



River and lake nutrient targets that support ecological status: European scale gap analysis and strategies for the implementation of the Water Framework Directive



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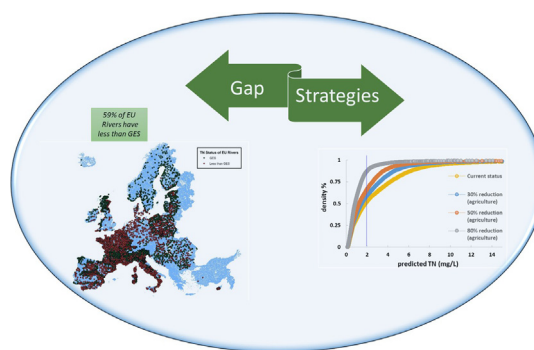
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HIGHLIGHTS

- Developed EU wide and regional nutrient targets for rivers and lakes
- 59% of the TN and 57% of the TP river sites have less than good ecological status.
- 64% of the TN and 61% of the TP lake sites have less than good ecological status.
- Developed regression model to quantify reductions for reaching the nutrient targets
- Significant agriculture load reductions cause small improvement in GES status.

GRAPHICAL ABSTRACT



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ABSTRACT

Eutrophication caused by an excessive presence of nutrients is affecting large portions of European waters with more than 60% of the surface water bodies failing to achieve the primary ambition of water management in Europe, that of good ecological status (GES) with diffuse emission from agriculture being the second most important pressure affecting surface waters. We developed EU wide and regional nutrient targets that define the boundary concentrations between good and moderate status for river and lake total P (TP) and total N (TN) and assessed the gap between actual nutrient concentrations and these targets and considered strategies of nutrient reductions necessary to achieve GES and deliver ecosystem services. The nutrient targets established for rivers ranged from 0.5–3.5 mg/L TN and 11–105 µg/L TP and for lakes 0.5–1.8 mg/L TN and 10–60 µg/L TP. Based on the EU wide targets, 59% of the TN and 57% of the TP river monitoring sites and 64% of the TN and 61% of the TP lake monitoring sites exceed these value and are thus at less than GES. The PCA and step-wise regression for EU basins clearly showed that the basin nutrient export is predominantly related to agricultural inputs. In addition, the step-wise regression models for TN and TP provided the ability to extrapolate the results and quantify the input reductions necessary for reaching the nutrient targets at the EU level. The results suggest that a dual water management strategy would be beneficial and should focus a) on those less polluted rivers and lakes that can easily attain the GES goal and b) on the more highly polluted systems that will improve the delivery of ecosystem services.

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1. Introduction

Eutrophication caused by an excessive presence of nutrients is still affecting adversely large portions of European (Ibisch et al., 2016), USA (Stevenson et al., 2008; Evans-White et al., 2014), Canadian (Chambers et al., 2012), Australian (Davis and Koop, 2006) and Chinese (Huo et al., 2014, 2018) fresh and coastal waters. During the past 50 years, countries have been developing nutrient criteria at various spatial scales (Harmel et al., 2018; Smith et al., 2003) in an effort to assess the attainment of policies in achieving certain level of water uses for freshwaters and coastal areas (Poikane et al., 2020; Evans-White et al., 2014). While numerous approaches have been proposed and employed to derive nutrient criteria (Evans-White et al., 2013; Huo et al., 2018), the two most commonly used are:

- percentile analysis aiming at identifying baseline (i.e. reference) conditions using the 75th percentile of reference sites or the 25th percentile of a general population (e.g., Herlihy and Sifneos, 2008; other approaches have been employed to define baseline levels, see Dodds and Oakes, 2004; Smith et al., 2003);
- pressure - response models linking nutrients with response variables aiming to find a change-point in regression models or to link nutrient concentrations to predetermined ecological outcome (Stevenson et al., 2008).

It should be noted that in several cases the terms *reference* (i.e. *background*, *baseline*) *nutrient levels* and *nutrient targets* are not differentiated and are used largely as synonyms. However, for EU water policy (and of this work) this is not the case and the two concepts are clearly distinguished:

- *reference levels* equal to conditions with no or minor human impact and depict high ecological status (Nöges et al., 2009), these are set using reference sites, paleolimnological analyses, modelling etc;
- *nutrient targets* equal to good status conditions (i.e. slight deviation from reference conditions) derived using pressure-response relationships and intercalibrated good status boundaries (Poikane et al., 2019a, 2019b, 2021; Dolman et al., 2016).

The impact of nutrient loads in European surface water bodies was assessed by the European Environment Agency (EEA, 2018) who found that 60% of the surface water bodies fail to achieve the ecological conditions necessary to meet the Water Framework (WFD) definition of “good ecological status” (GES) with diffuse emission from agriculture being the second most important pressure affecting surface waters. In addition, 25% of groundwater bodies were also classified as having poor chemical status with nitrate the dominant factor in water degradation. To combat these excessive nutrient loads in the environment, the European Commission has adopted a wide range of legislation during the past 30 years.

The Urban Waste Water Treatment Directive (Council Directive 91/271/EEC) aims at controlling pollution from point sources requiring advanced treatment in areas located in sensitive zones. The implementation of this Directive since 1991 has led to a significant decrease of nitrogen (N) and phosphorus (P) emission from wastewater treatment plants. The Nitrates Directive (91/676/EEC) aims at controlling nitrate emission from agriculture both by implementing Action Programmes in Nitrate Vulnerable Zones, setting the maximum nitrate concentration in water at 50 mg/L NO₃-N and limiting the use of manure in agricultural lands. The Directive was very successful in particular in reducing the pressure from fertilization with significant decreases of the N and P surpluses throughout Europe. The Water Framework Directive (WFD; 2000/60/EC) and the Marine Strategy Framework Directive (MSFD; 2008/56/EC) have been applied within a broader framework of catchment management, aiming at achieving good ecological/environmental status. The Water Framework Directive (2000/60/EC) (WFD; EC 2000) mandated, among other things, a comprehensive ecological status assessment of all surface waters based on a variety of biological, hydromorphological, chemical, and physicochemical quality elements (EC 2000, Annex V 1.1

and V 1.2). Many of these quality elements are traditionally used for assessing eutrophication, in particular nutrients, phytoplankton, macrophytes and phytobenthos. The Directive requires Member States (MS) to classify the ecological status of surface water bodies into one of five ecological status classes where high status equals to reference conditions (no or minor human impact) while good status constitutes the aim of water management and equals to slight deviation from reference conditions with no significant undesirable disturbance to the aquatic ecosystem (EC, 2009; Poikane et al., 2014a).

The Conventions for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) and of the Baltic Sea Area (HELCOM) required the contracting parties to reduce actual N and P loads. HELCOM set a reduction of nutrient load by 50% to be achieved in 1995. However, contracting parties failed to achieve the target. HELCOM set up the Baltic Action Plan with stricter targets (Boesch, 2019). The required annual load reductions for N were 3, 28 and 8% for Finland, Poland and Sweden, respectively (Piniewski et al., 2020). The efforts on P were stricter with reductions ranging from 60% in Poland to 10% in Finland (Piniewski et al., 2020). OSPAR has also adopted a reduction target of nutrient loads between 1885 and 1995; however, the reduction target was only achieved for P (Lenhart and Große, 2018). Similarly, Med-Pol, the scientific branch of the Mediterranean Action Plan's (MAP) objective is, among other, to support Mediterranean countries in implementing plans for limiting pollution mainly from land-based sources (Karydis and Kitsiou, 2012), however, with no clear target.

The ambitious European Green Deal is the response of the European Union (EU) at the challenges posed by climate change and environmental degradation. In particular, it aims at enhancing an efficient use of resources, restoring biodiversity, and cutting pollution. The Farm to Fork and Biodiversity Strategies, both pillars of the European Green Deal in the protection of the environment, aim at cutting nutrient losses by 50% thus leading to a cut of at least 50% of the use of fertilizers.

