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Knowledge-based Sustainable Management for Europe's Seas

Deliverable 7.2 Cost-efficient and Cost-Benefit analyses of reduction of eutrophication in the Baltic Sea

Part I: Cost-efficient management measures for the mitigation of eutrophication in the Gulf of Finland

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Part II: Cost-benefit analysis of reduction of nutrient emissions in the Baltic Proper – a Bayesian Belief Network approach

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CONTENTS

SUMMARY

This deliverable presents two innovative bio-economic models for studying eutrophication and its reduction in the Baltic Sea context. The models are presented as separate reports.

Part I studies cost-efficiency of management measures for the mitigation of eutrophication in the Gulf of Finland. The innovative aspect of the analysis is the optimal timing of the implementation of the management measures. The analysis takes into account two important time lags: 1) the nutrient loads to the sea are not reduced immediately when the measure is implemented, and 2) the nutrient concentrations in the sea are not reduced immediately when the load is reduced. Further, marine ecosystem response can be either positively or negatively correlated with reduction of external load.

The results show the timing of the implementation of the management measures. The analysis works on the premise that the coastal countries of the Gulf of Finland (Finland, Estonia and Russia) can choose a cost-efficient combination of ten management measures. Which country is using which management measure, and when, is dependent on the management objective. If the objective is to reduce algae then the largest share of the budget should be used in Russia. However, from 2035 onwards, the majority of the budget should be used in nitrogen fertilization reduction in Finland. If the target is cyanobacterial reduction, Russia is the most important target for money in all the time periods.

Part II of this deliverable introduces a probabilistic approach to study costs and benefits of combating eutrophication in the context of the Baltic Proper. The analysis focuses on the effects of implementation of HELCOM's Baltic Sea Action Plan when benefits are gained from cod fishery and recreational use of the sea. The strength of the probabilistic approach is that it allows an analysis under critical uncertainties, which are many in the Baltic Sea. This approach also provides a framework for explicitly depicting uncertainties in one approach, which is a valuable input to decision making. The ecosystem's behaviour has many stochastic features and built-in feedbacks, which increase uncertainties. Economic variables like benefits and profits are also very uncertain, as is the response of the fish industry and tourism.

The main objective of Part II is to develop the probabilistic model. An initial model run is conducted and its results show that the benefits of implementation of the Baltic Sea Action Plan very probably will be higher than the costs. It also shows that this finding is relatively robust, even given the amount of uncertainty. Benefits will come especially through increasing recreation value. The model indicates also that eutrophication development influences cod stocks, but it is just one of many factors (e.g. fishing, salt water inflow from the North Sea and climate change). The model run shows that combating eutrophication increases catches in cod fisheries, but not by much.

Part I

Cost-efficient management measures for the mitigation of eutrophication in the Gulf of Finland

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1. Introduction

Presently, nearly 12.4 million people live in the catchment of the Gulf of Finland. Although the three riparian countries, Finland, Estonia and Russia, have been collaborating to protect the gulf since the 1950s, it is the most nutrient-enriched sub-basin of the Baltic Sea. The increased nutrient concentrations enable excessive growth of algae, frequent blooms of cyanobacteria, some of which are toxic, and oxygen depletion including hypoxia in the bottom. The nutrient load enters into the gulf as waterborne (rivers and direct point sources) or as airborne (deposition) input. The waterborne (land-based) input is currently nearly 6,000 tonnes of phosphorus and 110,000 tonnes of nitrogen annually. The present figures are one-third of the 1990s loads; however, despite the decreased loads the trophic status of the gulf has not changed correspondingly. In addition to the present flows of nutrients, the trophic status is dependent on the nutrient stock in the bottom sediment. The excess nutrients stimulate excessive growth of aquatic plants, which creates a large pool of organic matter that sinks to the bottom. This organic matter consumes oxygen in the near-bottom water and fuels the release of previously stored nutrients from the bottom sediment back to the water. For instance, the estimate of the internal phosphorus loads in the gulf varies between 4000 and 8000 tonnes per year (Rintala and Myberg 2009).

The present report is part of the deliverable D7.2 and the aim is to explore cost-efficient management measures for the mitigation of the eutrophication in the Gulf of Finland. The analysis is carried out using dynamic optimisation, but the problem can also be addressed using a DPSWR-cycle. Here we first describe the problem in general using the DPSWR approach (Table 1) and then explain the model framework and results from the numerical analysis. The main purpose of this report is to describe the methodology we are using, while the results presented here are only preliminary. We aim at publishing the final numerical analysis in a scientific journal.

By definition, cost-efficiency means that a policy goal is achieved with the minimum cost. In policy analysis cost-effectiveness is commonly used as a synonym for cost-efficiency. A policy is more cost-effective than another if its conservation outcome is higher for given total costs. Alternatively, a policy is more cost-effective than another if an equal conservation outcome is attained at lower total costs (see e.g.Wätzold et al. 2010). Our analysis is a combination of efficiency and effectiveness analyses so that it provides a frontier of cost-efficient solutions for a policy (see Figure 4), while not taking a stand on which of the efficient solutions is the most effective (Balana et al. 2011).

In practice, when negotiating and implementing international environmental agreements, budget considerations and target setting precedes decisions related to the management measures. The earlier studies addressing the cost-efficient mitigation of eutrophication have not considered a budget constraint, but they have calculated the cost minimising solution given the target. We take here an approach that considers the budget limitations seriously. The set of measures are then applied with the limits of the agreed budget.

The starting point of our analysis is therefore the definition of what can be interpreted as a response (R) in the DPSWR-cycle. The three littoral countries of the Gulf of Finland (GoF), Finland, Estonia and the Russian Federation, agree on a common budget that is used to mitigate the eutrophication. By assumption, they define the management objective in terms of reductions in algae and cyanobacterial biomass. A change in this eutrophication indicator (W) may lead to a change in the budget (R). Our numerical analysis illustrates which targets can be achieved by which budgets. The other important innovation of our analysis is that we consider both the nutrient stock (S) at the sea and the flow (P) to the sea. The earlier studies addressing similar problems have only considered the flows. Due to ecosystem feedbacks (state changes), management measures (drivers) based on analysis of nutrient flows only, can be less effective than originally thought. Furthermore, handling the timing of nutrient emission reductions and respective effects is an enhancement in comparison to previous research.

Table 1. *DPSWR structure of the cost-efficiency analysis*

Driver	Control variables that can be tuned to change the pressures (see Table 2)
Pressure	Loads of nitrogen (N) and phosphorus (P) from the catchment to the Gulf of Finland (GoF)
State	Stock of N and P at the sea
I/**W**	The change in the level of eutrophication with the given budget
Response	Budget of the three littoral countries Finland, Estonia and the Russian Federation allocated to the mitigation of eutrophication in the GoF.

The rest of the report is structured as follows. The next section explains the modelling framework that is a combination of catchment and marine models. Section three defines the boundaries for the cost-efficiency analysis. We define the budget and the target and explain the management measures. Section four reports the results and section five concludes the study.

2. Modelling framework

The modelling framework combines catchment and marine models. We apply the model developed by Ahlvik et al. (2012) but focus only on the Gulf of Finland and extend their analysis by considering a larger set of abatement measures and solving the optimal timing of the use of the measures. The catchment model estimates the effect of nutrient abatement measures on loads and abatement costs. The marine model estimates the effect of given loads on the nutrient concentrations in the sea and calculates algal and cyanobacterial biomasses.

2.1 Catchment model (Drivers and Pressures)

Eutrophication in the Gulf of Finland is driven by the lifestyle of the people living in its catchment area. One underlying force is demand for food, which again creates demand for agriculture production that is the main source of nutrients in the Baltic Sea (HELCOM 2007). Therefore, our focus is mainly on the cost-efficiency of different measures used to reduce nutrient loads from agriculture to the sea. In addition we explore the costs and effects of wastewater treatment plants and the use of phosphate-free laundry and dishwasher detergents. Thus, in our analysis, we define drivers as control variables that can be tuned to change the pressures. These measures can be taken in all three countries and our results will show the cost-efficient allocation of the abatement measures between the countries.

Table 2 presents the abatement measures of which some (e.g. use of fertilization) can be adjusted annually and some (e.g. construction of wastewater treatment plants) are fixed. Figure 1 illustrates the catchment model and how the interdependencies of measures are taken into account such that use of one measure affects the costs and effects of other measures. For instance, reducing fertilization reduces the efficiency of wetlands, since the fewer nutrients leaching into the water the less the wetland can retain. Reducing the use of inorganic fertilizer or manure reduces nutrient loss from fields, but has a cost in the form of decreased crop yields. The total cost of agriculture-related measures consists of the sum of net present values of loss of crops, opportunity costs of pigs, cattle and poultry, and the money saved by the decreased consumption of inorganic fertilizers.

Table 2. *Drivers defined as abatement measures*

Annual	Fixed
N & P fertilisers	Wetlands
Number of cows, pigs and poultry	Phosphorous ponds
Use of catch crops	Wastewater treatment plant
Phosphate-free detergents	

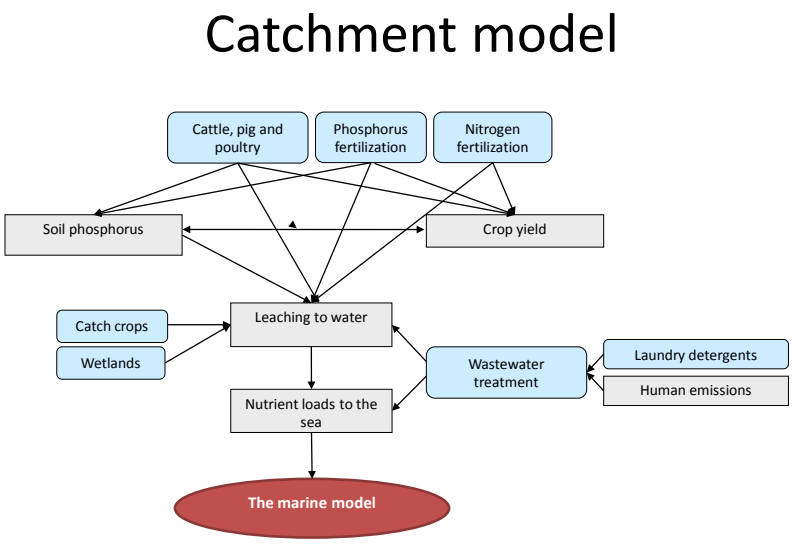

Figure 1. *Schematic presentation of the catchment model. The drivers are shown in blue.*

2.2 Marine model (State)

Nitrogen and phosphorus are the main components limiting the growth of phytoplankton in the Baltic Sea. In the marine model the phytoplankton is divided into algae and cyanobacteria and the model is able to calculate the biomass of both given the nutrient loads from the land and ecosystem feedbacks on the loads (Figure 2). These feedbacks are functions of nutrient stocks in the sea. We will base the existence of these feedbacks on a few well-known features of the marine ecosystem:

1. As nutrient concentrations increase, a constant or progressively smaller fraction of nitrogen is removed from the system (Seitzinger and Nixon 1985)
2. The capacity of sediments to retain phosphorus is decreased when the amount of benthic organic matter increases (Lehtoranta et al. 2008)
3. Nitrogen fixation depends linearly on the amount of cyanobacteria, which is largely determined by the nitrogen to phosphorus ratio (Vahtera et al. 2007)

Ecosystem response can be either positively or negatively correlated with reduction of external load. If the correlation is positive, the effect of nutrient abatement is amplified by the ecosystem, and the effect is called *positive feedback*. Respectively, negative correlation between the ecosystem response and nutrient abatement attenuates nutrient abatement, and the feedback is *negative*. These positive and negative feedbacks make the effective ecosystem management more complex than a straightforward external load optimization problem. Positive feedbacks can also cause regime shifts in the ecosystem; once a certain pollution level has been reached, positive feedback maintains eutrophication even if the external load is reduced back to its old level (Figure 3c). Feedbacks can thus cause significant delays or even disruptions in the ecosystem's responses to reduction of nutrient loading (Rönnberg and Bonsdorff 2004).

In order to find the true effect of load reduction and solve the real cost-efficient solution, we will find numerical estimates for magnitudes of feedbacks based on three stylized facts. The feedbacks are based on a deterministic version of the marine model introduced in Ahlvik et al. (2012), which is calibrated using nutrient concentration data for the years 1980-2000 and validated for years 2001-2008. The outline of the model is illustrated in Figure 2. Nitrogen and phosphorus stocks for the Gulf of Finland are measured by

$$Q_{N,t+1} = Q_{N,t} + \sum_{i=1}^{3} L_{N,t}^{i} + F\left(c_N^{BP} - \frac{Q_{N,t}}{V}\right) + D_N - Den\left(Q_{al,t}, Q_{cy,t}\right) + Fix\left(Q_{al,t}, Q_{cy,t}\right) \tag{1a}$$

$$Q_{P,t+1} = Q_{P,t} + \sum_{i=1}^{3} L_{P,t}^{i} + F\left(c_P^{BP} - \frac{Q_{N,t}}{V}\right) + D_P - Sed\left(Q_{al,t}, Q_{cy,t}\right), \tag{1b}$$

where Q_N and Q_P are masses of nitrogen and phosphorus, L^i is the riverine nutrient load from country I, F is the exchange of water between Gulf of Finland and the Baltic Proper, c^{BP} is the nutrient concentration in the Baltic Proper, V is the volume of the sea basin, D is the atmospheric deposition, *Den* is denitrification, *Fix* is nitrogen fixation and *Sed* is phosphorus sedimentation. We assume that an algal spring bloom consumes the nitrogen and phosphorus until the limiting nutrient runs out. If the sea is nitrogen-limited the excessive phosphorus is used by the nitrogen-fixing cyanobacteria. Biomasses of phytoplankton and cyanobacteria are calculated as

$$Q_{al,t} = \max\begin{cases} R_m \rho_{DIP} \dfrac{Q_{P,t}}{V}(1+\eta)V_{up} \\ \rho_{DIN} \dfrac{Q_{N,t}}{V}(1+\eta)V_{up} \end{cases} \tag{2a}$$

$$Q_{cy,t} = R_m \rho_{DIP} \frac{Q_{P,t}}{V}(1+\eta)V_{up} - Q_{al,t}\,, \tag{2b}$$

where R_m is the Redfield ratio for masses; ρ_{DIN} and ρ_{DIP} refer to basin-specific shares of inorganic dissolved nitrogen and phosphorus, which are assumed to be constant; η is the share of other, non-limiting components of phytoplankton; and V_{up} is the volume of the uppermost 20-meter layer of a subbasin. The nitrogen fixed by cyanobacteria is converted into ammonia, which can be recycled by algae blooms in forthcoming years:

$$Fix(Q_{cy,t}) = \frac{1}{(1+\eta)R_m} Q_{cy,t}\,. \tag{3}$$

Denitrification and phosphorus sedimentation are both assumed to follow a quadratic function. Both are functions of total algal biomass that decomposes in the bottom of the sea. Because cyanobacteria are lighter than algae, its effect is assumed to be 10 % of the effect of algae. Values

$$Sed(Q_{al,t}, Q_{cy,t}) = k_{Sed}\left[(Q_{al,t} + 0.1Q_{cy,t}) - \gamma_{Sed}(Q_{al,t} + 0.1Q_{cy,t})^2\right] \tag{4a}$$

$$Den(Q_{al,t}, Q_{cy,t}) = k_{Den}\left[(Q_{al,t} + 0.1Q_{cy,t}) - \gamma_{Den}(Q_{al,t} + 0.1Q_{cy,t})^2\right], \tag{4b}$$

where k_{Sed} , k_{Den} γ_{Sed} , and γ_{Den} are parameters that are estimated by the least squares method using observed data. Values for the parameters are shown in Table 3, and the relationship between denitrification, phosphorus sedimentation and benthic algal decomposition is illustrated in Figure 3.

As can be seen in Figure 3a, increased algal biomass also increases the amount of denitrification, but with decreasing marginal denitrification. In Figure 3b it can be noticed that phosphorus sedimentation processes have already suffered such that increase in phytoplankton biomass decreases the absolute amount of sedimentation. In Figure 3c we visualize the regime shift that has happened in the Gulf of Finland: even by reducing nutrient loads to some historical lower level, we cannot reach the same level of eutrophication because of decreased net sedimentation. If we want to reach some historical good state of the sea, we will have to reduce nutrient input even more to overcome the effect of internal loading.

Based on Equations (1) and (2) and parameters in Table 3, we can calculate positive and negative feedbacks of nitrogen and phosphorus load reductions. Nitrogen load reduction decreases the amount of nitrogen-limited algae and increases phosphorus sedimentation, therefore having a positive feedback on phosphorus abatement. The amount of excessive phosphorus after the spring bloom, however, increases, which increases cyanobacterial blooms and nitrogen fixation. This causes negative feedback on total nitrogen abatement. Phosphorus load reduction decreases cyanobacterial blooms, which causes less nitrogen fixation and more phosphorus sedimentation.

That is, phosphorus load reduction has positive feedbacks on both nitrogen and phosphorus budgets. These feedbacks are shown in Table 4.

