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Reduction of nutrient emission from Polish territory into the Baltic Sea (1988–2014) confronted with real environmental needs and international requirements

by

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Abstract

Eutrophication of the Baltic Sea is considered a major threat to its ecological status. We present and discuss Polish riverine flow normalized loads of total nitrogen (TN) and total phosphorus (TP) discharged into the Baltic Sea in (i) 1988–2014, (ii) periods of maximum TN (1992–1994), TP (1988–1991) emission, (iii) the reference period (1997–2003) established by the Helsinki Commission (HELCOM), (iv) 2012–2014, last years of our study. Despite considerable nutrient load reductions prior to the HELCOM reference period, Poland is expected to reduce TN and TP loads by 30% and 66%, respectively. In the light of our historical and up-to-date findings defining ecological status of the Baltic Sea, we suggest that the proposed TP load reduction is overestimated and its realization may lead to (i) undesirable consequences for the Baltic ecosystem, (ii) would require a decline in TP concentrations to 0.067 mg P dm^{-3} (the Vistula River) and 0.083 mg P dm^{-3} (the Oder River), values reported for pre-industrial times. The current nutrient concentrations in the Vistula and Oder safely comply with the requirements of the Water Framework Directive. We also comment on the top-down and bottom-up effect resulting in quantitative and qualitative reorganization of the Baltic ecosystem, a phenomenon already observed in the Baltic Sea.

Key words: Baltic, Vistula, Oder, nutrient loads, eutrophication, abatement measures

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Introduction

The tremendous progress in all areas of life and at the same time growing threats to the natural environment, are characteristic features of the 20th and the beginning of the 21st century. The Anthropocene is a recent and informal geologic chronological term that serves to mark the evidence and extent of human activities that have had a significant global impact on the Earth's ecosystems, particularly regarding the climate (Meybeck 2001a,b; Crutzen & Stoermer 2000). In the last few decades, human activities encompassing intensive agriculture with excessive use of fertilizers, deforestation, mining, urbanization, industrialization, irrigation, and damming have caused dramatic changes in riverine geochemistry and supply of constituents from rivers to the estuarine and coastal ecosystems (Meyer & Turner 1992; Vitousek et al. 1997; Antrop 2004; Meybeck 2002; 2004; Meybeck & Vörösmarty 2005). An early estimate made by Larsson et al. (1985) shows that starting from the beginning of the 20th century, a 4-fold and an 8-fold increase in total nitrogen (TN) and total phosphorus (TP) loads discharged into the Baltic Sea have been observed. The development of dynamic nutrient models revealed that the TN and TP loadings have instead increased by a factor of 2 and 3, respectively (Savchuk et al. 2008). Håkanson & Bryhn (2008) and Håkanson et al. (2010) presented the first dynamic mass-balance model for phosphorus with a unitary set of calibration constants, which delivered good predictions for all of the major Baltic basins: the Bothnian Sea, the Bothnian Bay, the Gulf of Finland, the Gulf of Riga, and the Baltic Proper. They estimated that the TP loading has increased by about 50% during the last 100 years. Numerous authors report decreasing silicon (Si) fluxes to coastal ecosystems in the 20th century resulting in adverse consequences in ecosystem functioning (Rahm et al. 1996; Humborg et al. 2006; Conley et al. 2008; Danielsson et al. 2008).

Although nitrogen and phosphorus as such do not pose any direct hazard to marine organisms, their excessive inputs may disturb the balance of the ecosystem. Recent decades have witness a massive increase in coastal eutrophication globally, leading to widespread hypoxia and anoxia, habitat degradation, alteration of food-web structure, loss of biodiversity, and increased frequency, spatial extent, and duration of harmful algal blooms (Anderson et al. 2008; Heisler et al. 2008; Howarth 2008; Smith & Schindler 2009). These findings are genuine, but nowadays the qualitative and quantitative changes in ecosystem structure and functioning must also be related to the top-down and bottom-up effects, and the resulting regime shifts (Möllmann et al. 2009; Diekmann & Möllmann 2010; Daskalov 2011; Möllmann 2011). The regime shifts, which have been documented in numerous regions, the Baltic Sea included, may be difficult or impossible to reverse (Duarte et al. 2009; Scheffer 2009), not to mention the required drastic and very costly interventions in ecosystem management (Scheffer et al. 2001; Suding et al. 2004).

Recent studies point to the fact that the term "eutrophication", which must be exclusively related to the process, is very often wrongly understood, and therefore mixed up with the "eutrophic" state. The ecosystem may experience the process of eutrophication but it may still remain oligotrophic or mesotrophic as has been explicitly put forward by Nixon (1995) and Nixon & Fulweiler (2009). This issue is of tremendous importance when it comes to nutrient mitigation measures, which can be overestimated if the state of ecosystem is wrongly evaluated, as for example heavily eutrophic. Costs of implementation of nutrient mitigation measures can be very high, as pointed out by Håkanson & Bryhn (2008), Szoke et al. (2009), and Wulff et al. (2014), particularly when the decision is not preceded by cost-effective studies (Håkanson et al. 2010).

The ecological and economic importance of inland and coastal waters with their goods and services (Botsford et al. 1997), superimposed on the degradation of natural environment, has evoked public awareness of environmental issues. Protection of European waters against degradation has become high on the agenda of the European Commission, and locally on the agenda of the Helsinki Commission (HELCOM) (HELCOM 2007; 2013a; Jadczyszyn & Rutkowska 2012). Since the beginning of the 1990s, the European Union (EU) has implemented various directives to control and reduce nutrient loads into receiving waters: surface waters, transitional and coastal waters, and groundwater. The following EU directives should be listed here: the Nitrate Directive (EEC 1991a), the Urban Waste Water Treatment Directive (EEC 1991b), the Water Framework Directive (WFD) (EC 2000), and the Marine Strategy Framework Directive (EC 2008). HELCOM has introduced the Baltic Sea Action Plan (BSAP), which can be perceived as the implementation of the EU Marine Strategy Framework Directive in this region, and which was adopted by the member states in 2007 and updated in 2013 (HELCOM 2007; 2013a). This initiative encompasses a number of obligations for the Baltic countries, which were to be undertaken not later than 2016 in order to achieve a good ecological status of the Baltic Sea by the year 2021. The Maximum Allowable Inputs (MAI) were calculated in order to meet the previously defined water quality targets of different Baltic sub-basins (HELCOM 2013c). Knowing MAI and loads of TN and TP discharged by each country during the reference period (1997–2003), Country Allocated Reduction Targets (CART) were established (HELCOM 2013a,b).

The overall objective of this study is to take a stand on Poland's feasibility of nutrient loads reduction established by HELCOM (2013a) and scrutinize and critically examine other factors than TN and TP loads, which may contribute to imbalance in the Baltic ecosystem functioning. In order to fulfill this task, we wish: (i) to present the level of reduction of nutrient concentrations and flow normalized nutrient loads at the lowermost monitoring stations on two largest Polish rivers (the Vistula and Oder): (a) in 1988–2014, (b) between the reference period (1997–2003) and 2012–2014, (c) between the maximum TN load (1992–1994) and TP load (1988–1991) and those in 2012–2014; (ii) to collate the calculated target TN, TP concentrations (after implementation of CART reductions) with monitored nutrient concentrations in 1988–2014 and with officially accepted target nutrient concentrations resulting from implementation of the WFD in Poland; (iii) to use the authors' own data to demonstrate that in the light of recent findings and definitions, the Baltic Sea as a whole is not eutrophic and that measures proposed by HELCOM are not adequate to threats; (iv) to comment on the top-down effect, the bottom-up effect (driven by climate change), and the regime shift in the Baltic Sea resulting in quantitative and qualitative reorganization of the Baltic ecosystem, a phenomenon already observed in the Baltic Sea.

Materials and methods

Study area

Almost all of Poland (99.7%) is located within the Baltic Sea drainage basin (Fig. 1). Most of the land belongs to the large drainage basins of the Vistula River (194 424 km^2, with 168 699 km^2 within the Polish borders, constituting ca. 54% of the Polish territory) and the Oder River (118 861 km^2, with 106 056 km^2 within the Polish borders, constituting ca. 34% of the Polish territory). Small, Pomeranian rivers, carrying their waters directly to the Baltic Sea, drain the remaining area (> 11%) (HELCOM 2004). The annual water runoff by the Vistula and the Oder in 1951–1998 ranged from 21.9 to 50.8 km^3 and from 9.5 to 25.8 km^3, respectively. The average water flows in the Vistula and the Oder, estimated for the period of 1951–1990, reached 1081 m^3 s^{-1} and 574 m^3 s^{-1} (Fal et al. 2000).

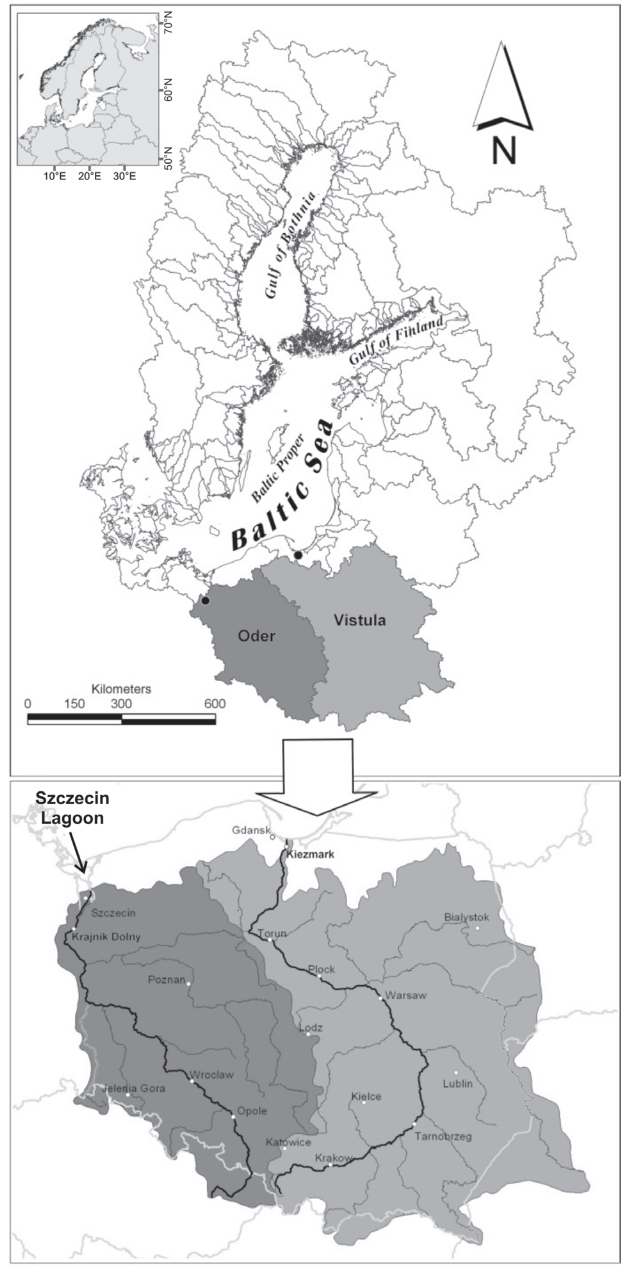

Figure 1

The Baltic Sea catchment and the location of the investigated Vistula and Oder basins; two dots in the upper map indicate the lowermost Oder (Krajnik Dolny) and Vistula (Kiezmark) monitoring stations (source: the upper part of the combined map was produced and kindly made available by Dr. Erik Smedberg from BNI, Stockholm University, Sweden)

In 1975–2012, the annual water discharge from the Polish territory into the Baltic Sea varied from ca. 45 km^3 to ca. 90 km^3. Four wet periods can be distinguished: at the turn of the 1970s and 1980s, in the late 1980s, in 1997–2002, and in 2010–2011. Although all these periods are classified as wet, they differ by 20 km^3 in the volume of water discharged. Very dry periods were identified in 1983–1985, 1989–1993, and 2003–2009, and these were characterized by a drop in riverine water discharge down to 40–50 km^3. The difference in riverine water discharge between dry and wet periods is significant and amounts to ca. 40 km^3; this volume is close to the volume discharged from the Polish territory in the dry period of 1989–1993 or in 2015 (Pastuszak 2012; GUS 1991–2016). A distant location of the Oder lowermost monitoring station from the open Baltic waters (106 km) is conditioned by the fact that the Oder River discharges its waters into the Szczecin Lagoon, the main component of the Oder estuary (Fig. 1).