Despite these initiatives nutrients in European water remain an issue of concern, and to facilitate appropriate management it would be advantageous to have a clear set of nutrient target concentrations. The WFD provides a common framework for water management in Europe but it does not provide Europe wide nutrient standards or targets; rather each MS is required to establish national targets for N and P that would support GES. Thus, despite the importance of nutrients in determining the ecological status of European waters, there are no common nutrient standards which can be used to make a Europe wide assessment of the actions needed to determine GES. However, one major achievement of WFD implementation has been the establishment of a common view of ecological status through the intercalibration exercise (Birk et al., 2013; Poikane et al., 2014b; Kelly et al., 2014). In this process, class boundaries of ecological assessment systems designed to assess departure from natural conditions were compared, brought to a common scale and harmonized. This has ensured a robust view of GES can be used as the starting point for the development of nutrient targets both on national and European level (Phillips et al., 2019; Kelly et al., 2021).

According to the requirements of WFD, all MS have established **type-specific nutrient standards** supporting intercalibrated boundaries of GES (see examples Dolman et al., 2016; Free et al., 2016; Kagalou et al., 2021). These nutrient standards are used for the classification and management of eutrophication and thus constitute actual nutrient management targets across Europe (Poikane et al., 2019a). However, there are several difficulties in applying these standards at European scale as they are set for national river and lake types which are very numerous (amounting to a total of 1599 river types and 673 lake types), and defined by different factors not readily available at European scale, such as lake stratification or river flow variability (Lyche-Solheim et al., 2015; Munné and Prat, 2004; Søndergaard et al., 2020). On the other hand, recent EU guidance proposed a modelling approach to determine nutrient standards using pressure response relationships (Poikane et al., 2019b, 2021; Kelly et al., 2021). Using this approach, nutrient concentrations are related to ecological status allowing to estimate the TP and TN concentrations at which water bodies achieve good ecological status. Such a standardized approach would be ideal, but the availability of data and the relatively high level of uncertainty of these models, particularly for rivers (Poikane et al., 2021) limits their use

at the European scale. Further work is thus needed before a robust range of ecologically appropriate nutrient boundary values that would support GES can be proposed. Therefore, to make a Europe wide assessment, it is necessary to establish **Europe-wide nutrient standards** for a standardized classification of rivers and lakes types such as the European broad surface water types (Lyche Solheim et al., 2019).

Thus, for the analysis presented in this work, we harmonized the national type-specific targets within pan-European broad water body types (Lyche Solheim et al., 2019) and then confirmed and validated these available values with results derived from European-wide empirical modelling.

The overall objective of this work was a) to assess the gap between actual nutrient concentrations and nutrient targets (corresponding to good/moderate TN and TP boundaries) for rivers and lakes at continental scale, and b) to assess strategies of nutrient reductions necessary to achieve GES and ecosystem services.

2. Material and methods

2.1. Harmonization of current national Boundary values

National surfacewater-type specific boundary values were collated and grouped by country and then matched to a common typology that has been used to characterise European lakes and rivers, the European Broad Waterbody types (Lyche Solheim et al., 2019). There was significant variation in the values used by different countries, even within this typology (Poikane et al., 2019a) and a method was needed to harmonise them. The simplest approach to using these data would be to base a broad type standard on an appropriate summary statistic of values categorised by broad type, for example the median or other quantile. Initial data exploration showed significant regional variation in these boundaries could be linked to a latitudinal gradient, thus we used a general additive model (GAM) to predict broad type specific good/moderate boundary concentrations after allowing for the effect of country (expressed as the latitude of the country centroid) and water body type. Prior to modelling, extreme type specific values were removed, as these were likely to be associated with specific uncommon national types. The predicted national boundary values for each broad type were then summarized by median and interquartile range.

Type specific pan-European and regional concentration values were determined using the median of the predicted values for each broad type. Regional values were determined by splitting the predicted data into 3 groups by latitude (>60 N Scandinavia, 55–60 N Baltic, and < 55 N Central/Southern). Where the predicted type specific boundary value fell outside the range reported, it was replaced by either the maximum or minimum reported value, the only exceptions being for water body types with fewer than 2 reported records.

The resulting type specific boundary values were then compared with values available from empirical modelling (Phillips et al., 2018; Poikane et al., 2021) with the values obtained from the analysis of the MS boundaries replaced by those predicted from empirical modelling where these were lower. For a few types insufficient data were available to reliably use either approach, particularly for TN values. In these cases, if TP boundary values were available the TN boundary was predicted using the mean TN/TP ratio of available boundary values for the type. If that was not possible, then expert judgment was used by applying either the minimum boundary values reported for the type by a country or the predicted median value from the most similar type.

2.2. Methodology of gap analysis

Three different sets of spatial data were used to conduct the gap analysis of nutrient targets. These were:

- The river network data were obtained from the WISE WFD reference spatial data sets in shapefile form (WISE WFD Reference Spatial Datasets reported under WFD 2016 - PUBLIC VERSION - version 1.4, Apr. 2020 <https://www.eea.europa.eu/data-and-maps/data/wise-wfd-spatial-3>).

- The broad type typology of rivers and lakes obtained from the PANGAEA Data Publisher for Earth & Environmental Science (Globovnik, 2019). A water body broad type is defined as having common natural ecological conditions which are not affected by human intervention, represent fixed abiotic conditions, and explain natural variability. The size of the basin, underlying geology, and elevation are the 3 descriptors that define the 20 Broad Types of Rivers and 8 Broader Type of Lakes.
- The nutrient data obtained from the EEA Waterbase – aggregated Water Quality database. The database contains data on the status and quality of Europe's rivers, lakes, groundwater bodies, and transitional, coastal, and marine waters.

The average TP and TN concentrations for 2000–2017 for rivers and for 2000–2014 for lakes were extracted from the database for each river and lake sampling location, respectively. Most of the entries were from the 2007–2012 period. The 3 databases were linked together in ARCGIS, and every river segment and lake was associated with a broad type of rivers or lakes, average TN and TP concentrations as well as nutrient targets. The nutrient targets for each broad type were also included in the database, and the nutrient gap between the actual average concentration of TN and TP and the nutrient target for each river segment and lake was calculated and used in this analysis to identify the extend of freshwater bodies that exceed the targets.

Finally, we evaluated the factors driving this gap at the basin level. Basin level parameters for each basin of the continent used in the analysis for N and P included: area, average annual flow, average annual precipitation, air temperature and water temperature, average annual irrigation, average lake area, land use area (harvested, crops, fodder, grassland, forest, shrub, urban, bare, water), mineral fertilizer input (on crop, fodder, grassland), net mineral fertilizer input (on crop, fodder, grassland), losses of mineral fertilizer input (on crop, fodder, grassland), manure input (on crop, fodder, grassland), N fixation (on crop, fodder, grassland), atmospheric deposition, N point sources, uptake, transported via erosion, surplus, potential delivery, lithological classes, and total, urban, and rural population. This database for 6028 basins was obtained from the work of Malagó et al. (2019) and Malagó and Bouraoui (2021).

Given that these data were collected using a grid of 10 by 10 km², we selected the basins with an area greater or equal of 1000 km² and linked them to the previous database containing the chemical and ecological data and associated each basin with the sample points nearest to the outlet of the basin. In this way, we can relate the basins with available TN and TP concentrations with morphological parameters and TN and TP loads. Once the combined database was obtained, we regrouped the data (i.e. sum of all mineral fertilizer) and scaled to convert from absolute value to pressure (i.e. normalize amount of fertilizer to the agricultural area). Then we conducted the following analyses: a) correlation matrix to evaluate the coherence of the data, b) Principal Component Analysis (PCA) to identify the factors that impact the chemical status of the basins and c) step-wise linear regression to develop statistically robust models enabling TN and TP concentrations prediction at the outlet of all EU basins (including those with no TN and TP measurements) and thus assess the impact of nutrient loads to EU seas as well as the policy implications for reach the nutrient targets at the EU level. The statistical analysis was conducted using the R statistical package (R Development Core Team, 2011). These statistical models can then be used to develop strategies for nutrient reductions and assess the impact of such reductions to the implementation of the WFD at continental scale.

