

# European assessment of eutrophication abatement measures across land-based sources, inland, coastal and marine waters



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**Cover photo:** Biological sewage treatment plant (Pflanzenkläranlage) Langenreichenbach, Germany, May 2001

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## Glossary

BQE	Biological Quality Element
CAP	Common Agricultural Policy
CIS	Common Implementation Strategy
EAP	European Environmental Action Programme
EU	European Union
HELCOM	Helsinki Convention on the Protection of the Marine Environment of the Baltic Sea Area
LRTAP	Long-range Transboundary Air Pollution
MSFD	Marine Strategy Framework Directive
N	Nitrogen
NAP	Nitrate Action Programme
ND	Nitrates Directives
NVZ	Nitrate Vulnerable Zone
OSPAR	Convention for the Protection of the Marine Environment of the North-East Atlantic
P	Phosphorous
p.e.	Population Equivalent
RBMP	River Basin Management Plan
UWWTD	Urban Waste Water Treatment Directive
WFD	Water Framework Directive
WWTP	Wastewater treatment plant

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## Executive Summary

Rivers, lakes and groundwater as well as coastal and marine waters have suffered from nutrient pollution already since the 1950s. Excessive nutrient releases from agricultural and urban areas contribute to water quality degradation in many parts of Europe. Eutrophication is the result of excessive nutrient enrichment, which may cause accelerated growth of algae and plants and a wide range of impacts on aquatic ecosystems such as the loss of aquatic biodiversity and proliferation of toxic algal blooms.

European policy has consistently identified eutrophication as a priority issue for water protection, in particular through the Urban Wastewater (UWWTD) and the Nitrates Directives (NiD), as well as the more recent Water Framework Directive (WFD) adopted in 2000, and the Marine Strategy Framework Directive (MSFD) adopted in 2008. In addition to EU legislation, there are a number of international conventions on river basin management e.g. for the protection of the River Rhine (ICPR) or the Danube River (ICPDR) as well as conventions for the protection of the marine environment, e.g. for the North-East Atlantic (OSPAR), the Baltic Sea Area (HELCOM), the Black Sea (Bucharest Convention) and the Mediterranean Sea (Barcelona Convention). These conventions are complemented by the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP) and the EU Directive on national emission ceilings for atmospheric pollutants. All of these conventions aim at controlling pollution from land-based sources, air pollution and for the marine areas also for controlling pollution from maritime transport. These international frameworks are supplemented by national legislation.

The general objectives of European and national policies, such as good ecological status under the WFD or the good environmental status under the MSFD, need to be broken down to more specific objectives and quantitative targets to guide eutrophication abatement. This includes the setting of nutrient load reduction targets, such as the 50 % reduction target that was set by the countries cooperating under OSPAR for the Greater North Sea region. Nutrient concentration targets, also termed nutrient standards are important management tools, when linked to direct and indirect ecological impacts of eutrophication, such as algal blooms and oxygen depletion. In water bodies currently exceeding the nutrient standards, these must be translated into targets for nutrient load reductions. Nutrient load reduction targets, using the desired state of the water body as the aim, are more ecologically sound than targets set with respect to some reference year. This is the basis for the movement away from aspirational targets to more ecologically based targets seen in e.g. the HELCOM, OSPAR and major river conventions.

Indicators are a widely used instrument for monitoring the eutrophication status and to document the progress of policy implementation over time. Eutrophication indicators comprise pressure indicators (nutrient emissions and loads), state indicators (nutrient concentrations in the water) and impact indicators describing the health of aquatic ecosystems (such as biological indicators, oxygen and chlorophyll-a concentration, or Secchi depth).

A targeted program of measures to reduce nutrient pollution pressures can be established once the most significant pressures in relation to eutrophication have been identified. The program of measures is the central element of an integrated water resources management plan that ideally fulfils the requirements of different water related policies like WFD, UWWTD, NiD and national legislation. The program of measures has to be cost-effective and consulted with stakeholders to become operational. This report provides an overview of measures to reduce nutrient pollution from point and diffuse sources.

In order to exemplify different eutrophication abatement strategies eight case study regions (territory of Member States or river basins) were selected for a detailed analysis: Denmark, Finland, the Po river basin, the Ebro river basin, the Rhine river basin, Lake Constance, the Danube river basin and the Baltic Sea. Eutrophication was a significant problem in these regions and they illustrate how the responsible authorities approached the pollution problem, and how they developed and implemented

eutrophication control programmes. The case study regions cover i) different European regions, ii) lakes, rivers, coastal and marine waters, iii) national and transboundary settings, and iv) examples of interlinkages between freshwater and coastal / marine water management.

The case studies are analysed in view of different aspects such as governance issues (cooperation between sectors and the role of river basin commissions), the role of research, stakeholder participation, choice of measures and funding instruments (e.g. voluntary versus mandatory measures) and time lag effects of measures taken. The report demonstrates the lessons learnt from the case studies and concludes with an analysis of progress achieved and remaining implementation gaps.

Overall, the implementation of European and national policies over the past decades has greatly reduced the levels of nutrients in Europe's waters. Nevertheless, eutrophication is still considered as a major threat to significant parts of Europe's waters. Strong and coordinated actions are needed in the future to combat the eutrophication problem in order to reach the ambitious goals of WFD and MSFD. This includes better cross-sectoral integration of water policies, stronger enforcement, burden sharing between sectors, innovative instruments, and alternative concepts such as payment for ecosystem services.



# 1 Introduction

## 1.1 Eutrophication – still a problem in Europe’s waters

Eutrophication is a major problem in Europe’s waters despite substantial management efforts in the last decades (EEA, 2012; EEA, 2015a). The concentrations of nutrients in many European waters are still excessively high which promotes eutrophication in surface waters with a series of negative impacts and consequences. Starting with an increasing anthropogenic nutrient input in the 1950s and 1960s, partly caused by the application of phosphate-containing detergents, the resulting unnaturally high loads led in many waters to eutrophication, associated with a significant increase of high nutrient concentrations. Eutrophication and associated negative symptoms were recognized in the 1970s as a socially relevant problem and first policies tackling nutrient pollution were adopted. The following examples highlight that important European lake, sea and river basins are affected by eutrophication:

- Eutrophication of European lakes began in the 1960s as a result of population and industrial growth. In many cases, water quality deteriorated to a critical level within only one decade with frequently occurring massive algal blooms and depleted oxygen levels. This was in many cases the primer for the development of policies and implementation of measures. Yet in 2009, around 20% of European lakes suffered from nutrient enrichment (EEA, 2012).
- First signs of eutrophication in the Baltic Sea were visible already in the 1950s and the ecosystem changed from an oligotrophic clear-water sea into a highly eutrophic marine environment with wide-spread algal blooms (Andersen et al., 2014). In the North Sea, eutrophication effects peaked in the 1970s and continue to be a significant problem in the coastal areas, enclosed estuaries and embayments (OSPAR, 2010).
- The Mediterranean which is generally regarded as a highly oligotrophic sea receives excessive loads of nutrients from sewage effluents, riverine inputs, aquaculture farms, fertilizers and industrial facilities especially in the northern parts of the basin leading to intense eutrophic phenomena in the coastal zones with many adverse effects for the marine ecosystem and humans (UNEP, 2007; Kyradis and Kitsiou, 2012). Similar concerns exist for the Black Sea environment which has been severely damaged by eutrophication since the 1970s (UNDP-GEF, 1999).
- Large and important European river basins such as the River Rhine, River Elbe, Po River or River Danube were affected by excessive nutrient input resulting in eutrophication symptoms especially during the 1970s and 1980s (Wilken, 2006; Hardenbicker et al., 2014; Pirrone et al., 2005; UNDP-GEF, 2006). Despite major improvements, nutrient pollution continues to be a significant pressure to the aquatic ecosystem.
- Additionally, high concentrations of nitrates are a widespread phenomenon in Europe’s groundwater resources showing that diffuse pollution, particularly from agriculture, continues to be a problem (EEA, 2012). High nitrate concentrations in groundwater impair the use of the resource for human consumption and were therefore not in the centre of this report.

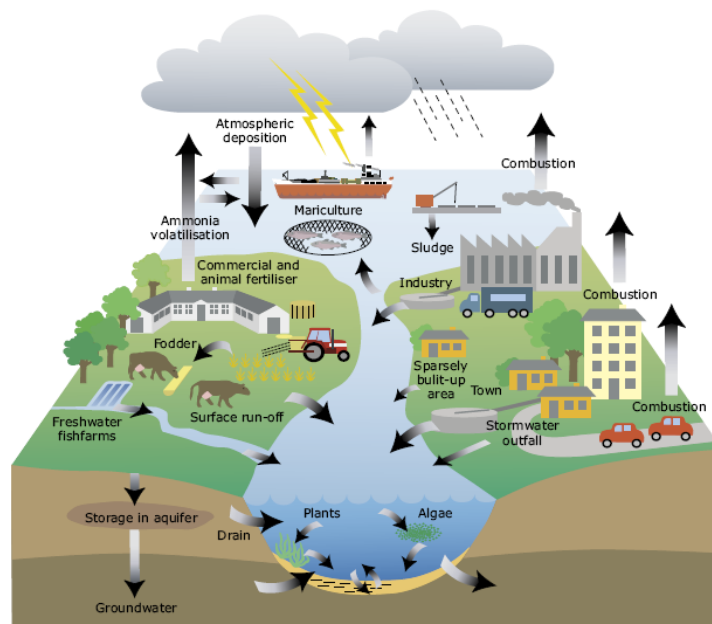
## 1.2 Eutrophication and sources of nutrient pollution

Eutrophication is the result of excessive nutrient enrichment of water resources and this may cause accelerated growth of algae and higher forms of plants resulting in a wide range of impacts on aquatic ecosystems (Carpenter et al., 1998; Smith, 2003; Smith et al., 2006; Schindler, 2006; Dodds, 2007; Dupas et al., 2015). This includes the loss of aquatic biodiversity, toxic algal blooms and dead zones. High phytoplankton concentration leads to reduced water transparency and once algae are dying the degradation of biomass causes oxygen depletion in deeper zones of lake and seas. In addition, an increase of dissolved organic carbon and bacteria concentrations lead to poor water quality associated

with odour and taste disturbances. Eutrophic water bodies are therefore only limited useable for local recreation, and drinking water purification gets more cost intensive. The final stage of eutrophication is a hypertrophic water body, with depleted oxygen content causing the death of fish and other organisms.

To understand the nutrient enrichment problem, it is important to understand where the pollution originates and through which pathways nutrients are transported to aquatic ecosystems (Figure 1.1). Nutrients can enter water bodies through different pathways, arising from point and diffuse sources. Point sources have a distinct point of discharge such as inflows of urban wastewater, industry and fish farms. In contrast to this, diffuse source do not have a distinct point of discharge, such as nutrient losses from agriculture, nutrient releases from scattered dwellings and atmospheric deposition on surface waters. The actual contribution of different sources to the total amount of nutrients entering waters is done during source apportionment studies (EEA, 2005).

**Figure 1.1: Nitrogen cycle in the aquatic environment, where all relevant sources and pathways are shown. The phosphorous cycle shows similar sources and pathways, though phosphorus is not emitted to the air, which is why anthropogenic deposition is of no importance.**



**Source:** Ærtebjerg et al., 2003.

Agriculture is the most important sector for nitrogen releases to aquatic systems and contributes typically 50-80 % to the total load (Table 1.1; EEA, 2005). Modern agricultural practices very often involve an intense use of mineral fertilisers and manure, leading to high nutrient surpluses that are released to aquatic ecosystems. The more intensive agricultural areas are managed the higher is the total area-specific load. The hot-spot areas in Europe with the highest N-fertilizer applications are the agricultural areas in the Western parts of Europe (Bouraoui et al., 2009; EEA, 2012).

Households and industry are the most significant source for phosphorus inputs to aquatic systems. While urban wastewater sources constitute about 50-90% of the point source discharges (Table 1.1), industrial sources constitute about 17% (EEA, 2005). However, due to improved wastewater treatment of municipal and industrial wastewater, the amount of discharged pollutants has decreased significantly since the 1970s. Both, treatment technologies and the number of inhabitants connected to sewer and wastewater treatment systems have increased markedly. As these point source discharges have been reduced during the last previous years, agriculture becomes in some regions the most important phosphorus source (EEA, 2005). Here, soil erosion and diffuse emissions of phosphorus bound to soil particles becomes the most important pathway.

Atmospheric depositions directly on surface waters, such as nitrogen oxide and ammonia depositions or phosphorus fall out as dust generally play a greater role in marine waters than in inland waters. In the Baltic Sea, atmospheric nitrogen deposition amounted to 27% of the total nitrogen input and atmospheric phosphorus deposition constituted 7% of the total phosphorus input (year 2012; HELCOM, 2015a). For marine waters, nitrogen oxide emissions from shipping are also an important nutrient source and the amounts are increasing due to the increase in ship traffic. In the Baltic Sea and North Sea shipping was the second largest source of nitrogen oxides deposition (EMEP 2015). In the Danube river basin, direct nitrogen atmospheric deposition contributes about 2 % while atmospheric deposition of phosphorous accounts for around 0.8 % of total emissions (ICPDR, 2015).

**Table 1.1: Estimated nutrient discharges and losses from land-based sources (atmospheric deposition not included) in selected European lakes, rivers and seas.**

Catchment	Nitrogen				Phosphorus			
	Total Diffuse sources (%)	Therefrom agricultural sources (%)	Total Point sources (%)	Therefrom urban waste water sources (%)	Total Diffuse sources (%)	Therefrom agricultural sources (%)	Total Point sources (%)	Therefrom urban waste water sources (%)
<b>North Sea</b>	69	85	31	89	44	85	56	77
<b>Baltic Sea</b>	45	70–90*	12	ca. 90	45	60–80*	20	ca. 90
<b>Po River Basin</b>	80		20	57	93		7	90
<b>Ebro River Basin</b>	64 <sup>#</sup>						88 <sup>#</sup>	
<b>Rhine River Basin</b>	69		31	84**	51		49	89
<b>Danube River Basin</b>	84	42	16	25	67	28	33	51
<b>Lake Constance</b>	81		19		88		12	

**Note:** Not included in this table are natural background loadings and unspecified loads

\*agricultural contribution to total riverine load

\*\* diffuse urban loads included

# agricultural non-point sources account for 64% of NO<sub>3</sub> loads while urban and industrial point sources are responsible of 88% PO<sub>4</sub>-P loads (Torrecilla et al. 2005)

**Sources:** IGKB, 1999; EEA, 2005; Pirrone et al., 2005; Torrecilla et al. 2005; HELCOM, 2011; ICPDR, 2015; IKSIR, 2015.

Another important nutrient source is the internal release of nutrients from sediments. The excessive anthropogenic nutrient discharges of the past 100 years have led to an accumulation of nutrients in the sediments of lakes, rivers, estuaries and coastal and marine waters, building up a “nutrient legacy”. The amount of internal nutrients released can be substantial and can exceed the amount of external direct, atmospheric and waterborne nutrient inputs. In the Baltic Sea, for instance, it has been estimated that the amount of phosphorus released from the sediments during hypoxia is approximately one order of magnitude greater than the anthropogenic total P loading (Conley et al., 2002). Hence, to combat internal eutrophication ecotechnological or biomanipulation methods have been applied in particular in lakes, but also in the Baltic Sea (see chapter 5.6).

### 1.3 Motivation for this report

European policy has addressed the eutrophication problem since the 1970s. When eutrophication became a major problem in the North and Baltic seas and parts of the Mediterranean in the late 1980s, the Urban Waste Water Treatment Directive (UWWTD) and Nitrates Directive (ND) were both adopted in 1991. The two directives led to a significant improvement of water quality in the past decades. Most

notably, the Water Framework Directive (WFD), adopted in the year 2000, sets provisions for a significant step forward in combating eutrophication by setting targets for achieving good ecological status, ensuring nutrient levels that do not entail accelerated growth of algae or aquatic plants. Member States need to incorporate cost-effective measures to tackle all sources of nutrient pollution into their River Basin Management Plans.

In addition to the EU Directives, the 7<sup>th</sup> European Environmental Action Program (EAP) has amongst others the objective to ensure that by 2020 "the nutrient cycle (nitrogen and phosphorus) is managed in a more sustainable and resource-efficient way. This requires, in particular: taking further steps to reduce emissions of nitrogen and phosphorus, including those from urban and industrial wastewater and from fertilizer use, inter alia, through better source control, and the recovery of waste phosphorus".

Despite all the efforts in the past and sound objectives in European environmental legislation, eutrophication abatement remains a major challenge for water management. The progress in implementing existing directives is in many cases slower than expected and the objectives have only partly been achieved (EC, 2013b, 2015a). Therefore, it is time to take a closer look at why European policies have not been as powerful as expected.

The aim of this report is therefore to analyse the implementation of European policies for eutrophication abatement of inland, coastal and marine waters. Examples from policies and case studies are used to illustrate the different steps in policy implementation. The assessment provides also experiences with the implementation of measures under the Water Framework Directive (WFD) and the Marine Strategy Framework Directive (MSFD).

Within this report eight case studies were selected from across Europe: Denmark, Finland, the Po river basin, the Ebro river basin, the Rhine river basin, Lake Constance, Danube river basin and the Baltic Sea region. The analysis was done to illustrate how measures have been implemented on the ground to reduce nitrogen and phosphorus releases and to show success stories as well as remaining challenges. The case study regions cover i) different European regions, ii) lakes, rivers, coastal and marine waters, iii) national and transboundary settings, and iv) examples of interlinkages between freshwater and coastal / marine water management.

The study focusses on measures related to nutrient abatement from diffuse and point sources. Additional beneficial impacts from restoration measures such as improvements in hydromorphology are not covered. The report concludes with some lessons learnt and an outlook on the challenges ahead. The Annex contains a systematic overview of the selected case studies providing information on the driving forces for eutrophication, pressures exerted, the state and impacts on water and response measures taken to combat eutrophication.

This report is structured around the following questions:

- Why is eutrophication a problem in Europe and why this report? – chapter 1
- Which European Directives, international conventions are in place to tackle the eutrophication problem? – chapter 2
- What qualitative objectives and quantitative targets have been set in eutrophication management programmes? – chapter 3
- What indicators are used to assess eutrophication and to measure changes in pressures, impacts and state of the aquatic environment? – chapter 4
- What mitigation measures are available to combat eutrophication and how have these been implemented? – chapter 5
- What are the lessons learnt and the challenges ahead in combating eutrophication in Europe? – chapter 6

## 2 European policies and other legislative frameworks

### 2.1 An overview of European policies for eutrophication abatement

The development of EU legislation to tackle eutrophication has a long history, starting with the 1<sup>st</sup> Environmental Action Programme (EAP) (1973-1976) which stipulated the setting of environmental targets to tackle water pollution, including nutrients (Hey, 2006). Despite major improvements in the quality of European waters, the current 7<sup>th</sup> EAP of the European Union (2013-2020) actively supports further efforts to manage the nutrient cycle, calling for more cost-effective, sustainable and resource-efficient approaches, in particular regarding the efficient use of fertilisers.

The first major legislative action on eutrophication abatement at European level followed from the European Council Resolution of 28 June 1988 which identified the need for **more control on emissions** from municipal and industrial sewage, and from agriculture (EC, 1988). This led to the enactment of the Urban Waste Water Treatment Directive (91/271/EEC) and the Nitrates Directive (91/676/EEC) (Table 2.1). The more recent Groundwater Directive (2006/118/EC) aims to prevent or limit inputs of pollutants into groundwater. It is important to note that the Directives presented in Table 2.1 require Member States to monitor parameters relevant to eutrophication and set ecologically relevant guideline values; however, only the UWWTD and the Nitrates Directive have an explicit requirement to assess eutrophication (EC, 2009).

The **Water Framework Directive** (2000/60/EC) sets ambitious environmental targets, aiming for "good status" of all freshwater, transitional, coastal water bodies and for groundwater by 2015. Amongst other things, the definition of each ecological status class describes the extent to which biological and physico-chemical quality elements may differ in that class compared to their reference (or high status) conditions as a result of human activity. The assessment of eutrophication is strongly implied in the classification of water bodies, with quality elements such as "phytoplankton" and "macrophytes and phytobenthos" and the requirements to avoid their accelerated growth (EC, 2009). Eutrophication abatement is required when nutrient enrichment affects these quality elements to the extent that the water body is classified less than good status or is at risk of deterioration. As the UWWTD and the Nitrates Directive do not specify any methods or guideline values for assessing eutrophication, Member States have developed their own assessment systems and criteria. The WFD provides an avenue for intercalibration based on status classes and quality elements (EC, 2009; see chapter 3 on Objectives and Targets).

The **Marine Strategy Framework Directive** (2008/56/EC) provides for a streamlined approach to the management of European marine waters. The MSFD aims at achieving and maintaining "good environmental status" by 2020. The good environmental status refers to the intrinsic conditions of the ecosystem. One of the eleven descriptors of good environmental status directly refers to eutrophication abatement (Descriptor 5). The assessment of eutrophication in marine waters needs to take into account the assessments made for coastal and transitional waters under the WFD, in a way which ensures comparability, taking also into consideration the information and knowledge gathered and approaches developed through the four marine conventions mentioned above. The assessment should take into account nutrient levels as well as primary and secondary effects of nutrient enrichment. Because most sources of nutrients in marine waters will originate inland, the MSFD considers nutrient loads from relevant rivers, and encourages close integration of freshwater and land-based legislation (e.g. WFD) and cooperation with land-locked Member States.



**Table 2.1: Main European directives relevant for eutrophication abatement.**

Directive/policy	Main objective relevant for eutrophication abatement	Main requirements relevant for eutrophication abatement
<b>Freshwater, transitional and coastal waters</b>		
<b>Bathing Water Directive (76/160/EEC) and (2006/7/EC)</b>	To monitor the state of bathing waters by measuring concentration of bacteria (e.g. intestinal enterococci, <i>Escherichia coli</i> , cyanobacteria or bluegreen algae), Emissions of bacteria is associated with waste source such as animal feedlot or septic tank which also emit high concentrations of nitrate and phosphorus and lead to eutrophication.	Action to be taken when there is a tendency for bacterial proliferation and a health risk has been identified or presumed.
<b>Freshwater Fish Directive (78/659/EEC)</b>	To monitor and set nutrient standards to safeguard waters from the harmful consequences from discharge of pollutants.	Action to be taken when phosphorus exceeds guideline values. Phosphorus values explicitly set for reducing effects of eutrophication.
<b>Urban Waste Water Treatment Directive (91/271/EEC)</b>	To identify as sensitive areas water bodies in a eutrophic state (or future state without measures), taking into account nitrate and phosphorous, and tackle point sources of nutrient pollution (urban wastewater, and discharges of food-processing industry).	Designation as sensitive areas results in action regarding wastewater treatment independent of the origin of the pollution.
<b>Nitrates Directive (91/676/EEC)</b>	To designate catchment areas of water bodies that are eutrophic (i.e. water with nitrate concentration of 50 mg l <sup>-1</sup> ) or will become eutrophic if no measures are taken, and select mandatory measures to reduce emissions.	Designation as Nitrate Vulnerable Zone results in establishment of measures, which focus on nitrate pollution from agriculture.
<b>Water Framework Directive (2000/60/EC)</b>	To achieve Good Ecological Status (GES) in all inland, transitional and coastal waters. GES for example includes an absence of undesirable disturbances due to accelerated growth of algae	Action to be taken when water bodies are classified as worse than GES or judged at risk of deterioration. For groundwater, significant and sustained upward trend in the concentration of pollutants must be reversed.
<b>Groundwater Directive (2006/118/EC)</b>	To establish quality standards for nitrates and pesticides, support the establishment of additional threshold values, and implement measures to reduce emissions.	Action to be taken when quality standards are not met.
<b>Marine (and coastal) waters</b>		
<b>Marine Strategy Framework Directive (2008/56/EC)</b>	To achieve Good Environmental Status in all marine waters, including eutrophication as core descriptor (n°5).	Action is required for waters classified as worse than good environmental status.
<b>Air / atmospheric emissions</b>		
<b>National Emission Ceilings for Atmospheric Pollutants Directive (2001/81/EC)</b>	To establish national emission ceilings for ammonia and NO <sub>x</sub> emissions and report on emissions.	Action to be taken when national emissions are exceeded.

Source: EC, 2009.

**International conventions** exist for some major international river basins in Europe including the Elbe (1991), Danube (1994) and Rhine (1999), and seas, including the Baltic, North Atlantic and Mediterranean (Table 2.1Error! Reference source not found.). International conventions and their operating Commissions provide an arena to discuss coordination action between countries outside the processes of EU directive implementation (see text box below).

**Box 2.1: National and European policies for eutrophication abatement: the case of the International Commission for the Protection of the River Rhine (ICPR)**

After the Sandoz accident in Basle in November 1986, the states bordering the River Rhine agreed on the Rhine Action Programme for its ecological rehabilitation. This programme of 1987-2000 aimed to reduce nutrient input by 50%, mostly based on the application of the state of the art in technology. The subsequent management program “Rhine 2020” was adopted in 2001 by the Rhine Ministers and was seen as an implementation of the requirements of the EU WFD and the Swiss water policy. In 1985 only about 85% of the inhabitants were connected to urban wastewater treatment plants while in 2000 the percentage has increased to 95%. The nitrogen loads to the river decreased between 1985 and 2000 from ca. 500,000 tons to 360,000 tons.

Despite the fact that the Rhine Action Plan 1987-2000 was not legally binding, most of its emission reduction goals have been achieved. The success of the Rhine Action Plan is often explained by the more business-like and non-hierarchical approach that was adopted. Some clear success factors were that (1) the targets had been defined with precision, that (2) all Rhine bordering countries implemented their decisions and reported periodically on progress and deficits. Moreover, the European water directives have been very influential from the 1970s onwards, in addition to other international fora, such as the North Sea Ministers Conferences, domestic legislation, the activities of environmental NGOs and the waterworks in the Rhine basin, growing environmental awareness, technological innovation and changes in the structure of the industry in the basin.

In some cases, co-ordination between Member States for the achievement of the WFD or MSFD may be facilitated via **international commissions**. For example, the International Commission for the Protection of the Danube River (ICPDR) - a commission of the 14 riparian states of the Danube and the EU - acts as the body for co-ordinating both the implementation of the Danube River Protection Convention and the WFD International River Basin Management Plan. When the WFD came into force in 2000, the majority of the Danube riparian states were not yet members of the EU and therefore not required to implement EU legislation. Nevertheless through the ICPDR, riparian countries agreed to the development of the RBMP. In 2009 when the plan was adopted, five riparian states had already become EU Member States. The measures which the plan foresees aim at a substantial reduction of nutrient pollution in the Danube river basin (EC, 2012). A joint management approach is detailed that contains as basic measures the implementation of measures to fulfil the Nitrates Directive and UWWTD in EU Member States and the implementation of the code of good agricultural practice established by the ICPDR (ICPDR, 2015).

Regarding marine waters, a series of **four conventions** for the protection of European marine waters were signed in the 1970s (OSPAR for the North-East Atlantic, HELCOM for the Baltic Sea, and Barcelona Convention for the Mediterranean) together with the Bucharest Convention in 1992 for the Black Sea. These conventions regard eutrophication abatement as a major priority and this led to the signature of specific protocols establishing advanced approaches to assessing and monitoring marine eutrophication and controls on land-based sources of nutrients. Additional international conventions are relevant. The UNECE Convention on Long-range Transboundary Air Pollution (LRTAP) (signed in 1979) contributes to eutrophication abatement by controlling atmospheric emissions of ammonia and nitrogen oxides. The LRTAP is complemented in the EU by the National Emission Ceilings (NEC) for the Atmospheric Pollutants Directive (2001/81/EC) (EC, 2005a), which was developed alongside. Both the convention and the Directive have similar ambitions for emissions, and help address problems of air pollution and eutrophication on a broad regional basis.

### Box 2.2: Eutrophication abatement in the North-East Atlantic region

The OSPAR Commission developed targets to reduce nutrients as part of its Eutrophication Strategy. OSPAR reports that of the nine countries bordering the North and Celtic Seas, six have achieved a 50% or greater reduction in their phosphorus emissions between 1985 and 2005. This is primarily due to systematic implementation of urban wastewater treatment (OSPAR, 2010). With regards to nitrogen, reductions of approximately 30% since 1985 have been achieved, mostly due to measures targeting agricultural use of fertilizers and increasing urban waste water treatment. While reductions have been achieved, many countries are not reaching their targets. Only Denmark reached its 50% reduction target between 1985 and 2005 (OSPAR, 2010). Reasons for the failure to meet targets stem from several sources. Measures to reduce nutrient sources were often taken later than envisaged. Also, there is a lag in response from ecosystems to show nutrient reductions, as nutrients can collect in sediments and continue to contribute to eutrophication over time. The transport of nutrients from other areas also contributes to eutrophication problems.

**Table 2.2: Main European conventions relevant for eutrophication abatement.**

Directive/policy	Main objective relevant for eutrophication abatement	Main requirements relevant for eutrophication abatement
<b>Freshwater, transitional and coastal waters</b>		
Convention for the Protection of the Rhine Rhine River Basin	To strengthen cooperation between the Community and the Rhine riparian States in order to preserve and improve the ecosystem of the river.	To reduce pollution by harmful substances through emission reduction and securing standard in wastewater treatment.
Danube River Protection Convention  Danube River Basin	To achieve sustainable and equitable water management in the Danube Basin. Agreement to reduce pollution loads to the Black Sea.	
Elbe River Protection Convention  Elbe River Basin	To prevent the pollution of the Elbe River and its drainage area	
<b>Marine (and coastal) waters</b>		
Convention on the Protection of the Marine Environment of the Baltic Sea Area (1974, 1992).	To restore the Baltic marine environment to good environmental status and unaffected by eutrophication by 2021.	To quantify and assess nutrient emissions, discharges, losses and inputs to, as well as concentrations and effects.
Convention for the Protection of the Mediterranean Sea Against Pollution (1975).	To assist Mediterranean countries to formulate and implement monitoring programmes, pollution control measures and action plans. First protocol on eutrophication signed in 1980.	The associated Strategic Action Plan aims to reduce inputs of nutrients from land-based sources.
Convention on the Protection of the Black Sea against Pollution (1992).	To reduce and control pollution from land-based sources to reduce risk of eutrophication.	The associated Strategic Action Plan has provisions to monitor and reduce the inputs of nutrients.
<b>Air / Atmospheric emissions</b>		
UNECE Convention on Long-range Transboundary Air Pollution (LRTAP).	To control and reduce emissions of nitrogen oxides and ammonia. Depositions shall not exceed the critical loads of nutrient nitrogen that allow ecosystem recovery	Emission reduction commitments are set per Convention Party, based on estimates of critical loads reported by the Parties.

