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Are historical conditions reference conditions? Revising the modeled riverine nutrient input into the German North Sea and Baltic Sea around 1880

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Abstract

Background The German legislation sets two targets for riverine nitrogen concentrations in North Sea and Baltic Sea tributaries as well as river type-specific phosphorus thresholds. The current target for the Baltic Sea as well as the thresholds for the good status were derived from modeled riverine and atmospheric inputs around 1880. However, the calculated nitrogen balance differed between the model applications for the German North Sea and Baltic Sea. Existing nitrogen targets for North Sea tributaries are likely insufficient for environmental objectives according to recent model and data analyses. We used a harmonized approach to model nutrient inputs to the German Seas around 1880 and discuss these outcomes in the context of stricter requirements needed for ecological objectives in the North Sea.

Results For river basins entering the German North Sea and Baltic Sea, we modeled emissions, concentrations, and loads of total nitrogen and total phosphorus around 1880. The historical riverine inputs to the North Sea were 180 kt N yr^{-1} and 4.20 kt P yr^{-1} and to the Baltic Sea 22 kt N yr^{-1} and 0.45 kt P yr^{-1}, respectively. These loads corresponded to annual mean concentrations of 1.36 mg N l^{-1} and 0.032 mg P l^{-1} (North Sea) as well as 1.11 mg N l^{-1} and 0.022 mg P l^{-1} (Baltic Sea). Modeled nitrogen concentrations at river mouths were lower than the previous German model results but exceeded published reference concentrations. They were, however, partly in agreement with ecology-based concentrations for major North Sea tributaries based on published reduction needs.

Conclusions The modeled nutrient concentrations at river mouths confirm the inconsistency of German model applications with regional applications. For the North Sea, they support a more stringent basin-wide nitrogen target and thresholds for the good status of coastal and marine waters. As the historical conditions exceeded reference conditions, the offset of 50% to the historical concentration for the good ecological status should be revised for both sea basins. According to ecology-based target concentrations, only + 30% may be acceptable for North Sea tributaries, corresponding to 1.8 mg N l^{-1}. Any revision of the German legislation should acknowledge the inherent uncertainties.

Keywords Eutrophication, Good ecological status, Historical concentrations, Marine Strategy Framework Directive, MoRE model, LARSIM model, Nitrogen, Phosphorus, Target concentrations, Water Framework Directive

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Background

Anthropogenic eutrophication occurs, where the enrichment of mainly nitrogen (N) or phosphorus (P) yields negative ecological and societal consequences due to the excessive production of organic matter [1]. It is a key topic in marine studies around the world [2]. Its impacts on coastal and marine ecosystems and their ecosystem services have been studied extensively albeit rarely in combination with other pressures [3].

The Water Framework Directive (WFD) and the Marine Strategy Framework Directive (MSFD) of the European Union (EU) aim at achieving the good ecological and chemical status (WFD) and good environmental status (MSFD) of surface waters by applying holistic approaches [4, 5]. Both directives target the ecological impacts of nutrient emissions and nutrient inputs (henceforth "emission" refers to the amount of nutrients entering the river network and "input" to the total river load entering the sea) as one end of the pollution continuum [6]. They complement source-oriented policies on wastewater, agriculture, and atmospheric nitrogen emissions. Despite decades of regulation and management, eutrophication remains a major problem of terrestrial and marine waters in the EU with diffuse nutrient pollution being the major pressure [7–9]. Of the German coastal and marine water bodies, for instance, 87% in the North Sea and 100% in the Baltic Sea are still eutrophic according to the latest assessment of the MSFD [10]. Nonetheless, reducing the nutrient input via rivers and the atmosphere has contributed to improving the eutrophication status of both seas during the last decades [8, 11–13].

The WFD demands the "good ecological status" of water bodies according to a list of quality elements which (in)directly reflect eutrophication, such as the composition and abundance of phytoplankton and supporting physico-chemical quality elements, such as nutrient and oxygen conditions. In contrast, the Descriptor 5 according to Annex I of the MSFD explicitly addresses the (minimization of) eutrophication and its ecological impacts. These goals are achieved under near-natural (reference) conditions. EU member states determined a range of criteria and threshold values specifically for their water-body types. Appropriate threshold values were derived, e.g., by analyzing pressure–response relationships, from expert judgement, or with models [14, 15]. This variety hampers the harmonization between countries [16]. For the North Sea and the Baltic Sea, various marine models were applied to derive coastal and marine reference conditions from (modeled) riverine nutrient loads (inputs) or concentrations (e.g., [17–22]) as data-based approaches were restricted by the lack of near-natural conditions and suitably long time series [23].

Germany is among the few EU countries without N targets for freshwaters [23] except for the monitoring stations at the transition between freshwater and brackish seawater at low tide or at the national border (henceforth termed "limnic–marine border"). The catchment model MONERIS [24] was applied to estimate the nutrient inputs in 1880 from rivers entering the North Sea and Baltic Sea [25–27]. The model results of Hirt et al. [26] complemented previous estimates for the Baltic Sea [28]. They served as the basis for the current German target annual average concentration of 2.6 mg N l^{-1} at the limnic–marine border to achieve the good ecological status of the Baltic Sea ([19], Table 1). In contrast, the target annual average concentration of 2.8 mg N l^{-1} for German rivers to the North Sea was originally derived for river Rhine at the German–Dutch border from the Dutch threshold value for dissolved inorganic N in coastal waters [29]. Scenario analyses suggested later that the intercalibrated thresholds for the good/moderate boundary for the coastal and marine waters could be achieved if this target concentration was achieved at the limnic–marine border [30]. The historical MONERIS results [27] supported partly

Table 1 German approaches for riverine N targets and thresholds for the good/moderate boundary in coastal waters

	Baltic Sea 1880	North Sea 1880
Modeling approach for riverine inputs	MONERIS, N balance without N fixation [26]	MONERIS, N balance with N fixation [27]
Approach for reference conditions in coastal areas	ERGOM–MOM with atmospheric and riverine inputs, relative change 1880 to 2000–2008 applied to observed data [19]	Riverine nutrient concentrations interpolated along salinity gradient, 50% N retention in estuaries, chlorophyll-a estimated with relationship to total N (1980–2010) [31]
Good/moderate boundary in coastal waters	50% added to area-specific reference values for nutrients and chlorophyll-a	50% added to area-specific reference values for nutrients and chlorophyll-a
Management target at limnic–marine border	Maximum nutrient input calculated to achieve chlorophyll-a target, 2.6 mg N l^{-1} derived by assuming 20% lower atmospheric input, verified that orientation values for P sufficient	Previously derived as 2.8 mg N l^{-1} from the Dutch threshold value for dissolved inorganic N in coastal waters [29, 30], tested for agreement with good ecological status for chlorophyll-a

its good agreement with the German threshold values for chlorophyll-a in the German North Sea ([31, 32], Table 1).

For both the German North Sea and Baltic Sea, the historical nutrient inputs at the limnic–marine border were extrapolated to coastal and marine waters either using a modeling approach or by linear correlations along the salinity gradient to obtain reference conditions. 50% was added to the reference conditions of the coastal and marine areas to set the threshold for the good/moderate boundary following common practice [23, 33, 34]. The two riverine N targets were later translated to local N targets for inland waters for the German river basin management plans by considering the in-stream retention as modeled with MONERIS [35, 36]. They were also used to, e.g., derive the national N target of about 1000 kt N yr^{-1} [37].

According to earlier studies, the current German target value for river Rhine—the major tributary of the North Sea—suffices in fact only a few ecological objectives (cf. [38]). Likewise, the thresholds for the good ecological status for chlorophyll-a as derived from the historical nutrient input to the North Sea cannot be achieved everywhere as the retention of the N input in the estuaries is lower under current compared to historical conditions [31]. Recent data analyses and ecosystem modeling for the German–Dutch Wadden Sea supported these findings [39]. Furthermore, at least 33% lower N loads compared to 2010–2017 are needed in river Rhine to make seagrass permanently return in the southern and western Wadden Sea according to data analyses and habitat suitability models [40], although the river already meets the target concentration [41]. These analyses were supported by a broad evaluation of safe ecological boundaries which included the (absence of) algae blooms in the Wadden Sea, the O_2 level in the Elbe estuary, the phytoplankton blooms in the Elbe river, and riverine molar N:Si ratios of about 1 during winter to enable N-limited diatom blooms in coastal and marine waters ([42], see also [43]).

