Nutrients in European ecosystems

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Cover design and layout: J. H. Schultz Grafisk A/S

A great deal of additional information on the European Union is available on the Internet.
It can be accessed through the Europa server (http://europa.eu.int)

Cataloguing data can be found at the end of this publication.

Luxembourg: Office for Official Publications of the European Communities, 1999

ISBN:

Printed in

Printed on recycled and chlorine-free bleached paper

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Contents

Preface

This report presents a pan-European overview of the geographical distribution, and severity of adverse effects, of excessive anthropogenic inputs of nutrients in European ecosystems. As such it deals with lakes and reservoirs, rivers, transitional and marine waters, and also terrestrial ecosystems. It has been produced by the European Topic Centre on Inland Waters (ETC/IW) on behalf of the European Environment Agency (EEA). The project was led by the International Office for Water (France) with the assistance of the Water Research Centre (UK), National Environmental Research Institute (Denmark), the Centro de Estudios y Experimentación de Obras Públicas (CEDEX) (Spain), and the European Topic Centres on Nature Conservation and on the Marine and Coastal Environment.

An important element of the project was the collection of data by means of a questionnaire distributed to 42 European Countries through the EEA's National Focal Points, the PHARE Topic Link (PTL/IW) and other components of the EIONET. Data and information provided from 30 of these countries have been used in this report. Supplementary information from literature, reports (for example, national state of the environment reports, and reports produced by international organisations such as Eurostat, OSPAR and HELCOM) was also used when appropriate.

This report is also the source document for information on surface water aspects used in two environment reports published by the European Environment Agency: *Europe's Environment: The Second Assessment* published in June 1998 and *Environment in the European Union at the turn of the century* published in June 1999.

The report aims to provide information for policy and decision makers at both the national and European level. For example, it will aid the European Commission's review of the progress made by implementing the Fifth Environmental Action Programme 'Towards Sustainability'. It will also be of value to Non-Governmental Organisations, and of general interest to informed members of the public.

At the time of preparation of this report there was no harmonised European monitoring or information network for water through which comparable information could be obtained. Thus this report is based on the best available information and has been validated, where possible, through review by the EEA's National Focal Points. It is, however, important for the reader to bear in mind the context and limitations of the information presented.

For example, background information was requested on the type of river and lake sampling stations at which quality data was measured, but it was not always clear whether they represented natural background situations, high contamination areas or gave a representative view of quality for a particular river length or lake. Some river monitoring networks might concentrate on relatively impacted areas on nationally large rivers. Alternatively they might be designed to give an overall national assessment of quality, in which case 'impacted' and 'background' quality rivers would be included. Comparison of results from different sampling stations may lead, therefore, to potentially wrong conclusions.

The problems mentioned above are of particular importance if information is compiled and compared at the European level as it is in this report. Data aggregated at the country level may not fully reflect the actual national status and level of risk to aquatic and terrestrial ecosystems within a country.

However in spite of these limitations, the information network of the EEA, the Environmental Information and Observation Network (EIONET), has been successfully used to collect information on the impacts of anthropogenic nutrients on aquatic and terrestrial ecosystems on a pan-European scale. The implementation of EURO-WATERNET, the EEA's information and monitoring network for inland water resources, across Europe will improve the reporting of information on the state of surface fresh waters and the pressures placed upon it.

Executive summary

The Dobrís Assessment of Europe's Environment clearly identified eutrophication as a major environmental issue in both freshwater and marine waters. The atmospheric deposition of nitrogen oxides was also identified as a serious threat to terrestrial ecosystems. The presence of excessive amounts of nutrients has direct effects, such as high concentrations of nitrogen compounds making water unsuitable for human consumption, and indirect effects, such as eutrophication[1], on ecosystems and water uses.

Purpose

This Environmental Assessment Report, produced by the European Topic Centre on Inland Waters (ETC/IW) on behalf of the European Environment Agency (EEA), provides an overview of the geographical distribution and severity of adverse biological effects of excessive anthropogenic inputs of nutrients into rivers, lakes and reservoirs, estuarine and coastal waters, as well as into sensitive terrestrial ecosystems. Only those nutrients (also called 'macronutrients') that are important in terms of nutritive value and nuisance levels are assessed. They are in practice phosphorus and nitrogen compounds. Those compounds that may limit the growth rate or biomass yield of aquatic plants are given special attention.

Information used

Information was obtained from published and 'grey' literature. Data used in calculations were obtained only through the distribution of questionnaires to the EEA's National Focal Points and designated points of contact in central and eastern European countries. Geographically the assessment, thus, covers pan-Europe, and includes EEA member countries, Switzerland and the so-called PHARE and TACIS countries. The data and information obtained from countries are very patchy in terms of quality, comparability, suitability and the ecosystems covered. The report is, thus, based on the best information available to the ETC/IW and the EEA.

The assessment method used followed, as far as possible, the DPSIR (**D**riving forces, **P**ressures, **S**tate, **I**mpact, **R**esponse) assessment framework adopted by the EEA. However, considering the fundamental differences in the ecosystems considered and also the lack of the appropriate and specific information required, it was not possible to undertake a comprehensive DPSIR assessment.

Sources of nutrients

There are both natural and anthropogenic sources of nutrients to all ecosystems. If there were no human activities, water composition would be determined only by hydrological and geochemical factors. Apart from the run-off producing water flow, the main processes involved are the weathering of bedrock minerals (which determines the basic composition of water), aerial deposition of dust and salts (which contributes to the nitrogen load), and natural leaching of organic matter and nutrients from soils and decaying biological material. The major source of phosphorus in pristine rivers is considered to be erosion of soil particles.

In general the loads of nutrients increase with increasing human activity in catchments of aquatic ecosystems – these loads produce pressures on the environment. During the last century, industrial production and household consumption (driving forces) have increased at a rapid rate, producing greater loads of nutrient-rich waste water. The extent to which this is discharged into surface waters depends on the proportion of the waste discharges connected to sewerage systems and on the sewage treatment facilities available.

Activities within the agricultural sector (another driving force) have also changed with the result that intensive farm manage-

[1] The term 'eutrophication' is defined in section 4.3.

ment practices are considered normal. Since the 1940s, the area of ploughed land and the use of commercial inorganic fertiliser has increased dramatically. This, in conjunction with increased livestock densities, has resulted in the production and application of much greater loads of manure to cultivated land. Higher livestock densities also result in greater emissions of ammonia which, in turn, lead to greater atmospheric deposition of nitrogen to land and surface waters. Changes in arable farming practices have also increased the rate of soil erosion, with a related increase in phosphorus run-off. In many areas, much of the agricultural land is drained and a large number of Europe's marshes, wetlands, ponds and lakes have disappeared. This has considerably reduced the capacity of freshwater ecosystems to store and eliminate nutrients and many pollutants.

Most of the dissolved phosphorus loading of inland surface waters is attributable to discharges from point sources, especially municipal sewage and industrial effluent. The emissions of phosphorus are decreasing in some parts of Europe. Results from some large river catchments, or national emission inventories, show that there has been a reduction of typically 30-60 % since the mid-1980s. Improved waste water treatment and the substitution of phosphorus in detergents have resulted in a decrease of phosphorus concentrations in some European surface waters over the last 10 to 20 years. Nevertheless, the anthropogenic contribution to phosphorus loading is generally far beyond the natural load in most parts of Europe. To reduce eutrophication, further reductions are needed in phosphorus emissions, both from point and from diffuse sources.

In contrast to phosphorus, nitrogen loading is primarily from agricultural activity, especially the use of nitrogen fertilisers and manure. The nitrate level in most European rivers has increased during the last 10 to 20 years, mainly as a result of increasing or high use of nitrogen fertilisers, and intensification of crop production on thin and fragile soils. In those river systems draining catchments in the central and western part of Europe, 46-87 % of the nitrogen load to inland waters is related to agriculture. In some catchments, point sources of nitrogen (predominantly municipal sewage treatment plants) also play an important role, accounting for 35-43 % of the total discharge.

The impacts of excessive nutrients are either manifested as a direct effect (for example, the nutrient acting as a toxin) or through an eutrophication effect (where the nutrient acts directly on the trophic structure). These impacts are detailed in later chapters.

Lakes and reservoirs

The density and size of natural lakes throughout Europe vary considerably. In terms of numbers the majority of natural lakes in Europe occur in Norway, Sweden and Finland with 85 000 and 56 000 lakes over 1 hectare in Sweden and Finland, respectively. It has been estimated that over 9 % of the land area of Finland and Sweden is covered by freshwater lakes. Significant numbers of natural lakes also exist in Iceland, Denmark, Ireland and the UK.

Reservoirs are distributed quite differently over Europe, often with the highest densities in regions with low rainfall, particularly southern Europe. In other countries with increasing demands for water and energy, reservoirs are also numerous. It has been estimated that there are about 3500 large reservoirs with a dam height exceeding 15 m in the EEA member countries. Spain, France, UK and Italy have the largest number of major reservoirs (more than 400 in each case). Nordic countries have lower numbers of reservoirs, but these are generally of larger capacity.

Impacts arising from excessive nutrients in still fresh waters are usually caused by phosphorus. In unaffected lakes and reservoirs the concentration of phosphorus is generally below 25 µg l^{-1}. A large proportion of the lakes in most parts of Europe has phosphorus concentrations exceeding this limit, thus indicating a significant anthropogenic influence. Only in sparsely populated regions, such as parts of the Nordic countries, Ireland, parts of the UK and central European mountain regions, is there a high proportion of lakes with lower phosphorus concentrations. Primarily because of improved urban waste water treatment, there have been quality improvements in some lakes since the 1970s. In particular the proportion of heavily polluted lakes and reservoirs has decreased. In some of the latter cases observed concentrations were three or more orders of magnitude above ‘reference’ (unimpacted) values. At the same time, the proportion of pristine or near pristine lakes has tended to shrink. The state of European lakes and reservoirs is, thus, still heavily affected by anthropogenic nutrient pollution and the condition of many lakes is far from satisfactory.

Rivers

River catchments in Europe are particularly numerous because of the catchments’ geological structure and shape. However by world standards, only a few catchments are drained by very large rivers. Only 31 catchments exceed 50 000 km^2, and these drain 6.63 million km^2, approximately two thirds of the continent’s area of 10.2 million km^2.

Nitrate, ammonium and its toxic form, ammonia, are present in excessive concentrations in almost all river monitoring stations for which information was received. Available data suggest that this situation is slowly improving, although this is mainly at the worst quality locations, whilst in contrast the number of good quality stations has tended to decrease.

Phosphorus is present in excess at practically all monitoring stations for which information was provided. In terms of eutrophication, this excess generates unwanted plant growth at a large proportion of stations. It is very likely that stations, where no excessive quantity of biomass can be detected (directly or indirectly), are not sensitive to eutrophication because of other controlling factors (e.g. turbidity or river velocity). The imbalance in main nutrients is exaggerated by human inputs: the most polluted sites have N/P ratios favourable to cyanobacteria growth, which is often observed in great quantities during drought periods, especially in slow-flowing water bodies.

Marine and coastal waters

Between 1990 and 1995 the load of total nitrogen from rivers and direct discharges to seas in the western and northern part of Europe appears to have increased for the North and Baltic Seas, while there appears to have been no changes in the other seas for which information is available. No changes in the atmospheric deposition of nitrogen were measured in the North, Mediterranean and Black Seas, while in the Baltic Sea there appears to be decreasing deposition. The North East Atlantic Ocean shows variable levels of load.

Trends in total phosphorus loads over the same period appear to be somewhat different from nitrogen loads. The North Sea data show an increase in phosphorus load, the Iberian Coast gives a variable picture, loads to the Irish and Celtic Seas have been steady since 1991, and the Baltic Sea shows a decrease between 1990 and 1995. The three northern-most seas show no changes.

Marine eutrophication is a widespread and transboundary phenomenon in almost all European seas with eutrophication related phenomena being observed over increasing areas and with (perhaps) increasing frequency. It affects marine biodiversity and fish or shellfish stocks as well as human health and recreational uses of marine coastal zones.

Terrestrial ecosystems

Terrestrial ecosystems receive a range of nutrients from anthropogenic sources. The most important pathways from anthropogenic sources to natural and semi-natural terrestrial environments is through atmospheric deposition, and through lateral groundwater flow or surface run-off from agricultural fields. The terrestrial environments susceptible to increased availability of nutrients are, however, mainly influenced by atmospheric deposition. In these ecosystems, the largest effects are expected from increased availability of nitrogen, partly because growth and competition might be controlled by nitrogen, partly because the atmospheric input of nitrogen has increased more than inputs of, for example, phosphorus, potassium or magnesium.

Those species belonging to terrestrial ecosystems most sensitive to excessive nitrogen deposition are mainly, or completely, dependent upon nutrient input from the atmosphere. Thus these species are adapted to survive at very low nutrient levels. Since many of these plants use their entire surface for uptake of nutrients from air or precipitation, they are directly exposed to excess inputs of nitrogen and other nutrients as well as toxic compounds from the atmosphere. Examples of such ecosystems are the raised and blanket bogs of north-western Europe, Arctic ecosystems with a very low rate of mineralisation, lichen and moss dominated heathlands, and plant communities dominated by lichens and mosses. Epiphytic lichen communities are probably those most sensitive to excess nitrogen.

Current estimates, based on 1990 deposition values, indicate that 38 million hectares of nature areas in the EU15 countries, or 34% of the nature areas[2] considered, receive nitrogen deposition above their critical loads for eutrophication. The corresponding figures for the whole of Europe are 77 million hectares or 18% of the nature areas considered. It is expected that by 2010 the area receiving nitrogen deposition above the critical load for eutrophication will be reduced to 19% within the EU15 countries and 11% for the whole of Europe under currently agreed abatement measures.

Impacts on water use

In addition to the impact on the ecosystem, the effects of eutrophication cause problems for the use of surface waters. Public water supply is particularly vulnerable to eutrophication effects. For example, it may lead to problems in the water treatment system such as filter clogging and undesirable tastes, odours and colour, both because of excessive algae. Other uses are less sensitive to eutrophication, though it may cause problems for hydroelectric, irrigation or fish hatchery purposes.

Eutrophication may also render a lake unsuitable for recreation because of the unpleasant appearance of water caused by high turbidity/low transparency, odours or algal masses. Furthermore the presence of toxic cyanobacteria may pose a health risk. During the summer of 1989, major blooms of toxic cyanobacteria were reported in many reservoirs in UK, Finland, Norway and Sweden. Frequent blooms are reported in the Netherlands and France as well.

Responses

Concern about elevated nutrient concentrations and adverse eutrophication effects has prompted the introduction of many reduction strategies at international, national and local levels. An important element of the strategies are the goals set, particularly where these are numerical targets, since these allow progress to be measured and the success of the policies adopted to be assessed. In addition to goals set internationally, some countries, or parts of countries, have adopted their own national targets, particularly in countries where eutrophication problems are at their greatest. These tend to be more specific, in that they focus immediately on areas of national concern. The Danish Action Plan, for example, focuses on reducing the loss of nitrate from agricultural land, setting a target of 50% reduction in 1987, and establishing policies on farming practices to achieve these targets. Policies were refined in 1991, 1996 and 1998 in the light of progress.

Measures to reduce point source discharges are focused on urban waste water treatment plants, and also on some key industries which can have a large contribution to nutrient discharges (production of fertilisers, food industry, etc.). At an international level, traditional command and control approaches are used widely. For example, common standards as well as common principles are adopted in some cases. At a national level, instruments such as charging schemes and voluntary agreements are being used to supplement the more traditional approaches.

[2] These areas are not necessarily protected by national legislation or other international conventions. The choice of areas implies, however, that these are the individual countries want to see protected from transboundary air pollution.

As described above, the nutrient load from agriculture represents a high proportion of the total anthropogenic load of nutrients to many waters, directly via run-off and leaching, and indirectly via ammonia and NOx emissions to air. However, reducing nutrient inputs from agriculture has been a difficult issue to tackle, from both a technical and political point of view.

Probably the single most significant policy to control nutrient inputs from products has been limitation of the use of phosphorus in detergents, for which several international measures have been introduced.

There are a number of international policies which set overall reduction targets for atmospheric emissions of NOx as a whole. For example, the Sofia Protocol (under the LRTAP Convention) requires participating countries to stabilise NOx emissions at the 1987 level by 1994. In addition, NOx emission limits and Air Quality Standards for the EU have been set for vehicles and combustion plants.

Measures to safeguard habitats and terrestrial ecosystems have focused on preventing inputs by controlling developments. The level of control is determined by the sensitivity of the areas. All areas of the EU are covered by the Environmental Impact Assessment Directive but additional measures exist in areas requiring special protection. For example, the Habitats Directive (92/43/EEC) requires that Member States designate a series of Special Areas of Conservation for habitats and species of 'Community interest' in which the habitats and species must be maintained in a 'favourable conservation status'.

Progress of control and reduction measures

The effectiveness of the policies to prevent or cure elevated nutrient levels is in general difficult to judge at a European scale because of the disparities in data concerning primary causes, emissions and ecosystem status. Therefore, in this report only a general and qualitative evaluation can be provided. There are, however, some notable exceptions to this general statement, where even at an international level, considerable efforts have been made to establish good and comparable information on sources, effects and inputs. Thus policy effectiveness can be assessed and fed back into the policy formulation process, hence completing the management loop. Good examples of approaches containing a strong information element are those adopted by the Paris and Helsinki Conventions. These can perhaps provide models for policy formulation in other parts of Europe.

Despite the general lack of information, it is possible to observe trends in the successes and failures of the systems, and the following general conclusions can be made:

- The application of nutrient reduction policies across Europe is 'patchy';
- A report by the European Commission indicates that six years after its adoption the status of implementation of the Nitrate Directive in most Member States is unsatisfactory;
- There are different deadlines for the application of the requirements of the Urban Waste Water Treatment Directive. However, the responses by Member States have also been slow and variable, so much so that infringement procedures against some have been instigated, by the European Commission, for no legal transposition of the Directive into national law or non-conformity of the legal transposition;
- Integrated polices have been developed for many areas of Europe, for example the North Sea, the Baltic Sea, the rivers Rhine and Danube. Other Europe-wide policies focus on specific sources;
- Application of integrated policies in the Paris Convention area were expected to result in a 50% reduction (25% in the case of France) of the input of phosphorus into nationally defined problem areas in the North Sea by 1995;
- However, the equivalent 50% reduction target for nitrogen were not expected to be achieved by 1995. Here the relevant North Sea states were expected to achieve reductions of between 20% and 30% into the potential problem areas;

- Although a similar approach was adopted for the Baltic Sea, under the Helsinki Convention, the lack of investment in some countries has meant that the overall targets will not be achieved until 2010 because of the lack of funds for policy implementation;

- Considerable success has been achieved in reducing nutrient inputs from point sources, in particular by targeting 'hot spots' – key emissions often responsible for significant inputs. An exception to this has been in central and eastern Europe, again mostly because of lack of investment;

- Agricultural inputs have been difficult to control. Even where integrated approaches have been adopted, as in the Helsinki and Paris Convention countries, limited success has been achieved, because of difficulties in implementation, and because of the time-lag between response and effect;

- Fertiliser usage has been reduced since the 1990's. In central and eastern Europe this has been largely due to economic reforms and recession, and in EU countries due to changes in the Common Agricultural Policy (CAP).

- Although policies are often formulated at a national level, the success of these measures depends on the extent to which they are applied and enforced. Thus, for example, the impact of directives such as the Urban Waste Water Treatment and Nitrate Directives is dependent upon the extent to which Member States designate sensitive areas and vulnerable zones, respectively.

Future actions

The European Commission's Fifth Environmental Action Programme discusses broadening the range of instruments to bring about changes in current trends and practices, and to involve all sectors of society. Whilst traditional command and control type measures still dominate the approaches used to control nutrient inputs, many interesting examples of alternative measures, which have been applied effectively, are reported for point sources. Examples include the charging schemes for discharges in Germany, France and the Netherlands, and for emissions to air in Sweden.

The European Commission has also proposed a Framework Water Directive (COM(97)49 final) which, when adopted, will have an overall environmental objective of reaching and/or maintaining 'good' status in all surface waters and groundwaters within a certain timescale (originally by 2010). Once implemented, the Directive should have a significant effect on reducing the impact of excessive nutrients on all surface waters.

These, and other approaches, may pose an effective way forward, particularly for agriculture as a way of overcoming the difficulties of effectively enforcing command and control instruments. For example, although the use of fertiliser taxes is not widely accepted, they have been used in Sweden, and studies suggest that doubling the price of fertiliser could reduce inputs by 25%.

Fertiliser and advice programmes were among the most successful approaches for controlling inputs from agriculture in the Paris Convention countries. One of the preferred measures for the Paris Convention's future programme is to balance fertiliser inputs on crop needs on a voluntary basis, although even this is proving to be politically a difficult agreement to make. There is also a requirement to develop 'good agricultural practices' under the Nitrate Directive.

Tradable quotas for fertiliser such as those adopted for milk production in the EU is another strategy which has not so far been adopted but might be considered. Such an approach has been used successfully in the USA for controlling diffuse sources.

These observations all reinforce the necessity of strengthening the European Environment Agency's EIONET for data exchange and for implementing the EUROWATERNET. The latter will enable the collection of relevant status information from those rivers, lakes, coastal waters and groundwater that are likely to be most affected by implementing the Urban Waste Water Treatment and Nitrate Directives.

Status information collected over a number of years will be related to changes in the pressures within the catchments, pressures which should be reduced by implementing the requirements of the Directive. Thus, once EIONET and EUROWATERNET are fully established, information will be provided at a European level by which the effectiveness of Directives, and other policies to reduce nutrient inputs, can be judged, and will also help to identify what other policies and practices might be required.

1. Introduction

1.1. Context and purpose of this assessment report

The first Dobris assessment report clearly identified excessive anthropogenic nutrients in all European ecosystems as a major environmental issue (EEA, 1995). The presence of excessive amounts of nutrients has direct and indirect effects on ecosystems and water uses.

Direct effects are observed when the concentrations of nutrients exceed standards or safe values for particular ecosystems. All nutrients are of potential concern and any effects are assessed using monitored concentrations. The assessment of indirect effects has been limited to eutrophication of water bodies.

Eutrophication of inland and marine waters deserves special attention as it significantly affects both the biodiversity and the beneficial uses and aesthetics of surface waters. Surface waters are widely used for domestic and industrial water supply: eutrophication may lead to unwanted tastes and odour problems. In this report, eutrophication is defined in its wider sense as an environmental adverse perturbation caused by an excess rate of supply of organic matter, including excessive primary production.

This Environmental Assessment Report provides an overview of the geographical distribution and severity of adverse biological effects of excessive anthropogenic inputs of nutrients into rivers, lakes and reservoirs, estuarine and coastal waters, as well as into sensitive terrestrial ecosystems.

The objectives of the report are to:

1. Evaluate the sources of excessive anthropogenic nutrients present in European ecosystems, with particular emphasis on phosphorus and nitrogen.

2. Assess the implications of excessive anthropogenic nutrient inputs into European ecosystems, in terms of observed impacts and estimated risk of nutrient enrichment problems.

3. Assess the past observed and future expected impacts of European, national and multi-national protection measures, including those laid down in the Urban Waste Water Treatment Directive (91/271/EEC) and the Nitrate Directive (91/676/EEC).

4. Evaluate the effectiveness of these protection measures against possible strategic goals.

For all of the assessments presented in this report, the authors have aimed to use the most up-to-date, validated, representative and comparable information available. When the available data and documented scientific evidence suggest possible gaps or uncertainties, these gaps have been indicated, especially if alternative policies could be derived from the interpretation of uncertain data.

1.2. Approach used in the report

The approach is broadly based on the DPSIR integrated environmental assessment approach (**D**riving Forces, **P**ressures, **S**tate, **I**mpact and **R**esponse). However, it should be noted that few European databases contain all of the required information for a full DPSIR assessment. For example, there are no databases on surface water status (this gap will be filled with the implementation of the EEA's EUROWATERNET (EEA, 1996; 1998a). The EEA database on large reservoirs (ELDRED) is not complete in relation to water quality issues (the development of ELDRED has been carried out in parallel with the report (EEA, 1999)). The same is true for lakes and small reservoirs. Thus a comprehensive DPSIR assessment is not feasible at this stage.

Data and information collection have been carried out in two ways. First of all questionnaires were prepared and forwarded to all of the EEA's contact points in all European countries. These questionnaires emphasised the importance of relevant national and regional reports, but also required data which would permit national and major catchment comparisons. The questionnaires comprised six sections dealing respectively with:

1. Pressures from municipal discharges,

2. State and trends of lakes and reservoirs,

3. State and trends in selected rivers,

4. Adverse effects observed in coastal and marine areas,

5. Documentation regarding terrestrial ecosystems,

6. Information describing policies and curative measures enforced by states.

The responses received by the ETC/IW, expressed in four classes according to their relevance for the report, are summarised in Appendix A.

Available published documentation was also sought to complement the data collection, especially where aggregated and interpreted data were only needed.

2. What are nutrients?

2.1. Definitions

A nutrient is 'a chemical compound (carbon, nitrogen, phosphorus, etc.) which can be used directly by living cells for nutrition or can be assimilated without prior digestion' (Encyclopaedia Universalis). Strictly speaking, the word 'nutrients' therefore refers to a large range of substances with variable concentration ranges, behaviour and effects on the environment.

This report deals with only those nutrients (also called 'macro-nutrients') that are important in terms of nutritive value for plants and nuisance levels. They are in practice phosphorus and nitrogen compounds. Those compounds that may limit the growth rate or biomass yield of aquatic plants will be given special attention. However, in order to illustrate the deliberately limited scope of this approach, a much wider range of compounds which can be considered to be nutrients in the three main ecosystems under consideration is briefly reviewed in the following sections.

In all the ecosystems under consideration, the nutrient requirements depend on the elementary composition of the living organisms in the ecosystem. The same chemical compounds are, therefore, found in all the ecosystems, albeit they are met in different proportions. The following sections describe the nutrients whose concentrations or bioavailability can be substantially modified by human activities.

2.2. Terrestrial ecosystems

It is well known that carbon is the fundamental nutrient for all living matter. The atmosphere is practically the exclusive source of easily assimilated carbon dioxide. The evolution of the atmospheric stock of carbon dioxide has been influenced by human activities, but as this falls into the field of climate change studies, it is outside the scope of this report.

In the soil, the major nutrients are the forms of nitrogen which can be used by plants (organic nitrogen, ammonium nitrogen, oxidised nitrogen). Although nitrate in solution in the soil is the form mainly used by nearly all terrestrial plants, the other forms of nitrogen can be transformed into nitrate by soil bacteria. The other major nutrients are phosphorus, calcium and potassium. The presence or absence of calcium in particular can significantly modify the composition of plant communities.

Although present at very low concentrations, micro-nutrients are also essential – these are, in no particular order, sulphur, silicon, metals (Cu, Mo, Mn, Zn, Co) and boron.

Many of these nutrients can have a positive or negative (even toxic) effect, depending on their concentration, but also on the proportions present in the ecosystem.

2.3. Surface fresh waters

Nitrogen, phosphorus and inorganic carbon (dissolved carbon dioxide and carbonic acid) are major nutrients in surface fresh waters. The presence of inorganic carbon is controlled by the calco-carbonic equilibrium. Calcium is, therefore, an important factor of carbon availability. Silicon is essential to diatoms, and determines the type of planktonic or epiphytic microscopic algae population present in a water body. Iron is also an essential nutrient, and can be present at very variable concentrations (Clasen and Bernhardt, 1974).

Unlike terrestrial plants, freshwater algae have a marked preference for ammonium nitrogen, which is energetically more efficient for incorporating into proteins. Aquatic macrophytes (filamentous algae and vascular plants) have similar requirements to benthic, epiphytic and planktonic unicellular algae. All major forms of nitrogen (ammonium, nitrite, nitrate, urea, etc.) are ultimately bioavailable. They can be incorporated by plants or are assimilated after bacterial mineralisation.

Phosphorus is more or less bioavailable from different particulate or dissolved compounds. Mineralisation, sorption onto suspended matter or sediment, and precipitation with calcium partly controls dissolved phosphate concentrations in fresh and marine waters.

The actual bioavailability of nutrients in surface fresh waters is extremely complicated to assess, because of varying levels of alkalinity, ionic balances, particulate matter and nutrient concentrations. For example, macrophytes with roots obtain their phosphorus from sediment, and their growth is, therefore, weakly related to available phosphorus in water. In contrast their epiphytic diatoms are more sensitive to the nutrients in surrounding waters.

However, the growth of phytoplankton of inland waters not affected by human influence is limited only by the lack of available phosphorus, when considering only nutrient impact.

2.4. Marine and coastal waters

The relatively constant composition of sea water tends to limit the number of nutrients with an influential role. Calcium, boron and potassium are present in significant quantities. The amount of bioavailable phosphate is determined by the speed of mineralisation of organic phosphorus, and by the exchange with suspended matter or sediment (which depends, for example, on particulate matter composition, pH, and Eh), levels of calcium, magnesium (Dauvin, 1997), pH and Eh. In practice the available phosphorus is, therefore, regulated and buffered to a large extent. This regulation makes nitrogen and silicon exert an important influence on marine phytoplankton and seaweed production. In coastal waters, nitrogen levels derive mainly from human-related loads (Dauvin, 1997). In contrast, silicon is quite independent of human activity and comes from fresh water inputs into coastal waters, or from recycling in open seas. In the sea, iron may also be a significant nutrient (Aubert and Aubert, 1986).

In contrast to the situation in the open sea, in many coastal lagoons or closed coastal zones major perturbations of the trophic cycle seem to be caused by the excessive production of organic matter stimulated by nutrient inputs (Nixon, 1995). In these cases carbon is a bacterial substrate which influences oxygen levels. It must be borne in mind that the oxygen, carbon, nitrogen and phosphorus biochemical cycles are closely inter-linked.

2.5. The choice of nutrients considered in this report

The previous sections have shown that the concept of ‘nutrients’ can include a wide range of compounds and can be interpreted in different ways. Nevertheless, the number of nutrients common to all ecosystems is quite small if one only considers them as a source of potential nuisance, and if their presence in large quantities is clearly related to human activities.

The choice of nutrients considered in this report was made considering the following criteria:

- nutrients which can cause direct or indirect nuisances by stimulating excessive plant growth; and,

- the concentrations of these nutrients (or their respective ratios) can be attributed to human activities.

Table 2.1 relates these two criteria to the most significant nutrients for the three main ecosystems under consideration.

Table 2.1 Qualitative assessment of nutrients in the three main ecosystems

Nutrient	Ecosystem					
	Terrestrial		Surface fresh waters		Marine and coastal waters	
	Effects or nuisances	Present due to human activities	Effects or nuisances	Present due to human activities	Effects or nuisances	Present due to human activities
Ammonium	Diversity (Biomass)	yes	Uses (Biomass)	yes	(Direct) Biomass	yes
Nitrate	Biomass Diversity	yes	Uses (Biomass)	yes	Uses Biomass	yes
Phosphorus	Accumu-lation (toxicity)	yes	Biomass	yes	(Biomass)	yes
Calcium	Diversity	sometimes	*(Biomass,(*) bio-availability)*	no	none	no
Potassium	Biomass	yes	negligible(**)	sometimes	none	no
Iron	?	no	Biomass Bio-availability	locally, (most frequently depends on bedrock composition).	Biomass	no
Mineral carbon (possibly including sources of organic carbon)	no (except heterotrophic organisms)	yes	Bio-availability, if other nutrients	(yes)	(Biomass) Diversity	

Legend:
- Biomass: an increase in the nutrient can result in an increase in biomass.
- Uses: an increase in the nutrient can result in an increase in disturbance of river or water uses.
- Diversity: an increase in the nutrient can modify ecosystem biodiversity.
- Bioavailability: change in the bioavailability of other nutrients.
- (*) in fresh waters, calcium is not a direct nutrient, but it controls the quantity of available carbon (thus often determining the final biomass) and it determines to a large extent phosphorus bioavailability. Iron (according to its oxidation level) and aluminium control phosphorus post-depositional mobility in sediments.
- (**) no effect as nutrient, locally may be involved in salinity problems.

3. Sources of nutrients

3.1. Introduction

Nutrient loading generally increases with increasing human activity in the catchment. Figures 3.1 and 3.2 illustrate the main aspects of the nitrogen and phosphorus cycles in the environment. During the last century, industrial production and household consumption have increased at a rapid rate, producing greater loads of nutrient-rich waste water. The extent to which this is discharged into surface waters depends on the proportion of the waste discharges connected to sewerage systems and on the sewage treatment facilities available, as well as the nutrient content of the item(s) produced or consumed. Activities within the agricultural sector have also changed with the result that intensive farm management practices are considered normal. Since the 1940s, the area of ploughed land and the use of commercial inorganic fertiliser have increased dramatically. In parallel, the sharp increase in livestock densities has resulted in the production and application of much greater loads of manure to cultivated land, and from there to surface waters. A great deal of the food used to grow livestock is imported from overseas, thus exceeding the potential recycling capacity of cultivated land. Higher livestock densities also result in greater atmospheric emissions of ammonia which, in turn, lead to greater deposition of nitrogen to land and surface waters.

Rain may add substantial amounts of nitrogen to marine waters enhancing eutrophication.
Photo: Svend Tougaard/BIOFOTO

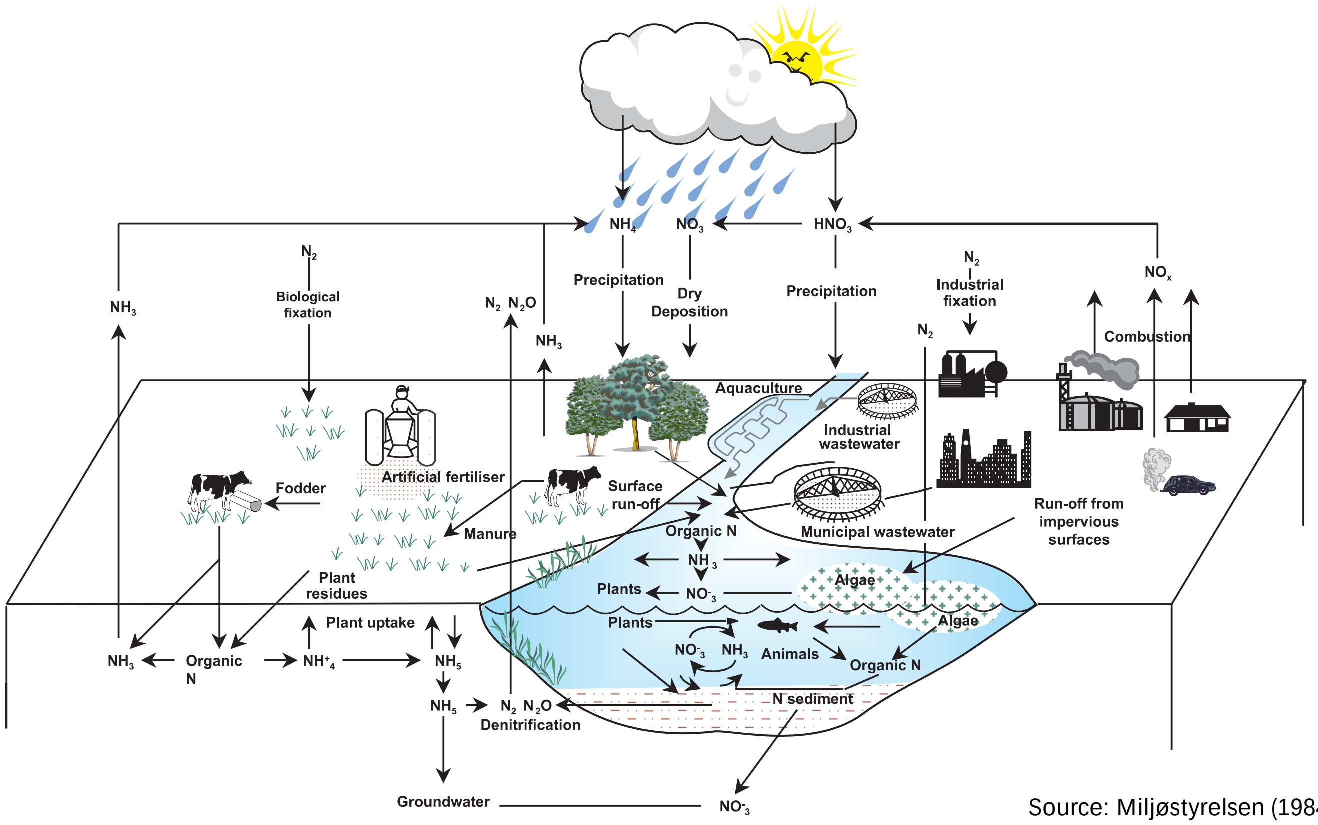
Both increases in chemical fertiliser use, and in animal manure to be disposed of, constitute a potential source for run-off of nutrients to inland waters. Changes in arable farming practices have also increased the rate of soil erosion, with a related increase in phosphorus run-off. In many areas much of the agricultural land is drained and a large number of Europe's marshes, wetlands, ponds and lakes have disappeared. This has considerably reduced the capacity of freshwater ecosystems to store and eliminate many contaminants, including nutrients.

Most of the phosphorus loading to inland surface waters is attributable to discharges from point sources, especially municipal sewage and industrial effluent, whilst nitrogen loading is primarily from agricultural activity, especially from the use of nitrogen fertilisers and manure. Dissolved phosphorus concentrations in some European surface waters have decreased during the last 10 to 20 years as a consequence of improved waste water treatment and the substitution of phosphorus in detergents. In contrast to phosphorus, the nitrate level in many European rivers has increased during the last 10 to 20 years, mainly as a result of increasing or high use of nitrogen fertilisers (Kristensen and Hansen, 1994), and intensification of crop production of thin and fragile soils.

3.2. Driving forces and pressures

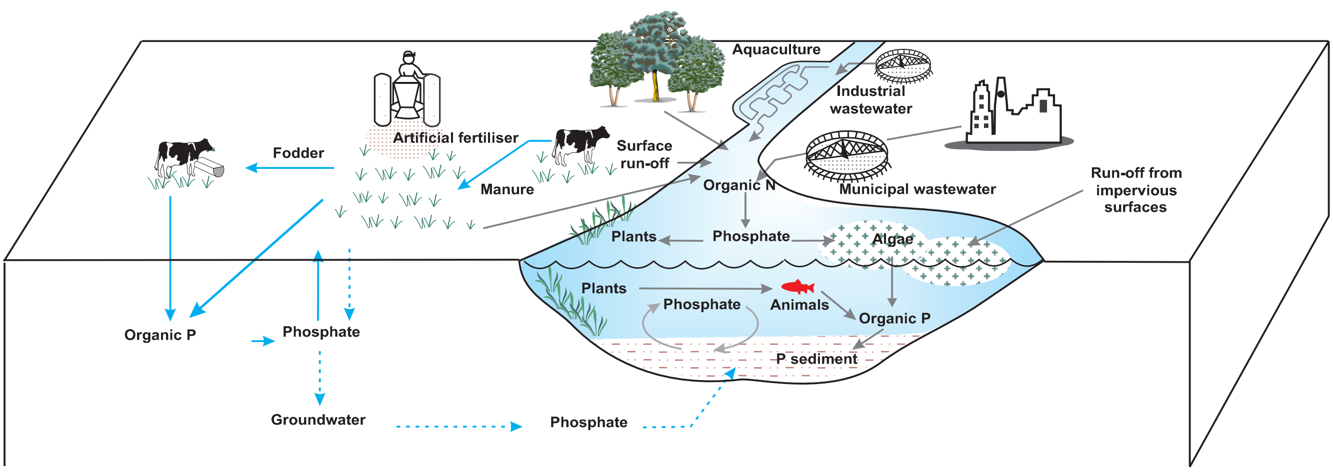
The pressures on the environment are the result of production or consumption processes which, in a macro-economic context, are structured according to economic sectors such as agriculture, energy, industry, transport and consumers. Several economic activities in society are responsible for substantial loadings of nutrients to the environment. At the European scale important anthropogenic sectors are agriculture,

Figure 3.1 Illustration of the nitrogen cycle in the environment.



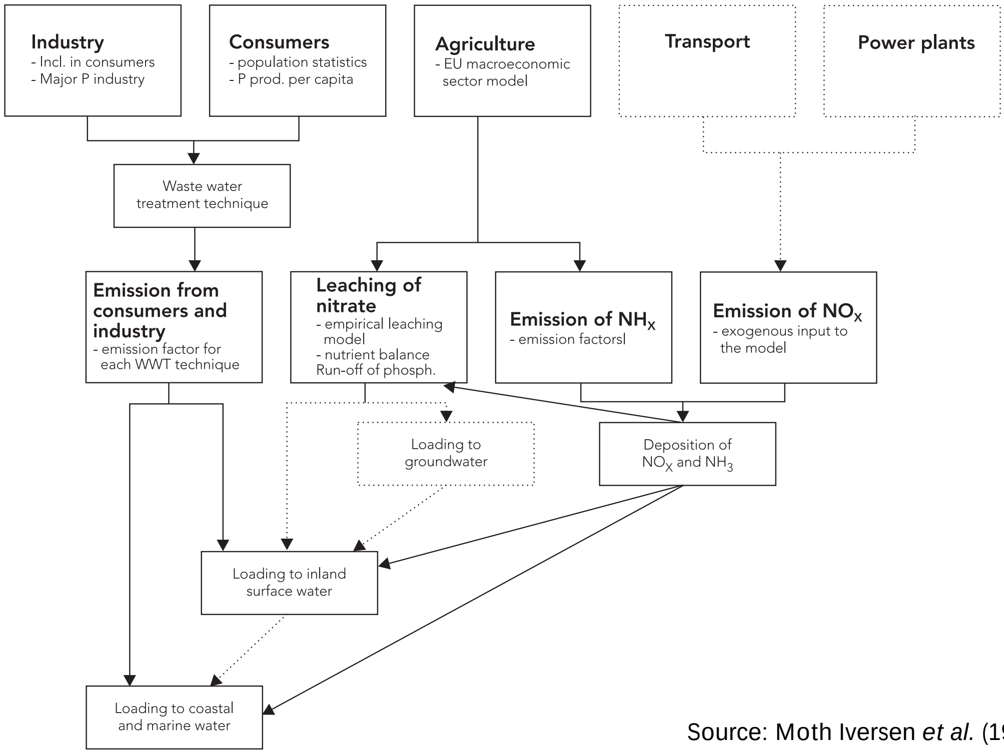
Source: Miljøstyrelsen (1984)

Figure 3.2 Illustration of the phosphorus cycle in the environment.



Source: Miljøstyrelsen (1984)

The basic framework of the driving forces/pressure relationship of the assessment. **Figure 3.3**



Source: Moth Iversen *et al.* (1997)

Note: The Transport and Power plant sectors are only described briefly, and the groundwater pathway is not included in the assessment

consumers, industry, energy production and transport: most of these are key sources of nutrients to the environment (EEA, 1998b). At a more local or small catchment level other anthropogenic factors can be of importance, for example, tourism or fish farming in lakes, small rivers or enclosed coastal areas. The driving forces-pressure relationship in the DPSIR integrated environmental assessment framework is illustrated in Figure 3.3.

The environmental *pressures* resulting from *driving forces* are a function of two types of variable: first the *level* of these activities and secondly the *technology* used in these activities. The description of pressures thus needs to take into account these two types of variable.

The variables accounting for the *level* of activities are economic in nature, reflecting the level of production and consumption. However, in some cases the relationship between the level of the economy and production is rather weak, e.g. the amount of phosphorus per capita in waste water is not fully related to the economy; the quantity due to human metabolism is quite constant across Europe. The *technology* variables will be reflected by emission factors; for example, empirically estimated relationships between fertiliser application and nutrient leaching, or emission factors depending on the purification process applied in waste water treatment plants.

In the following sections a brief description of the major sectors, their level of activity and technology involved is provided.

3.3. Consumers and industry

The European population has increased markedly during this century. The urban population has increased at a faster rate during the same period – from less than one-third to more than two-thirds of the population. Tap water facilities have been installed at a high rate, as well as sewerage systems that have been constructed to transport the increasing volume of sewage from the dwelling areas. For a long time, the only purpose of that was the control of water borne diseases. Natural waters were supposed to self-purify, whatever the polluting load they received. This has resulted in extremely large quantities of waste water, and hence of nutrients being disposed of to surface waters, and, in some circumstances to groundwater. Because of the lack of efficiency in sewage management and purification in many areas of Europe (see Figure 3.6), the majority of this waste water is discharged into surface waters with little or no pollution reduction. The disposal of urban waste water is, therefore, an important threat to surface waters.

In densely populated areas most of the phosphorus loading to surface waters is derived from human waste, phosphorus production being 1 to 1.5 kg P capita^{-1} year^{-1} in industrialised countries (Jones *et al.*, 1979). The extent to which this is discharged into surface waters depends on the level of sewage treatment. Sewage treatment plants (primary treatment and classical activated sludge treatment) removes only a minor part of the phosphorus from waste water, whereas plants with specific biological treatment and chemical precipitation of phosphorus may remove more than 95%. Waste water treatment plants designed for phosphorus removal have been constructed in Europe over the last 30 years, with an increased rate for the last 15 years, especially in the Nordic and western European countries. However, at present the majority of European waste water treatment plants have only limited ability to remove phosphorus. At the same time many countries have lowered the phosphorus content of detergents, thereby lowering phosphorus loading of surface waters.

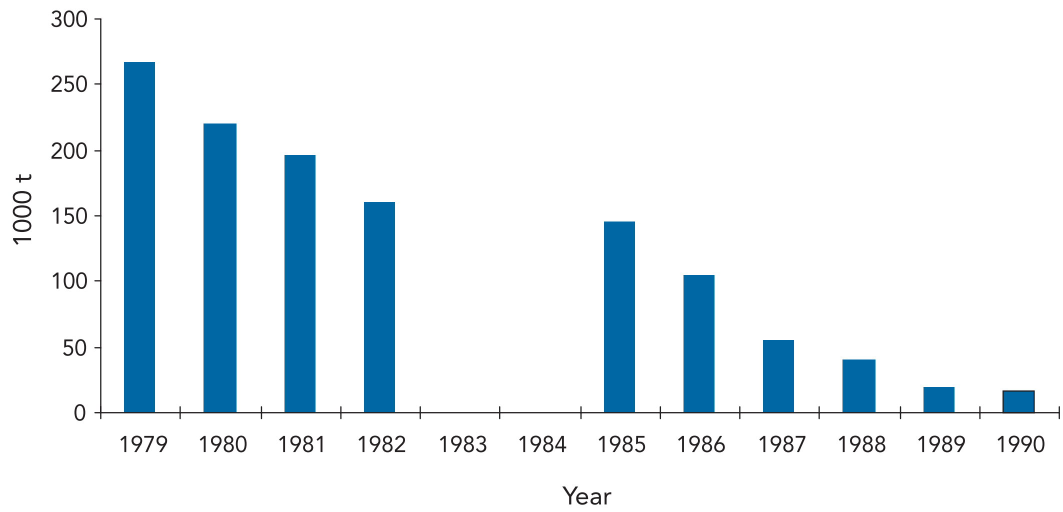
The nitrogen production derived from human waste products is estimated to be in the range 2.2 to 5.5 kg N capita^{-1} year^{-1} in European countries (EEA, 1995 and IFEN personal communication). Waste water treatment plants incorporating nitrogen removal have also been constructed in Europe over the last 15 years, but to a lesser extent compared to phosphorus removal. Generally nitrogen loads from waste water make up less than half of the total nitrogen load to surface waters. However, part of the nitrogen output from urban waste water treatment plants is in the ammonium form, which is especially harmful to aquatic life and human uses. Moreover, it contributes to the oxygen demand in river water, since it is readily oxidised into nitrate in natural waters.

3.3.1. *Level of activity*

3.3.1.1. Households

The amount of waste water produced is proportional to the population. The phosphorus content of waste water from households is primarily determined by excreta from humans and phosphorus from deter-

Figure 3.4 **Consumption of polyphosphate in detergents – Old German Länder.**
Source: Umweltbundesamt (1994)



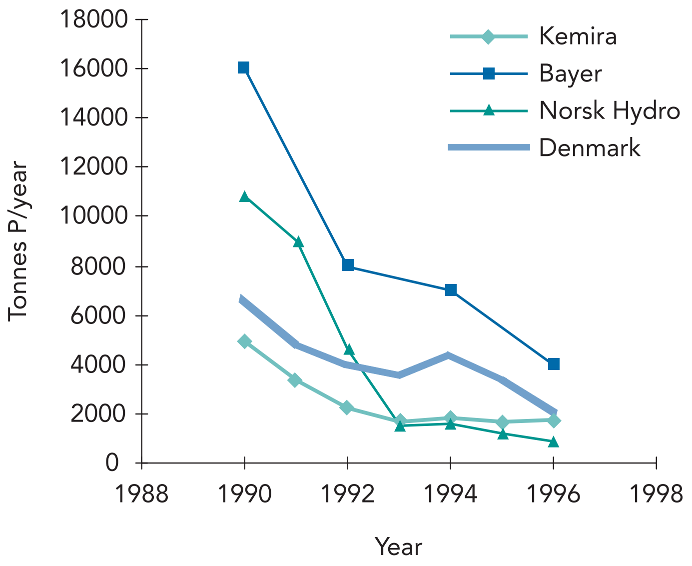
gents. Over the last 10 to 15 years the phosphorus content in detergents has been markedly lowered in many countries. For example, in the Old German Länder, phosphorus loading in household detergents was reduced by 94 % between 1975 and 1990 (Figure 3.4).

3.3.1.2. Industry
Large industrial facilities, in particular producers of phosphorus fertilisers, often emit quantities of phosphorus that are of the same order of magnitude as the total emissions from small countries (Figure 3.5). Emissions from such facilities have significantly decreased during the 1990's due to improved technology and waste water treatment.

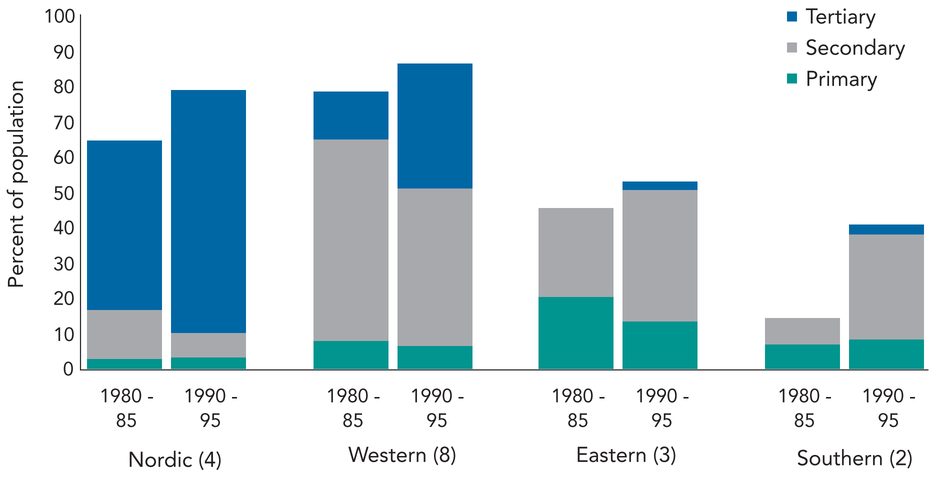
3.3.2. Technology variables

The extent to which the nutrients in waste water from industries and consumers are discharged into surface waters depends on the waste water treatment facilities available. Traditional waste water treatment plants were designed primarily for the reduction of organic matter, and the nutrients load was largely unaffected. Modern waste water treatment greatly increases the removal of nutrients. Today the percentage of the population served by waste water treatment varies from about 50 % in southern and eastern Europe to about 80 % in Nordic countries and western Europe (Figure 3.6). The treatment of municipal

Phosphorus emissions from large industries. Total emissions from Denmark added for comparison. Source: Industrial WWW homepages **Figure 3.5**



Waste water treatment in regions of Europe between 1980/85 and 1990/95. Only countries with data from both periods included for the temporal change analyses, the number of countries in parentheses. Sources: Eurostat (1995) and ETC/IW questionnaires **Figure 3.6**



Box 3.1 Definition of European regions

Unless otherwise indicated regional analyses have been undertaken in this report according to the following groupings of countries:

- Nordic (NO): Finland, Iceland, Norway, and Sweden.
- Eastern (EA): Belarus, Bulgaria, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Moldova, Poland, Romania, Russian Federation, Slovak Republic, and Ukraine.
- Southern (SO): Albania, Bosnia-Herzegovina, Croatia, Cyprus, Greece, Italy, Malta, Portugal, Federal Republic of Yugoslavia, Slovenia, Spain and the Former Yugoslav Republic of Macedonia.
- Western (WE): Austria, Belgium, Denmark, France, Germany, Ireland, Liechtenstein, Luxembourg, the Netherlands, Switzerland, and UK.

waste water in Europe has significantly improved during the past 10 to 15 years, especially in southern Europe. A larger proportion of the population has been connected to treatment plants and the treatment level has changed. In eastern and southern Europe, there has been a pronounced change from primary (mechanical) to secondary (biological) treatment. In western Europe and Nordic countries, the introduction of tertiary treatment, most often with phosphorus removal[3], has been a predominant trend over the last decade.

3.4. Agriculture

Agricultural activities can produce run-off (leaching and erosion) of nutrients from the land area used for plant production. In addition, emissions of ammonia to air result in deposition of reduced nitrogen on

Modern waste water treatment plants remove large proportions of nutrients from the sewage. Photo: K-A Larsson/BIOFOTO

[3] Some tertiary treatment processes aim only at nitrogen removal or disinfection.

surface waters and on land, which in turn may be washed out into surface waters.

3.4.1. Level of activity

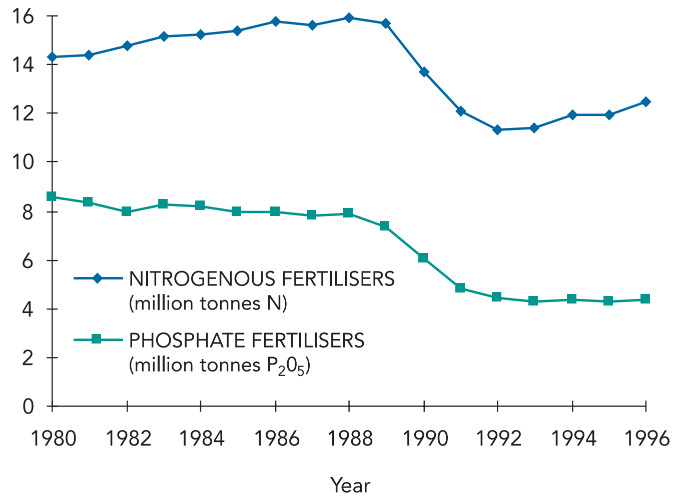
The productivity of the agricultural sector has increased markedly in this century. Today more than 30% of the land area of Europe is used for agricultural production, although there are large regional differences in the percentage of farm land, farming intensity and type of crops grown. For example, agricultural land constitutes about 81% of the total land area of Ireland, 65% of Denmark and 8% of Sweden. However, whilst 60% of the agricultural land area is arable in Denmark, only 18% is arable in Ireland (most of Ireland's agricultural land is used for grazing).

