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Eutrophication: causes, consequences and control

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Foreword

The book "*Eutrophication: Causes, Consequences and Control*" is an ambitious and laudable attempt to summarize different aspects of eutrophication, highlighting both the extent and severity of the phenomenon in different parts of the world as well as efforts to control or mitigate its biological effects. The 19 chapters of the book also underscore fundamental differences in the way that different aquatic ecosystems respond to over-enrichment and stoichiometric imbalances of nutrients, an observation that has been amply reinforced in documented studies of Lake Washington in the USA, the classic experimental lake program studies in Canada, and studies of the Chesapeake and Delaware bays in the USA.

Different chapters are built upon different objectives and offer a variety of spatial and temporal scales; some of them offer inter-disciplinary approaches, newer methods to study the problem, and linkages/interactions with other stressors, such as climate change, drought or arid climates, dam projects, waterborne disease vectors, and heavy metal contaminants. Still the primary messages from the book chapters may be summarized as follows:

- Environmental concerns associated with nutrient-enhanced eutrophication are quite varied and potentially severe; they include low or nearly non-existent dissolved oxygen in the water column, changes in aquatic food webs, biomass and diversity, alteration of sediment geochemistry, contributions to smog and greenhouse effects, and loss of amenities and ecosystem services.
- Economic factors have played a big role as both a cause and a consequence of nutrient over-enrichment and consequent eutrophication in aquatic ecosystems; however, economic analyses have not factored into many eutrophication-related analyses.

In well-studied eutrophic waterbodies, it has been possible to estimate maximum allowable amounts of nutrients to attain water quality objectives, such as concentrations of nutrients close to natural levels, clean and clearer waters, natural levels of algal blooms and dissolved oxygen, and natural distributions and abundance of plants and animals. For example, a recently adopted Baltic Sea Action Plan (2007) calls for limiting the total input of nitrogen to 600,000 t of nitrogen and 21,000 t of phosphorus. These values represent substantial reductions when compared with long-term average input of nitrogen (737,000 t) and phosphorus (36,000 t) into the sea. The signatories to the plan will be expected to develop measures and implement activities that would attain their designated targets, including management of agricultural runoff, improved sewage treatment, and reduction of phosphorus in detergents.

In summary, the book offers a glimpse of eutrophication research and management approaches in the developed and developing countries and recognition of its impacts on renewable resources and amenities. There is clearly a need to sponsor and support research on eutrophication and related factors that take into consideration the hydrology, atmospheric flows, and climate conditions typical of the study regions, preferably within an integrative framework and explicitly linking the study outputs to the desired management outcomes.

National Oceanic and Atmospheric Administration Jawed Hameedi
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Preface

Adopt the pace of nature: her secret is patience.
Ralph Waldo Emerson 1803–1882

Degradation of water quality, loss of biodiversity, increased waterborne disease transmission, excess sedimentation, and major trophic cascades from altered biological food webs are of major concern at a time when global water supply is facing rapidly increasing demand. A recent report indicates that an enormous amount of our global water supply, approximately 10,800 km^3, has been impounded in the world's artificial reservoirs alone. Many of the new reservoirs have resulted from hydroelectric dam projects. Much of the total water impounded is located in sub-tropical and tropical regions with a very high probability for accelerated eutrophication and water quality degradation.

Eutrophication is the natural process driving the ecological succession of freshwater, estuarine, and marine ecosystems. Although eutrophication involves the integration of complex biological and geological processes that are often studied as individual entities, many useful studies have involved larger scale studies of the land use changes, aquatic biota, and biogeochemical cycles that regulate the structure, function, and gross productivity of human-impacted aquatic ecosystems.

The natural eutrophication process is extremely variable and subject to major site-specific characteristics such as nutrient stoichiometry, biodiversity, climate-related factors, and geomorphology. Human or cultural influences have profound ecological effects on the natural functions and rates of many of the major processes in the aquatic processes influencing eutrophication and often result in accelerated rates of eutrophication that produce significant changes in the structure and function of aquatic ecosystems.

The result of human activities that have increased the rate of eutrophication in different aquatic ecosystems around the world is the main topic of this book. We hope it will be useful to aquatic ecologists, engineers, and water quality scientists and managers faced with both the traditional challenges of accelerated eutrophication and new challenges catalysed by global climate change. This book provides case histories of eutrophication events in sub-tropical, tropical, and temperate zone freshwater, estuarine and marine ecosytems along with studies on the biomanipulation, phytoremediation, restoration, and control of impacted ecosystems.

The editors and contributing authors hope that the results of publishing this book will include a practical update on our knowledge of eutrophication on a global scale and lead to new discussions and efforts to deal with the threat of accelerated eutrophication during the process of global climate change.

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Chapter 1

Eutrophication and Climate Change: Present Situation and Future Scenarios

Martin T. Dokulil and Katrin Teubner

Abstract Anthropogenic eutrophication still is a major threat to inland waters in large parts of the world although re-oligotrophication has largely progressed in several regions. Climate change now poses a new risk effectively endangering inland waters. Impacts of climate change on inland waters are already well documented and established. Direct effects are mainly through changes in temperature and associated physico-chemical alterations. Indirect effects mediated through processes such as extreme weather events in the catchment include increased nutrient loading among others. The combination of all impacts will lead to severe and significant changes in the physical structure and the biological configuration of the waters depending on future climate scenarios. Since water quality and quantity will be affected, concern is necessary. Possible adaptation and mitigation mechanisms are discussed.

Keywords Eutrophication · Climate impacts · Inland waters · Rivers

1.1 Preamble

Historically, the German Weber (1907) invented the terms *eutrophe, mesotrophe*, and *oligotrophe* to describe different states of the flora in peat bogs as they changed with time through accumulating material. Weber called the first bog vegetation eutraphent, requiring high concentrations of essential elements in the soil. As the bog level built up above its surroundings, and became more leached an oligotraphent flora covered the bog, composed of species tolerating very low nutrient concentrations at the end. This process in bogs is essentially the reverse from what happens in freshwaters. Naumann (1919) used Weber's terms to characterise waters in Sweden as oligotrophic, mesotrophic, or eutrophic depending on their content of mineral nutrients similar to freshwaters or brackish and marine waters depending on their salt content. Since hydrochemical techniques at that time did not allow nutrient concentrations to be reliably measured, Naumann redefined the categories according to the appearance of lakes in summer and their most abundant types of phytoplankton algal groups. Other studies of European lakes supported the idea about distinctive floras. The desmids were seen as the characteristic phytoplankters of oligotrophic lakes in mountainous regions while diatoms and cyanobacteria were seen as the characteristic phytoplankters of eutrophic lowland lakes. Analysing data from Northern Germany, Thiennemann (1918) showed that some stratified lakes may experience severe oxygen depletion in their hypolimnion. These findings, together with Naumann's definitions, resulted in the *oligotrophic–eutrophic* paradigm (Fig. 1.1).

In this paradigm an oligotrophic lake is deep receiving nutrient-poor effluents from its drainage basin. Production of organic matter is small in the well-illuminated epilimnion. Therefore, the amount of material sinking into the hypolimnion is small and little oxygen is consumed there during the summer. In contrast, a eutrophic lake is often, but not necessarily, shallower, the drainage basin is richer, and rivers

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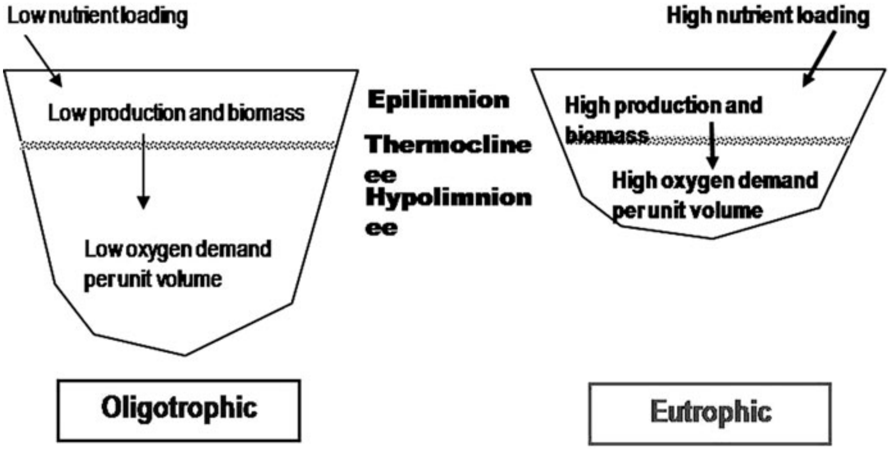

Fig. 1.1 The *oligotrophic–eutrophic* paradigm. For explanation refer to the text

and groundwater discharging into its epilimnion therefore contain more nutrients. Primary production is higher compared to the oligotrophic situation. Hence, more organic material settles into the hypolimnion, which gets more easily depleted of oxygen. As a consequence, the deep water layer of an eutrophic lake becomes anoxic during summer. Following this paradigm, eutrophication in freshwater systems can be defined as a syndrome of ecosystem responses following the increase in concentration of chemical elements (nutrients) to an extent that the primary productivity of the ecosystem is enhanced. The nutrient enrichment can be natural or artificial, usually caused by human activities. Depending on the degree of nutrient enrichment, many subsequent negative environmental effects may occur. The EC definition in the Urban Waste Water Treatment Directive (1991) applies equally to freshwaters. Here eutrophication is 'the enrichment of water by nutrients especially compounds of nitrogen and phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms and the quality of the water concerned'. Nutrient enrichment tends to stimulate the development of phytoplankton biomass development in lakes because micro-algae and cyanobacteria usually grow faster than larger algae or plants. The resulting biomass absorbs light and so shades out benthic micro-algae or macrophytes. In flowing waters, phytoplankters tend to be washed downstream and attached plants and benthic algae may be stimulated instead. Since algal growth primarily determines the trophic status of lakes, it is best defined in terms of primary production – i.e. the amount of organic carbon produced by photosynthesis within an annual cycle. Rodhe (1969) delineated trophic levels by the carbon uptake in g C m^{-2} yr^{-1}:

oligotrophic 7–25
eutrophic (natural) 75–250
eutrophic (polluted) 350–700.

Because the measurement of primary production requires specialized skills and equipment, the trophic status has also been defined, using more easily measured variables such as the concentration of the limiting nutrient (usually P or N), chlorophyll-a and Secchi-disk transparency to categorize the trophic status of freshwaters (OECD 1982).

1.2 The Wax and Wane of Lake and River Eutrophication

As described in the preamble, the discovery of trophic development of freshwaters and their terminology dates back to the early twentieth century. The consequences of anthropogenic-induced eutrophication of freshwaters, however, were not anticipated until they became evident in the 1940s and 1950s. Severe deterioration of surface waters at that time raised remarkable public concern and triggered expanding scientific interest. Starting from the pioneering work of Vollenweider (1968) and an early overview by Rohlich

(1969), an enormous number of books, research publications and articles dealt with water pollution, eutrophication and related management issues (e.g. OECD 1982, Henderson-Sellers and Markland 1987, Ryding and Rast 1989, Harper 1992, Thornton et al. 1999). After an initial debate on which nutrient is primarily responsible for limiting productivity in lakes and rivers/lakes, known as the limiting nutrient controversy (Likens 1972), freshwater scientists have largely concentrated on phosphorus as a key element in controlling eutrophication.

Potential effects of cultural eutrophication, caused by excessive inputs of phosphorus and, to a lesser extent, nitrogen to lakes, reservoirs, rivers and coastal oceans include:

- Increase of biomass of phytoplankton and macrophyte vegetation
- Shift to bloom-forming algal species that might be toxic or inedible
- Increase of biomass of benthic and epiphytic algae
- Change in species composition of macrophyte vegetation
- Increase of biomass of consumer species
- Increase of incidence of fish kills
- Reduction in species diversity
- Reduction in harvestable fish biomass
- Decrease in water transparency
- Oxygen depletion in the water body
- Taste, odor, and drinking water treatment problems
- Decrease in perceived aesthetic value of the water body

In the late 1980s eutrophication became less fashionable and was replaced by focusing on other freshwater issues than nutrient enrichment problems. The declining perception of the eutrophication problem by the public and the politicians was largely a result of the success of the restoration measures. It improved water quality through extensive invention of treatment plants for sewerage and waste water decreasing mainly organic load and particularly of phosphorus which was identified as prime target. As a consequence of the reduction in anthropogenic nutrient load and additional measures, eutrophication in many lakes in the industrialized countries was effectively stopped and reversed (Sas 1989). The declining in-lake concentrations of chemical elements, particularly of phosphorus ultimately resulted in re-oligotrophication which can be described by empirical models (Jeppesen et al. 2005, Jensen et al. 2006). Most of these undoubtedly great achievements in lakes largely resulted from the consequent treatment of point sources and removal from the catchment (EC 2002). At the same time eutrophication of rivers became more evident because of the effective reduction of organic load, again as a result of large-scale sewage treatment.

Early water quality studies in running water systems focused on organic load, mainly carbon enrichment from untreated sewage. Excessive emissions of organic waste made rivers completely anoxic and still does in many developing and industrializing countries. Later on, as sewage treatment progressed and streams became less polluted, it became evident that nutrient enrichment can also occur in rivers and streams as a direct result of human alteration of land use. Depending on geology, topography, and land use, various regions are expected to have distinct baseline amounts of nutrients. Based on observations in the United States, Omernik (1977) defined the concept of *nutrient ecoregions*. Naturally occurring reference conditions form the basis of many current efforts to regulate stream nutrients.

Compared to point sources, the management of diffuse sources is far more problematic because of the difficulty of controlling nutrients as runoff water from agricultural land and urban areas (Thornton et al. 1999). In terms of annual loads however, most P transfer from land to water tends to occur in runoff and erosion during winter storm events. This phosphorus from diffuse sources may therefore play a relatively minor role in eutrophication in rivers because the timing of the transfers does not usually coincide with the period of maximum biological demand. Instead, an important element in understanding eutrophication may lie in identifying and quantifying P sources during periods of low flow. Undesirable symptoms of river eutrophication primarily occur during the plant growing season (spring and summer), when low flow, high water residence times, abundant light levels and high water temperatures promote rapid algal growth. During the growing season, the eutrophication risk in rivers mainly originates from point discharges, a major source of high concentrations of dissolved, bioavailable phosphorus fractions. At times, when diffuse agricultural runoff contributions are in general relatively low, river water P concentrations from point sources become even higher, as a result of reduced effluent dilution. Moreover, these phosphorus concentrations may remain well above levels likely to bring about ecological improvement, even after P-stripping of the treated effluents (Jarvie et al. 2006). According to Meybeck (1982, 2003) streams and rivers have worldwide doubled their nitrogen and phosphorus concentrations, with local increases of up to 50 times. Overall, artificial eutrophication of rivers is a widespread phenomenon which has, however, received far less attention than eutrophication of lakes. This is partly because effects of elevated nutrient levels in rivers are often less dramatic since factors other than nutrients limit algal growth. Despite some progress, there is still no conceptual understanding of how eutrophication develops in rivers and streams. Hydraulic flushing of nutrients, light limitation and water velocity are certainly important in controlling algal growth interacting in complex ways. This suggests that short retention time rivers (<3 days) will show different effects compared to long retention time impounded rivers or riverine lakes (>3 days). Based on a comparison with lake systems (>30 days retention time) and considering some of the factors motioned above Hilton et al. (2006) proposed a conceptual model of how eutrophic conditions develop in rivers. Since many natural streams are net heterotrophic, Dodds (2006) proposed to divide the trophic state of rivers into autotrophic, nutrient regulated and heterotrophic, external carbon controlled state. The autotrophic state in flowing waters depends mainly on phosphorus and nitrogen levels. Algal biomass is positively correlated to gross primary production in streams and rivers. Based on the nutrient reference concept, boundaries for trophic levels were developed by Dodds (2006).

A further general problem in eutrophication can be the (co)limitation of nutrients other than phosphorus, most often nitrogen (Howarth and Marino 2006). Recently, the interest and the debate on the role of nitrogen in eutrophication were revived because high levels of nitrogen in rivers are increasingly recognized as a driver of eutrophication in estuaries and coastal oceans (Schindler 2006). Therefore Conley et al. (2009) argue that it might be necessary and in most cases essential to implement a dual nutrient-reduction strategy when implementing measures to control eutrophication. A focus on only P or N reduction should not be considered unless it is clear that downstream ecosystems will not be affected. This paper in fact has started a lively debate on the subject. Schindler and Hecky (2009) argue for more data and the demonstration that removal of nitrogen from sewage is effectively reducing eutrophication before implementation is imposed on society particularly in the developing countries. This argument is supported by Bryhn and Håkonson (2009) who aim for modeling and prediction before acting. Schelske (2009) focuses on P-only strategies, while Jacoby and Frazer (2009) claim to adjust expectations because even reduction of both nitrogen and phosphorus may not yield the desired response.

Another question is if groundwater phosphate can be an unrecognised driver for eutrophication? In contrast to nitrate which is a problem in many areas, the common assumption for phosphorus is that groundwater is usually relatively low in P because little P is expected to be transferred from soils to groundwater. In fact, however, a number of anthropogenic sources can be responsible for elevated P levels in groundwater (Carlyle and Hill 2001, Whelan 2008). These include:

- ‘saturated’ P concentrations in agricultural soils resulting from historical application of mineral fertilizer and manure in excess of crop demands
- manure heaps and unlined slurry storage facilities
- leaking main water pipes
- leaking sewers
- leaking septic tanks
- leaking waste disposals

Eutrophication problems worldwide have been summarized by UNEP (1994) and are depicted in Fig. 1.2. In Spain for example, 80% of the lakes, 70% of the reservoirs and 60% river sites were eutrophic in the 1990s with hypertrophy increasing downstream (Alvarez Cobelas et al. 1992).

For the European countries, a recently published assessment by the EEA (2009) concludes that P concentrations in rivers and lakes are usually low in northern and alpine countries. Countries in central and southern Europe have generally higher proportions of rivers and lakes with high phosphorus concentrations (Fig. 1.3).

Both soluble reactive phosphate (SRP) and total phosphorus (TP) showed statistically significant decreasing trends between 1992 and 2005 in about 34 and 18% of the monitoring stations for rivers and lakes, respectively. These reductions were possible because of the advancements in the scientific understanding and

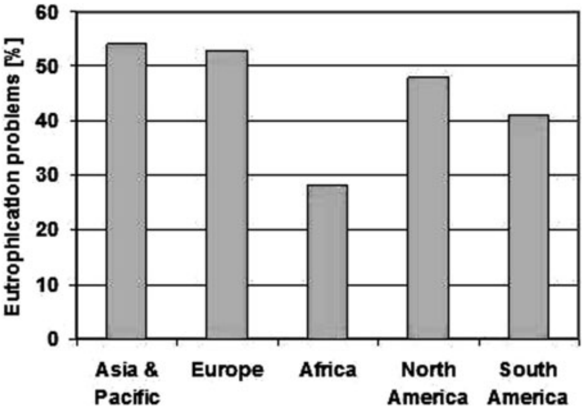

Fig. 1.2 The percentage of lakes and reservoirs with eutrophication problems in the 215 lakes and reservoirs covered by the UNEP/ILEC surveys (UNEP 1994)

management of eutrophication since the late 1960s. The control of point sources of phosphorus-reduced algal biomass prevented blooms in many lakes. Diffuse nutrient sources from land use changes and urbanization in the catchments of lakes have proved possible to control but require many years of restoration efforts. Internal nutrient loading can be controlled by reducing external loading, although the full response of lakes may take decades. Additional in-lake treatments or biomanipulation can help to shorten the time needed for full restoration.

The successful story of eutrophication control in freshwaters involves reduction of P inputs primarily from external sources, such as sewage, and additionally the internal, recycling of phosphorus from sediments in lakes. New studies are necessary to determine the abilities of estuaries, coastal marine systems or engulfed seas (e.g. the Baltic Sea) to recover from enrichment with both nitrogen and phosphorus (Schindler 2006, Smith and Schindler 2009). In the developing world however, most cities have no or limited sewerage systems. To construct, manage and

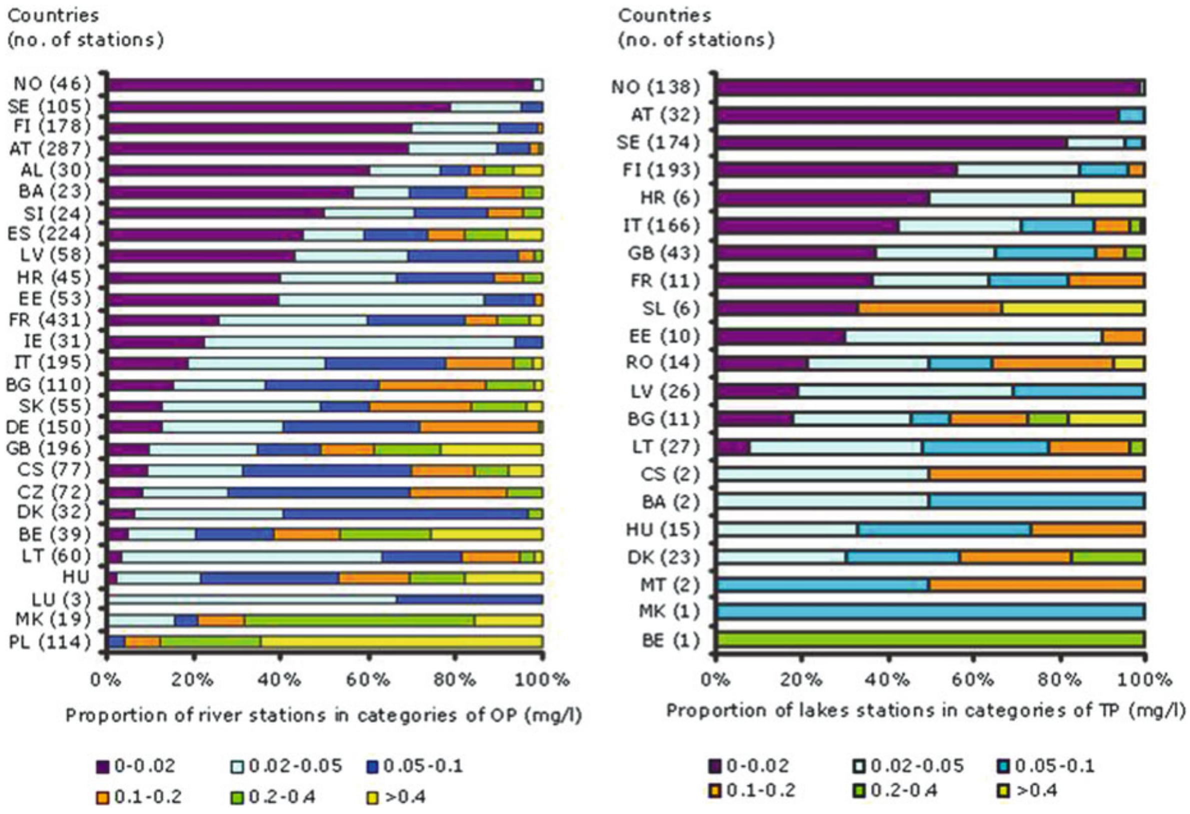

Fig. 1.3 Concentration of phosphorus in rivers (*left*; soluble reactive phosphate) and lakes (*right*; total phosphorus) in European countries. The number of monitoring stations in each country is given in brackets. Figures are based on the most recent year for which data are available: this is 2005 for all countries except for France (lakes: 2000), Czech Rep. (lakes: 2003), Bulgaria and Denmark (rivers and lakes: 2004). Reproduced from EEA (2009)

maintain such systems in the developing countries, large investments of money and time are necessary. Even then, it will take many years or even decades to restore or recover freshwaters because many inland waters in the developing countries are extremely eutrophic or polluted. Many of the native plants and animals they once supported have now disappeared. These systems do not function properly and do not provide service any more. In several regions erosion of soil particles into streams and lakes is an important driver of eutrophication particularly when phosphorus concentrations are high in the soil. In agricultural regions, the flux of phosphorus from over-fertilized soils may be even more important for eutrophication. This type of eutrophication is not easily reversible. Substantial changes in soil management, reducing erosion rates or technologies for P reduction of enriched soils are needed to improve water quality (Carpenter 2005).

1.3 Evidence of Climate Change – Does It Matter?

Although there are still a large number of skeptics who deny significant human-induced global warming, evidence is growing and substantiated that it is so as uncertainties become smaller (see e.g. the latest IPCC report 2007). Warming of the climate system is indisputable, as it is evident from observations of increases in global average air and sea surface temperatures (Fig. 1.4) as well as many other variables (Burroughs 2001, Pittock 2005). Moreover, twenty-first century global warming projections far exceed the natural variability of the past 1,000 years and is greater than the best estimate of global temperature change for the last interglacial (Crowley 2000).

Eleven of the last twelve years (1995–2006) rank among the warmest years in the instrumental record since 1850 of global surface temperature. The temperature increase is widespread over the globe (Easterling et al. 2000, IPCC 2007) but is greater at higher northern latitudes and in the Alpine region (Giorgi et al. 2001, Beniston et al. 1994). Temperatures in these regions have increased at almost twice the global average rate in the past 100 years.

Key drivers of climate change are greenhouse gas emissions (GHGE) which have particularly increased through anthropogenic forcing. Freshwater ecosystems have an important role in these emissions (see e.g. Tremblay 2009). Alterations in the climate have far reaching consequences for freshwaters. All observational records and climate projections provide abundant evidence that freshwater resources are vulnerable and have the potential to be strongly impacted by climate change, with wide-ranging consequences for human societies and ecosystems (Bates et al. 2008).

Major concerns related to water therefore include (Pittock 2005):

- *Risk to unique or already threatened ecosystems*: Wetlands, ponds and unique water bodies will be at risk due to changing hydrology. Of all ecosystems, freshwater aquatic ecosystems appear to have

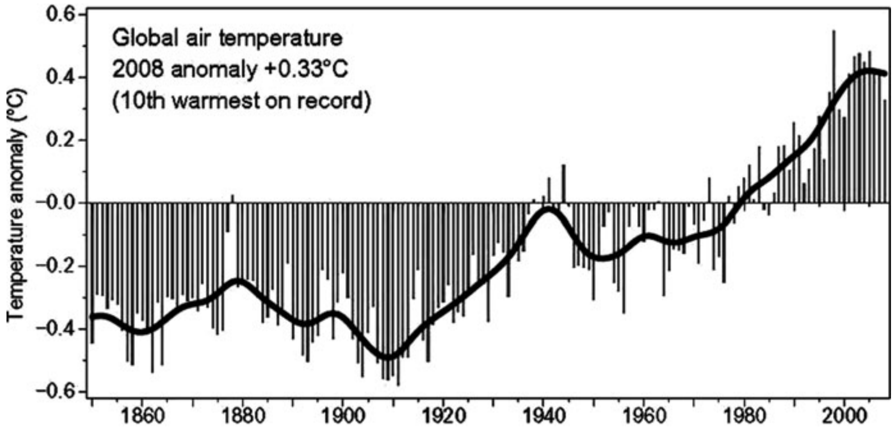

Fig. 1.4 Time series of the combined global land and marine surface temperature record from 1850 to 2008. From: Phil Jones, http://www.cru.uea.ac.uk/cru/info/warming/ (Brohan et al. 2006)

the highest proportion of species threatened with extinction by climate change.

- *Risks from extreme weather events*: The frequency and severity of extreme events are expected to increase which will likely have direct and indirect impacts on freshwater ecosystems.
- *Distribution of impacts*: The likely impacts of climate change will not be distributed equally. Natural water resources in low-latitude and/or developing countries will likely be more severely and earlier affected than those in mid- and high-latitude developed countries.

Climate change has already altered the water cycle in the United States affecting where, when and how much water is available (Karl et al. 2009). Global warming will affect surface water quality and groundwater quantity and will impose additional burdens on already stressed water systems. Therefore, the past century can no longer be a reasonable guide to the future for water management.

The anthropogenic global warming signal will partially be offset by internal variability for the next few years. However, climate will continue to warm, with at least half of the years after 2009 predicted to exceed the warmest year currently on record (Smith et al. 2007).

1.4 What Do We Know About Climate Impacts on Inland Waters?

Already quite some knowledge on climate impacts on freshwaters has been accumulated over the past two decades. Especially for Europe, these findings have been summarized by Eisenreich et al. (2005), George et al. (1998) and George (2009). A summary of the impact and the consequences of climate change on lakes in Central Europe is provided by Dokulil et al. (2009). The ecological consequences of a warmer climate have a large variety of facets depending on the geographical position on earth and the different seasons of the year. Ice cover of lakes and rivers during winter is a key component of cold regions. Ice often controls aquatic habitats, biological productivity and species diversity. It also poses major challenges such as floods, for example (Prowse et al. 2007). Reductions in freshwater-ice cover have largely mirrored trends in air temperature. Changes in the duration of ice cover are characterized by later ice-on and earlier ice-off dates. Over the last 150 years, freeze dates on average moved later by 5.8 days per 100, and breakup dates were 6.5 days per 100 years earlier. Variability in both freeze and breakup dates has increased since 1950 (Assel and Herche 1998, Magnuson et al. 2000). The shifts in the timing of ice breakup in temperate regions have now been substantiated in many cases (e.g. Livingstone 2000, Dokulil and Herzig 2009). In colder zones, these shifts can be less dramatic resulting in large differences in timing along regional temperature gradients. The date of ice breakup determines the timing of many physical, chemical and biological lake processes, which may lead to an alteration in the diversity of lake types in many areas (Weyhenmeyer et al. 2004). Knowledge of how climate affects ice processes such as ice-cover composition, thickness and breakup dynamic remains poor.

Winter warming and the timing of ice breakup have complex effects on the physical and chemical processes in lakes (e.g. Straile et al. 2003). As a consequence, biomass and species composition of the plant and animal plankton are affected (e.g. Straile and Adrian 2000) which ultimately influence fish populations as well (Straile et al. 2006). In this context, the timing of the spring phytoplankton peak and the clear water phase (Fig. 1.5) are extremely critical for the success of many components of the food web (Adrian et al. 1999, Straile 2002). The timing of the spring peak, the species composition and their biomass can affect zooplankton as well as benthic invertebrate abundance through food quality and food availability (Visconti et al. 2008, Goedkoop and Johson 1996). In general, the coupling of habitats has important consequences for nutrient cycling, predator–prey interactions, food web structure and stability. Nutrient excretion by the benthic macro- and meiofauna, for example, can substantially add to the pelagic nutrient cycles. Moreover, benthic resources subsidize carnivore populations that have important predatory effects on plankton communities. Anthropogenic disturbances, such as eutrophication and climate change, may alter these habitat connections fundamentally (Schindler and Scheuerell 2002).

During the summer stratification period, nutrients are released from the sediments in many lakes which accumulate in the hypolimnion. Summer phytoplankton assemblages, especially under elevated nutrient

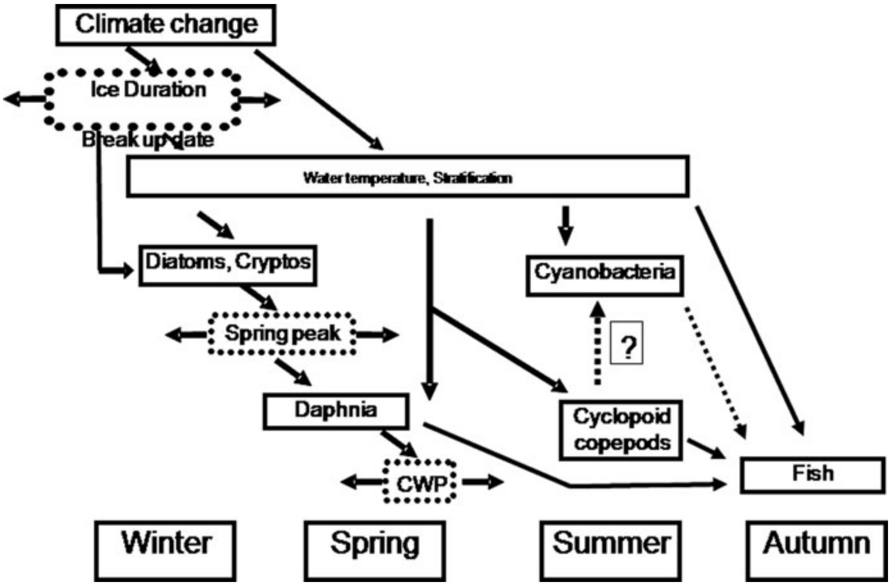

Fig. 1.5 Conceptual diagram of the effects of climate change on deep lake ecosystems. Modified from Dokulil et al. (2009). CWP, Clear water phase

conditions, are frequently dominated by cyanobacteria often filamentous, nitrogen-fixing and/or potentially toxin-producing species. These species are often too large to be grazed by zooplankton and therefore constitute a sort of 'dead end' for the trophic web (Blenckner et al. 2002). In mountainous regions of the Alps climate warming is proceeding at rates well above the global mean (Beniston et al. 1994). Temperature rise affects the thermal regimes of lakes (Livingstone 2003) as well as rivers (Hari et al. 2006) with coherent response to climatic fluctuations in summer (Livingstone and Hari 2008). Both surface (epilimnetic) and deep-water (hypolimnetic) temperature effects in European freshwaters largely depend on the climate which can be described by various climate signals, particularly the North Atlantic Oscillation (Livingstone and Dokulil 2001, Dokulil et al. 2006). In addition, oxygen concentrations in deep lakes respond to interannual variations in winter climate (Rempfer et al. 2008). Long-term data from Lake Constance suggest alteration in heat exchange, vertical mixing, stratification and water level (Wahl 2008). The possible consequences of climate change on large lakes were summarized by Hollan (2000). Because of the particular features of freshwater which are different from the solid and the gas phase (high heat content, density maximum at 4°C, low molecular conduction), climatic impacts to lakes depend on high internal heat transfer by turbulent and convective processes. Water temperature and thermal stratification are therefore an essential controlling factor for a variety of processes in lakes. The renewal and the regeneration of the oxygen content in the hypolimnion critically depend on the extent of convection and circulation, density stratification of inflowing river water and subduction of colder surface water from shallow in-shore areas. All these processes are highly temperature dependent.

Extreme events, such as e.g. unusual hot years, can have deleterious effects on freshwater ecosystems (Jankowski et al. 2006). Droughts, heavy precipitation or floods, mainly influence water systems via catchment processes while storms can have direct and indirect effects. Storms directly affect lakes by cooling, mixing and possibly de-stratifying of the water column (Berger et al. 2006). Indirect effects result from forest damages leading to increased erosion and hence nutrient input to surface and ground waters. Climate events affect not only the input (recharge) and output (discharge), but also the quality of the groundwater (Dragoni and Sukhija 2008).

Cyanobacterial blooms occur more frequently in lakes under increased warming (Le Blanc et al. 2008). These blooms are often linked to human health problems because of the capacity of some cyanobacteria to produce toxins dangerous to humans. Toxins occur either free in the water or bound cyanobacterial cell. It is difficult to remove free toxins from the water by the normal treatment processes. In most cases it is much easier to remove cyanobacterial cells than free toxins. The resistance and persistence of toxins in the environment depend on the nature of the compound. To date there are more than 50 identified cyanobacterial taxa able to produce toxins. The most frequently observed genera in freshwaters during blooms are *Microcystis, Anabaena, Aphanizomenon, Oscillatoria, Nodularia*, and *Nostoc*. Cyanotoxins can be classified into three groups: hepato-, neuro- and dermatotoxins affecting the liver, nervous system or the skin respectively. People may be exposed to toxins through the consumption of contaminated drinking water, direct contact with freshwater or the inhalation of aerosols (Chorus and Bartram 1999, WHO 2002).

1.5 Consequences of Climate Change for Inland Waters – Future Scenarios

In most of the contemporary studies, reports and books future water-related problems are mainly treated quantitatively while quality problems are often underrepresented (e.g. Gleick 1993, 2009, GEO4 2007). Globally however, the most prevalent water quality problem still is eutrophication. As we have discussed in Section 1.2, rivers, lakes and wetlands are worldwide under intense pressure from multiple use, particularly eutrophication and pollution. The services that aquatic ecosystems can provide to society have therefore been reduced and will further decline. Trends affecting the decline of water quality include:

- Population growth
- Urbanization and poverty
- Industrial expansion
- Agricultural pollution
- Water supply development
- Changes in land use

The progressing deterioration of aquatic ecosystems is stopped in Europe through the principal legal instrument of the Water Framework Directive (WFD), which aims at restoring aquatic ecosystems back to good status. This Directive, however, does not consider climate change. Any attempts to restore aquatic ecosystems will be counteracted by climate change adding additional threats, e.g., complex interactions with other stressor types, such as eutrophication. Climate changes will certainly be associated with

- increased amounts of rainfall and snowfall
- changes in water temperature
- alterations of mixed layer depth
- changes in species composition

The EC-WFD as well as other management plans will need constant updating as changes progress. In a gradually warmer climate, the critical load level for obtaining a good ecological status of lakes will in many parts of the world result in stricter demands on the acceptable level of nutrient loading from cities and cultivated fields.

Many effects of climate change will be unpredictable and may have different effects on freshwater systems from different geological backgrounds or ecoregions because these waters vary in their physical and chemical properties. Moreover, climate and certain environmental condition at a specific location can differ from the ‘average’ conditions of the region. The understanding of eutrophication and its management has evolved from simple control of nutrient sources to recognition that it is often a cumulative effect problem that will require protection and restoration of many features of a lake’s community and its catchment (Schindler 2006). The effects of population increase and economical development in a gradually warmer climate are the key environmental threats particularly on small lakes and ponds during the next 25 years. Small water bodies have a higher risk of increased eutrophication for the following reasons:

- Situated in coastal zones (greater risk for salinisation), agricultural areas or cities
- Vulnerable to droughts and alternations in water tables
- Less favoured by the public than deep lakes
- Receive less attention by the politicians than large lakes and reservoirs

Salinisation will further increase in arid areas, leading, in the worst cases, to the disappearance of water bodies. Using available climate scenarios, analysis of the combined effects of climate change and socio-economic driving forces on the future distribution of the world’s freshwater resources clearly show

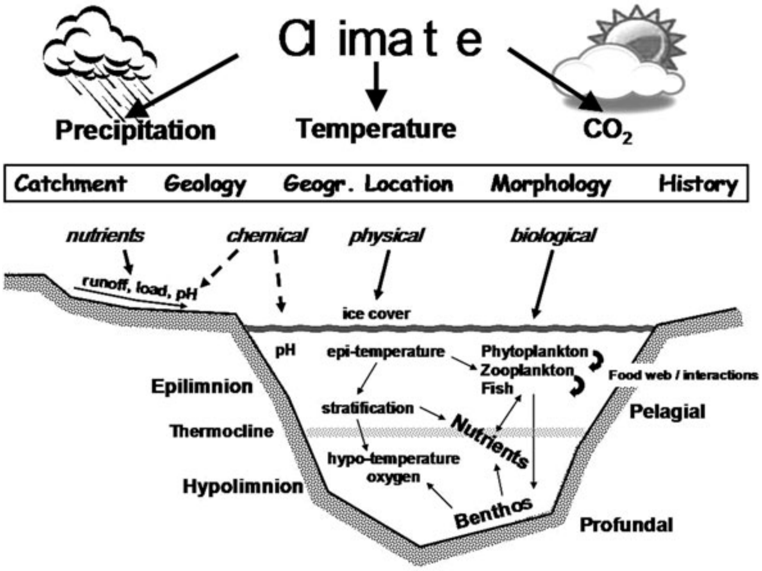

Fig. 1.6 Conceptual diagram visualizing the main components of climate change and their major impacts on freshwaters. The framed terms are the essential ‘effect filters’ sensu Blenckner (2005) which define and modify the responses of the variables given below

increasing water stress due to growing water withdrawals. Some regions might benefit however from climate change because of increased water availability and hence reduced water stress (Arnell 2004, Menzel et al. 2007). The impacts of climate variation and the changes in the associated processes on water ecosystems are summarized conceptually in Fig. 1.6 using the ‘effect filter concept’ of Blenckner (2005).

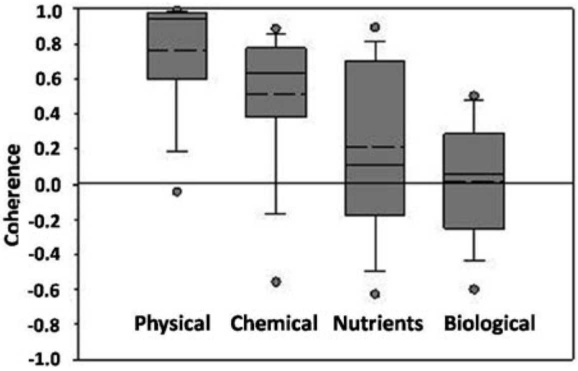

Fig. 1.7 Regional coherence expressed as correlation coefficients for each of the four groups of variables stated in Fig. 1.6 shown as box-whisker plots. Box limits are the 25th and 75th percentile; whiskers indicate the 10th and 90th percentile. In these boxes, the *solid line* is the median, the *dashed line* the mean. From Dokulil et al. (2009)

Main consequences of changes in the climate relevant to freshwater systems are shifts in precipitation, increase in air temperature and elevated greenhouse gas concentrations, particularly CO_2. Alterations in these atmospheric variables are modified by several characteristics for a specific catchment, water body or a group of lakes and streams, here called *effect filter*. These effect filters influence limnological variables or groups of variables differently. Waters with similar effect filters, however, should respond similar to climate variability. The coherent response of lakes within a certain lake district is shown in Fig. 1.7. Similarities are highest among the physical variables while biological reactions are most diverse. In other words, climate impact response cascades down and fades away from physical parameters via chemical and nutrient variables to biological entities. The morphology, geology, land use and anthropogenic pressure affect processes in the catchment via point and particularly non-point sources. Timing and amount of runoff together with erosion primarily determine the nutrient concentration in streams and rivers which in turn supply and load standing water bodies with nutrients and other substances. Depending on the geology of the catchment, CO_2 may affect freshwaters by lowering the pH value leading to acidification of both running and stagnant waters.

The geographic position, the morphometry and the history of a lake determine and further modify its reaction to climate variability. Future projected changes in the timing and duration of the ice season will depend on latitude or altitude. Hydrodynamic patterns are influenced largely by the depth and size of the lake affecting temperature stratification during summer, the annual heat budget, the concentration of oxygen in the hypolimnion which in turn can trigger internal loading with nutrients. The retention time, a factor depending on morphometry and through-flow, determines if internal or external processes dominate. All these various influential determinants affect pelagic and profundal biological components of the food web and their interactions directly (e.g. temperature) and indirectly (e.g. nutrients). In addition, the history of a lake can be an important factor in the response to climate change possibly enhancing eutrophication via nutrient release, for example. Lakes with different history will however react differently to climate impacts depending in which phase they are.

If winters in the northern hemisphere get warmer and wetter, and summers hotter and dryer as anticipated (Easterling et al. 2000, IPCC 2007) shifts in the hydrological runoff regime will occur. Greater discharges are expected in winter and spring as more precipitation will come down as rain rather than snow. Peak flow in streams and rivers are likely to shift earlier. Larger runoff combined with more frequent extreme rainfalls may result in floods, increased erosion and wash out of nutrients which ultimately lead to eutrophication of rivers and lakes. In glaciated regions discharge will first increase due to more melt water and later decrease when glaciers have disappeared. In subtropical and tropical regions, intensified rainy seasons might create storms and floods. Increased summer temperatures in temperate and high-latitude countries will lead to extreme weather situations, reduced stream and river flow and stronger and prolonged thermal stratification. Oxygen will be less available in these warmer lakes as solubility declines with increasing temperature. In addition, rates of decomposition will increase consuming more oxygen which may lead to deoxygenation in deeper parts of lakes. As a result of dwindling inflows and hence increasing water residence time eutrophication in waters receiving point sources of nutrient will be intensified. Together with decreased silica supply (Teubner and Dokulil 2002), this process will replace diatoms by cyanobacteria, some of them toxic, nitrogen fixing or both, leading to impoverished ecological quality, reduced biodiversity and reduced drinking water quality (Dokulil and Teubner 2000). The demand for water, however, will increase enormously, particularly in the developing part of the world. The problem is even worse in subtropical and tropical countries, where the majority of the developing population is concentrated, because these regions strongly depend on surface water as drinking water. Deep tropical lakes are excellent climate monitors because annual mixing is shallow and flushing rates are low, allowing heat to accumulate during climatic warming. A sharpened density gradient as a result of warming has slowed vertical mixing and reduced primary production in Lake Tanganyika, the oldest of the East African Rift Valley Lakes and the second deepest lake in the world. During the coming century, increased warming rates will continue to slow mixing and further reduce productivity in Lake Tanganyika and other deep tropical lakes (Verburg et al. 2003). Recent analyses of water temperature and ice cover indicate that Lake Baikal—the world's largest, oldest and most biotically diverse lake—strongly responds to climate change. The climate of the Baikal region will be warmer and wetter by the end of this century, particularly in winter. The average ice thickness in the lake has decreased, and the ice-free season has increased. Changes in ice cover will alter the food web structure and function. Ice duration affects the lake's dominant primary producers, the endemic diatoms, which bloom, unlike most diatoms, under ice in springtime and are highly dependent on ice cover for their reproduction and growth. These diatoms are the principal food of the crustaceans in this lake, and these in turn are preyed upon by the lake's fish. Shortened periods of ice cover will ultimately also harm the top predator in Baikal, the world's only freshwater seal, because the seals mate and give birth on the ice. As secondary effects of climate change, ecosystem function will be affected by the melting permafrost exacerbating additional anthropogenic stressors such as industrial pollution and cultural eutrophication (Moore et al. 2009).

Increased erosion due to high winter runoff combined with higher water temperatures and more prolonged stratification in summer will, almost certainly, lead to widespread, climate-related eutrophication (Dokulil et al. 2009). Increased soil erosion and runoff will enhance nutrient load particularly from nitrogen to rivers and lakes. While increased P loads will mainly affect lakes, elevated nitrogen concentrations are unlikely to have any effect. In rivers however, increased input of N will lead to higher transport and accumulation downstream ultimately affecting estuaries, costal zones and engulfed seas such as the Baltic Sea.

Greenhouse gases, particularly CO_2, have little or no effect on well-buffered systems (calcareous geological regions). In crystalline and bed-rock areas, elevated CO_2 concentrations will lead to acidification of freshwaters similar to what is anticipated for the oceans (Caldeira and Wickett 2003). The effects of increased concentrations of CO_2 on algal assemblages, however, will depend on many factors including to some extent adaptation processes (Collins and Bell 2004) particularly in cyanobacteria which have evolved an extremely effective CO_2 concentrating mechanism (Badger and Price 2003). If atmospheric CO_2 doubles in the future, air temperatures are projected to rise by 4.3°C in the warm season between April and October. Because of the close coupling of annual average air and water temperature, this increase will translate into an average 4.1°C stream temperature rise, if stream shading remains unaltered (Pilgrim et al. 1998). Research on the effect of climate change on inland waters must increasingly amalgamate findings from field-based investigations with results from carefully controlled experiments as Dunne et al. (2004) suggested. Moreover, predictions of future changes must use and rely on ecosystem models (e.g. Mooij et al. 2007). Future scenarios for inland waters will largely depend on future climate projections which are incorporated into numerical simulation models (e.g. Stefan et al. 1993).

1.6 Concerns, Adaptation and Mitigation

The drivers of eutrophication are expected to increase for the foreseeable future. Specifically:

- World population will continue to grow (estimated 9.2 billion by 2050)
- Cities and suburbs will continue to grow larger
- Intensive agriculture, including the use of fertilizers, and land use conversion—for crops, livestock, and aquaculture—will increase, especially in the developing world. Consumption is expected to increase by 54%.
- Energy consumption is expected to grow 50% from 2005 to 2030. Fossil fuels, which release greenhouse gases into the environment when burned, will continue to be the dominant fuel source in this century.

These increasing global trends in population growth, energy use and agricultural production will increasingly impact freshwater and coastal systems by eutrophication and pollution, especially in the developing world (Selman and Greenhalgh 2009). Eutrophication nowadays must be seen as multitude and cumulative effects of several sorts of abuses of water ecosystems leading to deterioration and impairment of the services they normally provide. All these impacts are magnified by climate change as extensively summarized by Schindler and Vallentyne (2008, Chapters 10, Chapters 11 and Chapters 12). At the same time, we must be aware that nearly every system worldwide responses differently to eutrophication leading to site-specific problems. The manifestations of the ultimate effect of local eutrophication depend largely on physical boundary conditions, and the transformation and retention of nutrients. The real challenge of this millennium is to establish a balance between nature and mankind including its population size. Therefore, eutrophication must be reduced back to levels acceptable to both humans and natural ecosystems because all available data indicate that natural background concentrations of nutrients (pristine conditions) were much lower than today (De Jong et al. 2002, Reynolds 2003). To foster the challenges imposed by climate variability adaptation strategies at all levels are necessary especially in developing countries (UNFCCC 2007, US Aid 2007). Water-related adaptation has many facets. Adaptation has to cope with ecological, political, social, economic and cultural factors (IPCC 2007). Changes in water availability and quality impose practical reaction and planning not only on water managers but also on individual users of the water environment including industry, farmers and individual consumers. Examples of ‘concrete’ actions in the water sector to adapt specifically and solely to a changing climate are however very rare. This is partly because climate change may be only one of many drivers affecting strategies and investment plans, and partly due to uncertainty in projections of future hydrological changes which still exist. It is now generally accepted that P inputs must be decreased to mitigate eutrophication of lakes, reservoirs, streams and rivers. However, reactive N is often thought to be the key controlling factor for eutrophication of estuaries or coastal oceans. Yet, to decrease eutrophication, control of reactive N alone is not sufficient because reduced nitrogen input will have disastrous consequences by favoring nitrogen-fixing cyanobacteria as a response of the phytoplankton community to extreme seasonal nitrogen limitation (Schindler et al. 2008). Control of phosphorus is essential and must be included in management programs designed to decrease eutrophication of freshwaters and coastal zones (Carpenter 2008). Many concepts, methods and techniques exist nowadays to mitigate eutrophication including biological control (Khan and Ansari 2005).

1.7 Epilogue

Climate change will create risks but may also produce opportunities worldwide affecting various sectors such as forestry, agriculture and water management. The water sector, however, is likely the most delicate and vulnerable of all resources. Everyone on earth requires water in sufficient quantity and quality. Any scarcity or shortage will result in water crisis which can create serious conflicts (UNDP 2006, 2007). Individuals and societies can reduce these risks by understanding, planning for and adapting to a changing climate.

The consequences of climate variability and climate change are potentially more significant for the poor in developing countries than for those living in more prosperous nations. In any case, climate-induced eutrophication will affect water resources and degrade water quality with the potential risk of water-borne diseases and health problems associated with contaminated water. Vulnerability to the impacts of climate change is a function of exposure to climate variables, sensitivity to those variables and the adaptive capacity of the affected community (WWAP 2006, 2009).

Fighting climate change and particularly its impact on water resources and availability is a cross-generational exercise which can only be successful if greenhouse gas emissions and particularly CO_2 emissions are drastically reduced in the near future. Only then will sustainable development of freshwater resources be possible to avoid poverty in many regions of the globe.

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Chapter 2

Controlling Eutrophication in the Baltic Sea and the Kattegat

Lars Håkanson and Andreas C. Bryhn

Abstract The basic aim of this work was to motivate a realistic strategy to combat marine eutrophication in north-eastern Europe. Data from the Kattegat (located between Sweden and Denmark) were used to illustrate basic principles and processes related to nutrient fluxes. We have applied a process-based mass-balance model, CoastMab, to the Kattegat and quantified the nutrient fluxes to, within, and from the system. Several scenarios aiming to decrease eutrophication in the Kattegat have been modeled. By far the most dominating nutrient fluxes to the bioproductive surface-water layer in the Kattegat come from the south (from the Baltic Proper), which should be evident just by comparing the catchment area for the Baltic Sea, including the Baltic States, parts of Russia, Belarus and Germany, Poland, Finland, and Sweden in relation to the relatively small catchment area draining directly into the Kattegat (from SW Sweden and parts of Denmark). The dominating deep-water fluxes come from the north (from the Skagerrak). The strategy that one should ask for should concur with some evident practical constraints, e.g., it is not realistic to reduce all anthropogenic P or N discharges. For countries where major investments in nutrient reductions have already been made, it will become increasingly expensive to reduce the remaining tons. In the "optimal" scenario discussed in this work, about 10,000 t year^{-1} of P is being reduced and also N reductions that would lower the N concentration in the Baltic Proper by 10%. The cost for this "optimal" strategy was estimated at 200–420 million euro year^{-1} given that the focus will be on the most cost-effective P reductions connected to the most polluted estuaries and coastal areas. To achieve cost-effectiveness, one can assume that most of this would go to upgrading urban sewage treatment in the Baltic States, Poland, and other former East Bloc countries. The costs to reduce 15,016 t year^{-1} of P and 133,170 t year^{-1} of N according to the HELCOM strategy (agreed upon by the Baltic Sea states in November 2007) would be 3,100 million euro year^{-1}. That is, 2,680–2,900 million euro year^{-1} higher than the "optimal" strategy advocated in this work.

Keywords: Eutrophication · Nitrogen · Phosphorus · Kattegat · Baltic sea

2.1 Background and Aim of the Work

Validated process-based mass-balance models are – categorically – the only tool to quantify fluxes, concentrations, and amounts and to make predictions of how nutrient concentrations would change in response to reductions in nutrient loading where the given reduced flux is put into a context where all other fluxes influencing the given concentrations are quantified in an appropriate and realistic manner. The aim of this work has been to do the following:

- Discuss fundamental aspects related to eutrophication in aquatic systems and using data from the Kattegat to illustrate basic principles and processes. The ultimate aim is to motivate the most realistic strategy to combat eutrophication. We have applied the CoastMab model (a process-based

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mass-balance model using ordinary differential equations giving monthly fluxes) to the Kattegat directly and without any "tuning" to quantify the nutrient fluxes to, within, and from system. This model has been described in detail in many other contexts (Håkanson and Bryhn 2008a, 2008b, Håkanson 2009) and the basic aim here is not to repeat the motivation and testing of the equations but to focus on the principles in more general terms and how to use the model in finding the best possible remedial strategy. We will, however, describe the basic structure of the model (i.e., how the water and sediment compartments are defined).

- Present key driving variables related to salinity, water temperatures, water discharges, and nutrient concentrations and trend analysis for the study period (1995–2008) for the Kattegat system to stress that similar background information should be at hand for all aquatic systems in contexts where remediation of eutrophication is discussed from a mass-balance perspective. Boesch et al. (2008) has given a literature review related to the conditions in the Kattegat.
- When the presuppositions have been defined, several remedial scenarios will also be given, which are meant to demonstrate how the given system would likely respond to changes in tributary P and N loading.
- Finally, based on those results, recommendations will be given for a remedial strategy to reduce the eutrophication in the case study area, the Kattegat.

The transport processes in aquatic systems are general and apply for all substances in most aquatic systems, but there are also substance-specific parts (e.g., related to the particulate fraction, criteria for diffusion and denitrification). Note that the model used to quantify these transport processes in this work, CoastMab, is general so this is not a model where the user should make any tuning and calibrations or change model constants when the model is applied to a new aquatic system. The idea is to have a model based on general and mechanistically correct algorithms describing the monthly transport processes (sedimentation, resuspension, diffusion, mixing, etc.) at the ecosystem scale (i.e., for entire defined basins) and to calculate the role of the different transport processes and how a given system would react to changes in inflow related to natural changes and anthropogenic reductions of water pollutants.

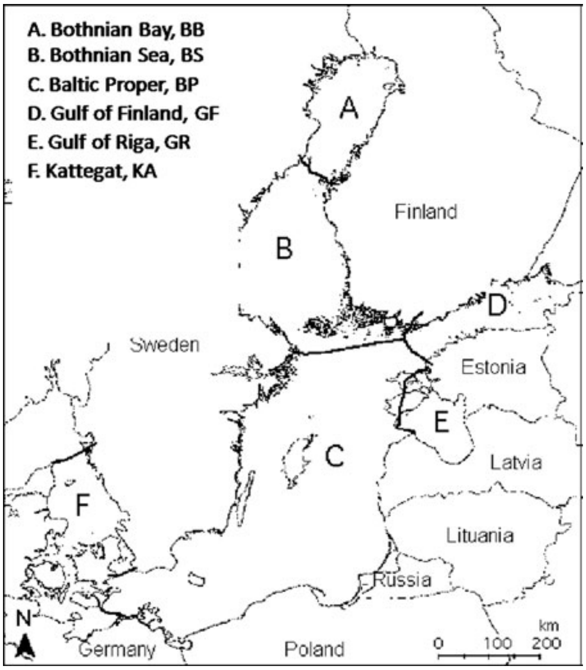


Fig. 2.1 Location map of the Baltic Sea

For persons not familiar with the Baltic Sea system, Fig. 2.1 gives a geographical overview and the names of the main basins. The salinity decreases from over 30 psu in Skagerrak to about 3 psu in the northern part of the Bothnian Bay. It is easy to imagine the enormous water dynamics of the system which is responsible for the inflow of salt water from the south (Kattegat and Skagerrak), the freshwater outflow and the rotation of the earth (the Coriolis force), the variations in winds and air pressures that cause the necessary mixing, and water transport casing this salinity gradient. These salinities demonstrate that the Baltic Sea system including the Kattegat is a very dynamic system. The catchment area of the entire Baltic Sea system is many times larger than the Swedish and Danish areas draining into the Kattegat, and the water from the entire Baltic Sea system will eventually also flow into the Kattegat. The basic structure of the work done and some of the main features of the CoastMab model are illustrated in Fig. 2.2. First (at level 1), the coastal mass-balance model for salt, which is explained in detail in Håkanson and Bryhn (2008a) for the Baltic Sea basins, will be used to quantify the water fluxes to, within, and from all the sub-basins and vertical layers in the Kattegat, including mixing and diffusion.

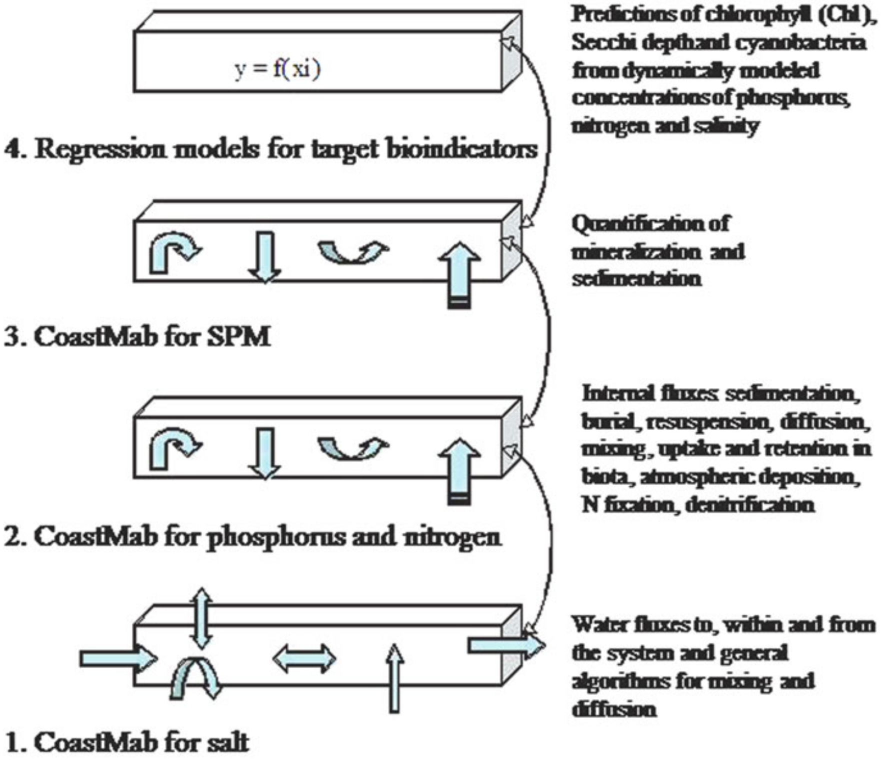


Fig. 2.2 Illustration of the basic structure of the process-based mass-balance model (CoastMab)

The main results will be given in Section 2.3. It should be stressed that the CoastMab modeling has been tested in many coastal areas and lakes and also discussed in Håkanson and Bryhn (2008a, 2008c). This model will calculate the water fluxes needed to explain the measured salinities. This means that data on salinities in the inflowing water to the Kattegat from the Baltic Proper and Skagerrak are needed to run the model and in the following simulations, data from the period 1995–2008 will be used. This modeling also needs morphometric data (mean depth, volume, form factor, dynamic ratio, etc.) and the hypsographic curve and those data are discussed in Section 2.2. The size and form of a given aquatic system, i.e., the morphometry, influences the way in which the system functions, since the depth characteristics influence resuspension and internal loading of nutrients, the nutrient concentrations regulate the primary production, which in turn regulates the secondary production, including zooplankton and fish (see Håkanson and Boulion 2002). At level 2, CoastMab for phosphorus is used (see Håkanson 2009). One should note that many of the algorithms to quantify the transport processes for phosphorus, salt, and nitrogen are also valid for other substances, e.g., inflow, sedimentation of particulate phosphorus and SPM, mixing, diffusion of salt and dissolved phosphorus and nitrogen, resuspension, and burial. There are also substance specific transport processes. For example, for nitrogen, atmospheric deposition, gas transport (nitrogen also appears in a gaseous phase), atmospheric N_2 fixation, and denitrification. Nitrogen modeling is included in this work and data from Eilola and Sahlberg (2006) (see also Håkansson 2007) have been used for the atmospheric N deposition. At level 3, CoastMab for SPM (suspended particulate matter) is used. This means that the inflow, production, sedimentation, burial, and mineralization of suspended particulate matter are quantified on a monthly basis (Håkanson 2006). Sedimentation is important for the oxygen consumption and oxygen status of the system, especially for the oxygen conditions in the deep-water layer below the theoretical wave base and for the diffusion of phosphorus from sediments to water. At level 4, general regression models to predict how the two key bioindicators in eutrophication studies, the Secchi depth (a standard measure of water clarity and the depth of the photic zone) and the concentrations of chlorophyll-a (a key measure of both primary phytoplankton production and biomass and the driving variable for the foodweb model, CoastWeb; see Håkanson and Boulion 2002, Håkanson 2009), would likely change in relation to changing phosphorus and nitrogen concentrations, salinities, SPM values, temperature, and light conditions.

2.2 Basic Information

As a background to this work, Figs. 2.3 and 2.4 show maps related to the areal variations in two of the target bioindicators for eutrophication, the concentration of chlorophyll-a and the Secchi depth.

These two maps provide an overview of the areal distribution patterns of two important variables and from maps such as these one can identify "hotspots," i.e., areas with high algal biomasses expressed by the chlorophyll-a concentrations and areas with turbid water and low Secchi depths, which should be targeted in remedial contexts related to eutrophication. And vice versa, these maps also provide key information related to areas where reductions in anthropogenic nutrient input should not have a high priority. One can note that the conditions in the Kattegat are significantly better than in, e.g., the Gulf of Finland, the Gulf of Riga, and the estuaries of Oder and Vistula. However, this does not imply that nothing should be done to improve the eutrophication in the Kattegat. From Fig. 2.3, one can note typical chlorophyll-a concentrations in the Baltic Sea and parts of the North Sea. Values lower than 2 $\mu g\ L^{-1}$ (oligotrophic conditions; see Table 2.1) are found in the northern parts of the Bothnian Bay and the outer parts of the North Sea, while values higher than 20 $\mu g\ L^{-1}$ (hypertrophic conditions) are more often found in, e.g., the Vistula and Oder lagoons.

The hotspots shown in the map outside the British coast may be a result of data from situations when algal blooms are overrepresented. This map shows that at water depths smaller than 10 m, the Baltic Sea has typical chlorophyll concentrations between 2 and 6 $\mu g\ L^{-1}$ during the growing season (May–September), which correspond to the mesotrophic class. Figure 2.4 shows that several areas with low Secchi depths can be observed, e.g., in the Gulf of Riga and along the North Sea coasts of Holland, Belgium, and Germany. However, some of the observed patchiness may be a result of the interpolation method rather than a true patchiness. In the following, the utilized morphometric data for the Kattegat will first be presented. It will also be explained why and how the given morphometrical parameters are important for the mass-balance calculations. This has been discussed in more detail for lakes by Håkanson (2004). The idea here is to provide a background illustrating how morphometric parameters are used in the CoastMab model.

Compilations of data on salinities, phosphorus, nitrogen, temperature, oxygen concentrations, Secchi depths, and concentrations of chlorophyll-a will also be given. The water fluxes will be presented in the next section. They are used for quantifying the transport of the nutrients. The dynamic mass-balance model for suspended particulate matter (CoastMab for SPM) quantifying sedimentation will also be used. SPM causes scattering of light in the water and influences the Secchi depth and hence the depth of the photic zone; SPM also influences the bacterial decomposition of organic matter, and hence also the oxygen situation and the conditions for zoobenthos, by definition an important food source for benthivorous prey fish. This section will give trend analyses concerning all the studied water variables for the period 1995–2008. An important aspect of this modeling (at the ecosystem scale) concerns the use of hypsographic curves (i.e., depth/area curves for defined basins) to calculate the necessary volumes of water of the defined vertical layers. This information is essential in the mass-balance modeling for salt, phosphorus, nitrogen, and SPM. If there are errors in the defined volumes, there will also be errors in the calculated concentrations since, by definition, the concentration is the mass of the substance in a given volume of water. This section also presents an approach to differentiate between the surface-water and the deep-water layers. Traditionally, this is done by water temperature data, which define the thermocline, or by salinity data, which define the halocline. CoastMab uses an approach which is based on the water depth separating areas where sediment resuspension of fine particles occurs from bottom areas where periods of sedimentation and resuspension of fine newly deposited material are likely to happen (the erosion and transportation areas, the ET areas). The depth separating areas with discontinuous sedimentation (the *T* areas) from areas with more continuous sediment accumulation (the *A* areas) of fine materials is called the theoretical wave base. This is an important

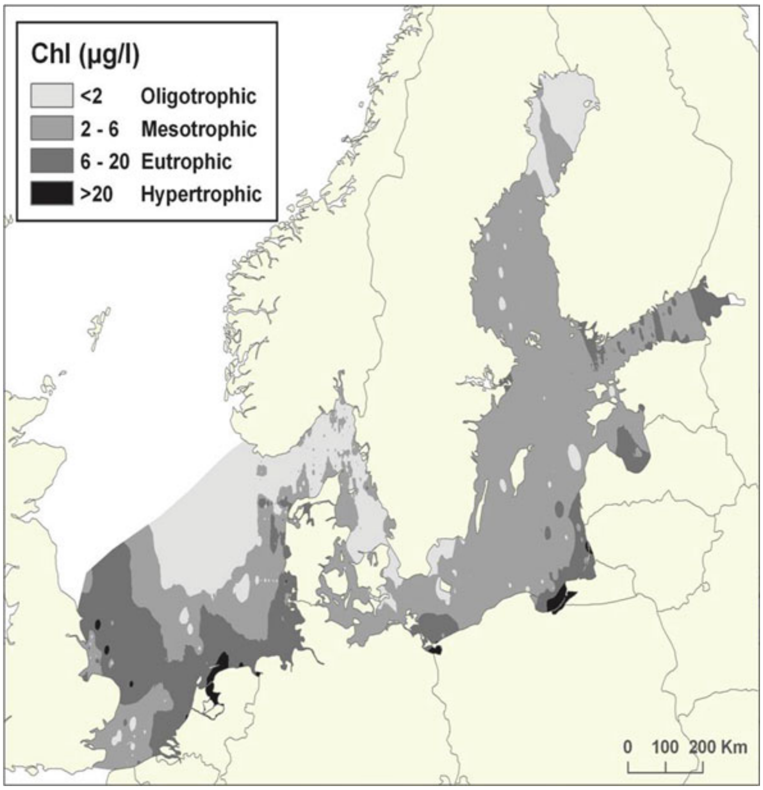


Fig. 2.3 Areal distribution of chlorophyll-a concentrations in the Baltic Sea and parts of the North Sea during the growing season (May–September) in the upper 10 m water column for the period from 1990 to 2005 (from Håkanson and Bryhn 2008a)

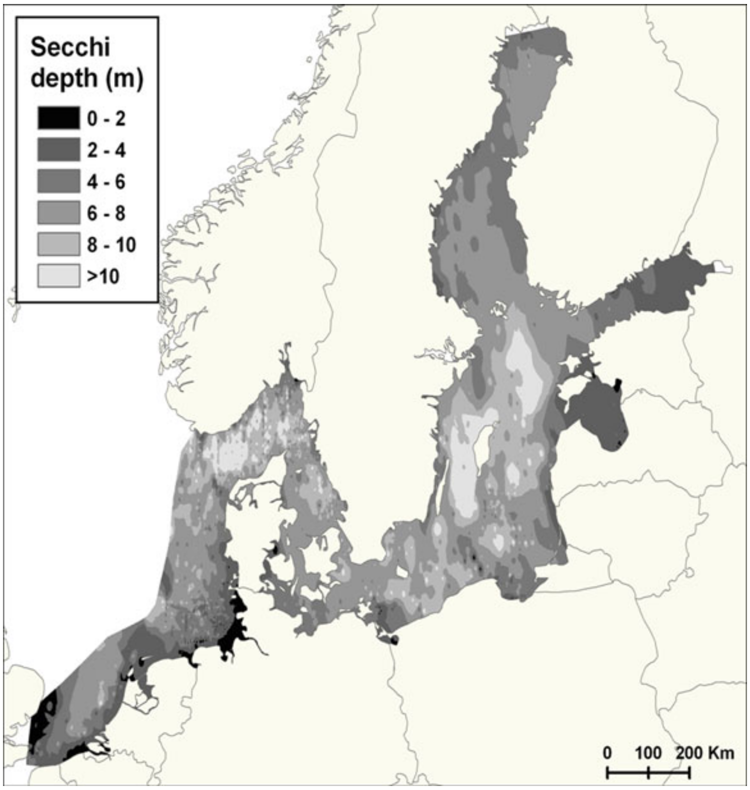


Fig. 2.4 Average annual Secchi depths in the Baltic Sea and parts of the North Sea in the upper 10 m water column for the period from 1990 to 2005 (from Håkanson and Bryhn 2008a)

Table 2.1 Characteristic features in (**A**) freshwater-dominated systems, (**B**) brackish systems, and (**C**) marine coastal systems of different trophic levels (see also OECD 1982, Håkanson and Jansson 1983, Wallin et al. 1992; Håkanson and Boulion 2002, Håkanson et al. 2007). All data represent characteristic (median) values for the growing season for the surface-water layer

Trophic level	Secchi[a] (m)	Chl-a (μg L^{-1})	Total N (μg L^{-1})	Total P (μg L^{-1})	Cyanobacteria[b] (μg ww L^{-1})
A. Freshwater-dominated systems, salinity < 5 psu					
Oligotrophic	>5	<2	<60	<8	<2.2
Mesotrophic	3–5	2–6	60–180	8–25	2.2–250
Eutrophic	1–3	6–20	180–430	25–60	250–1,400
Hypertrophic	<1	>20	>430	>60	>1,400
B. Brackish systems, salinity 5–20 psu					
Oligotrophic	>8	<2	<70	<10	<9.5
Mesotrophic	4.5–8	2–6	70–220	10–30	9.5–380
Eutrophic	1.5–4.5	6–20	220–650	30–90	380–2,500
Hypertrophic	<1.5	>20	>650	>90	>2,500
C. Marine systems, salinity >20 psu					
Oligotrophic	>11	<2	<110	<15	<55
Mesotrophic	6–11	2–6	110–290	15–40	55–680
Eutrophic	2–6	6–20	290–940	40–130	680–4,040
Hypertrophic	<2	>20	>940	>130	>4,040

Relationships between chlorophyll, TP, TN, and salinity calculated from Håkanson (2006)

[a] Secchi depth calculated from Håkanson (2006)

[b] Concentration of cyanobacteria (CB) calculated using the model from Håkanson et al. (2007) when TN/TP is set to 15 (weight ratio), surface-water temperature to 17.5°C, and the salinity to 2.5, 12.5, and 36, respectively, for freshwater, brackish, and marine systems

concept in mass-balance modeling of aquatic systems (see Håkanson 1977, 1999, 2000). The theoretical wave base will also be used to define algorithms

- to calculate concentrations of matter in the given volumes/compartments,
- to quantify sedimentation by accounting for the mean depths of these compartments,
- to quantify internal loading via advection/resuspension as well as diffusion (the vertical water transport related to concentration gradients of dissolved substances in the water),
- to quantify upward and downward mixing between the given compartments, and
- to calculate outflow of substances from the given compartments.

Empirical monthly values of the salinity for the period 1995–2008 have been used to calibrate the CoastMab model for salt and those calculations provide data of great importance for the mass balances for phosphorus, nitrogen, and SPM, namely

- The fluxes of water to and from the defined compartments.
- The monthly mixing of water between layers in the given basin.
- The basic algorithm for diffusion of dissolved substances in water in each compartment.
- The water retention rates influencing the turbulence in each compartment, and hence also
- The sedimentation of particulate phosphorus, nitrogen, and SPM in the given compartments. So, this section will provide and discuss the data necessary to run the CoastMab model.

2.2.1 Morphometric Data and Criteria for the Vertical Layers

Basin-specific data are compiled in Table 2.2 for the case study area, the Kattegat, and will be briefly explained in this section. This table gives data on, e.g., total area, volume, mean depth, maximum depth and the depth of the theoretical wave base (D_{wb} in m), the fraction of bottoms areas dominated by fine sediment erosion and transport (ET areas) above the theoretical wave base, the water transport between the Kattegat and the Baltic Proper (see Håkanson and Bryhn 2008a), sediment characteristics (water content and organic content = loss on ignition; mainly

Table 2.2 Basic data (and abbreviations) for the three basins studied in this work. Bold values are used in the modeling

		Kattegat (KA)	Skagerrak (SK)	Baltic Proper (BP)
Area (A)	(km^2)	**21,818**	**33,400**	211,100
Theoretical wave base (D_{WB})	(m)	**39.9**	40.9	43.8
Area above WB (ET areas)	(km^2)	**18,684**	13,190	87,600
Area below WB (A areas)	(km^2)	**3,134**	3,510	123,500
ET areas (ET)	(%)	**86**	79	47
Max. depth (D_{Max})	(m)	**91** (130)	–	459
Volume (V)	(km^3)	**522.7**	–	13,055
Volume above WB (ET areas)	(km^3)	**487.5**	–	–
Volume below WB (A areas)	(km^3)	**35.2**	–	–
Mean depth (D_{MV})	(m)	**23.96**	–	61.8
Form factor (V_{d})	(–)	**0.79**	–	0.40
Dynamic ratio (DR)	(–)	**6.16**	–	7.43
Water transport from Kattegat to Baltic Sea (Q)	(km^3 year^{-1})	**29**	33.2	250
Water transport to Kattegat from Baltic Sea (Q)	(km^3 year^{-1})	**889**	33.2	250
A-sediment water content (W, 0–10 cm)	(% ww)	**70**	–	75
Fresh sediment water content (W)	(% ww)	**85**	–	–
A-sediment organic content (IG, 0–10 cm)	(% dw)	**10**	–	–
Fresh sediment organic content (IG)	(% dw)	**15**	–	–
Old sediment organic content (IG)	(% dw)	**7.5**	–	–
Latitude	(°N)	**57**	–	–

based on data supplied by Prof. Ingemar Cato, SGU, Uppsala), and latitude.

There are more than 15,000 measurements on water temperature, salinity, TN and TP concentrations, and chlorophyll and about 14,000 data on Secchi depths and oxygen concentrations for the period from 1995 to 2008 used in this work from the entire Kattegat. The theoretical wave base is defined from the ETA diagram (see Fig. 2.5; erosion–transport–accumulation; from Håkanson 1977), which gives the relationship between the effective fetch, as an indicator of the free water surface over which the winds can influence the wave characteristics (speed, height, length, and orbital velocity).

The theoretical wave base separates the transportation areas (T), with discontinuous sedimentation of fine materials, from the accumulation areas (A), with continuous sedimentation of fine suspended particles. The theoretical wave base (D_{wb} in m) is, e.g., at a water depth of 39.9 m in the Kattegat. This is calculated from Eq. (2.1) (Area = area in km^2):

$$D_{\mathrm{wb}} = (45.7 \cdot \sqrt{\mathrm{Area}})/(\sqrt{\mathrm{Area}} + 21.4) \qquad (2.1)$$

It should be stressed that this approach to separate the surface-water layer from the deep-water layer has been used and motivated in many previous contexts for lakes (Håkanson et al. 2004), smaller coastal areas in the Baltic Sea (Håkanson and Eklund 2007), and the sub-basins in the Baltic Sea (Håkanson and Bryhn 2008a, 2008c). This approach gives one value for the theoretical wave base related to the area of the system. The validity of this approach for the Kattegat is demonstrated in Fig. 2.6a for the salinity, Fig. 2.6b for the oxygen concentration, and Fig. 2.7 for the TN/TP ratio (TN = total nitrogen; TP = total phosphorus).

From Fig. 2.6a, it may be noted that for the Kattegat the surface-water (SW) salinity is clearly different

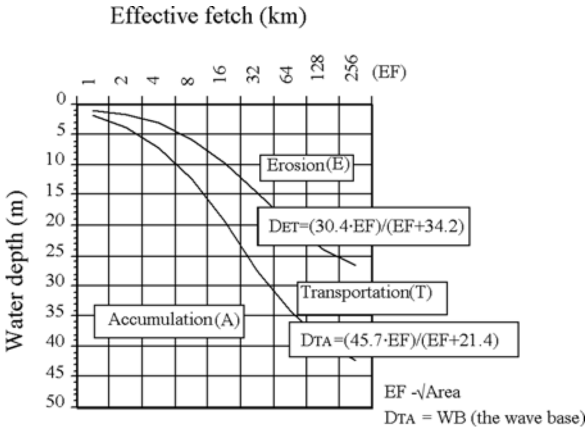


Fig. 2.5 The ETA diagram (erosion–transportation–accumulation; redrawn from Håkanson 1977) illustrating the relationship between effective fetch, water depth, and potential bottom dynamic conditions. The theoretical wave base (D_{wb}; 39.9 m in the Kattegat) may be used as a general criterion in mass-balance modeling to differentiate between the surface-water layer with wind/wave-induced resuspension and deeper areas without wind-induced resuspension of fine materials. The depth separating *E* areas with predominately coarse sediments from *T* areas with mixed sediments is at 25 m in the Kattegat

from the deep-water (DW) salinity. The mean SW salinity is 24.6 psu (see Table 2.3, which also gives monthly mean values and coefficients of variation, CV), whereas the mean DW salinity is 33.3 (the CV value is very low, 0.02; CV = coefficient of variation, CV = SD/MV; SD = standard deviation, MV = mean value). Tables 2.3 and 2.4 give mean monthly values and coefficients of variations not just for salinity but also for water temperatures, oxygen concentrations, phosphate, TP, nitrite, nitrate, ammonium, and TN, and Table 2.3 gives the corresponding data for PON (particulate organic nitrogen), POC, chlorophyll, and Secchi depth.

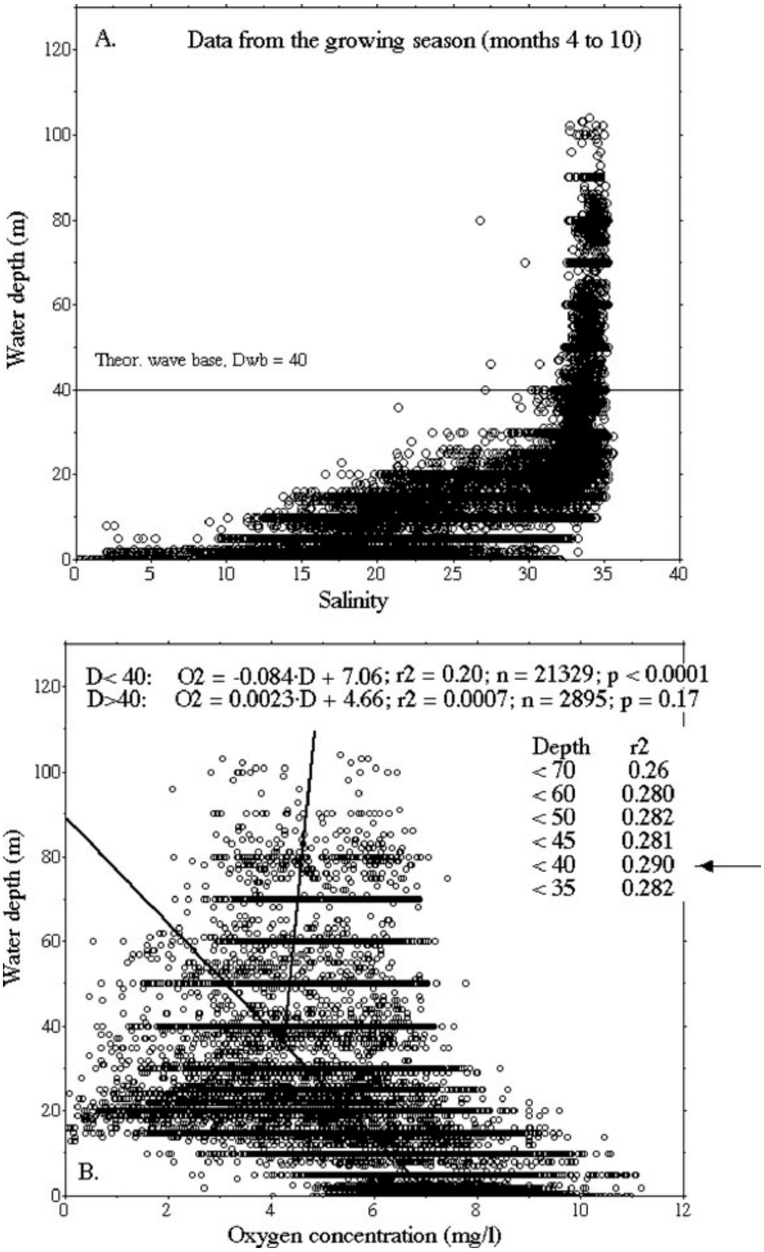
The aim of the modeling is to describe these empirical salinities as close as possible and to predict the given TP, TN, chlorophyll concentrations and Secchi depths so that the predicted values agree with the empirical data. Note that the basic aim is to predict the mean annual values rather than the monthly data because (1) annual and not monthly nutrient fluxes from the Baltic Proper are used in this modeling and (2) annual and not monthly nutrient fluxes from land (from HELCOM 2000) are used. So, in this modeling, the case study system (KA) has been divided into two depth intervals: (1) the surface-water layer (SW), i.e., the water above the theoretical wave base; (2) the deep-water layer (DW) defined as the volume of water beneath the theoretical wave base. It should be stressed that the theoretical wave base at around 40 m in the Kattegat describes average conditions. During storm events, the wave base will be at greater water depths (see Jönsson 2005) and during calm periods at shallower depths. The wave base also varies spatially within the studied area. From Figs. 2.6 and 2.7, it is evident that the depth of the wave base describes the conditions in the Kattegat very well. Figure 2.8 gives the hypsographic curve for the Kattegat and how the areas above and below the theoretical wave base are defined.

One can note that the area below the theoretical wave base (D_{wb}) at 39.9 m in KA is 3,134 km^2 and the total area is 21,818 km^2. The volume of the SW layer is 487.5 km^3 and of the DW layer only 35.3 km^3; the entire volume is 522.7 km^3. The maximum depth is 130 m, but from Fig. 2.8, one can see that the area below 91 m is very small so 91 m has been used as a functional maximum depth in this modeling. Among the morphometric parameters characterizing the studied sub-basin, three main groups can be identified (see Håkanson 2004):

- Size parameters: different parameters in length units, such as the maximum depth, parameters expressed in area units, such as water surface area, and parameters expressed in volume units, such as water volume and SW volume.
- Form parameters (based on size parameters) such as mean depth and the form factor.
- Special parameters, for example, the dynamic ratio and the effective fetch.

The CoastMab model uses several of these variables. They are listed in Table 2.2. The volume development, also often called the form factor (V_{d}, dimensionless), is defined as the ratio between the water volume and the volume of a cone, with a base equal to the water surface area (A in km^2) and with a height equal to the maximum depth (D_{Max} in m):

$$\begin{aligned} V_{\mathrm{d}} &= (A \times D_{\mathrm{MV}} \times 0.001)/(A \times D_{\mathrm{Max}} \times 0.001 \times 1/3) \\ &= 3 \times D_{\mathrm{MV}}/D_{\mathrm{Max}} \qquad (2.2) \end{aligned}$$



Depth	r2
< 70	0.26
< 60	0.280
< 50	0.282
< 45	0.281
< 40	0.290
< 35	0.282

Fig. 2.6 The relationship between (**a**) water depth and salinity in the Kattegat and (**b**) between water depth and oxygen concentration. The two figures also show the theoretical wave base at about 40 m in the Kattegat. Data from SMHI. The statistical analyses given in Fig. 2.6b demonstrate that the theoretical wave base at 40 m is also the threshold depth for the oxygen concentrations

The form factor describes the form of the basin. The form of the basin is very important, e.g., for internal sedimentological processes. In basins of similar size but with different form factors, one can presuppose that the system with the smallest form factor would have a larger area above the theoretical wave base and more of the resuspended matter transported to the surface-water compartment than to the deep-water compartment below the theoretical wave base compared to a system with a higher form factor. This is also the way in which the form factor is used in the CoastMab model.

The dynamic ratio (DR; see Håkanson 1982) is defined by the ratio between the square root of the water surface area (in km^2 not in m^2) and the mean depth, D_{MV} (in m; DR = Area/D_{MV}). DR is a standard morphometric parameter in contexts of resuspension and turbulence in entire basins. ET areas above

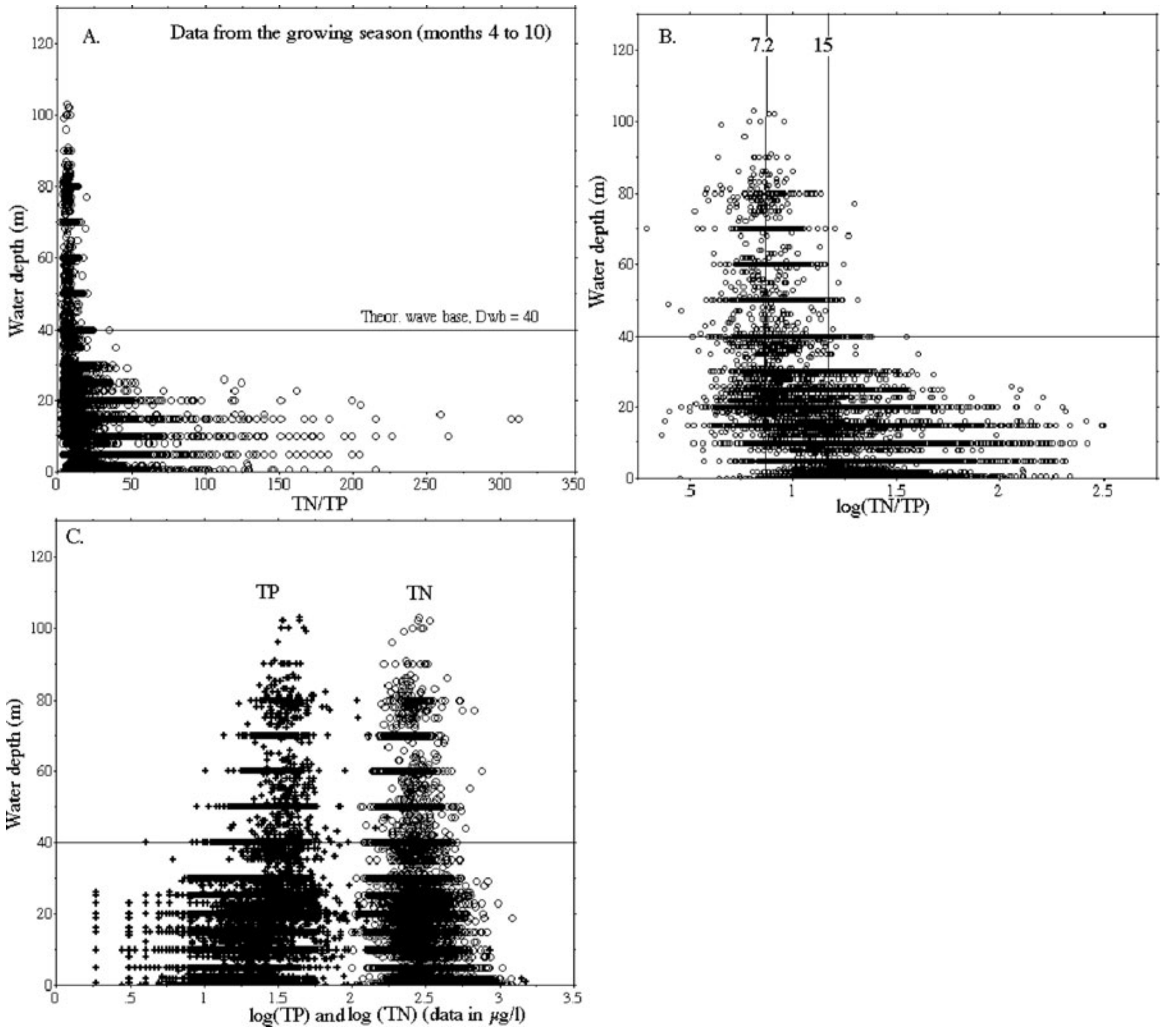


Fig. 2.7 The relationship between (**a**) water depth and the TN/TP ratio in the Kattegat, (**b**) between water depth and log values for the TN/TP ratio, and (**c**) between water depth and log(TP) and log(TN), respectively. The figures also show the theoretical wave base at about 40 m in the Kattegat. Data from SMHI

the theoretical wave base (i.e., areas where fine sediment erosion and transport processes prevail) are likely to dominate the bottom dynamic conditions in basins with dynamic ratios higher than 3.8. Slope processes are known (see Håkanson and Jansson 1983) to dominate the bottom dynamic conditions on slopes greater than about 4–5%. Slope-induced ET areas are likely to dominate basins with DR values lower than 0.052.

One should also expect that in all basins there is a shallow shoreline zone where wind-induced waves will create ET areas, and it is likely that most basins have at least 15% ET areas. If a basin has a DR of 0.26, one can expect that in this basin the ET areas would occupy 15% of the area. If DR is higher or lower than 0.26, the percentage of ET areas is likely to increase. Basins with high DR values, i.e., large and shallow system, are also likely to be more turbulent than small and deep basins. This will influence sedimentation. During windy periods with intensive water turbulence, sedimentation of suspended fine particles in the water will be much lower than under calm conditions. This is accounted for in the CoastMab model and the dynamic ratio is used as a proxy for the potential turbulence in the monthly calculations of the transport processes. It should be stressed that the form factor and

Table 2.3 Mean monthly values (MV) and coefficients of variation (CV) for variables in the surface-water layer of Kattegat for the period 1995–2008 for surface-water temperature (SWT), salinity (Sal), oxygen concentration (O_2), phosphate (PO_4), total phosphorus (TP), nitrite (NO_2), nitrate (NO_3), ammonium (NH_4), total nitrogen (TN), particulate organic nitrogen (PON), particulate organic carbon (POC), and chlorophyll-a concentrations (Chl)

Month	Secchi (m)	SWT (°C)	Sal (psu)	O_2 (mL L^{-1})	PO_4 (µmol L^{-1})	TP (µmol L^{-1})	NO_2 (µmol L^{-1})	NO_3 (µmol L^{-1})	NH_4 (µmol L^{-1})	TN (µmol L^{-1})	PON (µmol L^{-1})	POC (µmol L^{-1})	Chl (µg L^{-1})
1	5.7	4.4	26.5	7.4	0.63	0.88	0.34	7.60	0.71	21.7	2.18	18.8	1.55
2	5.1	3.4	26.2	7.7	0.60	0.89	0.30	8.96	0.69	23.5	2.35	18.9	2.09
3	5.0	3.3	24.0	8.1	0.39	0.77	0.19	7.52	0.78	24.1	3.94	29.5	5.01
4	6.4	5.0	23.5	7.7	0.28	0.61	0.14	5.58	0.97	22.1	3.69	25.9	2.27
5	5.9	7.9	23.5	7.0	0.27	0.58	0.15	4.87	1.03	21.0	3.36	24.3	1.98
6	6.2	11.6	23.6	6.5	0.21	0.60	0.14	3.33	0.92	20.0	3.40	23.8	2.09
7	6.3	14.8	23.4	5.8	0.18	0.50	0.15	2.45	0.88	19.2	3.40	23.8	1.93
8	6.8	16.6	24.1	5.4	0.21	0.53	0.16	1.87	0.82	18.5	3.17	22.8	2.00
9	7.5	15.5	25.9	5.1	0.26	0.60	0.22	1.87	0.78	18.6	3.70	22.3	2.08
10	6.1	12.9	24.1	5.7	0.31	0.69	0.23	2.62	1.03	19.9	3.14	23.1	2.59
11	5.7	9.5	25.4	6.3	0.39	0.75	0.31	3.87	1.30	21.2	3.21	22.8	2.45
12	6.1	7.0	25.1	6.8	0.51	0.79	0.32	5.52	1.02	21.2	2.26	17.0	1.83
MV	6.08	9.33	24.60	6.61	0.35	0.68	0.22	4.67	0.91	20.9	3.15	22.7	2.32
1	0.46	0.50	0.22	0.15	0.29	0.23	0.95	0.64	1.87	0.38	0.55	0.62	1.96
2	0.48	0.56	0.22	0.13	0.35	0.28	0.76	0.73	1.69	0.40	0.65	0.69	1.49
3	0.46	0.49	0.29	0.16	0.77	0.40	0.77	1.05	2.14	0.40	0.67	0.73	0.92
4	0.41	0.24	0.35	0.16	1.01	0.46	0.72	1.30	1.03	0.42	0.53	0.57	0.99
5	0.40	0.28	0.35	0.15	1.08	0.53	0.83	1.41	1.11	0.41	0.60	0.69	0.93
6	0.36	0.29	0.33	0.13	1.10	3.19	0.94	1.67	1.45	0.42	0.45	0.46	0.82
7	0.37	0.24	0.30	0.15	1.16	0.52	1.10	2.05	1.59	0.42	0.46	0.48	0.83
8	0.38	0.21	0.30	0.20	1.21	0.62	1.26	2.32	1.78	0.37	0.64	0.74	0.97
9	0.38	0.16	0.26	0.27	1.06	0.48	1.43	2.01	1.65	0.37	0.68	0.71	0.96
10	0.41	0.13	0.29	0.25	0.93	0.51	1.23	1.73	1.93	0.40	0.53	0.58	0.86
11	0.45	0.19	0.23	0.18	0.72	0.52	0.88	1.51	2.18	0.48	0.68	0.76	1.29
12	0.48	0.31	0.25	0.18	0.45	0.36	1.06	1.01	1.17	0.39	0.59	0.59	0.97
MV	0.42	0.30	0.28	0.17	0.84	0.68	0.99	1.45	1.63	0.40	0.59	0.63	1.08

Table 2.4 Mean monthly values (MV) and coefficients of variation (CV) for variables in the deep-water layer of Kattegat for the period 1995–2008 for deep-water temperature (DWT), salinity (Sal), oxygen concentration (O_2), phosphate (PO_4), total phosphorus (TP), nitrite (NO_2), nitrate (NO_3), ammonium (NH_4), and total nitrogen (TN)

Month	DWT (°C)	Sal (psu)	O_2 (mL L^{-1})	PO_4 (µmol L^{-1})	TP (µmol L^{-1})	NO_2 (µmol L^{-1})	NO_3 (µmol L^{-1})	NH_4 (µmol L^{-1})	TN (µmol L^{-1})
1	4.44	26.51	7.36	0.63	0.88	0.34	7.60	0.71	21.69
2	5.86	33.64	6.41	0.76	0.91	0.27	9.26	0.25	18.53
3	5.33	33.87	6.43	0.75	0.87	0.24	10.75	0.70	20.77
4	5.50	34.19	6.13	0.73	0.85	0.23	9.09	0.99	19.20
5	5.88	34.37	5.88	0.75	0.90	0.25	8.41	1.32	18.12
6	6.51	34.11	5.46	0.72	0.86	0.26	7.67	1.03	17.69
7	7.66	33.81	4.56	0.79	0.92	0.22	8.47	0.69	17.99
8	9.62	33.76	3.83	0.84	0.97	0.34	7.42	0.88	17.38
9	11.14	33.86	3.62	0.83	0.98	0.42	6.70	0.71	16.45
10	11.66	34.09	4.26	0.76	0.92	0.26	6.40	0.28	15.27
11	11.22	33.69	4.85	0.74	0.91	0.40	5.49	0.42	15.00
12	9.52	33.80	5.47	0.72	0.85	0.43	6.30	0.24	15.30
MV	7.86	33.31	5.35	0.75	0.90	0.30	7.80	0.69	17.78
1	0.17	0.03	0.08	0.14	0.17	1.25	0.20	1.53	0.16
2	0.17	0.03	0.08	0.14	0.14	1.19	0.18	1.41	0.18
3	0.15	0.02	0.07	0.17	0.19	0.59	0.31	1.16	0.23
4	0.14	0.02	0.07	0.23	0.28	0.38	0.30	0.91	0.23
5	0.11	0.01	0.07	0.20	0.22	0.43	0.32	0.88	0.22
6	0.14	0.01	0.08	0.26	0.24	0.66	0.42	0.85	0.19
7	0.21	0.01	0.13	0.24	0.24	0.50	0.34	1.05	0.16
8	0.27	0.02	0.20	0.30	0.28	0.67	0.45	0.95	0.21
9	0.21	0.02	0.24	0.30	0.27	0.74	0.41	1.07	0.17
10	0.19	0.02	0.18	0.27	0.25	0.77	0.39	1.36	0.19
11	0.12	0.02	0.13	0.21	0.22	0.74	0.43	1.24	0.17
12	0.10	0.02	0.08	0.20	0.19	1.78	0.28	1.90	0.19
MV	0.16	0.02	0.12	0.22	0.22	0.81	0.34	1.19	0.19

the dynamic ratio provide different and complementary aspects of how the form may influence the function of aquatic systems. The effective fetch (see the ETA diagram in Fig. 2.5) is often defined according to a method introduced by the Beach Erosion Board (1972). The effective fetch (L_{ef} in km) gives a more representative measure of how winds govern waves (wave length, wave height, etc.) than the effective length, since several wind directions are taken into account. Using traditional methods, it is relatively easy to estimate the effective fetch by means of a map and a special transparent paper (see Håkanson 1977). The central radial of this transparent paper is put in the main wind direction or, if the maximum effective fetch is requested, in the direction which gives the highest L_{ef} value. Then the distance (x in km) from the given station to land (or to islands) is measured for every deviation angle a_i, where a_i is ± 6, 12, 18, 24, 30, 36, and 42°. L_{ef} may then be calculated from

$$L_{\mathrm{ef}} = \sum x_i \cdot \cos(a_i) / \left(\sum \cos(a_i)\right) \cdot \mathrm{SC}' \qquad (2.3)$$

$\sum \cos(a_i) = 13.5$, a calculation constant.
SC' = the scale constant; if the calculations are done on a map in scale 1:250,000, then $\mathrm{SC}' = 2.5$.

The effective fetch attains the highest values close to the shoreline and the minimum values in the central part of a basin. This relationship is important in, e.g., contexts of shore erosion and morphology, for bottom dynamic conditions (erosion–transportation–accumulation), and hence also for internal processes, mass-balance calculations, sediment sampling, and evaluations of sediment pollution. For entire basins, the mean effective fetch may be estimated as Area (see Fig. 2.5). In a round basin, the requested value should be somewhat lower than the diameter ($d = 2r$; r = the radius); the area is πr^2 and hence $d = 1.13\cdot\mathrm{Area}$ and the mean fetch approximately Area.

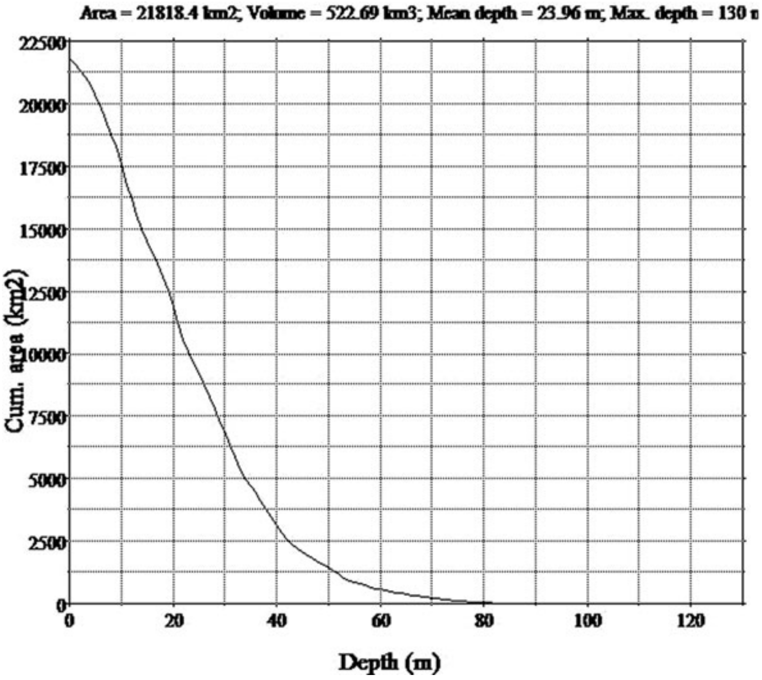


Fig. 2.8 Hypsographic curve for the Kattegat. Based on data from SMHI

2.2.2 *Sediments and Bottom Dynamic Conditions*

As stressed in Fig. 2.5, the theoretical wave base may also be determined from the ETA diagram. This approach focuses on the behavior of the cohesive fine materials settling according to Stokes' law in laboratory vessels:

- Areas of erosion (*E*) prevail in shallow areas or on slopes where there is no apparent deposition of fine materials but rather a removal of such materials; *E* areas are generally hard and consist of sand, consolidated clays, and/or rocks with low concentrations of nutrients.
- Areas of transportation (*T*) prevail where fine materials (such as the carrier particles for water pollutants) are deposited periodically (areas of mixed sediments). This bottom type generally dominates where wind/wave action regulates the bottom dynamic conditions. It is sometimes difficult in practice to separate areas of erosion from areas of transportation. The water depth separating transportation areas from accumulation areas, the theoretical wave base, is, as stressed, a fundamental component in these mass-balance calculations.
- Areas of accumulation (*A*) prevail where the fine materials (and particulate forms of water pollutants) are deposited continuously (soft bottom areas).

Generally hard or sandy sediments within the areas of erosion (*E*) often have a low water content, low organic content, and low concentrations of nutrients and pollutants. These are the areas (the "end stations") where high concentrations of pollutants may appear (see Table 2.5). The conditions within the *T* areas are, for natural reasons, variable, especially for the most mobile substances, like phosphorus, manganese, and iron, which react rapidly to alterations in the chemical "micro-climate" (given by the redox potential) of the sediments. Fine materials may be deposited for long periods during stagnant weather conditions.

In connection with a storm or a mass movement on a slope, this material may be resuspended and transported up and away, generally in the direction toward the *A* areas in the deeper parts, where continuous deposition occurs. Thus, resuspension is a most natural phenomenon on *T* areas. It should also be stressed that fine materials are rarely deposited as a result of simple vertical settling in natural aquatic environments. The horizontal velocity is generally at least 10 times larger, sometimes up to 10,000 times larger, than the

Table 2.5 Mean values (MV) and coefficient of variation (CV) for TP, TN and loss on ignition in surficial (0–2 cm) accumulation area sediments (*A*) and erosion and transport sediments (ET) in Kattegat (data from Prof. Ingemar Cato, SGU)

0–2 cm	TP (mg g^{-1} dw)	TN (mg g^{-1} dw)	IG (% ww)	Water depth (m)
ET/SW				
n	67	13	67	67
MV	0.99	1.69	9.87	22.6
CV	0.29	0.46	0.72	0.47
A/DW				
n	40	17	40	27
MV	0.88	2.43	11.02	52.6
CV	0.16	0.34	0.30	0.16

vertical component for fine materials or flocs that settle according to Stokes' law (Bloesch and Burns 1980, Bloesch and Uehlinger 1986). An evident boundary condition for this approach to calculate the ET areas is that if the depth of the theoretical wave base $D_{\mathrm{wb}} > D_{\mathrm{Max}}$, then $D_{\mathrm{wb}} = D_{\mathrm{Max}}$.

In CoastMab, there are also two boundary conditions for ET (= the fraction of ET areas in the basin):

If ET > 0.99 then ET = 0.99 and if ET < 0.15 then ET = 0.15.

ET areas are generally larger than 15% (ET = 0.15) of the total area since there is always a shore zone dominated by wind/wave activities. For practical and functional reasons, one can also generally find sheltered areas, macrophyte beds, and deep holes with more or less continuous sedimentation, that is, areas which actually function as *A* areas, so the upper boundary limit for ET may be set at ET = 0.99 rather than at ET = 1. The value for the ET areas is used as a distribution coefficient in the CoastMab model. It regulates whether sedimentation of the particulate fraction of the substance (here phosphorus, nitrogen, or SPM) goes to the DW or ET areas. The sediment data are compiled in Table 2.6.

One can note the following:

Most TP values from the upper 2 cm of the accumulation area sediments below the theoretical wave base vary in the range from 0.7 to 1.1 mg TP g^{-1} dw (the mean value is close to 0.88 mg g^{-1} dw; dw = dry weight); the TN data from 2.1 to 2.8 mg g^{-1} dw (MV = 2.4 mg g^{-1} dw); the organic content is about 10–11% ww (ww = wet weight).

1. Due to substrate decomposition by bacteria and compaction from overlying sediments, the TP, TN concentrations and the organic content (loss on ignition, IG) decrease with sediment depth in the accumulation areas (see Håkanson and Jansson 1983). In all of the following simulations, a sediment depth of 0–10 cm will be used and this means that the reference values for the water content, organic content, TP and TN concentrations will be adjusted to this. The reference values for the 0–10 cm layer are set to be 33% lower than the P and N values given in Table 2.4 for the 0–2 cm layer.
2. The bulk density (*d* in g cm^{-3} ww) is between 1.1 and 1.3.
3. The water content (*W* in % ww) has been set to 70% for the upper 10 cm accumulation area sediments in the Kattegat (0–10 cm) and to 85% for the newly deposited SPM on the ET areas.
4. The organic content (= loss on ignition, IG in % dw) is set to 10% for the upper 10 cm accumulation area sediments in the Kattegat. The IG value in underlying clayey sediments is around 7.5% dw.

The area of erosion (Area$_{\mathrm{E}}$) is calculated from the hypsographic curve and the corresponding depth given by the ETA diagram (Fig. 2.5). This means that the depth separating *E* areas from *T* areas is given by

$$D_{\mathrm{ET}} = (30.4 \cdot \sqrt{\mathrm{Area}})/(\sqrt{\mathrm{Area}} + 34.2) \qquad (2.4)$$

Note that the area is given in km^2 in Eq. (2.3) to get the depth in m.

2.2.3 Trends and Variations in Water Variables

This section will present and discuss empirical data in the Kattegat for the period 1995–2008 (data from SMHI) as a background to the subsequent modeling.

Table 2.6 The relationship between bottom dynamic conditions (erosion, transportation, and accumulation) and the physical, chemical, and biological character of the surficial sediments. The given data represent characteristic values from marine coastal areas based on data from 11 Baltic Sea coastal areas (from Håkanson et al. 1984). ww = wet weight; dw = dry weight

	Erosion	Transportation	Accumulation
Physical Parameters			
Water content (% ww)	<50	50–75	>75
Organic content (% dw)	<4	4–10	>10
Nutrients (mg g^{-1} dw)			
Nitrogen	<2	10–30	>5
Phosphorus	0.3–1	0.3–1.5	>1
Carbon	<20	20–50	>50
Metals			
Iron (mg g^{-1} dw)	<10	10–30	>20
Manganese (mg g^{-1} dw)	<0.2	0.2–0.7	0.1–0.7
Zinc (μg g^{-1} dw)	<50	50–200	>200
Chromium (μg g^{-1} dw)	<25	25–50	>50
Lead (μg g^{-1} dw)	<20	20–30	>30
Copper (μg g^{-1} dw)	<15	15–30	>30
Cadmium (μg g^{-1} dw)	<0.5	0.5–11.5	>1.5
Mercury (ng g^{-1} dw)	<50	50–250	>250

Figure 2.9 first gives data on the target bioindicators, Secchi depth, oxygen concentrations, and concentrations of chlorophyll-a in the surface-water layer in Kattegat.

This figure and the following figures also give statistical trend analyses (regression line, coefficient of determination, r^2, and number of data, n). From Fig. 2.9, one can note the following:

- There is a very weak trend for these three bioindicators, as revealed by the small slope coefficients (–0.00776 for Secchi depth, –0.0021 for oxygen, and –0.0028 for chlorophyll) and the low r^2 values (0.21, 0.0052, and 0.0027). So, for this period, the conditions have been rather stable in the Kattegat for these three key variables.
- One can also note the clear seasonal pattern for oxygen, no evident seasonal pattern for Secchi depth, and a fairly distinct pattern for chlorophyll. One might have expected a more evident seasonal pattern for chlorophyll with peak values in the spring and fall.

The corresponding information is given in Fig. 2.10 for surface-water temperatures, salinity, TP and TN concentrations, and the TN/TP ratio.

The TN/TP ratio addresses the question about "limiting" nutrient, which is certainly central in aquatic ecology and has been treated in numerous papers and textbooks (e.g., Dillon and Rigler 1974, Smith 1979, 2003, Riley and Prepas 1985, Howarth 1988, Evans et al. 1996, Wetzel 2001, Newton et al. 2003, Smith et al. 2006, Håkanson and Bryhn 2008a, 2008c). The average composition of algae ($C_{106}N_{16}P$) is reflected in the Redfield ratio (N/P = 7.2 by mass). So, by definition, algae need both nitrogen and phosphorus and one focus of coastal eutrophication studies concerns the factors limiting the phytoplankton biomass, often expressed by chlorophyll-a concentrations in the water. Note that the actual phytoplankton biomass at any given moment in a system is a function of the bioavailable nutrient concentrations, light, and predation on phytoplankton by herbivorous zooplankton minus the death of phytoplankton regulated by the turnover time of the phytoplankton (see Håkanson and Boulion 2002). From Fig. 2.10, one can note the following:

- All trends are weak. The strongest is the decrease in TN concentrations; the increase in temperature is also interesting in these days when global warming is on the agenda; the changes in salinity, TP, and TN/TP are very small. It should be stressed that all these changes are statistically significant because the number of data is so large. These data support the conclusion that there have been no major changes in the Kattegat system during the last 18 years regarding the variables in Fig. 2.10.

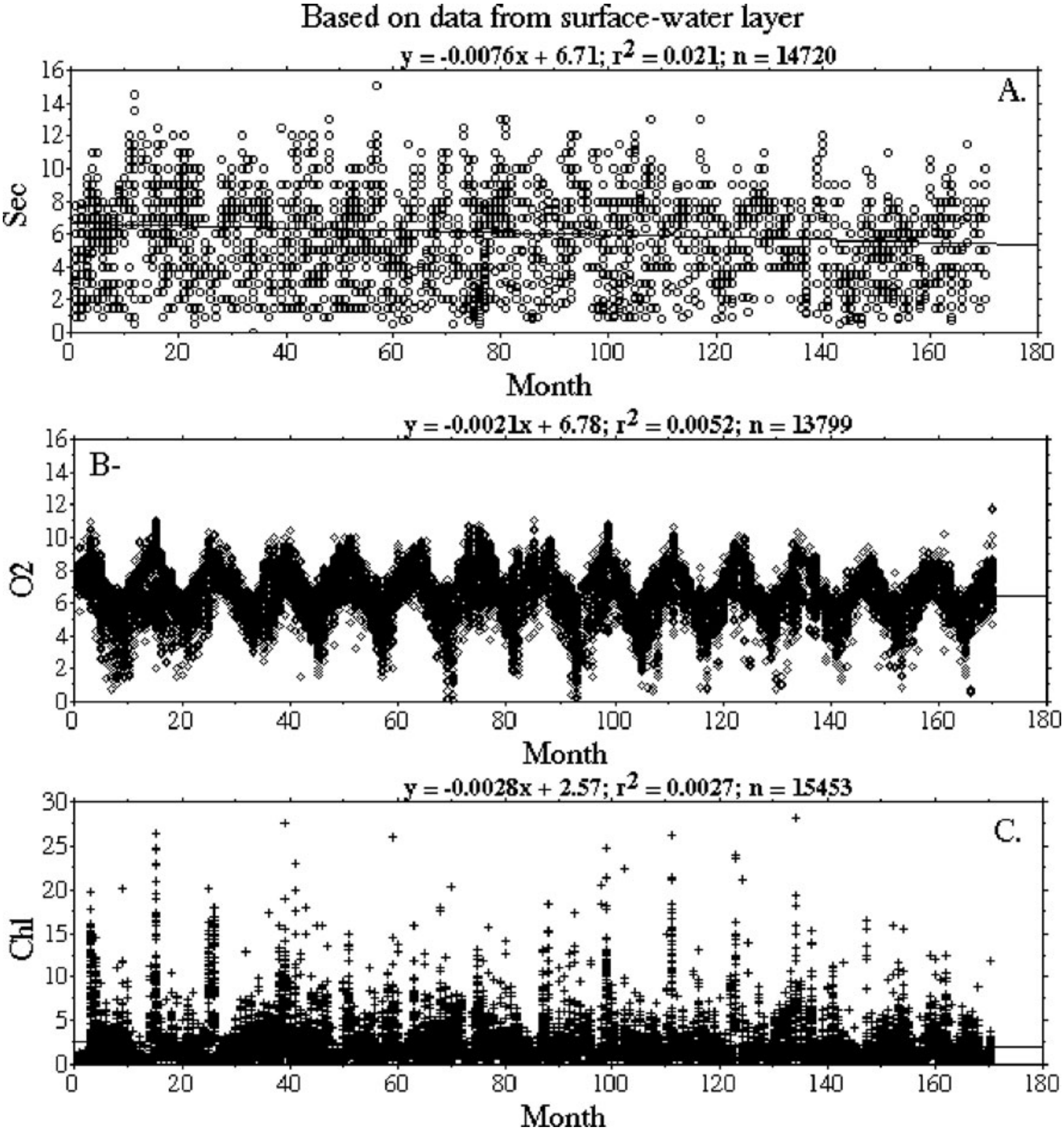


Fig. 2.9 The temporal variation in (**a**) Secchi depths (m), (**b**) oxygen concentrations (O_2), and (**c**) concentrations of chlorophyll-a ($\mu g\ L^{-1}$) in the surface-water layer of the Kattegat in the years 1995–2008 (month 1 is January of 1995). The figure also gives statistical trend analyses (regression line; coefficient of determination, r^2, and number of data, n; data from SMHI)

- Figure 2.11 gives the temporal (monthly) trend in tributary water discharge from Swedish rivers entering the Kattegat. Here, one can see a characteristic seasonal variation with high water discharge in spring, but also this trend is very weak.
- Figure 2.12 illustrates another problem related to the concept of "limiting" nutrient. Using data from the Baltic Proper, this figure gives a situation where the chlorophyll-a concentrations show a typical seasonal "twin peak" pattern with a pronounced

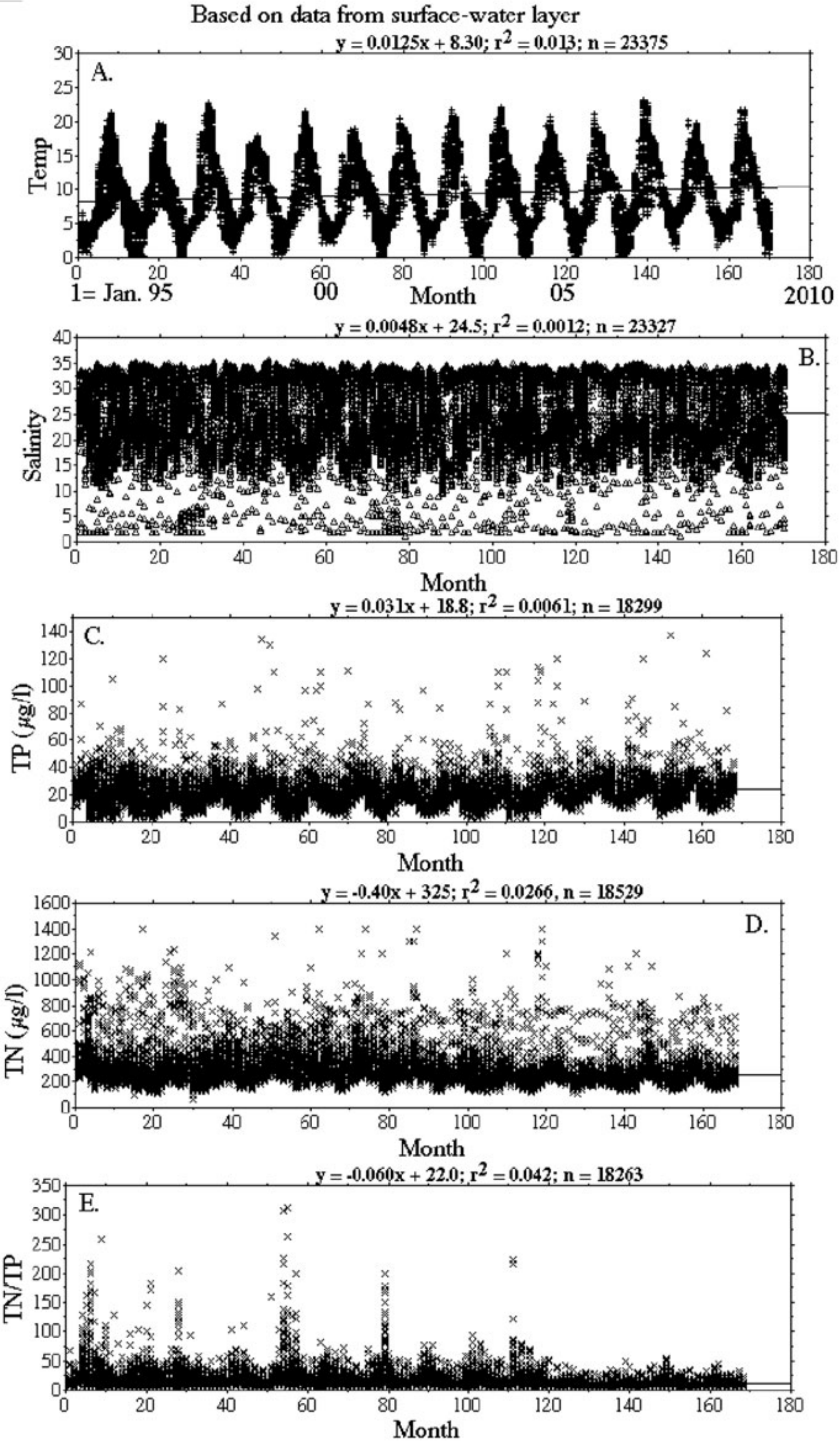


Fig. 2.10 The temporal variation in (**a**) temperatures, (**b**) salinities (psu), (**c**) TP concentrations, (**d**) TN concentration, and (**e**) the TN/TP ratio in the surface-water layer of the Kattegat in the years 1995–2008 (month 1 is January of 1995). The figure also gives statistical trend analyses (regression line; coefficient of determination, r^2, and number of data, *n*). Data from SMHI

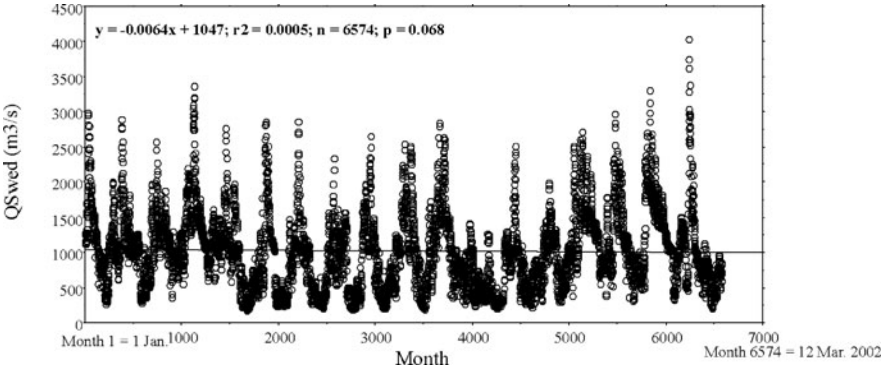


Fig. 2.11 The temporal variation in monthly tributary water discharge from Swedish rivers entering the Kattegat in the period 1985–2002. The figure also gives tatistical trend analyses (regression line; coefficient of determination, r^2, and number of data, n). Data from SMHI

peak in April. The higher the primary production, the more bioavailable nitrogen (nitrate, ammonium, etc.) and phosphorus (phosphate) are being used by the algae (the spring bloom is mainly diatoms) and eventually the nitrate concentration drops to almost zero and the primary production decreases – but the important point is that the primary production, the phytoplankton biomass, and hence also the concentration of chlorophyll-a remain high during the entire growing season!

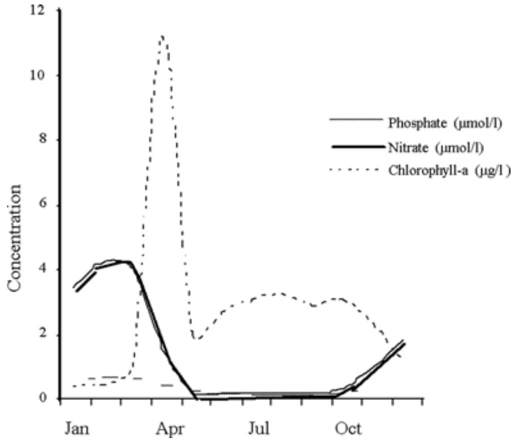


Fig. 2.12 Variations in chlorophyll-a concentrations, phosphate, and nitrate in the Baltic Sea (using data from the Gotland deep between 1993 and 2003; data from SMHI, Sweden)

Trends in nutrient inputs to the Kattegat have to some extent been investigated by Carstensen et al. (2006). They found a significant decrease from 1989 to 2002 in TP inputs to Kattegat, Öresund, and the Belt Sea from the catchment but no changes in TN inputs from land or from the atmosphere during this period. Carstensen et al. (2006) also correlated changes in nutrient inputs from land with changes in nutrient concentrations of Kattegat waters, but failed to account for any trends in nutrient inputs from the Skagerrak and the Baltic Proper. Carstensen et al. (2006) dismissed the possibility of explaining nutrient trends in bottom waters of the Kattegat by nutrient trends in the Skagerrak on the grounds that nutrient concentrations in the Skagerrak are very low and scantly influenced by inputs from land.

However, although nutrient *concentrations* are low in the Skagerrak and the Baltic Proper compared to concentrations in many tributaries, nutrient *fluxes* from the Skagerrak and the Baltic Proper are very large in a mass-balance context, which has been noted by Eilola and Sahlberg (2006) and which will be further elaborated in this work. Comprehensive trends in TN and TP inputs to the Kattegat from land plus inputs from the atmosphere, the Skagerrak, and the Baltic Proper have to the best of our knowledge not been studied.

2.2.4 The Dilemma Related to Predictions of Cyanobacteria

Figure 2.13 illustrates this dilemma using data for the Kattegat. The figure gives the TN/TP ratio on the y-axis and the surface-water temperature on the x-axis. It has been demonstrated by analyses of empirical data from many systems that there exists a threshold value for blooms of cyanobacteria when the TN/TP ratio is lower than 15 and when the SW temperatures are higher than 15°C (see Håkanson et al. 2007).

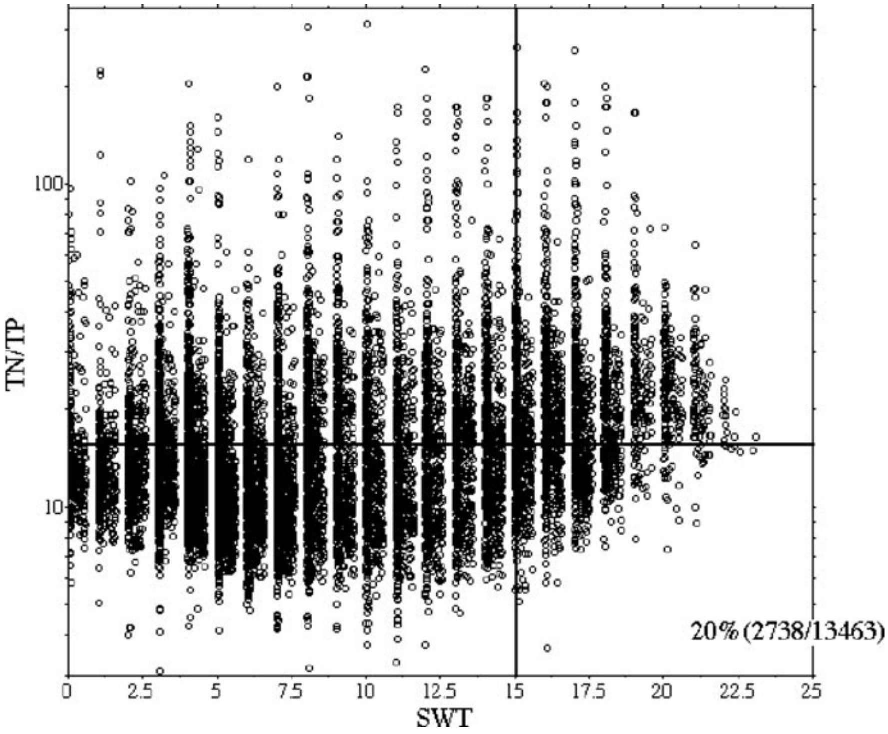


Fig. 2.13 The relationship between temporal TN/TP ratio and surface-water temperatures in the Kattegat in the years 1995–2008 (month 1 is January of 1995). The figure also illustrates threshold temperatures and TN/TP ratios (at 15) for cyanobacteria. Data from SMHI

Based on this, one should expect that the conditions in the Kattegat would favor cyanobacteria in about 20% of the time (Fig. 2.13). However, cyanobacteria do not seem to abound in Kattegat but they certainly abound in the Baltic Sea (see Håkanson and Bryhn 2008a, 2008c). In hypertrophic lakes, the biomass of cyanobacteria can be very high with concentrations of about 100 mg L^{-1} (Smith 1985). Howarth et al. (1988a, 1988b) found no data on N-fixing planktonic species in estuaries and coastal seas, except for the Baltic Sea and the Peel-Harvey estuary, Australia. Also results from Marino et al. (2006) support this general lack of N-fixing cyanobacteria in estuaries. There are more than 10 nitrogen-fixing cyanobacteria species in the Baltic Proper (Wasmund et al. 2001). A field study in the Baltic Sea (Wasmund 1997) indicated that in this brackish environment cyanobacteria have the highest biomass at 7–8 psu and that the blooms in the Kattegat and Belt Sea are more frequent if the salinity is below 11.5 psu (see also Sellner 1997). A laboratory experiment with cyanobacteria from the Baltic Sea supports the results that the highest growth rate was at salinities in the range between 5 and 10 psu (Lehtimäki et al. 1997). So, the scarcity of cyanobacteria in the Kattegat may be related to the relatively high salinity of about 25 psu in this system. This also means that in this mass-balance modeling for nitrogen, there is no atmospheric nitrogen fixation.