The existing modeled historical riverine inputs to the Baltic Sea and North Sea suffer from inconsistent modeling approaches, primarily regarding the estimation of agricultural N balances in the river basins. Unlike the Baltic Sea, N fixation by leguminous plants was considered for the North Sea which increased the average N balance from 1 to 23 kg ha^{-1} [27]. The authors also applied their approach to the Baltic Sea and obtained an average riverine N concentration of 1.2 mg N l^{-1} instead of 0.8 mg N l^{-1} with the original approach used by Hirt et al. [26] and, subsequently, for the German N target concentration. In addition, the P input to the North Sea modeled with MONERIS replaced the estimates of the E-HYPE model (European Hydrological Predictions for the Environment) for German rivers in the modeling of pre-eutrophic conditions for the whole North Sea around 1900 [21]. The substantial model deviations between MONERIS and E-HYPE were attributed to methodical aspects (model choice, scenario implementation) rather than the year of the historical scenario (1880 versus 1900) [20]. The significantly lower riverine P input according to MONERIS resulted in lower pre-eutrophic coastal concentrations of chlorophyll-a in German waters compared to the E-HYPE model. These discrepancies do not only exemplify the uncertainty in model results, but also support the criticism of setting thresholds based on catchment models used for arbitrary historical periods and of the notion that coastal waters were still close to reference conditions at the end of the nineteenth century (cf. [18, 44]).

Compared to other countries around the North Sea and Baltic Sea, using 1880 as reference year is early [45, 46]. The range from 1880 to the 1960s across these countries may reflect differences in data availability and national perceptions of pre-eutrophic water quality [14] rather than in the development of agriculture and industry. Like other European countries at this time, Germany underwent a phase of industrialization, urbanization, and the early beginnings of agricultural intensification [47]. Based on historical records, however, it was assumed that water transparency was still high and macrophytes abundant in German coastal waters [19, 48, 49]. Accordingly, signs of eutrophication were still scarce in the Baltic Sea and the Wadden Sea in the early twentieth century [40, 50].

The overall goals of this study were to reliably reconstruct the historical riverine emissions and inputs of N and P in a harmonized way and to discuss the deviations from previous model outcomes in the light of stricter targets required to achieve ecological objectives in the North Sea. Specifically, we addressed the questions: (1) are the model outcomes consistent with the existing basis for the German N targets and the reference conditions of German coastal areas? (2) Which deviations from the modeled historical N concentrations can we accept to achieve ecological objectives in the German North Sea and its tributaries? They are linked to the question whether the derived nutrient concentrations in the late nineteenth century indeed represent riverine and, thus, coastal reference conditions to which the common offset of 50% should be applied to derive thresholds for the good status of coastal and marine waters.

Since its original applications, the MONERIS approaches—re-implemented in the MoRE model (Modeling of Regionalized Emissions)—were not only extended to other pollutants [51] but also regularly applied, e.g., for the official reporting on nutrient sources to international entities. At the same time, MoRE was further developed which helped, e.g., to reduce the dependency on (historical) N balances [52]. With the revised modeling, we also strived for results on historical conditions being more consistent with current model applications and recent data sets as well as more consistent among the sea basins.

Materials and methods

General modeling approach

We applied the current version of the catchment model MoRE with a common database for the Baltic Sea and North Sea. MoRE quantifies the emissions of total N and total P to surface waters via different pathways as well as their concentrations and loads in surface waters [51]. The model is used for the national reporting to the regional sea conventions HELCOM and OSPAR and to quantify relevant nutrient sources for reporting under the MSFD. MoRE has been continuously improved (e.g., [51, 53, 54]) and was recently extended to integrate runoff components from the hydrological model LARSIM–ME (Large Area Runoff Simulation Model–Middle Europe) [53]. MoRE operates at an annual time step and was adapted to reflect average conditions for the year 1880 in this study.

The spatially distributed process-based model LARSIM was originally developed by Bremicker [55] and has been continuously further developed by various public authorities in Germany and neighboring countries (cf. [56, 57]). A LARSIM model for Central Europe (LARSIM–ME) was set up on behalf of the Federal Institute of Hydrology [58] and recently improved regarding the runoff separation into components needed for nutrient modeling [53]. We applied a LARSIM–ME configuration which is based on the meteorological forcing provided by the Federal Institute of Hydrology, and which was successfully validated for 1998 through 2018 [53].

Spatial input data

To reconstruct the historical conditions, we derived spatial model input from various international data sets (Table 2). For the hydrological conditions, we considered the differences of land use and imperviousness (i.e., land surface sealing) between 1880 and today [59, 60] as well as the change in meteorological forcing. LARSIM–ME could not be re-calibrated for the historical period due to the lack of reliable meteorological forcing and the small number of historical discharge data (cf. [56]). Therefore, to avoid model inconsistencies, we derived temporally and spatially averaged climate change vectors from long-term reanalysis data, which cover the historical and validation periods (Table 2). The vectors were derived from the differences between the 15-year periods for 1873–1887 and 2001–2015. LARSIM–ME was re-run with the meteorological input adapted to the period 1873–1887. The change in land use and imperviousness was considered in the post-processing of the LARSIM results as described by Morling et al. [53], finally yielding the average historical water balance for MoRE.

Model assumptions and adaptations

The existing MoRE setup was adapted based on general assumptions and specific literature values.

Table 2 Spatial data sets used to update the model input data for historical conditions

Description (Unit)	Data set	Period[d]	Resolution	Source
Land use and cover (–)	HANZE 2.0.3	1880	100 m	[59, 60]
Imperviousness (%)				
Population (capita)				
Soil erosion outside Germany[a] (t ha^{-1} yr^{-1})	Historical soil erosion	1871–1889	0.125°	[61]
Atmospheric N deposition (kg ha^{-1} yr^{-1})	CCMI v2.0 for input4MIPs	1873–1887	2.5° · 1.9°	[62]
Aquifer type (–)[b]	IHME1500, v1.2	n/a	1:1,500,000	[63]
Precipitation (mm yr^{-1})	20CRv3 Reanalysis	1873–1887, 2001–2015	0.7°	[64]
Air temperature (°C)[c]				
Global radiation (W m^{-2})[c]				
Wind velocity (m s^{-1})[c]				
Relative humidity (%)[c]				
Atmospheric pressure (hPa)[c]				

[a] Complements the German soil-erosion data in MoRE

[b] As proxy (see text for details)

[c] Only for LARSIM–ME

[d] Multi-year periods were averaged

First, we assumed that agricultural soils were not systematically overfertilized and local livestock production was constrained by feed production, i.e., by available agricultural land. Consequently, the soil-surface N balances of agricultural land were commonly negative [65] and groundwater generally unaffected by nitrate [66]. Based on the latter study, the recent groundwater N concentrations [52] were limited to a maximum of 2.257 mg l^{-1}, i.e., the threshold of 10 mg NO_3^- l^{-1} for groundwater considered as unaffected by human impacts. We also used 2.257 mg l^{-1} as tile drainage concentration, thus replacing the estimation from N balance for the latter. The average share of tile drainage on agricultural land around 1880 was lowered to 6% on average [27]. For P, we assumed that its concentration in groundwater and tile drainage is determined by geology rather than agriculture given its sorption to soil particles. Without historical data, we used the current concentrations [54] with a Germany-wide mean of 0.022 mg P l^{-1} which is close to the historical MONERIS application (e.g., [26]). The N and P concentrations were extrapolated to areas outside Germany using the average values per aquifer type (Table 2).