3. Results

3.1. Nutrient targets to reach GES for EU Rivers and lakes

A summary of the best available estimate of good/moderate boundary concentrations for river TP and TN is presented in Table 1 for Europe. Detailed target values are presented in Tables S1–S4. As there were significant regional variations, these tables present targets for the whole of Europe as well as for three regions (Scandinavia, Baltic, Central/Southern Europe).

Table 1

Summary of best available estimate of good/moderate boundary concentrations for river and lake TP ($\mu\text{g/L}$) and TN (mg/L) for Europe (EU).

River targets			
River broad type	Code	EU TP Target, $\mu\text{g/L}$	EU TN target, mg/L
Very Large	RT-01	89	2.1
Lowland	RT-02 – RT-07	40–105	2.1–3.5
Mid-altitude	RT-08 – RT-13	47–70	0.8–2.4
Highland	RT-14 – RT-16	11–27	0.9–1.3
Mediterranean	RT-17 – RT-20	21–41	0.5–1.1
Lake targets			
Lake broad type	Code	EU TP Target, $\mu\text{g/L}$	EU TN target, mg/L
Very Large	L-01	20	0.7
Lowland	L-02 – L-05	17–60	0.5–1.8
Mid-altitude	L-06	17	0.5
Highland	L-07	10	0.5
Mediterranean	L-08	19	0.6

3.2. Gap analysis between EU nutrient targets and measured values

3.2.1. Rivers gap analysis (all Europe targets)

For annual mean concentration in rivers, there were 33,430 records for TN and 49,910 records for TP, mostly for 2007–2012. These records correspond to 8857 sites for TN from 31 countries and 10,473 sites for TP from 35 countries. The distribution of sites per country for TN and TP varied significantly. Spain, France, Italy, and Poland have the highest number of sites reported in the database. Dense monitoring networks were reported in the database in the central European countries, the Baltic countries, Portugal, Scotland, and some parts of the UK (Figs. 1 and 2). Comparing these TN observed values with the respective Europe wide targets, it was found that 41% of the monitoring sites achieve GES while 59% were less than good (Fig. 3). Similarly, for TP data, 43% of the monitoring sites have GES and 57% less than good (Fig. 3). The average TN concentration of all river types is 2.58 mg/L (range 0.66–4.81 mg/L), and the average target is 1.76 mg/L (range 0.5–3.5) resulting in an average gap between measured concentration and the target of 0.82 mg/L (Table S4) with 5 river types showing a negative gap and the rest positive. The percentage of the monitoring points achieving GES ranged from 8.7% for the Mediterranean RT20 river type to 100% for mid-altitude RT13. Similarly, the average TP concentration of all river type is 0.565 mg/L (range 0.042–1.527 mg/L),

and the average target is 0.05 (range 0.011–0.105 mg/L). The average gap between measured concentration and the target is 0.5 mg/L (range, 0.023 to 1.477 mg/L) with only one river type showing a negative gap and the rest positive (Table S6). The % of the monitoring points with GES ranged from 0% for river type RT13 to 88.2% for the very small mid-altitude RT12.

Grouping the rivers into larger aggregated types allows some broad generalisations to be made. The Mediterranean rivers (RT17–20) had the greatest gap with the highest average TN concentration of 3.8 mg/L followed by the lowland (RT2–7) and very large rivers (RT1) with 3.09 and 2.872 mg/L , respectively. There was also an altitudinal TN concentration gradient with lowest value at the high altitude and highest in the lowland types. Regarding the TP concentrations, the very large rivers (RT1) had the highest TP concentration of 1.445 mg/L followed by Mediterranean rivers and lowland with 0.74 and 0.652 mg/L , respectively. A similar altitudinal TP gradient was also observed for TP as for TN.

Figs. S1 and S2 present the distribution of TN Gap Concentrations for Broad River Types RT1 to RT10 and RT11 to RT20, and Figs. S3 and S4 present the distribution of TP Gap Concentrations for Broad River Types RT1 to RT10 and RT11 to RT20. For TN the gap was negatively correlated with the target concentration ($r = -0.55$, $p = 0.012$), but for TP there was no significant correlation with the target concentrations. However, for both TN and TP there were much higher positive correlations with the mean concentrations (TN $r = 0.81$, $p < 0.001$, TP $r = 0.99$, $p \leq 0.001$) demonstrating that the gap for different river types is primarily driven by variation in the measured concentration, not by differences in the targets, particularly for TP.

To place the gap into context, it is helpful to consider the normalized values (gap as a percentage of the target), where a doubling from the target is represented by 200% (Fig. 3). The distributions of the normalized gap for TN and TP are remarkably similar demonstrating that the relative pressures from these nutrients are similar (Fig. 3). For both nutrients roughly 40% of sites have a negative gap (in good or better status), while for TN 17% and for TP 22% of the monitoring sites have mean concentrations greater than 200% of the target value. The only notable difference being that some sites have an extremely large gap (>1000%) for TP (Fig. 3). These relationships can be used to quantify the proportion of river sites with excessive TN and TP concentrations where significant load reductions need to occur to improve their status.

There is significant variability among the river types regarding the fraction of river sites that are above the 200% TN and TP target (Tables S7, S8). The Mediterranean rivers have between 59 and 45% of the river sites above

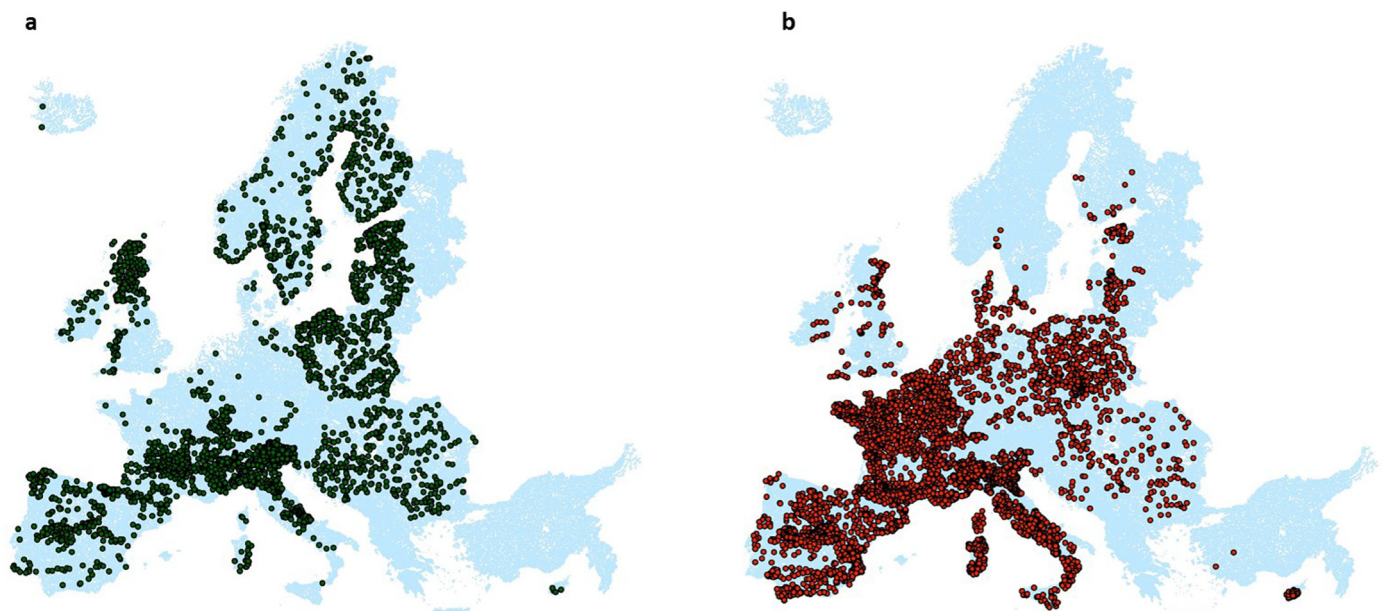


Fig. 1. Spatial distribution of TN Status of European Rivers: a) river segments with GES and b) with less than GES.

