Eutrophication

Manifestations, causes, consequences and predictability

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Reflection of a photographer in a water bubble: © ALAIN JOCARD / AFP; Microscopic observation (x 200) of a freshwater cyanobacterium, Planktothrix: @ Alexandrine Pannard / University of Rennes 1; Arrival of green algae on the Brittany coast, Trezmalacouen, Kerlaz, 23 June 2013: @JP Guyomarc'h; Aerial photo of a phytoplankton bloom at Pointe Saint Gildas, Préfailles, 1 August 2014: @Yves Le Medec / Minyvel-Environnement; Ducks on a pond: @ Alexandrine Pannard / University of Rennes 1; photo p. 83: @ Christian Chauvin.
Foreword

The massive introduction of organic matter and nutrients (nitrogen, phosphorus) in surface water upsets the natural balance of aquatic ecosystems. These external inputs, caused mainly by direct discharges of effluents (domestic, industrial and agricultural), runoff waters contaminated by flowing through agricultural and non-agricultural surfaces, and atmospheric deposition, pollute both inland waters and coastal marine waters.

The most visible manifestation of these types of pollution is eutrophication of river, lake or coastal water bodies, which takes the form of excessive algal growth and depletion of water oxygen concentration, resulting in a higher risk of mortality for some aquatic organisms.

In order to better understand the complex and socially sensitive issue of eutrophication, the Ministry of Ecological and Social Transition (MTES), together with the French Agency for Biodiversity (AFB) and the Ministry of Agriculture and Food (MAA) mandated France’s National Centre for Scientific Research (CNRS) to coordinate a joint scientific appraisal in order to provide proven scientific and technical elements. The purpose is to produce a comprehensive situational analysis of the scientific knowledge on the complex and controversial issue of eutrophication and the role played by nutrients (particularly N and P) in this phenomenon so as to inform public policies. The need to carry out this joint scientific appraisal on eutrophication is motivated notably by the implementation of the European directives for the protection of aquatic environments: the Nitrates Directive, the Water Framework Directive (WFD), the Marine Strategy Framework Directive (MSFD) and the Urban Wastewater Directive (UWD).

This joint scientific appraisal was conducted within the framework of a close partnership between the CNRS, Ifremer, INRA and Irstea, whose scientific activities cover various aspects of eutrophication. Other partners (notably universities) also took part in the appraisal with the voluntary involvement of scientists, each contributing their unique, recognized and complementary expertise. Nearly 50 scientists worked for more than two years to bring a fully comprehensive vision of advances in knowledge on eutrophication phenomena, analyzing the world scientific literature on eutrophication’s causes and consequences as critically, but also as neutrally, as possible. Based on past experimentation, these scientists identified the potential levers that, in a collaborative approach, could be activated to combat the emergence of eutrophication situations or seek to define remediation approaches. They also identified knowledge gaps as well as observation and experimentation systems necessary for a better understanding of the processes involved in eutrophication phenomena.

The results of this scientific analysis, based on a critical study of a bibliographic corpus of more than 4,000 international articles reviewed by peers, were also complemented and objectified by the findings of an international scientific seminar which was held at the CNRS headquarters from 18 to 20 April 2017.

CNRS, Ifremer, INRA and Irstea wish to express their sincere gratitude to all the staff who took part, on a voluntary basis, in this large-scale scientific analysis, in compliance with the national appraisal charter.
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Joint scientific appraisal – Authors and editors
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1. Introduction

1.1. The issue in context

People tend to settle in communities in valleys, on the banks of rivers and lakes or along the coast. Water is vital to human life and development, but it is also used—often unintentionally or indirectly—as somewhere to dispose of waste. However, population growth and the development of these communities has gradually exceeded the limits of water’s natural purification capacities, so water quality has deteriorated.

Eutrophication is one of the most common forms of pollution in marine and fresh waters. It is caused by the enrichment of water with an excess amount of nutrients and results in much greater productivity in aquatic ecosystems. The best-known signs of eutrophication are toxic cyanobacterial bloom in lakes and rivers, and the proliferation of green macroalgae in coastal areas. These phenomena cause major disruption in aquatic ecosystems and have an impact on the related goods and services, the ensuing business activities and human health. Intensified agriculture and urbanization have been identified as the main contributors to excess nutrient inputs in aquatic ecosystems.

Eutrophication phenomena were observed early in the 20th century in aquatic environments near the large urban and industrial areas of industrialized countries in the northern hemisphere. Political initiatives to address the problem were introduced several decades ago. The issues tackled by these policies, the cognitive and technical frameworks applied to address the problems, their social visibility, and the knowledge and instruments on which they are based have considerably evolved since the 1990s. On a European scale, these developments have resulted in the set-up of regional conventions such as the OSPAR convention, or standards such as the Water Framework Directive (WFD).

A significant outcome of these changes is that, in certain contexts, eutrophication has become a sensitive social issue; in other words, it involves an array of stakeholders with contrasting values and interests and cannot be approached and tackled by the establishment of scientific proof alone. While eutrophication can be observed all over the world, the actual phenomena remain local, materialized in various forms and according to different trajectories, and of varying sensitivity and severity. The public measures taken to tackle eutrophication highlight the social and political challenges, on the one hand because of the diverse regulatory constraints they entail and, on the other hand, because they feed into the public debate on activities pinpointed as contributing to or having decisively contributed to the nutrient enrichment of aquatic environments. This is especially true with regard to farming in Europe and North America.
1.2. Questions put to the CNRS, Ifremer, INRA and Irstea

In light of the socially sensitive issues surrounding eutrophication, and with a view to improving the consistency, relevance and effectiveness of public action, the ministries in charge of ecology and agriculture in France were keen to establish a sound scientific basis bringing together all the knowledge available on the topic. The CNRS, Ifremer, INRA and Irstea were thus called on to produce a critical appraisal of scientific knowledge from across Europe and the rest of the world, on the causes, mechanisms, consequences and predictability of eutrophication phenomena. The term eutrophication is used by both the scientific community and public policymakers, but with a myriad of definitions. The research institutes were therefore asked to clarify the definition of eutrophication, given the requirements and operational challenges of public action. The ministries also asked that consideration be given to the land-water continuum, in other words the transfer system between watersheds and fresh, brackish and marine aquatic ecosystems, given that this continuum is a factor when characterizing the eutrophication risk. Finally, the four organizations were asked to identify any scientific obstacles that require the acquisition of new knowledge and could therefore be the subject of research. The scope of the appraisal does not include a detailed analysis of the impact of human activities (agricultural systems, water transfer conditions, etc.) on eutrophication. Only the global aspects of their impacts and effects on a large spatial scale have been examined, to provide an integrated vision of the processes across the country.

More than simply understanding the biological, physical and chemical processes at work, it appeared essential to the review’s coordinators to deal with eutrophication as a social issue for two main reasons. Firstly, because the worsening of eutrophication phenomena appears to be closely linked to the growth dynamics of human societies. Secondly, because the phenomenon is becoming increasingly visible to people. The scope of the literature studied therefore encompasses all the human and social scientific work looking at these issues, especially economic, legal, political science, sociology and management science work that may provide key factors for understanding and analysing public policies to tackle eutrophication and the context of their application.

1.3. Method

The founding principles of joint scientific appraisals (which go by the abbreviation ESCo in French) are defined by the national appraisal charter established on 22 December 2009 under the aegis of the ministry of research and signed by the CNRS, Ifremer, INRA and Irstea, and by the specific charters of these four institutes. The purpose of this type of appraisal is to provide the authorities with a base of certified scientific knowledge that they can draw on during their policy-making processes. An ESCo appraisal means collecting the international scientific literature available on a given topic and identifying areas of certainty and uncertainty, the shortcomings and any questions that are subject to scientific controversy. A joint scientific appraisal is not designed to provide the experts’ opinions or turnkey technical solutions to the questions raised by the authorities, but is instead aimed at identifying levers for action.

The soundness, quality and objectiveness of an appraisal are based on the founding principles set out in the national charter and in the institutes’ own charters, namely: competency, plurality and impartiality of the expert group and transparency in the process. Competency is guaranteed by the method used to select the experts, based on their specialities related to the issue and on their peer-reviewed publications. The quality of the appraisal is ensured by the diversity of disciplines, institutions and nationalities represented and the variety of analyses. In addition, this enables a confrontation of viewpoints and the identification of any controversies, thus meeting the need for partiality, a principle that is also ensured by the fact that each expert signs a charter of ethics and completes a declaration of interests. Finally, the principle of transparency is guaranteed by the traceability and reproducibility of the method and the fact that the sources used are made public.
The Eutrophication ESCo group comprised a project team of four scientific coordinators, a coordination manager and five documentary assistants, and a panel of experts made up of 39 scientists (the list of people involved in the group can be found at the end of this document). The group’s areas of expertise range from ecology, hydrology, bio-geochemistry and the biotechnical sciences, to social sciences, law and economics. They cover the various types of aquatic ecosystems: watercourses, bodies of water, estuaries, coastal and offshore marine environments, and the notion of continuum between these systems with specialists in the watershed approach.

The panel of experts produced the different components of the appraisal by analysing and synthesizing the appropriate documentary corpora. All the experts’ contributions have been published together in an appraisal report, on the basis of which the project team wrote this summary (documents available in full on the open-access website: www.cnrs.fr/inee). The panel of experts is responsible for the report. The project team guarantees the conditions under which the appraisal process is conducted: quality of documentary work to update bibliographical sources, transparency of discussions between experts, coordinating the working group and drafting summary documents and communication materials.

The corpus of literature on which the appraisal was based was put together using requests via the Web of Science (WoS), an online academic information service that provides access to several bibliographic databases containing a very high proportion of the world’s published and ‘certified’ scientific literature. For human and social sciences, additional research was conducted using the Scopus and Econlit bases. With support from the documentary assistants, the experts were able to use a combination of keywords to search the databases and extract the literature relevant to their particular question. Where there was little documentation available on the questions, the experts referred to grey literature, i.e. literature not endorsed by an academic reading committee. In those cases, they only used literature from governmental organizations (scientific or technical reports from or commissioned by public research agencies or state services). For law and monitoring, legal texts and technical reports in the field of applied ecology were included. Academic scientific literature on signs of eutrophication in the French overseas departments is rather limited and most of the information available is found in grey literature. As a result, this literature review mainly concerns mainland France.

The appraisal is based on a corpus of literature of approximately 4,000 references. Given the mass of information in existence and its dispersal, we cannot guarantee that the review is exhaustive. However, the rigour applied to the process and the reliability of the documentary sources used means we can guarantee the representativeness of the corpus of literature with regard to the questions covered.

1.4. Structure of this summary

Our summary is a cross-cutting view of the questions dealt with in the report. It is built around four sections: 1) characterization of eutrophication phenomena, 2) their evolution in relation to human activities, 3) their monitoring, and 4) their management. The first section sets out the causes, mechanisms and main signs of eutrophication, and the role played by the transfer of nutrients along the land-water continuum. It also proposes a definition of eutrophication. The second section explains the evolutions in nutrient flows and the signs of eutrophication with regard to human activities, while the third section looks at the capacity of existing tools to measure these phenomena. The final section covers the integrated management of eutrophication phenomena, focusing on the predictability of these phenomena and ways to remedy and prevent them.

In all four sections, the elements that are common to both marine and freshwater environments (e.g. mechanisms, reflections and the procedures concerning monitoring tools) are dealt with first, followed by the differences between the two systems (e.g. signs of eutrophication).
2. What is eutrophication? Why and how does it occur?

The eutrophication of aquatic ecosystems is a series of biological processes triggered by an excess of nutrient inputs. It results in complex responses in all fresh, brackish or salt-water ecosystems. It may be gradual or sudden. The most significant effects are the proliferation of primary producers (aquatic plants, algae and cyanobacteria), phenomena of toxicity or anoxia (lack of oxygen), and biodiversity loss. Eutrophication is often perceived by the public as a very localized environmental issue, with some highly publicized phenomena such as the ‘green tides’, depositing vast quantities of toxic algae on the Brittany coast.

However, the factors that control eutrophication do not work just locally or occasionally. The processes are controlled by factors that also operate on broad spatio-temporal scales. Nutrients that arrive in the coastal zone may be transported from upstream watersheds several hundred kilometres away, before being diluted in the sea.

Eutrophication processes occur in all aquatic environments, but the phenomenon varies in its expression and dynamics depending on whether it occurs in fresh, brackish or salt water.

The aim of this chapter is to present 1) a description of the eutrophication process and the factors that trigger it, underlining the similarities and differences in the main types of aquatic environment; 2) a systematic analysis of the definitions found in the literature to establish a common framework for scientific and technical use of the term; 3) an analysis of the nutrient transfer mechanisms along the land-sea continuum, from the watershed headwater to fresh, brackish and salt water ecosystems; 4) a critical analysis of the literature concerning the current controversy within the scientific community on the respective roles of nitrogen and phosphorus in triggering eutrophication; 5) an application of the concept of vulnerability to eutrophication.
2.1. The mechanisms of eutrophication

2.1.1. Factors controlling eutrophication

Regardless of the aquatic ecosystem in question, the primary producer compartment, i.e. the compartment of organisms that produce their own organic matter, is always the first to be affected in the eutrophication process. Chain reactions then occur in the other biological compartments with consequences on biogeochemical cycles, the dynamics of the biological communities and, ultimately, the evolution of the aquatic ecosystem as a whole. To truly understand eutrophication and its control factors, we first need to recall how the primary producer compartment functions. We will then examine the factors that affect its functioning and therefore control eutrophication.

The primary producer compartment

Aquatic primary producers are organisms such as cyanobacteria, algae, higher plants (phanerogams), aquatic ferns (pteridophytes) and moss (bryophytes). While they are of very high phylogenetic diversity, they are all able to use solar energy to produce organic carbon and oxygen from carbon dioxide and water. This diversity is reflected in the very broad diversity of physiological mechanisms, metabolic pathways and morphological structures among photosynthetic aquatic organisms.

Primary producers can be classed into functional groups that sometimes contain very varied taxonomic groups. They are either microphytes or macrophytes. Microphytes are microscopic primary producers including cyanobacteria and microalgae. Macrophytes are visible to the naked eye and this term encompasses macroalgae, higher plants, aquatic ferns and moss. Primary producers may develop freely in the water column (phytoplankton) or grow on a substratum (phytobenthos) or on other organisms (epiphytes or epibionts).

These primary producers convert light energy into chemical energy which they store in the form of organic carbon via the biological process known as photosynthesis. In this process, primary producers reduce the carbonic gas (CO₂) in the atmosphere or dissolved in the water and release oxygen (Fig. 2.1). Net primary production (organic matter production) on the planet is estimated at 10¹⁷ g.C.an⁻¹ with 56 10¹⁵ and 48 10¹⁵ g.C.an⁻¹ respectively for terrestrial and marine ecosystems. The main driving force behind organic matter decomposition in terrestrial and aquatic environments is the respiratory process. Aerobic respiration produces energy for the organism through the oxidation of organic carbon in the presence of oxygen. This reaction produces carbon dioxide (CO₂). Where there is no oxygen (anoxic conditions), micro-organisms are able to produce energy from organic matter using other oxidizing molecules (nitrates, iron and manganese oxides, sulphates); this is called anaerobic respiration.

![Figure 2.1. Photosynthesis (or primary production, blue arrows) and respiration (organic matter decomposition, red arrows).](image-url)
Living plant matter produced through photosynthesis comprises several chemical elements. The five main elements in organic matter are carbon (C), hydrogen (H), oxygen (O), nitrogen (N) and phosphorus (P). Magnesium (Mg), potassium (K), calcium (Ca) and sulphur (S) are macro-nutrients (> 0.1% of the organism’s dry weight) deemed to be essential in plants. Micro-nutrients, such as iron (Fe), boron (B), manganese (Mn), zinc (Zn), copper (Cu), nickel (Ni), chlorine (Cl) and molybdenum (Mo) are also essential. However, within the different primary producer groups, dependency on these various nutrients varies. Among algae, for example, C, H, O, N, P, Mg, Cu, Mn, Zn, Mo and Fe are deemed essential for all phyla. S, K, and Ca are necessary for all algae and higher plants, but may be partially replaced by other elements. Finally, sodium (Na), cobalt (Co), selenium (Se), silica (Si), chlorine (Cl), boron (B) and iodine (I) are only essential for some algae. silica, for example, is vital for diatoms (microalgae that have a silica cell wall).

The mass ratios between the various nutrients (stoichiometry) are relatively stable at a global level. In the marine environment, in 1934 Redfield demonstrated that the average compositions of water and the phytoplankton biomass of the Atlantic Ocean were highly stable, characterized by atom number ratios of 106:16:1 for carbon, nitrogen and phosphorus respectively; this means that for one phosphorus atom used in biosynthesis, 16 atoms of nitrogen and 106 atoms of carbon are consumed. In 1985, Brzezinski completed these ratios, adding silica, an element vital for the growth of diatoms, with C:Si:N:P ratios of 106:15:16:1. In fresh water, the ratios between elements are not quite as homogeneous as they are in the world’s oceans, and C:N, C:P and N:P ratios are higher. Recently, authors have demonstrated that the element composition varies according to the main phytoplankton groups and environmental constraints. Some authors have shown that the Redfield N:P ratio of 16 is not a universal biochemical optimum but results from the average of the N:P ratios of species.

**Nutrient input**

In primary production, growth is controlled by the scarcest nutrient resource. This concept was originally based on Liebig’s law of the minimum (1850) developed in agricultural science and applied in the post-industrial era to optimize higher plant crop yields by adding fertilizers (Fig. 2.2). The law applies to photosynthetic organisms in aquatic environments and is completed by Liebscher’s law of the optimum. This dates from the late 19th century and highlights the importance of complying with optimal element ratios to optimize growth. We now know that these optimum ratios can vary considerably from one plant to another.

These principles have been applied to aquatic ecosystems very frequently over recent decades. However, the phenomena of co-limitations, synergies and interactions between elements, especially nitrogen and phosphorus, are very pervasive among terrestrial and marine primary producers. When characterizing phytoplankton limitations, application of the law of the minimum is therefore currently specified by the generic concept of co-limitation, recently introduced to define interactions or simultaneous limitations by several resources. The various concepts have significant implications when it comes to understanding the effects of nutrient inputs on primary producer growth, especially in the context of eutrophication. Along the land-sea continuum, there is a gradient of natural enrichment in nutrients (figure 2.3). As we shall see in chapter 2.4, nitrogen and phosphorus are generally identified as factors limiting the development of primary producers, and therefore as the main contributors to the onset of eutrophication.

Affinity and the speed with which nutrients are incorporated are two physiological variables that determine algae species growth strategies. It is generally considered that the large microalgae species have a weaker affinity for nutrients than smaller-sized species (e.g. in ocean communities), which makes the latter more competitive when a nutrient becomes scarcer in its environment.
environment. In addition, some species are able to incorporate and store large quantities of nutrients in their cells very rapidly when there are occasional nutrient inputs, as in coastal ecosystems. Consequently, plant growth is not directly linked to the concentration of nutrients in the surrounding environment, because they are able to build up reserves to provide a buffer against the sometimes suddenly fluctuating nutrient levels in the aquatic environment. The plant’s internal content of a given nutrient, known as the cellular quota, will therefore vary. It is acknowledged that there is a minimum cellular quota, corresponding to the intracellular reserve threshold below which the cell loses all its growth capacity. Cellular quotas therefore serve as an indicator to the reserve capacities and nutrient requirements of each species. They can also be used to determine the physiological advantage of one species over another, which will eventually lead to its dominance over other species. In morphological terms, nutrient absorption depends on the surface area/volume ratio of organisms: a high surface area to volume ratio means a larger area of contact with the environment and thus optimized nutrient absorption. For example, the microalgae associated with proliferation phenomena often have foliose or filamentous morphologies.

**Water residence Time**

The residence time of water in an aquatic ecosystem or in part of it affects the development of algae proliferation. The longer the water residence time, the less quickly the plankton formed will be evacuated; there will be more exchanges between photosynthetic organisms and more nutrients will be dissolved, potentially increasing the risk of eutrophication. Ponds, natural lakes and dams, millstreams, gently sloping sections, oxbow lakes and lagoons are therefore particularly vulnerable to eutrophication as they are ‘closed’ (Fig. 2.3). In marine environments, the same is true of inlets, bays, mudflats where water is decanted along estuaries or on straight foreshores sheltered by a string of islands, like the Dutch and German Frisian areas. The low gradient in these areas is often accompanied by very low flow rates, making for long water residence times. The ebb and flow of tides only have a minor impact on water residence time in these cases. On the contrary, bays that are naturally linked to the ocean by a narrow channel, such as the Bay of Brest or the Gulf of Morbihan in Brittany, generate intense currents associated with the tide; they quickly dilute inputs from the watersheds and carry them out to sea.

In addition to the lateral containment of these environments, related to their geographic position, there may also be a vertical containment when there is thermal stratification in the water, as in deep lakes, for example. Vertical containment may also be due to a salinity gradient in coastal waters, resulting from limited mixing of fresh water from a river with sea water, as is the case for the Vilaine or Loire. Vertical containment resulting from salinity recedes as we move away from the coast but in summer, in depths of 100 metres or more, it is replaced by vertical thermal stratification as in deep lakes. This stratification triggers the start of phytoplankton proliferation in spring by blocking plant cells in a well-lit, nutrient-rich layer a few metres below the surface.

![Figure 2.3. Diagram showing the evolution of eutrophication control factors along the land-sea continuum.](image)

1. Intertidal zones
Light

We have already seen that photosynthetic organisms need light to provide the necessary energy for primary production. Along the land-sea continuum, zones frequently subject to the re-suspension of sediments in the water column, such as estuaries, are zones where lack of sunlight limits the development of primary producers (Fig. 2.3). Phytoplankton, and floating plants and algae usually have better access to light than benthic plants. Hence, the growth of rooted photosynthetic organisms, those drifting near the ocean bed or epibiotic plants mainly occurs in shallow or clear water, such as the tranquil waters of small, non-shaded watercourses, shallow lakes or watercourses, and coastal zones. Meanwhile, water on the surface of the continental shelf, in estuaries (even where quite turbid), marine waters, and the surface waters of deep lakes in periods of thermal stratification are all favourable to the growth of phytoplankton and floating plants or algae.

Temperature

All biological activities are stimulated by an increase in temperature. Cell development, photosynthetic activities and the respiratory processes are no exception. In addition, an increase in water temperature leads to a fall in dissolved oxygen concentrations in water, prompting the development of anoxic conditions. This explains why algal blooming and anoxic conditions mainly occur in spring and summer, when temperatures are high.

2.1.2. The main mechanisms at work

The factors controlling eutrophication can ultimately be summarized as the converging of all or some of the following interacting elements: an excess in nutrient inputs, long water residence time, sufficient light and favourable temperatures. Under the effects of these factors, the functioning of aquatic ecosystems is modified, triggering a complex response from the ecosystems.

The basic mechanism is common to both freshwater and marine ecosystems: the increase in nutrients triggers a high increase in primary production. The aquatic ecosystems are then transformed from a system with limited nutrient inputs into a system that is gradually saturated in nutrients, where the limiting factor then becomes light. The primary production zone thus concentrates towards the surface of the water column at an increasingly shallow depth, as light penetration is diminished by self-shading as the biomass produced increases.

The responses engendered by a disruption will initially be detected at a physiological/biochemical level of an individual, then at a morphological or behavioural level, and finally at the level of the population or community. As we saw in 2.1.1 above, a rise in nutrient input or an imbalance in the ratios between nutritive elements causes the growth of species that are more competitive in physiological and morphological terms in response to these changes. These modifications then have a significant impact on the system’s species composition and primary production.

Changes in primary producer communities

In communities, increased nutrient inputs lead to a succession of different primary producer functional groups and bring about changes to the communities’ structure and functioning, even resulting in considerable biodiversity loss in the aquatic ecosystems. Even though marine and freshwater systems do not comprise the same species, the successions of primary producer functional groups follow a similar path. Unlike phytoplankton, floating plants and algae, and epiphytic species, rooted macrophytes can draw nutrients from sediments. They are not therefore as dependent on nutrients found in the water column. A such, rooted macrophytes are better placed when it comes to nutrient acquisition, whereas phytoplankton, and floating plants and algae are better off with regard to light. In simple terms, therefore, rooted macrophytes dominate in nutrient-poor environments. When the environment is enriched, epibions then emerging macrophytes, opportunistic floating macrophytes and/or phytoplankton proliferate, to the detriment of submerged macrophytes, which no longer have access to light.
In the marine environment, depending on the level of nutrient input, plant successions can be divided into four phases (Fig. 2.4, left). In phase I, nutrient availability in the environment is low. Perennial benthic macrophytes dominate with, for example, phanerogam meadows on loose substrata or perennial macroalgae on hard substrata. In phase II, when nutrients and turbidity increase, epiphyte species proliferate, along with the grazing organisms associated with macrophytes, to the detriment of the latter. In phase III, the environment tends towards a eutrophic state, and opportunistic macroalgae that drift in the water column and/or phytoplankton proliferate, diminishing light penetration in the water column. Epiphyte numbers drop sharply and benthic macrophytes disappear. Phase IV is the final stage in eutrophication, and can most notably be seen in estuaries and contained bays with low hydrodynamic activity. Phytoplankton and drifting macroalgae are the dominant primary producers and form substantial organic deposits.

In fresh water, there are significant differences depending on the hydrological parameters of the ecosystems, especially when comparing lentic systems with stagnant water or very low flow rates, such as ponds, pools, lakes or very slow-flowing watercourses, and lotic systems with faster flowing waters, such as rivers and streams. Generally speaking, when the environment’s nutrient availability is low, the sediment-based periphyton² tends to dominate (Fig. 2.4, right). When nutrient inputs increase, rooted and submerged perennial macrophytes dominate, such as phanerogams or charophyta-type macroalgae, along with a myriad of epiphytes and drifting macroalgae living between the macrophytes (metaphyton). At an even higher nutrient level, in lentic systems where water is rarely renewed, we see a dominance of phytoplankton or floating plants such as *Lemna* or *Azolla*, and sometimes harmful invasive species; in systems with a high water renewal rate and shallow depth, submerged rooted macrophytes such as filamentous macroalgae dominate.

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**Figure 2.4.** Left: diagram showing changes in the marine environment: A) dominance of primary producers, B) relative levels of physical, chemical and biological parameters and C) structure, distribution and functioning of the various parameters according to the level of eutrophication. Source: Lemesle 2015, according to Schramm 1999.

Right: Diagram showing changes to the relative dominance of primary producers according to the level of eutrophication in fresh water. Adapted from Brönmark & Hansson 1998 and Wetzel 2001.

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² Periphyton on sediment: Epiphytes & metaphyton
Emergent macrophytes
Phytoplankton
Floating macrophytes
Emergent macrophytes

Increasing productivity or eutrophication

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2. Living microalgae attached to a submerged surface and forming a biological cover or biofilm.
Biodiversity loss, toxicity and anoxia

Changes in primary producer communities cause an upheaval in the community structure of the entire ecosystem, affecting biodiversity. For example, within the water column, in both fresh and sea water, flagellate species such as cryptomonads and dinoflagellates generally take the place of chrysophyceae and diatoms in nutrient-rich zones. Dinoflagellates are a nutritional resource of lesser quality than diatoms and cryptomonads. This cycle changes the quality and quantity of primary producers for zooplankton grazers, leading to alterations across the entire trophic network. The same applies to cyanobacterial blooms found in fresh water; the nutritional quality of cyanobacteria is often impaired compared with other phytoplankton groups.

The term ‘harmful algal blooms’ (HABs) refers to any proliferation of microalgae which causes a nuisance to man or the ecosystem. This phenomenon is also known as red tides, as algal blooms often turn the water red, although the species that cause them are not always harmful. In fact, some harmful blooms occur when the water is perfectly clear. There are blooms of species producing toxins affecting humans or other communities of organisms in the ecosystem (cf. 2.1.3); blooms that are harmful because their high biomass generates anoxia or mechanical damage (gill irritation, foam).

During massive blooms, of both microphytes and macrophytes, the deterioration of the large biomasses generated causes a depletion in the environment’s oxygen, potentially leading to hypoxia or anoxia. Organisms living in aquatic ecosystems, sediments and wet zones are not directly in contact with oxygen gas, and only have access to dissolved oxygen in the water. Oxygen diffusion in water is around 1,000 times slower than in the atmosphere and its concentration decreases with an increase in temperature and salinity. As a result, demand for oxygen for oxidation of organic matter in aquatic environments may be higher than supply through diffusion; the outcome is partial or total anoxia. In the absence of dissolved free oxygen, some heterotrophic micro-organisms are able to use the oxygen present in the nitrate molecule ($\text{NO}_3^-$) to oxidize organic matter; this process is known as denitrification (Fig. 2.5). When all the nitrates are consumed (reduced), different micro-organisms are able to use the oxygen found in manganese oxides ($\text{MnO}_2$), iron oxides ($\text{FeO}_3$) and sulphates ($\text{SO}_4$). Finally, when all these oxidants are reduced, methanogenic micro-organisms are able to break down organic matter and produce methane ($\text{CH}_4$). These oxido-reduction reactions (redox) produce carbon dioxide ($\text{CO}_2$), hydrogen sulphide ($\text{H}_2\text{S}$) or methane ($\text{CH}_4$).

These oxido-reduction reactions produce gases such as carbon dioxide, hydrogen sulphide or methane, which can be released into the water and then into the atmosphere (Fig. 2.5). In addition, the reduction of iron oxides found in sediments leads to the discharge of the phosphorus previously adsorbed, making it bioavailable for algae growth once again. This is a positive feedback loop that reinforces primary producer bloom development.

The release of $\text{H}_2\text{S}$ and $\text{CH}_4$ gases into the water causes the death of most pluricellular organisms that live attached to the substrate in a matter of minutes, creating temporary or permanent ‘dead zones’, as has occurred in the central part of the Baltic Sea, for example. Mobile species, especially herbivores, leave the environment if they are able to, amplifying the production of plant biomass in the absence of ‘top-down’ regulation. This final stage in eutrophication is a threat for the economy because it can result in the loss of shellfish production, for example. It can also be harmful for human health, because the anaerobic decomposition of green algae on beaches can produce lethal gas ($\text{H}_2\text{S}$).

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3. Freshwater and marine aquatic ecosystems that are subject to excessive nutrient inputs are subject to two antagonistic mechanisms. A ‘bottom-up’ mechanism or ‘resource effect’ whereby the nutrient enrichment encourages phytoplankton and also stimulates annual algae that proliferate to the detriment of perennial algae. A ‘top-down’ mechanism or ‘consumer effect’ in which filter feeders and grazers limit respectively the abundance of phytoplankton and opportunistic annuals. Indirectly, they thus help to maintain perennial algae. On the flip side, predators affect the filter feeders and grazers and indirectly encourage the growth of phytoplankton and opportunistic annual algae.
It should be noted that anoxia is not always caused by the decomposition of matter produced by algal bloom. In estuaries, for example, water turbidity prevents the development of plant blooms but the mineralization of organic matter and nutrient input from upstream generate oxygen demand that exceeds supply by diffusion, leading to partial or total anoxia. This was the case in the Seine estuary, which was subject to sewage discharge from the Paris conurbation prior to improvements in effluent treatment processes. The basis eutrophication mechanisms are therefore common to all aquatic environments. However, expression and dynamics vary according to ecosystem, location and the species present.

2.1.3. The most common signs of eutrophication

Proliferation of macroalgae in marine environments

The proliferation of opportunistic green macroalgae (Chlorophyta) is the most common phenomenon in coastal areas. The majority of green algae proliferations reported across the world are from the Ulva genus, which encompasses over a hundred species. More occasionally, there are green algae proliferations of the Cladophora genus, red macroalgae of the Gracilaria species, and brown macroalgae of the Pylaiella genus. Three ‘types’ of opportunistic macroalgae can develop; all three have been observed and described on the English Channel/Atlantic coasts of France. Type I, characteristic of the green algae invasions in Brittany, is the development of algae suspended in water with no real prior fixed phase. They are usually found in shallow, sandy bays with low water renewal. The resulting arrivals of algae are monospecific, i.e. mainly comprised of a single species. Free-living algae may be local and subsist in the form of a limited stock during periods that are unfavourable to growth, then develop again when growth conditions are more favourable. With regard to this type of development, work conducted in Brittany has established a connection between nitrogen flows arriving on the coast between May and August and the resulting green algae phenomena in August and September. Type II, found in the Seine Bay, is associated with opportunistic algae whose main growth phase occurs while attached to a rocky substratum before becoming detached and arriving onshore on nearby beaches, mixed with other algal species. Type III is associated with the growth of opportunistic algae on silty substrate. At the end of the algae growth season (spring/summer), some of the algae will be buried in the sediment, where they will remain dormant through winter. Next season, growth will begin again from this winter stock. The presence of high densities of decomposing green algae leads to a fall in oxygen concentration in the water. This causes the death of macro-invertebrates and triggers a change in the structure of their community. It has therefore been established that an Ulva spp biomass superior to 110 g /m² (dry weight) present on a sandy or silty substrate for more than two weeks induces a loss of key benthic invertebrate functional groups. Anoxia of sulphur-loaded sediments also causes the disappearance of a significant portion of the benthic meiofauna (worms, molluscs, etc.). In offshore marine environments, green tides can also have an impact on fishery resources. Three-quarters of commercial catches in the North Atlantic come from species dependent on coastal and estuary environments for at least one phase in their life cycle. Macroalgae proliferations trigger physiological and behavioural responses in fish, leading to a reduction in food uptake, growth and energy reserves. The composition of the fish community is affected, with a gradual decline in fish density and eventually local disappearance in the event of significant or prolonged algal proliferation. In the Mediterranean lagoons, anoxic crises known locally as ‘malaigue’ give the water a milky white appearance and produce hydrogen sulphide. These waters become toxic for animals and plants and can cause shellfish loss at aquaculture sites. In the coastal waters around the islands of the French overseas territories, characteristic of nutrient-poor lagoon and tropical reef environments, nutrient inputs aid the proliferation of certain macroalgae. The combination of this enrichment with other trophic factors (higher mortality among herbivores due to disease or overfishing, development of corallivore species prompted by overfishing of their predators, etc.) may rapidly transform a reef ecosystem into a macroalgae-dominated environment. This transformation has been observed in many coral ecosystems and in the tropical lagoons. Indeed the reef environments have particularly low resilience when faced with these impacts. These changes of regime have major consequences on the rich fauna of the reef system, especially its fish.
Massive arrivals of Sargassum occurred on the Antilles islands and in French Guiana in 2011, 2012, 2014 and 2015. Arrivals like this are not a new occurrence as the Sargasso Sea is known to be the source of banks of Sargassum, dispersed in the Atlantic by the currents, but the extent, duration and intensity of the phenomena reported since 2011 have increased manifold. They also affect the coasts of Brazil and West Africa. This would therefore appear to be a new phenomenon, affecting the entire tropical sector of the Atlantic ocean. Two appraisal reports have recently been produced in France, reviewing the information currently available in the literature on the causes of these phenomena (Florenne et al. 2016, ANSES 2017). The causes have not been clearly established for the moment. A combination of climatic factors and local nutrient enrichment may be at work. A study project is currently underway, backed by the French ministry of environment with scientific coordination by IRD.

Harmful phytoplankton proliferations in the marine environment

A hundred or so marine microalgae species producing toxins affecting humans and other animal species are known at present. The most frequently encountered toxic microalgae on the French coast are species from the Alexandrium, Dinophysis and Pseudo-nitzschia genera, respectively producing paralysing, diarrheal and amnesic toxins that are harmful for shellfish consumers. There is no general link between abundance and toxicity, even for one given species. Many cases of proliferations causing hypoxia or anoxia as a result of their substantial biomass have been recorded across the world. The Baltic Sea and the Gulf of Mexico are the two largest sites the most regularly affected by these phenomena. In France, only a few coastal and estuary sites are subject to hypoxia phenomena. For example, along the coastal waters of the North Sea, the colonial nanoflagellate Phaeocystis globosa proliferates virtually every spring, turning the water slimy and producing foul-smelling yellowish-white accumulations along the coast. Coloured water is also regularly observed in the Loire estuary plume. In spring, diatom blooms of Cerataulina pelagica can turn the water brown, clog fishing gear and cause anoxia that is fatal for fauna living in sheltered coves. Red water from Noctiluca scintillans dinoflagellates can occur in late spring followed by green water from Lepidodinium chlorophorum dinoflagellates, although these do not cause acute hypoxia. There is a worldwide consensus that the diversity, frequency, consequences and geographic extent of harmful microalgal bloom have increased over the past few decades. Although it is still difficult to extrapolate the trends observed from one region to another, the link between eutrophication and the increase in these blooms has often been proven. For example, recurring Prorocentrum minimum blooms are 10–100 times more substantial on the heavily eutrophic US Atlantic coast than a few decades ago, and they occur after rain or use of nitrogen or phosphorus fertilizers. The development of Phaeocystis in the North Sea is linked to increases in N and P inputs in the environment and the stability of silica inputs, while the distribution of Ostreopsis in the Mediterranean is linked to areas where anthropogenic pressure is high. For many species, toxin production is influenced by the quantity of N or P and by the N:P or Si:P ratio, as the stress provoked by the limitation of one of the nutrients triggers toxin production. This is the case with Pseudo-nitzschia, whose toxin secretions are triggered by an excess of N or P in relation to silica, for Alexandrium tamarense, which produces toxins when P is in limited supply, and for Dinophysis acuminata, which produces toxins when N is the limiting factor. Although the link between eutrophication and substantial biomass bloom, toxic or otherwise, has often been proven, the link between less extensive toxic bloom and eutrophication is much more tenuous. Hydrodynamic conditions play a key role in bloom, limiting certain phenomena by dispersion or aiding others even when eutrophication cannot be invoked (water column stratification, upwellings, etc.).
Macrophyte proliferation in freshwater systems

In lotic systems, proliferations of embryophyta (spermatophytes, ferns, mosses) and filamentous macroalgae such as charophyta or Cladophora and Vaucheria can occur. In fast-flowing, shallow watercourses, the proliferation of this kind of macroalgae is generally seen as a major problem as these opportunistic, highly productive organisms can develop a substantial biomass in a very short time, depending on changes in hydrometeorological conditions. For biomass reaching 3 kg/m², filamentous macroalgae can give almost complete coverage of the entire water surface. In addition, the ecological preferences of cohorts may enable this group to occupy the habitat in successive waves, for example Vaucheria sp. at the start of the growing season, then Cladophora sp. when the temperature rises. However, in these fast-flowing systems, it is difficult to establish a clear link between proliferation and eutrophication. In fact, nutrient inputs come from both the water and the sediment, and the effects of these inputs are obscured by variations in hydrological conditions (flow rate, water column height, etc.) and light availability.

In lentic systems, there may be problems with significant macrophyte proliferations in shallow zones, such as shallow lakes, the shore area of deep lakes, and bays in major river systems. Generally, only higher aquatic plants (angiosperms) and some ferns cause problems, especially if they are floating plants or submerged plants with the majority of their biomass in a canopy at the water’s surface. For example, in the bays of the major river systems and in certain lakes, the water chestnut (Trapa natans) has caused huge problems including hypoxia and anoxia of the system, as the floating biomass prevents gas exchanges between the water and the atmosphere and blocks access to light for primary producers, which see their growth diminished (phytoplankton and submerged macrophytes). Similar effects have been observed with water ferns of the Azolla genus, which includes invasive exotic species such as Azolla filiculoides. Other cases of large-scale floating macrophyte domination are to be expected in a context of global warming.

Cyanobacteria proliferations in freshwater lentic systems

In lentic ecosystems, cyanobacterial proliferations are the direct consequence of eutrophication. In phytoplankton communities, cyanobacteria use an array of adaptive strategies to make them particularly competitive when it comes to accessing nitrogen and phosphorus resources and light. For example, cyanobacteria have a high affinity for phosphorus; certain species are able to fix atmospheric nitrogen while others can use organic forms of nitrogen and phosphorus. Many cyanobacteria species produce gas vacuoles, meaning they are able to float and remain close to the surface where light is at a maximum. Several species are also able to produce dormant cells that help increase recruitment the following year. There are several types of proliferation. Development at the water’s surface, as with Microcystis for example, or development dispersed in the water column, as is the case for species adapted to low light conditions, e.g. Planktothrix or Oscillatoria. Development may also be benthic, associated with biofilms, for species such as Oscillatoria and Phormidium. The visibility and ecological consequences of these blooms differ. For example, species that develop in the deep layers may suddenly appear at the surface in early autumn, under the combined effects of a fall in light and the wind stirring the water. Species that develop and are dispersed in the water column may also rapidly concentrate at the water surface at the end of their bloom if their floatability is poorly controlled.