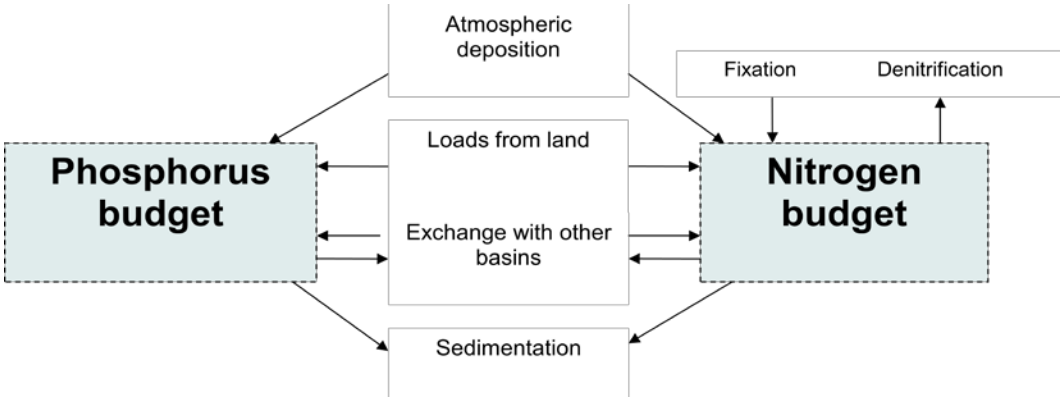

Figure 2. *Schematic presentation of the marine model*

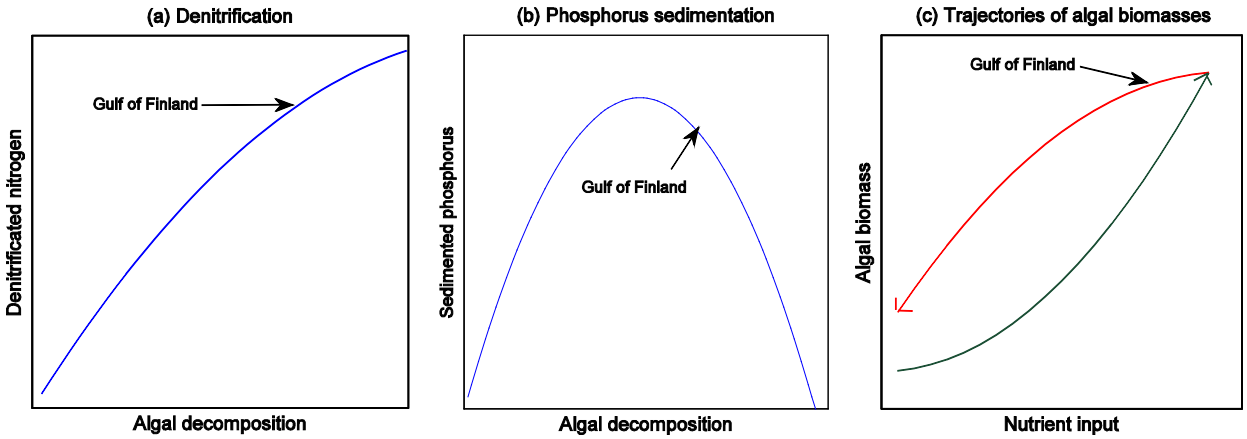

Figure 2. *(a) Denitrification and (b) phosphorus sedimentation as a function of biomass of decomposing algae (Ahlvik et al., 2012), (c) Trajectories resulting from a regime shift in response to nutrient loads; with increasing (green) and decreasing (red) nutrient (Duarte et al. 2009, Ahlvik et al. 2012)*

Table 3. *Parameters for denitrification and sedimentation function in each sub-basin (Ahlvik et al., 2012)*

	BB	BS	BP	GoF	GoR	DS	KT
k_{SedP}	0.5983	0.0845	0.094	0.1362	0.0017	0.0018	0.438
k_{DenN}	9.6893	1.2662	4.1026	1.5594	4.0773	3.6082	2.5747
γ_{SedP}	0.29 10^-4	0.02 10^-4	0.01 10^-4	0.135 10^-4	0.04 10^-4	0.22 10^-4	0.17 10^-4
γ_{DenN}	0.82 10^-4	0.01 10^-4	0	0	0.16 10^-4	0	0.01 10^-4

BB=Bothnian Bay, BS=Bothnian Sea, BP=Baltic Proper, GoF=Gulf of Finland, GoR=Gulf of Riga, DS=Danish Straits, KT= Kattegat

Table 4. *Feedbacks of reduction on external nutrient load*

	Feedback on nitrogen	Feedback on phosphorus
Nitrogen load reduction	Negative	Positive
Phosphorus load reduction	Positive	Positive

3 Optional policies

3.1 Policy objective (W)

The key in every cost-efficiency analysis is to determine a measurable target indicator, which can be either money or an index. In this study we use an indicator that is henceforth called the environmental index. The choice of such an index can be ambiguous, because decision makers might differ in their willingness to avoid different symptoms of eutrophication. In this study, we use algae and cyanobacteria to represent two different types of problems caused by eutrophication. Algae causes non-toxic algal blooms, loss of benthic oxygen and fish kills (Rönnberg and Bonsdorff, 2004), and it is determined by the amount of limiting nutrient, which is currently nitrogen in the Gulf of Finland. Cyanobacterial blooms are occasionally toxic and they can decrease the recreational value of the sea. Due to nitrogen fixing, the amount of cyanobacteria is defined by the amount of phosphorus.

In the environmental index we depict the attitude of the decision maker toward these two problems with a weighting parameter $\beta \in [0,1]$. Also, we assume that the decision maker values each time period up to T with equal weight. Thus, the environmental index is of the following form:

$$I(a,\beta) = \sum_{t=1}^{T} \left(\beta Q_{al}(a) + (1-\beta) Q_{cy}(a) \right), \tag{5}$$

where a is the set of abatement measures. If β =0 then the objective is to minimize the algal biomass and if β =1 then the objective is to minimize the cyanobacteria biomass. In the DPSWR-cycle this indicator can be interpreted as a change in the ecosystem state, which can be further interpreted as a change in human welfare. So far, direct welfare change estimates have not contributed to the setting of management objectives, which have been based instead on indicators describing the ecosystem state.

The cost-efficiency analysis is carried out so that the environmental index is minimized given different budgets and varying the values for β by solving

$$\begin{aligned} &\min \{ I(a,\beta) \} \\ &s.t. \quad C(a) \le B, \end{aligned} \tag{6}$$

where B is the given budget and C is the cost function.

3.2 Budget (R)

The earlier studies addressing the cost-efficient mitigation of eutrophication have not considered a budget constraint, but they have calculated the cost minimising solution given the target. We argue that a more realistic case is that countries first agree on the budget and then allocate this amount cost-efficiently between the management measures. This would be more realistic in the sense that budget allocations are an important factor in determining which management measures are taken in the end. Presently the budget allocations for measures to reduce nutrient emissions are set through a complex web of political negotiations that, in effect, take place in several processes on multiple levels (e.g. allocation of agri-environmental scheme funds, building and licencing of waste water treatment plants, banning of phosphates in detergents, limiting airborne emissions, etc.). The results will illustrate a cost-efficient solution for a hypothetical budget of 3 billion euros over the years 2010-2050. This is hypothetical since due to the complex policy setting underlying the implementation of management measures, there are no negotiations on a single budget at the national or international level.

4 Results and discussion

In this section we solve the optimization problem in Equation (6) for different ecological targets (values of weighting parameter β) and budget B. By fixing the budget and solving for all β we can derive the Pareto frontier, that is, a surface where it is not possible to reduce the environmental index for any β without increasing it for some other β. A rational decision maker in the framework of this study will always choose some point from the Pareto frontier depending on his or her β. The only way to further decrease the environmental index for all β is to increase the mitigation budget.

In Figure 4 we illustrate the Pareto frontier for budgets of 3 billion euros, which is 126 million euros annually on average over the simulation period of 40 years. According to the model, the amount of algae can be reduced to 93.6% – 95.2% of the original amount and the amount of cyanobacteria to 65% – 80% with the given budget. Even if the target is to reduce only algae, the amount of cyanobacteria is also reduced. This is due to two reasons. First, some abatement measures such as wastewater treatment plants reduce both nitrogen and phosphorus loads, and therefore phosphorus loads and cyanobacteria are reduced on the side. Second, nitrogen fixation depends on the amount of cyanobacteria. Therefore, reducing cyanobacteria is an effective way to reduce nitrogen input and the amount of nitrogen-limited algae.

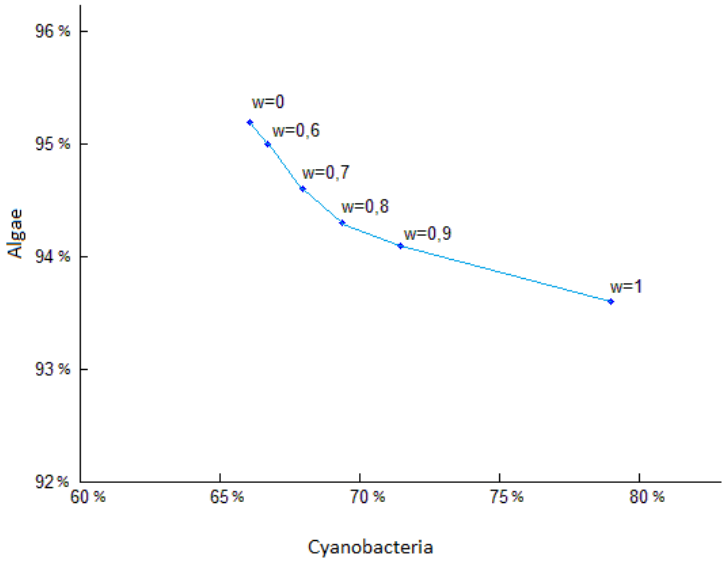

Figure 4. *The Pareto frontier for a budget of 3 billion euros*

In Figure 5 and Figure 6 we illustrate the division of a budget among the three countries and between abatement measures. We consider the two extremes: when the objective is to reduce only cyanobacteria (β=0) or algae (β=1). The most effective set of abatement measures to reduce cyanobacteria consists of improved wastewater treatment in Russia, a ban on phosphate-containing detergents in all countries, reduced use of phosphorus fertilizer and construction of phosphorus ponds in Finland and some use of catch crops in Finland and Estonia. If the goal is instead to reduce algae, we should decrease nitrogen fertilization in all the countries, construct some wastewater treatment plants in Russia, build wetlands instead of phosphorus ponds in Finland and ban phosphate in detergents only in Estonia and Russia, where the level of wastewater treatment is lower.

According to the results of this study, reduction of cattle, pigs or poultry is not part of the cost-efficient set with any β if the budget is three billion euros. Also, there are a few abatement measures that should be carried out regardless of the policy goal. These measures are part of the optimal solution for all weighting parameter β. Such measures are some use of catch crops in Finland and Estonia, a complete ban on phosphorus-containing detergents in Russia and Estonia, and improving wastewater treatment in the Russian coastal zone.

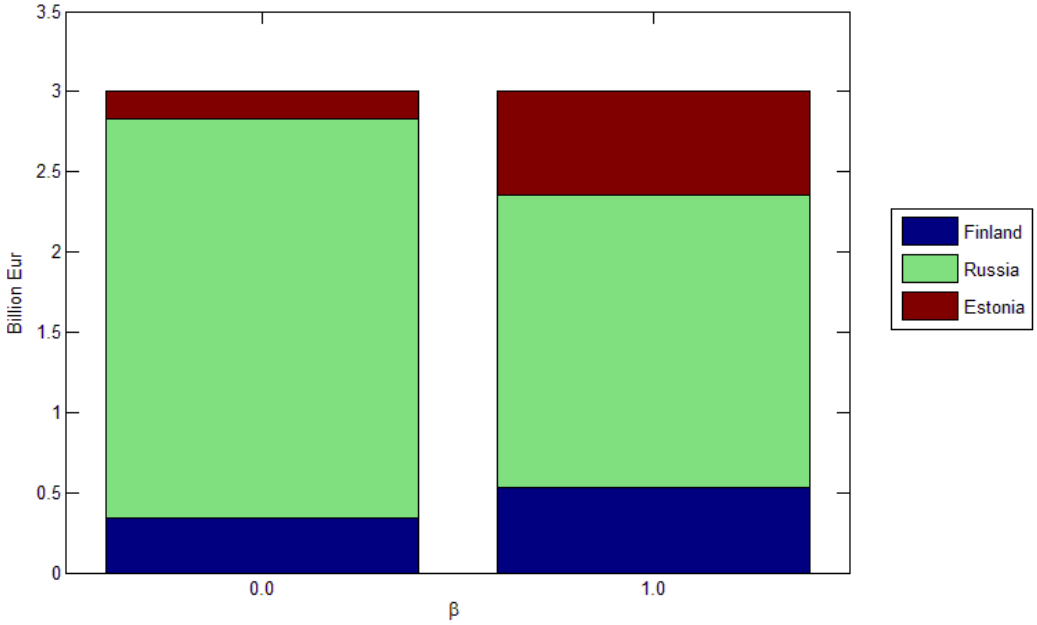

Figure 5. *Allocation of the common budget among three countries*

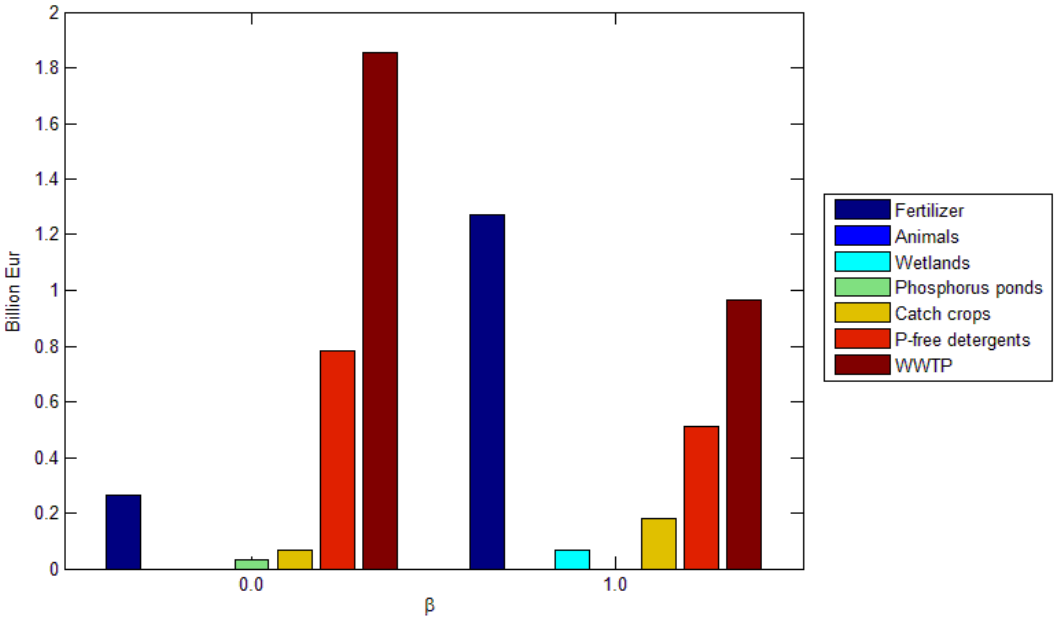

Figure 6. *Allocation of the common budget among abatement measures*

The annual costs of the optimal abatement program for algae and cyanobacteria targets are shown in Figures 7 and 8. The investment cost in wastewater treatment in Russia is the single most important cost component in both abatement objectives. In the beginning of the program to reduce algae, the largest share of the budget should be used in Russia. However, from 2035 onwards, the majority of the budget should be used in nitrogen fertilization reduction in Finland. If the target is cyanobacteria

reduction, Russia is the most important target for money in all the time periods. Interestingly, Finland can even profit from reducing phosphorus fertilization during the first time periods. This is because the soil fertility keeps up crop yields without having to add more fertilizers. The costs are not seen until the soil phosphorus stock decreases to a level where the cost of lost crops exceeds the cost of buying fertilizers.

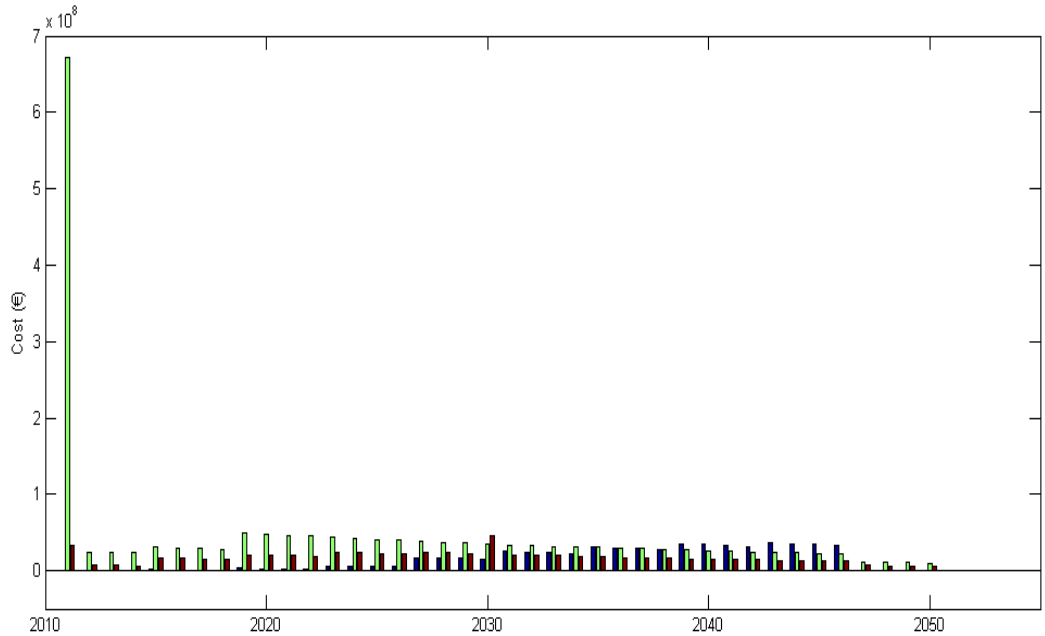

Figure 7. *Annual cost of optimal abatement program with algae target in Finland (blue), Russia (green) and Estonia (red).*

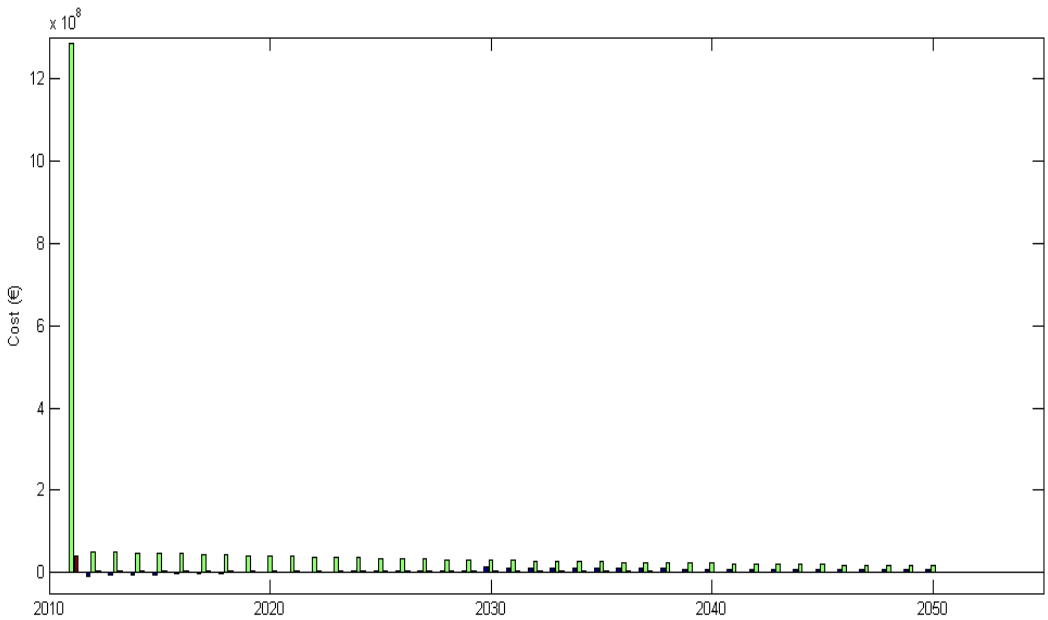

Figure 8. *Annual cost of optimal abatement program with cyanobacteria target*

5 Conclusion

We have addressed the cost-efficient mitigation of the eutrophication of the Gulf of Finland in an international framework. The underlying ecological-economic model is a combination of catchment and marine models. The catchment model estimates the effect of nutrient abatement measures on loads and abatement costs. The marine model estimates the effect of given loads on the nutrient concentrations in the sea. A clear innovation of the present analysis is that the time lags are included in the model in two different ways. First, there is a lag in phosphorus abatement measures. When phosphorus fertilization is reduced, loads are not reduced respectively immediately because of the soil phosphorus stock in the fields. Only after the phosphorus stock discharges, the full effect of the abatement measures is seen. Another component of lag is the ecosystem response. Due to large stocks of nutrients already existing in the Baltic Sea and the long residence time of water in the Baltic Sea, the nutrient concentrations are not reduced immediately when the load is reduced. This lag is less significant for nitrogen, because a relatively large share of the total nitrogen stock is buried or denitrificated. These lags are more important for phosphorus, because a smaller share of phosphorus is buried, and in addition phosphorus can be stored in sediments for a long time releasing only slowly.