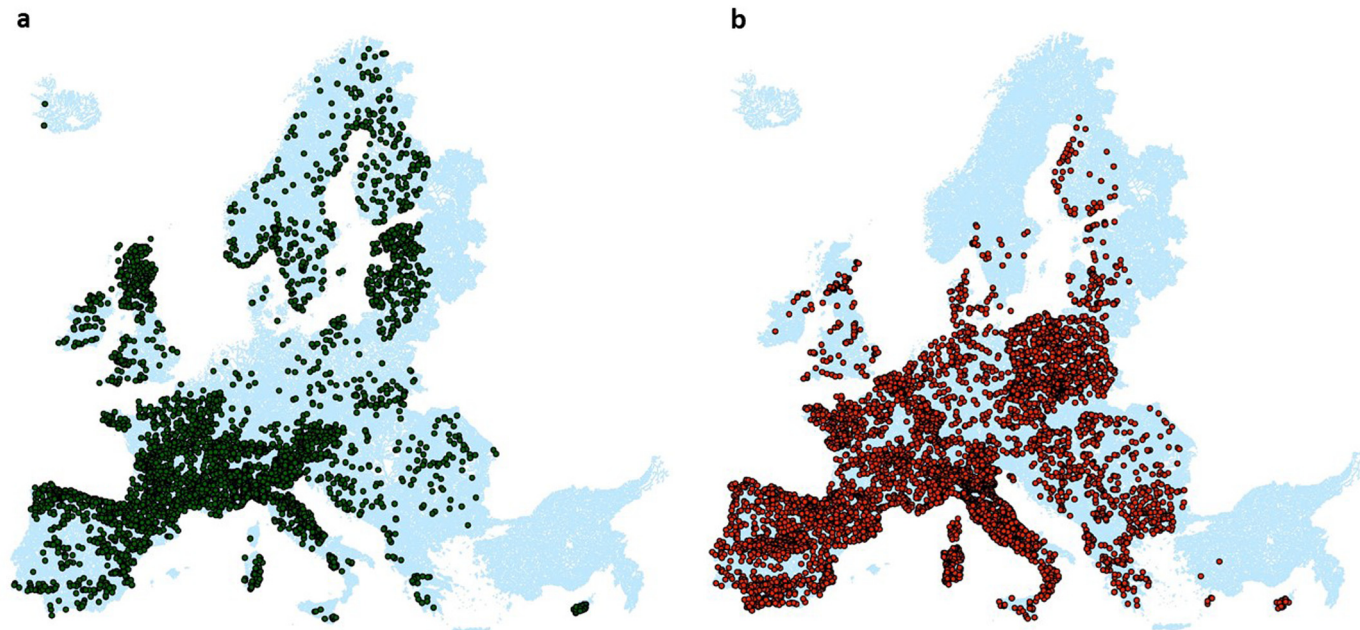


Fig. 2. Spatial distribution of TP Status of European Rivers: a) river segments with GES and b) with less than GES.

the 200% level for TN and 38–50% for TP. The other river types have less than 10% for TN and less than 15% for TP sites exceeding the 200% level. It is important to analyse those sites above the 200% level to identify the sources of TN and TP loads and the special conditions that cause such intense pollution. If the major causes of pollution at these sites is related to point sources, then it might be easy to address the problem and improve the ecological status. If not, then it will be very hard to impose non-point source reduction loads to improve the situation.

3.2.2. River gap analysis (regional targets)

The same gap analysis was conducted using the regional targets for Scandinavian, Baltic, and the Central-South Europe sites. Fig. 5 compares the TN and TP normalized gap distributions using the whole EU targets and the regional targets for the three regions. For the Central-South Europe region, the distribution was presented in two parts due the large range of the gap (Fig. 4c., 4d., 4g., 4h.).

The reductions in the TN and TP targets in the Scandinavian region causes deviation from the gap distributions of the whole EU targets. The differences in the other two regions were less marked. Using the lower regional targets for the Scandinavian region, the % of monitoring sites with good TN status was 70% as compared to 93% using the whole EU targets while for the good TP status was 64% compared to 84% respectively

(Fig. 4a., e.). Similarly, for the Baltic region, the % of monitoring sites with good TN status was 71% as compared to 72% using the whole EU targets while for the good TP status was 69% compared to 78% respectively (Fig. 4b., f.). Finally, for the Central-South Europe region, the % of monitoring sites with good TN status was 44% as compared to 37% using the whole EU targets while for the good TP status was 47% compared to 40% respectively (Fig. 4c., d., g., h.).

3.2.3. Lakes gap analysis

There were 1626 lakes with TN values and 1959 lakes with TP values (9917 records for TN and 11,367 for TP). These lake records were less evenly distributed than for rivers with more data from northern latitudes, but they show similar percentages of lake sites at less than good status as for rivers (Figs. 5–6).

The average TN concentration of all lake types is 1.19 mg/L (range 0.53–2.32), and the average target is 0.72 (range 0.5–1.8 mg/L) resulting in an average gap of 0.46 mg/L (range – 0.17–1.72 mg/L). Only 1 aggregated lake type (LA-04 Lowland & mid-altitude siliceous and humic lakes) have an average negative gap (good or better status) with the other types positive.

Similarly, the average TP concentration of all lake types is 0.076 mg/L (range 0.021–0.231 mg/L), and the average target is

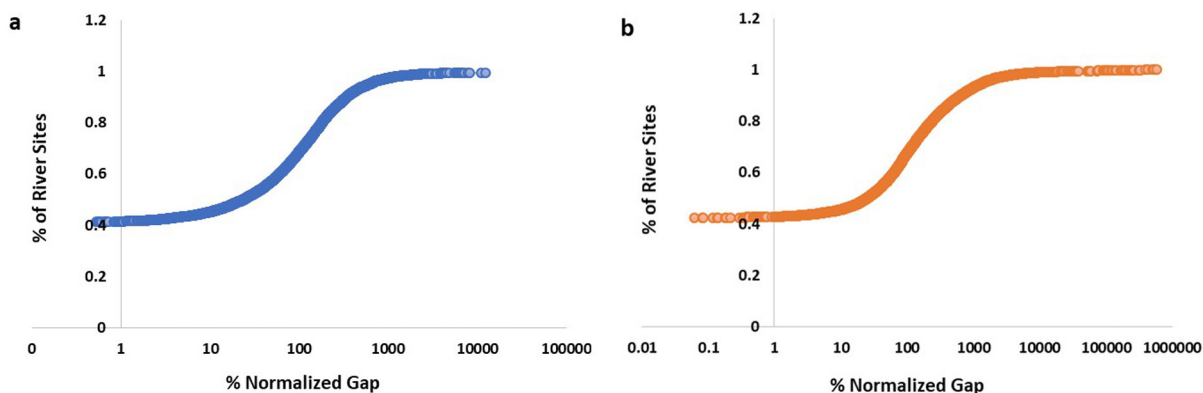


Fig. 3. a) TN and b) TP cumulative distribution of the normalized gap for the Less than GES.

