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Increasing the cost-effectiveness of water quality improvements through pollution abatement target-setting at different spatial scales

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Abstract: In this paper, we investigate the potential gains in cost-effectiveness from changing the spatial scale at which nutrient reduction targets are set for the Baltic Sea, focusing on nutrient loadings associated with agriculture. Costs of achieving loadings reductions are compared across five levels of spatial scale, namely the entire Baltic Sea; the marine basin level; the country level; the watershed level; and the grid square level. A novel highly disaggregated model, which represents decreases in agricultural profits, changes in root zone N concentrations and transport to the Baltic Sea is proposed, and is then used to estimate the gains in cost-effectiveness from changing the spatial scale of nutrient reduction targets. The model includes 14 Baltic Sea marine basins, 14 countries, 117 watersheds and 19,023 10-by-10 km grid squares. A range of policy options are identified which approach the cost-effective reductions in N loadings identified by the constrained optimization model. We argue that our results have important implications for both domestic and international policy design for achieving water quality improvements where non-point pollution is a key stressor of water quality.

Keywords: cost-effectiveness, nutrient pollution, agricultural run-off, Baltic Sea, eutrophication

JEL codes: Q52, Q53, Q18, Q25, F53, R52

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1. Introduction

In this paper, we investigate the potential gains in cost-effectiveness from changing the spatial scale used for targeting of nutrient (N) reduction targets in the Baltic Sea, focusing on nutrient pollution associated with agriculture. Non-point nutrient inputs from agriculture constitute one of the main sources of total nutrient inputs to the Baltic Sea, and thus are one of the main determinants of eutrophication (HELCOM, 2011; Reusch et al., 2018). Intuitively, one would expect that a policy, which sets a target reduction at the largest area of spatial aggregation – in this case, the drainage basin for the entire Baltic Sea – would be more cost-effective than a policy, which imposes targets at lower levels of spatial disaggregation, since this would provide greater opportunity to take advantage of options in areas where loading reductions from agriculture into the Baltic can be achieved at a relatively low cost. Conversely, the highest cost per unit of N loading reduced – the least cost-effective policy – should be associated with a situation when targets are imposed at the lowest level of spatial aggregation (here, a 10-by-10 km grid square).

Complicating any assessment of the total cost of reducing nutrient loadings is the observation that the marginal abatement costs for reducing N loadings entering the Baltic Sea area would be expected to show large spatial variations, due to spatial variation in the three factors that underlie these abatement costs, namely (i) the value of N as fertilizer or as associated with livestock herds to farm profits – its marginal value product; (ii) the relationship between N fertilizer/ manure use and N concentrations in the root zone, which determines how reductions in N applications translate into reductions in potential pollution; and (iii) the extent and speed with which N in the root zone is transported to the Baltic as increased loading, translating potential pollution loads into actual impacts on ambient water quality.

This paper presents a novel, highly-disaggregated model which represents decreases in agricultural profits, changes in root zone N concentrations and transport to the Baltic Sea. The model is used to estimate the gains in cost-effectiveness from changing the spatial scale of N reduction targets. The model includes 14 Baltic Sea marine basins, 14 countries, 117 watersheds and 19,023 10-by-10 km grid squares. Costs of achieving loadings reductions are compared across five levels of spatial scale, namely the entire Baltic Sea; the marine basin level; the country level; the watershed level; and the grid square level. Cost-effectiveness indeed turns out to be highest when targets are set at the largest area of spatial aggregation, and lowest when targets are set at the smallest level. We argue that our results have important implications for the design of both domestic and international policy for achieving water quality improvements where non-point pollution is a key stressor of water quality. This point is re-enforced by a discussion of policy options which can get close to the idealized cost-effective reductions in N loadings as identified by the constrained optimization model.

2. Literature review

In this paper, we assume that the regulator (policy designer) wishes to achieve target reductions in water pollution at the lowest aggregate cost – that is, in a cost-effective manner. This does not imply that other criteria are unimportant in assessing policy options: merely that cost-effectiveness is a focal and relevant consideration in how pollution control options are assessed by society.

2.1 Previous work on cost-effectiveness in nutrient pollution control

A very early insight into how to improve the cost-effectiveness of pollution reduction was that allowing for varying levels of abatement across individual sources of pollution would result the achievement of a pollution reduction target at lower total abatement costs than a policy which required all sources to reduce pollution by the same amount (Baumol and Oates, 1971). This result arises from a situation where the marginal (incremental) costs of reducing pollution, known as marginal abatement costs, varied across sources of pollution. This finding underpins claims for the relative cost-effectiveness of "economic instruments" for pollution control, such as pollution taxes and tradeable permit markets, over regulatory alternatives (Hanley, Shogren and White, 2006). Studies on the relative cost-effectiveness of alternative policy options to control non-point source nutrient pollution from agriculture showed that this variation in marginal abatement costs across farms (due to differences, for example, in farm productivity) was key to explaining differences in the costs of achieving pollution loading reductions (Shortle and Horan, 2001). Another important factor driving this heterogeneity in abatement costs is variation across time and space in the physical processes linking nutrient application and manure deposition to a leachable pool of nutrients within the root zone; and variation in the processes linking nutrients in the root zone to loadings discharged into receiving water bodies and ambient water quality indicators (Aftab, Hanley and Baiocchi, 2010). However, finding policy instruments which can achieve target improvements in water quality at lowest aggregate cost is more complex for non-point pollution than for point source pollution, due to the un-observable nature of actual emissions from farmland (Hanley, Shogren and White, 2006).

A key insight from the work noted above is that one would expect that a policy which allows for greater scope in varying required emission reductions (or changes in management practices expected to reduce emissions) across sources leads to lower costs than one which restricts such flexibility. Since greater spatial scale is almost certainly correlated with greater variation in marginal abatement costs, the spatial scale at which controls are implemented should have implications for aggregate abatement costs. We use this insight to design the policy experiment reported in the methods section.

2.2 Previous studies on the Baltic Sea

The literature on cost-effective solutions for reducing nutrient pollution in the Baltic Sea has been comprehensively reviewed by Elofsson (2010). Therefore, we review previous studies only briefly, focusing on the choice of methodology used. The three most important factors that differentiate published studies are (i) the number of drainage basins included in the modeling, (ii) estimated nutrient reduction costs and (iii) leakage/retention functions included (which describe the linkages between nutrient application, the leachable pool of nutrients within the root zone, and the transport of those nutrients to the Baltic Sea). While the number of drainage basins is an indicator of the spatial resolution of the model, the types of the cost and leakage/retention functions and the methodology used to derive these functions' parameters define the models' accuracy. The characteristics of the models are compared in Table 1.

Table 1. Comparison of modelling approaches for estimating costs of nutrient loading reductions to the Baltic Sea

Study	Number of measures	Number of target regions	Number of target levels	Cost function	Leaching function, data
Gren, Jannke and Elofsson (1997)	5	14	8	nonlinear	linear, various sources
Ollikainen and Honkatukia (2001)	----	9	1	quadratic	----
Schou et al. (2006)	5	24	1	quadratic	linear, PLC-4 HELCOM data
COWI (2007)	9	24	1	linear	linear, based on distance
Gren (2008)	9	24	6	nonlinear	linear, PLC-4 HELCOM data
Hasler et al. (2014)	5	22	1	nonlinear	nonlinear, high resolution (117 watersheds) DAISY model
Ahlvik et al. (2014)	10	23	1	nonlinear	nonlinear coupled with marine model
Our approach	10 x 2[a]	19,023[b]	5 x 10[c]	nonlinear	non-linear, very high resolution (19,023 grid squares), DAISY model with grid-level combined groundwater and surface water retention, data from various sources

Notes: a) Our model includes mineral fertilizer and manure application levels to 10 crop types: common wheat and spelt, rye, barley, oats, potatoes, sugar beet, rape, turnip and other oil-seed or fiber plants, forage plants (temporary grass), pasture and meadow, and rough grazings; b) At its highest spatial resolution, our model allows for separate targets for each of the 19,023 10-by-10 km overland grid squares in the Baltic Sea drainage area. c) In this paper we present the results for 10 relative reduction levels for N reduction targets set at each of the 5 different spatial scales.

In the very first study of this type Gren, Jannke and Elofsson (1997) showed, using a model that included a wide spectrum of reduction measures, that the aggregate cost of nutrient reduction (i.e., reductions in both nitrogen and phosphorus) using uniform reductions across countries can be as much as four times higher than the cost-effective solution. The number of marine basins used for this study was 14. The nutrient reduction cost function was non-linear with parameters derived from econometric estimation of a fertilizer demand function. The leakage function, however, was linear and used, as described by the authors, a "rough" estimation of retention rates. The study showed that even using large drainage basins and relatively simple leakage functions, the gains from attaining cost-effective rather than uniform solutions to nutrient reductions were considerable.

Ollikainen and Honkatukia (2001) approached the problem more generally and estimated the aggregate cost of nutrient abatement, thus circumventing the problem of choosing control measures and calculating the leakage and cost functions. They compared a 50% load reduction for uniform and cost-effective allocations. The calculated minimum cost was much higher than that reported by Gren, Jannke and Elofsson (1997); however, the conclusion about the importance of cost-effective allocations of abatement was similar. A source-disaggregated approach was used by Schou et al. (2006). In this study cost functions were estimated for multiple measures including livestock density reductions (linear) and fertilizer usage reduction (quadratic) for 24 drainage basins with the same linear leakage function. In this case, results were presented for only one level of target nutrient reduction and the calculated cost was only slightly higher compared to the result of Gren, Söderqvist and Wulf (1997).

