# Modelling nutrient retention in the coastal zone of an eutrophic sea

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# Abstract

The Swedish Coastal zone Model (SCM) was used at a test site, the Stockholm Archipelago located in the northern part of the central Baltic Sea, to study the retention capacity of the coastal filter on nitrogen (N) and phosphorus (P) loads from land and atmosphere. The efficiency of the coastal filter to permanently retain nutrients determines how much of the local nutrient loads actually reach the open sea. The SCM system is a NPDZ-type model coupled to a horizontally integrated, physical model in particular suitable for estuaries. In this study the Stockholm Archipelago consisting of 86 sub-basins was divided into three sub-areas: the inner, the intermediate and the outer archipelago. An evaluation of model results showed that the modelled freshwater supply agrees well with observations. The nutrient, salinity and temperature dynamics simulated by the SCM model are also found to be in good or acceptable agreement with observations. The analysis showed that the Stockholm Archipelago works as a filter for nutrients that enter the coastal zone from land, but the filter

1    efficiency is not effective enough to retain all the supplied nutrients. However, at least 65 %

2    and 72 % of the P and N, respectively, are retained during the studied period (1990-2012). A

3    major part of the retention is permanent, which for P means burial. For N almost 92 % of the

4    permanent retention is represented by benthic denitrification, less than 8 % by burial, while

5    pelagic denitrification is below 1 %. Highest total amounts of P and N are retained in the

6    outer archipelago where the surface area is largest. The area specific retention of P and N,

7    however, is highest in the smaller inner archipelago and decreases towards the open sea. A

8    reduction scenario of the land loads of N and P showed that the filter efficiencies of N and P

9    increase and the export of N from the archipelago decreases. About 15 years after the

10   reduction the export of P changes into an import of P from the open sea to the archipelago.

# 1  Introduction

The worldwide increase of coastal eutrophication and anoxia has spread exponentially since the 1960s. Coastal oxygen depletion is associated with dense population areas and large river loads of nutrients (Diaz and Rosenberg, 2008). The use of industrially produced fertilizer started in the late 1940s and has since then been contributing to the anthropogenic fertilization of the global marine system (Galloway et al., 2008). The river load of nutrients originating from agriculture activities has been shown to be controlled by the size of the river flow, e.g. the flow from the Mississippi River has a large impact on the oxygen conditions in the northern Gulf of Mexico, which suffers from severe hypoxia with "dead zones" as a result (Rabalais et al., 2002).

With ambition to diminish eutrophication there has been a lot of efforts around the world to reduce the land load of nutrients to sea, but the expected results of a healthier environment have not been accomplished in all places (Kemp et al., 2009). The responses of eutrophication and the extent of hypoxic area for changes in nutrient loads are different in different types of systems. Also changes in climatic and hydrodynamic conditions might lead to a non-linear recovery (Kemp et al., 2009). Nutrients transported from land to sea first enter the coastal zones and are then further transported towards the open sea. However, not all of the supplied nutrients reach the open sea as they are retained in the coastal zone (Fig. 1), which acts as a filter (McGlathery et al., 2007). The retention capacity depends on different chemical, physical and/or biological processes that involve nutrients e.g. denitrification, permanent burial, algae and plant assimilation (Duarte and Cebrián, 1996; Voss et al., 2005). The filter efficiency of the coastal zone might be of large importance for the water quality in open waters.

*Fig. 1.*

Retention capacity is, however, not well defined. Johnston (1991) discussed that retention processes are of different magnitudes and irreversibility, e.g. plant uptake and litter decomposition provide short- to long-term retention of nutrients. Billen et al. (2011) and Nixon et al. (1996) defined retention as the net effect of temporary and permanent removal from the water phase through different biogeochemical processes. Burial and denitrification lead to a permanent removal of nutrients from the ecological system (Voss et al., 2005). Plant assimilation of nutrients and sedimentation of organic material might influence the temporary retention, a build-up of active nutrient pools in the water and in the sediment. Some of the

organic material is more refractory than others e.g. parts of root systems, which also can influence biogeochemical processes by enhanced sediment oxygen, nutrient and dissolved organic material concentrations (McGlathery et al., 2007). Thus, temporary retention depends on the release rates, translocations, and the longevity of plants, which causes variations in retention capacity depending on the time scale of the study. The net effect of nutrient retention in an area can be studied by the simple method of subtracting the output of nutrients from the input (Johnston, 1991). This simple method of calculating the retention capacity of nitrogen (N) and phosphorus (P) has been used in a number of studies (e.g. Eilola et al., 2014; Hayn et al., 2014; Karlsson et al., 2010; Nixon et al., 1996; Sanders et al., 1997) for different areas of the world. The retention capacity has been discussed to be related to the residence time and depth in different water systems (Balls, 1994; Hayn et al., 2014; Nixon et al., 1996). Hence, the longer a water parcel and its nutrient content stays within a system, the more the containing nutrients are affected by the internal transformation and retention processes.

Filter efficiency is in the present study referred to as the capacity of the studied area to retain the local nutrient loads from land and atmosphere (see Section 2.4). It is distinguished between the permanent removal and temporal retention of which the latter is caused by changes in the N and P inventory (Fig. 1). There are studies of nutrient retention in different coastal zones around the world, but there are not enough estimates to evaluate and understand its effect on the environmental status of coastal seas. Quantification of the filter efficiencies in different coastal ecosystems as estuaries, archipelagos, lagoons and embayments would increase the understanding and the knowledge necessary for managing the coastal zone. Numerical models have been used to a larger extent for studies in lakes and freshwater catchment areas (e.g. Ahlgren et al., 1988; Hejzlar et al., 2009) than for retention and filter efficiency studies in coastal areas where only a few studies seem to exist in the literature (e.g. Fennel et al. 2006; Seitzinger and Giblin, 1996;  Xue et al, 2013).

The Baltic Sea (Fig. 2), located in northern Europe, is an example where the enhanced land load of nutrients to the sea (Gustafsson et al., 2012) has led to eutrophication and consequently increased frequency and intensity of cyanobacterial blooms, expanding bottom hypoxia and dead bottom zones (e.g. Bergström et al., 2001; Conley et al., 2009; Diaz and Rosenberg, 2008; Vahtera et al., 2007). Actually, the largest anthropogenically induced hypoxic area in the world is found in the Baltic Sea (Carstensen et al., 2014), where it varied between 70000 and 80000 km$^2$ during year 2010-2014 (Hansson and Andersson, 2014). In the

Baltic Sea, most of the coastal zones and the open sea still suffer from eutrophication in spite of reduced nutrient loads since the 1990s (HELCOM, 2010).

The aim with this study is to quantify the filter efficiency in the eutrophic Stockholm Archipelago (see section 2.1) of N and P and to discuss the relative importance of different physical and/or biological processes using the Swedish Coastal zone Model (SCM). In addition, changes in the filter efficiency along the land-sea continuum, from the inner archipelago, through the intermediate and outer archipelago to the open Baltic Sea, will be studied in order to evaluate the effect of the size of the archipelago on the filter efficiency.

After a description of the model system (Section 2) and an evaluation of the results of SCM (Section 3.1), the filter efficiency of the coastal zone is calculated and the effects of a reduced land load of N and P are analyzed (Section 3.2). Conclusions finalize the study (Section 4).

## 2  Methods

### 2.1  Study site

The brackish archipelago of Stockholm (Fig. 2), located at the east coast of Sweden, is the largest archipelago in Sweden and the second largest in the Baltic Sea. The archipelago is a continuation of the river Norrström with an average discharge of about 160 $m^3$ $s^{-1}$ from Lake Mälaren (Lindh, 2013). The river outflow carries about 2600 metric tons (t) of N and 120 t of P annually to the coastal basin "Strömmen" in the inner archipelago (Lännergren, 2010). The rocky islands in the archipelago are surrounded by basins of different sizes and depths which are connected by straits. In this study the archipelago has been divided into three areas: the inner, intermediate and outer archipelagos. Several large islands form a natural border between the inner and the intermediate archipelagos and the limited water exchange occurs through five narrow sounds with shallow sills. The outflow from the inner to the intermediate archipelago passes through the sounds in the surface layer, while inflows of more saline water mainly occur at larger depths. The border between the intermediate and the outer archipelagos follows the chain of islands in north-south direction with several connections between the areas (Fig. 2).

*Fig. 2.*

The largest point sources of nutrients to the inner archipelago originate from waste water treatment facilities of Stockholm, which is situated at the outlet of the Lake Mälaren. Signs of

1  eutrophication in the Stockholm Archipelago have been observed as increased ratio of

2  laminated sediments from the 1930s (Jonsson et al., 2003) and the eutrophication status in the

3  inner Stockholm Archipelago was in the early 1970s classified as highly eutrophic

4  (Lännergren et al., 2009). In the 1970s the sewage treatment facilities in Stockholm started to

5  chemically precipitate P, which reduced their P load from about 600 t yr$^{-1}$ to about 100 t yr$^{-1}$

6  (Fig. 3 in Lücke, 2014). The reduction led to some improvements of the marine environment

7  (Brattberg, 1986), but in the 1990s the areas were still eutrophic with poor bottom water

8  oxygen conditions (Jonsson et al., 2003; Rosenberg and Diaz, 1993). In the mid-1990s there

9  was a further reduction of the P to about 25 t yr$^{-1}$ and the sewage treatment facilities started to

10  reduce the N as well, from about 3000  t yr$^{-1}$ to 1250 t yr$^{-1}$ (Fig. 3 in Lücke, 2014), which led

11  to further improvement of the eutrophication status. In 2008 the bottom oxygen conditions

12  had clearly improved in the deeper parts and only enclosed bays, such as e.g. Stora Värtan,

13  suffered still from anoxia (Karlsson et al., 2010 and references therein). However, the annual

14  monitoring status report of the environmental status of the inner Stockholm Archipelago in

15  2014 still classified the area as unsatisfactory eutrophic (Lücke, 2015) according to the

16  national directives by the Swedish Environmental Protection Agency and the Swedish Agency

17  for Marine and Water Management (Naturvårdsverket, 2007; HaV, 2013) based on the EU

18  Water Framework Directive. The area still suffered from reduced water transparency, high

19  concentrations of phytoplankton chlorophyll and areas without any bottom fauna due to low

20  oxygen concentrations.

21  ## 2.2  Model description

22  The Swedish Coastal zone Model (SCM) is a multi-basin 1D-model based on the equation

23  solver PROgram for Boundary layers in the Environment (PROBE;Svensson, 1998), coupled

24  to the Swedish Coastal and Ocean Biogeochemical model (SCOBI; Eilola et al., 2009;

25  Marmefelt et al., 1999). The model system was developed to calculate physical and

26  biogeochemical states in Swedish coastal waters. The inner, intermediate and outer Stockholm

27  archipelagoes (Fig. 2) are represented by 16, 44 and 26 sub-basins, respectively (see figure in

28  Supplement 1).

29  ### 2.2.1  PROBE

30  The physical model PROBE calculates horizontal velocities, temperature and salinity profiles

31  (Svensson, 1998; Omstedt, 2015). The surface mixing is calculated by a $k$-$\varepsilon$ turbulence model

1  and the bottom mixing is a parameterization based on the stability in the bottom water. Light

2  transmission, as well as ice formation growth and decay, are also included in the model. The

3  vertical grid resolution is half a meter in the uppermost layers, one metre from 4-70 m, and

4  two metres between 70-100 m. The general differential equation of the PROBE solver is

5  formally written as

6  $$\frac{\partial \phi}{\partial t} + \frac{\partial}{\partial x_i} u_i \phi = \frac{\partial}{\partial z}\left(\Gamma_\phi \frac{\partial \phi}{\partial z}\right) + S_\phi \qquad\qquad (1)$$

7  Here $\phi$ is the dependent variable, $t$ time, $z$ vertical coordinate, $x_i$ horizontal coordinates, $u_i$

8  horizontal velocities, $\Gamma_\phi$ vertical exchange coefficient, and $S_\phi$ source and sink terms. Vertical

9  advection (and moving surface) is included accounting for vertical transport in sub-basins due

10  to in and outflows. The sources and sinks determined by the ecosystem model are added to $S_\phi$.