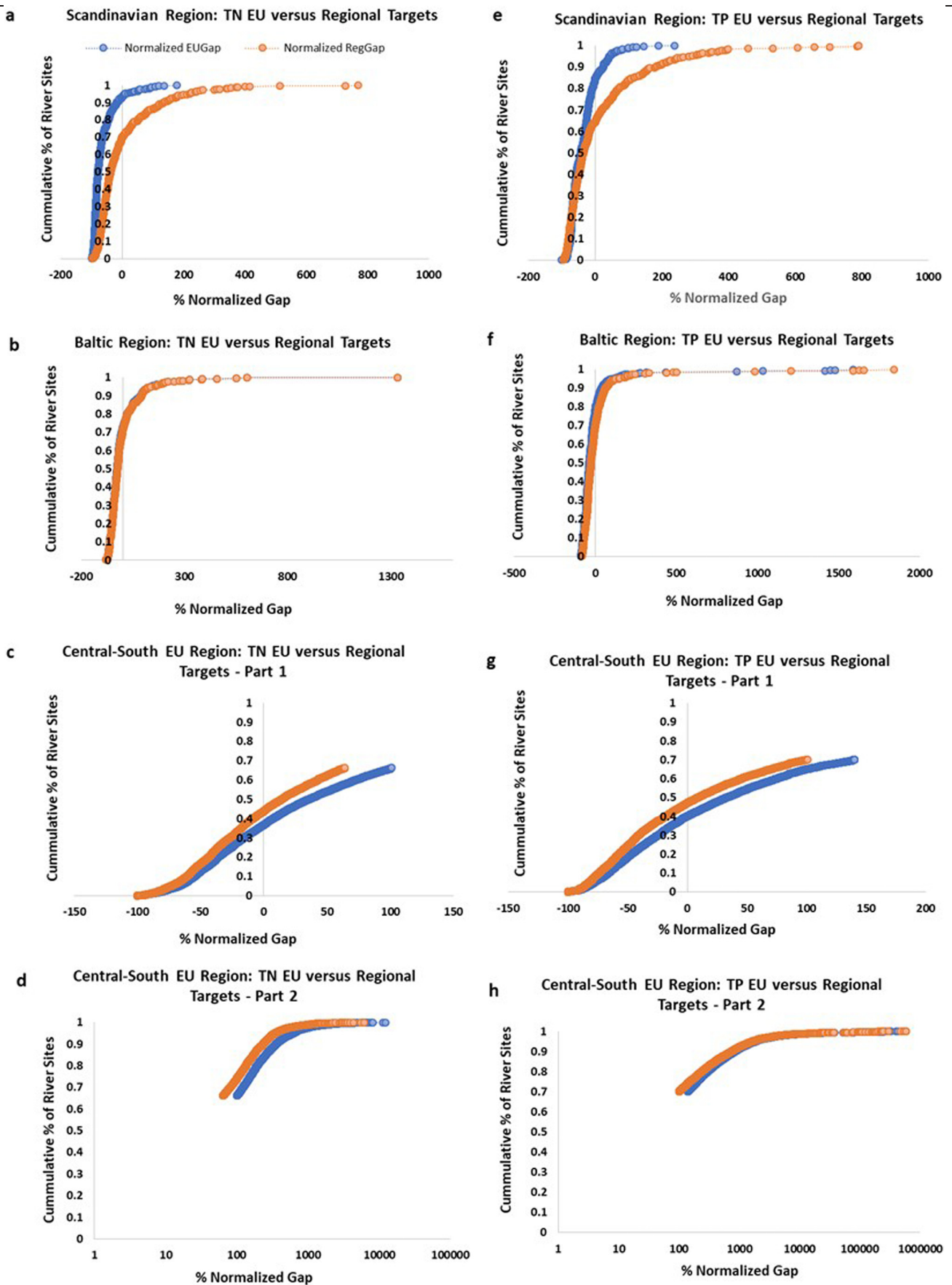


Fig. 4. Comparison of the TN and TP normalized gaps distributions of the Whole EU versus the Regional Targets for the Scandinavian Region (a and e), Baltic Region (b and f) and Central-South Europe Region (c, d, g and h).

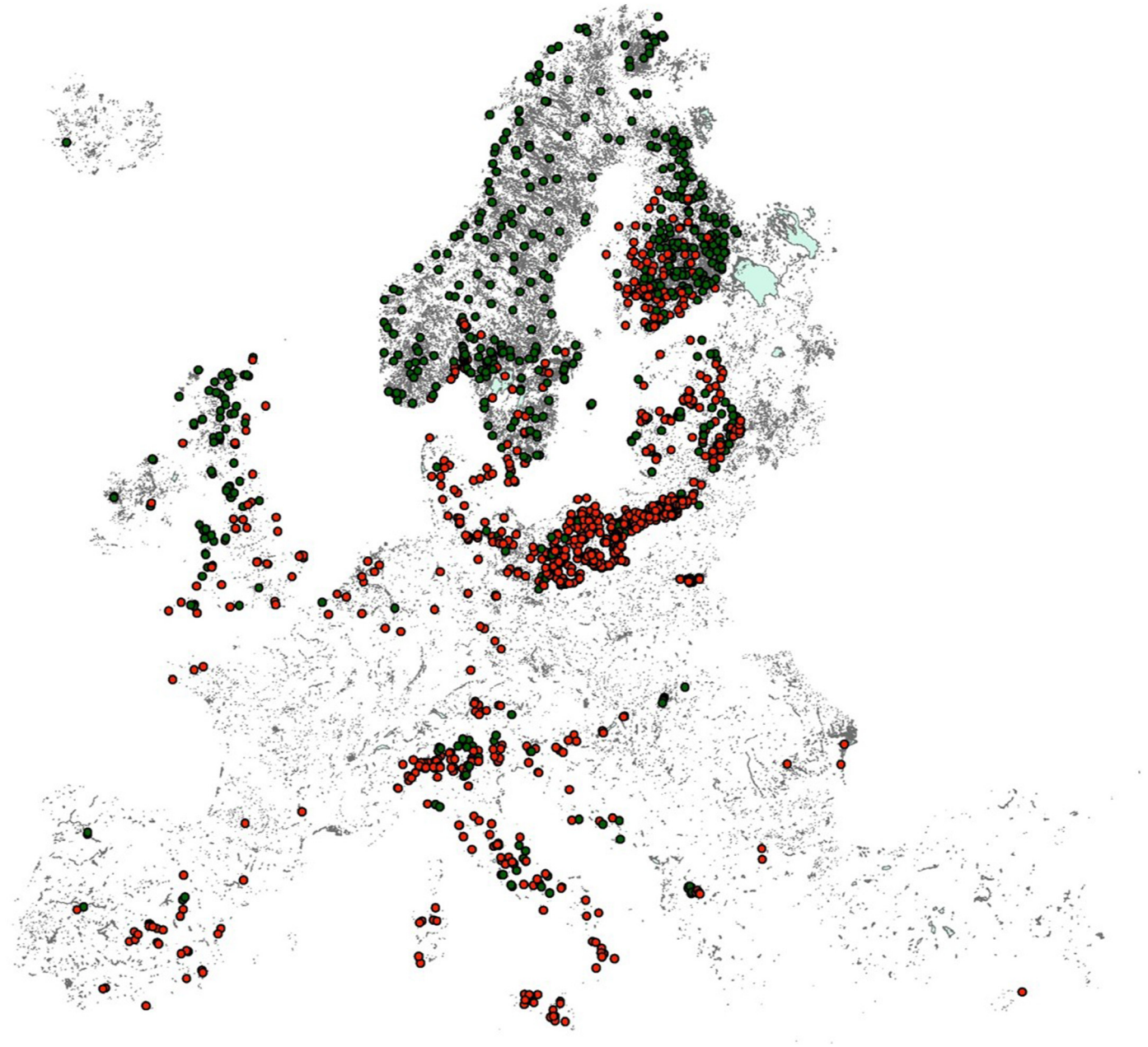


Fig. 5. Spatial distribution of TN Status of European Lakes. Green symbol depicts lakes with GES and red symbol depicts lakes with less than GES.

0.023 (range 0.010–0.060 mg/L for L02) with an average gap of 0.054 mg/L (range -2.2–0.212 mg/L). Again, one lake type (LA-04 Lowland & mid-altitude siliceous and humic lakes) had a negative gap and the rest were positive.