The results illustrate the cost-efficient management measures in two extreme cases where the decision maker's preferences differ in terms of the symptoms of eutrophication. If decision makers are more worried about non-toxic algal blooms, oxygen depletion at the bottom or fish kills, then nitrogen fertilization should be decreased in all three countries: Russia should construct wastewater treatment plants, Finland should build wetlands and Estonia and Russia should ban phosphate in detergents. If the decision makers prefer to reduce cyanobacterial blooms then Russia should construct wastewater treatment plants, phosphate-containing detergents should be banned in all countries, Finland should reduce phosphorus fertilizer and construct phosphorus ponds and Finland and Estonia should use catch crops. Further, the dynamic analysis shows that irrespective of the decision maker's preferences, the investment cost in wastewater treatment in Russia is the single most important cost component and the measure should be implemented in the first year of the management programme. If the objective is to reduce algae then the largest share of the budget should be used in Russia. However, from 2035 onwards, the majority of the budget should be used in nitrogen fertilization reduction in Finland. If the target is cyanobacterial reduction, Russia is the most important target for money in all the time periods.

6 References

Ahlvik L., Pitkänen H., Ekholm P.P., H. and Hyytiäinen K., 2012. An integrated economic-ecological model to evaluate the impacts of nutrient abatement measures in the Baltic Sea. Submitted.

Balana B.B., Vinten A. and Slee B., 2011. A review on cost-effectiveness analysis of agri-environmental measures related to the EU WFD: Key issues, methods, and applications. Ecological Economics, 70:1021-1031.

Duarte C.M., Conley D.J., Carstensen J. and Sánchez-Camacho M., 2009. Return to Neverland: Shifting baselines affect eutrophication restoration targets. Estuaries and Coasts, 32:29-36.

HELCOM, 2007. HELCOM Baltic Sea Action Plan.

Lehtoranta J., Ekholm P. and Pitkänen H., 2008. Eutrophication-driven sediment microbial processes can explain the regional variation in phosphorus concentrations between Baltic Sea sub-basins. Journal of Marine Systems, 74:495-504.

Rintala J. and Myberg K. (Editors), 2009. The Gulf of Finland. Finnish-Russian-Estonian cooperation to protect the marine environment. Ministry of the Environment of Finland, Sälekarin Kirjapaino Oy, Somero, 119 pp.

Rönnberg C. and Bonsdorff E., 2004. Baltic Sea eutrophication: area-specific ecological consequences. Hydrobiologia, 514:227-241.

Seitzinger S.P. and Nixon S.W., 1985. Eutrophication and the rate of denitrification and N_2O production in coastal marine sediments. Limnology and Oceanography:1332-1339.

Vahtera E., Conley D.J., Gustafsson B.G., Kuosa H., Pitkänen H., Savchuk O.P., Tamminen T., Viitasalo M., Voss M. and Wasmund N., 2007. Internal ecosystem feedbacks enhance nitrogen-fixing cyanobacteria blooms and complicate management in the Baltic Sea. AMBIO: A journal of the Human Environment, 36:186-194.

Wätzold F., Mewes M., van Apeldoorn R., Varjopuro R., Chmielewski T., Veeneklaas F. and Kosola M., 2010. Cost-effectiveness of managing Natura 2000 sites: an exploratory study for Finland, Germany, the Netherlands and Poland. Biodiversity and Conservation, 19:2053-2069.

Part II

Cost-benefit analysis of reduction of nutrient emissions in the Baltic Proper – a Bayesian Belief Network approach

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1. Introduction

This part of the deliverable presents a method for a cost-benefit analysis (CBA) of the Baltic Sea Action Plan's eutrophication segment. We study the costs of meeting the set eutrophication reduction targets and the benefits that can be gained. In this work we test and develop a method for conducting a CBA. For that purpose, we take cod fishery and recreational use as benefits for further scrutiny.

The problems of eutrophication in the Baltic provide an excellent example of the complexity of ecologically and economically important interactions (Turner et al, 1999 and HELCOM, 2004). Our analysis focuses on the relationship between nutrient reduction measures, changes in environmental conditions including cod stocks and the costs and benefits of these changes to humans. Under the Baltic Sea Action Plan (BSAP) and the Marine Strategy Framework Directive (MSFD) there are clear environmental targets for reduction of nutrient loads (BSAP, 2007). Meeting targets for the reduction of nutrient loads will have clear costs and these should be balanced by the resulting benefits. Understanding the ecological consequences of nutrient reduction and their effects on human welfare requires an understanding of the physical, chemical and biological dynamics of the Baltic Sea ecosystem as well as an understanding of how these processes will result in changes to human welfare.

However, in both the economic as well as in the biological areas, various types of uncertainties exist. (For example, see Brandt and Vestergaard, 2011.) In particular, the consequences of nutrient reduction policies are fraught with uncertainty both in ecological and economic terms.

The aim of this deliverable is to assess the economic consequence for society of achieving a good environmental status in the Baltic Sea. Given that policies are already in place, our task is to evaluate whether the agreed-upon policies that focus on Good Environmental Status (GEnS) are a good idea for society, then calculating the costs and benefits of this policy and potentially indicating whether more or less regulation or changes in the design of the regulation are appropriate.

In our analysis we apply Bayesian state-space modelling to evaluating policy options under conditions of uncertainty. By using the BBN (Bayesian Belief Networks), the focus is not to avoid or ignore the prevailing uncertainty, but to quantify it by discretizing, that is to derive discrete probabilities for possible outcomes for each of the stochastic processes or other uncertainties. In order to include both the economic and environmental dimension of the problem and the attached

uncertainties on equal terms, we apply a BBN approach. Given the amount of uncertainty, the analysis is done in probabilistic terms by including the prevailing inherent uncertainties into an analysis evaluation using BBN.

Several papers have used this approach. Bromley et al. (2005) provide a good example of a BBN model for integrated water resource planning, while Borsuk et al. (2004) use a Bayesian network of a eutrophication model in particular focusing on prediction and uncertainty analysis. Levontin et al. (2011) also focus on the strengths of the BBN to apply an interdisciplinary approach to evaluate a management plan for regulation of salmon. Barton et al. (2008) emphasize the advantage of a BBN to structure (and to give a common understanding of) the probabilistic information of existing studies, varying from bio-chemical models, over non-market evaluation to various expert options and model assumptions. The central theme of all these papers is the integration of a large amount of very diverse data, information, or disciplines into one comprehensive analysis through the BBN representation.

Costs of reducing emissions and other social costs will be studied as extensively as possible, by taking the results of the existing studies as starting points. All benefits will be identified and the relevant ones are quantified if data or existing studies are available. Cod is given a closer scrutiny to exemplify possible benefits to be gained from reduction of eutrophication.

The costs of protecting the Baltic Sea have been studied extensively. Cost-effectiveness of policies and ways to optimize policies are an important societal goal as well as a valid academic question. In recent years as research methods have evolved, more interest has been shown toward studying the benefits of protecting the sea. Research of costs and benefits together can help us to ask whether chosen policies are worthwhile efforts for societies.

Using the BBN shows that the BBN provides important information about the social value of the BSAP. Our model shows that the result of implementing the BSAP compared to the reference scenario is positive. Perhaps more importantly, it also shows the uncertainties attached to this type of assessment and as we show, it is an excellent platform for making sensitivity analysis.

This deliverable starts with a short description of the main problem – eutrophication. The topic was described extensively in deliverable D7.1. The rest of the report focuses on the CBA. Cost-benefit analysis is, however, a complicated endeavour, as is shown in this deliverable. We start that section with a brief introduction to the topic. After that we present a deterministic approach for doing CBA. That methodological choice presents several challenges that need to be overcome. We discuss ways to handle the challenges and in the end we present an alternative probabilistic method for conducting a CBA.

2. Dynamics of Eutrophication

Eutrophication in the Baltic is a highly complex phenomenon, involving interactions and feedbacks between the physical, chemical and biological components of the ecosystem. The intricacies of these interactions have several consequences which increase the resilience of the system against remediation efforts (Österblom et al., 2010).

At lower trophic levels inputs of excess nutrients, principally from agriculture and domestic waste water, have caused undesirable blooms of algae and cyanobacteria. These are sometimes toxic and carry health risks for humans and animals. Decomposition of this excess algal material has resulted in reduced bottom oxygen concentrations and large areas of hypoxia or "dead zones". Two important feedback loops directly affect eutrophication status by increasing non-anthropogenic nutrient loads. These feedback loops are: nitrogen fixation by cyanobacteria from the air which enhances eutrophication under phosphorus-limited conditions, and internal loading of phosphorus from benthic sediments which is stimulated by the anoxic conditions (themselves the result of the eutrophication process) (Neuman, 2007, Mort et al., 2010).

This eutrophication phenomenon in turn has effects on higher trophic level species and in particular cod. Over the last five decades increased nutrient loading has led to higher primary production and increasing food availability for cod. However, the dead zones, which have extended due to eutrophication, are void of macroinvertebrates, an important food for cod. More importantly, the large-scale anoxia together with low salinity cause unfavourable conditions for hatching of cod eggs. The oxygenation of the Baltic water column is critical to the recruitment of cod, since cod eggs settle on a density layer determined by salinity and temperature. The amount of water with suitable density and oxygen conditions is known as the cod reproductive volume (MacKenzie et al., 2000). If the layer where the cod eggs settle is anoxic, the eggs do not develop (Figure 1), which leads to a decline in cod stocks over and above declines caused by fishing and natural mortality. Thus eutrophication can have both beneficial and detrimental effects on cod stocks. Unfortunately the latter effect has probably been stronger in recent decades (Köster et al., 2003).

Natural climatic fluctuations exert control over physical conditions in the Baltic, and recovery from hypoxic conditions, with its associated consequences for cod recruitment, is related to the rate of replenishment of waters within the Baltic Sea from North Sea inflows. Flows of well-oxygenated saline waters from the North Sea through the Kattegat are important determinants of oxygenation levels in the deep waters of the Baltic (Matthäus and Franck, 1992).

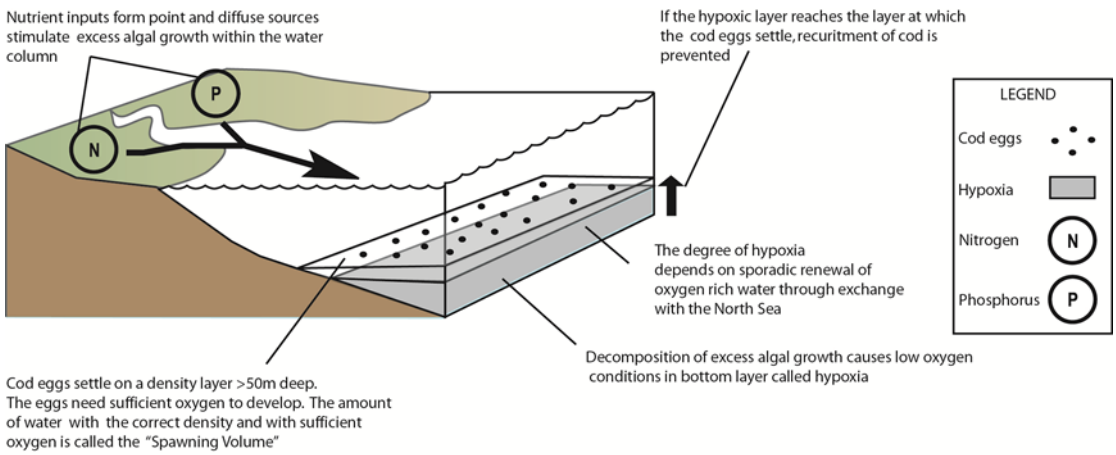


Figure 1. *Conceptual diagram showing the relationship between hypoxia and cod recruitment.*

These saline inflows are weather-driven stochastic events. Figure 2 shows a schematic diagram of how sporadic inflow events could alter the trajectory of recovery in an unpredictable and stepwise fashion.

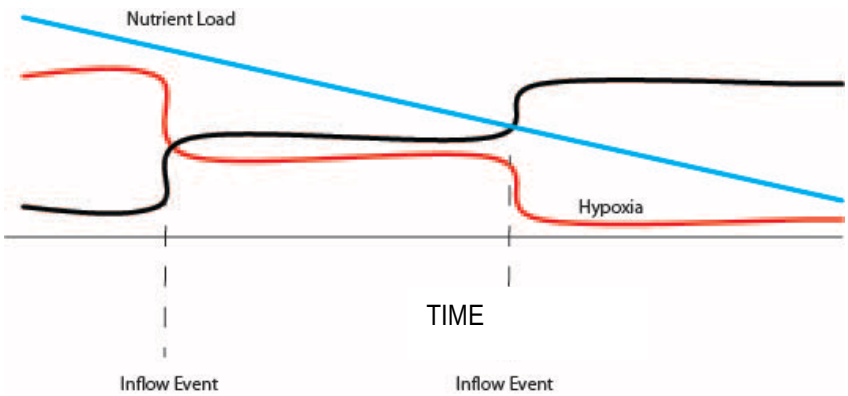


Figure 2. *Diagram illustrating the importance of stochastic inflow events in controlling the environmental state of the Baltic Cod (Black) and Oxygen (red conditions). The blue line illustrates decreasing nutrient load nutrient load (blue).*

The Baltic is characterized by shallow and narrow connections to the North Sea and relatively large river runoff, which together result in low salinity, long residence time and strong stratification. The main implications of long residence time (of 30 years) is that nutrients are not exported in significant amounts and are retained in the Baltic. Increased primary production and the building up of a phosphorous pool in the deep basins of the Baltic Sea may have caused a biogeochemical regime shift from oligotrophic to eutrophic conditions (Osterblom et al. 2007). This new biogeochemical state is stabilized by feedback, where phosphorous released from the sediment in response to eutrophic conditions stimulates the growth of N-fixing cyanobacteria. Cyanobacterial decomposition promotes further increases in oxygen consumption and the release of phosphorous from the sediment (Conley et al., 2009). An increase in summer phytoplankton production (and thus deep-water oxygen demand, at least in the Baltic Sea) is further strengthening the feedback mechanisms between unfavourable environmental conditions and overfishing. Sub-basins closer to the input of the comparably nutrient-poor North Sea water and with shorter water residence times generally respond faster to nutrient load reductions (Savchuk and Wulff, 2007).

Stochasticity and feedbacks inherent to the system can create considerable delays to the ecological systems' response to reduction of nutrient loads. A further delay between applying protection measures and gaining positive ecosystem response is generated by agriculture lands. Agriculture is a significant source of nutrients in catchments of the Baltic Sea. This delay is due to the slowness of the phosphorous cycle in the system. Nitrogen in the form of nitrate is easily soluble and is transported in runoff rather quickly. Phosphate is only moderately soluble and is not very mobile on soils. However, erosion (caused for example by rain) can transport considerable amounts of sediment-absorbed phosphate to surface waters. During heavy rain not even a buffer strip can completely prevent the runoff (Iho and Laukkanen, 2009).

If soils have been over-fertilized, rates of dissolved phosphorus losses in runoff will increase due to the build-up of phosphate in the soil (Shortle and Adler, 2001). Nutrients are coming from the whole field area; only small proportions of the surplus not used in plants are immediately released into surface water courses (Vagstad et al., 2004). Larger fractions are retained in the soils and slowly released into groundwater and, after a considerable time lag, eventually reach the sea. These time lags vary significantly between regions due to differences in climate, hydrology, erosion, and soil types (Vagstad et al., 2004). Therefore, results of many nutrient-reducing efforts by farmers can be seen only after a long time, which does not help raise the motivation of farmers to implement reduction measures (Iho and Laukkanen, 2009).

Superimposed on these eutrophication and natural climate-driven changes to the ecosystem are the anthropogenic changes caused by fishing, as well as trophic interactions between cod and other fish species. In the Central Baltic Sea a regime shift from a cod-dominated ecosystem to one dominated by clupeids has been observed. The trophic interactions between cod, herring and sprat may periodically have a strong influence on the state of the fish stocks in the Baltic, depending on the abundance of cod as the main predator of herring and sprat. The prey-predator relationships in the Baltic Sea are complicated (Casini et al., 2007). During the past two decades, the cod stock has declined from a historic high (in the early 1980s) to its lowest level on record (at the beginning of the 1990s), while the sprat stock increased to historic levels during the 1990s. Decreased predation on planktivorous fish then allows growth of these fish populations that may feed on cod eggs and larvae and compete with juvenile cod for prey, thus reinforcing the shift from cod to clupeids. This is the so-called Prey to Predator feedback (Möllmann et al., 2008). The regime shift has been largely attributed to overfishing and nutrient loading concurrent with natural climatic fluctuations (Österblom et al., 2010, Lindegren et al., 2009, Möllmann et al., 2008).

