Comparing least cost solutions from three Baltic wide models; lessons learnt for end-users

Authors: Janne Helin, Berit Hasler

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Name of the scientific representative of the project's coordinator, title and organisation:	Senior researcher Berit Hasler, Aarhus University.
Tel:	+45 8715 8637
Email:	bh@envs.au.dk
Project website address:	http://go4baltic.au.dk/

1. Preface

Three Baltic wide models (Gren et al 2013, Hasler et al 2012 and Ahlvik et al 2014) are developed for minimisation of the costs of achieving nutrient load reduction targets set for the Baltic sea. In this report we compare the three models, and use a choice of model assumptions from these three models to set up a detailed catchment model for Swedish catchment in the western part of Sweden. The main part of this deliverable constitutes the description of the detailed catchment modelling, which builds on the three Baltic wide models but which allows for more detailed modelling of both economic and natural scientific catchment data. The model results and discussion provide results that recommend end-users to use detailed data and modelling approaches, whenever possible, but also that the choice of model and spatial scale of the modelling depends on the purpose of the study.

The work presented in section 4 describing the catchment model in Southwestern Sweden has been described in a working paper, presented at the World conference for environmental economics, Gothenburg, Sweden in June 2018. Helin, Janne: "Developing improved methods for identifying the cost-efficient abatement set in the water protection of the Baltic Sea region". The full paper is included in this deliverable for illustration of how more detailed modelling in a catchment can be used to explore the most cost-effective abatement allocation and choice of measures.

Hereby the deliverable contribute together with deliverable 1.4. "Report on effects of socioeconomic scenarios on nutrient loading, GHG emissions and soil organic carbon" to the assessment of BSAP least cost solutions. As mentioned data from the existing cost- minimization models for the entire Baltic Sea basin are used to build the Swedish catchment cost-minimisation model.

The final deliverable has been compiled and edited by Berit Hasler. Section 4 is solely written and edited by Janne Helin, and section 2 is written by Janne Helin and edited by Berit Hasler. Section 3 is writteh by Berit Hasler. Janne Helin was appointed as post doc at AU, Denmark for 2 years (2015-2017) and thereafter he has been appointed as senior scientist at LUKE, Finland. The work was done while he was appointed at AU.

2. The three Baltic wide cost minimization models

Author: Janne Helin, edited by Berit Hasler

This report compares three Baltic wide economic models of aquatic nutrient flows that have been recently published (Gren, Savchuk & Jansson 2013, Ahlvik et al. 2014, Hasler et al 2014).

We describe the model structure and discuss the implications on results and the Baltic protection policies. The report is organised as follows: it first describes basic model structures, and then the results and their policy implications.

Comparison of model structures

All of the models are economic models integrating environmental processes in different ways. The following sections compare the different aspects of the model setup.

Time frame

The choice of how to model time is one of the fundamental issues in model building. Static models do not include equations connecting processes over time, while dynamic models capture explicitly the changes between modelled periods. The Finnish model developed by Ahlvik et al 2014 and the model developed by Gren et al (2013) are dynamic, hence allowing the description of changes in the nutrient stocks over period of time, while Hasler et al. (2014) use a static model setup for the BALTCOST model.

Modelled timeframe can influence the cost results since the responses in nutrient cycles are not immediate and a longer time span can reduce the costs by allowing natural or low cost processes to reach the abatement targets slowly. In addition, with a future target for sea water quality, a dynamic model implies that some of the effect of early abatement is lost, because of in-sea adjustment processes. Furthermore, discounting future costs will reduce their present value. Together, this implies that in a dynamic model with a target fixed at a future date, actions to reduce nutrient loadings are delayed, and undertaken closer to the target date.

Both Gren, Savchuk & Jansson (2013) and Ahlvik et al (2014) use one year as the model period, meaning that the nutrient (and economic) processes within the year have been abstracted from. This implies that all three models are unable to account in detail for abatement measures affected by the timing of the measure. For example the abatement effect of the commonly utilised limitations for dispersal of manure during winter on some calendar months depends on the propensity of the nutrients to runoff or leach from agriculture determined by variable weather conditions. The abatement costs for this example can also vary over the year because costs can depend on the quantity to be dispersed at a given point in time (due to limitations of dispersal capacity), opportunity costs of labor (related to the amount of other tasks to be carried out at the farm) and to the potential effects on the crop harvest (due to the impact on sowing time and, hence, length of the growth period). Other models with smaller geographical scale have been constructed to show that the intra-annual temporal distribution can provide cost efficient abatement options. For example modelling the effects of splitting total nitrogen fertilisation to smaller dozes during the growth period requires temporal dynamics that the Baltic wide models are not tailored to cope with (Helin, 2013).

The inter-annual dynamics are important in the Baltic Sea models due to the state of the sea responding slowly to the abatement measures. Gren, Savchuk & Jansson (2013) set the target year for 2082 to allow flows between the modelled basins to influence the optimal results based on estimated 40 years of minimum "response time" and added 20 years for (cost-saving) flexibility calculated from 2021, which is the proposed deadline for the implementation of the BSAP by HELCOM. Ahlvik et al (2014) interpret 2021 as the state target year and give 40 years for adjusting the nutrient stocks and the loads, which also depend on slow responding stocks in arable land. The model of Hasler et al. (2014) is static and estimates the costs for reaching 2021 updated Country Allocated Reduction Targets instead of the stock targets used in the dynamic models. The reference period for BSAP targets is 1997–2003, and since BALTCOST uses data from year 2005 as baseline there is some inconsistency with the reference period. For the dynamic models, the lack of explicit BSAP target year for the nutrient stocks increases the difficulties to compare cost estimates, since shorter time ranges require more expensive abatement options.

Load functions

Riverine nutrient loads consist of emissions from different sources. By nutrient load we here refer to nitrogen and phosphorus at the point where they reach the stream, may it be the end of the wastewater discharge pipe or a ditch at the edge of the field or at the rivermouth. Particularly the diffuse load from catchment areas is subject to uncertainty, and the processes that transport nutrients from land to rivers can be described in different ways that imply different distributions between the sources, such as agriculture and forestry. The transport process can also divided into different flow paths like the surface and through the soil matrix.

Gren, Savchuk & Jansson (2013) split the catchment area to 24 regions, Ahlvik et al. 2014 into 23 areas and Hasler et al. (2014) into 22 regions. For modelling of the nutrient load, this spatial structure implies that the smaller river systems have been aggregated to the larger ones. In Hasler et al (2014) the regions are also further disaggregated to 117 subcatchments, as well as further downscaled to a 10x10 km grid across the whole Baltic Sea drainage basin. In all of the models, the catchment areas (in BALTCOST the sub-catchments) are homogenous model units based on average propensity of nutrients to leach and runoff. Therefore by construction, the models will have a tendency to overestimate the costs of relatively small abatement efforts that could be reached by targeting abatement to for example more vulnerable soils or steep slopes. On the choice of more cost-effective abatement measures, compared to the average, see for example Helin & Tattari, 2012.

As a rough way of avoiding biggest discrepancies with respect to nutrient transports within each catchment, Gren, Savchuk & Jansson (2013) divide the model to upstream and downstream regions with individual composition of load sources. The nutrient load is based on Gren, Jonzon and Lindqvist (2008), which in turns refers to Elofsson (1997). The national fertiliser statistics are regionalised to the catchment area specific fertilisation quantities which determine the average load for each of the modelled catchment areas.

Gren, Savchuk & Jansson (2013) rely on Gren, Jonzon and Lindqvist (2008) for the nutrient loads from point sources. According to Gren, Jonzon and Lindqvist (2008) the "*Discharges of N and P from households are estimated based on data on annual emission per capita in different regions, and on connections of populations to sewage treatment plants with different cleaning capacities* "

For describing the diffuse nutrient load sources, the models are further apart. Ahlvik et al. 2014 uses linear load function for nitrogen based on Gren et al. (1997). Leaching of phosphorus is a nonlinear function of soil phosphorus stock and total fertilization (Helin et al., 2006). Load from cropland in all countries is parameterised using barley as a reference crop. The approach in Ahlvik et al. (2014) allows for soil phosphorus stock adjustments over time, which is not considered in the other studies.

The nitrogen loads in BALTCOST is estimated based on application of the DAISY model. The assessments are described in Andersen et al. (2016), which is a report from the RECOCA project.

Retention

The term retention is normally used to describe the natural processes which prevent the nutrients from reaching the sea. These include the processes that remove the nutrient from water such as volatilisation and sedimentation, and the processes that retain the nutrients in the water and just delay them from reaching the sea such as more temporary deposition and eventual resuspension. This is also how retention is defined in all the three Baltic wide cost-minimisation model. In other types of assessments, like in Wulff et al (2014), retention is defined as the difference between net anthropogenic inputs and observed riverine nutrient exports to the sea, i.e. adding up both the retention in ecosystems, abatement, and removal by crop harvests. In economic modelling, abatement carries a cost, and is treated separately from the natural processes for estimating the cost-efficient solutions.

In the BALTCOST model developed by Ahlvik et al (2014), the MESAW-modeled retentions are used, documented in Stålnacke et al. (2011). These retentions are modelled for soil, groundwater and surface water on a detailed 10x10 km level,. Gren et al (2013) use retention parameters for which references are available in, e.g., Gren (2008), Elofsson (2010) and Schou et al. (2006) for the 23 drainage basins, not separating between soil, ground- and surface water.

In all three studied models the amount that is retained does not re-enter the model, and thus the modelled retentions should reflect the long term share of nutrients that never reaches the sea. In Gren, Jonzon, and Lindqvist (2008), which contains the more detailed description of measures used in Gren, Savchuk & Jansson (2013), the term retention is also used to describe the effectiveness of wetlands, which is an abatement measure, e.g. construction of artificial wetlands.

Gren, Savchuk & Jansson (2013) divide the nutrient sources into two classes, one without any retention (located at the coast) and the other with a drainage basin specific retention defined by hydrogeochemical and climate conditions.

In Ahlvik et al. 2014 the retention is modelled in two stages. First, a share of both N and P is retained in the ground water; and second, of the remaining load, a part is retained during the transport in the surface water.

BALTCOST calculates nutrient-specific, abatement measure-specific, drainage basin-specific retentions in ground water and surface waters derived from relevant combinations of modelled surface and ground water nutrient retentions at the 117 subcatchment resolution (Hasler et al 2014).

BALTCOST has the highest spatial resolution for the retention processes, and the authors also have made sensitivity analyses of the assumptions related to the retention.

Marine Transfers

Currents and other bio-physical processes transfer nutrients spatially in the sea. Thus, reaching given target nutrient concentration and its costs depend on measures taken at the onset of this spatial transfer process, which can be described in several ways.

Ahvik et al. (2014) include dynamic marine transfer equations in the model, based on water flow, salinity, and basin specific nutrient stocks, while Gren, Savchuk & Jansson (2013) use a static transfer matrix of nutrients based on Gren and Wulff (2004) and Savchuk (2005) and nutrient stocks not influencing the transfers. Hasler et al. 2014 does not contain marine transfers, but takes the BSAP distribution of load reductions as the target for each (therefore separated) basin. Their argument for doing so is that the transfers are already accounted for in the distribution of load reduction targets between sea basins, and including them in the model would lead to double counting. But one could argue that this type of static approach oversimplify the modelling of the biochemical processes and therefore might bias the results (Ahlvik et al. 2014).

Technical implementation

The model of Ahlvik et al. (2014) is written in Mat lab. The models described in Gren, Savchuk & Jansson (2013) and Hasler et al (2014) are written in GAMS using the solver CONOPT. Cost functions in Hasler et al. (2014) are in Excel worksheets, and cost functions are described in supplementary material to Hasler et al. (2014).

Modelled abatement measures
When estimating abatement costs, the choice of which abatement measures are included in the model can have a large impact on the estimated abatement costs. If low cost methods are not covered, the model results will overestimate the costs given that the omitted measure has a non-neglible effect on the total load. Despite several similarities, the recent Baltic wide models differ considerably in the choice of which abatement measures have been included. These differences can be traced to the used data sources, albeit for some differences there is no clear theoretical or empirical justification.

The model developed by Gren et al. (2013) includes 13 different abatement measures for nitrogen reduction and 11 abatement measures for phosphorous reduction in each drainage basin and time period. Most measures are agricultural, but also measures to reduce emissions from wastewater treatment, industry and transports are included.

The BALTCOST model (Hasler et al 2014) only includes S measures, directed to agriculture and waste water treatment. The reason for not including more measures was missing data and knowledge of the effects of other measures at a Baltic wide level, as the nitrogen and phosphorus load reduction levels were estimated by agronomists and catchment modellers. Changes in tillage, changes in animal feeding, manure dispersal technologies as well as lime or potassium treatments or manure utilisation improvements are considered important, but Baltic wide effects could not be measured and applied.

Distribution of optimal abatement measures
From economic theory, we know that least-costs solution is required for the social optima. When costs are heterogeneous, the optimal abatement quantities differ between the abating industries or as in this case countries, which are used to derive the costs of abatement. The significance of this heterogeneity for the Baltic Sea nutrient abatement problem has been empirically shown in several past studies. From Gren et al. (1997) the conclusion that proportionally more abatement should take place in the ex-communistic countries. Conclusions in the more recent Baltic wide abatement models are the same despite more advanced process nutrient process descriptions (Ahlvik et al. 2014) or some more detailed spatial data sources (Hasler et al 2014).

Differences in results between Hasler et al (2014) and Ahlvik et al (2014) can be explained by Ahlvik et al. having included phosphorus abatement effects for all measures, whereas Hasler et al. (2014) did not because phosphorus effects are variable between locations and catchments. Hereby abatement costs of achiving both N and P targets are overestimated in Hasler et al (2014) but treated as uncertain in Ahlvik et al.

Concluding remarks

(Author Berit Hasler)

The cost-minimisation models have data and functionalities in common, but do also differ in a number of ways illustrated in the assessment presented above. Other comparisons are presented in Hyytiainen et al (2014), Elofsson 2010 and by BalticStern (undated).

