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Estimated trophic state effects and abatement costs in connection with improved urban sewage treatment in the Gulf of Riga

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Abstract
Environmental conflicts of interest are important to account for when environmental policies are designed. This paper explores the quantitative connection between urban waste water treatment, coastal eutrophication, and fish biomass in the mesotrophic Gulf of Riga (northern Europe). The probable effect on the water quality from one clearly defined abatement measure, improved urban sewage treatment has been studied. Furthermore, the implementation cost and the likely effect on total fish biomass have also been assessed. Computer simulations using the previously published model CoastMab suggested that good water quality according to the EU Marine Strategy Framework Directive could be achieved if urban sewage treatment would be upgraded to Nordic and German standards, and not only around the Gulf of Riga but in the whole Baltic Sea drainage basin. The Secchi depth would double according to these simulations while total phosphorus and summer chlorophyll concentrations would decrease by 54% and 53%, respectively. The total fish biomass should be expected to decrease by about 42% if "good" water quality (as defined in European Union directives) should be achieved. However, changes in total fish biomass could also be offset by changes in other important determinants such as climate related variables or fishing pressure. The study estimated that it could take about 20-40 years after abatement action for the trophic state in the Gulf to stabilise again. Upgrading urban sewage treatment to this extent would cost 468-1,118 million euros per year. Treatment could have substantial positive effects on the water quality of the Gulf but could also have adverse side effects on the total fish biomass.

Keywords: eutrophication; Gulf of Riga; sewage treatment; water quality; fish biomass

Introduction

Urban waste water treatment, coastal eutrophication and fisheries have been subject to much environmental regulation in the European Union and elsewhere. Related to these issues are conflicts of interest between environmental goals such as low levels of eutrophication and thriving fisheries (Hansson et al. 2007). When environmental regulation is introduced or revised, it is important to assess expected beneficial environmental effects from each action alternative and to weigh such effects against all probable adverse side effects.

Eutrophication, manifested as decreased Secchi depth (water clarity) and intensified phytoplankton blooms, has been considered a serious environmental problem in the Gulf of Riga (northern Europe; see fig. 1) for many decades. The mesotrophic Gulf is one of the most nutrient polluted sub-basins of the Baltic Sea (Håkanson and Bryhn 2008a; Kotta et al. 2008). Urban sewage treatment is the most effective, and also a comparatively cost-effective way to decrease the total phosphorus (TP) loading to the Gulf of Riga (Bryhn, 2009). The average waterborne TP loading to the Gulf during 1997-2003 was 2,180 tonnes per year while the TP loading to the Baltic Sea (excluding the Kattegat and the Danish Straits) was 33,328 tonnes per year according to HELCOM (2007). Table 1 lists expected TP reductions if the EU Urban Waste Water Treatment Directive (UWWTD) would be implemented in the drainage area of the Baltic Sea, including the drainage area of the Gulf of Riga (from HELCOM 2007). More than one third of the TP loading to the Baltic Sea could thus be reduced by implementing this abatement measure, which would also include that about 54% of the TP loading from the Gulf of Riga catchment would be cut, while the TP loading to the Baltic Proper and the Gulf of Finland would decrease by approximately 42% and 47%, respectively.

Nutrient concentrations and phytoplankton biomass are also connected to fish productivity. Fish stock size depends on the food availability to the fish. Primary producers may be consumed directly by fish or by various secondary producers which fish in turn prey on. Fish production appears to be proportional to primary production in coastal waters as well as in lakes, estuaries and oceans (Nixon 1982; Iversen 1990;

Houde and Rutherford 1993; Ware and Thomson 2005; Jennings and Brander 2010; Kaiser et al. 2011).

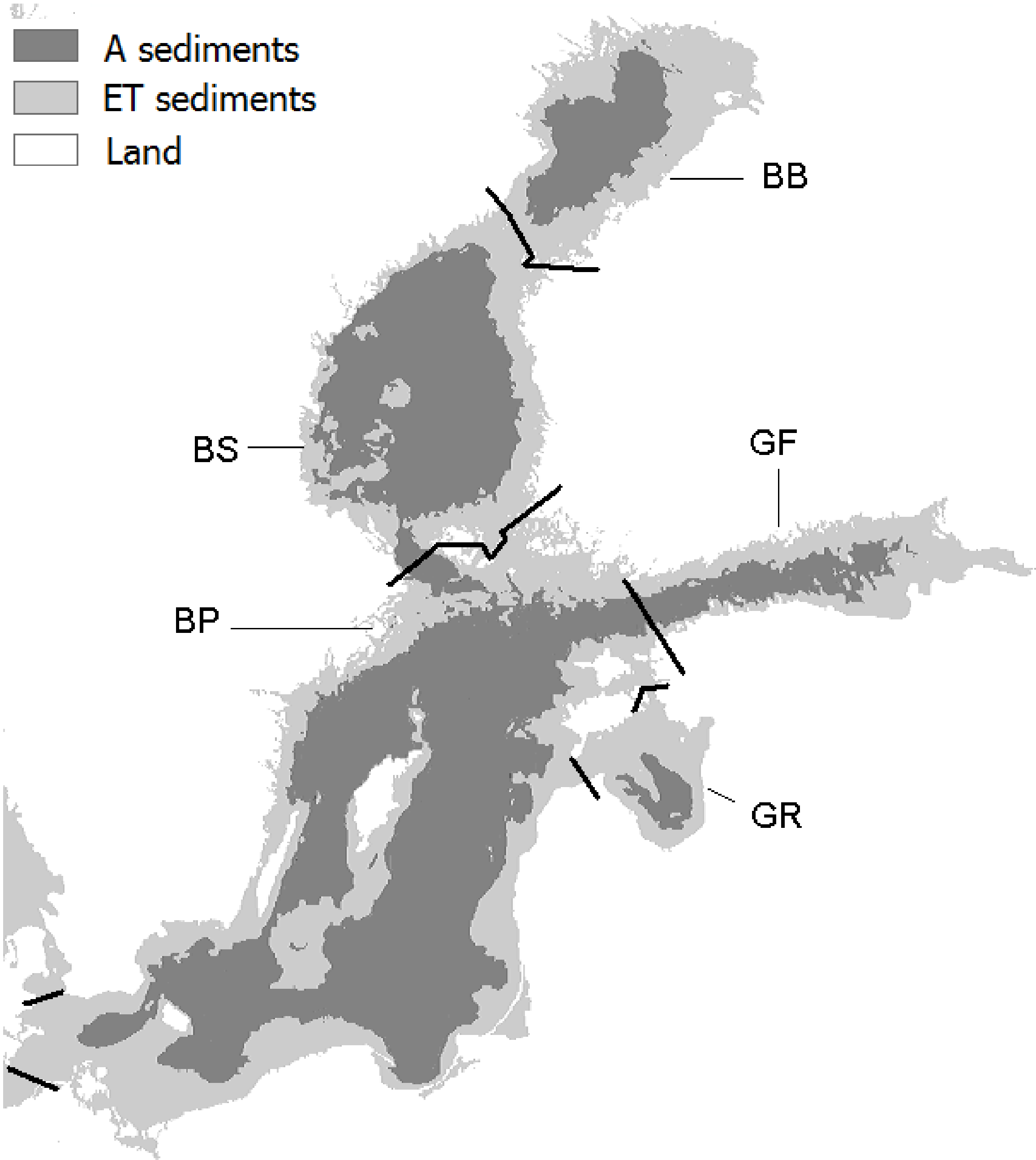

Figure 1. Location of study area. The Baltic Sea and its subbasins Bothnian Bay (BB), Bothnian Sea (BS), Baltic Proper (BP), Gulf of Finland (GF), and Gulf of Riga (GR). Subbasin limits are marked by black lines. Accumulation (A) sediment areas are distinguished from erosion and transport (ET) sediment areas. Modified from Bryhn and Håkanson (2010).

