RESEARCH ARTICLE

Long-term eutrophication history and ecosystem changes in a large Baltic river basin – estuarine system

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Abstract

1 - The Odra river basin, the Szczecin Lagoon and its coastal waters in the southern Baltic region cover an area of about 150,000 km². We apply a spatial integrative approach to analyze the eutrophication history and functional changes of this area between early 1960s and 2000s. For this purpose, a river basin model is linked to a coastal and marine 3D-hydrodynamic ecosystem model. Annual riverine nutrient loads increase from the 1960s until the mid 1980s to 110,000 t nitrogen (N) and 15,000 t phosphorus (P) and decrease afterwards, but show a strong inter-annual variability. The contribution and location of emission sources in the catchment change significantly during the investigated 40 years. Nutrient and chlorophyll concentrations in the estuary follow changes in riverine N loads quickly. Therefore, river basin management measures should focus on both, P and N. Strong spatial concentration gradients in the estuary are always present. Nutrient availability, N/P ratios as well as processes like denitrification and N fixation show spatial and temporal changes during the 40 years. In the lagoon, N fixation does not play an important role and riverine N load reductions are not compensated by increased N fixation. The lagoon's quantitative N retention is stable over the 40 years. However, in the early 1960s, about 26% of the total N load into the lagoon is removed by denitrification, compared to only about 15% during 1999–2002.

Introduction

The Baltic Sea is one of the world-wide largest brackish water bodies (412,000 km²) with a water residence time of about 25–30 years, a drainage basin of 1,734,000 km² and a population in the drainage basin of about 85 million people. According to the Baltic Sea Action Plan (HELCOM, 2007): 'eutrophication is a major problem in the Baltic Sea, caused by excessive inputs of nitrogen and phosphorous which mainly origin from inadequately treated sewage, agricultural run-off and airborne emissions from shipping and combustion processes. The plan's objectives for eutrophication include: concentrations of nutrients close to natural levels, clear water, natural levels of algal blooms, natural oxygen levels, and natural distributions and abundance of plants and animals'. According to HELCOM (2005), over 90% (70%) of phosphorus (nitrogen) reaches the Baltic Sea via rivers. Of this

amount 53% (47%) of the phosphorus loads (nitrogen loads) enter along the south coast of the Baltic Sea (Germany, Poland and the three Baltic States). Therefore, large rivers like Odra (German: Oder), Vistula, Nemunas and Daugava in the southern Baltic region are of outstanding importance for Baltic Sea eutrophication and management. The southern Baltic coast is characterized by soft sediments, highly dynamic coastlines with long sandy beaches and a complex pattern of land and water. All large rivers do not enter the Baltic Sea directly but discharge their nutrient load into coastal estuaries, bays or lagoons. The quantitative significance of these coastal waters with restricted water exchange for Baltic Sea management is, in detail, not well known. Coastal waters serve as converters for entering nutrient compounds, as sinks and retention ponds and thereby control the amount and composition of nutrients entering the Baltic Sea.

One of the most important polluters along the southern Baltic coast is the river Odra with its complex and heavily eutrophied estuary system. Especially during summer, eutrophication effects like cyanobacteria blooms or fish kills can cause serious economic damage for tourism (Dolch and Schernewski, 2002; Wasmund, 2002). Because of its size, state of pollution as well as economic and ecological importance, the Odra estuary system has been intensively investigated. Lampe (1999) and Meyer and Lampe (1999) analyze the nutrient cycles, budgets and retention capacity, but exclude the entire Polish part of the estuary and focus on a few years only. Grelowski *et al.* (2000) on the other hand focuses on the Polish part only. Pastuszak *et al.* (2005) extends the view towards the Pomeranian Bay, but base all their nutrient transformation and retention calculations on a simplified gradient method instead of using hydrodynamic models. The first model simulation with the aim to analyze the long-term eutrophication in the lagoon has been carried out by Humborg *et al.* (2000), using a box model. Spatial aspects and the Pomeranian Bay are not considered in this approach and the sediment is included in a very simplified manner. Behrendt *et al.* (2008) criticize the calculations of the historic Odra river loads, which are based on fertilizer application data only. They state that this approach underestimates historic loads by far, and is not suitable to reproduce the eutrophication history of the lagoon. An improved box-model for the Szczecin lagoon (Wielgat and Witek, 2004) and a comprehensive load compilation (Wielgat, 2002) allow reliable nutrient budget calculations but are only applied for short periods.

Objectives of the present study are (a) to reconstruct the eutrophication history of the Odra estuary between 1960 and 2000, (b) to analyze the functional changes in the estuary during this period and their consequences for the Baltic Sea, (c) to assess the relationship between external loads and the water quality status, nutrient availability, limitation and algal biomass, (d) to improve our understanding about sources and spatial origin of the nutrient loads, and (e) to contribute to the discussion, on which nutrient a management should focus to reduce eutrophication in the Odra estuary.

Study site: the Odra/Oder river basin and estuary

The Odra estuary is located at the German/Polish border. It consists of the Szczecin Lagoon (Oder Lagoon) and the Pomeranian Bay (a part of the Baltic Sea) (Fig. 1). The lagoon is large (687 km^2) but shallow (average depth of 3.8 m) and can be subdivided into an eastern bay (Wielki Zalew) on the Polish territory and the Kleines Haff in the west, on the German side. Three outlets link the lagoon with the Pomeranian Bay. The tidal range is

below 10 cm and tidal water exchange can be neglected. The salinity in the lagoon is low (1–3 PSU) compared to 5-8 PSU in the bay. The entire estuary is controlled by the discharge of the Odra River which contributes at least 94% to the lagoon's water budget and dominates the nutrient budgets, as well. With a length of 854 km, a river basin of 118,861 km², an average annual discharge between 275 m³/s (1990) and 878 m³/s (1977) as well as a long-term average discharge of 542 m³/s (Löser and Sekścińska, 2005; IKSO, 2010), the Odra is the third largest river in the Baltic region. About 89% of the river basin is located in Poland (IKSO, 2010).

The Odra is a typical lowland river. Glacial sediments with intensive agriculture cover 90% of the river basin, which is inhabited by about 16 million people (Löser and Sekścińska, 2005). According to HELCOM (2009), several large cities and industries along the river are still major pollution 'hot-spots' in the Baltic Sea Region.

Methods

Until late 1970s, monitoring data are scarce and due to analytical differences and independent sampling strategies in Germany and Poland do not allow a comprehensive picture of the ecological situation. To be able to analyze, assess and evaluate the impact of the Odra river basin on coastal water quality during 40 years (1960-2000) models are needed, which extend into the past, increase temporal data density and ensure a comprehensive spatial picture. This means we need a spatially integrated, large-scale model approach covering the entire river basin – river – coastal water – sea system. For this purpose, the river basin model MONERIS is linked to the three-dimensional hydrodynamic and ecosystem model of the Baltic Sea, ERGOM. To be able to link both models, ERGOM is spatially extended from
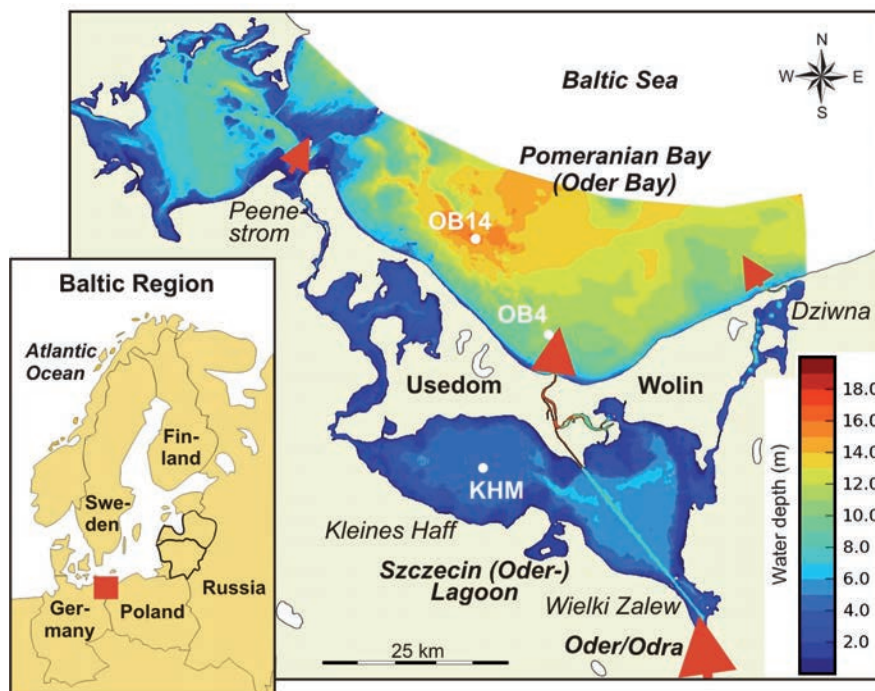

Figure 1. The Oder/Odra Estuary located at the German/Polish border (54°N, 14°E). Data from the three indicated stations are used in this study: the central Szczecin Lagoon (KHM), the coastal Pomeranian Bay (OB4) and the open Pomeranian Bay (OB14).

the Baltic Sea into the Szczecin Lagoon to cover the entire estuarine system.

MONERIS – nutrient flux model for the river basin

Reliable and consistent data on river discharge and nutrient loads are imperative for model applications in the estuary. Previous studies on nutrient inputs into the Odra River (Behrendt and Dannowski, 2005) already identified the most important sources, pathways, stocks and sinks of nutrients within the Odra and allowed the development of the river basin model MONERIS. This model has been applied to estimate the nutrient emissions by point and diffuse sources and the resulting Odra river loads over almost 50 years from 1955-2000 (Behrendt *et al.*, 2008).

Model outputs are annual data for river discharge as well as nitrogen (N) and phosphorus (P) concentrations for the period between 1960 and 2002.

MONERIS is able to represent the source as well as the pathway of nutrients in the river basin.. To be able to run the model, large amounts of spatial information had to be compiled: The river system, catchment and administrative borders, land use classifications, soil maps, topographical information, ground water tables, hydro-geological and hydro-meteorological information as well as data on atmospheric deposition, river flow and water quality. Point discharges from waste water treatment plants and industry enter the river system directly, but diffuse emissions into surface waters have very different pathways and have to be modeled separately. Altogether six diffuse pathways (sub-models) are considered in MONERIS: atmospheric deposition, erosion, surface runoff, groundwater, tile drainage and paved urban areas. Transformation and retention processes are taken into account along the pathway from the emission source into the river.

The use of a GIS allows a regionally differentiated quantification of nutrient emissions into river systems. In contrast to the study of Behrendt and Dannowski (2005) the simulations used in this study have a much higher spatial resolution of the river basin. Altogether nearly 500 river sub-catchments are calculated separately. These data are aggregated to larger units and finally to the entire river system. Because of data availability and funding reasons, detailed model calibrations and validations took place for the periods 1993–1997 and 1998–2002 (Behrendt *et al.*, 2008).

To calculate river loads and discharge for more than four decades, additional historical data and statistics especially about major human activities have to be taken into account. Official country statistics of the FAO (FAOSTAT 2004) supply data on population and land use changes as well as on fertilizer application, livestock and crop harvests. The nutrient surplus on agricultural areas is estimated according to OECD (1997). Sewer systems are the major point source. The degree of connection to sewer systems and waste water treatments plants (WWTPs) are calculated on the basis of statistical data for Germany, Poland and Czech Republic. Changes in the nutrient elimination of WWTPs for individual countries are estimated based on elimination rates and proportions of WWTPs with primary, secondary and tertiary treatment. Schmoll (1998) and Behrendt (1994) provide data on long-term specific human P emissions, P in detergents and indirect industrial discharges into sewer systems. For the period 1960 and 1990, we assume that the situation in Poland and the Czech Republic was similar to that in eastern Germany. For 39 monitoring stations in the sub-basins nutrient loads are calculated using the method by OSPAR (1996). The observed discharge time series at several stations are used as input data to adjust the model to the flow conditions. Additionally, all calculations are repeated with long-term

observed discharges. Detailed statistics and historical data are presented in Behrendt *et al.* (2008). Venohr *et al.* (2010) show that the average deviation between MONERIS simulated annual loads and observations (period 1990–2005; station Krajnik Dolny) is 15.5% for total N, 21.5% for inorganic N and 20.1% for total P. These deviations are very close to the uncertainties associated with observed loads. The five year averaged MONERIS loads are similar to averaged observed loads. Further descriptions of the model, including processes as well as the calibration and validation are given in Behrendt and Dannowski (2005).