3. Comparing results from Balticwide cost-minimisation modelling

Author: Berit Hasler

A number of studies have addressed the problem of cost-effective nutrient reduction to the Baltic Sea. Apart from the three model studies referred to in chapter 2 (Ahlvik et al 2014, Hasler et al 2014 and Gren et al 2013) a large number of previous studies exist: Gren et al 1997; Turner et al. 1999, Gren 2001, Gren and Folmer 2003, Schou et al. 2006, NEFCO 2007, Gren 2008, Gren et al. 2008, Elofsson 2010. Among these Gren 2008 and Elofsson 2010 adress the BSAP targets specifically. While the revised BSAP targets in 2013 (HELCOM, 2013) are used in Hasler et al 2014 and Hyytiäinen & Ahlvik 2015, all other studies have been applied using the initial HELCOM BSAP targets from 2007. When comparing the results of the models the BSAP 2007 targets are therefore used.

In these studies the nutrient loads from 1997-2003 is used as a base year, and a comparison of estimated total costs of achieving HELCOM BSAP from 2007 is presented in Table 1 below. As it appears from the table the choice of the initial load compilation-years plays a role for the cost estimation, as it influences the load reduction target and the distribution between countries and sea-basins.

Table 1. Comparison of estimated total costs between studies

	Costs, million EUR/yr				
	MTT cost model (Ahlvik et al 2013)	Hasler et al 2012 (BALTCOST)	Hasler et al 2014 (BALTCOST)	Gren 2008	Elofsson 2010
Aggregate load reduction target for nitrogen t/yr	102624	135000	102624	135000	135000
Aggregate load reduction target for phosphorus t/yr	10555	15250	10555	15250	15250
Initial loads	2004-2008 (HELCOM PLC-5)	1997-2003 (HELCOM 2007)	2004-2008 (HELCOM PLC-5)	1997-2003 (HELCOM 2007)	1997-2003 (HELCOM 2007)
Denmark	629	472	84	96	451
Estonia	78	32	74	132	25
Finland	23	17	16	79	7
Germany	480	371	113	42	39
Latvia	85	227	231	172	96
Lithuania	101	406	141	377	161
Poland	544	2386	373	3313	2204
Russia	105	507	277	205	962
Sweden	290	272	84	84	-585
All countries	2336	4689	1430	4251	4533

(Source: The table is adapted from Hyytiainen et al 2014)

Sensitivity analysis on the effect of the baseline crop distribution for the effect of measures

Sensitivity analyses can be used to explore the importance of assumptions about parameters and functional relationships, and such sensitivity analysis was performed in Hyytianen et al. (2014) to explore the effects of such uncertainty in assumptions.

One such assumption is that the Ahlvik et al. model assumes that the response to reductions to nitrogen applications to crops can be modelled by a using barley response function, while crop specific response curves are applied in BALTCOST, assuming a mix of crops as measured in the baseline dataset for the model. The sensitivity analysis is performed by applying both the crop specific response and the spring barley type of response in BALTCOST. When the assumption is spring barley only, and not representing the mix of crops, the costs of fulfilling the BSAP targets as anticipated in the BalticSTERN assessments are estimated to 1798 million EUR/yr. The cost is lower when anticipating a realistic land use with a crop distribution as observed in 2005 : 1431 million EUR/yr. The difference can be explained by the fact that the effect on nutrient load reductions from spring barley is lower compared to other crops, because of lower fertilisation level amongst other reasons. This sensitivity analysis indicate that the assumption of spring barley as the representative crop might overestimate the costs, and that the total costs – when looking at this assumption all else equal – of achieving the BSAP targets therefore might be lower than estimated by the model developed by Ahlvik et al (2014).

Sensitivity analysis on the retention-parameters

In the BALTCOST model developed by Ahlvik et al (2014) the MESAW-modeled retentions are used, documented in Stålnacke et al. (2011). These retentions are modeled for soil, groundwater and surface water on a detailed 10x10 km level, while former retention parameters used in e.g. Gren (2008), Elofsson (2010) and Schou et al. (2006) used a single retention figure for each of the 22 drainage basins, not distributed on soil, ground- and surface water. As a sensitivity analysis of the effects of different retentions we have therefore explored the effects of using the MESAW retentions (Stålnacke et al, 2011) compared to the "Schou retentions", both retention sets used in BALTCOST, all else being held constant. The MESAW retentions are distributed on groundwater and surface water retention, while the Schou retentions are only measured for surface water. More details about the assumptions and modelling of the retentions can be found in Hasler et al. (2012) and in Schou et al. (2006).

As a starting point the effects on the load reduction capacity in the sea regions is compared using the MESAW retentions and the "Schou retentions" in BALTCOST.

Table 2. Comparison of maximum reduction capacity with the modeled measures in BALTCOST with two sets of different retention assumptions.

	Max reduction capacity with MESAW retentions		Max reduction capacity with the "Schou retentions"	
	Nitrogen, load reduction (tons / year)	Phosphorus, load reduction (tons / year)	Nitrogen, load reduction (tons / year)	Phosphorus, load reduction (tons / year)
BB	32037	295	28644	360
BS	21804	332	21649	399
BP	188933	9294	309836	17108
GF	47121	2289	52993	3012
GR	40130	828	55887	1371
DS	13139	406	28089	874
KT	26756	679	43924	1128
Total	369920	14123	541021	24252

(Source: Adapted from Hyytiainen et al, 2014)

As the comparison in Table 2 shows, the maximum reduction capacity is higher for phosphorus when using the Schou retentions in all sea basins. The same is true for nitrogen in all the catchments except for Bothnian Bay and Bothnian Sea, where BSAP does no assign any nutrient reductions.

The higher maximum reduction capacity is one important explanation for why the costs modeled by Gren (2008) and Elofsson (2010) are lower than the BALTCOST solutions for the same load reduction targets (BSAP 2007 in HELCOM 2007). The total costs of achieving the Alternative I targets are also compared using the Schou and MESAW retention in the BALTCOST model. The costs of achieving the BSAP 2007 targets are much lower when using the Schou retentions compared to the MESAW retentions. The fulfillment of the full BSAP targets that are achievable when using the Schou retentions are 910 million EUR, which should be compared to the costs of achieving the maximum load reductions that BALTCOST can deliver with MESAW retentions: 4680 million EUR. When comparing the results from the Baltic wide models the differences in assumptions are therefore essential to consider.

The different results presented in this chapter are subject to many kind of uncertainties. Broadly, these uncertainties can be divided into natural uncertainty (uncertainty on the effectiveness of abatement measures), technological uncertainty (uncertain abatement capacity of some measures) and economic uncertainty (uncertainty regarding the cost of abatement measures).

Many important caveats still remain, however. The models consider a significant number of abatement measures, but there are many measures that are excluded from this analysis.

The reasons for not including more measures than the five measures in BALTCOST are that we don't have reliable data for all relevant measures for all the countries in the Baltic Sea catchment (e.g. storage and management of manure). These are measures that can be locally very effective but perhaps not useable on a large scale so that the local specific capacity should be known (e.g. buffer strips).

More detailed catchment models can more easily be applied to capture more measures, because data might be more accessible. In the remainder of this deliverable, Chapter 4 presents a catchment specific model, developed for the catchment in the south-west region of Sweden.

4. Developing improved methods for identifying the cost-efficient abatement set in the water protection of the Baltic sea

Author: Janne Helin

This chapter comprises a model development and analysis in a Swedish coastal catchment. The paper was presented at the World Conference of Environmental Economics (WCERE) conference in Gothenburg 22-29th June 2018, session Agriculture and Water Management, Wednesday 27th. The full paper is available from the homepage http://fleximeets.com/wcere2018.

Developing improved methods for identifying the cost-efficient abatement set in the water protection of the Baltic Sea region

Janne Helin

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Abstract

Economic nutrient abatement models analysing the Baltic Sea protection policies commonly operate on a large scale, grouping river systems to large catchment areas. However, operating at the level of large catchment areas has consequences for policy recommendations. In particular, averaging the in-stream capacity of river systems to retain nitrogen from reaching the sea, removes the opportunity of targeting measures to the most vulnerable regions within the catchment, while overestimating the capacity of abatement measures in the upstream areas. In this study we build a model to show what kind of bias in the optimal abatement set is caused by the assumption of spatial homogeneity. We classify catchment area to zones with increasing distance from the coast and solve the model with and without the zones. We find that while assuming homogeneity prevents from using abatement measures where they would be the most effective (typically close to coast), it also leads to ignoring spatial limitations that are more relevant to a subset of abatement measures, such as the wetlands and buffer zones. Therefore, the bias for setting economic instruments optimally is not only derived from overestimating the costs due to underestimated efficiencies, but also from overestimating the abatement measure capacities relative to the average efficiency. We illustrate this outcome with numerical Swedish data on reaching the good ecological status for the South-West coastal waters.

1 Introduction

Baltic wide models of nutrient abatement commonly operate on a large scale, grouping river systems to roughly 20 catchment areas (Gren, Savchuk, and Jansson, 2013; Ahlvik et al., 2014; Hasler et al., 2014). Since measured environmental data is available on the scale of individual rivers, aggregating it, ignores some existing data that could influence decisions on where and how the nutrient load should be reduced. For example, it can be shown that small catchment areas retain typically less nutrient on average than large catchments, and therefore the optimal abatement, all other things equal, should be higher in small than large catchments. While modelling nutrient abatement at the catchment level might seem like a well-founded choice for management purposes, describing catchment properties with averages could lead to wrong conclusions and policy recommendations.

Averaging heterogeneous load parameters to a single catchment value, cuts the use of within catchment variation from the abatement set, reducing the efficiency that the model describes compared to what would be possible in reality. Therefore, the opportunity of targeting measures to the most vulnerable regions within the catchment is lost. Besides the heterogeneous nutrient load sources, the impact of retention, which reduces the quantity of nutrients reaching the sea as the distance to the estuary increases, is averaged. Thus, a model with uniform retention is not capable of recommending more abatement downstream than upstream even though this might allow considerable cost savings for equal nutrient abatement.

In the large, aggregated catchment areas of the Baltic Sea abatement models, the average retention includes by definition upstream areas far from the sea with a high retention rate. Therefore, the overall effectiveness of all abatement measures decreases compared to a spatially disaggregated catchment area. This can lead to subsidising unnecessary abatement measures. In some large scale applications, like the country-level abatement model in Sweden developed by Hart and Brady (2002), retention is excluded from the abatement problem entirely, which leads to overestimating the effectiveness of measures and potentially not reaching the optimal abatement levels because of inadequate policy response, for example not covering all the required measures.

Several more detailed abatement models exist for various smaller areas around the Baltic Sea (Helin and Tattari, 2012; Helin, 2014; Konrad et al., 2014). The problem in up-scaling many of these models lays in specific data sources used to set up the models that are not available for the entire Baltic Sea catchment. This could lead to building models based on excessive amount of assumptions instead of representing the catchment characteristics realistically.

One solution, which trades some accuracy for an easier generalization of parameters, is to break down the catchment areas to several sub-catchment areas and create a model structure with intra catchment heterogeneity derived from sub-catchment level data. The sub-catchment area is a spatial unit formed similarly to the main catchment area except on a finer scale splitting the river network between areas drained by various tributaries. Sub-catchment areas can be formed for any watershed in the Baltic Sea region with freely available digital elevation models. The idea of breaking down the watershed to several sub-catchments and using it in optimisation has been demonstrated in US for example by Rabotyagov et al. (2010). While this work demonstrates the significance of spatial resolution for the least-cost abatement, it does not contain some of the abatement measures that are considered to play a part in the least-cost solution in the Baltic models, such as wetland and vegetated buffer zones.

Gren, Savchuk, and Jansson (2013) divide each catchment of Baltic Sea into two; upstream and downstream. This allows to differentiate between what is assumed to be no retention at downstream, and uniform retention at upstream. Hasler et al. (2014) compare the effect of different spatial structure on retention, but the cost-minimisation for the catchment areas uses a catchment wide average retention rate.

We use Swedish data as a case study on how the use of sub-catchment data will change the optimal abatement measure set for reaching the good ecological status (GES) as defined in the Water Framework Directive (WFD). In Sweden predefined sub-catchment areas are associated with the elementary nutrient load data as well as the information on estimated retention rates. We include main abatement options considered as cost-effective in the past abatement literature and show that many of the previous estimates are subject to the modelled geographical scale and other assumptions that can be questioned by use of higher resolution data.

2 Material and methods

The impact of the model scale on the cost-efficient allocation of nutrient reduction efforts is investigated by setting up a numerical optimisation model with two different specifications. The more detailed version, referred as the heterogeneous specification divides catchment areas to subsections, while in the less detailed, homogeneous specification, the catchment area contains no further spatial disaggregation of input parameters or decision variables. The latter case represents modelling results that the existing Baltic Sea wide nutrient abatement models lead to, although the numerical results for homogeneous model specification are still overestimating the accuracy of Baltic wide models in which smaller catchment areas would be pooled to a bigger one.

Both of these model specifications aim to reach GES with the least-costs by using the same set of abatement measures and data sources. To keep the method section concise and to avoid burdening the paper with excessive notation, we present the model structure formally only for the heterogeneous model specification. The heterogeneous specification reduces to the homogeneous specification when the subdivision of catchments is removed except for the wetlands which have been modeled as in BALTCOST in the homogeneous model version since the more complex dynamic approach in the heterogeneous specification cannot be applied without some spatial interactions within the catchment areas.

2.1 Model

We take the target nitrogen abatement level required to meet GES of coastal water as a given external parameter. This target can be derived with ecological models that connect nitrogen concentrations in the sea with the loads from land. The abatement targets are defined for delimited areas of the sea which are located close to the coast. We denote the total abatement required for each of these littoral stretches as $\bar{A}_l$, i.e. this is the quantity of abatement required in total for the stretch to reach the GES. We allow the littoral stretches to receive nitrogen loads from several catchment areas and several littoral stretches to be connected to a single catchment area. The share of nutrient load for the stretch l, from catchment area i, is denoted with $\psi_{i,l}$. Thus,

$$\bar{A}_l = \sum_i \{\psi_{i,l} A_i\} \forall \; l \tag{1}$$

The total abatement over each catchment area A_i is a sum of abatement a over different measures m within different distances d from the coast.

$$A_i = \sum_{d,m} \{a_{i,d,m}\} \tag{2}$$

The optimal abatement level for the catchment A_i^* is given by solving the cost-minimisation problem in equations 1-16:

$$\min C \| \bar{A}_l = \sum_{i,d,m} \{c_{i,d,m}(a_{i,d,m})\} \tag{3}$$

Abatement measures have costs, $c_{i,d,m}$, which depend on the level of abatement. We look for an optimal solution for the abatement levels that minimises the total costs C of reaching the fixed target level set by the good ecological status target for each littoral stretch. We will assume m to consist of some common measures which belong to the cost-efficient set in the past literature (Brady, 2002; Hart and Brady, 2002; Helin and Tattari, 2012; Hasler et al., 2014; Ahlvik et al., 2014). The cost functions are split to two categories $m = \{1, 2\}$ corresponding to centralised waste water treatment (WWT) and agriculture.

