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DELIVERABLE 4.2

SAFE ECOLOGICAL LIMITS

Work Package 4
Ecosystem Health

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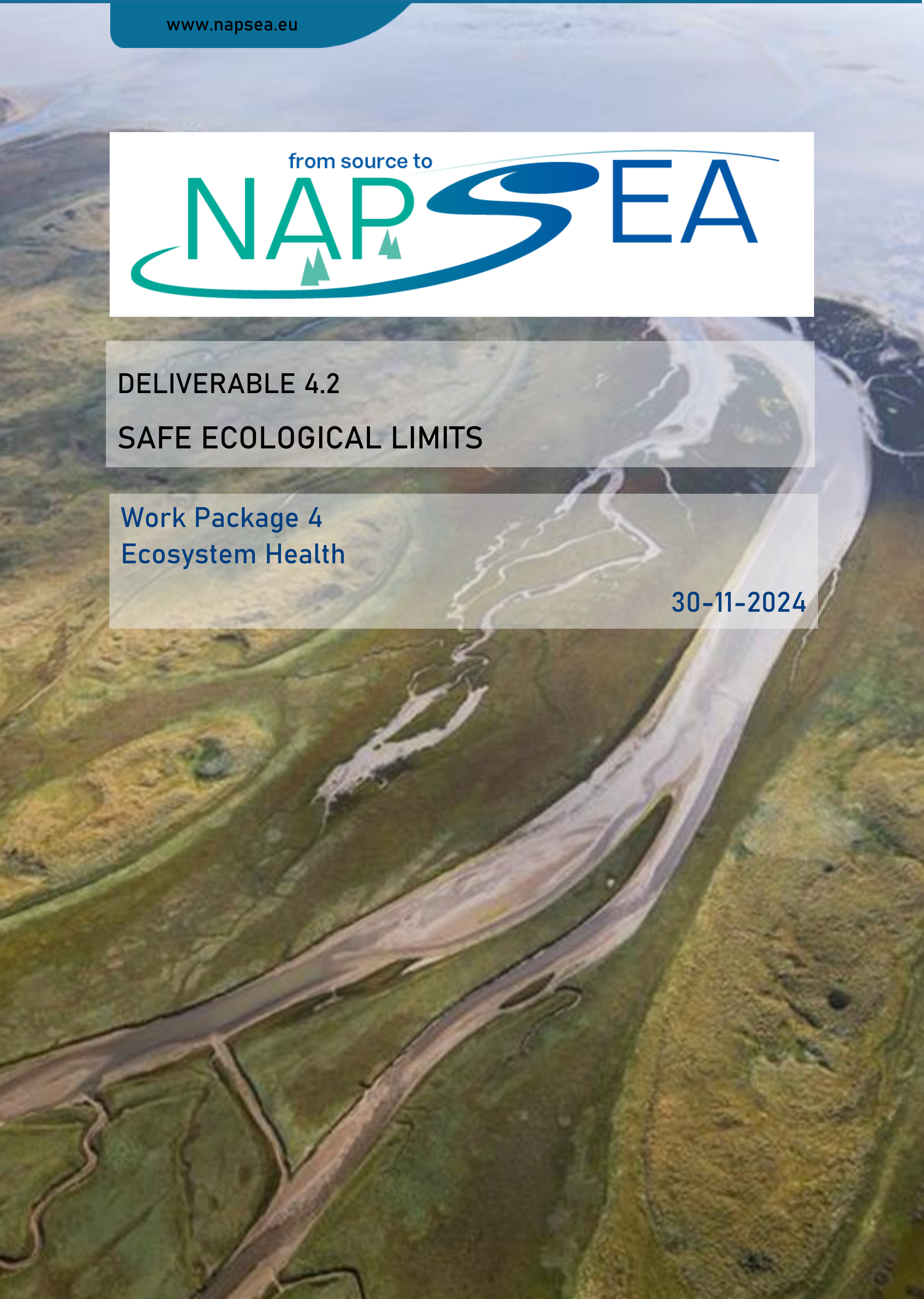

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Abstract	This deliverable addresses the Safe Ecological Limits in the four NAPSEA case studies. For the Wadden Sea, new limits were derived from seagrass and phytoplankton composition. The reduction needed to reach these Safe Ecological Limits were put into perspective of three riverine case studies (Rhine Catchment, Elbe estuary, Hunze catchment) each with their own unique environmental setting and reduction needs.
Keywords	Safe ecological limits; Rhine; Elbe; Hunze; Wadden Sea

Contents

1. ACRONYMES

N	Nitrogen
TN	Total Nitrogen
P	Phosphorus, mostly as PO_4
Si	Silicate, mostly as SiO_4
WFD	Water Framework Directive
OGweV	Oberflächenwasserverordnung

2. EXECUTIVE SUMMARY

In this deliverable, we address Safe Ecological limits. We developed new indicators for the Wadden Sea as the ultimate receiver of nutrients from the rivers Rhine, Elbe and Hunze, and discuss the Wadden Sea reduction needs in relation to Safe Ecological Limits for the Rhine catchment, the upper Elbe Estuary and the Hunze catchment, each with their own unique ecological settings. The indicators and proposed reduction needs generally refer to 2010 – 2017 as a reference period.

In the Wadden Sea case study, it was shown that N is the main limiting nutrient for phytoplankton growth. Two new indicators were developed in addition to existing indicators for nutrients and chlorophyll: seagrass recovery and Si/N ratios in winter. Seagrass recovery in the northern Wadden Sea accelerated around 2000 but no clear recovery was observed in the southern Wadden Sea. The eutrophication conditions (based on phytoplankton biomass) that prevailed during the recovery in the northern Wadden Sea were projected on the southern Wadden Sea. It was estimated that in comparison to the period 2010 – 2017 the riverine TN loads should be reduced by at least 1/3 (range: 34 – 46%). The second indicator is based on the ecological role of dissolved silicate (Si). During pre-eutrophication conditions the spring bloom was limited by N before it shifted to a Si limitation of diatoms during increased eutrophication and leading to increased blooms of algae not depending on Si. To return to a N-limited spring diatom bloom, a reduction in riverine N loads of 30% (Elbe) – 55% (Ems) is needed.

The Elbe Case Study focussed on the present O_2 problems in the upper estuary caused on the one hand by the extreme large phytoplankton blooms in the riverine part of the Elbe and on the other hand by light limitation in the estuarine part due to dredging. When these riverine blooms enter the upper part of the upper estuary, grazing decimates the phytoplankton standing stock whereas light limitation precludes phytoplankton growth to compensate for the losses. This lead leads to severer O_2 problems during summer. Based on models, a reduction need in organic matter loading of about 45% was estimated to reach Safe Ecological Limits of 7 mg O_2 /l. Present phytoplankton levels (March – October) in the Elbe just upstream of the estuary of about 109 µg Chl a/l are well above the Safe Ecological Limits of 40.1 µg/l implying a reduction by 63%.

The Rhine and Elbe are the two major rivers draining a large part of Germany and the Netherlands and impacting the Wadden Sea. The two rivers are very different. First, the Rhine is dominated by large amounts of melting water (about 48% of total discharge) from the Alps with low nutrient concentrations, whereas the Elbe is a rain-fed river. Furthermore, a high number of invasive filter feeders inhabit the riverbed suppressing phytoplankton growth in the Rhine in contrast to the Elbe, where large phytoplankton blooms occur. Whereas most stations in the main stem of the Rhine indicate a good status for both N and P, most of the stations in the sub-catchments fail the good status. For N, the average reduction needed for a good status in the sub-catchments amounts to 44%, for P about 50%. In the Dutch part of the Rhine basin, nearly 50% of the water bodies had a moderate or worse status for P and about 30% had a moderate or worse status for N.

The Hunze, located in the northeast of The Netherlands, south of Groningen, drains directly into the Zuidlaardermeer, that is connected to the Wadden Sea through various waterways. The ecological status improved during the last decades and several nutrient targets are nearly met (except for NH_4). Still, problems remain including blue-green algae (cyanobacteria) compromising local bathing water quality and submersed vegetation growth. To change lake Zuidlaardermeer from an algae-dominated state into a clear state dominated by submersed vegetation, a critical P-load threshold of 2.75 mg P/m^2/d is derived from model calculations, implying a reduction need of almost 40%. As in many other freshwater systems, the focus is on P to reach a good environmental status. To support Safe Ecological Levels for the Wadden Sea an N reduction of 34% is needed.

Despite a wide range of ecological settings the reduction needs are in a similar range, between 30 and 63% for both N and P. Whereas our focus was on aquatic habitats we also discuss the reduction needs in the context of the environmental problems of N deposition on terrestrial ecosystems where reductions in N deposition in a similar range are needed.

3. GENERAL INTRODUCTION

After a strong increase in riverine nutrient concentrations since the late 1940s, many policy-driven measures have been taken during the 1970s and 1980s to combat the adverse effects of N and P enrichment in both marine waters and in freshwater systems in Europe (e.g. de Jong, 2007). Riverine nutrient concentrations have decreased since then leading to an improvement of the environmental status (e.g. van Beusekom et al., 2019). However, in most aquatic (marine and freshwater) systems, a good environmental status (as defined by the MSRL and WFD) has not been reached yet (see Deliverable 4.1). The NAPSEA goal is to develop a more holistic view on eutrophication by looking at the N- and P- flows from the source to the sea. The aim of this deliverable is to propose new Safe Ecological Limits and discuss them in relation to the present eutrophication status based on existing indicators.

We define Safe Ecologic Limits as the limits of factors that change an ecosystem in an undesirable way. This implies that indicators must be developed, adopted or modified to enable a quantification between driver and ecological response. This is a very challenging task as many natural factors (or “noise”) and human driven factors impact the status of an ecosystem. This also implies that Safe Ecological Limits have to be defined locally. This is supported by the review of the indicators (Deliverable 4.1) which observed that many local indicators and limits are defined, and that these indicators do not form a clear continuum. For example, different indicators are used for groundwater, surface water, and coastal waters and for surface water the EU member states and local water authorities also use different indicators for their WFD status assessments.

Since NAPSEA focuses on the Wadden Sea as the ultimate receiver of a large part of the nutrients released in northern Europa, we will discuss the newly proposed safe ecological limits in three terrestrial aquatic systems: Rhine, Elbe and the Hunze watershed (the Netherlands) all debouching in or –especially in the case of the Rhine/Meuse and Elbe- strongly impacting the Wadden Sea.

Basically, N and P loads are strongly coupled to the water flow starting with rain or melting snow and ice that contain only low nutrient levels from e.g. atmospheric sources, and that are continuously enriched with nutrients on their way to the sea. During transport from the source to the sea, nutrients may be taken up by primary producers, may be released from detritus or may be removed from the biogeochemical cycle by denitrification (N only) or burial (permanent sedimentation and/or dredging) of N- and P-containing particles. Especially nutrient uptake exerts a strong impact on the local ecological conditions because of the corresponding increases in organic matter loading (e.g. Nixon, 1995).

First, we focus on the Wadden Sea as the ultimate receiver of nutrients from the two major north-European rivers (Rhine and Elbe) as major nutrient sources (e.g. van Beusekom et al., 2001). Until now, phytoplankton biomass has been a major indicator to evaluate the eutrophication status of the Wadden Sea for instance in the WFD or in the MSFD as it shows a clear relation with riverine nutrient inputs (e.g. van Beusekom et al., 2019). However, defining proper chlorophyll levels is challenging: For instance: why would a level of 4 instead of 5 µg chlorophyll / liter be a good indicator for a safe ecological limit? To circumvent such arbitrary choices, we suggest two approaches that look at discontinuities in the ecosystem response to changes in riverine nutrient loads: seagrass recovery and Si limitation of the spring phytoplankton bloom. Then we evaluate ecological conditions and discuss ecological indicators for instance as used in the WFD for the two major river systems -Elbe and Rhine- each with their own specific ecological settings and for a small lowland catchment with a small lake (Hunze/Zuidlaardermeer) entering the Wadden Sea.

In the last chapter of this deliverable, we summarize the results from the case studies and draw some overarching conclusions. and discuss whether current environmental goals as formulated in the WFD are able to bring both the Wadden Sea and the contributing river basins within Safe Ecological Limits.

4. SAFE ECOLOGICAL LIMITS FOR THE WADDEN SEA

4.1 Introduction

The Wadden Sea is a unique intertidal coastal ecosystem stretching along the Dutch, German and Danish North Sea coast which was declared as a World Nature Heritage Site in 2004. The Wadden Sea is a relatively young ecosystem that developed about 7500 years ago after the end of the last glacial, and human impacts have changed the Wadden Sea since about 1000 years (Lotze et al., 2005). Eutrophication is one of the more recent human pressures impacting the Wadden Sea. Our understanding of the Wadden Sea eutrophication was shaped especially by the investigations since the 1950s of the Netherlands Institute for Sea Research in Den Helder and since 1969 on Texel shaped: First signs of increased eutrophication were already described in the 1970s. Van Bennekom et al. (1975) already noted the strong increase in nutrients in the Rhine. De Jonge and Postma (1974) showed a 3-fold increase in phosphorus import from the North Sea into the Wadden Sea between the 1950s and 1970s. In this chapter, we will summarize our understanding of Wadden Sea eutrophication, its long-term trends and regional differences, review potential indicators and suggest Safe Ecological Limits. Phytoplankton (Chlorophyll a) is a widely used eutrophication indicator. One of the goals of the NAPSEA project is to suggest alternative indicators that may guide Safe Ecological limits for the Wadden Sea. We will focus on 1) the role of silicium (Si) as a limiting nutrient for phytoplankton and 2) on seagrass response to eutrophication.

4.2 Area description

The Wadden Sea is a shallow coastal sea with a length of about 500 km and a width of 10-30 km between Den Helder in the Netherlands and the Skallingen peninsula in Denmark (Figure 1). Most of the Wadden Sea is protected from the North Sea by barrier islands. Tides play a dominant role in shaping the Wadden Sea: Tidal range is between 1.5 and >3.5 m. Highest ranges are found in the central Wadden Sea. Tidal ranges > 3 m prevent the formation of barrier islands. About 50% of the Wadden Sea are intertidal flats emerging during low tide. Sediments in most of the Wadden Sea intertidal and subtidal areas are dominated by sand, but in the more protected areas, muddy sediments prevail (e.g. Dijkema, 1991; Baptist et al., 2019).

Freshwater impacts the Wadden Sea both directly and indirectly. Important direct freshwater sources into the Wadden Sea are lake IJssel (fed by the Rhine) as well as the rivers Ems, Weser, Elbe, Eider and Vårde A. The most important indirect nutrient source is the combined outflow of the rivers Rhine and Meuse. The salinity is about 30 psu but clear salinity gradients exist near river mouths and salinity can also be lower near sluices.

Postma (1954) already pointed out that the Wadden Sea is a heterotrophic area importing organic matter produced by phytoplankton in the coastal North Sea (see also van Beusekom et al. 1999). This is the reason, why nutrient discharges via the river Rhine and Meuse are so important for the Wadden Sea eutrophication especially in the southern part (e.g. van Beusekom et al., 2001).

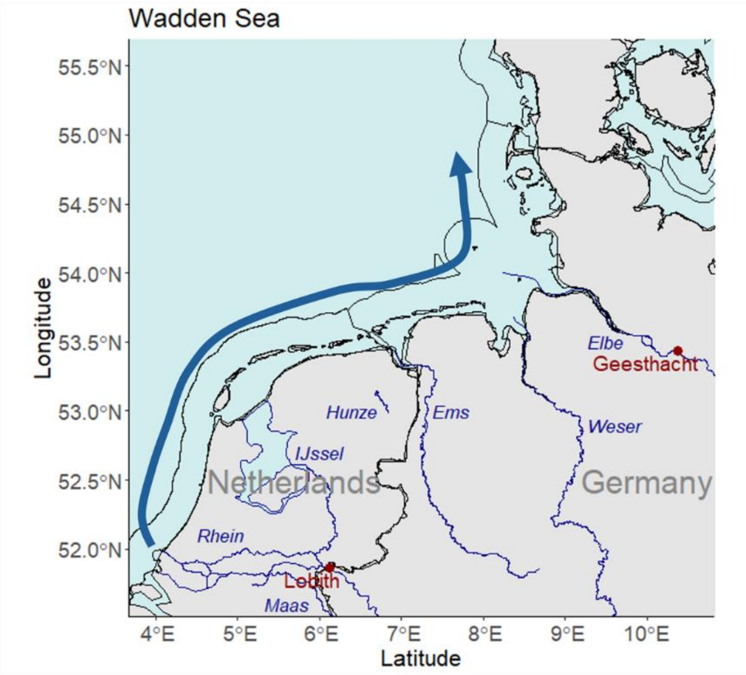


Figure 1. Map of the Wadden Sea showing the most important rivers. The blue line indicates the residual currents transporting Rhine/Meuse river water towards the Wadden Sea. (an updated version is being prepared).

4.3 Eutrophication history

Riverine nutrient concentrations sharply increased after WWII, peaked during the 1980s and 1990s, and decreased since then (Figure 2). At present, TN concentrations are about 50% and TP about 75% below their maximum levels, (van Beusekom et al., 2019). The increase in riverine nutrient loads is clearly reflected in the nutrient concentrations observed in the Wadden Sea (e.g. van Beusekom et al., 2001).

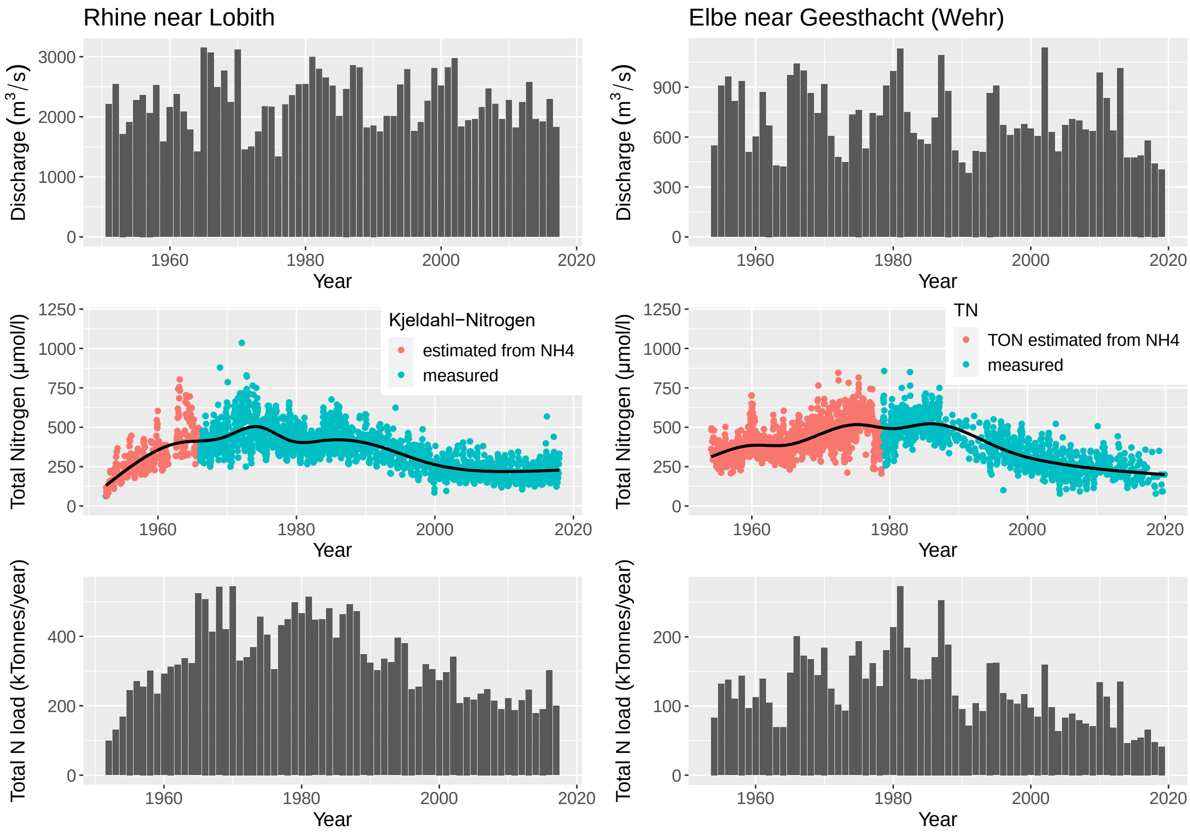


Figure 2. Historical discharges, total nitrogen (TN) concentrations and TN loads in the rivers Rhine (measured at Lobith; panels at the left) and Elbe (measured near Geesthacht; panels at the right). TN loads were either measured (green dots) or estimated based on measured DIN values and correlations between NH_4 and Organic N (Elbe)or Kjeldahl-N (Rhine). Source: van Katwijk et al. (2024).

Already during the 1970s, researchers from the NIOZ observed that increased riverine nutrient loads affected the Wadden Sea: De Jonge and Postma (1974) mentioned a three-fold increase in organic P concentrations between the 1950s and 1970s. Helder (1974) observed no changes in the NO_3^- concentrations but a clear increase in NH_4^+-concentrations compared to the first N measurements in the Wadden Sea (Postma, 1966) indicating an increased organic matter (OM) turnover.

Ecological consequences of the increased nutrient concentrations were indicated by two time series in the Western Dutch Wadden Sea on primary production and macrobenthos: Benthic primary production doubled between 1968 and 1981 from around 100 gC m^{-2} y^{-1} to more than 200 gC m^{-2} y^{-1} around 1980 (Cadée, 1984), pelagic primary production also more than doubled in the Marsdiep from about 150 gC m^{-2} y^{-1} during 1964-1976 to ~350 gC m^{-2} y^{-1} in the 1980s (Cadée & Hegeman, 1993)). Macrobenthos biomass and annual production doubled between 1970 and 1984 (Beukema & Cadée, 1986).

4.3.1 Negative effects of eutrophication

In the Wadden Sea and elsewhere, seagrass ecosystems are important habitats delivering ecosystem services like nursery habitat, improved water quality, coastal protection, and carbon sequestration (Valdez et al., 2020). Negative effects of the increased nutrient loads were observed for seagrass showing a downward trend in the Dutch Wadden Sea (den Hartog and Polderman, 1975), in the Lower Saxonian Wadden Sea (Michaelis, 1987) and in the northern Wadden Sea (Dolch et al., 2013, see also van Katwijk et al., 2024). Conversely, especially since the late 1980s, massive green macroalgae blooms were observed both in the Lower Saxonian Wadden Sea and in the northern Wadden Sea (Reise, 1994; Reise & Siebert, 1994; 1997). Negative effects are among others anoxic sediments (Neira & Rackemann, 1996) and accompanying mortality of macro- and meiobenthos (Reise & Siebert, 1994; Neira & Rackemann, 1996).

A one-time event related to eutrophication was the occurrence of black spots covering a large part of the Lower Saxonian Wadden Sea in 1996. Large parts of the sediment were completely anoxic which turned out to be a unique phenomenon (Farke, 1997; Michaelis, 1997). Possibly, the large amounts of macroalgae played a role (compare Reise, 1994; Farke, 1997).

4.3.2 Increased monitoring in the international Wadden Sea: an international view on the Wadden Sea eutrophication

During the 1970s and 1980s, several national monitoring programs were started to observe ecological and environmental changes in the Wadden Sea. In addition, the Biologische Anstalt Helgoland extended its time series to the Wadden Sea Station Sylt starting in 1984. In 1987, the trilateral Wadden Sea secretariat was founded among others to assess the ecological status of the North Sea as documented in several Quality Status Reports (QSR) (e.g. de Jong et al., 1993, 1999). In this context, Wadden Sea-specific Eutrophication Criteria were developed including summer chlorophyll (mean monthly chlorophyll a concentration between May and September) and autumn remineralization products (NO_2 + NH_4) as monthly means between September and November. These were applied during the subsequent QSRs (van Beusekom et al., 2005; 2009; 2017). The QSR analyses were extended with model data to show 1) maximum riverine nutrient loads during the 1990s decreasing by about 50% (TN) and 75% (TP) to present levels, 2) maximum eutrophication levels during the 1990s followed by a decreasing eutrophication and 3) regional differences in the eutrophication levels between the less eutrophic northern Wadden Sea and the more eutrophic southern Wadden Sea driven by a higher import of organic matter from the coastal to the southern Wadden Sea as compared to the northern Wadden Sea (van Beusekom et al., 2019).

4.3.3 Limiting nutrients

The main nutrients determining phytoplankton growth are Si, P and N. All three nutrients can be limiting in the Wadden Sea at different times in the season. Diatoms use Si for their frustules. Diatom growth is limited by Si availability. Already during the 1970s, van Bennekom et al. (1974, 1975) and Gieskes and van Bennekom (1973) showed for the Dutch coastal zone and the Western Dutch Wadden Sea that the spring phytoplankton bloom was dominated by diatoms. After the bloom, enough N and P was available to enable an ensuing bloom by algae not depending on Si. Especially *Phaeocystis* blooms increased during the period of increasing eutrophication (Cadée & Hegeman, 2002).

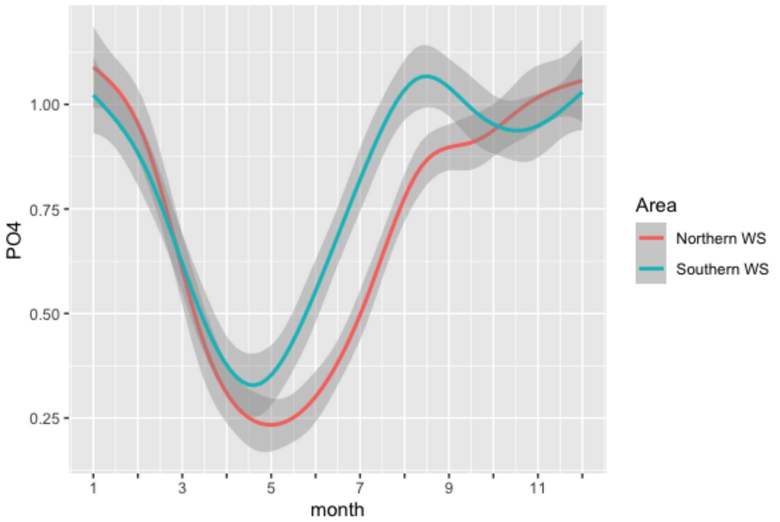


Figure 3. Mean seasonal cycle of phosphate (in µmol/l) in the Southern and in the Northern Wadden Sea (2000-2017). Limiting concentrations can be reached during April and May. For details on the stations used: see van Katwijk et al., (2024).

During the spring bloom, PO_4 concentrations can reach limiting levels during April and May (Figure 3). Ly et al (2014) carried out experiments that supported P limitation during the spring bloom in the Western Dutch Wadden Sea

However, despite decreasing P loads from the river Rhine, no decrease in annual primary production was observed suggesting that primary production in the Western Dutch Wadden Sea was not limited by P (Cadée & Hegeman, 1993). This is probably due to internal cycling: Leote et al. (2016) identified internal cycling as a major P source. De Jonge et al. (1993) showed that enough P was available in sediments to support an annual primary production of 150 gC m^{-2} y^{-1}.

The P increase after the spring blooms minimum starts earlier in the southern Wadden Sea than in the northern Wadden Sea (see Figure 3), and in the southern Wadden Sea, a small summer maximum is still reached, whereas in the northern Wadden Sea, a continuous increase is observed. This is in line with a higher eutrophication level (more import from the North Sea) in the southern Wadden Sea.