COWI (2007) calculated the cost of nutrient reduction for several measures and scenarios including investments in wastewater treatment plants and multiple agricultural measures. The cost function for fertilizer use reduction was linear and estimated separately for all nine Baltic Sea littoral countries. The leakage function was also linear but calculated for 24 different drainage basins. In another study conducted by Gren (2008) an increased number of measures were applied. Once again, the fertilizer reduction cost function was nonlinear, while the leakage function was linear. All functions' parameters were updated, with respect to their 1997 work, and calculated for 24 drainage basins. This study reports the lowest total abatement cost of all studies conducted to date.

All the research described above applies fairly similar approaches to the estimation of the costs and the effects of applied measures based on aggregated data. Two most recent studies used methods that are more novel. Hasler et al. (2014) applied a bottom up approach1 to calculate abatement cost functions for each watershed, which were then used in optimization. The cost functions were based, where possible, on microeconomic analysis at the farm level (e.g. nonlinear yield functions link the usage of fertilizer with the value of agricultural production). The optimization process itself was conducted using aggregated data for 22 drainage basins.

Ahlvik et al. (2014) used a combination of ecological and economic modelling to analyze the cost of nutrient abatement proposed in the Baltic Sea Action Plan (BSAP). Using an ecological marine model allowed for a more accurate assessment of the effects of nutrient

1 The leaching functions were estimated by applying the DAISY model (described in more detail in section 3.1 below) to high resolution girds.

reductions at the source on the ecological state of different regions of the Baltic Sea. The cost calculations were conducted for 23 drainage basins.

The above literature illustrates that the main advances to date in research on cost-effectiveness of nutrient abatement around the Baltic include increasing the accuracy of cost and leakage functions (e.g., by using more accurate, nonlinear functions), using more disaggregated models and data, and increasing the variety of abatement measures applied (e.g., including both agricultural and municipal sources of nutrient inputs). Needless to say, the authors of all the studies reviewed above emphasize the approximate nature of the models used, stress the need for caution in the usage of estimated costs for policy making, and call for further research devoted to improving models. The model described in this paper provides a further step in this direction.

Interestingly, the studies reviewed above show that uniform reduction targets applied at the national rather than at the Baltic Sea level reduce cost-effectiveness. In what follows, we show that this is the case for uniform (proportional) targets set at any spatial scale – basins, countries, watersheds or smaller regions. Applying a highly disaggregated bottom-up approach, based on modelling optimal measures for each 10-by-10 km grid square in the Baltic Sea drainage area, we show that the truly cost-effective policy should fulfill Baltic-wide reduction targets, while distributing the applications of measures in a more region-specific manner. We illustrate the scale of effectiveness improvements if relevant region-specific factors, such as retention, climate, soil, prices, and livestock intensity are taken into account in planning the optimal (cost-minimizing) pattern of abatement in non-point source nutrient loadings.

3. Methodology

In this section, we describe the modelling framework used for identifying cost-effective policies for reaching nutrient reduction targets. Our model is highly regionalized, in the sense that it uses a bottom-up approach – for any set of N reduction targets (specified at any spatial scale) it allows for identifying cost-effective implementation levels of each abatement measure to be implemented in each of the 19,023 10-by-10 km grid squares, into which the Baltic Sea drainage area is divided.2

The model takes three components into account and evaluates them simultaneously: the effectiveness of applying a particular measure in a grid square (in terms of reduced leaching); the retention coefficient for each grid square (i.e., the proportion of nutrients leached from each grid square that does not reach the Baltic Sea); and the cost of applying the measures in each grid square. Each of these components can be disaggregated into more detailed sub-components, each produces a non-linear relationship between the scale of application of the measure and its effect, and each uses grid square-specific parameters. Perhaps most importantly, the model takes grid square-level interactions between the measures into account – the effect that has been largely ignored in previous studies.

Approaching the problem of nutrient reduction using such a highly spatially-disaggregated approach has not been attempted before. It allows us to identify a cost-effective solution for targets specified at any spatial scale (sea basins, countries, watersheds, and down to the grid square level). Nutrient loadings, leakage and retention, as well as the cost of a proposed reduction, are calculated for each grid square individually and can later be aggregated, to calculate combined effects and costs for any desired sub-division of the Baltic Sea drainage area. In what follows, we describe the details of our modeling approach.3

3.1. Rootzone nitrogen leaching

The first building block of our model is a rootzone N leaching function, which is used to calculate the total leaching from agriculture in each grid square (N_leach) depending on the variable N inputs from two sources – mineral fertilizer (N_fert) and manure (N_man) .4 The function is non-linear and it utilizes multiple grid square- and crop-specific parameters, such as: acreage of agricultural land for each of the 10 crop types (ha), additional nitrogen inputs resulting from fixation (N_fix), seed content (N_seed) and atmospheric deposition

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2 While our model only considers reductions of N, we believe that similar conclusions apply to joint reductions of N and P. While modelling P reductions may not yet be feasible at this scale, considering the spatial scale for the cost-effective solution would also be evident for a study that considered, for example, the effects of urban/rural town sewage treatment plants.

3 We aimed to make this description self-contained and easy to follow. If additional information is required, we refer the reader to the RECOCA project reports (http://www.bonusportal.org/about_bonus/bonus_and_era-net/bonus_2009-2011/bonus_projects/recoca), which contain a more detailed description of the model.

4 Since each of these two variables is crop-specific, there are effectively 22 independent arguments. The reductions in mineral fertilizer and manure inputs for each crop (the potentials of each measure) are constrained by the current levels observed in the grid square.

(N_dep), and bio-geographic characteristics, such as the soil clay $(clay)$ and carbon $(carbon)$ content. In addition, some of the parameters can differ for three management intensity types (m). Using g and c subscripts to indicate the grid square- and crop-specific inputs, the rootzone N leaching function can be represented mathematically in the following way:

$$N\_leach_g\left(N\_fert_{c,g}, N\_man_{c,g}\right) = \sum_{c=1}^{10}\left(ha_{c,g}\left(b^0 \exp\left(b^1_{c,m} + b^2_{c,m} clay_g + b^3_c carbon_g + b^4_{c,m}\log\left(N\_input_{c,g}\right)\right)\right)\right),$$

where:

$$N\_input_{c,g} = N\_fert_{c,g} + N\_man_{c,g} + N\_fix_{c,g} + N\_seed_{c,g} + N\_dep_g.$$

(1)

The rootzone N leaching function was developed and parameterized using the state of the art soil-vegetation-atmosphere model "DAISY" (Abrahamsen and Hansen, 2000): DAISY is a 1 or 2 dimensional field-scale model, process-based, dynamic and tested in many applications. The N leaching rates found in the Baltic Sea drainage basin were identified using the following drivers: precipitation, temperature, soil types, farm types and levels of inputs of fertilizer and manure to crops. This allowed us to construct more than 11,000 combinations of these drivers to describe the variations found in the Baltic Sea drainage basin, and then apply the DAISY model to these combinations using a time series of climate data (1995 – 2006). From the DAISY simulations, the annual N losses from the rootzone for each combination of drivers were calculated. The N leaching function was developed from this data set by multiple regression analysis (see Andersen et al., 2016 for further details).