3.4.2. Technology variables

3.4.2.1. Fertiliser application – nutrient balances

Nutrient application to agricultural land includes inorganic fertiliser and manure. Whereas the consumption of nitrogen fertiliser generally increased from 1970 to 1988, the consumption has decreased in recent years (Figure 3.7). There are large variations in Europe from about 180 kg N ha^{-1} agricultural land in the Netherlands in 1991 to 30 to 70 kg N in southern Europe. The use of phosphorus fertiliser in Nordic

Development in the consumption of artificial fertilisers in Europe. Source: FAO (1996) **Figure 3.7**



Locating dung heaps close to water bodies is not good agricultural practice. Photo: André Maslennikov/ BIOFOTO and western Europe has generally decreased since 1970 to a level of 20 to 50 kg P ha^{-1} of agricultural land. In southern Europe the consumption of phosphorus fertiliser has remained relatively stable at a level of 10 to 30 kg ha^{-1}.

Similarly, the application of manure to agricultural land varies widely between European countries from >200 kg N and >100 kg phosphate (as P_2O_5, i.e. 44 kg P ha^{-1}) per hectare agricultural land in the Netherlands to <40 kg N and <20 kg phosphate (as P_2O_5, i.e. 8.8 kg P ha^{-1}) in southern Europe.

Nitrogen balance studies on agricultural land in the EU12 countries have shown that the surplus (difference between input and output) varies from over 200 kg N ha^{-1} $year^{-1}$ in the Netherlands to about 15 kg N ha^{-1} $year^{-1}$ in Austria (Figure 3.8). In general, there is an increasing surplus with increasing inputs, reflecting increasing potential leaching with increasing inputs.

3.4.2.2. Leaching of nitrate

In general, higher nutrient loads are applied than the quantity incorporated with crops, which in turn is greater than the quantity finally harvested and exported. A large part of this nitrogen surplus leaches out of the root zone either to the groundwater and later to surface waters, or directly to surface waters, for instance through drains. Nitrate leaching from the root zone is generally a function of natural factors, such as soil type and climate, and anthropogenic factors, such as crop type, the level and type of applied fertiliser (commercial fertiliser, manure, sewage sludge), and the utilisation regime of manure (e.g. spreading of manure in autumn or spring).

Run-off of melting water and slurries from fields adds nutrients and oxygen consuming substances to water bodies. Photo: Elvig Hansen/ BIOFOTO

3.4.2.3. Phosphorus run-off

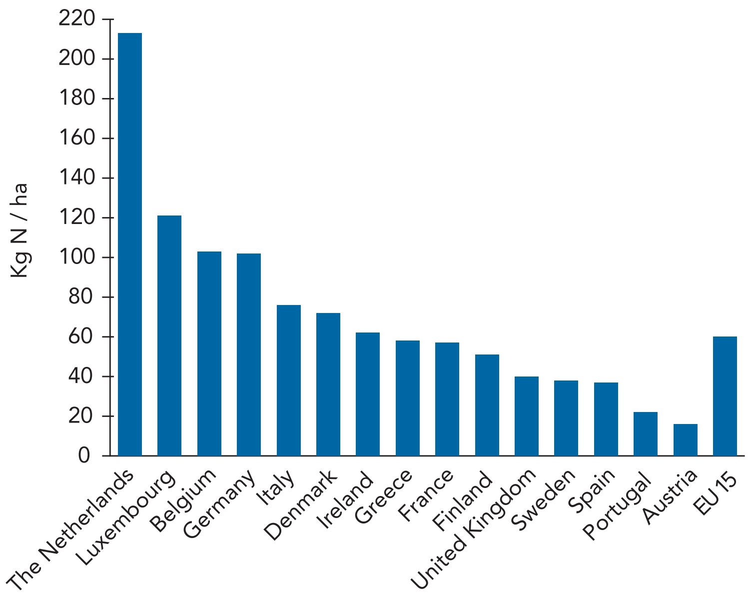
Phosphorus in soil is bound in inorganic and organic phosphates or adsorbed to oxides and hydroxides, clay minerals and organic matter. However, in the Netherlands, for example, the high amounts of phosphorus applied for many years have resulted in about 20% of the agricultural land, mainly sandy soils, being saturated with phosphorus and hence increasing significantly leaching of phosphorus. Such excess occurs in other places, such as in Brittany, France, where excess in phosphorus fertilisation results from the application of mineral fertiliser and manure. Similarly, very high levels of phosphorus (>1 mg l^{-1}) have recently been reported in the major aquifer lying beneath the Po valley flood plain, Italy. Here, the major source of the problem appears to be phosphorus fertiliser applied to overlying rice fields.

3.4.2.4. Emission of ammonia

Ammonia emissions arise mainly from the production and spreading of animal manure and are, therefore, closely related to livestock population. Historic trends in livestock populations present a mixed picture (Figure 3.9). The overall livestock population increased considerably over the period 1961 to 1996, especially pigs. Cattle numbers decreased. Sheep and goat populations decreased at first, then increased during the 1980s, and declined in the 1990s. This largely reflects changes in consumer preferences. The decrease in cattle numbers is also likely to be due to the first adjustments in the common market organisations introduced in the mid and late 1980s, particularly milk quotas. The distribution of livestock (heads ha^{-1}) is very unbalanced across Europe, with the highest concentrations in north-western coastal areas.

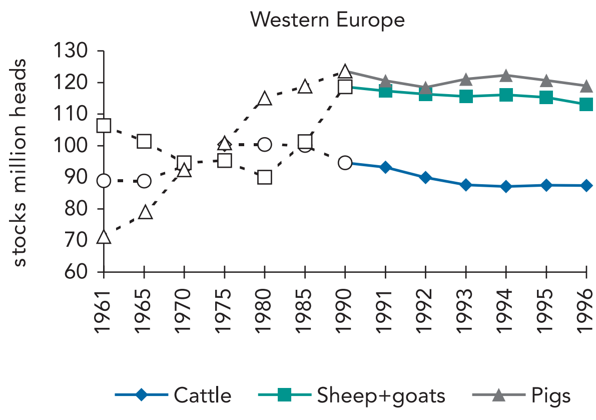
Atmospheric emissions of ammonia (NH_3) in the period 1980 to 1990 have been more or less constant. From 1990 to 1995 the emissions of NH_3 in Western Europe decreased by about 8%. Agriculture is by far the largest contributor to atmospheric emissions: estimates suggest that over 90% of total emissions are from agricultural activities.

Average nitrogen surplus (difference between input by atmospheric deposition, fertiliser and manure and output by harvested crops). Source: Eurostat (1997) **Figure 3.8**



A. Trend in number of livestock: cattle, pigs,sheep and goats and B: Emission of ammonia in Western Europe. Sources: FAOSTAT and Tsyro (1997) **Figure 3.9**

A



B

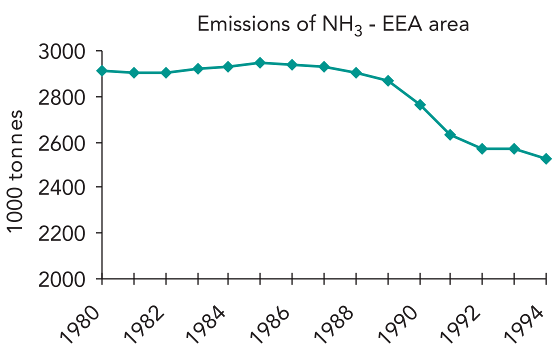
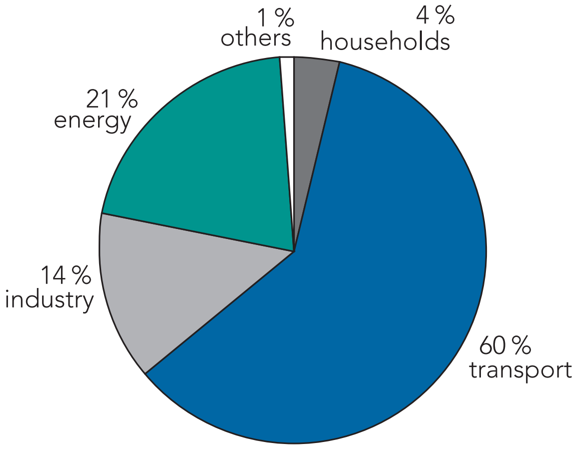


Figure 3.10 **Atmospheric emissions of NO_x per sector, 1994/95. Source: EEA (1998)**



3.5. Emission of nitrogen oxides from transport and power plants

Nitrogen oxides (NOx) are a by-product of combustion of fossil fuels. Similarly to ammonia (from agriculture), the deposition of nitrogen oxides to land and surface water may contribute to increased N loads and eutrophication.

The main source of NOx is the transport sector accounting for about 60 % (Figure 3.10). The continued growth in total goods transported in western Europe between 1985 and 1995 has been accompanied by a growth in road transport (EEA, 1998). Only 17 % of the goods transported is now by rail. In eastern Europe the total goods transported dropped rapidly as a result of economic restructuring, but as in western Europe, a growing proportion is transported by road. Also passenger transport has increased. In the EU, there has been an increase of 82 % in air transport and 46 % by car. The increasing pressure arising from car transport has to some extent been reduced by the introduction of catalysts for petrol cars.

The emissions of NOx within the EMEP (Co-operative Programme for Monitoring

Marine aquaculture, Salmon production on the Faeroe Islands. Photo: Karsten Schnack/BIOFOTO and Evaluation of the Long Range Transmission of Air Pollution in Europe) area were relatively stable until 1990, but since then there has been a 15% decline in total emissions (Figure 3.11). In western Europe the decline has only been 8%, whereas in eastern Europe it has been 29%.

3.6. Fish farming

Fish farming produces effluents rich in nutrients. In watersheds heavily stocked with fish farms, ammonia can reach high concentrations (potential toxic effects). In semi-enclosed coastal areas benthic organic enrichment has been observed if local hydrodynamic conditions do not allow favourable dispersion.

3.6.1. Level of activity

Fish farming, especially cages in sheltered coastal areas, has evolved rapidly over the last 15 years because of the development of new technologies (offshore cages) and a better market distribution network for the fish produced.

3.6.2. Technology variables

The outflow of nutrients from inland fish farms can generally be controlled by settling ponds and by reducing excessive use of fish feeds. Cages and net-enclosures can be equipped with sludge-collection funnels to reduce their pollution effects.

In land based fish farms, there have been noticeable changes in food composition, which has resulted in lower ammonia excretion by trout.

3.7. The significance of different sectors

3.7.1. Phosphorus

In relatively sparsely populated areas with relatively low agricultural activity, such as in Sweden, only around 50% of the phosphorus loading is related to human activities (generally point sources and including industrial sources). Half of the loading is derived from diffuse run-off from undisturbed land. In these areas, atmospheric deposition may contribute significantly to phosphorus loading. This is, for example, the case in Sweden where lakes represent 7% of the total area. Here, atmospheric deposition to inland surface waters is responsible for a relatively high proportion of the phosphorus budget compared to other catchments.

Atmospheric NOx emissions in Europe, 1980 to 1995. Source: EEA (1998) **Figure 3.11**

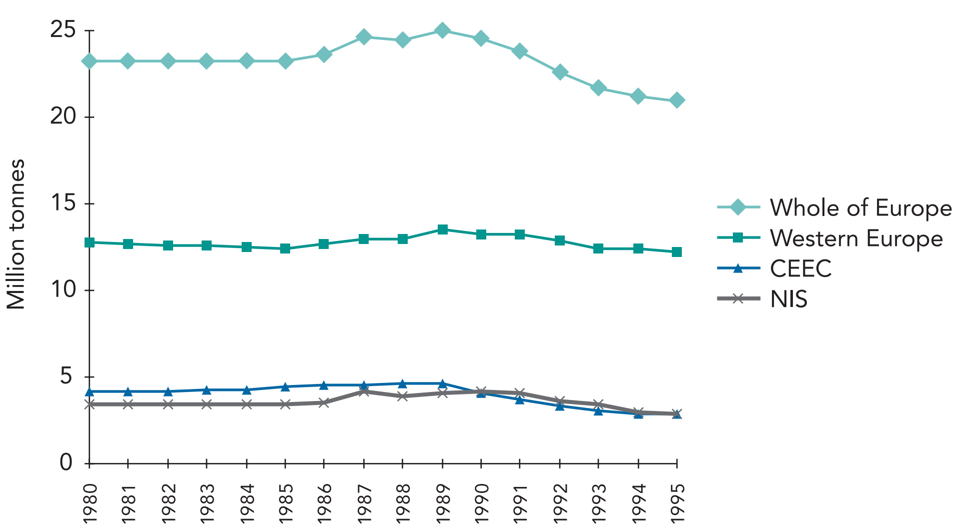


Table 3.1 **Apportionment of N and P budgets for the Po catchment. Source: WRc/TEI (1997)**

Source	% N budget	% P budget
Civilian population	21.3	43.8
Industry	10.4	6.2
Livestock	22.4	12.5
Fertiliser application	27.3	24.1
Atmospheric deposition (N)	18.6	
Natural P load		13.4
Total	100	100

Phosphorus loading generally increases with increasing human activity in the catchment. In the more densely populated areas (Figure 3.12), 50-76% of the dissolved phosphorus load to inland waters is derived from point sources, while agricultural activity generally accounts for 20-40%. If there was no human activity, phosphorus levels would only be 5-10% of current levels. In these densely populated catchments, municipal sewage discharges generally account for the major part of the total point source discharges. However, in the Dutch part of the Rhine catchment, industrial effluents account for more than 75% of the point source discharge (RIVM, 1992), even though the agricultural sector remains the major overall contributor to the total load.

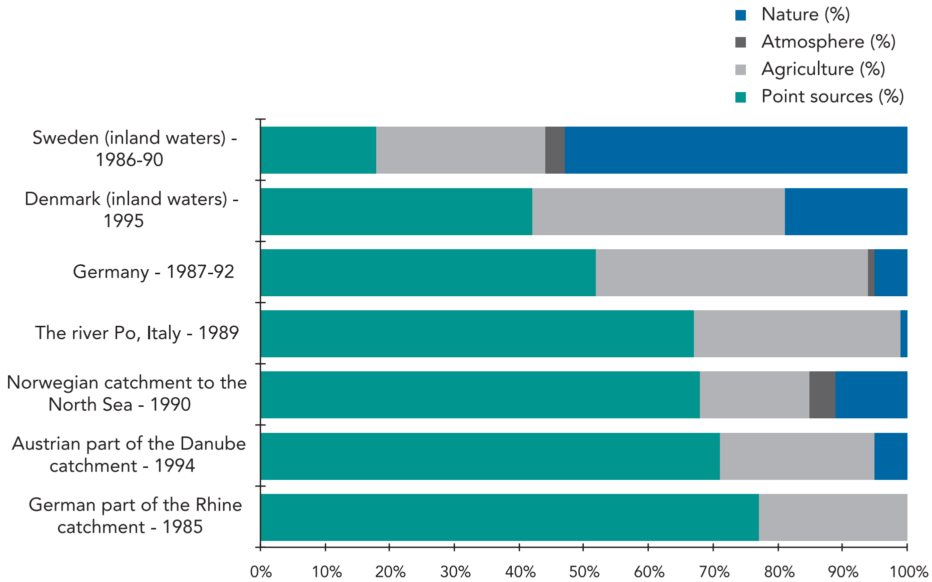
More detailed data are available from a modelling exercise for the Po catchment, Italy (Table 3.1). These results (or at least the total loads) were validated against measured in-river loads wherever possible, and lie within the 90% confidence limits calculated at a number of flow gauging/ water quality sampling stations. The results are based on the Italian (ISTAT) 1990 agricultural census data.

Emissions of phosphorus are decreasing in many parts of Europe. Results from large river catchments or national emission inventories show that there has been a reduction of typically 30-60% since the mid-1980s (Figure 3.13). The emissions from the industrial sectors of Denmark and the Netherlands have decreased by as much as 70-90%. Nevertheless, the anthropogenic contribution to phosphorus loading is generally far beyond the natural load in most parts of Europe. To reduce eutrophication, further reductions are needed in phosphorus emissions, both from point and diffuse sources.

3.7.2. Nitrogen

Nitrogen pollution is usually dominated by diffuse sources, in particular agriculture (Figure 3.14). In those river systems draining catchments in the central and western part of Europe, 46-87% of the nitrogen load to inland waters is related to agriculture. In some catchments point sources of

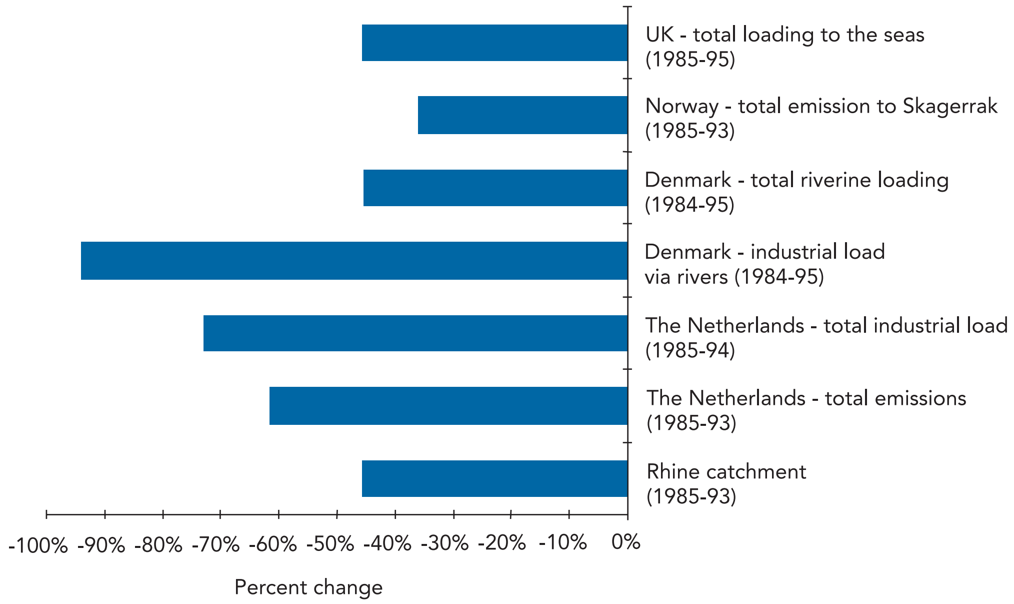
Figure 3.12 **Source apportionment of phosphorus load. Source: compiled by ETC/IW from state of the environment reports, Windolf, 1996; Swedish EPA, 1994; Umweltbundesamt, 1994; BMLF, 1996; Ibrekk et al., 1991; Italian Ministry of the Environment, 1992**



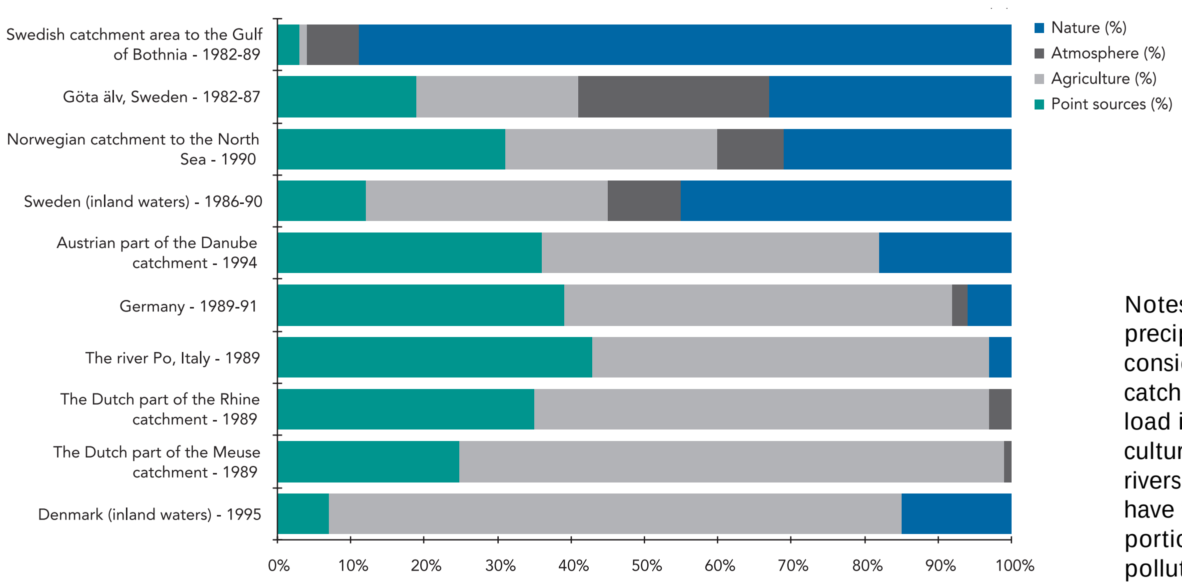
Note: Atmospheric contribution considered only for some catchments. The lower bars have the highest proportion of point source pollution.

nitrogen (predominantly municipal sewage treatment plant) also play an important role, accounting for 35-43% of the total discharge. In the two Swedish example catchments the differences in percentage of agricultural land and population density are reflected in the nitrogen budgets. In the Göta älv catchment area, where about 10% of the land is cultivated and the population density is about 30 inhabitants km^{-2}, human activities account for 41% of the nitrogen discharge (some of the atmospheric deposition can also be related to human activities). In the Gulf of Bothnia catchment area, where about 1% of the land is cultivated and the population density is only 1-3 inhabitants km^{-2}, most of the nitrogen discharge is related to diffuse run-off from forested and uncultivated areas.

Changes in phosphorus loading since the mid 1980s. Sources: RIVM, 1995; Miljøstyrelsen, 1996; Windolf, 1996; SFT, 1996; WRc, 1997 **Figure 3.13**



Source apportionment of nitrogen load. Source: compiled by ETC/IW from state of the environment reports: Windolf, 1996; Swedish EPA, 1994; Umweltbundesamt, 1994; BMLF, 1996; Ibrekk *et al.*, 1991; Italian Ministry of the Environment, 1992; RIVM, 1992; Löfgren and Olsson, 1990 **Figure 3.14**



Notes: Atmospheric precipitation only considered for some catchments. Natural load included in agriculture for the Dutch rivers. The lower bars have the highest proportion of agricultural pollution

4. What constitutes an excess of nutrients?

4.1. A working definition

In Chapter 2, the nutrients that can pose a risk for each type of ecosystem, or which can arise through human activities, were described. Chapter 3 outlined the main sources of the two principal nutrients (nitrogen and phosphorus) considered in this report. In this chapter, what constitutes an excess of nutrient levels is considered. It should be noted that later chapters examine the principal nutrients in each of the main ecosystems. All data related to nutrients in this report are expressed as the main element (for example: N-NO_3, P-SRP, etc.) unless otherwise stated.

The aim of defining the idea of 'excess' is to provide a reasonable basis for data evaluation. In order to do this, reference values are necessary, as are graded scales of values. An excess is thus a measure of the difference between the observed levels and the established reference levels.

If this reasoning is pursued, a measure of nutrient excess from human origins can be defined as

'A nutrient load supplied in such a quantity as to result in:

- *direct effects which can be immediate or predictable (due to nutrient accumulation) and which can be observed directly in the ecosystem or in the river/water uses; and/or*

- *an increased production of biomass which exceeds the recycling (aerobic mineralisation) capacity of the ecosystem.'*

As a first approach it can, therefore, be considered that there is an excess of nutrients when the natural (or slightly disturbed) concentrations are significantly and constantly exceeded, and that this causes a nuisance or an ecosystem modification.

Agricultural crops should be excluded from this environmental approach. The scope of this report does not include terrestrial or aquatic ecosystems which have been modified in order to increase production. It should be noted, however, that these ecosystems are often the source of excessive nutrients for disturbed natural ecosystems. These types of human-modified ecosystems and the natural ecosystems are often geographically interwoven and environmentally interdependent.

The impacts of excessive nutrients are either manifested as a direct effect (for example, the nutrient acting as a toxin) or through an eutrophication effect (where the nutrient acts directly on the trophic web).

4.2. Excessive nutrients with direct effects

Under certain circumstances, each of the nutrients included in Table 2.1 can exert a direct negative impact, which are described in the following paragraphs. It is important to note that the concentrations producing these impacts can be quite different from those inducing eutrophication effects.

4.2.1. Ammonium

Ammonium loads to soils (fertilisers, sewage, atmospheric deposition) do not result in a specific direct effect. In soil ammonium is either fixed to clay minerals or organic matter, converted into nitrate by microbes or taken up directly by plants and micro-organisms. Small amounts are lost by volatilisation as ammonia gas. Thus like organic nitrogen, ammonium is not very mobile in soils and, therefore, contributes to the soil's fertility as total nitrogen.

In contrast, ammonium in fresh waters is in equilibrium with its dissolved (un-ionised) gaseous phase, the proportion of which depends on the water's pH, temperature, and total dissolved solids (TDS). An increase in TDS reduces the proportion of un-ionised ammonia.

In fresh waters, a practical assessment of the un-ionised fraction (NH_3) can be deduced from tables (Train, 1979) or alternatively from a simple equation in relation to pH and temperature (IFEN, personal communication), as follows.

proportion in $NH_3 = 1/(1+10^{(10-pH-0.03t)})$,
where t is the temperature in °C.
Equation 1

In this report, no ammonium data in relation to TDS were available. Therefore, Equation 1 has been used to estimate the un-ionised fraction from measured ammonium concentrations, temperature and pH. Un-ionised ammonia is the most toxic form to aquatic organisms. Ammonia becomes toxic for aquatic animals at concentrations of 0.2 to 2 mg l^{-1} (WRc, 1988). In most inland waters, the pH range results in the ionised fraction (NH_4^+) dominating. In alkaline waters, or in waters where the pH is significantly raised by eutrophication, toxic values of ammonia may occur. The recommended values are, therefore, calculated allowing for a certain margin of safety.

The Directive 78/659/EEC *'on the quality of fresh waters needing protection of improvement in order to support fish life'* prescribes guideline and limit values for ammonia of 0.005 and 0.025 mg l^{-1} respectively, for salmonid and cyprinid waters. The same Directive also provides guideline and maximum limits for total ammonium of 0.04 and 1 mg l^{-1} for salmonids and 0.2 and 1 mg l^{-1} for cyprinids, respectively.

Total ammonium directly impacts drinking water production, since it reacts with the chlorine added to oxidise organic matter. Chloramines are formed so that higher concentrations of chlorine are required to reach an effective level. The fact that ammonium is not destroyed by chlorine dioxide nor by ozone (both reagents that are widely used as an oxidiser in tap water preparation) means that low ammonium concentrations have been defined in the two relevant EEC drinking water directives: Directive 75/440/*EEC (quality required of surface water intended for drinking water abstraction*) and 80/778/*EEC (quality of water intended for human consumption*). The maximum admissible concentrations of ammonium (MACs) defined in these Directives are 1.5 mg l^{-1} for untreated waters and 0.5 mg l^{-1} for treated waters. From this, it is obvious that an ammonium concentration in the order of 4 mg l^{-1} indicates a significant excess**.**

In sea water, the presence of ammonia is also dependent on the pH and the salinity. At normal temperatures and salinity, ammonia represents about 3-5% of total ammonium (Spotte and Adams, 1983). There do not appear to be general data concerning international standards for ammonium in sea water though some reference values have been proposed in some countries (e.g. UK).

4.2.2. Nitrate

In terrestrial ecosystems, nitrate is generally found in the dissolved form in soil porewaters. In this form, it contributes to the soil's fertility, since it can be easily assimilated through the roots of most plants. Since some plants (e.g., *Solanaceae*) prefer ammonium to nitrate, the concentration of nitrate in soils and the ratio $N\text{-}NH_4/N\text{-}NO_3$ are, therefore, major factors determining the composition of plant communities. In addition, inorganic nitrogen inhibits symbiotic fixation by leguminous plants and tends to eliminate oligotrophic species. An abundance of nitrogen allows nitrophilic plants (such as nettles) to dominate and large excesses lead to disease in grasses, which represent the major crops. The nitrogen cycle in plants and soils is highly complex and further details can be found in the terrestrial ecosystem chapter.

The range of concentrations of nitrate observed in ecosystems does not pose a toxic risk to aquatic animals (Nixon, 1995). In highly polluted ecosystems, nitrate is reduced and can partially compensate for the lack of oxygen needed to oxidise organic matter. Nitrite is a transitional form of nitrogen (between ammonium and nitrate) and is highly toxic even at low levels. The EC Freshwater Fish Directive 78/659/EEC stipulates guide values of 0.01 and 0.03 mgl^{-1} NO_2^- respectively, for salmonids and cyprinids. These recommended values are much lower than the US EPA guideline value of 5 mg l^{-1} (Nixon, 1995).

The direct potential nuisances caused by nitrate are essentially related to drinking water production or industrial water use. In the case of drinking water, there is a risk of methaemoglobinaemia in vulnerable groups such as young children whose stomach acidity is insufficient, resulting in a transformation of nitrate to nitrite in the gut (OMS, 1980). This risk has prompted authorities to regulate (Directive 75/440/EEC) and specify guideline and MAC values of 25 and 50 mg l^{-1} NO_3^-, respectively, principally supported by the precautionary principle. The actual risk and relevance of these values are controversial (L'Hirondel, 1993a; 1993b; Duncan *et al*., 1997). Suitable maximum concentrations for animals' drinking water are not well researched. Much lower concentrations (a few mg l^{-1}) are necessary for industries using fermentation processes (such as breweries).

4.2.3. Other nutrients

Phosphorus compounds do not cause any direct impacts in water. However, in intensively farmed regions, phosphorus accumulates in over-fertilised soils. An often-neglected aspect is the risk of accumulating heavy metals through phosphorus fertiliser applications. In the case of animal manure-spreading, these include copper (which is a growth factor used in pig-farming) and zinc (which prevents parakeratosis, a skin disease of pigs) (Simon, 1992). In areas where only mineral fertilisers are applied, metallic contamination from cadmium, arsenic and uranium in the phosphate fertiliser can occur. Research in this field is not conclusive, but seems to indicate the toxicity problems could appear in the long term (decades).

Loads of other nutrients (such as potassium, calcium and iron) from human-related activities seem to be negligible when compared to natural sources (the exception is elevated salinity produced by potassium pollution).

4.3. Eutrophication

4.3.1. Definition and approach

The most important role of nutrients is for the growth of plants. In water, the excessive amplification of trophic cycles is generally referred to as 'eutrophication', despite the fact that this is etymologically incorrect. According to modern concepts in freshwater ecology, it is clearly preferable to consider the whole extent of each ecosystem (for example, the lake ecosystem comprises the water body, the sediments as well as the catchment). This concept is also applied to running waters, but with some adjustments. In this way, a eutrophic aquatic system is one where the potential concentration of nutrients is excessive, even if the instantaneous concentration in the water is low.

In order to embrace all media covered by this report, the following operational definition of eutrophication has been used:

'*Nutrient enrichment of the aquatic environment leading to increased rate of supply of organic matter, including primary production. This enrichment leads to environmental perturbation and changes in ecological quality, and ultimately reduces the utility of the aquatic body.'* [modified from Moth Iversen et al.,1997].

Thick layer of *Aphanizomenon flos-aqua* (a cyanobacteria) covering the surface of a French reservoir. Photo: Philippe Crouzet

This definition encompasses more limited definitions, used for specific context, such as urban waste water discharge control policies.

Eutrophication is also defined in the Urban Waste Water Treatment Directive (UWWT) (91/271/EEC) as:

'the enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned.'

The Nitrate Directive (91/676/EEC) defines eutrophication in a similar way to the UWWT Directive with one significant difference, the definition is restricted to eutrophication caused by the compounds of nitrogen.

The nutrients under consideration in this report (Table 2.1) are in fact limited to nitrogen and phosphorus. The other elements are only incidentally considered, when they have an important influence on a particular eutrophication phenomenon.

4.3.2. Aquatic photosynthesis

Plant growth results from photosynthesis, which can be summarised as the plant's ability to capture photons and to use part of their energy to reduce inorganic carbon to organic carbon compounds and at the same time to take up a balanced amount of nutrients from the water. All the nutrients incorporated are in ionic forms. In water photosynthesis, therefore, substantially modifies the ionic equilibrium.

Two general chemical equations of photosynthesis (Stumm and Morgan, 1981) are given below, and depend on the source of nitrogen being used:

$$106\ C + 16\ NO_3^- + HPO_4^{2-} + 122\ H_2O + 18\ H^+ \rightarrow (CH_2O)_{106}(NH_3)_{16}(H_3PO_4) + 132\ O_2 \quad \textbf{Equation 2}$$

$$106\ C + 16\ NH_4^- + HPO_4^{2-} + 122\ H_2O \rightarrow (CH_2O)_{106}(NH_3)_{16}(H_3PO_4) + 107\ O_2 + 14\ H^+ \quad \textbf{Equation 3}$$

When nitrate is used, the pH rises, as a result of the incorporation of 18 moles of acidity. The oxygen content rises too, and may reach very elevated super-saturated values. When ammonium is used, the pH should drop, according to Equation 3. In fact, since the ammonium ions primarily result from gaseous ammonia hydrolysis, the pH increases too, but the final consumption of acidity is only 2 moles, instead of 18 when nitrate is used. In rivers, nitrate is nevertheless the main source of nitrogen, with the exception of water bodies receiving large amounts of urban sewage where ammonium is an important source of nitrogen.

The overall composition of aquatic plant tissue according to the equations above is $C_{106}H_{263}O_{110}N_{16}P$, yielding a C:N:P w:w:w ratio of 41:7:1.

The ratio of major components of phytoplanktonic species is considered as quite constant, despite large variations around average ratio values. Hence, this ratio is often named 'the Redfield ratio', after the author who established it through observations of marine phytoplankton around 1930 (in Strickland, 1960). In fact, the N:P ratio depends on the species (ranges of 3:1 to 20 :1, w:w) and on the physiological state of the cells, since cellular divisions tend to evolve the N and P concentrations, as well as the nutrient reserves. Macrophytes also present values falling in this range; for example, annual average values in the range 6.1±0.9 :1 to 11.9±4.3 :1, w:w have been observed on highly contrasting reaches of the river Doubs in the east of France (Nauleau, 1988).

Finally, the weight ratio of N:P in water is optimal for algae growth in the range of 8:1 to 12:1, i.e., when slightly higher than the Redfield ratio in plant tissues.

The N:P ratios in terrestrial plants are different to those found in aquatic plants. An average value of 15 :1 w :w is frequently quoted (Korselman and Meulman, 1996). However, requirements of terrestrial plants may be very different, from nitrophilic plants to ones demanding oligotrophic conditions.

By itself, the N:P ratio makes sense only if both nutrients are quite limiting. When there are large excesses in nutrient supply, the changes induced by nutrient ratios are negligible. However, it is important to bear in mind that control of eutrophication by nutrient limitation should consider both nutrient concentrations (the major engine of control) and nutrient ratios to prevent the occurrence of adverse effects.

It should not be forgotten that plant growth processes have a time dimension. Plants accumulate and, with bacteria, mineralise nutrients at different rates, so certain sequences of nutrient use can occur and, in particular, some nutrients can be used up. Since some algae are capable of storing more phosphorus than they require, and since organic phosphorus mineralisation is much quicker than mineralisation of carbon and nitrogen incorporated in plant material, the N:P and C:P ratios in the water phase decrease with the result that carbon or nitrogen availability can become a limiting factor. However, the open cycles of carbon and nitrogen in fresh water allow uptake from the atmosphere, whilst phosphorus is always confined to the water or sediment.

Real time observations of nutrient concentrations can, therefore, result in an incorrect understanding of the actual roles of carbon, nitrogen and phosphorus in eutrophication. In fresh waters with little or no pollution, phosphorus will always represent the most limiting nutrient – hence the N:P ratios will mainly be greater than the Redfield ratio.

The prominent role of phosphorus in inland water eutrophication has been understood for many years. The most famous on-site demonstration was by Schindler (1974) in whole lake experiments. The rivers studied by Carbiener *et al.* (1990) in the Alsace plain (France) have very low organic matter but varying degrees of pollution by nitrogen and phosphorus. They have shown that whatever the nitrogen concentration may be in these rivers, increases in plant growth are related to phosphorus concentrations. These types of observation, and examples of phosphorus and other nutrients, can be found in a wealth of literature on the subject, and can be distilled into two different approaches.

1. From a purely scientific point of view, Carbiener (1992) identifies phosphorus as the fundamental cause of eutrophication, but distinguishes other symptomatic factors (nitrogen, silicon, other nutrients, temperature, light, hydrodynamics, chemical effects etc.) which determine the type of plants involved and the quantity, timing and successions of the biomass produced. From this point of view, the idea of having a limiting factor is of secondary importance and only represents a clue that helps to understand a specific eutrophication occurrence.

2. Using a similar starting point, Chiaudani and Premazzi (1986) consider that phosphorus is the key element, even though many factors determine aquatic ecosystem plant biomass. Nitrogen plays a secondary role, but can become important at a high level of eutrophication. In this case, freshwater cyanobacteria are able to fix atmospheric nitrogen and the resulting nuisances are much more significant than eutrophication caused by other types of algae (Korselman and Meuleman, 1996).

It is now widely accepted that the systematic elimination of phosphorus is the only practical way to combat eutrophication in fresh waters, either because the other factors are beyond our control, or because phosphorus availability determines the influence of nitrogen and carbon.

4.3.3. Still waters

Nutrients are usually the limiting factor to the growth of algae in still water bodies, at least at low depths, where light is not limiting. The first effect of increased availability of nutrients may be an increased growth of weeds in shallow lakes. Progressive eutrophication causes phytoplanktonic algal growth to increase and accumulate biomass, thus making lake water more turbid. During extreme algal blooms lake water may appear to be covered by green paint. Such blooms often consist of blue-green algae (cyanobacteria), some of these species are toxin-producing.

Eutrophication causes substantial changes to the whole lake ecosystem. In deep lakes, the phytoplankton concentration is approximately proportional to the phosphorus concentration. In shallow lakes, biological interactions have a significant impact. In such lakes, the change from bottom-dwelling weeds to planktonic algae alters the conditions for the zooplankton living on algae, and in turn the composition of the fish stock. The ecosystems are conservative and a drastic change will not arise until eutrophication has become sufficiently severe. The altered ecosystem also exhibits mechanisms that make the system conservative again, establishing a new steady state. Algae that settle will decompose and cause high oxygen consumption in deep water layers, the hypolimnion. On certain occasions, oxygen is depleted and the whole fish stock of a lake may die, a fish kill incident has happened. Apart from these indisputable adverse impacts, the change from crystal clear water to a more turbid state is generally seen as a deterioration of the recreational and aesthetic value.

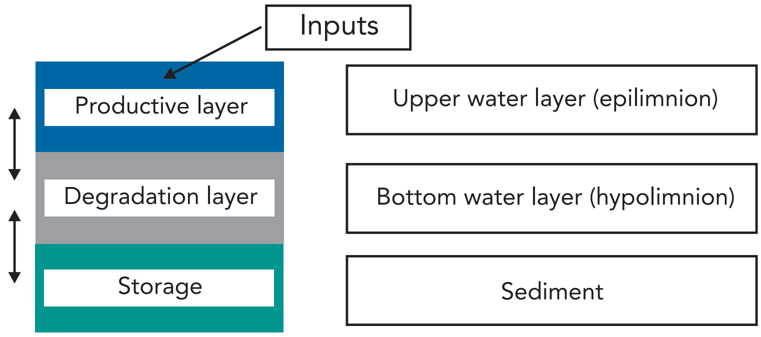
Most lakes are phosphorus limited. However, some seem to be limited by nitrogen, possibly because of a large excess in phosphorus. In an Italian study (Gaggino *et al.*, 1985) about 85% of the investigated lakes had a N:P ratio greater than the Redfield ratio indicating that algal growth was limited by phosphorus. In contrast about 7% seemed to be nitrogen limited while both nutrients playing equal roles in the remaining 8%. Nitrogen limitation is considered to promote dominance by blue-green algae.

The main problem concerning eutrophication of lakes is thus phosphorus pollution. Susceptibility to eutrophication depends upon morphometrical, chemical and hydrological characteristics. The longer the hydrological residence time, the lower the resulting concentration from a certain input concentration. Thus deep lakes with a long residence time generally need a higher input of nutrients than shallow lakes to reach the same degree of eutrophication. However, even though a deep lake may become less turbid than a shallow lake, a reduction of transparency to some metres may still be considered a serious deterioration of water quality compared to a state with crystal clear water with a top to bottom view even at many meters depth.

Eutrophication of a water body is also dependent upon exchanges between compartments (schematically illustrated in Figure 4.1). In aerobic conditions, and provided that sufficient amounts of iron, aluminium and calcium are present, phosphorus is trapped in the sediments and the total amount of phosphorus increases progressively. The mineralisation of the organic matter formed by photosynthesis in the upper layers results in falling oxygenation levels in the deep lake layers. Consequently, the phosphorus trapped in sediments may be leached in large amounts when the oxygen concentration is reduced to low levels. If any subsequent movement of the water masses carries this dissolved phosphorus towards the upper layers secondary eutrophication occurs. This global mechanism is called ‘internal loading’. Once a lake has become seriously eutrophic, it may take several years to return to a better state after the pollution has stopped because there is often a significant internal loading from phosphorus accumulated in the sediment. This internal loading may last for many years after external loading has been stopped. However, the mechanisms are influenced by several chemical and morphometrical factors such as iron content, residence time and mixing of the water.

Water composition changes according to seasonal and interannual patterns, partly determined by the morphometry of the lake and climate that governs vertical mixing. For example, the maximum nutrient concentration is observed prior to the start of the photosynthesis period. The decrease in bottom layer oxygen concentrations depends on initial concentrations which can require several years to recharge from previously low levels.

Compartments in still waters **Figure 4.1**



Models established by Vollenweider (1968; 1976), and Dillon and Rigler (1974) provide a relationship between the phosphorus load and the biomass produced by photosynthesis. These models were developed from observations of the biomass / phosphorus concentration relationship and from the hypothesis that phosphorus concentrations can be estimated from phosphorus loads. Even though the models at best are approximations, they enable the excessive nutrient load to be defined for any particular lake.

This scientific approach has led to the development of assessment criteria, which have been transposed into indicative limit values by the OECD (1982) (see Table 4.2). It has been proposed that the practical application of these models should take into account the likely natural phosphorus concentration, which can be estimated using the morphoedaphic index (MEI) (Premazzi and Chiaudani, 1992) calculated from the alkalinity (or conductivity) divided by the mean depth.

Table 4.2 **Indicative limit values for phosphorus and related parameters in still waters**

OECD trophic level	Ultra-Oligotrophic	Oligotrophic	Mesotrophic	Eutrophic	Hypereutrophic
Total P concentration (annual average) (µg P l^{-1})	<4	4-10	10-35	35-100	>100
Chlorophyll (µg Chla l^{-1})					
• mean	<1	<2.5	2.5-8	8-25	>25
• max	<2.5	<8	8-25	25-75	>75
Secchi disk disappearance depth (m)					
• mean	>12	>6	6-3	3-1.5	<1.5
• min	>6	>3	3-1.5	1.5-0.7	<0.7

Tables 4.3 and 4.4 give expected natural phosphorus concentrations derived from mean depth and conductivity or alkalinity of a still water body. The potential trophic classes from the natural values are highlighted by colours identical to those used in Table 4.2.

The difference between the OECD indicative limits (which define the trophic status) and the probable natural concentrations (using the MEI), can be taken as an estimate of the 'excessive' concentration due to human activities. Some indicative values are provided in Table 4.3 and 4.4.

Table 4.3 **Estimated "natural" P concentrations (µg P l^{-1}), according to the morphoedaphic index method (conductivity)**

Conductivity classes (mS cm^{-1})	Mean depth of the lake (m) 10	30	50	100	150	200
50	11.5	8.5	6.9	6.2	5.5	5.1
500	21.3	15.8	13.8	11.5	10.3	9.5
1000	25.7	19.1	16.7	13.8	12.4	11.4

[*] calculations by G. Premazzi using the equation Log Pn (µg l^{-1})=0.87+0.27(±0.11) Log (Conductivity / average z).

Table 4.4 **Estimated "natural" P concentrations (µg P l^{-1}), according to the morphoedaphic index method (alkalinity)**

Alkalinity classes (meq l^{-1})	Mean depth of the lake (m) 10	30	50	100	150	200
0.2	8.3	5.8	4.9	3.9	3.4	3.1
2.0	17.8	12.4	10.3	8.3	7.4	6.6
4.5	23.2	16.2	13.6	10.8	9.0	8.7

[*] calculations by G. Premazzi using the equation Log Pn (µg l^{-1}) =1.48+0.33(±0.09) Log(alkalinity / average z).

Shallow and alkaline lakes have potentially relatively high natural phosphorus concentrations, which may make these water bodies very sensitive to any supplementary input, since they are likely to be naturally mesotrophic. This means that the margin for human inputs is very dependent on the type and composition of the lake under consideration. However, in hard water lakes rich in calcium, a large amount of phosphorus may be precipitated as calcite and will then be bound to the sediment.

It must be emphasised that the search for natural conditions is very difficult, since the natural phosphorus input depends on many catchment characteristics such as bedrock composition, weathering factors, vegetation cover and uncertainty in the data and history of the catchment studied. Therefore, the reference values used later in the report will be mean values, which may, by definition, be exceeded in some circumstances.

Until recently, the predictive capacity of eutrophication models was limited to quantitative assessment of produced biomass. However, one of the most threatening issues in still water eutrophication is the excessive development of toxic phytoplankton. Cyanobacteria are frequently involved in fresh water blooms of toxic phytoplankton. *Microcystis aeruginosa*, is one of the most widespread organism involved and releases powerful toxins containing amino-acids specific to cyanobacteria. It appears preferentially in hypereutrophic water bodies, principally when the N:P ratio falls below five. Recently, the biology and behaviour (including vertical migrations) of this unwanted species have been modelled in order to develop control methods for the hypereutrophic Villerest reservoir in France (Devaux, 1994).

4.3.4. Rivers

4.3.4.1. Mechanisms

River eutrophication is the result of a severe perturbation of the normal balance between the supply of external organic matter, the internal photosynthetic activity of river plant communities and the capacity to mineralise this organic matter. These perturbations are generally the result of damming, organic matter pollution and excessive nutrient inputs.

In undisturbed rivers, plant communities develop along an upstream-downstream gradient, according to both the hydrodynamic conditions and the general increase in available nutrients in downstream reaches.

High velocity rivers provide adverse conditions for the development of river plankton, since they are swept downstream faster that they can multiply. Reynolds (1984), reviewing hydrodynamic conditions for river plankton development, suggested that a water velocity greater than 0.5 m s^{-1} would limit planktonic growth. However, the Loire river, which often flows at more than 1 m s^{-1} in summer in most of its middle reaches, presents one of the best examples of hyper-eutrophic rivers with high phytoplankton biomass.

Since phytoplankton growth requires sufficient time, the actual residence time of water in the river (from a few hours to several weeks in largest rivers) plays a major role. In small, shallow rivers the residence time is too low to allow any significant phytoplankton development. Very deep rivers do not offer the best conditions for phytoplankton growth over the whole water column. Hence, the morphometric characteristics of a river are a major factor affecting its sensitivity to eutrophication, since they may not be suitable for maximum growth in spite of nutrient availability.

4.3.4.2. Perturbations because of river eutrophication

River eutrophication results in two types of perturbation of water characteristics. First, because the photosynthesis and respiration of the biomass is considerably increased, the diurnal variations in dissolved oxygen levels (photosynthesis / respiration) and pH (see equations 2 and 3) are dramatically increased.

Phytoplankton may behave as organic pollution requiring oxygen for its mineralisation, and thus increasing the total organic matter content of water. This is, for example, the case when eutrophic water is used to cool power plants, or when turbidity is increased, or when the river reaches its estuary. Long-established observed ratios between biochemical oxygen demand (BOD_5) (a common indicator of biodegradable organic matter) and chlorophyll concentrations are generally around 50-100 and quite stable. [100 µg l^{-1} of chlorophyll indicates a potential BOD_5 of 5-10 mg l^{-1}, (Crouzet and Bertru, 1987)]

Chlorophyll a concentrations are widely used as an estimate of biomass, and for assessing the trophic status of rivers. Where chlorophyll measurements are not available an indirect assessment can be made using pH and oxygen variations. The latter reflect biomass activity and thus constitute an indicator of probable eutrophication. The variation in pH is less sensitive, since it depends on both the form of nitrogen available to the plant, and on the buffering capacity of the water body. The latter is determined by its alkalinity or hardness. Such an indirect assessment is used in Chapter 6.5.1.

Values indicating excessive phosphorus in rivers are not directly available from similar approaches to those described for lakes. Many 'reference' values are, however, available. For example, the Freshwater Fish Directive (78/659/EEC) gives maximum phosphorus concentrations of 0.065 and 0.13 mg P l^{-1} for salmonid and cyprinid waters, respectively. The Surface Water Directive (75/440/EEC) stipulates guidance values of 0.17 and 0.31 mg l^{-1} for A1 and A2/A3, levels of treatment, respectively.

According to Carbiener (1992) these values are already within the range of hypertrophy. It would appear likely that the significant differences in the phosphorus 'limits' proposed by different authors derive from the different approaches. For example, either high values for nuisances judged to be unacceptable might be used, or lower values for the measurable effects on disturbed ecosystems.

4.3.5. Estuaries

Estuarine waters are the result of mixing of river and sea water. The dilution of sea water depends principally on the river flow and tidal regime. The major event occurring in large estuaries is the flocculation of organic matter and clay, and the death of river phytoplankton along the salinity gradient. This has been well documented in the river Loire estuary.

Uncultivated buffer zones along rivers reduce the input of nutrients from adjacent farmland, Uggerby, Denmark. Photo: John Nielsen/ BIOFOTO

The relationship between increased nutrient levels in estuaries and increased algal standing crops is an extremely complex issue, and one which can be prevented or mediated by a host of other factors. These include flushing time, substrate suitability for macroalgal growth, light penetration (determined principally by suspended solids), mixing depth (affected by stratification – temperature and salinity), grazing, growth rates, and desiccation of intertidal reaches (Parr and Norton, 1994; Parr, 1994).

Two macronutrients are commonly referred to as being potentially limiting in estuarine or coastal waters: nitrogen and phosphorus. However, care should be taken over the use of the word ‘limiting’, since other factors (often light or residence time) usually have a more limiting effect on algal growth. At N:P ratios of <8 (w:w), predominantly in the lower reaches of estuaries and coastal waters, nitrogen is said to be limiting. At N:P ratios of >8 (w:w), often in the upper reaches of estuaries where water quality is dominated by the freshwater inflow, phosphorus is said to be limiting. The cut-off value of 8, in the range of the Redfield ratio, is based on the ratio found in a typical alga growing in ideal conditions but which does not have any nutrients accumulated from luxury uptake. In practice, however, for different algal species there is a range of values at which the N:P ratio can be considered to be ideal (OECD, 1982; Reynolds, 1984). Use of the N:P ratio is made even more complex by the fact that the ratios should be based on bioavailable nutrients and on their differential uptake, but bioavailable nutrients can only be measured using bioassays (see Parr, 1993).

Other nutrients have also been reported to limit algal growth, of which silicon is probably the most widely reported, albeit in a different manner to nitrogen or phosphorus. With silicon, the growth of only certain types of algae like diatoms are limited, so a shortage of silicon tends to affect what grows, rather than how much grows. Reports of algal growth being limited by a shortage of micronutrients, such as vitamin B_{12}, are much less frequent and tend to be species-specific typical of, for example, autotrophic flagellates.

Recently, however, there has been increased discussion over the possible limitation of phytoplankton growth by a shortage of iron, particularly with regard to diatoms, in oceanic waters and warmer climates (e.g. Martin *et al.*, 1994). Phytoplankton requires more iron than any other trace metal (Wells *et al.*, 1983), since it is a cofactor for many enzymes, notably with regard to oxidase systems (i.e. the oxygen evolution step of photosynthesis). However, estuarine waters tend to contain higher levels of iron than the open sea, since land is a major source of this metal, as it is for nitrogen and phosphorus. It is, therefore, considered unlikely that low iron levels would often be a major factor limiting algal growth in many estuaries.

In estuaries where algal growth is stimulated, the effects may be seen in the water column as increased frequency of algal bloom formation, more intense algal bloom formation, or sometime by normally sub-dominant members of the phytoplankton community becoming dominant and causing problems such as diarhetic or paralytic shellfish poisoning (DSP or PSP) events in river plumes. In other cases the problems are more concerned with increased growth of opportunistic and nitrophilic benthic algal genera, notably *Enteromorpha* and *Ulva*. These may colonise tidal flats or oust natural populations of slower-growing, more sensitive species, such as eelgrass. The latter is a particular problem off the coasts of Germany and Denmark (OSPAR, 1992) and many other places.

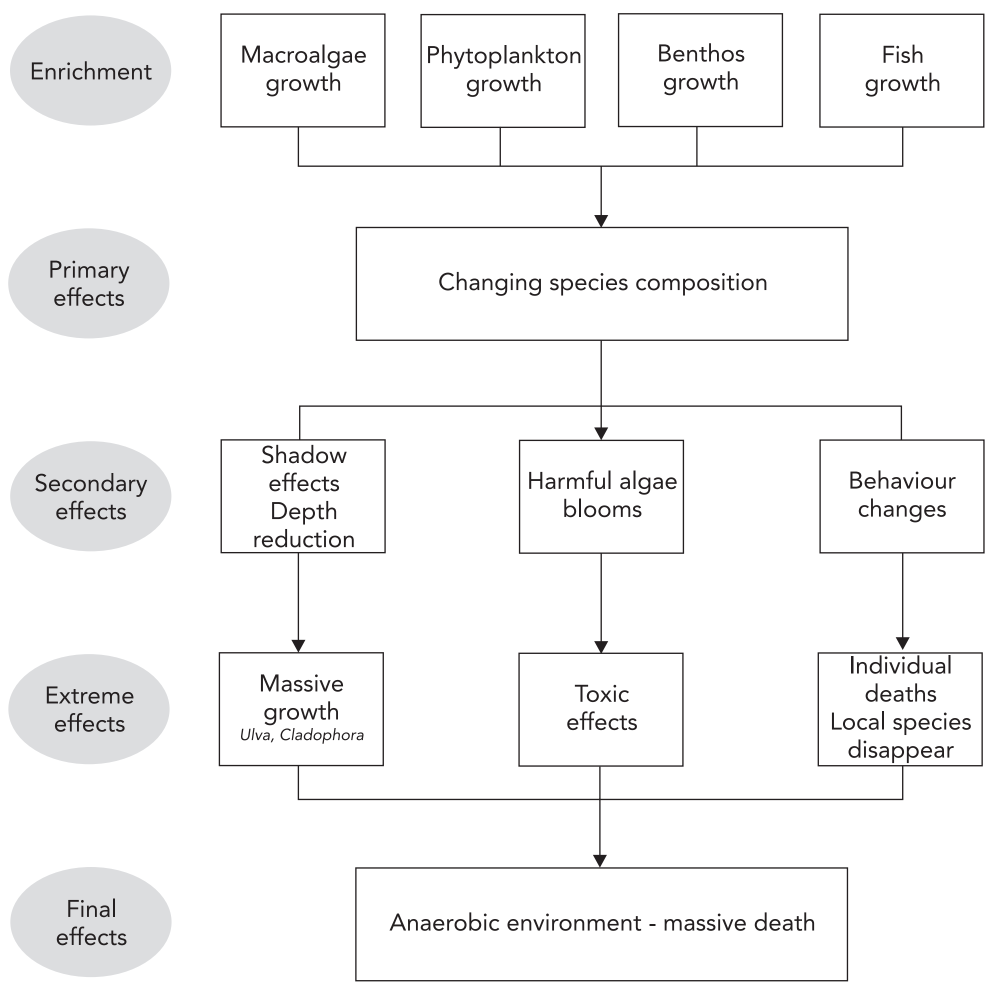
Nutrient enrichment of estuaries, as in freshwater lakes, tends to result in one or the other of these problems – plant standing crops tend to increase either in the water column or on the substrate, both of which result in lower dissolved oxygen levels. When *Enteromorpha* and *Ulva* standing crops increase, dissolved oxygen levels in surrounding sediment decrease due to respiration of the associated bacterial microflora, and some invertebrates are driven away. For example, in the Ythan Estuary in the UK, this has had a major effect on the *Corophium* population (Raffaelli *et al.*, 1991), with feared repercussions to the status of this internationally renowned bird reserve. Dissolved oxygen sags, as a result of phytoplankton blooms, occur after the bloom has collapsed when the organic load associated with the algae increases the sediment oxygen demand. In estuaries, this situation may then be exacerbated by high turbidity (during spring tides), lower river flows and elevated temperatures in summer.