A further layer of complexity is added to the system when climate change is considered. A number of key species/taxonomic groups have been affected by the recent warming, e.g. copepod *Acartia* spp. (Möllmann et al., 2008) and the pelagic planktivore spratt (Köster et al. 2003). Other key species such as *Pseudocalanus acuspes* and the main fish predator in the pelagic food web, cod, have in parallel suffered from reduced salinity and oxygen levels (Möllmann et al., 2008, Köster et al., 2003). Effective exploitation has further reduced the abundance of the predator. The drastic increase of zooplanktivorous sprat, a consequence of predation release from cod and climate warming, has also strongly affected the zooplankton community (Casini et al. 2007). All together, these processes have caused changes in the structure and function of the food web and eventually contributed to an ecosystem regime shift (Möllmann et al., 2008, Casini et al., 2007).

Overall the system is dominated by complexities of nutrient cycling, enhanced primary production and oxygen consumption, trophic interactions between species and the physical processes of saline water inflows all against a background of gradual warming and freshening caused by climate change. Feedback loops both in the eutrophication process (nitrogen fixation and phosphorus remineralisation) and in the trophic interactions (between fish species) add resilience to the ecosystem which has already undergone a regime shift. All these factors lend uncertainty to the trajectory of recovery for the Baltic.

3. Combating eutrophication – Baltic Sea Action Plan (BSAP)

In our study we focus on the Baltic Sea Action Plan (BSAP) as a policy to combat eutrophication. The BSAP is a programme of measures for the protection and management of the Baltic Sea. It was adopted in 2007 by a ministerial meeting of HELCOM (HELCOM 2007). The Action Plan is structured around a set of Ecological Objectives used to define indicators and targets, including effect-based nutrient input ceilings, and to monitor implementation. The Action Plan strongly links Baltic marine environmental concerns to important socio-economic fields such as agriculture and fisheries and promotes cross-sectoral tools including marine spatial planning. Due to complementarities with the European Union (EU) Marine Strategy Framework Directive, the Action Plan is in essence a pilot for this process without neglecting the important role of the Russian Federation - the only Baltic coastal country which is not a member of the EU.

BSAP introduces an ecosystem approach to protection of the Baltic Sea with a target of restoring good ecological status (Backer et al. 2010). BSAP's vision is “a healthy marine environment, with diverse biological components functioning in balance, resulting in a good ecological status and supporting a wide range of sustainable human activities”. This target is further separated into four main thematic topics: 1) eutrophication, 2) hazardous substances, 3) loss of biodiversity and 4) maritime activities. Under each of the four themes a clear set of ecological objectives is defined that provides the base of the Baltic Sea Action Plan. The goals are to be achieved by 2021 (HELCOM 2007).

For the work presented in this report the first theme – eutrophication – is the most relevant. A set goal for this topic is to achieve a status of "Baltic Sea unaffected by eutrophication". The goal has five more detailed objectives:

- Concentrations of nutrients close to natural levels,
- Clear water,
- Natural level of algal blooms,
- Natural distribution and occurrence of plants and animals,
- Natural oxygen levels

An indicator for clear water was chosen as the main indicator to follow progress towards achieving the objectives for the eutrophication theme. BSAP states that "transparency of seawater integrates many of the concrete effects of eutrophication" (HELCOM 2007, 76). Summer time Secchi depth is the actual indicator, for which sub-basin specific target levels are further specified (Table 1).

Table 1. *Water transparency indicator Secchi depth. Reference levels, Targets and the present situation for the Baltic Sea's sub-basins.*

Sub-Basin	Summer time Secchi depth (June-September), metres		
	Reference, m	Target, m	Present situation, m
Bothnian Bay	7.5	Present situation	5.8
Bothnian Sea	9.0	Present situation	7.0
Gulf of Finland	8.0	6.0	4.1
Gulf of Riga	6.0	4.5	3.4
Kattegat	10.5	Present situation	8.5
Baltic Proper	9.3	7.0	6.3

Reference levels reflect "historical, non-impacted status", but in setting the actual target the action plan considers as high as 25% deviation from reference levels as acceptable. Methods that are used to define reference conditions include use of historical data, modelling, empirical research on the relationship of Secchi depth to levels of chlorophyll-*a* or nitrogen concentrations and historical depth data of macrophytes. Reference conditions are defined differently in each country (HELCOM, 2009).

Table 2. *Emission reduction targets for phosphorous (P) and nitrogen (N) in the sub-basins (HELCOM, 2007).*

Sub-basin	Maximum allowable nutrient input, tonnes		Inputs in 1997-2003, tonnes		Needed reductions, tonnes	
	P	N	P	N	P	N
Bothnian Bay	2,580	51,440	2,580	51,440	0	0
Bothnian Sea	2,460	56,790	2,460	65,760	0	0
Gulf of Finland	4,860	106,680	6,860	112,680	2,000	6,000
Gulf of Riga	1,430	78,400	2,180	78,400	750	0
Kattegat	1,570	44,260	1,570	64,260	0	20,000
Baltic Proper	6,750	233,250	19,250	327,260	12,500	94,000
Danish Straits	1,410	30,890	1,570	64,260	0	20,000
Total	21,060	601,720	36,310	736,720	15,250	135,000

The BSAP stipulates how much nutrient emissions should be reduced to reach the set objectives for the eutrophication goal. The emission reduction targets are set for each sub-basin (Table 2) and each country (Table 3). The emission reduction calculation is based on the MARE NEST model. The NEST decision support system combines different models: a drainage basin model to cover water runoff, sediment and nutrient inflows from the Baltic Sea catchment area, and a biogeochemical model to simulate nitrogen and phosphorous cycles in the Baltic Sea's sub-basins (see e.g., Wulff et al, 2007). The contracting parties have agreed that the actions to reach the set goals are taken no later than 2016 (HELCOM, 2007). The national implementation plans will be evaluated in the 2013 ministerial meeting (HELCOM, 2010).

Table 3. *National emission reduction targets (HELCOM, 2007).*

Country	Phosphorous, tonnes	Nitrogen, tonnes
Denmark	16	17,210
Estonia	220	900
Finland	150	1,200
Germany	240	5,620
Latvia	300	2,560
Lithuania	880	11,750
Poland	8,760	62,400
Russia	2,500	6,970
Sweden	290	20,780
Transboundary common pool	1,660	3,780

4. Costs and benefits of reducing eutrophication

There is an increasing realisation that effective management of the marine environment requires information not just about how marine ecosystems function but also about how human activities rely on stocks and flows of ecosystem services (Constanza, 1997; MEA, 2005). This realisation has led to the development of the Ecosystem Approach, which recognises the concepts of social-ecological systems (Tallis, 2010). An integrated understanding of the relationships between the human and natural elements of social-ecological systems requires transdisciplinary cooperation combining the specialised skills and knowledge of natural as well as social scientists. Bridging the disciplinary gap poses a unique challenge, since different disciplines have different foci, and essential questions in one discipline may seem to be of only peripheral relevance to another (see Haapasaari et al., 2012). Recognition and reconciliation of the distinct yet related areas of expertise should lead to a fuller understanding and more practically applicable research outputs (Gowdu and Carbonell, 1999).

Mechanistic modelling of biological processes gives us insights into the causes and consequences of ecosystem change and can help predict likely future environmental conditions. Yet biological modelling often tells us little about how these future conditions will affect human welfare and may not necessarily be of direct use to managers. Similarly, economic modelling can allow us to understand the relationship between changes in ecosystems processes and changes in human welfare, but oversimplification of such relationships may lose important detail. For the ecosystem modeller, trophic flows, positive and negative feedbacks and other intricacies of ecological systems are essential, while human activities may be considered only peripherally in terms of the pressures and environmental state changes they produce. Similarly, for the economic modeller the functioning of the ecosystem is not the main focus; rather the interaction between the supply of ecosystem services and the welfare benefits these generate as well as the values of the externalities caused by human activities are of primary interest.

Given the two different foci for research there is a risk that essential elements of the ecological system can be overlooked by economic modellers, while the benefits to humans from exploitation of ecosystems may not occur to the ecologist. Omission of either the ecological or social detail from models can result in products which are too simplistic to be credible or useful. Collaboration across disciplines can therefore improve the relevance of our models, resulting in more highly focused, more realistic and more credible outputs of improved value to decision makers.

An analysis of the costs and benefits of combating eutrophication is offered as an approach for linking ecological and economic parts of the problem described above. A Cost-Benefit Analysis (CBA) is a bio-economic modeling exercise to study whether a project or policy initiative is, from an economic point of view, worthwhile to undertake. This is relevant, if e.g., policy makers already have agreed upon a reductions target, and we want to make an economic appraisal of this project. A project would be worthwhile undertaking for society if the total benefit (B) from the project is larger than the total costs (C) of undertaking this project. If in C all costs, including opportunity costs, are included, then we can say that when $B/C > 0$, then the project is worthwhile undertaking, while when $B/C \leq 0$, it is not worthwhile. Consider again that uncertainty about the position of B and C is present, as shown in Figure 3a and 3b.

In Figure 3a, a policy in L^1 is good ($B/C > 0$), in L^2 is probably good (the sign of B/C cannot be determined), and in L^3 is bad ($B/C < 0$). However, if the uncertainty is very large, we cannot by an ordinary CBA conclude that any reduction is certainly good, even though there are probably reduction levels that could be expected to be good, as illustrated in Figure 3b.

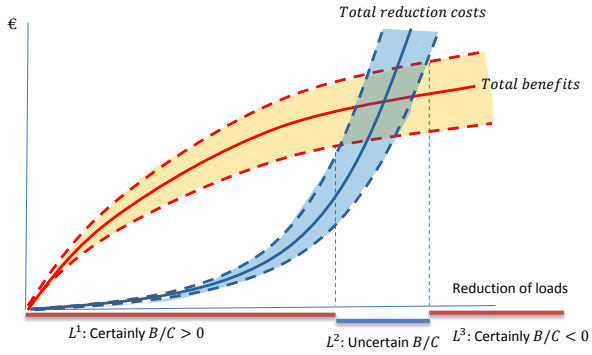


Figure 3a. *CBA under "small" uncertainty*

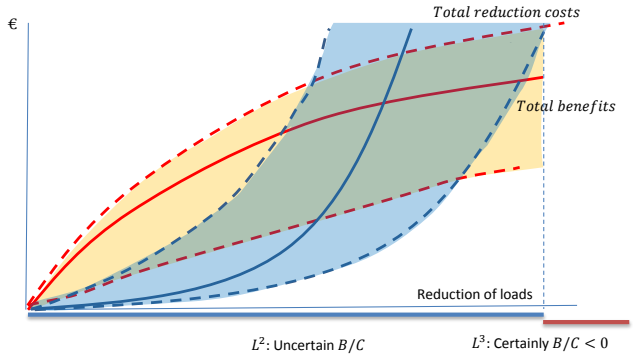


Figure 3b. *CBA under "large" uncertainty*

Incorporating the uncertainty will again be the approach that we will follow when conducting a CBA of the eutrophication of the Baltic Sea.

The situation shown in Figure 3b provides a huge challenge, in particular because different options for the cost and benefits can either support or reject the chosen policy. In section 5 we will explain a way to balance different options and modelling results, such that we can evaluate the policy in a probabilistic way. This seems, given the amount of uncertainty, the only reasonable way to present the consequence of a chosen policy.

4.1 Costs and benefits – a deterministic modelling example

To illustrate the point of uncertainties further, we present here an example of a deterministic modelling approach and challenges related to it. After that we describe how taking a probabilistic approach will help to deal with the uncertainties we face with the CBA. We undertook a CBA implementing a policy of good environmental status in the Baltic Sea regarding the eutrophication problem. Figure 4 summarizes the model structure, showing the relevant variables associated with reduction in eutrophication for the Baltic and the links between them.

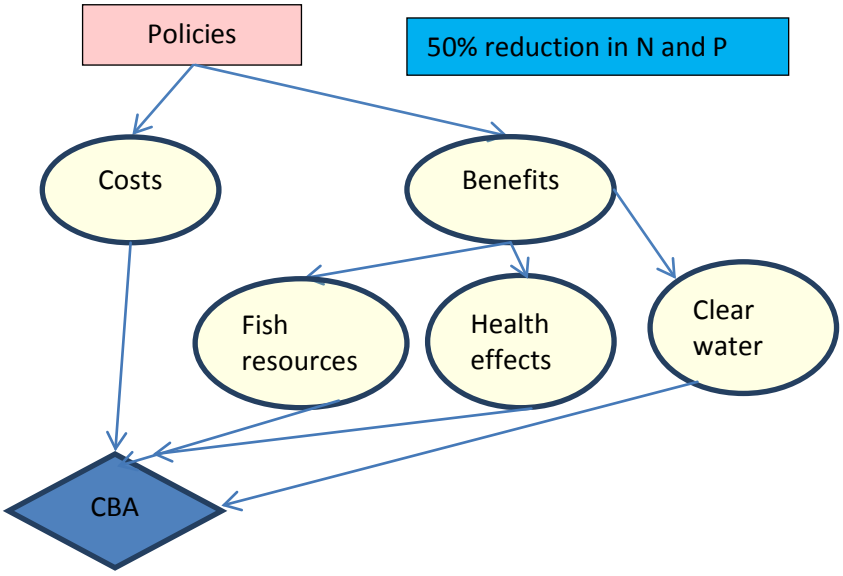


Figure 4. *Model structure for CBA on nutrient emission reduction*

For the example here, the goal set for eutrophication will be achieved by a 50% reduction in N and P. **Costs** of meeting this target have been studied, for instance, by Gren (2008). The study is a cost-efficiency analysis, i.e., a calculation of the minimum costs of achieving a pre-determined reduction target. **Benefits** in the figure come through changes in cod stocks and in water clarity and reduced risk to health. Thanh (2011) estimates a negative relationship between profit in fishery and level of N. Assumptions are that the Total Allowable Catch (TAC) of the cod fisheries is set equal to the optimal yield, given a discount rate of 4% per year. A bio-economic model is used to model the bio-economic dependency (using a best fit procedure). Kosenius (2010) estimates the minimum yearly household Willingness To Pay (WTP) for scenarios with a focus on the reduction of blue-green algae blooms and an increase in water clarity at about 165 euros and 210 euros respectively. Given the household size (2.34), WTPs for these two scenarios are about 70.5 euros and 89.7 euros per person per year respectively. Table 4 illustrates a result of using such studies as a basis for approaching a deterministic CBA.

Table 4. *Costs and benefits of reducing eutrophication in the Baltic Sea*

	Year (million euro)	NPV (40 years, i = 4%)
COSTS		
Reduction costs		
For N (50% reduction)	-2504	-49559
For P (50% reduction)	-2405	-47605
Total	-4909	-97164
BENEFITS		
Improvement in fish stock	+11	+218
Improvement in water quality	+13203	+261324
Improved health	+10373	+205310
Total	+23587	+466852
TOTAL	**+18678**	+ 369688

This results in a conclusion that a reduction of 50% of nutrient loads would give a positive CBA of + 18667 million euros or a B/C ratio of 4.8, indicating that (on average) each action that yields a cost of 1 provides 4.8 benefits. Given these numbers, such a project is worthwhile undertaking. One must notice, however, that the calculation assumes cost-efficient measures. In reality the costs would be higher.

The deterministic approach presents us with several challenges. The most critical challenges pertain to variables which need to be well understood but where there is severe uncertainty and causal links between variables are not straightforward to establish.

4.2 Main issues and challenges in a deterministic approach

There are several studies of the costs of reducing nutrient loads. First, these studies give very different results for the costs. Second, all such studies do assume cost efficiency. This poses a critical challenge for CBA. Regarding the fish stocks, for one, the challenge is that the size of a fish stock is influenced by several factors. Much of the variation in fish stocks is caused by climate variations and fishing effort, but experts have different opinions on that. Moreover, models give deterministic results for complex models. Similarly, the relationship between stock size and profit is not easy to establish as the relationship is influenced by several factors. One possibility is to use historical data. Another is to assume a functional relationship between stock and profit.

Additional challenges include how to measure water quality and health, and how to value it. One possibility is to use WTP studies (travel costs, benefits transfers). In the existing studies values build up to be very large, but still the existing studies result in different values. A further complication is the difference in water quality near the coast and in open sea. A second question is whether WTP studies are more applicable for coastal conditions or rather on the scale of the whole sea. Timing of costs and benefits is also an important issue. As discussed in section 2 there are critical delays in the system, which create unpredictably long gaps between occurrence of costs of protection measures and emergence of the benefits. Other questions include: How should delays be accounted for, and should costs and benefits be valued equally? Should existence values for e.g., fish stock also be included? Finally, how should utilities be assigned to the welfare generating variables?

Below we will address these questions in building our BBN approach.

5. Costs and benefits – a probabilistic approach

In this section we present an alternative approach that is more suitable for assessing costs and benefits in a situation characterized by several critical uncertainties. The following section (5.1) presents our method of doing a probability-based CBA and presents the model structure. Sections 5.2 to 5.7 describe how different parts of the model are populated. The method is summarized in Figure 12. Results of the model run are presented in section 6.

5.1 Methodological approach

The overwhelming complexity in both the socio-economic and ecological aspects of the social-ecological system requires an approach that can adequately encompass the uncertainty inherent in each domain complexity. We consider the Bayesian Belief Network (BBN) approach to have this potential and that it may be an appropriate tool to properly evaluate policy options regarding reduction of eutrophication in the Baltic Sea. Our aim is to define a number of policies, using the BBN-framework to calculate a probability distribution over states for each policy, and evaluate them by an agreed utility function. The main task is to identify reasonable groups, intervals or outcomes for each non-deterministic part in the cause-effect chain.

A BBN can be described as an acyclic directed graph. It consists of a quantitative component (the graph) and a qualitative component, the conditional probability tables. The quantitative component gives the cause-effect relationship between the variables in the network. Ellison (1996) gives a useful introduction to Bayesian inference for research related to our analysis.

5.1.1 Conditional probability tables

A core feature of the BBN is the Conditional Probability Table (CPT). In this section we discuss ways to derive these CPTs on the basis of eliciting expert opinions.

One of the strengths of the BBN is that while conditional probabilities attached to each node can be quantified independently, all of the variables (nodes) are connected. This allows each variable to be populated with the best data available, including expert opinion, simulation results or observed data. It also allows the information to be easily updated as better data become available.

In Hamilton et al. (2005) it is described how a BBN can be used to identify important variables and represent causal relationships, through using workshops to elucidate expert knowledge and opinions, which in many ways resembles our approach to the problem. Uusitalo et al. (2006) collect expert opinions from different experts on probability statements and on this basis derive a CPT, an approach that is similar to our approach of eliciting expert knowledge. Kuhnert et al. (2005) argue that that expert information might play an integral part in the modelling and analysis of a BBN. It is, however, a far from trivial task to extract probabilistic statements from one or more experts. As noted by e.g., Burgman et al. (2006), a first and important step is to identify who is an expert.