In France, Microcystis is the cyanobacteria genus that most frequently proliferates in lakes, with Planktothrix, Dolichospermum (formerly Anabaena), Aphanizomenon, Oscillatoria, Lyngbya and Nodularia. All cyanobacteria orders have species capable of producing toxins and bioactive compounds. For example, microcystin, a hepatotoxin, is produced by Microcystis, and by other species such as Planktothrix and Dolichospermum; anatoxin, a neurotoxin, is mainly produced by Dolichospermum, but also by Aphanizomenon and Oscillatoria. With the current state of knowledge, it is impossible to predict the potential toxicity of a proliferation that develops in an aquatic environment. Within the same cyanobacterial species, certain strains will produce toxins while others will not. In addition, the proportion of cells containing the genes that enable toxin synthesis may vary considerably during a bloom, or from one bloom to another, even when these proliferations develop in ecosystems that are geographically very close.
2.1.4. Eutrophication trajectories

When disturbed, an ecosystem can, in theory, respond in a linear manner, gradually and proportionally to the forcing factors. In reality, however, it usually responds in a non-linear way, either by becoming saturated or by changing abruptly when conditions near a critical level or when the ecosystem has more than one stable state within the same range of environmental conditions. In the case where the system contains accumulator compartments (e.g. nutrient storage in groundwater, soils and sediments), the system's memory means that the recovery pathway will be different from the eutrophication trajectory (the phenomenon known as hysteresis). We can illustrate this with an example (Fig. 2.6A): when nutrient concentrations are low, only benthic macrophytes dominate, but at the highest nutrient concentrations, only phytoplankton is present. Both states are possible along an intermediary nutrient concentration gradient (hysteresis), and a change from one of these two states to a given nutritive level requires a relatively significant disturbance to shift the system into one state or the other. While phytoplankton turbidity is low, benthic macrophytes may persist. If critical turbidity is reached, the system shifts towards a phytoplankton-dominated state and accumulates nutrients in the sediment. To return to a macrophyte-dominated state, the nutrient richness will need to be reduced to a value that is lower than the starting value, to offset the portion of the nutrient that is diffused from the sediment reserves.

By combining the various responses possible from the ecosystems—linear or abrupt, with or without hysteresis, with or without a change of state—there are many potential trajectories for aquatic systems subject to excess nutrient inputs (Fig. 2.6B). In other terms, while the causes and mechanisms of eutrophication are similar, the trajectories, speed of development and consequences depend on the history of the aquatic ecosystem in question, along with its intrinsic physical and biological characteristics and those of its watershed. The complexity of the phenomena means they are difficult to predict from an ecological viewpoint. For example, we cannot predict which species will dominate at a given site or in a given year. In cases of marine macrophyte eutrophication, the proliferating biomass may sometimes comprise brown algae of the *Pylaiella* or *Ectocarpus* genera and sometimes green algae of the *Ulva* or *Monostroma* genera.

The notion of resilience is a dynamic property of ecosystems subject to multiple disturbances. In ecology, the notion is generally defined as the capacity to absorb the changes cause by a disturbance without altering the structure and functioning of the system.

The mechanisms of eutrophication are therefore complex and its expressions diverse and multiple. We will now see how we can summarize this complexity and diversity in a consensual, generic definition.

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**Figure 2.6.** (A) Theoretical representation of a change in state during a eutrophication process. Nutrient concentrations in the water (unbroken line) fluctuate around the averages (dotted black line) in both states: a state dominates by benthic macrophyte and a state dominated by phytoplankton. The dashed blue and green lines show the state change thresholds. Resilience is the gap between the state average and its threshold. Adapted from Carpenter et al. 2006.

(B) Diagram showing six hypothetical trajectories for the system's response (vertical axis) after changes to the nutritive conditions (horizontal axis). Source: Kemp et al. 2009.
2.2. A definition of eutrophication

Many definitions have been put forward for eutrophication in scientific literature. They reflect the dynamics and diversity of the knowledge produced on the subject. However, the current, very broad use of the word eutrophication covers a scientific concept, a water management issue and a media topic, so it can cover several meanings. In light of this, the panel of experts were asked to ‘clarify the definition of eutrophication to take into account the operational requirements and challenges for public action’. In an attempt to meet this objective, the spectrum of definitions of eutrophication found in the literature was characterized, and the content of the definitions analysed according to their context. In all, 170 definitions were collected, 118 of which came from scientific publications and books and 52 from technical reports and websites.

As we have seen earlier, eutrophication is a complex process. The conceptual diagram shown in Fig. 2.7 is widely used in the scientific realm to illustrate this complexity. In this diagram, the changes induced by nutrient inputs are separated into direct and indirect effects. The review of the definition content was conducted using this conceptual diagram: each of the 170 definitions was treated as an assembly of key words identifying (or otherwise) the type of nutrient input (N, P, natural, anthropogenic, etc.) and the changes engendered by these inputs on the system status, differentiating between direct effects (greater growth and/or biomass of higher plants, microalgae, macroalgae, etc.) and indirect effects (drop in diversity, reduced oxygen, toxicity, water quality, etc.). A statistical analysis was then conducted to compare the definitions, on the one hand taking the most generic and representative definitions from the full entire corpus and, on the other hand, the definitions taken from articles the most often quoted within the selected corpus.

Figure 2.7. Conceptual diagram showing eutrophication, for all types of surface water bodies. Feedback in red. Adapted from Claussen et al. 2009.
2.2.1. Description and analysis of the array of definitions

The first scientific documents dealing with eutrophication date from the early 20th century, looking at European lakes. Eutrophication is described as nutrient enrichment associated with the ageing of a lake over geological time scales (several thousand to several million years). Here, the term ‘enrichment’ refers to the process of trapping and concentrating nutrients naturally made available to the aquatic ecosystem by the watershed and the atmosphere. This so-called ‘natural’ eutrophication is not a reaction to an increase in external nutrient inputs, but a gradual and natural evolution of aquatic ecosystems. Some authors consider that for this process, the term «ontogenesis» would be more relevant than that of eutrophication. After the Second World War, the notion of ‘cultural eutrophication’ emerged, highlighting the role of human activity in the nitrogen and phosphorus enrichment of lakes. Then, the term ‘enrichment’ referred to an increase in external nutrient inputs to ecosystems. The term ‘anthropogenic’ was later preferred to ‘cultural’. Definitions of eutrophication in marine environments appeared in the 1980s.

A review of the content of definitions shows that there is no separation between definitions according to origin (marine or freshwater environment) or the target audience (technical report or scientific publication). However, the review points to a contrast between generic, consensual definitions of little informative value on the one hand, and definitions that provide more information but are further removed from the main consensus (Fig. 2.8).

The most generic definitions reflect scientific knowledge applicable to all situations (natural and/or anthropogenic eutrophication) and all types of aquatic ecosystem. They refer to a process of evolution of the ecosystem’s trophic state, i.e. the process of evolution of its capacity to produce organic matter, after nutrient enrichment of the environment. Organic carbon production per unit of time and volume is a criterion used to categorize the states of aquatic ecosystems, with an index ranging from oligotrophic to hyper-eutrophic. For example, Rabalais et al. (2004) define eutrophication as ‘the increase in the rate of carbon production and carbon accumulation in an aquatic ecosystem’. While Smith et al. (1999) say that: ‘Eutrophication is the process by which water bodies are made more eutrophic through an increase in their nutrient supply’. The publication from which this definition comes is, at the time of this review, one of the most frequently quoted. However, the definition, which is somewhat tautological, does not cover the multiple effects caused by nutrient inputs. The same can be said of other generic definitions which steer away from an encyclopaedic survey but omit certain impacts.

![Figure 2.8. Example showing the array of definitions taken from multiple correspondence analysis (MCA). The definitions are illustrated by points in space, according to their content (nutrient input type, direct and indirect effects of those inputs on the ecosystem). Generic definitions made up of the most common key words are usually positioned in the centre of the graph, while the least common definitions are furthest from the centre.](image-url)
On the other hand, the definition by Carpenter et al. (1998), also taken from the most frequently quoted scientific resources in this appraisal, takes an approach that is markedly opposed to the generic definitions. Those authors define eutrophication through its most significant direct and indirect effects: ‘Eutrophication is characterized by blooms of noxious algae, excessive growth of aquatic macrophytes, episodes of anoxia, dominance of the zooplankton by small, inefficient grazers, and dominance of the fish biomass by benthivores’. Yet the effects mentioned do not apply to all cases of eutrophication. They are also far from representative of the full range of effects caused in ecosystems by an increased nutrient input. Some definitions specify the type of nutrient input. Take, for example, the definitions from documents on environmental management and aquatic ecosystem development, often more closely linked to anthropogenic eutrophication. ‘Nutrient pollution’ is hence the term adopted by the American EPA (Environmental Protection Agency) and the National Oceanic and Atmospheric Administration (NOAA) websites when referring to eutrophication. In Europe, the set-up of the Water Framework Directive spurred researchers to revise generic definitions to put the focus on the environmental aspect: ‘the enrichment of water by nutrients, especially nitrogen and phosphorus and organic matter’ (Andersen et al. 2006). The definition in the nitrate directive is one of the furthest removed from the centre in Fig. 2.8. This is mainly because this definition only mentions nitrogen as a nutrient, and does not refer to phosphorus.

Our review highlights the fact that one of the main difficulties in accurately defining eutrophication lies in the fact that the phenomenon occurs over both geological time, with no human involvement, and over time scales of less than one century, impacted by human activities. Another difficulty is summarizing the multitude of bio-geochemical and biological responses caused by nutrient inputs in just a few words. The notion of syndrome, which is defined as a set of symptoms or direct/indirect effects, is sometimes used in the literature to summarize this complexity.

### 2.2.2. Scientific definitions put forward during the ESCo appraisal

Although similar in terms of the mechanisms at play, ‘natural’ eutrophication and the anthropogenic form do not occur over the same time scales. They do not therefore have the same bio-physical and social implications. For anthropogenic eutrophication, nitrogen and phosphorus inputs have been clearly identified as the factors triggering an imbalance in the ecosystems over short time scales (a few decades). With ‘natural’ eutrophication, phosphorus and nitrogen are no different from other chemical elements; the aquatic ecosystem accumulates and concentrates them over geological time scales as and when the aquatic environment receives organic and mineral inputs. With regard to the speed with which nutrients are accumulated in the environment, the evolution of ecosystems from an oligotrophic state to a eutrophic state also occurs on a different time scale: in the case of anthropogenic eutrophication, the ecosystem’s response is felt very quickly (counted in hours, days, weeks, months or years) with multiple symptoms (syndrome), while in ‘natural’ eutrophication, the ecosystem evolves very slowly and the changes can rarely be seen within a human lifetime.

The polysemy of the word ‘eutrophication’ is therefore an issue in that it associates a process with a neutral–or even positive–connotation (natural evolution) with a negative connotation (pollution). As a result, within the framework of this appraisal where the goal is to ‘give consideration to the requirements and operational challenges of public action’, it appears necessary to point out this marked difference in time scale and put forward two definitions:

**‘Natural’ or ‘geological eutrophication**: An increase in the production of organic matter as an aquatic ecosystem evolves over a geological time scale; lakes may accumulate so much sediment that they eventually fill in.

**Anthropogenic eutrophication**: Syndrome* of an aquatic ecosystem associated with the overproduction of organic material induced by anthropogenic inputs of phosphorus and nitrogen.
The notion of syndrome, which is defined as a set of symptoms, is used to overcome the difficulty of summarizing the multitude of bio-geochemical and biological responses engendered by nutrient inputs in just a few words. It covers all the direct and indirect effects induced by these inputs (cf. Fig. 2.7) including the sometimes toxic plant proliferations, hypoxia and anoxia, modifications to the biological community structure, trophic networks, bio-geochemical cycles, and the alteration of the diversity and ecological functioning of aquatic ecosystems.

The arbitrary nature of this separation should be noted, as an aquatic ecosystem evidently combines the effects of natural eutrophication as it reaches a certain age and those of anthropogenic eutrophication. Also note that the use of the term ‘input’ in the anthropogenic definition avoids the ambiguity of the term ‘enrichment’.

Nota Bene: these definitions do not take into consideration one-off cases of eutrophication resulting from upwelling phenomena, bird concentrations, or organic matter concentration caused by the mechanical effect of currents and storms, where the eutrophication phenomena are not due to an excess of nutrients of anthropogenic origin. For example, in the marine environment, upwellings convey nutrient-rich water to the surface and thus boost phytoplankton growth and an accumulation of organic matter, beneficial to the trophic network but also generating anoxic zones. Upwellings and their impacts are often seasonal but can also be pretty much permanent, such as the phenomena occurring off the coasts of Peru and Namibia.

2.3. Inputs, transfers, retention and transformation of N and P along the land-sea continuum

Of the four factors that control eutrophication, water residence time and light are mainly dependent on local conditions. Water temperature and nutrient inputs are partially local too, but also depend on the watercourses and watersheds that feed them. Nutrient inputs can thus come from several hundred or even several thousand kilometres away and their transit time from the source zone to the recipient aquatic ecosystem can take several decades. The risk of eutrophication of an aquatic ecosystem and the measures required to restore it to its normal functioning state therefore depend on local conditions and nutrient inputs from its watershed.

We need to remember that the water cycle forms a very strong link between terrestrial and marine ecosystems since the evaporation of fresh and marine waters under the effects of solar energy forms clouds that then give precipitation. Rain water can be stored in ice, soils, wetlands, aquifers, ponds and lakes for a certain length of time, but ultimately it is returned via watercourses to estuaries, lagoons and the sea. This water carries dissolved and particulate elements, with nutrients (nitrogen, phosphorus, silica, etc.) generated in the watershed through erosion or dissolution of soils, sediments, rocks and alterites (Fig. 2.9).
2.3.1. Estimated N and P inputs from watersheds

The nitrogen and phosphorus cycles

Nitrogen and phosphorus are often taken together as they are both seen as the main factors triggering eutrophication phenomena in aquatic ecosystems. However, there is a major difference between nitrogen and phosphorus cycles. The nitrogen cycle comprises gas phases, the main one of which is inert (N₂) but there are other, highly reactive phases (N₂O, NO, NH₃ in particular); there are no gas phases in the phosphorus cycle.

Nitrogen is found in two states in the natural world: in a free state in the form of N₂ gas, making up 79% of the air that we breathe, and in a combined state in the form of a dissolved mineral – ammoniac, nitrite or nitrate – or in organic form. Nitrogen can be fixed in its atmospheric gas form by certain organisms, known as diazotrophs. Being able to fix atmospheric nitrogen gives these organisms an advantage over other species, especially when dissolved nitrogen in mineral form is in short supply. There is a great diversity of diazotroph organisms: the nitrogen-fixing function can be found in most of the major bacterial groups. However, only some cyanobacteria and some photosynthetic bacteria are able to fix carbon via photosynthesis and nitrogen via diazotrophy. Diazotrophic cyanobacteria are found in fresh and salt water, although the ability to fix atmospheric nitrogen is greater in freshwater areas. In fact, unlike in inland waters, nitrogen fixing in a marine environment is limited by slower growth among cyanobacteria and lower bioavailability of molybdenum (a vital co-factor in the functioning of the enzyme responsible for fixing and converting nitrogen, dinitrogenase). The discovery of the Haber-Bosch method made it possible to fix atmospheric nitrogen using an industrial process, which is the basis for mineral nitrogen fertilizers now used in farming.

Nitrogen absorption by plants and micro-organisms mainly occurs in dissolved mineral form (ammoniac NH₄⁺ or nitrate NO₃⁻). These forms are naturally present as a result of the symbiotic fixing of nitrogen by plants, or of the mineralization of organic matter (ammonification and nitrification) in soils and sediments (Fig. 2.10). Nitrogen may also be provided by mineral (ammonium) and organic (livestock manure, sludge, compost and the like) fertilizers.

Nitrate nitrogen can be turned into molecular nitrogen gas (N₂), primarily through the process of denitrification by bacteria and fungi. These organisms are able to use the oxygen contained in the nitrate molecule (NO₃⁻) as an electron acceptor during respiration in the absence of free oxygen (O₂) (anoxia), and thereby convert nitrates into inert nitrogen (N₂) or nitrous oxide (N₂O). Emissions of this later gas contribute to destruction of the ozone layer. Potentially therefore, denitrification can be responsible for a transfer of pollution from water (nitrates) to the atmosphere (N₂O). It should be noted that annamox, the microbial reduction of ammonia and nitrate into N₂, can be an important process in nitrogen reduction in the marine environment.

The largest portion of mineral nitrogen is transferred to watersheds in the form of dissolved nitrate; organic nitrogen is also largely transported in its dissolved state.

Phosphorus is preferentially absorbed by living organisms in the dissolved phosphate form (PO₄³⁻), naturally drawn from the soil or from inputs; the same is true of nitrogen provided in mineral or organic fertilizers (livestock manure, sludge, compost and the like). The complete phosphorus cycle occurs in liquid or solid form. Most phosphorus is therefore transported via surface runoff and subsurface flows into watercourses. Although phosphorus cannot be eliminated as an inert gas like nitrogen, it can be transported long distances by air, adsorbed on very fine particles, and can considerably enrich oligotrophic lake or marine ecosystems. The
The absence of a gas phase in the phosphorus cycle leads to a relative enrichment in phosphorus, compared to nitrogen, in aquatic environments (Fig. 2.10).

In watercourses, dissolved forms of phosphorus are transported either by diffusion or by advection, and are subject to exchanges with solid matter (particles, banks, etc.), primarily by adsorption-desorption processes. The particulate forms of phosphorus are transported by erosion-deposit processes and may sediment mechanically, depending on local hydraulic conditions (millstream, threshold, flood zone, etc.); while the dissolved forms tend towards an ionic balance with phosphorus concentration in sediment interstitial water.

**HOW ARE NUTRIENT CONCENTRATIONS EXPRESSED?**

Nutrient concentrations are expressed in different ways. In an inland environment, values are expressed as a mass concentration (mg/l). They may refer to the concentration of an element (e.g. N-NO₃ for nitrate nitrogen) or its molecule (NO₃⁻). A concentration of 50 mg/l NO₃⁻ is equivalent to a concentration of 11.3 mg/l N-NO₃ (the molar mass of nitrogen being 14 and that of oxygen 16, a rule of 3 can be used to calculate the proportion of the weight of the nitrogen atom compared to that of three oxygen atoms in a mole of nitrate). In a marine environment, values are usually expressed as a molar concentration (µmol/l). A concentration of 50 mg/l NO₃⁻ is equal to 806 µmol/l (given that nitrate weighs 62 g/mole (or 62 mg/mmol), 50 mg of nitrate corresponds to 806 µmoles).
Forms and quantities of nutrient inputs

Nitrogen and phosphorus from human activity can be transferred to watercourses on a point or non-point basis. Point sources (such as industrial discharge or sewage plants) are relatively easy to identify and quantify, as long as measurements are taken and made available, because the discharge is clearly defined in spatial terms. Non-point sources are much more difficult to determine and quantify with regard to space and time. However, they account for the largest share of nitrogen inputs and a considerable portion of phosphorus inputs in watercourses. The consequences of point and non-point sources on eutrophication are similar. However, their transit time from pollution sources to aquatic ecosystems differs. Point sources are, by definition, directly discharged into aquatic ecosystems while non-point sources are transferred over or through soils, by surface runoff or through subsurface or deeper ground flow. This means that the retention of inputs from direct point sources is only effective in aquatic ecosystems and their extensions (flood zones, oxbow lakes, sediments, etc.), while non-point sources can be regulated in watersheds and in semi-natural and agricultural areas, in the various landscape structures that they contain.

Estimates of nutrient flows and their origins vary from one publication to another, depending on the approach and the databases used. Some figures are calculated using pressure factors, which may be discharge data, quantities of fertilizer inputs, or agronomic surpluses. Other figures are established by calculating flows to the sea, estimated by modelling their origins. Taking estimates of flows out to sea and the most recent models applied at a global level, agriculture is the dominant source of nutrients (50% of N released and 55% of P) when it comes to surface water. On a global scale, mass balances at outlets between nitrogen inputs and outputs in dissolved or particulate form show that a third of nitrogen inputs and two-thirds of phosphorus inputs are retained in terrestrial ecosystems. At a national level, specific flows (kg/ha/year from the outlet of a watershed) may vary from one watershed to another, by an order of magnitude of 10; nitrogen flows are on average 40 times higher than phosphorus flows. This variability is related to pressures and to the watersheds’ retention capacities. For nitrogen, retention capacity first depends on effective rainfall, water residence time in aquifers, and the extension of wetlands; for phosphorus, it depends on the connectivity of runoff between source and watercourse. Dissolved nitrate is the dominant transported form of nitrogen while particulate phosphorus is the dominant transported form of phosphorus.

2.3.2. Main N and P apparent retention hotspots

Soils: more effective in retaining phosphorus than nitrogen

Soils are the main interface between non-point source nitrogen and phosphorus inputs and watercourses. They are the principal landscape unit for nitrogen and phosphorus retention in watersheds. Soils receive mineral (ammonium-nitrate) and organic (livestock manure, sludge, etc.) fertilizers. The positively charged ammonium ion is barely mobile in soil because it is adsorbed on clay. On the other hand, the nitrate, negatively charged, is highly mobile and easily transported by leaching in soils and later subsurface or deeper waters. As a result, action to retain or eliminate nitrogen must first be taken in soils, for example by limiting inputs (mineral and/or organic), introducing winter plant cover to assimilate nitrogen and transfer it more slowly, by selected rotations that absorb nitrogen as it is produced, thus avoiding its transport via leaching at the surface or in underlying ground water, or by returning carbon-rich plant residues to the soil, aiding microbial reorganization of the soil’s nitrogen. Denitrification can be significant in soils, even in generally well-aired conditions. For example, rain may temporarily saturate all or some of the soil. Very often, anaerobic microsites can be found in generally aerobic soil matrices, creating zones conducive to denitrification at a local level. High organic matter as well as clay and silt contents favor the retention of nitrogen in soil and its elimination by denitrification, leading to a net loss for ecosystems.

The capacity of soil and sediment to retain phosphorus depends on pH, mineralogy and the organic matter they comprise. Where well-aired soils are maintained, with a high percentage of organic matter, silt and clay, phospho-
Phosphorus retention is favored. Phosphorus adsorption sites are manifold but finite in quantity, because the capacity of soil and sediment to retain phosphorus is also dependent on the quantity of phosphorus already adsorbed. Phosphorus losses in soils are mainly attributed to soil erosion, as phosphorus is transported in particulate form, adsorbed to clay and mineral elements (primarily iron, aluminium and calcium). Hence, maintaining plant cover on soil helps retain phosphorus by limiting erosion. While measurements of P content extractable from soil are frequent, given their link to the question of plant biomass production, total P content measurements are less common. Estimates for soil P saturation rates are only common in a few countries, mainly those where eutrophication of lakes is an issue. The array of methods used to measure extractable P from soils does not facilitate the comparison of the P retention capacity of soils. The notion of soil P saturation rate exists but the way in which it is estimated differs. It should be noted that while soil is an effective system for storing phosphorus, it does not eliminate it. This means that it can be remobilized according to the environmental conditions described above, and can be transported along watersheds and down to the watercourses that drain them. Soil is therefore an effective but temporary retention system for phosphorus.

**Water tables: long-term nitrate reservoirs**

Nitrate are the dominant form of nitrogen transfers from soils to aquifers. Excess nitrates in soils, i.e. nitrates that are not absorbed by plants and micro-organisms or that are not denitrified, are leached in the root zone and move out into aquifers. Aquifers are the main reservoir for nitrates in the inland environment. The average age of the water in the aquifers may be up to several decades in small watersheds (basin heads) or several hundred years in deep aquifers feeding the major watercourses. The lengthy retention of water in aquifers, along with nitrates, partially explains the very gradual decrease in nitrate concentrations in watercourses, despite the measures taken to restrict inputs. Most runoff from watercourses in our temperate oceanic climates comes from runoff from subsurface or deep aquifers. A very small proportion of organic carbon in soils percolates in the water tables, limiting the possibility of heterotrophic denitrification by organisms using organic matter as an energy source. However, in some aquifers, there may be chemolithotrophic denitrification by bacteria using the oxidation of minerals as an energy source in the presence of pyrite (FeS₂), which is oxidized to sulphate (SO₄) while nitrates are reduced into nitrogen gas (N₂). The usually long transit times (several decades) in deep aquifers, and even in basin heads, can aid the process, as long as there is pyrite present, which is not easy to predict.

The high capacity for phosphorus retention in soils, apart from in very sandy soils, limits its transfer to aquifers. Phosphorus is therefore rarely present in groundwater.

**Wetlands: natural systems for nitrogen removal and phosphorus retention**

Wetlands trap fine sediment and organic matter transported during flood events. Natural and constructed wetlands are ideal sites for denitrification and phosphorus retention. Wetlands are usually the site of high organic matter production, matter that stocks phosphorus and retains it in organic form given the low levels of organic matter decomposition.

Riparian woodlands are wet wooded areas along the banks of rivers and lakes. Given their topographic position, between slopes and aquatic systems, they function as an interface for the transfer of water and dissolved and particulate matter. Their bio-geochemical functioning is dependent on local hydrodynamics. Fluctuations in the water table or river flood events affect soil aeration in riparian woodlands to varying degrees, depending on local topography. Low-lying zones near lakes or rivers will be more frequently waterlogged and subject to temporary or permanent anoxia, conducive to denitrification; zones that are higher up may experience anoxia on a more temporary basis. Riparian woodlands may therefore act as buffer zones for non-point nitrogen pollution from watersheds.

Flood zones have a similar bio-geochemical function as riparian woodlands. They provide an interface between aerobic and anaerobic conditions during flooding, conducive to denitrification and the accumulation of sediment and organic matter where phosphorus can be retained. Flood zones are rare in watershed heads but large flood zones can be found in bigger watersheds when watercourses are not diked. They are primarily found downstream of hydrosystems.
**Watercourses: intermediate storage systems for phosphorus**

Watercourses can retain and use phosphorus in their sediment, through the action of micro-organisms and algae. This retention capacity also depends on the concentration of phosphorus in the water and sediment, and on the complexity of the geomorphological structure of the watercourse. An increase in water residence time in a section of the river, caused by a slowing of the current, can lead to suspended phosphorus-containing mineral and organic matter being deposited. Water residence time also increases the time of contact with phosphorus associated with sediment and biological organisms, aiding its adsorption or absorption. Generally, anything that slows down the flow of river water and aids exchanges between the water and sediment, be it the presence of riffles and pools, meanders, secondary channels or logjams, is conducive to nitrogen removal through denitrification. At the scale of the hydrographic networks, the denitrification rate is higher in small, shallow watercourses than in the major rivers. In addition, the relative overall length of small rivers (70–80% of the total length of the hydrographic networks) further increases the prominence of denitrification in basin head river sediments. It should however be noted that water retention in watercourses can trigger a eutrophication risk in the sections concerned if there are excess nutrient inputs.

**Lakes and dams: phosphorus traps and nitrogen removal systems**

The sediment in lakes and dams can host an anoxic zone of varying depth where active nitrogen removal can occur through denitrification. Here again, denitrification capacity depends on water residence time and the ratio between the sediment surface and water volume.

Lakes, ponds and artificial dams are also important areas of phosphorus storage. It should be pointed out that phosphorus most often remains trapped in sediment for decades; it may be released depending on aeration conditions in the sediment and the concentration balances between water and sediment. The release of phosphorus feeds the eutrophication process and can explain the often mixed results of water quality restoration programmes in lakes, even after considerable limitation of inputs.

Natural lakes and dams are also silica retention systems; silica is a vital nutrient for diatom growth. Where there are several lakes or dams within an agricultural area, the imbalance between the natural N, P and Si ratios already caused by fertilizer inputs, may be reinforced as N:Si and P:Si ratios are increased. The outcome is limited diatom growth in fresh or marine water aquatic ecosystems downstream.

**Estuaries: Denitrification and phosphorus transfer zones**

Fresh and marine waters, subject tide and flood cycles, mix in estuaries. This causes sediment to be suspended all along the water column, and the formation of what is generally called the silt plug. This turbid mixture is not conducive to algal growth because of the lack of light, but it is a very active bio-geochemical reactor within the water column.

Anoxic conditions can therefore be frequent and very marked, mainly due to the bacterial decomposition of dissolved organic and particulate matter from upstream watercourses. The development of anoxic conditions is also related to the capacity for oxygen dissolution in the water, which fails as salt content rises and thus with the arrival of marine water. Ammonium, which is most often of urban or industrial origin near estuary zones, can be nitrified in zones where oxygen is present, thus contributing to a fall in dissolved oxygen in the silt plug. Anoxia phases in the estuary affect native invertebrates and fish but are also a physiological barrier to the migration of heritage species such as migrating fish (salmon, eels, etc.).

In estuary zones that often have low oxygen content, nitrates from watercourses can be denitrified. The bioavailability of phosphorus is aided, as its desorption from sedimentary clays is facilitated in the highly reductant, quasi-anoxic context caused by the decomposition of sedimented organic matter. Non-denitrified nitrates and non-consumed phosphorus, along with non-decomposed organic matter, are transferred to the marine environment and may form brackish water plumes over tens or hundreds of kilometres off or along the coasts.
Lagoons: denitrification and phosphorus accumulation zones

Lagoons exchange directly with the sea via one or several permanent or temporary inlets, and sometimes indirectly if there is a groundwater table underneath the spit. Depending on freshwater inputs and exchanges with the sea, a lagoon is generally brackish. With their capacity to store water, lagoons buffer fluctuations in flows from the sea and help decant particulate matter from watersheds, while enabling gradual transition towards saline conditions. Nitrogen transfer to lagoons occurs in nitrate form from late autumn to early spring, gradually via marine inputs or in bursts when watercourse levels rise, which may be very brief but intense in Mediterranean climates.

The transfer of ammoniacal nitrogen and phosphorus mainly occurs via urban discharge and is spread over the year, but can be more intense in the summer tourist season. In summer, a fall in inputs, consumption by algae and denitrification in deep hypoxic waters reduces dissolved nitrogen nutrient stocks; meanwhile large quantities of phosphate, which come from the remineralization of the considerable phosphorus sediment stock found at shallow depths, are diffused from the bed, especially in warm climates, maintaining algal bloom. The presence of shellfish farming contributes to the transfer of some of the phytoplankton nitrogen and phosphorus to the sediment, via bio-deposits.

Coastal areas: variable nutrient retention zones

The continental shelf has always received nutrient inputs from continental rivers, resulting in a natural enrichment gradient decreasing from the coast out to the ocean. In temperate zones, the nitrogen-rich situation can be caricatured by considering that if we take the average ocean concentration as a unit, that of an estuary non-enriched by human activity will be around 10 times higher, while that of a river draining a watershed where farming dominates will be 100 times higher. A considerable transfer of nitrogen nutrients (mainly in nitrate form) therefore occurs in river plumes, the reach of which can stretch several hundred kilometres for large, fast-flowing rivers and which are gradually diluted. The geographic extent of phosphorus enrichment is much more confined around the hypoxic estuary due to the high level of re-adsorption of phosphorus on clay particles when they are found in a well-oxygenated marine context where they sediment quickly.

Rocky or sandy areas with strong hydrodynamics do not accumulate N or P while sheltered bays and coastal gyres, conducive to sedimentation, can accumulate detritic organic nitrogen and phosphorus adsorbed on organo-mineral particles in their bed, where they remain trapped as long as the superficial sediment remains well oxygenated. In winter, these sedimentary stocks are only remobilized sporadically during very severe storms, with no major risk of eutrophication. In summer, in contained, shallow eutrophic sites, anoxia at depth can release high quantities of phosphorus and nitrogen at a time when algal flora lacks these nutrients, thus fuelling the eutrophication process. The transfer of nitrogen from inland areas to the coastal ocean surface can also occur with the atmospheric deposition of nitrogen as a gas or via aerosols, as occurs in the central zone of the North Sea.

2.3.3. Assessment of the cumulative effects of P retention and denitrification hotspots in watersheds

Here we should recall the fact that phosphorus is less mobile than nitrogen and is primarily retained in soils and sediments. Nitrogen is more mobile and largely transferred in dissolved nitrate form. Nitrites can be denitrified in sediments and soils and can accumulate in aquifers. Nitrogen may be retained in soils in organic and ammoniacal form. The differences in transfer mechanisms, retention times and elimination capacities for nitrogen and phosphorus engender differences in the mass ratio of these two elements from watershed heads and all along the land-sea continuum. This also explains why the assessment of phosphorus retention capacities and nitrogen retention and elimination capacities in a watershed is currently so difficult and marred by a high degree of uncertainty, even in watershed heads which are the most conducive areas for this kind of estimation. There
is a high degree of variability in watershed heads, but the relationships between landscape structures (land use spatial arrangement) and the quality of water in the rivers that drain them have not been established. While one or several structures may be assessed using specific equipment and measurements, it is not possible to quantify all landscape configurations. Nor is it possible to extrapolate the rates measured from one site to another because of the specific hydrological, hydro-geomorphological and bio-geochemical features of each site, leading to wide spatio-temporal variability in denitrification and phosphorus retention.

2.4. The N and P controversy

The literature contains some controversy over the respective roles of N and P in triggering eutrophication. Some authors believe that the regulation of eutrophication phenomena in freshwater requires the restriction of P inputs only. In their view, this element is the only factor limiting the growth of primary producers since nitrogen acquisition can occur via the activity of diazotroph cyanobacteria.

For fresh water, this hypothesis was mainly formulated on the basis of results from studies involving the long-term monitoring of nitrogen, phosphorus or nitrogen and phosphorus enrichments of a nutrient-poor lake, Lake 227 in northwest Ontario in Canada, then the monitoring of its evolution after enrichment was stopped. The studies clearly demonstrated that the removal of phosphorus inputs limited eutrophication, which led to the authors’ recommendation to limit phosphorus inputs only to eradicate eutrophication in both fresh water and brackish ecosystems. The authors of these studies back up their arguments with 1) the success of this unique, long-term experiment, 2) the fact that studies demonstrating the co-limitation of nitrogen and phosphorus are mainly based on laboratory trials or mesocosms and that these short-term experiments do not take into consideration ecosystem complexity, 3) the fact that the regulation of nitrogen inputs would be too expensive, and 4) they would not be useful as they would be offset by cyanobacterial atmospheric fixation.

This focus on regulating phosphorus inputs alone emerged in the United States several years ago, most notably with regard to Lake Erie. However, many recent studies have challenged this option and recommend a restriction of both nitrogen and phosphorus inputs in terrestrial, brackish and salt water. The main arguments endorsing the co-regulation of nitrogen and phosphorus to curb eutrophication are as follows:

- The oligotrophic lakes of North Canada are not representative of the great diversity of terrestrial, estuary, lagoon, coastal and marine aquatic ecosystems.
• It has been repeatedly demonstrated that algal blooms in coastal environments, especially green macroalgae, are highly influenced by nitrogen inputs, especially in summer, because while both nitrogen and phosphorus are not particularly abundant in marine surface water, the mineralization of detritic organic nitrogen from spring bloom is much slower than that of detritic organic phosphorus, usually leading to nitrogen deficiency in marine waters in summer.

• Several studies (including one at Lake 227 in Ontario again) have shown that nitrogen fixing by cyanobacteria is not equivalent to the inputs that usually come from watersheds.

• As phosphorus does not really have a gas phase, it accumulates faster than nitrogen in anthropized aquatic ecosystems. Its steady build-up in the sediments of watercourses, lakes, estuaries and lagoons means it is increasingly bioavailable for aquatic plants and algae, which is why it is also necessary to control nitrogen inputs, as nitrogen becomes the element limiting primary producer growth.

• The strategy based on reducing phosphorus inputs only as a means of stemming eutrophication has not been successful to date. For example, the Apopka and Okeechobee lakes in the USA, Lake Erie between the USA and Canada, and the Taihu and Donghu lakes in China have experienced algal blooms recently, despite considerable efforts to limit phosphorus inputs only.

• When phosphorus only is controlled, the disproportionate transport of nitrogen compared to phosphorus leads to transfers out to sea, as has occurred in the Neuse estuary in North Carolina or in the Wadden Sea between the Netherlands and Denmark.

There is now a panel of objective knowledge supporting the consensus widely shared among scientists that we need to limit both nitrogen and phosphorus inputs in aquatic ecosystems, with regard to point and non-point sources of urban, industrial, agricultural origin or stemming from the use of fossil fuels.

The controversy over the factor(s) controlling eutrophication underscores several elements that are to be taken into consideration when addressing this complex process. Firstly, it shows the difficulty in extrapolating results obtained from one type of ecosystem to another. Every ecosystem is unique and has its own history and dynamics, which are themselves related to local geological, geomorphological, hydrological, ecological and climate conditions, to past and present anthropogenic pressure, to their type and to the sociological and economic contexts to which they belong. Secondly, aquatic ecosystems are complex, open systems and their response times to anthropogenic inputs can be very long (several decades) and non-linear, therefore requiring long-term monitoring. Thirdly, nutrient cycles are not isolated from one another and measures taken to control one element will have consequences on the others and, ultimately, on ecosystem balance.

2.5. The concept of vulnerability

The notion of vulnerability is used in various bio-physical, ecological and socio-economic subject areas to measure a risk of direct or indirect alteration of human well-being\(^4\) or ecosystems. The main worldwide assessments, such as the Millenium Ecosystem Assessment (MEA), or the Intergovernmental Panel on Climate Change (IPCC) use the concept within the framework of the United Nations.

Vulnerability analysis implies 1) a target, part of an overall system (e.g. riverbank ecosystems with regard to floods, coastal ecosystems with regard to erosion, segments of a watercourse with regard to pollution), 2) a risk of alteration of that target (e.g. for floods and erosion, we will look at meteorological hazards and their reflection in hydraulic characteristics; for pollution, we will look at pollutant loads), 3) the sensitivity of the target to alteration, i.e. its capacity to withstand the disruptive event(s) and their intensity, and 4) a capacity to rebuild itself subsequently (resilience).

These components in the notion of vulnerability vary according to the intrinsic properties of recipient ecosystems and to the social issues at stake. Vulnerability must therefore be carefully defined for these different compo-

\(^4\) In the sense of the MEA, this notion of well-being encompasses freedom of choice and action, health, good social relations and security. The situations that people perceive depend on local geography, culture and ecological contexts.
The concept of vulnerability applied to eutrophication

The notion of vulnerability is used in the global sphere of watercourse management because as a tool, it can help to adapt remedial actions to the physical and human realities of the situations under review. The majority of risk analyses put forward in the literature to date are developed on a hydro-geochemical basis: they calculate a transfer of nutrients from watersheds to watercourses according to different models, from the most rudimentary to the most sophisticated (chapter 5), and estimate a eutrophication risk solely on the basis of flow thresholds and nutrient concentrations. The relevance of the tools deployed is linked to mesh calculations of flows, the richness of the calibration data, and the explicitness of the link to the eutrophication phenomenon. For example, at European level, the Joint Research Centre (JRC) has published eutrophication risk defined a priori on N concentrations expressed in mg/l: 0.5 = low risk; 0.5 to 1.5 = moderate risk > 1.5 high risk. Bearing in mind that P is not taken into consideration, most notably for inland water bodies, and given the concentrations recorded on the European continent, which are generally much higher, this risk—calculated at too broad a scale (180 km²)—is theoretically unsuitable for precise identification of inland situations at risk and to define appropriate remedial actions.

Broadly speaking, there are still very few published examples in scientific literature covering both aspects (hydro-geochemical hazard and ecological vulnerability) of hydrosystem vulnerability to eutrophication; firstly, because the disciplines operate in distinct universes and do not communicate much, or because they operate at different scales (e.g. basin hydro-geochemistry of watersheds versus hydrosystem ecology) and, secondly, because the final target mainly concerns the marine environment, which leads to rather general overviews of continental hydrosystems, sometimes looking at N only, and more rarely N and P.

At this point, we should point out, with regard to the assertions put forward in Chapter 3, that scientific questioning is now very much focused on the capacity to account for situations subject to multiple stress factors and how to link scales; these are rapidly developing areas of research today.

Eutrophication risk considering the biological vulnerability of hydrosystems: an area to be developed

A review of the literature shows that the construction of a eutrophication risk analysis responds to different objectives depending on environments, selected targets, their vulnerability, the overall complexity to be tackled and, to a certain extent, the data that can be potentially used and their relevance. As a result, we have put forward the analytical framework, shown in Fig. 2.12.

Figure 2.12. Eutrophication risk analytical framework, including the climate hazard and the hydro-geochemical hazard, and ecological vulnerability of the recipient systems, which depends on their position on the land-sea continuum and on their structure, functioning and ecological resistance. In italics: the properties of the terrestrial and aquatic systems that contribute to the eutrophication process and condition its intensity.
SM: suspended matter; OM: organic matter.
In Fig. 2.12, the priority areas of knowledge to be developed concern the links between contingent properties and aquatic environments and the notions of vulnerability and ecological resilience, because of the co-existence of non-linear relationships between elements (not a conventional relationship of dose and effect, and the existence of threshold effects), lag effects (events with delayed or accumulative effects in time) and long-distant relationships along the land-sea continuums (the effect of basin heads and their flow of non-metabolized nutrients in coastal areas, for example). We also need to further explore the links between hazard and vulnerability, especially in their spatial and temporal dimensions (periods and spatial structures).

To handle this complexity, especially in a comprehensive risk analysis, it will be necessary to call on spatial coupling that makes use of geographic information systems and their related data layers, and presumably combine measured data, data from proxy approaches, estimated data from models along with their uncertainties, and expert opinion. The various disciplines are increasingly making use of probabilistic type models such as Bayesian networks, which have the advantage of being flexible when it comes to handling this kind of relationship, which set out causal patterns and which can be revised without extensive changes of architecture when probabilities are refined after in situ measurements or greater experience.

There is also a shift in the management paradigm, with construction decisions and implementation becoming incumbent on the operational sector. This suggests that any new policy should attempt to coordinate model construction and the strategic collection of data to calibrate the models and reduce uncertainties, while linking to the existing operational networks.

Finally, many questions have risen about the future behaviour of systems in the light of climate change (Chapter 3.4). The analytical framework provided in Fig. 2.12 is suited to the study of climate change scenarios (modification of hydrological, hydro-chemical and thermal hazards, changes in light period intensity, duration and frequency, and seasonality) or changes in land use and practices in terrestrial environments.
3. What are the trends in eutrophication?