Poland is one of the largest suppliers of N and P loads to the Baltic Sea (HELCOM 2004; 2011). Because nutrient loads are the result of multiplication of water outflow by nutrient concentrations, it is obvious that one must look into these two parameters and objectively evaluate their magnitude. Among the Baltic countries, Poland has one of the four largest catchments that drain into the Baltic (the other three belong to Sweden, Russia and Finland). The size of the catchment is directly related to water outflow, and Poland is the fourth largest supplier of riverine water to the Baltic Sea (HELCOM 2004; 2011). Nutrient concentrations in Polish rivers are extensively commented in the Results and the Discussion sections. The agricultural land in Poland constitutes over 50% of the overall agricultural land in the entire Baltic catchment and, as shown in the studies carried out in Finland and the USA, nitrogen loss from agriculturally active regions is 8 times (USA) or even 10 times (Finland) higher than from forested areas (Rekolainen et al. 1995; Hatfield & Follett 2008). A recent study, comprising all 117 river basins in the Baltic Sea drainage area, showed that the unit-area loss of nitrogen from agriculture is five times higher than from forest (Stålnacke et al. 2015). The urban area in Poland is the largest one in the Baltic basin, and that in turn is directly related to the population figure; 45% of the entire Baltic basin population inhabits the territory of Poland (HELCOM 2004; Pastuszak 2012).

Data sources and the statistical approach

River monitoring at the lowermost monitoring stations on the Vistula and Oder (Fig. 1) as well as chemical analyses are conducted by the laboratories of the Inspectorate of Environmental Protection in Gdańsk and Szczecin, under auspices of the National Environment Monitoring Program. Both laboratories have Accreditation Certificates (No. AB 177 issued by the Polish Accreditation Center), which meet the requirements of the PN-EN ISO/IEC 17025 standard. The scope, the frequency of sampling (at least once a month), and the analytical methods are regulated by the Water Law, which is also rooted in the EU Water Framework Directive (WFD) 2000/60/EC (Jadczyszyn & Rutkowska 2012). The Journal of Laws of the Republic of Poland (Dz. U. of 5 August 2016, item. 1178) specifies the forms and methods of monitoring the homogeneous surface waters and groundwater (all in compliance with EU regulations), and provides the lists of chemical procedures applied in the determination of nutrients in freshwaters in Poland. The Inspectorate of Environmental Protection in Gdańsk and Szczecin and the Institute of Meteorology and Water Management in Warsaw provided all river monitoring data presented or commented in this paper. The monthly data, covering river nutrient concentrations and water flows, are also reported in monthly bulletins of the Institute of Meteorology and Water Management in Gdynia and Warsaw (IMWM, 1989–2001, 1990–2002, 2003–2015). Aggregated annual nutrient concentrations and loads are submitted by Polish administration to international organizations such as HELCOM.

In this study, we analyzed the following chemical species: total nitrogen (TN), nitrates (NO_3-N), ammonium (NH_4-N), total phosphorus (TP), phosphates (PO_4-P), and Other P (Other P = TP − PO_4-P). For the most part, the sampling frequency for nutrients was twice a month, and we calculated the average monthly values of nutrient concentrations. Nutrient loads were calculated based on the average monthly nutrient concentrations and the average monthly water flows, and in the graphs that follow they are presented as "estimated" loads. Prior to the formal statistical test, a visual inspection of all the time series was performed. Environmental data, such as nutrient concentrations and loads in rivers, often exhibit substantial natural variability caused by the weather conditions at or prior to the sampling occasion, e.g. shown by the temporal variability in water discharge. In the exploratory phase of the analyses, both the Mann-Kendall-test (Hirsch & Slack 1984) and the partial Mann-Kendall test (Libiseller & Grimvall 2002) were used to test the long-term monotonic trends (including linear trends) in annual riverine inputs and concentrations. The latter method has its methodological basis in the Mann-Kendall-test with the difference that explanatory variables can be included. In our investigation, only a partial Mann-Kendall test, using water discharge as an explanatory variable, is presented in order to assess the reasons for the trends. This test also includes a correction for serial correlation up to a user-defined time span; in our case a span of two years (Wahlin & Grimvall 2010). The method also offers convenient handling of missing values. Moreover, a trend line, given as a smoother, was obtained by statistical cross-validation that minimizes the residuals in the statistical modeling, determined according to the method originally developed by Wahlin (2008). This method uses cross-validation to obtain the optimal statistical compromise between a good fit and a smooth function. Confidence intervals for the fitted values are also computed using residual re-sampling (bootstrap). New datasets (bootstrap samples) are generated by adding error terms drawn by sampling with replacement from the observed model residuals. However, this "smoother" should be interpreted with caution; it is included to give a visual picture of the most likely long-term trends in the flow-normalized loads. The software (VBA-macros in Excel) that was used for the statistical significance test and flow-normalization, including the smoother, was downloaded from http://www.ida.liu.se/divisions/stat/research/Software/index.en.shtml.

In our study, in addition to the statistical testing mentioned above, the trend assessment was performed by comparing the calculated loads with the flow-normalized loads, based on a semiparametric regression method by Stålnacke & Grimvall (2001) and further developed and improved by Hussian et al. (2004). This normalization method aims at removing the natural fluctuations in loads and concentrations caused by variability in water discharge or other weather-dependent variables. Simply speaking, the purpose of flow-normalization is to estimate the load at the "normal" water discharge. Such removal or reduction of irrelevant variation in the collected data can help to clarify the impact of human pressures on the environment (in our study, the impact on riverine loads of nutrients). It has also been shown that semiparametric normalization models were almost invariably better than ordinary regression models (Hussian et al. 2004).

Results

Vistula and Oder water flows in the years 1988–2014

The historical data indicate that annual cycles of water flows in the Vistula and Oder showed similar patterns, with maxima during freshets in spring, and minima in summer (Fal et al. 2000). Owing to climate change, this well-established historical pattern has been significantly disturbed over the last decades (Figs 2, 3). The spring maximum in water outflows in the rivers became shifted to February/April as in 2002, 2006, or even to June as in 2013 in the Oder River. The summers of 2001, 2010 (great flood), and 2011 were exceptionally wet in both river basins and water flows were two/three times as high as the seasonal average. In June 2010, the record high flows were measured and they amounted to ca. 3200 m^3 s^{-1} in the Vistula and 1400 m^3 s^{-1} in the Oder. In both basins, the 2010 flood continued until January/February 2011 (Figs 2, 3).

Trends in nutrient loads and concentrations in the Vistula and Oder in 1988–2014

Nitrogen species

A substantial fraction of the inter-annual variation in TN loads was added or removed in both rivers when the load data were flow normalized (Fig. 4), especially in years with very low, as in the 1990s, or extremely high and long lasting water flows, as in 2010 and 2011

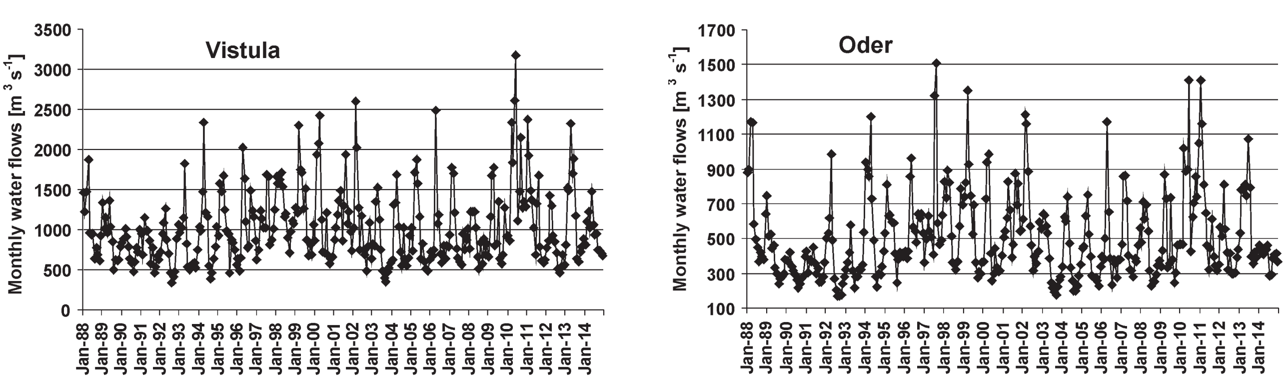

Figure 2

Average monthly water flows in the Vistula and Oder River in 1988–2014 (please note different scales)

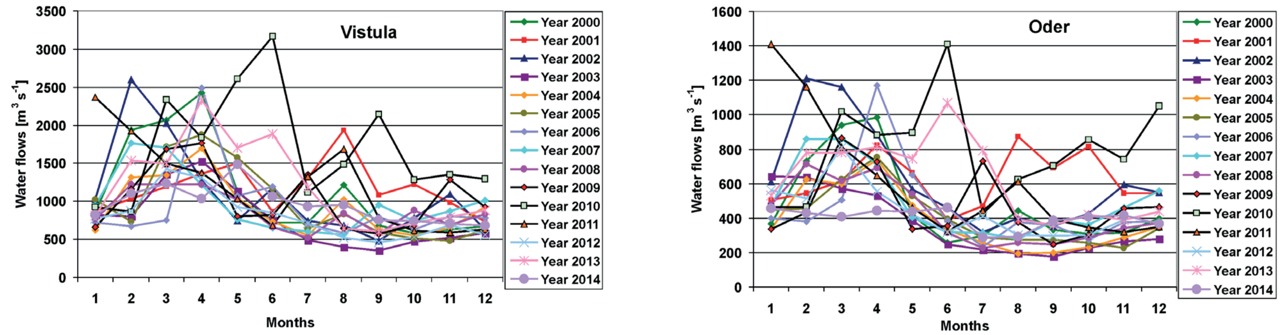

Figure 3

Seasonal variability of water flow in the Vistula and Oder River in 2000–2014 (please note different scales)