The theoretical foundation of deriving the CPT is specified in Borsuk et al. (2004). They define the conditional variable X, derived using a functional relationship of the form:

$$X = f(\boldsymbol{p}, \theta, \epsilon)$$

$\boldsymbol{p}$ is the set of immediate causes (or parents) of X, θ is a vector of parameters of the function relating $\boldsymbol{p}$ and X, and ϵ an error (or disturbance) term.

E.g.:
$Algal\ density\ =\ f(water\ temperature,\ oxygen, salinity, nitrogen\ concentration)$
$Cod\ stock\ =\ f(fishing\ pressure, recruitment, nitrogen, salinity\ and\ oxygen)$

Since the true functional form is most likely unknown, and is the distribution of the error term, we need to proceed by making various assumptions. The assumption in a BBN is that the relationship between a parent and child variable can be described using probabilities and that it is possible to discretize the probabilities to fit into the CPT.

There are several options of extracting a CPT out of expert judgments. One approach presented by Dawes et al. (1989) is simply to ask experts directly. Uusitalo et al. (2005) derive a discrete probability table directly asking experts about pre-smolt density capacity. Ten intervals were given, and the experts assigned probabilities to each interval. Hereafter the authors calculated the aggregate probability table by giving each expert the same weight.

To illustrate this, for instance in the case of eutrophication in the Baltic Sea and consequences of reducing the nutrient loads, we can ask experts to enter probabilities into the following table. The table asks experts what the probability is under each of the scenarios (BAU, REF and BSAP, see section 5.2 for explanation), that, for instance, reduction of nutrient loads will lead to an increase in cod stock.

	BAU	REF	BSAP
Very high			
High			
Moderate			
Small			
Negligible			

As indicated by Burgman et al. (2006) the subjective (qualitative) names of the intervals might influence the judgement of experts. Intervals can also be made quantitative by asking the same experts to assign quantitative values to the categories (see table below). Knowing the experts' understanding of the intervals is important for comparability of different opinions, since an increase of cod stock by 5% might be considered “very high” by one expert, while another expert might consider it as “moderate”.

	INTERVAL	BAU	REF	BSAP
Very high				
High				
Moderate				
Small				
Negligible				

Burgman et al. (2006) raise a question regarding the use of experts: How is an expert defined? A problem is that experts have a specific view on the subject; they may not see the limits of their own expertise and can therefore be overconfident. For this reason, one should include opinions of several experts and also use available studies to generate CPTs.

Another way of deriving a CPT is by using earlier research as a source for filling in CPTs. This approach is used here to generate a CPT from a situation where various studies give different estimates for the variable in our BBN.

Consider that we have a number of studies, each giving an estimate of the value of the variable, for instance, the costs of implementing the BSAP. We assume that each study is equally valid (sound) and assume that these three estimations are independent draws from a normal distribution with $(N(\mu, \sigma)$. A way to estimate μ and σ is to use the maximum likelihood estimators of μ and σ given by:

$$\hat{\mu} = \frac{\sum_{i=1}^{n} x_i}{n} \text{ (equal to the average)}$$
$$\hat{\sigma}^2 = \frac{\sum_{i=1}^{n} (x_i - \hat{\mu})^2}{n}$$

From this it is possible to derive the CPT.

An example of how this can be made operational follows. Assume that for a 50% reduction in loads of N into the Baltic Sea, there are three studies that use different approaches to calculate the cost-efficient way to reduce the N load by 50%. The costs are shown in Table 5.

Table 5. *Assessments of 50% red N*

Study	Values NPV € (billions)
Gren 2008	73
Gren 1997	85
Wolff et al. (2001)	41

This yields the following estimates:

$$\hat{\mu} = 66$$
$$\sigma^2 = 345$$
$$\sigma = 19$$

The next step is to find relevant intervals for the costs, and then, given the estimated parameters of the normal distribution, calculate the probability that the cost falls into each interval:

Groups	Intervals	probability	normalized
	<0	0.00	
Very large	x>100	0.03	**0.03**
Large	50<x<100	0.78	0.78
Medium	20<x<50	0.18	**0.18**
Small	<20	0.01	**0.01**

Note: Taking into account that costs cannot be negative, the probabilities are in bold.

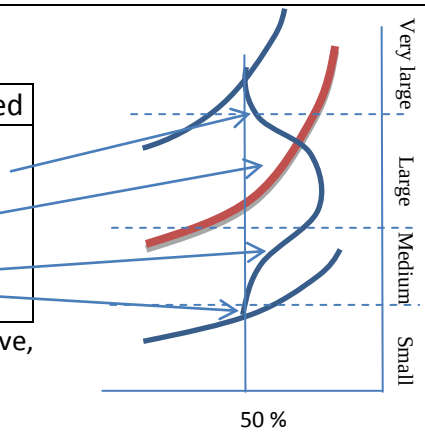


Figure 5. *Deriving intervals for the costs*

Modelling can also be used in filling in the CPTs. In this example, we use modelling as well as expert opinions and existing studies in deriving the CPTs. How to use modelling in creating CPTs is described below in detail.

5.1.2 Utility functions – BBN results

After the CPTs are ready, the model can be run. In our approach the results come out as utilities. Which type of utility function to choose depends on the context. As noted by Levontin et al. (2011), specification of utility functions can be based on information about preferences, risk attitudes, and values in either monetary or other units. Two polar cases in how utility is specified can be found in Barton et al. (2008) and Levontin et al. (2011).

Barton et al. (2008) use BBN to analyse whether GEnS implies disproportionally high costs (compared to benefits) and therefore express their results in a CBA framework and in expected monetary units.

One of the strengths of the BBN is to integrate various disciplines and handle very diverse data, including both quantitative and qualitative data. For such purposes, a utility approach that is not linear in (dependent on) monetary units is appropriate. Levontin et al. (2011) specify all (local) utility functions in their model to be based on hypothesized preferences, such as commercial fishers preferring greater profit and recreational fishers preferring that more salmon reach the rivers. More precisely, utility functions take normalized values: 1 is assigned to states that are most preferred, and zero to those that are unacceptable; intermediary states are given values based on expert judgment. The weakness of this is the arbitrariness of the values attached to the states, e.g., it assumes that the most preferred state has the same utility for all local utility functions.

When using a utility approach, the underlying functional form of the utility function must be agreed on. In economics, a concave function is usually used, expressing the diminishing marginal utility from the variable of interest.

Since in our BBN, all the CPT are stated as intervals, we assume that direct assignment of utilities to each state is most appropriate. In our default setting, we simply use the following “linear” utility assignment (Tables 6 and 7).

Table 6. *Attaching utility to the variable x if x is a benefit*

Groups	Intervals	Utility
Very large	x>100	1
Large	50<x<100	0.75
Medium	20<x<50	0.5
Low	<20	0.25

Table 7. *Attaching utility to the variable x if x is a cost*

Groups	Intervals	Utility
Very large	x>100	-1
Large	50<x<100	-0.75
Medium	20<x<50	-0.5
Low	<20	-0.25

The second issue to note is that we use local utility functions, that is treat the utility from each variable separately, and then simply add up these utility functions.

$$U = u(benefit\ 1) + u(benefit\ 2) + u(benefit\ 3) + u(costs\ 1)$$

This actually implies that all benefits and costs are evaluated compared to a status quo level. In such a setting, the utility function proposed by Kahneman and Tverski might be the most appropriate, which implies concavity in gains and convexity in losses.

This can be represented with the following alternative utility assignment (Tables 8 and 9).

Table 8. *Alternative attachment of utility to the variable x if x is a benefit*

Groups	Intervals	Utility
Very large	x>100	1
Large	50<x<100	0.85
Medium	20<x<50	0.7
Low	<20	0.4

Table 9. *Alternative attachment of utility to the variable x if x is a cost*

Groups	Intervals	Utility
Very large	x>100	-1
Large	50<x<100	-0.85
Medium	20<x<50	-0.7
Low	<20	-0.4

However, since the assignment of utilities is inherently subjective, a second approach is to ask policy makers about the utility assignments to cost and benefits. A major challenge here is to identify who are the right policy makers, and secondly, how to state the questions in a meaningful and understandable way. This approach fits into the idea of the BBN to collect all available information.

5.1.3 Model structure

Studying the costs and benefits of combating eutrophication produces a model structure that has seven parts:

1. The policy options (in our case the BSAP): achievement is based on goals for eutrophication by 2021 in terms of reduced nutrient loads compared to the reference situation against which BSAP nutrient load reduction targets are set

Five conditional probability tables:

2. Effects of policy to cod stocks
3. Benefits gained from changes in the cod stock
4. Benefits gained from improved water clarity
5. Benefits gained through reduced cyanobacteria blooms

6. Costs of implementing the policy

7. CBA presented as utility

The BBN for our study is structured as follows (Figure 6).

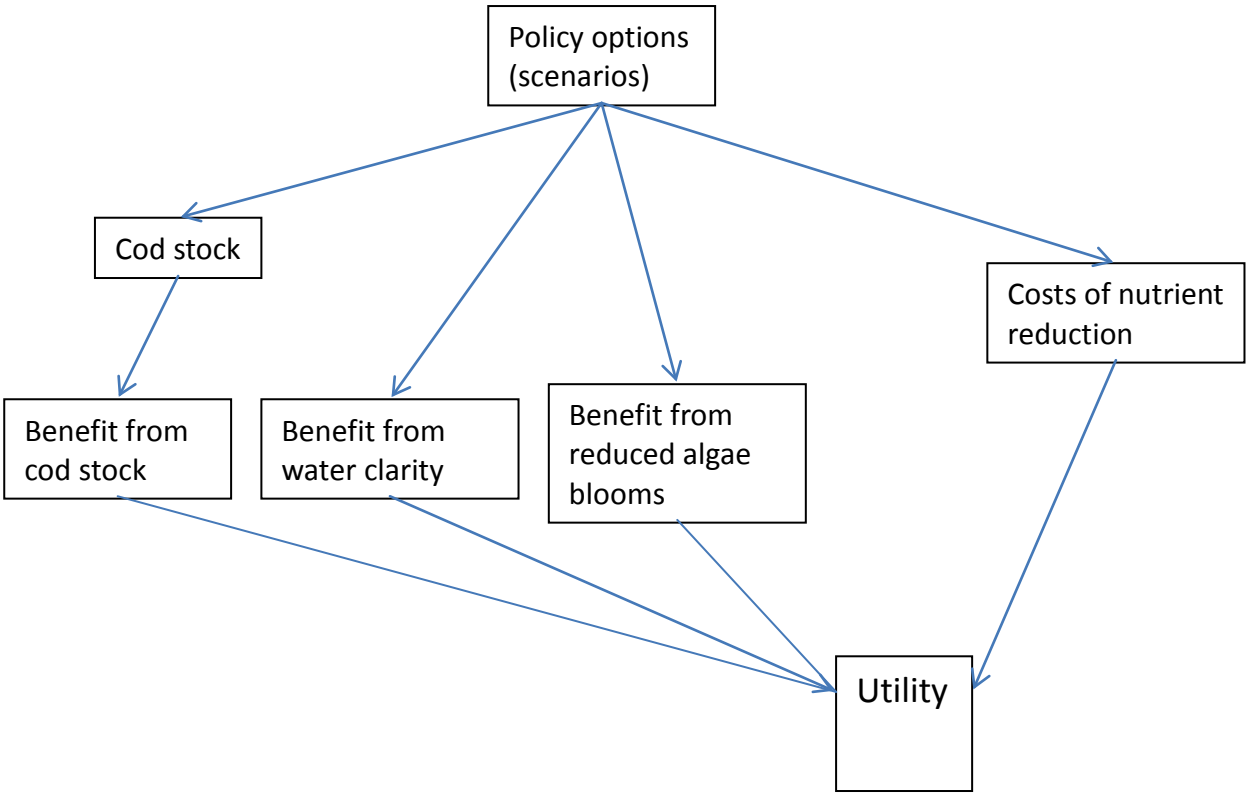


Figure 6. *BBN structure for our case study*

5.2 Policy options

Here we study the costs and benefits of BSAP's eutrophication goals that were described above in section 4. In our study we focus on the Baltic proper for which BSAP has set a goal of reaching a water clarity depth (measured as Secchi depth) of 7 metres. The present water clarity is 6,3 metres, which is our reference condition. In terms of annual nutrient loads, reaching the target requires considerable reductions: 12,500 tonnes of phosphorous and 94,000 tonnes of nitrogen. The average reference loads from 1997 to 2003 were 19,250 tonnes of phosphorous and 327,260 tonnes of nitrogen.

Cost-benefit analysis is a comparison between at least two options. Therefore, one needs to have scenarios to compare. Our approach includes three scenarios that are the same as used in BALTSEM model (see Meier et al. 2011):

REF
The reference scenario simply indicates human pressure as being constant in recent years, based on the average between 1995 and 2002. Riverine loads will however change in response to varying river runoff as described below.

BSAP
In the eutrophication segment of the Baltic Sea Action Plan, the maximum allowable loads for each major sub-basin are presented together with corresponding load reductions compared to a reference period of 1997–2003. In our case we make use of the reductions instead of the maximum allowable loads, which gives slightly different loads. We combine these reductions with a possible further decrease in atmospheric nitrogen deposition of 50%.

BAU
This scenario is based on the assumption of expanding agriculture, especially in transitional countries. In Humborg et al. (2007), the anticipated increase of the loads under these assumptions has been estimated to be about 340,000 tonnes of N per year and 16,000 tonnes of P per year.

The modelling also takes into account climate change as "minimum", "median" and "maximum" values. These values are based on two downscaled climate scenarios (A2, A1B). This climate forcing has been used for the BALTSEM model which was then used again as a forcing for the food-web model (see Meier et al. 2011).

The test model run presented below (section 6) runs for the scenario "BSAP" only, which produces a comparison between a fully implemented BSAP and the situation that prevailed when BSAP was launched in 2007.

5.3 From policy to cod stock

Here we describe the method of filling in a CPT for the cod stock. We utilized a deterministic computer simulation for which the results were extended to probabilistic modelling by using a Gaussian approach. The BNI Ecopath with Ecosim model simulates the food-web dynamics of the Baltic Proper from Bornholm to the Åland islands, excluding the Gulf of Finland and Gulf of Riga (ICES Sub-Divisions (SD) 25-29, without the Gulf of Riga). For forward cod stock projections, the food-web model is combined with a biogeochemical model (BALTSEM) and climate scenarios. The combined model referred herein as BNI-BALTSEM can be run under three scenarios: BAU (business as usual), BSAP (N and P is reduced according to the Baltic Sea Action Plan) and REF (N and P loads are >BSAP<BAU). In this deliverable we present only an initial model run (section 6) that has only one scenario of BSAP that is compared to the situation that prevailed when BSAP was initiated.

Figure 7 shows the observed cod stock biomass in years 1974–2006 and the projected stock size in years 2007–2060. The figure shows that cod stock is slightly higher under the BSAP than under the other scenarios. However, as of 2011, the model's projected cod biomass is significantly lower than the observations. The units are in of number of tonnes of cod in the entire modelled area (i.e. tonnes/$2.4 \cdot 10^5$ km^2).

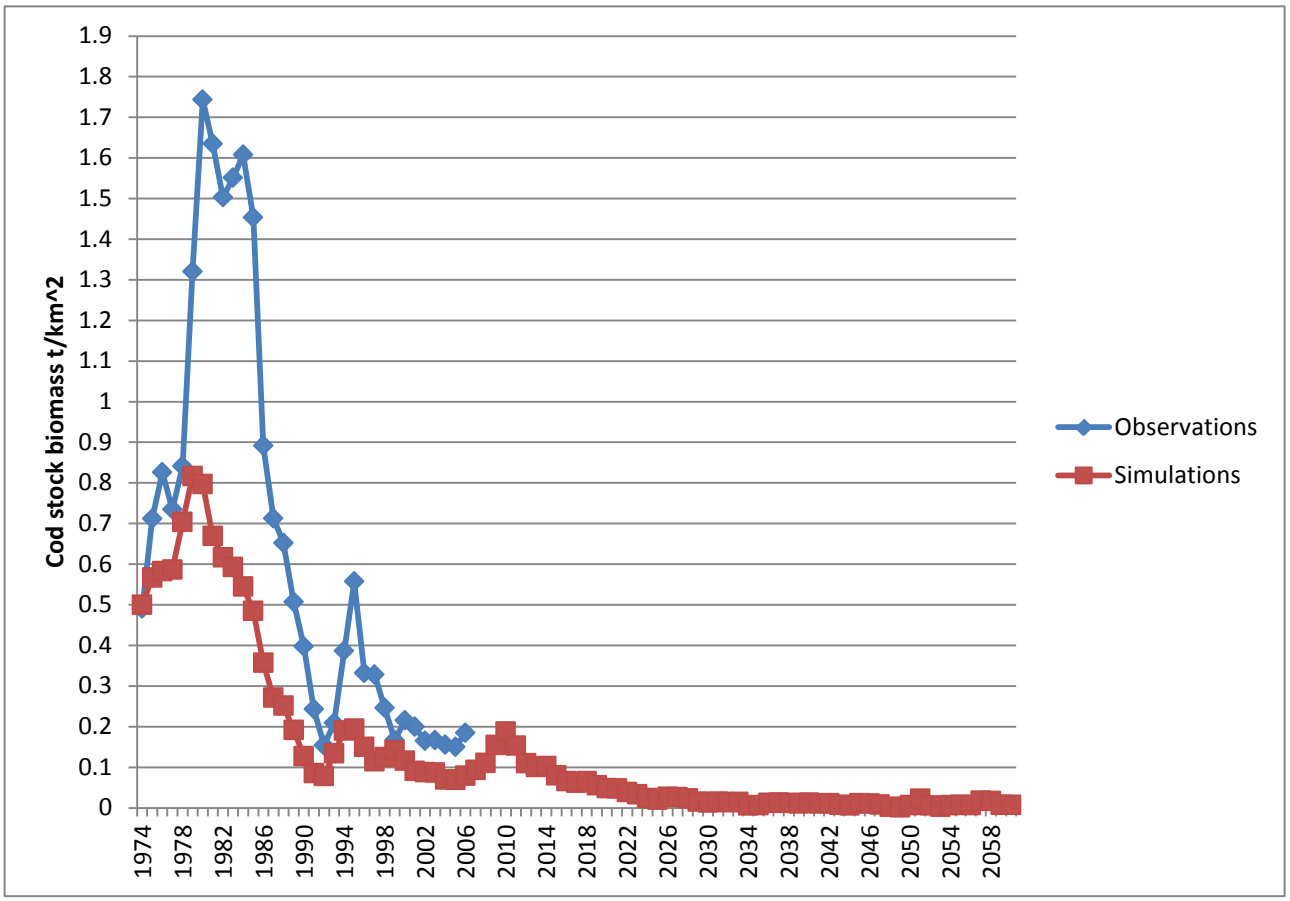


Figure 7. *Observed cod stock biomass in years 1974–2006 and predicted cod stock biomass in tonnes per square kilometre in years 2007–2060*

In order to apply Bayesian Belief Networks (BBN) to the cost-benefit analysis reference points for the cod, stock size is needed. In our approach we try to define indicators that would describe cod stock size when it is low, medium or high. As Figure 7 shows, the model predicts stocks sizes that are lower than historically observed. None of the existing reference points that are based on single-species models opposite to our food-web model provides reference points for adult cod stock size. Thus, we derive the reference points so that cod stock size in year 2021 is considered low if it is lower than 80% of the lowest observed stock size. Cod stock is considered high if it is higher than 50% of the highest observed stock size and is considered medium if the stock size is 50% of the limit of the very high stock size.