2.2.5 The Reasons Why This Modeling Is Not Based on Dissolved Nitrogen or Phosphorus

At short timescales (seconds to days), it is evident that the causal agent regulating/limiting primary production is the concentration of the nutrient in bioavailable forms, such as DIN (dissolved inorganic nitrogen) and DIP, nitrate, phosphate, and ammonia. Short-term nutrient limitation is often determined by measuring DIN and DIP concentrations or by adding DIN and/or DIP to water samples in bioassays. However, information on DIN and DIP from real coastal systems often provides poor guidance in management decisions because

- DIN and DIP are quickly regenerated (Dodds 2003). For example, zooplankton may excrete enough DIN to cover for more than 100% of what is consumed by phytoplankton (Mann 1982). In highly productive systems, there may even be difficulties to actually measure nutrients in dissolved forms because these forms are picked up so rapidly by the algae. Dodds (2003) suggested that only when the levels of DIN are much higher than the levels of DIP (e.g., 100:1), it is unlikely that DIN is limiting and only if DIN/DIP < 1, it is unlikely that P is the limiting nutrient. He also concluded that DIN and DIP are poor predictors of nutrient status in aquatic systems compared to TN and TP.

- Phytoplankton and other primary producers also take up dissolved organic N and P (Huang and Hong 1999, Seitzinger and Sanders 1999, Vidal et al. 1999).
- DIN and DIP are highly variable in most aquatic systems including the Kattegat (see Håkanson and Bryhn 2008a, 2008c and Tables 2.3 and 2.4) and are, hence, very poor predictors of phytoplankton biomass and primary production (as measured by chlorophyll concentrations; see Fig. 2.14).
- Primary production in natural waters may be limited by different nutrients in the long run compared to shorter time perspectives (see Redfield 1958, Redfield et al. 1963). Based on differences in nutrient ratios between phytoplankton and seawater, Redfield (1958) hypothesized that P was the long-term regulating nutrient, while N deficits were eventually counteracted by nitrogen fixation. Schindler (1977, 1978) tested this hypothesis in several whole-lake experiments and found that primary production was governed by P inputs and unaffected by N inputs, and that results from bioassays were therefore irrelevant for management purposes. Redfield's hypothesis has also been successfully tested in modeling work for the global ocean (Tyrrell 1999) and the Baltic Proper (Savchuk and Wulff 1999). However, Vahtera et al. (2007) have used a "vicious circle" theory to suggest that both nutrients should be abated to the Baltic Sea since they may have different long-term importance at different times of the year.

So, the concentrations of the bioavailable fractions, such as DIN and DIP in µg L^{-1} or other concentration units, cannot as such regulate primary phytoplankton production in µg day^{-1} (or other units), since primary production is a flux including a time dimension and the nutrient concentration is a concentration without any time dimension. The central aspect has to do with the flux of DIN and DIP to any given system and the regeneration of new DIN and DIP related to bacterial degradation of organic matter containing N and P. The concentration of DIN and DIP may be very low and the primary phytoplankton production and biomass can be high as in Fig. 2.12 because the regeneration and/or inflow of DIN and DIP is high.

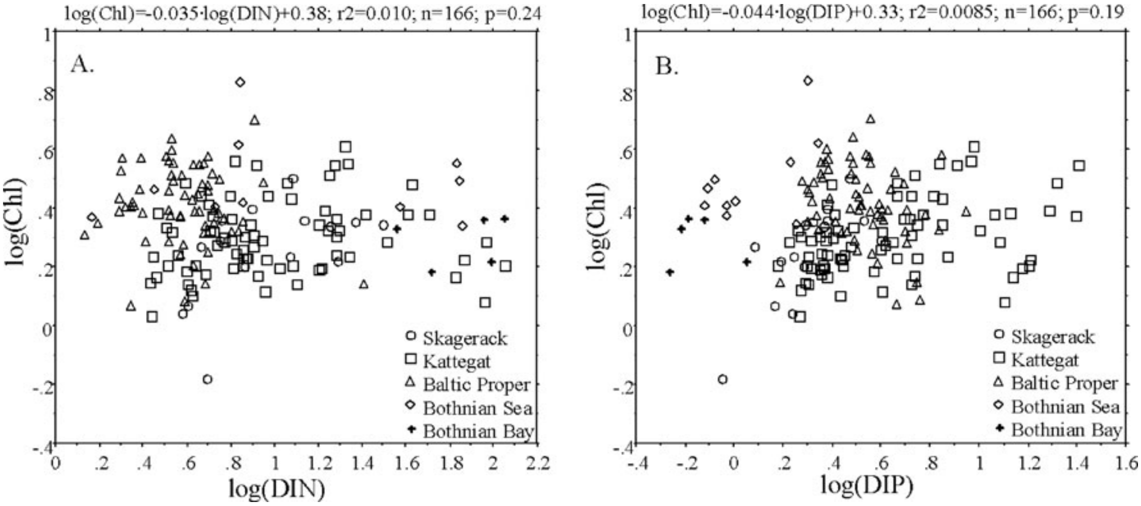


Fig. 2.14 Empirical data from the Baltic Sea, Kattegat, and Skagerrak on mean monthly chlorophyll-a concentrations (logarithmic data) versus empirical data (log) on DIN and DIP, respectively. The figure also gives the equations for the regressions and the corresponding r^2 values (from Håkanson and Bryhn 2008a)

The regeneration of DIN and DIP concerns the amount of TN and TP available in the water mass, i.e., TN and TP represent the pool of the nutrients in the water, which can contribute with new DIN and DIP. It should be stressed that phytoplankton has a typical turnover time of about 3 days and bacterioplankton has a typical turnover time of slightly less than 3 days (see Håkanson and Boulion 2002). This means that within a month there can be 10 generations of phytoplankton, which would need both DIN and DIP in the approximate proportions given by the Redfield ratio (7.2 in grams).

2.2.6 The Reasons Why It Is Generally Difficult to Model Nitrogen

There are four highlighted spots with question marks in Fig. 2.15 indicating that for many coastal systems, it is very difficult to quantify some of the most important transport processes in a general manner for nitrogen. Three of them are denitrification, atmospheric wet and dry deposition, and nitrogen fixation, e.g., by certain forms of cyanobacteria.

Figure 2.15 also highlights another major uncertainty related to the understanding of nitrogen fluxes in coastal systems, the particulate fraction, which is necessary for quantifying sedimentation. Atmospheric nitrogen fixation may be very important in contexts of mass-balance calculations for nitrogen (see Rahm et al. 2000) and in this modeling; the same value for atmospheric nitrogen deposition has been used as in the OSPAR model by SMHI. The data on atmospheric nitrogen deposition for the Kattegat should be reasonable in terms of order-of-magnitude values. Without empirically well-tested algorithms to quantify nitrogen fixation, crucial questions related to the effectiveness of the remedial measures to reduce nutrient discharges to aquatic systems cannot be properly evaluated, since costly nitrogen reductions may be compensated for by nitrogen fixation by cyanobacteria. However, this is a problem in many systems, such as the Baltic Sea, but not in the Kattegat where there seem to be no significant amounts of cyanobacteria.

2.2.7 Comments and Conclusions

Traditional hydrodynamic or oceanographic models to calculate water fluxes to, within, and out of coastal areas generally use water temperature data (the thermocline) or salinity (the halocline) to differentiate between different water layers. This section has motivated another approach, the theoretical wave base as calculated from process-based sedimentological criteria, to differentiate between the surface-water layer and lower vertical layers and this approach gives one characteristic value for each basin. Morphometric data for

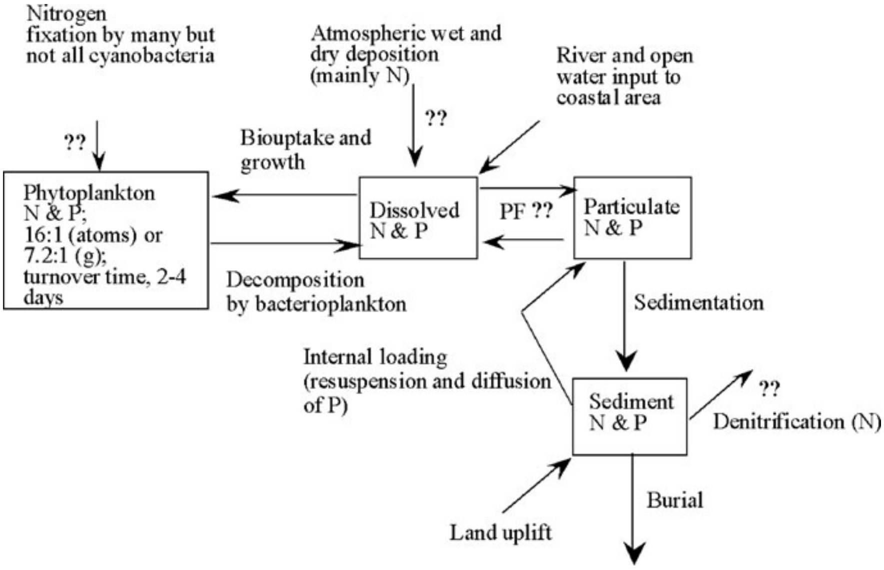


Fig. 2.15 Overview of important transport processes and mechanisms related to the concept of "limiting" nutrient (from an illustration for the Baltic Sea from Håkanson and Bryhn 2008a)

the Kattegat and the hypsographic curve have been used in the CoastMab modeling. The basic aim of this section has been to present empirical data from the Kattegat on total phosphorus (TP), total nitrogen (TN), chlorophyll, Secchi depth, water temperature, and salinity. The empirical data from the Kattegat show the following:

1. All relevant water variables in the SW layer of the Kattegat have been fairly stable in the period between 1995 and 2008.
2. There is a small increase in surface-water temperatures in the Kattegat (compare global warming).
3. The salinities have also been fairly stable since 1995.
4. The concentration of chlorophyll-a shows a very slowly decreasing trend in the surface-water layer of the Kattegat since 1975. The seasonal pattern in monthly median chlorophyll-a concentrations is relatively obscure.
5. The water column has been divided into two layers, separated by the theoretical wave base. This describes the conditions very well.

The long-term trends in TN and TP inputs to the Kattegat from land plus inputs from the atmosphere, the Skagerrak, and the Baltic Proper are, however, largely unknown.

2.3 Water, SPM, Nutrient, and Bioindicator Modeling

2.3.1 Background on Mass Balances for Salt and the Role of Salinity

The salinity is of vital importance for the biology of coastal areas influencing, e.g., the number of species in a system (see Remane 1934) and also the reproductive success, food intake, and growth of fish (Rubio et al. 2005, Nissling et al. 2006). Furthermore, a higher salinity increases the flocculation and aggregation of particles (see Håkanson 2006) and hence affects the rate of sedimentation, which is of particular interest in understanding variations in water clarity within and among coastal areas. More salt in the water, greater the flocculation of suspended particles. This does influence not only the concentration of particulate matter, but also the concentration of any substance with a substantial particulate phase such as phosphorus and nitrogen. The salinity also affects the relationship between total phosphorus (TP), total nitrogen (TN), and primary production/biomass (chlorophyll-a; Håkanson and Bryhn 2008a, 2008c). These relationships are shown in Figs. 2.16 and 2.17 and they are used in this work to calculate chlorophyll-a concentrations from dynamically modeled salinities in the different sub-basins, from dynamically modeled phosphorus and nitrogen concentrations, and from information on the number of hours with daylight. The salinity is easy to measure and the availability of salinity data for the Kattegat is very good.

So, Figs. 2.16 and 2.17 illustrate the role of salinity in relation to the Chl/TP and Chl/TN ratios. The figures give the number of data in each salinity class; the box-and-whisker plots give the medians, quartiles, percentiles, and outliers; and the table below the diagram provides information on the median values, the coefficients of variation (CV = SD/MV; SD = standard deviation; MV = mean value), and the number of systems included in each class (*n*). These results are evidently based on many data from systems covering a wide salinity gradient. An interesting aspect concerns the pattern shown in the figure. One can note the following:

- The median value for the Chl/TP ratio for lakes is 0.29, which is almost identical to the slope coefficient for the key reference model for lakes (0.28 in the OECD model; see OECD 1982).
- The Chl/TP ratio changes in a wave-like fashion when the salinity increases. It is evident that there is a minimum in the Chl/TP ratio in the salinity range between 2 and 5. Subsequently, there is an increase up to the salinity range of 10–15 and then a continuous decrease in the Chl/TP range until a minimum value of about 0.012 is reached in the hypersaline systems. From the relationship between the Chl/TN ratio and the salinity, one can identify differences and similarities between the results presented for the Chl/TP ratio.
- At salinities higher than 10–15, there is a steady decrease also in the Chl/TN ratio (note that there are no data on TN from the hypersaline Crimean lakes).

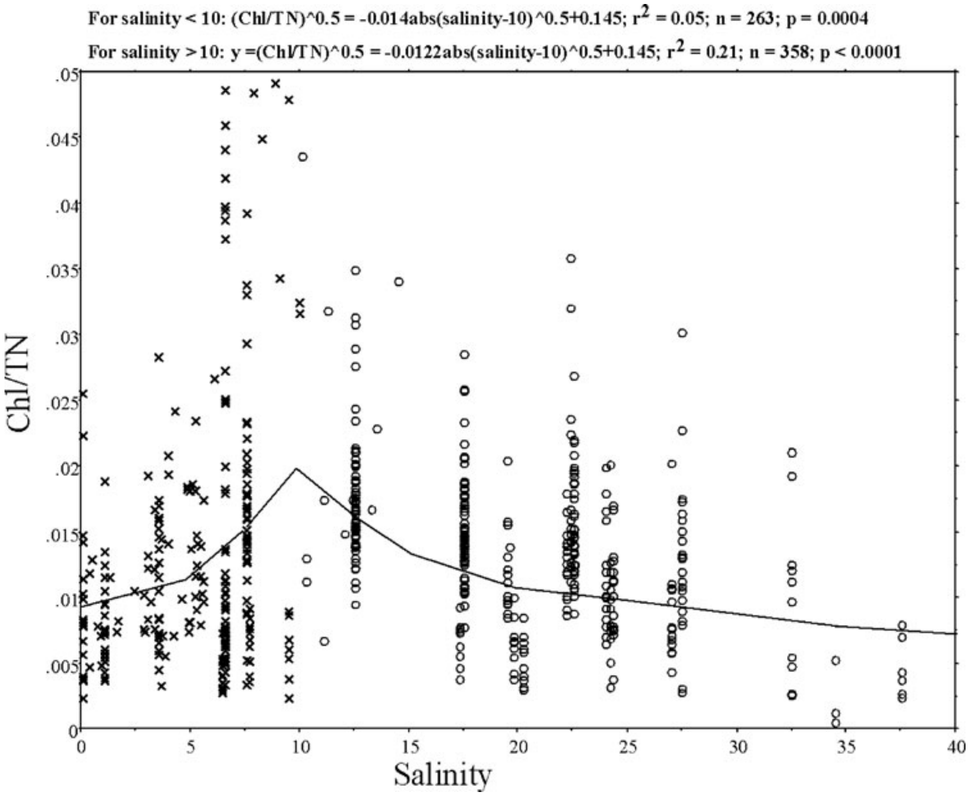
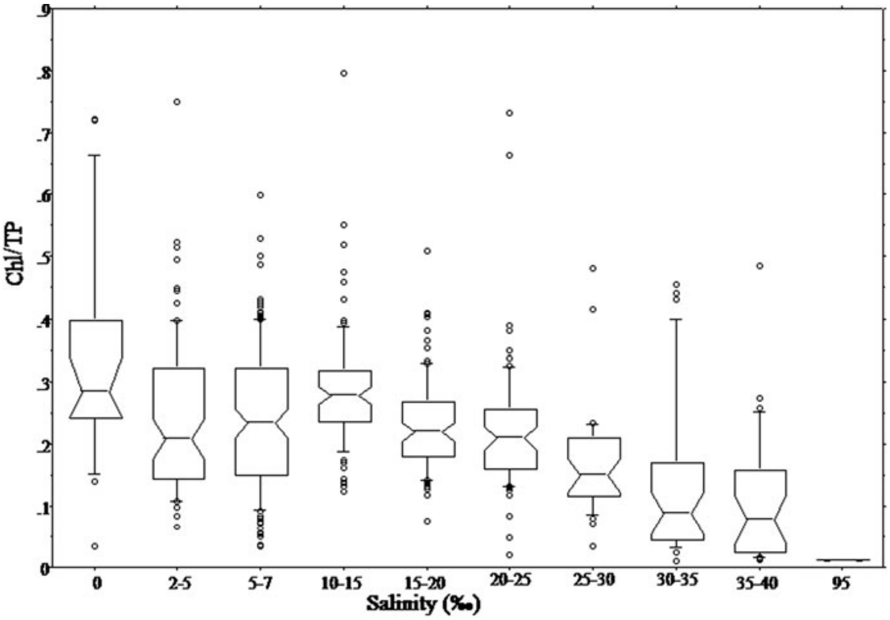


Fig. 2.16 Scatter plot of all available data relating the ratio Cl/TN to salinity (psu). The figure also gives two regressions for salinities either below (*crosses*) or higher than the threshold value of 10 (*circles*) (from Håkanson and Bryhn 2008a). Note that for the Kattegat, the surface-water (SW) salinity is about 25 psu; if TN is 300 μg L^{-1}, this gives Chl $\approx$ 3 μg L^{-1}. The scatter around the given regression partly depends on light, uncertainties in data, and uncertainties in the particulate coefficient for nitrogen

- The Chl/TN ratio attains a maximum value for systems in the salinity range between 10 and 15 and significantly lower values in lakes and less saline brackish systems.
- The table in Fig. 2.16 gives the median Chl/TN values and they vary from 0.0084 (for lakes), to 0.017 for brackish systems in the salinity range between 10 and 15, to very low values (0.0041) for marine coastal systems in the salinity range between 35 and 40.

The water exchange in the Kattegat is calculated using the CoastMab model for salt. This section will present monthly budgets for water and salt in the Kattegat. Mass-balance models have long been used as a tool to study lake eutrophication (Vollenweider 1968, OECD 1982) and also used in different coastal applications (see Håkanson and Eklund 2007, Håkanson and Bryhn 2008c). Mass-balance modeling makes it possible to predict what will likely happen to a system if the conditions change, e.g., a reduced discharge of a pollutant related to a remedial measure. Mass-balance modeling can be performed at different scales depending on the purpose of the study. A large number of coastal models do exist, all with their pros and cons. For example, the 1D nutrient model described by Vichi et al. (2004) requires meteorological input data with a high temporal resolution, which makes forecasting for time periods longer than 1 week ahead problematic.

The 3D model used by Schernewski and Neumann (2005) has a temporal resolution of 1 min and a spatial resolution of 3 nm (nautical miles), which means that it is difficult to find reliable empirical data to run and validate the model. Several water balance studies have also been carried out in the Kattegat and the Baltic Sea, see, e.g., Jacobsen (1980), HELCOM (1986, 1990), Bergström and Carlsson (1993, 1994), Omstedt and Rutgersson (2000), Stigebrandt (2001),



Median:	0.29	0.21	0.24	0.28	0.22	0.21	0.15	0.11	0.079	0.012
CV:	0.54	0.52	0.48	0.32	0.29	0.42	0.53	0.89	0.98	0.16
n:	20	74	140	82	79	61	30	28	24	3

Fig. 2.17 Box-and-whisker plot (showing medians, quartiles, percentiles, and outliers) illustrating the Chl/TP ratio for 10 salinity classes. The statistics give the median values, the coefficients of variation (CV), and the number of data in each class (from Håkanson and Bryhn 2008a). Note that for the Kattegat, the surface-water (SW) salinity is about 25 psu; if TP is 20 μg L^{-1}, this gives Chl $\approx$ 3 μg L^{-1}. The scatter around the given regression partly depends on light, uncertainties in data, and uncertainties in the particulate coefficient for nitrogen

Rutgersson et al. (2002), Omstedt and Axell (2003), Omstedt et al. (2004), and Savchuk (2005). The result of such mass-balance calculations for salt or for other substances depends very much on how the system is defined and how the model is structured.

Within the BALTEX program (BALTEX 2006, BACC 2008), the water and heat balances are major research topics and estimates on the individual terms in the water balance are frequently being revised (e.g., Bergström and Carlsson 1993, 1994, Omstedt and Rutgersson 2000, Rutgersson et al. 2002). The major water balance components in the Baltic Sea are the in- and outflows at the entrance area, river runoff, and net precipitation (Omstedt et al. 2004). Change in water storage needs also to be considered at least for shorter time periods. The different results depend on the time period studied and the length of the period. Several studies have also divided the Baltic Sea into sub-basins and from the water and salt balances estimated the flows (e.g., Omstedt and Axell 2003, Savchuk 2005).

The necessary empirical data on salinity (and other water variables) to run the CoastMab model have originally been obtained from SMHI (the Swedish Meteorological and Hydrological Institute) and data from the period 1995 to 2008 have been used in this work. There are inter-annual and seasonal variations in both net precipitation and riverine water input to the Kattegat (HELCOM 1986, Bergström and Carlsson 1993, 1994, Winsor et al. 2001) as well as in the exchange of water with the Kattegat and the salinity of this water (Samuelsson 1996). This work has focused on a period when there is access to comprehensive data for the mass balances for salt, but also for this period there are inherent uncertainties in the data. This is shown by the CV values in Tables 2.3 and 2.4.

The fluxes and retention rates for the different sub-basins and compartments of the Kattegat, as defined in this mass-balance modeling for salt, will be used in the following mass-balance modeling for phosphorus, nitrogen, and SPM. The basic structuring of this model (CoastMab) enables extensions not just to substances other than salt, but also to systems other than the Baltic Sea and the Kattegat.

2.3.2 Water Fluxes

Figure 2.18 illustrates the basic structure of the model with its two water compartments (SW and DW in the Kattegat) and also results of the modeling for water fluxes. Note that this modeling is done on a monthly basis to achieve seasonal variations, which is important in the mass-balance models for phosphorus, nitrogen, and SPM.

All the water fluxes in Fig. 2.18 are given in km^3 year^{-1} to get an overview. This figure also shows water fluxes from Swedish and Danish tributary rivers, precipitation, and evaporation. For the tributary fluxes data from SHMI for the period 1995–2008 have been used. The salinities in the inflowing water from Skagerrak have been calculated using data exemplified in Table 2.7 for the surface-water inflow.

The model quantifies the fluxes needed to achieve steady-state concentrations for the salinity that correspond as closely as possible to the empirical monthly salinities in the two compartments. All equations have been given by Håkanson and Bryhn (2008a), and they are compiled in Table 2.8.

One can note from Fig. 2.18 that the greatest water fluxes into the Kattegat are the deep-water (DW) flux from Skagerrak (SK) (2,165 km^3 year^{-1}), the surface-water (SW) flux from the Baltic Proper (BP) (960 km^3 year^{-1}); the tributary inflow, precipitation, and deep-water inflow from the Baltic Proper are relatively small (30, 51, and 47 km^3 year^{-1}, respectively). Since this is mass balance for salt, the fluxes out of the system should be equal to the inflow at steady state. These fluxes provide a very important interpretational framework for the other mass balances (for phosphorus, nitrogen, and SPM). From the fluxes of water, one can also define the associated retention times (*T*) and retention rates (1/*T*). The retention rates for water may be used in mass-balance models for, e.g., nutrients since these rates indicate the potential turbulence in the given compartment, and the turbulence regulates the settling velocity for suspended particles – the higher the potential turbulence, the lower the settling velocity for particulate phosphorus (Håkanson and Bryhn 2008a). The retention time for water in each compartment is defined from the total inflow of water (m^3 year^{-1}) and the volume of the compartment (m^3). Empirical salinity data are compared to modeled values in Fig. 2.19a. The inherent empirical uncertainties in the mean monthly salinity values (the CV values) are small, about 0.28 in the SW layer and very small in the DW layer, 0.02 (see Tables 2.3 and 2.4).

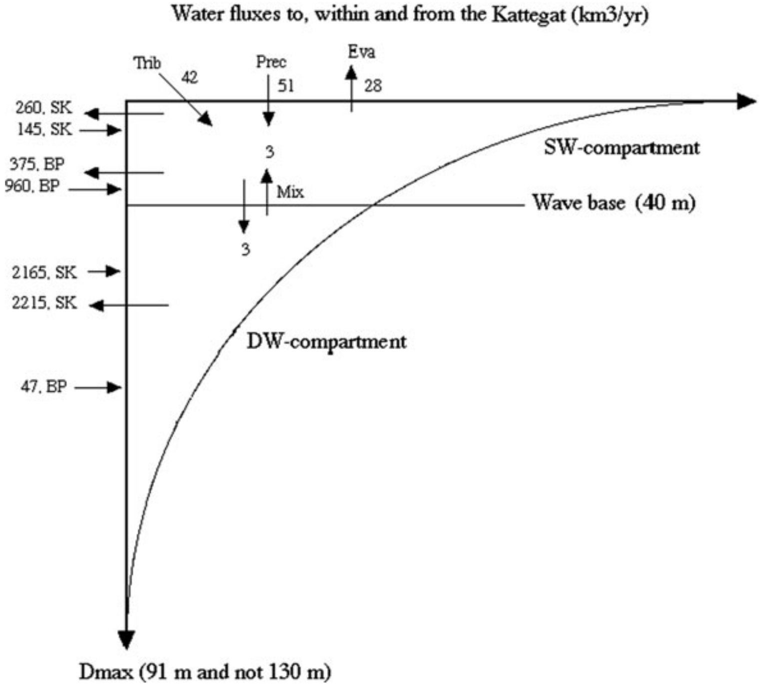


Fig. 2.18 Characteristic annual water fluxes to, from, and within the Kattegat for the period 1995–2008

Table 2.7 Mean monthly values (MV) and coefficients of variation (CV) for variables in the surface-water layer of Skagerrak for the period 1995–2008 for Secchi depth, surface-water temperature (SWT), salinity (Sal), total phosphorus (TP), and total nitrogen (TN)

Month	Secchi (m)	SWT (°C)	Sal (psu)	TP (μg L^{-1})	TN (μg L^{-1})
1	7.8	5.6	32.4	21.8	233.2
2	6.0	4.5	32.0	22.7	261.3
3	6.8	4.6	31.3	18.0	249.4
4	8.3	5.9	30.7	14.6	225.7
5	7.9	8.4	30.3	13.5	207.5
6	6.4	11.3	30.9	12.5	195.3
7	8.4	14.8	31.3	11.4	181.7
8	9.2	16.0	31.9	11.2	163.0
9	8.1	15.2	31.4	13.0	165.8
10	8.1	12.8	30.3	15.1	175.9
11	6.5	10.3	32.4	17.5	186.2
12	9.7	7.9	31.9	21.6	213.8
MV	7.75	9.77	31.39	16.07	204.9
1	0.37	0.27	0.07	0.23	0.30
2	0.32	0.35	0.09	0.17	0.23
3	0.31	0.27	0.11	0.30	0.38
4	0.21	0.18	0.15	0.33	0.27
5	0.25	0.25	0.17	0.35	0.26
6	0.26	0.26	0.14	0.38	0.32
7	0.20	0.20	0.10	0.33	0.29
8	0.27	0.17	0.09	0.40	0.25
9	0.25	0.10	0.11	0.34	0.24
10	0.18	0.12	0.13	0.26	0.18
11	0.18	0.13	0.06	0.30	0.21
12	0.15	0.21	0.09	0.24	0.26
MV	0.25	0.21	0.11	0.30	0.27

The excellent results shown in Fig. 2.19a are not a result of a blind test, rather a result achieved after many calibrations. To understand how the Kattegat system, or any aquatic system, responds to changes in, e.g., loading of toxins, salt, or nutrients, it is imperative to have a dynamic process-based perspective quantifying the factors and functions regulating inflow, outflow, and internal transport processes and retention rates. This section has demonstrated that this modeling using the theoretical wave base rather than traditional temperature data to define the surface-water and deep-water compartments can give excellent correspondence between empirical and modeled data for the salinity. It is often stressed in contexts of marine eutrophication that it is important to develop practically useful general dynamic mass-balance models based on the ecosystem perspective to be able to give realistic evaluations of how systems will respond to changes in nutrient loading or other remedial actions (Smith 2003). The basic aim of this section has been to present data on the fluxes of water and the theoretical retention times for water and salt since those values give fundamental information on how the system reacts to changes in, e.g., nutrient loading. The idea with this modeling is that these water fluxes, water retention rates, and the algorithms to quantify vertical mixing and diffusion among the defined layers should be structured in such a manner that the model can be used to quantify also fluxes of phosphorus, nitrogen, and SPM. This places certain demands on the structure of this model, which are different from oceanographic models, e.g., in quantifying resuspension, mixing, and diffusion and in the requirements regarding the accessibility of the necessary driving variables.

Table 2.8 A compilation of equations in the mass-balance model for salt (CoastMab) for the Kattegat. Abbreviations: F for flow (kg month^{-1}), R for rate (1 month^{-1}), C for concentration (‰ = psu = kg m^{-3}), DC for distribution coefficients (dimensionless), M for mass (kg salt), D for depth in m, A for area in m^2, V for volume in m^3; ET stands for areas with erosion and resuspension (advection) of fine sediments above the theoretical wave base; T is the theoretical retention time (years); flow from one compartment (e.g., SW) to another compartment (e.g., MW) is written as F_{SWMW}; mixing flow is abbreviated as F_{xDWMW}; Q is water discharge (m^3 month^{-1})

Surface water (SW)

$M_{\mathrm{SWKA}}(t) = M_{\mathrm{SWKA}}(t\text{–d}t) + (F_{\mathrm{xDWSWKA}} + F_{\mathrm{tribKA}} + F_{\mathrm{precKA}} + F_{\mathrm{dDWSWKA}} + F_{\mathrm{SWBPKA}} + F_{\mathrm{SWSKKA}} - F_{\mathrm{xSWDWKA}} - F_{\mathrm{evaKA}} - F_{\mathrm{SWKABP}} - F_{\mathrm{SWKASK}})\cdot \mathrm{d}t$

Inflows

$F_{\mathrm{xDWSWKA}} = M_{\mathrm{DWKA}}\cdot R_{\mathrm{xKA}}\cdot V_{\mathrm{SWKA}}/V_{\mathrm{DWKA}}$; mixing flow from DW to SW in KA (kg/months)

$F_{\mathrm{tribKA}} = Q_{\mathrm{tribKA}}\cdot C_{\mathrm{tribKA}}$; tributary inflow to KA (kg/months)

$F_{\mathrm{precKA}} = Q_{\mathrm{precKA}}\cdot C_{\mathrm{precKA}}$; flow to KA from precipitation (kg/months)

$\mathrm{F}_{\mathrm{dDWSWKA}} = M_{\mathrm{DWKA}}\cdot R_{\mathrm{dDWSWKA}}\cdot \mathrm{Const}_{\mathrm{diff}}$; diffusive flow DW to SW in KA (kg/months)

$F_{\mathrm{SWBPKA}} = Q_{\mathrm{SWBPKA}}\cdot C_{\mathrm{SWBP}}$; SW flow from BP to KA (kg/months)

$F_{\mathrm{SWSKKA}} = Q_{\mathrm{SWSKKA}}\cdot C_{\mathrm{SWSK}}$; SW flow from SK to KA (kg/months)

Outflows

$F_{\mathrm{xSWDWKA}} = M_{\mathrm{SWKA}}\cdot R_{\mathrm{xKA}}$; mixing flow from SW to DW in KA (kg/months)

$F_{\mathrm{evaKA}} = M_{\mathrm{SWKA}}\cdot Q_{\mathrm{evaKA}}\cdot 0$; evaporation from BP (kg/months)

$F_{\mathrm{SWKABP}} = Q_{\mathrm{SWKABP}}\cdot C_{\mathrm{SWKA}}$; SW flow from KA to BP (kg/months)

$F_{\mathrm{SWKASK}} = Q_{\mathrm{SWKASK}}\cdot C_{\mathrm{SWKA}}$; SW flow from KA to SK (kg/months)

Deep water (DW)

$M_{\mathrm{DWKA}}(t) = M_{\mathrm{DWKA}}(t\text{–d}t) + (F_{\mathrm{xSWDWKA}} + F_{\mathrm{MWBPKA}} + F_{\mathrm{DWSKKA}} - F_{\mathrm{xDWSWKA}} - F_{\mathrm{dDWSWB}} - F_{\mathrm{DWKASK}})\cdot \mathrm{d}t$

Inflows

$F_{\mathrm{xSWDWKA}} = M_{\mathrm{SWKA}}\cdot R_{\mathrm{xKA}}$; mixing flow from SW to DW in KA (kg/months)

$F_{\mathrm{DWBPKA}} = Q_{\mathrm{DWBPKA}}\cdot C_{\mathrm{DWBP}}$; DW flow from BP to KA (kg/months)

$F_{\mathrm{DWSKKA}} = Q_{\mathrm{DWBPKA}}\cdot C_{\mathrm{DWBP}}$; DW flow from SK to KA (kg/months)

Outflows

$F_{\mathrm{xDWSWKA}} = M_{\mathrm{DWKA}}\cdot R_{\mathrm{xKA}}\cdot V_{\mathrm{SWKA}}/V_{\mathrm{DWKA}}$; mixing flow from DW to SW in KA (kg/months)

$F_{\mathrm{dDWSWKA}} = R_{\mathrm{dDWSWKA}}\cdot M_{\mathrm{DWKA}}\cdot \mathrm{Const}_{\mathrm{diff}}$; diffusive flow DW to SW in KA (kg/months)

$F_{\mathrm{DWKASK}} = Q_{\mathrm{DWKASK}}\cdot C_{\mathrm{DWKA}}$; DW flow from KA to SK (kg/months)

2.3.3 Mass Balances

2.3.3.1 Phosphorus Dynamics

To combat eutrophication, it is fundamental to try to identify the anthropogenic contributions to the nutrient loading. HELCOM (see Table 2.9) has presented very useful data regarding the natural, diffuse, and point source discharges of phosphorus and nitrogen to the Kattegat. Evidently, the natural nutrient fluxes should not be reduced, only a certain part of the anthropogenic fluxes from point sources and diffuse emissions.

As a background to the discussion to find the best possible remedial strategy to mitigate the eutrophication in the Baltic Sea, Table 2.10 shows central aspects of the strategy proposed by HELCOM (2007b), which was also accepted by the Baltic States in November 2007. Based on costs for building water treatment plants in the Baltic States and the St. Petersburg area (20,000 euro t^{-1} P; HELCOM and NEFCO 2007), the action alternative motivated in Håkanson and Bryhn (2008a; about 10,000 t phosphorus year^{-1}) would cost 0.2–0.4 billion euro year^{-1}, or about 10% of the cost of the Baltic Sea Action Plan.

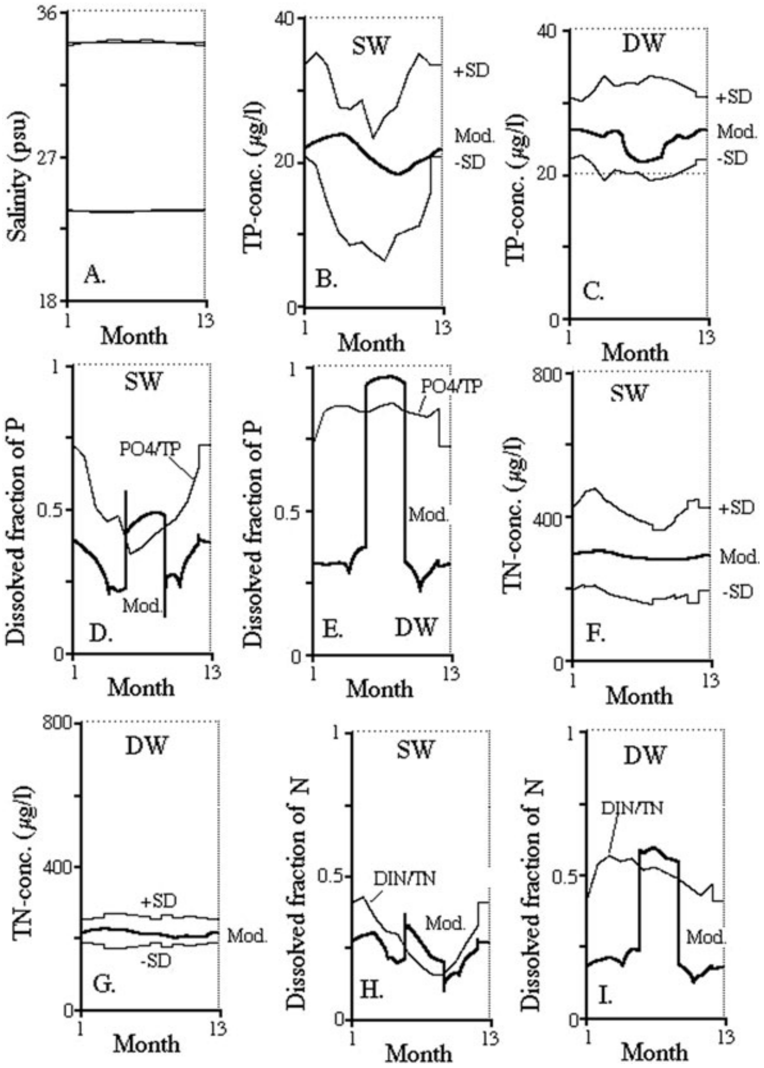


Fig. 2.19 Empirical data versus modeled values in the Kattegat. (**a**) Salinities (the two *upper lines* give the DW salinities, the two *lower lines* the SW salinities), (**b**) modeled TP concentrations in the surface-water (SW) layer versus ±1 standard deviation (SD) of the mean empirical value, (**c**) modeled TP concentrations in the deep-water (DW) layer versus ±1 SD, (**d**) modeled dissolved fractions of phosphorus in the SW layer versus PO_4/TP ratio, (**e**) modeled dissolved fractions of phosphorus in the DW layer versus the PO_4/TP ratio, (**f**) modeled TN in SW layer versus ± 1 SD, (**g**) modeled TN in DW versus ±1 SD, (**h**) modeled dissolved fractions of N in SW versus the DIN/TN ratio, (**i**) modeled dissolved fractions of N in DW versus DIN/TN

Table 2.9 Transport of phosphorus and nitrogen from land to the Kattegat in the year 2000 (t; from HELCOM 2000)

Nutrient	Natural	Diffuse	Point sources	Total load	From Sweden (%)
Phosphorus	363	1, 063	387	1, 813	46.8
Nitrogen	13, 561	53, 661	6, 452	73, 674	54.3

In the requested budgets for nitrogen and phosphorus for the Kattegat, it is essential to include all major transport processes in order to understand the situation and especially to know how remedial measures reducing nutrient loading to the system will likely change nutrient concentrations in water and sediments. The importance of the internal fluxes and the transport between basins compared to the anthropogenic nutrient input from land has also been shown by Christiansen et al. (1997) in a study of parts of the Kattegat. The transport processes (sedimentation, resuspension, burial, diffusion, mixing, biouptake, etc.) for phosphorus, nitrogen, and SPM quantified in the CoastMab model are general and apply for all substances in all/most aquatic systems (see Fig. 2.20), but there are also substance-specific parts (mainly related to

Table 2.10 Required nutrient reductions according to HELCOM (2007a, 2007b)

	Phosphorus (t)	Nitrogen (t)
Denmark	16	17,210
Estonia	220	900
Finland	150	1,200
Germany	240	5,620
Latvia	300	2,560
Lithuania	880	11,750
Poland	8,760	62,400
Russia	2,500	6,970
Sweden	290	20,780
Transboundary pool 1	660	3,780
Sum	15,016	133,170

the particulate fraction, the criteria for diffusion from sediments, and the fact that nitrogen appears with a gaseous phase).

So, these processes have the same names for all systems and for all substances:

- Sedimentation is the flux from water to sediments or to deeper water layers of suspended particles and nutrients attached to such particles.
- Resuspension is the advective flux from sediments back to water, mainly driven by wind/wave action and slope processes.
- Diffusion is the flux from sediments back to water or from water layers with high concentrations of dissolved substances to connected layers with lower concentrations. Diffusion is triggered by concentration gradients, which would often be influenced by small-scale advective processes; even after long calm periods, there are currents related to the rotation of the earth, the variations of low and high pressures, temperature variations between day and night, etc.; it should be noted that it is difficult to measure water velocities lower than 1–2 cm s^{-1} in natural aquatic systems.
- Mixing (or large-scale advective transport processes) is the transport between, e.g., surface-water layers and deeper water layers related to changes in stratification (variations in temperature and/or salinity).
- Mineralization (and regeneration of nutrients in dissolved forms) is the decomposition of organic particles by bacteria.
- Primary production is creation of living suspended biomass from sunlight and nutrients.
- Biouptake is the uptake of the substance in biota. In the CoastMab/CoastWeb model, one first calculates biouptake in all types of organisms with short turnover times (phytoplankton, bacterioplankton, benthic algae, and herbivorous zooplankton) and from this biouptake in all types of organisms with long turnover times (i.e., fish, zoobenthos, predatory zooplankton, jellyfish, and macrophytes) to account for the fact that phosphorus circulating in the system will be retained in these organisms and the retention times for phosphorus in these organisms are calculated from the turnover times of the organisms.
- Burial is the sediment transport of matter from the biosphere to the geosphere often of matter from the technosphere.

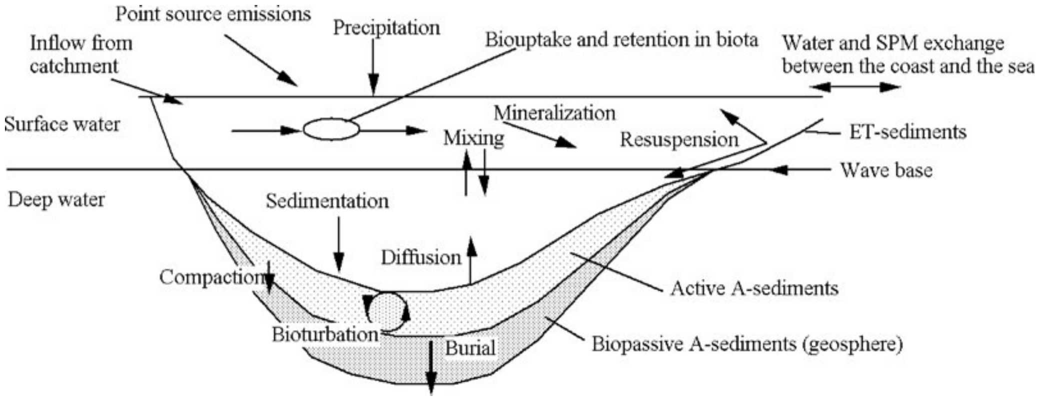


Fig. 2.20 An outline of transport processes (= fluxes) and the structure of the dynamic coastal model (CoastMab) for phosphorus, nitrogen, salinity, and suspended particulate matter (SPM). Note that atmospheric nitrogen fixation and deposition and denitrification are not shown in this figure

- Outflow is the flux out of the system of water and everything dissolved and suspended in the water.

Figure 2.19b, c gives the modeled annual TP concentrations in SW and DW water against the corresponding empirical data. The results in Fig. 2.19 are well within the uncertainty bands given by ±1 standard deviation for the empirical data and one cannot expect better results given the fact that there have been no calibrations and that the dominating transport from the Baltic Proper is based on the mean annual transport. The modeled mean annual TP concentrations in *A*-sediments (0–10 cm) are given in Fig. 2.21a and also these modeled values fall within the requested empirical range (0.5–0.66 mg TP g^{-1} dw). The annual fluxes of phosphorus are shown in Fig. 2.22. These fluxes give information of fundamental importance related to how the Kattegat reacts to changes in phosphorus loading. It should be noted that the phosphorus fluxes to and from organisms with short turnover times (BS) are very large compared to all other fluxes, but the amount of TP found in biota is small compared to what is found in some other compartments.

This illustrates the classical difference between "flux and amount." In the ranking of the annual fluxes for the Kattegat from Fig. 2.22, it is evident that the most dominating fluxes are the ones to and from biota with short turnover times (about 320 kt $year^{-1}$), whereas the average monthly amount of TP in all types of plankton is just about 1.7 kt. Most phosphorus is found in *A*-sediments (104 kt), on ET areas (10 kt), and in the SW layer (5 kt). Looking at the TP fluxes to the Kattegat, the DW flux from Skagerrak is the dominating one (47 kt $year^{-1}$), followed by the SW inflow from the Baltic Proper (20 kt $year^{-1}$), DW inflow from the Baltic Proper (5.4 kt $year^{-1}$), SW inflow from the Skagerrak (2.4 kt $year^{-1}$), tributary inflow (2 kt $year^{-1}$), and atmospheric precipitation (0.1 kt $year^{-1}$). Sedimentation in the SW layer is also important, 3.1 kt $year^{-1}$ to the DW layer and 19 kt $year^{-1}$ to the ET sediments (Fig. 2.23).

Sedimentation in the DW layer is relatively small (4.2 kt $year^{-1}$) since about 50% of the phosphorus in the SW layer and about 85% of the phosphorus in the DW layer (see Table 2.12 and Fig. 2.19d, e) are in dissolved forms, which do not settle out. Figure 2.19d, e gives a comparison between modeled dissolved fractions and empirical ratios between phosphate and total phosphorus. It should be stressed that the dissolved fraction (DF) as defined in the model from the particulate fraction (DF = 1 – PF) is not the same thing as phosphate.

There are several different dissolved forms of phosphorus often abbreviated as DP (DIP + DOP), and Fig. 2.19d, e illustrates that the overall correspondence between modeled DF and the ratio between phosphate and total phosphorus in the Kattegat is reasonable. Together with the relatively high oxygen concentrations in the entire Kattegat, this also implies that diffusion of phosphorus from the *A*-sediments is small in the Kattegat (only 0.008 kt $year^{-1}$). The diffusive flux in the water from the DW compartment to the SW compartment is also small (0.01 kt $year^{-1}$). Burial, i.e., the transport of TP from the sediment biosphere to the sediment geosphere, is 5.1 kt $year^{-1}$.

2.3.4 SPM Dynamics

The dynamic SPM model (CoastMab for SPM) has been described by Håkanson (2006, 2009). The model gave very good results for the tested 17 different Baltic Sea coastal areas. The mean error when empirical data on sedimentation (from sediment traps) were compared to modeled values was 0.075, the median error was –0.05, the standard deviation was 0.48, and the corresponding error/uncertainty for the empirical data was 1.0, as given by the coefficient of variation. This means that the uncertainties in the empirical data set the limit for further improvements of model predictions. The error for the modeled values was defined from the ratio between modeled and empirical data minus 1, so that the error is zero when modeled values correspond to empirical data. There are different sources for SPM:

1. Primary production, which causes increasing biomasses for all types of plankton (phytoplankton, bacterioplankton, and herbivorous zooplankton) influencing SPM in the water.
2. Inflow of SPM to the surface-water layer in the Kattegat from the Baltic Proper and Skagerrak.
3. Inflow of SPM to the deep-water layer (i.e., from Baltic Proper and/or Skagerrak).
4. Tributary inflow.

Table 2.11 gives the panel of driving variables for the dynamic SPM model. These are the site-specific

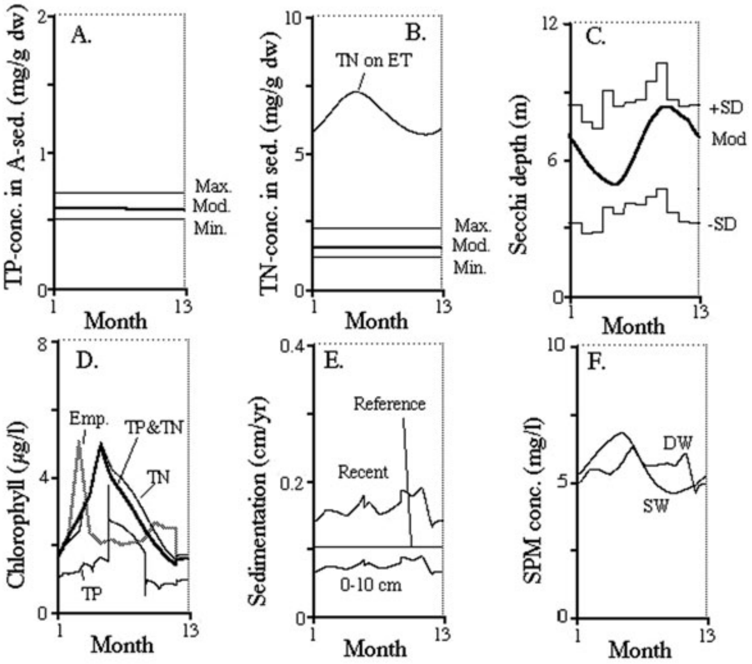
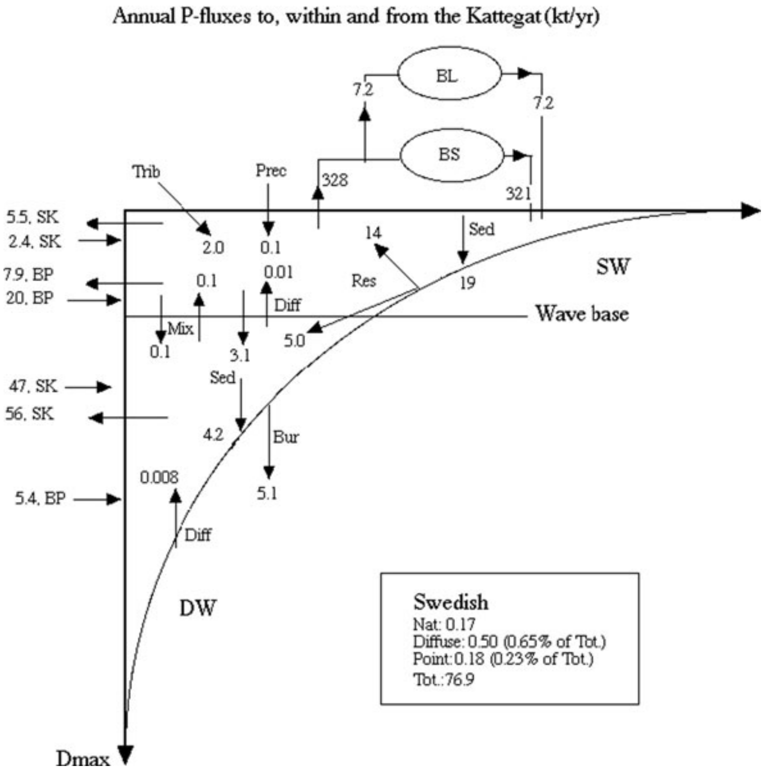


Fig. 2.21 Empirical data versus modeled values in the Kattegat. (**a**) Modeled TP concentrations in the accumulation area sediments (0–10 cm) versus empirical maximum and minimum values, (**b**) modeled TN concentrations in the accumulation area sediments (0–10 cm) versus empirical maximum and minimum values and modeled TN concentrations in recently deposited matter on ET areas, (**c**) modeled Secchi depths versus ±1 standard deviation (SD) of the mean empirical value, (**d**) empirical mean concentrations of chlorophyll, modeled chlorophyll concentrations based on only TP, on only TN, and on both TP and TN (*bold*), (**e**) modeled sedimentation based on the water content of recently deposited matter and on the mean water content in sediments from the upper 10 cm sediment layer and compared to the mean annual sedimentation in the Baltic Proper, and (**f**) modeled SPM concentrations in the surface-water layer and in the deep-water layer in the Kattegat

data on variables needed to run the dynamic SPM model. No other parts of the model should be changed. Figure 2.22 shows the annual SPM fluxes to, within, and from the Kattegat. It is evident that the most dominating abiotic SPM inflow is DW inflow from the Skagerrak (about 12,000 kt year^{-1}), followed by tributary inflow (2,000 kt year^{-1}), SW inflow from the Baltic Proper (1,850 kt year^{-1}), SW inflow from the Skagerrak (800 kt year^{-1}), and DW inflow from the Baltic Proper (100 kt year^{-1}). Sedimentation in the SW layer is also important with 5,600 kt year^{-1}. Sedimentation of SPM from the SW to the DW layer is 950 kt year^{-1}. The flux related to internal loading (resuspension) is 915 kt year^{-1} from ET areas to the SW layer and 325 kt year^{-1} to the DW layer. Burial, i.e., the transport of SPM from the sediment biosphere to the sediment geosphere, is 1,500 kt year^{-1}. The total SPM production is 9,000 kt year^{-1}.

Previous knowledge regarding the SPM concentration, its variation, and the factors influencing variations among and within sites was very limited for the Kattegat. The results discussed here represent a step forward in understanding and predicting SPM in the Kattegat and also in other similar systems. Evidently, it would have been preferable to have access to a large database on SPM, but it is very demanding (in terms of costs, manpower, ships, etc.) to collect such data, especially under storms. It should also be noted that bioturbation, fish movements (Meijer et al. 1990), currents (Lemmin and Imboden 1987), and slope processes (Håkanson and Jansson 1983), as well as boat traffic, trawling, and dredging, might all influence the SPM concentrations and how SPM varies among and within sites. These factors have, however, not been accounted for in this modeling, which does not concern sites but entire basins.

Fig. 2.22 Characteristic annual phosphorus fluxes to, from, and within the Kattegat for the period 1995–2008. Note that the net inflow of phosphorus from the Baltic Proper is 17.5 kt year^{-1}, SMHI (Håkansson 2007, the OSPAR assessment) gives 14 kt year^{-1}



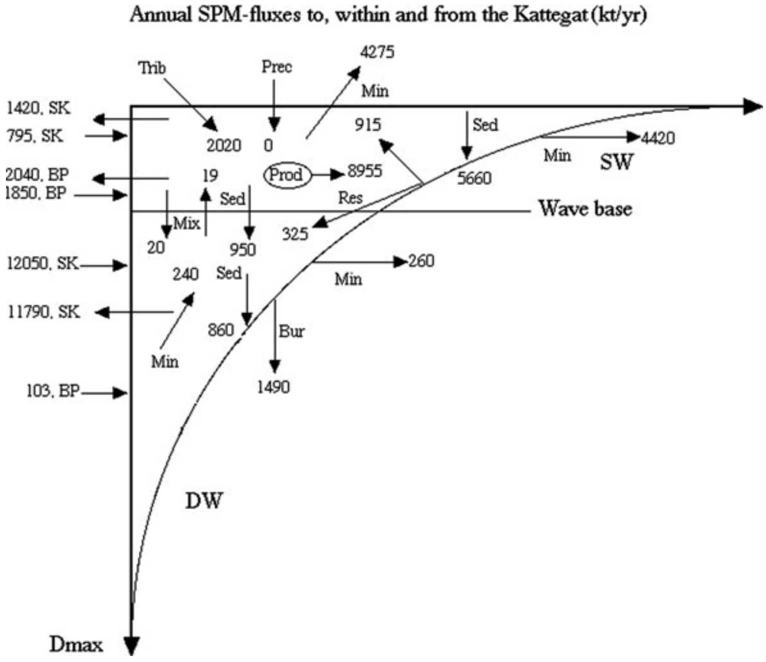


Fig. 2.23 Characteristic annual SPM fluxes to, from, and within the Kattegat for the period 1995–2008

Table 2.11 Panel of driving variables for the dynamic SPM model

A. Morphometric parameters
1. Hypsographic curve
B. Chemical variables
2. Data on salinity, TP, TN concentrations, Secchi depths, and/or SPM concentration in the inflowing water to the coastal area
3. Data and tributary inflow of TP, TN, and SPM
C. Other variables
4. Tributary water discharge or latitude and annual precipitation and evaporation

2.3.5 Nitrogen Fluxes

The dynamic modeling of the nitrogen fluxes uses the same CoastMab model and the same water fluxes (to, within, and from the Kattegat) and the same mixing rates and diffusion rates, as given by the CoastMab model for salinity; it uses the same algorithms for sedimentation, resuspension, biouptake, and retention in biota as the CoastMab model for phosphorus. However, for nitrogen, the following substance-specific modifications have been applied:

1. The particulate fraction of nitrogen (PN) in the SW layer is calculated using the same basic algorithm as used for phosphorus except that for the dissolved fraction of nitrogen in the SW compartment, the monthly correction factors given in Table 2.12 have been used (i.e., the (DIN/TP)/(PO_4/TP) data have been multiplied with the monthly modeled DF value for phosphorus). These modeled values are compared to the empirical DIN/TN values in Fig. 2.19 h and there is a good general agreement.

Table 2.12 Mean monthly values (MV) for the surface-water layer of Kattegat for the period 1995–2008 for the ratios between phosphate (PO_4) and total phosphorus (TP), dissolved inorganic nitrogen (DIN = NO_2+NO_3+NH_4) to total nitrogen (TN), and the ratio between these two ratios

Month	PO_4/TP	DIN/TN	(DIN/TN)/(PO_4/TP)
1	0.72	0.40	0.55
2	0.68	0.42	0.63
3	0.50	0.35	0.70
4	0.45	0.30	0.67
5	0.47	0.29	0.61
6	0.34	0.22	0.64
7	0.36	0.18	0.51
8	0.40	0.15	0.39
9	0.43	0.15	0.36
10	0.46	0.19	0.43
11	0.52	0.26	0.50
12	0.64	0.32	0.51
MV	0.48	0.26	0.54

2. The particulate fraction of nitrogen (PN) in the DW layer in the Kattegat is calculated using the same approach. Table 2.13 gives the monthly correction factors [i.e., (DIN/TP)/(PO_4/TP)]. The modeled values are compared to the empirical DIN/TN values in Fig. 2.19 l and also these values are in relative good agreement with the measured DIN/TN values.
3. Since there are no or very small amounts of nitrogen-fixing cyanobacteria in the Kattegat, N_2 fixation is not accounted for in this modeling.
4. The nitrogen inflow from Skagerrak is based on the same water fluxes as the ones used for the salinity, phosphorus, and SPM, the empirical data for the SW layer in Skagerrak.
5. The nitrogen inflow from the Baltic Proper is based on the same empirical data (TN in $\mu g\ L^{-1}$) for the SW layer (from HELCOM 2007a, 2007b) as presented and used by Håkanson and Bryhn (2008a), i.e.,

Jan.	298.7	Jul.	270.4
Feb.	292.1	Aug.	266.9
Mar.	292.8	Sep.	265.5
Apr.	280.5	Oct.	283.7
May	264.7	Nov.	278.8
Jun.	273.2	Dec.	305.7

For the DW inflow from the Baltic Proper to the Kattegat, the following mean annual value has been used (also from HELCOM 2007a, 2007b): 314 $\mu g\ L^{-1}$.

6. The tributary inflow of nitrogen to the Kattegat is based on the values from HELCOM given in Table 2.10.
7. The denitrification in the Kattegat (in water and sediments) has been calculated as a residual term to satisfy the mass balance for nitrogen. This means that denitrification in the SW layer has been calculated by

$$F_{\mathrm{denitSW}} = 0.01 \cdot (\mathrm{SWT}/9.33) \cdot M_{\mathrm{TNSW}} \cdot V_{\mathrm{SW}}/V \tag{2.5}$$

Table 2.13 Mean monthly values (MV) for the deep-water layer of Kattegat for the period 1995–2008 for the ratios between phosphate (PO_4) and total phosphorus (TP), dissolved inorganic nitrogen (DIN = NO_2+NO_3+NH_4) to total nitrogen (TN), and the ratio between these two ratios

Month	PO_4/TP	DIN/TN	(DIN/TN)/(PO_4/TP)
1	0.72	0.40	0.55
2	0.84	0.53	0.63
3	0.86	0.56	0.65
4	0.86	0.54	0.62
5	0.84	0.55	0.66
6	0.84	0.51	0.60
7	0.86	0.52	0.61
8	0.87	0.50	0.57
9	0.84	0.48	0.57
10	0.83	0.45	0.55
11	0.82	0.42	0.52
12	0.85	0.46	0.54
MV	0.83	0.49	0.59

where 0.01 is a calibration constant (a denitrification rate for the water with the dimension 1 month^{-1}); denitrification is assumed to be temperature dependent (SWT) and 9.33 is the mean annual temperature and SWT/9.33 is a dimensionless temperature moderator; M_{TNSW} is the mass (amount) of TN in the SW layer (g); V_{SW} is the SW volume; and V is the total volume (m^3) so V_{SW}/V is a dimensionless moderator for the SW layer.

Denitrification in the DW layer is given by

$$F_{\mathrm{denitDW}} = 0.01 \cdot (\mathrm{DWT}/9.33) \cdot M_{\mathrm{TNDW}} \cdot V_{\mathrm{DW}}/V \tag{2.6}$$

For the ET sediments, denitrification has been calculated from

$$F_{\mathrm{denitET}} = 3 \cdot M_{\mathrm{TNET}} \cdot (\mathrm{SWT}/9.33) \tag{2.7}$$

where 3 is a calibration constant (a denitrification rate for the ET sediments with the dimension 1 month^{-1}); M_{TNET} is the mass (amount) of TN in the ET sediments (g).

Denitrification in the *A*-sediments (0–10 cm) is given by

$$F_{\mathrm{denitA}} = 0.003 \cdot (\mathrm{DWT}/9.33) \cdot M_{\mathrm{TNA}} \tag{2.8}$$

It should be stressed again that all the denitrification constants are determined from calibrations to satisfy the mass balance for nitrogen and they are not based on general, tested, algorithms which have been proven to work well in many coastal systems. This means that the predictions using the mass-balance model for nitrogen are more uncertain than the predictions of salt, phosphorus, and SPM. The diffusion of dissolved nitrogen from the deep-water layer to the surface-water layer is small in the Kattegat because the concentration gradient is small; the diffusion is calculated with the same algorithm as used for salinity and phosphorus. The predictions for the TN concentrations in the SW and DW layers in the Kattegat are compared to empirical monthly data in Fig. 2.19. Since these modeled values are based on calibrated denitrification rates, the modeled values are close to the empirical data. Annual fluxes of nitrogen are shown in Fig. 2.24. These fluxes give important information of how the Kattegat system likely reacts to changes in nitrogen loading. It should be noted that also the nitrogen fluxes to and from organisms with short turnover times (BS) are very large compared to all other fluxes, but the amounts of TN found in biota are small compared to what is found in other compartments.

In the ranking of the annual fluxes to the Kattegat, the most dominating abiotic fluxes are the TN flux to DW layer from the Skagerrak (435 kt year^{-1}), followed by the SW inflow from the Baltic Proper (270 kt year^{-1}), tributary inflow (82 kt year^{-1}), SW inflow from the Skagerrak (30 kt year^{-1}), atmospheric precipitation (18 kt year^{-1}), and DW inflow from the Baltic Proper (15 kt year^{-1}). Sedimentation in the SW layer is 30 kt year^{-1} to the DW layer and 180 kt year^{-1} to the ET sediments. Sedimentation in the DW layer is 17 kt year^{-1}; about 25% of the nitrogen in the SW layer and about 50% in the DW layer (see Fig. 2.19 h, i) of the nitrogen appear in dissolved form. Figure 2.19 h, i gives a comparison between modeled dissolved fractions and empirical ratios between DIN and TN. It should be stressed that the dissolved form (DF), as defined in the model from the particulate fraction (DF = 1 – PF), is not the same thing as DIN. Figure 2.19 shows that the overall correspondence between modeled DF and the ratio between DIN and TN in the Kattegat is quite good, especially for the SW layer. From Fig. 2.24 one can note that the diffusion of nitrogen from sediments to water and from the DW layer to the SW layer is very small. Denitrification, on the other hand, is large: 13 kt year^{-1} from SW, 170 kt year^{-1} from ET, 8.3 kt year^{-1} from *A*-sediments, and 0.1 kt year^{-1} from the DW layer. Burial of TN from the *A*-sediments is 12 kt year^{-1}.

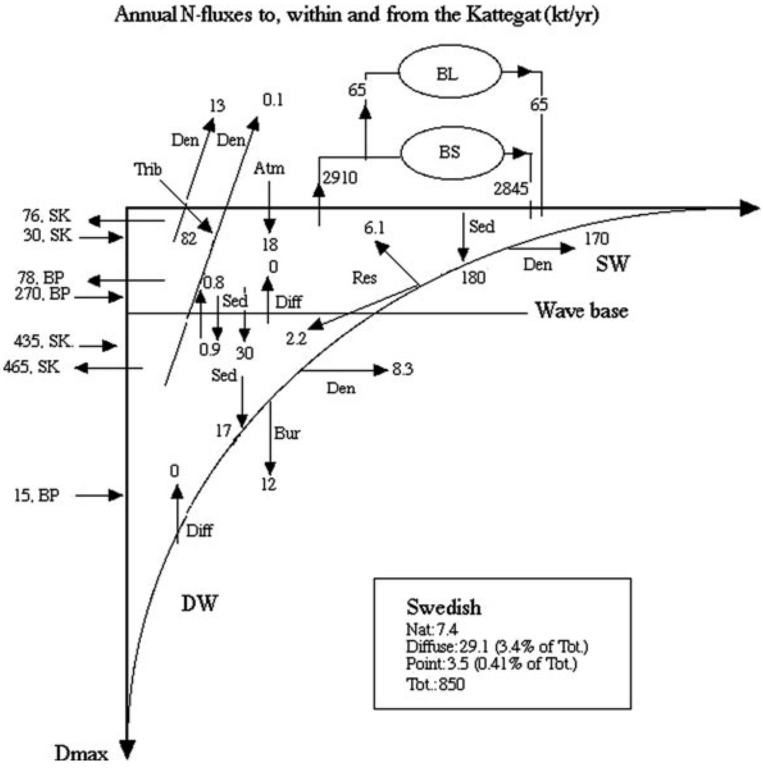


Fig. 2.24 Characteristic annual nitrogen fluxes to, from, and within the Kattegat for the period 1995–2008. Note that the net inflow of nitrogen from the Baltic Sea is 207 kt year^{-1}, SMHI (Håkansson 2007, the OSPAR assessment) gives 190 kt year^{-1}

2.3.6 Predicting Chlorophyll-a Concentrations

Values of chlorophyll-a concentrations in the surface-water layer drive the secondary production (including the production of zooplankton and fish), which means that it is very important to model chlorophyll as accurately as possible. This section will first describe the approach used to model chlorophyll and then present results describing how well modeled values correspond to measured data. Typical chlorophyll-a concentrations for the Kattegat and parts of the North Sea are shown in Fig. 2.3. Values lower than 2 μg L^{-1} (oligotrophic conditions) are found in the northern parts of the Bothnian Bay and the outer parts of the North Sea, while values higher than 20 μg L^{-1} (hypertrophic conditions) are often found in, e.g., the Vistula and Oder lagoons.

Concentrations of chlorophyll-a represent one of the most important bioindicators related to eutrophication. Håkanson and Bryhn (2008a, 2008c) discussed several approaches to predict chlorophyll in the surface-water layer:

1. From regressions based on empirical TN concentrations and light conditions (see, e.g., Fig. 2.25)
2. From regressions based on modeled or empirical TP concentrations (see, e.g., Fig. 2.26), light, salinity, and boundary conditions related to surface-water temperatures

Approaches applied in this work are also given in Table 2.14.

To obtain seasonal/monthly variations, the following calculations will use three approaches, which will be compared to empirical data:

1. Chl from TP, TN, and salinity. This is the approach given in Table 2.14a, which has provided an r^2 value (r^2 = coefficient of determination) of 0.76 and is based on data from 493 systems from many parts of the world. The relationship between TN and TP concentrations for these data is shown in Fig. 2.26 and the results shown in this figure are important in contexts of remedial strategies, since the figure demonstrates that there is generally a significant co-variation between TN and TP

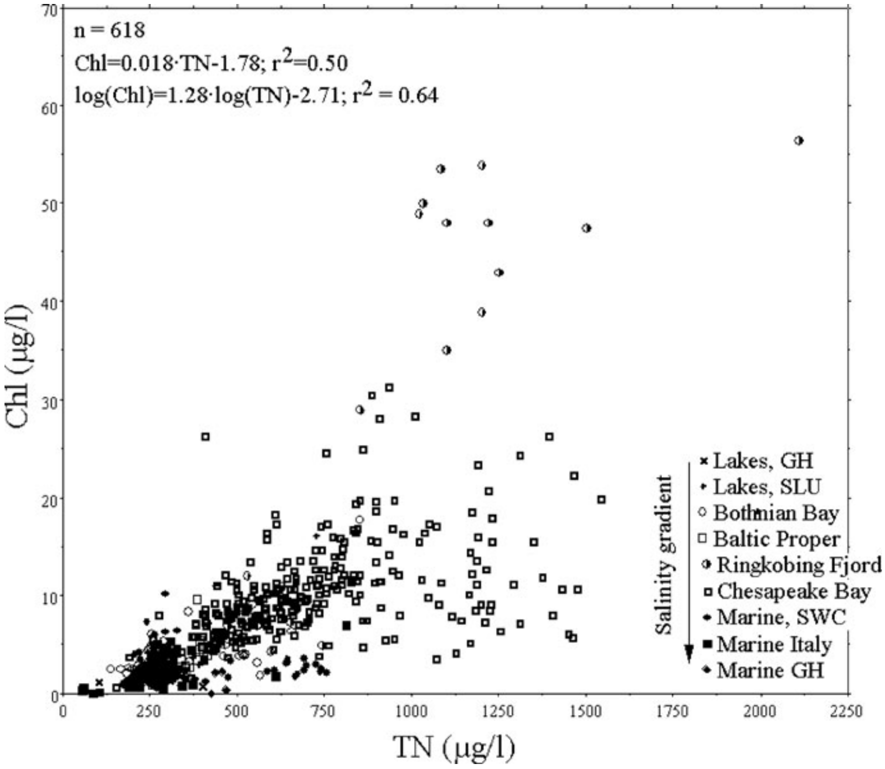


Fig. 2.25 Scatter plot between chlorophyll and TN. The figure also gives regressions for the actual data and log-transformed data for the 618 data points (from Håkanson and Bryhn 2008a)

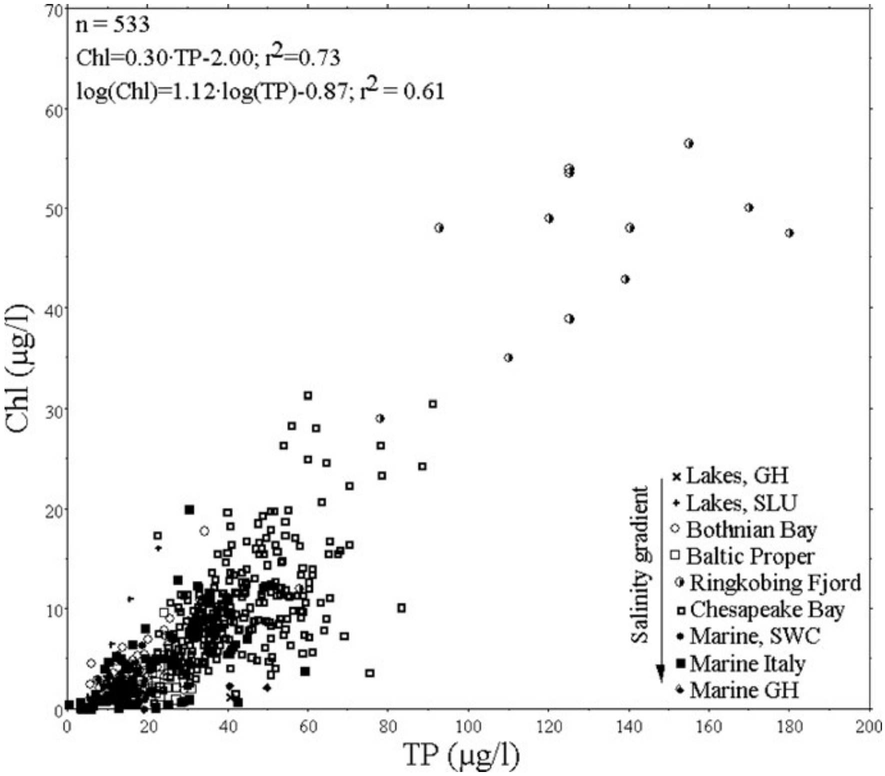


Fig. 2.26 Scatter plot between median surface-water concentrations of chlorophyll and total P (TP) for the growing season from 10 sub-groups constituting a salinity gradient. The figure also gives regressions for the actual data and log-transformed data for the 533 data points. How much of the scatter in this diagram depends on variations in salinity is discussed in the running text (from Håkanson and Bryhn 2008a)

Table 2.14 Compilation of regressions for predicting (A) concentrations of chlorophyll-a in the surface water from TP- and TN concentrations and surface-water salinities using a comprehensive database from 493 coastal systems from many parts of the world and using data from the growing season (see Håkanson and Bryhn 2008a), (B) TP concentrations in the surface water from TN concentrations and surface-water salinities using the same comprehensive database, (C) TN concentrations in the surface water from TP concentrations and surface-water salinities using the same database. $F > 4$ in all cases

Step	r^2	*x*-variable	Model
A.			
1	0.66	TP	log(Chl) = 1.17·log(TP)–0.94
2	0.72	TN	log(Chl) = 0.66·log(TP) +0.73·log(TN)–2.16
3	0.76	Sal	log(Chl) = 0.67·log(TP)+0.57·log(TN)–0.02·abs(Sal-12)–1.62
B.			
1	0.66	TN	log(TP) = 0.923·log(TN)–0.012
2	0.72	Sal	log(TP) = 1.011·log(TN) +0.2245·log(1+Sal)–1.478
C.			
1	0.65	TP	log(TN) = 0.70·log(TP)+1.668
2	0.75	Sal	log(TN) = 0.668·log(TP)–0.0092·Sal+1.830

concentrations and this indicates that one would often reduce also TP concentrations in receiving water systems if remedial measures focus on nitrogen reductions, and vice versa. To achieve realistic seasonal patterns, the dimensional moderator ($Y_{\mathrm{DayL}} = \mathrm{HDL}/12$) based on the number of hours with daylight each month (from Table 2.6) has also been applied in all the following predictions using the regression in Table 2.14a.

2. Chl from TP and salinity. This approach used the results shown in Fig. 2.27 and also modeled values on the dissolved fraction of phosphorus, since this is the only fraction that can be taken up by phytoplankton and since values of the dissolved fraction of phosphorus in the SW layer ($\mathrm{DF_{SW}}$; dim. less) are automatically calculated by the CoastMab model for phosphorus and are thus available for predicting chlorophyll.

   This modeling also uses a boundary condition related to low water temperatures given by

$$\begin{aligned}&\text{If SWT} > 4^\circ\text{C, then } Y_{\mathrm{SWT}} = 1\\ &\qquad \text{else } Y_{\mathrm{SWT}} = (\mathrm{SWT} + 0.1)/4\end{aligned} \tag{2.9}$$

   This water temperature moderator will not influence modeled chlorophyll values when the surface-water temperature is higher than 4°C, but it will lower predicted chlorophyll values during the winter time, and since there is also primary production under ice, the constant 0.1 is added. This moderator has been used and motivated before (see Håkanson and Eklund 2007). This means that using this approach Chl (μg L^{-1}) is predicted from

$$\mathrm{Chl_{Mod}} = \mathrm{TP_{SW}} \cdot \mathrm{DF_{SW}} \cdot Y_{\mathrm{DayL}} \cdot Y_{\mathrm{Sal}} \cdot Y_{\mathrm{SWT}}$$

   $\mathrm{TP_{SW}}$ = TP concentration in SW water in μg L^{-1}.
   $Y_{\mathrm{Sal}} = Y4$ a dimensionless moderator for the influence of salinity on chlorophyll calculated from:
   $Y1$ = if Sal < 2.5 psu then (0.20–0.1·(Sal/2.5–1)) else (0.20+0.02·(Sal/2.5–1))
   $Y2$ = if Sal< 12.5 then $Y1$ else (0.28–0.1·(Sal/12.5–1))
   $Y3$ = if Sal > 40 then (0.06–0.1·(Sal/40–1)) else $Y2$
   $Y4$ = if Y3 < 0.012 then 0.012 else $Y3$.

3. Chl from TN. This approach is similar to the algorithm given in Eq. (2.9) but the basic relationship between Chl, TN, and salinity is the one given in Fig. 2.16.

Figure 2.21d compares the modeled values using the three approaches with the mean monthly empirical chlorophyll values from the Kattegat for the period 1995–2008. There is generally relatively good correspondence between the modeled values and the empirical data and in all following simulations, the regression based on both TP and TN will be used. It should be stressed that the empirical chlorophyll values are quite uncertain; the average monthly CV value is as high as 1.08, so all model predictions are well within ±1 standard deviation of the empirical mean values.

2.3.7 Predicting Water Clarity and Secchi Depth

The Secchi depth is an important variable since the water clarity defines the depth of the photic zone. In all the following calculations, the depth corresponding to two Secchi depths is used to define the entire depth of the photic zone (see Håkanson and Peters 1995). There exists a close relationship between SPM, Secchi depth, and salinity (see Håkanson 2006) – the higher

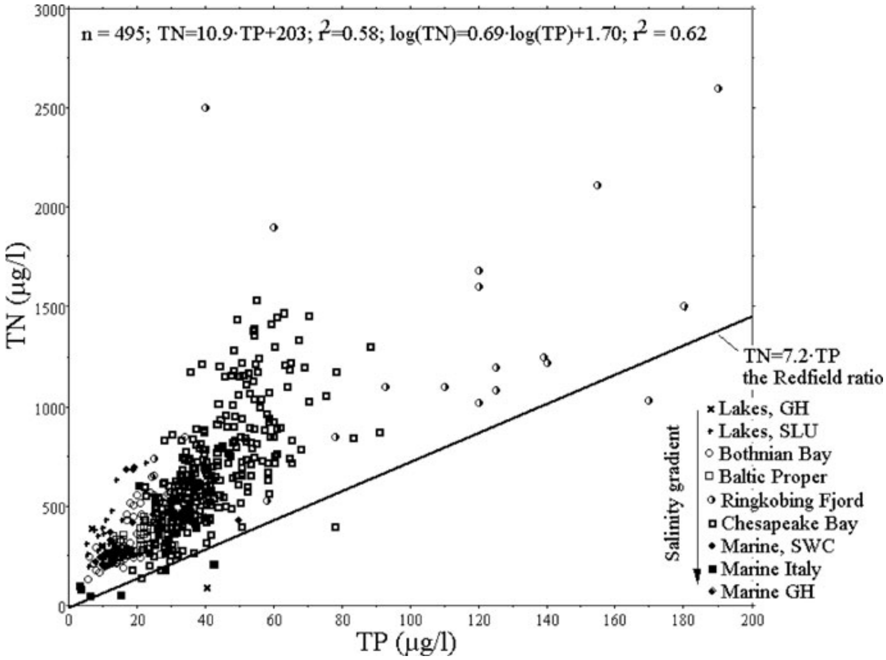


Fig. 2.27 Scatter plot between concentrations of total P (TP) and total N (TN) for the growing season from nine sub-groups constituting a salinity gradient. The figure also gives regressions for the actual data and log-transformed data for the 495 data points (from Håkanson and Bryhn 2008a)

the salinity, the higher the aggregation of suspended particles, the larger the particles, and the higher the water clarity. An SPM concentration of 10 mg L^{-1} would imply turbid conditions in a freshwater system, but relatively clearer water in a saline system. The relationship between Secchi depth (Sec in m), SPM_{SW} (mg L^{-1}), and salinity (Sal_{SW} in psu) is given by

$$\begin{aligned}\mathrm{Sec} = {} & 10\hat{}(-((10\hat{}(0.5\cdot\log(1+\mathrm{Sal}_{\mathrm{sw}})+0.3)-1)\\ & +0.5)\cdot(\log(\mathrm{SPM}_{\mathrm{sw}})+0.3)/2+(10\hat{}(0.15\cdot\\ & \log(1+\mathrm{Sal}_{\mathrm{sw}})+0.3)-1))\end{aligned} \qquad (2.10)$$

The SPM concentrations in the SW layer (SPM_{SW} in mg L^{-1}) are predicted from the dynamic SPM model. It should be noted that this approach is also used to predict SPM concentrations in the SW layer in the Skagerrak from empirical data on Secchi depth in Skagerrak (and from empirical salinities, as already explained). The results of these model predictions for the Secchi depth in the Kattegat are compared to measured data in Fig. 2.21c. The modeled values are close to the empirical values and within the uncertainty band given by ±1 standard deviation for the empirical data. These results give further empirical support to the general validity and predictive power of the CoastMab modeling.

2.3.8 Conclusions

To understand how the Kattegat system, or any aquatic system, responds to changes in, e.g., loading of toxins or nutrients, it is imperative to have a dynamic process-based perspective, quantifying the factors and functions regulating inflow, outflow, and internal transport processes and retention rates. This section has demonstrated that this modeling approach, using the theoretical wave base rather than traditional temperature and salinity data to define the surface-water and deep-water compartments, can give excellent correspondence between empirical and modeled data on the salinity. This section has presented budgets for water, salt, TP, TN, and SPM in the Kattegat. This process-based mass-balance modeling has used empirical data (from SMHI) for the period 1995–2008. An aim of the first part of this section was to present data on the fluxes of water and the theoretical retention times for water and salt in the defined sub-basins of the Kattegat since those values give fundamental information on how the system reacts to changes in, e.g., nutrient loading. This places certain demands on the structure of this model, which are different from oceanographic models, e.g., in quantifying resuspension, mixing, and diffusion and in the requirements regarding the accessibility of the necessary driving variables. This section has also discussed empirically based models, which have been added to the process-based dynamic CoastMab model. These are the sub models for Secchi depth and chlorophyll-a concentrations. When tested against empirical data for the Kattegat, there was good overall correspondence between predicted values for Secchi depth and chlorophyll-a concentrations and the dynamic SPM model predicts sedimentation, SPM concentrations, and burial in accordance with existing, but rather scattered, data.

2.4 Management Scenarios

This section will present several scenarios, which are meant to focus on key problems related to a sustainable management of the trophic state in the Kattegat. The same principles and questions discussed in this section should apply to most systems in contexts of remediation of eutrophication. The last scenario will put the results together and discuss an "optimal" management plan for the Kattegat related to realistic nutrient reductions to lower the eutrophication. The first scenario is logical in the sense that the main focus is on the largest nutrient flux to the surface water in the Kattegat. If very costly remedial actions reducing 10,000–100,000 t nutrients (P and N, respectively) annually to the Baltic Sea including the Kattegat are needed at a yearly cost in the range of 1,000–30,000 million euro year^{-1}, the model should be able to predict the expected changes in the surface-water layer (the bioproductive layer) not just for the nutrient concentrations but also for key bioindicators of eutrophication, such as the Secchi depth and the concentration of chlorophyll-a. So, scenario 1 is the first logical step in an attempt to find an "optimal" abatement plan to reduce eutrophication. Comprehensive analyses based on very large data sets on the conditions in the Kattegat have shown (in Section 2.2) that the anthropogenic nutrient emissions have not altered the eutrophication in the Kattegat markedly during the last 15–20 years. It is, however, well documented (see, e.g., a compilation of data and literature references in Håkanson and Bryhn 2008a, 2008c) that the eutrophication in the Baltic Sea increased significantly in the period between 1920 and 1980. The second and third scenarios will focus specifically on phosphorus and nitrogen reductions in the catchments of the rivers entering the Kattegat from Sweden. The Baltic Sea Action Plan (see Table 2.10), which the governments of the Baltic countries agreed upon in November 2007, implies that 15,000 t of phosphorus and 133,000 t of nitrogen of the total riverine nutrient fluxes to the entire Baltic Sea (including the Kattegat) should be reduced annually, including 290 t year^{-1} of phosphorus and 20,780 t year^{-1} of nitrogen from Sweden. The second and third scenarios will address how such reductions would likely influence the Kattegat. The fourth scenario will be based on the results from the first three scenarios and on the results presented in this work on the water fluxes, salt fluxes, nutrient fluxes, and fluxes of suspended particulate matter to, within, and from the Kattegat as well as the results related to how the two key bioindicators (Secchi depth and chlorophyll) would likely respond to changes in nutrient concentrations in surface water of the Kattegat. The basic idea is that this scenario should motivate an "optimal" remedial strategy to improve the eutrophication in the Kattegat. Nutrient reductions are ultimately related to political decisions. One can safely assume that it is practically impossible to remediate all human emissions of nutrients to the Baltic Sea. The 15,000 t year^{-1} suggested by HELCOM (2007b) represent a reduction of 50% of the 30,000 t year^{-1} of phosphorus transported via rivers/countries to the Baltic Sea. From countries that have already carried out costly measures to reduce nutrient discharges to the Baltic Sea, only a smaller part of the remaining anthropogenic nutrient fluxes can realistically and cost-effectively be reduced. The costs for nutrient reductions are essential to quantify for optimizing the cost-effectiveness of nutrient abatement strategies. Cost-effectiveness is not only a means for saving money, but also a means for increasing the chances that the selected strategy will be fully implemented. Less expensive measures are easier to undertake than expensive measures (Bryhn 2009). One point made in this section is that there are major differences in cost-effectiveness among the different options. Comparing cost-effectiveness between options is really important and the CoastMab model can be a useful complementary tool in such contexts to address the "benefit" side of the cost–benefit analysis.

Target variables which should be used for measuring benefits should not be the reductions in nutrient input from countries or tributaries related to a given remedial action, neither the reductions in nutrient concentrations in the Kattegat system, but rather the change in the target bioindicators in the system: How would a certain remedial strategy for reducing *X* tons of phosphorus for *Y* euro in river *Z* change the water clarity, the Secchi depth; reduce the risks of blooming of cyanobacteria (e.g., in the Baltic Proper); and reduce the maximum concentration of chlorophyll-a in the Baltic Proper and/or the Kattegat? To address such issues, one needs a validated, process-based mass-balance model. No such model is at present available for nitrogen, but the CoastMab model presented in this work may be used to address the target issues related to how the Kattegat would respond to changes in phosphorus input and also, with the given reservations, for nitrogen in the Kattegat and for the key bioindicators, and this will be demonstrated in this section.

2.4.1 Reductions in Tributary Phosphorus Loading to the Baltic Sea

This scenario is based on the following two key arguments:

- The focus is set on the dominating fluxes to the surface-water layer in the Kattegat. That is, on the nutrient fluxes from the Baltic Proper (see the annual budgets presented in Fig. 2.22 for phosphorus and in Fig. 2.24 for nitrogen). By far the most dominating nutrient loading to the bioproductive surface-water layer in the Kattegat comes from the Baltic Proper, which should be evident just by looking at the catchment area for the entire Baltic Sea, including the Baltic States, parts of Russia, Belarus, and Germany, Poland, Finland, and Sweden in relation to the relatively small catchment area draining directly into the Kattegat (from south-western Sweden and parts of Denmark).

  The focus will also be set on phosphorus and not on nitrogen because

- It is not possible to provide scientifically relevant predictions how the Baltic Sea system would respond to reductions in nitrogen loading since there are many major uncertainties related to the quantification of nitrogen fixation, wet and dry deposition of nitrogen, the algorithm regulating the particulate fraction for nitrogen and hence also sedimentation of particulate nitrogen and denitrification. For the Kattegat, on the other hand, atmospheric nitrogen fixation is neglected in this modeling because there are no significant amounts of N-fixing cyanobacteria in this system; the atmospheric deposition used in this modeling for the Kattegat comes from the OSPAR model (see Eilola and Sahlberg 2006, Håkansson 2007) and should be reliable in terms of order-of-magnitude values. Quantifying the denitrification is uncertain also in the Kattegat and it has been treated as a residual term in the mass balance for nitrogen so that the modeled concentrations in the surface-water layer, the deep-water layer, in the ET sediments, and the *A*-sediments should correspond to empirical data. No such calibrations have been done in the mass-balance calculations for phosphorus (i.e., the basic, validated CoastMab model is used directly without any tuning) or for the mass-balance calculations for SPM.
- In the Baltic Sea, and especially in the Baltic Proper, nitrogen reductions are likely to favor the blooming of harmful algae (cyanobacteria), and such events should be avoided. This means that reductions in tributary nitrogen loading to the Baltic Sea may, in fact, even increase the nitrogen concentration in the water (see Håkanson and Bryhn 2008a).
- So, there are no general, process-based mass-balance models for nitrogen, neither for the Baltic Sea basins, the Kattegat, or for any other coastal areas in the world, which have been tested (validated) for independent coastal systems and been demonstrated to yield good predictive power.
- In spite of the fact that costly measures have been implemented to reduce nitrogen transport from agriculture, urban areas (e.g., from water purification plants), and industries, the nitrogen concentrations in the surface water in the Kattegat have remained largely constant for the last 15–20 years.

So, the focus is set on the mass-balance modeling of phosphorus in scenario 1.

Figure 2.28 gives the results from three simulations:

- When half of the total phosphorus reductions have been carried out (i.e., a reduction of 7,500 t TP year^{-1}) for the tributaries to the Baltic Proper (as if 7,500 t TP year^{-1} was suddenly reduced from Polish rivers entering the Baltic Proper). Evidently, it is not realistic to implement such large and sudden reductions. These simulations illustrate the dynamic

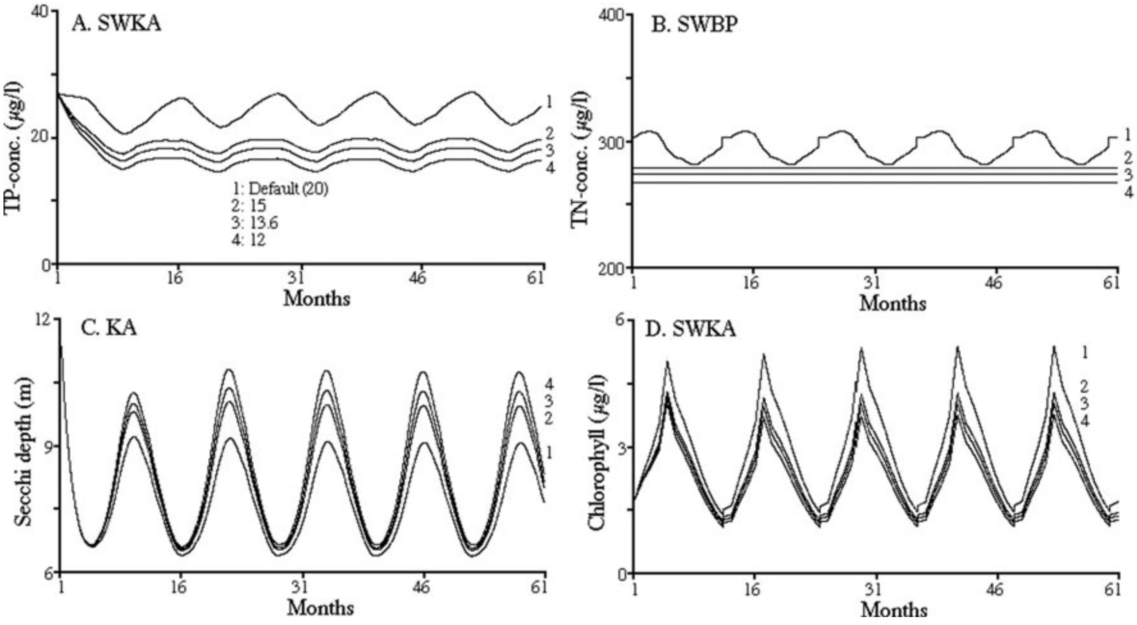


Fig. 2.28 Scenario 1 – changes in nutrient concentration in the Baltic Proper (BP). Curve 1 gives the default conditions, when the mean TP concentration in the Baltic Proper is 20 μg L^{-1}; curve 2 when the value is 15 μg L^{-1} (in SWBP) corresponding to a reduction in TP loading of 7,500 t $year^{-1}$ to the Baltic Proper; curve 3 when the value is 13.6 μg L^{-1} corresponding to a reduction in TP loading of 9,775 t $year^{-1}$ (the optimal scenario according to Håkanson and Bryhn 2008); curve 4 when the value is 12 μg L^{-1}. (**a**) Corresponding modeled TP concentrations in the surface water (SW) of the Kattegat (KA). (**b**) Corresponding TN concentrations in the surface water (SW) of Baltic Proper (BP). (**c**) Probable changes in Secchi depth in the Kattegat. (**d**) Corresponding likely changes in chlorophyll-a concentrations in the Kattegat

response of the Kattegat system to such a sudden P reduction into the Baltic Proper delivering its water to the Kattegat.

- When 9,775 t TP $year^{-1}$ from the tributaries entering the Baltic Sea have been (suddenly) reduced. Many tests have been presented by Håkanson and Bryhn (2008a) to try to find an optimal strategy for Baltic Sea management. Such a strategy should also concur with some evident practical constraints. For example, it may be very difficult and costly and maybe damaging for agriculture, urban development, and industry to reduce more than 60–70% of the anthropogenic point source and diffuse discharges of TP in Russia, Poland, and the Baltic states. There was also a focus on the conditions in the "hotspots," i.e., the Gulf of Finland, the Gulf of Riga, and the Baltic Proper, and not on smaller coastal areas and not on the oligotrophic basins (i.e., the Bothnian Bay and the Bothnian Sea). The total phosphorus reduction of 9,775 t $year^{-1}$ advocated in this management strategy was allocated accordingly: inputs to the Baltic Proper would be reduced by 6,625 t $year^{-1}$ (48% of anthropogenic emissions), in addition to reductions of 2,725 t $year^{-1}$ from the rivers entering the Gulf of Finland (corresponding to 60% of the anthropogenic input) and 425 t $year^{-1}$ of TP to the Gulf of Riga (or 46% of the anthropogenic input to this basin). Effective and cost-effective measures available to meet such reductions will be discussed in Section 2.4.5.
- This would give an average Secchi depth of 7 m in the Gulf of Finland and this is what the Secchi depth was in the Gulf of Finland before 1920. It would also give a Secchi depth of almost 10 m (9.7 m) in the Bothnian Sea, of about 8 m in the Bothnian Bay, 5.6 m in the Gulf of Riga, and almost 8 m (7.9 m) in the Baltic Proper.
- When 15,000 has been reduced according to the Baltic Sea Action Plan. One can estimate that this would create a mean annual TP concentration in the surface water of the Baltic Proper of about 12 μg L^{-1}, as compared to the default value today of about 20 μg L^{-1}. Case 1 (a reduction of 7,500 t TP $year^{-1}$) would give an annual mean TP concentration of 15.2 μg L^{-1}; case 2 (when 9,775 t

TP year^{-1} is being reduced as described) would give a mean annual value of 13.6 μg L^{-1} in the surface-water layer of the Baltic Proper (see Håkanson and Bryhn 2008a).

From Fig. 2.28, one can note that one should expect major reductions in the TP concentration (Fig. 2.28a) in the SW layer in the Kattegat if these remedial actions were carried out; see also Table 2.15, which gives the corresponding mean annual values for the Secchi depth and the chlorophyll, TP, and TN concentrations in the surface-water layer in the Kattegat. In these simulations, it is assumed that reductions in TP loading would also imply reductions in SPM loading. This may not be the case if the TP reductions would mainly relate to the building of water treatment plant, which could target specifically on phosphorus removal.

So, if that would be the case, the improvements in SPM concentrations and the related improvements in water clarity (Fig. 2.28b, c) would be smaller. This would also affect the predicted changes in chlorophyll-a concentrations, but to a lesser extent. So, the results would depend on the way in which the remedial actions are carried out and the results shown in Fig. 2.28 are meant to represent what one would "normally" expect. One can also note from Table 2.15 that the TN concentrations should increase slightly (from 281 to 290 μg L^{-1}) as a consequence of the reductions in SPM concentrations and the related increases in Secchi depths (from 6.5 to 8.4 m); the lower SPM concentrations would decrease the settling velocities for particulate nutrient forms (nitrogen and phosphorus). One can conclude from this scenario (and the following scenarios) that no other realistic actions will improve the eutrophication in the Kattegat more than reductions in phosphorus loading to the Baltic Sea. This is, in fact, evident from looking at the phosphorus fluxes (Fig. 2.22) into the surface-water layer in the Kattegat, since this action addresses the largest TP flux into the surface-water layer in the Kattegat.

2.4.2 Reductions in Tributary Phosphorus Loading to the Kattegat from Sweden

From Fig. 2.22, one can also see that the total Swedish contribution from diffuse sources corresponds to 500 t year^{-1} or 0.65% of the total TP inflow to the Kattegat; the TP contribution from Swedish point source emissions amounts to 180 t year^{-1}, or 0.23% of the total annual TP inflow to the Kattegat (76,900 t year^{-1}). So, what could one expect if half the Swedish BSAP quota of 145 t year^{-1} or if all of the Swedish quota (290 t year^{-1}) would be directed (rather unrealistically) to the catchment areas of the Swedish rivers entering the Kattegat. It is evident from Fig. 2.29 that this is not an effective strategy to improve the eutrophication in the Kattegat. It should be stressed that more or less the same results as shown in Fig. 2.29 would be obtained if 145 or 290 t phosphorus year^{-1} would be reduced from any inflow to the Kattegat system, whether this is from Sweden, Denmark, the Skagerrak, or the Baltic Proper.

2.4.3 Reductions in Tributary Nitrogen Loading to the Kattegat from Sweden

Figure 2.24 gives the annual budget for nitrogen and Fig. 2.30 three simulations in analogy with the results for phosphorus in Fig. 2.29. As an important background, one can note that the total contribution from Swedish diffuse sources corresponds to 29,100 t TN year^{-1} or 3.4% of the total nitrogen inflow to the Kattegat; the TN contribution from point sources amounts to 3,500 t year^{-1}, or 0.41% of the total annual TN inflow to the Kattegat (850,000 t year^{-1}). If half of the Swedish BSAP quota of 10,390 t year^{-1} or the entire Swedish quota (20,780 t year^{-1}) were (hypothetically) reduced from the tributaries or other inflows to the Kattegat, the environmental gain would be very small, as shown in Fig. 2.30a.

The improvements for the Secchi depth and for the phytoplankton biomass (the chlorophyll-a concentration) would also be very small indeed. This is also evident by looking at the nitrogen fluxes to the Kattegat in Fig. 2.24.

2.4.4 An "Optimal" Management to Reduce the Eutrophication in the Kattegat

How would a more "optimal" remedial scenario for the Kattegat look? Many alternatives have been tested

Table 2.15 Mean annual values for Secchi depth, chlorophyll-a, TP and TN concentrations in the surface-water layer in the Kattegat related to scenario 1

	Reductions (t phosphorus to the Baltic Sea)			
	Default	7,500	9,775	15,000
Secchi depth (m)	6.5	7.5	7.9	8.4
Chlorophyll (μg L^{-1})	2.7	2.3	2.2	2.0
TP (μg L^{-1})	21	17	15	14
TN (μg L^{-1})	281	286	288	290

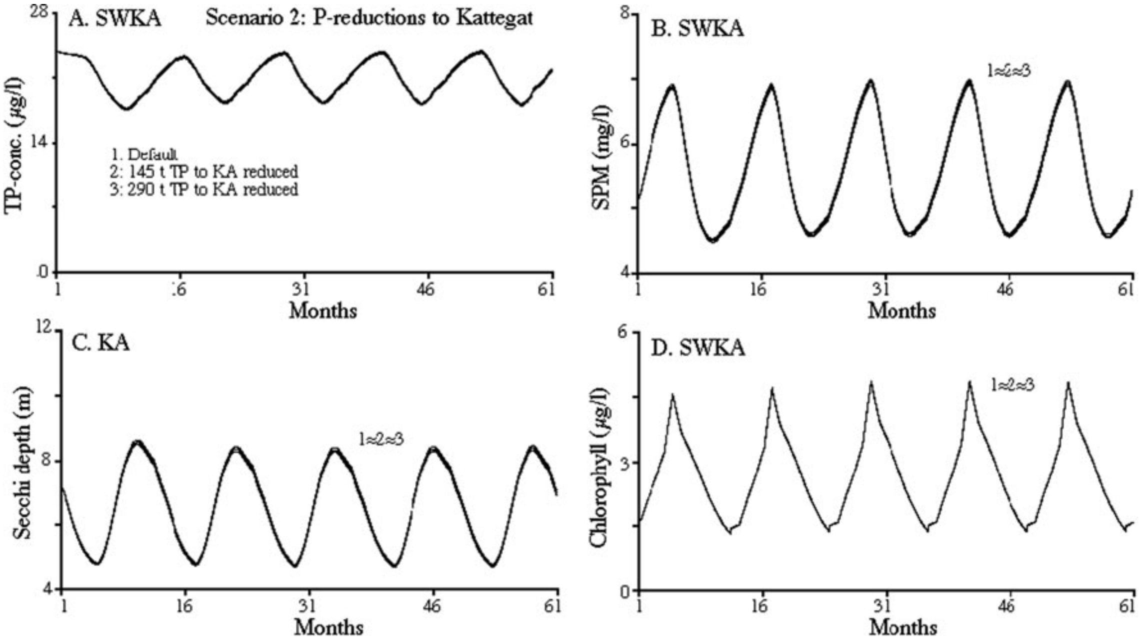


Fig. 2.29 Scenario 2 – curve 1 gives the default conditions; curve 2 the modeled response when 145 t year^{-1} (half the Swedish BSAP quota) of the tributary TP inflow to the Kattegat have been removed; and curve 3 the modeled response when 290 t year^{-1} of the tributary TP inflow to the Kattegat has been removed. (**a**) TP concentrations in the surface water (SW) of the Kattegat (KA). (**b**) The corresponding SPM concentrations in the surface water (SW) of the Kattegat. (**c**) Probable changes in Secchi depth in the Kattegat. (**d**) Corresponding likely changes in chlorophyll-a concentrations in the Kattegat

and it seems clear from the results already given that the first focus should be on phosphorus reductions in the rivers entering the Baltic Proper. The second focus could be on remedial actions for phosphorus that would also reduce the nitrogen transport to the Baltic Proper, although it is difficult to predict how such nitrogen reductions would actually change the nitrogen concentrations in the Baltic Proper. It is also, evidently, very important to seek remedial measures that would reduce phosphorus and nitrogen emissions in a cost-effective manner; the costs per removed kilogram nutrient may vary with a factor of 10–100 depending on the selected approach; and if the same approach is carried out in different Baltic Sea countries and whether the reduction concerns the “first kg” or the “last kg” in a long-term remedial strategy removing 10,000–100,000 t year^{-1}. It should also be stressed that nutrient reductions in the Baltic Proper would be beneficial for the entire Baltic Sea systems, where there are several “hotspots” (e.g., the Gulf of Finland, the Gulf of Riga, the area outside Kaliningrad, the Oder and Vistula estuaries) with significantly worse conditions than in the Kattegat system (see Figs. 2.3 and 2.4). Reductions in the “upstream” Baltic Sea system would also clearly benefit the Kattegat system. Figure 2.31 gives results from simulations when 9,775 t TP year^{-1} has been reduced (as described and motivated by Håkanson and Bryhn 2008a) and when also

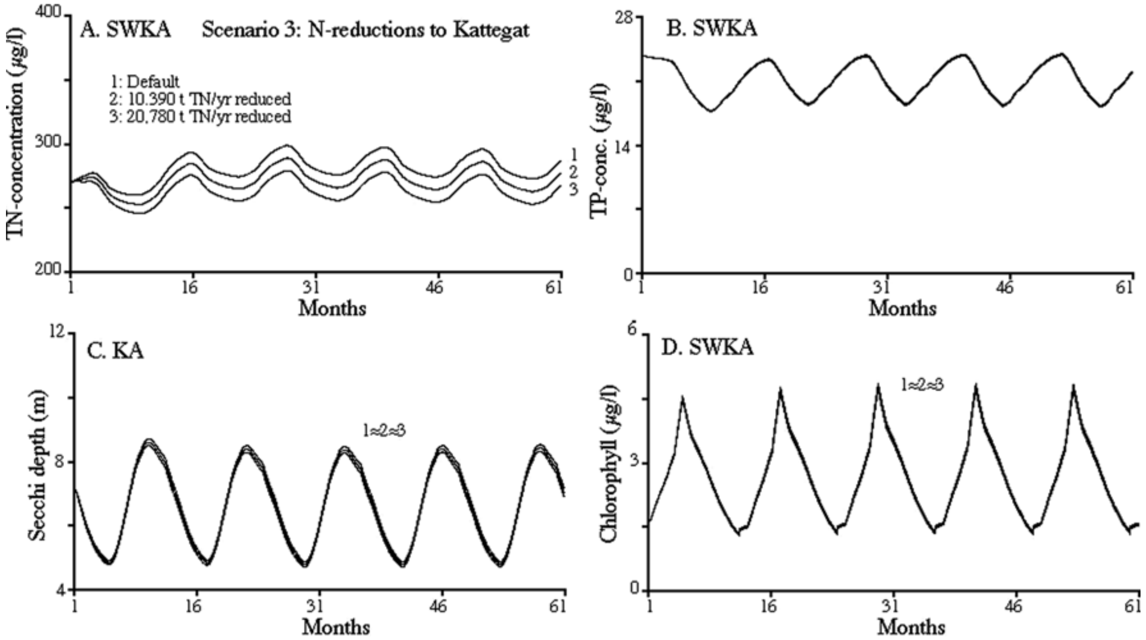


Fig. 2.30 Scenario 3 – curve 1 gives the default conditions; curve 2 the modeled response when 10,390 t year^{-1} (half the Swedish BSAP quota) of the tributary TN inflow to the Kattegat have been removed; and curve 3 the modeled response when 20,780 t year^{-1} of the tributary TN inflow to the Kattegat have been removed. (**a**) TN concentrations in the surface water (SW) of the Kattegat (KA). (**b**) The corresponding TP concentrations in the surface water (SW) of the Kattegat. (**c**) Probable changes in Secchi depth in the Kattegat. (**d**) Corresponding likely changes in chlorophyll-a concentrations in the Kattegat

the average nitrogen concentration in the Baltic Proper has been hypothetically lowered by 10% (from 281 µg L^{-1} on an annual basis to 253 g L^{-1}). This would significantly lower the TP concentrations in the SW layer in the Kattegat (Fig. 2.31a) and also reduce the TN concentrations in the SW layer in the Kattegat (Fig. 2.31b) and if those measures would be carried out in a manner that would also reduce SPM emissions to the Baltic Proper (in a "normal" way), then there would also be clear reductions in the SPM concentrations in the SW layer in the Kattegat and corresponding increases in water clarity and lower chlorophyll-a concentrations, as shown in Fig. 2.31.

It should be noted again that the modeled changes in TP concentrations are more reliable than the other changes shown in Fig. 2.31 and that the reductions in the TN concentrations in the Baltic Proper in this scenario are hypothetical. If the reductions in TN concentrations in the Baltic Sea would be even lower than 10% (which is suggested in the Baltic Sea Action Plan) this would create even smaller changes than the already small changes related to this scenario. "Optimal" in this scenario means that this is probably the best results one could realistically hope for.

2.4.5 *Effective and Cost-Effective Nutrient Reductions*

The "optimal" strategy advocated in Fig. 2.31 should appear more attractive when presented in combination with substantiated measures which could meet this strategy in an effective (decreasing the loading with a sufficient number of tons) and cost-effective (at the lowest possible cost) manner. An initial benchmark may be the Baltic Sea Action Plan, described in Section 2.3.3.1 and Table 2.10, whose full implementation would require a wide array of measures, including construction of wetlands, improved sewage treatment, and decreased agricultural production (Swedish 2008). The plan also includes measures for the Kattegat and the yearly cost of the plan has been estimated at 3.1 billion euro t year^{-1} (in 2008 prices; HELCOM and NEFCO 2007). According to calculations by the Swedish Department of Agriculture, N reductions, which Sweden has agreed to undertake in the Baltic Sea Action Plan, cannot be fulfilled unless a large part of the agricultural sector in the country would be permanently shut down, an option which would

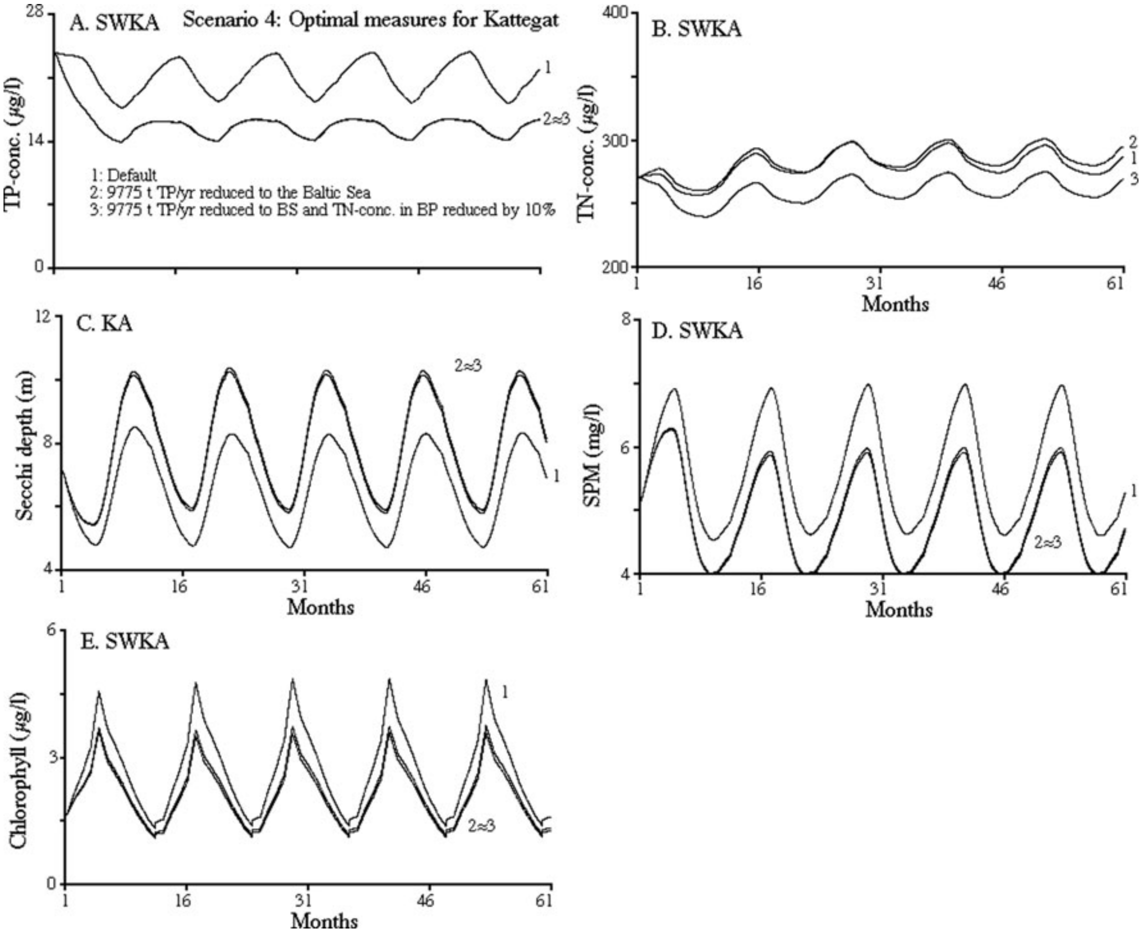


Fig. 2.31 Scenario 4 – the "optimal" management scenario. Curve 1 gives the default conditions, when the mean TP concentration in the surface-water layer in the Baltic Proper is 20 μg L^{-1}; curve 2 when the value is 13.6 μg L^{-1} corresponding to a reduction in TP loading of 9,775 t year^{-1}; curve 3 when also the TN concentration in the surface-water layer in the Baltic Proper has been reduced by 10% (from 281 to 253 μg L^{-1}). (**a**) TP concentrations in the surface water (SW) of the Kattegat (KA). (**b**) The corresponding TN concentrations in the surface water (SW) of the Kattegat. (**c**) Probable changes in Secchi depth in the Kattegat. (**d**) Connected changes in SPM in the surface-water layer in the Kattegat. (**e**) Corresponding likely changes in chlorophyll-a concentrations in the Kattegat

eliminate tens of thousands of jobs. Sweden, which is presently a net exporter of grain, could have to become a yearly net importer of millions of tons of grain (Swedish 2008), which would be associated with additional environmental pressure and transportation costs. Two particularly cost-effective measures for decreasing P inputs to the Baltic Sea are improved urban sewage treatment in former East Bloc countries and a ban on phosphates in detergents (Gren and Elofsson 2008, Bryhn 2009). Regarding the latter measure, however, attention must also be paid to the regional differences. On the one hand, in former East Bloc countries where urban sewage treatment is poor, a ban on phosphates would be very cost-effective, at least in the short run (Bryhn 2009). In Sweden, on the other hand, where sewage treatment has been implemented with relatively ambitious standards, marginal costs for P abatement are higher than those of many projects regarding urban sewage treatment in Poland, Russia, and the Baltic states, and a phosphates ban would probably have much lower cost-effectiveness in more countries if they would first upgrade their sewage treatment to Swedish standards (Bryhn 2009). It should also be noted that alternatives to phosphates in detergents may have their own adverse environmental effects. One of the most viable alternatives, Zeolite A, produces greater volumes of sludge which cannot be recycled in the same manner as phosphorus in sewage sludge can be used as a fertilizer in agriculture. Thus, with effective sewage treatment in place, phosphates may actually be the most environmentally friendly option in a life-cycle perspective (Köhler 2006), which would imply that many available marginal cost estimates for a phosphate ban may be greatly underestimated. Wetland construction and agricultural measures often have higher marginal P abatement costs than improvements in urban sewage treatment, and in some cases the cost difference may be a factor of 100 (Bryhn 2009). So, how much P can be removed by means of upgraded urban sewage treatment and how much would this cost? Helcom (2007a) estimated that advanced (tertiary) treatment was performed on sewage from 34% of the Estonian and Polish population, from 18% of Latvians and Lithuanians, and from 0% of Czechs, Russians, and Belarusians. Corresponding figures for Sweden, Finland, Germany, and Denmark were 86, 80, 85, and 81%, respectively. By upgrading urban sewage treatment in the former East Bloc countries, Helcom estimated that 12,400 t year^{-1} of phosphorus may be removed, which actually exceeds the TP abatement goal according to the "optimal" strategy motivated in Section 2.4.4. Thus, it appears to be possible to decrease the TP loading to the Baltic Sea with 10,000 t year^{-1} by means of upgrading urban sewage treatment. The cost is highly dependent on the available sewage pipe system in urban areas. According to Bryhn (2009), improved urban sewage treatment including the pipe system in former East Bloc countries had a typical marginal cost of 42 euro kg^{-1} P (2008 prices) while the typical marginal cost was only 20 euro kg^{-1} P when pipes were in an acceptable shape. This would mean that the TP abatement goal in Section 2.4.4 would cost 200–420 million euro year^{-1}, an estimate which corresponds to 6.5–14% of the cost of the eutrophication part of the Baltic Sea Action Plan. As previously stressed in this chapter, it is not yet possible to predict the extent to which TN loading reductions would be needed to decrease TN concentrations in the Baltic Proper and so the cost of unknown reductions in TN loading is therefore likewise difficult to estimate. However, since curves 2 (TP reductions) and 3 (TN+TP reductions) in Fig. 2.31c (Secchi depth) and Fig. 2.31e (chlorophyll) are quite close to each other, the cost-effectiveness of separate N treatment in addition to P treatment should be quite low in any case. Nevertheless, it is worth mentioning that decreased TP inputs to the Baltic Proper may also decrease nitrogen fixation in surface waters (Savchuk and Wulff 1999), and upgrading P treatment in sewage treatment plants may in addition retain some of the N in the sewage effluent.

2.4.6 Comments and Conclusions

In this section, the wisdom of the HELCOM strategy to reduce the eutrophication in the Baltic Sea (including the Kattegat) has been challenged. Nitrogen reductions may fail to give lower N concentration in the water because of compensatory increases in the nitrogen fixation by cyanobacteria, especially in the Baltic Proper. The results presented in this section indicate that a reduction of 15,000 t year^{-1} of phosphorus would likely create what may well be an undesired oligotrophication of the Baltic Sea system in the sense that the trophic status, as revealed by the operational bioindicators (Secchi depth and chlorophyll), would approach a lower level than Baltic Sea managers should realistically ask for. An alternative remedial strategy to reduce the eutrophication in the Kattegat based on the following cornerstones has been presented and motivated:

- Many remedial measures in agriculture, urban areas, or industry would remove both nutrients and when substance-specific methods are available, they should target on phosphorus removal; less substance-specific methods may reduce both phosphorus and nitrogen and if such remedial measures could be carried out in a cost-effective manner, it would be advantageous. The effects of nitrogen reductions cannot be predicted with any certainty in the Baltic Proper, but with some certainty in the Kattegat.
- A remedial strategy where 3,180 t year^{-1} of the phosphorus to the Gulf of Finland, 550 t year^{-1} to the Gulf of Riga, and 5,000 t year^{-1} to the Baltic Proper (and no reductions at all to the Bothnian Sea and the Bothnian Bay) has been motivated as the most effective approach to reduce also the eutrophication in the Kattegat system. Evidently, it would take a long time to implement such reductions in the Baltic Sea system (including the Kattegat). The Baltic Sea system could face several changes in

that time (e.g., related to climatic variations such as increased water temperatures and reductions in ice cover). This means that these recommendations should be taken with due reservations and that they should be adjusted to such possible future changes. The CoastMab model applied in this work could be a useful tool in such contexts.

2.5 Summary and Recommendations

To develop scientifically warranted programs of conservation, management, and remediation is a great challenge. In this situation, quantitative models are essential to predict, to guide assessment, and to direct intervention. The CoastMab model used in this work may be regarded as a tool for water management. It is also an approach to handle "trade-offs" and test working hypotheses concerning aquatic transport processes and interactions. The fact that the CoastMab model, in spite of its breadth and complexity, may be driven by relatively few readily accessible variables and that it is based on a general algorithm which may be repeated for different substances gives a certain robustness and attractiveness to the model and provides a framework for its practical usefulness and predictive power, which are essential components in models for aquatic management.

Section 2.2 gave basic information on the conditions in the case study area, the Kattegat, e.g., on the morphometry including the criteria to define the limit for the surface-water layer from the theoretical wave base. Section 2.3 presented the water fluxes to, within, and from the Kattegat system. These water fluxes are important for the quantification of all fluxes of salt, phosphorus, nitrogen, and SPM regulating all monthly concentrations. Section 2.3 also gave approaches to predict chlorophyll-a concentrations and Secchi depths from dynamically modeled values of phosphorus, nitrogen, SPM, and salinity and monthly light conditions.

These approaches are of fundamental importance in the Coast Web modeling because the food web model is driven by chlorophyll-a concentrations and the Secchi depth is a measure of the depth of the photic layer. The water fluxes determined from the CoastMab model for salinity are used throughout this modeling. It has been demonstrated that the CoastMab model for phosphorus, which prior to this work has been validated for many independent aquatic systems and been demonstrated to predict very well, also predicts TP concentrations in the Kattegat very well. It has been shown how the CoastMab model predicts TP and TN concentrations in water and sediments and also the target bioindicators. In fact, the inherent uncertainties in the available empirical data used to run and test the model for salt, phosphorus, SPM, and the two target bioindicators set the limit to the predictive power of the model.

It should, however, be noted that it is not possible to provide scientifically relevant predictions how the Baltic Sea system would respond to reductions in nitrogen loading since there are major uncertainties related to the quantification of nitrogen fixation, wet and dry deposition of nitrogen, the algorithm regulating the particulate fraction for nitrogen, and hence also sedimentation of particulate nitrogen and denitrification. For the Kattegat, on the other hand, atmospheric nitrogen fixation has been neglected in this modeling because there are no significant amounts of N-fixing cyanobacteria in this system; the atmospheric deposition used in this modeling for the Kattegat comes from the OSPAR model (SMHI) and should be reliable in terms of order-of-magnitude values; however, the denitrification is uncertain also in the Kattegat and it has been treated as a residual term in the mass balance for nitrogen so that the modeled concentrations in the surface-water layer, the deep-water layer, the ET sediments, and the *A*-sediments should correspond to empirical data. No such calibrations have been done in the mass-balance calculations for phosphorus (i.e., the basic, validated CoastMab model is used directly without any tuning) or for the mass-balance calculations for SPM. It is sub-optimal to give reduction quotas to different countries (such a strategy is based on political considerations rather than science). A more scientific strategy should be based on the identified "hotspots," and so the strategy should rather be to target on basins (generally estuaries) with a high degree of eutrophication and reduce nutrient input to such systems. From the maps given in Section 2.1, one can identify the Gulf of Riga, the Gulf of Finland, the Oder and Vistula estuaries, and the coastal area outside of Kaliningrad as hotspots. Because of major changes in population structure, agriculture, species composition, fishing/trawling, etc., it is not possible to carry out measures that would bring the Baltic Sea ecosystem including key structural and functional characteristics, functional groups, and species back to the conditions as they were, say 100 years ago, but it would be possible to reduce nutrient inputs so that the Secchi depth in the Gulf of Finland could return to about 7 m as it was between 1900 and 1920. To reach such a specific goal, there must also be major reductions not just in the rivers entering the Gulf of Finland, but also in the rivers entering the Baltic Proper, since the water and nutrient exchange between the Baltic Proper and the Gulf of Finland is intense (which can be seen from the salinity maps for the entire Baltic Sea including the Kattegat). In this work, a realistic remedial scenario has been presented that would considerably improve the conditions not just in the Kattegat but also in the Gulf of Riga and the Gulf of Finland as well as the Baltic Proper and the entire Baltic Sea. The default conditions using the CoastMab model have been described in detail for water fluxes, salinity, phosphorus, SPM, chlorophyll, Secchi depth and it has been demonstrated that the general approaches used here (without any tuning or calibrations for the Kattegat system) also generally showed good correspondence between modeled values and empirical data. The nitrogen modeling also showed good results, but the CoastMab model for nitrogen includes calibrations related to denitrification so the results related to the mass balance for nitrogen are not as reliable as the other predictions. Many tests have been carried out to find a strategy to reach the goal that the eutrophication in the Kattegat system could be reduced. By far the most dominating nutrient loading to the bioproductive surface-water layer in the Kattegat comes from the Baltic Proper, which should be evident just by looking at the catchment area for the entire Baltic Sea, including the Baltic States, parts of Russia, Belarus, Germany, Poland, Finland, and Sweden in relation to the relatively small catchment area draining directly into the Kattegat (from south-western Sweden and parts of Denmark). The final results are given on a monthly basis in Fig. 2.31. Evidently, it is not realistic to implement such major reductions in nutrient P loading suddenly, and these curves are meant to illustrate the relatively fast dynamic response of the Kattegat system in this hypothetical remediation scenario.

One can note from these tests, and also from Håkanson and Bryhn (2008a), that a reduction of 15,000 t year^{-1} of phosphorus to the Baltic Sea, as suggested by HELCOM (see Table 2.10) and agreed upon by the Baltic Sea states in November 2007, would likely increase the Secchi depth in the Gulf of Finland beyond the mean or median values around the year 1900. One hundred years ago, the nutrient loss from human activity was already substantial in the Baltic Sea catchment (Savchuk et al. 2008). Natural fertilizers were used in agriculture, and horses were intensively used for transportation in urban and rural areas. Sewage systems were constructed to prevent outbreaks of cholera and other diseases in the cities but sewage treatment was absent or very ineffective in many areas until after the Second World War. This indicates that a reduction by 15,000 t year^{-1} is likely "overkill." A lowering of the primary production in the Baltic Sea and the Kattegat will imply also a reduction in the secondary production, including zooplankton and fish; it would increase the acidification (since this is related to the primary production); it would also increase the concentration of organic toxins in fish – "in the clearest waters swim the most toxic fish." This is a well-established fact called biological dilution (see Håkanson 1999, 2000). It relates to the definition of the average concentration of toxins in fish, $C = M/\mathrm{BM}$, where M is the total mass of a given toxin in fish (in g; e.g., total PCB, total dioxins, methyl mercury) and BM is the total biomass of the fish (e.g., prey or predatory fish, or a given species of fish, such as cod; in kg). If BM decreases as it does in this oligotrophication scenario, C should increase if there are no simultaneous reductions in the loading of toxins to the system. There is evidently no point to lower the trophic status of the Baltic Sea or the Kattegat system to levels where the environmental drawbacks become larger than the benefits, and every action could potentially include benefits as well as drawbacks.

The strategy that one should ask for should also concur with some evident practical constraints. For example, it is not really realistic to reduce all anthropogenic TP or TN discharges. And for countries where major investments in nutrient reductions have already been made, it will become increasingly expensive to reduce the remaining tons. So, by a search for an optimal strategy, one could, for example, limit TP reductions to 60–70% of the anthropogenic emissions in coastal systems where few costly remedial actions have been implemented, and to much less in countries such as Sweden, Finland, and Germany. So, the wisdom of the HELCOM strategy to reduce eutrophication in the Baltic Sea may be challenged.

It should also be stressed that given the conditions in the Baltic Proper, nitrogen reductions may fail to give lower N/P ratios in the water because of compensatory increases in the nitrogen fixation by cyanobacteria (see Håkanson and Bryhn 2008a, 2008c). If nitrogen reductions lower the N/P ratios in the surface water, this could increase the competitiveness of cyanobacteria in relation to other algae even more, which is a clearly negative consequence of an expensive remedial strategy implemented to improve rather than worsen the conditions in the Baltic Sea. Conversely, P reductions may increase N/P ratios, thereby decreasing both the competitiveness of cyanobacteria and the fixation of atmospheric nitrogen (Savchuk and Wulff 1999, Tyrrell 1999).

In the "optimal" scenario, about 10,000 t year^{-1} of phosphorus is being reduced and also nitrogen reductions that would lower the TN concentration in the Baltic Proper by 10%. The costs for this would likely be about 200–420 million euro t year^{-1} if this is done in a cost-effective manner, which means a focus on improved phosphorus removal in urban sewage which is discharged into the most polluted estuaries and coastal areas. The costs to reduce 15,016 t year^{-1} of TP and 133,170 t year^{-1} of nitrogen according to the HELCOM strategy would be 3,100 million euro year^{-1}. That is, 2,680–2,900 million euro year^{-1} higher than the "optimal" strategy discussed in this work.

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Chapter 3

Eutrophication Processes in Arid Climates

Elias Salameh and Sura Harahsheh

Abstract Eutrophication is the process by which a water body becomes increasingly rich in aquatic plant life such as algae and aquatic macrophytes (water weeds). Eutrophication is characterized by the presence of sufficient plankton, algae, and water weeds, which cause water quality impairments for domestic water supply such as tastes and odors, shortened filter runs, THM precursor's formation. Lakes are often classified according to their trophy or degree of enrichment with nutrients and organic matter. They are classified by their trophic state with the main classes of oligotrophic, mesotrophic, eutrophic, and dystrophic. The main cause of eutrophication is the large input of nutrients to a water body and the main effect is the imbalance in the food web that results in high levels of phytoplankton microalgae, with a silicon skeleton (diatom) biomass in stratified water bodies which can lead to algal blooms. King Abdullah Canal is the main surface water source of potable and agricultural water supply in Jordan. Two main problems seem to face the use of KAC for domestic and agricultural purposes. These problems are eutrophication and formation of chlorinated hydrocarbons during water treatment. Both problems develop as direct or indirect results of anthropogenic activities.

Keywords Eutrophication · King Abdullah Canal · Natural Organic Matter · Nutrients · Trihalomethanes

3.1 Introduction

Two main problems seem to face the use of surface water for domestic purposes in semi-arid regions. These problems are eutrophication of surface water bodies and the formation of chlorinated hydrocarbons during water treatment. Both problems develop as direct or indirect result of anthropogenic activities (Salameh 1987). Nutrients, such as nitrogen, phosphorus, and silicate, in lakes, reservoirs and some streams, rivers, and near-shore marine water are prerequisites for life and do not form an environmental problem. Nutrients are not pollutants but life givers. They become a problem when too large inputs affect the original character, properties, or functions of the ecosystem (Finnish Institute of Marine Research 2002). When this occurs, it is referred to as "eutrophication of water bodies." Eutrophication is characterized by the presence of sufficient plankton, algae, and water weeds, which cause water quality impairments for domestic water supply such as tastes and odors, shortened filter runs, THM precursor's formation.