11  The water exchange between the sub-basins is controlled by the baroclinic pressure gradients.

12  The net flow through the sounds will be the same as the river discharge from land in order to

13  preserve volume. Inflowing water to a sub-basin is interleaved into its density level without

14  any entrainment, and heavy surface water in one sub-basin may thus reach the bottom level in

15  an adjacent basin. The sea level variations outside the boundary are of minor importance for

16  the SCM results and are therefore not included in the forcing. The water exchange across the

17  boundary between the coastal zone and the open sea is assumed to be in geostrophic balance,

18  since this boundary is open with a width greater than the internal Rossby radius.  A time step

19  of 600 seconds was used in the present simulations.

20  **2.2.2  Biogeochemical model (SCOBI)**

21  The SCOBI model describes the biogeochemistry of marine waters in the Baltic Sea and

22  Kattegat (Eilola et al., 2009). Nine pelagic and two benthic variables (Fig. 3) are described in

23  the SCM-SCOBI model. In the pelagic zone three different phytoplankton groups (diatoms,

24  flagellates and others, and cyanobacteria), one zooplankton group, one pool for detritus and

25  three inorganic nutrients pools (nitrate, ammonium and phosphate) are represented. The

26  model also calculates oxygen and hydrogen sulfide concentrations, of which the latter are

27  represented by "negative oxygen" equivalents (1 ml $H_2S$ $l^{-1} = -2$ ml $O_2$ $l^{-1}$) and includes the

28  conversion of sulfate into hydrogen sulfide (Fonselius, 1969). Thus, the negative oxygen

29  corresponds to the amount of oxygen needed to oxidise the hydrogen sulfide. The sediment in

30  the present model is parameterized by one vertically integrated bulk sediment layer (level 3 in

Soetaert et al, 2000). Organic material that sinks to the sediment is divided into one benthic nitrogen pool (NBT) and one benthic phosphorus pool (PBT). SCOBI has been used and validated in several studies, both coupled to the basin scale Baltic Sea model PROBE-Baltic (e.g. Marmefelt et al., 1999) and to the three dimensional Rossby Center Ocean model (RCO; e.g. Meier et al., 2011).

In the model the processes of phytoplankton assimilation, mortality and nitrogen fixation, zooplankton grazing, excretion of detritus and dissolved inorganic nitrogen (DIN) and phosphorus (DIP), the oxygen and temperature dependent mineralization of detritus, benthic N and benthic P, nitrification and denitrification are described. Phytoplankton assimilates carbon (C), N and P according to the Redfield molar ratio (C:N:P=106:16:1) and the biomass is represented by chlorophyll (Chl) according to a constant carbon to chlorophyll mass (mg) ratio (C:Chl=50:1). Light attenuation depends on background attenuation due to water and humic substances and a variable attenuation caused by particulate organic matter (phytoplankton, zooplankton and detritus). All particulate variables sink downward through the water column. Predation is used as a closing term to parameterize interactions with higher tropic levels in the ecosystem and move matter from zooplankton to the detrital and inorganic pools. Resuspension of sediment that is important in the open Baltic Sea (Almroth-Rosell et al., 2011) has not yet been implemented in this SCOBI version, but the sediment releases dissolved inorganic nutrients back to the water mass, with the release of phosphate being redox dependent. Some fractions of benthic N and P are assumed to be buried in the sediment as a permanent sink, and are hence removed from the system. For further details of the SCOBI model the reader is referred to Eilola et al. (2009; 2011).

*Fig. 3.*

### 2.2.3 Forcing

The SCM-SCOBI model system is forced by weather, the conditions in the sea outside the archipelago, point sources, discharge of freshwater and nutrients from land and atmospheric deposition of nutrients. The initial values for both the pelagic zone and the sediment are derived from spin-up simulations.

There are two types of land derived forcing; discharge of water and nutrients from both rivers and surface run-off from the drainage area given by the S-HYPE model (Lindström et al., 2010) and point sources representing sewage plants and industries. The run-off is added to the surface water of each basin and no reduction of river nutrients due to precipitation at river-

mouths is assumed in this model setup. The point sources of nutrient loads are assigned to the depth levels mostly resembling the actual depth of the discharge. The inorganic riverine nutrient loads are added as DIN and DIP to the SCM. The organic nutrients in the land loads are calculated from the difference between total nitrogen (TN) and DIN, and total phosphorus (TP) and DIP, respectively. The bioavailability and the composition (dissolved or particulate) of the organic nitrogen and phosphorus loading from land are generally not known. In the present model configuration the fraction of organic nutrient loads that follows the Redfield ratio are assumed to be bioavailable and will be added to the detritus pool in the model, while the remaining fractions of nutrient loads are treated as conservative tracers in the model..

The weather forcing consists of solar insolation, air temperature, wind, relative humidity and cloudiness. The insolation and all the radiation and heat fluxes across the water-air interface are calculated by the PROBE model. The weather variables are taken from a gridded database developed at the Swedish Meteorological and Hydrological Institute (SMHI), using 3-hourly meteorological synoptic monitoring station data, and the depositions of nitrogen species (NHX and NOX) are calculated by the MATCH model (Robertson et al., 1999). For the deposition of phosphate, a literature value of 0.5 kg m$^{-2}$ month$^{-1}$ is used (Areskoug, 1993).

The boundary conditions to the open Baltic Sea is set by vertical mean profiles calculated by a one dimensional PROBE setup for each Baltic open water area and assimilation of monitoring data. The monitoring data used in the assimilation are extracted from the stations MS4, US5B, SR5, BY31 and BY29 (Fig. 2) depending on depth and time, to get the best representation of the open sea's influence on the SCM model domain.

## 2.3 Evaluation strategy

To quantify the fit between modelled values and observations a correlation coefficient, $r$, was calculated (Eq. 2).

$$r = \frac{\sum_{i=1}^{n}\left(P_i - \overline{P}\right)\left(O_i - \overline{O}\right)}{\sqrt{\sum_{i=1}^{n}\left(P_i - \overline{P}\right)^2 \sum_{i=1}^{n}\left(O_i - \overline{O}\right)^2}} \qquad\qquad 2$$

where $P$ is model value, $O$ is observation of the analyzed parameter, $i$ is the data number and $n$ is the total number of data points. Two series of observations and model values that are identical will lead to an $r$ value equal to one, while uncorrelated data result in a $r$ value close

1     to zero. In addition to the *r* value, the average cost function (C) values (Eq. 3) for the different

2     parameters were used in the evaluation of the SCM results.

4     $$C = \frac{\sum_{i=1}^{n} \left| \frac{P_i - O_i}{sd(O_i)} \right|}{n}$$                      3

5     A cost function describes the proximity of model results and observations by normalizing the

6     difference between them with the standard deviation (sd) of the observations. If average

7     model results fall within the standard deviation of observations, C is below one which is

8     regarded as good. Results that are within two standard deviations will be regarded as to be on

9     an acceptable level. The corresponding simulation levels, good and acceptable, for the

10    correlation coefficient are achieved when r is higher than two thirds (0.66) and one third

11    (0.33), respectively. This approach using *r* and C has been used in earlier studies (Edman and

12    Omstedt, 2013; Edman and Anderson, 2014) and is based on methods by Oschlies (2010).

13    The outflow from Lake Mälaren is three orders of magnitude larger than the sum of all other

14    S-HYPE fresh water components to the inner Stockholm Archipelago. The output from S-

15    HYPE of fresh water and nutrient loads from Mälaren to the Stockholm Archipelago was

16    therefore used in the evaluation of the fresh water forcing to SCM. Observations of freshwater

17    discharge were retrieved from the Baltic Environmental Database (BED, 2015) at the Baltic

18    Nest Institute, Stockholm University. The correlation between monthly mean of observed and

19    simulated discharge for the period (1990-2012) was then calculated.

20    In the evaluation of the results of the SCM in different basins, the long term averages (1990-

21    2012) of the vertical distribution of salinity, DIN, DIP and oxygen during winter (November-

22    February) and summer months (May-August) were compared to corresponding observations

23    for the whole modelled period. Further, the correlation *r* and the mean cost function C of the

24    vertical distribution of observations and model output were calculated. Also the long term

25    averages of the seasonal variations in surface temperature, DIN, DIP and bottom water

26    oxygen concentrations were used in the evaluation by calculating the corresponding *r* and C

27    values.

28    Observations from the Stockholm Archipelago (Fig. 4) were provided by Stockholm City and

29    Stockholm University. For the quantitative validation described above the quality of

30    observations from each site (Table 1) had to fulfil three requirements to be used in the

1 validation process; 1) period coverage: 80% of the years sampled; 2) annual coverage: at least

2 7 of the 12 months sampled; and 3) vertical data coverage: at least 5 depth levels frequently

3 measured over the full depth of the basin. In addition at least 3 months with observations were

4 required for the evaluation of winter and summer conditions. Average values were then

5 calculated for periods and depth levels with dense data distribution. The model output was

6 used in the same way as observations, and the modelled averages were calculated for the same

7 time intervals and depth ranges.

8 *Fig. 4.*

9 *Table 1.*

## 2.4 Calculation of retention

11 The retention of P and N in a region can be calculated as the difference between the load and

12 the outflow (Almroth-Rosell et al., 2015; Hayn et al., 2014; Johnston, 1991; Meier et al.,

13 2012). The input of nutrients is the sum of inflows from outer areas, rivers, land runoff, point

14 sources and atmospheric load, while the outflow of nutrients is the export from the area to

15 outer seas (Fig. 1). $N_2$-fixation is another process that needs to be taken into account as it is a

16 source of bioavailable N to the system. Retention in the present study can be temporal or

17 permanent. Permanent retention removes the nutrients permanently from the pool of nutrients

18 in the modelled system. Burial is the only retention process that permanently removes P. For

19 N, in addition to burial, also benthic and pelagic denitrification is considered as permanent

20 removal. The temporal retention during a studied period can be negative or positive

21 depending on changes in the pelagic and benthic inventory of nutrients. The nutrient pools

22 include both the inorganic and organic nutrients. Factors that affect the benthic N and P pools

23 are the sedimentation of organic material from the water column, the decomposition of

24 organic material and the release of inorganic nutrients back to the water column, as well as

25 burial of nutrients. The pelagic N and P pools are affected by the supply from land, the export

26 of organic material to the sediment, the release of nutrients from the sediment to the water

27 column and to the net export of nutrients to downstream areas.

28 The different processes that affect retention have been calculated separately, as they are

29 included in the biogeochemical model SCOBI. Total retention ($R_{tot}$) is the sum of both

30 permanently and temporally retained P and N. Area specific retention, $R_{AS}$, is the retention

31 normalized to the area and is calculated from eq. 4:

1    $R_{AS} = \dfrac{R_{tot}}{A}$       4

2    where $A$ is the size of the area. $R_{AS}$ can be used to compare the retention in basins of different

3    sizes. The filter efficiency, $F_{eff}$, is calculated from eq. 5:

4    $F_{eff} = \dfrac{R_{tot}}{Nu_{land}} \times 100$       5

5    where $Nu_{land}$ is the sum of the nutrient load from land and the deposition from air. The $F_{eff}$ is

6    an estimate of the proportion (%) of the nutrients from land and atmosphere that is retained

7    within the area. Similarly the retention efficiency ($R_{eff}$) can also be calculated, defined as the

8    proportion of the total nutrient load (sum of all sources, including import from surrounding

9    waters) that is retained within the area. In the present study, however, the focus is on the filter

10    efficiency.

11    The total retention efficiency was calculated for the entire Stockholm Archipelago, and also

12    separately for the inner, intermediate and outer archipelagos in order to investigate the spatial

13    gradient of retention capacity from the inner coastal zone towards the open Baltic Sea.