Based on the observed cod stock size we derive the limits (reference points) for the states of the cod variable as follows. We consider three states: low, medium and high. The stock size is low if the cod stock is 80% of the observed minimum value (0.12 t/km^2). History shows that from the lowest observed values the stock is able to recover. The stock size is high if it is 50% from the observed maximum (0.872). Finally, the stock size is at a medium level if it is 50% of the high stock size (0.436).

In order to derive the conditional probabilities given the reference points, we need to extend the BNI-BALTSEM model into a probabilistic model so that we can obtain uncertainty estimates for

the predicted cod stock size. We follow Kennedy and O'Hagan (2001) and O'Hagan (2011) and utilize Gaussian processes to model the bias on the simulator and the residual between the simulator prediction and the observations (Figure 7).

First we define some terminology. With *simulator* we mean the BNI-BALTSEM model that produces the cod stock size estimates for future years given some management decision (e.g. future nutrient loadings). The simulator prediction for year t is denoted by s(t). With *observations* we mean the historical cod stock size data. We have the observations, y(t), for years 1974–2006. The *Gaussian process model* stands for the extension of the simulator into a probabilistic model. The Gaussian process model, f(s,t), is a function of the simulator output and time and it is assumed to represent the true stock size.

The simulator, the Gaussian process model and the observations are related as follows:

$$y(t) = a\, s(t) + \rho(t) + \varepsilon$$

where a is the multiplicative bias of the simulator, ρ(t) is a temporally correlated additive bias and ε the observation error. The observation error is assumed to be i.i.d Gaussian, that is

$$\varepsilon \sim N(0, \sigma^2).$$

The multiplicative bias is given a zero mean Gaussian prior

$$a \sim N(0, v^2),$$

and the additive bias a zero mean Gaussian process prior (Rasmussen and Williams, 2006)

$$\rho(t) \sim N(0, K)$$

where the covariance matrix K incorporates the temporal association between different years. It can be represented as a function that describes the decay in correlation between pairs of points with distance in time. We will use the squared exponential covariance function to define the covariance matrix

$$K_{ij} = \exp(- (t_i - t_j)^2 / l^2)$$

where l is the length scale parameter that governs how fast the correlation decreases. We give a prior for the length-scale.

Since both a and ρ(t) are given Gaussian priors the sum a s(t) + ρ(t) is also Gaussian distributed for which reason we call f(s,t) = s(t) + ρ(t) the Gaussian process model.

For the years 1974–2006 we have both the observations y(t) and the simulator predictions. Thus, we can infer the posterior distribution of the parameters σ^2, v^2 and l and the function f(s,t). In practice we use an empirical Bayes approach for the hyperparameters $\Theta = (\sigma^2, v^2, l)$ and fix them into their maximum a posterior estimate (MAP) Θ_{MAP}. After this we evaluate the conditional posterior of f which is Gaussian

$$f(s,t) \mid (y(t), s(t))_{t=1975,\ldots,2006} \sim N(m, \Sigma).$$

Notice that we obtain a posterior distribution over *function* f(s,t). Thus we can evaluate the posterior distribution of f(s,t) for any combination of simulator prediction s and time t.

We present the simulation and the Gaussian calculation that gives a posterior distribution for the cod stock size in Figure 8. The posterior probability distribution is discretized using the reference points and the probability mass falling into each of the bins (CPT) is shown in Table 10.

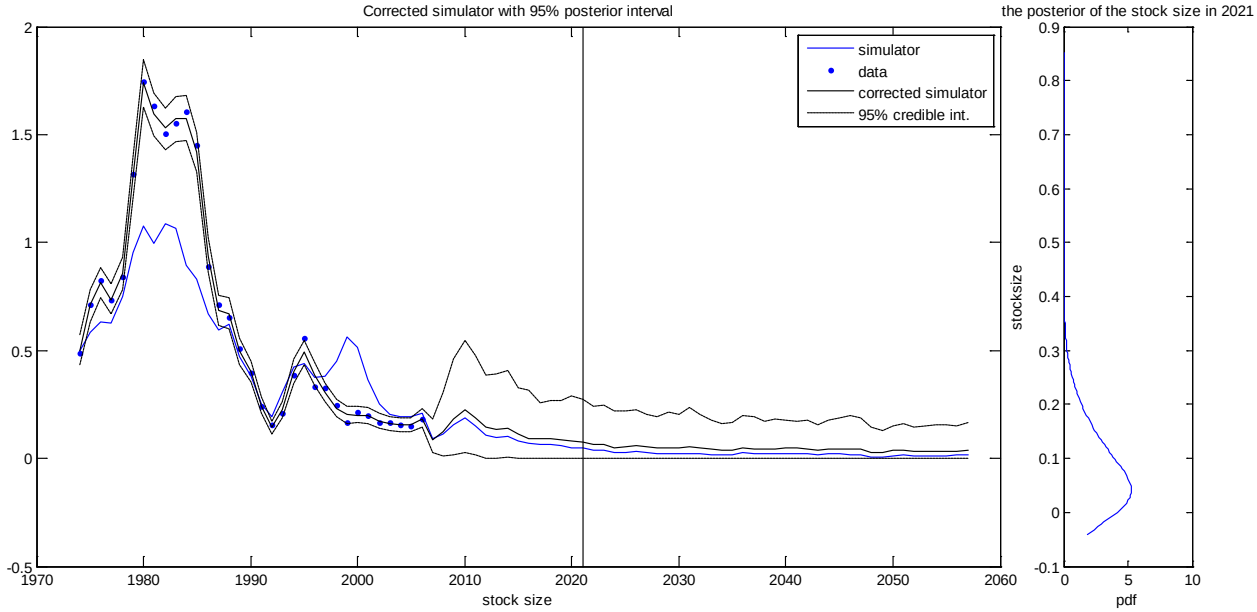


Figure 8. *Stock simulation and correction (graph on the left) and the posterior of the cod stock (graph on the right).*

Table 10. *Conditional probability table for cod stock size. BSAP MED is used for the model run in this deliverable.*

Year 2021	BAU			REF			BSAP		
Reference points	MIN	MED	MAX	MIN	MED	MAX	MIN	MED	MAX
p(stock < 0.12)	0.890	0.770	0.774	0.882	0.780	0.702	0.904	0.776	0.750
p(0.12 < stock < 0.436)	0.110	0.228	0.224	0.118	0.218	0.296	0.096	0.224	0.250
p(0.436< stock < 0.872)	0	0.002	0.002	0	0.002	0.002	0	0.000	0
p(stock > 0.872) *	0	0		0	0	0	0	0.000	0

In future work it is essential that the reference points be elaborated. We will need to consult more of the fisheries experts in order to gain more qualified probabilities. Similarly a model run needs to calculate all scenarios to compare between multiple alternatives. The initial model run in this deliverable uses only the BSAP MED scenario.

5.4 From cod stock to increased benefits

Since there are no modelling or literature studies concerning expected profits from increasing cod stock, we relied on expert judgments only. Value of processed cod (in Poland) and respective probabilities were used to populate that part of our model. Note that using values of processed fish (or landings, which would be considerably lower) as a proxy for profits gives an overestimate as the costs of fishing (or processing) is not counted.

To elicit expert opinion we asked experts to fill in probabilities for each column (except for profit) of the table below (Table 11). We also asked them to fill in the possible profit intervals for the Baltic Proper measured in euros.

Table 11. *CPT for Profits from a given cod stock*

CPT Profit given cod stock	Profits (interval)	Low S < 0.1	Medium 0.1 < S < 0.2	Large 0.2 < S < 0.6	Very large S > 0.6
Very high	220 mill. euro	0.0	0.0	0.4	0.05
High	170 mill. euro	0.1	0.1	0.1	0.25
Medium	150 mill. euro	0.2	0.1	0.0	0.30
Low	>110 mill. euro	0.7	0.8	0.5	0.40

Note that the profits are only for Polish fishers (need to know their share of total profit from cod fishery in BP). Therefore **all values should be scaled** to the whole Baltic Proper (Germany, Sweden, Denmark, Poland, Estonia, Lithuania, Russia, and Latvia). However, our expert states that in his opinion the Polish values can be representative even for the whole main basin because the Polish fleet and industry have a substantial share in the cod industry around the Baltic and land the largest cod catches: in 2006 this was 70,000 tonnes of the total 160,300 tonnes (Burns and Stöhr 2011). However, the profits are very small. Even calculating Net Present Value (NPV) the profits may not exceed 20 billion, which then amounts to the state “Low” in our CPT frame.

Moreover, Thanh (2011) calculates the NPV of the cod fishery in the Eastern Baltic Sea to be in the most optimistic scenario, with optimal harvest and optimal nitrogen level to be of an order of 3 billion euros (2000 prices). We conclude that under no scenario will the profit from the fishery exceed an NPV of 20 billion, giving the following CPT (Table 12).

Table 12. *Profits – BSAP*

States	Intervals	Probability
Very large	Profits > 100	0
Large	50< Profits < 100	0
Medium	20 < Profits < 50	0
Low	Profits < 20	1

5.5 From policy to benefits as clear water

Secchi depth is taken as the indicator for the eutrophication goal for the BSAP. Nutrient reduction allocations in the BSAP are modeled based on desired Secchi depths. A problem with connecting the Secchi depth to benefits is that the modelling handles the open sea conditions (as does the BSAP goal), but benefits are experienced mainly in coastal areas. In the coastal areas the Secchi depth is often lower due to higher loads of sediments and nutrients, which enhance the phytoplankton biomass and by this decrease the Secchi depth. Therefore, when asked about willingness to pay the respondents might have been thinking of coastal water clarity rather than in the open sea. This makes it difficult to combine available modeling directly with WTP results. Therefore, a probabilistic approach is again required.

A large number of studies consider the value or WTP for clean water (covering both Secchi depth and health-related valuation). General considerations can be found in Goffe (1995), Georgiou et al. (1998 and 2000), Bell et al. (2002), Hanley and Kriström (2002), Machado and Mourato (2002), Hoagland et al. (2006), and finally Tuhkanen et al. (2010) for a study about public attitudes toward clean water. Specific studies can be found in Atkin et al. (2000) for Denmark, EFTEC (2002) for bathing water quality in Wales and England, Hoagland et al. (2002) for the US, and Sandström (1996) and Soutukorva (2001) for Sweden.

We choose the method to derive the CPT that is described in earlier sections. For now, we rely on only two studies, but as already discussed the BBN allows us to revise our estimates as new information becomes available. Obviously, we could ask some of the experts listed above.

We do not have any direct information about the likely change in water clarity due to the proposed policy measures, and therefore we again ask the experts in our group about their opinion about the probability that specific Secchi depth targets will be reached in 2012 for the three policy options we analyse. These probabilities based on expert opinion are presented in Table 13. The actual Secchi depth in the Baltic Proper is 6.3 m and the target is 7 m as stated for BSAP.

Table 13. *Expert opinion on changes in Secchi depth*

Secchi depth	BAU (2021)	REF (2021)	BSAP (2021)
8	0	0	0
7.5	0	0	0
7 (target)	0	0	0.3
6.5	0.5	0.8	0.6
6	0.5	0.2	0.1

Since we essentially need to measure the changes that a policy implies compared to a situation without that policy, the benefit from implementing the BSAP should be compared to the reference situation REF.

For the BSAP, we see that there is only a 30% probability that the target will be reached in 2021, and a 70% probability of small or non-increase in Secchi depth. Therefore, taking this into account will reduce the benefit from water clarity significantly compared to the deterministic case. Assuming linearity, the change in water clarity follows three paths, as shown in Figure 9, where the y-axis indicates changes in the Secchi depth relative to 6.3 meters.

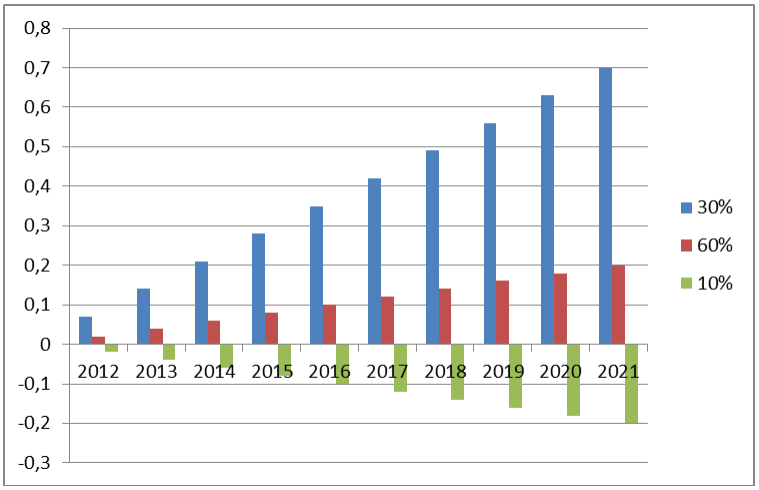


Figure 9. *Change in Secchi depth over time in the three possible scenarios*

Our first willingness to pay (WTP) estimate is based on the study of Vesterinen et al. (2010), a travel costs analysis of WTP for clearer water in coast of Finland (we call this Study 1). In the paper they present a low and a high value, replicated in Table 14.

Table 14. *Benefit from + 1 meter in Secchi depth (Vesterinen et al., 2010)*

	Low	High	Average
Per person			37.76
(euro/person/year)	18.79	56.72	

To find the NPV for the whole Baltic Sea (here we include all 147 million people), we multiply the numbers from Table 14 with 147 million (assuming that this benefit transfer is valid), and take an average of the low and high values. Finally we attach the weights shown in Table 15.

Table 15. *Calculations for Study 1*

Year	30%	60%	10%	Weighted	Average benefits (per person)	Total benefits (billion euro)
2012	0.07	0.02	-0.02	0.031	4.56	0.67
2013	0.14	004	-0.04	0.062	9.13	1.34
2014	0.21	0.06	-0.06	0.093	13.69	2.01
2015	0.28	0.08	-0.08	0.124	18.26	2.69
2016	0.35	0.1	-0.1	0.155	22.82	3.36
2017	0.42	0.12	-0.12	0.186	27.39	4.03
2018	0.49	0.14	-0.14	0.217	31.95	4.70
2019	0.56	0.16	-0.16	0.248	36.52	5.37
2020	0.63	0.18	-0.18	0.279	41.08	6.04
2021	0.7	0.2	-0.2	0.31	45.65	6.72
					NPV	**28.20**

For example, in year 2013, the expected increase in Secchi depth will be 6.2 cm, which gives an average benefit of 9.13 euro per person. Multiplying by 147 million yields a total expected benefit of 1.34 billion euro. In total, the NPV for a 4% discount rate over 10 years yields 28.2 euro. However, such "benefit transfer" includes large uncertainties (Luisetti and Turner, 2011a).

Now we assume that a benefit extracted from a study in Finland is directly applicable to other parts of the Baltic Sea. The uncertainty included here can be illustrated by results from a large survey reported in 2010 (Söderqvist et al., 2010a and 2010b). The results show differences between countries in people's understanding of the state of the Baltic Sea in general. There is also a question about how the impact of bad water quality on recreational use differs between countries. A more accurate benefit transfer would thus require further analysis for its application; for instance, by using other research to extract weightings for transferring values from one country to another. In this study we use a direct benefit transfer.

Study 2 is the study reported in the deterministic result, giving a Baltic Sea–wide yearly benefit of clear water of 13.203 billion euro (Luisetti and Turner, 2011). Assuming that the target of the BSAP, which is based on good environmental status, will give clear water, as defined in Study 2, we use the weights from Table 15 to calculate the expected benefits over the next 10 years and calculate the NPV (Table 16).

Table 16. *Calculations for Study 2*

Year	Weights	Benefits (billion euro)
2012	0.031	0.41
2013	0.062	0.82
2014	0.093	1.23
2015	0.124	1.64
2016	0.155	2.05
2017	0.186	2.46
2018	0.217	2.87
2019	0.248	3.27
2020	0.279	3.68
2021	0.31	4.09
	NPV	17.19

Now we have two studies for the same problem giving two results:

	NPV (billion euro)
Study 1	28.20
Study 2	17.19

Using the methodology and putting equal weight to the two studies yields the following estimates

$$\hat{\mu} = 22.7$$
$$\sigma^2 = 30.31$$
$$\sigma = 5.51$$

The next step is to find relevant intervals for the benefits, and then, given the estimated parameters of the normal distribution, calculate the probability that the benefit falls into each interval, which then gives the following CPT (Table 17).

Table 17. *Benefit - Secchi Depth - BSAP*

States	Intervals	Probability
Very large	Benefits > 100	0.000
Large	50< Benefits < 100	0.000
Medium	20 < Benefits < 50	0.688
Low	Benefits < 20	0.312

5.6 From policy to benefits as reduced risk of algae blooms

Links from existing models to benefits gained from reduced cyanobacteria are very difficult to gain. Cyanobacteria blooms are a natural phenomenon in the Baltic Sea (Bianchi et al., 2000), but anthropogenic nutrient emissions are increasing the frequency of their occurrence.