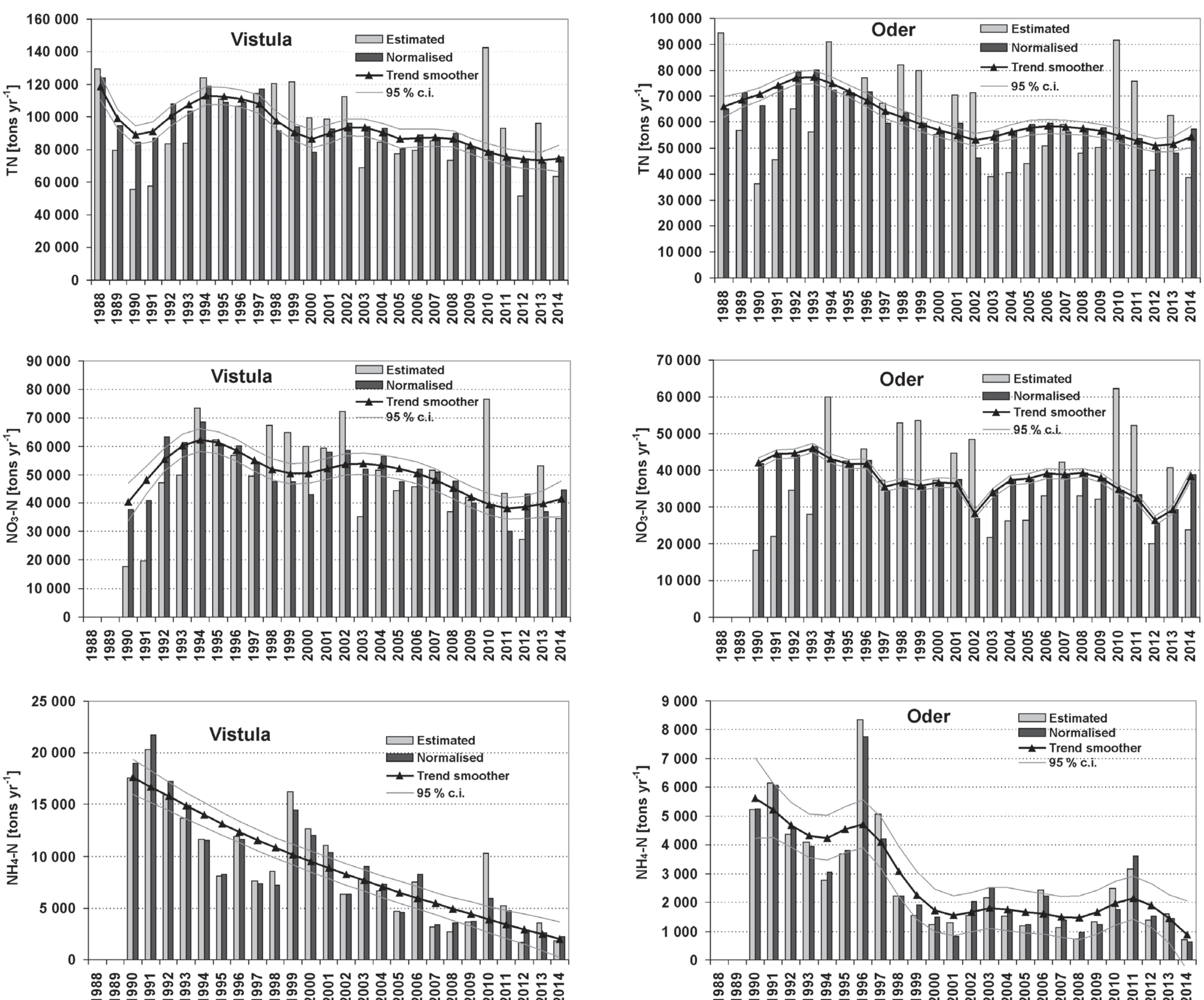

Figure 4

Estimated and flow normalized loads of nitrogen species at the lowermost river monitoring stations on the Vistula and Oder River in 1988–2014 (please note different scales)

(Figs 2, 3). A statistical test of the flow normalized loads and concentrations showed that the decline was statistically significant in the Vistula and Oder ($p < 0.05$). Over the years 1988– 2014, flow normalized TN loads in the Vistula River declined from a maximum of 125 000 tons N yr^{-1} in 1988 to ca. 74 000 tons N yr^{-1} in the last two years of our study. Due to the extensive long-lasting flood in 2010 (Fig. 3), the estimated TN loads discharged by the Vistula and Oder were the highest in the entire period of study (over 140 000 tons N yr^{-1} and over 90 000 tons N yr^{-1}, respectively). Flow normalized NO_3-N loads in the Vistula declined from 68 000 tons N yr^{-1} in 1994 to ca. 40 000 tons N yr^{-1} over the last three years, whereas loads of NH_4-N dropped from over 21 000 tons N yr^{-1} in 1991 to slightly over 2000 tons N yr^{-1} in 2014. The years 1991–1992 were extremely dry and even flow normalized NO_3-N loads were by 30 000 tons N yr^{-1} lower than those in the years that followed (Fig. 4). In the Oder River, flow normalized TN loads dropped from ca. 80 000 tons N yr^{-1} at the beginning of the 1990s to ca. 55 000 tons N yr^{-1} in the last years of observations. Flow normalized NO_3-N loads in the Oder declined from ca. 46 000 tons N yr^{-1} in 1993 to a minimum of 27 000 tons N yr^{-1} in 2002, then increased to ca. 38 000 tons N yr^{-1} in 2007–2009 to decline again to 25 000 tons N yr^{-1} in 2012, and increase again to over 38 000 tons N yr^{-1} in 2014. Loads of NH_4-N in the Oder showed a tenfold decrease (from ca. 6000 to ca. 600 tons N yr^{-1}) over the period of our studies (Fig. 4).

TN concentrations in the Vistula River decreased from 3 mg dm^{-3} in 1988 to 2 mg dm^{-3} in 2014. Nitrate concentrations in the Vistula River showed an increase from ca. 1 mg dm^{-3} to 1.5 mg dm^{-3} in 1998–2003, and that maximum was followed by a decline to ca. 0.9 mg dm^{-3} in 2014. In the Oder River, TN concentrations declined from 4.6 mg dm^{-3} in 1988 to 2.9 mg dm^{-3} in 2014. Nitrate concentrations in the Oder decreased from 2.1 mg dm^{-3} at the beginning of the 1990s to 1.7 mg dm^{-3} in 2014 (Fig. 5).

Phosphorus species

Contrary to the Oder, flow normalized TP loads in the Vistula showed quite substantial interannual variability (Fig. 6) and that affected the trend line showing a decline but incomparably weaker than in the case of the Oder River. TP loads in the Vistula dropped from max 7700 tons P yr^{-1} in the 1993 to ca. 5000 tons P yr^{-1} in 2013–2014, whereas in the Oder

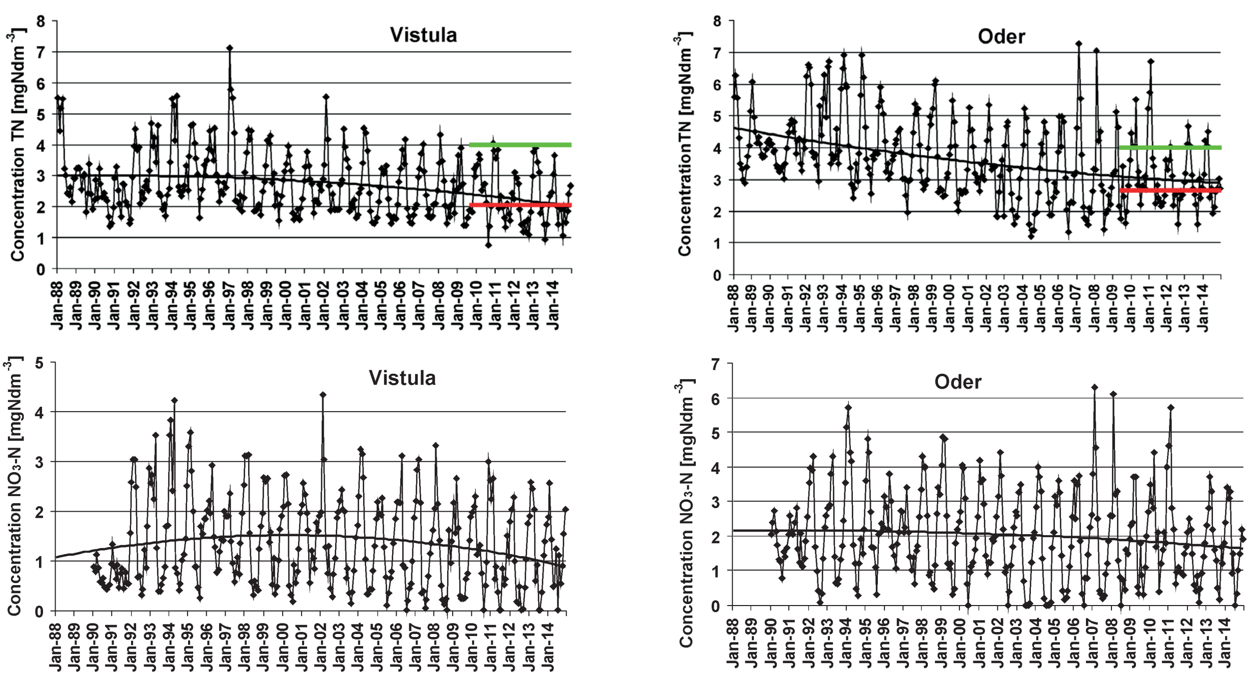

Figure 5

Concentrations of nitrogen species at the lowermost river monitoring stations on the Vistula and Oder River in 1988–2014; red lines in the TN graph indicate the calculated target concentrations with adopted HELCOM load reduction; green lines indicate target concentrations established for lowland large rivers (type 21; good ecological status acc. to the WFD) (see the Discussion; please note different scales)

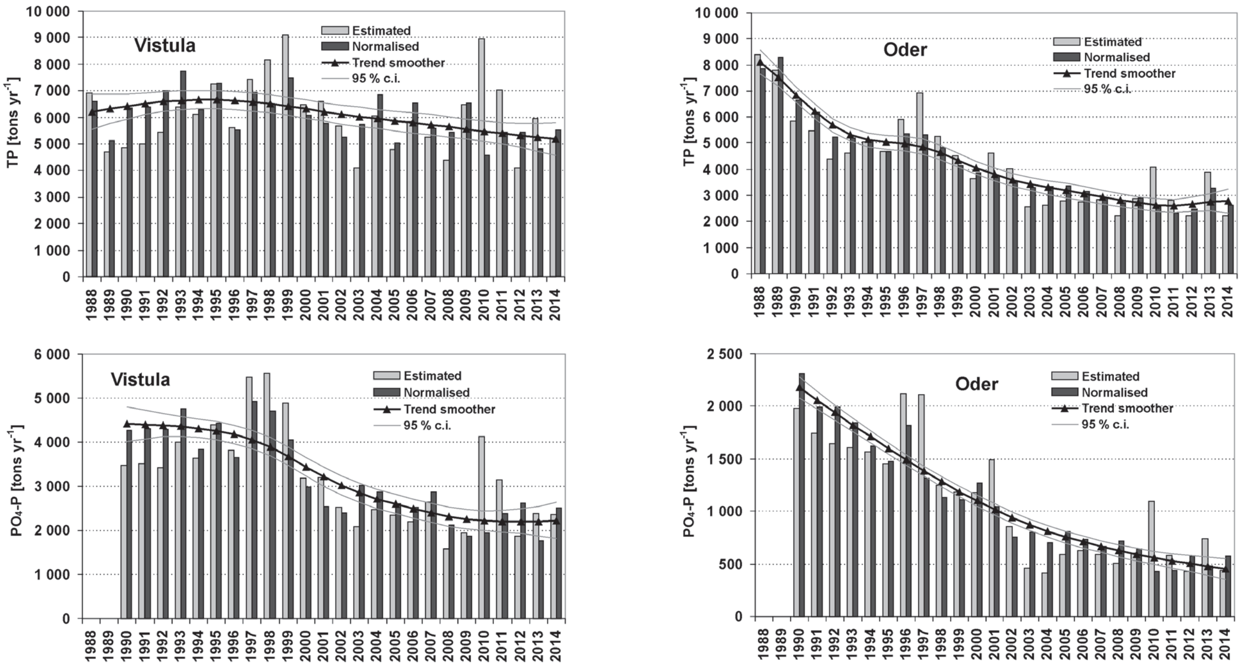

Figure 6

Estimated and flow normalized loads of phosphorus species at the lowermost river monitoring stations on the Vistula and Oder River in 1988–2014 (please note different scales)

– from ca. 8300 tons P yr^{-1} in 1989 to ca. 2600 tons P yr^{-1} in 2014. The TP load time-trend was statistically significant ($p < 0.05$) in both rivers (Fig. 6).