During summer, NO_3 gets more and more depleted (van Beusekom et al., 2001). For instance, in the northern Wadden Sea, van Beusekom et al. (2009) observed an increasing number of days with NO_3 concentrations below 0.5 µM between 1985 and 2005. The decreasing DIN levels and increasing PO_4 levels result in N/P ratios clearly below 16 (Redfield ratio) indicating N limitation during summer (July-September; Figure 4). Despite similar DIN/PO_4 ratios in winter, the ratios in the Northern Wadden Sea are extremely high during the spring bloom indicating a smaller P source in the Northern Wadden Sea sediment (cf. Figure 3)

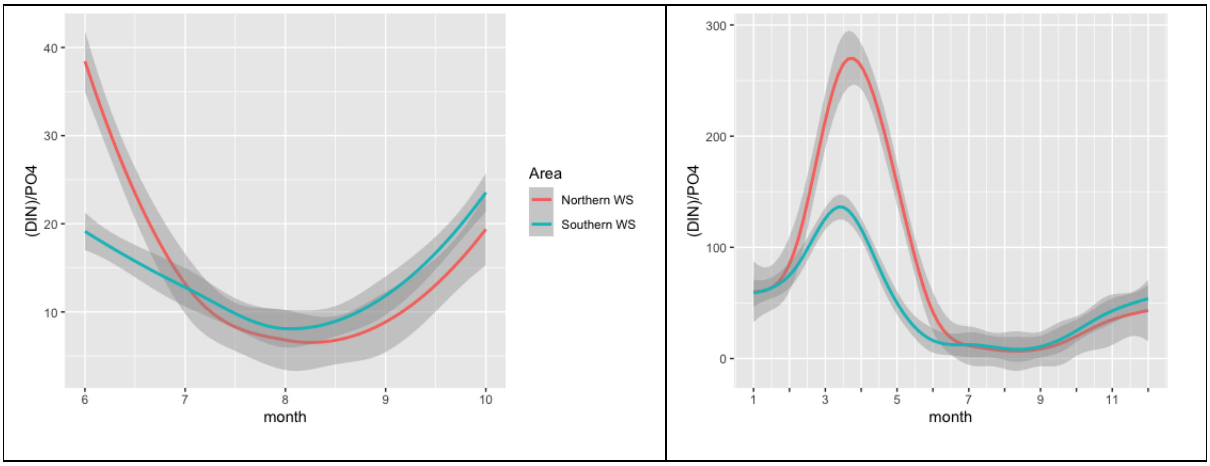


Figure 4. DIN/P ratios in the Wadden Sea (based on monthly means; 2000 – 2018) from June- September (left and during the entire year (right).

In summary, we see a clear seasonal cycle in limiting nutrients from Si in spring combined with a possible P co-limitation and a clear N limitation during summer. This pattern is reflected in the phytoplankton composition as observed in the northern Wadden Sea (Sylt). Here, a spring diatom bloom peak is reached when Si reaches limiting concentrations and the following *Phaeocystis* bloom is limited by N depletion (Figure 5, compare Loebl et al., 2007 and van Beusekom et al., 2009).

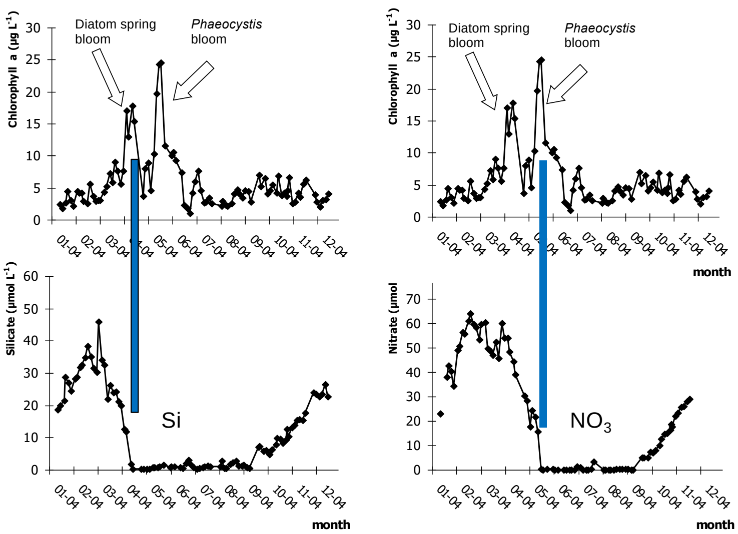


Figure 5. Seasonal pattern in phytoplankton blooms and nutrient limitation in the northern Wadden Sea (Sylt-Romo Bight), where the diatom bloom is limited by Si depletion (left) and the Phaeocystis bloom by NO3 depletion (left). The blue lines connect the peak in phytoplankton biomass with the nutrient depletions observed. (after Loebl et al., 2007).

4.4 Safe Ecological Limits

Goals to bring the Wadden Sea or the North Sea within safe ecological limits have so far mostly been based on phytoplankton biomass, indicated by chlorophyll-a concentrations. Phytoplankton biomass often directly responds to increased nutrient levels. A major challenge however is to define phytoplankton levels that ensure safe ecological conditions. We observed a linear response of the summer phytoplankton biomass to changes in riverine TN loads (albeit with variability of about 40% at a given nutrient load (e.g. van Beusekom et al., 2019). But which level marks a Safe Ecological Limit? For instance, is a total N level of 50% above the natural background safe, but a level of 75% not? To circumvent such arbitrary choices, we suggest two approaches that look at discontinuities in the ecosystem response to changes in riverine nutrient loads: seagrass recovery and Si limitation of the spring phytoplankton bloom.

4.4.1 Seagrass recovery as a safe ecological limit

Van Katwijk et al. (2024) analyzed 160 years of seagrass data from the Wadden Sea. The first eutrophication-related deteriorations were observed in the 1970s in the entire Wadden Sea. The first signs of recovery were observed in the less eutrophic northern Wadden Sea during the late 1990s after riverine TN loads started to decrease. The recovery accelerated around 2000 (Figure 6 left).

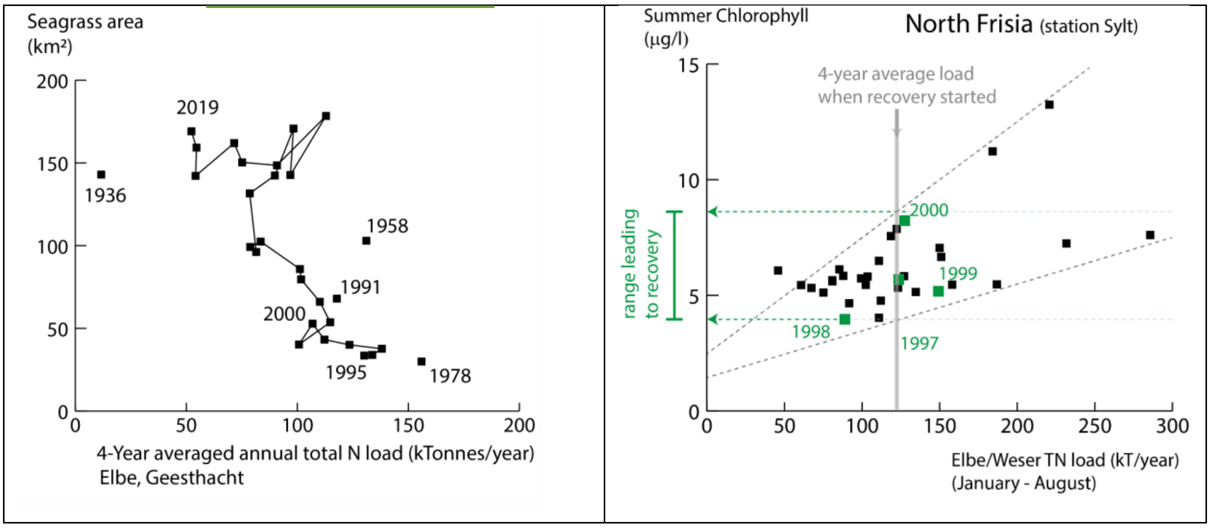


Figure 6. Left: Seagrass response to long-term changes in riverine TN loads at Geesthacht, situated near the Weir separating the river from the estuary since 1960. Right: the relation between TN loads via Weser and Elbe (annual load from January – August) and summer chlorophyll (mean of monthly means of May – September). Indicated is the range of summer chlorophyll levels that prevailed when seagrass recovery accelerated around 2000 (both Figures from van Katwijk et al., 2024, where further details can be found).

Van Beusekom et al., (2019) used two eutrophication indicators -summer chlorophyll levels (May-Sept.) as an indicator of phytoplankton growth potential and autumn levels of the sum of NH_4 and NO_2 as an indicator of organic matter remineralization potential. Both describe the long-term trends and regional differences in Wadden Sea eutrophication. In the following, phytoplankton levels are thus used as indicators of nutrient availability. During 1997 – 2000 (the years preceding the acceleration of seagrass recovery), eutrophication levels as indicated by phytoplankton levels ranged between about 4 and 8 µg chlorophyll. These general eutrophication conditions enabled seagrass recovery occurring at TN loads of about 115 kT/year. Van Katwijk et al. (2024) suggested that eutrophication levels similar to those that prevailed around 2000 in the northern Wadden Sea when seagrass returned would be necessary to also enable full recovery in the southern Wadden Sea (Figure 7). By projecting these summer chlorophyll levels on the relations between chlorophyll and Rhine/Meuse TN loads in the Western Dutch and Lower Saxonian Wadden Sea, they suggested that for seagrass recovery TN loads should be reduced by at least 34% (Western Dutch Wadden Sea) and 39% (Lower Saxonian Wadden Sea between Ems and Jade) compared to the input levels of 2010 (Figure 7). Given the uncertainties in the estimates, van Katwijk et al., (2024) suggested as a management goal to reduce the river loads by at least one-third compared to the levels between 2010-2017.

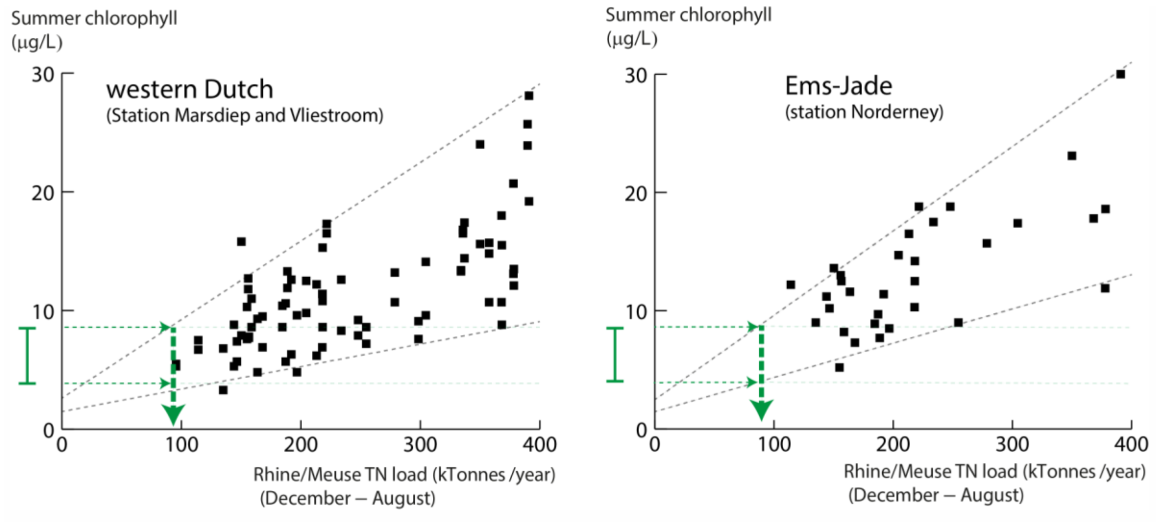


Figure 7. Estimation of reduction needs in riverine TN loads to reach eutrophication levels (as indicated by the summer chlorophyll values) in the Western Dutch Wadden Sea (stations Marsdiep and Vliestroom) and in the Lower Saxonian Wadden Sea between the Ems and Jade Bay (station Norderney). Relation between summer chlorophyll levels and TN loads by the rivers Rhine/Meuse from December–August, for the long-term stations 'Marsdiep noord' and 'Vliestroom' in the western Dutch region, 1977–2017 (left) and at 'Norderney' in the Ems-Jade region, 1985–2016 (right), and the predicted riverine TN loads that lead to seagrass recovery – provided that other habitat and seed availability requirements are met for each region. The grey dotted lines envelop most of the data and can be interpreted as the maximum and minimum values to be expected at a certain riverine nutrient load. The critical summer chlorophyll concentration (vertical green range, its derivation is depicted in Figure 6) is projected on the river TN load of Rhine/Meuse to come to this prediction (green dotted arrows). Source: van Katwijk et al. (2024).

Large regional differences in eutrophication status exist in the Wadden Sea (e.g. van Beusekom et al., 2019). Regional differences in mean summer chlorophyll levels are shown in Figure 8. The green horizontal bar depicts the range of summer chlorophyll levels that enabled seagrass recovery in the northern Wadden Sea. It shows that in the Vliestroom area values are currently within the range of enabling a permanent recovery. It is interesting to note that recent seagrass restoration was successful in that area (Gräfnings et al., 2022). The temporary recovery in Ems-Jade as well as recent restoration success at one out of three restoration sites in the western Wadden Sea (Govers et al., 2022; Gräfnings, 2022) show the vulnerability, but also the potential of seagrass in these regions.

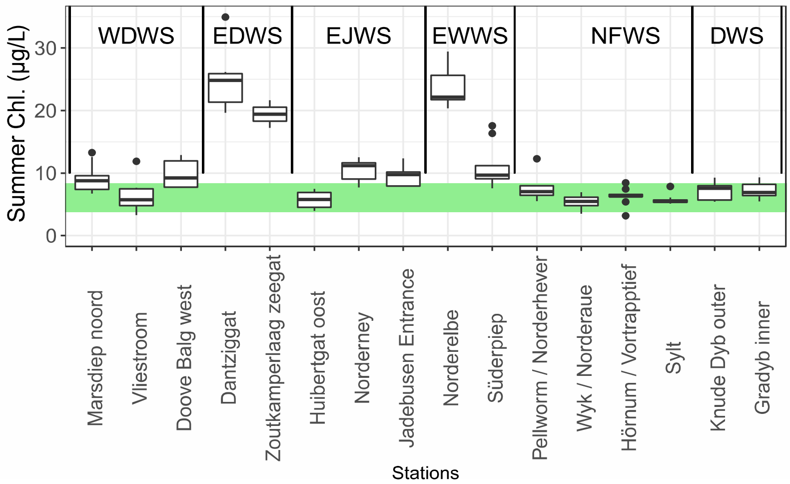


Figure 8. Regional difference in summer chlorophyll in the years 2008–2016. Summer Chlorophyll is the mean of the monthly means in May–September. DWS, Danish Wadden Sea; EDWS, eastern Dutch Wadden Sea; EJWS, Ems-Jade; EWWS, Elbe-Weser; NFWS, North Frisia; WDWS, western Dutch Wadden Sea. The green bar depicts the eutrophication threshold, that is the range between the minimum and maximum prevailing concentrations when seagrass recovery started to accelerate in the North Frisian region. Zoutkamperlaag zeegat: only 2008 and 2009 (from van Katwijk et al., 2024).

4.4.2 Si limitation as a safe ecological limit

At present, Si limits the spring diatom bloom in the Wadden Sea, and enough N is left over to fuel an ensuing non-diatom bloom. In the following, we will first derive the winter maximum Si concentrations in rivers, then derive a first estimate of N/Si uptake ratios by Wadden Sea phytoplankton and based on this we will suggest maximum riverine winter concentrations which will not allow non-diatom blooms.

In winter, maximum winter concentrations of about 200 µM Si are reached in rivers Schelde, Ems and Elbe (Figure 9-Figure 10; for the Ems, see Helder and de Vries, 1986). In the Rhine/Meuse, lower concentrations of about 135 µM are reached due to algae growth in the large lakes (like lake Constance) in the upstream part of the Rhine (e.g. van Bennekom et al., 1974; compare Figure 11) and due to the dilution with melting water from the Alps (compare chapter on Safe Ecological Limits in the Rhine).

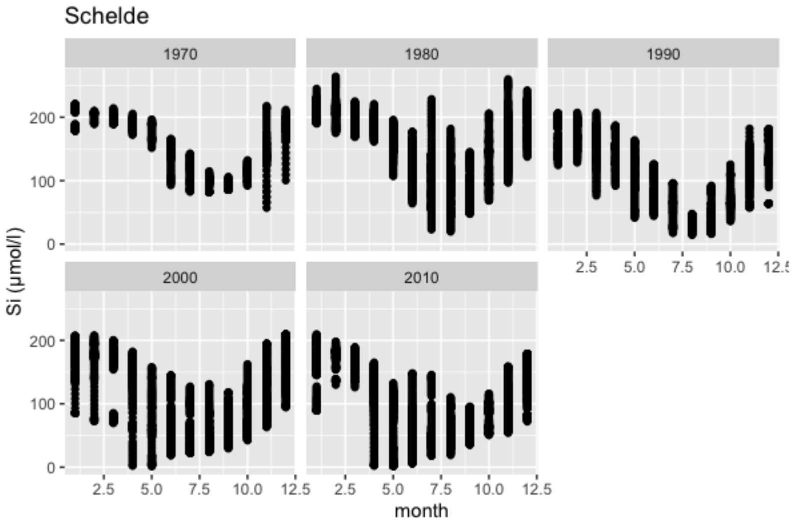


Figure 9. Seasonal cycles and long-term trends in the river Schelde. Data: Pätsch (2024)

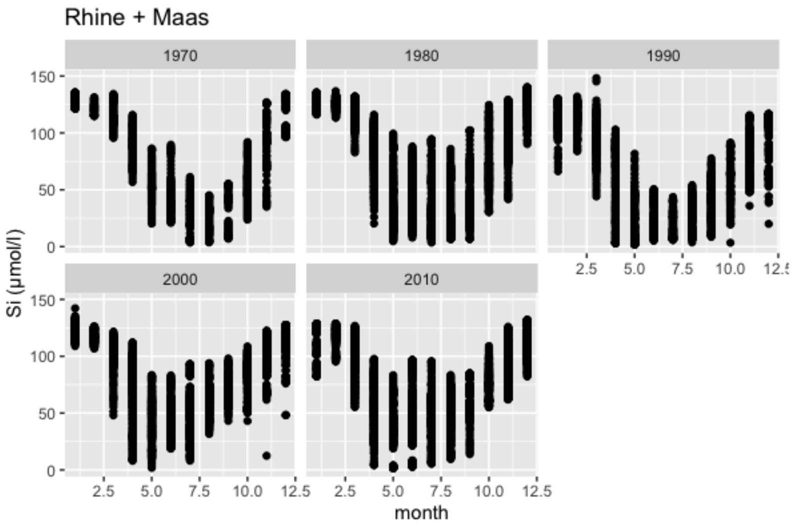


Figure 10. Seasonal cycle and long-term trends of Si in rivers Rhine and Meuse. Data: Pätsch (2024)

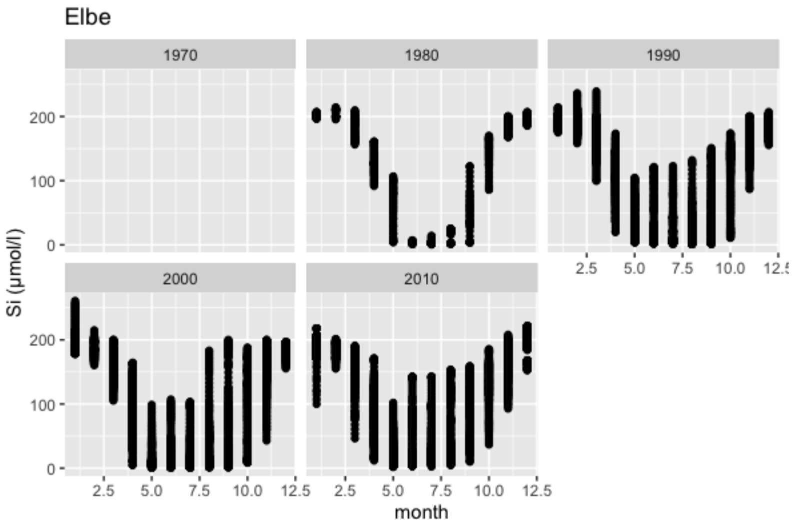


Figure 11. Seasonal cycle and long-term trends of Si in river Elbe. Data: Pätsch (2024)

The silica-content of diatoms shows a wide range. First of all, freshwater diatoms have a larger Si content than marine diatoms – probably an adaptation to different sinking strategies (Conley, 1989). For marine diatoms, Brzezinksi (1985) reported Si/N ratios of around 1 depending on diatom size and growth conditions. We used the Sylt, Norderney and Marsdiep/Vliestroom time series to check the ratio of Si and N removal during the spring phytoplankton bloom. The monthly means of February – April were used. We excluded Si values < 2µM as they may reflect N uptake under Si limited conditions. A plot of DIN versus Si shows a ratio between about 0.84 (Sylt, Figure 12) and 1.24 (Marsdiep/Vliestroom). Given a standard error of about 0.1, we use an average DIN/Si uptake-ratio of 1.

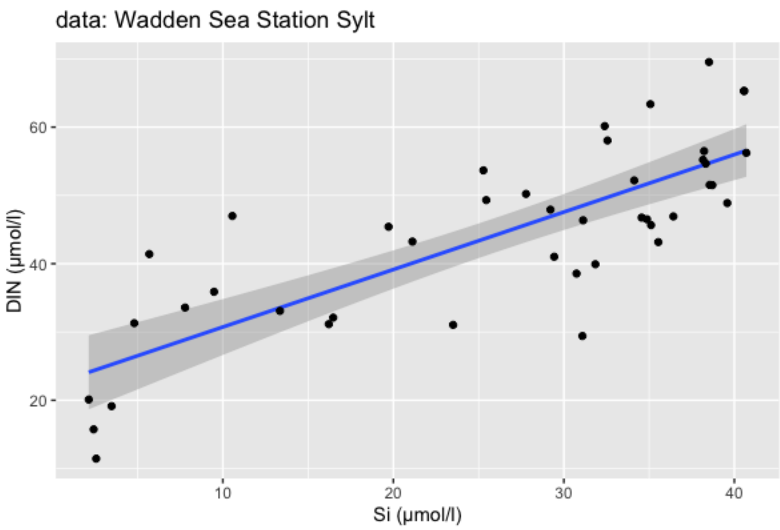


Figure 12. Correlation between Si and DIN based on monthly means (February, March, April; 2000-2017) from the Sylt time series (AWI, Wadden Sea Station Sylt) were used. Si concentrations below 2 µM were excluded as they may reflect N uptake under Si-limited conditions. The blue line is the best fit, the grey area marks the uncertainty of the regression.

Only limited pre-eutrophication observations for N are available for rivers discharging into the Wadden Sea. Van Bennekom and Wetsteyn (1990) mention NO_3 and NH_4 values below 50 µM and DIN values clearly below 100 µM for the 19th century and the beginning of the 20th century. Given winter (January-February) Si values of 135-200 µM, it can thus be assumed that N was the main limiting nutrient for the spring bloom before eutrophication started during the 1950s.

With increasing eutrophication, a discontinuity emerged in the response of the phytoplankton to increased nutrient levels, when a N surplus was left over after the initial spring diatom bloom enabling blooms of phytoplankton not depending on Si. Given an average N/Si uptake ratio of 1, this discontinuity probably happened after riverine concentrations exceeded this N/Si ratio of 1. Thus, given the maximum winter Si concentrations of about 200 µM in Elbe, Ems and Schelde, we suggest DIN concentrations of 200 µM as a Safe Ecological Limit for rivers, noting that for the Weser, no data are available. Given the lower Si levels in the Rhine, we suggest a winter DIN concentration of 135 µM as a Safe Limit for the Rhine.

Table 1. Summary of reduction needs of DIN concentrations (µM) or TN loads (kT/year) for 1) transition from Si to N limited spring phytoplankton blooms and 2) seagrass recovery. Reference are the years 2010 – 2017.

River	Winter Si max	Winter DIN max	DIN Reduction to reach Si/N of 1	Reduction in annual TN loads for seagrass recovery Western Dutch Wadden Sea[5]	Reduction in annual TN loads for seagrass recovery Lower Saxonian Wadden Sea[5]
Rhine/ Meuse	135µM[1]	260µM	50%	38% (34-43%)	43% (39 – 46%)
Ems	200µM[2]	370µM	55%	NR	43% (39 – 46%)
Weser	200µM[3]	330µM	40%	NR	NR
Elbe	200µM[1]	290µM	30%	NR	NR

Notes
1) Based on Pätsch (2024)
2) Pätsch (2024), Helder & de Vries (1986)
3) No data available. We assume similar values as for the Elbe
4) NR: Not relevant
5) Reduction in total nitrogen (TN) loads of the Rhine/Meuse required for seagrass recovery was based on 12 scenarios to derive thresholds: 3-, 4- and 5-year averaged TN loads, and the year 2000 as start of the acceleration and 2003 as the start of the recovery setting through (van Katwijk et al., 2024).

Since 2010, Elbe maximum DIN concentrations are reached in February and amount to about 290 µM. This would imply a reduction of 30% assuming winter Si concentrations of 200 µM (see also Table 1). Weser maximum DIN concentrations since 2010 are reached in January and amount to about 330 µM. This would imply a reduction of about 40% assuming winter Si concentrations of 200 µM. Ems maximum DIN concentrations since 2010 are reached in January and amount to about 370 µM. This would imply a reduction of about 55% assuming winter Si concentrations of 200 µM. In the Rhine/Meuse, maximum DIN concentrations since 2010 are reached in February and amount to about 260 µM. This would imply a reduction of almost 50% assuming winter Si concentrations of 135 µM.

The winter Si concentrations suggested above are a first estimate. A more in-depth analysis of Si dynamics is needed to understand interannual differences in winter concentrations. Also, investigations are needed on particulate biogenic Si (dead frustules from diatoms) as this may be an additional source of dissolved Si.

4.5 Discussion

4.5.1 Loads versus concentrations in setting reduction goals

Two approaches were used to estimate reduction needs: 1) based on TN loads from winter to summer enabling seagrass return in the southern Wadden Sea and 2) based on Si/DIN ratios in winter. At first sight, these goals may seem to be unrelated since they target quite different aspects, but from a management point of view, N input has to be reduced forboth of them in a similar amount (30-50%). After all, the difference between the two approaches is riverine discharge (loads are concentrations multiplied by discharge). If we assume that long-term, yearly discharge levels remain the same, measures to reach the suggested reduction should be similar. However, it should be noted that if long-term changes in discharge occur, this will change the TN loads. Changes in frequencies and durations of climate extremes can also change riverine loads as e.g. droughts will increase the N retention (Schulz et al., 2023).

4.5.2 Relative importance of rivers impacting the Wadden Sea

The statistical analysis of the relation between eutrophication status and riverine TN loads only used the Rhine/Meuse (for the southern Wadden Sea) and the Weser/Elbe (for the northern Wadden Sea). These were chosen as they represent the largest riverine sources for the Wadden Sea. This, however, does not preclude that other rivers are also impacting the Wadden Sea. From a statistical point of view, it is extremely difficult to distinguish between the contribution of the different rivers because of correlation of the TN loads among the different rivers. Also, other sources such as atmospheric deposition and the import of organic matter must be taken into account.

Table 2. Relative magnitude of the TN loads of rivers impacting the Wadden Sea based on correlation with the annual TN loads (1979 -2022) from the Rhine and Meuse.