The distributions of the normalized gaps for lake TN and TP were similar to each other and to those for rivers (Fig. 7). Based on TN data, 36% of the monitoring sites achieve GES and 64% less than good. Similarly, based on TP data, 39% of the monitoring sites are at GES and 61% less than good. Additionally, 17% of the monitoring sites have mean TN concentrations greater than 200% of the target value. Similarly, 25% of the monitoring sites have mean TP concentrations greater than 200% of the target value. This is the group of lake sites that require significant TN and TP reductions to improve their status.

The L03 (shallow (stratified) calcareous) and L08 (Mediterranean) lake types have the fewest GES lakes (<25%) and the highest percentage of lakes

(greater than 30%) above the 200% gap for both TN and TP. The L04 and L06 (siliceous lowland and mid-altitude) type have the most GES lakes and like the remaining of the Broader aggregated Types have less than 20% of the lakes exceeding the TN and TP targets, with the exception of L05 lakes (lowland siliceous) for TP.

The shallow (stratified) calcareous lakes (LA-03) and the Mediterranean lakes (LA-08) have the fewest good status lakes ($\leq 0\%$ gap) for both TN and TP (Tables S11 and S12) and the most sites above 200%, while the siliceous lowland and mid-altitude lakes (LA-04, LA-06) have the most GES lakes and the least above 200%.

Fig. 7 presents the TN and TP distribution of the % normalized gap up to 200% of the target level. Both distributions show sites with greater than 200% gaps. These graphs can be used to identify realistic expectations of ecological status improvements due to established Programs of Measures in the MS.

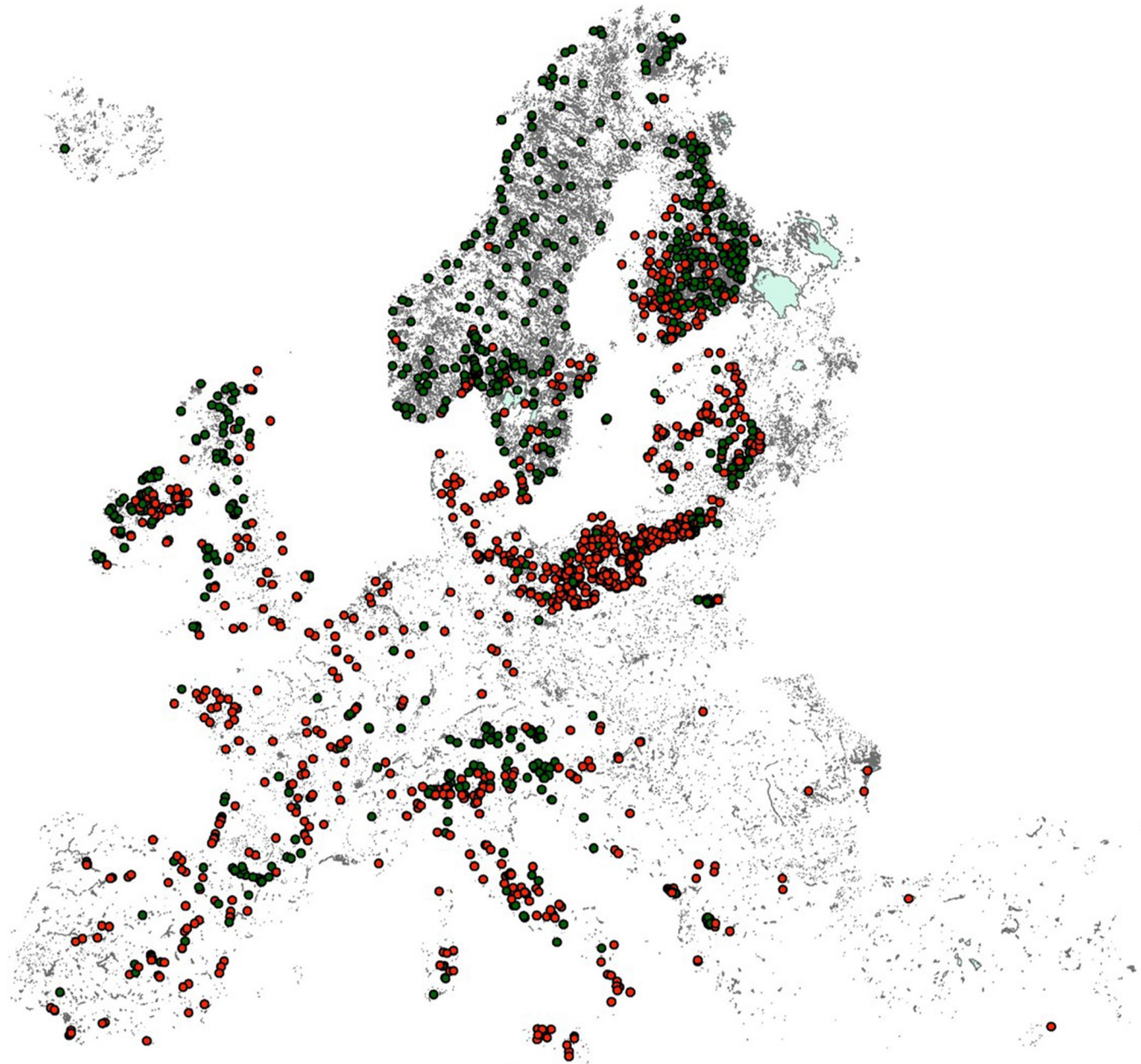


Fig. 6. Spatial distribution of TP Status of European Lakes. Green symbol depicts lakes with GES and red symbol depicts with less than GES).

3.3. Assessment of factors affecting the nutrient gap

The linking of the two databases resulted in 174 basins with TN data and 208 basins with TP data. As mentioned earlier, estimates of the point source N and P loads and non-point source N and P surpluses and the other descriptive parameters for each basin were obtained from Malagó et al. (2019) and Malagó and Bouraoui (2021). N and P surplus is the difference between all the N and P input loads (such as fertilizer, manure etc.) and the output (such as crop uptake) in an agricultural land. The sum of point source loads and nutrient surplus loads represent the potential net input of TN and TP pressures. The regrouping and scaling of the data resulted in a matrix of 25 variables for TN and 23 variables for TP.

The cross correlation between TN and TP and the other variables confirmed they were coherent, meaning that the positive correlation was with those variables that one is expecting to see an increase in the chemical concentration with an increase in the value of the variable. For instance, the

TN concentration was correlated positively with the % crop area and the amount of irrigation (Fig. S9).

The PCA for N showed that the first seven components (eigen value threshold of one was used) explained 84% of the variance of the explanatory variables evaluated. The first two components describe 43% and 42% of the variability. The score plots of the first two components of the PCA are presented in Figs. S10 and S11, and the colour scheme represents the TN and TP concentration (left graph) and the respective status (right graph). Two groups of basins are identified. Group 1 is characterized by higher values of concentrations of TN and TP (Figs. S.10 and S.11, respectively), where the GES of the monitoring sites is less than good (right side of the graphs). Group 2 is characterized by lower TN and TP concentrations (Figs. S.10 and S.11) and GES of the sites (left side of the graphs). The results showed that TN and TP concentrations were significantly positively correlated with the % crop and fodder area, N deposition, N fixation, population, mineral fertilization, surplus, and mean temperature. On the