4.3.6. Marine waters

Marine waters are those which salinity is not influenced by fresh water discharges. Eutrophication of marine waters is caused by nutrient enrichment occurring where other favourable conditions are met (low current velocity, light, temperature, etc.). The process of nutrient enrichment and its effects in the marine environment are illustrated in Figure 4.2.

The key plant nutrients in the sea are nitrogen and phosphorus, but other nutrients such as silica and trace elements are also important. When favourable conditions are met, nutrient enrichment leads to higher primary productivity of phytoplankton and phythobenthos (given enough light), followed by higher secondary productivity of animals. The phenomena of eutrophication in the marine environment is much more complex than in fresh waters. Although it is clear that excessive nutrients are the fundamental cause of the problem, the relationships between these excesses and the resulting problems in the ecosystem depend on various factors, like hydrodynamic environment, and are not well understood. Hence, it would not seem possible to identify one single nutrient as the cause: of eutrophication: nitrogen and phosphorus both appear to be involved. However some authors argue that nitrogen rather than phosphorus is usually the potentially limiting nutrient for primary pro-

Figure 4.2 **Illustration of the process of nutrient enrichment and its effects on the marine environment. Source: ETC/IW**



duction (Jørgensen and Richardson, 1996). The impact of other factors, such as 'mediators' (Aubert and Aubert, 1986) on the ecosystem will not be discussed in this report: they have less influence on the modes of eutrophication occurrences.

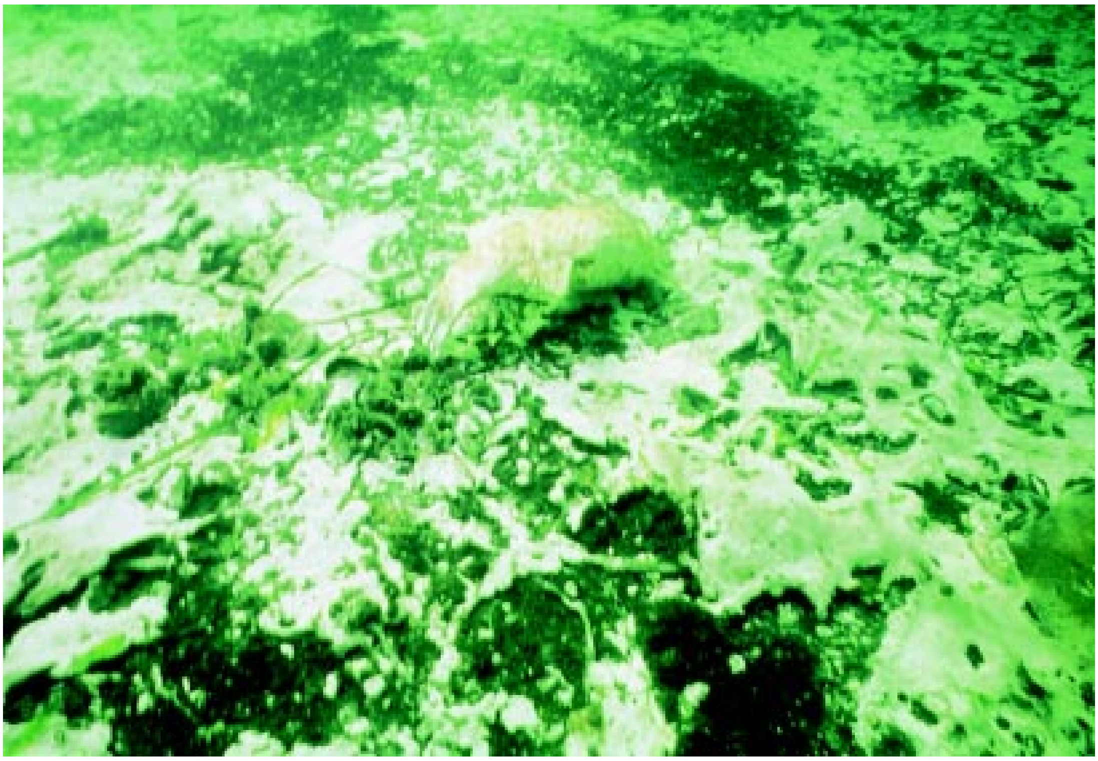
While nutrient enrichment may at first be regarded as positive, further eutrophication may be negative. Large phytoplankton 'blooms' and large growths of seaweeds develop, which cannot be controlled by zooplankton and herbivore grazing. Much of the produced phytoplankton sinks to the bottom, where oxygen deficits in the bottom water can occur as the organic matter decomposes. In extreme cases all the oxygen is consumed and hydrogen sulphide produced. Hydrogen sulphide is toxic to marine life and thus high mortalities can occur. Such oxygen deficiencies are the ultimate signs of severe eutrophication.

The effects of eutrophication are numerous: deaths of fish and other marine animal which can reduce fishing production, significant ecosystem disturbances, impacts on tourism, human health risks and costs to implement curative measures.

However, the marine environment evolves slowly and this may give rise to difficulties in understanding the eutrophication processes. In some areas, substantial increases in commercial fishing catches because of eutrophication could be interpreted as increased productivity, but in fact are a consequence of progressive anoxia in deep waters, which displaces benthic and semi-benthic animals (Gray, 1992).

There is a huge diversity of phenomena which are symptoms of marine eutrophication, and they are still poorly understood (Vollenweider, 1992). Like fresh waters, marine water eutrophication is a major source of organic pollution with resulting effects on the ecosystem (in particular the oxygen balance), which are added on top of the impacts from other biodegradable pollutants.

In marine waters, available phosphorus is principally bound to sediment. It may be released when there is sufficient sediment disturbance, or just a low concentration in the water column. Hence, there is a certain active stock of bioavailable phosphorus, which often corresponds to the quantities required by algae. Atmospheric nitrogen fixation has been demonstrated, at least in coastal lagoons, salt marshes and intertidal areas. The possible importance of atmospheric fixation in the open sea is probably limited, and this mechanism does not seem



The ultimate result of coastal eutrophication. A 'dead' sea bottom devoid of oxygen covered by sulfur bacteria, Bay of Aarhus, Denmark.
Photo:
M.B. Rasmussen/
BIOFOTO

to be as important as it is in freshwater environments. Hence, the reduction of nitrogen loads seems to be a possible way of controlling coastal marine eutrophication. For this reason, the control of eutrophication in marine environments cannot be achieved solely through phosphorus load reduction, but must also address other nutrients like nitrogen, or pollutants which are involved in phosphorus mobilisation.

To summarise, researchers have not been able to propose indicative values related to nutrient excesses, but instead tend to describe values related to impacts, in particular the oxygen balance, organic carbon concentrations and the C/N ratios in sediments. More research on this subject is obviously required. It is not, therefore, possible to provide guidance on loads or concentrations of excess nutrients in the marine environment. Instead an approach based on impacts has been followed in this report.

4.4. Summary

Table 4.5 gives values considered to be 'excessive' but these are intended to be only for guidance. The concept of 'excess' should be considered in relation to a hypothetical natural state for a particular observed nutrient sensitivity. Where several values are indicated, they refer to different ecosystems. Excess is only considered as such when it is caused by human activities.

Table 4.5 Indicative values for nutrients above which concentrations can be considered excessive

Ecosystem	Nutrients		
	$N\text{-}NO_3$	$N\text{-}NH_4$	Phosphorus (as Total P)
Terrestrial (kg ha^{-1} $year^{-1}$)	10? (oligotrophic) 200? (cultivated)		10 ? 100? (cultivated)
Surface fresh waters (direct effects) (mg l^{-1})	Water intended for the abstraction of drinking water • 5.7, guideline value • 11.4, mandatory value	Water intended for the abstraction of drinking water • 1.17, mandatory value Drinking water • 0.39, maximum allowable concentration Freshwater Fish (NH_3) • 0.005, guideline value • 0.025 mandatory value	Water intended for the abstraction of drinking water • 0.17 guideline value (A1 treatment) • 0.31 guideline value (A2/A3 treatment) Freshwater Fish • 0.065 limit value for salmonids • 0.13 limit value for cyprinids
Surface fresh waters (eutrophication) (mg l^{-1})	1 (for P=0.1). in general a value of total available nitrogen which is greater than 10 times the P concentration limits the risk of developing N-fixing cyanobacteria.		Scientific evidence: 0.01 rivers, 0.025 lakes
Marine waters (eutrophication)	Location specific concentrations (and hence not defined here) ensuring that dissolved oxygen saturation is greater than 30 % in the deepest layers.		

? tentative

5. Status and impacts in lakes and reservoirs

5.1. Characterisation of European lakes and reservoirs

The density and size of natural lakes throughout Europe varies considerably (Table 5.1). In terms of numbers, the majority of natural lakes in Europe occur in Norway, Sweden and Finland with 85 000 and 56 000 lakes over 1 hectare in Sweden and Finland, respectively (Swedish EPA, 1992; Wahlström *et al*., 1993). It has been estimated that over 9 % of the land area of Finland and Sweden are covered by freshwater lakes. Significant numbers of natural lakes also exist in Iceland, Denmark, Ireland and UK. Many lakes in the northern part of Europe were formed during the last glaciation period and are often rela-

Number of lakes and reservoirs of different sizes in European countries. Source: ETC/IW — **Table 5.1**

	Size distribution				
	0.01-0.1 km^2	0.1-1 km^2	1-10 km^2	10-100 km^2	>100 km^2
Austria	9000 (<1km^2)	17	7	2	
Bulgaria	53	175	288	14	0
Croatia	-	1	3	0	0
Czech Republic	116 ‘large’, 560 ‘smaller’ and 23 468 ‘small’ reservoirs. Few natural lakes				
Denmark	365	269	69	6	0
Estonia	750	209	41	1	3
Finland	40 309	13 114	2283	279	47
France (*)	24 068	2011	201	25	2
Germany (**)	~4700	~1300	~250	~24	2
Greece	-	-	-	>16	1
Hungary	-	-	-	2	2
Iceland	~7000	1650	176	17	0
Ireland	~5500		~100	14	3
Italy	-	>168	>82	13	5
Latvia	3039	636	119	16	0
Lithuania	2010	15	56	13	0
The Netherlands	>100	>100	100	10	2
Norway	116 218	16 417	2039	164	7
Poland	1057 (> 0.5 km^2)		46		
Portugal (***)	-	30	40	15	0
Romania	-	-	24	5	0
Slovenia	66	24	11	1	0
Spain (***)	482	330	247	63	2
Sweden	71 693	20 124	3512	369	23
Switzerland	-	111	40	13	5
United Kingdom	478	197	146	27	2

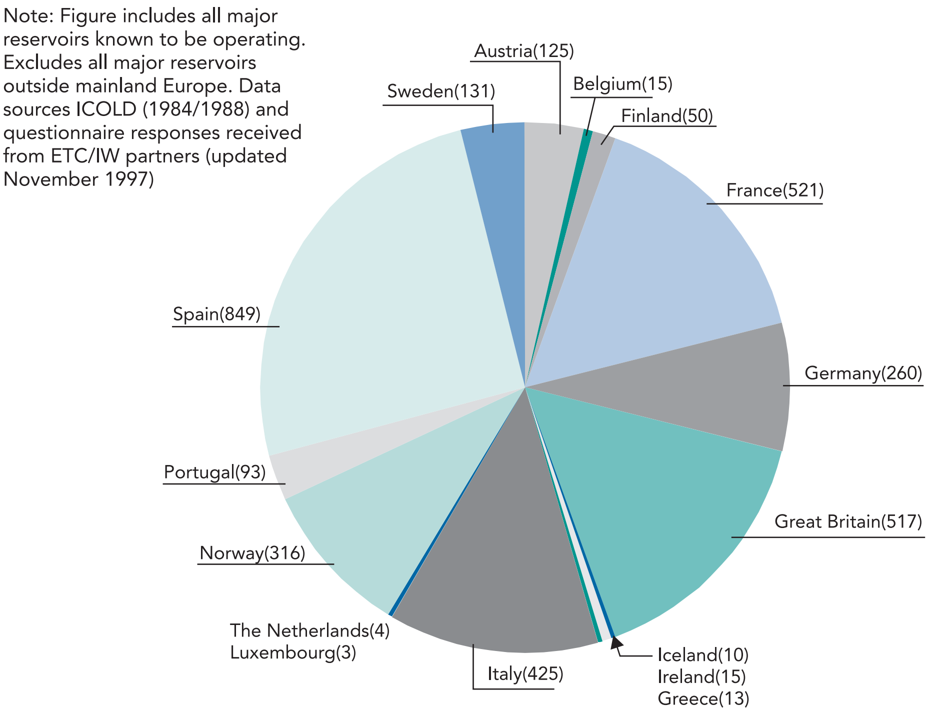
Notes: (*)Incl. lagoons and reservoirs; (**) Only natural lakes; (***) Reservoirs;

tively shallow. In the Alps and other mountainous regions, there is also a high density of natural lakes. These are usually formed by tectonic processes and are often very deep. A special type of lowland lake results from the separation of coastal lagoons previously connected to the sea. Such lakes can be observed on the Atlantic and Mediterranean coasts. Some countries such as Portugal, Spain and Belgium have very few natural lakes.

Reservoirs are distributed quite differently over Europe, often with the highest densities in regions with low rainfall, particularly southern Europe. In other countries with increasing demands for water and energy, reservoirs are also frequent. It has been estimated that there are about 3900 large reservoirs with a dam height exceeding 15 m in Europe excluding the former Soviet Union (Boon, 1992). Spain, France, UK and Italy have the largest number of major reservoirs (more than 400 in each case). Scandinavian countries have lower numbers of reservoirs, but these are generally of larger capacity.

Reservoirs are similar to lakes in many respects, but there are some general differences. Most reservoirs can be considered as intermediate between rivers and lakes. The residence time is normally higher than in most rivers but lower than in many lakes. Natural lakes can generally be situated in upstream as well as downstream parts of a river basin, whereas many reservoirs, constructed by damming a river, are elongated and usually placed at the downstream boundary of a relatively large watershed. Nutrient and sediment loading rates are commonly much higher for reservoirs than natural lakes. While natural lakes have the outflows at the surface, reservoirs have usually several artificial outlets affecting the internal mixing processes. The position of

Figure 5.1 **Distribution of major reservoirs in countries in the EEA area. Source: ICOLD, ELDRED and questionnaire returns.**



these outlets varies greatly according to the design and purpose of the dam. A large number of dams constructed for hydro power have their outlets close to the surface of the water, whilst dams made for other water uses may have operating outlets at different levels, including at the bottom of the dam wall.

The water level of natural lakes varies with precipitation, evaporation, natural outflows, etc., the level changes are limited and slow. On the contrary, the water level, of reservoirs is directly dependent on water inputs and management rules. These rules may yield all types of level variation. For example, in reservoirs devoted to flood control, a 10 m change within a few days is possible. In reservoirs used for water supply in dry countries, the water body may be reduced to a small fraction of the maximum volume after several years of water shortage. In all cases, these changes affect the littoral and aquatic plants and animal communities.

Reservoirs are today as important as natural lakes as recreational areas, but in addition have high importance for flow rate regulation, flood control, electric power generation, water supply, irrigation, etc. [See EEA (1999) for more details of reservoirs in the EEA area].

5.2. Nutrient concentration in lakes and reservoirs

5.2.1. Reference levels of phosphorus

The natural state of clean and unaffected lakes depends on their size and depth, and on the geology of the catchment. Some soils release more salts and nutrients than others, and thereby potentially give rise to a higher natural production of algae. The

Phosphorus concentration in "pristine" lakes. Source: EEA (1995) **Table 5.2**

Site	Country	Number of Lakes	Total P (µg P l^{-1})
Sierra Nevada	Spain	10	3.9 *
Pyrenees	Spain	102	~15
Tatra Mountains	Slovak Republic	10	5.7
Northern Apennines	Italy	43	14
Southern Alps (Pennine-Lepotine)	Italy	50	19
Italian Alps	Italy	320	<10 (85%)
Reference-lakes – Sweden	Sweden	154	<15 (80%)
Forest lakes, northern Sweden	Sweden	59	13.2
Forest lakes, Finland	Finland	135	10
Reference lakes, Norway	Norway	1008	3

*Dissolved reactive phosphorus.

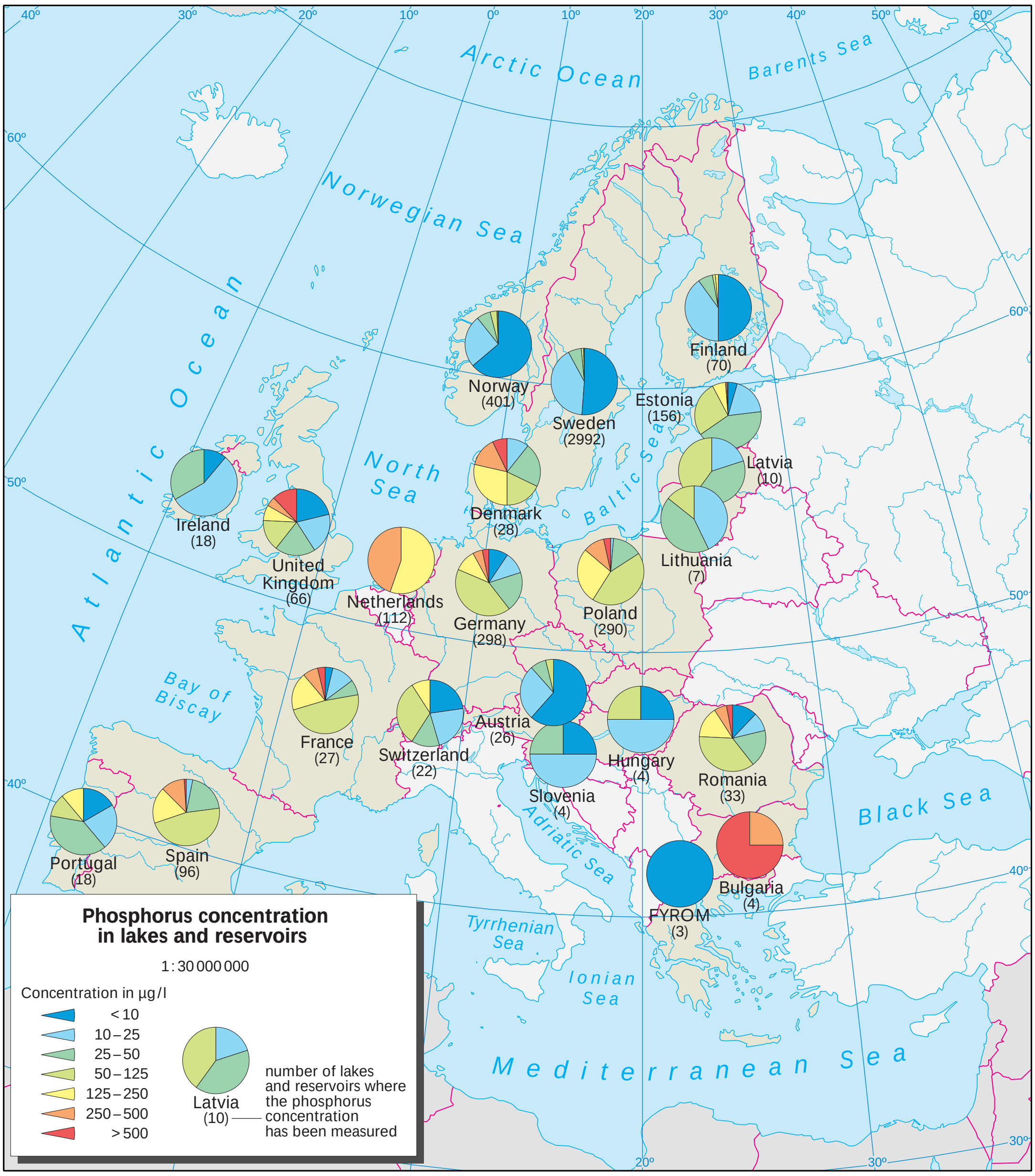
Nutrient-poor lakes dominated by rosette species of plants are becoming rare because of eutrophication (e.g. Lake Grane, Denmark). Photo: Kaj Halberg/Biofoto natural concentration of phosphorus in a lake can be estimated using the morphoedaphic index (MEI) based on alkalinity or conductivity of lake water (see Chapter 4).

The estimates of natural phosphorus concentrations (see Tables 4.3 and 4.4) are relatively high, and sometimes greater than the sampling data from a number of lakes considered to be pristine (Table 5.2). However, compared to the variation within European lakes, having phosphorus concentrations of up to more than 500 µg P l^{-1}, the variation between calculated and measured natural phosphorus concentrations is small. In general, in this report it is considered that shallow and hard-water lakes are 'clean' when their total phosphorus concentration is less than 25 µg P l^{-1}. Conversely, deep, soft water lakes are considered as clean when their total P concentration is close to, or less than, 10 µg P l^{-1}.

5.2.2. Observed phosphorus concentrations

The distribution of phosphorus in lakes and reservoirs within each country is illustrated in Map 5.1. The frequency distributions are mostly based on nation-wide lake surveys covering a large number of lakes. For some countries the distribution is esti-

Distribution of average total phosphorus concentrations in European lakes and reservoirs by country. Source: ETC/IW **Map 5.1**



Number of lakes per country: A(26), BG(4), CH(22), D(~300), DK(28), EE(156), E(96), FIN(70), F(27), H(4), IRL(18), I(7), LV(10), MK(3), NL(112), N(401), PL(290), P(18), R(33), S(2992), SLO(4), UK(66).

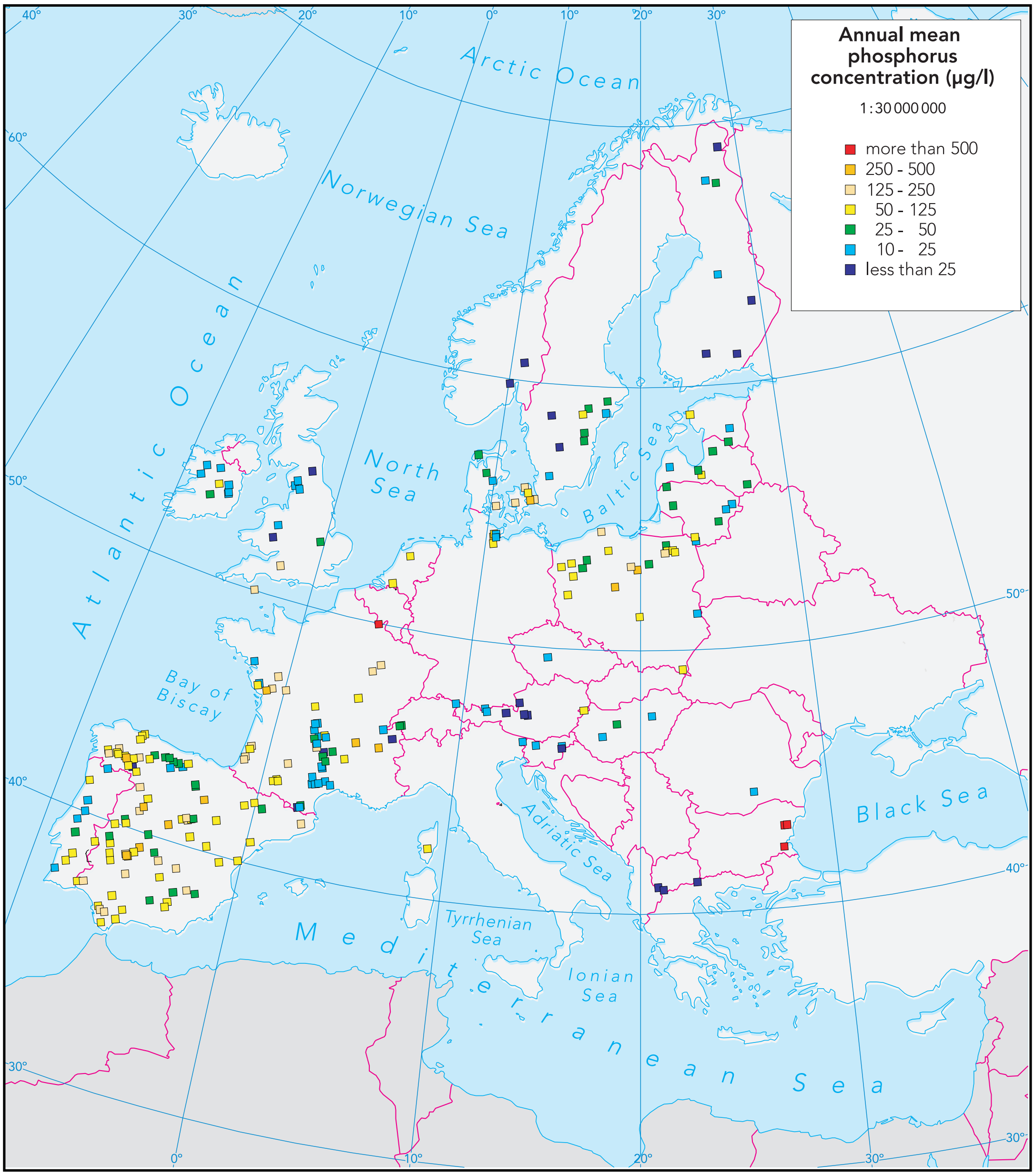
mated from information for a relatively small number of lakes. Though there has been no distinction made between natural lakes and reservoirs, the previously described pattern is also valid on this map, e.g. the Spanish pie is based on reservoir data.

Nutrient-poor lakes are mainly found in sparsely populated regions such as northern Scandinavia or mountainous regions such as the Alps, where many lakes are situated away from populated areas or fed by unimpacted rivers. In densely populated regions, primarily western and central Europe, a large proportion of lakes are affected by human activities and, therefore, relatively rich in phosphorus. The Baltic states, Poland and Spain have an intermediate quality in the majority of their lakes and reservoirs.

The downstream location of many reservoirs often equates to a large catchment which will almost always have a significant human influence. Large catchments tend to smooth out local variability. In densely populated regions only small catchments are unaffected by man.

There is often a high public awareness and concern for large and nationally important

Map 5.2 **Total phosphorus concentration in large or important lakes and reservoirs. Source: EEA (1999)**



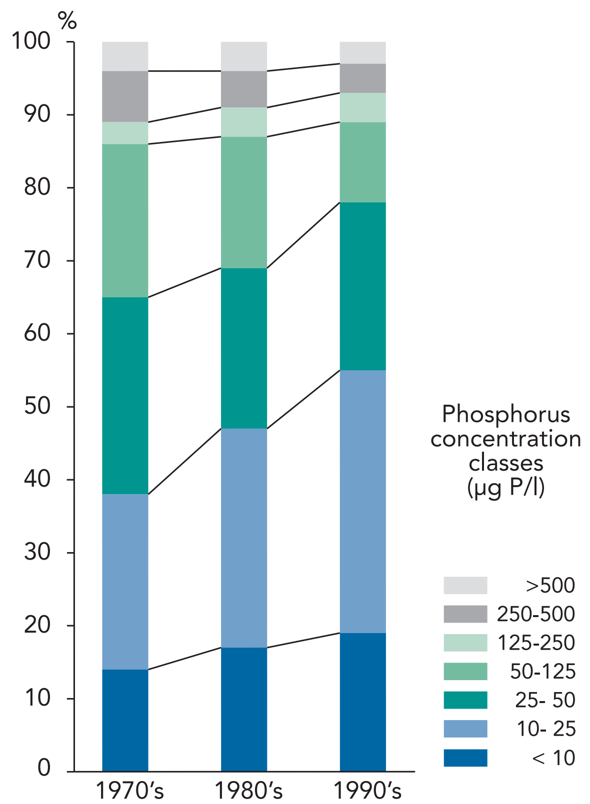
lakes and reservoirs. Map 5.2 shows the phosphorus concentration in some European lakes larger than 1 km^2. Large lakes have generally been monitored more regularly than smaller ones and can provide good time-series. Lakes and reservoirs with time-series from the 1970s have been used for analysis of trends in phosphorus concentration (Figure 5.2).

Over the past decades the environmental quality of some lakes has generally improved (Figure 5.2). The proportion of lakes rich in phosphorus has decreased and the proportion of lakes of near-natural quality (assumed to be below 25 µg P l^{-1}) has increased. Figure 5.2 should only be interpreted in relation to changes in those specific lakes as these were selected for their long time-series rather than for their representativeness. Environmental degradation up to the 1970s has often turned to improvement during the 1980s and 1990s, as illustrated for a number of selected lakes (Figure 5.3). The chart has been divided into three on the basis of initial phosphorus concentration. The cleanest lakes (<10 µg P l^{-1}) have generally remained unchanged, intermediately polluted lakes shows diverse trends and the most polluted lakes have improved since the 1980s.

Much of the change can be ascribed to improved treatment of urban waste water (as described in the pressure chapter), having taken place, most noticeably in the Nordic and western countries. Industrial pollution has similarly been substantially reduced. In addition, the discharge of waste water has in some instances been moved to a point downstream of the lake. In certain lakes and reservoirs, special action programmes in the catchment have been put in place and some are beginning to have positive effects.

Further improvement in lake quality water will require continued reductions in pollution from point sources. As pollution from these sources is reduced, the diffuse pollution from erosion and leaching becomes relatively more important, and there will

Temporal change in distribution to total phosphorus classes (in µg P l^{-1}) amongst selected European lakes. Source: compiled by ETC/IW from questionnaire returns **Figure 5.2**



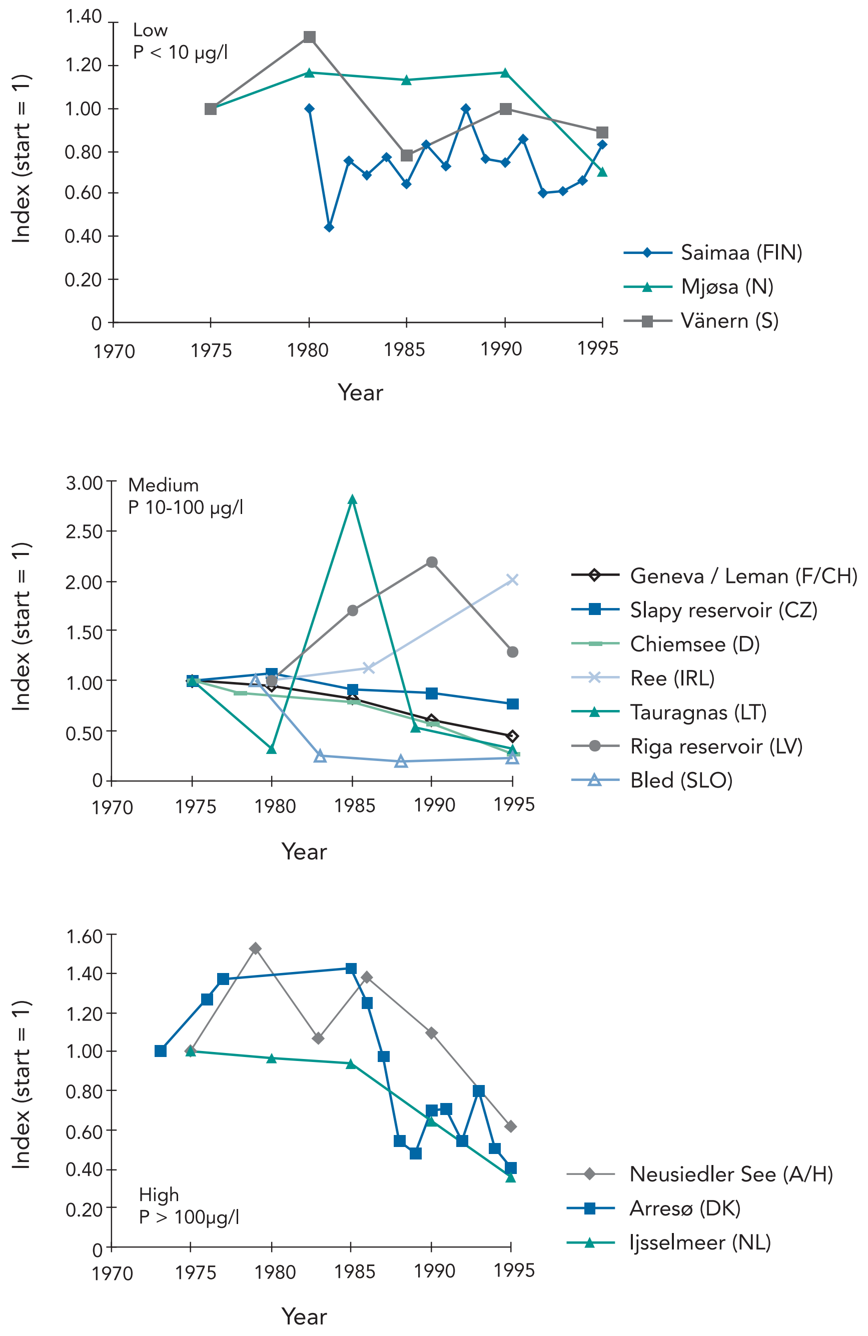
Notes: To avoid disproportional large influence from Danish and Finnish lakes, these have been weighted by factors of 0.25 and 0.1, respectively. The lakes in Figure 5.2 were selected for their long time-series rather than for their representativeness. Number of lakes per country: A(3), CH(2), CZ(1), D(4), DK(20), FIN(70), F(1), H(3), IRL(3), LT(1), LV(2), NL(2), N(3), P(1), S(9), SLO(1) be a need to reduce these contributions that usually arise from agriculture.

The current lack of good geographical and historical data makes regional comparisons at a European scale difficult and the currently available data are probably unrepresentative. This is because data are generally more available for lakes having problems than those without problems.

5.3. Ecological state in lakes and reservoirs

Eutrophication produces a shift in the biological structure of reservoirs and lakes. A growing phytoplankton community feeds on the increased amounts of available nutrients and produces a turbid environment, which affects higher life forms, including certain fish species. Decaying

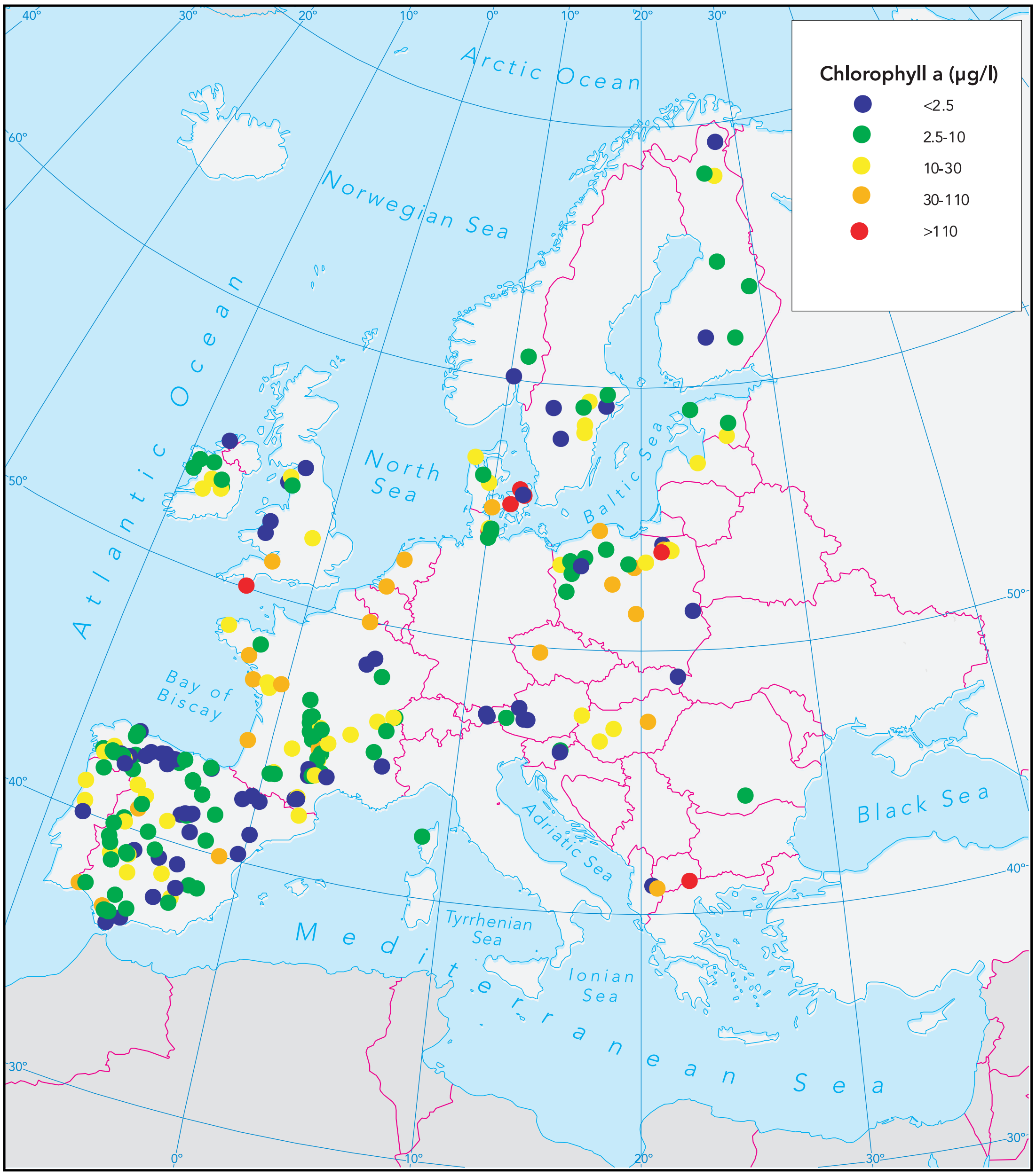
Figure 5.3 **Temporal trends in selected European lakes with low, medium and high total phosphorus concentrations. Source: ETC/IW**



phytoplankton, too abundant for the algal grazers to consume, reduces dissolved oxygen concentrations, which may become too low to support fish life and benthic invertebrates. The low dissolved oxygen concentrations also tend to enhance the release of additional phosphorus from the sediment, thereby increasing the available nutrients. Dissolved oxygen and pH in the surface layers often show significant daily variations, due to the response of the algal community to variations in light. Under these conditions, the ecosystem changes drastically and generally experiences a significant reduction in biodiversity. Nevertheless, it should be borne in mind that ultra-oligotrophic water bodies have a natural low diversity.

The amount of algae is often expressed in terms of the concentration of the photosynthetic pigment chlorophyll *a*. The reference (natural) level of chlorophyll *a* is usually below 3 µg l^{-1}. Chlorophyll is a very useful indicator for eutrophication and it is related to the concentration of phosphorus. Many European lakes have high levels of chlorophyll (Map 5.3) due to past and current emissions of nutrients.

Chlorophyll concentration in large or important lakes. Source: ETC/IW **Map 5.3**

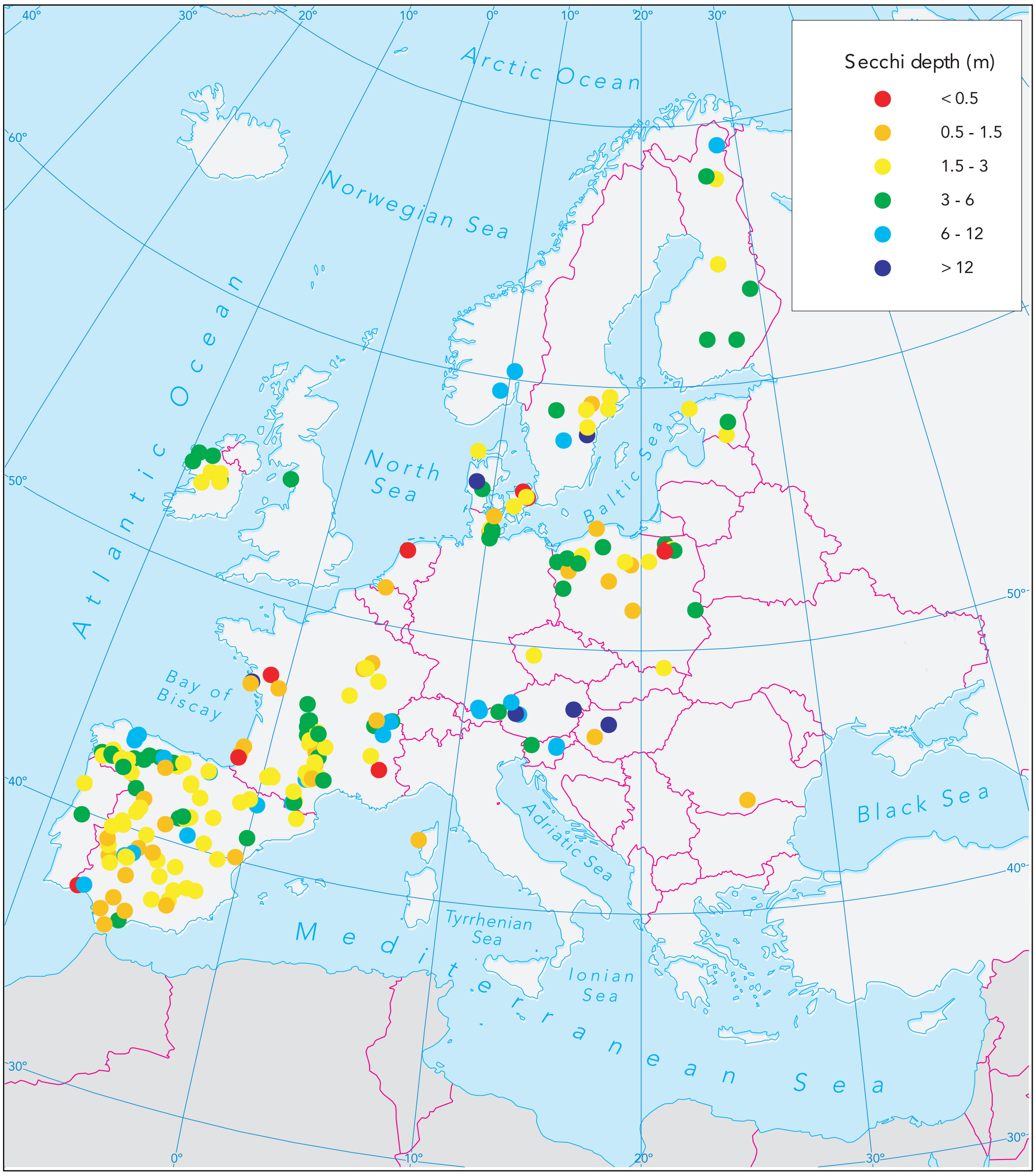


High concentrations of chlorophyll diminish the transparency of water. Turbid water is unpleasant from a recreational point of view, and the reduced availability of light degrades the conditions for bottom-dwelling plants and certain fish species. Transparency is measured as Secchi depth, the water depth where a 30 cm diameter white disc is no longer visible. When there is a top-to-bottom visibility, Secchi depth is obviously not a good indicator of transparency. The Secchi depth in large or important European lakes is shown in Map 5.4.

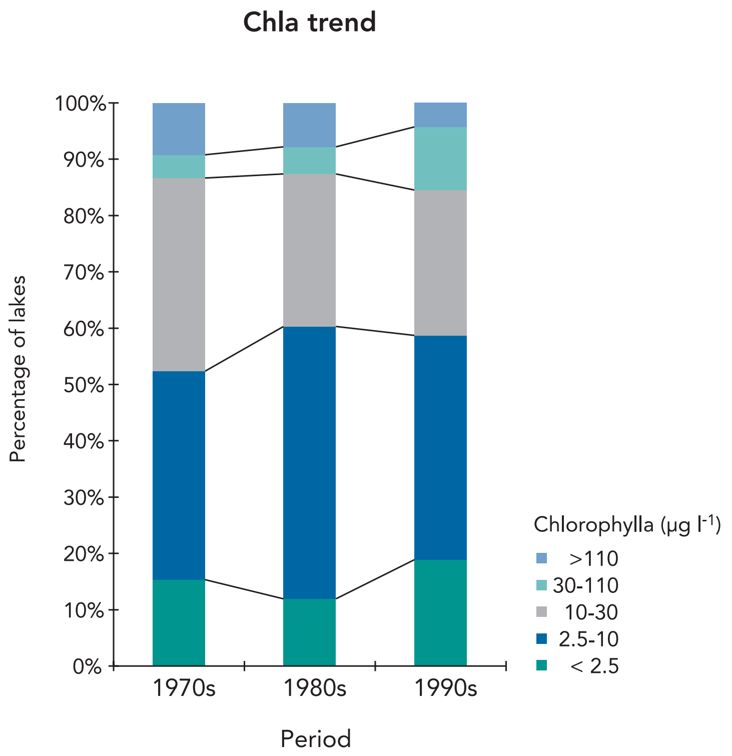
The biomass of algae has reduced in many lakes, as suggested by decreasing chlorophyll concentrations (Figure 5.4). The improvement is more pronounced in the previously most polluted lakes. Moderately polluted and clean lakes present, respectively, erratic or stable changes with time.

The recovery of previously polluted lakes with regard to both the total phosphorus concentrations and chlorophyll concentrations may take several years as illustrated by the Danish lake Furesøen (Figure 5.5).

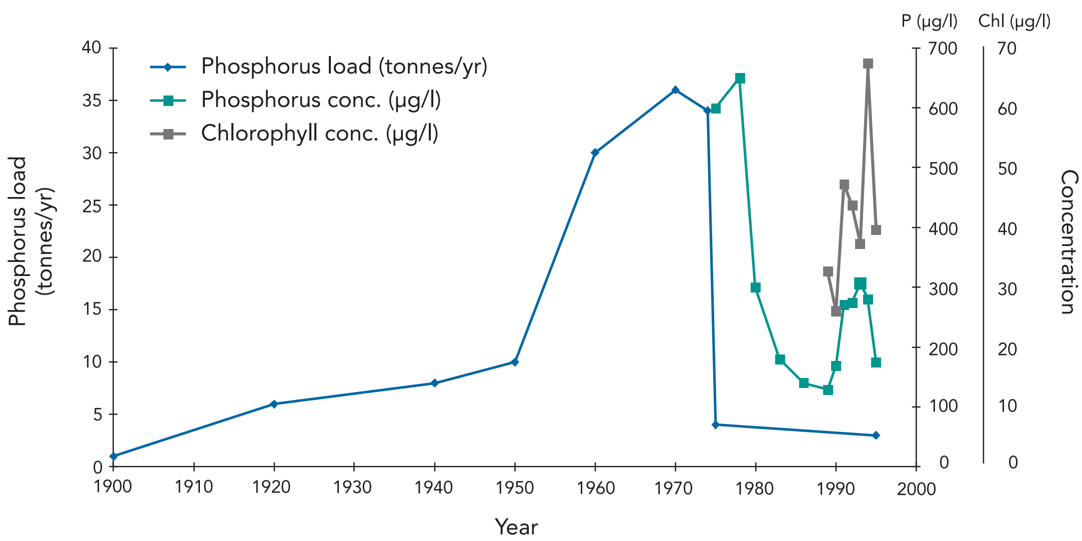
Map 5.4 **Transparency expressed as Secchi disk disappearance depth in large or important lakes. Source: ETC/IW**



Temporal change in distribution to chlorophyll a classes amongst selected European lakes Source: Compiled by ETC/IW from questionnaire returns **Figure 5.4**



Phosphorus load and in-lake concentrations of phosphorus and chlorophyll a in Lake Furesø, Denmark. Source: compiled by ETC/IW. **Figure 5.5**



Over this century the lake received increasing amounts of poorly treated urban waste water. In 1975 the discharge of waste water was stopped. Since then the lake has slowly recovered but it is still far from a near natural state, and there have been considerable fluctuations during the recovery period.

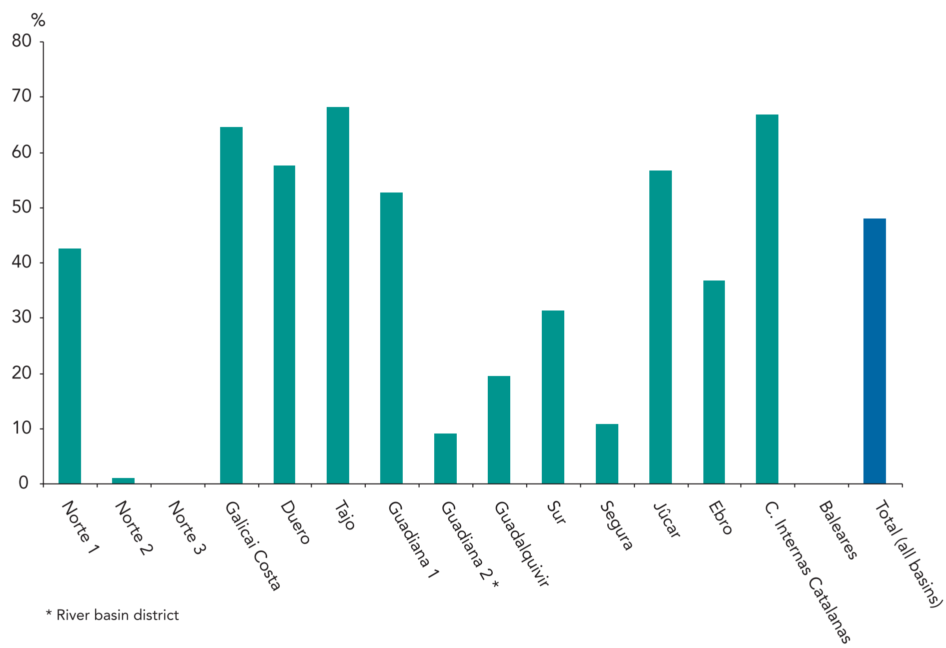
It has been estimated that nearly half of the total volume of Spanish reservoirs is severely impacted (eutrophic or hyper-eutrophic). In certain catchments such as the Tajo, up to two-thirds of the total volume may be affected (Figure 5.6). However, when the water quality is compared to the requirements for each reservoir's use, a slightly lower total volume (27 %) is found to be unsuitable for the desired use. This is nevertheless a significant impact on the reservoir resource in Spain.

5.4. Impacts due to eutrophication in lakes and reservoirs

In addition to the impact on the ecosystem, the effects of eutrophication cause problems for the use of water from lakes and reservoirs. Public water supply is particularly sensitive to eutrophication. It may lead to problems in the water treatment system such as the following (Meybeck *et al.*, 1987):

- Clogging of filters in water treatment plants, clogging of drip irrigation systems because of algae;
- undesirable tastes, odours and colour caused by algae;
- iron, manganese, ammonium, sulphur and carbon complexes, caused by hypolimnetic deoxygenation, which can cause problems in the treatment process and require elimination;
- seasonal and daily water quality variations, requiring frequent adjustment of the water treatment process (in particular, diurnal pH variations which cause problems in flocculation);
- formation of chlorophenols during chlorine disinfection due to phenolic substances liberated by cyanobacteria;

Figure 5.6 **Proportion of total reservoir volumes in Spain which are eutrophic/hyper-eutrophic (EEA, 1999)**



- increased chlorine requirements, formation of organochlorine compounds and a possible bacterial growth due to high dissolved organic matter;

- presence of toxins liberated by certain cyanobacteria;

- pipe corrosion.

Other kinds of usage are less sensitive to eutrophication, though it may cause problems for hydroelectric, irrigation or fish hatchery purposes.

Eutrophication may render the lake unsuitable for recreation because of the unpleasant appearance of water caused by high turbidity / low transparency, odours or algal masses. Furthermore the presence of toxic cyanobacteria may pose a health risk. During the summer of 1989, major blooms of toxic cyanobacteria were reported in many reservoirs in England, Finland, Norway and Sweden. This is believed to have been caused by a mild winter, high mid-summer temperatures and sunshine and a long period of stable weather in July. In Spain there has also been a decrease in the quality and value of fisheries with salmonid and coregonids being replaced by less valued cyprinid fish (UNECE, 1992).

5.5. Conclusions

The impacts of nutrients in still waters are usually caused by phosphorus. In unaffected lakes and reservoirs the concentration of phosphorus is below 25 µg l^{-1}. Large proportions of the lakes in most parts of Europe have phosphorus concentrations exceeding this limit, thus indicating a significant anthropogenic influence. Only in sparsely populated regions such as parts of the Nordic countries, Ireland, UK (Scotland) and central European mountainous regions is there a high proportion of lakes with lower phosphorus concentration. The impacts of excessive nutrient concentrations are widespread and includes both deterioration of the recreational value and implications for other uses of the water resource. Primarily because of improved treatment of urban waste water, there has been some improvement in some lakes since the 1970s. The improvements are reflected in the decreasing proportion of the most polluted lakes and reservoirs. However, the proportion of lakes with median and low levels of total phosphorus has remained quite stable. The state of European lakes and reservoirs is still markedly affected by anthropogenic nutrient pollution and the condition in many lakes is still far from satisfactory.