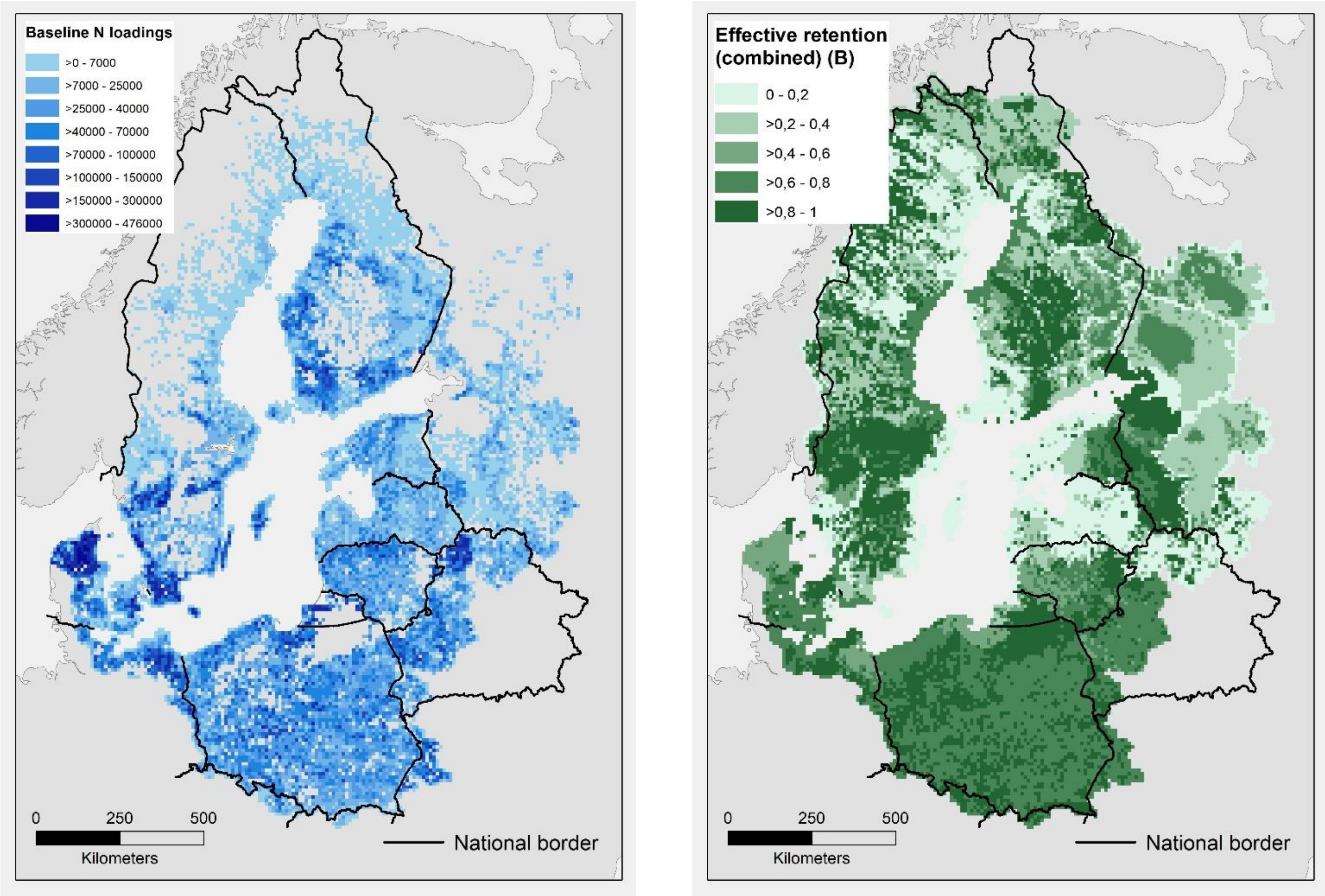
3.2. Retention

In order to estimate how much of the nitrogen that leaves the rootzone eventually reaches the Baltic Sea one needs to take account of nutrient retention in ground and surface waters. Retention is the permanent removal or temporary storage of nutrients within a system (von Schiller et al., 2008). Depending on the hydrological pathways along which nutrients are routed through the catchment, retention processes may significantly alter the concentration of these elements before they reach the marine recipient (Stalnacke et al., 2003). For nitrogen, denitrification, a microbial dissimilatory process in which dissolved nitrate is reduced to gaseous forms of nitrogen (Seitzinger, 1988), is often the most important process, and denitrification is strongly dependent on the hydraulic residence time in groundwater and surface waters.

Independent estimates of surface water nitrogen retention were provided by the MESAW model (Grimvall and Stålnacke, 1996; Stålnacke et al., 2015) for the Baltic Sea drainage basin subdivided into 117 watersheds. MESAW is a statistical model for source apportionment of riverine loads of pollutants and has been applied to the calculation on nitrogen retention in surface water bodies in the Baltic Sea river basins (Lidén et al., 1999; Vassiljev and Stålnacke, 2005; Vassiljev, Blinova and Ennet, 2008). The surface water nitrogen retention estimates were further disaggregated to each 10 km grid square by a procedure involving the flow distance from each cell to the sea and the total surface water area between the cell and the sea, thus accounting for the hydraulic residence time. Estimates for nitrogen retention in groundwater for each of the 117 watersheds were calculated by Andersen et al. (2016) as the difference between total watershed rootzone nitrogen leaching and the total riverine nitrogen loss from the watershed corrected for surface water retention.

The left panel of Figure 1 presents the distribution of baseline N loadings to the Baltic Sea. They are clearly unevenly distributed, with agriculture-heavy regions contributing the most. Another driver of the baseline N loadings is retention – in general, N pollution in areas that are closer to the sea causes more impact. The right panel of Figure 1 presents retention associated with each gird square. How much nitrogen reaches the Baltic Sea from each grid cell depends on the ground water retention for this grid cell and the surface water retention on its way to the sea. As a result, the N pollution in grid squares located further away from the Baltic Sea, with long slow-flowing rivers or lakes along the way, causes less impact than N pollution in areas close to the sea.

Figure 1. Baseline N loadings to the Baltic Sea (left) and effective (ground and surface water) retention (right)



3.3. *Costs of N reduction measures*

To find a cost-effective combination of abatement measures to deliver a given level of N reduction an appropriate method of cost assessment is needed. In our model we estimate the cost of applying an abatement measure on a particular scale by calculating the foregone profits resulting from its application. This approach is relatively straightforward, and results in substantially diversified cost coefficients between grid squares because country-specific price data and non-linear yield functions are applied, calibrated for each grid square.

The point of departure for estimating grid square-specific yield functions were the Danish experimental yield functions reported by Pedersen (2009). We adapted these functions to all the grid squares in the Baltic Sea drainage area by applying a calibration procedure similar to that described by Brady (2003). In essence, our approach was based on scaling the experimental yield functions horizontally, so that the observed levels of fertilization rates in each grid square were profit-maximizing, considering the local prices of crops and fertilizer. The resulting yield functions were grid square-specific, and took the following quadratic form:

$$Y_{c,g}\left(N\_frt_{c,g}, N\_man_{c,g}\right) = a_{c,g}^{0} + a_{c,g}^{1}\left(N\_frt_{c,g} + N\_man_{c,g}\right) + a_{c,g}^{2}\left(N\_frt_{c,g} + N\_man_{c,g}\right)^{2} \quad . \quad (2)$$

The cost of fertilizer or livestock reductions can then be calculated in the following way: for each grid square and for each crop we used a crop and grid square-specific yield function $(Y_{c,g})$ to calculate the reduction in yields resulting from reducing the amount of mineral fertilizer (N_fert_c) and the reduction in the amount of manure applied to this crop (N_man_c). Next, the resulting differences in yields per hectare were valued using average crop prices in the country $(p_{c,g})$. From this we subtracted the avoided cost of mineral fertilizer – the reduction in fertilization rate for each crop was multiplied by the average fertilizer price in the country $(p_{f,g})$.5 These "per ha of crop" reductions in revenues (from yield reductions) and the avoided costs (from fertilizer reductions) were then multiplied by the area of each crop in each grid square. This procedure assumes that fertilization levels can be exogenously constrained for each of the crop types, but farmers do not change their optimal cropping mix in response to a nutrient control policy. Finally, we subtracted the reduction in net profits (measured as standard gross margins6) resulting from the reduction of each of the three types of livestock included in the model – cattle, pigs and poultry (L_l).

Mathematically, the cost of reducing the amount of mineral fertilizer and/or livestock in each grid square can be calculated as the difference between baseline profits (Π_g^0) and the profits implied by the amount of mineral fertilizer (N_fert_c) and manure (N_man_c) used

5 The data on prices of crop outputs and fertilizer inputs were obtained from Eurostat. The areas of each crop in each country and drainage basin were provided by Andersen et al. (2016).

6 The Standard Gross Margin (SGM) is the average value of output minus certain specific costs of each agricultural product (crop or livestock) in a given region (http://ec.europa.eu/agriculture/rica/methodology1_en.cfm). Values of SGM for livestock have been calculated as a weighted average for each of the NUTS 2 regions based on FADN (SGM for all animals groups) and Eurostat data (structure of the heard). To avoid bias caused by fluctuations (in production, e.g. due to bad weather, or in input/output prices) we averaged these indicators for 3 successive years for each region.

in this grid square applied to each crop on its respective acreage, while taking savings in the cost of mineral fertilizer, and profits resulting from reduced livestock into account:

$$TC_g\left(N\_fert_{c,g}, N\_man_{c,g}\right) = \Pi_g^0 - \sum_{c=1}^{10}\left(ha_{c,g}\left(p_{c,g}Y_{c,g}\left(N\_fert_{c,g} + N\_man_{c,g}\right) - p_{f,g}N\_fert_{c,g}\right)\right) - \sum_{l=1}^{3}\left(sgm_{l,g}L_{l,g}^{0}\frac{\sum_{c=1}^{10}\left(N\_man_{c,g}\right)}{\sum_{c=1}^{10}\left(N\_man_{c,g}^{0}\right)}\right). \quad (3)$$

Regarding nutrient emissions related to livestock, we note that the amount of manure available in each grid square was determined by the number of each type of livestock held in that square. The rate at which manure can replace mineral fertilizer was assumed to be country-specific and a function livestock management intensity, manure storage capacity, timing of application of manure and application technology. It is anticipated that 70% of the N applied is utilized and substitutes for commercial fertilizer N in Denmark and Sweden, and 50% in the other riparian countries (Andersen et al., 2016). The difference in N utilization from manure is explained by variation in application methods, timing, manure storage and handling systems, as well as the type of livestock involved. We also assumed that available manure was distributed among the crop types in each grid square following a schedule favoring forage crops and with supplementary addition of mineral fertilizer. Finally, we assumed that any reductions in livestock were relative, i.e. they affected all cattle, pigs and poultry in each grid square in a proportional way.

