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# Economic effects of the invasive worm *Marenzelleria* spp. in the Baltic Sea

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# Economic effects of the invasive worm *Marenzelleria* spp. in the Baltic Sea.

*Abstract.* Impacts of alien marine species on ecosystem services have been documented in numerous studies, but studies on the estimation of costs with an explicit quantification of the impact channels are lacking. We calculated costs of the invasive worm *Marenzelleria* in the Baltic Sea, which affects sea bottoms capacity to store nutrients. A reduced capacity implies higher costs for reaching nutrient reduction targets. Therefore, the costs were calculated by taking the difference in minimum costs of attaining nutrient targets with and without the presence of the invasive species. To this end, we developed a simple dynamic model which captures the effect of the invasive species on the decay rates of nutrient pools in different and interconnected sea basins in the Baltic Sea. Evidence and numerical estimates were found for a decrease in sequestration of phosphorus in the different sea basins caused by the occurrence of the invasive worm. The results indicated a cost of this effect that ranges between 167 billion SEK and 732 billion SEK depending on the effect of *Marenzelleria* on sequestration of phosphorus. This cost corresponds to 0.02-0.08% of total gross domestic product of the nine coastal countries. However, the cost is unevenly distributed among the countries, where Poland bears the largest part because of its large phosphorus loads and access to low cost options.

*Key words*; invasive species, Marenzelleria, costs, nutrients, eutrophication, dynamic cost effectiveness, Baltic Sea

JEL: Q250, Q280, Q530

# 1. Introduction

Numerous studies have pointed out effects of alien aquatic species on ecosystem services (e.g Vila et al., 2009; Katsanevakis et al., 2014), and a few have estimated costs (see reviews in Lovell and Stone, 2005; Marbuah and Gren, 2014). Relatively many studies have calculated costs of aquatic invasive species in terms of their impact on provisioning and cultural ecosystem services, in particular on production losses in commercial fishery and decline in recreational values of sport fishery (Knowler and Barbier, 2000; Lovell and Stone, 2005; Adams and Lee, 2007; Carlsson and Kataria, 2008; Horsch and Lewis, 2009). For example, Rockwell (2003) shows that the costs of aquatic weeds in the US from production losses in fishery and decreases in recreational values amounts to 10 billion USD.

However, aquatic invasive species also affect supporting and regulating services, such as ocean nourishment and climate regulations of a sea (e.g. Katsanevakis et al., 2014). The estimation of costs of such impacts are usually difficult since it requires numerical modelling of impacts on the ecosystem, associated provision of several types of ecosystem services, and measurement of these services in monetary terms. Nevertheless, it is of importance to have some understanding of the magnitude of costs created by different aquatic alien species in order to prioritize mitigation actions. The purpose of this study is to provide and demonstrate a method, the so-called replacement cost method, for the calculation of costs which does not need quantified information on the effects of several ecosystem services and their value in monetary terms. Instead, the method rests on governmental decisions on marine targets and costs for achieving these targets. It is applied to the estimation of costs of the invasive worm *Marenzelleria* spp., which affects the nutrient enrichment of the Baltic Sea. The estimate of costs of the species is then based on calculations of the impacts of the species on nutrient enrichment and on existing intergovernmental targets on nutrient loads.

The polychaete genus *Marenzelleria* (Mesnil) was introduced into the Baltic Sea in the mid 1980’s (Bick and Burckhardt 1989), most likely via ballast water (Bastrop and Blank, 2006). Since then, *Marenzelleria* spp. has become a significant component of benthic faunal communities across the whole Baltic Sea (Zettler et al. 2002). Currently, three species of *Marenzelleria* are known to be present in the Baltic Sea and they are known to have separate ecological niches regarding bioturbation (Renz and Forster, 2013). It is currently largely unclear what role *Marenzelleria* itself plays as prey species in the Baltic Sea food web but it is known that old deposited organic material is an important food source (Karlson et al., 2015).

*Marenzelleria* bury much deeper than native deposit-feeding fauna and can thereby access organic material which is unavailable for the native species (Kotta et al., 2006), leading to decreased deposition rates. Additionally, experimental and field studies using stable isotopes also show that *Marenzelleria* increases the community resource use efficiency for deposited organic matter (i.e. a higher fraction of the nutrients in the organic matter is recycled) (Karlson et al., 2011; Karlson et al., 2015), further reducing organic matter deposition. *Marenzelleria* can also increase the release of organic contaminants from sediments (Granberg et al., 2008). Currently, there is a lack of field and model studies that consider the effects of *Marenzelleria* on relevant spatial and temporal scales. Norkko et al. (2012), however, have performed a study showing that effects of *Marenzelleria* on phosphorous cycling are potentially very relevant also at a local/regional scale.

The effect of *Marenzelleria* on sea bottoms' storage of nutrients can affect the eutrophication status of the Baltic Sea through impacts on the concentrations of nutrients in the sea water (e.g. Maximov et al., 2015). Eutrophication is regarded as the major threat to coastal water quality worldwide (Selman et al. 2008). Between the early 1960s and 2007, the documented number of eutrophied coastal zones in the world has increased from approximately 10 hypoxic areas to at least 169 (Selman et al. 2008). Symptoms of eutrophication are oxygen depletion, harmful algal blooms, and changes in the composition of fish species at the disadvantage of commercial species. For example, excess blooms of a toxin-producing flagellate killed large numbers of fish in the Baltic Sea and on the Atlantic coast of the USA (Burkholder and Glasgow, 1997), and one algal bloom destroyed 90% of the fish population in farms in Hong Kong (Selman et al., 2008). Worldwide, management plans have been implemented in order to combat damages from eutrophication. However, the presence of the invasive *Marenzelleria* is likely to influence the costs of these plans through the species' effect on the nutrient recycling in the sea bottoms which, for a given level of external nutrient loads to the sea, affects the concentration of nutrient in the sea. The direction of change can be either an increase or decrease in costs depending on the effect.

This study estimates economic impacts of *Marenzelleria* through the effect on the costs of reaching international agreements for mitigating eutrophication in the Baltic Sea in Northern Europe, which contains the largest hypoxic zones in the world (Conley et al. 2009). International agreements for the Baltic Sea are set for future achievement of reductions in nutrient loads. The most stringent targets are set for phosphorus pools, which need to be reduced by at least 50% (Helcom, 2013a). Data are available on impacts of *Marenzelleria* on phosphorus dynamics, but not on nitrogen dynamics. Therefore, we calculate minimum costs for reaching phosphorus targets in the Baltic Sea with and without *Marenzelleria*. To this end, we construct a dynamic optimisation model, which accounts for differences in decay rates of nutrients, i.e. the rate at which a certain amount of phosphorus is decomposed, between the sea basins.