Although good oxygen conditions and low levels of organic matter decomposition by bacteria may promote fish survival in the young stages, high food availability appears to be more important for subsequent stages and for the cumulative (net) effect of primary production on the total fish stock size (Hansson et al. 2007).

Table 1. Expected TP loading reductions (tonnes/year) from improved sewage treatment compared to 2004. The final column lists how much of the reductions to the Baltic Sea could be performed in the Gulf of Riga drainage basin. The average annual TP loading 1997-2003 was 2,180 tonnes to the Gulf of Riga and 33,328 tonnes to the Baltic Sea including the Gulf of Riga but excluding the Danish Straits. Data from HELCOM (2007).

Country	**Baltic Sea drainage basin**	**Gulf of Riga drainage basin**	**% of Baltic Sea reductions in the Gulf of Riga drainage basin**
Belarus	1,977	523	26
Czech Republic	391	0	0
Estonia	133	17	13
Lithuania	615	46	7.5
Latvia	187	162	87
Russia	3,829	431	11
Poland	5,292	0	0
Total	12,424	1,179	9.5

General cross-systems relationships between primary production and fish yield are displayed in fig. 2 using data from two different studies (Håkanson and Boulion 2002; Chassot et al. 2007). The relationship in fig. 2A suggests that if primary production is reduced by 10%, the fish yield can be expected to decline by about 9%. Similarly, the regression in fig. 2B indicates that a 10% reduction in primary production of the Baltic Sea should result in a 15% lower fish yield. Patterns similar to those depicted in fig. 2 have been documented by Nixon (1982), Iversen (1990), Ware and Thomson (2005) and Jennings and Brander (2010).

It should, however, be stressed that there are other crucial determinants of total fish biomass which may dampen or counteract effects from trophic state changes on changes in biomass. Some of these determinants are fishing pressure (Möllmann et al. 2009; Jennings and Brander 2010), predation by seals or other marine mammals (Thurow, 1997), and climatic factors such as variations in water inflow from more saline seas (Möllmann et al. 2009), wind speed and ice cover (Kotta et al. 2009). Additional factors related to eutrophication which may influence the relationship between primary production and fish yield are fish and zoobenthos kills due to deepwater hypoxia, and changes in macrophyte cover. Shallow waters with high macrophyte density may serve as crucial and vital providers of food and shelter for fish (Sandström and Karås 2002).

Nevertheless, there has been a sharp long-term increase in cod, herring and total fish production in the Baltic Sea during the last century which has partly been attributed to the cumulative effects of eutrophication (Thurow 1997; Ojaveer and Lehtonen 2001).

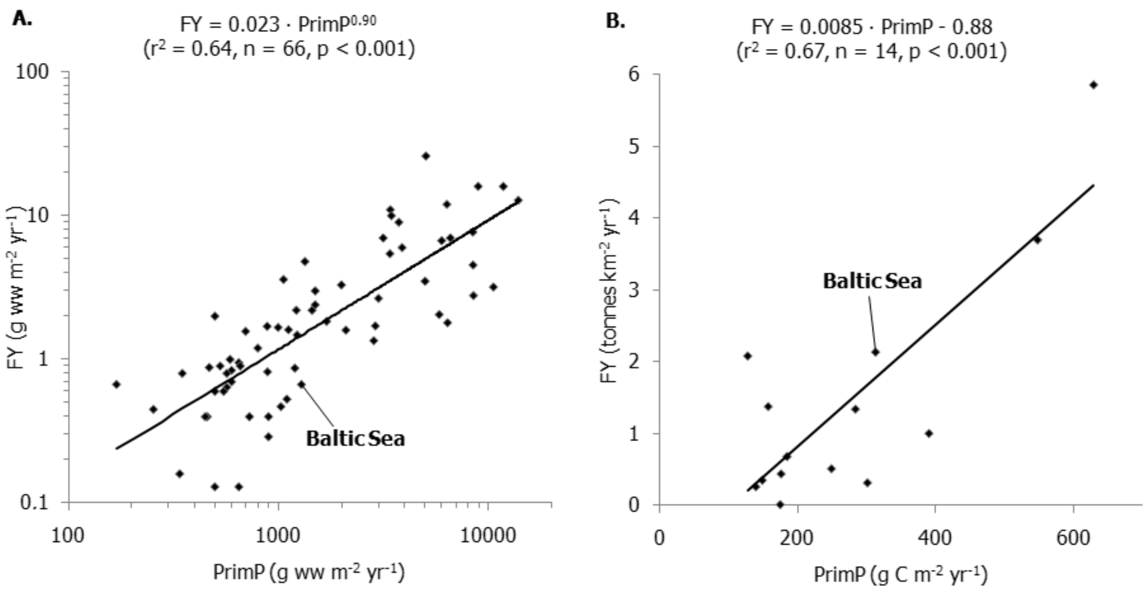

Figure 2. Two cross-systems statistical models of primary production (PrimP) and fish yield (FY). A. Freshwater and marine ecosystems in various parts of the world, from Håkanson and Boulion (2002). B. European marine ecosystems, from Chassot et al. (2007). Data pairs regarding the Baltic Sea have been marked.The difference in units and scale projections should be noted.

Previous studies have shown that good water quality in the Gulf of Riga could be achieved by means of substantially decreased TP loadings (HELCOM 2007; Håkanson 2009). However, the present study concerns three new aspects related to this issue; (1) what will the effect be on the trophic state in the Gulf from improved urban sewage treatment, (2) how will the total fish biomass be affected, and (3) what is the abatement cost. The dynamic mass-balance model CoastMab (Håkanson et al. 2010) will be used for investigating effects from implementing the UWWTD in (a) the Gulf of Riga catchment only and (b) the whole Baltic Sea catchment, on the trophic state in the Gulf of Riga. The idea of using two scenarios is to examine whether one or both of these management options would be enough to achieve “good” water quality according to the EU Marine Strategy Framework Directive (Anon, 2008). In addition, effects on preconditions for fishery in the Gulf will be studied and discussed in relation to changes in trophic status. Finally, a separate cost estimate for implementing the UWWTD in the Gulf of Riga catchment and for achieving good water quality will be made and will be compared to cost estimates made in connection with the Baltic Sea Action Plan, which was signed in 2007 by the Ministers of Environment of the Baltic Sea States (HELCOM 2007).

Background

The Gulf of Riga (58°N, 23°30'E; Fig. 1) has a surface area of 16 700 km^2, a mean depth of 24.5 m and a maximum depth of 56 m (Håkanson and Bryhn 2008a). The water of the Gulf is brackish as the salinity in most parts ranges from 5.5 to 6.0 psu (Kotta et al. 2008). The Gulf of Riga is bordered by the Estonian and Latvian mainlands, the two islands of Muhu and Saaremaa, the Irbe Sound and the Suur Strait. The drainage basin covers 137,200 km^2 (Laznik et al. 1999) and includes parts of Estonia, Latvia, Lithuania, Belarus and Russia. The largest river in this area is River Daugava with a drainage basin of 87,900 km^2 (Laznik et al. 1999). River Daugava originates in Russia, flows through northern Belarus, drains parts of those countries as well as parts of Latvia and Lithuania and joins the Gulf at the Latvian coast. Water, nutrients and other substances are also transported to the Gulf via several smaller rivers and streams such as River Lielupe (drainage basin located in Latvia/Lithuania), River Gauja/Koiva (Latvia/Estonia), River Pärnu (Estonia/Latvia), and River Salaca/Salatsi (Latvia/Estonia).

The Gulf is highly exposed to fluxes from the adjacent Baltic Proper, as fig. 1 suggests. The gross water flux to the Gulf from the Baltic Proper is 3-4 times greater than the discharge from the catchment. Although the Gulf is a net exporter of both water and TP to the Baltic Proper, the gross TP flux from the Baltic Proper to the Gulf of Riga is about 4,300 tonnes per year (Savchuk 2005; Håkanson and Bryhn 2008a), and is thus considerably larger than the TP loading from the drainage basin.