3.1.1 Eutrophication Process

The common link of algal blooms, fish kills, inedible shellfish, blue algae, and public health is water eutrophication (WHO 2002). Eutrophication is the process by which a water body becomes increasingly rich in aquatic plant life such as algae and aquatic macrophytes (water weeds). It is driven by increasing input of aquatic plant nutrients, especially compounds of nitrogen and phosphorus, from natural

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and anthropogenic sources that induce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water. Natural eutrophication processes take place over geological time and they mean aging or dying of water bodies.

3.1.1.1 Natural Eutrophication

The runoff water brings silt, rich in organic nutrients and minerals, into the newly formed lakes. Wind-borne bacterial, protozoan, and algal spores soon establish population and form the base of a food pyramid which, in time, builds to support larger plant and animal population (Eutrophication 2005a). Decay by bacteria breaks down the remains of dead plants and animals at the bottom of lakes. In addition, sediments of surface water runoff from the surrounding land continue to supply organic nutrients and minerals to the lake ecosystem. Algae and bacteria grow in the nutrient-rich water. As long as there is adequate dissolved oxygen in the water the decomposition remains primarily aerobic. However, with the continued influx of silt and the build up of organic matter, the aerobic decay bacteria use the oxygen and a different kind of bacteria begin to dominate the decay process. These are the anaerobic bacteria whose products of decay are methane and hydrogen sulfide gases. Natural eutrophication is the adding of nutrients to an ecosystem, resulting in a decrease of available oxygen in the water. It is the force that drives the succession from lake to marsh to meadow to forest. This natural succession process may take place anywhere in a period of 500–10,000 or more years depending on the initial condition of the area (Eutrophication 2005a).

3.1.1.2 Eutrophication by Human Activities

Human activities can accelerate the rate at which nutrients enter ecosystems. Runoff from agriculture, urban, and industrial development, especially from septic systems and sewers and other human-related activities increases the flux of both inorganic nutrients and organic substances into terrestrial, aquatic, and coastal marine ecosystems (Eutrophication 2005a).

3.1.2 Eutrophication Classification

Lakes are often classified according to their trophy or degree of enrichment with nutrients and organic matter. They are classified by their trophic state with the main classes of oligotrophic, mesotrophic, eutrophic, and dystrophic.

3.1.2.1 Oligotrophic

Oligotrophic lakes are poorly supplied with plant nutrients and support little plant growth. As a result, biological productivity is generally low. The waters are clear, and the deepest layers are well supplied with oxygen throughout the year.

3.1.2.2 Mesotrophic

Mesotrophic lakes are intermediate in characteristics. They are moderately well supplied with plant nutrients and support moderate plant growth.

3.1.2.3 Eutrophic

Eutrophic lakes are richly supplied with plant nutrients and support heavy plant growths. As a result biological productivity is generally high. The waters are turbid because of dense growths of phytoplankton or they contain an abundance of rooted aquatic plants; deepest waters exhibit reduced concentrations of dissolved oxygen during periods of restricted circulation (Lee 2005).

3.1.2.4 Dystrophic

In the dystrophic stage the water is highly polluted and no desirable species can be supported due to lack of oxygen and presence of toxins (Salameh 1987).

3.1.3 Causes of Eutrophication and Supporting Factors

More plant-available nutrients in the water imply increased algal growth (primary production). In the photosynthesis process, green plants (including algae of all sizes and lifestyles) need chlorophyll, sunlight, carbon dioxide, and nutrients in order to produce oxygen and biomass. The most important nutrients are nitrogen, phosphorus, and silicate, but micro-nutrients like potassium, sulfur, iron, and molybdenum are also needed (Finnish Institute of Marine Research 2002). The deeper down the sunlight can penetrate the water, the deeper algae can grow. Green plants need chlorophyll to bind energy from the sunlight. By measuring the concentration of chlorophyll in water the quantity of microscopic algae can be determined in the water in winter, before the spring bloom (Finnish Institute of Marine Research 2002).

3.1.3.1 Nutrients

The main causes of eutrophication is the large input of nutrients to a water body, and the main effect is the imbalance in the food web that results in high levels of phytoplankton microalgae, with a silicon skeleton (diatom) biomass in stratified water bodies which can lead to algal blooms. In addition to carbon, oxygen, and hydrogen that plants can find directly in the water and carbon dioxide in the atmosphere, two major nutrients are necessary for the development of aquatic life, namely nitrogen (N) and phosphorus (P). A third one, namely silicate is necessary for the development of diatoms. During eutrophication, the concentrations of nutrients in the water change. In some cases one out of the three nutrients may be totally bound to the aquatic life and will not be available for further growth of algae (WHO 2002). This nutrient is then called the limiting factor. P is more often a limiting nutrient for phytoplankton in freshwater bodies, while N is often the limiting nutrients in marine water especially in summer. Intermediate areas such as river plumes are often P limited during spring, but may turn to silicate or N limited in summer. When P is the limiting factor, a phosphate concentration of 0.01 mg L^{-1} is enough to support plankton and concentrations of 0.03–0.1 mg L^{-1} or higher will likely promote blooms. Growth rate-limiting concentration for phosphorus is on the order of 2–8 $\mu g\ L^{-1}$ available P and for nitrogen 15–20 $\mu g\ L^{-1}$ available N. It is important to recognize, however, that even growth rate-limiting concentrations can support appreciable algal biomass if there is sufficient time for algal growth to occur (Lee and Lee 2005). It is difficult to say that N truly limits growth because N-fixing cyanobacteria (blue-green algae) can proliferate, creating nuisance condition in N-depleted environments.

In coastal areas, the growth and proliferation of diatoms is promoted by the presence of silica; when the silica concentration is low diatoms cannot develop. The ratio of N to P compound in a water body is an important factor determining which of the two elements will be the limiting factor and, consequently, which one has to be controlled in order to reduce a bloom. Table 3.1 shows the N/P ratio to serve as limiting factors.

To determine which nutrient is the limiting factor for algal growth in a particular water body, some have relied on the comparison of the concentration of nitrogen and phosphorus to the stoichiometric ratio of these elements in algae (16:1 atomic basis) (Eq. 3.1) (Lee and Lee 2005).

$$106\,CO_2 + 16\ N + 1\ P + \text{Trace element} = \text{Algae} + O_2 \tag{3.1}$$

3.1.3.2 Availability of Nutrients

N and P exist in aquatic systems in many different forms, only some of which can be used by algae and aquatic plants. Therefore, in assessing the limiting nutrient in a water body or evaluating the control of nutrient input to a water body, it is essential to consider the forms in which N and P exist in the loading sources and water body. Algal available forms of nitrogen are nitrate (NO_3^-), nitrite (NO_2^-), ammonia (NH_3), and some of the organic nitrogen. The organic nitrogen availability depends on its source and age, and some blue-green algae can fix atmospheric nitrogen gas (N_2) dissolved in water and use it as a source of nitrogen for growth.

Table 3.1 Nitrogen/phosphorus ratio (WHO 2002)

	N-limiting ratio, N/P	Intermediate ratio, N/P	P-limiting ratio, N/P
Freshwater	≤4.5	4.5–6	≥6
Estuarine coastal water	≤5	5–10	≥10

Total P (TP) includes some or all the following fractions: crystalline, occluded, absorbed, particulate organic, soluble organic, and soluble inorganic P. Out of these fractions the two biological available P fractions listed in order of decreasing availability are soluble reactive phosphorus (a mixture of dissolved inorganic and organic species, soluble uncreative P, some include dissolved P, fed by per sulfate oxidation, and available for phytoplankton by enzymatic hybridization which frees organically bound fraction) and labile phosphorus association with soil particles (Lee 2005). Generally, soluble orthophosphate (PO_4^{3-}) is the form of phosphorus available to support algal growth. Most particulate P and organophosphorus compound, and oxygen–P polymer chain and ring compounds do not support algal growth (Lee and Lee 2005).

3.1.3.3 Factors Supporting the Development of Eutrophication

Besides nutrient inputs, some physical conditions support eutrophication development. Thermal stratification of water bodies (such as lakes and reservoirs), temperature, and light influence the development of aquatic algae. Increased light and temperature condition during springs and summer explain why eutrophication is a phenomenon that occurs mainly during these seasons (WHO 2002). Eutrophication itself affects the penetration of light through the water body because of the shadow effect coming from algae and other living organisms and this reduces photosynthesis in deep water layers.

3.1.3.4 Sources of Nutrients

In order to gauge how to best prevent eutrophication from occurring, specific sources that contribute to nutrient loading must be identified. There are two common sources of nutrients and organic matter: point and non-point sources.

Point Sources

Point sources are directly attributable to one influence. In point sources the nutrient waste travels directly from source to water or it is any discernible, confined, and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock concentrated animal feeding operation or vessel or other flouting craft, from which pollutants are or may be discharged (Ongley 1996). Examples are factories which have a waste discharge pipe directly leading into a water body. Point sources are relatively easy to regulate. The following are point sources:

- Wastewater effluent (municipal and industrial)
- Runoff and leachate from waste disposal system
- Runoff and infiltration from animal feed lots
- Runoff from mines, oil fields, unsewered industrial sites
- Overflows of combined storm and sanitary sewers

Non-point Sources

Non-point source water pollution occurs when “diffuse or runoff” source pollution arises from a broad group of human activities for which the pollutants have no obvious point of entry into receiving water courses (Ongley 1996). Non-point sources are difficult to regulate and usually vary spatially and temporally with season, precipitation, and other irregular events. The following are non-point sources:

- Runoff from agriculture/irrigation
- Runoff from pasture and range land
- Urban runoff from unsewered areas
- Septic tank leachate
- Runoff from construction sites of more than 20,000 m^2 construction area
- Runoff from abandoned mines
- Atmospheric deposition over a water surface
- Other land activities generating contaminants (Eutrophication 2005b)

3.1.4 Effects of Eutrophication

The effects of eutrophication on the environment may have deleterious consequences on the health of exposed animal and human population through various pathways. Specific health risks appear when freshwater extracted from eutrophic areas is used for the production of drinking water. Severe impact can also occur during animal watering from eutrophic waters (WHO 2002). The following are the symptoms and impacts of eutrophication.

- Increase in production and biomass of phytoplankton, attached algae, and macrophytes
- Shift in habitat characteristic due to changes in assemblage of aquatic plants
- Replacement of desirable fish by less desirable species: Eutrophication has been shown to cause competitive release by making abundant or otherwise limiting nutrients; this causes shifts in the composition of ecosystems. For instance, an increase in nitrogen might allow new, more competitive species to invade and compete with original species (Eutrophication 2005b).
- Production of toxins by certain algae: Some algal blooms especially blue-green algae are toxic to plants and animals. This toxicity can lead to decreased biodiversity or it can manifest itself in primary products making its way up the food chain, and marine animal mortality has been observed (Anderson 1994) and may pose a threat to human.
- Increasing operating expenses of public water supplies, including taste and odor problems, especially during periods of algal blooms: When raw water supplies contain large amounts of algae and some other aquatic plants, the cost of treatment increases and the quality of the product may decrease. Planktonic algae can shorten filter runs. They can also release organic compounds that cause tastes and odors and, in some instances, serve as trihalomethanes (THMs) and halo acetic acid (HAA) precursors (Lee and Lee 2005). The compounds react with chlorine during the disinfections process and are considered as human carcinogens.
- Deoxygenating of water, especially after collapse of algal blooms, usually resulting in fish kills: When a body of water experiences an increase in nutrients, primary producers reap the benefit first. This means that species such as algae experience a massive population bloom. The increase in algae bloom would increase the amount of oxygen present in the water because oxygen is a product of photosynthesis. Under eutrophic condition, dissolved oxygen is reduced by the dense population. But too much algae block sunlight from reaching deep in the lake. These algae die and become food for the bacteria which use up the oxygen while eating the dead algae. When dissolved oxygen levels decline, especially at night, when there is no photosynthesis, hypoxia occurs and fish or other marine animals may suffocate.
- Infilling and clogging of irrigation canals with aquatic weeds (water hyacinth problem of introduction, not necessarily of eutrophication).
- Loss of recreational use of water: Excessive growth of attached algae and aquatic macrophytes can impair swimming, boating, and fishing by interfering with water contact; severe odor problems can also be caused by decaying algae, water weeds, and algal scum (Lee and Lee 2005).
- Violations of water quality standards: During daylight, algal photosynthesis removes CO_2 from water which increases the pH, algal respiration in the night releases CO_2 and lowers the pH. In late afternoons the pH of excessively fertile water can be found to exceed the water quality standard for pH. Algae produce oxygen during photosynthesis, but they consume it during respiration. Also due to bacterial and other organism respiration, dissolved oxygen concentrations can be below water quality standards for the protection of fish and other aquatic life. Excessively fertile water bodies that thermally stratify often exhibit dissolved oxygen depletion below the thermocline due to bacterial respiration and consumption by dead algae. Richards (1965) showed that one phosphorus atom, when converted to algae, which subsequently dies, can consume 276 oxygen atoms as a part of the decay process (Lee and Lee 2005). Although oxygen depletion in lakes leads to the death of fish and benthic organisms, the production of undesired chemical species (NH_3, H_2S, CH_4) accelerates cycling of pollutants from sediments, especially P. Oxygen depletion is one of the most important and commonly observed water quality problems in lakes (CE4505 Surface Water Quality Engineering 2006).
- Water clarity (water transparency): Water clarity is defined by the depth of the water body at which the bottom sediments can be seen from the surface. Water bodies with high degrees of clarity (the bottom can be seen at depths of 20 or more feet) have low planktonic algal content. In more eutrophic water bodies, the sediments can only be seen at a depth of a few feet. The greenness of water, inorganic turbidity, and high level of planktonic algal chlorophyll diminish water clarity.

- Impediments to navigation due to dense weed growth.
- Economic loss due to change in fish species, fish kills, and shellfish.

3.1.5 *Trihalomethanes*

3.1.5.1 Disinfection

Disinfection of drinking water is defined as a treatment process for the purpose of the destruction or inactivation of human pathogens, up to a given level of safety that should be maintained throughout water storage and distribution. The process depends on the type and concentration of the microorganisms, and the physical and chemical properties of the source water (Department of Health and Community Services 2001). Chlorination disinfection by-products (DBPs) are chemical compounds that form when water containing natural organic matter (NOM) reacts with chlorine during water treatment. There is a variety of disinfection methods utilized worldwide for the treatment of water. Some of the main disinfection techniques are listed in Table 3.2 (Cornell 1996).

The most commonly used disinfectants are

- Chlorine
- Chloramine
- Chlorine dioxide
- Ozone
- Ultraviolet radiation
- Mixed oxidants
- Iodine

Why Is Chlorination the Most Widely Used Method of Disinfection?

- It is effective against a broad range of pathogens including bacteria, viruses, and protozoa.
- It provides residual protection by preventing microbial re-growth after the treated water enters the distribution system.
- 3-Chlorination of drinking water reduces tastes and odors.
- It eliminates slime bacteria, molds, and algae. Chlorine controls these nuisance organisms, which typically can grow in reservoirs, on the walls of transmission water mains and in storage tanks.
- Chlorine in water treatment disintegrates hydrogen sulfide and removes ammonia and other nitrogenous compounds that have unpleasant tastes and hinder disinfection (Chlorine Chemistry Council 1997)
- The technology associated with chlorine disinfection is simpler and cheaper than other disinfection technologies and can be utilized in treatment plants of all sizes.

Chlorine can be administered to a water system in both gaseous and liquid forms such as gas (Cl_2), liquid (sodium hypochlorite), and solid/water such as calcium hypochlorite. All forms of chlorine invariably react with the water to form hypochlorous acid which in turn dissociate in the water to give hypochlorite ion and hydrogen ion depending on pH and temperature (Eq. 3.2)

$$\begin{gathered} Cl_2 + H_2O \rightarrow HOCl + HCl \\ HOCl \rightarrow H^+ + OCl^- \end{gathered} \tag{3.2}$$

The order in which the chlorine will be used is as follows:

Stage 1: The hypochlorous acid will first react with dissolved ions, hydrogen sulfide, and other inorganic materials.

Stage 2: After reaction with the impurities in Stage 1, the next set of reactions will be with reducing compound and organic material.

Stage 3: After the Stage 2 reaction, the next set of reactions will be with background ammonia levels in the water. The exact reactions depend on the pH of the

Table 3.2 Disinfection techniques

Disinfection methods	Example
Physical	Heat, storage
Light	Ultraviolet radiation
Metals	Silver
pH	Acids, alkalis
Oxidants	Chlorine, chlorine dioxide, ozone, iodine, and chloramines
Others	Surface active agents

water. For water supplies with natural ammonia levels, chloramines will form.

Stage 4: As more chlorine is added, the chloramines that were formed in Stage 3 are destroyed. After that, any more chlorine added to the water remains as hypochlorous acid or the hypochlorite ion (free chlorine residual). The chemical products of Stages 3 and 4 will result in working as disinfectant, while products from Stages 1 and 2 have no real disinfecting power (Department of Health and Community Services 2001).

3.1.5.2 Natural Organic Matter (NOM)

The complex matrix of organic material (humic acid and fulvic acid) is present in all natural surface waters (CWRS Workshop Notes and Center for Water Resources Studies 1995). NOM results from the decomposition of organism remains from the environment surrounding the watershed such as leaves, aquatic plants, and human and animal wastes. Water quality parameters such as water color and total organic carbon (TOC) are considered as good indicators of the presence of NOM in natural water. TOC is comprised of dissolved organic carbon (DOC) and particulate organic carbon (POC), of which DOC makes up approximately 99% of TOC (CWRS Workshop Notes and Center for Water Resources Studies 1995).

It has been determined that increased levels of water color and TOC or DOC indicate that there is a significant amount of NOM present. The disinfection by-products (DBPs) formation potential is directly proportional to the disinfectant type and amount used to disinfect the water. However, in addition to the disinfectant and NOM, other parameters such as pH, water temperature, water turbidity, disinfectant dose, chlorine demand, sunlight, and contact time (i.e., the time that the organisms are in contact with the disinfectant) also affect the formation of DBPs (Steven et al. 1989). Many of these factors may be working at the same time and may be working against each other.

There are a variety of DBPs, such as trihalomethanes (THMs), haloacetic acids (HAAs), and haloacetonitriles formed from all methods of disinfection by both halogen substitution and/or oxidation reactions (Singer 1999). The DBPs result from the most common methods of disinfection including chlorination, chloramination, ozonation, and disinfection with chlorine gas (Singer 1999). Among all of these DBPs, trihalomethane has been the most controversial issue of DBPs with a significant amount of available information.

3.1.5.3 Trihalomethanes

In the early 1970s, Rook (1974) and Bellar et al. 1974) discovered that the reaction of chlorine with dissolved organic materials in water formed a class of chlorination disinfection by-products (DBPs) called trihalomethanes (THMs) (Cornell 1996). The by-products formed during the chlorination of natural waters can be divided into two classes of compounds. The first class consists of the trihalomethane (THM) compounds, which are chlorinated and brominated derivatives of methane. The four compounds commonly included in this class are chloroform ($CHCl_3$), dichlorobromomethane ($CHCl_2Br$), chlorodibromomethane ($CHClBr_2$), and bromoform ($CHBr_3$) (Rathbum 1995). The chemical structure of each of these forms of THM is shown in Fig. 3.1. Among the four THMs, chloroform is the most common and detected in the greatest concentration.

The second class consists of a complex mixture of relatively nonvolatile compounds that have higher molecular weights than THM compounds. Because of the large number of different compounds present in this mixture at small concentration, identification and quantification of the individual compounds are difficult. Consequently, the concentration of these compounds is determined as a bulk parameter called the non-purgeable total organic halide (NPTOX) concentration.

Trihalomethanes are the most common DBPs in chlorinated drinking water (Singer 1999). THMs are

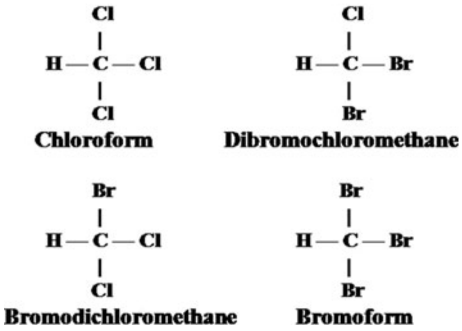

Fig. 3.1 The chemical structure of THMs

considered to be an indicator of the possible presence of other DBPs. They are known to be carcinogenic in laboratory animals and are probably carcinogenic to humans (Health Canada 1996). New epidemiological (human) studies had been published which reported associations between THMs and bladder and colon cancer and adverse pregnancy outcomes including miscarriage birth defects and low birth weight (Government of Newfoundland and Labrador 2000). Also positive correlations were observed for brain cancer in both males and females and for lymphoma and kidney cancer in males (Rathbum 1995).

3.1.5.4 THM Formation Potential

There are numerous factors that affect the formation of THMs in public water supplies. Both natural and human factors influence the quality of a water source and consequently the THM formation potential.

Natural Factors

The natural characteristics of a surface water source and characteristics of its watershed surrounding area influence water quality, including the potential formation of THMs. Among the natural factors hydrology, topography, geology, soil, vegetation, and climate are considered as key parameters to influence the water quality (Department of Health and Community Services 2001).

Human Factors

Human factors affecting water quality are generally associated with the type and the level of development activities being undertaken within the catchments of a water body. The presence of human activity in a public water supply area is considered as a potential risk to water quality impairment. In addition, human activity may increase the THM formation potential in water supply, due to disturbance to the natural watershed setting and surface runoff pattern. For this reason, it is important to consider the level of human activities within water sources catchments used for public supply (Table 3.3; Department of Health and Community Services 2001). Table 3.4 presents general land use and potential pollutant analysis matrix which provides additional details on different types of pollutants associated with each land use activity. Overall, both natural and human factors influence the quality of the source water.

Levels of THM are generally highest in treated water from source with high organic matter content, such as rivers and lakes. Lower levels of THMs are usually formed when the source water is groundwater. THM level can vary within a single water supply depending on the season, water temperatures, amount of natural organic matter in water, pH, amount of chlorine added, point of chlorination, time in distribution system, and other factors, such as treatment processes used (Government of Newfoundland and Labrador 2000).

3.1.6 Control of Disinfection By-product

Disinfectants react with natural organic matter (NOM) to form organic disinfection by-products (DBPs). Treatment techniques are available that provide water suppliers with the opportunity to maximize potable water safety and quality while minimizing the risk of DBP formation. One of the best methods to control DBPs from any disinfection process is to remove NOM precursors prior to disinfection.

3.1.6.1 Organic Precursor Removal

There are there ways to effectively remove NOM precursors (Chlorine chemistry council 1997).

Coagulation and Clarification

Most treatment plants optimize their coagulation processes for turbidity (particles) removal. Coagulation processes can, however, be optimized for NOM removal. Precursors are removed when alum or ions of salts are used as coagulants for turbidity control. Further precursor removal is usually achieved by reducing the pH prior to or during the addition of these coagulants.

Adsorption

Adsorption processes have been used successfully in some applications for removing DBPs precursor

Table 3.3 Comparison of disinfectants and their application (Department of Health and Community Services 2001)

Characteristics Classification	Chlorine Primary and secondary	Chloramines Secondary	Ozone Primary	Mixed oxidants Primary and secondary	UV light Primary	Iodine Primary and secondary
Effectiveness						
Bacteria	Very good (HOCl)	Poor	Excellent	Very good	Very good	Very good
Viruses	Very good (HOCl)	Poor	Excellent	Very good	Very good	Good
Protozoa	Fair Good	Very poor	Very good	Good	Fair	Good
Helminthes		No information	Excellent	Good	No Information	No formation
Water quality						
pH	Strong effect	Strong effect	Little effect	Moderate effect	No effect	Strong effect
Turbidity	Strong effect	Strong effect	Strong effect	Strong effect	Strong Effect	Strong effect
Temperature	Strong effect	Strong effect	Strong effect	Strong effect	Strong Effect	Strong effect
Organic matter	Strong effect on	Little effect	Ozone Demand	Less effect than chlorine	No effect	Little effect
THMs	Chlorine demand can develop with precursors	Not formed	Little Formation	Less than chlorine	None	Not formed
Experience	Wide experience	Little	Limited	Limited	Limited	Limited
Cost of other disinfectants relative to chlorine	1.00	3.4	3.5	0.8–1.5	3.5	6.10

Table 3.4 Land use and potential pollutant analysis matrix (Department of Health and Community Services 2001)

Land use sources	Turbidity sediment	pH	NP	Algae	Viruses parasites	Bacteria	THM potential	Pesticides	SOCs[a]	VOCs[b]	HMs[c]	Fe/Mn
Cropland runoff	X		x	X	x	x	x	x			x	x
Dairies/feedlot	X		x	X	x	x	x					
Grazing	X		x	X	x	x	x					
Recreation					x	x						
Forest management	X		x	x		x	x	x				x
Roads	X		x	x			x					x
Mining	X	x									x	x
Industrial discharge	X	x	x	x		x	x	x	x	x	x	x
Wastewater discharge	X	x	x	x	x	x	x	x	x	x	x	x
Septic tanks		x	x	x	x	x	x	x				
Urbanization	X	x	x	x	x	x	x	x	x	x	x	x
Hazardous materials								x	x	x	x	x
Acid rain		x										

[a] SOCs, other synthetic organic chemicals
[b] VOCs, volatile organic chemicals
[c] HMs, heavy metals

material. Activated carbon can provide adsorption, and significant research has been dedicated to determining the available capacity of activated carbon for dissolved organic and specific micropollutants. Both granular-activated carbon and powdered-activated carbon perform this function.

Membrane Technology

Membranes have been used historically for desalination of brackish waters. The process uses hydraulic pressure to force the liquid through a semipermeable membrane. This technology has demonstrated excellent removal of THM precursors.

3.1.7 King Abdullah Canal (KAC): A Case Study

3.1.7.1 Introduction

The Hashemite Kingdom of Jordan is located in the heart of the Middle East between 29°11′ and 33°22′ N latitudes and 34° 59′ and 39° 12′ E longitudes with an area of 89.210 km^2 (Ministry of Municipal and Rural Affairs 1991). It is bordered by Syria in the north, Iraq in the northeast, Saudi Arabia in the southeast, and Israel and West Bank (Palestinian authority) in the west. The country is landlocked except for a 27 km stretch of coastline bordering the Gulf of Aqaba on the Red Sea in the extreme southwest (Fig. 3.2).

Water in fact is the origin of life, and it is the most important factor of human life and the economics in all countries. The King Abdullah Canal is a significant water project in the country, since it is a main municipal and agricultural water project for the Kingdom of Jordan. However, its water quality and quantity depend on the water supplied from outside sources in Israel and Syria. This canal extends from the upper part of the valley at Adasiyah down to the Dead Sea with a total length of 110 km. The water originates from the Yarmouk River, the international Jordan border with Syria (Fig. 3.3). The other sources are the Mukhaibeh wells, the Peace Conveyer water, from Israel, and along the canal the water from the side wadis and dams established on some of the side wadis such as wadi Al Arab, Ziglab, and King Talal dams.

The KAC construction started in 1958 with the excavation of Addasiyah Tunnel diverting some of the Yarmouk River water in a southerly direction on the eastern side of the Jordan Valley. The KAC was 70 km when it was first commissioned in 1961 and was planned to serve irrigation purposes only. It was extended later on by 8, 18, and 14.5 km between 1969 and 1989 making the total length of this canal about 110 km with a head discharge capacity of 20 $m^3\ s^{-1}$ and a tail discharge of 6 $m^3\ s^{-1}$. It transfers water from the north to the south to reach the northern shores of the Dead Sea. It irrigated 123,000 dunums in 1966, additional, 20,000 dunums in 1968, 37,000 dunums in 1978, and finally 60,000 dunums in 1988 (Mahasneh 2000). Presently, 45 MCM $year^{-1}$ of water is pumped from KAC to Amman with the provision for treatment at Zai plant, located between Deir Alla, the water

Fig. 3.2 Location of the Hashemite Kingdom of Jordan

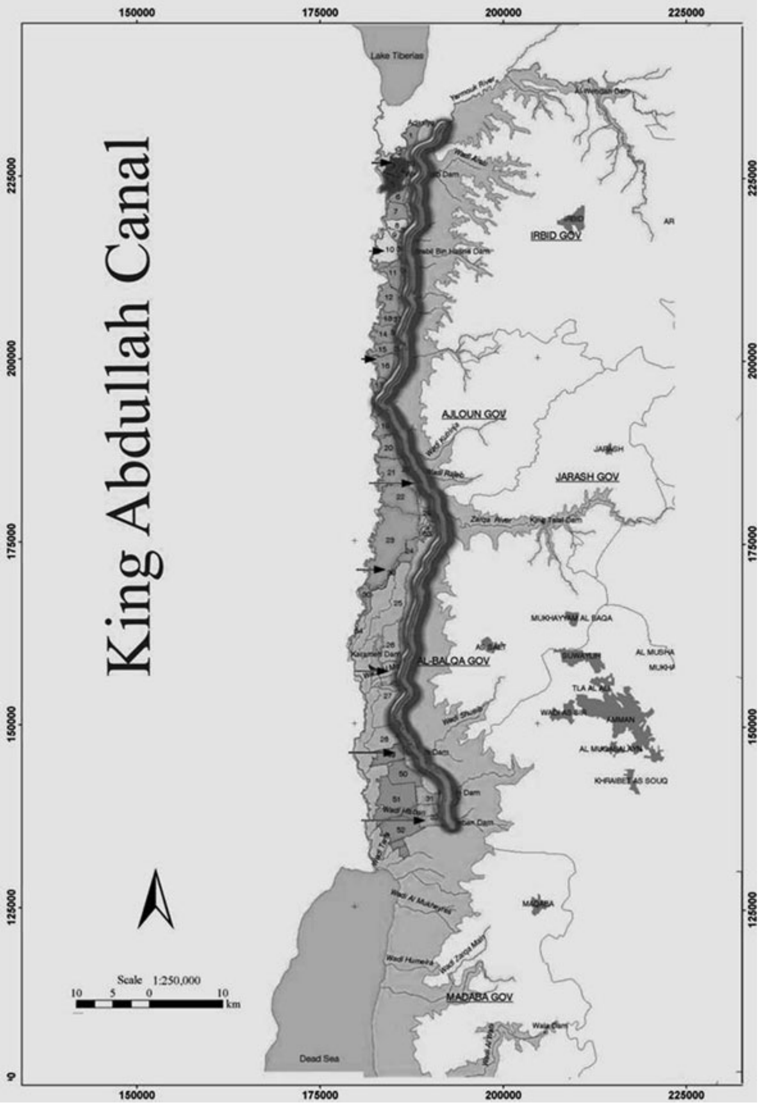

Fig. 3.3 Location of the King Abdullah Canal

intake site, and Amman. This amount is planned to be doubled in the near future, i.e. 1–2 years from now (JVA and WAJ (open files)).

3.1.7.2 The Study Area

The study area extends along KAC from its intake at the Yarmouk River to Deir Alla, at an elevation of 235 m below sea level to the south, along the left-hand side of the Jordan River terraces with a length of 65 km. The coordinates of the canal are 32°10′ 3.53″ N to 32°39′ 48.70″ N and 35°34′ 49.70″E to 35°36′ 13.72″ E in Universal Transverse Mercator (UTM) grid (Fig. 3.4).

KAC with a length of 65 km between the Yarmouk River and Deir Alla consists of a concrete trapezoidal canal with a base width of 3 m and surface width that varies from 10 to 11 m. The southern portions of the canal are narrower than at their beginning. The slope of the canal is 0.00018 (JVA) and its capacity at the initial entrance is 20 m^3 s^{-1}, reducing to 6 m^3 s^{-1} in Deir Alla site (Fig. 3.5). KAC receives water from a variety of sources (Fig. 3.6), in addition to the Jordanian-Israel Peace Treaty water of 1995.

King Abdullah Canal is the main surface water source of potable and agricultural water supply in Jordan. Two main problems seem to face the use of

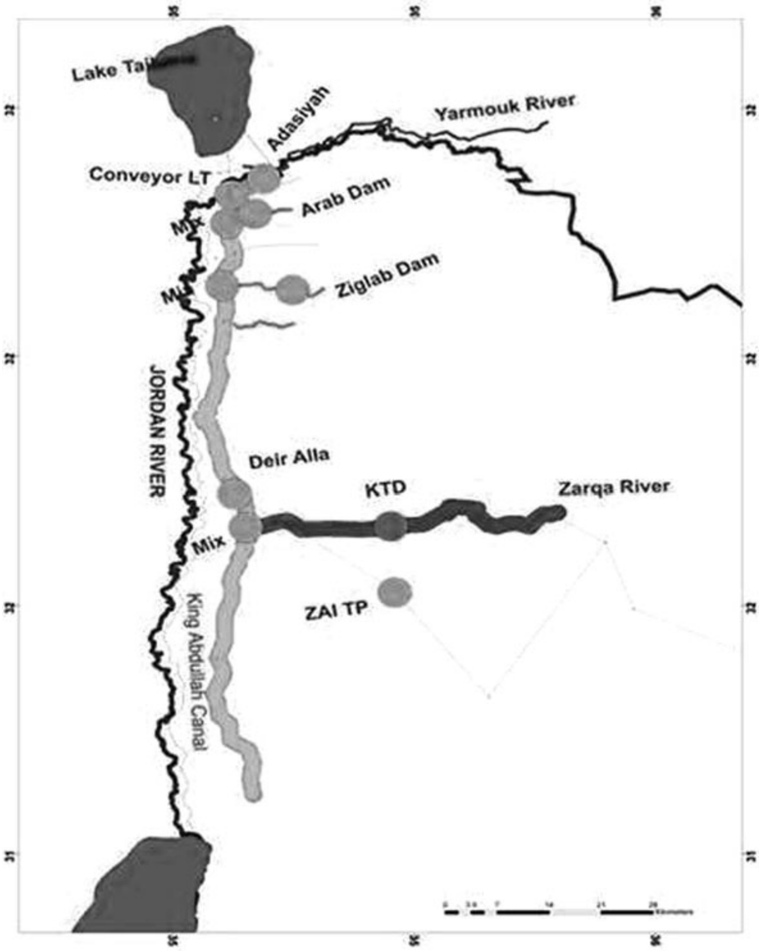

Fig. 3.4 The studied part of KAC and the water carrier to Zai purification plant

KAC for domestic and agricultural purposes. These problems are eutrophication and formation of chlorinated hydrocarbons during water treatment. Both problems develop as direct or indirect results of anthropogenic activities. The main objectives of this case study are to investigate the organic compounds in KAC especially those leading to the formation of trihalomethanes and the possibility to control them before reaching Zai treatment plant. Organic compounds resulting from activities of organisms in KAC water have not been analyzed. Such compounds disintegrate along the canal and cause upon chlorination in Zai treatment plant nuisance problems (AL-Harahsheh 2007). The produced water may have odor and taste and concentrations of trihalomethanes (THMs) which are considered carcinogenic substances. The secondary objectives are as follows.

Study the parameters in the water which lead to algal activity, especially blue-green algae, phytoplankton and zooplankton, and bacteria, and the eutrophication processes such as PO_4,$^{-3}$, $NO_3{}^-$, light, high temperature, and certain amounts of heavy metals which are generally available in urbanized catchments. Water samples were collected from May 2005 to August 2006, one time every month, along KAC from Yarmouk River at Nafq site to Deir Alla, KTD, and from the mixture of KAC with KTD. Table 3.5 and Fig. 3.7 show the sites.

3.1.7.3 Results

The Nutrients

The average $PO_4{}^{3-}$ and $NO_3{}^-$ concentrations in winter months is higher than in summer months due to

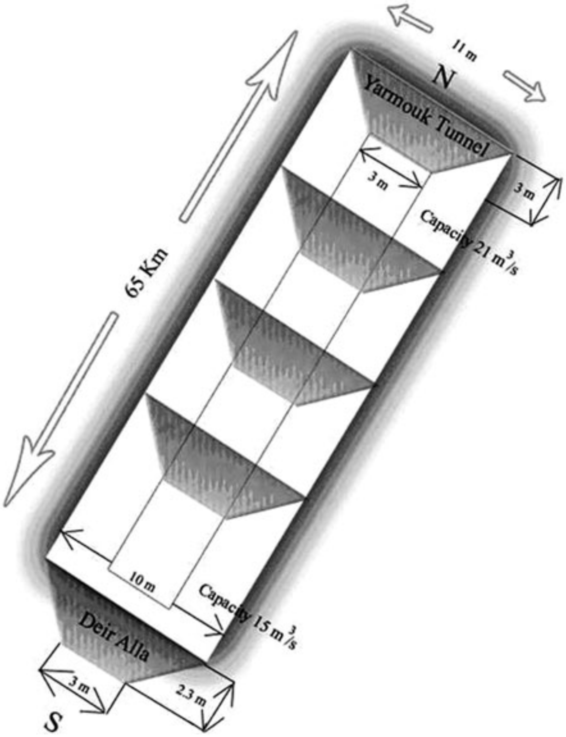

Fig. 3.5 Geometry of KAC (Al-Khoury 2005)

the flushing of the soils by the runoff. The highest concentration of phosphate ions (PO_4^{3-}) was detected in Yarmouk River while the lowest concentrations were found in the mixture of KAC and Ziglab Dam water (Fig. 3.8).

Deir Alla site and the mixture of KAC and Arab Dam have the highest concentration of nitrate ions. The lowest concentration of water was found in the Yarmouk River (Fig. 3.8).

Biological Parameters

Biological oxygen demand (BOD_5), chemical oxygen demand (COD), total organic carbon (TOC), total coliforms and *Escherichia coli*, chlorophyll-a, and plankton count were analyzed to present the biological parameters in the waters of the study area.

COD

The COD values ranged from 1 mg L^{-1} in KAC (Deir Alla) in November 2005 to 50 mg L^{-1} in the mixture of KAC and Peace Water in March 2006.The highest COD concentration value in these locations mixture of KAC and Arab Dam, Deir Alla, KTD and mixture of KAC during winter months while in summer months the Yarmouk River, mixture of KAC and Peace Water, and Arab Dam and mixture of KAC showed the highest COD concentration (Fig. 3.9).

BOD_5

The BOD_5 concentration values ranged from 0 mg L^{-1} in specific locations to 39 mg L^{-1} in the mixture of KAC and KTD. The average BOD_5 values has higher concentration in summer months in all locations except mixture of KAC and KTD in winter months (Fig. 3.9).

TOC

The TOC concentration values ranged from 0.92 mg L^{-1} in Yarmouk River during March 2006 to 9.64 mg L^{-1} in the mixture of KAC and KTD during December 2005. The average TOC has higher concentrations in summer months, except the Yarmouk River and the mixture of KAC and Peace Water which have higher concentrations in winter months (Fig. 3.9).

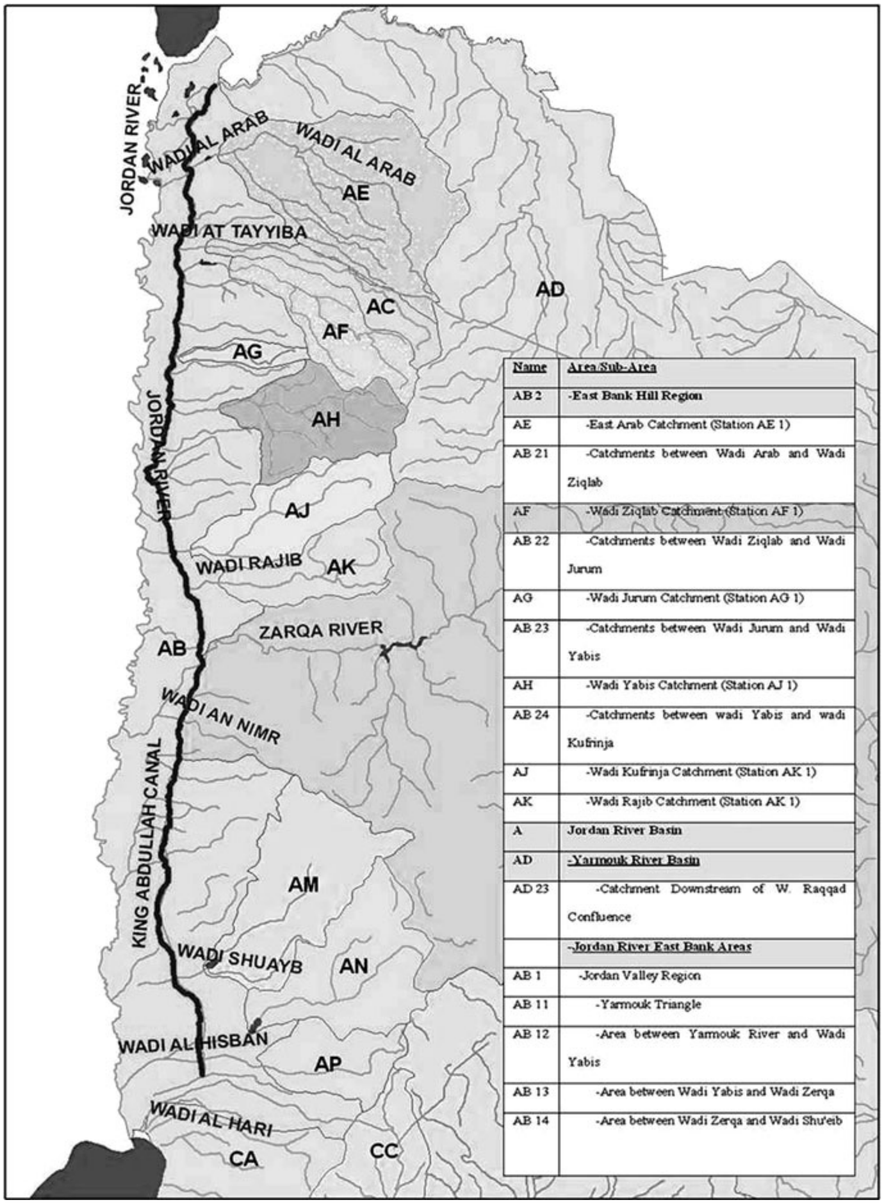

Fig. 3.6 Catchment areas and main wadis in the study area

Name	Area/Sub-Area
AB 2	**-East Bank Hill Region**
AE	-East Arab Catchment (Station AE 1)
AB 21	-Catchments between Wadi Arab and Wadi Ziqlab
AF	-Wadi Ziqlab Catchment (Station AF 1)
AB 22	-Catchments between Wadi Ziqlab and Wadi Jurum
AG	-Wadi Jurum Catchment (Station AG 1)
AB 23	-Catchments between Wadi Jurum and Wadi Yabis
AH	-Wadi Yabis Catchment (Station AJ 1)
AB 24	-Catchments between wadi Yabis and wadi Kufrinja
AJ	-Wadi Kufrinja Catchment (Station AK 1)
AK	-Wadi Rajib Catchment (Station AK 1)
A	**Jordan River Basin**
AD	**-Yarmouk River Basin**
AD 23	-Catchment Downstream of W. Raqqad Confluence
	-Jordan River East Bank Areas
AB 1	-Jordan Valley Region
AB 11	-Yarmouk Triangle
AB 12	-Area between Yarmouk River and Wadi Yabis
AB 13	-Area between Wadi Yabis and Wadi Zerqa
AB 14	-Area between Wadi Zerqa and Wadi Shu'eib

Table 3.5 Sites of the sampling

No.	Site name
1	Yarmouk River (Nafq)
2	Mix (KAC +Lake Tiberias)
3	Mix (KAC +Arab Dam)
4	Mix (KAC +Ziglab)
5	Deir Alla
6	Mix (KAC + KTD)

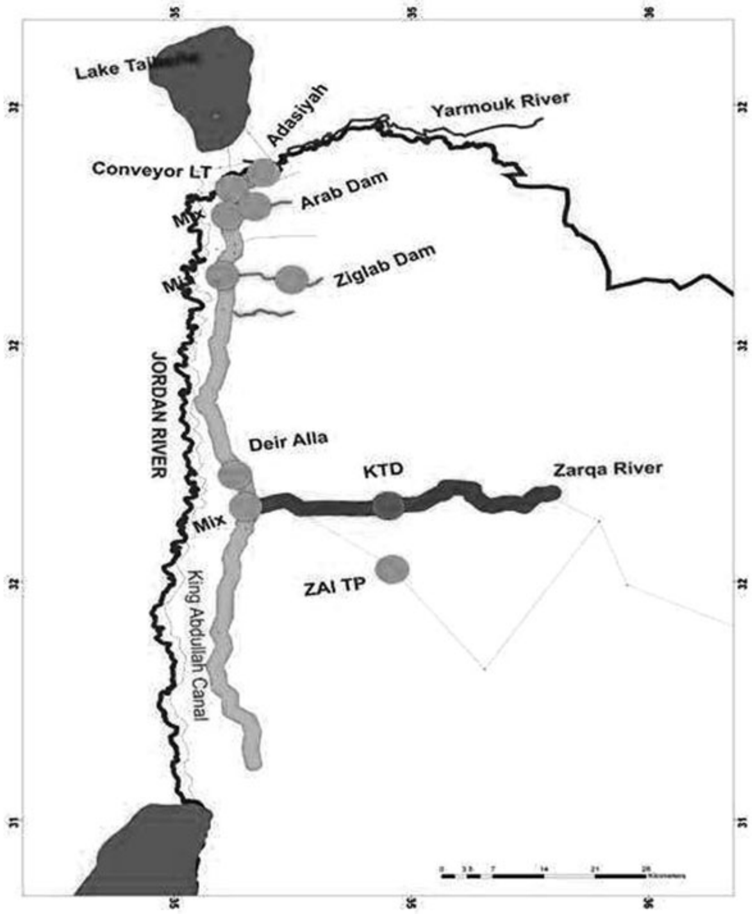

Fig. 3.7 The studied part of KAC and the water carrier to Zai purification plant

From the above result, it can be concluded that the mixture of KAC and KTD has the higher concentration in COD, BOD_5, and TOC followed by the mixture of KAC and Arab Dam water and Deir Alla.

THMs

The THM formation potentials for the samples show high diversity due to the different sources of the water body and the different characteristics of inflows. Brominated THM is found to be dominating in KAC water, especially $CHBr_3$ and $CHClBr_2$. It is well known in the literature that the toxicity of brominated hydrocarbons is 50–100 times of that of chlorinated hydrocarbons (Hua et al. 2006). This might be a source of considerable concern for water agencies using KAC water as a source for drinking water (Table 3.6).

The mixture of KAC and KTD water has the highest THM potential of 155.2 because a good part of its flow is composed of effluents from the overloaded wastewater treatment plant of Khirbet Samra while the KAC at Deir Alla has the lowest THM potential.

Total coliforms and *E. coli*

Bacterial counts play an important role in the biological process because heterotrophic bacteria break down organic matter into smaller molecules and carbon dioxide. The presence of coliforms, particularly fecal coliforms, is an indication of the presence of associated pathogens. Thus, determination of coliforms and fecal coliforms in water is essential as water quality parameters. The total bacterial count was determined by incubating on a nutrient in an agar plate at 28°C for 48 h. Total coliforms count and fecal coliforms were determined using multiple tube fermentation. The total

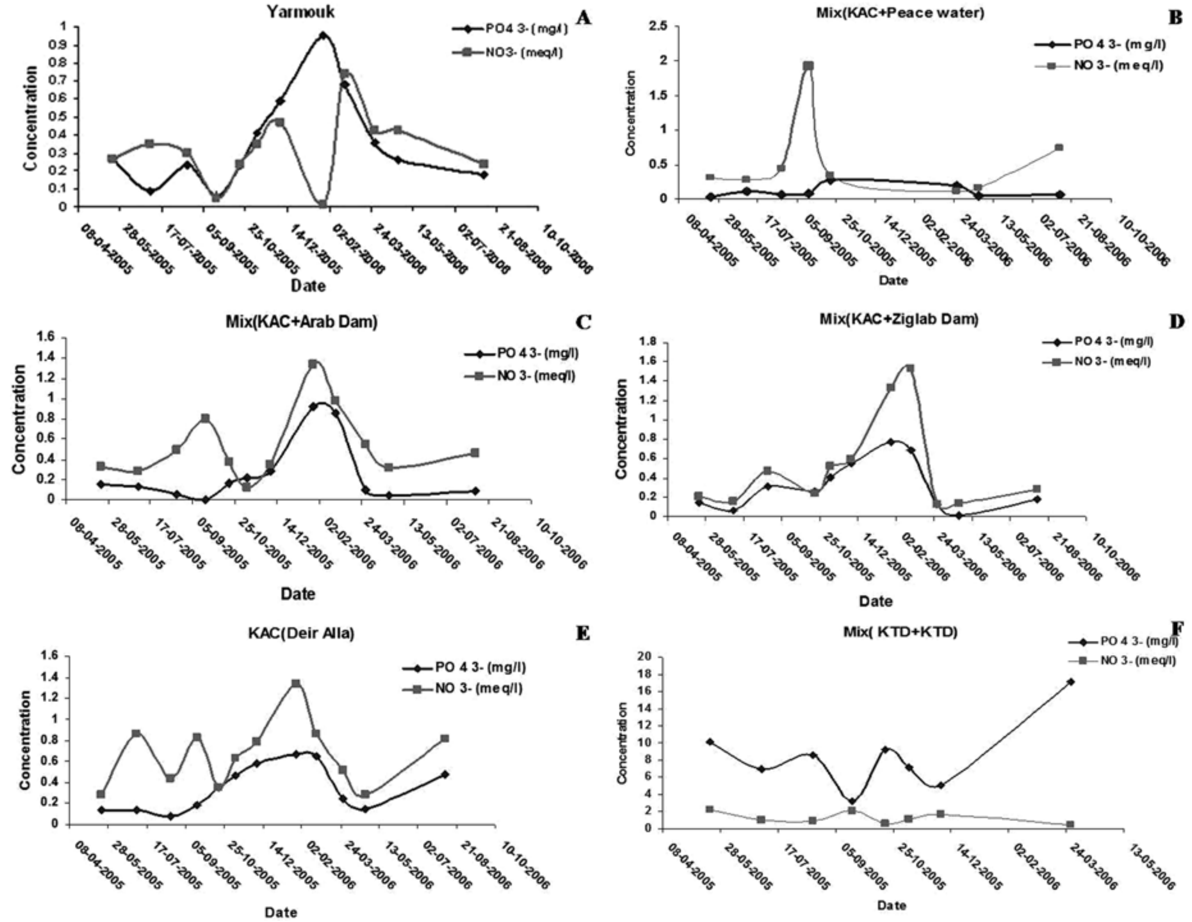

Fig. 3.8 Concentrations of $PO_4{}^{3-}$ and $NO_3{}^{-}$ for all sampled locations in the study area. (**a**) Yarmouk River; (**b**) mixture of KAC and Peace Water; (**c**) mixture of KAC and Arab Dam water; (**d**) mixture of KAC and Ziglab Dam water; (**e**) KAC water at Deir Alla; and (**f**) mixture of KAC and KTD

coliform numbers were estimated in a Lauryl tryptose broth and the tubes were incubated at 37°C for 24–48 h. For fecal coliforms count, *E. coli* was incubated at 44.5°C for 24 h.

The analysis of total coliforms and *E. coli* is presented in Table 3.7 for three locations only: Yarmouk River, mixture of KAC and Peace Water, and KAC at Deir Alla. The average total coliforms (MPN/100 mL) in winter months was higher than in summer months, while *E. coli* numbers were higher in summer months than in winter months except for the Peace Water. The mixture of KAC and Peace Water showed the highest total coliforms followed by KAC at Deir Alla and the Yarmouk River, respectively, while the mixture of KAC and Peace Water has higher *E. coli* numbers than the Yarmouk River and KAC at Deir Alla, respectively. The total coliforms ranged from 230 MPN/100 mL to >16,000 MPN/100 mL in three locations. *E. coli* number ranged from 20 MPN/100 mL in Yarmouk River to >16,000 MPN/100 mL in the mixture of KAC and Peace Water.

Chlorophyll-a and Plankton Count

Regarding the algae all genes found in the last three locations (Yarmouk River, mixture of KAC and Peace Water, and KAC at Deir Alla) were classified into six main groups (A–F) as illustrated in Table 3.8 and Appendix C.

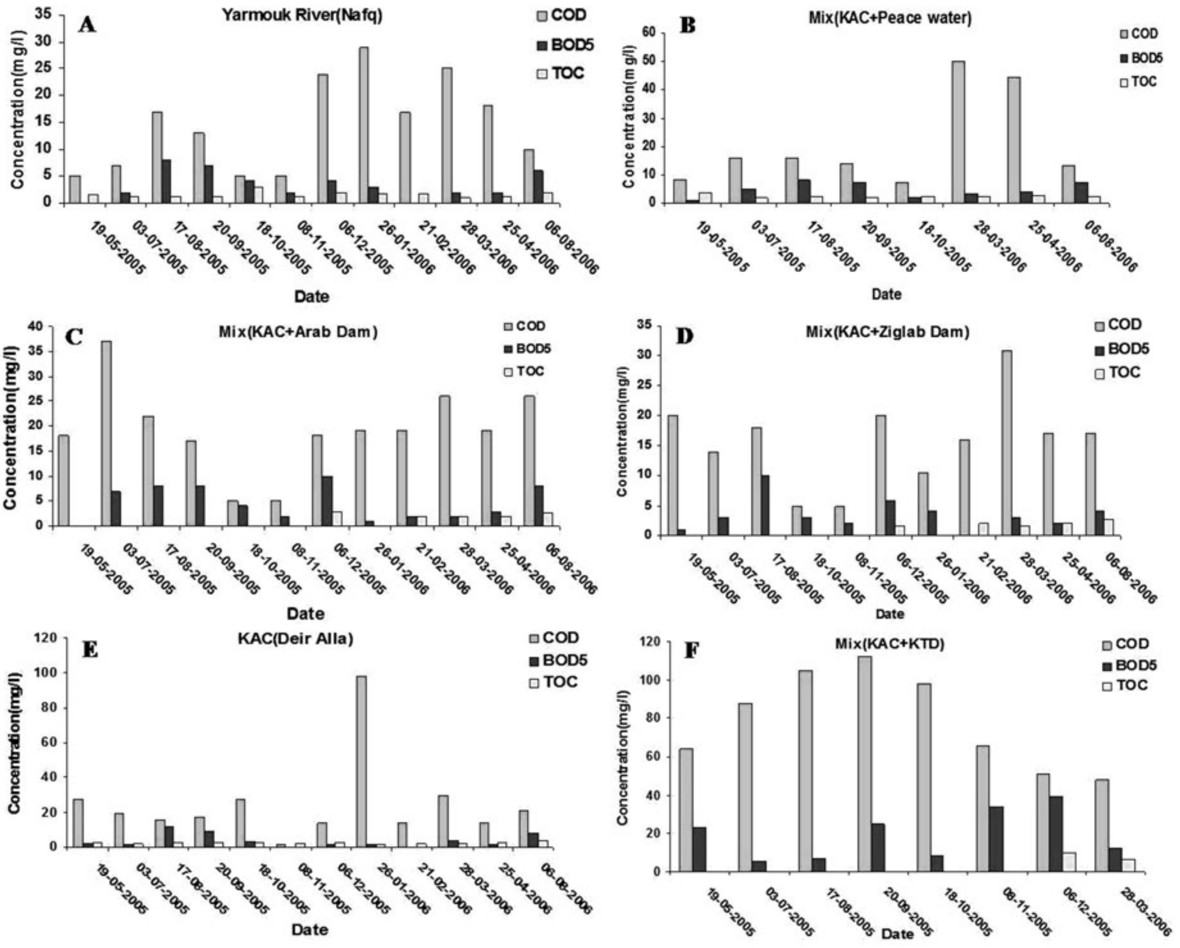

Fig. 3.9 Concentrations of COD, BOD$_5$, and TOC in the sampled locations of the study area. (**a**) Yarmouk River; (**b**) mixture of KAC and Peace Water; (**c**) mixture of KAC and Arab Dam water; (**d**) mixture of KAC and Ziglab Dam water; (**e**) KAC water at Deir Alla; and (**f**) mixture of KAC and KTD water

Table 3.6 THMs concentration in some samples

Location	$CHCl_3$ (μg L^{-1})	$CHCl_2$ Br (μg L^{-1})	$CHClBr_2$ (μg L^{-1})	$CHBr_3$ (μg L^{-1})	Total THMs (μg L^{-1})
Yarmouk River	1.2	8	27.7	33.8	70.7
Mix KAC + Peace Water	1.6	10.9	35.4	47.6	95.5
Mix KAC+Arab Dam	1.7	15.2	48.1	56.2	121.2
Mix (KAC+Ziglab Dam)	1.2	12.5	41.5	73.4	128.6
KAC Deir Alla	3	2	1.6	1.7	6.5
Mixture of KAC and KTD water	4.6	22.2	61.1	67.3	155.2

Table 3.7 Total coliforms and *E. coli* in the different sites of the study area

	Yarmouk River		KAC + Peace Water		KAC (Deir Alla)	
Date	Total coliforms (MPN/100 mL)	*E. coli* (MPN/100 mL)	Total coliforms (MPN/100 mL)	*E. coli* (MPN/100 mL)	Total coliforms (MPN/100 mL)	*E. coli* (MPN/100 mL)
19/05/2005	1,300	230	5,000	1,100	3,000	500
08/06/2005	300	300	5,000	240	3,000	300
03/07/2005	500	300	≥ 16,000	900	5,000	300
17/08/2005	800	300	9,000	800	500	40
20/09/2005	1,300	800	≥ 16,000	≥ 16,000	500	500
18/10/2005	300	20	16,000	500	800	40
08/11/2005	2,400	230	500	500	1,300	300
06/12/2005	16,000	300			≥ 16,000	230
26/01/2006	500	230			230	40
21/02/2006	1,100	230			9,000	500
28/03/2006	3,000	500	230	230	1,300	800
25/04/2006	2,400	800	9,000	1,100	9,000	300

Table 3.8 The algal genus found in the main water sources of KAC

Yarmouk River		Mixture of KAC and Peace Water		Deir Alla	
Euglena	E	Diatoms	B	*Navicula*	B
Chlamydomonas	C	*Gomphonema*	F	*Chlamydomonas*	C
Coleochaete		*Trachelomonas*	B	*Zygnema*	E
Oikomonas		*Oikonomos*		*Oikomonas*	
Navicula	B	*Staurastrum*	A	*Dinomonas*	
Pinnularia	D	*Cladophora*	F	*Pinnularia*	D
Oedogonium	F	*Tetraedon*	C	*Cocconeis*	D
Dinomonas		*Euglena*	E	*Pelomyxa*	
Anisonema		*Gomphnema*	C	*Anisonema*	
Pelomyxa		*Senedsmus*	E	*Heliozoans*	
Cyclotella	B	*Spirogyra*	B	*Navicula*	B
Gomphonema	F	*Volvox*	A	*Cyclotella*	B
Scenedsmus	E	*Phacus*	C,E	*Scenedesmus*	E
Tetracystis				*Coelastrum*	E
Chlorococum	C			*Pediastrum*	E
Achley(funji)				*Cosmarium*	
Ceratium	A			*Chlorella*	B
Gomphonema	C			*Schizochlamysm*	
Chlorococcus				*Stylosphaeridium*	
Oscillatoria	B,C*			*Chlorococcus*	
Diatoms	B			*Tetraedon*	C
Nitzschria				*Asterococcus*	
Micrasterias	D			*Tetraspora*	F
Clorella	B			*Euglena*	E
Spirogyra	B,C			*Spirogyra*	B,C
Cladophora	D,F			*Doctylococcopsis*	
Codium				*Aphanocapsa*	
Vaucheria	F			*Micasterias*	D
Tetraedon	C			*Gleocystis*	
Chaetophora	F			*Anisomonas*	
Gymodium				*Dionomonas*	
Staurastrum	A			*Chaetophora*	F
Crucigenia					

**Oscillatoria* is a blue-green characteristic alga for eutrophic water body (Ryding and Rast 1989).

A: Taste and odor algae; B: Filter clogging algae; C: Polluted water algae; D: Clean water algae; E: Plankton and other surface water algae; F: Algae growing on reservoir wells

Fig. 3.10 Concentration of chlorophyll-a and plankton counts in the study area. (**a**) Yarmouk River; (**b**) mixture of KAC and Peace Water; and (**c**) KAC water at Deir Alla

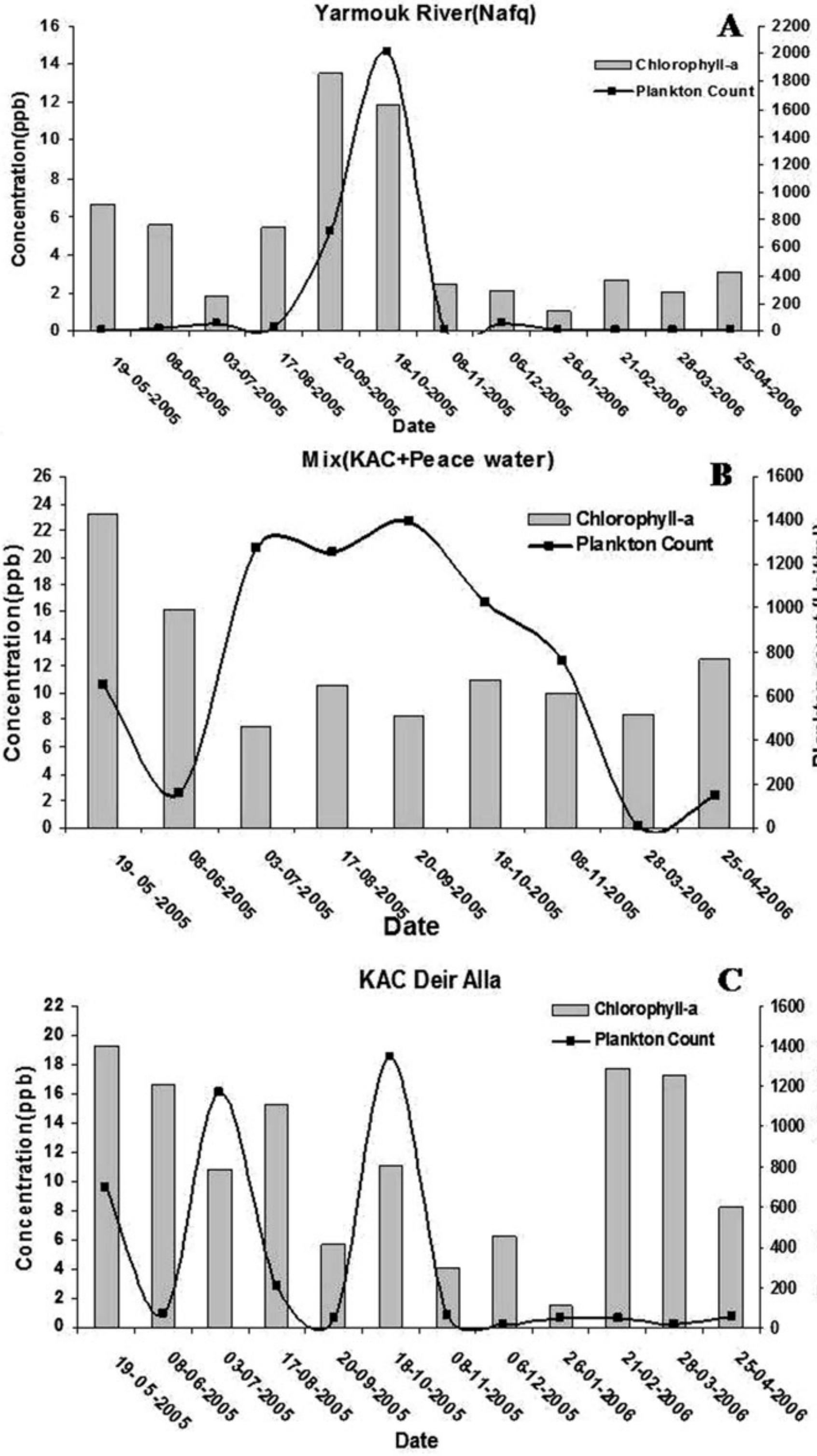

Table 3.9 Concentrations of chlorophyll-a and plankton count in the test sites

	Yarmouk River		KAC + Peace Water		KAC (Deir Alla)	
Date	Chlorophyll-a (ppb)	Plankton count (Unit mL^{-1})	Chlorophyll-a (ppb)	Plankton count (Unit mL^{-1})	Chlorophyll-a (ppb)	Plankton count (Unit mL^{-1})
19/05/2005	6.69	11	23.27	649	19.26	694
08/06/2005	5.55	17	16.24	160	16.67	67
03/07/2005	1.84	53	7.49	1, 275	10.83	1, 169
17/08/2005	5.42	28	10.54	1, 257	15.24	203
20/09/2005	13.5	718	8.33	1, 397	5.64	45
18/10/2005	11.85	2007	10.93	1, 023	11.08	1, 347
08/11/2005	2.47	14	9.94	762	4.11	63
06/12/2005	2.12	52			6.18	17
26/01/2006	1.00	14			1.51	44
21/02/2006	2.66	8			17.70	49
28/03/2006	2.04	6	8.40	8.61	17.30	19
25/04/2006	3.11	5	12.41	145	8.22	53

The concentration of chlorophyll-a and plankton count were high during summer months than in winter months in all the locations due to high temperatures, high nutrients, and high light intensity, which encourage algal growth in the canal resulting in taste and odor problems. Phytoplankton plays an important role in material and energy fluxes in eutrophic water systems, like KAC, and often controls the DO and the nutrient budget as well as the pH values. During the day, algae consume CO_2 from the air by photosynthesis, which leads to decreases in the concentration of carbonic acid; consequently, the pH value increases and O_2 concentration in the water increases. On the other hand, during night the algal activity stops and the pH and DO values decrease.

Thus the difference in pH, DO, and temperature values between summer months and winter months can be used as a good indicator for the presence of algae in the canal water. The highest concentration of chlorophyll-a in KAC at Deir Alla is followed by that of the mixture of KAC and Peace Water, while the mixture of KAC and Peace Water showed higher plankton counts, followed by Deir Alla. Chlorophyll-a concentration ranged between 1 ppb during January 2006 in the Yarmouk River and 23.27 ppb during March 2005 in the mixture of KAC and Peace Water while the plankton count ranged from 5 unit mL^{-1} during April 2006 in the Yarmouk River to 2007 unit mL^{-1} in the same location during October 2005 (Table 3.9 and Fig. 3.10).

3.1.8 Conclusions

The case study shows that sun illumination, NO_3 concentrations of more than 10 mg L^{-1}, PO_4 concentrations of 1 and more mg L^{-1}, micronutrients of K, Fe, and Mn in addition to the slow water movement along KAC can be made responsible for the eutrophication processes in the KAC water. The algal residues and other organic precursors originating from the catchment areas produce upon chlorination of the water THMs, which are carcinogenic substances. The canal water rich in bromide produces higher percentages of bromoform than the other THMs. Bromoform has a much higher toxicity than the other THMs; hence, the danger is in not limiting the formation of the THMs, but by reducing the eutrophication processes and the amount of THM precursors in the raw water.

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Chapter 4

Eutrophication and Restoration of Shallow Lakes from a Cold Temperate to a Warm Mediterranean and a (Sub)Tropical Climate

Meryem Beklioglu, Mariana Meerfhoff, Martin Søndergaard, and Erik Jeppesen

Abstract Freshwaters are crucial to the establishment of any human community and to the conservation of all land-based wildlife. The majority of the world's freshwater area is shallow, typically $\leq$3 m deep, and polymictic. Freshwaters are rather versatile in their use and have great economical and conservation value. They can be/are often dominated by littoral communities as the water depth is generally shallow enough to support the growth of submerged macrophytes, and the species richness of invertebrates, fish and waterfowl communities is more diverse than in deep lakes. Eutrophication or nutrient enrichment affects the entire aquatic system by altering trophic structure, biodiversity and biogeochemical cycles, as well as seasonal dynamics. In shallow lakes with low nutrient concentrations (<0.025 mg TP L^{-1}), a clear state dominated by submerged plants often occurs as phytoplankton is nutrient limited. In north temperate shallow lakes, the TP range for coexistence of alternative states varies from ca. 0.025 to ca. 0.15 mg TP L^{-1}. In these lakes, submerged plants stabilize the clear water state through several physico-chemical buffer mechanisms of which some have proven to be weak in warm lakes. An abrupt shift to a turbid state may occur after surpassing a lake-specific nutrient threshold. In warm lakes, internal nutrient cycling is relatively more important and external loading relatively less important than in cold temperate lakes. Warm lakes tend to be more productive than cold lakes with similar nutrient concentrations. With a diverse and abundant omnivorous fish community, the predation pressure on zooplankton is strong, top-down control becomes less important and nutrient control thus emerges as a decisive factor for the water clarity. For cold lakes, restoration methods encompass both bottom-up and top-down controls; however, for warm lakes bottom-up or nutrient control methods appear to be most significant for eutrophication control. As for climate change, warming is likely to exacerbate some symptoms of eutrophication in both cold and warm climates. The structure and functioning of cold temperate shallow lakes are expected to become more similar to those of (sub)tropical shallow lakes, as the temperature increase will enhance the top-down controls of omnivorous and benthivorous fish as well as the nutrient cycling.

Keywords Bottom-up control · Climate change · Fish manipulation · Salinity · Submerged plants · Top-down control · Water level fluctuation

4.1 Shallow Lakes

Freshwaters are crucial to the establishment of any human community and to the conservation of all land-based wildlife. All civilizations are founded close to a reliable supply of freshwater, often large lakes or rivers that are used as potable supply and for irrigation, for harvesting fish and other food sources, and for recreational activities such as boating, swimming and angling (Moss 1998). Among the various bodies of freshwaters, perhaps because of their grandeur and clarity, we tend to prize large, deep bodies of water and tend to view open water as an ideal (Moss et al. 2005, Moss 1998). However, the majority of the world's

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freshwater area is shallow and confined mostly to lowlands with different origins, ranging from geological upheavals to glaciations, landslides, and damming of rivers either by nature or by humans (Moss 1998, Wetzel 2001). Shallow lakes are rather versatile in the uses to which they have been put, but generally have great economical and conservation value. On a world scale, the monetary value of shallow lake wetlands is immense due to the natural goods and services they provide (Costanza et al. 1997). The total value of the world's natural systems amounts to over 30 trillion US\$ year^{-1}, or three times the combined gross domestic products (GDP) of all the world's economies combined (Costanza et al. 1997). The value of shallow lakes and wetlands proves to be greater than that of the forests and grasslands and totals trillions of US\$ year^{-1}. The contribution of freshwater wetlands is over 10%, and of wetlands as a whole over 25% (Costanza et al. 1997).

Shallow lakes are typically <3 m deep and polymictic, implying that thermal stratification is not stable, and that the entire water column mixes more or less continuously (Moss et al. 1996, Moss 1998). They can be dominated by littoral communities as the water depth is generally shallow enough to support sufficient underwater light for the growth of submerged macrophytes. Submerged macrophytes may cover a large part of the lake area, in contrast to deep lakes where areas for growth of submerged plants are limited and confined to near-shore areas (Moss 1998, Scheffer 1998, Jeppesen et al. 1998). In shallow lakes and wetlands, submerged plants are significant as both habitat and food for numerous consumers, and they have an essential structuring effect on the whole ecosystem (Moss 1998, Scheffer 1998, Meerhoff and Jeppesen 2009). In combination, the large littoral zone and shallow depth result in a greater impact of sediment processes on the water column dynamics and an increased benthic–pelagic coupling (Vadeboncoeur et al. 2002).

Shallow lakes are more productive per unit area of water than deep ones as a result of greater recycling of nutrients that become directly available to primary producers, especially phytoplankton and periphyton (Jeppesen et al. 1997, Moss 1998). Shallow lakes are more species rich compared to deep lakes due to a strong effect of a wide littoral zone and the associated macrophytes (Carpenter and Lodge 1986, Jeppesen et al. 1997, 2003). Thus, shallow lakes contribute significantly to biodiversity on earth and act as important foraging areas for terrestrial animals and waterfowl than turbid lakes (Brönmark and Hansson 1999, Jeppesen et al. 2000, Burks et al. 2006, Declerck et al. 2005, De Meester et al. 2006, Moss et al. 2009). Despite their great abundance and value, research in shallow lakes was largely neglected until the 1980s. Early European shallow lake research was mainly conducted by British, Danish and Dutch scientists who stated hypotheses on the structure and functioning of shallow lakes in the north temperate region, some of which (with later updates by numerous groups around the world) are discussed in the following.

4.2 North Temperate "Cold Shallow Lakes"

We first concentrate on lakes in the north temperate climates, referred to as "cold". They are mainly located in Europe and North America where the roots of shallow lake ecology were established.

4.2.1 Alternative Stable States

Shallow lakes do not respond to external forces in a linear but rather in a non-linear and abrupt fashion (Scheffer et al. 1993, Jeppesen et al. 1998, Moss 1998). The "alternative stable states" hypothesis asserts that shallow lake ecosystems can exhibit at least two alternative states, a turbid state with phytoplankton and/or suspended matter dominance and a clear state with dominance of submerged plants, under similar conditions (e.g. intermediate range of nutrient availability) and with a certain degree of stability against external forces. Shallow lakes dominated by submerged plants have higher biodiversity, from invertebrates to fish and waterfowl. Several physical, chemical and biological stabilizing feedback mechanisms, many of which are based on interactions with submerged plants (see Section 4.2.2), may help to provide stability against perturbations. Enhanced nutrient availability or eutrophication is the most thoroughly explored external force, especially in north temperate shallow lakes. Indirect as well as direct effects of enhanced total phosphorus (TP) levels may act as the trigger promoting such a shift. In warm (e.g. tropical and subtropical) shallow lakes, free-floating plants may represent

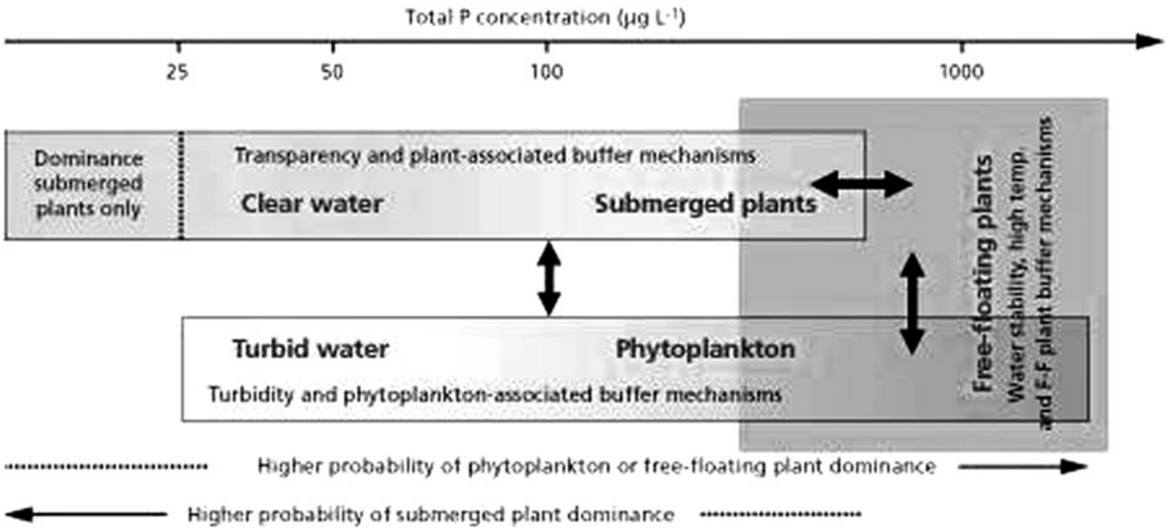

Fig. 4.1 General model of the alternative states in shallow lakes over a gradient of nutrients (phosphorus) where the three main alternative states: phytoplankton-dominated, submerged plant-dominated and free-floating plant-dominated may occur. Modified (adding free-floating plants), with permission, from the original model for temperate lakes published in Moss et al. (1996) and taken from Meerhoff and Jeppesen (2009) with kind permission from Elsevier

another relatively stable alternative state (Scheffer et al. 2003, Meerhoff and Jeppesen 2009) (Fig. 4.1).

In pristine waters, TP concentrations are around a few micrograms to a few tens of micrograms per liter, while total nitrogen (TN) concentrations will usually be ca. 10–20 times as high (Moss 1998, Moss et al. 1996). Whether nitrogen (N) or phosphorus (P) limits primary production in shallow lakes depends on their relative availability in the lake ecosystem. This, in turn, depends on both the amount of N and P entering the system and on in-lake processes. Nutrient loads are strongly affected by hydrology and catchment characteristics such as soil type and land use (Downing and McCauley 1992, Downing et al. 1999, Kosten et al. 2009a), which may vary with climate. P is usually scarce in aquatic systems and frequently limits algal growth (Schindler 1977). Thus, the availability of P has traditionally been used to determine the threshold for a switch between alternative states (Scheffer et al. 1993). At low nutrient concentrations (usually <0.025 mg TP L^{-1}), a clear state dominated by submerged plants occurs as phytoplankton is nutrient limited, whereas submerged plants can take up nutrients from the sediments. In north temperate shallow lakes, the TP range for coexistence of alternative states varies from ca. 0.025 to ca. 0.15 mg TP L^{-1}; however, the upper limit can be much higher if N is a limiting nutrient, particularly in fishless small lakes (Jeppesen et al. 1997, 2003, 2005a). At high nutrient levels, lakes are turbid and submerged plants rare. A shift to a turbid state may occur abruptly after surpassing a lake-specific nutrient threshold (see Section 4.2.2). A recent analysis of data from 782 lake years in different climate zones in North America, South America, and Europe has confirmed the earlier findings that submerged macrophyte coverage (30% of the lake area) decreased in a sigmoidal way with increasing TP concentrations, falling most steeply between 0.05 and 0.2 mg L^{-1}. Substantial submerged macrophyte coverage was also rare in lakes with TN concentrations above 1–2 mg L^{-1}, except for lakes with very low TP concentrations (Kosten et al. 2009b). The maximum TP concentration allowing substantial submerged macrophyte coverage was clearly higher in cold regions with more frost days. This is in agreement with other studies revealing marked influence of ice cover duration on shallow lake ecology through partial fish kills that may improve light conditions for submerged macrophytes by cascading effects on periphyton and phytoplankton (Kosten et al. 2009b, Jackson et al. 2007).

Enhanced growth of periphyton and phytoplankton results in a deteriorated underwater light climate and a subsequent collapse and loss of plants (Phillips et al. 1978, Jeppesen et al. 1997, 2005a, Jones and Sayer 2003). Nutrients alone, however, do not appear to determine whether epiphyton can reach sufficient densities to negatively impact macrophytes. Experimental work suggests that other mechanisms are important for the loss of macrophytes (Jones et al. 2000, Williams et al. 2002). These experiments, previous work (Brönmark and Vermaat 1997) and a survey of 17 macrophyte-dominated lakes in the United Kingdom (Jones and Sayer 2003) have resulted in the proposal that the strength of a trophic cascade in the littoral zone determines the fate of macrophytes. This further augments phytoplankton growth through nutrient release. Also, increased abundance of plankti-benthivorous fish stimulates phytoplankton growth via predation on the major grazers. Other factors, such as pronounced changes in water level and wind exposure, are among the reasons suggested for triggering a shift of ecosystem state in shallow lakes (Jeppesen et al. 2005b, Meerhoff and Jeppesen 2009).

4.2.2 Role of Vegetation

In north temperate shallow lakes, submerged plants stabilize the clear water state through several physico-chemical buffer mechanisms (see Moss et al. 1996, Jeppesen et al. 1997, Scheffer 1998):

- the growth of submerged plants causes enhanced fixation of nutrients in macrophytes and associated epiphytes, leaving less nutrients available for the phytoplankton, especially in summer (Sand-Jensen and Borum 1991);
- the increase in plant biomass greatly helps reducing turbidity induced by sediment resuspension, which otherwise entails increased nutrient release to the water column (Barko and James 1998, Hamilton and Mitchell 1996). Reduced sediment resuspension significantly promotes the clear water state with consequent occurrence and dominance of submerged plants, especially in large lakes subjected to wind exposure;
- roots and plant surface areas, open to colonization by nitrifying and denitrifying bacteria, boost denitrification and, in turn, induce nitrogen-limited conditions (Weisner et al. 1994, Erikson and Weisner 1999); and submerged macrophytes may diminish phytoplankton by shading (Wetzel 2001). The effect of plants on nutrients and light does not entirely explain why submerged plants promote clear water conditions at similar P concentrations. Several direct mechanisms may offer an explanation. Submerged plants also reduce fish-induced resuspension through their effect upon fish community structure. Common Northern Europe fish species, such as the benthivorous bream (*Abramis brama*) and carp (*Cyprinus carpio*), stir up sediment when foraging and this may substantially increase the amount of suspended matter (and associated nutrients) in the water (Meijer et al. 1990, Breukelaar et al. 1994). These fish are often abundant in eutrophic plantless lakes. Furthermore, laboratory experiments have provided evidence for chemical suppression of phytoplankton growth by submerged plants such as *Ceratophyllum, Myriophyllum* and *Chara* spp. (Wium-Andersen 1987, Hilt and Gross 2008).

Submerged plants indirectly influence fish and invertebrates, particularly zooplankton, with cascading effects on the phytoplankton in temperate shallow lakes (Brönmark and Weisner 1992). Macrophytes favour predatory fish over zooplanktivorous fish, such as north European perch (*Perca fluviatilis*) which have a competitive advantage over roach (*Rutilus rutilus*) within the plant beds, as the former forage better within a structured habitat (Jacobsen et al. 1997, Perrow et al. 2002). Thus, in shallow lakes with substantial plant coverage, perch stand a better chance of reaching the predator stage. Strong predator fish like pike (*Esox lucius*) also thrive within vegetation (Grimm 1994). Multiple predatory fish can exert a strong top-down control on benthi-planktivorous fish, mainly cyprinids (Grimm and Backx 1990). This leads to lower predatory control of zooplankton, especially large-bodied species (e.g. *Daphnia*) (Moss et al. 1996, Jeppesen et al. 1997). Furthermore, submerged plants may act as a daytime refuge for large-bodied pelagic zooplankton species (e.g. *Daphnia* and other cladocerans), enabling them to avoid fish predation (Timms and Moss 1984, Lauridsen and Buenk 1996, Burks et al. 2002, 2006). At night the predation risk diminishes as several of the predators are visual hunters, and the zooplankton then moves to open water to graze down algae. Thus, plant refuges help zooplankton to control phytoplankton biomass and, thus, maintain a clear water state. Also, large mussel grazers (e.g. *Anodonta*, *Unio*, *Dreissena*) can be critical for clearing the water of phytoplankton (Ogilvie and Mitchell 1995, Dionisio and Van Donk 2002).

4.2.3 Eutrophication

Eutrophication is the result of excessive nutrient loading to the receiving lakes reflecting mainly human-inflicted impacts (Smith et al. 1999, Jeppesen and Sammalkorpi 2002). Shallow lakes, mostly confined to lowland areas, are very vulnerable to nutrient enrichment through the conversion of most lowlands to agricultural or urban land, with major effects on nutrient flows (Wetzel 1990, Moss 1998, Wetzel 2001, Smith et al. 1999, Jeppesen et al. 2003). Natural pristine systems generally conserve nutrients and store organic matter, whereas all agricultural systems lack such conservation mechanisms and leak nutrients to streams and other water bodies (Moss 1998). On a world scale, the key negative influence on lakes is eutrophication due to high external nutrient inputs deriving from domestic sewage, intensive agricultural activities and industry, ultimately breaking the tight nutrient cycles of terrestrial ecosystems in the catchment (Moss 1998, Wetzel 2001, Smith et al. 1999, Jeppesen and Sammalkorpi 2002). Eutrophication leads to a whole series of biological changes in lakes, with negative effects on their ecological state. Eutrophication also has practical implications, ranging from increased problems of using water for domestic supply and irrigation, toxic algal blooming and reduced value for recreational and commercial fisheries. Especially nowadays, many developing countries face an alarming intensification of eutrophication as a result of fast economic development and non-sustainable exploitation strategies. In the western world substantial efforts have been made to combat eutrophication and/or its symptoms (see Section 4.4). Today, in the developed world, the major source of nutrient loading to lakes is diffuse sources in the catchment, mainly intensive farming, while sewage water and fish farm waste play a major role in the developing countries (United Nations 2006, Jeppesen and Sammalkorpi 2002). Furthermore, increased demands for water by a growing human population, as well as the effects of global warming, are expected to escalate the eutrophication problem on a global scale.

However, eutrophication is not only a direct response of algae to increasing nutrient levels, but also a result of major changes in the food web. Typical changes in north European temperate shallow lakes are illustrated in Fig. 4.2 (for details, see Moss et al. 1996, Gulati and Van Donk 2002, Jeppesen et al. 1998, 2005a, 2009a, 2009b, Jeppesen and Sammalkorpi 2002). At the top of the food web, fish respond markedly to increased nutrient levels. A shift takes place from dominance of predatory fish, such as perch and pike, at low nutrient levels to dominance of cyprinids, especially roach, carp and bream, at high nutrient concentrations, while at intermediate levels the predatory fish resist and their biomass increases. Consequently, cyprinids exert a strong predation pressure on zooplankton and macroinvertebrate grazers (e.g. snails). The increased abundance of benthivorous, bottom-feeding fish promotes sediment resuspension and, in turn, enhances turbidity. Due to the high predation pressure, zooplankton biomass declines, leading to a lower ratio of zooplankton to phytoplankton biomass, in northern temperate lakes from around 0.5 to 0.8 in mesotrophic lakes to less than 0.2 when phosphorus concentrations are above 0.1–0.15 mg TP L^{-1}.

This implies that the zooplankton is incapable of controlling the phytoplankton in nutrient-rich lakes. An increased nutrient supply associated with decreased grazing pressure by zooplankton implies a strong increase in phytoplankton biomass, resulting in reduced water transparency (Secchi depth). Furthermore, due to the increased predation pressure, the zooplankton community becomes dominated by less efficient small-bodied grazers (e.g. rotifers). Thus, in eutrophic lakes, large-sized phytoplankton taxa (e.g. filamentous cyanobacteria) may thrive. The underwater light climate will be further impoverished for submerged plants through a fish predation-mediated decrease in the abundance of snails that are efficient grazers of epiphytes on plants. This decline is further accelerated by waterfowl feeding on macrophytes and macroinvertebrates. The conservation and biodiversity values are muted in turbid lakes where a few cyprinid fish species are dominant and the phytoplankton largely consists of cyanobacteria and green algae, with only few or no submerged plants and a greatly reduced diversity of waterfowl (excluding fish-eating birds).

4.3 Shallow Lakes in Different Climatic Regions

Our knowledge of shallow lakes in other climates is unfortunately limited, although the climate regime induces important differences in lake structure and

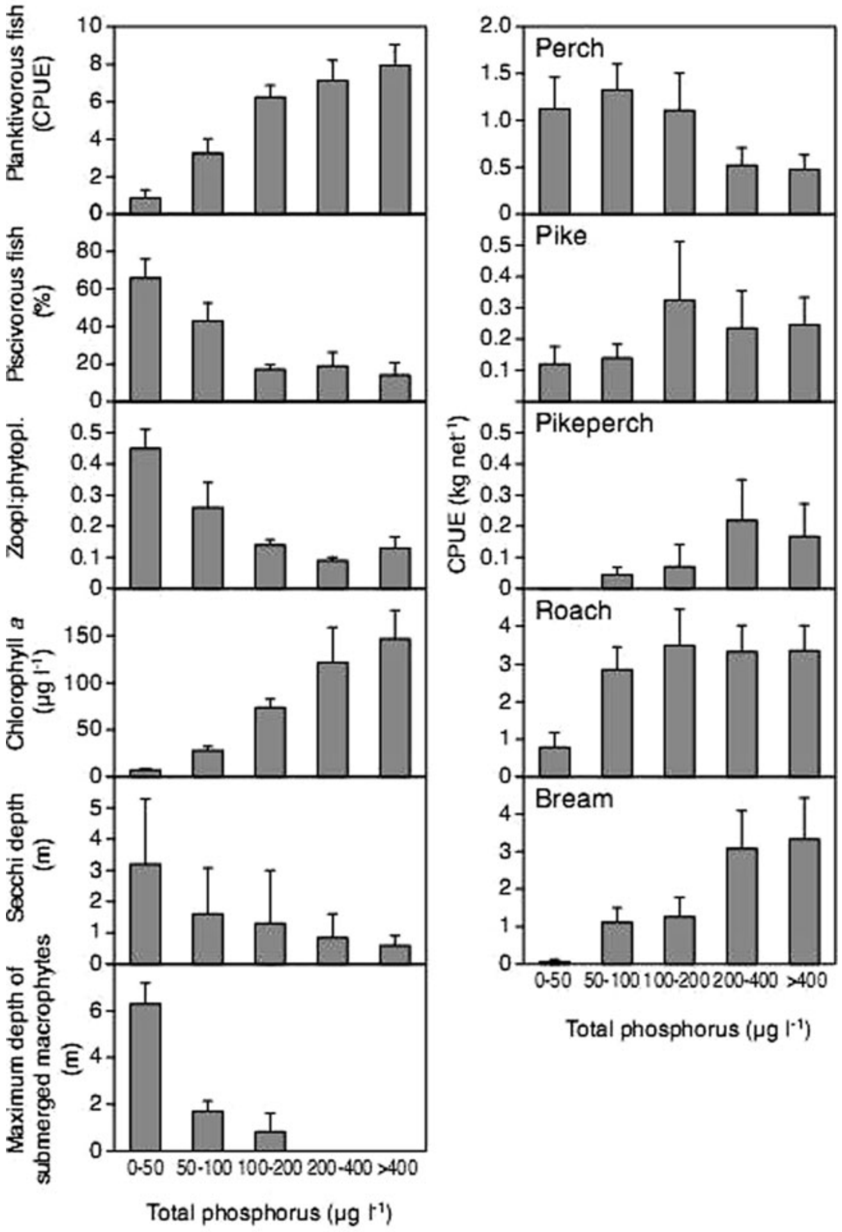

Fig. 4.2 *Left*: August biomass of zooplanktivorous fish (measured as CPUE: catches from 14 different mesh sizes 6.25–75 mm) versus summer mean lake water total phosphorus (TP). Also shown are percentage of piscivorous fish, summer mean (1 May–1 October) of zooplankton:phytoplankton biomass ratio, chlorophyll *a*, Secchi depth and maximum depth of submerged macrophytes versus the lake water TP concentrations. Mean±SD of the TP groups is shown. *Right*: Biomass (CPUE) of various quantitatively important fish species in Danish lakes versus summer mean TP. The first three species are piscivorous, while the last two are plankti-benthivorous. Taken from Jeppesen et al. (2005a) with kind permission from Science Publishers

functioning. There is a 50-fold range of light energy available from the poles to the equator, and the higher temperatures at decreasing latitudes may create greater nutrient recycling and mineralization, as well as production. With everything else equal, internal nutrient cycling in warm lakes is relatively more important and external loading relatively less important than in cold temperate lakes (Beklioglu et al. 2007, Meerhoff and Jeppesen 2009). Warm lakes tend to be more productive than similar cold lakes. In this chapter we focus on Mediterranean and (sub)tropical lakes.

4.3.1 *Functioning and Eutrophication of Mediterranean Shallow Lakes*

The Mediterranean climatic region spans over 8° of latitude (32° to 40° north and south of the equator) and has two well-separated seasons, a wet winter, during which most of the precipitation is concentrated, and an arid summer with no precipitation but high evaporation (Bolle 2003). The Mediterranean climatic region covers five major regions in the world. In contrast to north temperate shallow lakes, Mediterranean lakes are very sensitive to natural changes in hydrology, as these imply major fluctuations in water level (WLF) and hydraulic residence time, which are largely controlled by intra- and inter-annual variations in rainfall and groundwater discharge or re-charge in alternating drought and wet periods (Coops et al. 2003, Naselli-Flores and Barone 2005, Alvarez-Cobelas et al. 2005, Beklioglu et al. 2007). The high evaporative loss that is not balanced by an adequate water income results in large WLF. Moreover, high water abstraction in summer (e.g. more intensive water use mainly for agriculture, due to climate warming) enhances the natural variability caused by the regional climate variation. Many Mediterranean small lakes and ponds are temporary and often dry out in summer.

The role of WLF in shallow lakes remains to be fully elucidated. A few studies from north temperate to semi-arid/arid and (sub)tropical regions have shown that pronounced changes in water level may act directly or indirectly as catastrophic disturbances leading to state shifts (Coops et al. 2003). A high water level during the growing season may reduce the light availability for submerged plants, while a too low water level may damage plants via ice and wave action in winter and desiccation in summer. The depth profile (e.g. slope) and periodicity (e.g. range and frequency) of WLF can be very decisive for submerged plant growth (Fernandez-Alaez et al. 2004, Van Geest et al. 2005, Beklioglu et al. 2006, Tan and Beklioglu 2006). For instance, shallow lakes with low depth variation have the potential to show stronger hysteresis compared to lakes with high slope (Scheffer 1998). On the other hand, hydrology is an important element in nutrient mass balances (Downing and McCauley 1992, Downing et al. 1999). Changes in hydrology may have strong implications for major ions and nutrient dynamics, as well as lake nutrient retention, not only directly through changes in loading and hydraulic retention time, but also indirectly through alterations in trophic structure, including changes in macrophyte coverage and community (Talling and Lamolle 1998, Talling 2001, Beklioglu and Özen 2008, Jeppesen et al. 2009a, 2009b). Concentrations of the major ions and nutrients (nitrogen and phosphorus) vary with the water level and significantly increase during drought periods (Beklioglu and Özen 2008). During wet years, salinity and nutrient availability exhibit strong seasonality, whereas salinity and concentrations of TP and DIN (dissolved inorganic nitrogen: ammonium as well as nitrate) may increase substantially during dry years (Figs. 4.3 and 4.4).

During drought periods the concentrations of TP and DIN may increase markedly despite lower external nutrient loading. In-lake TP and DIN become more dependent on internal processes (evaporation and internal loading) than on the external loading (Beklioglu and Özen 2008). Droughts with low water levels and prolonged water retention time enhance the risk of eutrophication in shallow lakes in the Mediterranean region (Romo et al. 2004, 2005, Naselli-Flores and Barone 2005). An increase in nutrient concentrations and salinity has strong implications for the ecological state in that saline lakes tend to show more symptoms of eutrophication and turbid waters than freshwater lakes at similar nutrient levels (Jeppesen et al. 1994). This partly reflects that zooplankton grazing on algae is reduced due to higher fish and shrimp predation.

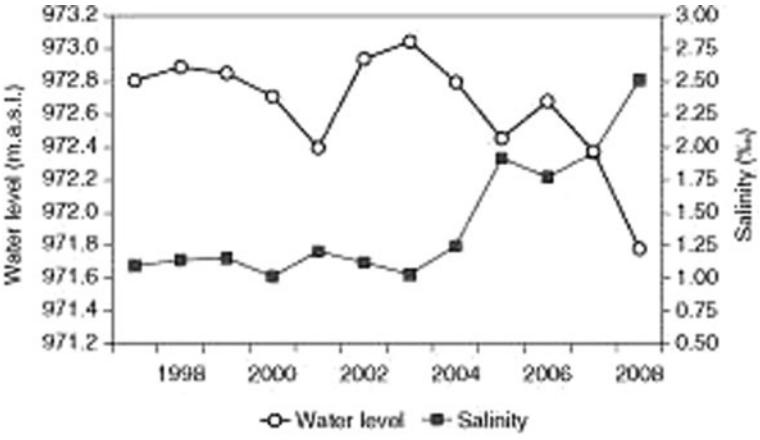

Fig. 4.3 Changes in water level (m.a.s.l) and salinity (‰) in Lake Mogan from 1997 to 2008. Water samples were collected fortnightly in spring, summer and autumn and at monthly intervals in winter, lake level being recorded daily from a fixed gauge

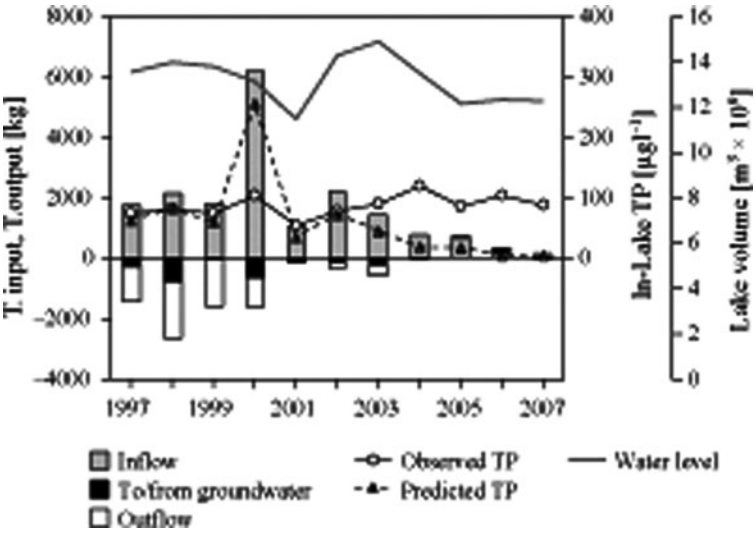

Fig. 4.4 Mass balance of total phosphorus (TP) on Lake Mogan, showing annual input, output and retention of TP as well as annual mean lake TP concentration and lake volume. The mass balance is based on water samples collected fortnightly in spring, summer and autumn and at monthly intervals in winter. The lake level was recorded daily from a fixed gauge, flow rates were measured in inlet and outlet, and ground water was determined from input calculations. Taken from Jeppesen et al. (2009) with kind permission from J Environ Qual

Moreover, salinity may limit the presence of efficient grazers such as Cladocera, implying that copepods gain a competitive advantage at salinities above 0.5‰ (Jeppesen et al. 1994). However, some large daphnids, such as *Daphnia magna* and *Daphnia mediterranea*, tolerate a rise of salt concentrations in shallow lakes. Changes in salinity also directly affect vegetation community composition (Boronat et al. 2001).

Similarly, through nutrient enrichment or eutrophication, shallow Mediterranean lakes shift rapidly from dominance of submerged macrophytes to periphyton and phytoplankton dominance, reducing the underwater light climate for plants. Macrophytes seemingly disappear when the TP concentration exceeds 0.1 mg L^{-1} (Romo et al. 2004, 2005, Naselli-Flores and Barone 2005). In some cases, however, plants may remain resilient to the shading provided by periphyton and phytoplankton when there is a substantial drop in water level (Özkan et al. 2009). In shallow Mediterranean lakes, nutrient enrichment often results in dominance and persistence of cyanobacteria (Romero et al. 2002, Romo et al. 2005). Zooplankton grazing is inefficient as small-sized species of microzooplankton dominate in eutrophic Mediterranean lakes, whereas omnivorous copepods usually dominate the zooplankton biomass in mesotrophic lakes (Beklioglu et al. 2003, Romo et al. 2004, Fernandez-Alaez et al. 2004, Romo et al. 2005). Large *Daphnia* may appear in early spring and autumn, except in temporary Mediterranean waterbodies where macrozooplankton may dominate in the almost absence of fish. Thus, control of phytoplankton by zooplankton grazing appears to be weak in most of the lakes with permanent water. The fish communities of Mediterranean shallow lakes are dominated by omnivorous and benthivorous species due to their frequent spawning and the absence of piscivores, which may otherwise exert a strong and negative effect on the zooplankton (Blanco et al. 2003, Fernandez-Alaez et al. 2004, Romo et al. 2005).

A high planktivorous fish:zooplankton biomass ratio evidences the ability of fish to control zooplankton. A study of 84 shallow European lakes from Northern Sweden to Spain showed that the ratio of fish biomass (expressed as catch per net-night in multi-mesh sized gillnets) to zooplankton biomass increased southwards, while the zooplankton:phytoplankton biomass ratio decreased in the same direction, both substantially (Moss et al. 2004, Gyllström et al. 2005). Mediterranean shallow lakes can thus remain in a turbid state, with zooplankton exerting only a weak control of algal biomass due to the far more intense fish predation here than in temperate lakes, for a given nutrient level (Beklioglu et al. 2007).

4.3.2 Functioning and Eutrophication of Subtropical and Tropical shallow Lakes

Subtropical and tropical shallow lakes hereafter named (Sub)tropical shallow lakes share some characteristics with Mediterranean lakes (Meerhoff and Jeppesen 2009). Most notably, the fish species richness and degree of omnivory are frequently high, while large numbers of fish (both juveniles and adults) are strongly associated with the submerged plants (Lazzaro 1997, Meschiatti et al. 2000, Meerhoff et al. 2003, 2007a, Teixeira-de Mello et al. 2009). Also density, but not necessarily biomass, is substantially higher (1–2 orders of magnitude) than in comparative temperate lakes, independently of trophic state. Due to high fish predation (Fig. 4.5), the zooplankton communities in tropical and subtropical lakes are frequently dominated by copepods, rotifers and small cladocerans (e.g. *Diaphanosoma*, *Ceriodaphnia* and *Bosmina*) (Jeppesen et al. 2005a, 2005b, 2007a, Meerhoff et al. 2007b). Also, the densities of littoral macroinvertebrate communities are lower compared with similar temperate lakes (Meerhoff et al. 2007a, 2007b). The trophic web appears more truncated in (sub)tropical than in temperate lakes, likely due to the omnivorous nature of most fish species and the structure of the predatory assemblages (Meerhoff et al. 2007a). In the subtropics, both low and high nutrient loaded systems are typically dominated by numerous small omnivorous fish exerting an intense predation on zooplankton, preventing the vegetation from acting as a daytime refuge to the zooplankton (Meerhoff et al. 2006). In temperate lakes plant-associated macroinvertebrates are important grazers on periphyton and sometimes also on phytoplankton in the water column of cold lakes. However, their numbers are comparatively lower in (sub)tropical lakes, likely due to the

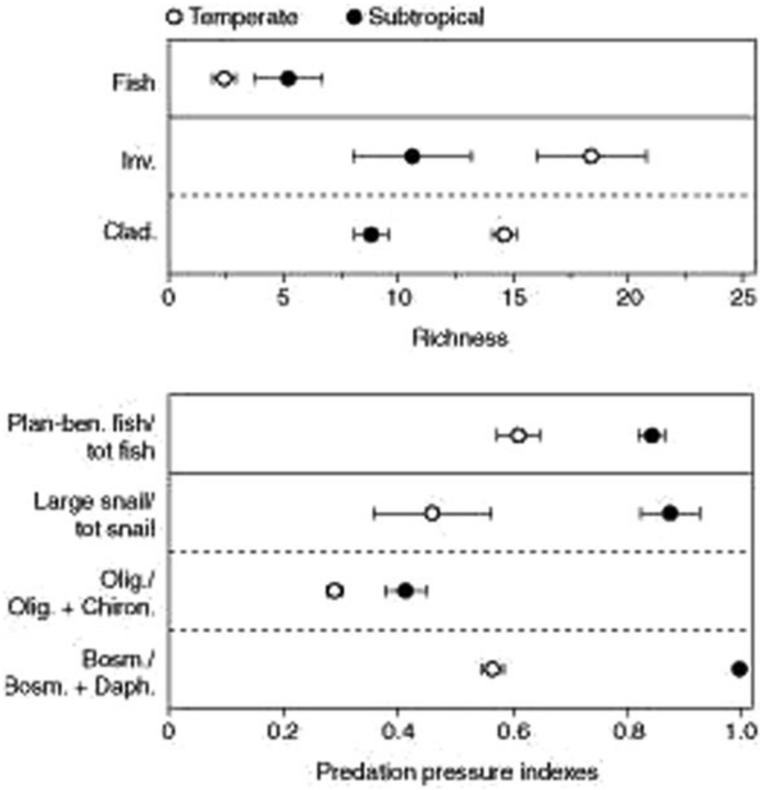

Fig. 4.5 Comparison of littoral areas in similar temperate and subtropical shallow lakes. *Above*: taxon richness of fish (species), macroinvertebrates (Inv., genera) and cladocerans (Clad., genera). *Below*: indexes of predation pressure in the littoral, considering the density of different taxonomic groups. *From top to bottom*: ratio of plankti-benthivores to total fish density, ratio of large snails to total snail density, ratio of oligochaetes to sum of oligochaetes plus chironomids and ratio of *Bosmina* to sum of *Bosmina* plus *Daphnia* density. In all cases, when the ratio is closer to 1 the predation pressure is higher. In all cases, the data represent the sample means of five lakes in each climate zone ± 1 SE, except the last index, as not even *Bosmina* was found in two out of the five lakes in Uruguay. Modified with permission from Meerhoff et al. 2007a (Glob Change Biol)

much stronger fish predation (Fig. 4.5). Periphyton biomass is not higher than in cold temperate lakes, however, probably due to the high omnivory of most fish species here.

4.3.3 Role of Vegetation in Mediterranean and (Sub)Tropical Shallow Lakes

Several critical feedback mechanisms of macrophytes for stabilizing clear water conditions have been thoroughly described for cold temperate shallow lakes. However, in the (sub)tropical and Mediterranean lakes, effects of plants on trophic interactions seem more complex (Meerhoff and Jeppesen 2009). Firstly, all forms (emergent, submerged, floating leaved, free-floating) of macrophytes can be prominent, but have quite different effects on water quality and trophic interactions. The aquatic plants are not temperature limited as in cold temperate lakes and may thus occur all year round in these systems. In particular, the effects of free-floating (e.g. *Eichhornia crassipes*, *Salvinia* spp., *Pistia stratiotes*) plants on water transparency seem much weaker than those of submerged plants, whereas a large cover of floating plants may seriously hamper biodiversity due to the strong decrease in oxygen concentrations. Furthermore, there are only weak positive impacts of plants on water clarity, and nutrient-rich lakes with high plant biomass are often turbid (Bachmann et al. 2002, Meerhoff et al. 2007a, 2007b). Provision of refuge for large-bodied grazers against fish predation is one of the main indirect stabilizing buffer mechanisms of submerged plants in shallow temperate lakes (reviewed in Burks et al. 2002). However, the predator-avoidance behaviour of zooplankton seems different in (sub)tropical lakes, and diel vertical migration appears to be more frequent than in similar temperate shallow lakes, where diel horizontal migration from open water to submerged plants, and vice versa, usually prevails (Meerhoff et al. 2006, 2007b). Furthermore, for large-bodied grazers water turbidity may serve as a disguise, especially in very eutrophic lakes.

Denitrification is thought to be a very critical stabilizing buffer mechanism of submerged plants in warm lakes for reducing the N availability for phytoplankton, as denitrification is known to increase strongly with increasing temperatures (Golterman 2000, Pinay et al. 2007, Herrman et al. 2008). Therefore, it has often been hypothesized that N may be more limiting in warm lakes than in temperate lakes (Lewis 1996, 2000, Downing et al. 1999). However, extensive growth of submerged macrophytes does not result in lower inorganic N concentrations in Mediterranean and subtropical shallow lakes. A recent comparative study of shallow lakes from the (sub)tropical to the tundra in South America also revealed inefficient denitrification in the warmer lakes?/possibly due to low quality organic matter (Kosten et al. 2009a, 2009b). This topic needs further clarification, as does the potential for (sub)tropical macrophytes to suppress phytoplankton due to release of allelochemicals. As evidenced above, shallow lakes and ponds located in warm climate regions seem much more sensitive than cold lakes to external impacts such as temperature increases (due to climate warming), water level changes (natural or anthropogenic), and nutrient loading increases (eutrophication).

4.4 Restoration of Eutrophicated Cold and Warm Shallow Lakes

During more than 50 years, eutrophication has constituted the most serious environmental threat to lakes worldwide. In recent decades, mainly in many western European countries and in North America, large efforts have been devoted to combat eutrophication by reducing the external loading of phosphorus (Jeppesen et al. 2005a, 2005b, Søndergaard et al. 2005). Accordingly, the P loading from sewage and industrial sources has declined significantly since the 1970s. Phosphorus is often in excess compared with nitrogen and is more easily controlled, but in several cases also N removal may be effective. The external loading reduction has led to improvements in many lakes. In Danish shallow lakes, for example, chlorophyll *a* has declined markedly, partly as a result of reduced nutrient concentrations, partly due to enhanced zooplankton grazing, as evidenced by an increase in the zooplankton:phytoplankton ratio and in cladoceran body size. This might be a result of stronger control by piscivores over cyprinids since the percentage of piscivorous fish has risen after a marked TP reduction (Jeppesen et al. 2005a, 2005b, 2007a, 2007b, Søndergaard et al. 2000a, b, 2005). Experience with lake recovery in warm lakes is far less advanced. Some studies conducted both in (sub)tropical and Mediterranean lakes have shown that a nutrient loading reduction may lead to improvement of the ecological state via declining algal biomass and increased water transparency (e.g. Lake Paranoa, Brazil; Lake Apopka, USA; Lake Albufera, Spain; Lake Eymir, Turkey) Coveney et al. 2005, Romo et al. 2005, Beklioglu et al. 2003, Jeppesen et al. 2007a, Beklioglu and Tan 2008).

Lakes tend not to respond immediately to a nutrient loading reduction, and even when the P loading has been sufficiently reduced, resistance may occur (Sas 1989, Marsden 1989). Although such internal loading delays recovery, a new equilibrium with respect to total phosphorus (TP) is typically reached after 10–15 years in cold temperate lakes; however, much longer response times have been observed in some lakes (Jeppesen et al. 2005b). The resistance can be "*chemical*" when the P concentrations remain very high due to P release from the nutrient pool accumulated in the sediment at high loading. This surplus pool of P is either released or permanently buried; the duration of the release depends on both the thickness of the nutrient-enriched sediment layer, the nature of the P binding site in the sediment and on the flushing rate in summer when P accumulates in the lake water (Søndergaard et al. 2000, 2002, Søndergaard, et al. 2003, Jeppesen et al. 2009a). In contrast, the response times to reduced N loading are typically <5 years, as less N is stored in the sediment (most is denitrified) (Weisner et al. 1994, Jeppesen et al. 2005a). Biological resistance also affects the internal P loading and the physico-chemical environment of the lakes (Breukelaar et al. 1994, Søndergaard et al. 2002a, 2002b). Particularly, planktivorous and benthivorous fish contribute to "*biological resistance*" (Moss et al. 1990, Scheffer et al. 1993). These fish may exert a continuous predation pressure on large-bodied grazers (e.g. *Daphnia*) and thereby reduce the grazing pressure on phytoplankton, diminishing also the number of benthic animals that stabilize and oxidize the sediment. Furthermore, benthivores (e.g. carp, bream) that stir up the sediment while feeding may help maintaining high internal P loading and high inorganic turbidity (Meijer et al. 1990, Breukelaar et al. 1994). Moreover, grazing by herbivorous waterfowl like coot (*Fulica atra*) and mute swan (*Cygnus olor*) may create resistance by delaying recolonization of submerged plants (Perrow et al. 1997b, Mitchell and Perrow 1998). The effect of waterfowl grazing on the exponential growth of plants early in the season is usually low compared with the plant growth rate; however, birds tear up more plants than they can consume, and grazing in autumn and winter on tubers, turions, etc. by migrating birds may reduce the plant density of the following year and consequently delay the shift to a clear water state (Mitchell and Perrow 1997). Several biological and physico-chemical restoration measures have been developed. Here, we will briefly present those most frequently applied to temperate shallow lakes (Søndergaard et al. 2007, Jeppesen et al. 2009a).

4.4.1 *Biological Methods*

Various biological methods, generally termed "biomanipulation", have been developed to overcome biological resistance in order to enhance the top-down control of phytoplankton and reduce resuspension (Duncan 1990, Benndorf 1995, Perrow et al. 1997, Hansson et al. 1998, Drenner and Hambright 1999, Mehner et al. 2002, Jeppesen et al. 2007c, Søndergaard et al. 2008). One of the most widely used methods is fish manipulation.

4.4.1.1 Fish Manipulation

Fish manipulation typically involves removal of plankti-benthivorous fish. Fish removal has been extensively utilized in north temperate lakes in Europe during the last 30 years. Removal of approximately 75% of the benthi-planktivorous fish stock during a 1–2 year period has been recommended to ensure recovery and to stimulate the growth of potentially piscivorous fish (Meijer et al. 1999, Hansson et al. 1998) (Fig. 4.6). An alternative or supplementary method to fish removal is stocking with piscivores, such as 0^+ pike (Prejs et al. 1994, Berg et al. 1997, Skov and Berg 1999), or with pelagic fingerlings of, for instance, zander (*Sander lucioperca*) or walleye (*Sander vitreum*) to control newly hatched plankti-benthivorous fish (Benndorf 1995, Lathrop et al. 2002). Stocking with zander or walleye should preferably be accompanied by catch and mesh size limits for fishing. Fish manipulation is often cheap compared to the physico-chemical methods described below and therefore attractive, though its long-term stability is

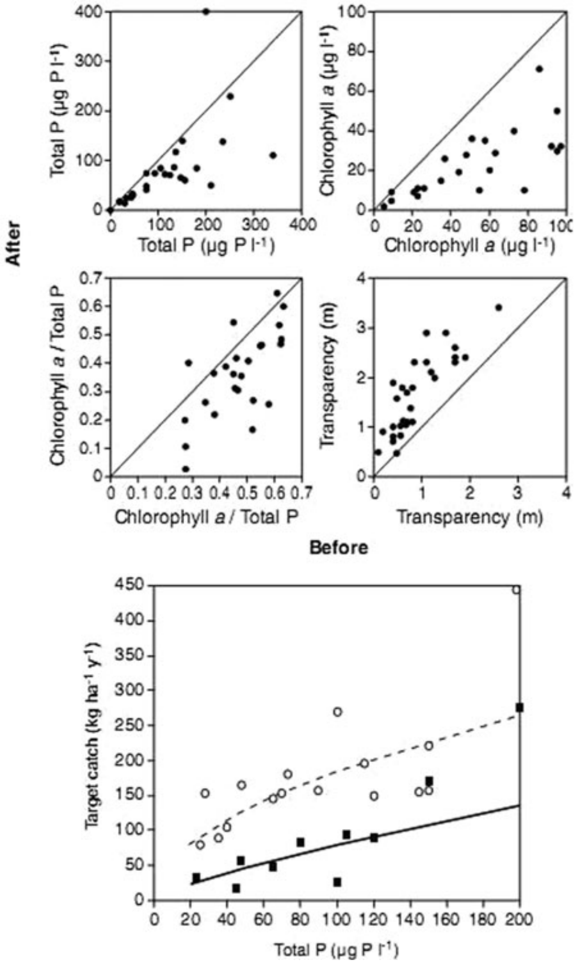

Fig. 4.6 Transparency, chlorophyll *a*, TP and chlorophyll *a*:TP ratio in lakes before and a few years after effective fish removal (*upper*). *Below*: the amount of fish removed during biomanipulation of eutrophic European lakes dominated by planktivorous and benthivorous fish. *White circles* denote the annual catch where effective fish removal led to improved water quality (increased transparency and reduced biomass of cyanobacteria), at least in the short term, or an increase in piscivorous perch. *Black dots* indicate lakes in which fish removal was too limited to have an effect on water quality or fish density. Taken from Jeppesen et al. (2009a) Lake Reservoir Management with kind permission from Elsevier

uncertain (Jeppesen and Sammalkorpi 2002). Dramatic short-term effects by fish manipulation on top-down control are generally achieved in eutrophic lakes and reservoirs, reflected in depressed phytoplankton biomass, dominance by large-sized zooplankton and higher transparency (Jeppesen et al. 2009a) (Fig. 4.6). These effects may cascade to the nutrient level, and ca. 30–50% reduction in in-lake TP concentrations has been recorded even when macrophytes were still absent (Søndergaard et al. 2008). A significant contributory factor is increased growth of microbenthic algae owing to improved light conditions at the sediment surface. More benthic algae and less sedimentation of phytoplankton due to intensified grazing, and more benthic animals due to reduced fish predation, may all result in a higher redox potential in the surface sediment, potentially reducing the P release (Søndergaard et al. 2000, 2005).

However, to date the long-term success or stability of fish manipulation is less promising, as a gradual return to a turbid water state with high abundance of zooplanktivorous fish has often been reported 5–10 years following the removal (Jeppesen and Sammalkorpi 2002, Søndergaard et al. 2008). Fish removal must therefore be repeated at regular intervals to maintain the clear water state. If fish manipulation is undertaken when the TP concentration is reduced to <0.05 mg P L^{-1} during summer, chances of obtaining long-lasting effects may be higher, at least in shallow temperate lakes. However, the threshold for manipulation success varies along a climatic gradient and depends also on the external N loading. If N loading is low, fish manipulation can have a positive effect at higher/at medium-high TP levels (Moss et al. 1996). Furthermore, low N availability contributes to achieving a stable and diverse macrophyte community (Moss et al. 1996, James et al. 2005).

It is debatable whether the fish manipulation approach used in cold temperate lakes can be used with success in warm lakes (Jeppesen et al. 2005a, 2005b). Here, high species richness and high densities, with dominance of omnivores, a few efficient predators and several cohorts, lead to higher predation on zooplankton (Lazzaro 1997, Meschiatti et al. 2000, Meerhoff et al. 2003, 2007a, 2007b, Teixeira-de Mello et al. 2009). It is therefore likely that a removal-induced reduction of the biomass of planktivorous fish will be compensated by fast adjustment of the remaining population, and the impact will therefore be of only short duration (Jeppesen et al. 2009a). Hence, it may be more difficult to provoke and maintain a trophic cascade effect in subtropical and tropical lakes than in temperate lakes. However, lakes at high altitudes that are warm in summer but covered by ice in winter often present a very different fish community structure from that of permanently warm lakes, implying that the success of top-down control might be stronger here (Beklioglu et al. 2003, Beklioglu and Tan 2008). In several regions of the world the composition and diet of many fish species are still unknown, and their potential use for biomanipulation therefore requires further studies.

4.4.1.2 Protection of Submerged Plants and Transplantation

Construction of exclosures to protect macrophytes against waterfowl and fish grazing has been used as a supplementary (or alternative) tool to fish manipulation (Søndergaard et al. 1996). In the exclosures, macrophytes can grow in a grazing-free environment where they spread seeds, turions or plant fragments, augmenting the chances of successful colonization (Søndergaard et al. 1996, Mitchell et al. 1998, Lauridsen et al. 2003). Furthermore, plants in exclosures may serve as a daytime refuge for the zooplankton against fish predation. This method seems most promising in small shallow lakes.

4.4.1.3 Combating Nuisance Plant Growth

Although re-establishment of macrophytes is the main goal of many shallow lake restoration projects, dense plant beds may occasionally be considered a nuisance in that they reduce the recreational value of the lakes (e.g. by impeding navigation and angling.). Especially, invasive species like the Eurasian milfoil, *Myriophyllum spicatum,* and the free-floating *Pistia stratiotes* or *Eichhornia crassipes* in many lakes in the United States, Europe, Asia and Africa, or the North American *Elodea canadensis* in Europe, may substantially alter lake ecosystems and become a threat to the native flora and fauna (Engel 1990, Pieterse and Murphy 1990). Manual harvesting or introduction of specialist phytophagous insects (e.g. weevils) or herbivorous grass carp (*Ctenopharyngodon idella*), water level drawdown and coverage of sediment with sheets blocking the emergence of plants from the sediments have been commonly applied. Grass carp introduction can have strong negative effects on plants and thus initiate a shift to a turbid water state; therefore, this method should be very carefully analysed before application.

4.4.2 Physico-Chemical Methods

Several physico-chemical methods have been applied, especially in northern temperate shallow lakes, to reduce the internal P loading. These include sediment removal, chemical treatment of sediment with aluminium, calcium or iron salts, and hydrological alterations. The results presented here are from Western Europe and North America (Jeppesen et al. 2009a). Sediment removal by dredging can be useful to reduce the internal phosphorus loading provided that the full thickness of the P-enriched layer is removed. However, disposal of the sediment, which may contain toxic substances, and the costly process pose the main concerns. Disturbance to wildlife and especially benthic fauna may be yet another factor to consider (Hinsman and Skelly 1987). Treatment of sediment with aluminium, iron and calcium, providing new sorption sites for phosphorus on top of the surface sediment to reduce the internal P release, can be effective, especially after stringent external nutrient control. Phosphate adsorbs readily to calcite ($CaCO_3$) and hydroxides of oxidized iron (Fe^{3+}) and aluminium (Al^{3+}). Phosphate precipitation with calcite has been used in hardwater bodies (Boers et al. 1994, Rydin et al. 2000, Reitzel et al. 2005). Phosphate adsorption onto Fe^{3+} and Al^{3+} is widely used to precipitate P in waste water treatment plants. However, Fe^{3+} and Al^{3+} form hydroxides with lesser binding capacity for P in lakes. This hydroxide formation may lead to a marked drop in lake water pH, depending on the alkalinity level. Thus, the dose of the metal ions treatment should be adjusted to lake water alkalinity to avoid a pH decrease below 6. Aluminium addition has been used for restoration in some 120 lakes in the United States and Europe, and the longevity of positive effects varies from a few to 10–15 years or longer (Jeppesen et al. 2009a). Of the three chemicals listed here, treatment with aluminium seems to be the most effective and provides a cost-efficient solution compared to sediment removal (Reitzel et al. 2005).

Water level management has also been widely used as a tool to improve the habitat for waterfowl and to promote game fishing (Scheffer 1998). There is often a unimodal relation between the water level and submerged plant growth, though this depends on the lake morphometry. Pronounced effects of the water level on submerged macrophyte growth are well known in warm lakes in arid and semi-arid regions, such as the Mediterranean basin, as described above. Changes in water level may also indirectly influence lakes by affecting fish recruitment. Complete drawdown, which has been used to control nuisance plant growth, may also facilitate a shift to clear water conditions in nutrient-rich turbid lakes, at least in the short term, as drying out may consolidate the sediment (Cooke et al. 2005). Drawdown may facilitate fish kill, thus enhancing zooplankton grazing on phytoplankton, which in turn improves water clarity. The disappearance of non-target fauna poses a clear disadvantage to be evaluated. Flushing may potentially control phytoplankton biomass, since lakes with very low hydraulic retention time tend to be clearer than expected from their nutrient levels. Flushing may also be critical for nutrient availability at prolonged residence times, especially in dry climates (see below) (Hosper 1985). Typically, less than 3–5 days are required for phytoplankton biomass to be controlled in summer, and the method is therefore not applicable at this time of the year. In contrast, winter flushing of lakes with relatively high concentrations of "overwintering" cyanobacteria may potentially be a valuable method to promote higher transparency (Jeppesen et al. 2009a). Prior to initiating lake restoration, the strategy to be employed should be carefully considered. A suggested procedure is as follows (Moss et al. 1994, 1996, Jeppesen et al. 2005a, 2005b, 2009a):

- Define the target of restoration
- Determine the annual P and N loadings from direct measurements or area coefficient models
- If the calculated mean TP concentration is higher than 0.05 mg L^{-1}, the external nutrient loading from point sources and/or diffuse catchment loading should be reduced
- If a sufficiently low external loading is achieved and the lake remains in a turbid state and if the measured summer mean TP concentration is considerably higher than the critical values calculated from models, the internal loading is probably still high. Physico-chemical methods may be considered
- If the concentration is close to 0.05 mg L^{-1} and if the fish density is high and the stock dominated by benthi-planktivorous fish; or if the chlorophyll *a*/TP ratio is high, biomanipulation can potentially be used
- Introduce and protect plants, if necessary and
- Monitor the results

4.5 Climate Change Gives Future Challenges

Eutrophication affects the entire aquatic system by altering trophic structure, biodiversity and biogeochemical cycles, as well as seasonal dynamics (Jeppesen et al. 2009a, 2010). In turn, trophic structure and ecosystem processes are water and temperature dependent. The expected increasing temperatures and changes in hydrology in the future might be expected to interact strongly with existing increased nutrient flows, and thus create new problems or lead to intensification of existing ones, as warming is likely to exacerbate some symptoms of eutrophication (Mooij et al. 2005, Blenckner et al. 2006, Jeppesen et al. 2007a). The structure and functioning of cold temperate shallow lakes are expected to become more similar to (sub)tropical shallow lakes, as the temperature increase will enhance the top-down controls of omnivorous and benthivorous fish as well as the nutrient cycling (Jeppesen et al. 2009a). Macrophyte growth might be stimulated by climate warming due to the higher temperature, and in the Mediterranean region also due to a reduced water table (Beklioglu et al. 2006). However, we might expect reduced plant diversity and species richness, an increase in warm-water exotics, such as *Lagarosiphon major*, and expansion and predominance of free-floating plants, such as lemnids (Jeppesen et al. 2009b). Moreover, even when submerged macrophytes abound, the positive effect of the plants on water clarity will likely be less pronounced at higher temperatures due to the different impact of fish communities in warmer lakes, leading to lower transparency and higher internal loading of nutrients (Meerhoff et al. 2007a, 2007b). Nuisance algae, such as cyanobacteria and most notably the N-fixing forms, are highly sensitive to increases in temperature, which may prolong their dominance (Jeppesen et al. 2009b). At the higher temperatures in Southern Europe the predicted decrease in precipitation and higher evaporation will result in less runoff and likely, as a result, lower nutrient loading to freshwaters. However, this reduction is not expected to compensate for the negative consequences of water loss that leads to a concentration of nutrients and reinforces eutrophication in aquatic ecosystems (Beklioglu et al. 2007, Beklioglu and Tan 2008). Moreover, in Southern Europe, drought and reduced discharge into inland waters will result in greater salinization, to be exacerbated by increased evaporation and greater use of water for irrigation (Zalidis et al. 2002, Beklioglu and Tan 2008, Beklioglu and Özen 2008). Adaptations in the northern temperate zone could include a more sustainable agriculture with less loss of nutrients to surface waters and, where appropriate, re-establishment of lost wetlands. In the arid Mediterranean region and likely also in (sub)tropical areas, restrictions on the human use of water are needed, particularly in irrigated agriculture, supplemented with reduced intensification of agriculture and drought control (Olesen and Bindi 2002, Alcamo et al. 2007).

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Chapter 5

Trophic State and Water Quality in the Danube Floodplain Lake (Kopački Rit Nature Park, Croatia) in Relation to Hydrological Connectivity

Vesna Peršić, Dubravka Čerba, Irella Bogut, and Janja Horvatić

Abstract The investigated floodplain lake is located in the marginal part of the Kopački Rit Nature Park, Croatia, a floodplain area of the Danube. In this chapter, we have assessed the influence of hydrological connectivity with the main channel on the trophic state and water quality in the Lake Sakadaš. The evaluation of trophic state combines a number of physical, chemical and biological parameters with the objective to select the relevant indicators for the floodplain lakes. The following environmental parameters have been considered: water temperature, nutrients, dissolved oxygen, water transparency, chlorophyll-*a* concentration, phytoplankton primary productivity, benthic and periphytic communities and laboratory experiments (nutrient enrichment bioassay). An integrative approach, which includes in situ research and laboratory experiments, indicated strong correlations of abiotic and biotic components with a hydrological regime. Also, during the biological monitoring, occurrence and spread of invasive species was noted as an indication of ecosystem disturbance (hydrological and/or anthropogenic). Therefore, our results contribute to the perception of importance of the hydrological connectivity (by inundation) on the eutrophication process in the investigated floodplain lake.

Keywords Danube floodplain · Water level fluctuations · Nutrients · Productivity · Invertebrate fauna · Invasive species

5.1 Introduction

Floodplain–river ecosystems are natural fragmented systems with periodic hydrological connections (Thoms et al. 2005). Physical and chemical conditions within the floodplain are affected by the size, position and hydrological connectivity of floodplain water bodies with the parent river (Junk et al. 1989, Junk and Furch 1993, Junk and Wantzen 2004). Hydrological connectivity greatly contributes to the exchange of nutrients and organic matter between the river and its floodplain (Tockner et al. 1999). In addition, hydrological connectivity plays a crucial role for species that require different habitat conditions during their life cycle (Amoros and Bornette 2002). Therefore, riparian floodplains are among the most valuable, but unfortunately among the most degraded ecosystems in the world.