In this study, the output of MONERIS is used as an input for ERGOM. We transformed the annual river load data from MONERIS into monthly data by applying a typical seasonal riverine discharge dynamics. This simplified approach fails to represent single extreme discharge events, like the Odra flood in 1997. However, long-term discharge data do not show extreme values for the focus periods of this study (1961–1964, 1985–1988, 1999–2002).

ERGOM – the 3D ecosystem model for coastal waters and the Baltic Sea

The ecosystem model ERGOM is an integrated biogeochemical model linked to a 3D circulation model covering the entire Baltic Sea. The circulation model is an application of the Modular Ocean Model (MOM 3) code and includes an explicit free surface, an open boundary condition to the North Sea and freshwater discharge by rivers. The applied ERGOM model grid has a horizontal resolution of 3 nautical miles in the south-western Baltic Sea. Towards north and east the grid size gradually increases up to 9 nm. Altogether 77 vertical water layers are assumed. The upper 100 m are resolved with a layer thickness of 2–3 m and below 100 m a constant layer thickness of 6 m is applied. To be able to link MONERIS to ERGOM and

to use the river loads and discharge data for the Baltic ecosystem model, the ERGOM grid is extended towards the lagoon applying the same high resolution as for the south-western Baltic Sea. The shallow, nearly always fully mixed lagoon on average is sub-divided into two vertical water layers. Horizontally, the lagoon is represented by about 25 grid cells. In the model, the lagoon is linked to the Baltic Sea by only one inlet representing the Swina Channel, which is, in reality, responsible for about 70–75% of the discharge into the Baltic Sea. Two other minor inlets are neglected. This approach allows us to simulate retention and transformation processes and reflects general horizontal gradients in the lagoon. However, it allows only a general representation of the lagoon and does e.g. not fully reflect the exchange between lagoon and bay.

The biogeochemical model consists of nine state variables. The nutrient state variables are dissolved ammonium and nitrate, which are summarized as dissolved inorganic nitrogen (DIN), as well as dissolved inorganic phosphorus (DIP). Primary production results from three functional phytoplankton groups: diatoms, flagellates and cyanobacteria. Diatoms represent larger cells which grow fast in nutrient-rich conditions. Flagellates represent smaller cells with an advantage at lower nutrient concentrations especially during summer conditions. The cyanobacteria are able to fix and utilize atmospheric N, and therefore, the model assumes P to be the only limiting nutrient for cyanobacteria. N fixation by cyanobacteria is a N source for the system. A dynamically developing bulk zooplankton variable causes grazing pressure on phytoplankton. Dead particles are accumulated in a detritus state variable. Detritus is mineralized into dissolved and bio-available ammonium and phosphate during the sedimentation process. A certain amount of the detritus reaches the bottom, where it is accumulated in the sedimentary

detritus. Detritus in the sediment is buried, mineralized or resuspended into the water column, depending on the velocity of near-bottom currents. The development of oxygen in the model is coupled to the biogeochemical processes via stoichiometric ratios. Oxygen concentration controls processes such as denitrification, nitrification and the release of ferric phosphates, which are a separate state variable. The biogeochemical model is coupled to the circulation model via advection diffusion equations for the state variables. Neumann (2000) provides a detailed model description and Neumann and Schernewski (2005), Schernewski and Neumann (2005), and Neumann (2007) present model applications and details about the data requirements as well as possibilities and limits of the model.

To ensure correct lateral transport and mixing processes, ERGOM always simulates the entire Baltic Sea, even if only smaller areas are of interest. However, in this study we restrict ourselves spatially and present only water quality results of the Szczecin Lagoon and the southern Baltic Sea. The simulation covers the years between 1960 and 2000. The most important input data are the water discharge and the nutrient loads of the Odra River, supplied by the river basin model MONERIS. Data for atmospheric N deposition between 1960 and 2000 are calculated on the basis of HELCOM (2005). Meteorological data are taken from the ERA-40 re-analysis for the entire period between 1960 and 2000. Therefore, all simulations between 1960 and 2002 are based on real and comparable weather, discharge and load data. A specific model calibration for the application in the lagoon has not been carried out.

Monitoring data
Model simulations are compared to German/ Polish monitoring data by the Polish and German State agencies Wojewódzki Inspektorat Ochrony Środowiska w Szczecinie (WIOS) and Landesamt für Umwelt, Naturschutz und Geologie (LUNG), respectively. Bangel *et al.* (2004) compiled all existing monitoring data for the Polish as well as the German part of the estuary and presented changes in nutrient concentrations during about 20 years as well as spatial gradients in the estuary. This dataset is used in the present study. The regular hydro-biological, chemical, and physical monitoring in the estuary started in the 1970s. In the lagoon, altogether 12 stations are sampled; additional 4 stations are sampled in the Pomeranian Bay. The sampling frequency is one month and the samples are taken in 1 m water depth. Total P (total N) is analyzed as phosphate (nitrate) after micro-wave pulping with peroxodisulfate. After filtration (0,45 µm), phosphate and nitrate are determined using automated photometric methods. For phosphate, ammonium molybdate is used (DIN EN 1189) and nitrate is determined as NO_2-N after Cd-reduction (DIN ISO 13395). The detection limits are 0.02 mmol/m³ for total phosphorus (TP) and P as well as 0.1 mmol/m³ for total nitrogen (TN) and NO_3-N. A spectral-photometric determination method based on filter residues, with a detection limit of 0.1 mg/m³, is applied for chlorophyll-a. Inter-calibration exercises as well as harmonized sampling dates ensure reliability and comparability between German and Polish data. The three stations with the longest and most complete data record (sampling started in 1975) are chosen for this study and the locations are indicated in Fig. 1.

Results

Long-term nutrient emission changes in the river basin
The river basin model MONERIS is able to calculate and visualize spatial nutrient

emission patterns. The spatial distributions of nutrient emissions into surface waters reflect the geologic and anthropogenic conditions in the sub-basins. Further, they strongly depend on weather conditions and therefore show a high interannual variability. To be able to present stable long-term spatial changes, we average the MONERIS model results for the years 1961–1964, 1985–1988 and 1999–2002 and compare these three periods. The following interpretation is based on the resulting MONERIS model maps (Fig. 2).

With respect to N emissions, groundwater, drainage of agricultural land via ditches and pipes as well as urban sources (sewerage systems and waste water treatment plants) are of major importance in the Odra river basin and, according to MONERIS, responsible for over 80% of the total emissions (Fig. 2). The spatial differences of urban N emissions reflect differences in population density as well as the quality and connectivity of the sewerage system and sewage treatment plants. The map for 1999–2002 suggests that even larger cities are no outstanding emission hot-spots any more. Diffuse emissions from the country-side dominate.

Areas with intensive agriculture have a high N surplus (difference between fertilizer application and uptake by plants and removal through harvest) and are usually N emission hot-spots. The central Warta region near Poznan is a good example, especially during the period 1985–1988. On the other hand, the geologic substrate and high ground water levels play an important modifying role. The lowlands of the Odra basin are formed by porous sediments. During transport through soil and sediments towards groundwater, dissolved N is subject to denitrification and lost to the atmosphere. Especially in flood plains, this process efficiently reduces the N loads into rivers. This is the reason for the very low N emissions from Odra, Warta and Notec river lowlands in northern Poland.

The mountains with bed rock in southern Poland and the Czech Republic have only a poor N retention and denitrification capacity. Therefore, these areas show the highest N emissions. Only 6% of the Odra basin is in the Czech Republic, but on average this area is responsible for 14% of the total N emissions (Fig. 2).

The intensification of agriculture between the 1960s until the 1980s is well visible. Improved sewage treatment and reduced fertilizer application in agriculture are most important for spatial differences between the 1980s and the years around 2000. The N surplus in Poland increases to about 6 t/km² until the mid 1980s, drops in early 1990s and increases steadily again to about 3 t/km² in the late 1990s (Behrendt *et al.* 2008). Despite a much lower N surplus around 2000 (compared to the 1980s), the total N emissions into the surface waters are only slightly lower. The reason is the long water residence time of 25 years in unsaturated soil layers and groundwater in the river basin. The N loads observed in the 1990s and even until 2002 are largely a result of agricultural practices during the 1980s.

According to MONERIS, soil erosion, point sources (e.g. industries and waste water treatment plants) and urban areas are of highest importance with respect to P emissions into surface waters (Fig. 3). Model simulations show a close positive relationship between population density and P emission. The emissions from point sources differ among the countries. About 6% of the Odra river basin is located in Germany but Germany is only responsible for 2.2% of the total P emissions from point sources. On the other hand, Poland contributes over 91% of the point sources emissions. One reason is the difference in quality (P elimination) and quantity of waste water treatment between the two countries. The other reason is the difference in P emissions per capita. Behrendt

and Dannowski (2005) assume an emission of 1.8 g P per person and day for Germany and a value of 3.26 g P per person and day for Poland, because phosphate-free detergents are standard in Germany but less common in Poland. The mountains in the southern and south-western part of the river basin are areas with high P emissions. Slopes and intensive rain are responsible for increased soil erosion. Especially for the 1980s, the MONERIS map shows high P emissions from areas all along the rivers Odra and Warta. They are a result of dense population and a high number of industries along these rivers (Fig. 3).

Nutrient loads and concentrations in the estuary between 1960 and 2002
The riverine nutrient loads at the Odra river mouth are based on MONERIS model simulations, as well. MONERIS calculates the emissions into the surface waters and the retention in the river and takes into account observed and modeled discharge data. To analyze the effect of N loads on coastal waters, not the total loads but dissolved inorganic (bio-available) nitrogen (DIN) and phosphorus (DIP) concentrations are of highest importance and are used in coastal and marine model simulations. The following results and interpretations are again entirely based on MONERIS model simulations.
Figure 4 shows the average annual DIN and DIP concentrations as well as the molar DIN/DIP ratio in river loads in relationship to the observed average annual Odra river discharge for all years between 1960 and 2002. The relationship between concentrations and river discharge is similar for N and P. The 1960s are characterized by low concentrations and the late 1980s and early 1990s by the highest concentrations. Recent years show significantly reduced concentrations compared to the decade before. In the 1960s, the concentrations in the river are largely independent of the river
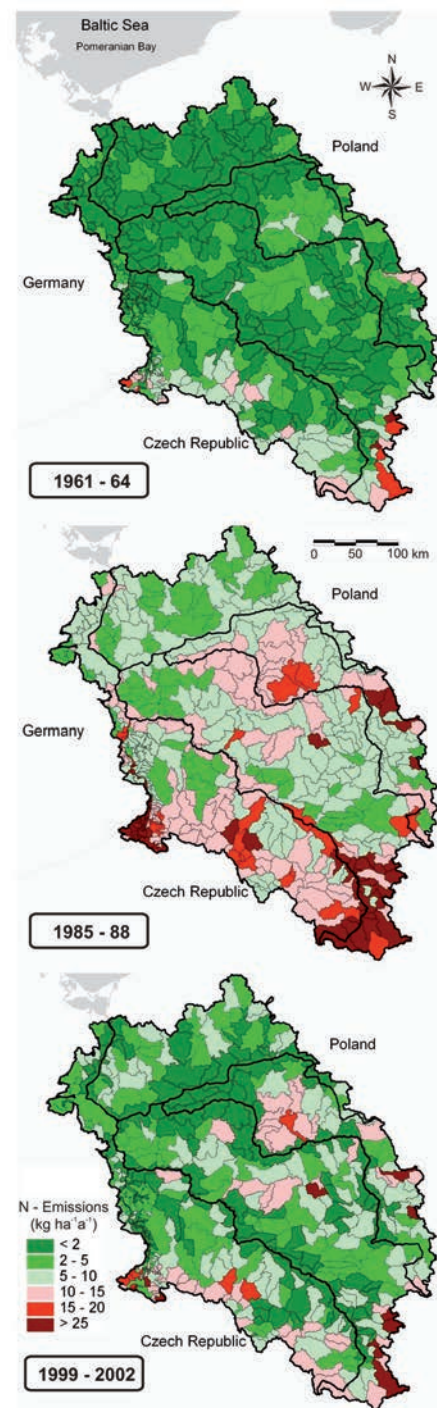