**Source:** EC, 2009.

This overview of the European policy framework on eutrophication abatement clearly shows that future EU and national interventions can rely on an already extensive set of policies. The current 7<sup>th</sup>

Environmental Action Programme (2013–2020)<sup>1</sup> identifies excessive nutrient emissions in the atmosphere and in water bodies as a key threat to the European environment. It calls for more action on inefficient fertiliser use, adequate wastewater treatment, and points towards stronger enforcement of existing legislation, a tightening of standards and more holistic and integrated approaches to reduce emissions. Better coordination in the implementation of European legislation (see Table 2.2) is encouraged so as to avoid shifting nutrient emissions across media and maintain the risk of further eutrophication in the future.

## 2.2 Key features of European policies for eutrophication abatement

To simplify the complex legislative framework presented above, this sub-chapter will focus on European policies that explicitly tackle eutrophication abatement. Regarding fresh and coastal waters, the WFD acts as the main Community instrument for implementing an integrated approach, taking into account the requirements of the UWWTD and NiD. Regarding marine waters, the MSFD is the main Community instrument, although implementation of the first cycle is still on-going and available information is thus yet limited. A brief comparison between the four key directives for eutrophication abatement shows the following.

**Targeted pressures and drivers:** The WFD recognises point and diffuse sources of nutrient pollution from all drivers, while the MSFD recognises all land-based eutrophication pressures and drivers, as well as atmospheric deposition from transport and shipping emissions. The UWWTD focuses more specifically on phosphorus and nitrogen emissions from urban areas, while the Nitrates Directive focuses on nitrate emissions from fertiliser and manure use in crop production and manure storage at livestock keepings.

It is important to note that the UWWTD does not cover the collection and treatment from private sewers (septic tanks), which can represent in some rural catchments a significant source of nutrients into freshwater water bodies. Traditionally, in many countries, the responsibility for the operation and maintenance of private sewers is held by individual householders. However, such arrangements have been characterised by poor performance and maintenance leading to nutrient leakage and eutrophication. Some countries have now stepped up their approach to private wastewater treatment. For example, the UK Government moved in 2011 the legal responsibility for managing private sewers from householders to the water industry. This led to an additional 220,000 km of sewers to be managed by water utilities (DEFRA, 2012). It is argued that the involvement of water utilities in sewer management will improve their environmental performance.

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<sup>1</sup> Decision No 1386/2013/EU of the European Parliament and of the Council of 20 November 2013 on a General Union Environment Action Programme to 2020 “living well, within the limits of our planet”. L 354/171 (28/12/2013).

### Box 2.3: Strengthening eutrophication abatement in the Ebro river basin

Eutrophication of the Ebro River started around the 1960s and 1970s with increasing population, economic activity and per capita water consumption. Sewage collection systems were installed in most Spanish localities during the 1950s and 1960s, and industrial wastewaters were added to these urban sewages since the 1970s. Since the 1990s, there is a trend of reduced nutrient loading likely due to improved water treatment and land management. The construction of treatment facilities in the main villages in the early 1990's, which was triggered by EU Directives the UWWTD, caused a very clear decrease in PO<sub>4</sub>-P content.

The Ebro River Basin Management Plan 2010-2015 includes 131 measures to reach the water quality objectives, to improve sanitation and water treatment and makes reference to the measures implemented under the EC Nitrates Directive (Ebro River Basin Authority, 2010). It responds to the EU WFD and the Spanish Water Act, and establishes more demanding water quality objectives than in previous times. The Ebro River Basin Management Plan has 12 key elements, and includes the ambitious environmental objective that at least 83% of river water bodies will attain the WFD good ecological status by 2015.

At the same time the plan responds to the Spanish Water Act, which sets the regulations to fulfil the water demands for different sectors. The Water Act also covers water quality, but also follows the EU WFD, which has been incorporated into the Spanish Water Act.

Describe the program of measures instead that the plan responds to some reporting requirements

**Spatial coverage of management units:** The UWWTD focuses on agglomerations of at least 2000 population equivalents (PE) and their receiving waters, as well as sensitive areas. Sensitive areas are water bodies in a eutrophic state, or at risk of becoming eutrophic without adequate measures, taking into account both nitrate and phosphorous parameters. The Nitrate Directive focuses on Nitrate Vulnerable Zones (NVZs) defined as catchment areas of all water bodies that are eutrophic or will become eutrophic if no measures are taken. A eutrophic state is defined as water with nitrate concentration of 50 mg l<sup>-1</sup>.

Member States have to adopt a Nitrates Action Programme for their NVZs or can choose to design those programmes for their entire territory. In addition, Codes of Good Agricultural Practices – to be established as part of the Nitrates Directive- become compulsory in NVZs (see Chapter 5.2).

Countries such as Germany, Denmark, Finland, Malta and Slovenia decided to designate their whole territory as NVZs. Several countries such as France and Spain opted for a regionalised approach where only catchments meeting the requirements described above are designated. In those countries that opted for a regionalised approach, the coverage of NVZ ranges between 76.2% (Belgium) and 4.4% (Portugal) with an average of 28.6% of their state territory (EC 2013b). Designation of NVZs should be primarily based on existing nitrates concentration and the risk posed by agricultural activities on future concentrations. In addition, legal cases indicate that regions more susceptible to eutrophication through phosphorous should also be considered in the designation of NVZ (EC 2009).

Under the WFD, eutrophication abatement is taken into account in River Basin Management Plans (RBMPs). NVZs under the Nitrates Directive and Sensitive Areas under the UWWTD are recognised as Protected Areas under the WFD. Whenever pressures addressed by these directives are present, the list of water bodies subject to WFD Programme of Measures should be coherent with these designations. However, additional measures may be necessary to achieve the environmental objectives of the WFD (i.e. good water status or potential at water body level). Management within the MSFD is according to the Member State's so-called Exclusive Economic Zones. In some cases e.g. in Spain, the management is split into additional sub-units. The MSFD mostly relies on the WFD for tackling land-based sources of nutrient pollution.



**Implementation timeline:** The WFD sets in place an iterative policy cycle of RBMP development, implementation and revision every 6 years from 2009 to 2027. The UWWTD does not set an iterative planning framework, but sets deadlines between 1998 and 2005 (until 2023 for Member States which have accessed the EU more recently), depending on the size of settlements and the sensitivity of the receiving water body. The designation of sensitive areas has to be reviewed every four years based on an eutrophication and surface water monitoring. Nitrates Action Programmes under the Nitrates Directive should be revised every four years, which leave substantial possibilities for an integrated implementation with the WFD. The MSFD establishes a 6-year cyclical approach to the development, implementation and revision of Programmes of Measures, starting 2015. The policy cycles of the WFD and MSFD are synchronised, which provides opportunities for co-ordinated implementation.

The different approaches in the European directives with regards to measures are highlighted in chapter 5.2.

### 2.3 National policies for eutrophication abatement

Eutrophication abatement policy at European level is closely intertwined with the development of **policies at the national level**, which in part started much earlier. Already in 1959, for example, the neighbouring regions of Lake Constance founded the Commission for the Protection of Lake Constance and committed themselves in 1967 to reduce phosphorous releases into the lake by constructing sewage collection and treatment facilities. Since 1985, Denmark has implemented a set of measures to reduce nitrogen and phosphorus loads by 50% in order to combat the eutrophication of coastal waters.

Nearly all European countries have taken policy measures for tackling eutrophication, although there is a wide diversity of approaches as presented by the approaches in case studies selected for this report. Some focused on urban and industrial emissions (e.g. Lake Constance agreement) whereas others focused exclusively on agriculture (e.g. Finnish Agri-Environmental Programme). Yet others have taken more inter-sectoral approaches between agricultural, urban and industrial sectors from the onset (e.g. Danish Action Plan) while others have progressively integrated new sectors with time (e.g. agriculture in the Lake Constance agreement).

It is difficult to disentangle the influence of national and European initiatives in the development of the policy framework for eutrophication abatement. In the case studies reviewed, there is often a close interaction between national and European levels. The Rhine Action Programme for example was an early and ambitious initiative outside the strict realm of the European Community regulatory environment, but it clearly also relied on European legislation (see textbox above). Another example is the Danish Action Plan for the Aquatic Environment. The first Action Plan was established before the enactment of the NiD, covering already part of the later Directives aims. The second review of the Danish Action Plan for the Aquatic Environment was accepted as the Aquatic Action Programme under the NiD. Its third review is linked with the Danish WFD RBMPs, and in part, funded through the Common Agricultural Policy (CAP). The interaction between EU Policies and the Finish Agri-Environmental Programme is even more complex. The first Finish Agri-Environmental Programme was enacted in 1995, and thus after the NiD came into force. Nevertheless, it already covered several NiD's objectives although it was not designed as the national Nitrate Action Programme. Its subsequent development presented also a blend between EU water and agricultural policies.

## 3 Objectives and targets for eutrophication abatement

### 3.1 Characterising objectives and targets for eutrophication abatement

#### 3.1.1 *The process of setting objectives and their importance for policy implementation*

The policies and legislation described in chapter 2 have the common objectives to reduce eutrophication and thereby securing healthy ecosystems, safe bathing or drinking waters. Various specific qualitative objectives in line with this common objective are specified for various policies and for the different major river conventions and sea conventions in chapter 2, tables 2.1 and 2.2 above, as well as in the case studies in the appendix. These objectives define the direction we want to head towards. However, to implement the different policies in single river basins or in single water bodies, these qualitative objectives must be translated into quantitative targets, such as nutrient concentration targets (chapter 3.2) and to nutrient load reduction targets (chapter 3.3). Setting qualitative objectives and translating them to quantitative targets are vital to reduce eutrophication of European waters, because they provide both direction and the level of ambition for action. The objectives are also important because they represent a common understanding of the acceptable situation both in qualitative and in numerical terms. Without such objectives and targets it is impossible to know whether potential nutrient reduction measures would be sufficient to provide recovery from eutrophication. Once the targets have been quantified, the distance to target can be monitored by eutrophication indicators (chapter 4) and translated into nutrient concentration reductions needed and further to emission reductions needed. The mix of nutrient reduction measures can then be planned and evaluated to assess whether they are sufficient to reach the target (chapter 5).

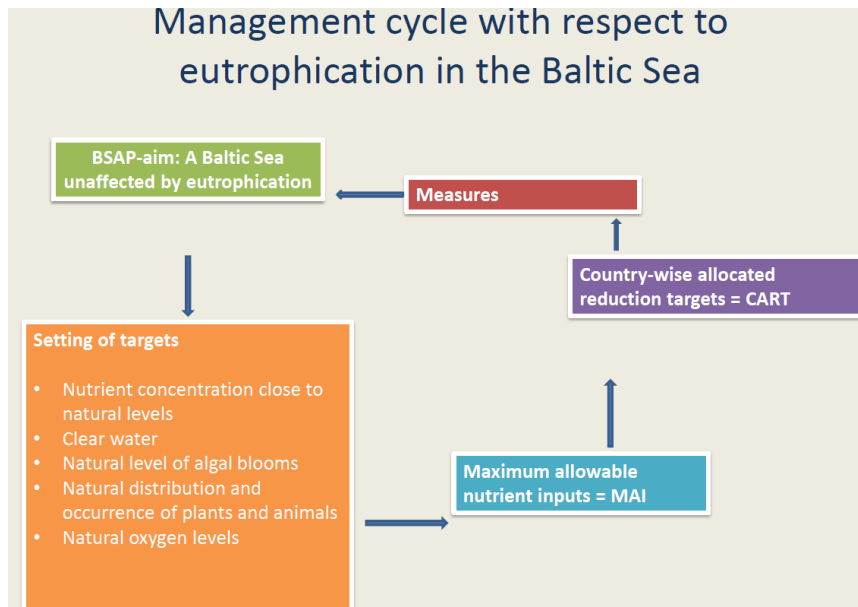
Nutrient concentration targets, also termed nutrient standards are important management tools, because they are quantitative targets to which the current conditions can be compared. In water bodies currently exceeding the nutrient standards, these must be translated into targets for nutrient load reduction in order to make decisions on the extent of measures needed. Such load targets, using the desired state of the water body as a starting point, are both more accurate and ecologically sound than targets set as a relative reduction in loads with respect to some reference year. This is the basis for the movement away from aspirational targets to more ecologically based targets seen in e.g. the HELCOM and OSPAR Conventions (see example in section 3.3).

It should also be taken into account that even if nutrient standard targets are achieved in upstream areas, they may still be exceeded further downstream, especially at a change in water category. The management of nutrients in upstream areas must thus take into account also the load these water bodies supply to downstream areas, and whether that load exceeds the nutrient load target for downstream waters, including lakes and coastal waters.

The Danish action plans on the aquatic environment (case study 1 in the appendix) are good examples of the process of transferring qualitative objectives into quantitative targets. Observed habitat deterioration and rising public and political concern led to the identification of general, policy-based qualitative objectives. In the first action plan these objectives were transferred into aspirational quantitative targets for load reductions, supported by new regulations and suggested specific measures to reach these targets. The effect of the policy was monitored and the targets and measures adjusted in the following plans. In total the action plans have led to substantial reductions in N and P loads to the aquatic environment.

The Baltic Sea Action plan is another good example on how to set nutrient load reduction targets, which is further described in chapter 3.3.2. The process of setting objectives and targets as a basis for nutrient load reduction measures is illustrated in Figure 3.1.

**Figure 3.1: Process of setting objectives and quantitative targets as a basis for nutrient reduction measures in the Baltic Sea.**



Source: HELCOM, 2013c.

### 3.1.2 Setting eutrophication targets in-line with the WFD good ecological status objective

The CIS Guidance on eutrophication (EC, 2009) provides guiding principles defining the WFD concept of ecological status in the context of eutrophication. These guiding principles are based on the WFD normative definitions for the different ecological status classes for nutrient sensitive biota (so called biological quality elements), primarily phytoplankton, phytobenthos and macrophytes. Good ecological status for the algal and plant quality elements will not be achieved if there is a significant probability of undesirable disturbances due to accelerated growth, such as massive algal blooms especially those dominated by toxic phytoplankton, and degradation of blooms causing oxygen depletion. This qualitative description is translated into quantitative targets representing the good/moderate class boundary, using various indicators for phytoplankton, macrophytes (macro-algae and angiosperms) and phytobenthos and their relationship to nutrients (dose-response curves). These boundaries have been developed by the countries and are intercalibrated between neighbouring countries within the same region of Europe (the Geographical Intercalibration Groups) (JRC, 2014). The boundaries are now legally binding (EC, 2013c).

## 3.2 Setting nutrient standards as targets for eutrophication abatement

### 3.2.1 Nutrient standards in rivers and lakes

Chapter 3.1 outlines the importance of setting quantitative nutrient concentration targets as a basis for implementing relevant policies aiming at reducing eutrophication problems in European waters. In the context of the WFD, nutrient concentration targets (termed nutrient standards) should be clearly linked to the good ecological status target for the nutrient sensitive biological quality elements. The CIS guidance on Classification of Ecological Status (EC, 2005b) prescribes that the nutrient concentrations targets should be neither more stringent nor less stringent than required to support the achievement of good status for the biological quality elements and the functioning of the ecosystem.

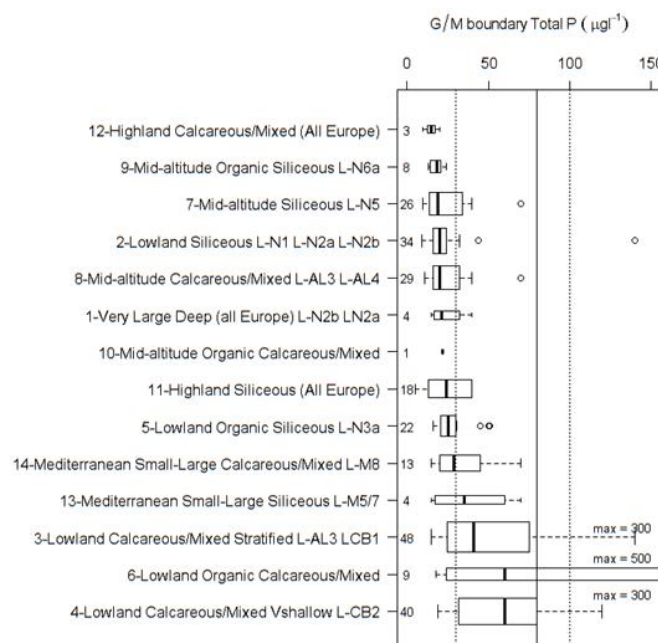
But also before the WFD, nutrient standards have been used for water management in different countries under different legislation, going back more than 50 years. For example, historic nutrient standards for lakes have been a prerequisite for lake management in Europe, North-America and Australia for decades (Cardoso et al., 2009). Emphasis has been given to the Vollenweider model estimating lake specific critical phosphorus concentrations and phosphorus loads (Vollenweider, 1976).

Under the WFD, nutrient standards are generally set by the Member States as part of their ecological status assessment methods. Many differences exist in the standards derived and the methodologies/assumptions used to derive those standards. The methodologies range from dose-response relationships between a nutrient parameter and a numeric expression of a biological quality element or statistical distribution of nutrient concentrations in water bodies classified to different ecological status classes to a statistical division of the whole nutrient gradient or simply expert judgement (Phillips and Pitt, 2015).

Different water categories have different sensitivities to nutrients: the same nutrient concentration does not necessarily have the same effect e.g. in small rivers versus lakes, or in freshwaters versus coastal and/or marine waters. Therefore, when setting nutrient standards it is important to consider the water category, as well as the type of water body within each category. Nutrient standards vary from one type of surface water to the next as the biological, physico-chemical and flow characteristics are also type-specific. In order to enhance comparability of surface waters types across Europe, European broad types have been developed for freshwater (Lyche Solheim et al., 2015). In this way nutrient standards can be compared across Member States.

Data provided in 2014 by 28 countries on WFD boundary values for nitrogen and phosphorus in lakes and rivers, and methods used in deriving the values, were collated and analysed (Phillips and Pitt, 2015). For lakes, there is a large range of boundary values in use, with the majority in the range of 5-100  $\mu\text{g l}^{-1}$ . The total phosphorus standards for lakes are lower in highland or mid-altitude types than in lowland, calcareous lakes, a pattern which is mostly consistent with the type-differences found for the biological good/moderate status boundaries (Figure 3.2 **Error! Reference source not found.**). For most countries, the differences in the good/moderate boundary for comparable types are on average less than  $\pm 20 \mu\text{g l}^{-1}$ .

**Figure 3.2: Range of reported good/moderate boundary total P values for lakes grouped by broad types. Numbers show the number of national types allocated to each broad type. Types ordered by median value of reported boundary.**



Source: Phillips and Pitt, 2015.

For rivers, there was much less evidence of type specific discrimination for river phosphorus boundary values, and the majority falls within a range of 10-500  $\mu\text{g l}^{-1}$ , the most common river total phosphorus standards are found within a range of 100-200  $\mu\text{g l}^{-1}$ . The overall results thus show that the boundary values for lakes are in general lower and also more comparable between countries than for rivers.

The eutrophication response in rivers is more variable than in lakes due to the large variability in river flow. This may explain the larger variation in boundary values between Member States for rivers than for lakes. River phosphorus boundaries were in the range of 10–500  $\mu\text{g l}^{-1}$ , while the range for river nitrogen was 0.25  $\text{mg l}^{-1}$  to 35  $\text{mg l}^{-1}$ . For nitrogen, however, several Member States reported values that appear to be linked to guideline values from the Nitrate Directive (91/676/EC), the Drinking Water Directive (80/778/EC) and Surface Water Abstraction Directive (75/440/EC). These are unlikely to support WFD good ecological status. The same is true for the highest nutrient standards reported for both phosphorus and nitrogen both in rivers and in lakes.

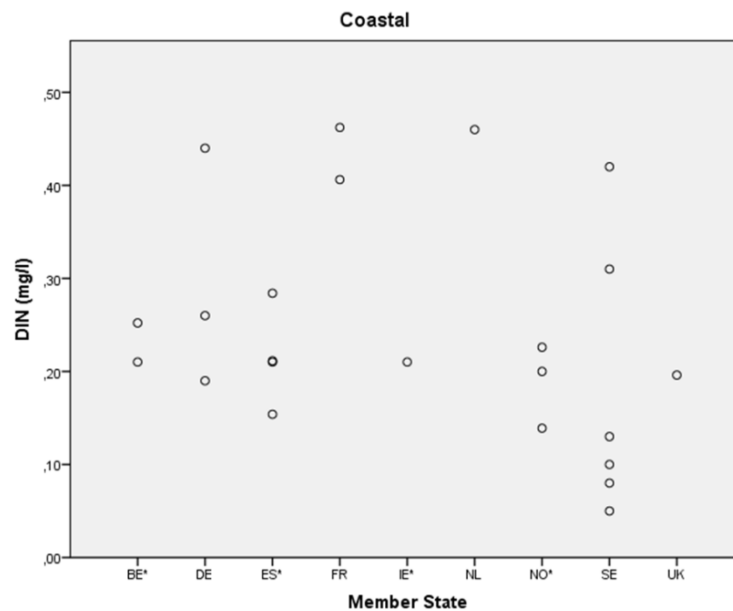
If nutrient standards are used to set the nutrient targets to achieve good ecological status, they will also probably be used as a basis to estimate nutrient load reduction in the river basin management plans. The large variations found in nutrient standards among countries for comparable types of water bodies, in particular for rivers, suggest that there is a need for harmonisation of methods and assumptions at the European level on how to set the nutrient standards, as well as on how they are used for water management. This is particularly important in transboundary river basins.

### 3.2.2 *Nutrient standards in coastal and marine waters*

For transitional, coastal and marine waters a comparison of nutrient standards has proven much more challenging since there was a large heterogeneity in the nutrient parameters assessed by Member States (Dworak et al., 2016). While some assessed the dissolved nutrients (inorganic nitrogen, phosphate) others assessed total nutrients (total nitrogen, total phosphorus). In addition, the assessment time (summer, winter or all year round) varied between Member States and there were also differences in the statistic used for the assessment (mean, median or 90th percentile). These differences did not only exist between Member States but often also within a Member State between transitional, coastal and marine waters and also within common types. For example, in the Baltic Sea, Sweden assesses DIN in summer and winter using mean methods in all three water types while Germany assesses total nitrogen year round using median methods for coastal and marine waters. The rest of the Member States in the Baltic Sea either assess different parameters or monitor the parameters at different times of the year. Where comparisons between nutrient standards used by Member States could be made these standards showed generally wide ranges (Figure 3.3).



**Figure 3.3: Comparison of good/moderate boundary values used in nine member states for winter DIN in coastal waters in the North East Atlantic.**



**\*Notes:** values for BE, IE, ES are taken from Common Procedure for the Identification of the Eutrophication Status of the OSPAR Maritime Area (Reference number: 2013-8).

**Source:** Dworak et al., 2016.

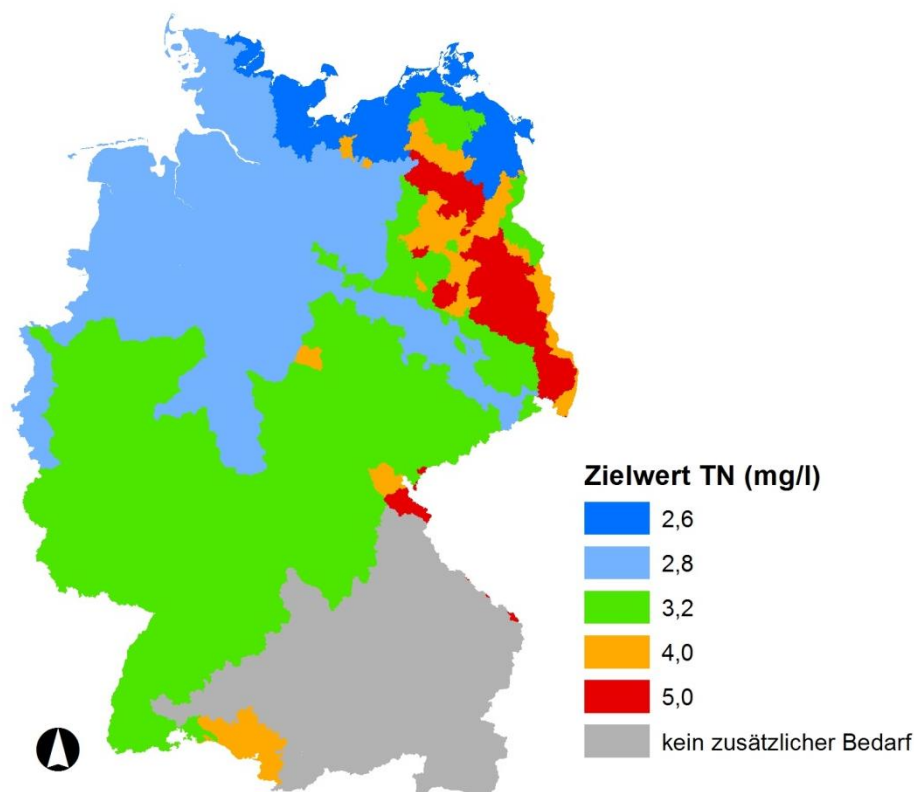
Concerning the methods used to derive nutrient standards most often a mixture of approaches was used and while expert judgment played an important role, it was predominantly used in combination with other, more quantitative approaches (use of existing sites with minor disturbance, historical data and information, modelling). Interestingly, and by contrast to the situation in rivers and lakes, many Member States, in particular in the North East Atlantic and the Baltic Sea have used historic riverine nutrient inputs or historic nutrient concentrations to derive nutrient reference conditions rather than pressure-response relationships with biological quality elements. The reason for applying this approach is a general lack of current near-pristine conditions in coastal and marine waters as well as a lack of historic data on biological quality elements that go far enough back to represent near-pristine conditions. The further the historic nutrient concentrations go back in time the more they were derived by modelling rather than looking at time-series of in-situ data. With respect to the historic conditions, it is interesting that even within a region and between neighbouring Member States there have been very different historic years used to base reference conditions upon (e.g. 1880, 1900, 1930, 1950s, 1960s). While this might be due to data availability, it also appears that there are very different notions among Member States of what constitutes water quality conditions not yet affected by eutrophication. Based on these historic reference conditions good/moderate nutrient boundaries have often been derived by adding an “acceptable deviation” to the reference conditions. Mostly, 50% have been used as a deviation since this percentage reflects the high natural variability and the “slight deviation” from reference conditions as demanded by the WFD. However, a number of Member States have also used much higher deviations (up to 300%) (Dworak et al., 2016).

These described large differences in methodologies are the main reason for the observed large heterogeneity in nutrient standards especially in rivers and coastal waters, and will lead to largely diverging nutrient load reduction targets based on these standards. Hence further harmonisation of nutrient standards is of great importance for arriving at pan-European nutrient abatement efforts that are characterised by a comparable level of ambition.

### 3.2.3 Aligning nutrient boundaries for freshwater and saline waters

Waterborne nutrient inputs should be managed using nutrient reduction targets that enable the achievement of good status for all surface water bodies, including rivers, lakes, transitional, coastal and marine waters. Hence these reduction targets need to be driven by the water category that is most susceptible to nutrient inputs, which are often transitional and coastal waters. So far, few Member States have addressed this issue. Often, nutrient standards for freshwater and saline waters are set separately using different approaches and there is little communication between experts on these two different subjects. Furthermore, Member States use different nutrient parameters in freshwaters versus saline waters (Dworak et al., 2016). One “best practice” example in this respect comes from Germany where a “management target” for the concentration of TN in rivers was set at the limnic-marine border that allows for the achievement of good status in transitional, coastal and marine waters (2.8 mg/l TN for rivers entering the North Sea and 2.6 mg/l TN for rivers entering the Baltic Sea; Figure 3.4) The target was then translated to inland waters by using the catchment model MONERIS and by considering retention. Maximum allowable TN concentrations upstream have been calculated that should be considered in the River Basin Management Plans (Figure 3.4). Based on the ‘management targets’ for TN, nutrient load reduction targets have been calculated considering freshwater discharge. They range between 30% (14 440 t) for the river Weser and 49% (4413 t) for the Schlei/Trave (LAWA, 2014).

**Figure 3.4: Average annual total nitrogen (TN) concentrations required for German rivers entering the North Sea or Baltic Sea to achieve good status in the German Baltic and North Sea coastal and marine waters.**



**Notes:** “Zielwert” = “management target” for TN in mg/l. For white areas, there are no nutrient reduction requirements to achieve good status in coastal and marine waters.

**Source:** LAWA AO 2016 “Empfehlung zur Übertragung flussbürtiger, meeresökologischer Reduzierungsziele ins Binnenland”. LAWA AO Geschäftsstelle Magdeburg

### 3.3 Use of quantitative nutrient load reduction targets

Nutrient load reduction targets are the ultimate requirement to achieve recovery from eutrophication. Such targets are the direct basis for planning measures that will be sufficient to bring the eutrophied waters back to good ecological status in the WFD context and also to achieve the aims of several other policies (e.g. NiD and UWWTD). The nutrient load reduction targets should be linked to the nutrient concentration targets in order to become sufficient and appropriate for recovery.

There are several ways to translate nutrient concentration targets (chapter 3.2) into nutrient load reduction targets: one must either calculate a concentration reduction target, based on the difference between the observed concentrations and the target concentrations, and recalculate this into a load, or the target concentration is recalculated into a load and subtracted from the observed load to get to the load reduction target. In both cases there is a need to establish a link between loads and concentrations, i.e. to calculate the effects of the incoming loads on the concentrations in the water body. The calculation will also be water body specific, as it must take into account the hydrological and morphological properties of the water body receiving the load.

Models can support this process. These may be complex dynamic models or simpler empirical models, depending on the purpose of the modelling and the data and resources available. The choice of the model will also depend on the water category and the complexity of the system. For rivers and streams, the load-concentration relationship is mainly a result of the discharge, but sedimentation and in-stream nutrient conversion processes also affect the final concentration, especially in larger rivers. For lakes, the internal processes become increasingly important, depending on lake residence time. This is also the case for transitional, coastal and marine waters, and here also mixing effects with nearby waters through currents and upwelling must be taken into account.

To achieve the nutrient load reduction targets a detailed nutrient pollution account must be calculated, including a source apportionment to identify the different emission sources. From this pollution account the most cost-efficient nutrient load reduction measures can be taken. The nutrient in focus in freshwater eutrophication is mainly phosphorus, as that is most often the limiting nutrient for primary production, while nitrogen is the main focus in reducing eutrophication in coastal and marine waters

The below examples illustrate different approaches for translating nutrient concentration targets into nutrient load reduction targets, for different water categories and at different scales.

#### 3.3.1 *Nutrient load reduction targets in rivers and lakes*

With the implementation of the WFD and the requirement to define a programme of measures for a large number of water bodies there is a need for simple, yet reliable tools to estimate the nutrient load reduction target as a basis for measures to reach the WFD good ecological status or other policy objectives. Below are some examples for rivers and lakes on various applied methods and tools.

A methodological study of how to select economically justifiable measures to reach WFD objectives was carried out using a sub-catchment of the River Dee in Scotland as case study (Martin-Ortega et al., 2015). A concentration reduction target was set based on distance to the WFD nutrient standard for phosphorus. This was translated into a load reduction target using the catchment model INCA-P (Wade et al., 2002; Wade et al., 2007). The model routes water and nutrients through the terrestrial and aquatic compartments of the catchment and estimates stream loads and concentrations. The model was also run with the different potential mitigation measures, and the effect evaluated both in terms of the concentration and load reduction targets. This was then followed by an economic analysis.