Hydrologic fluctuations coupled with periodicity, water chemistry and community structure are the key elements affecting habitat and species diversity in river–floodplain systems (Schemel et al. 2004), such as floodplains of the Kopački Rit Nature Park in Croatia. The hydromorphology of natural nets of channels and lakes in the Kopački Rit floodplain area depends strongly on the flood dynamics of the Danube. Depending on the Danube water level, the hydrological regime within the Kopački Rit switches between two phases: the limnophase and the potamophase (Mihaljević et al. 1999, Palijan and Fuks 2006). The flood pulse concept proposed by Junk et al. (1989) emphasizes the importance of an alteration between periods of flooding and drought in the aquatic floodplain ecosystem. Although the presence of a periodic flood pulse is a key factor in maintaining a healthy river

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ecosystem, flooding has a destabilizing effect on nutrient dynamics and trophic interactions in floodplain wetlands (Junk and Wantzen 2004).

This chapter summarizes results based on the literature review, and also by using unpublished data, of long-term limnological studies in the Danube floodplain lake, Kopački Rit Nature Park (Croatia) with the objective to explore

- the influence of hydrological connectivity with the parent river on water quality and trophic state
- the importance of the appropriate key nutrients determination for possible eutrophication control
- the relationship of hydrological connectivity and the trophic status with biological parameters (bacterioplankton and sediment bacteria, phytoplankton, protozoans, sediment and weed-bed invertebrates, macrophytes) and
- possible antropogenic effect on the abundance and composition of the invertebrate fauna

5.2 Study Area

The Kopački Rit Nature Park is situated in northeastern Croatia (45°35′ N, 18°51′ S) at the confluence of the Danube and the Drava river. The protected floodplain area of the Danube is located between 1,410 and 1,383 r. km, with a surface of 177 km^2 (Fig.5.1). The complex structure of the Kopački Rit floodplains consists of big depressions permanently under water (floodplain lakes), occasionally flooded areas (sloughs) and recesses with flowing water (backwater channels). Hydrology of its natural networks of channels and lakes directly depends on the water level fluctuations of the Danube, while the Drava river regime has an indirect influence. The investigated floodplain lake Sakadaš is located in the marginal part of the Kopački Rit floodplain (some 10 km away from the main Danube flow). The Lake Sakadaš is hydrologically connected through a system of natural channels with the Danube at the east and bordered by the Drava–Danube flood-dyke from the west (Fig. 5.1). The lake has a characteristic dendritic shape with relatively steep slopes and represents the deepest water depression in the Kopački Rit Nature Park with a mean depth of 7 m, maximum depth of 11 m and surface area of 0.12 km^2. The lake was formed during the flooding of the Danube in 1926, which changed the configuration of the Kopački Rit floodplain.

The investigated area is located in a geographical region of Pannonian Plain Danube, Middle Danube, and according to hydrological records, high water levels in the Danube floodplains are characteristic for spring and the beginning of summer (Mihaljević et al. 1999).

5.3 Sediment Biota (Research Review 1997–2002)

Research of the sediment quality was conducted in the eulittoral of the Lake Sakadaš at three sites (Fig. 5.2), once a month from November 1997 till November

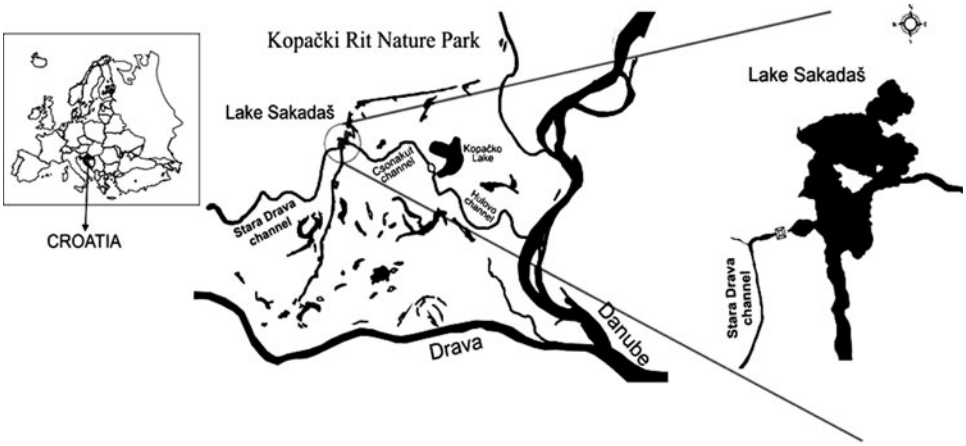

Fig. 5.1 Location of the studied Lake Sakadaš in the Kopački Rit Nature Park, Croatia

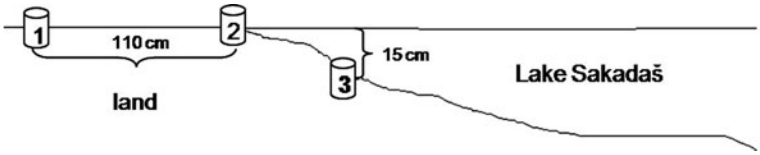

Fig. 5.2 Illustration of three sites in the eulittoral zone of the Lake Sakadaš during the research period from illustration of three sites in the eulittoral zone of the Lake Sakadaš during the research period from November 1997 till November 1998. *Mark 1* represents the land (emergent) sampling site, *mark 2* is land–water interface and *mark 3* is submerged site in the lake

1998 (Vidaković et al. 2001, Bogut and Vidaković 2002a) and in the central part of the lake from January 1998 to November 2000 (Vidaković and Bogut 2004). These research have been conducted in the scopus of Research Project: Protection of the Kopački Rit Nature Reserve Water bodies (no.122009; supported by the Ministry of Science, Education and Sports of the Republic of Croatia)

At the beginning of investigation (November–December 1997), high mortality of fish was observed (Mihaljević and Novoselić 2000), while the cause of fish kill was not determined. Due to its shallowness, the water was getting warm or was cooling down rapidly in that period – thus only indicating a thermal instability of this part of aquatic biotope. Recorded values for ammonium and nitrates indicated deterioration in the water quality, but dissolved oxygen never dropped below 6 mg L^{-1} what would point out to organic pollution. High trophic status (1998: $TSI_{SD} = 60$, $TSI_{Chl\text{-}a} = 68$, there were no data for total phosphorus; 1999: $TSI_{SD} = 59$, $TSI_{Chl\text{-}a} = 67$, $TSI_{TP} = 83$; 2000: $TSI_{SD} = 60$, $TSI_{Chl\text{-}a} = 67$, $TSI_{TP} = 87$) was recorded due to a high organic input from a large cormorant colony on the island in the middle of the lake and fallen tree leaves (Vidaković and Bogut 2004). At eulittoral zone, lowest amounts of organic matter were recorded in August during high temperatures and low water level and higher amounts of organic matter during high water level. At the submerged site, a significant correlation existed between the organic matter in the sediment and the water level of the Lake Sakadaš ($r = 0.91$, at $p < 0.001$) as well as between organic matter and the moisture content in the sediment ($r = 0.47$, $p < 0.05$) (Bogut and Vidaković 2002a). Origin of organic matter in the sediment is mostly autochthonous while allochthonous matter is brought by the Danube, shore drainage and, when the dam is lifted, from the Stara Drava channel, which collects waters from nearby villages and fields.

To assess the water quality and status of the lake's ecosystem, the changes on lower trophic levels (sediment bacteria, protozoa and meiofauna) were monitored since they are the most sensitive to disturbance. Significant increase of eutrophic bacteria abundance was recorded from November 1997 to January 1998. Ten groups of meiofauna: nematodes, rotifers, gastrotrichs, oligochaetes, copepods (incl. nauplii), cladocerans, tardigrades, turbellarians, insect larvae and acarines were found. Nematodes were the dominant group of meiofauna at all study sites and represented between 75 and 93% of total meiofauna. There was no clear temporal pattern in the meiofaunal community, although maximum for the invertebrate abundance was found in April and minimum in August (Bogut and Vidaković 2002a). The recorded changes in the meiofauna abundance, we can link with the feed source and abiotic factors. Namely, the meiofauna abundance maximum values were established at the beginning of the research when the number of bacteria and protozoa was larger and this indicated a larger quantity of feed in the sediment. The first minimum value was recorded in August as a consequence of a very high temperatures and a low water level causing a withdrawal of the meiofaunal community into the deeper sediment layers (McIntyre 1969). According to Wasilewska (1973), periodic flooding and drying-up, fluctuations of the water level, have an important influence on the meiofauna diversity and abundance. The relatively high number of meiofauna groups with a low abundance was the consequence of the physical and chemical characteristics of the study area, primarily of the sandy structure of sediment (the sediment texture consisted of sand (95%), low percentage of silt (4.2%) and clay (0.8%)). Number of recorded Nematoda species was 46, what is more than other authors found for several lakes (Prejs 1977, Schiemer 1979, Traunspurger 1996) while Andrassy (1990) found 119 species in Lake Balaton. During the winter/spring period of the eulittoral research, the maximum diversity of nematoda species was found on every site: site 1–17 species, site 2–19 species and site 3–22 species. The minimum diversity (only few species) was registered in autumn (Bogut and Vidaković 2002b).

High number of species in that period can be explained by the quality and quantity of the feed available in the sediment (i.e., bacteria and protozoa) since, according to Wetzel (2001), a higher amount of various feeds makes possible the development of a greater diversity of nematodes. The minimum diversity of nematofauna can be caused by an increase in stressful habitat conditions (Brinkhurst 1974, Warren et al. 1995), such as the increase of water level which can cause a decrease in the number and abundance of nematode species. The correlation between water level and abundance of *Thornia* sp. as well as *Tylencholaimus* sp. was recorded. Highest abundance of *Brevitobrilus stefanskii*, *Mononchus aquaticus* and *Tobrilus gracilis* occurred. According to the Shannon and Pielou indices (<2.5, respectively <0.50), all sampling sites were eutrophic during the investigated period except in November 1997, September and December 1998 (for Pielou index) (Vidaković et al. 2001, Bogut and Vidaković 2002b). *M. aquaticus* and *T. gracilis* are cosmopolites, typical for sediments of the eutrophic lakes, and *T. gracilis* is often found in reduced sediments with anoxic conditions (Schiemer et al. 1969, Pehofer 1989, Ocana and Picazo 1991). The only species that Bogut et al. (1999) and Vidaković and Bogut (2004) recorded in the central part of the lake during the summer months with anoxic conditions (<1 mg L^{-1}) in the sediment was *T. gracilis*. Eulittoral zone supported highest abundance of nematodes and consequently lowest abundance of protozoans but also as a result of less water in interstitials. Ciliata genus tolerant to anoxic conditions, *Colpidium* and *Metopus*, were the most abundant at the beginning of the research during the period of low temperatures and high water level, at what time there was a high number of bacteria recorded. The highest number of eutrophic and oligotrophic bacteria was found in the winter/spring period of 1998. In later months, lower number of eutrophic bacteria and presence of oligotrophic bacteria indicate a slow process of remineralization of organic matter (Romarenko 1985). The correlation – mutual influence by sediment biota can be seen from the relations presented below. At the submerged site, a significant correlation existed between the number of total bacteria and protozoa ($r = 0.82$, $p < 0.001$). There was a significant correlation between the sediment moisture content and the total meiofauna ($r = 0.51$, $p < 0.05$) as well as nematode abundance ($r = 0.52$, $p < 0.05$) at the land–water interface. A correlation existed between the protozoa and meiofauna ($r = 0.98$, respectively $r = 0.92$ at $p < 0.01$) and protozoa and nematode abundance ($r = 0.83$, respectively $r = 0.89$ at $p < 0.01$) for both interface and emergent site. Relationship between bacteria, protozoa and meiofauna is shown in Fig. 5.3.

The following research of the sediment in the Lake Sakadaš was conducted from March to December 2002. Anoxia and hypoxia during summer and early autumn months, characteristic for the period 1998–2000 indicates trophic state of the lake as eutrophy with tendency to hypertrophy (Table 5.1).

Anoxic conditions were recorded in contact zone in July (0.96 mg L^{-1}), August (2.79 mg L^{-1}) and September (1.48 mg L^{-1}). Amount of nutrients, as well as the number of eutrophic (CFU-E) and oligotrophic (CFU-O) bacteria, indicated the balanced dynamics and continuous processes of remineralization of organic matter as the input of organic matter in the Lake Sakadaš increased when it was brought in by water from the Danube and at the end of the vegetation season. Again, important factor was water level fluctuations influenced by irregular flooding temporal frames of the Danube and the Drava river (Vidaković et al. 2001, Bogut et al. 2003).

5.4 Hydrological Regime (2002–2005)

To determine the influence of hydrological connectivity on the trophic state and water quality in the Danube floodplain lake, surface water samples were collected at monthly intervals (March–November) during a 4-year period, 2002–2005. In the investigated section of the Danube (1,425 and 1,381 r. km), the altitude ranges from 78.08 to 80.64 m above sea level. Land configuration, small slope (0.055‰) and average current velocity of about 0.4 m s^{-1} in this section of the Danube provide adequate conditions for meandering. The Danube discharge is mainly influenced by alpine flow conditions and peaks in spring and early summer due to snowmelt. The minimal discharge of the Danube is registered in autumn and the maximal of the nival

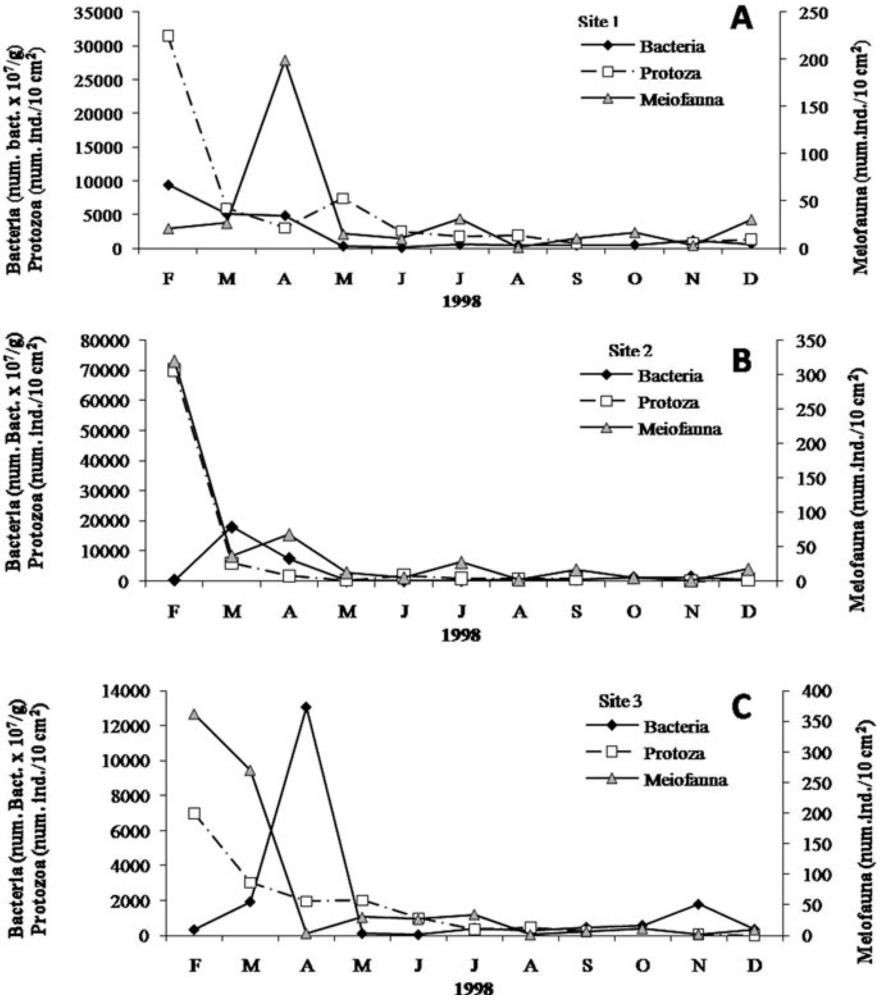

Fig. 5.3 Relationship between bacteria, protozoa and meiofauna in the eulittoral zone of the Lake Sakadaš at emergent site – site 1 (**a**), land–water interface – site 2 (**b**) and submerged site – site 3 (**c**) during the period from February till December 1998. Based on the data from Bogut (2000)

Table 5.1 Mean values of relevant water quality parameters in floodplain lake during 3-year monitoring period measured in the central part of the lake: Danube water level – Dwl (m), water depth – WD (cm), transparency or Secchi depth – SD (cm), water temperature – WT (°C), dissolved oxygen – DO (mg L^{-1}), oxygen demand – OD (mg L^{-1}), total phosphorus – TP (mg L^{-1}), electric conductivity – EC (μScm^{-1}) and the concentration of phytoplankton chlorophyll-*a* – Chl-*a* (μg L^{-1})

	1998 Mean (min–max)	1999 Mean (min–max)	2000 Mean (min–max)
Dwl (m)	1.84 (0.74–4.53)	2.69 (0.61–4.64)	2.31 (0.80–4.08)
WD (cm)	438.90 (300–600)	466.30 (280–610)	517.13 (312–800)
SD (cm)	97.60 (70–135)	118.30 (59–262)	104.50 (62–160)
WT (°C)	14.80 (4.5–20)	16.30 (7–25)	15.10 (5.5–25.5)
DO (mg L^{-1})	4.87 (0.83–13.17)	5.22 (0.91–13.29)	6.17 (2.04–11.03)
OD (mg L^{-1})	68.67 (15.66–418.72)	28.01 (3–46)	36.71 (10–82)
TP (mg L^{-1})	–	0.23 (0.23–0.23)	0.43 (0.11–1.02)
EC (μS cm^{-1})	–	584.13 (332–747)	506.14 (405–619)
Chl-*a* (μg L^{-1})	48.58 (12.18–73.94)	52.59 (6.89–86.85)	55.95 (8.25–126.21)
TSI_{SD}	60 (56–65)	59 (46–68)	60 (53–67)
TSI_{TP}	–	83 (83–83)	87 (72–104)
$TSI_{Chl\text{-}a}$	68 (55–73)	67 (50–74)	67 (51–78)

Data from Vidaković and Bogut (2004)

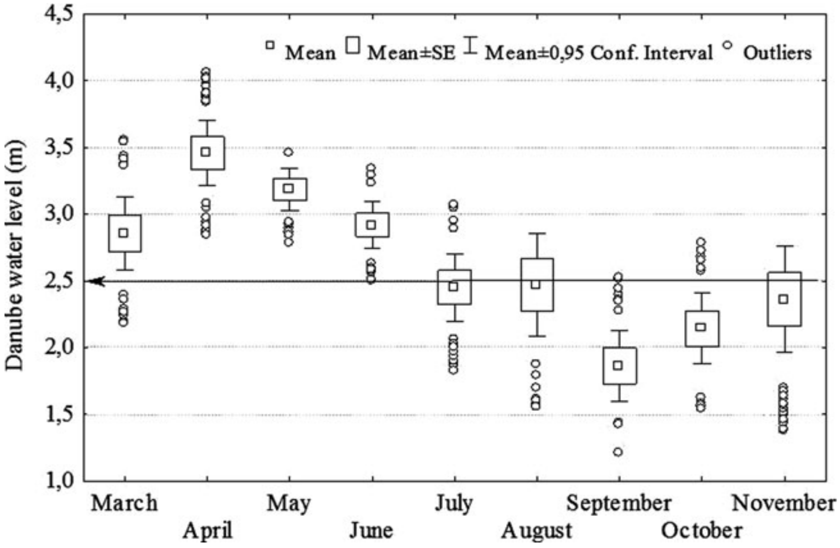

Fig. 5.4 Mean monthly water levels of the Danube near Apatin (1,404.1 r. km) in the period from March to November 2002–2005. The inflow/outflow boundary level of the Danube waters into the Danube floodplains of Kopački Rit was 2.5 m

regime in April/May, while of the glacial regime in June/July. During the investigated period, the highest river discharge was characteristic for spring and early summer (March–June) and the lowest for September and October (Fig. 5.4).

The discharge was measured only at Batina/Bezdan gauge with an average value of 2,268 m^3s^{-1}, and minimal and maximal values varying between 742 and 8,360 m^3s^{-1}. When the Danube water level (Dwl) reaches 2.5 m at Apatin gauge (1,404.1 r. km), floodwaters enter channels, which fill the lakes of the floodplain. Furthermore, when the Danube water level reaches 4 m, water in the floodplain channels starts to overflow the surrounding lowland area. Therefore, the inflow/outflow boundary level of the Danube waters into the Kopački Rit Nature Park was 2.5 m, and accordingly two hydrological phases were distinguished: the limnophase, when the floodplain remains isolated from the parent river and lentic conditions prevail, and the potamophase, a flood period characterized by more lotic conditions. During the period of investigation, flooding conditions (Danube water level >2.5 m) lasted for 195 days in 2002, 25 days in 2003, 131 days in 2004 and 145 days in 2005. An unusual flooding was recorded in the summer and early fall 2002 with maximal discharge observed in August. From March to November 2002, an average value of the Danube water level was 3.4 m (Fig. 5.5), with the

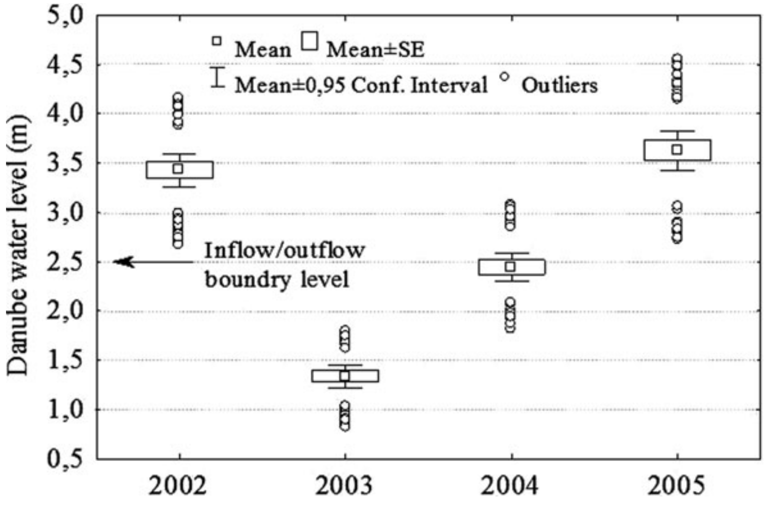

Fig. 5.5 Average values of the Danube water level (1,404.1 r. km) from March to November in 2002, 2003, 2004 and 2005. The inflow/outflow boundary level of the Danube waters into the Danube floodplains of Kopački Rit was 2.5 m

average amplitude of flood pulses 2.6 m. In this study, amplitude presents the difference between the maximal Danube water level and the specified threshold level when flooding occurs. In 2003, the Lake Sakadaš was completely isolated from the parent river (average value of the Danube water level from March to November was 1.3 m). Flood pulses in 2003 occurred (maximal Danube water level >2.5 m) in March (duration of 15 days, amplitude 1.1 m), May (duration of 4 days, amplitude 0.3 m) and in October (duration of 6 days, amplitude 0.7 m). Without natural flood disturbances, the floodplains of the Danube have a tendency towards geographical and temporal uniformity with a resulting reduction in biodiversity (Tockner et al. 1998, Ward and Tockner 2001). The flooding in 2004 started at the end of March and lasted till the beginning of August (average amplitude was 1.3 m, and total duration 131 days). On the other hand, in 2005, flooding was recorded throughout the investigating period with an average value of the Danube water level 3.6 m (average amplitude of flood pulses 3.2 m, total duration of flooding 145 days).

5.5 Water Quality Parameters

Water quality parameters for the floodplain lake during this study are summarized in Table 5.2. According to statistical analysis, fluctuations of the Danube water level significantly influenced the physical and chemical characteristics of the investigated floodplain lake waters.

In the investigated period, the increase in Danube water level was positively correlated with lake water depth ($r = 0.55$, $p < 0.001$, $N = 56$), Secchi depth ($r = 0.62$, $p < 0.001$), nitrate ($r = 0.32$, $p < 0.05$) and total nitrogen concentrations ($r = 0.37$, $p < 0.05$) while negatively correlated with ammonium-N ($r = –0.49$, $p < 0.001$), orthophosphate ($r = –0.58$, $p < 0.001$), dissolved oxygen ($r = –0.27$, $p < 0.05$) and chlorophyll-a concentrations ($r = –0.60$, $p < 0.001$). Therefore, hydrological connectivity with the parent river mainly influenced nutrient concentrations, water transparency and phytoplankton of the studied floodplain lake. An increase in the duration of hydrological connection with the parent river provided a continuous input of high NO_3–N and TN concentrations. In contrast, longer flooding conditions were reflected in reduced ammonium-N and orthophosphate concentrations. This decrease was likely due to low concentrations of the same nutrients in the source waters, as well as due to some physical processes of the biogeochemical cycles. According to Forshay and Stanley (2005), several possible biogeochemical processes can affect each fraction of organic-N, ammonium-N and nitrate-N that arrives during floods from the river into the floodplain. The average ratio of nitrate-N to ammonium-N during the studied period ranged from 1.3 in 2003 (low water level – dry conditions) to

Table 5.2 Mean ± standard deviation of concentrations for relevant water quality parameters in floodplain lake water during this study: WD (cm), SD (cm), WT (°C), pH, and average concentrations (mg L^{-1}) of total nitrogen (TN), nitrate-N (NO_3–N), ammonium-N (NH_4–N), nitrite-N (NO_2–N), total phosphorus (TP), orthophosphate (PO_4–P), dissolved oxygen (DO) and the concentration of phytoplankton chlorophyll (Chl-*a*, Chl-*b*, Chl-*c*; µg L^{-1})

	2002	2003	2004	2005
WD (cm)	5.62 ± 1.56	3.51 ± 1.27	4.49 ± 1.68	6.81 ± 1.73
SD (cm)	1.24 ± 0.50	0.73 ± 0.36	1.09 ± 0.65	1.38 ± 0.47
WT (°C)	19.0 ± 6.8	19.3 ± 7.3	16.4 ± 5.7	19.3 ± 4.7
pH	7.7 ± 0.3	7.9 ± 0.4	7.5 ± 0.5	8.0 ± 0.3
TN (mg L^{-1})	1.21 ± 0.80	0.87 ± 0.37	1.16 ± 0.35	1.54 ± 0.74
NO_3–N (mg L^{-1})	1.33 ± 0.49	0.79 ± 0.33	0.76 ± 0.40	1.46 ± 0.80
NH_4–N (mg L^{-1})	0.48 ± 0.26	0.61 ± 0.38	0.07 ± 0.07	0.02 ± 0.01
NO_2–N (mg L^{-1})	0.02 ± 0.001	0.02 ± 0.01	0.02 ± 0.01	0.02 ± 0.01
TP (mg L^{-1})	0.30 ± 0.41	0.24 ± 0.20	0.20 ± 0.38	0.40 ± 0.30
PO_4–P (mg L^{-1})	0.07 ± 0.06	0.07 ± 0.04	0.08 ± 0.13	0.04 ± 0.02
DO (mgO_2 L^{-1})	7.0 ± 3.8	11.8 ± 5.4	7.4 ± 3.8	9.8 ± 3.3
Chl-*a* (µg L^{-1})	52.24 ± 38.67	66.08 ± 42.57	68.27 ± 54.75	27.33 ± 11.36
Chl-*b* (µg L^{-1})	18.48 ± 24.15	1.18 ± 1.22	4.57 ± 5.74	4.02 ± 2.77
Chl-*c* (µg L^{-1})	4.50 ± 4.64	5.66 ± 4.10	12.00 ± 14.13	11.01 ± 5.99

94.2 in 2005 (high water level – flooding). High concentrations of ammonium-N and low ratio of NO_3–N/NH_4–N during dry conditions could be the consequence of several pathways of N cycle, i.e. algal and macrophyte uptake (and later conversion to organic-N), ammonification of organic-N and/or dissimilatory nitrate reduction to ammonium. On the other hand, high ratio of NO_3–N/NH_4–N in 2005 suggests that almost all ammonium is oxidized microbiologically to nitrate. The pattern of increasing N concentrations during flooding, followed by decreasing N concentrations as hydrologic connectivity between the river and floodplain was lost, has been observed in other river–floodplain systems i.e., Rhine floodplains (Van den Brink et al. 1994), Danube floodplains in Austria (Tockner et al. 1999) and the Missouri in the United States (Knowlton and Jones 1997). On the other hand, larger amplitude of flood pulses and greater intensity of N enrichment levels in 2002 and 2005 caused the increase of nitrogen retention in the floodplain lake.

For floodplains to be effective N sinks, denitrification must be a dominant process during flooding. However, timing and duration of flooding influences the contribution of denitrification (Forshay and Stanley 2005). Low ratio of (NH_4+NO_3+NO_2)/TN and PO_4/TP in 2004 (a year with regular exchange of flooding and isolation cycle) may indicate an increased phytoplankton control over both nutrients associated with the observed increase in algal abundance (during isolation period), or microbial-mediated denitrification (during flooding conditions).

5.5.1 Phytoplankton Chlorophyll

The highest ratio of Chl-*b*/Chl-*a* (1.07) was determined in September 2002 after the high intensity flood pulse (the largest amplitude). As a result of high intensity flooding, allochthonous Danube phytoplankton became dominant in the investigated floodplain lake. According to Mihaljević et al. (2004), Bacillariophyceae and Chlorococcales were dominant in phytoplankton biomass at that time. The lowest concentrations of phytoplankton Chl-*a* were determined in the Lake Sakadaš in 2005 during continuous hydrologic connectivity with the Danube. Therefore, under conditions of higher connectivity of the floodplain lake and its parent river, the flooding acts as a disturbance factor for the phytoplankton. This is also confirmed by the negative correlation between Danube water level and phytoplankton Chl-*a* concentrations. In addition, a high ratio of Chl-*c*/Chl-*a* in 2005 indicates that Chl-*c* containing algae were dominating phytoplankton biomass. In 2003 (dry conditions – low water period) high concentrations of phytoplankton Chl-*a* indicated increased phytoplankton biomass, while the highest average value of Chl-*a* concentration was recorded in 2004 (mean water level). According to Mihaljević et al. (2009), dry conditions in 2003 and low water conditions in 2004 were characterized by the cyanobacteria dominance. Among the factors which may stimulate the development of cyanobacteria is the low light availability. The Secchi depth in the Lake Sakadaš was quite low (0.3–1.4, average 0.7 m, Table 5.1) during dry conditions in 2003, and 0.6 m in low water conditions during August–October 2004. A very low ratio of Chl-*b*/Chl-*a* suggests that contribution of green algae during that period is negligible. Also, during that time, nitrogen limitation occurs and most of the cyanobacteria get the advantage in competition. The usual dominant phytoplankton taxa in the Lake Sakadaš during the spring time, regardless of hydrological conditions, are Bacillariophyceae. Moreover, spring flooding is a stimulating factor for the development of Bacillariophyceae (Mihaljević et al. 2009). On the other hand, prolonged flooding acts as a disturbance factor for phytoplankton biomass in the Lake Sakadaš. Therefore, hydrology is recognized as one of the major factors that regulates plankton biomass in the floodplain waters of the Lake Sakadaš.

5.5.2 Bacterial Abundance

During extremely dry conditions in 2003, eutrophic bacteria dominated in the Lake Sakadaš, while in the period of repeated inundation and isolation cycles (in 2004 and 2005) the trophic structure of bacterioplankton was dominated by oligotrophs (Palijan and Fuks 2006, Palijan et al. 2008). Flooding provides organic carbon of terrestrial origin with lower bioavailability (Hein et al. 1999, 2003), and in such conditions it supports the development of abundant oligotrophs (Palijan et al. 2008). Therefore, the floodplain lake Sakadaš is a heterotrophic ecosystem during high hydrologic connectivity with the Danube (Palijan et al. 2008).

5.6 Primary Productivity

Primary productivity in the Lake Sakadaš was measured only in 2002 from March to August. The highest values of areal NP in the Lake Sakadaš were determined in June 2002 (Fig. 5.7). Negative values of areal NP in the Lake Sakadaš were determined in July 2002 (–0.0549 g C m^{-2} d^{-1}), at the same time when maximal values of community respiration were determined (Fig. 5.6). Although the values of Chl-*a* concentration (129.58 μg L^{-1} in July 2002) indicated a favourable phytoplankton development, at the same time, all of the oxygen produced by primary producers was consumed by the present aquatic biota.

Cole et al. (2000) found that allochthonous contribution of respiration is most obvious in the lakes with negative and low NP values and P/R ratios <1. In the Lake Sakadaš, during the investigated period (March–August 2002), respiration was generally higher than production, thus indicating a tendency to heterotrophy. At the Lake Sakadaš, the specific rate of biomass production (PB) or the assimilation ratio, as a measure of gross photosynthesis rate per unit of phytoplankton chlorophyll-*a*, ranged from 1.1 mg C mg Chl-a^{-1} h^{-1} in July to 6.7 mg C mg Chl-a^{-1} h^{-1} in March 2002. The assimilation ratio has also been used as an index of phytoplankton community nutrient status (Harding 1997). Higher values of assimilation ratio indicate more enriched water and less nutrient limitation (Calijuri and Dos Santos 2001). According to Curl and Small (1965) cited in Calijuri and Dos Santos (2001), values of the assimilation ratio less than 3 indicate an environment where depletion of nutrients occur (in the Lake Sakadaš PB <3 was in June–August 2002). Values between 3 and 5 indicate a limit where nutrient deficiency might occur (May 2002) and values between 5 and 10 indicate environments that are rich in nutrients (March and April 2002). The highest values of assimilation ratio were determined in early spring, indicating favourable nutrient conditions (high nitrate-N and total phosphorus concentrations) for algal growth, but low water temperature (9–13.5°C) and flooding conditions negatively influenced on phytoplankton development.

5.7 Trophic State in Relation to Hydrological Connectivity

Trophic State Indices of Secchi depth ($\mathrm{TSI}_{\mathrm{SD}}$), total phosphorus ($\mathrm{TSI}_{\mathrm{TP}}$) and phytoplankton chlorophyll-*a* ($\mathrm{TSI}_{\mathrm{Chl}\text{-}a}$) were calculated using the equations described by Carlson (1977) and TSI values of total nitrogen ($\mathrm{TSI}_{\mathrm{TN}}$) were calculated according to Kratzer and Brezonik (1981). The lowest average value during the investigated period was determined for $\mathrm{TSI}_{\mathrm{TN}}$ (mean 55, 95% confidence interval 53–58) suggesting the eutrophic state of the floodplain lake. The highest average value was determined for $\mathrm{TSI}_{\mathrm{TP}}$ (mean value 79, 95% confidence interval 75–82) indicating the hypertrophic state of the Lake Sakadaš. The $\mathrm{TSI}_{\mathrm{Chl}\text{-}a}$ (mean 67, 95% confidence interval 64–69) and $\mathrm{TSI}_{\mathrm{SD}}$ (mean 61, 95% confidence interval 59–63) indicated eu-hypertrophic trophic state. A significantly positive correlation among trophic state parameters was determined only between $\mathrm{TSI}_{\mathrm{Chl}\text{-}a}$ and $\mathrm{TSI}_{\mathrm{SD}}$ ($r = 0.73$, $p < 0.001$). On the contrary, a negative correlation was

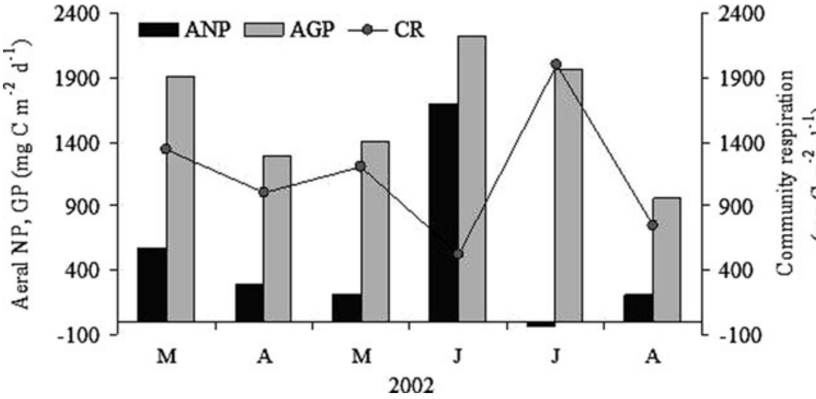

Fig. 5.6 Areal rates of planktonic primary productivity in the Lake Sakadaš during the period March–August 2002

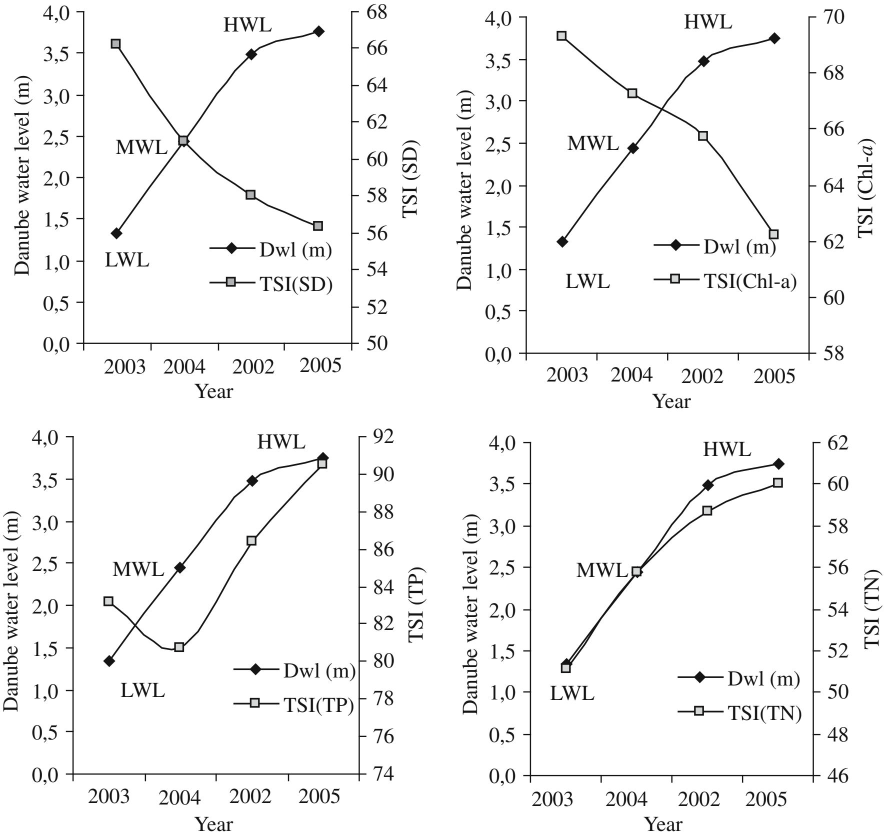

Fig. 5.7 Average values of the Danube water level (Dwl, m) in relation to the trophic state indices based on Secchi depth (TSI_{SD}), phytoplankton chlorophyll-*a* ($TSI_{Chl\text{-}a}$), total phosphorus (TSI_{TP}) and total nitrogen concentrations (TSI_{TN}). LWL is low water level observed in 2003, MWL is mean water level observed in 2004 and HWL is high water level observed in 2002 and 2005

determined between $TSI_{Chl\text{-}a}$ and TSI_{TN} ($r = -0.30$, $p = 0.02$). According to nutrients, trophic state in the Lake Sakadaš increased with increasing the Danube water level (Fig. 5.7). A significant positive correlation was determined between Dwl and TSI_{TN} ($r = 0.30$, $p = 0.02$). As for TSI_{TP}, the lowest trophic state index was observed in 2004 when the exchange of flooding and isolation cycles occurred.

In 2003, during extremely dry period when the Lake Sakadaš was completely isolated from the Danube, possible internal loading of phosphorus could have been related to low N input or/and increased sediment loading (Søndergaard et al. 2003). When the river transports suspended sediment discharges into a lake, the particulate P in the suspended sediments begins to re-equilibrate with the dissolved phosphorus in receiving waters. If the concentration of dissolved phosphorus is high, P is bound to the suspended sediments and if the concentration of dissolved phosphorus is low, P is released from the suspended sediments. According to Hein et al. (2004), in river–floodplain systems, the mean duration of connectivity was significantly related to geochemical conditions, nutrients and particle concentrations. The inverse relationship between the Danube water level and trophic state indices based on phytoplankton chlorophyll-*a* concentration and water transparency indicates that hydrological connectivity is a dominant factor. Therefore, when assessing eutrophication status of the floodplain lakes, it is important to take into consideration the influence of hydrological connectivity with the parent river. To examine the relationship between four trophic state indices, we have calculated the deviations between TSI by subtracting the TSI_{TP}, TSI_{TN} or TSI_{SD}

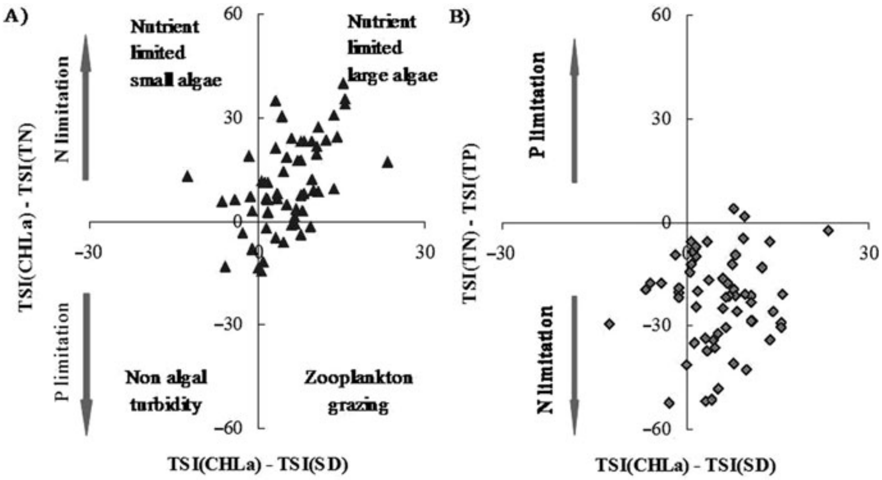

Fig. 5.8 Deviations between the Lake Sakadaš trophic state indices. Deviations $TSI_{Chl\text{-}a}–TSI_{TN}>0$ (**a**) suggest possible nitrogen limitation and larger deviations $TSI_{TN}–TSI_{TP}<0$ indicate increasing possibility of nitrogen limitation (**b**). *Points* represent monthly values during the period of investigation (2002–2005) based on the format described by Carlson (1992)

from $TSI_{Chl\text{-}a}$ (Carlson 1992), Fig. 5.8. Theoretically subtracting $TSI_{Chl\text{-}a}$ from other TSI indices should be about zero. According to Carlson (1992), zero line is related to total nitrogen to total phosphorus (TN/TP) ratios greater than 33:1 (molar ratio). In reality, predictable deviations between $TSI_{Chl\text{-}a}$ and TSI_{TN}, TSI_{TP} or TSI_{SD} can be used to assess the type and the degree of nutrient limitation (Carlson 1992, Matthews et al. 2002). The deviations between the Lake Sakadaš TSI indices suggest that phytoplankton productivity may be nitrogen limited throughout the investigated period ($TSI_{Chl\text{-}a} – TSI_{TN}>0$), Fig. 5.8.

A better interpretation would be that as the positive deviation decreases, there is a greater probability that something other than nitrogen limits algal growth increases. Slight deviations around the zero line would not truly indicate N or P limitation. A combined phosphorus and nitrogen TSI deviation was used to eliminate the effects of nitrogen as well as phosphorus limitation (Fig. 5.8b). The points that are below the zero line suggest an increasing possibility of nitrogen limitation in the investigated floodplain lake. On the other hand, $TSI_{CHL\text{-}a}$ and TSI_{SD} deviations (Fig. 5.8a and b) on the right of the *Y*-axis indicate situations where the transparency is greater than expected from the $TSI_{Chl\text{-}a}$. These deviations may occur if large particulates, such as cyanobacteria, dominate (i.e. in 2003 and 2004), and transparency is less affected by the particulates, or if zooplankton grazing removes smaller particles and leaves only large ones (Carlson 1992).

5.8 Nutrient Enrichment Bioassay

Nutrients essential for phytoplankton growth are of great importance in the successful control of eutrophication (Smith et al. 1999). Phosphorus and nitrogen are key nutrients that are managed for to improve water quality. In freshwaters, phosphorus (P) has been regarded as the most probable limiting nutrient for phytoplankton growth (Hecky and Kilham 1988, Wetzel 2001) but co-limitation by P and nitrogen (N) is not uncommon (Elser et al. 1990, Jansson et al. 1996). The limiting nutrient can be determined using different methods, for example, by measuring ambient nutrient concentration and their ratios, by determining intracellular nutrient concentration (Redfield et al. 1958, Järvinen et al. 1999) or by enrichment experiments (Elser and Kimmel 1986, Hecky and Kilham 1988, Levine and Whalen 2001, Dzialowski et al. 2005, Peršić et al. 2005, Horvatić et al. 2006, Peršić et al. 2009). Although nutrient limitation of phytoplankton is common in aquatic ecosystems, it is difficult to carry out its quantification. Evaluating the growth response of algae after nutrient additions in laboratory conditions makes the quantification of nutrient limitation possible. If a nutrient is limiting the growth rate of *Chlorella kessleri*, there is not enough of the nutrient to maximize algal growth. By experimentally adding nutrients to such a sample, we expect to see an immediate growth response of the tested algae proportional to the intensity of limitation of that particular nutrient. Therefore, a bioassay measurement is based on comparing the growth of algae between untreated (control) samples and treated (nutrient addition) samples. These results reflect algal growth responses to increased nutrient availability under specific experimental conditions. They are comparable with in situ nutrient limitation of algal growth in the absence of other limiting factors (Elser and Kimmel 1986). Therefore, the interpretation of the degree of algal growth response provides a quantifiable measure of nutrient limitation that is comparable across studies (Downing et al. 1999). The N/P ratios can be used to estimate the relative importance of the most frequent limiting nutrients. According to Dzialowski et al. (2005), TN/TP ratios reflect only the potential for nutrient limitation, while actual limitation should be determined by concentrations of available dissolved inorganic nitrogen and phosphorus. However, the ratios of inorganic N to P are not reliable indicators of nutrient availability (Dodds 2003). Therefore, according to Dodds (2006), when possible, total N/total P should be used to characterize stoichiometry of nutrient loading. High N/P of total N and P inputs indicates high availability of N relative to P (tendency to P limitation), and low N/P indicates low availability of N relative to P (tendency to N limitation), Fig. 5.9. Nitrogen limitation of surface waters in the investigated floodplain lake during low water conditions is not desirable because it favours bloom-forming cyanobacteria.

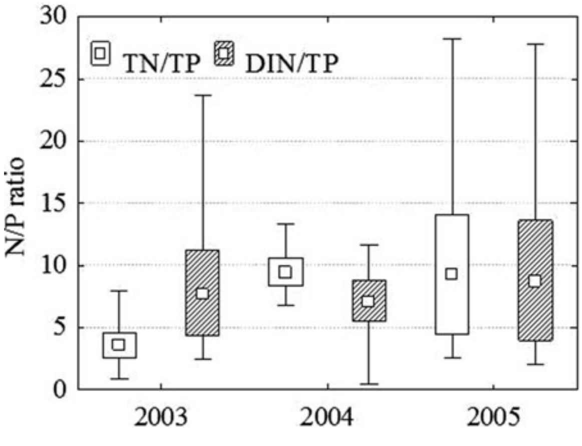

Fig. 5.9 The N/P ratio in surface waters of the Lake Sakadaš from May to October 2003 and 2004, and from March to July 2005. *Points* represent average value, *boxes* standard error and *whiskers* min–max values

Even though the N/P ratio indicates N limitation in 2003 (Fig. 5.9), and N limiting conditions correspond with observed dominance of cyanobacteria (Mihaljević et al. 2009), the nutrient addition to these samples did not have a stimulating influence on the *C. kessleri* growth rate (Fig. 5.11). Higher growth rates of *C. kessleri* in the control water samples from Lake Sakadaš were proportional to phosphorus, nitrate-N and especially high ammonium-N concentrations (Fig. 5.10), indicating nutrient saturation of *C. kessleri* growth.

A slower growth rate of *C. kessleri* in the water samples from 2004 is due to a reduction in the efficiency of a metabolic process because of restricted availability of nutrients, primarily ammonium-N and TP (Fig. 5.10). This can be acclimated condition of *C. kessleri* provided that the rate of supply of the nutrient matches the growth rate, or a result of abundant macrophyte development during 2004. Also, nutrient addition revealed significant N limitation of *C. kessleri* growth rate in 2004 (Fig. 5.11). On the other hand, in 2005, nitrate-N concentrations are higher, but at the same time, N is the limiting nutrient for the growth rate of *C. kessleri* with lower degree of N limitation than in 2004 (Fig. 5.11). Higher nitrate-N concentrations may have been the cause for the lower degree of N limitation. In fact, *C. kessleri* could have developed some N limitation as a result of very low ammonium-N concentrations. Also, intracellular P storage may have reduced bioavailability of P involved in algal growth limitation (Fig. 5.10). Nevertheless, when adding nitrogen (or phosphorus), we often stimulate an increase in the biomass. According to Liebig limitation nutrients control biomass, not the growth rates. On the other hand, Blackman limitation refers to the effect of nutrients on the growth rate. Therefore, it is possible that nutrients are limiting the rate of production, but not the total biomass.

Since the growth rate of *C. kessleri* in N-enriched samples is directly proportional to the input of limiting nutrient, we can conclude that ammonium-N and nitrate-N concentrations in surface waters of the Lake Sakadaš are good predictors of responses in bioassays. The bioassay results emphasize the importance of ammonium as a primary nitrogen source for phytoplankton in shallow wetland waters (Kinnear and Garnett 1999, Horvatić et al. 2006, Palijan and Fuks 2006). Furthermore, nutrient ratios do not always indicate the real limiting nutrient. For example, ambient

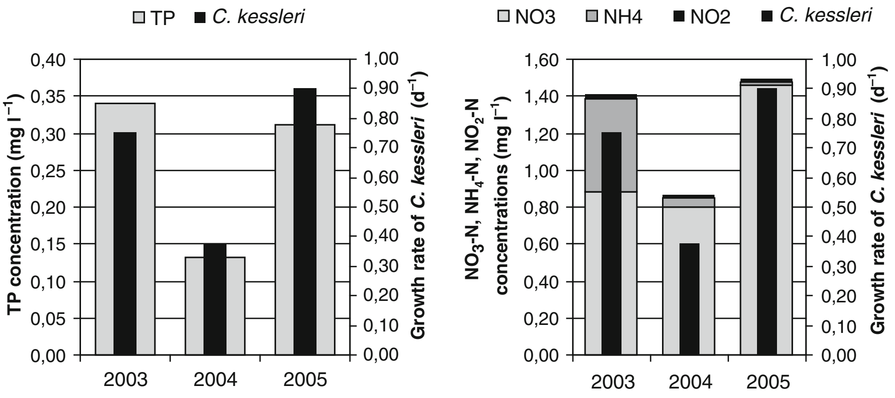

Fig. 5.10 Average values of the growth rate of *Chlorella kessleri* in comparison to the average concentrations of nutrients (total phosphorus and orthophosphate, as well as nitrate-N, ammonium-N and nitrite-N) in the sampling period May–October 2003, 2004 and March–July 2005

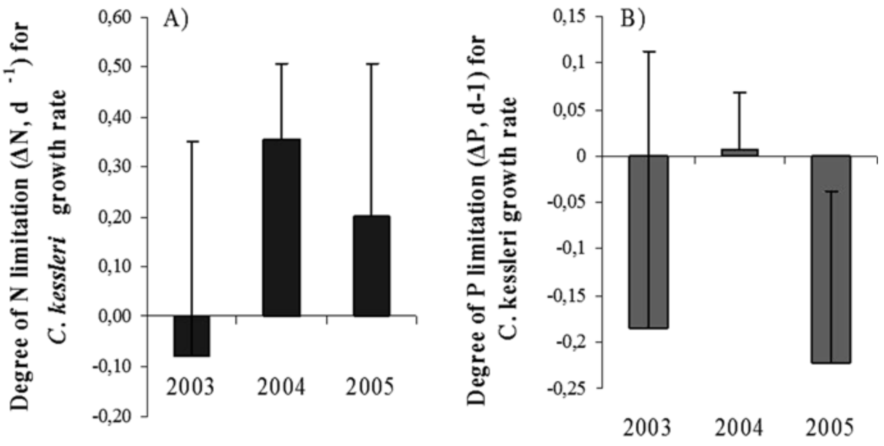

Fig. 5.11 Average and maximal values of the degree of N, P limitation in water samples of the Lake Sakadaš during the sampling period May–October 2003 and 2004, as well as March–July 2005

N/P ratios in Australian reservoirs correctly indicated if N or P is limiting in 33% of bioassay experiments (Kobayashi and Church 2003). Similarly, in the Great Salt Lake, USA, TN/TP ratios normally suggest P limitation, but bioassay experiments routinely demonstrate N limitation of phytoplankton (Wurtsbaugh 1988). Algal growth potential can be useful in projecting potential nutrient limitation and maximum biomass of experimental algae in the investigating waters (Horvatić and Lukavský 1997, Horvatić et al. 2003, Horvatić et al. 2006, Peršić et al. 2009). On the other hand, this type of experimental approach utilizes a single species which may not always be representative for the investigated waters. Increased algal growth of individual species may or may not reflect nutrient limitation of the entire phytoplankton community. For example, the cyanobacteria dominate in the community mostly at N/P ratios from 5 to 10, while high N/P ratio stimulates growth of Chlorophyta (Schindler 1977). The best N/P ratios for green algae are those>29 (Smith 1982). Also, bioassays are performed over a number of days under controlled conditions. Therefore, factors other than nutrient limitation may be minimized or magnified (Holland et al. 2004). Other factors such as temperature, light and grazing also need to be considered (Elser and Kimmel 1986, Beardall et al. 2001).

Nevertheless, nutrient addition bioassays indicate only limitation under predefined conditions (Peršić et al. 2009).

5.9 Weed-Bed Invertebrates Characterize Trophic State

In 2004, the shift from the phytoplankton to macrophyte-dominated lake was detected. Large stands of *Myriophyllum spicatum* L. and *Ceratophyllum demersum* L. appeared for the first time in June 2004 (Vidaković and Bogut 2007). The two species created adjacent but clearly separated beds with a maximum distance of half a meter. The three sites where those stands appeared were selected as the sampling sites (Fig. 5.12). The sampling started in July and lasted till September 2004 when all stands disappeared at the end of the vegetation season. Samples were taken on the weekly basis. According to Peršić and Horvatić (unpublished data) in April and June 2004, the highest heterogeneity of nitrates was recorded after the overbank flooding and there was a gradual decrease in nitrogen concentrations towards the edge of floodplain. Decrease in nitrogen concentrations could be explained with an uptake of nitrogen by macrophytes (Hamilton and Lewis 1990, Unrein 2002, Olde Venterink et al. 2003). High amounts of nitrogen brought by the flooding waters or sediment resuspension related to the inflow of flooding waters, could be what enabled macrophyte development. On the shoreline there were emerged species *Typha* sp. and *Carex* sp., *Myriophyllum spicatum* and *Ceratophyllum demersum* represented submerged macrophytes and there were also free-floating *Spirodela* sp. and *Nymphoides peltata* (S. G. Gmel.) O. Kuntze. Amphibious species *Polygonum amphibium* All. was recorded sporadically, as well as free-floating *Lemna* sp., *Trapa natans* L. and *Potamogeton gramineus* L. (Fig. 5.12). The mentioned species (with exception of *M. spicatum* and *C. demersum*) did not form large and stable stands and thus were not suitable for fauna spatial and temporal distribution research (Vidaković and Bogut 2007, Čerba et al. 2009).

Periphyton on submersed macrophytes provides a complex habitat for invertebrates and, according to Gressens (1995) supplies more nutritious substrate than sediments or vascular plant tissues. Each macrophyte species has a specific spatial structure, different length, width and surface area of stem (Kuczynska-Kippen and Nagengast 2006, Kuczynska-Kippen 2007). Cyr and Downing (1988) found no differences for plants of different leaf structure, while Bogut et al. (2007a) found that macrophytes with higher leaf complexity provide better shelter and support for weed-bed invertebrate fauna. The biomass of different aquatic plants may depend on nutrient concentration, prevailing turbidity and light condition, their species-specific ability to grow towards the water surface, and the particular growing season which directly influence macrophyte development (Pieczynska 1988). The dominant pattern suggests that our samples collected in July and beginning of August are separated from those collected in middle August and September (Bogut et al. 2010). The reason could be the change in the Danube water level since the flooding period lasted till beginning of August. Depth of the lake, Secchi depth and temperature had the most influence on weed-bed fauna collected in the first period, while in the second period concentrations of dissolved oxygen and chlorophyll-*a* were more influential. These results indicate that the important factors for the development of periphytic community at the beginning of the season are light availability, lower growth rates of phytoplankton, available nutrients and suitable lake depth. As the season continued and the community formed, available feed for the periphytic invertebrates became more important (Bogut et al. 2010). Beside macrophyte morphology, factors influencing diversity, abundance and community composition are water quality, water level and water flow (Gregg and Rose 1985, Timms 1981, Strayer et al. 2003).

C. demersum formed denser stands than *M. spicatum* and according to Sandilands and Hann (1996), denser stands of macrophytes harbour large numbers of invertebrates. As shown in Fig. 5.13.

C. demersum supported higher invertebrate abundance (at $p < 0.06$). Although similar plant morphology exists in *M. spicatum* and *C. demersum*, there are some differences. *M. spicatum* has long stalks with most of its leaves clustered towards the water surface, while *C. demersum* has denser dissected leaves evenly distributed along the entire plant length, consequently creating more surface for periphytic organisms.

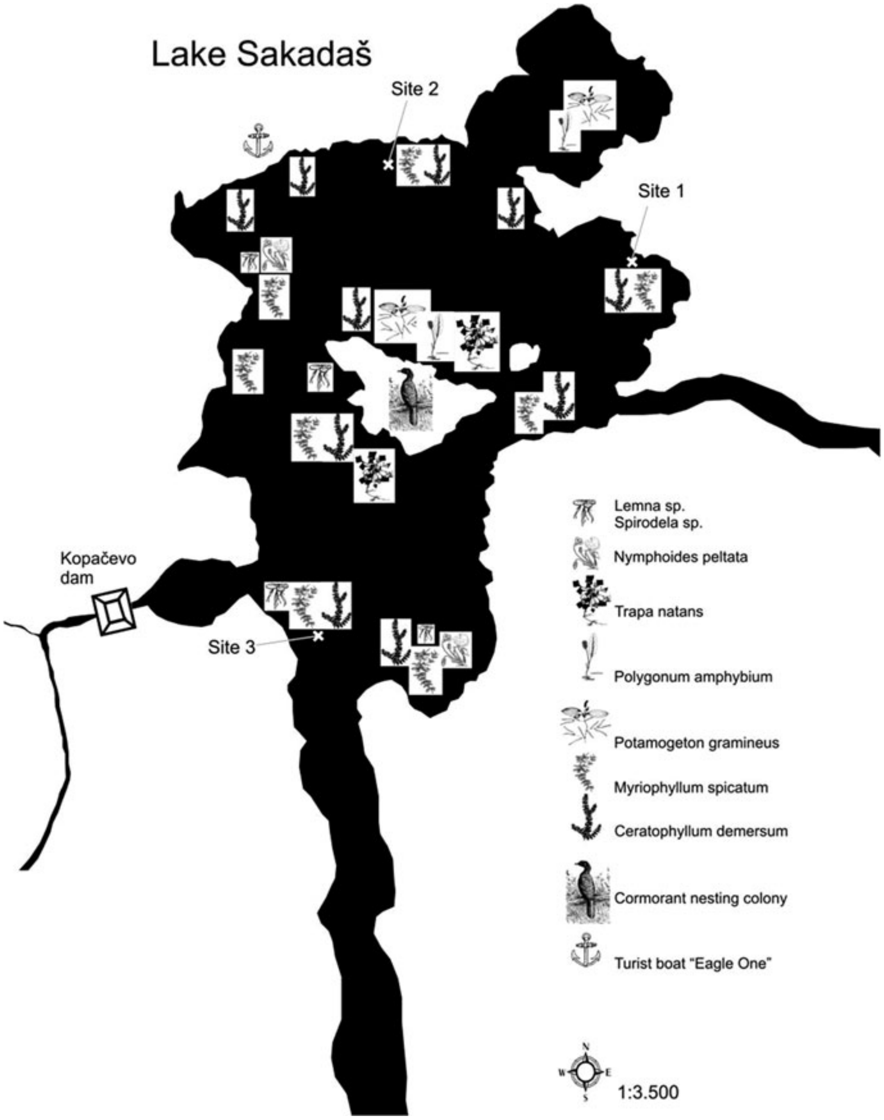

Fig. 5.12 Schematic display of the macrophyte position in the Lake Sakadaš during the vegetation season in 2004

In total, 25 taxonomic groups were recorded which belonged to the following main groups: insects larvae (chironomids, biting midges, mayflies, shoer flies, true bugs, Odonata (damselflies and dragonflies), stoneflies, caddisflies, flies, water tigres, backswimmers, curculios), crustaceans (cladocerans, ostracodes, copepodes including nauplii, mysid shrimp – *Limnomysis benedeni*), leeches, oligochaets (Naididae, mostly *Chaetogaster* sp.), nematodes, turbellarians, gastropods, acarines, hydrozoans (*Hydra* sp.) and water spider (*Argyroneta aquatica*) while in *M. spicatum* stands the total of 21 taxonomic groups were found with the difference in presence of Ephydridae, Notonecta, Curculionidae, Ostracoda and Turbellaria. The most abundant taxonomic group was chironomids in both species stands; they made in average 67% on *M. spicatum* and 79% on *C. demersum*. Chironomids associated with *C. demersum* had statistically higher abundance (at $p<0.03$) than those recorded for *M. spicatum* (Bogut et al. 2010; Čerba et al. 2009), what could be the result of the denser stands, higher amount of periphyton developed on *C. demersum* or allelopathic compounds produced by *M. spicatum*. Different types of feed were recorded in digestive systems of chironomids, several species of algae, several head cases of other chironomids and what appears to be

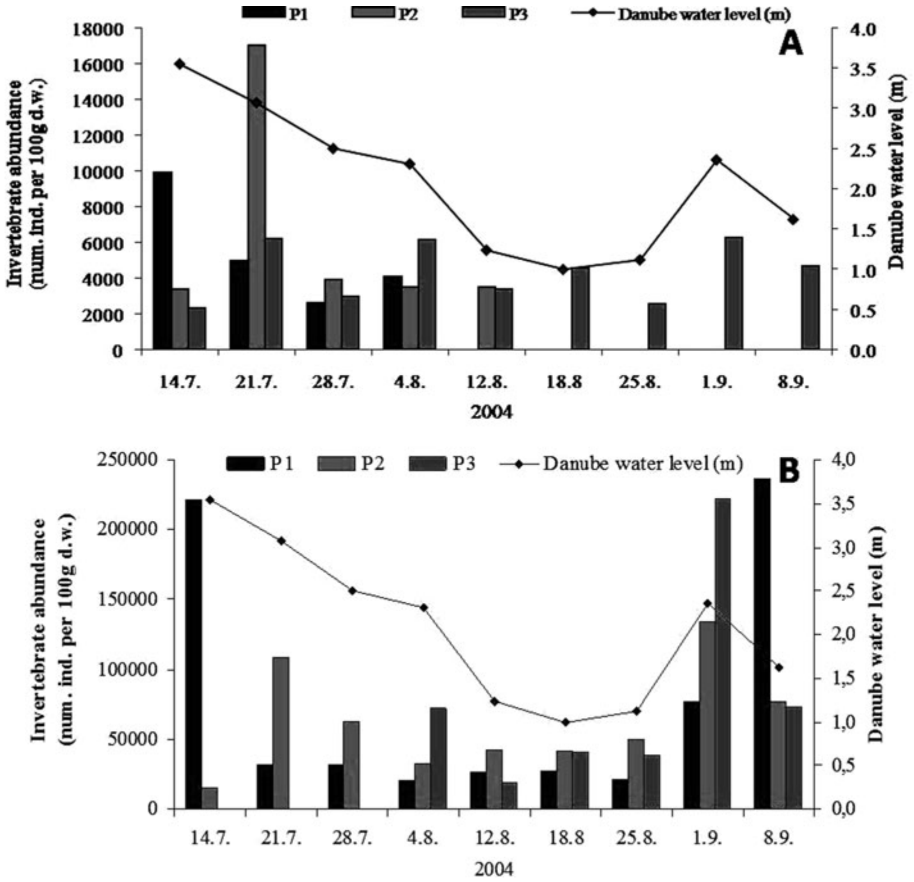

Fig. 5.13 Relation between invertebrate abundance, from both macrophyte species stands, *Myriophyllum spicatum* (**a**) and *Ceratophyllum demersum* (**a**) and *Ceratophyllum demersum* (**b**), and Danube water level during the investigation period in 2004

detritus and/or plant tissue. Chironomid larvae are very often found as the most abundant group of macrofauna in most aquatic (freshwater) habitats (Cranston 1982, Kornijów 1989, Epler 2001). The larvae have diverse diet: algae, detritus and associated microorganisms, macrophytes, wood debris, and other invertebrates of adequate size (Armitage et al. 1995, Dvorák 1996, Nazarova et al. 2004). Piezynska et al. (1999) found greater abundance of Chironomidae in highly eutrophic Lake Mikołajskie than in other less eutrophicated lakes in Poland where chironomid larvae constituted more than 95% of all macroinvertebrates. They state that Chironomidae abundance is more related to the trophy than to community composition and colonisation. Oligochaets and nematodes made a significant percentage in the weed-bed invertebrate fauna. In *M. spicatum* stands, they presented in average 10.4%(oligochaets) and 12.5% (nematodes) but at the Site III the percentages go up to 21% respectively 28%. In association with *C. demersum* nematodes had the same share as predatory larvae Zygoptera (6%) and oligochaets made 5% of total fauna. The majority of taxa recorded in association with *C. demersum* and *M. spicatum* are fairly widespread and characteristic for eutrophic waters (Van der Berg 1999; Mackie 2001).

5.10 Occurrence of Invasive Invertebrates

According to Wittman (2006), *Limnomysis benedeni*, Czerniavsky 1882 (Crustacea, Mysidacea) belongs to the original endemics of Ponto-Azov and Caspian basin and Ricciardi and Rasmussen (1998) classify it as a high-invasion risk species. This was the first record of this species in Croatian inland waters during the 2004 research. Several specimens of *L. benedeni* were found in association with both *C. demersum* and *M. spicatum* stands. The influence of this species on Kopački Rit ecosystem has not yet been perceived and still remains unknown (Bogut et al. 2007b). There are three migration corridors in Europe: northern, central and southern corridor (Bij de Vaate et al. 2003). Most probably, *L. benedeni* was brought to Croatia via Danube–Rhine route (south corridor), connected by the Main Danube Canal since Nehring (2006) states the southern corridor as the most important connection between Western Europe and Ponto-Caspian area.

The presence of this species was determined with only a few specimens but this is not surprising since, according to Lindén (2006), *M. spicatum* and *C. demersum* excrete toxic compounds which make them repellent for Mysidacea. Prof. Dr. Karl J. Wittmann found few specimens in the rivers Drava (city of Osijek) and Danube (city of Vukovar). Until 2004 *Dreissena polymorpha* was the only Ponto-Caspian species recorded in Kopački Rit when it overgrew submerged objects and branches. Based on the currently available literature, Ponto-Caspian invaders have spread throughout the European water bodies. These species can have a great ecological and economic influence on the native fauna and, therefore, it is of great importance to monitor their occurrence (Bij de Vaate 2003). The presence of the invasive Ponto-Caspian species in Kopački Rit reflects antropogenic influence and disturbance of Danube, the parent river and consequently the floodplain.

5.11 Conclusion Remarks and the Basis for Future Research

A decrease in connectivity between the floodplain lake and its parent river (i.e. absence of flooding in 2003) causes a decrease in total nitrogen concentrations. The consequence is an increase in nitrogen fixation (dominance of cyanobacteria in 2003), but without a significant effect on bioavailable nitrogen concentrations ($NO_3 + NO_2 + NH_4$). On the other hand, an increase in the duration of hydrological connectivity provides a continuous input of high NO_3–N and TN concentrations. However, longer flooding conditions reflected in reduced ammonium-N and orthophosphate concentrations, as well as reduced phytoplankton biomass. The inverse relationship between Danube water level and $TSI_{Chl\text{-}a}$ as well as TSI_{SD}, and positive relationship between Danube water level and the trophic state based on nutrients suggests that hydrological connectivity is a dominant factor that influences eutrophication process in floodplain lakes.

Therefore, when assessing eutrophication status of the floodplain lakes it is important to take into consideration the influence of hydrological connectivity with the parent river. An identification of limiting nutrients is necessary for the selection of appropriate nutrient control measures to reduce nutrients as well as to slow down eutrophication process in aquatic systems. Such control efforts must be directed towards the nutrient which is currently limiting or can be made to limit algal growth. To assess the type and the degree of nutrient limitation, we used the water quality parameters in situ (N/P ratio, predictable deviations between trophic state indices) and nutrient enrichment bioassay in the laboratory. Deviations of trophic state indices suggested nitrogen as the most probable limiting nutrient for phytoplankton productivity. Furthermore, the bioassay results emphasized the importance of ammonium as a primary nitrogen source for phytoplankton in shallow wetland waters and nitrogen as the most probable limiting nutrient for the growth rate of *C. kessleri*. Nitrogen limitation of surface waters in the investigated floodplain lake during low water level is undesirable because it favours bloom-forming cyanobacteria. Pulsed flooding, on the other hand, could be an effective way to control or to reduce eutrophication process. The regular exchange of flooding and isolation cycle indicates increased phytoplankton control over nitrogen and phosphorus forms (especially during the isolation period), or microbial-mediated denitrification (during flooding conditions).

The trophic state greatly influences taxonomic composition in the investigated floodplain lake. As a consequence, the dominance of chironomids and oligochaets, as well as low abundance of microcrustaceans in association with macrophytes is seen. As a consequence of limited hydrological connectivity, the competitive advantage of flood-tolerant traits is reduced, allowing the invasion of upland species. An overbank flooding is a major source of nutrients which are either deposited together with sediment or transformed into biomass (algae and macrophyte). High amounts of nitrogen brought by the overbank flooding waters or sediment resuspension related to the inflow of flooding waters could be what enabled macrophyte development and shift the Lake Sakadaš from phytoplankton-dominated lake to macrophyte-dominated lake. Trophic state, nutrients and hydrologic connectivity have an impact on bacteria, and phytoplankton composition and abundance, consequently influence protozoans, meiofauna and respectively macrofauna and higher trophic category organisms. The presence of anoxic and hypoxic conditions in the sediment during summer months causes an increase of tolerant species abundance.

The presence of the invasive species in Kopački Rit indicates an antropogenic influence and disturbance of the Danube and, consequently, the entire floodplain area. There is a lack of information about the influence of their presence on the floodplain ecosystem. The biological traits of these species (Bij de Vaate 2003), especially short life span and generation time as well as no specific feed preferences, make a good basis for their survival and number increase. Therefore further monitoring of the occurrence of these species in the Kopački Rit floodplain is necessary.

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Chapter 6

Mediterranean Climate and Eutrophication of Reservoirs: Limnological Skills to Improve Management

Luigi Naselli-Flores

Abstract Sicily is the largest Mediterranean island and one of the most densely populated areas in the region. To counteract the periodic oscillation in water availability and to fulfill drinking and agriculture needs, about 30 dam-reservoirs, impounding in total 750×10^6 m^3, were built in the last 60 years. However, the inherent features of the Mediterranean climate and the total lack of awareness from local authorities have contributed to a progressive increase in the trophic state of these human-made lakes. In particular, the water is stored during the winter rainy season and then it is intensively used in the dry summer. These operational procedures, in particular the time separation between filling and emptying phases, contribute to increase the internal loadings. Under this hydrological regime, reservoirs act as a sink for nutrients. Moreover, the wide water-level fluctuations which characterize these environments further contribute to enhance eutrophication processes by promoting the instability of the thermal stratification and by establishing a state of atelomixis during the summer. Consequently, most Sicilian reservoirs are affected by intense cyanobacterial blooms, which impair their water quality and cause concern due to their toxicity. Since a reduction in external loadings, which involves economic and political decisions, is far from being achieved, some procedures based on the management of the hydrological patterns of these reservoirs have been proposed. These procedures are aimed at avoiding excessive dewatering and at maintaining summer stratification. A reduction in phytoplankton biomass and a structural change in phytoplankton composition with a reduction of toxic cyanobacterial blooms have been observed as a result of the implementation of the above-mentioned procedures.

Keywords Water-level fluctuations · Phytoplankton · Cyano bacteria · Zmix/zeu · Sicily

6.1 Introduction

With a surface area of nearly 26,000 km^2, Sicily is the largest island in the Mediterranean Sea and, with about 6 millions inhabitants, one of the most densely populated areas in the region. Its territory not only is dominated by a hilly landscape (61% of the surface area) but also includes a fair amount of mountain ranges (25%). Lowlands cover about 14% of the whole area of the island. The average annual rainfall in Sicily falls between 350 and about 1,200 mm y^{-1}, with an average value of 750 mm y^{-1}, and the average annual temperature ranges from 4 to 20°C. These differences, related to the different altitude and exposition, allow the island to be divided into four zones according to the De Martonne aridity index, from semi-arid to humid (Fig. 6.1); the more humid area is located along the north coastal chain and on Mount Etna, the highest European volcano, while the more arid zones are located in western Sicily and along the southern coast of the island. Semi-arid climate characterizes most of the hilly and lowland parts of the island. In general, these areas are the most exploited for intensive agriculture. To fulfill irrigation needs and supply drinking water, 30 dam-reservoirs, impounding a total of 750×10^6 m^3, were built in the last 60 years. However,

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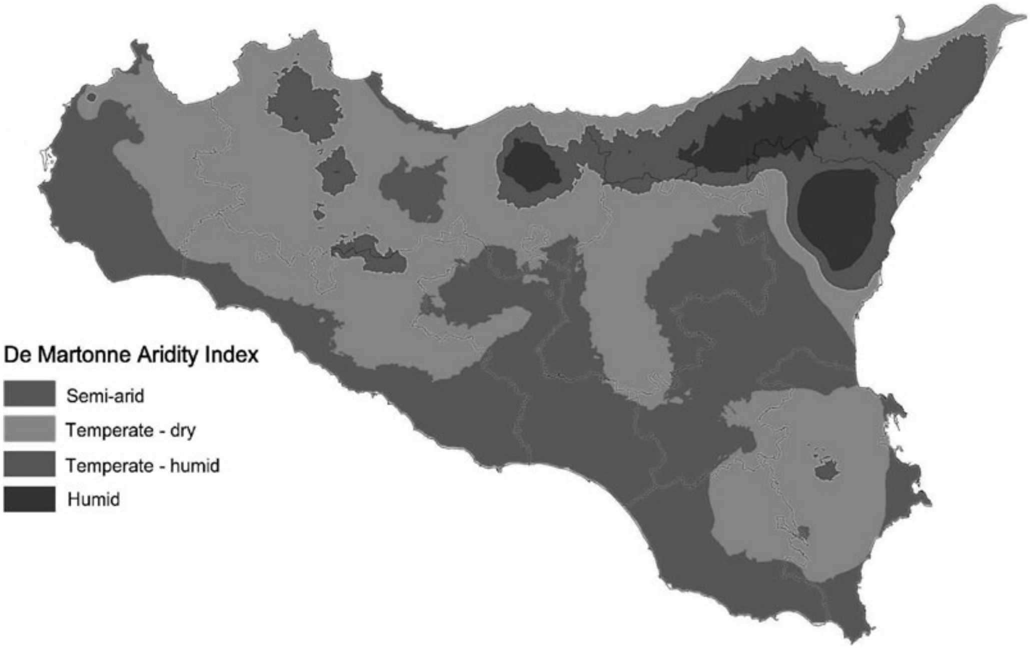

Fig. 6.1 Map of Sicilian climatic zones according to De Martonne aridity index

the lack of any plan to cut off nutrient loads to recipient water bodies (sewage diversion, buffer zones, etc.) has rapidly driven all of these aquatic ecosystems to eutrophication.

Eutrophication has undoubtedly been the most challenging global threat to the quality of freshwater resources in the last 40 years. Several European countries have successfully responded by managing human nutrient emission (low-P detergents, P precipitation at sewage treatment plants, decreased fertilizer application, erosion control, etc.). This European tendency has been strengthened in the last years by several directives issued by the European Union, the most famous being the so-called Water Framework Directive (European Parliament 2000), aimed at protecting inland water and at setting quality standards for freshwater compatible with their drinking, recreation, industry, or agriculture usage. Unfortunately, a counter tendency has been observed in Sicily and almost all of its water bodies, both natural and man-made, are still suffering the noisy effects of eutrophication such as the undesirable disturbance to the balance of organisms (structural and functional changes, a decrease in biodiversity, higher vulnerability to invasions, fish kills, etc.) and to the quality of water (cyanobacterial blooms, oxygen depletion, liberation of corrosive, and greenhouse gases, toxins, etc.). The causes of this inverse trend are complex and involve political and socioeconomic aspects (e.g., Giglioli and Swyngedouw 2008), which are beyond the purposes of this chapter. However, it has been observed that eutrophication processes proceed much faster in man-made lakes than in natural ones because the operational procedures applied to reservoirs have a negative impact and contribute to worsening water quality (Naselli-Flores 1999). Thus, current water management policies in Sicily, far from considering reservoirs as complex ecosystems, are only addressed to storing, selling, and distributing water, which often further enhance eutrophication in the Mediterranean climate.

The aim of this contribution is to summarize the results achieved in 20 years of investigations carried out in Sicily on Mediterranean reservoirs and to underline the importance of

- understanding how the Mediterranean climate influences aquatic ecosystem functioning and
- setting proper water management strategies based on ecological paradigms specifically developed for Mediterranean aquatic ecosystems (Alvarez-Cobelas et al. 2005, Naselli-Flores and Alvarez-Cobelas 2007).

6.2 Effects of the Mediterranean Climate and Insularity on Eutrophication Patterns in Sicily

Located between 32 and 42° latitude, the Mediterranean basin is placed between temperate and tropical regions and gives its name to a peculiar climate type. Beyond areas surrounding the Mediterranean Sea, this climate (also known as dry summer subtropical) prevails in much of California, in parts of West and South Australia, in southwestern South Africa, and in parts of central Chile. It is characterized by a relatively mild winter with temperatures generally above 0°C and snow occurring only rarely at sea level, but often in surrounding mountains because of wet conditions. Summer temperatures are variable depending on the region and they range from mild to very warm, depending on the distance from the open ocean, elevation, and latitude. Even in the warmest locations with a Mediterranean-type climate, however, temperatures usually do not reach the highest readings found in adjacent desert regions because of cooling from water bodies, although strong winds from inland desert regions can sometimes boost summer, or even winter, temperatures. Regarding precipitation, during summer these regions are dominated by subtropical high-pressure cells, with dry sinking air capping a surface marine layer of varying humidity and making rainfall impossible or unlikely except for occasional thunderstorms, while during winter the polar jet stream and associated periodic storms reach into the lower latitudes of the Mediterranean zones, bringing rain. As a result, these areas receive almost all of their yearly rainfall during the winter season. Schematically, these regions are thus characterized by a dry and warm summer semester, alternating with a wet and rainy winter one. The length of the dry period can vary and recurrent periods of prolonged drought with a periodicity of 11–12 years are also typical.

Due to the insularity of Sicily and according to its morphology, the river network of the region is mainly formed by torrent-like systems, whose discharge is strictly dependent on precipitation. Most of these streams are temporary and/or saline due to the presence of evaporite outcrops. All Sicilian reservoirs were realized by damming freshwater streams. Thus, their hydrology is also strictly linked to precipitation. In general, these reservoirs are characterized by wide water-level fluctuations because of the alternate periods of water storing and use: the wet winter season and the dry summer season (Naselli-Flores 1998). As a rule, reservoirs reach their maximum level in April/May at the end of the rainy season. During the filling phase, no significant water abstraction occurs. Afterward, the absence of precipitation, the rapid raise of temperature, and the evapotranspiration losses generate a strong water request for irrigation and drinking purposes, which may cause a volume contraction of more than 90% of the water stored (Naselli-Flores and Barone 1994). During this emptying phase, water abstraction is not compensated by any inflow. In this way, a summer drawdown occurs and the water level reaches its minimum value in October/November. The intensity and extent of water abstraction may differ conspicuously from year to year among the reservoirs and a variety of physical structures are precipitated by high flushing rates and abrupt fluctuations in water level. The strength of these hydrological events can be considered an important factor governing both the biotic and abiotic compartments of these ecosystems, whose effects are rather complex and involve top-down and bottom-up patterns. Figure 6.2 shows a 50-year-long time series of filling and emptying in a Sicilian reservoir. It is possible to see how climate fluctuations with a periodicity of about 11 years are superimposed over annual fluctuations due to water storage and distribution.

6.2.1 *Top-Down Effects Caused by Water-Level Fluctuations*

One of the most remarkable consequences of water-level fluctuations commonly observed in Sicilian reservoirs is the lack of a structured littoral zone. The wideness of such fluctuations does not allow the growth of aquatic macrophytes along the shores. These are stony, sandy, or muddy and do not provide refuge for fish, fry, and zooplankton. In addition, the stabilizing effect of macrophytes on sediment resuspension is totally lacking. In Sicilian reservoirs, due to Mediterranean temperature trends, fish spawning often occurs at the end of February when water temperatures are above 15°C (Naselli-Flores and Barone 1997). Moreover, precipitation may still occur in this period and as a result of surface runoff silt deposition on fish eggs may increase egg mortality and thus decrease

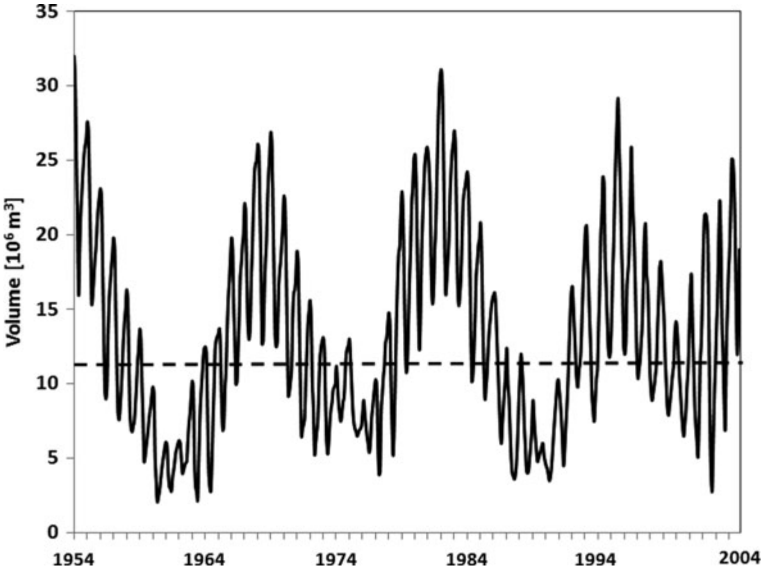

Fig. 6.2 Inter- and intra-annual variability of water stored in a Sicilian reservoir over a 50-year period. The *horizontal line* indicates the dewatering threshold above which the water body maintains its summer stratification

spawning success (Zalewski et al. 1990). It is very well known that fish fry play an important role as consumers of zooplankton in general and in particular of large cladocerans. In Sicilian reservoirs it has been observed that a decreased predation pressure on zooplankton, and the consequent increase both in numbers and in body size of these herbivores, may contribute to select large-celled or colonial inedible phytoplankton (Naselli-Flores and Barone 1997, 2000), thereby profoundly influencing the structure of the entire food web.

6.2.2 Bottom-Up Effects Caused by Water-Level Fluctuations

Water-level fluctuations, especially summer drawdown, interfere with the periodicity and stability of stratification in reservoirs. The deep location of outlets, typical of reservoirs, allows summer dewatering causing a depth decrease, but leaving the thickness of the epilimnion unchanged by dragging down the thermocline. However, if over a certain amount of water abstraction occurs, the stability of the water column is compromised, thermocline breaks, and the removal of water results in a deepening of the mixed layer (Naselli-Flores 2003). Stability of the water column can be calculated in several ways (e.g., Padisák and Reynolds 2003). However, an easy estimation of the relative water column stability can be achieved by comparing the density difference between bottom (D_b) and surface (D_s) water to the density difference between 4°C (D_4) and 5°C (D_5) of pure water:

$$\mathrm{RWCS} = \frac{D_b - D_s}{D_4 - D_5}$$

Water density can be calculated from temperature values using any of the water density calculators available on the Internet. Figure 6.3 schematically shows the relative water column stability trend in a Sicilian reservoir when the thermocline is eroded by dewatering (solid line) and when it persists throughout summer (dashed line). In the first case, at the beginning of July the thermocline weakens and rapidly reaches values below 50 at the beginning of August. This limit has been observed to be the threshold between stratified and circulating reservoirs (Naselli-Flores and Barone 2003, 2005). Ultimately, the breaking of the thermocline modifies the mixing depth–euphotic depth ratio of these water bodies. Actually, the deepening of the mixed layer may be analogized to the effect of the truncation of the euphotic depth due to an increase of phytoplankton biomass, which is generally taken to be the consequence of a shift toward a higher trophic state (Naselli-Flores 2000). The effect is to increase the value of the mixing depth–euphotic depth ratio, thus modifying the underwater light climate (Fig. 6.4).

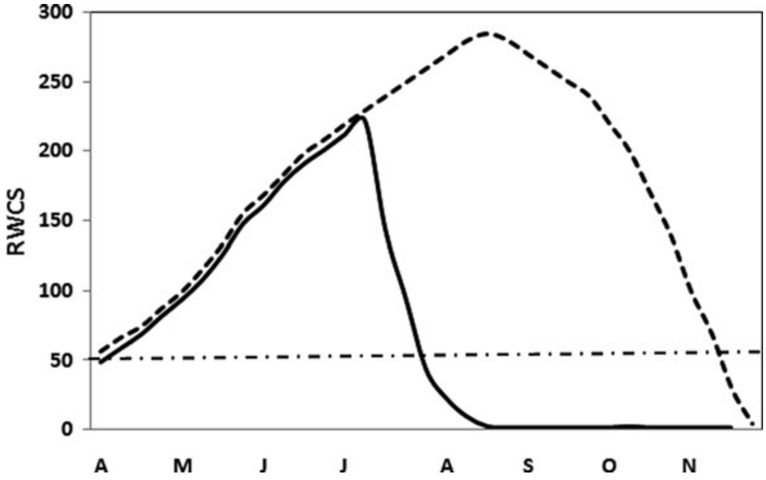

Fig. 6.3 Relative water column stability (RWCS) during the stratification period in years when the thermocline breaks down in mid-summer (*solid line*) and in years when stable stratification occurs throughout summer (*dashed line*). The *horizontal line* represents the lower value below which the water body is circulating

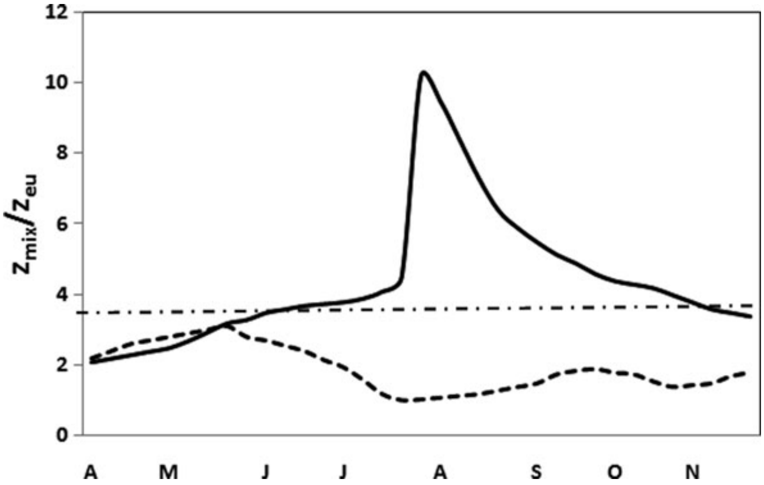

Fig. 6.4 Mixing depth–euphotic depth ratio trends in years when the thermocline breaks down in mid-summer (*solid line*) and in years when stable stratification occurs throughout summer (*dashed line*). The *horizontal line* represents the value above which the probability to have cyanobacterial blooms is higher

The structure of phytoplankton assemblages (Barone and Naselli-Flores 1994, García de Emiliani 1997, Bormans et al. 2005) and their seasonal cycles (Harris and Baxter 1996, Huszar and Reynolds 1997, Reynolds 2006) are profoundly affected by the underwater light climate and thus a change in the value z_{mix}/z_{eu} is generally followed by a modification in the structure of phytoplankton assemblage. Reynolds (1997) showed how mixing times in natural environments depend on both wind speed and epilimnion depth. Under comparable wind conditions, the variations of the mixing depth–euphotic depth ratio cause a modification to the time that a phytoplanktonic alga has to spend in the dark as it travels within the mixed layer. Thus, if an assemblage is formed by species which maximize their opportunities of growth under a certain rate of time of dark/light cycles, a change in the ratio between the mixing zone and the euphotic zone may require a special adaptation by a given species to enhance its opportunities of growth or, ultimately, its replacement. It is well known that some phytoplanktonic algae are better adapted than others to well-mixed or turbid environments (Reynolds et al. 2002). This involves physiological adaptations, which are very often accompanied by a well-defined algal shape and size that sustain and maximize them. The idea that the diverse ecological strategies adopted by phytoplankton can be related to differences in their morphology is now widely accepted and morphological responses to light and nutrient availability as well as to grazing are more and more often recognized and described (Weithoff 2003, Padisák et al. 2006, Naselli-Flores et al. 2007b, Cymbola et al. 2008). In particular, shape and size selection by light availability were found to be strongly effective in Sicilian reservoirs (Naselli-Flores and Barone 2007) as well as in reservoirs located all around the Mediterranean basin (Albay and Akcaalan 2003, Rojo et al. 2009, Caputo et al. 2008, Hoyer et al. 2009) and in subtropical/tropical climate (e.g., Figueredo and Giani 2001, O'Farrell et al. 2007, Becker et al. 2008, Zhang et al. 2008, Soares et al. 2009).

The quality of turbidity should also be considered. In those water bodies only characterized by phytoplankton turbidity, the species with a higher degree of relative entrainment, such as chlorococcalean Chlorophyceae, will be favored (Reynolds 2006) since they have a major probability of remaining and traveling in the mixed layer; conversely, in those water bodies where turbidity has a non-algal origin, flagellated algae will be more advantaged. In Sicilian reservoirs, transparency values are generally correlated to phytoplankton biomass (Naselli-Flores 1999) with chlorococcalean dominating in moderately eutrophic water bodies. This is a further consequence of the Mediterranean climate and of its lack of summer precipitation which would avoid the transportation of allochthonous matter from the catchment during the period of higher phytoplankton growth; therefore, algae entrained in the mixed layers all have the same probability of accessing light as they travel along the water column.

However, the drawdown caused by the intense summer usage, to supply water for drinking and irrigation, either prevents the formation of a thermocline or advances its breakdown in early summer. In this way, both the reservoirs in the upper part of the trophic spectrum, with high phytoplankton biomass and low transparencies, and in the mesotrophic range, with higher transparencies, are subject to the progressive increase of their mixing depth and ultimately of their z_{mix}/z_{eu}. The consequent decrease in light availability is generally accompanied by sharp changes in the structure of the phytoplankton assemblages, and the organisms which form the spring assemblage are replaced by phytoplankton species with an elongated needle shape which enables them to tolerate highly unbalanced dark/light cycles or, when the underwater environment is too dark and $z_{mix}/z_{eu}>3.5$, which can regulate their buoyancy (Fig. 6.5). In this latter group, gas-vacuolated, toxin-producing cyanobacteria are the best represented phytoplankton. These are generally hardly edible for zooplankton, thus impairing the control exerted by planktonic herbivores on primary producers.

The importance of underwater light climate, as described by the z_{mix}/z_{eu} ratio, in shaping phytoplankton assemblage in Mediterranean reservoirs has been highlighted not only in those having a insular location (Naselli-Flores 2000), but also in those located on

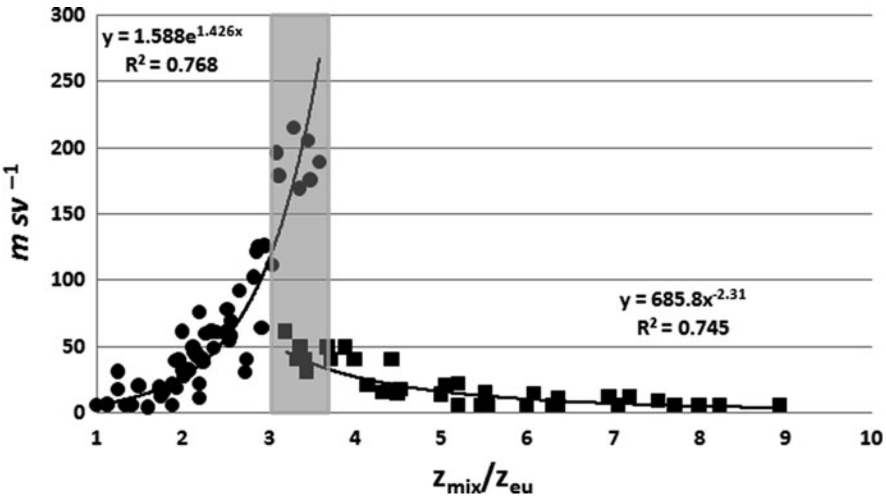

Fig. 6.5 Relationship between a morphological descriptor ($m\ sv^{-1}$: m = maximal linear dimension, s = surface, v = volume of the single species or colony) and the mixing depth–euphotic depth ratio in Sicilian reservoirs. The *shaded area* indicates z_{mix}/z_{eu} values where cyanobacteria with both elongated (high values of $m\ sv^{-1}$) and spherical or ellipsoidal shapes ($6<m\ sv^{-1}<70$) may appear depending on the successional pattern of phytoplankton assemblages and on the size of their inocula

mainland and connected to a well-developed river network (Caputo et al. 2008, Moreno-Ostos et al. 2008, Rojo et al. 2008). Moreover, $z_{\text{mix}}/z_{\text{eu}}$ thresholds very similar to those triggering cyanobacteria development and dominance in Mediterranean reservoirs have been recorded in several subtropical aquatic ecosystems in Australia (Bormans et al. 2005).

6.3 Phosphorus Loadings in Sicilian Reservoirs

Most of the nutrient loading into Sicilian reservoirs comes from untreated urban wastes and agriculture. In spite of the directives emanated by the EU, only a few urban centers have treatment plants. These are generally located along the coasts of the island and are for tourist purposes. Inland waters, even those utilized for drinking, often receive untreated water coming from urban discharge. In addition, intensive agriculture significantly contributes fertilizers as well as the wastes coming from wine and olive oil factories, ending up in freshwater. Animal farming may also significantly contribute to nutrient loadings. The quantity of phosphorus reaching an "average" Sicilian reservoir (storing 24×10^6 m^3 of water and with a surface area of 2 km^2) can be estimated in 12 t y^{-1}, which approximately correspond to 0.5 g m^{-3} y^{-1} (Calvo et al. 1993) when the reservoirs reach their maximum capacity. This value is highly variable and unpredictable depending on the amount of water reaching the water body year by year. Especially in years of drought, the dilution effect is reduced and higher loads reach the water bodies (Fig. 6.6). On a broader scale, the Mediterranean climate affects nutrient loading temporal patterns and nutrients from the catchment can reach the water bodies only during winter, when precipitation occurs. Since, according to temperature, productivity is low and no water outflow occurs from the reservoirs in this season, these environments act as a phosphorus sink; as a consequence, their internal loading is constantly increasing. The high amount of organic matter causes a very rapid oxygen depletion in the hypolimnion at the onset of stratification and the pH decreases. This reduced environmental condition promotes the release of phosphorus trapped in the sediment. The dragging down of the thermocline caused by the summer dewatering allows the "migration" of anoxic zones of reservoir bottom into the circulating part of the water body. Several hectares per week of reservoir bottom can be re-exposed to circulation and release their nutrient content to the upper layers, as demonstrated by the frequent pulses of reactive phosphorus recorded in Sicilian reservoirs in summer, when there is not any water input from the catchment (Naselli-Flores 1999, Naselli-Flores and Barone 2005). This pattern can be worsened when thermocline breaks. In fact, due to the high summer temperature, reservoirs may show an atelomictic behavior. This daily circulation pattern further contributes to nutrient release from the reservoir bottom, sustaining phytoplankton growth throughout the summer. An attempt to evaluate the phosphorus balance in Sicilian reservoirs has shown that the amount of phosphorus entering the reservoirs during the rainy season is higher than that flushed out during the summer emptying phase (Naselli-Flores and Barone 2005), causing the progressive increase of internal loadings.

Thus, the anticipated breaking of the thermocline does contribute to make these reservoirs optically deep as well as to supply them with nutrients which sustain phytoplankton biomass accumulation. However, biomass accumulation is inversely related to water transparency and euphotic depth, further contributing to high values of the mixing zone–euphotic zone ratio. Ultimately, nutrient supply promotes secondary modifications in the physical environment through the decrease of light availability, again selecting those species better adapted to darker environments.

6.4 Consequences of Eutrophication on Public Health

The combined factors of a Mediterranean climate, insularity and limitation of the river network, operational procedures applied to water storage and distribution management, all contribute to modify the structure of phytoplankton assemblages in Sicilian reservoirs. As shown, all of these factors push toward the development, and often the dominance, of both nitrogen-fixing (*Anabaena*, *Aphanizomenon*, *Cylindrospermopsis*, etc.) or non-fixing (*Planktothrix*, *Microcystis*, etc.) cyanobacteria. Most of these genera were found to develop

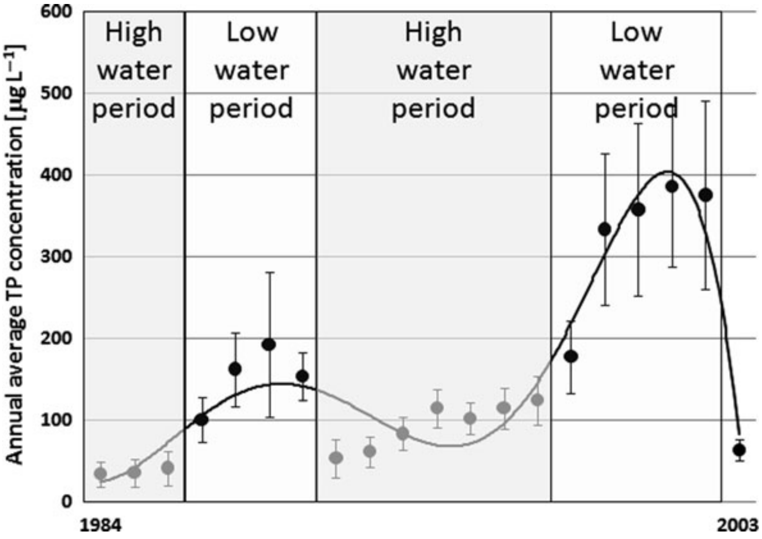

Fig. 6.6 Annual trend of average total phosphorus concentrations over two decades in a Sicilian reservoir

toxic strains in Sicilian reservoirs (e.g., *Microcystis aeruginosa*, *M. panniformis*, *Planktothrix rubescens*, *Aphanizomenon ovalisporum*) and, thus, large blooms may directly harm both other aquatic organisms and humans (Naselli-Flores et al. 2007). The reasons for the evolutive success of cyanobacteria are widely discussed in scientific literature. However, as summarized by Istvánovics (2009) each bloom-forming species possesses a certain combination of these traits, aimed at ensuring the competitive success under environmental conditions that are often stressful for other phytoplankton. Schematically, these traits involve the following:

Efficient light harvesting: Some bloom-forming cyanobacteria (e.g., *P. rubescens*, *Cylindrospermopsis raciborskii*) show the lowest light saturation values of both photosynthesis and growth and their light requirement is in general much lower than that of the eukaryotic algae. In addition, their gas vesicle regulation favors their placement in the light gradient along the water column, especially during calm conditions and/or stratification.

Nutrient uptake: Bloom-forming cyanobacteria are extremely efficient in exploiting nutrient reserves that are not readily available, for most other algae. Buoyancy regulation helps them to exploit vertical nutrient gradients as well as their fast maximum rate of P uptake and their high affinity for NH_4 (along with the ability to fix nitrogen). Carbon concentration mechanisms are highly efficient both for CO_2 and $HCO^-{}_3$, allowing them to be competitive within a wide pH range. Moreover, the so-called luxury uptake and their large (compared to other bacteria) size enable them to store C, N, and P in excess and consequently to buffer environmental fluctuations in the nutrient supply.

Low biomass loss: Through buoyancy regulation, healthy cyanobacteria avoid sinking loss even under calm conditions. Large size and morphology (large colonies, filaments) reduce zooplankton grazing to negligible levels. Furthermore, their persistence in the aquatic ecosystem is guaranteed by the production of resting stages (as akinetes or non gas-vacuolated overwintering stages into the sediments).

In addition to these points, cyanobacteria, although their forms are limited to relatively simple shapes, show a high phenotypical variability with respect to length of trichomes and their coiling or dimension and shape of the colony, which allow them to persist under fluctuating environmental conditions once that dominance has been established (Naselli-Flores and Barone 2003, 2007, Naselli-Flores et al. 2007). In particular, *Microcystis* morphotypes are able to exploit and store nutrients very efficiently and their buoyancy regulation and variability in colony shape and size offer additional advantages in an underwater climate with highly fluctuating light levels. In Sicilian reservoirs, one of the most striking consequences of nutrient accumulation in the sediments and their summer release caused by particular hydrological patterns was the establishment of very heavy cyanobacterial blooms beginning at the turn of the century. From 1979 onward, the relative importance of *Microcystis* and other planktonic cyanobacteria gradually increased at the expense of green algae previously dominating those reservoirs (Barone and Naselli-Flores 1994, Naselli-Flores and Barone 2003, 2007). From 2000 onward *M. aeruginosa* made up 50–100% of the total phytoplankton biovolume in 70% of the water stored in Sicilian reservoirs. Because of Mediterranean climate characteristics, longer growing periods are achieved and *Microcystis* usually starts its growing season at the end of February continuing as late as October/November. Fresh weight biomasses higher than 1 g L^{-1} of *Microcystis* spp. were measured in these environments and microcystin concentrations, expressed as LR equivalents, higher than 2.7 mg L^{-1} (Naselli-Flores et al. 2007). These values, 27 times higher than the maximum recommended by the WHO (Chorus and Bartram 1999), cause concern and may seriously endanger public health.

At the beginning of its growing season, *Microcystis* recruitment from the sediments in the shallow parts of the reservoirs is promoted by high light levels (e.g., Reynolds 2006). The maximum growth rate observed in Sicilian reservoirs is 0.3 day^{-1}. In spite of this rather low value, populations can reach densities above 10^{10} cell L^{-1}. These values largely depend on the size of the inocula present in the sediments and on dewatering. In fact, the proceeding of the latter allows new portions of sediments to be illuminated and thus favors recruitment. Dewatering thus not only influences light availability and nutrient dynamics in Sicilian reservoirs, making them more suitable for cyanobacteria growth, but also sustains and favors their biological cycle. In addition, toxin production rates of *Microcystis* spp. were found to be linearly correlated with the growth rates in Sicilian reservoirs, thus suggesting that the microcystins produced by cyanobacteria may improve their photosynthetic performance. This is consistent with the results achieved by Sedmak and Elersek (2006) who suggested that the biological activity of microcystins toward the morphological traits (increased aggregation and cell size) and light-harvesting complex (overproduction of photosynthetic pigments) of photobionts can be interpreted as a signal announcing the worsening of light conditions due to the massive proliferation of cyanobacteria. At the end, the reduced light availability typical of Sicilian reservoirs, by favoring cyanobacteria, may harm public health because of the enhanced toxin production of these organisms.

6.5 Eco-friendly Procedures to Control Eutrophication and Their Effectiveness

Although a huge amount of literature has been produced in the last 40 years on reservoir management (e.g., Cooke et al. 2005, Jørgensen et al. 2005, Rast 2007 and references therein), regional peculiarities may require the adjustment of general managing procedures when these have to be applied to a local context. In particular, climatic, morphological, and hydrological patterns of given water body are important features which must be carefully evaluated to ensure the effectiveness of any management plan. As formerly discussed, in Sicily the operational procedures to which reservoirs undergo, in relation to Mediterranean climate, cause the breakdown of the thermal stratification and the release of nutrient-rich hypolimnetic waters into the upper layers. In this way, phytoplankton growth is sustained throughout the summer and harmful algal blooms may be favored. As shown in Naselli-Flores and Barone (2005), to test the hypothesis that too widely fluctuating water levels may impair the ecological functioning of the reservoirs by pushing the thermocline to vanish, the management board of a Sicilian irrigation reservoir, Lake Arancio, was asked to slightly modify their water storage and distribution operational procedures. In particular, this reservoir was refilled at the end of summer by using the waters coming from another man-made lake temporarily out of use. This refilling generally took place when Lake Arancio had reached its minimum holding and the spilling out of water was thus impossible. All of the implications of such a huge dewatering were particularly evident in Lake Arancio, which at the end of the irrigation season had the appearance of a dense pea soup because of *Microcystis* growth and concentration. In order to avoid the anticipate breaking of the thermocline, the proposed modification of the operational procedures consisted in the refilling of the reservoir throughout the entire irrigation period instead of only at its end. This could be realized using the same amount of water (about 13×10^6 m^3) generally pumped from the other reservoir. The water-level fluctuations in Lake Arancio were thus much less pronounced and the thermocline was maintained throughout the summer period through mid-November. Moreover, a minor area of reservoir bottom was influenced by circulation and hypolimnetic nutrient reserve was more confined and less available to permit the phytoplankton biomass increase.

The most striking consequence of this procedure was the strong reduction of *Microcystis* bloom and the re-establishment of a phytoplankton assemblage dominated by green algae. This was attributed to the lowered phosphorus inputs from the sediment and to the reduced nutrients (mainly P and N-NH_4) availability in the epilimnetic layers due to algal consumption. Although high phosphorus concentrations were still present in the hypolimnion, the persistence of a stable stratification limited algal growth and significantly reduced biomass accumulation. This allowed the $z_{\mathrm{mix}}/z_{\mathrm{eu}}$ ratio to maintain values well below 3.5 throughout the summer and thus opened the competitive arena to algae (e.g., fast growing chlorococcalean greens) other than cyanobacteria. A decreased dewatering may also limit *Microcystis* recruitment from the sediment. Maintaining thermal stratification as a procedure to counteract cyanobacteria blooms contrasts what had been shown in lakes located at higher latitudes, where promoting circulation was successfully applied to reduce cyanobacteria growth (Visser et al. 1996). Actually, *Microcystis* does not easily tolerate prolonged darkness (Furusato et al. 2004, Reynolds 2006). Conversely, this species is highly efficient in exploiting nutrients even though it is characterized by rather low growth rates (Reynolds 2006). However, in Sicilian reservoirs circulation events are promoted by dewatering; the water-level decrease not only continuously moves portions of reservoir bottom into the circulating layers but also favors the establishment of atelomictic patterns, thus promoting nutrient inputs from the sediments. Conversely, the stable segregation of epilimnetic waters from the hypolimnion was observed to be effective in both reducing biomass accumulation and improving light availability. Furthermore, the flushing out of phosphorus was observed to be higher under stable stratification conditions (Naselli-Flores and Barone 2005). This is probably due to the deep location of the reservoirs' outlets, which more efficiently remove hypolimnetic nutrients when mixing does not occur.

6.6 Conclusion

In conclusion, a more integrated eutrophication management strategy must be developed. The goal of the strategy should be to reduce the frequency and intensity of cyanobacterial blooms and other water quality problems associated with nutrient pollution in Mediterranean reservoirs. To meet this goal, the management of hydrological patterns would only be a palliative method. Eutrophication phenomena in Sicilian and Mediterranean reservoirs can be counteracted promoting an integrated lake basin management (ILEC 2007) planned:

- to reduce nutrient concentrations reaching the recipient water bodies (i.e., implementing waste treatment plants and reducing fertilizers);
- to improve operational procedures and to minimize the effects of water-level fluctuations;
- to increase the community's awareness of eutrophication problems and their associated risks;
- to promote research in order to obtain better information and scientific knowledge about the influence of local factors on ecosystem functioning.

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Chapter 7

Eutrophication: Threat to Aquatic Ecosystems

Abid A. Ansari, Sarvajeet S. Gill, and Fareed A. Khan

Abstract Eutrophication was recognized as a pollution problem in European and North American lakes and reservoirs in the mid-twentieth century. Since then, it has become more widespread. Surveys showed that most of the lakes in Asia, Europe, North America, South America, and Africa are found in eutrophic state. Eutrophication leads to significant changes in water quality. It lowers the value of surface waters for the industrial and recreational uses. The overpopulation of algae makes water unfit for swimming. The algae growing in long strands often twine around boat propellers and make boating difficult. Eutrophic waters tend to be scummy, cloudy, or even soupy green. The rapidly growing aquatic plants may wash onto the shores in storms or high winds, where these plants die, decay, and produce a bad smell all around such water bodies. The eutrophication in an aquatic ecosystem also causes significant changes in biodiversity. The eutrophication causes an increase in plant and animal biomass, frequency of algal blooms, growth of rooted plants, and decreases the species diversity. Due to eutrophication, an increase in turbidity and anoxic conditions occurs. Because of the high density of aquatic organisms in a eutrophic system, there is often a lot of competition for resources. This high degree of competition and high chemical or physical stress make the struggle for the survival in eutrophic systems higher. As a result the diversity of organisms is lower in eutrophic than in oligotrophic systems.

Keywords Eutrophication · Aquatic ecosystem · Biodiversity · Nutrients

7.1 Water

Water is the most precious natural resource for the sustenance of life on the earth. Water is used in day-to-day activities for drinking, bathing, washing, recreation irrigation, fishing, and industrial purposes. Freshwater would continue to be abundantly available. Of the total estimated water on earth and in its atmosphere, 95% is locked in the lithosphere and sedimentary rocks. Only 50% is actually available for free circulation and about 99% of free water is in oceans. Thus only a minute quantity is available for terrestrial life-supporting system. The earth is the only planet where liquid water exists in substantial quantities. Oceans, lakes, rivers, glaciers, and other bodies of liquid or solid water cover more than 70% of the earth's surface. Earth's atmosphere is unique in trapping water vapours and maintaining most of it in liquid state due to its temperature ranges.

The hydrologic cycle describes the water circulation, its evaporation from land, water bodies, and organisms. The hydrologic cycle includes circulation of water in the atmosphere via condensation and precipitation on the earth's surface. The water cycle includes the way water moves underground by infiltration and overland by runoff into rivers, lakes, and seas. The total water quantity on the earth is maintained year to year (Fig. 7.1). The hydrologic cycle is a simple transfer of water from one source to another. This process supplied freshwater to the land masses, which play a vital role in creating a habitat climate and moderating world temperatures. Movement of water back to

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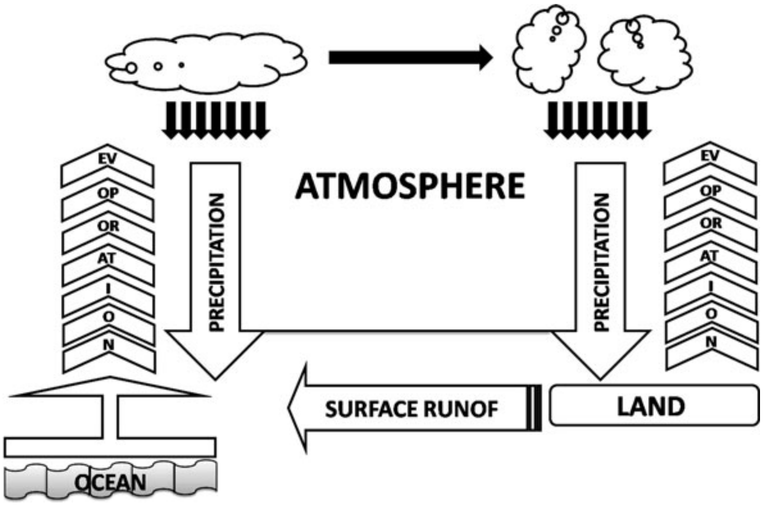

Fig. 7.1 Water cycle showing movement of water in the atmosphere

the sea, rivers, and glaciers is a major geological force that shapes the land and re-distributes materials. Plants play important role in the water cycle, absorption of groundwater, and its pumping into the atmosphere by transpiration. In tropical forests, as much as 75% of annual precipitation is returned to the atmosphere by plants. Solar energy dries the hydrologic cycle by evaporating surface water. Evaporation is the process in which liquid is changed to vapour at temperature well below its boiling point. Water can also move between solid and gaseous states without ever becoming liquid in a process called sublimation. On bright, cold, windy winter days, when the air is very dry, snow banks disappear by sublimation, even though the temperature never gets above freezing. This is the same process that causes freezer burn of frozen food.

In both evaporation and sublimation, molecules of water vapour enter the atmosphere, leaving behind salts and other contaminants, and thus creating purified freshwater. This is essentially distillation on a grand scale. The rainwater was once known for its purity. It was considered as a standard against which pollution could be measured. Unfortunately increasing amounts of atmospheric pollutants are picked up by the water vapour as it condenses into rain.

A cloud is an accumulation of condensed water vapour in droplets or ice crystal. Normally cloud particles are small enough to remain suspended in the air when these cloud droplets grow large enough and gravity overcomes uplift air currents, and eventually cause clouds to rain.

7.2 Eutrophication

Eutrophication is a kind of nutrient enrichment process of any aquatic body, which results in an excessive growth of phytoplanktons and macrophytes. This undesirable overgrowth of aquatic plants and their subsequent death forms a greenish slime layer over the surface of the water body. The slime layer reduces light penetration and restricts re-oxygenation of water through air current. The death and decay of aquatic plants produces a foul smell and makes the water more turbid. Eutrophication is one of the serious kinds of water pollution directly affecting the flora and fauna due to the loss of dissolved oxygen. It leads to an early and relatively faster mortality rate of fishes and thus spoils the desired water qualities of ponds and lakes. The fishing operation and navigation in eutrophic water becomes difficult due to enmeshed and heavy growth of plants. The hydroelectric generation from such water storages is adversely affected as nutrient-rich water acts chemically upon the turbines (Khan and Ansari 2005). At the end of an algal bloom, the decomposing debris also spoils the desired water characteristics and may result in the growth of disease-causing bacteria. An uncontrolled eutrophication leads to a rapid upwelling of a water body. The limited storage and water-recharging capacity of smaller freshwater bodies is reduced by silting. Small lakes and many ponds steadily lose their aquatic entity and become permanently terrestrial in nature.

Eutrophication leads to significant changes in water quality. It lowers the value of surface waters for the industrial and recreational uses. The overpopulation of algae makes water unfit for swimming. The algae growing in long strands often twine around boat propellers and make boating difficult. Eutrophic waters tend to be scummy, cloudy, or even soupy green. The rapidly growing aquatic plants may wash onto the shores in storms or high winds, where these plants die, decay, and produce a bad smell all around such water bodies (Khan and Ansari 2005).

The problem of eutrophication besides being dependent upon the nutrient inputs is greatly influenced by the number of environmental factors. Eutrophication induced significant change in the biodiversity. The change in biodiversity directly affects the trophic structure of the ecosystem. The floral and faunal diversity is threatened in the coastal areas receiving direct input of nutrients from some of the major rivers like the Amazon, Nile, Ganges, Mississippi, Brahmaputra, and Thames. Some effective control measures like biological control, mechanical control, legislative measures, and the awareness programmes pertaining to the present threat to water resources on the blue planet need to be carried out effectively. The water loses its normal dissolved oxygen level resulting in increased fish mortality. The fishing operation and navigation becomes difficult due to enmeshed and heavy growth of plants. Continued eutrophication leads to rapid upwelling of the water body. The limited capacity of small freshwater bodies is reduced by silting. Many ponds and lakes permanently lose their aquatic entity and become terrestrial in nature (Khan and Ansari 2005). An increase in living standard agriculture and development of local industries resulted in meso-eutrophication of Lake Taihu in China (Weimin et al. 1997). The runoff fertilizers in the stable ecosystem of Pagasitikos Gulf (Greece) resulted in eutrophic condition showing algal bloom (Triantafyllou et al. 2001). The excessive addition of nutrients to water causes quality problems and is one of the major causes of eutrophication in freshwater ecosystems (Ansari and Khan 2007). Eutrophication is the enrichment of water bodies with plant nutrients and precursors, typically nitrogen, phosphorus, and organic matter (Ansari and Khan 2009a). There exists a natural and slow eutrophication, which over geological times turns a lake into a marsh and then dries it entirely (Tusseau-Vuillemin 2001). Due to ubiquitous eutrophication and fragmentation, many plant species are actually threatened in Europe. Most ecosystems face an overall nutrient input leading to changes in species composition. Increasing species had better means of long-distance dispersal and are more competitive than decreasing species (Romermann et al. 2008).

Eutrophication is a natural process that occurs in all the aquatic systems and takes thousands of years. However, a high rate of input of nutrients due to anthropogenic activities significantly develops this condition in a very short period of time. This is called an artificial eutrophication (Rovira and Pardo 2006). The acceptable level of total inorganic phosphate in water is 0.03–0.04 mg L^{-1}. In most of the lakes and rivers where eutrophication is encountered, the principal causes of excessive enrichment of water are phosphates and nitrates. In and around cities and industries, phosphate contents increased 20- to 25-fold during the last 10–15 years (Muller and Helsel 1999).

The eutrophication causes an increase in plant and animal biomass, frequency of algal blooms, growth of rooted plants, and decreases the species diversity. As a result, an increase in turbidity and anoxic conditions occur. Because of the high density of aquatic organisms in a eutrophic system, there is often a lot of competition for resources. This high degree of competition and high chemical or physical stress make the struggle for survival in eutrophic systems higher. As a result the diversity of organisms is lower in eutrophic than in oligotrophic systems (Ansari 2005).

7.3 Eutrophication: A Global Scenario

Eutrophication has become a major cause of concern in the developing as well as developed countries. The water bodies of the highly populated countries like China, India, Bangladesh, Pakistan, Indonesia, and industrialized countries of Europe and Great Lakes states of USA and Canada are under the direct threat of eutrophication. The population increase coupled with economic development and changes in lifestyle during this period has added to the problem. The availability of freshwater for human consumption will be one of the great issues of the twenty-first century. The scientific interest in eutrophication has consistently been significant during the last 25 years. The emphasis in this area of research has suddenly risen in recent years. It is predicted that eutrophication, acidification, and contamination by toxic substances are likely to increase as threats to freshwater resources and ecosystems.

Eutrophication was recognized as a pollution problem in European and North American lakes and reservoirs in the mid-twentieth century. Since then, it has become more widespread. Surveys showed that 54% of lakes in Asia, 53% in Europe, 48% in North America, 41% in South America, and 28% in Africa are found in eutrophic state (Colin et al. 2007). Lake of Okeechobee, USA, is a shallow mixing basin. Total annual phosphorus concentration in the lake ranged from 50 to 100 μg/L. The seasonal minima in orthophosphorus and inorganic nitrogen were found limiting. Owing to seasonal and spatial gradations in P and N, the models based on annual phosphorus loadings were found inadequate to predict chlorophyll or other trophic state variables in the lake (Schelske 1989). Human-induced eutrophication degrades freshwater systems worldwide by reducing water quality and altering ecosystem structure and function. When current total nitrogen (TN) and phosphorus (TP) concentrations for the US Environmental Protection Agency nutrient ecoregions were compared with estimated reference conditions, all values for rivers and lakes exceeded reference median values. Over 90% of rivers currently exceed reference median values. Potential annual value losses were calculated in recreational water usage, waterfront real estate, spending on recovery of threatened and endangered species, and drinking water. The combined costs were approximately $2.2 billion annually as a result of eutrophication in US freshwaters. The greatest economic losses were attributed to lakefront property values ($0.3–2.8 billion/year, although this number was poorly constrained) and recreational use ($0.37–1.16 billion/year) (Dodds et al. 2009). Shallow brackish lake Hickling Broad, Norfolk (USA), shows changes from a clear macrophyte-dominated state in the decades previous to 1970 to turbid phytoplankton-dominated state by the mid-1970s due to increasing eutrophication. In the 1970s the lake diversity began to change again and during the 1980s submersed plants showed a recovery (Bales et al. 1993). Lake Tohopekaliga is a part of Kissimmee River system in central Florida. The discharge from various point source into this lake and its tributaries begin in the 1950s. The discharge of secondary wastewater effluents of four treatment plants containing huge amounts of P and N began in the early 1960s. Rapid population growth by 1969 expanded the treatment plant discharges. The excessive effluent inputs deteriorated the water quality, aquatic habitat, and biocommunities. By 1979 annual P loading to the lake was 11 times higher than under natural conditions (Williams 2001). Lake Apopka became eutrophic due to seasonal production of detritus by water hyacinth (*Eichhornia crassipes*) (Reddy and Be Bush 1991). Lake Chapala (Mexico) is the third largest lake of America. Large quantities of domestic and industrial sewage were largely treated and agricultural runoff from the entire Lerma–Chapala basin was drained into the lake. The effluent input caused significant changes leading to eutrophication (De Anda et al. 2002). The shallow urban city park lake located in Baton Rouge, Louisiana, has been reported hyper-eutrophic and suffered from frequent algal blooms and fish kill (Ruley and Rusch 2002).

Lake Apopka is a large (area of 125 km^2) shallow (mean depth 12.5 m) lake in Florida (USA). The phosphorus loading from flood plain farms made the lake hyper-eutrophic. The lake is reported to have high levels of nutrients, phytoplanktons (Chl-a, 80 μg/L), and suspended matter (Conveney et al. 2002). The lakes of Pampa plains (Argentina) are shallow polymictic, eutrophic, or hypertrophic and fluctuate greatly in water renewal time and salinity. The lakes are highly turbid due to organic discharge (Quiros et al. 2002). Characteristics of distinct beaches of the lower river Luján (Argentina) were monitored. Upstream shows lowest depth, discharge of highest nutrients, dissolved heavy metals, and Chl-a; downstreams showed consistent decrease in organic pollution level (Ferreira et al. 2002).

During the second half of the last century, the Lake Victoria (Kenfa) ecosystem has undergone drastic changes. Most noticeable changes were the decline in the populations of many endemic fishes. However, the remnants of some species that had disappeared from Lake Victoria occur abundantly in the Yale Swamp lakes (Aloo 2003).

Lake Greifensee, Switzerland (a dimictic lake), was oligotrophic 120 years ago but became hypertrophic in the early 1970s due to increase in population in the catchment area and related phosphorus input from sewer systems (Burgi et al. 2003). Over 30-year period, the habitat condition of the mesotrophic lake Mikolayskie (Poland) deteriorated and the phytolittoral zone was reduced from a 6-m-wide zone to only 2 m (Solinska and Symoides 2001). Increasing nutrient concentration and massive blooms of cyanobacteria were recently observed in Alte Donau, a eutrophic urban lake in Vienna, Austria. The first half of 1994 was characterized by relatively clear water and a high diversity of the phytoplankton composition due to flushing of the lake with water of better quality by the end of 1993 (Mayer et al. 1997).

A sediment core from the shallow hypertrophic lake Sobygaard (Denmark) indicated a succession from a naturally eutrophic state to hypertrophic state during recent centuries (Broderson et al. 2001). Lake Sakadas (Croatia) received untreated wastewater during the 1980s. The improvement of water quality and decrease in eutrophication were recorded in the period from its revitalization (controlling of wastewaters and sediment removal) till the beginning of war activities in 1991 (Mihaljevic and Novoselic 2000). Varying plant communities and environment regimes were observed in Lake Jaroslawieckie in summer of 1996; most habitats of this lake were eutrophic (Pelechaty et al. 1997). A year-long survey of Lough Bunny Lake (Ireland) was conducted during 1992–1993. The high water transparency observed in eutrophic zone may have been feasibly caused due to the shift of blooms at times to the deepest part of the lake (Pybus et al. 2003). In Danube Delta lakes, species changes occurred due to accelerated eutrophication of the lakes, with increased phosphorus loading and a reduction in N:P ratio (Schelske 1989). Lake Constance (France) showed a period of fast eutrophication during the late 1970s which subsequently altered the trophic structure. The eutrophication was caused by the excessive use of phosphate-based detergents (Schmieder et al. 1995). Lake Geneva (Switzerland) underwent rapid eutrophication until 1980 followed by a reversal that is still in progress (Lehmann and Lachavanne 1999). Surface runoff rich in agricultural wastes and underground seepage from urban and rural areas enriched Lake Kastoria (Greece) with nutrients and thus intensified the problem of eutrophication (Kaussouris et al. 1991).

The nitrate concentrations in three tropical coastal lagoons of Yucatan (Mexico) suggested that trophic status was influenced by freshwater springs. On the basis of nitrogen status, the Celestron lagoon was found eutrophic, Chelem lagoon was oligotrophic, and Dzilam lagoon was mesotrophic. But NH_4^+ concentrations indicted that all three lagoons were mesotrophic owing to some impact on remineralization process. On the basis of PO_4^{3-} concentrations, Celestron and Dzilam were mesotrophic and Chelem was oligotrophic (Herrera et al. 2002). Lake Peipsi (3,555 km^2, mean depth 7.1 m) consisting of three parts (Lake Peipsi s.s., Lake Pihkva, and Lake Lammijarv) is located on the border of Estonia and Russia. Lake Peipsi s.s. is eustralified and eutrophic. Lake Lammijarv has some dyseutrophic feature, while Lake Pihkva is a typical unstratified eutrophic lake. The mean concentration of total phosphorus and nitrogen in the surface water was 42 and 768 mg/m^3, respectively. The biomass of phytoplankton fluctuated between 1.0 and 125 g/m^3 and that of zooplankton from 0.088 to 6.344 g/m^3 with a summer average of 3.092 g m^{-3}. The dominant species were typical indicators of eutrophic lakes, including *Phragmites australis* (Haberman et al. 2000). The majority of Danish lakes are highly eutrophic due to high nutrient input from domestic sources and agricultural activities. Reduced nutrient retention and more rapid removal in catchments as a result of agricultural drainage of wetlands and lakes and channelization of streams also play a role in eutrophication (Jeppesen et al. 1999). Studies were conducted on oligotrophic Grane Langso Lake (Denmark) to examine that the lake has maintained a high water quality in relation to change in pH, nutrient conditions, and transparency over the past 40 years (Riis and Sand 1998). Most freshwater lakes in the Netherlands are very shallow (<2 m). They vary in area from a few hectares to a few thousand hectares. The external input to the lakes of phosphorus and nitrogen and of polluted waters from the river and canals has been the major source of eutrophication (Yin 2002). In the Lake Lugano (located at the border between Italy and Switzerland), a faster rate of eutrophication is reported due to excessive discharge from human settlements around the lake owing to the population increase and immigration (Berbeiri and Simona 2001).