The average Secchi depth in the Gulf has decreased from 5 m in the early 1960s to 3 m during the first years of the 2000s (Fleming-Lehtinen and Kaartokallio 2009). Regularly measured and reliable data on many other water quality indicators are, however, only available for more recent decades and changes have been fairly modest during this time. July-August mean concentrations of chlorophyll-a (a pigment used as a proxy for phytoplankton concentration or productivity) have fluctuated between 4 and 6.5 μg/l in surface waters during 1998-2007 (EC-JRC, 2008). Since the late 1980s, TP concentrations in surface waters have increased slightly from about 27 μg/l to approximately 31 μg/l (Bryhn 2010). Kotta et al. (2009) noted two TP loading peaks in the late 1980s and in the late 1990s but could not find any clear inter-annual loading trend. The TP concentration in the Baltic Proper has also remained fairly stable since the 1980s with concentrations fluctuating around 21 μg/l in the 1980s and around 22 μg/l during 2000-2007 (Bryhn 2010). The lack of water quality improvement in the Gulf is likely due to relatively insignificant changes in TP loadings from the catchment as well as from the Baltic Proper.

However, the TP input to the Baltic Sea as a whole has decreased by more than 40% since the 1980s although surface water TP concentrations have not decreased in most of the Baltic Sea subbasins (Savchuk and Wulff 2009; Bryhn 2010). A common explanation is variations in major saltwater intrusions from the North Sea. After a major saltwater intrusion, redox conditions in deep waters and sediments of the Baltic Sea change due to the influence of the added oxygen-rich water which has a high salinity and density compared to surface waters and is therefore mainly transported downwards (Matthäus 2006). Redox conditions affect phosphate diffusion from deep sediments (Conley et al. 2002). Intensive saltwater intrusions have occurred at a historically low frequency since the 1980s (Matthäus 2006), after which a larger fraction of the settling particulate phosphorus (P) in the water appears to have returned as internal phosphate loading from deep sediments (Neumann and Schernewski 2008; Savchuk and Wulff 2009). Conley et al. (2002) correlated oxygen and phosphate concentrations in hypoxic waters of the Baltic Sea 1970-2000, noting more than 50% higher phosphate concentrations during years with low oxygen concentrations compared to years with less severe deepwater hypoxia. Another common partial explanation to the lack of substantial TP concentration changes is that the Baltic Sea reacts slowly to changes in nutrient input (Savchuk and Wulff 2007). The TP content in Baltic Sea water is about 550,000 tons (Savchuk 2005). The TP content in shallow sediments exposed to wind and wave action has been estimated at 200,000 tonnes while the P which may be released from deep sediments unaccessible for waves may be 900,000 tonnes (Håkanson and Bryhn 2008). Altogether, waters, bioactive deep sediments and wave-exposed shallow sediments contain almost 50 times more P than what is added to the Baltic Sea annually from its drainage basin.

Primary production promotes food availability for fish and thereby also affects fish production (fig. 2). The Baltic herring (*Clupea harengus membras)* is the most abundant amongst more than 50 other species in the Gulf of Riga and herring also accounts for about 90% of the commercial catch value. The herring stock is quite stationary in the Gulf (Kotta et al. 2008) and fig. 3 shows the annual variations in landings and spawning stock biomass of herring in the Gulf of Riga 1977-2007. Since the early 1990s, the spawning stock biomass has been quite stable. One can, however, note a strong increase in spawning stock biomass between the mid-1980s and the early 1990s, a change which concurred with increasing nutrient concentrations and primary production in this area (Kotta et al. 2008), and also with the end of a temporary extreme peak in prevalence of the migratory Baltic cod (Ojaveer et al. 1999; Eero et al. 2008). However, since trophic state changes during this period were very limited, the impact from these changes on the herring stock has been difficult to distinguish. Kotta et al. (2009) found no eutrophication effects on the herring stock in the Gulf during 1977-2006 but instead correlated herring stock changes to changes in climate variables. Regarding the development before this period, Thurow (1997) estimated that the herring biomass in the Baltic Sea had increased by a factor of 15-16 from the 1920s until the mid-1970s, although estimates for the Gulf of Riga were not specified.

Urban sewage discharge in the EU is regulated by the UWWTD (Anon 1991). The Directive requires that waste water be collected and subjected to secondary treatment in all agglomerations with more than 2,000 population equivalents. Agglomerations with more than 10,000 population equivalents and which drain into sensitive areas should have more advanced urban sewage treatment. Exceptions to these general rules are described in Anon (1991). Table 1 shows that substantial TP reductions to the Gulf of Riga could be made if the UWWTD would be implemented throughout the drainage basin, although it is important to note that a considerable part of the reductions would have to be made in the non-EU states Russia and Belarus.

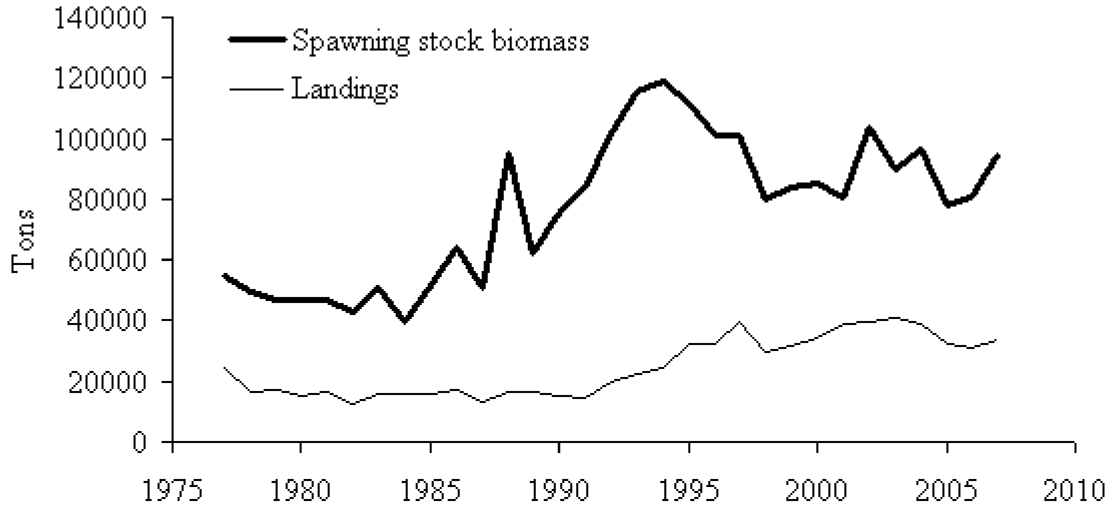

Figure 3. Spawning stock biomass and landings of herring in the Gulf of Riga, 1977-2007. Data from ICES (2008).

The Marine Strategy Framework Directive (MSFD; Anon 2008), requires EU member states to take necessary measures to achieve (or maintain) "good" or "high" water quality in marine waters. Marine waters are defined in the MSFD as waters, including their seabed and subsoil, from the coastline "to the outmost reach of the area where a Member State has and/or exercises jurisdictional rights". Good water quality should be achieved by the year 2020, albeit some waters may be exempted if this goal is out of reach. The MSFD applies to the whole Gulf of Riga which consists of Estonian and Latvian territorial waters. Good and high water quality were defined in Anon (2000) and the definitions are provided in table 2.

Table 2. Ecological status classification according to the EU Water Framework Directive (Anon 2000).