To the best of our knowledge, there exists no study using the replacement cost method for estimating the cost of an invasive species. Instead, costs have been estimated as damages on facilities, such as the clogging of zebra mussels on drain-pipes in the Great Lakes (Finnof et al., 2005) and loss of property values (Lovell and Stone, 2005; Horsch et al., 2009). Another commonly applied method is the use of expenses for combatting spread and damage of invasive species, such as public authorities' cost of harvesting an invasive aquatic weed and vaccination of salmon against the furunculosis caused by invasive bacteria (Gren et al., 2009). The idea is that such costs would not be borne by the authority unless they are covered by the avoided costs of the invasive species. However, the replacement cost method has been much applied to the valuation of regulating and supporting ecosystem services, such as nutrient retention by wetlands and coastal zones, and carbon sequestration by forests (e.g. Breux et al., 1995; Byström et al., 2000). One reason for the lack of applications on invasive species affecting such services can be the requirement of relevant environmental targets, data and numerical models of the species' impact on a regulating or supporting ecosystem service of relevance for the achievement of the targets, and information on costs for measures reducing the pollutant loads.

The paper is organised as follows. First, we present the structure of the numerical dynamic model which is used to identify possible economic effects of the presence of *Marenzelleria*. Next, data are briefly described, and the results are presented in Section 4. The paper ends with a summary and tentative conclusions.

## 2. Conceptual approach and the dynamic cost effectiveness model

The replacement cost method assigns value or a cost to a species only if this species affects the abatement costs of reaching specified environmental targets. The cost effective solutions are, in turn, determined by a numerical model for the minimization of total costs for achieving nutrient targets in a certain time and space. The construction of a cost minimizing model is thus crucial for the determination of value or cost of the species. In the following, we therefore give an intuitive presentation of the replacement cost method before presenting the structure of the cost effectiveness model, which is used to identify parameters affecting the value or cost of the invasive *Marenzelleria*.

## 2.1 Conceptual framework

The value or cost of the invasive species is determined by its impact on the cost effective solution to a predetermined nutrient target. High cost of abatement measures, such as reductions in nutrients from sewage treatment plants and changed land use practices in agriculture, results in high cost or value of the species. This simple principle is illustrated in Figure 1 for phosphorus reduction and target when the species creates a cost.

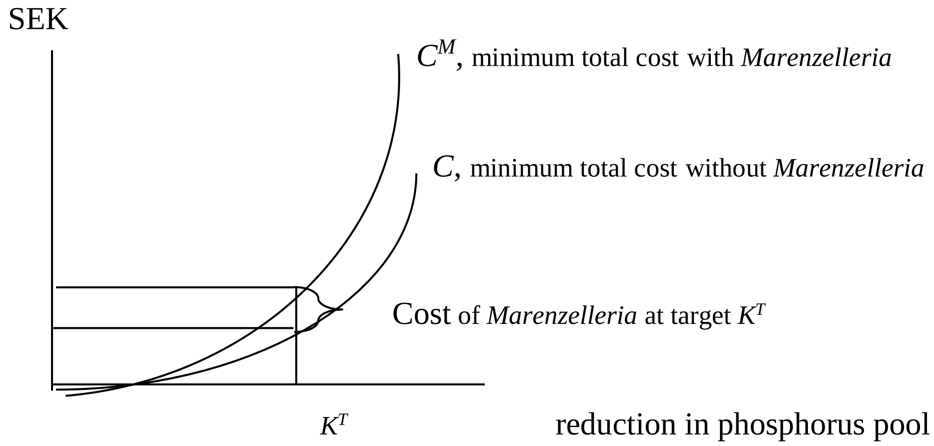

*Figure 1: Illustration of the calculation of cost of Marenzelleria in a cost effectiveness framework*

The horizontal axis illustrates reductions in the pool of phosphorus in the sea, and $K^T$ is the target to be achieved. The vertical axis shows the cleaning cost for different pool reduction levels. The curve $C^M$ illustrates the minimum costs for achieving different pool reduction targets when *Marenzelleria* is present in the sea, and $C$ illustrates the minimum costs without the species in the sea. Each point on $C^M$ and $C$ respectively reflects the allocation of all abatement measures that reaches the target at minimum costs.

The cost of *Marenzelleria* at the target $K^T$ is now determined by the difference in total minimum costs with and without the species, which corresponds to the distance $C^M$-$C$ in Figure 1. The cost of the species is then determined by the costs of the abatement measures. The larger the difference between the abatement costs with and without *Marenzelleria*, the higher is the cost of the species. This is, in turn, determined by the stringency in the phosphorus pool target since the costs of all abatement measures increase at higher pool reduction levels.

## 2.2 Dynamic cost effectiveness model

The basis for the model is an international sea shared by several countries *g=1,..n*, with a common agreement on achieving future nutrient targets, such as those in the HELCOM (2013a) and OSPAR (2010). Since the ecological conditions differ for each of the marine basins, *i=1,..,k*, in the sea, the targets also differ. In order to account for this, we adopt the model developed by Gren et al. (2013), but add the impact of *Marenzelleria* on nutrient dynamics in the marine basins.

