Predicting Ecosystem Response from Pollution in Baltic Archipelago areas using Mass-balance Modelling

OLOF MAGNUS KARLSSON
Dissertation presented at Uppsala University to be publicly examined in Hambergsalen, Villavägen 16, Uppsala, Friday, April 8, 2011 at 13:00 for the degree of Doctor of Philosophy. The examination will be conducted in Swedish.

Abstract

Baltic archipelago areas have high nature values despite being polluted from various anthropogenic activities within the Baltic Sea catchment area and from long-range transport of airborne substances. The discovery of environmental problems in the Baltic Sea in the 1960s led to countermeasures that gradually gave results in reducing the toxic pollution, e.g. from PCBs. Today, much of the environmental management is focused on reducing the effects of eutrophication. There is a demand from society on science to develop strategies that can direct remedial actions so that the cost-effectiveness is maximised. This work focuses on how mass-balance models can be used to understand how coastal ecosystems are controlled by abiotic processes and to predict the response to changes in loading of different substances. Advection, sedimentation and burial are examples of general transport processes that are regulated by morphometrical characteristics, e.g. size, form, effective fetch and topographical openness. This is why different coastal areas have different sensitivity to loading of pollutants.

A comparison of six phosphorus and chlorophyll models of different complexity showed that the model performance was not improved with more state variables of total phosphorus (TP) than two water and two sediment compartments. Modelling chlorophyll as a separate state variable did not improve the results for individual values compared to a simple regression against total phosphorus in surface water. Field investigations of the phosphorus content in accumulation sediments along the coast of Svealand showed a distribution pattern that probably is related to differences in the redox status. The average content of mobile phosphorus was much higher than previously found in offshore Baltic sediments indicating that sediments may play an important role for the phosphorus turnover in Baltic archipelago areas.

A one-year field study to measure the levels of polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs) in water, sediments and fish during different seasons was carried out in Kallrigafjärden Bay. The collected data set was used to test a mass-balance model for PCDD/F-turnover. It was possible to reproduce the concentrations of different PCDD/F-congeners with high accuracy using a general model approach, including one water compartment and two sediment compartments, indicating that the applied model has the necessary qualifications for successful predictions of PCDD/F-turnover in Baltic coastal areas.

Keywords: Baltic Sea, archipelagos, aquatic ecosystems mass-balance modelling, sediment-water dynamics, phosphorus, dioxins, PCDD/Fs

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From *Grupp Krilon* by Eyvind Johnson, 1943

*To my son Gustaf with love*
This thesis is based on the following papers, which are referred to in the text by their Roman numerals.


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In addition, the following papers, related to this thesis but not appended to it, have been published during the PhD study.


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### Abbreviations

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<th>Description</th>
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<tbody>
<tr>
<td>A-areas</td>
<td>Accumulation areas</td>
</tr>
<tr>
<td>BOD</td>
<td>Biochemical oxygen demand</td>
</tr>
<tr>
<td>Chl-a</td>
<td>Chlorophyll-a</td>
</tr>
<tr>
<td>COD</td>
<td>Chemical oxygen demand</td>
</tr>
<tr>
<td>CV</td>
<td>Coefficient of variation</td>
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<tr>
<td>DIP</td>
<td>Dissolved inorganic phosphorus</td>
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<tr>
<td>DOP</td>
<td>Dissolved organic phosphorus</td>
</tr>
<tr>
<td>DP</td>
<td>Dissolved phosphorus</td>
</tr>
<tr>
<td>DW</td>
<td>Dry weight</td>
</tr>
<tr>
<td>( D_{wb} )</td>
<td>Theoretical wave base</td>
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<tr>
<td>E-areas</td>
<td>Erosion areas</td>
</tr>
<tr>
<td>EIA</td>
<td>Environmental impact assessment</td>
</tr>
<tr>
<td>ET-areas</td>
<td>Erosion and transportation areas</td>
</tr>
<tr>
<td>LOI</td>
<td>Loss on Ignition</td>
</tr>
<tr>
<td>ODE</td>
<td>Ordinary differential equation</td>
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<tr>
<td>P</td>
<td>Phosphorus</td>
</tr>
<tr>
<td>PCDD/Fs</td>
<td>Polychlorinated dibenzo-p-dioxins and dibenzofurans</td>
</tr>
<tr>
<td>PF</td>
<td>Particulate fraction</td>
</tr>
<tr>
<td>PP</td>
<td>Particulate phosphorus</td>
</tr>
<tr>
<td>PSU</td>
<td>Practical salinity units</td>
</tr>
<tr>
<td>SD</td>
<td>Standard deviation</td>
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<tr>
<td>SPM</td>
<td>Suspended particulate matter</td>
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<tr>
<td>STP</td>
<td>Sewage treatment plant</td>
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<tr>
<td>T-areas</td>
<td>Transportation areas</td>
</tr>
<tr>
<td>TEQ</td>
<td>Toxicity equivalents</td>
</tr>
<tr>
<td>TOC</td>
<td>Total organic carbon</td>
</tr>
<tr>
<td>TN</td>
<td>Total nitrogen</td>
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<td>TP</td>
<td>Total phosphorus</td>
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Introduction

Coastal ecosystems contribute to more than 30% of the goods and services provided by the ecosystems of the world (Costanza et al., 1997). The coast is also a zone of conflicts where different interests, e.g. those of industrialists, fishers, armed forces, shipping, recreation and nature conservation compete for the available resources, (Bruntland, 1987; Cicin-Sain, 1993). Many coastal waters around the world are under severe environmental pressure and degradation (Lindeboom, 2002; Diaz & Rosenberg, 2008).

The Baltic Sea has a densely populated (approximately 85 million inhabitants) and heavily industrialised drainage area covering 1.7 million km$^2$. Water exchange with the adjacent sea occurs only through three narrow and shallow sounds. Therefore, the Baltic Sea including many of its coastal areas is polluted by different types of chemical substances such as organic carbon, nutrients, trace metals including radionuclides and organic toxins (HEL- COM, 2010). Major environmental threats to the Baltic Sea ecosystems include eutrophication (Larsson et al., 1985; Elmgren, 2001; Wulff et al., 2001; Lundberg, 2005; Johansson, 2006; Håkanson & Bryhn, 2008), overfishing (Österblom et al., 2007; Aps & Lassen, 2010), ecotoxicological disturbances and biological uptake of toxic substances (Södergren, 1989; Bengtsson et al., 1999; Kiviranta et al., 2003; Olsson et al., 2004) and invasion of alien species (Leppäkoski et al., 2002).