River	R^2	Intercept (kT N/year)	Slope (=Fraction of Rhine/Meuse TN Loads)
Schelde	0.62	3.5	0.083
Lake IJssel	0.82	10	0.17
Ems	0.57	5.1	0.045
Weser	0.80	1.2	0.19
Elbe	0.80	-15	0.48

Table 2 summarizes the correlations between the annual TN loads from the different rivers impacting the Wadden Sea with the TN loads from the Rhine/Meuse. It clearly shows the Rhine/Meuse as the largest riverine N source for the Wadden Sea and adjacent coastal zone of a similar magnitude as Schelde, IJsselmeer, Ems, Weser and Elbe together. The close correlation suggests that EU legislation and similar management measures have impacted the European river TN loads in a similar way.

We also correlated trends in N deposition into the Wadden Sea with Rhine/Meuse river loads. To correct for interannual differences in TN loads driven by discharge, we calculated flow-normalized concentrations (total annual TN load / total discharge). Again, good significant correlations were found for the Dutch ($r^2 = 0.58$), Lower Saxonian ($r^2 = 0.44$) and Schleswig-Holstein Wadden Sea ($r^2 = 0.41$) suggesting that EU legislation and similar management measures impacted riverine loads and atmospheric deposition in a similar way.

Model exercises show that the relative amounts of N from different sources can be traced back to their sources (e.g. Troost et al., 2013). This might open the possibility to specifically manage those sources that are the main drivers of Wadden Sea eutrophication. However, the main driver of Wadden Sea eutrophication is the import of N-containing organic matter produced in the adjacent coastal zone. This complicates the relative attribution of the different sources including atmospheric deposition as particles behave non-conservatively. For instance, particles can settle in different water masses, or nutrients released from particles end up in different water masses.

As a starting point, we therefore suggest that riverine TN loads and atmospheric sources must be reduced for all river basins impacting the southern Wadden Sea by a similar magnitude as suggested in Table 1 to enable permanent seagrass recovery. As a second step we suggest taking into account the winter DIN/Si ratios. This indicates that stronger measures are needed for the river basin of the Ems as here highest DIN/Si ratios are found.

4.6 Outlook

Within the NAPSEA project, two new approaches to assess Safe Ecological Limits for the Wadden Sea were developed based on seagrass dynamics and on phytoplankton limitation by dissolved Si. The focus was on reducing the N loads to the Wadden Sea as presently, N/P-ratios in winter are extremely high, high N/P ratios have a negative effect on food quality and N is ultimately limiting phytoplankton growth. In the other case studies, the role of P is discussed in more detail as P is an important nutrient potentially limiting phytoplankton growth in freshwater systems.

At present, not enough data are available to test the impact of reduced nutrient loads and changed nutrient ratios (N/P ratios) on the entire food web (compare Philippart et al., 2007). Such information will be necessary to evaluate the interacting effects of reduced primary production but also changed N/P ratios on food quality availability and their impact on the carrying capacity of the Wadden Sea.

5. SAFE ECOLOGICAL LIMITS FOR THE ELBE ESTUARY

5.1 Introduction

Both the riverine part of the Elbe and its estuary have been subject to many man-made changes including pollution and eutrophication and morphological changes like weirs, diking and dredging. In this chapter, we will focus on eutrophication and the consequences of large phytoplankton blooms in the riverine part of the Elbe for the Hamburg port area in the upper part of the Elbe estuary, where nowadays very low oxygen levels are reached down to levels which impact fish populations. We expect that mitigating this issue will also bring the Elbe loads towards the Wadden sea withing Safe Ecological Limits. In addition, N and P reductions in the headwater tributaries of the Elbe are needed to solve the oxygen problems in the estuary, although we do not consider the local safe ecological limits within these headwaters in this chapter.

5.2 Area description

The Elbe is the largest river in Northern Germany (Figure 13). It originates in the Giant Mountains region in the north of the Czech Republic and has a catchment area of 148268 km^2 and a total length of 1094 km (IKSE, 2005). The free-flowing part of the Elbe ends at Elbe-km 586 at the weir of Geesthacht, which was built between 1957 - 1960. The long-term mean annual discharge at Neu Darchau (1903 – 2019) is 694 m^3/s (data: FGG Elbe).

Figure 13. The Elbe catchment. https://en.wikipedia.org/wiki/File:Elbe_basin.png

Figure 14Figure 14

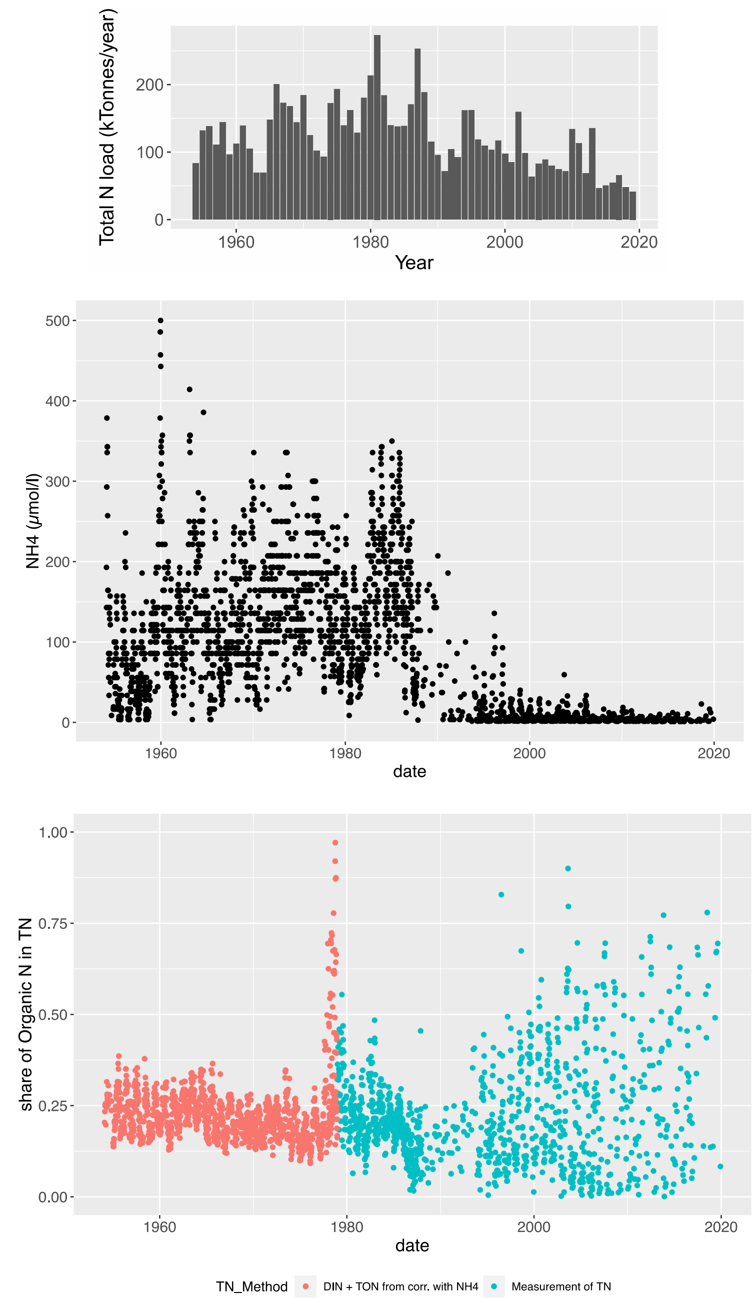


Figure 14. Changes in annual total nitrogen loads (upper panel), the NH_4^+ concentrations (middle panel) and share of organic N in the Elbe near Geesthacht (lower panel). For details on the estimation of total organic nitrogen (TON) before 1979: see van Katwijk et al., 2024. Data: data portal of the FGG Elbe (2020). After 2016, the no measurements were carried out at the Geesthacht stations and only data from the transects through the estuary (~4/year) are available.

5.3 Eutrophication history

Large changes in water quality have been documented in the Elbe. Eutrophication already started in the 1950s and culminated during the 1980s (e.g. van Beusekom et al., 2019; van Katwijk et al., 2024). In the Elbe estuary, O_2 concentrations during the 1980s were low due to high loads of organic matter and NH_4^+ (e.g. Kerner, 2000, compare Figure 14). From 1990, political and economic changes in Czechia and former Eastern Germany allowed for the quick adoption of measures leading to the elimination of the most critical sources of pollution (e.g. Adams et al. 1996; Langhammer, 2010; Rewrie et al., 2023). For instance, a rapid decline in organic matter and NH_4^+ loading (Figure 14) was observed. Improved water quality enabled the development of large summer phytoplankton blooms in the riverine part of the Elbe since the 1990s (e.g. Rewrie et al, 2023) with lowest levels during winter (week 45 – week 10) and highest levels in summer (week 20 – week 30) with averages of around 90 - 130 µg Chl a/l and maximum values up to 300 µg Chl a/l (Figure 15). This is reflected by organic N compounds increasing to reach levels up to >75% in summer in contrast to values <5% in winter (compare the large variability in Figure 15 since about 2000 with lowest values in winter and highest in summer). During spring and summer, phytoplankton levels constantly increase in the Elbe from Schmilka, where the river enters Germany, to the end of the riverine part (Weir at Geesthacht 30 km upstream of Hamburg; see Figure 16 left). Factors that support the riverine phytoplankton blooms are low water depth in the river (better light conditions) and long residence times which both occur during low discharge events (e.g. Scharfe et al., 2009; Kamjunke et al., 2021).

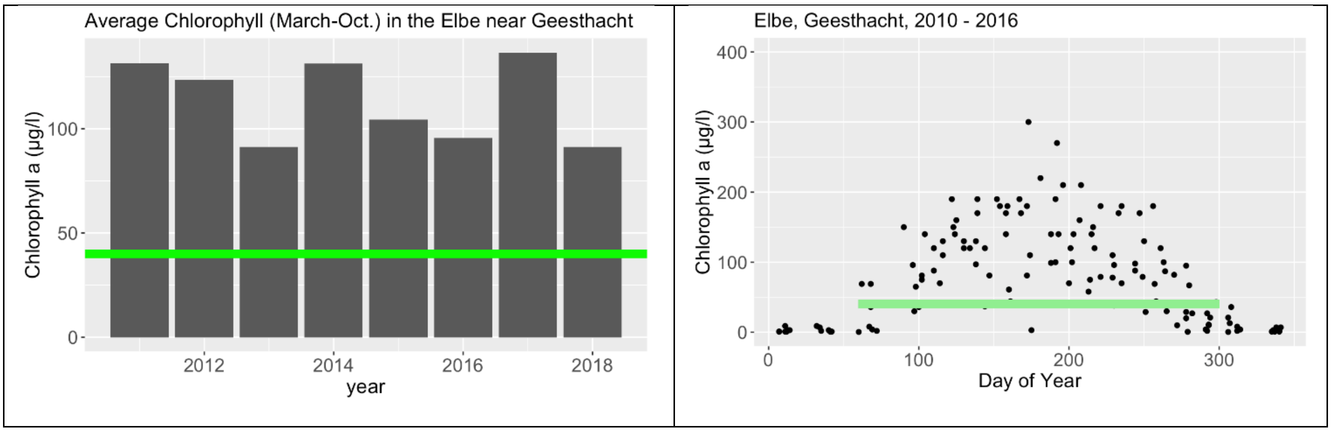


Figure 15. Left: Average Chlorophyll levels (March-October; 2010 - 2016). The green line shows the environmental goals for the WFD (OGewV). Right: Seasonal dynamics of phytoplankton (as Chlorophyll a) between 2010 and 2016 at the end of the river Elbe near the Weir at Geesthacht. The green line shows the environmental goals for the WFD (OGewV).

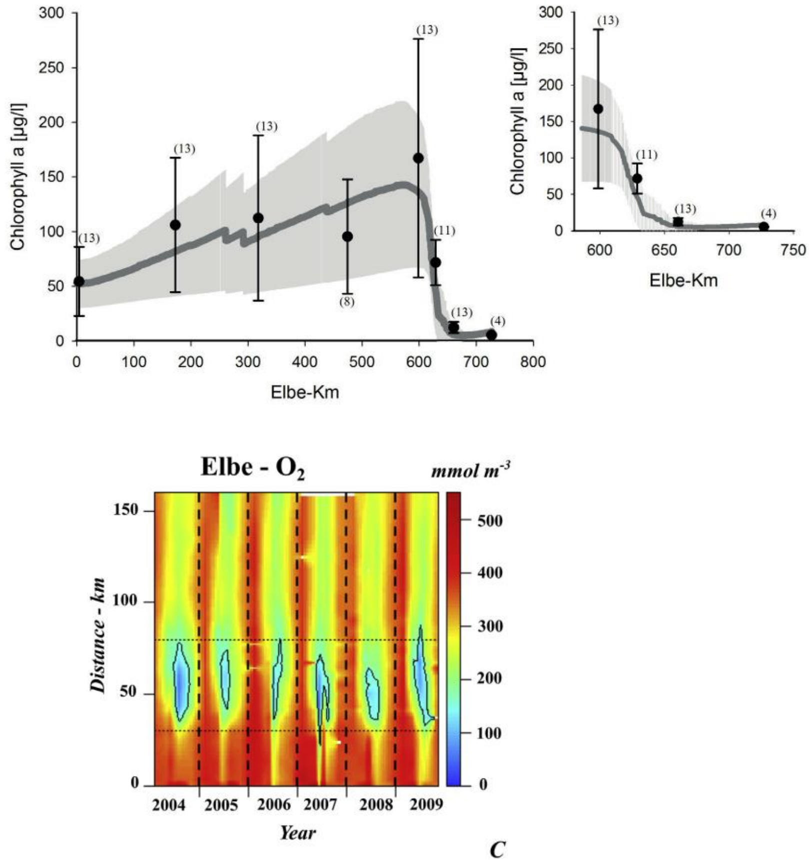


Figure 16. Left: Longitudinal profile of measured (black dots) and modelled (dark grey line) seasonal means of chlorophyll a (µg/l) from Schmilka (km 0) to Cuxhaven (km 727) for May to October 2006. Number of measurements is given in brackets. Standard deviations of measurements reflected by bars and of modelled data (n = 184) by the light grey area (from Schöl et al., 2014).

In the upper estuary, water depth are around 4 m but increase sharply in the Hamburg port area to levels of about 18 m due to dredging. Once the Elbe phytoplankton blooms enter the deepened part of the estuary, grazing and light limitation decimate the phytoplankton standing stocks. High degradation rates lead to low oxygen concentrations in the Hamburg port area clearly below the level of 219 µmol/l (7 mg/l) as set by German law (Schöl et al., 2014; Geerts et al., 2017, see Figure 16 right).

5.4 Safe Ecological Limits

As a first safe ecological limit for the Elbe estuary, we propose to use the value of 7 mg O_2/l (219 µmol/l) as set by German law (OGewV). Based on model calculations we will suggest reduction needs to keep oxygen levels in the estuary within Safe Ecological Limits. As a second approach, we use the environmental goals (OGewV) for phytoplankton biomass aiming at a mean of 40 µg Chl a/l for the period March – October. The reduction needs will be discussed in relation to Safe Ecological Limits for the Wadden Sea.

5.5 Modelling the O_2 dynamics in the Hamburg port

5.5.1 Methods

To model the oxygen dynamics in the Elbe estuary, we use the existing SCHISM-ECOSMO model for the Elbe estuary. It uses an unstructured model grid covering the Elbe estuary with a computational mesh with a resolution between 30 m in the port of Hamburg and 500 m at the model boundary in the German Bight. As hydrodynamical core it uses the Semi-implicit Cross-scale Hydroscience Integrated System Model (SCHISM, Zhang et al., 2016) as hydrodynamical core, which is coupled with an ecological model, the ECOlogical System Model (ECOSMO, Yumruktepe, 2022).

With the help of SCHISM, researchers have successfully modelled both idealized and realistic estuarine domains tackling research questions in the areas of hydrodynamics, sediment dynamics and ecology (Pein et al., 2021a, b; Stanev et al., 2019; Ye et al., 2018). The SCHISM solves the Reynolds-averaged Navier–Stokes equations on unstructured meshes assuming hydrostatic conditions. The model predicts water elevation, horizontal currents, vertical exchange and tracer transport.

The biogeochemical model ECOSMO has successfully been applied to simulate lower trophic level biogeochemical dynamics in the North Sea (Schrum et al., 2006) and Baltic Sea (Daewel and Schrum, 2013). In the coupled physical-biogeochemical model, the local concentration of an ecological tracer changes according to

$$C_t + (\mathbf{v}\nabla)C + (w_d)C_z = (A_v C_z) + R_C, \quad (1)$$

where *C* represents an ecological state variable, **v** refers to the 3D velocity field, w_d is a constant settling velocity and A_v is the turbulent diffusion coefficient.

In ECOSMO, the pelagic prognostic state variables comprise four nutrients, three functional groups of primary producers, herbivorous and omnivorous zooplankton, detritus, opal and dissolved organic matter. The term in Eq. (1) represents the biogeochemical sources and sinks modifying the respective state variable concentration *C*.

In the Elbe estuary, the coupled modelling framework SCHISM-ECOSMO can simulate the physical-ecological dynamics on tidal to seasonal to intra-annual scales (Pein et al., 2021a). With the help of this model, it was demonstrated that tidal pumping leads to trapping of organic particulate matter in the port region that is remineralised in the same area, whereas summer heating leads to enhanced process rates and water column stratification. These processes exacerbate the oxygen depletion in the deep channels and basins in the port region (Pein et al., 2021a).

5.5.2 Response of the O2 dynamics to reduced organic matter loads

Here we used the validated model configuration for the year 2012, to investigate the effect of nitrogen load reduction at the tidal weir on oxygen levels in the port of Hamburg (Figure 17). The scenario approach was as follows: Two historic scenarios adopted the ratio of contemporary to historic N loads reported by Serna et al., 2010 for the 1960s and pre-industrial times, respectively. These resulted in our historic scenarios E1960 and E1860 in which both inorganic and organic nitrogen loads were modified according to the findings of Serna et al., 2010. The historic scenarios were complemented by two idealised scenarios reducing the total nitrogen loads by 50% and 75% respectively resulting in our scenarios E-50 and E-25.

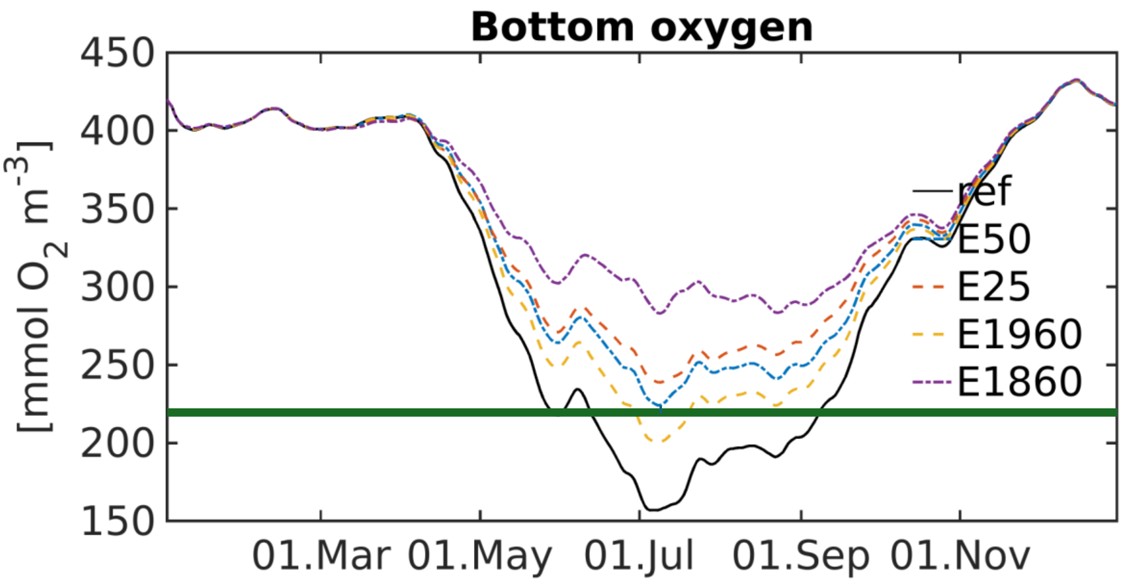


Figure 17. Reduction scenarios for the effect of reductions in the organic matter loading from the riverine part of the Elbe on the oxygen dynamics in the Hamburg port area. The green line marks the level of 7 mg O_2/l (219 µmol O_2/l) specified as Good Environmental Status in German law (OGewV). Note that the y-axis starts at 150 mmol O_2/m^{-3}.

5.6 Reductions needed to stay within Safe Ecological Limits

At present, O_2 levels in the Elbe reach very low levels during summer being clearly below Safe Ecological Limits of 219 µmol O2/l or 7 mg O2/l as indicated by the green line in Figure 17. We estimated the response of the O_2 dynamics to organic matter loads by stepwise reducing the load into the estuary (Figure 17). As reference, we used the year 2012. A 50% reduction in organic matter loading leads to O_2 levels just above 7 mg O_2/l. From this we estimate a reduction need of the organic matter loading by about 40 – 45%. This range is lower than the reduction needed to bring phytoplankton biomass within Safe Ecological Limits as proposed by the WFD/ OGewV: At present (2010-2016) average phytoplankton biomass at the end of the riverine stretch of the Elbe near Geesthacht (March – October) is 109 µg Chl a /l. To reach Save Ecological Limits of 40 µg Chl a/l, a reduction of 63% is needed, which is clearly higher than the reductions of 40-45% needed to keep O_2 levels in the upper estuary at levels of 7 mg O_2/l (219 µmol/l).

All reduction levels are relative to the year 2012. The mentioned reduction requirements however are assumed to also apply to the present conditions as since 2010 no clear trends in riverine flow-normalized concentrations have occurred (except for discharge driven changes; see previous chapter on Wadden Sea Safe Ecological Limits).

5.7 Discussion

The reductions in summer organic matter loads from the river Elbe to keep O_2 levels in the upper estuary above 7 mg /l O_2 of 40 – 45% are lower than the reduction needs of about 60% to keep mean phytoplankton biomass (March-October) below 40 µg Chl a/l. An important question is, which factors determine phytoplankton bloom size. The major factor presently determining the phytoplankton dynamics is river discharge (Kamjunke et al., 2021) and Si availability (Scharfe et al., 2009). Apart from Si availability that limits diatom growth, P can become limiting during low discharge events (Kamjunke et al., 2021). Measures to reduce riverine N-loads have led to decreasing annual TN loads (Figure 14) and in summer to minimum concentrations decreasing from around 300 µmol DIN /l during the early 1990s to less than 50 µmol DIN /l since 2003 (Schulz et al., 2023). Recently, indications of a potential N limitation were observed during periods of extreme low discharges (Schulz et al., 2023). It remains an open question, whether future N reductions may lead to an N limited phytoplankton bloom.

5.7.1 The role of Si

Scharfe et al. (2009) pointed at the role of Si in Elbe phytoplankton dynamics. Figure 18 shows the relation between DIN and Si. Both N and Si are taken up with a ratio of about 1:1. A similar ratio was observed in the Wadden Sea (see Chapter on Wadden Sea Safe Ecological Limits). Decreasing riverine N loads since the 1990s led to increasingly lower amounts of N left over after Si reached limiting concentrations.

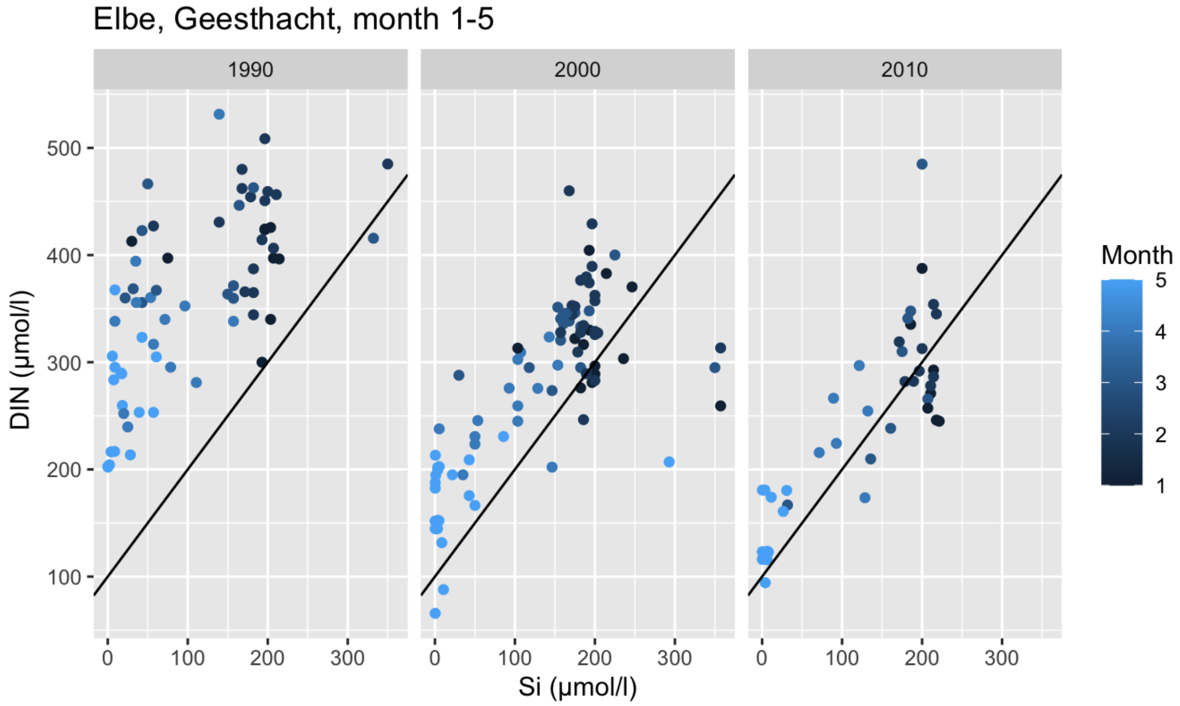
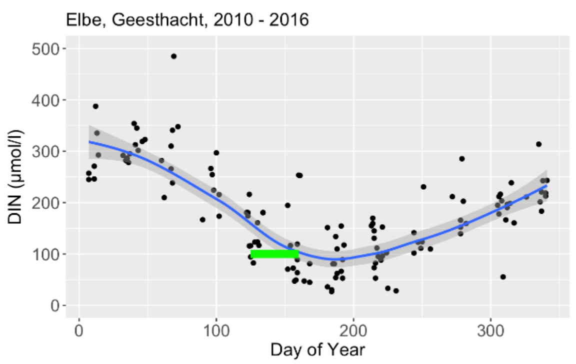


Figure 18. The relation between Si and DIN during the spring blooms in the 1990s, 2000s and 2010s. The black line shows the 1:1 ratio between DIN and Si at a y-axis intercept of 100 µmol DIN/l. Data: FGG Elbe. The colours indicate the month of the year (dark = January, light blue = May)

5.7.2 The role of N and P

Kamjunke et al. (2021) highlighted that mostly N and P are not limiting phytoplankton dynamics. However, during low discharge conditions, causing low water levels and long residence times, large phytoplankton blooms developed during which P can become the main limiting nutrient in the lower reaches of the Elbe during low discharge events.

An important question is whether also N can become a limiting nutrient in the future. This is relevant as N is the main element limiting phytoplankton dynamics in the Wadden Sea from June/July onward (Chapter on Wadden Sea Safe Ecological Limits). Schulz et al. (2023) showed that since 2018 during low discharge conditions, extreme low NO_3^- values were reached suggesting that N can become limiting if N levels are further reduced. First indications on the potential consequences of N limitation were observed in 2022: Already in early June 2022 NO_3^- levels of about 3 µmol/l and DIN levels of about 5 µmol/l were observed (Lempges, 2023). During the following weeks, the phytoplankton bloom collapsed leading to low O_2 levels of around 1 mg/l in the riverine part of the estuary before the water entered the deepened port area (data: Hygiene Institute, Hamburg).



