Paper on effects of socioeconomic scenarios on nutrient loading, GHG emissions and soil organic carbon (manuscript)

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Water quality management and climate change mitigation: cost-effectiveness of joint implementation in the Baltic Sea region

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Highlights

- We model the scope for a cost-effective strategy to simultaneously tackle nutrient abatement and climate change mitigation using selected measures
- The results show that up to a certain degree it is economically efficient to integrate climate change mitigation target and water regulation
- Asymmetry in biophysical and economic effects between countries of the realization of the joint strategy further supports the call for regional cooperation
- The results highlight the opportunity for collective, cost-effective strategy for joint water and climate regulation

Abstract

This paper explores the scope for simultaneously managing nutrient abatement and climate change mitigation in the Baltic Sea (BS) region through the implementation of a selection of measures. The analysis is undertaken using a cost-minimisation model for the entire BS region, the BALTCOST model. In the present research, the model has been extended to include greenhouse gas (GHG) emissions effects enabling us to analyse the trade-offs between cost-effective GHG- and nutrient load reductions. Our findings show that implementing land-based measures to meet the HELCOM's Baltic Sea Action Plan nutrient abatement targets (2013) produces climate change mitigation co-benefits equivalent to 2.3 % of the 2005 emission level (from agriculture and waste water combined) for the entirety of the BS region. Further climate change mitigation benefit (i.e. up to 5.4%) can still be obtained at a marginal cost that is comparable to mitigation costs reported by other studies. Our findings show that the cost and the outcome of the implementation vary between countries. All in all the present study illustrates the need to develop a joint regional strategy for water and climate regulation that fully considers the asymmetry in both the expected effects and cost distribution across the countries in the region.

Key words: trade-off; water management; climate change mitigation; co-benefit; cost-minimisation; economic-hydrological model

1. Introduction

The Baltic Sea is an example of an international sea area where collective actions have been agreed by intergovernmental fora to regulate environmental quality. One of these fora is the Helsinki Commission (HELCOM). Nutrient loads to the sea are responsible for the eutrophication of large parts of the central and coastal Baltic Sea area, and HELCOM has declared eutrophication as one of the most serious threats to obtain healthy ecosystems in the Baltic Sea and the delivery of important ecosystem services. The HELCOM contracting parties (the riparian countries and EU) have agreed on nutrient reduction targets[1] and adopted in the Baltic Sea Action Plan (BSAP) (HELCOM 2007, 2013). The BSAP defines maximum levels of total phosphorus and nitrogen loads to the sea such that the sea ecosystem can recover and a good environmental status can be reached in the future. The load quotas are measured as maximum allowable inputs from each of the riparian countries and to each of the 7 sea basins. The BSAP targets were first agreed in 2007, and revised later in the HELCOM Copenhagen Ministerial Meeting in October 2013 (HELCOM, 2013).

The HELCOM contracting parties have agreed to align the implementation of the BSAP with other policy objectives in order to enhance efficiency and to reduce conflicts between policies (HELCOM 2013). One of these policies is the international and EU policy to reduce greenhouse gas emissions (GHG). The implementation of policies and measures to control both nutrient losses and GHG emissions might lead to conflicts, but also synergies. Most policy evaluations to date deal with the assessments of individual policies, but coherence and coordination of policies are required to attain efficient outcomes (Bennear & Stavins 2007).

Previous research have developed models to analyse the cost-effectiveness of nutrient reduction policies to the Baltic Sea at various spatial scales and with different types of data (Elofsson, 2010; Gren et al., 2013; Wulff et al., 2014; Hasler et al., 2014, Hyytiainen et al., 2014; Ahlvik et al., 2014, Gren et al., 1997, Turner et al., 1999, Ollikainen and Honkatukia 2001, Schou et al., 2006; Gren 2008). Elofsson (2010) provides a comprehensive review of this research. These Baltic wide studies on nutrient load reduction conclude that restoring the Baltic Sea will be expensive. It is therefore of policy relevance to explore how national and regional/international implementation of collective actions to

[1] One of the main foci of the BSAP is combatting eutrophication, but BSAP also sets targets to reduce the loads of hazardous substances, to improve biodiversity and to regulate maritime activities. In this report we focus only on eutrophication.

reduce both nutrient loads and GHG emissions, individual and simultaneous, influences the total costs of achieving the environmental objectives, the distribution between the countries, as well as to how synergies and potential conflicts influence the effects of the policies. Beyond total aggregate costs and costs distributed on countries, it is also of interest to explore spatial differences in nutrient abatement and GHG emission mitigation, as well as related costs. Spatially explicit modelling of abatement effectiveness and abatement cost has proven to be essential for identifying cost-effective combinations of abatement measures (Konrad et al., 2014; Iho, 2005; Iho and Laukkanen, 2012). In the Baltic Sea region this is especially important because of the heterogeneity in catchment characteristics, land use and agricultural production as well as differences in the sea regions capacity for receiving nutrient loads.

GHG reductions are regulated at an international multilateral level according to the United Nations Framework Convention on Climate Change (UNFCC), but also EU has made a unilateral commitment on 20% reductions of GHG emissions from 1990 levels by 2020. The EU policy includes the Emission trading scheme (ETS), which regulates industry, and the Effort sharing decision scheme (ESD). The ESD includes most sectors not included in the EU ETS, such as agriculture and waste, as well as buildings and transport (except aviation and international maritime shipping) (EC, 2016). Emissions and removals from land use, land-use change and forestry (LULUCF) are currently not included in the ESD (EC, 2015). The difference between the ETS and the ESD is that while sectors are allowed to trade across country boundaries in the ETS, the ESD implies binding annual GHG emission targets for Member States. Similar to the BSAP the allocation of the GHG reduction targets are set according to equity and fairness, but not cost-effectiveness (De Cara & Jayet (2011). According to De Cara & Jayet's analysis the costs of a 10% reduction in EU could be reduced by a factor of two to three compared to the fixed targets; if a flexible cap-and-trade system were introduced and a more cost-effective distribution among these countries could be obtained. Their results indicate that an introduction of cap and trade would imply that new member states (PL, LT, LA and EE) would sell permits to old (DK, GE, FI and SE), and thereby the allocation of emissions reductions would change considerably.

A literature review by Balana et al. (2011) highlights that existing studies on cost effectiveness analysis of implementing measures to mitigate water pollutants have focused solely on the direct impacts, and neglected potential co-benefits and unintended consequences. Since this review in 2011 only a few studies have been accomplished addressing the costs and effects of implementing nutrient and climate policy objectives simultaneously (e.g. Eory et al., 2013; Gren & Säll, 2015; Konrad et al., 2017). The studies have very different spatial coverage. Eory et al. focused on the UK. The work of

Konrad et al. studies a catchment (Limfjorden catchment in Denmark), whereas the work of Gren & Säll (2015) covers the entire BS region. Gren and Säll (2015) analyze cost-effective multi-target management of nutrient and GHG emissions in the Baltic Sea, and state that simultaneous management of targets on both nutrients and GHG-emission reduce costs compared with separate management, if measures are complementary in pollutant abatement and the same source emits more than one pollutant. The study includes sources being inside and outside the ETS. They conclude that the multi-target implementation reduces total costs by 11% compared to separate management. The Gren and Säll (2015) study use data on nutrient emissions from Gren et al. (2008) and on GHG emissions from Gren et al. (2012). Gren and Säll's study therefore seems to compare results from minimization of the costs of abating GHG emissions and nutrient loads from model versions with different assumptions, being run for different years. Gren and Säll furthermore claim that the location of the source does not matter for the climate impact, which we agree on in principle, but GHG emission effects of land-use measures do, contrary to Gren and Säll's expectations, vary according to climate zone, soil types etc. A more spatial approach is therefore justified, taking heterogeneity between catchments into consideration for the optimal localization of measures.

A number of land-use changes and measures are spatially specific in terms of the effects and costs and some measures cause changes in both GHG emission and nutrient load levels. We therefore find that it is of high relevance to investigate the scope for jointly delivering cost-effective nutrient abatement and reductions in GHG emissions within a spatial modelling framework for the Baltic Sea. By applying the analysis to the Baltic Sea the paper contributes as an example of how international collaborative agreements can be improved by cost-effective allocation. This is done by comparing the cost-effective and flexible allocation of measure with the country specific allocations of emission targets agreed on in the BSAP and ESD. The analysis is undertaken using and further developing a cost-minimisation model for the entire Baltic Sea region, the BALTCOST model (Hasler et al 2014), which is an economic-hydrological model applied with high spatial resolution data for the entire Baltic Sea catchment. In the current paper, the model has been extended to include GHG emissions at the same spatial resolution as the nutrient load modelling. More specifically the present paper aims to model scenarios for cost-effective, joint water and climate strategies; investigating the economic consequences of different implementation scenarios compared to the current policies.

The remainder of the paper proceeds as follows. Section 2 describes the additions made to the BALTCOST model to include GHG effects, together with the data sources and methodologies that were used to estimate cost functions, effect functions on GHG and nutrient loads, capacity constraints and nutrient retentions for each abatement measure in each main drainage basin. Section 3 presents the modelling results. Section 4 discusses the results and section 5 concludes the paper.