3.4. The optimization problem

With these building blocks in place, we are now able to specify the optimization problem. Our model allows for designing a cost-effective policy for annual N load reduction targets specified at differing spatial scales – from the Baltic Sea as a whole, through individual sea basins, countries, watersheds, to each grid square separately. Using T_r to denote the N reduction target specified for each of the $r = 1...R$ "regions" (referring to the spatial scale at which the targets are specified) the optimization problem can be described as finding a cost-minimizing set of mineral fertilizer reductions $\left(\Delta N\_fert_{c,g} = N\_fert_{c,g}^{0} - N\_fert_{c,g}^{*}\right)$ and livestock reductions leading to the decrease in total manure $\left(\Delta N\_man_{c,g} = N\_man^0_{c,g} - N\_man^*_{c,g}\right)$ for each crop and grid square, such that the N reduction targets are attained (at the specified spatial scale) i.e.:

$$\min \sum_{r=1}^{R} \sum_{g_r=1}^{G_r} TC_{g_r}\left(\Delta N\_fert_{c,g_r}, \Delta N\_man_{c,g_r}\right) \quad \text{s.t.} \quad \begin{cases} \forall_{r=1..R} \quad \sum_{g_r=1}^{G_r} \left(1-\theta_{g_r}\right) N\_leach_{g_r}\left(N\_fert^*_{c,g_r}, N\_man^*_{c,g_r}\right) \geq T_r \\ \forall_{g=1..G} \quad \forall_{c=1..11} \quad 0 \leq N\_fert^*_{c,g} \leq N\_fert^0_{c,g} \\ \forall_{g=1..G} \quad \forall_{c=1..11} \quad 0 \leq N\_man^*_{c,g} \leq N\_man^0_{c,g} \end{cases}$$

(4)

We note that since $TC_g(\gamma)$ functions are convex, and $N\_leach_g(\gamma)$ functions are concave, the Lagrangian is concave in $\Delta N\_fert_{c,g}$ and $\Delta N\_fert_{c,g}$, and the Kuhn-Tucker necessary conditions define the optimal solution. The cost minimization problem can thus be solved using any statistical package suited for optimization problems of this size. In our case, the model was implemented in GAMS using CONOPT as a numerical optimization routine, and additionally in Matlab, which resulted in better convergence of the model.7

4. The results

In what follows we apply the model to a number of illustrative policy scenarios in order to compare the total cost of reaching N reduction targets specified at different spatial scales . We compare the scenarios with N reduction targets specified at the following spatial scales: (1) Baltic Sea drainage basin as a whole, (2) separate targets for each of the 14 sea basins, (3) separate targets for each of 14 countries in the Baltic drainage basin, (4) separate targets for each of 117 watersheds or (5) separate targets for each of the 19,023 grid squares. Every scenario was evaluated for decreases of between 5 and 50% (with 5 percentage point increments) of the maximum potential decrease from current N loads8 for each region

7 The data and software codes used for this study are available at http://czaj.org/research/supplementary-materials.

8 Our model predicts, that the total loads of N (from agriculture) to coastal waters are now at about 338 000 Mg per year, which is in agreement with the 285,000 – 370,000 Mg estimated by HELCOM (2011).

$\left(\sum_{g_r=1}^{G_r}(1-q_{g_r})\left(N\_fert^0_{c,g_r}+N\_man^0_{c,g_r}\right)\right)$.[9] For any such relative target it is possible to calculate the corresponding absolute N load reductions required.

The summary of the results is provided in Table 2.[10] The total baseline N loads to the Baltic Sea from the sources included in our model are approximately 340 Gg (thousands of metric tons) per year. Some of these loads (e.g., atmospheric deposition) cannot be reduced with the measures considered and hence the relative reduction targets are expressed with respect to the theoretically maximal reduction levels given the focus on non-point N run-off from farmland. For each reduction target the minimized total cost of abatement is calculated, given the constraints imposed by satisfying the target for the Baltic Sea as a whole (Overall), for each sea basin separately (Basin), for each country (Ctr), each watershed (Wts) and each grid square (Grid).[11]

Table 2. Total cost of reaching N reduction targets specified at different spatial scales

Reduction relative to maximum possible	0%	5%	10%	15%	20%	25%	30%	35%	40%	45%	50%
Absolute reduction [Gg]	0	14.11	28.22	42.33	56.44	70.55	84.66	98.77	112.88	126.99	141.10
N load [Gg]	337.68	323.57	309.46	295.35	281.24	267.13	253.02	238.91	224.80	210.69	196.58
Total cost [million EUR] for targets specified at:											
Baltic Sea level – *Overall*	0.0	0.18	1.03	3.31	7.35	13.30	21.59	32.67	47.11	65.58	89.05
sea basin level – *Basin*	0.0	0.25	1.26	3.71	8.23	15.18	25.02	38.23	55.40	77.33	105.02
country level – *Ctr*	0.0	0.29	1.37	4.05	9.00	16.58	27.13	41.03	58.89	81.45	110.18
watershed level – *Wts*	0.0	0.34	1.67	4.67	10.12	18.47	30.11	45.35	64.83	89.50	121.19
grid square level – *Grid*	0.0	0.45	2.25	6.14	12.85	23.41	39.23	61.24	90.62	126.23	169.34

The differences in the total cost of reaching different N reduction targets allocated at different spatial scales are illustrated in Figure 2. In all the scenarios the cost functions are increasing and strictly convex. Under all scenarios, total cost is lowest when the target is specified as an overall reduction for the Baltic Sea; as expected, satisfying regional constraints pushes the solution away from overall optimality. The costs are roughly 20% higher in the solutions with basin constraints and 30% higher with country constraints. Stronger

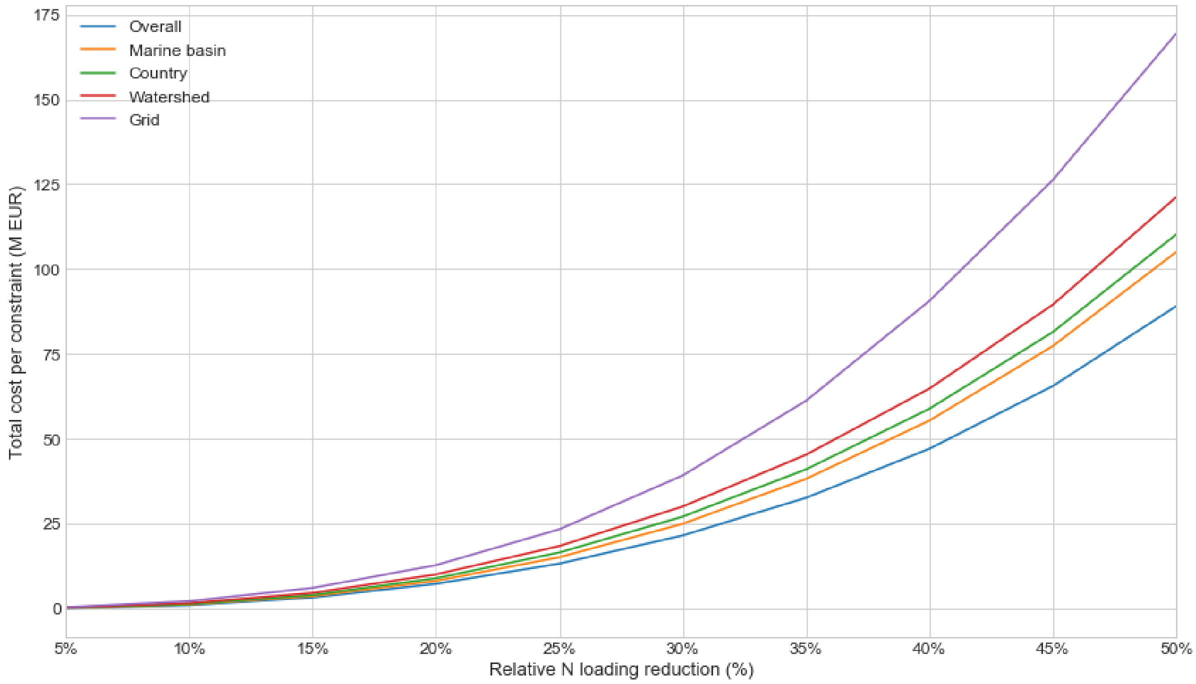
9 For example, a 20% reduction target for each country requires that every country reduces its current total N load to the Baltic Sea by 20% of what is theoretically possible (i.e., 20% of the reduction that would take place if there were no artificial fertilizers and livestock). The within-country reductions produced will be cost-efficient (i.e., achieved by implementing combinations of abatement actions at grid square resolution that deliver the N reduction target for that country at the lowest possible cost).

10 Detailed results are available in the online supplement to this paper.

11 Note that even when the N reduction targets are specified separately for each grid square, the model minimizes the cost of reaching the required reduction by decreasing mineral/manure fertilization of each of the 11 crops, relative to their baseline levels in each grid square.

constraints increase costs further – taking them about 45% higher for when the same relative reduction targets are set for each watershed separately and nearly twice as high when targets specified for separately for each grid square. Notice that the ranking of spatial targeting on cost grounds is stable across all of the reduction scenarios modelled. The ratio of the total cost for solutions with constraints imposed at different spatial scales provides vital information for policy makers, as it shows that hundreds of millions of euro can be saved through cooperation and high-resolution planning – at the level of the Baltic Sea as a whole.

Figure 2. The total cost of reaching the same relative N load reduction, with N reduction targets specified at different spatial scales.