Figure 2. Averaged total nitrogen emissions into surface waters of the Oder/Odra river basin for the periods 1960–64, 1985–1988, 1998–2002 according to MONERIS model simulations (Input data have been supplied by the Institute of Meteorology and Water Management, Wroclaw, Poland).
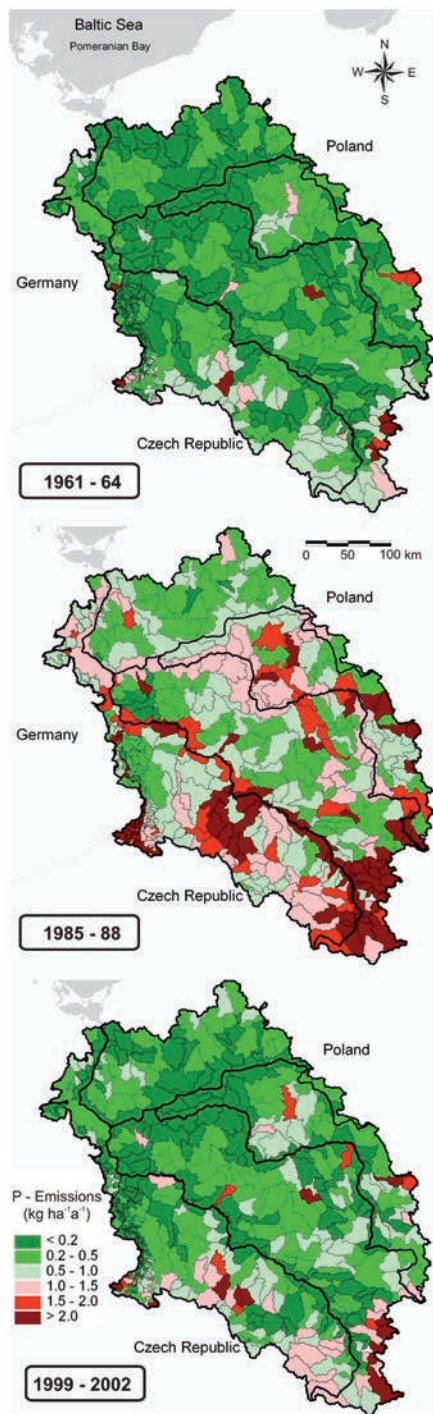
Figure 3. Averaged total phosphorus emissions into surface waters of the Oder/Odra river basin for the periods 1961–64, 1985–1988, 1999–2002 according to MONERIS model simulations (Input data have been supplied by the Institute of Meteorology and Water Management, Wroclaw, Poland).

discharge. Consequently, years with a high observed average water discharge, like 1967 with 764 m³/s cause a much higher load compared to drier years like 1964 (421 m³/s). In 1967, an estimated 45,690 t N (3,541 t P) enters the lagoon. This is more than 60% higher than in 1964, when loads of 27,981 t N (2,132 t P) are calculated. The highest annual discharge ever observed is the reason why 1977 became one of the years with the highest pollution loads (81,616 t N; 5,592 t P). Riverine loads are a major driving factor for processes in the estuary. When discussing single years in the estuary, it is important to keep in mind the high variability of river discharge and river loads from year to year. According to model and data, nutrient concentrations in the river increase steadily until the late 1980s and decrease afterwards. In years with average or high discharge (above 500 m³/s), there is no obvious relationship between discharge and N/P ratio. Years with very low discharge (below 400 m³/s), like 1983–1984 and 1989–1993, show significantly higher DIN and DIP concentrations than all other years. In these years, the N/P ratio is lower, indicating a relatively higher availability of P. Model results show that N loads are dominated by diffuse agricultural sources, like drainage, runoff and groundwater. These sources depend directly on the amount of precipitation, therefore dry years cause relatively low N loads. P is mainly coming from urban and industrial point sources and is related to human water consumption, which are not much affected by dry years. There is no systematic tendency in N/P ratios between 1960s and mid 1990s. Only most recent years (1998–2002) show a higher N/P ratio. Improved sewage treatment plants cause a decrease in P loads while ongoing intensive agriculture keeps the N loads high (Fig. 5 and 6). During the 1990's, the N availability in river loads increases relative to P and this process is ongoing.
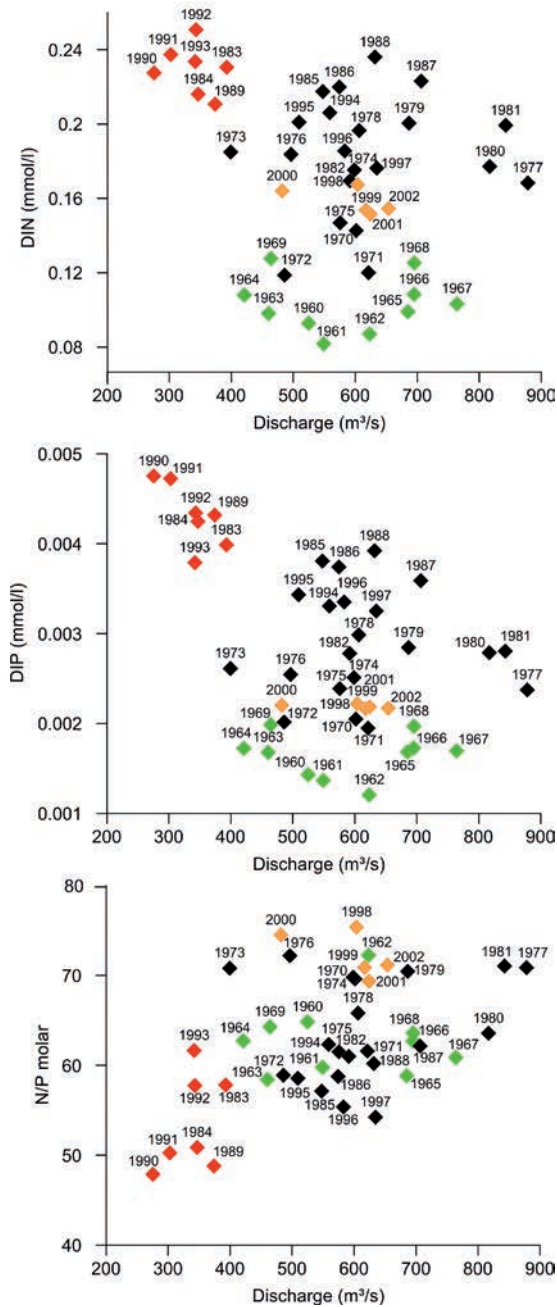
Figure 4. Average annual Dissolved Inorganic Nitrogen (DIN) and Phosphorus (DIP) concentrations as well as the molar DIN/DIP ratio in relationship to the average annual Odra river discharge. The data cover the period 1960-2002 and were calculated by the model MONERIS for the station Krjnik Dolny (south of the river mouth). Indicated are dry years (red), the 1960s (green) and the years 1998–2002 (orange).

According to MONERIS results (Fig. 5), the estimated total annual N loads (5 year averages) increase from nearly 50,000 t in the early 1960s up to over 110,000 t in the mid 1980s and declines to slightly below 100,000 t in 2000. MONERIS long-term monthly data show a strong annual cycle, with high loads during winter and low loads during summer months (Fig. 5). Again, the inter-annual variability is very high. Despite much higher total annual loads, summer N loads during 2000 do not differ much from summer loads in the 1960s.

MONERIS output serves as a consistent input for a long-term simulation with the marine and coastal model ERGOM. Figures 10–12 give an impression about the model performance. There, averaged observed data are compared to modeled monthly averaged data, while in Fig. 5–7 all observations are plotted and against simulated monthly averaged data. With respect to N, ERGOM results are well in agreement with observed data (Fig. 5). This is especially true for the period 1999–2002 and the lagoon. Changes in river loads cause similar changes in DIN concentrations in the lagoon. This is obvious both in modeled and observed data (Fig. 5). In most years, N is depleted (observed concentrations below 1 mmol/m³) in the water column of the central lagoon (station KHM, Fig. 1) in late summer. It can be regarded as a short and potentially limiting element for primary production. Only during the period 1977–1997 several years show excess N in the water even during summer. Uptake, sedimentation and denitrification in the lagoon reduce N concentrations. These processes determine the amount of N which is available for an export to the Baltic Sea. Annual maximum concentrations in the lagoon are approximately six times higher compared to the coastal Baltic Sea (station OB14, Fig. 1).