Algal blooms are a natural phenomenon in the Baltic Sea, which makes it complicated to quantify expected benefits of nutrient emission reductions in the form of reduced algae blooms as one cannot ask how much people are willing to pay if there were no algae blooms at all. The question would be phrased incorrectly as such a situation is not possible to reach. We will, therefore, approach the question as the likelihood for occurrence of negative health effects, which corresponds more closely to the natural conditions in the Baltic Sea.

Kosenius (2010) estimates the minimum yearly household WTP for scenarios with a focus on the reduction of blue-green algal blooms at 210 euros. Given the household size (2.34), WTPs for this scenario is about 70.5 euros per person per year respectively. Table 18 shows the aggregated WTP.

Table 18. *WTP for reduced Blue-green algae bloom in Baltic Sea*

Countries	Population at 2011*	WTP chosen (euro/person/year)	WTP aggregated for the Baltic Sea and related countries (million euro/year)
Finland	5,375,276	70.5	379
Sweden	9,415,570	70.5	664
Estonia	1,340,194	70.5	95
Latvia	2,229,641	70.5	157
Lithuania	3,244,601	70.5	229
Poland	38,200,037	70.5	2694
Germany	8,1748,892	70.5	5764
Denmark	5,560,628	70.5	392
Baltic Sea			**10,373**

*Data from Luisetti and Turner, 2011.

As explained in section 2, there are many factors that create delays in the response of the Baltic Sea ecosystem to reduction of nutrient loads. Therefore, benefits will emerge slowly. The exact timing of reduction in blue-green algal bloom is uncertain and depends on, among other things, unpredictable conditions like natural and possibly man-made variations in climate. We therefore use the method described in 5.5 and asked for direct expert opinion from our group about the timing (occurrence of the benefits) following the method adopted by Uusitalo et al. (2005).

Table 19. *Expert opinion on negative health effects in 2021 from three policy scenarios*

Negative health effects	BAU (2021)	REF (2021)	BSAP (2021)
150%	0.8	0	0
125%	0.2	0	0
100%	0	1	0
75%	0	0	0.2
50%	0	0	0.6
25%	0	0	0.2
0%	0	0	0

We asked our experts to fill probabilities into the table above. By negative health effect we mean changes compared to the reference, that is, the present situation. Zero percent means no negative health effects, which corresponds to the WTP estimates from Kosenius (2010), while more than 100% indicates a worsening. The probabilities reflect the experts' best guess about the state of blue-green algae bloom related negative health effects in 2021 for the three possible policy scenarios.

Calculating the CPT for the BSAP

We assume linearity in the way the benefits materialize. As an example, for the BSAP our experts state that there is a probability of 60% that in 2021 the negative health effects will have been reduced to 50%, which then given linearity implies an improvement of 5 % every year. For the valuation, this means that in year 1, benefits are only 5% of the values in Kosenius (2010), in year 2 10 % and so on, for the 60% scenario. (This also means that we assume that benefits are linear in changes in negative health effects). The development in benefits for three scenarios is shown in Figure 10.

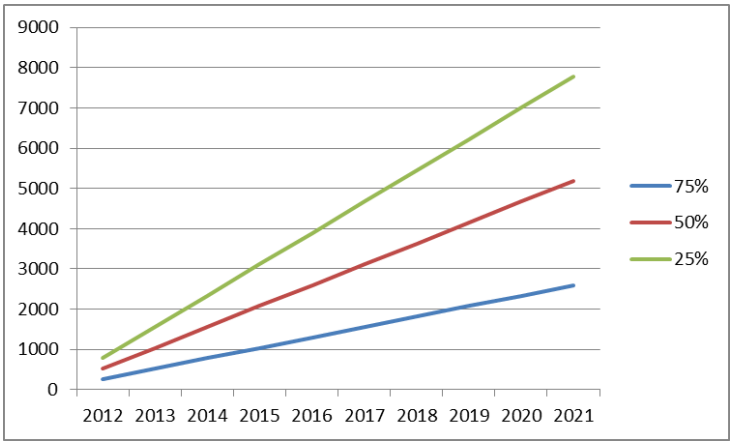


Figure 10. *Development in benefits for three scenarios*

For the three relevant cases in the BSAP, the NPV given discount rate of 4% is also calculated as

Scenario	75%	50%	25%
NPV	10889.64	21779.28	32668.92

Since we have no other study included, we build our CPT from this data, and it indicates that there is an 80% chance that benefits will be medium, and 20% that they will be low. The CPT is shown in Table 20.

Table 20. *Benefit - Secchi Depth - BSAP*

States	Intervals	Probability
Very large	Benefits > 100	0
Large	50< Benefits < 100	0
Medium	20 < Benefits < 50	0.8
Low	Benefits < 20	0.2

5.7 Costs of implementing policies

First we need to find the functional relationship between reductions and costs. Several papers argue that the costs (both total and marginal) are convex, increasing with the level of reduction. See Elofsson (2010) for a general overview. We take as a point of departure the results from Gren (2008) on the nitrogen reduction to a marine basin (Baltic Proper) presented in Table 21.

Table 21. C*ost of reducing N and P*

N reduction (%)	Per year costs	NPV	P reduction (%)	Per year costs	NPV
0	0	**0**	0	0	0
10	64	**519**	10	75	608
20	197	**1598**	20	288	2336
30	479	**3885**	30	720	5840
40	1131	**9173**	40	1493	12110
50	2504	**20310**	50	2405	19507

For example, the Net Present Cost (NPC) of a 50% reduction of the inflow of both P and N to the Baltic Proper (measured as a 10 year NPV and discount rate of 4%) and assuming all the measures are in place at period 1 would be 39.8 billion euros (assuming steady state values in millions of euro) (Gren 2008, table D2).

Figure 11 shows functional forms for NPV of for various levels of P and N. For both pollutants, reduction above 25% implies high marginal reduction costs, and therefore a steep increase in total costs of reduction at such high levels.

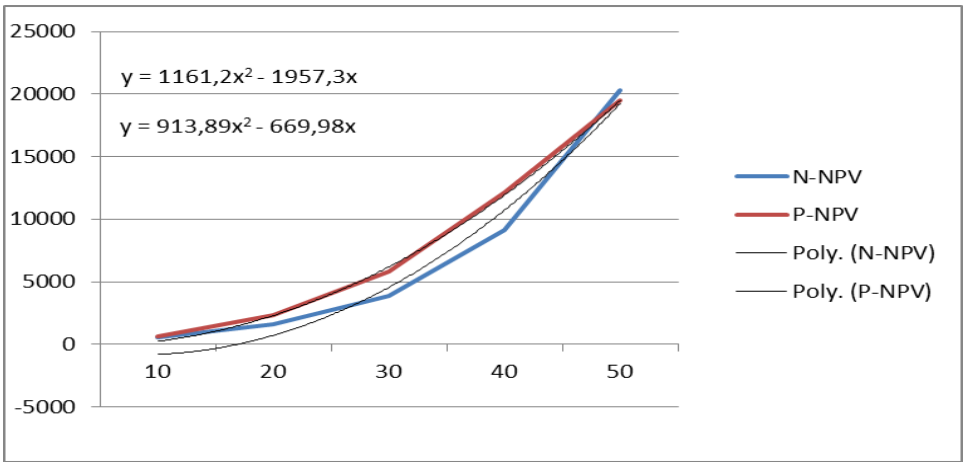


Figure 11. *Graphical view of one study's costs of reducing N and P. (Gren, 2008).*

A method of deriving the costs is explained above in section 4.1. Here we found three studies that either report costs for implementing the BSAP or can be used to calculate the costs. Cowi (2007) gives an estimate of 3 billion euro per year to implement the BSAP, while Hasler et al. (2012) provide an estimate of 3.65 billion euros. Gren (2008) has calculated the costs of reducing N and P, which is reproduced in Table 4 (in section 4.1).

Table 22. *Summary of cost calculation for N and P reductions*

N-NPV	$y = 913{,}89x^2 - 669{,}98x$	reduction 29 =	5745,715
P-NPV	$y = 1161{,}2x^2 - 1957{,}3x$	reduction 65 =	36507,25
		Total NPV	42252,96

From the reported numbers, we can find the best fit for nitrogen given by $y = 913.89x^2 - 669.98x$, where y measures the costs and x is the percentage reduction of N. For the BSAP we need a 29% reduction in N, giving a NPV of costs of 5.74 billion euro. For P we find: $y = 1161.3x^2 - 1957.3x$. Here we need a 65% reduction, giving NPV for costs of 36.5 billion euro. NPV is calculated over 10 years with a discount rate of 4%. The results are shown in Table 23.

Table 23. *NPV of implementing BSAP in BP*

	per Year	NPV
Gren (2008)		42.25
Hasler et al. (2012)*	3,867	31.45
Cowi(2007)	3	24.33

*RECOCA project's draft results

We find no reason to weight the studies differently according to the criteria stated in the methodology section, even though the Hasler et al. (2012) estimate is only a preliminary estimate. Using the methodology and putting equal weight to the two studies yields the following estimates:

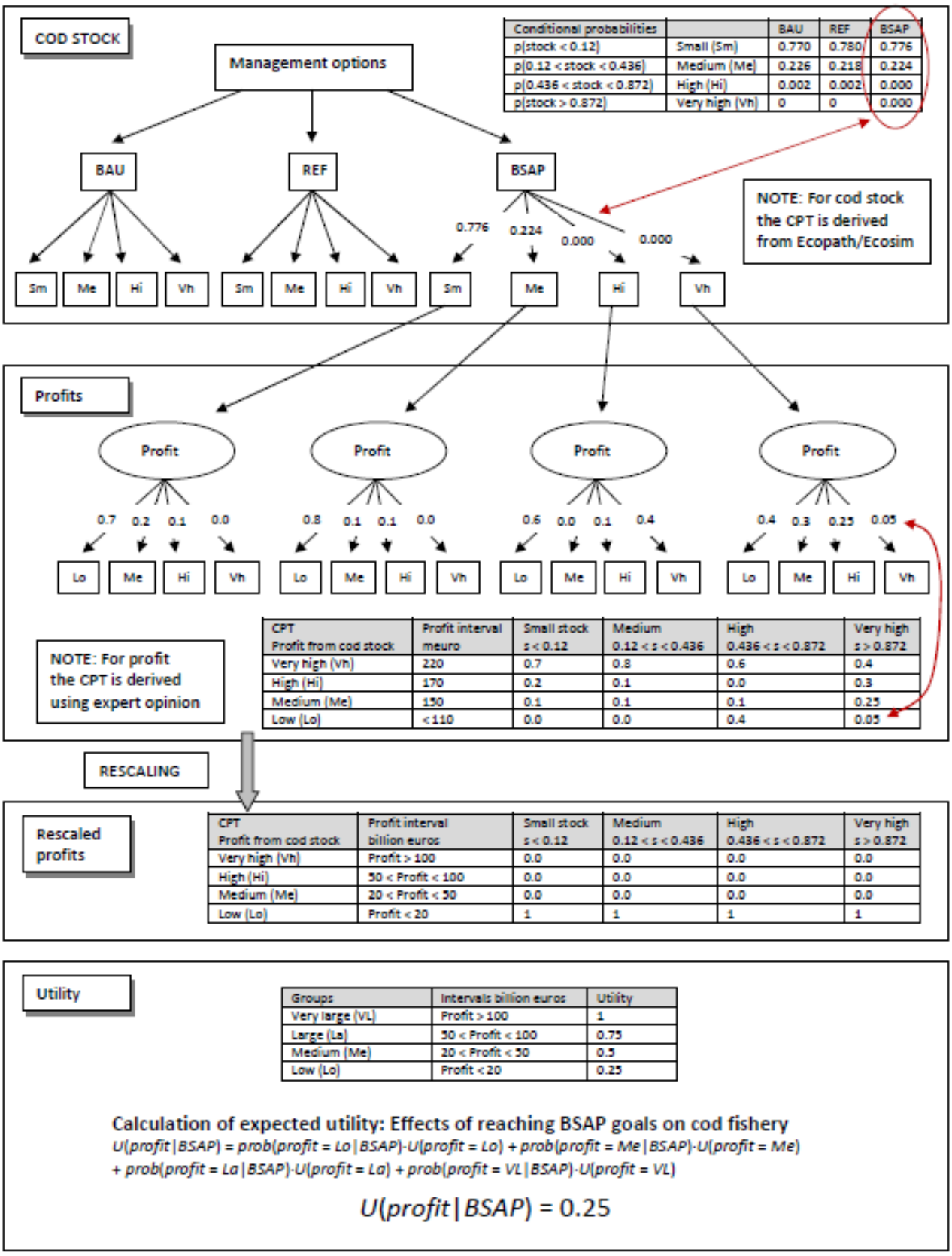
$\hat{\mu} = 32{,}68$
$\sigma^2 = 54{,}27$
$\sigma = 7{,}37$

The next step is to find relevant intervals for the costs, and then given the estimated parameters of the normal distribution calculate the probability that the cost falls into each interval, which then gives the following CPT:

Table 24: *Costs - BSAP*

States	Intervals	Probability
Very large	Costs > 100	**0**
Large	50< Costs < 100	**0.009**
Medium	20 < Costs < 50	**0.948**
Low	Costs < 20	**0.043**

The following figure (Figure 12) summarises our BBN method by using the sequence from cod stock to expected utility as an example.



Conditional probabilities		BAU	REF	BSAP
p(stock < 0.12)	Small (Sm)	0.770	0.780	0.776
p(0.12 < stock < 0.436)	Medium (Me)	0.226	0.218	0.224
p(0.436 < stock < 0.872)	High (Hi)	0.002	0.002	0.000
p(stock > 0.872)	Very high (Vh)	0	0	0.000

CPT Profit from cod stock	Profit interval meuro	Small stock s < 0.12	Medium 0.12 < s < 0.436	High 0.436 < s < 0.872	Very high s > 0.872
Very high (Vh)	220	0.7	0.8	0.6	0.4
High (Hi)	170	0.2	0.1	0.0	0.3
Medium (Me)	150	0.1	0.1	0.1	0.25
Low (Lo)	< 110	0.0	0.0	0.4	0.05

CPT Profit from cod stock	Profit interval billion euros	Small stock s < 0.12	Medium 0.12 < s < 0.436	High 0.436 < s < 0.872	Very high s > 0.872
Very high (Vh)	Profit > 100	0.0	0.0	0.0	0.0
High (Hi)	50 < Profit < 100	0.0	0.0	0.0	0.0
Medium (Me)	20 < Profit < 50	0.0	0.0	0.0	0.0
Low (Lo)	Profit < 20	1	1	1	1

Groups	Intervals billion euros	Utility
Very large (VL)	Profit > 100	1
Large (La)	50 < Profit < 100	0.75
Medium (Me)	20 < Profit < 50	0.5
Low (Lo)	Profit < 20	0.25

Figure 12. *The sequencing of BBN approach: example with cod by using BSAP MED scenario*

6. The final model and interpretation of the model run

Here we present the output of our methodology development and results of an initial model run. The final BBN is as follows (Figure 13).

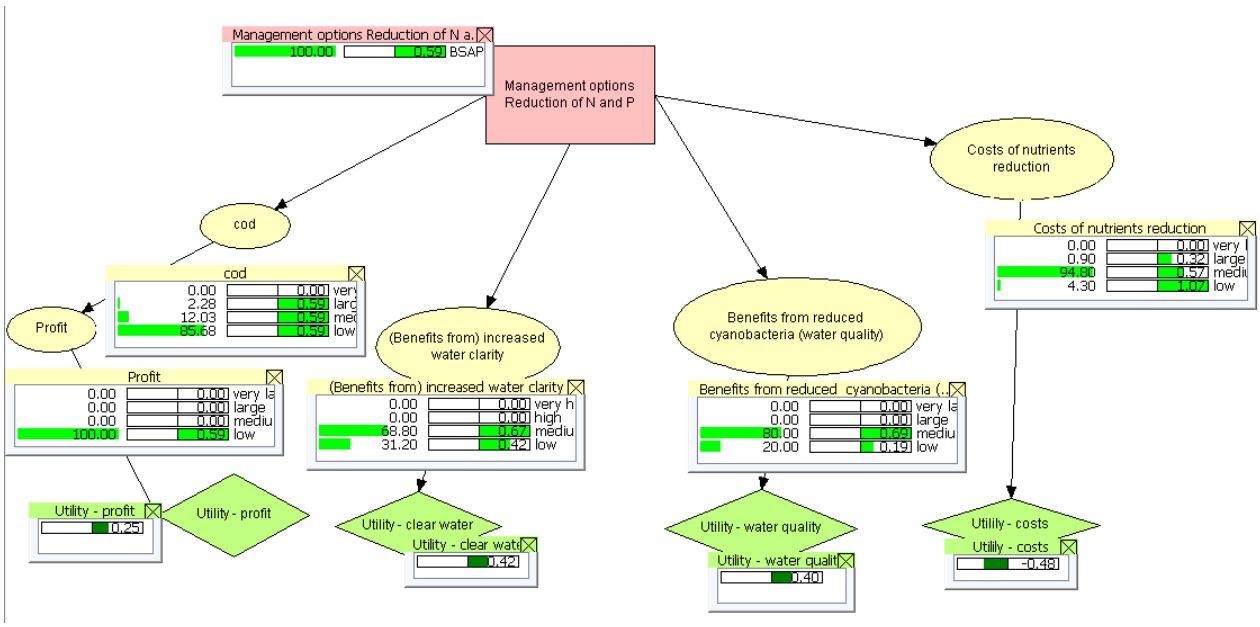


Figure 13. *Our BBN model*

Table 25 summarises the methods used and presents results that are further discussed below.

Table 25. *Summary of method, the CPTs and the results from the BBN*

Variables	Approach	CPT for BSAP			Utility	
Costs	Use existing studies to generate CPT	States	Intervals	Probability	$U(Costs\|BSAP)$	-0.48
		Very large	Costs > 100	0		
		Large	50< Costs < 100	0,009		
		Medium	20 < Costs < 50	0,948		
		Low	Costs < 20	0,043		
Profit from cod	Use direct expert opinion (See Figure 11 for the structure to do this)	States	Intervals	Probability	$U(profit\ cod\|BSAP)$	0.25
		Very large	Profits > 100	0		
		Large	50< Profits < 100	0		
		Medium	20 < Profits < 50	0		
		Low	Profits < 20	1		
Benefit from improved water clarity	Use direct expert opinion combined with existing studies and benefit transfers to generate CPT	States	Intervals	Probability	$U(clear\ water\|BSAP)$	0.42
		Very large	Benefits > 100	0.000		
		Large	50< Benefits < 100	0.000		
		Medium	20 < Benefits < 50	0.688		
		Low	Benefits < 20	0.312		
Benefit from reduced negative health effects	Use direct expert opinion and benefit transfers for benefit	States	Intervals	Probability	$U(Health\|BSAP)$	0.40
		Very large	Benefits > 100	0		
		Large	50< Benefits < 100	0		
		Medium	20 < Benefits < 50	0.8		
		Low	Benefits < 20	0.2		
Management: BSAP					U(BSAP)	**0.59**

The CPT values presented in Table 23 were used in a model run. The model calculated a scenario that the nutrient load reduction goal within the BSAP will be achieved in 2021. This leaves as an alternative scenario the situation that existed when BSAP was designed. The model works on the change that occurs as a result of implementing the BSAP.