*Marenzelleria* affects the sequestration of nutrient in the sediments, which we model as an impact on the decay of nutrient pools. However, the dynamics of the nutrient and the impact of *Marenzelleria* are relatively poorly understood in quantitative terms. We therefore assign a simple relation between the quantity of the species, $M^i$ , and the decay rate of nutrient in basin *i*, where *i=1,..,k* basins, which is written as $\alpha^i(M^i)$ where the sign of the derivative $\partial\alpha^i/\partial M^i$ can be zero, positive or negative. That is, marginal changes in $M^i$ have no impact, increase or decrease the decay rate. The stock, or pool, of phosphorus in period *t+1* in basin *i*, $S^i_{t+1}$ is also given a simple representation as a linear function of the stock in earlier period, $S^i_{t+1}$, decay rate $\alpha^i(M^i)$, and nutrient load $B^i_t$, according to

$$S^i_{t+1} = (1-\alpha^i(M^i))S^i_t + B^i_t \qquad \text{for } i=1,..,k \tag{1}$$

$$S^i_0 = \overline{S}^i .$$

The nutrient load to each sea basin, $B^i_t$, consists of discharges from its own catchment, $R^i_t = \sum_g D^{ig}_t$ , plus transports from other sea basins. The discharge from each country *g* is a result of nutrient load from and abatement at different emission sources, such as agriculture, sewage treatment plants, and industry. The total load of a nutrient without abatement, or the business as usual (BAU) loads, from each country *g* to a sea basin *i* at a specific time, $I^{ig}_t$ is then the net of BAU emission and abatement, $A^{ig}_t$. This is written as

$$D_t^{ig} = I_t^{ig} - A_t^{ig} \qquad (2)$$

With respect to transports of nutrients from other marine basins, we follow Gren et al. (2013) and assume that nutrient transports among basins can be described by a coefficient matrix. Each coefficient $a^{ij} = R^{ij} / R^i$ in the matrix shows the transports in nutrient loads from basin *i* to basin $j$, $R^{ij}$, as a share of total loads in basin *i*, $R^i$ These coefficients are time invariant and calculated as nutrient loads among marine basins without any abatement. Total nutrient load to a sea basin *i*, $B_t^i$, is then written as

$$B_t^i = \sum_j a^{ji} R_t^j \qquad (3)$$

Given a certain year, *T*, when the nutrient load targets are supposed to be achieved, we can express the nutrient targets by solving the integral (2) for *T* and define maximum nutrient pools, $K_T^i$, for each marine basin according to

$$S_T^i \leq K_T^i \qquad \text{for } i=1,..,k, \qquad (4)$$

where $S_T^i = \sum_\tau (1-\alpha^i(M_t^i))^{T-t+1}(B_t^i + S_0^i)$

where $S_0^i$ is the initial pool of a nutrient in a basin *i*. We can see from (4) that the impact of a given nutrient load, $B_t^i$, and inherited nutrient pool, $S_0^i$, has a smaller impact on the stock in the target year for a high than a low $\alpha^i$. The decay rate can be regarded as a self-cleaning capacity of the sea, and when this is relatively large, there is less need for costly abatement reducing the load to the basin. Thus, if $\partial\alpha^i / \partial M^i > 0$ an increase in *Marenezelleria* creates values from the increase in the self-cleaning capacity which reduces the need for costly abatement. On the other hand, when $\partial\alpha^i / \partial M^i < 0$ the opposite occurs and an increase in *Marenzelleria* instead creates costs from the need of more costly abatement in order to reach the nutrient constraint in the target year.

Each country faces an abatement cost function for reducing nutrient loads to a certain sea basin which shows the minimum cost for achieving different abatement levels, $C^{ig}(A_t^{ig})$. We thus assume that each country implements cost effective allocation of abatement among different measures directed towards agriculture, industry, sewage treatment plants and airborne emissions in each drainage basin. We impose the requirements that the cost function is increasing and convex in its arguments. Further, restrictions are assigned to the abatement capacity in each country and period of time, $A_t^g \leq \overline{A}^g$. They are imposed to avoid drastic structural changes in the sectors, the analysis of which would require a general equilibrium framework.

The decision problem is then formulated as the choice of abatement in the different countries, $A_t^{ig}$, which minimises total cost for reaching the nutrient stock restrictions in time *T*, according to

$$Min \qquad \sum_t^T \sum_i \sum_g C^{ig}(A_t^{ig})\rho_t \qquad (5)$$

s.t. (1)-(4)

where $\rho_t = \frac{1}{(1+r)^t}$ is the discount factor with *r* as the discount rate.

### 2.3 Determinants of costs of *Marenzelleria*

The determinants of the cost of *Marenzelleria* are derived from the first-order conditions of a cost effective solution to (5). These are obtained by constructing the Lagrange expression, which is written as

$$L = \sum_t \sum_i \sum_g \sum_f \rho_t C^{ig}(A_t^{ig}) + \lambda_T^i (K_T^i - S_T^i) + \gamma_t^g (\overline{A}^g - A_t^g) \qquad (6)$$

where $\lambda_T^i$ are the Lagrange multipliers of the restrictions on nutrient stocks in the marine basins and $\gamma_t^g$ are the multipliers on the abatement capacity constraints. The necessary conditions for minimising costs in eq. (6), subject to eqs. (1) to (5) are written as

$$\rho_t \frac{\partial C^{ig}}{\partial A_t^{ig}} = \sum_\tau \sum_j \lambda_T^i (1-\alpha^i(M^i))^{T-t+1} a^{ij} + \gamma_t^g \tag{7}$$

$$for \;\; t = 0,...,T-1, \;\; f = 1,..,c; \;\; g = 1,..,k; \; i = 1,..,n;$$

$$\lambda_T^i (K_T^i - S_T^i) = 0 \qquad for \qquad i = 1,..,n \tag{8}$$

$$\gamma_t^g (\overline{A}^{ig} - A_t^{ig}) = 0 \quad for \quad t = 0,...,T-1, \;\; f = 1,..,c; \;\; g = 1,..,k; \; i = 1,..,n \tag{9}$$

Equation (7) is of main interest for us. The conditions show that marginal abatement cost in present terms, in other words, the term at the left hand side of eq. (7) shall be equal to the sum of marginal impacts on the sea basins weighted by the Lagrange multipliers plus the marginal cost of the capacity constraint, second term at the right-hand side.

In order to show the impact of $M^i$ let us for the moment assume that there is a phosphorus target only for one sea basin. The cost-effective spatial allocation of the measures in each period is then determined where their marginal costs of achieving the targets are equal and correspond to the present value multiplier $\lambda_T^i$. For analytical convenience, we assume an interior solution where $\gamma_t^g = 0$, which does not affect the qualitative results. The first-order condition in this most simple case is then written as

$$\frac{\rho_t \partial C^{ig} / \partial A_t^{ig}}{\sum_\tau (1-\alpha^i(M^i))^{T-t+1} a^{ii}} = \lambda_T^i \tag{10}$$

The marginal costs of reducing nutrient load to the sea is determined by the marginal abatement cost in the numerator of eq. (10) and the marginal nutrient reduction in the target year as shown by the denominator. The higher the marginal nutrient reduction, the lower is the marginal cost of achieving the nutrient restriction in the target year.  We can then see that the marginal cost of nutrient reduction decreases when $\partial \alpha^i / \partial M^i > 0$ and increases when $\partial \alpha^i / \partial M^i < 0$. It can also be noted that the effect of an increase in $M^i$ in period *t* is reinforced for relatively long time period until the target time *T*. The decay rate then acts for a relatively long period.