Increasing population and urban concentration, agricultural industrialization related to phosphorus mining and to the chemical manufacturing process of mineral nitrogen (Haber-Bosch method) have led to an increase in flows and concentrations of nutrients in the environment, and ultimately in aquatic environments. The nitrogen cascade concept illustrates the sequence of effects on the soil, water and biodiversity, often mentioned in the literature, and that has been better assessed over time. Some of the recent literature refers to the Anthropocene as a new era during which human activity has influenced the environment. This literature broadly refers to the disturbance of biogeochemical cycles as a major component of the trends seen in recent decades. Changes in these biogeochemical cycles have caused major ecological disruption. Many iconic places around the world are subject to recurring eutrophication episodes: the Baltic Sea, the Laurentian Great Lakes, Chesapeake Bay, the Gulf of Mexico, the Venice Lagoon, a large number of lakes and coastal areas in China, Lake Victoria, the Brittany coast, etc. Most of these sites have been continuously and on a long term monitored. The collected data allows a detailed analysis, within a specific geographic area, of the dynamic and cross-correlations between pressures, hydro-climatic variables (temperature and discharge), environment (habitat and water quality) and biological responses. Regulatory monitoring networks do not always provide such a diversity of information and temporal depth, and therefore, provide a complementary vision via geographically aggregated data, indicating average statuses and variabilities. The major episodes mentioned above contribute to making eutrophication a public issue, becoming iconic “hydro-social” stages, highly investigated by human and social sciences. On the other hand, the non-point source pollution that causes some of these disruptions falls outside the scope of social sciences: it originates far away from these iconic places, in the vast territories of headwater catchments, and over the long term; this pollution is less dramatic and its consequences are locally less perceptible.
In this chapter, we examine the trends of all types of manifestations of eutrophication, whether iconic and visible, or more diffuse. We will start by charting the evolution of biogeochemical cycles at global scale, specifying the role played by human activity. We will then describe trends in manifestations of eutrophication, basing ourselves on the monitoring of emblematic or high-stake locations, well referenced geographically and duly instrumented over the long term. Systems’ responses to the implementation of remediation plans, when such plans exist, are well detailed: how do environments respond, and over what response time? The analysis is complemented with the findings from large-scale monitoring networks, covering large territories. We focus on the effects of global changes and the intrusion of invasive species: what are the first findings? Can they be predicted? Lastly, these trends in the manifestations of eutrophication are address in perspective with societal perceptions and responses.

3.1. Overall trend in nutrient fluxes, role of human activity

World population quadrupled in the twentieth century from 1.5 to 6 billion, currently standing at 7.4 billion (2016). 54% of the world population lives in urban areas, a proportion set to increase to 66% by 2050. In many areas, urbanization has densified around large mega-cities, often located near water, major rivers and especially in coastal areas. More than a billion people – mostly in Asia – live in coastal plains, mainly in cities and urbanized areas. These developments raise the question of waste and wastewater treatment, which is essential but poses a particular challenge in coastal areas.

Agriculture, for its part, has profoundly evolved, driven by three main factors: mechanization, raw materials and food transport, and the use of synthetic (N) and fossil fuel-derived (P) fertilizers. These changes led to a significant increase in productivity and lower production costs, as well as to a strong specialization of agricultural regions, disconnecting crop and livestock activities. This resulted in major changes in nitrogen and phosphorus cycles at local and global level: significant quantities of nitrogen and phosphorus were introduced in the form of animal feed in livestock systems and mineral fertilizers in arable lands. In both cases, these products stem predominantly from other geographic areas, or even other continents, often leading to significant, but very variable, surpluses from one region to another.

Nutrient fluxes vary from one publication to another, due to the chosen approach and the databases used. Based on the latest models deployed globally, the contribution of agriculture to outflows to the sea reportedly increased from 20% to 50% for nitrogen and from 35% to 55% for phosphorus in the twentieth century (figure 3.1).

![Figure 3.1. Change in sources of nitrogen (a) and phosphorus (b) sources contributing from terrestrial surface to sea water at global scale over 100 years. In mauve: atmospheric deposit; very light green: aquaculture; light green: wastewater; dark green: floodplain vegetation; circles: groundwater (agriculture); triangles: groundwater (natural); light blue: surface runoff (agriculture); dark blue: surface runoff (natural); orange: rock alteration. Source: Beusen et al. 2016.](image)
The massive introduction of nitrogen in continental surfaces is related to the discovery of the Haber-Bosch nitrogen fixation process in 1909, allowing the industrial manufacture of nitrogen fertilizers for agriculture. We should also mention the use of fossil fuels. These two types of inputs related to crop fertilization and food and feed are largely dominant, especially in Northern Europe and the United States. At European level, over the past century, biomass production increased fourfold while leakage to the atmosphere and ocean increased threefold (figure 3.2).

Since the twentieth century, the phosphorus found in continental surfaces has essentially been of mineral origin. Inputs to terrestrial surfaces as a result of mining increased substantially, to the extent that resource supplies now appear limited to a few decades. Changes in inputs in the form of crop fertilizers is well quantified (figure 3.3). Like for nitrogen, these changes disrupted the entire phosphorus cycle, notably by increasing soil phosphorus stocks and flows to continental and marine waters. However, the literature does not include charts similar to those developed for nitrogen due to the greater difficulty in estimating phosphorus flows and stocks around the world (food, emissions to aquatic systems).
3.2. Analysis of long-term trend in different systems

In scientific literature, the analysis of long-term trends is often limited to main types of aquatic ecosystems, as communities of scientific ecologists, hydrologists and biogeochemists typically specialize on one type of ecosystem (lakes, rivers, coastal areas/sea) and, within these ecosystems, focus their activity on a few landmark sites. On these sites, the nutrients studied are generally nitrogen, phosphorus, carbon, and their speciations. Silica is not often studied as it is considered to be stable and substrate-specific, even though changes can be observed very indirectly and slowly in connection with the increase in the rock weathering process due to human activity.

3.2.1. Changes in lakes

Lakes played the role of sentinels with regard to eutrophication processes. They were the first to raise the alarm. Being semi-closed systems, lakes are more sensitive to the phenomenon of eutrophication. Identifying their role as a sentinel with regard to global changes (climate, land use, etc.) is an important finding not only for eutrophication, but also for water monitoring. The scientific literature on these systems dates from the 1960s. At the time, it focused on the Laurentian Great Lakes, before making a quantum leap in the early 1970s and gradually extending to different lake systems. We will examine here in detail the cases of Lake Erie and Lake Geneva, which both have original, well-documented trajectories, notably because they have been monitored for several decades.

The case of the Great Lakes cannot be extended to all semi-closed systems, however, as these water bodies differ widely in size and residence time, incoming flows, positioning of their outlets for hydroelectric dams (water release) or fish ponds (production), and serve very diverse purposes (drinking water reservoirs, agricultural reservoirs, wetlands of ecological value, fish ponds, etc.). Such water bodies are seldom the subject of regular monitoring which would help characterize changes in nitrogen and phosphorus concentrations and risks of manifestations of eutrophication, as pointed out in the joint scientific appraisal (ESCo) on the cumulative impact of reservoirs.

Lake Erie, from oligotrophication to re-eutrophication

Lake Erie, the fourth-largest and the shallowest (19 to 64 meters) of North America’s Laurentian Great Lakes, has a surface area of 25,800 km² and a watershed of 64,000 km². The average residence time of its waters is very short at only 2.2 years. Twelve million people live in the Lake Erie watershed and are mainly supplied with drinking water from the lake. The lake faces highly pressing issues and has a fraught history: in the 1950s and 1960s, it was known as “North America’s Dead Sea”, and monitoring and remediation programmes were implemented from a very early stage (International Joint Commission of Canada and the United States). It has been in gradual remission since the 1990s thanks to the reduction of phosphorus inputs, the objective of 11,000 tons per year having virtually been met. However, Lake Erie experienced extremely severe episodes of blue-green algae (cyanophytes) blooms in 2011 and 2014, as a result of which the city of Toledo (population 400,000) had to distribute bottled water to its inhabitants for several days due to excessive concentrations of microcystins (2.5 µg/L) in tap water (estimated cost $2.5m). These episodes recurred in 2015 and the risk is now such that in sensitive periods, forecast bulletins are published every week by several departments, including the NOAA (National Oceanic and Atmospheric Administration, satellite images), based on the predictions of various models combining meteorology, watershed, lake hydrodynamics and ecosystems (phytoplankton, cyanobacteria, cladocera, mussels (Dreissena), etc.).
This alarming situation prompted an investigation into the causes of this phenomenon in a watershed where significant efforts had been made, followed by a large number of publications. A consensus conference brought together over one hundred national and global experts in 2015 and the findings were published.

The relative proportion of non-point sources, in particular that of dissolved phosphorus, appears to be increasing constantly. This temporal difference in the nature of inputs, with a rise in a more bioavailable form of phosphorus, is a factor that could explain the resurgence of manifestations of eutrophication (more blue-green algae washed up on beaches, hypoxia, cyanobacterial blooms). This is compounded by climatic phenomena such as rising temperatures (intensity, duration, impact on phenology), a decrease in snow cover (volume, duration and earlier thaws) or the upward trend in rainfall, playing a role in the transfer of nutrients from agricultural land. In addition, the increase in non-point source inputs is also attributable to new agronomic practices carried out in the watershed, even though they are identified as good practices (BMP Best Management Practices: zero or minimum tillage, mulch keeping P in the surface layer of soil and making it more easily mobilizable), as well as an increase in aerial fertilizer spraying (increase in surfaces covered in relation with agricultural intensification).

Researchers have also reconsidered the dominant paradigm of the role of phosphorus as the primary limiting factor for phytoplankton. Thanks to contributions from molecular biology and field data, researchers demonstrated that the proliferation and toxicity of non-diazotrophic cyanobacterial blooms could be controlled by nitrogen: for instance, *Microcystis* has physiological adaptations that allow it to dominate in P-poor environments and whose development is controlled by N. This notably explains blooming dominated by *Microcystis* and *Planktothrix*. These results, supported by several years of observation of the western part of Lake Erie, show that microcystin peaks co-occur with peaks of inorganic nitrogen, and not orthophosphate, and that they are weaker in years with the lowest concentration in N. The US Environmental Protection Agency now recommends being vigilant about the N:P ratio and building common remediation strategies for the two elements, including in continental waters.

The analysis of Lake Erie was made possible by the quality of historical series and ongoing scientific support. This is a territory in which models have been deployed and combined (watersheds, lakes) to predict biological responses (macroinvertebrates and fish are used as indicators of the ecological status). These models help represent the system's temporal and spatial complexity and are used to test remediation scenarios *in silico*, contributing to making recommendations and finding answers to the question: what is an achievable ambition in the field of biophysical disciplines and what is its feasibility from a socio-economic point of view?

Two initiatives are consistent with a more holistic approach to hydrosystem management and remediation: (1) the characterization of cumulated stress factors for the Great Lakes, based on the analysis of their prevalence and the relative severity of their potential impact, combined with their mapping, and (2) the characterization and mapping of environmental services (recreational services taken into account: attendance at parks and beaches, boating, fishing, observation of wild fauna and flora), notably for the consistency of restoration policies, which currently amount to $1.5bn.

**Lake Geneva, a lake in a re-oligotrophication phase**

Lake Geneva's trajectory is part and parcel of its location on the Swiss border and of the long-standing dialogue established between scientists and administrators at CIPEL, the Franco-Swiss Commission responsible for monitoring the quality of the water in Lake Geneva, created in 1963. The concentration of research centres within a single space with high stakes (lifestyle, tourism and water supply) and the early implementation of extensive monitoring made it possible to diagnose the lake's state of degradation and its causes in the 1960s and to engage in a remediation programme for urban-related phosphorus emissions (almost one million residents, 500,000 tourists) based on two pillars: ban on phosphate detergents (Switzerland in 1986, France in 2007) and phosphorus removal equipment at water treatment plants. This action plan is tailored to the situation of Lake Geneva (endogenous source, high in nitrogen and low in phosphorus content; deep lake, predominant role of exogenous source of urban-related phosphorus): in this case, controlling P could pay off.

The first trend was the decrease in phosphorus concentrations, from 80-90 µg/l in the period 1975-1980 to 20 µg/l today (figure 3.4), CIPEL having set a target of 10-15 µg/l by 2020. A rapid increase in phosphorus concentrations over two decades was therefore followed by a slower response, with a decrease in phosphorus concentrations over forty years.
The lake’s trophic status is gradually improving, but the trend is far less pronounced for phytoplankton, for example, than it is for phosphorus. The composition of Lake Geneva’s fish population changed considerably between the period 1950-1975, when catches consisted mostly of perch, and today, when the bulk of the biomass caught consists of whitefish (around eight times more than in 1950-1970).

These developments reflect deep changes in the lake’s trophic functioning, which are not adequately represented by global indicators such as chlorophyll a. A better indicator is the composition of phytoplankton: a true sign of the re-oligotrophication of Lake Geneva is the increase in small organisms and mixotrophs, more characteristic of oligotrophic situations such as Lake Annecy’s. Recent, very significant improvements have also been noted at the Lac du Bourget, which also benefited from a remediation plan, and where the spectacular blooms of *Planktothrix rubescens*, more commonly known as the Burgundy-blood phenomenon, have stopped occurring. This particularly well-researched case study shows that trends are not attributable to just one forcing factor. Only detailed limnological analyses can help understand the dynamics at play, by combining information on lakes’ chemical memory accumulated in sediments, on climate forcings, on the intensity, duration, frequency and phenology of blooms (early springs, water exchange in the winter, benthic temperature, etc.), and on trophic relationships (phyto- and zooplankton, sizes, crustaceans, benthic compartment, fish).

### 3.2.2. Changes in streams and rivers

**General trends**

Streams and rivers have also recorded an increase in nitrogen and phosphorus levels. A recent compilation of 22 rivers in the United States shows a rise in nitrate levels going back a very long way (figure 3.5). Long series of historical data going back to the beginning of the twentieth century remain exceptional, but they all show a significant increase in nitrogen and phosphorus. Scientific literature focused on this phenomenon at a later stage, in the 1980s, and there was really an explosion of publications in the 1990s.
In Europe, records date back to the 1960s-1970s. Nitrate and phosphate concentrations multiplied by a factor of roughly 4 for nitrogen and 10 for phosphorus, on an annual average over 20 years, in a few major rivers (Rhine, from 1 to 3.5 mg N L⁻¹ and from 60 to 400 µg P L⁻¹ between 1954 and 1976; Seine, from 2 to 8 mg N L⁻¹ and from 10 to 130 µg P L⁻¹ between 1960 and 1980). Similar statistics have been published on the Thames, upstream from London. Compared with the background level considered as relative to non-man-made or little anthropized systems, nitrogen and phosphorus have increased by a factor of 2 on a global scale; in the United States and Western Europe, locally, increases are sometimes by a factor of 10 or more.

There is little research on the possible link between changes in land use and human activity on the watershed and concentrations in rivers and streams, whether resulting in a deterioration or an improvement, despite the implementation of regulatory follow-up. This is mainly because these changes are often very gradual and their effects are felt even more slowly. Only major changes and very long records, sometimes aggregated on a set of rivers, would provide information as to a possible correlation. For example, in the Thames, a connection between the rapid increase in nitrate concentrations in the 1970s and the widespread grassland destruction in the United Kingdom was established, based on modelling and a reconstitution of land use over more than a century. Climate events could also have been triggers (1976 drought and nitrate content in Europe), suggesting threshold effects and interaction between climate and agricultural activity. A study of 250 river stations (watershed size over 1,000 km²) in the United States, between 1974 and 1994, indicates that total phosphorus and nitrogen contents decreased in almost half of the stations while continuing to increase in some stations. Improvements occur mainly in watersheds dominated by woodland and meadows, stagnation in primarily agricultural watersheds.

Like for lakes, there are very little data available for a retrospective analysis of headwater catchments (Strahler stream order of less than 4). Given the lack of data, it is not possible to build a differentiated remediation strategy or set targets tailored to development trajectories, current situations and territorial issues related to these headwater catchments.

The Loire, a river of marked trends

The Loire is a river particularly sensitive to eutrophication, with strong agricultural and urban pressure, pronounced low-water periods, and a braided river morphology that slows the flow during low-water periods, giving phytoplankton plenty of time to develop. At the end of the 1970s, conditions were particularly eutrophic (figure 3.6), with
extreme concentrations of phytoplankton (> 250 µg L\(^{-1}\) of chlorophyll a during the summer) in the lower part of the Loire (mid-Loire and downstream) and a great diversity of phytoplankton. At the time, the Loire estuary was recognized as a regularly anoxic area in the summer due to the degradation of large amounts of labile organic matter. An analysis of trends in algal biomass and nutrients since 1980 in the mid-Loire and along the length of the Loire highlighted a decrease by a factor of almost 2 in orthophosphate concentrations since the mid-1990s. This was in evidence along the length of the Loire River and its main tributaries, clearly reflecting efforts to reduce point-source inputs (urban, industrial and agricultural) across the whole of the Loire basin. As a result, more attention has been given to non-point source inputs in recent years, as their proportion could now be relatively greater than that of point-source inputs. Indicators of phytoplankton biomass, summer concentrations of total pigments, daily variations in concentrations of dissolved oxygen and pH decreased by a factor of 2.5. However, to this day, still excessive nitrogen inputs continue to produce green tides and coloured offshore waters in the coastal area swept by the Loire plume.

Like for other European rivers, work on the mid-Loire highlighted a significant rise in water temperature in the last three decades (+0.9°C between the periods 1978-1987 and 1999-2008), together with a decline in flows over the same period (-80 m\(^3\) s\(^{-1}\)), mainly related to global atmospheric warming. According to a number of projections, these climate changes are likely to lead to a sharp increase in nutrient concentrations in streams and to related eutrophication phenomena. For the time being, the changes in water quality expected in the mid-Loire in this context have been offset by local efforts to reduce phosphorus concentrations by improving wastewater treatments. This has limited the development of phytoplankton communities, stimulated by the invasion and colonization of most of the Loire’s surface water bodies by corbicula, an invasive bivalve filter feeding mollusc.

With lower levels of phytoplankton, the water column gets more light, encouraging the development of macrophytes, which are havens for a fauna of invertebrates that is more diversified, but different from that existing prior to the eutrophication period. A growing proportion of taxa develop resistance or resilience strategies adapted to hydro-climatic changes (for example, small organisms develop a system known as ovoviviparity). It is also likely that primary production is transferred from phytoplankton to epibenthic biofilms and macrophytes. This increase in diversity also seems to be increasingly widespread at broader scales. A study on a selection
of around one hundred sites across France, monitored annually over 25 years (1987-2012), highlights a significant increase in the specific wealth of macroinvertebrates over the period, including for so-called “benchmark” sites preserved from the various sources of disruption. Based on an analysis of temporal changes in various food groups, it would seem that here too, these changes result from a double set of factors: the gradual rise in temperature and the general improvement in the chemical quality of streams, with a cascade effect on the complexification of the food web (longer food chains and diversification) – in other words a trophic magnification – within communities.

3.2.3. Changes in coastal waters

General trends

Scientific research on eutrophication in coastal waters started a little later, in the 1990s, but has been increasing steadily since then. Changes in marine nutrient concentrations are more difficult to trace than in continental waters due to the dilution phenomenon. Analyses focus either on outflows to sea, or directly on marine manifestations of eutrophication (chlorophyll, oxygen concentrations, etc.).

For outflows to sea, data at global level show variable trends. At the national level, the Ifremer network monitoring the chemical quality of French coastal waters (RNO, then ROCCH) was initiated in 1974 to monitor trends in nutrients. Two national summaries covering the periods 1974-1984 and 1985-2003 indicate that for ammonium, the situation clearly improved in sites with abnormally high contents (Dunkirk, Seine Bay and Fos-Berre), while nitrate levels continued to increase until 1993, especially in the Seine Bay and in the Brest Bay, before stabilizing or even slowly decreasing subsequently. Concerning phosphate, the Seine Bay, a notorious black spot, recorded a fourfold decrease in concentrations between 1985 and 2003, while other sites with lower contents also recorded a downward trend. The French national observatory of the sea and coastline (ONML) concluded that for the four major French rivers (Seine, Loire, Rhône and Garonne), nitrate-related nitrogen flows declined by 40% from 1999 to 2003-2004, then more or less stabilized until 2011 before going back significantly on the rise in recent years. Phosphorus flows were divided by four between 2000 and 2005 due to the improved performances of water treatment plants and the lesser use of phosphate fertilizers.

Regarding manifestations of eutrophication, the global number of hypoxia cases rose from a dozen reported in the 1960s to 415 cases in 2009. An interactive global map created in 2011 now lists 762 coastal areas affected by eutrophication, of which 479 affected by hypoxia or anoxia, 55 in remission, and 228 subject to other symptoms such as algal blooms, loss of species, or impacts on coral reefs. The case of the Chesapeake Bay, which had the grim privilege of being the first place on the planet to be declared a dead zone in the 1970s due to anoxia, is detailed below. For France, we examine the case of green tides on the Brittany Coast.

Chesapeake Bay, the first marine area to be declared a dead zone

The Chesapeake Bay is a particularly emblematic case of the synergistic action of different anthropic pressures, and particularly of the combined action of excessive inputs of nutrients and overfishing of filter-feeding bivalves. A great deal of research has been conducted in the bay, reconstructing the dynamics of the ecosystem since the arrival of the first European settlers and adding up to what is probably a fairly representative story of what happened over a longer period in very many other coastal ecosystems. Chesapeake Bay spans more than 300 kilometres inland. A shallow bay, it was covered widely with sea grass and populated with large numbers of fish, sharks, marine mammals and turtles until the beginning of the 18th century. A first change took place in the ecosystem when settlers undertook major deforestation on adjacent watersheds. At this time, sediment inputs increased in the bay, and the sea grass present on the bed diminished, leaving room for the development of phytoplankton. No significant eutrophication manifestation was observed during this period, however, as large natural oyster beds constantly filtered the bay’s water to feed, renewing it every three days on average and thus controlling algal proliferation.
In 1870, with the introduction of dragging, there was a boom in the exploitation of oyster stocks, with catches quickly reaching hundreds of thousands of tons annually. In just a few decades, the natural beds were destroyed and the fisheries collapsed. Oyster density became too low to have an impact on the bay’s filtration and the first episodes of eutrophication were recorded in the 1930s. These eutrophication episodes multiplied due to rising nitrate inputs in connection with industrial and urban development and agricultural intensification, leading to anoxia in the bay and ultimately to it being declared a dead zone in the 1970s. Since then, efforts to restore the water quality have proved highly inadequate and the ecosystem remains extremely degraded.

The green tides of Brittany

Marine macroalgae are widely present on the rocky coasts of the Atlantic and the English Channel. By their very mass, they naturally dominate the biocenosis of shoreline substrates and also control their biodiversity. As they are close to the coastline, they are the first in line when nutrient flows arrive. The consequences can be decreases in large brown algae belts (Fucales, Laminariales) or algal blooms involving small opportunistic species. Green algae blooms, consisting of species of the *Ulva* genus, are currently the most characteristic and most common. The first written report on a massive arrival of *ulva* in Brittany is to be found in a request for assistance from the town council of Saint-Michel-en-Grève to the prefect of the Côtes d’Armor, dated July 1971. Annual monitoring of green tides in western France, carried out since 2002 by the Algae Assessment Centre (CEVA), shows significant inter-annual variability of the blooms. The amounts of algae washed up can vary by as much as 100%, mainly as the result of two factors: 1) fluvial input of nitrogen nutrients during the growing period (May to August), 2) algal biomass present in early spring (residue of biomass produced the previous year). Overall, the total tonnage of green algae observed in Brittany quickly increased from the end of the 1960s to the mid-1980s before fluctuating around an average, broadly stabilized level, for the two reasons mentioned above.

### 3.3. Changes measured by monitoring networks

The data collected by monitoring networks are essential to assess situations in vast territories. These networks, some of which were created as early as the 1970s, gradually improved in terms of coverage density and homogeneity of the data collection protocol. In this respect, the EU Water Framework Directive (WFD) was an accelerator in spreading this homogenization across Europe, a requirement to compare different situations. The same applies to data collected at European level via the monitoring networks of OSPAR (since 1992) and of the Marine Strategy Framework Directive (MSFD, assessment of initial status in 2011) specifically for the marine environment.

#### 3.3.1. Findings of European monitoring networks

Bilan à l’échelle de l’Europe

The European Union published a first WFD assessment in 2009, stating that 42% of waterbodies were in good condition. Lakes and coastal waters are in better condition than streams and rivers and transitional waters. For freshwater, the most affected regions are in northern and central Europe, particularly northern Germany, the Netherlands, northern and western France and Belgium, while for coastal marine and transition waters, the Baltic Sea and the North Sea are the regions most impacted (figure 3.7).
Non-point nutrient pollution and the degradation of hydromorphology, which alter habitats, are identified as the main pressures responsible for surface waters failing to meet objectives. When analyzing these factors, it appears that degraded conditions for surface waters are attributable to non-point pollution in a proportion of 30% to 50%, mainly agricultural pollution (this observation of the European institutions is supported by the scientific literature). More than 40% of coastal and estuarine water bodies are affected by non-point sources, and 20%-25% of them are also prone to point-source pollution. Watersheds with more than 40% of farmed land and more than 100 inhabitants/km² represent two thirds of the water bodies not achieving good status.

There is a consensus to the effect that public policies to fight domestic pollution (water treatment plants equipped with third-generation technology and ban on phosphate detergents) have resulted in a 54% decrease in phosphorus concentration in water bodies. Very simplified projections even indicate that good status could be reached for phosphorus by the end of the last WFD cycle in 2027 (figure 3.8). There is no change in nitrates over the same period. In coastal and estuarine areas, European statistics do not show any changes in nitrates and chlorophyll a between 1985 and 2010.

OSPAR has developed a Common Procedure (COMPP) to assess the status of marine water eutrophication. The second assessment, conducted in 2007, shows that the problem is limited mainly to the North Sea and to the embayments and estuaries along the coast of the Celtic Seas, the Bay of Biscay and the Iberian Coast (figure 3.9). OSPAR assessments confirm the downward trend in phosphorus flows, sometimes far in excess of the initially targeted 50% for some member countries. Nitrogen inputs decrease more slowly and more erratically.
Summary reports produced by European institutions provide far greater detail than those selected here, but their analysis in connection with the issue of eutrophication would need to be on a finer scale: a lot of the data relate only to the district, which can be useful when assessing outlets in relation to coastal waters, but is far too coarse for the large number of continental basins. Furthermore, there are huge disparities in the sizes of water bodies defined by each country and even between regions within a single country.

Figure 3.9. Assessment of eutrophication in the OSPAR maritime area after implementation of the Common Procedure set up by the Convention. Source: OSPAR Commission 2009.

Status report at French level

In France, the monitoring of fertilization-related pressures shows a stable situation in the use of organic fertilizers and a moderate decrease in the use of mineral fertilizers (a small decrease for nitrogen, bigger for phosphorus) over the last 20 years. Despite the reduction in inputs and the more rational use of fertilizers, there are still high surpluses in breeding regions due to livestock density.

In the absence of a soil framework directive, there is no harmonized Europe-wide soil monitoring. A French initiative, RMQS (Réseau de Mesure de la Qualité des Sols), a nation-wide soil quality monitoring network implemented as part of the GIS Sol network, made it possible to map soil phosphorus contents in the 2000s. A second ongoing campaign will help understand trends. Studies in Brittany, using the BDAT soil analysis database, indicate an increase in soil-extractable phosphorus in the last 20 years, an increase that has slowed down in recent years. This work is currently being extended to the national level and shows a decrease in soil phosphorus. The decreases differ according to analytical methods. The decline is sharper for Olsen P which is more easily extractable (and therefore responds faster) than for Dyer phosphorus, which is still marked by past inputs.

In France, WFD control reference networks (RCR) and monitoring reference networks (RCS) were implemented in 2004-2006 and 2007, respectively. France is in the European average for the ecological quality of its water bodies, with 42% of surface water bodies in good condition, all types taken together (41.4% in 2010 and 43.4% in 2013). In 2013, physical chemistry was the most frequent factor involved in downgrades, for 18% of streams and rivers and 35% of lakes, followed by phytoplankton for inpondments (24%) and transition waters (16%), and macroalgae for coastal waters (14%). However, ecological status varies widely across the national territory. Densely populated or primarily agricultural areas (crops and livestock) generally have fewer waterbodies in good ecological condition. We find the same temporal trends for nitrogen and phosphorus in France and in Europe, for the same reasons: orthophosphates have declined since 1998, while nitrates are relatively stable. Groundwater nitrate trends are also stable.

OSPAR assessments confirm this trend, showing a 50% reduction in river inputs of phosphorus in coastal waters between 1990 and 2007. On the other hand, no significant change in nitrogen inputs has been observed.
3.3.2. Findings of monitoring networks in the United States

The United States has a long tradition of monitoring water bodies, regularly supported by technical and scientific summaries.

For inland surface waters, the reports for the 2008-2009 campaign show that 55% of waterways do not meet quality criteria for aquatic life, mainly due to nutrient contents or poor habitat conditions; 23% are in fairly good condition; 22% are in good condition and are home to healthy biological communities. Adding up the last two categories, we get an order of magnitude 45%, a level of good ecological condition comparable with European and French figures. Of the 55% of waterbodies in poor condition, 24% are degraded by artificialized riparian corridors (lack of vegetation and presence of infrastructure) and 15% by an excess of fine sediments. Lakes in bad environmental condition represent 22% of the total number (negative factors are the poor state of banks and excess nutrients).

A comparison with 2004 shows a 7% decrease in waterbodies in good condition and a 19% increase in the total length of rivers with above-standard values of phosphorus (9% increase for nitrates). On the other hand, 17% of the overall waterway length improved in terms of habitat and 12% for riparian corridors. Nitrogen and phosphorus levels in waterways are clearly identified as being responsible for the deterioration of water quality: 46% of waterways have above-standard values of phosphorus as defined by the US EPA (41% for nitrates). Over the same period (2004 versus 2008-2009), the ecological status (macroinvertebrates) declined slightly by 9%, as did the proportion of sites in good condition in terms of phosphorus content (14%), nitrogen remaining almost unchanged. Like in the comparison made for France, five years is too short a period to reveal trends, particularly for variables like phosphorus, which are highly dependent on water regimes. One should also bear in mind that large data collection systems on a large scale involve a degree of uncertainty.

These general statistics are accompanied by an interesting analytical approach, consisting in calculating the prevalence of pressures in waterways (by combined length or by number), to assign an impact intensity against a biological target (periphyton, macroinvertebrates and fish) and, lastly, assign a weighted and relative impact risk for each pressure. The results point out different regional patterns and especially reflect the predominantly negative impact that nutrients have on macroinvertebrates, fish and periphyton.

For marine waters, of the 1,100 stations studied in 2010 across the United States, a water quality index shows that only 36% have a water column in good condition (half of them are in fair condition). The index shows excess phosphorus in freshwater running into the sea (only 40% in good condition for phosphorus), significantly linked with agricultural intensity in watersheds, but no excess nitrogen (70% of the water is in good condition with respect to nitrogen).

3.4. Impact of global changes

3.4.1. Global changes and eutrophication

The impact of global changes on the mechanisms of eutrophication and on the future intensity of its manifestations is a crucial issue. The question has been much debated in the last twenty years and a substantial body of scientific literature already exists that would merit summarising. Virtually all environmental agencies worldwide have produced contributions on this issue for the attention of decision-makers, with messages on the increased risk of eutrophication.

One of the titles of this abundant literature sums up the general consensus: “Allied attack: climate change and eutrophication”. Climate change should not be considered as merely an added stress factor, but as a general disturbance that will impact all of the mechanisms involved in eutrophication and amplify its symptoms. Transfers wit-
hin watersheds, nutrient loads reaching hydrosystems, the physical chemistry of environments, especially oxygen, pH and discharges of P and metals from benthic sediments, the metabolism of nutrients in aquatic environments, the niche of organisms and their distribution, the dynamics of trophic networks, primary, secondary and tertiary producers; all of these processes are likely to be modified by forecast climate changes (changes in thermal and water regimes (rainfall) as well as changes associated with terrestrial landscapes).

In this synthesis, we will limit ourselves to addressing the main features of projected changes and their main effects likely to act on the eutrophication process (figure 3.10). Many changes are forecast in temperate regions, where there are many geographical nuances and major uncertainties. In addition to global warming, the intensity and seasonality of rainfall will be affected, with consequences on nutrient transfer patterns; for instance, heavier rains could increase erosion, land transfers and, ultimately, nutrient loading into aquatic environments. Conversely, longer dry periods will weaken summer flows, increasing the proportion contributed by more nutrient-rich aquifers to waterways. In aquatic environments, the effects will notably be influenced by their morphometry (submersion and exposure times of riverbank areas), the residence time of their water, their initial state in the trophic gradient, the complexity of their trophic chains and the P concentration of their sediment. All the symptoms of eutrophication could therefore be impacted: turbidity, either due to seasonal excesses of minerals in suspension (increase of erosion episodes), inhibiting phytoplankton production or, conversely, increasing this production at other times; summer stratification, particularly for dimictic lakes in which the stratification period could increase, accompanied by phases of prolonged hypoxia; increased risk of algal blooms, particularly cyanobacteria (blue-algae) blooms due to high temperatures and changes in the N/P ratio; lastly, alterations of the trophic structure and changes in fish production, to the benefit of species with shorter life-cycles that consume large quantities of zooplankton and to the detriment of cold-water salmoniform species, which are often carnivorous.

These descriptions are not purely speculative; they notably draw on the long experience accumulated in shallow northern lakes, their evolution as a result of recent warming, as well as studies of mesocosms (in test controlled experimentations) and long-term monitoring of estuaries.

Figure 3.11 highlights the functional situation of the different processes described, with their direct and indirect actions. Direct relationships include temperature, which influences the system’s general metabolism and each aquatic organism’s thermal comfort for the various physiological functions (gamete maturation, breeding, feeding and growth). Depending on future rises in temperature, both in average annual trend and in seasonal trends, which can be described in intensity, duration (in particular the sum of degree days favourable to a physiological function), frequency, but also speed of change, communities and their interrelationships will be altered. The most often-cited trivial change is a gradual transition from cold-water communities to temperate to hot-water communities, giving invasive species (§ 3.4.2) an opportunity to compete with native species. There could also be more complex evolutions depending on changes in the contributions of the various groups of algae orphanerogams to primary production: for instance, the triad of fixed submerged vegetation, floating vegetation and phytoplankton (microalgae, cyanobacteria) will enter into a temporal competition for light and nutrient concentration and will depend on the predatory pressure exerted by one on the other, particularly by zooplankton on phytoplankton. The latter process is detailed in figure 3.12. It is governed by a two-way check, ascending and descending: an increased load of nutrients in the aquatic environment, combined with a temperature stimulating the metabolism on the whole, will generate more algal biomass; this algal biomass is kept in check by zooplankton, itself a prey of planktivorous fish, which in turn are regulated by predatory fish (often salmonids). If predatory fish are disadvantaged by the new thermal conditions, planktivorous fish, mainly cyprinids (as well as whitefish in alpine lakes) will prey increasingly on zooplankton, as a result of which algal biomass will be less regulated.
Other possible developments are also mentioned in the literature and represented in figure 3.11, namely (1) an alteration in host/parasite relationships (increased growth with rise in temperature and increased virulence) or in epizootic diseases, potentially affecting both fish and zooplankton, (2) the unpredictable role of indigenous or introduced benthic filters (§ 3.4.2), and (3) the modification of the forms of available nitrogen, $\text{NH}_4^+$ or $\text{NO}_3^-$, as the proportions of the reduced forms can increase during episodes of hypoxia. Concerning this last point, recent research on marine phytoplankton notably indicates that the relative proportion of these two forms of $N$ and other stoichiometric relations influence the diversity of phytoplankton. This somewhat upsets the paradigm of a simple relationship between algae, chlorophyll $a$, total nitrogen and total phosphorus.

Finally, the most commonly reported risk concerns the increased development of cyanobacterial blooms. The manifestations already observed can be used to describe the main mechanisms (high temperature $>25^\circ$C, change in stoichiometric relations, dominance over other algal forms, less grazing due to low edibility and toxicity). Paragraph 3.4.3 is devoted to this topic.

Figures 3.11 and 3.12 synthesize work focused mainly on lakes. The same mechanisms are involved in the marine environment, except that other physical factors such as wind mixing and swell should be considered (also the sea level (not shown), which could also influence coastal lagoons and their submersion rate) as well as the role of the acidification of the oceans. Other factors to be taken into account are the negative role of the “shadow” cast by excessively produced algae on fixed benthic vegetation and corals and, lastly, the contribution to the greenhouse effect of the products ($\text{CO}_2$, $\text{CH}_4$ and $\text{N}_2\text{O}$) of benthic physico-chemical reactions in the event of prolonged hypoxia (this is also a possible effect in lakes).
To sum up, in the future, global changes could increase the risk of eutrophication and exacerbate some of the most harmful symptoms, notably the development of cyanobacteria. The message conveyed by the literature is clear: changes in the forms and proportions of nutrients will alter biodiversity; the problem of harmful algal blooms will become more acute, with more blooms, more toxins, more often and in more places. We still need to acquire a sharper understanding of the phenomenon, but we must shift from the paradigm of a simple relation between nutrient concentrations, chlorophyll a and algal biomass, to a new paradigm taking into account a greater complexity of relative nutrient forms, their proportions and the different levels of organization of living beings, from physiology to ecology.

To what extent could we predict them? Mechanistic models of the entire process chain seem out of reach for now, or at the very least tainted with major uncertainties, ranging from the various global change scenarios, their downscaling and breakdown into regional changes in temperature and rainfall, to the transformation of rainfall into runoff modulated by the contribution of aquifers, changes in terrestrial nutrient transfer, with the possibility of changes in land use, in residence time, freezing and stratification times and, lastly, changes in species distribution and trophic rearrangements of communities.

It is nevertheless possible, as proposed in the scientific literature, to build simplified scenarios. For instance, one can consider growing disparities of temperature and flow rates, annual or seasonal (e.g. +/-5%, +/-20% +/-30%) and in nutrient concentrations, and model their consequences in relation to a targeted risk of increased cyanobacterial development, changes in fish communities, etc. (Examples of large-scale projects: estimation and mapping of the risk of cyanobacterial development for all the lakes in the United States monitored as part of the NLA (National Lake Inventory); temperature variation projections for 2,500 lakes in Wisconsin and modelling of trends relating to their productivity and to the gradual replacement of the American walleye (Sander vitreus), a temperate water species, by the largemouth bass (Micropterus salmoides), a hot water species, both of great importance for the fishing industry).

Possible actions? All the scientific literature agrees on the need to start by reducing the nutrient loads (N and P), applying stricter standards and bearing in mind that the risks have increased. We also note that some remediation actions have lost their effectiveness (e.g. in the Lake Erie basin), negatively affected by global changes. Other recommendations obviously include the widespread creation of buffer zones, the defragmentation of portions of ecosystems and the reduction of residence times. From this point of view, the rapid increase in the number of reservoirs along the total length of waterbodies risks making water temperature rise as a function of residence times and the surface of water mirror. There is also a risk of development of algal blooms, particularly cyanobacte-
ria, that needs to be assessed. Lastly, given the uncertainties reported and the future values of temperature and flow parameters, at levels that ecosystems have never experienced, it is difficult to make accurate predictions. Implementing and consolidating appropriate monitoring is therefore essential.

3.4.2. Role of invasive species in the trajectories observed

We limit the question to invasive species that can play a role in the eutrophication process by changing the system's biogeochemical cycles or by acting on one of its biotic compartments.

For continental waters, invasive filter-feeding molluscs fall into this category. They have a great capacity for dispersal, spread quickly, and breed in large quantities. They can cover large areas of substrate and reach significant biomass densities. Several cases have been reported in lakes (Lake Constance, Lake Maggiore, the Laurentian Great Lakes, especially Lake Huron and Lake Michigan), as well as in great slow-flowing rivers (Huron, Potomac). Incriminated species notably include Dreissena, like the quagga mussel (*Dreissena rostriformis bugensis*) in the United States, and corbicula like *Corbicula fluminea* in Europe, which has propagated to several French rivers in the western part of the country, including the Loire, since the end of the 1980s.

Invasive species like filter-feeding molluscs can have a strong influence on phytoplankton in the environments where they grow. Filtered volumes can be quite impressive: for instance, the zebra mussel can process 10% to 100% of the daily flow of a river like the Hudson (600 m$^3$/s in New York), while corbicula can filter five times the annual flow of the main tributary of Lake Constance, i.e. total filtration of the entire volume of water above the thermocline in 64 days, which corresponds to 2,000 times the filtration capacity of the Daphnia already present. Scientific literature offers a few explanations and functional assessments, when there are sufficient series covering invasion periods. In Lakes Michigan and Huron, an extension of the quagga mussel population was observed from 2004, with significant effects: 35% reduction in phytoplankton from 2007, while at the same time total P also reduced.

Phytoplankton's filtration potential is generally at its highest in the ascending phase of colonization. It subsequently decreases, either because hydrological conditions limit populations (effect of floods on adults in streams and rivers), or because predatory pressure increases and the food web readjusts. Taking Lakes Michigan and Erie, the total biomass of salmonids remained at a high level throughout the 2000s. This suggests that compensation mechanisms operate within complex food webs. One of the hypotheses is a greater development of the benthic algal biomass fixed in shallow belts which get more light. In similar areas on Lake Erie, a development and diversification of benthic macroinvertebrates on hard substrates has been observed as a result of the new food supply provided by mussel excreta and by the growth of a benthic biomass exploited by species capable of feeding on it. The indigenous amphipod *Diporeia sp.* followed the opposite trend to that of the quagga mussel: its numbers plummeted, reflecting the transformations under way in the food web.