According to MONERIS model results presented in Fig. 6, the annual total P loads
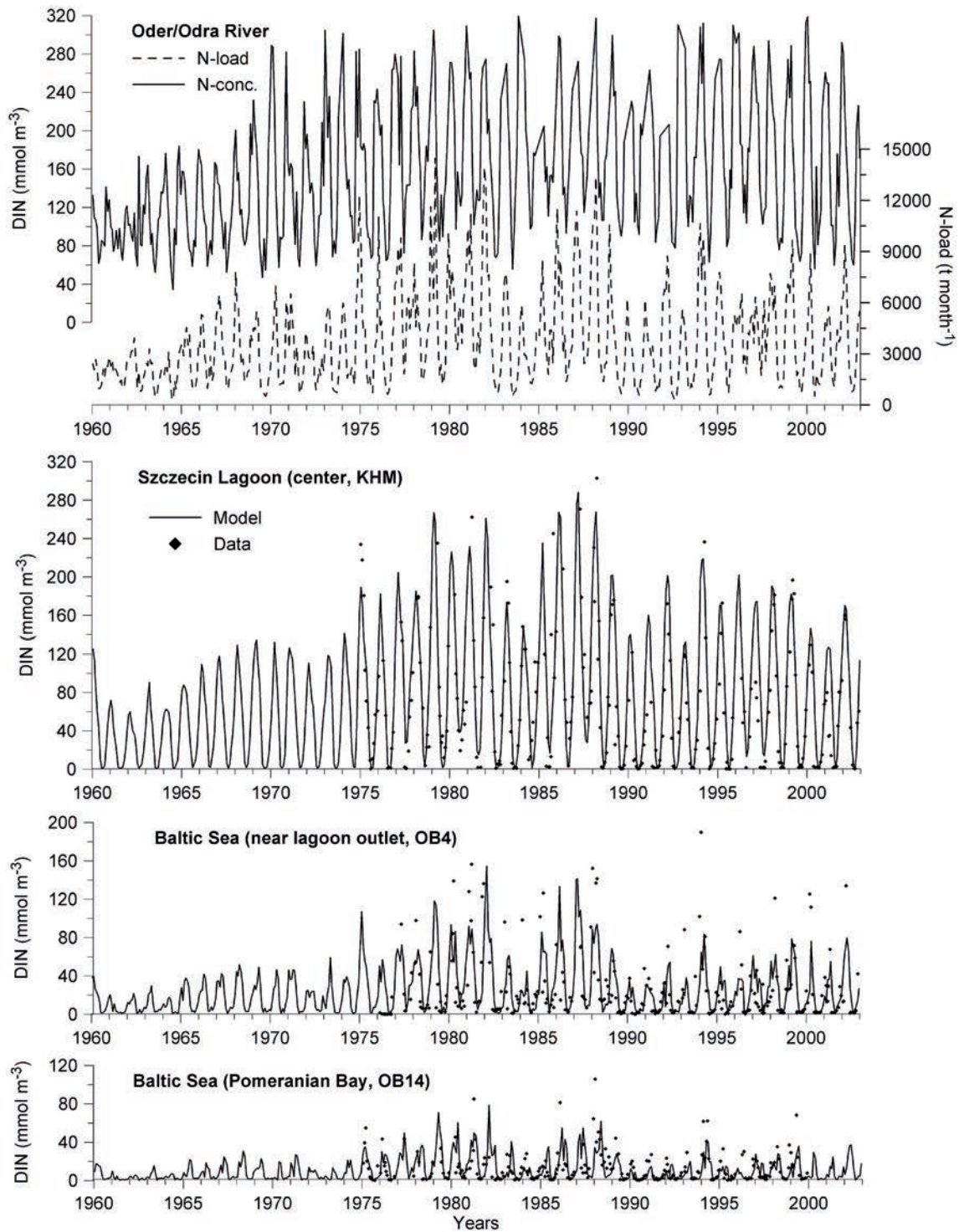
Figure 5. Nitrogen (Dissolved Inorganic Nitrogen, DIN) loads and concentrations in the Oder/Odra river and estuary. The labels and years indicate the 1st of January. Oder/Odra river loads are based on MONERIS model simulations. In the estuary, concentrations simulated with the ERGOM model are aggregated to monthly averages, while the measured data represent single samplings near the water surface (data source: LUNG).
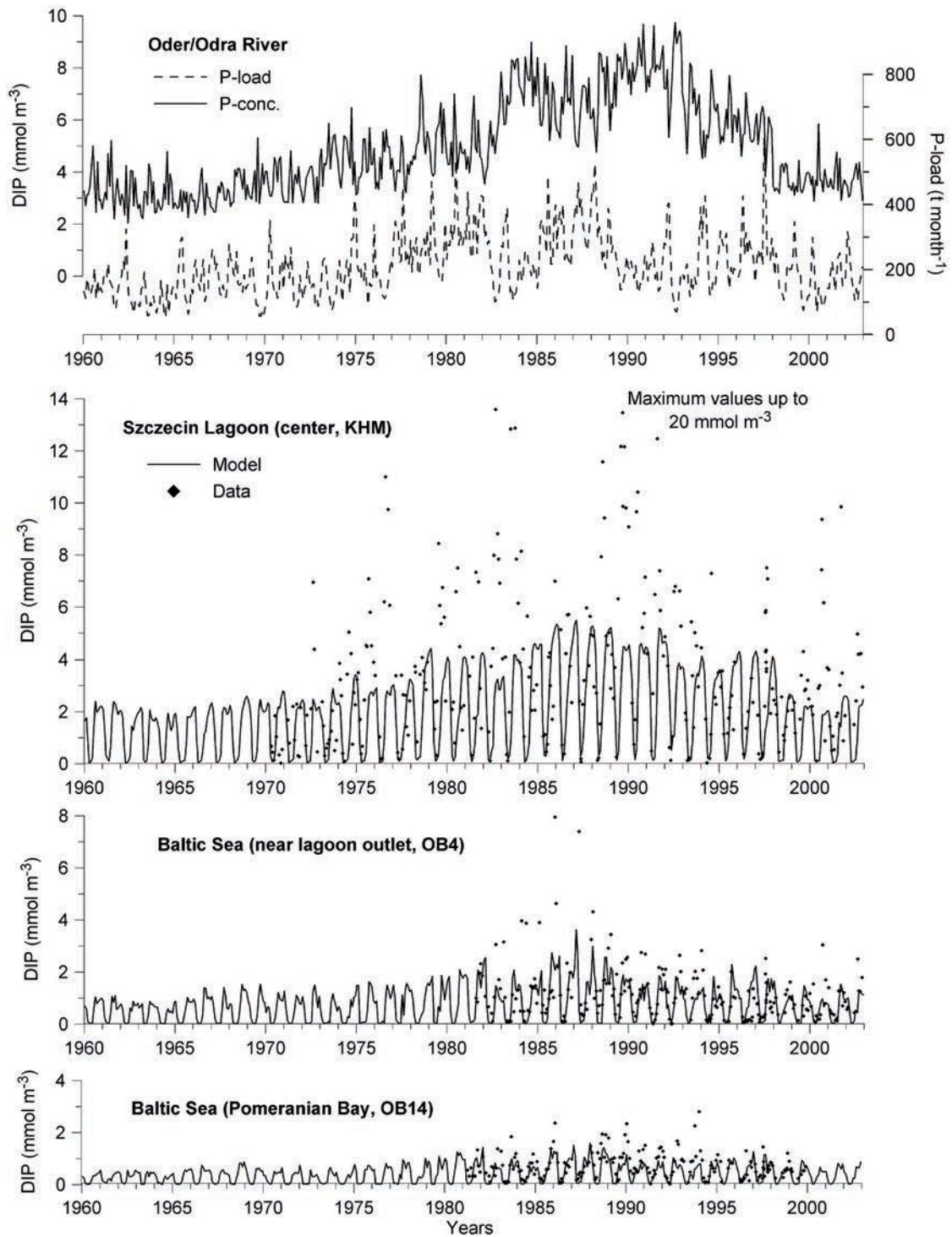
Figure 6. Phosphorus loads (bio-available P) and concentrations (Dissolved Inorganic P, DIP) in the Oder/ Odra river and estuary. The labels and years indicate the 1st of January. Oder/Odra river loads are based on MONERIS model simulations. In the estuary, concentrations simulated with the ERGOM model are aggregated to monthly averages, while the measured data represent single samplings near the water surface (data source: LUNG).

(5 year averages) increase from nearly 6,000 t in the early 1960s up to over 15,000 t in the mid 1980s and decline to values below 9,000 t in 2000. In general, the increase of the loads until the late 1980s and the decrease afterwards is reflected in the DIP concentrations in observed data as well as in model simulations. Only in the lagoon, differences between model and observed data are obvious during summer (Fig. 10).

One reason is that data are single samplings, while the model results are aggregated to monthly values. Another reason are short-term summerly hypoxia events, with subsequent P release from the sediment. This process is included but not sufficiently reproduced in the present spatial resolution of our model. . However, this short-coming does not have implications for our model-based analysis, because strong internal eutrophication events take place only in several years, over a short time of days and do not have lasting ecological consequences in the lagoon.

DIP concentrations show a steep decrease from the lagoon towards the coastal Baltic Sea (Fig. 6). Annual maximum concentrations in the lagoon are about four times higher compared to the coastal Baltic Sea. Nutrient retention processes and a strong water exchange with the open Baltic Sea are the reasons for these pronounced spatial gradients.

Fig. 7

ERGOM is able to simulate long-term changes in chlorophyll-a concentrations (an indicator for phytoplankton biomass) in the entire estuary (Fig. 7). The differences between simulation results and monitoring data, especially in the central lagoon, are largely due to strong lateral and vertical phytoplankton patchiness and the fact that ERGOM model results are aggregated to monthly averages, while monitoring data represent single samplings near the water surface. Again, a better insight into model performance is given in Fig. 10–12. The model reflects long-term changes in riverine nutrient

loads on summer chlorophyll concentrations, showing an increase until the 1980s and a decrease around 2000. According to the model, chlorophyll can exceed extremely high monthly average concentrations of 100 mg/m³ in the lagoon. Single observations even show concentrations above 160 mg/m³. In the coastal Baltic Sea (station OB14), average monthly chlorophyll-a values above 10 mg/m³ are rare and single observed data hardly exceed 20 mg/m³.

Figures 8 and 9 show the spatial distribution of modeled concentrations in the surface water layer of the Pomeranian Bay (coastal Baltic Sea, Fig. 1). ERGOM model results are averaged over the periods 1961–1964, 1985–1988 and 1999–2002 for summer and winter. The size of the plume depends on season, discharge and nutrients loads. In general, high nutrients loads during the 1980s cause the largest river plumes. During summer, primary production in the lagoon removes dissolved N from the water columns and transforms it into organic, particulate compounds. Consequently, the transfer of inorganic N from the lagoon into the sea is relatively low during summer and inorganic N is the element with the lowest availability in the lagoon as well as in the bay. Inorganic N concentrations in the coastal Baltic Sea are very similar between 1960s, 1980s, and 2000. With respect to inorganic P concentrations in summer, the situation is different. High concentrations in the lagoon during summer and relatively low concentrations in the Baltic Sea cause steep spatial gradients in coastal waters. During winter, increased riverine N and P loads are passing the lagoon without transformation and only minor retention. As a result, the model suggests high winter concentrations in the coastal Baltic Sea.

Figures 8c and 9c show the relative increase of dissolved nutrient concentrations between the early 1960s and 1999–2002 for winter and summer based on model results.

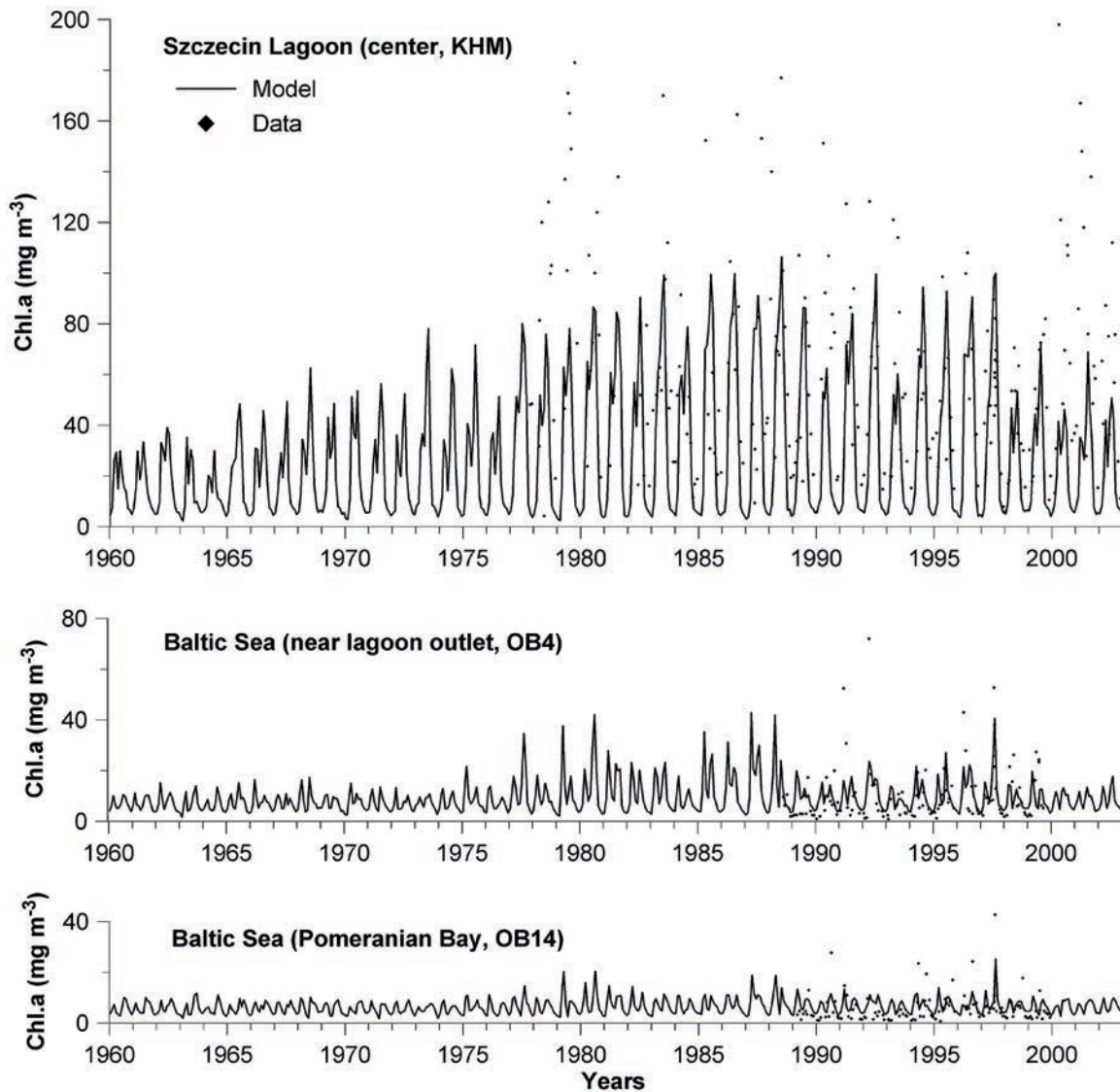During winter, the N concentrations in the

Figure 7. Chlorophyll a concentrations in the Oder/Odra river and estuary. The labels and years indicate the 1st of January. Oder/Odra river loads are based on MONERIS model simulations. In the estuary, concentrations simulated with the ERGOM model are aggregated to monthly averages, while the measured data represent single samplings near the water surface (data source: LUNG).