Ecological status	**Definition**
High	No, or only very minor, anthropogenic alterations
Good	Low distortion levels resulting from human activity, which deviate only slightly from levels associated with "high" status
Moderate	Conditions deviate moderately from those at "high" status and are significantly worse than those at "good" status
Poor	Major anthropogenic alterations compared to "high" status
Bad	Severe anthropogenic impacts under which large portions of the relevant biological communities have disappeared

Methods

The limiting nutrient in primary production of the Gulf of Riga is P (Kotta et al. 2008), so a delicate task in predicting effects on the trophic state from abatement action is to accurately assess external and internal P fluxes to, from and within the Gulf. The Baltic Sea Action Plan requires P abatement to the Gulf of Riga but no nitrogen abatement (HELCOM 2007). If P concentrations would decrease gradually in the surface waters of the Gulf, a gradually aggravated P deficiency could be expected for the phytoplankton communities which would thereby be forced to decrease their activity, reproduction and biomass (Tyrrell 1999; Håkanson and Bryhn 2008a; Schindler et al. 2008).

Mass-balance models are the only means by which changes in trophic state and internal P fluxes can be predicted from changes in P loading and in other external factors (Håkanson et al. 2010). The model used in this study is called CoastMab and consists of mass-balance submodels of salt, TP and suspended particulate matter (SPM). Masses, concentrations and fluxes are simulated at monthly time steps using Euler's method and ordinary differential equations. There are also other sub-models connected to these mass-balance submodels; e. g., a Secchi depth submodel, a chlorophyll submodel and also a foodweb model framework (CoastWeb) with predator-prey interactions (Håkanson et al. 2010). For simplicity and because no foodweb interactions will be simulated, total fish biomass changes will not be estimated by means of foodweb modelling in this study but will instead be predicted from TP concentrations and salinities. Thus, the CoastMab model (published in Håkanson et al. 2010) will be used for predicting i) TP concentrations, ii) chlorophyll-a concentrations (Chl; in µg/l), iii) the Secchi depth (in m) and iv) total fish biomass. Predictions of present levels of TP, Chl and Secchi depth will be compared to empirical measurements described in Håkanson (2009).

A well-known alternative to CoastWeb and the approach described above is the Ecopath with Ecosim (EwE) framework (Christensen and Walters 2004; Hansson et al. 2007). EwE allows construction and combination of modelling blocks where the user provides constants and other information about the modelled group of organisms. However, EwE does not include a mass balance model related to eutrophication and nutrient inputs, which makes EwE less well suited for the present study. An alternative to modelling average subbasin conditions related to the trophic state is to use 3D models which simulate concentrations and fluxes of nutrients on a small spatial and temporal scale (Håkanson et al. 2008a; Neumann and Schernewski 2008). Yet, modelling changes in average subbasin conditions on longer (monthly, annual and decadal) timescales should be sufficient for the purpose of the investigations in this paper and the CoastMab approach will therefore be selected.

In the CoastMab model, water masses and sediment areas are divided into categories according to location in relation to the halocline and to location in relation to the theoretical wave base (fig. 4). The depth of theoretical wave base (D_{WB}) is the depth at which wind and wave action normally reaches, stirs up sediments, and mixes water. Above this depth, erosion and transport (ET) sediments dominate bottom areas, while accumulation (A) sediments dominate areas below D_{WB}. Waters above D_{WB} are referred to as surface waters in CoastMab, while deep waters are defined as being located below D_{WB}. Two subbasins of the Baltic Sea (the Baltic Proper and the Gulf of Finland) also display a marked halocline at about 75 m, so waters and sediments below D_{WB} in these subbasins are divided into two categories; sediments and waters above the halocline (middle waters and mid-range A sediments) and below the halocline (deep waters and deep A sediments). TP and SPM fluxes and masses in the Gulf of Riga are thus simulated by CoastMab in the same manner as two other subbasins, the Bothnian Sea and the Bothnian Bay, using four state variables which represent surface waters, deep waters, ET sediments and A sediments. Six state variables are used for the Baltic Proper and the Gulf of Finland: surface waters and ET sediments, both above D_{WB}; middle waters and mid-range A sediments, both located between D_{WB} and the halocline; and, finally, deep waters and deep A sediments located below the halocline.

Some major simulated TP and SPM fluxes in CoastMab are sedimentation, mixing, resuspension (including erosion), diffusion, burial, biouptake and outflow. Sedimentation is the downward flux of SPM and particulate P due to gravity. Mixing is defined as the wind and wave driven TP and SPM fluxes between surface waters and lower water layers.

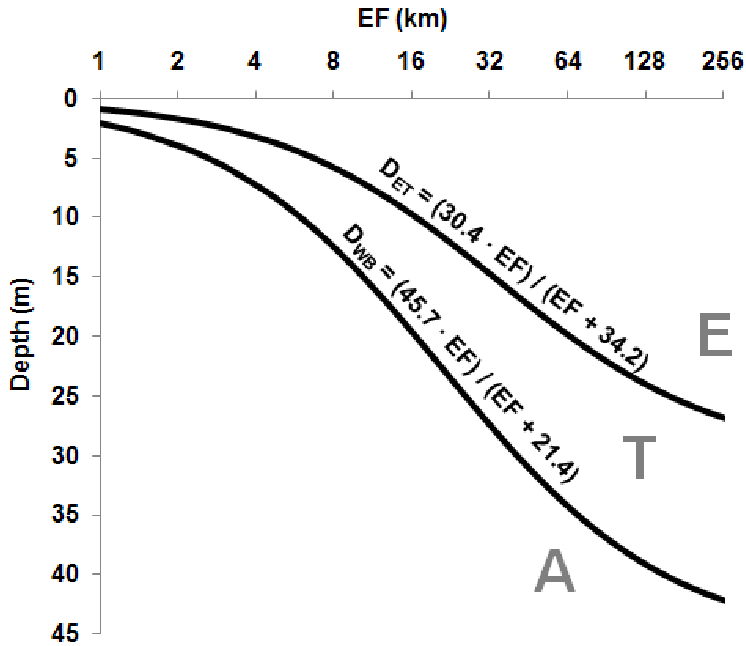

Figure 4. The ETA-diagram. Functional division lines between bottom areas of erosion (E), transportation (T) and accumulation (A). D_{ET} is the divider between E and T areas while D_{WB} (the depth of the theoretical wave base) is the divider between T and A areas. Both D_{ET} and D_{WB} are determined from the effective fetch (EF ≈ √[surface area] for whole basins). From Håkanson and Bryhn (2008a).

Resuspension refers to substance fluxes from ET sediments to the water column and such fluxes are also mainly driven by wind and wave action, and are parameterised in the model according to the morphometry of each subbasin and the P or SPM content in the ET sediments of each subbasin. Diffusion concerns dissolved P fluxes from water layers and sediments with high dissolved P concentrations to water layers with lower dissolved P concentrations. The SPM diffusion is zero since SPM is by definition particulate and not dissolved matter. SPM and TP pools in A sediments which do not return to the water column instead eventually enter the geosphere as they are buried by more recently precipitated material and this particle flux to the geosphere is commonly referred to as burial. Biouptake is the incorporation of dissolved P by primary producers during photosynthesis. Outflow is the TP and SPM export from the Baltic Sea to the adjacent Kattegat.

One flux category included in CoastMab but not in most other Baltic Sea nutrient or foodweb models is related to rising land and sediment areas. The isostatic land uplift after the last glaciation period occurs at a rate of 0-9 mm/year in the Baltic Sea region (Eronen et al. 2001). Land uplift gradually lowers the wave base and exposes new bottom areas to increased wind and wave driven erosion and resuspension. This means that large amounts of TP and SPM are continuously added to the system as a result of geological events (Jonsson et al. 1990). The eroded material primarily consists of glacial and postglacial clay, and the clay particles appear to have a clarifying effect on the Baltic Sea water. This effect has been used in the CoastMab model for improving the quantitative explanation to why waters are clear and oligotrophic in the northern Baltic Sea where land uplift is high while nutrient concentrations are much higher in the south where land uplift is low. Water mixing between subbasins is intensive, so geographical variations in nutrient loadings from the catchment cannot alone be the reason for the strong nutrient gradient in the Baltic Sea (Håkanson and Bryhn 2008a; Bryhn and Håkanson 2010).