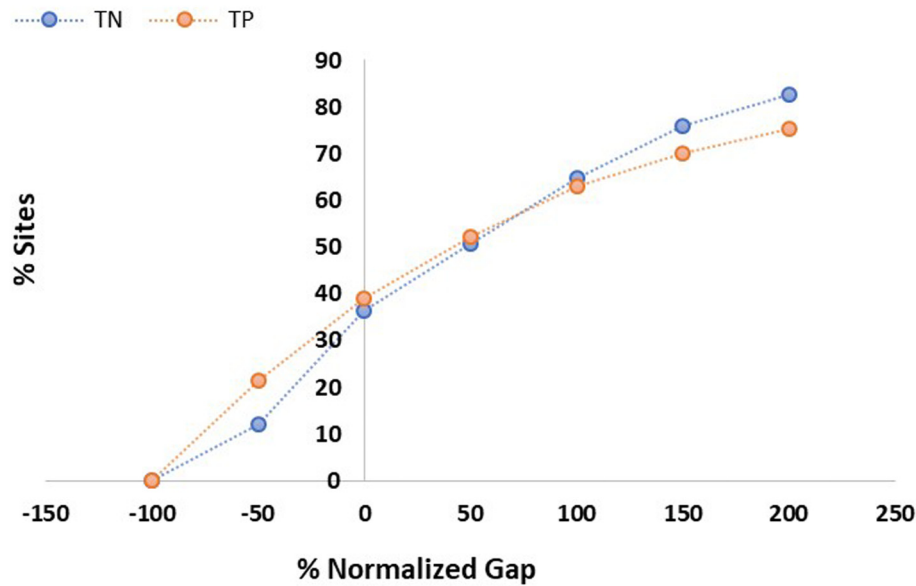


Fig. 7. TN and TP cumulative distribution of the normalized gap.

other hand, a negative correlation is found for both TN and TP with the water extent (%) indicating the role of surface water bodies in reducing nutrient concentrations (retention phenomena). Finally, TN and TP concentrations were also significantly negatively correlated with the % grassland and forest area and mean precipitation (Fig. S9).

The explanatory variables were used also to perform a step-wise regression. After an analysis of the distributions, all data were transformed as follows: add the median, take the natural logarithm, and then divide by the maximum value of the variable. The results for TN are summarized in Table 2. The step-wise regression yielded an R² of 0.736 and an adjusted R² of 0.723. Based on the standardized values, crop extent is the dominant factor controlling TN concentration. It also appears that precipitation has a dilution effect (negative correlation) and that water extent also reduces TN concentration (retention). The Variance Inflation factor (VIF) indicates that there is no significant collinearity (VIF of greater than 10 indicates collinearity) (Table 2). The most significant TN pressures that affect the basin TN concentration are crop area, manure and mineral fertilizer loads, precipitation, irrigation, water area, and population.

Similarly, the results of the step-wise linear regression for TP (R² = 0.63) are summarized in Table 3. Based on the standardized values, the crop extent is also the dominant factor controlling TP concentration. Results also show that flow has a dilution effect (negative correlation) and that the water extent also reduces TP concentration (retention). The VIF indicates that there is no significant collinearity (Table 3). The most

Table 2
Step-wise linear regression model for TN.

TN	Estimate	Std. error	t value	Pr(> t)	Standardized estimate	VIF
(Intercept)	4.596	2.267	2.027	0.044		
Crop area (%) ^a	0.563	0.159	3.533	0.001	0.287	4.03
Manure (kg/ha UAA) ^a	0.365	0.113	3.235	0.001	0.166	1.61
Precipitation (mm/yr) ^a	-1.335	0.298	-4.483	0.000	-0.229	1.60
Water area (%) ^a	-0.184	0.065	-2.811	0.006	-0.144	1.60
Human Population (head/ha) ^a	0.433	0.158	2.734	0.007	0.240	4.70
Irrigation (mm/yr) ^a	0.219	0.088	2.497	0.014	0.140	1.92
Mineral fertilizer loads (kg/ha UAA) ^a	0.561	0.221	2.536	0.012	0.188	3.36
Urban area (%) ^a	-0.205	0.117	-1.745	0.083	-0.136	3.70

^a Data were transformed first by adding the median, then taking the natural logarithm, and then by dividing by the maximum.

significant TP pressures that affect the basin TP concentration are crop area, shrub and forested areas, manure loads, mean flow, and population.

Fig. 8 presents a comparison of the predicted logTN and logTP from the step-wise linear regression versus the measured logTN and logTP data for the selected basins (174 for TN and 208 for TP). It is noteworthy from Fig. 8a that higher concentrations of TN are often associated with the NVZ (Status 1).

The step-wise multiple linear regressions for TN and TP were used to predict the TN and TP concentrations for the 6028 European basins. Fig. 9 presents a cumulative distribution of the predicted TN and TP values using the respective step-wise linear regressions. These predictive statistical models are valuable for assessing the impact of nutrient loads to EU seas by quantifying the input from all basins and the policy implications for reaching the nutrient targets at the EU level.

4. Discussion and policy implications

Twenty years after launching a major initiative to improve the quality of European waters by signing the WFD into law, the status of many European freshwater bodies is still inadequate. According to the WFD, GES should be achieved in all waters by 2027 except in areas where this is not feasible and thus less stringent objectives are defined. However, recent data indicate that only 41% of rivers are in high and GES, 36% moderate, 17% poor and bad status, and 5% unknown (EEA website). Similar results were obtained from our river analysis where 59% of the TN monitoring sites and

Table 3
Step-wise linear regression model for TP.

TP	Estimate	Std. error	t value	Pr(> t)	standardized estimate	VIF
(Intercept)	0.735	1.164	0.631	0.529	0	
Crop area (%) ^a	0.984	0.170	5.779	0.000	0.427	2.284
Human Population (head/ha) ^a	0.761	0.189	4.027	0.000	0.387	3.863
Flow (mm) ^a	-0.543	0.172	-3.153	0.002	-0.186	1.448
Manure (kg/ha UAA) ^a	0.461	0.137	3.365	0.001	0.179	1.178
Urban area (%) ^a	-0.322	0.156	-2.067	0.040	-0.181	3.217
Water area (%) ^a	-0.199	0.091	-2.199	0.029	-0.133	1.519
Forest area (%) ^a	0.640	0.253	2.523	0.013	0.163	1.754
Shrub land area (%) ^a	0.137	0.061	2.236	0.027	0.121	1.215

^a Data were transformed first by adding the median, then taking the natural logarithm, and then dividing by the maximum.

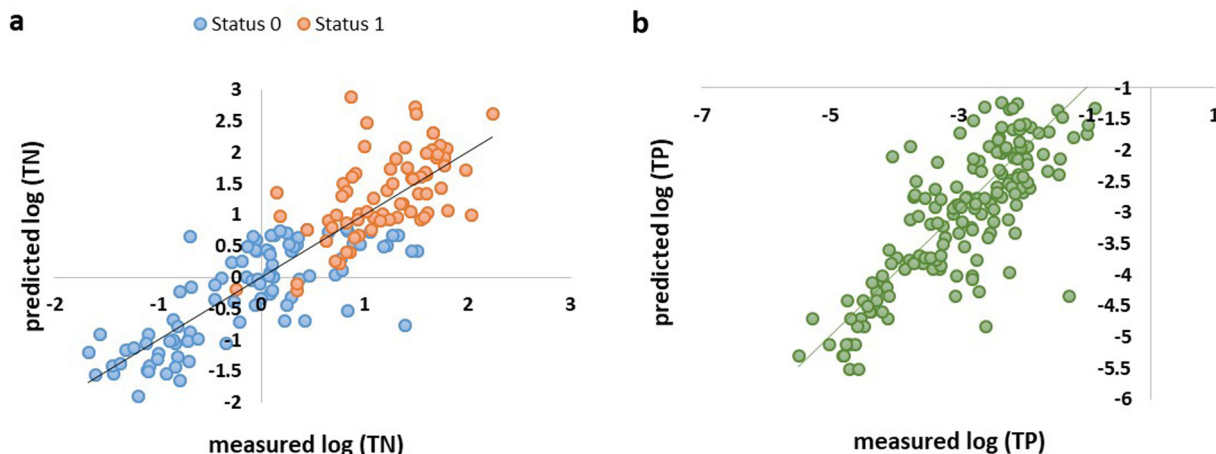


Fig. 8. Comparison of a) predicted logTN and b) logTP from the step-wise linear regression versus the measured logTN and logTP data for the selected basins. Status 1 indicates that the monitoring point is in a Nitrate Vulnerable Zone while Status 0 indicates that the monitoring point is outside the NVZ area.