PO_4-P loads in the Vistula dropped from 4000–4900 tons P yr^{-1} in the 1990s to ca. 2000 tons P yr^{-1} in 2013–2014, whereas in the Oder – from 2000 tons P yr^{-1} at the beginning of the 1990s to ca. 500 tons P yr^{-1} over the last three years. The PO_4-P load patterns were quite different in the rivers studied, with a decreasing tendency observed the Vistula River in 1996 and a steep decline in loads in the Oder River during the whole period studied. The PO_4-P load time-trend was statistically significant ($p < 0.05$) in both rivers however, the statistical significance values were stronger in the Oder River ($p < 0.0001$) (Fig. 6). Due to the long-lasting flood in 2010 (Fig. 3), the estimated TP and PO_4-P loads discharged by the Vistula were very high, reaching ca. 9000 tons P yr^{-1}, and over 4000 tons P yr^{-1}, respectively.

TP concentrations in the Vistula declined from 0.22 mg dm^{-3} to 0.16 mg dm^{-3}, while in the Oder – from ca. 0.55 mg dm^{-3} in 1988 to ca. 0.17 mg dm^{-3} in 2008–2011, and slightly increased to 0.19 mg dm^{-3} in 2014. PO_4-P concentrations in the Vistula decreased from ca. 0.17 mg dm^{-3} in 1990 to 0.07 mg dm^{-3} in 2012–2014. In the Oder, PO_4-P concentrations dropped from ca. 0.18 mg dm^{-3} in 1990 to 0.03 mg dm^{-3} in 2008–2012, and then slightly increased to 0.04 mg dm^{-3} in 2014 (Fig. 7).

Combined concentrations of PO_4-P and Other P (Other P = TP – PO_4-P) in the Vistula and Oder (Fig. 8) lead to the following findings (i) the contribution of the Other P fraction to TP concentration was higher in the Oder River as compared to that in the Vistula River, particularly up to the year 2004; (ii) concentrations of the Other P fraction were higher in the Oder River throughout the study period; (iii) there was a three-fold decrease in concentrations of the Other P fraction in the Oder River; (iv) there was a significant decline in PO_4-P concentrations in both rivers.

Discussion

There are three sources of nutrient losses into surface inland waters in the Baltic catchment: (i) losses from diffuse sources, with agriculture playing a key role (N – 59%; P – 53%), (ii) discharges from point sources (N – 10%; P – 20%), and natural background losses (N – 31%; P – 27%) (HELCOM 2004). Hydrological water flows provide the medium for transport and biogeochemical processing, via surface runoff or flow through aquifers, streams, lakes, reservoirs and wetlands (Bouwman et al. 2013). Studies carried out by numerous researchers have shown that nutrient concentrations in water discharge from river basins are the result of several interacting processes, including

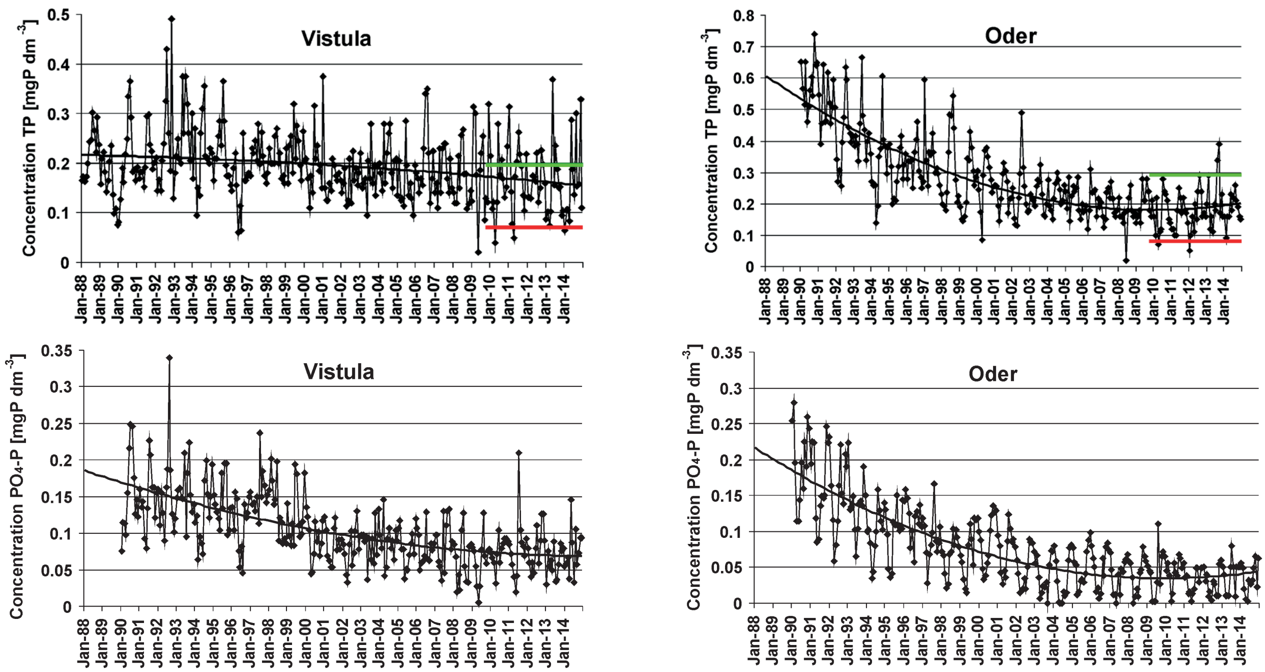

Figure 7

Concentrations of phosphorus species at the lowermost river monitoring stations on the Vistula and Oder River in 1988–2014; red lines in the TP graph indicate the calculated target concentrations with adopted HELCOM load reduction; green lines indicate target concentrations established for lowland large rivers (type 21; good ecological status acc. to the WFD) (see the Discussion; please note different scales)

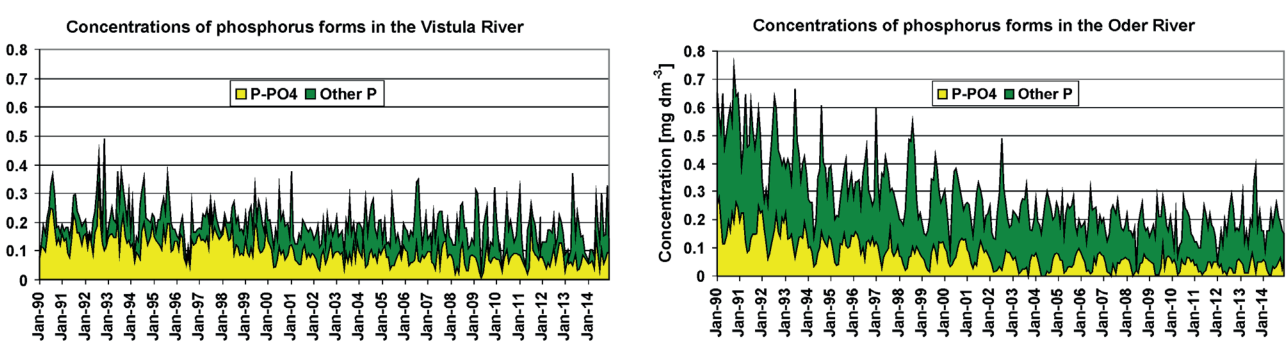

Figure 8

Concentrations of the phosphorus species at the lowermost river monitoring stations on the Vistula and Oder River in 1990–2014

exchange between cycles in the terrestrial, aquatic, and atmospheric environment (Haycock et al. 1993; Nixon et al. 1996; Behrendt et al. 1999; 2005; 2008; Arheimer & Lidén 2000; Mander et al. 2000; Kronvang et al. 1999; Stålnacke et al. 2003; Vagstad et al. 2004; Grizzetti et al. 2005; Pastuszak et al. 2005; Lepistö et al. 2006; Voss et al. 2005; 2006; Moreno et al. 2007). These processes can be categorized into: (i) nutrient release e.g. through mineralization, weathering, fertilization, atmospheric deposition, sewage effluents; the release must be referred to the type of crops, techniques of soil cultivation, intensity of fertilization (related to increased N and P surplus and possible P accumulation in soil), types of soils and their permeability; (ii) water transport (transit time and flow paths, with geomorphology playing a very important role); (iii) transformation and immobilization e.g. denitrification, sedimentation and adsorption taking place in riverine systems, estuaries, and lakes. The influence of different catchment processes varies spatially and temporarily, because they may be favored by conditions in basins such as: land use, land cover, physiographic features, management, meteorology, and hydrology. The pool of nutrients that enters the surface waters largely depends on N and P pathways from the soil surface and the root zone to the surface recipient of water (Behrendt et al. 2005; 2008; Grizzetti et al. 2005).

Models are useful assessment tools for quantification of pollution pressures by nutrients (De Wit 2000) and they are essential to improve our understanding of the interactions between multiple processes in different landscape elements in river basins and to better predict the transfer of nutrients from land to sea (Bouwman et al. 2013). Modeling studies (model MONERIS) have been carried out in the Vistula (Kowalkowski & Buszewski 2006; Kowalkowski 2009) and in the Oder basin (Behrendt et al. 2005; 2008; Kowalkowski et al. 2012; Gadegast et al. 2012; Pastuszak et al. 2014). They not only made it possible to estimate the overall annual N and P emission into the river basins, but also the changing relative percentage contribution of e.g. WWTPs pathway, considerably declining over the last 20 years, and consequently causing a relative increase in the percentage contribution of agriculture-related pathways.

Poland's efforts during the transition period to reduce pressures on natural environment

Since 1989, i.e. the beginning of the transition period, Poland's economy has undergone a profound transformation from a centrally planned economy to a market economy. This transition has led to major institutional and economic structural changes over the last years. Reforms initiated in the early 1990s provided opportunities to revise and implement environmental management and legislation, and to achieve greater integration of environmental concerns into other policies (Jadczyszyn & Rutkowska 2012). Polish agriculture has benefited from the financial assistance from the EU funds, and the main objectives of direct agricultural assistance from public sources included e.g. modernization of farms, compliance with environmental protection regulations, as well as standards of good agricultural practice (Kata & Miś 2006). In connection with the accession of Poland to the EU in 2004, the EU has been an actively influential agent in the Polish transition through a variety of mechanisms. Foremost amongst these has been a variety of multilateral and bilateral aid and loans programs implemented since 1990 (Ners 1996; Steves 2001).