14    The residence time is defined as the average time water, or a dissolved substance, spends

15    within a particular basin (Bolin and Rodhe, 1973). In the present study the residence time of

16    the freshwater is calculated to relate the filter efficiency to physical characteristics of the

17    archipelago as described by Nixon et al. (1996). A fresh water tracer in the model is used to

18    determine the freshwater volume ($V_{qf}$) in the different parts of the archipelago. The fresh

19    water residence time is estimated by the flushing time calculated from the freshwater volume

20    divided by the fresh water discharge received from land ($Q_f$) as in the freshwater fraction

21    method discussed by Sheldon and Alder (2006). The filter efficiency was calculated for; the

22    inner, the sum of the intermediate and inner archipelago, and the entire Stockholm

23    Archipelago.

24    **2.4.1   Oxygen reduction scenario**

25    In the model, denitrification is an $O_2$ dependent process that has a maximum rate at $O_2$

26    concentration of about 45 µmol $l^{-1}$ (~1 ml $l^{-1}$) while denitrification halts under anoxic

27    conditions. Also P is affected by oxygen since P has an oxygen dependent adsorption

28    behaviour on particulate iron(III)oxyhydroxides (Mortimer, 1941). The adsorption of P on

29    particles can lead to higher burial rates during oxic conditions compared to anoxic conditions

30    when the release rate of P from the sediment is higher (Viktorsson et al., 2012; Almroth-

1    Rosell et al., 2015). This $O_2$ dependent adsorption behaviour is also simulated by the model

2    (Eilola et al., 2009) using a reduced release of P from the sediment when $O_2$ is present in the

3    bottom water. The effect of the $O_2$ concentration on the filter efficiency is studied in an

4    experiment where the $O_2$ concentration was reduced in the SCOBI model with a fixed amount,

5    134 $\mu$mol $l^{-1}$ (3 ml $l^{-1}$), during the simulation period (1990-2012).

6    ### 2.4.2 Nutrient load reduction scenario

7    The SCM is also used to investigate the effect of a reduction of the nutrient load from land to

8    the Stockholm Archipelago. The reductions are applied to the forcing from 2010 with a river

9    load of 4027 t N $yr^{-1}$ and 163 t P $yr^{-1}$, and a load from point sources of 1805 t N $yr^{-1}$ and 30 t

10   P $yr^{-1}$ to the entire Stockholm archipelago. Reductions of point sources were estimated from

11   realistic minimum discharge concentrations of N and P from sewage treatment facilities based

12   on technical feasibility, but not on economic or resource sustainability (Table 2, Kerstin

13   Rosén Nilsson, County Administrative Board of Stockholm, personal communication). Point

14   sources from different industries are assumed to decrease their discharge of N and P by 10 %.

15   The minimum discharge concentrations, and the 10 % reduction from industries, resulted in

16   reductions by approximately 51 % of N and 34 % of P from point sources. The reductions of

17   N and P from land runoff, e.g. due to decreased nutrient load from agriculture and increased

18   use of small sized sewage treatment plants by individual households, are set to 15 % for N

19   and 10 % for P. The combined reductions in rivers and point sources results in a total decrease

20   of N and P load by 20 % and 12 % relative to 2010. A SCM model spin-up run period of 45

21   years, with forcing from year 2010 provides the steady state initial conditions used for the

22   reduction experiment. After the spin-up period the reductions of the nutrient loads are

23   implemented.

24   *Table 2.*

# 3 Results and discussion

## 3.1 Validation

The variability of the modelled discharge of water and nutrients by the S-HYPE model agrees well with observations (Fig. 5 and Table 3) for the simulated period (1990-2012). A good description of river runoff is needed because the nutrient loads are strongly related to the magnitude of river outflow ($Q_F$) as seen in Fig. 5. The model seems to slightly underestimate the spring discharge and overestimate low flow regimes relative to observations. However, overall it captures a realistic annual variation of the discharge, which is reflected in high correlation coefficients (Eq. 2) for all evaluated parameters (Table 3). Highest correlation coefficients are found for $Q_F$ and TN, compared to the slightly lower values for TP, DIP and DIN, which is in accordance with previous studies (Grimvall et al., 2014; Strömqvist et al., 2012). An extensive validation is also available in Sahlberg et al. (2008).

*Fig. 5.*

*Table 3.*

Datasets from eight stations (Table 1) fulfilled the requirements of good data availability and were used in the evaluation of the SCM model results. There are aspects that are important to have in mind when comparing model results and observations. In the model the state variables are horizontally averaged in each basin, while observations are measured at one station at a certain location. The Stockholm Archipelago has relatively large spatial salinity gradients and the representativeness of a station when compared to model results can be somewhat limited if e.g. the position of the station is close to an out- or inlet of the basin. Observations may in general also be influenced by local conditions, e.g. sewage effluents, high sediment fluxes or stagnant conditions, which are smeared out in the average results of the model. Still we assume for the present study that the station data are good enough for the quantitative model validation and give a background for discussions about model strengths and weaknesses. As an example, validation results are shown for one of the basins where the number of observations is large enough both during summer and winter periods to be included in the validation process. The example is from station Blockhusudden (Position G in Fig. 3), where the largest data set of observations was found. The station is situated at the boundary between the innermost basin Strömmen and the next adjacent sub-basin.

The objective correlation coefficients (Eq. 2) and the cost function value (Eq. 3) for the different state variables implied correspondingly that the model manage to simulate the average vertical winter and summer profiles with good or acceptable skills in the basin Strömmen (Fig. 6g), except for the average seasonal value of DIN that was described as not good. The differences between model results and observations of DIN may be a result of the location of the monitoring station.

The long term average summer depth profiles of modelled salinity and oxygen in the basin Strömmen correlate well with observations, while the winter values of salinity were too low, especially in the surface layers (Fig. 7a,b). This difference is partly due to the fact that the salinity of a station at the entrance to the basin is more reflecting the boundary conditions of the downstream basin than the mean conditions in Strömmen. The surface winter concentrations of oxygen were too high, but decreased with depth and became too low in the lower layers (Fig. 7b). It might be expected that winter surface oxygen concentrations in observations should be higher than in summer because of the temperature effect on oxygen saturation concentrations as seen from the model results. However, the number of observations during winter are limited and occurred mostly in November and February, which may influence the average values of the observations.

The results indicate that there is an impact from local conditions at the monitoring station that are not captured by the model setup. The modelled DIN depth profiles show higher values at about 15 m depth during both winter and summer (Fig. 7c), while the DIP profiles values seems to be satisfactory at all depth and periods (Fig. 7d). Also the individual observations show higher concentrations of both DIN and DIP around 15 m depth which is where the halocline has its largest vertical gradient. This depth level corresponds to the depth where two sewage water treatment plants relieve their sewage water in the model. The winter stratification was stronger in the model because of the lower surface salinity. This hampers the vertical transports of oxygen and has an influence on the winter oxygen conditions in the deep water that were lower in the model compared to the observations from the more well ventilated entrance area.

The average seasonal variation of the surface temperature and the bottom water oxygen concentrations was captured by the model, but not the increase of surface nutrients, especially DIN, during autumn (Fig. 8). The surface salinity was overall somewhat low, which probably is a result of the location of the monitoring station, as described above.

In the other basins used in the evaluation (vertical and seasonal profiles are not shown) of the SCM state variables during winter, summer and season were simulated with good or acceptable skills, except for the average vertical summer profiles of DIN in the basin Solöfjärden (Fig. 6c) and oxygen concentration in the basin Sandöfjärden (Fig. 6a). The combined model skills, which were calculated as the average of the individual $r$ and C values, were good in six of the eight evaluated basins (purple cross in Fig. 6). In the remaining two basins the skills were considered as acceptable.

*Fig. 1.*

*Fig. 2.*

*Fig. 3.*

## 3.2 Retention of nutrients in the Stockholm Archipelago

The load and the inventories of N and P may change and vary between the beginning and the end of a studied period, thus the determination of total nutrient retention depends on the time scales of consideration as discussed in section 3.2.3. During the period 1990-2012 on average 174 t P $yr^{-1}$ and 5846 t N $yr^{-1}$ entered the inner archipelago, mainly from the Lake Mälaren. That is a major part of the 217 t P $yr^{-1}$ and 8288 t N $yr^{-1}$ which entered the entire Stockholm Archipelago (Fig. 9). The P load from point sources was clearly lower than the river load (Fig. 10). However, the N load from point sources was higher than the river load in the beginning of the studied period (Fig. 10b), but decreased in the middle of the 1990s due to the implementation of a more effective method to remove N in the waste water treatment facilities. The P supply to the intermediate archipelago mainly originated from runoff from land, while for N there were also some point sources that contributed to the land load on the same level. In the outer archipelago the nutrient load from land was almost negligible and most of the nutrients were deposited from the atmosphere.

*Fig. 9.*

*Fig. 10.*

Largest amounts of P and N in the model were retained in the outer archipelago compared to the intermediate and inner archipelagos (Fig. 9). The retentions of all supplied P and N, including the net import from upstream areas, within the inner, intermediate and outer Stockholm archipelagos amounts to 18 %, 23 % and 48 % for P, respectively, and 14 %, 26 % and 60 % for N, respectively. The area of the three zones increases from inner (109 $km^2$), to

1    the intermediate (759 km$^2$) and to the outer archipelago (2360 km$^2$) and thus, the retention of

2    nutrients seems to increase with increased area. On the other hand, the average of the area

3    specific retention of P and N was for the simulation period highest in the inner archipelago,

4    and decreased towards the open sea (Fig. 11). The permanent retention was relatively stable

5    during the simulated period, while fluctuations in the temporal retention reflect the effect of

6    varying riverine nutrient input (Fig. 10c, d). The water depth and the residence time affect the

7    retention of nutrients, which will be further discussed in Section 3.2.2. The largest part of the

8    total retention in the entire Stockholm Archipelago was permanent, which for P means burial.

9    For N benthic denitrification represented as much as almost 92 % of the permanent retention,

10   burial for less than 8 % and pelagic denitrification was below 1 %.

11   *Fig. 11.*

12   Karlsson et al. (2010) found in their empirical study for 1982-2007 that about 15 % of the

13   total input of N and 10 to 13% of the total input of P were retained in the inner Stockholm

14   Archipelago. However, their numbers are based on the total input, thus both the land load and

15   an estimated input from outer areas, i.e. the intermediate Stockholm Archipelago. A

16   recalculation from the given numbers in their study resulted in a filter efficiency of about 25

17   and 24 % for N and about 21 and 30 % for P of the nutrient load from land and atmosphere

18   for the periods 1982-1995 and 1996-2007, respectively. These numbers of the filter efficiency

19   are higher than the numbers in the present model study. To be able to compare the numbers a

20   recalculation of the filter efficiency in the SCM for the latter period (1996-2007) in the inner

21   archipelago was performed, but did not change the SCM results considerably. The largest

22   difference between the two studies is caused by the calculation of net exchange of nutrients

23   through the sounds. The transport through the sounds was in Karlsson et al. (2010) calculated

24   from average volume flows estimated from mass balance calculations for salt together with

25   budget calculations using observations of average nutrient concentrations. In the present study

26   the exchange of nutrients between the inner and the intermediate archipelago was part of the

27   dynamic model calculations in the SCM. The SCM net outflow from the inner archipelago for

28   N and P was about 11 % and 8 %, respectively, larger compared to the net outflow of the

29   nutrients in the Karlsson et al. (2010) study. Another difference between the two studies was

30   the land load of P, which was about 8 % lower in the SCM. The difference in land load of N

31   was only about 1 %. Thus, calculations from an empirical model based on Knudsen´s

32   relations (Knudsen, 1900) and calculations using long term average values resulted in about

33   10 % higher retention efficiency values compared to the calculations from SCM, a coupled

1    numeric physical-biogeochemical model with high vertical resolution and small time step. In

2    spite of the difference in models the result are surprisingly close.