Adding the possibilities of targets on phosphorus pools in several basins, the condition in eq. (7) can be written as

$$\frac{\rho_t \partial C^{ig} / \partial A_t^{ig}}{\sum_{\tau} (1-\alpha^{i}(M^{i}))^{T-t+1} a^{ii} + \sum_{j \neq i} \lambda_T^{j} \sum_{\tau} (1-\alpha^{j}(M^{j}))^{T-t+1} a^{ij})} = \lambda_T^{i} \tag{10’}$$

We can see from eq. (10’) that the effects of $M^i$ are reinforced by the dispersals of nutrients to other sea basins. These effects are larger for relatively high values of $\lambda_T^i$, which, in turn, depend on the stringency of the nutrient pool targets. Effects of occurrences of *Marenzelleria* in one marine basin can then be larger in other basins with more stringent nutrient targets. These effects depend on the openness of the marine basins, i.e. the levels of $a^{ij}$ for $j \neq i$ compared with $a^{ii}$.

The existence of *Marenzelleria* thus affects total cost of achieving the target by an impact on the marginal nutrient reduction cost in each period of time. The species can also affect total cost by influencing the timing of abatement. This is, in the most simple case with a nutrient target in one basin, written as

$$\left. \frac{\partial Ci^{g}}{\partial A_t^{ig}} \right/ \frac{\partial C^{ig}}{\partial A_{t-1}^{ig}} = \frac{1+r}{1-\alpha^{i}(M_t^{i})} \tag{11}$$

When $\alpha^i=0$, i.e. when there is no self-cleaning capacity in the sea basin, eq. (11) is similar to the Hotelling’s rule for optimal depletion of an exhaustible resource where the net return in present terms are equal for all periods. This means the abatement increases over time at a rate corresponding to the discount rate *r*. Equation (11) shows that this delay in abatement is reinforced when $\alpha^i>0$. A delay in abatement by one period not only reduces cost in present terms because of the discount rate but also because of the cleaning provided by the sea. When $\partial \alpha^{i} / \partial M_t^{i} > 0$ cost savings are made from increased self-cleaning capacity, and vice versa when $\partial \alpha^{i} / \partial M_t^{i} < 0$. This is also true in the case with binding targets for several basins, which is written as

$$\left. \frac{\partial C^{ig}}{\partial A_t^{ig}} \right/ \frac{\partial C^{ig}}{\partial A_{t-1}^{ig}} = \frac{(1+r)\sum_{j} \lambda_T^{j} a^{ij}}{\sum_{j} \lambda_T^{j} (1-\alpha^{j}(M_t^{i}))a^{ij}} \tag{12}$$

Similar to the effect in each time period as shown by (10’), the effects are reinforced over time by the consideration of all marine basins.

In sum, we can conclude that the occurrence of *Marenzelleria* creates cost(values) when it decreases(increases) the self-cleaning capacity, that is when $\partial\alpha^{iE} / \partial M_t^i < 0$ and $\partial\alpha^{iE} / \partial M_t^i > 0$ , respectively. The magnitude of these effects are determined by

- Timing of the introduction of *Marenzelleria*; an early introduction and a long period to the target year *T* implies higher cost(value) because of the longer period through which the effects of the species act.

- Stringency of the nutrient targets; abatement cost increases for large nutrient reductions which generate relatively large costs or values of *Marenzelleria* because of the decrease or increase in the self-cleaning capacity caused by the species.

- Openness of the affected marine basin; the effect of the species in a highly connected marine basin is relatively large because of the dispersal of nutrient loads to other basins.
- Discount rate; total abatement costs in present terms for achieving the nutrient targets are lower for a high discount rate because of the discounting of future abatement costs, which implies a lower cost or value of *Marenzelleria.*

## 3. Description of data

The numerical application of the model presented in Section 2 rests on two main studies; data on the impact of *Marenzelleria* on phosphorous cycling are obtained from biophysical modelling using GIS (Geographical Information Systems) in Nyström Sandman et al. (2016), and data on nutrient loads, pools and decay rates in each marine basin, and dispersal among basins from Gren et al. (2013). The spatial division of the Baltic Sea and its catchment is based on existing oceanographic models which divide the Baltic Sea into seven sea marine basins (e.g. Savchuck and Wulff 2007). These basins are delineated by differences in, among others, depth and currents (see Figure A1 in appendix). For the purpose of matching data on the costs of different abatement measures with nutrient transports into different sea basins, the entire

Baltic Sea drainage basin with its nine countries with coastal zones to the sea is divided into 24 drainage basins (see Figure A1 in the appendix).

### 3.1 Calculations of impacts on phosphorus fluxes of *Marenzelleria*

Data on the impacts of *Marenzelleria* on phosphorus fluxes are obtained from Nyström Sandman et al. (2016), where the calculations are made in two steps. First, the distribution and density of *Marenzelleria* in the Baltic Sea were sampled by collection of samples of the sea bottoms. Approximately 8960 samples were used, which contained information on biomass of *Marenzelleria* as measured by g/m$^2$and on salinity, sight depth, sea bottom topology and other environmental variables. A Random Forest model of *Marenzelleria* spp. biomass, based on relationships to these environmental variables was used to predict *Marenzelleria* biomass across the whole Baltic Sea at 200 m grid cell resolution (HELCOM, 2013b).

Second, the relation between the density of *Marenzelleria* and phosphorus fluxes between the sediment and water were estimated by combining the data on the species obtained in the first step with data from field measurements of phosphorous fluxes in intact sediment cores (Norling, 2007) and or with simulated data from Norkko et al. (2012). The change in phosphorous flux per year and grid cell was then calculated using GIS modelling. Known areas of permanent hypoxia were masked from the analysis as *Marenzelleria* spp. cannot live there. An important assumption was that effects on sediment fluxes in sandy sediments (sediment type according to EU Seamap) was half of that in muddy sediments. It was further assumed that the modelled density of Marenzelleria spp. was constant until the target year of 2081 (see under Section 3.2) as *Marenzelleria* spp. densities have been relatively stable for many years on a Baltic Sea level scale. The results from the measured and simulated data are presented in Table 1.