The water exchange between the coastal zone and the open waters of the Baltic Sea is generally rapid, with typical water residence times in coastal areas ranging from days to weeks (Håkanson et al., 1986; Persson et al., 1994; Engqvist, 1999; Engqvist & Andrejev, 2003). This means that “it is not possible to save the coastal areas unless one first saves the sea” (Håkanson, 1999). There are only few areas along the coast of the Baltic Sea where action against the local pollution load may significantly improve the environmental conditions (Wulff, 2006). Nevertheless, it is for several reasons important to quantify how internal processes, e.g. sedimentation and resuspension, influence the distribution and turnover of waterborne substances in the coastal zone. Firstly, sediment burial is one of three major pathways that can withdraw substances from aquatic ecosystems. The other processes are outflow and for organic substances also degradation. Secondly, enclosed coastal areas with a high percentage of soft bottoms and where the turnover of substances is not completely dominated by the exchange of water with the
adjacent sea often have high biological values in terms of production and species richness (Schiewer, 2008) and are often sensitive to pollution (Mann, 1982; Persson et al., 1993, Lindgren & Håkanson, 2011). Thirdly, the understanding of transport processes within defined coastal ecosystems may serve as a tool to understand how the Baltic Sea functions on a basin scale since the fundamental processes that regulate the turnover of substances in aquatic systems are general (Håkanson, 2006).

The general aim of this thesis was to increase the understanding of how abiotic processes affect the environmental status of Baltic archipelago areas regarding pollutants. Advection, mixing, deposition, sedimentation, resuspension, erosion, land uplift and burial are all examples of processes that create substance flows that should be quantified before decisions about, e.g. remedial actions are taken. The pervading methodology in this thesis is to use mass-balance models as the tool to structure the processes and the resulting ecosystem responses into decision support systems. The applied modelling philosophy prefers practical usefulness before detailed physical understanding. For example, the empirical data used for parametrisation and forcing of the models should be easily accessible from nautical charts and regular environmental monitoring programs. This demand may limit a model's exactness but will increase its applicability to environmental management.

Considering these assumptions the more specific aims were

- to investigate recent structural changes of coastal sediments coupled to changes in redox conditions including distribution patterns for phosphorus

- to improve the knowledge of factors crucial for phosphorus modelling in Baltic coastal areas

- to develop a mass-balance model for polychlorinated dibenzo-\(p\)-dioxins and dibenzofurans (PCDD/Fs) turnover in coastal areas
Materials and methods

The rationale for quantitative predictive ecology

The pioneering work of Richard Vollenweider in the late 1960s (Vollenweider, 1968; Vollenweider, 1975; OECD, 1982) concerning load models for phosphorus in lakes showed that simplicity is the key to get predictive power and practical applicability. Vollenweider demonstrated, by means of simple mass-balance models and statistical regressions, that in most lakes, eutrophication could be reversed by reducing the input of TP to the lake so that the mean lake concentrations of TP could be lowered. A number of works related to the phosphorus loading concept (e.g. Dillon & Rigler, 1974; Imboden, 1974; Chapra & Reckhow, 1983; Håkanson & Peters, 1995; Malmaeus, 2004a; Bryhn, 2008) have shown that quantitative predictive limnology is an instrumental tool for eutrophication control in lakes (Fig. 1).

In coastal waters however, nutrient effect-load-sensitivity models have, with some exceptions (Wallin, 1991; Nilsson & Jansson, 2002; Dowd, 2005; Håkanson & Eklund, 2007) less frequently been used as a tool to understand and manage the ecosystems. Furthermore, there are examples of where coastal load models have been applied in the environmental impact assessment of point-source effluents (Karlsson & Håkanson, 2001; Karlsson, 2002; Karlsson & Lidén, 2003; Malmaeus & Karlsson, 2007). On the other hand, a rich flora of sophisticated coastal ecosystem models often coupled to hydrodynamical models and including a large set of state variables exist (e.g. Cero & Cole; 1993; Ferreira et al. 1998; Humborg et al., 2000; Duarte et al., 2003; Moll & Radach, 2003; Marmefelt et al., 2007; Nobre et al., 2010).

Environmental modelling can be divided into two main categories, physical and empirical modelling. Physical models are intended to represent a detailed description of system behaviour, often transformed into partial differential equations of multiple dimensions operating on short time scales (minutes-days). Empirical models are often one-dimensional formulated as ordinary differential equations operating on medium time scales (month-year) and use empirical data for calibration and validation of the model.
Figure 1. Illustration of the classical approach to ELS (Effect-Load-Sensitivity) models for lake eutrophication. Modified from Chapra and Reckhow (1983).

From a management perspective, complex systems such as coastal areas may be more predictable on a monthly or yearly basis than on short time scales, integrating the effects of non-predictable (but regularly appearing) meteorological events and internal interactions. Models predicting average conditions within clearly defined and confined water bodies also integrate spatial variation. It has been shown mathematically (Monte, 1996) that the optimal model size for ecosystem models contains rather few variables and that the predictive power is in general not improved by detailed analysis (Fulton et al., 2003).

High predictive power is an important goal in itself since it is a direct indicator of how well we understand scientific relationships (Peters, 1991). The ability to predict important target variables such as toxic content in fish or algal bloom intensity is also very useful in practice as a communication tool between researchers and stakeholders (Mattila et al., 2010). It is also evidently useful to direct remedial measures in ecosystems where the environmental quality is unsatisfactory.
Study area

Waters near the coastlines of land masses often form transitional zones between freshwater and marine ecosystems especially pronounced in river mouth areas, so called estuaries. Coastal areas are generally very productive (Mann, 1982; Rosenberg, 1984). Taking a system approach the general ecology of coastal waters is well described by Mann (1982). A compilation of the different types of coasts and coastal ecosystems that occur in the Baltic Sea has been done by Schiewer (2008). Several coastal types such as boddens, fjords, archipelagos and seabeds can be distinguished (Fig. 2).

