European Environment Agency
European Topic Centre on Water

Freshwater Eutrophication Assessment

Background Report for EEA European Environment State and Outlook Report 2010

ETC Water Technical Report 2/2010

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1. Introduction

This report provides an assessment of eutrophication in freshwaters throughout Europe. It is intended as extended background material to the nutrient and eutrophication issues described in a more condensed form in the Water Quality assessment within the EEA's 2010 State of Europe's Environment report.

This report describes:

● detrimental impacts that eutrophication causes (ecosystem health, economic costs etc.);
● factors driving eutrophication including the various sources of nutrients and the pressure on the water environment that they exert;
● current state and recent trends in freshwater nutrient quality as reflected by nutrient concentrations;
● impacts of excessive nutrient levels upon freshwater biota and ecosystems – including the current biological state;
● an analysis of the response or measures that can be implemented to tackle eutrophication, including case studies that illustrate the impact of measure(s) upon the state and impact.

Currently, concentrations of nutrients in many European freshwaters are excessively high – and have posed a major environmental concern for some decades. Excessive nutrient levels promote eutrophication in surface waterbodies with a proliferation in the growth of nuisance phytoplankton (algal blooms) and macrophytes and the associated loss of 'desirable' plant and animal species. The generation of huge amounts of organic matter, oxygen depletion, presence of algal toxins, reduced water clarity and fish kills can also result (Figure 1.1). Excessive nutrient levels, leading to water with a murky 'green soup' appearance, can also impact upon freshwater recreation (cover photo). The presence of suspended algae in freshwater represents an economic cost to the water industry that is passed onto citizens; expensive treatment is required – using energy and chemicals - where abstraction occurs to supply drinking water.

Climate change has the potential to exacerbate these impacts, particularly with respect to the predicted increases in frequency and intensity of (winter) rainfall, leading to enhanced flushing of nutrients from land to waterways. This includes both the agricultural and urban environments; e.g. more frequent and higher magnitude 'combined sewer overflows' will occur without appropriate action (CSOs can lead to a rapid and severe/fatal depletion of oxygen levels in receiving waters). Other climate change impacts include changes (increase) in soil mineralization rates, changes to denitrification in soils and water and, increases in water temperature that impact e.g. oxygen levels. Climate change will also impact (reduce) water quantity, particularly in summer months which, in turn, leads to diminished water quality.

As a result of the extent, severity and impacts of eutrophication across Europe, all of the current freshwater related legislation addresses at least one aspect of nutrient pollution. Most notably, the Water Framework Directive (WFD) represents the single most important piece of legislation relating to the quality of Europe's freshwater environment, requiring the attainment of good ecological and chemical status by 2015. Paramount to the achievement of good ecological status, however, will be a marked reduction in the excessive nutrient levels currently observed in freshwaters across much of Europe. River basin management plans under the WFD, due to be operational by 2012, will incorporate a suite of cost-effective 'measures' to tackle all sources of nutrient pollution.

Full compliance with other water-related legislation, in particular, the Urban Waste Water Treatment Directive (UWWTD) and Nitrates Directive (ND) will improve nutrient water quality and aid, although not necessarily guarantee, the achievement of good ecological status under the WFD. The UWWTD requires the collection and treatment of waste water in all agglomerations above a certain

size. As a rule the Directive provides for biological (secondary) waste water treatment and, therefore, a substantial removal of both the biodegradable and nutrient pollution in wastewater. Additionally, however, in catchments with waters sensitive to eutrophication, the legislation demands more stringent (tertiary) treatment to remove most of the nutrient load from wastewater.

The ND is a key piece of legislation with respect to the protection of waters against pollution caused by nitrates specifically from agricultural sources. Fundamental to the Directive is the designation of Nitrate Vulnerable Zones (NVZs), established where nitrate levels in ground or surface water are excessive and that require action plans detailing a suite of measures to be implemented to reduce nitrate pollution.
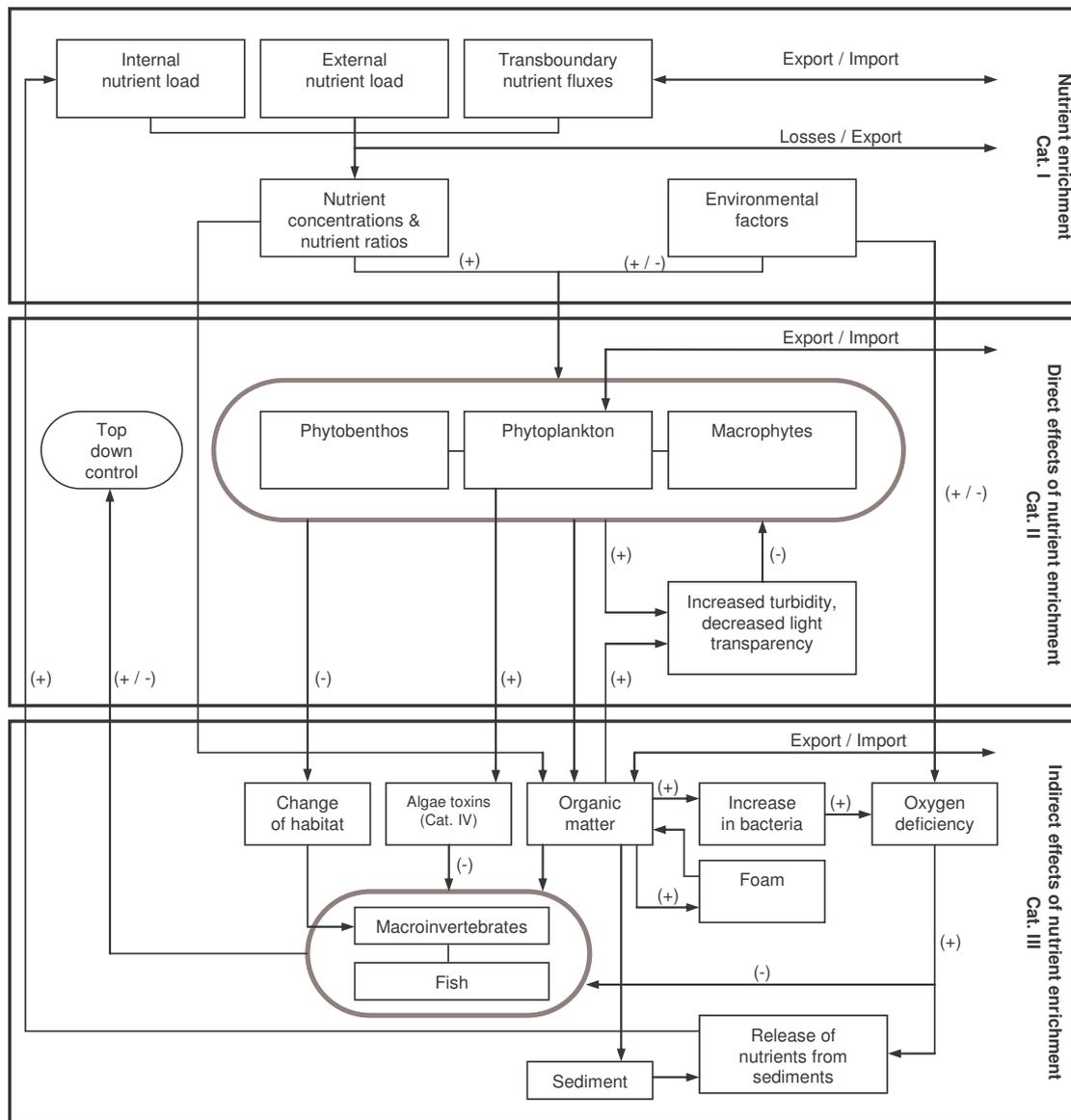
Figure 1.1 - **General conceptual framework to assess eutrophication in all categories of surface waters**. '+' indicates enhancement, '-' indicates reduction. Round boxes indicate biological quality elements of WFD (European Commission, 2009).

2. Sources and pressures of nutrients

The major sources of nutrients causing eutrophication of European waters are from agricultural activities and urban waste water. Additional sources of nutrients are atmospheric inputs, scattered dwellings with septic tanks unconnected to municipal wastewater treatment plants, industries (e.g. food processing facilites), forestry and aquaculture.

2.1. Agricultural Nutrients

2.1.1. Background

The European agricultural sector has changed dramatically over the last 50 years; mechanisation, increased use of fertilisers, farm specialisation, land drainage and advancements in animal feed, have all led to agricultural intensification in most European countries. This process has been exacerbated over recent decades by subsidies under the Common Agricultural Policy (CAP) that, until relatively recently, were strongly linked to production.

2.1.2. Nutrient Inputs to Agricultural Land

Whilst nitrogen fixation, atmospheric deposition and the application of treated sewage sludge can all be of importance, typically, the major nutrient inputs to agricultural land are from inorganic mineral fertilisers and manure from livestock. Today, the highest total nutrient application rates (mineral and manure combined) occur in Western Europe, with N inputs across the region generally exceeding 80 kg/ha/yr but reaching in excess of 170 kg/ha/yr across much of Denmark, Belgium and the Netherlands, Brittany, the Po valley, Western England, Eastern Ireland and North-Western Germany (Map 2.1a).
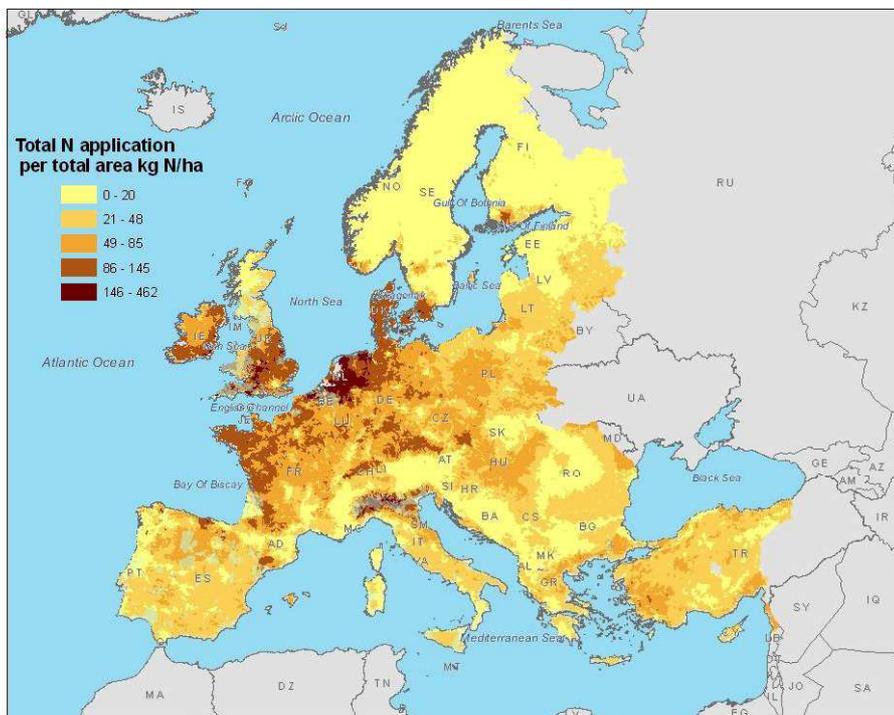
*Map 2.1a - **Overlay of potential NVZs (light green) and total application of N** (kg/ha) per sub-basin. (Bouraoui et al,. 2009).*

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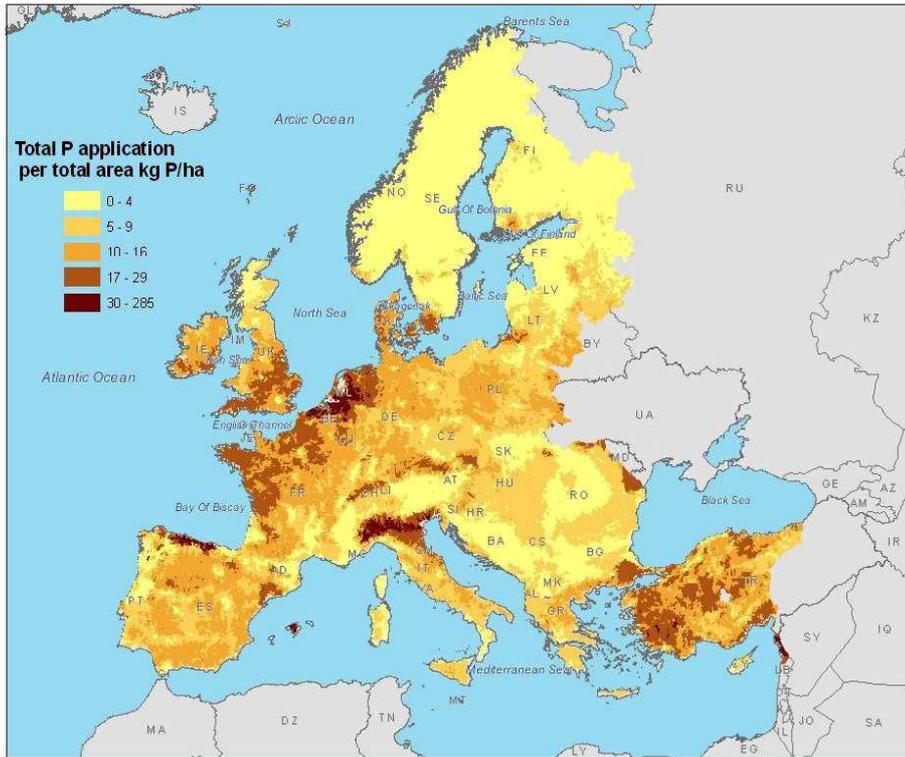
Map 2.1b - Total P fertilizer application (kg/ha) *per sub-basin (Bouraoui et al., 2009).*

A similar pattern is apparent for P, with inputs across Western Europe generally exceeding 15 kg/ha/yr with localised 'hotspots' apparent where applications are currently more than 40 kg/ha/yr. (Map 2.1b).

2.1.3. *Nutrient Surplus*

These current inputs of nutrients to agricultural land across Europe are generally in excess of that required by crops and grassland, resulting in the generation of a nutrient 'surplus'. The magnitude of the surplus reflects the potential for detrimental impacts on the environment since it is 'available' for gaseous loss to the atmosphere (as ammonia), transfer to the nearest receiving waterbody or, to be built up within soil pools over time. Not surprisingly, the pattern of N surplus across Europe reflects the magnitude of inputs via mineral and manure fertilisers; whilst much of the agricultural land across Europe (EU15) has a N surplus of at least 30 kg/ha, values in excess of 120 kg/ha are apparent, particularly in Belgium, the Netherlands, Denmark, north-west Germany and the Po valley (Map 2.2).
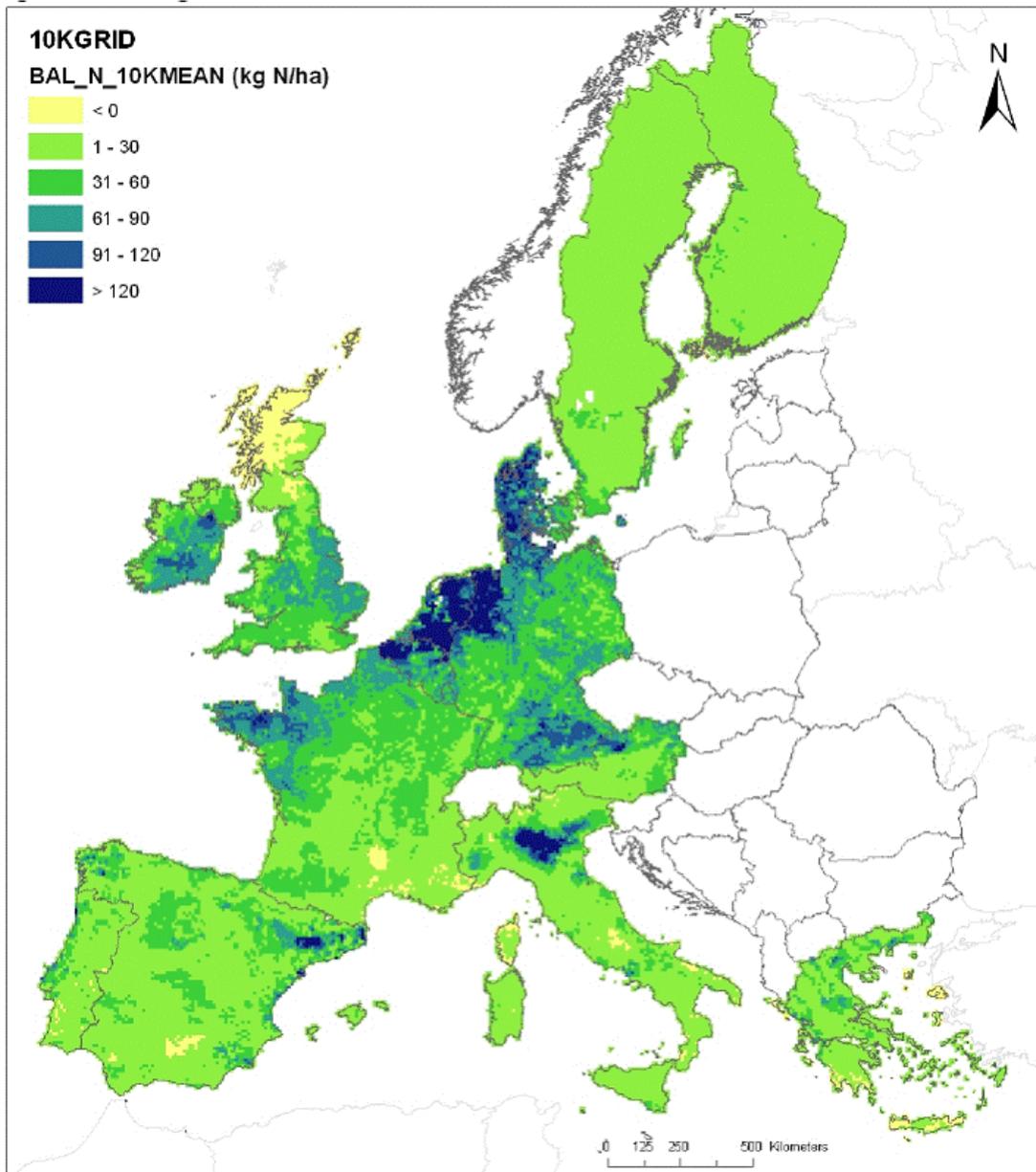
*Map 2.2 - **European map of nitrogen balance per total surface in EU15**, average on 10 km2 area (Grizzetti et al., 2007).*

2.1.4. Pathways and Retention

The degree to which a nutrient surplus is transferred from agricultural land to a waterbody is dependent upon a range of factors including rainfall, soil type, and the physical characteristics of a catchment. Where high intensity rainfall occurs, particularly on poorly drained soils (e.g. clays), water is less able to infiltrate, leading to the generation of surface runoff that can readily wash both soluble nutrients and those attached to soil (particulate) over the land surface.

In contrast, low intensity rainfall, falling on well drained soils (e.g. sands) promotes the infiltration of water and nutrients into the soil. P typically builds up within the soil horizons, although where application rates have been excessive over a number of years, levels can reach saturation, triggering a subsequent leaching of P. Typically, a substantial proportion of the N surplus will be leached – as nitrate - downwards to groundwater and subsequently to surface waters. Where artificial drains underlie pas-

toral land, this downward movement is interrupted and nutrients are instead rapidly transported to surface waters via ditches.

Depending on hydro-geological characteristics, the transport of nitrate from soil to groundwater and subsequently to surface waters can take several decades, for example in chalk. Where such groundwater 'residence' times are lengthy, a significant lag will exist between the implementation of changes in agricultural practices and the change in water quality that they may impart.

In addition to the build up of P in soil, a number of other processes exist within catchments to 'retain' nutrients and prevent their transport to waterbodies, including physical entrapment and gaseous loss via denitrification. In this respect, small ponds, marshes and carbon rich wetlands play an important role. Unfortunately, however, drainage of much of Europe's agricultural land has resulted in a considerable loss of such features, reducing the rural landscape's capacity to store and attenuate all agricultural pollutants.

2.1.5. *Nutrient Emissions to Water*

Despite the often large potential for attenuation within catchments, typically, a significant proportion of any nutrient surplus does reach a receiving waterbody. The magnitude of such agricultural 'emissions', together with those from other sources, determine the resulting nutrient quality observed in Europe's freshwater.

Across Europe, the magnitude of diffuse N and P emissions from agriculture to freshwater broadly reflects the pattern of surplus, with emissions exceeding 30 kg/ha/yr and 0.5 kg/ha/yr for N and P respectively in South-West Norway, Ireland, Western England, Wales, Belgium, Netherlands, Brittany in France, Southern Germany, Galicia in Spain and the Po valley (Map 2.3). However, the relationship between surplus and emissions is not always a strong one. In eastern England, for example, the N surplus is relatively high, but it is the west of England and Wales that exhibit the highest N emissions. This difference is explained by the spatial pattern of rainfall in the UK; the higher rainfall of the west is able to transport a much higher proportion of the total N and P surplus to freshwater.

2.3a
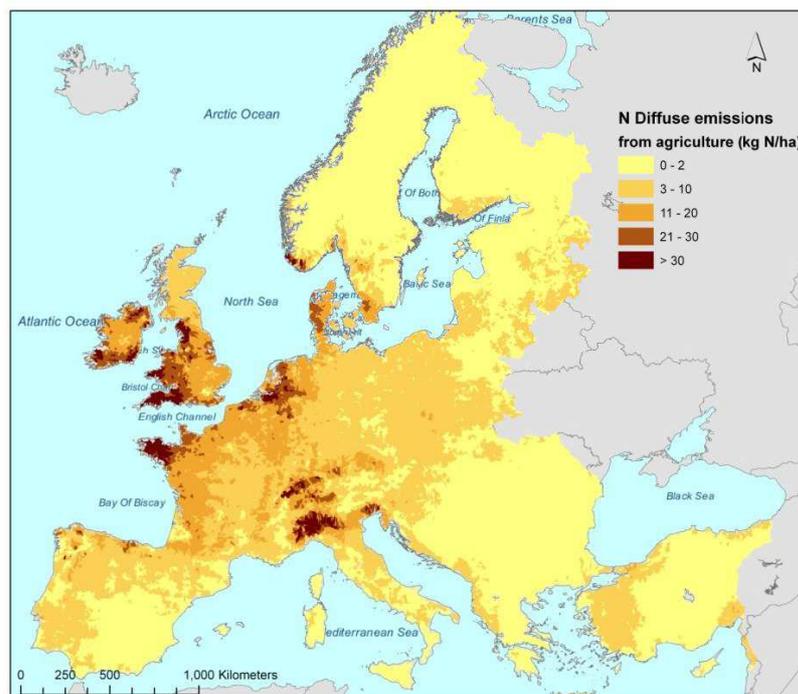
2.3b
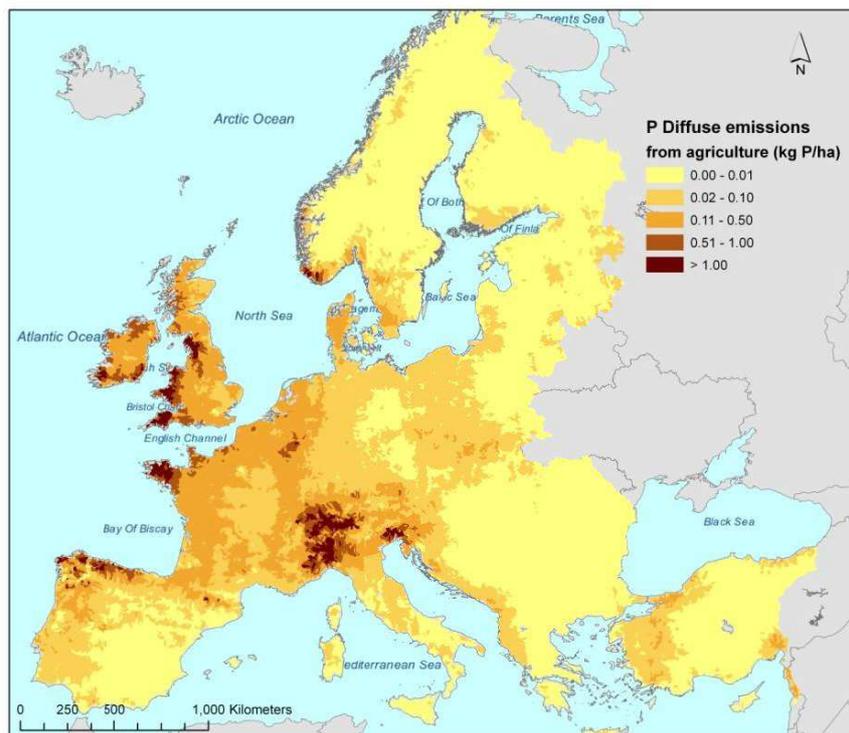
Map 2.3 - Total nitrogen (a) and total phosphorus (b) diffuse emissions from agricultural sources per sub-basin for the year 2000 (Bouraoui et al., 2009).

2.2. Urban waste water and emissions from scattered dwellings

Nutrients originate from numerous sources within the urban environment including from human excretion, the use of household chemicals, industry and, atmospheric deposition, with much of the N associated with the latter resulting originally from vehicle emissions. With respect to P, both domestic and industrial detergents can account for a substantial proportion of emissions in the urban environment.

The pathways and ultimate fate of these urban nutrients varies, with the amount finally discharged to freshwater being dependent upon a number of factors; in particular, whether each source is connected to a sewer and to what extent treatment occurs following collection.

2.2.1. *Municipal sewage treatment*

Driven by implementation of the UWWTD, an increasing proportion of Europe's population is now connected to a municipal treatment works via a sewer network (Figure 2.1). Connection rates in Northern Europe now exceed 80% whilst in Central Europe the figure has risen beyond 95%. Elsewhere in Europe, however, connection rates are lower, although in the case of the newer MS this is explained by the later compliance dates agreed in the Accession treaties. The UWWTD has also driven improvements in the level of treatment, with the designations of 'sensitive areas' requiring tertiary treatment that can remove much of the nutrient load received at the treatment works.

Despite the current increase in nutrient removal across Europe as a whole, urban wastewater discharges are estimated to contribute around 50% of P emissions to freshwater, while for N, the contribution is lower, but still substantial at around 30%.

A recent JRC report (Bouraoui et al., 2009) shows that the average N input to surface water from point sources and scattered dwellings ranged from < 1 kg N/ha in Scandinavia and the Baltic countries to almost 200 kg N/ha along the coast of the former Yugoslavian countries in the year 2000. High inputs (20-50 kg N/ha) also occurred in England, Belgium and the Netherlands. The same pattern is seen for P inputs ranging from < 0.1 kg P/ha to ca. 35 kg P/ha, with values up to ca. 10 kg P/ha in England, Belgium and the Netherlands.

Since the year 2000 these inputs from point sources and scattered dwellings are reduced due to increasing proportion of the population connected to urban wastewater treatment plants and to improved waste water treatment with more tertiary treatment than in the year 2000 (Figure 2.1).
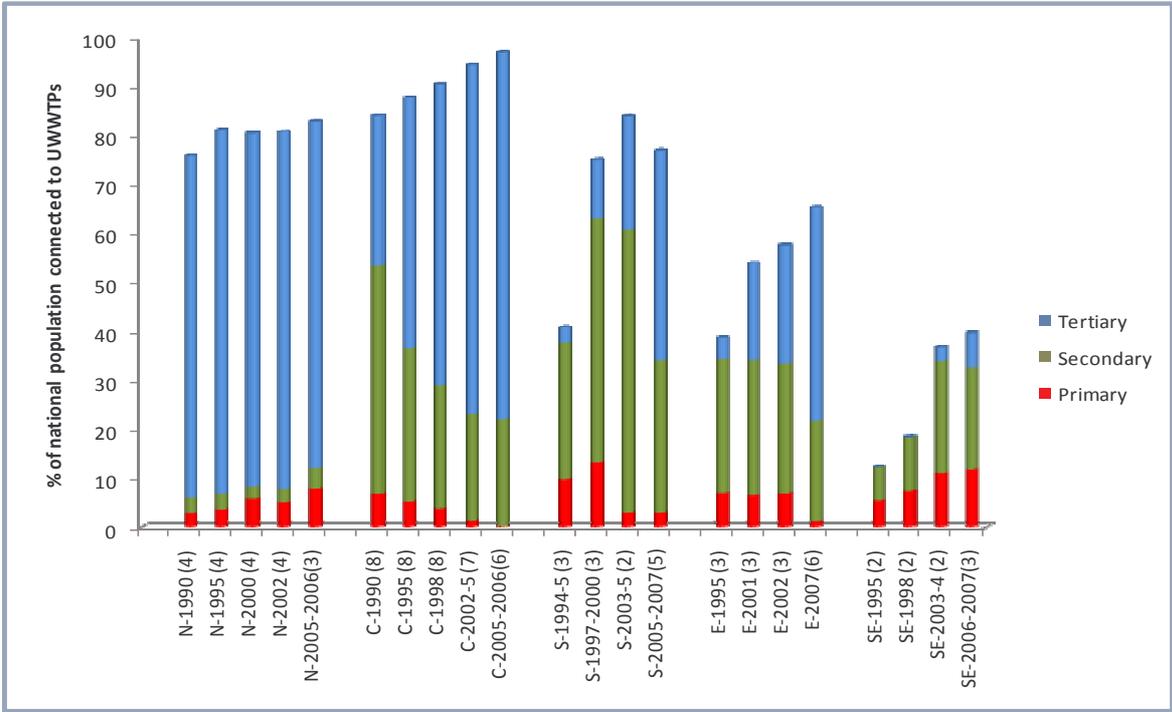
Figure 2.1 - Changes in wastewater treatment in regions of Europe between 1990 and 2007
(CSI024)
Note: *Only The numbers of countries are given in parentheses. Regional percentages have been weighted by country population.*
N-North:*Norway, Sweden, Finland and Iceland*
C-Central: *Austria, Denmark, England & Wales, Scotland, the Netherlands, Germany, Switzerland, Luxembourg and Ireland. (For Denmark no data has been reported to the joint questionnaire since 1998. However, according to the European Commission, Denmark has achieved 100 % compliance with secondary treatment and 88 % compliance with more stringent treatment requirements (with respect to load generated) under the UWWTD (EC, 2009). This is not accounted for in the figure.)*
S-Southern: *Cyprus, Greece, France, Malta, Spain and Portugal (Greece only up to 1997 and then since 2006)*
E-East: *Czech Republic, Estonia, Hungary , Latvia, Lithuania, Poland, Slovenia, Slovakia*
South Eastern: *Bulgaria , Romania and Turkey*
The percentage values have been weighted with country population when calculating the group values.
Data source: *EEA-ETC/WTR based on data reported to OECD/EUROSTAT Joint Questionnaire 2008 (July 2010 update)*

2.2.2. *Nutrients and organic loads from large municipal sources and industry*

The best available information about large point discharges of pollution is from the European Pollutant Release and Transfer Register (E-PRTR) based on Regulation (EC) No 166/20061. This register provides details of pollutant releases from industrial installations in 65 different sectors of activity and municipal waste water treatment plants > 100 000 p.e. Emissions include release to water only (not transfer of waste water before treatment) – treated or untreated above threshold values defined for every pollutant. The threshold value for total nitrogen is 50 000 kg per year, 5 000 kg per year for total phosphorus and 50 000 kg per year for TOC (total organic carbon as total C or COD/3). Emissions below these thresholds are not reported to E-PRTR. The emissions data presented here are from the year 2007 and they are measured, calculated or estimated by operators of each facility. All reported emissions were summarized per national River Basin District and calculated as annual specific emissions (g/km2/year). Discharges from urban waste water plants and from industrial facilities with emissions above the given threshold values are visualised in the maps in different ways according to their size (Map 2.4 a, b, c).