Main channel of the River Danube (Slovakia) forms a frontier between Slovakia and Hungary and shows succession on its left bank after the construction of a dam which resulted in the changes in spatial distribution and species composition of macrophyte Otahel and Valachavic (2002). Danube River contributes 80% of annual river discharge into the northwestern Black Sea. From 1988 to 1991, on the basis of available observations, inorganic nutrients, phytoplankton biomass, dissolved oxygen, etc. contribute to the eutrophication process (Garnier et al. 2002). Karasu River of Turkey was polluted by industrial and urban wastes (Gurbuz and Kivrok 2002). Lake Kasumigaura, Japan, showed that trophic state index ranged monthly from 55 to 75 and the limiting factors were chemical oxygen demand and total phosphorus (Wei et al. 2000). Phosphorus stripping at a wastewater treatment plant on the River Charles (Massachusetts) was felt to be one of the main reasons of eutrophication of the river (McIntyre et al. 2003). Lake Taihu, China, is a mesotrophic lake. The urban, agricultural, and industrial developments have raised the living standard of people. In the main eutrophicated area the COD recorded in 1993 was 4.63 mg/L and total N and P were 3.93 mg/L and 0.107 mg/L, respectively, in 1995 (Weimin et al. 1997). Eutrophication is one of the main factors causing increased growth of green algae and turbid water in Donghu Lake in China (He et al. 2002). Lake Mirik in Darjeeling, Himalayas (India), revealed a very high input of nutrients in certain pockets of the lake. This anthropogenic nutrient input has spoiled the quality of drinking water (Jha and Barat 2003). The impact of industrial and sewage discharge on the water quality in the middle stretch of River Ganga (between Kanpur and Varanasi) was studied for 2 years from 1993–1994 to 1994–1995. The river water in this stretch was characterized by high BOD, COD, chloride, hardness, specific conductivity, alkalinity, nitrate, phosphates, free CO_2, low dissolved oxygen, and pH (Singh et al. 1999). Lake Dalavayi around Mysore in India shows concentration of heavy metals exceeding WHO standard, indicating metal toxicity. Additional inputs of soaps and detergents carried by the sewage water have caused eutrophication in the lake (Sujatha et al. 2001). Bellandur Lake is one of the major lakes of Bangalore in India. The addition of effluents through city drainage has changed the characteristics of the lake from a natural oligotrophic to an artificial reservoir of domestic sewage and industrial effluents (Chandrashekhar et al. 2003). A natural lake in Chennai (India) receives tannery effluents which made the lake eutrophic (Koteswari and Romanibai 2003).

A study was conducted in three lakes (upper lake, lower lake, and Mansarovar reservoirs) in Bhopal (India), and potential fertility of lentic water was assessed. The calcium contents ranged between 41.6 and 74.4 mg/L in upper lake, 30.0 and 78.0 mg/L in lower lake, and 47.2 and 992 mg/L in Mansarovar reservoirs. The data proved that eutrophication was highest in Mansarovar reservoirs (Garg and Garg 2002). Water chemistry of Udai Sagar Lake (India) showed high phosphates. Enormous algal blooms in the lake and its tributary Ahar River as well as occasional fish mortality indicated eutrophic condition. Eutrophication was caused by the discharge of pollutants from phosphate mines, chemical factories, distilleries, sewage, and domestic waste from settlements and hotels seen throughout the length of the river (Das 1999). Studies on eight shallow lakes, namely Salon, Gambhir Bandh, Ratoi, Devrishi Deval, Chandu, Majuwajagat, Bandsideh, and Bhagnaija, located in Azamgarh, Mau, Basti, and Siddharthnagar district of Uttar Pradesh (India) were carried out. All these beels (small lakes) were diatom rich, eutrophic, and mildly polluted to heavily polluted.

7.4 Nutrients in Aquatic Ecosystems

Nutrients especially phosphorus and nitrogen from various sources are the major causes of eutrophication and degradation of aquatic ecosystems (Ansari and Khan 2006a) (Fig. 7.2). Excessive phosphorus loading from agricultural areas in a watershed lake ecosystem causes eutrophication (Anderson et al. 2000). The P was considered as deriving force for increased diatom population in the hypertrophic lake Lough Neagh (Foy et al. 2003). Phosphate turnover in eutrophic lake is dominated by seasonal vertical fluxes, while in mesotrophic lake it is more conservative and its consumption and regeneration are restricted mainly to metalimnion in Great Masurian Lake (Poland) (Kufel 2001). Phosphorus enrichment in rivers can degrade the plant community by altering the competitive balance between different aquatic plant species, including both higher plants and algae (Mainstone and William 2002). Phosphorus played important role in limiting periphyton growth in the coastal streams of Florida, USA (Notestein et al. 2003). The ultimate phosphorus source is the mineral matter of rocks, but largely being insoluble, its incorporation into water system from rock sources is very slow. But in water bodies around villages, towns, and cities, enough phosphorus is reached from detergents, agricultural fields, and sewage and industrial effluents. Fertilizers applied to

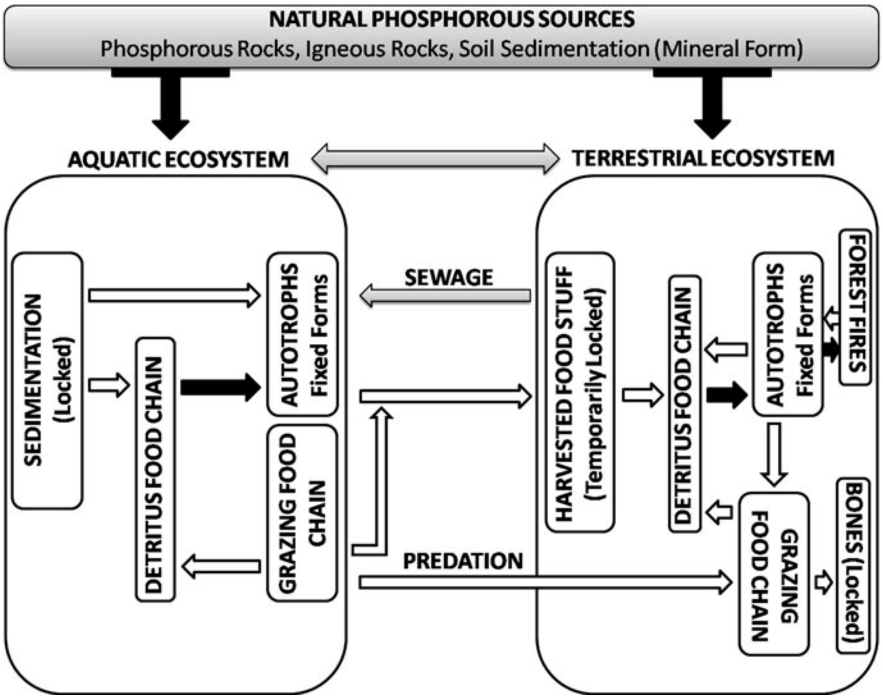

Fig. 7.2 Natural phosphorus cycling between the source, terrestrial, and aquatic ecosystems

agricultural field also contribute phosphates to the nearly water bodies. Over 60% of the phosphate in the US waterways is due to municipal sewage (Khan and Ansari 2005, Penelope and Charles 1992).

Phosphate mines and fertilizer runoff contain large amounts of phosphates. Domestic sewage is high in phosphate (about 50% from human waste and 20–30% from detergents). Animal wastes are the sources of both nitrate and phosphates (Penelope and Charles 1992). Diffuse phosphorus sources from agriculture are the major contributors to phosphorus enrichment in riverine sediments, where it can be utilized by benthic algae and rooted plants (Mainstone and William 2002). Surface runoff rich in agricultural wastes and underground seepage from urban and rural areas enriched Lake Kastoria (Greece) with nutrients and intensified the problem of eutrophication (Kaussouris et al. 1991). Application of manure to agricultural land is reported to have increased the concentration of phosphorus in soil (Sharpley 1999). The raw sewage discharges around most polluted sectors of River Arias-Arenales (the saprobity indices) varied between 10 and 11 during two hydrological regimes (period of drought from May to October and period of flood from November to April), which indicated a high degree of pollution level. The Shannon and Whilm–Dorris indices were not sensible enough to detect moderate pollution (Sadusso and Morana 2002).

The phytoplankton showed a marked difference in the diversity and abundance on fertilization with cow dung, pig dung, and poultry manure (Barua and Ghosh 2003). A shallow eutrophic wetland receiving domestic sewage and agricultural runoff was heavily infested with water hyacinth in Ujjain city (India) (Billore et al. 1998). In lentic and lotic water bodies, there is an intimate relationship of nitrogen and phosphorus cycling and sunlight. The water body having phosphate to nitrate ratio of 1:15 but about half of the nitrate is used up by the aquatic plants. The phosphates in such water body become limiting and nitrate accumulates in abundance. However, the water bodies with nitrate:phosphate enrichment level at the ratio of 4:15 used all its nitrate and the nitrate became limiting (Muller and Helsel 1999).

The contamination of a sewage outfall into Moa Point Bay (New Zealand) was closed and 3 months later signs of recovery of algae (*Ulva lactuca*) were noted. The fauna (limpets and blue mussels) showed slower recovery than the sea weed (*U. lactuca*) (Rogers 2003). Enormous algal blooming was observed in the Udai Sagar Lake (India). The lake becomes eutrophic due to the discharge of pollutants from phosphate mines, chemical factories, distilleries, and domestic wastes (Das 1999). The nutrients to aquatic environment were brought not only from rock weathering, soil leaching, and rain (natural sources) but also from agricultural runoff and domestic sewage. Some food processing industries (meat, vegetable, and cheese processing) also contribute significantly to the phosphorus budget, even though the pollution may be due to the floor and utensil cleaning rather than due to direct food wastes (Tusseau-Vuillemin 2001).

In Uruguay, detergents contributed 58% of daily P load (2.5 t) to the Rio de La Plata and Monte Vida. The detergents in Uruguay contain 0.6–11.3% phosphorus (Sommaruga et al. 1995). It is estimated that 55% of the P was from metabolic sources and 45% from detergent and cleaning products in Lake Lugano (Switzerland) Berbeiri and Simona (2001). The input of agricultural, industrial, and domestic effluents into Najafgarh drain in Delhi alters the algal diversity. Only highly resistant species could survive in the Najafgarh drain due to excessive effluents (Sinha 2001). Surface runoffs rich in agricultural waste and rural areas have been found to enrich Lake Kastoria (Greece) with nutrients (Kaussouris et al. 1991). The fertilizers used in agroecosystem have also been suggested to cause eutrophication of rivers and lakes (Egli et al. 1990). The increased consumption of P-based fertilizers from 1959 to 1970 with a decrease from 1975 to 1979 became stable from 1985 to 1990 in Uruguay with a total of 40,000 t/year. The phosphorus-sensitive water body was in the central, south, west, and east in Uruguay (Sommaruga et al. 1995).

Spatial and temporal variability of nutrients and suspended solids was investigated in agriculture headwater streams which are greatly influenced by human activities. It was found that higher concentration of nutrients occurred in the months with higher precipitation and intensive agricultural activities. Results showed that most important sources of pollutants are agricultural runoff or base flow, which accounted for more than 90% of the whole stream release. In the catchment areas of Lake Lugano (Switzerland), the external nutrient load derived from anthropogenic (85%), agricultural (5%), and industrial (10%) sources (Berbeiri and Simona 2001).

7.5 Eutrophication and Aquatic Environment

The physical processes within a water source have major implications for controlling eutrophication in aquatic bodies (Lau and Lane 2002a, 2002b, Khan and Ansari 2005, Ansari and Khan 2006b), while nutrients, temperature, pH, dissolved oxygen, carbon dioxide, and light limit the growth and development of aquatic plants (Shen 2002).The environmental factors like salinity, nutrient, inorganic N, dissolved Si, dissolved oxygen, and oxygen saturation were considered to work out pressure and state of eutrophication. However, the nutrient ratios (N:P and Si:P), besides capacity of the system to produce and sustain organic matter, were found as relevant descriptor for certain eutrophic sites in Bulgarian Black Sea coastal zone (Moncheva et al. 2002). The water temperature and physicochemical parameters were found to be important environmental factors determining macrophyte community distribution in 79 small Danish lowland streams (Baattrup et al. 2003). The drought resulted in the increase in water temperature and more penetration of light, which in turn improved the diversity of aquatic plant and enhanced the vegetation cover as well as biomass in the Lake Choughu in China (Peng et al. 2003).

The Baltic Marine Environment Protection Commission reported that the eutrophication is the main threat to the aquatic environment. The response of biological communities has reacted to an increase in man-induced multisource stresses. Changes in plankton, benthic animals, and macroalgal assemblages and fish communities have been documented in most parts of the Baltic Sea (Tremp 2007). The effluents of bleached kraft mill released into the Vouga River (Portugal) increased water temperature, pH, colour, suspended solid, conductivity, and COD. These effluents in summer season reduce the diatom diversity (Ferreira et al. 2002). The light intensity and temperature in addition to N and P modified algal growth in Zhejiang River network, China (Shen 2002). The sterile filaments of *Spirogyra* and *Zygnema* were more widely distributed in thermal water. The green algae played important role in the thermal algal communities forming the main bulk of biomass of floating macrophyte (Stoyneva 2003). In Lake Taihu of China, the *Microcystis* spp. constitute 85% of algal biomass in summer (Weimin et al. 1997).

The variation in inflow water temperature into Lake Eokiri (Japan) regularly affected the lake water quality similar to Lake Calhoun, USA (Imteaz et al. 2003). In the drainage channel Sussex (UK), the pH of water is increased due organic loading which resulted into an excessive growth of *Potamogeton crispus* and suppression in growth of *Groenlandia densa* (Thomas and Daldorph 1994). Lowering of salinity in shallow brackish lake Hickling Broad, USA, was suggested for the displacement of large phytoplankton biomass as a measure of its restoration (Bales et al. 1993). The effluents of bleached kraft mill released into the Vouga River (Portugal) increased water temperature, pH, colour, suspended solids, conductivity, and COD. These effluents in summer season reduce the diatom diversity (Ferreira et al. 2002). The aquatic mosses dominated over-rooted *Isoetes* probably due to acidification of oligotrophic Grane Langso Lake (Denmark). The *Polygonum amphibium* increased the pH near the bottom of shallow lakes of Kushiro (Japan) during summer (Takamura et al. 2003).

The 245 taxa of diatom community were studied in 39 streams in Lake Victoria, Australia, and were found to be strongly correlated with secondary salinization. The diatom community in this drainage was found directly related with the P concentration and specific conductance (salinity) of water (Blinn and Bailey 2001). Nine species of cyanobacteria, eight species of Chlorophyta, and three species of Bacillariophyta were found frequent in aridisols of Rohtak (India) having pH of 7.8 in dry season and 8.4 in rainy season (Manchanda and Kaushik 2000). The internal eutrophication (without nutrient loading) was noted in the semi-aquatic habitat receiving high alkaline and sulphate-rich water from River Rhine (the Netherlands). The alkalization of sediments is caused by the production of carbonate owing to the reduction of sulphate and alkaline nature of inlet water (Smolders et al. 1995). In Donghu Lake, *P. crispus* was grown in enclosures to compare water quality reference. It was found that the growth of *P. crispus* enclosed water transparency, increased pH, and decreased electric conductivity probably due to suppression of green algae. The periphyton photosynthesis has been found to increase with pH locally by 1 unit leading to precipitation of calcium phosphates and concurrent deposition of carbonate and phosphate complexes in freshwater ecosystem. This phenomenon results in long-term phosphorus sedimentation (Dodds et al. 2009). Low Secchi transparency has been recorded in Lake Sakadas as a result of high concentration of ammonium, nitrate, and organic matter as contained in the wastewater downloaded into the lake (Mihaljevic and Novoselic 2000). During drought period, improvement in aquatic plant diversity was found related to increase in light intensity in Lake Changhu in China (Peng et al. 2003). The occurrence of *Chara* sp. was positively related with average Secchi depth in Lake Veluwemeer (the Netherlands), while the occurrence of *Potamogeton pectinatus* was negatively related. Both the species showed a negative sigmoid response to water depth of lake. The dominance of *P. pectinatus* was restricted to turbid water and *Chara* appeared as stronger competitor in clear water (Berg et al. 1999).

The Secchi depth transparency data showed that Danish coastal water was heavily eutrophic. The studies revealed that Secchi depth was predictable with the precision from measurement of chlorophyll and suspended matters (Nielsen et al. 2002). The light stresses accelerated adoption strategy of *Potamogeton maackianus* (submersed macrophyte) resulting in the elongation of main and lateral shoots causing increase in their density in the Donghu Lake (China) as revealed in aquarium experiment. The disappearance of this species from Donghu Lake was attributed to further light stresses (Ni and Xie 1999). The nutrients increased eutrophication and promoted phytoplankton development which in turn reduced water transparency, growth of rooted plant, and accelerated silt input to a river connected to a wetland. All these factors promoted plant diversity (Amoras and Bornette G 1999). The increase in water transparency was considered to be an important factor in lake restoration plan so as to allow the re-establishment of submersed macrophyte. The lake water was re-circulated for 29 months and removal of suspended solids and particle-bound nutrients was assessed (Conveney et al. 2002). The increase in inlet water in Lake Veluwemeer increased water transparency because of lower turbidity. The water transparency further improved after the death of benthivorous fishes. The minimum internal charophyte coverage was the requirement to stabilize clear water transparency of at least 1 m (Porteilje and Rijsdijk 2003).

The clear lakes of relatively transparent water and high relative abundance of large piscivorous fishes were having relatively low phytoplankton biomass and abundant rooted macrophyte. The green turbid lakes of low water transparency with high abundance of visual planktivorous fishes had high inorganic turbidity and scarce aquatic macrophyte abundance and phytoplankton biomass. Large part of Pampa plain lake (Argentina) had seasonal phases of clear water. The complex interaction between climatic factor and land use may be related to non-periodic changes in the typology of a given lake. It may be greatly influenced by decrease in lake surface area (Quiros et al. 2002). Periphyton can cause supersaturated O_2 concentration near the sediment surface, causing increased metal phosphate deposition, but anoxia in the night associated with periphyton respiration may offset this phenomenon Dodds et al. (2009). The COD and BOD were found negatively correlated with dissolved oxygen in Karasu River basin in Turkey (Gurbuz and Kivrok 2002). The *Trapa japonica* in shallow lakes of Kusharomoore (Japan) formed an environment with steeper extension of light, higher concentration of dissolved organic carbon, and lower concentration of dissolved oxygen near the bottom in comparison to other taxa. The *P. amphibium increased* pH but dissolved oxygen near the bottom did not decrease to the level as seen in *T. japonica* community during summer (Takamura et al. 2003).

The silicate supply and strong tidal mixing of water current determined year-round dominance of diatoms over dinoflagellates in Urdaibai estuary of Spain (Maria and Orive 2001). The drought periods have temporarily reduced nutrient loading in nine lakes of Michigan (USA). During drought period the macrophyte community in normally eutrophic Shoe Lake has shifted from *Ceratophyllum*-dominated low diverse community to more diverse community similar to that in less-productive Graham Lake (Zalewski et al. 2003). The growth and the development of the aquatic plants in the Lake Choughu (China) were accelerated by drought. The turion germination and the shoot length growth of *P. crispus* and yellow leaves in *Trapa incisa* were higher in the drought period (Peng et al. 2003). The natural eutrophication of reservoirs resulting from reduction in total water capacity owing to drought has been found to be influenced by climate change and affected dissolved oxygen concentration. It was observed that global climate change such as drought linked to El Nino effects can influence phytoplankton population in continental water as proven in marine ecosystem (Tunner et al. 2003). Seasonal monsoon served as flushing mechanism in two ways: (i) monsoon reduced seasonal eutrophication effect during summer and (ii) monsoon prevented (long-term) accumulation of organic matter in the sediment due to nutrient enrichment (Yin 2002). The eutrophication in estuarine and coastal water shaft caused by nutrient loading is greatly influenced by climatic perturbations (e.g. drought, hurricanes, and floods). This climate stressor greatly influences phytoplankton diversity, productivity, and biogeochemical cycling (Paerl et al. 2003). The water transpiration by 14 wild plant species specifically *Cryptotaenia japonica*, *E. crassipes*, *Phragmites communis* (*P. custralis*), *Poa annua*, and *Oenanthe javanica* was found to be directly related to mineral absorption from eutrophic water (Huang et al. 1998).

7.6 Eutrophication and Aquatic Biodiversity

The floral and faunal diversity was influenced by nutrient loading in the coastal environment (Deegan et al. 2002). Some small lakes were studied in south-eastern Michigan, USA. It was observed that nitrogen (nitrate and ammonia) loading have a strong relationship with the dynamics of macrophyte community. The nitrogen loading itself appeared to be related to annual and seasonal precipitation. The drought causes reduction in nutrient loading to the lakes causing changes in species diversity (Tracy et al. 2003).

Eutrophication was found to have reduced the number of rare species and increased the abundance of meso- to hyper-eutrophic species. The reduction of total phosphorus restored the diatom assemblages of 1930 as studied in the experiments utilizing different situation of water quality in Nieuwkoopse Plassen in the Netherlands (Dam and Mertens 1993). Impact of three N:P ratio has been studied on 78 genera of algae. It was revealed that averaged total nitrogen value showed no disfrequency among three treatments. The phosphorus, however, varied in all three N:P applications which had significant impact on abundance and diversity of phytoplankton (Ahmadi and Bani 1999). Anthropogenic activities changed the composition of diatom assemblages of south-western Baltic Sea during the last century. These changes show increase in species that thrive in eutrophic water and these indicated that salinity and availability of inorganic nutrient increased in the water (An and Kim 2003).

Studies conducted along marsh and slough areas of Northern Everglades (USA) shows a significant relationship between species diversity and nutrient enrichment. The marshes with low concentration of nutrients were dominated by *Cladium jamaicense* where as nutrient enriched areas were dominated by *Typha domingensis*. Open and low nutrient water areas were characterized by *Eleocharis* sp. and nutrient enriched open areas were dominated by floating plants and *Polygonum* sp. The increase in phosphorus status accompanied gradual establishment of macrophyte and a variety of floating plants (Vaithiyanathan and Richardson 1999). The charophyte and benthic diatoms in Lough Bunny Lake (Ireland) actively utilized plant nutrients which led to the water clarity and small phytoplankton population (Pybus et al. 2003). The community ordination technique showed that specific conductance (salinity and P) interacted to determine stream diatom community structure in drainage with high secondary salinization in Australia. The drainage with a wide range of salinities showed strong association with certain diatom taxa (Blinn and Bailey 2001). The increased composition of sediments due to eutrophication of Lake Balaton (Hungary) increased the establishment of *Typha augustifolia* and *Typha latifolia*. Due to degradation of reed beds (*P. australis*) the aquatic plant showed disappearance and reduced species diversity (Kovacs et al. 1996). It was reported that imperical models based on annual phosphorus loading or concentration were not adequate to predict chlorophyll concentration or other trophic state variables due to large-scale seasonal changes in nutrient status. It was suggested that seasonal phytoplankton nutrient model coupled with hydrodynamic models may provide predictability of nutrient load (Schelske 1989). The ponds contaminated with organic matter and sewage water resulted in covering of water surface with aquatic weeds and thus the predation of mosquito larvae by larvivorous fishes was avoided. In such pond the larvae population increased (Lee and Lee 2002). The submersed macrophyte-dominated area guards against threats of eutrophication and loss of diversity (Nehring and Welling 2003).

The field studies conducted on 12 cutoff channels of the River Rhone (France) revealed that low or high nutrient levels decreased species. The intermediate nutrient levels favoured the co-occurrence of species contrasting nutrient requirements. The species richness was high because of their safety against extinction by long-term isolation of the channel and cold groundwater supplies. The most frequently flooded channels have highest species richness and occurrence of rare and fugitive species (Bornette et al. 1998).

A spatial heterogeneity was detected in La-Safar coast with small and shallow water bodies in Spain. The area was found highly eutrophic due to high amounts of phosphorus, Chl-a, and varied water body features. The eutrophication resulted in a high diversity of aquatic habits with 128 species of microalgae and 1–21 species of phytoplanktons. The water body types in the region were found to be a function of total phosphorus flux of water, water transparency, presence of macrophytes, and oxygen. These conditions contributed to the limitation of biodiversity loss often observed in eutrophic condition (Rodrigo et al. 2003). The decreased species diversity and disappearance of aquatic plants were noted in the reed beds of Lake Balaton (Hungary) as a result of eutrophication (Kovacs et al. 1996).

The submersed macrophytes have been reported to be equally sensitive to the process of eutrophication and oligotrophication (acidification of water) caused by man. These submersed macrophytes were reported to be the most suitable for the assessment of lotic (running) water (Tremp et al. 1995). Due to eutrophication in the river flood plains, wetlands of upper Rio Parana in Brazil, marshes in river deltas and mangroves near aquador border with Peru, significant changes in species diversity was recorded (Leon and Young 1996, Murphy et al. 2003). However, from water qualities within phosphate, light, pH, depth, and sediment have been considered in surveys of the Brazilian water bodies (Murphy et al. 2003). Aquatic macrophytes play an important role in the structure and function of aquatic ecosystems. Many of the threats to freshwaters (e.g. climate change and eutrophication) will result in reduced macrophyte diversity and will, in turn, threaten the faunal diversity of aquatic ecosystems and favour the establishment of exotic species, at the expense of native species (Chambers et al. 2008). Saugany Lake (USA) has also been conducted on the similar line for the future reference of the species diversity surveys (Alix and Scribailo 1998). The Saugany Lake has been referred to have natural community with high species richness. Similar studies on species diversity, density, and seasonal distribution of aquatic plant in Harike Lake (India) have been carried out. *Eichhornia* was found to cover 70% of the lake; in total 34 species were recorded (Bath et al. 1998). The species richness index, Palmer's generic index, Margalef's index, and water quality assessment indicated that a freshwater pond in Tamil Nadu (India) was highly eutrophic (Hariprasad and Ram Krishnan 2003). In a study conducted in restored grassland ditch vegetation in the Netherlands, it was noted that number of terrestrial plant species increased and new terrestrial species emerged in response to cessation of fertilizer application, which increased the terrestrial species number from 72 to 122 and the aquatic species number to 16 (Best et al. 1995). Macrophytes, namely *Najas graminea*, are most abundant submersed wild plant in rice agroecosystem. This species served as keystone species for its crucial role in maintaining the organization in diversity of aquatic community (Martin and Saverborn 2000). The succession occurred in Danube River, Slovakia, because of its damming. It resulted in change in spatial distribution and species composition of macrophyte. In North America, *Nuttali* spp. spread rapidly (Otahel and Valachavic 2002). The plant species composition in a study showed significant correlations with potash concentrations, winter and summer groundwater level, water depth, and elevation in bank vegetation. The aquatic vegetation showed correlation with pH, chloride, organic carbon, ammonium concentration, and water temperature (Best et al. 1995). The changing diversity of phytoplankton composition in response to flushing of the Alte Donau Lake (Austria) with water of better quality led to high phytoplanktonic diversity (Mayer et al. 1997). Alteration in the species diversity during floods and drought period in the eutrophic Shoe Lake (USA) has been observed. The nitrogen loading during normal precipitation promoted dominance of non-rooted species (Tracy et al. 2003).

In a study on densely populated lowland of Europe and its comparison with the record of last 100 years, most of freshwater lakes containing diverse submersed vegetation now have high phytoplanktonic biomass and summer transparencies below 2 m, a typical characteristic of eutrophication. The majority of 17 lakes have lost all or most of their submersed species during the past 100 years. Low habitat diversity and frequent disturbance in streams favoured the robust and fast-growing species capable of re-growth after weeding (Sand et al. 2000). Some of these diversity losses besides non-selective fishing were changes in drainage basin, increased eutrophication, and invasion of water hyacinth (Aloo 2003). Many ditch banks on modern dairy farms in the western peat district of the Netherlands contain species-rich vegetation. The impact of some agricultural factors including N supply on grassland, land use, slope aspects, soil type, pH, P and K of top soil, and ditch cleaning processes was assessed in relation to floristic richness (Twisk et al. 1994). In a chain of small lake in south-eastern Michigan, USA, macrophyte community dynamics has shown strong relationship variations in nitrogen loading (Tracy et al. 2003). Changes in aquatic systems as a result of nitrate leaching have led to decreased drinking water quality, eutrophication, hypoxia, and decrease in aquatic plant diversity (Mosier et al. 2001). A shift in primary producers from eelgrass to macroalgae was a record which in turn alters the food web. The increase in nitrogen load increased eelgrass shoot density and biomass (Deegan et al. 2002).

The increased organic loading in drainage with dominant submersed macrophyte in Sussex (UK) resulted in rapid diversions towards either phytoplankton or *Lemna* dominance at the cost of submersed macrophyte. There was an increase in the cover of *P. crispus* and a decrease in *G. densa*, on treatment with excessive organic nutrients. These nutrients increased conductively and decreased oxygen, pH, and chlorophyll values in the water column (Thomas and Daldorph 1994). The diatom communities (245 taxa) were found strongly correlated within land use and practices, i.e. historic clear-felling, and secondary salinization (Blinn and Bailey 2001). The difference in water current velocity at two sites in downstreams of Kisto Bazar Nala (India) resulted in the variation in the density of biotic communities which indicated the possibility of eutrophication into the impound water (Bhatt et al. 2001). In Tisza River (Yugoslavia), increasing trend of organic pollution was worked out utilizing the bioindicating values of floating, submersed, and emergent hydrophytes (Stojanovic et al. 1999). An inverse relationship was recorded between tannery effluent concentration and surviving planktons (Chlorophyta, Bacillariophyta, Cyanobacteria, Rotifera, Copepoda, and Cladocera) as studied in microcosm (Koteswari and Romanibai 2003). In a north temperate estuary, nutrient loading alters habit, physical, chemical structure, and food webs of the ecosystem. The microalgal biomass increased with nutrient loading, but the biomass and shoot density of eelgrass decreased. The fish population, abundance, biomass, and diversity decreased considerably. The macroalgae had marked impact on eelgrass. The fish had greater survivorship than eelgrass and the macroalgae. The watershed-derived nutrient increased macroalgal population and degraded eelgrass habitat, thus reducing the capacity of estuaries to support secondary consumers (Deegan et al. 2002).

The period with less than 250 mg/m^3 P showed dominance of chlorophytes and cryptomonad (phytoplankton) with lesser amount of diatoms and dinoflagellates. With declining P load, biomass of cyanobacteria increased, while chlorophytes and dinoflagellates decreased. The diatoms also attained substantial proportions at lower P loads. Non-motile planktons could survive during enhanced turbulence during clear water period, whereas motile planktons dominated during thermal stratification (Sand et al. 2000). The construction of reservoir and hydroelectric power station on River Danube (Slovakia) changed the diversity. The tank construction resulted in the onset of succession (Otahel et al. 2002). The construction of reservoir lakes and hydroelectric power generation increased the genetic diversity (Mori et al. 1999). The analysis of sediment core of a hypertrophic Lake Sobygaard (Denmark) showed a succession from a naturally eutrophic state to a hypertrophic state during recent centuries (Broderson et al. 2001). Distinct changes in chironomid communities reflected the eutrophication process and the succession of macrophyte through *Chara*, *Ceratophyllum*, and *Potamogeton* dominance to the existing state, with complete loss of submersed vegetation and dominance of phytoplankton (Broderson et al. 2001). Davies (1997) noted that submersed macrophytes facing a direct threat from eutrophication and saltation/turbidity played important role in maintaining the diversity and abundance of the invertebrate. The chironomid taxa were the best indicator of highly productive Danish lakes lacking abundant submersed vegetation (Broderson et al. 2001). The aquatic plant diversity is reported to be sensitive to a large number of other factors, viz. pollutant, resulting in reduced diversity of macrophytes, phytoplankton, and several aquatic fauna (Nicholls et al. 2003). In some oligotrophic Danish lakes, the high quality of water was maintained in the last 40 years. However, aquatic mosses now dominate the vegetation at the cost of rooted *Isoetes* due to slight change in pH (summer 5.6 in 1958–1963 and 5.07 in 1976–1994). The annual pH fluctuation in the lake was 0.3 unit 40 years ago and is 0.9 units presently. The population of *Nitella flexilis* has disappeared which once dominated the bottom vegetation in hypolimnion (Riis and Sand 1998).

Eutrophication can profoundly change rocky shore communities. These changes often cause the replacement of perennial, canopy-forming algae such as *Fucus* spp. with annual, bloom-forming algae such as *Enteromorpha* spp.; grazing, however, can counteract eutrophication by eliminating the annual algae's susceptible recruits (Worm and Heike 2006). In three shallow lakes with varying vegetation cover and nutrient concentration in Kushiro (Japan), *T. japonica* had a wide range of distribution in varying nutrient levels. The pH was higher in *P. amphibium* community. The Chl-a concentration at a given nutrient level was significantly lower in water with submersed macrophytes than in water without them. The presence and absence of submersed macrophytes was related to difference in phytoplankton community (Takamura et al. 2003). Lake Geneva (Switzerland) which underwent rapid eutrophication until 1980 and followed a reversal is still in progress. The submersed macrophytes along 20 km of the lake shore were studied in 1972, 1984, and 1995 and it was revealed that *P. pectinatus*, *Potamogeton perfoliatus*, *Potamogeton lucens*, and *Elodea canadensis* did not show significant changes in their distribution, with the two former species dominant throughout. However, *Chara* sp. and *Myriophyllum spicatum* increased in abundance in 1995. The abundance of *Potamogeton pusillus* consistently increased from 1972 to 1995. A new species *E. canadensis* appeared for the first time in the lake in 1995. The macrophytes were used to assess the water quality (Lehmann and Lachavanne 1999). The eutrophication in De Nieuw Roopse Plassen in the Netherlands reduced the number of rare species and increased the abundance of meso- to hyper-eutrophic species (Dam and Mertens 1993). The flora of Novi Sad-Selo Canal (Yugoslavia) was found diversed with several submersed floating and emersed hydrophytes. The existing species was assessed for their bioindicating values (Stojanovic et al. 1999).

The eutrophication and reduced N:P ratio in Danube Delta lakes caused species changes showing increased relative abundance of colonial blue-green algae (Vadineau et al. 1992). In meso-eutrophic Lake Taihu (China) the *Microcystis* species in summer occupied 85% of total algal biomass and appear to be the main blooming species among *Anabaena*, *Melosira*, *Cyclotella*, and *Cryptomonas* (Weimin et al. 1997). The phosphorus application in a pond induces significant difference in the abundance and diversity of phytoplankton including 78 genera from five phyla of algae (Ahmadi and Bani 1999). Proportions of algal biomass consisting of cyanophytes, filamentous chlorophytes, halophilic diatoms, and diatoms utilizing nitrogen heterotrophically were greater in eutrophic river segment than in less nutrient-enriched segment. The benthic flora composition indicated mesotrophic or eutrophic conditions throughout the upper Illinois River basin (USA). Few diatoms were indicative of hypertrophic water. The P regression calibration model with benthic algae abundances was found significantly in predictable models (Leland and Porter 2000). The disturbed sites with very high nutrient loading were characterized by a low vascular plant richness and the presence of filamentous algae in this mineralized stream (Theibaut and Muller 1998).

The logging of forest trees on the coastal areas promoted algal population due to increased availability of nutrients, light, and mixing depth (Nicholls et al. 2003). In Moa Point Bay (New Zealand), the sewage discharge affected the population of flora and fauna. The biodiversity was limited in the immediate areas surrounding the sewage discharge (Rogers 2003). The Venice lagoon subjected to high propagule pressure and anthropogenic forcing had higher diversity of non-indigenous species compared to other Mediterranean lagoons (Occhipiniti and Savini 2003). The oxygen depletion caused mass fish mortality for a short time but had a significant consequence on the lake ecosystem. The ammonia, nitrate, and organic matter were high in Lake Sakadas, Croatia (Mihaljevic and Novoselic 2000). The macrophyte species richness and abundance increased along an upstream to downstream zonation which was characterized by an increase in mineralization and nutrient level in Northern Vosges Mountains in north-east France (Theibaut and Muller 1998). In Jaroslawieckie Lake the phytoplankton sample and bottom sediments showed a succession of algae corresponding to the increasing trophic level of the lake. Most habitats of this lake were found eutrophic (Pelechaty et al. 1997). The phosphorus enrichment in the marsh and slough area of the Northern Everglades (USA) caused distinct changes in species frequency. The marshes in unenriched and enriched areas were dominated by *Cladium jamaicense* and *T. domingensis*, respectively. The unenriched open waters were characterized by *Eleocharis*, *Utricularia* sp., *Chara zeylanica*, and *Nymphaea odorata*. The enriched open water areas were dominated by *Polygonum* sp. and floating plants. The increase in P status in slough caused gradual establishment of macrophytes like *Mikania scadens*, *Sarcostemma clausum*, *Polygonum* sp., *Acrostichum danaeifolium*, and several floating plants. The characteristic slough indicators of unenriched slough were *Utricularia purpurea*, *Eleocharis elongata*, and *Eleocharis cellulosa* (Vaithiyanathan and Richardson 1999).

Lake Peipsi consisting of three parts (Lake Peipsi s.s., Lake Pihkva, and Lake Lammijarv) is located on the boarder of Estonia and Russia; the dominant species were typical of eutrophic lake including *P. australis*, *Potamogeton gramineus*, *P. pectinatus*, *P. amphibium*, *Eleocharis palustris*, *Sagittaria sagittifolia*, and *Butomus umbellatus*. Among these three parts, Lake Peipsi s.s. is an unstratified eutrophic lake, Lake Lammijarv has some dyseutrophic features, and Lake Pihkva is a typical unstratified eutrophic lake (Haberman et al. 2000). The effluents caused reduction in diatom diversity in the Mediterranean climate of central Portugal. The invertebrate densities and diversities decreased downstream (Ferreira et al. 2002).

7.7 Eutrophication in Wetland Ecosystems

A wetland is a place where water is the primary factor controlling the immediate environment. Wetlands can be as small as a child's wading pool or as large as a lake. Wetlands generally occur where land and water meet and underground water is at or near the surface or where land is covered by water less than 6 ft deep. The water level in a wetland rises and falls. This shift may depend on the location, the weather, the climate, or surrounding ecosystems. The area may be temporarily saturated and then dry up until another watery inundation. Meanwhile, a wetland provides a rich home to many animals and plants. Wetlands provide many important ecosystem services to human society, which may depend on how plant diversity influences biomass production and nutrient retention (Engelhardt and Ritchie 2001).

In rivers overflow of water promote diversity in wetland, reducing competition between the species.

The eutrophication results into phytoplankton development which in turn reduces water transparency and suppress the growth of rooted plants. The silt inputs into the water body cause terrestrialization. These multiple effects of eutrophication were combined into a model, predicting diversity and life history traits of aquatic plants (Amoras and Bornette 1999).

The drought caused increased biodiversity and increased vegetation coverage in Lake Choughu, China, due to increased light intensity and water temperature, although emergent vegetation disappeared from the lake during drought but floating and submersed vegetation expanded significantly (Peng et al. 2003). Studies conducted in 79 small Danish lowland streams reported the weed removal to be an important factor in reducing macrophyte communities (Baattrup et al. 2003).

The disturbance of wetland across the ecoregions of Great Lakes Basin has been found directly related with land use for agricultural and urban activities. The structure and functions of wetland ecosystems are affected by sedimentation/turbidity, retention time, eutrophication and changes in hydrologic time. (Detembeck et al. 1999).

- shift in plant species composition
- reduction in fauna
- decreased biodiversity
- reduction in secondary production
- increased flood peaks/frequency
- increased above ground production
- loss of aquatic plant species with high light compensation point

The high disturbance activities in wetlands of Canada and USA were reflected in terms of species loss and it was suggested that wetland restoration strategies shall cover diversed wetland type, including wet meadows, fens, and forested swamps. The vegetation removal and site disturbance caused plant invasion. The enclosement activities in marshes and ponds of southern ecoregions of Great Lakes were common causes of disturbances (Detembeck et al. 1999).

The riverine wetlands of River Rhone (France) were studied and divided into three different groups on the basis of species richness in water body. First with poor species richness, high nutrients concentration and turbid water. Second with intermediate species richness. The third group was species poor because of excessive groundwater supply. Srong dominance of *Eichhornia azurea* was recorded in the riverine flood plain wetland of upper Rio Parana River, Brazil (Murphy et al. 2003). The degradation of a reed bed of the Lake Balaton (Hungary) causes disappearance of aquatic plant or reduces species diversity and chemical composition diversity of the plant in the reed beds (Kovacs et al. 1996). Due to increased eutrophication of the mesotrophic lake Mikolayskie in over 30 years, the number of submersed macrophyte species has decreased by 50% and the frequency of most species has declined several fold (Solinska and Symoides 2001). Nutrients increased eutrophication and development of phytoplanktons which have been found to reduce transparency of water and reduce the growth of rooted plant. These effects are combined into one model that predicts diversity in land history effect of aquatic plant (Amoras and Bornette 1999). On the other hand due to high flooded period the periphyton shows the differential growth rate in hydrologic cycles which are reported to play an important role in the wetland diversity (Novelo and Tavera 2003). The Undasa wetlands of Ujjain (India) did not show full biological potential due to excessive pollution inputs from nearby industries (Shrivastava et al. 2003). In Spanish semi-arid wetland, sedimentation played a key role in nutrient dynamics. Internal loading was much higher than external loading and nutrients showed seasonality. N and P reduced in humid years due to higher water level. Spatial heterogeneity in nutrient content was ascribed to hydrological fluctuations. The input sites show higher mean value in rainy season. The water level-dependent emergent macrophytes, namely *Cladium mariscus* and seagrass, showed significant relationship with N and P (Sanchez Carllo and Alvarez 2001). During the second half of the last century, the biodiversity of Lake Victoria (Kenya) has changed considerably owing to changes in drainage basin, increased eutrophication, and invasion of water hyacinth. But some of the species that disappeared from Lake Victoria have been reported to occur abundantly in the Yale Swamp lakes (Aloo 2003). The heterotrophy resulted in adverse biodiversity and productivity in most aquatic and wetland systems in the River Danube (Cristofor et al. 2003).

The downstream water quality has been found to be related to the impact of lose strip (*Lythrum salicaria*) on the North American wetland which affected the nutrient cycling and decomposition rates of various other species (Emery and Perry 1996). The zooplankton index of wetland quality indicated that the pollution-tolerant open water taxa which included *Brachionus* and *Moina* were abundant in degraded wetlands (Lougheed and Chow 2002). Field studies conducted in 12 cutoff channel of River Rhone (France) showed that intermediate connectivity to river results in propagule input to wetland. The excessive connectivity impeded propagule input and insufficient connectivity eliminated less competitive species without replacement by another. The high nutrient level decreased species and intermediate nutrient favoured co-occurrence of species. The nutrient enrichment level, water turbidity, and connectivity of wetland to river affected the diversity and succession of species in wetland accordingly (Bornette et al. 1998).

7.8 Biological Monitoring and Impact Assessment

Bioindicators for ecosystem management provide information on the occurrence of ecological processes and structures. Bioindicators also provide information on the ecosystem condition if compared with a reference level of good ecological functioning and on cause–effect relationship within an ecosystem. The bioindicators for the river ecosystem and the lake ecosystem differed in types and responses (Lorenz et al. 2003). High diversity of phytoplankton composition due to flushing of the lake Alte Donau (Austria) with water of better quality has been reported; ciliates and metazooplankton constituted 10% of the total biomass. The high water temperature increases the dominance of cyanobacteria. The bacterial population was highest in November 1994 when the algal bloom finally ceased (Mayer et al. 1997). Phytoplankton biomass, dominance, grazing pressure, plankton diversity index and trophic state index are among the strong indicators of eutrophication in aquatic ecosystem (Moncheva et al. 2002). The anthropogenic forcing in Venice lagoon caused higher diversity of non-indigenous species. The human interventions in the Mediterranean Sea environment resulted in a fertile ground for mass bioinvasion of aquatic nuisance species capable of altering ecosystem equilibrium, ultimately resulting in global change (Occhipiniti and Savini 2003). The Shannon–Weaver and Whilm–Dorris diversity indices were not very sensitive parameters to detect moderate pollution. The saprobity indices of Pautle and Buck and diatom assemblage index of Waternabe et al. were useful in detecting intermediate organic load. The saprobity index of Waternabe et al. was found to be more precise for deteriorated part of river Rosario in Argentina (Sadusso and Morana 2002). Danilov and Ekelund (1999) worked out seven diversity and one similarity indices utilizing phytoplankton data for their usefulness in eutrophication studies in some central Sweden lakes. The lakes of different eutrophication level were monthly investigated during the autumn of 1998. Several diversity indices calculated failed to distinguish between different eutrophication levels. However, similarity index on the basis of the presence and the absence of matrices of phytoplankton species proved its high efficiency for eutrophication studies in the lake in central Sweden (Danilov and Ekelund 1999). The diversity and density indices were higher in saline alkaline ponds applied with organic fertilizer. The pond enclosures with fish population had relatively lesser diversity and density indices of phytoplanktons and zooplanktons (Zhao et al. 2000).

Out of 70 coastal and inland marsh samples during 1995–2000, the more open water was dominated by water pollution-tolerant taxa. However, wetland zooplankton index was found to be more useful than several diversity indices in Laurentian Great Lakes Basin (Lougheed and Chow 2002). The studies in the Lake Kasumigaura, Japan, showed that trophic state index ranged monthly between 55 and 75 and COD and total P were limiting factors. The Chl-a was found to be a suitable biological indicator of eutrophication in the lake (Wei et al. 2000). The algae diversity of Najafgarh drain in Delhi (India) receiving effluents from agricultural, industrial, and domestic sources indicated that the drain was excessively eutrophic. The large number of genera and species present in the drain was highly tolerant to organic pollution. These species have been suggested to be suitable for biomonitoring and phytoremediation (Sinha 2001). The diversity and abundance of plankton responded well to the fertilization of Cisterns (Barua and Ghosh 2003). The Lake Greifensee (Switzerland) in a dimictic lake was oligotrophic in the 1970s. The elimination of sewage treatment plants and the ban on P-containing detergents reduced the P load; the primary productivity decreased but showed a continuing eutrophic conditions. In this lake when the concentration of P was more than 250 mg m^{-3} phytoplanktons were dominated. There was reduction in population of diatoms and dinoflagellates. The population of cyanobacteria was almost negligible. Further increase in P concentration there was an increase in cyanobacterial biomass at the cost of dinoflagellates. The chrysomonads and pennate (diatoms) increased substantially. However, no change was recorded in zooplankton community. The main biomass of zooplankton was due to herbivorous crustaceans and only 20% were due to carnivores (Burgi et al. 2003). In Australian river systems the biofilms have been used as a suitable parameter to assess long-term functional changes. The biofilms are assemblages of algae, fungi, and microorganism which covers rocks, woods, and sediments in aquatic systems. This parameter facilitates in historical comparison with the previously recorded data sets of ecosystem function and structure (Burns and Ryder 2001). The plants of *Chlorella vulgaris* were detected in water samples with high nutrient content during eutrophication sensitivity experiment on some algae in three inland lentic resources of Bhopal, India. In these lakes, *Crucigenia crucifera* was highly tolerant to Ca, Mg, K, and P but moderately tolerant to nitrogen. Some other species of *Crucigenia*, viz. *Crucigenia quadricauda* and *Crucigenia rectangularis,* performed well in Ca-, Mg-, and K-rich models. The *Pediastrum* sp. grew well in calcified and nitrated water. The *Cosmarium granatum* was adversely affected by phosphorus and other nutrients as well. The *Merismopedia punctata*, *Melosira granulata*, and *Microcystis aeruginosa* were tolerant to highly eutrophic condition (Garg and Garg 2003). Eutrophication causes predictable increases in the biomass of algae in lakes, reservoirs, streams, rivers, and coastal lagoon ecosystems. Consistent and predictable eutrophication cause increase in cyanobacterial dominance of phytoplankton worldwide in natural lake. Similar trends were found both for phytoplankton in turbid reservoirs and for suspended algae in large river (Smith 2003). The fast-growing ephemeral algae in shallow coastal waters worldwide were considered to be the indicators of coastal eutrophication as studied in two shallow waterways in the west coast of Sweden (Sundback et al. 2003). In sewage sludge-treated Lake Dalavaya at Mysore (India), the metal toxicity due to input of heavy metals and cations in addition to soap and detergent carried in by sewage water caused eutrophication. The *E. crassipes* utilized Co and Cd as micronutrients and showed luxurious growth. The water hyacinth (*E. crassipes*) acted as bioindicator. The luxurious growth of water hyacinth affected the other life forms (Sujatha et al. 2001). A 50% reduction in submersed macrophytes population in past 30 years has been reported in the mesotrophic lake Mikolayskie (Poland) which is under increasing eutrophication process (Solinska and Symoides 2001). Submersed macrophytes have been suggested as an important tool for long-term monitoring of running water quality. The submersed macrophytes react equally to eutrophication and oligotrophication (acidification) of waters (Tremp et al. 1995). The *Potamogeton* sp. in 17 Danish lakes has been reported to be a suitable bioindicator of water quality. *Potamogeton filiformis* and *Potamogeton polygonifolius* are typical indicators of oligotrophic condition. Some other species common to oligotrophic condition are slow-growing *Potamogeton alpinus*, *P. lucens*, *Potamogeton praelongus*, and *Potamogeton perfoliatus*. The transient species of *Potamogeton* in these lakes are *Potamogeton aqualifolius*, *Potamogeton colouratus* (*Potamogeton coloratus*), *Potamogeton densus*, and *Potamogeton rutilus* (Sand et al. 2000). The dominance of *Ceratophyllum demersum* among the submersed plants, Lamnacae members among the floating plants, and *Typha* and *Phragmites* species among the coastal vegetations was found to be the indicator of increasing organic pollution in Tisza River, Serbia in Yugoslavia (Stojanovic et al. 1999). A wider spatial sampling including different channel types and systems was suggested to be required to analyse the usefulness of bryophytes as water quality bioindicators in irrigation channels (Martinez et al. 2002). In France the rivers are highly polluted with monthly average of ammonium and phosphates varying from 24 to 1,200 and 60 to 1,400 μg L^{-1}, respectively, which induce development of hypertrophic and pollution-tolerant macrophyte communities. The aquatic macrophyte communities were suggested to be the bioindicator of eutrophication (Schnitzler et al. 1996). In a eutrophic Shore Lake of Geneva (Switzerland), two methods of bioindication of water quality by macrophytes were compared, the macrophyte index based on nutrient loads and the saprobic index measuring organic pollution. The saprobic index was found sensitive to small-scale changes in species composition and reflected the changes in eutrophication (Lehmann and Lachavanne 1999).

Primary productivity was high at Celestron lagoon. It was found that indices based on nutrient concentration and phytoplankton biomass were useful indicators of trophic status in such coastal lagoons influenced by surface water (Herrera et al. 2002). In the Lake Sobygaard of Denmark, the radiometric dating to P^{32} of the upper 20 cm sediment core (1932–1993) indicated that sediment rate doubled in recent decades. The changes in chironomids were in close agreement with the changes in diatoms and macrophytes reflecting eutrophication process. A macrophyte succession was recorded from *Chara* to *Ceratophyllum* to *Potamogeton* dominance to presently complete loss of submersed vegetation and dominance by phytoplankton. The data were compared with the data of other 25 Danish lakes. A significant difference was recorded in chironomid assemblages among lakes in different macrophyte classes (Jeppesen et al. 1999). The diatoms have been recognized as bioindicators in temperate streams in Kathmandu valley, Middle Hills of Nepal, and northern India. In Kathmandu valley, richness and diversity increased with potassium, chloride, sulphate, and nitrate but declined with aluminium, iron, surfactant, and phenol. The species richness increased with Si, Na, and phosphate but declined with increasing pH, Ca, and Mg. Diatom assemblages in Kathmandu valley reflect water chemistry (Juttner et al. 2003). The silicate availability of river supply and strong tidal mixing seem to determine year-round dominance of diatoms over dinoflagellates in Urdaibai estuary, north Spain (Maria and Orive 2001). In the Netherlands, the diatoms did not indicate significant changes in fluoride concentration that occurred between 1934–1958 and 1989. The eutrophication, however, reduced rare species and increased the abundance of meso- to hypereutrophic species, particularly *Fragilaria berolinensis.* Considerable reduction in P concentration restored diatom assemblages of the 1930s (Dam and Mertens 1993). The specific conductance (salinity) and phosphorus were found to determine diatom community structure in drainages with high secondary salinization. The drainage with low nutrient but a wide range of salinity showed strong association with diatom taxa *Amphora coffeaeformis*, *Cymbella pusilla*, and *Entomoneis paludosa.* The streams with high nutrient contents were abundant of *Bacillaria paradoxa* and *Aulacoseira granulate*. Similarity indices depending upon presence and absence of matrices of phytoplankton are commonly used to determine the level of eutrophication (Danilov and Ekelund 1999). The diatoms indicated a long-term eutrophication trend in lower Lough Erne (Ireland, UK) and showed that changes in the geochemistry and diatom assemblages were expected to be linked with increased nutrient supply from sewage and diffused agricultural sources (Gibson et al. 2003). In the most organically polluted sites of Karasu River basin (Turkey), three diatom species, namely *Gomphonema parvulum*, *Nitzschia palea*, and *Navicula cryptocephala*, were found in high densities; COD, BOD, and concentrations of nutrients were co-related negatively with dissolved oxygen (DO). Except GI (generic index), all diatom indices were directly correlated with COD, BOD, DO, and nutrients. The diatom indices indicated that the Karasu River basin was eutrophic and organically polluted (Gurbuz and Kivrok 2002). The epilithic diatoms (Bacillariophyta) in two streams of Ontario (Canada) were found to be directly related to total nitrogen and phosphorus. The regression models showed its utility of indicating eutrophication in these two lowland water streams (Winter and Duthie 2000).

The chlorophyll in the Great Masurian Lakes of Poland was found strongly correlated with nitrogen and phosphorus indicating eutrophication. The chlorophyll–nutrient relationship varied with the trophic status of the lake (Kufel 2001). The pigment Chl-a was found to be a suitable biological indicator to show the trend of eutrophication in Lake Kasumigaura of Japan (Wei et al. 2000). The primary productivity of phytoplankton was found indicative of eutrophication and other functioning of the coastal ecosystem change in Neuse River estuary, NC, and Galveston Bay, Texas (USA). It was suggested that photopigment indicator can be routinely incorporated in assessing water quality, ecosystem structure, and function (Paerl et al. 2003). Modification with biocoenosis (ecological communities) is an inevitable result of dense human population which introduced an alien species into German water. Until 2003, 96 alien species have been permanently established as a result of anthropogenic activities. About every fifth species was invasive in nature (Nehring and Welling 2003). Studies on 39 streams in Victoria, Australia diatom communities were strongly correlated with land use practices

(Blinn and Bailey 2001). In Jaroslawieckie Lake, most habitats were eutrophic. The analysis of phytoplankton and bottom sediment shows the succession of algae, which correspond to the increasing trophic level of lake (Pelechaty et al. 1997). The radiometric dating of the upper 20 cm of the sediment core of the shallow, hypertrophic lake Sobygaard (Denmark) indicated doubling of the sediment accumulation rate in the recent decade. The changes in chironomid assemblages found directly related with changes in diatoms and macrophyte community (Broderson et al. 2001). The increased nutrient loading was found responsible for coastal eutrophication and climatic perturbations (drought, hurricanes, floods) when studied in Galveston Bay, USA (Paerl et al. 2003). Flooded and dry periods affected the accumulation of inorganic nitrogen in soil, periphyton, and water in Eden wetland in Quintana Roo (Mexico). The phosphorus in periphyton decreased at the end of flooded period but the loss and recovery of phosphorus in the wetland took place in each hydrologic cycle. The transition of lower Danube wetland towards hypertrophy affected biodiversity and productivity in wetland ecosystem. Aquatic macrophyte played a crucial role in ecosystem re-organization (Novelo and Tavera 2003). The increased nutrient loading causing eutrophication. There may be direct and indirect effects of eutrophication which resulted in one or two equilibrium states including (i) a turbid phytoplankton-dominated lake in a meso- or eutrophic state and (ii) a clear macrophyte-dominated lake in a meso- or oligotrophic state. The balance between these two states depends upon a bottom-up force determined by producers and top-down force determined by consumers (Lorenz 2003). The anthropogenic stresses and open water habitat increased exotic species abundance in inland wetland of southern Great Lakes. Vegetation removal and site disturbance caused plant invasion (Detembeck et al. 1999).

The experiments conducted on 220 wetland mesocosms revealed that after clipping the macrophyte species, richness did not affect respiration and biomass and vice versa. The presence of specific species scientifically affected macrophyte biomass, respiration, and the resilience and functioning of ecosystem (Engelhardt et al. 2001, Engelhardt and Ritchie 2001). The community ordination technique showed that specific conductance (salinity) and phosphorus interaction determined diatom community structure as studied in 39 streams located in drainage throughout Victoria, Australia (Blinn and Bailey 2001). High water transparency in the eutrophic zones in majority of lakes may be due to the utilization of plant nutrients by the charophytes and benthic diatoms, which is indicated by the data collected from Lough Bunny Lake of Ireland (Pybus et al. 2003). The biofilm assemblage of algae, fungi, and other microorganisms in the sediments of aquatic system are the functional parameters, which provide an integrated long-term measure of ecosystem functioning with structural attributes like biomass and diversity. Such studies are helpful in historical comparison of previously recorded data sets (Burns and Ryder 2001). The successions of floral and faunal diversity were noted in the river and lake ecosystems during 20 years up to 2001, which were caused by ecological conditions and industrialization in Moldova (Toderas et al. 2001). Total annual nitrogen retention has been found higher in eutrophic lake (20–30%) than in oligotrophic lake (<5%). The nitrogen retention in Newegian lakes, Eikeren watercourse, and Bjerkreim watercourse was limited by phosphorus (Berge et al. 1997). In the Netherlands, eutrophication in aquatic environment reduced macrophyte diversity and their former habitat was characterized by non-floating duckweeds like *Lemna* sp., *Spirodela polyrhiza*, and *Azolla filicoides* (Smolders et al. 1995). A mild and short period of eutrophication in Uatuma River (Brazil) resulted in a succession pattern of *Eichhornia* → *Utricularia* + Cyperaceae → *Salvinia*. This succession was observed after the closure of dam in 1987 at Baleina, Brazil (Walker et al. 1999; Ni and Xie 1999). The littoral primary producers were found benefited from the increased P input to a shallow mesotrophic lake. Some of the phosphorus fixed by primary producer was transferred to detrital pool and increased pigment concentration of sedimentary organic matter. The abundant aquatic macrophytes increase the resistance of lake to eutrophication by assimilating nutrient. Observable changes in phytoplankton communities were noted during 20 years as a result of loading from numerous cottages (Kauppila and Valpola 2003). Ecosystem resistance to a single spacer relies on tolerant species that can compensate for sensitive competitor and maintain ecosystem process, such as primary production. The resistance to additional stressors depends increasingly on species tolerance which is positively correlated (Vinebrooke et al. 2004).

7.9 Biological Restoration of Eutrophic Waters

Some species of aquatic plants, namely *C. demersum*, *E. canadensis*, and *Lemna minor,* were found highly efficient for the phytoremediation of polluted waters (Rice et al. 1997). In Najafgarh drain of Delhi (India), receiving effluents from agriculture, industrial, and domestic sources, 68 species of 32 genera (belonging to Chlorophyta, Euglenophyta, Chrysophyta, and Cyanophyta) have been suggested to be used for the control of water pollution (Sinha 2001). Utilization of plant nutrients by the charophytes and benthic diatoms of Lough Bunny, a Karst lake (Ireland), may be responsible for the water clarity and small phytoplankton population (Pybus et al. 2003). *Eichhornia crassipes* and *Salvinia auriculata* can remove nitrogen and phosphorus from the environment. *Eichhornia crassipes* exhibited the highest rates of nutrient reduction from waters. This information can be helpful in developing adequate management strategies for aquatic macrophytes in order to reduce the eutrophication processes (Petrucio and Estaves 2000). The aquatic weeds *Typha*, *Phragmites*, and *Glyceria* species were reported to remove nutrients from eutrophic water (Beltman et al. 1990). Harvesting of dominant macrophytes (*E. canadensis* and *Elodea nuttallii*) would be helpful to manage the process of eutrophication caused by a power plant (Wychera et al. 1990). The roots of water hyacinth were found to remove particulate matter and nitrogen in a natural shallow eutrophicated wetland (Billore et al. 1998). The potential of *E. crassipes* (water hyacinth) for phytoremediation of various pollutants in water has been determined (Xia and Xiangjuan 2006, Mishra et al. 2007, Mishra and Tripathi 2009). Water hyacinth is reported for its efficiency to remove about 60–80% nitrogen (Fox et al. 2008) and about 69% of potassium from water (Zhou et al. 2007). *Hydrodictyon reticulum* removed 67.3% nitrogen and 91% phosphorus over the 6-day study period under different environmental conditions. The removal rates of total nitrogen and total phosphorus were 70 and 50%, respectively (Wang et al. 1999). Some phytoplanktivorous fishes have been used in weed management and controlling eutrophication (Opuszyunski and Shireman 1995). Water peanut (*Alternanthera philoxeroides*) improves the transparency of eutrophic lake water (Wang et al. 1999).

Biological restoration measures have been employed in 17 Danish lakes. Macrophyte refuges and transplantation were reported to be the most successful (Phillips 1999). Dissolved phosphorus in irrigation drainage channel can be removed by planted floats. These floats are designed to implement horizontal spreading of aquatic plants on the surface of irrigation drains, fields, or treatment ponds. The float technique was opted for dissolved and residual phosphorus and also to facilitate the harvest of the standing crop. In the float technology the creeping stem water plants were utilized to remove soluble reactive phosphorus from the water column (Wen and Recknagel 2002). Seaweeds can remove up to 90% of the nutrients. Mass culture of commercially valuable seaweed species is likely to play an increasingly important role as a nutrient removal system to alleviate eutrophication problems due to fed aquaculture. Seaweed *Gracilaria* in intensive mariculture system could remove 50% of the dissolved ammonium released by the fishes in winter, increasing to 90–95% in spring. *Gracilaria* cultivated on ropes near a fish cage farm assimilated 6.5% of the released dissolved nitrogen (Troell et al. 1999). Three species of estuarine macroalgae (*Ulva rotundata*, *Enteromorpha intestinalis*, and *Gracilaria gracilis*) were used as biofilters for the removal of phosphate from the effluent of a sea bass (*Dicentrarchus labrax*). All the three species reduced substantial amount of P at low water flow (Martinez et al. 2002). The intensive fish farming in the Orbetello lagoon (Italian West Coast) had affected the environment resulting in a large-scale cycle development of opportunistic macroalgae. The flooded areas used for phytotreatment led to the growth of huge quantities of macroalgae which in turn removed nitrogen and phosphorus from the effluent (Porrello et al. 2003). It is reported that partial recovery from an algae to a macrophyte-dominant state in a eutrophic freshwater system required managed phosphorus limitation and unmanaged macrophyte growth (Conveney et al. 2002). Highest nitrogen and phosphorus removal was recorded from Swine lagoon by *L. minor*. Duckweed-mediated removal of nitrogen and phosphorus was effective at low nutrient concentration (Cheng et al. 2002a). The aerial microalga *Trentepohlia aurea* was investigated and found to have high wastewater purification potential. It had the ability to remove nitrate, nitrite ammonium, and phosphate ions from the wastewater. The biomass of this species was also found 1.5 times higher in medium with higher N and P sources than in ordinary medium with low nutrient value. The removal ratio of nitrite and nitrate peaked in 30-day culture reaching 37 and 32%, respectively. It shows that *T. aurea* has the potential for use in the purification of wastewater (Abe et al. 2002). Duckweed species are promising macrophytes for use in sustainable wastewater treatment due to their rapid growth case of harvest and feed potential as a protein source. Ammonia both ionized and unionized forms are toxic for duckweeds. But they can be used for ammonium-containing wastewater treatment up to a certain pH level (Korner et al. 2003). Abundant aquatic macrophytes of Lake Kaljasafari have made the lake resistant to eutrophication of assimilating nutrients providing refuges from zooplankton and having an allelopathic effect on phytoplankton (Kauppila and Valpola 2003). Exploiting allelopathy between organisms suggests ecological friendly ways to restore the original aquatic ecological system in the Donghu Lake, China (Ruley and Rusch 2002).

Periphyton removes phosphorus in shallow freshwater ecosystem. The phosphorus removal process includes uptake, its deposition, and filtering from the water (Dodds et al. 2009). In domestic wastewater treated with duckweeds, 77% phosphorus and 94% nitrogen removal were observed every fourth day. Periphyton also removed 18% phosphorus and 50% nitrogen (Vermatt and Khalid 1998).

Nutrient removal capacity of some important macrophytes, i.e. *E. crassipes*, *L. minor*, and *Azolla pinnata*, has been tested individually as well as in combination. Maximum removal was observed in combination studies involving *E. crassipes* and *L. minor* (Tripathi et al. 2003). *Lemna minor* is efficient in removing solids and nutrients from the wastewater (Rose and Sedwin 2000). Aquatic weeds like *Lemna*, *Spirodela*, and *Eichhornia* were found very efficient in removing nutrients from eutrophic waters if used in controlled environmental conditions (Ansari and Khan 2008, 2009b). Duckweed (*L. minor*) was cultured in laboratory under controlled environmental conditions for 14 days. The plants remove 7% of nitrogen and 10% of phosphorus from the nutrient medium (Smith et al. 2004). The primary productivity of aquatic plants is considered to be a strong indicator of eutrophication (Smith 2007). Duckweed (*L. minor*) was reported to remove 73–97% of nitrogen and 63–99% of phosphorus from nutrient medium in a 3 day's experiment. *Lemna gibba* can remove 30–47% of the total nitrogen. When grown together they accounted for 35–46% and 31–71% of the total N and P removal (Korner and Vermatt 1998). Grass carp farming in Chinese shallow lakes has been reported to remove large amounts of bionutrients from the lake through macrophyte harvesting and thus inhibit secondary pollution and eutrophication (Li et al. 1998). Rehabilitation of submersed macrophytes could efficiently reduce the nutrient level and thus control the overgrowth of the undesirable algae and improve water quality in eutrophic lakes (Zhenbin et al. 2003). During autumn and winter, macrophytes could increase the transparency twice. Total nitrogen concentration decreased by 43.7% in controlled enclosures and 59.4% in open wastewaters, respectively, after 3 weeks and 61.6 and 79.7% after 6 weeks. In controlled and open wastewater, the total phosphorus concentration decreased by 50.3 and 57%, respectively, after 3 weeks (Cheng et al. 2002). In Trout Lake, Wisconsin (USA), biogenic silicon analysis indicated that relatively high rates of phosphorus removal were associated with the sedimentation of siliceous algae (diatoms) from the water column. This nutrient removal through diatom sedimentation indicated that this process can reduce primary production by decreasing the amount of nutrient remineralization in the water column during the stratified period (Poister and Armstrong 2003). Freshwater organisms also reflect quality of water bodies and can thus be used to monitor changes in ecosystem health (Ansari and Khan 2002, Xu et al. 2007).

7.10 Engineered and Technological Correctives

Attempts have been made to reduce nutrient loading in Danish lakes by intervening at the source level and improving the retention capacity of catchment areas. The former measures include phosphorus stripping and nitrogen removal at sewage works, increased use of phosphate-free detergents and regulations concerning animal fertilizer storage capacity, fertilizer application practices, and fertilization of plants and green covers in winter. In order to improve nutrient retention capacity of catchments, wetlands and lakes have been re-established and channelized streams have been re-meandered. These measures have resulted in 73% reduction of total phosphorus (Phillips 1999).

Various strategies to control the growth of the dominant phytoplankton of the Lake Taihu (China) have been proposed so as to check the eutrophication. These strategies include cutting down the exogenic loading, dredging the sediments, collecting and treating the harvested algal blooms, ensuring the recovery of aquatic plants in suitable places, and proper fish culture (Weimin et al. 1997). PhoslockTM is used for phosphorus binding where the phosphorus (P) is released from sediments which enhances the algal bloom in water. PhoslockTM applied in slurries from two western Australian waterways resulted into reduction in dissolved P concentration in water (Robb et al. 2003). Inputs of nutrients in coastal watersheds must be managed to protect the diversity and functioning of the full range of estuarine marshes (Crain 2007). Changes in macrophyte communities due to eutrophication can be quantified using indices, such as the TI and the LTR indexes. Especially in lakes known to be only exposed to eutrophication pressure, such as is often the case in the Nordic GIG lakes, the macrophyte community changes are reflected in the resulting index values (Penning et al. 2008). An identification of limiting nutrients is necessary for the selection of appropriate nutrient control measures to reduce nutrients as well as to slow down eutrophication process in aquatic systems. Combining the information from both water quality parameters in situ and nutrient enrichment bioassay in laboratory could provide an integrative approach that would enable extrapolation of laboratory results to the real environmental conditions (Persic et al. 2009). Extensive use of phosphate-based detergent is one of the main causes of worldwide eutrophication of rivers and lakes (Ansari and Khan 2005). For the control, non-phosphorus-containing complexing agents are practiced in several countries and have been suggested to substitute phosphate-containing detergents (Egli et al. 1990). It is estimated that a complete ban on phosphates in detergents would remove about 20–30% of the phosphates in sewage. In several countries, a complete ban on or a reduction in the amount of phosphate in detergents has been tried to reverse eutrophication (Sweeney 1979). Detergent phosphorus bans in Maryland and Michigan states of America are reported to have significantly reduced phosphate loading to the Chesapeake Bay and the Detroit River, respectively (Penelope and Charles 1992). The P concentration of Lake Greifensee (Switzerland) declined due to P elimination in sewage treatment plants and a ban on P-containing detergents (Burgi et al. 2003).

Assessment and control of eutrophication in the UK have proceeded to focus mainly on certain localized problems or on the implementation of ECD directive, notably those relating to the control of urban wastewater and agricultural nitrate, which provide only partial solution to the problem. The Environmental Agency of England and Wales planned to develop a strategy on eutrophication using a well-coordinated and integrated approach. Key components of strategy planned were related to management, communication, and research and development. The important feature of the regulatory initiatives was to make its best use. However, promoting a partnership approach to the management of eutrophication is equally important (Leaf and Chatterjee 1999). The anticipated increase in C:P ratios depends on lake-specific properties such as morphometry, water residence time, water temperature, and trophic structure (Donk et al. 2008). On the other hand, lake managers have opted increased macrophyte abundance as control measure to restore eutrophic water bodies. It was aimed to improve water quality, increase water transparency, and reduce phytoplankton biomass. Low levels of macrophyte coverage; phytoplankton biomass was only limited by nutrients (total phosphorus and Si) availability while nutrient (Si) as well as grazing (zooplanktons and *Daphnia*) factors was both limiting phytoplankton levels in macrophyte-free water (Lau and Lane 2002b). The ecosystem of shallow lakes is analysed with PCLake, a dynamic model of nutrient cycling and biota including phytoplankton, macrophytes, and a simplified food web. The model was used to calculate the switch points in terms of critical phosphorus loading levels for a number of lake types. It turned out that the predicted critical phosphorus loadings differ per lake type. The results may be useful for lake management by comparing the critical loadings for a given lake with the actual loading (Janse et al. 2008).

To control the internal nutrient release, physical, chemical, biological techniques, and even bionic techniques could be selected. The idea of ecological restoration for a eutrophic lake is to shift the ecosystem from phytoplankton-dominant state to macrophyte-dominant state. To realize the shift of ecosystem state, environmental condition improvement is the fundamental work (Boqiang et al. 2006). The presence of excessive amounts of biologically reactive nitrogen in coastal bays and estuaries has been recognized as a major environmental issue. For the most part, this is due to its role in enhancing phytoplankton growth that, in turn, can lead to algal blooms, oxygen consumption in seawater and on the seabed, altered patterns of primary productivity, and changes in species composition. There is a lack of coherent monitoring programmes that can provide a continuum of observations from the watershed to the coastal ocean and in different environmental matrices to calculate or simulate source attribution of analytes of concern. Even after years of focused research on the consequences of nitrogen over-enrichment in US coastal waters and estuaries, several key issues remain, including the magnitude, spatial, and temporal extent of eutrophication and associated water quality and habitat impairments attributable to increasing nitrogen inputs to estuaries and coastal waters (Hameedi et al. 2007). Three potentially complementary ecotechnological measures are

- prevention of sediment resuspension in lakes by creating islands in order to minimize the wind fetch to reduce the wave amplitude;
- allowing greater water-level fluctuations (WLFs) in lakes as planned in lowland countries like the Netherlands; WLFs are likely to allow more space for water and may lead to improved water quality and higher biodiversity;
- the grazer populations, herbivorous zooplankton to regulate phytoplankton, particularly to control cyanobacterial blooms (Gulati et al. 2008).

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Chapter 8

Eutrophication Problem in Egypt

Mohamed M. Dorgham

Abstract The coastal area of Egypt on the Mediterranean Sea extends for about 1,200 km. It hosts a number of important residential and economic centres, like the cities of Alexandria, Port Said, Damietta, Rosetta, Matruh, and AL-Arish. The coastal strip between Alexandria and Matruh hosts tens of tourist villages, which are usually crowded by visitors during summer. Many activities are known in the coastal area, including fishing, industrial, tourism, trading and agricultural, oil and gas production, and transportation. There are five large ecologically different coastal lagoons connected to the sea coast, representing together about 25% of the total area of the Mediterranean wetland. These lagoons are considered as reservoirs for agricultural, industrial, and municipal wastes, which are discharged from surrounding cities and cultivated lands. The Egyptian Mediterranean coast receives huge volumes of wastewaters every year through the coastal lagoons and from other land-based effluents. These wastes are loaded by variable amounts and types of pollutants, in addition to great amount of nitrogenous and phosphorous compounds, which in turn cause high level of eutrophication along a significant part of the Mediterranean coast, particularly of both the Nile Delta region and Alexandria coast. Eutrophication is an importunate problem to the Egyptian Mediterranean coast, resulting in fundamental changes in the structure of the planktonic and benthic communities as well as fish mortality. Eutrophication was accompanied by the appearance of several harmful algal species at several hot spots along the coast. The level of eutrophication demonstrated wide variation along the Egyptian coast relative to the variations in the volume and contents of discharged wastes.

Keywords Eutrophication · Egyptian mediterranean coast · Abu qir bay · Mex bay · Eastern harbour · Dekhaila harbour · Western harbour

8.1 Introduction

There is clear evidence that nutrient loading to lakes, estuaries, and coastal oceans has greatly increased through human activities over the past few decades and that this has caused or enhanced many of the symptoms of the aquatic ecosystem transformation known as eutrophication (Bishop et al. 2006). According to recent suggestion of Andersen et al. (2006), eutrophication can be defined as the enrichment of water by nutrients, especially nitrogen and/or phosphorous and organic matter, causing an increased growth of algae and higher forms of plant life to produce an unacceptable deviation in structure, function, and stability of organisms present in the water and to the quality of water concerned, compared to reference conditions. Eutrophication is a serious environmental and economic problem in coastal marine ecosystems worldwide. In general, the results of nutrient over-enrichment tend to be negative, with beneficial effects being rare or accidental (Fisher et al. 1995). In particular, eutrophication often has a strongly negative economic dimension (Segerson and Walker 2002). The economic consequences of estuarine and coastal marine eutrophication can be very substantial and are expected to increase over time worldwide as human

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population numbers grow and move into coastal communities (Andersen et al. 2006). Eutrophication has been recognized as an environmental problem for aquatic habitat in Europe and North America since the mid-twentieth century (Rodhe 1969) and becomes more widespread all over the world, whereas 54% of lakes in Asia are eutrophic; in Europe 53%; in North America 48%; in South America 41%; and in Africa 28% (ILEC 1988–1993). Eutrophication also appears as a natural process in numerous inland waters due to climate change, geology, and other external influences (Whiteside 1983, Walker 2006). The investigation from the UNEP (United Nation Environment Programme) indicates that about 30–40% of the lakes and reservoirs have been affected more or less by water eutrophication all over the world resulting from anthropogenic nutrient enrichment. The human activities caused eutrophication in Erie Lake (Reutter 1989); in USA in Washington Lake (Welch and Crooke 1987), Okeechobee Lake (Schelske 1989), City Park Lake (Ruley and Rusch 2002); the majority of Danish lakes (Jeppesen et al. 1999); and three lakes of Bhopal in India (Garg et al. 2002). Eutrophication constitutes a serious threat to many European lakes (Søndergaard et al. 2007), South African lakes (de Villiers 2007), and Chivero Lake, Zimbabwe (Nhapi 2004). In China, water eutrophication occurred in 67 lakes (51.2% of the total lakes) and Dianchi Lake in Yunnan is possibly the most hypertrophic lake in the world (Lu et al. 2005). Human activities can maximize eutrophication problem by increasing the amount of nutrients entering the aquatic ecosystem. It was reported that 80% of lake and reservoir eutrophication is restricted by phosphorous, about 10% of lake and reservoir eutrophication is relative to nitrogen, and the rest of 10% of lake and reservoir eutrophication is relative to other factors (Zhao 2004). In freshwater lakes, phosphorous is often the main cause of eutrophication, since great amounts of phosphorous are introduced into the aquatic habitats through intensive application of agricultural fertilizer. According to Carpenter et al. (1998), between 1950 and 1995, 600 million tons of phosphorous was applied to Earth's surface, primarily on croplands. In coastal waters, eutrophication is mainly caused by nitrogen enrichment and estuarine areas are usually eutrophic due to high nutrient enrichment of the marine environment through the land runoff. In addition, anthropogenic nitrogen can enter the seawater from the atmosphere, accounting for about one-third of the ocean's external (non-recycled) nitrogen supply and up to 3% of the annual new marine biological production (Duce et al. 2008). The widespread increased nutrient loading to freshwaters has also led to increased nutrient loading to coastal areas worldwide, thereby transferring the eutrophication problem to the coast (Nixon 1995). The great attention given to coastal eutrophication was raised from the widespreading of eutrophication symptoms over the world coasts. Eutrophication has caused major problems in both the Baltic Sea (Cederwall and Elmgren 1990, Schulz and Nehring 1991), the Mediterranean Sea, particularly the Adriatic Sea (Nespoli 1988, Vollenweider et al. 1992) and the Black Sea (Gomoiu 1992). Water eutrophication is widespread all over the world and the severity is increasing, especially in the developing countries like China (Xiao-e Yang et al. 2008). The growth of the research effort on coastal eutrophication worldwide reflects the increased incidence of eutrophication as one of the major threats to the integrity of coastal ecosystems (Vollenweider 1992, Nixon 1995).

Eutrophication of inland waters has been a major problem affecting the quality of freshwater bodies over the past four decades (Vollenweider 1968). The algal bloom corresponds to high nutrient levels leading to generation of more organic matter, which on decomposition causes decrease of dissolved oxygen. Human activities in the watershed resulting in eutrophication can lead to loss of dominant species and functional groups, high nutrient turnover, low resistance, high porosity of nutrients and sediments, and the loss of productivity (Liu and Qiu 2007). Fish kills have occurred in the northern Adriatic as a result of noxious phytoplankton blooms and anoxic conditions, while in the northern Mediterranean, there is a decline of fish species and species diversity due to anthropogenic impacts of untreated sewage and an increased incidence of toxic blooms (Caddy et al. 1993). Planktonic blooms and sewage contamination of coastal waters have also caused health problems associated with the ingestion of contaminated shellfish (UNEP/FAO 1990, Caddy et al. 1993). Some algal blooms are toxic to plants and animals and their toxins find their way to the food chain, resulting in animal mortality (Anderson 1994), and pose neuro- and hepatotoxin threat to humans (Lawton and Codd 1991, Martin and Cooke 1994), through shellfish poisoning (Shumway 1990). The

World Resources Institute has identified 375 hypoxic coastal zones in the world, concentrated in coastal areas in western Europe, the eastern and southern coasts of the USA, and east Asia, particularly in Japan (Selman and Mindy 2007). Eutrophication problem on the Egyptian Mediterranean coast is attributed mainly to great volumes of wastewaters discharged through land-based effluents, derived not only from the Nile River water but also from agriculture and sewage water through coastal lakes that are connected directly or indirectly to the Mediterranean coastal area. These effluents are loaded by variable amounts of nutrient salts, which promote intensive phytoplankton growth. The level of eutrophication appeared to be significantly different along the Egyptian coast, due to the variability of quality and quantity of discharged wastes as well as their loads of nutrients. The high nutrient levels and high chlorophyll concentration as indication of eutrophication were reported at several areas along the Alexandria sea coast, like Mex Bay, Western Harbour, Dekhaila Harbour, Eastern Harbour, Abu Qir Bay, and others (Fig. 8.1). However, each of these basins is characterized by its own environmental conditions and sources of eutrophication.

Despite its negative effect on the coastal environment, the anthropogenic nutrients lead to improvement of the Egyptian fisheries, particularly during the past few years. The highly productive coastal Mediterranean fishery of the Nile River delta collapsed after the completion of the Aswan High Dam in 1965 while they have been recovering dramatically since the mid-1980s, coincident with large increases in fertilizer application and sewage discharge in Egypt (Oczkowski et al. 2009). A recent assessment of potential anthropogenic nutrient sources in Egypt also suggested that these sources may have more than replaced the fertility carried by the historical floodwaters (Oczkowski et al. 2009). Public water and sewer systems have expanded greatly (Nixon 2003), and annual fertilizer consumption has increased almost fourfold, from 3.4×10^5 to 13×10^5 tons (FAO 2008). Before 1965, the Nile flood delivered about 7×10^3 tons year^{-1} of nitrogen and $7–11 \times 10^3$ tons year^{-1} of phosphorous to the Mediterranean coast (Nixon 2003). Today, the Rosetta branch of the Nile alone discharges almost three times more dissolved inorganic nitrogen per year (2×10^4 tons) and about half as much bioavailable phosphorous (4×10^3 tons) into this oligotrophic region, and there are seven other major and countless minor drainage points along the coast (Abdel-Hamid et al. 1992, Awad and Youssef 2009). According to Hamza et al. (2003) the rate of discharge of main sources of the land runoff reaches high values during winter, particularly the Nile flood (through the Rosetta branch), which is

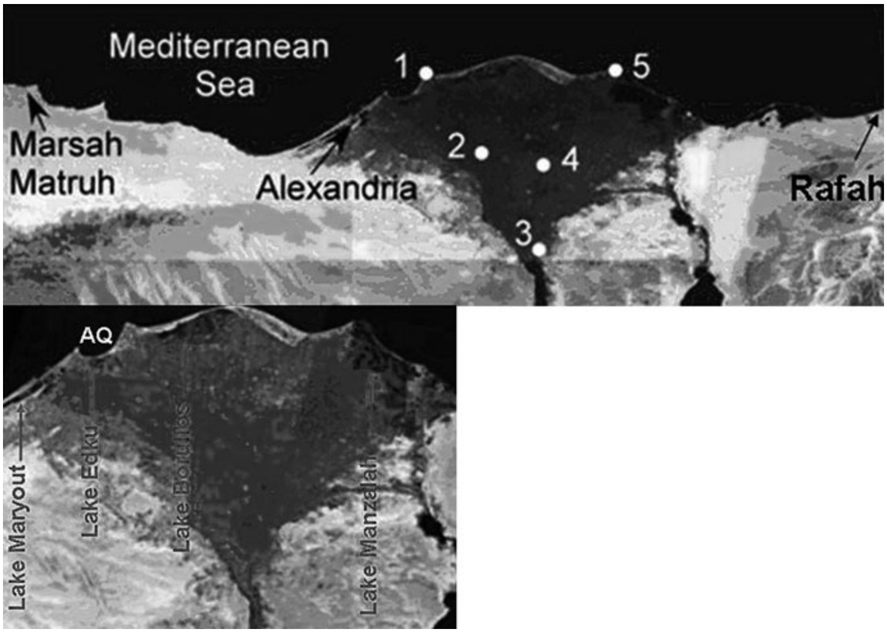

Fig. 8.1 Egyptian Mediterranean coast and coastal lagoons

responsible for up to 90% of the annual discharge, and the annual total load of nutrient salts from these sources is estimated to be 676.4 tons year^{-1} of dissolved nitrogen and 84.9 tons year^{-1} of dissolved phosphorous. These loads are carried by 17.1 km^3 year^{-1} of discharge water from various land runoffs, giving an overall flow rate of 453 m^3 s^{-1}. As compared to the offshore water, the near-shore water of the Egyptian Mediterranean is characterized by great variability of surface salinity, which in turn reflects the levels of nutrients as well as the N:P ratios. Since the surface salinity is a key factor moderating all environmental changes in the coastal waters, relative to the changeable volume of discharged wastewaters, the amounts of nutrient salts as well as phytoplankton biomass are expected to be different at the different areas along the Egyptian Mediterranean coast. Consequently the water quality and eutrophication level will be different at the different locations. In the following parts, we will discuss the environmental conditions of the more eutrophic embayment along the Egyptian coast.

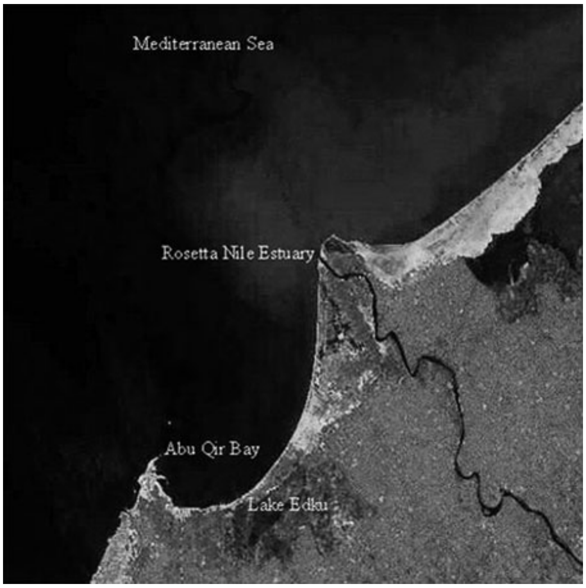

Fig. 8.2 Abu Qir Bay

8.2 Abu Qir Bay

Abu Qir Bay is a shallow semi-closed embayment lying about 20 km east to Alexandria city, between longitudes 30°03′ and 30°22′E and latitudes 31°16′ and 31°28′N. At the northeastern side, the bay is bordered by the Rosetta mouth of the Nile and at the southwestern side by Abu Qir headland, which recently was extended further seaward due to the construction of Abu Qir harbour. The average depth of the bay is 10–12 m with a maximum of 18 m at the central part of the bay–sea borders and a total area of 500–600 km^2, and water volume varies in literatures between 5 and 6 km^3 (Fig. 8.2).

Abu Qir Bay is one of the most ecologically and biologically important embayments along the Mediterranean coast of Egypt. The ecological importance of the bay is attributed to the fact that it receives several types of agricultural, domestic, and industrial wastewaters that cause fundamental changes in its water quality. The bay is also one of the high biologically productive basins; it was and is still considered among the highly fertile Egyptian coastal regions and provides suitable habitats for breeding of many commercially important fishes and growing fish fry. Due to the stress of discharged wastewater, the Bay became a hot spot area, suffering from acute eutrophication and other pollution sources. There are more than 5 × 10^5 m^3 of untreated domestic and industrial wastes discharged daily to Rosetta mouth of the Nile River, in addition to great volume of agricultural wastewaters loaded by pesticides and nutrients. Large amounts of these wastes reach Abu Qir Bay through an annual discharge from Rosetta branch of about 4 × 10^9 m^3 after the construction of the High Dam. The bay also receives about 3.5 × 10^6 m^3 day^{-1} of agricultural wastes from Lake Edku, besides discharged wastes from numerous paper mills, refineries, and chemical fertilizer plants (about 2 million m^3 day^{-1}) from El-Tabia pumping station. Furthermore, other sources of pollution are considered as source of threat to the bay, like domestic wastes of El-Maadi city and other villages as well as other urban activities, oil pollution from the fishing boats, the waste products of ammonia export, the water cooling of Abu Qir Electrical Power Station, and export of natural gas at Edku, southwest of Rosetta. The surface salinity in Abu Qir Bay appeared to be widely variable with time and space, relative to changes in the volume and direction of the discharged wastewaters. Abdel-Aziz et al. (2001) found that the surface salinity fluctuates between a minimum of 6.42‰ ppt in front of El-Tabia pumping station in March and a maximum of 39.05‰ ppt at the offshore water in August. The latter authors observed two water masses in the bay: one is restricted to the coastal strip, in the vicinity of the land-based effluents, with monthly average salinity <33.5‰, and the other one is found in the offshore part, with average salinity between >35 and 36.3‰. However, offshore water sustained higher salinity, similar to those usually found in open sea (<38.5‰). On the other hand, the effect of discharged wastewaters on the surface salinity is controlled by the direction and speed of water current in the bay, which is mostly directed seaward during spring and summer, and shows reverse direction in winter (Mohamed 1981, Abdel-Aziz and Dorgham 1999). Abu Qir Bay sustained low transparency over the year, mostly lower than 173 cm, particularly around the land runoff. The strong water current from both El-Tabia pumping station and Lake Edku is the main reason of vertical mixing of the water column, and consequently high turbidity in the bay, in addition to dense plankton population (Abdel-Aziz et al. 2001, Shams-El-Din and Dorgham 2007).

The bay water is usually well aerated, particularly in the part away from the land runoff. But low values of dissolved oxygen (down to 0.6 mg L^{-1}) usually here are reported over the year in front of Abu Qir Company of fertilizers and near El-Tabia pumping station. The dissolved oxygen was mostly lower than 4.5 mg L^{-1} over the year in the coastal waters under the stress of the land-based effluents. In few occasions, high concentrations of dissolved oxygen could be measured at the surface relative to the high phytoplankton biomass (Abdel-Aziz et al. 2001). The bay seems to be a highly eutrophic basin, since it sustains abnormally high concentration of nutrients. However, nutrient salts demonstrated exceedingly wide variations with time and space.

Nitrate attained pronouncedly high concentration of up to 51.11 μM, but three levels were observed over the year. The highest one occurred in August, moderately high during spring, November, January, and March, while comparatively low level was measured during the other months of the year (Abdel-Aziz et al. 2001). Recent observations (EEAA 2003, 2004, 2005) demonstrated lower concentrations of nitrate but it is still exceeding those given by Franco (1983) and Marchetti (1984) as criteria for eutrophication. Ammonia appeared to attain the highest concentrations and the widest range of variations as compared to other nutrients in the bay during 1999–2000, fluctuating between undetectable level and 338.5 μM, which was measured near the outflow of Tabia pumping station, in the vicinity of sewage water and industrial wastes (Abdel-Aziz et al. 2001). These authors recorded high monthly average concentration during spring (19.16 and 29.78 μM) and early winter (28.72–50.88 μM). The high level of ammonia in Abu Qir Bay is a good indication of great load of organic matter, which is derived mainly from the domestic wastewaters. On the other hand, the areas far from the land runoff usually sustained pronouncedly low ammonia as compared to the coastal strip, especially those observed during the period 2003–2005 (Table 8.1). The reactive phosphate recorded abnormally high concentration (14.64 μM) near El-Tabia pumping station, but the monthly average values fluctuated between 0.32 and 1.86 μM most of the year, the high values being found in the coastal waters (Abdel Aziz et al. 2001). Relative

Table 8.1 Several year observations of minimum, maximum, and average values of different environmental parameters in Abu Qir Bay

	Abdel-Aziz et al. (2001)	EEAA (2003)	EEAA (2004)	EEAA (2005)
	1999–2000	2003	2004	2005
S‰	33.41	–	–	–
Secchi depth (cm)	123.5	–	–	–
DO (mg L^{-1})	3.67	7.37	5.71	5.89
Nitrate (μM)	7.16	6.15	4.31	5.04
Ammonia (μM)	14.55	2	2.21	2.7
Phosphate (μM)	1.41	0.43	0.65	1.31
N/P	5.08	14.3	6.6	3.8
Silicate (μM)	16.74	11.71	–	–
Chl-*a* (μg L^{-1})	15.71	9.2	6.98	6.31
Phaeopigment (μM)	4.77	–	–	–

to the pronounced variability of inorganic compounds of nitrogen and phosphorous in the bay, N/P ratio showed exceptionally wide changes range of variations (0.07–79.9) and monthly average sustained high values (18, 21.3 & 15.6) in May, August and November respectively and comparitively low values (2.9–8.8) during most of the year (Abdel Aziz et al. 2001). During 2003–2005 the N/P ratio was comparatively low (Table 8.1). Silicate demonstrated high concentrations in the bay during 1998–1999, varying within the range of 11.6–33.34 μM, except the relatively low values (1.09–6.48 μM) in April, July, and November (Abdel Aziz et al. 2001). From the phytoplankton biomass it seems that the bay is highly productive all the year round. According to Abdel Aziz et al. (2001) the monthly average concentration of chlorophyll-*a* varied between 2.06 and 52.64 μg L^{-1}, demonstrating two distinguished peaks during spring and September, with high biomass (annual average: 19.4–24.8 μg L^{-1}) in front of the Lake Edku runoff (15.4–17.4 μg L^{-1}) in the vicinity of El-Tabia pumping station and Abu Qir Company of fertilizers. The other parts of the bay contained comparatively low chlorophyll (7.6–13.9 μg L^{-1}), but still higher than the eutrophication limits.

The seasonal variability in the volume and quality of the different wastewaters and their dispersion is of great importance in dynamics of the ecosystem of the bay. The wide variability of N/P ratio in Abu Qir Bay (up to 79.9) indicates that nitrogen supply is pronouncedly greater than phosphorous supply. The long-term observations revealed that dissolved inorganic nitrogen and phosphorous as well as the N/P ratio increased significantly in the past few years (Fahmy 1997, Gharib and Soliman 1998). This is related to the widespread utilization of nitrogenous fertilizers in agriculture in the adjacent cultivated land, in addition to the increase of the activities of the fertilizer company. In eutrophic waters, Secchi depth reading is usually less than 245 cm and chlorophyll-*a* is 14.3 μg L^{-1} (Wetzel 1983), NO_3–N amounts to 4 μM, NH_4–N amounts to 2 μM (Franco 1983), and PO_4–P amounts to 0.3 μM (Marchetti 1984). Relative to this criteria the annual average values of similar parameters in Abu Qir Bay reflect high eutrophication. It has been shown that the eutrophication or the red tide occurs when N concentration in water reaches 300 μg L^{-1} and P concentration reaches 20 μg L^{-1}. Richardson et al. (2007) reported that exceeding a surface water mean TP threshold concentration of 15 μg L^{-1} causes an ecological imbalance in algal, macrophyte, and macroinvertebrate assemblages as well as slough community structure in the Everglades areas. The eutrophication conditions in Abu Qir Bay reflected not only on the water quality but also on the biological ecosystem, particularly plankton community. The phytoplankton community structure experienced serious changes, particularly the increase in the number of freshwater assemblages, like those of Chlorophyceae, Cyanophyceae, and Euglenophyceae (Shams El-Din and Dorgham 2007). Chlorophyceae and Cyanophyceae are usually favored by increased nutrients and dissolved organic material characterizing areas subjected to freshwater discharge (Stirn 1988).

The deterioration of the water quality in the Abu Qir Bay during the past two decades leads to pronounced decrease in the phytoplankton count, whereas Shams El-Din and Dorgham (2007) reported markedly low count (annual average: 5 × 103 unit L^{-1}) as compared to that (36.4 × 10^3 and 780 × 10^3 unit L^{-1}, respectively) of El-Sherif and Gharib (1994) and El-Sherif and Mikhail (2003). However, the high chlorophyll-*a* concentrations are attributed in major part to picoplankton cells (Shams El-Din and Dorgham 2007). Of a few species responsible for the main bulk of phytoplankton count, two species are considered as indicators of eutrophication, namely the diatom *Nitzschia pungens* and the dinoflagellate *Prorocentrum micans* (Huang et al. 2004, Dorgham et al. 1987). The latter species is well known in all eutrophic areas along the Egyptian Mediterranean waters (Ismael and Dorgham 2003, Gharib and Dorgham 2006). The phytoplankton community experienced pronouncedly different patterns of species dominance in the bay even on the short-term scale. During the period of April 1998–March 1999, *P. micans* dominated in May, *N. pungens* in September, and the diatoms *Asterionella japonica* and *Thalassionema nitzschioides* in November (Shams El-Din and Dorgham 2007), while from December 1999 to November 2000, the green algae *Pediastrum clathratum* and *Pediastrum duplex* and the diatom *T. nitzschioides* were dominant in May, *Nitzschia longissima* and *A. japonica* in September, and the latter two species extended their dominance with the diatoms *Chaetoceros* sp. and *Leptocylindrus danicus* in November (El-Sherif and Gharib 1994). On the other hand, all dominant species reported in March and June by Samaan and Mikhail (1990) were completely absent from the study of Shams and Dorgham (2007). Furthermore, in 1997, Gharib and Dorgham (2000) reported high count of the fresh and brackish water forms of diatoms *Cyclotella meneghiniana*, *Nitzschia palea*, *Nitzschia microcephala*, *Cyclotella glomerata*, and *Skeletonema costatum* and the green algae *Scenedesmus quadricauda* and *Scenedesmus bijugatus*. However, later on these species were found either rarely or in low counts (Shams El-Din and Dorgham 2007). *C. meneghiniana* is considered as eutrophic indicator (Dorgham 1997, Gharib and Dorgham 2000). On the other hand, the phytoplankton community showed pronounced change in species composition as well as in the peak timing. Shams El-Din and Dorgham (2007) recorded 182 species with main peak in September, while EL-Sherif and Mikhail (2003) identified 241 species with main peak in May. Furthermore, the dominance of species in the two studies was mostly different (Table 8.2). Eutrophication may cause many effects, such as decreased biodiversity, changes in species composition and dominance, and toxicity effects.

The effect of eutrophication in Abu Qir Bay appeared also on the diversity index of phytoplankton, which experienced wide fluctuations (0–3.12 nats), indicating the instability of the area due to the effect of the discharged wastes (Shams El-Din and Dorgham 2007). This range is clearly wider than that (1–2.5 nats) found by Margalef (1964, 1978) for the actively growing coastal populations and eutrophic lakes. The diversity index is a suitable indication for water quality (Gharib and Dorgham 2006). Zooplankton community seems to be stressed by eutrophication in the bay. Copepods and tintinnids form the major zooplankton component in the bay as well as along the coastal waters of the Egyptian Mediterranean. The two groups are usually represented by low numbers of species (15–23 species and 13–46 species) in different near-shore waters and large numbers (132 and 99 species) in offshore waters. In the bay, copepods comprised 23 species and tintinnids 35 species. The number of species varied monthly throughout the bay, coincided with the direction and velocity of the water current, entering from Lake Edku through Boughaz El-Maadiya. However, zooplankton community appeared to be more diversified around the entrance of the Lake Edku current than the other parts of the bay, indicating a significant contribution of freshwater species from Lake Edku (Abdel Aziz 2000a, 2001). The long-term observations of zooplankton in Abu Qir Bay demonstrated significant changes in the community structure (Table 8.3). Although copepods are still the dominant group, their contribution to total zooplankton abundance decreased markedly from 71 to 72.6% in earlier studies to 52.6% recently. In the meantime, tintinnids, rotifers, and polychaete larvae played markedly greater roles than in the past.

Table 8.2 The dominant species during two successive years in Abu Qir Bay

Shams El-Din and Dorgham (2007)	El-Sherif and Mikhail (2003)
April 1998–March 1999	December 1999–November 2000
N. pungens Cleve	*Asterionella glacialis*
A. japonica Cleve	*S. costatum*
T. nitzschioides Grunow	*Rhizosolenia fragilissima*
Chaetoceros affinis Lauder	*Leptocylindrus minimus*
Chaetoceros didymus Ehrenberg	*C. affinis*
Chaetoceros sp.	*N. longissima*
Nitzschia pacifica Cupp	*Nitzschia seriata*
P. micans Ehrenberg	*C. meneghiniana*
Euglena acus Ehrenberg	*Cyclotella nana*
–	*P. triestinum*
–	*Scrippsiella trochoidea*
–	*Cardaria cordiformis*

Furthermore, the copepods *Isias clavipes*, *Oithona helgolandica*, *Oithona linearis*, *Paracalanus pygmaeus*, and *Paracalanus aculeatus* formed dense populations (Dowidar et al. 1983), while they were completely absent during 1998–1999. In contrary, *Oithona plumifera* was recorded in 1962 and 1963 (El-Maghraby and Dowidar 1973) and became one of the most abundant copepods later (Abdel Aziz 2001). On the long-term scale, zooplankton standing crop in the southwestern part as well as in the whole Abu Qir Bay revealed remarkable variations (Table 8.4), relative to the inter-annual changes in the water quality.

The effect of eutrophication on zooplankton community was reported in different localities of the world. The progress of eutrophication in the reservoir impacts the temporal dynamics of zooplankton (Pinto-Coelho 1998). A significant linear regression between biomass of cyclopoids and total phosphorous was found, suggesting that zooplankton reacts to eutrophication by increasing the biomass at least in some major groups in Pampulha reservoir, Brazil (Pinto-Coelho et al. 2005), and 49 lakes of various types, mostly in North America

Table 8.3 Relative contribution of different groups to the total zooplankton count in Abu Qir Bay on the long-term scale (1974–1999)

Group	Dowidar et al. (1983)		Anonymous (1984)		Abdel Aziz (2001)	
	1974		1980		1998–1999	
	Count m^{-3}	%	Count m^{-3}	%	Count m^{-3}	%
Copepods	9,775	71.4	6,913	72.6	47,352	52.6
Tintinnids	1,282	9.4	75	0.8	25,064	27.8
Rotifers	–	–	26	0.3	6,967	7.7
Larvaceans	914	6.7	616	6.5	1,425	1.6
Cladocerans	565	4.1	–	–	272	0.3
Polychaetes larvae	59	0.4	454	4.8	7,182	8.0
Cirripedes larvae	506	3.7	388	4.1	261	0.3

Table 8.4 Long-term observations of the annual average zooplankton crop in Abu Qir Bay

Area	Year	10^3 individual m^{-3}	References
Off Abu Qir	1962	22.4	EL-Maghraby and Dowidar (1973)
Off Abu Qir	1966	9.9	EL-Maghraby and Dowidar (1973)
Whole Abu Qir Bay	1974	13.7	Dowidar et al. (1983)
Outer part of Abu Qir Bay	1980	14.7	Anonymous (1984)
Southwestern Abu Qir Bay	1974	5–20	Dowidar et al. (1983)
Southwestern Abu Qir Bay	1980	10.5	Anonymous (1984)
Southwestern Abu Qir Bay	1998–1999	90.1	Abdel Aziz (2001)

and Europe (Hanson and Peters 1984). Eutrophication may affect the seasonal patterns of some zooplankton species (Pinto-Coelho 1998) and cause decreasing biomass or increasing species richness (Hulot et al. 2000). Calanoids are considered to be good indicators of oligotrophy in the tropical reservoirs of Brazil, and they are rare and often disappear from reservoirs that become eutrophic (Sendacz 1984, Arcifa 1984). The increase in the cyanobacterial biomass also induced important modifications in zooplankton structure and metabolism that has been discussed elsewhere (Pinto-Coelho et al. 2005).

8.3 Eastern Harbour

The Eastern Harbour is the most important embayment on the Mediterranean coast for Egypt as well as for the world heritage, since it contains thousands of the ancient Egyptian artefacts which encourage the idea of establishing a natural museum of underwater archeology in the harbour. It is a relatively small semi-closed basin on the central part of Alexandria coast, occupying an area of 2.53 km^2, with an average depth of 6 m. The harbour is isolated from the open sea by a high thick breakwater, but connecting to the sea through two narrow openings, Qayet Bey (Boughaz) and Silsila (Fig. 8.3). The harbour is exposed to several kinds of human activities, including fishing, yacht sport, anchoring of fishing ducks and ships, land-based effluents, boat building workshops, and recreation. Outside the harbour, the seawater is indirectly affected by domestic wastes (264×10^3 m^3 day^{-1}) discharged into Anfoushi area from Qayet Bey pumping station (El-Rayis and Hinkely 1999), which is located about 500 m west to the harbour. The large amount of nutrient salts, brought through the domestic wastes, caused eutrophication in the major part of the harbour since 1972–1973 (Halim et al. 1980a).

During the second half of the past century the ecosystem of the harbour passed through different phases of fundamental changes. The first phase was before the construction of the High Dam in 1964 when the Nile flood had been causing marked dilution of the harbour's water in late summer. The second phase occurred after the cessation of the Nile flood, with increasing effect of the sewage pollution reaching to the harbour directly through 11 submersible outfalls (Zaghloul and Halim 1990) and indirectly from Kyet Bey pumping station and Umum drain (Aboul-Kassim 1987). These conditions have made the water of the harbour eutrophic and the plankton production almost continuous all the year round at a high level (Ismael

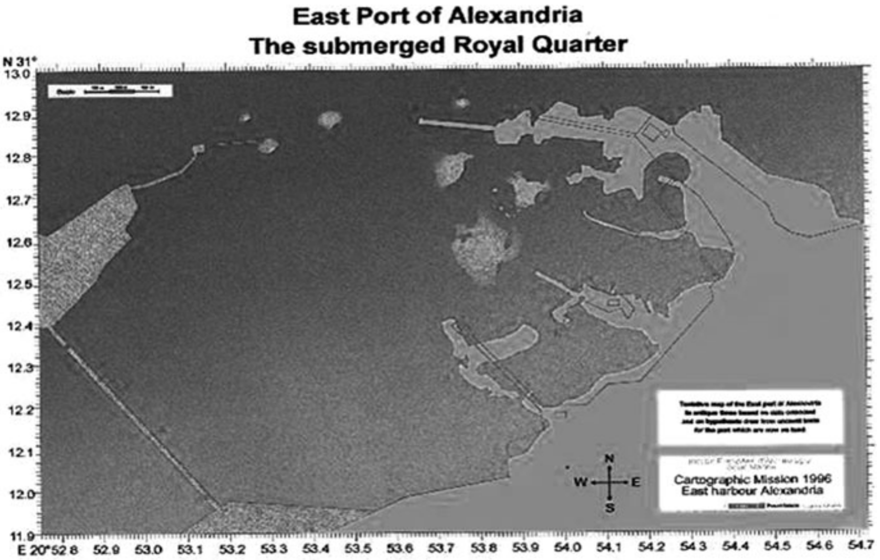

Fig. 8.3 The Eastern Harbour of Alexandria

1993). The third phase in the harbour started after 1993 when all sewers of the harbour were closed, except those at Qayet Bey and EL-Silsila, due to the diversion of sewage discharge into Lake Mariut. The long-term discharge of sewage waters to the Eastern Harbour has resulted in a high nutrient load, not only in the water column but also within the sediments, which is considered as the renewable nutrient source for the harbour ecosystem, especially after the development of the sewer system of Alexandria.

The Eastern Harbour has comparatively limited variations in its surface salinity, whereas it is usually higher than 38‰ during most of the year. However, it sometimes shows clear decrease due to the effect of discharged wastewaters down to 32‰. Throughout the half past century, salinity in the Eastern Harbour was decreasing during a part of the year, with a pronouncedly variable minimum (Table 8.5). Such decrease is attributed to the amount of the discharged waters reaching the harbour directly or indirectly.

The Eastern Harbour ecosystem experienced rapid and frequent pronounced variations due to the current regime, water exchange with the open sea, direction and speed of the winds, discharged wastes, and several other activities inside the harbour. There is active water exchange between the harbour and the open sea by seasonally variable currents, through Qayet Bey and Silsila openings (Abdalla 1979). These currents are affected by the direction and speed of the winds, which also experienced temporal changes (El-Geziry and Maiyza 2006). In addition, remobilization of nutrients from sediments to the water column could occur under the effect of variable environmental factors, causing eutrophication condition and abnormal flourishing of phytoplankton and that consequently play a crucial role in changing water quality of the harbour (Awad 2004). The long-term comparison of the fertility of the Eastern Harbour revealed that nutrient salts demonstrated pronounced variations over the past five decades (Table 8.6). The inorganic nitrogen forms (ammonia and nitrate) displayed pronounced increase during the past two decades, while phosphate illustrated relatively narrow changes. In contrast, silicate reported clear increase during the past decade. The levels of nutrients in the Eastern Harbour appeared to be higher than those proposed as indication of eutrophication, and chlorophyll-*a* concentration reflects clear eutrophication on the long-term scale (Table 8.7).

The long-term effect of anthropogenic activities on the environmental characteristics of the Eastern Harbour was resulting in changes in the standing crop and community structure of the phytoplankton. The available information showed clear discrepancy in the qualitative structure of the phytoplankton community throughout the last 40 years. As shown in Table 8.8 the total number of dinoflagellates decreased to 35 species as compared to those reported before the construction of the High Dam (62 species).

Table 8.5 The range of variations of the surface salinity in the Eastern Harbour during the period 1956–2007

Year	Min.	Max.	Average	References
1956–1957	31.44	38.91	37.77	El-Maghraby and Halim (1965)
1964	33.85	38.61	–	Hassan (1972)
1968	37.66	38.51	38.25	Megally (1970)
1969	37.64	38.4	38.4	Megally (1970)
1972–1973	38.11	38.86	–	Halim et al. (1980b)
1976–1977	37.6	38.73	38.08	El-Zawawy (1980)
1977–1978	37.62	38.73	38.09	Halim et al. (1980a)
1986–1987	28.2	38.8	35.83	Zaghloul and Halim (1992)
1989–1990	36.68	38.68	37.99	Ismail (1993)
1990–1991	35.83	38.88	37.28	Hussein (1994)
1996	34.03	39.83	36.83	Metcalf and Eddy Intern. (1997)
1999–2000	34.97	38.90	36.58	Tawfik (2001)
2003	37.3	38.14	37.7	EEAA (2003)
2006–2007	33.00	39.9	36.7	Mikhail and Halim (2009)

Table 8.6 Historical changes of the annual average concentrations of nutrient salts ($\mu M\ L^{-1}$) in the Eastern Harbour during the past five decades

Date	NH_4	NO_3	PO_4	SiO_4	References
1956–1957	–	–	0.23	–	El-Maghraby and Halim (1965)
1979	–	12.35	0.82	–	El-Nady (1981)
1985–1986	3.76	7.12	0.56	–	Aboul-Kassim (1987)
1989	1.5	4.12	1.08	11.2	Ibrahim (1999)
1990–1991	–	3.78	0.58	–	Hussein (1994)
1999–2000	3.2	2.71	0.66	4.3	Tawfik (2001)
2002–2003	5.33	10.48	0.6	3.36	Madkour et al. (2007)
2003	2.33	5.76	0.33	7.58	EEAA (2003)
2004	2.26	7.06	0.67	–	EEAA (2004)
2005	1.38	3.43	1.37	–	EEAA (2005)
2006–2007	2.33	2.31	1.1	4.33	Mikhail and Halim (2009)

Table 8.7 The minimum, maximum, and average concentrations of chlorophyll-*a* (mg m^{-3}) in the Eastern Harbour in different years from 1976 to 2007

Year	Min.	Max	Average	References
1985–1986	0.46	49.36	5.14	Aboul-Kassim (1987)
1986–1988	–	–	14.34	Nessim and Zaghloul (1991)
1987–1988	0.1	1.32	–	Emara et al. (1992)
1990–1991	0.71	13.18	4.9	Zaghloul (1996)
1991	30.4	135.0	63.84	Labib (1994)
1996	0.2	16.0	6.27	Metcalf and Eddy International (1997)
1999–2000	0.22	3.88	1.61	Tawfik (2001)
2003	1.71	17.18	8.93	EEAA (2003)
2004	3.01	38.32	13.7	EEAA (2004)
2005	3.69	24.97	8.89	EEAA (2005)
2006–2007	0.03	5.51	–	Mikhail and Halim (2009)

The dominance of species demonstrated wide variations, whereas some species lost their dominance while others were still dominant. The diatom *Chaetoceros socialis* was the predominant form before and after the construction of the High Dam, till 1977–1978. The dinoflagellate *Alexandrium minutum* was identified as a new species for the first time in the Egyptian Mediterranean waters, particularly in the Eastern Harbour in 1958 and predominated in the harbour for several years, especially during spring. Later on, the bloom of *A. minutum* diminished clearly for several years, but it appeared again in great numbers (24.4×10^6 cells L^{-1}) in 1994. The latter species was substituted by the dinoflagellate *Prorocentrum triestinum* in April 1993, amounting to 70×10^6 cells L^{-1} (Labib 1994). The characteristic feature in the Eastern

Table 8.8 The number of phytoplankton species in the Eastern Harbour from 1956 to 2003

Year	No. of species				References
	Diatoms	Dinoflagellates	Freshwater	Total	
1956–1957	57	62	–	119	El-Maghraby and Halim (1965)
1964	–	70	–	–	Hassan (1972)
1977–1978	88	–	–	–	Halim et al. (1980a)
1986–1988	–	–	–	77	Zaghloul and Halim (1992)
1989–1990	169	54	6	229	Ismail (1993)
1990–1991	76	35	20	131	Hussein (1994)
1999	–	–	–	75	Ismail and Halim (2000)
2000	45	31	–	76	Tawfik (2001)
2003	76	61	14	151	Madkour et al. (2007)

Harbour is the existence of the diatom *S. costatum* as a perennial dominant species throughout the last four decades but with a large variability in its contribution to the total phytoplankton crop. This species was encountered among the indicators of eutrophication in low water salinity. On the other hand, several freshwater algae were recorded in great numbers in the harbour, such as the diatom *C. meneghiniana, the* cyanophyte *Anabaena* sp., the euglenophyte *Euglena* sp., the chrysophyte *Amphicrysis* sp., and the chlorophyte *Pyramimonas* sp. The long-term observations on the dynamics of phytoplankton community in the Eastern Harbour reported recent increase in the number of species to 151 species (Madkour et al. 2007) as compared to those (75 and 76 species, respectively) found by Ismael and Halim (2000) and Tawfik (2001). This indicates that the harbour's phytoplankton started to restore its high diversity (229 species) which was recorded earlier by Ismael (1993) after stopping of the land runoff discharge. It is obvious that phytoplankton abundance cycle in the harbour has passed distinctive changes throughout the past five decades, in both the maximum values and the frequency of peaks. Several peaks over the year were recorded by Madkour et al. (2007), one peak by Ismael (1993) and three peaks by Tawfik (2001). According to Ismael (2003) in the last 10 years, however, organic pollution in the harbour has become increasingly more controlled, which might explain the gradual disappearance of *A. minutum* blooms since 1994. Harbour sediment cores, however, show peaks of cyst abundance corresponding to earlier blooms (Ismael et al. 2001).

Relative to the fundamental environmental changes and eutrophication, zooplankton community in the Eastern Harbour experienced pronounced variations during the past five decades. The number of zooplankton species increased from 45 species (Aboul-Ezz et al. 1990) to 79 species (Abdel Aziz 2004). Such increase was accompanied by analogous changes in the contribution of the different groups to the zooplankton abundance, whereas the role of copepods decreased during the last three decades from 65% to 36.8–38.5% (Abdel Aziz 2004). In contrast, the role of both protozoans and rotifers maximized from 8.3 and 1% (Dowidar and El-Maghraby 1970) to 40.5% and 8.8% (Abdel Aziz 2004). This may be attributed to the decrease of water stability from 1986 to 1990 in the harbour which caused marked drop in the crop of the dinoflagellate *A. minutum* and consequently the disappearance of the red tide phenomenon during that period (Ismael 1993). The flushing rate of the harbour water, which was estimated by 5 months (Aboul-Kassim 1987), may also play a role in such variations of zooplankton community. The dominance of zooplankton species in the Eastern Harbour reported clear long-term changes relative to the eutrophication conditions. According to Abdel Aziz (2004) there are four groups of dominant species occurred during the past half century. The first one includes species which demonstrated dominance during the past decade only, like the rotifers *Synchaeta oblonga*, *Synchaeta okai*, the tintinnids *Favella ehrenbergii*, *Favella markuzowskii*, *Favella serrata*, *Metacylis mediterranean*, *Tintinnopsis cylindrica*, and the copepod *O. plumifera*. The second group comprises species which lost their dominance in recent records, such as the copepods *Acartia latisetosa*, *Clausocalanus arcuicornis*, *and I. clavipes*. The third group was represented by species which kept their dominance throughout the past five decades, namely the copepods *Euterpina acutifrons*,

Table 8.9 Timing of zooplankton peaks in the Eastern Harbour during the period from 1957 to 2003

Date	Peaks timing		References
1957	April	July	El-Maghraby and Halim (1965)
1961	June	October	Dowidar (1965)
1962	May	August	Dowidar (1965)
1976	June	December	El-Zawawy (1980)
1986–1987	February	April and August	Aboul-Ezz et al. (1990)
1989–1990	March	August	Ismail (1993)
1991	May	October	Abdel-Aziz (1997)
1999–2000	February	October–November	Abdel-Aziz (2004)
2000–2001	March	September	El-Gindy et al. (2008)
2001–2002	December	July and September	El-Gindy et al. (2008)
2002–2003	March	October	El-Gindy et al. (2008)

Oithona nana, *Paracalanus parvus*, and *Acartia clausi*. The last group consists of species which altered the dominance in the successive annual records, like the rotifers *Brachionus calyciflorus*, *Brachionus plicatilis*, the tintinnids *Helicostomella subulata*, *Tintinnopsis campanula*, and the copepod *Centropages kroyeri*. The variation of eutrophication level was associated with variation in the dynamics of zooplankton abundance. In the period 1962–1963, before the appearance of eutrophication in the Eastern Harbour, zooplankton sustained remarkably high standing crop reaching 143.6 × 10^3 individuals m^{-3} (Dowidar and El-Maghraby 1970), dropped markedly to 29.7 × 10^3 individuals m^{-3} in 1976 (EL-Zawawy 1980) after the appearance of eutrophication for the first time in 1972, while during 1999–2003 it varied annually between 24.1 × 10^3 and 86.3 × 10^3 individuals m^{-3} (El-Gindy et al. 2008). On the other hand, the zooplankton abundance cycle exhibited clear inter-annual variations, particularly the timing of the seasonal peaks (Table 8.9).

8.4 Western Harbour

The Western Harbour of Alexandria (Fig. 8.3) is the largest harbour in Egypt and the second one in the Mediterranean after Marseille. It is a shallow semi-closed basin, receiving directly variable volumes (about 9,000 m^3 day^{-1}) of discharged wastes from Nubaria Canal and affected by the discharges of Umum drain. The harbour also suffers from intensive maritime activities, including the import of fertilizers, coal, cement, and export of oil.

The distribution of discharged wastewaters throughout the harbour as well as the environmental conditions in the Western Harbour is mainly governed by monthly exchange of water mass (El-Gindy 1986, Hassan and Saad 1996) through two subsurface currents (Farag 1982).

The Western Harbour ecosystem (Table 8.10) demonstrated pronounced variations on the long-term scale, relative to changes in the quality and volume of the discharged wastewaters. It is to be noted that the data of EEAA (2003, 2004, 2005) given in Table 8.1 were based upon on one sampling station only. Salinity is a good indication of the discharged wastewaters throughout the harbour. According to Dorgham et al. (2004) regardless of abnormally high salinity (40.7‰) in the recent records, the minimum as well as the annual average values appeared to be clearly lower (26.3‰) than those recorded earlier (29‰). Two water masses are distinguished: one occupied the area affected by the land runoff, with salinity < 35.1‰, and the second occurred in the inner part of the harbour, slightly away from the impact of the anthropogenic sources, with salinity > 35.1‰. This may indicate the greater role of Umum drain in salinity variations than Nubaria Canal, since the discharge of the latter is usually as seasonal pulses. However, both sources affect the harbour salinity on the long term. Water transparency is usually low almost over the whole harbour, with a maximum Secchi depth reading of 350 cm during the past decade (Table 8.10). However, both the minimum and annual average values reflect the continuous decrease of water transparency, which may be attributed mainly to the abnormal phytoplankton outburst, in addition to strong water mixing resulting from heavy ship traffic and water currents

Table 8.10 Long-term observations of ecological parameters at surface water of the Western Harbour

	Zaghloul and Nessim (1991)	Dorgham et al. (2004)	EEAA (2003)	EEAA (2004)	EEAA (2005)
	1989	1999–2000	2003	2004	2005
Salinity (‰)	36.26	35.12	–	–	–
Transparency (cm)	180	135	–	–	–
DO (mg L^{-1})	6.4	4.0	6.1	5.7	4.9
NO_3 (μM)	4.8	5.73	8.76	5.68	3.27
NH_4 (μM)	–	14.5	2.33	3.86	5.78
PO_4 (μM)	1.03	1.17	0.57	0.66	1.42
N/P	16	11.8	15.4	8.6	2.3
SiO_4	–	9.03	7.61	–	–
Chl-*a* (μg L^{-1})	4.2	33.82	8.14	7.74	5.43
Phaeop. (μg L^{-1})	–	10.39	–	–	–

(Dorgham et al. 2004). The harbour water was characterized by low oxygenation during most of the period 1999–2000, with monthly average concentrations of dissolved oxygen < 4 mg L^{-1} at the surface and near the bottom, and comparatively high values at the surface (6.0–8.2 mg L^{-1}) were found in April and September (Dorgham et al. 2004). On the vertical scale, the near-bottom water sustained lower oxygen by up to 0.6 mg L^{-1} than the surface during the low temperature period (October–March), while during the warm period (June–September), the difference between surface and bottom raised to 1.3–3.8 mg L^{-1} (Dorgham et al. 2004). The low oxygen near the bottom is attributed to increase of oxygen consumption in oxidation of the organic matter (Nessim and Tadros 1986) and the stagnation conditions prevailing in summer.

It seems that the oxygen concentration in the Western Harbour was mostly below the threshold level (<4 mg L^{-1}) proposed by Huet (1973), while near the bottom it was mostly close to the hypoxia conditions (Stachowitsch and Avcin 1988). These conditions classify the Western Harbour among the areas with critical limits of dissolved oxygen (3.5–4.2 mg L^{-1}) necessary for healthy growth of the biota in both cold and warm waters (Grundy 1971, Arin 1974). The Western Harbour sustained pronouncedly high nutritional level. Nitrate showed wide fluctuation along the water column, varying from 0.21 to 20.46 μM, with high concentrations at the surface from November to April (5.1–14.33 μM) and lower than 3 μM during the rest of the year (Dorgham et al. 2004). Near the bottom, nitrate fall within a range of 0.25–18.12 μM, the high concentrations (5.6–13.83 μM) appeared from December to March, dropped in late spring and summer due to uptake by abnormally high phytoplankton biomass. On long-term scale, nitrate experienced serious variations. However, the majority of the recorded concentrations of nitrate in the harbour exceed the levels (4 μM) proposed by Franco (1983) and Marchetti (1984) as indicator of eutrophication (Table 8.10). High concentrations of ammonia were found in the Western Harbour up to 57.46 μM in the surface water and up to 43.73 μM near the bottom. The surface water contained ammonia greater than 10 μM most of the year, with distinguished peak during winter, while near the bottom two peaks appeared in August and November (Dorgham et al. 2004). The higher surface ammonia in winter is related mainly to the domestic wastewaters that brought from the bottom by the winter mixing, but the higher values near the bottom in August and November may be produced from the decay of organic matter, accumulating during summer stagnation (Nessim and Tadros 1986). The records of EEAA (2003, 2004, 2005) demonstrated markedly low concentrations of ammonia, but they still higher than that (>2 μM) adopted as criteria of eutrophication by Franco (1983) and Marchetti (1984). N/P ratio showed wide range (0.3–170.5) over the harbour, the surface water suatained high values (13–58) during late autumn and winter and low values (<7.5) during spring and summer (Dorgham et al. 2004). However, the annual average values in earlier and recent studies were markedly lower. The wide variability of N/P is common along the Mediterranean coast of Egypt, particularly the areas exposed to land-based runoff (Zaghloul 1996, Abdel-Aziz et al. 2001). Phosphate sustained clearly high concentrations (up to 5.7 μM) at the surface water and up to 2.6 μM near the bottom. The surface phosphate showed irregular seasonal variation, with high concentrations (1.11–2.7 μM) during spring and summer and low concentrations (0.28–0.84 μM) during autumn and winter (Dorgham et al. 2004). In contrast, Nessim and Tadros (1986) reported the maximum phosphate in winter, while Saad (1973) found high phosphate in spring and summer. Near the bottom, the monthly pattern was similar to that at the surface but with lower values due to easily adsorption of phosphorous on fine sediments (Lucotte and Anglejan 1983) at low aeration conditions near the bottom (Mortimer 1971). Recent records (EEAA 2003, 2004, 2005) demonstrated relatively high values (Table 8.10), reflecting the continuity of anthropogenic phosphorous enrichment. Silicate demonstrated wide variations at the surface (0.3–36.3 μM) and near the bottom (0.5–38.4 μM). Three peaks were observed during summer, October, and February at both the surface and bottom, while the low silicate occurred during spring and autumn coincided with high phytoplankton biomass, particularly at the dominance of diatoms in both seasons, as they need silicate in their growth (Dorgham et al. 2004). The Western Harbour appeared to be highly productive, whereas phytoplankton biomass (Chl-*a*) reached a maximum of 219.4 μg L^{-1} and an annual average of 33.8 μg L^{-1} (Dorgham et al. 2004). The latter value is about eightfold that (4.2 μg L^{-1}) reported by Zaghloul and Nessim (1991), indicating the magnification of the eutrophication problem in the Western Harbour. However, lower chlorophyll concentrations were found during the period from 2003 to 2005 (Table 8.10), which are also higher than the minimum level of eutrophication proposed by Carlson (1977), Friligos (1988), and Stirn (1988). The abnormal phytoplankton growth was followed by intensive death, lead to high concentrations of phaeopigment (annual average: 10.4 μg L^{-1}), increasing at times pronouncedly high values (105.9 μg L^{-1}) (Dorgham et al. 2004). The latter authors stated that eutrophication has become a persistent problem in the harbour, with increasing level during the past two decades. Gharib and Dorgham (2006) reported pronouncedly high phytoplankton counts all the year round (439.4 × 10^3–1,960.4 × 10^3 units L^{-1}) in Western Harbour, with two peaks in May and July. Different abundance cycles were reported at the different stations, relative to spatial environmental differences. *S. costatum* showed absolute dominance (73–98.7%) in the outer part at salinity <35‰, while *A. minutum* and *P. triestinum* shared the dominance (50.4–59.5% and 38.2–44.4%, respectively) during July in the inner part at salinity >35‰. The abundance of the diatoms and dinoflagellates provides substantial succession in space and time, whereas diatoms illustrated absolute dominance throughout the harbour in winter and dinoflagellates dominated during summer.

The occurrence of two water masses with two different salinities leads to two different phytoplankton communities, a more diversified one (67–79 species) occurred in the outer part at surface salinity <35‰ and less diversified one (58–62 species) in the inner part at salinity > 35‰ (Gharib and Dorgham 2006). In addition, the structure of phytoplankton community showed little variations among the different seasons, except the drop in the number of diatom species in summer. In the inner part of the harbour, the dinoflagellates *A. minutum* and *P. triestenum* were the predominant species, which are considered as indicator of eutrophication in different areas (Pagon 1985, Zaghloul and Halim 1990). *A. minutum* has never been found earlier in the Western Harbour and its recent appearance as predominant form reflects more deterioration of the water quality in the harbour (Gharib and Dorgham 2006). The values of phytoplankton diversity index (0.1–2.63 bits) support that the harbour suffers from eutrophication all the year round, since these values are close to those (1–2.5 bits) recorded in eutrophic lakes (Margalef 1964). The long-term observations of phytoplankton community in the Western Harbour (Zaghloul 1994, Gharib and Dorgham 2006) reported marked decrease in the phytoplankton abundance during the past decade, associated with change of peaks timing from June and October in 1989 to July and September in 1999–2000. The contribution of dinoflagellates to total phytoplankton count increased from 25% in 1989 to 57% in 1999–2000 while that of diatoms decreased from 70 to 41% and the number of species became 107 instead of 68 species. The recent increasing role of dinoflagellates, which usually prefer relatively high temperature, explains the shifting in the peak timing of phytoplankton bloom toward the warmer period (27–29°C) in the harbour (Gharib and Dorgham 2006). Furthermore, the dominance of species experienced complete changes during 1999 (Table 8.11), which again indicates serious change in the characteristics of the harbour water.