Since the early 1990s, Poland has made remarkable environmental progress, meeting most of its environmental targets and decoupling a number of environmental pressures from economic growth. Together with economic structural changes, these efforts have contributed to progress on a number of fronts: (i) reductions in air pollutant emissions; in 2014, Poland achieved a reduction of 34% in the emission of greenhouse gases (expressed as a carbon dioxide equivalent) in relation to the base year, including in particular the emission of carbon dioxide by 35%, methane by 46%, and nitrous oxide by 32% (Błaś et al. 2008; GUS 1991–2016); (ii) declines in water withdrawal, and economical water utilization, which resulted in a two-fold decrease in the volume of industrial and municipal sewage (GUS 1991–2016); (iii) a ten-fold decrease in the volume of untreated sewage discharged into soil and surface waters in 1970–2014 (Pastuszak & Witek 2012; Pastuszak et al. 2012a,b; GUS 2015); (iv) since 1995, continuous construction of new tertiary waste water treatment plants (WWTPs); at present, nearly half of the sewage volume is treated with this technology (Pastuszak & Witek 2012; Pastuszak et al. 2012a,b; GUS 2015); (v) construction of 2580 WWTPs only between the years 1999 and 2014 and a considerable decline in N, P loads discharged from this source (Pastuszak & Witek 2012; Kowalkowski et al. 2012; Pastuszak et al. 2012a,b; GUS 1991–2016); in 2014, 93.9% of the urban population and 37.4% of the rural population was connected to WWTPs (GUS 2015); (vi) reduction of N, P emission from dispersed farmyards through construction of domestic wastewater treatment plants, serving up to 50 inhabitants; up to the year 2010, 82 632 domestic sewage treatments were constructed in Poland and it is anticipated that 587 000 new ones were/will be constructed in the years that follow (Przydomowe oczyszczalnie 2012); these plants are not covered by the official statistics; (vii) closure of obsolete factories and modernization of the remaining ones, elimination of "hot spots" (HELCOM 2009a); (viii) immense improvement of infrastructure in farming (Lipiński 2012; Pietrzak 2012); (ix) considerable decline in phosphorus surplus in the agricultural sector (Igras & Fotyma 2012); (x) the above-mentioned undertakings and actions resulted in a significant decline in N, P emission to the Vistula and Oder basins (Kowalkowski et al. 2012; Pastuszak et al. 2012a), and consequently in (xi) a considerable, statistically significant decline in nutrient concentrations and loads in the Vistula, the Oder, and the Pomeranian rivers supplying the Baltic Sea (Jarosiewicz 2009; Pastuszak & Witek 2012; Pastuszak et al. 2012a,b; Figs 4–8); nearly three-fold decline of $P\text{-}PO_4$ and Other P concentrations in the Oder River (Fig. 8) points to a decline in P emission not only from point sources but also from diffuse sources, the latter undoubtedly resulting from implementation of legal regulations of environmental protection in agricultural activities (Król 2015); (xii) tremendous improvement of water quality in the rivers studied, with the average Biological Oxygen Demand (BOD_5) dropping from 5.8 mg dm^{-3} to 2 mg dm^{-3} in the Vistula, and from 6.3 mg dm^{-3} to 3.5 mg dm^{-3} in the Oder (IMWM 1989–2001; 1990–2002; 2003–2015).

There are evident differences between nutrient concentrations and loads and their dynamics in the Vistula and Oder (Figs 4–8). Spatially differentiated natural factors, and spatially and temporally differentiated anthropogenic factors have been identified and found to be responsible for spatial and temporal variability in nutrient concentrations and loads carried by the largest Polish rivers. These issues have been extensively discussed in earlier publications (Kowalkowski et al. 2012; Pastuszak et al. 2012a,b; Pastuszak & Witek 2012; Pastuszak et al. 2014).

Reduction of nutrient loads discharged into the Baltic Sea – HELCOM vs. Håkanson et al. (2010) strategy

Eutrophication is perceived as one of the main threats to the Baltic Sea ecosystem health (HELCOM 2009b) and that made HELCOM introduce the Baltic Sea Action Plan (BSAP) (HELCOM 2013a), which is focused on the reduction of TN and TP loads (Table 1), and which was verified by using up-to-date knowledge and modeling approaches (Savchuk & Wulff 2007). New water quality targets of different Baltic Sea sub-basins have been developed. The level of load reduction proposed by HELCOM (Table 1) is referred to average flow normalized loads in the period of 1997–2003 (HELCOM 2013a,b,c). HELCOM modified the overall reduction of TN and TP loads over the period of 2007–2013 (HELCOM 2010, 2013a,b), with TN load dropping from 133 170 tons N yr^{-1} in 2007 to 89 260 tons N yr^{-1} in 2013, and TP load dropping from 15 016 tons P yr^{-1} in 2007 to 14 374 tons P yr^{-1} in 2013. The difference in TP load reduction in 2007 and 2013 is rather cosmetic (decrease by 4%), hence the conclusions from the studies by Håkanson and Bryhn (2008) and Håkanson et al. (2010) still hold.

The HELCOM (2013a,b) approach has been challenged by Håkanson and Bryhn (2008) and Håkanson et al. (2010) who developed a mass-balance model framework called CoastMab. One of the decisive advantages of the CoastMab model is that it has a unitary set of constants and algorithms for all sub-basins of the Baltic Sea, which indicates a robust model structure. Other model approaches either need sub-basin-specific calibration, or have poor resemblance between model data and empirical data in some sub-basins, indicating that some generic processes may not have been adequately captured. The outcome of the CoastMab approach is substantially different compared to the HELCOM strategy, with no TN load reduction recommended, and with the overall TP load reduction on the level of 8730 tons P yr^{-1} (Table 1), treated regionally and necessarily preceded by cost-effectiveness studies. Håkanson et al. (2010) give numerous arguments speaking for exclusive and much lower TP load reduction, and a couple of them should be listed here: (i) by lowering the nitrogen concentrations, the Baltic Sea is likely to favor additional blooming of harmful cyanobacteria; occasional high concentrations of cyanobacteria in the Baltic Sea may be quantitatively explained by high TP concentrations, high temperatures (higher that 15°C) and/or low TN:TP

Table 1

Country allocation of nutrient load reduction acc. to Country Allocated Reduction Target (CART) (HELCOM 2013a,b) and regional allocation of TP load reduction acc. to Håkanson et al. (2010)

Country	HELCOM (2013a,b) approach				Håkanson et al. (2010) approach		
	CART TN [tons yr^{-1}]	CART TP [tons yr^{-1}]	Percentage contribution		Region	TP [tons yr^{-1}]	TN [tons yr^{-1}]
			TN	TP			
Denmark	2890	38	3.24	0.26	Bothnian Sea	Not needed	Reduction not recommended by the authors
Germany	7670	170	8.59	1.18			
Poland	43 610	7480	48.86	52.04	Bothnian Bay	Not needed	
Lithuania	8970	1470	10.05	10.23			
Latvia	1670	220	1.87	1.53	Gulf of Finland	3180	
Estonia	1800	320	2.02	2.23			
Russia	10 380	3790	11.63	26.36	Gulf of Riga	550	
Finland	3030	356	3.39	2.48			
Sweden	9240	530	10.35	3.69	Baltic Proper	5000	
Sum	**89 260**	**14 374**	**100.00**	**100.00**		**8730**	

ratios (lower than 15 by weight; see also Schindler et al. 2008; 2016); (ii) the reduction of TP loads proposed by HELCOM (ca. 15 000 tons P yr^{-1}) will lead not only to lowering of primary production, but also secondary production, including zooplankton and fish, also fish with high commercial value; the total fish production would be much lower than today (see also Thurow 1997; Österblom et al. 2007), whereas the concentration of organic toxins, such as PCBs, dioxins etc. in fish, would likely be higher if nutrient concentrations decrease substantially (see also Gunnarsson et al. 2000; Skei et al. 2000).

Reduction of nutrient loads from Poland's perspective

The maximum of N and P emissions to the Baltic Sea took place in the 1980s, and these were followed by very steep load declines (Gustafsson et al. 2012). The HELCOM reference period (1997–2003) is positioned on these steep slopes. In 1997–2003, Polish TN and TP loads discharged into the Baltic Sea amounted to 169 648 tons N yr^{-1} and 11 548 tons P yr^{-1}, which means that by 2012–2014 Poland had reduced TN loads by 29 276 tons N yr^{-1} and TP loads by 2558 tons P yr^{-1} (Table 2; yellow columns). If Poland is to comply with HELCOM requirements, further reduction of TN load by 14 334 tons N yr^{-1} and TP load by 4922 tons P yr^{-1}, would be needed (Table 2; yellow and red columns). A significant nutrient load reduction in Poland took place before the HELCOM reference period (Figs 4 and 6). The maximum TN load discharged by Poland reached 209 243 tons N yr^{-1} in 1992–1994, whereas the maximum TP load reached 14 903 tons P yr^{-1} in 1988–1991. That means that by 1997–2003, TN and TP loads discharged by Polish rivers had declined: TN – by 39 595 tons N yr^{-1} (68 871 – 29 276 = 39 595 tons N yr^{-1}) and TP – by 3355 tons P yr^{-1} (5913 – 2558 = 3355 tons P yr^{-1}) (Table 2). It is obvious that these declines have not been accounted for in Poland's nutrient input budget in the HELCOM strategy. The reduction level of nutrient loads should be referred to the highest loadings of TN and TP fueling phytoplankton growth, thus the eutrophication process that HELCOM is trying to combat. The modeling studies carried out in the Oder estuary by Friedland et al. (submitted for publication) point to unfeasibility of the phosphorus load reduction in the Oder basin proposed by HELCOM. According to the authors, TP loads and concentrations in the Oder would have to reach values close to those in pre-industrial times, characterized by completely different agricultural activity and intensity, and a much smaller population.

Hereafter, we present TN and TP target concentration if Poland would follow the HELCOM strategy. TN and TP loads from the Polish territory are predominantly discharged by the Vistula and Oder, with the contribution of small rivers directly supplying the Baltic Sea constituting 11.6% and 11.4% of the added up TN and TP carried by the two largest rivers (Table 3). The contribution of the Vistula, the Oder, and the Pomeranian rivers to the overall TN, TP loads discharged from the Polish territory into the Baltic Sea, was used to proportionally split the Country Allocated Reduction Target (CART) (Table 1), and then subtract the obtained values from the average TN and TP flow normalized loads calculated for the HELCOM reference period of 1997–2003 (the Vistula and Oder) (Table 3). In order to estimate the approximate target concentrations in the two large Polish rivers, the obtained numbers were divided by the annual long-term Vistula and Oder water outflows i.e. 34 km^3 and 16.7 km^3, respectively. The obtained target CART concentrations, as well as the maximum allowable nutrient concentrations, established for Polish rivers in accordance with WFD, and meeting the good ecological status (Garcia et al. 2012), are presented in Table 3 and plotted in Figs 5 and 7 (red and green lines).

Table 2

Combined flow normalized TN and TP loads discharged by the Vistula, the Oder, and the Pomeranian rivers in the reference period (1997–2003) and in 2012–2014, and the difference in loads in these two periods (yellow columns); maximum loads observed in 1992–1994 (TN) and 1988–1991 (TP), and the difference between maximum loads and those in 2012–2014 (green columns); allocation of load reduction acc. to BSAP (HELCOM 2013b) (red column)

Nitrogen, phosphorus loads	In 1997–2003	In 2012–2014	Difference between 1997–2003 and 2012–2014	Maximal observed (years in brackets)	Difference between max. observed and 2012–2014	BSAP Poland
	tons yr^{-1}					
TN	169 648	140 372	**29 276**	209 243 (1992–1994)	**68 871**	**43 610**
TP	11 548	8990	**2558**	14 903 (1988–1991)	**5913**	**7480**

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Table 3

Average combined flow normalized TN and TP loads, calculated for the reference period (1997–2003) for the Vistula and the Oder, and approximated for Polish rivers directly feeding the Baltic Sea, and target TN and TP concentrations at CART assumptions (Table 2); concentrations in brackets are officially accepted maximum allowable values, meeting the good ecological status specified in WFD (Garcia et al. 2012)

Average flow normalized TN loads	Average flow normalized TP loads	Target CART and max allowable **WFD** concentrations TN	Target CART and max allowable **WFD** concentrations TP	Target CART and max allowable **WFD** concentrations TN	Target CART and max allowable **WFD** concentrations TP
Vistula + Oder + other rivers		Vistula		Oder	
tons N yr^{-1}	tons P yr^{-1}	mg dm^{-3}			
94 794 + 57 220 + 17 634 = 169 648	6250 + 4116 + 1182 = 11 548	2.06*) **(4.00)**	0.067 **(0.29)**	2.57*) **(4.00)**	0.083 **(0.29)**

*) target TN concentrations could be slightly lower if we took into account the reduction of direct atmospheric deposition to the Baltic Sea by Poland; such data are not available for our periods of calculations

The current TN concentrations in the Vistula (2 mg N dm^{-3}) are very close to the calculated target concentration with CART assumptions, whereas in the Oder (3 mg N dm^{-3}) they are higher than target CART values by ca. 0.4 mg N dm^{-3} (Fig. 5, Table 3). In order to achieve the HELCOM level, TN loads would have to be reduced by ca. 10 000 tons N yr^{-1} (at an assumed retention coefficient at the level of. 0.3), provided the very important role of the Oder estuary in natural nutrient retention (Pastuszak et al. 2005) is not taken into account. This is an approximate number, because the nutrient retention in rivers strongly depends on water flows (Pastuszak et al. 2014). It is worth noting that nitrate concentrations in Polish rivers (a key contributor to TN in agriculturally active catchments) (Fig. 5), have been very low as compared to corresponding values in eutrophic Western European rivers (5–8 mg N dm^{-3} in the Thames, the Rhine, the Elbe and the Seine) (Meybeck 2001a,b; Bouraoui & Grizzetti 2011; OECD 2008) or in the Humber catchment in north-western England (3.3–18.8 mg N dm^{-3}) (Neal et al. 2008).