3    The average temporary retention in SCM for the entire simulated period is negative in all

4    three parts of the archipelago for both P and N (Fig. 9 and Fig. 10). The reason for negative

5    temporary retention is mainly a decrease in the benthic nutrient pools during the period (Fig.

6    12). The largest decrease (29 %) is found in the pelagic pool of N in the inner archipelago,

7    which coincides with the decrease in N load from point sources (Fig. 10). In the intermediate

8    and outer Stockholm archipelagos the pelagic pool of N remains on about the same level

9    through the whole simulation period. The large decreases in the benthic pools of N and P (14-

10    18 %) occur in the intermediate and outer archipelagos, while there are only small changes in

11    the pelagic and benthic pools of P in the inner archipelago. Because of the nutrient retention

12    there is a reduced net transport of N and P from the inner archipelago towards the

13    intermediate and outer archipelagos and further to the open sea during the simulated period

14    (Fig. 9). The annual temporary retention of P in the entire Stockholm Archipelago increases

15    with time during the simulated period (Fig. 10). There is a change to positive values at the end

16    of the period, when there again is a build-up of the benthic pools of P (fig. 12). The build-up

17    is most likely a result of better oxygen conditions in the modelled deep water (not shown)

18    during the end of the simulation period, which lead to a lower release of P from the sediment

19    to the water column (Eilola et al., 2009). For the temporary retention of N there is no visible

20    trend in the variation with time. In addition to the nutrient load from land, and the net export

21    of nutrients to outer areas, there is also an extensive circulation of nutrients between the coast

22    and the open sea. The importance of imported nutrients into the coastal zones from sea have

23    been discussed in earlier studies (e.g. Humborg et al., 2003) in which it was concluded that

24    many estuaries has a net import of DIN and DIP from sea, e.g. Chesapeake Bay (Boynton et

25    al. 1995). This is shown also for e.g. the Mid Atlantic Bight where almost three times the

26    riverine input of N is denitrified (Fennel et al., 2006). In different parts of the shelf in the Gulf

27    of Mexico the denitrified proportion of the land input of N is in total 86 %, where locally on

28    the different part of the shelves the denitrification fraction of the supply from land varied

29    between 68 % and 341 % (Xue et al., 2013). Thus, in many cases the import is larger than the

30    export and the coastal zones works as a filter not only for the nutrients from land, but also for

31    the nutrients from the open sea as also discussed in section 3.2.3.

32    *Fig. 12.*

### 3.2.1 The coastal filter

From the present results it can be concluded that the Stockholm Archipelago works like a filter for nutrients that enter the coastal zone from land and atmosphere. However, a rather large area of the archipelago is needed to effectively retain the nutrients. About 82 and 86 % of P and N supplies, respectively, pass the small inner archipelago and are exported to the intermediate archipelago. In the intermediate and the outer archipelago all local supplies of nutrients from land and atmosphere are retained together with a fraction of the nutrients imported from the inner archipelago. The filter efficiencies increase with increased coastal area from land to the sea continuum (Fig. 13). However, the filter efficiency of the entire Stockholm Archipelago is not effective enough to retain of all the nutrients that enter the system from land and the atmosphere, but still, at least 65 % and 72 % of the supplied P and N, respectively, are retained. The total retention numbers (permanent and temporary) correspond to 141 t P yr$^{-1}$ and 5954 t N yr$^{-1}$ (Fig. 9). Since Stockholm Archipelago is the largest archipelago in Sweden it might be that most of the other Swedish coastal areas with a large run-off from land would be less effective as coastal filters and, thus, contribute to a larger extent to the eutrophication in the open sea. This is one question in focus of an on-going study where the entire Swedish coastal area will be evaluated similarly to the present study.

### 3.2.2 Processes affecting retention

The present study was performed in an area characterised as an eutrophic archipelago in an inland sea with basins having oxic, hypoxic and anoxic bottom waters. Nixon et al. (1996) showed that the retention of P and N correlated to the log scale of the ratio between the average depth and the residence time of the study areas, which is confirmed by the results from the studies by Billen et al. (2011), Hayn et al. (2014) and Nielsen et al. (2001) as well as by the present study (Fig. 13). The freshwater residence time in the Stockholm Archipelago is 48 days in the inner, 108 days in the middle and inner, and 185 days in the entire area. No clear relationship was found between the filter efficiency and the average depth, which vary between 17 m and 20 m for the three areas. These results are in agreement with Nixon et al. (1996) who showed that including the depth in the analysis of retention vs residence time did not much improve their regression. In the present study the change of the filter efficiency with residence time is about 0.5-0.6 % per day. The results of the present retention estimates are in agreement with results from previous studies (Billen et al., 2011; Hayn et al., 2014; Nielsen et

al., 2001; Nixon et al., 1996), but with somewhat higher values in the entire archipelago (Fig. 13). Their studies were performed in various types of systems: coastal lagoons, drowned river estuaries, coastal embayments, and inland seas in North America and in Europe. Those systems varied from being relatively pristine to systems with large point sources (eutrophic), and they also varied between oxic to hypoxic and/or anoxic conditions. In shallow areas larger parts of the sinking particulate organic material may reach all the way down to the sea floor where it can be exposed to retention processes such as burial and denitrification. On the other hand, in a much deeper area a larger part of the organic material may become re-mineralised within the water column on its way down to the sea floor. The nutrients can then be re-used by phytoplankton and/or be further transported out from the system. Long residence times in a system increase the time of exposure for biogeochemical transformation processes and sedimentation within the system and larger parts of the nutrients may be retained.

*Fig. 13.*

Denitrification increase the retention in areas with longer residence times (Nixon et al., 1996, Finlay et al., 2013) as also seen from Fig. 13. In the Randers Fjord the residence time was short (sex days) and the filter efficiencies of N and P were lower, 10 % and 9 %, respectively (Nielsen et al., 2001), compared to the Stockholm Archipelago where the freshwater residence time is longer. The denitrified proportion of the permanently retained N was also lower, about 60 % compared to in the Stockholm Archipelago (92 %). Oxygen is an important factor regulating the magnitude of denitrification. In waters with longer residence time the bottom water might be less ventilated, and, thus, the bottom water oxygen concentrations lower with higher denitrification as a result. As a result of the forced reduction of the oxygen concentrations with 134 μM the hypoxic areas increased by 49 km$^2$ (300 %) and the anoxic area increased by 13 km$^2$ (360 %) in the entire Stockholm Archipelago. The reduced oxygen concentration led to increased N retention (780 t yr$^{-1}$ or 14 %) due to increased denitrification and to decreased P retention (49 t yr$^{-1}$ or 28 %) due to higher release of P from the sediment. Denitrification increased the fraction of permanent retention from 92 % to 94 %, while the buried fraction decreased. The inner archipelago had the largest increase of hypoxic and anoxic areas and also the largest changes in retention of N and P. The N retention increased there by 243 t yr$^{-1}$ (29 %) and the P retention decreased by 9 t yr$^{-1}$ (38 %).

Benthic primary producers and benthic fauna are also important for the retention of nutrients in shallow coastal ecosystems (McGlathery et al., 2007; Norkko et al., 2012). Assimilation of nutrients during primary production does not directly change the inventory of N and P, but

transfer the nutrients into organic material. Plant uptake at the bottoms can e.g. lead to increased burial and also influence on the oxygen dependent biogeochemical processes in the sediment due the plant metabolism (McGlathery et al., 2007).These processes are not yet implemented in the SCM that only include pelagic primary production, and are therefore not included in the present study. Including these processes may have some impact on the model dynamics e.g. on bottoms where seagrasses and burrowing macrofauna might influence the decomposition of organic material and the permanent burial of nutrients and organic matter. The evaluation of forcing and model results indicate, however, that the model system is able to reproduce much of the observed physics and nutrient dynamics in the archipelago which give confidence to the budget estimations of nutrient retention in the area. A quantitative evaluation of the effect and the implementation of benthic flora and fauna to the model are therefore left for future work.

It is also important to know whether a system is in balance with the nutrient loads or not since it would affect the retention capacity. In this study the temporary retention is negative for both N and P in all three areas of the Stockholm Archipelago which implies that the system is not in a steady state. This imbalance is however expected since there are reductions of the nutrient loads in the first part of the simulation period (Fig. 10a, b). However, the possibility that the results may be influenced by unknown initial conditions of sediment concentrations should not be excluded. There are only few observations available and the knowledge about the amount of sediment nutrients involved in biogeochemical cycles is poor.

### 3.2.3 Response to nutrient load reduction

The fastest response in the nutrient load reduction experiment is seen in the pelagic pool of N which rapidly decreases, but reaches a steady state after about three years with reduced loads (Fig. 14). The pelagic pool of P decreases in the inner archipelago but increases slightly in the outer areas. The changes in P pools are slower compared to those in N pools. The large and fast decrease of pelagic N in the inner archipelago, results in a decreased N:P ratio (Table 4), as well as (not shown) lower chlorophyll concentrations, reduced sedimentation, and increased export of P from the inner archipelago to the outer areas and the Baltic proper. Also the anoxic areas decrease by about 30% as a result of the lower deposition of organic material on the sea floor (not shown). The changes in the benthic pools of N and P occur over a longer time period, and the benthic P pool does not reach a steady state until about 40 years after the reduction.

1    In the reduction scenario the transport of N to the open sea from the Stockholm Archipelago

2    decreases by 62% within four years (Table 4). The filter efficiency of N in the entire

3    archipelago increases at the same time from 79 % to 90 % as a result of the load reduction.

4    The longer response time of P compared to N is observed also in the filter efficiency (Fig. 14).

5    The filter efficiency of P at the end of the spin-up run is about 100 %. This implies that under

6    the 2010 conditions, all the P land load is retained in the Stockholm Archipelago when the

7    system is in steady state. This is not the case when the original model forcing is used, which

8    implies that the Stockholm archipelago is still adjusting to the load reductions already

9    implemented. Thus, the coastal region might under present conditions continue to improve

10   without further actions.

11   The filter efficiency of P decreases to 74 % during the first years after the reduction,

12   coinciding with the large decrease in the N pelagic pool and the decrease in N/P ratio. After

13   the initial decrease, the filter efficiency slowly increases to 106 % at the end of the simulation

14   period, i.e. retention is larger than the land and atmospheric load of P. As a consequence the

15   export of P from the archipelago to the Baltic proper decreases with time, and about 18 years

16   after the load reduction the direction of the transport changes. This coincides with the time

17   when the filter efficiency again reached 100 %. Thereafter the archipelago begins to import P

18   from the open sea. Thus, with the contemporary boundary conditions used at the open sea, P

19   from the Baltic proper is retained within the archipelago. For coastal management this

20   indicates the importance of the open sea nutrient conditions when effects of load reductions

21   are evaluated.

22   These results indicate that local nutrient load abatements can improve the environmental state

23   of a semi-enclosed coastal site (the inner archipelago) that is locally impacted by humans. The

24   results also imply that for the first 5-15 years, increased nutrient concentrations might be

25   expected locally. However, this effect largely depends on the water residence time and on

26   which nutrient limits the seasonal phytoplankton production initially. However, for the more

27   open coastal zone, represented in the present study by the intermediate and outer archipelago,

28   the response to further nutrient load reductions was minor. This exemplifies that for open

29   coastal areas the interactions between the open sea and the coastal zone is probably more

30   important than the land-sea connection.

31   The present study can conclude that even the eutrophicated Stockholm Archipelago can, after

32   further nutrient load abatements, act as a sink for open water phosphorous. Similar behaviour

1    was found in the Chesapeake Bay (Boynton et al. 1995), which acts as a sink for the total load

2    of P, thus, P from land, atmosphere and from the open sea.