Table 1: Effects of *Marenzelleria* on phosphorus sequestration in bottoms of different marine basins in tonnes. Increase in flow from sediment to water per year.

| **Marine basin** | ***Marenzelleria*, tonnes:** | **Increase in phosphorous flux, tonnes:** |
|---|---|---|

|  |  |  |  | Field data |  |  | Simulated data |  |  |
|---|---|---|---|---|---|---|---|---|---|
|  | **Mean** | **Min** | **Max** | **Mean** | **Min** | **Max** | **Mean** | **Min** | **Max** |
| **Baltic Proper** | 21132 | 15668 | 26866 | 2204 | 1647 | 2793 | 1491 | 1114 | 1889 |
| **Gulf of Riga** | 5988 | 5256 | 6747 | 738 | 656 | 823 | 494 | 438 | 550 |
| **Gulf of Finland** | 15932 | 14466 | 17406 | 1636 | 1485 | 1788 | 1079 | 979 | 1180 |
| **Bothnian Sea** | 23691 | 19654 | 27738 | 2589 | 2138 | 3041 | 1744 | 1441 | 2046 |
| **Bothnian Bay** | 6608 | 4926 | 8307 | 560 | 407 | 714 | 379 | 276 | 483 |

Source: Nyström Sandman et al. (2016)

The estimated increases in phosphorus fluxes from the presence of *Marenzelleria* are lower when simulated data are used. In order to calculate effects on costs we compare the reductions in fluxes with the phosphorus sequestration unaffected by the species. This is based on the data on the initial phosphorus stocks and decay rates in the different basin (Table 2). The new decay rates, which are affected by *Marenzelleria,* are then calculated by subtracting the increase in fluxes shown in Table 1 from the initial sequestration in presented the third column in Table 1 and divided by the initial phosphorus stocks shown in the first column in Table 2. The results are presented in the two last columns in Table 2.

Table 2: Initial bioavailable phosphorus stocks, decay rates, and sequestration, and calculated effects of *Marenzelleria* on sequestration in different marine basins

|  | **Initial stock, $S_0^i$, kton.**[a] | **Initial decay rate , $\alpha^i$.** [a] | **Initial sequestration, kton**[b] | **Ranges of decay rate with effect of *Marenzelleria*, in initial period, $\alpha^i(M_0^i)$** [c] |  |
|---|---|---|---|---|---|
|  |  |  |  | **Field data** | **Simulated data** |
| **Bothnian Bay** | 7.4 | 0.324 | 2.398 | 0.228 – 0.248 | 0.259 – 0.288 |
| **Bothnian Sea** | 71.2 | 0.042 | 2.990 | -0.001 – 0.012 | 0.013-0.022 |
| **Baltic Proper** | 434.6 | 0.041 | 17.886 | 0.035 – 0.037 | 0.037 – 0.039 |
| **Gulf of Finland** | 25.9 | 0.236 | 6.112 | 0.167 – 0.179 | 0.193 – 0.198 |
| **Gulf of Riga** | 12.7 | 0.164 | 2.083 | 0.099 – 0.112 | 0.121 – 0.130 |

a. Gren et al. (2013) Table 1
b. Calculated by multiplying initial decay rates with initial stock in Table 2
c. calculated by subtracting impacts on fluxes in Table 1 from initial sequestration in Table 2 and dividing by initial stock of phosphorus in Table 2

The largest relative effect for both types of data is obtained for Bothnian Sea where the sediment can turn into a source of phosphorus. The smallest relative effect is calculated for Baltic Proper where the decay rate could be reduced with a maximum of 15%, i.e. from 0.041 to 0.035.

However, as will be shown in the subsequent sections, this decrease has a large impact on the costs for reaching the HELCOM (2013a) nutrient targets because of the stringent phosphorus target for the Baltic Proper.

### 3.2 Phosphorus transports and targets

Gren et al. (2013) provide the source of data on BAU phosphorus loads from different emission sources and drainage basins into marine basins and of transport coefficients between marine basins. The emission sources included in each drainage basin are agriculture, industry, and sewage treatment plants. The BAU loads to the coastal waters of the Baltic Sea are measured in biologically available fractions which affect eutrophication. Poland is the largest contributor of both nutrients, accounting for 26% of nitrogen and 39% of phosphorus loads (Table 3).

The maximum levels of nutrient stocks and time for the achievement, i.e. $K_T^{iE}$ in Section 2, are obtained from the most recent intergovernmental agreement on nutrient load and pool restrictions presented in the eutrophication segment of Baltic Sea Action Plan (BSAP) (HELCOM 2013a; Backer et al., 2009). This agreement applies the so-called ecosystem approach where targets are determined by the ecological conditions in the marine basins. The agreement thus sets conditions for the maximum nutrient pools in each marine basin. Phosphorus reductions are required for three of the basins; Baltic Proper, Gulf of Finland, and Gulf of Riga.

Table 3: BAU phosphorus loads from the riparian countries[a] to marine basins[b] in kton, and BSAP target reductions in % from initial phosphorus pools

| | **BB** | **BS** | **BP** | **GF** | **GR** | **DS** | **KT** | **Total** |
|---|---|---|---|---|---|---|---|---|
| **DK** | | | | | | 0.9 | 0.8 | 1.7 |
| **EE** | | | 0.1 | 0.9 | 0.2 | | | 1.2 |
| **FI** | 1.5 | 1.2 | | 0.5 | | | | 3.2 |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **DE** | | | 0.2 | | | 0.3 | | 0.5 |
| **LV** | | | 0.3 | | 1.8 | | | 2.1 |
| **LT** | | | 2.4 | | | | | 2.4 |
| **PL** | | | 13.0 | | | | | 13.0 |
| **RU** | | | 1.2 | 4.9 | | | | 6.1 |
| **SE** | 1.0 | 1.1 | 0.8 | | | 0.1 | 0.7 | 3.7 |
| **Total** | 2.4 | 2.3 | 17.8 | 6.4 | 2.1 | 1.3 | 1.5 | 33.8 |
| **Target, % reduction in initial pools** | N.A. | N.A. | 50 | 33 | 34 | N.A. | N.A. | |

a) DK: Denmark, EE: Estonia, FI: Finland, DE: Germany, LV: Latvia, LT: Lithuania, PL: Poland, RU: Russia, SE: Sweden.

b) BB: the Bothnian Bay, BS: the Bothnian Sea, GF: the Gulf of Finland, BP: the Baltic Proper, GR: the Gulf of Riga, DS: the Sound , KT: the Kattegat,

Sources: Gren et al. (2013) Tables 1 and A3

Baltic Proper receives the largest load, 53% of the total, and has to meet the most stringent target, with a requirement of 50% reduction in the initial phosphorus pool. The role of Polish abatement then becomes important since it discharges 73% of the total load into Baltic Proper. The basin is also relatively open, approximately 55% of its load originates from other basins (Gren et al. 2013, Table A5). Phosphorus reductions to other marine basins may then be needed in order to meet the target for the Baltic Proper. Similarly, since Baltic Proper receives relatively much of total phosphorus load from its own catchment which is transported to other marine basins, reductions can be needed to achieve targets in the Gulf of Finland and the Gulf of Riga.