Another example is the far larger Danube catchment. Here target loads are set to ensure that the Black Sea ecosystems could recover to conditions similar to those observed in the 1960s, representing river loads under low pressures. According to the information available, they are ca. 250 kt/yr DIN, ca. 300 kt/yr TN and ca. 20 kt/yr TP (Adam Kovacs, ICPDR Secretariat, pers. comm. 2016). The target loads as

well as the current nutrient loads are based on the MONERIS model (Behrendt et al., 2005; ICPDR, 2015, Annex), which apply the same principles as the INCA-P catchment model used for the River Dee in Scotland described in the paragraph above.

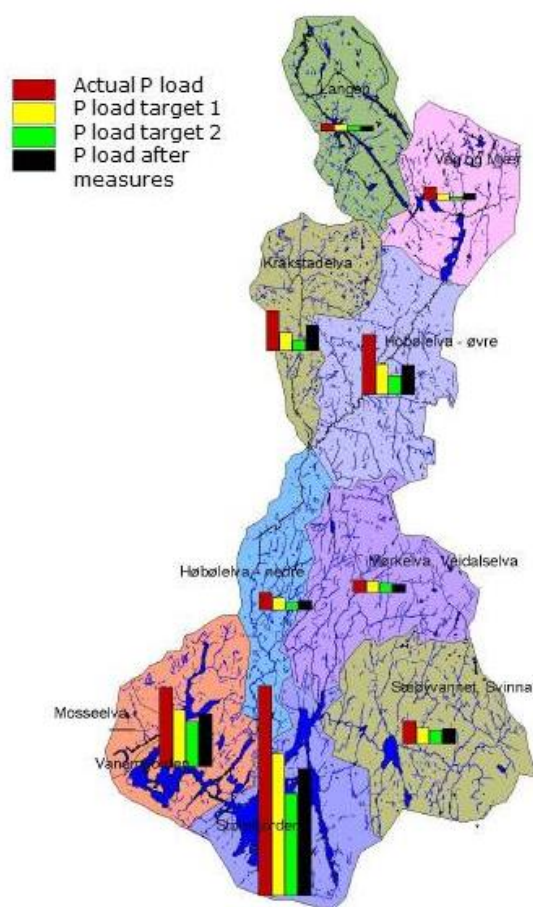
Mean total nutrient emissions in 2009-2012 in the Danube river basin sum up to ca. 610 kt N/yr and ca. 39 kt P/yr (annex to the 2<sup>nd</sup> RBMP for the Danube, ICPDR, 2015), showing that measures are still needed to achieve the target load for nitrogen (see more information in chapter 3 and in the case study description in the annex). Whether these target loads are sufficient to achieve good ecological status in all the Danube river water bodies is unclear.

For lakes, being separate units usually situated in one country only, there is no joint international approach such as for the much larger marine areas. For both rivers and lakes, the nutrients derive mostly from the upstream catchment. For highly eutrophied lakes, however, also the lake sediments are a nutrient source, causing internal loading. Below are some examples of calculation of nutrient load reduction targets for specific lakes, as part of a management strategy. See also the Lake Constance case study in the appendix.

One example of a relevant tool to estimate nutrient load reduction target is suggested by Kotamäki et al. (2015). The lake load response (LLR) tool calculates the nutrient load reduction that is needed to reach good ecological status in lakes in terms of nutrients and chlorophyll a concentration. This is done through combining an empirical nutrient retention model with a statistical chlorophyll a model. The tool also estimates the uncertainty of the calculations, which is particularly important when the amount and quality of input data is limited. As the authors point out, the tool has several shortcomings, and is to be developed further. The tool is also best suited for shallow lakes.

Another example is the nutrient load reduction targets made for the Norwegian Lake Vansjø, which was not assessed in detail for this report but the case study demonstrates clearly the use of load reduction targets in eutrophication abatement plans (Lyche-Solheim et al., 2001). The qualitative objectives agreed across all sectors and organisations in the catchment were that the main lake should become suitable as a drinking water resource, and that all lake water bodies in the river basin should be suitable as bathing water and for fishing, as well as for irrigation. These objectives were translated to quantitative targets using an empirical model relating total phosphorus concentration to algal blooms. There were also a series of other quantitative targets agreed for nitrogen, Secchi depth, oxygen, coliform bacteria, suspended solids and proportion of cyanobacteria in the phytoplankton. The phosphorus concentration targets were translated into maximum acceptable load targets using the model by Larsen and Mercier (Larsen and Mercier, 1976), taking retention into account. The phosphorus load reduction target was set by subtracting the estimated actual load from the target load. For each sub-catchment, the actual load was estimated both by a nutrient run-off model combined with point source emissions data, as well as by using observed nutrient concentrations and river discharge. The abatement measures focused on reduced surface runoff from agriculture, but also improved wastewater treatment, and their effect was estimated to be sufficient to achieve the needed load reduction. Implementation was done in two phases, first reducing the major phosphorus sources and then dealing with the remaining measures. The analysis revealed that the programme of measures was not quite sufficient to meet the needed load reduction, although significant improvement occurred in terms of reduced phosphorus load (Figure 3.5), and reduced cyanobacterial blooms in the Western basin.

**Figure 3.5: Phosphorus loads per sub-catchment in the Lake Vansjø catchment.**



**Note :** The bars in each sub-catchment show on a relative scale the total phosphorus load in the year 2000, the target load for the first phase and the second phase of implementation (P load target 1 and P load target 2), as well as the load estimated when all the load reduction measures have been implemented (P load after measures).  
**Source:** Lyche-Solheim et al., 2001.

### 3.3.2 Nutrient load reduction targets in coastal and marine waters

Considering the history of nutrient load reductions in these waters there has been a development away from aspirational political targets that were not linked to ecosystem requirements (e.g. 50% reduction) towards targets based on achieving a good eutrophication status as described by a number of eutrophication status parameters. Back in the early 1980s, when eutrophication problems were large and apparent, resulting in e.g. frequent algal blooms and fish kills due to large areas suffering from oxygen deficiency, the North Sea riparian countries held their first International Conference on the Protection of the North Sea in 1984 in order to tackle the large releases of various harmful substances through rivers, direct discharges and dumping of waste at sea. The 1<sup>st</sup> North Sea Conference agreed to investigate the relevance of nutrient inputs for the North Sea. Already at the 2<sup>nd</sup> North Sea Conference 1987 in London the North Sea countries decided “to take effective national steps in order to reduce nutrient inputs into areas where these inputs are likely, directly or indirectly, to cause pollution” and to “aim to achieve a substantial reduction (of the order of 50%) in inputs of phosphorus and nitrogen to these areas between 1985 and 1995”. Since there was no regional coordination platform or implementation mechanism associated with the North Sea Conferences OSPAR has taken up the 50% reduction goal and has regularly reported against its achievement (PARCOM Recommendation 88/2 on the Reduction in Inputs of Nutrients to the Paris Convention Area). In 2005, six out of nine reporting OSPAR Contracting Parties met the 50% reduction target for phosphorus while for nitrogen only Denmark achieved a 50% reduction by 2003 and Germany and the Netherlands achieved reductions close to 50% (Table 3.1). The substantial reductions in phosphorus inputs were largely achieved by



addressing point sources (e.g. by upgrading wastewater treatment plants, introducing phosphate free detergents). Since nitrogen stems to a larger proportion from diffuse sources, in particular agriculture, less progress was made in reducing these inputs. The reporting of OSPAR Contracting Parties against the 50% reduction targets ceased in 2005. Since there is to date no agreement in OSPAR about new quantitative nutrient reduction targets the reporting on nutrient discharges from different sources could not be revived yet.

**Table 3.1: Discharges/losses (in tonnes) of nitrogen and phosphorus from anthropogenic sources in 1985 and 2005 for OSPAR Contracting Parties of the Greater North Sea region and the reductions achieved.**

Country	Nitrogen			Phosphorus		
	1985	2005	% Reduction	1985	2005	% Reduction
Belgium	93 135	61 532	34	17 458	6 889	61
Denmark	NI	NI	NI	NI	NI	NI
France	NI	NI	NI	NI	NI	NI
Germany	806 000 <sup>1</sup>	418 020	48	65 790 <sup>1</sup>	17 390	74
Ireland	NI	39 275	NI	NI	2 506	NI
Netherlands	168 163 <sup>1/2</sup>	92 203 <sup>2</sup>	45 <sup>2</sup>	30 895 <sup>1/2</sup>	6 959 <sup>2</sup>	77 <sup>2</sup>
Norway	49 891 <sup>1</sup>	33 712	32	1 834 <sup>1</sup>	683	63
Sweden	47 540	38 245	20	1 626	1 320	19
Switzerland	39 749 <sup>1</sup>	28 656	28	3 004 <sup>1</sup>	1 468	51
United Kingdom <sup>3</sup>	76 000 <sup>*</sup>	61 500	19 <sup>*</sup>	4 300 <sup>3*</sup>	3 150	27 <sup>*</sup>

NI: No Information

<sup>1</sup> The figure for 1985 in the 2003 data report was different (OSPAR, 2006a).

<sup>2</sup> These figures include overflows and discharges of collected untreated rainwater and sewage from yachts and inland vessels.

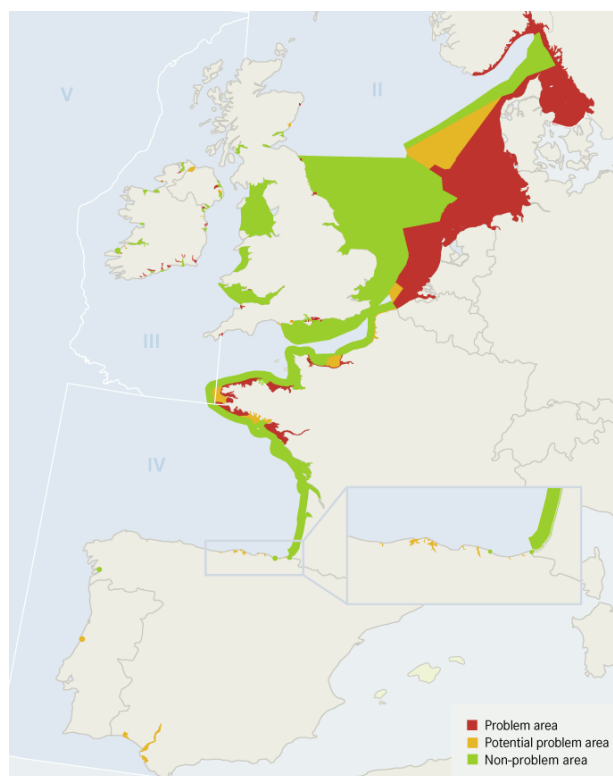
<sup>3</sup> These are the values for riverine and direct discharges from the sampling regions in which the United Kingdom's problem and potential problem areas are situated.

<sup>\*</sup> The United Kingdom data are for 2002; reduction figure is the reduction between 2002 and 2005.

**Source:** OSPAR, 2008.

OSPAR changed its strategy with respect to eutrophication in 2010 and agreed to “by 2011, quantify the reduction of nutrients to the maritime area required for individual eutrophication problem areas to achieve non-problem area status” (OSPAR North East Atlantic Environment Strategy). This agreement paved the way for the scientific derivation of country-based nutrient reduction targets based on a number of eutrophication parameters describing a non-problem area status with respect to eutrophication (mainly nutrients and chlorophyll-a). Nevertheless, no concrete nutrient reduction targets have been agreed until spring 2016 and the discussion in OSPAR is cumbersome despite an expert group of modellers that is prepared to derive nutrient reduction targets for the Greater North Sea by their appropriate models. There are a number of reasons for this slow progress. Firstly, model results have shown that there are still large reductions in nitrogen and phosphorus necessary to move eutrophication problem areas to the status of non-problem areas (partially up to 75% for nitrogen and 70% for phosphorus) while at the same time the potential for effective measures at point sources is almost exhausted and diffuse sources have proven difficult to tackle (OSPAR, 2013). Secondly, due to the strong anticlockwise circulation of water masses in the North Sea countries like the UK and France, although discharging large amounts of nutrients to the sea, have identified only small localised eutrophication problems (mainly in enclosed estuaries) and therefore see no immediate need to reduce nutrient inputs into their waters. Other countries such as the Netherlands, Germany and Denmark are substantially influenced by these transboundary nutrient transports leading to eutrophication problems not only in coastal waters but also far offshore (Figure 3.6). Hence, there is no notion of a “common eutrophication problem” in the OSPAR Greater North Sea Region and it is proving difficult to find a way forward that adequately takes the different interests and needs into account.

**Figure 3.6: Eutrophication status of the OSPAR Convention Area in the period 2001–2005.**



**Note:** Green= Non-problem areas with respect to eutrophication, orange = potential problem areas, red = problem areas. All other areas were screened in 2001 for obvious non-problem areas and were subsequently not assessed.

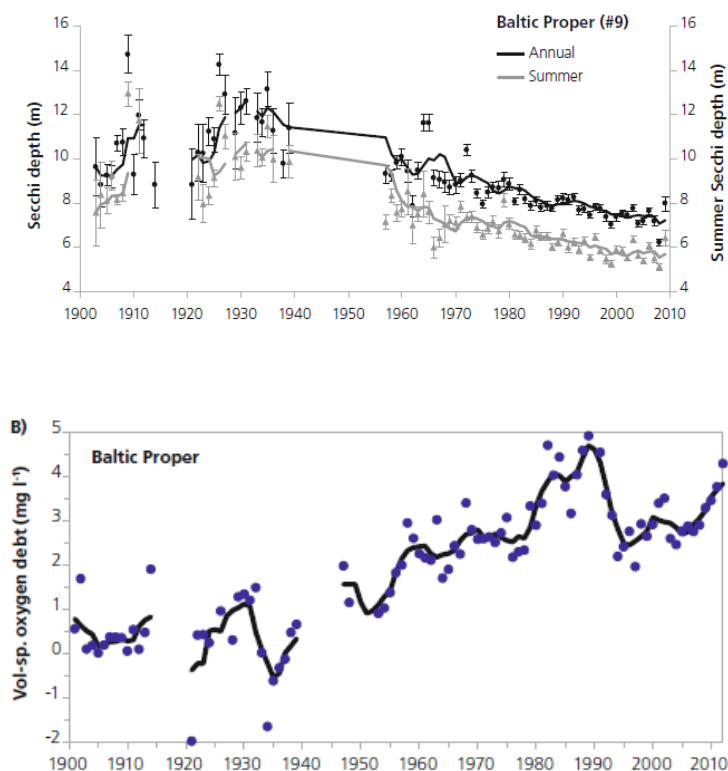
While eutrophication is strongly influenced by the hydrodynamic situation the differences in eutrophication status of adjacent areas are also caused by OSPAR Contracting Parties using national assessment levels which are not regionally harmonised.

**Source:** OSPAR, 2010.

It is interesting to compare this with the situation in the Baltic Sea. HELCOM was initially lagging behind in the process of setting nutrient reduction targets, but has quickly become the forerunner. In 1988 at the Ministerial Meeting in Helsinki HELCOM took on the aspirational 50% reduction target for nutrient inputs initiated by the North Sea Conferences. Already in the Baltic Sea Action Plan, adopted in 2007, quantitative country-wise nutrient reduction targets based on a modelling approach were agreed. These targets have undergone a thorough scientific revision starting in 2010. The eutrophication parameters nutrients, transparency (Secchi depth), chlorophyll-a and oxygen served as a basis for deriving quantitative nutrient reduction targets. In a major Baltic Sea wide data mining exercise supported by ensemble modelling, historic time series of these parameters were put together going as far back as to the 1900s (HELCOM 2013c; Figure 3.7). These time series were investigated for change points (abrupt changes in the parameter level caused by anthropogenic impacts) to identify the appropriate target levels (GES/sub GES boundaries). Relevant change points were in particular identified for oxygen and Secchi depth. Until around 1940, the Baltic Proper basin had clear water with a Secchi depth of around 10 m. The oxygen debt, defined as the missing oxygen relative to a fully saturated water column, was low until around 1940. Both parameters showed a rapid degradation after 1940 caused by excessive anthropogenic nutrient inputs. Therefore, the change point identified was used to describe the period before 1940 as a period where the Baltic Sea was unaffected by eutrophication and this was used as the basis for eutrophication target setting (HELCOM, 2013a). The concept of target setting based on change points developed by HELCOM was a challenge, but has a strong scientific basis and is a concept agreed by all Contracting parties. Nevertheless, the HELCOM approach also has a major weakness, because targets are only derived for open sea waters (>1 nautical mile) and not for coastal waters. The harmonisation of coastal and open water targets that have been set using different approaches, is left to the HELCOM

Contracting Parties, is scientifically challenging and only few have achieved such a harmonisation (see for instance Schernewski et al., 2014).

**Figure 3.7: Historic time series of annual and summer Secchi depth (in m) and of volume specific oxygen debt (in mg/l) in the Baltic Proper basin of the Baltic Sea.**



**Source:** HELCOM, 2013c, 2013d.

Based on the target levels set by HELCOM for Secchi depth, inorganic nitrogen and phosphorus concentration, chlorophyll a and dissolved oxygen for different basins the coupled physical-biogeochemical model BALTSEM was applied to identify the maximum allowable inputs (MAI) to the basins that do not lead to an exceedance of the eutrophication targets. The MAI were then used as the basis to derive country-specific nutrient reduction targets. The basic principle applied was the “polluter pays principle”, meaning that the reduction requirements to a sub-basin are based on the nutrient inputs of each HELCOM country. Besides waterborne nutrient inputs the revised nutrient reduction targets of the BSAP also consider airborne nitrogen deposition from HELCOM Contracting Parties as well as from other EU countries, third countries and from shipping. Concerning the waterborne nutrient inputs, contributions from upstream countries were explicitly considered and attributed to the respective polluters (countries). In this way the BSAP is able to assign quantitative nutrient reduction targets to all relevant polluters. The revised nutrient reduction targets of the BSAP were decided upon by ministers at the HELCOM Ministerial Meeting in 2013.

HELCOM has, as the first regional seas convention, been successful in formulating quantitative nutrient reduction targets, because nearly the whole Baltic Sea is seen as eutrophied and hence there is the notion of HELCOM Contracting Parties that they have a common problem necessitating actions. One important component of the success of the BSAP nutrient reduction scheme is a thorough follow-up based on regular reporting of HELCOM Contracting Parties of their waterborne and atmospheric nutrient inputs. The latest follow-up of the “country-allocated nutrient reduction targets” (CART) based on nutrient input data up to 2012 shows that no Contracting party has so far reached the nutrient reduction targets (see HELCOM CART follow up at <http://www.helcom.fi/baltic-sea-action-plan/progress-towards->

reduction-targets/). While countries such as Denmark, Germany and Sweden are on a good way having sustained nutrient reductions there are countries such as Latvia and Russia having increases in nutrient inputs in recent years. Poland, whose whole territory drains into the Baltic Sea Proper and has therefore been assigned the highest nutrient reduction targets, has so far only accepted the CART as indicative targets and are still holding national consultations. Another major challenge for the implementation of the BSAP is the fact that nutrient reduction targets derived under the WFD and the BSAP do not necessarily match, thereby impeding the development of a coherent nutrient management policy. The reason for this disagreement lies in the different approaches used to derive target levels for eutrophication determinants under the WFD and the BSAP so that coastal and offshore waters might require different nutrient reductions to achieve good status (see above).

## 4 Indicators to measure change in state and pressures

### 4.1 Indicators as management tools

Indicators are a widely used instrument for monitoring the eutrophication status and to document the distance to target and the progress of policy implementation over time. Eutrophication indicators comprise pressure indicators (nutrient emissions and loads), state indicators (nutrient concentrations in the water) and impact indicators describing the health of aquatic ecosystems (such as biological indicators, oxygen and chlorophyll-a concentration, or Secchi depth) (Table 4.1). The impact indicators used in connection with eutrophication vary depending on the water category, but they are generally sensitive to nutrients, either directly (biological indicators) or indirectly (physico-chemical indicators affected by biological processes). The indicators are also used as a basis to derive the actual quantitative targets for eutrophication described in chapter 3.

When measures are taken, the pressures should be reduced, the state improved and the impact lowered. The pressure indicators will reveal if the measures taken are actually adequate to lower the pressures according to the nutrient load reduction targets set. The state indicators will show if the lowered pressures have the desired effect. If not, there may be pressures that are overlooked or the processes coupling pressure and state are not correctly represented, suggesting that the load reduction targets may be re-evaluated. And likewise, the impact indicators serve as the final indication whether the measures taken are appropriate and that the links between pressures, state and impacts are correct. If the desired effects are not achieved, the response may be to adjust the measures, or the pressure or state targets. However, there will always be a lag in the response of the state and impact indicators.

The indicators and their use are briefly described and exemplified in the following sub-chapters.

**Table 4.1: Eutrophication indicators for nutrient pressure, state and impact in rivers, lakes, transitional, coastal and marine waters.**

Indicators	Rivers	Lakes	Transitional waters	Coastal waters	Marine waters
<b>Pressure indicators</b>					
Nutrient emissions, nutrient loads	x	x	x	x	x
<b>State indicators</b>					
Phosphorus concentrations (Total-P, ortho-phosphate)	x	x	x	x	x
Nitrogen concentrations (Total-N, NO <sub>3</sub> )	x	x	x	x	x
<b>Impact indicators</b>					
Ecological status (WFD)	x	x	x	x	
Environmental status (MSFD)				x	x
Phytoplankton (chlorophyll-a, biovolume)	x	x	x	x	x
Phytoplankton (community composition, toxic and nuisance algae)		x		x	x
Secchi depth		x		x	x
Macrophytes (lower growing depth)		x		x	
Macrophytes (community composition)	x	x	x	x	
Phytobenthos (benthic algae community composition)	x	x			
Macrozoobenthos (community composition, biomass)	x	x	x	x	x
Bottom water oxygen concentrations		x*		x	x

\*stratified lakes only

## 4.2 Pressure indicators

Nutrient emissions or load indicators are important performance indicators. Load indicators show how much nutrient loads have been reduced and whether the nutrient load reduction targets have been achieved. Nutrient emissions are assessed per sector and provide a direct link to the respective polluters. Data on sectoral nutrient emissions and the need for load reductions in each sector are gained by source apportionment. This is useful to identify the main contributors to the loads and where further measures would be most effective. Most of the case studies presented in the appendix use nutrient load indicators to illustrate policy effectiveness, shown as nutrient load reductions. The development over time also shows the effect of measures taken in particular sectors, e.g. the Danube case study and the Po river basin case study. Pressure indicators are also used to follow up on nutrient load reduction targets and projections, as illustrated by the Rhine case study, showing *inter alia* that the 50% load reduction target for nitrogen was not achieved within the Rhine Action Program until the year 2000.

Considering the historic development of the pressure indicators it is apparent that for decades only nutrient emissions or loads to surface waters have been considered, and only lately efforts have been made to also take atmospheric nutrient emissions into account, as for instance illustrated by the case study on the Baltic Sea Action Plan. In the North Sea and Baltic Sea roughly 25% of the nutrient inputs come from the atmosphere. Hence they are an important nutrient source that should not be neglected when concluding on measures. There is an important distinction made between nutrient loads that enter the sea (which are the product of freshwater discharge and nutrient concentration), which can be measured directly, and nutrient loads at source which are based on the quantification of the discharges/losses into inland surface waters within respective catchment areas and which are modelled using catchment models (also termed watershed models or hydrological models). The latter can distinguish different sources of nutrient inputs, e.g. waste water treatment plants, agriculture, atmospheric inputs, scattered dwellings etc. and are therefore directly relevant for nutrient management. Catchment models are highly complex. The model parameters often cannot be measured directly but need to be estimated. Across Europe, there is a large number of catchment models that have been applied, often providing largely diverging or even contradicting results. This can hamper an effective sectoral nutrient management.

## 4.3 State indicators

Changes in nutrient concentrations are often used to show changes in nutrient pressures or effects of nutrient reduction measures, since there is a direct link between inputs and concentrations that are quantifiable and can also be well described by modelling approaches. In this respect, nutrient concentrations are important performance indicators that indicate to what extent nutrient reduction efforts are successful. This importance contrasts with the fact that under the WFD nutrients are only supporting parameters for the assessment of “good ecological status” and how they have been applied often deviates from the provisions in the WFD CIS Eutrophication guidance (EC 2009). Following the CIS guidance a failure of achieving at least “good ecological status” for nutrients should result in an assessment of the ecological status that cannot be better than moderate (EC, 2009). In practice, it is sometimes the case that a WFD water body reaches good status even if nutrient standards are exceeded, which means that the one-out-all-out principle is not always applied. In other cases the water body might fail to achieve good status due to the nutrient standards being exceeded, but this does not always trigger measures if the biological quality elements are in good status.

In contrast to the WFD the HELCOM “Eutrophication Assessment Tool” (HEAT) assesses nutrients as one group of indicators besides direct and indirect eutrophication effects, and then the three indicator groups are combined by applying the one-out-all-out principle (meaning that the worst assessment result of any of the three groups is taken as the final result). In this way, the nutrient assessment directly impacts on the final assessment outcome. The OSPAR “Common Procedure” deals with nutrients differently. An area can be assessed as a non-problem area with respect to eutrophication even if the

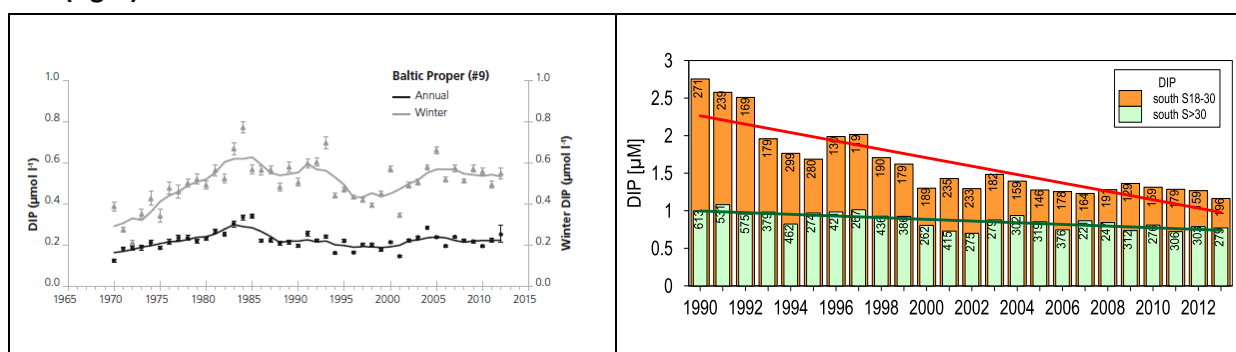


nutrient levels are high. The procedure acknowledges that increased nutrient enrichment in such areas may contribute to eutrophication problems elsewhere through transboundary nutrient transport. This procedure has hampered the derivation of nutrient reduction targets since the model approaches used have so far only been capable to produce trustworthy results for nutrient concentrations while results for biological parameters such as chlorophyll-a are less reliable (see chapter 3.3.2.) (OSPAR 2013).

That nutrient concentrations are a good performance indicator for the effects of nutrient reduction efforts can be seen from the case studies in this report (see appendix). These show reduced nutrient concentrations in response to reduced pressures / implemented measures. In the Danish example, N concentration is shown next to the N load, illustrating the coupling between reduced load and lower concentration (Figure A1.2Error! Reference source not found. in the appendix). The Ebro River example (chapter 5 in the appendix) also illustrates the coupling between measures and effects, but it also shows how measures may have different effects on different nutrient components. In addition, it shows the large variability observed in nutrient concentrations. This illustrates the need for frequent measurements to calculate representative average (e.g. annual) concentrations. This is important to correctly evaluate the overall effect of measures. Knowing the peak level is also important, as it can sometimes be a better indicator of the biological impact than the average concentration.

The Baltic Sea nutrient reduction efforts of the last decades have not yet resulted in a pronounced decline of nutrient concentrations in the sea, possibly due to releases of nutrients from hypoxic sediments. In contrast, in the coastal North Sea areas a significant decline could be detected (Figure 4.1).

**Figure 4.1: Long-term trend in annual (black) and winter (grey) surface DIP concentrations in the Baltic Proper (0-20m). Lines indicate the five-year moving average and error bars represent 95% confidence limits of the means (left). Time series of winter DIP concentrations in the coastal waters of the Southern North Sea and linear trend lines showing significant decreasing trends between 1990 and 2014 (right).**



**Source:** HELCOM 2013d; Draft of the OSPAR Common Indicator on winter nutrient concentrations.

#### 4.4 Impact indicators

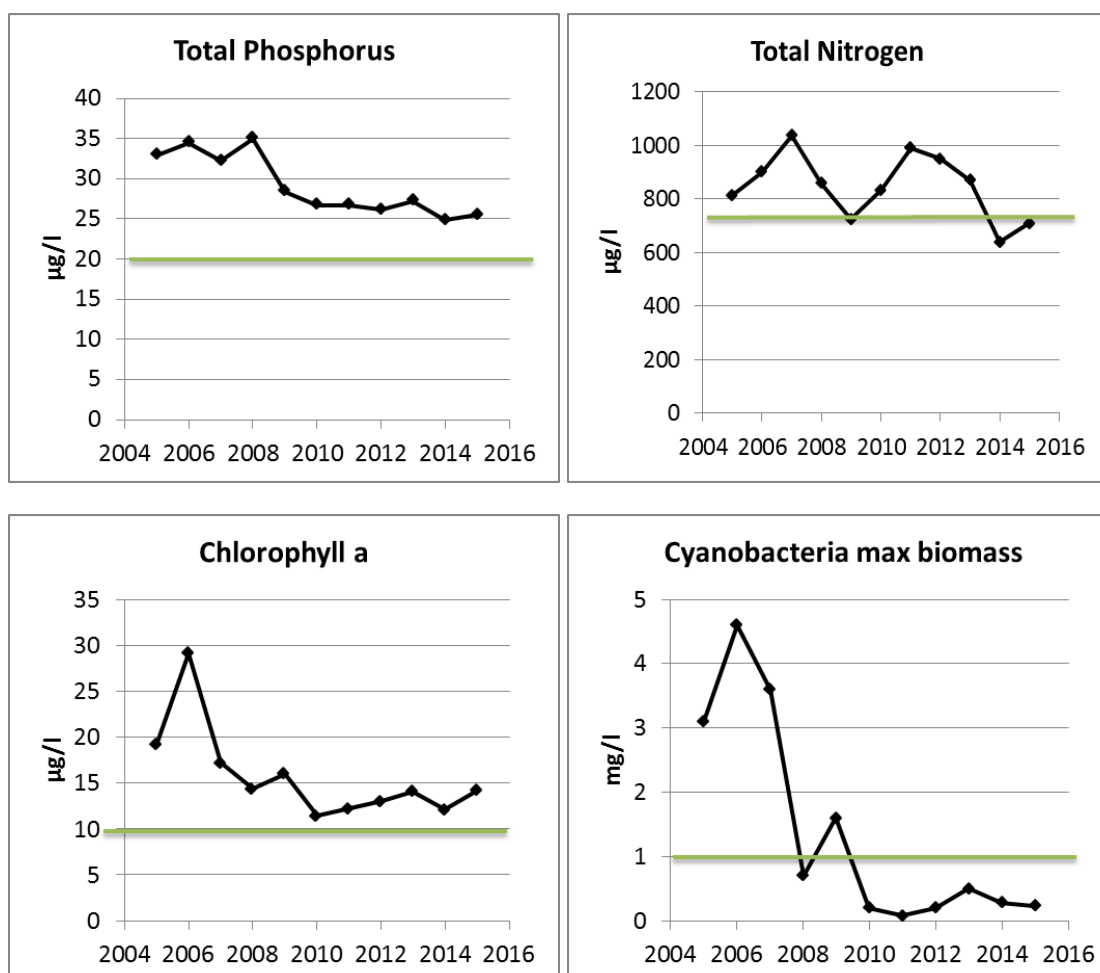
The impact indicators listed in Table 4.1 are used to reveal undesirable ecological disturbances, as described in chapter 4 of the WFD-CIS Eutrophication guidance (EC, 2009), and comprise both biological quality elements (BQEs), representing groups of organisms, biomass indicators like chlorophyll a and physico-chemical indicators like Secchi depth or oxygen, representing the effect of changes in biological conditions (in addition to direct physico-chemical changes). Individual species, genera or higher taxonomic units may also serve as impact indicators in certain contexts, such as cyanobacteria or other nuisance algae. Ecological status, as applied under the WFD, is a combined indicator, including several BQEs, but adjusted by physico-chemical indicators. Some of the latter are state indicators, meaning that ecological status is a combined state-impact indicator. The same is true for good environmental status (GES) with respect to eutrophication under the Descriptor 5 of the MFS. In general, state and impact indicators are frequently used together when the main purpose is to monitor the environmental conditions and the effects of measures.