57% of the TP monitoring sites have concentrations greater than the target supporting good status. Insufficient management measures and lack of cross-sector collaboration have been identified as the most important reasons hindering progress (Carvalho et al., 2019).

The shape of the normalized TN and TP distributions flattens out above 200%, suggesting that this is a critical point in the distributions above which the concentration gap is increasing while the number of sites in that range decreases. This group of river sites with excessive TN and TP concentrations need significant reductions to improve their status. In other words, these are the sites with diminishing returns if the main cause of the pressure is non-point sources. However, we need to consider that implementing nutrient reduction strategies for these sites will show a significant improvement in their ecosystem services even though they may not reach their ecological target.

A 200% gap means that the sites have on average less than 3 times the target concentration. In this group (0–200% the normalized gap) are 41% and 36% of the TP river and/or lake sites. On the average, these sites with TN and TP concentrations less than $3 \times 1.76 \text{ mg/L} = 5.28 \text{ mg/L}$ and $3 \times 57 = 171 \text{ } \mu\text{g/L}$ respectively, are the closest to achieving GES and the ones with the highest possibility of improving their ecological status with targeted actions.

Considering the importance of attaining GES, a legal requirement in Europe, it might seem logical that the MS and EU strategy to focus management efforts first on less polluted rivers and lakes (i.e. those with moderate status) where GES can be reached with comparatively less effort. However, there is another side to the coin. Given that there is a major link between ecological status and ecosystem service provision (Everard, 2012; Tolonen et al., 2014; Vlachopoulou et al., 2014), a bad/poor ecological status, where the biological communities are substantially or severely disturbed, due to human pressures result in reduced or absent provision of

ecosystem services and thus improvements in the ecological status from poor/bad to moderate status will result in significantly higher delivery of ecosystem services. This is especially true for cultural services providing recreational, aesthetic, and educational benefits and regulating services including water purification (Pouso et al., 2018; Grizzetti et al., 2019). In other words, waterbodies that are in a better condition provide more services, and thus improving their status from bad/poor to moderate is likely to result in great benefits to human well-being. Therefore, we suggest a dual strategy for WFD implementation, focusing separately on a) systems which can attain GES and b) highly polluted systems which may improve their ecosystem services delivery.

The PCA analysis and the step-wise regression analysis for 174 for TN and 208 for TP clearly showed that basin export is predominantly related to agricultural inputs. The step-wise regression models for TN and TP provided the ability to extrapolate the results beyond the 174 and 208 basins respectively to the 6028 European basins. With these statistical models, we can quantify the input reductions necessary for reaching the nutrient targets at the EU scale. Fig. 10 presents the improvement in the predicted TN distribution after a 30, 50 and 80% reduction in the pressure from agriculture (Fig. 10a) and after an 80% reduction in the point source pressure and 30 and 50% in agriculture (Fig. 10b). We can observe that the median TN value of the present-day distribution improved from 52% of the sites at present day to 60% for 30% reduction in agriculture which is increasing to 67% for 50% reduction and 87% for 80% reduction in agricultural pressures (Fig. 10a). This suggests that significant agriculture load reductions need to be implemented to have relatively small improvement in the ecological status. In addition, combination of point source load reductions together with agricultural load reductions can also achieve similar changes in the ecological status (Fig. 10b). However, we need to keep in mind that reductions in loads of high nutrient concentrations systems will result

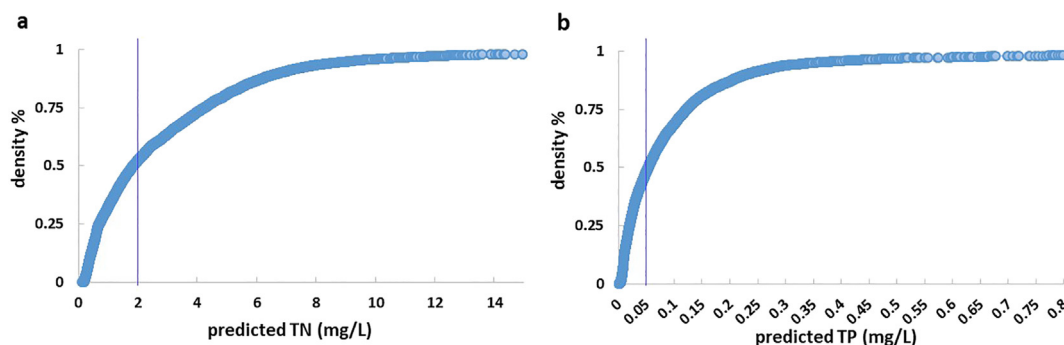


Fig. 9. Distribution of predicted a) TN and b) TP values using the respective step-wise linear regressions for the 6028 European basins. The blue lines identify the median target TN (2 mg/L) and TP (0.05 mg/L) values.

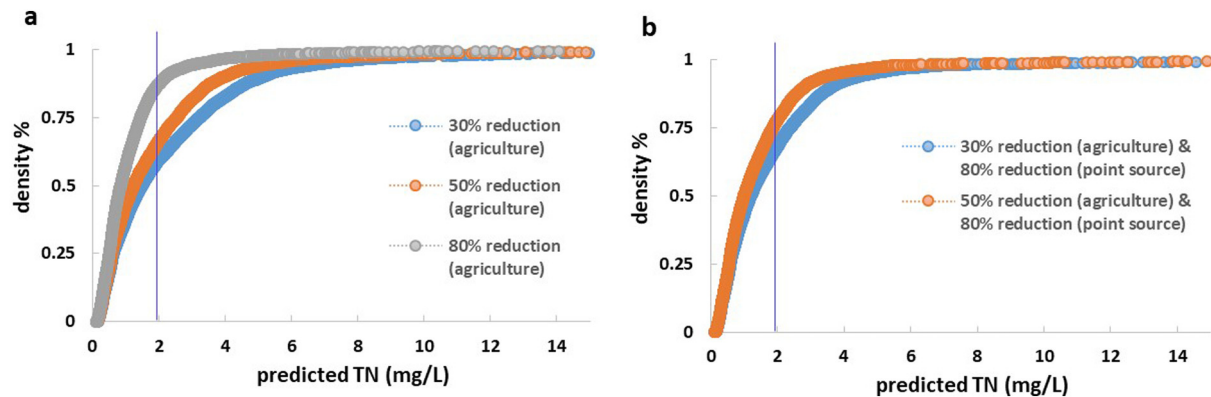


Fig. 10. Distribution of predicted TN using the step-wise linear regressions for the 6028 European basins for a) 30, 50 and 80% reduction in the pressure from agriculture and b) for 30 and 50% reduction in agriculture and 80% reduction in the point source pressure. The blue lines identify the median TN value of the present-day distribution.

in significant improvement in the ecosystem services provided by them. These statistical models can be used by MS to assess their strategy of WFD implementation and identify priority sites that will likely improve their ecological status and the delivery of ecosystem services and provide an example of a methodology that can be applied elsewhere.

Overall, this analysis has shown that agriculture is the main culprit for the current excessive N and P input to European rivers and lakes. Results also suggest that we should not only target and prioritize freshwater sites based on the feasibility of implementing nutrient reduction actions that will improve the water quality and ecological status, but also examine sites where measures will readily improve the ecosystem services even though they may still fail to achieve their GES target.

CRedit authorship contribution statement

Nikolaos P. Nikolaidis: Conceptualization, Methodology, Formal analysis, Writing – original draft, Supervision. **Geoff Phillips:** Methodology, Formal analysis, Writing – review & editing. **Sandra Poikane:** Formal analysis, Writing – review & editing. **Gábor Várbiro:** Methodology, Formal analysis, Writing – review & editing. **Fayçal Bouraoui:** Formal analysis, Visualization, Writing – review & editing. **Anna Malagó:** Formal analysis, Visualization, Writing – review & editing. **Maria A. Lilli:** Formal analysis, Visualization, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.scitotenv.2021.151898>.

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