E - PRTR: Volume of Nitrogen released to water in 2007

Total emission (kg/km2/year) per RBD
- RBDs without emission data
- < 25
- 25 - 100
- 100 >

Emission (t/year) for UWWTP
- < 500
- 500 - 5000

Emission (t/year) for facility
- < 500
- 500 - 5000
- 5000 >

0   250   500   750   1 000
Kilometers
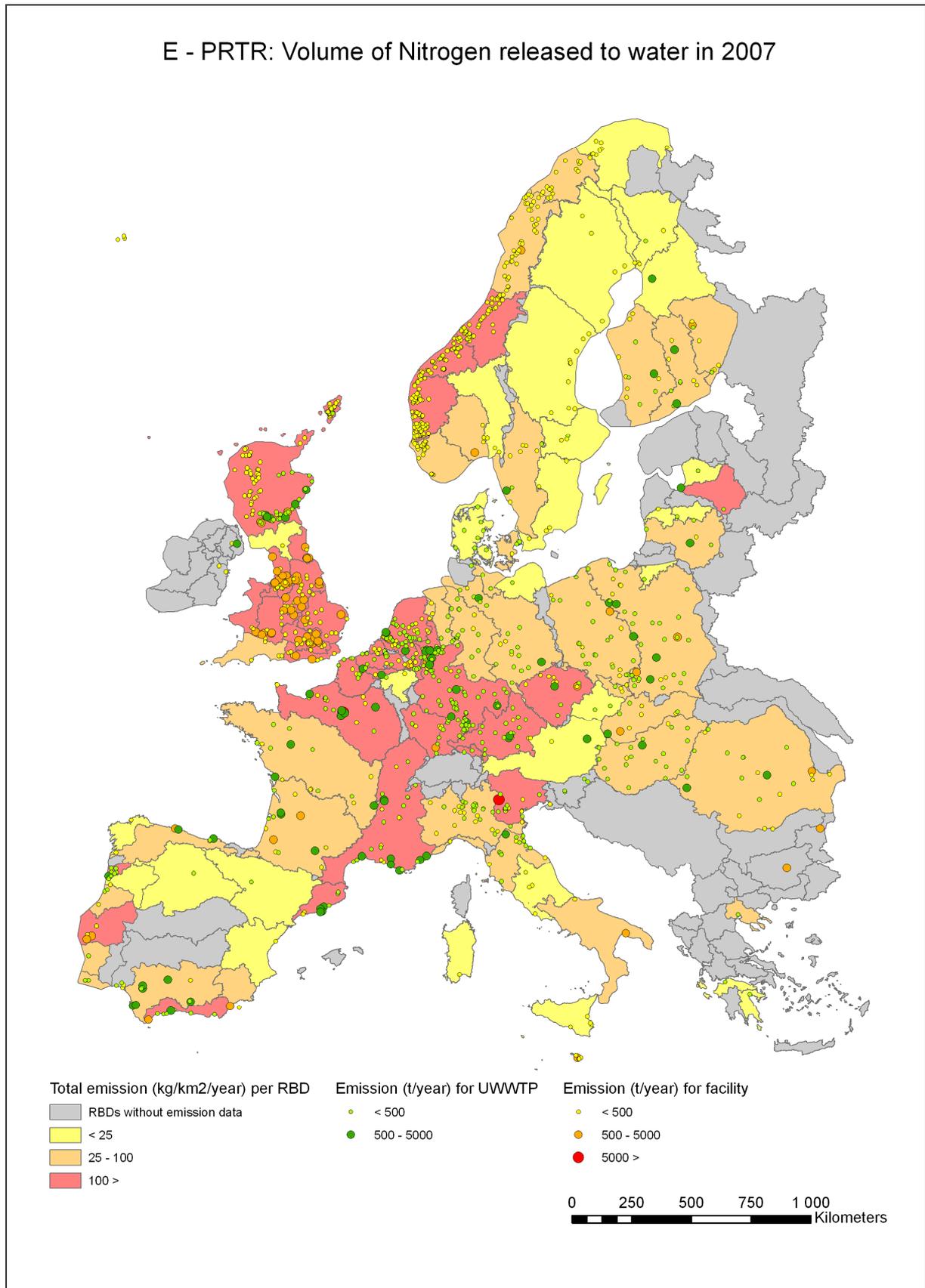
Map 2.4a - Total nitrogen emissions from E-PRTR facilities to water

E - PRTR: Volume of Phosphorus released to water in 2007

Total emission (kg/km2/year) per RBD
- RBDs without emission data
- < 1
- 1 - 10
- 10 >

Emission (t/year) for UWWTP
- < 50
- 50 - 500

Emission (t/year) for facility
- < 50
- 50 - 500
- 500 >

0   250   500   750   1 000
Kilometers
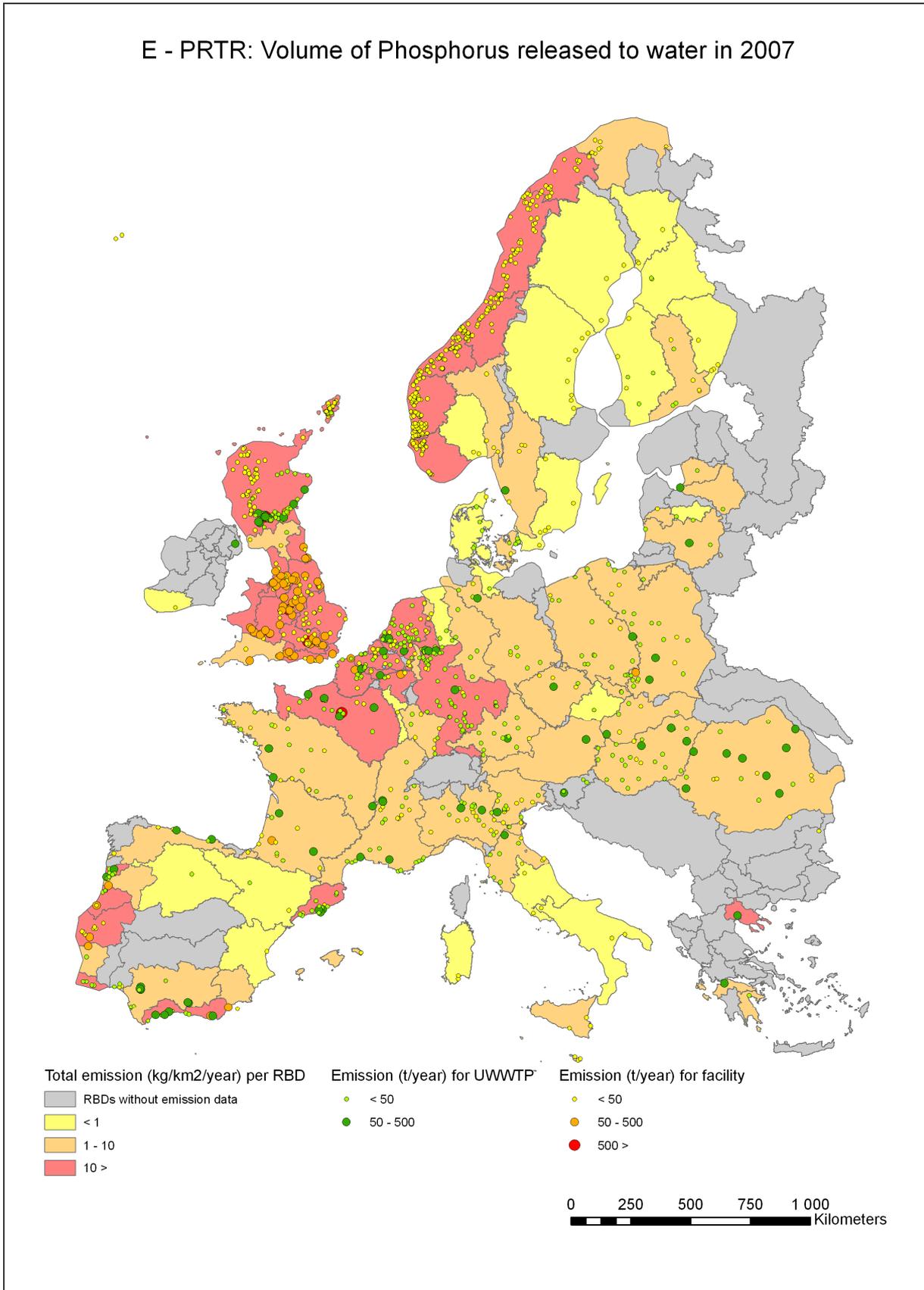
Map 2.4b - Total phosphorus emissions from E-PRTR facilities to water

E - PRTR: Volume of TOC released to water in 2007

Emission of TOC (kg/km2/year) per RBD
- RBDs without emission data
- < 50
- 50 - 200
- 200 >

Emission of TOC (t/year) for UWWTP
- < 1000
- 1000 - 10 000

Emission of TOC (t/year) for facility
- < 1000
- 1000 - 10 000
- 10 000 >

0   260   520   780   1 040
Kilometers
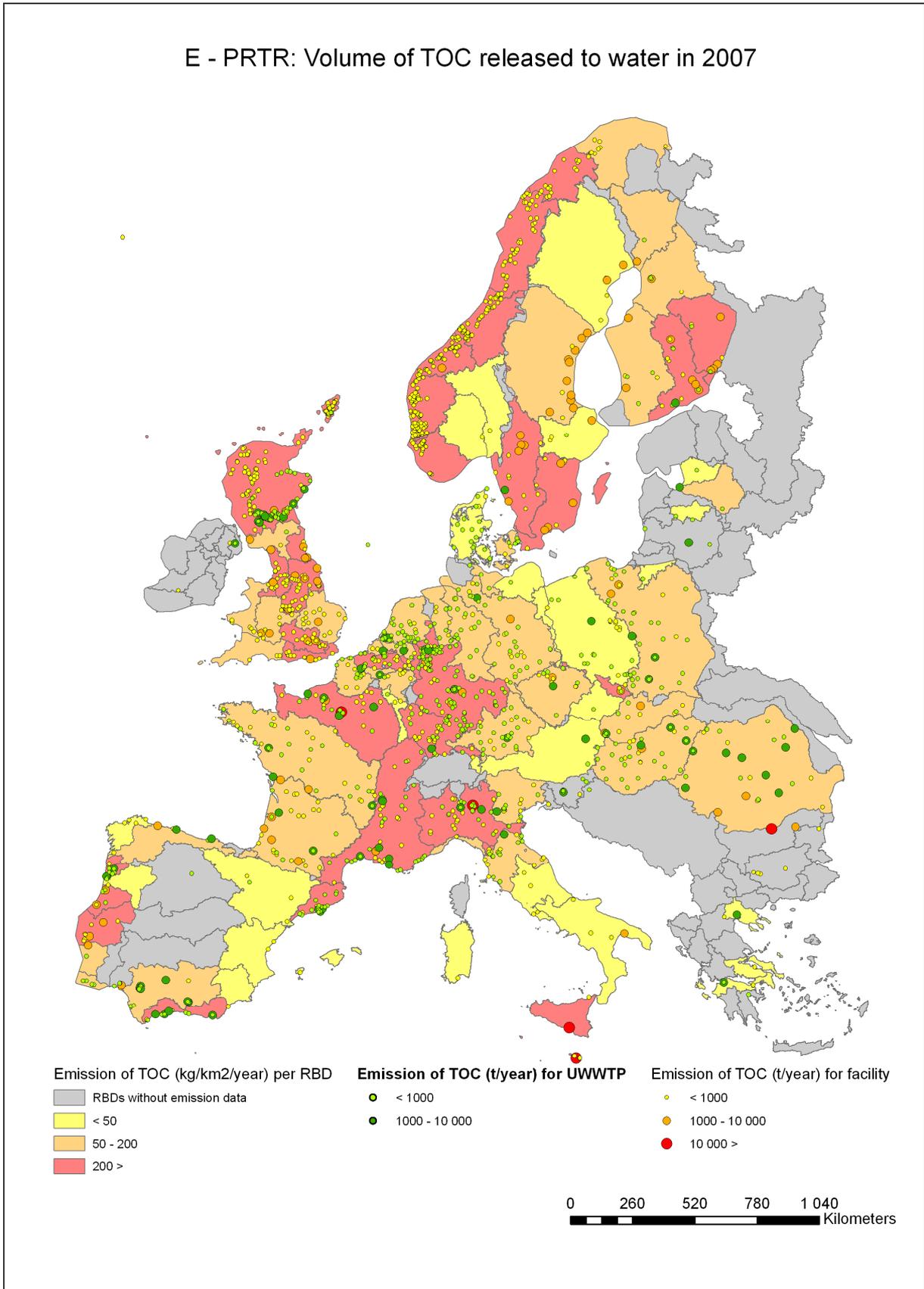
Map 2.4 c - Total organic carbon emissions from E-PRTR facilities to water

Total discharges per country and number of reported facilities are given in Figure 2.2.
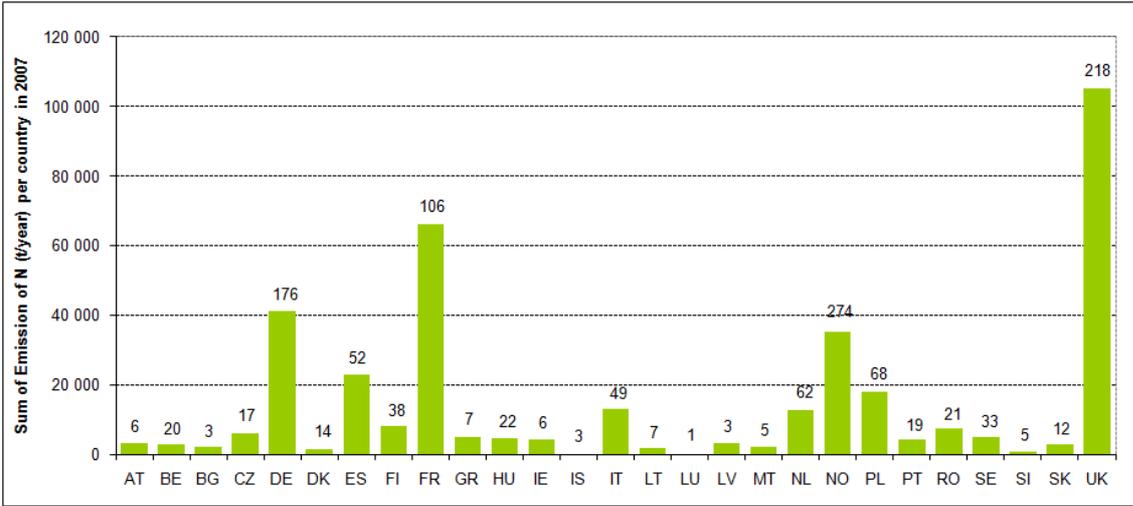
*Figure 2.2a - **Total nitrogen discharges and number of facilities** per country*
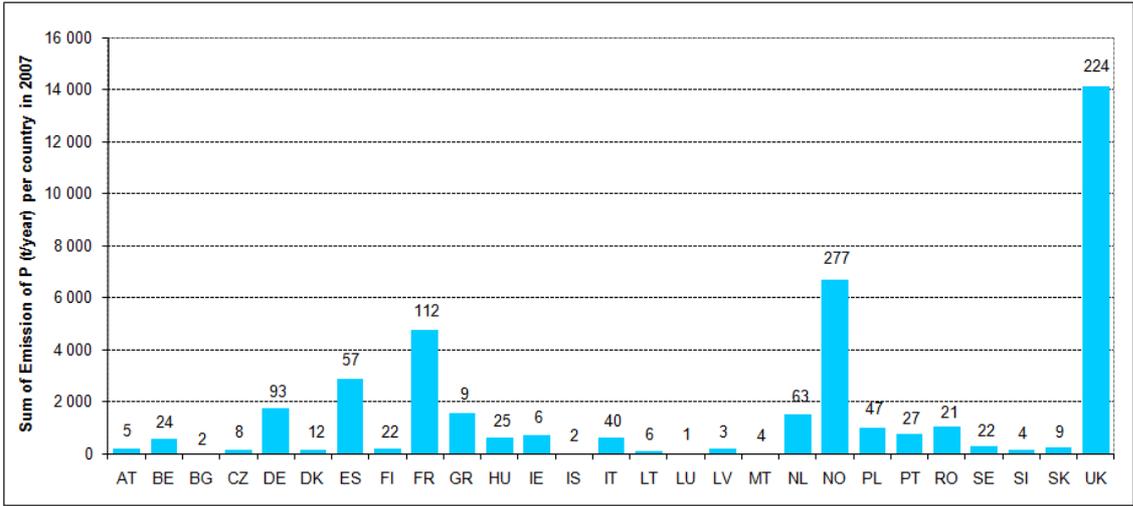
*Figure 2.2b - **Sum of total phoshorus discharges and number of facilities** per country*

Figure 2.2 shows that the countries with the highest point source emissions of N and P are UK, Norway, France, Germany and Spain. For Norway, the high emission is mainly due to a high number of aquaculture facilities along the coastline. Countries with a high number of reported facilities (and with higher specific emissions) are not necessarily worse then others but instead may simply reflect better reporting under E-PRTR. The figures therefore should be interpreted with caution.

The emissions of N, P and TOC from the largest facilities (according to the annual discharges) are in the Table 2.1, including the distribution of emission among the different main economic categories defined according to the NACE system. The data shows that it is the waste water sector that has the highest emissions of N, P and TOC, whereas the agriculture, forestry and fishing (incl. aquaculture) sector is the second highest sector for P and TOC emissions (see also Figure 2.3).

Table 2.1 - Distribution of total nitrogen, phosphorus and organic carbon emissions among major economic categories

NACE Main Economic Category	NACE Main Economic Category Name	Total N (kg/year)	% of Total Quantity	Count of Facilities	% of Facilities
A	Agriculture, forestry and fishing (incl.aquaculture)	31 403 000	8,2	318	25,5
B	Mining and quarrying	1 032 600	0,3	7	0,6
C10	Manufacture of food products and beverages	3 312 000	0,9	27	2,2
C	Other manufacture	53 933 700	14,1	233	18,7
D	Electricity, gas, steam and air conditioning supply	6 095 200	1,6	16	1,3
E	Water supply; sewerage; waste managment and remediation activities	278 685 000	72,6	631	50,6
S	Other	9 241 300	2,4	15	1,2
Total sum N		383 702 800	100,0	1 247	100,0

NACE Main Economic Category	NACE Main Economic Category Name	Total P (kg/year)	% of Total Quantity	Count of Facilities	% Of Facilities
A	Agriculture, forestry and fishing (incl.aquaculture)	6 350 350	15,6	332	29,5
B	Mining and quarrying	25 370	0,1	4	0,4
C10	Manufacture of food products and beverages	1 318 840	3,2	72	6,4
C	Other manufacture	2 890 190	7,1	149	13,2
D	Electricity, gas, steam and air conditioning supply	177 010	0,4	8	0,7
E	Water supply; sewerage; waste managment and remediation activities	29 021 120	71,1	545	48,4
S	Other	1 009 930	2,5	15	1,3
Total sum P		40 792 810	100,0	1 125	100,0

NACE Main Economic Category	NACE Main Economic Category Name	Total TOC (kg/year)	% Of Total Quantity	Count Of Facilities	% Of Facilities
A	Agriculture, forestry and fishing (incl.aquaculture)	126 166 660	4,5	328	22,0
B	Mining and quarrying	3 715 300	0,1	16	1,1
C10	Manufacture of food products and beverages	38 493 443	1,4	83	5,6
C	Other manufacture	2 376 594 804	84,1	422	28,4
D	Electricity, gas, steam and air conditioning supply	24 196 750	0,9	16	1,1
E	Water supply; sewerage; waste managment and remediation activities	243 134 507	8,6	612	41,1
S	Other	14 432 100	0,5	11	0,7
Total sum TOC		2 826 733 564	100,0	1 488	100,0

Figures 2.3 a, b show the distribution of total N and P emissions across the different economic categories.

Emissions to water reported under E-PRTR do not cover the smaller 10-20% of facilities. In RBD's with a number of such smaller facilities, E-PRTR data alone will significantly under-estimate total emissions.
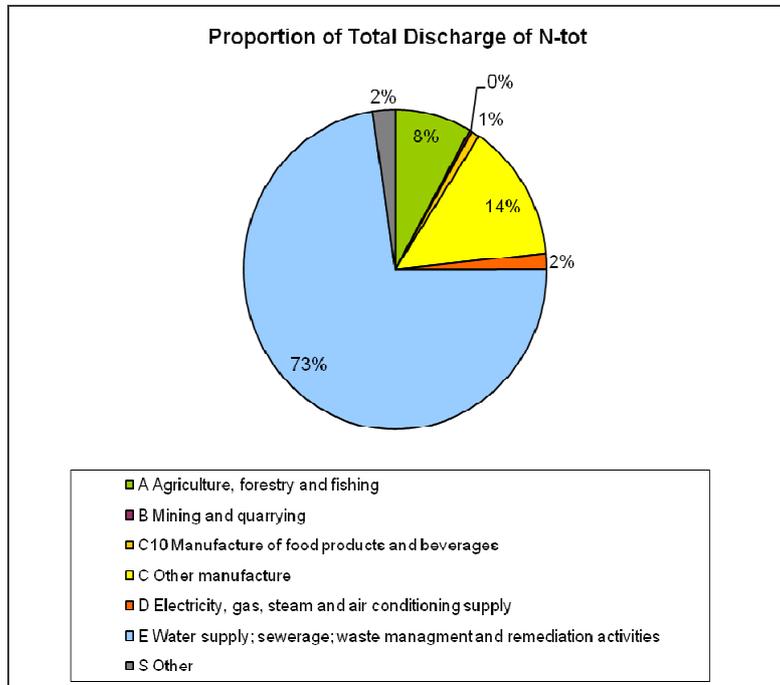
Figure 2.3a - Distribution of total N emissions on various economic categories (NACE)
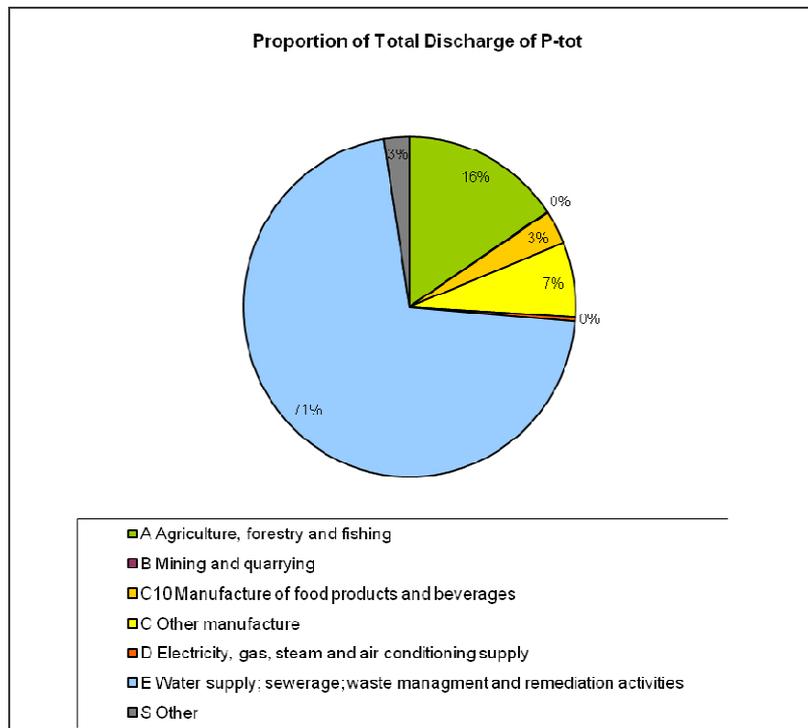
Figure 2.3b - Distribution of total P emissions on various economic categories (NACE)

2.3. Atmospheric deposition

2.3.1. *Nitrogen deposition:*

Large parts of Europe is exposed to fairly high levels of N-deposition with inputs in the year 1999 ranging from ca. 10 kg N /ha to 20 kg N /ha in Southern Norway and Sweden, as well as large parts of England, France and Eastern Europe, whereas inputs up to 34 kg N /ha occurred in large parts of Germany, Northern Italy, Belgium and the Netherlands (Bouraoui et al., 2009; Map 2.5). During the last decade the atmospheric deposition of N in Europe as a whole has decreased slightly, but there are large variations from year to year, so the trend is very uncertain (Fig. 4.1 in EMEP, 2009). The contribution of atmospheric N deposition to the total diffuse N loads is highest in the Northern countries including the Baltic Sea, but less in the rest of Europe, where high agricultural N emissions to water are dominating the diffuse N sources (Bouraoui et al., 2009).
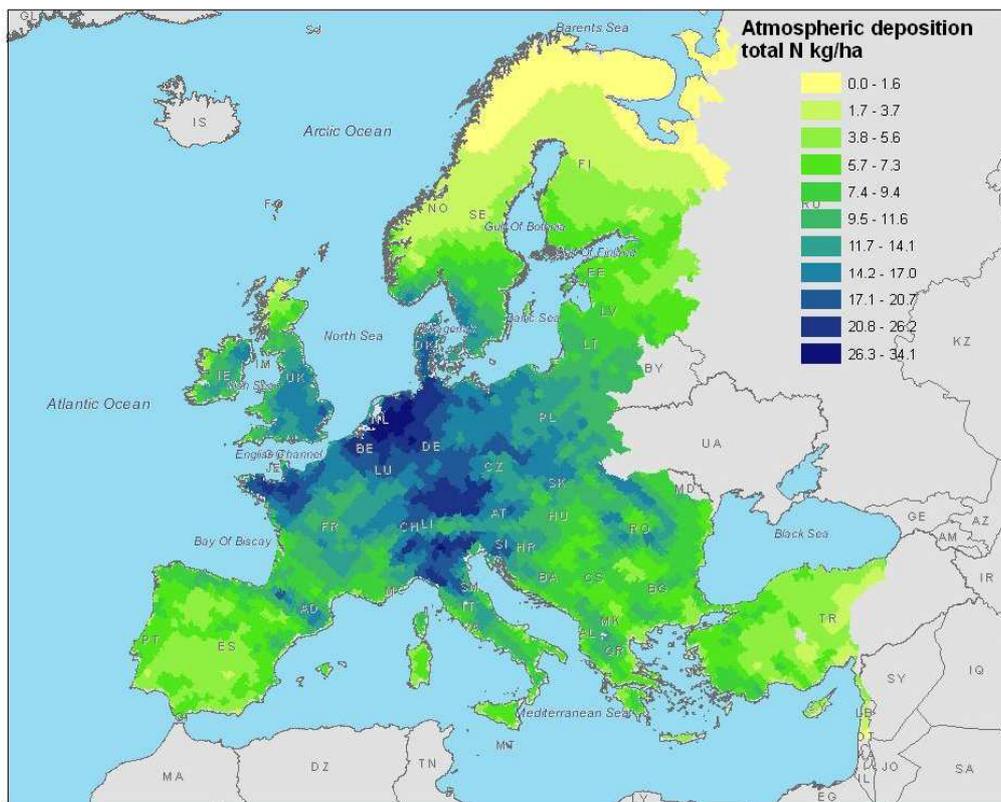
*Map 2.5 - **Atmospheric deposition of nitrogen** (total N). Data from EMEP 2001. Bouraoui et al. 2009, JRC-IES*

2.3.2. *Phosphorus deposition*

According to Rolff et al. (2008) the most comparable recent estimates of atmospheric phosphorus deposition are those reported by Knulst (2001), who found annual deposition rates of 5.8, 23.0 and 116.2 kg total P km^{-2} yr^{-1} at three Swedish inland sites (Knulst, 2001). An average for a selection of Norwegian monitoring stations in 1998 was 181 kg total P km^{-2} yr^{-1} (Solberg et al., 1999). Pollman et al. (2002) reported literature estimates of bulk depositions ranging from 4 to 230 kg total P km^{-2} yr^{-1} corresponding to and annual P deposition of 0.04-

2.3 kg /ha in a wide range of environments. The Helsinki Commission estimated the total load of phosphorus on the Baltic Sea in the year 2000 to be 34 600 tons (HELCOM, 2004), and assumed atmospheric deposition to be less than 5% of the total load. At the European level the contribution of atmospheric P deposition to the total diffuse P load is minor.

2.4.   Other sources

Forestry and aquaculture can also contribute to nutrient emissions to water, and may be major nutrient sources in areas where other sources are minor, such as in the Northern parts of Scandinavia. Aquaculture will primarily be a nutrient source in coastal waters, and has been found to constitute the largest nutrient source in Western and Northern Norway, where the number of fish farms is high and other sources are low (Skarbøvik et al., 2008).

2.5.   Total nutrient loads to coastal waters and proportion of diffuse sources

A predicted pan-European overview of total nutrient loads to European coastal waters is presented by Bouraoui et al., 2009 (Table 2.2). The total loads of N and P to coastal waters are shown in Maps 2.6 and 2.8, whilst the proportion of diffuse sources to total N and total P loads from different river basins is given in Maps 2.7 and 2.9. The observed data, against which the model is calibrated, are derived from marine and river conventions (HELCOM, OSPAR, ICPDR) and UNEP/MAP/WHO and originate from 1995-2002. The modelled data (FATE) are based on the year 2000 (Bouraoui et al., 2009).

*Table 2.2 - **Total nitrogen and total phosphorus annual loads into the European Seas** reported in other studies and estimated by JRC (further explanations in Bouraoui et al. 2009)*

Seas	Measured total N input (10$_3$ ton/yr)	Predicted total N input (10$_3$ ton/yr)	Measured total P input (10$_3$ ton/yr)	Predicted total P input (10$_3$ ton/yr)
Mediterranean (Europe)	786-2680 [b]	882	97-270 [b]	70
Baltic	745 [a]	564	34 [a]	35
North Sea	1267 [c]	1286	78 [c]	69
Atlantic (France, Spain, Portugal, UK, Ireland)	829 [c]	1502	64 [c]	75
Black Sea (Danube)	635 [d]	686	62 [d]	41

[a] Helsinki Commission (2003), [b] UNEP/MAP/WHO (1996), [c] OSPAR (2002), [d] ICPDR (2009),

Predicted total N loads vary from ca. 500 kt per year to 1500 kt per year, whilst, for total P, the predicted loads vary from ca. 30 kt yr^{-1} to ca. 70 kt yr^{-1} (Table 2.2). The ICPDR has published draft River Basin Management Plans under the Water Framework Directive for the Danube. In these it is estimated that the current total annual nutrient loads discharging through the mouth of the Danube river to be 686 kt N and 62 kt P (based on the MONERIS model) (ICPDR, 2009). These numbers are roughly the same for N inputs, but higher for P inputs than those estimated by the FATE model for the year 2000 for the Black Sea (Table 2.2), which may be caused by differences between the two models in terms of estimation of P inputs. Due to the large area of the Danube catchment, the load per unit area is rather low (Map 2.6).
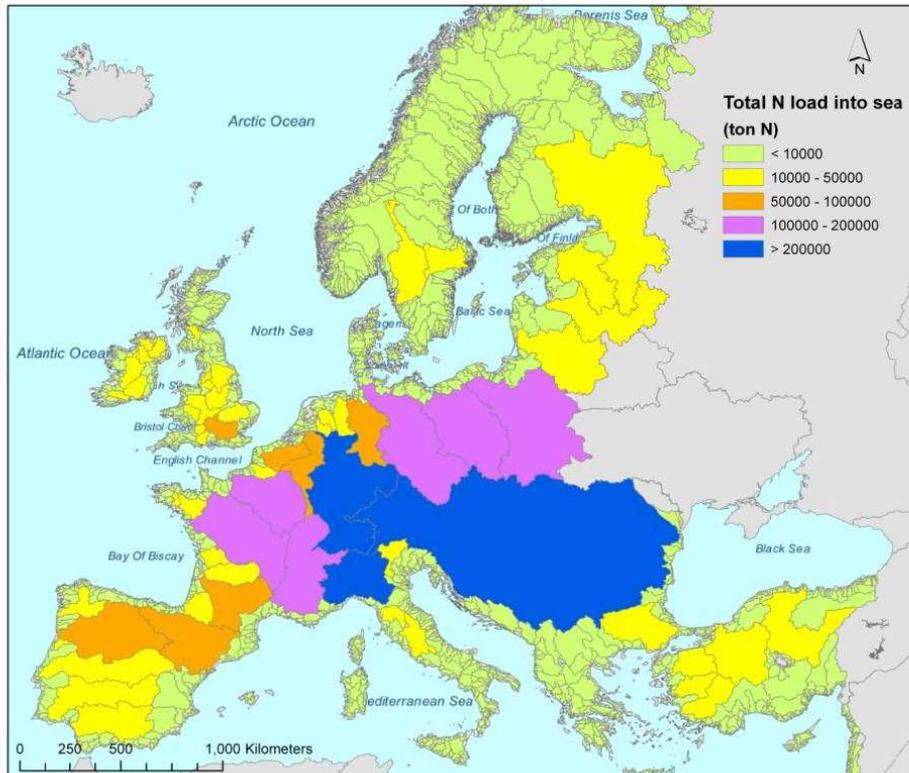
Map 2.6 - *Total nitrogen load* per unit area (tons N) (Bouraoui et al., 2009).