years around 2000 are about 50% higher compared to the early 1960s. Higher loads during the years around 2000 cause higher concentrations in the coastal sea. The same is true for spatial DIP surface concentrations during winter. After release into the coastal Baltic Sea, the Odra water and nutrient loads are usually transported with the prevailing currents along the Polish coast towards east. However, Fig. 8c and 9c show for both DIN and DIP highest winter concentrations along the Polish coast and not where the Odra enters the Baltic Sea. This is not an effect of riverine discharge or additional nutrient sources along the coast but of upwelling processes.
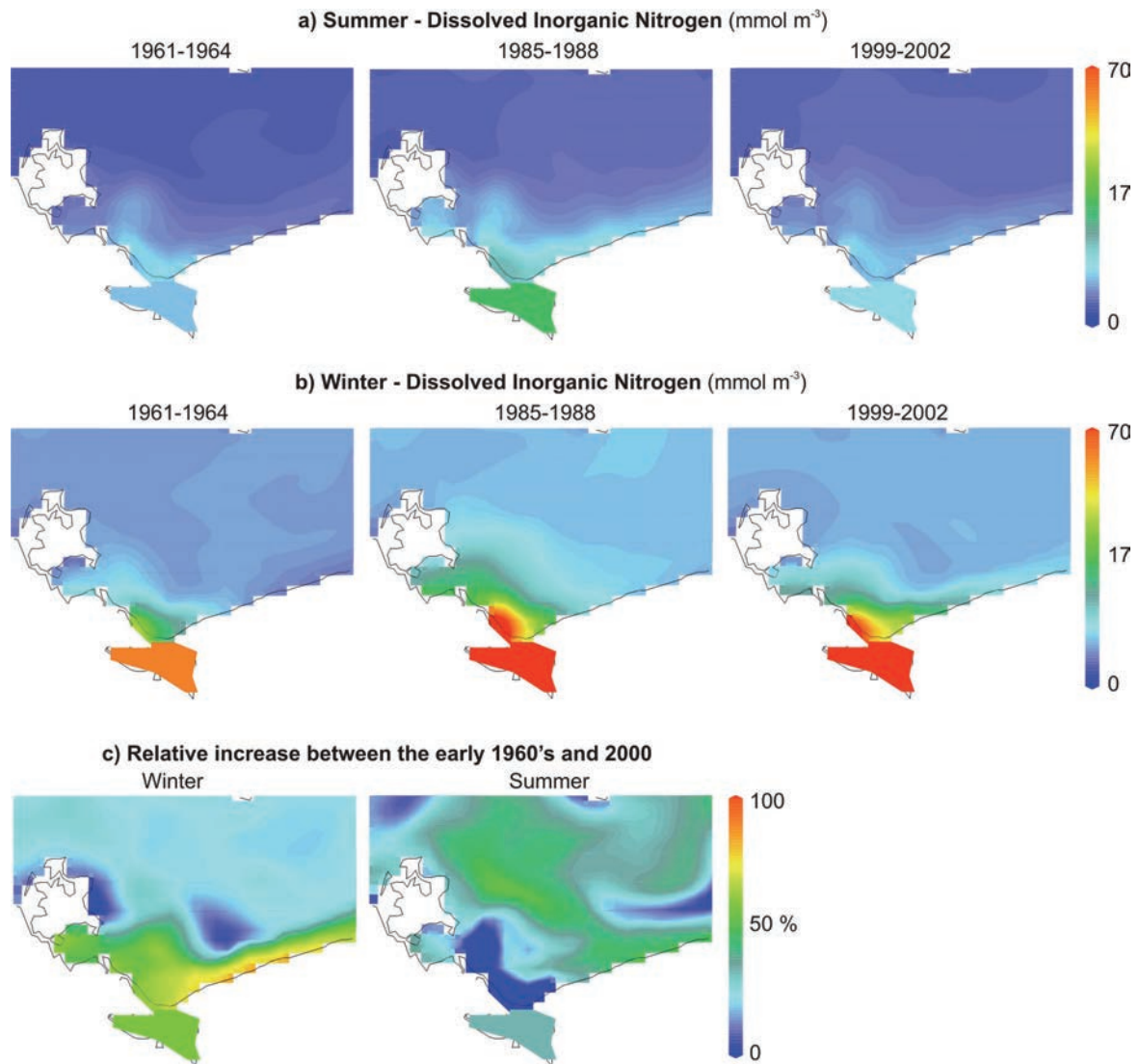
Figure 8. Spatial Dissolved Inorganic Nitrogen (DIN) concentrations (mmol/m³) in the surface water layer of the Pomeranian Bay. The data are ERGOM model results averaged over the periods 1961 – 1964, 1985 – 1988 and 1999 – 2002 for a) summer (July - September)and b) winter (December - February). c) Relative increase (%) between the early 1960s and 1999 – 2002 for winter and summer. In the lagoon, only spatial averages are shown.

The model calculates a slightly stronger upwelling of nutrient rich bottom water during the winters of the years around 2000 which result in locally increased concentrations. During summer, the situation is more complex. Although the DIN loads are higher in the years around 2000 than in the 1960s, model results show similar concentrations in the discharge plume and in the coastal sea during both periods. According to the Redfield ratio, N is the element with the lowest availability in the coastal water plume during summer. This holds for the early 1960s as well as the years around 2000. The model suggests that as soon as inorganic N enters the sea, it is taken up phytoplankton, independent of the
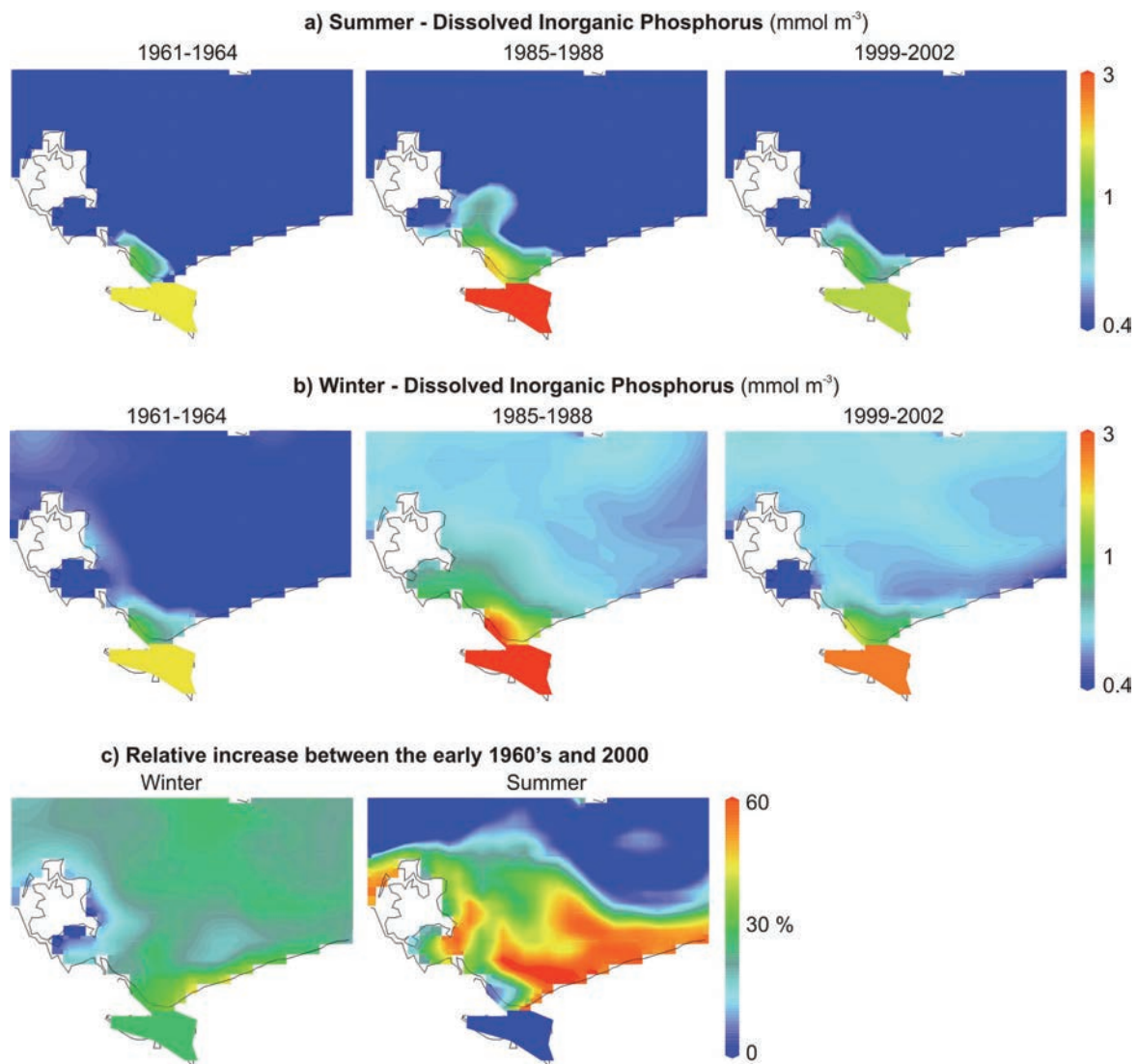
Figure 9. Spatial Dissolved Inorganic Phosphorus (DIP) concentrations (mmol/m³) in the surface water layer of the Pomeranian Bay. The data are ERGOM model results averaged over the periods 1961 – 1964, 1985 – 1988 and 1999 – 2002 for a) summer (July - September)and b) winter (December - February). c) Relative increase (%) between the early 1960s and 1999 – 2002 for winter and summer. In the lagoon, only spatial averages are shown.

load differences between the periods. Very low (and similar) DIN concentrations during both periods are the result.

On the other hand, parts of the open Baltic Sea and the Polish coast show an increase of summer DIN and DIP concentrations from the 1960s to the early 2000s. Again the explanation is slightly stronger upwelling of nutrient rich bottom water during the summers of the years around 2000.

Long-term changes of seasonal pattern in the estuary

In the central lagoon, the model suggests a similar seasonal pattern of DIN, DIP and chlorophyll-a among the early 1960s, the

mid 1980s and the years 1999–2002 (Fig. 10). Highest N concentrations are calculated and observed in February and March and lowest concentrations occur in August. In the 1960s and among the years 1999–2002, N concentrations drop below 1 mmol/m³ in late summer. During the 1980s, N is always abundant during the entire year.

Calculated and observed P-concentrations in the water column are high in winter and drop to low average monthly values in April and May. For the period 1961–1964, the model suggests average monthly DIP concentrations below 0.1 mmol/m³ in May, and in some years in April as well. In the mid 1980s, average monthly DIP concentrations in May are much higher, usually around 0.3 mmol/m³, and the time period with low concentrations is limited to only one month. The few monitoring data suggest higher average spring DIP concentrations in the 1980s compared to the model, but this is mainly an effect of one sample with high concentrations above 1 mmol/m³ in spring 1985. In other years, monitoring data at least once shows values below 0.1 mmol/m³ in May. For the years 1999–2002, the model suggests a situation close to the early 1960s in the lagoon (Fig. 10), with average DIP concentrations below 0.1 mmol/m³ in every May. In general, the monitoring data support this model outcome. In 1999 and 2000, data show concentrations below the detection limit. Only in 2001, DIP concentrations remain above 0.5 mmol/m³. Data are lacking for 2002. In general, the model suggests a significantly longer period of low concentrations in the 1960s than in the period 1999-2002. However, this temporal extension of low concentrations is not supported by the few available monitoring data. With respect to P in the lagoon, data and model results seem to differ in some periods, but these differences are mainly an effect of the few available monitoring data and single years with outstanding high concentrations. Nutrient concentrations in the lagoon are

directly controlled by riverine loads, but biological processes are responsible for the strong seasonal dynamics. The level of chlorophyll concentrations shows a close relationship to the nutrient concentrations in the lagoon. In the 1960s chlorophyll concentrations, especially during summer, are relatively low according to the model (30 mg/l), nearly triple until the mid 1980s and decrease again to an intermediate level around 60–70 mg/l. This long-term course of chlorophyll-concentrations is supported by monitoring data. Differences between model and data have two reasons: phytoplankton patchiness and, especially in winter, the lack of monitoring data (no sampling because of ice-cover). In warm winters, without ice-cover, diatoms can develop and cause high chlorophyll concentrations. The presented model results are averages over warm and colder winters and the colder winters are responsible for the lower chlorophyll concentrations.