3    *Table 4.*

4    *Fig. 14.*

# 4 Conclusion

Archipelagos are complex areas with many basins and several shallow sounds, which affect the transport of water and the dissolved and particulate nutrients. For the first time the SCM model was used to study the capacity of the coastal filter of nutrients. An evaluation showed that overall, model results agree with observations.

We focused our study in the northern Baltic proper and investigated retention of N and P in the Stockholm Archipelago. The main findings are described below.

- The coastal zone works as an efficient filter for the land loads of nutrients. Under prevailing conditions the total retention are 65 % and 72 % of P and N, respectively, supplied from land.
- A sensitivity experiment reducing the land load of nutrients showed that the retention capacity of N and P increased. In this case the export of N from the archipelago decreased and P was imported from the open sea.
- The average filter efficiency is dependent on the spatial dimensions of the coastal area. Thus, nutrient retention per area is largest in the inner archipelago and decreases towards the open sea.
- Average water depth and water residence time regulate the retention of nutrients that occurs mostly in the sediment due to processes such as burial and denitrification.
- The pools of nutrients in the water and in the sediment changes with nutrient loads on different time scales and affects the temporal nutrient retention in the area. N has a rather short response time of about three years while it takes about 40 years for P to reach balance in a system with constant forcing. Changing N:P ratios in the archipelago due to the different response time scales also have an impact on the nutrient retention capacity on decadal time scales.
- Coastal management needs to take the aspects of time and balance between nutrient loads and pools into account in the assessment of impacts from nutrient load abatements. On shorter timescales the retention capacity of P might be less effective when the nutrient load from land decreases.

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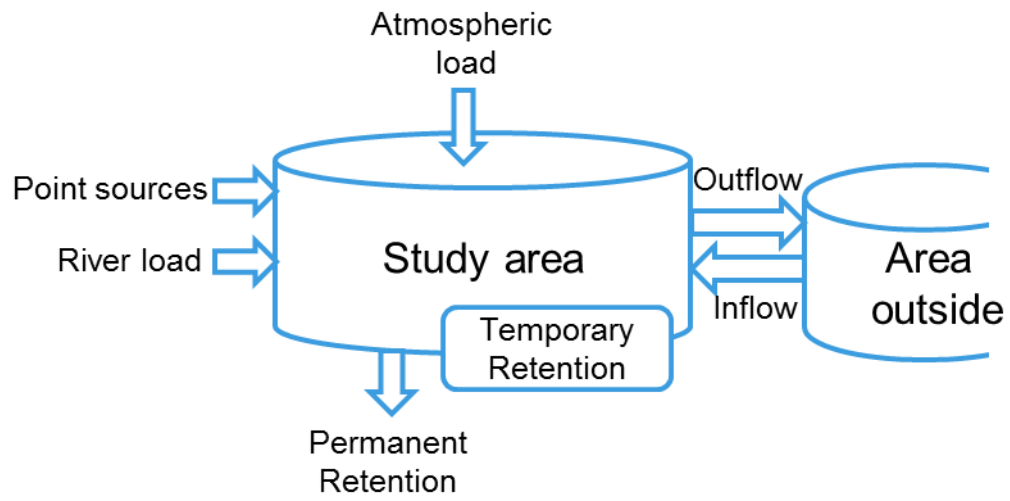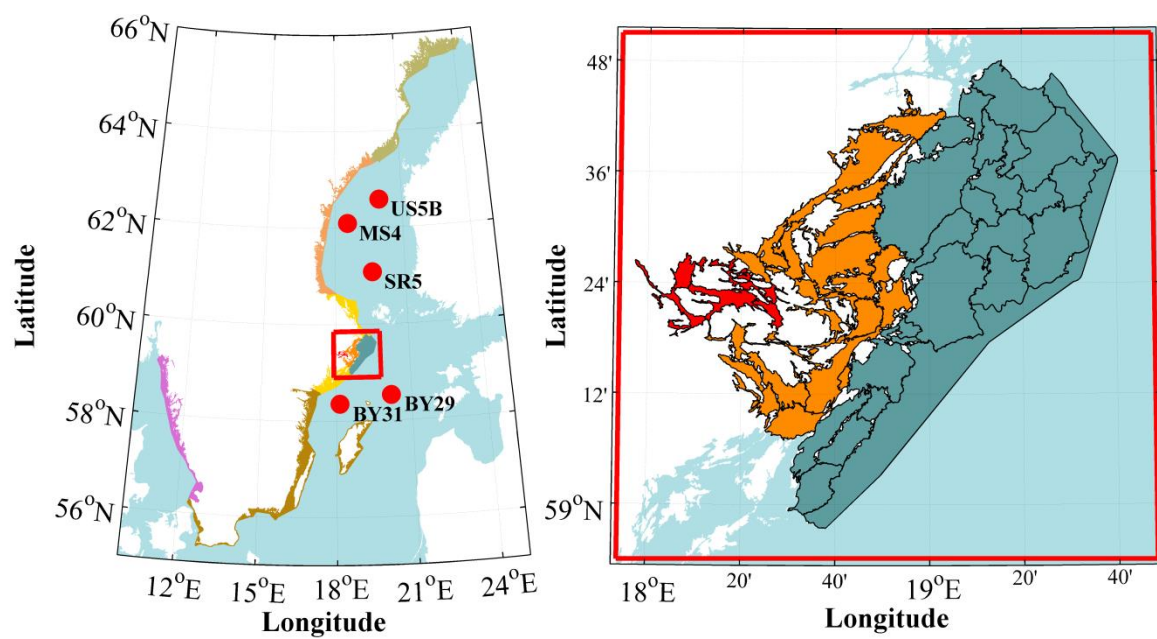
## 7 Figures

Fig. 1.