Second, the change in soil erosion depended presumably mostly on changes in land cover, tillage practice, and climate, while factors such as soil type and crop rotation were less important. Similar to the erosion map used for foreign areas (Table 2), we adjusted the C and R factors of the universal soil loss equation for Germany [54]. Compared to today, for instance, arable land was more abundant than grassland, also on steeper terrain. A supposedly higher erosion risk (S factor) was counter-balanced by smaller fields (L factor). We assumed 100% conventional tillage around 1880 and increased the current C factor—which reflects the actual share of conservation tillage [54]—by 41% based on the Germany-wide C factor calculated by Wurbs and Steininger [67]. As the current estimation of R factors reflects the increase in rainfall erosivity between 2001 and 2017 due to climate change (e.g., [68]), we used the empirical regression model of the previous German norm DIN 19708 (2017) with the mean annual precipitation for 1873–1887 as derived for LARSIM–ME. We, therefore, implicitly assumed that the statistical relationship of rainfall amount and rainfall erosivity for the period 1960–1990 holds valid for the late nineteenth century. In Germany, the sediment delivery ratio was estimated after Fuchs et al. [54] and elsewhere with the original MoRE approach based on Venohr et al. [24].

Third, fertilizer application likely changed the topsoil nutrient concentration since the late nineteenth century. The few available long-term experiments provide only site-specific data, which cannot untangle the complex interactions of soil properties, crop rotation, fertilizer application, and tillage over time and over different countries. As soil erosion is of minor relevance for N, we used the current input data of N content [53]. Following Ringeval et al. [69], who initialized their global reconstruction of P in agricultural soils since 1900 with recent P contents in (semi-)natural soils, we used the current P content of naturally covered areas [54] for historical agricultural land.

Fourth, we considered population density as a suitable proxy for the sewer-connection rate in urban areas following, e.g., Blauw et al. [45]. Sewer systems were gradually built in the rapidly growing urban areas at that time [70]. We assumed fixed connection rates for three classes of population density (< 10, 10–35, > 35 inhabitants ha^{-1}) which were split along the rounded minimum and median of the population of 37 agglomerations [71]. In contrast to Blauw et al. [45] for the year 1900, we assumed

Table 3 Adjusted model parameters with sources

Description	Value	Source
Atmospheric P deposition	75 g ha^{-1} yr^{-1}	[78]
N surface load from streets	4.49 kg ha^{-1} yr^{-1}	80 kg $capita^{-1}$ yr^{-1} litter [79], population and urban area from HANZE 2.0.3 (Table 2)
P surface load from streets	1.64 kg ha^{-1} yr^{-1}	
Water consumption in rural and urban areas (population density)[a]	55 l $capita^{-1}$ d^{-1}, 97 l $capita^{-1}$ d^{-1} (10–35, > 35 inhabitants ha^{-1})	[80]
Population connected to sewers (population density)[a]	0%, 30%, 80% (< 10, 10–35, > 35 inhabitants ha^{-1})	Based on HANZE 2.0.3 population density (Table 2)
Contribution from fertilizer application to concentration in surface runoff	0.1 mg N l^{-1}	Equal to currently implemented value for alpine areas
P concentration in surface runoff from vegetated areas	0.0125 mg P l^{-1}	average P leaching from unproductive vegetation [81]
Change in water temperature to 1998–2005	− 1 K	[82], approximated from change in air temperature

[a] The connection rates of urban population to sewers depended on population density (see main text for class boundaries)

that even in big cities not everyone was already connected to sewers around 1880 (Table 3). Very few cities treated wastewater by intermittent soil filtration on sewage farms [70, 72], the largest and most prominent one being located in Berlin, where we applied soil-retention rates of 65% and 90% for N and P. Elsewhere, collected wastewater entered the surface waters untreated [73, 74]. Human excreta collected in cesspits either infiltrated the soil or were still widely re-used as fertilizer [75, 76] and, thus, were part of the assumed nutrient concentrations in groundwater. Although industrial effluents were known to affect water quality (e.g., [77]), quantitative and qualitative data were unavailable in sufficient manner. Like in the previous modeling approaches, industrial emissions were neglected, i.e., assumed to be negligible for the overall nutrient emissions.

In addition, various model parameters were revised based on literature values (Table 3). The current settings for human excretion rates (12.0 g N capita^{-1} d^{-1}, 1.9 g P capita^{-1} d^{-1}) were found to be within the range of historical literature values and left unchanged. The water temperature was assumed to be 1 K lower than the period 1998–2005 due to the lower air temperature.

Study area

The study area comprised the basins of German rivers entering the North Sea of about 437,500 km^2 and the Baltic Sea of about 144,500 km^2 (excluding the Szczecin Lagoon, Fig. 1). The total area of 582,000 km^2 was subdivided into 3048 analytical units of variable area for MoRE (< 0.01–17,600 km^2, average 191 km^2).

Model evaluation and nutrient targets to achieve ecological objectives

We selected for each station at the limnic–marine border and the Rhine at the German–Dutch border which Germany reports to the EU [41, 83] the nearest MoRE analytical unit along the same river and manually verified the assignment by comparing reported and modeled catchment areas. Likewise, we assigned 11 discharge stations from the Global Runoff Data Center [84] which covered the historical period and the calibration period of LARSIM–ME sufficiently, i.e., less than 3 years were lacking (Fig. 1, Table 4). Annual runoff (q, mm yr^{-1}) was calculated from the original daily data if less than 15 days were missing. We calculated the percentage bias and the explained variance of the spatial variability (r^2) to assess the general agreement of observation and model results.

The modeled historical nutrient concentrations were compared to published reference concentrations and current concentrations as well as concentrations required to achieve ecological objectives in the North Sea basin. First, pristine values were derived by Hirt al. [26] from molarity originally collected by Topcu et al. [18]. However, we replaced the reported background concentrations from Behrendt et al. [85] with the values

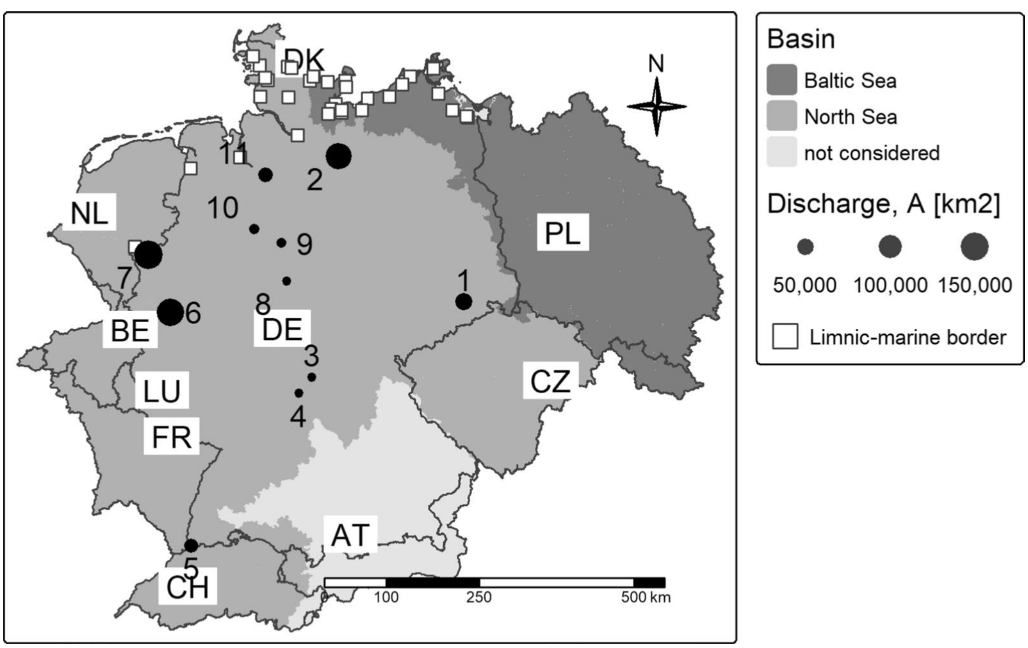

Fig. 1 Study area including monitoring stations at the limnic–marine border and with historical discharge. The point sizes depict the catchment area A of the stations; numbers refer to Table 4. Upper Danube basin and Szczecin Lagoon were modeled, but out of the scope of this paper. Country codes according to ISO 3166-1 alpha-2

Table 4 Available monitoring stations for evaluating the modeled average historical annual runoff (Source: [84])