Target TP concentrations reached 0.067 mg P dm^{-3} in the Vistula and 0.083 mg P dm^{-3} in the Oder (Table 3) and are well below the current values (0.16 and 0.19 mg P dm^{-3}) (Fig. 7, Table 3). For comparison, medians of phosphates concentrations (constituting part of TP concentrations) in Germany (89 rivers), Denmark (30 rivers), France (254 rivers), Great Britain (89 rivers) ranged in 1994–2010 from 0.06 mg dm^{-3} to 0.10 mg dm^{-3}; medians of TP concentrations, for the same years and settings ranged from 0.16 mg dm^{-3} to 0.26 mg dm^{-3} (European Environment Agency 2014). Average phosphate concentrations in the Humber catchment in north-western England were in the range of 0.023–1.96 mg P dm^{-3} (Neal et al. 2008). TP concentrations in large rivers in 25 European countries varied from a minimum of 0.007 mg dm^{-3} to a maximum of 0.200 mg dm^{-3} (Schöll et al. 2012).

Following the WFD requirements, Garcia et al. (2012) specified the maximum allowable concentrations of TN and TP in all Polish rivers, the Vistula and Oder (type 21 acc. to WFD) included. These concentrations for large lowland Polish rivers, meeting the criterion of good ecological status, are as follows: TN – 4.00 mg dm^{-3}, TP – 0.29 mg dm^{-3} (Table 3, numbers in bold; Figs 5, 7 – green lines). In 2016, these values were officially approved by Polish Parliament[1]. Schöll et al. (2012) provide the maximum acceptable TP concentration in large lowland rivers under least disturbed conditions, and this concentration amounts to 0.20 mg P dm^{-3}, thus the value close to that reported by Garcia et al. (2012).

Poland has already fulfilled the requirements of WFD with a great safety margin, but has and will have serious problems to meet the HELCOM requirements, particularly those concerning TP loads (Figs 5, 7). It is simply impossible to reduce TP concentrations to the values found in pre-industrial times, particularly in the country with 38 million inhabitants and with a large agricultural area, like Poland.

There are additional Polish arguments indicating that the HELCOM allocation of nutrient load reduction requires revision before the Baltic community is criticized by the EU for not having satisfactorily combated eutrophication, and this has already taken place (EU 2016). In the HELCOM approach (Table 1), Poland is responsible for ca. 49% and 52% of the overall reduction of TN and TP loads discharged into the Baltic Sea and these numbers strongly deviate from the Polish contribution to TN and TP emission to the Baltic Sea. TN and TP loads discharged by Polish rivers, as well as direct point sources constituted 26% and 37% in 2000, and 23.9% and 36.1% in 2006 of the overall riverine TN and TP loads reaching the Baltic

[1]Regulation of the Minister of the Environment (Dz.U. item 1187, 05 August 2016) on the methods of classification of homogeneous surface waters and on the environmental quality standards for priority substances

Sea (HELCOM 2004; 2011). Poland's contribution to TN and TP loading (actual estimates) in 2010 reached the values of 36% and 41%, whereas the contribution to flow normalized loads reached the values of 29% (N) and 33% (P) (HELCOM 2015). This indicates the importance of the flow normalization procedure, extensively adopted in the Baltic basin (Grimvall & Stålnacke 1996; Laznik et al. 1999; Stålnacke & Grimvall 2001; Stålnacke et al. 1999a,b; 2003; 2004; Grimvall et al. 2000; Hussian et al. 2004; Wahlin & Grimvall 2008; Pastuszak & Witek 2012; Pastuszak et al. 2012 b; Figs 4 and 6), but also the necessity of adequate data reporting by HELCOM, with graphs showing not only actual but also flow normalized loads, and most preferably with data not only related to the great flood in 2010 (HELCOM 2015), but also to droughts e.g. in 2015. It cannot be denied that large loads of TN and TP were discharged by Polish rivers in 2010 and supplied the Baltic ecosystem, but it cannot be denied either that very low loads of nutrients were discharged from Polish territory to the Baltic Sea in e.g. 2015, marked by severe drought (Somorowska 2016; Laaha et al. 2016; GUS 1991–2016). Poland experienced the catastrophic flood in 1997 and immediate studies carried out in the Pomeranian Bay as well as in the Gulf of Gdańsk showed that the large river discharge did not cause extreme nutrient transports into the Pomeranian Bay (Mohrholz et al. 1998), again indicating the importance of the Oder estuary in natural nutrient retention, even under flooding conditions. In both regions, pollutants accumulated within the coastal zones and did not spread beyond the bays borders into the open southern Baltic Sea (Trzosińska & Andrulewicz 1998).

In the applied MARE NEST model, and the subsequent model SANBALTS, HELCOM divided the whole Baltic into sub-regions, one of them comprising the Oder estuary, the Pomeranian Bay, and the Bornholm Basin (Savchuk et al. 2012; HELCOM 2013a,b). While passing through estuaries, the allochthonous dissolved and suspended matter, undergoes numerous biogeochemical processes leading to its substantial qualitative and quantitative transformation (Boynton et al. 1995; Howarth et al. 1995; 1996; Nixon et al. 1995; 1996; Pastuszak et al. 2005). The extent and intensity of these processes in estuaries cannot be compared with corresponding processes taking place in open deep waters. The HELCOM approach is tantamount to the fact that 45% of the TN load (ca. 37 000 tons N yr^{-1}), and 37% of the TP load (ca. 2200 tons P yr^{-1}) retained in the Oder estuary (Pastuszak et al. 2005) is not subtracted from the adequate loads carried by the Oder River, worsening not only Poland's nutrient input budget, but also making the reduction demands unrealistic, as stated in modelling studies performed in the Oder estuary by Friedland et al. (submitted for publication). Interestingly, if we take into account the role of the Oder estuary in TP retention (Pastuszak et al. 2005), it is obvious that the current Oder TP loads reaching the open Baltic waters, are below 1000 tons P yr^{-1} (Fig. 6).

Nixon et al. (1996) estimated that estuarine processes at the land-sea margin of the North Atlantic Ocean retain and remove 30–65% of the TN and 10–55% of the TP that otherwise would pass into the coastal ocean. The percentages of TN and TP retention in the Oder estuary (Pastuszak et al. 2005) are within the range of values reported by Nixon et al. (1996). Studies of Howarth et al. (1996) show that the amount of N retained in an estuary is relatively insensitive to trophic status; in contrast, the percentage accumulation of phosphorus in sediments decreases as an estuary becomes more eutrophic. When referring these findings to Polish studies (Pastuszak et al. 2005), it can be concluded that during the transition period N retention in the large Oder estuary remained unchanged, while P retention could have even increased due to a significant reduction in P emission into the Oder basin and consequently, a significant reduction in P loads introduced to the estuarine system (Pastuszak et al. 2012a,b).

Are the HELCOM measures adequately addressed?

The HELCOM load reduction is associated with high costs, reaching 4.7 10^9 Euro annually; as much as 50% of this cost is incurred by Poland (Wulff et al. 2014). Estimates of Håkanson & Bryhn (2008) and Håkanson et al. (2010) show that in the "optimal" scenario, i.e. when 8730 tons yr^{-1} of phosphorus and no nitrogen is reduced, and provided the most cost-effective measures are applied, the annual costs would be about 367 million Euro.

The assessment of the trophic state of any water reservoir is accompanied by controversy, resulting from the operational definition of "eutrophication" (a noun in English language), the term that is still used in fuzzy and often confusing ways by scientists and managers (Nixon & Fulweiler 2009). Eutrophication should be properly understood as a process, a change in the rate of supply of organic carbon and energy to an ecosystem, but not as the trophic state (Nixon 1995). Eutrophication indicators, such Secchi depth, chlorophyll *a* concentrations, TN, TP concentrations (Håkanson & Bryhn 2008), and/or organic carbon supply to the system (Nixon 1995), have been adopted for different trophic states i.e. oligotrophic, mesotrophic, eutrophic, hypertrophic (Table 4). Nixon & Fulweiler (2009) emphasize that some confusion arises from the fact that ecologists use the term "eutrophic" to characterize systems that have high primary production (the rate of carbon fixation or formation of new organic matter). There is a possibility that some marine waters may always have been eutrophic, including upwelling areas off the coast of Peru and parts of Africa. Many others have become eutrophic because of eutrophication brought on by human actions. For example, some parts of the Baltic may be undergoing eutrophication as their primary production increases from 20 to 40 g C m^{-2} yr^{-1}, but they are not eutrophic. By the same token, an estuary with relatively stable average production of 350 g C m^{-2} yr^{-1} may be eutrophic, but does not experience eutrophication (Nixon & Fulweiler 2009).

According to Håkanson & Bryhn (2008) and Håkanson et al. (2010), the open Baltic Proper is not eutrophic. With typical chlorophyll *a* concentrations around 2 mg dm^{-3} in the surface water layer, it should rather be classified as a system on the boundary between oligotrophy and mesotrophy (Table 4). These authors further conclude that the trophic conditions in the Baltic Sea have varied little during the last 30 years, and they are not getting worse, but marginally better. Only the Gulf of Finland, the Gulf of Riga, the area near Kaliningrad, the Vistula and Oder estuary, are eutrophic and need attention. Therefore, all the actions related to nutrient load reduction should be: (i) focused on these regions, but not on the entire Baltic Sea, and (ii) preceded by cost-effective studies. An integrated assessment of eutrophication in the Baltic Sea for the period of 1901–2012 (Andersen et al. 2015) shows that the ecological status was at its worst around 1985, and these conditions have since improved in a trend-wise manner, which is in line with the findings of Håkanson & Bryhn (2008).

Conclusions of Håkanson and Bryhn (2008) and Håkanson et al. (2010) find full confirmation in the Polish data encompassing chlorophyll *a* (Fig. 9), nutrient concentrations and primary production (Pastuszak et al. 2016; Tables 5, 6). Chlorophyll *a* concentrations in the southern Baltic slightly increased between the 1970s and the 1990s, but only very limited areas close to Vistula and the Oder outlets could be characterized as eutrophic (Fig. 9). A significant decline in nutrient loads discharged by the Polish rivers during the transition period (Figs 4 and 6) has resulted in a considerable decrease in chlorophyll *a* concentrations in the vicinity of the Vistula outlet, with maximum values not exceeding 6 mg m^{-3} in 2014 (Fig. 9, station UW1).