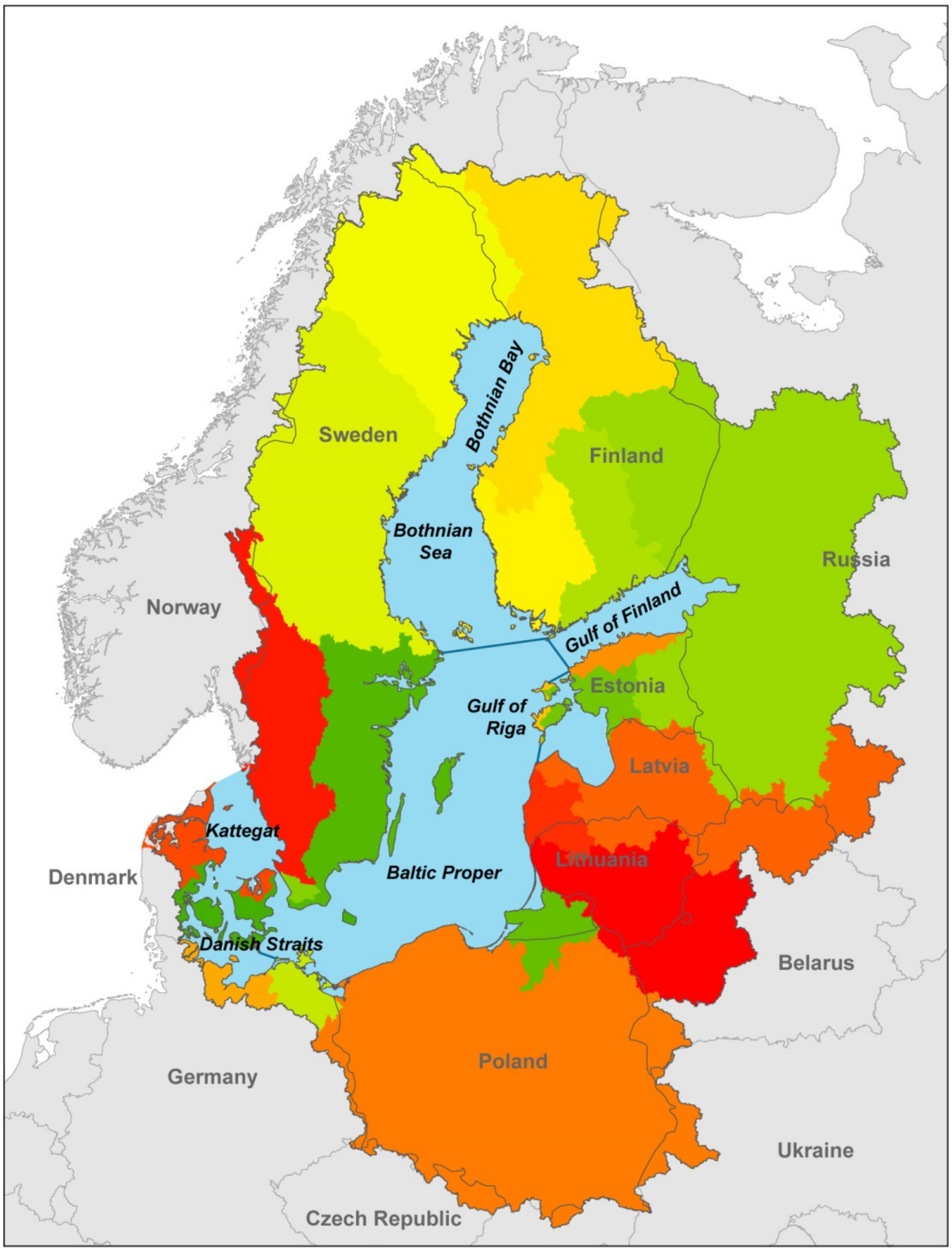
2. Methods

2.1. Modelling platform

To meet the objective of the paper, we extend the hydro-economic model BALTCOST (Hasler et al., 2014). The methodological contribution of the present paper is the incorporation of climate change mitigation objectives, effects and capacities for GHG reduction of each of the measures into the BALTCOST model.

BALTCOST is a non-linear optimisation model for the Baltic Sea developed collaboratively by natural scientists and economists (Wulff et al. 2014, Hasler et al. 2014). BALTCOST calculates the minimum total abatement cost incurred at main drainage basin resolution to enforcing delivery of Nitrogen (N) and Phosphorus (P) load reduction targets to the sea regions, as well as GHG-emission reduction target for the entire Baltic region and countries. The solutions are found, within modelled abatement capacity constraints of the measures included in the model (see below). The results are total costs (TC), total load and emission reduction effects for the whole Baltic as well as marginal abatement costs (MAC) and distribution of abatement measures. A map of the Baltic Sea region, the countries (9), the sea basins (7), and the drainage basins (22) is presented in Figure 1. For further details regarding BALTCOST, readers are referred to Hasler et al. (2014).

Figure 1. The 9 Baltic Sea riparian countries, the 7 Baltic Sea regions, and 22 main drainage basins (coloured).



BALTCOST includes 6 different abatement measures in each of 22 main Baltic drainage basins, where all of them are characterised by their costs, capacity for N, P and GHG reductions, effect on the N leakage and retention, the GHG emissions and for some of the measures also the P load reductions (Table 1). More information regarding the characteristics of the measures can be found in the Appendix. Cost- and effect-components in BALTCOST are tightly integrated as both elements draw on the same database of spatially-specific biophysical 10x10 km grid cell data where these data are used for the estimation of costs of each of the measures when implemented in the drainage basins.

BALTCOST regards the 6 abatement measures to be completely independent as most of these measures can be implemented at the same time and location[2]. BALTCOST identifies the most cost effective combination of abatement measures to deliver the desired load reduction targets for N, P and GHG simultaneously as well as one by one. Thus, where measures deliver N and P abatements as well as effects on the GHG emissions, reductions in both nutrients and the GHG emissions are accounted for when that measure is used. This method means that to comply with one reduction target, e.g. nitrogen, for one specific sea region, the reduction targets for the two other emissions/loads might be over-fulfilled.

Table 1. List of measures, N, P and GHG effects, costs and capacities

Measures	N	P	GHG	Costs	Capacity
Reductions in fertiliser application to arable crops	Crop and location specific leakage function (Andersen et al 2016) Retention in ground water and surface water added.	No effects (no yield function for P)	Direct emissions from synthetic fertilizers and manure application as well as indirect emissions, IPCC 2013.	Danish experimental yield functions for nitrogen application to crops (wheat, barley, rye, oats, rape, sugar beet, potatoes, clover grass, temporary grass), applied to crops in 10X10 km grid, country specific prices used for calculating yield loss costs (Eurostat)	20 percent of the initial N application level for the particular crop and drainage basin concerned. The 20 percent capacity limit reflects caution in extrapolating quadratic-form yield functions outside the range of their parameterisation data.
Catch crops under spring sown cereals	35% reduction in leakage, estimated for 10 x 10km grids. Retention in ground water and surface water added.	No effect	Increase. Larger increase at sandy soils than clay. Olesen et al 2013.	Additional seeds, no additional sowing and harvesting costs, no yield loss. Price of seeds: Eurostat	Equal to the drainage-basin specific area currently cultivated with spring barley and oats, calculated by summation of cropping areas at 10 x 10km resolution.
Restoring	Uniform:150	Uniform:	On-site CO_2-C emis-	Opportunity	GIS data on distribu-

[2] This does not hold for wetland, as the other measures fertilizer reduction and catch crops cannot be implemented at wetlands. On the other hand the spatial units used in the optimization are large and there are possibilities to implement more than one measure in each of these units.

wetlands on agricultural histosoils	kg N Retention in surface water added.	0.7 kg P Retention in surface water added.	sions/removals from the soil and non-tree vegetation, off-site CO_2-C emissions from dissolved organic carbon exported from rewetted organic soils, CH_4-C emissions from rewetted organic soils. Area distribution of climatic zones (boreal versus temperate), nutrient status of agricultural areas in drainage basins within each of the Baltic countries IPCC 2013.	costs of baseline land use in 10x10 km grid, costs: Eurostat	tion of histosoils, agricultural land use. 1.69 percent of the total Baltic drainage area; variation between 0.01 and 15.67 percent of the 22 catchments.
Reductions of the number of pigs	Modelled as difference between leakage from animal manure and commercial fertiliser, assuming that fertiliser input is substituted with commercial fertilisers. Livestock and crop specific in 10x10 km grids. (Andersen et al 2016). Retention in ground water and surface water added.		CH_4 emission from both enteric fermentation and manure management as well as N_2O emission from manure management (IPPC 2013).	Opportunity costs of pig production, Eurostat	20 percent of the current herd sizes in each drainage basin. Further reductions in livestock numbers would be likely to incur additional costs arising from unused animal housing and production facilities such as milking parlours, intensive rearing units etc. (sunk costs)
Reductions of the number of cows				Opportunity cost of cattle production, Eurostat	
Improving waste water treatment	Standard percentage reductions in country-specific N and P at-source loads per PE (Berbeka et al 2012) Retention in surface water added.		Difference in N_2O emission level per person with and without treatment. IPCC 2006.	Cost function based on Danish and Polish data.	Subcatchment-specific upgrading capacities derived at 10 x 10km grid cell resolution data on population sizes in combination with national resolution data on the percentage of population upgradeable to tertiary-level WWT. Summed to 22 catchments.

The 22 drainage basins each comprises between 1 to 16 smaller sub-catchments, of which there are 117 in total. The specification of the nutrient load reduction targets in BALTCOST is consistent with the BSAP (HELCOM 2013), and the targets for nitrogen and phosphorus load reduction are set for sea regions, of which there are 7 in all (See Figure 1 and Table 2).

Table 2. HELCOM'S 2013 BSAP nutrient load reduction targets.

Sea region	**N load reduction target (tons)**	**P load reduction target (tons)**
Bothnian Bay	0	0
Bothnian Sea	0	0
Baltic Proper	98920	10959
Gulf of Finland	14451	3908
Gulf of Riga	0	307
Danish Straits	0	0
Kattegat	4760	0
Total	118131	15174

- Adopted from HELCOM (2013).

As described in Hasler et al. (2014), BALTCOST does not account for nutrient transport between Baltic Sea regions as this transport is considered when the reduction targets for each sea basin were set. The BALTCOST results presented here thus report minimised costs for delivering the required nutrient load reduction targets for each of the 7 Baltic Sea regions separately, rather than for the Baltic Sea as a whole.

As for the GHG emission reduction an incremental range of targets is set for the entirety of Baltic region to explore how much the implementation of the measures can potentially contribute towards existing climate change mitigation policies (e.g. the EU targets for GHG emission reduction), at least cost, given the available capacities of the abatement measures. The maximum GHG reduction capacity is first calculated for each of the drainage basins connected to the 7 sea regions at full capacity of the measures. Then, the GHG reduction capacities are aggregated for the entirety of the Baltic region. The approach is hence different from the one applied for the nutrient abatement where the nutrient abatement targets are restricted to be met at individual Sea Region (SR), while for the GHG emission reduction the target is set for the entire Baltic Sea region. In the context of regional cooperation for climate change mitigation, what matters is how to achieve the GHG emission reductions for the Baltic region as a whole at the least cost, regardless of where in the region the reduction takes place. Moreover, it is important to note that the GHG emission reductions in the model should be interpreted as the reduction of emissions from the drainage basin (terrestrial) area to the air and thus it does not account for GHG emissions from the seas in the Baltic region.

- The GHG emissions are estimated as CO_2 equivalents using IPCC guidelines (IPCC 2006; 2013). The 10 x 10 km biophysical data, sampled and modelled in the BONUS project RECOCA (Wulff et al 2014, Andersen et al 2016) was utilized for the parameterisation of BALTCOST (Hasler et al 2014). In the present research the same data is used to down-scale the emission factors for country level obtained from IPCC's countries' reports into the level of drainage basins. The method used for estimating the GHG emission effects of the 6 selected measures is described in detail in the Appendix.