With respect to the timing for achieving the target, Helcom BSAP suggests 2021 to be the deadline for implementation of nutrient load reductions. However, the suggestion contains no discussion on when the targets are supposed to be achieved. Our calculations with different target times show that at least 40 years are needed to achieve the nutrient targets shown in Table 3. Since BSAP is supposed to be implemented at the latest in 2021, the earliest target is 2061. To obtain flexibility in choice of abatement and to obtain feasible solutions in the sensitivity analyses, we add 20 years and carry out calculations for a target time of 2081.

### 3.3 Abatement costs and discount rate

Abatement cost functions for each country are obtained from Gren (2016), who estimated the functions by generating data on minimum costs for reaching different levels of phosphorus reductions from the BAU level. The numerical model calculating cost effective solutions considers, for each country, 11 abatement measures which includes agriculture, industry, and sewage treatment plants. Most of the measures are directed towards agriculture since this is the main source of phosphorus loads. Included measures for this sector are reductions in use of fertiliser, livestock holdings (cattle, pig, and chicken), changed spreading time of manure from autumn to spring, cultivation of energy forest and catch crops, and construction of wetlands. In total, 500 observations are generated for each country with abatement cost as the dependent and phosphorus reduction as the independent variable. All costs are expressed in 2011 prices.

The final data component is the choice of discount rate. A large body of literature exists on the appropriate level of the social discount rate, which is determined by pure time preferences, growth in consumption opportunities, and utility of consumption (see e.g., Weitzman 2001). The level of the discount rate may differ between the riparian countries due to differences in consumption preferences and economic growth. It is also generally recommended to use a hyperbolic discount rate, in other words, a time-declining rate, for long-term projects exceeding 50 years. A simplification is made in this study by assigning a uniform discount rate for all countries and periods. Following the recommendations made for discounting future streams of net benefits, calculations of cost- effective solutions are made for a relatively low level, 0.015, of the discount rate (e.g., Newell and Pizer 2003).

## 4. Results

The numerical dynamic model is solved by using the GAMS Conopt2 solver (Brooke et al., 1998), which gives the optimal allocation of abatement and associated cost in different drainage basins and time periods. The calculated minimum costs reaching the BSAP targets without *Marenzelleria* amounts to 1025 billion of SEK (1 Euro = SEK 9.03 in average for 2011). This implies an annual average discounted cost of 17.1 billion SEK, which comes close to that obtained by Gren et al. (2013). Total gross domestic product (GDP) of the countries in the Baltic Sea catchment amounted to approximately 13700 billion SEK in 2011 (Gren and Säll, 2015 Table 1), which implies that the total abatement cost in the reference case corresponds to approximately 0.12% of GDP.

Minimum costs with *Marenzelleria* are calculated under the assumption that the species enter the marine basins as specified in Table 1 in time period t=0. Calculations are made for two assumptions of the occurrences of the species; in only one of the basins, or in all sea basins.

However, it turns out that the abatement costs are affected only when the species is in the Baltic Proper. The cost of the species then varies considerably depending on its effect on the decay rate (Table 4).

Table 4: Calculated costs in billion SEK of *Marenzelleria* at different average and range of phosphorus retention rates from the estimates based on field or simulated data.

| | **Total abatement cost:** | | **Cost of *Marenzelleria:*** | |
|---|---|---|---|---|
| | **Average** | **Range** | **Average** | **Range** |
| **Reference** | 1025 | | | |
| **Field data** | 1502 | 1391 - 1756 | 477 | 366 - 731 |
| **Simulated data** | 1287 | 1193 - 1391 | 262 | 168 - 366 |

The occurrence of *Marenzelleria* increases total abatement cost at the most by 731 billion SEK or 73%, which occurs at the minimum level of sequestration in the Baltic Proper. The cost of *Marenzelleria* then corresponds to approximately 0.08% of total GDP in the catchment. This cost decreases drastically to 168 billion SEK at the smallest impact of *Marenzelleria* when the decay rate is 0.039. At the average decay rate, the cost amounts to 477 and 262 billion SEK from the retention rates based on field and simulated data, respectively.

The reason for the differences in the costs of the species is the need of more and earlier abatement as compared with the reference case when there is no *Marenzelleria*, which is shown for the minimum and maximum estimates of the decay rates in Figure 2.