Excessive phytoplankton development in French reservoir (Villerest, Central France).
Photo:
Philippe Crouzet

6. Status and impacts of nutrients in rivers

6.1. Characteristics of European rivers

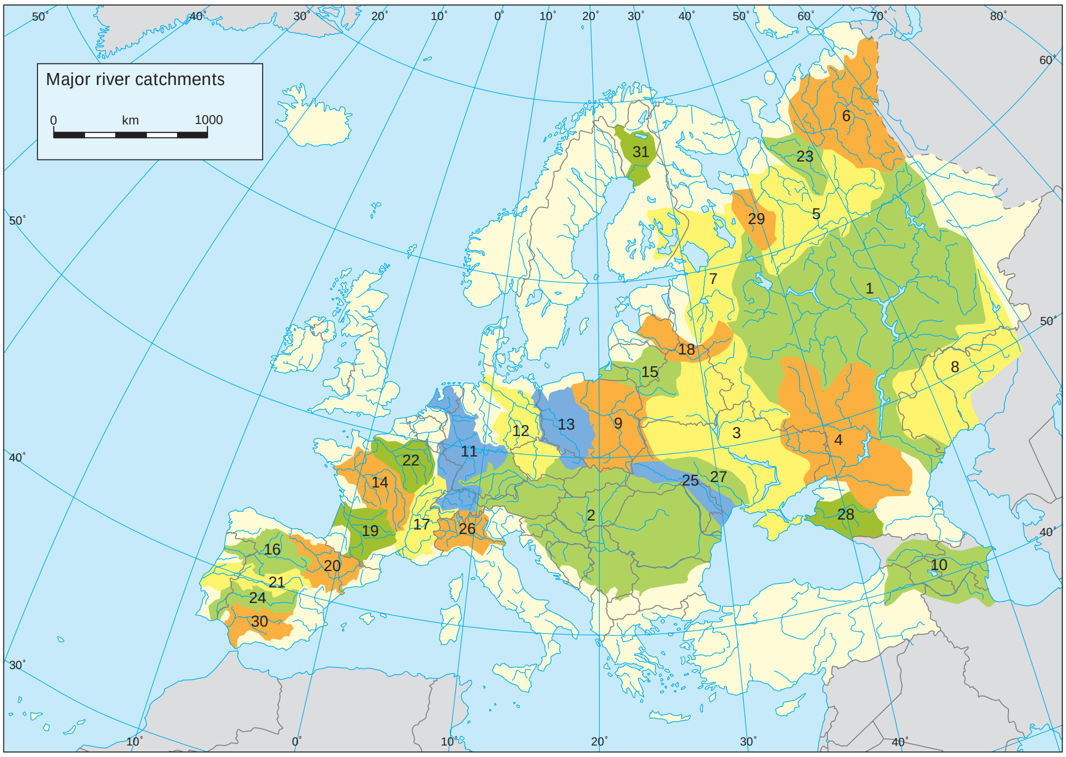
River catchments in Europe are particularly numerous because of the continent's geological structure and shape. According to world standards, only a few catchments are drained by very large rivers. Catchments exceeding 50 000 km^2 (which drain 6.33 million km^2, approximately 2/3 of the continent's area), equate to only 31 rivers, as shown in Map 6.1.

Most types of river systems, in terms of chemical composition, flow regime and human impact, are observed in Europe. For example, the main 31 river systems have a specific water flow ranging from 1.1 l s^{-1} km^{-2} (Guadalquivir, Spain) to 21 l s^{-1} km^{-2} (Po, Italy), with an average of 6.5 l s^{-1} km^{-2}.

According to the flow regime, the low water flow period may be in winter or in summer, resulting in different patterns of sensitivity to nutrients, and to eutrophication.

River regulation is a very important factor in relation to sensitivity of running waters to eutrophication. Small reservoirs and navigation works slow down the flow velocity and increase the residence time of water, thus favouring algal growth. Large reservoirs may act both as nutrient traps and sources of algal seeding to downstream waters. Small rivers, which would not naturally support planktonic growth, can, therefore, become eutrophic. Excessive macrophyte growth can also be observed at many sites and can cause many problems.

Map 6.1 **Large European river catchments exceeding 50 000 km^2. Source: EEA (1995)**
Basic catchment characteristics can be obtained from the table on the opposite page.



River	Country	Catchment area (10^3 km^2)	Mean discharge (km^3 $year^{-1}$)	Length (km)
1 Volga	Russian Federation	1360	230	3530
2 Danube	Germany, Austria, Slovak Republic, Hungary, Croatia, Federal Republic of Yugoslavia, Romania, Bulgaria, Ukraine, Switzerland*, Poland*, Italy*, Czech Republic*, Slovenia*, Bosnia-Herzegovina*, Albania*, Republic of Moldova*	817	205	2850
3 Dnepr	Russian Federation, Belarus, Ukraine	558	53	2270
4 Don	Russian Federation, Ukraine*	422	38	1870
5 Severnaya Dvina	Russian Federation	358	148	740
6 Pechora	Russian Federation	322	129	1810
7 Neva	Russian Federation, Finland*, Belarus*	281	79	75
8 Ural	Russian Federation, Kazakhstan	270	-	2540
9 Wisla	Poland, Slovak Republic*, Ukraine*, Belarus*	194	31	1050
10 Kura	Georgia, Turkey, Azerbaijan, Armenia*, Iran*	188	18	1360
11 Rhine	Switzerland, Austria, Germany, France, The Netherlands, Italy*, Luxembourg*, Belgium*	185	69	1320
12 Elbe	Czech Republic, Germany, Austria*, Poland*	148	24	1140
13 Oder	Czech Republic, Poland, Germany	119	16	850
14 Loire	France	118	32	1010
15 Neman	Belarus, Lithuania, Russian Federation, Poland*	98	22	960
16 Douro	Spain, Portugal	98	20	790
17 Rhône	Switzerland, France	96	54	810
18 Zapadnaya Dvina	Russian Federation, Belarus, Latvia, Lithuania*	88	21	1020
19 Garonne	France	85	21	575
20 Ebro	Spain	84	17	910
21 Tajo	Spain, Portugal	80	6	1010
22 Seine	France	79	16	780
23 Mezen	Russian Federation	78	26	970
24 Guadiana	Spain, Portugal	72	-	800
25 Dnestr	Ukraine, Republic of Moldova	72	10	1350
26 Po	Italy, Switzerland*	69	46	670
27 Yuzhnyy Bug	Ukraine	65	3	860
28 Kuban	Russian Federation	58	13	870
29 Onega	Russian Federation	57	18	420
30 Guadalquivir	Spain	57	2	675
31 Kemijoki	Finland	51	17	510

Note: * The country includes part of the catchment area but the main river does not run through it.

6.2. Criteria for assessment of nutrient levels in running waters

6.2.1. Reference values and criteria for nutrient assessment

If there were no human activities, water composition would be determined only by hydrological and geochemical factors. Apart from the run-off producing the discharge volume, the main processes involved are the weathering of bedrock minerals, (which determines the basic ionic composition of water), aerial deposition of dust and salts, (which contributes to the nitrogen load), and natural leaching of organic matter and nutrients from soils and decaying biological material. The major source of phosphorus in quasi-pristine rivers is considered to be soil erosion.

Only nitrogen and phosphorus are considered in this chapter as both are major nutrients whose concentrations in water are affected by anthropogenic activities. Silicate, carbonate and calcium are key nutrients as well, and they determine the type of eutrophication that can be observed. However, they are not derived significantly from human activities.

The natural levels in nitrogen and phosphorus species are not accurately known in European river systems. These river systems have been impacted by man for centuries, long before any measurement could be performed. Therefore, the natural levels must be derived from data presently monitored in quasi-pristine areas or deduced by indirect means. This second estimation method is necessary since the present pristine or quasi-pristine areas are situated in mountains or in northern regions, and do not necessarily represent the natural values of lowland or southern river systems.

Two criteria are relevant for water quality assessment:

1. Natural or pristine values supposing low human impact. They represent reference values for the environment. These values may be a result of scientific assessments when suitable rivers can be monitored or the result of modelling when only impacted environments are avail-

Table 6.1 **Reference values for nitrogen compounds. Concentrations in mg N l^{-1}.**

	Nitrate	Nitrite	Ammonium	Un-ionised ammonia
Low value	0.1 (*)	0.001 (**)	0.015(**)	0.0041(***)
High value	1 (*)	0.015(**)	?	0.0206(***)

Sources: (*) Meybeck, 1986; (**) Meybeck, 1982; (***) Agences de l'Eau, 1997; Train, 1979.

Table 6.2 **Standard values for nitrogen compounds. Concentrations in mg N l^{-1}.**

	Nitrate	Nitrite	Ammonium	Un-ionised ammonia
Guideline value	5.65 (*) (25 mg NO_3 l^{-1})		0.039 (*) (0.05 mg NH_4 l^{-1})	0.0041, 0.0206 (**) (respectively, salmonid and cyprinid waters)
Mandatory value	11.3 (*) (50 mg NO_3 l^{-1})	0.009, 0.09 (**) (respectively, salmonid and cyprinid waters)	0.389(*) (0.5 mg NH_4 l^{-1})	0.165, 1.65 (**) (respectively, salmonid and cyprinid waters)

Sources: (*) Directive 80/778/EEC, (**) Directive 78/659/EEC.

able. When recent data are projected to the past, the choice of the date may lead to a large range of reference values.

2. Standard values express the legal suitability of a water mass to various human or other uses. The threshold concentration for these criteria vary considerably, reflecting the different sensitivities of the natural environment on the one hand and the uses on the other.

6.2.1.1 Assessment values for nitrogen compounds

Reference values: Most nitrogen compounds present in river water are in the dissolved phase. Therefore, a reasonably good comparability of data may be assumed. The average level of nitrogen in pristine rivers is reported as 0.1 mg $N\text{-}NO_3$ l^{-1} and 0.001 to 0.015 mg $N\text{-}NO_2$ l^{-1} (Meybeck, 1982). These reference values can nevertheless be quite variable, since 1 mg $N\text{-}NO_3$ l^{-1} is a maximum value for rivers draining non-polluted catchments (Meybeck, 1986). The amount that eventually enters rivers is variable, with the areal loss coefficients for nitrogen laying in the range between 1 to 25 kg N ha^{-1} $year^{-1}$ (EEA, 1995) for temperate ecosystems. For an average water run-off in the range 100 to 500 mm $year^{-1}$, the resulting concentrations would be expected to reach a minimum of 1 to 0.2 mg N l^{-1}, respectively.

Reference values for ammonium are not often reported, principally because of analytical difficulties. Also ammonium is only a transient component in the N cycle in pristine waters.

A summary of the potential reference values for nitrogen compounds are given in Table 6.1.

Standard values: The values stipulated in Directives are used in this report (Table 6.2).

6.2.1.2 Assessment values for phosphorus

Reference values: The quantity of phosphorus in precipitation is normally low, in the range 10 to 60 kg total dissolved P km^{-2} $year^{-1}$ (Billen *et al.*, 1994), these rates being affected by human activities. In pristine conditions, practically the only sources of phosphorus are weathering that removes phosphorus from rocks, and airborne soil dusts (deposited directly by rain), which may represent a substantial input in uncontaminated areas (see Chapter 3.7 and Berner and Berner, 1996). However, dust re-deposition is not a primary source, since the corresponding loads are not necessarily exported to rivers. The leaching rate depends on bedrock chemical composition and denudation of soils. The natural loading rates to rivers are, therefore, reported as large ranges (0.05 to 0.12 kg total P ha^{-1} $year^{-1}$) in temperate areas (Billen *et al.*, 1994; EEA, 1995).

Unlike nitrogen compounds, phosphorus species may be strongly bound to suspended or settled material. The phosphorus content of rivers is thus governed by a 'spiral pattern' (Amoros and Petts, 1993) involving the sediment phase to which phosphorus species are bound and released from. Also as phosphorus is the most limiting nutrient in pristine areas, most of the available phosphorus is readily incorporated into organic material. Therefore, the ratio of soluble reactive phosphorus (SRP) to total phosphorus (total P) is shifted towards zero. This produces natural values in a range 0 to 0.010 mg P-SRP l^{-1}, and (0 to 0.005) to 0.050 mg total P l^{-1}, depending on the proportion of biological or inert material taken in the sample. Outside the period of biological activity, the P-SRP / total P ratios are in the range 0.4 to 0.7 with a median concentration of 0.017 mg total P l^{-1} (Meybeck, 1986).

Standard values: The standards in the Drinking Water Directive are extremely high, with respect to natural values: respectively 0.175 and 2.18 mg total P l^{-1} (values computed from the Directive limits which are expressed as P_2O_5).

Since the phosphorus standards are not relevant for eutrophication assessment, a very simple rule has been used. In the range of concentration in which phosphorus is the limiting factor, the biomass produced shows a linear relationship with the phosphorus concentration. According to plant composition and field observations, a very useful rule of thumb is that each mg of phosphorus may yield 1 mg of chlorophyll a. This value is obviously approximate, but practical for assessment purposes. It is supported by the OECD surveys (OECD, 1982).

6.2.1.3. Chlorophyll reference values

It is very difficult to propose reliable reference values for chlorophyll *a* in rivers. First, plant growth is a seasonal phenomenon that requires the choice of a standard period for data assessment. Also river water quality monitoring programmes do not operate in the same way for chlorophyll *a* assessment. Most monitor chlorophyll *a* only during the spring and summer, possibly measuring a single value during winter. Second, such values can represent only the planktonic content, which is not the only sign of eutrophication in rivers. Moreover, a classification of the algal content dealing only with river issues is not accurate when the river enters a reservoir where lake conditions prevail.

A specific assessment system is used in France, (Agences de l'Eau, 1997), where river eutrophication was long ago recognised as a widespread water quality problem. According to this system eutrophication class limits are established from the chlorophyll content of the river water indicating the level of planktonic proliferation and the assumption that 1 mg P 1^{-1} may yield 1 mg chlorophyll a 1^{-1}. For the purpose of this assessment report class limits are 10, 50, 100, 150 and 200 μg P 1^{-1} indicating pristine values, low, significant, high, excessive and hypereutrophication, respectively. These values are intended to be compared with the 90 percentile of the monitored values. When the 90 percentile is not calculable, which is the case in this report, the average of maxima over several years has been used instead.

6.3. Methods for the evaluation of nutrient related nuisances throughout Europe

Most countries have reports on nutrients or eutrophication. Respectively, 19, 18 and 20 countries out of 28 that responded to the ETC/IW questionnaire (Appendix A) indicated that national, regional and nutrient oriented reports are published.

Six different methods have been identified from the questionnaire responses. The most commonly used are, by decreasing order of occurrence: comparison of nutrient data with standards (14 countries), planktonic chlorophyll measurements (11), oxygen and pH perturbations (10), macrophyte observations (7), nutrients loads and modelling (6) and planktonic species counting, practised only in 6 countries. A country may use several methods concurrently (Table 6.3).

6.4. State and trends of nutrients in European rivers

6.4.1. Nitrogen compounds

6.4.1.1. Nitrate

Inorganic nitrogen constitutes on average 88% of total nitrogen in European river water, with the largest share being nitrate nitrogen (EEA, 1995). The data collected for this report allowed the calculation of an updated set of nitrate-N to total N ratios for 21 countries. These ratios were based on

Box 6.1: Historical values for eutrophication.

Since this report deals with nutrients and eutrophication, historical or reference values for plankton abundance or macrophyte cover are of interest. Historical values are extremely scarce.

Based upon plankton counts in 1920 in the downstream reaches of the Loire river (equivalent to 1-10 μg Chl a l^{-1}) historical concentrations of SRP have been estimated at 1-5 μg l^{-1}. Present day values are much higher - 100-400 μg Chl a l^{-1} and 60-200 μg SRP l^{-1}, respectively.

Although plant growth in quantities causing nuisances have been reported since time immemorial, the most important feature of eutrophic river systems is that eutrophication phenomena nowadays occur practically every year and last several months. In contrast, historically these phenomena were very infrequent and only lasted a few days.

Methods used for eutrophication assessment.
Source: Compiled by ETC/IW from questionnaire returns.

Table 6.3

Country	Use of method		Type of methods possibly used					
	Existence of a national classification system.	Number of methods in use	Comparison of sampling stations data with standard values	Nutrients loads and modelling	Plankton chlorophyll measure-ments	Plankton species counts	Macrophyte observations	Oxygen/pH variation measure-ments
Austria	NO	4	(X)		(X)		(X)	(X)
Belgium – Flanders	NO	1	(X)					
Bulgaria	NO	1	(X)					
Czech Rep.	NO	5		(X)	(X)	(X)	(X)	(X)
Denmark	YES	2	(X)	(X)				
Estonia	NO	4	(X)	(X)		(X)	(X)	
Finland	YES	none						
France	YES	4	(X)		(X)		(X)	(X)
Germany	YES	6	(X)	(X)	(X)	(X)	(X)	(X)
Hungary	YES	1			(X)			
Ireland	YES	2					(X)	(X)
Latvia	NO	4	(X)		(X)	(X)		(X)
Lithuania	NO	1	(X)					
The Netherlands	NO	6	(X)	(X)	(X)	(X)	(X)	(X)
Norway	YES	none						
Poland	NO	2	(X)		(X)			
Romania	YES	3	(X)		(X)			(X)
Slovak Rep.	NO	2		(X)	(X)			
Slovenia	unknown	4	(X)		(X)	(X)		(X)
Sweden	YES	3		(X)			(X)	(X)
Switzerland	NO	none	(X)					

(X) method used in the country. Only countries having responded are indicated, methods obtained from literature are not quoted.

Comments

Austria: Also assessment of ecological water quality. A classification system for eutrophication of rivers based on algae is under preparation.

Denmark: National classification by biotic index.

France: Regarding planktonic eutrophication, the most widely used method is the coupling of planktonic chlorophyll determination, pH measurements and oxygen saturation computation. In some cases, daily cycles of pH and oxygen may be available.

Germany: National classification method under development, taking into account Chl-a, P total, oxygen, pH, intensity of plant growth.

Norway: 155 rivers have been classified yearly since 1990 (map sent)

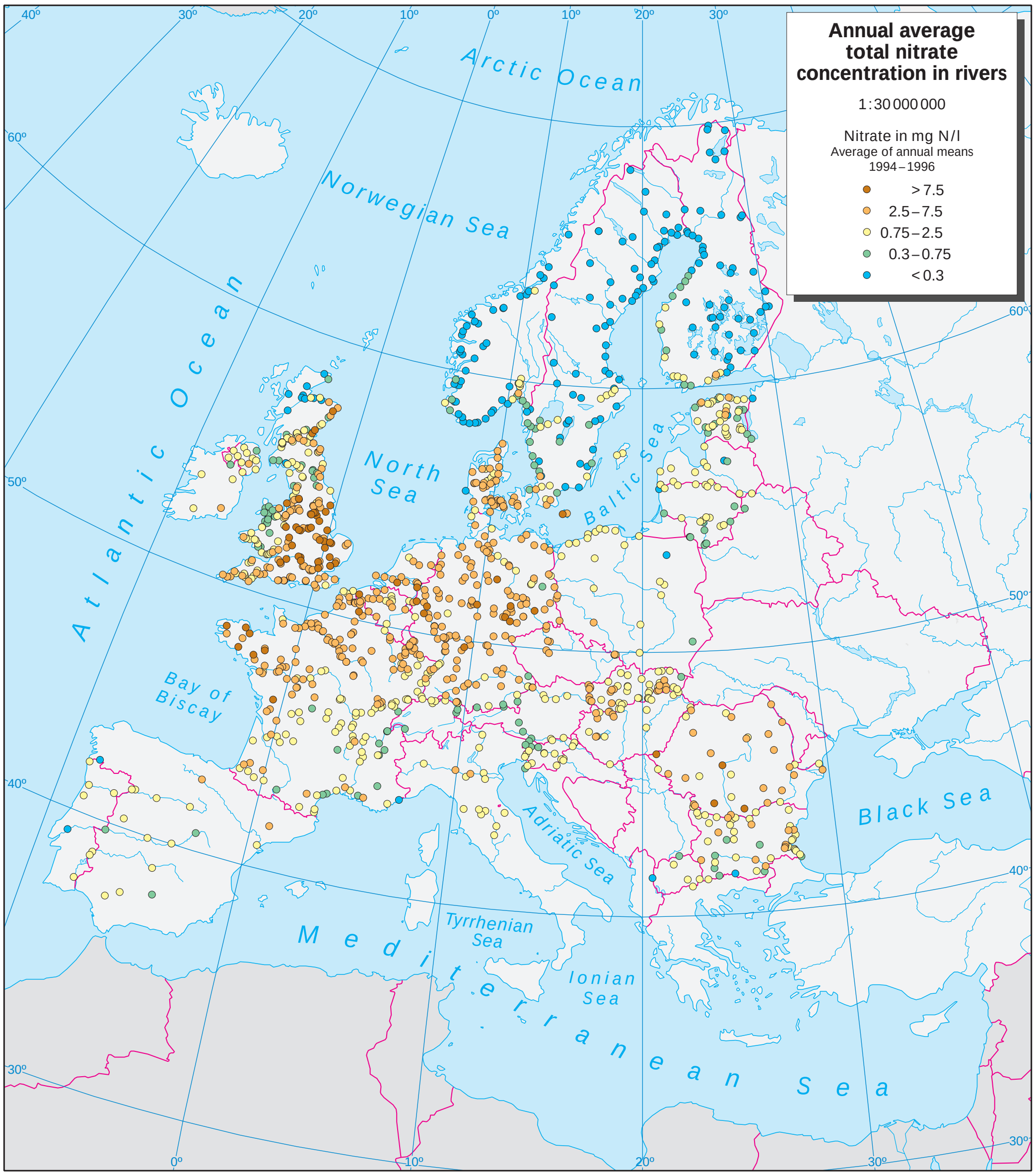
Sweden: The Swedish quality criteria for lakes and watercourses are commonly used for reporting and assessing water quality data. These criteria are currently being revised.

inter-annual averages and ranged between 28-89%. The observed range varies greatly within a country. In Denmark, over 23 years, the ratio has ranged from 52-84% (average 80%). In Norway, over 17 years, it has ranged between 21-81% (average 48%). In France and Hungary, the values lie between 42-70%, averaging around 60%. As a general rule, nitrate nitrogen constitutes 2/3-4/5 of total nitrogen.

The nitrate concentration in European rivers between 1992 to 1996 are shown in Map 6.1. River monitoring stations are classified into five concentration ranges. The data on which the map is based is derived from the ETC/IW questionnaires. Data collected under the EC Exchange of Information Decision (77/795/EEC) were also included.

Apart from the rivers in Nordic countries, (where 70% of points have concentrations below 0.3 mg l^{-1}, and 39.4% below 0.1 mg l^{-1}), 68 % of all the stations for which information was obtained had annual average nitrate concentrations exceeding 1 mg N l^{-1} over the period 1992 to 1996. Peak concentrations exceeding 7.5 mg l^{-1} were observed at about 15% of stations. The highest concentrations, over 20 mg N l^{-1}, were found in the northern part of western Europe reflecting the intensive agriculture in these regions. High concentrations also occur

Map 6.1 **Nitrate in European rivers 1992 to 1996. Sources: Compiled by ETC/IW from questionnaire returns and data reported under the Exchange of Information Decision 77/795/EEC.**



in eastern Europe, whilst southern Europe seems to have lower concentrations.

The distribution of available data is presented in Table 6.4. This table gives the concentration which is observed in 10%, 25%, etc., of monitoring stations. Thus in the period 1992 to 1996, 25% of stations recorded an annual average concentration of 0.72 mg $N\text{-}NO_3\,l^{-1}$ or less, and 90% of stations had concentrations of 5.89 mg $N\text{-}NO_3\,l^{-1}$ or less. Relatively clean waters are, thus, found at 25% of stations, whereas contaminated waters are observed in 10-25% of stations. The very contaminated stations equate to 1-10% of the total, which is likely to be an optimistic assessment, since the most contaminated rivers are small rivers, which are under-represented in this assessment.

The annual variability of nitrate concentration in rivers is very dependent on rainfall. Therefore, the annual average alone is not a good indicator of compliance with standards. The proportion of stations exceeding the assessment criteria based on the guideline and maximum admissible concentrations values specified by the Drinking Water Directive, of 25 and 50 mg $NO_3\,l^{-1}$ respectively, is given In Table 6.5.

The main source of nitrate is generally non point sources from agricultural activities

Table 6.4

Descriptive statistics for averages of annual mean and maximum nitrate nitrogen concentrations in European rivers 1975 to 1980, and 1992 to 1996. Data from 30 countries. Source: Compiled by ETC/IW from questionnaire returns

	Number of stations	Percentage of river stations with concentrations not exceeding (mg $N\text{-}NO_3\,l^{-1}$)					
$N\text{-}NO_3$ (mg l^{-1})		10 %	25 %	50 %	75 %	90 %	99 %
1975-1980 (average of annual means)	697	0.193	0.70	1.54	3.19	6.05	11.8
1992-1996 (average of annual means)	1525	0.193	0.720	1.73	3.53	5.89	9.78
1975-1980 (average of annual maxima)	685	0.392	1.23	3.12	5.66	11.40	24.4
1992-1996 (average of annual maxima)	1352	0.341	1.31	2.74	5.37	9.36	18.5

This table includes all stations, even if they are not reported on the maps because of lack of co-ordinates. For some stations, data include nitrate and nitrite concentration. In medium grey shading are the values worse then the guideline value for nitrate, and in heavy grey shading the values worse than the mandatory values given in the Drinking Water Directive. In light grey shading are values less than or equal to pristine values, arbitrarily considered to be at 1 mg N l^{-1}.

Table 6.5

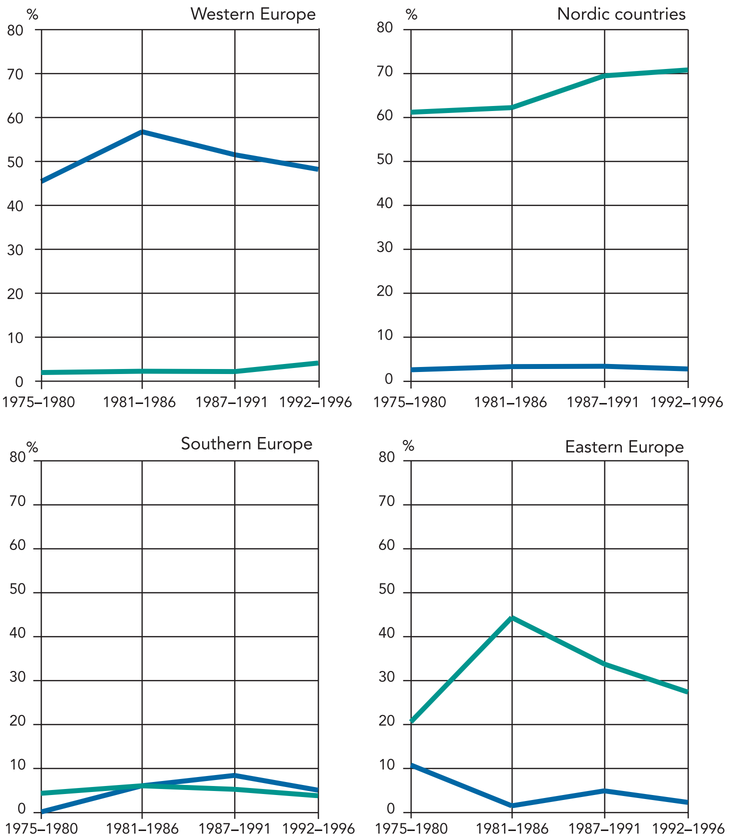
Proportion of river stations exceeding given assessment criteria. Source: Compiled by ETC/IW from questionnaire returns

Concentration values from the Drinking Water Directive[4]	Period 1975 : 1980		Period 1992 : 1996	
	Annual average	Annual max.	Annual average	Annual max.
Guide concentration 25 mg l^{-1} NO_3 (= 5.65 mg $N\text{-}NO_3\,l^{-1}$)	11 %	25 %	11 %	30 %
Maximum Allowable Concentration 50 mg l^{-1} NO_3 (= 11.3 mg $N\text{-}NO_3\,l^{-1}$)	1.4 %	8 %	0.6 %	6 %

This table includes all stations, even if they are not reported on the maps because of lack of co-ordinates. For some stations, data include nitrate and nitrite concentration.

[4] It should be noted that this assessment does not of course imply that the rivers assessed would be used directly for drinking water without prior treatment. The 'standards' in this context are used merely as assessment criteria against which comparison of river quality within Europe can be made. No judgement is made or implied about legal compliance or otherwise with standards.

Figure 6.1 **Evolution of mean nitrate expressed as percentage of stations according to their concentration values. (Data for 30 countries).[5] Source: Compiled by ETC/IW from questionnaire returns**



Number of stations per country group

Period	*WE*	*NO*	*SO*	*EA*
1975 – 1980	459	115	22	101
1981 – 1986	621	150	65	132
1987 – 1991	688	176	94	162
1992 – 1996	970	215	78	262

percentage of sampling stations with average under 0.3 mg N-NO_3/l

percentage of sampling stations with average over 2.5 mg N-NO_3/l

[5] Comments on Figure 6.1: For nitrate, the annual average is a good indicator of water quality issues, though it should be interpreted with the associated maximum values. In western countries, bad quality points are majority, and uncontaminated points seldom exist, which is the case for southern countries as well. In this latter group, the majority of points present an acceptable quality level. The eastern countries form a heterogeneous group, which locally behave like the western countries. First period data are purely indicative

which is highly dependent on precipitation. Generally speaking, nitrate release from soils requires substantial rainfall and permanent moisture of soils, that enhance soil nitrogen mineralisation. Anyway, as a result of weather fluctuations, nitrate concentrations vary widely between years, and a change observed during one period does not necessarily reflect changes in human activities. Computations performed on the available data sets suggested that in the period ~1970 to 1985, nitrate concentrations increased in 25-50% of the stations at rates in the range 1-10% per year. From 1987 to 1991, the number of points where the quality improves is offset by a similar number where it worsens, suggesting that the degradation rate has slowed down.

The data used suggest that, after two decades of rapid increase, annual maximum concentrations are approaching a steady state or even decreasing in western European river stations. At the same time, the minimum values are tending to increase at all stations, including Nordic river stations. This increase has not yet changed the proportion of points belonging to the best quality class, the concentrations observed being still very low. This tendency has already been reported (EEA, 1995).

To assess the changes in nitrate concentrations, countries have been grouped into four geographic regions using the earlier definitions of Nordic countries, southern, western and eastern Europe. Data have been assessed on a 5 years basis and the resulting trends are shown in Figure 6.1.

Nitrate represents a potential threat to water quality, although its environmental impacts are not so dramatic as those due to phosphorus or un-ionised ammonia. The points where excessive nitrate concentrations inhibit water uses represent only a few percent of the total resources. However, the proportion of points where nitrate contamination is negligible is diminishing. Excessive nitrate is, therefore, a good indication of general contamination of waters, including groundwater. Its evolution with time is thus a possible indicator of sustainable use of waters in Europe.

Table 6.6

Descriptive statistics for averages of annual mean and maximum ammonium nitrogen concentrations at European river stations 1975-1980 and 1992-1996. Data from 25 countries. Source: Compiled by ETC/IW from questionnaire returns

$N\text{-}NH_4$ (mg l^{-1})	Number of stations	Percentage of river stations with concentrations not exceeding (mg $N\text{-}NH_4$ l^{-1}) 10%	25%	50%	75%	90%	99%
1975-1980 (average of annual means)	688	0.027	0.065	0.163	0.506	1.65	9.14
1992-1996 (average of annual means)	1345	0.020	0.057	0.106	0.233	0.860	5.37
1975-1980 (average of annual maxima)	654	0.071	0.197	0.521	1.41	4.40	24.1
1992-1996 (average of annual maxima)	1308	0.063	0.160	0.320	0.760	2.31	13.4

Note: This table includes all stations, even if not reported on maps because of lack of co-ordinates. The light grey area includes stations with ammonium concentrations below the guideline value of Directive 80/778/EEC (see Table 6.2) and the dark shaded area those stations with concentrations above the mandatory value.

Table 6.7

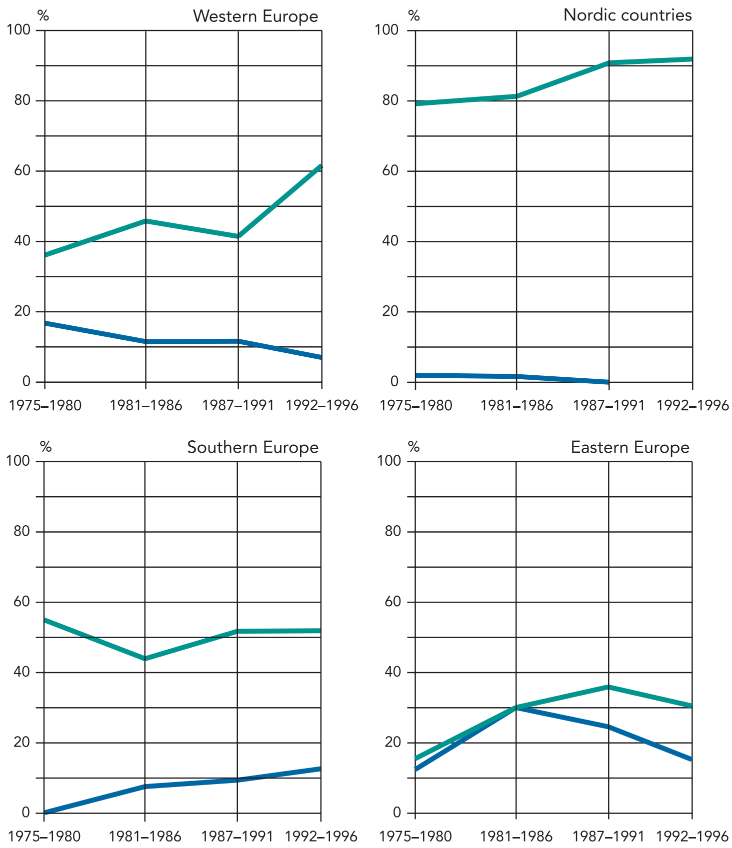
Percentage of river stations exceeding given assessment criteria for ammonium[6]. Source: Compiled by ETC/IW from questionnaire returns

	Period 1975-1980		Period 1992-1996	
Concentration values from the 80/778/EEC Drinking Water Directive	Annual average	Annual max.	Annual average	Annual max.
Guide level 0.05 mg NH_4 l^{-1} (0.039 mg $N\text{-}NH_4$ l^{-1})	86%	93%	82%	93%
Maximum allowable concentration 0.5 mg NH_4 l^{-1} (0.389 mg l^{-1} $N\text{-}NH_4$)	30%	59%	16%	42%
Concentration values from the 78/659/EEC Freshwater Fish Directive				
Guide level (Salmonids) 0.04 mg NH_4 l^{-1} (0.031 mg $N\text{-}NH_4$ l^{-1}) (*)	88%	95%	85%	94%
Guide level. (Cyprinids) 0.2 mg NH_4 l^{-1} (0.16 mg $N\text{-}NH_4$ l^{-1}) (*)	51%	79%	36%	75%
Mandatory level 1 mg NH_4 l^{-1} (0.78 mg $N\text{-}NH_4$ l^{-1})	18%	37%	11%	24%

(*) Guide concentration for salmonids is close to guide concentration for drinking water (0.031 instead of 0.039 $N\text{-}NH_4$ mg l^{-1}) This table includes all stations, even if not reported on maps because of the lack of co-ordinates.

[6] It should be noted that this assessment does not of course imply that the rivers assessed would be used directly for drinking water without prior treatment or designated as either salmonid or cyprinid rivers under the terms of the Freshwater Fish Directive. The 'standards' in this context are used merely as assessment criteria against which comparison of river quality within Europe can be made. no judgement is made or implied about legal compliance or otherwise with standards.

Figure 6.2 **Trends of ammonium in European rivers, expressed as percentage of stations according to their annual maximum concentration level. Source: Compiled by ETC/IW from questionnaire returns**



Number of stations per country group

Period	*WE*	*NO*	*SO*	*EA*
1975 – 1980	459	100	19	76
1981 – 1986	486	122	65	110
1987 – 1991	680	141	84	125
1992 – 1996	817	172	78	241

percentage of sampling stations with maximum under 0.4 mg $N\text{-}NH_4$/l

percentage of sampling stations with maximum over 3.1 mg $N\text{-}NH_4$/l

6.4.1.2. Ammonium and un-ionised ammonia

The presence of ammonium in rivers is normally from sewage effluents from cities or run-off (permanent or accidental spills) from fields where animal manure is spread. Ammonium is the preferred nitrogen source for many plants, since its integration into living material requires less energy than nitrate does. Ammonium levels in natural waters, therefore, vary rapidly. Total ammonium is highly undesirable both as an oxygen consumer, and because it is the basis of chloramine production during the chlorination stages of drinking water preparation. The toxic effect of ammonium on aquatic fauna depends on pH, total dissolved solids and temperature which govern the proportion of toxic un-ionised ammonia in water (see Section 4.2.1).

The distribution of ammonium concentrations at river stations over two periods (1975 to 1980, and 1992 to 1996) is given in Table 6.6. In most countries, a significant proportion of sampling stations have average and maximum concentrations which are greater than 1.2 mg $N\text{-}NH_4\ l^{-1}$, and may be over 6.2 mg $N\text{-}NH_4\ l^{-1}$. This is especially the case for many eastern European countries: Bulgaria, Czech Republic, Estonia, Lithuania, Poland, Romania and Slovak Republic (no data from the former Soviet Union). In Nordic countries, values over 1.2 mg $N\text{-}NH_4\ l^{-1}$ are exceptional. In southern countries, (Bosnia-Herzegovina, Former Yugoslav Republic of Macedonia, Italy, Spain, Slovenia), 1/4 to 1/3 of points are of bad quality with respect to ammonium. In western countries, only Austria and Switzerland have no values above 1.2 mg $N\text{-}NH_4\ l^{-1}$. The worst cases are found in Belgium and the Netherlands, although Denmark, France and United Kingdom have a significant number of high concentration values. This is reflected in the percentage of stations where the concentration exceeds the values laid down in the Drinking Water Directive (Table 6.7).

The trends of ammonium concentrations in rivers are quite close to those reported for organic matter (EEA, 1998). In western and Nordic countries (see Figure 6.2) the reported stations with high ammonium concentrations are improving, and those with low concentrations are getting worse. In southern countries the overall situation is slowly worsening and in eastern countries the proportions of both good and quality stations are falling.

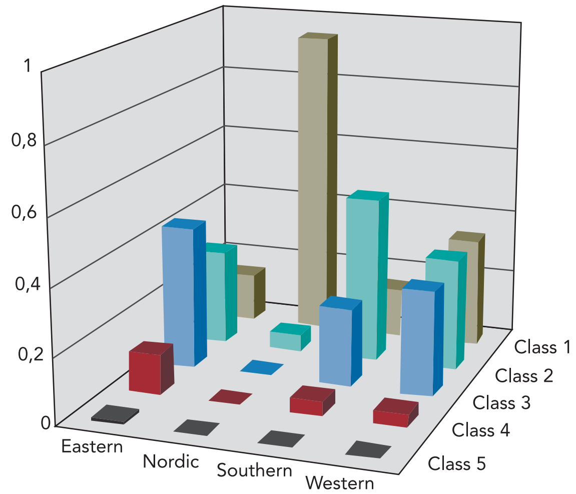
The assessment of the potential toxic effects of ammonium requires the un-ionised fraction to be computed. However, the available data only permit partial evaluation. This has been carried out assuming that peak pH would occur in summer when maximum ammonium is unlikely to be observed. Therefore, the assessment of un-ionised ammonium has been done by coupling *maximum pH* and *average ammonium* on the one hand, and *average values* for both parameters on the other hand. These results are taken to represent *pessimistic* and *optimistic* un-ionised ammonium concentrations, respectively. For presentation, the data have been assigned to classes according to the threshold values[7] explained in Section 6.2.1, and Table 6.2.

Because of the lack of data in many countries, especially the southern EU15 countries, only data from 22 countries have been analysed, this number falling, respectively, to 20 and 18 for the two most recent periods. Even if the data are restricted to one part of Europe, it is clear that the potential threat from high pH and elevated ammonium concentrations is far from negligible, with the exception of the Nordic countries where less than 1 % of the data exceed Class 2 (imperative value). The distribution of river stations per country group is represented in Figure 6.3 considering the 'pessimistic' approach. This graph suggests that more attention should be given to the monitoring of this form of ammonium, since it represents a greater factor of low quality than ammonium itself.

6.4.2. Phosphorus compounds

Phosphorus is measured both as total phosphorus (total-P) and SRP (soluble reactive phosphorus). The values of total-P include more or less suspended material, and are very dependent on the suspended solids behaviour in the river. Soluble phosphorus is readily incorporated in plant material. With exception of flood periods, when phosphorus contained in sediments is resuspended, phosphorus concentrations

Un-ionised ammonia in European rivers: Proportion of river stations according to their quality class.[8] Source: Compiled by ETC/IW from questionnaire returns **Figure 6.3**



[7] Class tresholds: Class 1/2 = 0.0041.mg N^{-1}: Class 2/3 = 0.0206 mg N^{-1}: Class 3/4 = 0.165 mg N^{-1}: Class 4/5 = 1.65 mg N^{-1}

[8] No country from EU15 are represented in the 'southern' group, since no suitable data were obtained from these countries. Available data do not permit computation of confidence limits for estimates. Computations by author of the chapter.

Table 6.8 **Descriptive statistics for averages of soluble (25 countries) and total phosphorus (24 countries). Annual average concentrations in European rivers 1975-1980 and 1992-1996. Source: Compiled by ETC/IW from questionnaire returns**

Note: This table includes all stations, even if they are not reported on the maps because of the lack of co-ordinates. An improvement can be observed between the two periods.

	Number of stations	Percentage of river stations with concentrations not exceeding (mg l^{-1} P)					
		10%	25%	50%	75%	90%	99%
1975-1980 Total phosphorus	105	0.086	0.150	0.317	0.683	1.020	2.834
1975-1980 Soluble phosphorus	657	0.007	0.032	0.091	0.276	0.811	2.832
1992-1996 Total phosphorus	546	0.050	0.100	0.172	0.290	0.576	2.219
1992-1996 Soluble phosphorus	1404	0.004	0.027	0.060	0.132	0.383	1.603

in rivers follow a dilution pattern. Hence, the maximum concentration is expected during low-flow periods.

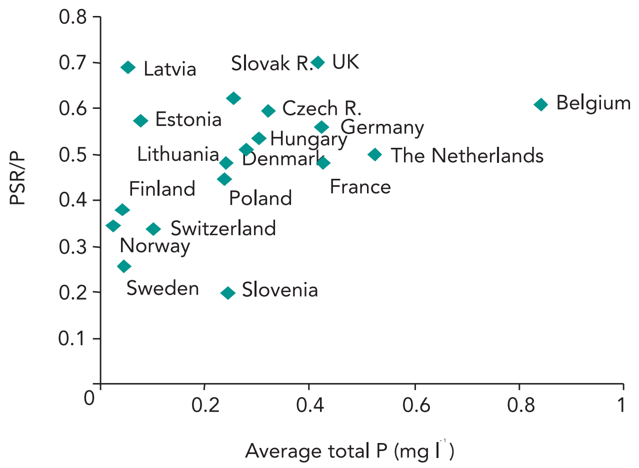
Monitored concentrations of SRP are widely influenced by the trophic state of the water body. During intense algal growth periods, the SRP content may be dramatically lowered by algal uptake. Therefore, although annual mean concentration is not a fully satisfactory indicator of the eutrophication risk in rivers, it is preferred to the annual maximum, particularly when the date of sampling is unknown, and may be biased by algal uptake. Algae generally only grow during spring and summer when most rivers have limited dilution capacity.

The lowest phosphorus concentrations found in the reported river stations are in Nordic countries, where 91.2% of stations have annual averages below 0.030 mg P l^{-1}, and 50% below 0.004 mg P l^{-1} reflecting both low population densities and high rainfall (Map 6.2). High phosphorus concentrations are found in the remaining countries' river stations, especially in a band stretching from southern UK across central Europe to Romania (and Ukraine). Western and eastern countries exhibit very similar distribution patterns, with high concentrations (50% of stations, respectively over 0.07 to 0.077 mg P l^{-1} in terms of annual average concentrations, and 1% greater than 4.30 to 4.40 mg P l^{-1}, respectively, in maximum concentration). The southern countries' stations, although also severely contaminated, show lower values. The sewerage connection ratio may have a more or less marked effect on the phosphorus export to surface waters. In the case of efficient septic tank systems (including disposal into soil), or in the case of use of sealed tanks (in which waste is kept until emptying), little or no phosphorus is eventually exported.

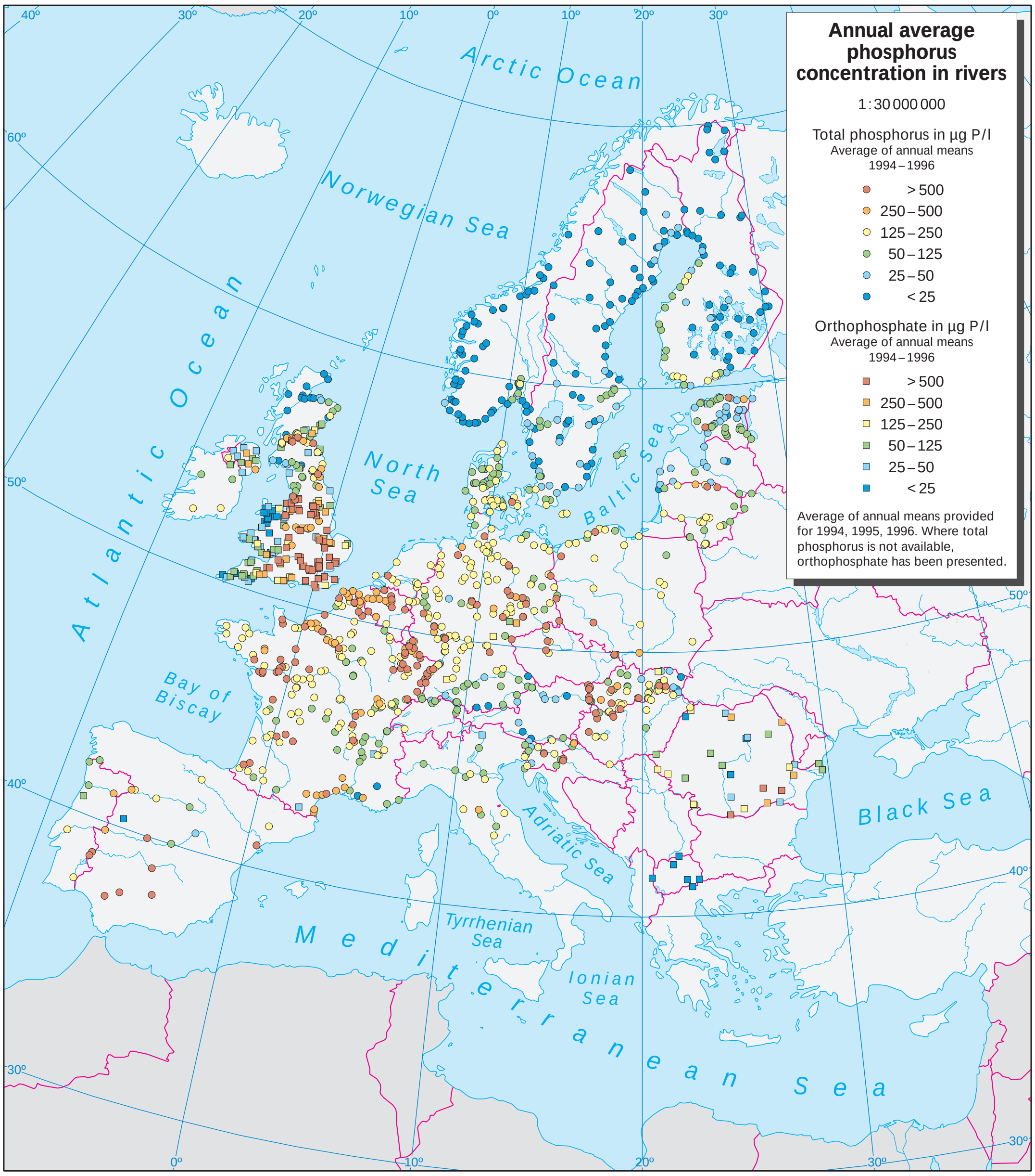
Phosphorus concentrations in European rivers have nevertheless greatly decreased between 1987 and 1991 and the present (1992 to 1996) (Table 6.8). The annual averages and maxima of total phosphorus and soluble phosphorus exhibit the same patterns. However, the maximum values found suggest that excessive concentrations may be recorded even at generally improving sites. In the 1990s significant improvements were observed in western Europe and some countries in eastern Europe. In Nordic countries, rising concentrations have been observed, although the concentrations generally are still very low. The overall improvement is caused by improved waste water treatment and by the overall reductions in phosphorus emissions from households and industry. (see Chapter 3).

The ratios between soluble reactive phosphorus (SRP) and total phosphorus are extremely variable. The proportion of soluble to total tends however to increase when the concentration of total phosphorus increases. The SRP / total P ratios range from 0.44 to 0.64 based on 4000 data points throughout Europe, according to the type of values computed (minimum / minimum, average / average, etc.) (Figure 6.4). The values computed from the data collected

Relationship between average total P and SRP/Total P ratio. Source: Compiled by ETC/IW from questionnaire returns **Figure 6.4**



Phosphorus in European rivers 1994-1996. Source: Compiled by ETC/IW from questionnaire returns. **Map 6.2**



for the report seem rather low. In the UK ratios calculated from data paired from the same sample, yield average SRP / total P ratios in the range 0.61 to 0.96 (Mainstone *et al.*, 1993).

No statistically validated trend is evident. However, comparing countries for which at least 30 data points are available, it appears that the higher total phosphorus concentrations are related to the maximum contribution of SRP. Nordic and Alpine countries, (Finland, Norway, Sweden, Slovenia, Switzerland) exhibit both low concentrations and low proportion of SRP. In western and eastern countries, where the total phosphorus content is higher, the contribution of soluble phosphorus increases with the total phosphorus. This is consistent with the general assessment that in catchments where the majority of phosphorus is derived from point sources (sewage, crude or treated), the SRP / total P ratio is higher than in catchments where the majority of phosphorus is derived from diffuse sources. This suggests that the human origin of phosphorus becomes more and more important with the increase of contamination, this increase being in the soluble form. The case of Baltic states (Latvia, Lithuania, Estonia) is not clear.

Since SRP is the bioavailable form of phosphorus to plants, it is very likely that eutrophication may be dramatically enhanced in rivers where the amount of total phosphorus is high, since the proportion of SRP is expected to increase at the same time.

The external limits to plant growth (namely self-shading) result in a concentration in phosphorus beyond which no increase in biomass can result from an increase in phosphorus; the water is simply *saturated in algal biomass*. An easy to remember figure is that phosphorus concentrations over 0.2 mg P l^{-1} encompasses all possible cases of plant requirements. In most cases 0.1 mg P l^{-1} is sufficient to saturate plant growth capacities. In practical terms this means that little phosphorus is needed to saturate a water body, that a change in eutrophication can be observed only if the available phosphorus concentration has been reduced below the previously indicated threshold concentrations.

The trend in phosphorus concentrations suggests some improvements. The trends shown in Figure 6.5 suggest that the proportion of 'worst cases' decreases slightly with time. However, these improvements

Table 6.9 **Proportion of river stations in relation to phosphorus nuisance limits defined by chlorophyll a classes. (It is assumed that 1 mg P l^{-1} may yield 1 mg Chla l^{-1}). Source: Compiled by ETC/IW.**

Soluble P data. Classes are expressed considering potential for biomass growth.	Period 1975-1980 Annual average	Period 1975-1980 Annual max.	Period 1992-1996 Annual average	Period 1992-1996 Annual max.
Pristine values (≤0.010 mg l^{-1} P)	12.0%	5.4%	16.6%	9.4%
Low eutrophication (>0.010 to ≤0.050 mg l^{-1} P)	22.2%	11.3%	28.1%	11.8%
Significant eutrophication (>0.050 to ≤0.100 mg l^{-1} P)	18.5%	12.4%	23.8%	17.7%
High eutrophication (>0.100 to ≤0.150 mg l^{-1} P)	9.6%	9.5%	8.8%	12.2%
Excessive eutrophication (>0.150 to ≤0.200 mg l^{-1} P)	6.7%	8.3%	5.4%	10.4%
Hyper-eutrophication (>0.200 mg l^{-1} P)	31.0%	53.1%	17.3%	38.5%

This table includes all stations, even if they are not reported on maps because of lack of co-ordinates.

are not sufficient to make water concentrations of phosphorus a limiting factor to algal growth.

An assessment against the criteria proposed for phosphorus in Section 6.2.1 has been carried out in Table 6.9.

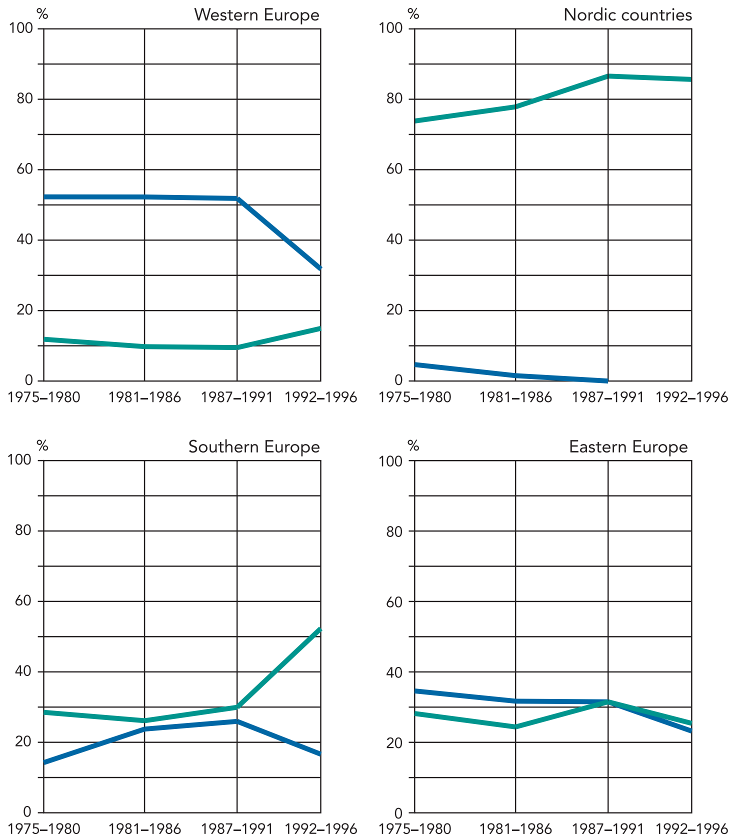
Data reported in Table 6.9 suggest an improvement pattern similar to the one described in the comments attached to Figure 6.5. They confirm that river stations (for which information was reported) with pristine or quasi pristine values are in the minority, whilst river stations with concentrations sufficient to support permanent eutrophication (over 0.060 mg l^{-1}) are the most frequent (50% to 75% of the stations). This statistic only considers soluble phosphorus, which may be only a fraction of total phosphorus. If total phosphorus is considered instead, the situation might be worse with 66% to 80% of stations supporting excessive plant growth.

The phosphorus loading in rivers is closely linked to human activity. However, this relationship is not always direct. In particular, the quantity of P stored in the sediment of large polluted rivers may represent several years of human discharge (AFES, 1997). Moreover, discharge loads for phosphorus are highly linked to high rainfall and floods. Several experiments and monitoring programmes exist to investigate the seasonality of river phosphorus (and sediment) retention. At the present moment, the riverine flux assessments cannot be correlated with known or estimated emissions.

The concentration of SRP in overlying water is determined by a permanent exchange between sediment and water. Depending on sediment characteristics, there is a SRP concentration in water for which no net exchange between sediment and water exists. This information would be of great importance to assess the lag between cuts in emissions and recovery of surface waters.

Evolution of mean soluble P concentration expressed as percentage of stations according to their concentration level. (Data from 25 countries). Source: Compiled by ETC/IW from questionnaire returns.