The station at the outlet of the lagoon, only a few kilometers off the coast (Fig. 11), is sometimes strongly influenced by the nutrient loads from the lagoon and sometimes dominated by large scale transport, mixing and exchange process in the Baltic Sea. The levels of nutrient and chlorophyll concentrations reflect its position in a gradient between lagoon and open sea. The general changes from relatively low, in the early 1960s, to a maximum in the mid 1980s, and back to intermediate concentrations in the years 1999–2002 is similar to what we obtain for the other stations. In the Pomeranian Bay (Fig. 12), the influence of the river Odra and the lagoon is less visible. However, the general seasonal pattern with minimum concentrations of N in August and of P in May is similar to the lagoon. In general, model results and monitoring data are very well in agreement in the coastal sea. The DIN concentrations of the period 1985–1988 are the exception.
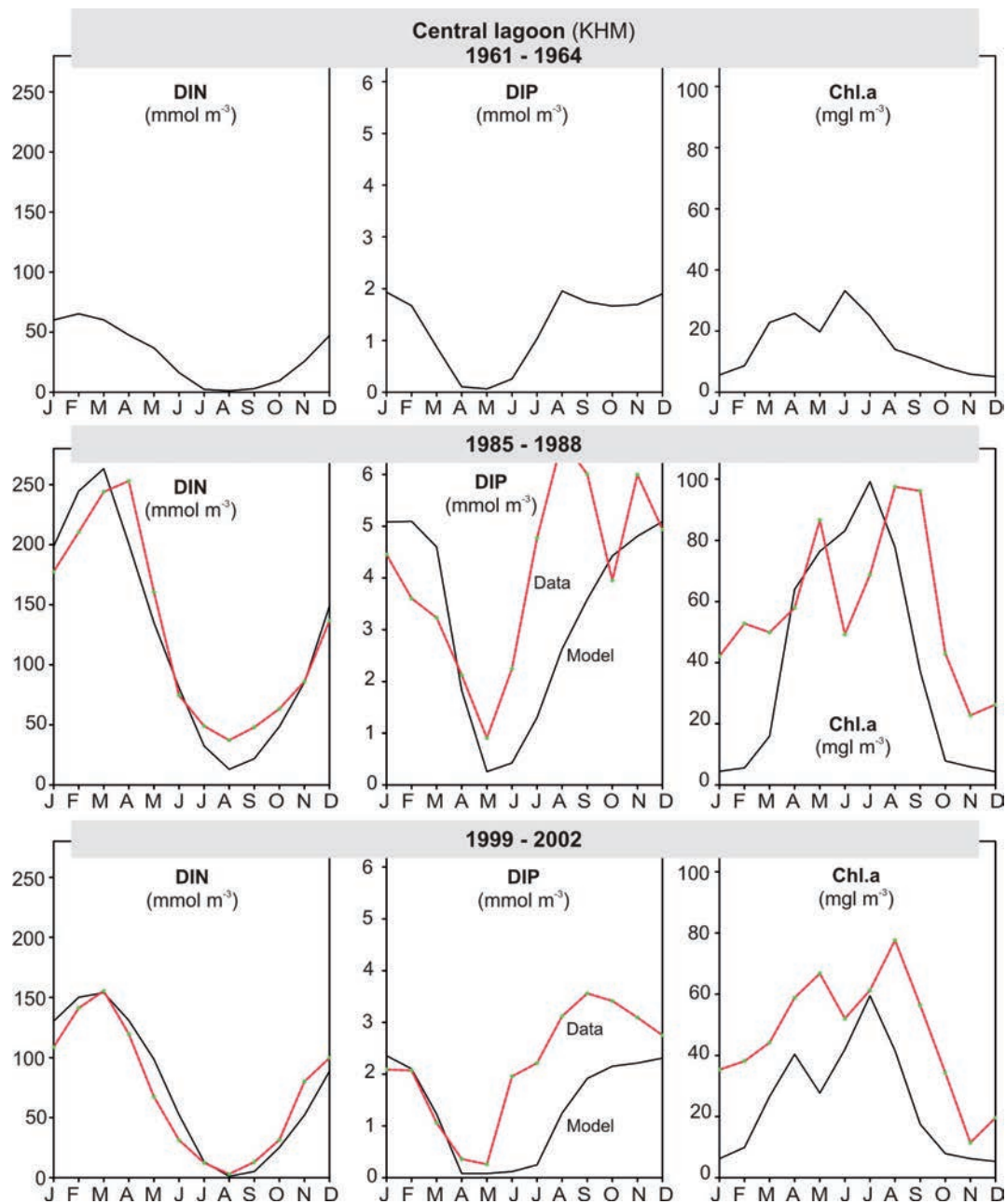
Figure 10. Central Szczecin Lagoon: monthly Dissolved Inorganic Nitrogen (DIN), Dissolved Inorganic Phosphorus (DIP) and Chlorophyll-a concentrations averaged over the periods 1961 – 1964, 1985 – 1988 and 1999 – 2002. Thin solid lines indicate model results and bolt, dashed lines averaged monitoring data.

Here, monitoring data show an extreme interannual variability. For example, the time of the observed maximum DIN concentration varies among the years (January 1988, February 1986, April 1985, and May 1987). The same is true for the absolute value of the maximum concentrations, which range between 21 mmol/m³ (1985) and 105 mmol/m³ (1988). The averaging over four years is responsible

for the smooth course of the observed DIN concentrations (1985–1988). It hides the extreme interannual variability of the data. The same is true for the summer situation. In 1988, monitoring data show extremely high DIN concentrations in August of 33 mmol/m³, compared to 7 mmol/m³ suggested by the model. This single value is responsible for the difference between model and data, because in all other years the average monthly DIN concentrations in August from monitoring data were between 1 and 3 mmol/m³. However, even these low concentrations are significantly higher than in the lagoon during August.

For the early 1960s the model suggests much lower concentrations between 0.4 and 0.9 mmol/m³ in July for the Pomeranian Bay compared to the 1980s. During the period 1999–2002 model and data show August-concentrations between 0.7–2 mmol/m³ DIN. This indicates that reduced riverine N loads are cascading through the coastal systems into the Baltic Sea and that N potentially could become a limiting element in coastal Baltic waters.

In the Pomeranian Bay in early 1960s, DIP simulations shows a minimum in April with concentrations between 0.016 and 0.054 mmol/m³. This is much less than in the lagoon and indicates that P is depleted from the water column. In the mid 1980s and for the years 1999–2002 the model indicates similar concentrations, but monitoring data show higher values, usually between 0.1 and 0.2 mmol/m³. In some years, the concentrations drop below the detection limit. However, the amount of P in the water column in winter seems to influence the intensity of the spring diatom bloom. Therefore, modeled chlorophyll concentrations are highest in mid 1980s, when the highest P concentrations in winter are observed (Fig. 12). In the coastal sea, the summer phytoplankton bloom, which is often dominated by cyanobacteria, does not differ much among the 1960s, the

1980s and the years around 2000. Differing N concentrations in the water column among the periods seem to have no effect.

Changes in ecosystem function and behavior
Baltic coastal waters are intensively used during summer and bathing tourism is the major source of income in the Odra estuary region. Therefore, algal blooms in summer are of great practical importance. This is especially true if cyanobacteria form these blooms and cause a risk of intoxication and beach closures. In the lagoon, which is known for intensive cyanobacteria blooms, Aphanizomenon spec., Planktothrix spec., Limnothrix spec., Anabaena spec., Microcystis spec., Woronichinia compacta and Snowella spec. are most common. These species often contribute 50% of the algal biomass in the lagoon in late summer and show a high bio-volume of usually 9 mm³/l. The same is true for the coastal Baltic Sea (OB 4), but the cyanobacteria bio-volumes are much lower, usually around 1.2 mm³/l. However, not all species possess heterocysts and are able to utilize atmospheric N. In the open Baltic Sea, Aphanizomenon, Nodularia and Anabaena are the most important N fixing heterocystous filamentous cyanobacteria (e.g. Larsson *et al.*, 2001). In the lagoon, only Anabaena and Aphanizomenon for sure have the ability to fix N. In several years, like 1990 or 1998, Aphanizomenon spec. bio-volumes above 10 mm³/l have been observed in the lagoon. However, there is evidence of N fixation by non-heterocystous cyanobacteria, as well (Wasmund *et al.*, 2001). Cyanobacteria and the intensity of N fixation during summer can serve as an indicator for N limitation. The model ERGOM does not differentiate between species. Cyanobacteria only form one state variable and we assume that all cyanobacteria are able to utilize atmospheric N. Background for this simplification is the assumption that heterocystous cyanobacteria will out-compete
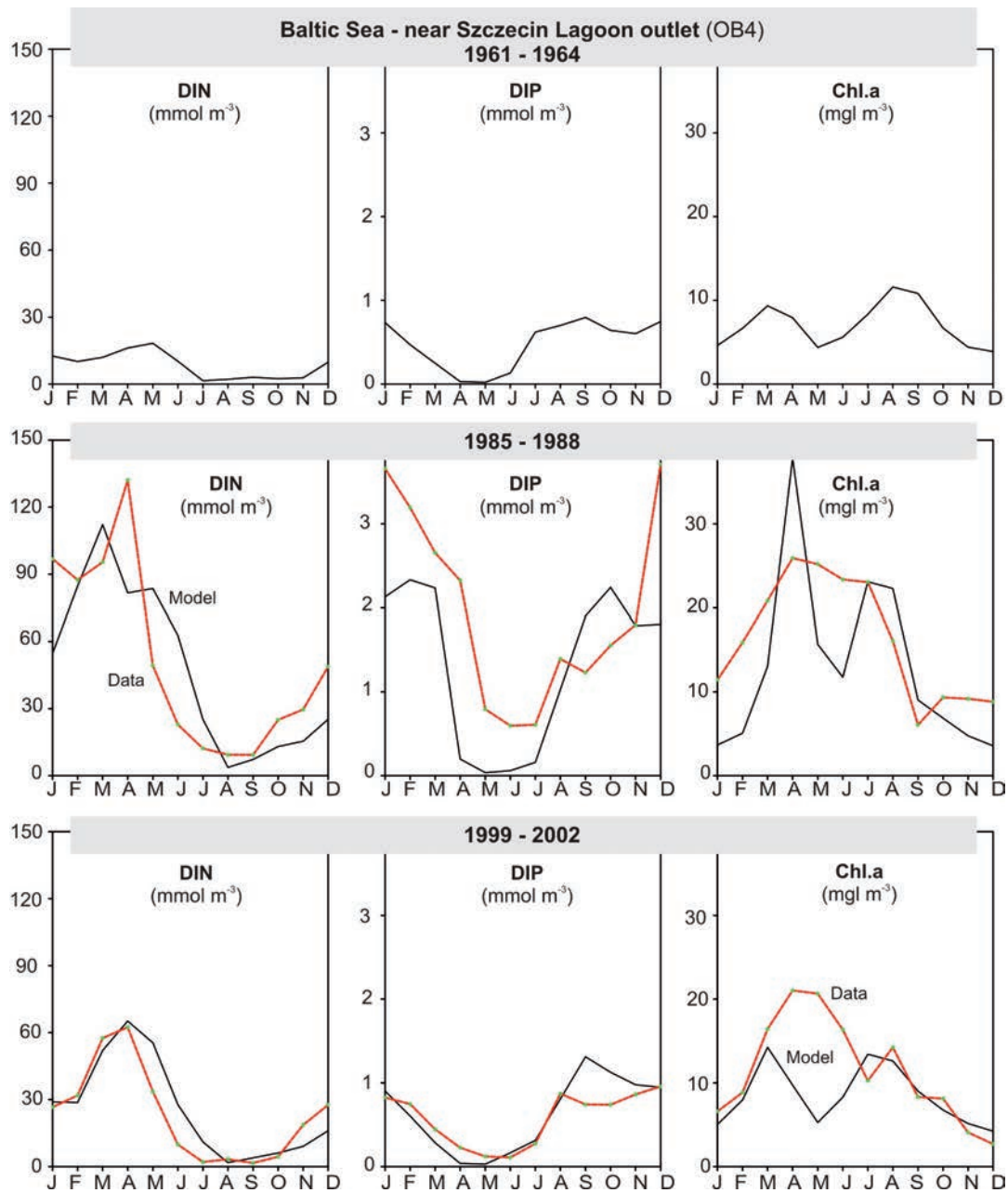
Figure 11. Coastal Pomeranian Bay (Baltic Sea): monthly Dissolved Inorganic Nitrogen (DIN), Dissolved Inorganic Phosphorus (DIP) and chlorophyll-a concentrations averaged over the periods 1961 – 1964, 1985 – 1988 and 1999 – 2002. Thin solid lines indicate model results and bolt, dashed lines averaged monitoring data.