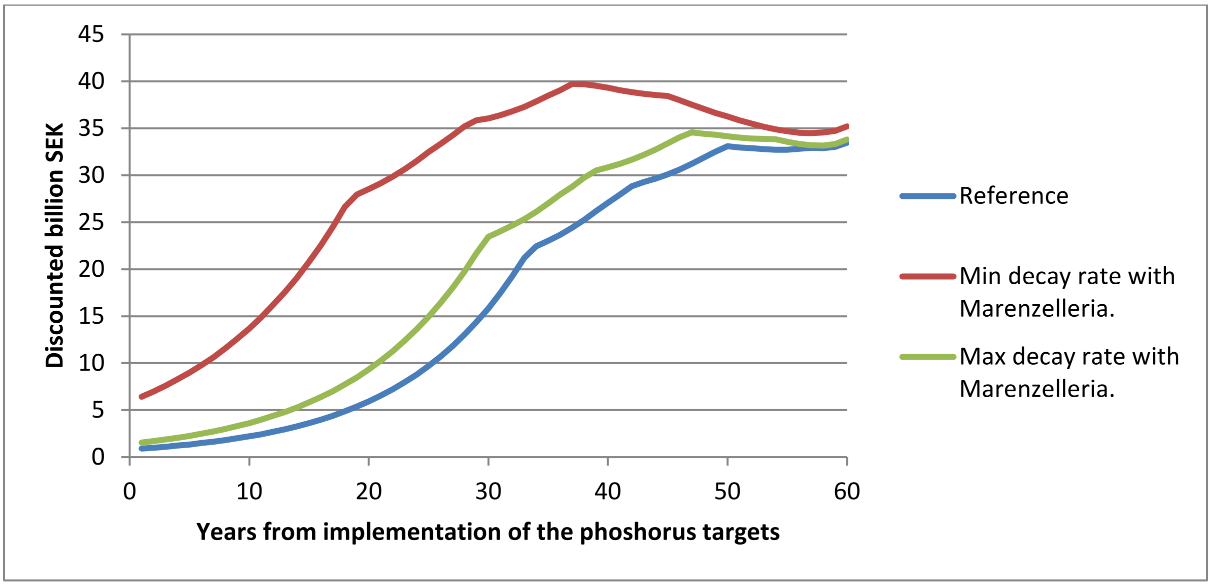

Figure 2: Cost effective time paths of annual total abatement cost for achieving the BSAP phosphorus targets listed in Table 4 with (min and max decay rates) and without (reference case) *Marenzelleria*.

The annual abatement cost reaches a peak in year 50 in the reference case, but is 5 billion larger and 13 years earlier for the minimum decay rate with *Marenezelleria*. The larger costs are created by the reduction in the decay rate, and the need for earlier abatement in order to reach the given phosphorus pool target in 60 years. The cost effective time path of the discounted cost of *Marenzelleria* is found by taking the difference in abatement costs between the reference case and the cases with *Marenzelleria*. The maximum annual cost of *Marenzelleria* occurs at *t=25* and amounts to 27 billion SEK

However, the cost of *Marenzelleria*, at both the minimum and maximum decay rates, is unevenly distributed among the countries. The increased costs created by the need for more abatement are born by countries with relatively large abatement capacity and low costs. Poland is the largest emitter of phosphorus to the Baltic Proper (Table 3), and faces the largest share of the costs of the invasive species (Figure 3).

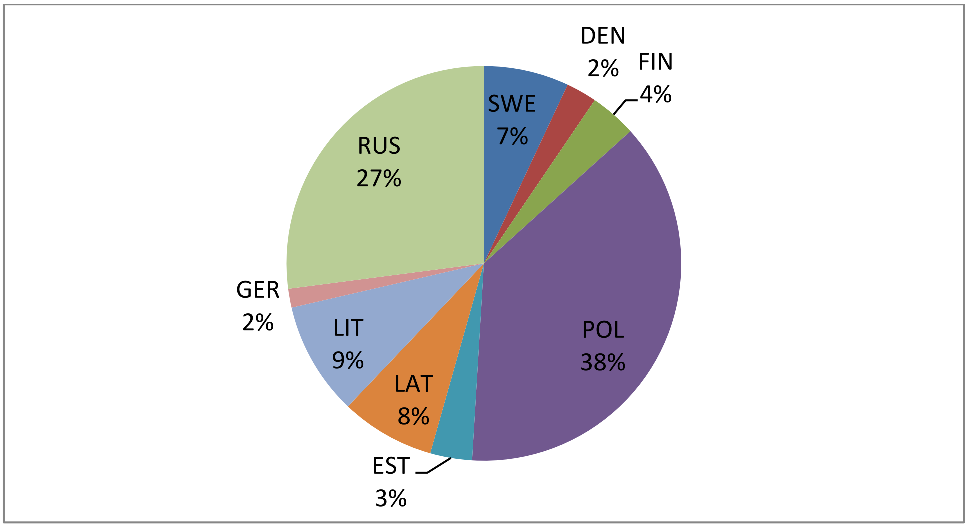

Figure 3 Allocation of costs of *Marenzelleria* among riparian countries at minimum phosphorus sequestration in Baltic Proper, % of total cost

If we instead of shares of total cost of *Marenzelleria* investigate relative increase in abatement costs for different countries caused by the species, we can notice that it is highest for Germany (Figure 4). The abatement cost can increase by twofold for this country. On the other hand, the increase occurs from a low level because of the relatively high abatement costs for this country.

The requirement of more abatement necessitates abatement also from relatively high cost countries.

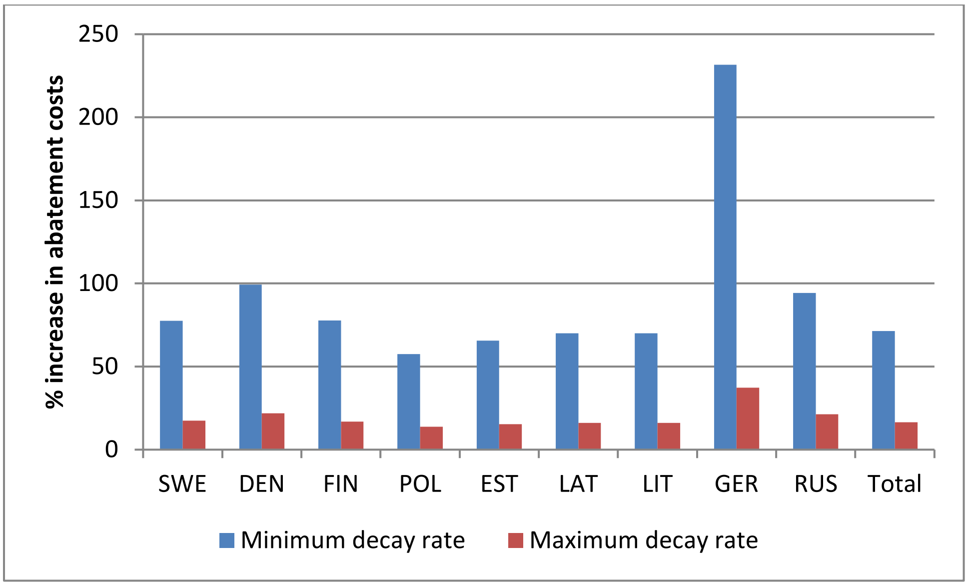

Figure 4: Costs for different riparian countries of *Marenzelleria* at the minimum and maximum phosphorus sequestration rate in Baltic Proper, % increase from abatement costs without *Marenzelleria*

As reported in the theoretical Section 2, the cost of *Marenzelleria* depends on several exogenous parameter values in our dynamic model. These include the timing of entrance of the species into the Baltic Proper, choice of discount rate, stringency of nutrient targets, and abatement costs. With respect to the entrance of the species, calculations have been made based on the current existence of the species in the basins. In order to illustrate the effects of timing of the species' appearance we calculate cost for the hypothetical case when it instead enters at $t=10$. Our choice of discount rate in the reference case is relatively low, and we therefore calculate costs of the species when it increases from 0.015 to 0.02. Without technological development, abatement costs are likely to increase in the future because of economic growth in the countries which raises the costs of several abatement measures. We therefore calculate costs of the invasive species for an arbitrary increase in abatement costs by 25%. Changes in the nutrient targets may be caused by climate change and/or by unexpected responses in the sea to changes in nutrient loads. Since we have no priors on the magnitude of such target alterations we calculate costs for an arbitrary tightening of the target by 10%, i.e. the phosphorus pool should be 10% lower than the targets set in Table 3.