Number[a]	Station Id	River	Station	Catchment area km^2	Observed $mm\ yr^{-1}$	MoRE input $mm\ yr^{-1}$
1	6340120	Elbe	Dresden	53,096	174.7	178.7
2	6340110	Elbe	Neu Darchau	131,950	172.0	172.2
3	6335301	Main	Schweinfurt, Neuer Hafen	12,715	275.7	301.6
4	6335500	Main	Würzburg	14,031	252.8	292.1
5	6935051	Rhine	Basel, Rheinhalle	35,897	884.0	777.2
6	6335060	Rhine	Köln	144,232	483.4	427.5
7	6335020	Rhine	Rees	159,300	456.1	423.1
8	6337400	Weser	Hannoversch–Münden	12,442	279.5	312.4
9	6337514	Weser	Bodenwerder	15,924	303.0	315.5
10	6337100	Weser	Vlotho	17,618	322.7	322.6
11	6337200	Weser	Intschede	37,720	279.9	287.3

[a] Corresponds to Fig. 1

Table 5 Reduction needs for German tributaries of the Wadden Sea compared to 2010–2017 (after [42])

Indicator	Main driver	Case study	River	Reduction needs, %
Sea grass recovery	River load of total N	Wadden Sea	Rhine	N: 34–43
			Rhine	N: 39–46
			Ems	N: 39–46
Absence of blooms by non-silicifying algae	Si:N ratio	Rhine	Rhine	N: 50
		Ems	Ems	N: 55
		Weser	Weser	N: 40
		Elbe	Elbe	N: 30
$O_2 > 7\ mg\ l^{-1}$	Import riverine organic matter	Elbe estuary	Elbe	N: 45
Phytoplankton biomass < $40.1\ \mu g\ l^{-1}$	Mainly P load (role N unknown)	Elbe river	Elbe	N: 63 P: 63

from the original study. Second, the most recent official German reporting for MSFD to the EU [41, 83] provided observed multi-annual average N and P concentrations for the stations at the limnic–marine border for the periods 2011–2015 and 2016–2020. The latter period was used to obtain the current concentrations. From the first period, we approximated mean annual ecological target concentrations for the tributaries to the North Sea. We assumed that the reduction needs to achieve safe ecological boundaries as derived by van Beusekom et al. [42] for the years 2010–2017 are representative for our period (Table 5). These values were complemented by earlier ecological N targets derived for river Rhine [38] (Table 6). In case of value ranges, we estimated the average value as the mean of the upper and lower boundary.

Given the inherent uncertainty of our large-scale model application, we focused on the sea basins rather than individual river basins and primarily discuss, e.g., flow-weighted average and median concentrations.

Table 6 Target concentrations for river Rhine based on various area-specific ecological objectives (after [38])

Area	Objective	Target, $mg\ N\ l^{-1}$
Coastal	Natural concentration	0.6
Coastal	50% algal biomass reduction in spring	1.8
Coastal	25% annual algal biomass reduction	3.0
Coastal	No oxygen depletion in stratified parts	3.0
Coastal	Maximum biomass of Phaeocystis < 5 µg chlorophyll-a l^{-1}	1.8
Coastal	N limitation in the North Sea	1.8
IJsselmeer	No dominance of blue–green algae	1.4
river Rhine	N-limited algal growth in river Rhine (chlorophyll-a < 25–40 µg l^{-1})	1.0
river Rhine	50% reduction of emissions	2.7–3.0
river Rhine	No N surplus (natural N:P ratio)	1.9

Results

Historical runoff and nutrient emissions into surface waters

Based on the climate data, the historical climate

1873–1887 was—on an annual time scale—characterized by significantly lower mean air temperature (−1.47 K) and, to a lesser degree, by lower global radiation and wind velocity (relative changes of 0.97 and 0.94) compared to the period 2001–2015. All other meteorological parameters in Table 2 changed negligibly across the study area. According to LARSIM–ME, these differences in climate conditions resulted in lower evapotranspiration and, consequently, led to an increase of discharge from unsealed areas in the historical period compared to 2001–2015. On the other hand, the lower degree of imperviousness in the nineteenth century resulted in lower modeled urban discharge compared to today. This effect was largest for the nowadays highly urbanized areas. On average, the modeled water discharge was 302 mm yr^{-1} (4188 $m^3 s^{-1}$) in the North Sea basin and 138 mm yr^{-1} in the Baltic Sea basin (632 $m^3 s^{-1}$, cf. lines in Fig. 3A).

The modeled total N emissions into surface waters ranged from 36 kt yr^{-1} (2.48 kg ha^{-1} yr^{-1}) in the basin of the Baltic Sea to 257 kt yr^{-1} (5.88 kg ha^{-1} yr^{-1}) in the basin of the North Sea (Figs. 2 and 3B, as well as the Supplementary Material A1). The dominant pathways in both sea basins were groundwater (47–49%) and interflow (28–35%, Fig. 2, left). Likewise, the total P emission was lower in the basin of the Baltic Sea than the North Sea, both in absolute (1.07 kt yr^{-1} and 8.21 kt yr^{-1}) and area-specific values (0.074 kg ha^{-1} yr^{-1} and 0.188 kg ha^{-1} yr^{-1}). In both sea basins, soil erosion was the dominant pathway (38–49%) followed by urban systems (23–25%, Fig. 2, right).

The modeled area-specific runoff and nutrient emissions differed widely across the river basins (Fig. 3A, B) which reflects differences in climate, terrain, and urbanization. For instance, the values for the Alpine river Rhine as the largest river basin were more than twice the values for the predominantly lowland Elbe River basin, which is the second largest.

Historical nutrient concentrations, loads, and inputs

After entering the stream network, 11–34% (flow-weighted average 30%) of the N emissions were retained in the river catchments entering the North Sea (cf. Fig. 3), and 35–39% (39%) entering the Baltic Sea. The modeled P retention was higher, ranging from 27% to 52% (49%) in the North Sea basins and from 38% to 62% (58%) in the Baltic Sea basin. The higher retention in the Baltic Sea basin corresponds to the generally lower runoff (i.e., lower hydraulic

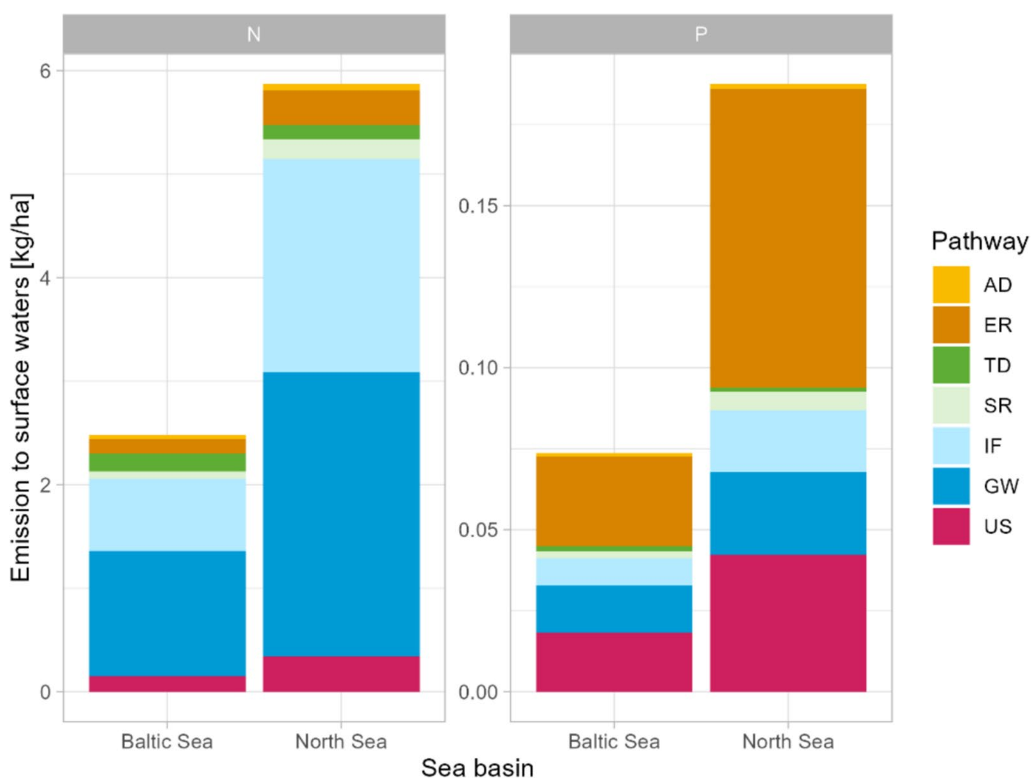

Fig. 2 Modeled area-specific emissions of total N (left) and total P (right) to surface waters around 1880. Colors depict the nutrient pathways atmospheric deposition (AD), soil erosion (ER), tile drainage (TD), surface runoff (SR), interflow (IF), groundwater (GW), and urban systems (US). See the Supplementary Material A1 for the absolute values