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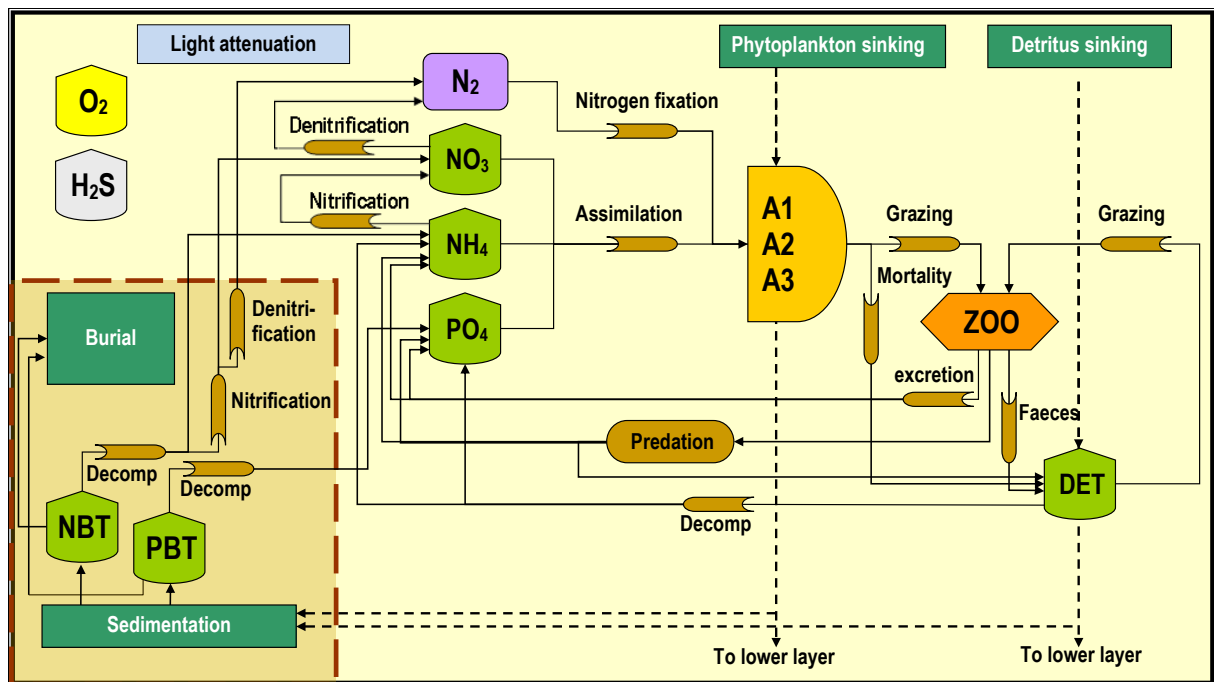
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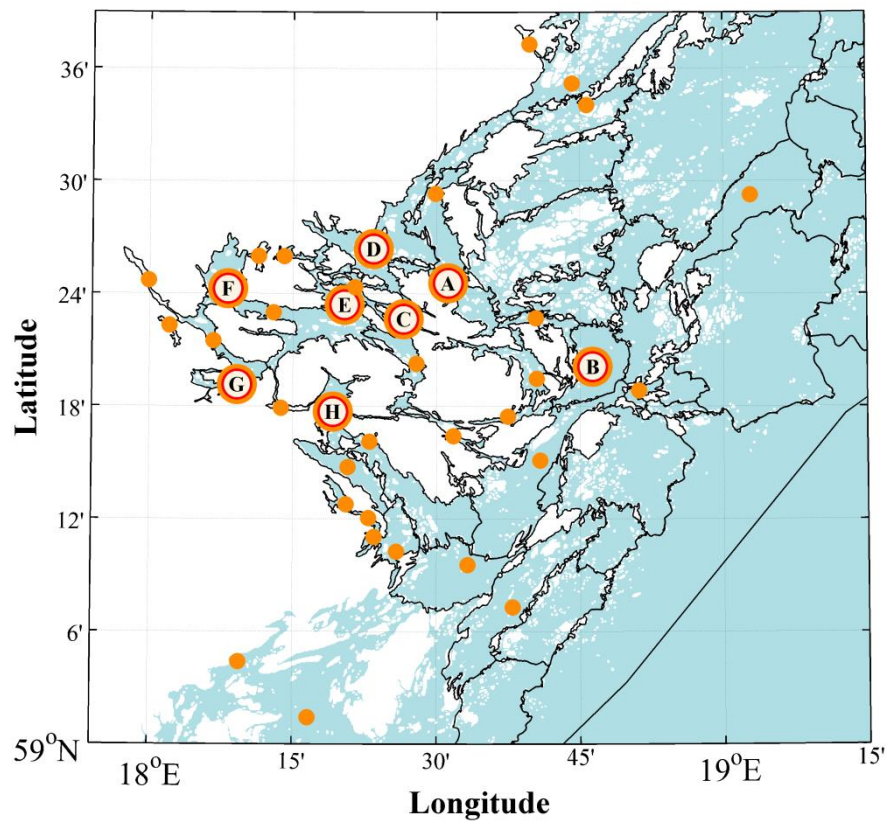
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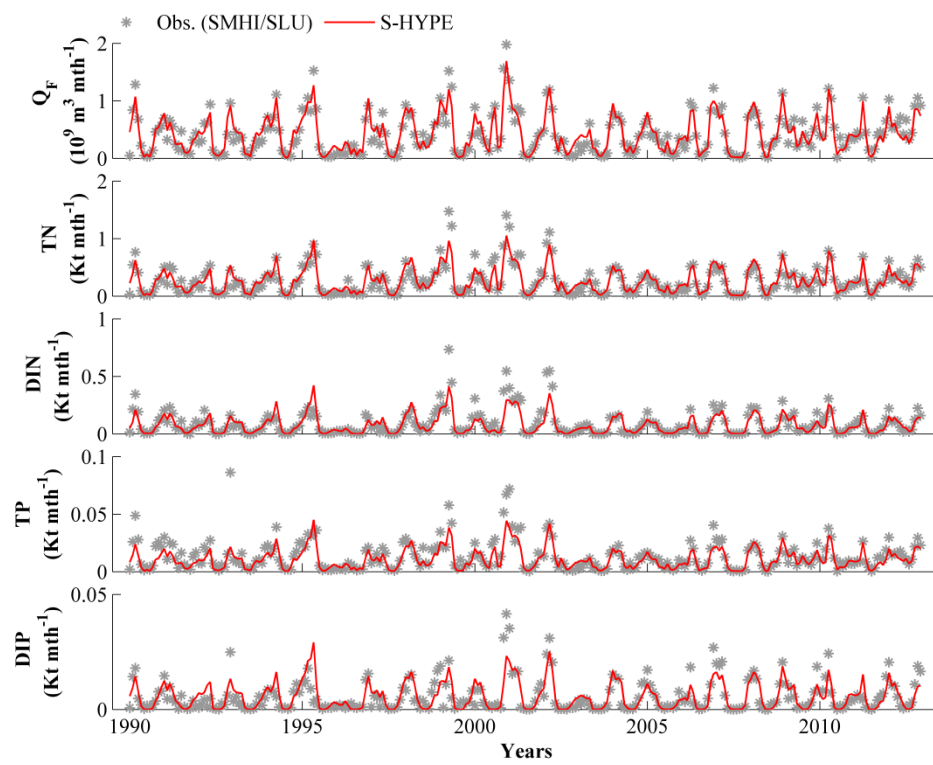
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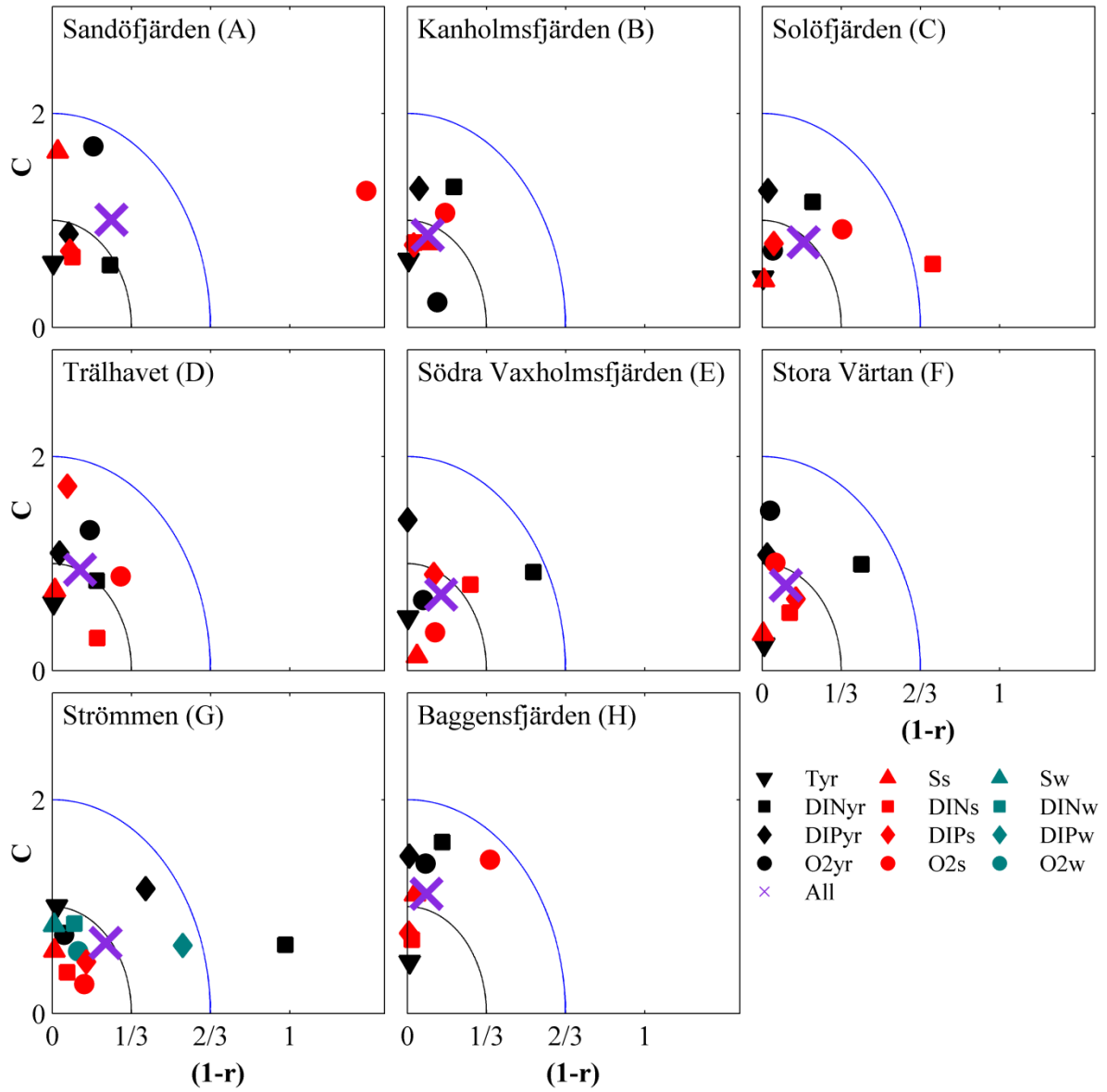
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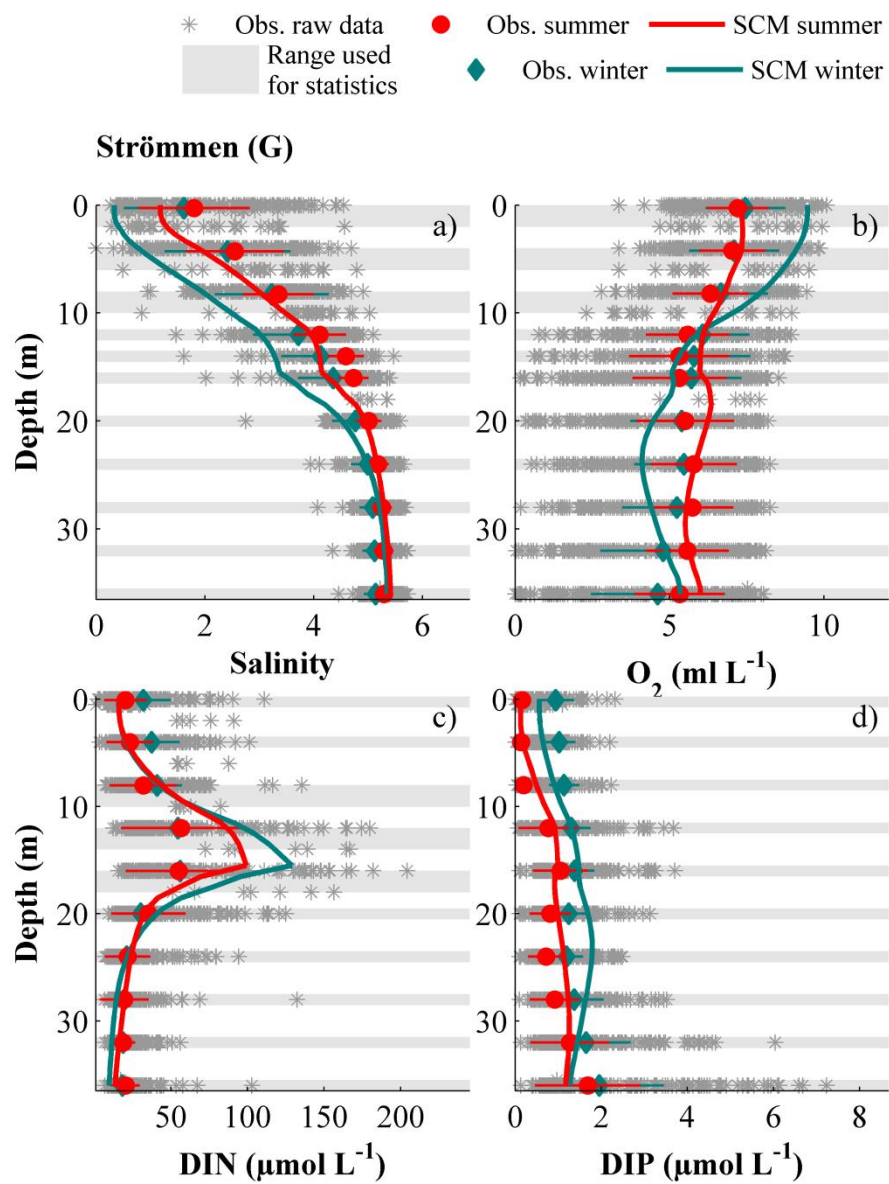
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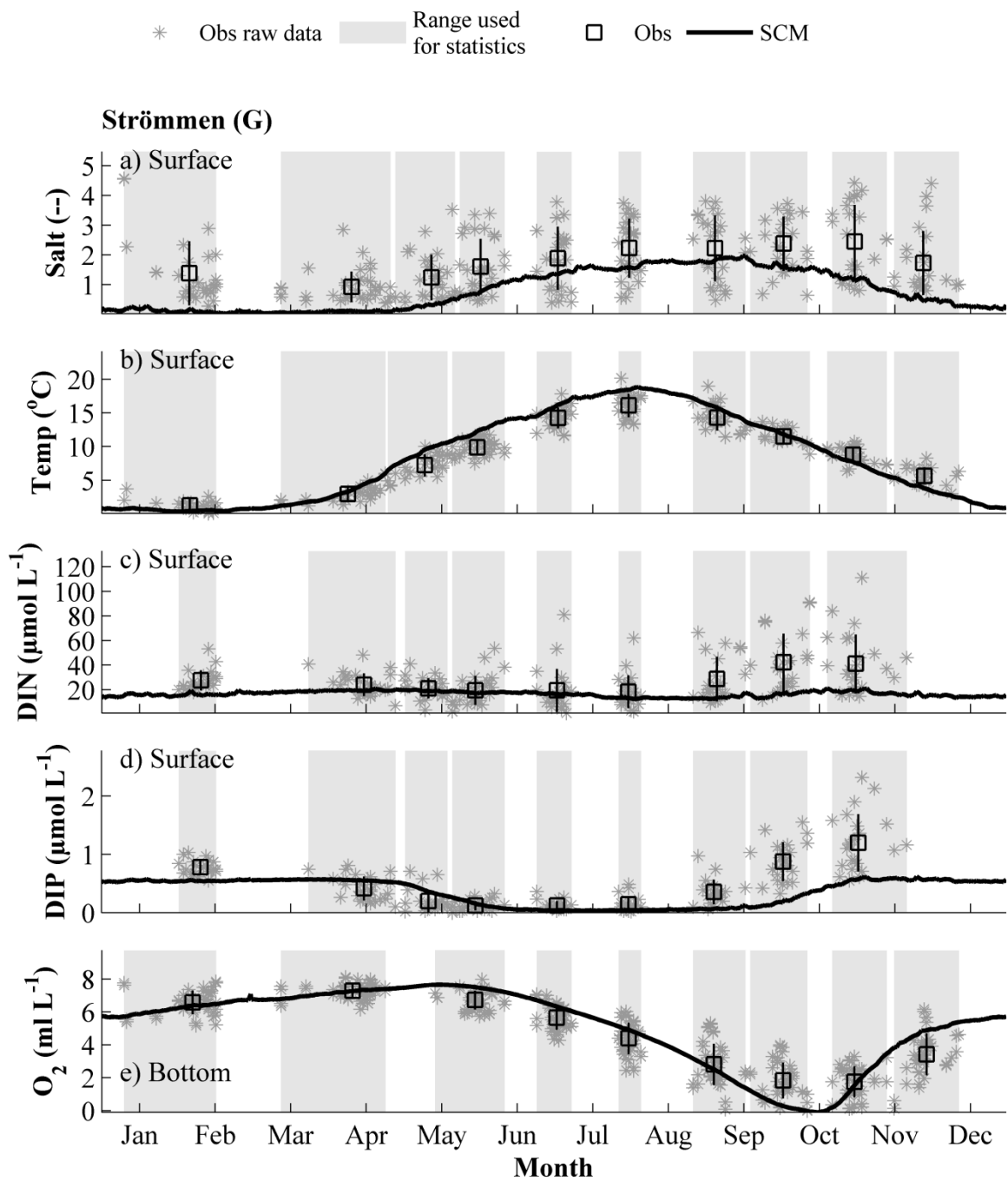