The highest absolute N load to European seas is observed from the large rivers; the Danube, Rhine, Elbe, and Po river, based on the data and modelling for the year 2000 (Map 2.6).

The proportion of diffuse nitrogen is highest in intensive agricultural areas and also where rainfall is rather high. In Southern Europe diffuse sources contribute less to the total N load than in other regions (Map 2.7) due to lower inputs of mineral and organic fertiliser, lower rainfall, and, generally, lower levels of connection to and treatment within municipal waste-water treatment plants.
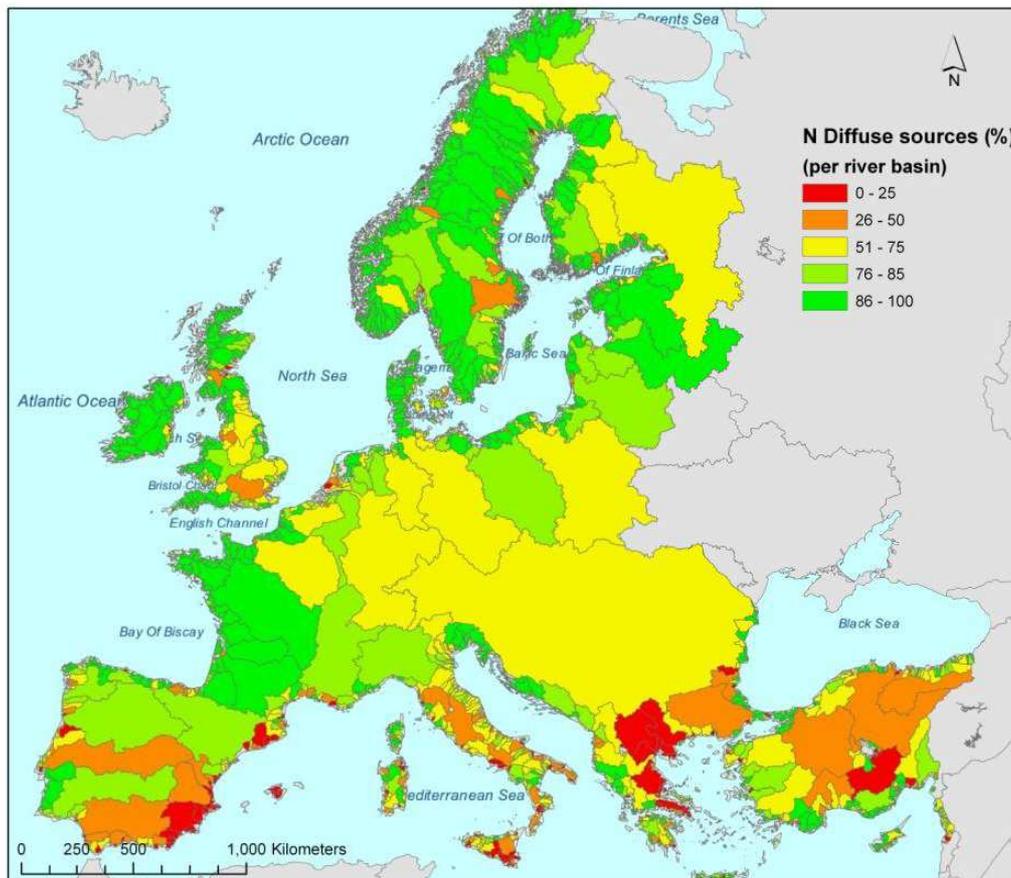
Map 2.7 - ***Contribution of diffuse sources to total nitrogen load into the sea*** *per basin. The green colour indicates a predominance of diffuse sources (represented mainly by agriculture), while the red colour signifies a higher contribution from point sources (represented mainly by the waste water discharges). (Bouraoui et al., 2009).*
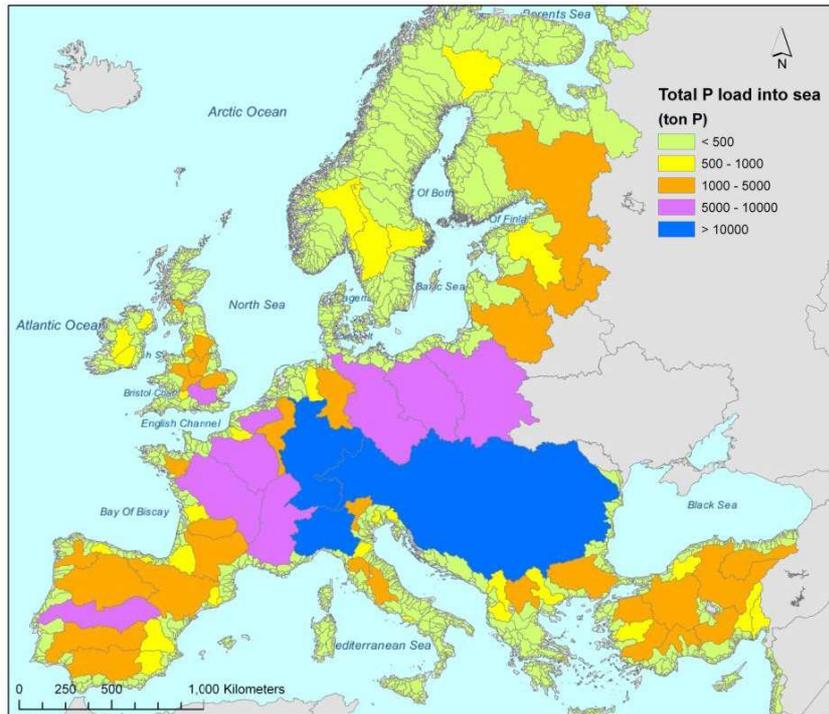
Map 2.8 - *Total phosphorus load* per unit area (ton P/km2). (From Bouraoui et al., 2009)
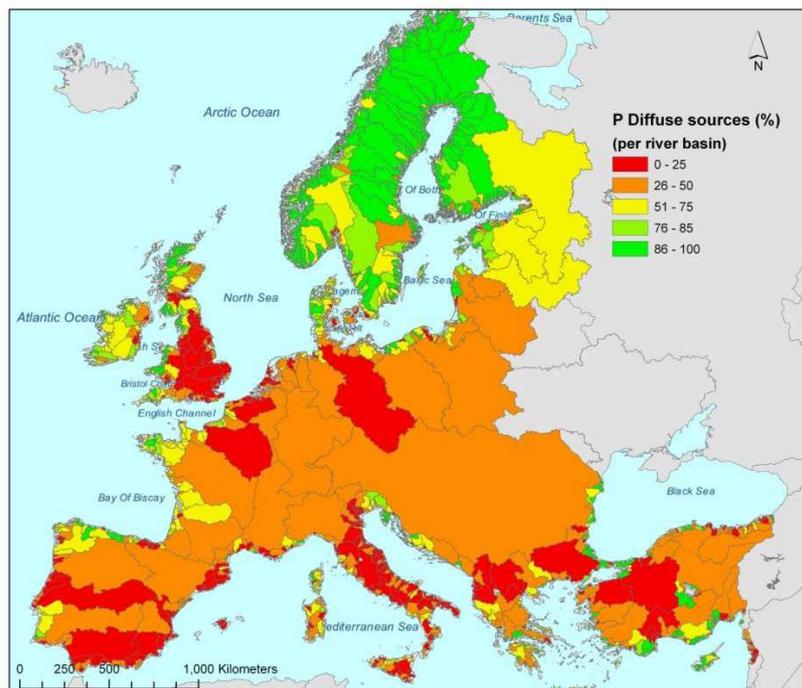
Map 2.9 - *Contribution of diffuse sources to total phosphorus load into the sea* per basin. The green colour indicates a predominance of diffuse sources (represented by agriculture), while the red colour signifies a higher contribution from point sources (represented mainly by the waste water discharges) (Bouraoui et al., 2009).

As for nitrogen, the highest absolute total phosphorus loads come from the large rivers; the Danube, Rhine, Rhone, Elbe and Po, but also from Southern England and one transboundary river basin encompassing Spain and Portugal (Map 2.8).

Diffuse sources contribute less to the total P load than to the total N load across most of Europe, except in the Northern (Scandinavian) countries where the population density is lower (Map 2.9). This is also confirmed by a previous EEA report concluding that run-off from agriculture is the principal source of N pollution contributing 50-80% of the load, whereas for P, point and diffuse sources across Europe as a whole each contribute about 50% (EEA, 2005).
These results are supported by more detailed, but locally focused, studies from the Euroharp project showing that agriculture is the major N source to European surface waters contributing > 70% in most of the catchments studied, whilst P loss from agriculture is the major source in ca. 50% of the catchments studied, contributing on average 50% to total loads. However, large variation between catchments is apparent (Kronvang et al., 2009).

Trends during the past 30 years shows that the nutrients coming from point sources have decreased whereas those coming from diffuse sources have remained at a rather constant level in many large river basins (Danube, Rhine, Elbe) (EEA, 2005).

More recent data and information on nutrient loads to freshwater and coastal waters are now available in the river basin management plans that have been submitted to the European Commission in March 2010. These data will be analysed as the basis for a forthcoming Water report at the European level in 2012.

Riverine nutrient loads to European seas are estimated to decrease in future years (Bouwman et al., 2005), in part, due to improved treatment of sewage and industrial emissions (OSPAR, 2008; HELCOM, 2009). However, future climate change may alter these projections, since in many areas more frequent and intense rainfall is predicted and is likely to lead to increased nutrient loss, particularly that which is transported in particulate form (attached to soil) on agricultural land and, via storm overflows in the urban environment. Both these processes may counteract the nutrient reduction measures currently taken to reduce nutrient loads to freshwater and coastal waters in the Central and Northern Europe.

3. Nutrient concentrations in rivers

3.1. Observed status and trends of nutrient concentrations in rivers

The pan-European assessment of the eutrophication status is based primarily on observed status and trends of nutrients and organic substances concentrations which are commonly monitored and assessed by EEA core set indicators 019 (oxygen consuming substances in water) and 020 (nutrients). The primary data source is the Eionet Rivers database.

3.1.1. Nitrate concentrations:

River nitrate concentrations aggregated at a river basin district scale (Map 3.1) are currently below the 11.3 mg l^{-1} NO_3-N limit (equivalent to 50 mg l^{-1} NO_3) of the Nitrates and Drinking Water Directives. Less than 1% of individual rivers across Europe exceed the limit. However, current concentrations are still sufficient to promote eutrophication in many of Europe's rivers (see Chapter 6.3), and also contribute to eutrophication of coastal waters (EEA Indicator CSI023). Generally, concentrations are lowest in Scandinavia (< 0.8 mg l^{-1} NO_3-N) and highest (> 3.6 mg l^{-1} NO_3-N; equivalent to > 16 mg l^{-1} NO_3) in parts of France, Belgium and the UK. For much of the rest of Europe, average concentrations lie between 0.8 and 3.6 mg l^{-1} NO_3-N.
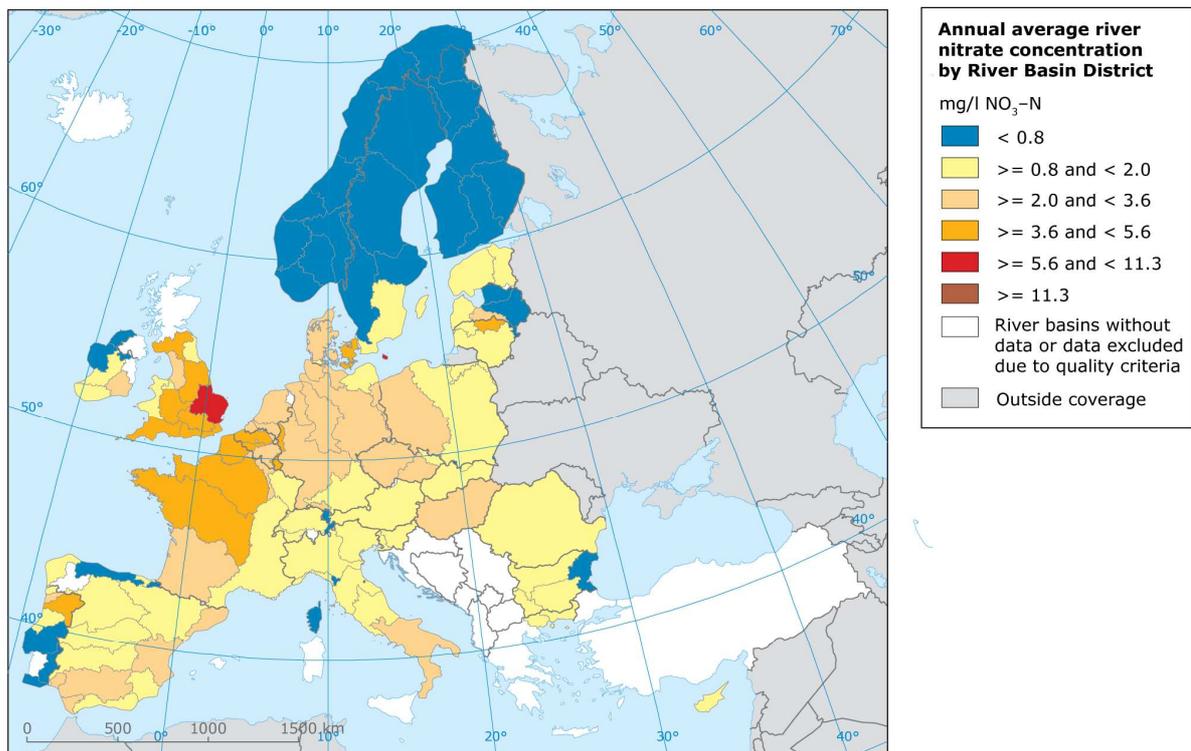
Map 3.1 - Annual average river nitrate concentration (mg l^{-1} NO_3-N) in 2008, averaged by river basin district. *Source; EEA/ETC-Water based on data reported to Eionet.*
The map is based on the most recent data available, in most cases from 2008 (see WISE interactive maps, "Map explanation" for more information). Data on total oxidised nitrogen are used in cases where nitrate data are missing. Concentration units are in mg l^{-1} N (= mg NO_3 divided by 62/14). Data from the following countries are included (number of stations): AL (51), AT (60), BA (19), BE (58), BG (98), CH (10), CY (18), CZ (69), DE (252), DK (41), EE (60), ES (1160), FI* (135), FR (1522), GR (14), HR (45), HU* (99), IE* (104), IS (2), IT (688), LI (1), LT (53), LU (4), LV (40), MK (19), NL (15), NO (46), PL (107), PT (39), RO (117), RS (76), SE* (119), SI (20), SK (85), TR (5), UK (139) (* = total oxidized nitrogen)*

Complete series 1992-2008
(3-year gaps interpolated)
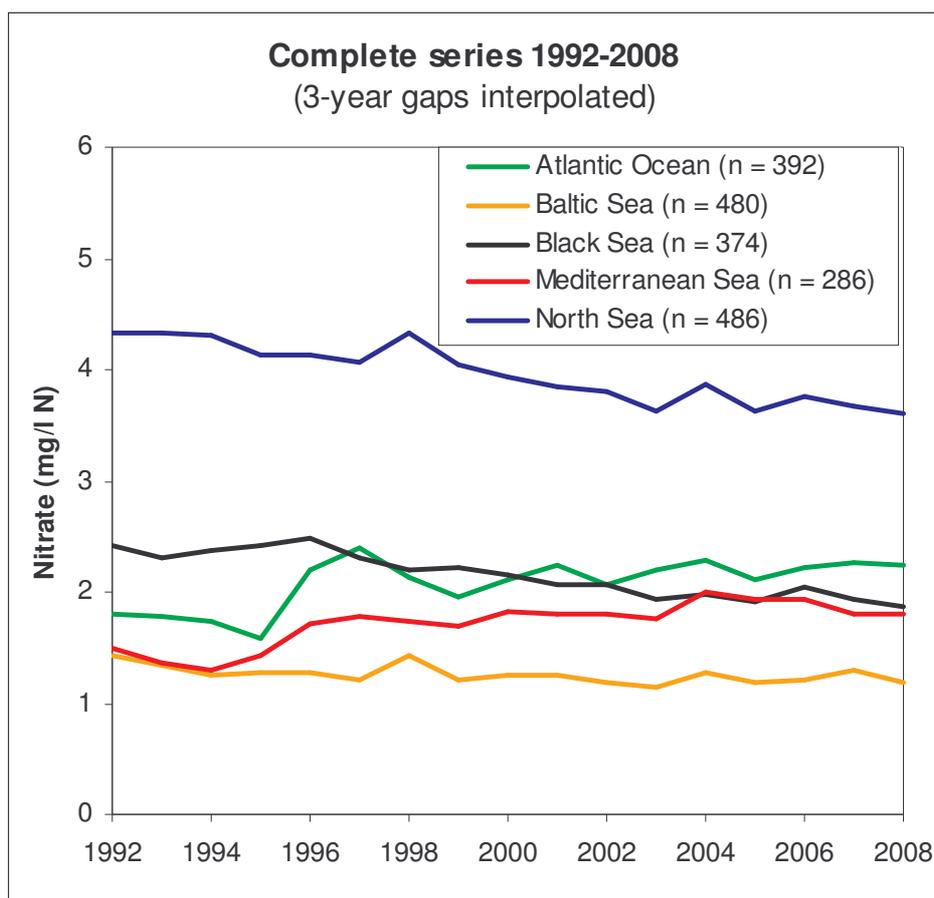
Figure 3.1 - NO3 concentrations in rivers in different sea regions 1992-2008. Graphs based on data reported to WISE/EIONET. The sea region *data series are calculated as the average of annual mean data from river monitoring stations in each sea region. The data thus represents rivers or river basins draining into that particular sea. Only complete series after inter/extrapolation are included (see* indicator specification*). Two regions are not shown in the figure due to a lack of data (Barents Sea: 1 station, Norwegian Sea: 2 stations); these stations have NO₃ concentrations below 0.8 mg l⁻¹ N.*
North Sea (national RBDs of AT, BE, CZ, DK, FR, LU, UK).
Atlantic Ocean (national RBDs of ES, FR, UK).
Baltic Sea (national RBDs of CZ, DK, EE, FI, LT, LV, SK).
Black Sea (national RBDs of AT, BA, BG, CZ, HR, HU, RO, SI, SK).
Mediterranean Sea (national RBDs of BA, BG, ES, FR, IT, SI).

Data source: Waterbase - Rivers (version 10)

Figure 3.1 shows time-series of NO_3 concentrations in European rivers aggregated at the major sea region level. The North Sea rivers have the highest concentration, whilst the Baltic sea rivers have the lowest. Declining nitrate concentration trends are most clearly observed in the North Sea and Black Sea regions, while there has been a slight increase for the Atlantic Ocean and Mediterranean Sea regions. Declining trends in river nitrate are likely to reflect a number of factors, including improved management of agricultural land, but also a reduced discharge of nitrogen from municipal wastewater treatment plants, as driven by implementation of the UWWT Directive. Rivers draining land with intense agriculture or high population density generally have the highest nitrate concentrations. In Mediterranean rivers also water scarcity and droughts counteract the effects of nitrate reduction measures.

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Overall, a statistically significant decrease in average nitrate concentrations is evident at 29% of river monitoring stations across Europe, whilst in 16%, a statistically significant increase has occurred. Significant decreases in river nitrate concentration during the whole timeseries period from 1992-2008 are particularly evident in Denmark, the Netherlands, the Czech Republic, Germany, Hungary, Slovakia, Latvia and Sweden (Map 3.2, left panel). However, more than 30% of rivers in Ireland, Estonia, Spain and Switzerland exhibit a rising trend over the same time period. Since the year 2000 NO_3 concentration has not changed much in the majority of stations (Map 3.2, right panel). Across Europe as a whole, the rate of improvement is still slow.
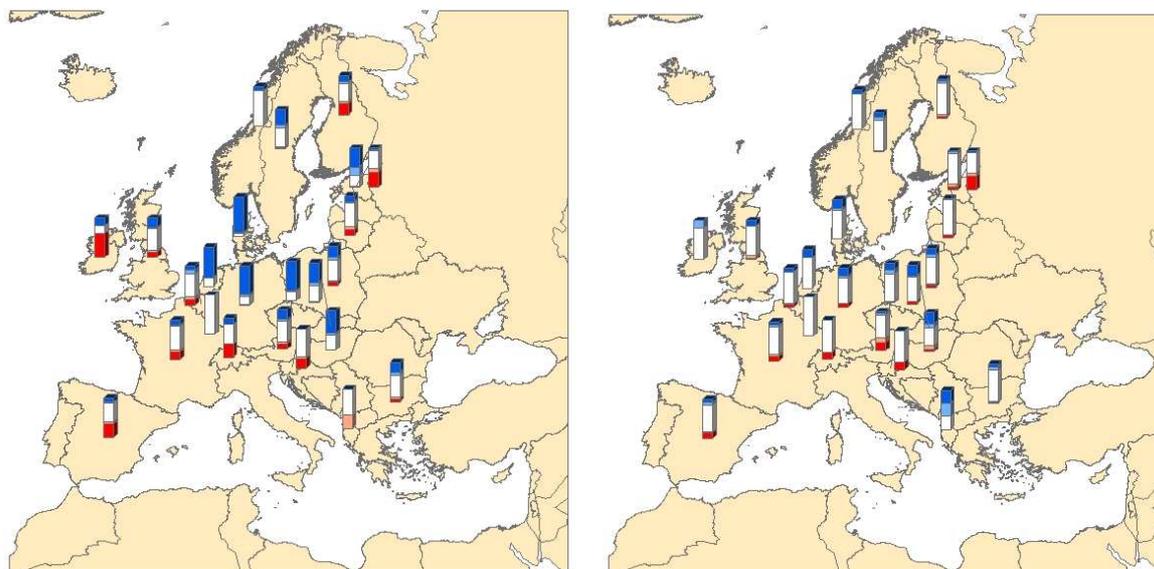
Map 3.2 - Trend analyses for NO3 concentrations in rivers in different countries. *Left panel shows trends for the whole period 1992-2008. Right panel shows trends for the last 9 years 2000-2008. Graphs based on data reported to WISE/EIONET. Station selection is the same as for the time series-plots in Figure 3.1. Blue and light blue colours show strongly and slightly declining trends, white shows no trend, whereas red and pink show strongly and slightly rising trends.*

3.1.2. Phosphate concentrations

The river basin districts with relatively low concentrations of phosphate in rivers are found in Northern Europe (Norway, Sweden, and Finland) and the Alps (Map 3.3), predominantly reflecting regions of low population density and/or high levels of wastewater collection and treatment. In contrast, relatively high concentrations (> 0.1 mg l^{-1} P) are found in several river basins with higher population densities and intensive agriculture, including: southeast UK, part of the Netherlands, Belgium, Southern Italy, central Spain and Portugal, western Poland, Hungary, Bulgaria, Macedonia, northern Greece. Given that phosphate concentration > 0.1 mg l^{-1} P is sufficiently high to promote freshwater eutrophication in rivers, the observed high values in many river basins of Europe are of particular concern.
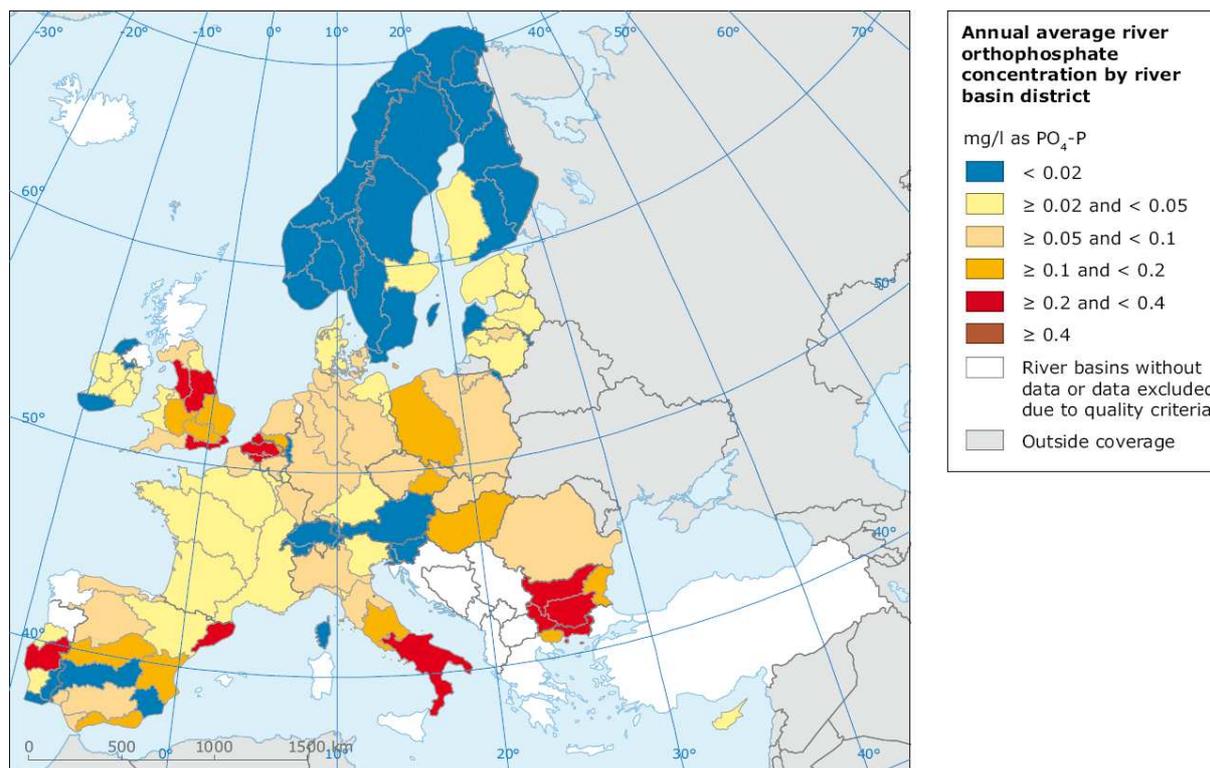
Map 3.3 - Concentration of phosphorus in rivers (ortophosphate) 2008: average concentration level per river basin district. *The map is based on the most recent data available, in most cases from 2008 (see WISE interactive maps, "Map explanation" for more information). Concentration units are mg l⁻¹ P. Data from the following countries are included (number of stations): AL (51), AT (60), BA (6), BE (58), BG (105), CH (10), CS (76), CY (18), CZ (68), DE (244), DK (41), EE (60), ES (1078), FI (134), FR (1521), GR (14), HR (45), HU (98), IE (158), IS (2), IT (792), LT (53), LU (3), LV (40), MK (19), NL (15), NO (46), PL (125), PT (26), RO (117), SE (116), SI (20), SK (85), TR (5), UK (140)*

Time-series of PO_4 concentrations in European rivers aggregated at the sea region level (Figure 3.2) show that the PO_4 concentrations are generally lowest for rivers draining to the Baltic Sea and (after 1999) highest for rivers draining to the Atlantic Ocean. The strongest decrease in PO_4 concentration is found for rivers draining to the North Sea, but the decrease is almost similarly strong in all other regions, except the Baltic Sea rivers.

The trend analyses done on the timeseries data confirm this picture: at 42 % of the river stations there has been a significant decline in PO_4 concentration since 1992 (an additional 5 % marginally significant), while there has been an increase at only 6% of the stations (an additional 2% marginally significant). This decrease reflects the success of legislative measures to reduce emissions of phosphorus such as those required by the Urban Waste Water Treatment Directive.

Since the year 2000 the decline in phosphate concentration is less pronounced in rivers draining to all of the sea regions, except the North Sea rivers, where the decline in concentration is still ongoing.

Complete series 1992-2008
(3-year gaps interpolated)
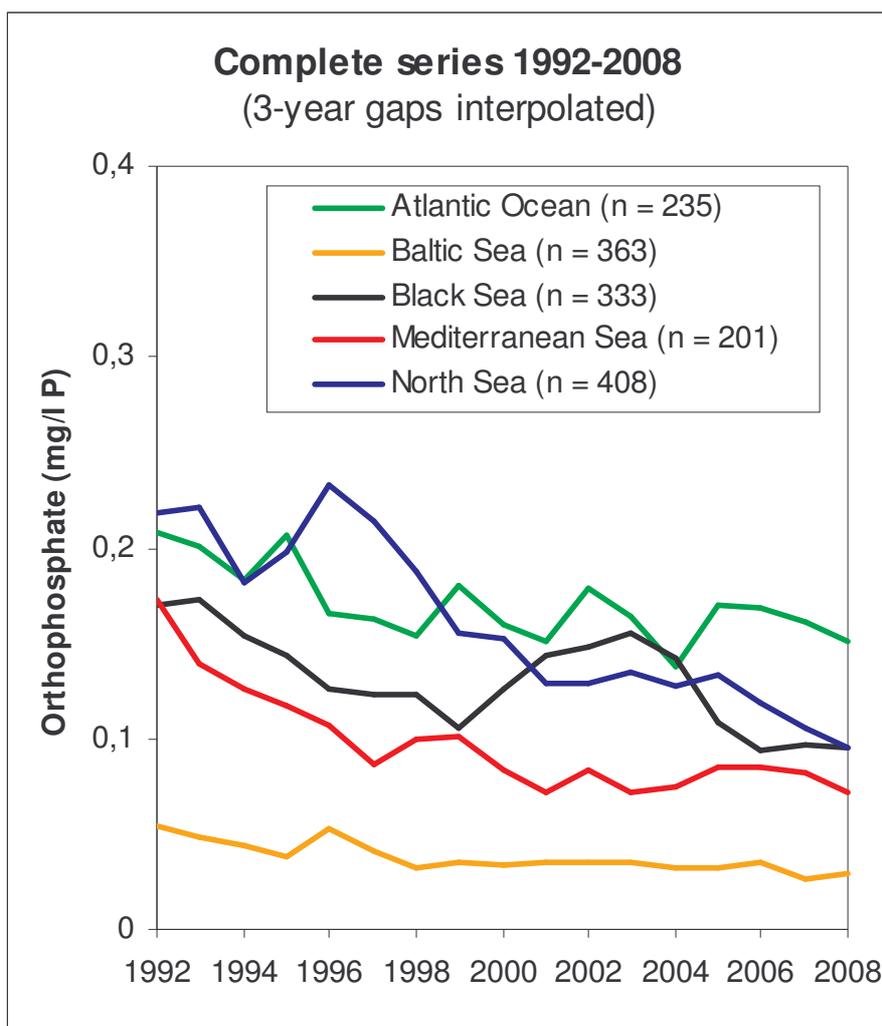
Figure 3.2 - PO4-P concentrations in rivers in different sea regions 1992-2008. Graphs based on data reported to WISE/EIONET. Number of data series given in parenthesis. The sea region data series are calculated as the average of annual mean data from river monitoring stations in each sea region. The data thus represents rivers or river basins draining into that particular sea. Only complete series after inter/extrapolation are included (see indicator specification). Two regions are not shown in the figure due to a lack of data (Barents Sea: 1 station, Norwegian Sea: 2 stations); these stations have PO$_4$ concentrations < 0.02 mg l^{-1} P.

North Sea (national RBDs of AT, BE, CZ, DK, FR, LU, UK).
Atlantic Ocean (national RBDs of ES, FR, UK).
Baltic Sea (national RBDs of CZ, DK, EE, FI, LT, LV, SK).
Black Sea (national RBDs of AT, BA, BG, CZ, HR, HU, RO, SI, SK).
Mediterranean Sea (national RBDs of BA, BG, ES, FR, IT, SI).