However, the precise role played by these invasive species on controlling phytoplankton, the concomitant increase in water transparency and the trophic impoverishment of the system, which could lead to a reduction in fish biomass, are still not well established and not sufficiently documented in France. Furthermore, these findings need to be put into perspective with the downward trend in phosphorus, sometimes with the decline in average summer flow rates for a river like the Loire, or rising water temperatures. This is a research field in its own right, requiring common relevant time series between forcing variables (climate, hydrology, hydrodynamics, temperature, chemistry) and ecological response variables.

In the marine environment, algal species related to marine eutrophication tend to be indigenous, apart for some toxic species of phytoplankton (*Alexandrium*) brought by ballast water or imported bivalves. Marine invasive species are not directly related to eutrophication (neither a cause nor a consequence). Suspension feeders (oysters, slipper limpets, etc.) can modulate the phytoplankton biomass, but in a fluid ecosystem open to the ocean, their regulatory capacity is always exceeded by the dispersive or confining capacity of local hydrodynamics.
3.4.3. Focus on cyanobacteria

In a context of global change, the scientific community is facing a number of new issues, as are the administrators in charge of managing the problems generated by cyanobacteria. The first of these issues concerns the effects of climate change on toxic microalgae and cyanobacteria blooms. This question started to be addressed in 2008, giving rise to many studies that helped modulate and enrich early findings predicting a widespread increase in cyanobacteria blooms under the effect of global warming. In particular, it appeared that interactions between climate change and eutrophication did not necessarily lead to increased cyanobacteria blooms, the result depending on the trophic status of ecosystems and of the cyanobacteria species considered. Moreover, over and above the direct effect of the temperature increase on the growth rate of cyanobacteria, several studies have shown that climate change appears to play a significant role in encouraging the proliferation of photosynthetic microorganisms through the effects of extreme weather events (strong rain, hurricanes, etc.) on nutrient contributions in aquatic ecosystems.

Like for the debate on cyanobacteria, or when referring to the example of macroinvertebrates in the Loire, it still appears difficult to easily predict systems’ future biological behaviour: firstly, because broad climate projections are tainted by many uncertainties, one of them being that the signal is built on a very coarse spatial grain in relation to the scale of ecological processes; secondly, because changes will concern not only climate forcing factors, but also the adaptation of landscapes and human practices within these landscapes; lastly, because these forcings will alter the intensities of complex functional relationships.

That said, various projected changes suggest the probability of greater development of cyanobacteria and dinoflagellates at the expense of diatoms and chlorophytes:

- firstly, warmer temperatures will favour cyanobacteria at the expense of diatoms and chlorophyceae, from 25°C and above,
- temperatures should be considered not only in terms of value, but also in terms of duration and seasonality, with new thermal regimes likely to considerably increase windows of opportunity for some species (e.g. *Alexandrium*),
- not only will changes in nutrient transfers modify N:P ratios, but the new physico-chemical conditions in aquatic environments will cause new proportions in the forms of N, notably a rise in the NH₄⁺:NO₃⁻ ratio in situations of O₂ deficiency (§ 3.4.1). When N:P increases and NH₄⁺:NO₃⁻ also increases, these conditions favour cyanobacteria and dinoflagellates,
- in reservoirs in the American Midwest, it has been shown that these new conditions also increase the production of secondary metabolites, notably microcystin-type toxins.

All of the potentially negative signals related to global changes indicating a possible increase in the prevalence of cyanobacteria blooms show the importance of being vigilant about this issue and reflecting on what needs to be done in terms of designing or strengthening monitoring and warning mechanisms.

3.5. Socio-political developments and public perception

3.5.1. Periodization of public action

Social sciences draw mainly on three areas to analyze socio-political developments: the Great North American Lakes on the US-Canada border, the Chesapeake Bay in Maryland and Virginia on the East Coast of the United States, and the Baltic Sea and some Baltic countries, in particular Sweden and Finland. European definitions of eutrophication tend to place more emphasis than North American studies on the sources and business sectors that account for the increase in nutrients. For example, in Scandinavia, research programmes have integrated human and social
sciences more systematically than elsewhere. In other European countries, and specifically in France, work dates mainly from the adoption of the WFD, which constituted a tipping point in terms of the use of the concept and of the highly inclusive approach taken. Notwithstanding marked differences in context, policies to combat eutrophication all seem to have developed in a fairly generic manner and which can be divided into three main periods.

**The first period**, initiated at the end of the 19th century, is characterized by the response to pollution in urban areas, with the development of collective sanitation infrastructure and the ban on direct discharges to the aquatic environment. This was primarily motivated by a health concern, the objective being to reduce nutrient and bacterial pollution in a context of high population density. In all the cases studied, pollution management policies achieved significant results, although this does not mean there were not lasting effects on the environment. In the first countries to be industrialized, pollution management gradually lost political and social visibility, although it translated into significant public investment. This phase is still ongoing in a large number of developing countries.

**The second period**, in the cases studied, was between the 1970s and 1990s: diagnoses became more precise, enabling a differentiated approach for each cause. Legislative and regulatory frameworks started to be implemented but initiatives remained predominantly local, focusing on managing the small water cycle. Most policies concentrated on point-source pollution. They involved direct actions on the part of public authorities (collective sanitation improvement), ad hoc regulations, or negotiations with easily identifiable industrial operators. Here too, significant results were achieved, although battles were often hard-fought and at the cost of very expensive programmes for public finances. This mode of action is still dominant in many countries, particularly in South-East Asia and China.

**The third period** dates back to the 1990s, in Europe and in the United States, and concerns non-point pollution affecting the large water cycle. Several factors explain this transition: the relative decline in point-source inputs, improved scientific knowledge highlighting inputs from slopes or the role played by atmospheric deposition in eutrophication, and the difficulty of obtaining results in relation to agricultural non-point pollution. This was a period of tighter regulations with respect to both emissions and the quality of receiving environments. In this context, most approaches were based on the voluntary engagement of stakeholders in local programmes for pollution control, including sedimentary inputs in aquatic environments.

This broad periodization deserves to be narrowed down based on local factors. Environmental factors seem to have a fairly limited effect on the type of policies implemented. Public action was structured by the identification of the different control factors (phosphorus in the Great Lakes, nitrates in the European Union). The instruments rolled out often had simultaneous effects on both sources, however. Moreover, in a number of countries, public policies have recently shifted towards a greater level of integration, under which nitrogen and phosphorus are considered as a combined whole.

### 3.5.2. Triggers

An analysis of the triggers of these public policies shows that it is often a combination of several dynamics that leads to putting eutrophication issues on the agenda, mirroring the conventional analysis of the social construction of public environmental issues: knowledge, temporal alignment, random elements, dramatic events and participation at grass-roots level of whistleblowers and people willing to take a stand for the cause (Fig. 3.13).

Most authors consider that research has played a key role in putting eutrophication issues on the agenda. More broadly, the organization of relations between science and politics formed the core of eutrophication governance mechanisms. Research institutions and researchers helped build frameworks for the consideration of these issues, ranging from targeted research and model designing to whistleblowing and the production of critical discourse.

Some authors place greater emphasis on the role of social movements, putting pressure on the authorities to get them to change their attitude towards powerful economic agents which may have an interest in preserving the status quo. Here again, a shining example is the regulation on phosphate levels in detergents. In France, local environmentalists were instrumental in getting the government to step up its efforts to combat coastal eutrophication, as underscored by the work on handling green tides in Brittany.
The influence of the government system and of the pollution regulation regime is often cited. A comparative study of the environmental history and socio-spatial characteristics of the Chesapeake Bay and Scotland underlines profound differences in pollution management between the old world and the new, as a result of very distinct temporalities and forms of artificialization of aquatic ecosystems. This means that administrators have to take different approaches to deal with the legacy of past pollution, which plays a big part in eutrophication. In the most recent period, however, several authors have highlighted the importance of disseminating frameworks and public policy initiatives from one region to another. Furthermore, on sides of the Atlantic, federal or community coordination has proved decisive in structuring public policies against eutrophication.

Figure 3.13. Periodization of the fight against eutrophication and theoretical responses of an ecosystem. Many trajectories are possible (linear or abrupt, with or without hysteresis, with or without change in status), depending on the history of the aquatic ecosystem, its physical and biological characteristics, as well as those of its watershed.

3.5.3. From changes in public perception to public policies

In industrialized countries, the perception of water quality is marked by a dichotomy between water consumed and water as an environment, but this dichotomy should not be seen as static. However, it does reflect a historical process in which aquatic environments have become gradually removed or even totally absent from the everyday life of the majority of the population. The immediate social visibility of nutrient pollution is generally limited. Visibility can increase, however, in certain circumstances related to eutrophication phenomena: visible algal blooms, for instance green tides, severe forms of hypoxia, manifested by changes in water's appearance or smell, in the type of dominant vegetation or the death of aquatic organisms, or swimming bans, water supply cuts, bans on the consumption of aquatic products, etc. There has been little work to analyse lay observers' perceptions and impressions when faced with this type of event. That said, some studies show that observers remember them as benchmark events for the assessment of the quality of water bodies. Environmental action, public action and media production contribute to calling attention to eutrophication and shaping the way in which the phenomenon is perceived, understood, recognized as a problem, and debated by all stakeholders in society. In other words, these actions help eutrophication gain a “secondary” social visibility, by giving it more prominence in public discourse aimed at raising vigilance and alerting to these issues. Media interest in eutrophication phenomena therefore intensified in some regions when events such as
visible, spectacular algal bloom made headline news. But in the case of eutrophication, events are generally shaped by the mobilization of stakeholders, giving journalists useful insights into understanding the issue. When eutrophication phenomena become the subjects of social and political action, the media agenda is also governed by the strengthening of monitoring and measurement systems and public policies rolled out over time. Initially a topic for experts, now attracting interest from a wider audience due to the escalation of environmental concerns, eutrophication has also become an important area of public action. It is identified as both a problem to be solved and a sign of the health of aquatic environments, giving rise to monitoring by increasingly finely tuned indicators.

3.5.4. Tensions and conflicts around eutrophication

In the same way as other environmental issues, eutrophication phenomena, in particular dystrophic crises expose and crystallize social tensions. The conflicts that surround the management of water quality and aquatic community health issues are just as much related to the harmful consequences of environmental degradation as to their institutional handling. These conflicts are key moments in the stakeholder coordination process. The intensity of conflicts therefore hinges less on the severity of eutrophication effects than on the social visibility of the phenomenon and on stakeholders’ level of structuring, whether they are institutions, professional groups whose responsibility is engaged or whose business is threatened, or the civil society. The impact of these conflicts on the effectiveness of eutrophication management, which has been discussed in the literature, is probably variable. In the abundant literature dedicated to the analysis of water conflicts, conflicts related to eutrophication remain very little studied. In this context, the case of green tides in France stands out as an exception: the work dedicated to this phenomenon shows that the coordination process between the agricultural world, local elected representatives and environmental protection associations is conflictual, fragile, and often breaks up. Since the 1970s, conflicts between and within these different groups have weighed on the management of the problem. These conflicts are complex: they confront not only economic interests, but also widely varying representations of the environment, conceptions of public action and social responsibility, and scientific knowledge.

The traditional interpretation of environmental conflicts, i.e. conflicts of use related to the issue of ownership of a scarce resource, does not account for their complexity and their contemporary dynamics. Environmental problems such as eutrophication have a certain degree of historical depth, are the subject of structured public policies, and are related to environments with a social and cultural value, in societies where scientific knowledge is accessible to a wide audience. Furthermore, dependence on a single water resource is at least as much of a factor in strengthening a sense of solidarity and negotiating capacity as in fomenting lasting antagonism. In this context, actors in social and political conflicts can use environmental issues as an argument and means of pressure. Environmental issues can also constitute an invisible infrastructure in which social, economic or political inequality is reproduced or accentuated, without resulting in explicit conflict. This is the case, for example, when the degradation of water quality mainly affects low-resource populations, forcing them to use other, more expensive forms of water supply or limiting their access to environmental amenities, their only affordable form of recreation.

In order to analyze these tensions, approaches favouring a strategic reading and those focusing on analyzing conflicting representations of the environment and the risks attached to its degradation turn out to be complementary. In the literature, however, they are often addressed on a mutually exclusive basis. Moreover, the aggravation of eutrophication phenomena being interwoven with growth dynamics, far more in-depth research would be required to put these conflicts back in the context of broader trajectories of structural changes in socio-ecosystems, for example by building on the concept of hydro-social cycle, as developed by the geographer E. Swyngedouw. This concept aims to account for the way in which water is socially and politically constructed, both in material terms and in terms of representation. It serves as a fulcrum for several authors of the corpus selected to report on the transformations affecting aquatic ecosystems in strongly anthropized environments. Some works show that in some parts of Southern Europe, the fear of drought results in water infrastructure management which, on the one hand, makes aquatic ecosystems more vulnerable to eutrophication, and on
the other hand rules out preventive interventions in practice. Pressure on the resource, related for example to the development of irrigated agriculture or tourism infrastructure, leads to an essentially quantitative water management and a de facto limitation of the consideration given to eutrophication issues. For example, studies conducted in Sicily showed that the failure of policies to tackle eutrophication was strongly correlated with a political framework under which water is primarily considered as a quantitative issue, to be stored in artificial reservoirs so as to ensure continuity of supply.

The complexity of the relationship between quantitative and qualitative water management is emphasized in many case studies. A case in point is France, in the watershed of the Charente, in a context of increased water withdrawals for agricultural purposes: while water quality appears to be the dominant issue for many players downstream of the watershed (shellfish farmers and tourism operators, for instance), negotiations between stakeholders are more likely to succeed when they concern decisions on the purchase of equipment or an effective allocation of the resource (for example, in terms of volumes withdrawn by user category).

The eutrophication of aquatic environments remains little studied from this perspective, however, despite the very close relations between growth dynamics, water management methods and accentuation of dystrophic crises. This close relationship can be studied on different scales, from local water management to major socio-economic trends observable on a global scale: improving knowledge of these processes would make it possible to better understand often complex conflict dynamics and put them in a longer perspective.

In Western Europe, public policies against eutrophication and the conflictual nature of social debate on the phenomenon are currently focusing on agricultural activities. Conflicts related to pollution of agricultural origin take various forms, but they are all marked by the same tension between available methods of coordination and the very weak levers for negotiating at local level. This weakness can be analyzed at several levels. Generally speaking, the coherence between agricultural policy and water policy is a determining factor in the effective development of production systems. However, while water policies are historically well-structured and applied at local level, notably due to the importance of the watershed scale, agricultural policies are designed and negotiated at other scales. At institutional level, local means and instruments of action on agriculture are limited and come within the scope of highly complex public action mechanisms, generic in nature, which local agents don’t always employ. In this context, specific governance efforts implemented in regions facing eutrophication issues can produce, in the short term, tensions and frustrations: they confront administrators, representatives of civil society and agricultural stakeholders with the structural limits of their own action. When these approaches are applied over the long term, however, they can foster a collective learning process and place agricultural issues back into a regional context. These positive effects can be seen in the management of coastal eutrophication in Brittany: since 2010, management of the green tide issue has been based on institutional recognition at different scales, leaving considerable room for the construction, by stakeholders in the bays affected, of territorial low-nutrient leakage projects, highly inclusive and differentiated from one bay to another. The second generation of local projects shows greater consideration given to the economic conditions of the transition to systems using fewer inputs and better integrated in the territory's economy, i.e. more sustainable and more resilient. These dynamics of change remain fragile, however, as they are highly dependent on the allocation of public funds and the volatility of agricultural prices, which work on different time scales and are marked by considerable uncertainty. They also depend on the quality of interactions between stakeholders, as denunciating the impacts of eutrophication now forms an integral part of a more comprehensive debate on agricultural activity and the mobilization of all those involved in managing the social and environmental consequences of farming.

Diffuse pollution from agriculture, which is currently one of the biggest causes of pollution in the European Union, involves players from different or even completely separate social spaces. This is especially the case for eutrophication, which, being by nature multifactorial and non-linear, raises the need for reflection and debate, not merely on an activity and its consequences, but more generally on the impact of human activity and the workings of nature. There can be no effective collective management without addressing the question of the many ways of apprehending the knowledge on which remedial action depends.
Countries first started to grow aware of eutrophication phenomena back in the 1970s. It then took a relatively short time to set up a legal framework to tackle the issue. Rules of international law, community law and national law were rapidly adopted. The implementation of some of these water management and protection policies is based largely on information from monitoring networks used to determine changes in environmental status under the impact of various influences (in particular climate change) and to measure the effects of management actions. These initiatives are very much on a “learn and adapt as you go” basis. In parallel, scientific research is based on observatories where investigation is more thorough and state-of-the-art instruments are tested over a long-term period. Long-Term Observation and Experimentation Systems for Environmental Research (SOERE) in France, Long Term Ecological Research (LTER) and Critical Zone Observatories (CZO) in the United States, and the global “Continuous Plankton Recorder” programme meet these objectives.

In this chapter, we describe developments in regulatory frameworks for water protection, then the observation networks related to this operational sphere, and we “question” their ability to respond to the issue of eutrophication. We supplement this analysis with a review of the biological groups that are or could be useful indicators of the eutrophication process. We then provide some examples of long-term networks. Finally, we look at the new technologies emerging in the areas of sensors and imaging, from which monitoring is set to benefit sooner or later, as well as fledgling citizen observation initiatives which, whether spontaneous or supported by scientists, will want a place in future debates between science and society.
Table 4.1. International and community legal instruments to combat nutrient discharges in the water and atmosphere.

<table>
<thead>
<tr>
<th>Date</th>
<th>Act/Protocol/Convention/Commission</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>22 March 1974</td>
<td>Baltic Marine Environment Protection Commission - Helsinki Commission (HELCOM)</td>
<td>First international text dealing with marine pollution from land-based sources. Since 1992, the convention has defined ‘risks of eutrophication from land-based sources’ as a criterion to identify, assess and categorize harmful substances (i.e. of which the introduction in the sea is likely to cause pollution).</td>
</tr>
<tr>
<td>4 June 1974</td>
<td>Paris Convention for the prevention of pollution from land-based sources in the North-East Atlantic</td>
<td>Sets a list of the substances to be limited/eliminated and establishes a commission tasked with examining the status of the seas and making recommendations. Combating eutrophication in coastal waters became a major concern for the Paris Commission as early as the 1980s.</td>
</tr>
<tr>
<td>13 Nov. 1979</td>
<td>Geneva Convention on Long-Range Transboundary Air Pollution (LRTAP)</td>
<td>Aims to reduce air pollution. Gothenburg Protocol of 30 Nov. 1999 to abate acidification, eutrophication and ground-level ozone: sets emission ceilings for various air pollutants (sulphur, nitrogen oxides, ammonia and VOCs) responsible for acidification and eutrophication.</td>
</tr>
<tr>
<td>21 May 1991</td>
<td>Directive 91/271/EEC concerning urban waste water directive (UWWT)</td>
<td>Concerns the collection, treatment and release of urban wastewater, as well as the treatment and release of wastewater from certain industrial sectors. This directive is intended to protect the environment from deterioration due to wastewater discharges. It requires States to identify sensitive areas requiring more advanced treatment.</td>
</tr>
<tr>
<td>12 Dec. 1991</td>
<td>Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources (Nitrates directive)</td>
<td>Aims to reduce water pollution caused or induced by nitrates from agricultural sources and to prevent pollution of this type. Requires States to define vulnerable areas and to define action programmes covering designated vulnerable zones.</td>
</tr>
<tr>
<td>22 Sep. 1992</td>
<td>OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic</td>
<td>A unified mechanism developed from the Paris Convention of 1974 and the Oslo Convention of 1972 for the prevention of marine pollution by dumping from ships and aircraft. The OSPAR Convention adopted a comprehensive approach for the protection of the marine environment and created the OSPAR Commission, tasked with drawing up action plans. It was responsible for the implementation of the OSPAR Eutrophication Strategy aiming to combat eutrophication based on a Common Procedure for the identification of the eutrophication status of the maritime area.</td>
</tr>
<tr>
<td>23 Oct. 2000</td>
<td>Water Framework Directive 2000/60/EC (WFD)</td>
<td>Establishes a community framework to achieve a good ecological status of surface waters and groundwater in all watersheds. Concerning eutrophication, bases itself on biological elements and physico-chemical parameters (dissolved oxygen concentration, temperature, salinity, turbidity, nutrient concentration, concentration in chlorophyll a, total nitrogen, total phosphorus, carbon, zooplankton, etc.).</td>
</tr>
<tr>
<td>23 Oct. 2001</td>
<td>Directive 2001/81/EC setting national emission ceilings for certain atmospheric pollutants (NEC)</td>
<td>Sets limits for total national emissions of four pollutants (sulphur dioxide, nitrogen oxides, VOCs, and ammonia), sources of acidification and eutrophication.</td>
</tr>
<tr>
<td>17 June 2008</td>
<td>Marine Strategy Framework Directive 2008/56/EC (MSFD)</td>
<td>Establishes a framework for community action enabling States to “take the necessary measures to achieve or maintain good ecological status in the marine environment by the year 2020 at the latest”. Establishes Descriptor 5 on eutrophication to define good ecological status.</td>
</tr>
</tbody>
</table>
4.1. Changes in the regulatory framework and associated monitoring

4.1.1. Changes in regulatory frameworks

The legal framework of eutrophication consists of international, European and national rules. International treaties concern themselves more particularly with eutrophication in seas and oceans and eutrophication related to air emissions. Some of these treaties set specific eutrophication criteria and provide for the implementation of action programmes. The rules of community law also apply to marine eutrophication and its proportion related to air emissions, as well as to freshwater eutrophication in the territory of the European Union. National laws contain a wealth of provisions, which have changed significantly under the influence of community law. The legal framework for eutrophication, initially introduced in the 1960s-1970s, is evolving with scientific knowledge and is gradually becoming stricter (table 4.1).

The large number of international and community provisions makes them fairly difficult to articulate. While all of these mechanisms aim to reduce nutrient concentrations, each one imposes its own method (identification of zones, action programmes, etc.) and results (threshold values, good ecological status, etc.). Each mechanism provides more or less leeway, which has prompted States to exempt themselves from their obligations on the grounds of an often fairly restrictive conception of eutrophication and related measures (leading to convictions by the Court of Justice).

It is important to keep in mind the rationale that underpinned the definition of standards: for instance, the Nitrates directive, which focuses on nitrates from agricultural sources, first set a threshold for nitrates against drinking water standards rather than ecosystem sensitivity standards, which are generally far lower, while aiming for environmental objectives to combat eutrophication. In support of this distinction between potability and eutrophication risk, a recent synthesis shows that national drinking water standards for nitrates range between 5.6 mg/L NO$_3$-N (Spain, Switzerland) and 11.3 mg/L NO$_3$-N (rest of Europe), while sensitivity thresholds for surface hydrosystems are lower: United States (0.1 to 2.1 NO$_3$-N), Germany (2.5 NO$_3$-N), Canada (3 NO$_3$-N) (table 4.2). The standard for drinking water in France, which is not shown in this table by Liu et al. 2017, is 50 mg/L NO$_3$-N, i.e. 11.3 mg/L NO$_3$-N, which also matches a WHO recommendation.

<table>
<thead>
<tr>
<th>Nitrogen (mg N/L)</th>
<th>Water quality standards for ecosystems</th>
<th>Total phosphorus (mg P/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Drinking water standards</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Canada</td>
<td>10 (NO$_3$-N)</td>
<td>3 (NO$_3$-N)</td>
</tr>
<tr>
<td>China</td>
<td>10 (NO$_3$-N)</td>
<td>1 (NO$_3$-N)</td>
</tr>
<tr>
<td>European Union</td>
<td>11.3 (NO$_3$-N)</td>
<td>5.6 (NO$_3$-N)</td>
</tr>
<tr>
<td>Germany</td>
<td>11.3 (NO$_3$-N)</td>
<td>3 (TN), 2.5 (NO$_3$-N)</td>
</tr>
<tr>
<td>Switzerland</td>
<td>5.6 (NO$_3$-N)</td>
<td>7 (TN), 5.6 (NO$_3$-N)</td>
</tr>
<tr>
<td>Netherlands</td>
<td>11.3 (NO$_3$-N)</td>
<td>0.12-1.8* (TN)</td>
</tr>
<tr>
<td>United States</td>
<td>10 (NO$_3$-N)</td>
<td>0.1-1.27 (TN, lakes), 0.12-2.17 (TN, rivers)</td>
</tr>
</tbody>
</table>

Table 4.2. Comparison of global quality standards for nitrogen and phosphorus. Note: Distinction between drinking water standards and water quality standards for ecosystems. TN: total nitrogen; * the value range is important because it includes the specific objectives for more than 500 water bodies. Source: Liu et al. 2017.
Currently, at national level, the legal arsenal against eutrophication consists mainly of provisions that protect the environment, by imposing the demarcation of territories within which environmental management is recommended, and provisions governing the activities likely to cause spills that could be a source of eutrophication.

The provisions that protect the environment against eutrophication consist mainly in two intervention techniques: zoning and planning. Zoning with regard to water resources makes it possible to protect zones already affected by pollution or with strategic water resources. Some zoning specifically concerns the fight against eutrophication: zones sensitive to eutrophication (as per the UWWD), zones vulnerable to nitrate pollution (as per the Nitrates directive), or watersheds experiencing severe green tides on beaches (art. L 211-3, 8°, French Environmental Code). Planning concerns water management and the protection of air quality. In France, master plans for water planning and management (SDAGE) and plans for water planning and management (SAGE), as well as action plans for the marine environment (PAMM), seek to achieve “good environmental status” in water bodies. Air planning tools are intended to combat air pollution.

Within these provisions aimed at protecting the environment against eutrophication, the concept of eutrophication is explicitly mentioned in the UWWD and Nitrates directives (which provide a definition of eutrophication) and in the MSFD (notably Descriptor 5 on eutrophication, used to define good environmental status), but more implicitly in the WFD (which only refers to the phenomenon in its indicative list of major pollutants in annex VIII). Limit levels for nutrient concentration are considered, either directly by setting threshold values, or indirectly by means of the good ecological status (table 4.3).

<table>
<thead>
<tr>
<th>Directive</th>
<th>General objective</th>
<th>Objective in terms of eutrophication: reduction in nutrient concentrations (threshold values where available)</th>
</tr>
</thead>
</table>
| UWWD      | Protect the environment from the negative impacts of urban wastewater | - Maximum average phosphorus concentration 2 mg/l (PE between 10,000 and 100,000) and 1 mg/L (PE over 100,000)  
- Maximum average phosphorus concentration 15 mg/l (PE between 10,000 and 100,000) and 10 mg/L (PE over 100,000) |
| Nitrates  | Protect the aquatic environment against pollution caused by nitrates from agricultural sources | - Nitrate concentration threshold: 50 mg/L in surface water and groundwater  
- Trophic status of water bodies (no quantified indicator) |
| WFD       | Achieving good ecological status in water bodies | Among many other parameters, annex V of the WFD cites nutrient concentration. The WFD does not set threshold values for nutrient concentrations, leaving it to the States to define such thresholds. It refers to other directives such as the Drinking Water Directive 98/83/EC: 50 mg/L of nitrates, 0.5 mg/L of nitrites  
It should be noted that subsequent sector-specific application directives set threshold values, e.g. the Freshwater Fish Directive for phosphates, nitrates and ammonia |
| MSFD      | Achieving good ecological status in marine waters | The MSFD does not set threshold values for nutrient concentration, but provides qualitative descriptors to define good environmental status (in particular Descriptor 5 on eutrophication) |

Table 4.3. Targets for reduction in nutrient concentrations per directive.
PE: population equivalent
The provisions framing activities that may cause spills in the environment and be a potential source of eutrophication are governed by several rules: some concern facilities, others products. Rules on facilities concern so-called “classified installations for environmental protection” and facilities that have an impact on the water and aquatic environments. Product-related policies were developed to regulate the marketing of products such as phosphate-based detergents, fertilizers, etc.

Eutrophication and the problems it causes fall within a fairly extensive national, community and international legal context, necessarily evolving and likely to become increasingly less sector-specific and more integrated (water/air, freshwater/sea, etc.). It is proving difficult, however, to put in place a coherent framework incorporating nitrogen and phosphorus. Furthermore, at territorial level, local players find it difficult to navigate the multitude of mechanisms. European guidelines and conventions sometimes apply in concert to a given territory. As a result, some countries, like France, have to manage up to four guidelines/conventions in their coastal waters (figure 4.1). In its 2010 report, the French Council of State said that water law was still akin to a “baroque construction, born of the sedimentation of disparate laws dealing separately with categories of water [...] or their respective uses, seeking to address the concerns of the moment or pursuing distinct objectives”.

With regard to eutrophication, there have been efforts to improve the interconnection between the various legal instruments concerned (government decree of 17/02/2014 on the interconnection between the WFD and MSFD, Ifremer’s work on this subject, etc.).
4.1.2. Management by thresholds or annual quantities not to be exceeded

A number of authors have analyzed biological data from monitoring networks to identify tipping points based on nutrient concentrations. This literature is difficult to analyze, and it is fairly complex to do so without taking into account the geographical contexts and contrasts in the data sets. Tipping points will initially depend on hydro-geological situations, the nature of soil in watersheds, the legacy of the past and nutrients stored in soils, groundwater, waterways and their geomorphology and, lastly, on the relation from watershed to estuaries and coastal areas and whether the water flow is free or contained.

Patently, given the spatial dependence between the various environments (land-sea continuum, land-wetlands, lakes, reservoirs, lagoons, groundwater-aboveground water bodies), a clear understanding is needed of these different spatial components, the flows that irrigate them and the relative nutrient concentrations, as well as the related environmental vulnerability thresholds, in order to define the levels of eutrophication risk.

This complexity is manageable, notably by using different models, often in combination, which offers a double advantage: (1) educational, by clarifying assumptions and providing a spatial illustration of results (scenarioic or linear mapping, animations), (2) operational, by circulating and comparing scenarios. A simplified translation of this analytical complexity can then be transformed into quantitative nutrient objectives not to be exceeded annually (e.g. the Mississippi basin, on which depends the anoxic zone of the Gulf of Mexico, the Great Laurentian Lakes, the Chesapeake Bay, the OSPAR convention) or into differentiated concentrations (e.g. modelling of the Seine or of the watersheds of the bays of Brittany).

The bio-physical perspective, however, is not the only one to be taken into consideration when drawing up standards which are not independent from the socio-political context, the cost of implementing remediation actions and the transformations that are feasible technically and humanly.

Regarding the standard of 50 mg/L of NO$_3^-$, it is clearly relative to water potability rather than to preserving environments from the eutrophication process. Situations of 1 to 3 mg/L are characteristic of zones with very low human pressure; some publications identify a tipping point at barely higher values, but here again, this is rather in the case of early changes in communities’ species composition. Transparency on assessment criteria and the related educational approach are essential to set threshold value ranges (cf. also paragraph on references). Before the implementation of the WFD, the French “inter water agencies” technical structure had produced a consensus grid for different environmental uses (water quality assessment system, or SEQ) and proposed various value guidelines in the range of 2 to 50 mg/L NO$_3^-$. It would be interesting to analyze the historical trajectory of these value guidelines and their territorial applications.

4.1.3. Regulatory monitoring networks

The adoption of the WFD in Europe, along the lines of the Clean Water Act (1972) in the United States, led to profound changes in water management policies, extending the assessment of the status of this resource to the monitoring of the biological communities that develop in it, and no longer merely to chemical measurements. The WFD applies to continental and coastal surface waters as well as groundwater. The directive also had a strong influence on the implementation of the MSFD. The MSFD currently advocates (work in progress) using the results deriving from the WFD at coastal level (< 1 nautical mile). Furthermore, in France, data from the WFD networks are reported at regional level in order to feed OSPAR and Barcelona assessments. This chapter will therefore be devoted mainly to “questioning” the WFD networks’ ability to respond to the issue of monitoring eutrophication.

Main features of general monitoring networks in France

For continental waters, a national watershed network (RNB, for réseau national de bassins) comprising around 1,700 observation points of the biological status of surface waters had been in place since the 1980s before being replaced by the WFD monitoring networks in 2007. Two significant changes were made, one to spatial cove-

1. It is common for a tipping point not to be found in a universe where all observable situations have a biology that has long become homogenized; not finding one, however, does not mean that these situations are not polluted and that nutrient values do not present a risk for areas further downstream.

2. This issue is not covered by the scope of this joint scientific appraisal.
rage, which needed to be more representative of all water bodies (basic spatial unit), without however reflecting their exact proportionality (headwaters are still under-represented), the other consisting in an increase in the parameters monitored, notably by systematically performing biological measurements to assess the ecological status. The monitoring system is now based on nearly 4,500 stations, all water body categories combined, on a total of roughly 11,500 water bodies (table 4.4), i.e. a total completeness level of 38%. Unmonitored water bodies are nevertheless assessed for the risk of not reaching environmental objectives (RNAOE, for risque de non atteinte des objectifs environnementaux) through a review of the pressures likely to have an impact on them and an assessment of their ecological status by modelling.

<table>
<thead>
<tr>
<th>Waterways</th>
<th>Mainland France</th>
<th>Overseas</th>
<th>France</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number</td>
<td>9,799</td>
<td>1,025</td>
<td>10,824</td>
</tr>
<tr>
<td>Number monitored</td>
<td>3,783</td>
<td>97</td>
<td>3,880</td>
</tr>
<tr>
<td>% monitored</td>
<td>39</td>
<td>9</td>
<td>36</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Water body</th>
<th>Mainland France</th>
<th>Overseas</th>
<th>France</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number</td>
<td>434</td>
<td>5</td>
<td>439</td>
</tr>
<tr>
<td>Number monitored</td>
<td>311</td>
<td>2</td>
<td>313</td>
</tr>
<tr>
<td>% monitored</td>
<td>72</td>
<td>40</td>
<td>71</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Transitional waters</th>
<th>Mainland France</th>
<th>Overseas</th>
<th>France</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number</td>
<td>84</td>
<td>12</td>
<td>96</td>
</tr>
<tr>
<td>Number monitored</td>
<td>77</td>
<td>9</td>
<td>86</td>
</tr>
<tr>
<td>% monitored</td>
<td>92</td>
<td>75</td>
<td>90</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Coastal waters</th>
<th>Mainland France</th>
<th>Overseas</th>
<th>France</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number</td>
<td>120</td>
<td>44</td>
<td>164</td>
</tr>
<tr>
<td>Number monitored</td>
<td>92</td>
<td>34</td>
<td>126</td>
</tr>
<tr>
<td>% monitored</td>
<td>77</td>
<td>77</td>
<td>77</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Total</th>
<th>Mainland France</th>
<th>Overseas</th>
<th>France</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number</td>
<td>10,437</td>
<td>1,086</td>
<td>11,523</td>
</tr>
<tr>
<td>Number monitored</td>
<td>4,263</td>
<td>142</td>
<td>4,405</td>
</tr>
<tr>
<td>% monitored</td>
<td>41</td>
<td>13</td>
<td>38</td>
</tr>
</tbody>
</table>

Table 4.4: Proportion of surface water bodies monitored compared with total water bodies in each category. Source: Eau France 2013.

These numbers are important to help keep in mind the orders of magnitude, the grain and the spatialization of data that can then be mobilized to analyze the eutrophication process, with their associated uncertainty and the pressure-status relations that can be defined.

For the marine coastal environment, WFD monitoring is based mainly on the Ifremer historical networks: the national network monitoring the quality of the marine environment (RNO, for réseau national d’observation), created in 1974, and the phytoplankton and phycotoxin monitoring network (REPHY, for réseau de surveillance du phytoplancton et des phycotoxines), created in 1984. The physico-chemical parameters (temperature, salinity, turbidity, nutrients, dissolved oxygen, chlorophyll a) initially measured by the RNO were integrated in the REPHY in 2008 when the WFD was implemented operationally, while the RNO was replaced by the chemical contamination monitoring network (ROCCH, for réseau d’observation de la contamination chimique). Monitoring of benthic macrophytes and invertebrates is carried out by the benthic population monitoring network (REBENT, for réseau de surveillance des peuplements benthiques), implemented in 2006 under the responsibility of Ifremer and involving a large number of players. The development of this network at national level was based on the REBENT pilot programme, initially implemented in Brittany in the early 2000s, as well as on the Posidonia sea grass monitoring network (RSP, for réseau de suivi des herbiers de posidonies), created in 1984, and on the lagoon monitoring network (RSL, for réseau de suivi lagunaire) created in 2000 for the Mediterranean coast.

Regarding the MSFD, a monitoring programme, part of the Action Plan for the Marine Environment (PAMM), has recently been proposed (in 2016) for each marine sub-region (Channel-North Sea, Celtic Seas, Bay of Biscay, western Mediterranean). This proposal was made in light of the results of the 2012 Initial Assessment and of the knowledge gained from observation and monitoring programmes, studies and research, and based on the opinion of local experts.
Monitoring systems focus on a global ecological status

The monitoring systems active in hydrosystems in Europe and North America are designed to collate simultaneous informations on the three hydrosystem compartments, namely chemistry, physics and biology. The primary objective is therefore to perform an overall assessment of the health status of environments, in accordance, respectively, with the European Water Framework Directive (WFD, 2000) and the Clean Water Act (CWA, 1972). Bio-indication, a concept born at the beginning of the last century, was initially designed to measure the effect of the major forms of pollution at the time, notably excess organic material discharged directly into the aquatic environment. It later followed developments in the 1980s to take into account a much broader range of environmental changes, using more inclusive multi-metric biological indicators, which were subsequently recommended in Annex 5 of the WFD. In the 2000-2010 decade, European researchers were largely involved in many programmes (Aquem STAR, Rebecca, Wiser, MARS for fresh water, North-East Atlantic and Mediterranean geographical inter-calibration groups working on WFD indicators on biomass, abundance and composition of marine phytoplankton, etc.), to improve and adapt existing methods, notably by converting them into multi-metric indicators and to inter-calibrate their results between the various member states.

The system, now operational, is therefore fairly complex insofar as it no longer merely detects exceedances of the physico-chemical thresholds defined to meet target uses, but also combines a number of biological indicators with different sensitivities and response times to pollution or pollution cocktails (e.g. fish, macroinvertebrates, macrophytes), using chemical and physico-chemical parameters to report on the ecological status of water bodies. Points of attention relate to working scales, the time lapses considered and the potential weight of pressures not well taken into account. Points for improvement concern improving the relations between pressures and biological responses and the bias that could be derived from biological characteristics, in particular features that could help diagnose situations.

This general system introduced by the new framework directives has improved the capacity to detect dysfunctions caused by multiple pressures. Eutrophication has become just one cause among others: in fact, the WFD only refers to the process in its highly technical Annex V, which defines the limits of status classes while emphasizing that the pre-existing directives on nitrates and urban wastewater still must be complied with. The issue of multiple stressors has become highly prevalent in the literature, particularly since 2010, and is currently one of the great scientific challenges in freshwater hydrobiology and oceanography, especially as recent climate changes (the last decade was the warmest on record) are changing the relations between forcing factors and hydrosystems’ ecological responses.

4.1.4. Reference in a regulatory framework

In the WFD like in the Clean Water Act in the United States, the benchmark relates to the best observable contemporary situations, which are therefore no longer historical situations, untouched by human presence, or “pristine”. The “good status” cursor, which is the WFD’s objective in terms of benchmark to be attained, was the subject of technical definitions, particularly in Annex 5 of the Directive, and of an inter-calibration of bioindication methods among member countries. European guides have been produced. The technical criteria were then transcribed into national decrees. This entire technical process, which is fairly complex, has resulted in a synthetic ranking of statuses using the now familiar range of qualifiers, “high, good, moderate, poor and bad”, associated with colour codes for mapping purposes. The benchmark statuses considered could change over time, particularly as a result of climate change. A general system incorporates this aspect insofar as a perennial reference network (RRP for réseau de référence pérenne) of around 300 monitored stations has been set up for this purpose in mainland France.

3. By way of illustration, the screening criteria were regionalized proxies such as the proportion of artificialized surfaces (< 0.8%) or the percentage of farmed land (< 25% intensive agriculture and < 30% non-intensive agriculture for the Nordic group; < 50% intensive agriculture for the Central Baltic group) as well as chemical criteria (O2 = 95-105%; N-NH4 = 0.10 mg/L; N-NO3 = 6.00 mg/L; P-PO4 = 0.040 mg/L), the average values for the Central Baltic Group.
4.2. Other networks more specifically dedicated to monitoring eutrophication

It is not possible to enter here into the detail of the many networks existing at international level or even to seek to perform a comprehensive inventory, but it is important to mention their existence as a large number of publications are based on the data they collect. In the United States, a specific research strategy addressed this issue, supplemented by various monitoring networks covering crisis areas (Chesapeake Bay programme, LUMCOM for the Gulf of Mexico, etc.), in areas long subject to often acute manifestations of eutrophication. In Europe, the United Kingdom has set up a eutrophication network to identify sensitive areas and monitor actions (ECAPs: Eutrophication Control Action Plans).

4.2.1. Networks dedicated to eutrophication in the United States

Nutrient pollution was identified as one of the major causes of water degradation in the 1970s, with landmark cases such as the Laurentian Great Lakes and the Chesapeake Bay (cf. chapter 3) or the Gulf of Mexico downstream from the Mississippi watershed (anoxia, fish mortality, toxic algal blooms).