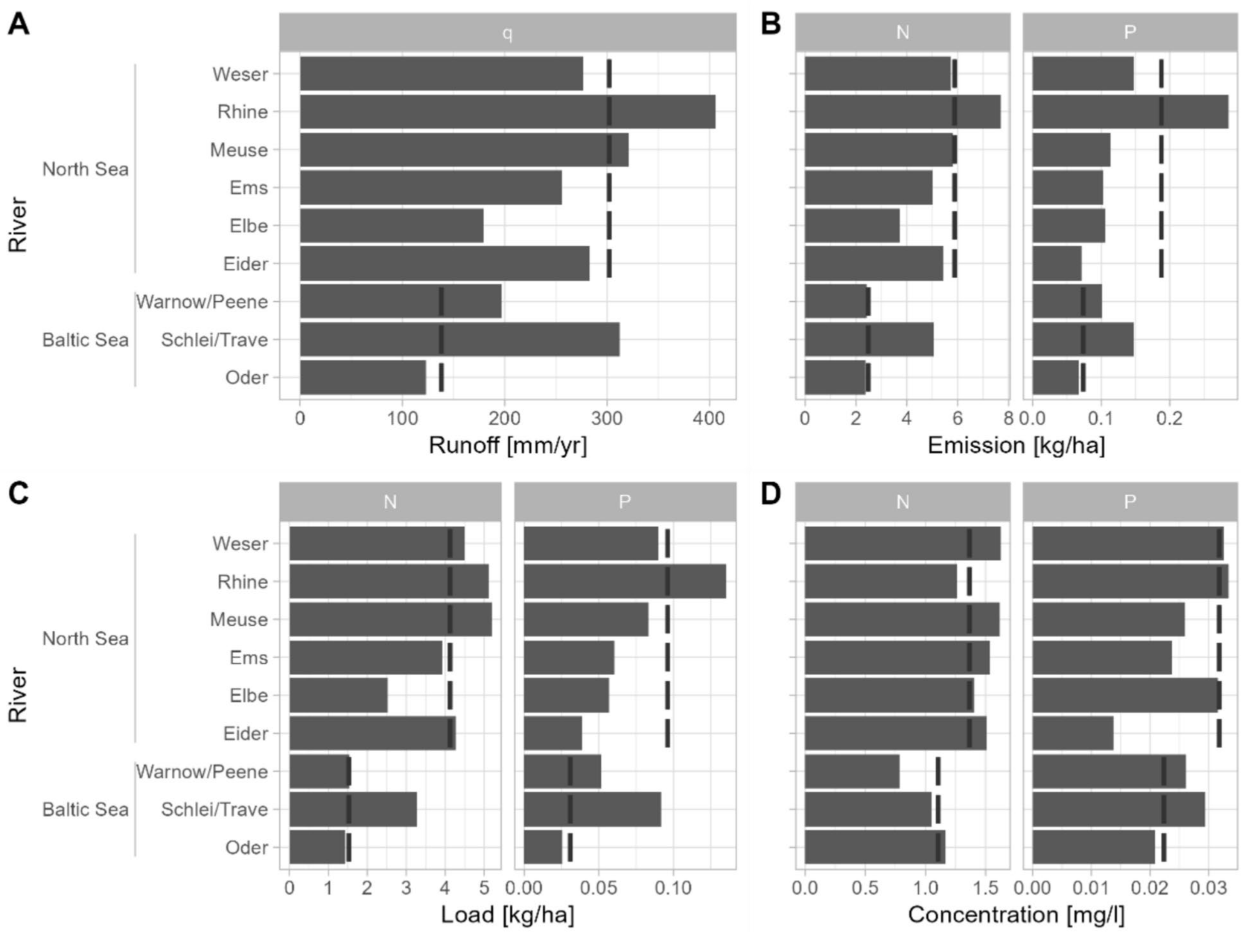

Fig. 3 Modeled results for river basins around 1880. The panels show the runoff (**A**), area-specific emissions (**B**) and loads (**C**), as well as concentrations (**D**) of total N and total P across the different river basins with broken lines showing the average values for the sea basins. Note the different value ranges of the x axes

load) compared to the North Sea basin. Consequently, the modeled overall riverine inputs to the North Sea around 1880 were 180 kt N yr^{-1} and 4.20 kt P yr^{-1} and the inputs to the Baltic Sea 22 kt N yr^{-1} and 0.45 kt P yr^{-1} (cf. Supplementary Material A1). These values correspond to area-specific N inputs of 4.12 kg ha^{-1} yr^{-1} and 1.53 kg ha^{-1} yr^{-1} as well as P inputs of 0.10 kg ha^{-1} yr^{-1} and 0.03 kg ha^{-1} yr^{-1} (lines in Fig. 3C). The flow-weighted average concentration of total N was 1.11 mg N l^{-1} in tributaries of the Baltic Sea and 1.36 mg N l^{-1} in tributaries of the North Sea. The P concentrations were 0.022 mg l^{-1} and 0.032 mg P l^{-1} in the Baltic and North Sea basins, respectively (lines in Fig. 3D). In comparison with the area-specific loads, the variability of the nutrient concentrations was not only less pronounced, but the ranking of the rivers also changed (Fig. 3C, D). For instance, river Rhine had the lowest modeled N concentration within the North Sea basin but the second highest area-specific load.

Model evaluation

The LARSIM–ME model explained reasonably well the spatial distribution of the few available historical observed runoff data (Fig. 4, $r2 = 0.998$). The average bias was −1.9% with deviations ranging from −12 to +16%. The strongest underestimation occurred at the Rhine gauge Basel.

As expected, the modeled historical nutrient concentrations at the limnic–marine border were significantly below current values. They overlapped with the reported reference concentrations. However, modeled N concentrations were partly above these reference concentrations, especially in the North Sea basin (Fig. 5).

Discussion

Consistency with previous model results

The historical N emissions modeled in this study were generally between the emissions obtained previously with MONERIS using a N balance without [25, 26] (only