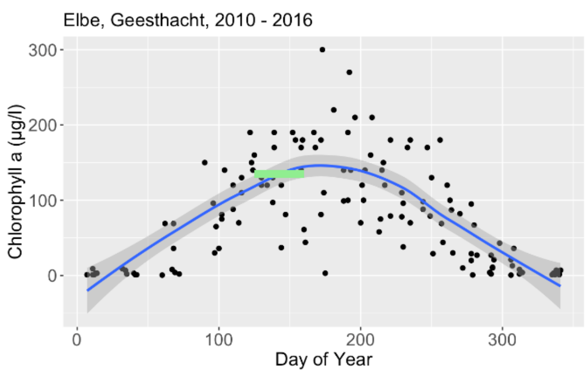


Figure 19. Recent seasonal cycles of DIN (left) and Chlorophyll a (right). The green bars in both graphs indicate the period when phytoplankton would have removed all DIN starting at winter DIN levels of 200 µmol/l. This level was chosen to reflect the situation at winter DIN/Si levels of 1 that were suggested to enable N-limited diatom blooms in the northern Wadden Sea. The blue line in both graphs show the smoothed seasonal dynamics.

5.7.3 How to reduce Chlorophyll and organic matter levels in the Elbe River

In the Chapter on Safe Ecological Limits for the Wadden Sea, a maximum winter level of 200 µmol DIN/l was suggested. We have estimated how Chlorophyll levels in the Elbe would respond when the winterly DIN levels were brought down from around 300 as presently observed to the 200 µmol DIN/l mentioned above. Note that in winter DIN is dominated by NO_3^-.

The green bar in Figure 19 (left) indicates the level of 100 µmol/l. If further management decisions would reduce the *winter* concentration to about 200 µmol DIN/l, limiting DIN levels could expected to be reached during the period indicated by the green bar. However, when DIN would reach these limiting levels, a Chlorophyll a level of about 140 µg /l can be expected (see the green bar in the right frame in Figure 19), which is still clearly above the goals set for the WFD (compare Figure 15).

It is at present unclear how the phytoplankton blooms will further develop after N limiting conditions will be reached. In contrast to the summer situation in the North Sea and Wadden Sea nutrients are constantly entering the river system thereby potentially sustaining the summer bloom. Hence, it is questionable that a 30% reduction in N loads will actually lead to levels of 40 µg/l Chl a and O_2 levels as envisioned in the framework of the WFD and demanded by the OGewV. Here, further research is clearly necessary.

Given that P is at present the main limiting element in the riverine part of the Elbe, some people may draw the conclusion that riverine P loads must be further reduced. From an ecologists point of view, however, a further reduction P without a concomitant reduction in N should be avoided as this will further increase the N/P ratios with negative consequences for the aquatic food web as the plankton food quality is negatively influenced by high N/P ratios (Malzahn et al., 2007).

5.7.4 Morphological adaptions

Alternative approaches to increase the resilience of the Hamburg port area to oxygen deficits include morphological adaptations as for example, the reconnection of a historic major river branch to the tidal system as proposed by Pein et al. (2024). Their model simulations demonstrate that proposed measures potentially reduce the siltation of the upper estuary and thus the need for extensive and costly maintenance dredging. Furthermore, the simulated measures also mitigate the consequences of eutrophication, such as the low oxygen content in the navigation channel.

5.8 Conclusions

To reach Safe Ecological Limits regarding Elbe phytoplankton blooms, even stronger reduction measures are needed than the ~30% reduction in DIN concentrations suggested to reach Wadden Sea Safe Ecological Limits. We suggest that a reduction of both N and P is needed. But the amount of reduction needed cannot be derived yet from the available data. Dedicated models and experiments are needed to estimate the extent of N and P reductions needed to limit phytoplankton biomass build-up the Elbe.

6. SAFE ECOLOGICAL LIMITS FOR THE RHINE BASIN

6.1 Introduction and area description

For the Rhine case study, we focused on the relation between the main river and the many tributaries in relation to the safe ecological limits. The Rhine basin is one of Europe's most significant and most intensively used river systems. It covers an area of approximately 185,000 km^2 and flows through several countries, including Switzerland, Germany, France, and the Netherlands, before draining into the North Sea/ Wadden Sea. Its catchment area is home to 58 million people of which 30 million people depend on the Rhine and its banks as a drinking water source (Plum & Schulte-Wülwer-Leidig 2014). Half of the Rhine`s catchment area is used for agriculture. The Rhine itself is intensively used as a traffic axis contributing to the development of heavy industries and chemical industries (https://www.iksr.org/en/topics/uses/industry).The Rhine origins in the Alps and drains into the North Sea/ Wadden Sea. It´s alpine and high parts in Switzerland cover around 20% of the total catchment area and has a mean annual discharge of 1060 m^3/s (gauging station Basel, Switzerland, Belz et al. 2007). It is characterized by a nival discharge regime with maximum discharge in June and July, although dampened by the large Lake Constance that the Rhine is flowing through. Lake Constance has a high surface area (539 km^2), large volume (48.53 km^3) and long residence time of water (4.5 years) (World Lake Database, https://wldb.ilec.or.jp/Lake/EUR-33).

In contrast to the upstream part, the Rhine at Lobith, at the German-Dutch border has a mean discharge of 2220 m^3/s and a mixed rain-nival regime with the dominant peak in February. Relative to the observation at Lobith, the alpine and high parts of the Rhine in Switzerland deliver around 48% of the total discharge. The largest tributaries of the Rhine are the Main, Moselle, Neckar, Aare and Ruhr (Belz et al. 2007). While the main stem of the Rhine is not dammed downstream of Switzerland, its tributaries such as Main and Moselle are not free flowing and have weirs and locks to make them navigable.

From the large number of inhabitants and connected wastewater inputs, the intense industry along the Rhine and from the agricultural usage of its catchment areas, the Rhine and its tributaries faced a long history of severe river pollution. While water quality greatly improved since the 1980s, the Rhine still faces challenges in chemical and biological water quality and still is a major source for nutrients and related eutrophication impacts in the Wadden Sea (van Katwijk et al. 2024). Below, we summarize ecological challenges in the Rhine and its tributaries in the frame of the NAPSEA project and discuss nutrient concentrations and loads as safe ecological limits.

6.2 Eutrophication in the Rhine basin

Within the NAPSEA project, we focus on the N and P concentrations and loads in the terrestrial, estuary and marine systems. Both N and P had been a much larger problem in the main stem of the Rhine in the past, especially in the 1970s and 1980s. Since then, much has been done to improve nutrient concentrations and loads in the river (Plum & Schulte-Wülwer-Leidig 2014). This is especially true for the regulation of point source inputs, that e.g. for N was reduced by a factor of 3.6 between the mid-1980s and the 2010s (according to our data, see Figure 20). At the same time, diffuse N inputs reduced by a factor of 1.8 due to measures in the agricultural sector and reduced atmospheric inputs. For P inputs, the ban of phosphates in textile detergents in the 1980s and the enhancement of wastewater treatment plants greatly reduced point source inputs into the Rhine. For the Rhine main stem, both resulting N and resulting P concentrations consequently improved (Figure 21, Figure 22).

Appendix A also gives the water quality status for N and P for the Dutch part of the Rhine. The 3-year summer average concentration target for Total-N of 2,4 mg/l is met since around 2010, except in 2019 and 2022. The target for Total-P of 0,14 mg/l has been permanently met since 2012. The trend assessment for Total-N and Total-P (Appendix B) shows significant downward trends between 1990 and 2020. Since around 2008, this trend flattens especially for Total-N. Similar to the German part of the Rhine basin, more severe eutrophication issues occur in the tributaries. In the Dutch part of the Rhine river basin district, nearly 50% of the water bodies had a Moderate or worse status for phosphorus and about 30% had Moderate or worse status for nitrogen in 2021 (IenW, 2022).

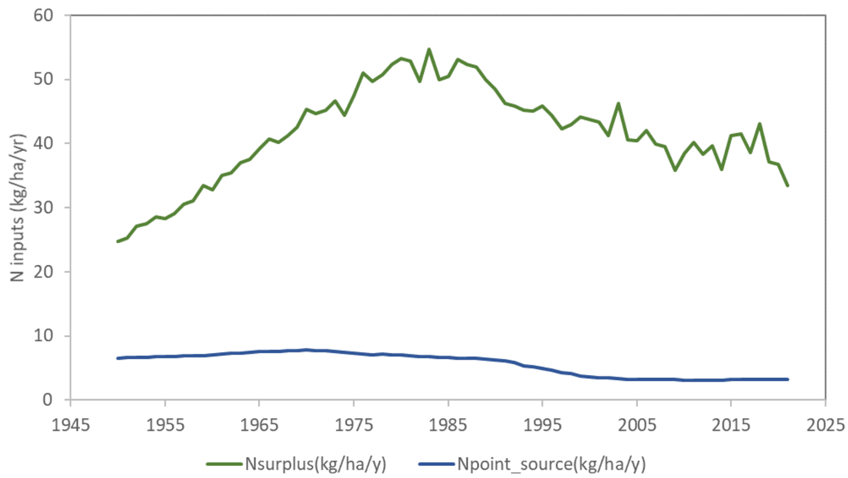


Figure 20. Nitrogen inputs from diffuse (Nsurplus) and wastewater point sources to the Rhine basin as quantified in the NAPSEA project, based on Batool et al. (2022), Häußermann et al. (2019), Büttner (2020) and Sarrazin et al. (2024).

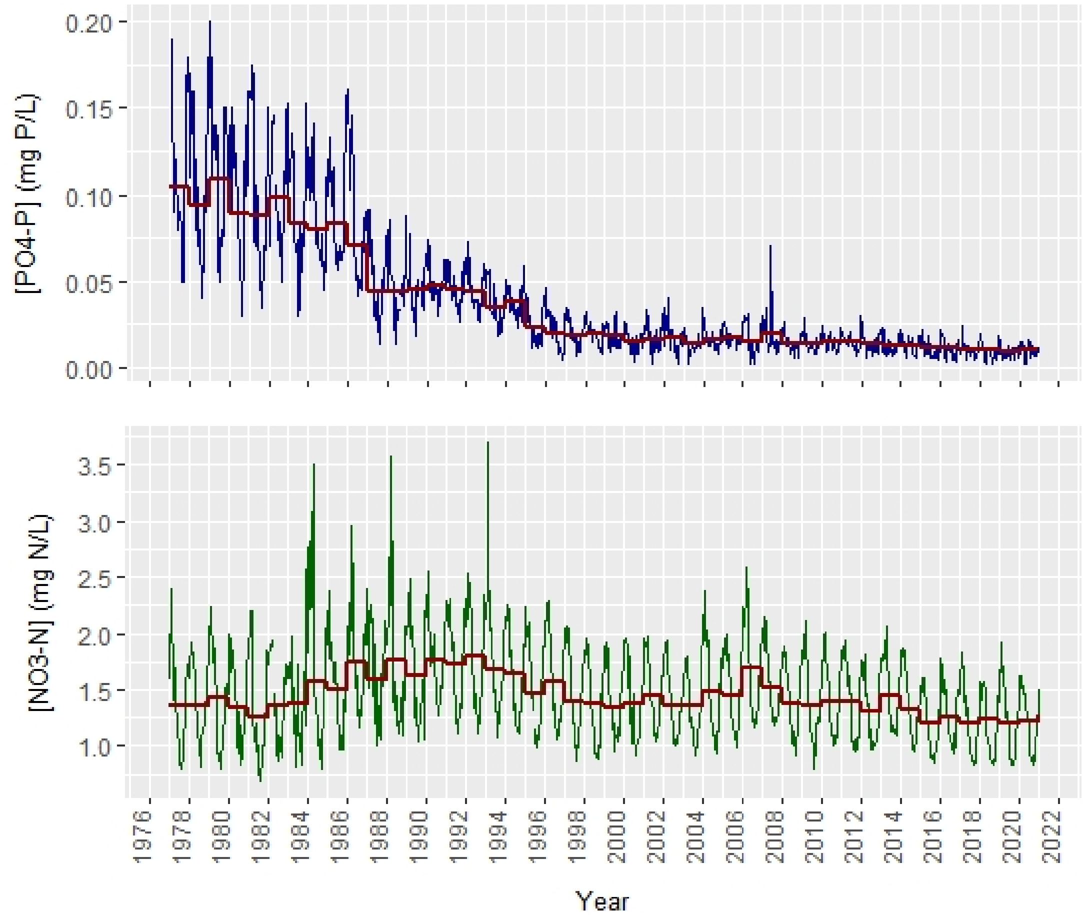


Figure 21. Development over time of concentration of orthophosphate-P (above) and nitrate concentrations (below) in the Rhine at Basel. The red lines indicate the annual average concentrations. Data: NADUF; the exact sampling location changed in 1995 (<1995: Village Neuf and ≥ 1995: Weil), https://www.bafu.admin.ch/bafu/en/home/topics/water/info-specialists/state-of-waterbodies/state-of-watercourses/water-quality-in-watercourses/nutrients-in-water.html.

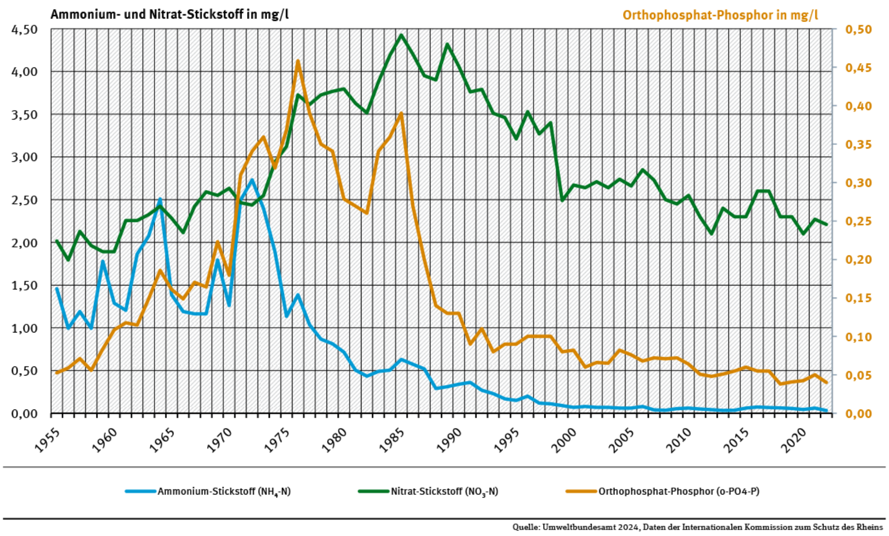


Figure 22. Temporal dynamics of concentration of orthophosphate-P (orange), ammonia (blue) and nitrate (green) concentrations in the Rhine at Lobith at the German-Dutch border. Data and figure from Umweltbundesamt (https://www.umweltbundesamt.de/daten/umweltzustand-trends/wasser/fliessgewaesser/naehrstoffkonzentrationen-in-fliessgewaessern).

At the Dutch-German border, o-PO4 concentrations reduced from 0.36 mg/L in the mid-1980s to 0.05 mg/L (by a factor of 7). At the same time NO_3-N concentrations reduced from 4.2 mg/L to 2.35 mg/L (factor 1.8 – identical with the change in diffuse N inputs). Trends in N and P concentrations after 2010 are neglectable.

6.2.1 Nitrate status

The German classification of the good NO_3-N status in streams was using the 90th percentile of concentrations measured within a year (LAWA 1998). The good (water quality class II) status is reached at stations that are below a 90th percentile of 2.5 mg/L. For this assessment, we translated this value into a value that is easier to work with and is directly modelled: mean annual concentration of NO_3-N. For all our available data that is also used for model calibration (Ebeling et al. 2022) we therefore derived annual 90th percentile concentrations and fitted a linear relation to the mean flow-weighted NO_3-N concentration of the respective years. The result of the regression analysis indicated that a mean annual NO_3-N concentration of 1.9 mg/L refers to a 90th percentile of 2.5 mg/L (R^2=0.9). This value of 1.9 mg/L is close to value of 2.0 mg/L used in the evaluation of Grizetti et al. (2012) and Vigiak et al. (2023) to assess NO_3-N concentration status across Europe (mean NO_3-N concentration considered low when <2.0 mg/L therein). This value also well compares to threshold suggested by an assessment of macrophytes and phytobenthos status under differing nutrient levels provided by Poikane et al (2021). Here, 1.0–2.5 mg/L TN is suggested as a threshold for reaching good ecological status. However, note that newest legislation in Germany (Oberflächenwasserverordnung OGweV from 2016) does no longer state thresholds defining NO_3-N status in rivers except the environmental quality standard for drinking water of 50 mg/L (11.3 mg/L NO_3-N) since NO_3-N is not seen as a limiting nutrient for algal growth and thus not relevant for mitigating eutrophication.

In the Rhine main stem, 13 stations with sufficient data are in our concentration database used in NAPSEA. Following the proposed annual threshold value, 10 out of 13 stations meet the good status with NO_3-N concentrations below 1.9 mg/L. The average concentration for all stations was 1.69 mg/L.

However, when looking at the tributaries to the Rhine, the picture differs from the main stem of the Rhine. Within NAPSEA, we are evaluating (and later modelling) 80 (partially nested) sub-catchments of the Rhine for NO_3-N concentrations with catchment areas between 22 and 98315 km^2 (mean 1942 km^2). Within these catchments, 71 out of 80 sub-catchments (89%) fail to reach the good nitrate state when looking at the observed flow weighted average concentrations between 2010 and 2020. The average concentration across all the sub-catchments in this period is 3.55 mg/L.

We would also like to point to the nitrate state in groundwater that is, in Germany, especially along the Rhine and in the lower Rhine area often above the drinking water limit of 50 mg/L Nitrate (11.3 mg N/L) (Figure 23).

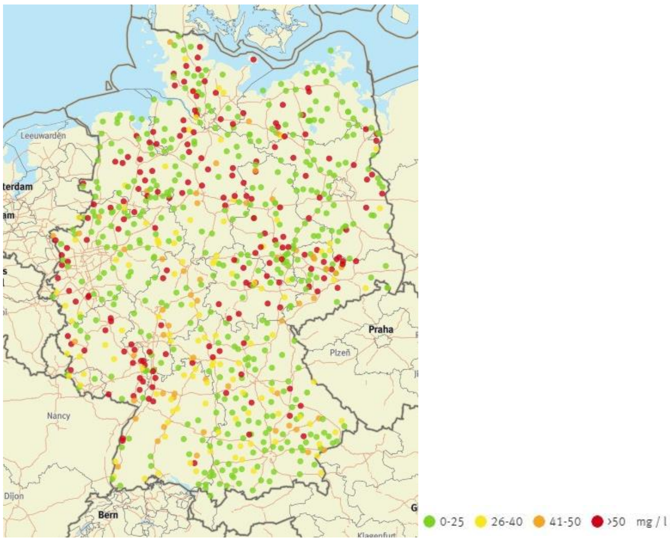


Figure 23. German groundwater NO_3-N concentrations in 2020-2022 (data and figure from Umweltbundesamt, https://www.umweltbundesamt.de/umweltatlas/reaktiver-stickstoff/wirkungen/grundwasser/wo-treten-probleme-nitrat-in-grundwasser-auf).

6.2.2 Phosphorus status

German classification of the good ortho-phosphate (o-PO_4) status was using the 90th percentile of concentrations measured in a year (LAWA 1998) similar to the assessment of NO_3-N described above. Here, the good (water quality class II) status is reached at stations that are below a 90th percentile of 0.1 mg/L. In the data used here (see also nitrate state section above) this value refers to a mean annual concentration of 0.055 mg/L (R^2=0.98). This value is at the lower end of the range suggested by Poikane et al. (2021) based on an extensive evaluation of macrophytes and phytobenthos state in different river types of central Europe (0.021-0.90 mg/L). The German surface water ordinance (OvGewV, 2016) defines the requirement for a 'very good' ecological state and the highest ecological potential of 0.02 mg/L mean annual concentrations of o-PO_4. A 'good' status is reached with mean annual o-PO_4 concentrations of 0.07 mg/L for most stream types (ranging between 0.05 mg/L for alpine and pre-alpine streams to 0.2 mg/L for marsh waters in the coastal lowlands).

Within NAPSEA, we are evaluating 386 sub-catchments with a mean catchment area of 1289 km^2 that have sufficient from 2010 onwards (>60 observations). Within that dataset, 84 stations (21%) reach the good o-PO_4 status or better (below 0.055 mg/L mean annual concentration). Across all stations the mean concentration after 2010 is 0.114 mg/L. Consequently, the PO_4 status across the tributaries to the River Rhine is slightly better than the nitrate state but still 78% of stations fail the good P state. In the Rhine main stem, 13 stations with sufficient data are in our concentration database used in NAPSEA. Here, 11 out of 13 stations meet the good status with PO_4 concentrations below 0.055 mg/L. Across all the 13 stations in the Rhine main stem the average concentration was 0.031 mg/L.

6.2.3 Chlorophyll-a status and invasive filter feeders

Here, eutrophication status based on observed chlorophyll-a (Chl-a) observations in Germany is evaluated. A threshold for the occurrence of algal blooms is defined based on Dodds et al. (1998) as Chl-a concentration above 30 µg/L. Specifically, data is evaluated after the year 2000 with measurements spanning the spring, summer and autumn season. First results (Hubig et al., 2024, EGU, Figure 24) indicate that the Rhine (42 stations) has a low probability of algal blooms (algal blooms only occur in less than 75% of the observations). Chl-a concentrations were found to be systematically higher in the lower order sub-catchments compared to the higher orders (main stem). This is different in comparison to the Elbe basin, where the highest probability of algal blooms occurred in the higher orders (main stem).

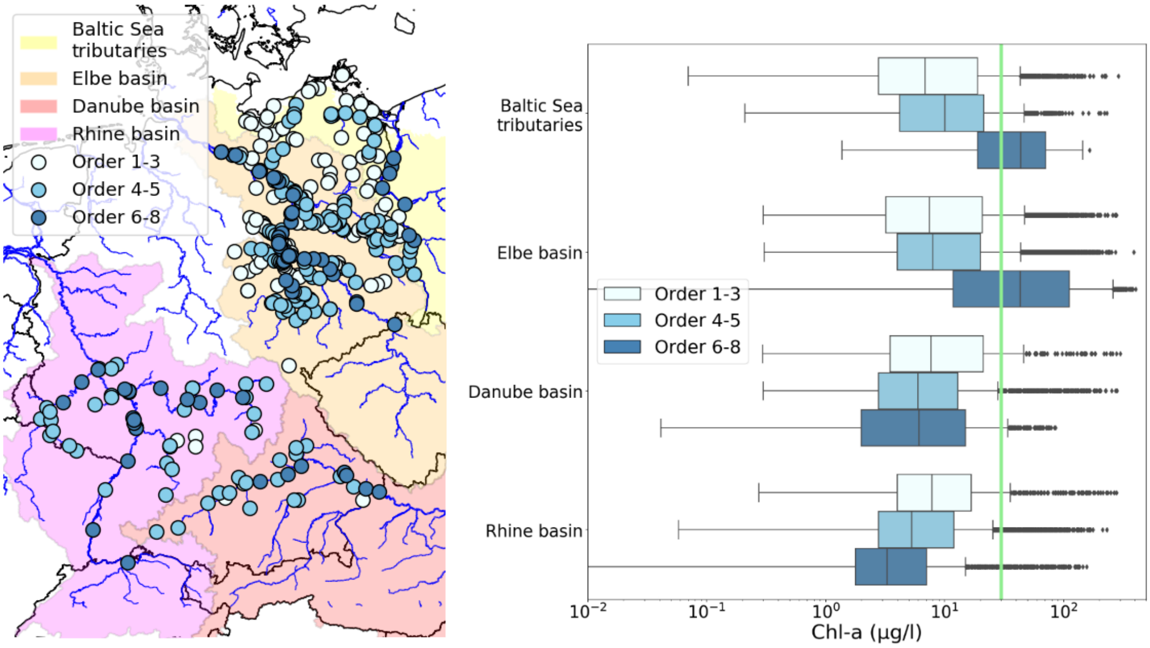


Figure 24. Evaluation of Chl-a concentration in Germany after 2000 (Hubig et al. 2024, EGU). The left plot shows the distribution of evaluated stations in the Germany. The right plot shows the observed Chl-a concentrations as boxplots on a logarithmic scale and occurrence of algal blooms (>30 µg/L Ch-a, Dodds et al. 1998).

This decrease of Chl-a can be attributed to the occurrence of filter feeders in the main stem of the Rhine. More specifically, two invasive mussel species (quagga and zebra mussel) have been spreading in the Rhine in the last decades, likely being introduced by the intense ship navigation (Leuven et al. 2009, Karatayev & Burlakova 2022). These species can occur in large masses and are very effective in biofiltration (Karatayev & Burlakova

2022). In the case of the Rhine main stem, and unlike the Elbe river, this biofiltration subdues the algal growth that would likely leading to Chl-a concentrations comparable or higher than in the tributaries (Hardenbicker et al., 2016; see also deliverable D3.2).

6.3 Safe Ecological Limits

We consider mean annual concentrations of 1.9 mg/L NO_3 and 0.055 mg/L PO_4 in river waters to be justifiable thresholds for good nutrient status and can be used as safe ecological limits in the Rhine basin. These values are adapted from the established LAWA classification (LAWA 1998) and are confirmed by newer studies (Poikane et al. 2021, Vigiak et al. 2023).

As discussed above, in the last years (2010-2021), these thresholds are largely met in the main stem of the river Rhine. However, within the Rhine *basin* only a small proportion of the river network including tributaries and lower order streams comply with this classification. For NO_3-N 89% and for o-PO_4 78% of the stations outside the main stem fail to reach a good water quality status. The main stem of the Rhine has a good status contrasting its tributaries because of several reasons: (a) there is a large dilution with water of good quality coming from the Alps with a nival discharge regime (main source is summer is snowmelt), (b) the upper Rhine flows through Lake Constance with a long residence time and large area, acting as a nutrient sink , (c) the Rhine is populated by invasive mussel species that are effectively filtering algae from the water column. We note that future climate change may interfere with these processes. Further warming will increase winter discharge (due to more winter rain) and decrease summer discharge (due to less accumulated snow) for the upper Rhine stemming from the Alps (Rottler et al. 2021). This can shift the seasonal dilution potential in the main stem.

In summary, the Rhine basin, as the sum of all parts of the river network, needs a further reduction of nutrient (N and P) inputs to ensure good water quality and to stay within safe ecological limits.

7. SAFE ECOLOGICAL LIMITS FOR THE HUNZE CASE

7.1 Introduction

The Hunze is a 250 km² catchment in the north of The Netherlands. Nutrient losses from the catchment have a direct impact on the ecology of the receiving lake Zuidlaardermeer. Further downstream, the drainage from the Hunze reaches the Wadden Sea through the Lauwersmeer estuary.

Compared to the Rhine and Elbe cases, the Hunze case focuses on a much smaller spatial scale. This enables more specific explorations on mitigation options to bring the nutrient losses within the safe ecological limits of the stream itself, the receiving Zuidlaardermeer and ultimately the Wadden Sea.

7.2 Area description

The Hunze is part of the Hunze and Aa waterboard, one of the largest waterboards in the northern Dutch provinces. The Hunze and Aa waterboard covers an area of about 2130 km² with about 3525 km of mainly artificial channels and ditches. Important functions of the area are agriculture, nature and recreation. The agricultural Hunze catchment (ca. 250 km²) drains directly into the recreational Zuidlaardermeer (Figure 25),a freshwater system of about 6.5 km², in which blue-green algae (cyanobacteria) are causing problems regarding the local bathing water quality. The nutrient losses from the Hunze catchment are directly linked to blue-green algae blooms in the Zuidlaardermeer, but to predict and reduce these losses quantitatively, more research is needed on travel times, flow route contributions, and nutrient retention. In addition, more specific effect quantifications are needed for measures within the agricultural landscape to reduce the nutrient losses, also for the effects on the ecosystem health of the Zuidlaardermeer and even the more downstream Wadden Sea.