Figure 6.5



Number of stations per country group

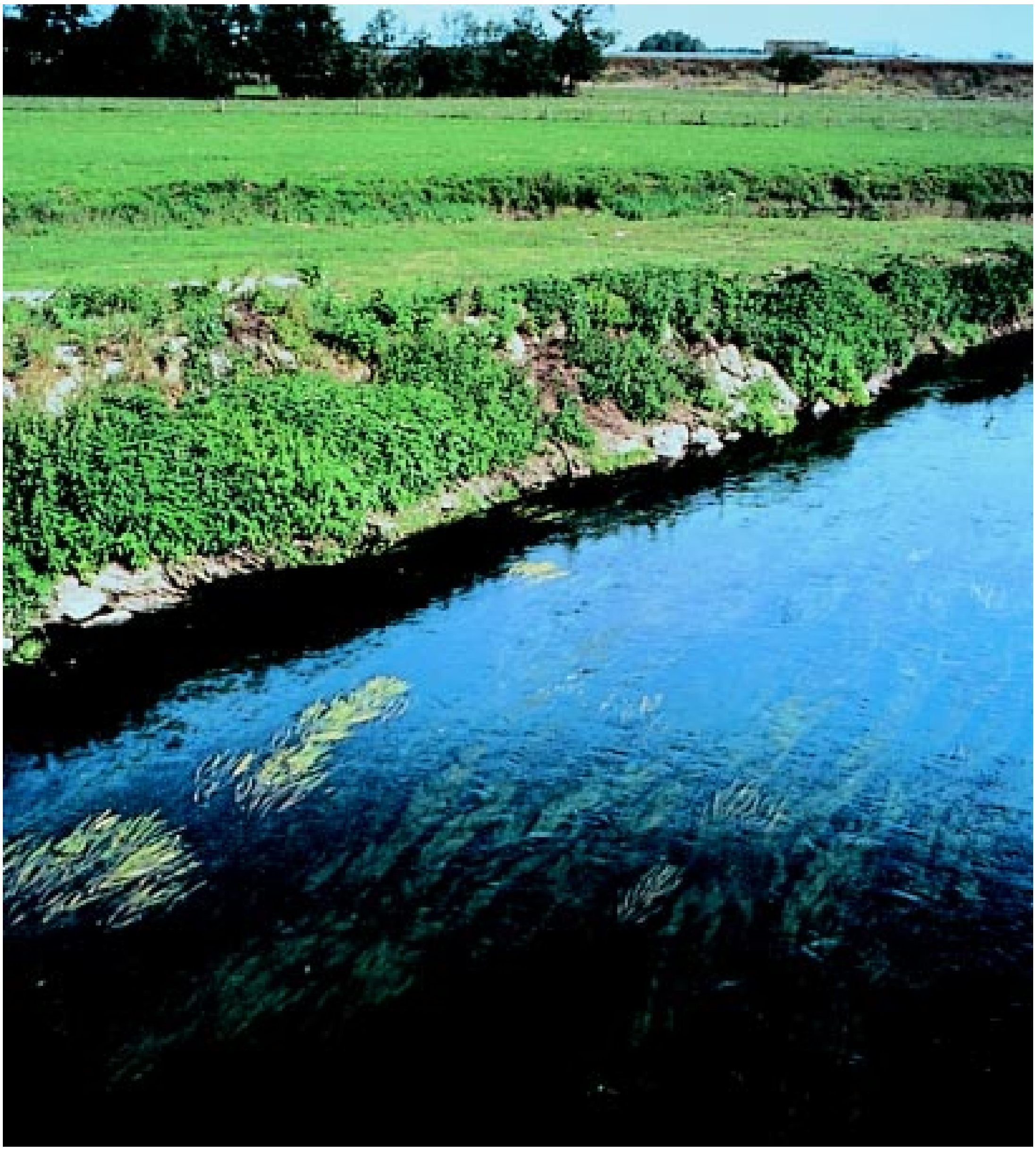
Period	WE	NO	SO	EA
1975 – 1980	454	106	20	77
1981 – 1986	613	130	41	81
1987 – 1991	672	178	49	91
1992 – 1996	968	215	41	180

percentage of sampling stations with average under 0.03 mg P/l
percentage of sampling stations with average over 0.13 mg P/l

Comments on Figure 6.5: For phosphorus, the annual mean provides a reasonably good indicator of potential eutrophication. As the available total P data were relatively few, soluble phosphorus data have been used in this figure. Two threshold phosphorus concentrations were then chosen to illustrate changes with time over different areas in Europe. These were based on the distribution of the available data, and the ability to discriminate between relatively high and relatively low concentrations. These threshold values were <0.03 mg P l^{-1} representing relatively uncontaminated stations, and ≥0.13 mg P l^{-1} as relatively contaminated stations (excessive concentrations). In western and eastern countries, stations with excessive concentrations are relatively numerous, although their proportion has been decreasing slightly over recent time. In Nordic countries, the quality of the very pristine points has tended to diminish, although their concentration is still lower than the 0.030 mg P l^{-1} threshold value. In southern countries, the situation appears to have improved, although this conclusion would need to be verified by a larger set of data.

6.5. Eutrophication of European rivers

As has already been described in section 6.3.1 the eutrophication state of rivers is expressed or assessed by countries in a number of different ways. These include assessing the coverage of the river bed by macrophytes (especially in small rivers), and measuring the planktonic chlorophyll concentrations (particularly in large, or slow flowing rivers). An excess of either in relation to what is expected might indicate eutrophication of the river. In both cases, photosynthesis and respiration of the plants and algae result in large diurnal variations in the oxygen saturation of the water. The pH of the water also increases irrespective of which nutrient is being used by the plants or algae (see Section 4.3.2 for more details). The resulting biomass produced in the water, especially planktonic biomass, can create a large oxygen demand once the plants and algae have died. An approximation, supported by numerous data,



Excessive nutrient input to small, shallow rivers can lead to proliferation of aquatic plants (eg. macrophytes. epiphytes and benthic algae). Photo: Philippe Crouzet is that the total chlorophyll concentration (in µg l^{-1}), divided by 20 or 10 gives an indication of the potential range of BOD_5. For example, a concentration of 100 µg l^{-1} chlorophyll would, on death and degradation of the plankton, equate to an oxygen demand of 5 to 10 mg l^{-1} (Crouzet and Bertru, 1987).

Even though the level of chlorophyll is the most commonly accepted indicator of eutrophication, it is not routinely monitored in rivers. For example, for this report, data on chlorophyll were obtained from only 363 river stations (52 eastern stations, 22 Nordic, 289 western, none in the south) for the period 1992 to 1996. By comparison, information on BOD was obtained from 1159 river stations for the same period. Therefore, an indirect method using two other determinands was used to give a more detailed assessment of the likelihood that any particular river station was eutrophic. The determinands were pH and DO saturation percentage (DO %) for which there were data from a larger number of river stations.

A relationship was derived between annual maximum chlorophyll and changes in dissolved oxygen (calculated as annual average DO % subtracted from the annual maximum DO %), and annual maximum chlorophyll and changes in pH (calculated as annual average pH subtracted from annual maximum pH) from stations that had synoptic measurements of each determinand. The derived relationships were used to predict equivalent maximum chlorophyll levels for those stations with only pH and / or DO % (averages and maximums).

The relationship relies on several factors that affect a river's response to eutrophication. They are:

1. Photosynthesis increases pH and the concentration of DO, and respiration decreases them;

2. In polluted rivers with large amounts of assimilable organic carbon, bacterial respiration may considerably worsen the oxygen conditions during night;

In large rivers excessive nutrient input results in massive development of phytoplankton. Photo: Philippe Crouzet

3. Diurnal changes in DO depend on the relationship between photosynthesis and total respiration and the capacity of water to exchange oxygen with the atmosphere (turbulent water becomes less super-saturated during daytime and less under-saturated during night than still waters);

4. The maximum values and average values were used as the best available information, more detailed data would have been preferable.

Nevertheless, and in spite of the obvious limitations of the method, very illustrative relationships were obtained between changes in pH and DO %, and the planktonic chlorophyll concentration (Figure 6.6). This relationship enabled all available data from stations with chlorophyll, only pH and / or DO % to be expressed within 6 classes of 'eutrophication likelihood' (see Box 6.2).

The results of this assessment are presented in Map 6.3. The map shows 401 river stations with chlorophyll data (interannual averages): 363 river stations for the 1986 to 1996 period, and 38 stations with chlorophyll data that was older. No significant change or trend in eutrophication likelihood could be detected within this data set, thus supporting the aggregation. The map also shows 640 river stations with no chlorophyll data, and for which the assessment of eutrophication likelihood was based on suitable pH and / or DO % data using the procedure described above and in Box 6.2. The risk of overestimating the likelihood of eutrophication by the use of maximum values was reduced by using a 10 year average for these values when assessing the classes.

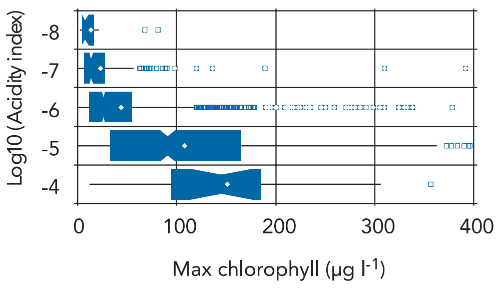
Overall the assessment demonstrates that eutrophication is a major issue at a large proportion of the river stations. This eutrophication is expressed as either an excessive phytoplankton biomass (indicated by chlorophyll concentrations), or by excessive biomass activity (affecting pH and DO saturation). Map 6.3 indicates that 33 % of stations are predicted to be 'highly' to 'hypereutrophic', 40 % 'significantly impacted', 25 % 'barely impacted', and only 2-7 % can be considered as 'unaffected'.

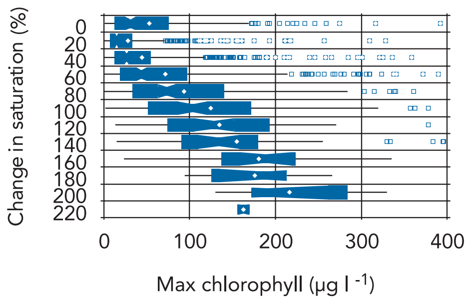
6.6. Conclusions

European rivers are greatly impacted by nutrients. Nitrate, ammonium and its toxic form, ammonia, are present in excess con-

Degradation of large amounts of filamentous algae in rivers uses dissolved oxygen in the water
Photo: Philippe Crouzet

Relationships between the maximum plankton biomass and the acidity index or the change in DO saturation percentage. Source: Compiled by ETC/IW. **Figure 6.6**





Note: this figure reports respectively changes in acidity (computed from pH) and DO saturation, related to monitored biomass. Details are explained in Box 6.2.

Box 6.2 Method for estimating the level or likelihood of eutrophication

In order to estimate the possible biomass at river stations where no such data (e.g. chlorophyll) were available, a step-by-step procedure was undertaken:

a/ the between-year maximum chlorophyll-*a* concentration per period (eg 1992 to 1996) was calculated. This averaging reduces the influence of any outlier data. The maximum was chosen because eutrophication is a seasonal phenomenon not so accurately assessed using annual averages.

b/ for these stations, mathematical relationships were sought using the only related data available: pH and DO saturation percentage (DO concentration/DO saturation concentration x 100). The DO saturation concentration is a function of water temperature and atmospheric pressure. It, therefore, incorporates the ambient conditions at the station when sampled, which DO concentration does not.

c/ Photosynthetic activity changes the concentration of hydroxyl ions [OH^-] and DO concentration. These changes were calculated by: (1) subtracting the average [OH^-] concentration (calculated from average pH) from the maximum [OH^-] (calculated from maximum pH), and (2) subtracting the average DO % from the maximum DO %.

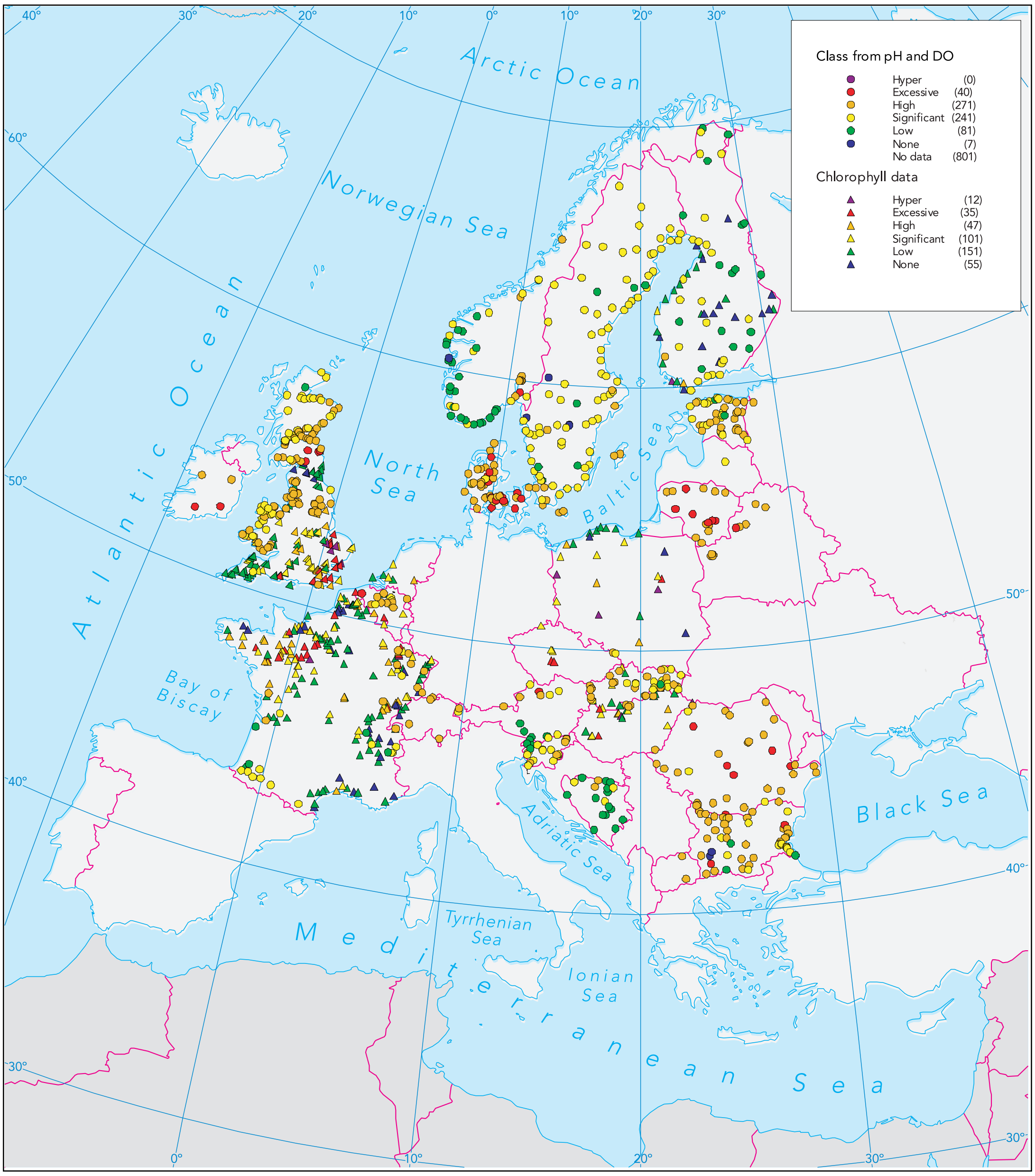
The relationships obtained were quite scattered but indicated some general relationships (see Figure 6.6). The scatter of the data points may arise, for example, from the lack of photosynthetic activity in a large biomass (dead or decaying), measurement errors and activity of fixed plants etc. This is why no statistically significant correlation could be found. However, an analysis of variance (ANOVA) performed on the 2628 available annual data points (not station averages) yielded 6 'classes'. The derived class thresholds equivalent to *none*, *low*, *significant*, *high*, *excessive* and *hypereutrophic* were, respectively, for chlorophyll, 10, 50, 100, 150, 200 µg l^{-1}, for DO saturation 10, 20, 80, 100% and for acidity index -8.5, -7.5, -6.5, -5.5 and -4.5. It should be noted that different chlorophyll classes were used from those described in section 6.2.1. An assessment of the accuracy of the procedure (by comparing predicted and measured chlorophyll) suggests that 30% of stations could be mis-classified, being either in a 'better' or 'worse' class than they should be.

centrations at almost all of stations for which information was reported. Available data suggest that this situation is slowly improving, although this mainly concerns the most polluted stations, whilst the number of pristine and less polluted stations tends, on the contrary to shrink.

Phosphorus is present in excess at practically all reported stations. Considering the sensitivity of water bodies to eutrophication, this excess yields unwanted plant growth in a large proportion of stations. It is very likely that stations where no excessive quantity of biomass can be detected (directly or indirectly) are not sensitive to eutrophication for reasons other than nutrient concentration. This may be the result of permanent and high turbidity, or presence of toxics.

The problems resulting from nutrient excess has become a very important environmental issue in European rivers, especially as pollution resulting from organic matter has significantly decreased (EEA, 1998).

Map 6.3 **Eutrophication (monitored or estimated) in water at European river stations. Source: Compiled by ETC/IW.**



7. Status and impact on marine and coastal waters

7.1. Characteristics of Europe's seas

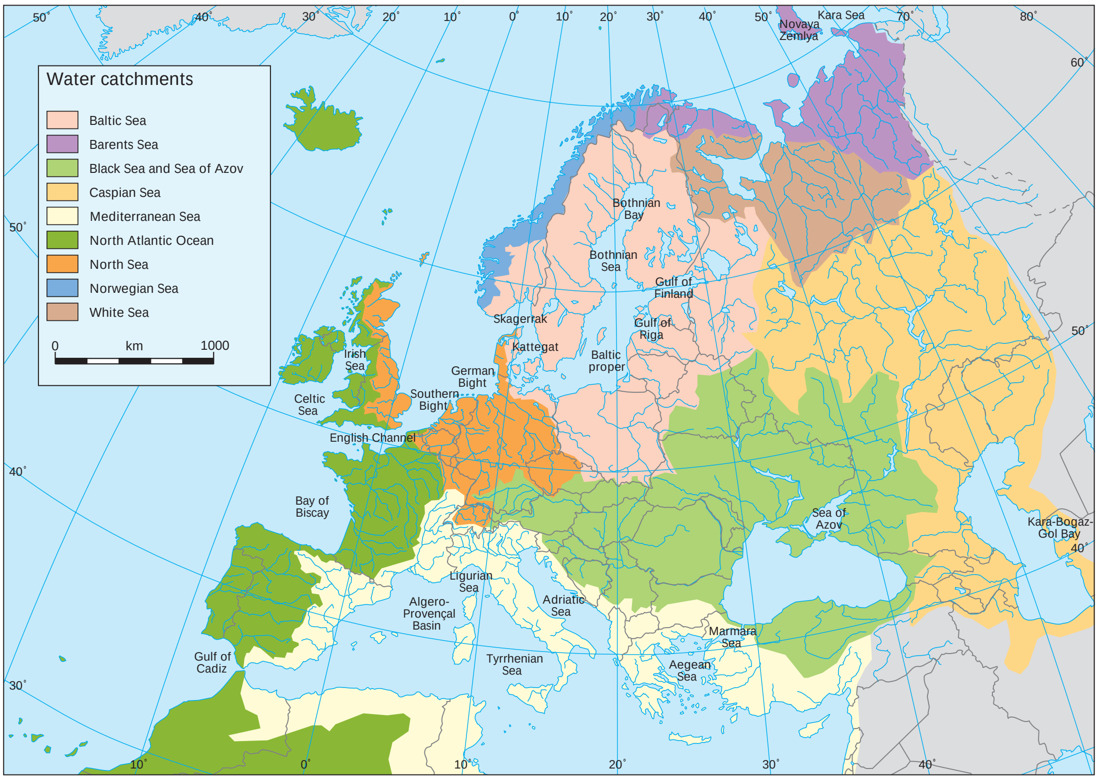
The seas and coastal environment of Europe are major economic and ecological resources. About one third of the European population lives within 50 km of coastal waters: urban, industrial developments and agriculture are resulting in significant degradation and mounting pressures on already hard-pressed areas. Map 7.1 shows Europe's Seas with the main subsidiary seas and bays, and their respective catchment and drainage area (EEA, 1995).

The seas primarily covered in this assessment report are: the Mediterranean Sea, Black Sea, Baltic Sea, North Sea, Barents Sea, Norwegian Sea and the north-east Atlantic Ocean. Europe's seas also include the Caspian Sea, White Sea and Arctic Sea. There appears to be little available information with regards to nutrients and eutrophication in these latter seas.

Most marine systems can be classified as being from oligotrophic to eutrophic in their natural states. Eutrophication can occur as a result of natural processes, for example, where there is upwelling of nutrient-rich deep water (below the photic zone) to nutrient poor but light rich surface water of the photic zone of the water column (Jørgensen and Richardson, 1996). Cultural eutrophication arising from anthropogenic activities is particularly evident in marine areas with limited water exchange such as semi-enclosed seas such as the Baltic and Adriatic Seas, fjords with sills that restrict water exchange, and in lagoons, bays and harbours.

Marine eutrophication has been described as 'one of the major causes of immediate concern in the marine environment' (GESAMP, 1990). While data coverage is incomplete, it remains a widespread phenomenon in Europe's seas and its effects have been reported in several areas.

Europe's seas with subsidiary seas and bays and catchments. Source: EEA (1995) — **Map 7.1**



7.2. Criteria for assessment of nutrient levels in transitional, coastal and marine waters

The key plant nutrients in the sea that can cause eutrophication are nitrogen and phosphorus, but other nutrients such as silica and trace elements are also important. Nutrient enrichment results in higher primary productivity of algae in surface layers and on the sea bed, followed by higher secondary productivity of marine animals. While some nutrient enrichment may thus be beneficial, excessive enrichment may result in large algal blooms and seaweed growths, oxygen depletion and the production of hydrogen sulphide which is toxic to marine life and can cause high mortality. Eutrophication phenomena also affect human health and recreational uses of marine coastal zones.

The concentration of nutrients above which eutrophication becomes an environmental problem depends on the topography and the physical and chemical nature of each sea and sea area. Thus 'excessive' nutrient levels would have to be relative to the 'background' or 'reference' level for a particular sea. Reference levels would be significantly different for the relatively oligotrophic Mediterranean Sea and the more eutrophic North Sea. Concentrations above 'reference' levels would then have to be related to the possibility or risk of there being a response in the ecosystem such as a change in trophic status. Such reference levels and excessive nutrient levels have not been established for European seas. In this report, therefore, the relative concentrations of nutrients have been defined as follows (Table 7.1). [Note that the classes in Table 7.1 do not cover the Mediterranean Sea].

7.3. Nutrient concentrations in Europe's seas

Figures 7.1 and 7.2 give an overview of data on nitrate / nitrite and phosphate concentrations in the surface water layer, mainly for the North and Baltic Seas. Limited data is available for the north-east Atlantic Ocean. There appears to be no data on nutrient concentrations in the Caspian, White and Arctic Seas.

Surface concentrations of nitrate / nitrite in most of the sampling areas in the North Sea appear to have been higher in 1995 / 96 than in 1980, possibly as a result of exceptional flooding in most of the rivers in the North Sea catchment area in 1995. The concentrations in the Baltic Sea did not show the same trend. High concentrations were recorded in some of the regions of the UK, but 1996 concentrations were lower than in former years. In the northern part of the North Sea and in the Thames estuary, phosphate concentrations seem to have been somewhat higher in the mid-1990s than in the early 1980s. The levels in the Rhine estuary and in the German Bight at Helgoland Reede fell from 1985 to 1994, while little or no change was found in other regions in the North and Baltic Seas and the north-east Atlantic Ocean.

Within the Baltic Sea there is a general spatial trend of decreasing nutrient concentrations with increasing distance from the coast, and from the inner parts of gulfs and bights towards the open sea. In the Kattegat and Belt Sea area, an additional nutrient source is water inflow from the Skagerrak with a salinity of about 33.5 ‰, and a nitrate concentration of 10 to 11 µmol NO_3 l^{-1}. In addition, the winter concentrations of phosphate and nitrate show

Table 7.1 **Criteria for assessment of nutrient levels in transitional, coastal and marine waters. Source: Compiled by ETC/MC**

Quality status	Nitrite+nitrate µmol l^{-1}	Quality status	Phosphate µmol l^{-1} (1)
Good	<6.5	Good	<0.5
Fair	6.5 to 9.0	Fair	0.5 to 0.7
Poor	9.0 to 16.0	Poor	0.7 to 1.1
Bad	>16	Bad	>1.1

[1] Please note that it is the convention to use µmol l^{-1} for concentrations of nutrients in marine waters rather than µg l^{-1} as used in fresh waters.

increasing trends in the surface layer of all sub-regions of the Baltic Proper for the period 1958 to 1993, and 1969 to 1993, respectively (HELCOM, 1996). The increases between 1969 and 1978 were particularly significant. In contrast to phosphate, the increase of nitrate continued until 1983. Thereafter, the concentration of both nutrients fluctuate strongly at a high level without significant trends.

There are, however, regional differences. For example, phosphorus concentrations have been decreasing in Germany's Baltic coastal waters since 1990. In 1996 / 1997 the mean concentrations were about 50 % lower than in 1990 (German NFP, 1998). The concentrations of nitrate / nitrite and phosphate in surface waters are measured routinely in the North and the Baltic Seas, but less frequently in the north-east Atlantic Ocean.

At the present there is no equivalent data from the Mediterranean Sea though less recent data reported by GESAMP (1990) reported typical background concentrations of nitrate of 0.5 µmol l^{-1}, in moderate eutrophic areas 1.5 µmol l^{-1} and in heavily eutrophic 5 µmol l^{-1} to greater than 7 µmol l^{-1}. Corresponding values for phosphate were 0.03, 0.15, and 0.3 µmol l^{-1}, respectively. It is generally the Mediterranean shore adjacent to large towns and tourist resorts (such as Adriatic coast) which show the highest nutrient concentrations. More recent surveys in the western basin have indicated that nitrate did not exceed 3.0 µmol l^{-1} and phosphate 0.12 µmol l^{-1}. In the surface water of the Ligurian Sea mean values of 0.02 µmol l^{-1} for nitrate and 0.04 µmol l^{-1} for phosphate have been reported (TEMPO project-MAST, 1991; Bethoux *et al.*, 1992).

Annual mean concentrations of nitrite/nitrate in the surface water of the North Sea, the Baltic Sea and some areas of the north-east Atlantic Ocean, 1980 to 1996. Source: EEA (1998) **Figure 7.1**

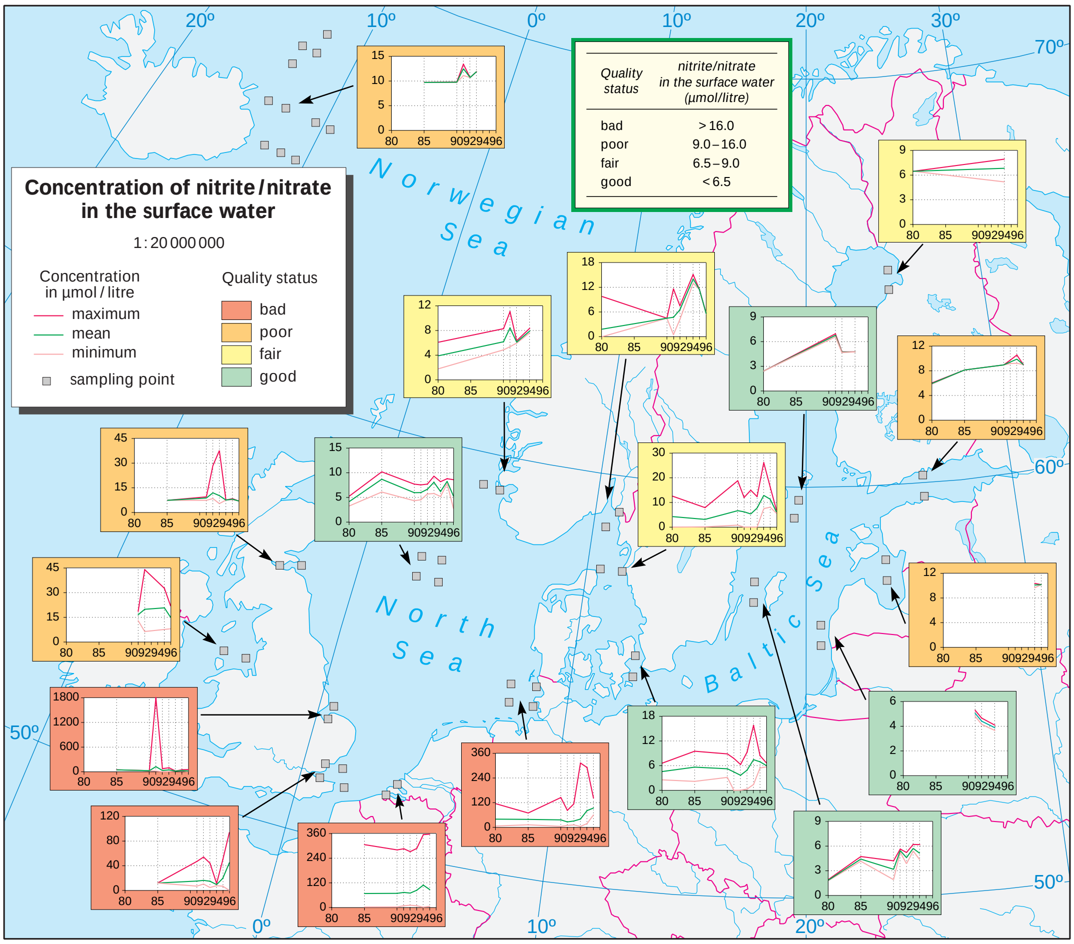
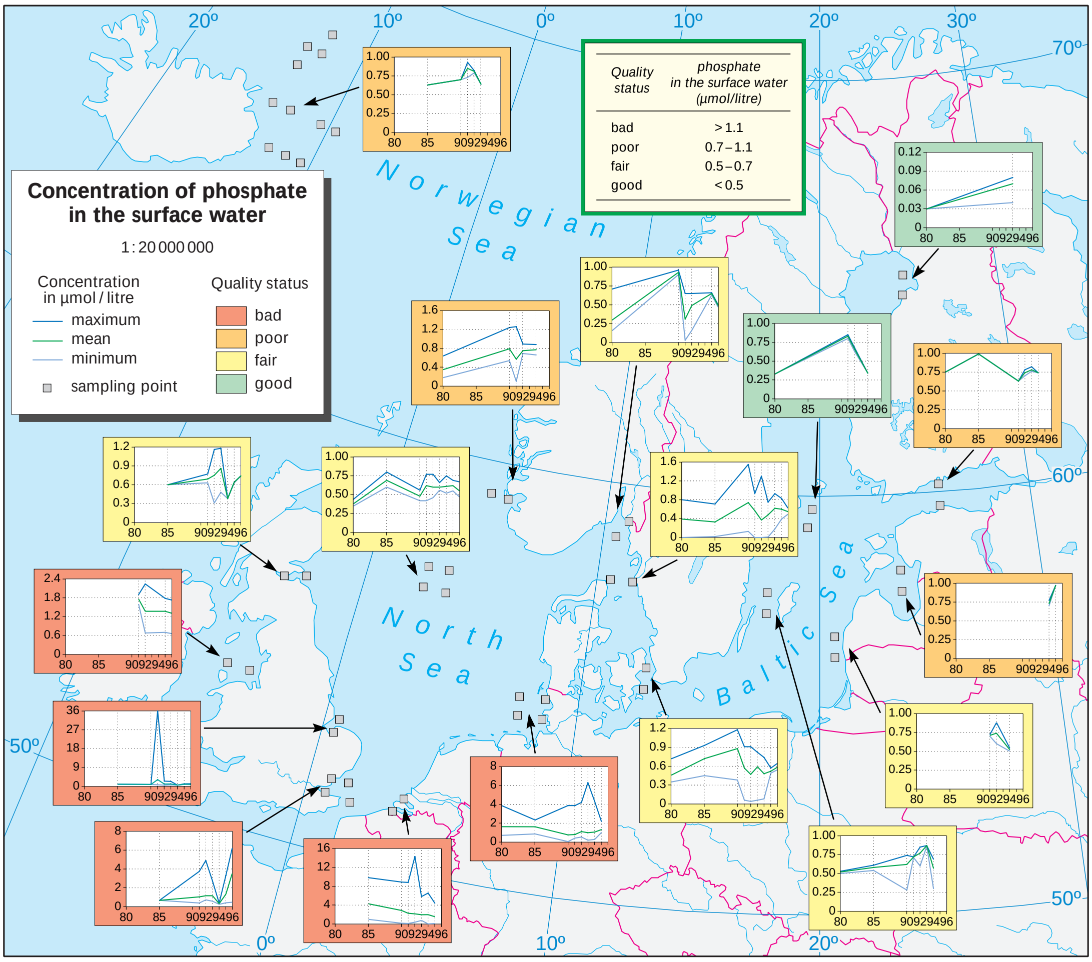


Figure 7.2 **Annual mean concentrations of phosphate in the surface water of the North Sea, the Baltic Sea and some areas of the north-east Atlantic Ocean, 1980 to 1996. Source: EEA (1998)**



Average nitrate and phosphate concentrations in the Black Sea increased, by factors of about 7 and 18, respectively, during the winter months between 1960 and 1992, probably because of the increased inputs from the Danube, Dnieper and Dniester (Cociasu *et al.*, 1996). Along the Romanian coastal waters of the Black Sea there has been an increase in the concentrations of nitrogen and phosphorous and a decrease in silicate. Between 1960 and 1992, in the winter months, the median of nitrate and phosphate concentrations changed from 1.2 to 7.9 µmol l^{-1}, and 3.9 to 6.9 µmol l^{-1}, respectively (Cociasu *et al.*, 1996).

The Co-operative Marine Science Programme for the Black Sea measured the concentration of nitrite / nitrate and phosphate at approximately 356 stations across the Black Sea in April 1993. The measured nitrite / nitrate concentrations at a 10 m depth ranged from <0.1-0.9 µmol l^{-1} offshore from the Danube delta to 1.46 µmol l^{-1} in the south-west of the sea off the Bulgarian coast. There was a general spatial pattern of higher concentrations closer to the coastline, and a trend of decreasing concentrations from the west to the east. The highest concentrations of phosphate at 10 m depth were in the north west basin of the sea reaching concentrations of up to 0.45 µmol l^{-1}. There were also relatively high concentrations in parts of the eastern basin reaching concentrations of up to 0.3 µmol l^{-1}.

7.4. Loads of nutrients discharged into Europe's seas

7.4.1. Introduction

Nutrients in the sea arise from direct discharge from industry, agriculture and sewage, by river transport, and by deposition from the atmosphere. These discharges are measured or estimated under a number of monitoring programmes. The completeness and accuracy, of the data vary for different countries and seas. There are many factors, both natural and anthropogenic, that can affect the load of nutrients entering seas each year. For example, riverine loads, particularly of nitrogen, can vary significantly in relation to river flow. Assessments of trends in annual loads must, therefore, take these factors into consideration.

Over the last decade the loads of nutrients from rivers and from direct discharge (industry, agriculture, sewage) into coastal waters and seas, and from atmospheric deposition have been measured or estimated under joint monitoring programmes directed by the marine Conventions, international organisations and the Co-operative Programme for Monitoring and Evaluation of the Long Range Transmission of Air Pollutants in Europe (EMEP). For some of the seas or nations, the available data provide trends over the last decade, for others it does not.

Total nitrogen and total phosphorus loads from direct discharges and riverine inputs, with trends over the years 1990 to 1995, are given in Figure 7.3 for the Barents, Norwegian, North, and Irish Seas and parts of the Iberian Coast. Figures 7.4 to 7.6 give a more detailed breakdown of direct and riverine loads into the North and Baltic Seas. For the Mediterranean and Black Seas only estimates of the total nitrogen and phosphorus loads are presented because of gaps in the data. Figure 7.7 shows the estimated atmospheric deposition of oxidised nitrogen over the years 1985 to 1995 into the Baltic, North, Mediterranean and Black Seas, and into the north-east Atlantic Ocean.

7.4.2. Direct discharges and riverine inputs

The load of total nitrogen from rivers and direct discharges to the western and northern parts of the European Seas seems to have increased between 1990 and 1995 for the North and Baltic Seas, while there appears to be no changes in the other Seas (Figure 7.3). The north-east Atlantic Ocean shows variable levels. There are no data available for the other four seas.

Trends in total phosphorus loads from direct discharges and riverine inputs appear to be somewhat different from those for nitrogen loads. Trends since the mid-1980’s indicate some significant reductions in loads to some seas and in some river catchments (Figure 3.13). However, more recent information (since 1990) seems to indicate different trends. For example, in the North Sea data show an increase in phosphorus load, the Iberian Coast gives a variable picture, loads to the Celtic Sea have been steady since 1991, and the Baltic Sea shows a decrease between 1990 and 1995 (Figure 7.3). The three northern-most seas show no changes.

The North Sea data show an increase of both phosphorus and total nitrogen discharges, mainly because of the run-off of surplus nutrients from agriculture. Discharges to the Iberian Coast have been

Total nitrogen and total phosphorus loads from direct discharges and riverine inputs. Source: Compiled by ETC/MC | **Figure 7.3**

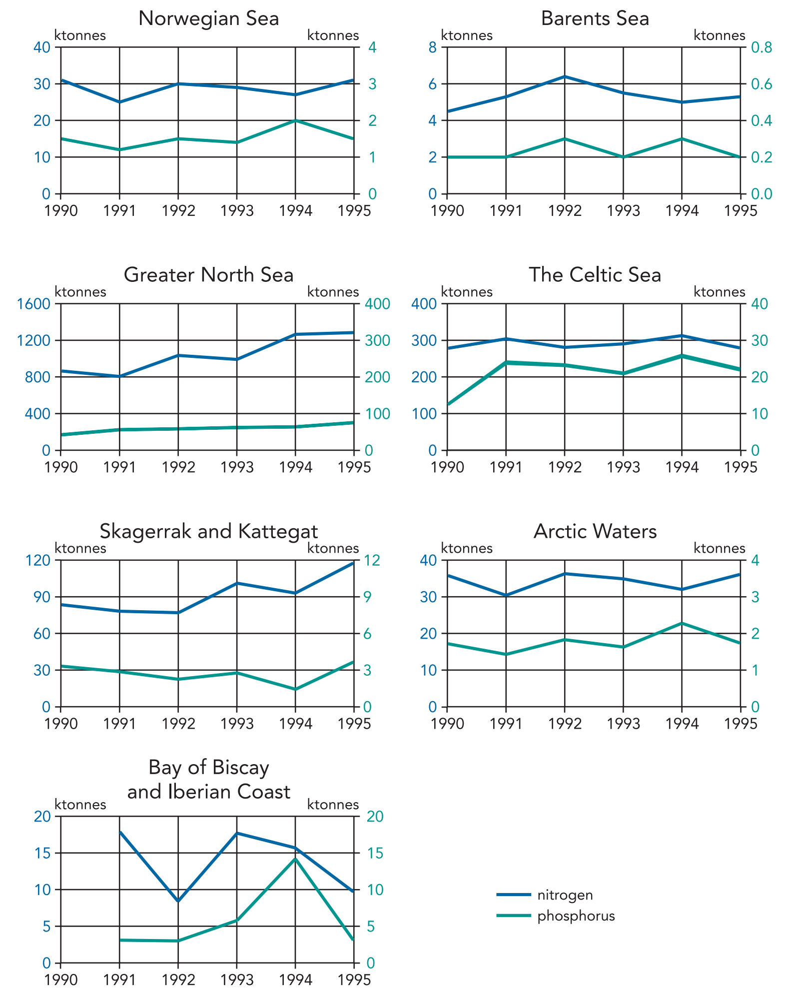
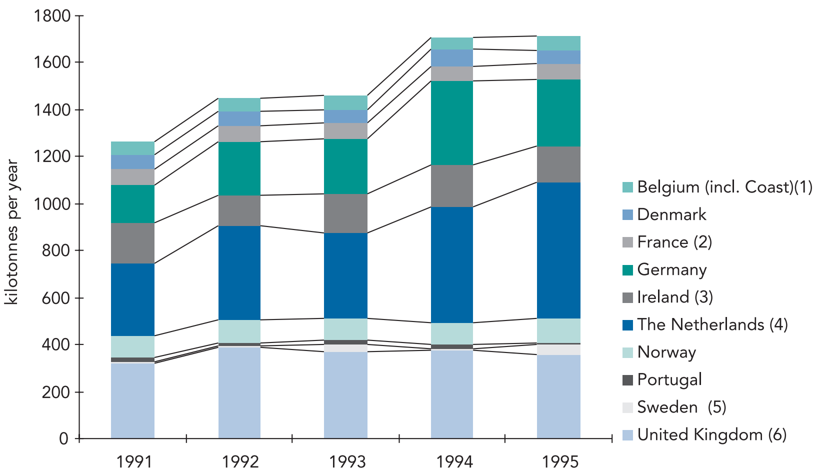
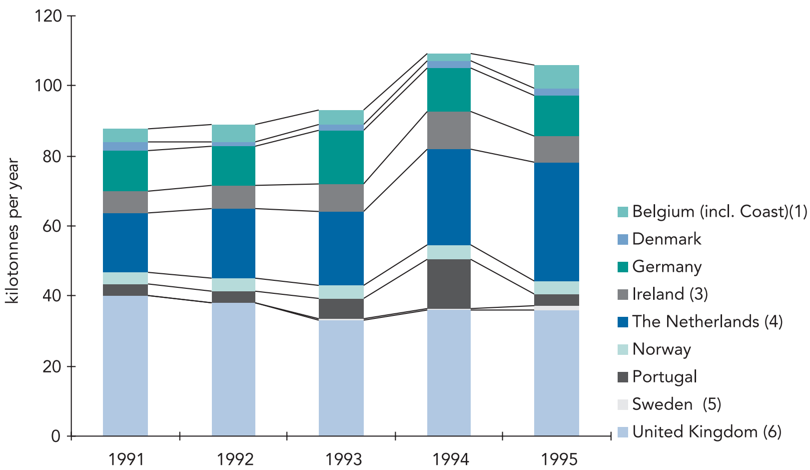


Figure 7.4 **Direct discharges and riverine inputs of total nitrogen and total phosphorus into the OSPAR area (ktonnes year^{-1}). Source: OSPAR (1997)**

a) Total nitrogen



b) Total phosphorus



1 High estimates used

2 Riverine inputs only and identical estimates each year

3 Identical estimates for direct discharges each year

4 No data was available for direct discharges in 1993/94. The estimated level is about 5000 tonnes year^{-1} for N, and 1000 tonnes year^{-1} for P

5 No data was available for riverine inputs in 1990/91/92/94. The estimated level is about 30 000 tonnes year^{-1} for N, and 1000 tonnes year^{-1} for P. Data for the OSPAR region only.

6 Phosphorus inputs are orthophosphate phosphorus. Data is for all seas around UK. No data for the Channel.

variable, those to the Celtic and Irish Seas have been steady since 1991, and those to the three northernmost seas have not changed significantly. For the Mediterranean and Black Seas, only estimates of the total nitrogen and phosphorus discharges can be given because of discontinuities in the data.

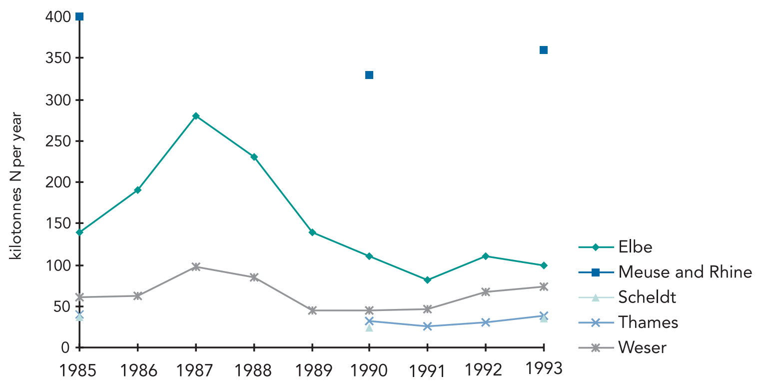
Figures 7.4 and 7.6 give a breakdown of the discharges of riverine and direct disharges into the North and Baltic Seas from the surrounding countries. There are only few long-term data series from which trends can be distinguished. It appears that the nutrient inputs (as total annual discharges) from Belgium, the Netherlands and Germany into the North Sea showed relatively high values during 1994 and 1995 (Figure 7.4). This increase correlates with the high precipitation and water flow of the main rivers during those years.

The nutrient inputs from 6 major rivers (Elbe, Meuse and Rhine, Scheldt, Thames and Weser) discharging into the North Sea between 1985 and 1993 are shown in Figure 7.5. The figures are not normalised for year to year climatic variations and may be based on somewhat different methodologies for calculating loads (North Sea Conference, 1995). They thus should be treated with some caution. Nevertheless these 6 rivers contributed approximately 60 % and 20 % of the estimated riverine and direct discharges of nitrogen and phosphorus, respectively, into the North Sea in 1993. The largest contributor of riverine loads of nitrogen and phosphorus from the 6 rivers are the rivers Meuse and Rhine[2]. Though there is evidence of decreased loads from all 6 rivers compared to 1985, since 1990 loads of both nitrogen and phosphorus appear to have been relatively unchanged though nitrogen loads in the Thames and Weser may have increased in more recent years. From these river examples it was concluded that it was not possible to draw any firm conclusions on the relationship between the reduction in discharges/losses of nutrients and the riverine input into the North Sea (North Sea Conference, 1995).

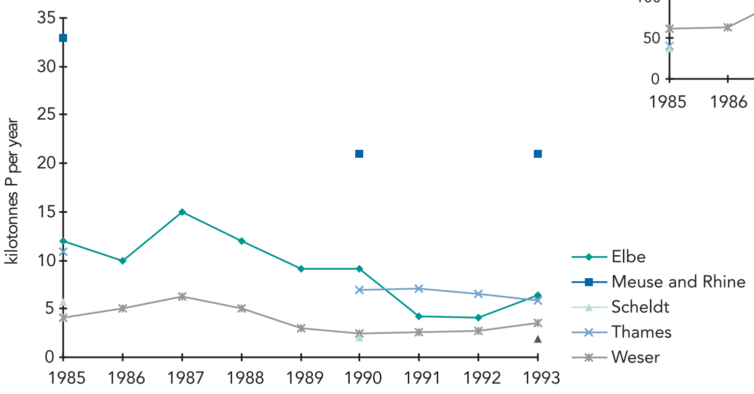
In the Mediterranean Sea, discharges of nitrogen and phosphorus are of the order of 270 000 and 24 000 tonnes year^{-1}, respectively, in the Adriatic region, which

Loads of nitrogen and phosphorus discharging from 6 major rivers into the North Sea between 1985 and 1993. Source: North Sea Conference (1995) **Figure 7.5**

a) Nitrogen

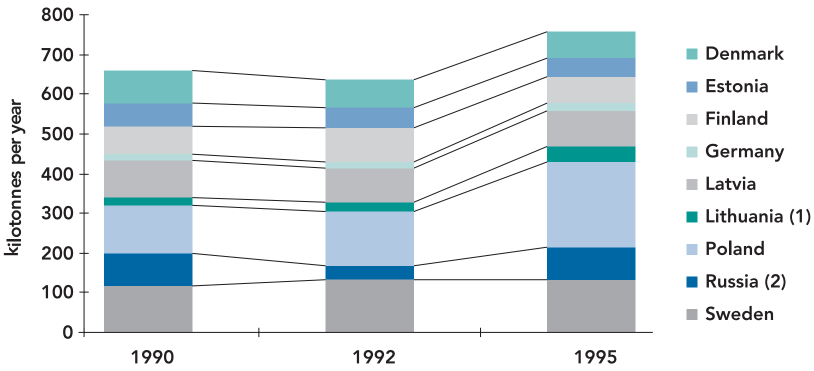


b) Phosphorus

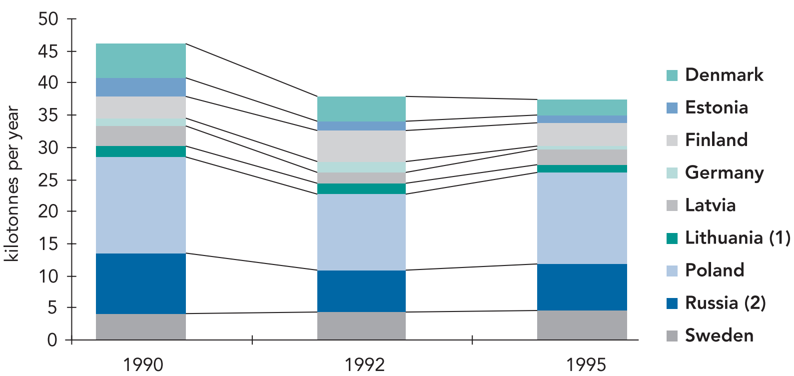


Direct discharges and riverine inputs of total nitrogen and total phosphorus (1000 tonnes year^{-1}) in the Baltic Sea area, 1990 to 1995. Source: HELCOM (1996) **Figure 7.6**

a) Total nitrogen



b) Total phosphorus



1 data for riverine total P from Lithuania is missing; for calculation the 1987 figure was used

2 data for riverine total P for Russia is incomplete for 1992

Eutrophic lagunes can be completely covered by green algae (*Ulva spp*.). Prevost lagoon, French Mediterranean coast. Photo: Philippe Crouzet

includes discharges from Italy, Croatia and Slovenia (UNEP, 1996). Polat & Turgul (1995) estimated that the north Aegean Sea receives annually 180 000 tonnes of nitrogen and 11 000 tonnes of phosphorus from the Black Sea, which are comparable with the inputs from land-based sources to the north-east of the Mediterranean (Yilmaz *et al.*, 1995).

In the Black Sea region, the annual discharges of the Danube alone were estimated to be 230 000 tonnes total nitrogen and 40 000 tonnes total phosphorus (GEF/BSEP, 1997). The total annual discharges for nitrogen and phosphorus from point sources located on the coast of sea are less than half of the total discharges from international rivers (Danube, Dniepr, Dniestr, Coruh, Don) (Table 7.2).

7.4.3. Atmospheric deposition

The atmospheric deposition of nitrogen can be a significant input to Europe's Seas. [The atmospheric deposition of phosphorus is relatively negligible]. Figure 7.7 shows the estimated atmospheric deposition of oxidised nitrogen over the years 1985 to 1995 into the Baltic, North, Mediterranean and Black Seas, and into the north-east Atlantic Ocean. There have

Table 7.2 **Annual discharges (1000 tonnes year^{-1}) in the Black Sea area, mid 1990s. Source: Black Sea Environment Program. Source: GEF/BSEF (1997)**

	Total nitrogen	Total phosphorus
Bulgaria	4.5	1.12
Georgia	1.6	0.43
Romania	89.7	0.51
Turkey	18.7	3.97
Russian Federation	13.5	1.04
Ukraine	41.8	5.43
International Rivers	236.2	43.274
Total	406	54.93

been no changes since 1990 in nitrogen deposition from the atmosphere measured in the North, Mediterranean and Black Seas. The north-east Atlantic Ocean shows variable levels of inputs, while the Baltic Sea appears to be receiving less nitrogen than in 1990-91. No data was available for the other four seas.

The third periodic assessment of the state of the Baltic Sea indicated that about 300 kilotonnes of nitrogen per year (between 1986 and 1990) reached the sea by wet and dry deposition with little inter-annual variations (HELCOM, 1996). About 40 % of this input consisted of the reduced forms of nitrogen, mainly ammonium, the remaining part occurring as NO_x forms. As a comparison it was estimated that 661 kilotonnes of total nitrogen reached the Baltic Sea in 1990 from direct discharges and riverine inputs.

The estimation of atmospheric deposition to the North Sea is based on a combination of data on contaminant concentrations reported by coastal measuring stations surrounding the North Sea, and the results of long range transport models (OSPAR, 1997). These estimates indicate that the annual total atmospheric input (wet and dry deposition) of nitrogen between 1987 and 1995 has varied between 298 kilotonnes and 371 kilotonnes with no trends apparent. In 1995, the estimated load was 327 kilotonnes, that is approximately 25 % of the estimated load from direct and riverine inputs.

7.5. Eutrophication of Europe's seas

The consequences of eutrophication can be an increased frequency of algal blooms (sometimes toxic), increased water turbidity, slime production, oxygen depletion in deep waters and mass fish and benthic fauna kills. The more common and apparent effects are increased turbidity and phytoplankton blooms; the extreme events are oxygen depletion, hydrogen sulphide (H_2S) emissions and fish kills. There are many intermediate situations over this range of events that vary in time and in space due to a combination of factors such as morphological and hydrodynamic characteristics, renewal time, natural fluctuations and climatic constraints.

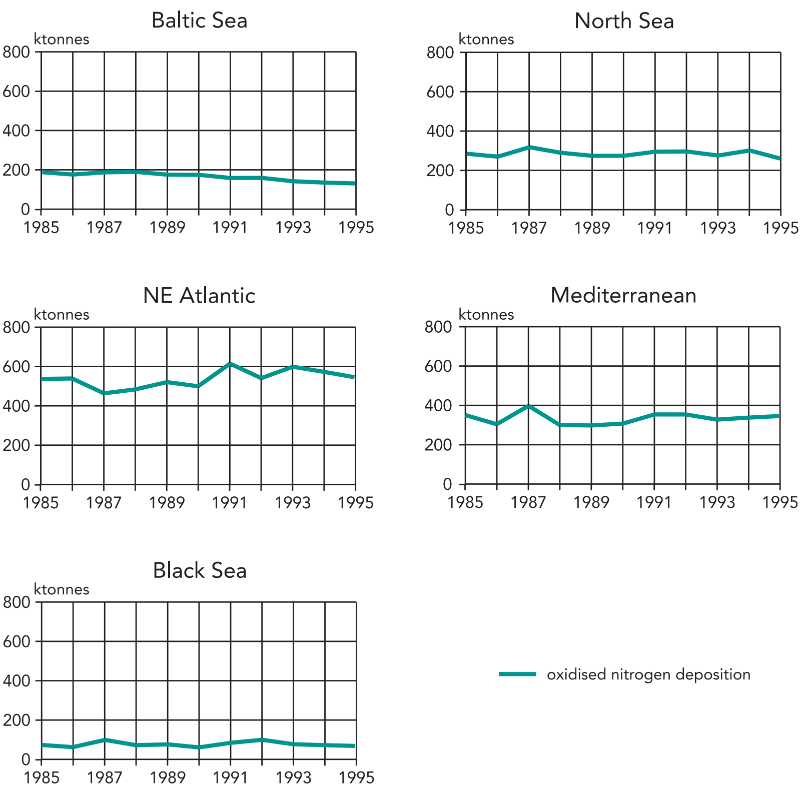
Forecasting the effects of eutrophication is a challenge for the future. However it should be recognised that the marine environment shows poorly understood short term fluctuations and even less well-understood long term changes due to natural causes (GESAMP, 1990).

The following text summarises the main manifestations of eutrophication in Europe's Seas and lists (in Boxes) the main 'eutrophication' episodes reported from each.