Th objective function in equation 3 is subject to constraints in equations 4 - 16, which describe the limitations on the capacities of abatement measures and define the cost functions for the abatement categories.

In the figure 1 we illustrate the basic spatial setting of the model, in which an estuary has been divided to littoral stretches, which correspond with boundaries defined in the national implementation of the WFD.

The nitrogen load reaching the coast depends on hydrology, which has to be simplified to reduce the model dimensions and facilitate finding of numerical solutions to the least-cost problem. The distance zones (d) are based on aggregating sub-watersheds formed by (pre-existing) flow direction maps. Therefore, the quantity of zones and their size vary between the watersheds. The streams flow from inland zones in the watershed through the downstream zones and within each zone nitrogen carried from the zone itself with the flow and the nitrogen received from upstream are subject to natural processes which retain nitrogen from reaching downstream such as denitrification.

The total annual load $e_{i,d}$ from each distance zone consisting of various sources such as the agriculture and waste water treatment plants (WTTPs) is an average value that can be estimated with nitrogen load monitoring data and hydrodynamic models.

$$e_{i,d}(a_{i,d,m}, x, b, z, n, \theta_{i,d}, \zeta_{i,d}) = (1-\rho_{i,d})(e_{i,d}(x, b, z, n, \theta_{i,d}) + \zeta_{i,d} - \sum_m (a_{i,d,m})) + e_{i,d-1} \tag{4}$$

where $\rho_{i,d}$ is the average retention for each distance zone. Distance zones are ordered from t to 1, from the most upstream zone (t_i) to the catchment outlet zone at $d = 1$. The total annual nitrogen load from the catchment area is given by the load of the zone closest to the sea $e_{i,1}$. To shorten the equation, the variables determining the load from agriculture, x, b, z, n, are presented here without their indices. They refer to field area, retention increasing measures, number of animals and fertilisation. The nitrogen load from waste water treatment in absence of abatement is given by a parameter $\theta_{i,d}$, which refers to person equivalent of biological oxygen demand (BOD). Sources of nitrogen load that are not covered by abatement methods, except of wetlands, are summed up to load $\zeta_{i,d}$.

Establishing new wetlands can be used to increase the hydrological residence time of the fresh water in the river systems. This allows for more denitrification

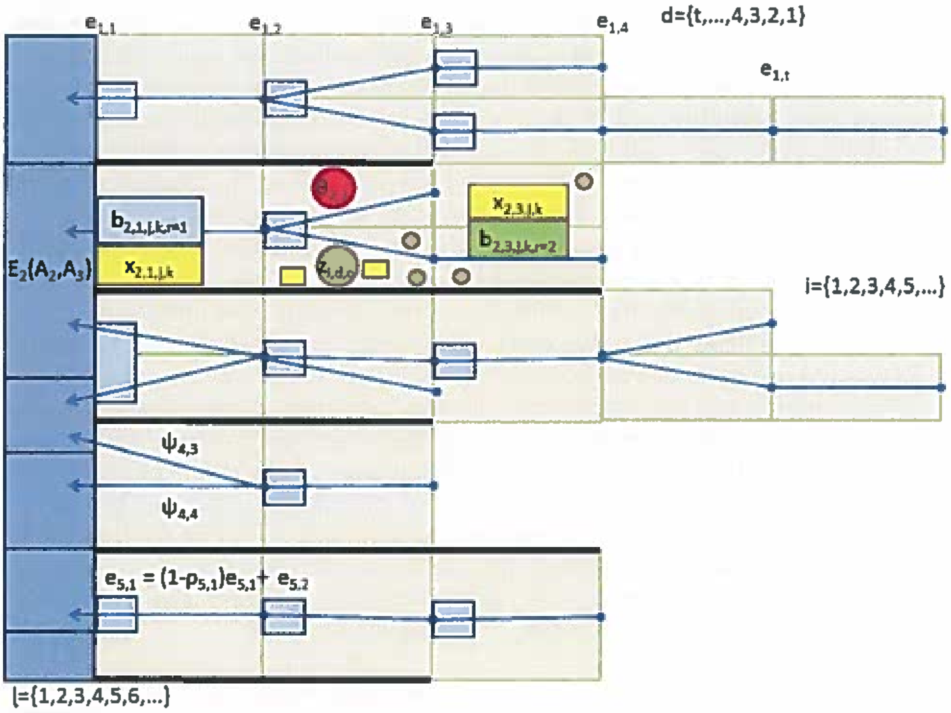

Figure 1: A diagram illustrating the type of spatial relations of catchment areas (i) to littoral stretches (l). For any given littoral stretch, the land based nitrogen load $e_{i,d}$ can come from one or more of the watersheds, which themselves consist of zones (d) with increasing distance from the coast. A single watershed can be connected to one or more littoral stretch. The share of nutrients received by the littoral stretch $\psi_{i,l}$ is determined by the share of watershed land area associated with each stretch. Nitrogen loads and their reduction costs are watershed specific and depend on farming area $x_{i,d,j,k}$, number of farm animals $z_{i,d,o}$, waste water treatment efficiency and the quantity to be treated $\theta_{i,d}$, and measures of increasing the retention of nutrients from the farm area with buffer zones $b_{i,d,j,k,1}$ and from the whole upstream catchment area with constructed wetlands $b_{i,d,j,k,2}$. Natural retention $\rho_{i,d}$ is accounted for each distance zone as an area weighted mean of the sub-watershed's retention in the distance zone.

to occur, as well as some other less important nitrogen removal/delay processes such as deposition. Therefore, wetlands reduce the quantity of nitrogen reaching the sea from all of the nitrogen sources that flow through it (Hammer, 1992). When the water flow to a wetland is high, the residence time is shorter. This reduces the effectiveness of downstream wetlands that capture all the stream flow. Besides moving the wetland location upstream and sacrificing the abatement effect for some sources left downstream, the effectiveness can be increased by making the wetlands relatively larger which allows the water flow to slow down more effectively. However, increasing the wetland size can be constrained by the area suitable for flooding it, for example relocating urban infrastructure would most likely turn out to be excessively costly compared to other available locations upstream or other means of abatement. Byström (1998) develop a model in which the the wetland efficiency is dependent on the wetland area and the incoming nitrogen load. Here we use more simple approach to incorporate the wetland problem to a more general abatement optimisation problem including various other measures described in the equation. Thus, wetland efficiency (per ha) is given only by its size relative to the catchment area upstream from it, but the incoming load is endogenous and subject to other abatement measures. Compared to a more general abatement problem in Gren, Savchuk, and Jansson (2013) we have endogenous wetland efficiency and more accurate spatial representation of the catchment area, which enables to compare solutions with spatial disaggregation to distance zones to the homogeneous model of the catchment area. In contrast to the constant efficiency postulated Byström, Andersson, and Gren (2000), we model the nitrogen removal efficiency of a wetland as an increasing function of the wetland size relative to its catchment area upstream.

Separating the abatement effect of wetland from the other abatement measures in agriculture allows to write its abatement effect as:

$$a_{i,d,3} = 100\omega \sum_{j,k} b_{i,d,j,k,1} x_{i,d,j,k} \upsilon [c_{i,d,m\neq 3} + \zeta_{i,d} - a_{i,d,m\neq 3}] \tag{5}$$

where wetlands affect the load from upstream sources including agriculture and WWT, but also the other sources $\zeta_{i,d}$ such as forests and lakes. The efficiency (in kg N removed per ha of wetland) depends on the location of the wetland. The further upstream the wetland is located, the less load passes through it, but the share that the wetland area covers from the upstream area increases, which slows down the water flow more. The efficiency relative to the size is given by ω. The upstream area, $\upsilon_{i,d}$, with respect to wetland location, is calculated for each zone assuming that within each zone the wetland is located at the most downstream point as illustrated in the figure 1. The total available land area for wetland construction is assumed to be given by a constant share of the field area that is distributed to the distance zones. This assumption is based on the idea that the most suitable wetland areas are typically fields that were created by draining wetlands in the past (Horvath et al., 2017). Furthermore, estimating the opportunity costs for other land use types besides agriculture would require additional empirical work. For the homogeneous model specification we use the upper limits based on the BALTCOST model, which assumes that wetlands can be only established on organic soil. Thus, in the homogeneous version of the equation 6 the way of defining the upper limit $\bar{a}_{i,3}$ is taken from the literature to better reflect assumptions used in the existing Baltic Sea abatement models.

$$a_{i,d,3} \leq \bar{a}_{i,d,3} \tag{6}$$

Using distance zones improves precision on how the abatement measures interact with each other. For example if wetland capacity downstream is limited by urban settlements, using upstream areas for wetlands reduces the abatement potential of wetlands for urban waste water. Furthermore, zoning differentiates the abatement efficiency of measures within the load sources as retention increases with the distance. Costs of waste water treatment for each catchment ($C_{i,1}$) are given by,

$$C_{i,1}(a^{\%}_{i,d,f,w,u,1}) = \sum_{d,f,w,u} (c_{i,d,f,1}\xi - \alpha_f a^{\%}_{i,d,f,w,u,1} + \beta_f (a^{\%}_{i,d,f,w,u,1})^2) \forall\ i \tag{7}$$

where f is the size class of the WWT facility measured by the person equivalents (PE) of biological oxygen demand (BOD), w is the water treatment technology and u is the number of specific type and size of plants. Size and technology classes and the distribution of the data is given in Appendix x. The cost function parameters (α,β,ξ) are from Hautakangas et al. (2013). Current abatement level $\bar{a}^{\%}_{i,d,f,w,u,1}$ is based on typical abatement percent for Swedish WWT facilities for different treatment processes (SEPA, 2010). The additional total costs from increased abatement are given by the difference of current and additional abatement cost $C_{i,1}(a^{\%}_{i,d,f,w,u,1}) - \bar{C}_{1,i}(\bar{a}^{\%}_{i,d,f,w,u,1})$. We assume that added cost of infrastructural investments in relocating plants are so high that the abatement problem reduces to choosing the abatement levels in existing plants. Furthermore, for each size class the maximum abatement percent is constant, set at 97 % based on the maximum values achieved in combining the different treatment technologies. This maximum capacity can be exceeded only by connecting previously unconnected households to the existing WWT facilities. The share of unconnected households has been estimated for each of the catchment by the connection percent given by the WWT facilities within the catchments.

$$\sum_{f,w,u} (a_{i,f,w,u,1}) \leq \theta_{i,d} \ \forall\ i, d \tag{8}$$

The current nitrogen loads are converted from the BOD capacities of the WWTPs (year 2011) by a multiplier of 4.2 (kg N per BOD PE following the BALTCOST model reported in Hasler et al. (2014).

Second, for nitrogen from diffuse sources, we base the analysis on the representative farm approach (Griffin and Bromley, 1982), which is defined by the profit-maximisation problem of a representative farm in agriculture, which is then upscaled to the catchment level. The choice of crop j to cultivate on the field area $x_{i,d,j,k}$ with technology k determines the profits from the fields jointly with the nitrogen fertilisation $n_{i,d,j,k}$ per area unit (ha). Further profit and nitrogen load is originating from farm animal production zi, d, o. Besides the changes in crop types and farming technologies as well as reduction in fertiliser quantities and animal numbers, the nitrogen load from agriculture can be reduced by allocating part of field edges with a grass strip $b_{i,d,j,k,r=2}$ that is not fertilised and creates a buffer between the cultivated areas and water bodies.

Similarly, a share of fields can be also used for establishing constructed wetlands $b_{i,d,j,k,r=1}$ to increase retention.

$$C_{i,2} = \pi_i^*(x_{i,d,j,k}^*, n_{i,d,j,k}^*, z_{i,d,o}^*, b_{i,d,j,k,r}^*) - \bar{\pi}_i^*(x_{i,d,j,k}^*, n_{i,d,j,k}^*, z_{i,d,o}^*, b_{i,d,j,k,r}^*) \forall\ i \tag{9}$$

The solution to the unconstrained (by nitrogen load) profit π^* is used to determine the baseline nitrogen load from agriculture. The optimal profit $\bar{\pi}_i{}^*$ in equation 9 is solved with a constraint on the allowed nitrogen load in the equation 1. All the variables are non-negative. The profits are increasing in animal numbers, field area and fertilisation. Total field area is fixed. The sum of retention measure shares $\sum_r b_{i,d,j,k,r}$ cannot exceed 0.98. In addition, both types of retention measures have their own capacity limits based on the catchment characteristics. For fertilisation, we assume quadratic yield response leading to a decreasing marginal revenue at high fertilisation rates. Animal numbers and field area have upper bounds set by the existing capacities. The nitrogen load from agriculture is increasing in field area, fertilisation and animal numbers, but decreasing in retention measures.

$$e_{i,d,2}(n(z), x, b) = h^1(h^2[1 - b_{i,d,j,k,2}^{0.2}] + h^3)exp^{0.71[1-b_{i,d,j,k,2}n_{i,d,j,k}/(\bar{n}_{i,d,j,k}-1)]} \forall\ i, d \tag{10}$$

where $\bar{n}_{i,d,j,k}$ is the reference level for nitrogen fertilisation for which the load from fertilisation is relative to and h^1, is a load factor incorporating the effect of rotation and tillage in the modelled load. Since only part of the nitrogen flows from the fields on surface to the streams and hence only this part can be captured by the buffer zone, the share of nitrogen in the surface flow of the total nitrogen load from the field is given by the h^2 and the remaining share by h^3. For a more detailed description of the basic profit and abatement function for the representative Swedish farm without livestock, we refer to the original sources (Brady, 2001; Brady, 2003) and equations 14, while the description and analytical solution of optimal buffer zones is presented in Helin, Laukkanen, and Koikkalainen (2006), and not repeated here. To incorporate the effects of livestock on the nitrogen load, we model animal density of the catchment by number of animals z of type o and fix nitrogen content of manure from each type of animal to be η_o. Therefore, the nitrogen available in manure for crops in each catchment in given by,

$$\sum_{j,k}(n_{i,d,j,k}^{manure}) = \sum_{o} \eta_o z_{i,d,o} \forall\ \ i, d \tag{11}$$

Each distance zone within the catchments has a maximum capacity of animal housing $\bar{z}_{i,d,o}$ based on the current capacity which the animal numbers cannot exceed,

$$z_{i,d,o} \leq \bar{z}_{i,d,o} \tag{12}$$

Thus, transport of manure is limited to the fields in the distance zone of the farm. We simplify the analysis by assuming that nitrogen in manure is a perfect substitute to mineral fertiliser use, but we take account that a constant share of manure nitrogen is volatilised. The optimal fertiliser use quantities of both sources depend on their prices (for manure the price of transport), prices of crops and yield functions (in Appendix X). As the equation 11 holds manure equal to number of animals multiplied by η_o, the only way of reducing manure nitrogen input is to reduce the number of animals. However, manure can be allocated between different crops that have different load propensities.