![Figure 2. A generalised picture of coastal types along the Baltic Sea coastline. From Schiewer (2008).](image)

The empirical foundation to this thesis emanates from the environmental conditions along the coast of Svealand on the Swedish east-coast (Fig. 3) representing the archipelago coast type (Fig. 4).
Figure 3. The coast of Svealand, the main study area.
The environmental characteristics of the coastal region of Svealand have been described by, e.g. Persson et al. (1993), Kautsky et al. (2000), Hill & Wallström (2008) and Kautsky (2008). The morphometrical attributes, i.e. mean depth, maximum depth, hypsograph and volume of different basins in the region have been calculated by SMHI (2003) using GIS and bathymetrical data from the Swedish Maritime Administration. Lindgren (2011) also used GIS to determine the openness and energy filtering in areas with various sheltering capacity. In some thirty areas of the region, Jonsson et al. (2003) classified the bottom dynamic conditions and the redox status of the sediments from seafloor mapping by means of side scan sonar and sediment echo sounder in combination with sediment sampling of cores from different bottom types. The water retention times for the different basins in the region have been calculated by Persson et al. (1993), Engqvist (1999) and Engqvist & Andrejev (2003). Håkanson et al. (1986) and Persson & Håkanson (1996) carried out empirical measurements of the water exchange in a number of areas in the region. Empirical studies on various substances and transport processes in the region have been carried out for, e.g. nutrients (Engqvist, 1996), organic carbon (Jönsson et al., 2005), suspended particulate matter (SPM) (Håkanson & Eckhèll, 2005), trace metals (Lindström et al., 2000), radiocaesium (Meili et al., 2000a) and PCBs (Meili et al., 2000b). The spa-
tial distribution of shallow areas with high production potential (Fig. 5) has been assessed by Lindgren (2010).

![Figure 5. Inner part of Kallrigafjärden Bay, an example of a shallow productive coastal area. Photo 2007-09-25.](image)

**Coastal processes**

Any coastal area is subject to a number of abiotic processes which determine its environmental characteristics as illustrated in Figure 6. To be able to make meaningful predictions about coastal ecosystem's responses to, e.g. change in loading of specific substances it is necessary to have figures on the fluxes (arrows) presented in Figure 6. The concentration of a substance in the water column of a coastal area depends on both advective forces (water exchange with the adjacent sea, riverine inflow), internal processes (stratification, sedimentation, resuspension) and biogeochemical processes (diffusion from sediments, biological uptake).
Modelling in coastal areas requires a technique that provides an ecologically meaningful and practically useful definition of the coastal ecosystem boundaries. It is often a straightforward process to identify the boundaries from nautical charts but in cases where it is not obvious, the topographical bottleneck method (Persson, 1999; Gyllenhammar & Gumbricht, 2005) may be applied. This means that the borderlines are drawn where the exposure or topographical openness towards the adjacent sea and/or coastal areas are minimised.

Sedimentological aspects
Most pollutants have a high affinity to cohesive fine material (Förstner & Wittman, 1981; Wu & Gschwend, 1986). An important step in ecosystem modelling is therefore to account for the bottom dynamical conditions, i.e. to separate areas where there is a net deposition of cohesive fine matter (soft bottoms) from areas where fine matter by time will be resuspended to the water column, generally generating hard or sandy sediments. The prevailing bottom dynamic conditions reflect the impact of various energy controlled processes due to, e.g. wave action, currents and slope (Sly, 1973; Sly, 1978). In this work, the following definition of bottom dynamical conditions from Häkanson & Jansson (1983) has been applied:

- Accumulation areas (A-areas) prevail where fine material with grain sizes less than 0.006 mm can be deposited continuously
- Transportation areas (T-areas) appear where there is a discontinuous deposition of fine particles/aggregates, i.e. where periods of

Figure 6. Principle illustration of abiotic processes that affect the turnover of substances in coastal areas. From Paper VII.
accumulation are interrupted by periods of resuspension and transport

- Erosion areas (E-areas) prevail where there is no deposition of fine materials

The water depth separating T-areas from A-areas is denoted as the theoretical wave base ($D_{wb}$). $D_{wb}$ is ideally determined from mapping using hydroacoustic instruments in combination with sediment sampling (Håkanson et al., 1984; Wright et al., 1987; Kenny et al., 2000; Jonsson et al., 2003; Sutton & Boyd, 2009). However, as mapping is both time consuming and costly information of this kind is available only for a limited number of coastal areas. Therefore, a number of regression models have been developed (Persson & Håkanson, 1995; Håkanson & Eklund, 2007; Bekkby et al., 2008; 2009; Lindgren & Karlsson, 2011) linking the percentage of A- or ET-areas to morphometrical characteristics such as the exposure, i.e. the openness towards the adjacent sea, the filter-factor, a measurement of the wave energy outside the area, the form factor, the water depth, the slope etc.

When addressing bottom dynamic conditions the role of the land uplift must also be discussed. As a late inheritance of the latest ice age, the coastlines of Finland and Sweden are still rising at a rate of up to 9 mm yr$^{-1}$, which over a long period leads to clear changes in the coastlines (Brydsten, 1985) and is of profound importance for the turnover of material in the Baltic sub-basins (Jonsson et al., 1990; Håkanson & Bryhn, 2008). Due to land uplift areas below the wave base are lifted above the same and become susceptible to resuspension. Moreover, the erosion pressure increases on existing ET-areas. On a coastal scale, however, studies have shown that for individual coastal areas the effects of the land uplift are generally of small importance for the turnover of particulate substances (Håkanson et al., 2004; Håkanson & Eklund, 2007).

There are several methods to estimate sediment growth which make it possible to calculate the sediment burial, one of two major exit pathways for substances in coastal areas (the second being outflow). The lamination pattern created in sediments that are not subject to bioturbation by macrobenthic fauna is considered to be annual (Renberg, 1986; Morris et al., 1988; Jonsson, 1992). The thickness of the varves in laminated sediment profiles can therefore be used to estimate the net sedimentation (Jonsson et al., 1990). The tragic nuclear reactor accident in Chernobyl, which led to a high deposition of radionuclides over large parts of Europe, has ironically provided a very useful tool to estimate sediment growth (Meili et al., 2000a). Jonsson et al. (2003) used both methods to estimate sediment growth (mm yr$^{-1}$) and dry matter deposition g m$^{-2}$ yr$^{-1}$ in A-areas of some fifty coastal areas along the Swedish coast. Håkanson et al. (2004) used empirical data on sedimentation
from coastal areas in Sweden and Finland to develop a dynamical model for sedimentation. In this work, both modelling (Paper II, Paper VII) and empirical approaches (Paper I, Paper IV) have been applied to quantify the sediment burial.

Advection

The settling velocity of particles in aquatic environments depends on a number of factors, e.g. salinity, turbulence and suspended solids concentration (Malmaeus, 2004b). A typical settling velocity for SPM in the open Baltic Proper is 0.5 m per month (Håkanson & Bryhn, 2008) whereas the default settling velocity in coastal areas has been calculated to 25 m per month for SPM (Håkanson et al., 2004) and 6 m per month for particulate phosphorus (PP) (Håkanson & Eklund, 2007), respectively. It should be stressed though that fine materials are rarely deposited as a result of simple vertical settling in natural aquatic environments. The horizontal velocity is generally at least 10 times larger, sometimes up to 10,000 times larger, than the vertical component for fine materials or flocs that settle according to Stokes' law (Bloesch & Uehlinger, 1986).