For continental waters, the US Geological Survey (USGS) has been entrusted with a research mission to understand and model the entire causal chain involved in the eutrophication process (sources, land transfer, river transfer). The USGS therefore rolled out an appropriate strategy based on a selection of 51 target watersheds in which a densified nutrient sampling was carried out. This sampling aimed to refine the nutrient balances of sources and better identify nutrient producing zones and the seasonality of transfer processes regulated by hydrology. These data are used to calibrate Sparrow-based (SPAtially-Referenced Regression On Watershed attributes) regionalized transfer and flow models, broadly deployed across the United States (cf. chapter 5). Another model, SWAT (Soil and Water Assessment Tool), developed by the US Department of Agriculture (USDA) and recently updated and enriched, is also deployed in some with predominantly agriculture-intensive watersheds (cf. Chapter 5). These models form the basis of retrospective and prospective scenarios to analyze or plan remediation measures for excess nutrients. This working strategy was followed for 10 to 15 years to refine the regional knowledge of nutrient dynamics, but it remained an essentially geochemical approach. It is only very recently that the literature in the United States suggested more systemic linkage relationships, extending the chain of causality right up to a probable alteration of the health status of waterbodies (biological indicators relating to macroinvertebrates and fish).

For marine waters, in 1999, the National Oceanic and Atmospheric Administration (NOAA) conducted the first synchronous assessment programme on the eutrophication status of 139 major national estuaries (National Estuarine Eutrophication Assessment), and in 2000, an academic commission recommended repeating this snapshot of the eutrophication status of American coasts every ten years. This commission also recommended improving the linkage relationship between monitoring measures and models so as to optimize the spatial and temporal positioning of samples, along the lines of what has been done in Denmark.

The US Environmental Protection Agency (EPA) is also conducting a water quality monitoring programme (National Aquatic Resource Surveys) and produces, approximately every five years, a report on the assessment of the quality of marine and estuarine coastal waters, as well as of the bays bordering the Great Lakes (National Coastal Condition Assessment). These reports cannot replace the precise assessment that each US state must perform under the Clean Water Act, but, at irregular intervals since the beginning of the 1990s, they provide a national snapshot, both synthetic and uniform, of the ecological status of the 56 coastal water bodies based on 1,100 sampling stations.
We conclude from these examples in the United States that focusing on a given alteration such as eutrophication requires a dedicated sampling strategy, and that modelling is necessary when dynamic cascading processes are involved. Modelling provides critical feedback on data and also makes it possible to integrate these data with spatial and temporal extrapolations while controlling related uncertainties. There are benefits to be gained from achieving complementarity between data from monitoring networks and modelling (for a more detailed presentation of modelling, cf. chapter 5).

### 4.2.2. Observatories in France

#### Lake observatories

In mainland France, the Alpine Lake Observatory (OLA), now designated as Observation and Experimentation Systems for Environmental Research (SOERE), was structured gradually based on initial questions relating to the development of eutrophication (1960s-1980s). The data series produced by the Observatory and SOERE are “long” (spanning several decades) and therefore far exceed average water residence times (table 4.5). They describe the ecological trajectory of three large deep lakes (Annecy, Le Bourget and Geneva), which share the same environment (same ecoregion and same type of economic development), but where a different approach was taken to implement measures against eutrophication (table 4.5).

<table>
<thead>
<tr>
<th>LAKE</th>
<th>Current status (mean residence time; maximum depth)</th>
<th>External P load control history</th>
<th>Manager</th>
<th>Date of follow-ups</th>
</tr>
</thead>
<tbody>
<tr>
<td>GENEVA</td>
<td>Mesotrophic (12 years; 310 m)</td>
<td>Progressive control from 1980 (de-phosphatation, TPP ban, etc.)</td>
<td>International Commission for the Protection of the Waters of Lake Geneva (CIPHEL)</td>
<td>Since 1957</td>
</tr>
<tr>
<td>ANNECY</td>
<td>Oligotrophic (3.8 years; 82 m)</td>
<td>Almost total from 1980 (circular sampler and discharges at the lake’s outlet)</td>
<td>Joint Association of Lake Annecy (SILA)</td>
<td>1967-1977, 1987-2010</td>
</tr>
</tbody>
</table>

Table 4.5: Dates of implementation of lake monitoring in connection with the partner management structure and trophic status history. According to Dorioz, 2010.

#### Watershed observatories

A dozen of small experimental watersheds located in several French regions are monitored by French research teams. The scientific objectives of these observation systems notably include understanding and modelling relations between watershed structures, land uses, nutrient and material flows. These watersheds form part of the research infrastructure supported by the Observation and Experimentation Systems for Environmental Research (SOERE) steered by the Allenvi alliance. They are also integrated in the Long Term Ecological Research (LTER) network and the Critical Zone Observation network. Groundwater monitoring is carried out by the Bureau of Geological and Mining Research, or BRGM (ADES database, recently included in the critical zone network), soil monitoring by the GIS Sol (BDAT soil analysis database and RMQS soil quality monitoring network). These monitoring networks are essential as they provide long-term data. In watershed-type systems, collecting high-
frequency monitoring data on water concentrations provides complementary information as these data make it possible to track nutrient exports precisely and over the long term. They can be used to determine optimal measurement times for chemical and biological elements which can then be applied in monitoring networks, as well as to better quantify relations between the landscape structure of watersheds and the quality of water, and the response time between changes in land use and nutrient flows.

Observatories in estuarine and marine environments

In mainland France, various monitoring initiatives of the coastal marine environment by university laboratories and marine stations have been grouped together and structured since 1997 under a national programme, the Littoral Observation Service (SOMLIT), approved by the CNRS's National Institute for Sciences of the Universe (INSU). Within the station and marine observatory network (RESOMAR), this monitoring programme covers 19 stations monitored by 11 marine laboratories along the coasts of the Channel, the Atlantic and the Mediterranean. Physical, chemical and biological measurements are performed twice a month. The main objective of this network is to study the impact of global change on coastal areas and its relative importance compared to local disturbances, including nutrient enrichment. Local monitoring networks dedicated to marine eutrophication have been launched by Ifremer and water agencies on the Eastern Channel (Normandy Coast Water Network, RHLN, since 2000 in Normandy and Regional Nutrient Monitoring, SRN, since 1992 in Artois-Picardy) and on all the French Mediterranean lagoons since 2000 (Lagoon Monitoring Network, RSL).

It should be noted that at global level, there is a phytoplankton monitoring programme, the Continuous Plankton Recorder (CPR), in which France is involved. Thanks to a plankton monitoring and analysis method that has remained unchanged since 1948, and to sampling instruments designed to be placed on merchant ships, this programme is one of the oldest and most extensive at world level. Several aspects of plankton ecology and dynamics are studied via this programme, including eutrophication and harmful algal blooms (HABs).

4.3. Indicators and methods for eutrophication monitoring

4.3.1. Pressure, status and impacts

Eutrophication indicators are generally classified into pressure indicators (nutrient emissions and flows), chemical status (water nutrient concentrations) and impact indicators (e.g. biological indicators, oxygen, etc., reflecting the health status of the aquatic ecosystem) (table 4.6). While status and pressure indicators are the same for all continental and marine environments, impact indicators vary according to environments.

The assessment of emissions and nutrient flows such as nitrogen, phosphorus, silica and carbon inputs, their retention in landscapes and their export to receiving environments (water bodies, estuaries, coastal areas) is based on a set of measures, indicators and models, each being tainted with uncertainty, rarely mentioned or commented. For example, measuring the effectiveness of measures implemented in watersheds to reduce N and P inputs involves uncertainty in the components of nutrient surpluses, little-known variability in point-source and non-point source discharges, calculation of estimated flows in receiving environments, with few measurements made, whereas the variability in concentrations and flow rates is significant. Determining balances of inputs-outputs in water bodies is complex when the differences reside in the margin of uncertainty of inflows and outflows. Lastly, performing a spatial extrapolation of emissions and flows to non-monitored zones using hydrological and biogeochemical models is delicate, despite their importance in quantifying input origin (urban, industrial, agricultural, point and non-point sources).
Status indicators (nutrient concentrations) can be used to link emissions and flows exported by watersheds with the concentrations measured in receiving environments. More generally, they are of major importance to establish a link with biological indicators. However, the use made of these indicators in the context of the WFD negates their significance; they are only considered as a support to biology. To overcome this problem, the Helcom Eutrophication Assessment Tool (HEAT) performs a separate assessment of the three groups of indicators (pressure, status, impact); downgrades are based on the worst indicator results. Some countries have taken highly proactive action to tackle the issue of eutrophication. For example, the Danish Aquatic Action Plan in Denmark and the Rhine Action Plan in Germany have proved successful in defining nutrient reduction targets, implementing measurement programmes, monitoring effects and revising programmes and objectives. While most European countries have adopted WFD-type monitoring, i.e. at a monthly frequency or lower, in Denmark, nutrients are measured at a frequency of up to 26 measurements per year in small agricultural and benchmark or forest watersheds, and 19 measurements per year in water bodies; the USGS is also taking this approach in its more in-depth analysis of nutrients (cf. paragraph 4.2.1). For cross-border watersheds or those with outflows to coastal areas (Netherlands, Germany and Finland), monitoring is even more frequent (52 measures per year). Measures taken less frequently and over a shorter time make it more complicated to detect trends, particularly in the case of temporary networks or rotating operations. One should also be cautious when comparing concentration indicators to target thresholds when using higher (C90) or lower (C10) quantiles, which are more uncertain than central values (C50, Cav) (cf. paragraph 4.4.2).

There are many more impact indicators and their interpretation is complex. They all share the specificity of responding to nutrient inputs, as we have seen in chapter 2, both directly (biological indicators on primary producer compartment) or indirectly (biological indicators on groups other than those in the primary producer compartment and physico-chemical indicators influenced by biological processes).
4.3.2. Impact indicators based on primary producer compartment

In the process chain, whether in a marine or continental environment, the primary producer compartment is logically the first one to be affected by the eutrophication process, with notably an increase in the phytoplankton biomass (which is usually measured by an increase in chlorophyll a) or abundant macroalgae. The recognized symptoms are identical in continental and marine waters except for the two groups that are very scarce in marine water in temperate environments, but highly developed in eutrophic fresh water, namely planktonic cyanobacteria and benthic phanerogams.

As explained in chapter 2, these manifestations are dependent on several other factors (residence time, light, temperature). The symptoms therefore have marked seasonal dynamics but that are not identified in monitoring networks, which mostly use annual frequencies for biological parameters.

Lakes

In the case of lakes, trophic classifications started to be documented as early as 1960 with the work of Vollenweider. They were later enriched to take into account other indicators (depth, turbidity, biomass expressed as a biovolume, cyanobacteria, structure and distribution of a taxonomic community, zooplankton) (tables 4.7, 4.8). The use of macrophytes appears to be appropriate for shallow lakes. The analysis of trends in toxic cyanobacteria, which could increase in water bodies in the years ahead due to climate change, is another forward indicator to be taken into account.

<table>
<thead>
<tr>
<th>Class</th>
<th>Designation</th>
<th>Average biomass between May and October</th>
<th>Biomass of peak blooming in August</th>
<th>Lake’s trophic status</th>
</tr>
</thead>
<tbody>
<tr>
<td>1a</td>
<td>Particularly small biomass</td>
<td>(\leq 0.1 \text{ mm}^3/\text{L})</td>
<td>(\leq 0.1 \text{ mm}^3/\text{L})</td>
<td>Ultraoligotrophic</td>
</tr>
<tr>
<td>1b</td>
<td>Very small biomass</td>
<td>0.1-0.5 \text{ mm}^3/\text{L}</td>
<td>0.1-0.5 \text{ mm}^3/\text{L}</td>
<td>Oligotrophic</td>
</tr>
<tr>
<td>2</td>
<td>Small biomass</td>
<td>0.5-1.5 \text{ mm}^3/\text{L}</td>
<td>0.5-2 \text{ mm}^3/\text{L}</td>
<td>Mesotrophic</td>
</tr>
<tr>
<td>3</td>
<td>Moderately large biomass</td>
<td>1.5-2.5 \text{ mm}^3/\text{L}</td>
<td>2-4 \text{ mm}^3/\text{L}</td>
<td>Eutrophic I</td>
</tr>
<tr>
<td>4</td>
<td>Large biomass</td>
<td>2.5-5 \text{ mm}^3/\text{L}</td>
<td>4-8 \text{ mm}^3/\text{L}</td>
<td>Eutrophic II</td>
</tr>
<tr>
<td>5</td>
<td>Very large biomass</td>
<td>&gt;5 \text{ mm}^3/\text{L}</td>
<td>&gt;8 \text{ mm}^3/\text{L}</td>
<td>Hypereutrophic</td>
</tr>
</tbody>
</table>

Table 4.7. Classification system of eutrophication status in Swedish lakes, based on biomass expressed as a biovolume of planktonic algae. According to Wilen 2000.

<table>
<thead>
<tr>
<th>Class</th>
<th>Designation</th>
<th>Total phosphorus (µg/L)</th>
<th>Cyanobacterial biomass in August (mm(^3)/L)</th>
<th>Number of genera of toxin-producing cyanobacteria</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Very small biomass</td>
<td>6-12.5</td>
<td>&lt;0.5</td>
<td>≤2</td>
</tr>
<tr>
<td>2</td>
<td>Small biomass</td>
<td>12.5-25</td>
<td>0.5-1</td>
<td>-</td>
</tr>
<tr>
<td>3</td>
<td>Moderately large biomass</td>
<td>25-50</td>
<td>1.25</td>
<td>3-4</td>
</tr>
<tr>
<td>4</td>
<td>Large biomass</td>
<td>50-100</td>
<td>2.5-5</td>
<td>-</td>
</tr>
<tr>
<td>5</td>
<td>Very large biomass</td>
<td>&gt;100</td>
<td>&gt;5</td>
<td>4</td>
</tr>
</tbody>
</table>

Table 4.8. Classification system of eutrophication status in Swedish lakes, based on phosphorus concentration and cyanobacterial blooms in late summer. According to Wilen 2000.

In France, the “fast diagnosis” method was developed in the early 1990s, placing emphasis on the study of a few parameters traditionally used in Europe or the United States to assess the trophic state of lakes (chlorophyll, transparency, dissolved oxygen, phosphorus, nitrogen). This method was tested by the Rhône-Mediterranean-Corsica water agency. Well aware of the limits of this approach, its authors were careful to supplement it by assessing sediment chemical quality as well as taking into account part of the biocenosis. This method is fairly comprehen-
sive given the descriptors used (physico-chemical and biological) and the compartments analyzed (open water, sediment). It draws on a fundamental principle of lake dynamics, which assumes there is a link between the physico-chemical water composition at the end of the winter exchange and the phenomena that it is likely to generate in the various compartments of the ecosystem during the ensuing plant growth period. In other words, the water body “fills up” on nutrients in late winter, when they are not yet used for plant growth, while water nutrient concentrations reach maximum levels following winter runoff and sediment discharge. In the spring and summer, these nutrients support primary production and more generally the entire trophic network, and the consequences on the environment can be observed more particularly at the end of the summer stratification, particularly at the level of the hypolimnetic oxygen reserve. It follows from this principle that the fast diagnosis method was reserved for water bodies lastingly stratified in the summer, logically ruling out shallower water bodies or those with shorter residence times. Fast diagnosis is also ill-suited to artificial reservoirs with significant extraction or drawdown.

Over the past 20 years, continuous measurements at several depths have increasingly been used in the scientific literature to characterize the physico-chemical dynamics of water bodies (dissolved oxygen, temperature, pH, conductivity, etc.) and local weather conditions (wind speed, solar radiation, etc.). These measurements also help to better understand the spatial and temporal complexity of water bodies as well as the metabolism and potential imbalances between photosynthetic production and consumption/respiration. It should be remembered, however, that France only implemented its water body network when the WFD was introduced in the early 2000s. Records are therefore very recent.

**Streams and rivers**

In the case of large rivers, the preferred measurement is chlorophyll a, quantified in the water body or in the benthos (table 4.9). These are the values typically used to determine trophic level classes, together with indicators of nitrogen and phosphorus concentration. Changes in vegetation, in connection with nitrogen or phosphorus inputs, are also identified (table 4.10).

<table>
<thead>
<tr>
<th>Trophic status</th>
<th>Average concentration in benthic chl-a (mg/m²)</th>
<th>Maximum concentration in benthic chl-a (mg/m²)</th>
<th>Concentration of chl-a in the water column (µg/L)</th>
<th>Total nitrogen (µg/L)</th>
<th>Total phosphorus (µg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Oligotrophic</td>
<td>&lt; 20</td>
<td>&lt; 60</td>
<td>&lt; 10</td>
<td>&lt; 700</td>
<td>&lt; 25</td>
</tr>
<tr>
<td>Mesotrophic</td>
<td>20-70</td>
<td>60-200</td>
<td>10-30</td>
<td>700-1 500</td>
<td>25-75</td>
</tr>
<tr>
<td>Eutrophic</td>
<td>&gt; 70</td>
<td>&gt; 200</td>
<td>&gt; 30</td>
<td>&gt; 1 500</td>
<td>&gt; 75</td>
</tr>
</tbody>
</table>

Table 4.9. Limits of classes of trophic status of streams and rivers.
According to Dodds et al. 1998.

For a more in-depth analysis of eutrophication, other impact indicators can be used. Daily variations in dissolved oxygen and pH, oxygen saturation rates are indicators that were recommended in the SEQ-Eau (water quality assessment system).

The WFD introduced other compartments (macrophytes, phytobenthos, invertebrates, fish), but a globalized use of ecological status indices cannot give a direct account of the eutrophication status. In other words,
what emerges from the assessment is a general ecological status in aggregated form. A twofold approach is required in order to establish to what extent degradation is attributable to eutrophication: (1) in the quality elements identified, e.g. diatoms or macrophytes, looking for traits of species showing an affinity for nitrogen or for phosphorus, or interpreting the functional metrics of the various biological communities, (2) looking for the origin of possible causes by analyzing pressures, which implies having the required depth of detail. However, networks’ usual temporal sampling may be insufficient for a proper apprehension of the dynamics of eutrophication. The other cautionary recommendation concerns the spatial representativeness of measurement stations, whose results are used by extrapolating them to water bodies. Water bodies have a marked internal spatial heterogeneity for parameters such as morphology or the tree cover of riparian area, which can have a local influence on the manifestations of eutrophication. A spatial contextualization of the stations in their broader environment is therefore essential.

In 1998-1999 and 2002-2003, the Rhône-Mediterranean-Corsica water agency had implemented a specific eutrophication monitoring network for rivers considered to be sensitive (around 200 stations): quantitative measurements of chlorophyll a were carried out in every summer month at stations with phytoplankton development, as well as a quantitative inventory of macrophytes (in particular proliferating species, recovery rate, biomass) supplemented by 24-hour recordings of pH and dissolved oxygen. Regrettably, after the overhaul of the WFD networks, this action was never repeated.

**Marine environment**

At global level, several methods have been developed to assess the magnitude of eutrophication effects (table 4.11). Most of these methods integrate physico-chemical and biological indicators and measure both the direct and indirect effects so as to assess water quality in relation to eutrophication pressure. In most of these methods, however, there are still persistent difficulties in linking nutrient contributions with ecosystems’ responses (table 4.11: last column). Generally speaking, there are many biological responses to multiple pressures of anthropogenic origin, but few specific biological responses.

<table>
<thead>
<tr>
<th>Name of the method</th>
<th>Biological indicators</th>
<th>Physico-chemical indicators</th>
<th>Link with nutritional load</th>
</tr>
</thead>
<tbody>
<tr>
<td>TRIX (TRophical IndeX for marine systems)</td>
<td>Chl</td>
<td>DO, DIN, TP</td>
<td>No</td>
</tr>
<tr>
<td>EPA NCA Water Quality Index (US Environmental Protection Agency National Coastal Assessment)</td>
<td>Chl</td>
<td>WC, DO, DIN, DIP</td>
<td>No</td>
</tr>
<tr>
<td>ASSETS (ASSeessment of Estuarine Trophic Status)</td>
<td>Chl, macroalgae, phanerogams, HAB</td>
<td>DO</td>
<td>Yes</td>
</tr>
<tr>
<td>TWQI/LWQI (Lake Water Quality Index ; TWQI : Transitional Water Quality Index)</td>
<td>Chl, macroalgae, phanerogams</td>
<td>DO, DIN, DIP</td>
<td>No</td>
</tr>
<tr>
<td>OSPAR COMPP (OSPAR Comprehensive Procedure)</td>
<td>Chl, macroalgae, phanerogams, HAB, phytoplankton</td>
<td>DO, TP, TN, DIN, DIP</td>
<td>Yes</td>
</tr>
<tr>
<td>DCE (Directive-Cadre européenne sur l’Eau)</td>
<td>Phytoplankton, Chl, macroalgae, benthic invertebrates, phanerogams</td>
<td>DO, TP, TN, DIN, DIP</td>
<td>No</td>
</tr>
<tr>
<td>HEAT (HELCOM Eutrophication Assessment Tool)</td>
<td>Chl, PP, phanerogams, benthic invertebrates, HAB, macroalgae</td>
<td>DO, TP, TN, DIN, DIP</td>
<td>No</td>
</tr>
<tr>
<td>RSL (utilisée dans le cadre du Réseau de suivi des lagunes méditerranéennes)</td>
<td>Chl, macroalgae, phanerogams, HAB</td>
<td>DO, WC, SRP, TP, TN, DIN, OM TN and TP of sediment</td>
<td>No</td>
</tr>
<tr>
<td>STI (Statistical Trophic Index)</td>
<td>Chl, PP</td>
<td>DIN, DIP</td>
<td>No</td>
</tr>
</tbody>
</table>

The measurement of chlorophyll a, used to assess changes in the phytoplankton biomass, is common to all of these methods, as is the measurement of the concentration in dissolved oxygen (except for the STI method). For algae and phanerogams, assessing the recovery and species richness of perennial species in relation to opportunistic species forms a common basis for all the indices available in the world. Local disturbances of anthropogenic origin are known to induce a shift in ecosystems from a preserved status dominated by perennial species to a degraded status dominated by opportunistic annual species.

Concerning harmful algal blooms (HABs), which refers to any proliferation of microalgae causing harm due to their high biomass or toxin production, the eutrophication effect is clearly linked to each phytoplankton species as well as to the hydrodynamic conditions of the environment. The HAB manifestations most commonly related to excessive enrichment, especially in nitrogen, are foam produced by the colonial flagellate Phaeocystis, ASP toxin produced by species of the diatom genus Pseudo-nitzschia and, more generally, an increase in the diatom/dinoflagellate ratio. These three indicators are measured regularly by some countries.

Concerning benthic invertebrates, the harmful effects of the decomposition of green or brown macroalgae on beaches are limited to the wildlife of the foreshores affected and juvenile fish usually present in the water strip at a depth of less than 2 metres. However, no specific indicator has been implemented yet for these disruptions. The impacts of hypoxia/anoxia on the sea floor can affect large areas, however, but virtually no regular measurement of faunal abundance in sensitive zones, or assessment of mass mortality episodes, has been implemented.

In France, the indicators used for eutrophication as part of the WFD’s monitoring reference networks (RCS) are: nutrient concentrations, measurements of dissolved oxygen and chlorophyll, phytoplankton cell counts, coverage of rocky areas by opportunistic algae, aerial monitoring of opportunistic green algal blooms on the sandy zones and mudflats of the Channel-Atlantic coast, and harmful algal blooms. In addition, regular monitoring of the internal nitrogen and phosphorus content of ulva algae is performed on the Channel-Atlantic coast within the scope of three of the WFD’s Operational Control Networks. This monitoring makes it possible to determine if algal growth is limited by nitrogen or phosphorus, and also to detect changes even before they are discernible on a larger scale. The interpretation of internal nitrogen and phosphorus quotas is based on the knowledge of critical (quota from which growth is no longer at its peak) and subsistence quotas (quota from which algal growth stops).

### 4.3.3. Impact indicators based on other biological groups

As described above in chapter 2, the cascade effects triggered by the eutrophication process contribute to:

1. an increase in biomass and changes in the species composition and productivity of zooplankton invertebrates, as grazers (e.g. Cladocera of the Daphnia genus) are privileged compared with species of other trophic groups,
2. changes in the biomass, productivity and species composition of benthic invertebrates due to the scarcity/disappearance of the more sensitive species and the proliferation of tolerant grazer species (e.g. gastropods of the Lymnaea genus) to the detriment of other trophic groups and
3. a loss of species diversity in zooplankton and benthic invertebrate populations.

The relation between nutrient concentration, algal biomass, and biomass of primary consumers is modulated by the water regime, so that the response of benthic macroinvertebrates to nutrient contamination in rivers can be considered to be closely dependent on physiography and climate, as it is strongly influenced by the water disturbance regime. Conversely, the effect of climate change on aquatic invertebrate populations can be largely modulated by local improvements in the physico-chemical quality of water (notably the reduction in phosphorus levels observed in many large European rivers). This can contribute to the return of pollution-sensitive species, even though the long-term trend is for a significant increase in water temperature and a reduction in water discharges. Similarly, the shadow effect generated by riparian woodland in small and medium-sized waterways can play a significant role in controlling eutrophication (up to 44% reduction in phytoplankton productivity), even though it has a limited impact on dissolved oxygen concentration, chemical oxygen demand, and levels of phosphate, nitrate and ammonium ions in water.

Using benthic macroinvertebrates as an indicator of eutrophication is therefore less direct than for impact indicators based on primary production. Research is currently under way to extract more specialized informa-
tion than that provided by general indices, thanks to the simultaneous analysis of several dozens of taxonomic or functional metrics characterising benthic macroinvertebrate assemblies (diagnosis rationale). In this way, based on the frequency of use of biocological metrics by benthic invertebrate assemblies, it is possible to characterize the risk of alteration by 16 types of anthropogenic pressures of hundreds of sites located in shallow waterways of mainland France. The sixteen types of pressure studied notably include “organic matter”, “nitrates”, “other nitrogen compounds” and “phosphorus compounds”.

4.4. Monitoring in the future

4.4.1. Measuring complementary parameters

It is generally accepted that there is a good qualitative knowledge of the processes involved in eutrophication, but that the quantitative influence on ecological processes and changes in community structures are not well known. Considering the complexity of the eutrophication process, it appears that eutrophication monitoring strategies do not take into account all of the control parameters with positive or negative effects.

Regarding nutrient contributions, there are few measurements of atmospheric inputs despite the fact that they can account for a not-insignificant proportion of total inputs in some regions. Soil phosphorus contents are also little measured, and it would be particularly useful to have this information. Assessments also take little account of sediment studies, and information obtained on the pelagic system and the water column cannot be interpreted as a whole.

The limitation of planktonic primary production could be investigated by studying the Redfield ratio, the proportion of carbon, nitrogen and phosphorus found in phytoplankton biomass, or by measuring the ratio of flows available to algae for the various elements, rather than the ratio of stocks available at the measurement time; in other words, moving from a static to a dynamic assessment of the limiting factor(s).

It would be interesting to combine the measurement of chlorophyll concentration with a measurement of processes such as primary production.

For phytoplankton, reflections revolve around a holistic assessment of plankton dynamics based on the joint use of three indicators incorporating: 1) the study of functional groups or life forms, 2) the use of temporal series of phytoplankton biomass and zooplankton abundance in order to monitor significant changes in planktonic production, 3) diversity indices taking into account the number or dominance of species.

Concerning harmful microalgal blooms (HABs), the challenge lies in understanding which blooming is related to eutrophication and why a particular species reacts to certain nutrient conditions, even though other factors also come into play.

It is important to study the relationship between eutrophication, climate change and sea water acidification and to consider the cumulative and synergistic effects when assessing ecosystems’ status. With respect to the effects of climate change and acidification, observation programmes do not systematically incorporate the system’s key parameters, namely pH, pCO$_2$, total alkalinity, and dissolved inorganic carbon. There is no therefore no indicator incorporating these parameters. Yet, these effects could very well neutralise the efforts made in terms of reducing nutrient inputs.

For biological indicators on groups other than primary producers, as referred to above, research is currently under way to extract more specialized information than that provided by holistic multi metrics indices.

4.4.2. Quantify uncertainties and optimize monitoring frequencies

Currently, waterbodies nutrient assessments use two types of indicators: on the one hand, in situ measurement of concentrations put into statistics (averages, extremes or ratios weighted by flows), and on the other
hand flow estimations, usually in relation to the watershed surface. These indicators are quantified per season, per year, or over several years, using models to reconstruct non-measured events. Apart from pilot monitoring programmes set up at a few specific sites, the assessment of these indicators is based on regular, regulatory monitoring, at a monthly or at best bi-monthly frequency. In many cases, however, this frequency is only appropriate for large watersheds and nutrients with little variation in concentration, as is often the case for nitrates, which have a fairly predictable seasonal behaviour. For smaller watersheds with a more responsive hydrology and for nutrients in particulate form, such as P, monthly sampling produces estimates of indicators with high levels of uncertainty.

In order to estimate nutrient flows, there are around twenty methods that can combine a high-frequency data element (flow rate) with a discrete data element (concentration in nitrogen, phosphorus, or any other element), generally divided into two major groups: regression methods, used to estimate concentrations of non-sampled days by empirical relations involving flow rate, season, sometimes turbidity for particulates; average concentration methods, which consist in multiplying average concentrations by flow rates over specific periods. These conventional methods are still widely used to assess flows, even though they are not entirely satisfactory and produce inaccurate estimates. This is particularly the case for sparse concentration data or when the data set does not reflect a representative range of concentrations and flow rates, or else when the relation is not linear (e.g., nutrients), or when the scatter graph is fairly dispersed and shows no particular relation between concentration and flow rate.

Two techniques are then employed. The first technique consists in dividing the general regression into several sub-regressions. This consists in identifying various relations depending on low- and high-water periods, seasons, rising or receding water periods, or over a multi-year period if a specific trend was identified in the data. This method therefore requires a preliminary analysis of the watershed’s functioning so as to identify homogeneous periods (similar export regime with a univocal relation between concentration and flow, etc.). The second technique consists in using a multivariate model involving several explanatory variables, without making assumptions about changes in concentration-flow relations.

Despite the many studies conducted, choosing a method, i.e., one that produces little-biased and accurate estimates, is not a simple matter. Recent studies propose a hierarchy of methods based on the exponent of the concentration-flow relation in the high-water period and the value of a flow variability indicator. Other studies suggest using a single method encompassing trend, seasonality, and relation with flow rates, with an automatic estimate of the weight of these components according to the data analyzed.

For a given method, however, current work is researching explanatory factors of uncertainty, the idea being to determine nomographs for the candidate indicator making it possible to link uncertainties with monitoring frequencies and indicators of flow and concentration variability. The nomographs can then be used in two ways: first, to assign uncertainty to the monitoring already performed over the last 20 to 30 years to study evolutionary trends, second, to suggest monitoring frequencies for specific sites and nutrients so as to obtain estimates with a low level of uncertainty.

For example, table 4.12 provides an indicative classification of uncertainties by type of nutrient (Si, NO₃, DOC) for monthly and annual flow estimates. We observe that low uncertainty (good error) is obtained for silica and DOC with a bias of < ± 3% and precision of < ± 5%, whereas for nitrate, thresholds are more significant (< ± 5% for bias and < ± 10% for inaccuracy). Similarly, by construction, an assessment of flows on a monthly scale is more uncertain than an assessment on an annual or multi-annual scale; errors tend to even out over longer periods.
4.4.3. New tools and new forms of monitoring

Sensors

*In situ* sensors are still little employed in operational and scientific monitoring despite recent technological progress for many parameters, e.g., greater detail, limited drift, continuity and representativeness of the measurement, broad autonomy, better robustness, enhanced protection against biofouling, data transmission, remote management, etc. Sensor performance is variable, however, depending on the parameter considered.

In marine and continental environments, chlorophyll fluorescence sensors are increasingly considered as sufficiently robust to determine phytoplankton biomass and gross and net primary productivity, although the non-photochemical quenching (NPQ) phenomenon (a mechanism protecting pigments against photodestruction) makes the fluorescence yield drop in the sunniest hours of the day, resulting in an underestimation of the chlorophyllous biomass; that said, these sensors can deliver more robust results than an episodic cell count to analyze long-term changes. Oxygen sensors can also diagnose periods of high algal production (oxygen supersaturation or undersaturation) or degradation of organic matter that can lead to hypoxia or even anoxia. These sensors present the advantage of monitoring impulsive phenomena, notably algal blooms. Biofilm cleaning systems are increasingly effective to mitigate or even cancel sensor drift, although calibrations in relation to one-off measurements are still required, especially at times of rapid plant growth (once or twice a month).

Turbidity, pH, temperature and conductivity sensors are also robust and help better understand physico-chemical conditions and changes, especially during periods of rapid plant growth and organic matter degradation.

Nitrate sensors have been starting to deliver tangible results for about five years. Figure 4.2 illustrates the difference between continuous nitrate data, conventional manual (albeit frequent) sampling, and a composite sample, which makes it possible to measure average flows per given time period. Automatic phosphorus measurement systems are still in the testing phase. In rivers where phosphorus is predominant in particulate form, linking conventional one-off phosphorus measurements and continuous measurement of turbidity or flows and rainfall can produce relatively reliable relations between turbidity and total phosphorus concentration.

![Figure 4.2. Comparison of three types of NO₃ concentration measurements, snapshot, average (SC Sampler) and continuous. Source: Rozemeijer et al. 2010. © American Chemical Society.](image)

These sensors can be installed almost anywhere: floating pontoons on rivers and water bodies, river banks, buoys at sea, estuaries and water bodies, boats or Autonomous Underwater Vehicles to monitor a vast marine zone. Multi-parameter sensors can also be submerged at different depths to study water body stratification, for instance.
Satellite images

Satellite images, which were first used in the marine environment, are starting to be used for inland water bodies and estuarine environments. Using them in addition to in-situ measurements offers several benefits: 1) higher frequency of satellite passages – several times per month; (2) spatial visualization of the area of interest (instead of one measurement of a single representative point); (3) increasingly competitive costs by comparison with in-situ measurements; (4) possibility of calibrating parameter models on in-situ measurements. In marine environments, chlorophyll concentration and water transparency are traditionally assessed by measuring chlorophyllous pigment and using the Secchi disk. These two variables can now be assessed by satellite images, by measuring water colour (Satellite ocean colour measurements) and using an attenuation coefficient at 490 nm, respectively. This is a fast-growing field. For example, for the last two WFD six-year assessment cycles, satellite data helped establish monthly surface chlorophyll maps as well as the 90th-percentile of surface chlorophyll over the March-October productive period for all coastal waters of mainland France (figure 4.3).

On land, until very recently, satellite images were still used indirectly (study of land use to deduce a site’s overall situation) or qualitatively (detection of changes between two images to perform an analysis by classification and to identify water body categories or the presence of invasive aquatic vegetation). The scientific literature has recently been reporting significant results indicating the possibility of accurately estimating water quality indicators (concentration in chlorophyll-a, concentration of suspended particulate matter, turbidity, water transparency) by reversing the radiometric signal measured by imaging- or radiometer-type satellite sensors (examples include MERIS, LANDSAT, SPOT, IKONOS, WORLDVIEW). The advantage with this type of measurement is that it is fast when analyzing a large number of lakes or when spatialized information is required, like for the most recent water body eutrophication models, which use satellite images to predict spatialized primary production. However, several articles point out that a simple measurement of chlorophyll is not sufficient. Notions of taxonomic composition are essential to perform a correct assessment of a lake’s eutrophication level. It is important to have knowledge of the algal classes that dominate the phytoplankton compartment: when assessing the trophic level, a biomass dominated by Desmids (often characteristic of meso- to oligotrophic environments) does not have the same signification as a biomass dominated by cyanobacteria (often characteristic of eutrophic environments). Furthermore, satellite imaging does not provide access to measurements at various depths, which is a limiting factor, especially for lakes with chlorophyll maxima at around 5 to 15 metres (e.g. alpine lakes).
New forms of monitoring

Another low-key trend has been observed in France in recent years with the emergence of eutrophication monitoring systems built in parallel or as a complement to scientific research or to regulatory and institutional action. These systems all focus mainly on in situ observation of eutrophication phenomena based on their most visible manifestations (animal or plant mortality, water clarity and colouring, algal growth, etc.). Photography is the most frequently-used medium to report on these manifestations, with or without metadata and positioning information. Data can be pooled online, offering players many possibilities for interaction.

These systems are of two kinds. The first originate from stakeholders committed to speaking out against the causes and consequences of eutrophication of anthropogenic origin. With the support of environmental protection associations, they mobilize both existing naturalist networks and more recently-involved players from more diverse cultures and backgrounds. These networks have developed especially since the end of the 2000s for the observation of green tides in Brittany and neighbouring regions.

The second build on scientific initiatives. These networks are still little developed, but examples worth citing are Ecoflux (focusing on water quality measurements at the outlets of coastal rivers) and Phenomer (focusing on the observation of microalgal blooms in the marine environment). Phenomer, created in 2013, combines several research institutes and associations specializing in marine observation and environmental education. The network seeks to encourage citizen contribution for the observation of microalgal blooms.

Although they have very distinct mechanisms and rationales, these two types of systems fall within the scope of participatory science in the sense that they rely on the mobilization and coordination of various types of expertise, notably scientific expertise, to broaden and disseminate scientific culture on ecosystem dynamics. The emergence of intelligence and monitoring systems bears witness to the social dynamics of addressing eutrophication issues, marked by a growing awareness of the phenomenon, which received a great amount of publicity in the 2000s.
5. The move towards remediation: predictability, remedial approaches and integrated management

Remedial approaches to eutrophication can be split into two main categories—those that treat the symptoms and those that deal with the causes. The symptoms remain difficult to tackle and this kind of measure is often reserved for small-scale hydrosystems. Measures that act at the source are, consequently, vital and to be applied ahead of other remedial approaches.

The policies, technologies and investments applied to treat point source pollution have proven effective. Improvements can still be made but methods are available and are clearly identified. For non-point source pollution, the solutions are clearly more complex, dependent on climatic, topographic, geological and soil factors, the structure of the landscape and the ecotones between terrestrial and aquatic areas, and current and past human activities (agricultural systems, urban development patterns, etc.), the drivers of which often lie outside the territory in question. These strategies must therefore form part of overall action plans and take account of impacts caused elsewhere, not only those affecting the target processes or goals.

Many watersheds are shifting towards green infrastructure policies1, which are well thought-out with regard to the territory and are largely beneficial. However, large-scale plans to apply good environmental practices (or BMPs—Best Management Practices), primarily on a voluntary basis, have demonstrated that the refusal of certain users to sign up to the policy at key points in the territory can diminish the effectiveness of the plans, as was the case, for example, in the Chesapeake Bay watersheds. These ambitious plans have also demonstrated that control strategies including the set-up of protected areas are not without their limits as even these areas can become saturated in phosphorus. These arguments all point to a need for a reduction of nutrients at the source, whether those sources are point or non-point.

1. http://www.eea.europa.eu/fr/articles/infrastructure-verte-mieux-vivre-grace. Green infrastructure is a network of natural, semi-natural and green spaces that provide various ‘ecosystem services’, the basis of human wellbeing and quality of life. Green infrastructure can provide multiple functions and benefits within a single territory. These functions may be environmental, such as conserving biodiversity or adapting to climate change, or social, e.g. providing water drainage or green and economic spaces, or job creation and higher property prices. Unlike grey infrastructure, which is typically limited to a single function such as drainage or transport, green infrastructure is valuable for its potential to tackle several problems at once.
5.1. Actions that treat the symptoms of eutrophication: principles and limits

Physical techniques used to tackle system symptoms have the following generic objectives: reducing residence time or destratifying the water column. In these cases, water is added to watercourses at key times, which is possible when there are sufficient reservoirs upstream and where tensions between use and transaction costs permit; otherwise mechanical action can be taken to stir water in a lake or reservoir. With both methods, action can only be taken on a case-by-case basis, as it is highly dependent on local context.

Several attempts to use chemical techniques have been described in the literature. One method to fight hypoxia or anoxia in deep zones is to re-oxygenate the environment artificially. This technique soon reaches its limits when applied to the Baltic Sea, for example, where the models calculate an annual requirement of 2–6 million tons of O₂ or 19,000–55,000 rail tanks of liquid oxygen! In addition, the rapid re-oxygenation of water can trigger colonization by burrowing worms which, through their burrowing action, transfer contaminants from the sediment to the water column. Other chemical engineering methods use substances that aid phosphorus precipitation such as lime, aluminium, sulphates or iron chloride. These procedures are used at water treatment plants but are costly and have collateral effects, particularly for the nitrogen cycle. Techniques used to extract and treat sediment have also been described, involving, for example, coal ash pellets or chemical substances such as aluminium polychloride. These techniques are also costly, have a limited geographic effect and their long-term impacts have not been fully understood, with potential uncontrolled leaching of other accumulated molecules, especially metals or phosphorus, which may incur risks.

Alongside these physical and chemical techniques, there are other methods that target the biotic compartment of the water body. Some aim to destroy algae directly, using algacides such as copper sulphate, which can be toxic.
for other organisms, or oxygen peroxide, which has a selective effect on cyanobacteria (no recognized effect on eukaryotic algae and higher plants) but which must be applied repeatedly. The latter substance is very expensive and can only be used in small systems (such as dams). Its effect on micro-organisms and their role in nutrient recycling remains unknown to date. In the marine environment, one method used at fish and shellfish farms is direct clay dispersion on toxic microalgae proliferations. However, the repeated application of clay appears to disrupt the growth of the Mercenaria mercenaria clam.