Chl is tightly correlated to TP and total nitrogen concentrations in aquatic systems. Furthermore, Chl increases at increasing temperatures, while salinity in the surface water layer (Sal_{SW}, in psu) affects the Chl/TP ratio (Håkanson and Bryhn 2008a; Håkanson et al. 2010). In CoastMab, Chl is predicted from monthly surface water temperatures (SWT, in °C; the annual average and seasonal variation of SWT is modelled from the mean latitude of each subbasin), long-term mean salinities and dynamically modelled TP concentrations in surface waters (TP_{SW}; in µg/l):

$$Chl = TP_{SW} \cdot DF_{SW} \cdot Y_{DayL} \cdot Y_{Sal} \cdot Y_{SWT,\,Chl} \tag{1}$$

where DF_{SW} is the dissolved fraction of phosphorus in surface water (dissolved phosphorus concentrations in surface waters divided by TP_{SW}), while Y_{DayL} is a dimensionless seasonal moderator expressing the number of daylight hours during an average day of the month. Y_{Sal} is a dimensional moderator for salinity and $Y_{SWT,\,Chl}$ is a dimensionless moderator which expresses the impact of surface water temperatures on Chl. Y_{Sal} is differently defined for different salinity ranges:

$$\begin{aligned}
&Y1 = \text{if } Sal_{SW} < 2.5 \text{ then } (0.20 - 0.1\cdot (Sal_{SW}/2.5 - 1)) \text{ else } (0.20+0.02\cdot(Sal_{SW}/2.5-1))\\
&Y2 = \text{if } Sal_{SW} < 12.5 \text{ then } Y1 \text{ else } (0.28 - 0.1\cdot (Sal_{SW}/12.5 - 1))\\
&Y3 = \text{if } Sal_{SW} > 40 \text{ then } (0.06 - 0.1\cdot (Sal_{SW}/40 - 1)) \text{ else } Y2\\
&Y_{Sal} = \text{if } Y3 < 0.012 \text{ then } 0.012 \text{ else } Y3
\end{aligned} \tag{2}$$

where Sal_{SW} is the salinity (in psu) in surface waters and Y1, Y2, and Y3 are dimensionless moderators used for defining Y_{Sal}. $Y_{SWT, Chl}$ is defined as:

$$Y_{SWT, Chl} = \text{If } (SWT > 4) \text{ then } (1) \text{ else } (SWT+0.1)/4 \qquad (3)$$

Secchi depth is predicted from salinity and from dynamically modelled SPM values. SPM, i. e., particles, prevent light from penetrating the water column, so SPM in surface waters (SPM_{SW}) and Secchi depth have an inverse relationship. The higher the salinity, the greater the flocculation of particles, the more rapid the sedimentation and the clearer the water. In the Baltic Sea, salinity may also be used as a proxy variable for colour, i. e., for allochthonous matter which prevents light dispersion and is transported by the freshwater discharge from the catchment (Håkanson et al. 2010). A basic, cross-systems approach for predicting Secchi depth is used for defining the model variable Sec_{Ref} (in m):

$$Sec_{Ref} = 10\text{^}(-((10\text{^}(0.15 \cdot \log(1 + Sal_{SW}) + 0.3) - 1)) + 0.5) \cdot (\log(SPM_{SW}) + 0.3)/2 + (10\text{^}(0.15 \cdot \log(1 + Sal_{SW}) + 0.3) - 1))) \qquad (4)$$

Sec_{Ref} is then used in combination with Y_{SalSec} (a salinity and water flux based dimensionless proxy variable for colour) to model Secchi depth while compensating for allochthonous influence:

$$\text{Secchi depth} = Y_{SalSec} \cdot Sec_{Ref} \qquad (5)$$

where Y_{SalSec}, is defined as

$$Y_{SalSec} = (Sal_{SW}/Sal_{Kattegat} + Q_{BPGR}/(Q_{trib}+Q_{BPGR})) / 2 \qquad (6)$$

in which $Sal_{Kattegat}$ is the salinity in Kattegat (in psu), Q_{trib} is the water discharge from the catchment (in m^3/month) and Q_{BPGR} is the water flux from the Baltic Proper to the Gulf of Riga (in m^3/month).

Håkanson et al. (2010) modelled total fish biomass values as deviations around a norm value for the water body in question. Such deviations may be different levels of fishing pressure, changes in consumption by marine mammals and changes in other foodweb interactions or climatic variables. This norm value will be referred to as total fish biomass in the present work and is given in tonnes as:

$$\text{Total fish biomass} = (Y_{Sal} / 0.28) \cdot \text{Area} \cdot 10^{-3} \cdot 590 \cdot TP_{SW}^{0.71} \quad (7)$$

where Area is the surface water area (in m^2) and Y_{Sal} was defined in eq. 2.

Predictions from CoastMab have previously been successfully tested against long-term TP, Chl and Secchi depth data from the Baltic Sea subbasins displayed in fig. 1 (Håkanson and Bryhn 2008a), against long-term TP data from 21 Baltic Sea coastal areas (Håkanson and Eklund 2007) and against a long time series of TP, Chl and Secchi depth data in Ringkøbing Fjord, Denmark (Håkanson et al. 2007). The model error was in general smaller than the relative deviation in a comparison of empirical data divided into two randomly selected datasets. Fish biomass data are, however, comparatively uncertain and difficult to predict; empirical values in Ringkøbing Fjord were predicted in the right order of magnitude by Håkanson and Bryhn (2008b) using the CoastMab/CoastWeb approach.

The cost for improving urban sewage treatment to UWWTD standards was taken from HELCOM and NEFCO (2007) and was provided in 2004 euros. Costs were converted to 2011 prices by adjusting for inflation according to rates reported by EUROSTAT (2011; see table 3). A separate cost estimate for the Gulf of Riga drainage basin was made using inflation adjusted costs from HELCOM and NEFCO (2007) multiplied by the ratio of possible nutrient reductions to the Gulf of Riga to possible nutrient reductions to the Baltic Sea, provided as percentage values in the final column of table 1. This operation rests on the assumption that the percentage of reductions that each country performs in the Gulf of Riga drainage basin compared to the whole Baltic Sea drainage basin (in table 1) equals the percentage of costs for reductions in this area compared to reductions in the Baltic Sea drainage basin.

Table 3.
Annual inflation rates (in %) in the euro zone. Data from EUROSTAT (2011).

Year	Inflation rate
2004	2.1
2005	2.2
2006	2.2
2007	2.1
2008	3.3
2009	0.3
2010	1.6

Results

Long-term model output values simulated at current TP loadings are compared to empirical long-term mean values in fig. 5. TP and Chl predictions were higher than empirical means, but the difference was less than one standard deviation higher than empirical means given in Håkanson (2009), see figs. 5A and C. Secchi depth was also predicted higher than the empirical mean value and the difference was between one and two standard deviations calculated from the empirical data (fig. 5B). Thus, while the model suggested a higher trophic state than empirical data in terms of TP and Chl, the modelled Secchi depth predictions, conversely, expressed a lower trophic state than measured values.