The supply of organic carbon to the system, evaluated based on measurements of primary production, is another indicator of eutrophication (Nixon 1995; Table 4). Although, average annual

Table 4

Eutrophication indicators in the brackish water systems (salinity 5–20 PSU) for four specified trophic states (source: Nixon 1995; Håkanson & Bryhn 2008)

Trophic state	Håkanson and Bryhn (2008)					Nixon (1995)
	Secchi depth	Chl-*a*	TN	TP	Cyanobacteria	Organic carbon supply
	m	µg l^{-1}			µg wm l^{-1}	g C m^{-2} yr^{-1}
Oligotrophic	> 8	< 2	< 70	< 10	< 9.5	< 100
Mesotrophic	4.5–8	2–6	70–220	10–30	9.5–380	100–300
Eutrophic	1.5–4.5	6–20	220–650	30–90	380–2500	301–500
Hypertrophic	< 1.5	> 20	> 650	> 90	> 2500	> 500

Table 5

Annual primary production [g C m^{-2} yr^{-1}] in various regions of the Baltic Sea and various periods of time (source: Renk 2000 – publication of NMFRI)

Area	Study period			
	1971–1974	1981–1985	1987–1991	1994–1998
Gulf of Gdańsk (average value)	140	156		
Gulf of Gdańsk (Gdańsk Deep)	107	129	172	190
Bornholm Basin (Bornholm Deep)	82	91	123	164
Gotland Basin (Gotland Deep)	92	116	141	140
Słupsk Furrow	88	103		

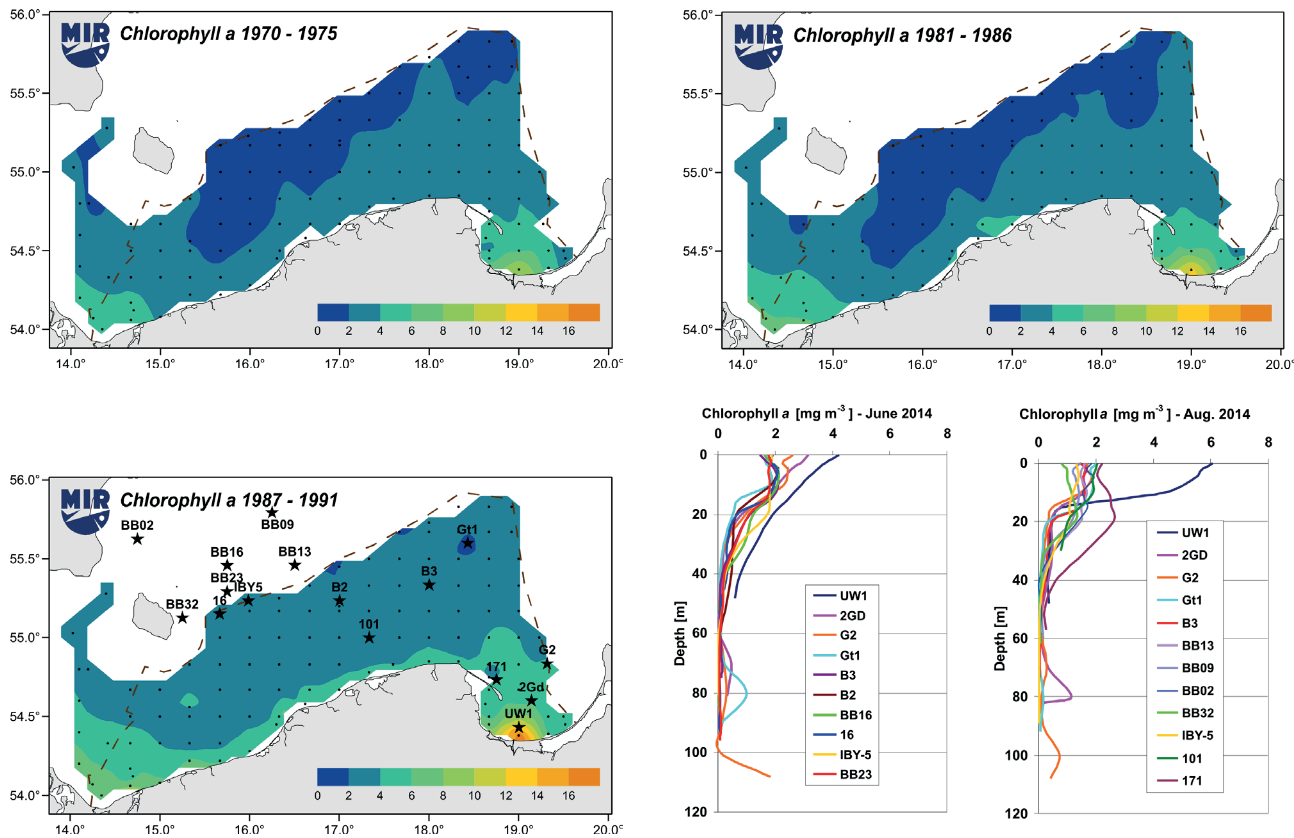

Figure 9

Average chlorophyll *a* concentrations (mg m^{-3}) (mg m^{-3} = μg dm^{-3}) in the summer season (July–September) in the 0–10 m layer in 1970–1975, 1981–1986, 1987–1991 (dots in the maps indicate sampled oceanographic stations) and vertical profiles of chlorophyll *a* concentrations (mg m^{-3}) measured in June and August 2014 at oceanographic stations marked with asterisks (source: Renk 2000; Pastuszak et al. 2016; NMFRI data and publications)

primary production in the Gdańsk Deep or Bornholm Deep has doubled between 1971–1974 and 1994–1998 (Table 5), the regions remain mesotrophic (Renk 2000). A similar conclusion can be drawn from the Table 6, covering a large spectrum of Baltic regions studied in 1954–2013. The average primary production, calculated for all these regions, amounts to 128 g C m^{-2} yr^{-1}, and that qualifies the Baltic Sea as mesotrophic. Out of 30 average annual production values, as many as 13 represent oligotrophic systems, whereas the remaining 17 – mesotrophic systems (with one exception, when the value exceeded by 4 units the upper limit of the mesotrophic value sensu Nixon 1995) (Renk 2000). The average primary production, calculated for the period of 1966–1995 reached the following values: 187 g C m^{-2} yr^{-1} in the Gulf of Gdańsk; 158 g C m^{-2} yr^{-1} in the Gdańsk Deep; 136 g C m^{-2} yr^{-1} in the Bornholm Deep; 183 g C m^{-2} yr^{-1} in the Gotland Deep (Renk 2000).

Top-down and bottom-up effects » regime shifts » a new challenge for the researchers

The problem of eutrophication of marine ecosystems must be studied at both the micro- and macroscale, otherwise the researchers' conclusions may be erroneous (Nixon 2009). Ecosystem functioning is based on numerous physical, chemical, and biological processes, which show nonlinear interactions (Nixon 1995; 2009; Botsford et al. 1997; Duarte 2009; Duarte et al. 2009). The macroscale encompasses the use of natural resources and climate change and their impact on the functioning of marine ecosystems. NAO and BSI are the acronyms of the North Atlantic Oscillation Index and the Baltic Sea Index (Lees et al. 2006; Wanner et al. 2001; Möllmann et al. 2005). The predominance of the positive NAO index, observed in the Baltic region over the last decades, indicates the predominance of westerly wind circulation characterized by transport of warm, moist air masses from above the Atlantic Ocean to Europe. Such circulation results in excessive precipitation and riverine water outflow. The negative NAO index indicates the easterly wind circulation bringing cold, dry air masses to Europe, which results in reduced riverine outflows and lower air temperatures. Both circulations extend over a large part of the globe, including Europe, the Baltic Sea and the North Sea (Wanner et al. 2001; Hurrell 1995), thus have strong impact on BSI (Alheit et al. 2004) (Fig. 10).

Applying a significant simplification, we may state that the Baltic ecosystem is characterized by a four-level food web, with predatory fish (cod, salmonid) feeding on smaller prey fish (Köster et al. 2003). Both, herring and sprat feed on zooplankton, the sprat being mainly a zooplanktivore irrespective of size, while herring may also use other food sources such as zoobenthos. Zooplankton feeds on phytoplankton, whose species composition is determined by, e.g. nutrient availability and salinity (Köster et al. 2003; Köster & Möllmann 2000; Möllmann 2011; Möllmann et al. 2004; Wasmund & Uhlig 2003) (Fig. 10).

The cascade top-down effect (Fig. 10) was generated by excessive, unsustainable catches of cod in the 1980s, which coincided with much worse cod recruitment. Reduced pressure on small prey fish caused a significant increase in sprat biomass, additionally strengthened by good recruitment of sprat at higher water temperature (Köster et al. 2003; ICES 2011). The increased biomass of prey fish imposed greater pressure on zooplankton, causing its biomass to decline. The lower zooplankton biomass triggered unsustainable consumption of phytoplankton and led

Table 6

Average annual primary production [g C m^{-2} yr^{-1}] in different regions of the Baltic Sea in 1954–2013 (source: Renk 2000 – publication of NMFRI; unpublished data of NMFRI)

Area	Study period	Primary Production	Author(s)
Kattegat	1954–1960	97.5	Steemann Nielsen 1965
	1964–1969	90.4	Gargas et al. 1978
	1988–1990	290	Richardson & Christoffersen 1991
Belt Sea	1953–1957	86	Steemann Nielsen 1965
	1975–1977	116.5	Gargas et al. 1978
Belt Sea – Sound	1972	70–77	Edler 1978
	1973	73–183	
Belt Sea – Kiel Bight	1971–1793	158	Bodungen et al. 1975
Bornholm Basin – Arkona Deep	1971–1974	85	Renk 1983
	1967–1978	94.3	Schulz & Kaiser 1973 and 1976
Bornholm Basin – Mecklenburg Bight	1969–1978	130	Kaiser et al. 1981
Bornholm Basin – Bornholm Deep	1967–1972	59–138	Schulz & Kaiser 1974 and 1975
	1971–1975	95	Renk 1983
	1987–1991	123	Renk 1997
Gulf of Gdańsk	1971–1974	140	
	1987	304	
	2004–2013	138–285	unpublished data, NMFRI
		average: 229	
Gulf of Gdańsk – Puck Bay	1965–1991	198	Renk 1997
Gulf of Gdańsk – Gdańsk Deep	1971–1974	107	
	1981–1985	129	
	1987–1991	172	
Gotland Basin – Gotland Deep	1970	38	Schulz & Kaiser 1973
	1973	91	Ackefors & Lindahl 1975
	1974	116	Lindahl 1977
	1987–1991	141	Renk 1991 and 1997
Aland Sea	1974–1976	66–94	Lindahl 1977
Bothnian Bay	1973–1974	18–70	
Gulf of Finland	1967–1971	30–65	Niemi 1975; Bagge & Niemi 1971
		78	Forsskåhl et al. 1982
Gulf of Finland – Helsinki area	1968	150–200	Bagge & Lehmusluoto 1971