In coastal areas, a number of factors can influence the water dynamics and thereby the advection of substances. In coastal regions influenced by tide, the water exchange with the adjacent sea is generally very effective (Fischer et al., 1979). However, due to its size and enclosedness, the tidal amplitude is low in the Baltic Sea, ~1 cm (Kullenberg, 1981; Keruss & Senikovs, 1999). Instead, surface water exchange in a majority of the Baltic archipelago areas is mainly driven by the transfer of momentum from drag forces of winds creating turbulent perturbations (Persson & Jonsson, 1997). In river mouth areas estuarine circulation can also be important depending on the size of the estuary and the discharge of fresh water (Fischer et al., 1979). In open coastal areas with large exposure to the adjacent sea, coastal currents driven by the rotation of the earth (the Coriolis effect) can also influence the water turnover (Persson et al., 1994). When Baltic coastal waters are thermally stratified the exchange of deep water is generally not as rapid as the surface water exchange. The deep water is often exchanged episodically whereas the mixing between surface water and deep water is small (Persson, 1999).

Håkanson et al. (1984) measured the water turnover in some twenty Swedish coastal areas using dye-tracing, water discharge measurements, gauges, gelatine pendulums, conductivity, temperature and depth profiles. The exchange of surface water was well correlated to the exposure, i.e. the topographical openness of the coastal area and statistical models were developed (Håkanson et al., 1986; Persson et al., 1994). Engqvist (1999) also took
a statistical approach when the water retention times for the whole coast of Sweden, resolved into basins according to SMHI (2003) were estimated. A three-dimensional model based on the concept of average age was used by Engqvist & Andrejev (2003) and Engqvist et al. (2006). In areas with fresh water inflow, i.e. estuaries, of such a magnitude that a salinity gradient is developed, the mass-balance for salt also known as Knudsen’s relations (Knudsen, 1900) can be used to calculate the water exchange. In a majority of the Baltic archipelago areas the surface water turnover is relatively rapid, with residence times typically less than a week. This means that quantifying the flux of matter generated by the water turnover with the adjacent sea is of profound importance for the modelling success. In this work, mostly the formula derived by Persson et al. (1994) and Knudsen’s relations have been applied but when different methods have been compared the agreement has been good (Karlsson & Håkanson, 2001; Karlsson, 2002; Paper I; Faxö et al., 2010).

The freshwater inflow is sometimes of importance for the water turnover depending on the morphometrical characteristics of the estuary and the magnitude of the discharge. The availability of run-off data is generally good for Swedish rivers by statistical records kept by the Swedish Meteorological and Hydrological Institute, e.g. SMHI (1993). However, in case the discharge data are lacking they can be modelled on an average monthly basis by knowledge of the drainage area and latitude (Abrahamsson & Håkanson, 1998).

Modelling

Basically, the modelling approach applied in this work calculates the concentration of a target variable in the water column or in the sediments using algorithms or empirical input data for the transport processes illustrated in Figure 6. The water volume is separated at the depth of the wave base into two compartments, surface water and deep water respectively. The water compartments are treated as completely mixed. Exchange with the adjacent sea is quantified either by Knudsen's relations, statistical models or empirical measurements (see preceding section). To account for the fact that particulate substances settle due to gravity a partitioning coefficient denoted PF (the particulate fraction) is included. To handle the sediment dynamics the modelled area is subdivided into A-areas and ET-areas, respectively. From A-areas substances can be withdrawn from the biosphere through burial whereas matter deposited on ET-areas by definition will be resuspended. A comprehensive description of the general model structure and equations are given in Paper VII.
To solve the set of the first order, ordinary differential equations (ODE) that arise when a mass-balance situation is to be modelled, the simulation computer program Stella® was used. ODEs were solved numerically using the Euler method. The time interval was set to 0.05, which means that the computer divides each time step (dt) into 20 units and for every unit approximates the derivatives with the difference through that time interval. The time step for simulations was normally set to 1 month primarily because that is a time step of ecological relevance and that much of the empirical data is collected monthly. It has also been shown that using the monthly time-scale often minimises the uncertainty in important coastal water quality variables (Håkanson & Duarte, 2008).

Field measurements

Field work was conducted from R/V Sunbeam, R/V Perca, R/V Alcidae and small work boats. Sediment samples were taken with a Gemini double corer (inner diameter 80 mm) and a Willner sampler (inner diameter 63 mm) which allows free water passage through the sample during descent and sediment penetration. Great care was taken to ensure that the sediment surface was intact, e.g., living macrozoobenthos, benthic macrofauna burrows, loose surficial sediment, bacterial film of *Beggiatoa* and clear supernatant water in the sampler above the sediment/water interface. The sediment cores were sliced in 2 cm sections immediately after sampling. Moreover, intact sampled cores were stored at 4° C until preparation in the laboratory. Before cutting the cores into two longitudinal halves, exposing the vertical structures, each core was put in a freezer for approximately two hours before extrusion to minimise disturbance of the very loose surficial sediments. This stabilised the outer 2-5 mm of the sediment column, leaving the central parts unfrozen and structures preserved. The section surfaces were examined, described and photographed (digital format, 4 mega pixels), and the lamina- tion patterns were recorded. Figure 7 shows an example of a documented sediment core from the Stockholm archipelago that was sampled in September 2008 (Karlsson et al., 2010).
Figure 7. Photograph of the stratigraphy in a sediment core from the Stockholm archipelago. From Karlsson et al. (2010). Photo: 2008-10-14, Per Jonsson.
Water samples for nutrient analyses were collected with a Ruttner sampler. The water salinity and temperature were measured online in the field using an YSI INC Model 30 M handheld salinity, conductivity and temperature device. The Secchi depth was measured using a standard Secchi disk. Water sampling for PCDD/Fs was conducted using a specially designed high-volume filtration system (Fig. 8) described in Broman et al. (1991). Approximately 800 L per sampling occasion was pumped through a prefilter, a glass fibre filter (pore size 0.7 μm) and finally a polyurethane foam (PUF) plug, which enabled separation between the particulate and the apparently dissolved fraction of PCDD/Fs. As the filters clogged during sampling, they were exchanged for new filters, and the number of filters used during each sampling varied between 2 and 10.

*Figure 8.* Sampling device for active PCDD/F sampling in water. Photo 2008-02-19.
Results and Discussion

Identifying the needs for improved models by analysis of changes in sediments (Paper I)

In 2008, a study was initiated in the Stockholm archipelago with the specific aim to investigate the distribution of “dead” seafloors, i.e. areas where benthic macrofauna is absent and laminated sediments prevail, and to compare the results with equivalent investigations from a decade ago (Jonsson et al., 2003). We found a remarkable improvement of the redox status in the sediments of large areas that had previously been covered by reduced surfaced sediments (Fig. 9). The portion of areas covered with laminated sediment in the investigated area had decreased from 17% to 4% between 1998 and 2008.