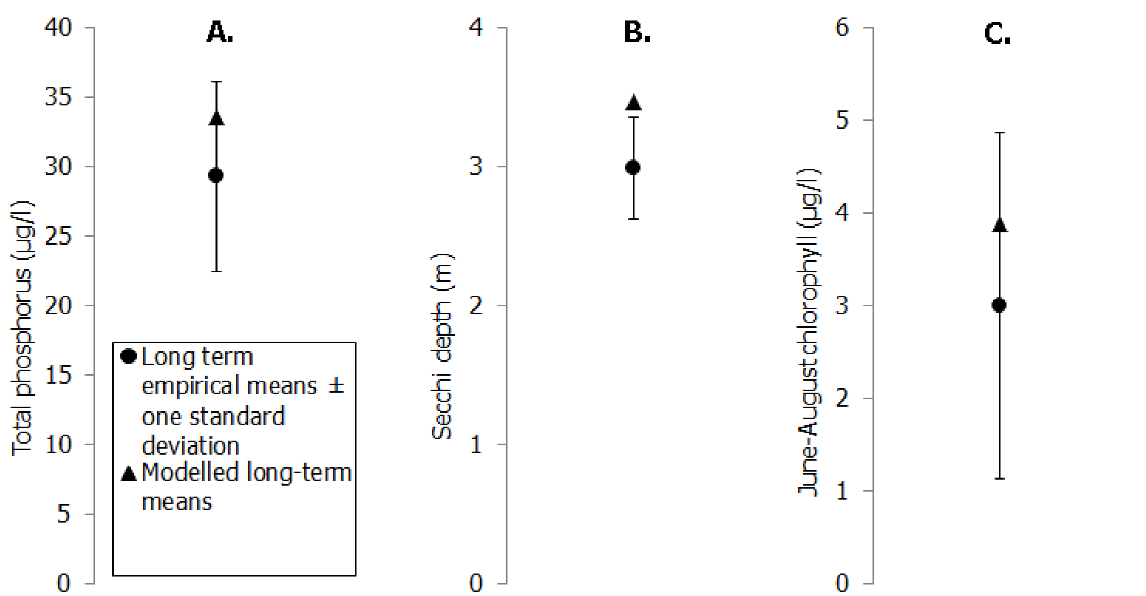

Figure 5. Model output compared to empirical long-term measurements. A. Total phosphorus concentration. B. Secchi depth. C. June-August chlorophyll concentration. Empirical means and standard deviations 1992-2005 from Håkanson (2009).

Fig. 6 shows simulated TP concentrations in surface waters of the Gulf of Riga during three different scenarios. Scenario 1 described the no action alternative, which is described as “Constant TP loading” in fig. 6. TP concentrations would in Scenario 1 stabilise at a long-term mean value of 33-34 µg/l. In Scenario 2, the TP input to the Gulf from the drainage basin would decrease by 54% in month 13, which would correspond to an instantaneous implementation of the UWWTD in the drainage basin. Scenario 2 is called “Reductions in Gulf of Riga drainage basin” in fig. 6. The TP concentration in surface waters would eventually decrease by about 17%, to approximately 27-28 µg/l along Scenario 2. Scenario 3 included the assumption that the UWWTD would be implemented in the whole Baltic Sea drainage basin in month 13, which would decrease direct TP emissions to the Gulf of Riga by 54%, emissions to the Baltic Proper by 42% and emissions to the Gulf of Finland by 47%. In this scenario, referred to as “Reductions in Baltic Sea drainage basin” in fig. 6, TP concentrations decreased by about 54% to 15-16 µg/l (fig. 6).

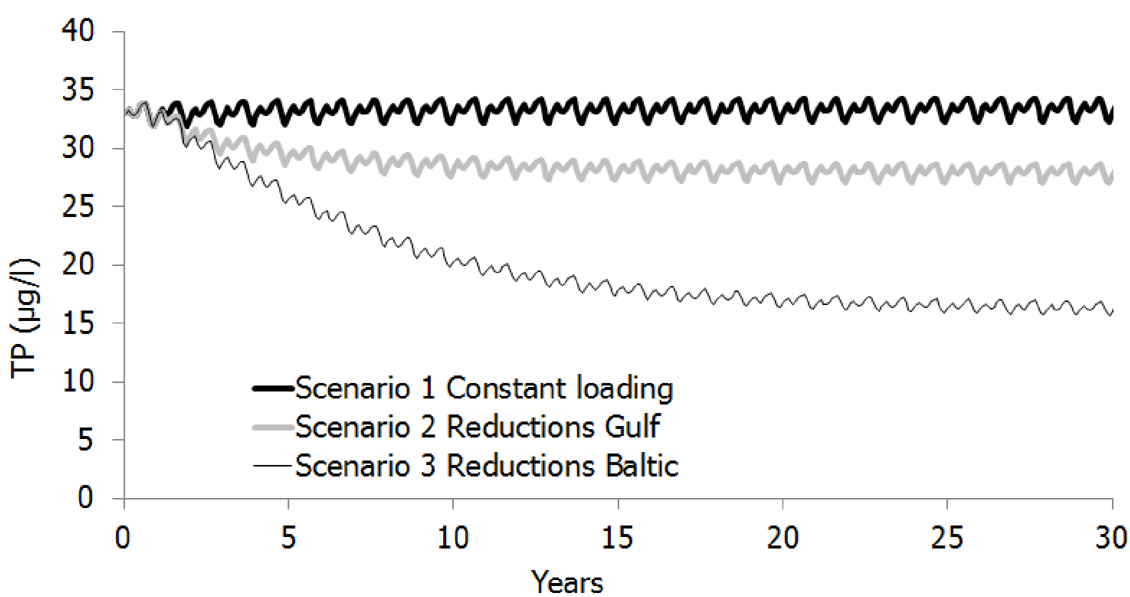

Figure 6. Simulated total phosphorus (TP) concentrations in surface waters of the Gulf of Riga from improved urban sewage treatment. Scenario 1: Constant TP loading. Scenario 2: Reductions in Gulf of Riga drainage basin. Scenario 3: Reductions in Baltic Sea drainage basin.

The Secchi depth simulated along the same three scenarios as those in fig. 6 is displayed in fig. 7A. In Scenario 1, the Secchi depth fluctuated around 3.5 m, while Scenario 2 led to a gradually increased Secchi depth values by some 45% to 5.0 m according to fig. 7A.

Even greater water clarity was achieved along Scenario 3, in which the Secchi depth was eventually doubled to 6.9 m. Simulated Chl concentrations during June-August are shown in fig. 7B. In Scenario 1, concentrations ranged from 3.5 to 4.3 µg/l, and decreased by about 13% to 3.1-3.7 µg/l during Scenario 2. Concentrations decreased even more in Scenario 3, by 53% to 1.7-2.1 µg/l. Average values in total fish biomass are depicted in fig. 7C. The average biomass fluctuated around 94,000 tonnes during Scenario 1 and decreased by 13% to about 82,000 tonnes along Scenario 2. A sharp total fish biomass decrease was noted in Scenario 3 as mean values decreased by 42% to 54,000 tonnes. Foodweb simulations of scenarios 1-3 using CoastWeb (Håkanson et al. 2010) yielded identical results for fish as in fig. 7C, since there were no foodweb interaction changes in any of the three scenarios.

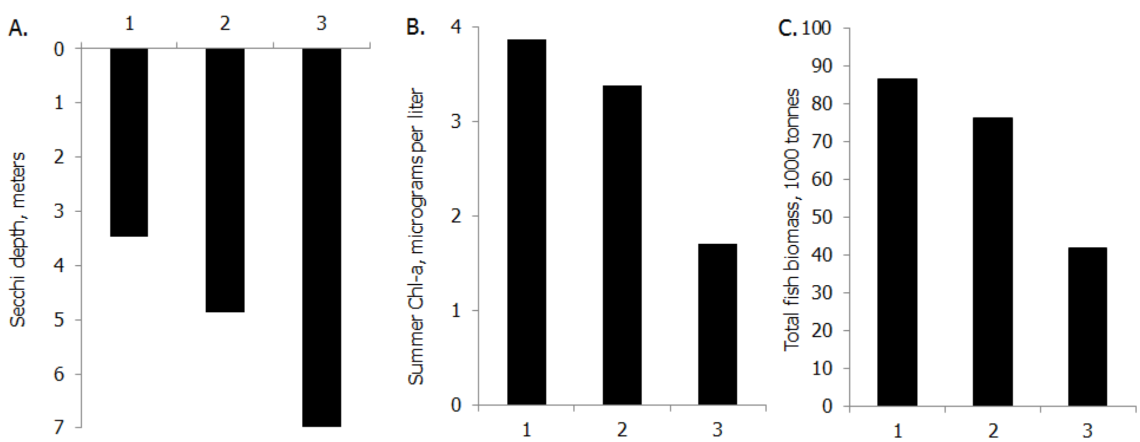

Figure 7. Simulated long-term environmental effects in the Gulf of Riga from improved urban sewage treatment. A. Secchi depth. B. June-August chlorophyll-a concentration. C. Total fish biomass. Scenario 1: Constant TP loading. Scenario 2: Reductions in Gulf of Riga drainage basin. Scenario 3: Reductions in Baltic Sea drainage basin.