Figure 25. Zuidlaardermeer from South to North with Hunze mouth on the forefront (source: www.haren-haren.nl/zuidlaardermeer/)

The Hunze is a slowly flowing, meandering lowland stream in a predominantly sandy catchment (Waterschap Hunze en Aa's, 2008; Schollema, 2020). Originally, the Hunze drained a large peat area (Figure 26). This peat has largely been excavated or was oxidized after implementing artificial drainage. At the west side, the Hunze catchment is bordered by a 20m high sandy ridge (Hondsrug) from which groundwater seepage feeds the Hunze. At the eastern side, the catchment covers a much flatter excavated peat area (De veenkoloniën). While the two southern Hunze branches (Voorste and Achterste Diep) are relatively steep and fast flowing, the area north of their confluence is relatively flat until the mouth into the Zuidlaardermeer. The Waste Water Treatment Plant of Gieten drains into the Hunze just north of the confluence and about 12 km from the mouth.

The Zuidlaardermeer is a natural freshwater lake downstream of the Hunze catchment. The lake and the nature reserves around it (in total 21 km^2) are assigned as Natura 2000 reserve. The wetland area is especially an important bird habitat. The lake has a surface area of 6,5 km^2 and an average depth of 1,16 m (Klomp, 2021). The average yearly hydrologic residence time is 35 days and the average residence time in summer is 55 days. The soil consists partly of peat and partly of sand.

The Hunze and the Zuidlaardermeer are reported as two separate water bodies for the Water Framework Directive (water body codes NL33HU and NL33ZM).

Figure 26. Maps of the Hunze catchment (present (from Schollema (2020) and historical, 1634 (from topotijdreis.nl))

7.3 Eutrophication history

In the latest available status assessment (2023) for the Hunze water body, the biological WFD targets were not all met. The scores for fish and for macrofauna did not comply and also did not in earlier assessments. The 'other aquatic flora' did comply in 2023, but did not yet comply in 2021. The targets for some specific chemical priority substances (ammonium, plant protection substances, heavy metals) were structurally not met. Regarding the nutrients, the 3-year summer (April to September) average concentration target for Total-P (0,11 mg/l) was just met in 2023 but has not always been met in recent years. The summer average Total-P concentration (2020-2022) at the outlet of the Hunze was 0,14 mg/l. The target for the summer average Total-N concentration (2,3 mg/l) was met with a larger margin in most recent years with a 2020-2022 summer average concentration of 1,5 mg/l.

For the receiving Zuidlaardermeer, the biological targets for macrofauna and other aquatic flora were met, although these targets were not yet met in 2021. The targets for fish and phytoplankton were still not met in 2023. This non-compliance is partly related to a relatively high turbidity which blocks sunlight and prevents aquatic flora growth (Klomp, 2021). Similar to the Hunze, the targets for some chemical priority substances (ammonium, plant protection substances, heavy metals) were also not met. The 3-year summer average concentration targets for Total N (1,3 mg/l) and Total P (0,09 mg/l) were not met. Note that these targets concentrations are lower compared to the WFD targets for the Hunze. The 2020-2022 summer average concentrations were around or just above these targets (Ntot: 1,3 mg/l; Ptot 0,09 mg/l).

All in all, the water quality in both the Hunze and the receiving Zuidlaardermeer has significantly improved over the last decades and seems to be close to compliance to the WFD targets. As an example, the downward trend in the Ntot concentrations is presented in Figure 27 (see Appendix C and D for more details). The reduction in nutrient concentrations can partly be explained by the national manure policy implementation in the late '80s. In addition, more catchment specific mitigation actions were implemented like: reducing sewage overflows, improving the wastewater treatment, and increasing wetlands for water and nutrient retention during high discharge.

The compliance for biological WFD parameters seems to lag behind both in Hunze and Zuidlaardermeer. This can have several causes, e.g.:

- Lower nutrient levels are needed for a switch to a good ecological state (alternative stable states, see shallow lakes theory (e.g. Scheffer & Nes, 2007))
- Toxicity effects of the non-complying chemicals or emerging contaminants
- The legacy of nutrients (especially P) still present in the sediment of Zuidlaardermeer and, more locally, in and around the Hunze
- Suboptimal stream and lake morphology, wave impact, sediment, turbidity

Apart from the nutrient mitigation actions mentioned above, large investments have been made in improving the stream and lake morphology. Many parts of the Hunze have been widened, re-meandered and its riparian zones were restored. In addition, fish migration has been promoted by removing or by-passing obstacles. Around Zuidlaardermeer, the riparian zones and wetlands have also been restored and/or re-connected to the lake.

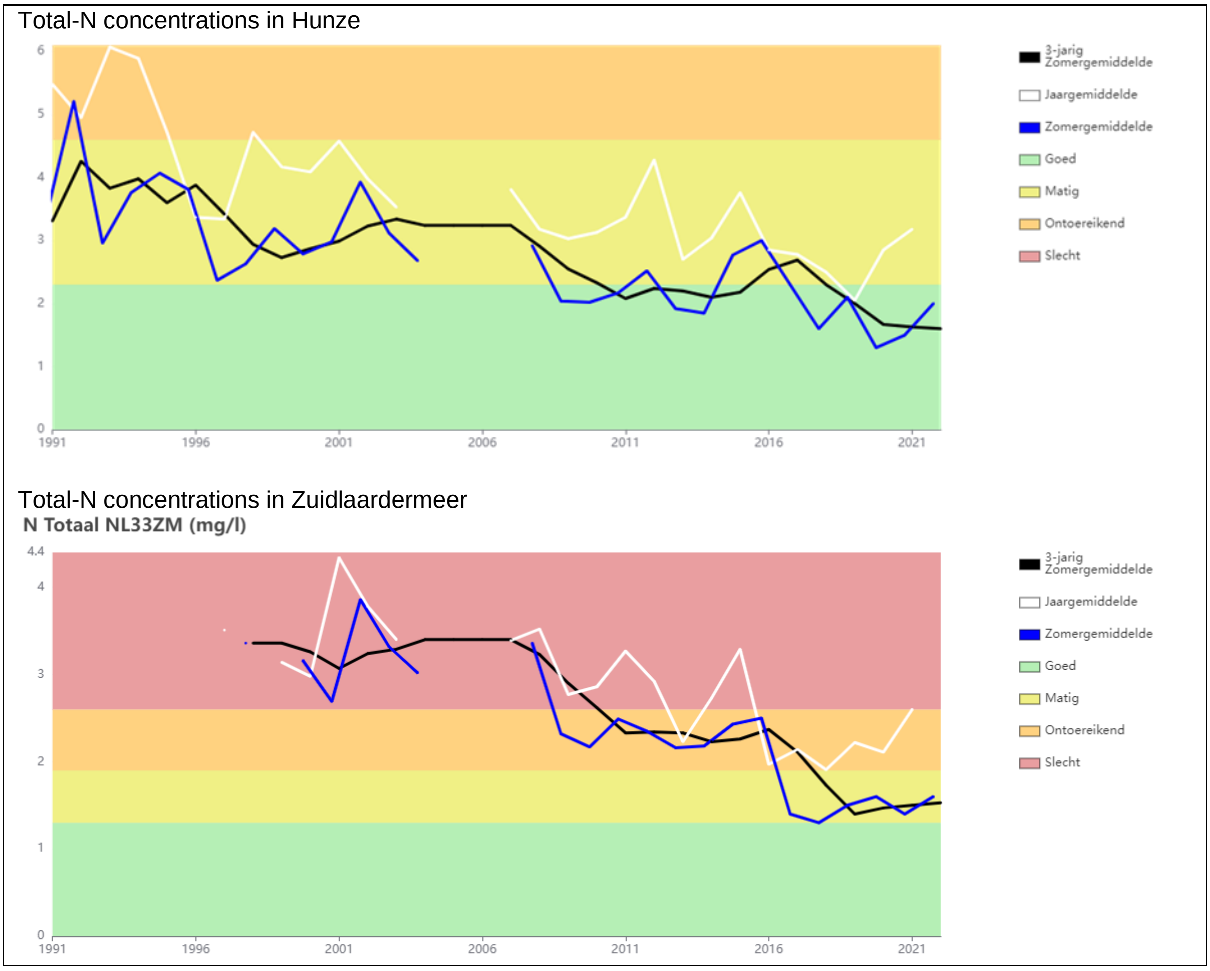


Figure 27. Total-N concentrations in Hunze and Zuidlaardermeer; the 3-year summer average concentrations (black line) are compared to the WFD concentration thresholds (indicated by green tot red colors).

7.4 Dutch WFD nutrient targets applicable to Hunze

7.4.1 Bioavailability

The water quality objectives in the WFD are supportive of the ecological objectives. The bioavailability of the various forms of N and P plays an important role in the effects on ecology. Inorganic forms of N and P such as ammonium, nitrite, nitrate and orthophosphate, are readily taken up by algae and are thus bioavailable. For uptake of N and P in organic form, these compounds must first be degraded. The degradability of organic N and P compounds varies greatly. Simple organic N and P compounds are usually easily absorbed. Difficult-to-degrade compounds such as humic and fulvic acids may contain N and P that is not available for algal growth.

The same is true for inorganic particulate P compounds. Phosphate adsorbed on the surface of inorganic particles is bioavailable, while phosphate in precipitates such as iron-(hydr)oxyphosphates and calcium phosphates is unavailable or hardly available to algae. Plants, however, can better release and absorb organically and inorganically bound P with their roots, including from larger soil or sediment particles. One way to do this is to excrete acid from the roots. Little is known about to what extent rooting aquatic or riparian plants, or plants with “floating roots,” such as duckweed, can also release and absorb phosphorus is not clear. When P-rich sediments become anoxic (usually in dry and warm periods), phosphate can be released from the iron(hydr)oxides.

7.4.2 Dutch WFD nutrient targets

The WFD targets for nutrient concentrations in Hunze and Zuidlaardermeer stem from the implementation process of the Water Framework Directive in the Netherlands. In the systematics of the Water Framework Directive (WFD), nutrients belong to the general physico-chemical quality elements. This means that the nutrient levels should not interfere with the achievement of the targets for biological quality elements. To come to the nutrients status classification system, summer averages of total-P and total-N are tested against biological targets (Heinis and Evers, 2007a). Nationwide nutrient standards have been derived that are differentiated by surface water type (e.g. the Hunze is water type R5; a slowly flowing headwater on sand, Zuidlaardermeer is M14; shallow buffered lake). The province of Groningen adopted these water type specific nationwide nutrient standards for Hunze (Schollema, 2020) and Zuidlaardermeer (Klomp, 2021).

The general classification system distinguishes between natural water types (Van der Molen et al., 2012), heavily modified water types (Evers and Van Herpen, 2010), and artificial water types (Evers et al., 2012). The water types ponds and lakes (M types) and flowing waters (streams and rivers, R types) can be classified as either natural or heavily modified. Ditches and canals (M types) are always artificial water types. For natural water types, the standard lies at the (lower concentration limit of the) quality class “Good Ecological Status” (GES). The ecological standard for heavily modified and artificial water types is the Good Ecological Potential (GEP). This standard is derived from the most similar natural water type.

In summary, the methodology used for setting the Dutch WFD targets came down to the following:

1. For ponds and lakes, a relationship was used between the summer mean nutrient concentration (total-N and total-P) and the concentration of chlorophyll-a and aquatic plants. Figure 28 shows the “example” figure used in almost all reports about the derivation of N and P WFD objectives. In these relationships the chlorophyll levels (y-axis) can be entered, belonging to a certain ecological quality ratio (EQR). The nutrient concentration (x-axis) that follows from this is the limit at which it can be stated with 90% certainty that the EQR in question is achieved. Figure 28 was derived from an inventory dataset of freshwater lakes in the 1990s (Portielje and Van der Molen, 1999). It is observed that the scatter in the relationship is large. Heinis and Evers (2007b) attribute this to the aspect that factors other than P are limiting for algal growth. This ignores the aspect that total-P is a sum parameter, in which the bioavailability of the various P components varies greatly and the concentration of the most bioavailable fraction (inorganic dissolved P) is often very low.

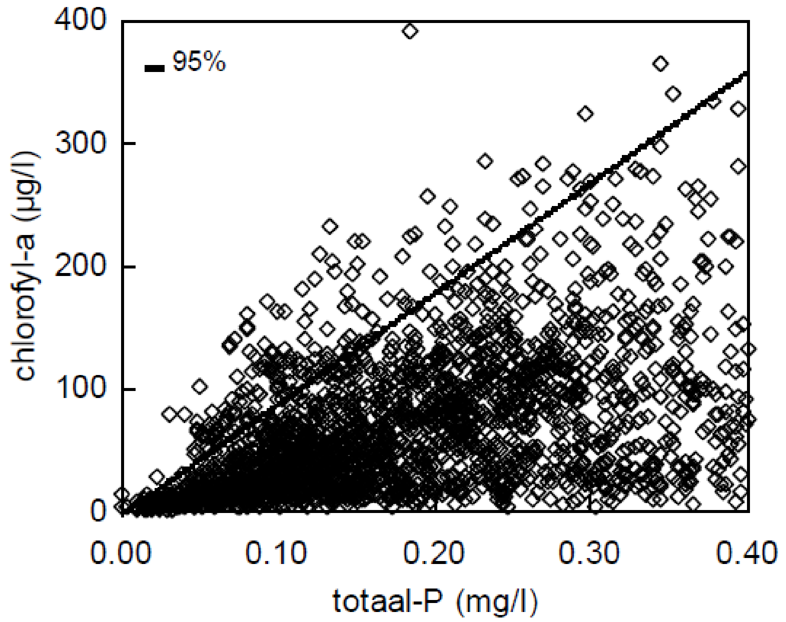


Figure 28. Derivation of the Dutch WFD objective for lakes based on the relation between P-total and chlorofyl-a concentrations; the line represents the 95 percentile of the ratio.

2. A different method was used for running waters. In this method, all phytobenthos samples from streams in the database 'Limnodata Neerlandica' were tested against the ecological quality ratios (EQRs) and linked to the summer average total-P and total-N concentrations. The concentration of total-P and total-N at which 90% of the biological samples meet at least the selected quality (as EQR) is then the WFD target (Figure 29). As for the lake approach, the scatter between the EQR and the summer nutrient concentrations is large.

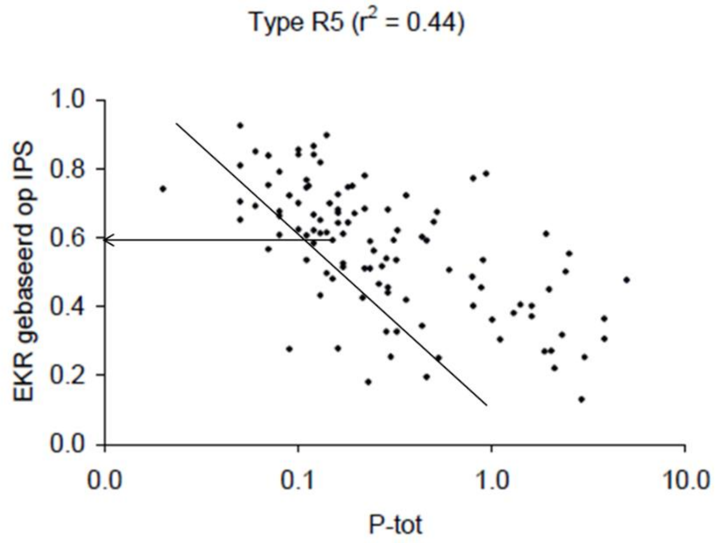


Figure 29. Derivation of the Dutch WFD objective in a running water based on the relation between the nutrient concentration (x-axis) and the EQR (Dutch: EKR) scores (y-axis).

Outside The Netherlands, separate standards for the P-total, dissolved P and soluble reactive P (SRP) are common in the EU (Philips & Pitt, 2015). For Nitrogen, together with Finland, The Netherlands are the only member state with just a standard for total-N. Other member states also have WFD standards for ammonium and/or nitrate. The Dutch focus on the growing season (standards for summer average concentrations) is more common for lakes; around half of the EU member states do the same, while the other half uses annual concentrations. For rivers, The Netherlands, Belgium (Flanders) and Poland are the only member states using growing season average values (Philips & Pitt, 2015).

7.4.3 Nutrient targets and nutrient concentration variability

Figure 30 shows the strong seasonality in the Total-N concentrations and in the N components NO3 and NH4 in one of the upstream Hunze tributaries (Voorste Diep). The winter Total-N concentrations generally exceed the Dutch WFD targets which are only applied to the summer average concentrations. Nitrate is the dominant N species, while both nitrate and ammonium show higher concentrations in winter. The high winter concentrations coincide with high discharges, causing the winter period to be dominant in the nitrogen loads to the Zuidlaardermeer and further downstream (canals of Groningen and the Wadden sea). The relevance of winter nutrient transport to receiving water systems is not recognized in the Dutch WFD targets for nutrients based on summer concentrations. Furthermore, efforts to reduce agricultural nutrient losses will mostly affect the drainage season (winter) concentrations.

The Total-P concentrations do not show a large seasonality. Still, the winter loads for Total-P can also be higher due to higher discharge. Figure 30 shows that particulate P is the dominant P fraction, especially at high concentrations. This particulate P is largely attached to iron(hydro)oxides (Particulate Inorganic Phosphorus) (Van der Grift, 2017). From high frequency monitoring in other catchments (e.g. Rozemeijer et al., 2010; Van der Grift et al., 2016, Barcala et al., 2020), it is known that very short total-P concentration peaks occur during discharge events, when P-rich sediment is remobilized. Between the events, iron and P from groundwater form iron(hydr)oxides and replenish the P-rich sediment. The typical sub-daily concentration dynamics are not captured by monthly sampling schemes (like in Figure 30) and load estimates from these monthly measurements are highly uncertain. In addition, the summer average P concentrations based on 6 measurements used for WFD compliance testing are highly uncertain (e.g. Wade et al., 2012; Halliday et al., 2015.

To get better insights into the nutrient concentration dynamics and loads from the Hunze to the Zuidlaardermeer, daily flow-proportional samples were collected at station ‘4206; Oostermoersevaart duiker in weg Zuidlaren-De Groeve’ for total-N and total-P analysis since 2019 (see Figure 31). These data show that daily averaged peak concentrations over 0,3 mg/l occur several times per year, while the base level total-P concentration is between 0,1 and 0,2 mg/l. For total-N, the seasonal pattern dominates, with summer concentrations around 1 mg/l and winter concentrations around 7 mg/l. The summers of 2019, 2020, and 2022 were relatively dry. In the more average summer of 2021, the total-N concentrations in the waters were higher (1-3 mg/l) due to more leaching and shorter residence times in surface water (resulting in less biochemical processing) compared to the dryer years. The summer of 2021 also shows higher base level total-P concentrations (0,15-0,3 mg/l) compared to the dryer summers.

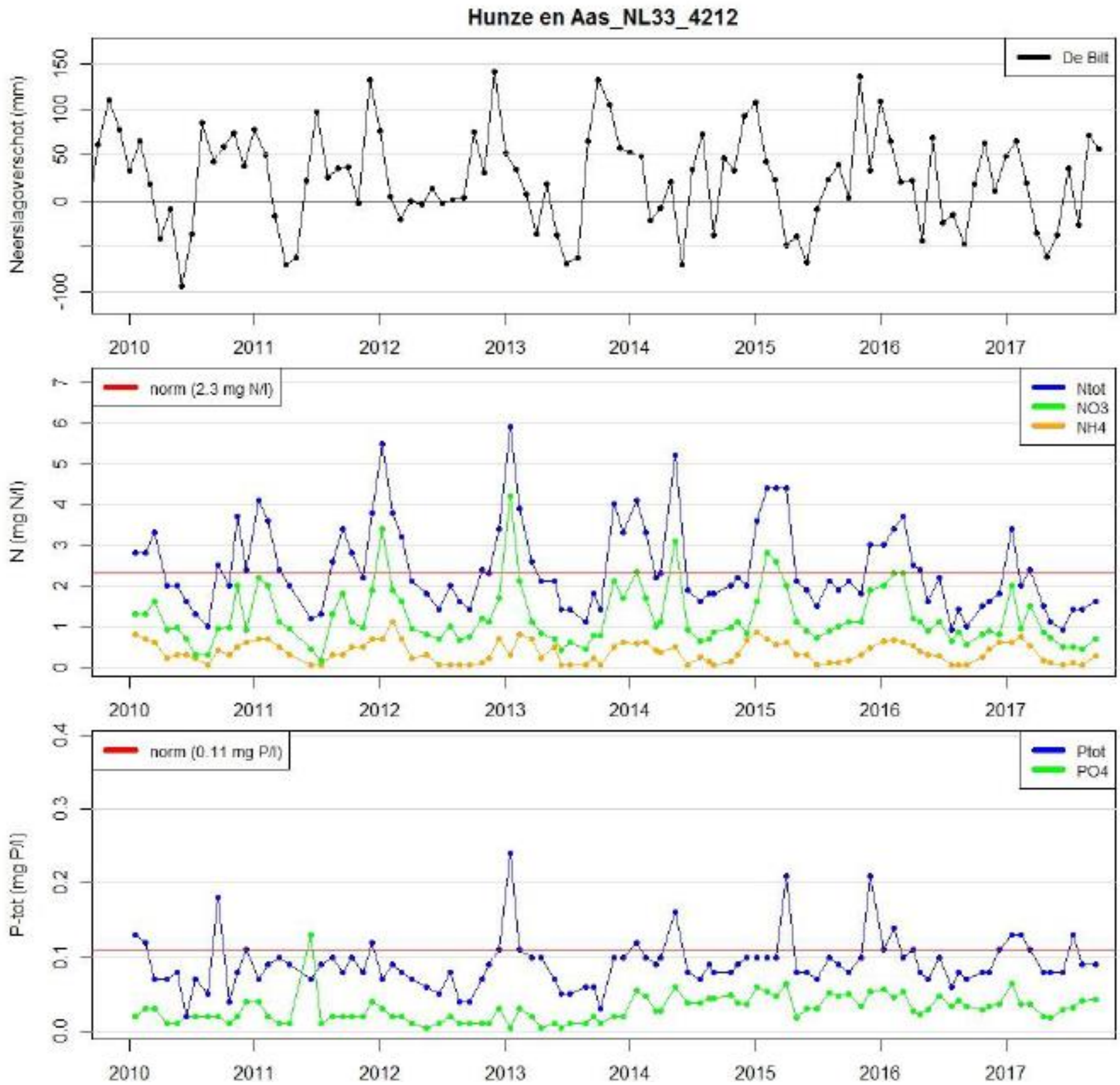


Figure 30. Dynamics in N and P species concentrations at Voorste Diep.

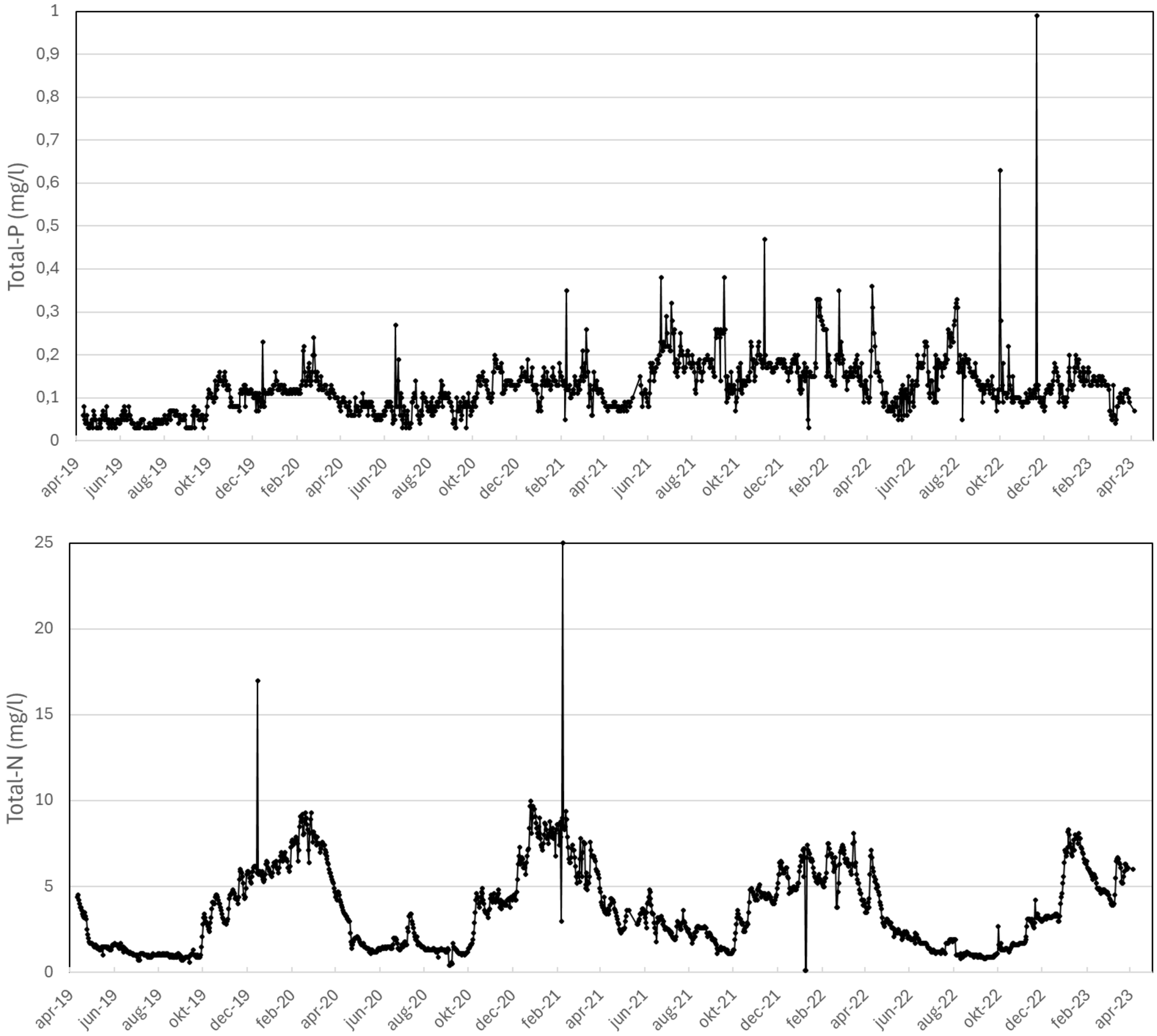


Figure 31. Daily flow proportional total-P and total-N concentrations at the outlet of the Hunze into Zuidlaardermeer.

7.4.4 Limiting nutrients

Zuidlaardermeer is mainly P limited (Klomp, 2021). The N/P ratio is increasing (see Figure 32), which means that the lake becomes increasingly P-limited. The measurements of Ptot and PO_4 in Zuidlaardermeer (Figure 33) show very low concentrations (below 0,01 mg/l) in summer. However, the inorganic N-species (NO_3 and NH_4) also show very low concentrations in midsummer (Figure 34). Figure 35 shows measured Chl-a concentrations in Zuidlaardermeer.