![Figure 9](image)

Figure 9. The distribution of reduced surface sediments in the inner Stockholm archipelago 1998 and 2008, respectively. From Karlsson et al. (2010).

Several hypotheses were formed to explain the observed improvements. One hypothesis was that a change in the nutrient load had occurred, either locally, by reduced discharges from sewage treatment plants (STPs) or regionally, by a reduced inflow of nutrients from the water exchange with the adjacent archipelago. To be able to address this hypothesis, a well-structured and validated mass-balance model with the capability to perform scenario analyses would be a natural choice. However, modelling the inner Stockholm archipelago is not a straightforward task. The area consists of several bays with different morphological characteristics connected through sounds of
various section areas. Lake Mälaren discharges some 300 m$^3$/s through River Norrström in the innermost part of the area whereas the main exchange with the adjacent archipelago goes through the sound Oxdjupet (Fig. 9). Altogether, the natural conditions create a hydrodynamical complex system with large inherent variability although estuarine circulation is the dominant forcing factor. Due to the complex flow patterns, we were not capable of modelling the nutrient dynamics in the inner Stockholm archipelago. Instead we established crude budgets for nitrogen and phosphorus turnover based on empirical data for different discharges, assuming the system being in steady-state and the sediments acting as a source or sink to be able to close the mass-balance (Fig. 10).

Figure 10. Nutrient budgets for the Stockholm archipelago, Tot-P = total phosphorus, Tot-N = total nitrogen, STP = sewage treatment plant. From Paper I.

Substance flow budgets are often the first step in developing mass-balance models (Wulff et al., 2001; Håkanson & Bryhn, 2008). The usefulness of
budget calculations should not be underestimated (e.g. Artioli et al., 2008) although one should be aware of its limitations. Budget calculations may be used to get an idea of a systems response to a change in one or several fluxes but are always site-specific since they require calibration.

Developing models (Papers II, III and IV)

Albert Einstein once stated, “Everything should be made as simple as possible but not simpler.” When practising ecosystem modelling in environmental impact assessments (EIA) of effluent discharges time and available resources may be a limiting factor and to seek for shortcuts is sometimes necessary. Moreover, the empirical frame for modelling, usually based on environmental monitoring programs, is seldom complete neither from a spatial nor a temporal perspective. One important task for applied environmental science should therefore be to develop sound models also from a management perspective. In Paper II, we examined the performance of models of different complexity when applied on data sets from 10 environmental monitoring programs along the Swedish east-coast.

We found that the best prediction of Chl-a concentration was established through a simple regression against TP-concentration (Fig. 11) similar to what have been found for lake ecosystems (Dillon & Rigler, 1974). Using the slope and $r^2$-values of modelled versus empirical values of TP-concentration as estimators of the model performance we found that the best fit was obtained when the coastal ecosystem was divided into four compartments, surface water, deep water, ET-sediments and A-sediments, respectively.

![Figure 11](image_url)

*Figure 11.* Summer average chlorophyll concentrations (Chl-a) versus summer average total phosphorus concentrations (TP) in 15 Baltic coastal areas. Modified from Paper II.
The particulate fraction may be regarded as the entry gate to mass-balance modelling since it regulates how much of a substance that could potentially settle on the sea bed due to gravity (Håkanson, 1997). In Paper III, the particulate fraction (PF) of TP in Baltic coastal waters was studied. The PF-value of TP has been studied in a wide range of freshwater ecosystems (Håkanson & Boullion, 2002; Cade-Menun et al., 2006; Ellison & Brett, 2006; Bryhn et al., 2007) but empirical data from marine ecosystems are scarce (Håkanson & Bryhn, 2008). The average value of PF for TP found in this study was 0.33 which is lower but of the same magnitude that have been found as an average for boreal glacial lakes (0.56, Håkanson & Boullion, 2002). Furthermore, it was found that orthophosphate (DIP) was a poor predictor of total dissolved phosphorus (DP) probably reflecting the quantitative importance of dissolved organic phosphorus (DOP). The importance of the PF-value for the predictive power of phosphorus mass-balance models was tested setting up two different model frameworks and applying the Monte Carlo simulation technique; 1) a typical coastal area of the Baltic Sea and 2) a sub-basin of the Baltic Sea. The simulations showed that the coastal ecosystem was insensitive to variations in the PF-value whereas the prediction of TP on a basin scale was sensitive to which PF-value that was applied. This can be explained by the advection driven by water-exchange that dominate the transport processes along the coast whereas internal processes, e.g. sedimentation are more important in areas with long water retention times such as Baltic sub-basins.

The predictive power of ecosystem models is not determined by the parts of the model that best describes the reality but by its weakest parts. Phosphorus concentrations and burial in A-areas have for a long time been recognised as difficult to predict (OECD, 1982), probably because of their patchiness and variability related to the redox status of the benthic compartment as described by Einsele (1936) and Mortimer (1941). Modellers often treat sediments like a “black-box” used to explain variations in the water mass that cannot be explained by other fluxes (e.g. Savchuk & Wulff, 2009). There are models that include algorithms for phosphorus leakage from A-areas in the Baltic Sea, e.g. Håkanson & Eklund (2007) and Håkanson & Bryhn (2008) but the empirical foundation for these algorithms is weak.

In 2008, a study was initiated where one aim was to increase the understanding of the water-sediment phosphorus exchange in Baltic coastal areas (Paper IV). The pool of mobile phosphorus in a dozen sediment-cores from A-areas along the Swedish east-coast (Fig. 12) where quantified with methods described in Rydin et al., (2011) including phosphorus fractionation (Psenner et al., 1988). In addition, some a hundred sediment cores were taken and analysed for TP content in surface and deep sediment layers (Tab. 1).
Figure 12. Positions of sediment sampling stations. Numbered crosses indicate stations where the mobile pool of phosphorus was quantified. Black dots show stations where concentrations of TP in surface and deep sediments were investigated. From Paper IV.
Table 1. Statistics (mean, median, minimum; Min and maximum; Max values, standard deviation; SD and number of samples; n) for total phosphorus (TP) concentration (μg·g⁻¹ dry weight) in surface sediments (0-2 cm) and deep sediments (2-5 cm layers collected at between 11 and 70 cm sediment depth). Statistics are shown for all data ('All') and for data grouped into 33 areas ('By area'). From Paper IV.