Thus, improved urban sewage treatment was projected to increase the Secchi depth but decrease TP and Chl concentrations and total fish biomass. It is apparent from fig 6 that the most accentuated changes in TP occurred during the first years after the loading decrease and that Scenario 2 would lead to stationary conditions more rapidly than Scenario 3.

Fig. 8 highlights the difference between Scenarios 2 and 3 with respect to percentual changes in modelled TP and Secchi depth values. The sharpest changes in both scenarios and both variables occurred 1-2 years after improved urban sewage treatment was introduced. The annual Secchi depth change became less than 1% six years after improved urban sewage treatment in Scenario 2, and less than 0.1% after eleven years. The absolute value of the annual TP change was lower than 1% after five years and lower than 0.1% after seventeen years (fig. 8A). During Scenario 3, it took 12 years after improved urban sewage treatment until the the Secchi depth change was less than 1% per year and 41 years until this change was less than 0.1%. The same scenario required 19 years to elapse from improved urban sewage treatment until the absolute value of the TP concentration change was lower than 1% per year and 45 years until this value was lower than 0.1% (fig. 8B).

Annual changes amongst TP and Secchi depth were then used for defining “stabilising conditions” so that the trophic state was considered to stabilise when the absolute value of the simulated annual changes was lower than 1%. Conditions were assumed to be stable when absolute values of changes were lower than 0.1%. Thus, “stabilising conditions” in Scenario 2 occurred during 11-16 years after improved urban sewage treatment in Scenario 2 and during 19-41 years after improved urban sewage treatment in Scenario 3. It is also quite apparent from fig 6 that TP was still changing during 11-16 years after improved urban sewage treatment in Scenario 2, but the significance of annual changes after that would have to be determined with statistical methods.

The estimated costs for implementing Scenarios 2 and 3 are listed in table 4. In this table, there are two additional cost scenarios (from HELCOM and NEFCO 2007): that all new investments are made in rural areas (high cost scenario) or that these investments are exclusively made in urban areas (low cost scenario). Implementing the UWWTD in the Gulf of Riga drainage basin should cost 43-103 million euros per year while implementing this directive in the whole drainage basin of the Baltic Sea should cost 468-1,118 million euros per year.

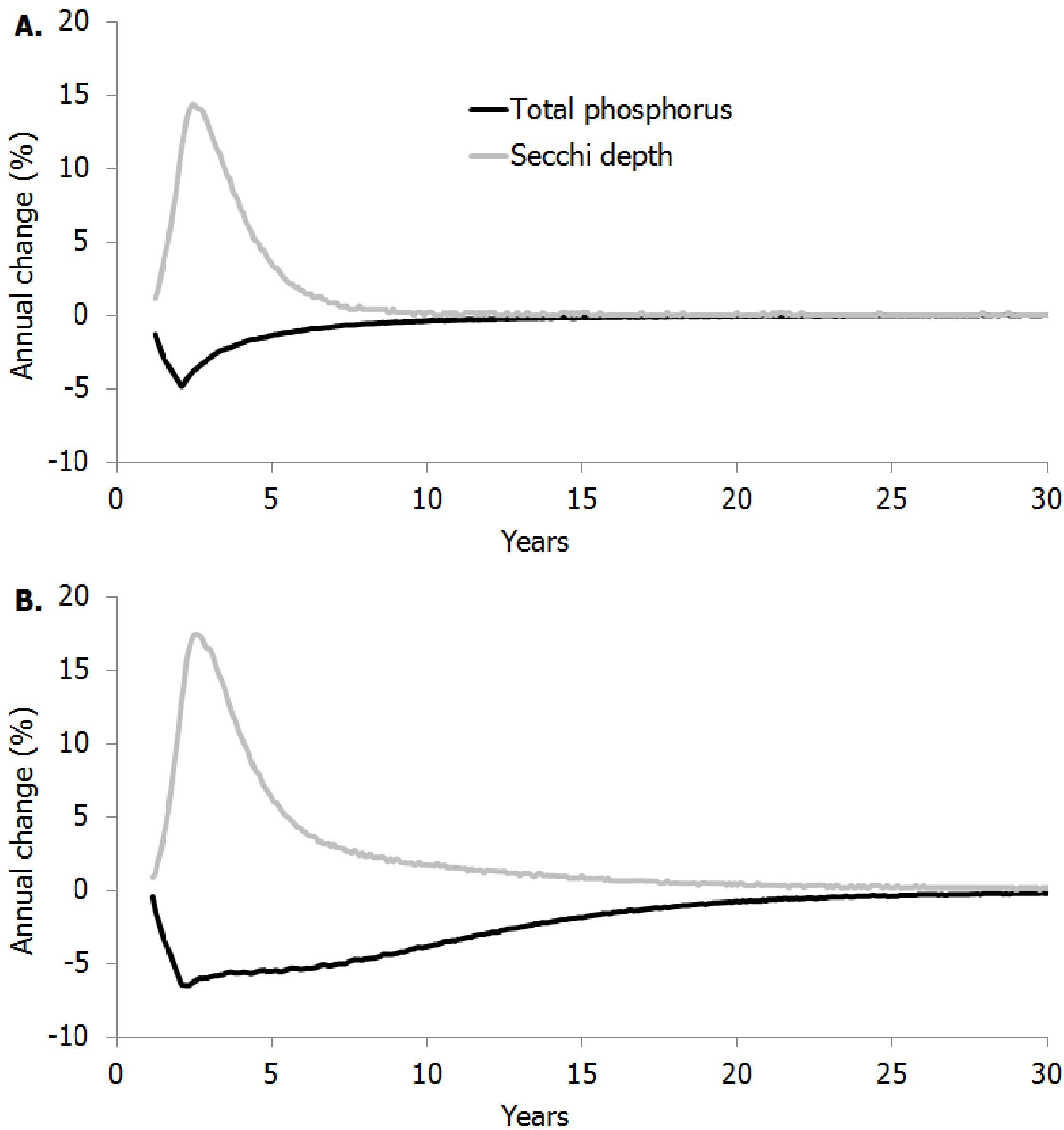

Figure 8. Annual change in modelled total phosphorus concentrations and Secchi depths. A. Scenario 2: Reductions in Gulf of Riga drainage basin. B. Scenario 3: Reductions in Baltic Sea drainage basin.

Discussion and conclusions

Simulated TP and Chl concentrations at present loading levels were less than one standard deviation higher than long-term empirical mean values (fig. 5) while simulated Secchi depth values were less than two standard deviations higher than the mean empirical Secchi depth. It should be noted that due to uncertainties in empirical data and in models, one cannot expect perfect predictions. CoastMab has been calibrated for making good overall predictions in all Baltic Sea subbasins in order to capture general patterns in the various flux algorithms (Håkanson and Bryhn 2008a; Håkanson et al. 2010). Simulating the phosphorus cycle in both the Baltic Proper and the Gulf of Riga was very important for predicting changes in the Gulf (figs 6 and 7). Therefore, it would have been a worse alternative to have calibrated the model against empirical data from the Gulf only, in comparison with the approach described in Methods.