Impact indicators used in connection with eutrophication should show a clear coupling to nutrient concentration, which is often the basis for the WFD good/moderate boundary for individual BQEs.

Oxygen in bottom waters is frequently used in lakes (see e.g. the Lake Constance case in the appendix) and coastal waters (e.g. Baltic Sea). The same is true for chlorophyll a. Phytoplankton community composition including toxic or other nuisance algae that are clearly associated with undesirable disturbances, are also commonly used as impact indicators, both in lakes and coastal waters. Toxic cyanobacteria in lakes have negative impacts on human health and ecosystem services (ETC, 2010 and literature cited therein; Carvalho et al., 2013). Trends in the development of cyanobacteria will also be important in the coming years to assess combined impacts of eutrophication and climate change, as several papers indicate an increase of cyanobacteria to be expected with climate change (Pearl and Paul, 2012).

The benefit of using state and impact indicators to monitor effect of nutrient load reduction measures can also be seen from the Norwegian lake Vansjø (see sections 3.3.), where a series of nutrient reduction measures were implemented from 2001 to 2005 (Figure 4.2).

**Figure 4.2: Status development towards the target by state and impact indicators in the Vanemfjorden basin of Lake Vansjø. The target is the WFD good/moderate boundary shown as a green horizontal line.**



**Source:** Skarbøvik et al., 2016

The indicators show the progress towards the WFD good status target. The measures have clearly been effective to get rid of the severe cyanobacterial blooms, although there may also be other mechanisms explaining the improvement of chlorophyll and cyanobacteria, such as a doubling of humic content over

the past 15 years. For the state indicator Total-P, more measures may be needed to achieve the target. The progress towards the Total-P target is however slowing down due to more frequent floods causing erosion and transport of P-rich soil particles to the lake. This example also illustrates the need to use a suite of indicators when evaluating the effect of nutrient load reduction measures.

With the implementation of the WFD and MFSD, the use of impact indicators is getting more harmonised, making it easier to compare and assess trends in more indicators, see chapter 4 in the WFD-CIS Eutrophication guidance (EC, 2009). Trends in eutrophication sensitive biological indicators and overall ecological status have already been shown in several case studies. Below (Figure 4.3) is an example of progress towards the WFD good status target in a river water body in the Netherlands. It shows improvement for several nutrient sensitive quality elements, such as phytobenthos, phosphorus and ammonium, but no change in the overall ecological status. This illustrates the necessity to look at the individual elements when identifying the cause of not reaching good status and deciding where further measures must be taken. In this case there does not seem to be a nutrient or eutrophication issue any more, rather it is the specific pollutant zinc that cause the overall moderate status.

**Figure 4.3: Improvements in quality element and overall ecological status of a river water body in the Netherlands.**

Classification Year :			2009		2013		2015	
Data period used :			2006-2007		2010-2011		2012-2013	
			QE Status	WB Status	QE Status	WB Status	QE Status	WB Status
ECOLOGICAL STATUS	Biological Quality Elements	Macrophytes	Good	Moderate	Good	Moderate	Good	Moderate
		Phytobenthos	Good		Good		Good	
		Invertebrates	Good		Good		Good	
		Fish	?		Good		Good	
	Physico- Chemical param.	O2	Good	Moderate	Good	Moderate	Good	Moderate
		Phosphorus	Good		Good		Good	
		NH4	Bad		Good		Good	
		NO3	Good		Good		Good	
	Specific pollutants	Cu	Good	Moderate	Good	Moderate	Good	Moderate
		Zn	?		Bad		Bad	
		2,4-MCPA	Good		Good		Good	

**Source:** van der Molen, 2015.

The nutrient reduction targets of the Baltic Sea Action Plan are mainly based on oxygen debt and Secchi depth. Biological parameters could not be considered due to the lack of historic data e.g. on macrophytes, macrozoobenthos or phytoplankton and due to the challenges still associated in modelling these parameters. This holds true for most efforts in linking state and impact indicators in coastal and marine environments, and implies that distance to target can only be formulated with respect to a very limited number of predominantly physico-chemical eutrophication indicators. The consideration of biological parameters is mainly restricted to chlorophyll-a.

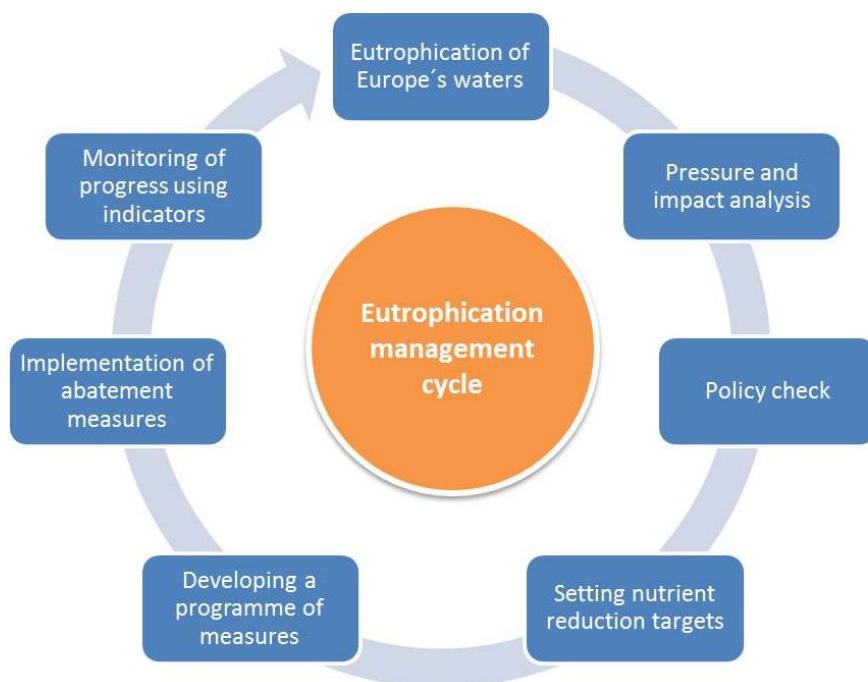
On a regional level HELCOM is regularly assessing chlorophyll-a concentrations, Secchi depth and oxygen (<http://helcom.fi/baltic-sea-trends/eutrophication/indicators/>) and is in the process of developing further impact indicators e.g. for cyanobacterial blooms and spring blooms. The latter two indicators do not require additional in-situ monitoring but are assessed based on satellite images. All HELCOM indicators are assessed using regionally-harmonised assessment levels that have been developed based on a sound scientific approach and with the participation of all Contracting Parties (see chapter 3.3.2). OSPAR has only recently started the development of common indicators and will be assessing chlorophyll-a, Phaeocystis and oxygen concentration in the Greater North Sea in the future. However, due to the lack of regionally-harmonised assessment levels the assessment is mostly based on trends.

## 5 Measures and management planning to reduce nutrient input

### 5.1 Establishment of Programmes of Measures – from quantitative targets to measures

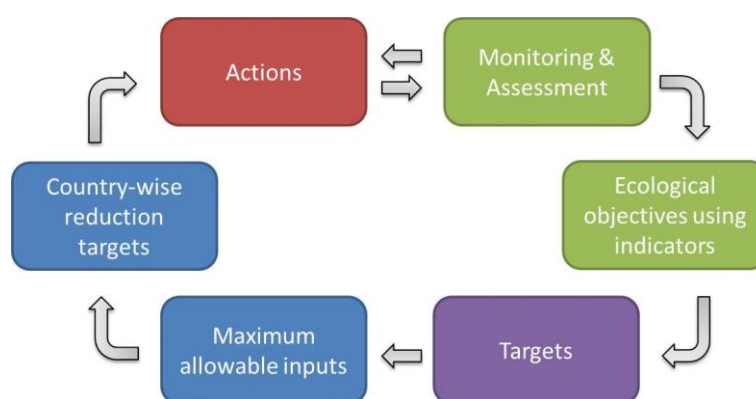
Once the most significant pressures have been identified (chapter 1.2) and nutrient reduction targets have been set (chapter 3), a programme of measures may be established. The program of measures is the central element of an eutrophication management plan that describes the activities to meet the overall objectives and the specific nutrient reduction targets (Figure 5.1). The measures should be both, targeted in terms of their type and extent to ensure that pressures are addressed appropriately and directly linked to the desired ecosystem status (see Baltic Sea Action Plan in the text box).

**Figure 5.1: Schematic representation of the eutrophication management cycle.**



### Box 5.1: Eutrophication management cycle of the Baltic Sea Action Plan

The eutrophication problem in the **Baltic Sea** was addressed by the Helsinki Commission (HELCOM) (see chapter 3.3). The Baltic Sea Action Plan (BSAP) represents an ecosystem-based approach to the management of human activities that incorporates the protection of the marine environment. The action plan puts the ecosystem at the center, by defining a desired status in the future and linking directly measures to the status of the Baltic Sea. The BSAP determines maximum allowable nutrient inputs to each Baltic Sea sub-region if good ecological quality is to be reached and sets country-wise nutrient reduction requirements (see chapter 3.3.2 and case study assessment in the appendix). HELCOM provided in close cooperation with science a catalogue on effective measures to curb eutrophication (HELCOM, 2007, 2013b). The measures take into account the unique environmental conditions and sensitivity of the Baltic Sea.



Eutrophication management cycle of the BSAP (Source: HELCOM, 2013c)

A task force group including all HELCOM members, as well as representatives from non-governmental and governmental organizations, coordinated the development of the actions and measures agreed upon in the BSAP (Pyhälä, 2012). The BSAP requires the countries to develop national implementation programmes, which outline the most cost-effective measures for achieving the targets in each country. The action plan also distinguishes between measures that can be implemented at national level, and measures that can only be implemented at EU level (e.g. Common Fisheries Policy) or globally (e.g. shipping controls).

Joint efforts by the HELCOM contracting parties have had positive results. A significant reduction in waterborne P loads was reached during the last decades, especially from point sources in the catchment area (Andersen et al. 2014). However, the latest follow-up of the “country-allocated nutrient reduction targets” (CART) shows that no contracting party has so far reached the nutrient reduction targets (see chapter 3.3.2) The Baltic Sea is still affected by eutrophication, except the Bothnian Bay and some coastal areas mainly in the north. The policy is working but the environmental objectives have not been reached until today (HELCOM 2013a).

There are various mitigation measures available to reduce nutrient pollution from point and diffuse sources that differ greatly in their specific suitability, cost-effectiveness and implementation perspective. The selection of measures and combination of measures is not an easy task and requires the consideration of technical and financial aspects as well as knowledge on the effectiveness of measures to reduce nutrient inputs into the water. The selection process is therefore to be organized as a structured decision making process (Dietrich and Funke, 2009; Stärz et al., 2016). After a first pre-selection of potential measures from established catalogues (e.g. DEFRA User Manual, 2007), experts are then involved to assess the effectiveness of specific measures and other impacts. These estimates may be based on expert knowledge, scientific results and scientific literature or on modelling the measures’ impacts (Klauer et al., 2012). The impact assessment should also include cost-benefit analyses as it is for

instance required in the WFD and MSFD in order to ensure that the chosen combination of measures is cost-effective, i.e. the maximum possible degree of ecological effectiveness is achieved at low cost (UBA, 2004).

After the creation of a program of measures by the competent authorities a stakeholder process is usually initiated to include views and opinions from other sectors and societal groups. The stakeholder process is intended to improve the quality of the decisions, to reach an agreement on contradicting opinions and to ensure acceptance of the chosen combination of measures during implementation (Mostert, 2003a; Özerol and Newig, 2008). During the stakeholder consultation process an adequate sharing of the burden should be reached between different sectors or different regions affected by the management plan.

It is important to monitor the progress in approaching the management objectives over time and to regularly update and adapt the program of measures. Therefore, monitoring and assessment are the tools for measuring the progress towards the ecological objectives, using a meaningful set of pressure indicators, state indicators and impact indicators which have been described in chapter 4. Regular monitoring and assessment are part of a procedural management cycle (Figure 4.3) as for example the review and update of River Basin Management Plans under the WFD. The management plans, which have to be updated every six years, document the progress in implementation and fine-tune the program of measures to changing conditions.

## 5.2 Nutrient reduction measures in European policies

The various European Directives (see chapter 2) have a different approach with regards to management measures and their meaning is slightly different. While the WFD, ND and MSFD give more room for the selection of specific and site adapted measures, the UWWTD defines clear technical and managerial measures to be taken.

The **EU-WFD** planning process distinguishes three categories of measures (EC, 2003): basic measures, supplementary measures and additional measures. In any given river basin, measures to abate eutrophication taken under existing Community legislation and/or national laws, e.g. under the UWWTD or Nitrates Directive are considered to be basic measures. Supplementary measures need to be taken if there is still a gap for achieving WFD good status of the water bodies in the river basin. Additional measures are needed if they basic and supplementary measures are not sufficient, i.e. also to reach more stringent environmental objectives e.g. under international agreements.

As part of the **Nitrates Directive**, Member States are expected to implement a series of measures, including: establishment of Codes of Good Agricultural Practices (CGAP); designation of NVZs; establishment of Nitrate Action Programmes (NAPs); and establishment of monitoring and reporting (EC, 2013b). CGAPs must target a number of activities leading to nitrogen emissions and loss, including closed periods of nitrogen fertilizer application on land and for example the application of conditions for fertilizer application (e.g. on steeply sloping ground, frozen or snow covered ground, near water courses). Member States establish and implement Nitrate Action Programmes for NVZs or the state's territory. NAP measures must go beyond CGAP measures; a number of pre-defined measures are outlined in the Directive such as maximum applicable amount of livestock manure (i.e. 170 kg N ha<sup>-1</sup>).

With respect to the **MSFD** most of the measures to abate eutrophication of marine waters will be implemented via other directives. Member States can fulfil the obligation to address eutrophication in marine waters through measures taken under the WFD, the ND or UWWTD. Annex VI of the MSFD lists eight types of measures (Art. 13): i) input controls, ii) output controls, iii) spatial and temporal distribution controls, iv) management coordination measures, v) measures to improve the traceability of marine pollution, vi) economic incentives, vii) mitigation and remediation tools and viii) communication, stakeholder involvement and raising public awareness.



In contrast to the aforementioned European Directives the **UWWTD** specifies concrete technical measures such as the licensing of discharges, the installation of collection systems and secondary treatment. Enforcement is differentiated between agglomeration size and types of measures to be implemented. Primary and secondary treatment shall be applied to all such urban wastewaters. In sensitive areas, advanced treatment (nitrogen and phosphorous removal) needs to be established for settlements of 10,000 inhabitants and more unless this more stringent treatment is implemented in the entire Member State.

### 5.3 Measures to reduce nutrient input from point sources

Measures to reduce nutrient inputs from point source include technical and organizational improvements in wastewater treatment, reduction and treatment of industrial discharges (e.g. Ranade & Bhandari, 2014), improved storm water management, the unsealing of urban surfaces (UBA, 2014) and improved management of fish farms and aquaculture (Bunting, 2013). Several more measures related to wastewater management are summarized in Table 5.1 **Error! Reference source not found.** (see also UBA, 2014, Nitrolimits, 2013).

Next to technical improvement steps in municipal WWTPs by introducing more stringent treatment via additional treatment steps such as phosphorous precipitation or denitrification, also modification in operation and management can lead to a significant reduction in nutrient loads (Metcalf et al., 2013). Besides technical upgrading of the WWTP itself, organizational issues are also important such as the establishment of larger centralized WWTPs with higher treatment standards or the connection of smaller plants to natural polishing and purifications systems like settling ponds or planted soil filters.

**Table 5.1: Overview of measures to address point sources.**

Strategy	Objective	Measures	Implementation options
<b>Removal of nutrients at source</b>		Introduction phosphate-free washing and cleaning agents via political regulations	Introduction of phosphate free detergents, replacement with harmless substances
<b>Decreased P and N input in the receiving water body</b>	Improved municipal wastewater treatment	Upgrading of WWTPs with additional treatment steps	Nitrification, Nitrification/ Denitrification, Phosphate elimination, modification of aeration stage,
		Downstream connection	Settling pond, biofilter, flocculation filtration, additional treatment of process water
		Increased connection rates of households to WWTPs	-
		Reduction of decentralized WWTPs, higher share of very big WWTPs	-
	Improved industrial wastewater discharge	Avoidance of wastewater, reduction of wastewater and its components, upgrading of industrial WWTPs	Reuse strategies, high performance bioreactors, flocculation/ precipitation, filtration, flotation, anaerobe, chemical or thermal approaches
	Improved storm water management (optimization and minimization)	Establishment of storm water overflow structures (rain overflow basins, rain purification basins)	Storm water overflow tanks, storm water sedimentation tank, soil retention filter, storage sewer
		Unsealing of urban surfaces and conversion to green areas	Removal of sealed floor covering, replacement of water impermeable covering by water permeable material
		Establishment of decentralized storm water infiltration structures	Swale infiltration, Swale-French drain infiltration, pipe infiltration, shaft infiltration, green roofs
	Improved management of fish farms and aquaculture	Wastewater collection and treatment, optimized feeding strategies, facility layout and management, modification of the receiving environment, water reuse	Nutrients adapted to nutritional demands, feeding regime tailored to behavior, stream channels modified to promote mixing and aeration, collection and treatment of wastewater after feeding (e.g. in constructed wetlands), culture water treated and re-used to retain nutrients

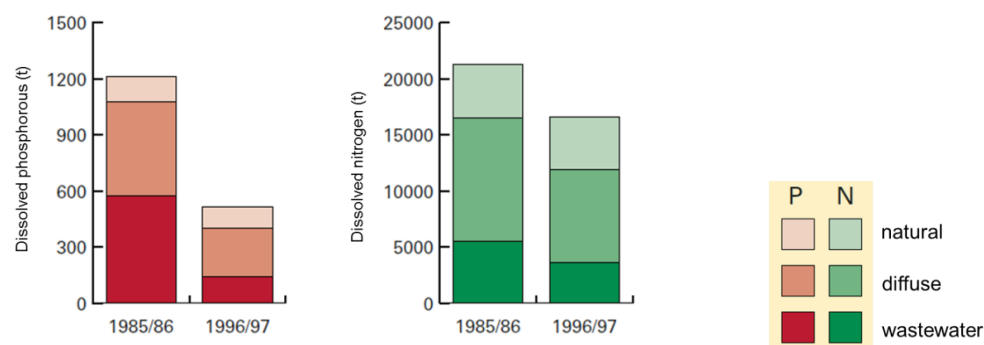
In the case study regions that were analysed for this report, significant improvements in wastewater treatment since the 1960s / 1970s have resulted in significant reductions in nutrient emissions from settlements and from industrial areas. The actions taken in the regions to install new WWTPs and to upgrade existing sewage and treatment system have been cost-intensive but successful (see text boxes with examples from Lake Constance and the Danube River Basin).

### Box 5.2: Case Study – Lake Constance

At Lake Constance, the International Commission for Protection of Lake Constance (IGKB) adopted the first “Guidelines for keeping Lake Constance clean” in 1967 and defined strict requirements for effluent concentrations of WWTPs (see Appendix 1 for concentrations). The first construction and investment program started in 1973 as a reaction to increasing nutrient loads caused by the discharge of highly polluted urban sewage (IGKB, 2004). The focus of the program was on the establishment of wastewater treatment facilities that included biological treatment and phosphorus elimination, the collection of wastewater via sewer networks in combined sewer systems and the comprehensive treatment of industrial wastewater in public and centralised wastewater treatment facilities. While in 1972 only 25 % of the population was connected to WWTPs with biological treatment, the connection rate in 1985 increased to 90 %. At the same time, the share of wastewater that was treated with phosphorous elimination improved from 24 % in 1972 to 88 % in 1985. In a second construction and investment program between 1986 and 1995 additional measures were implemented including the reduction of the phosphorous share in detergents and their replacement with environmentally friendly substances, the change in production methods in industry and the upgrading of storm water treatment. The latter included the upgrading of storm water overflow tanks, the establishment of storage sewer in combined sewer systems and the construction of storm water sedimentation tanks for separate drainage systems.

The efforts to reduce phosphorous and nitrogen releases into Lake Constance were considerable and resulted in the decline of the share of dissolved phosphorous in the treated wastewater from 577 t P in 1986 to 141 t P in 1997. The decline of phosphorous concentrations in wastewater is on one hand linked to the phosphate limit regulation that was adopted in 1980 that led to a P-decrease of 4.9 to 1.9 g per inhabitant per day between 1975 and 2000 (IGKB, 2004). On the other hand, the constantly increasing connection rate to WWTPs with integrated phosphate elimination contributed substantially to the reduction. In the year 2013 about 99.5% of the population in the catchment was connected to a central or decentral WWTP (IGKB, 2014). The nitrogen loads from urban areas showed the same behaviour of a substantial decline (see Figure 5.2).

**Figure 5.2: Development of dissolved phosphorous and nitrogen between 1985/86 and 1996/97 at Lake Constance**



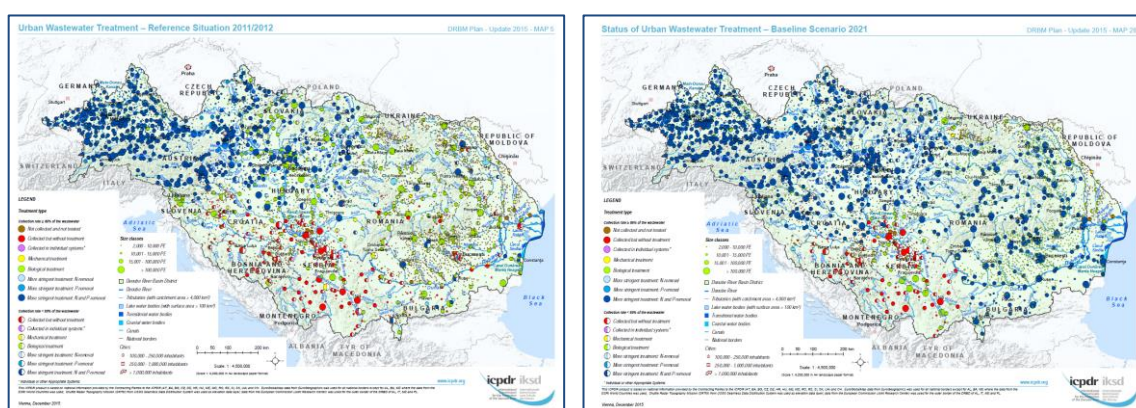
Source: IGKB, 2004.

### Box 5.3: Case Study – Danube River Basin

In the Danube River Basin, significant efforts have been made to reduce point source pollution by upgrading of WWTPs especially in the upstream countries (Germany, Austria and Czech Republic). In addition, a decrease of industrial discharge especially in the lower Danube countries was observed due to an economic break down in the 1990s and the closure of nutrient discharging industries. Today, agriculture (N: 42%, P: 28%) and urban water management (N: 25%, P: 51%) are the most important nutrient sources and are responsible for the majority of nutrient emissions indicating the necessity of appropriate measures to be further implemented in these sectors (ICDPDR, 2015). Untreated wastewater discharges are responsible for 28% (TN) and 39% (TP) of the total point source emissions (ICPDR, 2015).

Implemented measures to control point source emissions included nutrient removal at urban WWTPs, enhanced treatment technologies at industrial facilities and application of P-free detergents in the consumer laundry sector. Today, 1,827 agglomerations with a PE of about 50 million are equipped (at least partially) with tertiary treatment aiming at nutrient removal in the River Danube basin (ICPDR, 2015). Out of the 1,275 agglomerations with a size over 10,000 PE, 699 agglomerations (55%) have tertiary treatment already in place (Figure 5.3).

**Figure 5.3: Urban wastewater treatment in the Danube River Basin, left: reference situation 2011/2012, right: baseline scenario for 2021; brown dots = wastewater not treated and not collected, yellow dots = mechanical treatment, green dots = biological treatment, blue dots = more stringent treatment.**



Source: ICDPR, 2015.

Especially, in Austria efforts were undertaken to reduce nutrient inputs through better wastewater and storm water management. Since 1959, 18 000 municipal WWTPs, 11 000 km of storm water sewer system and 77 000 km of combined sewer system were built that led to an increase in the connection rate from 40 % in 1980 to 94.5 % in 2012. Today, 100 % of Austria's wastewater load is biologically treated, while in 98 % wastewater is also subject to more stringent treatment. Subsequently, the P loads from point sources have decreased by 50 % since 1990. The present level of P releases in the upper Danube is already in the range as it was in 1950. Nitrogen release from point source pollution decreased about 20 % since 1990.

Despite these successes, nutrient emissions from municipal WWTPs still contribute significantly to the total nutrient loads in the Danube River Basin (Schreiber et al., 2003; ICDPR, 2015). It can be assumed that the situation is similar in other parts of the Black Sea catchment area. This shows that the improvement of municipal wastewater management has a high potential for reducing nutrient loads to the Black Sea (UNDP-GEF, 2006). In this regard, the implementation of the UWWTD will significantly contribute to the further reduction of nutrient loads in the basin (ICPDR, 2015). In addition, the Danube River Basin Management Plan includes measures to introduce phosphate free detergents not only in Germany and Austria but also in the other Danube countries.

## 5.4 Measures to reduce nutrient input from diffuse sources

There are various measures to reduce nutrient pollution from diffuse sources, addressing different problems and considering different strategies (Table 5.2). Among the most effective measures is an effective **nutrient application management** in agriculture in order to reduce fertilizer application rates. The establishment of soil maps including nutrient balances in the soil-plant-matrix will help to balance nutrient uptake by crops and nutrient content in soils in order to avoid nutrient surplus. Fertilization plans, crop-rotation plans, as well as N mineralization assessments will support the farmer to reduce the excess of nutrients applied. A wise nutrient application rate not only refers to NPK-fertilizer but also to manure that should be immediately incorporated into the soil (Windolf et al., 2012). Also, the placement, the method and timing of manure and fertilizer is important (Schoumans et al., 2011; Schoumans et al., 2014): a successful strategy to avoid over-fertilization on agricultural land is to split nutrient fertilizer application into two phases, usually avoiding winter and autumn months, especially in high risk areas.

**Crop management** strategies will help to reduce nutrient losses to the surface and groundwater by planting catch and cover crops. While cover crops protect the surface of the soil from wind and runoff erosion, catch crops take up extra nutrients (Rubæk and Jørgensen, 2011; HELCOM 2013b; Andersen et al., 2014).

Losses of N and P to surface waters are often a problem in highly intensified agricultural areas associated with high **livestock** production. Therefore, the aim is to optimize overall livestock rates (Isermann, 2011), for example in the Danish Aquatic Action Plans a maximum stocking rate of 1.4 livestock units/ha on pig farms and max. 1.7 livestock units/ha on cattle/dairy farms was a regulated measure on the total agricultural area. Also, optimizing the daily nutrition of livestock, such as feeding adapted to lifecycles of livestock and reducing dietary N and P intakes will minimize the nutrient surplus in livestock production. The fencing off of rivers and relocation of gateways away from watercourses reduces the direct input of excrements and soil erosion by trampling (HELCOM, 2013b; Dorioz & Gascuel-Oudou, 2011).

Appropriate **soil tillage** has an influence on nitrogen and phosphorous mobilisation in the soil as it affects erosion and leaching of nutrients, especially during warm and wet conditions when soil tillage can promote mineralization processes within the soil (Schouman et al., 2001). In general, deeper and more intense soil management results in higher erosion risks (Schoumans et al., 2011). Minimal cultivation is therefore the best soil management to avoid losses.

**Water management in agricultural fields** may help to slow down the flow rate and to increase the length of the flow path within agricultural land, as this will promote sedimentation and infiltration in lower parts of the field. Another strategy is to avoid subsurface losses through leaching. Other measures are to establish controlled drainage systems or let drainage water irrigate meadows. The general idea is that a meadow or a riparian area filters water from tile drains that is polluted with particulate and dissolved nutrients. To avoid erosion and subsurface losses to rivers it is a common measure to establish riparian buffer strips that are located alongside watercourses (Stutter et al., 2012, Heeb and Johansson, 2013).

**Changes in land use and land use patterns** may help to reduce nutrient inputs into water courses by aiming at the identification of nutrient sinks and sources in the landscape. By relocating land use and changing land use patterns (like alternating grassland and arable land) the total buffer capacity of the catchment can be increased (Schoumans et al., 2014).

Other helpful measures in the field of surface water management are river, lake and wetland **restoration**. River restoration can significantly increase the retention times of water in the landscape. Increasing the water travel time in a catchment by 50% can result in 15-20 % higher N and P retention (Schouman et al, 2014). In addition, the re-establishment of lakes and construction of wetlands can significantly increase the retention of P and N since both function as a natural sink for nutrients (Zhi and Ji, 2012).