<table>
<thead>
<tr>
<th></th>
<th>Surface TP</th>
<th>Deep TP</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>All</td>
<td>By area</td>
</tr>
<tr>
<td>Min</td>
<td>840</td>
<td>870</td>
</tr>
<tr>
<td>Mean</td>
<td>1 817</td>
<td>1 651</td>
</tr>
<tr>
<td>Median</td>
<td>1 300</td>
<td>1 400</td>
</tr>
<tr>
<td>Max</td>
<td>7 100</td>
<td>4 900</td>
</tr>
<tr>
<td>SD</td>
<td>1 257</td>
<td>848</td>
</tr>
<tr>
<td>n</td>
<td>102</td>
<td>33</td>
</tr>
</tbody>
</table>

The difference between the TP level in the surface sediment and the deep sediment, respectively, reflects the amount of phosphorus that as time goes by shall be released to the water column. With knowledge of the sediment growth rate it is possible to use the information given in Table 1 quantitatively. Jonsson et al. (2003) measured the sediment growth in some fifty Baltic coastal A-areas using ¹³⁷Cs-dating and lamina counting. The average deposition rate was 2 300 g m⁻² yr⁻¹ equivalent to 17 mm yr⁻¹. The deposition rate multiplied by the average TP-levels in surface and deep sediments, respectively gave the average sediment-water fluxes in our studied A-areas (Fig. 13).

Figure 13. An illustration of the average sediment-water exchange of total phosphorus in A-areas along the Swedish east-coast. Modified from Rydin et al. (2011).
The pool of mobile phosphorus within the active sediment layer as illustrated in Figure 14 was calculated by integrating the surplus of TP in the surface layer of the sediment core compared to the burial concentration. The average pool of mobile phosphorus in our study area was 2.5 g m\(^{-2}\).

Figure 14. A principle illustration of the pool of mobile phosphorus (P; shaded) in a vertical sediment profile, DW = dry weight.

We also found a significant correlation (\(r^2 = 0.88\)) between the mobile pool of phosphorus and the content of TP in surface (0-2 cm) sediments (Fig. 15). Logarithmic transformations of the included variables gave a better approximation to the normal distribution but the \(r^2\)-value dropped to 0.72.

Figure 15. Regressions with 95% confidence bands to estimate mobile phosphorus pools from surface contents, \(n = 12\), TP = total phosphorus, DW = dry weight. From Paper IV.
Jonsson et al. (2003) also mapped the distribution of A-areas in a majority of the studied coastal areas by means of side scan sonar and sediment echo sounder. By extrapolation, the total mobile pool of phosphorus in our study area can be estimated calculated using Jonsson et al. (2003) data on distribution of A-areas and our calculated values of mobile phosphorus content.

Our estimates show that the coastal area between Oxelösund and Öregrund (Fig. 12) holds about 2 500 tonnes of mobile phosphorus that by time shall be released. This amount can be compared with the yearly riverine load of TP to the area of roughly 200 tonnes yr\(^{-1}\) and the equivalent TP-load from coastal STP discharges of 100 tonnes yr\(^{-1}\). Certainly, the eventual release of PO\(_4\)-P from sediments due to changes in redox conditions could have some implications for the trophic status of the coastal areas in question.

The findings of Papers III and IV raise expectations that existing mass-balance models for TP turnover (e.g. Håkanson & Eklund, 2007; Håkanson & Bryhn, 2008; Paper II) could be further improved by incorporating the empirical relationships found in these studies. It should be noted though that the results should be considered as preliminary and may be subject to modifications as the empirical ground grows by future sampling campaigns.
Model application (Papers V, VI and VII)

In 2007, a study was initiated with the aim of establishing a mass-balance for the PCDD/F turnover in a coastal area of south-western Bothnian Sea (Fig. 16). The background for the study was that PCDD/F contamination of Baltic fish still is a major environmental problem (Kiviranta et al., 2003; Vuorinen et al., 2004; Bignert et al., 2007) that restricts marketing of some species of fatty fish within the European Union although the levels have declined compared to the situation in the 1970s (Rappe et al., 1981, 1987). Why the decline in PCDD/F-levels in biota from the Baltic Sea seems to have ceased unlike in other classical organic pollutants (Olsson et al., 2004) is unclear. It is well known that high levels of PCDD/Fs in marine biota may locally be explained by point-source emissions (e.g. Ruus et al. 2006).

A sampling program was established to measure PCDD/Fs levels to, from, and within the studied system during one year (September 2007 to August 2010). The applied sampling technique (see Paper V) allowed us to separate the detected PCDD/F congeners in the water between the dissolved and the particulate phase (Tab. 2). PCDD/Fs are hydrophobic substances with a high affinity to particles which was reflected in the measured PF-values ranging between 0.69 and 0.95 for different PCDD/F-congeners.

Table 2. The proportion of different PCDD/F-congeners detected as particulate phase in the water (PF-value).

<table>
<thead>
<tr>
<th>PCDD/F-congener</th>
<th>12378-PeCDD</th>
<th>123478-HxCDD</th>
<th>123678-HxCDD</th>
<th>123789-HxCDD</th>
<th>1234678-HpCDD</th>
<th>OCDD</th>
</tr>
</thead>
<tbody>
<tr>
<td>PF-value</td>
<td>0.75</td>
<td>0.60</td>
<td>0.86</td>
<td>0.83</td>
<td>0.94</td>
<td>0.94</td>
</tr>
<tr>
<td>PCDD/F-congener</td>
<td>2378-TCDF</td>
<td>12378-PeCDF</td>
<td>23478-PeCDF</td>
<td>123478-HxCDF</td>
<td>123678-HxCDF</td>
<td>234678-HpCDF</td>
</tr>
<tr>
<td>PF-value</td>
<td>0.69</td>
<td>0.74</td>
<td>0.80</td>
<td>0.89</td>
<td>0.84</td>
<td>0.83</td>
</tr>
</tbody>
</table>

Figure 16. The study area Kallrigafjärden Bay in the south-western the Bothnian Sea including sampling stations. The red line demarks the operational border between Kallrigafjärden Bay and the adjacent sea. From Paper V.
The measured concentrations of PCDD/Fs from tributaries and the adjacent sea and estimated water fluxes in combination with literature values on the atmospheric deposition (Lohmann & Jones, 1999; Armitage et al., 2009) made it possible to calculate a PCDD/F-budget for Kallrigafjärden Bay where the residual was considered to be net sedimentation (Fig. 17).

Figure 17. Annual fluxes of PCDD/Fs in and out of Kallrigafjärden illustrated the as total sum of 17 toxic congeners (left y-axes) and toxic equivalents (WHO-TEQ; right y-axes). From Paper VI.