Our initial model run presented in this deliverable predicts that compared to a reference situation, the BSAP gives a positive benefit to society. The costs of implementing the necessary policy measures are valued lower than the benefits these policies generate in utility terms.

In order to put confidence into this result, it will be appropriate to make a sensitivity analysis, to judge the robustness of the result with respect to essential parameters and methods. As already discussed, the effect of a change in utility assignments will be important. If the result is not changed much for other types of utility assignments, this greatly increases the confidence in the result.

To see the importance of robustness analysis, consider the following scenario: The values for the benefits are highly uncertain, and are very dependent on whether the whole population of Germany is included in a benefit transfer calculation (see section 5.5). For the Secchi depth benefit, if

Germany were excluded from the calculations, the benefits would now fall 100% into the Low state, reducing utility from clear water by 0.17. The same underlying assumption is also done for the calculations of the reduction in health-related effects. If Germany is excluded, then as before 100% probability for the Low state will result. Thirdly, for the cost side, cost efficiency is assumed, but this assumption generates a minimum on the costs typically far from the real costs. If we scale up all costs by a factor of 2 then the (dis-)utility from costs would increase to -0.76. Adding these changes up to total utility, reveals that in this scenario, the BSAP produces a negative value to society, even though very small, as shown in Table 26.

Table 26. *Basic and alternative results for the BBN – model*

Variables	Utility	Basic run	Alternative scenario
Costs	$U(Costs\|BSAP)$	-0.48	-0.76
Profit from cod	$U(profit\ cod\|BSAP)$	0.25	0.25
Benefit from improved water clarity	$U(clear\ water\|BSAP)$	0.42	0.25
Benefit from reduced negative health effects	$U(Health\|BSAP)$	0.40	0.25
Management: BSAP	U(BSAP)	**0.59**	**-0.01**

This highlights the strengths of using a BBN. First of all, it can handle many very uneven variables at the same time and also handle the attached uncertainties; therefore it also makes it explicit where the critical assumptions are, as seen from the above considerations.

The result is only a first attempt to use the BBN and assess its applicability for this case. The group working on this has found it extremely fruitful to have the BBN structure to integrate the interdisciplinary nature of the problem and the researchers involved. Still, many improvements are needed.

Improvements that could validate our finding are:

- To have more experts' judgments to account for disagreements over probability statements
- To use Monte Carlo simulations to derive CPT for cod
- To have more valuation studies
- Include WTP estimates for the fish stock

It might seem that in this study the intervals for the states were too large, but this is because we only looked ten years ahead. If we take a longer perspective, e.g. forty years, NPV for both costs and benefits could increase to over 100 billion euro.

7. References

Atkins, J. P. and D. Burdon 2006. "An initial economic evaluation of water quality improvements in the Randers Fjord, Denmark." *Marine Pollution Bulletin* 53: 195-204.

Backer, H., J.-M. Leppänen, et al. 2010. "HELCOM Baltic Sea Action Plan - A regional programme of measures for the marine environment based on the Ecosystem Approach." *Marine Pollution Bulletin* 60: 642-649.

Barton, D. N., T. Saloranta, S.J. Moe, H.O. Eggestad and S. Kuikka (2008), 'Bayesian belief networks as a meta-modelling tool in integrated river basin management; pros and cons in evaluating nutrient abatement decisions under uncertainty in a Norwegian river basin', *Ecological Economics*, 66: 91–104.

Bell D., Hanley N., Alvarez-Farizo B. 2002. "Valuing the benefits of coastal water quality improvements using contingent and real behaviour". Discussion papers in Economics no. 0209, University of Glasgow

Bianchi, T.S., Engelhaupt, E. Westman, P., Andren, T., Rolff, C. & R. Elmgren 2000. Cyanobacterial bloom in the Baltic Sea: Natural or human-induced. *Limnol. Oceanogr.* 45: 716-726

Borsuk, M.E, C.A Stow and K.H. Reckhow 2004. 'A Bayesian network of eutrophication models for synthesis, prediction and uncertainty analysis'. *Ecological Modelling*, 173: 219-239.

Brandt, U.S. and N. Vestergaard (2011), 'Assessing Risk and Uncertainty in Fisheries Rebuilding Plans, IME working Paper, 107/11, University of Southern Denmark, Campus Esbjerg.

Bromley, J., N.A. Jackson, O.J. Clymer, A.M Giacomello and F.V. Jensen 2005,' The use of Hugin ® to develop Bayesian networks as an integrated water resource planning'. *Environmental Modelling & Software*, 20: 231-242.

BSAP 2007. "An approach to set country-wise nutrient reduction alloca tions to reach good marine environment of the Baltic Sea.", Helcom BSAP Eutro Expo/2007, Helsinki Commission, Helsinki, Finland

Burgman, M., F. Fidler, M. McBride, T. Walshe and B. Wintle 2006. 'Eliciting Expert Judgments: Literature Review', Report, Australian Center for Excellence in Risk Analysis.

Burns, Tom R. and Christian Stöhr 2011. Power, knowledge, and conflict in the shaping of commons governance. The case of EU Baltic fisheries. *International Journal of the Commons*, 5: 233-258.

Casini, M., Lövgren, J., Hjelm, J., Cardinale, M., Molinero J-C., and Kornilovs, G. 2007. Multi-level trophic cascades in a heavily exploited open marine ecosystem. *Proc. R. Soc. B.* doi: 1 0.1098/ rspb.2007.1752 Published online.

Daniel J. Conley, Svante Björck, Erik Bonsdorff, Jacob Carstensen, Georgia Destouni, Bo G.

Gustafsson, Susanna Hietanen, Marloes Kortekaas, Harri Kuosa, H. E. Markus Meier, Baerbel Müller-Karulis, Kjell Nordberg, Alf Norkko, Gertrud Nürnberg, Heikki Pitkaänen, Nancy N. Rabalais, Rutger Rosenberg, Oleg P. Savchuk, Caroline P. Slomp, Maren Voss, Fredrik Wulff, and Lovisa Zillén 2009. Hypoxia-Related Processes in the Baltic Sea. *Environmental Science and Technology*, 43: 3412-3420.

Constanza, R., R. d'Arge, R. deGroot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, G. R. Raskin, P. Sutton, and M. Van den Belt 1997. The value of the world's ecosystem services and natural capital. *Nature* 387**:**253–260

Cowi 2007. 'Economic analysis of the BSAP with focus on eutrophication: Economic analysis of the BSAP: Analysis of HELCOM scenarios', final report

Dawes, R., D. Faust and P.E. Meehl 1989. 'Clinical versus actuarial judgment', *Science*, 243: 1668-1674.

EFTEC 2002. "Valuation of Benefits to England and Wales of a Revised Bathing Water Quality Directive and Other Beach Characteristics Using the Choice Experiment Methodology", Economics for the Environment Consultancy Ltd

Ellison, A.M. 1996. 'An introduction to Bayesian inference for ecological research and environmental decision-making', *Ecological Applications*, 6: 1036-1046.

Elofsson, K. 2010. Cost-effectiveness of the Baltic Sea Action Plan. *Marine Policy* 34: 1043–1050.

Georgiou et al. 2000. "Coastal bathing water health risks: assessing the adequacy of proposals to amend the 1976 EC Directive". *Risk, Decision and Policy*, 5: 49-68.

Georgiou S., Langford I.H., Bateman I. Turner R.K. 1998. "Determinants of individual willingness to pay for reductions in environmental health risks: a case study of bathing water quality" *Environment and Planning A*, 30: 577-594

Goffe P. Le 1995. "The Benefits of Improvements in Coastal Water Quality: A Contingent Approach", *Journal of Environmental Management*, 45: 305–317

Gowdu, J.M. and Carbonell, A.F. 1999. Toward consilience between biology and economics: the contribution of ecological economics. *Ecological economics*, 29: 337-348

Gren, I. n.-M. (2008). Costs and benefits from nutrient reductions to the Baltic Sea, Swedish Environmental Protection Agency:

Haapasaari, P., S. Kulmala, and S. Kuikka. 2012. Growing into Interdisciplinarity: How to Converge Biology, Economics, and Social Science in Fisheries Research? *Ecology and Society*, 17: 6.

Hamilton, G, E. Abel, C. Alston, T. Chiffings, B. Hart and K. Mengersen 2005. 'Integrating science through Bayesian Belief Networks: case study of Lyngbya in Moreton Bay. In: Zerger, A. and

Argent, R. (Eds.) Proceedings of International Congress on Modelling and Simulation, Modelling and Simulation Society of Australia and New Zealand Inc, Melbourne, Victoria, pp. 392-399.

Hasler, Berit, James C.R. Smart and Anders Fonnesbech-Wulff 2012. Costs to reduce nutrient loads to the Baltic Sea. Some scenarios and examples from the model BALTCOST. http://balticnest.org/download/18.6018c17913483dc064280001063/RECOCA+_costs_BeritHasler_JimSmart_draftresultsinserted.pdf (accessed February 20, 2012)

HELCOM 2007. HELCOM Baltic Sea Action Plan, Baltic Marine Environment Protection Commission.

HELCOM 2009. Eutrophication in the Baltic Sea - An integrated thematic assessment of the effects of nutrient enrichment and eutrophication in the Baltic Sea region. Baltic Sea Environmental Proceedings. Helsinki, Baltic Marine Environment Protection Commission**:** 148.

HELCOM 2010. "Progress in implementing the Eutrophication Segment of the HELCOM Baltic Sea Action Plan." Retrieved 15/12, 2011, from http://www.helcom.fi/stc/files/Moscow2010/EutrophicationSegment.pdf.

HELCOM. 2004. Nutrient pollution to the Baltic Sea in 2000. Baltic Sea Environment Proceedings, 100

Hoagland P., Anderson D.M., Kaoru Y., White A.W. 2002. “The economic effects of harmful algal blooms in the United States: Estimates, assessment issues, and information needs.”. *Estuaries*, 25: 819-837

Hoagland P., Scatasta S. 2006. “The economic effects of harmful algal blooms. In: *Ecology of Harmful Algae”*, [E. Graneli and J. Turner (eds.)] Springer-Verlag, Dordrecht, The Netherlands, Chap. 29

Iho, A. and M. Laukkanen 2009. ‘Dynamically Optimal Phosphorus Management and Agricultural Water Protection’, Discussion Papers 54285, MTT Agrifood Research Finland.

Kennedy, M. and O'Hagan, A. 2001. Bayesian calibration of computer models (with discussion). *Journal of the Royal Statistical Society*, Series B. 63, 425-464.

Kosenius A.-K. 2010. “Heterogeneous preferences for water quality attributes: The Case of eutrophication in the Gulf of Finland, the Baltic Sea”, *Ecological Economics*, 69: 528–538.

Köster, F.W., Möllmann, C., Neuenfeldt, S., Vinther, M., St John, M.A., Tomkiewicz, J., Voss, R., Hinrichsen, H.-H., Kraus, G. and Schnack, D. 2003. Fish stock development in the Central Baltic Sea (1976-2000) in relation to variability in the environment. *ICES Journal of Marine Science*, 219: 294-306.

Kuhnert, P.M, T.G. Martin, K. Mengersen and H.P. Possingham 2005. ‘Assessing the impacts of grazing levels on bird density in woodland habitat: a Bayesian approach using expert opinion’, *Environmetrics*, 16: 717–747.

Lindegren, M., Möllmann, C., Nielsen, A. and Stenseth, N. C. 2009. Preventing the Collapse of the Baltic Cod Stock through an Ecosystem-based Management approach. *Proc. Natl. Acad. Sci. USA.*106: 14722-14727.

Levontin, P. S. Kulmala, P. Haapasaari and S. Kuikka 2011. ' Integration of biological, economic, and sociological knowledge by Bayesian belief networks: the interdisciplinary evaluation of potential management plans for Baltic salmon', *ICES Journal of Marine Science*, 68: 632–638.

Luisetti, T. and K. Turner 2011a. KnowSeas D 4.2: How water quality affects European coastal and marine recreation: a benefit transfer exercise to value health effects and eutrophication at regional sea level, UEA.

Luisetti, T. and K. Turner 2011b. KnowSeas D 4.1: An Economic Knowledge Base, UEA.

Machado F.S., Mourato S. 2002. "Evaluating the multiple benefits of marine water quality improvements: how important are health risk reductions?", *Journal of Environmental Management*, 65: 239-250

MacKenzie, B.R., Hirichsen, H.H. Plikshs, M. Wieland, K and Zezera, A.S. 2000. Quanitfying environmental heterogeneity: habitat size necessary for successful devlopmend of cod Gadus morhua eggs in the Baltic Sea. *Marine Ecology Progress Series* 193: 143-156.

Matthäus, W. and Franck H 1992. Characteristics of major Baltic inflows-a statistical analysis. *Con.t Shelf. Res.* 12: 1375-1400.

Meier, H. E. M., H. C. Andersson, K. Eilola, B. G. Gustafsson, I. Kuznetsov, B. Müller-Karulis, T. Neumann, and O. P. Savchuk 2011. Hypoxia in future climates: A model ensemble study for the Baltic Sea. *Geophysical Research Letters*, 38, doi:10.1029/2011GL049929

Millennium Ecosystem Assessment (MEA)**.** 2005. Ecosystems and human well being: Wetlands and water synthesis. World Resources Institute, Washington, D.C., USA.

Möllmann, C., Müller-Karulis, B., Kornilovs, G., and St John, M. A. 2008. Effects of climate and overfishing on zooplankton dynamics and ecosystem structure: regime shifts, trophic cascade, and feeback loops in a simple ecosystem. *ICES Journal of Marine Science*, 65: 302–310.

Mort, H.P. Slomp, C.P. Gustafsson, B.G and Andersen, T.J. 2010. Phosophorus recycling and burial in Baltic Sea sediments with contrasting redox conditions. *Geochemica et Cosmochemica Acta* 74: 1350-1362

O'Hagan, A. 2011. Probabilistic uncertainty specification: Overview, elaboration techniques and their application to a mechanistic model of carbon flux. *Environmental Modelling & Software*, doi:10.1016/j.envsoft.2011.03.003

Rasmussen, C. E. and Williams, C. K. I. 2006. Gaussian Processes for Machine Learning. The MIT Press.

Sandström M. 1996. “Recreational benefits from improved water quality: A random utility model of Swedish seaside recreation”, Working Paper No. 121, Stockholm School of Economics, Department of Economics.

Shortle, J.S., Abler, D. (Eds.), 2001. Environmental Policies for Agricultural Pollution Control. CABI, Wallingford. 240 pp.

Söderqvist T., Scharin H. 2000a, “The regional willingness to pay for a reduced eutrophication in the Stockholm archipelago”, Beijer Discussion paper No. 128, Beijer International Institute of Ecological Economics, The Royal Swedish Academy of Sciences

Söderqvist, T. and H. Scharin 2000b. The regional willingness to pay for a reduced eutrophication in the Stockholm archipelago. Beijer Discussion paper No. 128. Stockholm, Beijer International Institute of Ecological Economics, The Royal Swedish Academy of Sciences.

Soutukorva Å. 2001. “The Value of Improved Water Quality – A Random Utility Model of Recreation in the Stockholm Archipelago”, The Beijer International Institute of Ecological Economics, The Royal Swedish Academy of Sciences

Tallis, H., Levin, S.P., Ruckelshaus, M., Lester, S.E., McLeod, K.L. Fluharty, DD.L. and Halpern B.S. 2010. The many faces of ecosystem-based management: Making the process work in real places. *Marine Policy* 34: 340-348

Thanh, N. V. 2011. Bio-economic model of Eastern Baltic cod under the influence of nutrient enrichment. Ecosystem-based fishery management, PhD dissertation, Department of Environmental and Business Economics, University of Southern Denmark.

Tuhkanen, H. Alf Vanags and Natalia Volchkova 2010. BalticSurvey – a study in the Baltic Sea countries of public attitudes and use of the sea. Summary of main results. Report 6382. Swedish EPA.

Turner R.K., Georgiou S., Gren I.-M., Wulff F., Barrett S., Söderqvist T., Bateman I.J., Folke C., Langaas S., Żylicz T., Mäler K.-G., Markowska A. 1999. “Managing nutrient fluxes and pollution in the Baltic: an interdisciplinary simulation study”, *Ecological Economics*, 30: 333–352

Uusitalo, L. S. Kuikka and A. Romakkaniemi 2005. ‘Estimation of Atlantic salmon smolt carrying capacity of rivers using expert knowledge’, *ICES Journal of Marine Science*, 62: 708-722.

Vagstad N., Stålnacke P., Andersen H.-E., Deelstra J., Jansons V., Kyllmar K., Loigu E., Rekolainen S. and Tumas R. 2004. Regional variations in diffuse nitrogen losses from agriculture in the Nordic and Baltic regions. *Hydrol. Earth System Sci*., 8: 651–662.

Vesterinen J., Pouta E., Huhtala A., Neuvonen M. 2010. “Impacts of changes in water quality on recreation behavior and benefits in Finland”, *Journal of Environmental Management*, 91: 984–994

Wulff, F., Savchuk, O. P., Sokolov, A. V., Humborg, C. and Mörth, M. 2007. Assessing the past and the possible future of the Baltic. *Ambio* 36: 243-249.

Österblom, H., A. Gårdmark, L. Bergström, B. Müller-Karulis, C. Folke, M. Lindegren, M. Casini, P. Olsson, R. Diekmann, T. Blenckner, C. Humborg and C. Möllmann 2010. Making the ecosystem approach operational – Can regime shifts in ecological- and governance systems facilitate the transition? *Marine Policy*, 34: 1290-1299.

Österblom, H., Hansson, S., Larsson, U., Hjerne, O., Wulff, F., Elmgren, R. and Folke, C. 2007. Human-induced trophic cascades and ecological regime shifts in the Baltic Sea. *Ecosystems*, 10: 877–889.