**Table 5.2: Overview and summary of potential measures to reduce nutrient releases from diffuse sources.**

Strategy	Aim	Measures
Change in nutrient application management	Improved general nutrient application management	vegetative mining, use of fertilizer recommendation systems, scientific fertilization systems, integrate fertilizer and manure nutrient supply, reduce fertilizer application rates, reduce nutrient application based on soil status, fertilizer placement near crops, no fertilizer application to high risk areas at high risk times
	Improved inorganic fertilizer application management	Reducing P of common NPK fertilizer
	Improved manure production and application management	Increase capacity of manure/slurry stores, adopt batch storage of manure/slurry, minimize volume of dirty water, compost solid manure, change from slurry to solid manure, incorporate manure into soil
	Improved manure surplus management	Incinerate poultry manure
Change crop management	Avoid erosion and reduce surface runoff	Grassland instead of arable crops, grow deep-rooting crops
	Change cropping system	Introduce crop rotation and mixed cropping systems
	Avoid leaching	Apply catch crops, crop production without fertilizing
Livestock management and production of minerals in manure	Improved overall production	Optimize/reduce overall stocking rates on livestock farms
	Improved feeding management	Reducing content of N and P in nutrition, impact of nutrition on reduction of phosphate excretion in pigs
	Improved grazing management	Harvest of grassland for silage or hay instead of cattle grazing, manage interaction between livestock and rivers
	Reduced impact of point sources at farm scale	Place manure heaps away from water courses and field drains, place heaps on concrete and collect effluent
Soil management	Avoid transport of particles of particulate P	No tillage/direct drilling, shallow cultivation, contour ploughing, change from autumn tillage to spring tillage, reduce soil compaction and improve soil structure
	Avoid leaching of dissolved P in soil	Conventional ploughing or interspersing periods of ploughless tillage with conventional ploughing, bind soluble P with chemicals
	Reduce nutrient budgets and increase soil storage capacity by extensification and agro-forestry	Introduce crop rotation and mixes cropping systems, set-aside for several years
	Avoid transport of particulate P in tram lines	Tillage to avoid tramlines
Agricultural water management	Change runoff by blocking or reducing overland flow	Create ponding systems, construct grassed waterways, create sediment boxes, improve surface irrigation
	Avoid subsurface losses through leaching	Remove trenches and ditches or allow to deteriorate, install drains, controlled drainage systems, irrigation of meadows with drainage water
Land use management	Improve location of sinks and sources by changing land use patterns	Alternate grassland and arable land, avoid certain crops in hilly landscapes, plant crops with high nutrient uptake on bottom land
	Protect very vulnerable areas by nature development	Afforestation of agricultural land
Land infrastructure	Reduce direct losses from farm yards	Reduce volume of dirty water from farm, collect farm yard runoff
	Reduce direct losses from livestock	Prevent contact with surface water: fences and bridges
	Reduce surface runoff and erosion from field to field	Re-site gateways and paths, controlled access for livestock and machinery
	Intercept nutrients from runoff, erosion and subsurface losses	Vegetated buffer strips



<b>Measures in surface waters</b>	River maintenance/ river restoration to increase nutrient retention capacity	Limited cutting of vegetation and reduce regular removal barriers, re-meander, restoration re-connection of flood plains increases the retention of water and nutrients in the landscape
	Lake rehabilitation and restoration to reduce P concentration in lake water	Control P inlet and prolong residence time of water, apply chemicals to bind P
	Wetland restoration and constructed wetlands to retain nutrient losses from upstream fields in wetlands	Create wetlands in agricultural areas with substantial P losses

**Notes:** Measures were categorized following the EU initiative COST action 869 “Mitigation options for nutrient reduction in surface water and ground waters” (<http://www.cost869.alterra.nl/>).

**Sources:** Schoumans et al., 2014; JRC, 2013; HELCOM, 2013b.

In eight case study regions (see Appendix 1) that were used as a basis for this report, the mitigation of diffuse pollution played an important role. The adopted strategies including selection of mitigation measures, areal extent of measures and incentives for farmers to use the measures differ greatly between the regions (see example from Denmark and Finland in the text boxes). The Danish Action Plans present a comprehensive combination of measures. Sector specific reduction targets were established together with promoting improved wastewater treatment, best available techniques in industrial wastewater treatment and a series of mandatory agricultural management measures. The Finnish Agri-Environmental Programme (FAEP) subsidised Finnish farmers for carrying out environmentally beneficial agricultural practices.

#### **Box 5.4: Danish Action Plans for the Aquatic Environment**

**Denmark** was one of the first countries to start tackling nutrient problems. In the 1970s and the early 1980s, there was increasing concern about the effect of nutrient losses from agriculture. Since 1985 a series of Action Plans for the Aquatic Environment have been initiated (Kronvang et al., 2005, 2008; Windolf et al., 2012; OECD, 2012) with the aim of reducing nitrogen and phosphorus pollution from point and diffuse sources (see Table A1.2 in the Appendix). The three main components for regulating nitrogen pollution were (1) mandatory requirements for improving treatment measures on Waste Water Treatment Plants (WWTP's) including nitrogen removal on larger WWTP's; (2) mandatory fertilizer and crop rotation plans, with limits on the plant-available N applied to different crops, and (3) statutory norms for the proportion of manure N assumed to be plant available. The third component was controlled through annual farm fertilizer accounts. Increases in plant available N in manure should substitute chemical N fertilizer at farm level as specific N norms are given for each crop type.

The implementation of the policy has shown great reductions in N- and P-discharges to the aquatic environment. The applied measures have shown remarkable effects on improved N-utilization, reduction of N-surplus and reduction of N-leaching (see more details in the appendix). The effects of the policies were continuously monitored and the policy was continuously adjusted. The reduction targets could not be reached in the first Action Plan and therefore were tightened in 1991. The reduction target was maintained but the time frame was extended.

### **Box 5.5: The Finnish Agri-Environmental Programme (FAEP)**

The Finnish Agri-Environmental Programme (FAEP) formed the most important policy measure and was implemented when Finland joined the European Union in 1995. FAEP is part of the Common Agricultural Policy (CAP) of the European Union and is considered to be a major package of policy instruments to encourage farmers to protect and enhance the environment on their farmland, through payments for the costs of provision of environmental services. The goal of the FAEP was to ensure the change in agricultural practices towards higher sustainability including eutrophication objectives. Nowadays, the FAEP is also the main tool within EU Water Framework Directive (WFD) to control the nutrient load from agriculture.

The FAEP consisted of three different levels of measures: (1) basic measures that have to be adopted by all farmers participating in the programme, (2) additional optional measures, of which one has to be chosen and (3) special voluntary measures for which additional support is given. As to controlling P losses, the basic measures require the preparation of a farm environmental management plan, the balanced fertilization of arable crops and implementation of filter strips along the main ditches and watercourses. Additional measures for arable farming include an option to comply with a stricter fertilisation practices based on crop, expected yield and soil-test P and an option to increase plant cover in winter or apply reduced tillage. The special measures require more efficient environmental protection, e.g. establishment and management of riparian zones, wetlands or sedimentation ponds.

FAEP has substantially affected farming in Finland. It had i) increased the area of filter strips and riparian zones; ii) increased the plant cover on arable land in winter; iii) decreased the number of animals per hectare; and iv) decreased the use of P and N in chemical fertilizers. More than 90% of the Finnish farmers were participating in the FAEP. Although the effects on water quality were not pronounced in the first years, some recent assessments show positive trends: nutrient field balances of nitrogen and phosphorous decreased markedly (Aakkula et al., 2012). However, this is not considered to be sufficient with respect to the water quality targets.

## **5.5 Measures to reduce internal nutrient inputs and enhance self-purification**

In surface waters where internal nutrient loads (nutrients leaching from the sediment under hypoxic conditions (see chapter 1.2) substantially contribute to overall nutrient inputs, measures have been taken to reduce these loads. In this respect, eco-technological measures can be distinguished from bio-manipulation techniques. Most experiences with such measures have been gained from lakes, since these are small-scale systems with limited water exchange that can be manipulated much easier than large-scale open systems such as estuaries, coastal and marine waters. Measures to reduce internal nutrient load in lakes are collectively termed 'lake restoration' (Gulati et al., 2008).

Eco-technological measures addressing the lake water body are for instance chemical precipitation, adsorption to carrier materials, deep water drain, destratification or hypolimnetic oxygenation. There are also measures addressing the sediment, most commonly the removal of sludge and sediments, but also sediment conditioning and or sediment covering (Gulati et al. 2008). In the case of chemical precipitation, phosphorous compounds are adsorbed by precipitating agents (e.g. aluminium or iron salts, calcium compounds). These adsorbed compounds sink to the floor and sediment. As a consequence, P concentration in open water and re-dissolution from the sediment are reduced. In lakes with low alkalinity, the use of aluminium can lead to a lowering of the pH value rendering aluminium concentrations toxic to plankton, hereby negatively impacting the food web. There is the risk that the zooplankton grazing pressure on the phytoplankton diminishes, thereby increasing eutrophication and counteracting the measure. The removal of sludge and P-rich sediment removes the internal nutrient

reservoir, thereby preventing further nutrient leaching. Problems associated with this method are the stirring-up of organic substances, finding suitable sites to deposit the sludge on land and the high costs associated with this method.

While the eco-technological methods have mainly focused on reducing or immobilising the amount of phosphorus in the system, biomanipulation methods have aimed at reducing algal growth and improving secchi depth through food web changes. Removal of planktivorous fish stocks by targeted fishing efforts or by rotenone can increase zooplankton stocks that can, in turn, reduce the phytoplankton by more intensive grazing pressure (Jeppesen et al., 2012). Furthermore, it has been tried to stock fish-eating (predatory) fish to reduce the number of zooplankton-eating fish and increase the zooplankton biomass. Removing or reducing abundant planktivorous fish, such a roach, may further reduce phosphorus load to the pelagic zone from the littoral zone sediments through reduced fish migration between the littoral and pelagic zone in stratified lakes. Another type of bio-manipulation is the introduction of other organisms that consume nutrients, e.g. macrophytes. Lake shores have been planted with indigenous aquatic plants that compete with the phytoplankton for available nutrients. Extensive aquaculture, cultivating mussels, will also improve the self-purification capacity of the lake through increased grazing of phytoplankton.

Lake Trummen in Sweden is one of the earliest examples of a successful lake restoration. External loading was reduced at the end of the 1950s. In 1971/72, the P-rich upper sediment layer was removed and the lake's medium depth enlarged from 1.1 meters to 1.75 meters. This measure in combination with continuous manipulation of the fish stock brought about a lowered trophic status that could be maintained for over 30 years (Zerbe and Wiegand, 2008). A well-documented example of bio-manipulation in Germany is the dimictic lake Feldberger Haussee, covering 1.36 km<sup>2</sup> and of a mean depth of 6 meters. After 1980, the external loading due to wastewater inputs had been reduced by 90 % but the taken measures did not yield a perceptible amelioration of the water quality. To support the assimilative capacity of the lake, a bio-manipulation project was started. From 1985 onwards, planktivorous fish were regularly harvested and juvenile predatory fish stocked. Additionally, fishing such predators was absolutely forbidden between 1995 and 1997 and subsequently, fishing was generally controlled and subject to restrictions. As a consequence, secchi depth steadily increased since 1992 to a maximum annual mean of 2.8 meters in 1999. In 1995, the lake still was classified highly eutrophic, but already in 1997, it was classified slightly eutrophic (Nixdorf et al., 2004).

In general, bio-manipulation methods are much more cost effective than eco-technological measures. While in some lakes internal nutrient loads could be sustainably reduced, many lakes showed a return to a eutrophic state within 10 years or less. A prerequisite for such measures to be successful is the sustained reduction of external anthropogenic nutrient loads (Søndergaard et al., 2007).

#### **Box 5.6: Reducing the internal nutrient load of the Baltic Sea**

To speed up the recovery of the Baltic Sea environment various measures to reduce the internal nutrient loading have been proposed. In particular in Sweden considerable research and investments have been put into projects to investigate the potential and feasibility of direct measures in the sea. The measures proposed are similar to methods used for lakes. They fall into three categories – 1) elimination of phosphorus leakage from the sediment storage, 2) bio-manipulation of the food web and 3) the extraction of nutrients from the sea (HELCOM, 2015b).

To date only small-scale experiments have been conducted in fjords and coastal waters, but the effects of large-scale Baltic wide measures has been estimated by modelling. On a small-scale, the anoxic bottom waters of Byfjorden on the Swedish west coast have been oxygenated using electric pumps that pump down the oxygen-rich water (Stigebrandt & Liljebladh, 2011). There were positive results barely two months after pumping started. Oxygen concentrations and water flow increased, phosphorus concentrations above the sediment decreased and poisonous hydrogen sulfide was neutralized. Benthic animal life has been returning and there was only a minimal release of toxic substances from the oxygenated sediments.

While small-scale projects in fjords or coastal waters may, under certain circumstances may be a useful supplement to measures at source, upscaling those projects to cover large parts of the Baltic Sea will require negotiations and agreements between countries in the Baltic Sea region (HELCOM, 2015b). Furthermore, possible detrimental effects on organisms and environmental processes of sea-based measures need to be further studied to ensure their safe application (Conley et al. 2009).

### **5.6 Supplementary instruments**

While concrete and technical measures are implemented at the local scale, several other administrative, economic and informal supplementary measures can be implemented on a larger scale. These instruments serve to support the implementation of technical measures by creating incentives for the relevant players to modify their behavior. One consequence of this is that supplementary instruments have a more long-term, more widespread effect than technical measures, and require coordination at a higher administrative level. The following Table 5.3 gives an overview, categorizing the available supplementary measures into: charges/financial incentives, cooperation arrangements and advisory approaches.

**Table 5.3: Supplementary instruments to reduce nutrient pollution.**

Category	Instrument
Charges/financial incentives	Financial subsidising of organic farming
	Tax on organic fertilizers from non-free range farming
	Tax on mineral fertilizers
	Financial incentives for nature conservation
Cooperation arrangements	Creation of effective cooperation structures between farmers and water companies (This instrument represents a large number of possible cooperation arrangements between different players at the various levels).
	Nutrient trading
Information, knowledge production and communication	Advisory services to farmers on optimum operation from the viewpoint of water pollution control
	Advisory services to the competent authorities to optimize water body maintenance
	Awareness raising

Economic instruments are widely used to address diffuse source pollution, with the use of payments much more frequent than application of pollution taxes and with infrequent application of the Polluter-Pays-Principle to diffuse source pollution (OECD, 2012).

Taxes can affect the use of mineral and organic fertilizers dramatically. The Danish Aquatic Action Plans, for example, included a tax on nitrogen field fertilizers and taxes on phosphorous fertilizers added to feed but also subsidies to invest in farm infrastructures and subsidies to more organic farming, wetlands, extensification and afforestation. The tax system and subsidies were important incentives for investments in new and more environmentally friendly production systems and the use of mineral fertilizers has dramatically decreased due to the imposed taxes (Kronvang et al., 2008; Dalgaard et al., 2014)

Systematic on-farm advisory service is a common measure to support the implementation of agro-technical measures. A close co-operation between farmers and agricultural advisors is needed. Advisors give consultations on e.g. limited stock density, crop coverage over winter, intercropping, fixed values for nitrogen utilisation of farm manure, limited nutrient budget, fertilizer plans and nutrient balances. The method, which can reduce nutrient input significantly is easy to implement, but requires a dense system of advisors to support farmers.

## 5.7 Success of eutrophication abatement measures and policies

“The European environment — state and outlook 2015 report” (EEA, 2015b) shows that over the past two decades there has been a significant reduction in nutrient levels in Europe’s waters, although there are differences for nitrogen and phosphorous. Concerning measures to combat eutrophication the low-hanging fruits (the implementation of measures addressing point sources) have been picked and eutrophication abatement now faces the challenge to implement measures to reduce nutrient inputs from diffuse sources against a strong agricultural lobby.

### 5.7.1 Abatement of phosphorus pollution

Overall, phosphorous concentrations have halved in rivers of Western and Central Europe and to a lesser extent in lakes since the 1990s (EEA, 2015b). With regards to European seas, OSPAR reports that of the nine countries bordering the North and Celtic Seas, six have achieved a 50% or greater reduction in their phosphorus emissions between 1985 and 2005. In the Mediterranean, some studies suggest a reduction in eutrophication in critical areas, such as the mouth of the Po River in the Adriatic (EEA, 2015a). Eutrophication is still considered one of the major threats to the marine environment of the Black Sea although algal blooms were most significant in the 1970s and 1980s and have seen decreases since the 1990s (EEA, 2015a).

The decrease in phosphorus concentration reflects both improvements in wastewater treatment and the reduction of phosphorus in detergents through the use of phosphate-free detergents (EEA, 2015b). The countries showing the strongest decreasing trends for lakes and/or rivers are closely related to countries with a large proportion of the population connected to wastewater treatment (to a large extent tertiary) (EEA, 2015b). The UWWTD implementation has been accompanied by significant investments from the European Union and Member States, with a total of 8 billion Euros of committed funds from the EU cohesion and other funds between 2000 and 2006 and at least 15 billion Euros of committed national contributions in the same period (EC, 2010).

Because implementation has spanned over nearly 25 years, enforcement is differentiated between Member States based on their accession date to the EU. Implementation deadlines have now passed for 15 Member States producing 81% of the pollution (i.e. 12 Member States forming the European Community in 1991 and the three Member States joining in 1995). For the remaining 13 countries who joined the European Union after 1995, deadlines have been set between 2006 (e.g. Malta) and 2023 (e.g. Croatia) with most countries aiming for 2015 (Cerar, 2014). The most recent implementation report (EC, 2016) shows the following:

- Compliance regarding the installation of *wastewater collection systems* is high with 98% (EU-28), with only two Member States with a compliance rate below 60% (i.e. Bulgaria and Slovenia).
- Compliance regarding *secondary treatment* is at 92% (EU-28). 16 Member States are at a 90-100% compliance rate while three (Bulgaria, Malta and Slovenia) are below 50% compliance rate.
- Compliance regarding requirements for *more stringent treatment in sensitive areas* is at 88% (EU-28). As for secondary treatment, compliance vastly differed between Member States. 12 Member States are above 90% compliance rate while 9 Member States were below 50%, with some countries as low as 2% (i.e. Bulgaria).

Assuming further compliance with the UWWTD and other emissions reduction policies, WFD good status was foreseen to be achievable by 2015 and 2027 for total ammonium and total phosphorus respectively (EEA, 2012). However, Member States estimated that basic measures linked to the UWWTD were not enough to tackle significant pressures from point sources in 60% of RBDs (EC, 2015a). Future compliance cost of the UWWTD could nevertheless be minimised through the WFD, for example by engaging in supplementary measures aiming to reduce the pollutant loads received by treatment plants. This would for example necessitate further incentives or regulatory measures on the use of phosphate-free detergents in industrial and domestic uses.

Despite remarkable investments by public authorities over the last 25 years, the implementation of the UWWTD remains a significant challenge. It was estimated that at least 11 billion Euros were still needed to ensure compliance with key requirements of the UWWTD (EC, 2010). In addition, further actions are required on agricultural sources of phosphorus as they still represent an important source of eutrophication for many European lakes. A recent audit focusing on the Danube river basin by the European Court of Auditors concluded that “measures limiting phosphorus application on land have not been adopted or considered, and agricultural measures taken are mainly voluntary and do not directly improve water quality (...) The European Commission should consider introducing an obligation to set limits on the amount of phosphorus that can be applied on land, as is the case for nitrogen” (ECA, 2016).

### 5.7.2 Abatement of nitrate pollution

The average nitrate concentrations in European rivers have reduced steadily over the period 1992 to 2012 by about 0.8% per year (EEA, 2015b). Downwards trends in nitrate concentration in European rivers have resulted with reduced nitrate loads into European seas, with for example a reduction of approximately 30% since 1985 in the OSPAR region (OSPAR, 2010). Since 2005, average nitrate concentrations in European groundwater have declined and in 2011, the mean concentration had almost returned to the 1992 level (EC, 2015a). While reductions have been achieved in river loads, many countries are not reaching their targets, with only Denmark reaching its 50% reduction target between



1985 and 2005 (OSPAR, 2010). Also, 23 out of 32 countries had groundwater monitoring stations with an average concentration above 50 milligrams of nitrate per litre in 2012.

The reduction in nitrate concentrations is the result of increased wastewater treatment and, in some regions, declining mineral fertilisers' consumption and improved agriculture practices. Velthof et al. (2014) estimated that total emissions of nitrogen in the EU in 2008 were smaller due to the implementation of the Nitrates Directive by 3% for  $\text{NH}_3$ , 6% for  $\text{N}_2\text{O}$ , 9% for  $\text{NO}_{(x)}$ , and 16% for N leaching and runoff. Lower emissions were mainly due to the lower N inputs by fertilizers and manures. However, the Nitrates Directive has proven to be challenging to implement. While some Member States reported near to full compliance (e.g. Hungary, Finland, Cyprus), other Member States reported non-compliance rates of 88%, 82% or 60% (for single measures in Poland, Malta, and Portugal) (EC, 2013b). Ten cases of infringement have been opened since only 2013 by the European Court of Justice, involving France, Greece, Poland, Luxembourg, Bulgaria, Slovakia, Italy and Latvia. Cases focused on the incorrect designation of NVZs or on the lack of ambition of measures outlined in NAPs:

- Member States have taken different *approaches to the designation of NVZs*, with countries such as Germany, Denmark and The Netherlands designating whole territory while others, such as France, Portugal and Italy, designating regionalised NVZs. When using regionalised NVZs, Member States declare on average 28.6% of their state territory as NVZ. According to Grossman (2000), the decision to designate the entire country's territory as NVZ was initially based on the effort needed to monitor it. However, designation of NVZs should be primarily based on existing nitrates concentration and the risk posed by agricultural activities on future concentrations. In addition, phosphorus was usually not considered in the designation as it is not explicitly required by the Nitrates Directive. However, recent legal cases indicate that regions more susceptible to eutrophication through phosphorous should also be considered in the designation of NVZ (EC, 2009). The area covered by NVZs has nevertheless steadily increased, especially for Romania, Belgium (region of Wallonia) and Sweden.
- Most issues with the *lack of ambition in NAPs* include the appropriateness or volume of manure storage facilities, nitrate application limit and fertilizer accounting (EC, 2013b). Infringement cases in particular included "insufficient length of closed periods for fertilizer and manure application, insufficient requirements for manure storage capacity, insufficient and/or unclear rules for limiting the overall fertilization, insufficient rules for preventing water pollution through rules on fertilizer application to steeply sloping, frozen or snow-covered ground or near water courses" (EC, 2013b).

Nitrate pollution from agriculture is considered a significant pressure in more than 40% of all river and coastal water bodies in Europe (EEA, 2012) and 92% of the reviewed RBMPs refer to pollution pressures from agriculture (EC, 2012). In the first RBMPs (2009-2015) most measures on nitrate diffuse pollution related to the basic measures under the Nitrates Directive (e.g. CGAPs, NAPs). However, an assessment made in 2010 showed that basic measures linked to the Nitrates Directive were not enough to tackle significant pressures from diffuse sources in 63% of RBDs (EC, 2015a). Despite the gap, few supplementary measures targeted nitrates emissions outside NVZs. Furthermore, supplementary measures were mostly of a voluntary nature, such as training, awareness raising and research; less commonly were considered compensation for land use change, nutrient trading and fertiliser taxes (EC, 2012). Overall, the measures in the RBMPs for combating eutrophication were assessed as being vague, with insufficient description of the measures and a lack of detail on their implementation, such as share of responsibilities, timetable for implementation, or available funds (EC, 2012). Similar issues have been identified in a recent screening of the 2<sup>nd</sup> RBMPs 2015-2021 (EC, 2015a).

Gaps in implementing the Nitrates Directive and WFD impair the achievement of targets for eutrophication abatement stemming from international sea conventions (e.g. OSPAR; HELCOM). Other reasons may also explain these failures, including: the inability of international commissions to enforce agreements with contracting parties; the lack of commitment to reduce nutrient loads by some countries; the inefficiency of some of the selected policies; delays in the implementation of measures; or

the lag in response from ecosystems to show nutrient reductions, as nutrients can collect in sediments and continue to contribute to eutrophication over time (OSPAR, 2010). In the Mediterranean and Black Sea regions, the main policy challenges also include (Goulding et al., 2014): a lack of incentives to take action; a lack of awareness about the importance of the issue; political blockage to agree on common initiatives; and economic development without ensuring the precautionary principle (MIRA, 2012).

### 5.7.3 Common Agricultural Policy (CAP)

The Common Agricultural Policy (CAP) has been established in 1957 and has historically supported increased production and intensification of agriculture in the form of price control and agricultural subsidies. As a result, the CAP is commonly seen as one of the main causes of eutrophication in Europe. Since the 1990s, the CAP has undergone successive reforms. The CAP is now organised around two “pillars”: Pillar I was to support agricultural production while Pillar II was to support broader rural development aims. Support to eutrophication abatement can now be found under both pillars.

Under Pillar I, a 2003 reform led to the introduction of a general decoupling of agricultural subsidies from production and the establishment of cross-compliance mechanisms which farmers must comply with. “Statutory management requirements” (SMRs) include 18 regulatory requirements stemming from other European directives and regulations such as the Nitrates Directive, and “Good Agricultural and Environmental Condition of land” (GAECs) include 15 standards on farms. Three SMRs are directly relevant to eutrophication abatement<sup>2</sup>, while several GAECs are directly or indirectly relevant to eutrophication abatement<sup>3</sup>.

Under Pillar II, Member States must prepare Rural Development Programmes (RDPs) that outline activities for strengthening the competitiveness, social cohesion and environmental performance of agriculture and the rural economy. The activities are funded by the European Agricultural Fund for Rural Development and the respective Member State. By law, RDPs must integrate water policies and have thus considerable potential for supporting eutrophication abatement. Several “standard” measures are potentially relevant for eutrophication abatement<sup>4</sup>. However, Measure 10 on “agri-environment-climate scheme” is usually perceived as having most potential as it compensates changes in farming practices<sup>5</sup> and land use changes for environmental purposes<sup>6</sup>.

In its 2014 assessment of the linkages between the WFD and the CAP, the European Court of Auditors (ECA, 2014) conclude that cross-compliance and RDP funding have limited impact relative to the policy ambitions set for reducing the environmental impact of agriculture. With regards to cross-compliance,

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<sup>2</sup> SMR2 “protection of groundwater against pollution”, SMR3 “use of sewage sludge in agriculture”, SMR4 “protection of waters against pollution caused by nitrates from agricultural sources”

<sup>3</sup> Minimum soil cover, minimum land management reflecting site specific conditions to limit erosion, the establishment of riparian buffer strips, and adequate soil organic matter and soil structure

<sup>4</sup> Including so-called Measure 1 (“knowledge transfer and advisory services”), Measure 4 (“investments into physical assets”, e.g. installations for wastewater treatment on farms and in processing and marketing; modernisation of manure storage and handling facilities; investment related to hedgerow/wetland creation and landscape features for erosion control); Measure 8 (“forestry and agro-forestry measures”); Measure 11 (“organic farming”, both conversion and maintenance are eligible for funding); Measure 12 (“payments for compensating WFD implementation”, i.e. Article 38 which allows to compensate for costs incurred and income foregone from resulting disadvantages in the areas related to the implementation of the WFD) and Measure 13 (“funding for areas with environmental constraints”, which can maintain grazing systems and other low-intensive farming)

<sup>5</sup> For example, lower fertiliser use, soil management, winter catch crops, reduced livestock density, and manure management

<sup>6</sup> For example, riparian margins, buffer strips, hedgerows, conversion to grassland/pastures

the court observed the frequent breach of SMRs and GAECs (in particular SMR4 “protection of waters against pollution cause by nitrates from agricultural sources” and GAEC1 on buffer strips). A major barrier in the implementation of cross-compliance lies in monitoring compliance as many of the measures lack clarity or are ambiguous. With regards to RDPs, the Court observed that RDPs did not always take EU’s water policy into account. In addition, the implementation rates of water related-measures remained low.

A recent assessment of the 2014-2020 RDPs carried out by the European Commission highlighted an improvement in the consideration of water issues in RDPs but several weaknesses remained (EC, 2015c). For example, while nutrient pollution is usually examined in the environmental assessment of RDPs, it was rarely linked to specific agricultural pressures (e.g. livestock or cropland), thus ensuring appropriate targeting of measures.

## 6 Lessons learnt and future challenges in eutrophication abatement

### 6.1 Lessons learnt from case studies

The selected case studies show that in many cases nutrient concentrations in the water have been successfully reduced due to significant reductions in nutrient emissions and agricultural surplus. Different approaches have been taken in the selected case studies and it is useful to review the eutrophication abatement policies to discuss success factors but also the remaining challenges.

**Better eutrophication abatement may be achieved by effective water governance.** All eutrophication management plans in the case studies included the establishment of an organisational framework. The structure of the framework can be complex in some cases (e.g. Ebro River, HELCOM) and may require cooperation across sectors and administrative levels. In the *Ebro River Basin*, the Ebro Hydrographical Confederation, which was established already in 1926, is responsible for water management and planning. But there are several other strong actors in the region (National Directorate General for Water, Autonomous regions of Spain, national and regional departments of environment / agriculture, local governments, irrigation communities and others), which create a complex network of partners that do not always share common views (Bielsa and Cazcarro, 2015; Sánchez-Martínez et al., 2012). Such complex institutional frameworks cause lengthy coordination processes or even block effective implementation of eutrophication policies.

The *Baltic Sea* is a delicate case including a highly complex governance setting. The Baltic Sea is surrounded by nine riparian countries with five additional countries sharing parts of the catchment, making its eutrophication management politically complicated. On the one hand, the European Union adopted the first supra-regional strategy, the Strategy for the Baltic Sea Region (EUSBSR) in 2009. However, this strategy excludes Russia. Additionally, the EU directives play a decisive role in addressing the Baltic Sea eutrophication. Nevertheless, the countries bordering the Baltic Sea have their own national policies and legislations and the protection efforts varies greatly in terms of ambition, capacity, funding and continuity (Pihlajamäki and Tynkkynen, 2011), i.e. the national implementation programmes of the BSAP vary remarkably between the riparian countries (Tynkkynen et al., 2014). HELCOM provides a forum to discuss and negotiate eutrophication management strategies between all riparian countries. However, a main drawback of the HELCOM convention is that, contrary to EU directives, sanctions for non-compliance with the HELCOM recommendations do not exist. Mastering the complex governance setting in the Baltic Sea region in an intelligent way is therefore an important piece of the puzzle to improve the ecological conditions in the Baltic Sea.

**Intergovernmental river basin commissions for transboundary eutrophication management** can act as a platform for negotiations and cooperation and catalyse a constructive eutrophication management process (Mostert, 2003b). The International Commission for the Protection of the Danube River (ICPDR) and the International Commission for the Protection of the Rhine (ICPR) are often regarded as extremely successful in transboundary river basin management (Wilken, 2006). Both commissions were awarded the Thiess International River Prize, for the protection of the Danube River in 2007 for the protection of the Rhine in 2014. In the Danube river basin the challenge has been to develop common management goals for the largest river basin in Europe thereby uniting ideas and concepts from former socialist countries of the USSR with strategies and legal obligations of EU Member States. In the Rhine river basin, countries that went to war twice in the last century have managed to reach agreements on many water issues, and eutrophication has been reduced considerably. Both river commissions have fulfilled in an excellent way the function to moderate the process, to refine management goals to coordinate necessary research projects and to strengthen the process of cooperation.