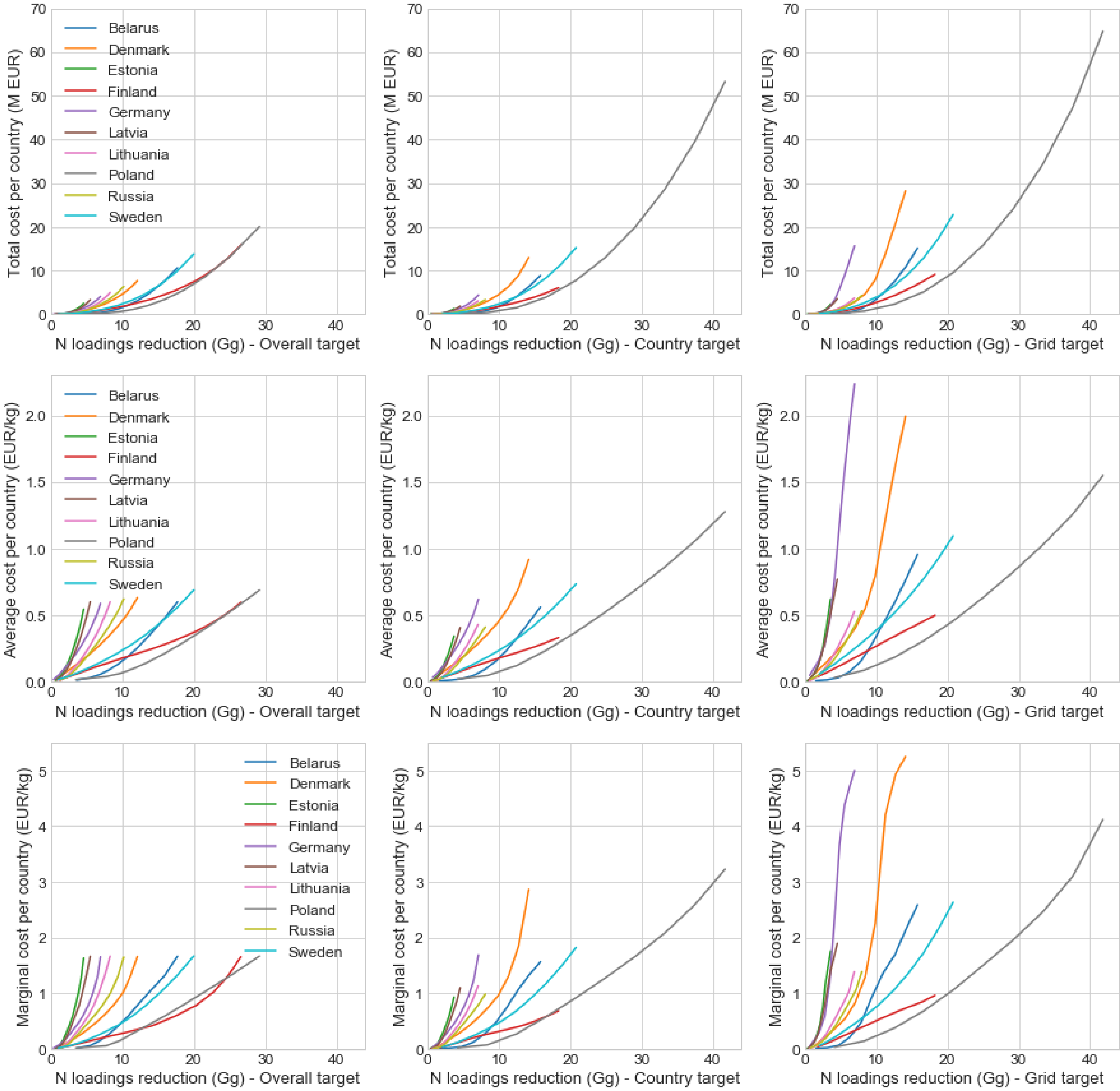


The same pattern can be observed for each country. Figure 3 presents the total cost (top), average cost (middle) and marginal cost (bottom) of delivering various N loadings reductions for each country if the N load reduction target is allocated at the Baltic Sea level (left), at individual country level (middle) and at grid-square level (right). The first thing to notice is that extreme total costs are significantly reduced in the optimal (Baltic Sea-wide) allocation, relative to imposing the same relative N reduction targets at the level of each country or each grid square. This is further illustrated by differences in the average costs of reducing N loadings that reach the sea for each country. While, with 50% N reduction targets specified at grid square resolution, the average cost of preventing 1 kg of N load reaching the sea can be as high as 2 EUR (Germany, Denmark), achieving the same overall target with a cost-effective distribution of measures across the whole of the Baltic Sea drainage basin yields average costs that are no higher than 0.7 EUR/kg. This is a result of reallocating costly (e.g., Denmark, Germany) or low-effectiveness (due to high retention, e.g., Poland) measures to more cost-effective locations. Finally, the bottom panel of Figure 3 presents marginal abatement cost estimates, that is, the cost of reducing an additional kg of N, given the current level of reductions. In an optimized allocation of abatement effort, the marginal cost in each country (and each grid square) would be equalized – otherwise it would be profitable to make lower reductions in areas with higher marginal costs, and substitute them with reductions in areas with lower marginal costs (Mäler, 1989). This is exactly what we find in the left-hand graph – the marginal costs corresponding to 50% reductions are roughly equal, while marginal costs associated with assigning N reduction targets at country or grid square scales vary substantially, by at least a factor of four.

The results presented in the bottom panel of Figure 3 also show that all the marginal costs functions are not only increasing but also convex (except at for very expensive marginal costs when N reduction targets are assigned at Grid scale). This information is valuable for two reasons. First, it implies that assuming linear marginal costs of nutrient reduction can produce high estimation errors. The scale of this problem is clearly visible when we analyze the values of marginal cost for different reduction targets. Marginal cost increases rapidly.12 To illustrate, marginal cost is around 4 times higher for the 50% reduction target than for the 25% reduction target. Second, marginal cost is the information that can be useful for evaluating economically effective reduction targets, and for identification of economic instruments that could be used to get close to the cost-effective solution (de Vries and Hanley, 2016). The N reduction targets that are set in BSAP are derived based on an evaluation from an ecological rather than an economic perspective. Due to the fact that marginal costs rise rapidly with increases in the ambition of the pollution reduction target, the economic significance of even small changes in the reduction targets can be profound. This is illustrated in the most tangible way by values of average cost (EUR per kg) that are more than 3 times greater for a 50% rather than a 25% reduction target (0.63 and 0.18 EUR respectively, when the N reduction target is set for the Baltic Sea drainage basin as a whole).

12 This is likely because the model does not allow for changes in the distribution of crops in response to the required reductions in fertilization.

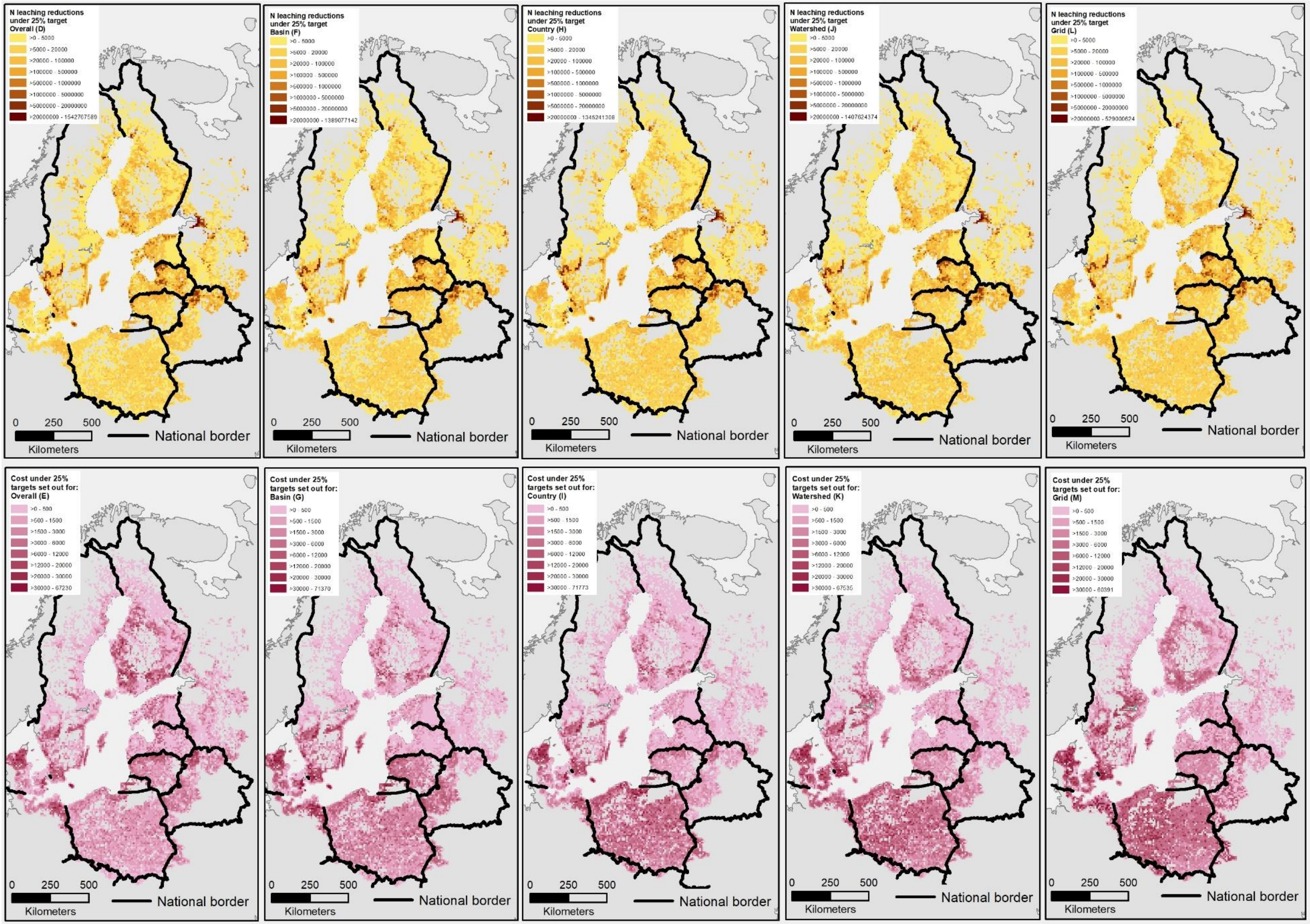
Figure 3. Total, average and marginal costs of N load reductions in each country with N reduction targets set at different spatial scales