Data source: *Waterbase - Rivers (version 10)*

Trends for individual countries (Map 3.4) show that the largest proportion of stations with declining trends are found in the countries of Central and Eastern Europe, as well as in the Baltic countries, UK, Denmark and Spain for the whole period 1992-2008. Since 2000, however, fewer declining trends are observed in most countries.
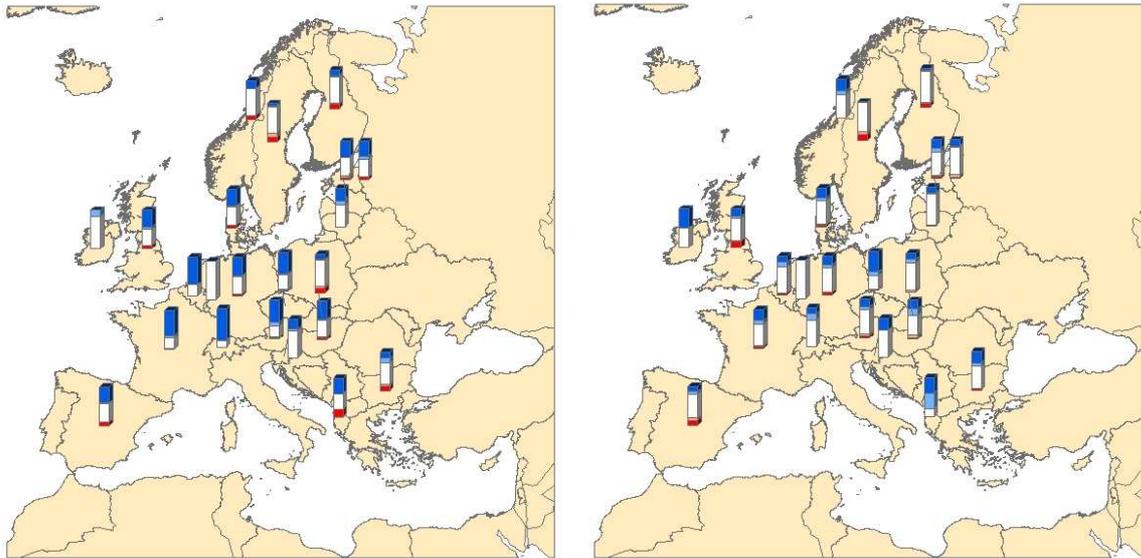
Map 3.4 - Phosphate (PO4-P) trends in rivers in different European countries. *Left panel shows trends for the whole period 1992-2008. Right panel shows trends for the last 9 years 2000-2008. Graphs based on data reported to WISE/EIONET. Station selection is the same as for the time series-plots in Figure 3.2. Blue and light blue colours show strongly and slightly declining trends, white shows no trend, whereas red and pink show strongly and slightly rising trends.*

3.1.3. Biological Oxygen Demand (BOD) and ammonium (NH4) concentrations

The highest BOD concentrations currently found in Europe are located in the Eastern part of the Danube, the Southern part of Italy and in Southern Spain (Map 3.5). The high values found in Southern Europe reflect insufficient wastewater treatment although water scarcity and drought are likely to also play a role whereby lack of water leads to a concentrating of pollutants from point discharges.
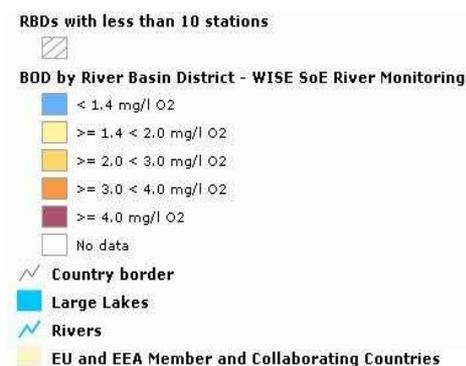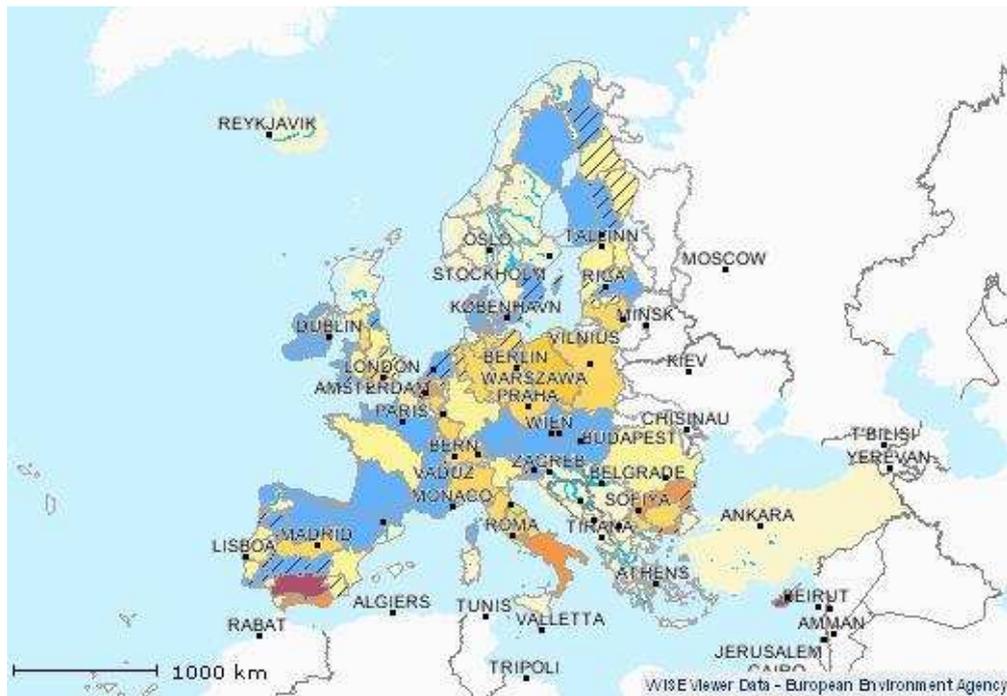
Map 3.5 - BOD concentrations per river basin districts (RBDs) in Europe, data from 2008.
Countries with more than 50 % of all stations with the lowest BOD concentrations (class 1) are
Finland, France, Croatia, Spain, Austria, Ireland, Denmark, the Netherlands and Slovenia. Countries
with more than 20 % of the stations with the highest BOD concentrations (class 5) are Italy, Cyprus,
Romania, Bulgaria, Hungary, Belgium, Albania, Macedonia and Turkey.
Countries with more than 50 % of all stations with the lowest total ammonium concentrations (class 1)
are Croatia, Estonia, Spain, Serbia, Slovenia, Bosnia and Herzegovina, Austria, Sweden, Ireland,
Finland and Norway. Countries with more than 20 % of the stations with the highest total ammonium
concentrations (class 5) are Greece, Bulgaria, Luxembourg, Romania, Macedonia, Albania and Bel-
gium.

The BOD concentration in European rivers was reduced substantially during the 1990s, and
has continued to decrease during the last decade for most regions, although the rate of reduc-
tion is less than in the 1990s (Figure 3.3, left panel). This decrease is attributed to the imple-
mentation of the Urban Wastewater Treatment Directive (UWWTD). The BOD concentration
in rivers going to different sea regions shows a clear downward trend for all regions.

Ammonium (NH₄) concentration in rivers (Figure 3.3, right panel) reflects the same pattern as for BOD with a general decline since the early 1990s. The rivers going to the Atlantic Sea (south of the North Sea) have the highest NH₄ concentration.
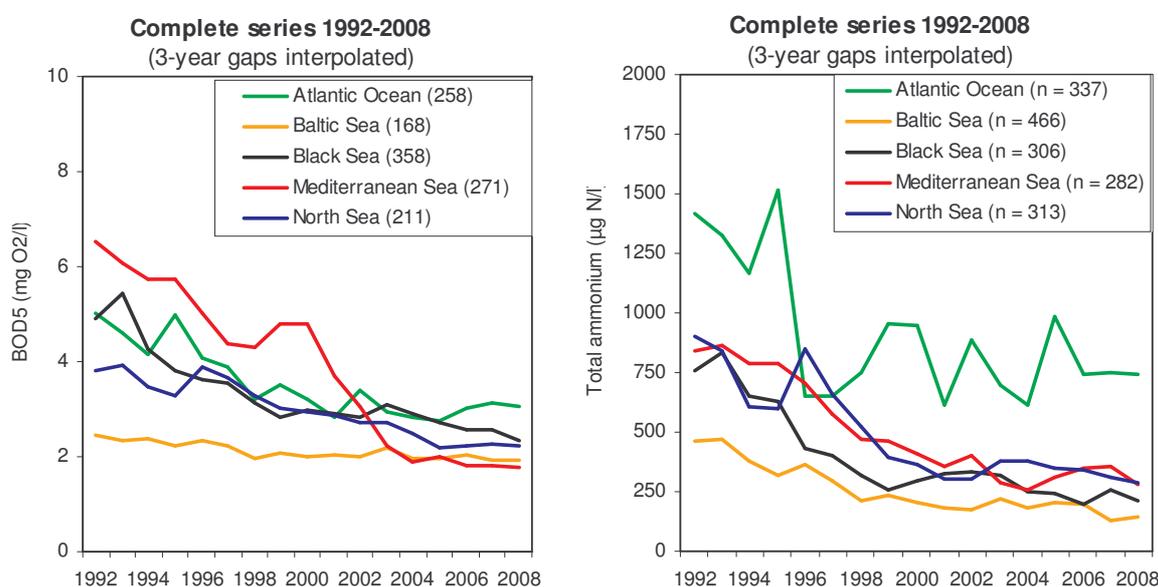
Figure 3.3 - BOD5 (left panel) and ammonium (right panel) concentrations in rivers between 1992 and 2008 in different regions of Europe. *Sea regions: Concentrations are expressed as annual mean concentrations. 3-year gaps of missing values have been interpolated or extrapolated. Only complete series with no missing values after this interpolation/extrapolation are included. Number of river monitoring stations included in analysis per region is noted in brackets. BOD7 data (EE, FI, LT (1996-2008), LV (1996-2001)) has been recalculated into BOD5 data.*

North Sea (national RBDs of AT, BE, CZ, DK, FR, LU, UK).
*Atlantic Ocean (national RBDs of ES, FR, UK).*
Baltic Sea (national RBDs of CZ, DK, EE, FI, LT, LV, SK).
Black Sea (national RBDs of AT, BA, BG, CZ, HR, HU, RO, SI, SK).
Mediterranean Sea (national RBDs of BA, BG, ES, FR, IT, SI).

***Data source****: Waterbase - Rivers (version 10)*

The largest decrease of **BOD** from 1992 to 2008 occurred in the rivers draining to the Mediterranean Sea (Figure 3.3, left panel). Due to this high decrease (above 70 %) their concentrations reached the lowest level of BOD among European rivers in recent years (less than 2 mg l⁻¹ O₂). The decrease in concentrations is somewhat lower in rivers draining to the Black Sea (above 50 %). The rivers draining to the North Sea and Atlantic Ocean have similar decrease (about 40 %). The smallest decrease occurred in the rivers draining to the Baltic Sea (above 20 %) with more or less stable concentrations (about 2 mg l⁻¹ O₂). The highest recent concentrations are found in the rivers draining to the Atlantic Ocean (about 2 mg l⁻¹ O₂), where the concentrations increased compared to 2005.

The largest decrease of **total ammonium** from 1992 to 2008 occurred in the rivers draining to the Black Sea (above 70 %) (Figure 3.3 right panel). The rivers draining to the Baltic Sea, Mediterranean Sea and North Sea have similar decrease in concentrations (above 65 %). The decrease in concentrations is the lowest in the rivers draining to Atlantic Ocean (less than 50 %) with very fluctuating and high concentrations (above 700 µg l⁻¹ N). In other European

rivers the recent concentrations are lower than 300 µg l⁻¹ N and are still decreasing. The lowest concentrations are found in the rivers draining to the Norwegian Sea and Barents Sea (3 and 5 µg l⁻¹ N in 2008), but the number of included stations is very low (two and one station respectively).

Overall there has been a significant decrease in BOD concentrations at 54.4 % of the stations (an additional 6.2 % marginally significant) on the European rivers between 1992 and 2008, while there has been a significant increase at only 1.2 % of the stations (an additional 3.5 % marginally significant). Similarly, there has been a significant decrease in total ammonium concentrations at 51.7 % of the stations (an additional 5.7 % marginally significant), while there has been a significant increase at only 1.2 % of the stations (an additional 4.5 % marginally significant).

The maps for NH₄ trends at the country level (Map 3.6) confirms this picture showing that most countries have downward trends for most of their reported stations over the whole data period. However, during the years after 2000 there is no trend for the majority of stations (white parts of the columns in the map). The proportion of stations with increasing trend is also higher after the year 2000 (Map 3.6, right panel).

River NH₄ trends 1992-2008   River NH₄ trends 2000-2008
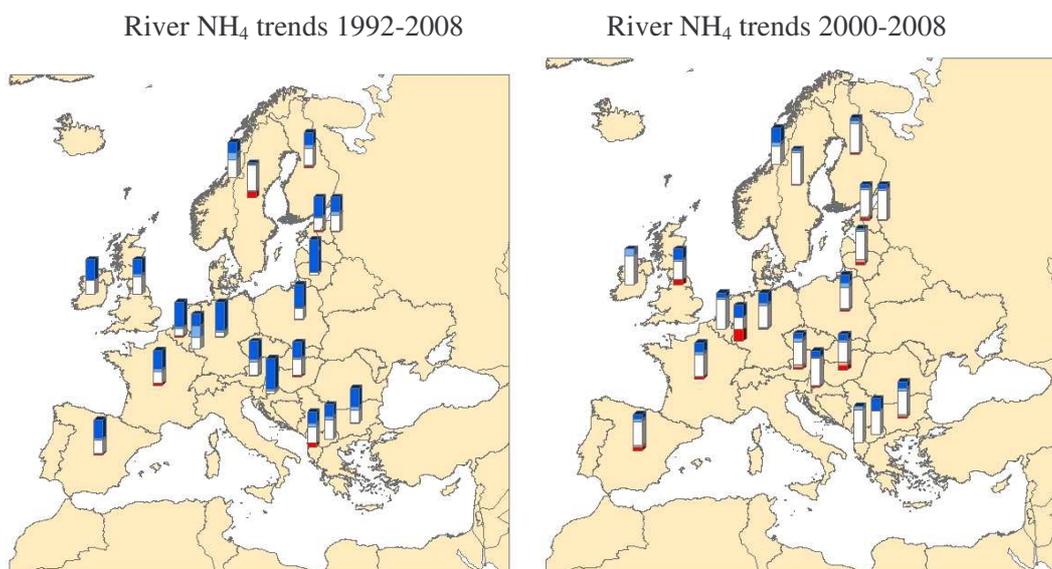
Map 3.6 - Ammonium (NH₄) trends in rivers in different European countries. Left panel: whole time period 1992-2008. Right panel: time period 2000-2008. Blue and light blue colours show strongly and slightly declining trends, white shows no trend, whereas red and pink show strongly and slightly rising trends.

3.2.   Distance to WFD target for nutrients

New standards setting the WFD target concentration for nutrients in rivers are currently available for a 18 countries, as reported by the member states for the official WISE reporting of the WFD Article 13 River Basin Management Plans .The data for these nutrient standards is taken from HTML Factsheets of 18 MS (BG; SK; RO; LT; AT; DE; PL; SE; NL; CZ; FR; EE; ES; LV; GB; FI; IE & BE). The comparability between these standards are limited, since different countries use different statistical expressions, e.g. mean, median, percentiles, etc. For the parameters included in Chapter 3.1 there are between 8 and 12 countries that have reported nutrient standards. Moreover, some standards are specific to the 'type' of river.

These new standards are presently being compiled and will be used for analysing the distance to target for the next European Water report to be published in 2012.

For the current report only a rough summary is presented of the range of preliminary standards available for river nitrate and phosphate. These standards were reported to the European Commission in 2008, as replies to a questionnaire meant to be included in an Annex to the Eutrophication guidance (see Table 3.1 and further info in EC, 2009). In theory, therefore it is possible to use these standards to compare against the current situation in a country giving some indication of the 'distance to target'. However, these draft WFD standards do not currently have any legal status so the use of them in this report can only provide a very preliminary evaluation of the distance to target for the different nutrients.

Moreover, they are compared here against data provided voluntarily to the EEA through the Eionet/SoE-WISE reporting process and **not**, therefore, with official and mandatory reporting required under water related Directives.

In the exercise below, the general range of draft standards across all countries reporting them is compared with the present day nutrient concentrations in rivers across Europe as shown in Maps 3.1 and 3.3.

Table 3.1 - *Summary of preliminary draft nutrient standards for the WFD given as replies to a questionnaire from DG Environment related to the Eutrophication guidance. Numbers are given as standard deviation of mean values for all countries and all types of rivers for which draft values are available. Data from Scandinavian countries are not available.*

Parameter	NO_3-N	PO_4-P
Countries with draft nutrient standards	AT, HU, IT, LT, RO, SI	AT, DE, FR, HU, LT, RO, UK
Range (standard deviation) (mg l^{-1})	2-5	0.06-0.16

These draft standards indicate that the current regional NO_3-N concentrations as reported to Eionet/SoE-WISE are below the higher end of the WFD target range in most of Europe, but exceeds the lower end of this range in many countries (Map 3.1),

For PO_4-P, the current observed concentrations exceed the lower end of the range of PO_4-P standards in most river basins districts (Map 3.3), and in many countries there are river basins exceeding the higher end of the range (Map 3.3).

4. Nutrient concentrations in lakes

4.1. Observed status and trends of Total Phosphorus in lakes and distance to WFD target

Lakes with high concentrations of total P (>0.05 mg l^{-1}) in 2008 are found mainly in RBDs in England, Belgium, the Netherlands, northern Germany, Poland, Hungary, Romania, Bulgaria, Spain, Portugal (Map 4.1). The lowest concentrations (< 0.02 mg l^{-1}) are found in RBDs in the northern countries: Norway, Sweden, Finland, Scotland, Northern Ireland, most of the Baltic countries, as well as in RBDs in the Alpine region, as well as the western part of Germany and southern Italy (very few lakes).
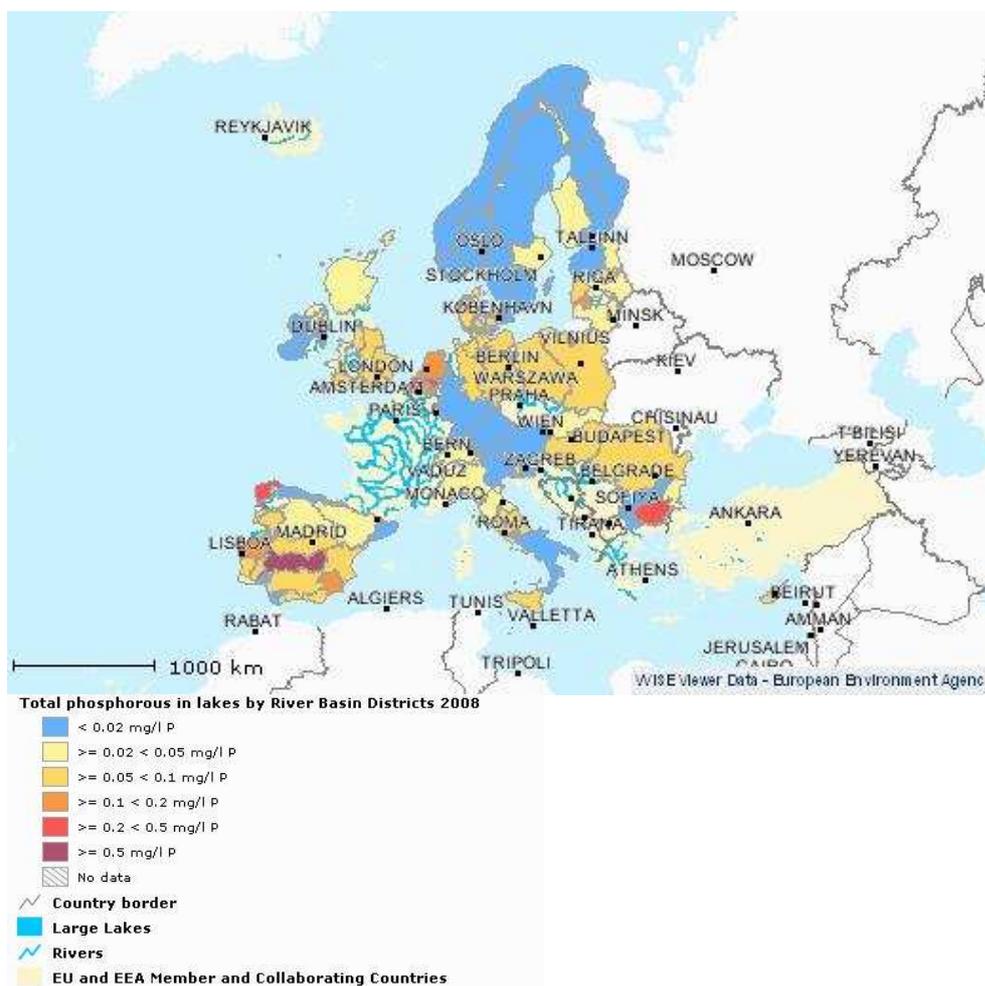
Map 4.1 - Total Phosphorus concentration in lakes. Average concentration level per river basin district based on data from 2008 or latest year of data. Taken from WISE viewer.(see WISE interactive maps, "Map explanation" for more information). Concentration units are mg l^{-1} P. Data from the following countries are included (number of stations): AL (5), AT (24), BA (3), BE (3), BG (6), CH (12), CY (9), DE (17), DK (19), EE (17), ES (268), FI (203), HR (9), HU (11), IE (72), IS (1), IT (142), LT (12), LV (10), MK (2), MT (2), NL (15), NO (109), PL (41), PT (24), RO (16), RS (70), SE (100), SI (8), SK (20), UK (40).

The time series of total phosphorus in lakes compiled from WISE/EIONET data (Figure 4.1) show that total P concentrations decreased in the western region, especially during the 1990s, but that there has been less change in the concentrations over the last decade. For Eastern Europe there is no clear trend; whilst for Northern Europe, the total P concentration is much lower than in the other regions, and there has been little change over the past two decades. For Southern and South-Eastern Europe the data is too limited to be evaluated, and is therefore not presented here. The overall mean concentration of total phosphorus is roughly 20 µg l^{-1} for lakes in the northern region and ca. 50-60 µg l^{-1} for lakes in the western and eastern regions during the last decade.
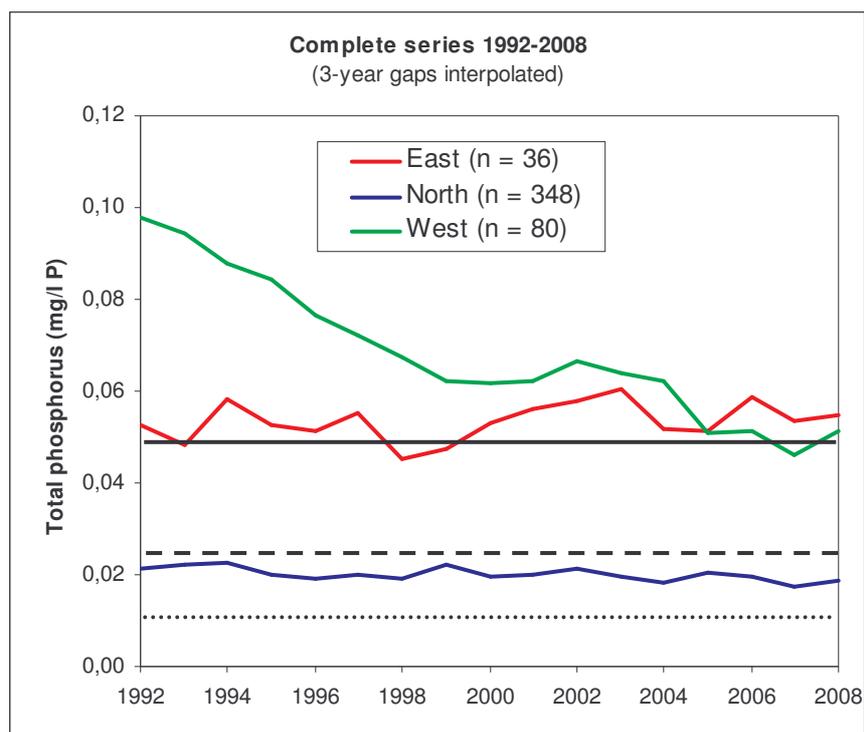
Figure 4.1 - Phosphorus concentrations in lakes (total phosphorus) between 1992 and 2008 in different geographical regions of Europe (the number of lake monitoring stations per region is given in parentheses). Black lines shows estimated mean WFD target for stratified non-humic lakes with low alkalinity (ca. 12 µg l^{-1}, dotted line), moderate alkalinity (ca. 25 µg l^{-1}, dashed line), and high alkalinity (ca. 50 µg l^{-1}, whole line), see text for further explanation..
Note: The data series per region are calculated as the average of the annual mean for river monitoring stations in the region. Only complete series after inter/extrapolation are included (see indicator specification). There were no stations with complete series after inter/extrapolation in the South and Southeast regions. The number of lake monitoring stations per country is given in parentheses:
        East: EE (8), HU (11), LT (4), LV (8), PL (1), SI (4),
        North: FI (183), SE (165),
        West: AT (5), CH (15), DE (7), DK (20), IE (8), NL (7), UK (18).
Data source: Waterbase – Lakes

Preliminary draft total P standards for lakes found in the WFD river basin management plans reported to the EU Commission in March 2010 are within the range 0.01-0.1 mg l^{-1} for most countries (see Chapter 3.2 for more info). An independent estimation of total P standards for lakes can be estimated from the WFD intercalibrated chlorophyll standards (Poikane 2009) and chlorophyll – total P regressions published for major lake types (Phillips et al. 2008).

These estimated total P standards indicate that a mean WFD target for stratified non-humic lakes with low alkalinity can be ca. 12 µg l^{-1} , for moderate alkalinity lakes ca. 25 µg l^{-1} , and for high alkalinity lakes ca. 50 µg l^{-1} .

The current total phosphorus concentrations reported for lakes are still considerably higher than the WFD target for stratified low and moderate alkalinity, non-humic lakes in the western and eastern regions, but close to the target for high alkalinity lakes, whereas lakes in the northern region are slightly below the target for moderate alkalinity stratified lakes, but still above the target for low alkalinity stratified lakes. The WFD target is higher for humic lakes and for non-stratified lakes in each of the alkalinity categories. Since the data does not specify lake type it is unclear what proportion of the lakes in the different regions belong to the different types. Thus, the distance to target indicated here must be interpreted with caution.

National scale trends (Map 4.2) reflect the regionalised data (Figure 4.1) and confirm that improvements have declined in recent years in most countries (lower proportion of blue colour in columns in Map 4.2 right panel compared to left panel in most countries).

Lake TP trends 1992-2008   Lake TP trends 2001-2008
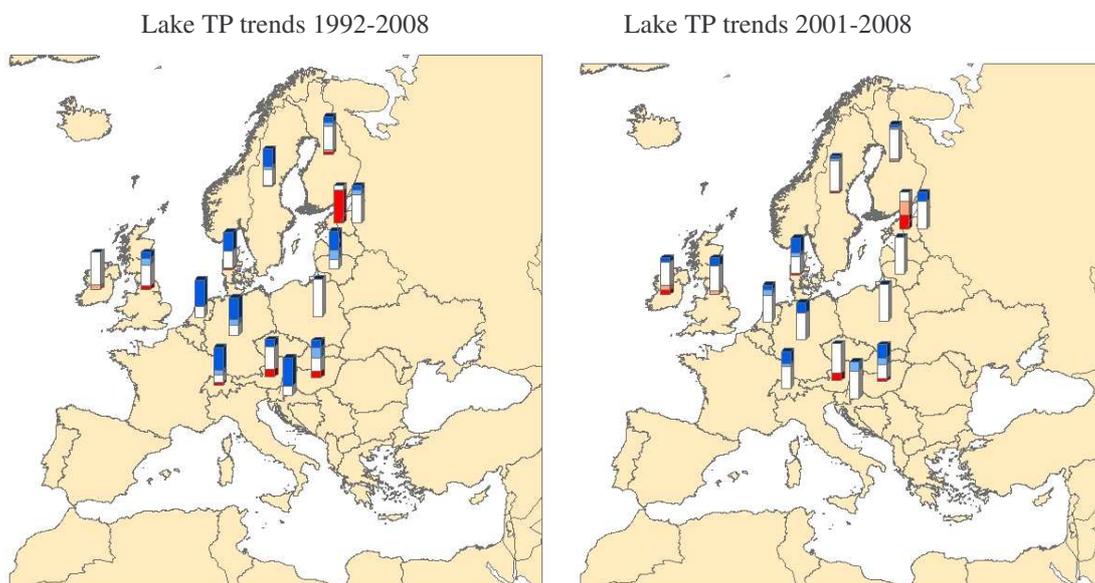
Map 4.2 - Total phosphorus trends in lakes in different European countries. Left panel: whole time period 1992-2008. Right panel: time period 2000-2008. Blue and light blue colours show strongly and slightly declining trends, white shows no trend, whereas red and pink show strongly and slightly rising trends.

A significant decline in total phosphorus concentrations has occurred since 1992 at 31% of the stations, while there has been a significant rising trend at 8% of the stations. Again, most of the decrease took place in the initial years. The Netherlands, Switzerland and Slovenia had the highest proportion of significantly decreasing lake total phosphorus trends.

The decline in total phosphorus in lakes primarily seen before the year 2000 are primarily due to measures implemented under the UWWTD, as well as the banning or substantial reduction of P in detergents. Agricultural P sources remain important, and further measures are required to ensure the achievement of good ecological status by 2015 as required by the WFD. Decrease in phosphorus levels in shallow lakes has been slow, partly because of internal P load from lake sediment stores. Such shallow lakes may require specific restoration measures.

5. Status and impacts in groundwater

5.1. Methods used for evaluation of status and impacts

Evaluation of the status of groundwater nitrate is based on compliance with groundwater quality standards over the period 2001-2008. Countries were grouped into regions according to Table 5.1 for the purpose of evaluation. The source of the data used for the evaluation was Waterbase ver. 10.

Southern, western and northern region represent mainly "old" EU members whereas eastern and southeastern regions represent "new" members and candidate countries.

Table 5.1 - Countries and regions used for Groundwater analyses.

Region	Country code	Region	Country code	Region	Country code
northern	FI	southeastern	AL	western	AT
northern	IS	southeastern	BA	western	BE
northern	NO	southeastern	BG	western	DE
northern	SE	southeastern	RO	western	DK
eastern	CZ	southeastern	RS	western	FR
eastern	EE	southeastern	TR	western	GB
eastern	HU	southern	CY	western	CH
eastern	LT	southern	ES	western	IE
eastern	LV	southern	GR	western	LI
eastern	PL	southern	IT	western	LU
eastern	SI	southern	ME	western	NL
eastern	SK	southern	MT		
		southern	PT		

5.2. Reference conditions of nutrients

Only nitrogen is considered to be a nutrient of concern in groundwater and whilst it can be found in three forms (nitrate, nitrite and ammonia) nitrate is typically dominant and predominantly sourced from agricultural fertilisers and manures. Natural background values of nitrate in groundwater are usually less than 10 mg l^{-1} and found in areas with little or no agriculture. However, concentrations up to 100 times background can be found in groundwater underlying intensive agriculture.

5.3. Environmental objective for nitrates

Environmental objectives for nitrates are set by:

1. Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources (Nitrate Directive) having objective of reducing surface and ground water pollution caused or induced by nitrates from agricultural sources and preventing further such pollution.
2. Directive 2000/60/EC establishing a framework for Community action in the field of water policy (Water Framework Directive) having with regard to groundwater purpose of protection of groundwater preventing its further deterioration by limiting of input of pollutants into groundwater.
3. Directive 2006/118/EC on the protection of groundwater against pollution and deterioration (Groundwater Directive) that establishes specific measures as provided for Directive 2000/60/EC in order to prevent and control groundwater pollution. This Directive also complements the provisions preventing or limiting inputs of pollutants into groundwater already contained in Directive 2000/60/EC and aims to prevent the deterioration of the status of all bodies of groundwater. The Groundwater directive sets value of 50 mg l^{-1} as groundwater quality standard for nitrates (same value as Directive 98/83/EC on the quality of water intended for human consumption (Drinking Water Directive).

Since observed concentrations of nitrates very often exceed the groundwater quality standard, programs of measures have been implemented under Water Framework Directive including "good agricultural practices" approach in order to prevent further contamination. Groundwater system dynamics are, however, typically very slow, thus the response of groundwater to measures is also very slow.