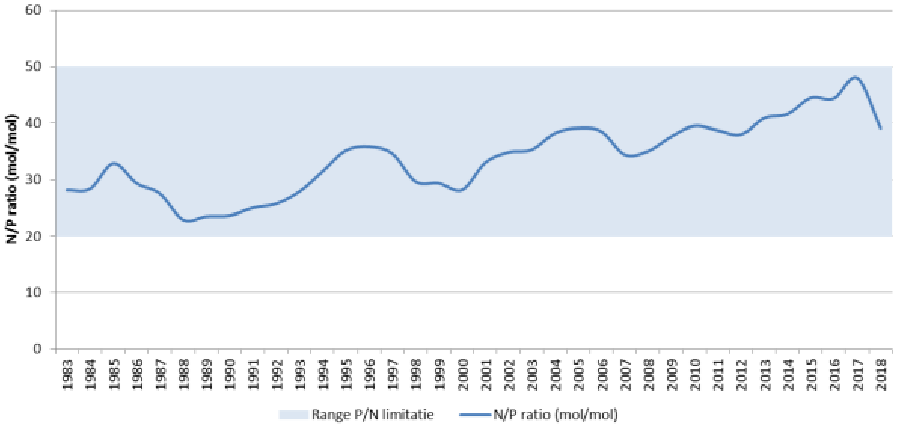


Figure 32. Summer average N/P ratio in Zuidlaardermeer (from Klomp, 2021).

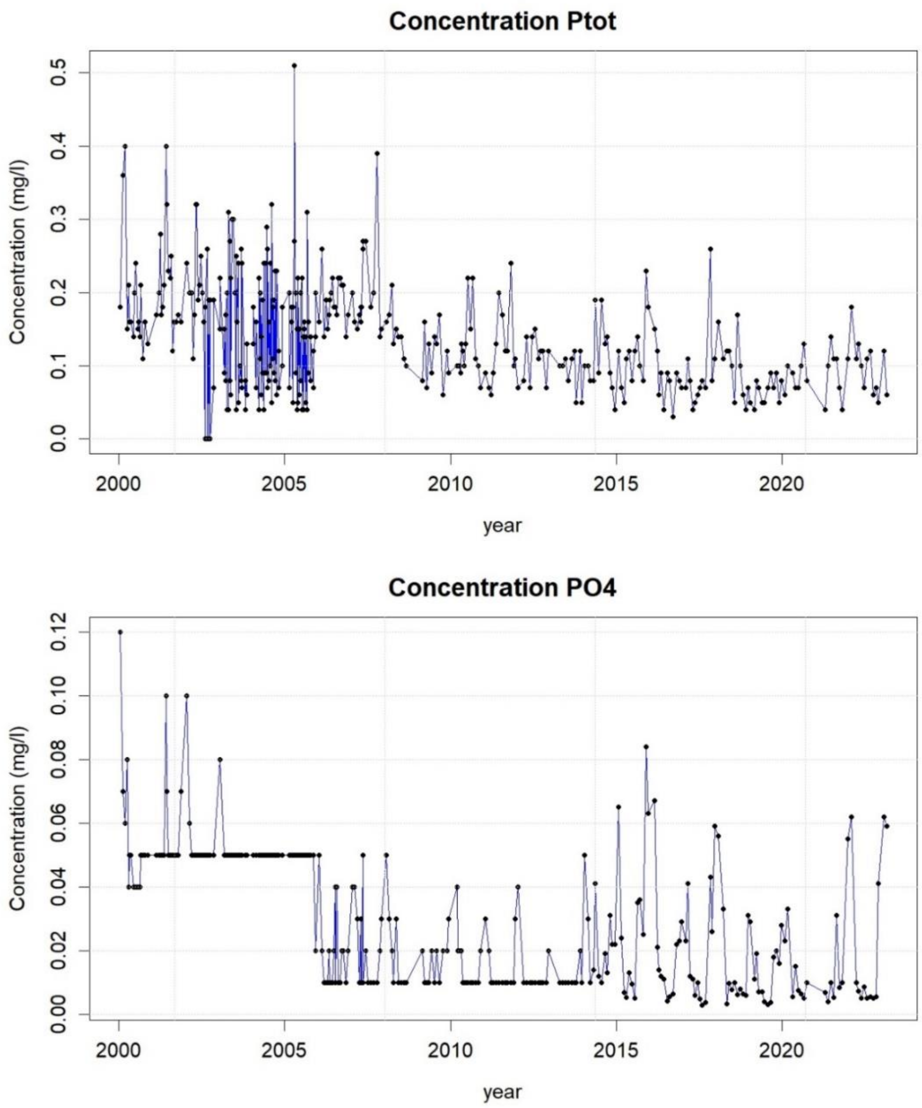


Figure 33. total-P and PO4-P concentrations in Zuidlaardermeer (location Zuidlaardermeer noord). Note that the PO4 detection limits were lowered from 0.05 mg/l PO4-P to 0.01 mg/l in 2006 and further down to 0.005 mg/l in 2014.

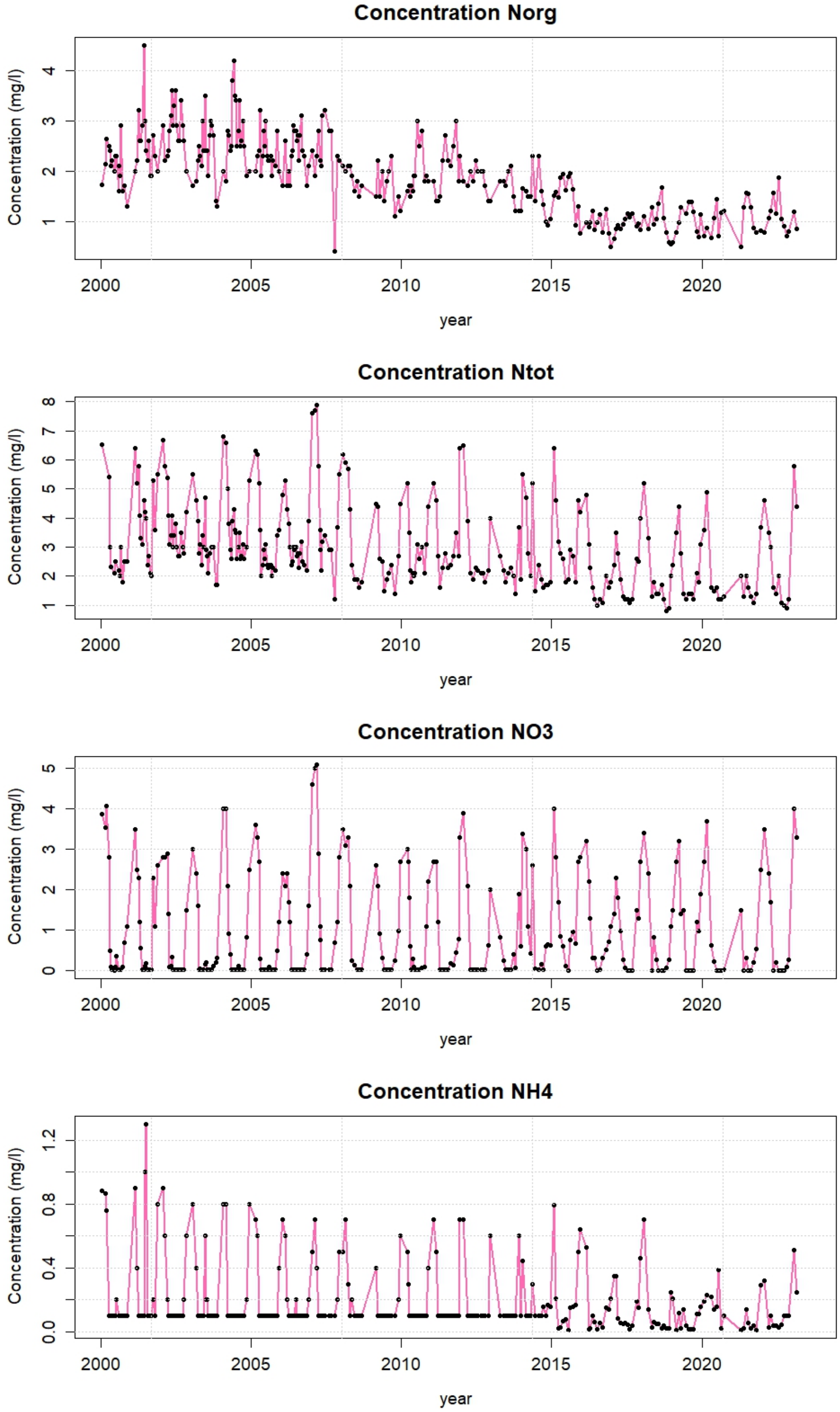


Figure 34. Norg, total-N, NO3 and NH4 concentrations in Zuidlaardermeer (location Zuidlaardermeer noord).

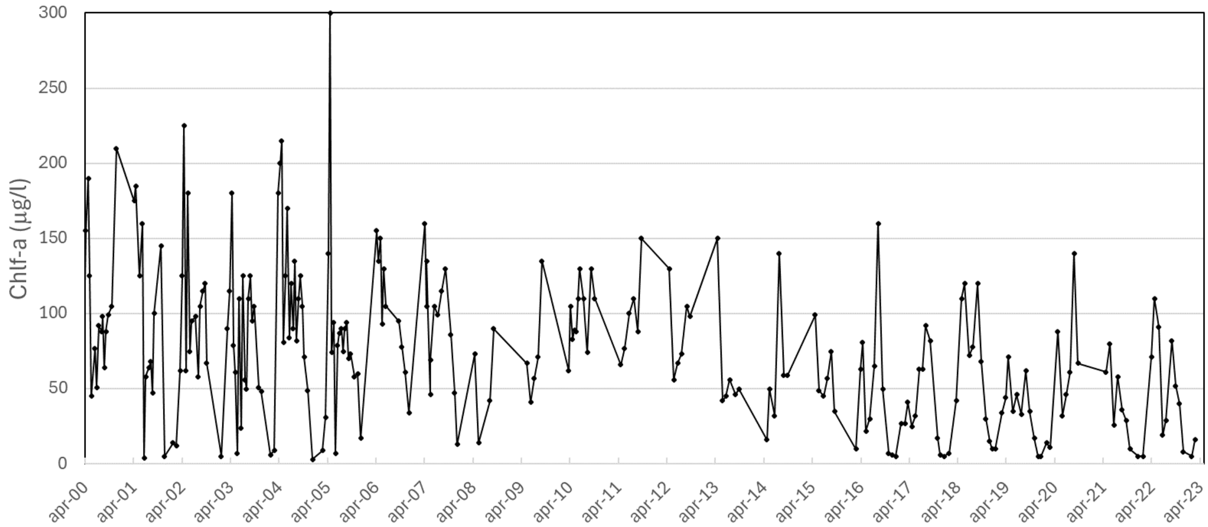


Figure 35. Chlorophyll-a concentrations in Zuidlaardermeer (location Zuidlaardermeer noord).

7.5 Safe Ecological Limits

Within NAPSEA, the aim is to connect the safe ecological limits of receiving water systems (like the Wadden Sea and the Zuidlaardermeer) with nutrient concentration and load targets for the contributing upstream catchments. For the Hunze catchment this means that the nutrient levels should be within the ecological thresholds for:

- the surface waters in the Hunze catchment itself
- the receiving Zuidlaardermeer
- the water bodies between downstream of Zuidlaardermeer (among which the channels around the city of Groningen and lake Lauwersmeer)
- the Wadden Sea

Here, we focus on the safe ecological limits for nutrients from the Hunze catchment to ensure a healthy aquatic ecosystem in the Zuidlaardermeer and in the Wadden Sea. Our focus is not on the water bodies downstream of Zuidlaardermeer. With the Zuidlaardermeer being a vulnerable water system for eutrophication, we expect that that no additional reductions are needed for the downstream water bodies. In other words: when the nutrient loads from the Hunze are within the safe ecological limits for the Zuidlaardermeer, we assume that these loads are also low enough to protect the terrestrial water bodies downstream. The same holds for the surface waters within the Hunze catchment itself, which have higher nutrient concentration targets compared to Zuidlaardermeer: when the nutrient loads from the Hunze are within the safe ecological limits for the Zuidlaardermeer, we also expect the nutrient concentrations to be within safe ecological limits for the Hunze river.

The impact of nutrients from the Hunze catchment on the Zuidlaardermeer is very direct. For the Wadden Sea however, the Hunze only provides a tiny fraction of the nutrient loading. A direct link between nutrient loads from the Hunze and the ecology of the Wadden Sea is therefore hard to make. However, for a healthy Wadden Sea, the sum of nutrient loads from all contributing catchments should be within safe ecological limits. When the Hunze catchment (or another one) does not comply, this would have to be compensated for by another area. To prevent rolling off nutrient reduction needs between areas, we assume that the safe ecological limits for the Wadden Sea can be translated to all contributing catchments, including the Hunze catchment.

7.5.1 Hunze safe ecological limits for the Zuidlaardermeer

Although the WFD targets (total-N and total-P) are related to the concentration of chlorophyll-a and aquatic plants, the scatter in the relationship is large, and the classification from good to bad states may appear arbitrary. Hence, here we propose to focus not on specific Chlorophyll concentrations but on whether the system is algae-dominated or plant-dominated (aka in the 'clear water state'). The critical P-concentrations at which the system switches from one state to another depend on the local properties of the system but can be derived using PCLake (Figure 36) . PCLake is an ecosystem model especially designed to study the effects of eutrophication on shallow lakes and ponds, which has been extensively applied and calibrated on Dutch lakes (Janse, 2005). The model is used to define the critical P-loadings per lake and to evaluate the effectiveness of restoration measures, while taking into account the phenomena of alternative stable states and hysteresis (e.g. Scheffer & Nes, 2007). For this purpose, also a meta-model has been developed (https://www.witteveenbos.com/nl/digitalediensten/pclake-metamodel). The meta-model can be used by water managers to derive an estimate of the critical loading values for a certain lake based on only a few important parameters, without the need of running the full dynamical model.

In order to change lake Zuidlaardermeer from an algal dominated state into a clear (waterplant dominated) state, the PCLake metamodel calculates a critical P-load threshold of 2.75 mgP/m^2/d. Timeseries based on measurements during the years 2003-2016 show an average P-load of 4.5 mgP/m^2/d, which would mean that (relative to that period) a reduction of almost 40% would still be needed. Once the system has reached a clear state, the PClake meta model indicates that the P-load should stay below 3.5 mgP/m^2/d in order not to switch back to a turbid state.

Assuming that the discharge from the Hunze catchment has not changed much since 2016 (apart from some uncommonly dry years resulting in odd outliers), a 40% load reduction would still be needed present day. This can be translated into a 40% reduction of the total-P concentrations within the Hunze catchment. The annual average Ptot concentration for 2003-2016 was 0,13 mg/l, so a 40% reduction would mean an average total-P concentration target of 0,08 mg/l.

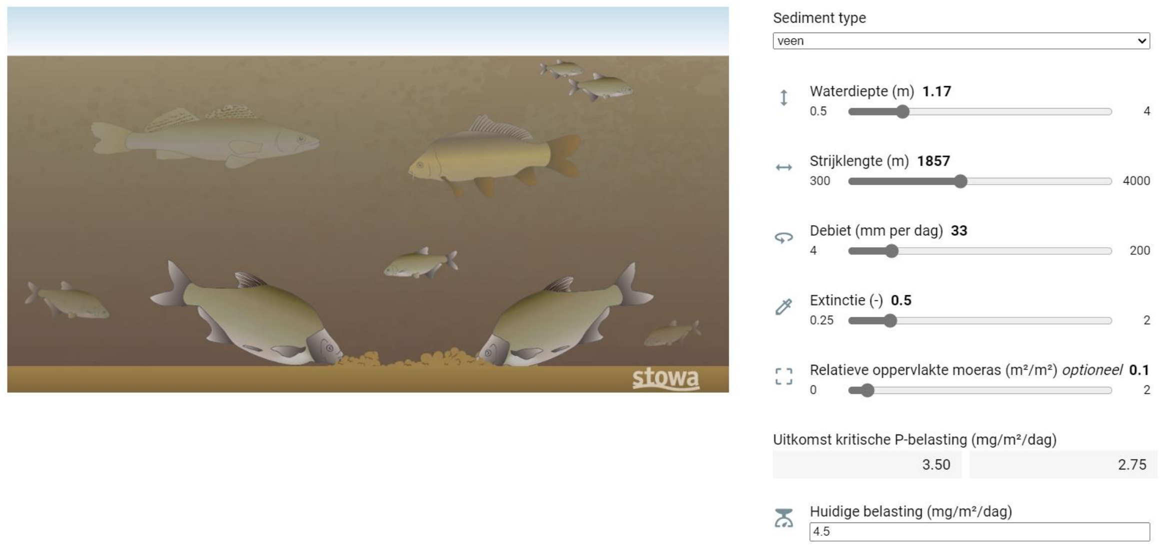


Figure 36. Screenshot from the website hosting the PClake metamodel, with the values used to represent lake Zuidlaardermeer.

7.5.2 Hunze safe ecological limits for the Wadden Sea

In chapter 4, two approaches were outlined to define safe ecological limits for the Wadden Sea. The first approach focusses on preventing *Phaeocystis* blooms in the Wadden Sea. To achieve this, a reduction of total-N loads and concentrations for the contributing rivers of 30-55% is needed, depending on the silica concentration levels (compared to 2010-2017). For the Hunze, this would mean a reduction from 3.1 mg/l (annual average total-N concentration 2010-2017) to 1,4-2,2 mg/l. As the winter dominates the total-N loads (with both higher concentrations and discharge), a winter concentration target may be more appropriate. In that case, the 2010-2017 average December-February concentrations of 4,5 mg/l should reduce to 2,0-3,2 mg/l.

The second approach for defining safe ecological limits for the Wadden Sea focusses on restoring sea grass (see chapter 4). For this, a 34%-39% reduction of N loads is needed (relative to 2010-2017 levels). For Hunze outlet this would correspond to a total-N concentration reduction from 3.1 mg/l (annual average 2010-2017) to 1,9-2,0 mg/l.

7.8 Discussion

We observe that some regional water authorities focus on P and some on N. However, to reach safe ecological limits in all water systems, both N and P are important. In general, most freshwater systems are P-limited, while receiving coastal waters are N-limited but N-limitation can also occur in freshwater systems.

There also is a discrepancy in the WFD targets. The Dutch WFD targets are based on summer average concentrations, while the targets in Germany are based on yearly averaged concentrations. The focus on summer concentrations may be logical for the local aquatic ecology, the winter concentrations and loads are generally higher and have more impact on downstream, receiving water systems with longer residence times.

A focus on summer concentration also has a relation with mitigation strategies. Agricultural losses are generally highest in the wet winter season. Most agricultural measures to improve water quality will therefore first and mostly affect winter concentrations. Realising reduced agricultural losses may therefore not lead to compliance with WFD targets based on summer concentrations. To avoid these discrepancies, water authorities may also consider setting goals for maximum yearly loads for catchments. Especially the impact on downstream water systems is better reflected by N and P loads from a catchment than by N and P concentrations.

8. TOWARDS SAFE ECOLOGICAL LIMITS: A SYNTHESIS AND DISCUSSION OF THE CASE STUDIES

8.1 Introduction

In the previous Chapters, case studies on Safe Ecological Limits in the Rhine, Hunze, Elbe Estuary and Wadden Sea were presented. The Wadden Sea case study was used to develop an alternative view on Safe Ecological Limits beyond just using total phytoplankton biomass. The Rhine case study developed a view on Safe Ecological Limits beyond just the Rhine river and explicitly looked at the entire basin including all tributaries. The Elbe case study focused on the low oxygen problems in the estuary. The Hunze case study looked at the entire source-to-sea spectrum including the Hunze river catchment, the receiving Zuidlaardermeer, the downstream water bodies and ultimately the receiving Wadden Sea.. Here, we will shortly summarize the main results and discuss them regarding the question whether the Safe Ecological Limits for freshwater ecosystems will also enable a Wadden Sea without eutrophication problems. The indicators and proposed reduction needs (reference: 2010 – 2017) are summarized in Table 3.

8.2 Summary of the Case Studies

8.2.1 Wadden Sea

In the Wadden Sea case study, it was shown that ultimately N is the main limiting nutrient for phytoplankton growth. Two new indicators were developed in addition to existing indicators for nutrients and chlorophyll: seagrass recovery and N/Si ratios in winter. Seagrass recovery in the northern Wadden Sea accelerated around 2000 but no clear recovery was observed in the southern Wadden Sea. The eutrophication conditions (based on phytoplankton biomass) that prevailed during recovery in the northern Wadden Sea were projected on the southern Wadden Sea. It was estimated that in comparison to the period 2010 – 2017 all riverine TN loads should be reduced by at least 1/3 (range: 34-46%). An alternative approach is based on the ecological role of dissolved silicate (Si): during pre-eutrophication conditions the spring bloom was limited by N, shifting to an Si limitation of diatoms during increased eutrophication and leading to increased blooms of algae not depending on Si. To return to conditions of an N limited spring bloom, a reduction in riverine N loads of 30% (Elbe) - 55% (Ems) is needed.

8.2.2 Elbe Estuary

The Elbe Case Study focussed on the present O_2 problems in the upper estuary caused by the extreme large phytoplankton blooms in the riverine part of the Elbe. When these blooms enter the upper part of the upper estuary, light limitation (increased depth due to dredging and high SPM concentrations) and grazing decimate the phytoplankton standing stock (and primary production of O_2) leading to increased O_2 demand and low O_2 levels. Based on models, a reduction need in organic matter loading (mainly both living phytoplankton and fresh phytoplankton detritus) of about 45% was estimated to reach Safe Ecological Limits of 7 mg O_2 /l as demanded by German law.

Present phytoplankton levels (March-October) in the Elbe just upstream of the estuary of about 109 µg Chl a/l are well above the Safe Ecological Limits of 40.1 µg/l a as demanded by German Law. The question of how to reduce these phytoplankton blooms is challenging: During the spring bloom in the Elbe, Si reaches limiting levels for diatom growth but enough N and P is available for other phytoplankton groups like green algae. At present, phytoplankton growth is most of the time not limited by N and P availability except for periods of low discharge when P limits phytoplankton growth in summer. As a first estimate we suggest that both N and P concentration levels in the Elbe have to be reduced by 45% for safe O_2 levels and by 63% for mean phytoplankton levels of 40.1 µg Chlorophyll a.

8.2.3 Rhine catchment case study

The Rhine and Elbe are the two major rivers draining a large part of Germany and the Netherlands impacting the Wadden Sea. The two rivers are very different: first, the Rhine is fed by large amounts of melting water (about 48% of total discharge) from the Alps with low nutrient concentrations. Furthermore, a high number of invasive filter feeders inhabit the riverbed suppressing phytoplankton growth in the Rhine in contrast to the Elbe, where large phytoplankton blooms occur (e.g. Hardenbicker et al., 2016).

The Rhine catchment case study is part of the NAPSEA modelling exercise, addressing the question which different measures on land contribute to the reduction of N loss to the environment. Focus is therefore on annual nitrate (NO_3-N) dynamics being part of the model.

The good (water quality class II) status is reached at stations that are below a 90th percentile of 2.5 mg/L (LAWA, 1998). For this assessment, we translated this value to a value that is also modelled – mean annual concentrations of NO_3-N - to assess NO_3-N concentration status across. 10 out of 13 stations meet the good status with NO_3-N concentrations below 1.9 mg/l. The average concentration for all stations was 1.69 mg/l. However, in 71 out of 80 sub-catchments (89%) the tributaries to the Rhine fail the good status. Between 2010 and 2020, the average concentration across all the sub-catchments in this period is 3.55 mg/l. This suggests an average reduction need of 44% in the subcatchments.

For phosphate, a similar picture as for nitrate emerges: whereas along the main stem of the Rhine 11 out of 13 stations meet the good status with PO_4 concentrations below 0.055 mg/l , the subcatchments fail the good status (0.55 mg/l) in most cases with a mean concentration after 2010 of 0.114 mg/l. suggesting a reduction need of about 50%.

In the Dutch part of the Rhine catchment nearly 50% of the water bodies had Moderate or worse status for phosphorus and about 30% had Moderate or worse status for nitrogen in 2021 (IenW, 2022).

8.2.4 Hunze case study

The Hunze is situated in the northeast of The Netherlands (south of Groningen). The agriculturally dominated Hunze catchment (ca. 250 km²) drains directly into the recreational Zuidlaardermeer, a freshwater system of about 6.5 km that is connected to the Wadden Sea through various waterways. The ecological status of the Hunze and the Zuidlaardermeer have improved during the last decades, but problems remain including blue-green algae (cyanobacteria) compromising local bathing water quality and submersed vegetation.

In the Hunze, the target for Total-P (0.11 mg/l for the summer average concentration) was just met in 2023 but has not always been met in recent years. The summer average Total-P concentration (2020-2022) at the outlet of the Hunze was 0.14 mg/l. The target for the summer average Total-N concentration (2.3 mg/l) was met with a larger margin in most recent years with a 2020-2022 summer average concentration of 1.5 mg/l, but the goals for NH_4 were not met. In the receiving downstream lake Zuidlaardermeer, goals for Total N (1.3 mg/l) and Total P (0.09 mg/l) are almost reached: the 2020-2022 summer average concentrations were around or just above these targets.

To change lake Zuidlaardermeer from an algae dominated state into a clear (dominated by submersed vegetation) state, a critical P-load threshold of 2.75 mg P/m^2/d is suggested based on model calculations. Presently (2003-2016), an average P-load of 4.5 mg P/m^2/d prevails, implying a reduction need of almost 40%.

As in many other freshwater systems, focus is on P to reach a good environmental status whereas for the Wadden Sea an N reduction is needed to reach Safe Ecological levels. For seagrass recovery, a 34%-39% reduction of N loads is needed (relative to 2010-2017 levels). For the Hunze outlet this would correspond to a reduction of about 34% from 3.1 mg/l (annual average 2010-2017) to 1.9-2.0 mg/l. This approach assumes an equal contribution in the required load reduction among the contributing catchments. However, the overall load reduction towards the Wadden sea can also be achieved by targeting catchments with the largest contributions or with the most cost-efficient mitigation possibilities.

Table 3. Overview of the resulting safe ecological limits

Case study	Indicator	Main driver	Reduction needs:	Comments?
Wadden Sea	Sea grass recovery	River TN loads	N: 34-43%	Generally applicable to all rivers
Wadden Sea	Absence of blooms by non-silicifying algae (e.g *Phaeocystis*)	Si:N ratio	N: 30-55%	Generally applicable to all rivers
Elbe estuary	O_2 >7 mg/l	Import riverine organic matter (phytoplankton)	N: ~45%	Local (estuary)
Elbe river	Phytoplankton biomass <40.1 µg/l	Nutrient loads (mainly P, but with an unknown role of N	N: 63% P: 63%	Local (estuary)
Hunze	Dominance of submersed vegetation in the Zuidlaardermeer	Incoming Phosphorus load	P: 40%	The local critical P-load depends on various factors and should thus be derived per system
Rhine catchment	N and P concentrations		N: 44% P: 50%	

8.3 Reduction needs

8.3.1 Summary of reduction needs

Table 3 summarizes the reductions in N and P proposed for the different case studies. All numbers are relative to the period 2010 – 2020, during which no large changes in riverine nutrient concentration occurred (see chapters on the Rhine and on the Wadden Sea and van Beusekom et al., 2019). The reductions needed to reach Safe Ecological Limits for N are between 30 and 63% and for P between 44% and 63% depending on location and indicator (Table 3). The highest reductions are needed in the Elbe based on reductions needed to reduce the immense phytoplankton blooms. However, it should be noted that mostly, both N and P are presently not limiting the blooms and both must be reduced to reach the WFD goals for the Elbe river. More research is needed on how to reduce the Elbe phytoplankton blooms. All other reductions are between 30 -55% for N both in the Wadden Sea and in the catchments and 40-50% for P in the catchments. No goals for P have been developed for the Wadden Sea as phosphorus concentrations have already decreased significantly since the 1990s. During the spring/early summer bloom N/P ratios are extremely high with possible negative consequences for higher trophic levels, which also underlines the need for N reductions.