Finally, biomanipulation emerged as a popular method for treating shallow lakes in the 1980s. As with re-oxygenation, the basic principle is logical: by intervening in the predator/prey relationship and encouraging the development of algae grazers, we can expect to reduce algal proliferations, increase the depth of light penetration and improve general clarity in the environment. Thus, large volumes (several tons) of planktivore fish were caught and removed from lakes, then introduced into eutrophicated hydrosystems. Feedback from these experiments has been very mixed, with sometimes spectacular short-term effects with the expected result, but a rapid return (within about 5 years) to pre-manipulated conditions. The authors of a recent appraisal concluded: ‘**Biotic manipulations in the lake are only useful to reinforce a restoration scheme, treat symptoms and temporarily maintain high environmental quality until the internal (nutrient) load has been significantly reduced**’. In the marine environment, the restoration of species that help control phytoplankton, such as natural bivalve filter-feeder populations, cannot really be used as a method to control eutrophication. However, it can be effective as part of a strategy to restore oligotrophic conditions in certain coastal ecosystems, as bivalve harvests are a way of exporting nutrients from the ecosystem. Yet bivalve farming modifies the nutrients’ biogeochemical cycles, especially the recycling of nutrients in their immediate environment, and shell fishing can sometimes trigger the factors that contribute to anoxia. Collecting washed-up macrophytes is another technique. It does not remedy the symptoms of eutrophication, but simply serves as a corrective action. It removes nutrients from the ecosystem and limits anoxia linked to plant decomposition. The piles of macroalgae collected can be exploited, but collection is costly.

In the end, none of these actions can be applied universally or provide a long-term solution. They can help remedy a symptom on a case-by-case basis in small-scale aquatic systems, with the figure of 50 ha having been quoted for physical and chemical actions. Before implementing these actions, prior impact studies looking at the potential effects on biological organisms found in the hydrosystems should be carried out systematically.

### 5.2. Actions that treat the causes of eutrophication: principles and limits

#### 5.2.1. Actions on domestic sources

Population growth and the concentration of people in villages or urban centres have resulted in the production of considerable volumes of wastewater. Population concentration raises the issue of wastewater and rainwater collection and treatment systems, on a collective or individual basis. Some elements in the chain have a technical side that is rarely dealt with in the scientific literature.

**To reduce at source**

One remedial approach is to reduce volumes at source, i.e. in the composition of domestic products consumed. For phosphorus, reduction at the source was introduced in 2007 with the regulations on domestic textile detergents. Since then, the load assignable to the source (washing textiles) can be considered negligible in France. If this reduction was also applied to domestic dishwasher detergents (European regulation 259/2012 on consumer use), we estimate an additional at-source reduction of 0.2 g of P per person, per day (10% of the flow produced),
which would have a positive impact on the financial and environmental cost of treating wastewater in sensitive zones. As the largest source comes from human physiology (urine and faeces), better adapted diets could be one way of reducing volumes at source.

**Better assessment of the volumes to be processed**

Treatment plants generally receive several sources of pollution of domestic, industrial and tertiary origin. Their objectives are defined in terms of volume to be treated. In built-up areas, a certain quantity of industrial pollution is often discharged into the collective network. This discharge, for additional treatment via the collective system, will be very clearly defined in an agreement (with financial conditions) between the industrial concern and the network operator. Ultimately, the flows generated by tertiary or industrial activities are difficult to document as they are all quite specific. These different sources lead to a high degree of variability in the sources entering water treatment plants which, depending on the requirements of their specifications, are sized to fulfil treatment objectives in diverse situations, mainly different climate conditions (dry weather, rain, etc.). The data from self-monitoring in water treatment systems is an important source of information when it comes to assessing the pollution flows to be treated.

The base unit used to characterize pollution from domestic sources takes the theoretical value defined in the European directive dated 21 May 1991 per population equivalent (PE): a PE means a five-day biochemical oxygen demand ($\text{BOD}_5$) of 60 g of oxygen per day. Nitrogen and phosphorus parameters are not defined in that directive. The domestic load per inhabitant has been much better estimated in recent data (daily mass of physiological origin of 1.4 g of total phosphorus (including 1.1 g of $\text{P-PO}_4^{3-}$) and 9.3 g of total nitrogen (including 8.4 g of $\text{N-NH}_4^+$), which can vary from one person to another, depending on diet, etc.).

While the theoretical number of inhabitants connected to the network and contributing to the pollutant load to be treated is known, the number of residents actually present is never certain. For example, in tourist areas, this is an economic and environmental issue for local councils who need accurate data on the actual number of inhabitants in the peak season and the number of full-time residents, so that they can optimize the dimensions of their facilities and adapt them to the pollution to be treated. Some adaptations are implemented in tourist areas for varying lengths of time, given that the time taken to achieve a balance is around one month.

With regard to individual water treatment facilities, it is thought that approximately 20% of the French population are concerned, either for their main home or their holiday home. The impact of individual treatment facilities on water bodies is considered negligible overall, especially since it mainly concerns scattered settlements where population density is low. In addition, the regulations only permit inflows of treated wastewater if the required technical solutions are in place. However, very locally, especially in situations well upstream, leaks can be seen in small aquatic ecosystems or small watercourses located near individual treatment facilities.

**Requirements for treated wastewater and evolutions**

The water treatment requirement is the outcome of the concentration of pollutant loads at an outfall, after a sanitation network has been established. It responds to environmental and public health concerns. In 2015, there were 21,079 wastewater treatment plants in France, representing a total load of 78 million PE for treatment capacity of 104 million PE. Urban areas with more than 2,000 PE accounted for 3,795 wastewater treatment plants and a pollutant load of 73 million PE, 18% of plants therefore treat 94% of the pollutant load.

Large plants work almost exclusively using what is known as the activated sludge process, based on the controlled development of certain bacteria, responsible for the decomposition of various forms of nitrogen by oxidizing the mineral forms in solution, then forming nitrogen gas. Phosphorus removal long relied on physical and chemical processes but is now possible using the biological ‘activated sludge’ method. When required by the receiving environment (areas identified as sensitive to eutrophication according to the regulatory definition of sensitive area), maximum discharge values are set for parameters other than carbonaceous matter, including total phosphorus and nitrogen. For a yield of 80%, the mass of residual phosphorus compounds discharged in water is measured at 0.35 g of P per person, per day in sensitive areas. In ‘eutrophication-sensitive areas’ and for plants treating daily flows exceeding 6,000 kg of $\text{DBO}_5$, requirements are stricter still with values in water of an annual average 1 mg
P.L. for phosphorus and 10 mg N.L\(^{-1}\) for nitrogen. This requires rigorous operation of the treatment plant and careful control of suspended matter discharge. For structures treating daily flows of less than 600 kg DBO\(_5\), nitrogen and phosphorus values are set by prefectural order as required, in response to environmental imperatives. The mass of phosphorus removed (and which does not therefore end up in the water) is contained in the sludge, which may be spread on agricultural land. If handled properly, this spreading is a way of recycling and making use of the phosphorus, forming part of the circular economy.

**Distinct treatment of human dejecta and other waste**, such as industrial waste, may be given priority. Likewise, the separate collection and treatment of human waste, with faecal matter on one side and urine on the other, may be a good way of recycling (examples have been seen in Sweden, Germany and Switzerland) as farmers can use healthy—i.e. germ-free—urine for fertilization, as mineral nitrogen is largely contained in urine.

### 5.2.2. Actions on agricultural sources

**Managing livestock manure**

Livestock manure management must be the first approach applied when balancing nutrients, and it should be handled per plot, per soil type and per cropping system. The effluent has to be quantified and qualified (concentrations, speciation, etc.), then the soil characteristics taken into consideration along with its ability to regulate inputs, using models and management/forecasting tools to predict nitrogen and phosphorus leaching. Knowledge and tools for nitrogen management are well established, but less so for phosphorus. On the scale of a farm or a territory, the assessment report will include an assessment linked to crops and to livestock, and explain all the related imports and exports, including animal feed. Indicators are provided in livestock farming regions to better estimate the dependency of farming systems on nutrient imports of every kind (fertilizers, feed, etc.). This assessment is not always done even though it is a crucial first step when looking for remedial solutions, which may be multiple (animal density, breed/productivity, feeding methods, etc.).

Nutrient fluxes from livestock manure are well documented along the chain, ranging from animal feed to agronomic use of animal waste. When mixed with water and litter, animal waste (urine and droppings) forms dung and slurry, known as livestock manure. This manure is mainly made up of water, complex organic matter and low quantities of mineral elements. From the time it is excreted by animals and at every stage of its processing, this organic matter undergoes various biological, chemical and physical reactions, engendering modifications to the chemical form of the elements and transfers to the natural environment, particular to the atmosphere. Direct nutrient losses to surface water through run-off during the storage/treatment phases are negligible, provided that storage volumes are adapted (in principle, this is stipulated in the regulations).

Nutrients present in animal waste come from their feed: the quantity rejected by the animals in their excrements is the quantity ingested minus the quantity assimilated by the animals. The majority of nitrogen and phosphorus ingested is excreted (55–95% for N and 58–80% for P), but this ratio depends on the type of feed and the quantity ingested, the type of animal and its physiological stage. Among these various factors and for the same type of animal and breed, feed plays a major role. Studies looking at pigs and dairy cows have demonstrated that a reduction of nitrogen and phosphorus in feed can significantly reduce levels in waste. Nitrogen is excreted in urine and faeces and the ratio depends on feed, while phosphorus is mainly excreted in faeces. Ground return—in full (floor production) or to a degree specific to each of these two forms of livestock farming (off-land production)—will therefore see different N:P ratios in organic matter inputs of animal origin to the soil.

**For nitrogen**, it should be noted that animal waste is subject to atmospheric volatilization during storage/treatment and spreading (ammoniac (NH\(_3\) flow varying from 15% to over 60% of the nitrogen excreted, much lower nitrogen protoxyde (N\(_2\)O) flows of just a few percent)). These emissions vary widely, depending on the intrinsic characteristics of the animal waste (pH, dry matter rates, etc.), which in turn depend on animal feed, the type of animal, the rearing infrastructures, husbandry practices, and are also linked to storage and spreading conditions (temperature, rainfall, etc.), soil and vegetation type during spreading, any treatment applied (stored in buildings, outside pit,
treatment, spreading, etc.). It is difficult to quantify the fluxes because of the differences in spatial and temporal coverage. These fluxes contribute to atmospheric deposits, bringing about a potential transfer of pollution from the water to the atmosphere (cf. 2.3.1), as the deposits concern both terrestrial and aquatic ecosystems.

For phosphorus, it should be noted that almost all the fluxes from animals or from effluent input from off-land production is excreted or spread on farmland, and thus becomes part of the soil. These inputs require cautious phosphorus management, depending on the phosphorus already found in the soil and its mobility (P saturation rate, P mobility). Where there is no controlled fertilization, ‘confined’ animal husbandry (in buildings) often results in manure spreading over a relatively limited surface, meaning quantities per surface area unit clearly exceed the plants’ requirements, so phosphorus accumulation in the soil (and on the surface) increases the risk of losses. This risk is higher for phosphorus because of the low N:P ratio in manure, with an average of 4 for manure (and 2.5 for poultry manure), compared to a very variable ratio for cultivated plants (cf. below). If the manure input is based on the crops’ nitrogen requirement, there is likely to be phosphorus accumulation in the soil. Indeed, the dose of livestock manure used is usually determined by N management for crops. Joint nutrient management is therefore vital.

**Fertilization management in field crop systems**

There is no simple relationship between the nitrogen and phosphorus quantities input, crop yield and leakage to groundwater. This is why fertilizer management is not simply a matter of addition or subtraction. We need to take other factors into consideration in the balance (leftover, mineralization of organic matter, atmospheric deposits, symbiotic nitrogen fixation, etc.). Many agricultural practices influence the relationship: choice of species and varieties, crop associations and cover crops, types of fertilizer and application mode, crop residue management, tillage, and so on. As is the case for livestock systems, an assessment is the first crucial step when seeking ways to optimize cropping systems and fertilization. In France, rational use of nitrogen fertilizers is determined using the element balance method. This means calculating the terms of the nitrogen balance of a crop over its main growth period to determine the provisional nitrogen dose to be applied to meet the crop’s requirements, according to the target yield. This balance is calculated on the basis of crop rotation, integrating the succession of crops and cover crops that play a positive role wherever their planting period is significant (see the summary study on cover crops). Useful methods have also been developed (cf. the French-language publication *Fertilisation et Environnement: Quelles pistes pour l'aide à la décision* from Pellerin et al. 2014). However, each term in the equation is a source of uncertainty, and we need to reduce that uncertainty to prevent excess inputs.

The first critical point is to determine the target yield, the basis for many rational usage methods; that yield may not be achieved however due, for example, to weather conditions that are not conducive to plant growth, something that cannot be predicted in advance, or due to over-fertilization increasing the post-harvest surplus. The solution to these difficulties is real-time management, which can be applied using several principles to assess the nutritional state of the crops. The now frequently used nitrogen credit method and the improved forecasting of nitrogen mineralization during crop growth enable optimized fertilization recommendations. An improvement in seasonal climate predictions could also contribute.

The second critical point is the heterogeneity of soils and crop condition. Remote sensing and proxi-detection can be used to monitor changes in the condition of a crop. It is estimated that this approach could improve the efficiency of fertilizer use (savings of 10–80%) and enable a 30–50% reduction in the nitrogen surplus, without affecting yield or quality. An accurate estimate of the mineralization of the soil’s organic matter is also necessary for precise fertilizer management: this flow is significant in terms of quantity but is only known approximately.

It is estimated that once the crop system, soil and climate have been fully described (which is not often the case), the surplus can be established for field crop systems with a 10–20% degree of uncertainty. Uncertainty over surpluses is hence much lower for field crop systems than it is for livestock systems (inputs, composition, time scale and spatial distribution of inputs), where it is estimated at 30–50%.

Methods for rational phosphorus fertilizer use are based on empirical models drawing on long-term trials and with low mechanistic content. Given the variability in soil analysis methods from one country to another, this range of
models results in a 1-3-fold variability in recommended dose for a given situation. The mineral content of harvests in a parameter used to calculate fertilization and also a source of uncertainty as it varies according to phosphorus availability in the soil. Fertilization has to give greater consideration to nitrogen and phosphorus. Nutrient cycles are more strongly coupled over time under grasslands than under crops where agricultural activity causes ruptures and decoupling. N:P ratios in harvests (which are a calculation base for element inventories) remain highly variable (from 20 for legumes and oil seeds to 1.5 for cereals).

It is crucial that fertilization management is optimized for a crop system in a given soil/climate context, with the aim of reducing nitrogen and phosphorus losses. However, this alone does not always suffice and changes to the crop systems are necessary. A balance approach, identifying emission systems or highly efficient systems and used together with the models to test scenarios, is useful when it comes to pinpointing changes in crop or production systems. We will discuss this below.

A move towards systemic changes on farms and in rural territories

The first approach is integrating eutrophication into other agricultural issues. For ecological and economic reasons, agriculture may aim to increase nutrient use efficiency and reduce dependency on inputs (nutrients and pesticides) in agricultural systems. The various approaches are based on agro-ecological principles and involve choices regarding the species and varieties grown, rotations that optimize nutrient use, alternating complementary crops over time and space and mixed crops, used to make better use of nutrient resources in the soil, fertilizer application adapted to local soil conditions, the development of inhibitors, green manure and crop residues, controlled nutrient release, growing legumes to fix atmospheric nitrogen, simplified tillage and permanent soil cover. The preference is for animal feed using local crop production, the recycling of nutrients from the livestock sector, the treatment of waste at wastewater plants that denitrify nitrogen and recover phosphorus, and the use of urban waste for soils and crops. Some of these solutions are based on agroecology principles and applied in organic farming systems. Yet precise nutrient management remains difficult in farming systems with very high organic matter inputs (livestock farming, vegetable production, etc.), even in organic farming. The second approach is integrating the eutrophication issue with other environmental challenges, within a context of the circular economy. Phosphorus has been declared as a limited resource by the European Union: it needs to be managed and recycled, if only from an economic viewpoint. In addition, against a background of global warming, energy consumption for the chemical synthesis of nitrogen and the transport of fertilizers and animal feed is a major concern at the current time. Likewise, the question of storing organic matter in the ground or the oceans, which implies the sequestration of atmospheric carbon, in synergy with nitrogen and phosphorus storage. The challenge is to bear in mind that the planet’s resources are limited and that climate change has to be mitigated, which is likely to encourage more holistic approaches as part of the ecological transition of agriculture and the waste treatment sectors. It is also a matter of repositioning humankind within the ecosystem, taking account of diets, the way in which human activity affects production patterns and the resulting flows.

5.2.3. Land use: aiding nutrient retention

Two concepts have been introduced for landscape management:

1) the concept of critical source area (CSA) is applied to an area with high, easy-to-harness nutrient stocks, where the permanent or temporary hydrological connection with the river system makes it the main source of pollutant losses. For particulate phosphorus, this area may be slopes along a watercourse or, for nitrate, drained plots where the outlets feed into watercourses. The operational purpose of this concept is to make a non-point source more akin to a point source, to facilitate action by concentrating it within space.

2) The concept of buffer zone applies to areas that are often non-productive or not used for farming, located between agricultural plots and the river system and the physical or biogeochemical properties of which are conducive to the trapping or conversion of nutrients. For particulate phosphorus, this can be grassy strips which aid
particulate deposits, for example, or for nitrates, riparian wetlands/woodlands that convert nitrate into biomass (plant uptakes) or gas (denitrification). The operational value of this concept is to design landscape structures that mitigate emissions from plots.

These concepts gave rise to the idea of supporting landscape retention capacities through landscape design. Countries in North America and parts of Northern Europe have developed models and risks indices based on this concept, especially for phosphorus. These indices integrate soil phosphorus availability, elements from the phosphorus inventories, factors in hydrological connectivity and transport (erosion, run-off, subsurface flows). In addition to the tools used to manage fertilization at plot level, we have other tools for landscape-level management. Transpositions have been applied in France (the Territ’eau project). These concepts cover some very different options: (1) protecting and enabling landscape structures and diversity (crop rotation, soil use), (2) recreating mounds and embankments, grassy strips or semi-natural buffer zones, (3) creating artificial wetlands, and (4) encouraging diversification in crop rotation and land use.

**Conserving and promoting structures and landscape diversity** should be a priority for eutrophication prevention policies since it has been proven that the simplification of landscapes systematically leads to reduced water quality. Although the benefits of creating structures that aid infiltration and reduce run-off (embankments, hedges or grassy strips on hillsides, perpendicular to the slope) have been clearly demonstrated for phosphorus, it appears to be quite limited for nitrogen. Conversely, the introduction of downslope features (grassy strips along streams or riparian buffer zones) has produced more moderate results for phosphorus, especially over the long term (tendency to saturate, creation of fast-flowing channels, etc.), while their benefits for nitrogen, serving as denitrification areas, have been clearly proven. Nonetheless, this benefit should not be overemphasized given that the share of nitric flows actually intercepted is very variable, standing at around 30% in the best cases. In addition, soil phosphorus management can be adapted (e.g. mowing meadows to enable phosphorus export). This potential reduction should also be put into perspective given the higher risk of pollution transfer, either to the air (N₂O during nitrification and denitrification) or to water (NH₄⁺, dissolved organic nitrogen), and dissolved phosphorus. Finally, these actions can be difficult to implement due to land management conditions in the areas in question and given the difficulty in obtaining reliable estimates of the resulting benefits. While these measures can be useful and should be encouraged, soil monitoring is important whenever there are substantial nutrient inputs. We should also mention the measures taken to protect watercourses from agricultural activities and direct livestock watering (fences), which are simple, inexpensive ways of reducing this significant source of phosphorus and particles, bacteria, drug products, ammoniacal and nitrogen, etc.

If well designed, the formation of artificial wetlands, which are areas that have been made wet or flooded intentionally during hydraulic operations (hollowing, installation of sills or dikes, capture in ditches, etc.) can be an effective way of reducing nitrate content in water, especially where water is rich. Given that substantial work is required to implement and maintain these systems, this solution is limited to one-off or occasional cases of pollution or high-risk situations. However, some countries, such as Sweden and Denmark, have introduced large-scale restoration programmes.

**Afforestation** (planting forest) and **long-term fallow** (‘diluting’ surfaces) are clearly efficient ways of reducing agricultural pollution, especially when they are used within catchment areas. Less radical landscape diversification measures (increasing permanently covered areas, longer rotations, increasing the number of varieties cultivated in crop rotation, etc.) can also be beneficial, although there have been few attempts to quantify them so far. Their effectiveness has however been demonstrated, particularly where they are combined with a rational approach to space (e.g. preferential location of extensive grasslands or specific crop itineraries in the bottom of valleys).

Knowledge of the processes at landscape level can provide very valuable input when designing preventive or remedial actions to tackle nutrient emissions in the environment. It completes action taken closer to the source (plot level). The main difficulties are, on the one hand, their very variable nature dependent on local context and, on the other hand, pollution transfer issues (between nutrients (N vs. P), between compartments (atmosphere-soil-water) or between chemical species (soluble vs. particulate P, NO₃⁻ vs. NH₄⁺ vs organic N, etc.).
5.3. Economic approaches to remediation

5.3.1. Assessing the impacts of eutrophication

Remedial trajectories have not been widely studied retrospectively to establish the costs engendered. The direct and indirect costs of environmental policies are actually quite complex to define and calculate. There are many difficulties: the increase in environmental constraints can have various effects on the competitiveness of agricultural sectors depending on country, even leading to relocation of agricultural production to countries with less stringent environmental standards; while it is relatively easy to identify the impacts of agriculture, it is much harder to assess the benefits (food production, biodiversity, cultural heritage, etc.); open spaces are often seen as acquired and farmers do not benefit from these externalities; costs are likely to change over time, with changes in technology and consumer or user demand; pollution often has several causes and the choice to control or assign it to one or several causes simultaneously or alternately is far from a neutral decision when it comes to economic results (taking action against nitrogen and/or phosphorus in agriculture, acting preferentially or simultaneously in the domestic or agricultural sector, or arbitrating between point and non-point pollutions).

Attempts to assess the impacts of eutrophication on continental, coastal and marine waters in monetary terms have been made over the last ten years. They cover four aspects: (1) impacts on water quality, the environment and climate change, (2) impacts on human health and well-being, (3) impact on business activities, and (4) how to reintegrate eutrophication into a circular approach. These studies indicate that it is not possible to attain an overall value for all the impacts linked to eutrophication.

The work done to identify partial assessments raises some questions and establishes certain orders of magnitude of the monetary impacts. These studies were mainly run in the United States and for the Baltic Sea. Very few economic assessment studies have been conducted in France. For continental waters, they mainly focused on the impacts in terms of environmental quality, health and well-being, and some of the impacts on economic activities. The economic assessment of environmental damage was estimated up to US$44 million per year in the United States for the prevention of aquatic biodiversity loss related to eutrophication, and up to US$2.2 billion per year in the United States for the potential loss of recreational uses. For coastal waters, the studies primarily covered the impacts of eutrophication on human health, economic activities and, to a lesser degree, the cost of collecting, treating and making use of algae. The incidence of disease from marine pathogens affecting human health varies from 1 to 1,000 depending on the scope of the study. The effects of eutrophication on economic activity varies from US$2–20.4 million per year in the United States. For marine waters, the only study identified looked at the Baltic Sea and deals with the fall in water quality only. The estimated economic impact varies, depending on the reduced nutrient input scenarios, from €25–54 million.

This work reveals the complexity of such an ambition and the fact that there is so little literature on the subject in France and Europe. However, the orders of magnitude show that these impacts are considerable and that they deserve to be brought to the attention to all stakeholders and users to spur action.

Over the past ten years or so, assessment approaches have endeavoured to identify the impacts of eutrophication, identify the ecosystem services affected in each environmental compartment (continental, coastal and marine waters) and the conventions applied to estimate the value of the environment from a socio-economic perspective. Approaches have been put forward to provide information—mainly monetary data—concerning the damage and benefits associated with eutrophication. In many papers, the authors point out that the costs calculated are still underestimated as they cannot include all the elements contributing to the impacts of eutrophication.

5.3.2. Different economic strategies for remediation

From an economic point of view, tackling eutrophication implies a compromise between two extreme situations: a situation where all signs of eutrophication are eliminated but that could induce very high costs for society (lack of economic activities, fall and variability of agricultural production, high cost for households), and a situation...
where eutrophication is accepted with no attempt to improve it, but that would be harmful for the environment, the economy and public health. To reach this compromise, cost/benefit or cost/effectiveness analyses of remedial actions can be carried out. For this purpose, we need to take into account 1) the constraints imposed on each party (fluctuating agricultural markets, tax burden, etc.), 2) the costs of environmental policies which are quite complex to calculate, and 3) all the benefits.

One important element arising from our review of the literature is that it is inefficient to take a two-phase approach, first estimating a target objective to limit eutrophication and then searching for the best way to reach it in economic terms. The economic aspect has to be taken into account from the outset. The literature shows that integrating economic, biogeochemical and ecological aspects at the start of the process allows complex phenomena to be taken into consideration as they arise, for example trajectory changes, irreversibility, or multiple and interconnected spatial and time scales.

Various economic leverages can be used, such as i) command and control systems, ii) taxation and/or subsidies, iii) ambient taxes, and iv) tradeable permits. These instruments can be coupled. The degree of irreversibility of the measures envisaged should also be taken into consideration.

Solutions that would merely imply defining an environmental standard (a pollution rate) or a minimum decrease in pollution levels are not usually effective, as it has been demonstrated in studies on the Baltic Sea or the Danube. In addition, sources of eutrophication are multiple and taking this into account can help reduce the cost of actions. Moreover, an excess of nutrients may have varied effects. Including then different sources and effects in the analysis could enhance a comprehensive approach to remediation.

Fighting non-point pollutions (from households or agriculture) is complicated because of the difficulty to measure them, the often high transaction costs, ‘free rider’ behaviors (benefiting from other people’s efforts without making any yourself), and the issue of asymmetric information (the polluter knows the cost of reducing waste, but the regulator does not). In addition, the impression that you are making an effort while others are not can be a barrier to action.

Ecosystem disturbances are often multiple (eutrophication, acidification, greenhouse effect, etc.) and effectiveness can be improved if this is taken into account. Likewise, if the multiple benefits generated by diminishing pollution are considered, solutions that would be barely profitable otherwise may be envisaged for reducing eutrophication.

In addition, the order in which one need to tackle a problem with multiple causes must consider the risks inherent to each source of pollution, the correlations between those risks, and the specific benefit of reducing each source.

5.3.3. Economic intervention methods for remedial action

In multi-causal situations, the decision to tackle one cause or several simultaneously or alternately influences the economic results, as several studies have shown. This applies to the decision to take action on nitrogen and/or phosphorus in farming, or to act on the domestic and/or agricultural sectors individually or simultaneously, or to deal with point and/or non-point source pollutions.

In a context where there is inevitably a degree of uncertainty and gradual knowledge acquisition, the solutions implemented may seek to strike a balance between performance and robustness. Adaptive management (updating objectives, tools and parameters on the basis of experiences) appears to be the best approach in this respect. Likewise, it is preferable to review action plans periodically. Generally speaking, the fluctuating character of pollution from one year to another and from one season to another, as illustrated, for example in wetlands’ effectiveness to abate pollutant concentrations, makes it necessary to take these temporal variations into account when determining the economic value of actions.

When it comes to non-point source pollution of agricultural origin, the main tools presented and discussed in the literature are taxes on fertilizers, or taxes on agro-environmental balances such as net (outflows minus inflows) phosphorus exports, non-linear taxes or subsidies, and more direct incentives to change crops or growing/production systems. One problem is avoiding ‘deadweight effects’, that can reduce the effectiveness of actions. In terms of application, the search for efficiency generally raises three questions: Who do we need to target? What should be targeted? What mechanism needs to be used?
Who do we target?

Targeting all polluters equally may appear to be the most immediate and easiest solution, for example by taxing them all in the same way. However, one cannot tackle household and agricultural pollutions using the same instruments. For agricultural sources, standardized measures can incur disproportionate costs because certain parties will not survive their implementation. In addition, in large areas, inspection costs will be high. Finally, it can be difficult to gain acceptance for the standardized application of instruments when each party contributes differently to pollution. For non-point source pollution, it is theoretically possible to target polluters differently, according to their environmental impacts and cost structures, but this strategy is hampered by the fact that individual contributions remain largely unknown. Putting transaction costs apart, case studies show that targeted strategies outdo undifferentiated strategies, often with some very big differences. Given that no single policy is the ideal solution for all farms, reduction policies can be defined by soil type, farm type or by geographic location.

What to target?

In the agricultural world, tax schemes based on inputs (fertilizers, manure, etc.) are usually insufficient because these inputs can be interchanged (if they are not all subject to restrictions). Nonetheless, this kind of taxation has other benefits. For example, a tax on nitrogen fertilizers is easy to set up and incurs low transaction costs. As long as this scheme is combined with other instruments, it can be useful, even though considerable differences have been observed in responses, varying according to type of nitrogen fertilizer and manure, and application method. River water quality is usually more relevant as an objective than input reduction. The value of a policy targeting either nitrogen or phosphorus as a priority, or both at the same time, will thus depend on context and specific situations. Policies restricted to a single fertilizer element are not generally very effective for limiting non-point source pollution. Targeting nitrogen and phosphorus together can therefore help reduce costs and improve the effectiveness of nutrient reduction measures.

What mechanisms to use?

The high level of variability and uncertainty regarding emissions can lead to the simultaneous introduction of several instruments, which will be effective within a variation range or in given circumstances. The choice of the instrument or set of instruments can significantly influence the result of the cost/effectiveness analysis of the agro-environmental policy. The same can be said of other parameters e.g. tax rates or subsidy levels. The effects on production costs and the survival of farms cannot be neglected. To alleviate some of these effects, the various instruments must be carefully coupled, for example subsidies may encourage environmentally-friendly habits, thus improving the effectiveness of policies by offsetting taxes with incentives to change practices and agricultural systems. Subsidies to encourage the introduction of measures can also be linked to the results obtained, as long as they are measurable and the measurement directly linked to the action (e.g. nitrogen residues at the end of winter, instead of monitoring concentrations in a watershed). Regulation (norms and standards) is another way of reducing point and non-point pollution (reducing excess nutrients during application, such as N and P for field crops). However, in agriculture, environmental regulations need to be harmonized multilaterally from one country to another because of the effects on farm competitiveness. Empirical studies show that it is more beneficial to base regulations on inputs, which are the most closely correlated with farm emissions. For example, the livestock stocking rate per hectare can be taken as a basis for regulation as it is strongly correlated with pollutant load. Explicit consideration of the diversity of farms can lead to a differentiated policy regarding regulation, although this can raise the issue of equality between farms. Market-based methods (e.g. tradeable permits) are theoretically more effective than the instruments that draw on regulations and restrictions. The efficiency of the environmental permit markets can vary widely, however, depending on the variety of pollution sources, pollutant activities and environments. In addition, the setting of a price for these permits is a key to the success of this instrument and for the structures that coordinate it (cooperatives and the like). When introducing a set of measures to reduce pollution sources, benefits must be compared to costs, even though some fundamental questions on the cost/benefit or cost/effectiveness analysis method remain, particularly with regard to the choice of the discounting rate, risk consideration, uncertainties, and spatial
and temporal variabilities (linked to the climate, for example). It is thus often recommended to keep transaction costs in mind to prevent distortions when evaluating costs and calculating cost/effectiveness ratios, and to devise policies that could reduce them, especially in a context of budget restrictions and a growing aversion for bureaucracy. Meanwhile, other works recommend the integration of opportunity costs instead, as they include direct and indirect, and private and public costs. To conclude, models that combine biophysical and economic aspects should be used to define reduction programmes, despite the uncertainties that arise from the incomplete knowledge of biophysical and ecological phenomena. Adaptive management that updates objectives, tools and parameters on the basis of experiences can be a pragmatic and effective solution. Far too often, exceedingly ambitious, and then inapplicable objectives result in programmes that are not cost-effective at all (ineffective because not introduced on the desired scale, and costly because introduced nonetheless). This does not mean that objectives should remain modest, but that they should be just attainable.

The comparative value of modest measures over wide geographic areas, or more ambitious measures within more restricted areas remains open for discussion. Nonetheless, spatially targeted instruments usually provide better results, and that raises the question of zoning and the scale on which to work. In some situations, ecological engineering solutions (buffer zones or wetlands) can be used as ways of limiting the effects of pollution over the short and long term, or of mitigating them, when applied alongside other measures. However, there are questions about the long-term effectiveness of these solutions with regard to delayed pollution from these zones.

In short, it is very unlikely that a solution that has proven its worth in one specific context can be reproduced to solve the same problems in another context. On the other hand, lessons can be learned from successes and failures in a variety of situations. There is no ideal solution but a range of policies that can help reduce pollution sources when targeted and designed for specific situations with instruments that are often developed for that particular case, and once the problems have been first correctly identified, analyzed and the various solutions have been assessed.

5.4. Modelling eutrophication: representations, understanding and accompanying action

5.4.1. Characteristics of eutrophicated ecosystem models

The eutrophication of fresh and marine waters is caused by terrigenous nutrient inputs. The modelling of eutrophication therefore starts on watersheds and continues in the aquatic ecosystems, which differ in terms of residence times and the dominant retention and transfer processes, and down to the coastal area. Generally, models start by marking out a clear land-sea continuum area and setting a question, then explaining the hypotheses that only integrate the processes deemed a priority for the question. Because there are multiple mechanisms and we only have fragmented knowledge of some of them, there is no real consensus on the degree of detail required to ensure effective representation of the processes, which are often formulated using simplifying assumptions.

Models were first developed in the 1970s for lakes, then later for watercourses, reflecting the order in which eutrophication phenomena appeared, with the aim of assessing the necessary reduction in nutrient inputs. These very reductionist models were gradually completed with more complex examples, explicitly describing the mechanics of the processes at work. In the 1990s, modelling also encompassed estuaries and the coastal sea. Today, we have two main types of tool which complement one another.

Statistical models seek to predict a target variable according to a number of causal variables measured in the field. These models benefit from semi-automatic aids (significance tests) that eliminate superfluous input variables. These models are not particularly generic but their calibration could be improved for a wider diversity of sites by enhancing the databases. We can combine them with models that call on fuzzy logic, neural networks, or Bayesian modelling.

Mechanistic (or deterministic) models use a more or less sophisticated theory based on the hydro-biogeochemical
and ecological mechanisms that lead to eutrophication, so these models should be more generic that the simple statistical approach. The equations quantify changes in the descriptors relevant to the ecosystem (called ‘state variables’) over time (ordinary differential equations) or over space if the heterogeneity of the ecosystem requires (partial differential equations).

Once the model structure has been selected, we need to set a number of parameters to reproduce the facts observed (calibration) and, if possible, validate the model using an independent data set. The model can then be used for operational simulations (the effects of nutrient input scenarios or climate change).

The choice of spatial representation of ecosystems (area covered, resolution, number of dimensions taken into account, etc.) will be based on an analysis of the spatial heterogeneity of the area in question. If we can reasonably consider that the eutrophic ecosystem comprises a single homogeneous body of water, we will not include the spatial dimension (hence the name 0D model). If there is only one main element of heterogeneity (horizontal for a river or small, well-mixed estuary, vertical for a deep lake or fjord), we will include one spatial dimension to give a 1 DH (horizontal) or 1 DV (vertical) model. River networks are modelled by using the 1DH model of the main river, together with as many 1DH models as there are tributaries. If we need to consider two main elements of heterogeneity (horizontal for a large bay or shallow lake, vertical for a stratified estuary), we will use 2DH and 2DV models. Finally, for complex ecosystems with a high degree of heterogeneity over the three dimensions of space (large lakes with several inlets and/or outlets, extended coastal areas with large river dilution plumes, etc.), we will need to represent the entire aquatic system in detail, using a three-dimensional or 3D model. The simulation will segment the system into elementary volumes (meshes) considered to be homogeneous for the ecosystem state variables. Fig. 5.1 shows some examples of these different possibilities for spatial modelling.

For the representation of the biogeochemical processes that lead to eutrophication, the models simulate all or part of the natural nitrogen (N), phosphorus (P) and silicon (Si) (for single-celled diatoms) cycles using the number of biogeochemical compartments deemed necessary and sufficient by the modeller (dissolved nutrients, algal biomass, grazers, litter, etc.), completed where necessary by compartments in the soils or sediments subject to erosion/deposits (Fig. 5.2). As one of the primary consequences of eutrophication for an ecosystem is the depletion of dissolved oxygen in the water, some models expressly introduce the main oxidation-reduction reactions (redox) that control dissolved oxygen content. Any process that is not explicitly described by an equation can be seen as implicitly summarized in the value of a parameter in the model’s equations, which is why it is necessary—and tricky—to calibrate the model, usually manually, on a trial and error basis. Variables that act on the system but are not modified by it in return are seen as “forcing” the model, which is represented either by a series of spatio-temporal measurements or by simple functions (constants, periodic time functions, etc.).
5.4.2. The main models used for the different environments

Watersheds

A model showing nutrient transfers in a watershed will take the type of land use and deduce the resulting nutrient concentration (or incoming flow) in the river system.

For watersheds with deep groundwater where field crops are grown, a simplified approach involves simulating water flows at an annual scale (AnnAGNPS, CREAMS and HYPE models) and using either a simplified agronomic balance of inputs and outputs, or statistical relationships between crop types and nutrient losses in the river system. Land use is described simply (forest, arable land, grasslands, built-up areas) and agricultural practices represented by average distribution. These models do not detail the soil/plant system and use monthly or annual time intervals, so they do not reproduce rapid variations in observed concentrations or erosion and absorption/desorption processes affecting phosphorus particles. These models can be used to simulate changes in the agronomic balance and thus the changes observed or simulated upstream of the model, but they cannot account for evolutions because they are limited to balanced situations.

In small watersheds, it is possible to take into account soil/plant/atmosphere/groundwater interactions using mechanistic models (ANSWER, EUROSEM, KINEROS, MEDALUS, RHEM, RUNOFF, SHETRAN, SWAT, INCA, HSPF,
AGNPS, Modcou-STICS, CAWAQS, TNT2) that, at the scale of a rainfall event or season, assess plant uptake, organic matter mineralization, nitrification and denitrification, adsorption/desorption of the solid phase of phosphorus, and transfers (Fig. 5.3) through run-off and erosion. To simulate the seasonal dynamics of nitrogen flows, controlled by the surface flow rate and exchanges with groundwater, the models usually distinguish between over-ground run-off and groundwater flows, or dynamically variable saturated and non-saturated domains. For phosphorus, which is massively adsorbed on soil particles that are transported during heavy rain, a proper simulation of high water peaks is needed for a realistic assessment of flows. Spatialization makes it possible to simulate downstream capture of the nutrients mobilized further upstream in the watershed. These very detailed models use a one-day time scale and are able to test more hypotheses and scenarios than the models described above. However, they are very time-consuming in terms of input database development, calculation and exploitation of results.

Streams and rivers

In reference to the Strahler stream order based on the number of confluences between the source and the segment in question, the modelling of water flow in the river system began with straight-line compartments, then branched 1DH or 2DH models to represent expansion zones (MIKE21) and full 3D models in some cases (MIKE31 and Delft3D). Initially looking simply at ‘water quality’ represented by the two state variables ‘biological demand for oxygen’ and ‘dissolved oxygen’, these models subsequently included ecological processes to study a watercourse, particularly the models used by the US-EPA (QUAL2EU, QUASAR/INCA, PROSE and KALITO); they can also look at the river system across a watershed (WASP, BASINS, EFDC, RIVERSTRAHLER, RIVE, PEGASE, AGIRE). These models use a dozen or more state variables to describe the N, P and O cycles or to simulate the different classes of algae, zooplankton, invertebrates, plants and fish. In addition, a sedimentary compartment is often present. Most river eutrophication models focus on the processes in the water column, and very few of them take a detailed look at the role of the compartments adjacent to the main channel and floodplain, the hyporheic zone and the water table. The models provide flow rates and concentrations at time scales ranging from ten days to one hour.

Lakes and reservoirs

Given their low flow velocity and the resulting confinement, lakes are particularly exposed and, where they are deep enough, are subject to thermal stratification in summer, which puts a stop to exchanges between the
bottom (hypolimnion) and the surface (epilimnion). Between 1975 and 1985, simple deterministic modelling by the OECD, applied to North American, Western Europe and Australia, was used to determine the vulnerability of lakes and reservoirs to eutrophication, explaining the state of balance using combinations of the main morphometric (the lake's depth and volume) and hydrological (average flow rates, annual nutrient inputs to the watershed) characteristics. Later, detailed mechanistic models were developed to include variable inputs over several decades and water column stratification. Since the turn of the millennium, 3D models have frequently been used (e.g. ELCOM). They aim to describe the annual average state of eutrophication in a balanced lake. In the 1970-80s, the biogeochemical part of these mechanistic lake models simply took the total chlorophyll biomass of phytoplankton, its main limiting nutrient in fresh water – phosphorus – and a dependent variable: dissolved oxygen. Now, simulation covers the different phytoplankton groups, their capacity for specific positioning in the water column and their decomposition, in order to predict, over a one- to two-week period (Fig. 5.4), cyanobacteria algae proliferations harmful to certain uses of the lake (bathing, leisure, drinking water, etc.). These mechanistic models, used alongside improved statistical models that are increasingly calibrated on high-frequency automatic measurements, can quantify the level of nutrient (phosphorus firstly, then nitrogen) input reduction level required per watershed to reach the good environmental status set by the WFD for the lake, with or without taking climate change into consideration.