5.4. Observed status and trends of nitrate concentrations

Groundwater nitrate concentrations primarily reflect the relative proportion and intensity of agricultural activity. Although there were no countries where the average nitrate concentrations exceeded the threshold value of 50 mg l^{-1} NO_3 in 2008, 13 out of 27 countries had groundwater bodies (GWBs) with average concentration above the threshold (Map 5.1). Spain, Belgium and Denmark had the highest proportion of GWBs with average concentration above the threshold, but there was also a significant number of GWBs above the threshold in Germany, Bulgaria, Romania and Portugal. Groundwater nitrate concentrations were generally low (most GWBs < 10 mg l^{-1} NO_3) in Norway, Sweden, Finland, Estonia, Latvia, Bosnia and Herzegovina and Serbia.
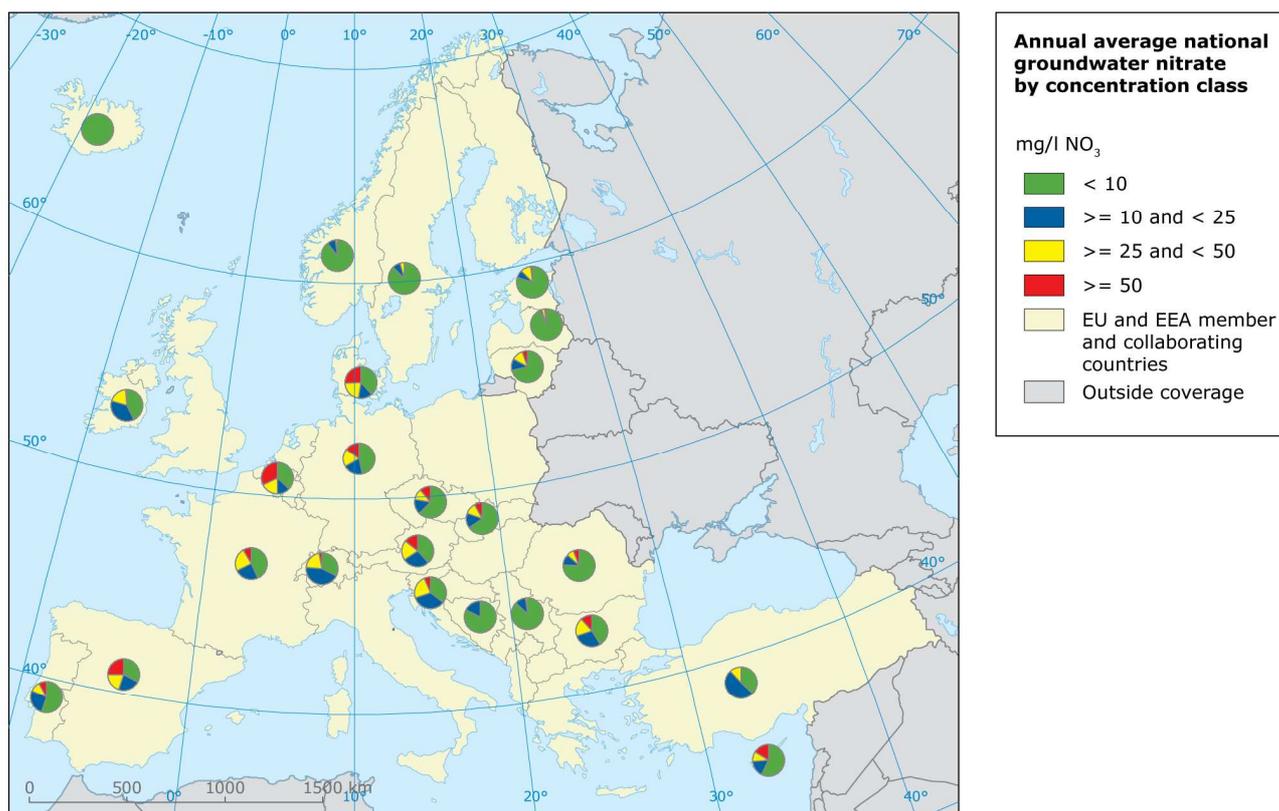
Map 5.1 - Annual average national groundwater nitrate (mg l^{-1} NO₃) by concentration class (2008). *Source; EEA/ETC-Water based on data reported to Eionet., based on WISE map viewer and CSI020. See "Map explanation" in WISE viewer for more information. Concentration units are mg l^{-1} NO₃. Data from the following countries are included (number of groundwater bodies): AT (28), BA (11), BE (49), BG (69), CH (27), CY (19), CZ (216), DE (348), DK (44), EE (5), ES (261), FI (38), FR (791), IE (136), IS (2), LI (1), LT (6), LU (4), LV (14), NO (1), PT (65), RO (24), RS (10), SE (21), SI (10), SK (81), TR (8).*

There is marked variation in groundwater nitrate concentrations between different geographical regions of Europe, with high concentrations in Western Europe and very low concentrations in Northern Europe (Figure 5.1). Overall nitrate concentrations in Western, Northern and Eastern Europe have remained relatively stable since 1992, although there is marked variation between different groundwater bodies (GWBs) (West: 26% negative, 29% positive; North: 11% negative, 9% positive; East: 26% negative, 37% positive; sum of significant and marginally significant trends). In the region of southern Europe there has been a marked decrease (3 out of 4 stations significant decrease), but it should be noted that this only reflects the development in Portugal. Likewise, southeastern Europe is only represented by Bulgaria. Here there was a marked increase until 1998, but since then the concentrations have stabilised (overall 10% negative trends and 17% positive trends, significant and marginally significant trends put together).

The lack of a general decrease is due to continued high emissions from agriculture.

41

Complete series 1992-2008

(3-year gaps interpolated)
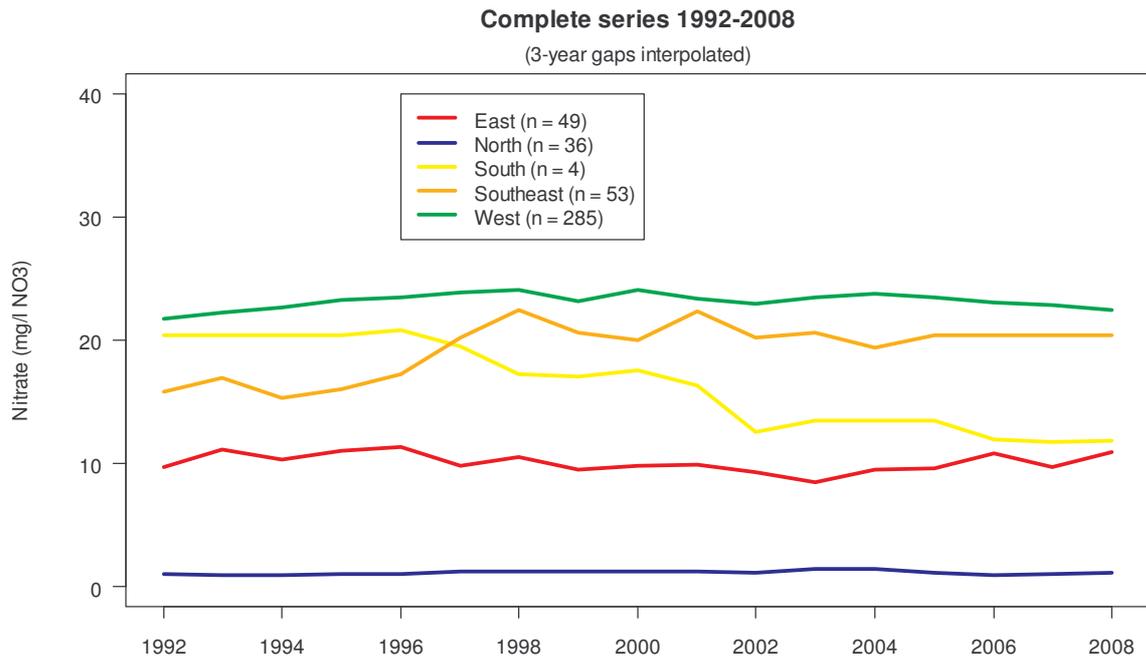
*Figure 5.1 - **Average annual concentration within each region.** See Table 5.1 for information on countries within the different geographical regions.*

Groundwater bodies nitrate trends 1992-2008   Groundwater bodies nitrate trends 2001-2008
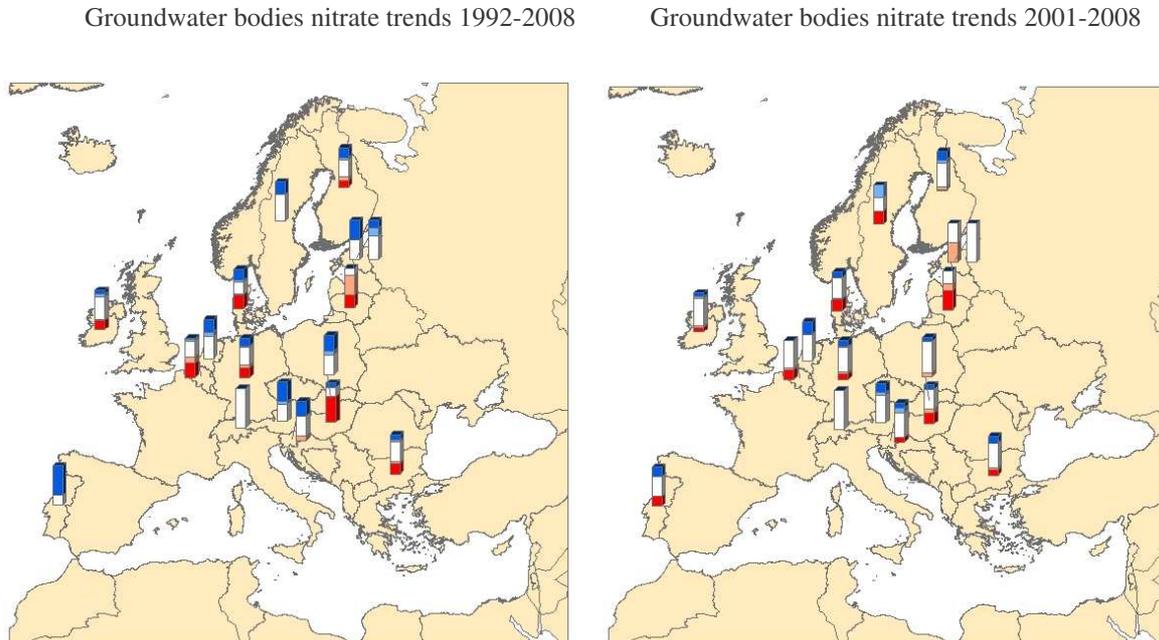
*Map 5.2 - **Nitrate trends in Groundwater bodies in different European countries.** Left panel: whole time period 1992-2008. Right panel: time period 2000-2008. Blue and light blue colours show strongly and slightly declining trends, white shows no trend, whereas red and pink show strongly and slightly rising trends.*

However, looking at individual GWBs there is wide variation in trends (Map 5.2), with 21% of the GWBs showing significantly decreasing nitrate concentrations since 1992 (an additional 5% showed a marginally significant decrease), while 24% of the GWBs showed significantly increasing concentrations (an additional 5% marginally significant). The countries with the highest proportions of GWBs with significant declining trends are Austria, Latvia and Portugal. The proportion of GWBs with declining trends have decreased since 2000 (Map 5.2 right panel).

5.5. Evaluation of level of exceeding the objective for nitrates

A comparison of proportion of reported groundwater bodies with respect to exceedance of the groundwater quality standard shows that the highest proportion of groundwater bodies exceeding the objective is found in the southern region, with 14-27% exceeding the objective within the 2003-2008 period (for which more data is available). For Europe as a whole, between 7 and 11 % of reported groundwater bodies exceed the groundwater quality standard within the 1992-2008 period (Figure 5.2).
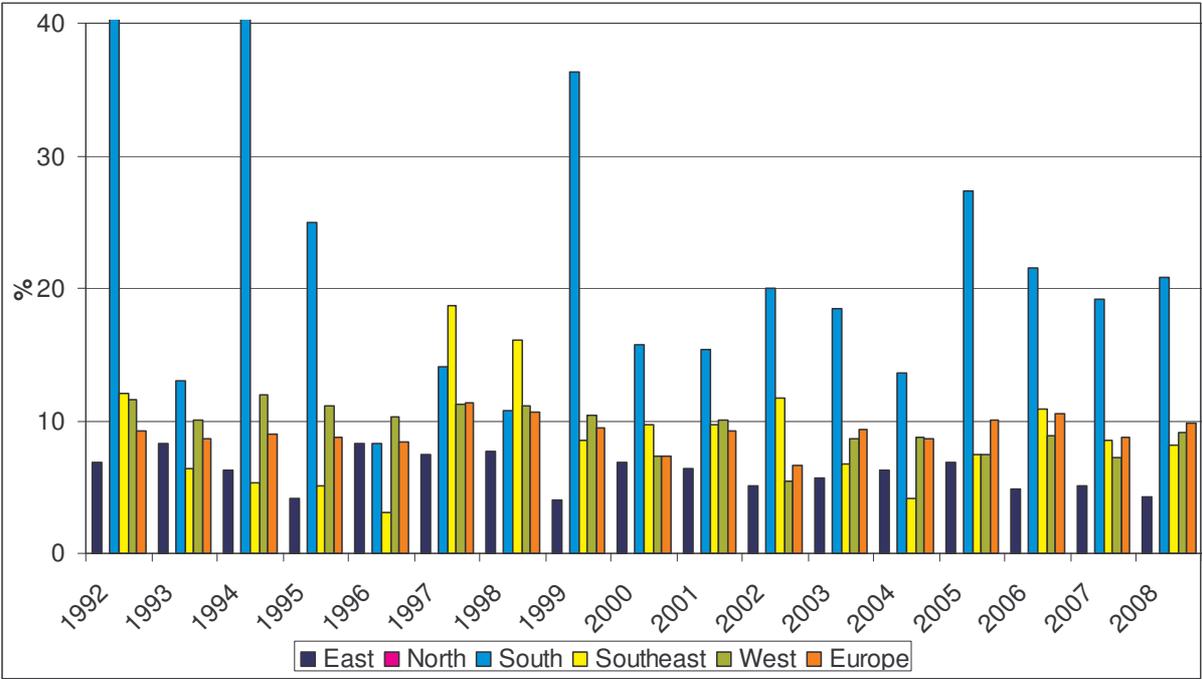
Figure 5.2 - Proportion of groundwater bodies exceeding groundwater standard value for NO₃.(50 $mg\ l^{-1}$*). The figure is based on all available data, i.e. the total number of groundwater bodies varies between years (generally increasing). See Table 5.1 for information on countries within the different geographical regions. Note: The y-axis is cut at 40%, but the bars for region South in 1992 and 1994 reach 100% (data from one groundwater body only in both years).*

The highest number of groundwater bodies (1428) exceeding the groundwater quality standard was reported for the western region in 2008, see Figure 5.3.
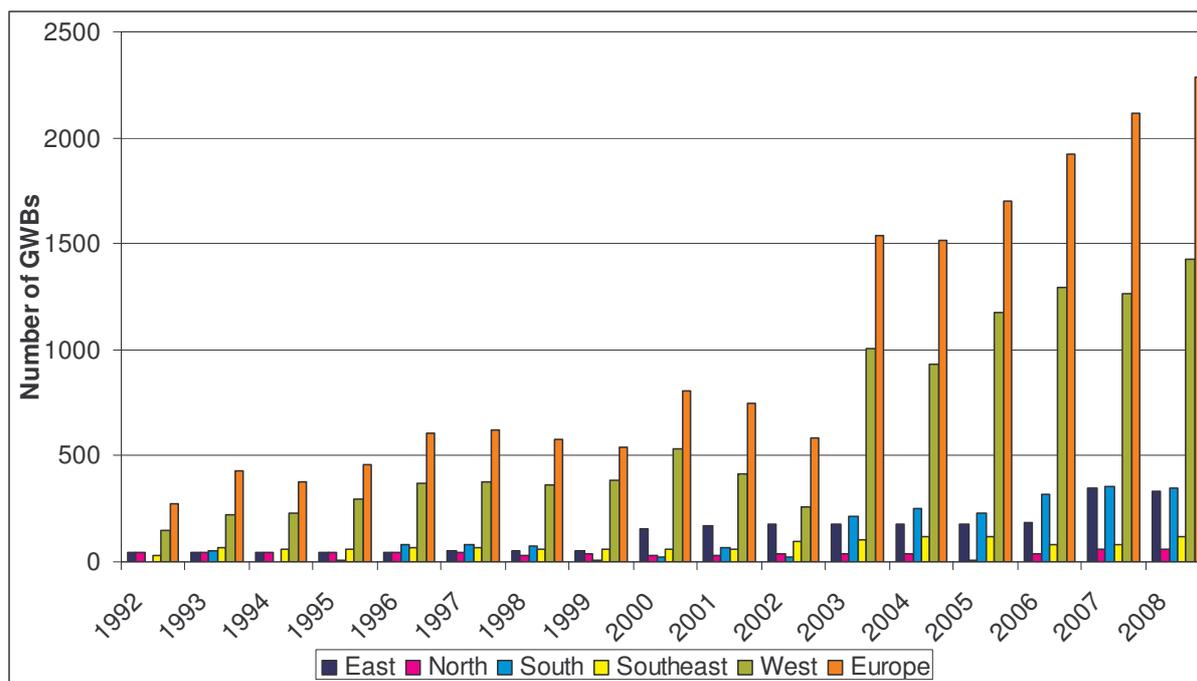
Figure 5.3 - Number of stations with concentrations exceeding groundwater standard value (50 mg l^1). The figure is based on all available data, i.e. the total number of groundwater bodies varies between years (generally increasing). See Table 5.1 for information on countries within the different geographical regions.

6. Impacts of eutrophication on freshwater ecology, biodiversity and human health

6.1. Overall description of eutrophication symptoms in freshwater.

Eutrophication causes increased biomass and altered species composition of all biological quality elements in rivers. The component most sensitive to eutrophication is benthic algae, whereas benthic fauna is sensitive to oxygen depletion caused by increased loads of oxygen consuming substances (organic matter and ammonium) from point sources along the river. In lakes the most prominent symptom of eutrophication is algal blooms, reduced water transparency, oxygen depletion of bottom waters (deep lakes) and sometimes even fish kills during winter in ice-covered lakes. In shallow lakes sensitive aquatic plants are reduced due to shading caused by high phytoplankton biomass. The fish community is also changed to a higher proportion of small cyprinids and lower proportion of perchids and salmonids.

6.2. WFD good status objective for diatoms and benthic fauna in rivers

The intercalibrated boundaries for good ecological status, sensu WFD, have been set for benthic diatoms and benthic fauna (van de Bund, 2009). These boundaries give the WFD target (e.g. the good/moderate class boundary) for the following quality elements:

- macroinvertebrates in rivers for all regions, given as ecological quality ratios (EQRs) of different indices.
- benthic algae in rivers for all regions, given as ecological quality ratios (EQRs) of mainly diatom indices.

The normalised WFD target for all quality elements is the EQR ratio of 0.6.

6.3. Impacts of eutrophication on ecological status and biodiversity in rivers

There is currently no overall European dataset available to show a representative picture of the ecological status of eutrophication sensitive biota of European rivers. A test data exercise for EIONET was done in 2009 for benthic fauna, but the data submitted are not acceptable for publication due to many shortcomings related to station selection and to the use of non-intercalibrated metrics for some countries. Therefore, the following section presents three case studies that are useful examples of ecological impacts of eutrophication in European rivers.

6.3.1. Case study: The Rhine

Recent information from the Rhine commission provides the following general picture of the ecological status of benthic diatoms and benthic fauna:

For benthic diatoms the varying species composition and frequency indicate a distinctly degrading ecological state from upstream to downstream. This correlates well with the trophic status and saprobity that are low in the High Rhine and increase further downstream. The analysed locations on the High Rhine are of very good ecological quality. While the sections of the Upper Rhine are largely assessed to be "good", the middle and lower part of the Rhine is largely characterized to be "moderate", although good status sites can be found also in this part of the river (Figure 6.1).
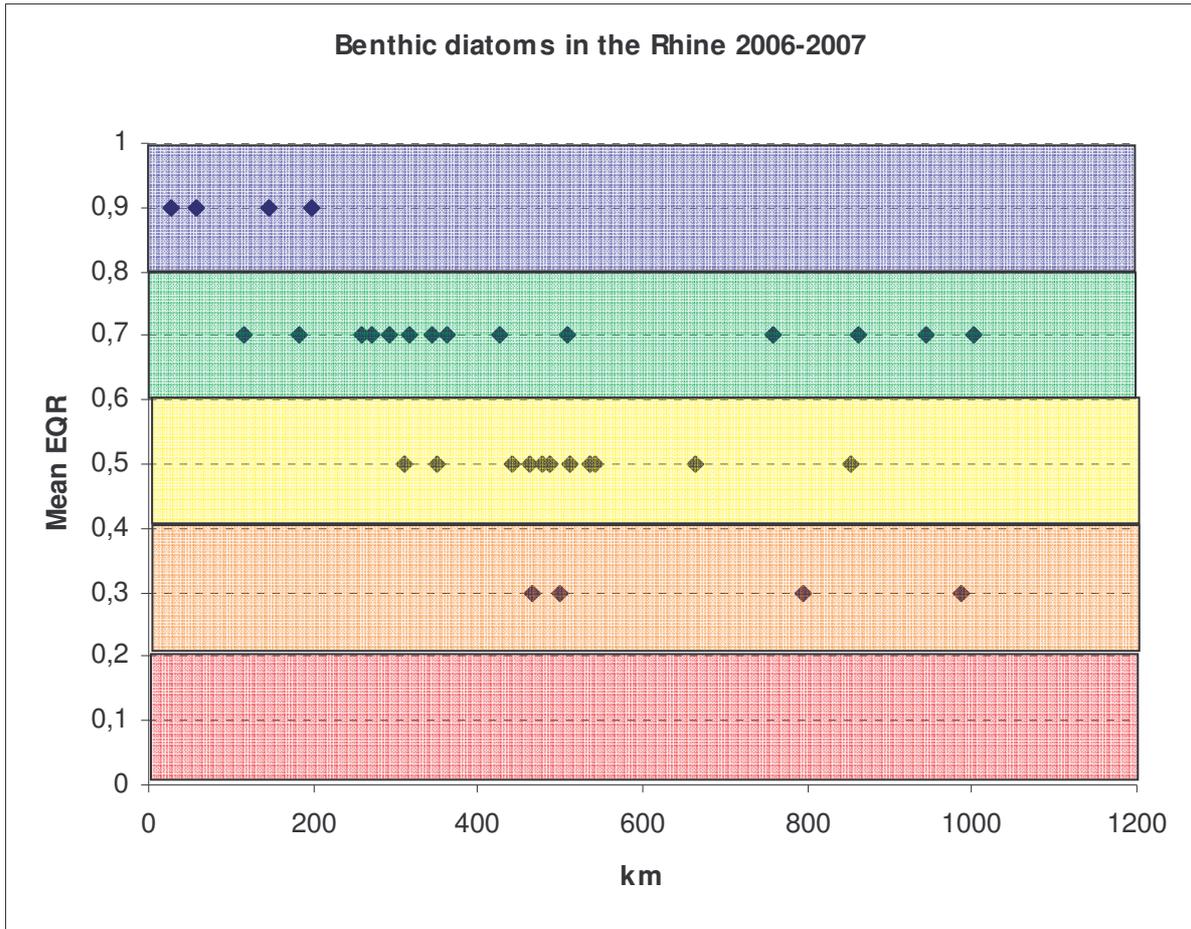
Figure 6.1 - Ecological status of benthic diatoms given as mean EQRs in the Rhine from the source to the mouth in 2006-2007*. The colours indicate high (blue), good (green), moderate (yellow), poor (orange) and bad (red) ecological status classes. The mean EQRs are estimated from classes reported in Rhine Commission report on benthic diatoms 2006-2007.*

For benthic fauna (Figure 6.2) the total number of species was low from the 1955-1980 due to low oxygen levels. From 1980-1995 the bioversity increased in parallel with the oxygen concentration and has been comparably constant during the last 15 years. This illustrates the success of wastewater treatment in the Rhine catchment. However, since 1995, the average number of species per sampling location is regressing. Presumably, the invasive species as a factor of biological stress are partly responsible. In addition, the absence of suitable habitats in the river itself prevents the return and spreading of a benthos fauna typical for the Rhine (Rhine commission report 2006-2007).
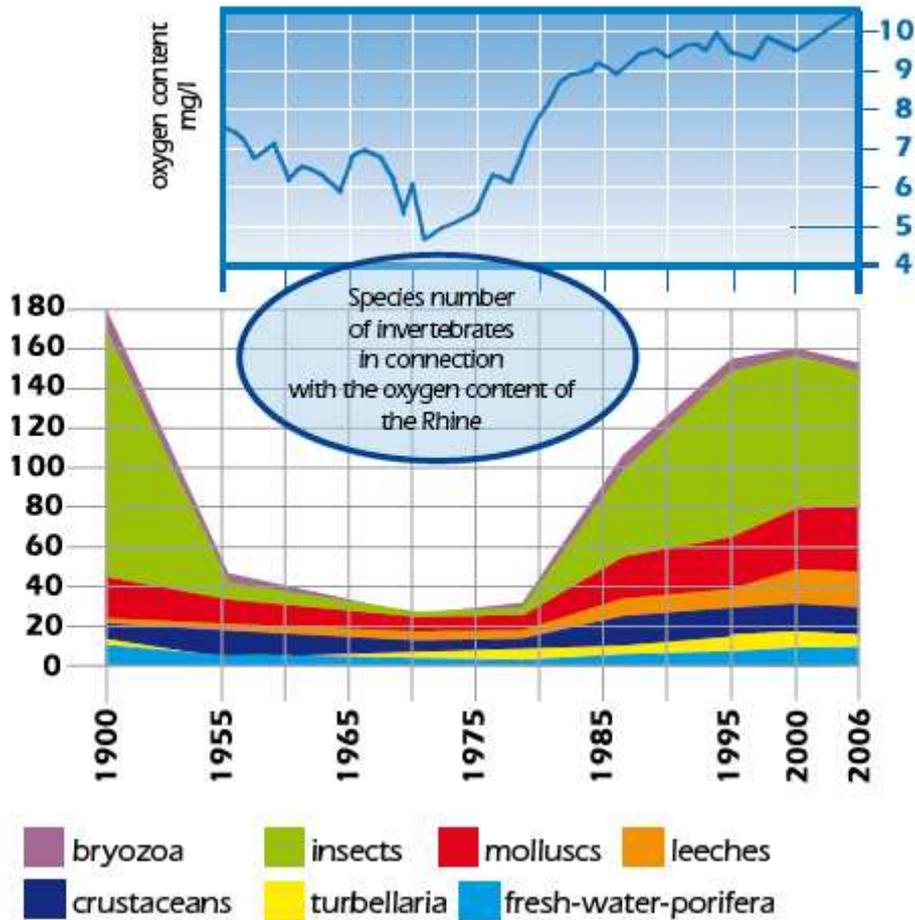
Figure 6.2 - Timeseries of oxygen concentrations and number of invertebrate species in the Rhine (Rhine commission report, 2006).

6.3.2. Case study: Scottish rivers

The monitoring results underlying the draft river basin management plans for Scotland are given in Table 6.1 (based on data from SEPA 2009). The number and percentage of river sites classified in different WFD ecological status classes are shown and indicate that a larger pro-portion of rivers fail the good status objective for the biological quality elements than for the chemical nutrient related quality elements, although there are clear correlations between the chemical parameters and the biological quality elements. For benthic algae, the proportion of sites failing the good status objective is > 30%, whereas for phosphorus, the proportion of sites failing this objective is merely 7 %. Similarly for benthic fauna 7% fail the objective, in contrast with 2% when using only the chemical parameters. The reason why the ecological status based on biology is worse than that based on chemistry alone can be that also other pressures affect the biology, e.g. pesticides or metals, in-appropriate habitat or a recent flood event, or that the nutrient standards are not well correlated with the biological good/moderate class boundaries. Benthic algae are generally known to be more sensitive to nutrient pressure than benthic fauna. If nutrient pressure is the most important pressure in these rivers, this higher sensitivity may also contribute to explain why a larger percentage of water bodies fail the objective based on benthic algae than on benthic fauna.

Table 6.1 - *Numbers (a) and percentage (b) of Scottish river sites classified in different ecological status classes for different water quality indicators (data from table B1, SEPA 2009).*

a)

Table B1: Results for indicators of water quality conditions in rivers in the Scotland RBD in 2008

Measured condition of indicator	phosphorus	bottom living algae	flowering plants and mosses	dissolved oxygen	ammonium	bottom-living insects and other invertebrates	acidity as pH	acid-sensitive bottom-living insects and other invertebrates
High	1,338	449	84	1,560	1,554	1,142	1,438	204
Good	202	181	69	38	68	357	190	81
Moderate	106	273	29	27	23	79	7	340
Poor	13	33	2	5	9	28	6	21
Bad	0	0	0	3	6	8	0	0

b)

class	TP	Benthic algae	O2	NH4	Benthic fauna
high	81 %	48 %	95 %	94 %	71 %
good	12 %	19 %	2 %	4 %	22 %
moderate	6 %	29 %	2 %	1 %	5 %
poor	1 %	4 %	0 %	1 %	2 %
bad	0 %	0 %	0 %	0 %	0 %

6.3.3. Case study: The Danube

In the summer of 2007 the 2nd Joint Danube survey was carried out. Selected results from this survey are shown below (Makovinska et al., 2008; Graf et al. 2008; ICPDR).

Benthic algae (based on Makovinska et al., 2008)
Cyanobacteria and green algae prevailed (from the point of view of relative abundance) at most of the sampling stations (Figure 6.3). However, at eight stations diatoms comprised the most abundant group. Phytobenthos biomass increased downstream, when comparing the average biomass of the Upper, Middle and Lower Danube (Figure 6.4).

A separate assessment was also done for benthic diatoms. In total 391 diatom taxa were found at 135 sites. A cluster analysis distinctly separated benthic diatom communities from the Upper Danube and the beginning of the Middle Danube (from Germany, the station upstream of Iller, to Slovakia, Bratislava station) from samples coming from the Middle and Lower Da-

nube downstream of Bratislava. The sites were mostly dominated by eutrophic to hypertrophic species indicating beta mesosaprobic to polysaprobic conditions.
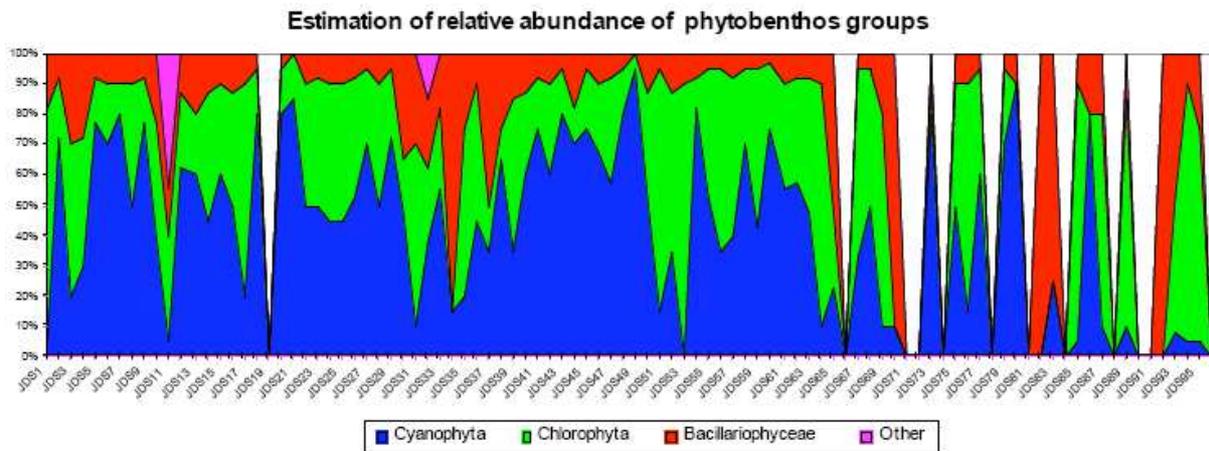
Figure 6.3 - Relative abundance of individual phytobenthos groups of cyanobacteria (Cyanophyta), diatoms (Bacillariophycea), green algae (Chlorophyta) and 'others' (including Rhodophytes - JDS11 and bacteria - JDS32) in living samples at each sampling site (Makovinska et al,. 2008).
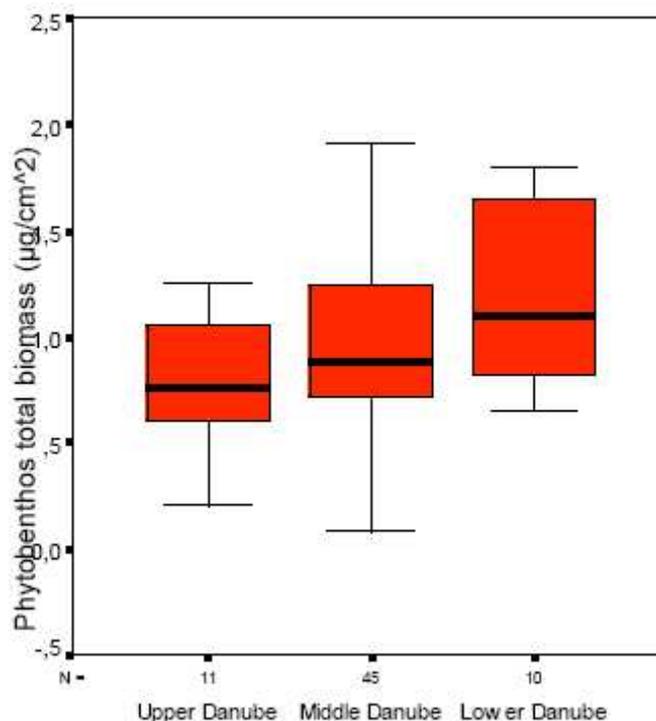
Figure 6.4 - Phytobenthos biomass in the Upper, Middle and Lower Danube (Makovinska et al. 2008)

In order to indicate ecological status (according to the WFD) based on results from the JDS2 sampling sites, the Slovak classification system for phytobenthos-based assessment was used (Figure 6.5). This system (Hlubiková et al., 2007) is based on a multimetric index comprising three separate diatom indices: IPS, CEE and EPI-D. For Ecological Quality Ratio (EQR) calculation, average values of three diatom indices were used. Reference values and class boundaries for this river type were derived by modelling. Further details are given in Makovinska et al., 2008.