8.3.2 The need for specific winter goals for riverine nutrients

Reduction needs are based on the above-mentioned ecological problems which are all related to nutrient overenrichment. These are typically related to too high levels of primary production, too large phytoplankton blooms and shifts in phytoplankton assemblages during spring and summer both in the freshwater systems and in the Wadden Sea and adjacent North Sea. To cover the relevant period, annual levels for nutrients or summer levels for phytoplankton are used. Also, for the Wadden Sea, summer indicators are used: seagrass surface or summer Chlorophyll levels. For phytoplankton growth in the Wadden Sea and North Sea, however, the riverine winter loads of nutrients are the main driving factor. One of the Safe Limits proposed for the Wadden Sea is based on N/Si ratios. Both TN and Si reach the highest levels during the first months of the year. With a maximum allowable TN concentration in winter, e.g. based on a N/Si ratio of 1, a more precise goal for riverine inputs can be formulated. Interestingly, the original suggestion (Gade et al., 2011) of maximum TN loads in rivers was based on a Si/N molar ratio of 1 (~ 200 µ N mol/l; ~2.8 mg N/l). However, later this threshold value was implemented in Germany as an annual mean which -given the strong seasonal cycle of N and Si- underestimates the importance of winter concentrations. High winter discharge levels amplify this effect. In the Netherlands, the N and P targets for summer average concentrations neglect the impact of the much higher winter loads on downstream ecosystems and ultimately the Wadden Sea.

8.3.3 The impact of N on terrestrial ecosystems

The reduction needs derived within NAPSEA to bring aquatic systems within safe ecological limits of about 30-60% may seem very high. However, reduction needs to bring terrestrial ecosystems within Safe Limits are within similar ranges. Nitrogen deposition on terrestrial ecosystems has a strong impact on biodiversity (Bobbink et al., 2010). De Vries et al. (2021) estimated N reduction needs for terrestrial and freshwater ecosystems based on critical N loads. They discerned between biodiversity effects mainly driven by atmospheric N-deposition, surface water eutrophication and groundwater contamination with NO_3 both driven by leaching from soils. Results are shown in Table 4. Reduction needs (reference: 2010) for biodiversity loss are about 31% at the EU level, but 39% and 46% for Germany and The Netherlands, respectively. Average EU reduction needs for reaching critical N levels in surface water are 57%, (51% for Germany and 74% for The Netherlands). The goals for surface water are based on a run-off concentration of 2.5 mg/l. This figure is based on the TN targets for rivers as defined by the WFD consisting of 2.5 mg/L (summer average) for the Rhine, and 2.8 mg/l (annual average) for the Elbe, Weser, Ems debouching into the North Sea.

De Vries et al. (2021) note that that several factors like wastewater input and loss within streams (denitrification, sedimentation) are not included in their analysis.

The Water Framework Directive does not require EU-wide uniform goals on N in surface waters but rather requires Member States to determine type-specific physicochemical criteria ensuring/supporting good ecological status (e.g. Poikane et al. 2019, Salas-Herrero et al. 2022). Poikane et al. (2019) suggest that currently established safe ecological limits for lakes and rivers show a wide range but should be much lower than 2.5 mg/l (0.7 – 1.5 mg N/l). The authors stress that defining nutrient thresholds should take into account biological responses to nutrient enrichment, but that in many cases other approaches have been followed.

Table 4. Actual (2010) N inputs (kt N yr^{-1}) and critical N inputs for all agricultural land, in view of critical thresholds for NH_3 emissions, N runoff to surface water and N leaching to groundwater, for Germany, The Netherlands and Europa (from de Vries et al., 2021).

Country	**N inputs (kton N yr^{-1})**			
	Actual	At critical NH_3emission	At critical N runoff to surface water	At critical N leaching to groundwater
Germany	3426	2096 (-39%)	1664 (-51%)	3143 (-08%)
Netherlands	695	378 (-46%)	182 (-74%)	467 (-33%)
EU	21791	15045 (-31%)	12409 (-57%)	22062 (+1%)

Table 5. DESTINO sub-indicators: Present status (2015) and goals. Only the bold numbers have been used for the DESTINO-goal (1050 kt N a^{-1}) and the present status (1574 kt N a^{-1}).

Environmental goals	N species	Present status	Goal	deviation (%)
Terrestrial biodiversity	*NH_3 emission*	*625.3 kt NH_3-N a^{-1}*	*441.1 kt NH_3-N a^{-1}*	*142%*
Terrestrial eutrophication	***Sum NH_3 + NO_xemission***	***625.3 kt NH_3-N a^{-1} 361.4 kt NO_x-N a^{-1}***	***396.4 kt NH_3-N a^{-1} 168.0 kt NO_x-N a^{-1}***	***175%***
Surface water	***TN loads***	***356.2 kt TN a^{-1}***	***314.0 kt TN a^{-1}***	***113%***
Ground water	***N surplus/NO_3***	***147.6 kt NO_3-N a^{-1}***	***126.6 kt NO_3-N a^{-1}***	***117%***
Climate	***N_2O Emissions***	***83.4 kt NH_3-N a^{-1}***	***47.8 kt NH_3-N a^{-1}***	***174%***
Human health	*NO_x Emissions*	*361.4 kt NO_x-N a^{-1}*	*235.8 kt NO_x-N a^{-1}*	*153%*

In the framework of the German project DESTINO (Heldstab et al., 2020, see also Geupel et al., 2021) a German Nitrogen-indicator was developed. As in de Vries et al. (2021), the goal was to quantify limits in N loads that enable to maintain biodiversity, avoid eutrophication, guarantee a good air, groundwater and surface water quality. For each of these goals, limits were developed. The indicators are presented in Table 5. Not all sub-indicators were used for the integrated Indicator: in case of similar pathways, the lowest goal was taken (Bold numbers in Table 5). The largest reduction needs concern terrestrial eutrophication, where NH_4 +NO_x loads have to be reduced by 75%. To reach the WFD goal set for TN discharge to the coastal zone 2.6 mg N/l for the Baltic, 2.8 mg/l for the North Sea), a reduction of only 17% is proposed. Other studies with a more local focus (e.g. Kunkel et al., 2017) come to similar conclusions. The Destino goals for surface water are based on the German WFD. Results from the case studies suggest that for rivers more stringent goals should be applied.

8.4 Conclusion

To reach Safe Ecological Limits in the Wadden Sea, TN loads of rivers impacting the southern Wadden Sea have to be reduced by at least 34% - 49% to enable seagrass recovery. For balanced Si/N ratios in the Wadden Sea in winter NO_3 concentrations must be reduced by 30 to 55% depending on riverine winter Si concentration (reference 2010-2017). These reduction needs seem to be high but compare to the reduction needs in terrestrial ecosystems based on critical N loads. In the Netherlands even higher reductions are needed amounting up to 74%.

For the Wadden Sea, no goals for P were developed. At present, N/P ratios especially during spring and early summer are high possibly impacting local food quality. But in freshwater aquatic systems P reductions of at least 40% are needed. Water quality standards for the terrestrial surface waters generally neglect the large impact of winter concentrations and loads on downstream water bodies and ultimately the Wadden Sea. Safe Ecological Limits should both ensure aquatic ecosystem health locally as well as in the receiving downstream water bodies.

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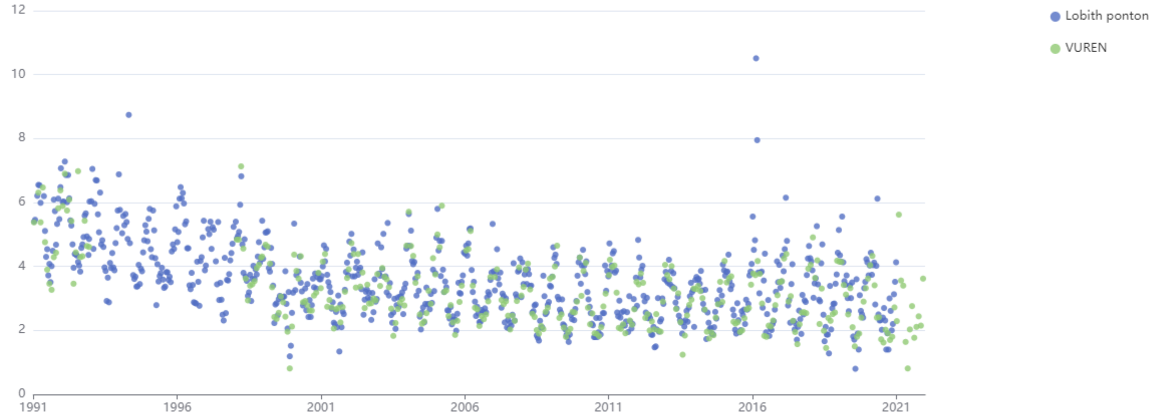
APPENDIX A

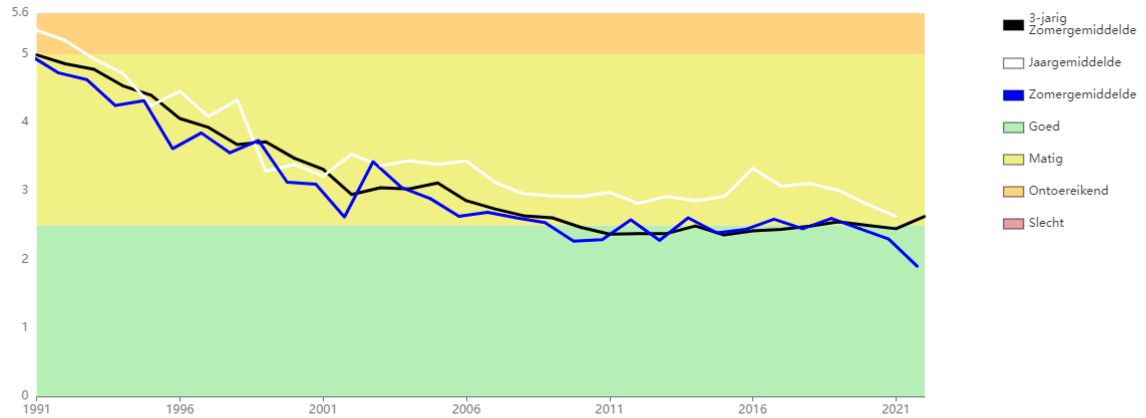
Nutrient concentrations and WFD status history of the upper Rhine (Lobith and Vuren)

Total N concentrations and WFD status

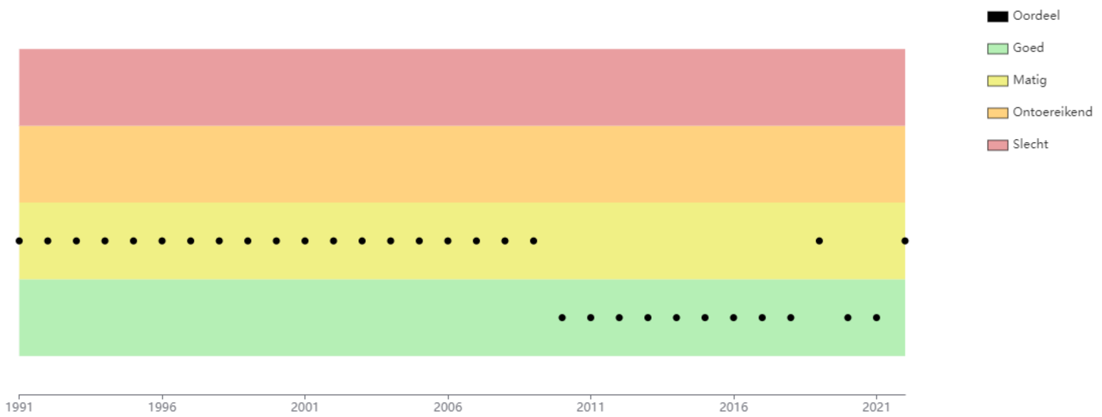
N Totaal NL93_8 (KRW monitoringslocatie in mg/l)



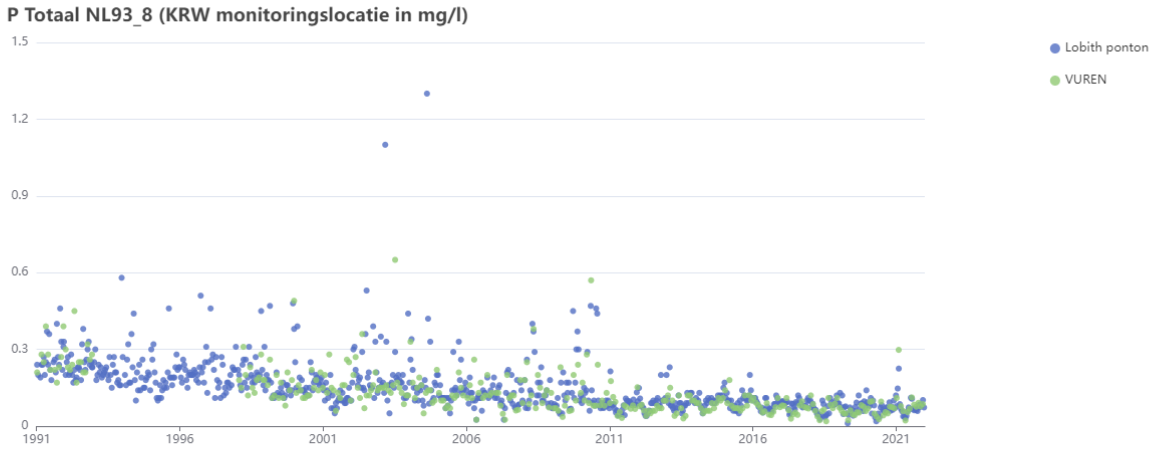
N Totaal NL93_8 (mg/l)

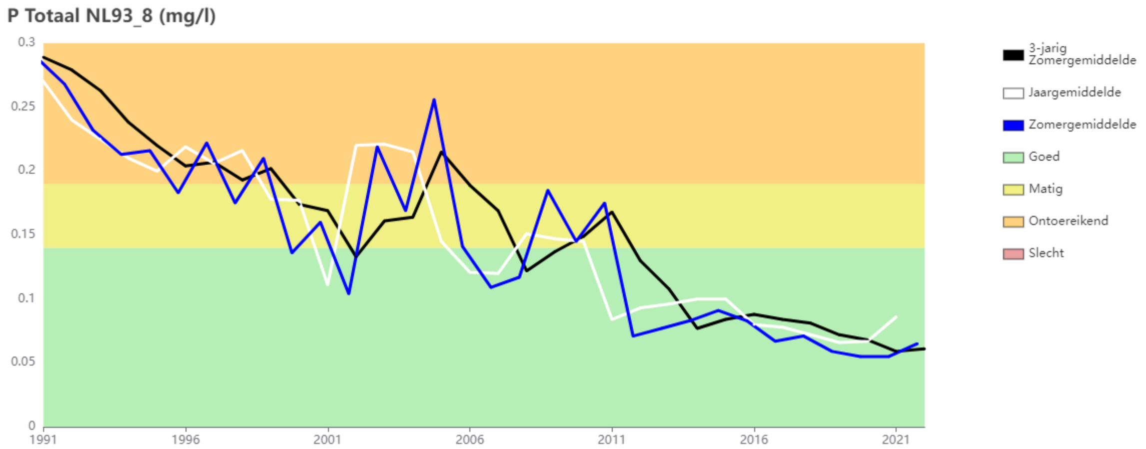


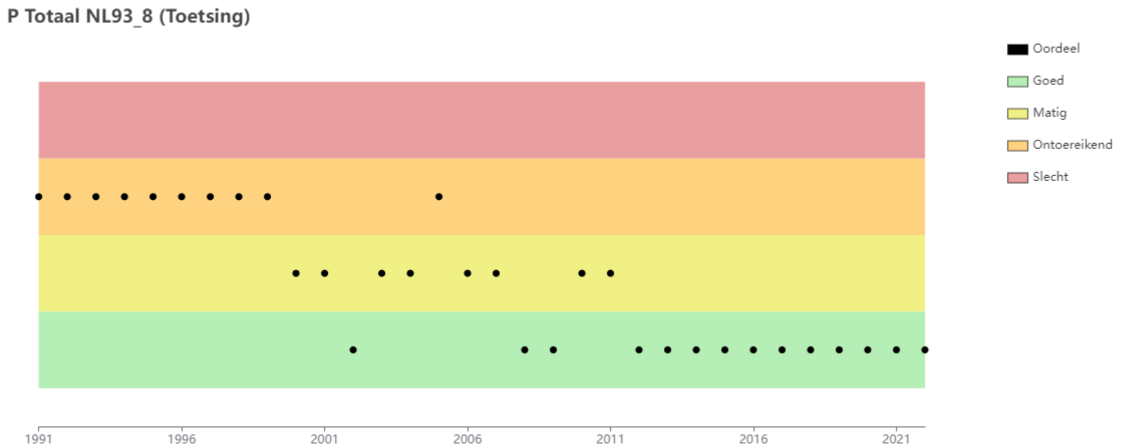
N Totaal NL93_8 (Toetsing)



Total P concentrations and WFD status



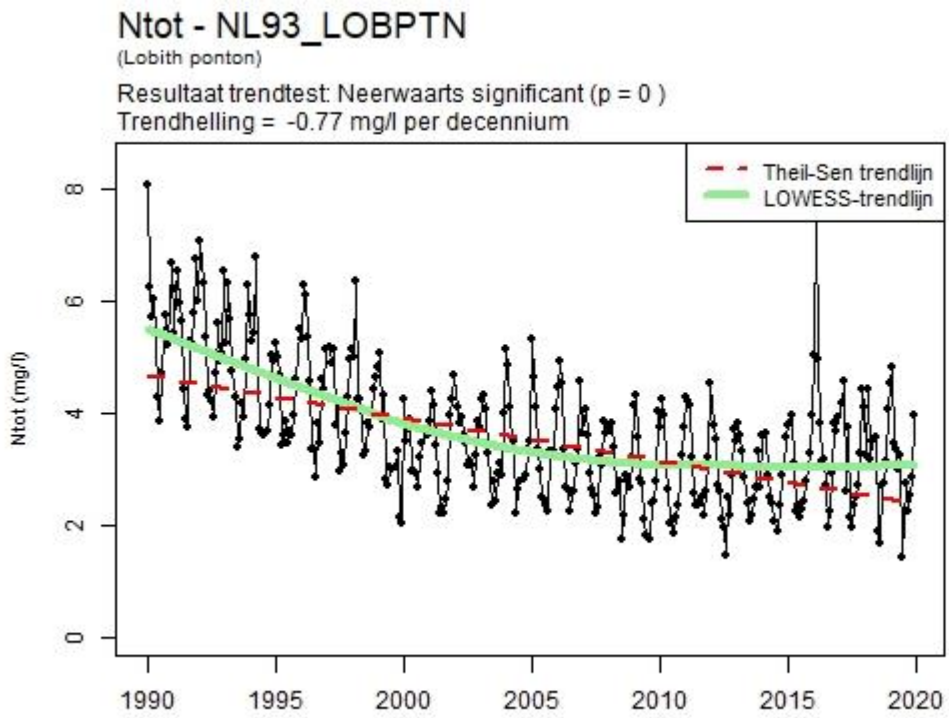


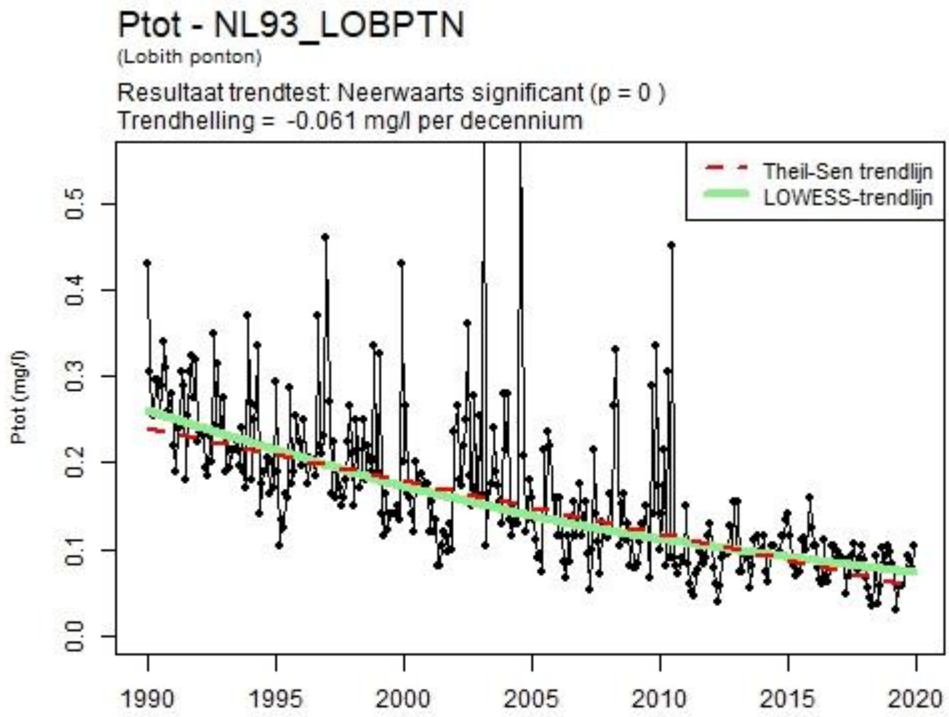


APPENDIX B

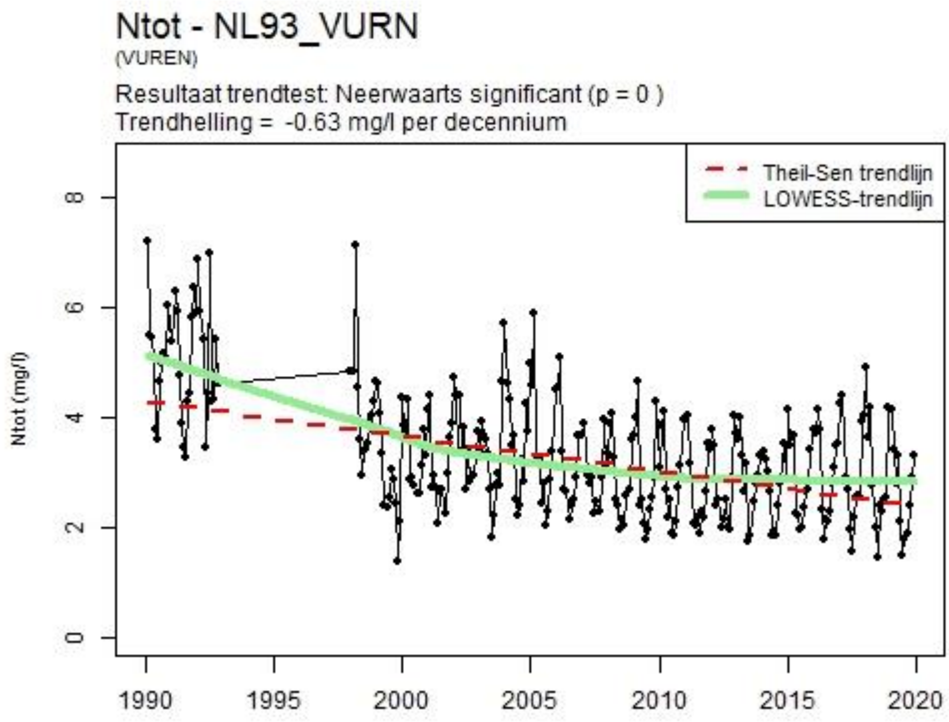
Trends assessment Total N and Total P for upper Rhine (Lobith and Vuren)

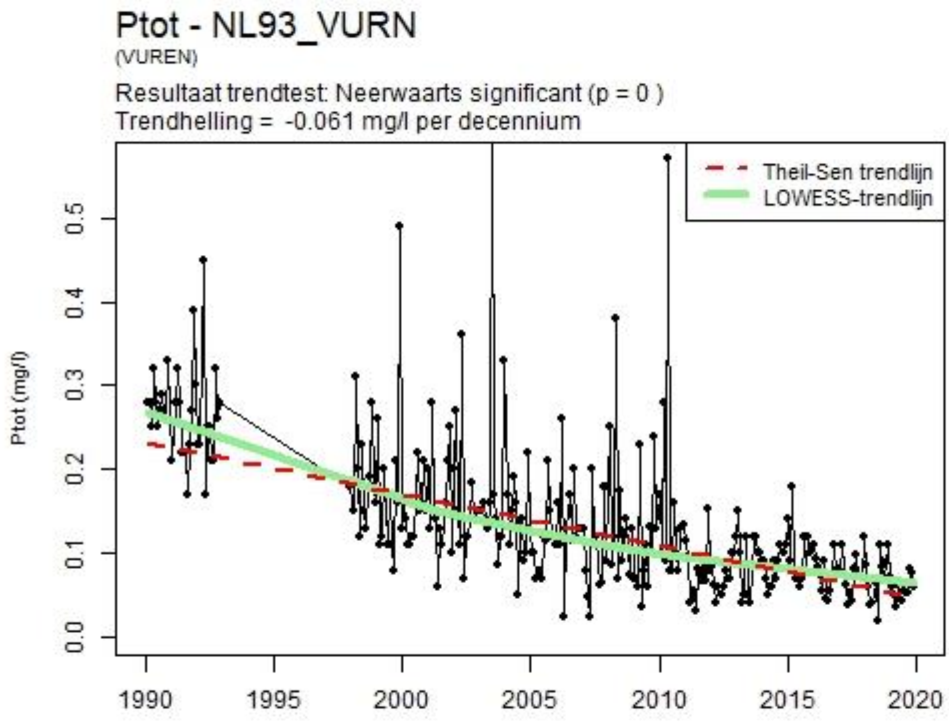
Lobith (NL/GE border)





Vuren (middle NL)