- Cost minimisation of the nutrient abatement is carried out separately for each of HELCOM's 7 Baltic Sea regions, subject to abatement capacity constraints in the river basins, which drain into that sea region. At the same time the cost minimization must meet the GHG emission reductions target for the entire Baltic region. BALTCOST uses the CONOPT v3 solver in GAMS to solve the following cost minimisation problem:

$$\min TC = \sum_{DB} \sum_{m=1}^{m=6} C_{DB,m}\left(a_{DB,m}\right)$$

Subject to the following constraints for each of the seven sea regions (SR):

$$(1) \quad \sum_{DB \in SR} \sum_{m=1}^{m=6} Ef_{DB}^{N}\left(a_{DB,m}\right) \geq T_{SR}^{N}$$

$$(2) \quad \sum_{DB \in SR} \sum_{m=1}^{m=6} Ef_{DB}^{P}\left(a_{DB,m}\right) \geq T_{SR}^{P}$$

And to this constraint for the Baltic region as a whole:

$$(3) \quad \sum_{DB \in BR} \sum_{m=1}^{m=6} Ef_{DB}^{GHG}\left(a_{DB,m}\right) \geq T_{BR}^{GHG}$$

And for each of the drainage basins:

$$\left(a_{DB,m}\right) \leq a_{DB,m}^{\max}$$

where

TC : total cost

$C_{DB,m}$: drainage basin-specific cost function of implementing each measure m

$a_{DB,m}$: intensity of the implementation of measure m in drainage basin DB

$a_{DB,m}^{\max}$: drainage basin-specific capacity constraints for each measure m

m : measures

Ef : biophysical effect of implementing each measure m

T : environmental target

DB : Drainage basin

SR : Sea region

BR : Baltic Region

N : Nitrogen

P : Phosporus

GHG : Greenhouse gas (reduction in emission)

- 2.2 Scenario analysis

Four scenarios are evaluated. The first scenario is referred to as the baseline scenario, where the *nutrient load reduction targets for BSAP* are met, and the magnitude of GHG emission reduction is a spill-over effect (Table 3). We consider the resulting GHG emission reduction spill-over effect as the climate change mitigation co-benefit of water quality management in the Baltic Sea region.

The second scenario is labelled as the *joint implementation strategy*. The notion of "joint" here refers to the fact that climate change mitigation objective is simultaneously considered along with the nutrient load reduction targets in the model run. The starting point of the joint implementation is the level of climate co-benefit obtained under the baseline scenario. As such, the objective of this scenario is to explore how much more climate change mitigation effect can be obtained cost-effectively from jointly implementing nutrient abatement and GHG emission reduction using the selected measures.

The third and fourth scenarios have an exclusive focus on a *climate change mitigation objective*. While the first climate scenario is run to fulfill the baseline GHG reduction, the second climate scenario fulfills the maximum GHG reduction capacity. Insights from these two climate scenarios allow comparison of the biophysical and economic implications of these climate focused policies and the alternative policy addressing climate and water regulations jointly.

We set up the BALTCOST model for the year 2005. The reported GHG emission levels of the Baltic region from agriculture and waste water treatment combined for the year 2005 was 304.6 million tons CO2eq. In both scenarios, climate change mitigation targets are expressed as frac-tions/percentages reduction of the GHG emissions (2005 level). These climate change mitigation targets are gradually increased until no feasible solution can be found.

3. Results

3.1. Overview

Table 3 provides a summary of the environmental and cost implications of implementation of the scenarios. The results show that a cost-effective nutrient abatement strategy to meet the BSAP target has a positive spill-over effect on climate change mitigation through a reduction of just over 7 million tons of GHG emission (Table 3, row 1). Fulfilling the full capacity of the climate change mitigation effect can deliver a further 11.5 million tons of GHG emission reduction at an additional cost of 2000 million Euros. The implementation of this scenario leads to substantial further reduction of the N load (i.e. almost double that of the BSAP target) (Table 3, row 2).

Under the climate focused strategy, such a reduction of 7.1 million tons of GHG can be achieved at costs far lower than the baseline scenario. However, the spill-over effect of implementing this climate change mitigation strategy on nutrient load reduction is less than half of the BSAP target for N and no P effect is achieved (Table 3, row 3). Reducing GHG emissions to the full capacity have significant spill-over effect on N abatement and the BSAP target is met as a spill-over effect; however this is not the case with regard to P (Table 3, row 4).

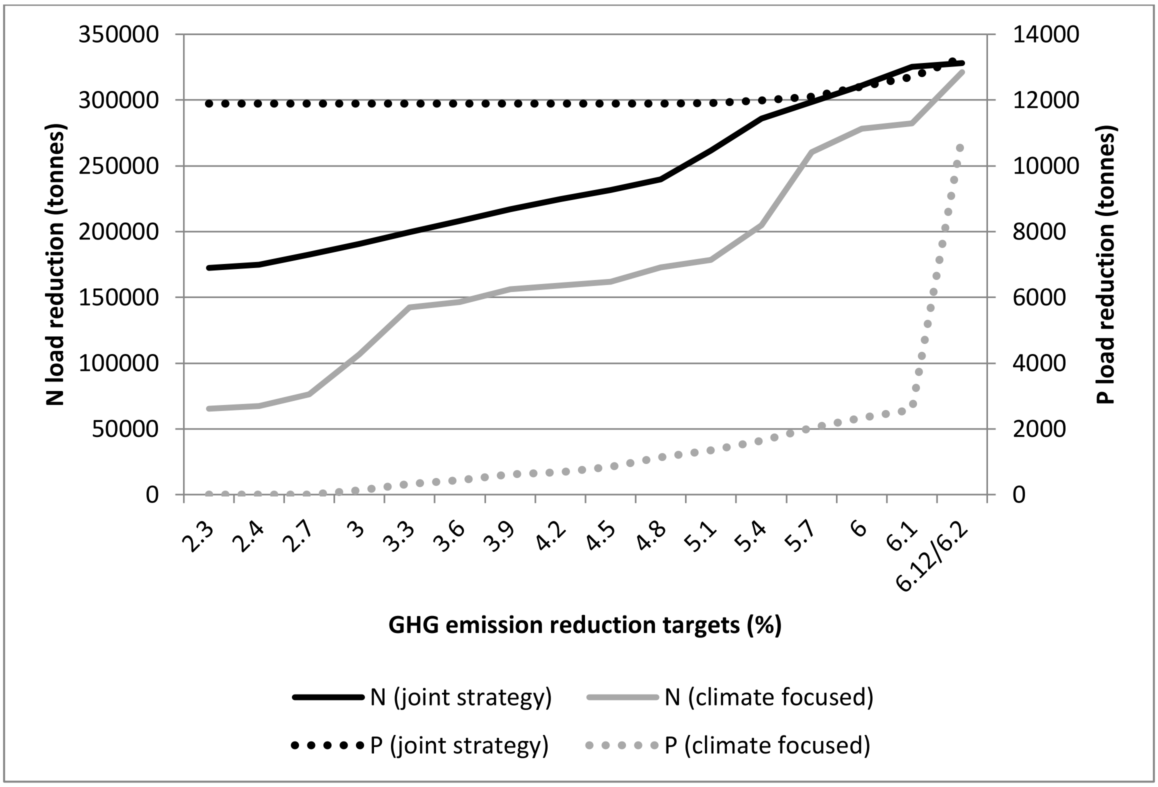
Table 3. Summary of nutrient load reductions, climate change mitigation effect, and total costs of implementing land based measures under different scenarios

Scenario runs	GHG emission reduction delivered (million tonnes)	N load reduction delivered (tons)	P load reduction delivered (tons)	Total cost of strategy implementation (million Euros)
A. Water quality management (meeting BSAP targets) – baseline scenario	7.1 (ca. 2.3 % of the 2005 emission level)	172469	11892	4169.6
B. Joint implementation (meeting BSAP targets and full GHG emission reduction capacity)	18.6 (ca. 6.1 % of the 2005 emission level)	328247	13279	6274.5
C. Climate focused strategy in order to achieve the GHG emission reduction as in baseline (scenario A)	7.1 (ca. 2.3 % of the 2005 emission level)	65232	0	130.9
D. Climate focused strategy implementing the full GHG reduction capacity	18.8 (ca. 6.2 % of the 2005 emission level)	321403	10897	5521.2

3.2. Biophysical impacts

Figure 2 depicts the relation between climate change mitigation and nutrient load abatement. The figure shows pronounced synergistic effect between increasing climate change mitigation target and N abatement under both scenarios (Figure 2. unbroken lines). This is however not quite the case for P abatement (Figure 2, dotted lines). Under the joint implementation scenario, there is no additional P effect of increasing the climate change mitigation target up to 5%. Meanwhile under the climate-focused strategies, gradual increase in P abatement is observed for climate change mitigation targets above 3% with a sharp increase when the climate change mitigation target exceeds 6%.

Figure 2. The effects of increasing GHG reductions on nutrient emissions (N and P load reduction)

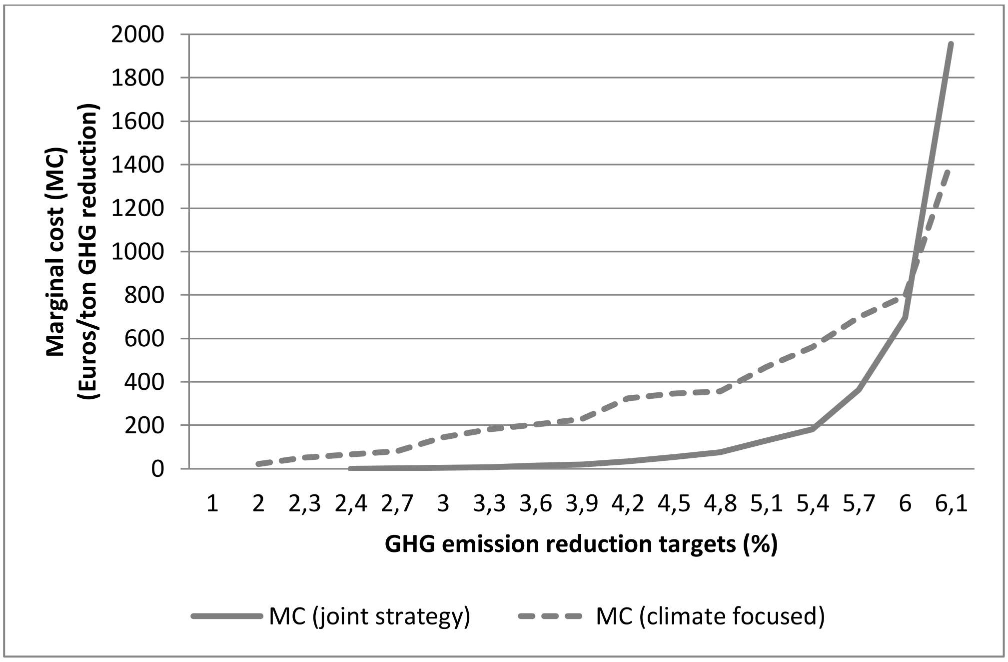


3.3. Cost implications

3.3.1. Marginal cost

The marginal cost under the joint implementation is in general lower than under the climate focused strategy although steep increase becomes evident when the climate change mitigation target is above 6% (Figure 3). When the targets of reducing GHG emission range between the baseline and below 5%, the marginal cost for implementing the combined water and climate regulation strategy ranges between 0.4 to 77 Euros per ton of GHG reduction. Within the same range of climate change mitigation target, the marginal cost under the climate only strategy ranges between 66 to 356 Euros per ton of GHG reduction. This counterintuitive finding results from the significant potential for exploiting the synergistic effects between meeting GHG reductions and the BSAP targets.