The benthic diatoms indicate that > 70% of the stations in the middle and lower part of the Danube fail the WFD good status objective due to eutrophication, whereas less than 50% fails the objective in the upper parts.
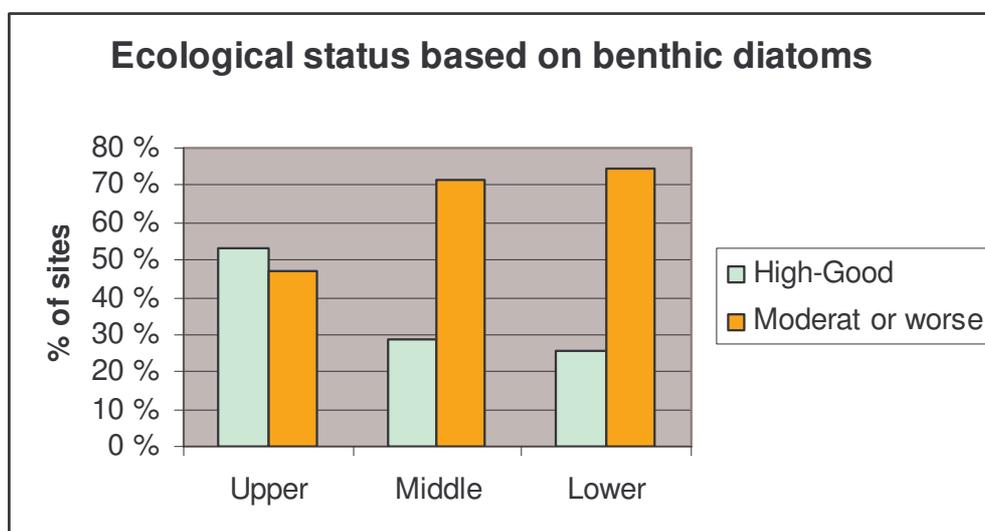
Figure 6.5 - *Assessment of the status of the Danube and the tributaries based on the Slovak classification system according to phytobenthos (benthic diatoms). The bars represent the upper, middle and lower parts of the Danube. Based on data reported by Makovinska et al., 2008.*

The results demonstrate that eutrophication is still a major problem in the Danube river.

Benthic fauna (based on Graf et al. 2008)
The benthic fauna diversity was significantly lower in the lower reach of the river, decreasing from 230-250 taxa in the upper and middle reaches to 170 in the lower reach (Figure 6.6 and 6.7). In the whole river the highest number of taxa was found in the groups Diptera and Oligochaeta, groups that normally indicate rather poor ecological quality.

Regarding abundance (ind./m2), Amphipoda are the dominant group in all Danube reaches and constitute up to 75 % while Isopoda (mainly *Iaera istri*) play an essential part in the Upper Reach and decrease downstream. Oligochaeta and Mollusca can be found in increasing numbers in the Lower Reach. EPT- Taxa (Ephemeroptera, Plecoptera and Trichoptera) were negligible – with the exception of the Upper Reach (sites 1 and 2). With regards to aquatic insects, only Chironomidae play a major role.

In terms of biomass, Mollusca are the most important organisms of the Danube and investigated tributaries. Due to their size Bivalvia make up more than 80% of the whole biomass, followed by Gastropoda (10% to 35%). Looking at the different reaches of the Danube, the increasing dominance of Mollusca from the Upper to the Lower Reach becomes evident. Although Crustacea are the most abundant group, they play only a minor role regarding biomass.

Figure 6.6 - *Kick and sweep sampling in the Danube during the 2nd Joint Danube survey 2007. Photo: Bela Csanyi, Vituki, Hungary*
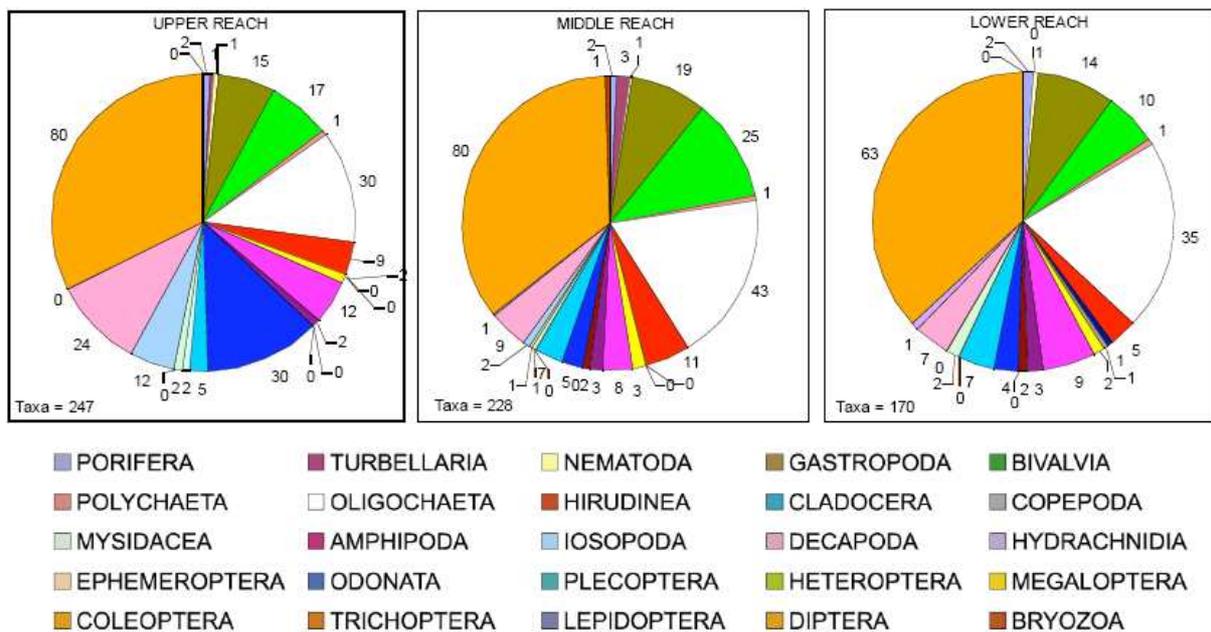
Figure 6.7 - *Number of taxa per taxa group along the different reaches of the Danube (Graf et al., 2008).*

The impact of eutrophication / organic pollution on the ecological status of the benthic fauna was measured with the saprobic index based on the deviation of the saprobic index from saprobic reference conditions (Stubauer & Moog, 2003).

The highest values of saprobic index, indicating serious organic pollution, were detected in the Danube downstream of Pancevo and at Giurgeni. Most sites (58 sites) were classified to be in good status, 9 sites even had high status. Moderate and poor status were found in 8 and

3 sites respectively. So in total only 11 of 78 sites (14%) are failing the good status objective based on this index. However, most of the fauna consisted of invasive species (Neozoa), such as the mussel, *Corbicula fluminea* occurring in 93% of the sites, followed by the crustaceans, *Corophium curvispinum* (90%) and *Dikerogammarus villosus* (69%). As Neozoa dominate the fauna, their classification is a crucial factor in assessing ecological status. Most of them indicate ß-mesosaprobic water quality due to their national classification, which results in an overall good ecological status due to their dominance. Omitting Neozoa from the analysis leads to zero-values for the saprobic index (SI) in some cases.

6.3.4. Case study: The Po river, Italy

Introduction
The longest river in Italy, the river Po, flows eastward across northern Italy from the Cottian Alps to the Adriatic Sea near Venice. The 652 km long river has a 74 000 km^2 drainage area of which 41 000 km^2 is mountainous and the remaining 29 000 km^2 is located in the Po valley encompassing the lowland plain of rich soil. There are 450 lakes in the drainage basin, and 141 tributaries add to the river that discharges an average 1540 m³/s into the Adriatic Sea through a wide delta. The amount of water discharged by Po is approx. 50% of the total freshwater input to the northern Adriatic Sea.

On the North side the flux of water is regulated by five large lakes. These lakes are directly connected to the main tributaries of the Po River, maintaining a continuous interchange of ground water and surface water between the lakes and rivers. The regulation itself influences the discharge pattern of the river and the retention of particulate material in these lakes influence the sedimentation of material on the plain. The landscape has changed due to increased urbanisation and intensified agriculture production. A large proportion of the natural vegetation (near 25%) along its banks has been replaced with plantations of poplars harvested for cellulose.

Due to an uneven precipitation pattern, the river is subject to periodic heavy flooding which is intensified by the fast runoff from the increasing urban areas. More than half the river length is controlled with a system of dikes to prevent flood damage.

Driving forces and Pressures
The Po River basin is a strategic region for the Italian economy, with significant agriculture, industry and tourism sectors. The lakes in the northern part are important for tourism but are affected by eutrophication.
The river passes a number of larger Italian cities and Turin is the industrial center of the region. Milan is indirectly connected to the river by a channel system.
More than 40% of the Italian workforce is employed in this region and produces nearly 40% of the national GDP. 55% of Italian livestock is found here and 35% of the agriculture production takes place in this region.
The principal farming areas are localised in the Po valley, covering 45% of the basin's total area. Most of the agricultural land in the Po valley is arable land, drained by artificial ditches, and 50% of agricultural land is irrigated during summer. The major crops that are grown are wheat, maize, fodder, barley, sugar beets and rice – with the latter being especially water demanding.

The United Nations World Water Development Report 3 (World Water Assessment Programme, 2009) summarises the water challenges in the Po River basin, concluding that the high level of regional development has put heavy pressure on water resources and led to degradation of surface and groundwater quality.

Averaged year flow rate is presently lower than water use permits, both for surface waters and for groundwater. Irrigation is by far the largest consumer of water, four times the consumption of industrial water and public water supply combined.

State

Surface and groundwater quality is affected by discharges and losses of pollutants from industrial, agricultural and household. Excessive nutrients and organics in surface water causes eutrophication in rivers and in lakes. Although a network of wastewater treatment facilities has stopped further degradation of water quality, it has not been sufficient to reverse the process. Groundwater resources continue to contain high concentrations of nitrates due to fertilizer use in agriculture, while excessive exploitation has caused salt intrusion into coastal aquifers and, in some places, ground subsidence.

More than half of the nitrogen and phosphorus loads in the river Po originate from diffuse sources – and the role of the intensive agriculture activities on the plain is obvious. The results of scenario analyses by the turn of the century indicated that the measures imposed by the EU Nitrates Directive and the EU Wastewater Treatment Directive may not be stringent enough to achieve a large reduction in the N and P loads in the river Po (de Wit M.1 and Bendoricchio G., 2001).

The nutrient enriched water has a negative impact on the ecological status of the main river and its tributaries (Figure 6.8), showing that the whole river basin, except a few minor tributaries, fails the WFD objective of good ecological status. Moreover, when the water discharges into the Adriatic Sea the high nutrient load leads to eutrophication of the coastal waters. Paleo analyses of marine sediments covering the period 1830–1990 have revealed that a progressive increase in eutrophication took place in the beginning in the 20th century - particularly marked between 1930 and 1978. After 1978, the situation has improved, but still not recovered (Sangiorgi and Donders, 2004).

Response

Although policy tools for managing and safeguarding water resources have been implemented at national level for some time, there have been long lasting problems with the implementation and enforcement of rules and regulations at a regional level.

A set of integrated water management plans as well as flood risk management plans are adopted in all regions of the Po river basin. In 2009 all these plans will be homogenised into the integrated river basin management plan at District level according to the WFD.
The Po Valley Project is a new integrated project that aims to integrate actions and measures for water quality, flood risk control, cultural and tourism requirements over a 6 year period from 2009.
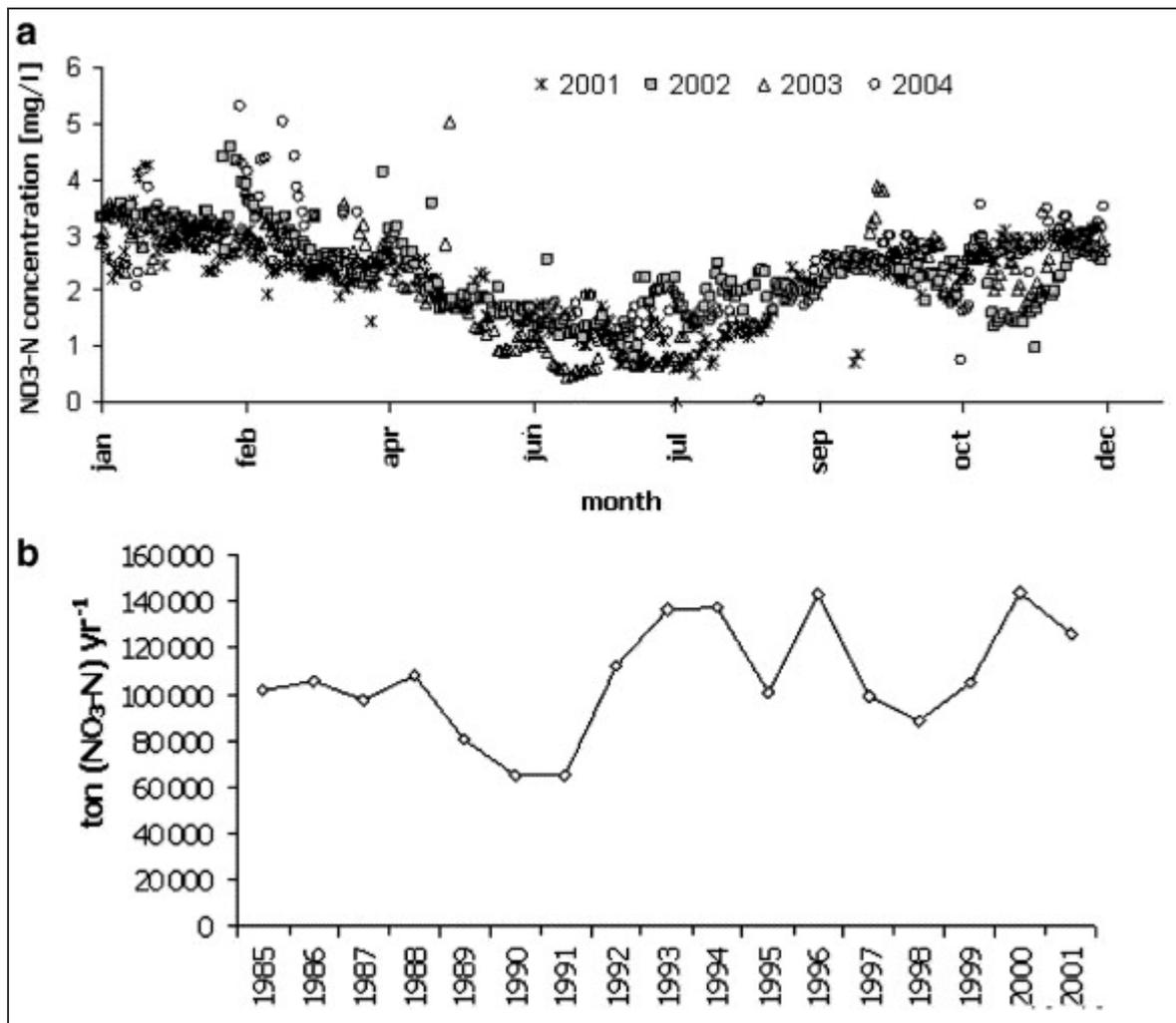
Figure 6.8 - Time serie (1985-2001) of nitrate concentration from the monitoring station at Pontela-goscuro (near the river mouth). Copied from Salvetti et al. (2006).
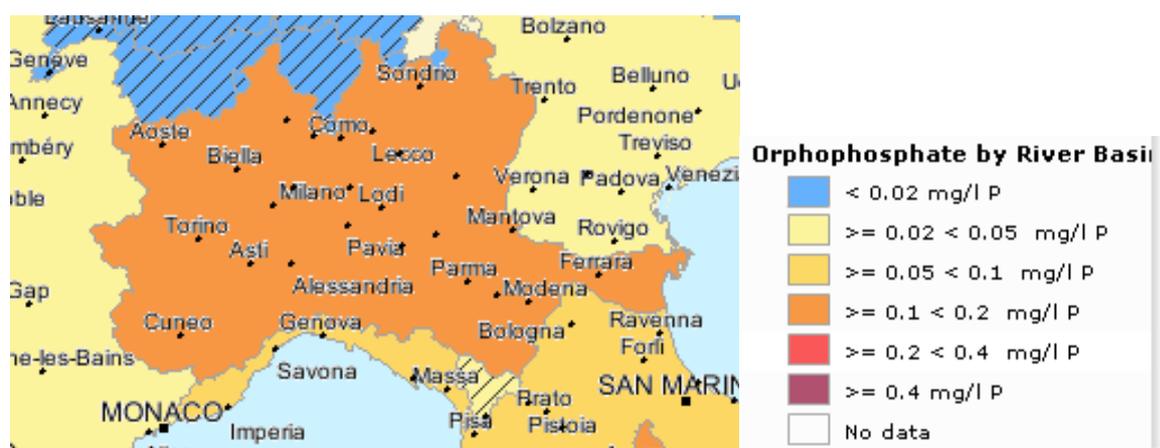
Figure 6.9 - Mean phosphate concentration for the Po river basin taken from WISE SoE.
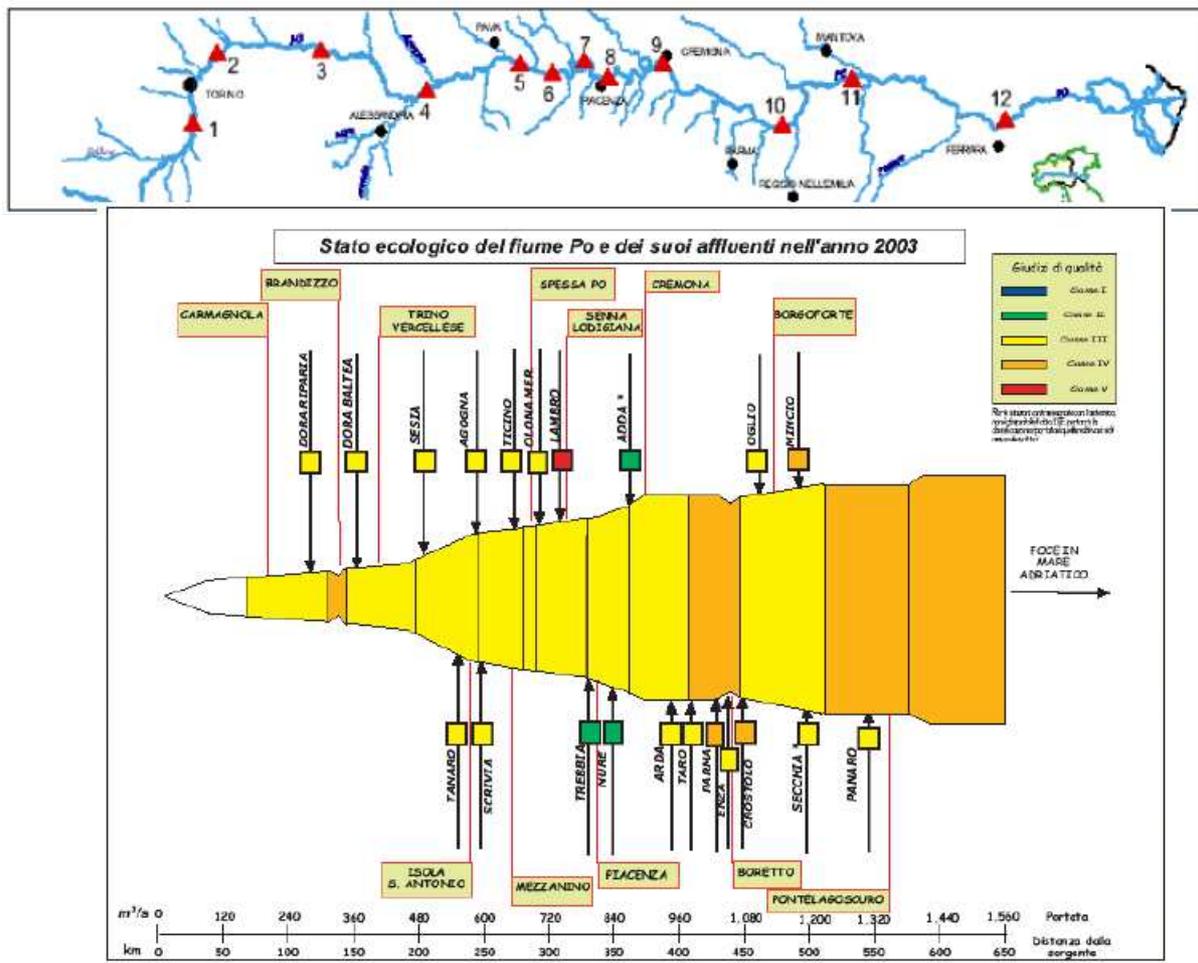
Figure 6.10 - *Ecological status of the Po river and its tributaries from the source (left side) to the mouth (right side) (Bortone, 2009).*

6.3.5. Conclusions and lessons learnt from the case studies in rivers

Based on these three case studies it is clear that eutrophication still has major impacts on river ecology in Europe, and that many river sites are far from the WFD good ecological status target for benthic flora and fauna. This is clearly seen in the middle and lower parts of the large rivers Rhine and Danube, as well as in the whole Po river basin in spite of implementation of large nutrient reduction measures. Even in Scotland ca. 30% of rivers fail the good status objective, although it is not clear whether this is mainly due to nutrient pressure or whether also other pressures contribute to this situation. Although improvements have occurred over the recent decades due to improved sewage treatment and thereby reduced nutrient load from point sources, the biology shows that further reduction of nutrients is still necessary, in particular concerning diffuse sources. The benthic flora seems to be the most sensitive quality element to eutrophication. For benthic fauna the status is also affected by other pressures such as invasive species and hydromorphological alterations.

6.4. WFD good status objective for biological quality elements in lakes

The intercalibrated boundaries for good ecological status, *sensu* WFD, have been set for a limited number of biological quality elements (Poikane, 2009). These boundaries give the WFD target (e.g. the good/moderate class boundary) for the following quality elements:

- Phytoplankton biomass in lakes, given as chlorophyll a for different lake types in all regions (Table 6.2) except the astern continental region (Hungary, Bulgaria and Romania), and species composition given as ecological quality ratios (EQRs) and the corresponding absolute value for different indices for the Alpine and Mediterranean regions only.
- Macrophytes species composition for northern, central and Alpine regions.

Table 6.2 - WFD targets for phytoplankton chlorophyll (Poikane, 2009).

GIG	Countries	Type	Type description	EQR		Chlf concentrations, µg/l		Mean target, µg/l
				H/G	G/M	H/G	G/M	G/M
Atlantic	IE, UK	L-A1/2	Lowland, shallow, high alk.	0.55	0.32	4.6-7.0	8.0-12.0	10.0
Alpine	AT, FR, IT ,SI, DE	L-AL3	Lowland, deep, high alk.	0.7	0.4	2.1-2.7	3.8-4.7	4.3
		L-AL4	Mid-altitude, shallow, high alk.	0.75	0.41	3.6-4.4	6.6-8.0	7.3
Central-Baltic	BE, DE, DK, EE, FR, LT, LV, NL, PL, UK	L-CB1	Lowland, shallow, high alk.	0.55	0.32	4.6-7.0	8.0-12.0	10.0
		L-CB2	Lowland, very shallow, high alk.	0.63	0.30	9.9-11.7	21.0-25.0	23.0
	BE, DK, EE, FR, LV, PL	L-CB3	Lowland, shallow, moderate alk.	0.57	0.31	4.3-6.5	8.0-12.0	10.0
Mediterranean	GR, FR, PT, ES, RO	L-M5/7	Deep reservoirs, low-moderate alk.	n.a.	0.21	n.a.	6.7-9.5	8.1
	CY, GR, FR, IT, ES, RO	L-M8	Deep reservoirs, high alk.	n.a.	0.43	n.a.	4.2-6.0	5.1
Northern	NO, SE, FI, UK, IE	L-N1	Lowland, shallow, mod. alk., clear	0.5	0.33	5.0-7.0	7.5-10.5	9.0
	NO, SE, FI, UK, IE	L-N2a	Lowland, shallow, low alk., clear	0.5	0.29	3.0-5.0	5.0-8.5	6.8
	NO, SE, UK	L-N2b	Lowland, deep, low alkalinity, clear	0.5	0.33	3.0-5.0	4.5-7.5	6
	NO, SE, FI, UK, IE	L-N3a	Lowland, shallow, low alk., meso-humic	0.5	0.30	5.0-7.0	8.0-12.0	10.0
	NO, SE, UK	L-N5	Mid-altitude, shallow, low alk., clear	0.5	0.33	2.0-4.0	3.0-6.0	4.5
	NO, SE, UK	L-N6a	Mid-altitude, shallow, low alk. mesohumic	0.5	0.33	4.0-6.0	6.0-9.0	7.5
	NO, SE, FI, UK, IE	L-N8a	Lowland, shallow, mod. alk., mesohumic	0.5	0.33	7.0-10.0	10.5-15.0	12.8

In Table 6.2, the WFD targets of phytoplankton chlorophyll are given for the common intercalibration types to allow the assessment of distance to WFD-target for lakes in different regions of Europe. The following general groups of lake types and WFD targets can be extracted from Table 6.2:
- low-moderate alkalinity and/or deep and/or mid-altitude lake types: 4-8 µg l^{-1},
- for moderate-high alkalinity and/or humic, shallow, lowland lake types: 8-12 µg l^{-1},
- for high alkalinity very shallow lowland lakes: 23 µg l^{-1}.

6.5. Impacts of eutrophication on ecological status and biodiversity in lakes

The only overall European dataset available to show a representative picture of the ecological status of eutrophication sensitive biota of European lakes is the EIONET data for chlorophyll. A test data exercise for EIONET was done in 2009 to get more data on phytoplankton and aquatic macrophytes, but the data submitted are not acceptable for publication due to many shortcomings related to station selection and to the use of non-intercalibrated metrics for some countries. Therefore, the following section presents the main time-trends of chlorophyll a for major European regions (country groupings as in Chapter 3 on nutrient concentrations), as well as the major state-of-the-art knowledge on occurrence of bluegreen algal blooms in Europe.

6.5.1. *Phytoplankton biomass trends in European lakes*

Data on algal biomass (estimated by seasonally averaged chlorophyll a concentrations) have been submitted by most European countries to EEA's Waterbase (CSI-020) for a number of years. However, the coverage is still uneven both in space and time. While data are available back to 1970 and earlier for a few countries, the records from others start after 2005, and in some of them data are yet available for only one year.

A summary of annual levels of algal biomass aggregated over geographic regions is shown in Figure 6.11. Data from 2007 were not included, as they were too sparse to be comparable with previous years. For the south and the south-east regions, the number of national datasets varied considerably between years (which explains erratic curves), and thus temporal trends are not reliable. **Reliable long-term trends are evident for the eastern and western region**, both of which show **a decline in algal biomass up to about 2000, but little change since then**. For the **northern region, no temporal trend is evident over the time interval shown**. In general, lakes of the northern region harbour lower algal biomass than the eastern and western regions, although the difference has been reduced since the early 1990s.
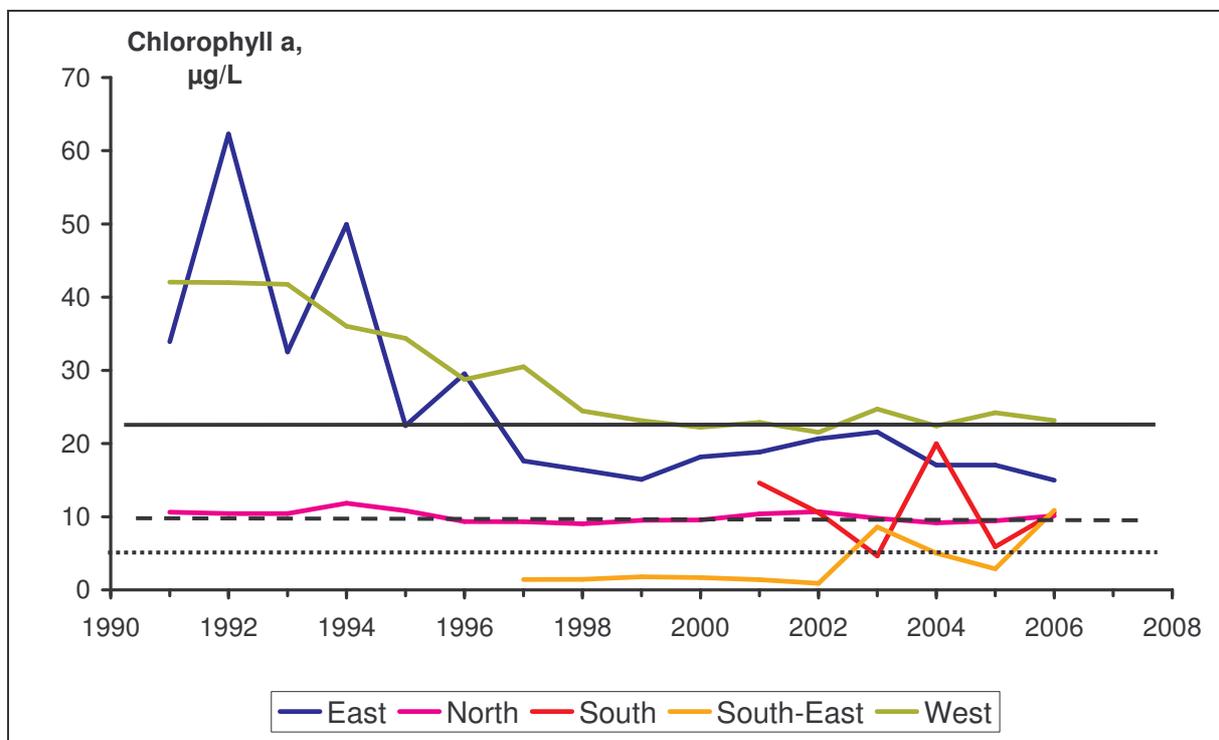
Figure 6.11 - Development of algal biomass (chlorophyll a concentrations) in European lakes aggregated to geographic regions for the period 1991-2006. Countries included in the regions are: East (EE, HU, LT, LV, PL, SI, SK); West (AT, BE, CH, DE, DK, FR, GB, IE, NL); North (FI, IS, SE); South (CY, IT, PT); South-East (BA, BG, HR, MK, RS, TR). Black lines shows mean WFD target for low-moderate alkalinity and/or deep and/or mid-altitude lake types (ca. 6 $\mu g\ l^{-1}$, dotted line), for moderate-high alkalinity and/or humic, shallow, lowland lake types (ca. 10 $\mu g\ l^{-1}$, dashed line), and for high alkalinity very shallow lowland lakes (23 $\mu g\ l^{-1}$, whole line).