Successful eutrophication abatement approaches typically follow **management cycles** including continuous revisions and updates of management action, this includes the establishment of clear timelines, the continuous monitoring of results, reporting, the review of actions and adapting the Action Programme. The Danish Aquatic Action Plans (Table A1.2 in the appendix) and the Rhine Action Plan (see appendix) are both successful examples of setting nutrient reduction targets, implementation of a program of measures, monitoring of effects, revision of the programmes and tightening of targets.

**A close cooperation between policy and science** will strengthen the overall process of eutrophication management and ensures that appropriate measures are put in place. Throughout the implementation of the Danish Aquatic Action Plans, the regulations have been designed in close dialogue with researchers, farmers and farmer associations, and have been followed-up by capacity development measures. In addition, the government supported considerable strategic research programmes e.g. on improving the understanding of nitrogen processes in soils and water, optimization of manure utilization, and organic farming production systems. The direct use of the research results in policy development has favoured a more targeted development of the Action Plans.

Setting regionally agreed, transparent and **science-based nutrient reduction targets** are a prerequisite for successful nutrient management, as shown in the Baltic Sea case study. The targets need to be accepted by all polluters and they require a regular and transparent follow-up. Furthermore, the targets should ensure that good status with respect to eutrophication can be achieved for all surface water categories. In this respect they need to be based on the nutrient reduction requirements of the water category most susceptible to nutrient enrichment (often coastal waters, but in some regions also groundwater or lakes). While aspirational targets (reduction targets not linked to the ecosystem requirements, e.g. 50%) are a good starting point they should be replaced by science-based targets as soon as the knowledge becomes available. Science-based targets should establish a link between nutrient inputs and the eutrophication status of surface waters and should be set so that the nutrient reduction allow the achievement of good status with respect to eutrophication. The Baltic Sea case study is a best practice example in this respect since the nutrient reduction targets are based on Secchi depth and oxygen debt and consider also nutrient concentrations and chlorophyll-a.

There are large differences between the countries regarding the incentives used to motivate farmers to implement agri-environmental measures. It differs between countries whether the mitigation measures are **regulated by legislation** or are implemented on a **voluntary basis**. In Denmark all measures are mandatory and are not subsidized (except restored wetlands and 10 m buffer strips) whereas in other Baltic countries the majority of measures are subsidized (Andersen et al., 2014). In Finland, many measures are implemented on a voluntary and subsidized basis. Denmark and Sweden adopted legislation on reducing N losses already in the late 1980s and both countries aimed at a 50% reduction. Both, Denmark and Sweden have had great success in reducing diffuse N losses from agriculture during the last decades. The long implementation period of more than 30 years and the mandatory status of measures may have triggered the success to achieve the reduction targets. In Finland however, the regulation showed less effect on N losses to the aquatic systems.

The analyses of the case studies reveal that eutrophication abatement can be successful if the **polluter-pays principle** is applied strictly and **financial burden is shared** by the sectors according to the relative pressure they exert. In Denmark, the national authorities reached an overarching agreement between sectors on the relative efforts to be made to achieve nutrient reduction. The negotiation was based on a quantitative assessment of the relative pressure (i.e. emission loads) from different sectors.

An example from Lower Saxony, Germany shows the collaboration between water supply companies and farmers aiming at nitrate reductions for drinking water protection. Lower Saxony decided in 1992 that water abstraction fees should be used to pay for measures that would reduce nitrate pollution in groundwater. This was one of the measures taken to comply with the Nitrates Directive. 74 partnerships between water supply companies and farmers have since then been established. Measures include additional advisory services for the agricultural, forestry and gardening sectors, compensatory payments

for farmers that adopt measures going beyond good agricultural practice, the monitoring of groundwater pollution as well as research for adaptive land use management. Voluntary actions, for which farmers receive payments are catch-crops, under-sown grass, reduced tillage, reduced nitrogen fertilisation, adapted crop rotation as well as result oriented compensation, in which farmers are compensated when they reduced nitrate input no matter which measures were implemented (Hartung et al., 2015).

The level of **public awareness** also seems to be an important supporting factor for pushing forward necessary eutrophication abatement measures and implementing them effectively. In the River Rhine, water pollution received attention from 1900 onwards and water quality issues were discussed between the riparian countries. The ICPR first met in July 1950 but the progress in achieving better water quality was only slow in the following years. A decisive factor that triggered the establishment of the Rhine Action Plan was generally growing environmental awareness since the late 1960s and the Sandoz disaster in November 1986 (Mostert, 2009). The environmental effects of the disaster were reported in great detail in the media, and two weeks after the chemical spill, a special Rhine Ministers Conference was organized in Zurich. This was the dawn for the Rhine Action Plan.

An example from Denmark shows also, that such disaster events can speed up the development of environmental action plans. A main factor that kicked-off the Danish National Action Plans with a regulation aiming at nutrient management in agriculture was a television report, which showed dead lobsters in the Kattegat Sea. The reason for the finding of dead lobsters was attributed to hypoxia resulting from algal blooms triggered by nutrient runoff from agricultural fields. The public awareness increased tremendously and people realised that something had to be done to reduce nutrient losses from agriculture. Against this background the first Danish Action Plan was initiated in 1985.

In many of the case studies, there was a significant **time lag** for the effects of agricultural measures to become visible and detectable in water quality monitoring. When looking at a comparably short time period of e.g. 5 to 10 years, a decrease in nutrient loadings may be masked by variations in the hydrological regime (Mitikka et al., 2005). This was apparently the case for the Finnish Agri-Environmental Programme which was reviewed after 10 years of implementation. Additionally, climate, soil properties and agricultural practices influence the amount of nutrient losses from agricultural land, especially phosphorus. Phosphorus is strongly bound by soil and large pools of phosphorus, built up during decades of 'over-fertilization', provide a considerable potential for P losses over several years (Ekholm, 2005). And finally, farmers might need time and a more profound understanding of how a change in their behaviour will contribute to eutrophication abatement.

## 6.2 Implementation gaps and future perspectives

European policies on eutrophication abatement have involved three stages: the setting of the first European-wide quality standards mainly in the 1970s, the establishment of stricter control on nutrient emissions mainly in the 1990s, and the search for greater policy integration between policies in the 2000s. The current European policy framework for eutrophication abatement in freshwater, coastal waters and marine waters is now comprehensive. Gaps in implementation prevail, calling for stronger enforcement and greater integration of policies.

The UWWTD has been accompanied by large investments from the European Union and Member States, but there is still a major gap in full compliance and effective reduction of nutrient loads in sensitive areas. While nutrient releases from urban wastewater have significantly decreased and a reduction of fertilizer application has been observed, diffuse emissions from agriculture remain a major challenge. Stronger implementation of the Nitrates Directive is needed to achieve further nutrient emission reductions from agriculture. A major challenge is the selection and design of measures within Nitrate Action Programmes, which lack ambition and are poorly enforced. Synergies exist between the UWWTD, Nitrates Directive and the WFD, which should be maximised to minimise implementation costs. This involves acting upstream by influencing behaviour and product development through appropriate regulatory (e.g. standards) and economic (e.g. taxes) instruments.



Translating European policies targets into qualitative objectives and quantitative nutrient reduction targets for eutrophication abatement is often achieved by looking at nutrient concentrations. There is a great need for harmonisation of target concentrations in freshwater (along different types of water bodies) and scientifically sound methods to derive these targets in order to achieve a level playing field in eutrophication abatement efforts. Additionally, when comparing nutrient standards for transitional, coastal and marine waters a wide range of parameters for different seasons and calculated with different statistics is in use across Europe, seriously hampering a comparison. Looking at the large heterogeneity of methods used to derive nutrient standards for those waters it can be concluded that the standards are not well harmonised, neither across Europe nor within Europe's Seas.

The development of nutrient standards needs to be based on a reference situation not influenced by eutrophication. This can be the reconstruction of a historic reference or the identification of existing reference sites. Especially in the Baltic Sea and the Greater North Sea nutrient reference conditions are mainly derived from historic nutrient loads, but there is no agreement on what constitutes historic conditions that were not yet affected by eutrophication. An exception to this are the efforts of HELCOM of deriving targets for eutrophication indicators based on a harmonised approach for the open Baltic Sea. In the last decade, a more and more sophisticated and regionalised approach is being applied considering the nutrient load reductions required for individual areas to achieve a good eutrophication status (as expressed by a number of indicators). While HELCOM has successfully set quantitative nutrient reduction targets in the Baltic Sea Action Plan for individual basins based on a Baltic-wide harmonised approach, OSPAR is still struggling to achieve this for the Greater North Sea.

Ideally, freshwater and salt water should be managed in an integrated way as excessive nutrient releases from the landscape harm all kinds of waters. Waterborne nutrient inputs are to be managed using nutrient reduction targets that enable the achievement of good status for all surface water bodies, including rivers, lakes, transitional, coastal and marine waters. However, in practice there have been only few efforts by Member States to align nutrient boundaries for freshwater and salt water, e.g. by setting a management target for rivers that enables achievements of good status for transitional, coastal and marine waters. However, the implementation of the MSFD is an opportunity to strengthen coordination between the implementation of freshwater, coastal and marine water directives, in particular the WFD, and implement more ambitious initiatives.

Pressure, state and impact analyses are important management tools to develop and implement a programme of measures. A series of effective management tools are well established and were described in chapter 5 of this report, ranging from technical measures to non-technical management instruments. However, eutrophication abatement measures of freshwaters should be combined with restoration measures, which were not reviewed in this report. Significant alterations in hydro-morphology and disturbances in hydrology in many European rivers contribute to the eutrophication phenomenon. Therefore, nutrient reduction measures should be combined with hydrological and hydro-morphological restoration efforts and synergies should be created to achieve policy objectives and to use budgets efficiently.

The implementation of measures under existing policies and international agreements by Member States is often lagging behind the ambitions. A mix of challenges is slowing the speed of implementation (financial resources, governance settings, time lags, etc.). Effective measures are available, but implementation of measures needs more efforts.

Measures to reduce internal nutrient loads in lakes and coastal waters can be a useful supplement to measures taken at source. However, great care needs to be taken that they do not serve as a replacement of such measures, thereby allowing polluters to escape from responsibility. This is especially important as some of the internal measures are considerably cheaper than land-based measures.

Recent proposals to establish nutrient trading systems in Europe, i.e. market-based programmes that involve the exchange of pollution allocations between sources, need to be reviewed carefully (NEFCO,

2008; SEPA, 2009). Such systems allow polluters to “pay to pollute” rather than cleaning up their own effluent. When applied at large-scale in coastal and marine waters there is a risk of creating eutrophication hotspots in areas where measures to combat eutrophication are expensive.

Taking a more general view, the eutrophication problem is part of a global problem on overuse of nutrients that is harming not only aquatic ecosystems but also the terrestrial environment, air, soil and the greenhouse gas balance. Excessive nutrient input to agriculture and use in animal production causes eutrophication problems. The over-fertilization of agricultural fields is part of an international imbalance in nitrogen production and use. During the last century the nitrogen input into the environment in Europe has significantly increased, what has greatly contributed to the increased productivity of agricultural land and the largely self-sufficiency in cereals in the EU. But 40-70% of the N-fertilizer is lost to the atmosphere and hydrosphere during cereal production and only 10-50% of the nitrogen is retained in animal farming (Sutton et al., 2011). The increasing production of biofuel crops will probably enhance this problem and provoke even more increased losses for nutrients to the environment. This highlights the necessity for improved nutrient use efficiency in crop and animal production. This includes improving field management practices and yields per input, nutrient recycling and optimal use of nutrient in manure (storage and application), optimal application rates and times adapted to crop demands.

Further efforts to manage the nutrient cycle in a more cost-effective, sustainable and resource-efficient way, and to improve efficient use of fertilisers are urgently needed (see 7<sup>th</sup> Environmental Action Programme of the EU). In particular, stronger enforcement of existing legislation, a tightening of standards and more holistic and integrated approaches to reduce emissions have to be taken. But this requires higher demands on interdisciplinary and consensus building between science, policy and stakeholders. Given the extensive policy framework on nutrient management, better coordination and burden-sharing across sectors are expected avenues for improved implementation in the future.

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## Annex 1 Case studies on eutrophication abatement in Europe

In order to exemplify different eutrophication abatement strategies we have selected eight case study regions: Denmark, Finland, Po river basin, Ebro river basin, Rhine river basin, Lake Constance and the Danube river basin. Eutrophication of water bodies played a significant role in these regions and they provide useful examples of how the relevant authorities approached the pollution problem and developed and implemented eutrophication control programmes. The case study regions were selected in order to cover i) different European regions, ii) lakes, rivers, coastal and marine waters, iii) national and transboundary situations and iv) to show the interlinkages between freshwater and coastal / marine water management. With respect to data and information availability, less documentation on eutrophication management strategies were found for the Ebro and Po River than for the other case studies (Table A1.1).

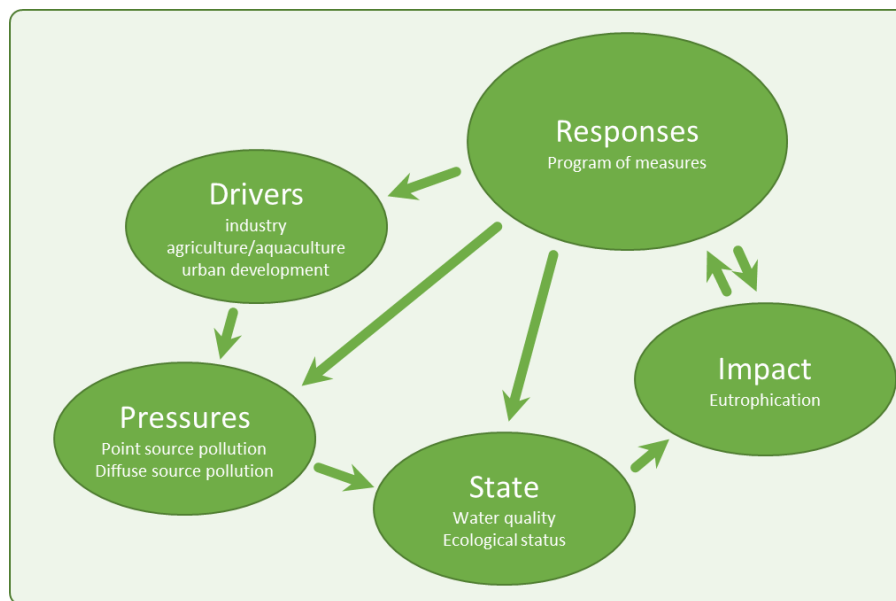
**Table A1.1: Overview of case study regions (RBD = River Basin District)**

Case study	Level	Suitability for this report	Available information
Danish Action Plan on the Aquatic Environment (APAE)	Country level	+	+
Baltic Sea Action Plan (BSAP)	Sea region	+	+
The Finnish Agri-Environmental Programme (FAEP)	Country level	+	+
The Po river, Italy	RBD level	+	+ (monitoring) - (measures)
Ebro River in Spain	RBD level	o	+ (monitoring) o (measures)
Rhine Action Plan 1987 – 2000 and Rhine 2020	RBD level	+	+
Lake Constance	Lake / water body level	+	+
Danube River Basin	RBD level	+	+

For the description of the case studies we followed the DPSIR assessment framework (EEA, 1999) that is illustrated in Figure A1.1. In the DPSIR framework the different steps are:

- The **Driving forces** are processes and anthropogenic activities (production, consumption, recreation etc.) able to cause pressures;
- The **Pressures** are the direct stresses, deriving from the anthropogenic system, and affecting the natural environment; typical indicators are nutrient loads from different sources
- The **State** reflects the environmental conditions of natural systems (e.g. water quality);
- The **Impact** is the measure of the effects due to changes in the state of environmental system;
- The **Response** is the evaluation of actions oriented to solve environmental problems in terms of management strategies.

**Figure A1.1: DPSIR assessment framework for eutrophication in freshwater, coastal and marine waters.**



**Source:** EEA, 1999; adapted by the authors.

The background of eutrophication, the management targets, development of programmes of measures and their implementation in the case study regions are examined in the following sections.

## 1 Denmark and the Danish Action Plan on the Aquatic Environment

### Background information

The Danish population is 5.6 million and the total area of the country is 43 321 km<sup>2</sup>. Intensive agricultural production caused problems of eutrophication in surface waters. The use of N fertilizer and manure is among the highest in Europe. The addition of surplus P to agricultural land started around 1900 and accelerated until the beginning of the 1980s.

### Driving forces, pressures and impacts

Danish inland and coastal waters have been under the influence of human activities since decades, leading to nutrient enrichment and changes in ecosystem structure, functioning and stability (Ærtebjerg et al., 2003; Nørring and Jørgensen, 2009). The discharge of urban and industrial wastewater directly to watercourses, lakes, and coastal areas increased during the past century, and agriculture became more and more intensive, with increasing inputs of nutrients from fertilizers and manure. Gradually, many aquatic habitats deteriorated, and in the 1970s and 1980s public and subsequently political concern arose (Danish EPA, 2000).

In Denmark, agriculture has always played an important role (Nørring and Jørgensen, 2009). More than 60% of the total area of Denmark is cultivated (Statistics Denmark, 2005) and livestock production and density is high, although with large geographical differences. In some parts of the country, animal density is very close to the upper limit allowed according to national legislation. Rieman et al., 2016 found that nutrient inputs from land were reduced by approximately 50% for nitrogen and 56% for phosphorus between 1990 and 2013 and these reductions led to a partial recovery of marine ecosystems. Nutrient and chlorophyll-a concentrations declined and eelgrass meadows expanded towards deeper waters in response to improved water clarity. Nevertheless, bottom oxygen concentrations did not improve,

presumably because more frequent stratification (due to a reduced summer wind speed) and higher water temperatures have counteracted the expected positive effects of reduced nutrient inputs.

## Responses – eutrophication management

Since 1985 Denmark has implemented a set of national measures to reduce agricultural nutrient pollution of water systems, especially to avoid eutrophication of coastal water (Danish EPA, 2000; Conley et al., 2002; Kronvang et al., 2005, 2008; Table A1.2). These measures have been in conformity with the European Union's Nitrate and Water Framework Directives, and in part, funded through the Common Agricultural Policy (CAP). The overall objectives of the policy approach were to ensure:

- that watercourses, lakes and marine waters are clean and of a satisfactory quality with regard to health and hygiene;
- that exploitation of water bodies and associated resources takes place in a sustainable manner;
- that the objectives of relevant international agreements will be fulfilled.

The primary means of achieving the quality objectives for surface waters was a reduction in nutrient discharges and emissions. In the January 1987 Action Plan on the Aquatic Environment and the April 1987 Report on the Action Plan on the Aquatic Environment the goal of reducing nitrogen and phosphorus loads to the aquatic environment was set to 50% and 80%, respectively. The reduction targets could not be reached in API and therefore were tightened in 1991. The reduction target was maintained but the time frame was extended to the year 2000.

The general objectives for Danish marine waters are based on the Consolidated Environmental Protection Act, the 1992 Helsinki Convention on Protection of the Marine Environment in the Baltic Sea Region and the 1992 Convention for the Protection of the Marine Environment of the Northeast Atlantic.

**Table A1.2: Policy measures to reduce agricultural nutrient pollution of the environment in Denmark**

Danish policy actions	Policy measures imposed
1985: NPo Action Plan to reduce N- and P-pollution	<ul style="list-style-type: none"> <li>• Target: general reduction of N and P</li> <li>• Minimum 6 month slurry storage capacity</li> <li>• Max. livestock density 2 LU/ha</li> <li>• Various measures to reduce runoff from silage clamps and manure heaps</li> <li>• Mandatory to establish a floating natural crust or artificial cover on slurry tanks</li> <li>• Ban on slurry spreading between harvest and 15 October on soil destined for spring crops</li> </ul>
1987: Action Plan on the Aquatic Environment I (Action Plan I)	<ul style="list-style-type: none"> <li>• Target: 49 % reduction of nitrogen leaching compared to the mid 1980s</li> <li>• Minimum 9 months slurry storage capacity</li> <li>• Ban on slurry spreading from harvest to November 1 on soil destined for spring crops</li> <li>• Mandatory to incorporate manure within 12 hours after application</li> <li>• Winter green fields required (percentage increasing from 45% to 65% through the period)</li> <li>• Mandatory fertiliser and crop rotation plans</li> </ul>

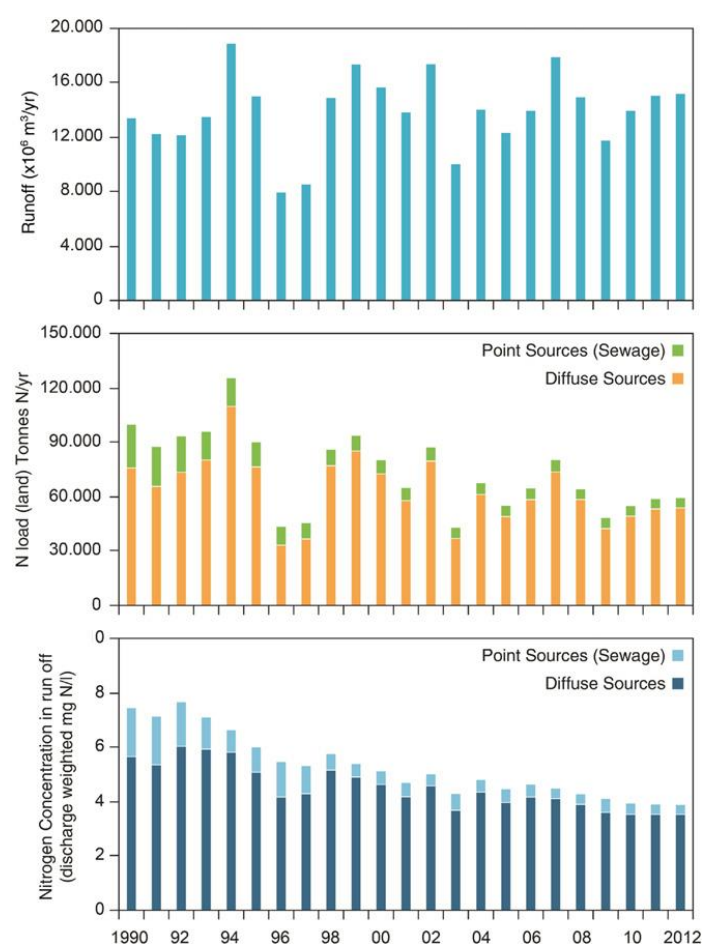
Danish policy actions	Policy measures imposed
<p>1991: Action Plan for Sustainable Agriculture (implemented to strengthen APAE I)</p> <p>1994: 10-Point Programme for Protection of Groundwater and Drinking Water</p> <p>1996: Follow-up on the Action Plan for Sustainable Agriculture</p>	<ul style="list-style-type: none"> <li>• Target: 49 % reduction of nitrogen leaching compared to the mid 1980s</li> <li>• Ban on slurry spreading from harvest to February 1st, except grass and winter rape</li> <li>• Compulsory fertiliser and manure accounts</li> <li>• Statutory utilisation rates for nitrogen in manure (pig slurry: 60 %, cattle slurry: 55 %, deep litter: 25 %, other types 50 %)</li> <li>• Winter green fields required (percentage increasing from 45% to 65% through the period)</li> <li>• Maximum limit on the plant-available nitrogen applied to different crops, equal to economic optimum.</li> </ul>
<p>1998: Action Plan on the Aquatic Environment II (Action Plan II)</p>	<ul style="list-style-type: none"> <li>• Target: 49% reduction of N-leaching compared to the mid 1980s</li> <li>• Mandatory catch crops (6% and 10% of the total area of the farm property depending on the amount of manure used per hectare). Catch crops included in fertilizer plan.</li> <li>• Nitrogen standards and norms with maximum limit on the plant-available nitrogen applied to different crops lowered to 10 % beneath economic optimal application rate.</li> <li>• Livestock density demands at 1.7 LU/ha for cattle and 1.4 LU/ha for pigs</li> </ul>
<p>2001: Midterm evaluation of Action Plan on the Aquatic Environment II</p> <p>2001: Action Plan for reducing Ammonia Volatilization from Agriculture</p> <p>2003: Final evaluation of Action Plan on the Aquatic Environment II</p>	<ul style="list-style-type: none"> <li>• Increased utilisation rates for manure through the period from APAE I to final rate in 2002 (pig slurry: 60 to 75 %, cattle slurry: 55 to 70 %, deep litter: 25 to 45 %, other types 50 to 65 %)</li> <li>• Improved animal feeding practice to improve utilization of feed</li> <li>• Tax on DKK 5 per kg nitrogen in fertiliser (farms are exempted if they register in the manure register – compulsory for most farms, but not possible for nonfarm nitrogen fertiliser users)</li> <li>• Reduced nitrogen standards for grassland and restrictions on additional N-application to bread wheat (2000)</li> <li>• Ban on slurry application by broadcaster spreader (2001)</li> <li>• Mandatory covering of all dung heaps</li> <li>• Slurry spread on bare soil must be incorporated within 6 hours (2001)</li> <li>• Optimisation of manure handling in animal housing</li> <li>• Ban on ammonia treatment of straw</li> <li>• Subsidies for: Forestation, conversion to organic farming, reduced nitrogen use in vulnerable areas, and establishment of wetlands</li> </ul>
<p>2004: Action Plan on the Aquatic Environment III</p> <p>2008: Mid-term evaluation of the Action Plan on the Aquatic Environment III considered further initiatives needed.</p>	<ul style="list-style-type: none"> <li>• Target: 13 % reduction of nitrogen leaching in 2015 compared to 2003 and 50 % reduction of phosphorous surplus in Danish agriculture by 2010.</li> <li>• Mandatory catch crops increased to 10 and 14 %</li> <li>• 50.000 ha 10 m. buffer zones along water courses and lakes before 2015</li> <li>• Improvement of utilisation of nitrogen and phosphorous in feed. For phosphorous encouraged by a tax on mineral phosphorous added to feed.</li> <li>• Based on research result further increase in utilisation rates for manure</li> <li>• 300 m protection zones around ammonia sensitive habitats</li> <li>• Strengthen and increase organic farming</li> <li>• Establish more wetland and support environmentally sensitive farming.</li> </ul>

Danish policy actions	Policy measures imposed
2009: Green Growth Agreement	<ul style="list-style-type: none"> <li>• 19,000 tonne reduction in nitrogen discharge to the aquatic environment from 2010 to 2015</li> <li>• 210 tonne reduction in discharge of phosphor to the aquatic environment from 2010 to 2015.</li> <li>• Permanent 10 m spraying-free, fertiliser-free and cultivation-free buffer zones</li> <li>• neutralisation of nitrogen effect when agricultural land is taken out of production</li> <li>• ban on ploughing grass fields at certain periods of the year</li> <li>• ban on certain forms of soil cultivation in the autumn</li> <li>• tightened regulation on existing APAE-catch crops and targeted use of catch crops</li> <li>• Wetlands and flooding of river valleys</li> </ul>

Source: Danish EPA, no date

### Status and effects of policy measures

**Figure A1.2: Temporal trends of run off (top), nitrogen loads (centre) and annual flow-weighted total N concentration in Danish surface water outflow to the sea (bottom) between 1990 and 2012.**



Source: Dalgaard et al., 2014.

The implementation of the Danish policy has shown great reductions in N- and P-discharges to the aquatic environment. Discharges of total nitrogen and total phosphorus from point sources have been reduced by 69% (N) and 82% (P) during the period 1989–2002 (Kronvang et al., 2005). In the intensive

Danish agriculture it has proven possible to reduce N-leaching by on average 33% and N-concentrations and loads in surface waters with on average 29–32% while maintaining at the same time crop yields and increasing livestock production significantly.

Significant reduction in N concentrations and total N loads could be observed in a study by Kronvang et al. (2008) (see Figure A1.2). The average reduction of total N concentration and total N load was 29 and 32%, respectively, for the period 1989–2004. In a study by Wiberg-Larsen et al. (2013, cited in Andersen et al., 2014) the temporal trends in N concentrations and loss in 53 streams draining agricultural catchments for 1989–2012 were analyzed. They found statistically significant downward trends in 51 streams with an average decrease in N loss of 51%.

In many Danish lakes an increased summer Secchi depth, decreased chlorophyll a and reduced phytoplankton biomass could be observed (Kronvang et al. 2005). But the observed and statistically verified reductions in nitrogen and phosphorus concentrations have led to only minor improvements at the ecosystem level (Andersen et al., 2014; Carstensen et al., 2006).

### **Lessons Learnt**

The implementation of the policy has shown great reductions in N- and P-discharges to the aquatic environment. The applied measures have shown remarkable effects on improved N-utilization, reduction of N-surplus and reduction of N-leaching. The effects were continuously monitored and the policy was adjusted. Nevertheless, a delay in catchment response was obvious.

## **2 The Baltic Sea and the Baltic Sea Action Plan**

### **Background information**

The Baltic Sea covers an area of 377,400 km<sup>2</sup> and is ecologically unique. The Baltic Sea drainage basin comprises 1,720,000 km<sup>2</sup> and can be divided into a northern boreal region that drains into the Gulf of Bothnia and a southern region that drains into the rest of the Baltic Sea. Due to its special geo-graphical, climatological, and oceanographic characteristics, the Baltic Sea is highly sensitive to anthropogenic impacts (Andersen et al., 2014).

### **Driving forces, pressures and impacts**

Human activities, both in the sea itself and throughout its catchment area, have over the last centuries put considerable pressure on its marine ecosystem (HELCOM, 2013a). First signs of eutrophication emerged in the mid-1950s and the central parts of the Baltic Sea changed from being unaffected by eutrophication to being affected (Andersen et al., 2015; HELCOM, 2013a). The Baltic Sea has changed from an oligotrophic clear-water sea into a highly eutrophic marine environment with wide-spread algal blooms.

Two studies have estimated the change in total nitrogen loading to the Baltic Sea over the last 100 years and found an increase by a factor of 2 or 2.4, respectively (Savchuk et al., 2008, Schernewski and Neumann, 2005). HELCOM estimated nutrient loads to the Baltic Sea via rivers and coastal point sources in 2012 to be 636,548 t TN yr<sup>-1</sup> and 28,656 t total phosphorus (TP) yr<sup>-1</sup> (<http://www.helcom.fi/baltic-sea-trends/indicators/inputs-of-nitrogen-and-phosphorus-to-the-basins/results/>). Rivers draining the southern cultivated part of the catchment account for the majority of the nutrient inputs to the Baltic Sea (HELCOM, 2004). A source apportionment for the total riverine TN loading of the Baltic Sea in 2000 reveals that natural background losses accounts for 28%, diffuse losses for 64%, and point source discharges for 8% (HELCOM, 2004). Agriculture accounts for 70–90% of the diffuse TN load (HELCOM, 2011). Almost 27% of the nitrogen load comes from atmospheric deposition (in 2012 235,100 t) (<http://www.helcom.fi/baltic-sea-trends/indicators/inputs-of-nitrogen-and-phosphorus-to-the-basins/results/>).