The structure of zooplankton community in the Western Harbour seems to be affected largely by

Table 8.11 Dominant phytoplankton species in the Western Harbour during the past decade

Zaghloul (1994) 1989	Gharib and Dorgham (2006) 1999–2000
C. meneghiniana	*A. minutum*
Pseudonitzschia delicatissima	*S. costatum*
Prorocentrum cordatum	*Prorocentrum triestinun*
P. micans	*Pseudonitzschia seriata*
Euglena granulata	*S. trochoidea*
–	*A. japonica*
–	*P. micans*

the discharged wastewaters, since of a total of 111 species recorded in the harbour, 39 fresh and brackish water forms were found. The eutrophication condition affected the community structure of zooplankton, particularly the copepods which usually form the major component of marine zooplankton. Copepods were represented in the Western Harbour by 13 species only, while tintinnids comprised 40 species. The seasonal variability in quantity of the discharged wastewaters into the Western Harbour was clearly detected from the variable role of freshwater forms in the zooplankton stock. The relatively high abundance of both copepods and tintinnids in the harbour as a polluted area is in agreement with Blanco et al. (1990). On the other hand, the pronounced occurrence of rotifers and ciliates could be considered as indicator of the freshwater discharge to the area, since the organic pollution enhances the survival of protozoa (Jahn and Jahn 1970). The abundance of zooplankton in the harbour was generally low except the relatively high peak at July, indicating the unfavorable conditions for healthy zooplankton population.

8.5 Dekhaila Harbour

Dekhaila Harbour was constructed in 1986 on the southwestern part of Mex Bay, west of Alexandria. The purpose of establishment of the harbour was to serve the iron and steel factory at Dekhaila, to facilitate international trade with Egypt, and to reduce the stress on the Western Harbour of Alexandria. The main basin of the harbour covers an area of 32.2 km^2 with an average depth of 12 m (Fig. 8.3). Before the construction of the harbour, the sea area was exposed to unknown extent to agricultural, sewage, and industrial wastewaters from Lake Mariut through Umum drain. After its construction, the environmental stress on the harbour increased due to several maritime activities, such as heavy ships traffic, export and import activities, and ship services. The infrastructure of the harbour caused crucial changes in the circulation pattern inside the Mex Bay, which are reflected on the distribution of surface salinity. Regardless of the minimum and maximum values (17.34 and 39.18‰) the surface of salinity in the harbour varied seasonally between relatively high values (31.86–38.33‰) from February to May and low values (22.2–24.5‰) from June to January (Fahmy et al. 2004). The inter-annual records show relatively small decrease in the surface salinity during the period 1990–2000 from 29.1 to 26.3‰ (Table 8.12). Such decrease coincided with increasing volume of the discharged wastewaters from Lake Mariut, in addition to wastewaters discharged frequently from giant ships anchoring for several days in the harbour. The harbour's water appeared to be turbid most of the year, with average Secchi depth < 106 cm, except slightly clear water (up to 270 cm) during late winter and spring (Fahmy et al. 2004). These values are markedly lower than that (Table 8.12) reported earlier by Abdalla et al. (1995), indicating more deterioration of the water quality in Dekhaila Harbour during the past decade. The high turbidity in the harbour is caused by extremely dense phytoplankton bloom (Ismael and Dorgham 2003), the active water mixing, frequently blown dust of scrap iron and coke which are stored as great heaps on the quays, and flour flying from grains milling inside the harbour (Fahmy et al. 2004). Changes in water transparency have been used to assess the rate of eutrophication (Lorentz 1980, Cruzado 1988).

The long-term observations of dissolved oxygen displayed relative narrow inter-annual fluctuation in the Dekhaila Harbour (Table 8.12). Fahmy et al. (2004) observed two aeration levels, one moderately high (4.9–7.2 mg L^{-1}) during intermittently 6 months and significantly low one (2.4–3.3 mg L^{-1}) during the rest of the year. The latter level is closed to the threshold level (< 2.8 ml L^{-1}) of well-aerated waters supposed by Huet (1973) and slightly higher than that (<1.4 ml L^{-1}) of hypoxia condition (Stachowitsch and Avcin 1988). Such levels are unfavorable for healthy populations of numerous aquatic organisms (Grundy 1971, Arin 1974). This means that the dissolved oxygen decreased to a level threatening the existing biota for

Table 8.12 Annual averages of hydrographic and eutrophication parameters recoded in the Dekhaila Harbour during the past two decades

	Abdalla et al. (1995)	Tayel et al. (1996)	Fahmy et al. (2004)	EEAA (2003)	EEAA (2004)	EEAA (2005)
Parameters	1990–1991	1993–1994	1998–1999	2003	2004	2005
Secchi depth (cm)	173	–	106	–	–	–
Salinity (‰)	29.1	26.3	26.7	–	–	–
DO (mg L^{-1})	2.9	7.5	4.6	4.98	4.98	5.08
NO_3–N (μM)	2.46	0.89	19.22	8.19	11.4	3.82
NH_4–N (μM)	8.12	9.85	38.69	9.19	7.59	3
PO_4–P (μM)	0.4	1.13	6.44	1.22	1.09	1.96
SiO_4–Si (μM)	5.7	22.2	49.52	17.3	–	–
N/P	6.15	0.8	5	7.5	11	1.9
Chl-*a* (μg L^{-1})	24.05	–	107.5	23.63	17.75	3.38

a significant part of the year. Although the amount of dissolved oxygen supposed to be high in the surface water due to intensive photosynthesis by abnormal phytoplankton bloom, large quantity of the produced oxygen in the harbour is consumed in oxidation of high load (3.63–10.07 mg L^{-1}) of organic matter (Tayel et al. 1996) and oil spills released frequently from ships and small motorized boats. The nutritional situation indicates high fertility in the Dekhaila Harbour, but the concentrations of nutrient salts experienced serious variations on the seasonal as well as the long-term scale (Table 8.12). Ammonia was generally high in the harbour during the past two decades, demonstrating inter-annual averages between 7.59 and 9.58 μM. Fahmy et al. (2004) reported distinctive peaks (52.71 μM) in June, (72.74–73.83 μM) in August–September, (61.08 μM) in November, and (64.19 μM) in January, attributing these high concentrations to high level of organic matter. On the long-term scale, the ammonia in the harbour is pronouncedly greater than that (2 μM) reported by Franco (1983) as indicator of eutrophication.

Nitrate demonstrated pronounced increase in the harbour during the past 20 years. According to Fahmy et al. (2004) it exceeds 10 μM over the year, increasing to a range of 28.79–43.31 μM from October to January, mainly due to the precipitation of flour of milled grains blown by strong northwest winds during this period. The annual average nitrate concentration in Dekhaila Harbour was for a long time higher than the indicating level of eutrophication (4 μM) suggested by Franco (1983). A regular study of reactive phosphate during 1998–1999 reported markedly high concentrations (up to 56.46 μM) in the Dekhaila Harbour, with peaks in August–September (13.32–17.84 μM), June (7.62 μM), November (7.75 μM), and January (8.74 μM) (Fahmy et al. 2004). These values have never been measured in the harbour since its construction (Table 8.12). The high amount of reactive phosphate during 1998–1999 is attributed to the precipitating flour of the milled grains, which are usually rich in phosphate, and also phosphate released (2.98 μM m^{-2} day^{-1}) from the bottom sediments (EL-Samra et al. 1984). The values of phosphate on the long-term scale (Table 8.12) appeared to be pronouncedly greater than the eutrophication limit (0.3 μM) assigned by Marchetti (1984) and Stirn (1988). During the past two decades, except that of Abdalla et al. (1995) the N/P values were low in the Dekhaila Harbour, corresponding to the relatively high phosphate content as compared to inorganic nitrogen compounds (Table 8.12). These conditions are in agreement with Welch (1980), who reported that the N/P ratio varies with trophic state and decreases with increased eutrophication. The harbour was generally rich in reactive silicate during the past two decades (Table 8.12). The highest concentration (148.2 μM) and highest annual average (49.52 μM) were recorded during 1998–1999, with monthly average higher than 14 μM over the year (Fahmy et al. 2004). The high contents of chlorophyll-*a* reflect abnormally intensive phytoplankton growth (Table 8.12). Extremely high chlorophyll-*a* (up to 1,322.7 μg L^{-1}) were found during 1998–1999, with monthly average peaks falling within the range of 123.8–444 μg L^{-1} from June to October and 8.2–40.7 μg L^{-1} during spring (Fahmy et al. 2004). The exceptionally high chlorophyll-*a* during the warm period (June–October) has resulting

Table 8.13 Long-term changes in composition and relative abundance of different phytoplankton groups in Dekhaila Harbour

	Zaghloul et al. (1995)		Ismael and Dorgham (2003)	
	1990–1991		1998–1999	
Group	Sp. no.	Relative abundance (%)	Sp. no.	Relative abundance (%)
Diatoms	77	83.7	52	7.8
Dinoflagellates	23	7.4	38	15.2
Cyanobacteria	10	5.7	5	58.6
Chlorophyceae	10	1.8	9	0.1
Euglenophycea	3	1.4	2	8.2
Total	123		107	

mainly by a great amount of planktonic cyanobacteria and green algae transferred from Lake Mariut through Umum drain (Ismael and Dorgham 2003). The high nutrients level and high chlorophyll concentration indicate acute eutrophication in Dekhaila Harbour during the past two decades.

It appeared that the dynamics of phytoplankton community in the Dekhaila Harbour experienced serious changes during the past decade. As shown in Table 8.12 fundamental changes could be reported in phytoplankton community structure as well as in the role of the variable. The number of diatoms species and their relative abundance decreased pronouncedly, while dinoflagellates showed clear increase in both the number of species and relative abundance. Regardless of negligible changes in the number of freshwater species, they demonstrated a greater role in the total phytoplankton count during the past decade, particularly the cyanophytes. The increasing richness of dinoflagellates may be explained by the increase of organic load in Dekhaila Harbour, since a significant part of dinoflagellates in some eutrophic bays (Eastern Harbour) is heterotrophic (Ismael and Halim 2000). In the meantime, the great contribution of the freshwater forms reflects the increasing volume of the discharged wastewaters to Mex Bay on the long-term scale, which lead to more deterioration of water quality in the harbour.

The phytoplankton cycle the harbour demonstrated in two peaks, but with different timing during the past decade. During 1990–1991 the phytoplankton peaks appeared in April and October (Zaghloul et al. 1995), while during 1998–1999, the peaks occurred in June and August (Ismael and Dorgham 2003) (Table 8.13). However, the cluster analysis of the standing crop over the year illustrated three seasonal communities with different standing crops (Ismael and Dorgham 2003). The diversity index in El-Dekhaila Harbour ranged from 0.08 to 2.41 for the different stations, while richness increased with decreasing salinity due to the introduction of brackish water species (Ismael and Dorgham 2003). Such pattern reflects eutrophication conditions in the harbour, as the values of the diversity index are close to those given by Margalef (1964) for eutrophic lakes.

Furthermore, the dominant phytoplankton species in the harbour experienced marked variations on the long-term scale (Table 8.14).

It seems that zooplankton have exposed to pronounced changes under the effect of environmental conditions in the Dekhaila Harbour. The distributional pattern of zooplankton was more or less related to that of salinity, since high standing crop (29,270 and 36,264 organisms m^{-3}) was found at stations experienced narrow salinity changes, while low count appeared at station with wide salinity range. The copepods *O. nana*, *E. acutifrons*, *and P. parvus* were among the dominant zooplankters in the southeastern Mediterranean usually found in remarkably great densities (Abdel-Aziz 1997, Hussein 1997), but in Dekhaila Harbour they demonstrated comparatively low counts (Abdel Aziz 2000b). *A. clausi* has been found previously as less important species (Dowidar and El-Maghraby 1970), but in Dekhaila Harbour it became a persistent species attaining sometimes relatively high counts. This means that the water

Table 8.14 Dominant phytoplankton species during the past decade in Dekhaila Harbour

Zaghloul et al. (1995)	Ismael and Dorgham (2003)
Merismopedia punctata	*Spirulina* sp.
Ankistrodesmus falcatus	*Oscillatoria* sp.
E. granulata	*S. trochoidea*
S. costatum	*E. acus*
N. microcephala	*S. costatum*
Protoperidinium minutum	*P. triestinum*

quality of the harbour became unsuitable for healthy populations of the above-mentioned dominant species, except *A. clausi*, which could withstand the eutrophication as well as pollution effect in the study area and it therefore may be considered as an indicator species. The permanent existence of the freshwater form *Acanthocyclops americanus* in high density in the Harbour reflects its ability to adapt to high salinity and establishes a well-growing population.

8.6 Mex Bay

El-Mex Bay is a relatively large coastal embayment west of Alexandria, at longitude 30°50′E and latitude 31°09′N, with an average depth of about 10 m and surface area of about 19.4 km^2. The bay is an important fishery ground as well as recreation area. It includes both the Western Harbour and Dekhaila Harbour (Fig. 8.4). The bay is one of the heavily polluted areas on the Egyptian Mediterranean coast, receiving huge amount of agricultural, industrial, and sewage wastes from the adjacent Lake Mariut through El-Umum drain. According to different estimations in literatures the volume of the wastewaters varied between 7×10^6 and 8×10^6 m^3 day^{-1}, which is supposed to increase with the growing population density of Alexandria city. In addition to El-Umum drain discharge, industrial wastes from several industries in the surrounding area, like chemical, chloro-alkali, tanneries, cement, and petroleum, are also discharged to the bay. These conditions cause pronounced eutrophication and drastic environmental changes. El-Mex Bay demonstrated wide-range variations in its surface salinity on the spatial scale relative to the dispersal pattern of the discharged wastes waters. The salinity of the near-shore waters sustained usually low values, increasing seaward to exceed 39.8‰ in the open part of the bay (Table 8.15). Relative to the surface salinity four water masses could be identified inside Mex Bay, but the area and position of each mass are usually governed by the wind direction and show seasonal pattern. These water masses include mixed land drainage with a salinity of <10.00 ppt, mixed water (10.0–30.0 ppt), diluted seawater (30.0–38.5 ppt), and Mediterranean Seawater (>38.50 ppt) (Soliman and Gharib 1998, Zakaria et al. 2007). During spring, the mixed water type occupied the near-shore area of El-Mex Bay, the diluted seawater occupied the central part, while pure Mediterranean water could be found at the both sides of the bay. During summer, the mixed land drainage type was spreading to a considerable distance off the drain outlet; the mixed water covers almost all the bay. During autumn, the diluted seawater occupied almost all the bay and the Mediterranean water type could

Fig. 8.4 Mex Bay including both the Western Harbour and Dekhaila Harbour

Table 8.15 Historical records of environmental parameters in Mex Bay

	1	2	3	4	5	6	7	1	2	3	4	5	6	7
	Minimum							Maximum						
	1983	1993	1995	1996	2003	2004	2005	1983	1993	1995	1996	2003	2004	2005
Salinity	5.2	20	3.68	0.6	–	–	–	38.44	39.8	38.5	39.6	–	–	–
Secchi depth (cm)	–	30	30	–	–	–	–	–	380	900	–	–	–	–
Dissolved oxygen (mg L^{-1})	-	1.6	0.23	2.6	–	–	–	–	9.8	8.2	9.6	–	–	–
Nitrate (μM)	6.3	2.0	4.81	0.0	2.37	3.97	5.23	21.0	16.6	58.0	71.0	12.6	11.5	5.97
Ammonia (μM)	5.78	2.5	0.0	2.13	1.5	1.95	3.29	34.6	65.8	132.1	127.8	19.6	5.1	9.4
Phosphate (μM)	4.2	0.6	0.28	0.32	0.07	0.37	0.99	19.43	4.4	17.2	48.0	3.31	1.68	3.47
Silicate (μM)	-	–	11.4	–	2.1	–	–	–	–	159.8	–	35.2	–	–
N/P	-	–	7.3	–										
Chl-*a* (μg L^{-1})	16.25	1.5	–	0.2	3.25	8.33	1.62	53.3	28.0	-	16.3	50.0	25.2	15.9

1 – Dorgham et al. (1987); 2 – Labib (1997), 3 – Soliman and Gharib (1998); 4 – Dorgham (1997); 5 – EEAA (2003); 6 – EEAA (2004); 7 – EEAA (2005)

only be found at the eastern part. During winter, Mex Bay was occupied by the four water types, whereas the land drainage water type occurred near El-Umum drain outlet, the mixed water occupied the southern half of the bay, the diluted seawater type reported at the northern part, and the western part has salinity of Mediterranean water (Zakaria et al. 2007). The water column of Mex Bay suffers from pronounced turbidity, particularly in front of the land runoff, whereas the Secchi disc readings were mostly <100 cm. Such turbidity is attributed to the strong mixing caused by discharged wastes, heavy traffic of fishing boats, and high count of plankton organisms. However, the open area of the bay shows comparatively high transparency (up to 9 m).

The long-term observations of the nutritional conditions demonstrated wide variability in the bay (Table 8.15), but the levels of all nutrient salts reflect high eutrophication. It is to be noted that the nutrients values given in Table 8.15 were based on different number of sampling stations and were collected at seasonal or bimonthly intervals. The markedly high nutrients reported during 1995 and 1996 (Gharib and Soliman 1998, Dorgham 1997) reflect the large amount of nutrient reaching the bay through the discharged wastewaters, since the maximum values were reported in front of the land runoff. In contrast, the comparatively low concentrations during 2003–2005 represented the amount of nutrients in area relatively far from the entrance of Umum drain. It is clear that Mex Bay is characterized by great load of organic matter on the long-term scale, whereas abnormally high concentration of ammonia reported during the past three decades (Table 8.15). The phytoplankton demonstrated pronouncedly intensive growth in Mex Bay, maximizing the level of eutrophication condition, since the inter-annual records over the past three decades indicate pronouncedly high chlorophyll concentration in the bay.

On the other hand, the phytoplankton community suffered from marked decrease in the species richness from 210 species during 1982–1983 to 158 species during 1995, while the total count reported abnormal increase from 43 × 10^3 to 31.4 × 10^6 units L^{-1} (Table 8.16).

Furthermore, the dominance of phytoplankton species demonstrated variable patterns throughout the past three decades. The diatoms *S. costatum* and *C. meneghiniana* kept their dominance over the whole period, while significant differences were reported for other species, particularly those of freshwater origin, transferred with the land runoff (Gharib 1998, Dorgham 1997). Zooplankton community in Mex Bay is pronouncedly affected by the dispersion pattern of discharged waters. The water masses in the bay showed different communities, relative to the salinity differences. The freshwater rotifers are more diversified and predominant zooplankton component in the water mass directly stressed by the Umum drain, while in contrast, copepods were less diversified and had low numerical density as compared to the water masses less affected by freshwater discharge (Table 8.17). On the other hand, the richness of protozoans experienced comparatively less spatial variation with salinity differences in the bay, but their abundance showed pronounced decrease seaward. The total zooplankton count reported marked decrease toward the open sea.

Table 8.16 The community structure and count of phytoplankton in Mex Bay on the long-term scale

	Dorgham et al. (1987)	El-Sherif (1989)	Gharib (1998)	Dorgham (1997)
	1982–1983	1988	1995	1996
Diatoms	119	83	83	41
Dinoflagellates	50	5	17	14
Chlorophytes	26	41	30	
Cyanophytes	11	26	20	
Euglenophytes	4	4	8	
Freshwater forms				11
Total sp. no.	210	159	158	66
Total count ($\times 10^6$ unit L^{-1})	0.043	0.097	0.94	31.4

Table 8.17 The community structure and count of zooplankton in Mex Bay during 2005

	S‰	Total	Rotifers	Protozoa	Copepods	Total count
First	<10ppt	47	19	25	2	106.6×10^3
Second	10–30	65	19	27	7	46×10^3
Third	30–38.5	64	12	35	10	20.1×10^3
Fourth	>38.5	41	2	23	12	$5.9\ 10^3$

Modified from Zakaria et al. (2007)

Table 8.18 Number of copepod and tintinnid species in Mex Bay on the long-term scale

Year	Copeods	Tintinnids	References
1982–1983	–	46	Dorgham (1987)
1995	19	6	Soliman (2006)
1996	33	13	Hussein (1997)
2005–2006	14	31	Zakaria et al. (2007)

The long-term changes of the environmental conditions in Mex Bay were reflected on the abundance and seasonal cycle of zooplankton. One peak (44.4×10^3 organisms m^{-3}) was reported in October 1996 (Hussein 1997) and two peaks in April and October with an annual average of 196.5×10^3 organisms m^{-3} during 1995 (Soliman and Gharib 1998). On the other hand, the recent records reported greater role for tintinnids compared to copepods (Table 8.18), indicating that copepods could not establish a healthy population in such an area of high eutrophication.

8.7 Conclusions

The Egyptian Mediterranean coast suffers from acute eutrophication, resulting from great amount of anthropogenic nutrients entering the sea through numerous land runoffs distributed mainly off the Nile delta region and Alexandria coast. These nutrients caused abnormally intensive phytoplankton blooms which together with great nutrients loads lead to deterioration of the coastal waters to a degree not favorable for healthy populations of different biota. In addition, these conditions caused fundamental changes in the dynamics of plankton community, including species composition, role of different groups, standing crop, seasonal cycles, and species dominance. Therefore, there is an importunate requirement to solve the problem of eutrophication not only in the Egyptian coasts but also everywhere in the aquatic habitats, through controlling the utilization of chemical fertilizers in the cultivated lands and reducing the discharge of such fertilizers to the marine ecosystem.

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Chapter 9

Freshwater Wetland Eutrophication

S. Sánchez-Carrillo, D.G. Angeler, M. Álvarez-Cobelas, and R. Sánchez-Andrés

Abstract The traditional perception of wetlands as nutrient sinks has led them to be used as wastewater disposal areas for a long time, resulting in a severe alteration of the structure and function by eutrophication. Nutrient loading is usually linked to hydrological alterations which encompass shifts in vegetation patterns and nutrient cycling. The eutrophication process in wetlands accelerates primary productivity and increases net accumulation of organic matter and nutrients but also enhances organic matter decomposition, microbial activity, and soluble nutrients in sediments. Internal loading becomes the main nutrient source to the wetland, even in the years of low external inputs, controlling the nutrient dynamics. Since soil phosphorus microbial biomass responds positively to phosphorus enrichment in wetlands, mineralized phosphorus in wetland soils appears as the most responsive microbial indicator to nutrient enrichment in wetlands. Therefore, phosphorus internal loading is the critical factor in regulating eutrophication status of wetlands. N_2O and N_2 emissions by wetlands can be enhanced in the future as nitrate availability in wetlands continues to be high due to increased pollution. The biological symptoms of wetlands degraded by eutrophication differ little from those observed in shallow lakes, these changes being consistent with predictions made by alternative state theory. The turbid state shows phytoplankton dominance and elevated biomass of planktivorous and benthivorous fish. Zooplanktivorous fish contribute to eutrophication chiefly via food web-mediated effects. Benthivorous fish increase the nutrient availability to phytoplankton chiefly by transferring sediment-bound nutrients to the water column during bottom foraging. Little of the methodology found to be useful in shallow lake restoration has been applied to wetland management. Whereas catchment nutrient management programs may be insufficient because of substantial storage of nutrients in wetland sediments, recent studies indicate that biomanipulation of fish standing stocks could contribute significantly to wetland eutrophication abatement. Therefore, biomanipulation, when appropriately timed in accordance with low water levels, combined with sediment dredging and other interventions, may be a low cost–high benefit tool for wetland eutrophication abatement.

Keywords Freshwater wetland · Eutrophication · Nutrient cycling · Alternative stable states · Biomanipulation

9.1 Introduction

Wetlands are known to function as filters in the landscape (Phillips 1996), retaining and transforming nitrogen and phosphorus and other compounds (Gunatilaka 1991, Sánchez-Carrillo and Álvarez-Cobelas 2001). Wetlands contribute as much as 40% to the earth's renewable ecosystem services, even though they cover only 1.5% of the planet's surface (Costanza et al. 1997). Biogeochemical processes in wetlands are so intensive to improve drastically the water quality of rivers, although the efficiency depends on landscape position and the wetland type (Whigham et al. 1988, Johnston 1991, Zedler 2003). Given this efficiency, many wetlands have been constructed,

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planted, and hydraulically controlled for the purpose of removing nitrate and phosphorus from wastewaters (Kadlec and Knight 1996). However, because of the intrinsic efficiency of nutrient retention, also natural wetlands have been used as wastewater disposal areas for a long time, resulting in a severe alteration of the structure and function by eutrophication (Vaithiyanathan and Richardson 1999, Álvarez-Cobelas et al. 2001). The symptoms of eutrophication in wetlands appear similar to those in shallow lakes (Angeler et al. 2003), although the high variability of wetlands associated with the hydrological disturbance regime comprises a marked difference to shallow lakes (Middleton 1999). Since hydrology is recognized as the primary force influencing wetland ecology (Mitsch and Gosselink 2000), hydrological fluctuations induce changes in nutrient budgets (Sánchez-Carrillo and Álvarez-Cobelas 2001) and, by extension, in the ecological functioning of wetlands (e.g., Sánchez-Carrillo and Angeler 2010). Recent studies, however, are beginning to highlight disadvantages when natural wetlands are exposed to prolonged nutrient loading as they result, for example, in net emissions of greenhouse gases to the atmosphere (Verhoeven et al. 2006), contributing to enhance the negative effects associated with global climate change.

The traditional perception of wetlands as nutrient sinks resulted in a focus on the effects of inflowing nutrient-enriched waters on wetlands rather than the other way round (Lowe and Keenan 1997). Because of the diversity of wetland types, the responses of these systems to eutrophication are varied. Nutrient enrichment has been shown to influence all trophic levels within a wetland and effects include changes in species abundance, displacement, biodiversity loss, and shifts in community structure and composition (Piceno and Lovell 2000, Álvarez-Cobelas et al. 2001, Guntenspergen et al. 2002, Liston et al. 2008). Abiotic shifts induce biotic changes and these, in turn, lead to complex feedback effects on ecosystem metabolism. However, the quantitative effects of nutrient enrichment on biological communities and how nutrient cycling is altered have not been studied sufficiently yet in wetlands. These topics are very important in describing wetland eutrophication patterns. The purpose of this chapter is to provide a general description of the effects of eutrophication on wetlands, with regard to biogeochemical processes, biological responses, and wetland ecosystem functioning. The final sections are devoted to the biological and chemical indicators of eutrophication as well as how to cope with eutrophication.

9.2 The Wetland Hydroperiod and Nutrient Transformations

Wetlands are unique hydrosystems where water level varies seasonally, sometimes interannually. This annual fluctuation of water level is known as wetland hydroperiod which depicts the length of time and portion of year the wetland holds water. Hydroperiod integrates all aspects of wetland water budget (rainfall, evapotranspiration, runoff from adjacent areas, flooding, net seepage of ground water) and is probably the main signature of a wetland ecosystem which impinges on functional (biogeochemical transformations) and structural (biota) characteristics (Brinson 1993). The duration and frequency of inundation of a wetland site vary according to its hydrodynamic setting, depending on regional differences in physiography and climate and on antecedent soil moisture conditions (Winter 1988, Skaggs et al. 1991, Brinson 1993, Mausbach and Richardson 1994). Since the wetland hydroperiod controls nutrient cycling through changes on soil redox transformations (oxygen availability as electron acceptor), fluctuations of water level are crucial for understanding wetland eutrophication processes.

In a general scope, wetlands can be grouped into three major hydroperiod categories: short, intermediate, and long-time inundation duration. Short hydroperiods have ephemeral or temporary wetlands (including vernal pools) that hold water for less than 4 months a year. These wetlands tend to dry during summer months (Fig. 9.1a). Intermediate hydroperiods have also ephemeral wetlands that hold water for at least 4 months and tend to dry in late summer or later, drying completely only in years with low precipitation. Therefore, these wetlands in some years may hold water year-round (Fig. 9.1b). Finally, long hydroperiods are those wetlands that never dry up; they always hold water (Fig. 9.1c). They are also called "permanent" wetlands such as lakes and some ponds.

Wetlands can be also discerned hydrodynamically as a function of its degree of ecosystem closure (Hopkinson 1992). It shows the wetland connectivity

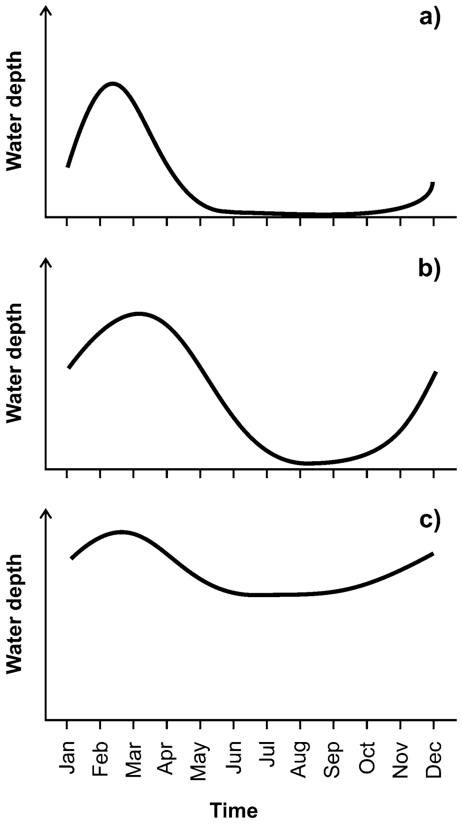

Fig. 9.1 Major wetland hydroperiod categories: short (**a**), intermediate (**b**), and long-time inundation duration (**c**)

to downstream and exerts strong influence on ecosystem dynamics as wetland exports water and chemical substances downstream, affecting retention and some chemical processes. The degree of closure of a wetland system is strongly related to its rate of water turnover (or water renewal time), which is strongly related to wetland nutrient loading (Mitsch and Gosselink 2000). In general, the openness of a wetland to hydrological fluxes determines nutrient loading, and it is an important determinant of wetland productivity, organic matter decomposition, and nutrient cycling.

9.2.1 Biogeochemical Transformations in Wetlands Under Anaerobic Conditions

The diverse hydrologic conditions in wetlands markedly influence nutrient biogeochemical processes (changes in the chemical forms and spatial movement of nutrients within wetlands, as water-sediment exchanges and plant uptake, and with surrounding ecosystems; Mitsch and Gosselink 2000). Because of the shallowness of wetlands, most nutrient transformations occur in wetland soils. While oxygen governs most of the biogeochemical reactions in upland soils, in wetlands during inundation, oxygen in soils is curtailed because soil pores are filled with water (oxygen diffusion in water is around 10,000 times slower than diffusion in air; Reddy and DeLaune 2008). The rate at which the oxygen is depleted depends on the ambient temperature, the availability of organic substrates for microbial respiration, and sometimes the chemical oxygen demand from reducing compounds such as ferrous iron. Under anaerobic conditions reduction of inorganic electron acceptors, accumulation of reduced compounds, accumulation of organic matter as a source of electron donor occur.

Oxygen is the preferred electron acceptor for microorganisms and, therefore, is always used first by them when it is available in the substrate. In the absence of oxygen, facultative anaerobes and obligate anaerobic microorganisms predominate in the substrate (Schlesinger 1997). These specialized microorganisms have the capacity to switch to other oxidants that replace oxygen in supporting biological oxidation of organic substances. The sequential thermodynamic reduction of inorganic electron acceptors according to changes in oxygen availability (as measured by the redox potential) occurs in a predictable sequence (Table 9.1). The reduction on the metabolic free energy (ΔG) determines the order of microbial processes under anaerobic conditions. The order of reductions in wetland soils starts with oxygen followed by oxides of nitrogen as nitrate (denitrification), oxides of iron and manganese, sulfate (sulfate reduction) and elemental sulfur, and carbon dioxide (methanogenesis). The rate at which these compounds are consumed in the wetland soil depends on their concentration, readily biodegradable organic compounds, and the microbial population involved in the process (Reddy and DeLaune 2008). The oxygen depletion (reduced conditions) in wetland soils appears quickly on the order of several hours to a few days after inundation (Turner and Patrick 1968, Fig. 9.2). Denitrification process develops during first aerobic stages but the rate increases strongly after oxygen is curtailed. Contrarily, reduction of iron does not begin until fully anaerobic conditions are achieved (Fig. 9.2).

Table 9.1 Thermodynamic sequence of reactions for reduction of inorganic substances by hydrogen (pH=7 and 25°C)

Reaction	Eh (V)	ΔG (kcal $\text{mol}^{-1}/\text{e}^{-}$)^a
Reduction (disappearance) of O_2	0.812	−29.9
Reduction of NO_3^-	0.747	−8.4
Reduction of Mn^{4+} to Mn^{2+}	0.526	−23.3
Reduction of Fe^{3+} to Fe^{2+}	−0.047	−10.1
Reduction of SO_4^{2-} to H_2S	−0.221	−5.9
Reduction of CO_2 to CH_4	−0.244	−5.6

Eh: redox potential, ΔG: free energy of reaction

^{a}Assuming coupling to the oxidation reaction: $\frac{1}{4}CH_2O + \frac{1}{4}H_2O \quad \frac{1}{4}CO_2 + H^+ + e^-$ and $\Delta G = -RT \ln(K)$

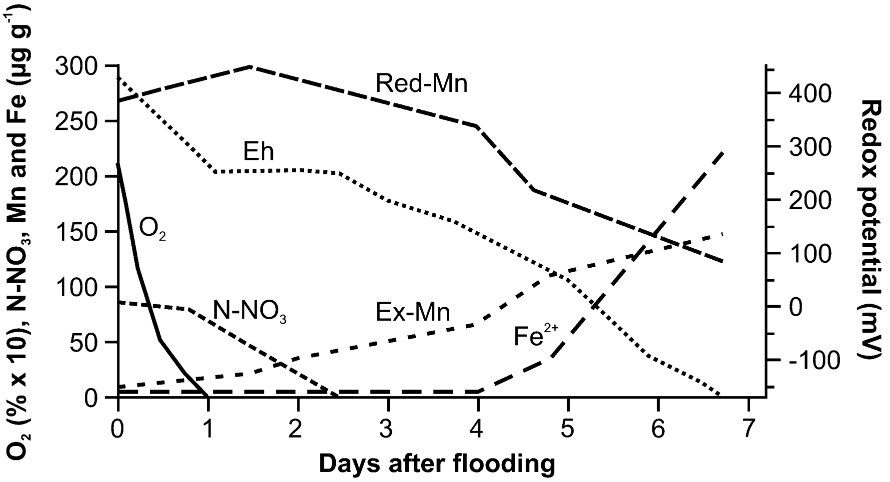

Fig. 9.2 Effects of flooding on the water chemical composition overlying a flooded soil. Redox potential (Eh) is expressed at pH 7. Red-Mn is the easily reducible Mn and Ex-Mn is the exchangeable Mn (modified from Turner and Patrick 1968)

Although the oxygen supply to the wetland soil is restricted during inundation, the oxygen demand continues to be high. These conditions result in the development of a thin layer of oxidized surface soil which has a significant effect on regulating nitrogen reactions and phosphorus sorption. It also functions as a sink for reduced compounds which diffuse from the underlying anaerobic soil layer (Reddy and DeLaune 2008).

9.2.2 Nitrogen and Phosphorus Cycling in Wetlands

Although very recent findings demonstrate some controversy about the role of N and P in limiting the productivity of aquatic ecosystems (Lewis and Wurtsbaugh 2008, Schindler et al. 2008), nitrogen is often but not only the most limiting nutrient in flooded soils. Because of the presence of anoxic conditions in wetlands, microbial denitrification of nitrates to gaseous forms of nitrogen in wetlands and their subsequent release to the atmosphere remain one of the main nutrient transformations occurring in wetlands. In addition, nitrates serve as one of the first terminal electron acceptor in wetland soils after oxygen disappearance, making them an important chemical in the oxidation of organic matter in wetlands. Nitrogen transformations in wetlands involve several microbiological processes. The presence of an oxidized zone in the soil over the anaerobic zone is critical for several of the nitrogen pathways (Mitsch and Gosselink 2000). Nitrogen retention is of particular interest in wetlands because they retain almost twice the amount of nitrogen than lakes (mean 64% of N inputs, Sanders and Kalff 2001). It has been shown that nitrogen retention depends mainly on both the nitrogen loading and the water residence time (Sanders and Kalff 2001). Water renewal rate in wetlands is lower depending on hydrodynamic openness, density and cover of macrophyte stands and those related to human intervention (damming, channelization, etc). Although nitrogen sedimentation could often be considered as the primary mechanism of nitrogen retention in wetlands, denitrification has been observed to be one order of magnitude larger than sedimentation. In an experimental wetland measured denitrification rates were 3.0–3.3 g N m^{-2} day^{-1} whereas sedimentation achieved values were 0.16–0.27 g N m^{-2} day^{-1} (Van Oostrom 1995). In a natural floodplain wetland, Saunders and Kalff (2001) also cited the same pattern between denitrification (1.3 g N m^{-2} day^{-1}) and sedimentation (0.6 g N m^{-2} day^{-1}). Denitrifying bacteria play an important role in the carbon cycle of wetlands as they contribute significantly to the carbon mineralization budget (up to 50% of the carbon mineralized in eutrophic freshwaters has been attributed to denitrifier activity, Andersen 1977, Christensen et al. 1990). The relative importance of macrophyte uptake as a nitrogen retention mechanism appears to be small compared to denitrification or sedimentation (Van Oostrom 1995). Despite the relatively small weight of plant uptake in total nitrogen retention, aquatic plants affect nitrogen cycling indirectly by retaining N during the growing season. It can influence the growth of phytoplankton by sequestering nitrogen during the highest algal demand period (Sanders and Kalff 2001). Also nutrient assimilation by plant wetlands impinges on nutrient sedimentation rates by contributing particulate matter to sediments during their senescence (Sánchez-Carrillo et al. 2001). Wetland vegetation contributes to create suitable environment for denitrification by increasing the supply of potentially limiting organic carbon and nitrate to denitrifying bacteria (Reddy et al. 1989, Brix 1997). Finally, as retention, nitrogen export downstream in open wetlands is recognized to depend on N loading and water renewal time. N export downstream in wetlands appears to be lower oscillating 10–40% of the N inputs (Kadlec and Knight 1996, Saunders and Kalff 2001).

Phosphorus is not a limiting factor in wetlands, although there are few exceptions (Reddy and DeLaune 2008). Phosphorus requirements of biota are usually much lower than that of nitrogen (mean N:P ratio of 16:1), as compared to available nitrogen in wetlands. Several wetland communities can assimilate phosphorus beyond their requirements through a process called "luxury uptake" and store phosphorus in their tissues as polyphosphates. It has been observed in sediment bacteria (Khoshmanesh et al. 2002) as well as in some wetland plants such as *Leersia oryzoides* (Kröger et al. 2007). Depending on water retention capability wetlands can operate as both a source and a sink for phosphorus. Usually, phosphorus-rich wetlands function as source of phosphorus to adjacent ecosystems. Phosphorus retention in wetlands has been cited to be regulated by macrophytes, periphyton and plankton, plant litter and detrital accumulation, soil physicochemical properties, water flow velocity, water depth, hydraulic retention time, length-to-width ratio of the wetland, phosphorus loading, and hydrologic fluctuations (Reddy and DeLaune 2008). Phosphorus retention mechanisms include uptake and release by aquatic vegetation, periphyton, and microorganisms; sorption and exchange reactions with soils and sediments; chemical precipitation in the water column; and sedimentation and entrainment (Reddy et al. 1999). Since the wetland environment provides nearly all of these favorable conditions, the ability of wetlands to accumulate phosphorus in soils is considered to be high, retaining between 40 and 90% of total phosphorus inputs, depending on the wetland hydrology, substrate composition, and redox conditions (Reddy et al. 1999). Water entering wetlands is not well mixed, resulting in typically high concentrations near the inflows, decreasing with the distance from the source (Reddy et al. 1993). Natural and constructed wetlands receiving inflows enriched in phosphorus exhibit same gradients. The reason for this spatial pattern is that phosphorus is carried in a plug flow fashion through the wetland (Reddy and DeLaune 2008), with high phosphorus sedimentation in wetland soil near to the inflow (Fig. 9.3, Sánchez-Carrillo et al. 2001). Seasonal retention rates of phosphorus usually peak at the end of summer and in fall, which match the senescent stage of macrophytes (Johnston et al. 1984, Sánchez-Carrillo et al. 2001), although higher rates have been also reported during the initial vegetation growth periods (Meeker 1996). One of the most controversial aspects of the buffer function of wetlands has focused on their ability to retain phosphorus in the long term. Wetland substrates can only hold a limited quantity of phosphorus. Since natural wetlands are also sites where sediment accretes, if the rate of sediment accretion is greater than the rate of phosphorus absorption, then wetlands can store phosphorus. In fact, wetlands tested as wastewater treatment systems became

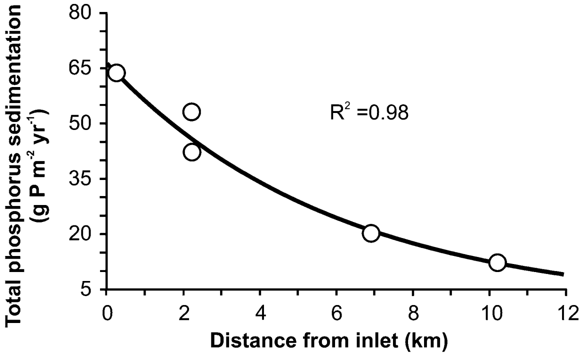

Fig. 9.3 Relationship between distance from the inlet and total annual phosphorus sedimentation in Las Tablas de Daimiel wetland during 1997–1998 (data source from Sánchez-Carrillo et al. 2001)

phosphorus-saturated in a few years, with the export of excessive quantities of phosphate (Richardson 1985).

Abiotic phosphorus retention by wetland soils is controlled by pH, iron, aluminum, and calcium content of soils, organic matter, phosphorus loading, and phosphorus concentration in soils (Reddy and DeLaune 2008). Changes in redox potential affect phosphorus solubility in wetland soils and therefore their retention. Ann et al. (1999) found that under reduced conditions, dissolved P concentrations increased in soils, which decreased with increasing redox potential. Several studies suggested that phosphorus sorption in wetland soils is related with amorphous and poorly crystalline forms of iron and aluminum (Richardson 1985, Gale et al. 1994, Reddy et al. 1998). Furthermore, total organic matter complexed with iron and aluminum also exerts an indirect effect on phosphorus sorption by wetland soils (Reddy et al. 1998). Because phosphorus retention (adsorption) in soils occurs at low concentrations, if concentration in soil pore water increases beyond the capacity of soil to adsorb phosphorus, it can precipitate some solid phases (e.g., under alkaline conditions: dicalcium phosphate, octacalcium phosphate, and hydroxyapatite; Reddy and DeLaune 2008). Similarly, continuous accretion of phosphorus in soils increases the dissolved phosphorus concentrations of soil pore waters, which results in flux from soil to the overlying water column (Reddy et al. 1999). Although periphyton and some macrophytes (e.g., submerged, floating, and some emergents as cattail and cut-sedge) have demonstrated statistically significant relationships with phosphorus concentration in the water column (McCormick et al. 1998, Sánchez-Carrillo and Álvarez-Cobelas 2001, Álvarez-Cobelas et al. 2010), plant uptake of phosphorus and incorporation into detrital tissues only must be considered as short-term storage as phosphorus is released after plants die off. Contrarily, the abiotic retention of inorganic phosphorus in soils can be considered as long-term storage of phosphorus by wetland ecosystems. Total phosphorus content in wetland soils varies from 30 to 500 mg P kg^{-1} in wetlands not impacted by anthropogenic phosphorus loading, until more than 10,000 mg P kg^{-1} in severely eutrophied wetlands receiving large discharges of phosphorus from urban wastewater and agricultural and animal operations (Reddy and DeLaune 2008). In spite of phosphorus not being a limiting nutrient in wetlands, phosphorus export downstream in open wetlands is generally low (10–60% of the total P inputs). Long-time exposition of wetlands to high nutrient loading has resulted in an increase of phosphorus concentration at the outflow during high inundation periods (Sánchez-Carrillo and Álvarez-Cobelas 2001).

9.3 Main Nutrient Sources to Wetlands: External Load vs. Internal Load

Since wetlands are hydrologically variable, nutrient inputs to the ecosystem come from various sources which can change seasonally or annually. Fluctuations in hydrological conditions induce large quantitative changes in nutrient inputs to the wetland (Sánchez-Carrillo and Álvarez-Cobelas 2001). However, the weight of the external vs. internal nutrient sources in the nutrient budget of the wetland often does not depend on the hydrologic regime. This is a distinctive feature between lakes and wetlands. While plankton controls nutrient cycling in lakes which mainly depend on external sources (Harper 1992), macrophytes and microbial communities usually control nutrients in wetlands through sedimentation and recycling (Mitsch and Gosselink 2000). Therefore, internal loading (i.e., release of accumulated nutrients to the water column) comprises the biggest nutrient fraction in wetlands as a result of continuous recycling by bacteria (Sánchez-Carrillo et al. 2001). External nutrient loading increases net accumulation of organic matter and associated nutrients in wetlands through accelerated primary productivity (Craft and Richardson

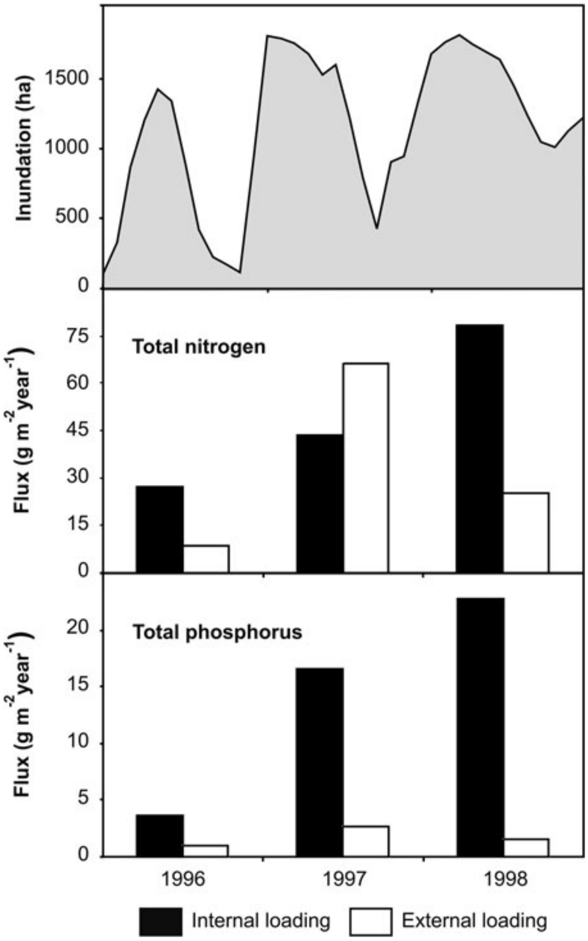

Fig. 9.4 Annual external and internal loadings of nitrogen and phosphorus and inundation in Las Tablas de Daimiel from 1996 to 1998 (data source from Sánchez-Carrillo and Álvarez-Cobelas 2001)

1993, Reddy et al. 1993). However, increased loading not only increases total nutrient content of soils but also increases soluble forms, which can potentially be released into the water column (Reddy et al. 1998). Mechanisms involved in phosphorus mobilization between soil and overlying water column are advection, dispersion, diffusion, seepage, resuspension, and bioturbation (see Reddy and DeLaune 2008 for further information).

Several studies have demonstrated the importance of internal loading of P in the nutrient dynamics of wetlands affected by long-term nutrient loading. In semiarid fluctuating wetlands such as Las Tablas de Daimiel (Central Spain), internal loading becomes the main phosphorus source to the wetland, even in the years of high external inputs (Fig. 9.4, Sánchez-Carrillo and Álvarez-Cobelas 2001). At this site, the ratio of internal vs external phosphorus loading experienced a roughly yearly doubling as humid conditions were imposed (3.93, 6.19, and 14.26, respectively). Also, in the Water Conservation Area 2A (WCA-2A) of the northern Florida Everglades (USA), Fisher and Reddy (2001) reported that these soils function as a source of P to the overlying water column (Fig. 9.5). These authors estimated that the measured P flux (2 mg P m^{-2} day^{-1}) would be sustained for a period of approximately 5 years, assuming that approximately 25% of the total P in the top 30 cm of soil is potentially mobile and can diffuse. Phosphorus internal loading is a critical factor in regulating eutrophication status of the wetland once external loads are curtailed.

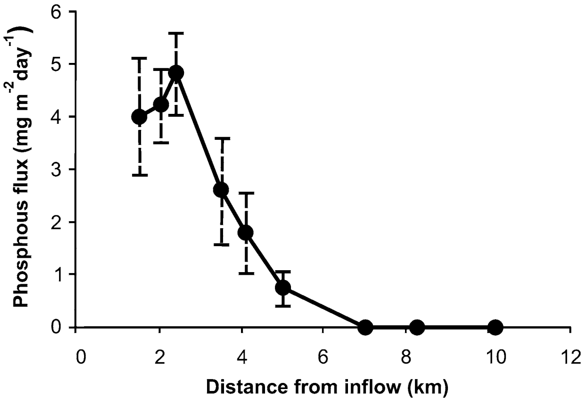

Fig. 9.5 Mean soluble phosphorus flux (±standard error) from the wetland soil to the water column measured in the Water Conservation Area 2A at the northern Florida Everglades (USA) (data source from Fisher and Reddy 2001)

9.4 Biogeochemical Responses of Wetlands to Nutrient Enrichment

As nutrient loading increases, biogeochemical processes in wetlands are altered, changing concentrations in water and soil and, therefore, nutrient fluxes. These ecosystem-wide changes modify conditions within the wetland and alter physical and chemical environment. The addition of limiting nutrients to ecosystems increases primary productivity and stimulates microbial processes. Organic matter decomposition and nutrient cycling in wetlands depend on the chemical and physical composition of soil, microbial activity, and nutrient availability (Wright et al. 2009). Therefore, changes affecting soil characteristics through increase of organic matter and accretion rates can result in severe physical and hydrological changes to wetlands, including disappearance by siltation to short term (Sánchez-Carrillo et al. 2001).

The first step of nutrient enrichment in wetlands is a productivity increase of wetland plants. High primary productivity associated with nutrient-rich areas has been observed in numerous wetlands (Davis 1991, Childers et al. 2003, Álvarez-Cobelas et al. 2010). It promotes increases of autochthonous organic matter accumulation in wetland soils (Sánchez-Carrillo et al. 2001). Usually, an increase of nutrient loading is linked to hydrological alterations and shifts in vegetation patterns can be expected (i.e., plant species replacement, Green and Galatowitsch 2002, Childers et al. 2003, Álvarez-Cobelas et al. 2008). Therefore, changes in organic supply and litter quality can be expected in high-nutrient scenarios (Engelhardt and Ritchie 2002). Because microbial activity responds to nutrient loading and hydrology (Wright and Reddy 2001), strong shifts in decomposition rates and nutrient cycling can also be expected as wetlands become eutrophic. We have previously commented that after prolonged exposure to high nutrient loading an increase of the wetland internal loading can be expected and hence a reduction in nutrient retention. Because sediment releases into the water column tend to increase the nutrient soluble forms during eutrophication (Reddy et al. 1998), more available nutrients promote more productivity, resulting in a positive feedback process which can last years. Thus, nutrient loading into these wetlands enhances organic matter decomposition and microbial activity which increase nutrient concentrations in water (Wright et al. 2008). Although decomposition in these wetlands appears to be dominated by anaerobic pathways (denitrification, sulfate reduction, and methanogenesis; Wright and Reddy 2001), some studies have shown that also aerobic heterotrophic microbial activity is often enhanced by nutrient loading (Aers and Toet 1997, Qualls and Richardson 2000). In the Everglades it has been observed that heterotrophic microorganisms are limited by the high C/P ratios of organic matter but external P loading removes this limitation and induces changes in microbial activity (DeBusk and Reddy 1998). However, aerobic CO_2 production could not be related to the content of dissolved nutrients in soils (Wright et al. 2009) whilst microbial biomass appeared positively related to nutrient loading (D'Angelo and Reddy 1999). While heterotrophic microbial activity is immediately enhanced after addition of oxygen in nutrient-rich wetland experiments, net soil organic carbon mineralization did not demonstrate significant differences using other electron acceptors as NO_3^-, SO_4^{2-}, or CO_2 under anaerobic conditions (D'Angelo and Reddy 1999). Soil phosphorus microbial biomass responds positively to phosphorus enrichment in wetlands (Qualls and Richardson 2000, Newman et al. 2003). Labile phosphorus (the most available P fraction to vegetation and microbial communities, Ivanoff et al. 1998) has been cited as the most responsive P fraction and sensitive to eutrophication in wetland soils and suspended sediments (consisted of algae, periphyton, and particulate organic matter; White et al. 2006). Carbon and nitrogen microbial biomass did not consistently respond to nutrient loading (Wright et al. 2008, 2009). Microbial C and N are linked to C and N cycles but cannot be considered as sensitive to nutrient loading. Although eutrophication promotes mineralization rates of both nitrogen and phosphorus, the latter has been cited as the most responsive microbial indicator to nutrient enrichment in wetlands (Wright et al. 2009). Mineralized phosphorus in wetland soils appears to be significantly related to soil total phosphorus and phosphorus microbial biomass.

Increased availability of sulfate in wetlands is known to cause serious eutrophication problems, as S^{2-} produced by SO_4^{2-} reduction interacts with Fe-PO_4^{3-} complexes in the sediment to produce FeS_2 and FeS, resulting in mobilization of phosphate (Smolders and Roelofs 1993, Lamers et al. 1998). High nitrate concentrations in groundwaters have been cited to inhibit eutrophication of sulfate-rich freshwater wetlands (Lucassen et al. 2004). Since NO_3^- is an energetically more favorable electron acceptor in anaerobic wetland soils than Fe and SO_4^{2-} (Table 9.1), high NO_3^- loads function as a redox buffer, preventing reduction of Fe and SO_4^{2-}. Therefore, limited SO_4^{2-} reduction prevents S^2-mediated mobilization of PO_4^{3-} from Fe–PO_4^{3-} complexes. At higher redox potential, reduced Fe is oxidized, increasing the content of Fe (III) capable of binding PO_4^{3-}. While the typical loading rates of nitrogen and phosphorus in natural and constructed freshwater wetlands exceed proposed critical loads to prevent eutrophication (total phosphorus: 10 kg P ha^{-1} $year^{-1}$, total nitrogen: 25 kg N ha^{-1} $year^{-1}$), wetlands continue to have serious problems preserving its structure (species composition) and functioning (nutrient cycling and retention; Verhoeven et al. 2006). The potential impact of climate change on wetlands shows that they will be significantly altered because temperature and precipitation are strong determinants of freshwater wetland ecosystem structure and function (Mitsch and Gosselink 2000). Lower water availability will reduce the flushing rate of wetlands as water renewal time increases. It probably will contribute to increase the high nutrient state of wetlands as biogeochemical processes will be promoted. In the same way, because wetlands are a major natural source of greenhouse gases to the atmosphere we could expect changes on the emissions of N_2O and N_2 as a consequence of eutrophication. Nitrogen emissions, which have a global warming potential 310 times that of CO_2 (IPCC 1996), can be enhanced in the future as nitrate availability in wetlands continues to be high due to increased pollution. When nitrate availability is high, reduction of nitrate instead of N_2O is energetically favorable for denitrifiers (Barnard et al. 2005). This potential negative consequence of nitrate-rich wetlands is often ignored or downplayed. Thus, there is a great need for additional information about the risk of nitrogen emissions from nitrogen-loaded wetlands (Verhoeven et al. 2006).

9.5 The Biological Effects of Wetland Eutrophication: Community Structure, Alternative Stable States, and Trophic Cascades

The biological symptoms of wetlands degraded by eutrophication differ little from those observed in temperate shallow lakes (Jeppesen 1998, Schutten and Davy 2000). Increased nutrient and water turbidity levels, shifts from submerged macrophyte dominance to phytoplankton dominance, decreased biomass of large-bodied zooplankton, elevated biomass of planktivorous and benthivorous fish, and losses of piscivores have been frequently reported in wetlands (e.g., Whillans 1996, Chow-Fraser et al. 1998, Álvarez-Cobelas et al. 2001). These changes are consistent with predictions made by alternative state (ASS) theory (Scheffer et al. 1993) and the trophic cascade concept (Carpenter and Kitchell 1993). The ASS concept posits that ecological systems may shift between contrasting states, the shifts being triggered either by altered community structure in similar abiotic environments (Chase 2003a, 2003b) or by changing abiotic environmental settings that cause major community shifts (e.g., Scheffer et al. 2001, Dent et al. 2002). In aquatic ecology, the ASS paradigm is based on research results from north-temperate, shallow lakes where cultural eutrophication is the main driver causing lakes to shift from the clear-water, submerged macrophyte-dominated state to the degraded, turbid, phytoplankton-dominated state (e.g., Scheffer et al. 1993, Jeppesen et al. 1997). This classic example from shallow lakes is an ideal starting point from which to reflect on the current status of the ASS concept in aquatic ecology and its particular application to wetlands. Our most pressing environmental problems demand consideration of diverse anthropogenic impacts that act in concert with global climate change, potentially pushing ecosystems irreversibly to other alternative states (Falk et al. 2006). This raises major challenges in managing novel ecosystems in ways that go beyond the simple control of nutrient loading and algal blooms and includes maintaining sustainability that guarantees the provision of ecosystem services to humans (Kremen 2005). Given the many ecosystem services which wetlands provide to humans and wildlife, understanding the processes that contribute to shift wetland ecosystems between ASS is relevant with regard to impact assessment and management interventions. In addition to alternative state theory, considerable evidence indicates that also concepts related to trophic cascading interaction are useful for understanding characteristics and responses of ASS in wetlands (Chow-Fraser 1998, Álvarez-Cobelas et al. 2001). As has been pointed out previously, the trophic status of aquatic ecosystems seems to mediate in the configuration of food webs and thus trophic cascading interactions from fish to plankton. Many aquatic ecosystems affected by agricultural or urban activities remain eutrophic, despite considerable reduction in external nutrient loading. Large quantities of phosphorus may be retained in sediments and subsequently work to maintain wetlands in the degraded state (Scheffer et al. 1993). Under such high-nutrient conditions, wetlands remain in a turbid (degraded) phase; phytoplankton is abundant and zooplanktivores and/or benthivores dominate the fish community. Under such conditions planktivorous and benthivorous fish can affect food web structure and ecosystem characteristics in different ways.

In shallow lakes and wetlands, zooplanktivorous fish contribute to eutrophication chiefly via food web-mediated effects, limiting zooplankton biomass and hence relieving phytoplankton from zooplankton grazing. The resulting high phytoplankton biomass contributes to high turbidity, which in turn constrains submerged macrophytes. Benthivorous fish, on the other hand, increase the nutrient availability to phytoplankton chiefly by transferring sediment-bound nutrients to the water column during bottom foraging (Meijer et al. 1990). Benthivorous fish, like for example common carp, also contribute to high non-algal turbidity via resuspension of sediment particles and by causing mechanical damage to submerged macrophytes (Crivelli 1983, Brabrand et al. 1990).

In contrast, community structure in shallow lakes and wetlands differs at lower nutrient concentrations, typical of the clear-water state. Submerged macrophytes dominate because of an improved light climate. Plants provide predation refugia for large daphnids, thus contributing to more control of phytoplankton via zooplankton grazing. Submerged plants also control phytoplankton via competition for light and nutrients (Balls et al. 1989, Ozimek et al. 1990, Van Donk et al. 1993) and/or by allelopathy (Wium-Andersson et al. 1982, Jasser 1995). Here, fish communities tend toward higher piscivore to planktivore ratios, ultimately relieving large daphnids from predation by zooplanktivores. At intermediate nutrient concentrations (50–150 μg L^{-1} total phosphorus (TP); Jeppesen et al. 1997), either alternative state (clear or turbid) may persist and system shifts are possible, although shifts may be resisted by internal stabilizing mechanisms, such as the activities of crayfish and planktivorous and benthivorous fish. It is not yet clear which factors contribute to the prevalence of either state during different years under intermediate nutrient concentrations; however, climatic factors (mild or harsh winters) may play an important role (Piet Verdonschot, Wageningen University and Research, The Netherlands, personal communication).

9.6 Biomanipulation of Wetlands as a Tool for Eutrophication Mitigation

Despite the similarity of symptoms, restoration strategies have been curiously different between eutrophied wetlands and shallow lakes. In shallow lakes, a step-by-step approach has generally been applied, starting with control of catchment nutrient inputs (e.g., wastewater treatment facilities, regulatory policies to reduce runoff), followed by in situ manipulations (biomanipulation, macrophyte implantation, sediment dredging, nutrient precipitation) (e.g., Annadotter et al. 1999, Madgwick 1999, Søndergaard et al. 2000). However, little of the methodology found to be useful in shallow lake restoration has been applied to wetland management (Kusler and Kentula 1990, Wilcox and Whillans 1999). Angeler et al. (2003) emphasized the potential usefulness of biomanipulation for wetland eutrophication abatement. In its original concept, biomanipulation encompassed manipulation of all lake biota and their habitats to improve water quality (Shapiro et al. 1975). More recently, the term is typically applied to top-down manipulation of lake fish communities, i.e., enhancement of piscivores or reduction of zooplanktivores and/or benthivores (Lammens et al. 1990) to achieve long-lasting improvements in water quality and shifts in plankton and benthic community dynamics (see reviews in Reynolds 1994, Perrow et al. 1997, Hansson et al. 1998, Drenner and Hambright 1999, Gulati et al. 2008). A strong conceptual basis for food web management exists, but related strategies have not melded well with other aspects of wetland ecology or management. Although the importance of biotic interactions as influences on wetland communities and ecosystem structure is becoming well documented (Zimmer et al. 2002), they have been largely neglected by scientists and managers. Recent studies, however, indicate that manipulation of fish standing stocks could contribute significantly to wetland eutrophication abatement (Chow-Fraser 1998, Lougheed et al. 1998, Angeler et al. 2002).

Catchment attempts to achieve eutrophication abatement should have primacy for restoring degraded aquatic ecosystems. However, studies of shallow lakes have shown that catchment nutrient management programs may be insufficient because of substantial storage of nutrients in sediments (Havens et al. 2001). Hence, lake internal management may be more useful to reduce eutrophication effects. By definition, wetlands are shallow ecosystems (<2 m), and this has important implications for management. Jeppesen et al. (1990) suggested that effects of planktivorous and benthivorous fish increase with decreasing water depth in shallow lakes, and similar assertions have been made for wetlands (Chow-Fraser 1999, Angeler et al. 2002). Such effects are manifest in (1) high zooplankton production per unit volume as a result of high primary production, hence strong top-down control of fish on zooplankton is likely; (2) the limitation of zooplankton refugia in shallow, turbid systems, especially when macrophytes are absent; (3) fish ingestion of sediment with high nutritive value because of low settling times in the water column, thus populations of obligate and facultative benthivores are sustained, even when invertebrates are scarce; and (4) the more pronounced effects of sediment resuspension by benthic-dwelling fish in shallower systems, because turbidity may affect the entire water column. Considerable evidence indicates that fish reduction schemes should have potential positive effects in many deteriorated wetlands (Table 9.2), and that fish catch per unit effort should increase with decreasing water column depth. Thus, it is reasonable to suggest that biomanipulation, when appropriately timed in accordance with low water levels, may be a low cost–high benefit tool for wetland eutrophication abatement.

Fish occupy a key position in lake food webs, and it may not surprising that the biomanipulation paradigm is mainly based on the notion that interventions in fish community structure have cascading effects toward lower trophic levels, ultimately improving water quality. The complexity of wetland food webs and the strong influence of abiotic factors require a focus on different communities and habitats, as initially proposed by Shapiro et al. (1975). In fact, recent studies suggest that the introduction of bivalves (e.g., *Dreissena polymorpha* Pallas) could be used to control cyanobacterial blooms (Gulati et al. 2008); however, the introduction of these species could cause more ecological damage than benefits; therefore, a thorough assessment of impacts is needed. The following examples highlight that targeting multiple communities could extend the usefulness of biomanipulations beyond water quality improvements to control key abiotic wetland processes.

A striking difference between many shallow lakes and riparian wetlands is the low ratio of open water area to space covered by emergent vegetation in marshes. With the exception of, for example, Lake Neusiedlersee (Austria/Hungary), emergent vegetation in shallow lakes is frequently limited to a littoral fringe. By contrast, vast areas of riparian wetlands are covered by large extents of emergent plants (Mitsch and Gosselink 2000), where they play important roles in many biological (primary production, decomposition, nesting ground for birds, refuge for fish and other wildlife) and abiotic processes (sedimentation patterns (Sánchez-Carrillo et al. 2000, 2001), wetland hydrology (Sánchez-Carrillo et al. 2004)). Thus, emergent vegetation clearly represents a key biotic component in this wetland, and the following example highlights that an extension of biomanipulations to cope also with other target communities than fish may be useful for wetland rehabilitation.

A case study is that of Las Tablas de Daimiel National Park, which is a semiarid floodplain wetland located in central Spain. Before large-scale degradation took place during the second half of last century, mainly in the form of wastewater discharge and groundwater abstraction, the emergent macrophyte community of this wetland was dominated by the evergreen *Cladium mariscus*. Nowadays, *Cladium* has largely been replaced by the annual *Phragmites australis*, which better tolerates the hypertrophic conditions and irregular flooding patterns in the wetland (Alvarez-Cobelas and Cirujano 2007, Alvarez-Cobelas et al. 2008). Sánchez-Carrillo et al. (2001) demonstrated that internal primary production, mainly through emergent vegetation, accounts for the considerable variability in sedimentation patterns of Las Tablas de Daimiel. The rates determined were substantial (max. 2.88 ± 1.2 cm year^{-1}), suggesting that, if current sedimentation patterns are maintained, the wetland will silt up and convert to a terrestrial ecosystem by the end of this century (Sánchez-Carrillo et al. 2000). This provides a strong argument in favor of intervention in the vegetation community structure, which could help to decrease the present sedimentation rates and extend the life span of Las Tablas de Daimiel.

Table 9.2 Comparison of selected water quality and biotic variables in response to biomanipulation in selected wetlands. The table has been taken from Angeler (2010)

Site	Secchi depth	Phytoplankton biomass	Total P	Total nitrogen	Turbidity	Cladocerans	Submerged vegetation	Intervention	Type of study/duration	Sources
Prairie wetland (USA)	n.d.	–6.1[a]	–1.5[a]	–2.0[a]	–4.7[a] as NTU	+176.8[a]	n.d.	Planktivore elimination with rotenone	Whole ecosystem study; 4 years	Zimmer et al. (2001)
Cootes Paradise Marsh (Canada)	n.d.	n.s.[b]	–1.6[b]	n.d.	–2[b] as NTU	n.s.[b]	n.d.	Carp exclusion	Enclosure study; 15 days	Lougheed et al. (1998)
Tablas de Daimiel floodplain (Spain)	n.s.	–2.6[b,c]	–2.3[b,c]	–1.4[b,c]	–3[b,c] as total suspended solids	+ ca. 250[b,c]	n.s.	Carp, sunfish, and mosquitofish exclusion	Enclosure study; 6 weeks	Angeler et al. (2002)
Ventura marsh (USA)	+2.9[d]	–11.5[d]	–ca. 1.2[d]	n.d.	n.s.	+ ca. 2.3[d,e]	+ (n.c.)	Benthivore elimination with rotenone	Whole ecosystem study; 14 months	Schrage and Downing (2004)
Major Lake (Hungary)	+2.1[d]	–(n.c.)	–1.3[d]	n.d.	n.d.	+ (n.c.)	+ 4.6[d]	Planktivore and benthivore removal, piscivore stocking	Whole ecosystem study; 4 years	Tátrai et al. (2005)
Delta marsh (Canada)	n.d.	n.d.	n.d.	n.d.	n.s.	n.d.	+11.9[b]	Planktivore and benthivore exclusion	Exclosure study; ca. 4 months	Evelsizer and Turner (2006)
Prairie wetlands	n.d.	n.s.	n.s.	n.d.	n.d.	+7.03[a]	n.s.	Piscivore stocking	Replicated whole ecosystem experiment; 3 years	Potthoff et al. (2008)

Values indicate the multiplicative change observed in each variable in response to the manipulation; ±, increase or decrease of value, respectively; n.d., no data; n.s., not significant; n.c., no calculations possible based on original study (but tends to increase or decrease after the manipulations are indicted)

[a] Changes observed between a treatment and a reference wetland

[b] Values calculated by comparing enclosures with highest fish stock and fishless controls

[c] Shown are carp data only, given that its effects were most deleterious for water quality

[d] Comparing periods before and after fish manipulations

[e] Values refer to body length and not biomass

9.7 Conclusion

In conclusion, even though wetlands are usually seen as efficient nutrient sinks, however, the prolonged exposition to high nutrient loading has demonstrated serious negative effects on wetland structure and functioning. Because the eutrophication symptoms in wetlands differ little from those observed in temperate shallow lakes, restoration strategies may be similar than those applied to shallow lakes. Sediment dredging and biomanipulation of fish standing stocks could contribute significantly to wetland eutrophication abatement, when appropriately timed in accordance with low water levels.

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Chapter 10

Effects of Contamination by Heavy Metals and Eutrophication on Zooplankton, and Their Possible Effects on the Trophic Webs of Freshwater Aquatic Ecosystems

Ana María Gagneten

Abstract In this chapter, the combined effects of eutrophication and of heavy metal contamination on the zooplankton community of a freshwater ecosystem are analyzed. Through biomonitoring, it was possible to study zooplanktonic attributes as indicators of environmental stress: species richness, species diversity, equity, and biomass. These attributes allowed the detection of structural and functional changes. There was an inverse relationship between stress situations and zooplankton body size with a proliferation of *r*-strategist species (rotifers) and opportunistic species (nauplii larvae), a dominance of tolerant species, and a decrease in the most sensitive ones, such as larger size crustaceans (copepods and cladocerans). The results of this study showed that zooplankton responds as a good descriptor of water quality, constituting an efficient tool to assess eutrophication and heavy metal contamination. A general diagram integrating possible effects of eutrophication and heavy metal contamination on the trophic webs of freshwater ecosystems is also included. Emphasis in biological control is suggested as a relevant control measure.

Keywords Aquatic ecosystems · Eutrophication · Heavy metals · Zooplanktons

10.1 Introduction

Unfortunately, the most spread and generalized use of surface water courses is as a means of transport to evacuate urban and industrial residual wastes. However, under the paradigm of the "multiple use of water – the precious fluid and a basis of life on the Earth, *sensu* Khan and Ansari (2005)", the use of a water body for a certain purpose should not damage other possible uses, as consumption, preservation of aquatic life, or recreation. As Moss (1999) pointed out, most freshwater systems have been seriously altered by human activities. We may wish to restore them to self-sustaining systems that provide conservation or amenity values or products such as poTable water or fish, which is completely impossible without profound understanding of their functioning. Water eutrophication in lakes, reservoirs, estuaries, and rivers is widespread all over the world and the severity is increasing, especially in developing countries like Argentina. The eutrophication of several water bodies leads to significant changes in the structure and functioning of the aquatic ecosystems (Khan and Ansari 2005). Eutrophication and various forms of pollution, which cause both foreseen and unforeseen problems, must be addressed and solutions must be found. However, this is a complex problem that cannot have a simple solution. In recent years it has become apparent that toxicity testing using single species is not adequate to assess the potential hazard of anthropogenic compounds and eutrophication. The studies of community-level impacts are a very useful tool for understanding the effects on the ecosystems. For example, Xu et al. (2001) proposed a set of ecological indicators for a lake ecosystem health assessment. The structural indicators included phytoplankton cell size and biomass, zooplankton body size and biomass, species diversity, macro- and micro-zooplankton biomass, the zooplankton–phytoplankton ratio, and the macrozooplankton–microzooplankton

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ratio. This case study demonstrated that this method provided results which corresponded with the lake's actual trophic state. In general terms, the studies on the change in structure, function, and diversity of the ecosystems have been used as parameters to assess the effects of contamination and eutrophication.

The objective of this study was to contribute to the knowledge of heavy metal–zooplankton interactions and the factors that condition the levels of heavy metals in zooplankton, such as the degree of eutrophication of systems that, due to their complexity, continue without a solution. Urbanization and intensive agriculture exploitation produce excessive nutrient inputs to lentic and lotic bodies, promoting algal proliferation and other eutrophication symptoms. This process has an adverse effect in water quality, because of the decrease in oxygen, the increase in turbidity, and interferences in water potabilization processes. In this study, we approached this problem taking a freshwater system with problems of eutrophication and contamination by heavy metals as an example. The effects of tannery wastewater with high contents of heavy metals, nutrients, and sulfide along a pollution gradient on the zooplankton assemblage in the lower Salado River basin (Santa Fe, Argentina) were assessed.

The lower Salado River is one of the most important basins in Argentina. It receives inputs of heavy metals, mainly from tanneries and metallurgic industries, thus representing an important segment of the economy. The Salado River runs along 2,010 km from northeastern Argentina, to the Santa Fe Province, where it joins the Paraná River. In the lower basin, where this survey was performed, it also receives nutrients of different sources, especially from agricultural origin. The levels of organic matter, dissolved oxygen, nitrites, nitrates, and phosphates showed that the system is eutrophicated. DBO values allow us to classify the studied systems as meso or polisaprobial. The values of chromium, copper, cadmium, and sulfide were higher than standard ones. Zooplankton density, biomass, species richness, and species diversity diminished along the pollution gradient. Cladocerans were the less tolerant organisms and *Eucyclops neumani* dominated the copepods. This survey allowed the understanding of the contamination of the ecosystem in terms of eutrophication and heavy metal concentration and their effects on zooplanktonic attributes. The aim of this chapter was to identify problems in a polluted freshwater environment, find general patterns, and extract recommendations for successful biomanipulation. Emphasis in biological control is suggested as a relevant control measure.

10.2 Methodology

Five sampling sites, considered to be polluted, were established along approximately 40 km (Fig. 10.1). The section was selected according to a pollution gradient: Salado River at Manucho (MSR), two sites in Las Prusianas Stream (LP1 and LP2), and two sites in the North Channel (NCH) and the South Channel (SCH). The reference site was located in the Salado River, 153 km upstream from San Justo city (SJSR).

In each sampling site, we measured pH, temperature, dissolved oxygen, turbidity, and conductivity. Sulfide and organic matter values, water hardness, chemical and biological oxygen demands (QOD, BOD), dissolved organic carbon (DOC), nitrates, nitrites, phosphates (NO_3, NO_2, P_3O_4), total suspended solids (TSS), and metal concentrations (Cr, Cr VI, Pb, Cu, and Cd) of river water samples were also recorded (see methodology details in Gagneten et al. 2007). To perform zooplankton analysis, five zooplankton samples (replicates) were taken at each site with a 20 L Schindler-Patalas trap of 45 μm mesh size, fixed and stained in situ. The quali-quantitative analysis of samples was carried out for mesozooplankton (adult copepoda and cladocerans) and for microzooplankton (rotifers and copepod nauplii). The attributes of the community selected as variables of response were total density (No ind L^{-1}) and by-group density (Copepoda, Cladocera, and Rotifera), micro and mesozooplankton density and biomass (μg L^{-1}). Species diversity through the Shannon–Weaver index and its components of richness (S) and equity (E) were also calculated.

One-way ANOVA with a significance level of $p \leq 0.05$ was conducted to determine whether the differences among concentrations could be significant between contaminated and control sites and for the comparison of the composition of the zooplanktonic assemblage. Data were normally distributed (Kolmogorov–Smirnov test). Hierarchical cluster analysis (Euclidean measures, UPGMA method) was used to study the different sampling sites based on physicochemical records, concentrations of metals,

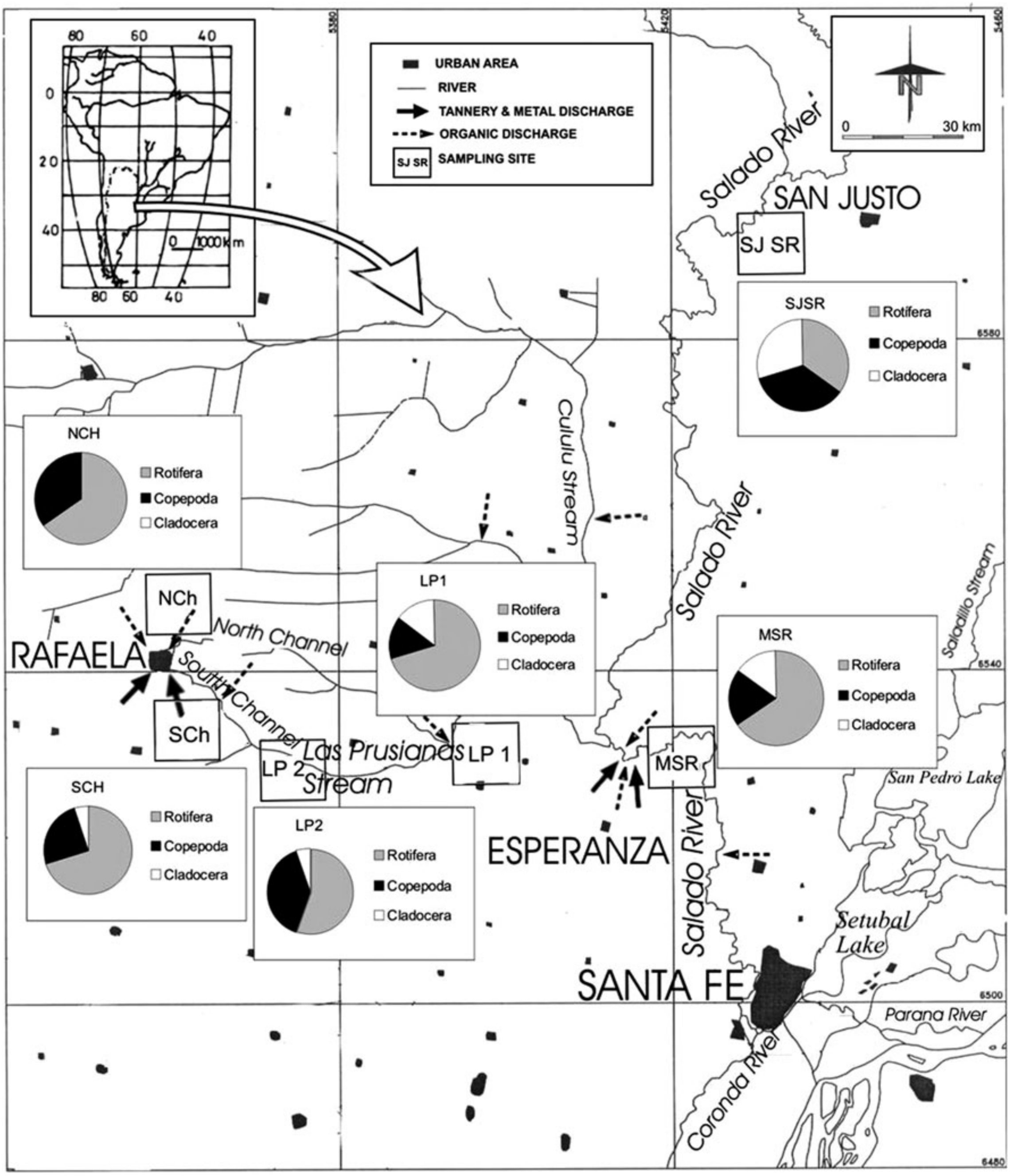

Fig. 10.1 Map of the Salado River basin showing location of the sampling sites, species diversity (H), total species richness (S), and relative richness of Rotifera, Copepoda, and Cladocera, recorded at each sampling site (Modified from Gagneten and Paggi 2009)

and community attributes (Zar 1984, Hair et al. 1999) using the program InfoStat (2007). For details, see Gagneten and Paggi (2009).

10.3 Results

10.3.1 Environmental Context

The spatial distribution of some physical and chemical parameters recorded in the water of the sampling sites is shown in Figs. 10.2 and 10.3.

Temperature changed throughout the study period (16–29°C), showing a normal seasonal dynamic. Depth was <1 m in the channels and Las Prusianas, but larger in the Salado River (MSR=3.50 m, SJSR=5.70 m). Turbidity showed high variability and great differences between sampling sites: 3–54 NTUs (Nephelometric Turbidity Units). High concentrations of TSS (Fig. 10.3) were recorded in the South Channel (mean 3.662 mg L^{-1}), intermediate values were found in Las Prusianas (mean 1.602–2.158 mg L^{-1}), and minimal values were recorded in the Salado River (mean 1.848 mg L^{-1} in SJSR and 2.698 mg L^{-1} in MSR). The median pH range was 7.5–7.8 (Fig. 10.2), with higher values in winter and lower values in summer at all sampling sites. Conductivity was relatively high (>1,000 μS cm^{-1}) at all study sites, a characteristic pattern of this river as it is suggested by its name ("salado" = salted). Highest values were recorded in Las Prusianas (3,000–7,100, mean 5,965 μS cm^{-1}) and in Salado River, Manucho (3,900–5,300, mean 3,260 μS cm^{-1}). Total hardness was high in the South Channel (mean 502.9 $CaCO_3$ L^{-1}, Fig. 10.3). This parameter showed minimal values in the North Channel and in the Salado River (mean 164 mg $CaCO_3$ L^{-1}). Very low values of dissolved oxygen were recorded, being extremely low in Las Prusianas in winter and spring

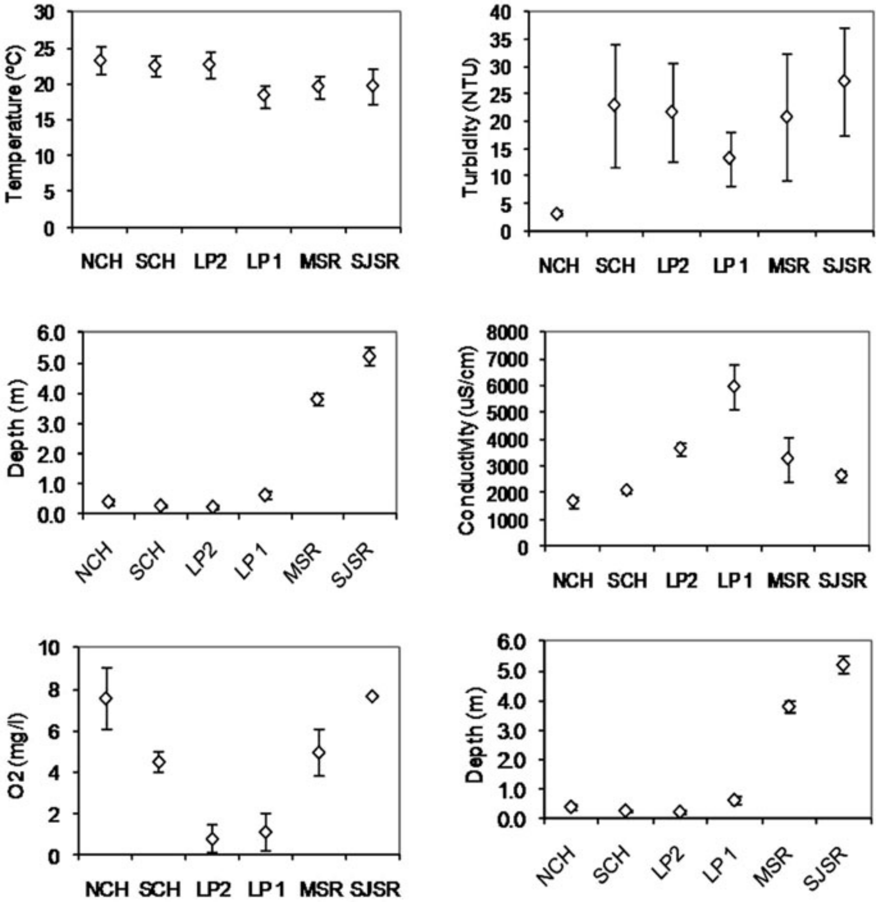

Fig. 10.2 Physicochemical parameters of sampling sites. Values correspond to the mean of four samples at each sampling site and the error bars represent one standard deviation (Modified from Gagneten et al. 2007)

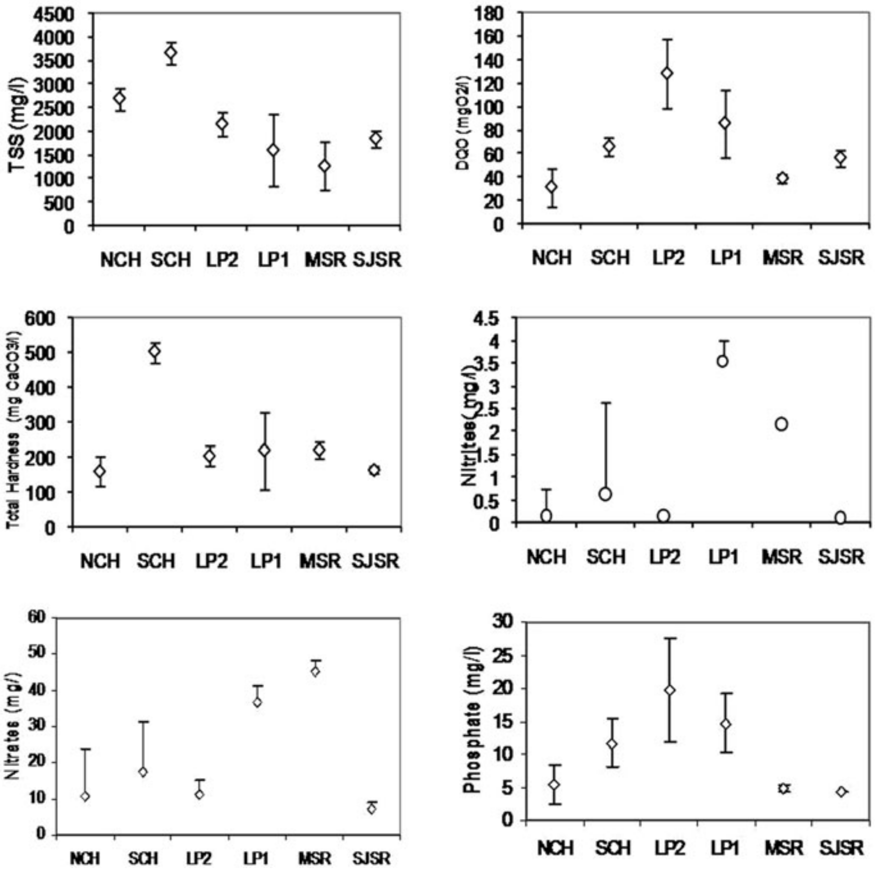

Fig. 10.3 Chemical parameters of sampling sites. Values correspond to the mean of four samples at each sampling site and the error bars represent one standard deviation (Modified from Gagneten et al. 2007)

(0.1–0.2 mg L^{-1}). This parameter only showed high values at the reference site (SJSR 8 mg L^{-1}) and in a few other cases, but mostly lower than 6 mg L^{-1}. QOD values showed higher concentrations at Las Prusianas (mean 65.6 mg O_2 L^{-1} at Las Prusianas 1 and 128 mg O_2 L^{-1} at Las Prusianas 2, Fig. 10.3) and lower values at MSR (mean 30.8 mg O_2 L^{-1}). Nutrients (N and P) were higher at all sampling sites than at the reference site (Fig. 10.3), indicating an eutrophication process. The results of previous research indicate that the ratio 0.95:1 between ammonium and nitrate in the Salado River is definitely lower that those found in unpolluted water bodies included in the Paraná River floodplain. This fact could be interpreted as the product of an unlikely higher biological productivity or the consequences of pollution from human activities (Maglianesi and Depetris 1970).

Variable seasonal levels of Cr were recorded (Fig. 10.4), the highest ones being in the South Channel (11 μg L^{-1}, mean 5.36 μg L^{-1}), Las Prusianas (13.6 μg Cr L^{-1}, mean 7.03 μg L^{-1}), and Manucho (13 μg L^{-1}, mean 8.32 μg L^{-1}). Cr VI was high in the South Channel and Manucho (4.6 and 4.8 μg L^{-1}, respectively) and in San Justo (2.5 μg L^{-1}). Cr VI always showed values above the standard, even at the reference site. Pb was higher than the detection limit only in the South Channel (maximum 6.1 μg L^{-1}, mean 4.74 μg L^{-1}) and in Manucho (mean 5.1 μg L^{-1}). Relatively high values of Cu were found in water at all sampling sites (maximum 22.9 μg L^{-1}, mean 13.0 μg L^{-1} in the South Channel), even at the control site (14.1 μg L^{-1}, mean 8.16 μg L^{-1}). Cd in water showed higher values than standard values in Manucho (maximum 1.9 μg L^{-1}, mean 0.85 μg L^{-1}).

Cr in water was sometimes higher than Canadian (8.9 μg L^{-1}, CEPA 2003) but not Argentine (44 μg L^{-1}, Subsecretaria de Recursos Hídricos de la Nación 2003) standards at sampling sites. On the other hand, Cr VI in water showed higher values than the Canadian standard (1.0 μg L^{-1}) and sometimes than the Argentine standard (2.5 μg L^{-1}) at all sampling sites. Standards for Cu were surpassed in the South

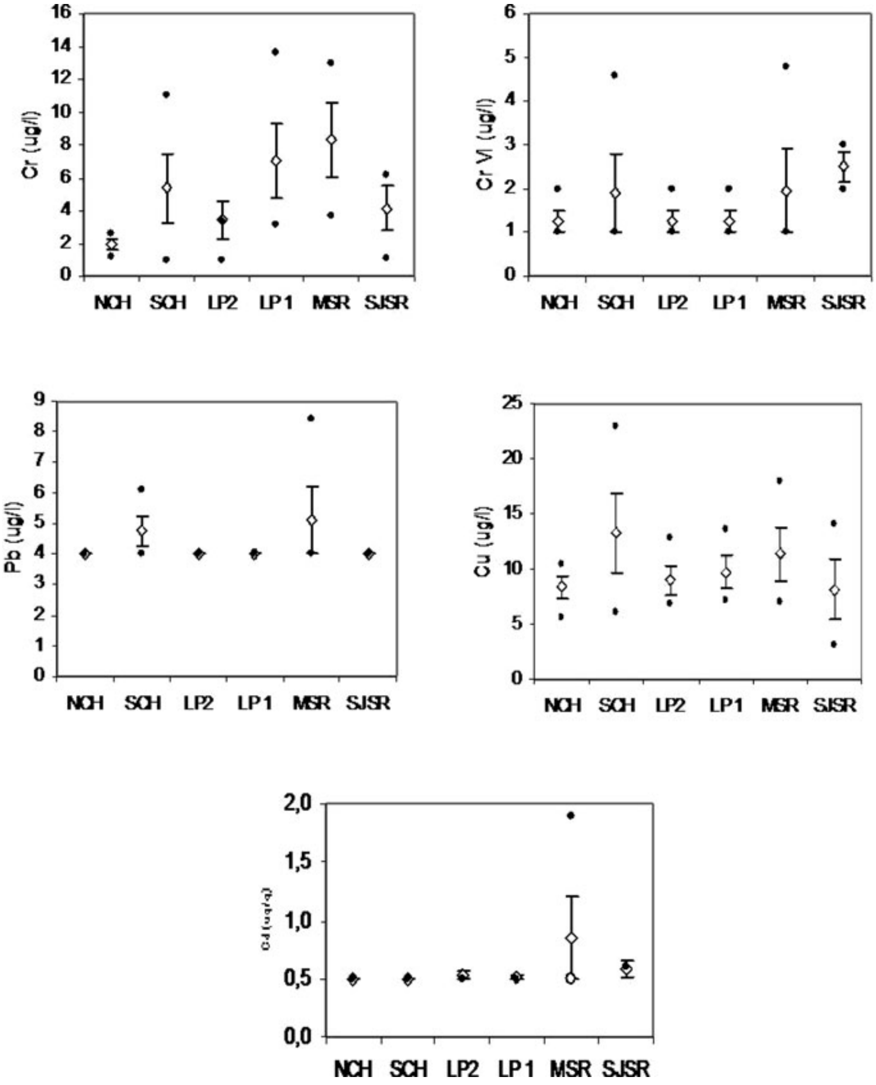

Fig. 10.4 Cr, Cr VI, Pb, Cu, and Cd content in water. Values correspond to the mean of four samples at each sampling site and the error bars represent one standard deviation. The dark circles indicate maximum and minimum values (Modified from Gagneten et al. 2007)

Channel and that for Cd exceeded the standard value in Las Prusianas. We can see that the pollution of the lower Salado River shows a close relationship with adverse impact of heavy metal contaminants and eutrophication. The water of Las Prusianas system and of the North Channel is contaminated with heavy metals if compared to the control site. Organic matter values were high (200–256 mg L^{-1}) although not very different between sampling sites (Table 10.1). BOD showed very high values in SCH and LP2 and high in NCH and LP1, corresponding to poly and mesosaprobial environments, respectively. Dissolved oxygen concentrations were very low (1.6 mgO_2 L^{-1}) in the sampling site closest to the effluent discharge (LP2), corresponding with the higher BOD (45.8).

Sulfide values (16–59.9 mg L^{-1}) allowed to recognize two environmental groups (Table 10.1): the furthest sites in the pollution gradient, with comparatively lower values (16–16.3 mg L^{-1}), and the closest sites in the pollution gradient, with higher values (59.5–59.9 mg L^{-1}). At all sites, however, sulfide concentrations were much higher than the reference level for surface freshwater (<1 mg L^{-1}). Total chromium concentration was highest at LP2 (215 μg L^{-1}), the site closest to the pollution source. This value was also much higher than permitted standards: 2 μg L^{-1} for protection of phyto and zooplankton and 20 μg L^{-1} for protection of fish (CEPA 2002).

In Table 10.2, correlation values between environmental variables and concentrations of

Table 10.1 Physicochemical parameters at North Channel (NCH), South Channel (SCH), Las Prusianas 2 (LP2) and Las Prusianas 1 (LP1) sampling sites

	pH	Sulfide (mg L^{-1})	Organic matter (mg L^{-1})	Depth (cm)	Transparency (cm)	Temperature (°C)	Total Cr (µg L^{-1})	Oxygen (mgO$_2$ L^{-1})	BOD	QOD	Nitrites (mg L^{-1})	Nitrates (mg L^{-1})
NCH	8.5 (0.3)	16.0 (0)	227.2 (17.5)	57.5 (14.0)	34.7 (14.0)	16.5 (4.5)	29.0	5.7	6.8	97.0	4.2	18.4
SCH	8.5 (0.1)	16.3 (0.7)	256 (45.2)	23.0 (2.8)	18.8 (6.3)	20.0 (4.5)	89	8.9	24.9	90.0	0.4	4.6
LP2	9.12 (0.8)	59.5 (0.7)	220.8 (87.2)	22.2 (8.0)	16.3 (2.7)	18.2 (3.5)	215	1.6	45.8	85.0	3.8	13
LP1	8.28 (0.4)	59.9 (1.0)	200 (16.0)	11.2 (2.0)	10.2 (3.6)	18.3 (2.9)	25.0	6.7	9.80	10.0	1.20	50.8

Table 10.2 Correlation values between water chemical variables and concentrations of chromium and sulfide

	Chromium	Sulfide	QOD	BOD	Oxygen	pH	Organic matter	Transparency
Chromium	–	0.392	0.38	0.989[a]	–0.714[a]	0.948[a]	0.036	–0.272
Sulfide	–	–	–0.659[a]	0.382	–0.587[a]	0.298	0.509[a]	–0.753[a]
QOD	–	–	–	0.339	–0.154	0.52	–0.217	0.703[a]
BOD	–	–	–	–	–0.614[a]	0.893[a]	–0.089	–0.361
Oxygen	–	–	–	–	–	–0.839[a]	–0.724[a]	0.025
pH	–	–	–	–	–	–	0.25	0.001
Organic matter	–	–	–	–	–	–	–	0.165

[a] Significant correlations

chromium and sulfide are shown. Positive correlations were found between Cr concentrations and BOD (0.989), pH and Cr concentrations (0.948), and pH and BOD (0.893). Negative correlations were registered between concentrations of Cr and O$_2$ (–0.714), O$_2$ and pH (–0.839), O$_2$ and BOD (–0.614), O$_2$ and organic matter (–0.724), and O$_2$ and Cr (–0.714). On the one hand, sulfide concentrations were negatively correlated to O$_2$ (–0.587) and QOD (–0.659). On the other hand, COD values were much higher than those of DO. This would mean an accumulation of organic matter, i.e., eutrophication as dominating condition.

10.3.2 Zooplankton Structure

10.3.2.1 Abundance

Total density of organisms was higher at the reference site (Salado River at San Justo, 0.86 ind L^{-1}) than at the more contaminated sites (0.31, 0.07, 0.03, 0.61, and 0.62 at the North Channel, South Channel, Las Prusianas 2, Las Prusianas 1, and Manucho, respectively). Copepods dominated the community in numbers. However, adult copepods were poorly represented quantitatively and qualitatively. The dominance observed at LP1, MSR, and SJSR is due to the great proliferation of larvae and juveniles (nauplii and copepodites). Nauplii reached densities of 6.9, 1.9, and 3.0 ind L^{-1} at LP1, MSR, and SJSR, respectively. In general terms, adult crustaceans were not as numerous as rotifers; the presence of cladocerans was very low or null at NCH, SCH, and LP2. The most frequent genera were *Bosmina, Ceriodaphnia,* and *Moina.* The most abundant species at LP1, MSR, and SJSR were *M. minuta, B. hagmani, Diaphanosoma spinulosum*, and *Macrothrix squamosa.* Among copepods, the most frequent genera were *Eucyclops* and *Metacyclops*, *Acanthocyclops* being represented in a lower proportion. The most frequent and abundant species was *E. neumani,* which was recorded in all environments and with a relatively high abundance, except at LP1.

Mesozooplankton was only well represented at San Justo, being scarce at Manucho and very scarce or null in the tributaries. Microzooplankton reached comparatively high values at Las Prusianas 1 (3.48 ind L^{-1}), caused by the abundance of nauplii, and was lower in the Salado River (1.10 and 1.55 ind L^{-1}) at Manucho and San Justo, respectively.

The high microzooplankton values are also explained by the abundance of rotifers, which were the best represented group, both qualitatively and quantitatively. The most frequent rotifer genera in relation to the number of species were *Brachionus* (10 species), *Lecane* (7 species), and *Keratella* (3 species). The most numerous species of the genus *Brachionus*, or with a more constant presence, were *B. quadridentatus*, *B. calyciflorus*, *B. plicatilis*, and *B. caudatus.* The latter, most of all abundant and frequent in the Salado River, was represented by different "varieties": *insuetus, provectus*, and *vulgatus*. *B. austrogenitus* and *B. alhstromi* were also frequent at Manucho and San Justo. The most numerous and frequent species of the genus *Lecane* were *L. lunaris* and *L. pyriformis*, and *K. americana* and *K. cochlearis* prevailed from the genus *Keratella.* The genus *Polyarthra* was recorded in the Salado River, with *P. vulgaris* showing a high density at San Justo (3.5 ind L^{-1}). Bdelloid rotifers (among them, *Philodina* sp.) were also frequent and abundant. Among the rotifer species of higher frequency, although represented with low density values, we can mention *Monostyla lunaris, Lepadella acuminata, Asplanchna* sp., and *Epiphanes* spp. Gagneten and Ceresoli (2004) showed significant negative correlations between zooplanktonic density with sulfide concentration ($r = -0.841$) and with Cr concentration ($r = -0.512$). These results show that both contaminants, and not only chromium, have important negative effects on the studied assemblage. Density showed significant positive correlations ($p < 0.05$) with depth, transparency, and temperature ($r = 0.941$; $r = 0.955$, and $r = 0.541$, respectively).

10.3.2.2 Biomass

Absolute biomass (B) was 17 μg L^{-1} for copepods (9.41, 4.24, 2.92, and 9.42 μg L^{-1} for Cyclopoida, Calanoida, Harpacticoida, and copepodites + nauplii, respectively), 4.2 μg L^{-1} for cladocerans, and 0.4 μg L^{-1} for rotifers. At Manucho and San Justo, zooplankton was constituted by the three main zooplanktonic groups: copepods, cladocerans, and rotifers, with high values of biomass. Biomass of copepods was high and constant (near 3 μg L^{-1}) at SJSR. It was somewhat lower at MSR. Biomass of Copepoda, concentrated in the river and at LP1, was distributed as follows: 55% Cyclopoida, 25% Calanoida, and 17% Harpacticoida. In decreasing order of importance, cladocerans showed biomass values between a minimum of 0.3 (LP1) and a maximum of 1.6 (SJSR), being absent at NCH. They were followed by rotifers, with comparatively lower values of biomass (0.01 at LP2 and 0.2 at NCH). Absolute biomass varied in the order SJSR>MSR>LP1>SCH>NCH>LP2 with 11.1, 4.9, 2.7, 1.5, 1.2, and 1.1 μg L^{-1}, respectively.

10.3.2.3 Species Richness and Species Diversity

A total of 74 species were recorded, from which 13.5% corresponded to copepods, 22.9% to cladocerans, and 63.5% to rotifers. At MSR a total of 59 species were recorded, while 56 species were recorded at SJSR, 38 at LP1, 17 at SCH, 16 at NCH, and 13 at LP2. Therefore, species richness decreased among the sampling sites in the following order: MSR>SJSR>LP1>SCH>NCH>LP2. In function of richness, two environmental groups can be formed: the tributaries, with lower species richness (between 13 and 36 species), and the main river course at MSR and SJSR, with almost twice the number of species (between 56 and 59 species). The dominant group was rotifers, which were present at all sampling sites. In the river (MSR and SJSR), 99% of all rotifer species were represented. At LP1, 50% of species were represented; at LP2, 22%; and only 24% at NCH and SCH, with some species being exclusive from these environments. Such is the case of *Anuraeopsis fissa* and *Euchlanis dilatata.* The second group was copepods, with low species richness [one to two species in the tributaries and somewhat higher (six to seven species) in the river], while cladocerans contributed significantly to the community only at the reference site (RSSJ), where they showed a more uniform abundance. Figure 10.1 shows the relative richness of Rotifera, Copepoda, and Cladocera when considering the 20 most frequent species recorded at each sampling site. In the direction of the basin current, i.e., from NCH to MSR and in relation to RSSJ, the absence of cladocerans was observed at NCH, with absolute dominance of rotifers and scarce copepods. This situation was maintained at the other contaminated sites, but the presence of cladocerans increased progressively toward the river at Manucho (MSR). A similar proportion (that means higher equity) for the three groups was found in the river at San Justo (SJSR). Species diversity showed low values (0.35–1.56) in the tributaries and higher values in the Salado River at Manucho (3.0) and San Justo (3.16) (Fig. 10.1).

10.4 Discussion

There were differences between the concentration of metals in water in the more polluted sites and the control site. Heavy metals, especially chromium, copper, and cadmium, appear to be an important problem to the studied freshwater environment. When the effects of euthrophication and heavy metal contamination were assessed on the zooplanktonic community, we found that total density, by-group density (Copepoda, Cladocera, and Rotifera), micro and mesozooplankton density, biomass, species richness (S), and species diversity (H) were all good indicatos of water pollution: total density of zooplankton was significantly higher in the river than in the channels and streams ($p < 0.001$), with dominance of rotifers but a higher copepod biomass. Calanoida dominated over Cyclopoidea and Harpacticoida. Total species richness was 74, the highest values (59 and 56) being shown at the points corresponding to the Salado River at localities Manucho and San Justo (MSR, SJSR) and the lowest ones in North and South channels (NCH and SCH with 16 and 17 species, respectively) and in the two sampling stations of Las Prusianas stream (LP1, LP2) with 13–38 species. The species diversity showed low values (1.8–2.3) in channels and streams but higher values (3.0) in the Salado River at Manucho and San Justo. Absolute biomass varied in the order SJSR>MSR>LP1>NCH>SCH>LP2, similar to absolute density, which varied in the order SJSR>MSR>LP1>NCH>SCH>LP2. The comparison of the content of heavy metals in water between the control site (SJSR) and the most contaminated sites showed significant differences with the North Channel and Las Prusianas 1 and 2 streams (ANOVA; $p=0.001$, 0.012, and 0.011, respectively) and non-significant differences, although close to the significance level, with the South Channel and Manucho ($p=0.08$, 0.059, respectively). The following positive correlations were found: depth with mesozooplankton density, H, and S ($p < 0.001$); temperature with microzooplankton density, H, and S ($p < 0.004$); and a negative correlation between dissolved oxygen with mesozooplankton density, H, and S ($p < 0.01$) but not with microzooplankton, indicating a higher tolerance of the organisms belonging to this zooplankton fraction. A negative correlation was found between biomass of copepods and concentration of Pb and Cu ($p < 0.05$ and $p=0.01$, respectively). Rotifers were the most tolerant to heavy metal contamination, followed by copepods and cladocerans. Species diversity values (H) allowed differentiating between pollution levels. We conclude that S and H are good indicators of stress in polluted systems. Species richness (S) allowed separating studied environments into two groups: the tributaries, with lower species richness, and the river, with higher species richness. The decrease in specific richness and diversity observed at stations closer to the effluent source was related to the increase in chromium and sulfide concentrations. This result suggests that both substances and not only chromium are highly toxic to this community, which is generally not considered when the effects of tannery effluents on biota are discussed. Another result found in this study was the decrease in zooplankton biomass at a higher concentration of heavy metals. This result indicates that this parameter is also a good indicator of polluted aquatic systems.

Compared to less polluted systems of the region, zooplankton density in this system was similar but zooplankton biomass was much lower. This indicates the settlement and proliferation of smaller size species (rotifers). Rotifers were the most tolerant species; copepods followed rotifers, while cladocerans only contributed significantly to the community at San Justo, where a higher equity was also registered. Cladocerans showed very low tolerance to the toxic action of heavy metals. The clustering of biological and physicochemical variables and the concentration of heavy metals in water resume the picture of the effect on zooplankton assemblage and show three groups of environments (Fig. 10.5): the first one was the main course of the river, with lower contamination by heavy metals and higher density, biomass, H, and S, which separated clearly from the other two groups of the tributaries composed by channels (SCH, NCH) and streams (LP2, LP1). In the tributaries, *r*-strategists and a few tolerant species, such as *E. neumani*, proliferated. In general, the river offered better conditions for the development of the community: a higher flow

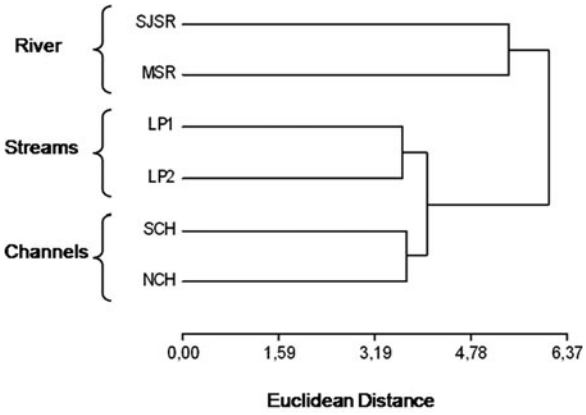

Fig. 10.5 Hierarchical cluster analysis (Euclidean measures, UPGMA method) based on biological zooplankton parameters and concentrations of heavy metals in water at the sampling sites (Modified from Gagneten and Paggi 2009)

and degree of dissolved oxygen than in the tributaries. This allowed the settlement of significant populations at Manucho, one of the polluted sites, and at San Justo, the initial reference site. Due to the high tolerance to tannery effluents and ubiquity of *E. neumani*, it is proposed as a water quality indicator species. In synthesis, the pollution gradient of the studied sites was Las Prusianas>Manucho>South Channel>North Channel>San Justo. The results of this study show that zooplankton responds as a good descriptor of water quality, constituting an efficient tool to assess eutrophication and heavy metal contamination. Data analysis shows the urgency to perform biological studies and to carry out remediation actions in the lower Salado River basin.

10.4.1 Integrating Possible Effects of Eutrophication and Heavy Metal Contamination on the Trophic Webs of Freshwater Ecosystems

When contamination by heavy metals is added to an eutrophication process it can turn out to be a very complex situation. As Clements and Newman (2002) pointed out, the studies of community-level impacts are a very useful tool for understanding pollution effects on the ecosystems. In this sense, responses of zooplanktonic species assemblage are a possible and reliable approach. This community, as it is constituted by organisms of different sizes and trophic habits and complex life cycles (parthenogenesis, sexual reproduction with larval and juvenile stages), is a valuable tool to characterize the environment biologically in areas with different degrees of anthropogenic impact (Fig. 10.5). Figure 10.6 summarizes the complex interrelations that can occur. Through biomonitoring, it is possible to study the attributes of the zooplanktonic community with a great potential as indicators of environmental stress: species richness (S), species diversity (H), equity (E), and biomass (B). These attributes allowed us to detect structural and functional changes. Structural changes: alterations in the community size structures are produced. Macrozooplankton reduces its number or disappears and rotifers, which would be the most tolerant species, increase markedly their density. Therefore, the composition changes, diversity decreases, and the community remains constituted by small size species, mainly smaller than 500 μm, i.e., rotifers, nauplii, and lower size cladocerans. The structure and size ranges in plankton are the first indicators of stress situations at the community level (Moore and Folt 1993). There is an inverse relationship between stress situations and zooplankton body size, with a proliferation of *r*-strategist species (rotifers) and opportunistic species (nauplii larvae), a dominance of tolerant species (*E. neumani*, in this example), and a decrease in the most sensitive ones, such as larger size crustaceans (copepods and cladocerans). Similarly, Takamura et al. (1999) registered a shift of zooplankton community structure from a *Daphnia–Acanthodiaptomus* community to a *Bosmina–rotifer* community, which probably led to a decrease of secchi disc transparency. The tolerance of rotifers would be determined by their lower sensitivity to toxics, their more rapid growth without molts, and their higher resilience (José de Paggi 1997). Functional changes: alterations in the intrazooplanktonic competition are produced. Macrozooplankton is substituted by microzooplankton. The selective elimination of larger size herbivore crustaceans (cladocerans and calanoid copepods) affects another trophic level, i.e., fish populations. Thus, changes in the trophic web are generated by the effects of the decrease in the available resource for larvae and juveniles of ichthyophagus fish and planktivorous adults. Similar results were obtained by Havens (1994) and Havens et al. (1993). Similar results were also recorded by Park and Marshall (2000) who

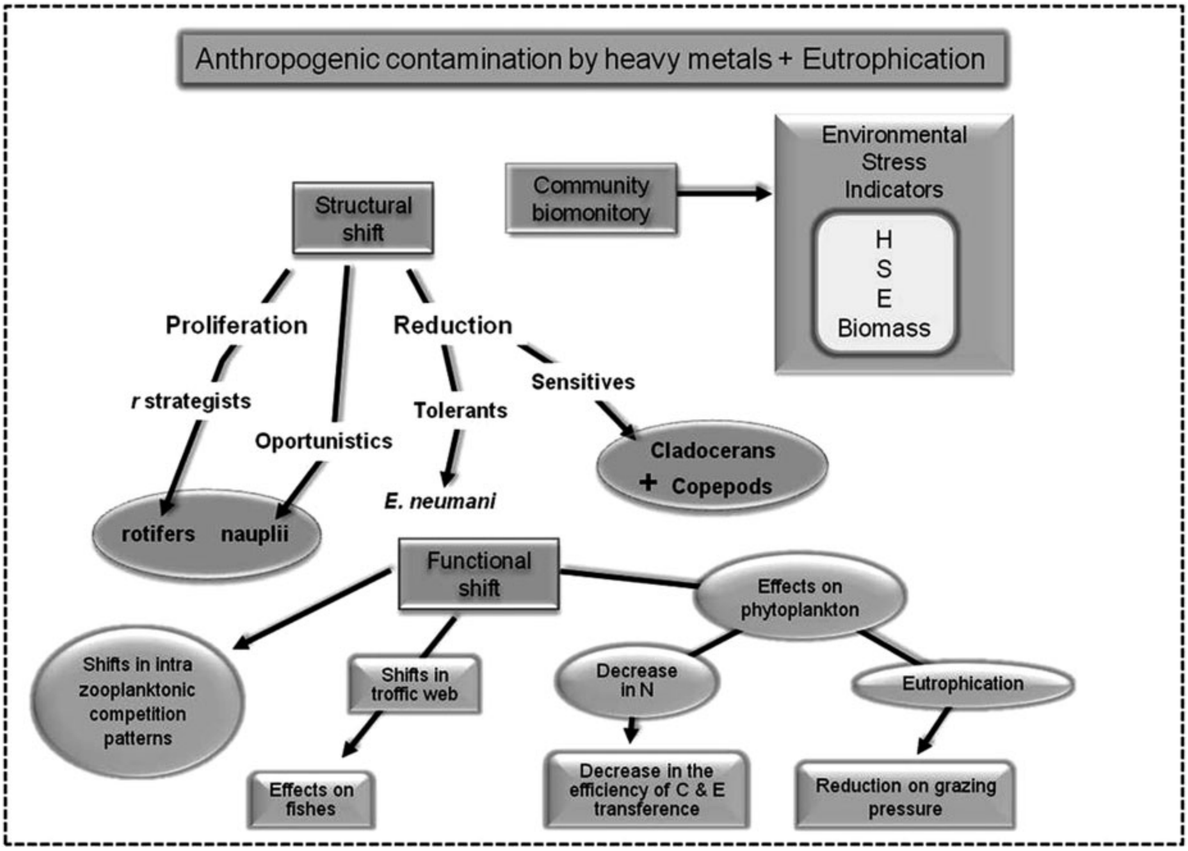

Fig. 10.6 Possible effects of eutrophication and heavy metal contamination on the trophic webs of freshwater ecosystems

investigated zooplankton and water quality parameters at the lower Chesapeake Bay and Elizabeth River to identify the changes of zooplankton community structure with increased eutrophication. The total micro- and mesozooplankton biomass decreased with the increase of eutrophication. However, the relative proportion of microzooplankton increased with increased eutrophication. Within highly eutrophied waters, the small oligotrichs (<30 µm) and rotifers dominated the total zooplankton biomass. However, tintinnids, copepod nauplii, and mesozooplankton significantly decreased with the increase of eutrophication. These patterns were consistent throughout the seasons and had statistically significant relationships. The authors also suggest that shifts in zooplankton community structure characterize an increasing eutrophication of an ecosystem. As shown in Fig. 10.6, cladoceran decrease generates changes in phytoplankton by decreasing the foraging pressure, which can increase the eutrophication process. This pattern was also addressed, among others, by Yang et al. (1998). A 12-year data analysis showed that since *Daphnia* feeds efficiently on phytoplankton, it could decrease concentration of Chl-a and enhance water transparency. Top-down control is an important type of interspecies interaction in food webs. Phytoplankton grazers contribute to the top-down control of phytoplankton populations, but chemical pollution may pose a threat to the natural top-down control of phytoplankton and water self-purification process (Ostroumov 2002, Bielmyer et al. 2006, Gama-Flores et al. 2006). On the one hand, a decrease in phytoplankton populations by direct toxic effect (Cu, for example, is a strong algaecide) can occur. This process can determine the decrease in the efficiency of carbon and energy transfer in the system. On the other hand, the decrease in phytoplankton affects negatively the filtering rate of cladocerans, which influences their growth rates. Water transparency and nutrient regeneration rate can also be affected (Somer 1998).

10.5 Summary

The contamination of the lower Salado River basin showed a very close relationship between the impact of heavy metals and the process of eutrophication on zooplanktonic assemblage. It is thus possible to conclude that zooplankters respond as good descriptors of water quality in complex situations, constituting an efficient tool together with other environmental parameters. Similarly, Beaver et al. (1998) suggested that abiotic factors which are known to directly affect phytoplankton may indirectly affect zooplankton composition in such a way as to use zooplankton assemblages as indicators of water quality. Moreover, pollutants finally reach the sea and can be found even in the traditionally less polluted environments. De Moreno et al. (1997) and Kahle and Zauke (2003) registered heavy metals in different groups of Antarctic invertebrates. The nutrient and metal removal from wastewaters through bioremediation using regional macrophytes such as *Eicchornia crassipes*, *Salvinia herzogii*, and *Pistia stratiotes* is proposed (Maine et al. 2004, 2005, 2006, 2007, Hadad et al. 2007). In other surveys, the growth response of *Lemna minor* and *Spirodela polyrrhiza* was studied for their possible application for remediating eutrophic waters (Ansari and Khan 2008, 2009). The biosorbent potential of algal cells for toxic metals also offers an effective and low-cost alternative to conventional methods for decontamination of industrial effluents containing metals (Rai et al. 2005, Baran et al. 2005, Beek et al. 2007, Regaldo et al. 2009, Gagneten et al. 2009). The most contaminant industries should be controlled in the tannery leather process, through the replacement of chromium salts by other less contaminant methods. In this sense, the United Nations Environmental Program (UNEP 1991) assessed that chromium salts should continue to be used because of their high affinity with carboxylic groups of the collagen fibers and of their price, which is comparatively lower compared to less contaminant methods. Finally, we should remember that the primary effects of contamination are exerted on the biota, *including* the human being. It is necessary to promote actions at different levels and sectors of the society, linked to the development of an adequate culture of water management, which will influence the improvement of life quality. In that sense, we can agree with the statement of Trevors and Saier (2009a, b): “We need to take care of the problems we currently recognize so that we are prepared to solve those about which we still have no inkling. The real rescue plan for the planet is environmental, not economic. Any rescue plan that does not aim to reduce the human population, the wasteful use of our resources and global pollution is doomed to failure.”

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Chapter 11

Impact of Eutrophication on the Seagrass Assemblages of the Mondego Estuary (Portugal)

Marina Dolbeth, Patrícia Cardoso and Miguel Ângelo Pardal

Abstract Human population has for long been attracted to live on the shores, imposing major pressures on transitional waters (including estuaries, lagoons) and adjacent coastal areas. A wide array of human impacts may be expected, colliding with the ecological function of these ecosystems and threatening their long-term integrity. Among major threats, eutrophication may be considered as a global ecological problem, affecting several worldwide coastal areas. The Mondego estuary (Portugal) is a coastal system, which has suffered eutrophication over the last three decades leading to major changes in environmental quality. Accordingly, this study addresses the responses of macrobenthic community and different key species to eutrophication, providing an insight on potential impacts for the whole ecosystem integrity. In the late 1990s a restoration plan was implemented in the system to control the eutrophication process and its main effects. A review on these major changes, occurring from 1993 to 2002, will be presented focusing on (1) nutrient dynamics; (2) seagrass and macroalgal dynamics; and (3) macrobenthic community biodiversity, density, biomass, production and feeding guilds composition, evaluating both the type and time of the response to the eutrophication effects. Additionally, six species will also be studied in more detail, which are representative of taxa commonly found at estuaries and other transitional waters, and important for the estuarine foodwebs: *Hydrobia ulvae* (Gastropoda), *Cyathura carinata* (Isopoda), *Scrobicularia plana* (Bivalvia), *Hediste diversicolor*, *Alkmaria romijni* and *Capitella capitata* (Polychaeta).

Keywords Eutrophication · Macrobenthic assemblages · Seagrass · Macroalgal Bloom · Multiple Stressors · Restoration

11.1 Introduction

Human development and the associated increasing population growth in watershed areas underlie many of the environmental problems occurring in freshwater, transitional (e.g. estuaries, lagoons) and coastal ecosystems. Nutrient enrichment (N and P) is one of the most prominent consequences directly related to the human activities (e.g. Kennish 2002, Paerl 2006), with eutrophication now considered as a global and worldwide problem in several aquatic ecosystems (Cloern 2001, Breitburg et al. 2009, Fox et al. 2009). The consequences of eutrophication may vary considerably, due to the integrating effect of the physical and biological characteristics of the ecosystem and climate combined with the nutrient loading itself (Cloern 2001, Rabalais et al. 2009). Several impacts on the biota have been studied, most of them addressing the effects on the structure (Raffaelli et al. 1998, Kennish 2002), and, to a lower extent, on the production (Dolbeth et al. 2003) and composition and stability of recipient food webs (Fox et al. 2009).

Similar to several transitional waters worldwide, the Mondego estuary, in the central Portugal (North Atlantic Ocean), has suffered from an ongoing cultural eutrophication process over the last 20 years. Several impacts were observed in the quality and

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quantity of the primary and secondary production levels, with huge increases in macroalgal productivity, concomitant with decreases in the macrophyte one (Lillebø et al. 2005, 2007, Leston et al. 2008), and general impoverishment of the biotic communities (macrobenthic communities: Dolbeth et al. 2003, 2007, Cardoso et al. 2008a, b; birds: Lopes et al. 2006; fishes: Leitão et al. 2007). In the meantime, a restoration programme was implemented in the system to control and reverse eutrophication and its main effects, which included experimental mitigation measures such as changes in hydrology to increase circulation and diversion of nutrient-rich freshwater inflow to the estuary. All these changes occurring in the Mondego estuary have been studied since the last 15 years, providing a large database and a comprehensive information background on the eutrophication event and on the efficiency of the measures to control eutrophication and restore the original seagrass community. Accordingly, this study will present a review on these changes, integrating information on nutrient dynamics, seagrass and macroalgal dynamics and macrobenthic fauna density, biomass, production and trophic structure. This information will be used to evaluate:

- type and time of the response to the eutrophication effects and
- how effective are being the measures to control eutrophication.

Additionally, the response of six estuarine species, representative of *taxa* usually found in estuarine systems, will be studied in more detail.

11.2 Case Study: The Mondego Estuary

The Mondego estuary (Portugal) locates in a warm temperate region, on the Atlantic coast of Portugal (40°08′N, 8°50′W), near to Figueira da Foz city. It is a small estuary (8.6 km^2 area), with two arms (north and south) of distinct hydrologic characteristics, separated by the Murraceira Island (Fig. 11.1). The north arm is deeper (4–10 m during high tide, tidal range 1–3 m) and constitutes the main navigation channel

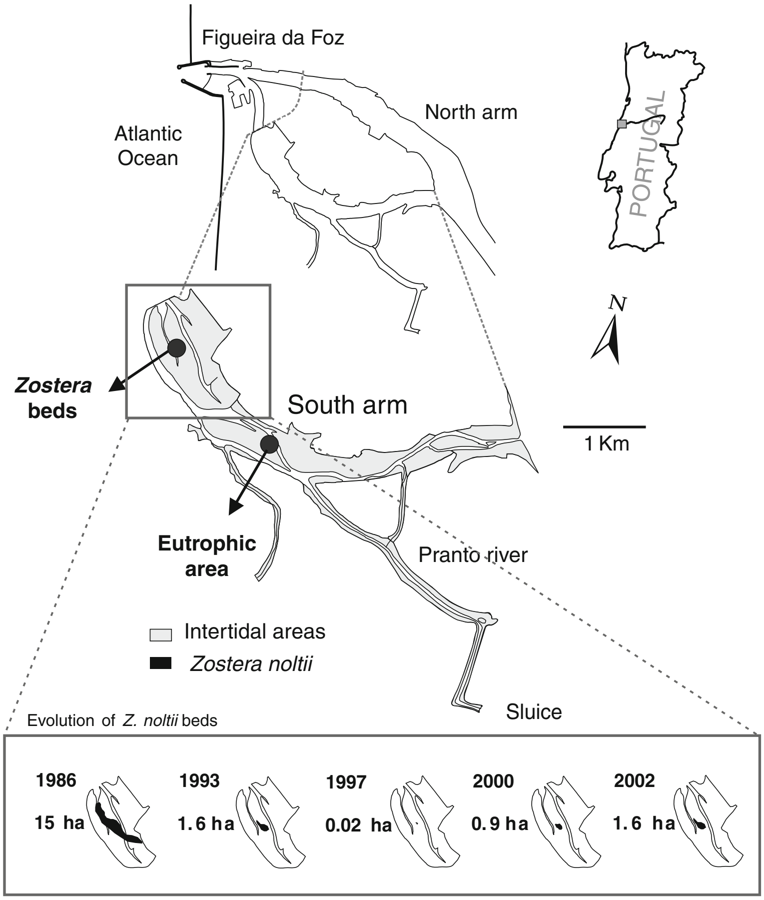


Fig. 11.1 The Mondego estuary, with indication of intertidal area, salt marshes and the seagrass bed (*Zostera noltii*) evolution in the south arm since 1986 until 2002 (*box*). Mapping of benthic vegetation is based in field observations, aerial photographs and GIS methodology (ArcView GIS version 8.2)

and the location of the Figueira da Foz harbour. The main freshwater inputs to the north arm are from the Mondego River (Fig. 11.1). The south arm is shallower (2–4 m during high tide, tidal range 1–3 m) and characterized by large areas of exposed intertidal flats during low tide (about 75% of total area). Before the implementation of the mitigation measures, water circulation in the south arm mostly depended on the tides and on the freshwater input from the Pranto River (Fig. 11.1), as the upstream areas were almost silted up, with only a small connection with north arm. The downstream areas of the south arm exhibit *Spartina maritima* marshes and *Zostera noltii* beds (Fig. 11.1). More details are available in Cardoso et al. (2008a) and Teixeira et al. (2008).

11.2.1 Anthropogenic Pressures

The Mondego estuary has a high regional value, sustaining several industries (mostly cellulose- and paper-related industries), aquaculture (several old salt ponds transformed into semi-intensive aquacultures), some traditional salt extraction, the location of the mercantile harbour of Figueira da Foz city and a marina (Ribeiro 2002). There is also an increasing human pressure in the area, with more than 60,000 inhabitants in Figueira da Foz city (Ribeiro 2002). Upstream the estuary, in the lower Mondego valley, there are agriculture fields (more than 15,000 ha of cultivated land), producing mainly rice (Lillebø et al. 2005), which drain nutrient-rich waters to the estuary. These anthropogenic activities have been the cause of high environmental pressure on the estuary mostly due to physical disturbance and to the high input of nutrients from agricultural fields and aquacultures (Lillebø et al. 2005, Cardoso et al. 2007, 2008a).

11.2.2 Eutrophication in the South Arm

The high input of nutrients from agriculture fields and aquaculture affected mostly the south arm. In fact, until the end of 1998, the south arm of the Mondego estuary was almost silted up in the upstream areas. Water circulation was mainly dependent on tides and on the freshwater input from the Pranto River (Fig. 11.1). This was artificially controlled by a sluice, according to the rice fields irrigation needs in the lower Mondego valley. The occasional freshwater flow to the estuary caused the high input of nutrients (discussed below) and high water residence time (about 5–7 days) (Lillebø et al. 2005). Consequently, since the 1980s, clear eutrophication symptoms were observed. The downstream areas maintained *S. maritima* marshes and the *Z. noltii* beds, but in the inner parts of the estuary the seagrass bed completely disappeared and blooms of the opportunistic macroalgae *Ulva* spp. were common (Lillebø et al. 2005, Cardoso et al. 2007, Dolbeth et al. 2007, Ferreira et al. 2007). The occurrence of macroalgal blooms was pointed out as one of the major reasons for *Z. noltii* decline, which was reduced from 15 ha in 1986 to 0.02 ha in 1997 (Fig. 11.1) (Dolbeth et al. 2007, Cardoso et al. 2008a).

11.2.3 Management Measures to Control Eutrophication

In late 1998, experimental mitigation measures were taken in order to control eutrophication process and restore the original seagrass beds of the Mondego's estuary south arm (Lillebø et al. 2005, Dolbeth et al. 2007, Cardoso et al. 2008a). These included the reduction of nutrient loading and the water residence time (from 5–7 days to 1 day) through:

- reduction of Pranto River sluice opening (most of the nutrient-enriched freshwater was then diverted through the north arm by another sluice located more upstream) and
- improving the hydraulic regime, by enlarging the upstream connection between the two arms and allowing water to flow from the north arm at high-tide conditions (Lillebø et al. 2005, Cardoso et al. 2008a).