For grass we consider endogenous demand unlike for other crop types for which constant prices p_j summarize the demand. Transporting grass is expensive and the demand for it depends largely on the quantity of cattle in the region of production. Since cattle quantity is endogenous in the model, the demand for grass could be reduced by the optimal abatement path, decreasing the value of grass production to zero. The grass demand is modeled in a simplified way as a function of total cattle numbers in each catchment.

$$\sum_{k,grass} y_{i,grass}(n_{i,d,j,k}, x_{i,d,j,k}) \leq \sum_{d,cattle} \sigma_{cattle} z_{i,d,cattle} \forall \ i \tag{13}$$

where *grass* is the subset of field use classes of grass production and *cattle* is the subset of animal types consisting of bovine species. The annual consumption of grass for cattle of each type is given by σ_{cattle}. We assume that maximum of 50% of the total grass consumption can be from pasture due to a limited growing season. For calibrating the modelled land use at the profit maximising level with the observed data we use a set of maximum constraints on specific crop types. We assume that no field area will be converted to other than agricultural uses since this would also reduce the single farm payment subsidies the farms are entitled to and therefore be expensive from the farmer's perspective. The modelled animal types have some interdependencies that are described in equations 15-16.

Profits from the animal production are calculated per head of animal as difference between animal type specific revenue and variable costs.

$$\begin{aligned} \pi_i = \sum_{d,j,k} s_j^{crop} - c_{j,k}^{crop} + \\ p_j \tau_k [(y_j^{\iota} + y_j^{\kappa}[n_{i,d,j,k} + n_{i,d,j,k}^{manure}] + y_j^{\lambda}[n_{i,d,j,k} + n_{i,d,j,k}^{manure}]^2)] \\ -(p_N n_{i,d,j,k} + c^{transport} D_i n_{i,d,j,k}^{manure}) \\ ] x_{i,d,j,k}(1 - b_{i,d,j,k,r}) + \sum_{j,d} (x_{i,j,k} b_{i,d,j,k,r}[s^{retention} - c_r^{retention}]) \\ - \sum_j p^{wage} l_j^{crop} \\ + \sum_{o,d} (z_{i,d,o} * p_o^{animal} - p^{wage} l_o^{animal}) \end{aligned} \tag{14}$$

The crops have different optimal nitrogen fertilisation intensities given by the yield parameters $y_j^{\iota}, y_j^{\kappa}, y_j^{\lambda}$ and prices of the inputs, in particular fertilisation, p_N and crop prices p_j. The effects of tillage type and timing on yield are modelled by τ_k following the specification in Brady (2001). Manure application costs are given per kilogramme of N dispersed $c^{transport}$ for average distance D_i. Cost per ha of crop farming c_j^{crop} include the other variable costs such as machinery, fuel and pesticides. Field area covered with crops or with buffer

Figure 2: Research area covers the South-West coast of Sweden. Cost-minimisation model described in the model section covers the catchment area of Göta river as well as the adjoining catchments draining both to Kattegat and Skagerrak.

zones or wetlands is considered to be eligible for the single farm payment of the EU common agricultural policy (CAP), respectively s_j^{crop} and $s^{retention}$. The cost parameter $c_r^{retention}$ covers the direct costs in establishing wetlands and vegetated buffer zones, while the opportunity costs depend on what type farming would be replaced determined by the $x_{i,d,j,k}b_{i,d,j,k,r}$, which is the area of farm land converted to retention measures.

Cows produce a single calf per year which determines the also the quantity young cattle that is not lactating. To produce piglets fixed quantity of sows and boars are needed based on the respective quantities in the statistics for years in data.

$$z_{calf} = \mu_{3,i}(z_{dairy} + z_{sucler}) \tag{15}$$

$$z_{piglet} = z_{grover} = \mu_1 Z_{sow}, = \mu_2 Z_{boar} \tag{16}$$

These simple and straightforward assumptions keep the ratios of different farm types fixed to avoid illogical reductions in animal types that depend on each other.

2.2 Data

To demonstrate the impact of improved spatial resolution on the estimation of abatement costs and optimal combination of measures, we use Swedish catchment data including the seventh biggest catchment area connected to the Baltic Sea. The Göta river, which is the biggest river in Scandinavia, flows to the Kattegat which is the entrance to the Baltic Sea. The research area is illustrated in figure 2

The included littoral stretches are receiving water flow and nitrogen from land area divided to 12 main catchment areas for the pollution reporting purposes including national authorities, EU and international cooperation bodies such as HELCOM. These main catchment areas are divided to 5900 sub-catchments, which are aggregated up to 108 distance classes as explained in the model description. We excluded 43 stretches (1.2 % of the total load) from the analysis since they have very small share of load originating from land based sources (areas connected only to small islands and at the edges of the research area). The required quantities of total abatement for each stretch (the table A.1) were obtained from Swedish regional authorities (personal communication Jan Petersson, Länstyrelse). They are based on Swedish Water Information System (WISS), which derives the data from the S-HYPE (Swedish Hydrological Predictions for the Environment) model (Lindström et al., 2010; SMHI, 2013; Brunell, Gustavsson, and Alavi, 2016). The boundaries for the catchments and the littoral stretches are open data from Geodataportal (Geodataportal, 2016).

The catchment/distance zone specific nitrogen retention $\rho_{i,d}$ is calculated as an area weighted average of sub-catchment specific retention estimates for each catchment based on the S-HYPE model results stored in the SMHI (Swedish Meteorological and Hydrological Institute)electronic archive (open data). The resulting catchment average retention rates for WWTP are in the table A.3. and for agriculture in the table A.4. The share of the catchment i land area connected to stretch l given by $\psi_{i,l}$ and distance zone retention $\rho_{i,d}$ are included in the electronic supplementary material. Besides retention, S-HYPE results have also been used as an input data source for nutrient load data for non-agriculture diffuse sources SMHI (2013) that have not been replicated here.

WWT volume and technology is from statistics Sweden SEPA (2010). There are 94 WWT facilities covered by the data source within the research area. For the share of population not connected to municipal WTT we use statistics Sweden data tables and Brånvall and Svanström (2011).

Land use data is collected at the municipal level and was available between 1998 and 2015. Field boundaries are from SBA (2015b) accessed through Geodataportal. Crop categories in agriculture were available until 2015 (SBA, 2016) at a municipal level. The allocation of the crop areas from the municipalities to the catchment areas was done by weighing the municipal crop area with the field share of municipality for each catchment. Since some crop types are not specified and others not covered by the yield and load parameter data, this remaining agriculture area was divided between the modelled crop types based on their share of the field areas in each of the catchments.

The table A.10 (in Appendix) contains the key to the crop type names.

The catchment field distribution derived from the municipal data is shown in the table A.5.

For the catchments 238 and 239 consisting of islands, the municipal boundaries that are used to allocate crop data to field polygons are not covering most of the area. The baseline crop allocation within these catchments represents the small share of land covered by the crop data.

Animal categories and numbers are from municipal data of 1990-2013 (SBA, 2015a), which has been distributed between catchments based on the location of animals in Geodataportal (2016) and number of animal holdings. We used a simple uniform distribution since municipal data on sizes of the farms was not available. The animal number summarised to animal units at the catchment area level are presented in the table A.4. For variable costs we use labour costs per animal, which have been estimated by using Eurostat (2015) wage and required labour per animal included in the table A.6.

SBA (2014) contains the data of the farm subsidies in Sweden that is used for calculating the baseline nutrient load levels. The excreted nitrogen is subject to volatilisation losses. We use constant loss factor of 0.32 to adjust the manure from animals ($"\eta_o"$) to crop available nitrogen (IPCC??).

We adopt an approach used by Brady (2001) and Brady (2003) for modeling the production function and diffuse nitrogen loads from Swedish agriculture. Therefore, the abatement options include reduced fertiliser use, catch crops, grass leys, fallow and tillage timing. We assume that efficiency of the abatement methods is same between east and west coast of southern-Sweden, since the parameters of Brady (2003) have been only published for the east coast(Brady, 2001). Yield parameters are summarised in the table A.7.

Symbols a, b, c are the crop yield response parameters to nitrogen fertilisation and the impact of tillage timing $\tau^1_{i,k}$and rotation on yield $\tau^2_{i,k}$. Parameter h^1 refer to the impact of tillage timing and rotation with respect their impact on the nutrient load. These are based on Brady (2001). Maize and rye have no parameter data and were not included in the model. Spring wheat is assumed to have yield parameters of spring barley. Both spring and winter rapeseed are modelled as winter rapeseed area since only winter rapeseed has yield parameters in Brady (2001). As Brady (2001) does not consider explicitly animal production, we modified grass ley to form two types of grass land i.e. silage and pasture. The municipal crop data contains multiple crop categories referring to pasture and grass production. Due to seemingly overlapping definitions these categories have been summed up and have been split equally between pasture and silage. Since we lack data to produce separate yield functions to pasture and silage, we use yield parameters of grass ley in (Brady, 2001) for both of these types. We assume that pasture receives only manure as a nitrogen input and that it requires less labour input than silage (parameters in the table A.6).

We extend the use of grass lays to specific grass buffer zones established at edges of surface water such as streams and lakes based on SLU (2013), which is an electronic database with estimated maximum areas suitable for buffer zones at the catchment level in Sweden. We assume that in addition to the maximum area defined in Fyris (summarised in the table A.4 at the catchment level), one percent of the total field area can be used for vegetated strips at the field edges. It is assumed that the share of buffer zone cannot be higher than 49 % from the area of any crop type. For the nutrient capture efficiency of the buffer zones we use the approach and parameters in Helin, Laukkanen, and Koikkalainen (2006). For fallow, pasture and silage we assume that the buffer zones do not constitute a further reduction in nitrogen load. Thus, we have fixed the buffer zone area to zero for these land uses. For other crop types the model has been set with a lower bound of 0.01 which reflects requirements within CAP to leave a one to three meter wide buffer strip at the field edge.

Lacking more detailed spatial data on potential areas suitable for constructing wetlands, we assume that maximum of 49 % of agricultural area can be used for wetland construction in each distance zone. Similar to the buffer zones, we assume that this limit has to hold for each crop type. To put this upper boundary in perspective, we analyse the distribution of fields in relation to stream data from Open Street Map (OSM, 2017) and Swedish lake data (SMHI, 2012). Within the research area, first 100 meters from surface water includes 7.5% of the total agricultural area, while 0.5 km and 1 km distances correspond to 26.8% and 46.3% of the agricultural area. Therefore, using the upper limit of 49 % reflects rather the maximum share that still retains the majority of the allocated as agriculture, than a maximum based on the spatial considerations of wetland locations.

For the homogeneous model specification we use the considerably lower upper limits based on organic soil area, which in Hasler et al. (2014) is taken as the capacity limit for restored wetlands. In the heterogeneous specification the efficiency of wetland to remove nitrogen is based on a simple regression between the wetland size (expressed as % of upstream area) and its nitrogen retention efficiency Puustinen et al. (2007), while for the homogeneous specification we use 300 kg N per wetland ha reported in Hasler et al. (2014). For the costs of constructing wetlands, we assume that the projects to establish them are similar to conditions (and prices) in Southern Finland and use averages of calculations included for specific projects reported in Majoinen (2005), which consist mainly of earthmoving, which we have annualised per hectare and divided over twenty year period. Crop prices in the table A.8 are from SBA (2017).

Nitrogen price is from Swedish recommendations for fertilisation and liming (Albertsson et al., 2015). We have updated variable costs (yield dependent) and fixed costs per ha in Brady (2001) with price indices (SBA, 2017). For pasture we assume, half of the costs of grass leys and for oats the fixed costs of barley since these were not covered in Brady (2001). Yield depended costs for pasture are taken as zero since yield on pasture is assumed to be directly consumed by cattle. For fallow, which in Brady (2001) is given negative costs, we use the relative difference between barley and green fallow in Helin, Laukkanen, and Koikkalainen (2006) to derive the cost from barley costs. The spring wheat costs are similarly derived from barley costs. For fixed costs of buffer zone, we use the costs of grass ley, since we assume annual cutting of buffer zone vegetation. The manure application costs have been derived using typical nitrogen concentrations (table A.6 to get the required volume to be transported and dispersed which has been divided to by transport volume of 10 m^3 to get the required quantity of deliveries with assumed speed of 28.08 km per hour for distance of 6 km for all non grass crops and 3 for grasses. The costs include wage and fuel (7 litres) as well as lubricant (0.09 litres) consumption per hour. We used fuel price of 0.68 eur per litre and lubricant price of 1.23 eur per litre.

Since all the modelled land uses are considered eligible for the CAP single farm payment, this subsidy is not affecting optimal allocation of abatement and left out from the revenue. National production subsidies for silage, pasture and potatoes are from Swedish Board of Agriculture (SBA) and included in the table A.8. Buffer zones are also regarded to receive the subsidy paid for grass land and not to distort the optimal abatement choice wetlands are considered to receive the same level of subsidy.