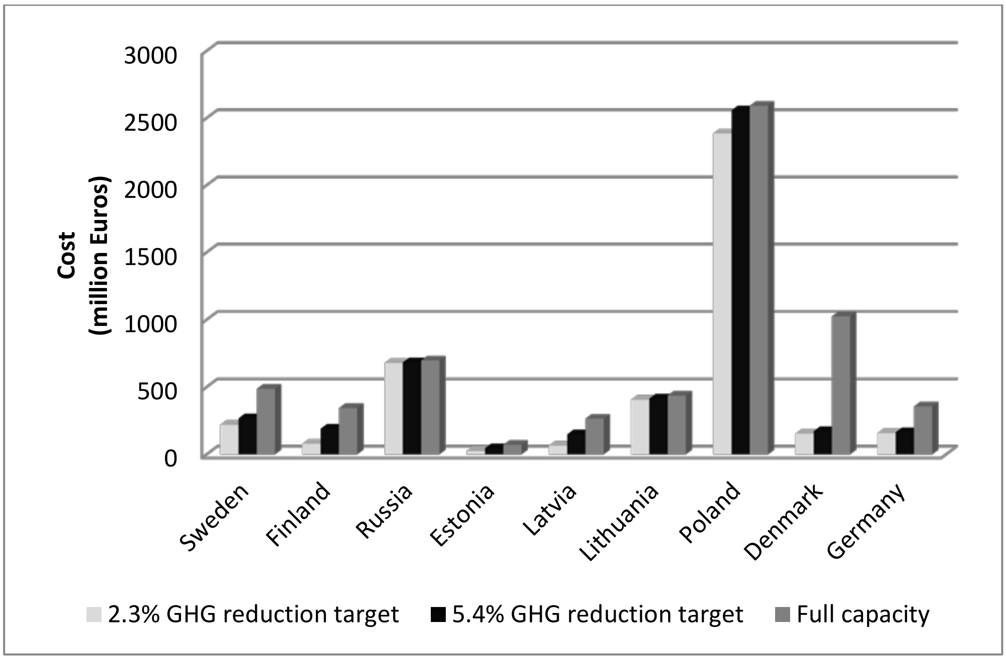
Figure 3. Marginal costs of GHG mitigation under two different strategies (joint versus climate focused)



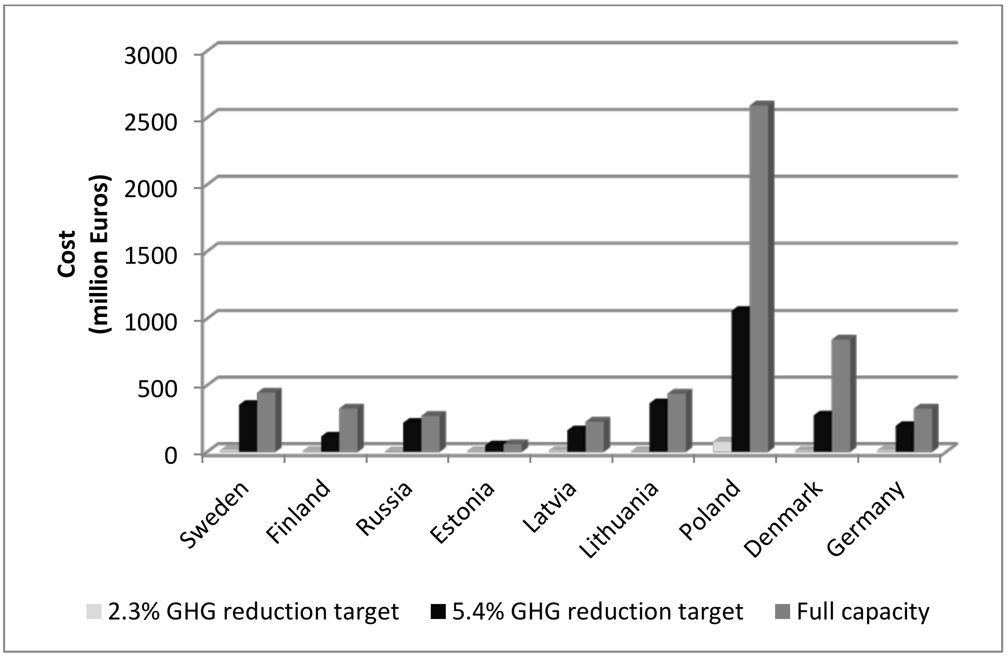
3.3.2. Distribution of costs by countries

For the scenarios under consideration, the modelling results reveal that the cost of implementing effective strategies that consider climate change mitigation by the studied measures for the Baltic region is not evenly distributed across the countries in the region (Figure 4). At the starting point of the joint implementation (i.e. equivalent to the baseline scenario), of the total cost of 4.2 billion Euros more than half is attributed to implementation in Poland (Figure 4a). Russia, Lithuania, and Sweden are placed second, third, and fourth in terms of cost allocation at 16, 9.7, and 5.3% of the total cost respectively. The lowest cost shares are found for Finland (1.9%), Latvia (1.5%) and Estonia (0.6 %). For delivering the climate change mitigation target of 5.4%, the relative shares of the costs for Poland, Russia, Lithuania, Sweden, Denmark, and Estonia follow the same pattern of the baseline scenario. Under this target, Finland and Latvia are to increase in cost share, to 4.1 and 3.6%, respectively.

Figure 4. Distribution of total cost of the implementation of strategies by countries for joint implementation (a) and climate-focused strategy (b) at three levels of climate change mitigation targets



a) Distribution of total costs by countries under joint implementation



b) Distribution of total costs by countries under climate focused strategy

For delivering the climate change mitigation target equivalent to the full GHG reduction capacity of the measures in all the countries, the operationalization of the joint implementation strategy leads to changes in the distribution of the total cost compared to the baseline scenario. Although at this maximum target Poland is still associated with the largest share of the cost, the share has dropped to

41%, however, compared to the baseline situation. Russia and Lithuania are also linked to lower share of the total cost compared to the baseline. As a consequence, increase in cost share is observed for other countries, and the most significant one is linked to Denmark where the share goes up from 3.7% to 16%.

In the case of the climate focused strategy, the largest portion of the total implementation cost at the 2.3% GHG emission reduction target (i.e. the level of emission reduction equivalent to the climate change mitigation co-benefit obtained under the baseline scenario) is also attributed to Poland at 57% (Figure 4b). In contrast to the joint implementation's starting point (i.e. the baseline scenario), here Russia is among the countries linked to the smallest share of the cost (1.2%) together with Lithuania (2.6%), Estonia (2.4) and Finland (4.2%). The other countries are linked to approximately equal share of the total cost which range between 7 to 10 %. To achieve 5.4% GHG reduction, important changes in the relative share of the implementation are linked to Lithuania and Russia (increase in share of cost to 12.2 and 7.3% respectively). At the full capacity of the measures, Poland remains associated with the largest share of the implementation cost albeit at a lower portion of 47% compared to the baseline. Significant increase in the share of the cost is attributed to Denmark (15%) and Lithuania (7.9%). A slight increase in the share of cost is observed for Finland and Russia. The share of the total cost for the remaining countries decrease at this maximum climate target (compared with the baseline situation).

3.3.3. Distribution of costs by measures

Table 4 presents the costs of implementing measures under the four scenarios explored and Figure 5 reflects the trajectory of the utilization of the six different land-based measures along the climate change mitigation targets gradient.

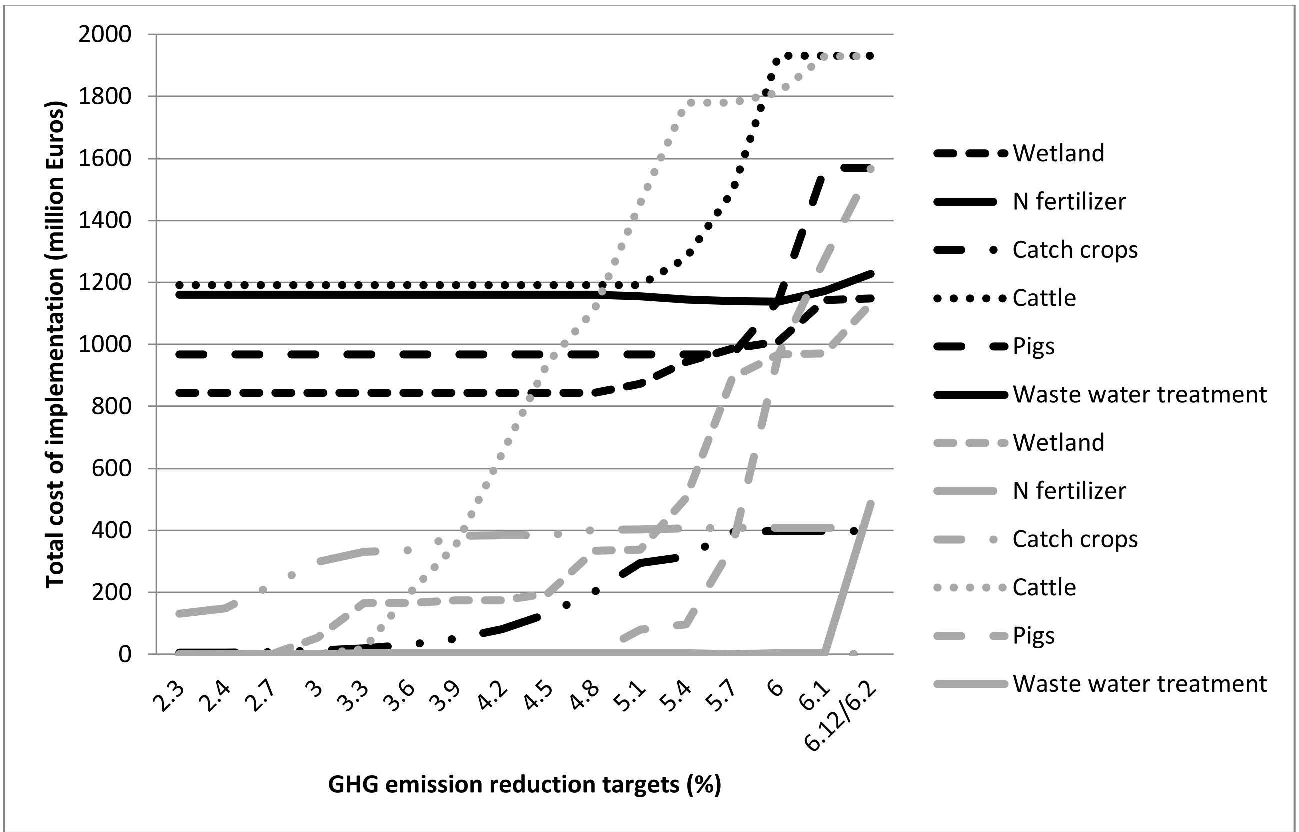
Table 4. Summary of costs of implementing measures under different scenarios (million Euros)