11.3 Materials and Methods

11.3.1 Sampling Programme and Laboratory Procedures

This study integrates the information taken from the south arm of the estuary from 1993 to 2002. Sampling occurred fortnightly from February 1993 to June 1994 and monthly during the rest of the study period, at low tide in two contrasting sites regarding an eutrophication gradient: (a) a non-eutrophic *Z. noltii* bed – *Zostera* area and (b) a sand flat where macroalgal blooms occurred – eutrophic area (Fig. 11.1). On each sampling occasion, 6–10 cores (141 cm^2) were taken to a depth of 20 cm for the study of the macrobenthic community and evaluation of the flora (algae and macrophyte) dynamics. Samples were washed in 500 μm mesh sieve bags. In the laboratory, the biological material was separated and preserved in a 4% buffered formalin solution. Plant material was sorted and separated into green algae and *Z. noltii.* For both faunal and plant material the ash-free dry weight (AFDW) was assessed, after combustion for 8 h at 450°C. In the same time, salinity was measured in the intertidal pools, while water samples were collected for analysis of dissolved inorganic nitrogen and phosphorus. At the laboratory, samples were immediately filtered (Whatman GF/F glass-fibre filter) and stored frozen, until analysis following standard methods described in Limnologisk Metodik (1992) for ammonia (NH_3–N) and phosphate ($PO_4{}^-$P) and in Strickland and Parsons (1972) for nitrate ($NO_3{}^-$N), and nitrite ($NO_2{}^-$N).

11.3.2 Macrobenthic Feeding Guild Assignments

The feeding guild composition of macrobenthic assemblages was also analysed. Each *taxon* was assigned to a feeding guild according to its food type. Feeding guilds used in this study were carnivores (C), herbivores (H), ominivores (O) and detritivores or deposit feeders (D), with this last group divided into subsurface-deposit feeders (SsDF), surface-deposit feeders (SDF) and suspension feeders (SuF), according to the following literature: Gaston and Nasci (1988, 1995), Sprung (1994), Oug et al. (1998), Mancinelli et al. (1998). Preliminary analysis included the snail *Hydrobia ulvae*, but it was also decided to analyse trophic structure omitting this species, since it occasionally occurred in very high numbers and its inclusion masked changes in other species.

11.3.3 Secondary Production

The secondary production of the macrobenthic community was evaluated, as described in Dolbeth et al. (2007). The methods used were cohort increment summation method (see below) for the dominant species, Brey (2001) method version 4-04 (worksheet provided in Brey 2001, www.awi-bremerhaven.de/Benthic/Ecosystem/FoodWeb/Handbook/main.htm) for other representative species and for the species with lower densities and biomasses, production was estimated by summing the increases in biomasses from one sampling date to the other. For more details, see Dolbeth et al. (2007). Each species production was then cumulated into community production and also analysed per feeding guild. Additionally, six intertidal benthic species, usually found in estuarine systems, were studied in more detail: *H. ulvae* (Gastropoda), *Cyathura carinata* (Isopoda), *Scrobicularia plana* (Bivalvia), *Hediste diversicolor, Capitella capitata and Alkmaria romijni* (Polychaeta). For *H. ulvae* and *C. carinata*, all individuals were measured and production was estimated by the increment summation method, after definition of cohorts through size-frequency distribution analysis of successive sampling dates (described in Ferreira et al. 2007, Cardoso et al. 2008b), according to:

$$P_{cn} = \sum_{t=0}^{T-1} \left(\frac{N_t + N_{t+1}}{2}\right) \times (\overline{w}_{t+1} - \overline{w}_t) \text{ and } P = \sum_{n=1}^{N} P_{cn}$$

where P_{cn} is the growth production of cohort n, N is the density (ind m^{-2}), $\overline{w}$ is the mean individual weight (g WW m^{-2}) and t and t+1, consecutive sampling dates. Population production estimates correspond to the sum of P_{cn} (each cohort production).

For the other species, annual production was computed using the empirical method of Brey (2001) version 4-04, without computing depth, following the recommendations of Dolbeth et al. (2005).

For the species whose production was computed by cohort increment summation method, the fortnightly production dynamics during the bloom will also be presented; for remaining species, whose production was assessed by an empirical method, the fortnightly biomass dynamics will be used.

11.4 Results

11.4.1 Climate

In the Mondego estuary there was a clear seasonal pattern of rainfall over the 10-year period, with the highest precipitation values in the winter (Fig. 11.2a).

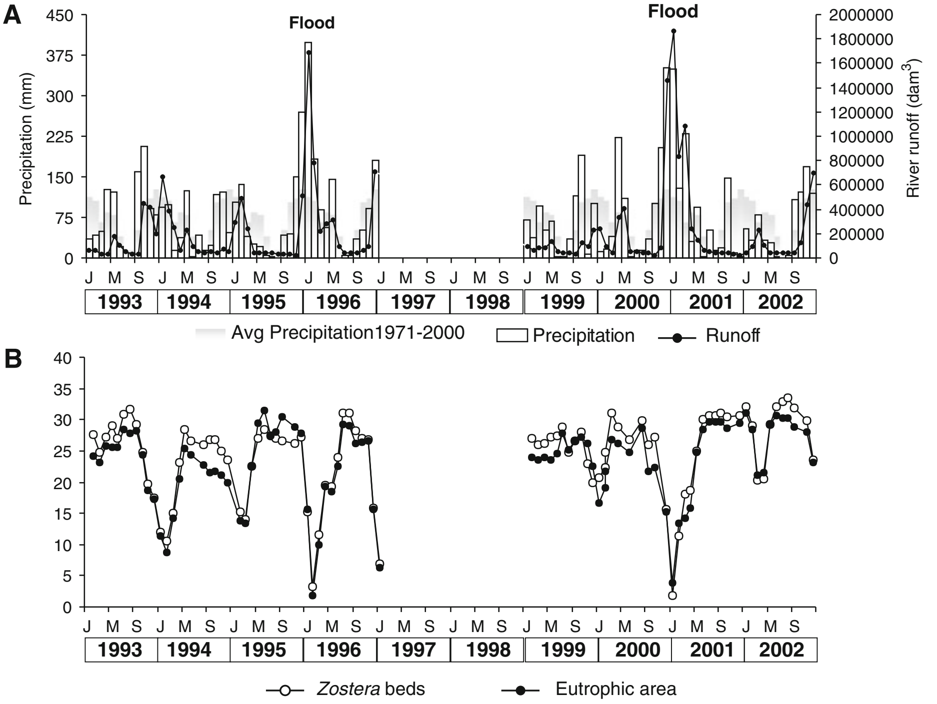


Fig. 11.2 Long-term variation in the (**a**) measured precipitation during the study period and mean precipitation for central Portugal during the period of 1971–2000 and (**b**) mean salinity for both study areas

Yet, taking into consideration the mean precipitation regime for central Portugal during the period of 1940–1997 (winter: 418 mm, spring: 265 mm, summer: 62 mm, autumn: 285 mm, INAG – http://snirh.inag.pt) some above-mean precipitation periods were evident (Fig. 11.2a). The hydrological years of 1993/94 (autumn: 593 mm) and 1995/96 (winter: 670 mm) were atypical, recording floods, and 2000/01 was even more atypical, with severe flooding occurring (winter: 767 mm) (INAG – http://snirh.inag.pt).

The seasonal pattern of rainfall and flooding reflected in the monthly and inter-annual variation of salinity in the south arm. During periods of intense rainfall, salinity declined severally (Fig. 11.2b), occasionally reaching <5 (Feb 96, Jan 97 and Jan 01). During these times of high rainfall there was an extensive opening of the Pranto River sluice, further contributing to the salinity decline (Lillebø et al. 2005).

11.4.2 Nutrient Dynamics

Regarding nutrient concentrations, there are two different scenarios (before and after the application of the mitigation measures) all over the study period. Concerning the dissolved inorganic nitrogen, there was a strong decline after the implementation of the mitigation measures, at both study sites (Fig. 11.3a). On the other hand, for the dissolved inorganic phosphorus, the pattern of variation was the opposite, with an increment after the management, especially in the eutrophic area (Fig. 11.3b). The *N*/*P* ratio showed a decline after the implementation of the management measures for both study sites (Fig. 11.3c).

11.4.3 Primary Producers

In the seagrass beds, *Z. noltii* showed an abrupt decline in total biomass during the pre-management period (Fig. 11.4a, $R^2 = 0.84$). After 1998, there seems to be a gradual recovery of its total biomass (Fig. 11.4a, $R^2 = 0.60$), reaching in 2002 similar biomass values to the ones registered in 1994 (Fig. 11.4a). The *Z. noltii* total biomass in 2002 corresponded to almost 50% of 1993 biomass. The green macroalgae *Ulva* spp. were more abundant in the eutrophic area, especially during the pre-management period (1993–1995). In the post-mitigation period, the biomass of green macroalgae tended to decrease, however, a small

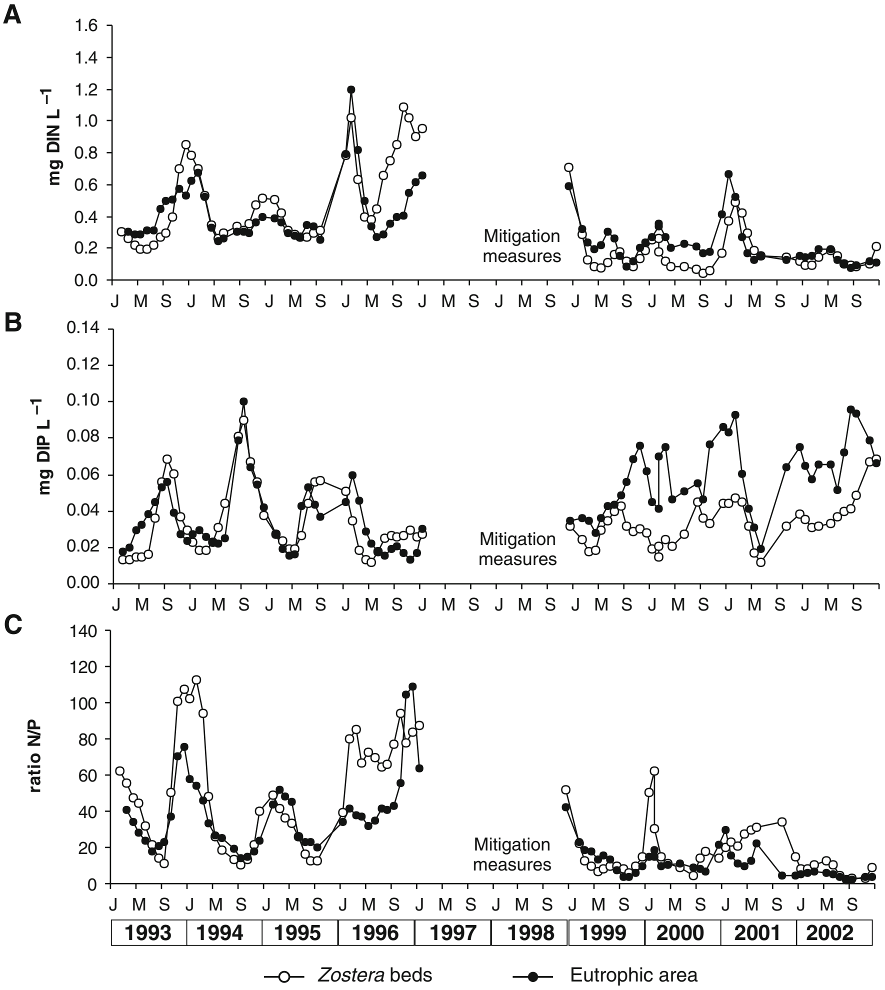


Fig. 11.3 Long-term variation in the nutrient concentrations: (**a**) dissolved inorganic nitrogen (DIN), (**b**) dissolved inorganic phosphorous (DIP); and (**c**) N/P atomic ratio, for both study areas

increase was recorded in the spring of 2000 (Fig. 11.4b). In the opposite, the red macroalgal *Gracilaria* sp. was more abundant in the *Zostera* beds and tended to increase all over the study period. In the eutrophic area, the biomass of *Gracilaria* sp. was vestigial (Fig. 11.4c).

11.4.4 Macrofauna Community General Trends

11.4.4.1 Changes in Diversity

Eighty different *taxa* were recorded over the 10-year period. The seagrass beds generally supported more species than the eutrophic area, with this difference more pronounced in the pre-mitigation period (Fig. 11.5a). Nevertheless, evenness was higher in the eutrophic area (Fig. 11.5b), mainly due to the dominance of *H. ulvae* in the *Z. noltii* beds, as detailed by Cardoso et al. (2008a).

For both areas, there was a clear decline in the number of species during the pre-mitigation period. Following the introduction of the management plan in 1998, there was a tendency for a species richness increment in both study areas. However in 2000/01, during the high rainfall event, there was a decline in species richness. After this extreme event, species richness only started to recover again in 2002 for both areas. Evenness recovery was more pronounced in the

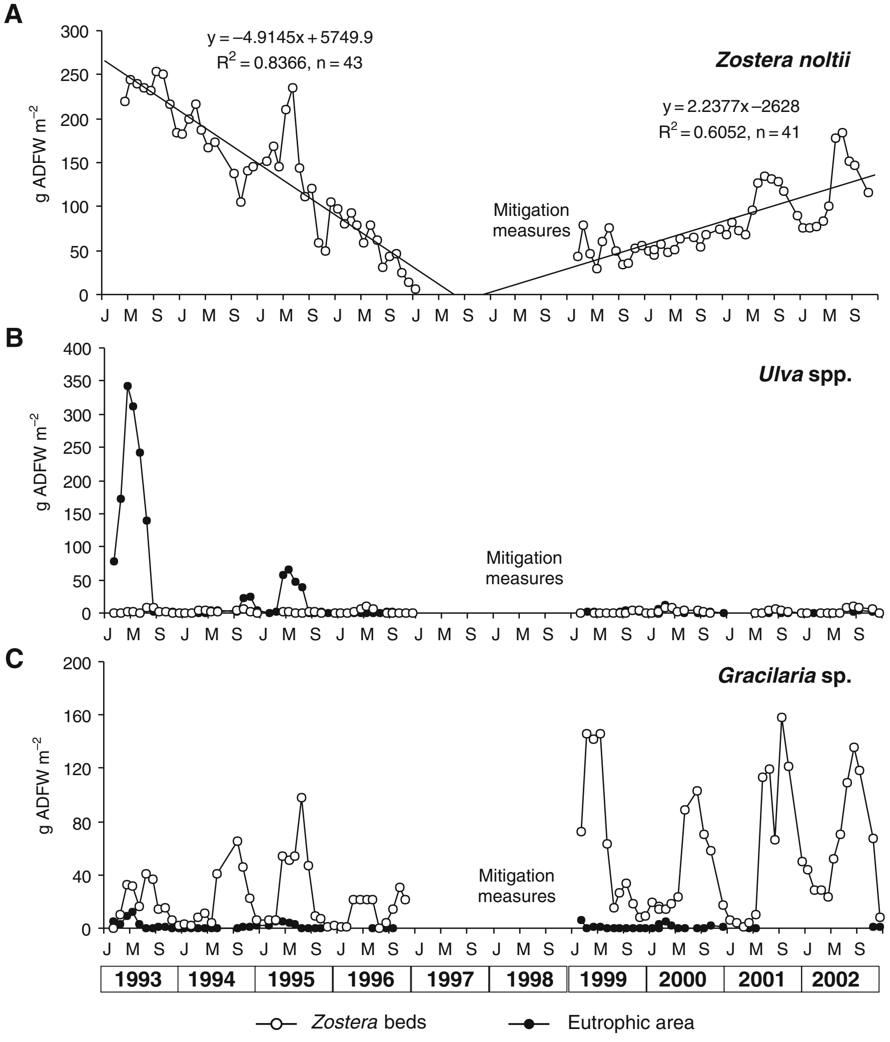


Fig. 11.4 Long-term variation of the seagrass and macroalgal biomass for both study areas: (**a**) *Z. noltii*, (**b**) *Ulva* spp. and (**c**) *Gracilaria* sp

recent years in the eutrophic area (Fig. 11.5b), while in the *Zostera* beds it remained quite stable all over the time.

11.4.4.2 Changes in Density, Biomass and Production

A similar tendency to the diversity changes occurred with density, biomass and annual production. In general, these parameters were higher in the *Zostera* beds than in the eutrophic area (Fig. 11.5c, d, Table 11.1), except in 1999, when higher annual production was obtained in the eutrophic area. Also, there was a general decline in density, biomass and production when comparing pre-mitigation and post-mitigation periods (Fig. 11.5c, d, Table 11.1).

Mean densities showed a clear seasonal pattern in the *Zostera* beds, with higher values in spring/summer throughout the study period, with the highest value observed in the spring of 1994 (Fig. 11.5c). In the eutrophic area, there was a general decline in total density all over the time (Fig. 11.5c), while total biomass increased considerably in the post-management period (Fig. 11.5d). Contrarily, in the *Zostera* beds, mean biomass declined in the beginning of post-management period. Afterwards it showed an increasing pattern until the occurrence of the flood in 2000/01 (Fig. 11.5d). In the eutrophic area alone,

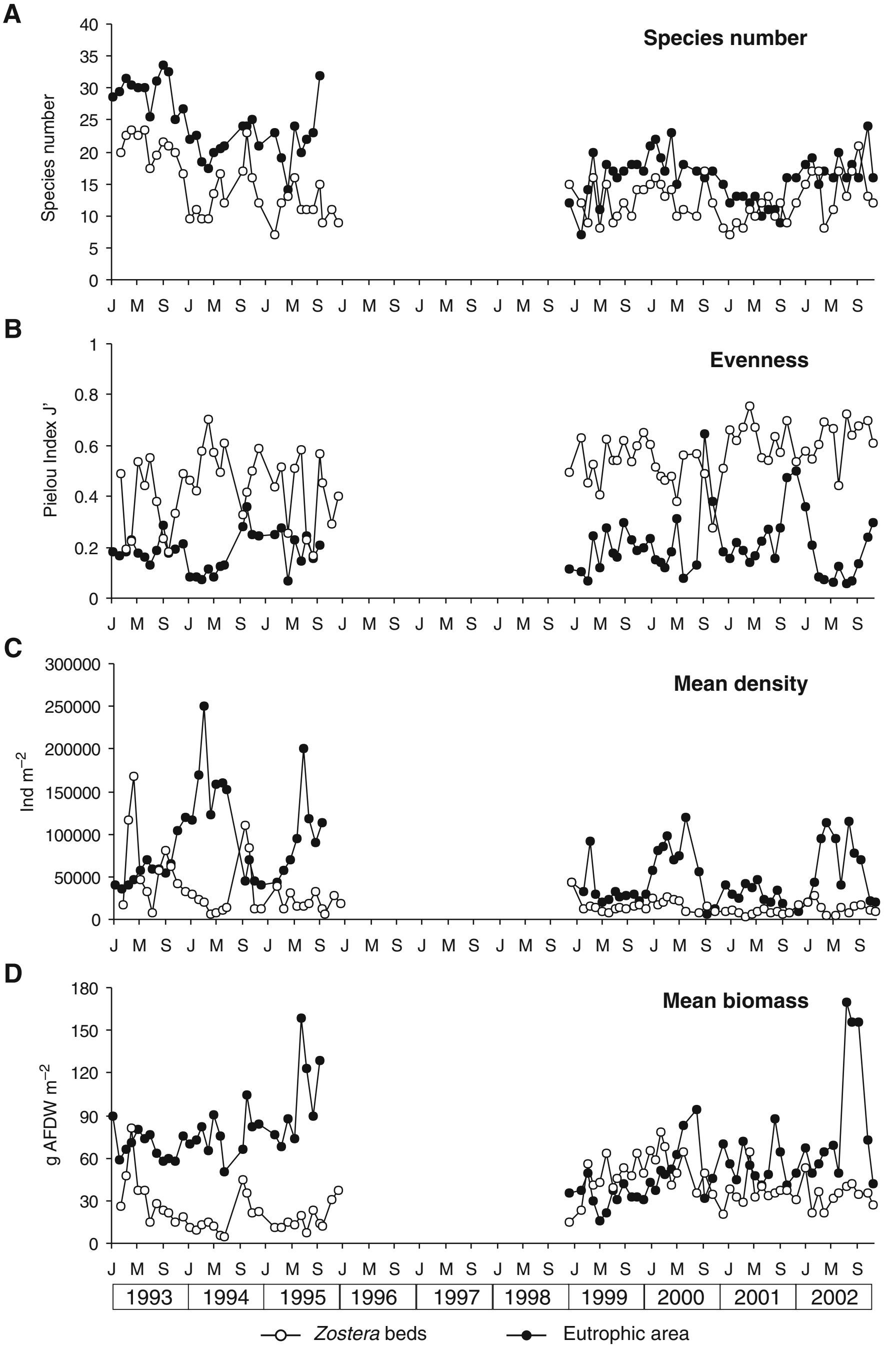


Fig. 11.5 Monthly long-term variation of: (**a**) species richness; (**b**) evenness; (**c**) mean biomass; and (**d**) mean density, for both study areas

mean density and biomass were higher during spring 1993 and spring 1995 (Fig. 11.5c, d), matching with the periods of higher macroalgal biomass (Fig. 11.2d). The highest value of annual production was obtained in 1993, the bloom year (Table 11.1). In the post-mitigation period, both mean biomass and annual production (P) were within similar values in all years, appearing to maintain a relatively stability, only with

Table 11.1 Community annual production (*P*) and mean annual biomass ($\bar{B}$) for *Zostera* beds and eutrophic area, during the 10-year study period

	Zostera beds		Eutrophic area	
	P (g AFDW m^{-2} y^{-1})	$\bar{B}$ (g AFDW m^{-2})	P (g AFDW m^{-2} y^{-1})	(g AFDW m^{-2})
1993	148.0	75.1	89.3	31.4
1994	222.9	84.6	45.8	15.8
1995	137.9	105.8	32.8	19.7
1999	45.9	34.9	57.0	45.0
2000	121.3	60.1	60.6	62.3
2001	88.4	54.4	51.7	37.8
2002	199.5	94.8	58.2	35.0

a slightly decrease in 2001, following the highest flood event.

The greatest annual production was obtained for the *Zostera* beds, in 1994 and 2002 (Table 11.1), matching the years when the area covered with *Z. noltii* was practically the same (1.4% of the intertidal area, Fig. 11.1).

11.4.4.3 Feeding Guilds Relative Composition

The analysis of the community feeding guilds relative composition including *H. ulvae* showed that for the *Zostera* beds the most representative groups, both in terms of density and production, were detritivores and herbivores (Fig. 11.6a). This result was due to the dominance of *H. ulvae* (considered as both as detritivore and herbivore) in the community in this area, with other groups comprising only a small fraction. For the eutrophic area, the herbivores percentage was not so high, with most of the community abundance and production represented by detritivores (Fig. 11.6b). Yet, it is worth to notice that about 25% of the community production was represented by carnivores, and that after the mitigation measures most of the detritivores production in the community increased (Fig. 11.6b). Analysing in detail the detritivore assemblage, surface-deposit feeders (SDF) were the dominant group in both study areas in terms of density, while the relative production contribution of SDF for

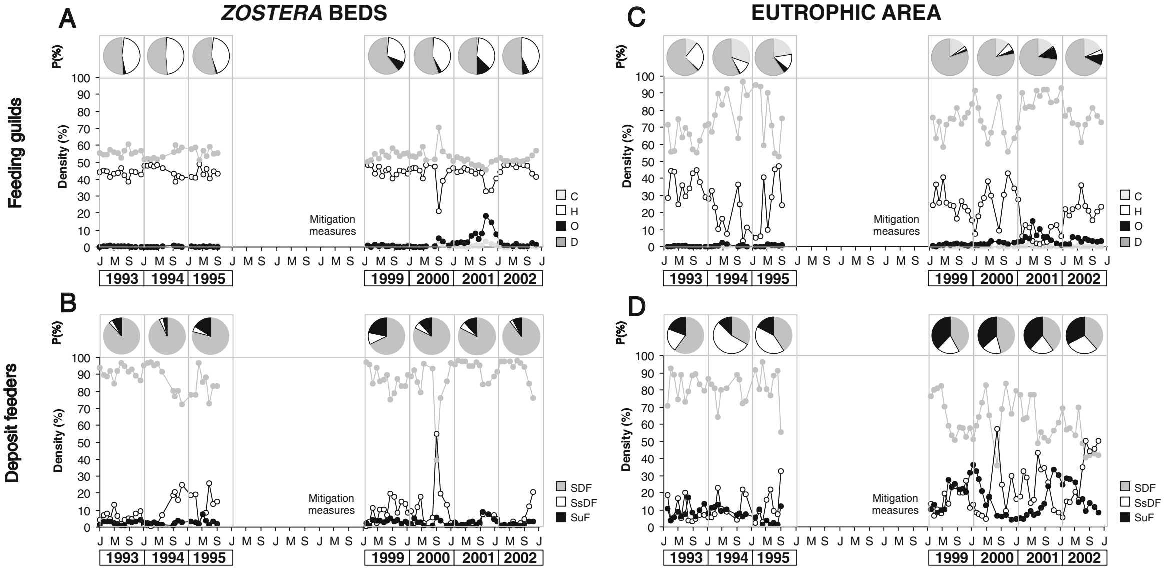


Fig. 11.6 Long-term variation of feeding guilds relative composition regarding density and production in the presence of *Hydrobia ulvae* for the (**a**, **b**) *Zostera* beds and (**c**, **d**) eutrophic area of the Mondego estuary, with indication of the detailed relative composition of the detritivores/deposit feeders (**c**, **d**). Legend: C, carnivores; H, herbivores; O, omnivores; D, detritivores/deposit feeders; SDF, surface-deposit feeders; SsDF, subsurface-deposit feeders; SuF, suspension feeders

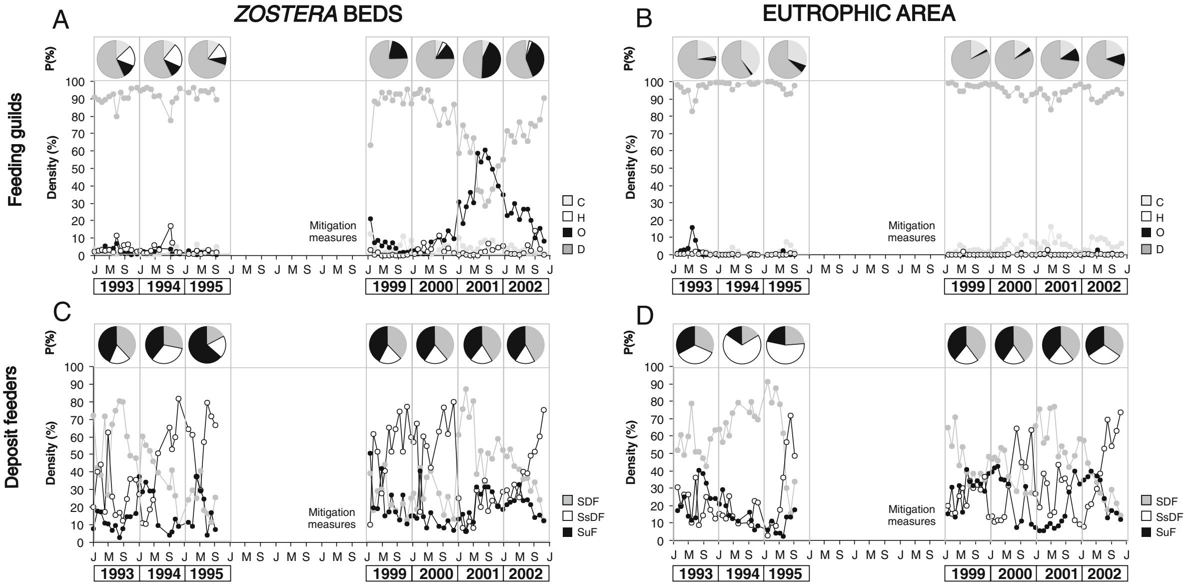


Fig. 11.7 Long-term variation of feeding guilds relative composition regarding density and production in the absence of *H. ulvae* for the (**a**, **b**) *Zostera* beds and (**c**, **d**) eutrophic area of the Mondego estuary, with indication of the detailed relative composition of the detritivores/deposit feeders (**c**, **d**). Legend: C, carnivores; H, herbivores; O, omnivores; D, detritivores/deposit feeders; SDF, surface-deposit feeders; SsDF, subsurface-deposit feeders; SuF, suspension feeders

the *Zostera* beds was considerably higher than for the eutrophic area (Fig. 11.6c, d). In addition, in the *Zostera* beds after the flood peak (December 2000) there was a large decline in subsurface-deposit feeders (SsDF) abundance and an increase in SDF (Fig. 11.6c). In the eutrophic area, there was a greater variability in the deposit feeders abundance composition from 1998 to 2002 compared to 1993 to 1995 period, which was mostly dominated by SDF abundance. This was evident for suspension feeders (SuF) abundance, which decreased by the large floods of 2000/01 (Fig. 11.6d). Yet an analysis of the production composition reveals a different scenario: while in the pre-mitigation period, the production had a higher percentage of SDF and SsDF, in the post-mitigation a higher percentage of SDF and SuF production contributing to the community was observed (Fig. 11.6c, d).

The analyses including *H. ulvae* were difficult to interpret, due to the masking effect of the large abundance of this species. Thus, an analysis of the feeding guilds composition was also performed for the community excluding *H. ulvae*. These analyses showed that the macrofaunal communities continue to be dominated by detritivores, which together accounted for more than 90% of the total macrobenthic abundance in both areas (Fig. 11.7a, b), with exception to the *Zostera* beds in 2001. In this year, it was observed a marked decline in the detritivores percentage, followed by a large increase in abundance of omnivores (e.g. *H. diversicolor*) (Fig. 11.7a). When analysing the community production, higher percentages of the other feeding guilds were observed, yet the highest percentage was still due to detritivores production in both areas (Fig. 11.7a, b).

Within the detritivore assemblages in the *Zostera* beds, SDF abundance declined from 1993 to 1999, following the decline of the seagrass *Z. noltii*, and started to increase again in 2001/2002 (Fig. 11.7c). In contrast, SsDF (mainly small polychaetes, such as *C. capitata*) showed the opposite pattern, dominating the community abundance in 1994–2000 and declining abruptly after the floods of 2000/2001 to start to increase again in 2002 (Fig. 11.7c). Yet, again the production analysis showed a slightly different picture, since the community was mostly dominated by SuF production from 1994 to 1995 and showed a similar pattern in the remaining years, being dominated by SuF and SDF production (Fig. 11.7c).

In the eutrophic area, in 1993 and 1994 there was an increase of the SDF abundance, accompanied by a gradual decrease of SsDF and SuF abundance. However, the production was not dominated by SDF (Fig. 11.7d). Instead, in 1993 similar percentages were obtained by the three deposit feeding guilds, while in 1994 the production was dominated by SsDF (Fig. 11.7d). By the end of the pre-mitigation period, in late autumn 1995, SsDF abundance increased, with this increasing tendency maintained over the 10-year period, except for 2001 (Fig. 11.7d). A similar trend was observed with SuF abundance, yet with lower percentage and with a marked reduction during the floods of 2000/2001, recovering over the following year (Fig. 11.7d). In the post-mitigation period, the relative production of the deposit feeders was similar, dominated by SuF and SDF, even in 2000/01, when the abundance of SuF decreased considerably (Fig. 11.7a).

11.4.5 Species-Specific Responses

11.4.5.1 *Hydrobia ulvae* (Gastropoda)

H. ulvae was the dominant species within the *Z. noltii* beds, where it attained the highest values of density, biomass and production. From 1993 to 1995, *H. ulvae* biomass was relatively constant (range 50–100 g m^{-2}), but had two density peaks in 1994 and 1995 (Fig. 11.8a). Following the decline of *Z. noltii*, biomass decreased considerably by approximately an order of magnitude (Fig. 11.8a), which implies a loss of a large number of reproductive adults during this time. *H. ulvae* population appeared to start to recover, until the fall/winter of 2000/01, when the period of prolonged and heavy rainfall was associated with a decline in both density and biomass (Fig. 11.8a). The population seemed to recovery again in 2002 (Fig 11.8a). The growth production followed the density and biomass pattern showing the highest values during the pre-mitigation period, especially in 1994, and in 2002 (Fig 11.8a). This species was highly productive within the community and in some of the years its production alone represented a major part of the total *Zostera* beds annual community production (Table 11.1).

In the eutrophic area, the pattern of change was completely different. Density and biomass of *H. ulvae* declined throughout the study period, with no obvious indication of recovery in the post-mitigation period (Fig 11.8b). Production reached the highest value in 1993, when the macroalgal bloom occurred. In fact, a closer view on the short-term dynamics of algae biomass and *H. ulvae* fortnightly production enabled to detect substantial increases in the production during the bloom occurrence (Fig. 11.8c). This increase in production occurred less than 1 month after the increase of algal biomass (when algae biomass attained ±300 g AFDW m^{-2}; Fig. 11.8c). In fact, 1 month production (April production) corresponded to 75% of the whole 1993 annual production. Afterwards, with the first signs of the macroalgae decline and following crash, the production also decreased considerably (Fig. 11.8c).

During the post-mitigation period, density, biomass and annual production were considerably lower than in the pre-mitigation period, with the highest increases in 2000 and 2002, coincident with the appearance of greater biomasses of green macroalgae (Fig 11.8b).

The $P/\bar{B}$ ratio of *H. ulvae* was much higher in the eutrophic area (between 2.7 and 4.8), where the population is mainly composed of juveniles, than in the *Z. noltii* beds (between 1.3 and 3.0), where it presents a more structured population (Table 11.2).

11.4.5.2 *Cyathura carinata* (Isopoda)

In the *Zostera* beds, *C. carinata* population was more unstable, showing a great variability all over the time than in the eutrophic area (Fig. 11.9a). At the beginning of 1993, *C. carinata* was absent from the *Z. noltii* beds; however, its population sprouted until the end of 1994 and 1995, reaching to more than 700 ind m^{-2} and with increases in biomass and annual production (Fig. 11.9a). Afterwards, the population decreased considerably by 1999, yet seemed to recover in the following 2000 and 2001, until a new decline by the end of 2002, being on the verge of disappearing from this area (Fig. 11.9a). The annual production of this species was much lower in the *Zostera* beds, reaching the highest value in 1994 (Fig. 11.9a). *C. carinata* presented a stable and consistent population in the eutrophic area all over the study period, exhibiting a characteristic annual pattern of variation. Density increased during summer, achieving maximum values in autumn/winter and then declined until late spring (Fig. 11.9b). Annual production was within similar values during the study period, with exception to a slightly higher peak in 1994 (Fig. 11.9b). During the

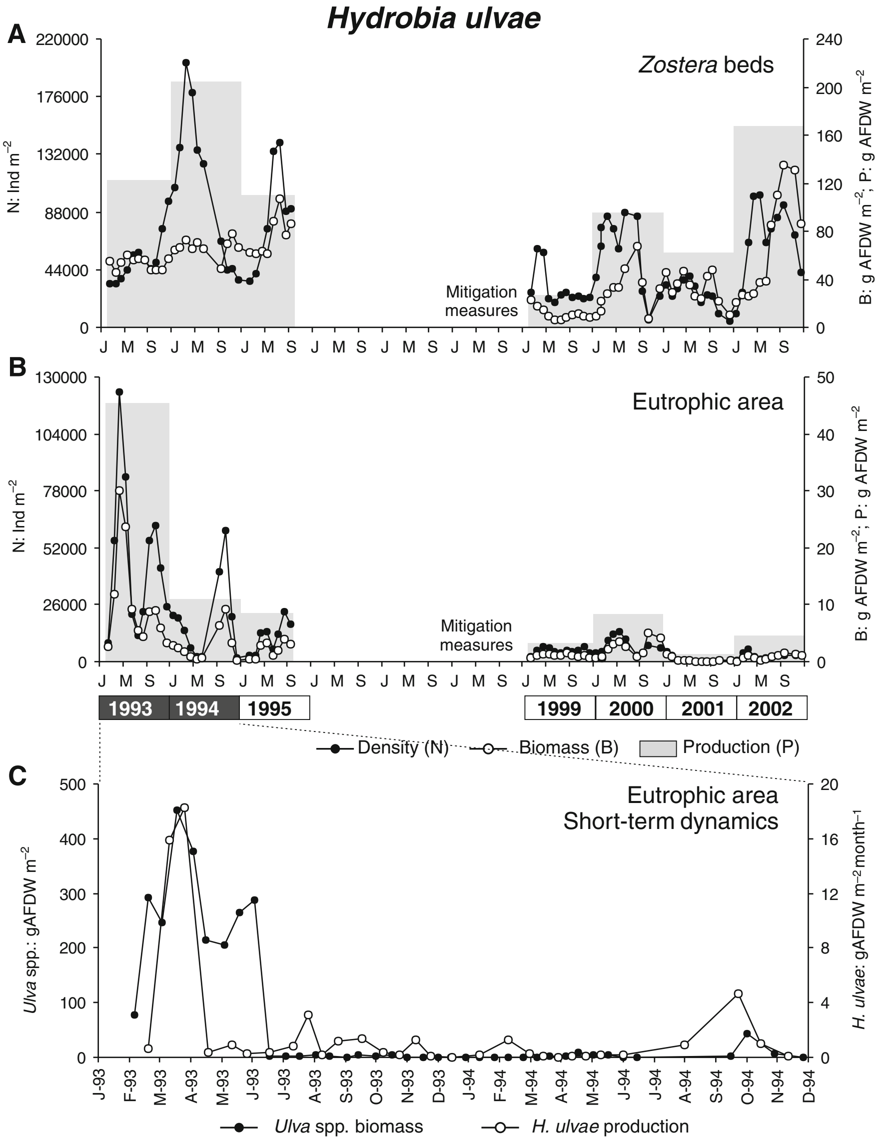


Fig. 11.8 Long-term variation of the monthly mean density, mean biomass and annual production of *H. ulvae* for (**a**) the *Zostera* beds and (**b**) eutrophic area, with indication of (**c**) the detailed short-term dynamics of *Ulva* spp. biomass and *H. ulvae* fortnightly production during the macroalgal bloom (1993) and the following year (1994)

macroalgal bloom, it was not clear whether *C. carinata* production increased due to the presence of the algal cover. As soon as the algal biomass started to increase in initial March (after ±80 g AFDW m^{-2} of algae), C. carinata production in the following sampling was high (less than 15 days) (Fig. 11.9c). Yet, during the highest algal biomass, the production decreased considerably and maintained low in remaining 1993. In 1994, with almost no algae, higher production peaks were obtained in March, similar to the previous year, and in August and November (Fig. 11.9c). The $P/\bar{B}$ ratio of *C. carinata* was around similar values for both areas and slightly higher in the pre-mitigation period than in the post-mitigation one (Table 11.2).

11.4.5.3 *Scrobicularia plana* (Bivalvia)

S. plana density, biomass and annual production were higher in the eutrophic area than in the *Zostera* beds (Fig. 11.10a, b). In both sites, an important increase in all parameters was observed after the introduction of mitigation measures (Fig. 11.10a, b, Table 11.1).

Table 11.2 $P/\bar{B}$ ratios ($P/\bar{B}$: y^{-1}) estimates for the main species for *Zostera* beds and eutrophic area, during the 10-year-study period

		1993	1994	1995	1999	2000	2001	2002
Zostera beds	*Hydrobia ulvae*	2.1	2.8	1.3	2.4	3.0	2.0	2.5
	Scrobicularia plana	0.9	1.8	1.3	0.7	0.8	0.6	0.8
	Cyathura carinata	4.6	2.9	3.0	2.4	2.0	2.2	2.2
	Hediste diversicolor	2.0	1.3	1.3	1.1	1.4	1.7	1.4
	Capitella capitata	6.9	9.2	8.9	5.8	7.0	8.1	5.5
	Alkmaria romijni	8.9	10.1	9.2	11.3	9.8	6.2	5.7
Eutrophic area	*H. ulvae*	4.8	4.5	2.4	3.1	2.9	4.1	2.7
	C. carinata	1.6	1.6	1.0	0.6	1.0	0.9	1.0
	S. plana	2.9	3.4	1.7	1.9	1.6	1.5	2.0
	H. diversicolor	2.0	1.8	1.5	1.1	1.4	1.9	1.9
	C. capitata	9.5	10.9	10.2	7.6	7.0	8.4	10.6
	A. romijni	13.4	11.8	10.3	9.0	8.5	10.6	9.5

For this bivalve, the biomass increase was more relevant than the density one (biomass increment: *Z. noltii* bed – 500%, eutrophic area – 250%) (Fig. 11.10a, b), which translated into lower $P/\bar{B}$ ratios in the post-mitigation period (Table 11.2). Like *H. ulvae*, after the extreme flood in 2000/01, there was a decrease in density, biomass and annual production, which increased again in the following year (Fig. 11.10a, b). During the macroalgal bloom, there were no specific increases in *S. plana* biomass, since the fortnightly variation pattern maintained similar in both 1993 and 1994, showing only a slight tendency to decrease with time (Fig. 11.10c).

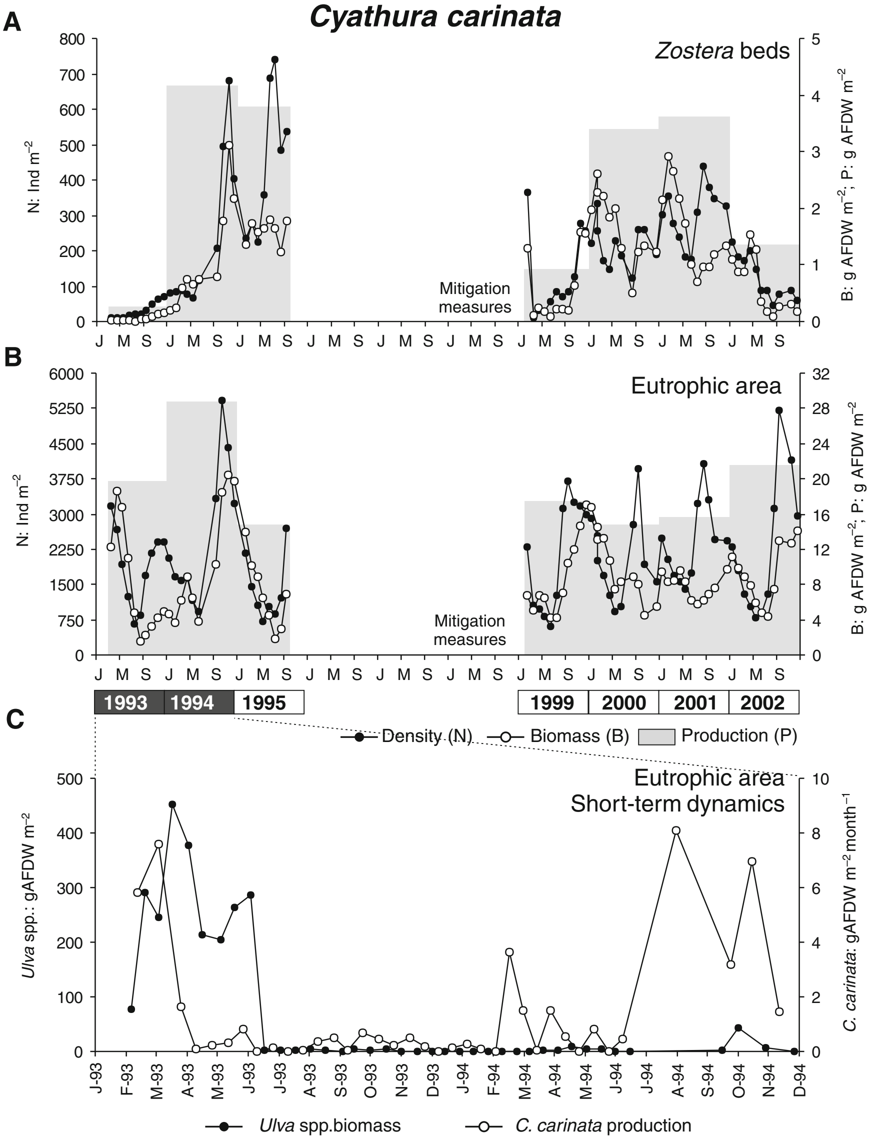
11.4.5.4 *Hediste diversicolor* (Polychaeta)

In the pre-mitigation period, *H. diversicolor* had slightly higher density, biomass and annual production in the eutrophic area than in the *Zostera* beds (Fig. 11.11a, b). After the mitigation measures, it presented a huge increment for both areas, especially for the *Zostera* beds. In fact, in this period, density, biomass and annual production were almost the double in the *Zostera* beds in comparison to the eutrophic area (Fig. 11.11a, b). During the macroalgal bloom, no specific increases in *H. diversicolor* biomass were observed; on the contrary, the biomass was nearly null and only increased after the bloom in the remaining period (Fig. 11.11c). The $P/\bar{B}$ ratios varied within similar values for the all study period, being slightly higher in the eutrophic area (Table 11.2).

11.4.5.5 *Alkmaria romijni* and *Capitella capitata* (Polychaeta)

A. romijni and *C. capitata*, together with *H. diversicolor* constitute the most abundant polychaete species in the Mondego estuary. Both had higher density, biomass and annual production in the eutrophic area (Figs. 11.12, 11.13), especially *A. romijni*, whose difference between areas is huge (Fig. 11.13). Over the study period, there was an important decline in all parameters in the post-mitigation period for both species and areas. This decline was quite high in 2001 for both species, after the extreme flood, with a slight increase for *C. capitata* in the following year 2002 (Fig. 11.12a, b). For *C. capitata*, the highest annual production was obtained in 1993 in the eutrophic area, during the macroalgal bloom (Fig. 11.12), while for *A. romijni*, the highest values were obtained in both 1993 and 1994, with similar annual production values (Fig. 11.13). A closer view on the macroalgal bloom short-term dynamics enables to detect *C. capitata* biomass increases about 1 month after the highest

Fig. 11.9 Long-term variation of the monthly mean density, mean biomass and annual production of *Cyathura carinata* for (**a**) the *Zostera* beds and (**b**) eutrophic area, with indication of (**c**) the detailed short-term dynamics of *Ulva* spp. biomass and *C. carinata* fortnightly production during the macroalgal bloom (1993) and the following year (1994)



increases of the algal biomass (Fig. 11.12c). A considerable decrease in production was observed about 1 month after the first signs of the macroalgae decline (Fig. 11.12c), suggesting that the algal biomass somewhat influenced the polychaete biomass. Regarding *A. romijni*, the pattern was not as clear, since there seems to be an increase following the algal biomass increases, yet the biomass did not decrease with the algal crash, and there were other biomass peaks following 1993 and in 1994 (Fig. 11.13c). These species had the highest $P/\bar{B}$ ratio values of the all the six species studied, with most values varying among 7–11 (Table 11.2). In general, these were higher in the pre-mitigation period for both species and both areas and were slightly higher for *A. romijni* (Table 11.2).

11.5 Discussion

11.5.1 Eutrophication Effects

Eutrophication may be defined as "the process of changing the nutritional status of a given water body by increasing the nutrient resources" (Jørgensen and Richardson 1996). Due to this nutrient enrichment,

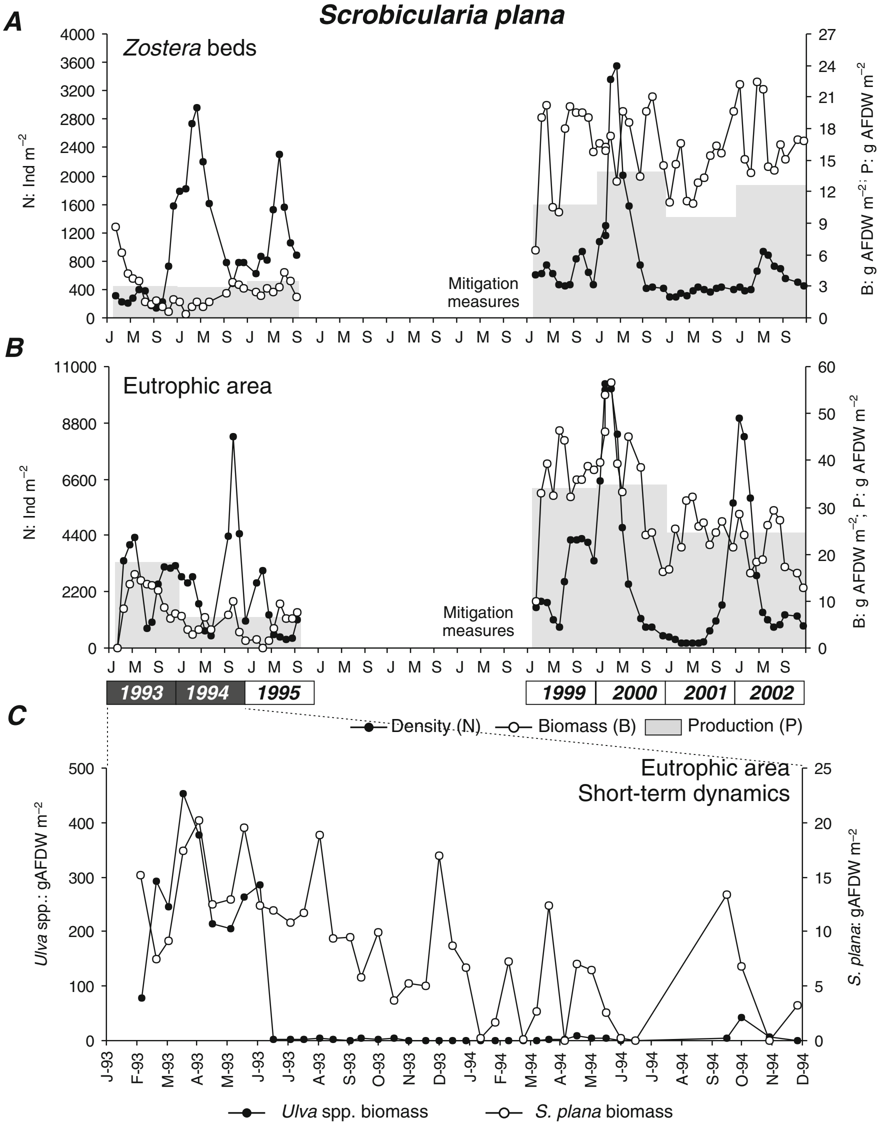


Fig. 11.10 Monthly long-term variation of the mean density, mean biomass and annual production of *Scrobicularia plana* for (**a**) the *Z. noltii* beds and (**b**) eutrophic area

most in the form of N and P, several impacts on the aquatic ecosystem may occur affecting primary and secondary production (Flindt et al. 1999, Dolbeth et al. 2003, Paerl 2006). The magnitude of the eutrophication effects depends on several aspects, from the hydrological characteristics of the system itself to climate (Cloern 2001, Paerl 2006, Rabalais et al. 2009). The Mondego estuary has a history of eutrophication, starting from the 1980s, and constitutes an important case study on this matter, since it has been monitored from 1993 to nowadays, and several hidden effects of eutrophication may only be revealed with long-term monitoring programmes. As discussed by Lillebø et al. (2005), nitrogen, in the form of ammonia, appears to be the limiting nutrient in the Mondego estuary, similar to other estuarine systems, and an increasing pattern of this nutrient was indeed observed in the pre-mitigation period. An important short-term effect of this nutrient enrichment, associated with high water residence time, was the development of macroalgal blooms, mainly *Ulva* spp., as also occurred in several other coastal ecosystems suffering from eutrophication (Raffaelli 1998, Prins et al. 1999, Feuerpfeil et al. 2004, Fox et al. 2009, Pravoni et al. 2008). Another consequence in the system was the replacement of *Z. noltii* beds by *Ulva* spp., which in turn affected the entire trophic

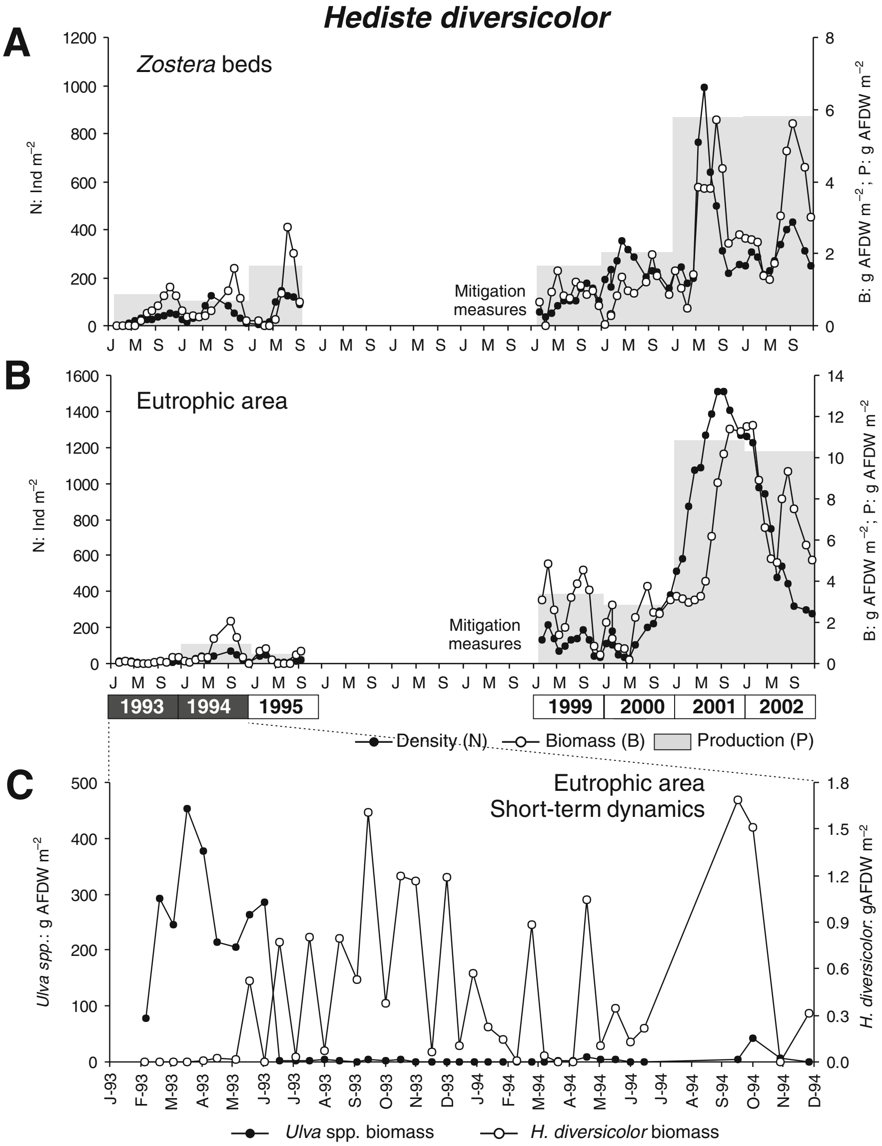


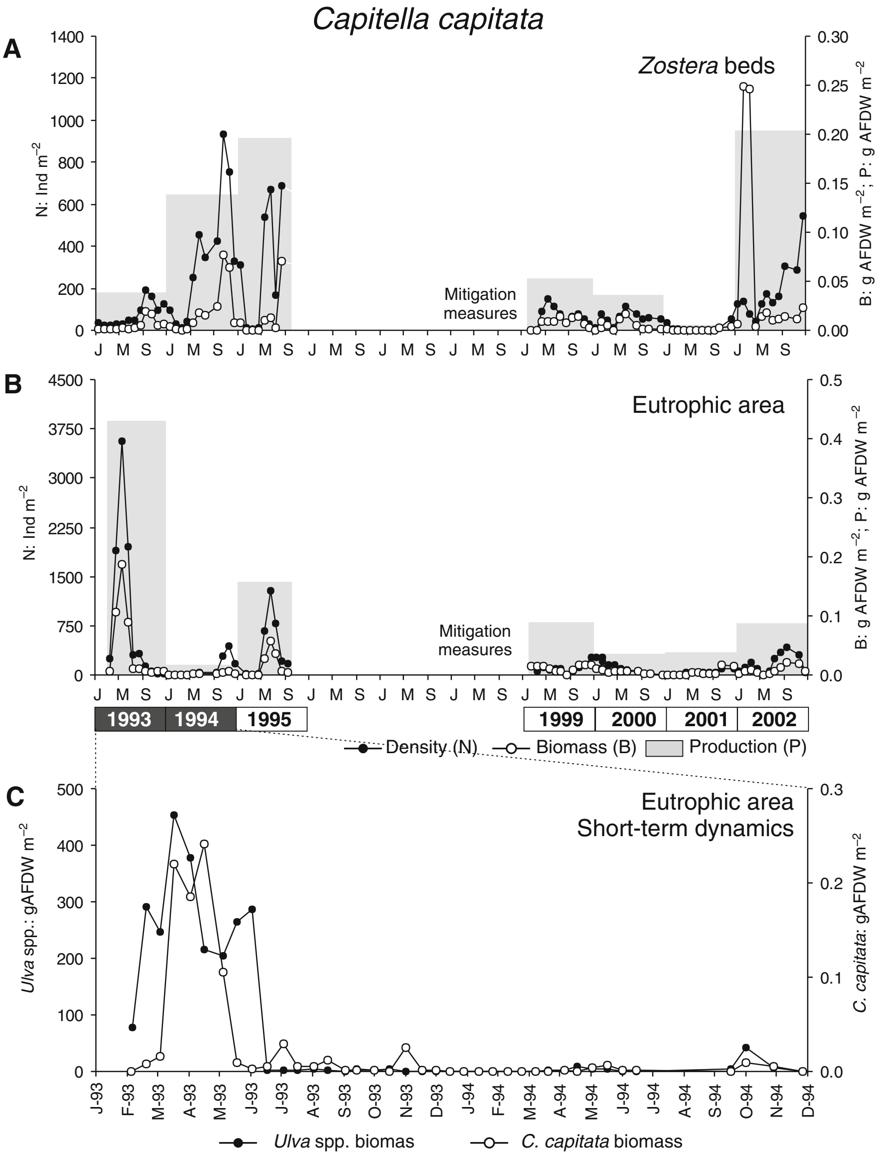
Fig. 11.11 Monthly long-term variation of the mean density, mean biomass and annual production of *Hediste diversicolor* for (**a**) the *Z. noltii* beds and (**b**) eutrophic area

structure of the associated benthic community. These effects have been reported worldwide, mainly in highly populated coastal areas, where the nutrient over enrichment has led to excessive production of organic matter in the form of algal blooms (fast-growing opportunistic algae such as epiphytic green macroalgae and/or phytoplankton), conducting to the gradual replacement of perennial benthic macrophytes (see review from Flindt et al. 1999, Cloern 2001), hypoxia and anoxia scenarios (Breitburg et al. 2009), in turn affecting the aquatic heterotrophic organisms depending on that primary production and living in the hostile generated environment (Oviatt et al. 1986, Raffaelli et al. 1998, Prins et al. 1999, Edgar and Barrett 2002, Dolbeth et al. 2003, Breitburg et al. 2009, Fox et al. 2009).

11.5.1.1 Macroalgal Bloom Dynamics in the Eutrophic Area

During the occurrence of the *Ulva* spp. bloom in the Mondego estuary in 1993, several species were able to take advantage of the extra food resources, habitat heterogeneity and protection against predation,

Fig. 11.12 Monthly long-term variation of the mean density, mean biomass and annual production of *Capitella capitata* for (**a**) the *Z. noltii* beds, and (**b**) eutrophic area



which the algae represent, as also discussed in other studies (Norkko et al. 2000, Dolbeth et al. 2003, 2007). In fact, mean annual density/biomass and production of the associated fauna from the eutrophic area, where the bloom took place, increased considerably. However, the response to the algal bloom is highly species specific (Dolbeth et al. 2003, Grilo et al. 2009), as also demonstrated experimentally (Cardoso et al. 2004). The link between changes in the primary and secondary production to the nutrient loadings in aquatic ecosystems is quite difficult, due to the interference of several other environmental factors (Cloern 2001, Edgar and Barrett 2002, Nixon and Buckley 2002). Yet, the analysis of the six species responses to the algal blooms enabled to set some hypotheses regarding which species could benefit from the algae. The gastropod *H. ulvae* and the polychaete *C. capitata* were the species that seemed to be influenced by the presence of the algal cover, with important positive changes in the density, biomass and production during the bloom year and with fastest response to the presence of algal cover (less than 1 month). These species are highly opportunist and seemed to optimize their population

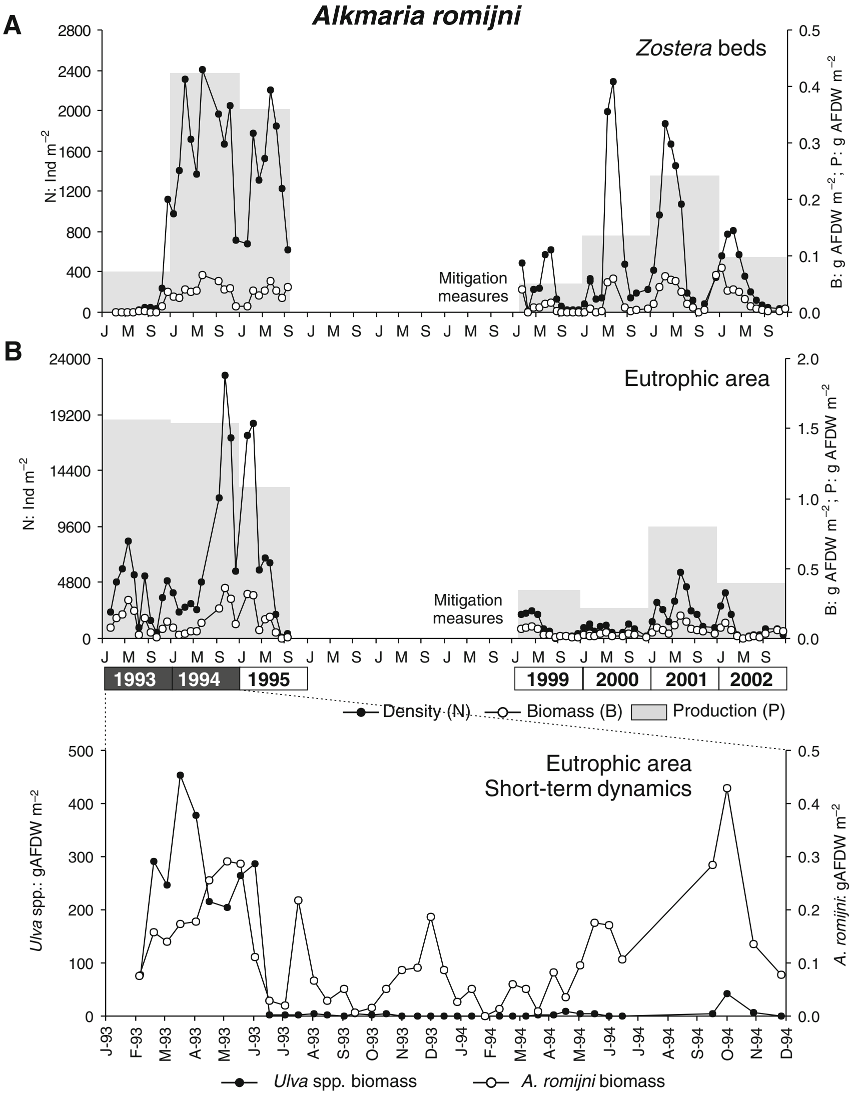


Fig. 11.13 Monthly long-term variation of the mean density, mean biomass and annual production of *Alkmaria romijni* for (**a**) the *Z. noltii* beds and (**b**) eutrophic area

growth at the eutrophic area, following the presence of the additional resource, as also observed by Fox et al. (2009). *H. ulvae* is a dominant species in the *Z. noltii* beds, where it presents a well-structured population (Cardoso et al. 2008b), with lower turnover ratios ($P/\bar{B}$ ratios), reflecting higher stability in this area. So, the eutrophic area, during the presence of the algal cover, probably worked as an alternative habitat, similar to the conclusions of Norkko et al. (2000) and Cardoso et al. (2004). *C. capitata* was generally more abundant in the bare sand flat eutrophic area. Yet, during the bloom, the increased habitat complexity and available organic matter probably worked as a driver for the observed increased biomass (reflected into higher annual biomass and production), emphasizing its *r*-strategist behaviour. Nevertheless, as also stated by Norkko et al. (2000), the effects of drift algal mats on infauna depend on the spatial and temporal extent of algal coverage. Once the algae grew beyond a certain threshold, the system collapsed into an algal crash, and both species suffered a strong decline in density, biomass and production, presenting low values in the following pre-mitigation period. This tendency was also observed for the community patterns, with declines in the community mean annual density, biomass and production in the following 1994 and 1995. These decreases were associated to the negative effects of macroalgal blooms and associated crash, which have been well documented. Among described effects are changes in sediment chemistry, nutrient release, rates of oxygen consumption and accumulation of metal sulfides and H_2S in sediment pore waters (Jorgensen and Richardson 1996, Cloern 2001), which in turn cause the invertebrate assemblages to collapse, as also reported elsewhere (Raffaelli et al. 1998), and ultimately affect the higher trophic levels depending on that invertebrate production (Dolbeth et al. 2008).

For the other studied species, the parallelism between algal growth and fauna-positive responses was not as clear or did not seem to occur at all. For instance, *S. plana, C. carinata* and *A. rominji* had increases in density and biomass during the bloom, yet the highest peaks in density and biomass were observed afterwards. So, the algae did not seem to have worked as an additional resource for those species populations development. *H. diversicolor* did not seem to be influenced by the presence of the algal cover at all. On the contrary, its biomass only increased after the bloom, although other studies have referred this species as being tolerant to hypoxia, anoxia and commonly found under algal mats (Pearson and Rosenberg 1978, Norkko and Bonsdorff 1996). In fact, during the bloom alone, *H. diversicolor* presented the lowest biomass values of the pre-mitigation period, which increased considerably after the bloom, suggesting that the algae did not act as a driver for the population development. For the Mondego estuary in particular, *H. diversicolor* was associated to a post-algae and recover phase (discussed below), rather than as an indicator of the organic matter enrichment, as also discussed by Cardoso et al. (2007).

The percentage of herbivores abundance and production at the eutrophic area was higher in the bloom year, in expected since the algae are a readily available food resource. However, it is worth to mention that most of this percentage is due to *H. ulvae* increase, which behaves both as herbivore and surface-deposit feeder. Also, a higher percentage of surface-deposit feeders was observed in 1993, and afterwards in the following pre-mitigation period, subsurface-deposit feeders dominate the community production. These differences are probably due to the fact that surface-deposit feeders prefer to feed on newly deposited organic matter, probably with highest amounts in 1993.

11.5.2 Differences Between Sites

The *Z. noltii* beds had in general higher species richness, density, biomass and production than the eutrophic area during the whole study period. These differences are in line in several studies elsewhere (e.g. Sprung 1994, Heck et al. 1995, Fox et al. 2009, Pranovi et al. 2008), comparing sediments covered with rooted macrophytes with bare and eutrophic sandflats, such as the eutrophic area, which has no seagrass cover for more than 15 years and has lower organic matter content. Nevertheless, the seagrass area presented lower evenness, mainly due to the dominance of *H. ulvae*, a common *taxon* in several estuarine and coastal areas (Norkko et al. 2000). Regarding the feeding guild composition, the dominance of this species reflected into the almost identical and dominant percentages of herbivores and detritivores abundance and production in seagrass area, especially in the pre-mitigation period; while in the eutrophic area the community was dominated essentially by detritivores. When excluding *H. ulvae* from the analysis, most of the community abundance and production was dominated by the detritivores alone (with few exceptions), which is a common feature in estuarine systems (Sprung 1994). This suggests that a great part of the energy/biomass enters the system via the detritus food chain, with only a small contribution from the grazing generated food chains. Differences in the detritivore composition were also registered among areas and throughout the study period (discussed below). When *H. ulvae* is accounted, the deposit feeders composition was more heterogenic in the eutrophic area; when not accounted, the pattern was similar in the two areas.

11.5.3 Pre-mitigation versus Post-mitigation Periods

A general conclusion that may be taken is that the macroalgal blooms temporarily increased production of specific *taxa*, contributing the overall increase in community production. Yet, this temporary gain is quite short lived, since the long-term effects of eutrophication and associated algal blooms necessarily imply a loss of the faunal production associated to the seagrass, conducting to an overall decrease in whole estuarine production and integrity, as also discussed by Dolbeth et al. (2003, 2007). In fact, the competition with the fast-growing algae has been considered responsible for the disappearing of the macrophyte beds, as also referred to occur in several other coastal areas worldwide (Flindt et al. 1999, Cloern 2001). So, if no mitigation measures would be taken in the Mondego estuary, the system would most probably collapse.

When comparing both areas, before and after the implementation of mitigation measures, several differences are noticeable in the structure and function of the benthic communities. While the nitrogen loading into the estuary was effectively reduced (lowering N/P ratio) since 1999, the benthic communities, both flora and fauna, took longer time to recover. The lowest biomass and extent of *Z. noltii* was observed in 1999, concomitant with the lowest values of species richness, density, biomass and production obtained for the *Z. noltii* area. Afterwards, the benthic community seemed to recover, until the occurrence of an extreme major flood, when the benthic community slowed down the recovery process, to re-start the recovery again after the flood. The eutrophic area, as a simple bare sandy/muddy habitat, appeared less variable over time in species richness, density, biomass and production. Similar to the *Z. noltii* area, during the extreme flood, the benthic community slowed down the recovery process. This occurrence highlights climate variability as an additional source of stress, whose frequency is becoming higher, and may intensify the course of eutrophication in estuarine and coastal waters, as discussed by Rabalais et al. (2009). In fact, ecosystems are being subjected more frequently to multiple stressors, which may act synergistically to lower the natural resistance and resilience of their inhabitants to disturbance, as observed for the Mondego estuary (Cardoso et al. 2008a, b, Dolbeth et al. 2007). Regarding the trophic organization, most of the community was dominated by deposit feeders, emphasizing the role of the detritus in the food chain. Yet, some differences were observed comparing pre-and post-mitigation periods for both areas. For the eutrophic area, generally, higher heterogeneity of the feeding guilds in the pre-mitigation period was observed, while in the post-mitigation period the community was essentially dominated by detritivores. These differences are probably associated to the higher diversity of food sources in the pre-mitigation period, while in the post-mitigation one the energy sources are probably more stable. The *Z. noltii* bed supported, in general, higher percentage of carnivores, herbivores and omnivores than the eutrophic area, which makes it functionally richer. For this area, there was an increase in omnivores in the post-mitigation period, especially during the flood year. This tendency was also observed for the eutrophic area, though in lower percentages. An increase in omnivores dominance may be regarded as an advantage, especially when subjected to a stress source, since omnivores have flexible generalist diets, being able to change their function in the benthic food web, according the available resource (Fox et al. 2009).

Compelling with the changes in the community in the latest years of the study was the evidence of succession from *r*-strategists towards *K*-strategists species in both study areas. *S. plana*, typically a slow growing species, increased considerably in the estuary after the mitigation measures, especially in the eutrophic area, probably responsible by the increase of the estuarine mean biomass in the estuary. *H. diversicolor* showed a similar development as *S. plana*, increasing considerably in estuary after the introduction of the mitigation measures, together with an overall decrease of the opportunist polychaetes *A. romijni* and *C. capitata*, considered as indicators of nutrient enrichment and pollution of estuarine systems (Cardoso et al. 2007). As discussed by Cardoso et al. (2007), in the Mondego estuary, *H. diversicolor* is not considered as an indicator of organic enrichment, but instead represents a measure of the ecosystem trajectory into a more nutrient-controlled system.

11.5.4 Evaluation of the Ecosystem Recovery

The measures undertaken in Mondego estuary south arm in the late 1990s included a set of restoration measures, such as the controlled use of fertilizers in the agriculture fields located upstream, the improvement of water circulation by enlarging the connection between the two arms, a better management of sluice openings, which led to strong modifications of the physico-chemical features of the estuary (lower turbidity, lower suspended organic matter and lower re-mineralization). These seemed to have some success, reflected in the dynamics of the primary producers and macrofauna. Nutrient loading was significantly reduced and no further macroalgal blooms were recorded, as also discussed by Lillebø et al. (2005, 2007). In addition, *Z. noltii* beds are gradually recovering, both in biomass and extent (4.7 ha in 2006), starting to re-colonize the upstream areas (personal observation). The biomass and production of the macrozoobenthic community also increased, together with a slight increase in biodiversity, as discussed in the previous section. Yet, although the cover extent of the *Z. noltii* is the same at the beginning and end of the study period (i.e. 1993 and 2002), biodiversity levels have still not recovered to the observed in 1993, which might have compromised the macrobenthic community after the extreme floods in 2000/01, as discussed by Dolbeth et al. (2007).

At the present time, the *Z. noltii* beds are recovering, but the eutrophic area restoration is still far away from the original habitat similar to the *Z. noltii* beds. Quite recently, some very small patches of *Z. noltii* appeared in the eutrophic area (personal observation), but the plant recovery is not fully taking place. Potentially this area may be reaching a new steady state community for bare sand/mud, which is consistent with the increase of the *K*-strategists species (e.g. *S. plana*) in this area.

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Chapter 12

Aquatic Plant Diversity in Eutrophic Ecosystems

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Abstract The chapter contains studies conducted on the impact of eutrophication on aquatic plant diversity. It covers the concept of eutrophication, its causes and effects on plant diversity within an aquatic ecosystem. A decrease in species diversity and disappearance of aquatic plants were noted in most of the water bodies of the world as a result of eutrophication. The plant diversity in eutrophic ecosystems was studied with special reference to phytoplankton and aquatic macrophytes. Studies on wetlands in western Europe showed that the nutrient enrichment or eutrophication leads to changes in species composition, declines in overall plant species diversity, and loss of rare and uncommon species. The nutrients are the major limiting factors for the aquatic plant diversity. Various environmental factors play a significant role in determining the aquatic plant diversity in a eutrophic ecosystem. In Europe about 17 lakes underwent eutrophication and have lost all or most of their submerged species and favoured the fast growing species capable of regrowth after weeding. The eutrophication process causes succession of macrophytes with complete loss of submerged vegetation and dominance of phytoplanktons. A direct relation was found between the succession of algae and trophic level of the water body. Few studies on measurements of plant diversity such as density, frequency, abundance, and diversity indices are included in this chapter.

Keywords Aquatic plants · Biodiversity · Eutrophication · Ecosystems

12.1 Introduction

Biodiversity refers to the huge variety of living organisms on this earth. More than 1.7 million species of organisms have been identified so far. This number is increasing by about 15,000 new species every year. Aquatic ecosystems and tropical rain forests support the rich biodiversity. The present biodiversity is a result of about 3.5 billion years of evolution. During this period innumerable species have evolved and the same numbers have extinct, leaving their evidences as fossils (Agarwal and Agarwal 2007). According to the US Office of Technology Assessment (1987), biological diversity is "the variety of variability among living organisms and ecological complexes in which they occur" (Sharma 2005).

The environmental modifications are the major causes of global species extinction as well as habitat destruction. The environmental constraints undergoing human modification include levels of nitrogen, phosphorus, calcium, pH, atmospheric CO_2, herbivore, pathogen, and predator densities, disturbance regimes, and climate. Extinction would occur because the physiologies, morphologies, and life histories of plants limit each species to being a superior competitor for a particular combination of environmental constraints (Tilman and Lehman 2001). The excessive addition of nutrients to water causes quality problems and is one of the major causes of eutrophication in freshwater ecosystems (Ansari and Khan 2007).

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Due to ubiquitous eutrophication and fragmentation, many plant species are actually threatened in Europe. Most ecosystems face an overall nutrient input leading to changes in species composition. Increasing species had better means of long-distance dispersal and are more competitive than decreasing species (Romermann et al. 2008). The plant species play a vital role in the functioning of soft-water lake ecosystems. Lake area, altitude, trophic state, and water quality have been found to be good predictors of macrophyte species richness for lakes. Major threats to the survival of soft-water lake vegetation are acidification, eutrophication, increased recreational use of lakes, and the effects of lake regulation for hydroelectric schemes. Due to such global changes, the plant diversity of soft-water lake in northern Europe is potentially severe (Murphy 2002). The disturbances like sedimentation, turbidity, retention time, eutrophication, and changes in hydrologic time resulted in shift in plant species composition, reduction in fauna, decreased biodiversity, reduction in secondary production, increased flood peaks/frequency, increased above-ground production, and loss of aquatic plant species with high light compensation point (Detembeck et al. 1999). Total annual nitrogen retention has been found higher in eutrophic lake (20–30%) than in oligotrophic lake (<5%). The N retention in Newegian lakes was limited by phosphorous (Berge et al. 1997).

Understanding the mechanisms that govern biological diversity in various environments is one of the greatest challenges for the scientific community of today. Compared to terrestrial and benthic habitats, mechanisms regulating species diversity in planktonic ecosystems have been dealt with relatively little. This is mainly due to the scarcity of the experimental evidence from field studies where the multitude of spatiotemporal scales has been covered with sufficient resolution. Various examples of the regulation of species diversity in the Baltic Sea showed peculiarities of the aquatic system as a growth environment for phytoplankton in comparison to terrestrial/connected systems. Eutrophication causes increase in availability of nutrients resulted in shift in the species composition of ecosystems (Kononen 2001). Eutrophication or nutrient enrichment is a condition in an aquatic ecosystem where high nutrient concentrations enhance the excessive growth of phytoplankton and macrophytes. Eutrophication is a natural process occurs in all the aquatic systems and takes thousands of years. However, a high rate of input of nutrients due to anthropogenic activities significantly developed this condition in a very short period of time. This is called an artificial eutrophication (Rovira and Pardo 2006). Lakes in the Netherlands also facing the problem of eutrophication and efforts have been made for their restoration (Gulati and Evan 2002). The main causes of eutrophication are the nutrients from the agricultural soil, weathering of rocks, mining, construction work, poor land use, detergents containing phosphates, partially treated or untreated sewage containing nitrates and phosphates in higher amounts. The eutrophication process directly or indirectly affects the physical, chemical, and biological characteristics of an aquatic ecosystem (Ansari and Khan 2006a). The eutrophication causes an increase in plant and animal biomass, frequency of algal blooms, growth of rooted plants, and decreases in species diversity. As a result an increase in turbidity and anoxic conditions occur. Because of the high density of aquatic organisms in a eutrophic system, there is often a lot of competition for resources. This high degree of competition and the sometimes high chemical or physical stress make higher the struggle for survival in eutrophic systems. As a result the diversity of organisms is lower in eutrophic than in oligotrophic systems (Ansari 2005). In a dimictic Greifensee lake (Switzerland) the analysis of cores revealed that the lake was oligotrophic for 120 years and became hypertrophic in the 1970s (Sand-Jensen et al. 2000). In France the rivers highly polluted with monthly average of $NH_3{}^{+}$ and $PO_4{}^{3-}$ varying from 24–1,200 to 60–1,400 $\mu g\ L^{-1}$, respectively, induced development of hypertrophic and pollution-tolerant macrophyte communities. The aquatic macrophyte communities were suggested to be as the bioindicator of eutrophication (Schnitzler et al. 1996). The degradation of a reed bed of the Lake Balaton (Hungary) causes disappearance of aquatic plant or reduces species diversity and chemical composition diversity of the plant (Kovacs et al. 1996).

Species-rich grassland and heathland communities still occur in low-intensity farming systems in many European countries. Such systems are intensively exploited, with a subsequent decrease in species numbers. Abiotic constraints like eutrophication and acidification influence the restoration of these communities (Bakker and Berendse 1999).

The increased nutrient loading caused coastal eutrophication and climatic perturbations (drought, hurricane, floods) in Galveston Bay, USA (Paerl et al. 2003). The great Masurian Lake in Poland was found highly eutrophic (Kufel 2001). An increase in living standard agriculture and development of local industries resulted in eutrophication of Lake Taihu in China (Weimin et al. 1997). The runoff fertilizers in the stable ecosystem of Golfo Pagasitikos (Greece) resulted in eutrophic condition showing algal bloom (Triantafyllou et al. 2001). The eutrophication caused a marked increase in the total biomass of macrobenthos. Heavy pollution resulted in the degradation of communities, a sharp decrease in the total biomass of plants and animals, and eventually a complete destruction of the biota (Kussakin and Tsurpalo 1999). The density of the epilithic algal community and the species diversity increased with the degree of eutrophication (Le-Bris et al. 1998). A comparison was made between the systems of brackish and freshwater lakes in the Norfolk Broadland of eastern England; most of which have undergone eutrophication and cause the lake community to change from submerged plant dominance to phytoplankton dominance at intermediate states of nutrient loading (Moss 1994). Species diversity has declined in ecosystems worldwide as a result of habitat fragmentation, eutrophication, and land use change. If such decline is to be halted, ecological mechanisms that restore or maintain biodiversity are needed (Collins et al. 1998).

12.2 Plant Diversity: Eutrophic Ecosystems

The Baltic Marine Environment Protection Commission reported that the eutrophication is a main threat to the aquatic environment. The response of biological communities has reacted to an increase in man-induced multisource stresses. Changes in plankton, benthic animals, and macroalgal assemblages and fish communities have been documented in most parts of the Baltic Sea (Tremp 2007).

A spatial heterogeneity was detected in La-safar coast with small and shallow water bodies in Spain. The area was found highly eutrophic due to high amounts of phosphorus. The water body types in the region were found to be a function of total phosphorus, flux of water, water transparency, and presence of macrophytes. The eutrophic conditions contributed to the loss of biodiversity (Rodrigo et al. 2003). The increased nutrient load in an aquatic ecosystem resulted in two equilibrium states, a turbid phytoplankton dominated meso or eutrophic state and a clear macrophyte dominated meso or oligotrophic state (Lorenz et al. 2003). Studies on lake Sakadas, Croatia, showed that over-exploitation of the ecosystem or degradation of its biotic structure alters ecosystem processes to the point where the ability of the ecosystem to produce desired resource is seriously diminished. The oxygen depletion caused mass fish mortally for a short time but had significant consequences on the lake ecosystem (Mihatjevic and Novoselic 2000). The species diversity of various water bodies in the river flood plains, wetlands of upper Rio Parana in Brazil, and natural aquatic ecosystems including marshes in river deltas, mangroves near Ecuador border with Peru have been worked out and water qualities like phosphate, light, pH, depth, and sediment have been considered in surveys of the Brazilian water bodies (Murphy et al. 2003).

Changes in aquatic systems as a result of nitrate leaching have led to decreased drinking water quality, eutrophication, hypoxia, and decrease in aquatic plant diversity (Mosier et al. 2001). The anthropogenic forcing in Venice lagoon led to higher diversity of non-indigenous species. The human interventions in Mediterranean Sea environment resulted in a fertile ground for mass bioinvasion of aquatic nuisance species capable of altering ecosystem equilibrium ultimately resulting in global change (Occhipinti and Savini 2003). Relationships between plant diversity and ecosystem properties can be explored by classifying component species into three categories – dominants, subordinates, and transients. Dominants reoccur in particular vegetation types, are relatively large, exhibit coarse-grained foraging for resources, and, as individual species, make a substantial contribution to the plant biomass. Subordinates also show high fidelity of association with particular vegetation types but they are smaller in stature, forage on a more restricted scale, and tend to occupy microhabitats delimited by the architecture and phenology of their associated dominants. Transients comprise a heterogeneous assortment of species of low abundance and persistence; a high proportion are juveniles of species that occur as dominants or subordinates in neighbouring ecosystems. When ecosystems become degraded by pollution or over-exploitation to a point where formerly dominant organisms are eliminated or debilitated, it is often possible to demonstrate a causal connection between losses in biodiversity and declines in ecosystem function (Grime 1998).

12.2.1 Phytoplankton Diversity

Eutrophication can profoundly change rocky shore communities. These changes often cause the replacement of perennial, canopy-forming algae such as *Fucus* spp. with annual, bloom-forming algae such as *Enteromorpha* spp. grazing; however, can counteract eutrophication by eliminating the annual algae's susceptible recruits (Worm and Heike 2006). At low concentration the P in aquatic ecosystem causes dominance of chlorophytes and Cryptomonad (phytoplankton) with lesser amount of diatoms and dinoflagellates. The diatoms also attained substantial proportions at lower P loads (Sand-Jensen et al. 2000). The changing diversity of phytoplankton composition in response to flushing of the Alte Donau Lake (Austria) with water of better quality led to high phytoplanktonic diversity (Mayer et al. 1997). In meso-eutrophic Lake Taihu (China) the microcystis species in summer occupied 85% of total algae biomass and appear to be main blomming species among *Anabena*, *Melocira*, *Cyclotella,* and *Cryptomonas* (Weimin et al. 1997). In Moa point Bay (New Zealand), the sewage discharge affected the biodiversity. The area was dominated by *Ulva lactuca*, seaweed (Rogers 2003). The eutrophication resulted in a high diversity of aquatic habits with 128 species of microalgae (in the wet and dry periods) 1–21 species of phytoplankton. The decreased species diversity and disappearance of aquatic plants were noted in the reed beds of Balaton Lake (Hungary) as a result of eutrophication (Kovacs et al., 1996).

Eutrophication causes predictable increase to the biomass of algae in lakes; reservoirs, streams, and rivers; and coastal lagoon ecosystem. Consistence and predictable eutrophication cause increase in cyanobacterial dominance of phytoplankton worldwide in natural lake. Similar trends were found both for phytoplankton in turbid reservoirs and for suspended algae in large river (Smith 2003). The bio-films assemblages of algae, fungi, and microorganism in the sediments of aquatic system provide an integrated long-term measure of ecosystem functioning with structural attribute-like biomass and diversity (Burns and Ryder 2001). Surveys on planktonic organisms in 31 fishless ponds in southern Michigan were conducted; the plant and herbivore diversity were correlated to pond nutrient levels. The density of plants (phytoplankton) showed a positive correlation with nutrient levels. The density of herbivores (zooplankton) was positively correlated with the density of plants. The trophic level changed significantly with eutrophication as indicated by these significant correlations (Leibold 1999).

12.2.2 Macrophyte Diversity

Aquatic macrophytes are aquatic photosynthetic organisms, large enough to see with the naked eye, that actively grow permanently or periodically submerged below, floating on, or growing up through the water surface. Aquatic plants are represented in seven plant divisions: Cyanobacteria, Chlorophyta, Rhodophyta, Xanthophyta, Bryophyta, Pteridophyta, and Spermatophyta. Species composition and distribution of aquatic macrophytes in the more primitive divisions are less well known than for the vascular macrophytes (Pteridophyta and Spermatophyta), which are represented by 33 orders and 88 families with about 2,614 species and 412 genera. These 2,614 aquatic species of Pteridophyta and Spermatophyta evolved from land plants and represent only a small fraction of the total number of vascular plants (Chembers et al. 2008). Kushiro, The *Trapa Japonica*, showed wide range of distribution with varying nutrient levels in Kushiro lake Japan. The pH was higher in *Polygonum amphibium* community. The Chl-*a* concentration at a given nutrient level was significantly lower in water with submerged macrophytes than in water without them. The presence and absence of submerged macrophytes were related with difference in phytoplankton community (Takamura et al. 2003).

Lake Peipsi consisting of three parts (Lakes Peipsi, Lake Pihkva, and Lake Lammijarv) is located on the border of Estonia and Russia where the dominant species typical of eutrophic lake including *Phragmites australis, Potamogeton gramineus, Potamogeton pectinatus, P. amphibium, Eleocharis palustris, Sagittaria sagittifolia,* and *Butomus umbellatus* were found (Haberman et al. 2000). Strong dominance of five aquatic macrophytes specifically *Eichhornia azurea* was recorded within the riverine floodplain wetland of upper Rio Parana river (Brazil) (Murphy et al. 2003). Transition towards hypertrophy has affected biodiversity and productivity of most aquatic and wetland systems in the Lower Danube Wetland System (LDWS) over the last two decades. The aquatic macrophytes have been deeply involved in ecosystem reorganization in these circumstances (Cristofor et al. 2003). The downstream water quality has been found related with the impact of loose strip (*Lydhrum salicaria*) into the North American wetland which affected the nutrient cycling and decomposition rates of various other species (Emery and Perry 1996). The increased composition of sediments due to eutrophication of Lake Balaton (Hungary) increased the establishment of *Typha augustifolia* and *T. latifolia*. Due to degradation of reed beds (*Phragmites australis*) the aquatic plant showed disappearance and reduced species diversity (Kovacs et al. 1996). The eutrophication in De Nieuw Roopseplassen in the Netherlands reduces the number of rare species and increases the abundance of meso to hypereutrophic species, *Fragilaria berolinensis* (Dam and Mertems 1993). Lake Geneva (Switzerland) which underwent rapid eutrophication until 1980 and followed a reversal is still in progress. The submerged macrophytes along 20 km of the lake shore studied in 1972, 1984, and 1995 revealed that *P. pectinatus, Potamogeton perfoliatus, Potamogeton lucens,* and *Elodea canadensis* did not show significant changes in their distribution, with the two former species dominant throughout. However, *Charo* sp. and *Myriophyllum spicatum* increased in abundance in 1995. The abundance of *Potamogeton pusillus* consistently increased from 1972 to 1995. A new species *E. canadensis* appeared for the first time in the lake in 1995. The macrophytes were used to asses the water quality (Lehman and Lachavanne 1999). In the Netherlands, eutrophication in aquatic environment reduces macrophytes diversity and their former habitat was characterized by non-floating duckweeds like *Lemna* sp., *Spirodela polyrhiza,* and *Azolla filicoides* (Smolders et al. 1995).

12.2.3 Wetland Diversity

Studies of wetlands in western Europe and of other terrestrial ecosystems in North America frequently show that nutrient enrichment leads to changes in species composition, declines in overall plant species diversity, and loss of rare and uncommon species (Barbara et al. 1999). In the riverine wetland of River Rhone (France) the cutoff channels with law sinuosity and intermediate flood frequency were divided into three groups. The first group was species poor, had maintained nutrient rich and turbid water due to backflows and close connection with river. The species richness in second group was intermediate due to lower river backflows. The third group was species poor because of excessive ground water supply, with limited species growth (Bornette et al. 1998). The Undasa wetlands of Ujjain (India) did not show full biological potential due to excessive pollution inputs from nearby industries (Shrivastava et al. 2003).

A study was conducted to examine the role of eutrophication or desiccation on two wetlands of the Zurich region (Sackriet, 3.8 ha, and Wollwisli, 0.8 ha). The present distribution of plant communities within both wetlands compared with the distribution mapped 20 years earlier and surveyed site conditions within the different vegetation units in order to determine which changes in site conditions might have caused past vegetation changes. The differences in vegetation suggest that the observed vegetation shifts were due to both eutrophication and desiccation, with the effect of eutrophication depending on water level and the effect of desiccation depending on nutrient supply. The vegetation in two wetlands of the Zurich region (Sackriet and Wollwisli) was found affected by eutrophication and desiccation (Bollens et al. 2001).

In the experiments conducted on 220 wetland mesocosms revealed that the presence of specific species scientifically affected macrophyte biomass, respiration, and the resilience and functioning of ecosystem (Engelhardt et al. 2001). The high disturbance activities in wetlands of Canada and the USA were reflected in terms of species loss and it was suggested that wetland restoration strategies shall cover diverse wetlands. The vegetation removal and site disturbance caused plant invasion. The encroachment activities in marshes and pond of southern ecoregions of great lakes were common causes of disturbances. The prevalence of anthropogenic stresses and open water habitat increased exotic species abundance in inland wetland of southern great lakes. Vegetation removal and site disturbance caused plant invasion (Detembeck et al. 1999).

12.3 Plant Diversity: Nutrient Limitations

The addition of nitrogen over a long time will affect the number of species, the type of species present, the amount of annual growth, and the change from year to year in the growth of each species in a plant community (Tilman 1996). The field studies conducted on 12 cutoff channels of the River Rhone (France) revealed that low- or high-nutrient levels decreased species richness by selecting specialized species. The sites of the channel farther from the river having excessive ground water supplies had limited species growth and requirement. The most frequently flooded channels have highest species richness and occurrence of rare and fugitive species (Bornette et al. 1998). In a 3-year nutrient addition experiment variation effects of nutrients were examined. Nutrient limitation shifted across estuarine salinity gradients; salt and brackish marsh vegetation was N limited, while oligohaline marsh vegetation was co-limited by N and phosphorus (P). Eutrophication by both N and P has the potential to greatly reduce the characteristic high diversity of oligohaline marshes. Inputs of both nutrients in coastal watersheds must be managed to protect the diversity and functioning of the full range of estuarine marshes (Crain 2007). The P was considered as deriving force for increased diatom population in hypertrophic Lake Laugh Neagh (Foy et al. 2003).

The nitrogen loading during normal precipitation promoted dominance of non-rooted species. In a chain of small lake in south-eastern Michigan, USA, macrophyte community dynamics have shown strong relationship variations in nitrogen loading (Tracy et al. 2003). Proportions of algal biomass consisting of cyanophytes, filamentous chlorophytes, hallophillic diatoms, and diatoms utilizing nitrogen heterotrophically were greater than vascular plants in eutrophic river segment than in less nutrient-enriched segment (Theibaut and Muller 1998). In Spanish semi-arid wetland, sedimentation played a key role in nutrient dynamics. Internal loading was much higher than external loading and nutrients showed seasonality. The macrophytes namely *Cladium marisceus* var. seagrass showed significant relationship with N and P (Sanchez and Alvarez 2001). The increased organic loading in drainage with dominant submersed macrophyte in Sussex (UK) resulted in rapid diversions towards either phytoplankton or *Lemna* dominance at the cost of submerged macrophytes. There was an increase in the cover of *Potamogeton crispus* and a decrease in *Groenlandia densa* on treatment with excessive organic nutrients. These nutrients increased conductively and decreased oxygen, pH, and chlorophyll values in the water column (Thomas and Daldorph 1994).

12.4 Plant Diversity: Environmental Factors

The physical processes within a water source have major implications for controlling eutrophication in aquatic bodies (Lau and Lane 2002a, b, Khan and Ansari 2005, Ansari and Khan 2006b), while nutrients, temperature, pH, dissolved oxygen, carbon dioxide, and light limit the growth and development of aquatic plants (Shen-Dong Sheng and Shen 2002). The plant species composition showed significant correlations with potash concentrations, winter and summer groundwater level, water depth, and elevation in bank vegetation. The aquatic vegetation showed correlation with pH, Cl^-, organic carbon, $NH_4{}^+$, concentration, and water temperature (Best et al. 1995). Studies in 39 drainage streams in Victoria (Australia) showed the diatom communities (245 taxa) were found strongly correlated within land use and practices. The streams were influenced by heavy irrigation practices and dryland farming at reduced species diversity and richness (Blinn et al. 2001). In some oligotrophic Danish lakes the quality of water was lost in last 40 years. The aquatic mosses dominated the vegetation at the coast of rooted *Isoetes* due to slight change in pH (summer 5.6 in 1958–1963 and 5.07 in 1976–1994). The annual pH fluctuation in the lake was 0.3 unit 40 years ago and 0.9 units presently. The population of *Nitella flexilis* has disappeared which once dominated the bottom vegetation in hypolimnion (Riis and Sand-Jensen 1998).

Rich algae flora in thermal strings and their main floods in Bulgaria consisted of more than 200 species of cyanoprokaryotes, glaucophytes, diatoms, and yellow green and red algae. There were 75 taxa of Chlorophyta which dominated the total algal diversity. Only the sterile filament of *Spirogyra* and *Zygnema* was more widely distributed in thermal water (Stoyneva 2003). Logging of forest trees on the coastal area promoted algae population due to increased availability of nutrients, light, and mixing depth (Nicholls et al. 2003). The biodiversity increased in Lake Choughu, China, due to increased light intensity and water temperature (Peng et al. 2003).

The silicate availability of river supply and strong tidal mixing seems to determine year-round dominance of diatoms over dinoflagellates in Urdaibai estuary, north Spain (Maria and Orive 2001). Studies in 39 streams in Victoria, Australia, showed that the drainage with low nutrient but a wide range of salinity showed strong association with diatom taxa (Blinn et al. 2001). The species richness increased with Si, Na, and $PO_4{}^{3-}$ but declined with increasing pH, Ca, and Mg. Diatom assemblages in Kathmandu valley reflect water chemistry (Juttner et al. 2003). At Cape Bolinao, Philippines, the seagrass species diversity, shoot density, and depth penetration declined with increasing amounts of suspended material and increasing water column light attenuation along the silt gradient (Bach et al. 1998).

A shift in primary producers from eelgrass to macroalgae was recorded which in turn alters the food web. The increase in nitrogen load increased eelgrass shoot density and biomass. In a north temperate estuary nutrient loading changed the habit, physical and chemical structure, and food webs. The microalgal biomass increased with nutrient loading, but the biomass and shoot density of eelgrass decreased. The removal of macroalgae increased eelgrass abundance, water column, benthic boundary, and oxygen concentration (Deegan et al. 2002). The ponds contaminated with organic matter and sewage water resulted in covering of water surface with aquatic weeds (Lee and Lee 2002).

12.5 Plant Diversity: Succession Pathways

The successions of floral and faunal diversity were noted in the river and lake ecosystem during 20 years up to 2001 which are caused by ecological conditions and industrialization in Moldova (Toderas et al. 2001). A mild and short period of eutrophication in Uatuma River (Brazil) resulted in a succession pattern of *Eichhornia* → *Utricularia* + Cyperaceae → *Salvinia*. This succession was observed after the closure of dam in 1987 at Baleina (Brazil) (Walker et al. 1999). In the mire of La Vraconnaz located in the Jura Mountains (Switzerland), a study showed the changes in vegetation that took place during the 11 years after the bog burst. The indicator values showed a general eutrophication and a closing of the herb layer in the slide area. The total species richness showed an increase. All pioneer species (first colonists) that appeared on bare peat after 2 years are perennials, either geophytes or hemicryptophytes and the majority of them are transient species. Species diversity in pioneer vegetation increased during the first 9 years and then become stable (Feldmeyer and Kulchler 2002).

In a study on densely populated lowland of Europe and its comparison with the record of last 100 years, majority of 17 lakes underwent eutrophication and have lost all or most of their submerged species. Low habitat diversity and frequent disturbance in streams favoured the fast growing species capable of regrowth after weeding (Sand-Jensen et al. 2000). The construction of reservoir and hydroelectric power station on River Danube (Slovakia) changes the diversity in the reservoir and the Danube River streams. The tank construction resulted in the onset of succession (Otahel and Valachovic 2002). The analysis of sediment core of a hypertrophic Lake Sobygaard (Denmark) showed a succession from naturally eutrophic state to hypertrophic state during recent centuries. Distinct changes in chironomid communities reflected the eutrophication process and the succession of macrophytes through *Chara*, *Ceratophyllum,* and *Potamogeton* dominance to the existing state, with complete loss of submerged vegetation and dominance of phytoplankton (Broderson et al. 2001).

In Jeroslawiec Lake the phytoplankton sample and bottom sediments showed a succession of algae corresponding to the increasing trophic level of the lake. Most habitat of this lake was found eutrophic (Pelechaty et al. 1997). The silicate availability in Urdaibai estuary (Spain) resulted into dominance of diatom over dinoflagellates round the year (Maria and Orive 2001). The weed removal has been found as an important factor in reducing macrophyte communities as reported in studies conducted in 79 small Danish lowland streams (Baattrup et al. 2003). The eutrophication promoted phytoplankton development which in turn reduced water transparency and impeded the growth of rooted plants accelerating terrestrial habitats

(Amoras and Bornette 1999). Field studies conducted on 12 cutoff channels of Rhone River (France). There was an intermediate connectively between channels and river which results into propagule input to the wetland. The high nutrient level cause decrease in number of species and intermediate nutrient level favoured the co-occurrence of species. The nutrient enrichment level, water turbidity, connectivity of wetland to the river was affected by diversity and succession of species in wetland ecosystems (Bornette et al. 1998). The succession occurred in Danube River, Slovakia, resulted in change in spatial distribution and species composition of macrophytes (Otahel and Valachovic 2002).

12.6 Plant Diversity: Assessment and Monitoring

Freshwater biodiversity is a much underestimated component of global biodiversity, both in its diversity and in its potential to act as models for fundamental research in evolutionary biology and ecosystem studies. Freshwater organisms also reflect quality of water bodies and can thus be used to monitor changes in ecosystem health (Xu et al. 2007). Few studies represent a basis for the assessment of plant species threat. Applying the deduced knowledge about the life history of decreasing versus increasing species to habitat-scale approaches it is possible to predict which species may become threatened in the future simply from the combination of their trait values (Candolin et al. 2008). The submerged macrophytes have been suggested as an important tool for long-term monitoring of running water quality and are equally sensitive to the process of eutrophication and acidification of water. These submerged macrophytes were reported to be most suitable for the assessment of lotic (running) waters (Tremp et al. 1995).

The algae diversity of Najafgarh drain in Delhi (India) receiving effluents from agricultural, industrial, and domestic sources indicated that the drain was excessively eutrophic. The large number of genera and species present in the drain were highly tolerant to organic pollution. These species have been suggested suitable for bio-monitoring and phytoremediation (Sinha 2001). The growth responses of common duckweeds *Lemna minor* and *Spirodela polyrrhiza* were studied at various temperature and pH levels for its possible application for remediating eutrophic water (Ansari and Khan 2008, 2009). The flora of Novi Sad-Selo Canal (Yugoslavia) was found diverse with several submerged floating and emerged hydrophytes. The existing species was assessed for their bioindicating values (Stojanovic et al. 1999). In Australian river systems the bio-films have been used as suitable parameter to assess long-term functional changes. The bio-films are assemblages of algae, fungi, and microorganism which cover rocks, woods, and sediments in aquatic system. This parameter facilitates in historical comparison with the previously recorded data sets of ecosystem function and structure (Burns and Ryder 2001).

From the herbarium specimen of macrophytes collected from diatom assemblages of man-made coastal dune between Blankenberg and Haiat (Balgium) dating from 1852 to 1929 and its comparison with the present collection revealed fairly eutrophic for phosphorus (Denys 2003). The diatoms (Bacillariophyta) sp. in two streams of Ontario (Canada) was found directly related with total nitrogen and phosphorus. The waited averaging regression models indicated had the utility of indicating eutrophication in these two lowland water streams (Winter and Duthie 2000). The diatoms indicated a long-term eutrophication trend in lower Lough Erne (Ireland, UK) and showed a shift from *Aulacaseira* sp. to *Stephanodiscus* sp. The changes in the geochemistry and diatom assemblages were expected to be linked with increased nutrient supply from sewage and diffused agricultural sources (Gibson et al. 2003). In a study conducted on Lake Greifensee (Switzerland) the ecosystem function was related with P decline. For the assessment of resilience of lakes, the species richness, biomass-based diversity, and evenness were best parameters; with reducing P content the species richness and diversity indices of phytoplankton increased (Burgi et al. 2003).

12.7 Plant Diversity: Indicator of Eutrophication

The plant diversity is a strong bioindicator for the river ecosystem and lake ecosystems differed in types and responses (Lorenz et al. 2003). The chironomid taxa were the best indicator of highly productive

Danish lakes lacking abundant submerged vegetation (Broderson et al. 2001). High diversity of phytoplankton composition due to flushing of the Lake Alte Donau (Austria) with water of better quality has been reported ciliates and metazooplankton constituted 10% of the total biomass. The high water temperature increases the dominance of cyanobacteria (Mayer et al. 1997). The free-floating duckweed *L. minor* is highly sensitive to a number of factors in its surrounding habitat. Its potential as an indicator of eutrophication has been determined (Ansari and Khan 2002). *Chlorella vulgaris* was detected in water samples with high nutrient content during eutrophication sensitivity experiment on some algae in three inland lentic resources of Bhopal in India. In these lakes *Crucigenia crucifera* was highly tolerant to Ca, Mg, K, and P but moderately tolerant to nitrogen. Some other species of *Crucigenia* viz *C. quadricauda*, *C. rectangulris* performed well in Ca-, Mg-, and K-rich models. *Pediastrum* sp. grew well in calcified and nitrated water. *Cosmarium granatum* was adversely affected by phosphorus and other nutrient as well. *Merismopedia punctata, Melosira granulata,* and *Microcystis aeruginosa* were found in highly eutrophic waters (Garg and Garg 2003). The fast growing ephemeral algae in shallow coastal waters worldwide were considered to be the indicators of coastal eutrophication as studied in two shallow water ways in the west coast of Sweden (Sundback et al. 2003). In sewage sludge-treated Lake Dalavaya at Mysore (India), the metal toxicity due to input of heavy metals and cations in addition to soap and detergent carried in by sewage water caused eutrophication. The *Eichhornia crissepus* utilized Co and Cd as micronutrients and showed luxurious growth. The water hyacinth (*E. crassipes*) acted as bioindicator (Sujatha et al. 2001).

The *Potamogeton* species in 17 Danish lakes have been reported to be a suitable bioindicator of water quality. *P. filiformis* and *P. polygonifolius* are typical indicators of oligotrophic condition. Some other species common to oligotrophic condition are slow growing *P. alpinus*, *P. lucens*, *P. praelongus,* and *P. zosterifolia*. The transient species in these lakes of *Potamogeton* are *P. angustifolius*, *P. colourates (P. coloratus)*, *P. densus, and P. rutilus* (Sand-Jensen et al. 2000). A wider spatial sampling including different channel types and system were suggested to be required to analyse the usefulness of bryophytes as water quality bioindicators in irrigation channels (Martinez-Aragon et al. 2002). The diatoms have been recognizing as bioindicator in temperate streams in Kathmandu valley, Middle Hills of Nepal, and northern India. In Kathmandu valley richness and diversity increased with K^+, Cl^-, $SO_4{}^{3-}$, and NO^{2-}, but declined with Al^{3+}, Fe^{3+} surfactant, and phenol (Broderson et al. 2001).

12.8 Plant Diversity: Measurements

The aquatic plant diversity can be measured with the help of frequency, density, and abundance of the plant communities within an ecosystem. The diversity index is a useful parameter for the study of biodiversity in an ecosystem.

12.8.1 Frequency

Frequency is the number of sampling units in which a particular species occurs. The phosphorus enrichment in the marsh and slough area of the Northern Everglades (USA) caused distinct changes in species frequency. The marshes in un-enriched and enriched areas were dominated by *Cladium jamaicense* and *Typha domingensis,* respectively. The un-enriched open waters were characterized by *Eleocharis*, *Utricularia* sp., *Chara zeylanica,* and *Nymphea odorata*. The enriched open water areas were dominated by *Polygonum* sp. and floating plants (Vaithiyanathan and Richardson 1999). It does not give correct idea of the distribution of any species, unless correlated with other characteristics, such as density.

12.8.2 Density

Density represents the numerical strength of a species in the community. The number of individuals of the species in any unit area is its density. It gives the degree of competition in an ecosystem. Studies on species diversity, density, and seasonal distribution of aquatic plant in Harike Lake (India) have been carried out. The *Eichhornia* were found to cover 70% of the lake in a total of 34 species recorded (Bach et al. 1998).

The variation in water current velocity at two sites in downstream of Kisto Bazar Nala (India) resulted in the variation in the density of biotic communities which indicated the possibility of eutrophication into the impound water (Bhatt et al. 2001). The effect of high population densities on the growth rate of *L. minor* (L.) was studied under laboratory conditions at 23°C in a medium with sufficient nutrients. At high population densities, there was a non-linear decreasing growth rate with increasing *L. minor* density (Haterdand and Heerdt 2007).

12.8.3 Abundance

The abundance is the number of individuals of any species per sampling unit of occurrence. The abundance and diversity of phytoplankton, zooplankton, and fishes were compared among limnetic and littoral habitats with (L^+) and without (L^-) hyacinths in Lake Chivero, a man-made hypertrophic reservoir near Harare (Zimbabwe). In addition, the littoral macrophytes community and macro-invertebrates associated with hyacinth mats were inventoried. The phytoplankton community was dominated by blue-green algae (mainly *M. aeruginosa*), typical for a hypereutrophic lake. There was an increasing importance of chlorophytes (*Staurastrum* sp. and *Pandorina morum*) and diatoms (*Cyclotella meneghiniana*) and a decreasing dominance of *Mycrocystis* along the discriminant axis from L^+, L^- to P (Brendonck et al. 2003).

The eutrophication and reduced N:P ratio in Danube delta lakes caused species changes showing increased relative abundance of colonial blue-green algae (Vadineau et al. 1992). The phosphorus application in a pond causes significant difference in the abundance and diversity of phytoplankton of 78 genera from 5 phyla of algae (Ahmadi and Bani 1999). The benthic flora composition indicated mesotrophic or eutrophic conditions throughout the upper Illinois river basin (USA). Few diatoms were indicative of hypertrophic water. The P regression calibration model with benthic algal abundance was found to be a significantly predictable model (Leland and Porter 2000). The macrophytes species richness and abundance increased along an upstream to downstream zonation which was characterized by an increase in mineralization and nutrient level in Northern Vosges Mountains in northeast France (Thiebaut and Muller 1998). Eutrophication was found to have reduced the number of rare species and increase the abundance of meso to hypereutrophic species in Nieuwkoopse Plassen in the Netherlands (Dam and Mertems 1993). The reduced abundance of species endemic or preferential to serpentine soils can be interpreted as a loss of "ecological quality" of the analysed community (Ricotta et al. 2004).

12.8.4 Diversity Indices

The diversity index of a community is the ratio between the number of species and the number of individuals in that community. The species richness index, Palmer's generic index, Margalef's index are useful for the assessment of water quality and eutrophic conditions of water (Hariprasad and Ramkrishnan 2003). Danilov and Ekelund (1999) worked out seven diversity and one similarity indices utilizing phytoplankton data for their usefulness in eutrophication studies in some central Sweden lake. The lakes of different eutrophication level were monthly investigated during autumn 1998. Several diversity indices calculated failed to distinguish between different eutrophication levels. However, similarity index on the basis of presence–absence matrices of phytoplankton species proved its high efficiency for eutrophication studies in the lake in central Sweden (Danilov and Ekelund 1999).

The studies in the Lake Kasumigaura, Japan, showed that trophic state index ranged monthly between 55 and 75. COD and total P were limiting factors. The Chl-*a* was found suitable biological indicator of eutrophication in the lake (Wei et al. 2000). The diversity and density index was found higher in saline alkaline ponds applied with organic fertilizer. The pond fish population had relatively lesser diversity and density index of planktons (Zhao et al. 2000). The wetland zooplankton index was found more useful than several diversity indices in Laurentian Great Lake basin (Lougheed and Chow 2002). The Shannon-Weaver and Whilm Dorris diversity indices were not found to be a very sensitive parameter to detect moderate pollution. The saprobity index of Waternabe was found more precise for deteriorated part of River Rosario in Argentina (Sadusso and Morana 2002). In a eutrophic shore lake of Geneva (Switzerland) two methods of bioindication of water quality by macrophytes were compared, the macrophytes index based on nutrient loads and the saprobic index measuring organic pollution. The saprobic index was found to be sensitive to small-scale changes in species composition and reflected the changes in eutrophication (Lehmann and Lachavanne 1999). The nitrate concentrations in three tropical coastal lagoons of Yucatan (Mexico) suggested that trophic status was influenced by freshwater springs. On the basis of nitrogen status the Celestron lagoon was found eutrophic, Chelem lagoon oligotrophic, and Dzilan lagoon as mesotrophic. But NH_4^+ concentrations indicted that all three lagoons were mesotrophic owing to some impact on remineralization process. It was found that indices based on nutrient concentration and phytoplankton biomass were useful indicators of trophic status in such coastal lagoons influenced by surface water (Herrera-Silveira et al. 2002).

In the most organically polluted sites of Karasu River basin (Turkey) three diatom species namely *Gomphonema parvulum, Nitzschia palea*, and *Navicula cryptocephala* were found in high densities. The COD, BOD, and concentrations of nutrients (NO_3^-–N, NO_2^-–N, NH_4^+–N, and PO^{-3}–P) were co-related negatively with DO, except the GI (generic index) which was directly correlated with COD, BOD, DO, and nutrients. The diatom indices indicated that the Karasu River basin was eutrophicated and organically polluted (Gurbuz and Kivrok 2002). Similarity indices based upon presence and absence of matrices of phytoplankton are commonly used for the determination of eutrophic state of lakes (Danilav and Ekelund 1999).

12.9 Discussion

In the last 50 years, eutrophication (natural or artificial) has become a widespread environmental pollution due to the growing population, rapid urbanization, and increased agricultural and livestock production (Rovira and Pardo 2006). Eutrophication has become a major cause of concern in the developing as well as developed countries. Eutrophication was recognized as a pollution problem in European and North American lakes and reservoirs in the mid-twentieth century. Since then, it has become more widespread. Surveys showed that 54% of lakes in Asia, 53% in Europe, 48% in North America, 41% in South America, and 28% in Africa are found in eutrophic state (Colin et al. 2007). The water bodies of the highly populated countries like China, India, Bangladesh, Pakistan, Indonesia, industrialized countries of Europe and Great Lakes States of the USA and Canada are under the direct threat of eutrophication. The population increase coupled with economic development and changes in lifestyle during this period has added to the problem. The availability of freshwater for human consumption will be one of the great issues of the twenty-first century. The scientific interest in eutrophication has consistently been significant during the last 25 years. The emphasis in this area of research has suddenly risen in recent years. It is predicted that eutrophication, acidification, and contamination by toxic substances are likely to increase as threats to freshwater resources and ecosystems.

The problem of eutrophication besides being dependent upon the nutrient inputs is greatly influenced by the number of environmental factors. Eutrophication induced significant change in the biodiversity. The change in biodiversity directly affects the trophic structure of the ecosystem. The floral and faunal diversity is threatened in the coastal areas receiving direct input of nutrients from some of the major rivers like the Amazon, Nile, Ganges, Mississippi, Brahmaputra, and Thames. Some effective control measures like biological control, mechanical control, legislative measures, and the awareness programmes pertaining to the present threat to water resources on the blue planet needs to be carried out effectively (Khan and Ansari 2005). The change in macrophytic flora in Lake Takkobu since the mid-1970s was briefly described. At that period floating-leaved plant communities dominated by species such as *Nuphar pumila* and *Potamogeton natans* proliferated in the littoral zone of the lake, and many submerged plants occupied the whole area of the lake. From the 1990s some species began to decline because of the progressive eutrophication of the water. Except for *T. japonica*, which has become the most dominant species, many other species are in decline. To date some species seem to have gone extinct while others are threatened with extinction. In these 30 years, both the total biomass and species diversity of aquatic macrophytes have drastically decreased (Kadono 2007).

Alteration in the species diversity during floods and drought period in the eutrophic Shoe Lake (USA) has been observed (Tracy et al. 2003). The aquatic plant diversity is reported to be sensitive to a large number of other factors, viz pollutant, resulting in reduced diversity of macrophytes, phytoplankton, and several aquatic fauna (Nicholls et al. 2003). The heterotrophy resulted in adverse biodiversity and productivity in most aquatic and wetland system in the River Danube (Cristofor et al. 2003). In the past second half century the biodiversity of Lake Victoria (Kenya) has changed considerably owing to changes in drainage basin, increased eutrophication, and invasion of water hyacinth. But some of the species disappeared from Lake Victoria have been reported to occur abundantly in the Yale Swamp Lakes (Aloo 2003). A 50% reduction in submerged macrophytes population in past 30 years has been reported in Lake Mikolyrkei (Poland) which is under increasing eutrophication process (Solinska and Symoides 2001).

Krabovaya Bay (Shikotan Island, South Kuril Islands) polluted by organic matter resulted in decline in species diversity. Changes in littoral flora as well as aquatic and swamp vegetation were analysed with increasing eutrophication of the mesotrophic Lake Mikolajski. The number of submerged macrophytes species decreased by 50% and the frequency of most of the remaining species declined several fold. No new species were encountered. Species retreating from the lake littoral included all *Chara* species, *Potamogeton obtusifolius, P. natans,* and *Hydrocharis morsus-ranae*. A significant lowering of the phytosociological diversity and species richness of aquatic and swamp communities was observed (Solinska and Symoides 2001).

Ecosystem resistance to a single spacer relies on tolerant species that can compensate for sensitive competitor and maintain ecosystem process, such as primary production. The resistance to additional stressors depends increasingly on species tolerance being positively correlated (Vinebrooke et al. 2004). The disturbances like sedimentation, turbidity, retention time, eutrophication, and changes in hydrologic time resulted in shift in plant species composition, reduction in fauna, decreased biodiversity, reduction in secondary production, increased flood peaks/frequency, increased above-ground production, and loss of aquatic plant species with high light compensation point. The Lake Greifensee (Switzerland) in a dimictic lake was oligotrophic in the 1970s. The elimination of sewage treatment plants and ban on P-containing detergents reduce the P load, despite the primary productivity decreased it showed a continuing eutrophic conditions. In this lake the phytoplankton biomass was dominated (Burgi et al. 2003).

Modification with biocoenosis (ecological communities) is inevitable results of dense human population which introduced an alien species into German water. Until 2003, 96 alien species have permanent established population. About every fifth species was invasive in nature (Nehring and Welling 2003). In shallow mesotrophic Kaljasjarbi Lake the abundant aquatic macrophytes increase the resistance of lake to eutrophication by assimilating nutrient, providing refuge for zooplankton and having allelopathic effect on phytoplankton (Kauppila and Valpola 2003). Inland aquatic ecosystems and their biodiversity in Asia represent a wide spectrum along a complex continuum of interacting ecological, economic, socio-cultural, and political gradients all of which determine their present and future. Whereas the diversity of biophysical environments ensures a rich inland aquatic biodiversity, their present status has been greatly influenced by human societies that have depended on them for millennia. The future of aquatic biodiversity in Asian countries will depend upon a radical change in national policies on water and upon research that can support the development of appropriate policies (Stehlik et al. 2007). An analysis conducted on spatial variation in species diversity of aquatic macrophytes. The species richness, species evenness, plant mass index, and similarity index in two morphometrically different lakes, Phewa and Rupa in subtropical Pokhara valley, Nepal, were compared. Vegetation mapping methods including GIS and semi-quantitative Kohler indexing approach were applied. A total of 56 species of aquatic macrophytes were reported along with higher species diversity of hydrophytes and helophytes in Lake Phewa and Lake Rupa, respectively. The study suggests that monitoring of influential factors like siltation and cultural eutrophication hold importance for the conservation and management of biodiversity of aquatic macrophytes in particular and health of subtropical lake ecosystem in general (Shrestha and Janauer 2000).

Phytoplankton dynamics in a shallow eutrophic lake was studied with respect to environmental forces which drive species composition and diversity. Application of a similarity–diversity model revealed that a high diversity was a transient non-stable state, whereas drastic changes or long-lasting stable environmental conditions are characterized by low diversity (Weithoff et al. 2001). Freshwater ecosystems are situated in depressions in the landscape. As a result they accumulate the impacts of human activities. Emission of pollutants can be controlled by technology. Modification of hydrological and biogeochemical cycles, that have negative consequences for the biota, can only be reduced by an integrated approach. The Restoration Ecohydrology Concept serves as an integrating mechanism for the restoration of physically modified freshwater habitats. There are two main approaches to restoration and mitigation: First actions at the catchment level connected with integrated management of abiotic factors including landscape planning, catchment management, forestation, phytotechnologies, and hydrology by impoundment and second actions at the level of the aquatic ecosystem include restoration of the diversity. These measures have to be oriented towards the control and regulation of the dynamic pool of nutrients, maintaining a fine line between eutrophication and impoverishment, to manipulate the productivity and diversity of the biota (Zalewski et al. 2001). Europe's largest lake (Lake Ladoga) and north European soft-water lakes are eutrophic and support an important plant component of the biodiversity resources of Europe. Within this region (comprising the British Isles, Scandinavia, and the north European Plain from Brittany to the Baltic States), some 24 soft-water lake euhydrophyte communities occur, including stress-tolerant communities which particularly characterize soft-water lake vegetation. Lake area, altitude, trophic state, and water quality are good predictors of macrophytes species richness (Murphy 2002).

Major threats to the survival of soft-water lake vegetation include acidification, eutrophication, increased recreational use of lakes, and the effects of lake regulation for hydroelectric schemes. All such pressures tend to reduce the diversity of typical soft-water species present in affected lakes. The implications of such global change for the maintenance of soft-water lake plant diversity in northern Europe are potentially severe (Murphy 2002).

Popular interest in protecting the world's plant and animal species has intensified during the last 20 years. Around the globe, biological communities that took millions of years to develop are being devastated by human activity. A new multidisciplinary science that has evolved to deal with the crises confronting biological diversity is the conservation biology, which has two goals: first, to investigate human impacts on biological diversity and, second, to develop practical approaches to prevent extinction of species (Sharma 2005).

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Chapter 13

Linking Anthropogenic Activities and Eutrophication in Estuaries: The Need of Reliable Indicators

Paula Castro and Helena Freitas

Abstract In recent decades we have witnessed human-induced alterations to coastal watersheds that caused or aggravated eutrophication problems in estuarine systems. Although eutrophication is a natural process in the aging of lakes and estuaries, human activities are greatly accelerating this phenomenon by increasing the rate at which nutrients and organic substances enter aquatic ecosystems from their surrounding watersheds. Despite there is a fundamental understanding of the processes that cause eutrophication we still need to develop new methods and indicators and improve others to detect incipient eutrophication in ecosystems, providing information before major changes in species composition and abundance take place. The need of reliable indicators that correlate estuarine changes with human pressures over time (urban, industrial, and agriculture) is essential to understand their dynamics and to ensure effective management of these ecosystems. This chapter aims to evaluate the relative utility of rooted macrophytes and chemical variables in assessing eutrophication of two Portuguese estuaries subjected to different degrees of anthropogenic inputs owing to varied human activities and soil uses on their watersheds. We also document nitrogen deposition history in each estuary.

Keywords Indicators · Macrophytes · Estuaries · Salt marshes · Din · Don

13.1 Introduction

Nutrient loading and poor water quality are serious environmental problems affecting aquatic systems (Howarth 2008, Nelson et al. 2009). Urban development, encompassing the expansion of human populations and associated industry, has led to the widespread discharge of both municipal and industrial effluents into surface and groundwaters worldwide (Umezawa et al. 2008). These pressures, along with changes in agricultural practices, have led to increased concentrations of nutrients in riverine and estuarine waters, particularly of nitrogen and phosphorus (EEA 2003, 2005).

Eutrophication is generally defined as a condition in an aquatic ecosystem where increases in anthropogenic supply of nutrients and organic matter change the composition of primary producers (Nixon 1995), particularly stimulating the blooms of algae. Such changes in producers then lead to other changes that propagate up through food webs (McClelland and Valiela 1998, Cabral et al. 1999, Raffaelli 1999, Deegan et al. 2002). Eutrophication is a widespread environmental problem in rivers, lakes, coastal oceans, but also can affect terrestrial ecosystems (Carpenter et al. 1998, Smith et al. 1999) and is arguably the biggest pollution problem facing estuaries all over the world (GESAMP 2001, EEA 2001). These conditions interfere with the recreational use of lakes and estuaries and the health and diversity of indigenous fish, plant, and animal populations (Valiela et al. 1992, Herman et al. 1996, GESAMP 2001, Deegan 2002). Productivity and species composition are altered, with a few opportunistic species proliferating. Nitrogen is a limiting nutrient for plant growth in most aquatic

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and terrestrial ecosystems, so it is not surprising that anthropogenic N addition enhance primary productivity. Increases in total production of vascular plants, changes in soil chemistry, nitrate leaching and accumulation in groundwater, and changes in plant and microbial community structure (see Smith et al. 1999) are some of the consequences from nitrogen enrichment. Although eutrophication is a natural process in the aging of lakes and estuaries, human activities are greatly accelerating this phenomenon by increasing the rate at which nutrients and organic substances enter aquatic ecosystems from their surrounding watersheds. Despite there is a fundamental understanding of the processes that cause eutrophication we still need to develop new methods and improve others to detect incipient eutrophication in ecosystems, providing information before major changes in species composition and abundance take place.

13.1.1 Estuaries and Salt Marshes

Marine and coastal areas are among the earth's most valuable resources providing a wide range of ecological services that benefit humans and other species (GESAMP 2001). In a river basin context, estuaries have considerable ecological, social, and economic value, for they are associated with nursery, breeding, and spawning grounds of some key marine and terrestrial species (Adam 1990). Estuaries are one of the most threaten ecosystems because of the large percentages of human population and increasing of urbanization in these watersheds (Valiela et al. 1992, Uncles et al. 2000, Ma and Aelion 2005). Pollutants are commonly derived from human activities and, in estuaries, usually result from intensification of agriculture, industrialization, fossil fuel combustion, discharge of waste and sewage, and infrastructural development and construction, and include persistent organic pollutants, nutrients, oils, radionuclides, heavy metals, pathogens, sediments, litters, and debris (Williams 1996). Estuaries are historical sources for a longtime diffusion of hazardous substances such as trace metals in soil and groundwater (Nriagu 1996, Audry et al. 2004, Caeiro et al. 2005, Cave et al. 2005, Caçador et al. 2009) which lead to serious contamination conditions. Free trace metals can be incorporated into organism's tissues (Stoltz and Greger 2002, Bonneris et al. 2005, Riba et al. 2005), bioaccumulated through the food web, and ultimately cause human health problems. Effects of pollution on aquatic ecosystems and aquatic living resources are reviewed in Goldberg (1995) and Islam and Tanaka (2004).

Salt marshes are found fringing in many of the world's coasts, occupying the interface between land and sea and may be defined as natural or semi-natural areas, vegetated by halophytic species (herbs, grasses, or low shrubs), bordering saline water bodies, and subjected to periodic flooding as a result of tidal fluctuation (Beeftink 1977). The characteristic floristic composition and structure of salt marsh areas are sufficient to maintain a distinction from surrounding areas. Salt marshes are one of the most productive natural systems (Adam 1990, 2002) with a total biomass production ranging from 0.008 to 11.34 g DW m^{-2} day^{-1} and around 2.5 g DW m^{-2} day^{-1} in 40° latitudes (Duarte and Chiscano 1999). Many of these communities are sheltered by intertidal estuaries and, like them, also have been stressed and degraded in the past decades due to humane use and development (Allen 2000, Adam 2002), despite their wide amplitude of adaptability to physicochemical and organic pollution. Land claim for agriculture, industry or residence, saline cultures, aquacultures, and pollution are effectively big major problems affecting these habitats (Adam 1990, 2002). In order to preserve them it is important to know their distribution, abundance, cover, as well as changes taking place.

Monitoring salt marshes is critical to understand their dynamics and to ensure an effective management of these ecosystems. Recent decades have witnessed a marked loss of coastal wetlands, but reliable data on salt marsh area are not available in Europe, despite the very long history of salt marsh studies (Allen 2000). Many arguments have been put forward for the preservation of salt marshes as useful parcels of coastal/seascape (Vince et al. 1981, Mitsch and Gosselink 1993, Valiela et al. 2001); salt marshes export materials important to food webs of deeper waters, act as a nurseries for many species of commercially important fisheries stocks, provide sources of harvestable shellfish and sites for aquaculture, intercept toxic contaminants, stabilize shorelines, and provide waterfowl refuges and nesting areas and stopover for migratory birds. There are also findings that salt marshes are net importers of some land-derived nutrients and organic matter (Flindt et al. 1997, Costa et al. 2001) protecting the quality of valued eelgrass habitats.