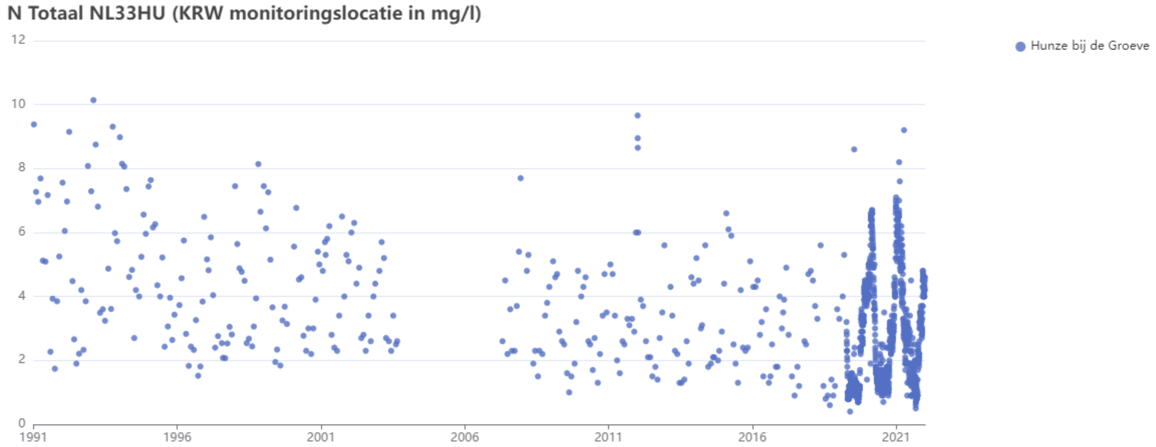


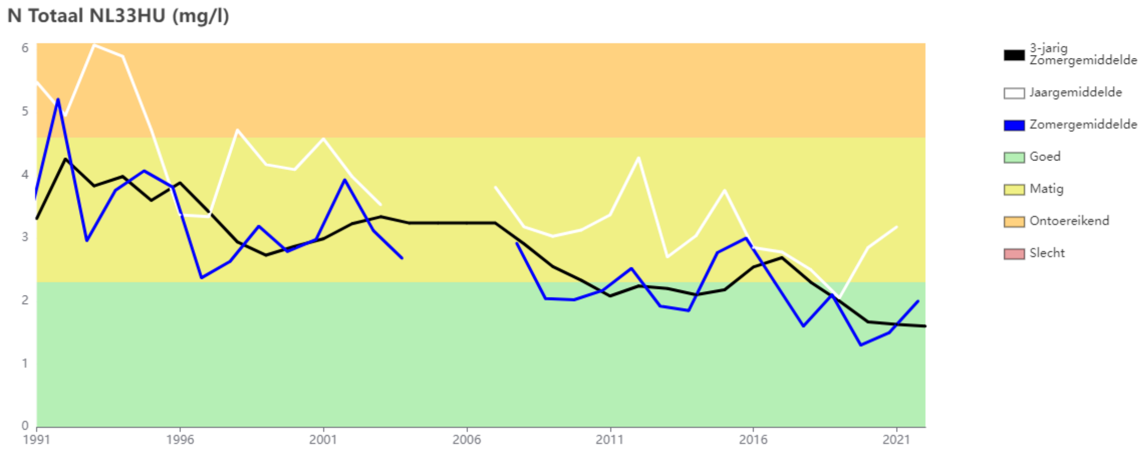


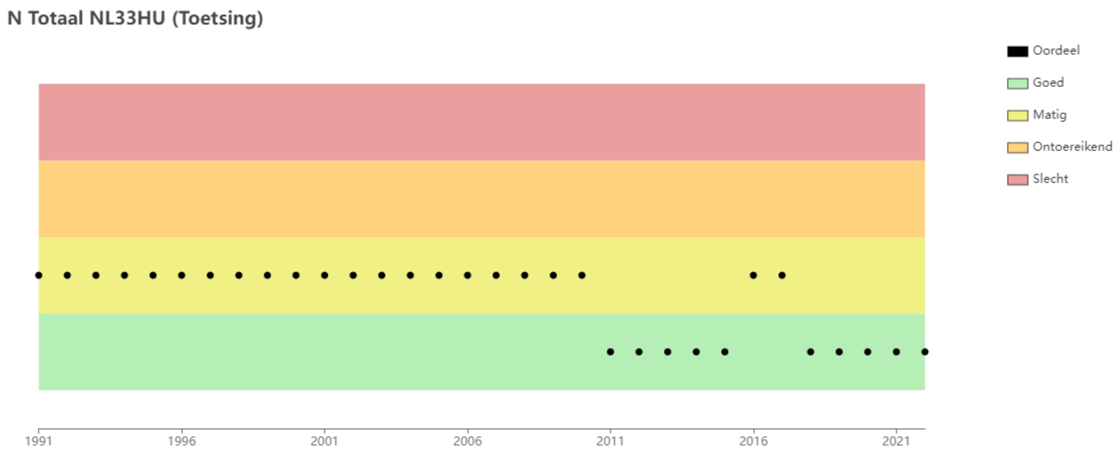
APPENDIX C

Nutrient concentrations and WFD status history of Hunze and Zuidlaardermeer

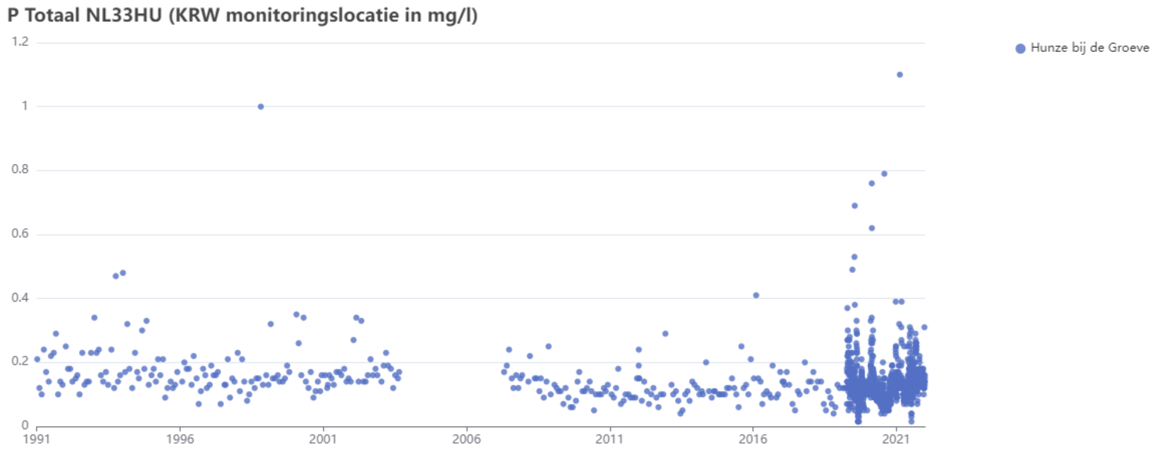
Total N concentrations and WFD status in Hunze

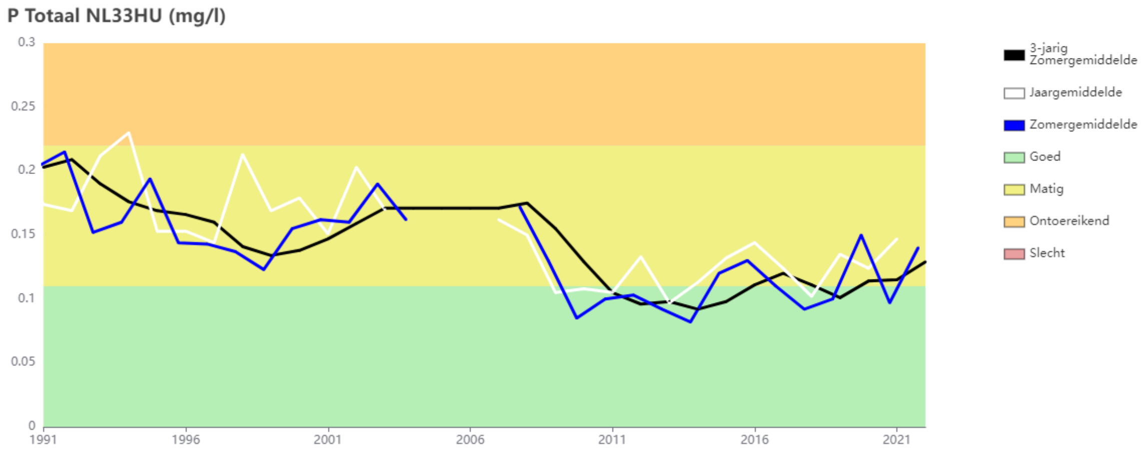


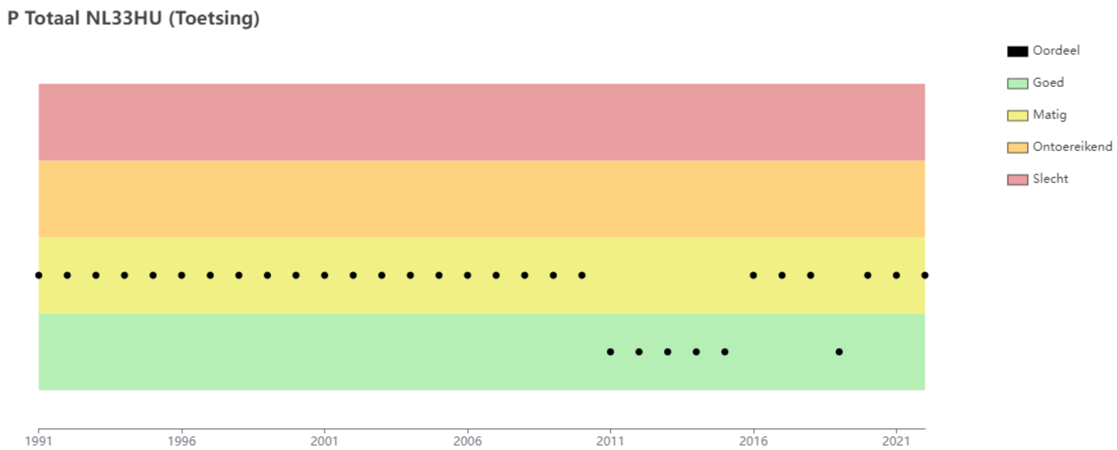




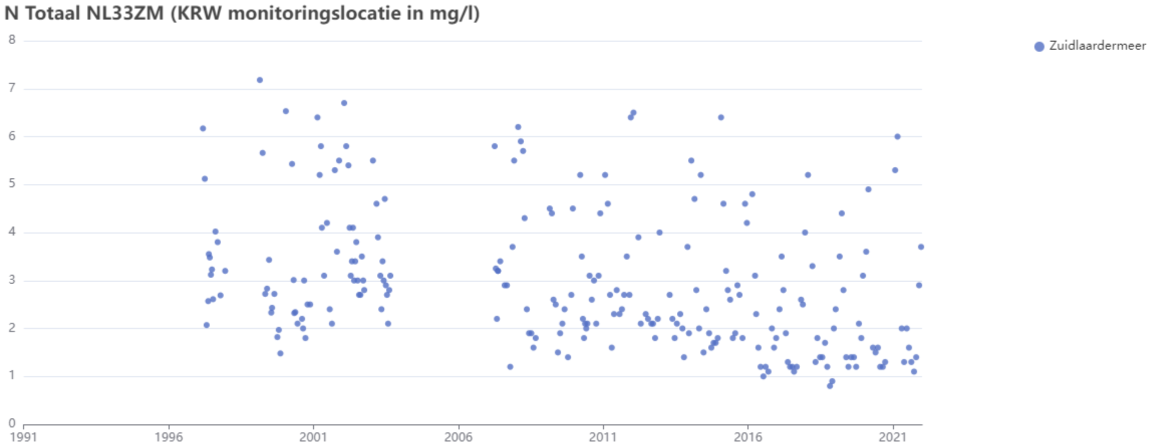
Total P concentrations and WFD status in Hunze

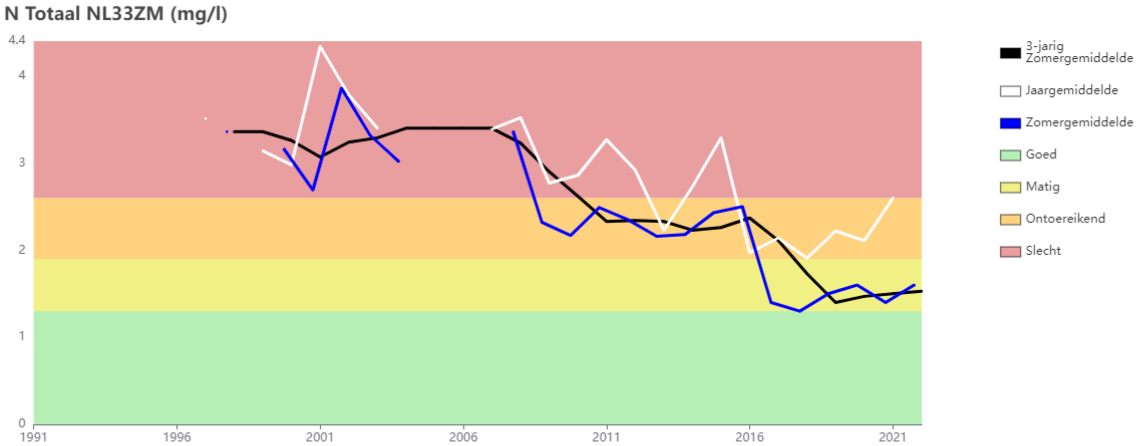


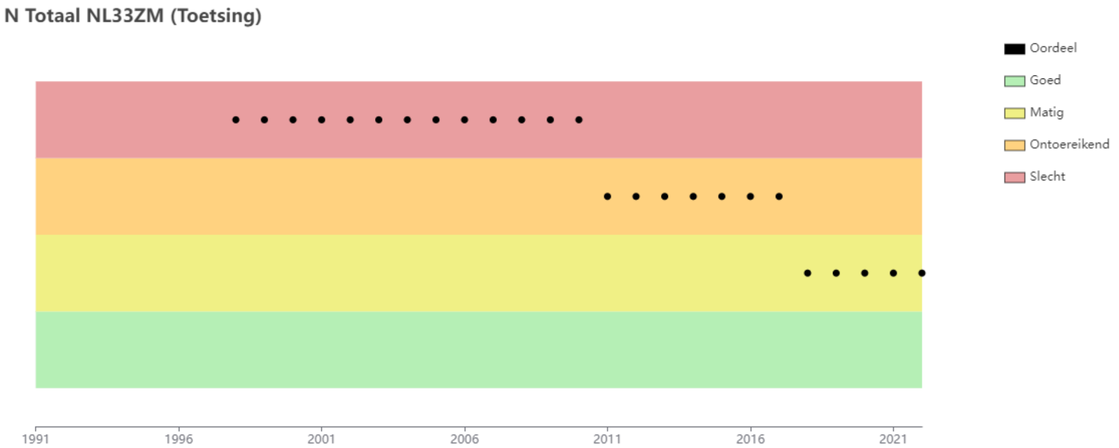




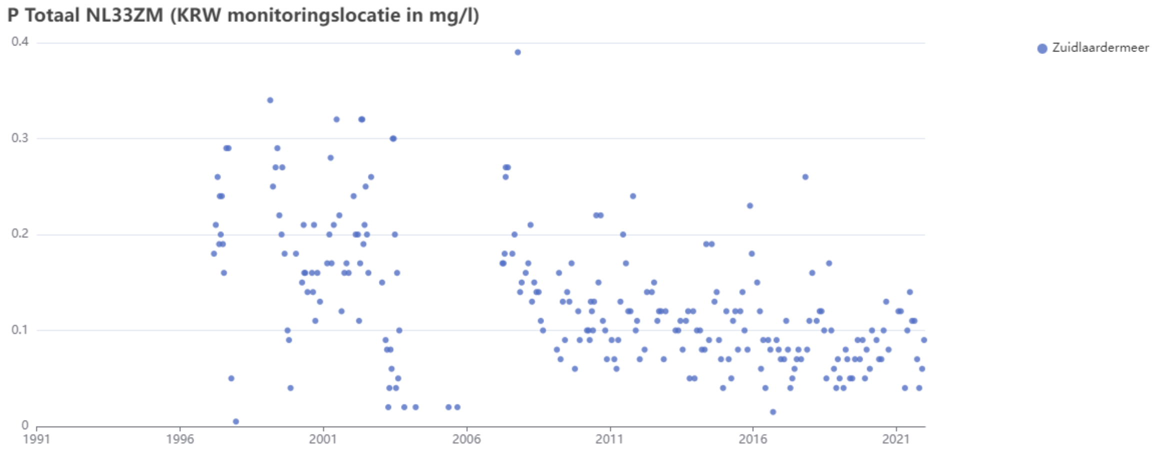
Total N concentrations and WFD status in Zuidlaardermeer

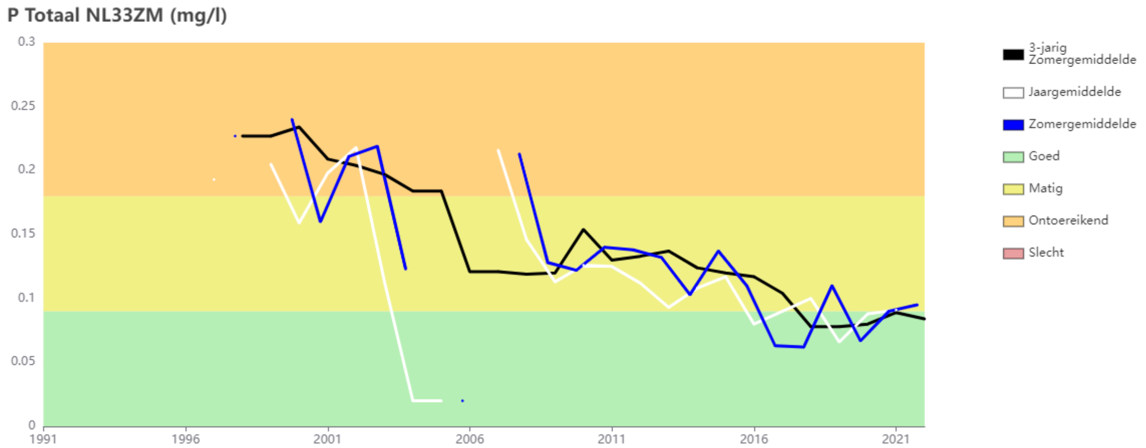


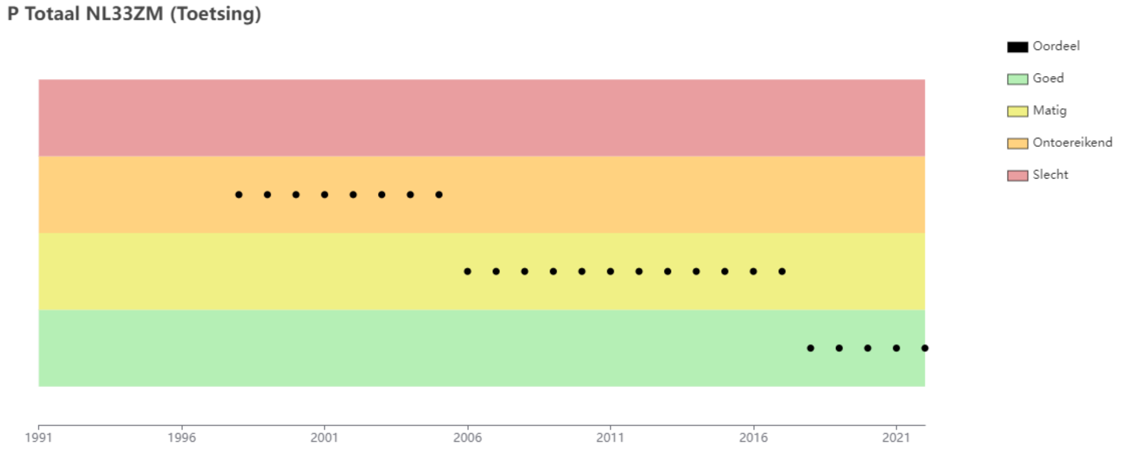




Total P concentrations and WFD status in Zuidlaardermeer

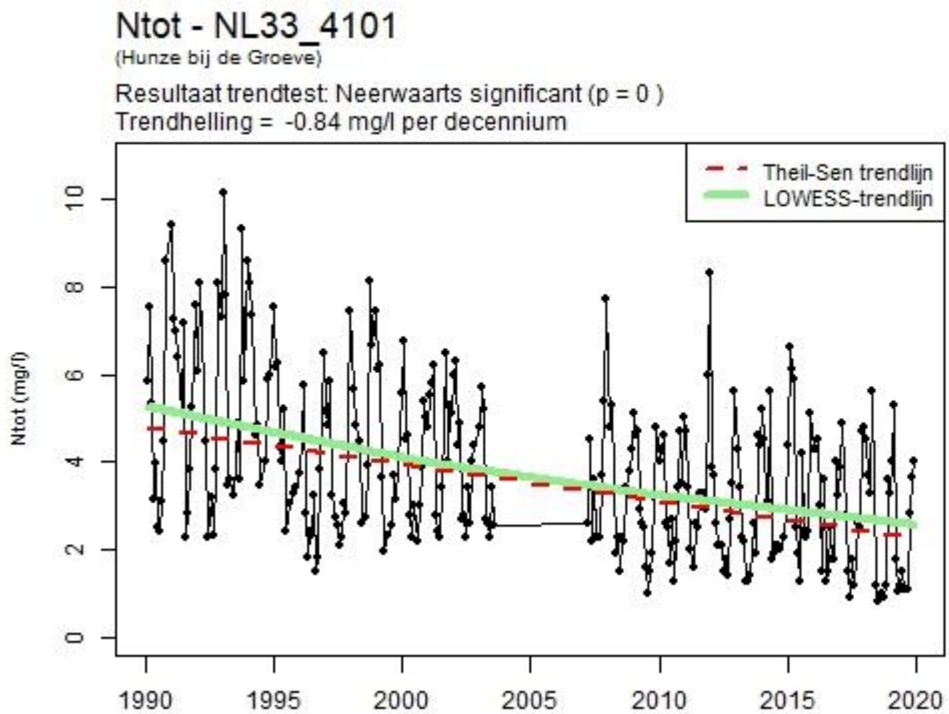


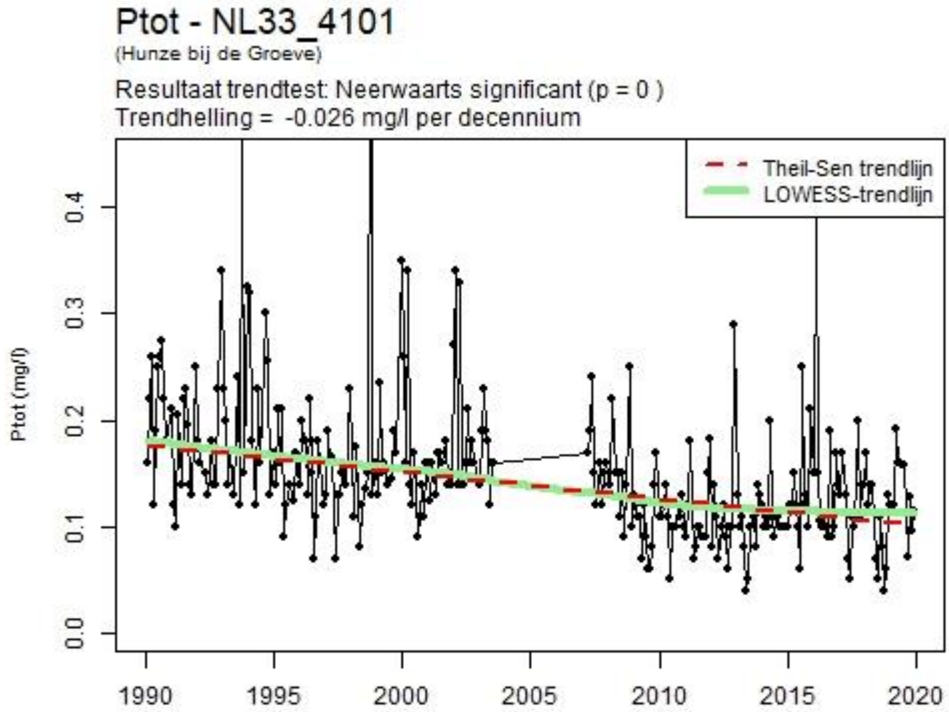


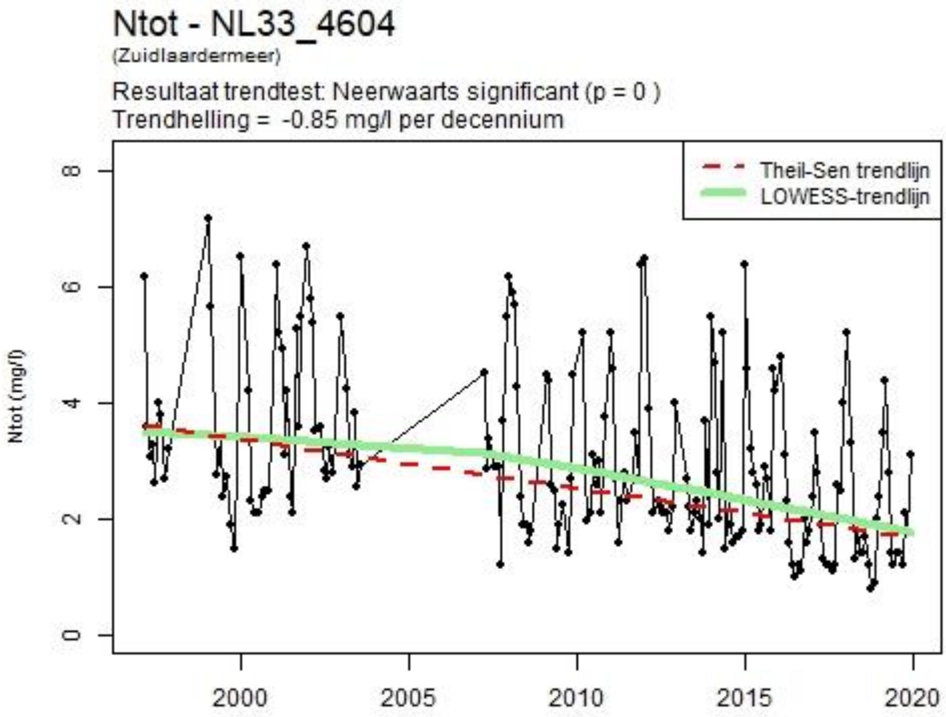


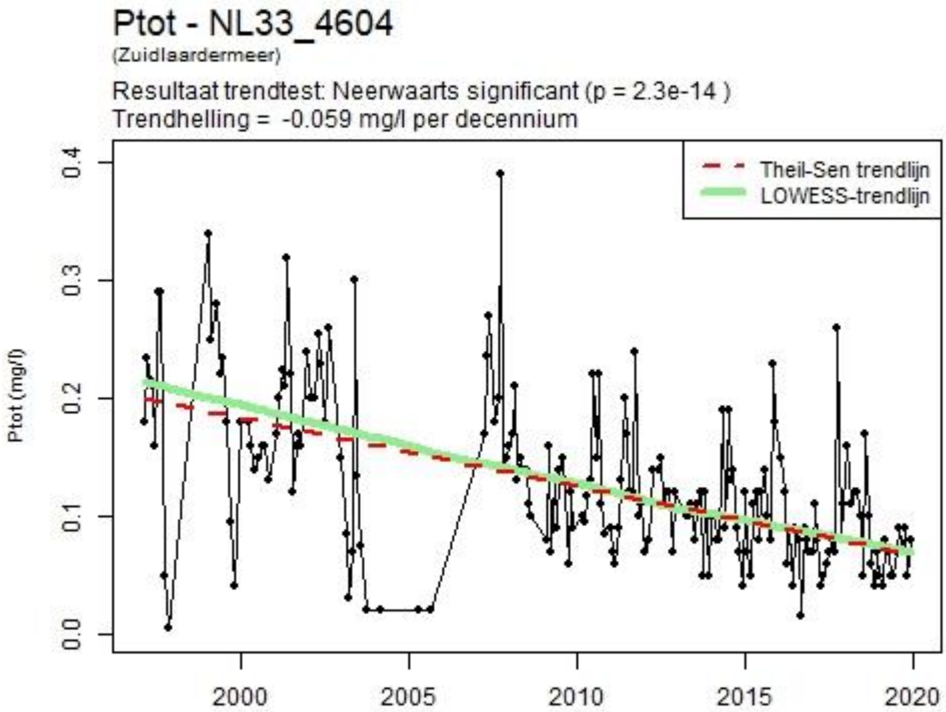
APPENDIX D

Trends assessment Total N and Total P for Hunze and Zuidlaardermeer






Ntot - NL33_4604
(Zuidlaardermeer)
Resultaat trendtest: Neerwaarts significant (p = 0)
Trendhelling = -0.85 mg/l per decennium
Theil-Sen trendlijn
LOWESS-trendlijn
Ntot (mg/l)
0
2
4
6
8
2000
2005
2010
2015
2020


Ptot - NL33_4604
(Zuidlaardermeer)
Resultaat trendtest: Neerwaarts significant (p = 2.3e-14)
Trendhelling = -0.059 mg/l per decennium
Theil-Sen trendlijn
LOWESS-trendlijn
Ptot (mg/l)
0.0
0.1
0.2
0.3
0.4
2000
2005
2010
2015
2020