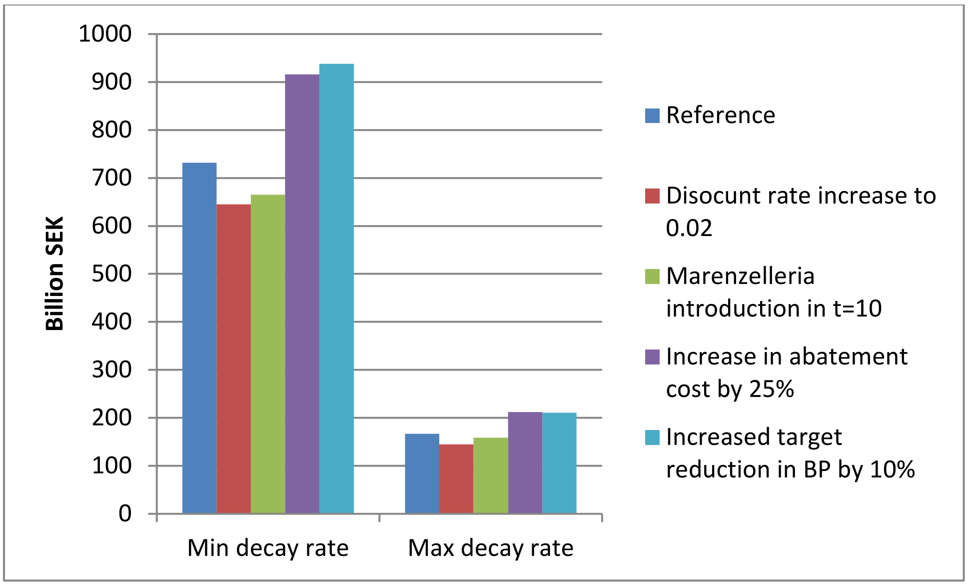

Figure 5: Calculated costs of *Marenezelleria i*n the reference case, and for different changes in exogenous parameter values.

The results in Figure 4 show, as expected, that costs of *Marenzelleria* decrease when discount rate increases and when the species enters at a later point in time. The cost increases when abatement cost and stringency of the target increase. The calculated cost of the species then varies between 145 billion SEK and 938 billion SEK depending on its effect on the decay rate and assumption of the values on the exogenous parameters.

## 5. Discussion and conclusions

The main purpose of this paper was to calculate the cost of the invasive species *Marenzelleria* in the Baltic Sea. This cost arises from the impact of the species on the phosphorus sequestration in the seafloor sediment. A change affects the dynamics of the phosphorus cycling, which, in turn, impacts the need for costly phosphorus load reductions in order to achieve predetermined targets. The costs were therefore calculated as the change in total abatement costs for achieving specific maximum pools of phosphorus in the Baltic Sea, which differs among its sea basins. To this end, we constructed a numerical dynamic optimisation model that accounts for interlinked sea basins that differ with respect to nutrient dynamics and eutrophication status.

The analytical results showed that the cost of *Marenzelleria* depends on its effect on the decay rate of phosphorus pools in the sea basins, stringency of the phosphorus targets, connectivity among sea basins, discount rate, and costs of abatement measures. Our empirical results for the current spread of *Marenzelleria* in different sea basins showed that the effect on the decay rate differs among and within sea basins. It turned out that *Marenzelleria* creates costs in only one sea basin, Baltic Proper. The reason is the relatively low decay rate, large phosphorus pool, and largest reduction need in phosphorus as set by HELCOM (2013a). The calculated cost ranges between 168 billion SEK and 731 billion SEK, which corresponds to a maximum of approximately 0.08% of total GDP in the Baltic Sea catchment. However, the results rest on assumed levels of several exogenous parameters, such as the discount rate and target stringency. Sensitivity analysis indicated a large range, from 145 billion SEK to 932 billion SEK. Common to all estimates is the uneven distribution of the costs among the riparian countries. The largest share, 37%, is born by Poland, which accounts for 73% of the phosphorus loads to the Baltic Proper and has access to low cost options.

Another important assumption is cost effective achievement of the phosphorus targets. Its implementation requires a system with differentiated charges or, under a permit market system, trading ratios among drainage basins and time periods. Such differentiated systems can be associated with high transaction costs due to the need for monitoring (e.g., Shortle and Horan 2001). The cost of *Marenzelleria* is then higher than our estimates when the transaction costs are increasing in the required nutrient reduction levels.

A factor not included in this paper is fair cost burden among countries, which is likely to be highly important for implementing any international agreement (e.g., Carraro 2000; Bérubé and Cusson 2002). The results in all scenarios in this paper showed that one country, Poland, bears the largest cost burden of *Marenzelleria*. This is explained by the size of Poland with respect to load of nutrients, the stringent target for the Baltic Proper, and the slow dynamics of this basin. As demonstrated by Gren and Destouni (2012), in a static context this implies an unfair allocation of costs when using fairness criteria such as costs per capita or costs related to the total economic outputs. On the other hand, the entrance of *Marenzelleria* increases abatement costs relatively more for wealthier countries as measured by GDP/capita. Despite the increase in overall abatement costs of the existence of the species it might have an advantage in mitigating unfair allocation of the abatement costs.

## Acknowledgements

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## Appendix: figure

*Figure A1: Drainage basins of the Baltic Sea. (Drainage basins in Denmark (2), Germany (2), Latvia (2), and Estonia (3) are not provided with name, but are delineated only by fine lines*

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