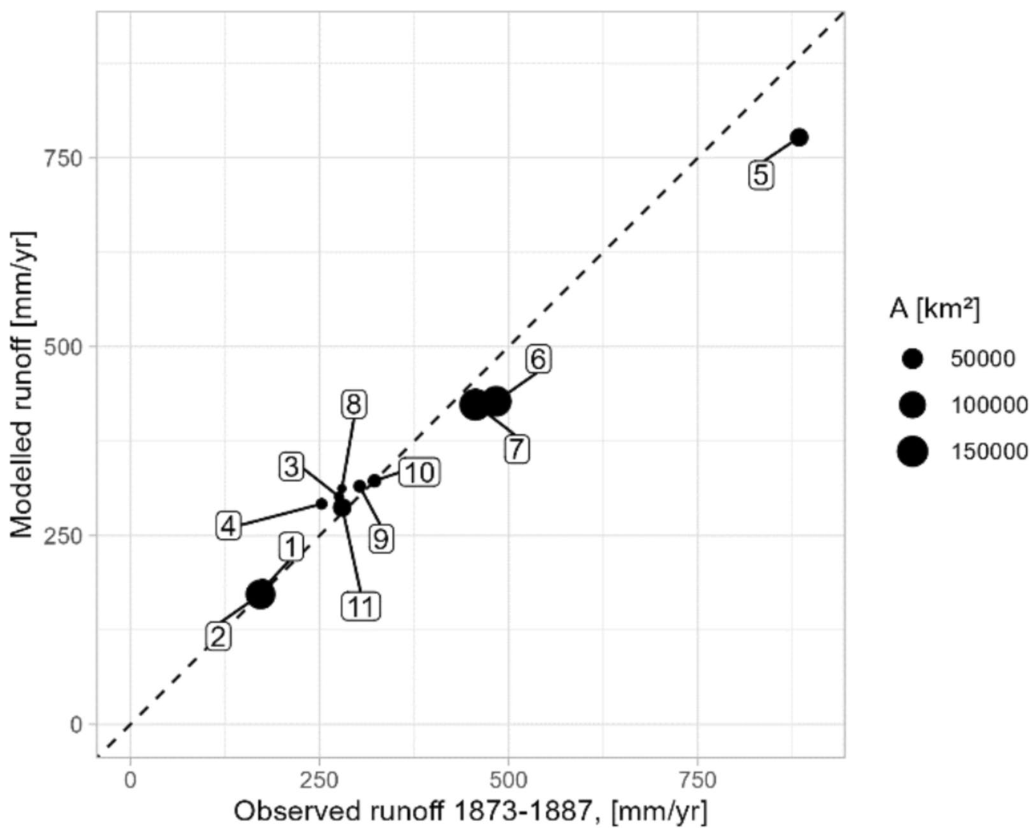

Fig. 4 Multi-year average observed and modeled runoff around 1880. Point sizes depict the catchment area A of the monitoring stations. Numbers refer to Table 4

Baltic Sea) and with N fixation [27] (Table 7). As MONERIS derived the N emissions via groundwater from these N balances [24], the significant deviations in estimated N emissions and N concentrations between these model applications confirmed the general importance of this pathway for the N emissions in Central Europe. However, the model choice affected the regional pattern of how important individual pathways are. For instance, the dominant pathway agreed with Hirt et al. [26] as well as Gadegast and Venohr [27], while Gadegast et al. [25] observed that urban systems were dominant in the Oder basin.

The deviating MONERIS results for the Baltic Sea also suggest that N fixation was a key element of historical N balances. Its omission resulted in the notion that the historical conditions still represented reference conditions (cf. [26]) which would have not been the case if N fixation was included (cf. the respective value ranges in Fig. 5 and Table 7). Coincidentally, our average riverine N concentrations were close to the MONERIS results without N fixation ([26], Table 7). The derived reference conditions of coastal and marine areas and, subsequently, the current German riverine N target of 2.6 mg N l^{-1} [19] are thus in principle valid despite the conceptual flaw. However, the lower revised riverine input to the North Sea compared to the previous historical modeling would result in lower thresholds for the good status in the North Sea if the original marine modeling was repeated as is. This expected decrease further strengthens the notion that the current N target of 2.8 mg N l^{-1} is too high (cf. [31]).

Despite the tendency to higher P emissions, flow-weighted concentrations at the outlets of the river basins were very similar to the previous model results with values below the P threshold of 0.05 mg P l^{-1} for the very good status of rivers according to the German Surface Water Protection Ordinance (Table 7). Regarding the dominant pathway, the model outcomes differed stronger from the previous results for P than for N. Instead of groundwater, soil erosion dominated the basin-wide P emissions for both sea basins according to our findings.

However, all these differences between the conceptually similar models MoRE and MONERIS were negligible compared to regional [45] and global estimations [86] for the year 1900. The discrepancies of our model to the E-HYPE and IMAGE–GNM models were higher

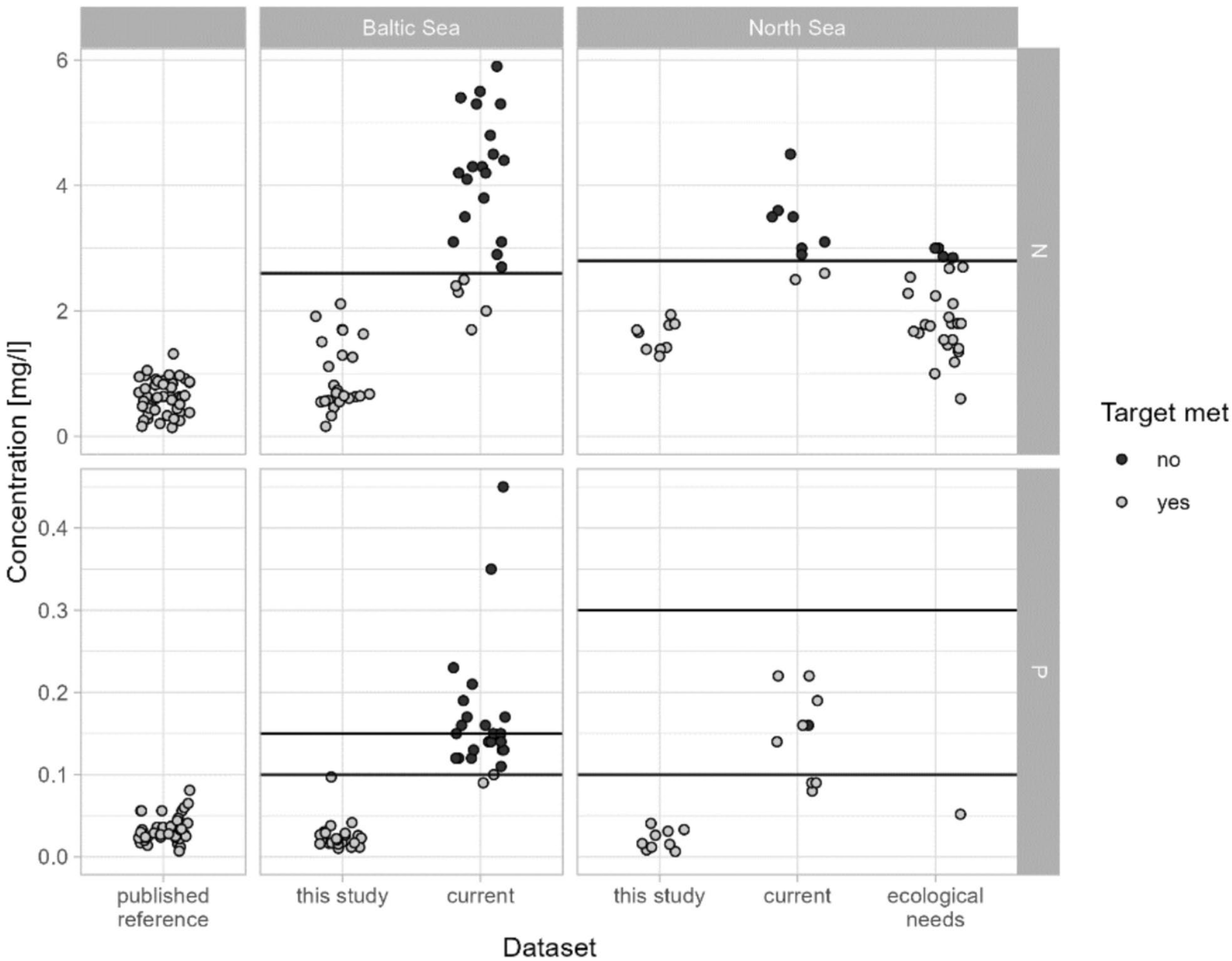

Fig. 5 Modeled riverine nutrient concentrations ("this study") compared to current observed concentrations and literature values. Results for total N (top) and total P (below) for the German monitoring stations at the limnic–marine border (cf. Fig. 1) in the basins of the Baltic Sea (center) and the North Sea (right) compared to published reference conditions in rivers (left) [26, 85]. Current average concentrations for the period 2016–2020 [41, 83], as well as concentrations required for ecological objectives in the North Sea basin as derived from Tables 5 and 6 and monitoring data for 2011–2015 [41, 83]. Color indicates whether target values according to the German Surface Water Protection Ordinance (horizontal lines) are met (gray) or not (black)

than to MONERIS (Table 7). This study thus supports that the inconsistency of the German models with model applications at larger scales was independent of the (changes in) input data, model assumptions, and the importance of nutrient pathways. In fact, E-HYPE implied that the P concentrations under pre-eutrophic conditions were already as high as the recent P concentrations in rivers Elbe, Ems, and Weser—unlike the N concentrations (cf. Fig. 5). The concentrations obtained with IMAGE–GNM were even higher: the P concentrations of 0.3 to 0.4 mg l^{-1}—one order of magnitude higher than the national model results—exceeded even recent observations.