To illustrate differences between various target allocations spatially, the top panel of Figure 4 presents the distribution of cost-effective N load reductions in the Baltic Sea drainage area for the 25% reduction target allocated for the Baltic Sea as a whole, for each sea basin, each country, each watershed and each grid square. The bottom panel of Figure 4 presents the corresponding costs incurred in each grid square. Overall, the larger the spatial scale over which the N load reduction target is specified (e.g., the Baltic Sea as a whole) the lighter the colors, indicating a reduction in the overall cost. However, the effect is not uniform for all regions, as a result of substituting abatement effort from less effective grid squares (e.g., with high retention) to more effective ones.13 This may mean some countries face the prospect of net losses (the difference between the costs and benefits of nutrient pollution control) at the national level from signing up to a Baltic-wide cost-effective program. This would create challenges for getting all countries to sign up to such a cost-effective policy (Kaitala, Mäler and Tulkens, 1995).

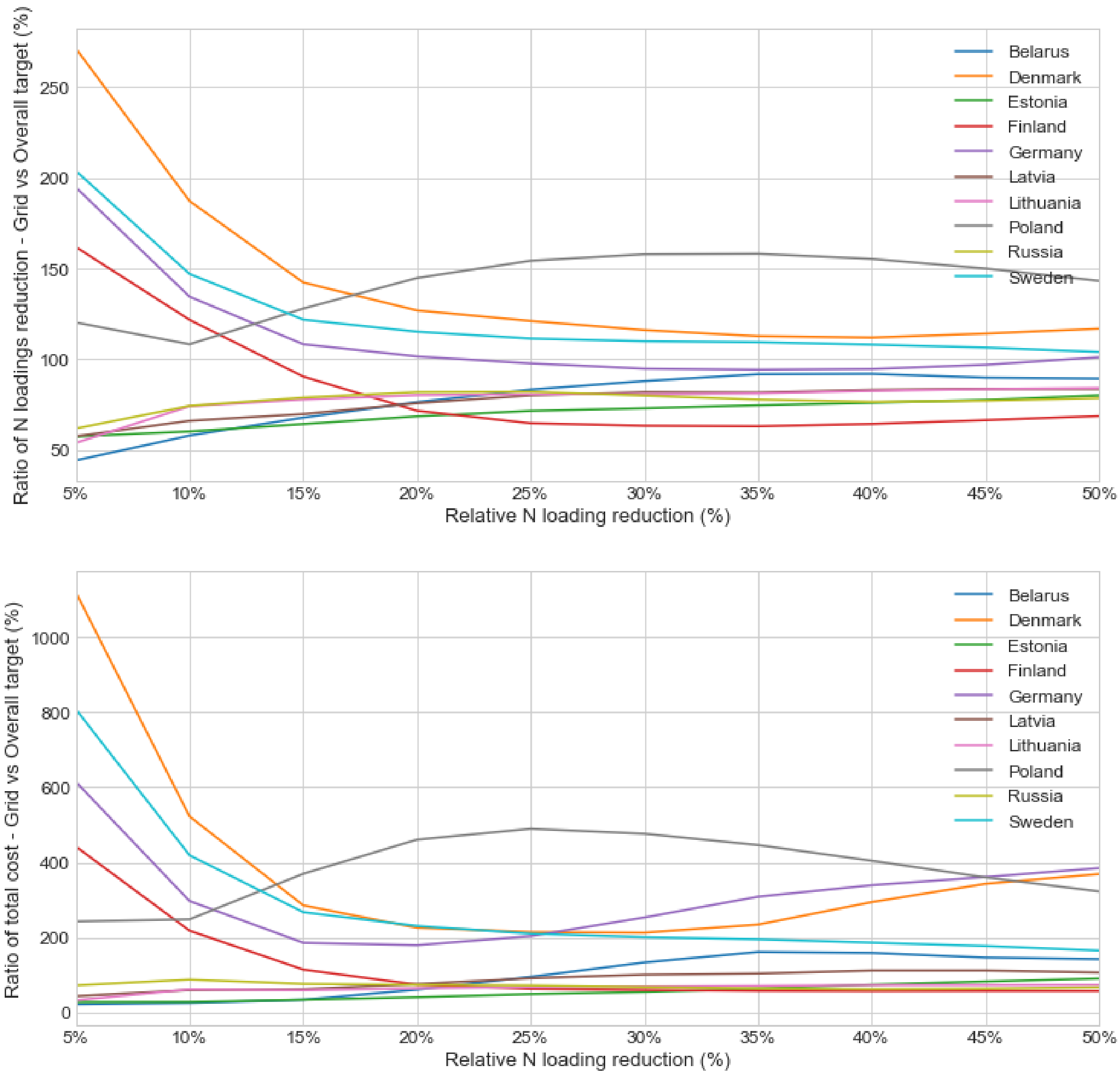
13 The greyed-out areas indicate grid squares with no changes in N leaching.

Figure 4. The distribution of N load reductions and the costs of reductions for the 25% reduction target specified at different spatial scales



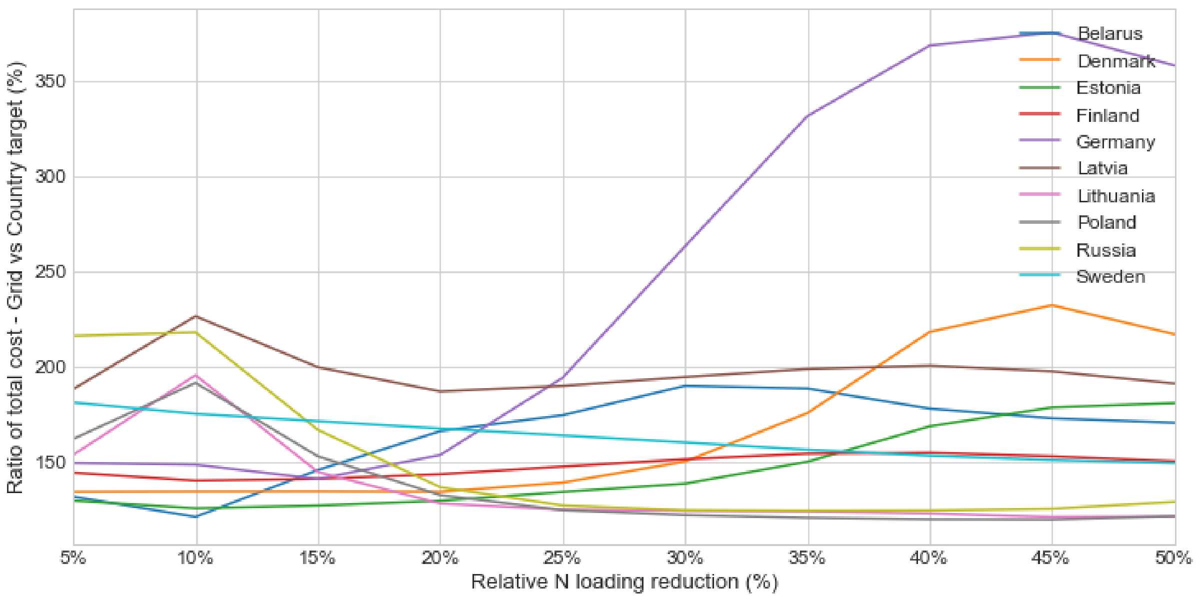
Another useful insight can be provided by the comparison of the ratios of the required nutrient load reductions and the associated costs in the grid square constrained (Grid) and optimal (Baltic Sea wide; Overall) scenarios incurred by countries (Figure 5). For Denmark, Sweden and Poland the cost-effective allocations would require lower N reductions than those required when N reductions are specified at grid square scale. For Germany and Finland, the outcome depends on the reduction target, as the curves intersect the 100% line, which corresponds to equal N reductions under allocations optimized for the Baltic Sea as a whole or within each grid square. The lower overall reductions in these countries would need to be compensated by increased reductions in the remaining countries. The lower panel of Figure 5 presents the equivalent ratios for the costs. This illustrates the importance of utilizing economic methods and international cooperation for deriving the least cost solutions to international environmental problems. Additionally, it can be seen that a high-resolution analysis is necessary for economic accuracy and effectiveness as the countries' share of total nutrient reduction changes significantly for different N reduction targets.

Figure 5. The ratio of overall N load reductions (top panel) and the costs (bottom panel) in each country with N reduction targets specified for each grid square, relative to targets optimized for the Baltic Sea as a whole



The benefits of allowing costs to be minimized over larger spatial scales are present even without any international cooperation. This can be seen in the comparison of the ratios of the costs incurred by individual countries under grid- and country-constrained scenarios (Figure 6). For any reduction target that is set for a given country significant gains can be realized by applying an efficient distribution of N abatement within that country – the cost ratios in Figure 6 are always above 100%, indicating that it is always cheaper to optimize N reductions within a country than to impose equal relative N reductions on each grid cell. Figure 6 illustrates the extent of this benefit, with the grid-scale targets leading to over 150% higher costs in more than 50% of the analyzed cases.