Our next step was to put the collected data set into a modelling context (Paper VII) by applying a general model for substance turnover in Baltic coastal areas originally developed for SPM (Håkanson et al., 2004) and TP (Håkanson & Eklund, 2007). By doing this it was possible to test the model's performance to handle sedimentological processes as the advective transport processes already were determined from empirical data. There was a strong positive correlation between the empirical and modelled logarithmic yearly mean values of individual PCDD/F congeners (Fig. 18) which to a large extent can be explained by the similarity between concentrations in the adjacent sea and the coastal area (Fig. 19). Since the water exchange between the coastal area and the adjacent sea is the dominant flux of PCDD/Fs it is logical that that the model performance is mostly determined by the capability to quantify this flux.
Figure 18. Modelled (Mod) versus empirical (Emp) yearly mean total levels of 15 PCDD/F congeners in the Kallrigafjärden Bay surface water (a) and sediments (b). From Paper VII.

Figure 19. A regression between empirical yearly mean total concentrations for 15 congeners of PCDD/Fs (pg TEQ m\(^{-3}\)) in surface water of the Öregrundsgrepen Bight (adjacent sea; C\(_{\text{sea}}\)) and Kallrigafjärden Bay (C\(_{\text{Kallriga}}\)). From Paper VII.
However, as shown in Figures 20 and 21, neglecting the sedimentological processes reduce the predictive power of the model. Figure 20 shows modelled versus empirical values when the algorithm for resuspension was “turned off” and Figure 21 shows equivalent values when the PF-value was uniform for all congeners. In both cases, the correlations between modelled and empirical values of individual congeners were lower compared to the original model set-up (Fig. 18).

**Figure 20.** Modelled (Mod) versus empirical (Emp) yearly mean values for 15 congeners of PCDD/Fs in the Kallrigafjärden Bay surface water (a) and sediments (b) when the model algorithms for resuspension have been turned off. From Paper VII.

**Figure 21.** Modelled (Mod) versus empirical (Emp) yearly mean values for 15 congeners of PCDD/Fs in the Kallrigafjärden Bay surface water (a) and sediments (b) when the particulate fraction values (PF) uniformly have been set to 0.56. From Paper VII.
The developed model for PCDD/F turnover (Paper VII) was built using the same philosophy and strategy as previously applied when modelling radionuclides in lakes and coastal areas (IAEA, 2000; Håkanson, 2005; Håkanson & Lindgren, 2009), trace metals in lakes (Lindström & Håkanson, 2001), TP in lakes (Malmaeus & Håkanson, 2004), coastal areas (Håkanson & Eklund, 2007) and Baltic Sea sub-basins (Håkanson & Bryhn, 2008) and SPM in lakes (Malmaeus & Håkanson, 2003) and coastal areas (Håkanson et al., 2004). The high reliability of all these models indicates that our view of fundamental transport processes within aquatic ecosystems has considerable predictive power in the sense that on the ecosystem level, important water quality variables can be predicted using a few, generally, easy accessible morphological and physic-chemical characteristics of the studied system.
Concluding remarks and outlook

My concise conclusions of the work behind this thesis are the following:

The soft bottom benthic community of the inner Stockholm archipelago ecosystem is recovering from hypoxia.

The summer levels of chlorophyll in Baltic coastal areas may be predicted through a simple regression with total phosphorus concentrations in surface water similar to what has been found in lakes.

The optimal size of dynamic mass-balance models for phosphorus in Baltic archipelago areas seems to contain a rather simple parametrisation of the coastal ecosystem into four compartments, surface water, deep water, ET-sediments and A-sediments, respectively.

The particulate fraction of TP on Baltic coastal waters seems to be lower but of the same magnitude as the average PF-value in boreal glacial lakes. DIP concentrations cannot be used to predict the PF.

The average pool of mobile phosphorus in studied soft bottoms along the coast of Svealand was 2.5 g m\(^{-2}\), which extrapolated to the total accumulation area of the studied region gives an amount of 2 500 tonnes of mobile phosphorus. In comparison, the yearly load of TP from tributaries and coastal STPs in the region is approximately 300 tonnes.

There was a strong correlation between the mobile pool of phosphorus and the content of TP in surface (0-2 cm) sediments from coastal A-areas.

It was possible to develop a mass-balance model for the PCDD/F turnover in Kallrigafjärden Bay that with high accuracy could reproduce the concentrations of different PCDD/F-congeners in the water mass without adjusting any algorithm previously used in modelling of TP and SPM.
There are several extensions of the work summarised in this thesis that could possibly increase our understanding and improve the management of Baltic coastal ecosystems. For example, it would be interesting to apply the model for PCCD/F-turnover in a historically contaminated site to investigate the importance of the sediment release to the contamination level in fish and the possible effects of mitigate measures similar to studies by Saloranta et al. (2008). It would also valuable to resolve the inner Stockholm archipelago into a number of basins following the salinity gradient and model the phosphorus dynamics. Since the background data regarding water quality are extensive and the external sources of phosphorus are quantified there are good prerequisites to further elaborate the sediment-water dynamics.

As earlier mentioned, “It is not possible to save the coastal areas unless one first saves the sea.” For management purposes it is therefore important to quantify the processes studied in this thesis also on a Baltic sub-basin scale. Furthermore, the value of comparative studies cannot be overemphasised. It is my belief that much is to be learnt by comparing the conditions in the open waters of the Baltic Sea not only with the conditions in the archipelagos but also with lakes, e.g. the large lakes of Sweden, the Great Lakes of North America or other enclosed seas, e.g. the Black Sea and the Bohai Sea.

One of the foundations bolts in science, including predictive ecology should be to falsify hypotheses with evidence of the opposite (Popper, 1972; Peters, 1991, Bryhn, 2008). Unfortunately, this has not always been the case in Baltic Sea research (Håkanson, 2010). It is my hope that this thesis has added some new information and insights on how Baltic archipelago areas function that by time will be refuted but will direct the management to the benefit of future generations. The archipelagos of the Baltic Sea are unique and invaluable world heritages that deserve a better environmental status than the present situation. History has taught us that environmental degradation, in most cases is a reversible process. By linking ecology to earth sciences and applying engineering methods, cost-effective remedial measures can be implemented. The signs of improving ecosystems during the last decades should further encourage us to make a considerable change in the management of our common sea.
Kustzonen, gränsen mellan land och hav, är barnkammare och skafferi för många av havets organismer och hyser i många avseenden stora ekologiska värden. Samtidigt bor en stor del av världens befolkning i närheten av kuster, varigenom trycket från olika mänskliga aktiviteter i många områden resulterar i förändrade miljöförhållanden. Östersjön är genom sitt instängda läge känsligt för tillförsel av föroreningar genom att det tar lång tid innan de transporterats ut ur systemet. En väl dokumenterad ökning i tillförseln av olika substanser till Östersjön under det senaste seklet till följd av industrialiseringen, rationaliseringar inom jordbruket och införandet av vattenburna avloppssystem har resulterat i storskaliga förändringar av miljön i öppna Östersjön. Detta har i hög grad också påverkat förhållandena i kustzonen då vattenutbytet mellan kust och hav i de flesta områden är betydande. Lokalt har Östersjöns kustområden också påverkats av tillförsel från tillrinnande vattendrag och kustlokaliserade tätorter och industrier.