Historical concentrations and ecology-based N targets

The modeled flow-weighted average historical P concentrations at the outlets of all river basins (Fig. 3D) matched the range of published reference values except for some outlets (cf. Fig. 5). Unlike P, the modeled N concentrations exceeded the upper threshold of riverine reference concentrations of about 1 mg N l^{-1} (cf. value range in the left panel of Fig. 5), especially in the North Sea basin. Despite the different input data and model assumptions, our results thus agreed here with the existing model results for 1880 and even more for 1900. The flow-weighted average of 1.4 mg N l^{-1} at the outlets of river basins (average 1.5 mg N l^{-1}) was close to the earlier

Table 7 Comparison of area-specific N and P emissions and flow-weighted concentrations at river outlets based on MoRE results ("this study") and previous model applications. The sea basins were sub-divided into the largest river basins and the remaining river basins

Region	Emission, kg ha^{-1}		Concentration, mg l^{-1}		Model: source	N balance including N fixation
	N	P	N	P		
Baltic Sea	1.84	n/a	1.0	n/a	MONERIS: [25, 26][a], [27][b]	No
	3.53	0.10	1.6	0.04	MONERIS: [27][c]	Yes
	2.48	0.07	1.1	0.02	This study	Not relevant
Baltic Sea, Oder	1.85	n/a	1.1	n/a	MONERIS: [25, 27][b]	No
	3.57	0.10	1.6	0.04	MONERIS: [27][a,c]	Yes
	4.65	0.65	4.5	0.4	IMAGE–GNM: [87]	Yes
	2.37	0.07	1.2	0.02	This study	Not relevant
Baltic Sea, except Oder	1.79	0.08	0.7*	0.03	MONERIS: [26]	No
	3.41	0.08	1.2	0.02	MONERIS: [27][c]	Yes
	2.92	0.095	0.9	0.03	This study	Not relevant
North Sea	6.92	0.17	1.6	0.04	MONERIS: [27]	Yes
	5.88	0.19	1.4	0.03	This study	Not relevant
North Sea, Rhine including IJsselmeer	8.58	0.22	1.4	0.03	MONERIS: [27]	Yes
	n/a	n/a	1.6	0.10	E-HYPE: [45][d]	n/a
	12.2	1.64	2.7	0.3	IMAGE–GNM: [87]	Yes
	8.03	0.31	1.3	0.03	This study	Not relevant
North Sea, except Rhine	5.69	0.14	2.0	0.04	MONERIS: [27]	Yes
	4.20	0.115	1.5	0.03	This study	Not relevant
North Sea, rivers Ems, Elbe, Weser without coastal areas	5.43	0.14	1.9	0.04	MONERIS: [27][c]	Yes
	n/a	n/a	2.6	0.13	E-HYPE: [45][d]	n/a
	5.10	0.65	3.9	0.3	IMAGE–GNM: [87]	Yes
	4.20	0.12	1.5	0.03	This study	Not relevant

[a] Calculated

[b] MONERIS setup without N fixation

[c] Complemented by unpublished database (M. Venohr, pers. comm.)

[d] Unscaled E-HYPE results from unpublished database (A. Blauw, pers. comm.)

* The value of 1.46 mg N l^{-1} shown in Schernewski et al. [19] was erroneously calculated from the emissions not the input

German threshold of 1.5 mg N l^{-1} for the lightly/moderately polluted boundary for rivers [88]. Given the in-stream retention, the N pollution further upstream was likely more pronounced. In consequence, the historical ecological state of Germany's coastal and marine waters likely did not fully represent reference conditions which confirmed similar conclusions for Danish coastal areas in 1900 [44].

The noticeable anthropogenic influence makes the common offset of 50% for the good/moderate boundary less appropriate for ambitious ecological targets. The strong overlap to the modeled historical outlet concentrations in Fig. 5 suggests that a significantly smaller offset would be needed to achieve various environmental objectives. For the German North Sea, the median of the ecology-based targets included in this study of 1.8 mg N l^{-1} (mean 1.9 mg N l^{-1}), for instance, would only be 30% above the modeled flow-weighted mean N concentration. Such a more ecology-based target concentration is more than one-third below the current German riverine N target.

Offsets to reference conditions are usually applied to the historical coastal and marine concentrations not riverine concentrations. However, ocean waters rapidly flush the North Sea, unlike the Baltic Sea. Therefore, the historical situation in offshore regions does not differ much from the current state. Under such circumstances, applying an offset to modeled historical conditions can result in thresholds of the good/moderate boundary which exceed current concentrations and can, thus, potentially worsen the ecological state. This counter-intuitive result could be avoided by applying the offset to the riverine input instead [89]. Based on our model results, the riverine N targets would be 2.1 mg N l^{-1} for rivers to the North Sea, if the common 50% increase is accepted as upper boundary. This concentration would already be substantially below the current N target of 2.8 mg l^{-1} and close to the 3rd quartile of the ecology-based target concentrations in this study (2.2 mg N l^{-1}). Lowering the target concentration for the North Sea requires a critical revision of the existing target concentration for tributaries of the ecologically more susceptible Baltic Sea. However, ecology-based assessments similar to the North Sea are lacking.

In fact, the formal offset of 50% is not based on ecological impacts related to nutrient enrichment [39, 90] but is a kind of expert judgement, another widespread approach to define the good/moderate boundary in EU member states [23]. Using offsets from reference conditions below 50% were already discussed for coastal areas, e.g., in various Danish studies [90–92]. For the Baltic Sea, regional targets for concentrations of dissolved inorganic N and P were estimated as average of the modeled pre-eutrophic state in 1900 and the eutrophic state at the beginning of measurements in 1970–1975 which resulted in variable deviations mostly below 50% [93]. The range of 1.8–2.1 mg l^{-1} for a suggested revised, more ecology-based N target at the limnic–marine border in Germany would also coincide with global literature indicating that riverine N concentrations should stay within 0.5–2.0 mg l^{-1} [94] or 1.0–2.5 mg l^{-1} [95]. The latter threshold of 2.5 mg N l^{-1} was, e.g., used to set terrestrial environmental boundaries for the agricultural N input across the EU [96] and globally [97]. Both studies derived relative reduction requirements for these inputs in the range of the reductions needed to achieve the environmental objectives for the North Sea considered in this study (Germany 51%, EU average 43%; Central Europe 54%).

Model limitations and uncertainties

The uncertainty related to the model assumptions and simplifications in combination with the lack of validation data cannot be quantified. The many methodical and data differences to MONERIS had only a minor impact on the basin scale, i.e., on the modeled concentrations and runoff at the basin outlets. Given the inherent uncertainty, our findings complement rather than replace the existing model results. Nonetheless, the plausible range of concentrations indicates that the national models provide more plausible results than the regional or global assessments. The data availability also limits the model-based historical reconstruction before 1880 to reduce the anthropogenic impact on the rivers and the riverine inputs to the seas.

Our hydrological model accounted for changes in climate, land use, and imperviousness but did not explicitly consider the extent of the Alpine glaciers. Around 1880, glaciers covered much larger areas of the Swiss Alps than today. At the same time, these vast glaciers strongly retreated from their maximum extent around 1850, leading to a considerable contribution of meltwater in river Rhine. The resulting underestimation of this meltwater at the Rhine gauge Basel as well as further downstream (Fig. 4) could likely be reduced by implementing the LARSIM glacier module into LARSIM–ME [98] and the historical glacier extent.