other species under N limiting conditions. Figure 13 shows ERGOM simulation results for potential N fixation in the lagoon and in the coastal Baltic Sea. Data on actual N fixation are not available. According to the model, N fixation in the lagoon is limited to July, August and September, and takes place to a minor degree even in October. The model suggests that N fixation does not play an important role in the lagoon in 1961 and
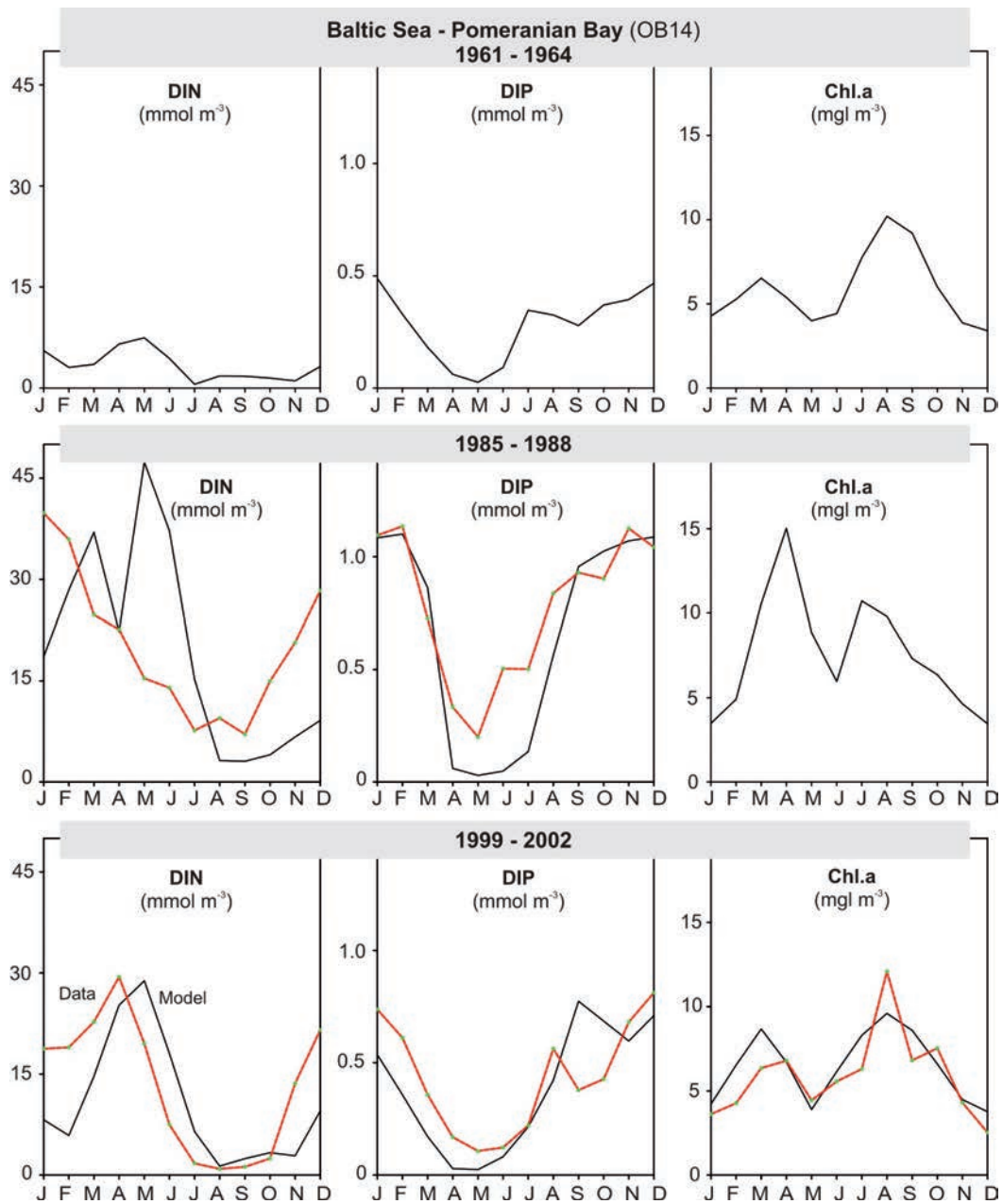
Figure 12.Pomeranian Bay (Baltic Sea): monthly Dissolved Inorganic Nitrogen (DIN), Dissolved Inorganic Phosphorus (DIP) and chlorophyll-a concentrations averaged over the periods 1961–1964, 1985–1988 and 1999–2002. Thin solid lines indicate model results and bolt, dashed lines averaged monitoring data.

1962, but intensive cyanobacteria blooms in 1963 and 1964 cause rates up to 5 mmol N m^{-2} d^{-1}. Due to the model, N fixation causes an average annual load of about 1,500 t N/a for the period 1961–1964. This is only 3%

of the annual riverine N load at that time. However, in August of single years like 1963, monthly loads by N fixation exceed 1,600 t N and can contribute more than 30% to the total monthly load into the lagoon.

According to the model, N is always sufficiently available in the lagoon during the 1980s. N fixation does not play a role at that time. During late 1990s the model suggests N fixation of 170 t N/a. This is much less compared to the 1960s. It can be concluded that decreasing riverine nutrient loads cause an increasing N fixation in the lagoon, but the compensation due to N fixation remains quantitatively marginal. However, the model shows that N fixation can become a temporary important N source. This is especially true for areas far off the river mouth. A strong and lasting limitation of primary production in late summer due to N depletion in the lagoon water is not obvious from the results of this study. It is likely that riverine loads, a fast nutrient turnover in this shallow system and mineralization processes

maintain a sufficient N supply in the lagoon even during intensive production in summer. The competitive advantage of an additional N source seems to be not high enough to carry out the energy-consumptive N fixation. In a larger temporal scale, model and monitoring data show that N loads have an immediate impact on N concentrations in the lagoon and determine the level of summer chlorophyll concentrations. One has to keep in mind, that the water transparency in the lagoon is very low during summer, with a Secchi depth between 0.3 and 0.8 m. Therefore, the availability of light is another important factor for primary production. In the coastal Baltic Sea, N fixation in the early 1960s and the late 1990s is, according to the model, on a comparable level. Average values of 2 mmol N m^{-2} d^{-1} are calculated for August (Fig. 13).
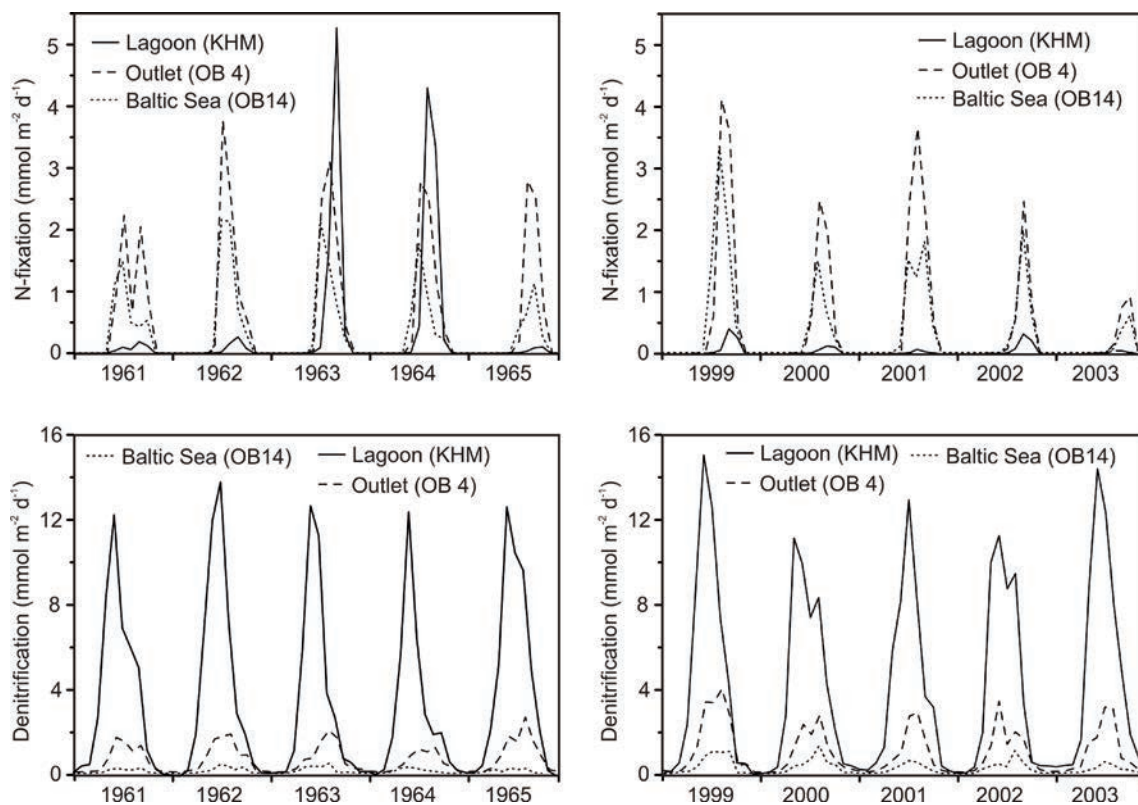

Figure 13. Changes of N fixation and denitrification between the early 1960s and the early 2000s for stations in the central Szczecin Lagoon (KHM) and the lagoon outlet in the Baltic Sea (OB4) and the coastal Pomeranian Bay (Baltic Sea) (OB14). The data are ERGOM model results.

During this month, N fixation is a very important nutrient source and allows cyanobacteria to overcome N shortages.

Our model simulation suggests that nutrient loads and the concentrations of P in the coastal Baltic Sea have a direct impact on the intensity of the phytoplankton spring bloom and that N is the limiting element during summer. In general, the lagoon, the outlet and the Pomeranian Bay show similar developments of N/P ratios during the year and this pattern does not change during the four investigated decades (Fig. 14). In spring, N is abundant and P is the short element and in summer the situation is opposite. In detail, there are differences between the systems. The P shortage in spring is more pronounced in the Baltic coastal waters compared to the lagoon. In the lagoon, the shortage of N in the water can be more intensive than in coastal waters during summer. The differences in the N/P ratios among the early 1960s, 1980s and the years around 2000 reflect changes in major processes like riverine loads, N fixation, sedimentation and denitrification. In the lagoon, the mid 1980s are characterized by a high availability of nutrients throughout the year. According to the model and monitoring data, there is hardly an N shortage in the coastal waters in the 1980s, not even in summer. This is opposite to the years around 2000 or the early 1960, where the model suggests an N/P ratio close to zero and a lack of N.

During the last decades, improved sewage treatment in the river basin caused a fast decline of riverine P loads, while riverine N loads, resulting mainly from agriculture, remained high. As a consequence, the N/P ratio in river loads is much higher in the years around 2000 compared to the 1960s this has immediate consequences for N/P ratios in the lagoon in spring. The P shortage is stronger and this might have an impact on the phytoplankton spring bloom, at least in some years. The model does not indicate significant changes
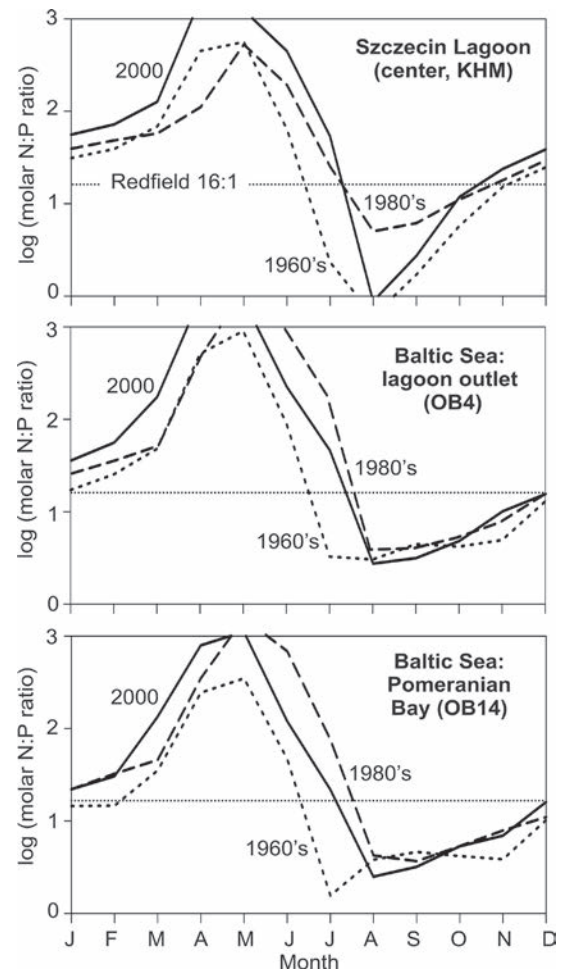

Figure 14. Long-term changes in nitrogen and phosphorus availability (dissolved inorganic components) represented in form of N/P ratios. The data are ERGOM model results averaged over the periods 1961–1964, 1985–1988 and 1999–2002 for three stations in the Szczecin Lagoon (KHM) and the coastal Baltic Sea (OB4, OB14).

in the recycling of P from the sediment. In the coastal Baltic Sea the model suggests a shift of N/P ratios in summer between early 1960s and the years around 2000 (Fig. 14). In the 1960s, the strongest N shortage is observed in July, whereas in 1999-2002 this occurs in August. Nitrogen shortages in the water last shorter in the years around 2000 compared to the 1960s. The most likely

explanation is a higher N transfer in riverine and lagoon water entering the coastal sea in July around the year 2000.