7.5.1. North Sea and north-east Atlantic Ocean

Seven North Sea riparian countries (Belgium, Denmark, France, Germany, the Netherlands, Norway and Sweden) have identified problem areas with regards to eutrophication, on the basis of national criteria. These largely lie along the eastern coast of the North Sea from the River Somme (France) in the south, along the Belgian, Dutch, German and Danish coasts

Oxidised nitrogen deposition from the atmosphere. Source: Compiled by ETC/MC **Figure 7.7**



to the southern tip of Norway (OSPAR, 1995). There are also problem areas in French coastal water adjacent to the River Seine and in areas around the Morlaix peninsula. The United Kingdom has identified 4 potential problem areas which were to be investigated further to establish their status.

The spread of eutrophication effects in the North Sea from the coastal regions that receive high nutrient loads to other regions of the North Sea is also an issue of concern (NSTF, 1993). The eastern Skagerrak and Kattegat are affected by both local and long-range transported nutrients and eutrophication. The Dogger Bank is also a region that may be affected by transport from coastal regions.

7.5.2. Baltic Sea

The Baltic Sea was for a long time considered to be an oligotrophic sea but over recent years that opinion has changed (EEA, 1995). Eutrophication is now identified as one of the most serious problems. For example, the increase in primary production and the subsequent increase in algal sedimentation and decomposition on the seabed have decreased the oxygen content of the deep water (Figure 7.8). Oxygen deficits have always occurred in special 'holes' in Danish marine waters, where the bottom water is not exchanged for long periods, but in the 1980s oxygen deficits have been more frequent, longer lasting and more serious than previously (Christensen *et al.*, 1994).

The mass development of microscopic algae over large areas of the Baltic drastically reduces water transparency and sometimes creates surface scum and odours. The reduction in water transparency has also affected the distribution and composition of benthic algae in particular.

Box 7.2 Eutrophication episodes

Anoxia in most of the deep basins in the Baltic Sea;
Changes in plant communities in important fish nurseries;
No exceptional algal blooms in the Baltic Sea in 1995, and more sporadic occurrences of a toxic species in 1995 compared to previous years.
Sources: Rosenberg *et al.*, 1990; Baden *et al.*, 1990; Ambio, 1990a, b; HELCOM, 1996; Leppänen *et al.*, 1995

Box 7.1 Eutrophication episodes

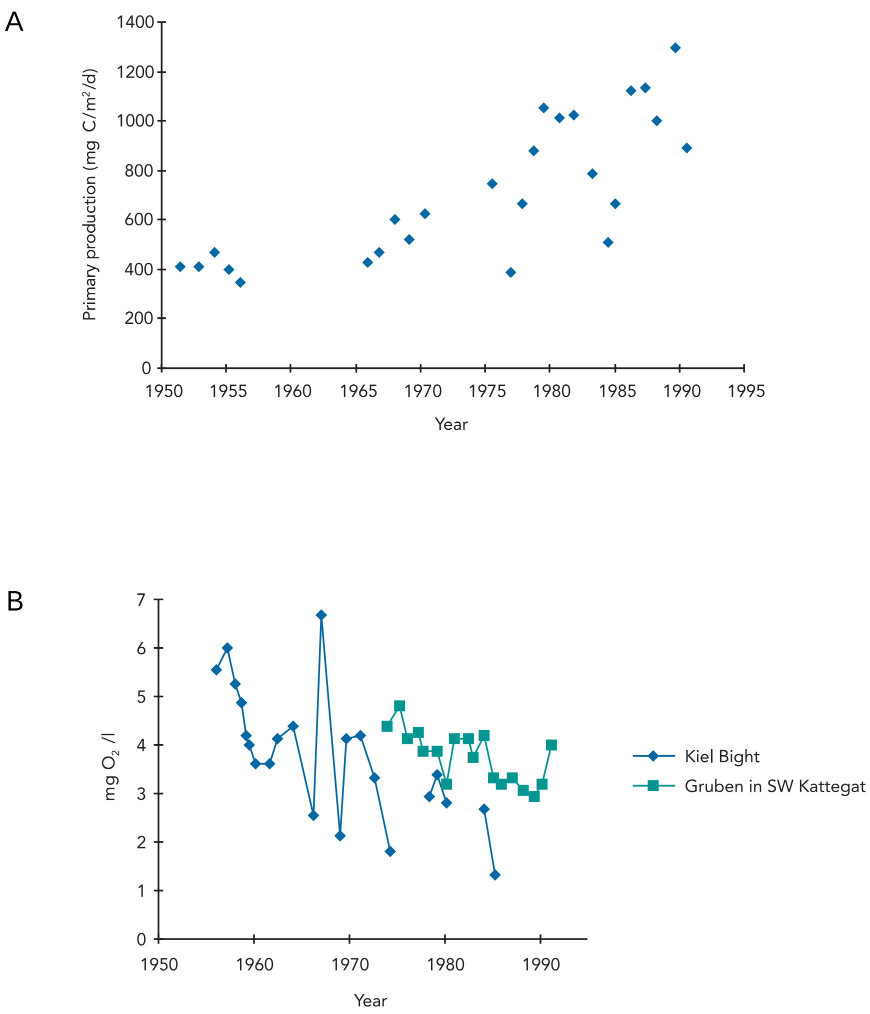
English Channel and Atlantic Coasts:
Eutrophication phenomena have been reported in bays along the coasts of the English Channel:

- Bay of Seine (France): High chlorophyll a concentrations are observed during spring and summer (max = 50 µg l^{-1}) (Guillaud and Ménesguen, 1998). Since 1975 green or red-tides are regularly observed;
- Bay of St. Brieuc, Bay of Lannion and Bay of Guisseny (France). Blooms of *Ulva sp*. have been observed before the 1970's, in 1973 and more or less from 1978 to 1995 (Ménesguen and Piriou, 1995);
- The estimated production of *Ulva sp*. in Brittany is about 100 000 – 200 000 tonnes wet weight per year (Piriou, 1996);
- The toxic species *Alexandrium minutum* is evenly encountered in "Abers" and in the bay of Morlaix, located on the north coast of French Brittany (Erard-Le Denn *et al*., 1993).

Atlantic coast:
Though less prevalent than in the North Sea, some bays along the south coast of French Brittany (Douarnenez, Concarneau), and the Arcachon Basin also have a large cover of green seaweed. The occurrence of toxic phytoplankton like *Dinophysis* or *Gymnodinium* cf. *nagasakiense* (ichtyotoxic) were observed along the Atlantic coast from 1983 to 1995 (Belin, 1993; Belin *et al*., 1989; 1995). A few hypoxic events which did not cause fish kills (except in the Vilaine river estuary in 1982) have been reported along the Atlantic French coasts (Ménesguen, 1990).
North Sea:
Regular major impacts in coastal waters, including the coast between Belgium and Skagen (Denmark), in Danish inlets, along the west coast of Sweden and in the outer Oslofjord.
The growth of macroalgae is effected in some UK estuaries.
(North Sea Task Force, 1993)

A. Trend in phytoplankton production over the period 1950-90 (Richardson and Ærtebjerg, 1991) and
B. Trend in bottomwater oxygen content in Kiel Bight 1957-86 (during September) and in the south-western Kattegat (Gniben) over the period 1974-92 (yearly average) (NERI, 1994)

Figure 7.8



7.5.3. Black Sea

Because of the long residence time of the water in the basin, the Black Sea is highly sensitive to eutrophication (Box 7.3).

There has been a major shift from diatom species towards blooms of non-diatom species, which probably relates to a considerable decrease in the silica / nitrogen ratio.

7.5.4. Mediterranean Sea

The inputs of nutrients into the Mediterranean Sea are estimated to be significantly lower than the outflow through the Gibraltar Strait, making it one of the world's most oligotrophic (nutrient-poor) seas. However, eutrophication problems occur in semi-enclosed bays, mainly because of poor water management (Box 7.4). A large number of coastal bays still receive large amounts of untreated sewage. In the eastern Mediterranean, uncontrolled expansion of fish farming might be causing local problems. The most endangered area is, however, the northern and west coast of the Adriatic sea, which receives the nutrient load of the River Po. Data is generally poor, with only some 'hot spots' being continuously monitored. Phosphate and nitrate concentrations near the surface are very low, tending to increase rapidly below 200 m (Bethoux *et al.*, 1992).

7.6. Conclusions

Eutrophication affects marine biodiversity, fish and shellfish stocks as well as human health and the recreational use of marine coastal zones. The main affected areas are the Black Sea, with severe anoxic effects at a basin scale due to the increase of nutrient discharges coming mainly from the Danube run-off, the Baltic Sea, because of excessive nutrients, topography and physical and chemical nature; the North Sea because of high nutrient discharges, particularly phosphorus; the Mediterranean Sea,

Box 7.3 Eutrophication episodes

Since early 1970s, large increase in algal bloom frequency and drastic reduction of shallow water species.
1980-90: 42 blooms recorded, with a strong increase in blooms of non-diatom species.
Reductions in some shallow water plant populations and distribution areas of long-life eelgrass species, perennial brown and red algae and all their associated fauna, but increases in some opportunistic species;
Mass mortality of numerous sea-bed species.
Mass development of jelly-fish and of predatory gelatinous species.
Every summer: hypoxia and anoxia phenomena reported, most severe effects in the north-western area.
Sources: Mee, 1992; Gomoiu 1992; Bodenau, 1992; Cociasu *et al.*, 1996; Leppakoski and Mihnea, 1996.

Box 7.4 Eutrophication episodes

Since the early 1970s, eutrophication in semi-enclosed bays: 34 cases along the coast line, 21 in lagoons, but the record is incomplete.
1975-97, Adriatic Sea: flagellate bloom, followed by anoxia and fish kill.
Every year since 1975, with increasing frequency, 15 mollusc species and 3 crustacean species have disappeared.
Sources: Montanari *et al.*, 1984; Margottini and Moiin, 1989; Rinaldi *et al.*, 1993; UNEP (OCA)/ MED, 1996.

but only in 'hot spots' in shallow and coastal waters with high nutrient discharges and favourable physicochemical conditions; and, at a basin scale, the Adriatic.

Measures against eutrophication need to be taken at international level, because of the transboundary, nature of the phenomenon. This will require uniform definitions and harmonisation of reporting and criteria for assessing eutrophication. The Oslo and Paris Commission (OSPARCOM), covering the north-east Atlantic Ocean, the North Sea, the Norwegian Sea and parts of the Barents Sea, has initiated a process to harmonise reporting of nutrient discharges from point and diffuse sources in the North Sea. The European Commission (CEC) and the European Environment Agency support this action to enable the process to be adapted for the other Member States.

The political goal in the OSPAR region and the Helsinki Commission (HELCOM) regions covering the Baltic Sea is to reduce discharges of nutrients by 50 % where these inputs are likely, directly or indirectly, to cause eutrophication.

In the Mediterranean Sea, the eutrophication of some areas ('hot spots' / semi-enclosed bays) is an issue for concern. The priorities of the Mediterranean Action Plan Assessment are to create an inventory of land-based sources and to encourage action on the factors controlling eutrophication, based on scientific knowledge of the functioning of the ecosystem.

The priority in the Black Sea Environment Programme is the control of nutrient inputs. primarily from river run-off.

Eutrophication (oxygen deficiency) caused by excessive nutrient input from agriculture eliminated the complete fish stock in August 1997 in Mariager Fjord, Denmark.
Photo: Gerth Hansen/ BIOFOTO

8. Status of terrestrial ecosystems

8.1. Introduction to terrestrial environments

8.1.1. Species and ecosystems endangered by excessive nitrogen deposition

The terrestrial environments considered for this report include those of potential interest for nature conservation, such as forests, heathlands and shrubs, permanent grasslands, bogs and marshes, and inland shallow standing water bodies. The classification used is illustrated in Box 8.1.

Box 8.1 Selection of natural and semi-natural terrestrial ecosystems (as proposed by ETC/NC)

Forests
- temperate broad-leaved deciduous forest
- temperate mixed forest
- temperate riparian and swamp forest and brush
- temperate broad-leaved evergreen forests

Scrubs
- temperate heath and scrub
- sclerophyllous scrub

Permanent grasslands
- steppes and dry calcareous grasslands
- dry siliceous grasslands
- alpine and subalpine grasslands
- humid grasslands and tall herb communities
- mesophile grasslands
- tundra

Bogs and marshes
- raised bogs
- blanket bogs
- fens, transition mires and springs

Inland shallow standing water bodies

Terrestrial ecosystems receive a range of nutrients from anthropogenic sources. The most important pathways from anthropogenic sources to natural and semi-natural terrestrial environments is through atmospheric deposition, and through lateral ground-water flow or surface runoff from agricultural fields. The terrestrial environments susceptible to increased availability of nutrients are, however, mainly influenced by atmospheric deposition. In these ecosystems, the largest effects are expected from increased availability of nitrogen, partly because growth and competition might be controlled by nitrogen, partly because the atmospheric input of nitrogen has increased more than inputs of, for example, phosphorus, potassium or magnesium.

The most important impacts of increased nitrogen deposition in terrestrial ecosystems are:

1. Direct effects of nitrogen gases on individual species,
2. Soil-mediated effects of acidification and eutrophication,
3. Increased susceptibility to secondary stress factors,
4. Changes in the competitive relationship between species.

The species most sensitive to excess nitrogen deposition are those belonging to terrestrial ecosystems which mainly or only depend upon nutrient input from the atmosphere. Species from these ecosystems are adapted to survive at very low nutrient levels. Often these ecosystems lack a soil compartment, which acts as a moderating factor in the flow of nutrients within the system, and the plant species present use their entire surface for uptake of nutrients from air or precipitation. Thereby these species are directly exposed to excess inputs of nitrogen and other nutrients, as well as toxic compounds from the atmosphere. Examples of such ecosystems are the raised and blanket bogs of north-western Europe, Arctic ecosystems with a very low rate of mineralisation, lichen and moss dominated heathlands, and plant communities dominated by lichens and mosses. Lichen communities, epiphytic or on base poor sand or stone are probably those most sensitive to excess nitrogen.

8.1.2. Effects of excess nitrogen on plant functioning

In ecosystems limited by nitrogen, for example, raised bogs and the heathland ecosystems of north-west Europe, the species exist in a balance with soil micro-organisms, often in a symbiotic relationship. When such ecosystems are exposed to increased loads of nitrogen, ecosystem structure or functioning may change in a number of ways. The changes may result in, for example, increased competition from nitrogen adapted species on less adapted species, or increased sensitivity to stresses as, for example, diseases, drought, frost, or pest insects. Insects tend to prefer leaves with high nitrogen concentrations. High nitrogen concentrations in leaves and needles are probably not directly associated with needle damage but may rather cause other nutrients to become deficient, especially phosphorus. On calcareous soils excess nitrogen will often change the ecosystem by increasing the production and decreasing the biodiversity. Rich fens, influenced by extensive grazing, and having a high biodiversity with orchids and other specialised floral elements, will change towards communities dominated by productive grasses. Grassland areas on calcareous soils may also loose biodiversity, especially some specialised groups of fungi *Hygrocybe*, *Nolanea* and *Leptonia* that are sensitive to nitrogen.

8.1.3. Effects of excess nutrients on species composition in ecosystems

In most ecosystems, a limited number of species benefit more than the average from excess atmospheric nitrogen deposition. These species are more tolerant to nutrient imbalances caused by the increased availability of nitrogen. Examples of such species are listed in Table 8.2.

Deposition of excess nitrogen on a heathland ecosystem may cause heather (*Calluna vulgaris*) to be overgrown by grasses. Photo: Anders Tvevad/BIOFOTO

Table 8.1 **Effects and results of excess nitrogen in terrestrial ecosystems**

Ecosystem	Threat	Effect	Result	Country	References
Raised bog	Excess N	Competition	Invasion of grasses and trees Frequency decrease of lichens Invasion of N tolerant species	DK UK	Aaby (1994) Risager and Aaby (1996) Risager and Aaby (1997) Lee et al. (1993)
		Stressed species	Decrease of sensitive *Sphagnum spp.*	DK UK	
Rich fen	Excess N Acidification	Stress and competition	Disappearance of rare mosses Biodiversity loss	NL	Kooijman (1992)
Heathland	Excess N	Competition	Overgrowing by grasses	NL	Aerts and Heil (1993)
			Loss of biodiversity	NL	Fennema (1990)
		Stress	*Calluna vulgaris* attacked by *Lochmaea suturalis*	NL DK	Aerts and Heil (1993) Riis-Nielsen (1997)
Grassland	Excess N	Competition	Increased frequency of nitrophilous plant species	NL	Bobbink (1991)
			Loss of lichen and moss species characteristic to chalk grasslands	NL	During and Willems (1986)
			Loss of *Hygrocybe* and *Leptonia spp*	NL	Arnolds (1988)
Deciduous forest	Excess N	Stressed trees with increased N conc. in leaves.	Unhealthy trees More exposed to parasitic fungi and insect pests	S	Balsberg Påhlsson (1992)
		Depodzolization	Changed soil fauna	F	Toutain et al. (1987)
		Symbiosis affected	Unhealthy trees Less mycorrhizal mushrooms on/in forest floor	UK S	Newton and Piggott (1991) Rühling and Tyler (1991)
		Nitrate leaching	Groundwater pollution	DK	Callesen et al. (1997)
Coniferous forest	Excess N	Stressed trees with increased N conc. in leaves.	Unhealthy trees More exposed to parasitic fungi and insect pests	S NL	Wästerlund (1982) Houdijk and Roelofs (1993)
		Reduced seedling survival	Changes in community composition	PL	Kowalkowski (1990/91)
		Symbiosis affected	Unhealthy trees Less mushrooms on/in forest floor	NL	Jentschke et al. (1991)
		Nitrate leaching	Groundwater pollution	DK	Callesen et al. (1997)

Table 8.2

Examples of species from natural and semi-natural ecosystems that may increase in frequency in sensitive ecosystems in case of increased nutrient availability

Species	Fertiliser	Ecosystem	State	Reference
Arrhenatherum elatius (grass)	NPK	Grassland	NL	Berendse *et al*. (1992)
Brachypodium pinnatum (grass)	N	Chalk grassland	NL	Bobbink, (1991)
Brachythecium spp. (moss)	N	Forest floor	S	Dirkse & Martakis (1992)
Campylopus introflexus (moss)	(N)	Heathland with lichens	NL	Greven (1992)
Chamaenerion angustifolium (herb)	N	Raised bog Pine-heath	DK S	Aaby (1994) Persson (1981)
Deschampsia flexuosa (grass)	N	Dry heathland	NL	Aerts and Heil (1993)
Deschampsia flexuosa	P	Dry heathland	DK	Riis-Nielsen (1997)
Deschampsia flexuosa	N	Forest floor	NL	Dirkse and Martakis (1992)
Dryopteris carthusiana (fern)	N	Forest floor		Dirkse and Martakis (1992)
Festuca gigantea (grass)	N	Forest floor	PL	Turnau *et al*. (1992)
Molinia coerulea (grass)	N	Moist heathland, heathland Heathland Raised bogs	NL DK DK	Aerts & Heil (1993) Degn (1996) Aaby (1994)
Molinia coerulea	P	Moist heathland	NL	Berendse and Aerts, (1984)
Paxillus involutus (fungi)	N	Deciduous forest	UK	Newton and Pigott (1991)
Pleurozium schreberi (moss)	N	Heathland with lichens	S	Persson (1981)
Scleroderma citrinum (fungi)	N	Deciduous forest	UK	Newton and Pigott (1991)
Sphagnum recurvum (moss)	N	Raised bogs	UK	Lee *et al*. (1993)

8.2. Geographical distribution of terrestrial environments

8.2.1. The land cover/land use of Europe

Large parts of the European landscape are clearly dominated by human activities like urban areas, agriculture and various infrastructure. In addition a large part of the forested area is production forest in more or less intensive management. Other potential nature types like heathlands and most pastures are semi-natural ecosystems demanding some kind of management to prevail. Map 8.1 shows the dominant land uses in Europe at a resolution of 10x10 minutes (Veldkamp *et al*., 1995). The distribution of land uses in Europe including the European part of the Russian Federation is listed in Table 8.3.

There are large differences in land cover/land use distribution in the different European countries. The land cover/land use in the central and Atlantic parts of Europe is dominated by arable land and grasslands, whereas land use in Sweden, Finland, and the northern part of the Russian Federation is dominated by coniferous forest. Permanent crops other than grassland are mainly found in the Mediterranean area.

8.2.2. Calculated critical loads and exceedances for nitrogen and for acidity

The concept of critical loads was first developed by the Canadian government in the eighties as a tool for ecological risk assessment. After further development, initiated by the Nordic Council of Ministers, the concept was adopted in 1988 as the basis for the development of effect oriented abatement policies under the UNECE Convention on Long Range Transboundary Air Pollution (LRTAP). The Convention has to date been ratified by 38 parties including the European Union. The most recent critical loads have been used as part of the scientific basis for the EU Community Strategy to Combat Acidification.

The currently used definition of a critical load is:

> *'A quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge' (UBA, 1996).*

Both structural and functional changes in ecosystems, either direct or mediated by changes in soil geochemistry, should be considered. There is, however, not necessarily a direct cause-effect relationship between critical load exceedances and observable effects in ecosystems.

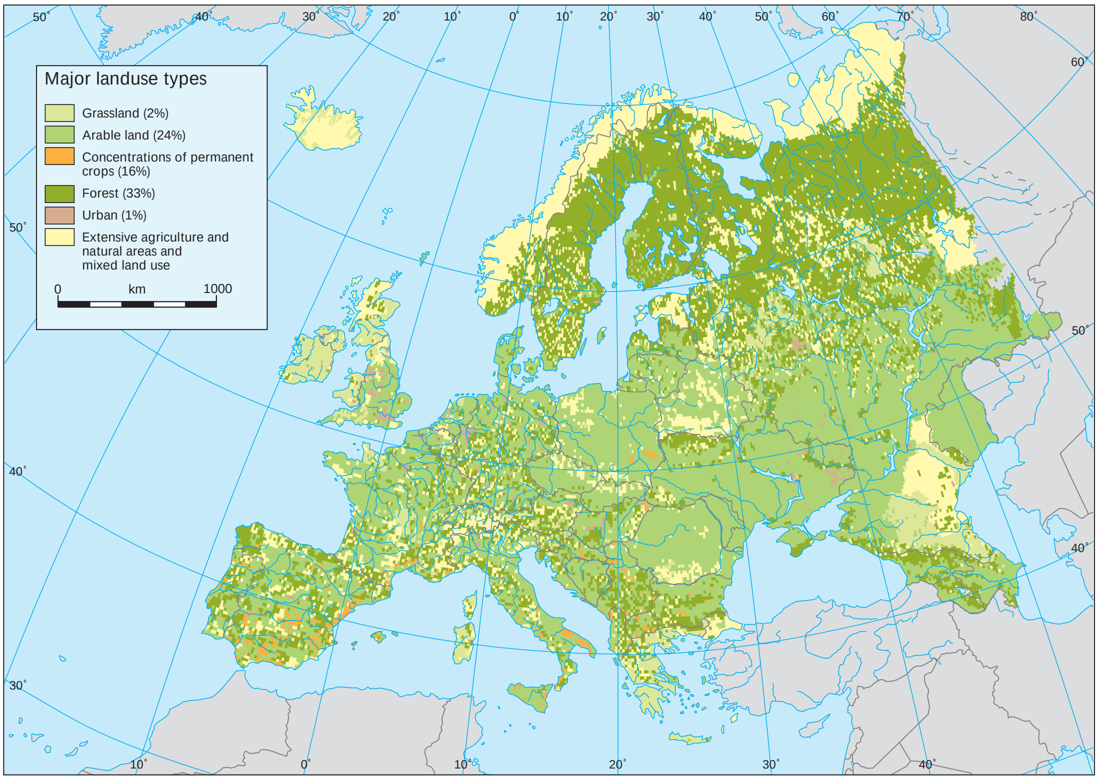
Soil acidification and nitrogen accumulation in ecosystems are long-term processes, and it can take decades of exceedance of the critical load before effects in the ecosystem can be observed. Ecosystem recovery can similarly take a long time after deposition levels have been reduced.

8.2.3. Critical load mapping

As part of the work under the LRTAP Convention, European maps of critical loads and critical load exceedances are calculated biannually for use in the ongoing negotiations on emission reductions. At present the mapping exercise considers the acidifying effects of nitrogen and sulphur, and the eutrophying effect of nitrogen in terrestrial ecosystems and poorly buffered fresh water. European maps are prepared by the Co-ordination Centre for Effects in the Netherlands on the basis of data prepared by 'National Focal Centres' in the countries which are parties to the convention. Most countries calculate critical loads for production forests and soft water (poorly buffered fresh water, especially lacking calcium and magnesium ions) lakes applying a steady state mass balance model. The most common criterion used in calculating the critical load for acidity is a ratio of 1 between base cations and aluminium in soil solution or, for poorly buffered fresh water lakes, a critical limit on acid neutralisation capacity (ANC) of 20 meq l^{-1}. For nutrient effects of nitrogen, a critical loading between 0.5 and 20 kg N ha^{-1} $year^{-1}$ is applied depending on ecosystem type. Only a few countries have reported empirically based critical loads for a larger range of ecosystems.

Deposition of air pollution in Europe is calculated by the Co-operative Programme for Monitoring and Evaluation of long range transmission of air pollutants in Europe (EMEP). EMEP has three tasks: collection of emission data, measurement of air and precipitation quality, and modelling of atmospheric dispersion. On the basis of emission data, meteorological data and func-

Dominant land use types in Europe (Veldkamp *et al*., 1995) **Map 8.1**



Distribution of land uses in Europe including the European part of the Russian Federation (Veldkamp *et al*., 1995) **Table 8.3**

Type of land use	area (km²)	%
arable land	2 343 838	24.3
coniferous + mixed forest	2 488 327	25.8
deciduous forest	664 587	6.9
grasslands	1 513 557	15.7
permanent crops	144 508	1.5
urban areas	117 887	1.2
inland waters	203 196	2.1
other (mountains, tundra, etc.)	2 175 516	22.5

tions describing the transformation and removal processes, EMEP calculates deposition of air pollutants across Europe. The resolution of the current EMEP model is 150x150 km, but a change to 50x50 km grid is under preparation.

The calculated critical loads have a much higher spatial resolution than the deposition values calculated by EMEP. Exceedance of critical loads is thus a comparison between a range of critical loads within a grid and a single calculated deposition value. This is either handled by applying a percentile of the critical load values, e.g. the 5 percentile, or by describing the percentage of the ecosystem protected within each grid. The 5 percentile critical load is the deposition threshold, where 95% of the ecosystems within the grid are understood to be safe.

Any potential effects of deposition on ecosystems will depend on the relationship between nitrogen (and sulphur) deposition and the critical loads for different ecosystem types. Thus:

- If the nitrogen deposition is lower than the minimum critical load [sum of nitrogen uptake and nitrogen immobilisation], nitrogen deficiency will occur.

- If the nitrogen deposition is higher than the minimum critical load, acidification occurs if denitrification is too low.

- Damage can occur if the total load of acidifying compounds, i.e. nitrogen and sulphur exceeds the critical load of acidity.

- If nitrogen deposition is higher than critical loads of nutrient nitrogen, too much nitrogen is leached leading to nutrient imbalances and vegetation changes favouring nitrogen-tolerant species.

On this basis, work reported by Posch *et al.*, (1995) using the EMEP maps, indicates that the most nitrogen sensitive ecosystems are found in Fennoscandinavia, Britain, and northern Italy.

8.2.4. Ecosystem threats

Exceedances of critical loads should be seen as an indicator of risk to ecosystems. Map 8.2 shows exceedances of the critical loads for eutrophying nitrogen. Maxima are observed close to the main emission

Almost all lichen species (e.g. *Cladonia* species) are very sensitive to air pollution. Dovrefjell, Norway. Photo: Ulla Koustrup/ BIOFOTO sources in western and central Europe. The level of exceedance is correlated with the risk of damage and the anticipated time before effects occur. The calculation methods does, however, not give a quantitative cause-effect relationship. Current reduction plans will increase ecosystem degree of safety in most of Europe. In the most polluted areas in the black triangle between Germany, Poland and the Czech Republic, and along the Kola peninsula, large areas will, however, remain under threat. It should also be noted, that the most sensitive ecosystems in the rest of Europe will remain effectively exposed, despite current reduction plans.

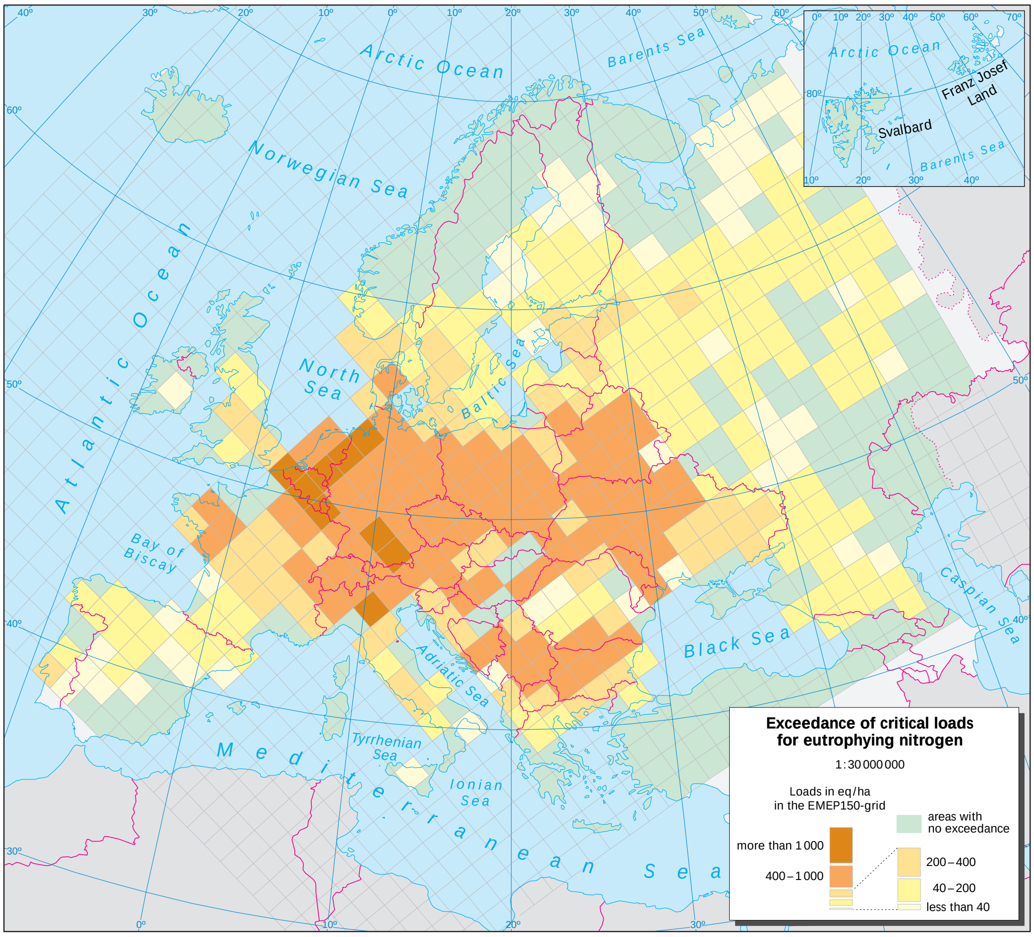
There are a number of limitations to the applied method of assessing ecosystem exposure. Critical loads are primarily calculated for forest ecosystems, and although the applied criteria might protect most of the forest species, the criteria are primarily set to protect forest production. Sensitive ecosystems like raised bogs and heathlands are not included in the calculation. The applied average deposition value within a grid-cell may cover a large local variation, especially in areas with high ammonia emissions. However, the overall picture of the magnitude and location of air pollution problems in Europe is probably valid.

8.2.5. *Threats to sites and species of special conservation interest*

The quality of the different terrestrial habitats can obviously not be seen from general land cover maps. One of the only indicators available on nature quality on an European scale is the designation of areas for special protection, e.g. RAMSAR areas, Sites of Special Scientific Interest, or Special Areas for Conservation to be included in the Natura 2000 network.

Used as a tool in risk assessment, calculated critical loads have been used to assess the risk of damage by acidification or eutro-

Exceedance of the 5 percentile critical loads for eutrophying nitrogen, 1995. (Sources EMEP/MSC-W and CCE) **Map 8.2**



phication to sites or species of special conservation interest. Tickle *et al.* (1995) reviewed the threat from air pollution to about 1300 species. Out of these 26 (2%) were both vulnerable to atmospheric deposition of nitrogen and sulphur, and occurred on an European red list of endangered species. In this study, genera such as *Sphagnum* and *Cladonia* were regarded as one species each, although these genera contain more species that are sensitive to air pollution.

In a detailed study performed in the United Kingdom by the national conservation agencies (Farmer, 1993; Farmer and Bareham, 1993), the susceptibility to acidification and various effects of nitrogen was investigated for Sites of Special Scientific Interest (SSSI) and statutory nature conservation sites. The investigation had a resolution of one square kilometre. The area and number of sites protected from acidification under different sulphur reduction scenarios were investigated. It was found, that even with a 80% reduction in sulphur emissions compared to a 1980 baseline, 250 sites in England and 86 sites in Wales were still left unprotected, partly due to deposition of nitrogen.

8.3. Empirically derived critical loads of nitrogen for terrestrial ecosystems

Empirically derived critical loads for nitrogen were first discussed at the UNECE workshop in Lökeberg in 1992 (Grennfelt and Thörnelöf, 1992). The concept has subsequently been adopted by the UNECE Task Force on Mapping, and was included in the manual for critical loads mapping (UBA, 1996).

In general three sources of information are available for an empirical assessment of critical loads. Firstly, vegetation and ecosystem monitoring covering areas with large gradients in atmospheric deposition, or long-term monitoring on sites where the nutrient level has changed, can give information on relationships between deposition and species composition. Secondly, long-term nutrient addition or clean rain experiments can, in addition, give information on the influence of different nutrient levels on ecosystem structure and function. Thirdly, dynamic models can be used in the interpretation of monitoring data and in forecast calculations showing long-term effects of nutrient addition. An overview partly based on the mapping manual (UBA, 1996) is given below.

8.3.1. Tree health

In the northern hemisphere the growth of trees has historically been limited by nitrogen, and nitrogen has been used as a fertiliser in limited areas of forest. But in the 1980’s, during the peak of reduced nitrogen deposition, positive effects on growth and biomass production were observed in many countries. It has, however, recently been recognised that excess nitrogen over a longer time scale can have adverse effects on forest health. When nitrified and leached, reduced nitrogen has an acidifying effect on the soil which, on poorly buffered soils, can lead to loss of base cations and mobilisation of aluminium. This can lead to nutrient imbalances. Root damage as a consequence of aluminium toxicity might lead to increased susceptibility to frost, drought, insect pests and pathogens. Another risk factor is an increase in shoot/root ratio as a consequence of increased nitrogen availability. Although changes in needle chemistry and nutrient circulation have been demonstrated in many studies, the complexity of the ecosystem and the long life-spans of trees make it difficult to set critical loads. A provisional value of 15-20 kg N ha^{-1} $year^{-1}$ for deciduous and 10-15 kg N ha^{-1} $year^{-1}$ for coniferous trees is considered further.

8.3.2. Ground-living and epiphytic lichens and algae

Epiphytic lichens are to a large extent dependent on the substrate on which they live, mostly the bark of different tree species. Apart from this they also depend on air nutrient supply. Most epiphytic lichens are more strongly affected by acidification than by nitrogen. However, 10% of the lichens that have cyanobacteria as photobionts are among the species most susceptible to increased nitrogen deposition. In the Netherlands, all cyanobacteria lichens present at the start of the 19th century are extinct today. The critical loads for these lichens are probably between 5-10 kg N ha^{-1} $year^{-1}$. As a contrast, the thickness and colonisation rate of layers of free living green algae on, for example, spruce needles increase with increasing nitrogen deposition. Results from Sweden and Norway indicate that these algae do not occur at deposition rates below 5 kg N ha^{-1} $year^{-1}$. At deposition rates above 10-20 N ha^{-1} $year^{-1}$, green algae can impede the photosynthesis of spruce trees (Bråkenhielm, 1991).

8.3.3. Forest ground vegetation, macrofungi and mycorrhizas

Vegetation changes in forest understorey vegetation following an increase in deposition, deposition gradients or nutrient addition have been shown in many studies. Long-term monitoring data from the Netherlands have shown a total disappearance of lichens and an increase in *Deschampsia flexuosa* and *Corydalis claviculata* as a response to an increase in nitrogen deposition from 20 kg N ha^{-1} $year^{-1}$ (in 1958) to 40 kg N ha^{-1} $year^{-1}$(in 1981) (Dirkse and van Dobben, 1989). In Sweden a distinct decrease was found for *Dentaria bulbifera*, *Pulmonaria officinalis*, and *Polygonatum multiflorum* as a response to a doubling in nitrogen deposition to 15-25 kg N ha^{-1} $year^{-1}$. At the same time an increase in 15 species, most of them nitrogen indicators, was observed (Falkengren-Grerup, 1986, 1995). Rosén *et al.* (1992) found a significant correlation between *Deschampsia flexuosa* in coniferous forest and the pattern of nitrogen deposition based upon data from a Swedish forest inventory. In Sweden the area cover of *Deschampsia flexuosa* in closed forest stands has been observed in 1973/77 (2000 observations) and in 1983/87 (13 000 observations). When mapped, the regional pattern of differences in area cover showed a striking similarity with the critical load exceedance map. The deposition level above which vegetation changes can be observed in this dataset is in the range 7-11 kg N ha^{-1} $year^{-1}$. A decrease in species diversity and abundance of macrofungi have been reported during the last two decades. The decrease might be caused by air pollution, especially the increased availability of nitrogen, but the cause-effect relationship is unclear (Arnolds, 1991). Similarly most studies show a dramatic decrease in the number of ectomycorrhizal species per forest. In the Netherlands, the number decreased from 71 between 1912 and 1954 to 38 between 1973 and 1982. Changes in mycorrhizal species composition or the loss of dominant mycorrhizal species in areas where diversity is already low may lead to increased susceptibility of plants to stress.

8.3.4. Heathlands

Heathlands can be divided roughly into five different categories according to broad differences in habitat: 1) dry heathlands, 2) wet heathlands, 3) species rich and acidic grasslands, 4) upland *Calluna* moorland, and 5) arctic-alpine heathland. Whereas heathland is a natural vegetation in montane and coastal areas of the Atlantic and sub-Atlantic parts of Europe, inland, lowland heathlands are semi-natural ecosystems formed as a result of agricultural practice in former times, although some have existed for centuries. On these heaths, the development towards woodland has been prevented by mowing, burning, grazing, sod cutting etc., and the heathlands are dependant on continuous management.

At present the critical loads for lowland heaths are mostly based on field data and experiments from the Netherlands. The effects of nitrogen deposition, nitrogen accumulation, outbreaks of the heather beetle (*Lochmaea suturalis*), management and species competition have been integrated in a dynamic ecosystem model, CALLUNA (Heil and Bobbink, 1993). The primary outcome of this model is the result of the competition between *Calluna* and the grass *Deschampsia*. It has been concluded from the simulations that the change from *Calluna* to grasses occurs at 15-20 kg N ha^{-1} $year^{-1}$. Western European lowland wet heathlands are dominated by *Erica tetralix*. On these heathlands, the cover of the grass *Molinia* increases with increasing nitrogen deposition. Berendse (1988) has on the basis of model calculations suggested a critical load of 17-22 kg N ha^{-1} $year^{-1}$. The model calculated critical load is set for the protection of the dominant vegetation. At this deposition level some sensitive species might, however, have already disappeared. A reduced species diversity has been observed on species-rich heath and acidic grasslands in recent decades. The decline or extinction of some herbaceous species, e.g. *Arnica montana,* has in the Netherlands been attributed to nitrogen deposition (Van Dam *et al.*, 1985). Furthermore, it has been suggested that upland *Calluna* moorlands and arctic heaths might be more sensitive to nitrogen deposition than lowland heaths, meaning that in these ecosystems effects of anthropogenic nitrogen may be more severe than in the above mentioned ecosystems.

8.3.5. Species-rich grasslands

Semi-natural grasslands are, like inland heathlands, man-made nature types that exist as a function of the (traditional) agricultural practice or management. Semi-natural grasslands are important habitats for many rare and endangered plant and animal species. Because of the low nutrient status of these ecosystems, which is a consequence of a long history of agricultural exploitation with low input of fertilisers, the semi-natural grasslands are expected to be susceptible to fertilisation and atmospheric deposition of nitrogen. A number of fertilisation experiments have been performed, especially on calcareous grasslands to determine the critical load.

8.3.6. Empirical critical loads for terrestrial ecosystems

Table 8.4 summarises the critical loads for terrestrial ecosystems based on empirical data. In addition to the actual ranges, the estimated uncertainty of the values is indicated ranging from reliable (based on experiments) to expert judgement.

8.4. Effects of increased nitrogen availability on soil fauna

Excess nitrogen acidifies the soil, and soil acidity is an important factor which affects the soil fauna both at species and group level (Hågvar, 1984). Commonly the species diversity of the soil fauna decreases with increased soil acidity. Most species of earthworm disappear at pH values lower than 5. As a response to soil acidification, lime and fertilisers are often used in forests in order to increase the mineralisation rates in the upper forest soil layers. An increase in pH value also increases the population of soil animals and might enable new species to establish. Huhta *et al.* (1967) reported that the common earthworm *Lumbricus rubellus* reacted to liming with a great population increase. This may, however, sometimes happen at the expense of those species specialised to low pH values.

8.5. Trends in the atmospheric deposition of nutrients

Atmospheric input is the most important anthropogenic source of nutrients in most natural and semi-natural terrestrial habitats. The most important nutrients contributed by atmospheric deposition are nitrogen and the base cations, calcium, magnesium and potassium. Monitoring and statistical data for modelling atmospheric deposition is in general not available from earlier than 1960. From 1960 to date, the RAINS model can be used to calculate deposition maps for Europe. Estimates of the development in deposition before 1960 can be based on historical data on economic activity, applied technology and estimates of emission-factors. The emissions of base cations is to some extent linked to sulphur emissions, because the anthropogenic sources are similar.

Table 8.4

Critical loads (CL) for terrestrial ecosystems based on empirical data. Uncertainty in the values is indicated ranging from reliable to expert judgement * = reliable, ** = quite reliable, * = expert judgement. After (UBA, 1996). See main references for detailed sources.**

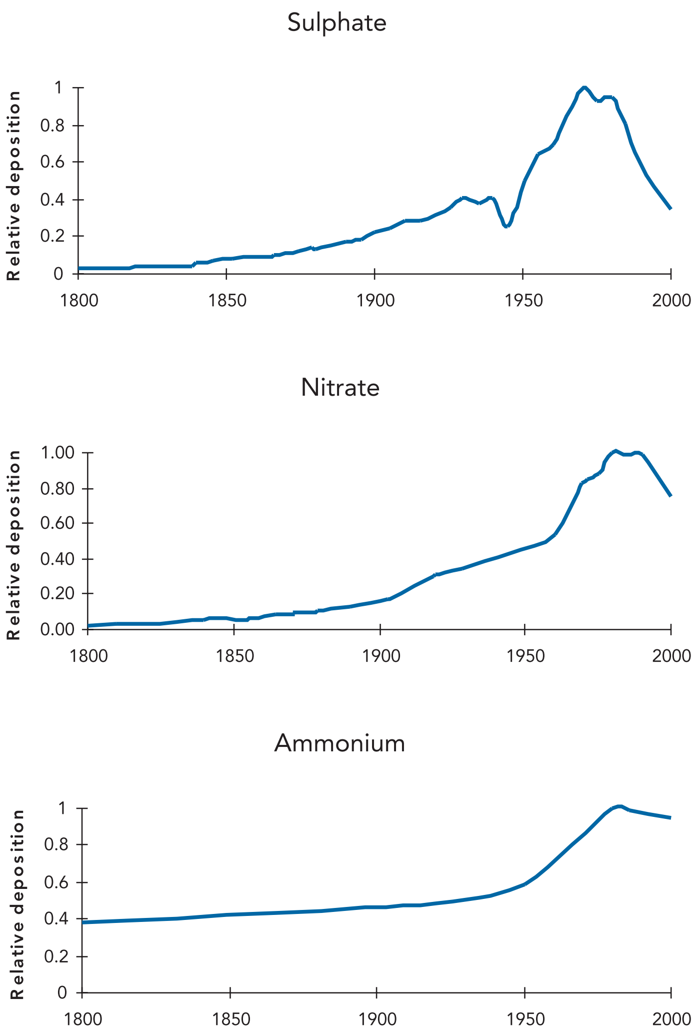
Ecosystem	CL kg N ha^{-1} $year^{-1}$	Relia-bility	Indication of exceedance
Trees and forest ecosystems			
Coniferous trees (acidic), low nitrification rate	10-15	***	Nutrient imbalance
Coniferous trees (acidic), high nitrification rate	20-30	**	Nutrient imbalance
Deciduous trees	15-20	**	Nutrient imbalance, increased shoot/root ratio
Acidic coniferous forest	7-20	***	Changing ground flora, mycorrhiza decrease, increased leaching
Acidic deciduous forest	10-20	**	Changing ground flora, mycorrhiza decrease
Calcareous forest	15-20	*	Changes in ground flora
Unmanaged acidic forests	7-15	*	Changing ground flora, mycorrhiza decrease, increased leaching
Trees and forest ecosystems			
Forests in humid climates	5-10	*	Epiphytic lichens decline, increase in free-living algae
Heathlands			
Lowland dry heathlands	15-20	***	Increasing grass dominance, functional changes, increased sensitivity to the heather beetle *Lochmaea suturalis*
Species-rich heaths/grasslands	10-15	**	Decline of sensitive species
Upland *Calluna* heaths	10-20	*	Decline of heather dominance, mosses and lichens; N accumulates
Lowland wet heathlands	17-22	**	Transition of heather to grass *Molinia coerulea*
Arctic and alpine heaths	5-15	*	Decline of lichens, mosses and evergreen dwarf-shrubs
Species-rich grasslands			
Calcareous grasslands, N limited	15-25	**	Increased mineralisation, N accumulation and leaching; change in
P limited	25-35	**	diversity and increase of tall grasses
Neutral – acid grasslands	20-30	**	Change in diversity and increase of tall grasses
Montane – subalpine grasslands	10-15	*	Increase of tall grasses, change in diversity

8.5.1 Model calculated European trends in deposition

The historical development in European deposition of SO_4, NO_X, and NH_X has been estimated by Walse *et al.* (1996). The trends are illustrated in Figure 8.1. A standard curve calculated from the average trend in sulphur emissions in 11 countries has been used to generate sulphur depositions from 1880-1960 (Mylona, 1993; Walse *et al.*, 1996). For the time before 1880, deposition measurements from the Rothamstead experimental station have been used as a trend indicator. Historical trends of ammonia emissions were estimated from European data from 1870 to 1980 (Asman and Janssen, 1987). Little data is available on the historical development in NO_X emission/deposition. These trends have been estimated from qualitative information and scaled to fit present day values. The RAINS model has been used to calculate the deposition from 1960 onwards.

The curves show a dramatic increase in the deposition of both nitrogen and sulphur from the start of the industrial revolution to peak values in the 1980s, only halted by a temporary decrease during the second world war. Following the first UNECE sulphur protocol, depositions of sulphur have been reduced by more than 50%. The reductions in NO_X, and particularly NH_X, depositions have been much smaller, and deposition values are from a historical point of view still very large.

Figure 8.1 **Model calculated curves for the average trends in atmospheric deposition of nitrogen and sulphur deposition from 1800-1990 (*Walse et al.*, 1996)**



8.5.2. Measured time series

Long time series of atmospheric deposition data are available from a few locations in Europe. The first rain gauge was built at Rothamsted in 1853, and precipitation / deposition data are available from that time. Total acid input has been reported by Johnston *et al.* (1986). The reconstructed deposition history is illustrated in Figure 8.2. (Sverdrup *et al.*, in press).

The measured time series from Rothamsted show a similar picture to the model calculated European trends. Deposition of sulphur dramatically increased until the 1980s where reductions in deposition began to occur. The deposition patterns for Ca, Mg, and K resemble the pattern of sulphur, although the percent change was much smaller. Nitrogen reductions were smaller than sulphur reductions.

Atmospheric deposition at individual sites has a large variation from year to year. Also the yearly variation in deposition can be substantial depending on the meteorological conditions and variations in especially local and marine sources. Figure 8.3 shows the yearly dynamic of litterfall, throughfall and stemflow of N, P, and K measured at the Polish monitoring area in the Ratanica river catchment (Godzik *et al.*, 1996). At a local scale, episodic events might have a substantial influence on ecosystem conditions.

Deposition history at the Rothamsted experimental station. Source: Sverdrup *et al.* (in press) **Figure 8.2**

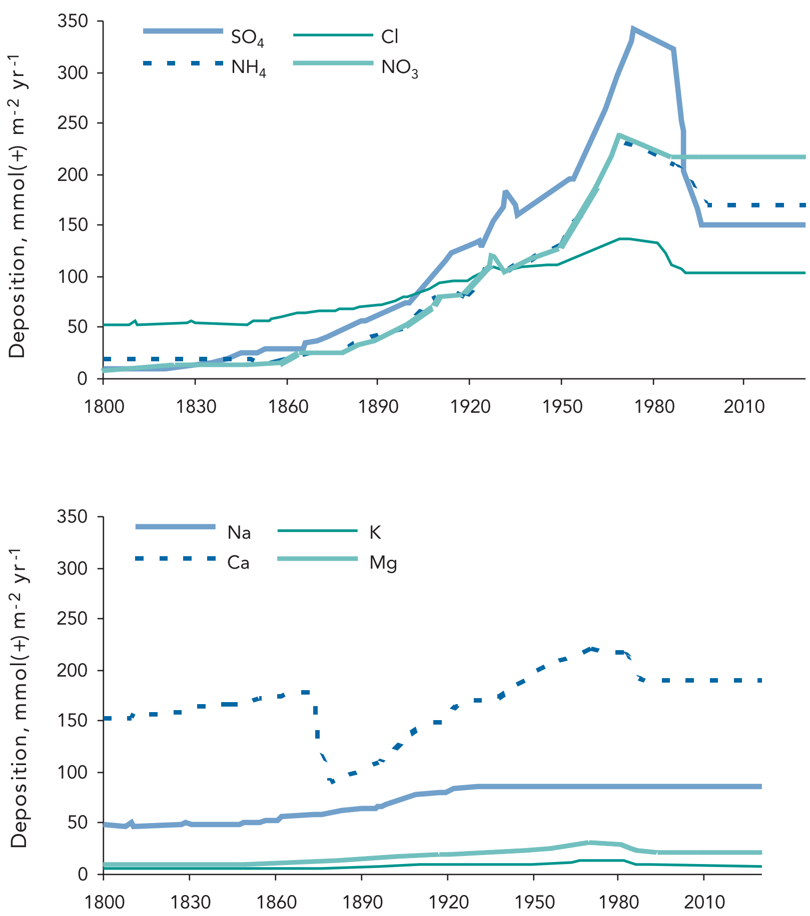


Figure 8.3 **Yearly dynamics of litterfall, through falland stemflow of N, P and K measured at the Polish monitoring area in the Ratanica river catchment. Source: Godzik *et al*. (1996)**

Note:
sp = stemflow:
op = throughfall:
os = litterfall

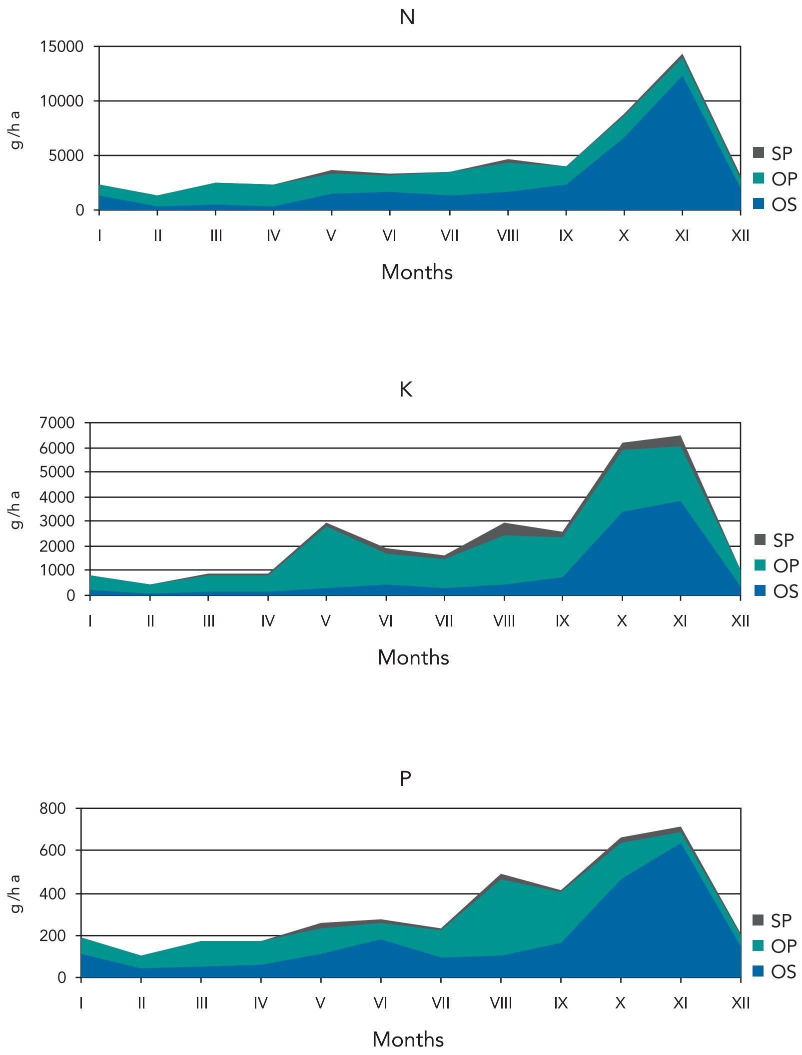
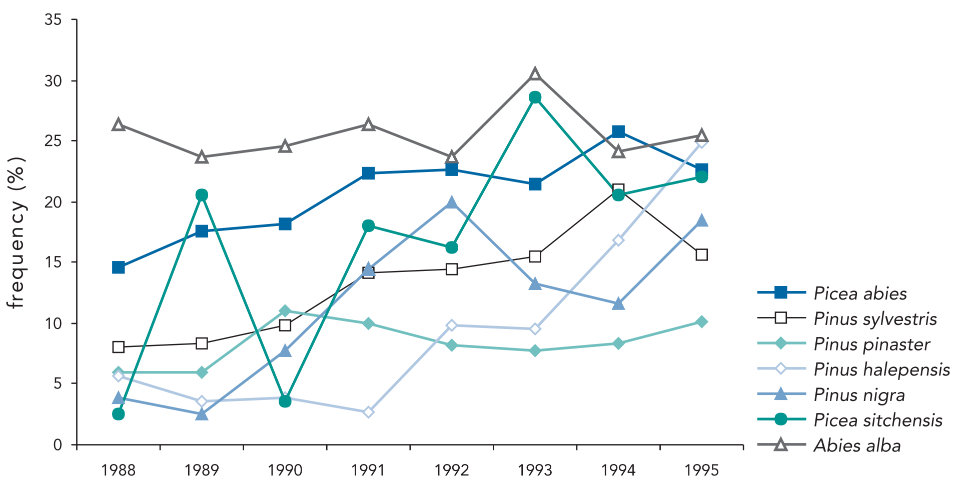


Figure 8.4 **Development in needle loss from 1988 to 1995 assessed on data from ICP forest. Source: EC-UNECE (1996)**



A statistical analysis of the changes in the flora in Scania in southern Sweden between 1938 and 1996 indicates that among the abiotic factors excess nitrogen gives the best explanation of the changes observed during the period. Of 400 taxa, 46 showed a decline of more than 75%. Changes in frequency of more than 15% had occurred for the majority of the taxa, the largest changes being observed for species growing in calcareous and in acidic fens as well as in unfertilised grasslands and oligotrophic lakes, while most forest plants had changed little. Large increases were observed in lake immigrants but also by some indigenous plants. The analysis of abiotic factors were based on Ellenberg *et al.* (1991) in Tyler and Olsson (1997).