Figure 6. The ratio of costs in each country under N reduction targets specified for each grid square relative to the targets specified for each country



5. Discussion and conclusions

This paper shows how the costs of delivering a targeted improvement in water quality in an internally-shared common pool resource – the Baltic Sea – depend critically on the spatial scale at which target reductions are set and delivered. Our study delivers the first nutrient reduction cost optimization model for the Baltic Sea at such a spatially-disaggregated scale, and is one of the first environmental economics models at such a disaggregation scale in general; see also Bateman et al. (2013). We show large and consistent reductions in the costs of reducing pollution in moving from localized targets to national and then Baltic Sea-wide targets. These cost savings come about due to variations in marginal abatement costs and leaching rates across and within countries. There is no reason to think that this finding on the benefits of targeting the highest spatial level for realizing shared water quality benefits does not hold in other, comparable contexts: we would thus argue that our findings are of general interest, and not confined to the case study system.

The average cost of abating 1 Mg of N from agricultural sources predicted from our model is approximately 500 - 2,000 EUR, depending on the target ambition and its allocation. It is difficult to compare this result with other studies, which typically include other measures and consider joint abatement of N and P loadings, however, the results are similar to the cost of

1,200 EUR/Mg N reported by, for example, Hasler et al. (2014), and in the range of costs referenced by Gren (2008) and Ollikainen and Honkatukia (2001). On the other hand, the estimated costs of spatially optimized reductions in fertilization tend to be lower than those reported in the literature, illustrating the advantages of policies tailored for high spatial resolution.

There are several limitations of our study that need to be acknowledged. First, the costs predicted by our model are based on experimental yield functions observed for Denmark. We assume that farmers apply economically optimal levels of fertilizer, which result in relatively low reductions in yields when fertilization is reduced (quadratic functions become relatively flat close to their maximums). In addition, these functions are not tailored to specific local (county-specific) conditions. Furthermore, we did not impose any restrictions on the extent of fertilization reductions; this is in contrast with many other studies that assume fertilization cannot be reduced by more than a specified percentage of the current level (e.g., Hasler et al., 2014; Wulff et al., 2014).

Another limitation of the proposed approach stems from the challenges in introducing spatially-differentiated regulation. Currently, there are no policies in place that would regulate farm activities with such a level of spatial heterogeneity. However, this is not to say such policies would be infeasible. One way to introduce such a governance solution could be provided by differentiating farm subsidies or payments for measures implemented in agri-environmental schemes, under Pillar 2 of the CAP or new “greening” measures, like the obligatory balancing of biogenic elements (N, P) proposed within New Delivery Model of the CAP by the European Commission (Wąs, Malak-Rawlikowska and Majewski, 2018). Introduction of fertilization constraints could be potentially offset by compensation payments to farmers, so that their profits remain relatively unchanged. This would make such a policy more politically feasible, and would limit the capitalization of respective constraints into land prices. However, it would also increase the total cost of achieving the N targets very considerably, and constitute an unwanted case of ‘beneficiary pays’ principle. On the other hand, it might be argued that ‘capitalization of nitrogen liabilities’ into land prices is long overdue. Agricultural land with higher N loss potential that is well-connected to receiving waters should only be subject to modest levels of N-fertilization. This limits its profit potential (at least in terms of agricultural production), but opens up an opportunity for a change of land use to more profitable purposes (from a societal perspective), including conversion to constructed N-treatment wetlands.

This raises the important question of how the spatial pattern of nitrate reductions which are identified by constrained cost minimization could actually be achieved in practice. It seems highly unlikely that farmers in each watershed in each Baltic Sea state could be instructed to reduce estimated emissions in the manner indicated by the least-cost outcome. Aside from obvious political barriers to regulating farmers in this way, the monitoring and enforcement cost implications are substantial. What would be more relevant would be to devise a set of differentiated economic incentives which farmers respond to in a manner which gets as close as possible to this least-cost solution. Early work on the economics of non-point source pollution control showed that a system of estimated emissions taxes on nutrients could, in principle, get close to this cost-minimizing pattern of abatement. However, implementing an estimated emissions tax would be difficult. Taxes on nitrogen inputs are simpler to administer, but do not reflect spatial variations in the transfer of nitrogen inputs to final loadings discharged, in this case, to the Baltic (Larson, Helfand and House, 1996). Two policy options that are more promising are either (i) to employ a combination of managerial standards with a nitrogen /manure tax, where the managerial standard can vary at the watershed level: such a combination has been shown to result in outcomes which get close to the cost-minimizing first-best solution for nitrate pollution at the catchment level (Aftab, Hanley and Baiocchi, 2010; 2017); or (ii) use an ambient tax/subsidy scheme where the payment made by or to an individual farmer depends on whether water quality at the basin or total Baltic Sea level falls below or above a target threshold at any point in time (Segerson, 1988; Suter et al., 2008). However, such ambient tax/subsidy schemes have been argued to depend on farmers' expectations on how their management actions will affect ambient pollution levels (in this case, nitrate loadings at the basin/Baltic Sea level).

Notwithstanding the limitations of our study, our results demonstrate the extent of the benefits resulting from spatial aggregation of N targets that allows fine-scale spatial differentiation of abatement measures. We show that tailoring the policy measures to finer spatial resolution (in our case, 10-by-10 km grid-squares), rather than imposing a uniform mitigation strategy at a larger spatial scale reduces the cost of reaching a particular nutrient abatement target by approximately 50%. The equivalent gains in comparison with setting uniform targets for all countries or watersheds (and then optimizing abatement measures within such policy units) are 22% and 31%, respectively. This shows that the potential cost effectiveness gains from using spatial differentiation strategies to reduce nutrient loadings to the Baltic Sea are larger than expected from earlier studies conducted at coarser spatial scales.

For example, Jacobsen and Hansen (2016) analyzed 23 watersheds in Denmark and found that spatial differentiation of targets could lead to 25% cost reductions, for a particular nutrient abatement target. Refsgaard et al. (2019) use case studies in selected Baltic Sea countries to demonstrate that spatially differentiated regulation (relocation of crops) can lead to up to 8% or 26% N-load reductions in Norsminde and Odense, respectively.

Finally, it is worth noting that if some countries need to make larger reductions (at higher costs) than other countries at the highest level of aggregation for target-setting (to achieve the overall cost-minimizing pattern of pollution reduction), then those “high reduction” countries may be opposed to agreeing to such a Baltic Sea-wide pattern of pollution cuts. This might not be the case if they are also the countries who gain the most benefits from reductions in eutrophication. Comparison of the cost per country (or per capita per country) with the benefits of recreational use of the Baltic Sea (e.g., Czajkowski et al., 2015) show, that this is not always the case. The countries that would have to pay the most (Finland, Sweden, Poland) or pay the most per capita (Finland, Estonia, Lithuania) are not necessarily the countries that currently derive the highest consumers surplus from the recreational use of the sea (Germany, Sweden, Poland), or the highest consumer surplus per capita (Sweden, Finland, Denmark). Similar conclusions arise when the costs are compared with the estimated benefits of improved condition of the Baltic Sea (e.g., Ahtiainen et al., 2014; Czajkowski et al., 2015) – the countries whose citizens are willing to pay the most are Germany, Sweden, and Poland, or in per capita terms, Germany, coastal Russia and Sweden. This indicates that adoption of cost-minimizing nutrient abatement policy for the Baltic Sea region would be politically difficult without between-country compensations (Kaitala, Mäler and Tulkens, 1995; Markowska and Żylicz, 1999).

Overall, our study demonstrates the significant potential of fine-scale spatial differentiation of abatement measures. We provide the estimates of theoretically-consistent gains resulting from allocating nutrient reduction targets to larger areas, while allowing for optimization of measures within these areas. In particular, we show that achieving overall cost effectiveness requires that nutrient abatement varies at finer scales than the current basin / country / watershed scales at which policies have typically been differentiated hitherto. To fully utilize these gains, however, more research is needed with respect to overcoming governance and economic challenges that arise between political jurisdictions for such spatially-tailored policies for inter-jurisdictional common pool resources.

References

Abrahamsen, P., and Hansen, S., 2000. Daisy: an open soil-crop-atmosphere system model. Environmental Modelling & Software, 15(3):313-330.

Aftab, A., Hanley, N., and Baiocchi, G., 2010. Integrated regulation of nonpoint pollution: Combining managerial controls and economic instruments under multiple environmental targets. Ecological Economics, 70(1):24-33.

Aftab, A., Hanley, N., and Baiocchi, G., 2017. Transferability of Policies to Control Agricultural Nonpoint Pollution in Relatively Similar Catchments. Ecological Economics, 134:11-21.

Ahlvik, L., Ekholm, P., Hyytiäinen, K., and Pitkänen, H., 2014. An economic–ecological model to evaluate impacts of nutrient abatement in the Baltic Sea. Environmental Modelling & Software, 55(0):164-175.

Ahtiainen, H., Artell, J., Czajkowski, M., Hasler, B., Hasselström, L., Huhtala, A., Meyerhoff, J., Smart, J. C. R., Söderqvist, T., Alemu, M. H., Angeli, D., Dahlbo, K., Fleming-Lehtinen, V., Hyytiäinen, K., Karlõševa, A., Khaleeva, Y., Maar, M., Martinsen, L., Nõmmann, T., Pakalniete, K., Oskolokaite, I., and Semeniene, D., 2014. Benefits of meeting nutrient reduction targets for the Baltic Sea – a contingent valuation study in the nine coastal states. Journal of Environmental Economics and Policy, 3(3):1-28.