The numerical version of the optimisation model is programmed in GAMS and was solved by using the CONOPT3 non linear solver (McCarl et al., 2016; Drud, 2016). Spatial analyses in order to produce the input parameters from various spatial and non spatial data sources was programmed in python using several packages such as arcpy, numpy and pandas. Crop type data was processed with R. Crop production in distance zones with less than 10 ha has been fixed to zero to facilitate solving the numerical problem with CONOPT3.

3 Results

3.1 Baseline results

The first set of results illustrates the model outcomes without the GES abatement targets in order to evaluate the model performance relative to data and other nutrient load estimation methods.

3.1.1 Land allocation

The modelled crops cover majority of the field use within the research area as can be seen from the table A.5 .

The table A.9 shows the profit maximising crop allocation of the heterogeneous model specification relative to the table A.5.

The constraints are set to allow limited freedom in allocating farm land between different crop types and conservation farming practises such as the catch crops and delayed tillage. Thus, the modelled land allocation does not reproduce an identical field area allocation between crops. Sugar beet or energy crop production are not profitable with the modeled parameters. The required amount of grass fodder to meet the cattle consumption (defined in equation 13) is produced using less field area than estimated from the municipal data. All the other crop types in production are at their maximum allowed levels defined by the constraints or close to it. The remaining area is allocated to fallow, which leads to overestimated shares of fallow for all the catchments except the largest, Göta river.

The area assumed as fallow to facilitate numerical solutions is highest in the catchment 215 where it totals 0.22 % of the catchment field area, while for all the others it represents less than a promille of the area.

3.1.2 Nitrogen load to coast

The majority of the modelled load is stemming from the Göta älv, which equals to approximately 79.86% of the total nitrogen reaching the sea within the research area. Correspondingly, the littoral stretch with the highest nitrogen input is Nordre river fjord, which receives majority of the Göta river's nitrogen load.

For the whole research area, the existing modelled abatement measures in WWTPs (1598 tonnes) and retention (9673 tonnes) reduce the total nitrogen load reaching the sea to 17264 tonnes. Reaching the good ecological status in the littoral stretches requires further reduction in the load reaching the sea by 6433 tonnes.

Model results for nitrogen load can be compared with the other estimation methods to demonstrate the model performance. This simple validation shows that despite the spatial aggregation of sub-catchments, the model produces comparable nitrogen load estimates to the official Swedish figures based on S-HYPE model.

All the columns in in the table 1 show the estimated nitrogen load for the catchment outlet. The first column (SHYPE) is a sum of individual S-HYPE sub-catchment nitrogen loads (obtained from the Vattenvebb data portal). The

Table 1: Nitrogen load to coast in tonnes per year (given different models)

	SHYPE	SHYPE2	SHYPE3	DynBL
208	274.35	270.71	274.69	223.80
209	358.71	316.62	357.63	299.14
211	187.49	184.03	187.62	150.82
212	146.66	142.05	149.14	146.93
213	15795.19	14732.89	15238.96	13787.63
214	900.59	885.20	899.96	1094.24
215	136.19	129.81	136.36	139.45
216	153.29	153.76	153.64	123.36
217	675.74	664.45	667.71	538.83
218	699.66	680.72	699.54	555.63
238	129.49	129.58	129.55	141.24
239	47.16	48.01	48.02	62.95

second column is using S-HYPE data of total load sources and multiplying them with average retention for each load category. Therefore, this column reflects the error caused by using the average catchment retention instead of sub-catchment specific retention. In the third column (SHYPE3), the S-HYPE retention is averaged over the distance zone instead of the whole catchment. In the DynBL column SHYPE is not used for estimating the source load from agriculture or waste water treatment, instead they are calculated as described in the equations and data above. From these results it can be seen that using the distance zone specific retention reduces the estimation error (deviation from S-HYPE) compared to using the average retention for the whole catchment. Non-synchronised input data and different calculation methods for agriculture and WWTP lead to larger deviations from the S-HYPE results than averaging the retention in either way. There seems to be no systematic bias to either direction compared to the S-HYPE results since five of the thirteen catchments are overestimating the load, while the rest are underestimating.

3.2 Least-cost abatement to reach GES

3.2.1 Summary for the whole research area

Differences in model specifications (homogeneous catchment with an average retention and heterogeneous catchment split to distance zones) lead to significant divergence in the the estimated costs of reaching the GES target. The homogeneous model specification which ignores spatial aspects (both the opportunities and challenges) produces the GES reduction with costs of 12.2 €/kg N (115.8 SKK/kg N) on average, while the heterogeneous allocation estimates 44.6 €/kg N (423.1 SKK/kg N)

Results in the table 2 compare the optimal model allocation to reach GES with the model baseline solution (BL, non-constrained by the equation 1) and between the homogeneous (subscript HO) and heterogeneous versions (subscript HE).

In the table abatement % refers to the quantity of nitrogen load reaching

Table 2: Summary of key results for the whole research area

	BL_{HO}	BL_{HE}	GES_{HO}	GES_{HE}
Retention %	35.68	35.91	35.32	36.32
Abatement %	0.00	0.00	36.31	37.26
Wetland effect	0.00	0.00	3909.21	2198.05
Wetland %	0.00	0.00	2.07	8.19
Buffer %	0.62	0.62	0.16	0.36
Buffer max %	5.65	5.65	1.45	3.26
Fallow %	5.12	5.41	46.73	53.07
% AU	100.00	99.97	100.00	81.40
% N input	100.00	100.00	91.24	65.37
Grass %	38.56	38.56	82.46	68.87
Delayed Tillage %	0.01	0.00	0.00	0.69
Catch Crop %	0.00	0.00	0.00	12.34
WWTP Count	0.00	0.00	1.00	32.00
WWT %	73.79	73.79	74.16	92.73

Retention % refers to natural retention without the constructed wetland. Abatement % includes all abatement measures including wetlands. The wetland effect is in metric tonnes of N. Buffer % is calculated from the combined area of CAP field edge vegetation and dedicated buffer zone are for non grass crop types relative to the total farm land. Similarly fallow % is the percent share from total farm land. AU % refers to animal units as % of the maximum capacity, while % N input for non grass crop types is relative to the economic optimal fertilisation of nitrogen. Grass % is a sum of grass crop types including the buffer zones and fallow relative to the total farm land. Delayed tillage and catch crop % are the respective shares of total farm land. WWTP count stands for the quantity waste water treatment plants in which the abatement is increased from the baseline, while WWTP % is the average abatement from waste water treatment plants.

the coast (reduced either in agriculture, waste water treatment or constructed wetlands) relative to the baseline load. The target reductions in both model specifications are the same (given in table A.1), but the baseline loads to coast of the homogeneous solution exceed the heterogeneous baseline load by 56.7 tonnes. The baselines are not identical because the distance zone specific constraints in the equation 12 for maximum animal capacity, the quantity of manure to be dispersed (the equation 11), the demand of grass and pasture and the upper limits for each crop type are simplified to catchment level in the homogeneous model specification. In addition to the slight deviation from the maximum animal capacity, the distance zone specific constraints force a slightly higher total share of fallow in the heterogeneous specification.

The total effect of wetlands is nearly twice higher in the homogeneous specification than in the heterogeneous model specification. Since a nearly four times higher share of area is allocated to wetlands in the heterogeneous model specification, it is clear that the homogeneous specification leads to a higher estimate of wetland efficiency than the heterogeneous one. In the heterogeneous catchment specification, the capacity within the distance zone sets limits to the potential size of wetlands, so that their efficiency cannot be very high in the downstream area, while in the upstream areas their effectiveness is limited by the smaller loads entering the zone and by the higher natural retention. Despite these limitations on wetland effectiveness, constructed wetlands still remain as a significant part of the optimal abatement set because other measures lack the capacity to reach the nitrogen reduction targets.

The limited effectiveness of wetlands (combined with the other effects due to heterogeneity) leads to a higher abatement in waste water treatment sector and in agriculture. The area required for fallow (% of agricultural area) increases in both of the optimal solutions to reaching GES. The buffer strips established for 1 % of the cereal and vegetable production cover a lower share of total agricultural area as the share of these crops from the total land allocation decreases. The optimal strategy for utilising the land removed from the cereal production depends on the model specification. Given the homogeneous specification, a larger share of land is allocated to producing grass than in the heterogeneous specification. Animal units are reduced in the heterogeneous specification more than in the homogeneous specification, for which the animal unit reduction is minor. Thus, the larger share of grass land in the homogeneous specification compensates for the quantity of area removed from production as wetlands and fallow, while compromising less the quantity of grass produced. In the heterogeneous specification, animal units, and consequently the grass share, are reduced more to avoid even larger economic losses in cereal and vegetable production. This is due to the wetlands being less effective in the heterogeneous model specification. The nitrogen fertilisation intensity of cereals and vegetables (as % of profit maximising level) as well as the overall nitrogen input, including grasses, is significantly lower in the heterogeneous than in the homogeneous specification.

Catch crops are utilised extensively, but not dominantly as an abatement measure given the heterogeneous specification, while delaying the tillage is optimal for only a very limited area. The reversal of these abatement measures in the homogeneous specification between the base line and the GES solution is likely due to a relaxed model tolerance for the optimal solution associated with a large difference in the variable sizes between the Göta catchment area and the other much smaller catchment areas.

When the load removed by constructed wetlands is not considered, the total natural retention actually increases from the baseline level in the heterogeneous specification. This results from the modelled capacity to target abatement measures to low retention distance zones and to retain higher nitrogen intensity crops in higher retention zones. Because of the upper limits set to constrain the maximum crop area based on the observed land use, the model offers limited possibility to relocate crops from lower retention zones to higher retention zones. However, the retention share in table 2 is calculated from the field edge to the sea and does not exclude the quantity removed by the constructed wetlands. Therefore, accounting for the wetland effect in the total retention estimate would reduce the natural retention depending on where the wetlands are located.

3.2.2 Catchment level

The spatial resolution of data, the modelling choices and the optimal allocation that follows from them, are manifested as differences in where the abatement ideally takes place. A figure 3 summarises the optimal solution for the least-cost nitrogen abatement required for reaching GES for each catchment given the different model specifications.

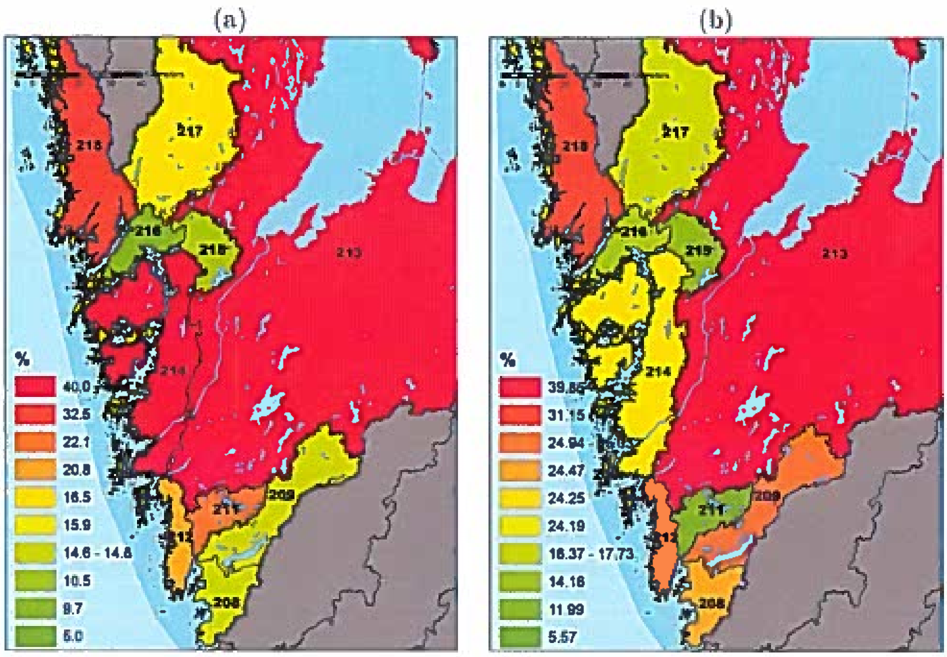

Figure 3: A comparison of the optimal abatement share summarised to the catchment level between the different model scales. Abatement % includes all nitrogen abatement including the effect of wetlands. Panel a) refers to the heterogeneous model specification, while panel b) depicts the homogeneous result.

The largest differences in the total abatement between the model specifications occur in the catchments connected to Byfjorden (582000-115270). In the heterogeneous model specification, the optimal allocation does not contain any wetland in the catchments (214 and 215) draining to Byfjorden, while in the homogeneous specification wetland in the catchment area 214 covers 1.98 % of agriculture area, which is the maximum allowed in the equation 6 and 0.54 % of are the in 215. The incoming nitrogen load in 214 from upstream areas is low, because the catchment area consists of short streams. Therefore in the heterogeneous model specification the removed quantities of nitrogen even by relative large wetlands would be modest, while the constant nitrogen removal effectiveness in the homogeneous specification does not depend on the quantity of load flowing to the wetland. In the homogeneous specification the few upstream areas in 214 with the high retention rate (average retention of 23.29 % in the fourth distance zone) increase the average retention and compared to the heterogeneous specification, the other abatement measures besides wetlands are less effective since in the heterogeneous specification the distance zone closest to the coast has an average retention of 5.67 % compared to the homogeneous catchment area average of 9.12 %. These differences in the model specifications lead to shifting the abatement effort in the homogeneous specification to the other catchments that are connected to same stretches, particularly to 216 in which fallow area in the homogeneous specification is nearly double to that of the heterogeneous specification and the overall abatement percent is nearly three times higher.

Looking at the largest catchment area of Göta river (213) in figure 3, the optimal abatement percent is similar between the model specifications. Irrespective of the specification, the abatement percent of the Göta river is the highest. The differences between the model specifications do not lead to differences in the abatement percent between Göta and other catchments, because the load reduction to the stretches can be only reached by significant measures in the largest catchment area, i.e. Göta river. However, the optimal allocation between the measures within the catchment area is very different depending on the model specification.