8.6. Monitoring of forest health

Annual monitoring of forest health condition in Europe, United States and Canada has been conducted since 1986 by the UNECE, International co-operative programme on forest health. The monitoring programme has since 1992 been undertaken in co-operation with UNECE, ICP-forest, and the European Union, under Regulation 3528/86 on the Protection of Forest Against Atmospheric Pollution. The monitoring of forest health is performed in EU countries at a resolution of 16x16 km. The work is co-ordinated by the Programme Co-ordination Centre West (crown condition), Forest Soil Co-ordination centre (soil data) and Forest Foliage Co-ordination centre (foliage chemistry). Data are at present collected from 25 170 observation points in 30 countries. Soil data are available from 4491 observation points. In addition to the extensive monitoring of forest health, intensive monitoring data have been collected from 643 permanent monitoring sites in 26 countries since 1994.

A 1996 report from the monitoring programme concludes that forest damage is a serious problem in Europe. From 117 035 trees evaluated in 1995, 25.3% had needle/leaf loss in excess of 25%, 10.2% of the monitored trees had discoloration on more than 10% of the leaves. The national reports point to several possible reasons for the present forest condition. Drought and warm weather have had a large impact. Pests, grazing and the influence of human activities in the forest are other important factors. In certain areas, especially in central Europe, direct and indirect effects of air pollution are identified as one of the major causes of forest decline (EC-UNECE, 1996).

The development in needle loss from 1988 to 1995 is illustrated in Figure 8.4. The health condition of conifers has been fairly stable, except for *Pinus halepensis*, where the proportion of damaged trees has increased from 2.7-24.8%. For deciduous trees (not illustrated in Figure 8.4), a small deterioration in health conditions was recorded from 1992, except for *Quercus suber*, where health conditions have been varying dramatically over the period. In some areas, improvement in health is also reported.

8.7. Effects on nitrogen cycling at integrated monitoring sites

The Integrated monitoring programme (ICPIM) is part of the Effects Monitoring Strategy under the UNECE LRTAP Convention. The main aim of the ICPIM Is to provide a framework to observe and understand the complex changes occurring in the external environment. This is done by intensive, integrated monitoring in undisturbed reference areas. The monitoring sites are divided into two categories, intensive sites and biomonitoring sites. At the intensive sites (IM), data are collected from many compartments of the ecosystem for the purpose of applying complex models. Biomonitoring sites are less intensively monitored primarily with the purpose of quantifying the variation between sites. The data collection on these sites are focused on crucial parameters as input/output budgets. Fifteen countries have one or more intensive sites, seven countries have one or more biomonitoring sites. The total number of sites in the programme is 57 (Kleemola and Forsius (eds), 1996).

The response to increased nitrogen deposition has been investigated on data from the IM sites. The response of catchments/plots is supposed to pass through three stages: i) an assimilation phase, where nitrogen is accumulated in biomass and soil organic matter, ii) a saturation phase, and iii) a decline phase, where nitrogen is leached from the rooting zone. In the decline phase nitrogen losses might also occur in the growing season and under all hydrological conditions (Gundersen, 1995). Using the data from the IM sites, significant correlations were found between nitrogen compounds in bulk and throughfall deposition, and between nitrogen fluxes in litterfall, and both bulk and throughfall deposition. A significant correlation was also found between the nitrogen flux in litterfall and the nitrogen content in the organic layer. A large number of variables were tested in a multiple regression analyses in an attempt to find a model predicting nitrogen leaching. Data from other experimental sites in Europe was also included in this analysis. An example of the results of this analysis is illustrated in Figure 8.5. The regression equation is:

$$ntot\_o = 2.11n\_org + 0.23ntot\_t - 28.24,$$
$$(n=11,\ r^2=0.89,\ p<0.001),\ \textbf{Equation 4}$$

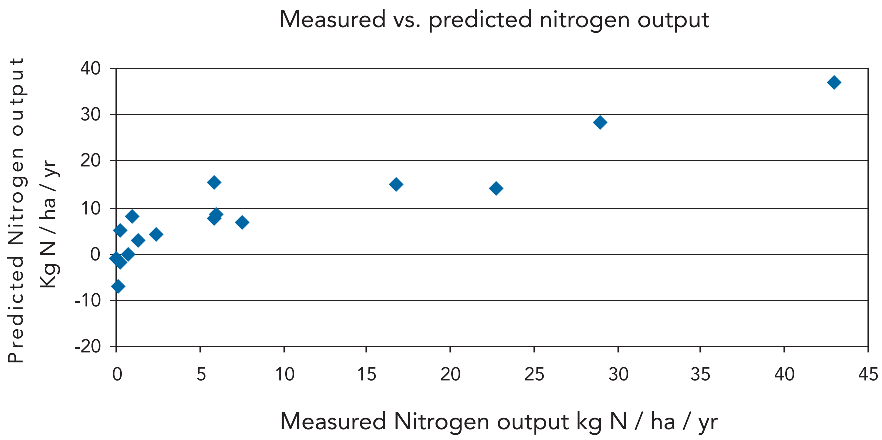
where: ntot_o = total nitrogen leaching, ntot_t = total nitrogen in throughfall, and n_org = nitrogen in soil organic layer. (Forsius and Kleemola, 1995).

The conclusion of the analyses on the IM data was that, in general, a critical deposition threshold of 8-10 kg N ha^{-1} $year^{-1}$ has to

Forest damage is a serious problem in certain parts of Europe, Karkonocze National Park, Poland. Photo: André Maslennikov/ BIOFOTO

Figure 8.5

Measured versus predicted nitrogen leaching. Data from the UNECE, IM programme with additional data from the EXMAN and NITREX programmes Effects of fertilisation on species composition in forests. Source: Forsius and Kleemola (1995)



be exceeded before nitrogen saturation occurs. This is in line with other findings, but does not exclude that nitrogen saturation can occur at lower deposition levels over a longer time frame. The output flux of nitrogen is strongly correlated with key ecosystem variables like nitrogen deposition, nitrogen concentration in organic matter and current year needles, and with nitrogen flux in litterfall.

Fertilisation with nitrogen in natural or semi-natural ecosystems only takes place in forests in order to increase production. Today this practice has been abandoned in areas where the atmospheric nitrogen deposition exceeds the amounts that are removed with timber. Numerous studies of the effects of nitrogen fertilisation have shown how, and in which cases, the understorey vegetation is affected (Becker *et al.*, 1992; Falkengren-Grerup and Lakkenborg-Kristensen, 1994; Huhta *et al.*, 1967; Kellner, 1993). The orchid *Goodyera repens* has been shown to disappear in nitrogen fertiliser experiments, whereas other species will recover after a few years, in the case of only one fertiliser application. It is known that other semi-natural ecosystems, such as oligotrophic shallow waters, raised bogs, wetlands, meadows and grasslands, are very sensitive to fertilisers. In these ecosystems fertilisers will change the community completely by an almost total elimination of the species special to the ecosystems.

8.8. Wetlands

Only those inland wetland ecosystems confined to waterlogged often peaty soils are dealt with in this section. In this context they comprise fens, bogs and shallow poorly buffered lakes. Bogs are peat moss based mires maintained by precipitation. Fens are often peat based either where monocotyledons invade and fill up lakes, or waterlogged areas supplied by surface or groundwater. Shallow poorly buffered lakes are nutrient poor lakes with clear water and a vegetation of bottom rosette plants (isoetids) i.e. *Littorella uniflora*, *Lobelia dortmanna* and *Isöetes spp*. Not dealt with further in this chapter are the coastal wetlands consisting of salt influenced ecosystems such as salt marshes and coastal deltas, or calcareous bogs.

8.8.1. Geographical distribution of wetlands

Because of the linkage between climate and peat formation, most peaty wetlands are situated in the northern temperate zone. The different kinds of bogs have their main distribution in different parts of this zone, but also occur in mountainous ranges throughout Europe.

Blanket bogs are favoured by extreme maritime conditions, where the climate is oceanic. They are widely distributed on the British Isles and on the west coast of Norway. The dominant species in blanket bogs are mosses i.e. ombrotrophic *Sphagnum* spp. and *Rhacomitrium lanuginosum*, the latter especially in the northern part of the zone.

Raised bogs are typical of the less oceanic, north-west corner of Europe, the boreo-nemoral and the southern part of the boreal zone. They are higher than their surroundings and receive all water and nutrients from rainwater. Hence they are naturally nutrient poor because of the low concentration of nutrients in rainwater. In the western part of the area they are primarily of the hummock-hollow type, whereas in the eastern part they are primarily of the forest-raised bog type.

Fens are widely distributed in the temperate zone of Europe wherever appropriate conditions appear. Their nutritional state depends upon the mineral content of the water supply. In oligotrophic fens or bogs, the water supply is poor in minerals and they are dominated by *Sphagnum spp.* They are typically formed along edges of nutrient poor lakes. They may eventually develop into raised bogs by cutting off the mineral water supply. They share a lot of species with the raised bogs. Mesotrophic fens are typical of alkaline to slightly acidic habitats in large parts of Europe. They are rich in *Carex spp*. and rare forbs e.g. *Pedicularis palustris* and several species of orchids. Shallow soft water lakes are oligotrophic water bodies distributed in the lowlands of western Europe on sandy soils poor in calcium carbonate. Because of their poor buffering capacity, and their nutrient poor state, they are sensitive both to acidification and eutrophication.

8.8.2. Protection of wetlands

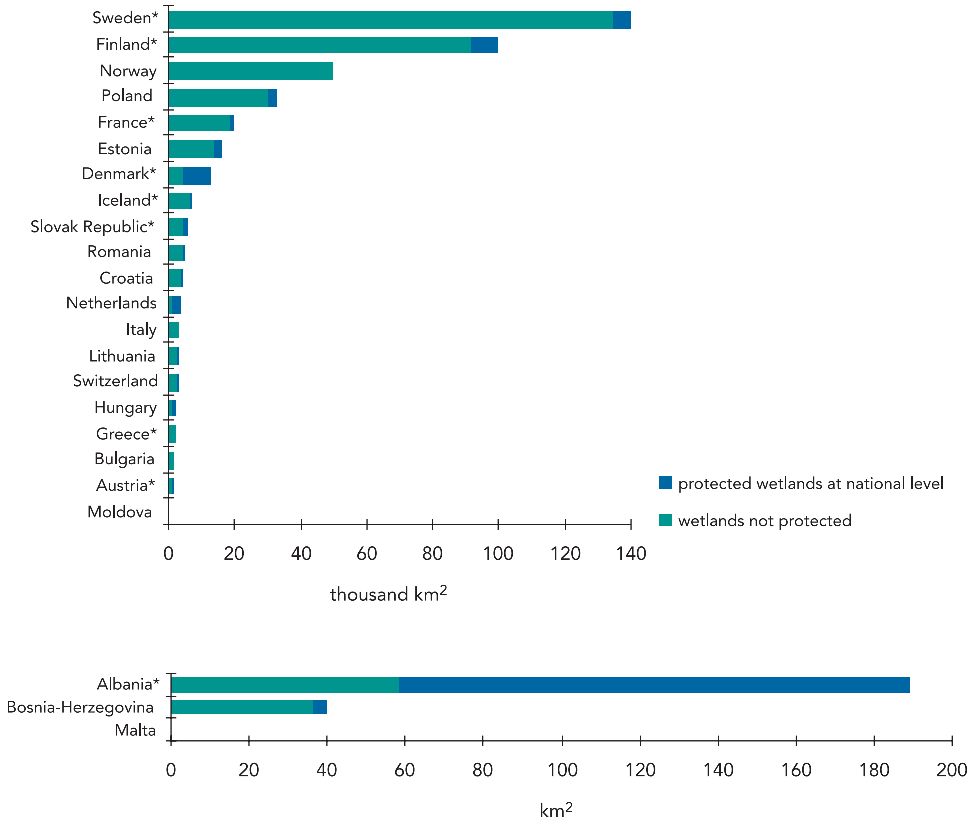
It was concluded in the Dobrís Assessment (EEA, 1995) that although most European countries are committed to extending the protection of wetlands, only a small fraction of the continent's wetland sites are directly protected. Figure 8.6 shows the latest compilation of data on protected wetlands in a number of countries (EEA, 1998). While in some countries the major part of all wetlands generally are protected as a habitat type, the proportion of these wetlands for which an excessive input of nutrients is a major threat cannot be assessed.

8.8.3. Empirically derived critical loads of nitrogen for wetlands

Empirical critical loads for wetlands are mainly based on British, Dutch and Scandinavian data. In the case of bogs and fens, observed effects from nitrogen addition experiments and vegetation monitoring programmes have clarified the relationship between nitrogen and vegetation change. The oligotrophic shallow water bodies resemble the poorly buffered Swedish lakes that suffered from acidification in

Excess deposition of nitrogen on oligotrophic fens can lead to the herbaceous layer become overgrown by bushes and trees. Photo: Niels Westergaard Knudsen/ BIOFOTO

Protected wetlands at national level. Source: Compiled by ETC/NC from questionnaire returns **Figure 8.6**



Notes: * estimated data
Wetland definition: Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres. (Art. 1.1 Ramsar Convention). Contrary to the Article 2.1 of the Ramsar Convention, 'riparian and coastal zones, adjacent to the wetlands, and islands or bodies of marine water deeper than six metres at low tide lying within the wetlands' are not included. Wetlands under general protection are not included.

Empirical critical loads for nitrogen deposition to wetlands. See comments on Table 8.4. Source: UBA (1996) **Table 8.6**

Ecosystem	CL kg N ha^{-1} year^{-1}	Reliability	Indication of exceedance
Wetlands			
Oligotrophic fens or bogs	5-10	*	Increased sensitivity to stress factors. Increased establishment of *Betula* and other trees. Increase of tall grasses.
Mesotrophic fens	20-35	**	Increase of tall grasses Decrease of biodiversity
Blanket bogs	5-10	*	Increase of *Sphagnum recurvum*, decrease of ombrotrophic species
Raised bogs	5-10	***	Increase of tall grasses and woody species. Increased mineralisation. Decrease of sensitive *Sphagnum* spp.
Shallow soft water lakes	5-10	***	Decrease of bottom rosettes (isoetid species)
21 Danish raised bogs	>21	**	Invasion by nitrophilic species from wet and dry habitats

Legend and source
*** = reliable,
** = quite reliable,
* = expert judgement.

the 1960s and 1970s. Hence the main effect of excess nitrogen supply to these systems is probably acidification. However, if phosphorus is not limiting, small amounts of nitrogen will influence such lakes before the critical load for acidification is exceeded, resulting in changes towards a mesotrophic state.

Table 8.6 summarises critical loads for wetland ecosystems based on empirical data. In addition to the actual ranges, an estimated uncertainty in the values is indicated ranging from reliable to expert judgement.

8.9. Liming of shallow poorly buffered lakes

Most European experience of liming soft water bodies has been gained in Sweden and Norway, where the bedrock weathers slowly, and where there is not enough capacity to neutralise man made acid deposition. The aim of liming these acidified waters is to detoxify the water from increased levels of hydrogen ions and metals like aluminium and cadmium, which are mobilised by the soil acidification, and to restore the water's chemical conditions so that natural aquatic life can survive and not only acid tolerant species like *Sphagnum* spp. and *Juncus bulbosus*.

A common criterion is that lakes and streams exposed to acid deposition exceeding 3-5 kg sulphur ha^{-1} $year^{-1}$, and with a water pH equal to or less than 6, may be treated (Henrikson and Brodin (Eds.), 1995). Liming increases the phosphorus availability and so the productivity of water. Liming, however, very seldom causes eutrophication of the treated lakes, as the natural levels of phosphorus and nitrogen are still very low. The ongoing forest soil acidification immobilises the leaching of phosphorus from soil to the water.

8.10. Conclusions

Some nature areas are sensitive to increased inputs of nitrogen and endangered by the present deposition load. The threat of air pollution to European ecosystems has been assessed as a basis for the negotiations under the Geneva Convention on Long Range Transboundary Air Pollution, and the EU Community Strategy to Combat Acidification. Current estimates based on 1990 deposition values indicate that 38 million ha of nature areas in the EU-15 countries, or 34% of the nature area considered, receive nitrogen deposition above their critical loads for eutrophication. For the whole of Europe the number is 77 million ha or 18% of the nature areas considered. Fifteen percent of the nature areas of Europe receive acid depositions above their critical loads for acidification. These nature areas are primarily forests, lakes, and, also for some countries, other sensitive ecosystems like heathlands, bogs and pastures. The areas are not necessarily protected by national legislation or other international conventions. The choice of areas (by National Focal Points) implies, however, that these are the areas the individual countries want to see protected from transboundary air pollution. Currently agreed abatement measures will in the year 2010 reduce the area receiving nitrogen depositions above the critical load for eutrophication to 19% within the EU-15 countries, and 11% for the whole of Europe (Amann *et al*., 1996).

This assessment has, however, mainly considered production forests. As described in this chapter, many natural or semi-natural ecosystems are sensitive to nitrogen deposition, and these ecosystems are not necessarily protected by the critical loads calculated for forests. Thus the problems concerning all terrestrial habitats are probably larger.

There is still an ongoing scientific debate concerning the long term effects of nitrogen depositions on forest health, (e.g. Binkley and Högberg, 1997). On the basis of existing monitoring data, it has not yet been possible to demonstrate a clear-cut relationship between critical load exceedances and effects on forest health. However, research findings and detailed studies on the nitrogen cycle at intensively monitored sites does indicate that long term effects can be anticipated where the deposition exceeds the critical load.

9. Protective and curative measures

9.1. Strategic goals

Concern about elevated nutrient concentrations and adverse eutrophication effects has prompted the introduction of many reduction strategies at international, national and local levels. An important element of the strategies are the goals set, particularly where these are numerical targets, since these allow progress to be measured and the success of the policies adopted to be assessed. Table 9.1 lists some of the goals set in the EEA area over the last 20 years.

9.1.1 International goals for reducing inputs to sea

In 1974, two important conventions were signed to prevent marine pollution from land-based sources in the North Sea and Baltic Sea areas, respectively, the Paris Convention and the Helsinki Convention. Both conventions adopted targets (for the North Sea initially agreed by Ministers at the North Sea Conference in 1987) to reduce inputs of nitrogen and phosphorus by 50% from 1985 to 1995 where these inputs are likely, directly or indirectly, to cause pollution (in 'problem areas with regard to eutrophication'). A key aspect of both Conventions was the agreement to share information on the status of the seas, on nutrient pressures and on appropriate responses. Thus under the Paris Convention, for example, information was sought to further identify the sources of nutrient inputs and improve information about eutrophication effects, and, Parties were required to provide all possible information, including national progress reports on the implementation of measures, and to elaborate a common basis for measurement.

PARCOM which administers the Paris Convention has made extensive recommendations addressing the major sources of nutrients, focusing in latter years on agriculture. As a result of the measures adopted, most Contracting Parties bordering the North Sea are expected to achieve a reduction in inputs to nationally defined problem areas, of the order of 50% for phosphorus, but only 20-30% for nitrogen mainly because the reductions expected from the agricultural sector have only partially been achieved.

Recent political changes allowed the Helsinki Convention in 1992 to be extended to include 6 countries from Central and Eastern Europe (CEE), and to be redrafted requiring the use of Best Available Techniques (BAT) for industrial sources, and best environmental practice for diffuse inputs. One hundred and thirty two 'Hot Spots' were identified for concrete action and recommendations were made for policy development, institutional strengthening and investment in both urban treatment works and agricultural remedies.

In the Baltic Sea, progress has been slow, despite the comprehensive work carried out. The 50% reduction in total nutrient loads called for in the 1988 Ministerial Declaration has been achieved by only a few of the countries. Less progress has been made in addressing industrial hotspots in the countries of Central and Eastern Europe, mostly because of lack of investment. As with the North Sea, agricultural inputs are proving the most difficult to control.

At the 4th North Sea Conference (1995), reduction targets for nitrogen were updated with the stipulation that the measures to be carried out in the area of municipal treatment plants and agriculture should be intensified. OSPAR (successor to PARCOM and the Oslo Commission) is currently working on measures to help achieve these aims. One of the key requirements is to achieve 'balanced fertiliser application'. However, problems in attaining a common operating definition is delaying recommendations and their implementation in these areas.

The Convention on Protection of the Mediterranean Sea (MEDPOL) was adopted in 1976. Although target reductions were not set, an assessment of the State of Eutrophication in the Mediterranean Sea made under the Convention has resulted in the formulation of recommen-

Table 9.1 **International goals for nutrient reduction. Source: Compiled by ETC/IW**

Objectives	Actions achieved
a) European Union 5th Environmental Action Programme	
• Surface fresh water – towards better ecological quality and safeguarding existing high quality	• Proposals for ecological quality of surface water (COM(93)680) incorporated into Water Framework Directive.
• Examine need for Directive to reduce phosphate	• No Directives developed; efforts to reduce phosphorus in urban waste water considered appropriate.
• 30% reduction in NOx emissions	• It is unlikely that the targets will be achieved in the timescales set. For NOx, a 20% reduction is expected by 2000 due to the introduction of catalytic converters on vehicles but the level of traffic is expected to grow
• Regional targets for NH_3 emissions	• Proposed – Inventory of NH_3 emissions, standards on new farm buildings
• Groundwater and surface fresh water – integration of resource conservation and sustainable use criteria into other policies including agriculture, land-use, planning, and industry	• The Commission adopted a proposal for an Action Programme for Integrated Groundwater Protection and Management. The plan addresses both qualitative and quantitative aspects of water management. One of the main themes of the programme is the integration of groundwater protection requirements into other policy areas, focusing in particular on the Common Agricultural Policy and on Regional Policy. • Nitrate Directive • Urban Waste Water Treatment Directive • Regulation 1765/92 – set-aside • Regulation 2078/92 – extensification
b) International Agreements **Danube Action Plan**	
• Maintenance of nutrient levels at 1995 levels By 1997:	• National Action Plans required
• emission limits for fertiliser plants, new industrial enterprises and livestock units,	• To date only one National Action Plans has been drafted
• establishment of national load reduction targets for high priority rivers By 2005:	• No integrated management plans have been completed • The evaluation of the nutrient discharges has not been undertaken
• regulations for fertiliser storage, handling and application, • environmentally sound agricultural policy reforms, • best environmental practice for the use of fertilisers and pesticides, • completion and application of pilot and demonstration projects for manure handling storage and disposal, • ban on phosphate detergents, • investment in priority waste water treatment plant	• To early to gauge success of measures
Rhine Action Plan	
• 50% reduction of total phosphorus and nitrogen by 1995 • 90% of communities connected to sewerage systems with subsequent biological treatment by 2000	• 50% reduction for phosphorus achieved 3 years early. • Only 20-30% reduction of nitrogen expected by 2000 • Diffuse inputs, particularly of nitrogen very difficult to achieve, resulting in failure to meet 50% reduction target • progress being made but still a lot to be done

continued

International goals for nutrient reduction. Source: Compiled by ETC/IW **Table 9.1**

Objectives	Actions achieved
Elbe Action Programmes	
• First action programme 1992 to 1995 aimed at: substantially reducing loads from Elbe catchment to the North Sea; achievement of near-natural aquatic ecosystem; and, to make river suitable for sophisticated uses. • long-term action programme for 1996 onwards aimed at further reduction of pollution in Elbe.	• marked improvement in Elbe water quality and decrease in loads to North Sea
Convention on the Protection and Use of Transboundary Watercourses and International lakes	
• To prevent control and reduce pollution of water causing or likely to cause transboundary impact • To ensure that transboundary waters are used with the aim of ecologically sound and rational water management, conservation of water resources and environmental protection	• Measures required for prevention, control and reduction of water pollution • Signed by 15 western European countries (except Iceland, Ireland and Liechtenstein) and 10 CEE countries. In addition, Croatia and Moldova have ratified but not signed • Convention came into force on 6 October 1996
• To ensure that transboundary waters are used in a reasonable and equitable way, taking into particular account their transboundary character, in the case of activities which cause or are likely to cause transboundary impact • To ensure conservation and where necessary restoration of ecosystems	• Information on progress not available
Strategic Action Plan for the rehabilitation and Protection of the Black Sea (October 1996)	
• reduction of nutrient loads in rivers (particularly the Danube) until Black Sea water quality objectives are met	• not known, Basin Wide Strategy proposed (links with Danube Action Plan)
• reduction of pollution from point sources by 2006: first progress report required by 2001	• list of high priority sites (hot spots) developed
• each Black Sea state to develop National Strategic Plan for point source reduction	• progress not known
• significant reduction of inputs of insufficiently treated sewage from large urban areas by 2006	• progress in developing comprehensive national studies not known
Helsinki Convention – Baltic Sea Joint Comprehensive Environmental Action Programme (1993 to 2012)	
• identification of all major point sources of pollution (hot spots)	• 132 initially identified, 47 of which were given priority status for action: 66% in the transition countries
• undertake remedial (preventive and curative) actions at hotspots	• progress not evenly distributed: well underway in Scandinavian countries, Finland and Germany, and also strong support in Baltic states and Poland • Actions at hotspots are anticipated to decrease loads by about 40% for phosphorus and 30% for nitrogen during the period 1991-2000
Helsinki Convention Ministerial Declaration 1988	
• 50% reduction of the total load of nutrients, heavy metals and toxic, persistent and bio-accumulating organic compounds to the Baltic Sea by the year 1995	• Although some countries have met the target, the overall 50% reduction will not be realised until the year 2020 • In some CEE countries the decrease in nutrient loading has been reached mainly due to decreased use of fertilisers and decreased agricultural production caused by structural changes and economic difficulties. Economic recovery might again lead to an increase in the agricultural run-off

continued

Table 9.1 **International goals for nutrient reduction. Source: Compiled by ETC/IW**

Objectives	Actions achieved
Oslo and Paris Commission (OSPAR) – North Sea Ministerial Conferences. Hague Conference 1990	
• Reduce inputs of nitrogen and phosphorus into (nationally defined problem) areas where these inputs are likely, directly or indirectly, to cause pollution, by around 50% by 1995	• Most countries were expected to achieve a reduction of 50% for phosphorus inputs, and 20-30% for nitrogen inputs, into problem areas by 1995 • overall reduction target for nitrogen inputs has not been reached mainly because losses from agriculture have proved to be more difficult to influence than anticipated, and because the measures adopted have been inadequate or inadequately implemented
North Sea Ministerial Conferences, Esbjerg 1995	
• To reach reduction targets set for 1995 as soon as possible	• Suite of instruments to be developed via other international agreements • implementation of the Urban Waste Water Treatment (UWWT) and Nitrate Directives, in particular, to apply the measures for sensitive areas under the UWWT and for vulnerable zones under the Nitrate Directive for the catchment of the North Sea (UK and F exceptions), and regulations on agricultural policy • the comprehensive strategies proposed by OSPAR • balanced fertiliser application subject to agreement on operational definitions (OSPAR to develop) • NOx emission reductions particularly from transport (via the EC and UNECE); • NH_3 emission reductions targets (to be developed by OSPAR)
Mediterranean Action Plan	
• To take all appropriate measures to prevent, abate, and combat pollution of the Mediterranean Sea area	• Information on progress not available or difficult to assess
Arctic Monitoring and Assessment Programme	
• to reduce and ultimately eliminate airborne and seaborne pollution such as that from heavy metals, greenhouse gases, PCBs, DDT and chlorinated hydrocarbons	• A State of the Arctic Environment report was published in 1997 • to soon to assess progress

dations to prevent and remediate nutrient inputs from sewage treatment, agriculture and industry. As with PARCOM and HELCOM, the MEDPOL Action Plan requires that monitoring programmes should be established to cover those areas showing clear signs of eutrophication, and that an inventory of land based sources be established to link to the monitoring programmes, and to provide scientific information as required for modelling and control policies.

The Strategic Action Plan for the rehabilitation and protection of the Black Sea (1996) also sets (non-numerical) targets for the reduction of pollutant loads to the Black Sea particularly of nutrients in rivers and also from priority ‘hot-spot’ point sources and from insufficiently treated sewage in urban areas. Significant reductions in the latter two categories are required by 2006 through the development and implementation of National Strategy Plans, and a first report on progress is required for a Ministerial Conference scheduled for 2001.

9.1.2. EU requirements

Numerical targets have also been set at an EU level in the Fifth Environmental Action Programme which, amongst other things, set target reductions for NOx emissions to air of 30%, and for zero nitrate levels in groundwater over the period 1985-2000. Progress towards these targets has not been good. NOx emissions increased between 1985-1990 across the EU by 7%, although reductions were achieved in Denmark, France, Germany and the Netherlands (EEA, 1995a), despite measures to restrict emissions from combustion plants and vehicles (see Section 9.5). Nitrate levels in groundwater are also continuing to rise in some EU Member States, although the Nitrate Directive, if effectively implemented, should help move towards this goal. There are also directives requiring Member States to reduce inputs from point sources, namely the Urban Waste Water Treatment (UWWT) Directive, and the Directive on Integrated Pollution Prevention and Control (IPPC) (see Section 9.2).

Other EU Directives set goals for protecting environmental quality in the form of standards for specified uses. Thus the Surface Waters, Drinking Water, Bathing Water, and Fresh water Fish Directives contain standards relating to nitrogen and phosphorus for different water uses (see

International standards for water quality — **Table 9.2**

Media/use	Parameter Phosphorus mg P_2O_5 l^{-1}	Nitrogen KjN (mg N l^{-1})	 Inorganic (mg l^{-1})	Ref
Drinking water	5 (as P_2O_5) MAC 0.4 (as P_2O_5) GL	1 (as KjN) MAC 0.1 (as NO_2) MAC	50 (as NO_3) MAC 25 (as NO_3) GL	Directive (80/778/EEC)
Drinking water abstraction	0.4 (90% ile Guide)A1 0.7 (90% ile Guide)A2/3	1 as KjN (90 Guide)A1 2 as KjN (90) A2 (90 Guide)A1 3 as KjN (90) A3	50 as NO_3 (95 M) (A1-A3) 25 as NO_3	Directive (75/440/EEC)
Fresh water fish			0.04[a] (95% ile) total as NH_4 Guide 1.0 (95% ile) total as NH_4 0.005 (95% ile) un-ion as NH_4 Guide 0.025 (95% ile)un-ion as NH_3	Directive (78/659/EEC)

Notes:
MAC Maximum admissible concentration
GL Guide level
a A value of 0.2 applies for cyprinid fish
Values are mandatory unless otherwise indicated.
un-ion Unionised
KjN Kjeldahl Nitrogen is a measure of reduced nitrogen in water (organic nitrogen + ammonium)

Table 9.2), and the Air Quality Framework Directive sets standards for levels of NO_x in air. Member States are required to take measures to ensure that these environmental quality standards are met and thus the measures taken can vary from country to country.

Recent proposals from the EU propose a more holistic approach for protecting waters. Thus the Commission recently prepared a Action Programme for Groundwater Protection and Management (COM/482/96) which focuses on changing practices in agriculture. It is also currently working on a proposal for a Water Framework Directive (COM(97)49 final) to protect surface water and groundwater quality and quantity in a co-ordinated way and incorporating many of the recommendations of the Groundwater Action Programme. A Directive to reduce phosphate inputs was considered but the controls exerted via the Nitrate Directive are currently considered adequate.

9.1.3. Other international agreements

Several other international agreements have addressed nutrient enrichment, for example:

- the Rhine Action Programme;
- the Elbe Action Programme;
- the Strategic Action Plan for the Danube River Basin;
- the Environmental Action Programme for Central and Eastern Europe; and,
- the UN Convention on the Protection and Use of Transboundary Watercourses and International Lakes.

The Rhine Action Programme, agreed in 1985 sets 50% reduction targets for inputs of total-phosphorus and NH_4-N to the river Rhine. Participating countries were required to reduce impacts from point sources through Best Available Techniques (BAT) and from agriculture through Best Environmental Practices (BEP).

The Action plan for the Danube River Basin and the Environmental Action Programme for Central and Eastern Europe do not set quantitative goals for reducing nutrient inputs, but they make recommendations or introduce requirements for signatory countries to take measures to reduce nutrient inputs. For example, the Strategic Action Plan for the Danube sets short, medium and long term targets for implementing policies to reduce nutrient inputs from urban waste water treatment plants, industry and agriculture as a framework for national action plans. The Action Plan supports and complements the Convention on Co-operation for the Protection and Sustainable Use of the Danube River Basin (DPRC) which had been signed and ratified by six countries (Austria, Croatia, Czech Republic, Germany, Hungary and Romania) by September 1997.

The Environmental Action Programme for Central and Eastern Europe, in setting priorities in 1993, identified the highest priority problems reflecting the limited resources available for environmental improvements in Central and Eastern Europe for the next 10 years or so. Primary concern was given to the damage to human health caused by poor environmental quality, including health impacts arising from nitrate in water from inadequately maintained or designed feed lots and agricultural enterprises, inappropriate fertiliser application, and rural septic tanks.

The UN Convention on the Protection and Use of Transboundary Watercourses and International Lakes (1992) was signed by 25 countries by end of May 1997 including all EEA countries (except Iceland, Ireland and Liechtenstein). Twenty countries had also ratified the Convention by this date. The Convention requires signatories to prevent, control and reduce pollution of water causing or likely to cause transboundary impact with the aim of ecologically sound and rational water management, conservation of water resources and environmental protection. The measures to achieve these objectives would incorporate, for example, application of BAT to reduce nutrient inputs from industrial and municipal sources, and best environmental practises for reduction of nutrients from diffuse sources (especially from agriculture).

9.1.4. National and local goals

In addition to goals set internationally, some countries, or parts of countries, have adopted their own national targets particularly in countries were eutrophication problems are at their greatest. These tend to be more specific, in that they focus immediately on areas of national concern. The Danish Action Plan, for example, focuses on reducing the loss of nitrate from agricultural land, setting in 1987 a target of 50 % reduction and establishing policies on farming practices to achieve these targets. Policies were refined in 1991, 1996 and 1998 in the light of progress.

9.2. Measures to reduce point source inputs

Measures to reduce point source discharges are focused on urban waste water treatment plants and also some key industries. At an international level, traditional command and control approaches are widely used. Common standards as well as common principles are adopted in some cases. At a national level, instruments such as charging schemes and voluntary agreements are being used to supplement the more traditional approaches.

9.2.1. Urban waste water

The Urban Waste Water Treatment Directive (91/271/EEC) is a key directive for water management in the EU. The Directive sets minimum standards for the collection, treatment and disposal of waste water dependent upon the size of the discharge, and the type and sensitivity of the receiving waters. The major requirements of the Directive are given in Table 9.3.

For discharges to areas ‘sensitive’ to contamination from nitrogen and/or phosphorus, secondary treatment with more stringent treatment must be installed.

Typical requirements for controlling discharges from Urban Waste Water Treatment Plants. Source: Compiled by ETC/IW **Table 9.3**

1. The Urban Waste Water Treatment Directive– sets emission limits for discharges to sensitive areas subject to eutrophication (one or both parameters – N and/or P – may be applied depending on the local situation).

	Annual average concentration	Or % reduction in relation to the load of the influent
Total P	2 mg P l^{-1} (10 000-100 000 PE)	80
	1 mg P l^{-1} (>100 000 PE)	
Total N	15 mg N l^{-1} (10 000-100 000 PE)	70-80
	10 mg N l^{-1} (>100 000 PE)	

2. PARCOM (1989) – Recommendations for new and large treatment works

Municipalities with waste water discharges of more than a few thousand PE should be connected to a waste water treatment plant with more than sludge separation.

New and enlarged UWWTP of >20 000 PE should be equipped with nitrogen and phosphorus removal so as to achieve a reduction efficiency of at least 70 %. For smaller new plants, provisions for a later installation of such facilities should be made. For nitrogen removal, specific difficulties due to cold climates will be considered.

3. MEDPOL

- emphasis on larger discharges – add requirements

4. HELCOM

Makes similar recommendations to PARCOM but in addition lists 132 Hotspots (mainly point sources) for action including many urban waste water treatment plants

Notes: PE= Population equivalent

Sensitive areas are broadly described as:

- surface waters (inland, estuaries and coastal waters) subject to eutrophication;
- fresh surface waters, intended for abstraction of drinking water, having, or likely to have a nitrate concentration of >50 mg l^{-1}; and,
- areas where further treatment is necessary to fulfil the requirements of other Directives.

The Directive defines common standards for treatment focusing on the control of nitrogen and/or phosphorus inputs from larger discharges to areas more sensitive to pollution. Member States must designate sensitive waters and the impact of the Directive will depend to a large extent on the extent of designation within each Member State. Thus, for example, Denmark, Finland, Sweden, Luxembourg and the Netherlands are not obliged to identify sensitive areas as they apply more stringent treatment (than required) throughout their territory (CEC, 1999). In contrast, Belgium, Germany, Spain, France, Ireland, Portugal, and the UK have designated a 'patchwork' of water bodies as sensitive areas, requiring treatment more stringent than secondary treatment only within these defined areas. Thus in effect nutrient removal will be required in all plants with a capacity above 10 000 PE, or an overall reduction of 75% for total nitrogen and phosphorus load must be reached. Austria has stated that there are no sensitive areas in its territory, and Greece and Italy had made no formal designations by October 1998. The UK and Portugal have also designated 'less-sensitive' areas allowing less stringent treatment in areas with high natural dispersion characteristics, and where there is low risk of eutrophication effects occurring. The Commission is in the process of verifying whether or not the criteria for the identification of sensitive and less sensitive areas have been respected by Member States. There is also some indication that Greece might designate sensitive areas, and Italy sensitive and less sensitive areas (EWWG, 1997).

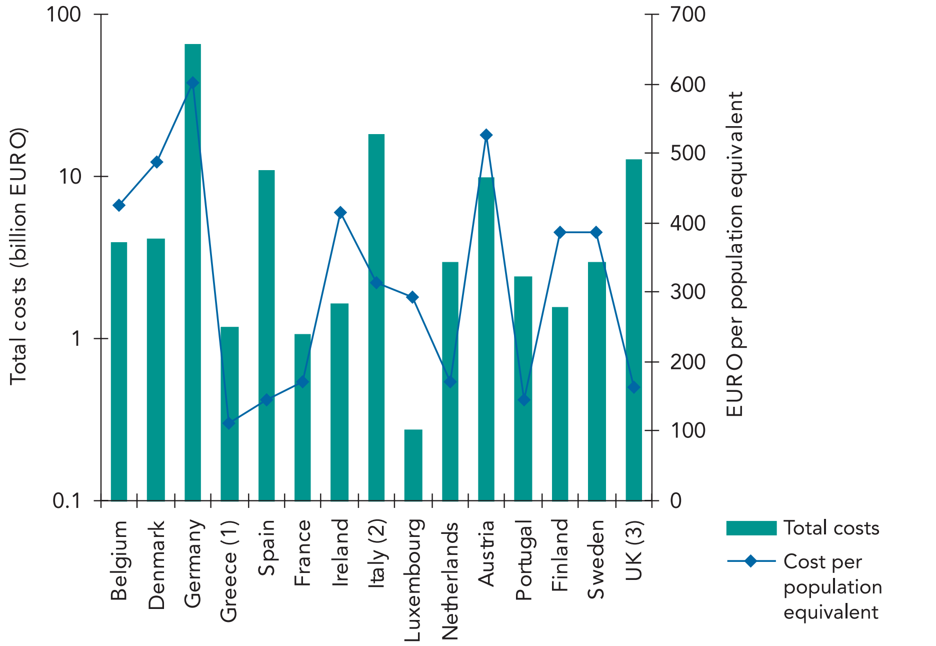
The costs of implementing the Directive are likely to be considerable particularly for countries such as Germany (due to the legacies of East Germany), Italy, Portugal, Spain, UK and France where major investment will be needed to build new infrastructure. This will result in higher water charges to users. The forecasted total investments in the 14 EU Member States, that provided information (Italy did not provide information) for a European Commission's report on progress with implementation of the Directive (CEC, 1999), amount to 130 billion EURO, 53% of which is for collecting systems and 47% for treatment plants. Figure 9.1 illustrates the total forecast costs and costs per population equivalent for each Member State. The forecasted costs per population equivalent (PE) varies between 112 EURO per PE in Greece and 602 EURO per PE in Germany, and equate to an average cost for the 14 Member States of 307 EURO per PE.

Many North Sea countries had already made a commitment to reduce nutrient levels from Urban Waste Water Treatment Plants under the PARCOM agreements. Thus in 1995, PARCOM reported that in most North Sea countries:

- the treatment capacity for municipal waste water had significantly increased between 1985 and 1995;
- 80% or more of households and industries connected to sewerage would have secondary and tertiary treatment;
- nutrient inputs from urban waste water treatment plants had been reduced by 8-74% and phosphorus discharges by up to 73% between 1985-1995.

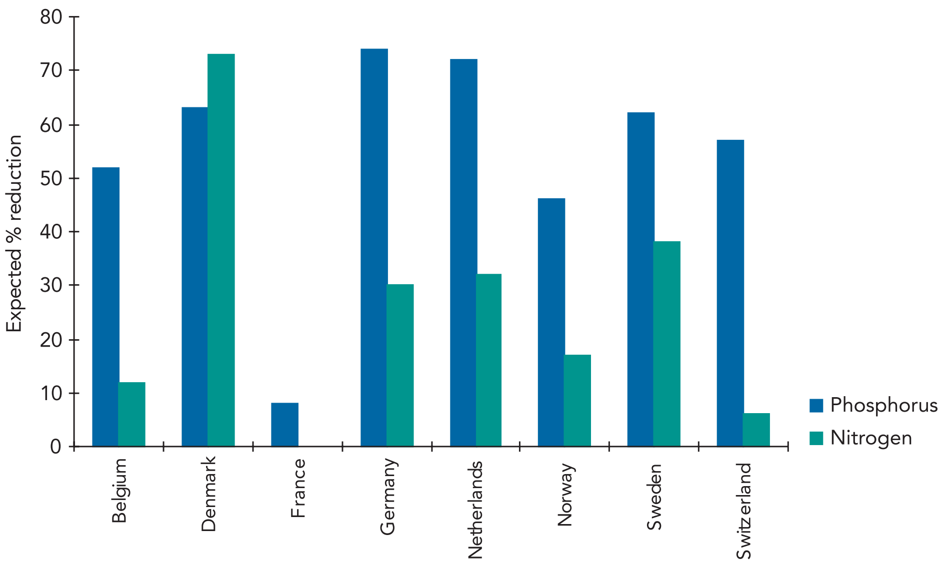
Figure 9.2 shows the expected reductions in nutrient loads from Municipal Treatment Plants between 1985 and 1995 in the OSPAR countries.

Forecast investment costs for the period 1993 to 2005 for implementing the requirements of the Urban Waste Water Treatment Directive in EU Member States. Source: CEC (1999) unless otherwise stated **Figure 9.1**



Notes:
Total costs for collecting systems and treatment plants at 1994/1995 values
1 1993 to 2000 only
2 Source: EWWG (1997). Costs are per person not population equivalent
3 Value 1996/1997 for UK

Expected reduction of nitrogen and phosphorus from Municipal Treatment Plants between 1985-1995. Source: OSPAR (1995) **Figure 9.2**



9.2.2. Industrial emissions to water

Emissions from certain industrial sectors have also been targeted for nutrient reduction generally by the application of Best Available Techniques (BAT). Table 9.4 lists the industries identified as significant sources of nutrients by the North Sea countries. Extensive measures to reduce discharges of nutrients from industrial sectors, for example, by setting stricter discharge limits or by introducing cleaner technologies have achieved significant reductions in emissions, often by controlling a small number of key emissions. In the PARCOM area estimated reductions for the period 1985 to 1995 of up to 96% were achieved for phosphorus (in Denmark) and up to 70% for the nitrogen discharges (in Norway).

Under Article 13 of the Urban Waste Water Treatment Directive, all the industrial waste waters from food industry plants of more than 4000 PE, and not connected to urban waste water systems, must be treated before the end of 2000.

In the future most industrial emissions in EU Member States will be subject to the Council Directive on Integrated Pollution Prevention and Control (96/61/EEC). The Directive addresses nutrients and NOx as important parameters requiring control through the application of BAT defined nationally by Member States. It supersedes directives previously governing emissions to single media, aiming to attain 'a high level of protection for the environment as a whole'. The Directive allows for common EC emission standards to be adopted at a later date but at present the effect of implementation on NOx emissions will depend on Member States interpretation of BAT.

9.2.3. Financial incentives – charging schemes

As well as command and control approaches, some countries have adopted alternative methods to control nutrient emissions from industry. Countries such as Germany and the Netherlands, for example, charge for discharges of, amongst other things, nutrients. In Germany, dischargers must pay 70 DM per pollution unit, where one pollution unit can comprise, 25 kg N or 3 kg P. Similarly, in the Netherlands, dischargers must pay 45.50 Dfl per pollution unit (p.u.) where 1 p.u.

= (Q (m^3) /136) x (chemical oxygen demand +4.57 Kjeldahl Nitrogen), where Q is the discharge flow in $m^3\,day^{-1}$, and chemical oxygen demand and Kjeldahl Nitrogen are in $mg\,l^{-1}$.

The revenue raised from these schemes is used in part to finance abatement measures through loans and grants. Both countries consider the charging scheme to be an important incentive to reduce nutrient loads in discharges.

In France, the water agencies charge urban and industrial discharges, basing the fee on the daily load of the maximum activity month. The monetary value of the fee is determined by the agency basin committee, according to the abatement programme. In 1995, the discharge of one kg of phosphorus was charged in the range of 286 to 940 FF. For nitrogen (only organic

Table 9.4 **Industries identified as important with regard to direct discharges of nutrients in the North Sea countries. Source: OSPAR (1995)**

- fertiliser and feedstuff production (N+P)
- pulp and paper (N),
- oil refineries (N),
- chemical industry, particularly ammonia production (N),
- pharmaceuticals (N+P),
- dairies (N+P),
- manufacture of leather, gelatine and bone products (N)
- fish and meat processing (N+P),
- pesticide production (N+P),
- petrochemical industries (N),
- phosphoric acid production (P),
- manufacture of sodium carbonate,
- potato and sugar processing (N),
- breweries (P).

and reduced nitrogen were charged, oxidised nitrogen has been charged since 1997) the discharge of one kg of nitrogen is charged in the range of 180 to 350 FF. In 1994, the total budget of the six water agencies was just over 10 billion FF, accounting for all revenues: organic matter, toxics, water abstractions, etc.

In Poland statutory regulations concern charges for standard exceedances, for example, 0.85 zloty per weight unit. A weight unit for nitrogen equates to 1 kg and for phosphorus 0.32 kg.

9.3. Measures to reduce inputs from agriculture

The nutrient load from agriculture represents a high proportion of the total anthropogenic load of nutrients to water, directly via run-off and leaching, and indirectly via ammonia emissions to air. However, reducing nutrient inputs from agriculture has been, from both a technical and political point of view, a difficult issue to tackle.

Technically, policy development in this area is complicated by the different approaches to reduce nutrient losses, by the different ways of calculating the nutrient load, and particularly by the time lag between action and response, which is an important problem when assessing the effectiveness of measures and the resulting reductions in nutrient inputs. Furthermore, once established policy measures are often difficult to enforce.

Politically, agricultural polices aimed at efficient food production often conflict with environmental concerns. Despite widespread recognition of the need to integrate environmental policies and agricultural policies more closely at all levels, it has been difficult to influence the strong agricultural lobby.

A variety of policies have been used at international, national and regional levels:

- legislation (e.g. establishing protection zones);
- economic instruments (e.g. incentives to adopt alternative farming practices); and,
- information (e.g. education programmes, codes of good agricultural practice).

Within the EU, key policies are the Common Agricultural Policy (CAP) and the Nitrate Directive.

9.3.1. EC agricultural policy

The EC Council of Agriculture Ministers have taken some steps towards reforming the Common Agricultural Policy taking into account environmental needs by changing the pricing policy as well as introducing specific measures to help protect the environment, in particular:

- oil seed rape payment mechanisms have been reformed to substantially reduce the economic optimum fertiliser application rate (oil seed rape was one crop which was associated with relatively high nutrient application rates under the previous economic optimum);
- a set-aside programme has been introduced to limit production of cereals and other crops forcing farmers to introduce fallow in their crop rotation.

Both policies will probably lead, directly or indirectly, to a decrease in the use of fertilisers, contributing to a reduction of nutrient surplus. However, proportional reductions in overall nitrogen loss might not be achieved, there might even be increases, in particular in the case of nitrate leaching from non-cultivated soils and the intensification of cultivated soils.

9.3.2. The Nitrate Directive

The objective of the Nitrate Directive (91/676/EEC) is to reduce or prevent the pollution of water caused by the application and storage of inorganic fertiliser and manure on farmland. It requires that Member States:

- identify areas vulnerable to pollution by nitrate (NVZs) (by December 1993);
- establish Action Programmes governing the time and rate of fertiliser and manure application, and conditions of storage of manure in vulnerable zones (to be reviewed every four years, initially for the period 1995-1999);

- implement monitoring programmes to assess the effectiveness of action programmes; and
- establish Codes of Good Agricultural Practice to be implemented by farmers on a voluntary basis in other areas (by December 1993).

As with most other directives, the impact of the Nitrate Directive will depend upon the interpretation of requirements by Member States, especially in interpretation of 'vulnerable' since this will affect the extent of the territory designated and subject to mandatory requirements (Table 9.5). Five Member States have designated the whole of their territory according to Article 3(5). This means that they are exempt from the obligation to identify specific vulnerable zone if they establish and apply action programmes throughout their national territory. The UK and Sweden have designated 69 and 5 NVZs, respectively, and at the other end of the spectrum, Ireland does not intend to designate any NVZs. In addition, the success of the Directive will depend upon the extent to which farmers co-operate since some of the rules will be difficult to enforce. In any case, the effects of the directive will not be clear until after the implementation in 1999.

A recent report on the implementation of the Directive (CEC, 1997) concluded that the status of its implementation in most Member States is unsatisfactory. This late implementation makes it impossible to assess the effectiveness or otherwise of the Directive. Although some Member States have made progress, many are already behind the implementation timetable. So far only five countries, Austria, Germany, Denmark, Luxembourg, and Sweden have submitted action programmes, required by December 1995, to the Commission.

9.3.3. PARCOM, HELCOM and MEDPOL

Other agreements making recommendations for measures to reduce nutrient losses from agriculture are PARCOM, HELCOM and MEDPOL (Box 9.1). Although sharing a common problem the approaches recommended reflect regional differences in the

Table 9.5 Designation [2] of vulnerable zones under Nitrate Directive. Source: CEC (1998)

Country	Area covered
Austria	Whole territory [3]
Belgium	No information received by the Commission
Denmark	Whole territory[3]
Finland	No official designation: expected that 11 inland and 3 marine vulnerable zones will be designated
France	46% of agricultural land [1]
Germany	Whole territory[3]
Greece	4 potential zones[1]
Ireland	No zones designated but designation exercise completed [1]
Italy	No information received by the Commission
Luxembourg	Whole territory[3]
The Netherlands	Whole territory[3]
Portugal	5 vulnerable zones[1]
Spain	No information received by the Commission
Sweden	5 vulnerable zones [1]
UK	69 vulnerable zones [1]

Notes:
1 Currently being considered by the Commission
2 Designation was required by December 1993
3 According to Article 3(5)

Box 9.1 PARCOM Recommendation 89/4 of 22 June – recommendations for agriculture:

- spreading of manure and chemical fertilisers should only take place in such a way and at such times that optimal use of nutrients by plants can be achieved, whilst ensuring minimum losses to the environment. Contracting parties should establish which time periods are appropriate in specific regions, and which limitations should apply.
- harmony should be established between the number of livestock and the area available for safe spreading of manure or the availability of environmentally acceptable alternatives.
- the capacity of manure storage facilities should exceed that required for the storage of ma nure throughout the longest period during which the application of manure is restricted.
- in order to reduce losses of nutrients through erosion and run-off, the application of manure and fertilisers should be more strictly regulated or possibly prohibited, for example in areas in close proximity to watercourses.
- the planting of winter cereal or catch crops should be encouraged
- measures should be taken to reduce atmospheric emissions of ammonia when handling manure by setting rules for storage (e.g. cover) and for ploughing-in as soon as possible after spreading on arable soil.
- the establishment of mandatory and chemical fertiliser handling plans for individual farms should be encouraged.

HELCOM Recommendations for reducing nutrients discharges (and pollution) from agriculture (7/2, 9/3, 13/7 to 13/11, 14/4 and 19/6). Recommendation 19/6 of 26 March 1998 aims at an amendment (expansion) of Annex III to the Helsinki Convention. It requires:

- measures to reduce the pollution from agricultural activities should take into account Best Environment Practice (BEP) and Best Available Technology (BAT).
- balance between number of animals and amount of land available for spreading manure, considering amount of nitrogen and phosphorus in manure and the crops requirements for nutrients.
- manure storage: losses prevented; sufficient storage capacity to ensure manure only spread when plants can utilise nutrients; minimum 6 months storage capacity; and efficient reduction of ammonia emissions.
- storage and treatment of agricultural waste water and silage effluents.
- application of organic manures that minimises loss of nutrients, with no spreading on frozen, water saturated or snow covered soils; and organic manures incorporated as soon as possible after application on bare soils; and with periods of ‘no application’ defined.
- application rates for nutrients not to exceed crops nutrient requirements; and national guidelines required taking into account factors such as soil condition, type and slope, climatic conditions irrigation, land use and agricultural practices.
- winter crop cover to reduce the loss of plant nutrients.
- establishment of surface water buffer zones and ground water protection zones, and maintenance and restoration of nutrient reduction areas such as wetlands.
- environmental permits for farms with livestock production above certain size.
- environmental monitoring to assess effects of measures and impacts of agricultural sector.
- promotion of systems for education, information and advice on environmental issues.

MEDPOL Recommendations for reducing nutrient inputs from agriculture and livestock:

- Rational use of fertilisers. Fertilisation should be reconciled with the nature of the soil and the nutritional requirement of the crops, adoption of ‘slow release’ synthetic fertilisers. Improvement of irrigation systems to minimise losses by wash-out and erosion.
- Erosional losses can also be minimised by different methods of strip farming. In hilly regions is ploughing and cultivation across to, not with the slops of terrains.
- Livestock: to treat sewage and to encourage manure-spreading. The first is costly and the second is easy and widely practised but extensive spreading may carry the risk of contaminants washing into surface waters and percolating down to the aquifers.

source and nature of the main emissions. Thus, whilst the three regions seek to control fertiliser applications, the Baltic and North Sea countries, with highly intensive agricultural systems and high livestock densities, place emphasis on the control of manure storage and application whereas controls on erosion are recommended in the Mediterranean area. The common element of the programmes is the need for improved information on nutrient levels and inputs to identify the most significant nutrient sources enabling policies appropriate to the local situation to be derived.