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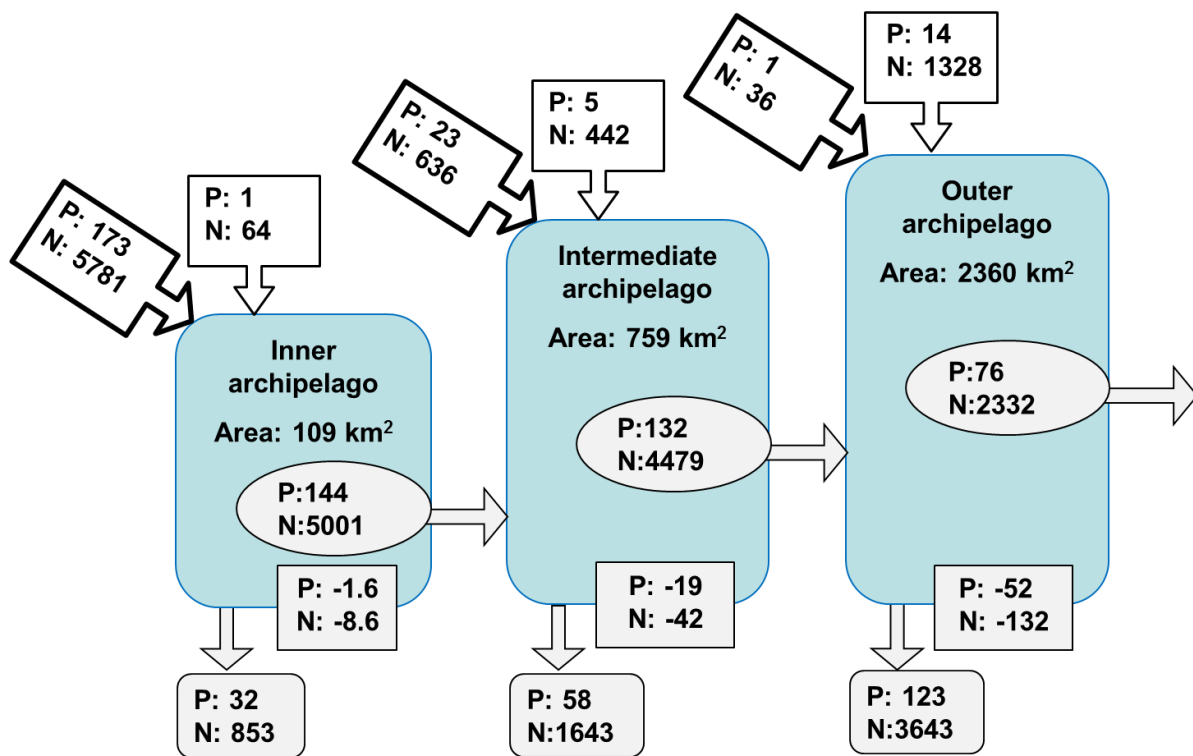
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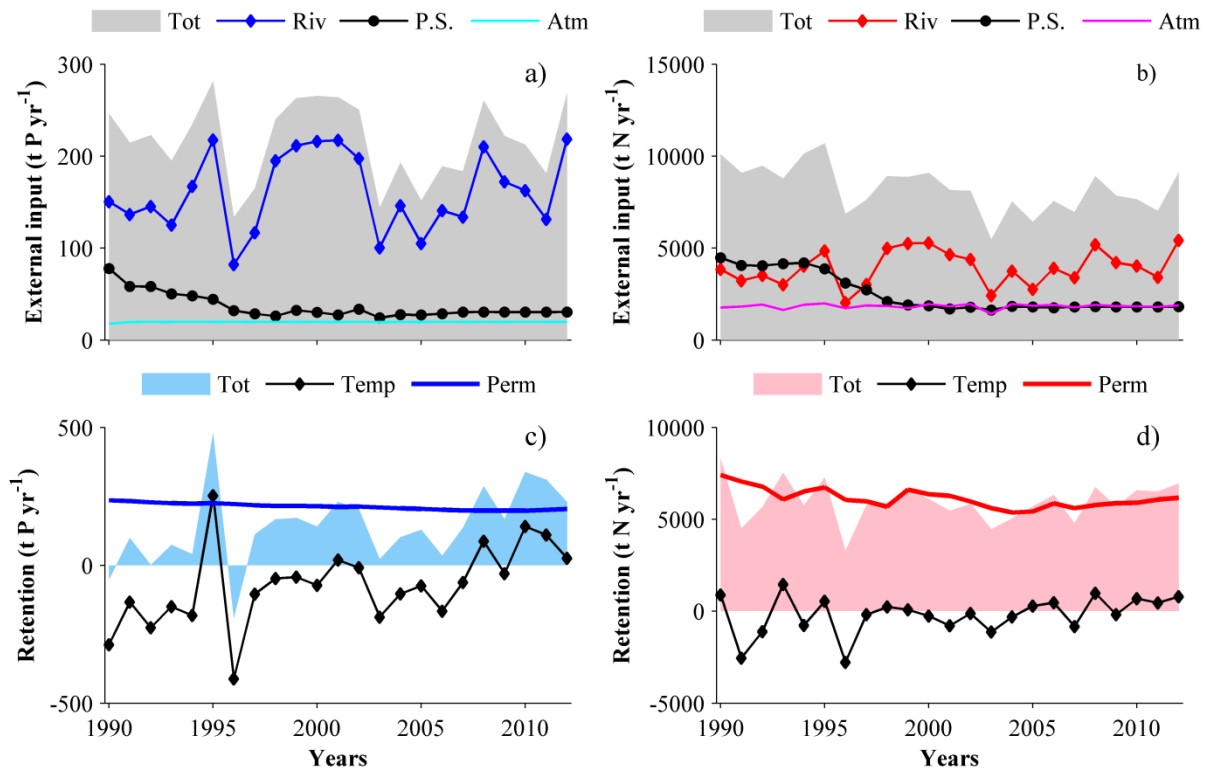
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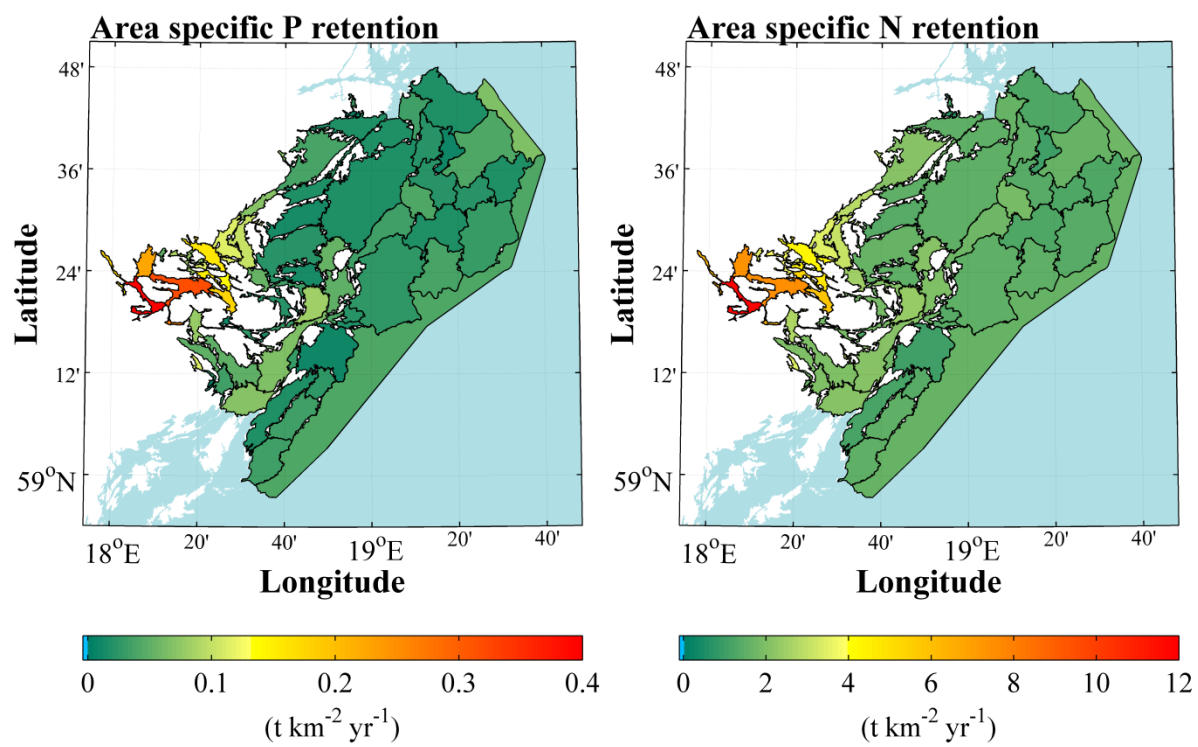
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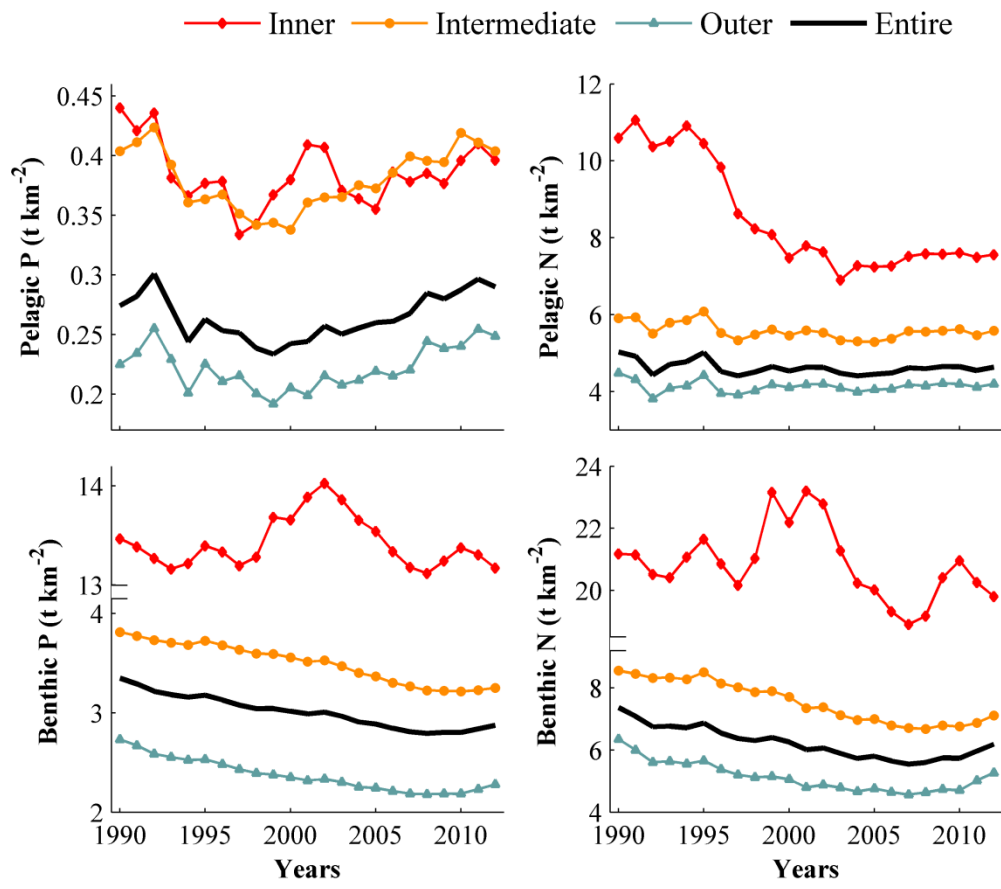
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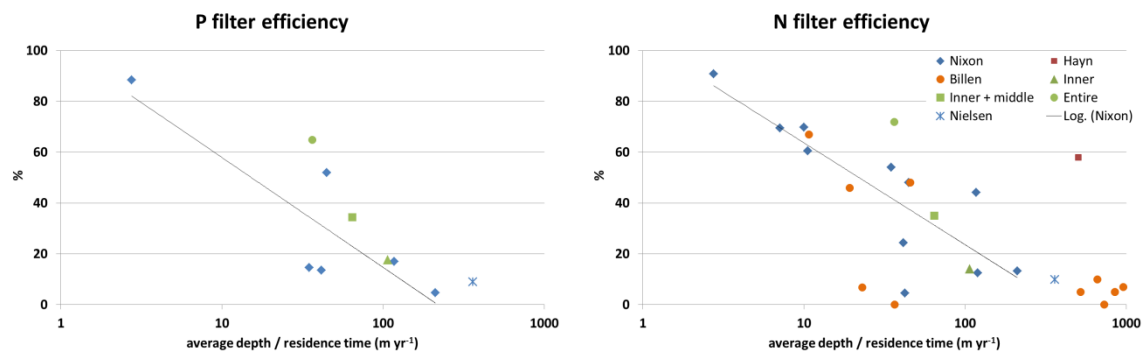
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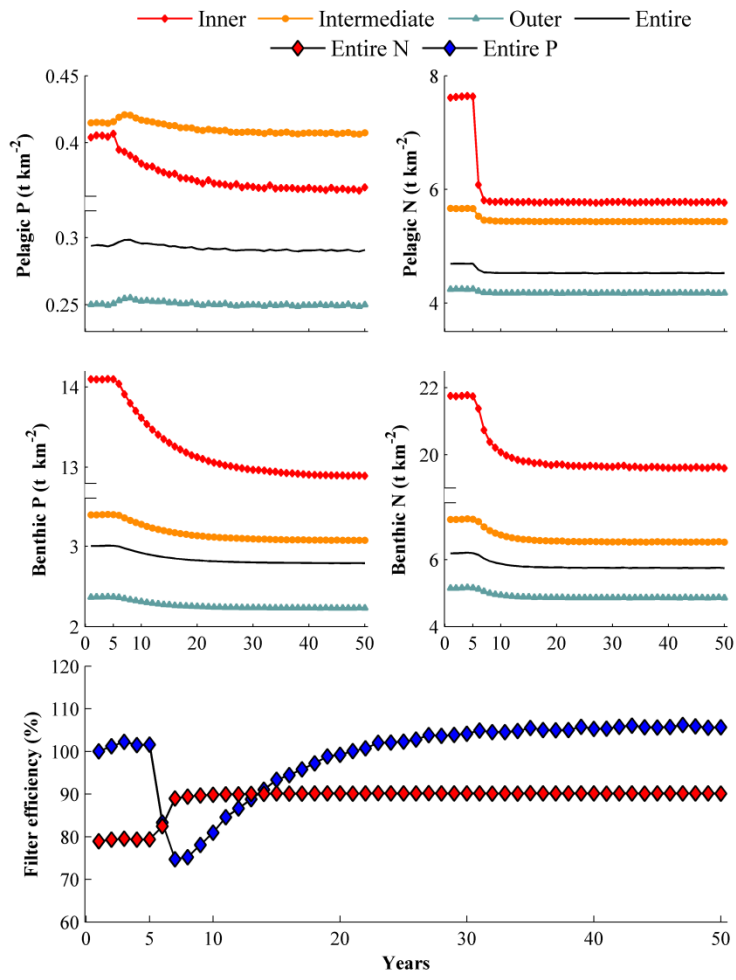
Fig. 14