Countries within regions vary considerably in the biomass levels reported. For instance, in the western region, DK and NL reported higher levels than other countries like DE, GB, CH and FR. Moreover, DK and NL also are the countries where the decline in 1990s is evident, while the levels in the other countries mentioned show little or no trends. For the eastern region, HU reported the highest levels, and also dominated the observed trend. No trends were discernible

for PL. LV, LT, while data for SK and EE only covered one single year. The data included from the northern region is comprised of SE and FI reports (IS only data for 2007, no chlorophyll data from Norway in the database). However, Norwegian data were recently reviewed by Lyche Solheim et al. (2008), who concluded that there had been only minor changes in eutrophication parameters (including chlorophyll a) before and after 1998. This result matches the lack of trend in FI and SE. In the southern region, the data cover only 2001-2006, and are predominantly from IT (CY only 2007, PT 2006-07). The data from the south-eastern region include several very short series. The longest record is from MK, and shows no trend.

For the three regions with data series covering 1991-2006, the chlorophyll a levels indicate the same trends or lack of trends as shown above for total phosphorus (section 4.1). This is to be expected since phosphorus usually is the primary limiting nutrient for freshwater phytoplankton.

In spite of a lack of trends for most time periods and regions, there are numerous examples showing that abatement measures are effective. Reducing the nutrient load on lake basins does result in lower biomass of algae, and also to lower frequency of cyanobacterial blooms. A lack of general trends therefore rather reflects that nutrient loads have not been sufficiently reduced on a large scale, except in some regions in the 1990s, and, of course, in several well-known single-lake cases.

6.5.2. *Evaluation of level of exceeding the WFD good status objective for chlorophyll*

A rough evaluation of the distance to the WFD chlorophyll targets is possible based on the grouping of related types of lakes given in Table 6.2. The lowest target (4-8 µg l^{-1}) is most applicable for clearwater deep lakes/reservoirs in the northern, Alpine and Mediterranean regions, whereas the middle target (8-12 µg l^{-1}) is most relevant for shallow lakes in the central region, as well as for humic lakes in the northern and central regions. The highest target is only relevant for very shallow lakes that mainly occur in the central region.

To relate these intercalibration regions to the geographical regions in Figure 6.11 the following coarse comparison is used:
Northern IC region ≈ northern region in Figure 6.11
Central-Baltic and Alpine IC regions ≈ western and eastern region in Figure 6.11
Mediterranean IC region ≈ southern and south-eastern region in Figure 6.11

For the northern region the current mean chlorophyll level is at the WFD target for moderate alkalinity or humic lake types. Although the lake types are not given in the current EIONET dataset, it is likely that a large proportion of the lakes reported are low-alkalinity lakes, since that lake type is dominant in the northern countries. If this is true, then the northern region is still above the WFD target for many lakes (dotted line in Figure 6.11) and more nutrient reduction measures are needed to achieve the target.

The western and eastern regions currently have chlorophyll levels close to or below the WFD target for very shallow (mean depth < 3m), high alkalinity lakes (whole black line in Figure 6.11). However, it is likely that a large proportion of the lakes reported belong to the shallow (mean depth 3-15 m), moderate to high alkalinity lake types. If so, then these regions still

have to reduce their nutrient inputs to lakes to ensure a further lowering of the chlorophyll levels to reach the WFD target for that lake type (dashed line in Figure 6.11).
For the southern and south-eastern region the data series are still too short and based on too few data to make an evaluation.

6.5.3. *Phytoplankton composition changes in lakes, incl. increase of toxic bluegreens*

Lake eutrophication leads not only to increased total algal biomass, but also to a general increase in the dominance of cyanobacteria in the phytoplankton community (Ptacnik et al. 2008). The threshold beyond which Cyanobacteria dominance can be expected is found already at chlorophyll levels of ca. 7-9 µg l^{-1} in low-moderate alkalinity, stratified, clearwater lakes (Figure 6.12), corresponding to a total phosphorus concentration of 15-25 µg l^{-1} (see also Figure 4.1 in Section 4.1). This response is the underlying scientific evidence for the WFD targets for chlorophyll and phosphorus in this lake type.
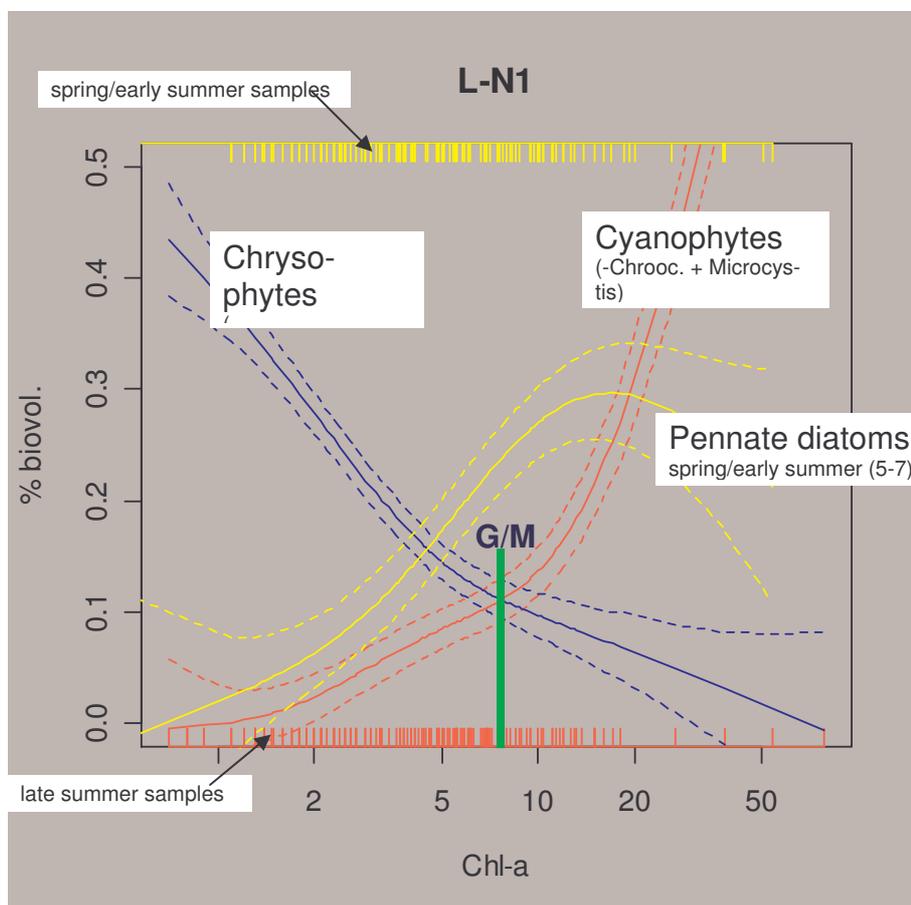
Figure 6.12 - Mean responses of phytoplankton indicator groups to eutrophication expressed as proportion of total biovolume (y-axis) along the trophic gradient, expressed by chlorophyll a (Chl-a, unit lg/l, x-axis). The vertical green line is the Good/Moderate boundary (WFD target) for the Northern lowland lake type L-N1 (moderate alkalinity, clear-water lakes) agreed by all countries in the Northern intercalibration group (NGIG),. The response curves are based on results given by Ptacnik et al. 2008. Data on chrysophytes and Cyanophytes are from late summer (July–September), whereas data on pennate diatoms are from spring/early summer (May–July). The distribution of the data along the chlorophyll gradient is shown as red and yellow bars at the bottom and top of the diagram, respectively. Modified from Lyche-Solheim, Ptacnik et al. (2008).

6.5.4. *Eutrophication impacts on aquatic submerged vegetation*

Due to shading from phytoplankton the aquatic submerged vegetation is also impacted by eutrophication. The most prominent response is the reduction and disappearance of the small isoetids at a phosphorus level 40-50 µg l^{-1} in softwater lakes (Figure 6.13). Similar response is found for sensitive charaphytes in hardwater lakes (Penning et al., 2008). This response has been used as a basis for setting the WFD target for submerged vegetation in lakes.
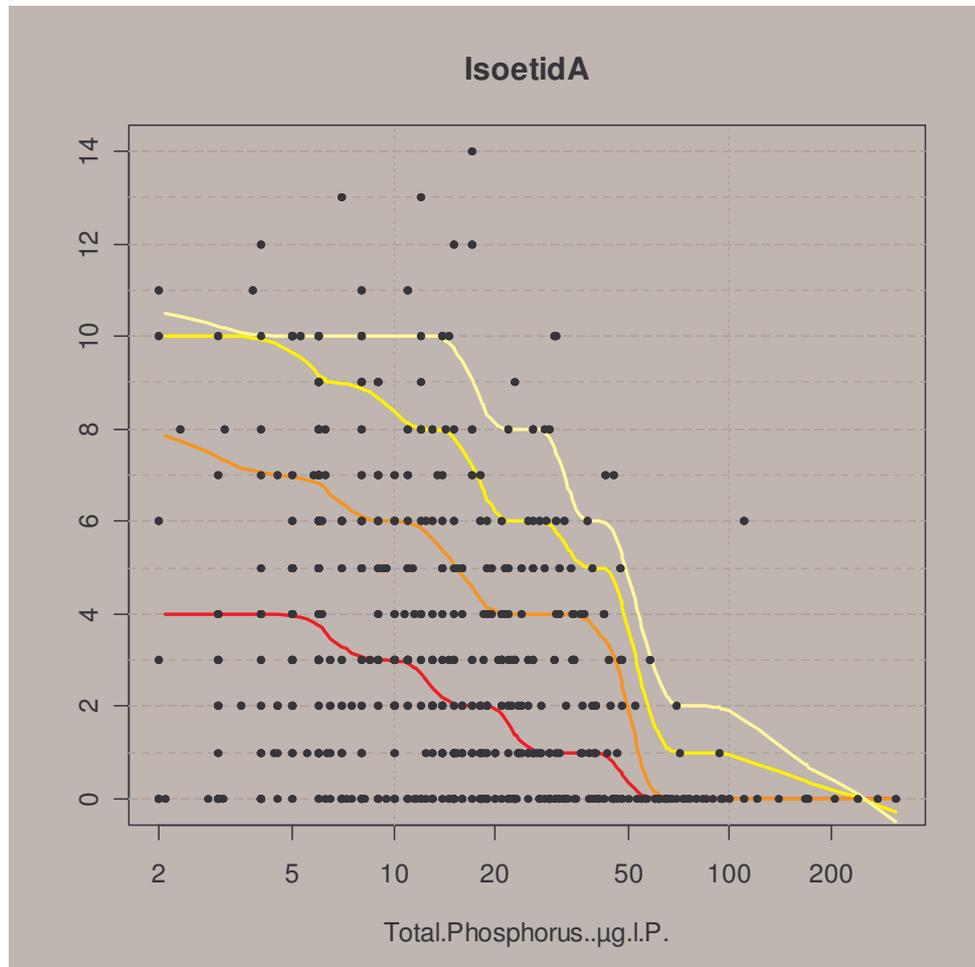
Figure 6.13 - Impacts on total phosphorus on submersed aquatic plants in lakes. Quantile regressions showing the reduction of the indicator plants, isoetids, at total phosphorus > 50 µg l^{-1} (Penning et al., 2008).

6.5.5. *Eutrophication impacts on aquatic macroinvertebrates*

Macroinvertebrates in lakes are also affected by eutrophication. In the profundal zone of stratified lakes, the most sensitive taxa disappear due to oxygen depletion, whereas in the littoral and sub-littoral zone sensitive taxa are also affected by eutrophication, but the response is often unclear due to complex trophic interactions and habitat changes (Solimini et al., 2006). Metrics to measure the impacts of eutrophication is under development in many countries, but so far no WFD target has been set for any lake type.

6.5.6. *Eutrophication impacts on fish in lakes*

Fish in lakes are affected by eutrophication in a number of ways: The total fish biomass increases, the species composition changes and age (size) structure changes from dominance of larger perchids and salmonids to dominance of smaller cyprinids. These changes aggravates the algal blooms due to reduced zooplankton grazing and cause further decrease of water transparency due to resuspension of sediments (Jeppesen et al.,1997). Fishing and stocking interferes with the responses and cause variability in the dose-response curves. Fish metrics to measure the impacts of eutrophication is under development in many countries, but so far no WFD target has been set for any lake type.

6.5.7. *Why do we see this status and this impact – relationships with nutrient pressure*

Most national environmental agency reports screened indicate that eutrophication problems in freshwaters are generally associated with agriculture, although sewage discharges frequently add to the problem. Sewage treatment has improved markedly in the latter decades in most countries, particularly in densely populated areas and cities. In rural districts, many small household discharges remain, however. In the northern region (Finland, Sweden, Norway), the association between lake eutrophication and agriculture is quite clear: the eutrophic lakes are mainly situated in agricultural districts, and this applies particularly to the larger lakes. Thus, improvements in lake eutrophication require regulations on farming practice and fertilisation regimes, as well as remedies that capture nutrients in runoff from agricultural fields. Continued efforts in sewage diversion and treatment will also be necessary in most countries.

High nutrient levels are clearly instrumental in the development of cyanobacterial blooms. Several national reports (e.g. Finland) mention high temperatures (or rather, long warm periods) as facilitating bloom formation. Synchronous occurrence in many lakes within a region further supports this notion, and some years blooms are more frequent. If generally true, temperature increase due to global climate change may clearly accentuate the problem of cyanobacterial blooms. Recently, a model study by Jöhnk et al. (2008) found temperature to be an important driver for *Microcystis aeruginosa* (a widespread species that often dominate in lakes, and sometimes produce microcystins). However, this result cannot readily be generalised, since temperature optima will vary between species and strains.

6.6. Impacts of toxic algal blooms on human health

Cyanobacteria represent perhaps the most undesirable eutrophication effect, since they tend to form surface scum and accumulate along the lake shores, and many species of cyanobacteria include toxin-producing strains. Cyanotoxins include hepatotoxic and neurotoxic compounds (microcystins, nodularins, anatoxin-a, homoanatoxin-a, saxitoxins, anatoxin-a(s) and cylindrospermopsin), and frequently represent a health problem for humans as well as domestic animals and wildlife, including birds and fish. Cyanobacteria and cyanotoxins may create problems in drinking water reservoirs, and human health incidents involving cyanotoxins have been reported from several European countries. While no human deaths have been recorded, there are several instances of cattle, sheep and dog deaths, and numerous bird and fish kills have been ascribed to cyanotoxins after drinking. Cyanobacteria occur not only in lakes, and similar problems occur in rivers as well as brackish waters.

At present, it is not possible to analyse trends in the frequency of cyanobacterial blooms or in cyanotoxin occurrence, since there has been no systematic recording of such data on a European scale. However, many European countries have adopted regulations to reduce risk of exposure to cyanotoxins based on biomass of cyanobacteria or directly on bioassays or measurements of concentrations of known cyanotoxins. WHO has developed guideline levels for cyanobacterial toxins in drinking water (Microcystin $< 1\,\mu g\,l^{-1}$) and bathing waters (Microcystin $< 10\,\mu g\,l^{-1}$) (Chorus and Bartram, 1999), which have been implemented by several countries.

As part of the International Hydrological Programme, Unesco supports CYANONET, a global network that aims inter alia to build a global database covering occurrences of cyanobacteria and cyanotoxins in water resources. Their first report was issued in 2005 (Unesco, 2005), and provided an initial situation assessment and recommendations. Europe was covered by a separate chapter, based on responses to a questionnaire as well as the literature.

The CYANONET 2005 report provides numerous examples of cyanobacterial blooms in European countries, dating as far back as the 19th century, as well as health incidences associated with some of these instances. Cyanobacterial blooms have been recorded in all 29 countries for which data or responses were available. Freshwater types include natural lakes, rivers, ponds, man-made lakes and canals, in addition to brackish waters such as the Baltic Sea, estuaries and lagoons. Among them were drinking water sources as well as waters used for livestock watering, aquaculture, recreation and tourism, and designated wildlife and conservation reserves. For instance, widespread occurrence was noted in at least 87 Spanish reservoirs since 1972, and in 54 of 94 lakes examined in Sweden 1984-1990.

Cyanobacterial blooms are not invariably associated with cyanotoxin production, although toxins are a common feature of blooms. At the present state of knowledge (2005), toxin-producing cyanobacterial blooms have been reported from at least 16 European countries. Hepatotoxic effects have been recorded more often than neurotoxic effects. It should be noted that bioassays have sometimes shown toxicity when analyses for known toxins have been negative, indicating that additional toxins occur, although not yet characterised and amenable to chemical analysis.

7. Mitigation and restoration measures

7.1. Reduction of nutrient pollution from urban waste water discharges

Continued future improvement in the level of nutrient removal from urban wastewater discharges is anticipated, driven by legislative requirements under the UWWTD, WFD and Habitats Directive. Whilst compliance with the UWWTD is already relatively high in the older MS, country-dependent deadlines for the newer MS, established in the Accession treaties, range between 2010 and 2018. As a consequence, improvements in both connection rates and treatment levels are likely to be realised for these countries over the coming years. Additionally, any further designations of 'sensitive areas' across Europe under the UWWTD will require the implementation of tertiary treatment and with it the substantial removal of nutrients.

Beyond the UWWTD, improved wastewater treatment could be triggered by the WFD, for example, where P concentrations in freshwater exceed levels required to achieve good ecological status. Furthermore, ongoing implementation of the Habitats Directive is likely to continue to identify Natura 2000 sites adversely impacted by wastewater discharges requiring, therefore, appropriate improvements in treatment.

Whilst at an early stage, research has been initiated in Europe to explore the potential for P 'recovery' from the wastewater treatment process with the aim of re-processing into fertiliser.

Whilst improved wastewater treatment has a key role to play in reducing nutrient emissions to freshwater, the process consumes energy and emits greenhouse gases. In addition, chemical removal techniques with respect to phosphorus involve the use of ferric or alum salts, raising the possibility that the subsequent discharge of these metals in treated effluent could contribute to the exceedance of EQS/consents in receiving waters. As a consequence of all these issues, the removal of nutrients at source, prior to their discharge to a treatment plant results not only in less energy consumption and chemical use, but has the potential to be more cost-effective as an approach. The use of P-free industrial and domestic detergents, in particular, has an important role to play in this respect and has been shown to significantly reduce P levels in influent received by wastwater treatment plants. Traditionally, sodium tripolyphosphate (STTP) has been the most common phosphate found in detergents, accounting for up to 50% of detergents by weight and playing a key role in the cleaning process. Phosphate-free alternatives to STTP, however, have been available for some time, in particular, zeolites, microporous crystalling solids that pose no ecotoxicological risk. To date no Europe-wide legislation has been developed to address P use in detergents. However, a number of countries have implemented either legislation or established voluntary agreements with detergent manufacturers at national level. Significant potential remains, however, for a greater use of P-free industrial and domestic detergents across many parts of Europe.

7.2.   Reduction of nutrient pollution from agriculture

To date, with a few regional exceptions, no clear improvement in nutrient water quality is evident in most agricultural catchments across Europe. However, this finding does not reflect a failure of current policy and legislation, rather that compliance with the WFD – the key legislation – is not required until 2015. Furthermore, appropriate measures under the Nitrate Directive (ND) have only recently been implemented in many countries whilst, for others, full compliance has yet to be achieved. Moreover, in many Nitrate Vulnerable Zones (NVZs), any decline in nitrate levels in freshwater resulting from these measures may take decades to occur due to the often slow movement of water and nitrate through groundwater.

Under the ND, a number of countries have designated their entire national territory as an NVZ, whilst in the remainder, NVZs typically encompass a substantial area of agricultural land. The ND requires, within NVZs, that fertiliser applications are balanced to crop requirements including compliance with a threshold limit for the application of N in manure of 170kg/ha/yr. Modelling studies have shown that compliance with the threshold will lead to substantial improvements in water quality; In the Brittany region of France, for example, reducing the application of N in pig manure to cereal crops from 2003 levels (280 kg/ha/yr) to the 170 kg/ha/yr limit is predicted to reduce the extremely high groundwater nitrate concen-

trations currently observed by more than 20 mg l^{-1}.

Other measures include the provision of a sufficient manure storage capacity that will not only lead to a reduction in nitrate pollution but will also reduce the emission of greenhouse gases. The Directive also requires restrictions on fertiliser application when soil conditions are unsuitable, for example, during prolonged wet weather or when the ground is frozen. In addition, applications near waterbodies or steeply sloping land are to be avoided.

Recent reforms (Agenda 2000 and the mid-term review) of the European Common Agricultural Policy (CAP) have decoupled agricultural subsidies from production levels and have, therefore, the potential to reduce the level of freshwater pollution by agriculture. The reforms also involve implementing a 'cross-compliance' mechanism that requires all farmers receiving direct payments under various schemes to comply with a set of 'statutory management requirements' in the areas of environment, animal welfare, animal diseases and public health. Payments are also dependent upon farmers keeping their land in 'good agricultural and environmental condition' with an aim of 'protecting water against pollution and run-off'. In addition, the regulation identifies the need to maintain existing permanent pasture and avoid its conversion into arable land. Adherence to this latter requirement, however, may be difficult to achieve given the Biofuels Directive and the likely future growing demand for energy crops.

In addition to decoupling and cross compliance, the CAP's rural development regulation includes the implementation of agri-environment and farm modernisation measures. These involve payments to farmers that carry out specific agri-environmental commitments that go beyond usual good farming practice. A range of measures are identified by the regulations with respect to the improvement of water quality, including improvement of manure storage, the use of cover crops, riparian buffer strips and wetland restoration.

Clear synergies exist between the rural developments measures of the CAP, those agri-specific measures to be identified within WFD River BasinManagement Plans and, the action plans currently being implemented under the ND. Maximising these potential synergies and achieving optimal outcomes will require strong integration between all relevant national and regional (including RBD) authorities.

A number of driving forces, some global and therefore beyond the direct control of national Governments, are likely to have a negative impact upon nutrient water quality over the coming decade and beyond. As a result, the implementation of more effective and extensive measures is likely to be required than would otherwise have been necessary in order to achieve compliance with water related Directives. These drivers impact particularly upon the agriculture sector and include climate change, global food demand and, the demand for biofuels.

7.3.   Restoration measures in rivers and lakes

In addition to reduction of external nutrient loads from urban waste water and agriculture (Sections 7.1 and 7.2) a number of internal restoration measures exist to speed up recovery of rivers and lakes from eutrophication. Some common restoration measures are:

- Restoration of wetlands and sedimentation ponds to enhance natural absorption of nutrient effluents;

- Establishing or widening buffer strips to enhance natural absorption of nutrient effluents;
- Remeandering of rivers;
- Harvesting aquatic vegetation along the shores of lakes to remove nutrients stored in the biomass;
- Biomanipulation of fish populations to enhance zooplankton grazing on phytoplankton;
- Oxygenation or artificial circulation of bottom waters of stratified lakes to prevent internal loading of phosphorus;
- Sediment dredging or covering with inert materials to prevent internal loading of phosphorus.

These restoration measures have been tried out in a number of rivers and lakes in Europe and elsewhere (Cairns, 1992). To be effective many of the measures must either be continuous over many years or repeated at certain intervals (harvesting vegetation, biomanipulation of fish, oxygenation, circulation). Their success-rate and cost-efficiency are questionable if the lakes are heavily degraded before introducing the measures. The most sustainable restoration measures in the long-term are the construction of buffer strips, restoration of wetlands and remeandering of rivers, but these measures may require complicated legal processes beforehand to expropriate areas close to rivers and lakes. Such measures will in any case merely be a supplement to reducing the external nutrient loads to rivers and lakes.

7.4. Response of lake ecosystems to measures (are they effective?)

Lake ecosystem response to P-reduction measures has been analysed using long-term data from 35 case studies all over Europe (Jeppesen et al., 2005). The following presents the summary of this paper.

1. This synthesis examines 35 long-term (5–35 years, mean: 16 years) studies of lake responses to reductions in external phosphorus loads. It covers lakes ranging from shallow (mean depth <5 m and/or polymictic) to deep (mean depth up to 177 m), oligotrophic to hypertrophic (summer mean total phosphorus concentration from 7.5 to 3500μg l^{-1} before loading reduction), subtropical to temperate (latitude: 28–65 ^{o}), and lowland to upland (altitude: 0–481 m). Shallow north-temperate lakes were most abundant.

2. Reduction of external total phosphorus (TP) loading resulted in lower in-lake TP concentration, lower chlorophyll a (chl a) concentration and higher Secchi depth in most lakes. Internal loading delayed the recovery, but in most lakes a new equilibrium for TP was reached after 10–15 years, which was only marginally influenced by the hydraulic retention time of the lakes. With decreasing TP concentration, the concentration of soluble reactive phosphorus (SRP) also declined substantially.

3. Decreases (if any) in total nitrogen (TN) loading were lower than for TP in most lakes. As a result, the TN : TP ratio in lake water increased in 80% of the lakes. In lakes where the TN loading was reduced, the annual mean in-lake TN concentration responded rapidly. Concentrations largely followed predictions derived from an empirical model developed earlier for Danish lakes, which includes external TN loading, hydraulic retention time and mean depth as explanatory variables.

4. Phytoplankton clearly responded to reduced nutrient loading, mainly reflecting declining TP concentrations. Declines in phytoplankton biomass were accompanied by shifts in community structure. In deep lakes, chrysophytes and dinophytes assumed greater importance at the expense of cyanobacteria. Diatoms, cryptophytes and chrysophytes became more dominant in shallow lakes, while no significant change was seen for cyanobacteria.

5. The observed declines in phytoplankton biomass and chl a may have been further augmented by enhanced zooplankton grazing, as indicated by increases in the zooplankton : phytoplankton biomass ratio and declines in the chl a : TP ratio at a summer mean TP concentration of $<100–150 \, \mu g \, l^{-1}$. This effect was strongest in shallow lakes. This implies potentially higher rates of zooplankton grazing and may be ascribed to the observed large changes in fish community structure and biomass with decreasing TP contribution. In 82% of the lakes for which data on fish are available, fish biomass declined with TP. The percentage of piscivores increased in 80% of those lakes and often a shift occurred towards dominance by fish species characteristic of less eutrophic waters.

6. Data on macrophytes were available only for a small subsample of lakes. In several of those lakes, abundance, coverage, plant volume inhabited or depth distribution of submerged macrophytes increased during oligotrophication, but in others no changes were observed d spite greater water clarity.

7. Recovery of lakes after nutrient loading reduction may be confounded by concomitant environmental changes such as global warming. However, effects of global change are likely to run counter to reductions in nutrient loading rather than reinforcing re-oligotrophication.

Lack of responses can be due to time-lags, climate change, alien species and habitat destruction. Time-lags may be critical to lake restoration and also to groundwater, and could be a key reason for a wave of derogations under the WFD.

The main results are shown in the Figure 7.1 below:

Shallow lakes

TP summer

Chla summer

Secchi depth summer

% Cyanobacteria

Deep Lakes

TP summer

Chla summer

Secchi depth summer

% Cyanobacteria

Legend:
- no trend
- up
- down
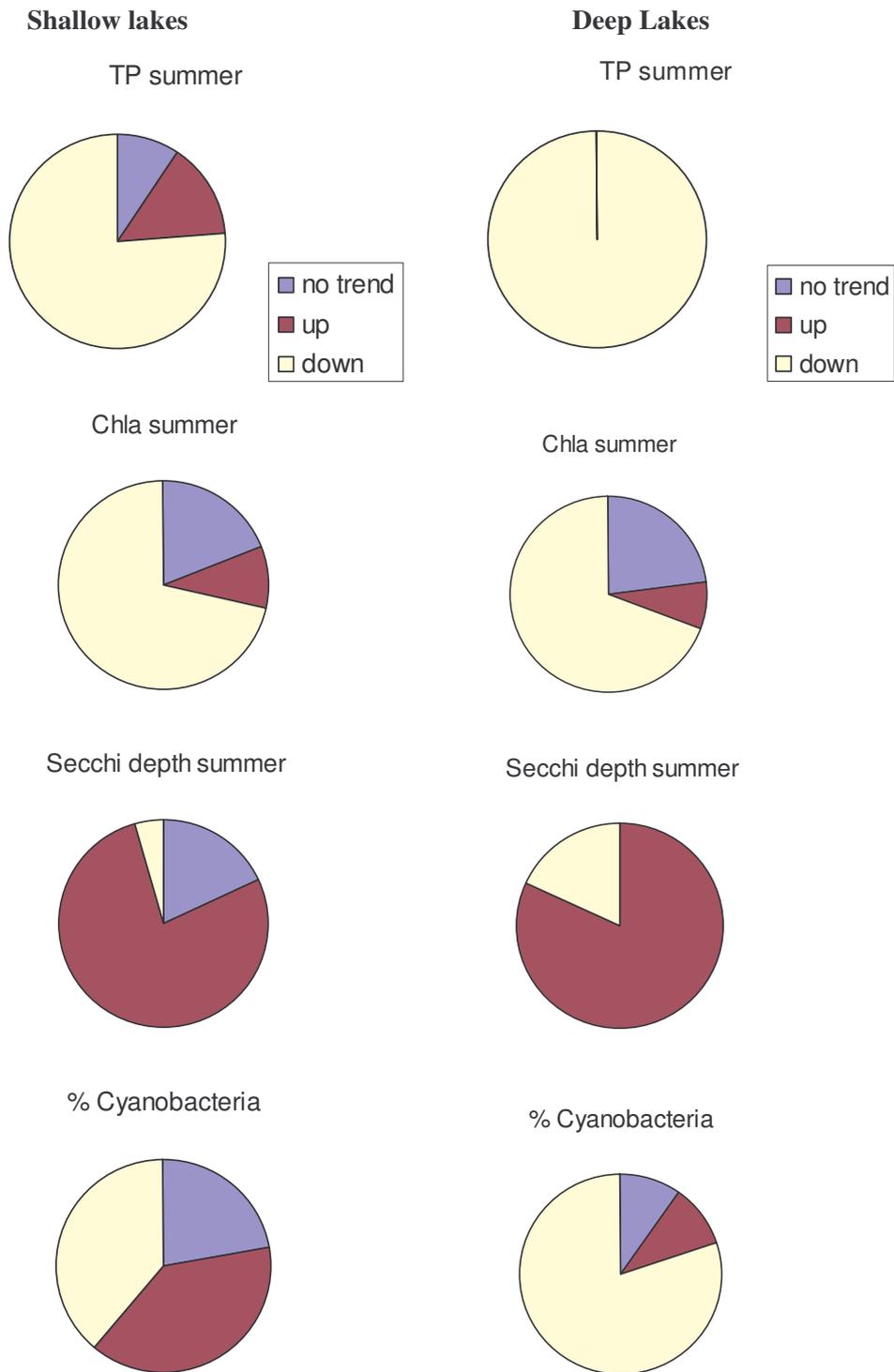
Figure 7.1 - Main trends of lake responses to phosphorus load reduction measures. Redrawn from Jeppesen et al., 2005. Left panel: Shallow lakes, Right panel: Deep lakes.

7.5. Response of river ecosystems to measures

Although a number of case studies exist, the experiences are not compiled in a comparable and quantitative way. The case studies included in Chapter 6.3 indicate however that river restoration is only partly successful so far, and that further measures to reduce diffuse nutrient loads, as well as other restoration measures against invasive species and hydromorphological alterations, must be implemented to enhance the likelihood of recovery.

The European Centre for River Restoration with national centres in several countries (UK, DK, IT, ES, RO, FI, BE) organised a workshop on river restoration in 2008. The most relevant conclusions and recommendations from the workshop are given below (Fokkens and Leummens, 2008):

1. Objectives which produce targeted and measurable outcomes for river restoration are rarely defined in advance;
2. Clear descriptions of reference conditions are used only occasionally as a basis for defining ecological objectives;
3. Commonly implementation of river restoration is dominated by the engineering approach of modifying hydro-morphological processes;
4. River restoration is mainly done on the local scale, focusing on the river, more rarely on (part of) the floodplain, hardly ever on the river basin. It is often uncertain whether local restoration efforts tackle the impact of relevant larger-scale regional factors;
5. During the latest decades new policy drivers have emerged, which link river restoration to pollution abatement, risk management, flood safety, renewable energy etc.;
6. River restoration implementation strategies are often focusing on selected species instead of whether integrated ecological objectives are being achieved.

8. Conclusions and key messages

Nutrient loads
Nutrient emissions to rivers and lakes primarily from agriculture and wastewater still cause eutrophication problems in freshwater ecosystems in Europe, and nutrient loads to coastal waters remain high from the major river basins of Europe. The proportion coming from diffuse sources (mainly agriculture) is ca. 70% for N and ca. 50% for P. Point source discharges remain important, particularly in Southern and Eastern Europe where wastewater treatment is not comprehensively implemented.