## Responses – eutrophication management

The Baltic Sea is surrounded by nine riparian countries with a further five in the catchment, making its management politically complicated. The International Council for the Exploration of the Sea (ICES), founded in 1901, documented severe environmental problems in the Baltic Sea in the 1970s (ICES, 1970). These led to the formulation and 1974 signing of the Helsinki Convention for the Protection of the Marine Environment of the Baltic Sea Area, which is governed by the Helsinki Commission (HELCOM). The eutrophication problems in the Baltic Sea were addressed in the 1988 “Declaration of the protection of the marine environment of the Baltic Sea Area” and plans to monitor, manage and reduce nutrient loads to the Baltic Sea by 50% before 1995 were adopted by the countries around the Baltic Sea (HELCOM, 1988). HELCOM revised the 1988 declaration in 1992 with the aim to accelerate the implementation of measures to substantially reduce nutrient loading. Following the revision, national plans have been adopted to combat the nutrient pollution.

As a consequence of the continuing non-satisfactorily ecological status of the Baltic Sea, HELCOM adopted The Baltic Sea Action Plan (BSAP) in 2007 (HELCOM, 2007). The BSAP determines maximum allowable nutrient inputs to each Baltic Sea sub-region if good ecological status is to be reached and sets country-wise nutrient reduction requirements based on a modelling approach that was agreed by all HELCOM Contracting Parties (revised in HELCOM, 2013a) (Table A1.3). The nutrient reduction targets consider both waterborne and atmospheric nutrient inputs and they account for nutrient inputs from upstream countries and for atmospheric nitrogen deposition from non-HELCOM Contracting Parties and shipping. Such explicit nutrient reduction targets enable a focussed nutrient management that explicitly addresses all polluters. Furthermore, HELCOM has established a scientifically robust, regular and transparent follow-up process to determine whether Contracting Parties have reached their reduction targets (<http://www.helcom.fi/baltic-sea-action-plan/progress-towards-reduction-targets/>). Concerning the implementation of measures to reduce nutrient inputs to the Baltic Sea it is up to each country to decide and implement these to reach the reduction targets set in the BSAP. Contracting Parties can also decide whether they reduce waterborne or airborne nutrient inputs. The individual measures taken in the Baltic Sea region are compared in greater detail in Andersen et al. (2014).

**Table A1.3: Country-wise nutrient reduction requirements (left) and maximum allowable inputs for Baltic Sea sub-regions as agreed in the HELCOM Baltic Sea Action Plan (Source: HELCOM 2013a).**

	Nitrogen	Phosphorus
Denmark	2890	38
Estonia	1800	320
Finland	2430 +600*	330 +26*
Germany	7170 +500*	110 +60*
Latvia	1670	220
Lithuania	8970	1470
Poland <sup>2</sup>	43610	7480
Russia	10380*	3790*
Sweden	9240	530
<i>The figures are rounded</i>		

Baltic Sea Sub-basin	Maximum Allowable Inputs		Reference inputs 1997-2003		Needed reductions	
	TN, tons	TP, tons	TN, tons	TP, tons	TN, tons	TP, tons
Kattegat	74,000	1,687	78,761	1,687	4,761	0
Danish Straits	65,998	1,601	65,998	1,601	0	0
Baltic Proper	325,000	7,360	423,921	18,320	98,921	10,960
Bothnian Sea	79,372	2,773	79,372	2,773	0	0
Bothnian Bay	57,622	2,675	57,622	2,675	0	0
Gulf of Riga	88,417	2,020	88,417	2,328	0	308
Gulf of Finland	101,800	3,600	116,252	7,509	14,452	3,909
<b>Baltic Sea</b>	<b>792,209</b>	<b>21,716</b>	<b>910,344</b>	<b>36,894</b>	<b>118,134</b>	<b>15,178</b>

<sup>1</sup> Finland's view is that according to HELCOM assessment open parts of the Bothnian Sea, Åland Sea and the Archipelago Sea are eutrophied and need reduction of nutrient levels, although BALTSEM model did not establish nutrient input reduction requirements to the drainage basins of these sea areas. Finland will address water protection measures to the drainage basins of these areas in its national plans.

<sup>2</sup> At this point in time Poland accepts the Polish Country Allocated Reduction Targets as indicative due to the ongoing national consultations, and confirms their efforts to finalize these consultations as soon as possible.

\* Reduction requirements stemming from

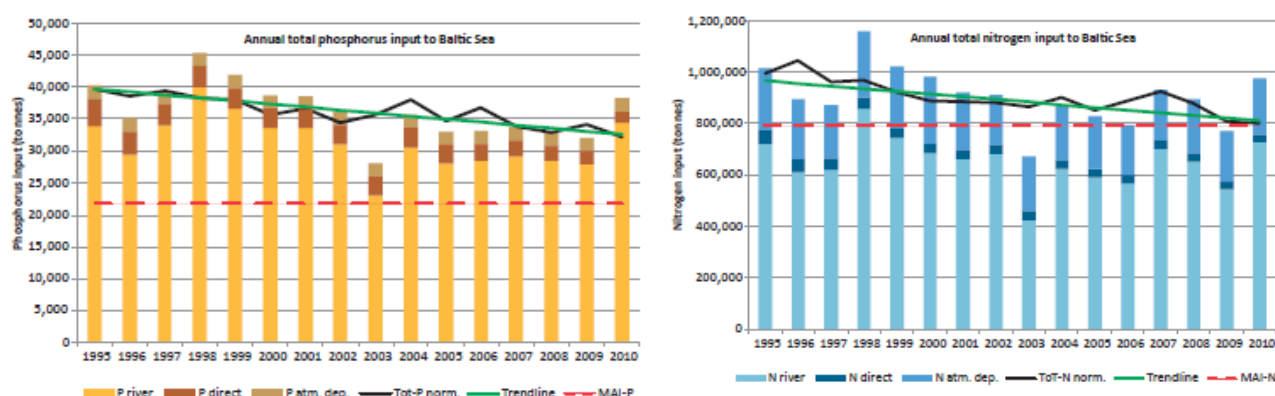
- German contribution to the river Odra inputs, based on ongoing modeling approaches with MONERIS;
- Finnish contribution to inputs from river Neva catchment (via Vuoksi river)
- these figures include Russian contribution to inputs through Daugava, Nemunas and Pregolya rivers

The figures for transboundary inputs originating in the Contracting Parties and discharged to the Baltic Sea through other Contracting Parties are preliminary and require further discussion within relevant transboundary water management bodies.

## Status and effects of policy measures

Joint efforts by the HELCOM contracting parties have had positive results including significant reductions in waterborne P loads, especially from point sources in the catchment area (Andersen et al. 2014), such that almost all HELCOM countries have achieved the 50% reduction target (Figure A1.3). However, the 50% reduction target for N reduction was not reached. Since the reference period in 1997–2003, which is the period the BSAP nutrient reduction targets relate to, phosphorus inputs have decreased by 13.6% and nitrogen inputs by 9.4% (based on data from 2010–12) (<http://www.helcom.fi/baltic-sea-trends/indicators/inputs-of-nitrogen-and-phosphorus-to-the-basins/results/>). Nevertheless, the inputs of nutrients, especially from diffuse sources such as agriculture, remain high and the Baltic Sea continues to suffer from the effects of eutrophication (HELCOM, 2009).

**Figure A1.3: Annual total (riverine, direct and atmospheric) inputs of nitrogen and phosphorus to the Baltic Sea between 1994 and 2010.**



**Notes:** Dark blue line = normalised total nutrient inputs; green line = trendline; red line = maximum allowable inputs.

**Source:** HELCOM, 2013a.

An assessment of the actions taken by countries to combat eutrophication (Figure A1.4) showed that all countries have tackled the problem of eutrophication and have initiated the implementation of the actions (WWF, 2013; based on reports of the HELCOM contracting parties). Still, out of the 10 actions analysed, only one action (the development of national programmes) had been fully implemented by all countries. Four actions are behind schedule and five have later deadlines. While measures to reduce nutrient loads from point sources, in particular the upgrading of wastewater treatment plants or the introduction of phosphorus free detergent, have made good progress measures that tackle excessive nutrient loads from agriculture are lagging behind. For those HELCOM Contracting Parties that have already sufficiently addressed point sources and can achieve larger reductions only by addressing diffuse sources it becomes increasingly difficult to maintain the downward trend in particular in waterborne nutrient inputs (reductions in airborne nutrient inputs are undertaken under the Gothenburg Protocol and show continued progress). In this respect it is interesting to observe that some of the latter Contracting Parties are pushing for “end of pipe solutions” to remove nutrients from the Baltic Sea that have already entered the sea (e.g. by mussel farming, removal of the top layer of the sediment, treating sediments with aluminium to bind phosphorus) or for technical solutions that counteract eutrophication effects (oxygenation of anoxic bottoms). Most of all, the yearly anthropogenic contribution of nutrients to the Baltic Sea is still much too high. While such measures may be justified since they can counteract the internal nutrient loads of the Baltic Sea (nutrients that have accumulated in the sediment and are now leaching) they also harbour the danger to compete with nutrient reduction measures at source.

**Figure A1.4: Assessment of the actions taken by HELCOM Contracting Parties to combat eutrophication.**

BSAP agreement	Action	Dead-line	DE	DK	EE	FI	LT	IV	PL	RU	SE
E-9	National programmes on nutrient reduction	2010	0	0	0	0	0	0	0	0	0
E-5	Actions to reduce nutrient load undertaken	2016	0	-1	0	0	0	-1	0	0	0
E-10	Inclusion of BSAP targets in national programmes	2009	0	-3	0	0	-2	0	-2	-2	0
E-11, E-12	Advanced municipal waste water treatment	2018	0	-1	0	0	0	0	0	0	0
E-11, E-12	On site treatment for scattered settlements (transitional)	2017	0	-1	0	0	-1	-1	-1	0	0
E-11, E-12	On site treatment for scattered settlements (final)	2021	0	-1	0	0	-1	-1	-1	0	0
E-13	Substitution of phosphorus in detergents	2010	0	0	-1	0	-2	0	0	-2	0
E-16	Designation of zones vulnerable to nitrogen	2021	0	-1	0	0	1	0	0	0	1
E-17	Permit systems for animal farms	2012	0	0	0	0	-3	-2	0	-2	0
E-19	List of agricultural hot spots	2009	0	0	-2	0	-3	0	-2	-2	-3
<b>Total score</b>			<b>0</b>	<b>-8</b>	<b>-3</b>	<b>0</b>	<b>-11</b>	<b>-5</b>	<b>-6</b>	<b>-8</b>	<b>-2</b>

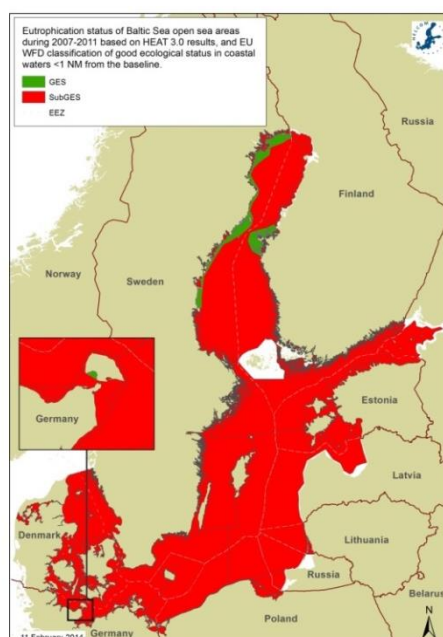
  

<b>Deadline not passed yet:</b>	<b>Deadline already passed:</b>
<b>1</b> Action implemented ahead of time	<b>0</b> Action implemented
<b>0</b> Implementation in progress	<b>-2</b> Implementation in progress
<b>-1</b> No action or unreported	<b>-3</b> No action or unreported

Source: WWF, 2013.

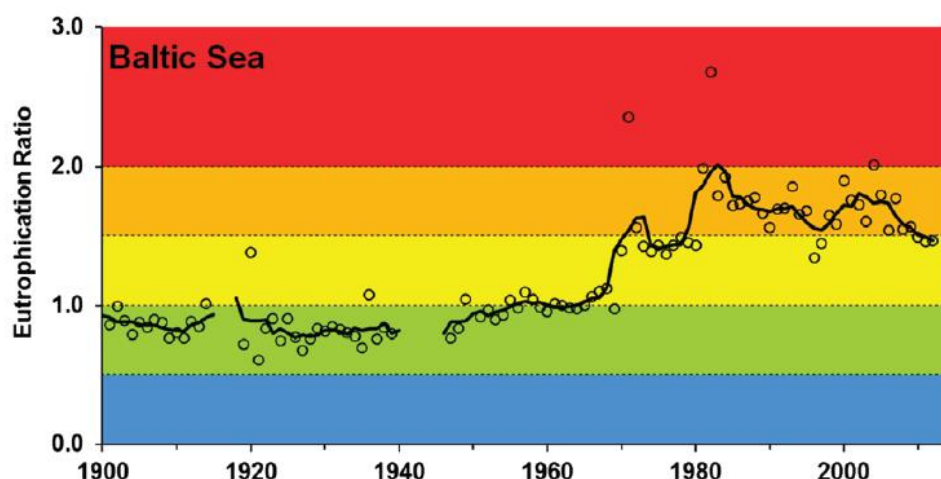
According to the latest HELCOM assessment on eutrophication (HELCOM 2014), in 2007-2011 almost the entire open Baltic Sea was assessed as being eutrophied and only the open Bothnian Bay was assessed as being unaffected by eutrophication (Figure A1.5). This result indicates that despite measures taken to reduce external inputs of nitrogen and phosphorus to the sea, good status for eutrophication has not been reached until today. Nearly the entire sea area is still affected by eutrophication. Nevertheless, a long-term study on the trend in eutrophication status of the Baltic Sea has shown that the status is slowly improving in recent years (Andersen et al. 2015; Figure A1.6).

**Figure A1.5: Eutrophication status in 2007-2011 was assessed as being affected by eutrophication in the entire open Baltic Sea (red colour, status less than good; sub-GES). Outer coastal waters of the Quark (Finland) and in Orther Bucht (Germany) were assessed by national experts to be in good ecological status).**



Source: HELCOM, 2014

**Figure A1.6: Integrated assessment of eutrophication in the Baltic Sea 1901-2012, combining 621 individual classifications of eutrophication status into a single assessment.**



**Notes:** The solid line is the 5-year average. Blue = high status, green = good status, yellow = moderate status, orange = poor status, red = bad status.

**Source:** Andersen et al. 2015.

### Lessons Learnt

Model predictions of recovery of the Baltic Sea show that once the BSAP nutrient reduction targets (HELCOM, 2007) are met, this will have a positive effect on the status of the Baltic Sea ecosystem and that nutrient concentration levels will decline during the following decades. While HELCOM has been successful in setting quantitative nutrient reduction targets incorporating all relevant nutrient sources and in maintaining a transparent follow-up process it has been less successful in ensuring that adequate measures are undertaken. In conclusion, the policy is working but the environmental objectives have not been reached until today. It will still take a long time to reach the nutrient reduction targets of the BSAP and good eutrophication status (HELCOM, 2013a).

## 3 The Finnish Agri-Environmental Programme (FAEP)

### Background information

Finland is rich in surface waters, with a total of ca. 187 000 lakes and ponds larger than 500 square metres, and rivers totalling 25 000 kilometres in length. Almost a tenth of the country's land area is covered by water. Many rivers of Finland are still largely in a pristine condition. However, nutrient loading has caused eutrophication and deterioration in the state of Finnish inland waters (esp. lakes) and the Baltic Sea since around the 1970s.

The latest ecological status assessment of Finnish waters from 2013 ([www.syke.fi](http://www.syke.fi)) showed that 85% of the surface area of the lakes and 65% of that of the rivers were in good or excellent condition (status is good or high). On the other hand, as much as three fourths of the surface area of Finnish coastal waters was in poorer condition. Eutrophication was assessed to be the most significant problem.

### Driving forces, pressures and impacts

Agriculture was identified to be the largest single source of nutrients into surface waters in Finland (Granlund et al., 2005 and literature cited therein). Mitikka and Ekholm (2003) reported that lakes with agricultural impact were the most eutrophied of the Finnish lake types. On the national scale, approximately 50% of the total nitrogen and 60% of total phosphorus loading originate from agricultural

sources (Silvo et al., 2002). The percentage of agricultural land is only 9% of the total land area of Finland, but in many drainage basins it exceeds 30% especially in the south and south-west. Sewage from dwellings accounts for approximately 20% and industrial sources approximately 5% of the total N and total P discharges into the water bodies.

### **Responses – eutrophication management**

The Finnish Agri-Environmental Programme (FAEP) formed the most important policy measure and was implemented when Finland joined the European Union in 1995. FAEP is part of the Common Agricultural Policy (CAP) of the European Union, is considered to be a major package of policy instruments to encourage farmers to protect and enhance the environment on their farmland, through payments for the costs of provision of environmental services. The goal of Finnish Agri-Environmental Programme (FAEP) was to ensure the change in agricultural practices towards higher sustainability including eutrophication objectives on reduction of pressures on surface waters, groundwater and air.

The FAEP consists of three different levels of measures (Valpasvuo-Jaatinen et al., 1997): (1) basic measures that have to be adopted by all farmers participating in the programme, (2) additional optional measures, of which one has to be chosen and (3) special voluntary measures for which additional support is given. As to controlling P losses, the basic measures require the preparation of a farm environmental management plan, the balanced fertilization of arable crops (involving crop-specific fertilizer limits that should not be exceeded) and implementation of 1–3 m wide filter strips along the main ditches and watercourses. Additional measures for arable farming include an option to comply with a stricter fertilisation practices based on crop, expected yield and soil-test P and an option to increase plant cover in winter or apply reduced tillage. The special measures require more efficient environmental protection, e.g. establishment and management of riparian zones, wetlands or sedimentation ponds. Nowadays, the FAEP is also the main tool within EU Water Framework Directive (WFD) to control the nutrient load from agriculture. Farmer participation in the FEAP was extensive. In March 2010, about 58 100 farms, or 90% of all farms (64 000), were committed to the basic measures defined for the programming period 2007-13 (Aakkula et al., 2012).

In 1998, the Finnish Council of State issued a Decision-in-Principle on the water protection targets to 2005. By the year 2005, P and N loads from field cultivation should be **reduced by 50%** from the estimated loads at the beginning of the 1990s.

### **Status and effects of policy measures**

An evaluation of the FAEP in 2004 (Ekholm, 2005) revealed that the FAEP has substantially affected farming in Finland. It had i) increased the area of filter strips and riparian zones; ii) increased the plant cover on arable land in winter; iii) decreased the number of animals per hectare; and iv) decreased the use of P and N in chemical fertilizers.

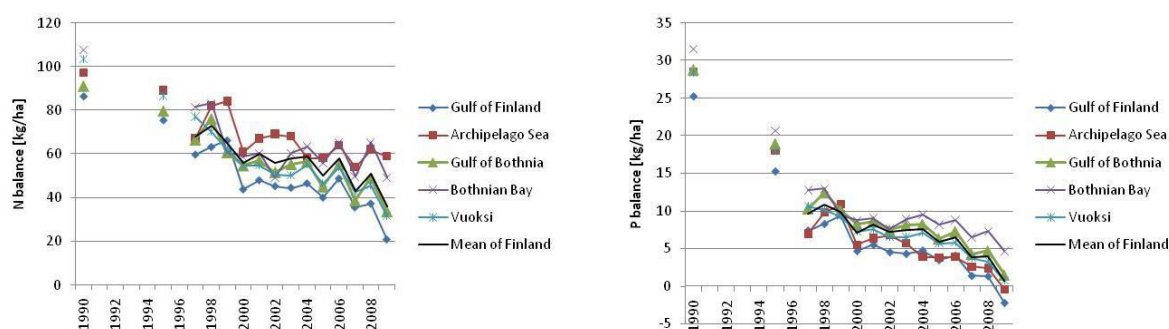
However, during the first years of FEAP the effects on water quality were not pronounced (Mitikka et al., 2005; Granlund et al., 2005). Several studies on water quality changes between 1992 and 2002 did not show clear and significant trends. Studies carried out in agricultural catchments confirmed only little or no effects on nitrogen leaching. In 2005 Granlund et al. (2005) concluded that the water protection targets assigned to agricultural nutrient loading (50% reduction for annual N and P loads from field cultivation by the year 2005) were far too optimistic. Ekholm et al. (2015) found that agricultural TP load decreased, whereas TN load generally increased between 1985 and 2006, despite the agri-environmental measures implemented since the start of Finnish Agri-Environmental Programme in 1995. At least, FAEP has significantly inhibited an increasing intensity of agricultural production. Even in southern Finland, the agricultural areas with intensive production would have produced significantly more nutrient leaching to the watercourses without the agri-environmental scheme (FAEP) (Lehtonen and Rankinen, 2015).

Only recently, some indicators show a more positive trend. The results of Aakkula et al. (2012) show significantly decreased nutrient field balances of nitrogen and phosphorous (Figure A1.7), and the FAEP is



concluded to play a major role in this development. Some reduction of nutrient leaching is reported which, however, is not considered to be sufficient with respect to the targets of water quality improvement.

**Figure A1.7: Change in field nutrient balances in Finland between 1990 and 2010 for nitrogen (left) and phosphorus (right).**



Source: Aakkula et al., 2012.

### Lessons Learnt

The results of Granlund et al. (2005) indicate that the Water Protection Targets assigned to agricultural nutrient loading were very optimistic. More than 90% of the Finnish farmers are participating in the FAEP, but the slow response of water quality shows that it may take years to achieve a clear reduction in nutrient loading. It is obvious that within a short time period (5–10 years) a possible decrease in potential loading may be masked by variations in the hydrological regime (Mitikka et al., 2005). Additionally, P is strongly bound by soil and the large P pool in soils, built up during decades of 'over-fertilization', provides a considerable potential for P losses due to leaching and erosion. It can be concluded that no dramatic changes in water quality in agricultural catchments is to be expected within a shorter period of 5–10 years.

The purpose of the agricultural support payments in FAEP has been to maintain farm incomes and keep land in good agricultural condition. This, however, implies direct or indirect production incentives. For example, large scale set-aside or even abandoning less favoured farmlands which might lead to reduced nutrient leaching without decreasing overall agricultural production, has not been a policy target. This might be a reason that nitrogen leaching in relatively less intensive regions did not decrease significantly (Lehtonen and Rankinen, 2015).

## 4 The Po river basin, Italy

### Background information

The longest river in Italy, the river Po, flows eastward across northern Italy from the Cottian Alps to the Adriatic Sea near Venice (Pirrone et al., 2002). The 652 km long river has a 74 000 km<sup>2</sup> drainage area of which 41 000 km<sup>2</sup> is mountainous and the remaining 29 000 km<sup>2</sup> is located in the Po valley encompassing the lowland plain of rich soil. There are 450 lakes in the drainage basin, and 141 tributaries add to the river that discharges an average 1540 m<sup>3</sup>/s into the Adriatic Sea through a wide delta. The amount of water discharged by Po is approx. 50% of the total freshwater input to the northern Adriatic Sea.

### Driving forces, pressures and impacts

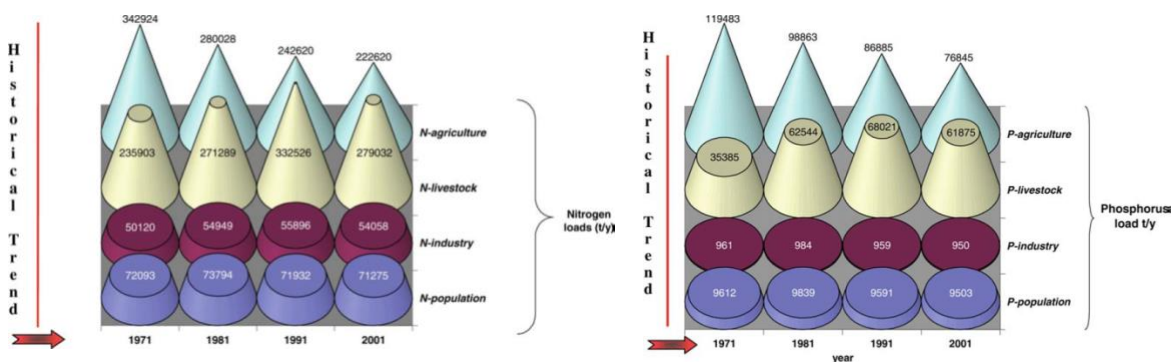
The high level of regional development with important agricultural, industrial and touristic sectors has put heavy pressure on water resources and led to degradation of surface and groundwater quality. More



than 40% of the Italian workforce is employed in this region and produces nearly 40% of the national GDP. 55% of Italian livestock is found here and 35% of the agriculture production takes place in this region (Pirrone et al. 2002). The principal farming areas are localised in the Po valley, covering 45% of the basin's total area. Most of the agricultural land in the Po valley is arable land, drained by artificial ditches, and 50% of agricultural land is irrigated during summer.

Excessive nutrients and organics in surface water caused eutrophication in rivers and in lakes. Groundwater resources contain high concentrations of nitrates due to fertilizer use in agriculture. Around the half of the nitrogen loads in the river Po originate from diffuse sources (48% agricultural and livestock activities) (De Wit and Bendoricchio, 2001; Pirrone et al., 2005; Figure A1.8), while for phosphorus the most significant contributions come from WWTP and urban systems (Salveti et al., 2006). When the water discharges into the Adriatic Sea the high nutrient load leads to eutrophication of the coastal waters (Pierluigi et al, 2015).

**Figure A1.8: Nitrogen loads (left) and phosphorus loads (right) between 1971 and 2001 calculated for each driving force acting in the Po river basin.**



Source: Pirrone et al. 2005.

### Responses – eutrophication management

A set of water management actions as well as flood risk management plans was adopted and implemented in all regions of the Po river basin (an overview of the law collection is given in Palopoli et al. 2002). Important legislative measures were implemented already in the 1980s, such as those aimed at the reduction of phosphates in detergents (Italian Ministry Decree 7/1986) and the improvement of wastewater treatment systems (Italian Law 319/1976). These measures were effective mainly for urban and industrial point sources, while the loadings from nonpoint agricultural and livestock sources remained high.

A multitude of Italian laws concerning water protection (see Palopoli et al. 2002) has been ordered by the **152/1999 Legislative Decree**, which transferred the European Directives 91/271/EEC (UWWTD) and the 91/676/EEC (ND) into national law. By the introduced measures the water bodies are expected to reach a good environmental status for surface and groundwater by December 31, 2016. The key contents of the 152/1999 Legislative Decree were (Palopoli et al. 2002):

- to define water quality goals with a particular attention for the water fauna;
- to define emission level limits;
- to define some policies to preserve, recycle and depurate water resources;
- to implement sewage systems.

An **eutrophication control plan** was adopted in 2001 that deals in particular with eutrophication problems in large Alpine lakes and the coastal waters of the Adriatic Sea. As a target, maximum admissible phosphorous concentrations were formulated (PRBA, 2002; Fresia and Mitidieri, 2005).

In 2009 all these plans were homogenised into the integrated river basin management plan at district level according to the WFD (Balzarolo et al., 2011). The **Po River Basin Plan** includes a program of measures to reduce and prevent pollution and to finally achieve good ecological status. The PoM distinguishes between measures from previous plans, such as the regional Water Protection Plans, and new measures (EC, 2012a).

### Status and effects of policy measures

The reactive N and P loadings steeply increased from the 1960s to the early 1980s (Table A1.4), raising the awareness of a causal link between P and N sources and loadings, and eutrophication. Since the implementation of nutrient reduction measures a significant decrease in the loadings is obvious (Pierluigi et al. 2015).

**Table A1.4: Main trend of reactive nitrogen and phosphorus loadings delivered by the Po river to the Northern Adriatic Sea. Q mean annual discharge rate (m<sup>3</sup>/s), DIN annual dissolved inorganic nitrogen loading (t N year<sup>-1</sup>), DIP annual dissolved reactive phosphorus loading (t P year<sup>-1</sup>), N/P molar ratio of DIN and DIP loadings.**

Year	Q	DIN	DIP	DIN/DIP
1968–70	1,378	53,028	1,825	64
1982–87	1,394	106,562	5,319	44
1995–96	1,596	136,100	3,310	91
2003–07	899	75,533	1,646	102

Source: Pierluigi et al. 2015

At the outlet of the Po river, the average concentrations of NO<sub>3</sub>-N decreased between 1998 and 2001 from 2.97 to 2.75 mg l<sup>-1</sup>, while NH<sub>4</sub>-N decreased from 0.169 to 0.10 mg l<sup>-1</sup> (during the period of the implementation of D.L. 152/1999). The concentrations of Total P remained constant (Pirrone et al., 2005). Despite these improvements in water quality eutrophication of the water bodies in the Po river and the Adriatic Coastal Zone is still a relevant environmental problem.

In 2003 the whole river basin was assessed to be in a moderate or poor ecological status, except a few tributaries (PRBA, 2006). In 2009, out of 2038 surface water bodies in the basin, around one third failed the WFD objective of good ecological status while for around half of the water bodies the status has not been determined according to WFD guidelines (EC, 2012). From these water bodies, 23.9% were assessed to be under significant pressure from point sources and 38.4% under significant pressures from diffuse sources.

## 5 Ebro River in Spain

### Background information

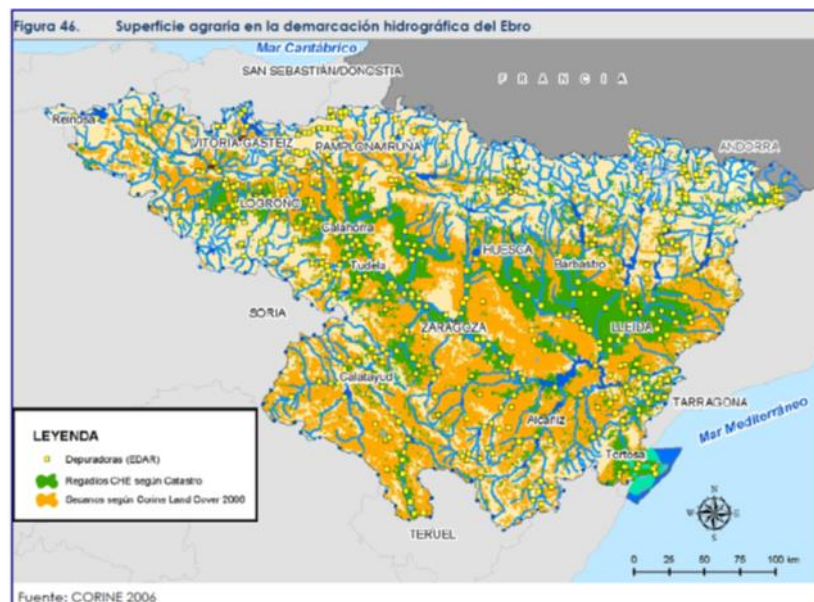
The Ebro basin is located in the NE of the Iberian Peninsula, with a surface of 85 550 km<sup>2</sup> (Comin, 1999; Ibáñez et al., 2008). The main river is 928 km long and its main tributaries are the Segre, Aragon, Cinca, and Gallego rivers. The mean annual flow near the mouth (Tortosa) was 592 m<sup>3</sup>/s at the beginning of the century. However, there is a continuous decreasing tendency since the 1970's (to ca. 400 m<sup>3</sup>/s) due to increasing water usage (Ibáñez et al., 2008). The population in the basin is about 3 million people, with a density of 33 inhabitants /km<sup>2</sup>.