# 8   Figure captions

*Fig. 1.  Simplified scheme of the retention calculations in the study area. Permanent retention is considered as a permanent removal of nutrients from the ecological system and includes burial and for nitrogen also denitrification. Temporary retention is defined as the changes in nutrient inventory in the active sediment layer and water column. The temporary retention may change sign depending on whether the nutrient inventory increases or declines.*

*Fig. 2. The Swedish Coastal zone Model can be used in different areas along the Swedish coast streching from the Norwegian border in the West to the Finnish border in the North (different colours, left). In the present study the SCM model covers the northen Baltic proper (marked with a red square) and has been used to estimate the coastal filter efficiency of nutrients in the Stockholm inner (red), intermediate (orange) and outer (blue) archipelagos (right).The outlet of river Norrström is marked with a black arrow and the different basins are shown by the black contours.*

*Fig. 3. Schematic figure of the Swedish COastal and BIogeochemical model, SCOBI. Oxygen and hydrogen sulphide are simplified for clarity.*

*Fig. 4. Available locations with observations (circles and dots) in the Stockholm Archipelago. Model evaluation of temperature, salinity, DIN, DIP and bottom water oxygen concentration was performed at selected stations (circles marked with letters), which are described in Table 1.*

*Fig.5. Observed (stars) and modelled (line) monthly outflow ($Q_F$) and nutrient loads from Lake Mälaren, through Norrström to basin Strömmen for the modelled period (1990-2012). DIN is the sum of nitrate and ammonium.*

*Fig.6. Average cost function (C) and correlation coefficients, adjusted (1-r) to the range 0-1, for an overview of the model skill at the eight different validation sites (A-G). The individual skills of the different parameters, average seasonal variation (black) and/or the vertical summer (red) and winter (turquoise) profiles of temperature (T), DIN, DIP and oxygen concentrations ($O_2$) are shown, as well as the combined model skills for all variables (purple cross). Variables within the inner quarter circle and between the two quarter circles are considered to be good and acceptable, respectively, while variables that are outside the quarter circles are not well simulated.*

*Fig.7. The SCM modelled (lines) and observed (circle and diamond) vertical average profiles (1990-2012) of salinity (a) and concentrations of oxygen ($O_2$; b), DIN (c) and DIP (d) in the basin Strömmen during winter (turquoise) and summer (red) months. Depth layers with dense number of observations (grey stars) determined the vertical depth intervals (grey shaded area) used in the profile calculations. The standard deviations (horizontal lines) were calculated for the summer and winter values of the observations.*

*Fig.8. Simulated (lines) and observed averages (squares) of the seasonal variation and the standard deviaton (vertical lines) of the observations in the basin Strömmen (1990-2012) of surface temperature (Temp), salinity, DIN and DIP and of the bottom water oxygen concentrations. Time periods with dense number of observations (grey stars) determined the time intervals (grey shaded area) used in the calculations.*

*Fig.9. Transport scheme of N and P (t $yr^{-1}$) from land (leaning top boxes) and atmosphere (top boxes), and the net exchange from the inner, intermediate and outer archipelago (ellipse) towards the open sea. Total retention is the sum of temporary retention (square) and permanent retention (square with round corners). For P burial is the only process that leads to permanent retention, while for N also denitrification removes N. Negative values for the temporary retention means a decrease in the benthic and/or pelagic pools of nutrients.*

*Fig.10. The external annual load and retention (t yr⁻¹) of P (a, c) and N (b, d) in the entire Stockholm Archipelago for the period 1990-2012. Total load (shaded area) and the contributions from the different sources; rivers and land run off (diamonds), point sources (circles) and atmosphere (solid line) is shown on the top row. The total retention (shaded area) as a sum of permanent retention (solid line) and temporary retention (diamonds) (c, d) is shown on the bottom row.*

*Fig.11. The retention per area unit (t km⁻² yr⁻¹) of P (left) and N (right) in each basin of the Stockholm Archipelago.*

*Fig.12. The total content (g m⁻²) of the pelagic (top) and benthic (bottom) P (left) and N (right) in the inner (diamonds), intermediate (circles), outer (triangles) and entire (black line) Stockholm archipelagos.*

*Fig.13. The filter efficiency of P (left) and N (right) versus the logarithmic ratio between the average depth and the freshwater residence time of the study areas (month yr⁻¹). Data from other studies are from Billen et al. (2011), Hayn et al. (2014), Nielsen et al, (2001) and Nixon et al. (1996). The straight line shows the logarithmic regression for the data from Nixon et al. (1996).*

*Fig.14. Pelagic (upper) and benthic (middle) pools of P (left) and N (right) in the inner (red), intermediate (orange), outer (turquoise) and entire (black) Stockholm Archipelago. The filter efficiencies (%) of N (red) and P (blue) load from land and atmosphere are shown for the entire Stockholm Archipelago (lower), where the small peaks derive from leap years.*

# 9 Tables

*Table 1. Number of sampling occasions (Occ) during the number of years, number of months during each year, and number of depths levels that was frequently sampled at the different stations used for validation of model results. The position of the stations can be seen in Fig. .*

| ID | Station name | Basin name | Occ | Years* | Months | Depths** |
|----|--------------|------------|-----|--------|--------|----------|
| A | Nyvarp | Sandöfjärden | 209 | 23 | 8 | 14 |
| B | Kanholmsfjärden | Kanholmsfjärden | 206 | 23 | 9 | 13 |
| C | Solöfjärden | Solöfjärden | 213 | 23 | 8 | 14 |
| D | TrälhavetII | Trälhavet | 215 | 23 | 9 | 13 |
| E | S. Vaxholmsfjärden | S. Vaxholmsfjärden | 131 | 23 | 7 | 8 |
| F | Blomskär | Stora Värtan | 141 | 23 | 8 | 9 |
| G | Blockhusudden | Strömmen | 249 | 23 | 11 | 16 |
| H | Baggensfjärden | Baggensfjärden | 173 | 20 | 9 | 10 |

* Entire period is 23 years; **Sampled at least half of the sample occasions

*Table 2. The maximum concentrations of P and N (mg l$^{-1}$) in the discharge from sewage treatment plants of different size (person equivalents, pe).*

| Sewage treatment facilities (pe) | P (mg l$^{-1}$) | N (mg l$^{-1}$) |
|---|---|---|
| >50 000 | 0.1 | 4 |
| 10 000-50 000 | 0.1 | 6 |
| <10 000 | 0.15 | 10 |

*Table 3. The correlation coefficients (r) between observations (obs) and model results (S-HYPE), and the long term (1990-2012) averages of river outflow (QF) and nutrient loads from Lake Mälaren.*

| Variable | Units | Average obs | Average S-HYPE | r |
|---|---|---|---|---|
| $Q_F$ | $10^6$ m³ month$^{-1}$ | 421 | 422 | 0.94 |
| TN | t month$^{-1}$ | 270 | 271 | 0.93 |
| DIN | t month$^{-1}$ | 83 | 76 | 0.86 |
| TP | t month$^{-1}$ | 13 | 11 | 0.87 |
| DIP | t month$^{-1}$ | 5.7 | 5.7 | 0.79 |

*Table 4. The total land load (rivers, land run-off and atmosphere) of P and N (t yr⁻¹) to the Stockholm Archipelago, the size of the benthic and pelagic N and P pools (t),the export from the area (t yr⁻¹) and the filter efficiency (F$_{eff}$) before and after the nutrient reductions, as well as their percentage changes. The system is in both cases in steady state, thus the benthic and pelagic pools are in balance with the nutrient load.*

|  |  | Unit | Initial values | End of period | Change (%) |
|---|---|---|---|---|---|
| **P** | Total load | t yr⁻¹ | 213 | 186 | -13 |
|  | Pool* | T | 10661 | 9952 | -7 |
|  | Export | t yr⁻¹ | -3.5 | -11 | -207 |
|  | F$_{eff}$ | % | 101 | 106 |  |
| **N** | Total load | t yr⁻¹ | 7690 | 6164 | -20 |
|  | Pool* | T | 35196 | 33216 | -6 |
|  | Export | t yr⁻¹ | 1585 | 609 | -62 |
|  | F$_{eff}$ | % | 79 | 90 |  |
| **N:P\*\*** | Molar ratio |  | 42 | 35 | -16 |

*The sum of benthic and pelagic pools. ** In the inner archipelago.