Andersen, H. E., Blicher-Mathiesen, G., Thodsen, H., Andersen, P. M., Larsen, S. E., Stålnacke, P., Humborg, C., Mörth, C.-M., and Smedberg, E., 2016. Identifying Hot Spots of Agricultural Nitrogen Loss Within the Baltic Sea Drainage Basin. Water, Air, & Soil Pollution, 227(1):38.

Bateman, I. J., Harwood, A. R., Mace, G. M., Watson, R. T., Abson, D. J., Andrews, B., Binner, A., Crowe, A., Day, B. H., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Lovett, A. A., Munday, P., Pascual, U., Paterson, J., Perino, G., Sen, A., Siriwardena, G., van Soest, D., and Termansen, M., 2013. Bringing Ecosystem Services into Economic Decision-Making: Land Use in the United Kingdom. Science, 341(6141):45-50.

Baumol, W. J., and Oates, W. E., 1971. The Use of Standards and Prices for Protection of the Environment. The Swedish Journal of Economics, 73(1):42-54.

Brady, M., 2003. The relative cost-efficiency of arable nitrogen management in Sweden. Ecological Economics, 47(1):53-70.

COWI, 2007. Economic analysis of BSAP with focus on eutrophication - Interim Report. Report 2(1) to HELCOM, COWI Group Consulting.

Czajkowski, M., Ahtiainen, H., Artell, J., Budziński, W., Hasler, B., Hasselström, L., Meyerhoff, J., Nõmmann, T., Semeniene, D., and Söderqvist, T., 2015. Valuing the commons: An international study on the recreational benefits of the Baltic Sea. Journal of Environmental Management, 156:209-217.

de Vries, F. P., and Hanley, N., 2016. Incentive-Based Policy Design for Pollution Control and Biodiversity Conservation: A Review. Environmental and Resource Economics, 63(4):687-702.

Elofsson, K., 2010. The Costs of Meeting the Environmental Objectives for the Baltic Sea: A Review of the Literature. AMBIO, 39(1):49-58.

Gren, I.-M., 2008. Costs and benefits from nutrient reductions to the Baltic Sea. Swedish Environmental Protection Agency Report no. 5877, Stockholm, Sweden.

Gren, I.-M., Jannke, P., and Elofsson, K., 1997. Cost-Effective Nutrient Reductions to the Baltic Sea. Environmental and Resource Economics, 10(4):341-362.

Gren, I.-M., Söderqvist, T., and Wulf, F., 1997. Nutrient Reductions to the Baltic Sea: Ecology, Costs and Benefits. Journal of Environmental Management, 51:123-143.

Grimvall, A., and Stålnacke, P. E. R., 1996. Statistical methods for source apportionment of riverine loads of pollutants. Environmetrics, 7(2):201-213.

Hanley, N., Shogren, J., and White, B., eds. 2006. Environmental Economics in Theory and Practice. Palgrave MacMillan.

Hasler, B., Smart, J. C. R., Fonnesbech-Wulff, A., Andersen, H. E., Thodsen, H., Mathiesen, G. B., Smedberg, E., Göke, C., Czajkowski, M., Was, A., Elofsson, K., Humborg, C., Wolfsberg, A., and Wulff, F., 2014. Hydro-economic modelling of cost-effective transboundary water quality management in the Baltic Sea. Water Resources and Economics, 5:1-23.

HELCOM, 2011. The Fifth Baltic Sea Pollution Load Compilation (PLC-5). Baltic Sea Environment Proceedings, 128.

Jacobsen, B. H., and Hansen, A. L., 2016. Economic gains from targeted measures related to non-point pollution in agriculture based on detailed nitrate reduction maps. Science of The Total Environment, 556:264-275.

Kaitala, V., Mäler, K.-G., and Tulkens, H., 1995. The acid rain game as a resource allocation process with application to negotiations between Finland, Russia and Estonia. Scandinavian Journal of Economics, 97(2):325-343.

Larson, D. M., Helfand, G. E., and House, B. W., 1996. Second-Best Tax Policies to Reduce Nonpoint Source Pollution. American Journal of Agricultural Economics, 78(4):1108-1117.

Lidén, R., Vasilyev, A., Stålnacke, P., Loigu, E., and Wittgren, H. B., 1999. Nitrogen source apportionment—a comparison between a dynamic and a statistical model. Ecological Modelling, 114(2–3):235-250.

Mäler, K. G., 1989. The acid rain game. In: Valuation Methods and Policy Making in Environmental Economics, H. Folmer and E. van Ierland, eds., Elsevier Science.

Markowska, A., and Żylicz, T., 1999. Costing an International Public Good: The Case of the Baltic Sea. Ecological Economics, 30:301-316.

Ollikainen, M., and Honkatukia, J., 2001. Towards Efficient Pollution Control in the Baltic Sea: An Anatomy of Current Failure with Suggestions for Change. AMBIO: A Journal of the Human Environment, 30(4):245-253.

Pedersen, J. B., 2009. Oversigt over landsforsøgene 2009. Forsøg og undersøgelser i Dansk Landbrugsrådgivning. Dansk Planteproduktion. Available at https://www.landbrugsinfo.dk/Planteavl/Landsforsoeg-og-resultater/Oversigten-og-tabelbilaget/Sider/pl_Oversigten_2009.pdf.

Refsgaard, J. C., Hansen, A. L., Højberg, A. L., Olesen, J. E., Hashemi, F., Wachniew, P., Wörman, A., Bartosova, A., Stelljes, N., and Chubarenko, B., 2019. Spatially differentiated regulation: Can it save the Baltic Sea from excessive N-loads? AMBIO.

Reusch, T. B. H., Dierking, J., Andersson, H. C., Bonsdorff, E., Carstensen, J., Casini, M., Czajkowski, M., Hasler, B., Hinsby, K., Hyytiäinen, K., Johannesson, K., Jomaa, S., Jormalainen, V., Kuosa, H., Kurland, S., Laikre, L., MacKenzie, B. R., Margonski, P., Melzner, F., Oesterwind, D., Ojaveer, H., Refsgaard, J. C., Sandström, A., Schwarz, G., Tonderski, K., Winder, M., and Zandersen, M., 2018. The Baltic Sea as a time machine for the future coastal ocean. Science Advances, 4(5).

Schou, J. S., Neye, S. T., Lundhede, T., Martinsen, L., and Hasler, B., 2006. Modelling Cost-efficient reductions of nutrient loads to the Baltic Sea. Model specification, Data and Cost functions. NERI Technical Report no. 592, Copenhagen, Denmark.

Segerson, K., 1988. Uncertainty and incentives for nonpoint pollution control. Journal of Environmental Economics and Management, 15(1):87-98.

Seitzinger, S. P., 1988. Denitrification in Freshwater and Coastal Marine Ecosystems: Ecological and Geochemical Significance. Limnology and Oceanography, 33(4):702-724.

Shortle, J. S., and Horan, R. D., 2001. The Economics of Nonpoint Pollution Control. Journal of Economic Surveys, 15(3):255-289.

Stalnacke, P., Grimvall, A., Libiseller, C., Laznik, A., and Kokorite, I., 2003. Trends in nutrient concentrations in Latvian rivers and the response to the dramatic change in agriculture. Journal of Hydrology, 283(1-4):184-205.

Stålnacke, P., Pengerud, A., Vassiljev, A., Smedberg, E., Mörth, C.-M., Hägg, H. E., Humborg, C., and Andersen, H. E., 2015. Nitrogen surface water retention in the Baltic Sea drainage basin. Hydrol. Earth Syst. Sci., 19:981-996.

Suter, J. F., Vossler, C. A., Poe, G. L., and Segerson, K., 2008. Experiments on Damage-Based Ambient Taxes for Nonpoint Source Polluters. American Journal of Agricultural Economics, 90(1):86-102.

Vassiljev, A., Blinova, I., and Ennet, P., 2008. Source apportionment of nutrients in Estonian rivers. Desalination, 226(1):222-230.

Vassiljev, A., and Stålnacke, P., 2005. Statistical modelling of riverine nutrient sources and retention in the Lake Peipsi drainage basin. Water Science and Technology, 51(3-4):309-317.

von Schiller, D., Marti, E., Riera, J. L., Ribot, M., Argerich, A., Fonolla, P., and Sabater, F., 2008. Inter-annual, annual, and seasonal variation of P and N retention in a perennial and an intermittent stream. Ecosystems, 11(5):670-687.

Wąs, A., Malak-Rawlikowska, A., and Majewski, E., 2018. The new delivery model of the Common Agricultural Policy after 2020-challenges for Poland. Problems of Agricultural Economics/Zagadnienia Ekonomiki Rolnej, 4(916-2019-2370):33-59.

Wulff, F., Humborg, C., Andersen, H. E., Blicher-Mathiesen, G., Czajkowski, M., Elofsson, K., Fonnesbech-Wulff, A., Hasler, B., Hong, B., Jansons, V., Mörth, C.-M., Smart, J. C. R., Smedberg, E., Stålnacke, P., Swaney, D. P., Thodsen, H., Was, A., and Żylicz, T., 2014. Reduction of Baltic Sea Nutrient Inputs and Allocation of Abatement Costs Within the Baltic Sea Catchment. AMBIO: A Journal of the Human Environment, 43(1):11-25.

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