Nutrient concentrations in rivers, lakes and groundwater:
The average nitrate concentration in European rivers has decreased slightly since 1992, reflecting improved wastewater treatment, reduced atmospheric inputs and, in some regions, lower agricultural emissions. Based on Eionet data reported to the EEA (rather than the mandatory information required under Directives).

Nitrate-concentrations in groundwater exceeded the compliance threshold of 50 mg l^{-1} under the Nitrates, Groundwater and Drinking Water Directives in ca. 10 % of reported stations over the 2001-2008 period. Most of these stations are located in western and southern regions. High NO_3 concentrations in groundwater are found in particular in Spain, Belgium, Germany, Romania, Cyprus, Italy and Malta. The timeseries indicate increasing concentrations in the south-eastern region, possibly coupled to water scarcity and drought issues.

Phosphate concentrations in European rivers have decreased over the last two decades, by more than 30% from > 0.15 mg l^{-1} in the early 90-ies to ca. 0.08 mg l^{-1} in 2008. Most of the decrease occurred in the 1990s, reflecting the general improvement in wastewater treatment and reduced phosphate content of detergents over this period. High concentrations (> 0.1 mg l^{-1} P) are found in several regions with high population densities and intensive agriculture, including southeast UK, part of the Netherlands, Belgium, Southern Italy, central Spain and Portugal, western Poland, Hungary, Bulgaria, Macedonia, northern Greece. Given that phosphorus concentrations greater than 0.1 mg l^{-1} are sufficiently high to promote freshwater eutrophication, the observed high values in some regions of Europe are of particular concern.

Total P in lakes has decreased only slightly since the early 1990s, and has been more or less stable since 2000. Lakes with high concentrations of total P (>0.05 mg l^{-1}) are found mainly in RBDs in England, Belgium, the Netherlands, northern Germany, Poland, Hungary, Romania, Bulgaria, Spain, Portugal. Concentrations reported under Eionet generally exceed draft WFD targets (0.01-0-05 mg l^{-1}) for many common lake types. Further reductions in diffuse emissions are needed to achieve WFD objectives.

Impacts of eutrophication on freshwater ecology and human health:
There are major biological impacts demonstrated in the large river basins of Europe (Danube, Rhine, Po) due to excessive nutrient levels.

For lakes there are reliable long-term trends for the eastern and western regions showing a decline in algal biomass up to the year 2000, but little change since then. For the northern region, no temporal trend is evident. The current chlorophyll levels are still too high to meet the WFD objective in most regions. As for total P, the WFD target for chlorophyll a is exceeded roughly by a factor of two for many common lake types.

Human health incidents involving toxic algal blooms (cyanotoxins) have been reported from at least 16 European countries. While no human deaths have been recorded, there are several instances of cattle, sheep and dog deaths, and numerous bird and fish kills have been ascribed to cyanotoxins after drinking untreated water. Hepatotoxic (liver) effects have been recorded more often than neurotoxic effects. WHO has developed guideline levels for cyanobacterial toxins in drinking water (Microcystin $< 1\,\mu g\,l^{-1}$) and bathing waters (Microcystin $< 10\,\mu g\,l^{-1}$). These toxins should be monitored regularly in lakes (and rivers) prone to bluegreen algal blooms.

Mitigation and restoration measures, are they effective?

The UWWTD has enforced considerable improvement in urban wastewater treatment causing a major decline in nutrients discharged from wastewater plants, in particular from the older EU-15 Member States. Additional national legislation including bans on P in detergents has also contributed to this decline. For the eastern European region further improvements are anticipated during the coming decade due to the implementation of this directive.

An increasing proportion of the nutrient load to European waters comes from agriculture. These diffuse nutrient sources are expected to be reduced in the coming years due to the implementation of the Nitrates Directive and the WFD, as well as through adoption of agri-environmental measures under the European Common Agricultural Policy (CAP), but so far, with some notable exceptions, little improvement of nutrient water quality is observed in most agricultural catchments across Europe. A number of driving forces, e.g. climate change, global food demand and, the demand for biofuels, are likely to counteract the effect of measures taken under these directives and policies. As a result, the implementation of more effective and extensive measures is likely to be required than would otherwise have been necessary in order to achieve compliance with water related Directives.

European rivers have only partly recovered from eutrophication as a response to reduced nutrient inputs from point sources. Further measures to reduce diffuse nutrient loads are needed for full recovery. Additionally, other restoration measures against invasive species and hydromorphological alterations must be implemented to enhance the likelihood of recovery.

The restoration of European lakes from eutrophication is far from completed. The slow recovery is due to insufficient reduction of nutrient loads, in particular from diffuse sources, internal nutrient loads, as well as timelags in ecosystem responses. Lakes that are only moderately eutrophied and have most of their nutrient load from point sources, have a higher likelihood of recovery than very eutrophied lakes with most of the nutrient load from diffuse sources.

For groundwater there are few clear signs of improvement in any region. NO_3 concentrations are stable in most regions but with some evidence of increasing trends in some locations.

Eutrophication problems still persist in a large number of European water bodies, affecting

ecological status, biodiversity, ecosystem services and human health. Further nutrient reduction measures and other restoration measures are needed to reduce nutrient inputs, in particular from agriculture, and to counteract the future impacts of climate change and other global change processes.

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I. Annex - Case studies

I.1. Lakes

I.1.1. Norway: Lake Mjøsa (success story),(J.E. Løvik, 2009)

Lake Mjøsa is Norway's largest lake. It is a deep fjord lake (max depth 453 m) situated in Southeast Norway. The surface area is 362 km^2, mean depth 153 m, and the theoretical residence time 5.6 years. Large parts of the catchment area consist of mountain regions. Several glaciers in this region feed the main tributary with a heavy silt load during June-August. The water flow from the river causes reductions in transparency, temperature and algal growth in Lake Mjøsa during early summer, especially in the northern parts of the lake (Holtan, 1979).

The lowland areas around the central part of the lake are one of Norway's best agricultural districts. The total population in the catchment area is ca. 200 000, of which about 150 000 live in the immediate surroundings of the lake. The lake serves as drinking water reservoir for approximately 80 000 individuals and has great importance for recreation and fishing activities, as well as for agricultural irrigation purposes and recipient for sewage treatment plants.

At the beginning of the 20th century Lake Mjøsa was oligotrophic, but as in many other lakes in Norway an accelerating eutrophication took place from the 1950s, caused by increasing nutrient loading from agriculture, domestic wastewater and industries (Holtan, 1979). The first severe actions to reduce nutrient inputs were taken in 1973, but the eutrophication continued and culminated in 1976 with a vigorous growth of cyanophytes. As a consequence of this event and increasing attention to the massive water quality problems a comprehensive nutrient reduction program was established in 1977 ("Save Mjøsa Campaign", 1977-1981). In the following years high priority was given to building of sewage pipe lines and municipal treatment plants, and new restrictions were introduced in order to reduce discharges from agriculture and industries. A ban was put on P in detergents. The total cost of these measures was ca. NOK 900 million (ca. EUR 110 million). The nutrient reduction actions led to a near extinction of cyanophytes and marked reductions of primary production and phytoplankton biomass during the 1980s (Rognerud & Kjellberg, 1990; Holtan, 1993). The long-term oligotrophication process continued during the 1990s and caused increased transparency and nearly acceptable water quality in the lake (Kjellberg at al., 2001), although considerable autumn blooms of diatoms have been observed even in recent years.

Total P loads
The total P loads from tributaries have been reduced from ca. 100-170 tons yr^{-1} during 1979-90 to ca. 65-100 tons yr^{-1} during the period 2001-2008. This means a 40 % reduction in average. The flood year 1995 peaks with an estimated total P load of ca. 290 tons. P loads were also relatively high in 2000 (153 tons). (Figure I1). Volume-weighted means of total P have declined from ca. 11-17 µg l^{-1} during the period 1979-90 to ca. 8-11 µg l^{-1} in recent years (2001-2008). The main inlet river Lågen provides ca. 50-75 % of the yearly total loads of phosphorus from rivers.
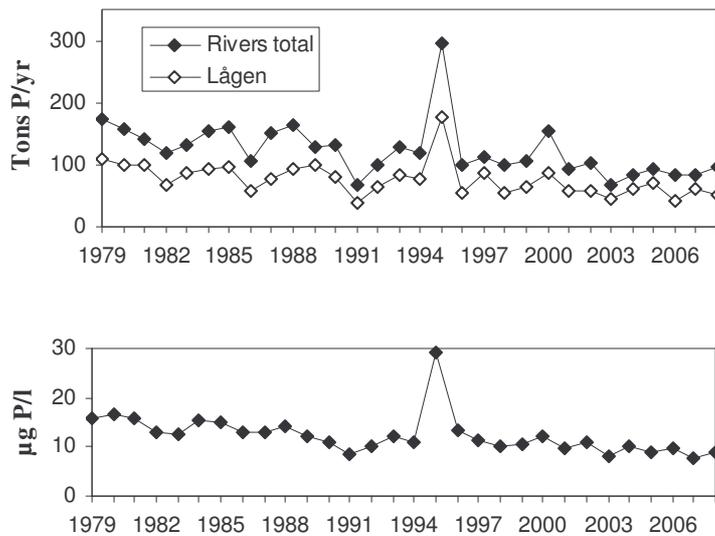
Figure I1 - *Upper panel: yearly P loads to Lake Mjøsa from the 6 most important tributaries including a 7 % estimated addition from tributaries where P measurements have not been carried out. Lower panel: yearly mean concentration of tot P (volume-weighted) in tributaries (sum of 6 inlet rivers).*

Nutrient concentrations

Phosphorus is the limiting nutrient for algal growth in Lake Mjøsa as in most lakes. Total P concentration in the lake water masses at late winter has shown a considerable decrease during the monitoring period 1972-2008, from ca. 8-12 $\mu g l^{-1}$ in the early 1970s (before the "Save Mjøsa Campaign", 1977-1981) to ca. 2-5 $\mu g l^{-1}$ in later years. Epilimnion (0-10 m) mean values for the algal growth season have shown a similar long-term trend with ca. 65 % decrease from the 1970s. The decline has been interrupted by periods with increase, like in the 1980s and during the years 1992-1997. According to the site-specific environmental goal for Lake Mjøsa, epilimnion P concentration in the algal growth season should not exceed 5.5-6.5 $\mu g \, l^{-1}$ in central and southern parts of the lake. After 1998 this goal has been fulfilled, with a mean tot P concentration varying in the interval 3.8-5.4 $\mu g \, l^{-1}$. A study of the long-term trend in Lake Mjøsa by means of the time series from the monitoring program and analyses of sediment cores indicates that "natural" total P yearly average has been lower than 5 $\mu g \, l^{-1}$, probably as low as 2 $\mu g \, l^{-1}$.

Thus the environmental objective of total P 6 $\mu g \, l^{-1}$ is compliant with the WFD objective for Northern deep lakes with an EQR value of 0.33.

The decline of P concentrations in Lake Mjøsa is a result of comprehensive measures to reduce nutrient inputs from agriculture, domestic sewage water and industries. This also includes local restrictions on P in detergents from 1977 and further enforcement in 1978.

The northern part of Lake Mjøsa has considerable lower total N concentrations than central and southern parts of the lake. This spatial difference is mainly caused by strong influence from the largest inlet river, Lågen, which enters in the northern part and usually has very low total N concentrations, while the central parts of the lake is more affected by supplies from agriculture and population from the local catchments. Total N concentration showed an in-

creasing trend during the 1970s until late 1980s, interrupted by a decline in the period 1979-1983. Since then, the N concentration curve has flattened off. Based on total N mean values for the algal growth season in 2008 the status is to be classified as "very good" at sampling station Brøttum in the northern part, as "good" at station Kise in the western part and as "moderate" at sampling stations Furnesfjorden and Skreia (main station) in eastern and southern part of the lake, according to WFD.
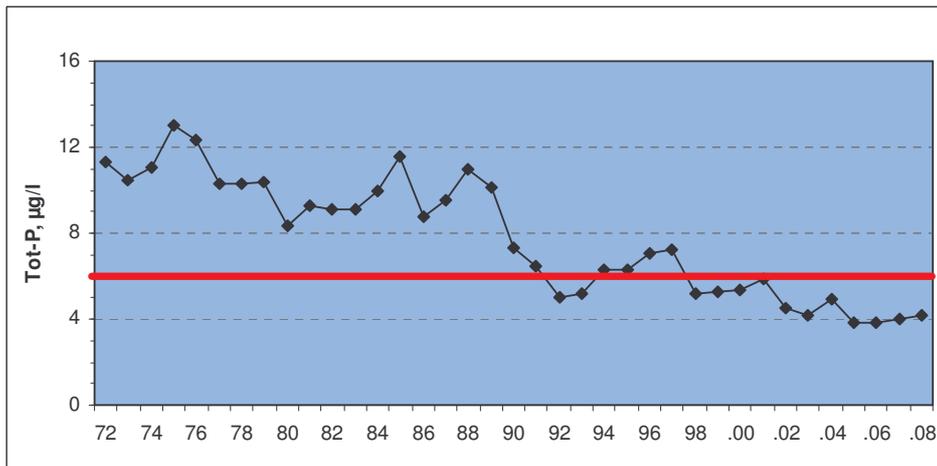

Figure I2 - Total P concentration development in Lake Mjøsa. Red line shows environmental objective according to local environmental authorities.

Phytoplankton biomass and species composition
As an effect of the pollution abatement actions, the phytoplankton biomass in Lake Mjøsa has been considerably reduced since the 1970s and the 1980s. In 2001-2008 mean phytoplankton biomass was ca. 65 % lower at the main station (Skreia) compared to the period 1972-1980. Phytoplankton biomass also showed a decreasing trend during years 2002-2007, both as chlorophyll-*a* (Figure I3) and as algal cell counts.
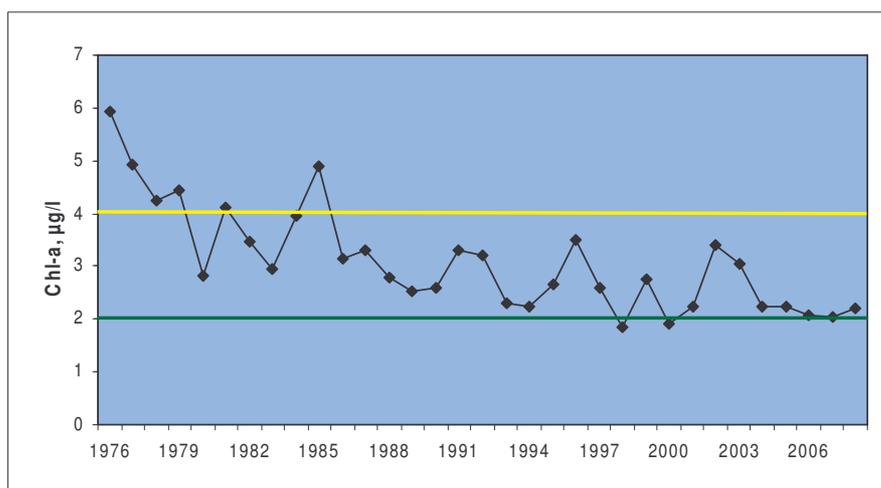

Figure I3 - Chlorophyll concentration development in Lake Mjøsa. Green line shows site-specific environmental objective (more strict than WFD-target for this lake type, see yellow line).

In 2008 the phytoplankton biomass was relatively low at all sampling stations until the middle of July, and the algal community was dominated by chrysophytes, cryptomonads, μ-algae and lesser amounts of diatoms and other groups, i.e. an acceptable composition of the algal community. Later in July the phytoplankton biomass increased considerably, and phytoplankton became more dominated by large diatoms like *Tabellaria fenestrata*. Periods with relatively large biomass of this species during late summer and autumn has been a quite common feature in Lake Mjøsa even in some recent years. The recorded biomass maximum in 2008 was higher than in 2006 and 2007 at all sampling stations. Diatoms accounted for a 64-76 % fraction of the total phytoplankton biomass at the different sampling stations. *Tabellaria fenestrata* is a good indicator for moderately eutrophicated lakes, i.e. oligomesotrophic and mesotrophic lakes. A high biomass of *Tabellaria* may lead to user problems by sticking to fishing nets or clogging water filters.

Transparency
As the phytoplankton biomass decreased in Lake Mjøsa, the transparency has improved substantially. In the 1970s Secchi disk transparency often varied in the interval 3-6 m in large parts of Mjøsa. Site-specific environmental objective states that Secchi disk transparency in central parts of the lake should be higher than 8 m. During recent years Secchi disk transparency has generally varied between 6 m and 11 m. During the summer of 2008 transparency was lower than the environmental objective at several sampling stations, mainly caused by periodically high diatom biomass and/or large inputs of water rich in humus and inorganic particles from the watershed.

I.1.2.   Norway: Lake Vansjø (problem story) (Skarbøvik et al., 2009)

Lake Vansjø in South-Eastern Norway is a fairly large eutrophied lowland lake with pressures from agriculture and scattered dwellings. The lake serves as drinking water source for ca. 60 000 people and is used a lot for recreation. The lake has been subject to considerable efforts (> 30 mill. Euro) to reduce phosphorus loads during the last 5 years, including a series of agricultural measures: changed tillage practices from autumn ploughing to spring ploughing, buffer strips, sedimentation ponds, reduced application of fertilizers, reduction of effluents from scattered dwellings, as well as an improved sewer system. In spite of these measures, the actual P load has not decreased. Accordingly, the lake has also not improved, either in terms of in-lake P concentration, or in terms of phytoplankton biomass (Figure I4).

Table I1. - Total P loads to Lake Vansjø eastern basin (Storefjorden) the years after P reduction measures were implemented.*

Major tributaries	2005	2006	2007	2008
Hobøl river	10,6	19,7	15,0	20,1
Svinna	2,3	2,0	2,5	2,7
Mørk river	0,9	0,9	1,0	1,0
Veidals river	1,1	1,1	1,1	1,3
Total P load	14,9	23,7	19,6	25,1

The P load to the western basin (Vanemfjorden) from the local catchment has decreased slightly during the same period, but it is too early to see any clear response to this reduction in

the lake. The coming years will show whether this decrease continues and whether this part of the lake will eventually respond to the local load reduction.
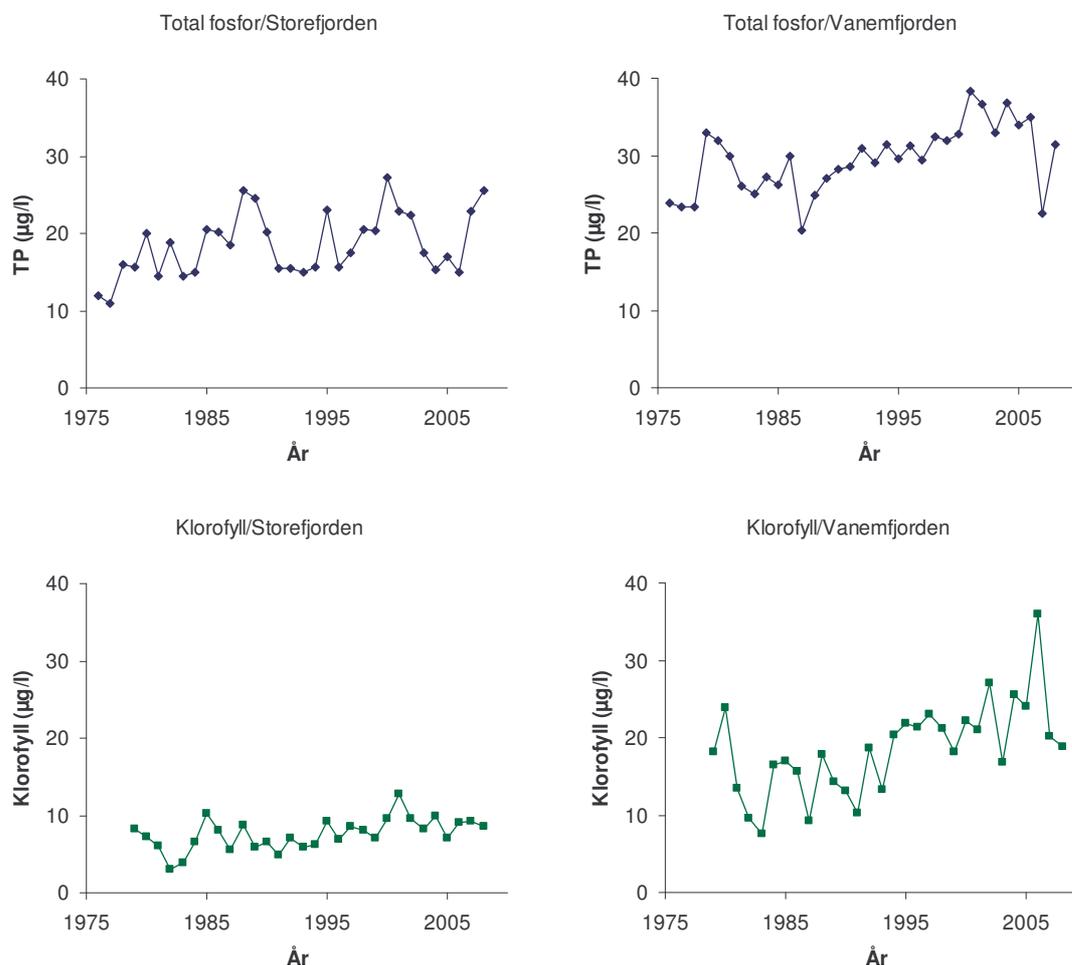

Figure I4 - Timeseries of total P (upper panels) and chlorophyll (lower panels) in the two main basins of Lake Vansjø (Skarbøvik et al., 2009).

According to the new WFD compliant classification system, the eastern mesotrophic basin (Storefjorden) is in moderate status, whereas the western more eutrophic basin (Vanemfjorden) is in poor status. The national.WFD target for chlorophyll is 7.5 µg l^{-1} for the eastern basin and 10.5 for the western basin, whereas the present concentrations are 8.5 µg l^{-1} and 25 µg l^{-1} respectively (Table I2).

The lake still has occasional blue-green algal blooms during summer, which cause problems for water supply and bathing, although the toxin level decreased to below the WHO guidelines in 2008 for the first time since the measurements started in 2005.

Table I2 - Ecological status of Lake Vansjø in 2008 according to WFD-compliant boundaries. The boundaries between good and moderate status for chlorophyll and phosphorus (WFD target) are given in parenthesis.

Parameter	Storefjorden	Vanemfjorden
Chlorophyll [µg l^{-1}]	8,9 (7,5)	25 (10,5)
Total phosphorus [µg l^{-1}]	21 (16)	29,6 (19)

The reason for the missing response to the P reduction measures is probably a combination of factors, including long time lags to reduce the catchment soil phosphorus content, sub-surface runoff in ditches, as well as climate change-related increase of river discharge, causing enhanced run-off, as well as increased erosion along the river banks. These processes delay or counteract the P reduction measures and maintain high P loads to the lake (Table I1). Without the P reduction measures, however, the P loads would have been even higher, and the lake would probably have become even more eutrophic than the present status.

This case study illustrates the problem of restoring eutrophic lakes under climate change (Battarbee et al., 2008).

I.1.3. *Italy: Lago Maggiore (success story of UWWT) (G. Morabito, CNR)*

Lago Maggiore, the second largest Italian subalpine lake, is oligotrophic by nature, as testified by early limnological studies (Vollenweider, 1965) and by the analysis of the sedimentary pigments (Guilizzoni et al., 1983; Marchetto et al., 2000). The eutrophication process started in the sixties: the algal nutrient concentration in the lake water started to increase and was soon followed by an increase of the phytoplankton abundance, biovolume and primary productivity (Ravera & Vollenweider, 1968; Morabito & Pugnetti, 2000). The lake reached a trophic state close to eutrophy in the late seventies, when the P loads peaked and the maximum in-lake TP concentration at winter mixing was recorded (around 30 µg l^{-1}; Mosello & Ruggiu, 1985). Since that time, the P loads have been gradually reduced by various means (Figure I5), among which the adoption of treatment plants and the reduction of total phosphorus in detergents were the most important. As a result, the values of TP at winter mixing gradually decreased to values around 10 µg l^{-1} in the most recent years (Ruggiu et al., 1998). The slow reversal of the trophic state of Lago Maggiore is documented by many papers: from a biological point of view, strong emphasis was put in the eighties on the apparent resilience of the plankton communities against falling phosphorus (de Bernardi et al., 1988). However, starting from 1987-88, major biological changes were at last manifest (Manca et al., 1992). Notable changes were recorded in the structure of the phytoplankton assemblages with oligotrophication (Ruggiu et al., 1998): among the most important of these, we must mention a remarkable decrease of the average cell size due to an increased importance of the smaller sized phytoplankton taxa.

In Lake Maggiore the biodiversity of phytoplankton proved to be a good indicator of the environmental changes: following the decrease of phosphorus input, the number of algal taxa gradually increased from about 50 total taxonomic units recorded at the end of the eighties to the almost 100 taxa found in the most recent years. The most significant rise of diversity took place among the cyanobacteria: from two or three blue-greens usually dominant in the eighties, to seven or eight dominant species in the most recent period. Cyanobacteria populations are among the most typical component of the pelagic microflora in the deep and large Italian subalpine lakes: their seasonal dynamics and species composition are however different in the lakes of the subalpine district, according the respective trophic evolution and present trophic state. The numerous investigations carried out on the phytoplankton assemblage of these

lakes, although discontinuous in some cases, show that *Planktothrix rubescens* is the most common and widespread cyanobacterial species: this organism, commonly taken as an indicator of moderate to high trophy (Reynolds, 1998), reached its maximum development in Lake Maggiore during the seventies, and remained among the dominant species until the end of the eighties, then gradually declined, leaving place to a more diversified cyanobacterial community. In particular, the decline of *P.rubescens* has been coupled with the increase of small Chroococcales (mainly *Aphanothece clathrata* and *Cyanodictyon planctonicum*). Within the algal class diatoms, the oligotrophication process resulted in an increase of the diversity, coupled with a decline of the dominant pennate taxa (such as *Asterionella formosa* and *Diatoma elongatum*) and an increase of many small centric species, belonging to the genera *Cyclotella*, *Cyclostephanos* and *Stephanocostis*.

The temporal development of chlorophyll-*a* in Lake Maggiore showed a clear tendency to decrease, together with the concentration of total phosphorus (Figure I6). From the end of the eighties until 1988 annual peaks were higher than 8 µg l^{-1}. Excluding two isolated cases in 1995 and 1996, corresponding annual maxima were constantly below 7 µg l^{-1} since the end of the 1990s.

This pattern is confirmed by the temporal development of the annual average of chlorophyll-*a*, which showed, except for a limited period in 1995–1996, a decrease from the 1980s (4.0–5.5 µg l^{-1}) to the end of the 1990s and the beginning of the new millennium: a minimum below 2.0 µg l^{-1}, with a corresponding shift of the lake from mesotrophy to oligo-mesotrophy (according to OECD scale), was recorded in 2002. The most recent years shows a slight increase of this parameter, although the nutrient availability did not change. Reasons for such apparent trend reversion need still to be evaluated.
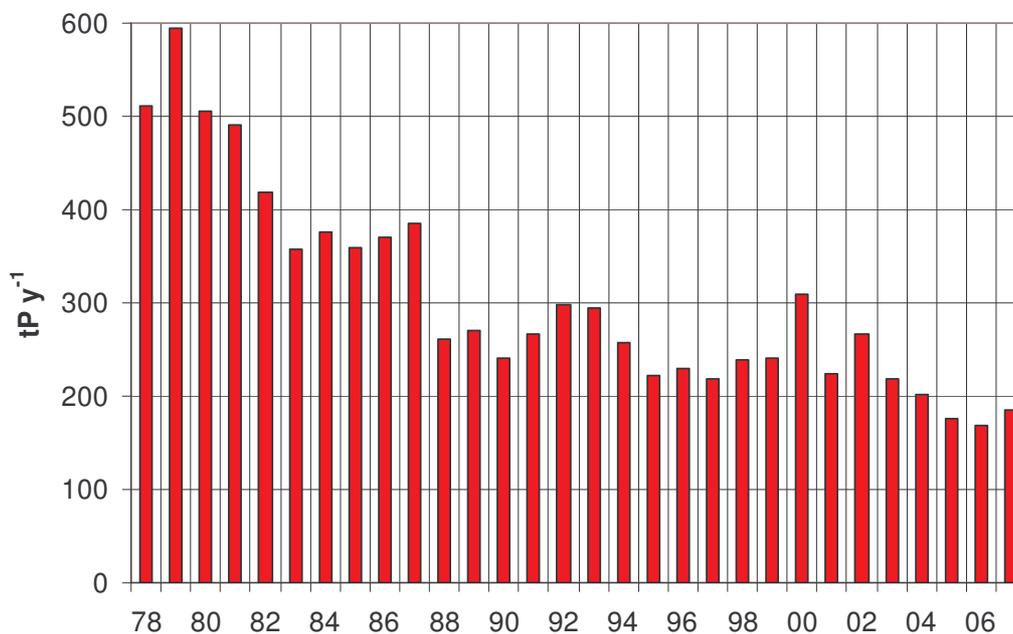

Figure I5 *- Phosphorus loading to Lago Maggiore in tonnes phosphorus per year (1978-2007).*
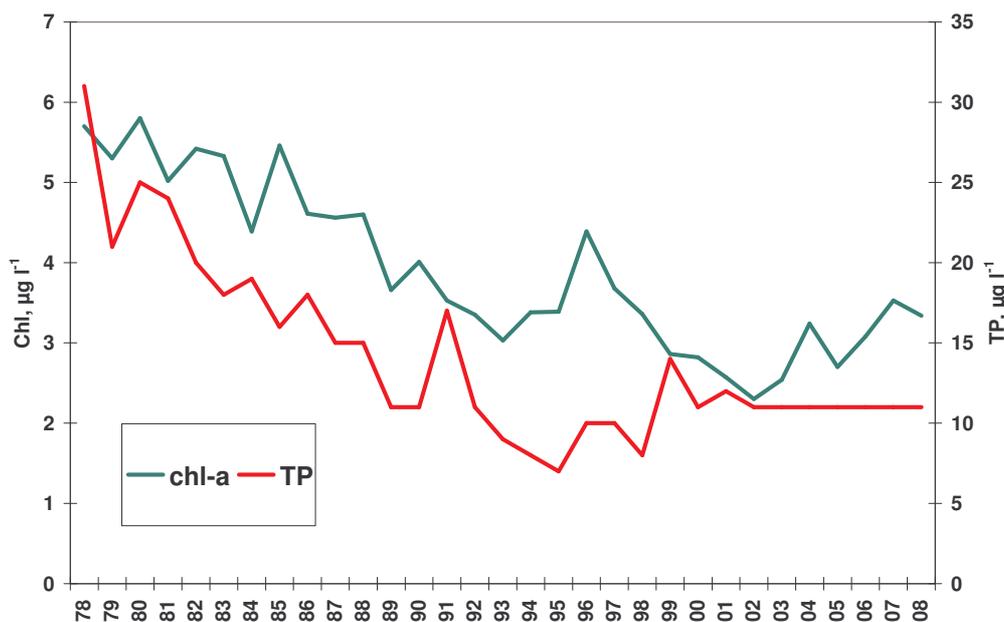
Figure I6 - *Lago Maggiore Chlorophyll a (yearly average) and Total Phosphorus (spring mixing) from 1978 to 2008.*

I.2.   References for Annexes

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Abbreviations

BOD	Biological Oxygen Demand
CAP	Common Agricultural Policy
COD	Chemical Oxygen Demand
CSO	Combined sewer overflows
E-PRTR	European Pollutant Release and Transfer Register
HELCOM	Helsinki Commission Baltic Marine Environment Protection Commission
ICPDR	International Commission for the Protection of the Danube River
JRC	Joint Research Centre
ND	Nitrates Directive
NVZ	Nitrate vulnerable zones
OSPAR	Oslo and Paris Conventions for the protection of the marine environment of the North-East Atlantic
RBD	River Basin District
TOC	Total Organic Carbon
UNEP	United Nations Environment Programme
UWWTD	Urban Wastewater Treatment Directive
UWWTP	Urban Wastewater Treatment Plants
WFD	Water Framework Directive
WHO	World Health Organisation