As to be expected from the total results in table 2 the results for the largest of the catchment area demonstrate diverging nitrogen removal effectiveness for wetlands for the different model specifications. In the heterogeneous specification nearly 25 % of the Göta catchment area is allocated as wetland, while in the homogeneous specification only 2.3% is used, but with much higher nitrogen reduction efficiency than in the heterogeneous specification. The main factor separating the Göta from the other catchment areas is the length of the river system, which leads to high average retention and large variation of retention between the distance zones. In the the heterogeneous model specification, this allows allocating abatement to lower than average retention zones that should reduce abatement costs compared to the homogeneous specification that uses only the (higher) average retention for all measures. However, the overall effectiveness of the abatement relative to the reduction target is so low, that measures are required also in higher than average retention zones, leading to higher overall abatement effort and costs for the Göta river. This is partly explained by the downstream urban areas, which already have their waste water treated with better than average effectiveness.

Despite the high abatement percent of the Göta catchment area, few of the smaller catchments have higher optimal abatement intensity (abatement kg per ha catchment); the highest (3.3 kg/ha) in the catchment area of 214 for the heterogeneous model specification. It is worth noting that the non-uniform GES targets for both of the model specifications and the spatial connectedness of some littoral stretches to a subset of catchment (lack of for some), implies that the optimum intensities for any catchment depend on the other connected catchments. Thus, the results do not show highest abatement intensity for example with the lowest average retention (catchment 238).

While the optimal abatement results summarise the differences between the catchments and model specifications, they do not reveal much about the optimal measures.

Table 3 shows the results for the constructed wetlands at the catchment level.

Table 3: Average catchment wetland share (as percent of total agricultural area)

	GES_{HO}	GES_{HE}
Total	2.07	8.19
208	0.00	1.91
209	2.33	9.97
211	0.00	5.59
212	1.33	8.79
213	2.25	24.67
214	1.98	0.00
215	0.54	0.00
217	0.37	4.76
218	1.20	10.15
238	2.66	7.95
239	2.47	1.21

As discussed above, wetlands have a significant role in the optimum abatement irrespective of the model specification. Besides the wetlands, the conversion of crop area to fallow is one of the more effective sources of abatement relevant for the whole research area irrespective of the model specification. The optimal fallow allocation between catchments is illustrated in the table 4.

Table 4 here

As shown already in the table 2, the total baseline levels of fallow between model specifications differ slightly. At the catchment level these differences are more visible, but within few % units of each other.

The assumptions between the model specifications lead to a diverging optimal share of fallow. The fallow share in the heterogeneous specification is higher since the wetland impact is lower than in the homogeneous specification. Compared to the baseline in the homogeneous specification, the fallow share decreases as it is in many catchments replaced by constructed wetlands. Compared to the baseline in the heterogeneous specification, the GES solution increases the share of fallow as wetlands are relatively less cost-effective in many catchment areas. The largest difference between the specifications is in the fallow allocation in the catchment 208, in which the homogeneous solu-

Table 4: Average catchment fallow share (as percent of total agricultural area)

	BL_{HO}	BL_{HE}	GES_{HO}	GES_{HE}
Total	5.12	5.41	46.73	53.07
208	14.50	16.31	64.12	13.11
209	21.10	23.49	50.33	32.28
211	46.98	45.73	75.58	56.01
212	22.80	24.58	63.05	42.32
213	0.00	0.00	44.13	54.66
214	36.95	39.03	56.80	46.76
215	14.15	16.14	32.61	31.55
216	33.69	35.58	63.64	33.72
217	34.88	36.61	64.75	51.78
218	26.93	28.88	59.89	48.29
238	64.88	65.34	71.22	81.72
239	65.78	65.95	81.66	85.24

tion contains significantly more fallow than the heterogeneous one. As can be seen from the figure 3, the total abatement required for the littoral stretch of Inre Kungsbackafjorden (572472-120302) is distributed differently between the catchment areas depending on the model specification. Moreover, the fallow share in the heterogeneous specification is smaller in all of the connected catchments (208,209,211,212) than in the homogeneous specification. In contrast the WWTP efficiency (211) and the wetland share (208,209,211) are higher and the nitrogen intensity is lower (208,209,211,212) in the heterogeneous specification. However, for the homogeneous specification in 209 the abatement effect of the smaller share of farm land allocated to wetland is larger than the the combined effect all wetlands in 208,209,2011 in the heterogeneous model specification. Furthermore, the effectiveness of wetland in the homogeneous specification is not decreased by conversion of the upstream area to fallow, leading to overestimated joint effectiveness of these abatement measures for the Inre Kungsbackafjorden.

The table 5 summarises the optimal allocation of buffer zones for the catchments.

Table 5 here

As seen from the overall results, the total significance of buffer zones decreases as a larger share of farm area is allocated as wetlands and fallow. The buffer zone capacity is not reaching its estimated limits in any of the catchment areas. Some variation between the catchments occurs due to the model specification.

The difference in optimal fertilisation for cereal, oilseed and vegetable crop at the catchment level is illustrated in the table 6

Table 6 here

In the homogeneous model specification, reduction of nitrogen intensity of the afore mentioned crops takes place in few of the catchment areas, while given the heterogeneous specification the fertilisation intensity is reduced in all catchment areas. With the exception of the catchment 239, the heterogeneous specification leads to larger fertiliser reductions than the homogeneous specifi-

Table 5: Average catchment buffer area (percent of maximum capacity)

	BL_{HO}	BL_{HE}	GES_{HO}	GES_{HE}
Total	5.65	5.65	1.45	3.26
208	24.78	24.78	0.00	50.78
209	2.72	2.72	0.00	4.21
211	2.31	2.31	0.00	3.52
212	42.49	42.49	0.00	40.64
213	5.99	5.99	1.62	3.14
214	3.25	3.25	0.00	3.15
215	1.99	1.99	1.89	2.09
216	3.34	3.34	0.00	6.77
217	1.84	1.84	0.00	1.67
218	4.42	4.42	0.00	3.64
238	26.12	26.12	48.89	14.28
239	30.90	30.90	25.64	15.50

cation. This stems from the large difference in the total quantity of catchment specific abatement effort illustrated in figure 3 and relates to the changes in the allocation of stretch targets between the catchments.

The distribution of animal units between catchments relative to the maximum capacity is illustrated in the table 7.

Table 7 here

The reduction of animal units is a part of the optimal solution only for the heterogeneous model specification, and even then only for the Göta river catchment area, for which the abatement targets are the highest. The share of crop cover measures are low in relation to the total modelled field area and not presented here.

The distribution of WWTP nitrogen removal % for catchments on average is presented in the table A.2.

Table A.2 here

The baseline for both of the model specifications includes different quantities and sizes of WWT facilities depending the catchment and therefore the average nitrogen removal shares vary. The increased overall WWTP abatement consists of individual catchment area specific abatement levels, highest in the watersheds 211,213 and 214, given the heterogeneous model specification. For the homogeneous specification, increasing the WWTP abatement % is optimal only in 214, and only for less than one % unit. Again these differences relate to the assumptions particularly concerning the effectiviness of wetlands and averaging of retention for the catchment areas.

Moving on from the catchment average values, the solution to the heterogeneous model specification can expressed at the model resolution i.e. with the distance zones. The figure 4 illustrates the distribution of the optimal abatement solutions within the catchments when sub-catchment data is used to optimise the abatement levels.

The high required abatement to reach the good ecological quality targets in Nordre river fjord (3415 tonnes) and Rivö fjord (2155 tonnes) leads to extensive

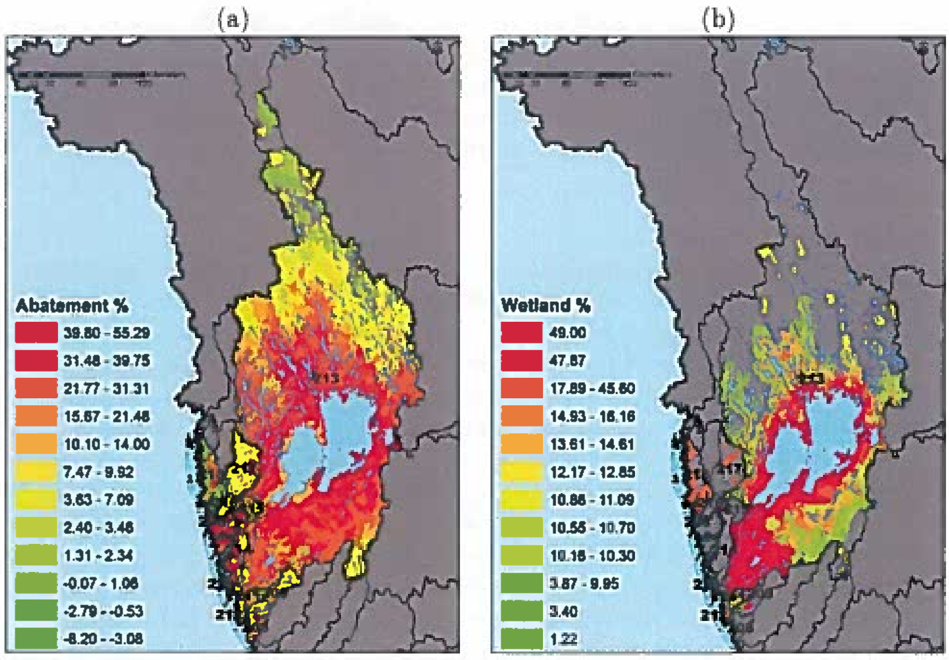

Figure 4: The distribution of source abatement and wetlands to distance zones. In the panel a, red colour implies high abatement % at the source (the effect of all measures except wetlands on the nitrogen load), while the green end of the spectrum implies low abatement. In the panel b, red shades imply higher shares of agriculture area used for the wetland.

Table 6: Average catchment N input (as percent of crop profit maximising level)

	BL_{HO}	BL_{HE}	GES_{HO}	GES_{HE}
Total	100.00	100.00	91.24	65.37
208	100.00	100.00	100.00	54.72
209	100.00	100.00	100.00	45.93
211	100.00	100.00	100.00	48.46
212	100.00	100.00	100.00	57.55
213	100.00	100.00	91.56	70.67
214	100.00	100.00	100.00	61.86
215	100.00	100.00	71.56	47.32
216	100.00	100.00	100.00	65.39
217	100.00	100.00	100.00	41.89
218	100.00	100.00	100.00	53.16
238	100.00	100.00	56.22	37.49
239	100.00	100.00	45.73	63.57

use of the abatement methods in the vicinity of lake Vännern. The wetland area reaches its capacity limits downstream of the lake and extra capacity has to be obtained upstream of the lake, where the natural retention is higher, reducing the efficiency of the abatement measures. Downstream of Göta river is urban area, as the river runs through the city of Gothenburg. Settlements leave limited capacity to implement wetlands. Improvements in the waste water treatment technology are limited since the existing abatement in the city is among the highest in the research area. Furthermore, the Göta river splits to two and establishing wetlands would only influence a part of the total riverine load after the split. These characteristics cannot be taken into account in the homogeneous catchment model calculations. In the upstream distance zones of Göta river the optimal nitrogen loads (at source) increase due to allocating more nitrogen intensive production from downstream distance zones to upstream zones. In this way the increased load from upstream will be reduced by both the natural retention and the constructed wetlands. This decreases the abatement costs compared to an uniform distribution of abatement in a heterogeneous catchment, but the effect on costs is smaller than the effect of using a constant wetland nitrogen removal rate and an average retention.

Some of the smaller rivers reach the GES targets of their coastal water with relatively less drastic nitrogen reductions, while the catchments consisting of islands (238, 239, partly 214) require a relatively high reduction. Due to smaller variations in retention in these catchments, the optimal abatement quantities between distance zones are more uniform. However, both the wetlands and the abatement percent at the source show a higher abatement effort closer to the coast than in the higher retention distance zones more inland. The constructed wetlands play a smaller role in the optimal nitrogen abatement of these catchments because the short streams do not receive high loads from upstream areas and thus there is not much benefit of the wetlands capturing also the upstream load compared to the higher costs in establishing them. The highest abatement per catchment area unit takes place in the catchment 214 in the heterogeneous

Table 7: Average catchment animal units (as percent of maximum capacity)

	BL_{HO}	BL_{HE}	GES_{HO}	GES_{HE}
Total	100.00	99.97	100.00	81.40
208	100.00	100.00	100.00	100.00
209	100.00	100.00	100.00	100.00
211	99.53	99.53	99.53	99.53
212	100.00	100.00	100.00	100.00
213	100.00	99.97	100.00	78.52
214	100.00	100.00	100.00	100.00
215	100.00	100.00	100.00	100.00
216	100.00	100.00	100.00	100.00
217	100.00	100.00	100.00	100.00
218	100.00	100.00	100.00	100.00
238	100.00	100.00	100.00	100.00
239	100.00	100.00	100.00	100.00

model specification. This follows from increasing the waste water treatment in the catchment area 214 to reach the binding abatement levels in two littoral stretches, Rivö fjord (212,213,214,239) and Byfjorden (214,215,216,238) that are connected to multiple other catchment areas. In 214, all the WTTP sites are located within the first distance zone closest to the coast (WWTP retention 1.1 %), and therefore in the homogeneous model specification, the use of average catchment retention parameter (3.8 %), decreases the relative efficiency of WWTP compared to the heterogeneous specification. The single WWTP plant within catchment area 211 is also within the first distance zone (WWTP retention 0.68 %), while the average retention for WWTP of the whole 211 catchment area is 12.1 %.