Generally there has been less success in reducing inputs from agriculture than from point sources. In the PARCOM area, the estimated reductions for the period 1985 to 1995 vary between 0% in the Netherlands and 40% in Norway for the phosphorus inputs; between 10% in France and 28% in Sweden for nitrogen inputs. Seen in combination with the relative importance of agriculture, this failure to reduce inputs from the agriculture sector, particularly for inputs of nitrogen, has resulted in overall reduction targets being missed by the Parties to PARCOM and the Rhine Commission. PARCOM reported that agricultural losses were more difficult to influence than anticipated, despite recommendations for a comprehensive suite of measures, because the measures adopted were inadequate or inadequately implemented in terms of:

- inadequate information and advisory services to farmers;
- ineffective control and enforcement of adopted measures;
- inadequate use and / or lack of financial instruments.

HELCOM also reported substantial differences between Contracting Parties with regard to achieving the goal of 50% reduction in nutrient discharges to the Baltic Sea by 1995 as required by the Helsinki Convention Ministerial Declaration of 1988 (HELCOM, 1996). Contracting Parties within the EU had not achieved the 50% target by 1995 and the need for additional measures, in particular with regards to nitrogen from agriculture, was identified. In some of the non-EU Contracting Parties the decrease in nutrient loading has been reached mainly due to decreased use of fertilisers and decreased agricultural production caused by structural changes and economic difficulties. However, economic recovery in the latter countries might again lead to an increase in the agricultural run-off.

Ammonia emissions, which originate mainly from agriculture, represent about 30-50% of the total emissions of nitrogen to the atmosphere (OSPAR, 1995). PARCOM recommendations included measures to reduce ammonia emissions which have resulted in mixed success. Thus whilst in the Netherlands a 31% reduction was achieved for the period 1985-1995, in Belgium ammonia emissions rose by 19%.

In 1992 PARCOM published a report reviewing the implementation of this recommendation (OSPAR, 1992). A wide range of approaches were used and found to be effective. Most of the Contracting Parties (5 or more from 7) found the following activities to be 'very effective' in reducing nutrient inputs:

- spreading manure and sludge to match nutrients per hectare on a farm basis, taking account of ground conditions;
- storing of manure and for silage, impermeable storage facilities, requirements for storage capacity, covering storage facilities; and,
- use of fertiliser, fertiliser advice programmes.

Similarly some measures were found to be not very effective, these included incorporation of straw into soil, organic farming, restoration of rivers, and treatment of stored manure.

Box 9.2 A successful advisory campaign in the UK

In the UK, an advisory campaign resulted in a reduction of the number of farmers applying nitrogen to wheat in autumn and winter (56% to only 10%), of fertiliser application rates (by 14% for nitrogen and 17% for phosphorus) and of nutrient surpluses (11% for nitrogen and 13% for phosphorus).

9.3.4. *Fertiliser taxes*

One much discussed approach is the use of fertiliser taxes such as those applied in Sweden. Although not a popular measure, a recent report from the Dutch Agricultural Economics Institute estimates that by doubling the price of fertiliser, nutrient inputs to the environment from agriculture could decrease by a quarter, depending on local circumstances (ENDS, 1997).

9.3.5. *Less intensive farming*

Many north-western countries (e.g. Denmark) have adopted complex strategies for controlling nutrient inputs from agriculture and have had some success in reducing inputs. In many other countries, where farming is less intensive, the environmental damage from agriculture is low compared to western European countries. For example, in Poland and Romania the agricultural sector is now centred around small private farms using mostly traditional practices and comparatively low levels of chemicals. As in many other CEE countries, the economic reforms and recession have meant that, especially since the early 1990s, pressures from agriculture have decreased. This has resulted in a reduction in agricultural activity and a significant drop in the use of agricultural chemicals. Fertiliser use in Poland fell by nearly 70% between 1989 and 1992 (Environmental Performance Reviews, Poland 1995). In Romania chemical fertiliser use has been reduced by more than 50% (e.g. nitrogen fertilisers: 2035 Kt in 1989, to 905 Kt in 1994). The privatisation of, and decrease in, agricultural production are the key factors of this change. Similarly in some of the Mediterranean countries, such as Portugal, the use of commercial fertiliser is much lower than in the rest of the European Union. Ireland has recently published its National Sustainable Development Strategy which targets a reduction of 10% per year in artificial phosphorus fertiliser usage over the next five years. This will reduce soil phosphorus levels down to recommended levels where at present they are excessive to crop requirements (Department of the Environment, Ireland, 1997).

9.4. Measures to reduce inputs from products

Probably the single most significant policy to control nutrient inputs from products has been the limitation of the use of phosphorus in detergents. Several international measures have been introduced:

- Council Regulation 880/92 encourages lower use of phosphate in detergents as a criterion for better ecological detergents, part of the eco-label award scheme;

- PARCOM (Recommendation 89/40) encouraged the development and marketing of phosphate-free detergents where it could be demonstrated that they were environmentally acceptable; and more recently;

- the Danube Action Programme recommends that a ban on phosphate containing detergents should be implemented by all contracting parties by 2005.

As a result of the PARCOM recommendation, there has been a significant increase in the use of phosphate free detergents, and efforts to reduce the content of phosphorus in detergents are being made by most North Sea countries either by legislation or by agreement with the industry.

In Belgium, Germany, the Netherlands and Norway there is practically no use of phosphorus in detergents, whereas in Denmark, France, Sweden and the United Kingdom the use of phosphate-containing detergents is declining. In Switzerland the use of phos-

Box 9.3 Examples of approaches to limit phosphate in detergents

- Belgium – voluntary agreement 18/9/88 between the Belgian Association of Soap Manufacturers and the Government – 100% phosphate free detergent by 1995;
- Denmark – recommendation for phosphorus-free detergents in areas where urban waste water treatment plants do not have phosphate removal, target for 50% phosphorus-free detergents by 1992;
- Switzerland – strict ban on phosphate detergents from 1986.

phates in detergents has been prohibited since 1986.

The decision to ban phosphates in detergents in Switzerland was part of a strategy to reduce phosphorus inputs to fresh waters along with phosphorus removal from urban waste water treatment plants. The removal of phosphorus from the urban waste water treatment plants around Lake Geneva began in 1967, initially as a 'recommendation', but then as a requirement by law in 1975. During this period total phosphorus concentrations had continued to rise in the lake. There followed a step-wise reduction in phosphate usage in detergents starting in 1981, and leading to a total ban in 1986. This simultaneous ban on phosphate in detergents and chemical phosphate removal in urban waste water treatment plants has resulted in a clear decline in phosphorus concentrations in Lake Geneva from 1979 (Figure 9.3).

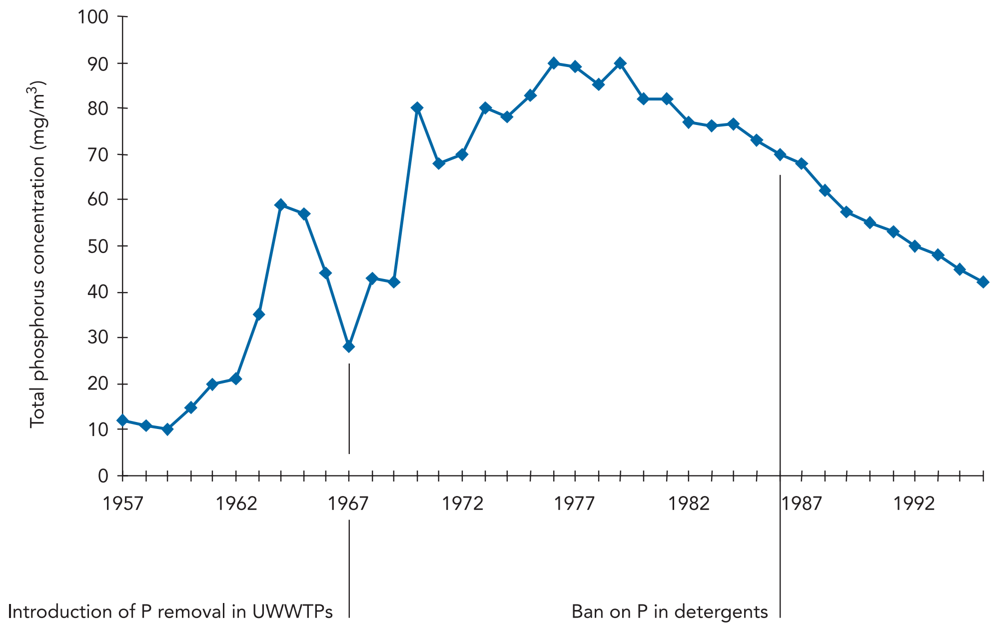
9.5. Measures to reduce inputs from atmospheric sources

There are a number of international policies which set overall target reductions for NOx emissions as a whole, for example, the Sofia Protocol 1988 which requires Parties to stabilise NOx emissions at the 1987 level by 1994. In addition, Air Quality Standards have been set at an EU-level for nitrogen dioxide as follows:

- limit values of 200 µg NO_2 m^{-3} (98% tile) for the mean values per hour; and,
- guide values of 50 µg NO_2 m^{-3} (50% tile) and 135 µg NO_2 m^{-3} (98% tile).

Proposals to tighten these standards are being discussed. Air Quality Standards have also been developed nationally, and are, in some cases, more stringent than those required by the EU.

Figure 9.3 **Phosphorus levels in Lake Geneva after phosphorus removal at sewage treatment plants and the phosphate ban in detergents. Source: EAWAG (1997)**



Other polices have been developed to tackle key emissions from point and diffuse sources, primarily NO_x emissions from industry, combustion plants and traffic, and also NH_3 emissions from agriculture (see Section 3.7). These have been adopted with mixed results. In the PARCOM area, for example, estimated changes for the period 1985 to 1995 vary between a 6% increase in nitrogen oxide emissions in Belgium and a 15% decrease in nitrogen emissions in Sweden. In Italy, emissions of NOx rose by 35% between 1985 and 1995. UNECE/EMEP (EEA, 1998) estimated that emissions of NOx decreased by 15% in Europe between 1990 and 1995. In EEA countries the reduction was 8% in this period.

By contrast, NOx emissions in much of CEE decreased considerably more, approximately 30% between 1990 and 1995, most probably because of a decrease in industrial activity. Traffic emissions are, however, beginning to rise in some of these countries.

9.5.1. Point source emissions

Approaches to reduce emissions from industry, principally from combustion plants include:

- **Target reductions**, for example, the EU Directive on Large Combustion Plants (88/609/EEC) requires Member States to reduce NOx emissions from existing large combustion plants, on average by 30% between 1980 and 2003 (28% by 1998);

- **Emission limits**, based on best available techniques have been developed in most EEA countries. Directive 84/360/EEC on combating air pollution from industrial plants required regulation of emissions of NOx for a wide range of industries. Emission limits for large combustion plants in the EU have been harmonised in Directive (88/609/EEC), and a revised Directive for Municipal Waste Incineration is also likely to include standards for NOx. In the future most industrial emissions in the EU will be subject to the Council Directive on Integrated Pollution Prevention and Control (96/61/EEC);

- **Voluntary agreements** have been made nationally by many countries in order to reach the target reductions established internationally. For example, Belgium has introduced a voluntary agreement with the electricity producers to reduce NOx emissions by 40% by 1998;

- **Environmental charges**. Sweden, for example, has introduced a charge of 40 SEK per kg NOx as of 1992 for all emissions from industry.

9.5.2. Emissions from vehicles

Directive 70/220/EEC and its amendments (91/441, 93/59, 94/12), and Directive 88/77/EEC impose severe limitations on NOx levels in exhaust emissions from respectively motor vehicles and diesel engines. The Commission predicts, from modelling, that the Directives will result in NOx emissions from on-road vehicles in the EU (excluding Austria, Finland and Sweden) being reduced from 5.8 Mtonnes in 1990, to 4.4 Mtonnes in 2000, and 3.2 Mtonnes in 2010.

HELCOM Recommendation 17/1 concerning reduction of emissions from the transport sector affecting the Baltic Sea recommends that Contracting Parties should incorporate a number of principles in national strategies and programmes for transport, including the introduction of best available technology for vehicle and fuels in all transport modes. In particular regulatory and technical measures are required by the year 2000, to gradually harmonise the emission standards for passenger cars, light duty vehicles and heavy duty vehicles, with a view to introducing at least the requirements specified in UNECE Regulations and EU-Directives. Though adopted in 1996, the Recommendation has not yet been implemented by Contracting Parties.

Although the NOx emissions per vehicle are being reduced by tightening emission standards, some countries are reporting that this is partly offset by an increase in NOx emissions as a result of increased volumes of traffic. Thus, for example, the UK estimates a 30% increase in NOx emissions from traffic over the period 1985 to 1990, and a subsequent decrease by 15%

between 1990 and 1995. Many countries have introduced, or are introducing measures, to try to restrict the numbers of vehicles and their emissions. Examples are given in Box 9.4.

Box 9.4

- Reduced speed limits, for example, in Sweden and Switzerland from 110 km hour^{-1} to 90 km hour^{-1};
- Adoption of Transport code, for example, in Austria, measures to reduce pollution from the transport sector have focused on reducing emissions from individual vehicles. The rail infrastructure is also being expanded and improved in urban and ecologically sensitive areas;
- Urban transport plans, for example, in France for cities > 100 000 inhabitants, and in the Netherlands.;
- Voluntary agreements, for example, in Italy the Ministry of the Environment and the automobile industry have signed voluntary agreements on low-emission public transport.

9.6. Measures to safeguard terrestrial habitats

Measures to safeguard habitats have focused on preventing inputs by controlling developments. The level of control is determined by the sensitivity of the areas. All areas of the EU are protected under the Environmental Impact Assessment Directive but additional measures exist in areas requiring special protection.

9.6.1. Environmental impact assessment

The Environmental Impact Assessment Directive (85/337/EEC) requires that developers carry out an environmental impact assessment as part of a planning application to undertake certain new developments. The assessment must include potential impacts and their significance on the aquatic and terrestrial ecosystems, on air quality and climate, and on humans, including indirect and secondary effects. Thus potential changes in nutrient status would have to be considered for many developments and any potential effects mitigated or reduced.

9.6.2. Special sites

There are several international measures requiring national governments to identify and protect special sites. Amongst other things such sites must be protected from impacts due to eutrophication. The EC Habitats Directive (on the conservation of natural habitats and wild flora and fauna 92/43/EEC) requires that Member States designate a series of Special Areas of Conservation (SACs) for habitats and species of ‘Community interest’. After agreement with the European Commission, national lists will be combined with Special Protection Areas designated under the EC ‘Birds Directive’ (79/409/EEC) to form a European network of protected sites called Natura 2000. Many proposed sites are aquatic or semi-aquatic, and vulnerable to nutrient enrichment. Measures will be required to protect such sites from nutrient inputs, if they are likely to damage them.

The UN Biodiversity Convention, drawn up at the 1992 Earth Summit in Rio de Janeiro, commits signatories to developing a national strategy for the conservation and sustainable use of biodiversity. Action plans contain objectives to conserve, and where possible, enhance: populations and natural ranges of native species; the quality and range of habitats; and, also the biodiversity of habitats where this has been diminished. This commitment provides impetus for the protection and enhancement of aquatic habitats, including measures to control eutrophication.

The Ramsar Convention signed in 1971 was drawn up to stop the progressive decline of wetlands through activities such as development and agricultural improvement. Signatories are required to list all wetlands of international importance, and, within these sites, to promote conservation, promote the establishment of nature reserves, and inform the Secretariat of the Convention of changes in status. Where such sites are impacted or threatened by nutrient enrichment, the obligation to take controlling action is inevitably increased.

9.6.3. Buffer zones and water protection zones

Buffer zones are often used to protect water resources against contamination from both point and diffuse pollution. Certain polluting activities are usually prohibited within such zones, for example, application of fertiliser or ploughing, or construction. In a similar way, water protection zones are sometimes used, usually to protect groundwater resources. Many areas in Germany take this approach and limit agricultural activities depending on the distance from the groundwater source. There is evidence of the success of such approach in reducing nitrate levels but there are considerable delays between action and effect.

9.7. Measures to remediate high nutrient levels

Once present in rivers, there is no satisfactory method of reducing excessive nutrient levels. The same can be said for estuarine and tidal waters, but in lakes a number of options are available. Box 9.5 (data obtained from Moss *et al.* (1996) and Sas (1989)) illustrates examples where some of these approaches have been used.

A number of chemicals can be added to the lakes (either to the lake surface via a sprinkler system, or dosed to the influent water stream) to flocculate phosphorus. Ferric sulphate, ferric chloride and aluminium sulphate are the most widely used flocculants. The reductions in phosphorus levels that can be achieved are largely dependent on the SRP:TP ratio, since it is primarily dissolved, loosely bound phosphate that is precipitated out of solution. SRP levels can be reduced to <10 mg l^{-1}, the levels to which phosphorus needs to be reduced to genuinely limit algal growth, but dosing needs to be undertaken continuously, and the lake sediment becomes a sink for potentially toxic heavy metals.

Natural sedimentation of phosphorus by calcium carbonate (calcite) has been taken a stage further by mixing calcite-rich and phosphorus-poor material from deep layers of sediment into the water column (Klapper, 1991). This provides a calcium-rich cover that inhibits release of phosphorus from the sediment below, as well as helping to bind phosphorus from the water above. Calcium appears to offer substantial advantages in terms of much lower toxicity when compared to the heavy metal alternatives, although the bonds formed with phosphorus are not as strong as those formed with iron and aluminium.

Many lake restoration schemes involve sediment removal or 'inactivation' to reduce phosphorus release from highly nutrient-enriched sediments. These methods rely either on removing the most phosphorus-rich surface layers of sediment to expose relatively nutrient-poor layers underneath, or placing a layer of material on top of the sediment. The latter provides a physical barrier to the release of dissolved phosphorus from interstitial water into the overlying water. Alternatively, chemical inactivation may be used by artificially increasing the sediment phosphorus adsorption capacity. Again, iron, aluminium and calcium are the most popular binding agents for this purpose, although it should be appreciated that calcium salts may serve a dual or triple role. When calcium nitrate is injected into the sediment (the 'Riplox' process), the calcium helps to bind phosphorus while the nitrate acts as an oxidising agent, reducing the sediment oxygen demand from the overlying water. Calcium hydroxide (slaked lime) is often added shortly after the addition of ferric chloride to sediment to help neutralise the acidifying effects of the iron salt (Klapper, 1991; Moss *et al.*, 1996).

If sediment removal or inactivation is not undertaken, but external nutrient loading to a lake is substantially reduced, the sediment will act as a net source of phosphorus, typically for up to 5 years (Sas, 1989), and in some case for longer (e.g. Lake Finjasjön, see Box 9.5).

Artificial flocculation is also used to remove phosphorus from the inputs to lakes. This can be done by circulating water from a pre-reservoir (which feeds a larger downstream reservoir) through a phosphorus stripping plant and back into the pre-reservoir, as at Wahnbach Talsperre in Germany (Bernhardt *et al.*, 1985; Sas, 1989). Alternatively, in the case of pump storage reservoirs, phosphorus can be stripped from the water after it has been abstracted from the river but before it is pumped into the reservoir.

The construction of pre-reservoirs without artificial phosphorus stripping may also be used as a method of nutrient removal – of particulate phosphorus by sedimentation and nitrogen by denitrification. However, the morphology required for optimal removal of nitrogen and phosphorus is very different – shallower reservoirs tend to favour sedimentation of phosphorus (except during high inflows), while deep, stratified reservoirs favour nitrogen removal by

Box 9.5 Control of eutrophication – Four European case studies. Source: Compiled by ETC/IW

Lake statistics	Problems	Actions	Results	Lessons
Lake Zwemlust, the Netherlands surface area = 1.5 ha, mean depth = 1.5 m, maximum depth = 2.5 m	In 1968 herbicide application to control macrophytes resulted in an almost immediate switch to phytoplankton dominance. Secchi depth decreased to 0.1-0.3 m.	Biomanipulation only; no nutrient control. The fish population was removed and the lake restocked with pike and rudd. Daphnia and yellow water lily were introduced. Bundles of willow twigs were fixed to the bottom of the lake to act as spawning sites for pike and refuges for zooplankton.	Successful in the short term, with Secchi depth increasing to 2.5 m. Summer chlorophyll maxima fell from >200 mg l^{-1} to <10 mg l^{-1}. Within 1.5 years, submerged plants dominated the lake and continued to do so for a further 3 years. However, a return to phytoplankton-dominance occurred after this time.	Several factors contributed to failure: uncontrolled nutrient loadings (N rather than P was the limiting nutrient); grazing on macrophytes by coots – simultaneous restoration of adjacent waterbodies (for alternative food supplies) would probably have helped; pike and rudd may have been introduced too early – delaying their introduction until after the plants had become established may have helped.
Lake Væng, Denmark surface area = 16 ha, mean depth = 1.2 m, maximum depth = 1.8 m	The lake was very eutrophic (Secchi depth below 0.8 m), and only minor improvement was seen after heavy loading with domestic sewage was diverted from the lake in 1982.	After sewage diversion (the phosphorus loading of the lake was reduced by 63%) biomanipulation took place in 1986-88 and 50% of the planktivorous fish were removed.	The Secchi depth increased from below 0.8 m to above 1.5 m. Chlorophyll concentrations were reduced threefold, and submerged macrophytes reappeared in 1989 covering more than half of the lake. The fish population became dominated by piscivorous species allowing planktivorous species to be maintained at a low level.	Phosphorus removal alone proved insufficient to reverse the adverse effects of eutrophication in the lake but combined with low-cost fish removal the lake returned to a much cleaner state than before biomanipulation was carried out.
Finjasjön, Sweden surface area = 11 km^2, mean depth = 3 m, maximum depth = 13 m	Dominance by blue green algal blooms – *Gloeotrichia echinulata* in the 1940s and *Microcystis aeruginosa* in the 1960s and 70s.	In 1977, phosphorus stripping was introduced at a UWWTP (serving the town of Hässleholm) which discharged into the lake. In 1987, dredging was started, but stopped 5 years later when only 25% of the lake had been dredged (at a cost of 5 million EURO). Between 1992 and 1994, some 430 tonnes of fish – mainly bream and roach – were removed (about 85% of the fish in the lake), at a cost of 0.63 million EURO. Narrow buffer zones (5 m wide) were established and a 30 ha artificial wetland was constructed to treat the UWWTP effluent. This cost 0.75 million EURO to build, with annual running cost of 0.125 million EURO.	Phosphorus stripping at the UWWTP lowered the annual external phosphorus load from 65 to 5 tonnes, but massive internal loading from the nutrient-rich sediment ensured that phosphorus levels in the lake were still 100-500 mg l^{-1}. Sediment dredging was discontinued because the dredged areas continued to release phosphorus at similar rates to undredged areas. Effluent from the artificial wetland contains only 0.1 mg P l^{-1} and 15 mg N l^{-1}. In 1994 and 1995 Secchi depth showed signs of increasing and chlorophyll-a levels show signs of decreasing (to a mean level of about 25 mg l^{-1}).	The enormous costs involved. The cost of dredging would have been better spent on an earlier fishery management programme. Dredging has clearly been of benefit in some lake rehabilitation studies, but the importance of chemical analysis of sediment core profiles before undertaking dredging is emphasised.
Lake Zurich Lower Basin, Switzerland surface area = 65.1 km^2, mean depth = 51 m, maximum depth = 136 m	Improvements to waste water treatment facilities in the catchment initially concentrated on reducing organic loadings to natural waters. Despite Secchi depth levels of about 3-11 m, nutrient enrichment was then considered a target for action.	Since the 1970s, phosphorus stripping has been progressively installed at UWWTPs in the catchment. A national ban on phosphates in detergent was introduced in 1986.	Little change has been recorded in Secchi depth or phytoplankton levels, despite substantial decreases in phosphorus levels. Primary productivity has dropped, but the reappearance of blue-green algae has led to a decrease in zooplankton.	Reducing the in-lake phosphorus concentrations by a factor of about three (from 1974 levels of 15-100 mg P l^{-1}) has had relatively little effect on algal standing crops.

denitrification. However, denitrification is not known to have been employed on a practical basis, only in experimental situations (See Klapper, 1990).

In lakes where the surface waters rapidly become nutrient-depleted and stratification is maintained during the growing season, abstraction of nutrient-rich hypolimnetic water (released into the lake outlet) has been used with some success. Here, the nutrients are not inactivated, but artificial short-circuiting is induced within the lake, so that the nutrients are 'discharged' to river, i.e. to an aquatic environment where the risk of them causing eutrophication effects is reduced. This was used, for example, as part of the restoration programme for Schlachtensee (Sas, 1989). This concept is similar to the direct diversion of nutrient-rich inputs from lakes to rivers or to the sea, such as in Little Mere, UK. Here sewage effluent was the major nutrient input to the lake, until the works was closed in 1991, and the raw sewage was treated at another works (Moss *et al.*, 1996).

In contrast to the previous paragraph, lakes and reservoirs may be artificially mixed as a method of eutrophication control. Numerous methods are used to do this, including fixed depth jetting (relatively common in pump storage reservoirs); variable depth mixing (whereby a mixing propeller is raised and lowered throughout the water column); and air diffuser mixing (the most widely used and, usually, the cheapest of the three methods).

Artificial mixing serves two purposes: lower waters remain oxygen-rich, thereby inhibiting the release of phosphorus from sediments (albeit at the expense of lower denitrification rates), and phytoplankton are circulated out of surface waters into deeper waters where light levels are too low for the algae to photosynthesise effectively. However, reductions in phytoplankton standing crop can only be expected where the depth of the water is at least twice as deep as the euphotic zone (usually assumed to be the depth at which light radiation (400-700 nm) is 1% of the level at the surface of the water, or 2.5-3.0 times Secchi depth).

In many cases, specially when phosphorus accumulation near the bottom is high, artificial mixing can actually increase the algal standing crop, rather than decrease it. Hypolimnetic aeration without breaking the thermocline, that is without destratification, can be used where this is expected (or has been shown) to occur. This keeps the hypolimnion well oxygenated, so the release of phosphorus, ammonia, iron, manganese, etc. from sediment is minimised, but has little effect on the phytoplankton community.

Biomanipulation (the artificial management of fish communities to control algal standing crops) has received an enormous amount of attention over the past decade or two. In western Europe the focus of attention has been to decrease the numbers of those fish which feed on zooplankton, or to provide habitats where the zooplankton are less likely to be preyed upon. Less predation on zooplankton results in greater grazing pressure on phytoplankton. However, in eastern Europe, much experimental work has been undertaken on the use of Silver Carp (*Hypophthalmichthys molitrix*) to control algal levels, particularly during the 1960s-1980s. This is an introduced species (Chinese in origin) which grazes directly on phyto- and zoo-plankton, but for which a wide range of stocking densities have been cited to achieve algal control (see Parr, 1992). Although used as a method of phytoplankton control in China and Israel, Silver Carp no longer appears to be used on a practical basis for algal control in Europe. The introduction of Grass Carp (*Ctenopharyngodon idella*) has been successfully used to control macrophytes in several countries, for example, in large canals in the USA.

Although biomanipulation is generally considered essential to the success of shallow lake restoration schemes, such programmes may require extensive (and expensive) monitoring. Fishery status needs to be carefully managed, since it is not only the numbers of individual species that are important, but also the number of each species in each age class. Other factors are also important, for example, de Bernardi and Guissani's (1990) review suggests that blue-green algae are a poor food source for zooplankton, which questions the utility of biomanipulation programmes in lakes dominated by blue-green algal blooms. Nevertheless, a number of useful reviews and 'user's guides' to biomanipulation have been produced, which learn from the mistakes made in previous studies and offer practical advice for future programmes (e.g. Moss *et al.*, 1996).

9.8. Summary

At international, national and local levels nutrient enrichment has been tackled through a wide range of approaches. Table 9.6 shows the scope of measures adopted across the EEA area.

Much success has been achieved in the battle to reduce nutrient inputs, particularly for phosphorus. In the PARCOM area for example, Belgium, Denmark, Germany, the Netherlands, Norway, Sweden and Switzerland, all expect to reduce phosphorus inputs of the order of 50% by the end of 1995 into the nationally defined eutrophication problem areas bordering the North Sea. France expects to achieve a 25% reduction in phosphorus inputs into areas where these inputs are likely, directly or indirectly, to cause pollution by the end of 1995. Nitrogen inputs are proving more difficult to tackle since diffuse pollution represents, in general, a larger proportion of the load. Most countries have only been able to achieve between 20 and 30% reductions of nitrogen inputs into problem areas.

Notes to table 9.6: Not all countries completed and returned the questionnaires requesting input to this assessment report. Whilst the returned questionnaires have been supplemented from a large number of sources, the information available varied in terms of age, and detail and large information gaps remain. Therefore the table provides only an overview of the measures taken providing no detail as to the extent or method of application (for example whether an approach was voluntary or compulsory). In addition, some of the boxes shown as blanks may result from a lack of information rather than lack of policy application.

The scope of measures applied across the EEA area to fight nutrient enrichment (■ = 'yes', white = 'no' or 'not known'). Source: Compiled by ETC/IW from questionnaire returns **Table 9.6**

Countries Approach	A	B	DK	D	E	FIN	F	GR	IRL	IS	I	NL	N	P	S	UK	BG	EE	LT	PL	R	SLK	SLO	CH
STRATEGIC																								
Goals established	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■		■	■	■
Target reductions of nutrients	■	■	■	■		■	■		■	■	■	■	■	■	■	■	■	■	■	■				■
EQS Water	■		■	■	■		■		■			■	■		■	■	■		■	■		■		■
NOx standards	■	■		■	■		■				■	■	■	■	■	■	■			■				■
POINT SOURCES																								
Point source reductions to water	■	■	■	■	■	■	■	■	■			■	■		■	■	■	■	■	■		■	■	■
UWWT Tertiary treatment		■	■	■	■	■	■		■			■	■		■	■			■	■		■		■
Industry – BAT	■		■	■												■							■	
Reduction policy	■	■	■	■	■		■		■			■	■		■	■	■					■	■	■
AGRICULTURE																								
Controls on agriculture	■	■	■	■	■	■	■	■	■			■	■		■	■		■	■	■	■		■	■
Manure restrictions	■	■	■	■	■		■	■	■		■	■	■		■	■								■
Manure storage	■	■	■	■			■	■	■			■	■		■	■								■
Silage storage			■	■				■	■			■	■		■	■								■
Fertiliser restrictions	■	■	■	■			■	■	■			■	■		■	■		■			■		■	■
Cultivation	■			■			■		■			■	■		■	■								■
Financial instruments	■	■		■			■		■			■	■		■	■		■	■		■			■
PRODUCTS																								
Eco-labelling	■		■	■	■		■				■	■		■	■									■
Detergents		■	■	■			■					■	■		■									■
ATMOSPHERIC INPUTS																								
Atmospheric sources	■	■	■	■	■						■	■	■	■	■	■				■		■		■
BAT Industry	■		■	■											■	■				■				
Voluntary agreements		■										■												
Combustion installations	■	■	■	■							■	■			■	■						■		■
Traffic	■	■	■	■	■					■	■	■	■		■	■				■		■		■
TERRESTRIAL HABITATS																								
Special protection areas	■	■	■	■	■	■	■	■			■	■		■	■	■								

Countries:

A	Austria	FIN	Finland	N	Norway
B	Belgium	F	France	PL	Poland
BG	Bulgaria	GR	Greece	P	Portugal
CH	Switzerland	IRL	Ireland	R	Romania
D	Germany	IS	Iceland	S	Sweden
DK	Denmark	I	Italy	SLO	Slovenia
EE	Estonia	LT	Lithuania	SLK	Slovak Republic
E	Spain	NL	The Netherlands	UK	United Kingdom

10. Conclusions

10.1. Assessment problems

In 1997/98, the European Topic Centre on Inland Waters, in collaboration with the European Topic Centres on Nature Conservation and on the Marine and Coastal Environment, carried out an assessment of excess nutrients in European ecosystems. The main conclusions are summarised in this chapter.

For each ecosystem under consideration, specific data concerning environmental, chemical and biological conditions are required in order to assess nutrient impacts. All the necessary datasets are not available at a European or national level, and do not even exist at all in some countries. Despite the important efforts made by National Focal Points during data collection for the Europe's Environment: the Second Assessment report and for this report, it was only possible to obtain a small fraction of the existing data, and this fraction is insufficient to produce a full assessment. Even for regions where monitoring networks are in place, they do not always include the numbers of points, parameters and time series necessary for establishing definitive assessments and trends.

Another obstacle to carrying out a satisfactory assessment of the issue concerns the large number of data errors, due to the lack of proper data description procedures. Unlike the local level where these difficulties can be overcome on a point by point basis, at European level it is impossible to rectify such errors because of the large numbers of data sources involved.

Apart from the limited availability of data concerning the ecosystems themselves, it has become clear that information concerning nutrient emissions pose even greater problems. It has not been possible to evaluate accurately emissions (atmospheric and/or to water) on a national basis or even at a catchment level, which should in theory be the basic assessment unit for relating pressures to state in aquatic environments. Work is, however, underway within the EEA and other organisations on developing European emission inventories.

10.2. State and trends in ecosystems

The aquatic and terrestrial ecosystems considered in this assessment report are all affected by nutrient excesses. The significance of the effect varies according to geographic region.

10.2.1. Still waters

Eutrophication is a major issue in still water environments (natural lakes and artificial reservoirs). However, despite the environmental and economic importance of such water bodies, and even after decades of scientific research, there are very few monitoring programmes in existence.

The information collected for this assessment report suggests that still waters, particularly lowland reservoirs, appear to be seriously affected by eutrophication. However, the overall impression of eutrophication problems being so widespread may possibly result from the fact that data are more often available for already impacted water bodies. Nevertheless, it is clear that many reservoirs serving essential uses such as public water supply and irrigation are amongst the most affected by eutrophication, since they are necessarily located within, or downstream of, areas of intense human activity.

It has not been possible to evaluate overall trends in reservoir water quality because of insufficient data, however an overall deterioration has been observed in several southern European countries. In natural lakes, it would appear that the significant efforts employed to reduce external phosphorus loads in the catchments of many lakes have resulted in improvements (decreases) in phosphorus concentrations in the lake. However, it is important to note that the effects of eutrophication, such as excessive algal growth and unfavourable oxygen balances, often continue in these lakes because of the large stock of accumulated nutrients.

10.2.2. Rivers

Rivers represent the main disposal site for many types of pollution. Although data concerning river water quality are generally more available than for other ecosystems, effort is still required to ensure that all the data are relevant and representative.

Rivers can be affected by the entire range of possible nuisances caused by nutrients. The majority of river stations studied in this assessment report was found to be impacted to some extent by these perturbations.

The analyses carried out for this assessment report have been able to confirm that excess ammonium is present at many river stations. Furthermore, the assessment of likely ammonia concentrations (the most toxic form of ammonium) suggests that this type of pollution is much more widespread than previously believed.

Excessive levels of nitrate, observed in many previous studies, also represents a widespread degradation of river water. Locally, nitrate concentrations may prevent human uses of water.

However, the information presented in this report shows that it is phosphorus pollution, which is not only the most widespread in geographic terms, but also the most significant compared to likely natural concentrations. Under suitable conditions, excessive phosphorus results in the development of large quantities of seasonal plant growth. The resulting excess in plant growth causes other types of impact; perturbed oxygen and pH cycles, organic pollution, and in cases of extreme ecological dis-equilibrium, massive growth of toxic algae.

These dysfunctions were used, along with the few chlorophyll data available, to assess the intensity and extent of river eutrophication. Although the results cannot be considered to be fully representative, they suggest that over half of all monitoring stations, for which information was provided, may be affected.

A general improving trend was observed for nitrogen compounds, although in some cases the trend appears to represent more of a halt in degradation than any real improvement. For phosphorus, it would appear that the most polluted stations have improved whilst the cleanest rivers show significant deterioration.

10.2.3. Coastal marine waters

The coastal environment is the end receptacle for pollution from many sources, including nutrients. It is therefore also affected by a wide range of eutrophication phenomena: green tides (wash-in and landing of massive algal growth); phytoplanktonic blooms, deep water anoxia, fish population changes, etc. The information available concerning eutrophication in coastal marine waters only enables a qualitative assessment of the situation. Although there is known to be a cause-effect relationship between nutrients, organic matter and eutrophication in the marine environment, it is not possible to establish quantitative correlations because of the chemical behaviour of nutrients in sea water.

Nevertheless, it is clear that the frequency and geographic extent of eutrophication phenomena are increasing, even to marine areas previously believed to be unaffected.

10.2.4. Terrestrial ecosystems

In this assessment report, only ‘natural’ and ‘semi-natural’ terrestrial ecosystems have been considered. The absence of relevant data should be emphasised, even more so than for the other ecosystems.

Sensitive nutrient-poor ecosystems affected by atmospheric pollution were studied in this assessment. These very special biological systems are irreplaceable, but also under threat, not only from direct nutrient excesses and the resulting changes in populations, but also from secondary effects such as the acidification of certain types of wetlands.

Although fields containing crops were deliberately excluded from the assessment report, there is a need for a specific approach to nutrient loads to such fields and their impact on other ecosystems. It was not, therefore, possible to consider the interface between cultivated ecosystems and their downstream wetlands.

Although it was only possible to examine a small fraction of nutrient impacts in terrestrial ecosystems, the study suggests that the problem may be more serious than eutrophication phenomena in aquatic ecosystems. In terrestrial ecosystems, the impact of excessive nutrient loads leads to species and ecosystem loss. This suggests that the recovery of impacted terrestrial ecosystems would be more uncertain and, in most cases, longer than the recovery of running water bodies, or even still water bodies.

10.3. Policy effectiveness and the way forward

The report has used the DPSIR framework, identifying the driving forces and pressures, assessing environmental state and impacts, and summarising common responses (policies).

The effectiveness of policies to prevent or cure elevated nutrient levels is, in general, difficult to judge at a European scale because of the disparities in data concerning primary causes, emissions and ecosystem status, and only a general and qualitative evaluation can, therefore, be provided. There are, however, some notable exceptions to this general statement, where even at an international level, considerable efforts have been made to establish good and comparable information on sources, effects and inputs. This allows policy effectiveness to be assessed and to be fed back into the policy formulation process thus completing the management cycle.

Good examples of approaches containing a strong information element are those adopted by the Paris and Helsinki Conventions, which could perhaps provide models for policy formulation in other parts of Europe. A key aspect of both Conventions was the agreement to share information on the state of the seas, on nutrient pressures and on appropriate responses. Thus under the Paris Convention, for example, information was sought to identify the sources of nutrient inputs and improve information about eutrophication effects. Contracting Parties were required to provide all possible information, including national progress reports on the implementation of measures, and to elaborate a common basis for measurement. The success of this approach can be clearly seen in terms of achieving reductions in inputs to eutrophication problem areas in the North Sea (most countries will achieve a reduction of 50% phosphorus from 1985 to 1995, and a 20-30% reduction in nitrogen). Policies drafted under the Conventions were gradually refined as further information on environmental pressures was obtained and feedback from the implementing Contracting Parties was received. Further policies are focusing on controlling exports, particularly of nitrogen from agriculture. The approach also encouraged information sharing amongst implementing countries allowing them to benefit from each others experiences.

State of current policies

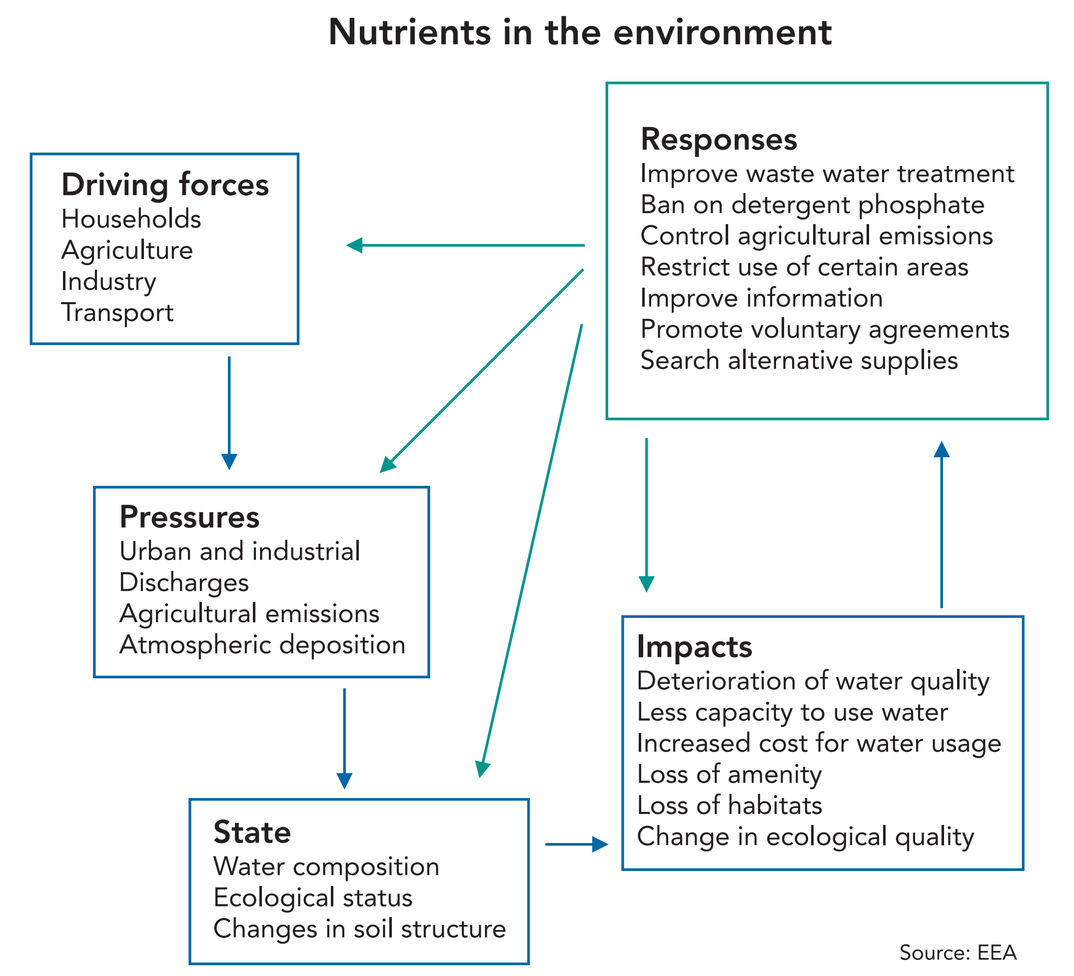
Despite the general lack of information it is possible to observe trends in the successes and failures of the measures and general conclusions can be made:

- the application of nutrient reduction policies is ‘patchy’;

- a report by the European Commission indicates that six years after its adoption the status of implementation of the Nitrate Directive in most Member States is unsatisfactory;

- there are different deadlines for the application of the requirements of the Urban Waste Water Treatment Directive. However, the response by Member States has also been slow and variable, so much so that infringement procedures against some have been instigated, by the European Commission, for no legal transposition of the Directive into national law or non-conformity of the legal transposition;

- integrated polices have been developed for many areas of Europe, for example the North Sea, the Baltic Sea, the Rhine and the Danube, other Europe wide policies focus on specific sources;



Assessing the effectiveness of responses is important in completing the environmental management cycle, providing feedback to policy makers so that policies can be refined as needed and environmental status further improved. Effectiveness can be assessed by monitoring the effect of the response in terms of reducing:

- environmental pressures, for example on nutrient loads from targeted sources (this assesses how the policy was implemented);
- environmental state, for example, on nutrient levels in the environment (this also implies a good understanding of the extent to which the targeted sources contribute to nutrient levels);
- the impact of nutrients in the environment (this also implies that the factors contributing to the impact of nutrients are well understood).

Indicators to measure the success or otherwise of policies have been or are being developed to assess effectiveness in each of these terms. Pressure indicators are important for monitoring policy progress since most EU measures are source-orientated. However, they do not provide an indication of whether the pressure addressed improved the environmental state or lessened the impact of the identified pressure. These can only be assessed in terms of state and impact indicators, which are often more difficult to measure as they suffer from chemical time-lags (i.e. only after a certain period can successes or failures of policies be noted) and because the effects of different pressures on status have to be identified and separated. An example of the differing information provided by the indicators is provided in by the phosphate ban for detergents introduced in Switzerland. The policy reduced both the pressure (P load from detergents) and improved the state (reduced levels of total phosphorus in lakes), however the impact was unchanged (algal levels were not affected significantly).

- application of integrated policies in the Paris Convention area are expected to result in a 50% reduction (25% in the case of France) of the input of phosphorus into nationally defined problem areas in the North Sea by 1995;

- however, the equivalent 50% reduction target for nitrogen will not be achieved by 1995. Here the relevant North Sea countries expect to achieve reductions of between 20% and 30% into the potential problem areas;

- although a similar approach was adopted for the Baltic Sea, under HELCOM, the lack of investment to fund policy implementation in some countries has meant that the overall targets will not be achieved until 2010;

- considerable success has been achieved in reducing nutrient inputs from point sources in particular by targeting 'hot spots', key emissions often responsible for significant inputs. An exception to this has been in Central and Eastern Europe again mostly because of lack of investment;

- agricultural inputs have been difficult to control. Even where integrated approaches have been adopted, as in the HELCOM and PARCOM countries, limited success has been achieved, because of difficulties of implementation, and because of the time-lag between response and effect;

- fertiliser usage has been reduced since the 1990's. In Central and Eastern Europe this has been largely due to economic reforms and recession, and in the EU countries due to changes in CAP.

In 1992, the European Commission published the 5th Environmental Action Programme (5EAP; COM 1992). The programme provides a strategy, objectives and targets up to the year 2000, and progress was reviewed in 1995 concluding that:

> *'the European Union is making progress in reducing certain pressures on the environment, though this is not enough to improve the general quality of the environment and even less to progress towards sustainable development. Without accelerated policies, pressures on the environment will continue to exceed (human health) standards and the often limited carrying capacity of the environment. Actions taken to date will not lead to full integration of environmental considerations into economic sectors or to sustainable development'*

This conclusion about the environment as a whole can equally well be applied to nutrient enrichment. The review goes on to state

> *'Point sources of pollution have been well targeted by regulations. Diffuse sources... have been targeted far less effectively'*

Although the strategy of the 5EAP is not fully implemented, it is clear that an increased response will be needed if targets are to be achieved. The way forward is indicated by some of the principles outlined in the 5EAP, for example by broadening the range of instruments and integrating environmental policy with other sectors.

10.4. The way forward

10.4.1. Broadening the range of instruments

The 5EAP discussed broadening the range of instruments to bring about changes in current trends and practices, and to involve all sectors of society. Whilst traditional command and control type measures still dominate the approaches used to control nutrient inputs, many interesting examples of alternative measures, which have been applied effectively, are reported for point sources, such as the charging schemes for discharges in Germany, France and the Netherlands, and for emissions to air in Sweden.

These, and other approaches, may provide an effective way forward, particularly for agriculture, as a way of overcoming the difficulties of effectively enforcing command and control instruments. For example, although the use of fertiliser taxes is not widely accepted, they have been used in Sweden and studies suggest that doubling the price of fertiliser could reduce inputs by 25%.

Fertiliser and advice programmes were among the most successful approaches for controlling inputs from agriculture in the PARCOM countries. One of the preferred measures for PARCOM's future programme is to balance fertiliser inputs on crop needs on a voluntary basis, although even this is proving to be politically a difficult agreement to achieve.

Tradable quotas for fertiliser, such as those adopted for milk production in the EU, are another strategy, which has not so far been adopted. Such an approach has been used successfully in the USA for controlling diffuse sources.

10.4.2 Integration with other policies

In talking about the achievement of the desired balance between human activity and development and protection of the environment, the 5EAP recognises that integration of environmental considerations in the formulation and implementation of economic and social policies as key. The programme identifies five target sectors for special attention. Of these, four, industry, energy, transport and, in particular, agriculture are key to achieving nutrient reductions. Although much has been achieved, better integration of environment in economic policies remains a challenge for the future. Of these, agricultural policy in particular will be key in tackling diffuse source inputs, but this continues to be both technically and politically difficult. Although the process of CAP reform has been used to integrate measures to reduce nutrient inputs, more will have to be done, for example to better focus policies such as set-aside to maximise environmental benefits. Such market controls will be essential as a report on CAP and the Environment (LEI-DLO, 1996) found that the effectiveness of measures largely depends on market regimes. The response by farmers participating in programmes under these measures depends, among other things, on incentives provided by alternative policies.

10.4.3. More effective implementation

Although policies are often formulated at a national level, the success of these measures depends on the extent to which they are applied and enforced. Thus for example the impact of directives such as the Urban Waste Water Treatment and Nitrate Directives is dependent upon the extent to which Member States designate sensitive areas and vulnerable zones.

10.4.4. Common targets and principals

The proposed Water Framework Directive tries to move away from a pressure focused approach to one which sets common objectives for environmental status, and common principles for assessing pressures and determining the measures to implement to deal with these. But it allows Member States freedom to identify measures most appropriate for improving water quality in their own areas taking into account local priorities. Extensive reporting is required which should help encourage policy learning through sharing experiences.

10.4.5. Sustainable development

As the cornerstone for environmental policies in the future, the need to take into account the sustainability of current practices relating to nutrients is key. For example, limitation of phosphorus use should consider the fact that all phosphorus ores come from mines situated in countries outside of the EU.

10.5. Needs for future policy assessment

Considerable effort is required in order to produce state, trend and policy assessments. The general mechanisms of nutrient pollution are now known with sufficient confidence to allow information needs to be satisfactorily defined but cause-effect relationships are not fully understood. It is important therefore to emphasise the necessity of improving information concerning pressures (emissions) at the same time as information concerning the state of the environment.

Emission studies require various techniques to be developed, including the modelling of administrative data, the use of discharge inventories and the use of nutrient balances in the environment. For the water environment, it is clear that there is no defined boundary between the data and techniques necessary to evaluate emissions, and to evaluate ecosystem status. Assessments should be based on surface and groundwater catchments. Co-ordination between the EEA and EUROSTAT should be ensured.

These observations all reinforce the necessity of strengthening the EIONET network for data exchange, and for implementing the EEA's monitoring and information network for inland water resources, EUROWATERNET. The latter will enable the collection of relevant pressure and state information from those rivers, lakes and groundwater that are likely to be most affected by implementing the UWWT and Nitrate Directives. State information collected over a number of years will be related to changes in the pressures within the catchments, pressures which should be reduced by implementing the requirements of the Directives. Thus once EIONET and EUROWATERNET are fully established information will be provided at a European level by which the effectiveness of Directives and other policies to reduce nutrient inputs can be judged, and will also help to identify what other policies and practices might be required.

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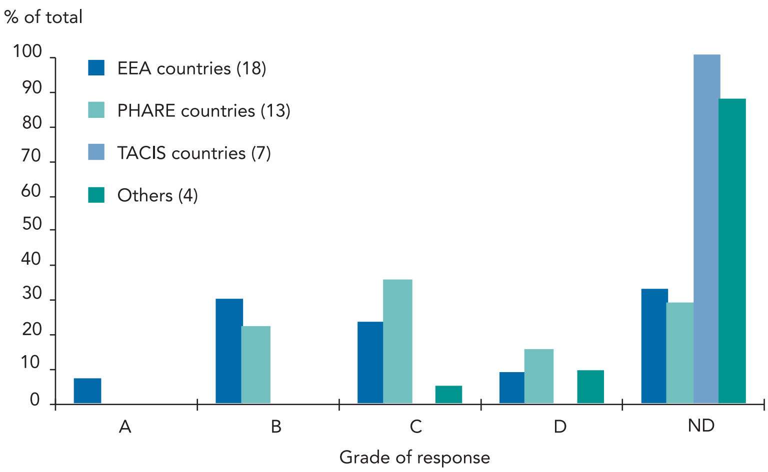
Appendix A

Figure A.1 **Percentages of responses by grade. Source: Compiled by ETC/IW**



Questionnaire responses from National Focal Points

Responses to the questionnaires were expected by May 1997. Information was in fact received between 26 February and 14 October 1997. The responses are summarised in Table A1.1. A further breakdown of the responses in terms of groups of European countries and is given in Table A.2 and in Figure A.1.

Table A.2 **Responses in terms of % of each grade/total number responses possible (that is 5 or 6 sections for 42 countries)**

Response grade (key as above)	A %	B %	C %	D %	ND %	Total number of responses possible
EEA countries (18)	7	30	23	9	31	105
PHARE countries (13)	0	22	35	15	28	74
Tacis countries (7)	0	0	0	0	100	39
Others (4)	0	0	4	9	87	23

Response to questionnaire/request for information as of October 1997 — **Table A.1**

COUNTRY	Section 1 Pressures	Section 2 Lakes	Section 3 Rivers	Section 4 Marine	Section 5 Land	Section 6 Policies
Albania	nd	nd	nd	D	D	nd
Armenia	nd	nd	nd	NR	nd	nd
Austria	B	B	B	NR	C	B
Azerbaijan	nd	nd	nd	nd	nd	nd
Belarus	nd	nd	nd	NR	nd	nd
Belgium	C [1]	nd	C [1]	C [2]	nd	C [1]
Bosnia and Herzegovina	C	D	B	C	nd	D
Bulgaria	C	D	B	D	nd	nd
Croatia	nd	nd	nd	nd	nd	nd
Cyprus	nd	nd	nd	nd	nd	nd
Czech Republic	B	C	C	NR	nd	nd
Denmark	B	A	A	B	C	B
Estonia	B	C	B	C	nd	nd
Finland	B	A	A	B	B	B
Former Yugoslav Rep. of Macedonia	nd	C	C	NR	nd	nd
France	B	B	A	B	nd	B
Georgia	nd	nd	nd	nd	nd	nd
Germany	B	C	C	C	C	C
Greece	C	D	C	nd	nd	D
Hungary	nd	C	C	NR	nd	nd
Iceland	nd	nd	nd	nd	nd	nd
Ireland	D	B	C	C	C	B
Italy	C	nd	D	nd	nd	nd
Latvia	C	C	C	C	nd	nd
Liechtenstein	nd	nd	nd	NR	nd	nd
Lithuania	B	C	C	D	D	C
Luxembourg	nd	nd	nd	NR	D	nd
Moldova	nd	nd	nd	NR	nd	nd
The Netherlands	B	B	D	C	C	B
Norway	B	B	B	B	C	nd
Poland	B	C	B	B	B	C
Portugal	nd	C	nd	nd	nd	nd
Romania	C	D	B	B	D	C
Russian Federation	nd	nd	nd	nd	nd	nd
Slovak Republic	B	nd	B	NR	C	C
Slovenia	C	C	B	B	D	nd
Spain	nd	nd	D	D	nd	nd
Sweden	B	B	B	B	C	B
Switzerland	C	D	D	NR	nd	nd
Turkey	nd	nd	nd	nd	nd	nd
Ukraine	nd	nd	nd	nd	nd	nd
United Kingdom	B	C	A	D	C	A

Key :
A = Very detailed
B = Detailed (sufficient)
C = Contains main elements
D = Some information (insufficient)
NR = Not relevant (no marine waters)
nd = No information