Icke desto mindre är det av flera skäl viktigt att studera och öka förståelsen för hur transportprocesser fungerar för olika ämnen i kustzonen. Sedimentation, resuspension och fastläggning av partikulärt bundet material är betydelsefulla processer för omsättningen av många substanser. I kustzonen regleras dessa processer i första hand av morfometriska parametrar som stryklängd, topografisk öppenhet och djupförhållanden. Eftersom dessa parametrar varierar avsevärt mellan olika kustområden varierar kustområdens känslighet för belastning. I inneslutna kustområden är förutsättningarna för sedimentation ofta relativt gynnsamma, vilket gör dem känsliga för belastning samtidigt som de ofta har hög biologisk produktionspotential.

Målet med detta arbete har varit att öka förståelsen för hur olika abiotiska faktorer reglerar omsättningen av potentiellt förorenande substanser i Östersjöns skärgårdsområden. Specifika frågor har varit att undersöka faktorer som styr fosfordynamiken mellan sediment och vatten samt att utveckla en dynamisk massbalansmodell för omsättning av dioxiner och furaner (PCDD/Fs). Studieområdet, där fältarbeten utförts har huvudsakligen förlagts till ett antal skärgårdsområden längs Svealandskusten.


En viktig fråga, inte minst ur ett avnämarperspektiv, är hur modeller med optimal storlek kan utvecklas med avseende på antalet ingående variabler och processer. Om en modell skall kunna användas i ett praktiskt fall, exem-
pelvis en miljökonsekvensbeskrivning av en verksamhets utsläpp, bör den kunna drivas med lättillgängliga empiriska data från befintliga recipientkontrollprogram. Simuleringar bör också kunna utföras på persondatorer inom rimliga tidsrymer. Å andra sidan får modellen inte förenklas så mycket att den prediktiva kraften hos målvariabeln går förlorad. I artikel II jämförs sex olika fosformodeller av varierande komplexitet. Modellerna tillämpades på tio kustområden längs den svenska ostkusten. Vi fann att det inte var motive­rat att använda fler än fyra tillståndsvariablen för att karaktärisera systemets trofiska status. Dessa var totalfosfor (TP) i yt- och djupvatten samt TP i ak­kumulationsssediment och erosions- och transportsediment. Vidare konstater­ades att koncentrationen av klorofyll-a i ytvattnet under sommarperioden var starkt korrelerad till ytvattenkoncentrationen av TP i studerade områden. Den påvisade korrelationen kan användas för att förutsäga sommarvärden av klorofyll i skärgårdsområden från Bråviken i söder till Kalixsälvens mynningsområde i norr.

En undersökning av fosfors fördelning i vattenmassan mellan lösta och partikulära förekomstformer redovisas i artikel III. I artikeln redovisas också känslighetsanalyser där variationen i TP-koncentrationen simulerats för olika värden på den partikulära fraktionen av fosfor. Sediments förmåga att hålla kvar fosfor är starkt kopplat till rådande redoxförhållanden. Många studier har visat att när syrgaskoncentrationerna sjunker längs botten, vilket oftast sker i slutet av stagnationsperioder, ökar utflödet av fosfor från i sedimenten i form av löst fosfat. Detta fenomen benämns internbelastning. Hur stort detta utflöde blir beror på det tillgängliga förrådet av mobila former av fos­for (huvudsakligen fosfor bundet till järn). Tidigare undersökningar av ak­kumulationsbottnar i öppna Östersjön har visat att förrådet av mobil fosfor i dessa bottnar är mycket begränsat, vilket kan förklaras av att det råder mer eller mindre permanent ansträngda syrgasförhållanden. Relativt få studier har genomförts för att undersöka fosforförrådets fördelningsmönster i kust­sediment samt vilka faktorer som styr fosforutbytet mellan sediment och vatten i kustzonen. Under 2008–2009 samlades därför ett hundratal sedi­mentkärnor in från olika ackumulationsområden långs Svealandskusten och deras innehåll av mobil fosfor analyserades (artikel IV).

Polyklorerade dibeno-p-dioxiner och dibensofuranser (PCDD/Fs) bildas oavsiktligt genom klorering av organiskt material vid olika typer av förbrän­ningsprocesser. Halterna av PCDD/Fs i biota från Östersjön minskar inte längre i samma takt som andra klassiska organiska miljögifter. I synnerhet fet fisk, t.ex. strömming och lax, från delar av Östersjön innehåller så pass höga halter av PCDD/Fs att de överskrida gällande gränsvärden för att få säljas. Orsaken bakom detta är oklar. För att bättre förstå hur PCDD/Fs om­sätts i kustzonen initierades ett forskningsprojekt i Kallrigafjärden långs Upplandskusten under 2007. Med speciell mätmetodik, utvecklad för de extremt låga koncentrationer av PCDD/Fs som uppträder i vatten (artikel V)
kunde flödena av PCDD/Fs från utanförliggande hav och tillrinnande vattendrag beräknas på årsbasis (artikel VI). Därefter tillämpade vi en generell modell för omsättning av partikulära substanser i kustområden för att simulerar utbytet av PCDD/Fs mellan vatten och sediment (artikel VII). Modellens förutsägelser av de olika PCDD/F-kongenernas koncentration i vattenmassan stämde väl överens med motsvarande empiriskt uppmätta koncentrationer.

Sammantaget har avhandlingsarbetet lett till nya insikter om hur föroreningar omsätts i kustområden. Ansatsen med empiriskt grundade massbalansmodeller har visat sig vara ett användbart verktyg för att beskriva, kvantifiera och förutsätta transportprocesser för olika substanser. Det bör finnas goda möjligheter att i framtida miljövårdsarbete använda metodiken för att identifiera de mest kostnadseffektiva åtgärderna mot vattenförrening i kustzonen.
Denna avhandling hade aldrig skrivits utan det stöd jag fått från ett stort antal personer. Jag vill rikta särskilt varma tack till

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