4 Discussion

4.1 Baseline nutrient load

Our results for the nitrogen load reaching the sea are partially derived from the S-HYPE model and partially based on applying earlier Swedish load models for agriculture and a Baltic wide model approach to WWT nutrient load with Swedish data. The focus on this study is in developing and comparing an economic model frameworks for nutrient abatement, and therefore we do not focus on calibration or validation issues of the nitrogen load that can be raised with any environmental model. However, we acknowledge that providing only very limited results for validation compared to strictly observed nitrogen concentration in river outlets of the research area, increases uncertainty of the abatement costs. Difference in nitrogen loads between the S-HYPE and the baseline in the abatement model follow from differences in the source loads and the use of average retention (over whole catchment or the distance zone) compared to individual sub-catchment retention in S-HYPE. The differences in source loads of agriculture can relate to crop choice, fertiliser input quantities, farming tech-

Table 8: Average catchment waste water treatment N removal (percent of incoming N load)

	BL_{HO}	BL_{HE}	GES_{HO}	GES_{HE}
Total	73.79	73.79	74.16	92.75
208	72.00	72.00	72.00	72.00
209	42.00	42.00	42.00	42.00
211	86.00	86.00	86.00	100.00
212	42.00	42.00	42.00	42.00
213	61.33	61.33	61.33	89.37
214	85.09	85.09	85.89	98.72
215	86.00	86.00	86.00	86.36
217	42.00	42.00	42.00	42.00
218	75.19	75.19	75.19	75.19
238	42.00	42.00	42.00	42.00
239	78.92	78.92	78.92	78.92

nology, animal husbandry or spatial disaggregation.

In the model baseline solution, the area used for grass production is underestimated, which influences the modelled nitrogen loads and the optimal abatement choices. Since the model does not contain data on pasture productivity and we have used parameters for grass ley to describe the yield from pastures, it seems as we have overestimated pasture productivity, and therefore underestimated the area the area required for fodder grass production.

Given that increasing grazing is the true profit maximising choice, a larger share of the load in the baseline would be linked to the cattle production. Depending on the load propensity of pasture compared to other crop types, the abatement costs could be either be over or under estimated. According to Ryden, Ball, and Garwood (1984) grazing could lead to 20 % increase in nitrogen leaching from grass land and thus substituting it with silage production could provide and additional abatement measure. However, the capacity of this measure could depend on the relative large nitrogen intensity as results in Ryden, Ball, and Garwood (1984) were derived with 420 kg N/ha input. Therefore, the capacity could be quite limited with lower nitrogen balances.

Matching the baseline land allocation with the observed crop data is challenging. In this case, replicating the existing land allocation is also influenced by the availability of crop data on the municipal level, which is particularly increasing uncertainty regarding the results for the island catchment areas (238 and 239), which were rather poorly covered by the available crop data. Furthermore, without demand functions or productivity differences of fields, the profit-maximising solutions paint too simple picture of land allocation. In representative farm models like (Brady (2003) and Helin (2013)) without demand or land productivity differences, the model solutions are forced to reflect the existing crop allocations with constraints that set maximum area limits to different crop types based on what is observed in the data. Using constraints will cut of some adaptation possibilities that exist when there are nitrogen load differences between crops. For this study we have not limited the adaptation possibilities very strictly as we allow crop areas to exceed the average crop allocation in the data by 50 %. This means that in the baseline model solution the most profitable crop types (wheat, barley) are over-presented while the least profitable are under-represented. The distribution of different crops is based on the municipality level data that has been split to catchment areas. This does not capture a possibly already existing distribution to protect vulnerable areas of the coast.

For the waste water treatment, the methods of estimating the incoming nitrogen in the untreated sewage and the limited input data on the existing treatment efficiency can reduce the reliability of the results. Given the scope of this study and magnitude of the difference between the modelling methods, an in-depth comparison or inter-calibration is not warranted. In further studies using more detailed data on the WWTP specific nitrogen removal efficiencies could improve the accuracy of optimal abatement solutions.

4.2 Optimal abatement

Wetlands have an important role in the modelled optimal allocation of nitrogen reduction measures. Compared to the two way split to upstream and downstream in Gren, Savchuk, and Jansson (2013), our approach splits the catchment in number of zones depending on the size of the catchment area. Particularly, for larger catchment areas like the Göta river, the size dependent zoning allows more accurate spatial relations between the polluting sources and wetlands, which in Gren, Savchuk, and Jansson (2013) are always located downstream and thus able to reduce all the modelled loads. This assumption is prone to lead overestimating the wetland abatement potential and to underestimate the abatement costs, since large settlements are typical in downstream are reduce the size of wetlands that can be constructed by presumably with smaller costs in agriculture than in urban areas. Furthermore, Gren, Savchuk, and Jansson (2013) does not consider the change in nitrogen removal effectiveness of the wetlands, which implies that smaller wetlands would likely have overestimated effect on the nitrogen load, while larger ones would seem to be constrained by the available area. Byström (1998) estimated the wetland effectiveness with respect to both area and the quantity of incoming load. Our approach differs as the abatement efficiency is increasing in area, but not increasing in the incoming load. Incorporating the reduced effectiveness of wetlands as the incoming nitrogen concentration decreases, would reduce the share of the wetlands from total abatement, lead to higher abatement costs and potentially to inadequate capacity to meet the GES reduction targets. However, in Byström (1998) the overall nitrogen removal efficiency of wetlands is significantly higher as the costs per ha are lower and effectiveness of wetland for smaller areas are higher.

Assuming that wetlands would be constructed only on agricultural area is a strong assumption considering the scaling of the nitrogen removal effectiveness. Our assumption is underestimating the capacity of the total wetland abatement potential. However, we have allowed a much larger area (49 %) of agriculture land to be allocated as wetlands than would be likely feasible in more detailed spatial analysis of potential wetland locations. The capacity limits are reached in several distance zones of the Göta river catchment, and therefore affect the overall average abatement costs and the optimal allocation. In some areas the opportunity costs of wetland creation could be overestimated in the model since we are deriving it from the land value of optimally managed agriculture. On the other hand, the construction costs could be also underestimated since only direct construction expenses have been accounted for and in practise more costs could incur from project planning and management. Compared to some Swedish estimates and the BALTCOST model Hasler et al. (2014), that did not consider any construction costs our wetland costs estimates are naturally higher. The data in Byström (1998) is from 1990-1994 and based on 50 years period, while shorter period of 20 years and more recent cost figures were used in this study. Given the prevalence of wetlands as a significant part of the cost-efficient abatement set in this and other earlier studies in the Nordic countries, a more detailed assessment using up-to-date cost data would seem to be called for. Furthermore, the use of non-agricultural areas for constructed wetlands could be implemented in further studies by GIS analysis and land prices.

We followed similar approach as in the BALTCOST model Hasler et al. (2014) in the reduction of livestock production as a measure to reduce nitrogen. This approach is simplified and might overestimate the costs since according to Helin (2014) reducing stocking density was the most expensive of the abatement methods considered at a dairy farm. For a more extensive model of manure nitrogen abatement opportunities, one should consider dispersal technologies and more detailed spatial patterns of livestock farms within the overall farm field structures Helin (2013). Omitting measures that could potentially be cost-effective leads potentially to overestimating the costs of reaching GES. However, the scope of the abatement achievable with the manure management technologies is smaller than what can be achieved by reducing the quantity of manure, and under the conditions in which relatively high abatement targets are required, (like for many of the modelled stretches), the target could be only attained by reduction of animal numbers. Thus, when facing high targets, omitting some measures like the manure dispersal would not bias the results significantly. Furthermore, including more decision variables to complex models reduces the model tractability and adds data needs and potential sources of modelling error that could also lead to biased estimates of costs of reaching GES.

Concerning the role of animal number reductions in environmental management, beef cattle poses a problem in modelling as well as a additional source of reduction potential. Given our simple approach in modelling animal profits by combining standard output with labour costs for animal care and the (opportunity) costs of growing grass instead of cash crops, the positive profitability of beef production depends on fairly optimistic interpretation on the production costs. Further research efforts would be needed to evaluate the environmental and economic outcomes of the farm land given up by beef production, which could under some circumstances be directed to more nitrogen intensive crops than the pasture typically associated with beef production.

4.3 Abatement costs

Since limited amount of information can be utilised in the optimisation models that estimate costs of water quality targets, economists often assume that the true adaptation costs are overestimated by the model results (for example Ahlvik et al. (2014), OTHERS). Our results show that using coarse spatial resolution can lead to underestimating the costs instead. The problem seems to lie in generalising empirical effectiveness results of abatement measures over the domain, in which we should assume a decreasing marginal effect. This is particularly critical for wetlands, for which the effectiveness depends on the nutrient content in the flow from upstream that is decreased by other abatement measures. While our model ignores some adaptation capacity due to less than ideal spatial resolution (lack of capacity to optimise the abatement for the sub-catchment level instead of the distance zone level), we expect that relaxing the spatial assumptions for wetlands would reduce their effectiveness further (further fragmentation of wetland area). Compared to earlier Swedish estimates, the abatement costs seem to be higher in our work. Hart and Brady (2002) estimate that 30 % reductions in nitrogen load could be achieved with only 3 % reductions in gross profits from farming, while our results show costs a magnitude higher than that. Compared to the optimal abatement set in Hart and Brady (2002), energy crops are fixed to zero in our results because of the negative profitability in the baseline and low share in the input data. As the energy crop profitability depends on energy prices and costs considerations with notable spatial dimension, a wide scale conversion of fallow to energy crops, as suggested by Hart and Brady (2002) seems like a more challenging source of nutrient abatement than, for example wetlands that we considered, outside the scope of Hart and Brady (2002). Ignoring the impact of retention can be shown to underestimate the abatement costs, although a more detailed analysis would be required to confirm how large share of the cost difference could be explained by just the omission of retention. Furthermore, assuming that all nitrogen is derived from agriculture in Hart and Brady (2002) can be expected to decrease the costs compared to our estimates, because of some high expense nutrient sources are treated as stemming from agriculture. In Gren, Savchuk, and Jansson (2013) the marginal abatement costs of nitrogen for Baltic Sea Action Plan (BSAP) do not reach 1 €/ kg N (10 SKK/kg N) in Kattegat area, which is the closest, but not precise geographical match to our research area. According to Elofsson (2010), the BSAP target for the Swedish Kattegat equals 8.6 % of abatement. Since the target is different it is not possible to do one-to-one comparison with Gren, Savchuk, and Jansson (2013), but it is worth noting the their analysis contains a "'self-cleaning"' property for the nitrogen load within the Baltic Sea, which is essentially an important free abatement method with some time delay. Our results, unlike Gren, Savchuk, and Jansson (2013), and to some degree Hart and Brady (2002), do not follow from inter annual dynamics and are based on external load reduction targets for the annual loads, which should reflect the required dynamic time frame for reaching GES. However, it is possible that these targets have been set incorrectly (sub-optimally) from the dynamic perspective and that extending the model developed here to contain the nitrogen stock in the sea, could reduce the costs of optimal abatement for reaching GES.

The marginal costs of reaching 15 % nitrogen reduction estimated in Elofsson (2003) vary between 52 - 145 SKK/kg N depending on the required certainty for reaching the target. The model in Elofsson (2003) does not include wetlands, and the measures combine to a maximum reduction of 25 % which would not therefore be sufficient in reaching the GES targets for all of the littoral stretches in this study. Besides the lack of wetlands, Elofsson (2003) uses more conservative limits for the abatement capacity in agriculture and in WWTPs.

Hasler et al. (2014) do not include costs estimates separate from phosphorus abatement since reaching the modelled BSAP targets for phosphorus leads to sufficient nitrogen abatement. For such a conclusion to hold, given the Swedish GES targets, the phosphorus GES targets would need to be binding, the measures with joint effectiveness should be a part of the cost-efficient solution and relatively effective in reducing the nitrogen compared to phosphorus. As we have shown above, the optimal abatement set for nitrogen in Swedish conditions is sensitive to model scale and related data. It is not far fetched to expect that also the optimal phosphorus abatement can diverge from the homogeneous solution when the data resolution increases. Therefore, both the nitrogen and phosphorus GES targets could be relevant at the same time, but at different locations. Investigating the impact of spatial resolution on optimal phosphorus abatement can utilise the model frame developed in this study, but the gathering of the relevant empirical data and accurate description of the nutrient interactions warrants writing a separate paper on the topic.

4.4 Policy implications of homogeneous catchment modelling

Homogeneous catchment modelling can lead to targeting different abatement sectors incorrectly. As demonstrated by our results, the optimal abatement allocation in the homogeneous model specification increased the quantity of nitrogen removal from waste water only by less than 1 %, while the heterogeneous specification leads to investing in several WWTPs, and an average reduction of more than 90 %. Therefore, policies based on homogeneous catchment models can lead to incorrectly targeting only agriculture. Situations in which this occurs could be prevalent around the Baltic, as the major population centres are located by the coast i.e. in low retention areas. However, the misallocation between the sectors is not evident in the existing Baltic Sea models, because they apply various rudimentary modelling strategies that differentiate the retention rate of the WWTPs from the retention rate of the diffuse sources Gren, Savchuk, and Jansson (2013) and Ahlvik et al. (2014). The Baltic models also use different sources for costs functions, which can elevate the importance of WWTP in the optimal abatement for a number of reasons not necessarily related to the homogeneous modelling of catchments. Furthermore, joint optimisation of nitrogen and phosphorus in these models seems to favour phosphorus reduction with nitrogen reduction as a complementary product Ahlvik et al. (2014) and Hasler et al. (2014), which thus shifts the optimal abatement strategy to depend on phosphorus cost functions.

In general, the required abatement effort was underestimated by the homogeneous model specification. For policy design, using such model would lead to underestimating the incentives required to achieve the water quality targets. The Nordic schemes for reaching the water quality targets have been generally found not reaching their targets. Our results suggest that this insufficiency of the existing policies is supported by the use of, what seems, too optimistic assumptions that overestimate the effects and underestimate the costs of nutrient reductions.

Besides underestimating the required scale of measures within the agricultural sector, using homogeneous catchment models can lead to under or over promoting particular measures. For example the role wetlands can be overemphasized at the expense of other measures such as reductions in animal units or tillage changes. All models that include wetlands as abatement measure seem to consider them as low cost measures that contribute a significant share of the nitrogen abatement effort Elofsson (2003), Ahlvik et al. (2014), Gren, Savchuk, and Jansson (2013), and Hasler et al. (2014). Our results suggest that wetlands can be more costly and have less total potential in nutrient abatement than previously considered for the Baltic Sea.