### Driving forces, pressures and impacts

The main land use is agriculture, with more than 1 million hectares of irrigation (90 % of water usage in the basin) and ca 3.9 million hectares of total agricultural area (Ebro River Basin Authority, 2013). The basin has been strongly regulated by the construction of ca. 200 dams and reservoirs, most of them built

between 1940 and 1970. The ecology of the lower Ebro (and as well hydrology and geomorphology) are strongly impacted by the existence, features and operation of these dams and reservoirs. Agricultural non-point sources account for 64% of NO<sub>3</sub> loads generated within the Ebro basin while urban and industrial point sources are responsible for 88% of PO<sub>4</sub>-P and 71% COD loads (Torrecilla et al., 2005).

**Figure A1.9: Agricultural land use in the Ebro River Basin (green = irrigation farming; yellow = dry farming; yellow dots = WWTPs).**



**Source:** Ebro River Basin Authority, 2015.

Eutrophication of the Ebro river started around the 1960s and 1970s (Ibáñez et al., 1995; 2008). The increasing population, economic activity and per capita water consumption during this period has also contributed to the nutrient enrichment of the lower Ebro River and delta. Sewage collection systems were installed in most Spanish localities during the 1950s and 1960s, and consequently rivers received the raw wastewater (including significant amounts of nutrients) of an increasing population. Industrial wastewaters were added to these urban sewages since the 1970s. Since the 1990s, there is a trend of reduced nutrient loading likely due to improved water treatment and land management (Torrecilla et al., 2005), but the response of the lower Ebro River and its estuary to nutrient reduction (higher water transparency, macrophyte spreading, etc.) has not yet been established.

### Responses – eutrophication management

The Ebro River Basin Authority (Ebro Hydrographic Confederation) which was already founded in 1926 carried out water management and planning for the whole catchment. Early water management plans before the 1940s were directed towards the development of irrigation and agriculture while the first plan prepared under the WFD implementation process included (amongst others) objectives to protect water quality, to prevent the deterioration of water quality status and to achieve the good ecological / good chemical status by December 2015 (Bielsa and Cazcarro, 2015). The Ebro River Basin Management Plan (or Hydrological Plan) 2010-2015 responds to the EU WFD and establishes more demanding water quality objectives than in previous times. At the same time the plan responds to the Spanish Water Act, which set the regulations to fulfil the water demands for different sectors. The Water Act also covers water quality, but also follows the EU WFD, which has been incorporated into the Spanish Water Act. The Ebro River Basin Management Plan has 12 key elements, and includes the ambitious environmental objective that at least 83% of river water bodies will attain the good ecological status by 2015.

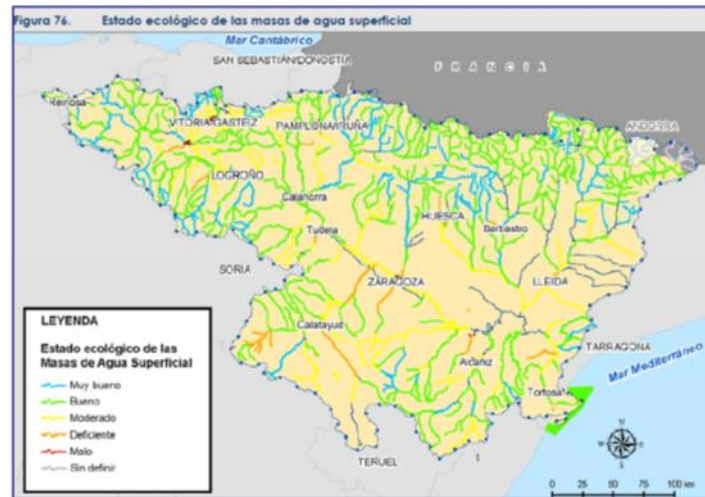
The Hydrological Plan for the Ebro River Basin 2010-2015 includes 131 measures to reach the water quality objectives, to improve sanitation and water treatment and makes reference to the measures

implemented under the EC Nitrates Directive (Ebro River Basin Authority, 2010). The detailed measures are set in in the regional plans of nine autonomous regions of Spain (e.g. Valencia, Cantabria, Castilla y León).

### Status and effects of policy measures

The ecological assessment of surface waters following the requirements of the WFD identified 694 surface water bodies in the entire Ebro River basin with ca. 71 % showing very good / good ecological status and ca. 29% failing the WFD objectives due to point and non-point pollution (Ebro River Basin Authority, 2015).

**Figure A1.10: Map of the ecological status of surface waters in the Ebro river basin.**



**Source:** Ebro River Basin Authority, 2015.

The construction of treatment facilities in the main villages in the early 90's (particularly in Tudela and Zaragoza in 1992), which was triggered by EU Directives, caused a very clear decrease in PO<sub>4</sub>-P content. Phosphate concentrations increased in the 1980s and showed a maximum at the beginning of the 1990's (mean annual maximum of about 0.8 mg/L), with a steady decrease until 1996, followed by a rather stable period of lower values until 2004 (mean annual values about 0.2 mg/l) (Ibáñez et al. 2008). Like phosphorus, ammonium also showed increasing values until the beginning of the 1990's and a statistically significant decrease since 1996, with a maximum annual value of 0.5 mg/L and a minimum annual value of 0.06 mg/L in 1993 and 1999, respectively. In contrast to this, nitrate did not show a clear trend between 1987 and 2004.

### Lessons Learnt

The implementation of measures to reduce nutrient input from point sources had clearly improved surface water quality in the Ebro river basin. At the moment, no conclusion can be drawn on the implementation of the water management plan 2010-2015 and the effects on eutrophication.

A Commission staff working document on the reporting of countries under the EU WFD (EC, 2015b) states that "it is complex or impossible to understand how the Programmes of Measures are linked and respond to the identified pressures and to the status assessment, and how the measures ensure the achievement of objectives. The measures to satisfy water demand – which use on average nearly half of the PoMs budgets - are not targeted to the WFD objectives.... According to aggregated information provided by Spain, measures addressing the WFD environmental objectives make up 46% of the PoMs budgets, measures for water supply 42%, floods and droughts 9% and 3% is targeting knowledge and governance. Among the measures considered by the RBMPs as contributing to the environmental objectives, there are many for which their contribution to achieve good status is unclear. In particular

the modernisation of irrigation takes a significant percentage of the budget of the measures to achieve environmental objectives, but its contribution is generally not assessed and not quantified“.

## 6 Rhine Action Plan 1987 – 2000 and Rhine 2020

### Background information

The Rhine is the largest river in north-western Europe with a total length of 1320 km. Its basin of 197,100 km<sup>2</sup> covers significant parts of Switzerland, Austria, Germany, France, Luxemburg, Belgium and The Netherlands, as well as small parts of Italy and the whole of Liechtenstein. About 60 million inhabitants live in the basin area (ICPR, 2005, 2015).

### Driving forces, pressures and impacts

The river Rhine was under considerable anthropogenic pressure since the middle of the 19th century. Major European industrial areas are located in the Rhine basin, including the Ruhr-area in Germany and the Rijnmond area near Rotterdam. The river itself is used intensively for shipping, waste disposal, drinking water production, and irrigation. In the years 1960–1980 water and sediment of the Rhine was so contaminated with pollutants – salts, nutrients, heavy metals, organotin compounds, and pesticides – that it could not be used neither for land reclamation nor farming nor drinking water purposes without costly treatment (Wilken, 2006). The Rhine used to be a very important salmon river, but in the 1930s salmon stock dwindled due to dam building, overfishing and deteriorating water quality.

### Responses – eutrophication management

After the Sandoz accident in Basle in November 1986, the states bordering the River Rhine agreed on the **Rhine Action Programme** for its ecological rehabilitation (ICPR, 2003). This programme of 1987-2000 had the following four aims, which should be realized by the year 2000: (1) to create conditions which will enable the return of higher species (such as salmon); (2) to safeguard Rhine water as a source for the preparation of drinking water; (3) to abate the contamination of sediments due to toxic compounds; and (4) to fulfil the requirements of the North Sea Action Plan, as the River Rhine flows into the North Sea. Water quality criteria have been developed for about 50 contaminants or contaminant groups and for phosphorus and ammonium. The main pillars of the Rhine Action Plan with regard to eutrophication were:

- Formulation of quantitative water quality objectives for the whole length of the Rhine.
- Emission reduction of 50% by 1995 compared to 1985 for 43 substances and groups of substances, based on the application of the state of the art in technology for different branches of industry.
- Elaboration of a draft inventory of diffuse pollution sources and a schedule for reducing diffuse pollution.
- Development and implementation of plans regarding hydrological, biological and morphological improvements

The subsequent management program “**Rhine 2020**” focused on ecology, nature protection, flood prevention and ground water protection. The program was adopted in 2001 by the Rhine Ministers and was seen as an implementation of the requirements of the EU WFD and the Swiss water policy. The main objectives of the program “Rhine 2020” with regard to eutrophication were

- Securing the standard in sanitation and wastewater treatment that already has been reached;
- Rehabilitation of the network of typical biotopes for the river Rhine and its connected wetlands;
- Water constituents should not have a negative effect on the aquatic communities;
- Meeting of the water quality norms for the river Rhine in the long-term perspective.

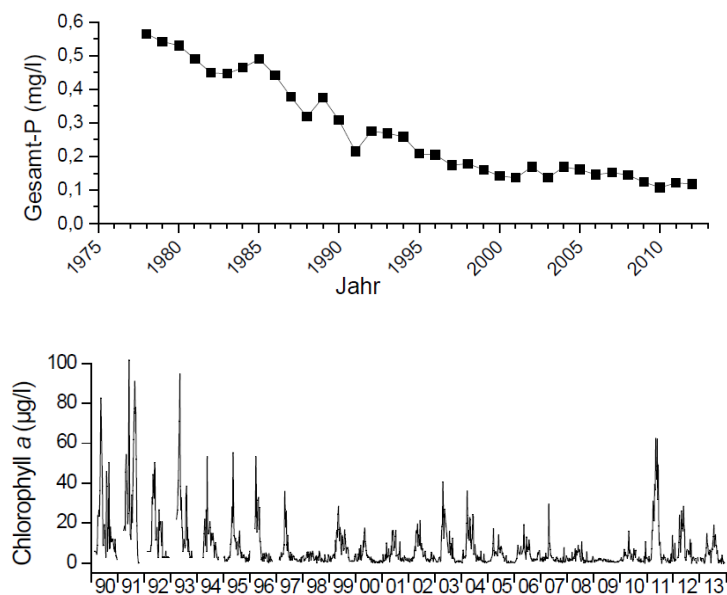


## Status and effects of policy measures

The management programmes and the coordination mechanism in the river Rhine basin – the International Commission for the Protection of the River Rhine, ICPR - can be regarded as a success story in European water management and international cooperation (Frijters and Leentvaar, 2003; Mostert, 2009). In particular the discharges of noxious substances from point sources (municipalities and industry) fell significantly. In 1985 only about 85% of the inhabitants were connected to urban wastewater treatment plants while in 2000 the percentage has increased to 95% (ICPR, 2003). The nitrogen loads to the river decreased between 1985 and 2000 from ca. 500,000 tons to 360,000 tons. However, the objective to half the loads by 50% has not been achieved within the Rhine Action Program until 2000. Diffuse nitrogen inputs as a result of leaching from agricultural fertilized soil via drainage and the ground water almost remained unchanged.

The concentrations of nutrients decreased significantly from the 1970s until today (ICPR, 2003; Friedrich and Pohlmann, 2009; Hardenbicker et al., 2014). The mean average concentrations of total phosphorus at the measuring site “Koblenz” decreased from 0.56 mg/l in 1978 to 0.12 mg/l in 2012 (Figure A1.11). While at the beginning of the 1990s phytoplankton peaked and concentrations of 80-100 µg/l chlorophyll-a were measured, such events did not occur since that time (Figure A1.11). It is assumed that rehabilitation of the River Rhine and grazing by mussels (*Dreissena sp.* and *Corbicula sp.*) additionally played an important role to re-establish clear waters (Weitere and Arndt, 2002).

**Figure A1.11: Concentrations of total phosphorus in the river Rhine (mean annual values) at the measuring site Koblenz between 1978 and 2012 (top). Concentrations of chlorophyll-a at the measuring site Koblenz between 1990 and 2013 (bottom).**



**Source:** ICPR, 2015.

The assessment of phytoplankton communities in the River Rhine according to the requirements of the EU WFD in 2012 (IKSR, 2015), resulted in a “very good” assessment for the Upper and Middle Rhine until the measuring site “Mainz-Wiesbaden”. More downstream the ecological quality based on phytoplankton changed to “good” and further downstream from the measuring site “Duisburg” it changed to “moderate” (ICPR, 2015). The transitional zones were generally assessed to be in a “good status” with stable results in the coastal zones and more variable results in the Wadden Sea (Figure A1.12).



**Figure A1.12: Ecological assessment of the coastal and transitional zones of the River Rhine based on a Dutch phytoplankton assessment methodology using Chlorophyll-a and *Phaeocystis*-blooms; blue = very good, green = good, yellow = moderate, red = poor status.**

Mess-station	Wasser-körper	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012
Noord-wijk 2	Holl. Küste	0,54	0,53	0,61	0,84	0,62	0,86	0,55	0,60	0,81	0,64	0,66	0,92
Ter-schel-ling 4	Watten-meer-küste	0,64	0,75	0,63	0,49	0,39	0,85	0,60	0,52	0,63	0,66	0,46	0,68
Dantzig-gat	Watten-meer Ost	0,41	0,48	0,47	0,47	0,54	0,51	0,52	0,24	0,52	0,44	0,50	0,48
Doove Balg West	Watten-meer West						1,00	0,56	0,69	0,61	0,74	0,65	0,72
Mars-diep Noord	Watten-meer West										0,68	0,56	0,74
	Watten-meer gesamt	0,41	0,48	0,47	0,47	0,54	0,76	0,54	0,47	0,57	0,62	0,57	0,65

Source: ICPR, 2015.

### Lessons learnt

Despite the fact that the Rhine Action Plan 1987-2000 was not legally binding, most of its emission reduction goals have been achieved. The success of the Rhine Action Plan is often explained by the more business-like and non-hierarchical approach that was adopted. Some clear success factors were that (1) the targets had been defined with precision, that (2) all Rhine bordering countries implemented their decisions and reported periodically on progress and deficits. Mostert (2009) argues that European co-operation in economic and other matters facilitated the conclusion of agreements on Rhine water quality and increased the costs of not reaching agreement. Moreover, the European water directives have been very influential from the 1970s onwards. Mostert (2009) concludes that there is no simple explanation for the water quality improvement of the Rhine, only a complex one. In addition to the ICPR and the non-legally binding Rhine Action Plan, mention should be made of the European Union and its binding directives, other international fora, such as the North Sea Ministers Conferences, domestic legislation, the activities of environmental NGOs and the waterworks in the Rhine basin, growing environmental awareness, technological innovation and changes in the structure of the industry in the basin.

## 7 Lake Constance

### Background information

Lake Constance is after Lake Geneva and Lake Balaton the third biggest inland lake in Central Europe with a catchment area of 11 500 km<sup>2</sup>. The three riparian countries Germany, Austria and Switzerland share sovereign rights without any fixed borders on the lake (IGKB, 2004). Lake Constance' natural trophic status is a nutrient-poor, oligotrophic pre-alpine lake (IGKB, 1985).

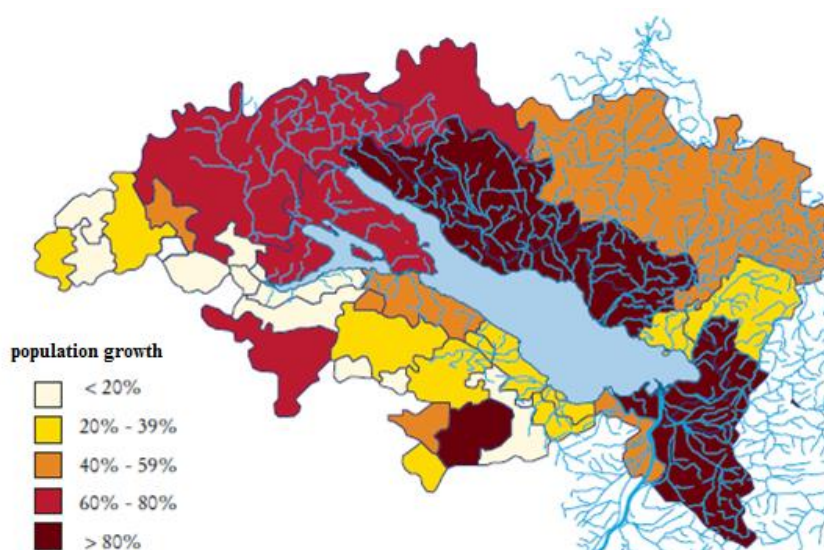
Despite all international concerns on the ecosystem health, Lake Constance could not escape from the negative influences of dynamic population and economic growth. The lake achieved in the second half of the last century a visible stress limit. Too many nutrients had entered the lake from different sources and caused enormous algal blooms and reduced water transparency (IGKB, 2004).

### Driving forces, pressures and impact

Until the 19<sup>th</sup> century Lake Constance was considered despite slowly increasing discharges of wastewater as a close-to-nature lake. Since the mid-19<sup>th</sup> century the utilization and pressure on Lake Constance and its catchment intensified due to high population growth (Figure A1.13). Bank consolidation, urban growth, regulating of tributaries' inflows as well as pollutant inputs from local towns, industry and

agriculture increased constantly over the years. The accelerated eutrophication of Lake Constance in the 1960s was mainly caused by highly polluted municipal wastewater. This led to very high phosphorous peak concentrations up to 87 µg/l in 1979 (IGKB, 2004). The annual primary production was at this stage 25 times higher. Also oxygen concentrations reacted in a way that oxygen peak values of 200% in the upper layer and minima of only 20% in the sediment-close areas were observed (IGKB, 1982). In 1972 only 25 % of the population that was suitable for a connection was connected to central wastewater treatment plants with an additional biological treatment step. In 1997 about 2.1 % of the total water inflow in Lake Constance consisted of treated wastewater. Also industry had a major impact on the nutrient pressure of the lake due to discharges of industrial wastewater into tributaries or directly into the lake. Diffuse nutrient inputs from agricultural fields played a minor role in the past (IGKB, 2004).

**Figure A1.13: Population growth in the Lake Constance area between 1950 and 1990.**



Source: IGKB, 2004.

## Responses – eutrophication management

In 1959 the International Commission for the Protection of Lake Constance (IGKB) was founded in which all partners (Baden-Württemberg, Bavaria, Austria, Switzerland) agreed on a joint cooperation to keep Lake Constance in a clean state. The commission adopted in 1960 the “Agreement on the Protection of Lake Constance against Pollution” and obliged all countries and cantons in the catchment to weigh carefully all water protection measures and implement them according to their national law. In 1967 the commission adopted (and partly revised them in a later stage) the first “Guidelines for keeping Lake Constance clean”. The guidelines defined stricter requirements compared to German Wastewater Regulation for effluent concentration of WWTPs, especially for phosphorous. Quantitative requirements with regard to wastewater effluent were described in dependence of the WWTP’s size class (SC):

- SC 1: national requirements
- SC 2: 1 mg/l, 90% purification capacity
- SC 3: 0,3 mg/l, 95% purification capacity

Next to the requirements for municipal wastewater management, it also contained requirements for the handling of industrial wastewater and recommendations on measures and regulations on how to deal with diffuse sources from agriculture in order to ensure the ecological functioning of the lake in a sustainable way. The amendment in 2005 contained the actual state of knowledge.

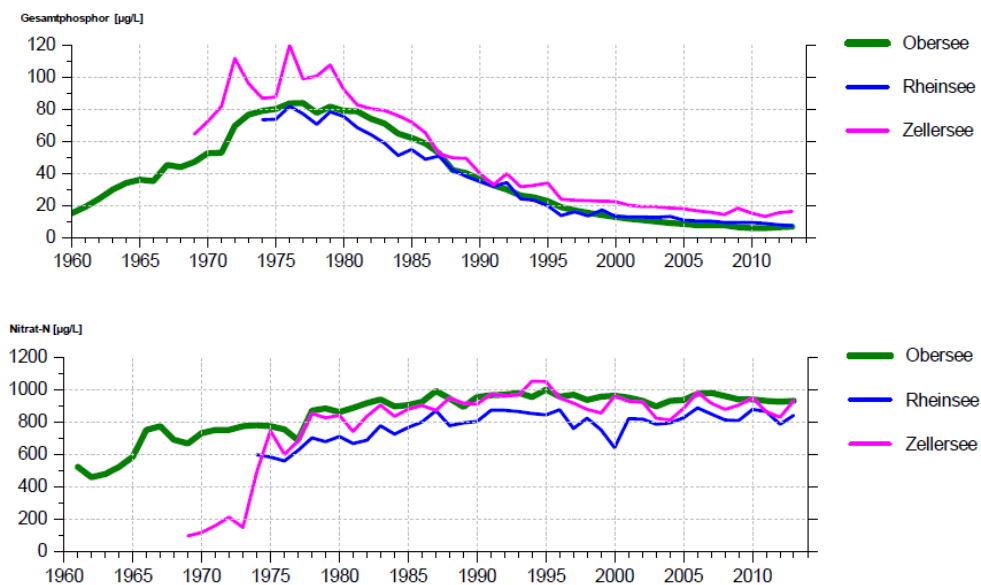
Next to the guidelines the IGKB adapted as a consequence of further increasing nutrient loads to Lake Constance that were mainly caused by the discharge of highly polluted urban sewage the first construction and investment program in 1973 (until 1985). The focus of the program was the establishment of wastewater treatment facilities including additional treatment, the collection of wastewater and the treatment of industrial wastewater. In a second construction and investment program between 1986 and 1995 an improved storm water treatment was implemented. Besides that, a number of other national regulations contributed to the protection of Lake Constance.

In 2013, 1.60 million out of 1.63 million people living in the catchment area of Lake Constance were connected to a central wastewater treatment system, 35,400 inhabitants were not connected at all (IGKB, 2014).

### Status and effects of policy measures

The implementation of the policies and programmes at Lake Constance has shown great reductions in N- and P loads to the aquatic environment and can be considered as very successful. From 1980 onwards, the phosphorus increase was stopped and its concentration in the lake fell from 87 µg/l in 1979 to 6.7 µg/l in 2013 (Figure A1.14; IGKB, 2014). The TP load that was entering the lake was reduced by 775 t/a between 1986 and 1997. The wastewater born load of bioavailable phosphorous was reduced from 536 t P/a in 1986 to 170 t P/a in 1997. Diffuse TP loads changed from 1719 t TP/a in 1986 to 1309 t TP/a in 1997. Nitrogen loads from settlement areas were reduced by 68% between 1986 and 1997 (Bühner et al., 2000). However, inorganic nitrogen concentrations in the lake remain on a rather stable level since the beginning of the 1980s (Figure A1.14; IGKB, 2014).

**Figure A1.14: Above: Total phosphorus concentration at Lake Constance, annual mean values in the central lake (mg/l); Bottom: Nitrate-N concentrations at Lake Constance, annual mean values in the central lake (µg/l) (green = Obersee, blue = Rheinsee, purple = Zellersee).**



Source: IGKB, 2014.

### Lessons learnt

The program worked very effectively in terms of improved wastewater treatment of municipalities, industry and storm water. However, a significant percentage of 60% of the nutrient inputs today origins from diffuse sources. For a further reduction of the nutrient loads also effective measures against diffuse sources need to be implemented. (IGKB, 2004).

## 8 Danube River Basin

### Background information

The Danube River is one of the world's biggest international freshwater resources. It stretches halfway across Europe and is shared by 17 nations. The Danube catchment covers an area of 817 000 km<sup>2</sup>. The ecological importance of the river also reflects in its estuary in the Black Sea. During the 1970s and 1980s, the trophic status of the Black Sea, and particularly the North-West Shelf increased dramatically, resulting in extended and extensive periods of hypoxia, with severely damaged pelagic (water column) and benthic (sediment) ecosystems (UNDP-GEF, 2006).

### Driving forces, pressures and impacts

Emissions of N and P lead to eutrophication in many surface waters of the Danube River Basin and contributed to eutrophication effects in the Black Sea. Since there is hardly any information available on the nutrient status of the Danube River in early times, results are often based on modelling efforts. According to Schreiber et al. (2005) and Behrendt et al. (2005) the diffuse source pollution of nitrogen has doubled in the period 1950 to the mid-1980s. The total phosphorous load in the Danube river basin had its peak in 1990 with an amount of 30kt/a. The dissolved inorganic nitrogen load amounted to 500 kt/a (GEF-UNDP, 2006). Main pressures for these high loads were wastewater from industry and municipalities but also diffuse nutrient inputs by agriculture.

### Responses – eutrophication management

A number of different action plans and programmes were implemented in order to increase nutrient control and to minimize the nutrient load from the Danube River into the Black Sea. Next to the Strategic Action Plan for the Danube River Basin (or Danube Pollution Reduction Programme) from 1995 to 2005 the more important step towards an efficient nutrient control was the foundation of the International Commission for the Protection of the Danube River (ICPDR) in 1998. The ICPDR was made responsible for the implementation of the Danube River Protection Convention (DRPC). After its foundation, a more coordinated action of all Danube countries was possible. Under the framework of the EU WFD, the ICPDR countries (including non EU Member States) adopted in December 2009 and December 2015 the 1st and 2nd Danube River Basin Management Plan that entailed a Joint Action Programme (JAP) of basin-wide importance and also set the framework for more detailed plans at the sub-basin or national level (ICPDR, 2009, ICPDR, 2015). The DRBMPs included assessments and measures towards the achievement of the 'good status' by 2015 and 2021. However, since the set of measures is until now not fully established their impact on the water quality can be not fully evaluated at this present stage.

Nevertheless, slightly decreasing nutrient trends in the basin are caused by a variety of measures that are not caused by a direct and objective-driven management plan but are partly a result of an economic break-down in the Eastern Danube countries that had a decreasing effect on agricultural and industrial pressure. On the contrary, stricter standards in terms of wastewater treatment and phosphate based detergents in the economically better situated countries of Germany, Austria and Czech Republic were applied. In Germany technical improvement aimed at nutrient reduction via additional treatment steps in WWTPs. Two wastewater treatment plants in Munich were upgraded. Also in Austria WWTP were enlarged and equipped with nutrient removal steps. Same applied for WWTPs in the Czech Republic. In Romania the reduction of industrial activities from several major polluters has led to general improvements in water quality. The consequences of the transition period in Bulgaria also caused an improvement of the water quality (ICPDR, 2001).

The JAP aims at "the balanced management of nutrient emissions via point and diffuse sources in the entire Danube River Basin District that neither the waters of the DRBD nor the Black Sea are threatened or impacted by eutrophication" (ICPDR, 2009). Concrete measures were specified in an "Interim Report on the Implementation of JAP" and its annexes which refers to the implementation and enforcement of the EU UWWTD and the EU Nitrates Directive in the Member States. The implementation includes

enforcement, codes of good practises, tariffs, awareness rising and voluntary agreements. The joint action program includes measures for the improvement of wastewater treatment and the application of BAT for industry and agriculture. It also includes measures to control diffuse nutrient pollution, measures to reduce phosphate emissions from household laundry and dishwater detergents, and, finally, measures addressing the nitrogen pollution from atmospheric deposition (JAP, 2001; ICDPR, 2012).

### **Status and effects of policy measures**

The economic break-down in the Eastern Danube countries and on the contrary the higher wastewater treatment requirements as well as the banning of phosphate based detergents in the economically better situated Danube countries led to a decreasing nutrient trend in the Danube River Basin between 1988 and 2005 (UNDP- GEF, 1999). But still the present level of total loads in the DRB is considerably higher than in the 1960s but lower in the late 1980s, according to modelling results of Schreiber et al. (2005) and Behrendt et al. (2005). Diffuse source pollution of nitrogen was reduced by about 23 % from 1980 to 1990. The point source pollution of nitrogen decreased within the 1990s by about 20 %. Total P-emissions reduced by about 40% during the 1990s, but the present level is still 1.6 times higher than in the 1950s. Phosphorous loads caused by point sources decreased by about 50% since 1990. The present level in the upper Danube is already in the range of what it was in the 1950s. Loads for dissolved inorganic nitrogen had its peak in 1985 with 500kt/a and were reduced to 300kt/a in 2005. For total phosphorous peak loads, these were observed in 1990. They accounted for 30kt/a and were reduced to 25kt/a in 2005. Also a reduction of nitrogen surplus was observed, especially in the countries in the middle and lower parts of the Danube. Large residence times in the groundwater will most likely lead to a further reduction of diffuse nitrogen emissions, if N surplus remains at the same level (IDRPC, 2004).

The reporting of the nutrient status of the Danube has been rather inconsistent in the past decades but has been substantially improved with the implementation of the EU WFD and the 1<sup>st</sup> and 2<sup>nd</sup> DRMBP which is why the effects of measures on eutrophication abatement on the nutrients status of the Danube are detectable in a more comprehensive and detailed way. Improvements of the nutrient status for the Danube are reported in the field of point and diffuse sources: For the reference year of the 1<sup>st</sup> DRBM Plan (2006) 130,000 tons per year TN load and 22,000 tons per year TP emissions were reported via direct urban wastewater effluents. The reported nutrient loads from UWWTPs are significantly lower compared to those of the 1<sup>st</sup> DRBM Plan. The TN and TP discharges declined by 32% and 45%, respectively, which is in line with the forecasted future achievements of the 1<sup>st</sup> DRBM. Moreover, the industrial emissions that are directly discharged slightly decreased by around 7% for TN. Industrial emissions for TP dropped by 50%. Diffuse emissions also substantially decreased by 8% for TN and 28% for TP due to the low agricultural intensity in many countries and the implemented measures. The total N emissions is reduced by 12% compared to the 1<sup>st</sup> DRBM Plan whilst, total P emissions decreased by 34% (ICDPR, 2015).

### **Lesson learnt**

Nutrient control has primarily focused on point source reduction, whereas there is an additional need to focus on diffuse sources in the future, as they clearly dominate the overall emissions having a contribution of 84% (TN) and 67% (TP) (ICDPR, 2015). However, there is still a high input of untreated wastewater from the downstream countries detectable (Deckere et al., 2011). In general agriculture (N: 42%, P: 28%) and urban water management (N: 25%, P: 51%) are responsible for the majority of the nutrient emissions. This indicates the need for appropriate measures to be implemented in these sectors (ICDPR, 2015).

Reduced nutrient loads within the past years are mainly due to political and economic break down that led in consequence to a change in agriculture from an economically driven production to self-sufficiency production. Also the closure of large industrial animal production plants and many fertilizer production plants contribute to the decreasing nutrient concentration in the Danube River Basin.



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