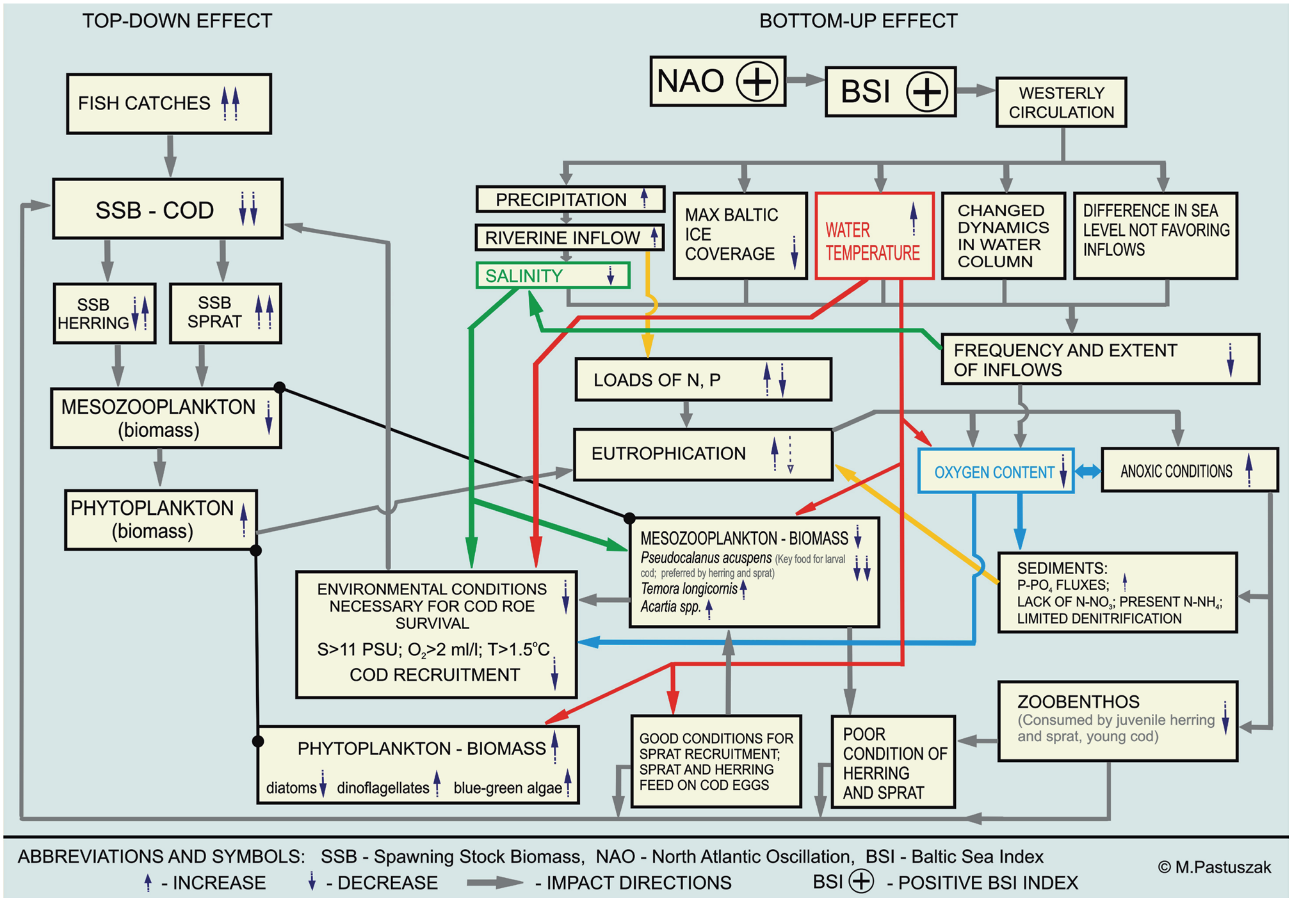

Figure 10

Top-down and bottom-up cascades and their interrelations in the Baltic Sea – biotic and abiotic parameters (source: graph was prepared based on the knowledge derived from the following publications: Alheit et al. 2004; 2005; Casini 2011; Casini et al. 2008; Cardinale & Arrhenius 2000; Cardinale et al. 2002; 2009; Conley et al. 2008; Dascalov 2002; 2011; Diekmann & Möllmann 2010; Flinkman et al. 1998; Håkanson et al. 2010; Hansson & Andersson 2015; Hare & Mantua 2000; Hinrichsen et al. 2002; Hurrell 1995; ICES 2011; Köster & Möllmann 2000; Köster et al. 2003; 2005; Lees et al. 2006; Mayer & Rietkerk 2004; Meier & Kauker 2003; Mohrholz et al. 2015; Möllmann 2011; Möllmann & Köster 1999; 2002; Möllmann et al. 2000; 2003; 2004; 2005; 2009; Overland et al. 2008; Rahm et al. 1996; Rönkkönen et al. 2004; Scheffer & Carpenter 2003; Schinke & Matthäus 1998; Scheffer et al. 2001; Wanner et al. 2001; Wasmund & Uhlig 2003)

to its excessive sedimentation on the bottom, thus worsening oxygen conditions, the situation that can be wrongly ascribed to the eutrophication process.

Due to climate change, there has been a dramatic decline in the number and intensity of inflows of saline and well-oxygenated waters into the Baltic Sea over the last decades (Schinke & Matthäus 1998; Meier & Kauker 2003; Mohrholz et al. 2015). Consequently, ventilation of stagnating bottom waters has considerably decreased thereby worsening the oxygenation of the near-bottom waters in deep basins of the Baltic Sea (Hansson & Andersson 2015), a phenomenon which should not be related exclusively to the eutrophication process but also to climatic factors.

Climate change has manifested itself in a decline in salinity and a significant increase in surface-water temperatures in the Baltic Proper (Möllmann et al. 2000; HELCOM 2009c; Håkanson et al. 2010). Temperature, salinity and oxygen have a huge impact on the functioning of the entire Baltic ecosystem. Changes in temperature and salinity triggered the disturbance in water dynamics in the upper layer, which resulted in a drastic decrease in the biomass of spring diatoms, a parallel increase in flagellates and total phytoplankton biomass in the Bornholm and Gotland Basin in 1988–1989 (Wasmund & Uhlig 2003; Alheit et al. 2004; Casini et al. 2008; Casini 2011; ICES 2011). A decline in diatom biomass, the most desired food of copepod, is explained by disturbance in water dynamics in the upper layer, more precisely in its greater stability favoring the development of flagellates but not diatoms (Alheit et al. 2004). The other reason of diatom biomass decline is a decrease in silicon concentrations, mainly due to river damming (Humborg et al. 2006; Conley et al. 2008). The causes of eutrophication should therefore be sought also in disturbed Si:N, Si:P, N:P ratios, but not exclusively in excessive loads of N and P (Rahm et al. 1996; Håkanson et al. 2010). The observed abiotic changes have also demonstrated their impact on zooplankton, which plays an extremely important role in the energy transfer between phytoplankton and fish. Zooplankton can be affected by top-down and bottom-up processes, and changes in its assemblages are reflected directly and rapidly in fish growth and stock conditions (HELCOM 2009c) (Fig. 10). Recent studies indicate that herring and sprat growth strongly depends on amount of copepods *Pseudocalanus* spp. and *Temora longicornis* (Rönkkönen et al. 2004; Möllmann et al. 2004; 2005; Cardinale et al. 2002; 2009). *Pseudocalanus* spp. is a high-energy food item zooplankton species in the central Baltic and it constitutes the main food for larval stages as well as adults of zooplanktivore fish (Hinrichsen et al. 2002; Möllmann et al. 2003; Möllmann & Köster 1999; 2002). There are numerous studies proving that owing to decline in salinity, worsening of water oxygenation and increased pressure from abundant sprat, a considerable drop in *Pseudocalanus* spp. has been observed (Möllmann et al. 2004; 2005; Möllmann & Köster 2002). The overall zooplankton biomass has shown a decline since the 1990's (Casini et al. 2008). Increasing water temperature in the Baltic Sea over the last years has favored a considerable development of zooplankton species that prefer higher temperatures e.g. *Acartia* spp. and *Temora longicornis* (Köster et al. 2003; Möllmann et al. 2000; 2003; 2005; Alheit et al. 2004) (Fig. 10). *Temora longicornis* is still positively selected by herring, but has lower energy content than *Pseudocalanus* spp. and *Acartia* spp. (Flinkman et al. 1998).

Sudden changes in the structure and function of marine ecosystems, e.g. in the Baltic Sea, occurring at multiple trophic levels (Fig. 10) and on large geographic scales are termed regime shifts. Regime shifts are commonly defined as abrupt changes between contrasting, persistent states of an ecosystem (Möllmann et al. 2009; Casini 2011; Möllmann 2011). The synchrony and time of occurrence of regime shifts in the northern hemisphere (the North Sea, the Baltic Sea, the Mediterranean Sea, the Black Sea, the Biscay Bay, the western part of the Atlantic Ocean, the northern Pacific) at the turn of the 1980s and 1990s suggest one common driving force i.e. climate change (Hare & Mantua 2000; Scheffer et al. 2001; Daskalov 2002; Scheffer & Carpenter 2003; Alheit et al. 2005; Overland et al. 2008; Casini et al. 2008; Möllmann et al. 2009). In the case of the Baltic Sea, multivariate analysis was performed in the following subregions: the Gulf of Riga, the Gulf of Finland, the Bothnian Bay, the Bothnian Sea and coastal regions. Despite large differences in environmental parameters among those regions, a change of regime was detected in all of them between 1987 and 1989 (Möllmann et al. 2009; Diekmann & Möllmann 2010). Trophic cascades triggered by overfishing are regularly involved, indicating the interaction of multiply drivers. Multiple drivers interact in a way that one undermines resilience (overfishing) and another (climate change) gives the final impulse. Ecosystem regime shifts can be extremely difficult to reverse when alternative stable states are involved (Möllmann 2011), and this fact should be kept in mind when attempts are made to divert the Baltic Sea to the state prior to eutrophication, i.e. prior to quantitative and qualitative reorganization of the Baltic Sea ecosystem.

Conclusions

Poland is one of the main exporters of total nitrogen (TN) and total phosphorus (TP) to the Baltic Sea, therefore we believe that the outcome of the presented paper is important for the decision-makers, as well as the research community in the Baltic region. The demands regarding the nutrient load reduction, resulting from regional agreements (HELCOM) and international regulations (European Union), should be optimally consistent and feasible to implement. This study has shown that this is not the case. Despite the tremendous progress in reducing N, P loads during the transition period, Poland does not meet the requirements established by HELCOM, but complies with the requirements of the Water Framework Directive, with nutrient concentrations in the Vistula and Oder (lowermost monitoring stations) below the target values established for good ecological status. If the HELCOM requirements were met, TP loads and concentrations in the Oder would have to reach values close to those in pre-industrial times, characterized by completely different agricultural activity and intensity, and a much smaller population. That would not be possible to achieve in any densely populated and agriculturally dominated country, like Poland, let alone the consequences for the agricultural sector and safety of food supply, or the related costs. This suggests an overestimation of the TP load reduction, a problem that has been identified in Håkanson & Bryhn (2008) and Håkanson et al. (2010). According to these authors, the reduction of TP loads proposed by HELCOM (ca. 15 000 tons P yr^{-1}) will lead not only to a decline in primary production, but also secondary production, including zooplankton and fish, also fish with high commercial value. We have identified several causes of the overestimation of nutrient load reduction requirements by HELCOM: (i) an inadequately assessed level of eutrophication; the Baltic Sea as a whole is not eutrophic; the Gulf of Finland, the Gulf of Riga, the area near Kaliningrad, the Vistula and Oder estuary remain eutrophic; (ii) HELCOM's mathematical model allowing establishing the level of nutrient load reduction has been challenged by Håkanson and Bryhn (2008) who are of the opinion that a reduction of TN load is not recommended, whereas TP load reduction should be by 40% lower than that proposed by HELCOM; (iii) an inadequate reference period (1997–2003) that does not represent the maximum TN and TP emission into the Baltic Sea; (iv) omission of the role of estuaries in substantial natural retention of nitrogen and phosphorus. Adoption of the 1997–2003 reference period indicates that up to 2012–2014 Poland has reduced TN and TP loads by 29 276 tons N yr^{-1} and by 2558 tons P yr^{-1}, respectively; adoption of periods with maximum TN (1992–1994) and TP (1988–1991) export gives the following levels of reduction: TN – by 68 871 tons N yr^{-1} and TP by 5913 tons P yr^{-1}. The causes of the currently unsustainable functioning of the Baltic ecosystem must be additionally sought in: (i) the impact of climate change, which is reflected in a drastically reduced number of refreshing inflows of saline, well-oxygenated waters from the North Sea; (ii) disturbed N:P, N:Si, P:Si ratios; (iii) top-down and bottom-up effects resulting in the detected ecosystem regime shift. Ecosystem regime shifts can be extremely difficult to reverse when alternative stable states are involved, and this fact should be kept in mind when attempts are made to divert the ecosystem of the Baltic Sea to the state prior to its quantitative and qualitative reorganization.

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