Measure	Scenario A (base-line)	Scenario B Joint implementation	Scenario C: GHG reduction as in baseline	Scenario D: Max GHG reduction capacity
Reductions in ferti-liser application to arable crops	6.0	398.28	130.91	407.92
Catch crops under spring sown cereals	0	0	0	0
Reductions of the number of pigs	967.99	1569.78	0	1566.23
Reductions of the number of cattle	1192.23	1930.98	0	1930.26
Wetlands	843.42	1147.90	0	1130.33
Waste water treatment	1159.96	1227.57	0	486.42
Total across measures	4169.6	6274.5	130.91	5521.2

In the case of joint implementation, the introduction of climate change mitigation target below 6 % does not seem to change the intensity of the utilization of wetland restoration, cattle reduction, pig reduction and sewage treatment at the respective costs of approximately 986, 1500, 970 and 1140 million Euros (Table 4, Figure 5). Instead, the consequence of intensifying the climate change mitigation target from the baseline level is the further activation of the measure involving reduction of N fertilizer as indicated by the rise in the implementation cost of this measure from 6 million Euros to close to 400 million Euros. Furthermore, it is only after the full capacity of N fertilizer measure is reached that the remaining capacities of the other measures are further utilized in the following sequential order: wetland restoration, then livestock reduction, and finally sewage treatment. Under the joint implementation scenario, the catch crop measure appears not to be utilized because of the negative effect on GHG.

N fertilizer reduction appears to be the first measure being utilized in the implementation of the climate focused strategy costing around 130 million Euros. The utilization of this measure continues to increase with increasing climate change mitigation target until the implementation reaches the full capacity of the measure at the cost of around 408 million Euros (at this point the GHG emission reduction is approximately 5.7 %). Wetland restoration becomes utilized when the climate target is at around 3% and its implementation continues to intensify thereafter. At climate target above 3%, other two measures come into use namely cattle reduction and sewage treatment. However, while the utilization of cattle reduction increases following the increasing climate target as indicated by growing cost from around 18 to 1930 million Euros, this is not the case for sewage treatment. The utilization of the latter remains unchanged at a cost of 4.4 million Euros until the climate target is at the maximum capacity. At that point the remaining capacity of sewage treatment becomes activated at a cost of 486 million Euros. The last measure to come into the solution is the pig reduction which takes place at a climate target above 5 % costing around 79 million Euros initially to 1566 million Euros at the maximum level. The catch crop measure does not enter the solution portfolio.

Figure 5. Intensity of strategy implementation in terms of cost by measure for joint implementation of BSAP and GHG emissions targets (black) and GHG only targets (Grey).



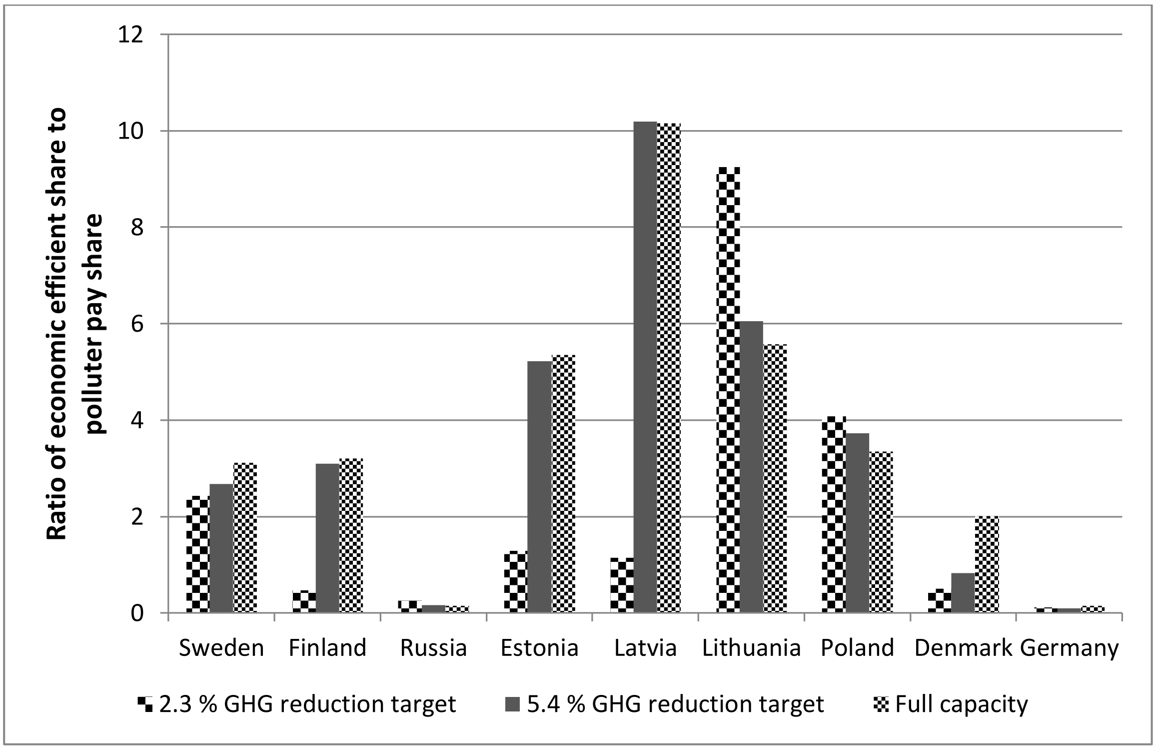
3.4. Comparing economically efficient allocation and polluter-pays-based allocation

Figure 6 shows the ratio between the economically efficient share of abatement between the different countries and the countries' actual share of the total emissions from agriculture and waste water for the entire Baltic region (2005 level). Ratios close to 1 would indicate that allocation of pollution targets according to historical emissions would obtain an efficient outcome. A ratio greater than 1 for a given country indicates its potential for efficient climate change mitigation beyond the level suggested from a polluter pays principles compared to other countries in the region. On the contrary, a ratio less than 1 reflects large actual emissions compared with other countries and that an economically efficient allocation of targets would be in the country's interest as it would reduce the targets compared to targets allocated using a polluter pays principle.

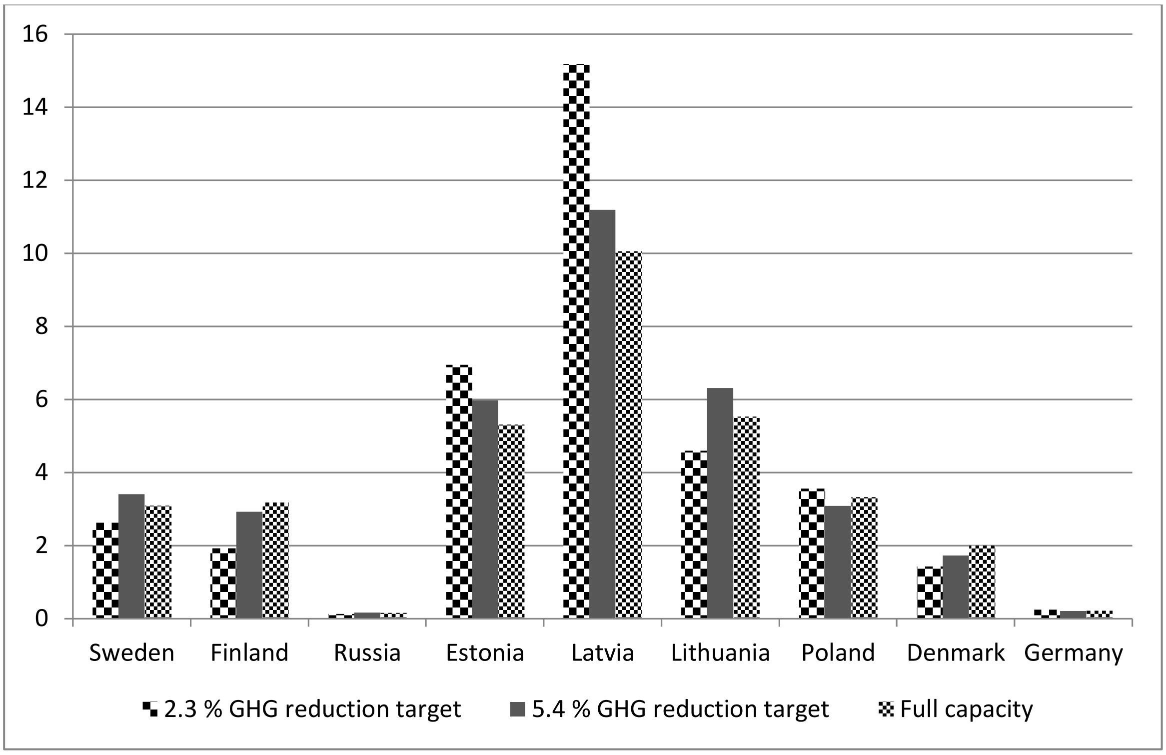
The results show that, under both joint and climate focused scenarios, Estonia, Latvia, and Lithuania are the top three countries in the region where significant share of reduction in GHG can potentially be delivered in an economically efficient way. It is interesting to note that, under joint implementation, for Lithuania the highest ratio is achieved at the starting point (i.e. with equivalent to 2.3 % of the GHG reduction as spill-over effect of meeting the BSAP target), the opposite trend applies for Estonia and Latvia (Figure 6a). This may indicate that achieving synergies between nitrogen abatement and climate change mitigation through the implementation of land-based measures is relatively cheaper in Lithuania than in Estonia and Latvia. On the contrary, when the focus is on climate change mitigation only, the realization of the strategy appears to be cheaper in Estonia and Latvia than in Lithuania (Figure 6b).