Table 4. Annual implementation cost (2011 prices, million euros) of the Urban Waste Water Treatment Directive. The calculation method is described in Methods. Estimates for the Gulf of Riga drainage basin are part of the estimates for the Baltic Sea drainage basin. Two cost estimates have been made for each case. High cost estimates are based on the assumption that all new investments are made in rural areas. Low cost estimates assume that all new investments are made in urban areas.

Country	**Baltic Sea drainage basin**		**Gulf of Riga drainage basin**	
	High	Low	High	Low
Belarus	267	113	71	29
Czech Republic	18	8.0	0	0
Estonia	1.1	0.25	0.14	0.03
Lithuania	85	35	6.4	2.5
Latvia	1.1	0.43	1.0	0.37
Russia	228	96	25	11
Poland	518	216	0	0
Total	**1,118**	**468**	**103**	**43**

Simulations in this study (figs. 6 and 7) demonstrated an important principle for the Gulf, that the water quality is highly dependent on the water quality in the adjacent Baltic Proper. Using a different modelling approach, Savchuk (2002) came to a somewhat different conclusion; that the TP import to the Gulf from the Baltic Proper was actually smaller (1,600 tonnes per year) than the TP import from the catchment. However, more recent studies by the same author (Savchuk 2005; Savchuk and Wulff 2007) covering all major subbasins of the Baltic Sea found that the TP import from the Baltic Proper was instead much larger than the TP import from the catchment, which has later been supported by Håkanson and Bryhn (2008a), Håkanson et al. (2010) and the present study. Thus, in order to substantially counteract the eutrophication of the Gulf, it would be necessary to decrease TP loadings to the whole Baltic Sea and not only from the drainage basin of the Gulf (figs 6 and 7).

HELCOM (2007) suggested a summer Secchi depth exceeding 4.5 m as an appropriate environmental goal for the Gulf while Håkanson (2009) proposed an annual Secchi depth of 4.6 m as a reference value. However, it is probably not even enough to return to the

1960s Secchi depth values of 5 m (see Background) to achieve good water quality in the Gulf of Riga according to the MSFD (good status means a very limited anthropogenic influence; see table 2). Anthropogenic nutrient fluxes into the Baltic have been significant since the early 1900s (Savchuk et al. 2008). There were decreasing long-term trends in the Secchi depth in the Baltic Sea already in 1919-1939 which subsequently continued (Sandén and Håkansson 1996). Because of the intensive nutrient exchange between the Gulf and the rest of the Baltic Sea, it is likely that eutrophication in the Gulf to a great extent concurred with eutrophication in the whole Baltic Sea both before and after the 1960s. Therefore, there may be reason to believe that urban sewage treatment according to Scenario 2 which resulted in a long-term Secchi depth of about 5 m (fig. 7A) would not be sufficient for complying with the MSFD. Good water quality may instead require a Secchi depth of about 6 m, i. e., twice as great as current mean values of 3 m. Scenario 3 resulted in a Secchi depth near 7 m (fig. 7A) which may be between the classifications "good" and "high" water quality according to table 2. However, given that the model yielded slightly exaggerated Secchi depth values (fig. 5), the TP reductions in Scenario 3 (reductions in the Baltic Sea drainage basin) could in any case be necessary for doubling the Secchi depth from 3 m to 6 m.

Considerably greater average Secchi depths than 5 m could be achieved within 10 years according to the simulations, but it may be difficult to reach a similar goal before year 2020 (as prescribed in the MSFD) because of the time it takes to motivate, initiate, and complete urban sewage treatment plants in the whole Baltic Sea drainage basin. To reach this goal by 2030 would probably be a more realistic ambition. If Scenario 3 should be realised (reductions in the Baltic Sea drainage basin), the TP concentration, the Chl concentration and the total fish biomass should be expected to stabilise at about half of their current values (figs 6, and 7B and 7C).

However, decreased biomass and production potential of fish may decrease profits and work opportunities in the fisheries sector, which is likely to be unpopular. A decreasing size of the total fish biomass would imply decreasing total allowable catches (TACs) of fish. It is also possible that a great trophic state decrease in the Gulf would trigger shifts in species composition, into an increasing percentage of highly economically valued species (Håkanson et al. 2010), although investigating such shifts has been beyond the scope of the present study. Modelling results suggested that total fish biomass changed to a slightly lesser percentual extent than Chl, which is supported by results in Håkanson and Boulion (2002; see also fig. 2A) but according to the regression model in Chassot et al. (2007; see also fig. 2B), percentual changes in total fish biomass could very well have been as great or greater in comparison with Chl changes. In this context it is worth noting that variations in herring biomass in the Gulf have been substantial during the past decades. The spawning stock biomass doubled from the period 1977-1982 until 1995-1990 (fig. 3), although these changes occurred when trophic state changes were modest and may instead have been driven by climatic factors (Kotta et al. 2009), or by the decreasing cod populations described by Ojaveer et al. (1999) and Eero et al. (2008). Large changes in trophic state may have clear effects on fish biomass (Thurow 1997; Håkanson and Boulion, 2002; Chassot et al. 2007; Hansson et al. 2007). However, effects from small changes in trophic state are difficult to distinguish from a multitude of other factors such as changes in fish landings or changes in climate related variables.

The time resolution for decreasing the TP concentration in the Gulf of Riga by means of decreasing TP inputs has to the best of the author's knowledge not been estimated previously. Fig 8 shows that the time between action and full effect (11-16 years during Scenario 2 and 19-41 years during Scenario 3) depends on the extent and location of the action. Håkanson et al. (2010) concluded that it would take 20 years or more until TP abatement action would have full effect on the Baltic Sea. Savchuk and Wulff (2007) estimated a much slower reaction; if all TP inputs into the Baltic Sea were to be stopped instantaneously, they concluded that it would take 50 years for the TP concentration to decrease by 50% and 250 years to decrease the concentration by 95%.

It is striking that one single type of abatement strategy, urban sewage treatment, accounts for such a large part of the potential TP reduction to the Gulf of Riga and the rest of the Baltic Sea (see table 1 and HELCOM 2007). No other option such as wetland construction or more efficient manure handling and application have individually or in combination been shown to be nearly as effective in terms of potential TP reduction compared to improved urban sewage treatment. The latter is also one of the most cost-effective measures (Gren and Elofsson 2008) and it may be difficult to achieve any substantial trophic state changes without relying fully or partly on this measure (Bryhn 2009). The present study has shown that improved urban sewage treatment at a cost of 468-1,118 million euros per year could make a profound impact on the trophic state of the Gulf of Riga. This cost should be compared to the cost of the Baltic Sea Action Plan which has been estimated at 3,400 million euros per year in 2011 prices (calculated from HELCOM and NEFCO 2007). The Baltic Sea Action Plan also requires many measures with low cost-effectiveness (Bryhn 2009). However, regardless of which nutrient abatement strategy is selected, the expected effects on fish production should also be weighed in together with other costs and it is possible that the population around the Gulf of Riga would not want any substantial nutrient reductions on behalf of decreasing fish stocks.

This study has investigated how the water quality and total fish biomass in the Gulf of Riga would be affected by improved urban sewage treatment around the Gulf and around the Baltic Sea. Simulations showed that good water quality in the Gulf according to the EU Marine Strategy Framework Directive could be achieved if urban sewage treatment would be performed to meet Nordic and German standards in all parts of the Baltic Sea drainage basin. The Secchi depth would approximately double while total phosphorus and chlorophyll concentrations would be cut by half. The total fish biomass would also eventually decrease by about 42% of its current size if good water quality should be achieved although such a change may be counteracted or exacerbated by changes in fishing pressure or climate. Improved sewage treatment to this extent would cost 468-1,118 million euros per year. These findings suggest that substantial international cooperation is necessary for reaching good water quality in the Gulf. Moreover, findings in this study support the suggestion by Hansson et al. (2007) that diminishing fish stocks as a possible adverse side effect from nutrient abatement needs to be acknowledged and investigated further.

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