Estuaries

Apart from some very large estuaries (Yangtze, Saint Laurent, Escaut, Seine, Tagus or the ‘super-estuary’ of Chesapeake Bay), it is mainly the smaller eutrophic estuaries that have been modelled or compared in American, Finnish and Irish studies. Due to their high turbidity and the often strong currents in their deep central channel, estuaries do not appear to be the most likely sites to show signs of substantial algal proliferation. However, downstream of the turbidity maximum, haline stratification can create a rather shallow, quite well-lit nutrient-rich surface layer. Estuary models therefore focus on the role of physics (flow rate, salt water intrusion and turbidity maximum) on phytoplankton proliferation and bottom hypoxia that can be caused by sedimentation of this biomass. Estuaries can be covered using 1DH models if they are well mixed, but they require two-dimensional vertical models (2DV) or 3D models (ELCOM, COHERENS, MOHID, ROMS, SHOC, SiAM3D, etc.) where there is clear salt water intrusion. The biogeochemical modelling of estuaries is complicated because of the transition between fresh water and salt water ecosystems: phosphorus switches from dissolved to particulate state, while the descending fresh water phytoplankton and the ascending marine phytoplankton die off, generating organic matter that is highly mineralized by bacteria, and phytoplankton production gradually switches from a phosphorus-limited condition in fresh water to a nitrogen-limited condi-

![Figure 5.4. Ten-day forecast of the evolution of cyanobacterial bloom observed in August 2015 at the western end of Lake Erie using a 2DH model and a 3D model, and compared with the satellite observation for the forecast day. Source: Rowe et al. 2016. © American Geophysical Union.](image)
tion in marine water. Estuaries are often shellfish farming sites (oyster and mussel beds) which need to be protected from summer mortality episodes caused by bottom anoxia. While certain statistical models use a series of causal variable measurements (stratification, terrigenous N and P inputs, percentage agricultural land use in the watershed, etc.) to predict variables that are characteristic of eutrophication (chlorophyll, primary production, bottom hypoxia), mechanistic models simulate the nitrogen and phosphorus cycle with fresh water and marine phytoplankton to better simulate the transition from fresh to marine water. Frequent simulation of dissolved oxygen generally makes use of a forced flow of consumption by sediment and, much more rarely, of explicit modelling of redox in sediment.

Coastal lagoons

Because they are shallow, contained and subject to long water residence times, lagoons are particularly vulnerable to eutrophication, which is as harmful to biodiversity as it is to fish farming. Three sites have been modelled more than any other since the 1990s: the Thau lagoon on the French Languedoc coast, and the lagoons in Venice and Orbetello in Italy. Some simulations have been run over several decades because of the low renewal rate of the sedimentary compartments. Even today, some authors do not believe that the spatial heterogeneity of lagoons is fundamental, and make do with 0D models. However, since the 2000s, half of all studies have used 2D or 3D hydrodynamic models, sometimes adding a layer representing sediment below the water layer(s). In the rare cases of low salinity lagoons, some authors have used the P cycle alone, distinguishing between cyanobacteria and other algae, but in the more common cases of brackish or salty marine lagoons, the models simulate the N cycle, alone or alongside P. In lagoons that experience summer anoxia, the models include redox in water and sometimes in sediment, so as to simulate sudden desorptions of phosphorus during anoxia.

Open coastal sea

Despite its very low natural nutrient content and high water renewal levels, the coastal sea can experience severe cases of eutrophication. Models have been used to reproduce the enrichment and vertical confinement of river dilution plumes and to understand that, even where there are strong tidal currents, some bays can be subject to confinement due to a lack of residual tidal flow, leading to eutrophication where there are moderate terrigenous inputs. Efforts to model marine eutrophication and anoxia have largely focused on five sites across the world: the Baltic Sea, the North Sea, the Gulf of Mexico, Chesapeake Bay and the Black Sea. The China seas, the Sea of Japan and the northern Adriatic have also been modelled frequently. The marine models are used to understand the role of various factors triggering eutrophication and to conceive scenarios for restoration. In the 1990s, they helped define remedial action to reduce the world’s large ‘dead zones’ at the lowest cost. Several studies of the English Channel/North Sea, the Baltic Sea and Chesapeake Bay have taken a chain of models of watersheds, river networks and the recipient coastal sea, sometimes forced by climate change scenarios. There are still some studies that make do with a statistical model or a 0D/1D mechanistic model today but, since the 1990s, there has been increasing use of mechanistic models using either compartments (sometimes with a double layer) interconnected or finely-meshed, Cartesian or curvilinear 3D models. The multiple sources of nutrients flowing into the marine environment (inputs from different sized rivers, ocean and atmospheric inputs, flows from sediments) prompted the digital marking of a chemical element in order to track it in the 3D trophic network from its source. The biogeochemical aspect of marine models (ERSEM, MIRO, ECOMARS) almost always simulates the nitrogen cycle (N being considered the primary limiting element in sea water), but often also looks at phosphorus and silicon, though rarely the carbon cycle. When dissolved oxygen is the target descriptor of the model, the oxidizing/reducing couples of iron, manganese and sulphur can be simulated in water and sediment. Because they are potentially harmful, certain algal groups (dinoflagellates, Phaeocystis flagellates, Ulvaceae macroalgae) define the specificity of many marine models, in the same way cyanobacteria define the fresh water models. Models have sometimes been used to develop eutrophication risk indices (e.g. EUTRISK).
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<td>Anthropogenic (landscape, crop system, climate)</td>
<td>WS green algae. Scenarios for landscape/crop evolution</td>
<td>Detailed data on crop systems</td>
</tr>
<tr>
<td></td>
<td>STICS-MODCOU</td>
<td>10 to 10,000 km² Daily</td>
<td>Deep water table. Groundwater transfer.</td>
<td>Biotransformation in soils (OM)</td>
<td>N. Seine WS context. N cycle and agricultural systems</td>
<td>Anthropogenic (landscape, crop system) and climatic</td>
<td>Seine WS Scenarios for evolution of the crop system.</td>
<td>Detailed data on water tables and crop systems</td>
</tr>
<tr>
<td></td>
<td>SWAT</td>
<td>10 to 10,000 km² Daily</td>
<td>Notion of hydrologic response unit (HRU)</td>
<td>Semi-empirical functions</td>
<td>N and P. Large international community. Platform. Modularity.</td>
<td>Anthropogenic and climatic per HRU</td>
<td>BMP (best management practice) scenarios</td>
<td>No. parameters Flaws in the predictions (setting)</td>
</tr>
<tr>
<td></td>
<td>INCA</td>
<td>10 to 10,000 km² Daily</td>
<td>Notion of ecosystem (including agricultural ecosystems).</td>
<td>Semi-empirical function</td>
<td>N and P. Above all devoted to ecosystems with little human intervention.</td>
<td>Anthropogenic and climatic by HRU</td>
<td>Scenarios on ecosystem distribution (including agricultural ecosystems)</td>
<td>No. parameters Flaws in the predictions (setting)</td>
</tr>
<tr>
<td></td>
<td>MONERIS</td>
<td>500 to 500,000 km² Inter-annual</td>
<td>Conceptual model with 3-4 compartments</td>
<td>Surplus N &amp; P Transfer/retenion function (groundwater)</td>
<td>N and P. Si. WFD reporting</td>
<td>Anthropogenic (surplus)</td>
<td>Europe, European WS (e.g. Danube) Export estimates</td>
<td>Simplistic scenario test (surplus)</td>
</tr>
<tr>
<td></td>
<td>NUTTING</td>
<td>500 to 5,000 km² Inter-annual</td>
<td>Conceptual model with two compartments Close to SPARROW and GREEN</td>
<td>Surplus N &amp; P Transfer/retenion function (surface)</td>
<td>N and P. WFD reporting</td>
<td>Anthropogenic (surplus)</td>
<td>France WFD water bodies. N and P export calculation. Eutrophic risk</td>
<td>Simplistic scenario test (surplus)</td>
</tr>
<tr>
<td>LAKES AND RESERVOIRS</td>
<td>OCDE</td>
<td>Lakes</td>
<td>0D morphometry</td>
<td>P and chlorophyll</td>
<td>Steady-state empirical statistical models</td>
<td>P inputs</td>
<td>Calculating maximum permitted P inputs</td>
<td>No transitional state, no algal groups</td>
</tr>
<tr>
<td></td>
<td>DYRESM-CAEDYM</td>
<td>Lakes</td>
<td>1D</td>
<td>N, R, Si and oxygen cycles</td>
<td>Different phytoplankton groups, including cyanobacteria</td>
<td>Inputs to the limits + weather</td>
<td>P reduction scenarios</td>
<td>Paid software</td>
</tr>
<tr>
<td></td>
<td>CE-QUAL W2</td>
<td>Lakes and stratified estuaries</td>
<td>Cartesian 2D with possible coupling of lake over estuary with branching</td>
<td>N, R BOD and oxygen cycles, pH</td>
<td>N phyto, zoo and macrophyte groups to be defined, sulphur and methane, hypoxic flow</td>
<td>ditto</td>
<td>Eutrophication in stratified water</td>
<td>Paid software</td>
</tr>
<tr>
<td>Environment</td>
<td>Name</td>
<td>Spatio-temporal scale</td>
<td>Physics</td>
<td>Biogeochemistry</td>
<td>Characteristics</td>
<td>Forcing factors</td>
<td>Main applications</td>
<td>Drawbacks</td>
</tr>
<tr>
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<td>RIVERS</td>
<td>QUAL2EU</td>
<td>River network</td>
<td>Branched 1DH for a regular regime</td>
<td>N, P, BOD and oxygen cycles</td>
<td>Varied sensitivity study tools</td>
<td>Water and substance inputs in each mesh + light level</td>
<td>Optimized point and non-point discharge</td>
<td>No vertical dimension, no intermediate regime</td>
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<tr>
<td></td>
<td>EFDC-WASP</td>
<td>Rivers and estuaries</td>
<td>1DH to 3D curvilinear</td>
<td>N, P, BOD and oxygen cycles, metals,</td>
<td>Simulation of sedimentary diagenesis</td>
<td>ditto</td>
<td>Input reduction scenarios</td>
<td></td>
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<tr>
<td></td>
<td>MIKE HYDRO river</td>
<td>Watercourses</td>
<td>1DH</td>
<td>C, N, P and oxygen cycles</td>
<td>User-friendly interface</td>
<td>ditto</td>
<td>Discharge modification scenarios</td>
<td>Paid software, little description and rigid</td>
</tr>
<tr>
<td></td>
<td>PEGASE</td>
<td>Watersheds and river network</td>
<td>Branched 1DH</td>
<td>N, P, BOD and oxygen cycles</td>
<td>Coupled with a GIS, with user-friendly interface</td>
<td>ditto</td>
<td>Point and non-point discharge modification scenarios</td>
<td>No groundwater or sediment</td>
</tr>
<tr>
<td></td>
<td>RIVERSTHRALER</td>
<td>River network, evolution of the eutrophication of big rivers over several ten-day periods</td>
<td>Branched 1DH</td>
<td>C, N, P, Si and oxygen cycles</td>
<td>Diatoms, chlorophylla, cyanobacteria; coupled with a GIS (pyNuts platform)</td>
<td>ditto + ten-daily measurements of flow rate at several points</td>
<td>Point and non-point discharge reduction scenarios</td>
<td>No intermediate states below the ten-day period, no groundwater</td>
</tr>
<tr>
<td></td>
<td>PROSE-RIVE</td>
<td>Watercourses</td>
<td>Non-branched 1DH</td>
<td>N, P, BOD, bacteria and oxygen cycles</td>
<td>Various types of detritic matter</td>
<td>ditto</td>
<td>Effects of discharge modifications on O2</td>
<td>No river network, no groundwater, no sediment</td>
</tr>
<tr>
<td>ESTUARIES, LAGOONS AND COASTAL SEA</td>
<td>MIKE 3 ECO</td>
<td>coastal</td>
<td>Cartesian 3D</td>
<td>C, N, P and oxygen cycles</td>
<td>User-friendly interface</td>
<td>Tide at the boundaries + weather + river inputs</td>
<td>Discharge optimization</td>
<td>Paid software, little description and rigid</td>
</tr>
<tr>
<td></td>
<td>DELWAQ</td>
<td>coastal</td>
<td>3D curvilinear volume</td>
<td>N, P, Si and oxygen cycles</td>
<td>N phyto and macrophyte groups, pH, trophic network</td>
<td>ditto</td>
<td>Lagoon and coastal eutrophication</td>
<td>Paid software</td>
</tr>
<tr>
<td></td>
<td>MIRO&amp;CO</td>
<td>coastal</td>
<td>Cartesian 3D</td>
<td>N, P and Si cycles</td>
<td>Phaeocystis phyto; N tracing</td>
<td>Ditto + atmospheric N inputs</td>
<td>Phaeocystis blooms, WFD scenarios</td>
<td>No sediment</td>
</tr>
<tr>
<td></td>
<td>ECOMARS3D</td>
<td>coastal</td>
<td>Cartesian 3D</td>
<td>N, P, Si and oxygen cycles</td>
<td>Ulvaceae, 3 toxic phyto-plankton, N and P tracing</td>
<td>Ditto + atmospheric N inputs</td>
<td>Green tides, WFD scenarios</td>
<td>No biochemistry in sediment</td>
</tr>
<tr>
<td></td>
<td>ERSEM</td>
<td>coastal</td>
<td>0D to 3D</td>
<td>C, N, P, Si and oxygen cycles</td>
<td>Overall quotas, three phyto groups, biogeochemistry in sediment</td>
<td>ditto</td>
<td>North Sea and Mediterranean eutrophication</td>
<td>Highly complex, difficult to calibrate</td>
</tr>
<tr>
<td></td>
<td>MOHID</td>
<td>Estuary and coast</td>
<td>Cartesian 3D</td>
<td>N, P, BOD and oxygen cycles</td>
<td>Calculating integrated flows per zone, coupling with watershed model</td>
<td>ditto</td>
<td>Estuaries and lagoons</td>
<td>Rudimentary ecology</td>
</tr>
<tr>
<td></td>
<td>WQMAP</td>
<td>coastal</td>
<td>3D curvilinear</td>
<td>N, P and oxygen cycles</td>
<td>Bathymetry on worldwide geographic information system</td>
<td>ditto</td>
<td>Industrial and urban discharge</td>
<td>No macrophytes, no trophic network</td>
</tr>
<tr>
<td></td>
<td>ECPATH-ECOSIM</td>
<td>Marine ecosystem</td>
<td>0D</td>
<td>Complex trophic network</td>
<td>Calculating indices of matter transfer within the network</td>
<td>Nutrient inputs, eco-physiological constraints</td>
<td>Response of species involved in eutrophication</td>
<td>No spatialized physics</td>
</tr>
</tbody>
</table>
5.4.3. Using the models

Controlling nitrogen and phosphorus nutrient inputs in aquatic environments

Given the reliable simulations of highly variable flow rates using the hydraulic component of the models, and the buffered concentration variations, the watershed models are able to produce realistic flows of dissolved N and P for various scenarios. However, the much more intermittent particulate flows remain imprecise and the simulation of future scenarios comes up against the uncertainty over anthropogenic pressure.

In the hydrographic system (streams and rivers) draining these watersheds, the models have reproduced the long-term evolution of nutrient flows under the effect of changes in human activity, making it possible to assess the range of variation in possible inputs. The PIREN-Seine research programme has replicated the evolution in pollutant and purifying activities in the Seine basin over several centuries, along with its impact on the level of eutrophication in the river. Modelling of the Danube revealed a striking example with a fall of 25% of nitrogen flows and 50% of phosphorus flows following the social changes in Central Europe in the 1990s, when industries and large livestock concerns were closed down and fertilizer consumption thus considerably reduced.

When the aim is to prevent the eutrophication of lakes and estuaries, where primary production is above all limited by access to light and the water residence time, the OECD’s approach has been used to estimate the maximum annual nitrogen or phosphorus flow that any balanced ecosystem can accept from the upstream watershed before exceeding an acceptable level of chlorophyll biomass. In the coastal sea, the issue is the respective role of the diverse nutrient sources; digital plotting is used to visualize the long-range action of certain big rivers (the Loire, Seine, Rhine, etc.), which sometimes cross borders, and to assess each one’s role in the eutrophication of Western European coastal areas, while showing the substantial (10–30%) proportion of atmospheric nitrogen deposits in offshore waters.

Controlling the plant biomass and toxic algae

In streams and rivers, residence time gets longer as we move down from the basin head (there may even be rapids in mountain areas) towards the lower reaches and the estuary. Models reproduce this growing risk of eutrophication from the source to the mouth and some of them show the competition between phytoplankton swept along by the current and the phytobenthos rooted in the bed. Contrary to phytoplankton, phytobenthos can grow in the faster-flowing waters of the more upstream sections of rivers, actually decelerating the current, which has been studied by certain models. At the end of the 20th century, given the high nutrient content of many rivers, the models rarely pointed to a limit on algal growth by P or N, with only Si limiting the growth of diatoms after their spring bloom, thus light and residence time remained the dominant control factors. Since the ban on phosphates in laundry detergents and widespread phosphorus removal at water treatment plants, several cases modelled show limitation by P which has partially reduced the very substantial proliferations visible in the middle reaches of rivers. Diatoms generally dominate in spring and autumn phytoplankton, while in summer, the fall in the flow rate and higher light levels aid the growth of non-siliceous algae, especially chlorophyta. To reduce eutrophication in streams and rivers in the strictest sense, the models therefore usually recommend reducing phosphorus inputs rather than nitrogen inputs. Modelling has also resulted in proposals for eutrophication criteria better adapted to rivers. While we can transfer the eutrophication criteria applicable to lakes (e.g. chlorophyll plankton biomass in the water column) to slow (lentic) river systems such as the Seine, they have to be adapted for faster-flowing (lotic) watercourses such as the upstream Garonne. Dynamic criteria such as primary production and respiration, along with their ratio, can be used to diagnose either the eutrophication of an autotrophic segment (Production:Respiration > 1) where production exceeds 1–2 mg C.m\(^{-2}\).j\(^{-1}\) or the organic pollution of a heterotrophic segment (Production:Respiration < 1) where respiration exceeds 1–2 mg C.m\(^{-2}\).j\(^{-1}\). In addition, if the ultimate goal is to reduce coastal eutrophication, the models can take the N:P:Si ratios of flows at the river mouth to establish risk indices (e.g. ICEP) for the development of toxic marine species (Pseudo-nitzschia diatoms or dinoflagellates in waters with a high N:Si ratio). Recent models therefore recommend a considerable reduction in nitrogen inputs in rivers because of this link with marine eutrophication.
In lakes, the models usually show that an increase in P inputs does not modify the diatom biomass significantly (as it is rapidly limited by the reduction in available light as a result of self-shading) but it stimulates cyanobacteria proliferation in summer. According to the models, the expected increase in water temperature linked to climate change is likely to bring forward spring diatom bloom, after which the epilimnion remains deficient in dissolved inorganic nitrogen for a longer period, which aids the proliferation of nitrogen-fixing cyanobacteria species, followed by a fall in herbivorous zooplankton because of the low nutritive value of cyanobacteria.

In estuaries, primary production is limited by light levels, because of high turbidity in the fresh water/sea water transition zone. However, many estuaries are also protected from plankton proliferations by low residence times. In fact, one model has been used to explore the effects of a partial deviation of the Yangtze River towards a faster-flowing channel, with the aim of reducing primary production and redirecting it towards less harmful species. Nonetheless, in 20 or so cases in Finland and around 15 in Canada, average chlorophyll levels or the oxygen deficit in the bottom water have been significantly correlated with the percentage of agricultural land use in the watershed. Several small estuary models have demonstrated that estuary eutrophication could be reduced with a fall in nitrogen river inputs, while others have shown that in large estuaries, P remains the controlling factor, as is the case in fresh water. The observation that virtually no estuary in the world nor any temperate coastal area (apart from the salty northern part of the Baltic Sea having a very low salinity) appears conducive to cyanobacteria proliferation, even though these species are able to overcome N limitation, poses a conundrum for the scientists, which two very simple 0D models have attempted to solve. Both these models show that at sea, cyanobacteria growth is too low to offset dispersion by the currents and predation by zooplankton and benthic filter-feeder invertebrates. This is partially due to the lower bioavailability of molybdenum and iron, which slows cyanobacteria growth to the point that they are no longer able to generate sufficient quantities of photosynthetic cells to supply energy to the cells that fix nitrogen gas.

In lagoons, models using increased nitrogen inputs do reproduce the successive replacement of eelgrass beds and small phytoplankton species with accumulations of green benthic macroalgae, then with proliferations of large phytoplankton species. They also show the transition from a phosphorus-limited state in spring to a nitrogen-limited state in summer. An oyster farming simulation in a lagoon creates a well of phytoplankton that drains nutrients well beyond the farming area, and generates a local source of recycled nutrients using organic bio-deposits that sediment on the seabed. Shellfish farms in lagoons therefore help fix locally at the bottom a portion of the nutrient inputs that come sporadically from watersheds. This benthic nutrient source is the cause of green macroalgae proliferation which, when they die, further increase the organic matter concentrated at the bottom.

In the coastal sea, green macroalgae proliferation has proven to be determined by the rich nitrogen content of coastal water. The aim of the models is to estimate the concentration thresholds that should not be exceeded at river outlets in order to contain ‘green tides’ at an acceptable level. At several other sites, the models suggest that a considerable reduction in nitrogen inputs could bring down flagellate (Phaeocystis) or dinoflagellate proliferation, and reduce the toxicity of Pseudo-nitzschia diatoms by lowering the N:Si ratio. Fishery resource management, using the ECOPATH steady-state trophic network model to simulate a range of animal compartments including fish, has shown that eutrophication can maintain the ecosystem in a productive but somewhat immature state, because the branched trophic network gives way to several small short-chain trophic networks.

Controlling dissolved oxygen in bottom water

In streams and rivers, until the mid-20th century, oxygen consumption was largely due to the decomposition of raw organic matter discharge from urban and industrial (e.g. tanneries) sources. The first stage in purifying this discharge led to release into rivers of the product of aerobic decomposition of the organic matter, i.e. for nitrogen, firstly ammonium then nitrate if the treatment procedure includes an aerobic phase. The models that simulate nitrification explain that this ‘clarified’ discharge was able to restore the river’s oxygenation immediately downstream from water treatment plants but that, along the river, slow bacterial nitrification of the ammonium discharged upstream transferred the anoxia problem, sometimes much further downstream close to the mouth (e.g. the Seine or Loire). Since the late 2000s, the systematic addition of a final nitrification/denitrification stage at water treat-
The simulation of ammonium and dissolved oxygen profiles over three successive periods shows the spectacular restoration of oxygenation all along the Seine, as and when its ammonium content is reduced.

In lakes, the models have been used to assess the quantitative effects to be expected from artificial re-aeration of the hypolimnion. In small reservoirs, water moves up from the hypolimnion in ascending bubbles which appear to destratify the water column locally and to introduce oxygen into the water through diffusion via the bubbles’ surface (diameter \( \approx 1 \) mm).

In estuaries, the anoxia risk comes from the isolation of a bottom layer that receives sedimenting detrital matter but that is unable to renew its dissolved oxygen from the atmosphere. Modelling has shown that making the estuary deeper by dredging can also turn a vertically well-mixed estuary into an estuary with salt water intrusion conducive to hypoxia if we do not reduce the quantity of detrital matter at the same time.

In lagoons, because of the very shallow water and the nearby sediment stocks, several models have focused on simulating the accumulation of organic matter in sediment, especially in oyster farming areas where bio-deposits are generated. The intense mineralization of these deposits triggers a nutritive flow towards the bottom water, which stimulates the proliferation of green macroalgae and high oxygen consumption in summer, which may cause anoxia that is fatal to fauna (the phenomenon known as malaigue). Integrating benthic nitrogen and phosphorus flows in the models is usually essential for a realistic simulation of the eutrophicated status of lagoons.

In both the statistical and mechanistic models, the hypoxic zones of coastal waters (in the Baltic Sea, Black Sea and Gulf of Mexico) were linked (in a non-linear manner) to growing nitrogen inputs from local rivers. This explains the appearance then the extension of the largest eutrophic and hypoxic zone (22,000 km²) in the world in the Gulf of Mexico in the 1970s. The more stratified the zone, the more the instantaneous respiration processes of phytoplankton and mineralizing bacteria in the dark part of the water column appear as processes that control this hypoxia, with the slow mineralization of the detritic sediment stock as an aggravating factor. The results of the various models can be used to complete the geographic coverage of the monitoring networks and to draw up a map of sea bed oxygenation status, as in the Baltic Sea.

Remediation scenarios tested by simulation

In watersheds, the most frequently tested scenarios concern the application of good agricultural practices, including the reduction of fertilizers. These scenarios do not lead to an adequate improvement in water quality so the models may test additional measures, such as the introduction of buffer zones (e.g. grassy strips, riverbank buffer zones). The models have also explored more radical scenarios such as developing organic farming, conservation or the conversion of arable land into permanent grasslands or forest. In large watersheds, the models show that the development of urbanization and conventional agriculture would increase nutrient loss, but they are more ambiguous about the effect of crop diversification, climate change and the increase in \( \text{CO}_2 \). For example, while warmer temperatures are supposed to reduce water flows together with dissolved flows through soil leaching, the flow of nutrients in particulate form could increase during more frequent extreme events; the rise in temperatures would affect both the source processes (nitrogen mineralization in the soil, less extensive wetlands in the watershed heads) and sink processes (denitrification). These simulations rarely take into account medium and long-term changes in land use brought about by climate change or otherwise, the impact of which could exceed the direct thermal effect on system functioning.

In streams and rivers, the models show that an improvement in the treatment of urban waste at water treatment plants (many of which are already very effective) would reduce plankton biomass by a few additional percentage points only. The models have helped focus action against eutrophication in rivers on a reduction of phosphorus waste. However, the recent inclusion of the effects of terrigenous inputs on marine eutrophication requires the testing of scenarios that reduce both nitrogen and phosphorus flows from the continent, and account for consider-
rable modifications to agricultural and food systems (a fall in the proportion of animal products in diets) (cf. the Livestock and Environment ESCo appraisal).

For lakes, the simulations assess the effectiveness of the various physical (re-aeration using air source arrays) or chemical measures (wastewater treatment, reduction in nutrients from livestock farming) and their combination to preserve lake water quality. For example, the models recommend an optimal number of nozzles, gas flow rate and bubble size (1 mm diameter) to induce summer destratification. They also show that an increase in plant-eating fish can help restore good ecological status by reducing the phytoplankton biomass.

For estuaries, several models have shown that reducing nutrient inputs by a fall in river flow would not cut down the estuary's phytoplankton biomass as much as a drop in concentrations in the river water. According to a multiple regression model calibrated on ten American estuaries on the Gulf of Mexico, giving the hypoxia level according to stratification intensity, nitrogen flow and the river’s P:N ratio, seven estuaries would require a fall of 50–95% in current nitrogen flows to recover a sufficient level of oxygenation. However, in the case of a moderately eutrophic estuary that is home to shellfish farming (e.g. mussel farms in the Limfjord in Denmark), one model has explained the fall in the mussel stock’s growth (and its negative economic impact) by too high a reduction in terrigenous inputs following recommendations issued to recover WFD good ecological status. With regard to the effects of climate change, one model has demonstrated that seaweed meadow surfaces will fall by a few more percentage points if the sea level rises by 30 cm, but they could increase by 50% if we reduce terrigenous nitrate inputs by 50% or more.

One of the main objectives of eutrophic lagoon models is the simulation of remedial operations based on reducing terrigeneous nutrient inputs, aiding water circulation through pumping or the widening of the channels to the sea, exploitation of the plant biomass, the modification of the location and density of oyster farm structures, or the removal of the superficial nutrient-saturated sediment layer. A fall in terrigeneous nutrient inputs (nitrogen for marine and brackish lagoons, phosphorus for lagoons that are mainly fresh water) generally appears to be the most effective remedial strategy, but the respective proportions of urban and agricultural sources need to be identified in these inputs. Where urban sources dominate, the strategy combining partial mineralization of organic matter at water treatment plants with a partial bypass of discharge to the sea appears, for a same level of effectiveness, to be less costly than full mineralization of organic matter at water treatment plants. In agricultural watersheds, the models are used to demonstrate that it is the non-point share of nitrogen inputs of agricultural origin that first needs reducing, as point-source inputs of N and P of urban origin often only account for a small proportion of all inputs. In shellfish lagoons such as the Thau lagoon, given that the malaigue phenomenon has its origins very close to the coast, the models have shown that distancing the oyster beds from the coast would considerably limit the risk of malaigue spreading and that a reduction in oyster farm biomass would restrict the self-sustaining of malaigue by reducing the volume of dead organic matter. None of the models has really simulated the expected effect of removing the superficial sediment layer.

In the open coastal sea, while P and Si are often limiting at the end of the spring bloom, N is the limiting factor in summer when the risks of toxic dinoflagellate proliferation or hypoxia in bottom water are higher. In northwest Europe, it can be argued that the OSPAR objectives to reduce nutrients have been reached or even surpassed for P thanks to the introduction of environmental measures, but for N, the models show that a radical change in agricultural practices is still required to reach the objectives. Except in enclosed seas (the Baltic Sea with very low salt content, and the low-salt Black Sea), where it is necessary to reduce P inputs, most of the models recommend reducing N inputs, often significantly. This is always the case where there are coastal green macroalgae accumulations, but also in cases of phytoplankton eutrophication, for example to limit Phaeocystis blooms in the North Sea or to reduce anoxia in the Gulf of Mexico. Some models estimate that, to attain a significant result, terrigeneous nitrogen inputs need to be reduced by 45–70%. Certain models also suggest that it is important to prevent nitrogen inputs from saturating the sediment's denitrification capacities, and the phosphorus inputs from saturating the iron oxides that adsorb phosphate, because that could lead to a sudden worsening of eutrophication (tipping point). The build-up of sedimentary organic matter stock is seen as a
potential retardant of the effects of remedial action, making the re-oligotrophication process slower than eutrophication (the hysteresis phenomenon): the modelling suggests that for some of the major enclosed sites such as the Baltic Sea, even if anthropogenic terrigeneous inputs were to come to a sudden and final halt, a return to a non-eutrophic situation could take a century. For the Baltic, the models tend to indicate that global warming will lead to an expansion of hypoxic and anoxic areas, unless we considerably reduce nutrient inputs. Increasing bivalve mollusc or grazer fish numbers is one remedial solution but it should be used with caution because of the indirect effects that it can induce: if the excess phytoplankton is eliminated from the water column, nutrient recycling can accelerate leading to increased primary production and more organic bio-deposits at the bottom, along with a greater risk of hypoxia.

5.4.4. The current limits of the models and possible improvements

For the physical side (hydrodynamics, temperature and salinity), the models are certainly now more realistic since they have moved on from a compartment approach (numerical diffusion multibox models) to a partial differential approach in the 1D, 2D then 3D space. When it comes to the biogeochemical and ecological aspects of aquatic ecosystems, there is no real consensus on the processes to be taken into account, their prioritization or, most importantly, the way to formulate them. The first models developed only simulated a single biogeochemical cycle (P in fresh water, N in marine water) and could not alone establish a preferential limitation by N or P. Today, however, most river, estuary and coastal ecosystem models simulate the N and P cycles alongside one another, and sometimes also include Si. They are able to indicate the success of the most limiting element over time and/or space. Hence, in marine environments, the models point to nitrogen as the limiting element in green macroalgae tides, while they show a more complex succession in the case of phytoplankton coloured waters: phosphorus and/or silicon are often the limiting factors in spring, while summer limitation comes from nitrogen. Although the forcing of marine models by river models is still infrequent, it can be used to explore scenarios for the coupled reduction of N and P. In addition, the early models often only used a single phytoplankton variable, while today’s models simulate competition between different algal types by introducing functional traits when formulating the phytoplankton or benthic compartments. The coupling of processes that take place in sediment with those found in the water column is becoming gradually more widespread for chemical descriptors, but much less so for biological descriptors: at the current time, very few models handle the dormant sedimentary cycles of cyanobacteria or proliferating micro- or macroalgae. Moreover, the nutrient flows between water and sediment estimated by the models are highly sensitive to the type of formulation (using a threshold or gradually variable) applied for the effect of oxygen content on bioturbation. Finally, very few eutrophication models include a detailed description of the trophic environment and there is currently a lack of studies on the biodiversity modifications brought about by recurrent eutrophication: stimulation of gelatinous plankton, regression of algal diversity, macroalgae succession (Cladophora → Pylaiella → Ulva) in a marine bay that is gradually enriched, or the effect of toxic substances from eutrophication, such as H2S, cyanotoxins or phytoplankton toxins. Likewise, very few models deal with the impact of contaminants (metals, pesticides or emerging pollutants) in the processes linked to eutrophication phenomena.

The existing uncertainty or sensitivity studies mostly show that the equation parameters have correlated effects and only a few of the parameters control the results. We also see a scale effect: the models of small ecosystems are more sensitive than those for large systems. Calibration is done using a very small number of parameters, partly due to the very lengthy calculation time required to carry out simulations using large 3D models, and the level of uncertainty of the models’ results is rarely quantified. Another source of uncertainty lies in the forcing factors used, which are often imprecise in the case of human activities because the data is difficult to access and update often because they vary widely over time and space. This is certainly true of agricultural activities: type of fertilizer, livestock manure transfers over short or long distances, type of cover crops, tilling, etc. In addition, the slow response of storage compartments (groundwater or sediment, for example) means that eutrophic ecosystems are more often in a transitional state than stationary, and few models benefit from the long-term series of observation data that would be required to calibrate/validate them. Finally, we should note that several of the
models’ state variables are only measured rarely, most notably the detrital variables in water and sediment. With all of this in mind, authors are advised to remain very cautious as to their conclusions. For the Baltic Sea and the North Sea, some recent articles have compared the results from several models, or have combined them in so-called ‘ensemble’ simulations. The disparity between models increases when we move away from physics (currents, temperature and salinity) to chemistry (nutrients) then biology (algae, grazers).

The application of the models to assess remedial scenarios can indicate the main trends after substantial modifications to the forcing factors, and often only considering one segment in the continuum, yet nutrient flows are transferred and transformed from the basin head down to the coastal waters. Because of the gradual shift from a limitation in phytoplankton growth by phosphorus in fresh water to limitation by nitrogen in sea water, we need to know where, on the salinity gradient, to find the plankton proliferations and hypoxia that we wish to reduce to assess the respective role of terrigenous N or P inputs. The systematic introduction of accumulator compartments with a long response time, which can be physical (soil, groundwater, the backwaters of a river, tidal marshes) or biogeochemical (soil and sediment stocks), is one way of improving models. Because non-point phosphorus transfers are intermittent and highly variable (moved by storm waters, river bank erosion, etc.), their modelling is still a challenge for scientists. The chaining of watershed models with those for river networks and then coastal sea models began in the 2000s, but often only concerns extensive systems and uses simplified formalization. These model chains need to be made more reliable and their uncertainties assessed before they can be used operationally to identify the maximum permitted flows that would guarantee a complete water continuum free from eutrophication.

5.4.5. Systemic modelling approaches: SFA, LCA and bio-economic models

Alongside these modelling approaches, methods to estimate flows and the potential eutrophication risks have been developed to form a better picture of the impacts of human activities on nitrogen and phosphorus flows. These approaches are especially interesting when comparing remedial scenarios.

The Substance Flow Analysis (SFA) approach is used to calculate flows at country or regional level, to highlight the contribution of certain production streams, and especially agricultural sectors. Flows in natural systems are based on often very roughly estimated coefficients so they are not meaningful at a local level. This approach has been widely applied to estimate phosphorus flows. For example, an inventory carried out in the United Kingdom compared phosphorus production in animal waste to crop requirements, indicating a slight surplus; it then differentiated the sources according to livestock sector. In France, a similar inventory of phosphorus flows has been run at national level and in two regions (Brittany and the Paris basin). It quantified the flows linked to animal food and waste, and calculated an excretion coefficient, then compared this to exports via crops. This approach gave an indication of the scale of the imbalance that can occur between inputs and export via crops.

The Life Cycle Analysis (LCA) approach and the calculation of a nitrogen footprint were developed to assess the potential impacts of production methods on eutrophication, particularly in agriculture and water treatment. These approaches have been applied to an agricultural production sector, to production methods and, more recently, to a territory. The advantage of these approaches is that they take into account all sources, including transport, infrastructure and so on, to situate sector impacts within the wider picture of production methods. The downside to these approaches is that they are not contextualized as they are based on normative impact coefficient approaches. The approaches can also be used to consider a set of impacts (climate change, acidification, etc.) and to analyse synergies and/or antagonisms between the various issues. The various studies conducted generally agree on the need to take into account the fate of emissions from soil or water, and their effects, particularly with the notion of critical load. The models are based on a simplification, taking the limiting element in reference to the Redfield ratio. The N footprint is a promising method, especially when it is coupled with life cycle analysis to take into account the anthropogenic system and the environment. It gives a holistic vision of nitrogen use and efficiency.

Our inventory of methods should also mention the still quite limited attempts to combine biophysical and economic models. Some of the results of these models have been mentioned above in the economic section.
5.5. Dealing with complexity: instruments and practices for integrated management

5.5.1. Integrated water management to improve treatment of eutrophication

Many of the studies published since the 1990s emphasize the benefit of approaching eutrophication processes within the framework of fundamental research activities and targeted research to produce interdisciplinary knowledge that encompasses biophysical sciences and human and social sciences. Most of them are related to applied ecology and management sciences. This evolution is not specific to the study of eutrophication issues; generally speaking, it concerns most environmental problems resulting from accelerated changes and the coupling of environmental changes with social shifts in complex socio-ecological systems. It is particularly well developed in the field of water and coastal management, where the principles of integrated management are included in various public policy tools. The disciplinary diversity of the authors and the shift in public policies are helping to structure scientific production within the domain of sustainability science, which takes new conceptual approaches to water management, where eutrophication is a textbook case. This rapidly expanding research field aims to build interdisciplinary knowledge to help understand and tackle sustainability issues effectively. It is highly structured in Scandinavia and North America but is still in its infancy in France.

Most of the work done in this regard is restricted to Europe and North America, geographic areas for which nitrogen and phosphorus pollution has long been subject to problematization and very structured public policies. Nonetheless, social science publications taking eutrophication as their main focus, rather than a contextual element, are still rare and often work with relatively old data. Although integrated management has long been recognized in France and Europe as a guiding principle for water management, it is covered far less extensively in targeted research in the social sciences than it is in biophysical sciences. As such, work on water policy remains largely disconnected from that on the integrated management of socio-ecosystems affected by eutrophication. Usually, the aim of research into integrated management is to produce generic models, which unfortunately do not focus much on the regulatory context and the specific public action mechanisms linked to the case studies.

Work on the integrated management of socio-ecosystems affected by eutrophication generally raises two points. Firstly, the approaches developed mainly respond to the principle of integrated water management, defined as the consideration of socio-ecological processes and their interactions within a given territory (e.g. a watershed). Secondly, in relation to this conceptual framework, there is the question of knowledge production as part of integrated management. What knowledge should we and can we draw on to account for the socio-ecological processes that contribute to eutrophication? How can we develop that knowledge and make it operational in management terms? How should we present it to stakeholders who are likely to make decisions or change their practices? The effectiveness and limits of the various ways of getting stakeholders involved are therefore discussed.

Dealing with the complexity of eutrophication processes

The call for more integrated management first came after highlighting and reviewing the complexity of the systems concerned and the eutrophication phenomena they undergo, calling for dialogue between the various scientific disciplines to look beyond the field of environmental sciences. The different dimensions of this complexity have been set out in the preceding chapters: eutrophication processes are multifactoral, non-linear and marked by irreversibilities. Eutrophication appears to be a multifaceted phenomenon, characterized by chronicity and the existence of crises and tipping points, meaning the traditional forms of action against environmental problems have to be reconsidered. Scientific literature on how to manage eutrophication therefore generally conveys the idea that the responses implemented will be integrated and systemic, i.e. they will put greater focus on the nature and multi-
plicity of interactions and feedbacks between social and political changes on one hand, and ecological changes on the other. The spatialization and temporality of the socio-ecological processes concerned make it necessary to deconstruct a linear, causality-focused way of thinking.

In the literature, the interdisciplinary experiments involving social science and environmental science and looking at eutrophication raise two major scientific challenges. The first challenge lies in the difficulty in obtaining data from different scientific fields to integrate them into a single scientific inquiry or model, either because the socio-economic data are lacking or because they come from qualitative and not quantitative approaches. The second challenge is that social science approaches, especially where they draw on interview-based surveys, ‘incorporate’ the eutrophication issue within a much broader discursive and cognitive system than the biophysical process alone. While this is of major interest for sociologists and anthropologists, it complicates things for interdisciplinary exchange, which is then required to develop a reflective dialogue between disciplines to try and formulate results that are not expressed in the same language and which concern different scales of interpretation. The social complexity is also widely questioned by social science researchers (sociologists, anthropologists, political scientists and geographers) who develop approaches in terms of the diverse social representations and perceptions, of potentially contradictory socio-economic and political issues and of contrasting public practices: in the short term, these approaches may sometimes seem less operational and may involve risk-taking for some stakeholders. They are, however, essential within the framework of effective public action over the long term.

**Assessing the effects of public policy designed to tackle eutrophication**

Research into integrated management has developed because of the hindsight researchers now benefit from when analysing eutrophication management policies and their effects. An analysis of the policies introduced to tackle eutrophication first points to the wide variety of trajectories taken and their high dependency on local contexts. This work provides valuable information on the context of the policies implemented since the early 20th century. When faced with complex phenomena that are very difficult to unravel, the authors note the persistent gap between the stated objectives, the resources committed and the poor results. Significant results with regard to controlling phosphorus and nitrogen flows have only been obtained in certain specific circumstances, on a temporary or more sustainable basis. However, the results are more limited when it comes to restoring environments and mitigating the consequences of eutrophication. These results are very similar overall to those produced by environmental scientists to date, although human and social science research usually takes a more critical viewpoint given the focus it puts on the socio-political conditions under which public policy mechanisms are devised and implemented. Research led by historians, sociologists, geographers and political scientists highlights the historical importance of the scoping work done by the industrial and agricultural stakeholders involved or by the public resource management agencies: this research covers the full range of activities that generate or define the meaning of ideas aimed at mobilization or countermobilization, in order to establish the terms used when adding a problem to the political agenda and when it becomes public. Because of the domination of technical approaches to the problem and the difficulty in influencing development models, which tend towards an increase in stress factors in hydrosystems, these scoping activities play a decisive role in the effective treatment of eutrophication: they have reduced the spectrum of the public policy instruments activated, narrowed the focus of certain activities, and led to a downward adjustment of the objectives set.