Under both scenarios, of all the countries, Russia and Germany exert the lowest ratio (Figure 6a and b). This is because the shares of the GHG emissions from these countries far outweigh the potential for emission reduction through the selected measures. These two countries' actual emissions respectively account for 50% and 24% of the total emission in the region. Meanwhile, Sweden, Finland, and Poland are more or less within the same range suggesting some potential for realizing efficient climate change mitigation in their own territory. For Denmark, there seems to be some potential for economically efficient climate change mitigation although slightly lower than that of Sweden, Finland and Poland. All in all, the findings highlight the importance of viewing the distribution of cost effective potential abatement effects across the region in relation to the actual emission of the different countries.

Figure 6. Ratio of economic efficient share to polluter pay share for GHG reduction; a) under joint implementation and b) climate focused strategy



(a)



(b)

4. Discussion

Both HELCOM (2013) and former cost-effectiveness studies on the implementation of nutrient reduction policies to the Baltic sea (e.g. Gren et al 2013; Elofsson 2014, Hyytianen et at 2014; Ahlvik et al 2014 and Hasler et al 2014) conclude that fulfilling these reduction policies is costly for the countries around the Baltic Sea. Cost savings will therefore be highly welcome, especially for those countries that have the highest reduction requirements. All EU member states around the Baltic have signed up to agreements to reduce both the emission of greenhouse gases to the air and the flow of nutrients to the aquatic environment, but there are only few studies addressing the costs and effects of these policies together (e.g. Gren & Säll 2015.). The present study seeks to fill this gap by analysing and measuring spillover effects from individual implementation of the policies on the other policy's targets, as well as synergies and potential conflicts when the policies are implemented simultaneously. The study also offers insights regarding the differences between countries in terms of the potential for and the environmental and economic consequences of a given policy implementation. These insights may be important when considering the opportunity for regional cooperation in environmental policy making in the Baltic region.

4.1. From co-benefit to joint strategy

The present research seeks to fill the described lack of studies on the cost-effectiveness of joint strategies between water policies and other policies, as highlighted by Balana et al. (2011), by exploring the potential for jointly managing nutrient abatement in water quality policies and climate change mitigation in the Baltic region through the implementation of selected measures in agriculture and waste water treatment.

The point of departure of our analysis is the fulfilment of the HELCOM's revised Baltic Sea Action Plan (BSAP) 2013 targets for N and P load reductions. Our present analysis shows that the implementation of the included measures to meet the BSAP nutrient load reduction targets induces positive spill-over or co-benefit on climate change mitigation through a reduction of GHG emission of 7.1 million tons CO2eq (i.e. equivalent to 2.3% of the emission from agriculture and waste water sectors). Additional 2% climate change mitigation can be delivered without pronounced increase in the total cost of implementation. This suggests that the integration of climate change mitigation target up to this level is achievable through reshuffling of the intensity of implementation of the different measures across the countries in the region. Furthermore, the results of our analysis show that significant increase in joint implementation cost occur when the GHG reduction target is above 5.4%.

The marginal cost of the additional climate change mitigation as a result of integrating climate change mitigation targets into the nutrient abatement implementation ranges from 0.4 Euros per ton of GHG reduction at baseline level to 181 Euros at 5.4% GHG reduction target. The marginal cost of implementing the selected measures for this combined nutrient abatement and GHG reduction targets is substantially lower than that of the case where the implementation of the measures is exclusively targeted towards climate change mitigation. However, our finding shows that the marginal cost of the joint implementation becomes higher when the GHG reduction target is above 6%.

It is also interesting to compare the cost implication of generating additional climate change mitigation from the joint implementation with marginal abatement costs of greenhouse gas emission reported in the literature. Elzen et al. (2007) provided an estimate of marginal abatement cost (MAC) of 23 to 93 Euros per ton CO2 within the EU27 in order to meet the EU 2020 target. Kuik et al. (2009) estimated the MAC for the year 2025 contingent upon the stabilisation targets: 37 to 119 Euros per ton CO2 for a stabilisation target of 500 ppm and 69-241 Euros per ton CO2 for a stabilisation target of 400 ppm. We refer to these two studies because the reported MACs are based on 2005 prices hence readily comparable to our findings. Besides, the referred studies cover the EU therefore the results are relevant for comparison to the regional scope of the present paper. The marginal cost of achieving additional climate change mitigation effect equivalent up to 5.4 % GHG reduction through joint implementation in our model is still within the range of the marginal abatement costs of GHG emissions reported by Elzen et al. (2007) and Kuik et al. (2009). This highlights the economic rationale for considering simultaneously multiple environmental objectives notably nutrient abatement and climate change mitigation.

As the abatement strategy builds upon a selection of measures, another important element of the analysis is to investigate the intensity of utilization of the different measures and how this develops as climate change mitigation target increases. Interestingly, some degree of similarity in the emerging patterns can be observed for both joint policy and climate focus strategy scenarios. It is evident that N fertilizer reduction is the most preferred measure when the objective is to reduce GHG emission followed by wetland restoration as the second preferred measure in the list. In the case of livestock measure, cattle reduction appears to be more competitive than pig reduction does. Sewage treatment turns out to be the least preferred measure.

Overall the findings from the present research demonstrate a stronger justification for the implementation of the selected measures in order to meet the BSAP nutrient abatement targets because doing so can deliver climate change mitigation co-benefit in terms of GHG emission reduction. This climate change mitigation co-benefit amounts to 2.3% of the 2005 GHG emission level from agriculture and waste water in the Baltic region. Our present research also shows that further climate change mitigation benefit can be obtained through joint implementation in a cost-effective manner up to 5.4% of the 2005 GHG emission level. The climate change mitigation effect (co-benefit and beyond) from the joint strategy is attributable largely to the synergistic effect of the selected land-based measures in delivering N load reduction and GHG reduction but not the case with P load reduction. This is also confirmed by the findings that a policy solely focused on climate regulation has the potential to deliver both significant reduction of GHG emission and N abatement but not able to meet the BSAP target for P abatement. As such, in environmental terms, it makes sense to develop policies that simultaneously addresses water regulation (both N and P load reductions) and climate change mitigation. It is clear however that the synergies between P and GHG are less than between N and GHG-effects, because N_2O and NH_4 reductions are included as CO_2 eq. in the GHG-reductions. Insights regarding cost for implementation, as previously discussed, also appear to support the case for joint policy development at least up to a certain level of GHG reduction target. We argue that by explicitly addressing such interlinkage between different objectives one can generate a more thorough picture regarding the economically efficient solution compared to a single policy objective perspective. Based on the results from our analysis, future policy development in an attempt to address nutrient load reduction to the Baltic Sea should seriously consider the potential synergies and trade-offs with other environmental goals. To this end, our paper specifically demonstrates the scope for cost effectively managing water and climate objectives through a small selection of measures. Our research findings show that up to a certain extent the additional benefits of implementing measures to simultaneously address nutrient abatement and GHG reduction are sufficient to justify the additional costs to be incurred.

4.2. Asymmetrical distribution of effects and cost across countries

Our analysis reveals that to fulfill a cost-minimisation solution the biophysical effects and the cost implications of implementing a strategy to jointly tackle nutrient abatement and climate change mitigation using land-based measures vary between countries in the Baltic region. This reflects that the asymmetry between the individual countries' intensity of agricultural production and state of waste water treatment technology and the economic efficiency of abatement affects the cost-effective distribution of effort, and the countries share of the effort. The actual emission levels also differ by countries, and in the year 2005, of the total GHG emission of around 304.6 million tons in the Baltic region, the largest shares of the emission were attributed to Russia (50.9%), Germany (24.2%) and Poland (13.9%). The results of the cost-minimisation indicate that, in comparison to other countries, it is cost-effective for Latvia, Lithuania, and Estonia to deliver shares of the GHG abatement outcome that exceed their share of actual emission. The results also indicate that the cost of realizing the joint abatement strategy in these three countries seem to be substantially lower than in the other countries. In other words these countries potentially can have larger economic efficient share of GHG reduction effects relative to their polluter pay share and the "fairness" share. In contrast, in the case of Russia and Germany, the potential effects of cost efficient abatement strategy, relative to their reported emissions, are much lower. These findings highlight the potential for economically efficient allocation and a scope for developing joint water-climate strategy through regional cooperation.

As presented in the introduction De Cara & Jayet (2011) have pointed at the need for revising the EU ESD target allocation, or utilising the inherent possibility, to make this regulation more flexible, and Amman et al (2011) also conclude that the cost-effectiveness of additional emission reduction measures is sensitive to whether equal targets across countries are employed or whether trade is allowed. When comparing the cost-effective allocation of GHG reductions from the present study to the emission reduction targets in the ESD, it is striking that the ESD allocation is not cost-effective and that cost-savings could be achieved with a more flexible allocation. The comparison is not straight forward however, as a 10% GHG reduction is not feasible with the limited number of measures included in the model, but the comparison is indicative for that De Cara & Jayet (2011) conclusions can be supported by the spatial analysis performed here.

The insights from the present study provide a basis for justifying a strategy for regional cooperation for example a mechanism that involves cap-and trade or other transfers of resources between countries to pave the way for implementation of a cost-effective, win-win strategy in terms of simultaneously tackling nutrient load reduction and climate change mitigation for the benefit of the region as a whole. The important role of transnational cooperation for water management has been highlighted in the literature (e.g. De Cara & Jayet 2011; Huisman et al., 2000; Phillips et al., 2006; van Rijswick et al., 2010). For the Baltic Sea region, the incentive for a regional cooperation involving the implementation of land based measures is even greater with the additional climate change mitigation benefit in terms of GHG emission reduction as demonstrated in this paper.

Shortcomings from the present study are that only a few measures are included in the model at this stage. Following MacLeod et al. (2015) the measures included in the analysis influence the results, and with additional measures with effects on either all pollutants or parts could alter the results if the capacity, effects or costs of implementing such measures are different between the countries. Further research in including measures and estimating reliable costs, effects and capacities of these abatement solutions, is crucial to enable more comprehensive advice on how costs can be reduced.

5. Conclusions

The present research focuses specifically on land-based agricultural measures and waste water treatment for abatement of emissions of nutrients and GHGs. The present study makes an important contribution to the scant literature on understanding the environmental and economic synergies and trade-offs between different policies objectives from a regional perspective. The overall conclusion of the present study is that there is scope for implementing measures that jointly deliver policy objectives concerning nutrient load reduction as well as GHG emission reduction in a cost effective way. The realization of the strategy will inevitably have differing distribution of consequences across the countries in the Baltic Sea region from both environmental and economic perspectives. This has important policy implications for the potential to save costs by developing a joint, cost-effective regional strategy for water and climate regulation that fully consider the asymmetry in the expected effects and cost distribution across the countries in the region.