The dominant pathways clearly show that the choice (and reliability) of agricultural data has been most pivotal for the nutrient modeling, be it, e.g., the upper limit of the N concentration in groundwater and the impacts on soil erosion in this study or the estimation of historical N balances for MONERIS. Indeed, the maximum N concentration in groundwater was conservatively derived from the reported class limit (< 10 mg NO_3^- l^{-1}). It is very likely that in many regions the actual concentrations and consequently the emissions via groundwater and interflow were lower. However, the small sample size (n = 278) raises the question how reliably the data captured the pollution of aquifers. The same threshold was used for tile drainages which was significantly lower than the measured N concentration of 12 mg N l^{-1} in root-zone percolates under Danish agriculture representative for the year 1900 [44]. However, the Danish values were similar to the mean concentration under pasture (8.6 mg N l^{-1}) and arable land (15.8 mg N l^{-1}) as modeled with MoRE from recent N balances in Germany [99]. As the share of tile drainage was assumed to be very low, the uncertainty in N concentration had only a small impact on N emissions.

Furthermore, industrial emissions were neglected in line with previous model applications. Given the long history of industrial development since medieval times in European cities, this gap, however, adds to the uncertainty in connection rates to sewers in urban areas. Approaches to identify point sources have been adopted in small scale urban studies [73] but required census data on industrial facilities and their workers which are unavailable at larger scales.

The MoRE/MONERIS concept is in principle applicable beyond the case study to foster the harmonization of historical riverine nutrient inputs to the North Sea and Baltic Sea (cf. [100–102]). Various data sets cover at least the EU member states. Apart from hydrology as core input, such an application also requires plausible assumptions or estimations linked to agriculture as the most relevant source of nutrient emissions in river basins. Specifically, this requires scrutinizing the assumed maximum groundwater N concentration of 2.257 mg N l^{-1} (10 mg NO_3^- l^{-1}) and the use of recent P contents in topsoil under naturally covered areas for agricultural land. According to an estimation of long-term N balances across Europe, large parts of the agricultural land had negative N balances around 1880 [65] thus hampering the use of modeling approaches based on historical N balances (cf. [44]). To refine the estimation of (maximum) historical N groundwater concentrations, other proxies would be needed.

In addition, using mean annual concentrations as management targets neglects seasonality and likely underestimates the relevance of winter concentrations for coastal productivity and phytoplankton blooms in spring (e.g., [103]). Winter nutrient concentrations are used as indicators for the ecological status of transitional, coastal, and marine waters by German and other national legislation as well as by OSPAR and HELCOM. Defining in addition maximum allowable winter N concentrations, e.g., based on a balanced seasonal N:Si ratio, may also help to stimulate a more consistent management across water types (cf. [23]).

Conclusions

In this study, we revised the nutrient emissions around 1880 into German rivers draining towards the North Sea and Baltic Sea. It complemented existing model applications by integrating the historical hydrology, more recent data sets on other historical conditions, modeling approaches being consistent with the national reporting, and harmonized approaches for the two sea basins. Two established hydrological and nutrient models were applied with a set of published data sets and plausible literature-based assumptions to fill data gaps. The good agreement between the modeled discharge and the few river gauge measurements available for the years around 1880 showed the valid hydrological basis of the nutrient model.

The model and data choices had variable impacts on emissions and dominant pathways within both sea basins. Nonetheless, the modeled concentrations at the outlets of the river basins were consistent with the basis of the current thresholds for the good environmental status of coastal and marine waters as well as the riverine N targets in Germany—if compared to existing regional and global assessments. This is especially true for P concentrations. Our modeling approach and the historical scenario are in principle transferable to other countries to foster international harmonization, and most input data are readily available. However, the model assumptions and required model input data have to be critically evaluated, in particular the P concentrations in topsoil and the N concentrations in groundwater—both reflecting the intensity of historical agriculture.

Our lower historical riverine concentrations for the North Sea basin, i.e., lower reference concentrations, could contribute to a more stringent target riverine concentration for nitrogen as well as thresholds for the good ecological status of the coastal and marine waters. However, with harmonized assumptions for both sea basins, this study underpins the perception that the riverine inputs to the German seas around 1880 already exceeded published reference conditions not only in the North Sea basin but also in large parts of the Baltic Sea basin. In consequence, the historical ecological state of coastal and marine waters likely also did not represent reference conditions anymore.

This exceedance makes the common 50% offset for the good/moderate boundary questionable. It is even more pronounced if previous national and regional model results are used. With relative reduction needs derived from data analyses and modeling, we calculated river-specific ecology-based N target concentrations to achieve various safe ecological boundaries in the Wadden Sea and its main tributaries. In addition to these mean annual concentrations, maximum winter concentrations should be included in future studies as they drive coastal productivity.

To meet 25% of these concentrations allows for an offset of 50% to the historical flow-weighted average concentration of riverine N of 1.4 mg l^{-1}, but only 30% for 50% of them. Although ecology-based riverine targets were unavailable for the Baltic Sea basin, we recommend revising the current N target concentration of 2.6 mg N l^{-1} in accordance to the North Sea.

Establishing (new) reference conditions and thresholds for the good/moderate boundary has societal consequences which asks for harmonization and transparency [16]. Given the many assumptions, the differences between the German models MoRE and MONERIS should be considered as uncertainty. Although the model resolution allows in principle deriving spatially variable target concentrations for riverine N, which would be in line with most EU countries, the inherent uncertainty in historical (reference) conditions asks for a robust setting of sea-specific targets rather than river-specific targets. Such robustness and reliability can be fostered using multiple models. For instance, the pre-eutrophic state of the North Sea was derived from a model ensemble which was weighted according to the recent model accuracy [21]. In absence of suitable observation data as historical evidence, such an approach has to assume that model deviations are stationary. Under these circumstances, more qualitative weight-of-evidence methods like “Best Professional Judgment” [104] or the precautionary principle (e.g., [105]) could be more appropriate to weight different model outcomes together with other information. Accordingly, the regional assessment of historical riverine inputs to the North Sea was replaced with the more stringent MONERIS results for German rivers [21]. The uncertainties in nutrient models but also in safe ecological boundaries as well as in future effects of climate change are best addressed by reliable “no regrets” measures and adaptative management in rivers basins [106]. Such measures will also contribute to minimize the impact of nutrient inputs on the terrestrial and freshwater ecosystems [96, 97].

Abbreviations

E-HYPE	European Hydrological Predictions for the Environment
EU	European Union
HELCOM	Helsinki Commission (Baltic Marine Environment Protection Commission)
MSFD	Marine Strategy Framework Directive
LARSIM–ME	Large Area Runoff SImulation Model–Middle Europe
MONERIS	MOdeling Nutrient Emissions in RIver Systems
MoRE	Modeling of Regionalized Emissions
N	Nitrogen
OSPAR	Oslo Paris Convention for the Protection of the Marine Environment of the North–East Atlantic
P	Phosphorus
WFD	Water Framework Directive

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1186/s12302-025-01185-8.

Supplementary Material 1.

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Author contributions
A.G., K.M., I.H., J.K., M.G., S.F., J.v.B., and W.L. contributed to the study conception and design. Material preparation, data collection, and data analyses were performed by K.M. (nutrients), M.G. (soil erosion), I.H., G.B. (hydrology), and A.G. (nutrient targets, model results). The modeling was conducted by K.M. (nutrients) as well as I.H., J.K., and G.B. (hydrology). W.L. and S.F. supervised the study. A.G. wrote the first draft of the manuscript. K.M., I.H., M.G., J.v.B., and W.L. reviewed the manuscript. A.G., K.M., and I.H. revised the manuscript. All authors read and approved the final manuscript.

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Data availability
The table of MoRE data created and analyzed during this study as well as the geometry of the modeling units are available in the ZENODO repository https://doi.org/10.5281/zenodo.15227585. The MoRE software is open source. An empty SQLite version of MoRE is available from KIT (https://www.iwu.kit.edu/wg/english/MoRE.php). The model input and output produced during the study is available on reasonable request. However, restrictions apply to data under license which is not publicly available. The software LARSIM is available from one of the members of the LARSIM developer community on reasonable request (https://larsim.info/). The LARSIM–ME model used in this study is the property of the Federal Institute of Hydrology (BfG) and has to be requested there (https://www.bafg.de/).

Declarations

Ethics approval and consent to participate
Not applicable.

Consent for publication
Not applicable.

Competing interests
The authors declare no competing interests.

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