Eutrophication management also means taking on-board the sometimes antagonistic interests and representations of social stakeholders concerning eutrophication issues. Given the social and political tensions surrounding eutrophication, its consequences and its management in certain territories, the involvement of non-institutional stakeholders—or interest groups who are less used to working with the public authorities—in problematization work remains rather limited and a source of distrust. Their involvement is nonetheless vital from the perspective of change because, over the long term, it prevents avoidance strategies from undermining the credibility of the institutions and the legitimacy of the environmental regulations. A lot of the work done in the Great Lakes region and in Europe emphasizes the importance of involving associations that defend users’ (such as fishers or watersports enthusiasts) quality of life or scientific whistle-blowers when implementing public action. Their inclusion in local water quality governance, as in the example of the local water commissions in France, forces certain key stakeholders to step up their efforts. This was the case in France when, in 2008–2009, local associations active in the fight
against green tides warned of the health risks from decomposing green algae on Brittany’s beaches: although the prevalence and seriousness of the risk was not recognized by all stakeholders, the shift in scope resulted in much greater involvement from the State and the launch of a solidly funded government action plan on green algae. Against this background, a failure to recognize the diversity of representations and ‘means of knowledge’ will be seen as a weakness in the policies negotiated. Social representations form and reflect the way in which environmental elements are assimilated, and how they are given a sense and a value. In other words, better knowledge of the perceptions of water quality and the problems that affect it could help bring on-board stakeholders with specific interests to contribute to the design and implementation of management policies.

The dynamics of knowledge and relationships between science and politics

Given the uncertainty and complexity of the issues surrounding eutrophication, a consensus has gradually been reached on the fact that their scientific and political aspects need to be tackled together. The agencies that manage habitats and the environment lie at the interface between scientific production and the application of measures designed to eradicate eutrophication processes. Cooperation between scientists and management agencies depends not only on the state of knowledge on eutrophication but also on the way in which that knowledge is disseminated and shared. The production and dissemination of knowledge useful to eutrophication management cannot be understood without linking it to the socio-political contexts within which it emerges and the places in which it is disseminated. In this respect, the done work focuses more particularly on the receipt and co-development of knowledge outside of the ‘scientist/manager’ sphere. This work mainly concerns farmers whose practices affect the aquatic environment and who are therefore regulated. In other, rarer cases, it looks at waste and wastewater management and therefore concerns a wider population and local public operators.

Eutrophication processes are thus dealt with as socio-political issues, within the framework of wider thinking on the quality of democracy in a situation of great uncertainty, as a condition for proper management of environmental problems. Rather than developing knowledge, it is more a matter of advancing a political model to establish dialogue based on the knowledge. This shift would have several benefits:

- making it easier to identify the links between local problems (water supply, local development, cohabitation of activities and uses, habitat quality, and so on) and global changes, especially disruptions to nitrogen and phosphorus biogeochemical cycles;
- enabling a better grasp of all the processes that affect water quality and the vast array of dimensions to be taken into consideration;
- taking a clearly defined approach in a context where management is heavily reliant on the production and interpretation of scientific data;
- taking care over the quality of communication between social stakeholders and, more generally, recognizing the variety of outlooks, knowledge and approaches to problematization, and making this a condition for the success of the policies implemented, going beyond dialogue between research and management agencies, to integrate the impact of human activities on eutrophication processes and the social and economic implications of remedial actions. Knowledge of eutrophication and its causes and impacts is central to the reflective processes, not only in terms of significance but with regard to the capacity for forging links and giving meaning to social relationships. More particularly, the way in which scientific knowledge is applied in policies designed to tackle eutrophication has been covered in very detailed, often critical analyses in the literature. The biggest challenge therefore lies in positively transforming the interfaces between science and politics to enable understanding and greater awareness, and the mobilization of stakeholders to get them involved in policies that are demanding in terms of objectives, take a long-term outlook and bring together different social groups which often have very different cultures and interests.

5.5.2. The principles of integrated management

The dissemination of integrated management has drawn on national and supranational experiences. In the case of France, the 1964 water act, which set up the various water boards, introduced the principle of territorialized management, based on watersheds and participatory democracy involving the different local bodies and frameworks,
such as the Schémas d’Aménagement de Gestion des Eaux (SAGE or water planning and management schemes). Management per geographic district was introduced in 2000 by the WFD, supported by management plans and various sets of measures to reinforce the principles of integrated management.

The Global Water Partnership (GWP) defines integrated water resource management as ‘a process which promotes the coordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems’. This definition was originally promoted by the international organizations that work with large private groups but is discussed and understood differently by researchers. Integrated water management encompasses several principles. Four of them relate to the studies into eutrophication: the socio-ecosystem approach, so-called ‘adaptive’ management, the consideration given to ‘lay’ knowledge, and the participatory approaches. In this context, the Driving force, Pressure, State, Impact, Response (DPSIR) framework has been adapted and updated several times since the mid-2000s. For example, the work done by C. Pahl-Wostl and her team insists on the value of this approach when it comes to increasing the ability of the system to cope with change and producing autonomous responses (Fig. 5.5). The whole process needs to be perceived as iterative, proceeding in cycles, in contrast to the quite linear and sequential approach that is often adopted when using the PSIR framework.

Given its participatory dimension and the wide diversity of social groups potentially involved, integrated socio-ecosystem management entails in-depth work to investigate the different ways in which a given situation is recognized and problematized. It also requires the definition of indicators that integrate the social aspect of pollution management and change processes. Political science, sociology, environmental psychology, ethnology, economics and management science can all make a useful contribution in this respect.

5.5.3. Integrated management practices and experience

Integrated management and modelling

Knowledge for management purposes is developed at the interface between environmental science and social science, meaning the ‘natural’ process has to be extended to include more general interaction between human society and other components in the socio-ecosystem. Thinking ‘at the interface’ means researchers and managers have to share tools for dialogue.

In environmental science literature, models have been extensively taken on-board and discussed. Similarly, since the 2000s, models are being increasingly used beyond the bio-economic sphere. Nonetheless, theoretical papers and empirical experiments that consider the political and social dimensions of eutrophication issues remain few and far between. For now, other than in economics, social science publications in this vein tend to look less at the data used in the models and more at their acceptance and use in the arenas of participation.
Figure 5.6. This diagram completes the general descriptive representation of eutrophication provided in Fig. 2.7 with a compartment reserved for the socio-economic sphere within the framework of a Drivers-Pressures-State-Impacts-Responses (DPSIR) conceptual model developed by the European Environment Agency regarding the eutrophication of the Baltic Sea. This type of representation including social dynamics remains marginal in the international literature, firstly because of the limited number of interdisciplinary research programmes resulting in analyses of integrated management and, secondly, because of the complexity of interpretation that arises after adding the socio-economic effects and management options to the model. Here the processes are split into i) causes, ii) primary effects, iii) secondary effects, iv) social responses, and v) management options. The causes are then divided into four categories linked to nutrient sources. Primary effects are grouped as physical and chemical or biological. The different colours show whether an effect has been detected, whether evidence as to its occurrence in the Baltic Sea is lacking, or whether that effect is typical of shallow zones. Source: Lundberg, 2005. © The Royal Swedish Academy of Sciences.
CONCEPTUAL MODELLING

The principles of integrated water management can be detailed using a range of conceptual approaches to better grasp the complexity of the socio-ecological processes that contribute to eutrophication. There is a broad consensus on the need to:

- deal with this type of pollution and its effects in a global, not local, manner;
- think jointly about the ecological and social dynamics with a view to sustainability (in the broadest sense).

This consensus operates on two levels. It questions scientific production as to its capacity to report on a particular systemic operation and/or recommends the implementation of this framework for discussion through management practice; these two levels are, of course, linked because many researchers are involved in applied research programmes. The conceptual approaches in the literature are most often represented diagrammatically, showing the permanent interactions between the various components in the system; they have been little tested in the field but serve as analytical frameworks. They were extensively developed to accompany the multi-annual development plans for pollution in the Great Lakes of North America, the North Sea and the Baltic Sea. However, few of them manage to reconcile a truly accurate description of interactions within the system with sufficient readability to increase the intelligibility of the processes and their levers for non-experts. The model below endeavours to demonstrate the possible links and trajectories between the various components, including socio-economic aspects. Here, as the authors point out, the representation is hampered by the delayed spatio-temporal effects of certain relationships. As such, certain relationships in this diagram are typical for shallow areas, while others are more likely to be found offshore. (Fig. 5.6).

OTHER STUDIES USE RETROSPECTIVE MODELLING

Researchers from the Global Alliance for Resilience (AGIR) put forward a systemic interpretation of eutrophication using the notion of adaptation cycle. This type of integrated modelling draws on long-term longitudinal analyses, documented using historical ecology data linked to contextual data (demographic, socio-economic and political changes). Retrospective modelling gives a better representation of the socio-ecosystem dynamics but remains largely descriptive. The case of Lake Mendota in the United States is a good example since it represents a long phase of deterioration followed by repeated attempts to restore water quality: the authors identify five main cycles, each one split into four phases (Carpenter et al. 2001):

- The first cycle starts with the arrival of European immigrants in around 1840. They developed agriculture, which diminished the lake’s resilience due to run-off of water and sediment from ploughed fields, reducing the transparency of the lake’s water. This operating state underwent a long transitional phase. The population grew—albeit slowly—and likewise the impact of agriculture on the lake. Intensification of agriculture and urbanization after the Second World War lead to phase three or the collapse of the system, bringing cycle one to an end.
- The second cycle began with a phase of renewal and reorganization. In the 1970s, there were several—largely unsuccessful—projects to divert wastewater. The episodic growth of invasive plant species led to more visible eutrophication once again. For the authors, cycle two ended with an institutional transition (the institutions taking charge of the problem) but did not result in ecological change.
- The third cycle was triggered by the lake’s managers in an attempt to reduce phosphorus within the sub-watershed. The rapid failure of this cycle can be partially explained by a low level of commitment from farmers.
- The fourth cycle started in the 1980s with a biomanipulation approach and the introduction of piscivorous fish (carnivores that create high predation pressure on planktivore fish, thus enabling more abundant zooplankton grazers which in turn apply pressure on phytoplankton and thus reduce algal concentration). This artificial manipulation only worked for a few years, hindered by fishing and heavy rainfall events accompanied by further episodes of erosion and sediment and phosphorus run-off into the lake, bringing the fourth cycle to an end.
- The fifth and final cycle was the outcome of collaboration between managers and researchers in the late 1990s to launch a new, ambitious action programme (50% reduction in P inputs) involving farmers, erosion control and the preservation or restoration of riverbank areas and wetlands. The level of phosphorus tended to stabilize or indeed fall (although the authors are not yet able to confirm categorically that this is linked to the actions taken).
A final body of work builds forward-looking scenarios that feed into goal-setting for public policies and help define development paths

In the field of economics, these approaches have highlighted the benefits of bio-economic coupling when setting goals for public policies. Some authors also recommend different approaches, such as simply trying to improve the current situation. In anthropology and sociology, studies on this topic are fewer in number and more critical. They tend to highlight the biases related to the choice of scenario, the underlying asymmetric information and powers, and the accompanying knowledge selection/exclusion process, particularly with regard to contextualized knowledge. With the current state of knowledge, the effects of these approaches in terms of stakeholder acceptance, mobilization and the effectiveness of public policies remain uncertain.

Co-construction with local stakeholders: non-point agricultural pollution sources

The co-construction experiments undertaken by researchers now mainly focus on agricultural areas, giving farmers a new forum for discussion and an alternative to the proposals previously headed by the institutions. However, the majority of experiments are conducted in close liaison with institutional players, which is a condition and source of support for research but it also introduces significant bias when assessing the results produced, since the assessment is linked to the prospects for transforming practices and regulation, and thus includes a strong normative aspect.

Multiple goals are set within the framework of this kind of approach:

- developing management alternatives that take into account the viewpoints of local stakeholders to solve conflicts over the water resource;
- establishing models covering nutrient inputs from agricultural areas until their discharge in the receiving water bodies to share knowledge with local stakeholders;
- building scenario-based tools to plan and estimate future management in economic terms.

From the institutional viewpoint of WFD implementation, most of the international articles on this topic come from Scandinavian researcher collectives. However, since the early 2000s, research has also been conducted in France and the United Kingdom, looking at the local level. A growing number of authors are thus taking an interest in agricultural practices and their potential modifications in the context of the implementation of regulations. Depending on country, these generate varying political tensions between the agricultural realm and the organizations in charge of protecting aquatic environments. All of these studies question the capacity of a model to perform in a participatory framework. Here there are two streams of research: one is based on the conventional theory of rational choices, focused on a cost/benefit calculation performed by individuals, while the other introduces a more qualitative and contextual dimension to the analysis. Broadly speaking, the most thorough surveys to date emphasize the importance of collective identities, the memory of social ties and the quality of relationships with the institutions as drivers of engagement in schemes to improve water quality. Sharing knowledge about eutrophication processes within the agricultural realm is fundamental in changing production and management practices. The importance of disseminating knowledge in the agricultural sector and the significance of its own operating rules are regularly highlighted. Trust, shared experimentation, and identity-forging and collective bonds should therefore be given consideration when studying mobilization in the agricultural realm.

The green tides in France have been the subject of management science work that is quite similar to that conducted abroad. The difficult balance between economic and public policy reasoning on the one hand, and between public policies themselves on the other remains a major obstacle in mitigating eutrophication. The type of commitment from farmers and other stakeholders fluctuates between localized and territory-based engagement and more extensive standardization. There is a risk of creating a system that is polarized between a culture of industrial ecology (that uses technology to limit impacts) and a series of alternative microsystems promoting systemic changes to production methods. However, another two points are more widely developed in the French case. Firstly, there are the views expressed by scientists involved in the appraisal systems. The second point highlights the symbo-
lic dimension associated with green algae, which have become a kind of ‘figure of disorder’, problematizing the environmental consequences of society’s choices. The fact that these algae have become a symbol of the harm caused by productivity-driven farming further complicates management and adds to the uncertainties surrounding the treatment of eutrophication on the Brittany coast.

The literature that reports on co-construction experiments in agricultural areas also questions the relevance and effectiveness of the models as participatory tools and the value in focusing on eutrophication-related knowledge sharing. Their widespread use of models is the source of several challenges for the social stakeholders involved: how are they interpreted in expert or lay (non specialist) terms? What about their ability to reflect loyalties, values and identities, to account for the complexity of socio-ecological systems and to result in new management actions?

The tensions surrounding the production and dissemination of knowledge

These cases of eutrophication thus illustrate the rising popularity of participatory methods as an alternative to more conventional approaches brought about by economic and/or regulatory control of environmental issues, as well as the limits of those same methods in terms of public action. Some authors even suggest that they may be instrumentalized.

The limits lie in the difficulty in representing all stakeholders, who do not share the same knowledge and are not equipped with the same resources to take part in consultation. Socio-economic and political interests can also contribute to the failure of a knowledge-sharing approach: the greatest leverage lies at other levels and some structural stakeholders (such as large economic agricultural organizations) are, with few exceptions, missing from the mechanisms in place.

This point concurs with the more general finding that there is a huge gap between the goals, instruments and governance methods applied to water policies on one hand, and those applying to agricultural policies on the other. Local stakeholders, including farmers, only have limited means at their disposal when it comes to influencing productive models that are not largely focused on local production but instead form part of the agricultural policies and markets developed at other levels. Against this background, securing the effective participation of all stakeholders in the sector should be seen as a way of evening out the contributory effort while supporting structural developments in the agricultural and agri-food sectors. However, this condition is not enough.

Generally, the issues associated with knowledge dissemination, sharing and co-construction within the spaces for negotiation cause a couple of pressure points. The first is due to the multidimensional character of these negotiating spaces. Certain have their origin in the experiments led by researchers as part of a programme, setting forth the ‘right’ conditions for directing a constructive dialogue but remain subject to the temporality of research and institutional frameworks. Others emerge directly from institutional dynamics (in which researchers participate or which they observe) and often struggle to do away with technocratic inflexibility (in terms of discussion, the issues raised and so on). The ‘ideal’ space for negotiation is ultimately highly ambiguous. The second pressure point is the scale of co-construction, ranging from the scale of the territory concerned—local or micro-local—to the more global scale of a hydrosystem and its watershed, for example. Knowledge, practices and loyalties do not contribute in the same way to these various scales of understanding eutrophication.

The tensions highlighted by participatory eutrophication management exercises indicate a ‘blind spot’ in the work conducted on the topic thus far. Little has been said about the professional culture of the stakeholders (non-researchers) responsible for these exercises, whether they come from the agricultural, environmental or administrative sphere. The dissemination of the ‘interface approach’ and its input in cognitive and relational terms for institutional stakeholders is rarely discussed in detail in the literature, in contrast to the contact between researchers and local stakeholders.

Hence, as they stand, these limits and tensions are not fundamentally any different for eutrophication management than for other water quality management issues, and they can all be partially addressed at watershed level through integrated water management as established and developed under French law since the foundation of the water boards in 1964. Nonetheless, several aspects of eutrophication management entail a further step in the integrated management process. The first aspect is joining up the different time scales. The second is the
different levels of management: while the biophysical mechanisms of eutrophication are mainly local, its socio-political impacts cannot be dealt with on a purely local level. Management policies have gradually taken on-board the need to prevent pollution at the source, but they now have to face a dual challenge when treating non-point pollution sources, firstly by reorienting agricultural policies at community and national level, then by rebuilding the links between farming and the territories in which it takes place, from an economic, political and social viewpoint. Given all of this, encouraging social and territorial innovation is more than just a form of change management in many areas where the agricultural sector is precarious; it is in fact a vital addition to the mechanisms of integrated water quality management at watershed level.
6. Conclusions

Two definitions of eutrophication have been proposed based on an analysis of the literature: one refers to so-called “natural”, or geological, eutrophication, and the other to anthropogenic eutrophication. We felt it was important to make a distinction between the two, as the description of natural eutrophication should not distract from the threat posed by human activity to phosphorus and nitrogen cycles, as well as to carbon cycles. The accelerated development of eutrophication of anthropogenic origin is currently the main focus of societal and management concerns. This joint scientific appraisal therefore discusses eutrophication of anthropogenic origin.

6.1. Findings

Aquatic ecosystems are complex systems whose functioning is governed by dynamic balances. Eutrophication is an imbalance in functioning, triggered by a change in the quantity, relative proportions or forms of nitrogen and phosphorus entering systems. The nature and intensity of responses also depends on environmental factors such as light, water residence time and temperature.

Both continental and marine water ecosystems share the same general response mechanism to these changes in nutrient flows: an increase in nitrogen and phosphorus causes an increase in plant biomass, gradually generating a decrease in the penetration of light in the water column. Aquatic ecosystems thus shift from a system with limited nutrient inputs to a system gradually saturated in nutrients, in which light becomes the new limiting factor. This mechanism leads to a cascade of chain reactions, notably a change in the structure of biological communities and trophic networks, as well as changes in biogeochemical cycles. These changes can occur gradually, in proportion to forcing factors, or, on the contrary, abruptly. The most notable effects of eutrophication are vegetal blooms, sometimes toxic, loss of biodiversity and anoxia, which can lead to the massive death of aquatic organisms.
Each ecosystem is unique and has its own history and dynamics, which in turn are related to geological, geomorphological, hydrological, ecological and climatic conditions, but also to past and present anthropogenic pressures and their nature, as well as to the sociological and economic contexts in which they have evolved. While the mechanisms of eutrophication are generic, its trajectories are diverse, related to the diversity of local situations, with threshold effects highly dependent on past and present situations. Aggravating conditions exist: for instance, links are suspected to exist between eutrophication, toxicity and biological invasions. Inputs of toxic elements alter trophic chains, which can create conditions more favourable to eutrophication. Changes in trophic chains can provide the right conditions for biological invasions. Vulnerability to eutrophication varies with the intrinsic properties of the receiving aquatic ecosystem. Vulnerability therefore needs to be defined by taking into account the entire direct and indirect causal chain that connects these properties.

This complexity makes it is very difficult to predict ecological and biogeochemical developments in aquatic systems. For instance, it is difficult to accurately extrapolate the results observed in one type of ecosystem to others. Furthermore, climate change is also likely to play a role in stimulating eutrophication. While the role of climate on nutrient flows is still a subject of debate in the scientific literature, as it depends on the interaction with land use and human activity, there is more of a consensus about the effect on biological communities: rising temperatures are clearly an aggravating factor in the development of plant biomass. Longer stratification times in slow environments, earlier nutrient consumption by phytoplankton and the resulting change in stoichiometric balances could also lead to greater frequency in the development of harmful algal blooms, particularly cyanobacteria blooms.

Land use and human activity (urbanization, industrialization, agriculture) in the last hundred years have drastically amplified pressures and transformed landscapes, impacting the quality of soil, groundwater and surface water. The majority of groundwater is polluted by nitrates while soils and sediments are often enriched in phosphorus. The transfer time of nitrogen from watersheds to aquatic ecosystems spans several decades and phosphorus bioavailability in soils and sediments has increased. This partly explains the limited decrease in nitrogen flows, and, to a lesser extent, in phosphorus flows to watershed outlets despite the efforts made to reduce inputs. Restoration trajectories must therefore be determined based on local contexts.

There is a growing consensus on the necessity of jointly reducing nitrogen and phosphorus inputs, even though some ecosystems are more sensitive to nitrogen or phosphorus. All systems (water and waste treatment, agricultural systems), biological data, and economic approaches emphasize the importance of considering N and P together in the remediation and prevention process. Strategies to limit nitrogen and phosphorus flows involve different levels of action depending on the vulnerability of aquatic ecosystems:

1. **Rational use of inputs** depending on the environment (soil, groundwater); there is still some leeway to achieve efficiencies in the use of nitrogen and phosphorus. These efficiencies are to be found both in waste and wastewater treatment and in agricultural fertilizer management (composition, soil management and control tools, plants, etc.) and animal feed (adapted composition, etc.). There is room for improvement at this level and it must be implemented. In a number of cases, however, this will not be sufficient, and other levers concerning the very design of systems (e.g. a change in the type of production), should be considered.

2. **Vegetation cover should be as continuous as possible**, in spatial and temporal terms (interculture or interplanting, combined crops, meadows, agroforestry, etc.), in order to provide for a continuous assimilation of nutrients by terrestrial plant biomass. This biomass will contribute to carbon sequestration in soils, thereby strengthening that of nitrogen and phosphorus. Animal densities must be reviewed and limited to soils’ capacity to receive effluents (directly or after treatment). Keeping soils, groundwater, sediment well ventilated will help improve phosphorus adsorption.

3. **Maintaining or restoring landscape diversity** (hedges, wetlands, riparian woodland, etc.) in order to limit leaks of nutrients to aquatic ecosystems by various processes (adsorption, denitrification, etc.). Conversely, any changes increasing flow velocities, such as stream drainage, rectification, containment and dredging, soil imperviousness (urban development, soil degradation and compaction), should be avoided.

The organization of public action in terms of water quality management covered three periods: firstly, sanitation, which has largely been completed in industrialized countries, but remains an urgent priority in fast-growing countries; secondly, the still ongoing phase of treating industrial and domestic pollution; and lastly, a phase that is now a priority, treating agricultural non-point source pollution. In industrialized countries, developments observed in freshwater have been fairly positive in recent decades, more so for phosphorus than for nitrogen, while
marine eutrophication phenomena do not appear to have diminished since the beginning of the 21st century. At global level, the number and footprint of hypoxic and anoxic zones in the marine environment has tripled since the 1960s. A 2010 census numbered nearly 500 of these areas, with a geographical footprint of 245,000 km². There has also been an increase in the diversity, frequency, size and geographical extent of toxic microalgae blooms in recent decades. Although it is still difficult to extrapolate trends from one region to another, the link between the increase in nutrients inputs and that of toxic blooms is often established.

Eutrophication is a process framed by several regulatory texts following different rationales. Several fragmented guidelines on uses, dating back to the 1990s and providing a framework for a given field, coexist with directives with a more comprehensive reach such as the WFD and the MSFD in the 2000s: the Nitrate Directive focuses on the agricultural origin of nitrates, with a threshold of 50 mg/L defined in relation to drinking water standards; ecosystem sensitivity must be considered explicitly in the characterization of water bodies for the definition of vulnerable areas, bearing in mind that the eutrophication process in continental waters also depends to a great extent on available phosphorus and relative relations between nutrients; the UWWD frames the collection, treatment and discharge of wastewater, with point source-specific emission standards, but no standards for the receiving environment; the WFD and the MSFD require the implementation of the measures necessary to maintain or achieve the objective of good ecological status in water bodies, notably by a regular census of the overall health state of hydrosystems. With the exception of the MSFD, guidelines provide no specific recommendations on eutrophication, which is considered as part of a set of potentially degrading pressures. To each of these texts correspond targeted monitoring systems, which are essentially used to check compliance with the standards in their field and which are insufficiently coordinated.

In the field of the characterization of eutrophication, two strong considerations stand out: firstly, systems implemented prior to the WFD were more targeted to frame the eutrophication process (e.g. fast diagnosis of lakes; eutrophication monitoring by the Rhône-Mediterranean-Corsica water board); a number of variables, their frequency or their spatialization provided a wealth of information (24-hour cycles for oxygen and pH, at different levels of the water column for deep-water environments, seasonal succession of phytoplankton or macrophyte communities, etc.). This assertion must be qualified for the marine environment, which is often governed by international conventions and for which protocols dedicated to eutrophication have been maintained. In addition, the biological indicators for each quality element in the WFD show a general status of water bodies under the effect of multiple pressures; it is this property that was optimized when these indicators were recently brought into conformity (multimetric indicators responding to the variety of possible degradations). The joint review of the various indicators strengthens the WFD's ambition to carry out a more holistic assessment, more readily understandable by the general public and administrators; on the other hand, it makes it more difficult to extract an individualized signal such as eutrophication. This is even more true for information on some of the possible symptoms that can occur seasonally (algae blooms or temporary anoxia).

The virtual absence of bioeconomic models makes it difficult to support remediation. These models would have to be built very early in the restoration process to integrate biophysical, ecological and economic aspects from the outset. Existing studies show that the impact-related costs that have been assessed are high, underscoring the importance of remediation and prevention. Studies on remediation plans indicate that there is no ideal solution, but only targeted policies, designed for specific situations, often with instruments developed for a specific purpose once the problems have been properly identified and analyzed and the various options available have been reviewed.

Similarly, environmental sociology is currently little developed in France. Whole sections are not studied, such as environmental advocacy and representations of natural environments. The case of green tides is an exception. Territorial transformation is no longer perceived as merely biophysical. Sociological aspects are starting to be taken into account, calling for differentiated approaches depending on the socio-ecosystems and their different spatial scales, and involving various stakeholders to tackle the issue of eutrophication.
6.2. Obstacles and future areas of investigation

6.2.1. Moving towards systemic research approaches

There is still a lack of highly inclusive research at territorial level to meet the different management issues of watershed heads, riparian corridors and coastal areas. Remediation of eutrophication should therefore strive towards systemic approaches integrating hydrosystems, agricultural and urban areas, and production, feeding and recycling practices. Generally, the issue of agricultural transition is closely related to that of eutrophication. Models combining both biophysical and economic aspects need to be developed as a discussion basis for the definition of remediation programmes, despite the uncertainties related to imperfect – albeit constantly improving – knowledge of biophysical, ecological and socio-economic phenomena. The relation between changes in eutrophication and changes in socio-ecosystems should also be better put into perspective, going beyond sector-related focuses such as that placed on agriculture in recent decades. Sharing knowledge can recreate bonds between social groups and business sectors which are currently set apart from each other. There needs to be a greater number and variety of interdisciplinary investigation sites (lakes, rivers, coastal areas) where biophysical and societal dynamics could be studied over the long term, and existing investigation sites should be perpetuated. Cooperation between biophysical and human and social sciences must be strengthened, paying special attention to the successful integration of disciplines working from quantitative and qualitative data.

6.2.2. Use of existing data and avenues for improving frameworks and regulatory monitoring networks

Although there have been many French experiments in remediation, going back a long way, the results are often buried in a grey literature of varying quality and difficult to access. They are likely to fade from collective memory if something is not done about it very quickly. One of the challenges lies in analyzing these experiments and publishing summaries in international journals. Long-term series provide invaluable information and legacy data need to be digitized, stored and made readily available. Analyzing trajectories of public issues according to the diversity of systems and governance methods implemented, especially on a European and international scale forms an integral part of the analyses of these experiments. Lastly, any existing series must be leveraged and interpreted (e.g. data on cyanobacteria blooms measured at national level within healthcare networks, etc.). Reflection on the regulatory dimension, its limitations in terms of effectiveness and applicability, and its potential harmonization, not only between environments, but also between neighbouring countries, is a key area of research for future investigation: how can one take better account of the land-sea continuum in which the eutrophication process occurs at varying degrees? Another issue is how to make better use of the information collected in monitoring networks, while making a clear distinction between four separate functions for the use of data: (1) compliance with applicable standards, (2) regular statistical reports on the environments and their spatial and temporal changes, (3) developing knowledge of a given theme and (4) monitoring the effectiveness of remediation actions. When the first three functions are assigned to the same data sets, there can be some lingering confusion and misunderstanding. The quality of the data mobilized (type, accuracy, spatial and temporal representativeness), both for chemical status variables and for biological variables, is essential: pressure data (past and present), chemical data, and their geographical contextualization. These reflections will make it possible to go beyond existing approaches, which define eutrophication thresholds on a mainly physico-chemical basis. Regarding the standard of 50 mg/L of nitrates, it clearly relates to water potability rather than to preserving environments from the eutrophication process. Before the implementation of the WFD, a consensus grid had been proposed with various value guidelines, in the range of 2 to 50 mg/L of nitrates. It would be interesting to analyze the historical trajectory of these values and their territorial applications. Transparency on assessment criteria and the related educational approach are
essential to set threshold value ranges. Scientific boards and interfaces between science and society should be put in place so as to create opportunities for discussion in the implementation of these diagnosis and remediation approaches. While eutrophication issues can be framed by normative approaches, their adaptability to the characteristics of watersheds must also be considered as part of a process of advancing knowledge and improving diagnoses and adaptive management. In France, in the field of water, citizen science has not yet gained much traction. And yet participatory monitoring by citizen scientists could contribute to growing knowledge, notably in the case of transient episodes undetected by existing monitoring networks. It could also have educational virtues in that it would initiate and encourage dialogue between scientists and volunteers.

Analyzing the social, health and cultural impacts of eutrophication requires undertaking case studies and producing highly contextualized knowledge, which is lacking today. From the perspective of social sciences, important factors to be considered are social action, conflicts and the multiple forms of problematization that come with the emergence of eutrophication as a public issue, on the one hand through its most harmful consequences, and on the other hand through the profound transformation in the social representations associated with ecosystems in today’s society. Existing work shows that the range of policy tools available and the effectiveness of policies depend on taking these factors into account.

The monetary assessment of eutrophication impacts remains a real question for research. Few references are available and the long-term impacts are difficult to estimate.

6.2.3. Developing a methodology for analyzing the eutrophication risk

Concerning study and monitoring systems, a debate should be initiated to identify future areas at risk and define what type of operational and scientific action can be undertaken in addition to existing measures. This is becoming increasingly necessary given the possible exacerbation of eutrophication phenomena in the future under the combined effect of global changes. An analysis of the literature suggests that a risk analysis methodology should combine 1) hydro-biogeochemical transfers and transformations along the land-sea continuum, 2) ecosystem vulnerability factors, and 3) climate hazards.

Taking into account the variability of inputs, residence times and nutrient transfers in watershed heads and more generally along the land-sea continuum is a challenge at both theoretical and applied levels. It requires stepping up soil and groundwater monitoring and, for this purpose, developing indicators on phosphorus mobility and nitrogen transfers in headwaters. Changes in soil and water need to be monitored as closely as possible to where action is taken so as to ensure their effectiveness. Another challenge lies in strengthening flow monitoring and assessment methodologies by analyzing the effect of measurement frequencies, periods and optimal zones, as well as uncertainties, and by combining data acquired by high frequency sensors. Future technologies, whether ground-based or airborne, will open up fields of investigation compared to existing situations, with either a high temporal frequency or a broader spatial extent, for both soils and waters. These technologies will have to be built into new monitoring strategies, while the water sampling strategy will also need to be revisited. Deterministic modelling will take an increasingly realistic approach to physical, chemical and biological factors and will continue to be one of the key tools in exploring nutrient inputs and climate scenarios. Another benefit it offers is the possibility of sequencing eutrophication models for various aquatic environments from upstream to downstream, i.e. from headwaters to the ocean. Its forecasts will need to be matched with confidence intervals.

Vulnerability needs to be defined by taking into account the entire direct and indirect causal chain that influences the inherent properties of the receiving aquatic ecosystem, in relation to the diversity of local situations and past and present contexts. The possible improvements in characterizing this vulnerability focus on several aspects. On the one hand, based on the extraction and calibration of information provided by biological indicators: (1) better understanding how some metrics already used in biological indicators or new metrics signal significant swings towards eutrophic situations, (2) identifying these swings in the trajectories of situations closely monitored over the long term and interpreting them in functional terms (nutrient flows and concentrations versus relations between biological groups). In parallel, some paradigms need to be revisited, particularly the relationship between C and Si nutrients and the production of plant biomass, which appears to be more complex when we seek to better
understand the determinism of algal blooms, community changes and trophic adjustments. For this purpose, works combining hydrology, geochemistry, physiology and ecology are required. 

**Global changes** involve changes in hydrological, sedimentary and thermal dynamics and changes in land use and human activity. Taking account of climate hazard is therefore essential in a methodology for characterizing the eutrophication risk. Determining the respective roles of climate and human activity is a central field for research. Modelling can contribute to advancing this work as a complement to and building on long-term observation. More information is needed on interactions between the climate and eco-physiological, biogeochemical (biotransformation in soils, transitional wetlands and aquatic environments), hydrology (connectivity, distribution of residence times and transfers) and ecological (food web) processes. The increase in CO₂ concentrations in the air must be taken into account, as it could stimulate productivity in terrestrial and aquatic ecosystems while intensifying cyanobacteria blooms. These elements create great uncertainty as to future changes in aquatic ecosystems, particularly in countries of the northern hemisphere. The **role of invasive species** in freshwater eutrophication should also be covered by investigations. The Loire is a case in point, but other rivers in western France, subject to these invasions, are sites to be studied in priority provided the data series are sufficiently rich (nutrient flows, flows of other biotic compartments) to carry out a functional systemic study.
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I - Context and issues

Eutrophication phenomena and the role played in these phenomena by nutrient inputs and their interaction with other factors are currently poorly or inadequately understood. Knowledge of these phenomena is not consolidated and still less shared. The ministries in charge of ecology and agriculture have therefore mandated a joint scientific appraisal (expertise scientifique collective, or ESCo)\(^1\).


In light of these issues and given the deadlines related to the timetables for implementation of the directives, the joint scientific appraisal will provide validated information that will help inform these public policies.

II - Purpose and scope of the joint scientific appraisal

A particular feature of eutrophication is that it is a concept used both by the scientific community and in public policies, for which there are multiple definitions. The appraisal will clarify the definition of eutrophication by taking into account operational issues and needs for public action. The analysis will produce a critical situational analysis of the scientific knowledge certified at European and global level on the causes and mechanisms of water eutrophication and identify the scientific obstacles that require acquiring new knowledge that could be the subject of further research.

It will cover the land-water continuum, i.e. the system of transfers from watersheds to the aquatic ecosystems included in the concept of watershed, insofar as they can be used to characterize the risk of eutrophication. The scope of the joint scientific appraisal does not include a detailed analysis of the impact of human activity (agricultural systems, water treatment methods, etc.) on eutrophication.

The appraisal will serve as a scientific basis to improve the consistency of the terms of implementation of the directives concerned (Nitrates Directive, WFD, MSFD, and UWD).

III - Questions and scientific themes

1. Situational analysis

On the basis of scientific publications, a global situational analysis of the forms and manifestations of eutrophication and related issues will be drawn up before focusing on the case of France (mainland and overseas departments and territories). This analysis will provide a context and link the theoretical analysis to the issues identified in France. Based on the analysis of the scientific literature, the joint scientific appraisal will draw up, on the basis of the knowledge available:

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1. This joint scientific appraisal was notably announced at a meeting on the Nitrates Directive with representatives of the agricultural profession on 24 July 2014.
a) A SITUATIONAL ANALYSIS OF THE FORMS OF EUTROPHICATION

- The forms of manifestations of eutrophication.
- A census and description of the environmental, economic and social impacts:
  - Environmental impacts (LCA and multi-criteria approaches).
  - Economic and social impacts of eutrophication phenomena and policies to combat eutrophication (cost of damages, cost-benefit analysis, etc.).

b) A SITUATIONAL ANALYSIS IN FRANCE

- The forms of eutrophication observed in France: manifestations of eutrophication (places and periods, conditions, etc.) and related issues (toxicity, trophic imbalances, etc.).
- The modalities of eutrophication monitoring in France: status of monitoring programmes, available data, degree of knowledge on the eutrophication status of waters and criteria applied.
  - Mobilizable data (national, regional and local) on water quality, biological monitoring, the structure of river systems, land use, etc., their relevance and quality. Analysis of gaps and difficulties, including spatial and temporal.

2 - Definition of eutrophication, causes and symptoms

The appraisal will make an analysis of the biotic and abiotic mechanisms responsible for eutrophication and identify the factors that are related to it. It will give a scientific definition of eutrophication, emphasizing the complex and multifaceted nature of this process and its manifestations in the main types of continental and marine aquatic ecosystems. This analysis will be balanced with a legal analysis of the definition of eutrophication in directives. It aims to enable public authorities to propose a relevant definition of eutrophication from a scientific point of view, adapted to the issues identified and making it possible to meet the regulatory requirements related to the implementation of European directives.

a) SCIENTIFIC DEFINITION

- Scientific definitions of eutrophication: the various forms of eutrophication (green tides; phytoplankton; etc.), description of related mechanisms, criteria (algae communities, abiotic factors, etc.) characterizing ecological disruptions.
- Factors responsible for eutrophication: identification of predominant environmental factors, variations according to context.
- Symptoms and manifestations of eutrophication: consequences on the various aquatic ecosystems and related risks.

b) LEGAL OR REGULATORY FRAMEWORK

On the basis of regulatory documents that will notably be provided by the Ministries for Ecology and Agriculture, the joint scientific appraisal will analyze the legal framework(s) relating to eutrophication: based on the publications, how the concept of eutrophication is defined in the regulations, what is their complementarity, and what are the criteria taken into account.
3 - Characterisation of environments’ vulnerability to eutrophication: appropriate tools and methods

The goal is to provide the public authorities with relevant information to help in defining the criteria used to characterize the vulnerability of aquatic environments to eutrophication and in selecting appropriate methods. These scientific issues will be dealt with separately for the specific cases of continental eutrophication and coastal and marine eutrophication.

3-1 Vulnerability of aquatic environments
a) Inventory of scientific knowledge on the most relevant criteria to characterize the vulnerability of aquatic environments to eutrophication (continental, coastal and marine eutrophication)
b) Inventory of existing methods or models to assess the vulnerability of environments to eutrophication, scope of validity and relevance
c) Identification of appropriate criteria and methods to characterize the vulnerability of continental and marine environments to eutrophication in relation to their richness in nutrients
   • Role played by nutrients in eutrophication phenomena, concept of risk linked to nutrients present in the environment and risk gradation, evaluation of nutrient flows in the river system and marine environment
   • Changes and probable developments with climate change
   • Existing studies and models, thresholds proposed by the literature: analysis of variabilities and relevance
   • Identification of the most appropriate criteria to measure nutrient levels in the environment (when and where to measure, how often, etc.), in connection with the biological functioning of ecosystems and the behaviour of nutrients in the environment

3-2 Risks associated with transfers in watersheds
How can the environmental eutrophication risk be characterized in connection with the properties of watersheds in relation to nutrient transfers?
Estimation of stocks and residence times of nutrients in watersheds.
Critical analysis of coupled and uncoupled models of carbon, nitrogen and phosphorus according to various scales and contexts.

4 - Overview and identification of needs in terms of knowledge

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Eutrophication affects many lakes, reservoirs, rivers and coastal areas in France and around the world. It generates major disruptions to aquatic ecosystems and has impacts on related goods and services, on human health and on economic activities. In some areas, it has become an urgent societal issue. Debates on the identification of the factors and risk levels of eutrophication, seeking to guide public policies, have led the ministries in charge of the environment and agriculture to mandate the CNRS, Ifremer, INRA and Irstea to conduct a joint scientific appraisal (Expertise Scientifique Collective, or ESCo) on the subject. This joint scientific appraisal establishes a critical situational analysis of the scientific knowledge available at the international level on the causes, mechanisms, consequences and predictability of eutrophication phenomena. It identifies concepts considered as certain and those still subject to uncertainties, gaps, issues subject to scientific controversies, as well as the existing action levers to curb eutrophication.