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APPENDIX

A. Characteristics of the abatement measures included in the BALTCOST model

This section provides short overviews of how the costs, the effectiveness and capacity of each of the abatement measures are modelled in BALTCOST. A more detailed documentation of the nutrient load reduction effects and costs for each measure is provided in Hasler et al (2014). The approach for estimating the GHG emission reduction effects for each measure are described in detail in Appendix B. The approach follows the IPPC guidelines (IPCC 2006; 2013).

Waste water treatment

BALTCOST regards improving wastewater treatment (WWT) as the connection of an additional 'person equivalent' (PE) pollution load to an improved level of WWT. The costs is estimated as a cost-function for tertiary treatment, and specific cost-functions are estimated and localised according to PE's treated for each of the 117 catchments, using costs of operation and maintenance from Danish WWT-plants (see Hasler et al 2014). Subcatchment-specific upgrading capacities, derived from 10 x 10km grid cell resolution data on population sizes in combination with national resolution data on the percentage of population upgradeable to tertiary-level WWT, sum to the 117 subcatchment-specific capacity constraints for WWT upgrading. Capacity constraints for the 117-subcatchments are aggregated to the 22 drainage basins. Following Berbeka et al.(2012), standard percentage reductions in country-specific N and P at-source loads per PE are assumed for the populations upgraded to tertiary WWT. The GHG emission reduction effect from the implementation of WWT is defined as the difference in N_2O emission level per person between two situations: with and without treatment. The estimation of the effect refers to the default emission values found in the IPCC guideline for 2006.

Livestock reductions

BALTCOST includes reductions in cattle and pig numbers. The costs of reducing pig and cattle numbers, respectively, reflects the loss in profit from reducing livestock numbers with an "average cow" and "average pig" using the data at 10x10km grid level, calculated using drainage basin-specific standard outputs (in EUR) for each livestock class of each species. The maximum capacities for the livestock reduction measures is set at 20 percent of the current herd sizes in each drainage basin, as it is assumed that further reductions in livestock numbers would be likely to incur additional costs arising from unused animal housing and production facilities. The costs and effects on N and P load reductions are estimated assuming that farmers substitute reductions in available manure input with increased mineral fertiliser applications, and this assumption of course reduce the effect of the measure but ensure independence between this measure and the reduced fertilisation measure. Effects on both N and P are based on assessments in the BONUS RECOCA project (Andersen et al 2016), assuming that the utilisation efficiencies of N and P from manure is lower than the N and P utilisation efficiencies from mineral fertilisers, in accordance with results from Webb et at (2010), an assessment made for the European Commission. In the RECOCA project the P loss reduction from livestock reductions were not modelled, but we assume a reduction of P. This is based on an understanding that all P in excess of harvested P (i.e. a positive P field balance) will accumulate in the soil and potentially lead to P losses. Substitution of manure by inorganic fertilizer will lead to reduced P inputs because when manure is substituted by inorganic fertilizer the farmer will most likely apply fertilizer according to the N requirement and then there will automatically be a much lower P input.

As the P effects are much more uncertain than the effects on N, because they vary more between locations than the N-effects, the P effects can be overestimated, because a homogenous effect across different locations cannot be assumed. Nonetheless, effects of livestock reductions on P can be anticipated. The effects of livestock reduction on GHG emission reduction is quantified by including CH_4 emission from both enteric fermentation and manure management as well as N_2O emission from manure management (IPPC 2013). These two sources are the main sources of CH_4 emissions from agriculture (Eurostat, 2012).

Fertiliser reductions

BALTCOST includes reductions in nitrogen fertiliser applications as a measure to reduce nitrogen loads into the Baltic Sea. This measure includes reductions in applications of both artificial fertilisers and animal manure. The cost-functions are estimated using crop production- and fertiliser application data at the 10x10 km resolution. The method is explained in detail in Hasler et al (2014), but some details are repeated here since the approach is important for the inputs to the GHG effect measurement. Since yield functions or data to estimate them were not available for the whole Baltic Sea region Hasler et al (2014) assumed yield loss functions equal to Danish conditions, i.e. we are using data from Danish experiments for the yield functions (Pedersen 2009). The same approach is used here. Opportunity costs are calculated as the difference between the profit arising at reduced levels of fertiliser application and the profit arising at the initial, profit maximising, N fertiliser application level. The capacity of this measure is constrained to 20 percent of the initial N application level for the particular crop and drainage basin concerned, as this constraint reflects caution in extrapolating quadratic-form yield functions outside the range of their parameterisation data. The effects for N leakage is estimated using a crop- and location-specific leaching function (farm type, soil type specific) which links the amount of fertiliser applied to a crop to the nitrogen load that leaches from the root zone below that crop (Andersen et al., 2016). The calculation of the GHG emission reduction effect takes into account direct emissions from synthetic fertilizers and manure application as well as indirect emissions (volatized N and N leaching from both synthetic fertilizers and manure application). NO_2 emissions mainly consist of emissions from the management of the manure, as well as emissions from crop residues in the soil, from cultivation of histosols as well as from nitrogen deposition and hydrological processes in the soil-water column (IPCC, 2013).

Catch crops

Catch crops are grown to retain N in the period between two main crops, and we assume that catch crops are rye grass undersown in spring barley and oats, assuming that the undersown catch crop does not reduce the yield of the main crop (Jensen et al. 2009). The capacity of the measure is set equal to the drainage-basin specific area currently cultivated with spring barley and oats, using the data at the 10 x 10km resolution. The costs of seed purchase for the catch crop is taken to represent the cost incurred in implementing this N abatement measure, since no additional sowing costs are incurred when the catch crop is sown together with the main crop. The effects for the N leakage are due to a better utilisation of nitrogen, and the N leakage is reduced by 35%. The values of GHG emission effect from catch crops in the reported model follow Olesen et al. (2013). The implementation of catch crops is expected to increase emission. The calculation of the effect takes into account the areal distribution of sandy and clay soils because the emission factors are dependent on the soil characteristics (i.e. emission factors are higher for sandy soils).

Restored wetlands

The wetland restoration N and P abatement measure in BALTCOST is defined as the restoration of wetlands in agricultural areas on organic soils that have been wet in the past, by de-commissioning drainage or by implementing other hydrological changes. Drainage basin-specific cost functions for wetland restoration reflect the opportunity cost of the arable land area lost. This opportunity cost is based on the Eurostat standard output of crops grown in each of the drainage basins in an equivalent approach to that used for estimating the opportunity costs of livestock reductions. The capacity were identified from GIS data based on the distribution of histosoils. The effects on N and P leakage is assumed uniform: 150 kg of N load and 0.7 kg of P load to surface waters, based on Hoffmann et al. (2006). The average N and P retentions in surface waters for the wetlands restoration measure within each drainage basin were calculated knowing the distribution of wetland restoration potential across constituent sub-catchments within each drainage basin. Wetland restoration can have varied effects on GHG emissions in different catchments depending on the climatic zones and the soil types, including the nutrient status of the soils where the restoration takes place (IPCC, 2013). For example, while wetland restoration on agricultural soils with rich nutrient level in boreal region is beneficial in terms of CO_2 emission reduction, the opposite effect is expected for the temperate region.

The GHG emission effect of restoring wetland takes into account on-site CO_2-C emissions/removals from the soil and non-tree vegetation, off-site CO_2-C emissions from dissolved organic carbon exported from rewetted organic soils, and CH_4-C emissions from rewetted organic soils. The estimation also factors in the areal distribution of climatic zones (boreal versus temperate) and nutrient status of agricultural areas in each of the drainage basins within each of the Baltic countries. The method used for estimating the effect is consistent with the IPCC's 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands - Methodological Guidance on Lands with Wet and Drained Soils, and Constructed Wetlands for Wastewater Treatment - (Wetlands Supplement).

B. Quantifying the GHG emission reduction effects of the selected measures

We estimated the potential contribution of implementing the selected measures towards GHG emission reduction in this paper by developing a framework that primarily builds upon the methodology of the IPCC Guidelines for National Greenhouse Gas Inventories. To operationalize the framework, we consulted national emission data from the IPCC Common Reporting Framework for all the countries in the Baltic region in order to obtain GHG emission factors that are specific for a given country and for a particular measure. To downscale the estimates of GHG effects of the different measures from the country level to drainage basins, we utilized 10 x 10 km datasets containing information regarding the agricultural land use and biophysical characteristics of the drainage basins across the Baltic region. Below we describe in detail the framework used for quantifying the GHG reduction effects of implementing the different measures.

Measure #1 Reduction in fertilizer applications to arable crops (N abatement)

The estimation of the GHG emission reduction as a result of reducing the amount of fertilizers applied to arable crop cultivation factors in both direct and indirect emissions. Indirect emissions include volatized and leached N from both synthetic fertilizers and animal manure applications.

Total emission reduction (in CO_2eq.) = $(EF^{direct\text{-}db} + EF^{indirect\text{-}db}) \times 44/28 \times GWP_{(N2O)}$

Direct emission:

$$EF^{direct\text{-}db} = (frtprop^{db} \times EF^{fertc}) + (manprop^{db} \times EF^{amred\text{-}c})$$

$$EF^{amred\text{-}c} = (EF^{amh\text{-}c} \times (N^{amh\text{-}c} / (N^{amh\text{-}c} + N^{amp\text{-}c}))) + (EF^{amp\text{-}c} (N^{amp\text{-}c} / (N^{amh\text{-}c} + N^{amp\text{-}c})))$$

Indirect emission:

$$EF^{indirect\text{-}db} = EF^{vol\text{-}db} + EF^{lc\text{-}db}$$

$$EF^{vol\text{-}db} = (frtprop^{db} \times N^{fertvol\text{-}c} \times EF^{fertvol\text{-}c}) + (manprop^{db} \times N^{amvol\text{-}c} \times EF^{amvol\text{-}c})$$

$EF^{lc\text{-}db}$ = ($frtprop^{db}$ x $N^{fertlr\text{-}c}$ x $EF^{fertlr\text{-}c}$) + ($manprop^{db}$ x $N^{amlr\text{-}c}$ x $EF^{amlr\text{-}c}$)

Where:

$EF^{direct\text{-}db}$ = Emission factor from direct emission for drainage basin db
$EF^{indirect\text{-}db}$ = Emission factor from indirect emission for drainage basin db
$frtprop^{db}$ = Proportion of synthetic fertilizer use to meet N requirement in drainage basin db
$manprop^{db}$ = Proportion of manure use to meet N requirement in drainage basin db
$EF^{fert\text{-}c}$ = Emission factor for synthetic fertilizer application in country c (kg N_2O-N/kg N)
$EF^{amred\text{-}c}$ = Emission factor for animal manure application in country c (kg N_2O-N/kg N)
$EF^{amh\text{-}c}$ = Emission factor for manure application from animal house in country c (kg N_2O-N/kg N)
$EF^{amp\text{-}c}$ = Emission factor for manure application from animal excretion on pasture, range, and paddock in country c (kg N_2O-N/kg N)
$N^{amh\text{-}c}$ = N input from manure application from animal house in country c (kg N /year)
$N^{amp\text{-}c}$ = N input from manure application from animal excretion on pasture, range, and paddock in country c (kg N /year)
$EF^{vol\text{-}db}$ = Emission factor for volatized N for drainage basin db
$EF^{lc\text{-}db}$ = Emission factor for leached N for drainage basin db
$N^{fertvol\text{-}c}$ = Fraction of N from synthetic fertilizers that volatize as NH_3 and NOx in country c
$EF^{fertvol\text{-}c}$ = Emission factor for volatized N from synthetic fertilizer application in country c (kg N_2O-N/kg N)
$N^{amvol\text{-}c}$ = Fraction of N from animal manure that volatize as NH_3 and NOx in country c
$EF^{amvol\text{-}c}$ = Emission factor for volatized N from synthetic fertilizer application in country c (kg N_2O-N/kg N)
$N^{fertlr\text{-}c}$ = Fraction of N from synthetic fertilizers that is lost through leaching and run off from fertilizers reduction application i country c
$EF^{fertlr\text{-}c}$ = Emission factor for N lost through leaching and run off from synthetic fertilizer application in country c (kg N_2O-N/kg N)
$N^{amlr\text{-}c}$ = Fraction of N from synthetic fertilizers that is lost through leaching and run off from animal manure application reduction in country c
$EF^{amlr\text{-}c}$ = Emission factor for N lost through leaching and run off from animal manure application in country c (kg N_2O-N/kg N)

Measure #2 Catch crops under spring-sown cereals (N abatement)

The implementation of this measure will result in an increase of GHG emission. According to Olesen et al. (2013), the emission level is higher for cultivation on sandy soils (113 kg CO_2 eq. per ha) than on clay soils (5 kg CO_2 eq. per ha). The estimation of the GHG emission effect of implementing this measure therefore takes into account the areal distribution of sandy and clay soils of agricultural area in each of the 22 drainage basins in the Baltic sea regions as follows:

Emission = (Areal proportion sandy soil x 113) + (Areal proportion of clay soil x 5)

Measure #3 Reductions in livestock number (N & P abatement)

The GHG emission reduction from reducing pig number takes into account CH_4 emissions from enteric fermentation and manure management as well as N_2O emission from manure management.

Total emission reduction (in CO_2eq.) = $((EF^{pigredferc} + EF^{pigredmnc}) \times GWP_{(CH4)}) + (N^{excpig} \times EF^{pigc} \times 44/28 \times GWP_{(N2O)}) + (EF^{fertincpig} \times 44/28 \times GWP_{(N2O)})$

$$EF^{pigc} = \sum_{k=1}^{m} \frac{N_k^{texcpig}}{\sum_{k=1}^{m} N_k^{texcpig}} EF_k^{pigc}$$

$EF^{fertincpig} = N^{wexcpig} \times Prop^{pigdb} \times N^{subt} \times (EF^{fertc} + (N^{fertvolc} \times EF^{fertvolc}) + (N^{fertlr} \times EF^{fertlrc}))$

Where:
$EF^{pigredferc}$ = Emission factor for CH_4 emission from enteric fermentation for pig in country c
$EF^{pigredmnc}$ = Emission factor for CH_4 emission from manure management for pig in country c
$GWP_{(CH4)}$ = Global warming potential of CH_4 (in CO_2 eq.)
$GWP_{(NO2)}$ = Global warming potential of NO_2 (in CO_2 eq.)
N^{excpig} = Average nitrogen excretion from pig in country c (kgN/head/yr)
EF^{pigc} = Weighted implied emission factor for pig in country c (N_2O-N/kgN)
$N_k^{texcpig}$ = Total N excretion from pig per animal waste management system k in country c (kgN/ yr)
EF_k^{pigc} = Implied emission factor for pig per animal waste management system k in country c(kg N_2O-N/kgN)
k = animal waste management systems (anaerobic lagoon, liquid system, daily spread, solid storage and dry lot, pasture range and paddock, other)
$EF^{fertincpig}$ = Emission factor for substitution to synthetic fertilizer
$N^{wexcpig}$ = Weighted N excretion per pig per year (kg N manure/year/animal)
$Prop^{pig}$ = Total number of pig divided by total number of livestock in drainage basin db
N^{subt} = N substitution effect (kg N synthetic/kg N manure)
EF^{fertc} = Emission factor for synthetic fertilizer application in country c (kg N_2O-N/kg N)
$N^{fertvolc}$ = Fraction of N from synthetic fertilizers that volatize as NH_3 and NOx in country c
$EF^{fertvolc}$ = Emission factor for volatized N from synthetic fertilizer application in country c (kg N_2O-N/kg N)
$N^{fertlrc}$ = Fraction of N from synthetic fertilizers that is lost through leaching and run off from fertilizers reduction application
$EF^{fertlrc}$ = Emission factor for N lost through leaching and run off from synthetic fertilizer application in country c (kg N_2O-N/kg N)

Measure #4 Reductions in cattle number (N & P abatement)

The GHG emission reduction from reducing cattle number takes into account CH_4 emissions from enteric fermentation and manure management as well as N_2O emission from manure management.

Total avoided emission = (($EF^{catredferc}$ + $EF^{catredmnc}$) x $GWP_{(CH4)}$) + (N^{exccat} x EF^{catc} x 44/28 x $GWP_{(N2O)}$) + ($EF^{fertinccat}$ x 44/28 x $GWP_{(N2O))}$

$$EF^{catc} = \sum_{k=1}^{m} \frac{N_k^{texccat}}{\sum_{k=1}^{m} N_k^{texccat}} EF_k^{catc}$$

$EF^{fertinccat}$ = $N^{wexccat}$ x $Prop^{cat}$ x N^{subt} x (EF^{fertc} + ($N^{fertvolc}$ x $EF^{fertvolc}$) + (N^{fertlr} x $EF^{fertlrc}$))

Where:
$EF^{catredferc}$ = Emission factor for CH_4 emission from enteric fermentation for cattle in country c
$EF^{catredmnc}$ = Emission factor for CH_4 emission from manure management for cattle in country c
$GWP_{(CH4)}$ = Global warming potential of CH4 (in CO_2 eq.)
N^{excpig} = Average nitrogen excretion from cattle in country c (kgN/head/yr)
EF^{pigc}= Weighted implied emission factor for cattle in country c (N_2O-N/kgN)
$N_k^{texcpig}$= Total N excretion from cattle per animal waste management system k in country c (kgN/yr)
EF_k^{pigc} = Implied emission factor for cattle per animal waste management system k in country c(kg N_2O-N/kgN)
k = animal waste management systems (anaerobic lagoon, liquid system, daily spread, solid storage and dry lot, pasture range and paddock, other)
$EF^{fertinccat}$ = Emission factor for substitution to synthetic fertilizer
$N^{wexccat}$ = Weighted N excretion per cat per year (kg N manure/year/animal)
$Prop^{cat}$ = Total number of cattle divided by total number of livestock in drainage basin db
N^{subt} = N substitution effect (kg N synthetic/kg N manure)
EF^{fertc} = Emission factor for synthetic fertilizer application in country c (kg N_2O-N/kg N)
$N^{fertvolc}$ = Fraction of N from synthetic fertilizers that volatize as NH_3 and NOx in country c
$EF^{fertvolc}$ = Emission factor for volatized N from synthetic fertilizer application in country c (kg N_2O-N/kg N)
$N^{fertlrc}$ = Fraction of N from synthetic fertilizers that is lost through leaching and run off from fertilizers reduction application
$EF^{fertlrc}$ = Emission factor for N lost through leaching and run off from synthetic fertilizer application in country c (kg N_2O-N/kg N)

Measure #5 Wetland restoration on agricultural soils (N & P abatement)

The GHG emission reduction effect from restoring wetland on agricultural soils is estimated following the IPCC 2013 supplement to the guideline for national GHG inventory for wetlands (IPCC, 2013). Assuming there is no burning involved, the total effect includes on-site CO_2-C emissions/removals from the soil and non-tree vegetation, off-site CO_2-C emissions from dissolved organic carbon exported from rewetted organic soils, and CH_4 -C emissions from rewetted organic soils. The calculation of the effect takes into account the biophysical characteristics of the drainage basins (climatic zones and nutrient status). The formulas to calculate each of the emission components are described below:

Annual on-site CO_2-C emissions/removals from rewetted organic soils

CO_2-C composite = $\sum_{c,n}(A.EFco_2\ )$

Where:
CO_2-C composite = CO_2-C emissions/removals from the soil and non-tree vegetation, tonnes C yr-1
Ac,n = area of rewetted organic soils in climate zone c and nutrient status n, ha $EFCO_2$c,n = CO_2-C emission factor for rewetted organic soils in climate zone c, nutrient status n, tonnes C ha-1yr-1.
C = climate zone; Boreal and Temperate
N = nutrient status; poor and rich.

Annual off-site CO_2-C emissions due to dissolved organic carbon from rewetted organic soils

CO_2-C DOC = $\sum_{c,n}(A.EF_{DOC\_REWETTED}\quad)$

Where:
CO_2-C DOC = off-site CO_2-C emissions from dissolved organic carbon exported from rewetted organic soils, tonnes C yr-1
Ac = area of rewetted organic soils in climate zone c, ha
EF DOC_rewetted, c = CO_2-C emission factor from DOC exported from rewetted organic soils in climate zone c tonnes C ha-1yr-1

Annual CH_4-C emissions from rewetted organic soils

CH_4-Csoil= $\frac{\sum_{c,n}(\mathrm{A}.EFCH4\ soil)c,n}{1000}$

Where:
CH_4-Csoil = CH4 -C emissions from rewetted organic soils, tonnes C yr-1
Ac,n = area of rewetted organic soils in climate zone c and nutrient status n, ha
EFCH4 soil = emission factor from rewetted organic soils in climate zone c and nutrient status n, kg CH_4-C
ha-1 yr-1

Measure #6 Improving wastewater treatment (WWT) (N & P abatement)

The GHG emission effect is estimated as the reduction of N_2O emission to the air as result of wastewater treatment. This is equal to the difference between N2O emission per person without treatment and the emission with treatment. The formulas to calculate the emission levels under the different conditions are as follows:

$EN_2O_{(\text{no treatment per person})} = 4 \text{ kg N/person} \times EF_{(\text{no treatment})} \times 44/28 \times GWP_{(N2O)}$

$EN_2O_{(\text{treatment per person})} = 4 \text{ kg N/person} \times EF_{(\text{treatment})} \times 44/28 \times GWP_{(N2O)}$

The emission factors (EF) are based on the default values following the 2006 IPCC guideline.