Several processes in coastal waters reduce transport of nutrients to the Baltic Sea and combat thereby eutrophication of the open sea: sedimentation causes nutrient retention and denitrification cause nutrient loss to the atmosphere. The question is whether the nutrient retention/nutrient removal capacity has changed from the beginning of the 1960s to 2002. Since reliable denitrification data are not available, the calculations are based on our model simulations only. Denitrification shows a clear gradient with highest values in the lagoon and lowest values in the coastal Baltic Sea (Fig. 13). The denitrification rate in the model depends on nitrate concentration, oxygen conditions, temperature and availability of organic matter. It includes denitrification in the water column and in the sediment. During the early 1960s and the period 1999–2002, the maximum denitrification rates in the lagoon are estimated with values between 11 and 15 mmol N m^{-2} d^{-1} in June or July. Later in summer, decreasing nitrate concentrations limit denitrification. The total average annual denitrification in the lagoon is estimated to around 13,000 t/a in the 1960s compared to around 15,000 t/a in the years 1999–2002.

These differences are not large and can mainly be explained by different water temperatures and nitrate concentrations. The model suggests that in the early 1960s, about 26% of the total N load into the lagoon is removed by denitrification, compared to only about 15% during 1999–2002. The lagoon's quantitative N retention is stable over the years, but the relative contribution of denitrification to nutrient retention drops. In the coastal Baltic Sea (OB14), the model calculates much lower denitrification rates with a strong interannual variability. The maximum denitrification rate is between 0.8–1.6 mmol m^{-2} d^{-1} (in the 1960s) and

1.1–2.9 mmol m^{-2} d^{-1} (1999–2002). Due to higher nitrate concentrations, the average annual denitrification rate increases from 0.4 (in the early 1960s) to 0.55 mmol N m^{-2} a^{-1} (1999–2002).

Due to its size, the Pomeranian Bay is, and always was, important for N retention and Baltic Sea protection. Neumann (2007) conclude that approximately 50% of the riverine N is denitrified in the Pomeranian Bay and that high denitrification rates in coastal waters are responsible for the strong N/P ratio gradient between coast and sea, with high values near river outlets and low values in the open Baltic Sea.

Discussion and Conclusion

The Odra river basin, the Szczecin lagoon and its coastal waters in the southern Baltic region cover an area of about 150,000 km². To analyze the long-term eutrophication history and functional changes in the entire estuarine system between early 1960s and the years around 2000, we apply a spatial integrative approach, which allows a comprehensive understanding of causes, effects and consequences. For this purpose, the river basin nutrient flux model MONERIS is linked to the 3D hydrodynamic and ecosystem model of the Baltic Sea, ERGOM. Our analyses are based on model simulations and a comprehensive German/Polish monitoring data-set. Although the models are suitable and sufficiently reliable for this analysis and its temporal and spatial resolution, some limitations are obvious.

In some years in July and August, even in this shallow lagoon, hypoxia above the sediment can occur. Hypoxia is limited to short periods between a few days and a fortnight and requires lasting calm weather conditions with wind below 5 m/s. The lagoon sediments are rich in iron phosphate and during hypoxia large amounts of P can be released from the sediments into the water body and cause extreme values of up to 20

mmol/m³ P in the water column within a few days (Schernewski and Wielgat, 2001). Monitoring data show that constantly more than 2 mmol/m³ dissolved P are available in the lagoon during summer (Fig. 10). Therefore, it is available in abundance and additional P does not play a role as a stimulating element for primary production. Further, after P release events from the sediment, a fast precipitation as FeIIIPO4 takes place in the water body and removes P from the water column again. This is indicated by the fast decline of P concentrations in the water after an anoxic release period. In general, the model includes these processes and is able to simulate oxygen depletion in water layers, but it shows low oxygen concentrations only in a few cases and cannot be regarded as sufficiently reliable in this respect. However, this short-coming does not have implications for our model-based analysis, because strong internal eutrophication events take place only in several years, over a short time of days and do not have lasting ecological consequences in the lagoon.

For detailed studies of single years, single processes or the calculation of reliable nutrient budgets, improvements are necessary. In the 3D ecosystem model ERGOM, the vertical and horizontal resolution, the exchange between lagoon and Baltic Sea and the parameterization of several processes have to be improved. Furthermore, quality and spatio-temporal resolution of the input data more and more becomes a determining factor for model performance.

According to our model simulation study, the nutrient concentrations in the river increase from the 1960s until the mid 1980s and decrease afterwards. The estimated annual loads of up to 110,000 t N and 15,000 t P reflect this situation but depend much on a very variable annual discharge. Long-term shifts between point and diffuse emission sources cause shifts in the N/P ratio in the river and this is reflected in processes in the

estuary as well. The availability of N and P as well as N/P ratios changes significantly during the investigated period from early 1960s to 2002s. However, the strong spatial gradients in the estuary remain.

Both the model and observations show that a reduction of riverine nutrient loads quickly reduces the average phytoplankton biomass in the lagoon and in the coastal sea. Despite the relatively high abundance of N in the lagoon, N load reductions affect summer phytoplankton biomass. In the shallow lagoon, riverine loads, interactions with the sediment and mineralization processes ensure a high flux and fast replacement of nutrients. A lasting limitation of primary production due to nutrient depletion cannot be assumed. This is supported by generally low calculated N fixation rates in the lagoon, which are well in agreement with existing literature (e.g. Rahm *et al.*, 2000; Neumann and Schernewski ,2008). Larsson *et al.* (2001) estimated fixation rates of 2.3–5.9 mmol N m^{-2} d^{-1} for the Baltic Sea, which is sufficient to sustain 30–90% of the June–August pelagic net community production. Wasmund *et al.* (2001) obtained similar average daily N fixation rates of 2.5 mmol N m^{-2} d^{-1} for the Baltic proper and the coastal Baltic Mecklenburg Bay in July/August 1997/1998.

Despite very low N concentrations in the lagoon, phytoplankton has no need to largely invest energy in N fixation because the subsequent supply of N is fast and seems to be mostly sufficient. According to the model, N fixation in the lagoon is lowest in the late 1990s. However, the simulations suggest that intensive cyanobacteria blooms could cause rates up to 5 mmol N m^{-2} d^{-1} in single months or up to 30% of the monthly N load into the lagoon. Decreasing riverine nutrient loads are estimated to cause a slightly increasing N fixation, but by far do not compensate riverine load reductions. According to the model, a relatively high N fixation rate in

the coastal Baltic Sea indicates a lasting N limitation during summer.

The estimated denitrification shows strong spatial gradients in the estuary (summer maxima of 15 mmol N m^{-2} d^{-1} in the lagoon and 2.9 mmol N m^{-2} d^{-1} in the coastal sea). According to the model, the N removal rate due to denitrification declines from 26% in the early 1960s to 15% during 1999–2002. Therefore, the relative function of the lagoon as a sink for nutrients decreases. Seitzinger *et al.* (2006) compiled denitrification rates in aquatic systems around the world and received a clear relationship between the N removal rate and water residence time as well as water depth. Compared to these results the removal rates between 15% and 26% of the present study can be considered as typical for a shallow lagoon.

In lakes, usually P is the primary limiting element and P reduction programs were successful in fighting eutrophication during the last decades (e.g. Schindler *et al.*, 2008). In estuaries and coastal waters the situation is different. Here, N is regarded as a primary cause of eutrophication worldwide (Conley *et al.* 2009). Optimal management of coastal eutrophication often suggests controlling both, N and P, because P can interact with the availability of N and silica (Si) in estuaries (Howarth and Marino 2006). In the brackish Baltic Sea we find a special situation. During summer, N is the limiting element in the open Sea. In some Baltic coastal waters and bays P is the element with the least availability and usually is a short element in spring. In the Baltic there has been a long and controversial discussion, whether management should focus on N, P or both (e.g. Elmgren and Larsson, 2001a, b). In 2005, an international committee examined the existing literature for the Swedish waters. In their report Boesch *et al.* (2006) stated that P inputs should be reduced to abate eutrophication and the extensive cyanobacteria blooms in the open Baltic Sea. Reduced N inputs were considered

to be necessary to fight eutrophication in Baltic coastal water, especially at the Swedish west coast. Boesch *et al.* (2006) did not reach a consensus whether reduced N loads are a benefit or a risk for the central Baltic Sea. Recent results by Vathera *et al.* (2007) indicate that coupled processes in the open Baltic Sea induce internal loading, N removal and the prevalence of N fixing cyanobacteria can create a vicious, self-sustaining circle, which maintains the cyanobacteria blooms at least on a medium-term up to decades. To effectively reduce cyanobacteria blooms and overall signs of eutrophication Vathera *et al.* (2007) suggest reductions in both N and P external loads. However, the dispute goes on. Schindler and Hecky (2009) state that 'N abatement is a very expensive shot in the dark that may favor cyanobacteria instead of the water quality.' This view is supported by Bryhn and Håkanson (2008), as well as. Our results show that external nutrient loads are the major divers for water quality in the entire Odra estuary and suggest that N load reductions have positive effects on water quality in the Oder estuary. Carstensen *et al.* (2006) show that this is true for an Danish estuary, as well. Therefore, not all our attention should be given to P and the open Baltic Sea. The inner and outer coastal waters are highly productive and ecologically valuable ecosystems, require protection and will benefit from N load reductions. We support the view of Pearl (2009) who suggests that controls on both nutrients are needed for long-term management of eutrophication along the freshwater–marine continuum. Eutrophication management measures should consider water quality needs and necessities in all aquatic systems, in the river, the inner- and outer coastal waters, as well as in the open Baltic Sea in a balanced manner. Further, the interactions between these systems and their different vulnerability towards pollution have to be taken into account.

It is obvious from other studies (e.g. Kemp

et al. 2005), that long-term changes in nutrient loads cause complex ecological interactions in estuaries. Further, the response of coastal ecosystems to nutrient abatement follows different trajectories than the pollution, and the systems fail to return to their historic state (Duarte *et al.*, 2009). Therefore, our model approach does not give a complete picture on eutrophication induced changes in the investigated ecosystem. It is obvious from data and model results that the abundance and composition of phytoplankton changes during the four investigated decades. However, seasonal, annual and long-term weather conditions and variations are a major driving factor and superimpose long-term developments of nutrient loads. Phytoplankton requires a separate, detailed analysis and shall not be discussed here. Rödiger (2003) and Gosselck and Schabelon (2007) compiled all available historical information about macrozoobenthos and macrophytobenthos in the Szczecin Lagoon. The lower boundary for the growth of macrophytes (mainly reed) nowadays is a water depth of 1 m and large areas with submersed macrophyte species vanished during the last decades. These structural changes had influences on macrozoobenthos communities. Further, the authors assumed that eutrophication caused changes in sediment structure and organic contents. As a consequence, macrozoobenthos biomass increased since the 1970s while species diversity decreased. The original communities, with many snail and mussel species (Radix auricularia, Bithynia tentaculata, Valvata piscinalis, Sphaerium ssp., Pisidium ssp., Unio, Anodonta), are largely lost. Nowadays a few species of tubificids (worms) and chironomid larvae (midges) dominate. However, it is not possible to link these changes to eutrophication history alone. Intensive anthropogenic impacts, such as the deepening of the shipping channel, and salt water penetration from the Baltic Sea are other important factors for observed changes in the lagoon. This is especially true for the fish community. The abundance of different fish species in the lagoon largely depends on fisheries. According to Wysokinski (1998), recent shifts in the fish population are a result of market requirements, changing target fish species and adapted fishing methods after the late 1980s. Decreasing nutrient loads and a recently improved water quality seem to play only a minor role.

Acknowedgements

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