However, we must acknowledge that the current resolution provided by Swedish data cannot be replicated for all catchments of the Baltic Sea without further applications of environmental models such as HYPE or SWAT that allow partitioning of the catchment parameters. Furthermore, we have not considered nitrogen dynamics, including the retention of nitrogen in the sea, which could reduce the need for load abatement from land. Since the required abatement effort to reach the reduction target set to reach GES is high (for example, conversion of nearly half of the farm land to fallow), ignoring the capacity of the sea to clean itself in setting the target levels, would be an expensive mistake for the Swedish environmental authorities to make. In terms of the compliance to WFD, it seems that the exceptions from reaching GES, granted based on excessive costs of implementation, would be influenced by the spatial scale of models used to calculate the costs. Therefore, in the European courts, it could matter whether for example EU level model was used in contrast to more local estimation methods.

5 Conclusions

The resolution of spatial data influences the distribution of optimal nitrogen abatement measures, given model structures and data sources relevant for water quality targets in Sweden. Partitioning the watersheds to sub-catchment areas, changes the optimal abatement set, in this case, increasing the required abatement effort and costs, as well as, it's geographical distribution. While *a priori*, increasing resolution opens up possibilities to target abatement measures more effectively, providing potential for cost decreases, the results show an overall cost increase due to improved understanding of interactions and limitations of the abatement measures. Since gaining effectiveness through targeting could imply further transaction costs, which were not included in the analysis, we conclude that reaching water policy targets in countries around the Baltic Sea is more costly than what has been previously communicated to policy-makers.

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A Model parameters

appendix

B Tables of symbols

Table A.1: Estimated N load reduction required to reach GES

	GES tonnes
571240-121000	256.50
571720-120640	17.21
572135-120141	6.60
572472-120302	81.83
572072-115880	0.59
572227-115662	0.05
572308-115550	3.17
572838-115515	0.74
572980-115576	5.07
573100-115580	1.84
573173-115587	0.29
573322-115478	6.27
573500-115150	8.15
573860-115000	1.02
574050-114780	2155.62
574000-114230	0.43
574370-114250	1.23
574650-114360	3415.46
574870-113795	14.35
575500-113750	6.45
575700-114240	21.24
575747-113237	0.42
580025-113168	3.50
580325-113500	1.98
580338-112901	0.94
580500-112970	0.21
580500-114725	8.46
580530-112700	0.61
580610-113615	12.20
580650-113000	0.33
580688-114860	4.74
580765-112501	0.28
580860-114560	0.03
581120-112680	2.34
581260-113220	1.53
581260-115280	3.38
581520-113750	0.05
581540-114000	1.70
581740-114820	6.55
582000-115270	23.10
581450-113140	0.24
581570-113040	0.14
581700-113000	7.96
582500-113890	91.65
581853-112736	0.76
582000-112350	0.14
582150-112530	6.89
582210-111880	2.72
582230-112255	7.17
582302-111451	0.49
582630-113515	8.57
582850-111760	4.29
583050-110650	0.53
583160-111551	0.75
583710-111535	29.73
584030-111400	2.36
584363-110971	0.26
584400-116000	0.39
584450-111445	6.60
584670-111300	2.02
584890-110950	6.53
585200-111140	2.43
580550-112460	0.03
581338-112332	0.01
581365-112910	0.25
582420-111370	0.07
585160-110880	0.06
585290-110830	0.29
585660-112590	0.33
564500-122601	118.77
570900-121060	40.40
574931-113131	0.10

Table A.2: Sum of current catchment and distance class treatment volume in person equivalents of BOD for different treatment technologies and size classes

	w2u1	w2u2	w3u1	w3u2	w3u5
208	2476	0	5303	0	0
209	4361	0	0	0	0
211	0	0	0	35150	0
212	7000	0	0	0	0
213	158586	193652	26943	249131	0
214	13730	0	0	12862	640157
215	0	0	0	31657	0
217	3844	0	0	0	0
218	8831	0	27104	0	0
238	4099	0	0	0	0
239	1572	0	8202	0	0

Table A.3: Catchment area input WWT and retention

	N load - WWT	BOD PE	ρ_i
208	0	7779	4.26
209	31	4361	9.05
211	32	35150	9.84
212	0	7000	11.11
213	1922	628312	7.84
214	22	666749	8.39
215	0	31657	12.83
216	0	0	5.24
217	22	3844	7.78
218	32	35935	7.14
238	2	4099	1.20
239	0	9774	2.80

Table A.4: Catchment area input data summary agriculture

	N load - agriculture	Field area (ha)	Animal Units	Buffer capacity (ha)
208	176	9246	5355	93
209	123	6678	3186	732
211	52	2692	642	313
212	60	3467	2227	0
213	9109	527330	250833	54364
214	460	25625	9474	1571
215	62	3598	1400	762
216	79	4725	1616	382
217	341	20229	6843	3218
218	368	19841	7707	1383
238	23	3868	316	0
239	3	1373	42	0

Table A.5: Crop area (ha) and the shares that they cover of total agricultural area in catchment

	wwhe1	sbar1	wrap1	pota1	beet1	gra1	oat1	swhe1	fal0	engy1	% F.A.
208	646	1019	60	23	0	5347	865	451	807	27	81
209	258	429	26	15	0	4735	517	204	480	13	94
211	115	134	9	4	0	1933	182	79	229	6	97
212	248	257	19	7	0	2083	300	151	388	14	78
213	64370	61866	10610	3523	22	242477	81660	16282	45131	1389	91
214	458	583	16	30	0	19441	2465	404	2209	19	85
215	172	224	14	8	0	2033	564	75	506	2	95
216	139	169	12	12	0	3064	568	54	705	2	92
217	251	1331	53	5	0	12153	2073	475	3864	23	96
218	616	837	212	31	0	11300	2423	535	3870	16	92
238	78	88	39	6	0	2776	393	70	416	3	20
239	32	43	0	0	0	978	170	37	111	0	6

Table A.6: Parameters for animal related data

	p_o	σ_{cattle}	η_o	LSU	l_o
Cow	3256.00	8690.48	112.00	1.00	47.97
Sucler	818.00	6952.38	63.00	0.80	0.00
Calf	504.00	1738.10	37.30	0.40	0.00
Beef	378.75	5214.29	58.00	0.83	1.90
Grover	694.00	0.00	9.00	0.30	0.20
Sow	99.00	0.00	19.00	0.50	15.00
Boar	99.00	0.00	9.00	0.30	0.00
Piglet	106.00	0.00	2.30	0.03	0.00
Chicken	26.24	0.00	0.00	0.01	0.99
Turkey	56.72	0.00	2.52	0.03	0.99
Broiler	26.24	0.00	0.50	0.01	0.99
Hen	24.26	0.00	0.59	0.01	0.99
Sheep	70.00	2607.14	14.29	0.10	1.00
RamEwe	0.00	0.00	0.00	0.10	0.00

Table A.7: Yield and nutrient load parameters for crops

	a	b	c	ttf	rf
wwhe1	3866.00	35.73	-0.10	1.00	1.00
sbar1	3411.50	23.95	-0.08	1.00	1.00
sbar2	3070.35	21.56	-0.07	0.70	1.00
sbar3	3070.35	21.56	-0.07	1.00	2.00
sbar4	2899.78	20.36	-0.07	0.70	0.70
wrap1	1948.20	14.04	-0.04	1.00	1.00
pota1	27109.10	151.10	-0.52	1.00	1.00
pota2	25753.64	143.54	-0.49	0.70	1.00
beet1	34769.30	144.03	-0.45	1.00	1.00
beet2	33030.84	136.83	-0.43	0.70	1.00
gra1	2023.00	39.44	-0.06	0.70	1.00
oat1	2248.40	27.89	-0.09	1.00	1.00
swhe1	3411.50	23.95	-0.08	1.00	1.00
swhe2	3070.35	21.56	-0.07	0.70	1.00
swhe3	3070.35	21.56	-0.07	1.00	2.00
swhe4	2899.78	20.36	-0.07	0.70	0.70
fal0	0.00	0.00	0.00	0.50	0.60
fal1	0.00	0.00	0.00	1.00	0.80
engy1	7583.70	50.75	-0.23	0.70	0.50
ley1	2023.00	39.44	-0.06	0.70	1.00

Table A.8: Price and cost data for crops (index corrected to year 2010 from Brady 2001)

	p_j	s_j^{crop}	c_j^{crop}	p_j^{wage}
wwhe1	1.52	0	2852.03	0.32
sbar1	1.28	0	2282.24	0.32
sbar2	1.28	0	2435.82	0.32
sbar3	1.28	0	2589.41	0.32
sbar4	1.28	0	2589.41	0.32
wrap1	3.02	0	4096.06	0.41
pota1	0.55	1750	15192.42	0.12
pota2	0.55	1750	15192.42	0.12
beet1	0.24	0	9709.51	0.04
beet2	0.24	0	9709.51	0.04
gra1	0.74	300	3455.62	0.19
oat1	1.16	0	2128.66	0.32
swhe1	1.52	0	2128.66	0.32
swhe2	1.52	0	2282.24	0.32
swhe3	1.52	0	2435.82	0.32
swhe4	1.52	0	2435.82	0.32
fal0	0.00	0	722.00	0.00
fal1	0.00	0	1001.36	0.00
engy1	0.34	0	2457.33	0.18
ley1	0.74	300	1727.81	0.00

Table A.9: Profit maximising crop allocation (as percent of crop area derived from the municipal crop data)

	wwhe1	sbar1	wrap1	pota1	gra1	oat1	swhe1	fal0	ley1
208	148.50	148.50	148.50	148.50	76.00	148.50	148.50	186.95	41.48
209	148.50	148.50	148.50	148.50	79.68	148.50	148.50	326.58	44.32
211	148.50	148.50	148.50	148.50	39.92	148.50	148.50	537.55	30.00
212	148.50	148.50	148.50	148.50	70.59	148.50	148.50	219.61	39.03
213	148.50	148.50	148.50	148.50	92.17	148.50	148.50	0.00	47.91
214	148.50	148.50	148.50	148.50	63.56	148.50	148.50	452.73	36.12
215	148.50	148.50	148.50	148.50	89.96	148.50	148.50	114.69	50.22
216	148.50	148.50	148.50	148.50	66.90	148.50	148.50	238.32	38.37
217	148.50	148.50	148.50	148.50	69.31	148.50	148.50	191.67	38.27
218	148.50	148.50	148.50	148.50	81.05	148.50	148.50	148.04	45.13
238	148.50	148.50	148.50	148.50	15.21	148.50	148.50	607.15	8.58
239	148.50	148.50	148.50	148.50	5.67	148.50	148.50	812.51	3.16

Table A.10: Key for crop and tillage types

Abreviation	Description
wwhe1	winter wheat
sbar1	spring barley
sbar2	spring barley delayed tillage
sbar3	spring barley undersown catch crop
sbar4	spring barley undersown catch crop and spring tillage
wrap1	winter rape
pota1	potatoes
pota2	potatoes with delayed tillage
beet1	sugarbeet
beet2	sugarbeet with delayed tillage
oat1	oat
gra1	grass silage
ley1	pasture grass
swhe1	spring wheat
swhe2	spring wheat delayed tillage
swhe3	spring wheat undersown catch crop
swhe4	spring wheat undersown catch crop and spring tillage
fal0	more than one year fallow
fal1	one year fallow

Table A.11: Table of symbols: Sets

Symbol	Description
i	Set of catchments
d	Set of distance zones within catchments
l	Set of littoral stretches
m	Set of abatement measures
f	Size class (in PE BOD) of the WWT facility
w	Set of WWT technologies
u	Number of WWTP (of specific type and size)
j	Set of farmed crop types
k	Set of farming technology
o	Set of farm animal types
r	Set of retention increasing measures

Table A.12: Table of symbols: Variables

Symbol	Description
E_i	Sum of N load from catchment area reaching the sea
E_l	N load reaching littoral stretch l
$e_{i,d}$	N load at outlet of distance zone in catchment
A_i	Abatement of N (for whole catchment i)
$a_{i,d,m}$	Abatement of N in distance zone in catchment for m
$c_{i,d,m}$	Cost of abatement
C_i	Cost of abatement for catchment i
$x_{i,d,j,k}$	Field (agriculture) area
$n_{i,d,j,k}$	Nitrogen fertilisation (chemical)
$z_{i,d,o}$	Quantity of animals
$n^{manure}_{i,d,j,k}$	Nitrogen fertilisation with manure
$b_{i,d,j,k,r}$	Share of buffer zones and wetlands from agriculture area
%	Refers to percent change in the variable
*	Refers to an optimal level of variable
$\bar{\cdot}$	Refers to a fixed level of variable
$\hat{\cdot}$	Refers to a level of variable associted with GES

Table A.13: Table of symbols: Parameters

Symbol	Description
$\rho_{i,d}$	Retention
$\psi_{i,l}$	Fraction of N load from catchment i to l
ω	Abatement efficiency of wetland
$\upsilon_{i,d}$	Upstream area
$\zeta_{i,d}$	Nitrogen load from all other sources
$\theta_{i,d}$	Amount of waste water to be treated (in PE BOD)
ξ	Coefficient for WWTP abatement cost
α_s	Coefficient for WWTP abatement cost
β_s	Coefficient for WWTP abatement cost
pi_i	Profit from agriculture
η_o	N content in manure
μ_o	Fixed ratio between animal types
σ_{cattle}	Grass consumption requirement for cattle
t	Number of distance zones
y_j	yield from crop j
y_j^ι	Crop yield N response function coefficient
y_j^κ	Crop yield N response function coefficient
y_j^λ	Crop yield N response function coefficient
τ_k	Crop yield function coefficient for tillage and catch crops
s_j^{crop}	Subsidy for crop j
c_j^{crop}	Cost per ha of crop cultivation
p_j	Producer price for crop j
p_o^{animal}	Standard output (revenue per animal)
p_N	Price of fertiliser per kg N
p_j	Producer price for crop j
p^{wage}	Price of agricultural labour
l_o^{animal}	Annual labour requirement per animal
$l_{j,k}^{crop}$	Annual labour requirement per crop
$c^{transport}$	Transport cost of manure
$c_r^{retention}$	Cost of retention increasing measure r
$s_r^{retention}$	Subsidy for retention increasing measure r
h^1	Impact of crop rotation and tillage timing on the N load
h^2	Share of N in the surface flow of the total nitrogen load
h^3	Share of N in the subsurface flow

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