13.1.2 *Nutrient Loading and Plant Responses*

Human-induced alterations to coastal watersheds have affected natural biogeochemical cycles, particularly natural N cycle through food and energy production, and have invariably resulted habitat loss and disturbance of the ecology of aquatic systems (Callaway et al. 1995, Vitousek et al. 1997, Galloway 1998, de Jonge et al. 2002, Erisman et al. 2003). Estuaries and salt marshes are key ecosystems impacted by intense human activities along the coast and tributary rivers (Uncles et al. 2000, Bowen and Valiela 2001, EEA 2001, Ma and Aelion 2005). Increased levels of nitrogen are an important contributor to eutrophication of estuarine waters resulting in the proliferation of opportunistic, fast-growing marine seaweeds, affecting basic habitat structure, energy flow, trophic ecology, and community composition (Deegan et al. 2002, Verdelhos et al. 2005, Teichberg et al. 2009), and can lead to a decline in the extent of seagrass meadows. Seagrasses, and particularly eelgrass, are highly sensitive to increases in N loads from land, so that the widespread losses of this habitat signal the onset of major changes in the quality of coastal waters. Thus, changes in eelgrass densities and meadow distribution could be considered as indicators of increased N loads (Milchakova 1999, Jagtap et al. 2003, Hauxwell et al. 2003, Pardal et al. 2004).

Salt marsh rooted macrophytes are important biological agents that control sediment redox structure, nutrient cycling, and the degree of eutrophication in estuarine systems as they link the nutrients in sediments with the overlaying water (Simas and Ferreira 2007, Koretsky et al. 2008, Sousa et al. 2008). Plant responses in these coastal environments depend on ambient conditions in water as well as in sediments, and given that estuarine systems are normally nutrient limited, the plant nutrient content, density, and growth rate usually correlate with the concentrations of available nutrients (Short 1987, Udy and Dennison 1997a, b, Osgood and Zieman 1998, Udy et al. 1999, Tyler et al. 2003). Seagrasses are able to take up nutrients from both water column and sediment (Pedersen and Borum 1993, Touchette et al. 2003, Rubio et al. 2007). The combined capacity for nutrient uptake is an important property that allows these species to show luxuriant growth in nutrient-poor coastal waters (Hemminga 1998). In spite of plant adaptations to estuarine systems, sediment properties, such as high porewater salinity and low sediment redox potential, particularly in warmer months, or water quality, and light availability, may in turn interact with nitrogen availability limiting plant production (Cartaxana et al. 1999, Edwards and Mills 2005, Pergent-Martini et al. 2005, Sousa et al. 2008). Nutrient enrichment also increases epiphyte growth on seagrass and impairs productivity (Neckles et al. 1993, Short et al. 1995).

13.1.3 *The Selection of Indicators*

Coastal eutrophication is a growing problem, and developments of effective indicators that provide early information and predict how ecosystems will react are needed. Common methods used to study nutrient enrichment rely on water column concentrations data, but sometimes poor relations can be found between them and productivity or abundance of primary producers (Fong et al. 1993, Lee et al. 2004). This chapter intends to analyze the consistency and effectiveness of nitrogen (N), carbon (C), plant aboveground biomass, nitrogen stable isotopes (δ^{15}N), and lead (Pb) as sensitive indicators that reflect the differences of human influence and eutrophication in two Portuguese estuaries with historical different nitrogen regimes.

Carbon and nitrogen measurements are commonly used to assess nutrient loading, trophic status, and primary productivity of estuaries (Smith et al. 1999, Abril et al. 2002, Burdloff et al. 2008). Since growth of primary producers in estuaries is typically N limited (Howarth 1988, Cloern 2001, Invers et al. 2004) the historical evaluation of N status of these ecosystems is of particular interest. Nutrient concentrations as well as plant density and biomass are expected to be correlated with concentrations of available nutrients (Rogers et al. 1998, Udy et al. 1999, Invers et al. 2004). Carbon content can also influence metal distribution in the sediments (Caçador et al. 2000, Fitzgerald et al. 2003) and its accumulation in sediments through time may be related to the supply of organic matter, the effects of dilution of other detrital materials, and preservation of sediment layers (Emerson et al. 1985).

Coupling information on the elemental and isotopic nitrogen provides a better understanding of anthropogenic activities in estuaries (Fry et al. 2003). Radioactive isotopes have limited lifetimes (from nanoseconds to thousands of years) and undergo a decay to form a different element. Stable isotopes indicate those isotopes of an element which are stable and that do not decay through radioactive processes over time. Most elements consist of more than one stable isotope. For the element nitrogen are known two stable isotopes: ^{14}N and ^{15}N, being ^{14}N the abundant form (Fry 2006). The use of stable isotopes is a powerful technique that is cost effective and informative and has been used in ecological research as chemical markers for tracing the origin of organic matter. They have also been suggested to be reliable indicators of relative eutrophication in freshwater and estuarine ecosystems (Peterson and Fry 1987, McClelland and Valiela 1998, Savage and Elmgren 2004, Baeta et al. 2009). These linkages are possible because different combinations of land uses convey different N isotopic signatures to receiving waters and reflect the level of anthropogenic influence (Cole et al. 2005, Umezawa et al. 2008, Bannon and Roman 2008). δ^{15}N signatures are related with DIN (dissolved inorganic nitrogen) concentrations in the receiving water and can be explicitly linked to N in estuarine macrophytes, becoming heavier as the N loads and contribution by wastewater increase (Cole et al. 2005, Cabaço et al. 2008a). Nutrient discharges from sewage treatment plants can contribute significantly to coastal and estuarine eutrophication (Savage et al. 2004) and have signatures between +15 and +30‰ (depending on the type of treatment), whereas artificial fertilizers largely applied in agriculture typically have negative (or close to zero) δ^{15}N values. In particular, natural abundance stable nitrogen isotopes have been used effectively to trace the influence of organic sewage as it is transported physically and biologically through ecosystems (e.g., Lepoint et al. 2008). The technique developed by Costanzo et al. (2001) of applying naturally occurring stable isotopes of nitrogen to map the spatial influence of sewage nitrogen in Moreton Bay (Australia) is an example of how stable isotopes may have applications in environmental management. Savage (2005) reviewed the use of stable nitrogen isotopes to delineate the influence of sewage nitrogen in coastal ecosystems that receive ^{15}N-enriched tertiary-treated sewage that is discharged mainly as dissolved inorganic N.

Vertical profiles of materials on coastal sediments have long been used as sources of information on time courses of environmental change, and metals (Caçador et al. 1996, Marcantonio et al. 2002, Shumilin et al. 2002, Cundy et al. 2003) have been used as markers that conveyed notions as to the relative influence of anthropogenic activities. A common characteristic of human activities is the emission of lead as a part of pollution and has long been introduced in the global environment. With the industrial revolution by the mid-eighteenth century and then by the automobile utilization with gasoline additives containing Pb, this element was emitted at large scale primarily distributed through atmospheric transport (Nriagu 1996, Hagner 2002, Smittenberg et al. 2005). Lead and isotope composition of Pb can be used as environmental indicators describing the affected environment, investigating the history of anthropogenic emissions over short as well as long time periods, identifying sources of contamination, and detecting trends of pollution. In particular, long-term trends in anthropogenic influence have been recorded by the presence of lead in sediments or ice (Caçador et al. 1996, Vallelonga et al. 2005, Smittenberg et al. 2005, Caçador et al. 2009). In Chesapeake Bay, Marcantonio et al. (2002) found that sediments deposited before the Industrial Revolution had Pb concentrations of about 10 ppm.

13.1.4 Scope and Goals

In the scope of the Water Directive (Directive 2000/60/EC) adopted by the European Parliament and the Council the framework for the quality management of all waters including surface and coastal waters, aiming to protect and enhance their status quality (chemical and ecological) is established. In this context, throughout Europe, monitoring networks and management plans should be established to create a coherent and comprehensive knowledge of ecological and chemical status and ecological potential of all water bodies by 2015. However, the formulation of loading targets should consider all sources and must be adapted to local situations (Ulén and Weyhenmeyer 2007). In addition, the European Union's Marine Strategy

Directive (Directive 2008/56/EC) also requires the reduction of eutrophication effects as to achieve good environmental status of marine waters.

Eutrophication from nutrient loading in estuarine environments has contributed to salt marsh and seagrass decline (Short and Burdick 1996, Castro and Freitas 2006) and has altered salt marsh communities and dynamics (Caffrey et al. 2007). However, linking anthropogenic activities and eutrophication effects in estuaries is not always linear and understandable, since these ecosystems respond in different ways to nutrient loading. Responses and methods of monitoring relevant indicators vary regionally, hindering interpretation of ecosystem state changes (McQuatters-Gallop et al. 2009). To successfully manage nutrient enrichment it is important to define and quantify sources of N to coastal waters and to identify estuarine responses to N loading. New methods and indicators are thus needed to effectively detect incipient pollution impacts before it has substantially altered the estuarine system.

Hoping to contribute toward a process-oriented monitoring and assessment strategies in river basins and estuaries, this work aimed to evaluate the relative utility of rooted macrophytes and chemical variables in assessing eutrophication of two Portuguese estuaries – Mondego and Mira – subjected to different anthropogenic inputs owing to distinctive land use/land cover in their watersheds. We also document long-term changes of nutrient deposition in each estuary. An integrated approach using biological as well as chemical indicators was established to assess and evaluate past and present anthropogenic pressures on water quality, nitrogen dynamics, and plant responses to eutrophication. We also compared indicator's performances to establish their relative ability to see if Mondego and Mira nitrogen regimes resulted in differences on the year and magnitude of plant growth and nutrient content.

13.2 General Approach

We present here a case study conducted in two Portuguese estuaries as a part of a research project funded by the Portuguese Ministry of Science, Technology, and Higher Education conducted between 2001 and 2005 whose main objectives were to use an integrative approach to (1) assess the changes in salt marshes throughout the past century; (2) assess the effect of eutrophication on species composition, biomass, and nitrogen content of salt marsh and seagrass meadow; and (3) establish links between land-derived nitrogen and estuarine producers.

13.2.1 Study Areas

The Mondego estuary (40°08′N/8°50′W) is located in the central coast of Portugal (Fig. 13.1). It consists of two arms separated by the Murraceira Island, with distinct sedimentology and hydrodynamics characteristics (Dinis and Cunha 1999): the north, deeper, and south, largely silted up in the upstream areas, causing the freshwater to flow essentially through the north arm. Consequently, the circulation in this arm depends mainly on tides and small freshwater inputs of the Pranto River.

Besides dredging activities related to harbor activities (in the north arm) which cause physical disturbance, this estuary supports several industries, many saltworks, and aquaculture farms (Table 13.1). The conversion of the salt marsh into non-tidal land has a very long history in this region. Embanking these areas to create agricultural land occurs in the Murraceira Island since early times. In 1947 all the island was already transformed for these activities. Salines and aquacultures were also observed in the 1940s (Castro and Freitas 2006).

A reduction of seagrass beds, mainly in the dwarf eelgrass (*Zostera noltii*) and their replacement by green macroalgae, has been commonly observed from the 1980s until the early of this decade (Flindt et al. 1997, Verdelhos et al. 2005) attributed to high nutrient loads from areas upstream (Table 13.1). From 1948 to 1998 there was a loss of about 17% of salt marsh area (Castro and Freitas 2006). In 1998, management measures were taken in place to reduce nutrient loading from the Pranto River and to increase freshwater circulation in the system from the north arm.

The Mira estuary (37°40′N 8°45′W) is a narrow shallow tidal estuary located on the southwest of Portugal (Fig. 13.1). It runs for over 32 km and is about 150 m wide in the lower part and 30 m in the upper part with a mean depth of about 6 m. In spring and summer the water varies from well mixed to slightly stratified during neap and spring tides, respectively

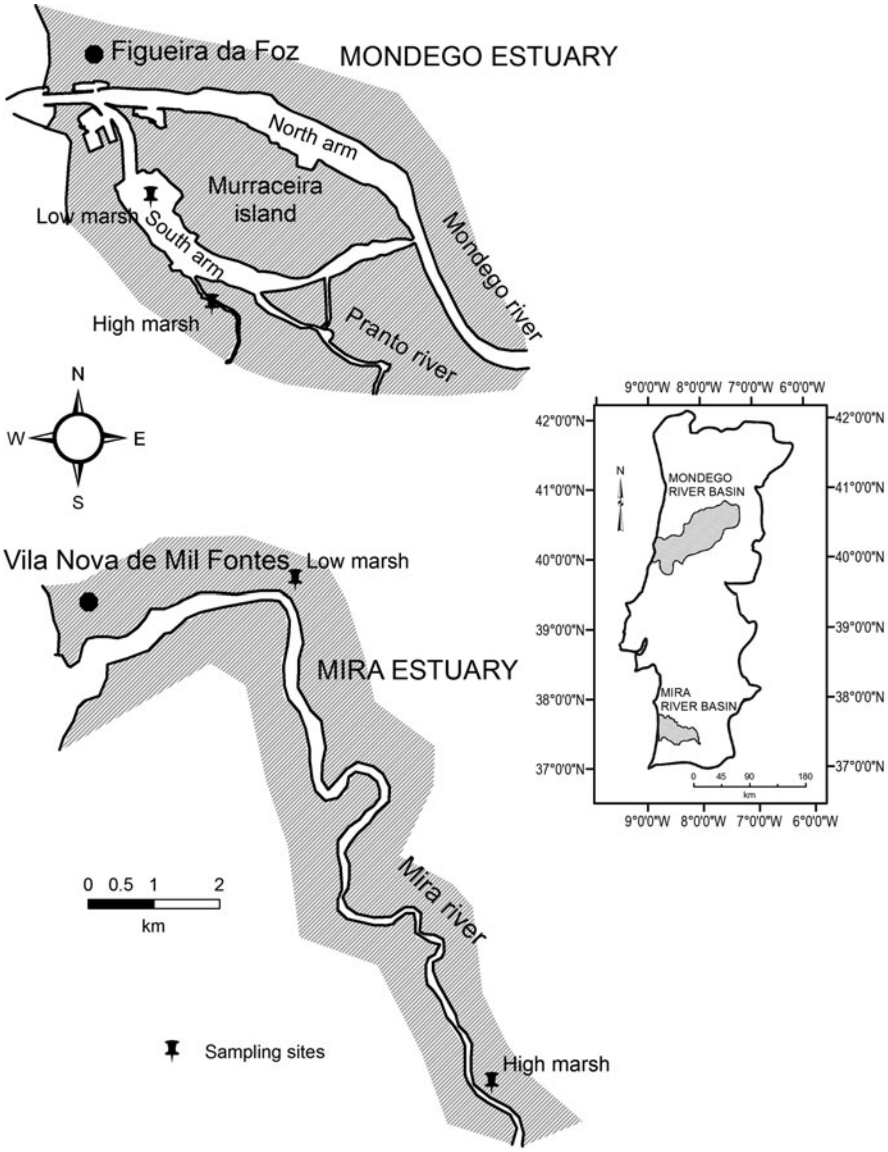

Fig. 13.1 Location maps of the Mondego and Mira river basins and estuaries

(Andrade 1986). Freshwater discharge is controlled by the Santa Clara Reservoir located ca. 50 km upriver. This ecosystem is relatively undisturbed by anthropogenic activities when compared to Mondego and marsh variation seems to follow a natural dynamics rather than been affected by impacts resulting from human pressure (Castro and Freitas 2006). However, human influence has increased in the last decades with growing of urban areas and construction of fish farms. Pollution sources affecting this reservoir were derived mainly from agriculture, aquaculture, cattle breeding, and also from wastewater of urban areas (Table 13.1). A maximum of 280 kg N ha^{-1} was estimated to enter the estuary each year (Costa et al. 2001).

13.2.2 Eutrophication Status: Comparison Between Estuaries

To assess the eutrophication status and the response of plants to ambient eutrophication we selected two salt marsh areas (low and high marsh) (Fig. 13.1) with similar abiotic and biotic conditions and determined, from 2001 to 2003 (every 3 months), %N, %C, and δ^{15}N in

Table 13.1 General characteristics of Mondego and Mira rivers and estuaries

		Mondego	Mira	Source
River length (km)		258	145	PBH Mondego (2002), PBH Mira (2002)
River basin area (km^2)		6,645	1,582	PBH Mondego (2002), PBH Mira (2002)
People km^{-2} in the river basin (1995)		101	14	PBH Mondego (2002), PBH Mira (2002)
People km^{-2} in the estuarine watershed (2001)		165	15	INE (2001)
Resident population in the estuarine watershed (2001)		62,601	26,106	INE (2001)
Number of residences in the estuarine watershed (2001)		37,596	18,336	INE (2001)
River flow (mean values)		80 m^3 s^{-1}	2.94–10 m^3 s^{-1}	Neto et al. (2008), Chainho et al. (2008), www.inag.pt (May 2009)
Water residence time (mean values)		1–2 days (south arm)	10 days	
Land-derived N loading (estimations)		Before 2000: 540 kg Nha^{-1} year^{-1} (South Arm) After 2000: 202 kg N ha^{-1} year^{-1} (South Arm)	280 kg N ha^{-1} year^{-1}	Neto et al. (2008), Flindt et al. (1997), Castro et al. (2007a), Costa et al. (2001)
N regimes (mean water column concentrations – μM)	$N–NO_3{}^{-}$	7	6	Castro et al. (2009)
	$N–NH_4{}^{+}$*	25	13	
	DON*	209	91	
PN	289	251		
Driving forces		Saltworks, aquaculture, industries, intensive agriculture, tourism, urban development, fisheries, harbor development	Tourism and urban development near mouth of river, aquacultures, some cattle breeding, fisheries	–

$NO_3{}^{+}$ – nitrate; $NH_4{}^{+}$ – ammonium; DON – dissolved organic nitrogen; PN – particulate nitrogen
*Significant differences between estuaries

leaves as well as plant aboveground biomass of three dominant macrophytes: *Zostera noltii* Hornem. in low marsh sites, and *Spartina maritima* (Curtis) Fernald and *Halimione portulacoides* (L.) Aellen in both upper and lower marshes. Nutrient concentrations were also measured in bare sediment and in sediments beneath each plant canopy. One-way ANOVA and Tukey post hoc tests were performed to see if there were differences among species and estuaries using the two stations per estuary as replicates. More detailed information about this methodological approach can be found in Castro et al. (2007a, 2009).

13.2.3 Historical Nutrient History

The assessment of estuarine changes over time is especially useful for ecological systems that lack long-term monitoring data. Although managers can measure current nutrient concentrations as well as seagrass and eelgrass distribution to examine the present status of an estuary, this method is not useful to determine its history. Methods to identify incipient eutrophication brought about by increased N loading would greatly help efforts by environmental managers to preserve critical coastal habitats. Recording historical nutrient history is also of great importance to set off reference condition standards with the objective of enabling the assessment of present ecological quality against these standards.

Vertical profiles of C, N, δ^{15}N, and Pb were measured in sediment cores taken in the Mondego and Mira lower marshes (Fig. 13.1). After sectioned into 1 cm intervals, sediments were dried until constant weight, ground to a fine powder, and stored for analysis. Dates of the different layers were estimated from

^{210}Pb radioisotope analysis. A number of residences (data from the Portuguese Institute of Statistics – INE) were taken as an approximate that reflected human influence in estuarine watersheds and were related with element levels as described in Castro et al. (2007b).

13.3 Results and Discussion

The nitrogen contents of the water in Mondego and Mira estuaries seem quite high when compared with other sites. From 28 observations (Fig. 13.2), the mean DIN concentrations in Mondego and Mira place them among the highest third of these case studies. Differences in intensity of land use in their watersheds are reflected in ammonium (NH_4^+) and dissolved organic nitrogen (DON) concentrations (Table 13.1). On average, NH_4^+ and DON were, respectively, 1.8 and 2.3 times higher in Mondego, and NH_4^+ was the dominant inorganic form being 3.5 and 2.1 times higher than NO_3^- in Mondego and Mira, respectively. The relative proportion of DON in the total N present makes this variable considerably important for eutrophication studies. DON is an important bacterial substrate and source of recycled DIN; it could contain a significant bioavailable component or be remineralized to inorganic N and then be taken up by the planktonic food web and converted to biomass (Seitzinger and Sanders 1997, Seitzinger et al. 2002, Capriulo et al. 2002). [See Castro et al. (2009) for discussion of annual and seasonal fluctuations of inorganic and organic forms of nitrogen in the water column].

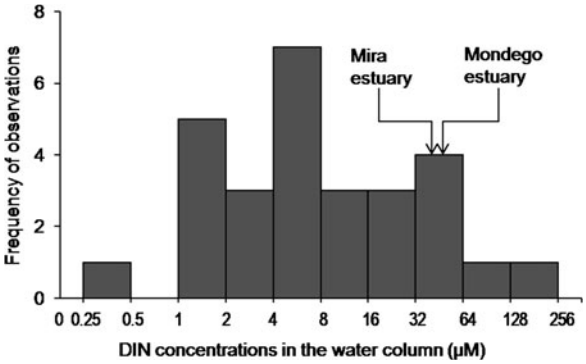

Fig. 13.2 Frequency distributions of dissolved inorganic nitrogen (DIN) in 28 aquatic systems. Literature data compiled from Pederson and Borum (1993), Fourqurean et al. (1997), McClelland and Valiela (1998), Stribling and Cornwell (2001), Cole et al. (2004, 2005), and González et al. (2008)

13.3.1 Eutrophication Status: Comparison Between Estuaries

13.3.1.1 Nitrogen and Carbon Concentrations

When comparing %N and %C values in bare sediments and in sediments collected beneath plant canopies (Table 13.2) it becomes clear the important role of vegetation in the dynamics of these nutrients in salt marsh habitats as the highest element concentrations were always observed in vegetated sediments. Rooted macrophytes are important agents in estuarine dynamics. They exhibit high rates of production (Adam 1990, Duarte and Chiscano 1999) which result in large amounts of detritus (Palomo and Niell 2009) and link the nutrients in sediments with the overlaying water influencing the degree of eutrophication (Lillebø et al. 2006, Koretsky et al. 2008, Sousa et al. 2008).

Sediment N levels in both estuaries (most in the 0.2–0.5% range) are found in other marsh sediments (Boyer et al. 2001, Tyler et al. 2003, Cabaço et al. 2008a). These values were higher when compared to developing marshes or habitats under severe N limitation (Van Lent and Verschuure 1994, Tyler et al. 2003), but were moderated compared with other studies (Craft et al. 1999, Struck et al. 2000, Voss et al. 2000), where sediments that have been historically receiving high nitrogen and organic contents have showed percentages of C and N raging from 4 to 11% and 0.3 to 1.3%, respectively. Considering %N and %C differences between estuaries, they were significant for all types of sediments (Table 13.2), with bare and *Z. noltii* sediments showing an opposite pattern when compared with *S. maritima* and *H. portulacoides* sediments. The comparisons of leaf N of the three species of plants again showed only modest differences between the two estuaries (Table 13.2) and none concerning %C. Leaf N values were consistent with published values (Table 13.3) and the range of %N (1.4–3.4 in Table 13.2) lies within the range found for global surveys (Duarte 1990). Nitrogen regimes in both estuaries were not differentiated by leaf nitrogen concentration of *Z. noltii* and *S. maritima*. High N loads and higher water residence times might be responsible for raising nitrogen availability in the Mira estuary. N content alone in seagrasses might not always clearly demonstrate significant differences in nitrogen availability, at least in these enriched estuaries. N regimes,

Table 13.2 Annual nutrient concentrations (mean ± sd) in sediments and leaves of the three rooted macrophytes collected in the Mondego and Mira estuaries

	Mondego			Mira			Differences between estuaries
Percentage of N in sediment below canopy							
Spartina maritima	0.34	±	0.08	0.26	±	0.05	$p < 0.001$
Halimione portulacoides	0.38	±	0.13	0.33	±	0.07	$p < 0.01$
Zostera noltii	0.2	±	0.02	0.23	±	0.02	$p < 0.001$
Percentage of C in sediment below canopy							
S. maritima	4.03	±	1.09	2.89	±	0.55	$p < 0.001$
H. portulacoides	4.71	±	1.9	3.49	±	0.83	$p < 0.001$
Z. noltii	2.54	±	0.27	3.13	±	0.68	$p < 0.001$
Percentage of N in leaves							
S. maritima	2.46	±	0.46	2.39	±	0.42	ns
H. portulacoides	1.78	±	0.53	1.44	±	0.3	$p < 0.001$
Z. noltii	3.36	±	0.33	3.31	±	0.42	ns
Percentage of C in leaves							
S. maritima	45.11	±	4.97	43.59	±	4.36	ns
H. portulacoides	32.73	±	5.16	33.71	±	4.65	ns
Z. noltii	41.97	±	4.64	40.82	±	4.98	ns
Percentage of N in bare sediment	0.18	±	0.02	0.19	±	0.02	$p < 0.01$
Percentage of C in bare sediment	2.32	±	0.42	2.56	±	0.54	$p < 0.01$

Table 13.3 Literature data on macrophyte (*Spartina* and *Zostera*) %N and %C in coastal lagoons and estuaries

Species	%N	%C	Site	Country	Source
S. alterniflora	1.4	–	Chesapeake Bay	USA	Stribling and Cornwell (2001)
S. alterniflora	1.8	–	Great Sippewisset	USA	Vince et al. (1981)
S. patens	1.7	–	Hog Island	USA	Windham and Ehrenfeld (2003)
S. patens	1.5	–	Great Sippewisset	USA	Vince et al. (1981)
Z. capricorni	2.1	40.5	Moreton Bay	Australia	Grice et al. (1996)
Z. marina	2	–	Øresund	Denmark	Pederson and Borum (1993)
Z. marina	2.3	36.3	Tomale Bay	USA	Fourqurean et al. (1997)
Z. marina	2.5	–	Venice Lagoon	Italy	Sfriso and Ghetti (1998)
Z. noltii	3.5	35	Plamones estuary	Spain	Pérez-Lloréns and Niell (1993)
Z. noltii	3	–	Kattendijke	Netherlands	Marbà et al. (2002)
Z. noltii	3.5	–	Cádis Bay	Spain	Brun et al. (2002)
Z. noltii	≈4	–	Ria Formosa	Portugal	Cabaço et al. (2008a)
Z. noltii	3	–	Ria Formosa	Portugal	Peralta et al. (2005)
Z. noltii	3	35	Cádis Bay	Spain	Brun et al. (2003)

however, in *H. portulacoides* were reflected in higher mean leaf N concentrations as well as in higher plant aboveground production in the Mondego estuary.

There were species-specific differences regarding N concentrations in leaf tissue (Table 13.2), but generally, these macrophytes seemed to respond positively to increases in sediment N (Fig. 13.3) except for *S. maritima* in the Mira estuary. It is well known that marsh plant growth is linked to available nutrient resources and mostly take up nitrogen from sediments together with some foliar uptake (Pedersen and Borum 1993, Touchette et al. 2003, Rubio et al. 2007).

Nitrogen concentrations in *S. maritima* and *Z. noltii* leaves were higher than the 1.8% median (Duarte

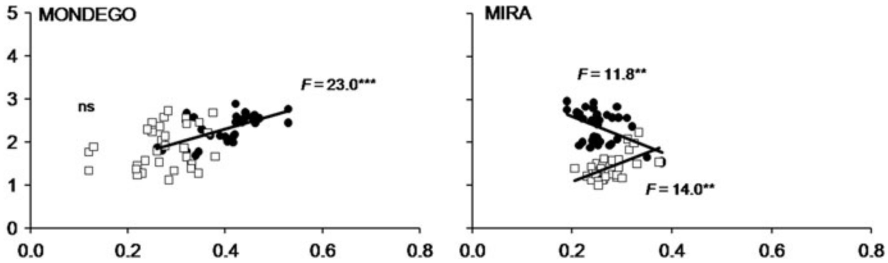

Fig. 13.3 Nitrogen concentration in leaves of *Spartina maritima*, *Halimione portulacoides*, and *Zostera noltii* compared to nitrogen concentration in sediments beneath plant canopy. $* = p < 0.05$, $** = p < 0.01$, and $*** = p < 0.001$

1990) associated with enriched growth conditions such as those we report for the Mondego and Mira estuaries. Only *H. portulacoides* showed values below this number (Table 13.2). *Z. noltii* considerably uptake more N when compared to other seagrasses (Marbà et al. 2002), probably to maintain high N content all year, plant growth, and primary production. Higher %N in *Z. noltii* could also be a result of the contribution of N fixation. Welsh et al. (1996) estimated that fixated N provides 6.3–12% of the total N need for a *Z. noltii* meadow. Nitrogen fixation rates are higher during light periods and during summer (Welsh et al. 1996, 1997) and together with leaf and sediment N uptake greatly contribute to maintain high N content in plant leaves. *H. portulacoides* is a C_3 plant with lower N uptake efficiency compared to C_4 species (Simas and Ferreira 2007).

Although there are some uncertainties and variations in the responses, sediment data and leaf N concentrations suggest a general pattern of high nitrogen regimes in Mondego and Mira and point to a somewhat higher delivery of nitrogen to Mondego. These results suggest that, at least for these highly enriched systems, nitrogen concentrations in sediment and plants were indicative, but not as sensitive indicators of level of land use as might be desirable.

13.3.1.2 Plant Aboveground Biomass

Both estuaries were very productive concerning aboveground biomass values of *S. maritima* and *H. portulacoides* and less for *Z. noltii* when compared to other seagrasses of the same genera (Table 13.4). The plant biomass data showed ambiguous evidences between Mondego and Mira. There were significantly higher biomasses of *H. portulacoides* and *Z. noltii* in the Mondego estuary and higher aboveground biomass of *S. maritima* in the Mira estuary (Table 13.4). These results may suggest that *H. portulacoides* might be nutrient limited in Mira and that other factors such as sediment stresses, water quality, water residence time, or epiphyte overgrowth on seagrass blades might decrease plant productivity in Mondego (Neckles et al. 1993, Burkholder et al. 2007, Leoni et al. 2008, Sousa et al. 2008). Plant aboveground biomass of these rooted macrophytes did not clearly demonstrate significant differences between estuaries and, thus, was not a sensitive indicator of nitrogen availability in those systems.

Aboveground biomass of seagrasses usually peaks in summer (Plus et al. 2001, Edwards and Mills 2005, Pergent-Martini et al. 2005, Trilla et al. 2009), clearly dependent on temperature, light availability and of growth and higher nitrogen mineralization rates during this period (Cartaxana et al. 1999), but a continuous growth throughout the year can also be observed in marshes where mean monthly winter temperatures do not fall to values limiting growth (Pierce 1983, Costa et al. 2001, Castro et al. 2009). Hussey and Long (1982) reported a seasonal biomass pattern for *H. portulacoides* in an England salt marsh with a maximum in summer months, but were less productive populations when compared to the Portuguese populations. *H. portulacoides* is a semi-deciduous shrub with no seasonal pattern in the amount of woody tissue (Hussey and Long 1982), and even if there was a marked seasonal pattern in leaf tissue, this was dissembled by the other tissue in Portuguese populations. Plant production can be affected by sediment stresses, particularly in warmer months, such as high porewater salinities and low sediment redox potentials (Osgood and Zieman 1993, Cartaxana et al. 1999). Light and temperature are key ambient factors as well that affect *Zostera*

Table 13.4 Data from this study and literature of aboveground biomass (mean annual values) of the macrophytes *Halimione*, *Spartina*, and *Zostera*

Species	Biomass (g DW m^{-2})	Site	Country	Source
H. portulacoides	113	Essex	England	Hussey and Long (1982)
H. portulacoides	410	Tagus estuary	Portugal	Caçador et al. (2009)
H. portulacoides	3,053	Mondego estuary	Portugal	This study
H. portulacoides	1,899	Mira estuary	Portugal	This study
S. alterniflora	1,000	Snow's Cut Marsh	USA	Craft et al. (2003)
S. alterniflora	800	Georgia Marshes	USA	Gallagher et al. (1980)
S. alterniflora	200	Georgia Marshes	USA	Gallagher et al. (1980)
S. alterniflora	500	Hog Island	USA	Edwards and Mills (2005)
S. alterniflora	200	Great Sippewisset	USA	Vince et al. (1981)
S. maritime	600	Algoa Bay	South Africa	Pierce (1983)
S. maritime	348	Tagus estuary	Portugal	Caçador et al. (2009)
S. maritime	1,085	Mondego estuary	Portugal	This study
S. maritime	1,313	Mira estuary	Portugal	This study
S. patens	300	Great Sippewisset	USA	Vince et al. (1981)
Z. marina	250	Øresund	Denmark	Pederson and Borum (1993)
Z. marina	150	Thau Lagoon (north site)	France	Laugier et al. (1999)
Z. marina	50	Thau Lagoon (south site)	France	Laugier et al. (1999)
Z. marina	450	Venice Lagoon	Italy	Sfriso and Ghetti (1998)
Z. marina	150	San Quentin Bay	Mexico	Cabello-Pasini et al. (2003)
Z. marina	50	Ojo de Liebre Lagoon	Mexico	Cabello-Pasini et al. (2003)
Z. marina	127	Øresund	Denmark	Middelboe et al. (2003)
Z. noltii	82	Arcachon Bay	France	Auby and Labourg (1996)
Z. noltii	119	Plamones estuary	Spain	Pérez-Lloréns and Niell (1993)
Z. noltii	82	Biguglia Bay	France	Pergent-Martini et al. (2005)
Z. noltii	82	Urbino Lagoon	France	Pergent-Martini et al. (2005)
Z. noltii	40	Thau Lagoon (north site)	France	Laugier et al. (1999)
Z. noltii	150	Thau Lagoon (south site)	France	Laugier et al. (1999)
Z. noltii	140	Ria Formosa	Portugal	Peralta et al. (2005)
Z. noltii	150	Thau Lagoon	France	Plus et al. (2001)
Z. noltii	400	Venice Lagoon	Italy	Sfriso and Ghetti (1998)
Z. noltii	60	Cádis Bay	Spain	Brun et al. (2003)
Z. noltii	94	Mondego estuary	Portugal	This study
Z. noltii	80	Mira estuary	Portugal	This study

production (Dennison 1987, Pérez-Lloréns and Niell 1993, Plus et al. 2001, Leoni et al. 2008). Water quality (clarity) indirectly controls light availability to plants and is reduced when turbidity levels increase (Moore et al. 1996, Pergent-Martini et al. 2005). In these two estuaries, DON and PN concentrations might have contributed to seagrass decline, especially during summer, and therefore masked estuarine difference. Excess of nutrient loading shorter eelgrass leaves (Short et al. 1995) and low water quality during summer could be factors that impede plant biomass to peak in this season. In addition, high concentrations of nitrogen in the water column, as observed for the Mondego and Mira estuaries, might also be directly toxic to *Zostera* plants (Burkholder et al. 1992, van Katwijk et al. 1997, Brun et al. 2002). The epiphytic community is a regular part of healthy seagrass systems, but nutrient enrichment has been known to induce epiphyte overgrowth on seagrass blades with consequences to plant productivity (Neckles et al. 1993, Short et al. 1995). In Mondego, epiphytes are very common in *S. maritima* (unpublished personal observations), but studies have thus far not been conducted to assess how this community might affect *Spartina* or other seagrasses.

13.3.1.3 Nitrogen Stable Isotopes

More contrast results were evidenced by the $\delta^{15}N$ ratios in sediments and plant leaves. The range of sediment $\delta^{15}N$ (2.5–7.5‰) we report in Fig. 13.4 was quite similar to the range of 3.1–7.7‰ found by Deegan and Garritt (1997). $\delta^{15}N$ between 4 and 6 were reported from Carmichael and Valiela (2005) in estuaries receiving heavier N loads and between 2 and 4 from less polluted estuaries.

More clear differences between estuaries were evidenced by nitrogen stable isotopes in plant leaves. $\delta^{15}N$ of all three species of macrophytes were heavier than $\delta^{15}N$ of unvegetated sediment (Fig. 13.4) and showed relative high values when comparing with literature for the same genera (Table 13.5). For all plant species $\delta^{15}N$ values from Mondego were heavier than those from Mira (Fig. 13.4). In addition, the $\delta^{15}N$ in plant leaves were heavier than those in the sediments directly beneath their canopies. These macrophytes evidenced a fractionation of about 3‰, with a wide range of up to 7‰ and was more than twofold larger in plants collected in Mondego compared to Mira, but not for the cordgrass.

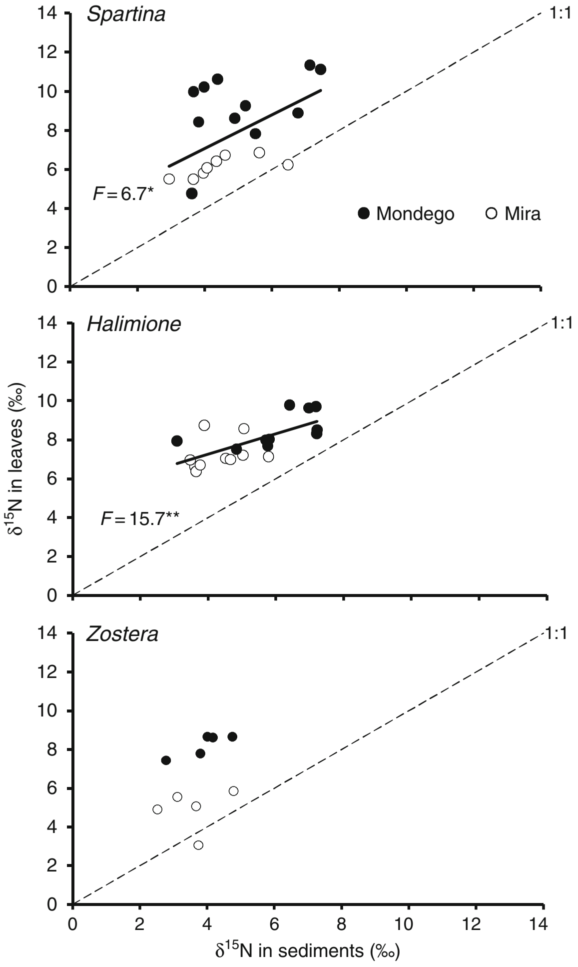

Fig. 13.4 $\delta^{15}N$ in plant leaves compared to those of sediments. $* = p < 0.05$ and $** = p < 0.01$

The differences of $\delta^{15}N$ ratios in plant leaves follow patterns observed in many other estuaries, where plant $\delta^{15}N$ values become heavier as the N loads and contribution by wastewater increase (see Section 13.1.3). Grice et al. (1996) reported higher $\delta^{15}N$ values in seagrasses from a more eutrophic site (8.6–8.8‰) when compared to a less anthropogenic influenced one (2.6–4.5‰). More recently Baeta et al. (2009) found a mean value of 9.5‰ in *Z. noltii* leaves in the Mondego estuary (8.2‰ was the mean value found in this study), which may reflect a more strong influence of wastewater nitrogen in the system. The parallel increase of $\delta^{15}N$ in leaves and sediments (Fig. 13.4) may indicate that both leaves and sediments followed and reflected the level of eutrophication despite responsiveness of $\delta^{15}N$ of sediments may be constricted by biogeochemical transformations (Brandes and Devol 1997, Altabet et al. 1999, Ohte et al. 2004). The $\delta^{15}N$ of leaves and to a less extent in sediments were therefore a far more sensitive reflector of the degree of human influence in these estuarine watersheds when compared with nutrient content or plant aboveground biomasses.

13.3.2 Historical Nutrient History

Element levels in sediment cores showed variation during the time course of burial (Fig. 13.5). From 1920 to 1960 values were very similar and therefore we decided to show the average for that period in Fig. 13.5.

All variables reflected the degree of human pressure in the Mondego and Mira watersheds. The variation of C in Mondego clearly indicated physical disturbance, consequence of several engineering works that have been carried out on the Mondego River since the 1970s (e.g., dam construction, dredging), that lead to a reduction in the supply of sediment and increased land erosion. Sediment erosion as well as burial have significant impacts on seagrass survival and on the plant allometric responses, and these processes were

Table 13.5 Literature data on $\delta^{15}N$ (mean values) from *Halimione*, *Spartina*, and *Zostera* species (leaves)

Species	$\delta^{15}N$ (‰)	Site	Country	Source
H. portulacoides	12.4	Bourgneuf Bay	France	Decottignies et al. 2007
H. portulacoides	8.5	Mondego estuary	Portugal	This study
H. portulacoides	7.2	Mira estuary	Portugal	This study
S. alterniflora	3.3	Sage Lot Pond	USA	Martinetto et al. (2006)
S. alterniflora	7.8	Quashnet River	USA	Martinetto et al. (2006)
S. alterniflora	8.2	Childs River	USA	Martinetto et al. (2006)
S. alterniflora	4.4	Childs River	USA	McClelland and Valiela (1998)
S. alterniflora	6.0	Quashnet River	USA	McClelland and Valiela (1998)
S. alterniflora	7.6	Childs River	USA	McClelland and Valiela (1998)
S. alterniflora	6.0	Sapelo Island	USA	Peterson and Howarth (1987)
S. alterniflora	6.8	Mashpee River	USA	Cole et al. (2005)
S. alterniflora	7.7	Great Pond	USA	Cole et al. (2005)
S. alterniflora	8.1	Green Pond	USA	Cole et al. (2005)
S. foliosa	10.3	Tijuana estuary	Mexico	Fry et al. (2001)
S. foliosa	10.5	San Francisco estuary	USA	Cloern et al. (2002)
S. maritima	8.8	Ria formosa	Portugal	Machás and Santos (1999)
S. maritime	9.6	Mondego estuary	Portugal	This study
S. maritime	5.9	Mira estuary	Portugal	This study
Z. capricorni	8.8	Moreton Bay	Australia	Grice et al. (1996)
Z. marina	−1.6	Sage Lot Pond	USA	McClelland and Valiela (1998)
Z. marina	0.5	Quashnet River	USA	McClelland and Valiela (1998)
Z. marina	2.0	Sage Lot Pond	USA	Martineto et al. (2006)
Z. marina	5.0	Childs River	USA	McClelland and Valiela (1998)
Z. marina	9.7	Tomale Bay	USA	Fourqurean et al. (1997)
Z. marina	14.1	Kwangyang Bay	Korea	Kang et al. (2003)
Z. noltii	4.5	Ria Formosa	Portugal	Machás and Santos (1999)
Z. noltii	8.2	Mondego estuary	Portugal	This study
Z. noltii	4.9	Mira estuary	Portugal	This study
Z. noltii	9.2	Kattendijke	The Netherlands	Marbà et al. (2002)

reviewed by Cabaço et al. (2008b). Castro et al. (2007b) pointed out that C variation could also be due to the reduction of vegetation cover in those decades as described in literature (Marques et al. 2003, Pardal et al. 2004). Long-term changes in the Mondego estuary revealed a clear reduction of 17% on the salt marsh area during the five decades analyzed by Castro and Freitas (2006). Increasing of urban areas in the Mondego River vicinities and intensive agriculture practices in upstream areas claimed permanent marsh land and contributed significantly to eutrophication. In the Mira estuary, however, vegetation cover seems to be the same since the 1950s (Castro and Freitas 2006), and changes in %C are probably due to the intensification of land use in upstream areas (mostly agricultural) provided by the construction of a dam in 1968 20 km up river (Castro et al. 2007b).

Nutrient loads over time were discernible by %N and N isotopic vertical profiles, but values of $\delta^{15}N$ began to diverge much sooner (Fig. 13.5). As early as the 1930s (see Castro et al. 2007b) there were evidently higher values in Mondego relative to Mira. Although there were near-parallel changes, the pattern consistently showed that Mondego sediments were higher in $\delta^{15}N$ reflecting human density surrounding the Mondego estuary. Higher $\delta^{15}N$ levels are commonly associated with high wastewater inputs (Savage et al. 2004), and since isotope fractionation of sediments is relatively unaffected by diagenesis (Altabet et al. 1999, Lehmanna et al. 2004), the $\delta^{15}N$ signatures in sediments most likely reflect the signatures of the sources, rather than internal transformations.

The concentration of lead, our proxy for industrial contamination, followed a pattern similar to that of $\delta^{15}N$, with Mondego having much higher concentrations in the sediment profiles (Fig. 13.5), indicating the degree of historical industrial contamination in the surrounding areas of both estuaries (Table 13.1). The trend of increased metal concentrations with time in recent decades, as observed for these two estuaries, has also been verified in other areas (Caçador et al. 1996, Marcantonio et al. 2002, Feng et al. 2004, Osher et al. 2006) reflecting the constant increase of anthropogenic contamination in coastal areas.

Relationships between time courses of variables are shown in Table 13.6. Increased carbon content and, presumably, more reduced sediments, seemed to be associated with less lead accumulations. However, no significant correlations were found. Opposite relationships were observed between temporal pattern of %C and the time course of %N and δ^{15}N. δ^{15}N increased significantly in parallel to N in sediments but the higher δ^{15}N indicated that Mondego probably received more wastewater N.

The connection between humans, expressed as the number of residences in the municipalities surrounding the estuarine area, was evident in both estuaries (Table 13.6). Generally, more people were linked to more N, δ^{15}N, and Pb. By contrast, the %C had poor relation to human pressure. There were estuary-specific factors that affected correlations between %N and the number of residences Castro et al. (2007b) which make %N a slightly less useful indicator, as it may be more sensitive to other variables. Increased concentrations of nutrients in estuaries usually are correlated to the number of human inhabitants of a riverine and watershed catchments system (Nedwell et al. 2002, Bowen and Valiela 2001, Deegan 2002), leading to disappearance of seagrass beds all over the

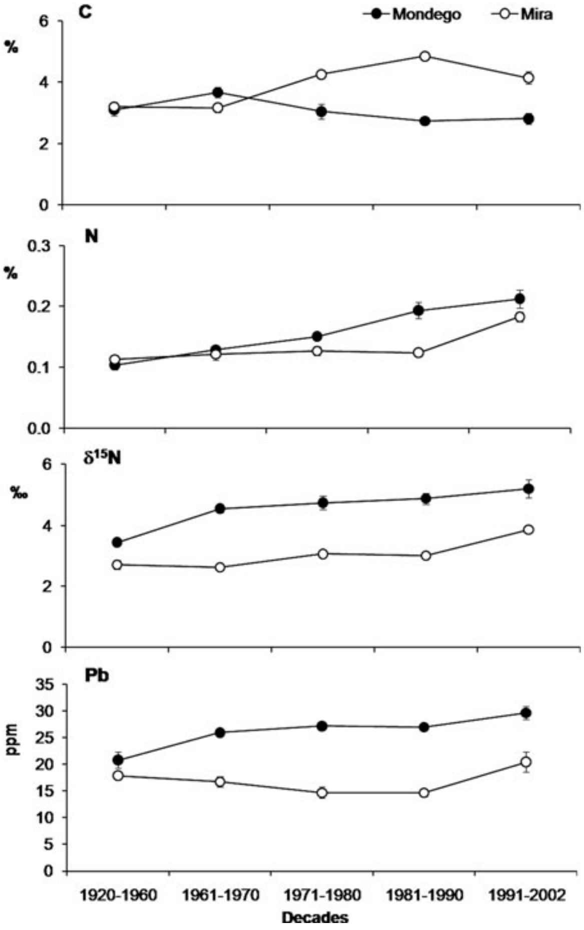

Fig. 13.5 Time courses of C, N, δ^{15}N, and Pb content (mean ± sd) of sediment from cores collected in the Mondego and Mira estuaries. (Results are shown by decade except for the first case that corresponds to the average of the first six decades)

Table 13.6 Relationships between time courses of variables and number of residences in the Mondego and Mira estuarine watersheds

Variables	δ^{15}N	C	N	Pb	Number of residences
Mondego estuary					
δ^{15}N	–	–0.432	0.954*	0.976*	0.889**
C	–	–	–0.441	–0.275	–0.348
N	–	–	–	0.950*	0.969**
Pb	–	–	–	–	0.893**
Mira estuary					
δ^{15}N	–	0.476	0.992*	0.231	0.832**
C	–	–	0.422	–0.655	0.451
N	–	–	–	0.297	0.837**
Pb	–	–	–	–	0.192

*Correlation is significant at the 0.01 level
** Correlation is significant at the 0.05 level

world (Short and Wyllie-Echeverria 1996, Milchakova 1999, Jagtap et al. 2003) and changing of dynamic of associated biota (Boström et al. 2002, Deegan et al. 2002, Verdelhos et al. 2005). These results convey the fact that land-derived N loads were associated with increased human population in watershed settlements and subsequent wastewater inputs and point out that Mondego is exposed to heavier anthropogenic influence when compared to Mira.

13.4 Concluding Remarks

Estuaries are very complex hydrodynamics bodies and the multifaceted biological, physical, and chemical interactions that occur impose some problems in predicting the extension of anthropogenic impacts. The selection of reliable indicators that accurately reflect the impact of human activities in estuaries requires careful attention and the interpretation of results should be done with caution. Estuarine- and plant-specific responses to nutrient loading are common and generalizations may not always be suitable. The results obtained from this project suggest that, at least for these enriched estuarine systems, nitrogen concentrations in sediment and in plant leaves were less sensitive indicators of the level of eutrophication. In contrast to the elemental results, N isotopic values showed significant differences between estuaries and seemed to reflect the degree of land use on the respective watersheds. This work also showed that vertical distributions of %C, %N, δ^{15}N, and Pb concentrations reflected the level of anthropogenic influence in the past decades. Recent increases in %N and again more clearly in δ^{15}N signatures showed to be linked to an increase of human pressure. δ^{15}N therefore appeared to be more sensitive sentinels of land-derived nitrogen loads, particularly of wastewater-borne nitrogen. Finding indicators of eutrophication to assess aquatic ecosystems quality is of great importance, in order to preserve and restore important functional habitats as salt marshes. Chemical analyses of water, sediments, and biota are valuable in estuarine studies, but the use of a more integrated chemical approach may be a useful complementary tool because it incorporates many of the nitrogen fluctuations in those sources.

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Chapter 14

Successful Restoration of a Shallow Lake: A Case Study Based on Bistable Theory

Martin T. Dokulil, Karl Donabaum, and Karin Pall

Abstract Eutrophication of shallow lakes provides an excellent example of alternative stable states. Clear water, macrophyte-dominated stages can alternate with turbid conditions characterized by high algal concentrations. Stable states can switch from one to the other domination through alterations of natural factors such as changes in water level and reduction in throughflow. Forward switches are often associated with anthropogenic pressure. In such cases, backward shift to the original, macrophyte-dominated stage may be difficult. Return times are often prolonged due to hysteresis as a result of resilience. The theory is exemplified with results from a shallow, urban, seepage lake 'Old Danube,' which is within the city limits of Vienna. Causes and consequences of switches between stable states including resilience and hysteresis are discussed. The remediation measures are explained and the success of the restoration is explained in detail.

Keywords Urban lake · Remidiation · Whole lake approach · Australia

14.1 Defining the Problem

Lake eutrophication is a worldwide environmental problem. The process of eutrophication and (re)-oligotrophication has manifold facets including many moderate and continuous disturbances originating in the lake, its watershed, or its airshed. In the normal dynamics of lakes, systems tend to maintain a given state through resilience against environmental perturbations (Carpenter and Cottingham 1997). Such 'resilient systems' have several mechanisms with different ecosystem components, distinctive temporal and spatial extents, and return times. These mechanisms buffer lake ecosystems against fluctuating perturbations. They maintain the reliability of ecosystem services, water quality, and productivity. Perturbations of freshwater systems are usually brief in duration, but may be extensive in space. Fluctuations of environmental variables in the catchment or lake influenced by weather conditions, variability of interacting populations, or fires that sweep through the watershed are examples. Resilience mechanisms that tend to restore the normal dynamics involve longer or larger scales, e.g., nutrient pulses which can be absorbed by food web dynamics or retained in wetlands. Extreme perturbations can destroy resilience but may also give rise to new resilience mechanisms and qualitative changes in the ecosystem.

Oligotrophic conditions are usually stable, because the growth of algae is limited by the nutrient input from the watershed and practically no return of nutrients from the sediments. Eutrophic conditions are stabilized by internal recycling of phosphorus within the lake particularly in shallow, thermally unstratified lakes. Many such lakes remain eutrophic for extended periods of time. Their persistent eutrophication can be due to many factors because the process of recovery from eutrophication is not entirely understood, can be slow, or do not succeed. Environmental fluctuations and natural or human-induced perturbations can result in big changes that occur during a relatively short period of time. These regime shifts are infrequent, modify ecosystem organization and dynamics, with prolonged

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consequences which may have large ecological and social consequences. Such regime shifts are difficult to predict in advance.

In some cases, a threshold may separate different regimes. When an ecosystem crosses a threshold it may switch from one alternative stable state to another. Therefore regime shifts are an important topic for ecosystem management (Carpenter 2003, 2005).

14.2 The Theory of Stable States – Reloaded

The existence of alternative stable states in natural ecosystems has already been hypothesized half a century ago (Dokulil and Teubner 2003). According to Lewontin (1969) a system possesses alternative stable states if it can return to one or more equilibrium after a disturbance. In several cases, compound disturbances may play a key role in changing ecosystem structure or composition of the community. Changes only occur when the severity of the disturbance exceeds the tolerance level, e.g., the 'normal' intensities a species assemblage may typically experience (Sousa 1984). Regime shifts can be caused by natural or anthropogenic perturbations and may be gradual or catastrophic (Scheffer et al. 2001, Scheffer and Carpenter 2003, Van Nes and Scheffer 2005). Two perspectives have developed to describe how communities shift from one stable state to another. One assumes a constant environment with shifts in variables such as population density, and the other anticipates changes to underlying parameters or environmental 'drivers.' Regeneration or recovery of the ecosystem is ensured by species adapting to a certain disturbance regime (Paine et al. 1998). The possibility of alternative stable states has been supported by ecological models (Holling 1973, Sutherland 1974) but their presence in the 'real' world has been much debated (Conell and Sousa 1983, Peterson 1984, Jasinski and Asselin 2004). Experimental evidence shows the alternating presence of persistent communities which are often difficult to observe directly (Petraites and Latham 1999, Bertness et al. 2002). The presence of alternative equilibria has been demonstrated for a number of aquatic and terrestrial ecosystems using a multitude of analytical and experimental techniques (Knowlton 1992, Hughes 1994, Baker and Walford 1995, Steele 1998, Hare and Mantua 2000, Van de Koppel et al. 2001, Bertness et al. 2002, Sedia and Ehrenfeld 2003, Rietkerk et al. 2004, Jasinski and Payette 2005). In freshwaters, especially when shallow, the theory of bistable states has been widely used, accepted, and modified (Scheffer 1990, 1991, 1998, Scheffer et al. 1993). Moreover, the concept has been adopted and expanded for use in lake management and restoration (Moss et al. 1996, 1997).

Two different situations can exist in lakes, as has already been pointed out by Uhlmann (1980): dense stands of submerged plants and clear water or algal blooms associated with high turbidity. Such stable states usually change gradually with changing environmental conditions in deep lakes. In shallow lakes, however, more abrupt switches from one situation to the other are common allowing alternative states at almost identical conditions (Fig. 14.1). The equilibrium trajectory is folded producing an unstable, never realized region (dotted line in Fig. 14.1) which marks the 'turning point' of the system called 'bifurcation.' Two mechanisms are involved: environmental disturbances partly absorbed by the resilience of the system (Gunderson et al. 2002), gradually modify internal structures. When the 'break-point' region is reached, a small further alteration may result in a 'catastrophic' shift to another stability domain. The folded region of the trajectory is therefore called 'catastrophic fault.' Strong perturbations can flip an ecosystem across the unstable region when disturbances are large enough to exceed resilience.

Natural ecosystems are never in equilibrium or steady state, because unpredictable changes in weather and hydrology, seasonal changes of light and temperature, and internal mechanisms in populations continuously affect the structure and function of freshwater systems. The theory of alternative stable states nevertheless is applicable if two main questions can be resolved:

- What are the reasons for the disappearance of macrophyte beds?
- Which factors are necessary to stabilize the macrophyte domination?

One reason for a forward switch from clear water and macrophyte domination to the turbid, algal-dominated state is an increase in nutrient concentration leading to enhanced growth of planktonic and epiphytic algae which in turn affects the underwater vegetation, especially when combined with an increase

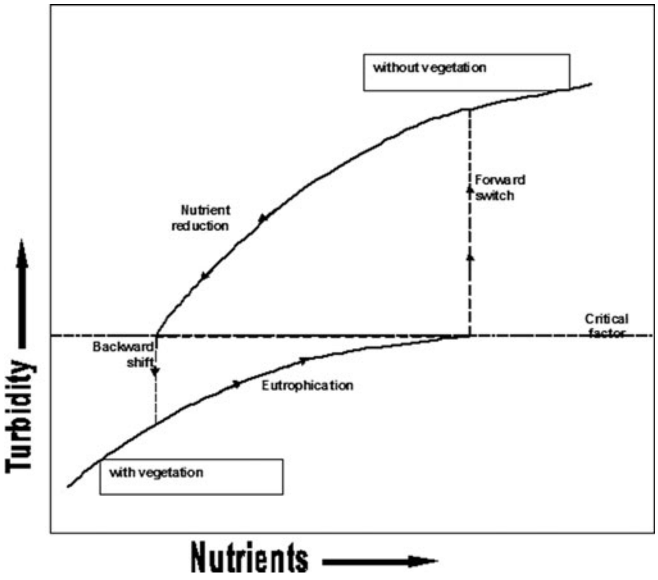

Fig. 14.1 Conceptual diagram of regime shifts (forward and backward switches) leading to alternative stable states under nutrient enrichment and decline

of algivorous and benthivorous fish species (Scheffer 1998). Massive stocking with carp and grass carp can result in abrupt changes in equilibrium conditions leading to a forward switch inducing a turbid algal-dominated situation. Grazing by waterfowl during summer, however, is considered to be of minor importance while migrating birds can damage overwintering macrophytes severely. Grazing by waterfowl may also strongly affect re-colonization by water plants. Other factors causing destruction and loss of macrophytes are disturbances by heavy storms, extreme frost at low water level, or a permanent increase in water depth.

The macrophyte-dominated stage is stabilized essentially by two mechanisms: uptake and incorporation of nutrients by macrophytes and their associated periphytic algae. Both strongly reduce nutrient availability for algae in the pelagic. Moreover, such systems have greater potentials for top-down control of the phytoplankton (Jeppesen et al. 1998).

14.3 The Study Site

Regulation of the river Danube at Vienna in 1875 resulted in the isolation of parts of the main river channel (Fig. 14.2)

The remaining backwater, now known as ‘Old Danube,’ became almost entirely dependent on groundwater seepage and precipitation because of no natural surface inflow or outflow. The lake soon developed into a famous recreational resort and has progressively been engulfed by the city. The nearby river Danube and especially the impoundment New Danube, which was built to protect the city from flood events, both influence the direction and dynamics of the groundwater. Today, the Old Danube is a shallow urban lake within the city of Vienna and a very popular recreation area (Fig. 14.1 and Table 14.1).

Before about 1990 water was clear with high Secchi disk transparencies, frequently down to the bottom. Large areas were covered with submerged macrophytes substantially influencing nutrient dynamics by their storage capacity. The abundant species were *Myriophyllum spicatum* L. and *Potamogeton pectinatus* L. The Charophytes *Nitellopsis obtusa* (DESV. IN LOIS) J. GROVES, *Chara tomentosa* L., and *Chara hispida* WOOD dominated macrophyte biomass. Over the years organic-rich sediments accumulated in several areas on top of the fluvial deposits as a result of internal processes. Parts of these sediments became anoxic because of respiration and reduced water exchange (Löffler 1988).

14.3.1 What Happened? Causes of Change

Available information on long-term changes in phytoplankton biovolume and submerged macrophyte biomass is summarized for Old Danube in Fig. 14.3, an

Fig. 14.2 Map showing the natural meandering situation of the river Danube in the mid-nineteenth century in Vienna (*upper right*) and the situation as it appears at present in 2009

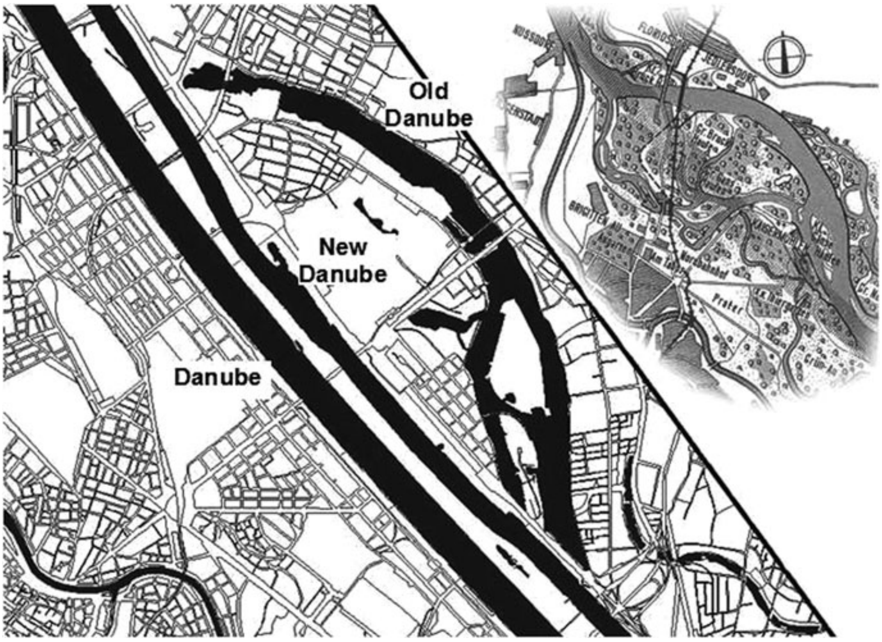

Table 14.1 Morphometric and basic chemical data for Old Danube

Altitude [m a.s.l.]	157
Area [km^2]	1.583
Volume [10^6 m^3]	3.697
Maximum depth [m]	6.8
Mean depth [m]	2.3
Average retention time [days]	190
Mixing type	Polymictic
pH	7.0–8.5
Conductivity [μS cm^{-1}]	321–608
Alkalinity [meq L^{-1}]	1.96–4.72
Ca^{++} [meq L^{-1}]	1.03–4.15
Mg^{++} [meq L^{-1}]	1.09–3.07
Cl^{-} [meq L^{-1}]	15.2–38.7

updated version from Donabaum et al. (1999). In 1987 the lake was in the clear water, macrophyte-dominated state as indicated by the high macrophyte biomass (721 tons dry weight biomass) and the low phytoplankton biovolume of 1.7 mm^3 L^{-1}. Phytoplankton composition at that time was dominated by cryptophytes (39%), dinophytes (17%), and chrysophytes (12%) while cyanobacteria comprised as little as 4%. Macrophyte biomass was mainly made up by a variety of Charophyte species almost entirely covering the bottom sediments. For the following 5 years we only have fragmented information. Certainly algal biovolume gradually increased at a moderate rate from 3 to 4 mm^3 L^{-1}. First symptoms of severe deterioration of water quality were detected during routine monitoring in the late 1980s. The filamentous cyanobacteria *Limnothrix redekei* (VAN GOOR) MEFFERT was first recorded in water quality samples in the year 1992. By early 1993 the lake had suddenly shifted to a turbid state dominated by the filamentous, cyanobacterial species *Cylindrospermopsis raciborskii* (WOLOSZ.) SEENAYYA ET SUBBA RAJU which potentially can fix atmospheric nitrogen and is also able to produce cyanotoxins harmful to men affecting also trophic interactions (Dokulil and Mayer 1996, Mayer et al. 1997). This compositional change was accompanied by a sixfold increase in annual average biovolume and a dramatic loss of macrophytes. In the years 1993 and 1994 only remnants of the previous submerged macrophyte stands and the extensive bottom cover of Charophytes were left (Dokulil and Janauer 1995).

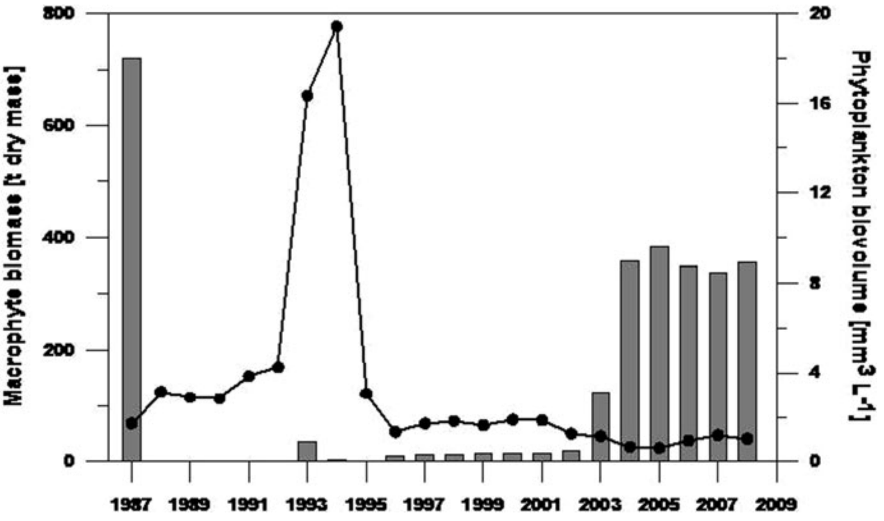

Fig. 14.3 Long-term changes of macrophyte biomass (*bars*) and phytoplankton biovolume (*line*) in Old Danube from 1987 to 2008

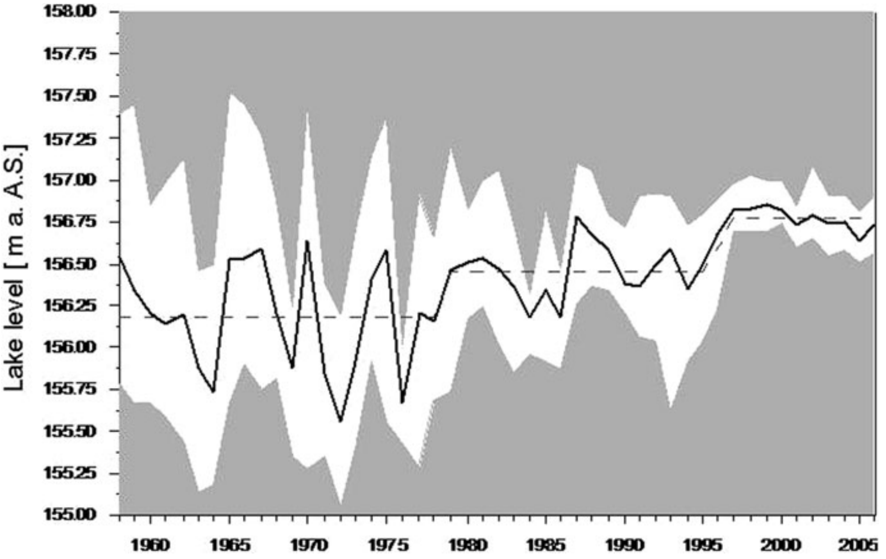

Fig. 14.4 Maximum, mean, and minimum water levels in Old Danube for the years 1958–2003. Water level is in meters above the Adriatic Sea

Reasons responsible for the rapid shift from clear water to the turbid state were believed to result from substantial nutrient flux from non-point sources such as leaking septic tanks and elution from a former dump site nearby. Nutrient input originating from the excretion of water fowl and from recreational activities was assumed to be of minor importance. Due to large stocks of benthivorous (cyprinids) and planktivorous fish in Old Danube, background turbidity was high and abundance of larger zooplankton species was low. The zooplankton was dominated by rotifers, small cladocerans, and copepods. The main trigger for the shift, however, was seen in the changes in water level dynamics in the second half of the twentieth century (Fig. 14.4.). The reduction in water level fluctuation and a permanent higher water level since the late 1970s in combination with nutrient influx were assumed to be the main reasons for the vanishing of the macrophytes similar to observations by Blindow et al. (1993).

14.3.2 How to Restore? The Concept of Remediation

As a consequence of the associated drop in water quality, the municipality asked for quick action and restoration early in 1993. As an immediate short-lived action, the lake was diluted with water of better quality which reduced algal biomass considerably in the first half year of 1994. After running a number of different pilot projects in 1994, we decided to use internal phosphorus precipitation as a restoration technique (Ripl 1976). As a whole-lake experiment, ferric chloride and calcium nitrate were successfully applied in a two-step process in spring 1995 and 1996 resulting in a drastic backward shift of all parameters. For details refer to Donabaum et al. (1999) or Donabaum et al. (2004). Secchi depth increased significantly mainly because algal biovolume declined to about 1 mm^3 L^{-1} and macrophytes started to re-colonize the system (Fig. 14.3). A period of 6 years from 1997 to 2002 and several additional measures including biomanipulation were necessary to finally re-establish macrophyte biomass. During this period phytoplankton composition changed from cyanobacterial dominance to a more mixed composition of various algal classes (Fig. 14.5.). Similarly, zooplankton biomass declined and the composition shifted from rotifer dominated to a greater preference of cyclopoida (Fig. 14.6). The main increase of macrophyte biomass occurred in 2003 and 2004 after growth has been promoted every spring through water level drawdown since 2002. This drawdown, to some extent, mimics the previous water level dynamics thereby providing better light conditions in shallow areas enhancing macrophyte growth. As a consequence, annual mean submerged macrophyte biomass has now stabilized around 350 tons dry weight (Fig. 14.3) which is less than in the 1980s. Species composition has changed and *M. spicatum* is now dominating making weed management necessary to avoid interference with swimming and boating. Therefore, the biomass stabilization is mainly a consequence of the aquatic weed control by cutting and removing aquatic weeds. At the moment, Charophytes are coming back at increased rates. The final goal therefore is to get a bottom cover of stonewort back similar to what it was previously.

Following Scheffer (1998), shifts in stable states can be visualized using conventional phosphorus–chlorophyll diagrams. The development of annual average TP versus chlorophyll-a is plotted for Old Danube in Fig. 14.7. The diagonal line in the graph separates macrophyte from algal-dominated years. The shift from one stable state to the other occurred in 1992. Trophic state dropped from hypertrophic to lower eutrophic in 1995 after the first phase of internal restoration. Mesotrophic conditions were reached in 2004. The development of chlorophyll-a versus

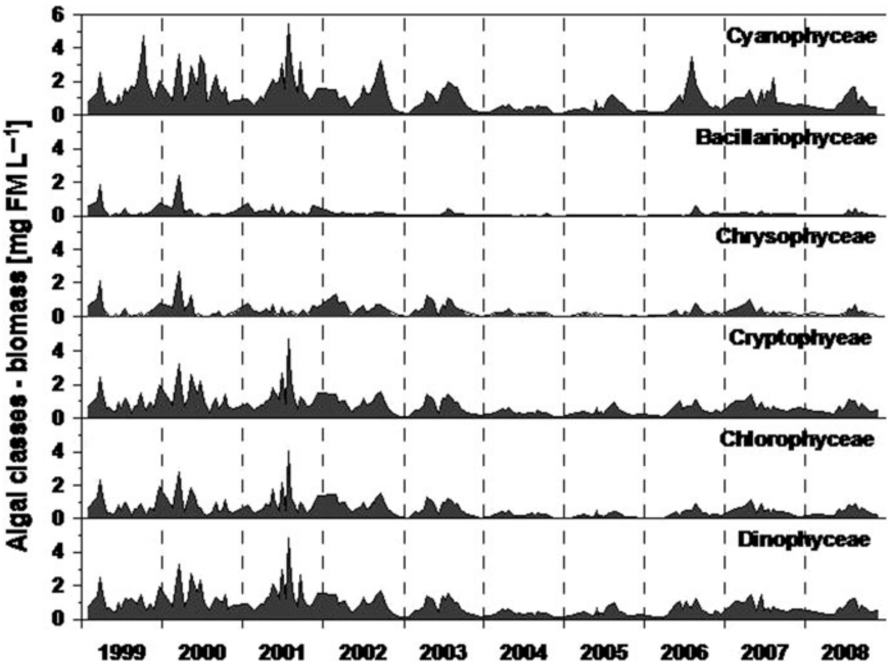

Fig. 14.5 Long-term succession of phytoplankton algal classes in Old Danube for the years 1999–2008

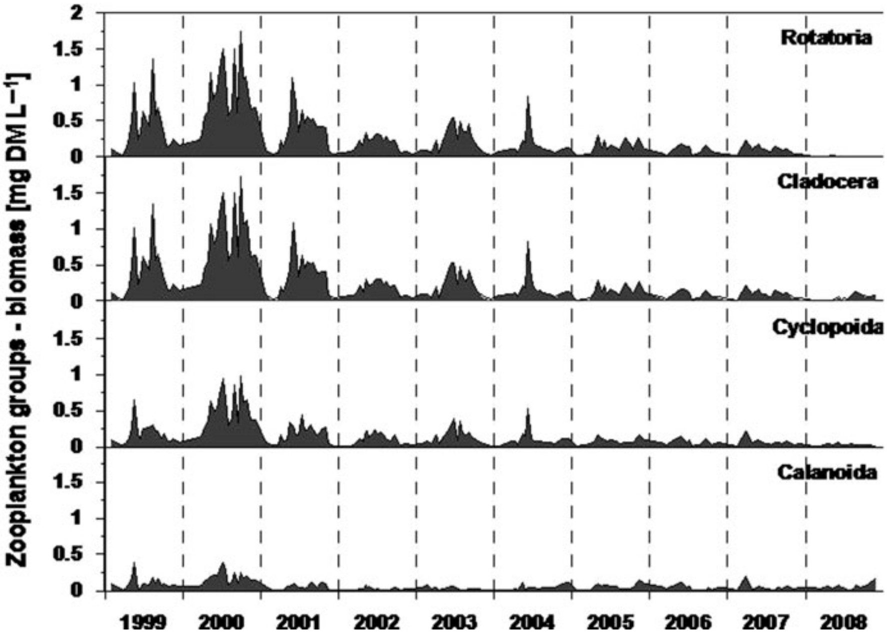

Fig. 14.6 Long-term changes of zooplankton biomass and groups in Old Danube for the years 1999–2008

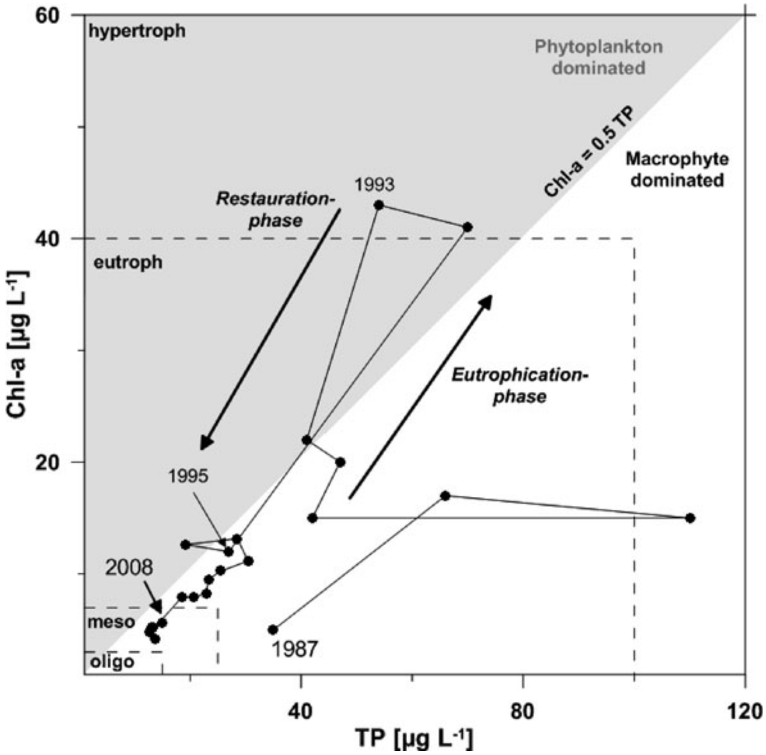

Fig. 14.7 Trophy plot of Chl-a versus TP for Old Danube for the years 1987–2008. Delineation of trophic levels follows Forsberg and Ryding (1980). Separation of phytoplankton from the macrophyte domination by Chl-a = 0.5 TP

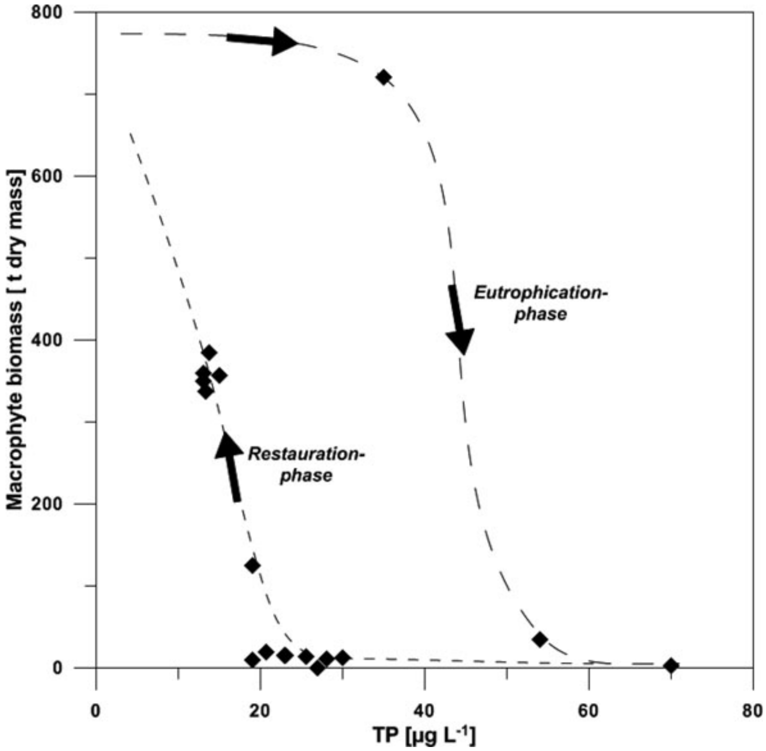

Fig. 14.8 Annual total macrophyte biomass in tons of dry matter versus annual mean total phosphorus concentration (TP) for the year 1987 and for the period 1993–2008

TP over time in this graph, however, does not show a great deal of hysteresis due to resilience of the system, most likely because of the rapid internal restoration.

In contrast to the phytoplankton, the history of macrophyte development clearly indicates considerable hysteresis in switching between stable states (Fig. 14.8). Also only limited information is available for the eutrophication phase; the switch from macrophyte dominated to the dominance of phytoplankton is clearly separated from the backward switch. Macrophyte development did not immediately respond to the reduction in total phosphorus concentrations. The resilience of the system produced strong hysteresis which to some extent probably was a result of the chemical perturbation during remediation forcing the system to switch back through technical intervention. Natural cyclic shifts between stable states have been recently observed and modeled by Van Nes et al. (2006).

14.4 Conclusions from a Successful Story

Alternative stable states and regime shifts can occur in shallow lakes as a result of natural or anthropogenic forcing. Such stable equilibria can occur within the same water body at various locations, at different times of the year, or during trophic development (Scheffer et al. 1994). The theory of alternative stable states and regime shifts can be applied successfully to the recovery of urban lakes as a consequence of ecohydrological changes (Hosper 1998). When systems are forced back to the original, macrophyte-dominated stage through, e.g., internal technical intervention resilience produces hysteresis and consequently return times become longer, particularly for macrophytes. Here we have shown that state transitions from one stable state to another were associated with significant changes in species composition of the phytoplankton assemblage. Shifts in either direction immediately altered total biovolume and algal composition without hysteresis. In contrast, considerable hysteresis occurred during macrophyte recovery making additional remediation measures necessary. Regime shifts induced by internal restoration techniques may require many years of monitoring for improvement and additional efforts to return to the previous stable state. Summarizing the experiences of the remediation project Old Danube, the theory of alternative stable states can be applied successfully to the restoration and recovery of shallow urban lakes.

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Chapter 15

Biomanipulation in Lake Årungen, Norway: A Tool for Biological Control

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Abstract This chapter deals with whole-lake experimentation for the restoration of the water quality and reduction of pollutants in fish from the eutrophic Lake Årungen, Norway, through biomanipulation of the top predator fish, the northern pike (*Esox lucius*). The manipulation was carried out during 2004–2006, with a main effort to remove large cannibal pike (>65 cm), and the cascading impacts on the major fish community were evaluated. The main fish species in Lake Årungen are roach (*Rutilus rutilus*), perch (*Perca fluviatilis*), pike, and rudd (*Scardinius erythrophthalmus*) with some additional fish species in low numbers. The main assumptions behind the removal of large pike were to obtain (i) increased number of small pike and large perch due to reduced cannibalism and predation pressure; (ii) reduced abundance of roach and small perch due to increased number of piscivores, i.e., small pike and large perch, and (iii) reduced levels of pollutants in the remaining fish community by increased growth rates and changes in diet. The recruitment success of pike after removal of large pike was evaluated by the Petersen mark–recapture method. The habitat utilization of the major fish species was studied based on gillnet catch data. The trophic position of different fish species was determined based on both gut content analyses and analyses of stable isotopes of nitrogen ($^{15}N/^{14}N{=}\delta^{15}N$) and carbon ($^{13}C/^{12}C{=}\delta^{13}C$). Important biological factors influencing the dynamics of mercury and organic pollutants through the fish community in the lake were identified as well. The estimated numbers of small pike (45–64.9 cm, mostly age class 3+) significantly increased in 2005–2006, after the experimental removal of the large pike in 2004 and 2005, and thus assumed to be a cannibal-driven response. The dominant prey fish of pike was roach, both in terms of numbers and biomass. The relative abundance of piscivores increased and as a consequence the relative abundance of small roach declined in 2005–2006. A marked shift in habitat utilization of roach and perch was recorded after the manipulation of the pike population. The gut content analysis was in agreement with the food web analysis based on the stable isotopes of nitrogen and carbon in 2005–2006. Pike (exclusively piscivores) and perch (feeding mainly on zooplankton, macroinvertebrates, and fish) occupied the top trophic positions in Lake Årungen, with pike at the highest position. Roach and rudd occupied lower trophic positions, with rudd at the lowest. Although there was segregation in diet between roach and perch, the increased importance of zooplankton in their diet during the summer 2006 may indicate an increased abundance of zooplankton. Mercury content in fish muscle increased with size and weight (bioaccumulation), as well as with an increased position in the food web (biomagnification), with pike having the highest concentrations. Growth rate of pike in age classes 1–3 increased in 2005, probably due to the lower predation risk, and thereby a more extensive use of the most favorable pike habitats. The faster growth rate might have played a role for a significant decrease in total mercury concentrations, probably as a result of growth biodilution (~50% reduction between same size groups of pike). None of the fish analyzed in 2005 had a mercury concentration which would have exceeded the FAO/WHO *provisional tolerable weekly intake* value (1.6 μg kg^{-1}), whereas the mercury concentration in only four

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analyzed pike in 2003 would have exceeded the recommended limit. The metabolite *pp*'DDT formed the main part (88%) of the total concentrations of the DDTs (dichlorodiphenyltrichloroethane) in fish liver. Both ΣDDT and ΣPCB (polychlorinated biphenyls) were significantly higher in pike compared to perch and roach, whereas HCB (hexachlorobenzene) and HCE (heptachlor epoxide) were detected mainly in pike. In pike, males had significantly higher concentrations of all organochlorine compounds compared to females. Such a sex-dependent differentiation in the concentrations of organochlorine compounds may be attributed to physiological differences, for example, a higher elimination of the lipophilic pollutants in roe during spawning by female pike and higher growth rate of female pike. The lower levels of organochlorines in pike and perch in 2005 compared to analyses performed in 2004 might be an effect of the biomanipulation of Lake Årungen as well. Therefore, selective exploitation of a large top predator fish, e.g., large pike, may serve as a tool to reduce the concentrations of mercury and organochlorine compounds in the remaining fish community. To obtain a sustained positive effect both on the trophic status of the lake and on the pollutant levels in the fishes, as obtained by the manipulation of the large pike in 2004–2006, a high fishing pressure on the large pike has to be persistent and carried out annually.

Keywords Biomanipulation · Biological control · Fish · Selective exploitation · Pollutant reductions · Lake Årungen

15.1 Introduction

Eutrophication may be defined as a condition of excessive growth of phytoplankton leading to algal bloom due to nutrient over-enrichment in a water body causing anoxic condition (Carpenter 2005). This condition causes changes in chemical, physical, biological, and ecosystem characteristics in those water bodies. Although it is a natural process, it has been accelerated by anthropogenic activities mainly due to increased population and changing economic activities in the present world by increasing the nutrient input in water bodies (Khan and Ansari 2005). Nutrient enrichment is one of the vital processes for enhancing eutrophication in lakes. Experiments in different lake environments have indicated that phosphorus is the vital nutrient for enhancing eutrophication. Phosphorus control measures may be expected as rapid abatement of eutrophication (Schindler 1974, Carpenter 2008). Success of lake restoration depends on the collaborations between scientists and managers (Carpenter and Lathrop 1999). Different control measures have been proposed for the restoration of lakes from eutrophic conditions by many researchers. Removal of algal blooms, development of macrophyte-dominated states, introduction of phytoplanktivorous fish are few of the biological remediation practiced throughout the globe (Khan and Ansari 2005). Biological methods have gained popularity in recent decades. It basically includes the conversion of turbid water state of shallow lake waters into a clear water state mainly by increasing the macrophyte growth in biomanipulated lakes (Tátrai et al. 2009).

Fish manipulation is one of the easiest ways of restoration process compared to the manipulation of nutrients, phytoplankton, and zooplankton in a lake (Lammens 1999). The mass removal of fish together with conventional pollution control methods involving the local people in controlling the non-point nutrient loading is much more successful and inexpensive compared to the expensive chemical and technical methods such as chemical treatment and dredging of the profundal sediment (Kairesalo et al. 1999). In addition, nutrient loadings from the watershed, climate, and hydrology should be taken into consideration while doing fish manipulation for the lake restoration (Sosnovsky and Quirós 2009). This chapter deals mainly with the fish manipulation and its cascading effects on fish community and pollution levels of mercury and persistent organic pollutants in fish flesh.

15.1.1 Why Lake Biomanipulation?

Until the 1980s, bottom-up effects in the food chain, particularly the phosphates were regarded as the major factor in the process of eutrophication (Langeland 1990, Lammens 2001). Although the eutrophication of lakes is generally considered as biological effects as a function of their nutrient status (Harper 1992, Scheffer et al. 1993), many other factors including biological activities (e.g., *Daphnia* grazing, fish activities) have significant impacts on primary production in a lake (Brabrand et al. 1990, Cooke et al. 1993). The elimination of excess algae by nutrient diversion in lakes, as a conventional method, has been successful in several lakes (e.g., USA, Sweden), but characterized by high costs and low applicability (Shapiro and Wright 1984). Biomanipulation is an alternative approach proposed for lake management and aimed at manipulating the lake trophic structure (Shapiro and Wright 1984). The main assumption behind such manipulation of shallow eutrophic lakes is a substantial reduction of the planktivorous fish stocks, producing a trophic cascade (Carpenter et al. 1985, Wysujack and Mehner 2002) leading to an increase in macrozooplankton and a subsequent reduction in biomass and production of phytoplankton (Langeland 1982, Langeland and Reinertsen 1982, Shapiro and Wright 1984, Sanni and Wærvågen 1990, Hansson et al. 1998, Søndergaard et al. 2000). Generally, shallow lakes can have two alternative stable states over a range of nutrient concentrations: a clear state with dominance of aquatic vegetation and a turbid (eutrophic) state having high algal biomass (Scheffer et al. 1993). In addition, there can be an intermediate stable state where clear (above vegetation beds) and turbid (deeper parts) water coexist for a number of years on route to full recovery, as a result of resistance of the turbid state (Ibelings et al. 2007). A eutrophic lake can be converted into a clear water state by reducing the nutrient load (Van Nes et al. 2007); however, such a process may have little impact on lake restoration (Benndorf et al. 2002). Biomanipulation, although regarded as one of the controversial applications of ecological theory, has been suggested as one of the management practices to cope with the eutrophication of lakes (Langeland 1990, Hansson et al. 1998, Persson and Hansson 1999, Benndorf et al. 2002, Skov et al. 2002). There can be many different approaches (efforts and methods) in such whole-lake biomanipulations; however, the main processes include either the removal of planktivorous fish or addition of piscivorous fish or a combination of both (Shapiro and Wright 1984, Hansson et al. 1998, Wysujack and Mehner 2002). Biomanipulation should be regarded as a triggering force that may produce other secondary cascading effects on the food web (Hansson et al. 1998, Benndorf et al. 2002). The main goal of biomanipulation is therefore to obtain a change in the aquatic ecosystem, not only change in the fish community (Lammens 2001). The success of biomanipulation in lakes is a combination of fish management and a considerable reduction in external phosphorus load to the lakes (Langeland 1990, Reinertsen et al. 1990, Cooke et al. 1993). Food web manipulations (biomanipulation), after an initial reduction of nutrient loads, thus seem to bring a lake back to a more stable clear state (Scheffer et al. 1993, Phillips et al. 1999, Søndergaard et al. 2000, Benndorf et al. 2002).

The manipulations of planktivorous fish stocks started in Norway around 1978 to study trophic interactions, with an aim to improve the water quality (Langeland 1990, Lammens 2001). Biomanipulation experiments conducted in shallow eutrophic lakes (max depth <20 m) in Norway until 1990 concluded that a substantial reduction in algal biomass is possible through removal of planktivorous fish biomass (Langeland 1990, Lyche et al. 1990, Sanni and Wærvågen 1990). In Norway, rotenone was used for the biomanipulation, with complete removal of fish from Lake Asklundvatn, Lake Haugatjern, Lake Helgetjern, and Lake Vikvatn (Lyche et al. 1990, Reinertsen et al. 1990, Lammens 2001), whereas piscivores (pike-perch *Stizostedion lucioperca*) were stocked in Lake Gjersjøen (Lyche et al. 1990).

15.1.2 Increased Piscivory: A Target of Biomanipulation

The major part of young age classes and small fish in a lake may be consumed by piscivorous fish (Popova 1978, Tonn et al. 1992, Craig 1996, Paukert and Willis 2003, Svenning and Borgstrøm 2005). Some piscivores, e.g., pike (*Esox lucius*), show cannibalism and may consume a high number of its own young ones (Le Cren 1965, Kipling 1983, Giles et al. 1986, Bry et al. 1992, Grønkjær et al. 2004) and play an important role as top predators in the aquatic biotic community (Miller and Senanan 2003, Paukert and Willis 2003). Although the pike population structure is affected by environmental factors (Hassler 1970, Kipling 1984, Margenau et al. 1998), an extensive reduction (or exploitation) of large pike may result in increased year class strength (Casselman and Harvey 1975, Mosindy et al. 1987). Northern pike is one of the most thoroughly studied fish species (Craig 1996); however, the cascading effects of exploitation of large pike in eutrophic lakes is a less studied aspect.

15.1.3 Prey Fish Behavior: A Role of Piscivory

The patterns of habitat utilization may be species specific as a response to habitat availability in a lake (Brosse et al. 2007). However, the activities and spatial distribution of freshwater fishes may be governed by a complex of biotic conditions such as food availability (Keast and Harker 1977, Persson 1983b, Persson 1987, Vinni et al. 2000), intra- and inter-specific food competition (Wang and Appenzeller 1998, Persson and Hansson 1999), as well as predation pressure/risk (Brabrand and Faafeng 1993, Eklöv 1997, Haertel et al. 2002, Paukert and Willis 2003, Jacobsen et al. 2004, Vainikka et al. 2005). Thus, food availability and safety are two major biological factors directly affecting fish distribution. Roach (*Rutilus rutilus* L.) and perch (*Perca fluviatilis* L.) are common co-occurring fish species in Scandinavia, with increasing dominance of roach in eutrophic lakes (Persson 1983a, Bjerkeng et al. 1991, Mehner et al. 2005) due to its ability to feed on dead organic matters (Persson 1983a, Persson and Greenberg 1990). Both roach and perch exploit the littoral habitat to a great extent (Lewin et al. 2004, Jeppesen et al. 2005), and they have the ability to coexist at relatively high densities by niche separation, both in the spatial dimension and in the food utilization (Horppila et al. 2000, Kahl and Radke 2006). Perch have a potential to undergo size-related ontogenetic niche shifts in diet and starts piscivory when becoming large (Johansson and Persson 1986, Hjelm et al. 2000, Amundsen et al. 2003). However, roach does not show such a size-related shift in diet due to its omnivorous nature (Lehtiniemi et al. 2007). Piscivory controls habitat utilization of smaller individuals of many prey fish species (Werner and Gilliam 1984, Persson 1986, Brabrand and Faafeng 1993, Paukert and Willis 2003, Romare and Hansson 2003, Schulze et al. 2006). Thereby, a shift towards greater dominance of piscivores can lead to a predator-mediated control of zooplanktivorous cyprinids (Jeppesen et al. 2005), and such a decline in prey fish abundance can change the overall fish composition (He and Kitchell 1990). Roach, an omnivorous cyprinid (Vinni et al. 2000, Lehtiniemi et al. 2007), try to avoid its potential predator in lakes (Schulze et al. 2006), and such avoidance can lead to a shift in the diet toward increased consumption of less nutritious food, such as detritus (L'Abée-Lund and Vøllestad 1987, Hölker et al. 2002).

15.1.4 Effects of Biomanipulation on Pollutants

15.1.4.1 Mercury

Global anthropogenic activities and long-range transfer of mercury (Hg) through the atmosphere are responsible for the elevated mercury concentrations in the environment (Johnels et al. 1967, Fjeld et al. 1994, MacDonald et al. 2000, Pacyna et al. 2006, Swain et al. 2007). Mercury is accumulated in organisms, including fish, in the form of neurotoxic methylmercury (MeHg) (Clarkson 1993, Watras et al. 1998, Gustin et al. 2005, Kidd 2005). Exposure to high mercury levels can damage many organs (e.g., the nervous and immune systems, heart, kidney, and lungs) of people (FAO/WHO 2006). The bioaccumulated MeHg in organism biomagnifies through the food web (Cabana et al. 1994, McIntyre and Beauchamp 2007), and in freshwaters, maximum values are found in top predatory fish species (Kidd et al. 1995a, Bowles et al. 2001, Rognerud et al. 2002, Lockhart et al. 2005). However, growth biodilution of Hg has been reported in some fast growing fish species (Kidd et al. 1999, Cizdziel et al. 2002, Campbell et al. 2003). In addition, biological factors, especially food web structure is considered important in explaining variations in Hg concentrations in and between organisms (Rask and Metsälä 1991, Vander Zanden and Rasmussen 1996, Gorski et al. 2003, Simoneau et al. 2005). The carbon isotope signatures ($^{13}C/^{12}C=\delta^{13}C$) have been used to determine the sources and flow of carbon through a food web; and the nitrogen isotope signatures ($^{15}N/^{14}N=\delta^{15}N$) have been used to determine the relative trophic position (Cabana and Rasmussen 1994, Campbell et al. 2006) and to quantify the rate of biomagnification through the food web and the uptake rate within a species (Kidd 1998, Campbell et al. 2006).

15.1.4.2 Persistent Organic Pollutants (POPs)

Organochlorine compounds (OCs) such as polychlorinated biphenyls (PCBs), hexachlorobenzene (HCB), heptachlor epoxide (HCE), and dichlorodiphenyltrichloroethane (DDT) are persistent organic pollutants (POPs) with a potential threat to organisms due to their lipophilic character and persistence in nature (Holden 1966, de Voogt and Brinkman 1989, Burreau et al. 2004). These chemical compounds have various adverse toxicological effects on the human body (Longnecker et al. 1997, Alexander et al. 2007), and due to their volatile and semi-volatile properties they have a global distribution, including pristine waters of the Arctic and Antarctic polar areas where their enrichment has been caused due to cold condensation effects (Wania and Mackay 1993, Allen-Gil et al. 1997, Rosseland et al. 1999, Grimalt et al. 2001, Vilanova et al. 2001, Weber and Goerke 2003). Fishes are exposed to these compounds through gill membranes as well as via food uptake (Holden 1966, Campbell et al. 2000, Burreau et al. 2004, Vives et al. 2005). Generally, these organochlorine compounds increase in tissue concentration with increased age and size of fish (bioconcentration and bioaccumulation) (Rognerud et al. 2002, Burreau et al. 2004, Alexander et al. 2007). In addition, some of these compounds have been found to biomagnify (Kidd et al. 1995b, Kidd et al. 1998, Rognerud et al. 2002).

15.1.5 Aims and Objectives

Biomanipulation of Lake Årungen, situated in the county of Akershus, southeast Norway, has been initiated by Ås municipality to increase lake transparency by controlling algal blooms. To accomplish this goal, exploitation of large pike has been initiated in 2004 to increase the proportions of large piscivorous perch and small-sized pike, thus increasing predation pressure on roach. Estimates of the pike population (mainly the mature fraction) were carried out both before and during the experimental exploitation of large pike to study the change in the population structure of pike due to an extensive removal of large pike (>65 cm). A reliable method for age validation and backcalculation of length at age in pike was needed in a study of pike population dynamics. Since the metapterygoid bone had been proposed by Filipsson (1972) and Appelberg (2000) for ageing of pike, a validation study of the applicability of this bone was initiated. Furthermore, the hypothesis that a change in the population structure of a top piscivorous fish can generate cascading effects on the density, habitat use, and diets of its prey fishes, mainly roach and perch, was studied simultaneously. Due to increasing concern about the environmental status of Lake Årungen, we investigated the concentrations of total mercury (THg) and organochlorine compounds in the major fish species in the lake. The main hypotheses for the study of mercury in the fish community were that changes in feeding and habitat use may influence patterns of mercury transfer through the fish community, and also change growth rates of small pike due to the removal of larger pike, resulting in changes in THg concentrations in both pike and its prey fishes. Although no individual data on organochlorine compounds from the fish community of Lake Årungen were available before the biomanipulation, the composite samples of each pike ($n=10$) and perch ($n=10$) analyzed in 2004 by Haarstad and Borch (2004) are used as the reference organochlorine levels before exploitation of large pike. In addition, stable isotopes of nitrogen and carbon were used in 2005 to get a better insight into the transfer pattern of organochlorine compounds within a fish community. Including a study of organochlorine pollutants, together with mercury, the study also gives a more complete overview of the risk connected to human consumption of fish from a eutrophic lake such as Lake Årungen. The main objectives of the study can be summarized as follows:

- Study the change in the population dynamics of pike as a result of selective exploitation of large pike in Lake Årungen
- Study the change in density, habitat use, and diet of roach and perch after the selective exploitation of pike
- Quantify and evaluate the concentrations of mercury and organochlorine compounds and their transfer patterns through the fish community of Lake Årungen

15.1.6 Study Area

Lake Årungen (area: 1.2 km^2; max. depth: 13.2 m) is situated 33 m a.s.l. in Akershus county, southeast Norway (Fig. 15.1). The lake has a length of about 3 km, with an average depth of 8.1 m (Skogheim and Abrahamsen 1980). The 52 km^2 catchment area is dominated by agriculture (51.5%) and forest (39.8%) areas (Eie and Borgstrøm 1981). Five main tributaries feed the Lake Årungen: Vollebekken, Fosterudbekken, Storgrava, Bølstadbekken, and Østbybekken. The

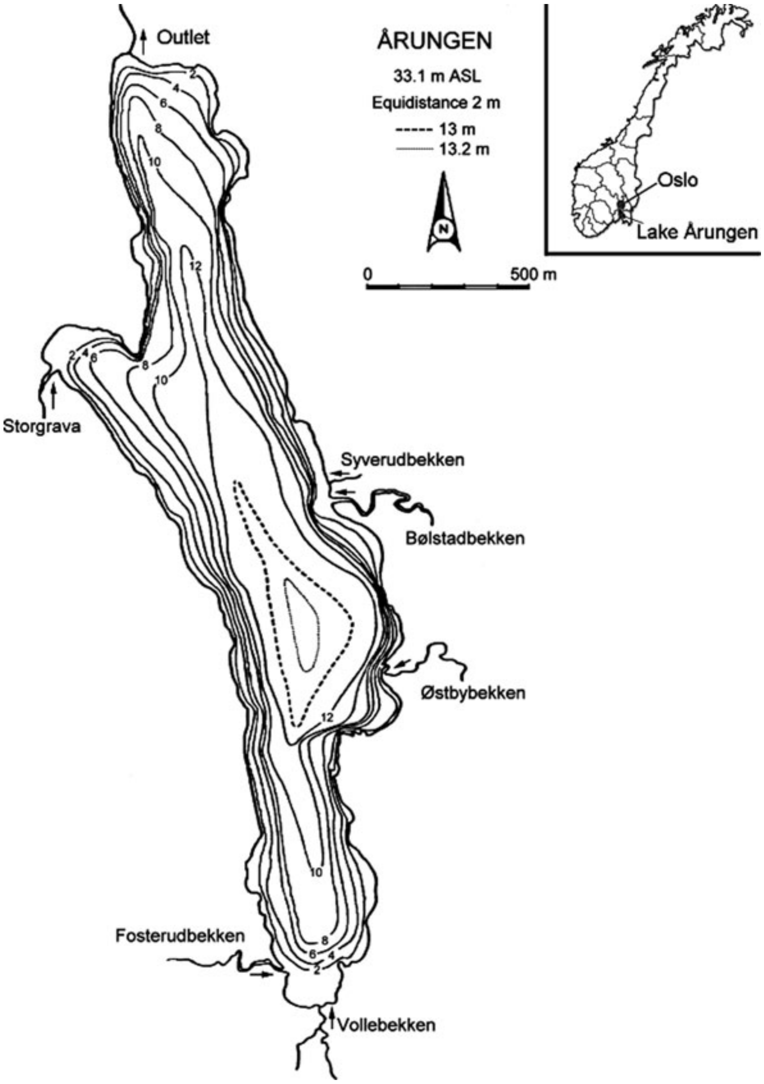

Fig. 15.1 Bathymetric map of Lake Årungen (redrawn after Skogheim and Abrahamsen 1980). Lake Årungen is situated in the southeast Norway

outlet stream from the northern part of Lake Årungen enters the Oslo Fjord.

In addition to pike, perch, and roach, the recorded fish species in Lake Årungen are rudd (*S. erythrophthalmus*), Crucian carp (*Carassius carassius*), tench (*Tinca tinca*), and European eel (*Anguilla anguilla*). From the 2 km long outlet stream a few sea trout (*Salmo trutta*) and Atlantic salmon (*Salmo salar*) ascend the lake during high discharge in the autumn. There are no point sources of mercury and organic pollutants to the lake at present. However, a part of the catchment enters Lake Årungen through the Bølstad stream. This stream receives 31,000 m^3 drainage water each year from Bølstad landfill site which was in operation during 1962–1994. Although none of the pesticides analyzed in the fish samples has been reported used since 1993, 11 different pesticides of 14 previously used were detected in 2004 (Ludvigsen and Lode 2005). Now, the pollution to the lake originates mainly from agricultural runoff, municipal waste discharges, and long range transported air pollution. Total phosphorus (TP) concentration in Lake Årungen has decreased considerably in a period from 1976 to 1985 due to changes in the agricultural practices, with less autumn ploughing (Yri 2003). Such a practice helped to stabilize the TP concentration and maintained it below 55 μg TP L^{-1} (Yri 2003). Although blooms of blue green algae together with deposition

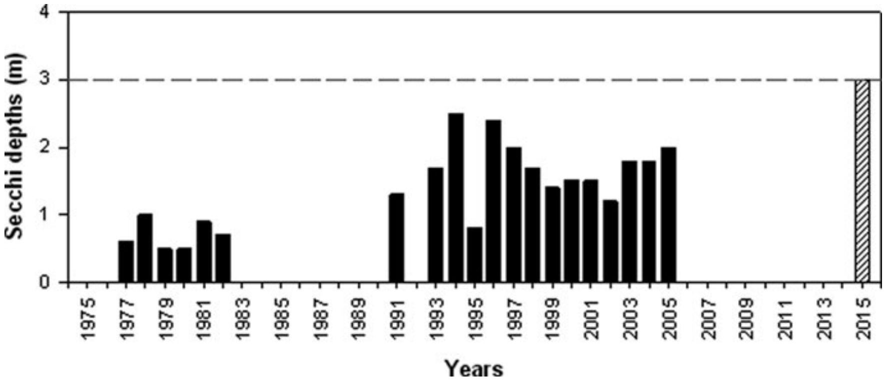

Fig. 15.2 Secchi disc transparency in Lake Årungen during summer starting from the year 1977. Year 2015 indicates a goal to obtain a transparency of 3 m (Borch et al. 2007)

of marine clay from the inlet streams make the lake transparency below 2 m during the summer, there was a considerable improvement in the Secchi disc readings during the 1990s compared to 1980s (Fig. 15.2). Pronounced oxygen depletion occurs in the deeper strata of the lake both during summer and winter (Eie and Borgstrøm 1981, Gunnhild Riise 2007, personal communication).

15.2 Materials and Methods

15.2.1 Exploitation of Large Pike and Its Population Recruitment

The pike population in Lake Årungen has frequently been heavily exploited, for example, in 1978–1980, by fyke nets and also by forks during the spawning period. Similarly, commercial fishermen captured more than 1 ton of large pike by gillnetting in 1991–1992. However, reliable statistics from these exploitation events are lacking. The recent experimental exploitation (extensive fishing) in Lake Årungen was initiated in 2004 by using gillnets with large mesh sizes (45–58 mm knot to knot) with a target to remove large-sized cannibal pike (>65 cm), but with largest efforts with mesh sizes of 52–58 cm. Before the manipulation of the pike population, large pike up to 100 cm was frequent; however, the number of large pike declined significantly in 2005 and 2006, after the beginning of manipulation in 2004 (Fig. 15.3). Metapterygoid bones were used to determine the age of pike (Sharma and Borgstrøm 2007).

Simple Peterson mark–recapture experiments were performed to estimate the number of pike (Ricker 1975). Numbered Floy anchor tags were used at the base of the dorsal fin to mark pike during the spawning period (April–May). Recapture of pike was performed

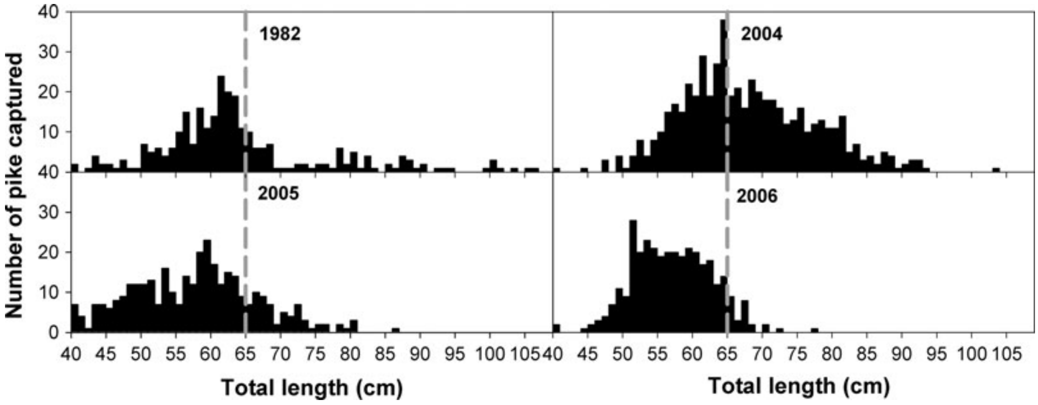

Fig. 15.3 Length frequency distributions of pike in gillnet catches in May 1982 and 2005–2006 in Lake Årungen (Sharma and Borgstrøm 2008a)

during an extensive size selective harvest of large pike ($\geq$ 65 cm) in the years 2004–2006 from the middle of May to beginning of June. Chapman's modified Petersen formula was used to estimate the number of pike (Ricker 1975) as follows:

$$\widehat{N} = \left[\frac{(M+1)(C+1)}{(R+1)}\right]$$

where $\hat{N}$ is the population estimate, M the number of marked pike in population, C the number of individuals captured for control, and R the number of recaptures (see Sharma and Borgstrøm 2008b, for details).

15.2.2 Relative Abundance and Habitat Use of Perch and Roach

Gillnet samplings were performed to find the relative abundance of roach, perch, and pike and to study the winter and summer habitat use of the species as shown in Fig. 15.4. Monofilament gillnet fleets consisting of 25 m long gillnets with mesh sizes 16, 19.5, and 29 mm were used in littoral (1.5 m high), sub-littoral (4–6 m high), and pelagic/profundal (6 m high) habitats. The gillnet fleet was set 8, 10, and 5 times in 2004, 2005, and 2006, respectively. In 1982, the efforts varied in littoral (16 mm: 9, 19.5 mm: 10, and 29 mm: 11 settings) and pelagic habitats (16 mm: 3, 19.5 mm: 12, and 29 mm: 6 settings). Catch per unit effort (CPUE) was calculated as number of fish per 100 m^2 gillnet fleet area per hour, and proportional relative abundances in different years were analyzed by chi-square tests.

In the littoral, gillnets were set at the bottom down to the depth of 4–6 m; in sub-littoral, nets were set at the bottom down to the depth of 6–8 m; and in the pelagic/profundal, the nets were set between the surface and the 12 m depth (Eie and Borgstrøm 1981, Sharma and Borgstrøm 2008a). The catches from the littoral and sub-littoral habitats were merged together due to the trough-like shape of the lake and short distance from the shore to the depths around 8 m (Fig. 15.1). Monthly samplings from December 1978 to July 1980 as well as from December 2005 to July 2006 were used for the habitat use analysis. During the period of ice cover (December–April), sampling lasted for 3–5 h, whereas during May–July, the sampling lasted only for 0.5–2 h. A total of 7,103 roach and 1,106 perch during 1978–1980 and 857 roach and 243 perch during 2005–2006 were caught for the analysis.

Statistical analyses were performed by SigmaStat 3.0.1 statistical software and the figures were prepared by Sigmaplot 9.0 software. The relative abundances of perch and roach in different habitats were analyzed by chi-square test. The difference in habitat utilization between perch and roach was tested by Kruskall–Wallis test. All pairwise comparisons were

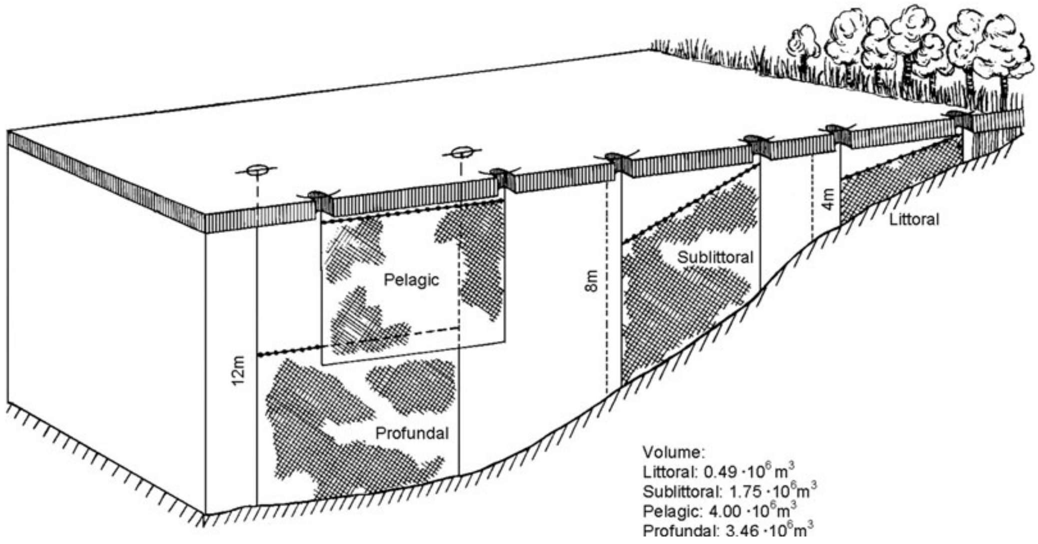

Fig. 15.4 Diagram of the gillnet settings in the littoral, sub-littoral, pelagic, and profundal zones of Lake Årungen (redrawn after Eie and Borgstrøm 1981)

performed with Dune's method (Zar 1996). Mann–Whitney rank sum test was performed to compare only two habitats. The significance level for all the tests was 0.05 ($p < 0.05$).

15.2.3 *Diet Analysis*

Random sub-samples of different size classes of fishes were collected every month by gillnetting. The gut contents of pike (152), perch (120), roach (259), and rudd (17) were sampled from December 2005 to July 2006, and were analyzed under a dissecting microscope (see Sharma and Borgstrøm 2008a, for details). The relative volume percentages of food items were estimated by the method described by Hyslop (1980). The diet data from 1978 to 1980 were obtained from Eie and Borgstrøm (1981). Diet overlap between roach and perch was calculated as percent overlap using Schoener's index (Schoener 1968, Wallace 1981) using the formula:

$$S = 1 - 0.5\left(\sum_{i=1}^{n}\left|P_{xi} - P_{yi}\right|\right)$$

where S = percent overlap in diet; P_{xi} = proportion of food category i in the diet of species x; P_{yi} = proportion of food category i in the diet of species y; and n = number of food categories. The diet overlap between perch and roach was analyzed for different seasons, viz., winter (December–April) and summer (May–July). Comparisons were made for all habitat types. Percent overlaps in diets were considered biologically significant when S value exceeded 0.60 (see Wallace 1981).

15.2.4 *Food Web Analysis Using Stable Isotopes of Nitrogen and Carbon*

The muscle samples of fishes were collected by the standard methods described in the EMERGE protocol (Rosseland et al. 2001). In addition to fish muscles, primary producers, zooplankton, and macroinvertebrates were also collected for the analysis of stable isotopes of nitrogen (δ^{15}N) and carbon (δ^{13}C). Composite samples were used for each taxon other than fishes. The δ^{15}N signatures were used to determine the relative trophic position of organisms in the food web of Lake Årungen. Nitrogen and carbon were analyzed at the Environmental Chemistry Section, Dept. of Plant and Environmental Sciences (IPM), Norwegian University of Life Sciences (UMB). Homogenized and freeze-dried muscle samples were subjected to combustion in a flash elemental analyzer (EA). The combustion gases (CO_2 and N_2) were separated with a Poraplot Q column and transferred to a Finnigan Delta$^{\text{Pluss}}$ XP continuous-flow isotope ratio mass spectrometer (CF-IRMS). See Sharma et al. (2008) for further details of the analysis methods and quality control. The isotopic ratios of nitrogen and carbon (^{15}N/^{14}N and ^{13}C/^{12}C) were expressed as delta values:

$$\delta^{15}\text{N and }\delta^{13}\text{C }(‰) = \left[\left(\frac{R_{\text{SAMPLE}}}{R_{\text{STANDARD}}}\right) - 1\right] \times 1000$$

where $R = {}^{15}\text{N}/{}^{14}\text{N}$ for δ^{15}N or $R = {}^{13}\text{C}/{}^{12}\text{C}$ for δ^{13}C.

15.2.5 *Total Mercury Concentrations and Its Transfer Patterns*

Sampling of pike ($n = 66$ in 2003, $n = 34$ in 2005), perch ($n = 25$ in 2003, $n = 28$ in 2005), roach ($n = 22$ in 2003, $n = 20$ in 2005), and rudd ($n = 6$ in 2005) for Hg analyses was done by gillnetting. The analysis of THg (wet weight, ww) was performed by using the Perking-Elmer model FIMS 4000 Flow Injection Mercury System at IPM-UMB. Anton Paar microwave oven was used for dissolution of fish muscles. The calibration curves of four different synthetic standards were used to calibrate the equipment. The DORM-2 (piked dog fish *Squalus acanthias* L.), certified reference material of National Research Council of Canada, Ottawa, was used to control the accuracy of the method (see Desta et al. 2006, Sharma et al. 2008, for further details). Biomagnification rate of THg through the fish community was determined by regressing log-transformed THg (logTHg) concentrations against δ^{15}N values for all fishes, with the slope of the regression as a measure of the quantitative biomagnification rate (Kidd 1998). The slope of the regression within a species was used as the uptake rate of THg for that species (Kidd 1998, Campbell et al. 2006). The *provisional tolerable weekly intake* (PTWI) of mercury through fish meal size of 250 g for a 60 kg person was computed, based on the FAO/WHO (2006) guideline (see Desta 2007).

15.2.6 Persistent Organic Pollutants (POPs)

The liver samples of fishes (pike, perch, and roach) were collected by the standard procedure given in the EMERGE protocol (Rosseland et al. 2001). The samples were then deep frozen until analysis of POPs. A total of 30 liver samples, 10 samples from each species, were analyzed at BIOFORSK, the laboratory of Norwegian Institute for Agricultural and Environmental Research, Norway. The samples were analyzed for 7 PCB congeners and 12 OC pesticides (including five metabolites). For OC analysis, each liver sample was ground with activated anhydrous sodium sulfate and transferred to a pre-cleaned glass fiber cartridge (Munktell ET/MG 160, 30×77 mm). The samples were extracted in dichloromethane for 4.5 h in an automated Soxhlet extractor (Tecator Soxtec Avanti, 2050). Samples were analyzed for content of OC pesticides (dieldrin, aldrin, endrin, heptachlor, heptachlor epoxide, hexachlorobenzene, and DDT/DDD/DDE) and PCBs (PPCB7; no. 28, 52, 101, 118, 138, 153, and 180) by gas chromatography coupled to mass spectrometry (HP6890 GC–MS) (detailed methodology for the analysis of OCs is described in Sharma et al. (2009)). Comparison of OC concentrations in different fish species was performed by ANCOVA (statistics F) using age as the covariate. Sexwise comparisons were also performed for pike and perch by the same method. Data on OC concentrations were log transformed prior to the analysis due to skewed distribution. Spearman rank order correlations were performed to analyze the relationships between different variables.

15.3 Results

15.3.1 Recruitment of Pike After Population Manipulation

A total of 63% ($n = 304$), 55% ($n = 66$), and 64% (167) of the estimated population of large pike were removed from the lake during 2004, 2005, and 2006, respectively. In terms of estimated biomass, total pike removal due to experimental biomanipulation accounted for 1,225 kg (10.2 kg ha^{-1}) in 2004, 402 kg (3.4 kg ha^{-1}) in 2005, and 609 kg (5.1 kg ha^{-1}) in 2006. The dominating age class of pike in length class 45–64.9 cm was 3+, except in 1982 and 2004, and the number was particularly high in the years 1981, 2005, and 2006. No significant difference in the total estimated number of pike was observed between the 1980s and 2004. However, the estimated number of small pike (45–64.9 cm) increased significantly from 2004 ($n = 634$) to 2005 ($n = 1520$) ($H = 17.112$, d.f. = 2, $p < 0.001$; Fig. 15.5), with a further increase in 2006 ($n = 1590$). The estimated numbers of large pike (≥65 cm) decreased significantly from 2004 ($n = 485$) to 2005 ($n = 121$) ($H = 19.535$, d.f. = 2, $p < 0.001$; Table 15.1), but showed a small increase in 2006 ($n = 260$). In total, there was a significant increase in the estimated numbers of pike from 2004 to 2005 and 2006. According to Sharma and Borgstrøm (2008a), the mean length of prey fish in the stomachs increased with an increase in pike length.

The mortality rate of pike (>45 cm) due to the experimental fishing was very small during 1980–1983 (7% in 1983–19% in 1981), but increased to 49% in 2004 with a decline in 2005 and 2006 to 19 and 18%, respectively. The total annual mortality (experimental as well as other reasons) of pike during 2004–2005 and 2005–2006 were 86 and 56%, respectively (Table 15.2).

15.3.2 Relative Abundance and Habitat Use

CPUE by gillnetting in June indicated that the proportions of large perch had significantly increased during 2004–2006 compared to 1982 (Sharma and Borgstrøm 2008a). The proportion of small roach declined during 2005–2006, after the heavy exploitation of large-sized pike in 2004 and was much lower than in 1982. The number of small pike (< 55 cm), with age ≤ 3 years, as indicated by CPUE in June, was significantly higher in 2005–2006 compared to 1982, with intermediate abundance in June 2004 (Fig. 15.6).

In winter 1978–1980, perch showed a significantly low preference for the pelagic habitat, and they were not caught in this habitat at all during 2005–2006. The CPUE of perch in profundal and littoral habitats were

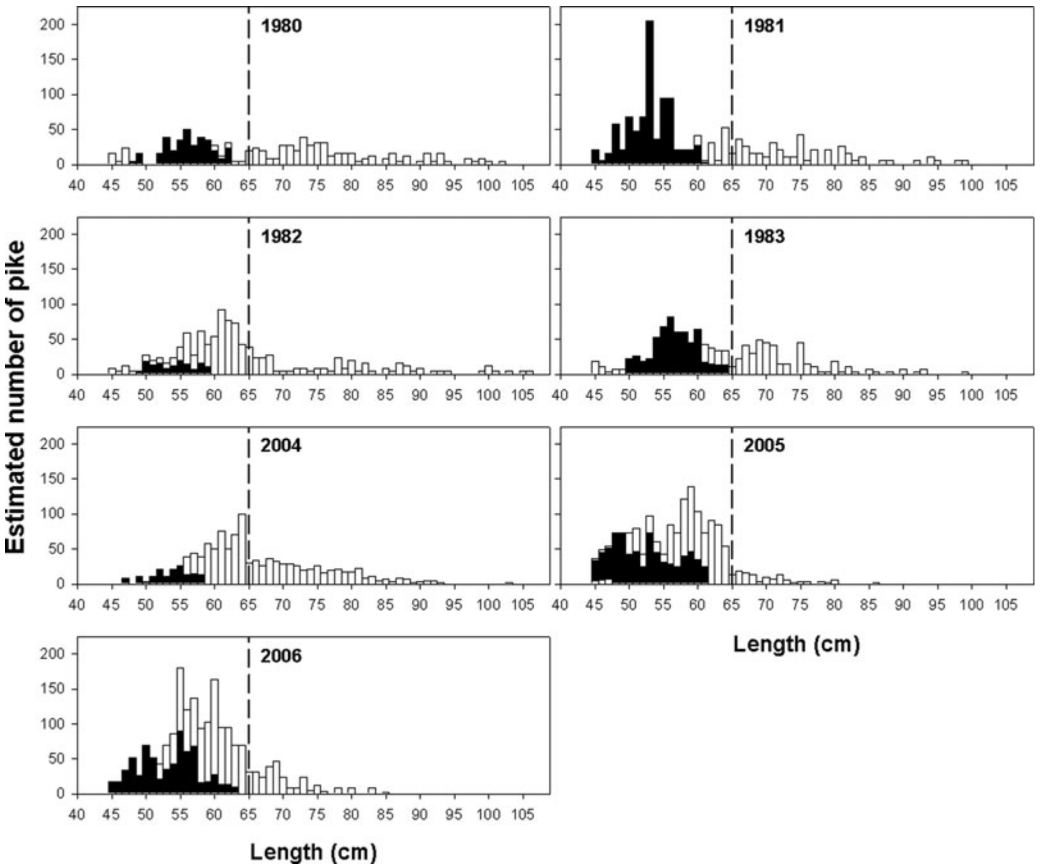

Fig. 15.5 Estimated length distribution of pike (>45 cm) in Lake Årungen during April–first week of May 1980–1983 and April–first week of May 2004–2006. *Black bars* indicate age 3+ pike. Note that size selective harvest of large pike started from the middle of May in each of the years from 2004 to 2006 (Sharma and Borgstrøm 2008b)

Table 15.1 Number of tagged, controlled, and recaptured pike >45 cm during 1980–1983 and 2004–2006, with corresponding estimated numbers of pike with confidence limits (0.95), mortality rates due to the experimental exploitation, total biomass, and average weight

Year	Length class (cm)	Number tagged (*M*)	Number examined for tags (*C*)	Number recaptured (*R*)	Estimated number (*N*)	Confidence limits (0.95)	Mortality due to experimental exploitation	Total biomass (kg)	Average weight (kg)
1980	>45.0	99	85	9	860	475–1720	0.09	2, 249	2.62
1981	>45.0	57	249	10	1318	747–2544	0.19	2, 180	1.65
1982	>45.0	136	173	22	1036	697–1611	0.17	2, 274	2.19
1983	>45.0	123	82	8	1144	651–3027	0.07	2, 010	1.76
2004	45.0–64.9	103	243	39	634	467–884	0.38	2, 257	2.02
	65.0–103.9	142	304	89	485	395–595	0.63		
2005	45.0–64.9	186	248	30	1502	1063–2196	0.17	1, 790	1.10
	65.0–86.9	35	66	19	121	79–193	0.55		
2006	45.0–64.9	185	170	19	1590	1039–2544	0.11	2, 503	1.35
	65.0–85.9	33	167	21	260	172–406	0.64		

Source: Sharma and Borgstrøm (2008b)

Table 15.2 Annual mortality rates of pike in Lake Årungen from 2004 to 2005 and from 2005 to 2006, based on the number of tagged and recaptured pike in different years (Ricker's method)

Period	Tagged in year 1 (M_1)	Tagged in year 2 (M_2)	Recaptures in year 2 from first year (R_{12})	Recaptures in year 2 from second year (R_{22})	Annual mortality rate
2004–2005	245	221	7	49	0.87
2005–2006	221	218	18	40	0.56

Source: Sharma and Borgstrøm (2008b)

not significantly different in winter during both of the periods. In summer, perch were distributed equally in the littoral and profundal habitats in 1978–1980, but were highly abundant in the littoral habitat compared to all other habitats in 2005–2006 (Fig. 15.7). In summer, the CPUE of roach was similar in the littoral habitat both in 1978–1980 and in 2005–2006; however, the catch declined significantly in the pelagic and profundal habitats. In winter 1978–1980, the lowest catch of roach was obtained in the profundal habitat, but no significant difference in CPUE was found in summer. During 2005–2006, a significantly higher CPUE of roach was obtained in the littoral habitat in summer compared to the pelagic and profundal, with no difference in CPUE in all the habitats in winter. Pike were highly abundant in the littoral habitat throughout the study period; however, the CPUE increased in the pelagic habitat in 2005–2006 compared to 1978–1980 (for detailed explanation, see Sharma and Borgstrøm 2008a).

15.3.3 Diets and Food Web Structure

The most preferred prey fish for pike was roach throughout the study period, but perch was also important, and cannibalism was evident, with 2+ pike forming part of the diet in 2004 and 2006 (Fig. 15.8).

Both according to δ^{15}N and stomach content analysis in 2005–2006, pike (feeding exclusively on fish) and perch (consuming zooplankton, fish, and aquatic insects) occupied the top trophic position, with pike having a slightly higher δ^{15}N than perch (Fig. 15.9). Roach had slightly higher δ^{15}N values than rudd. Roach had a wide range of food items, including detritus, zooplankton, aquatic plants, and macroinvertebrates. The δ^{15}N signature values suggest that the food web contained up to three to four trophic levels above the primary producers (δ^{15}N>12‰). The fish community had at least two trophic levels (δ^{15}N>5.4‰); pike and perch at the top, roach at the middle, and rudd at the lower level. Individuals of roach had a large variation in δ^{15}N values spreading them into two different trophic levels (δ^{15}N>3.4‰).

Two co-occurring fish species, perch and roach, exhibited a significant segregation in their diets (SI < 0.60) during most of the study period, with a high diet overlap only in the winter during 1978–1980. However, both species had a large proportion of zooplankton in their diet during summer 2005–2006 (Fig. 15.10).

15.3.4 Hg Concentrations and Biomagnification

A significant reduction in THg concentrations were observed in 2005 compared to 2003 in all fish species (Fig. 15.11), e.g., pike (ANCOVA: $F_{1,97} = 127.9$; $p = 0.000$), perch ($F_{1,50} = 39.8$; $p = 0.000$), and roach ($F_{1,39} = 17.3$; $p = 0.000$). There was also a significant reduction of THg concentrations in 2005 compared to 2003 for same age classes of pike between 1 and 5 years (Fig. 15.12). In addition, a significant faster growth rate was observed in young pike (1+ to 3+) in 2005 compared to 2003 (Fig. 15.13; see Sharma et al. 2008, for details). Pike had a much larger length compared to perch at the same age demonstrating the normal faster growth rate in pike (Fig. 15.14).

In 2005, the biomagnification rate of THg in individual fish from all analyzed species in Lake Årungen was 0.163 (per ‰ δ^{15}N) with a significant relationship between logTHg and δ^{15}N ($F = 157.01$; d.f. = 1, 86; $p = 0.000$; Fig. 15.15) producing the regression equation as

$$\text{Log THg (mg kg}^{-1}) = -3.82 + 0.163\delta^{15}\text{N}$$

The highest rate of THg uptake (0.232) was found in perch, the moderate (0.151) in pike, and the lowest

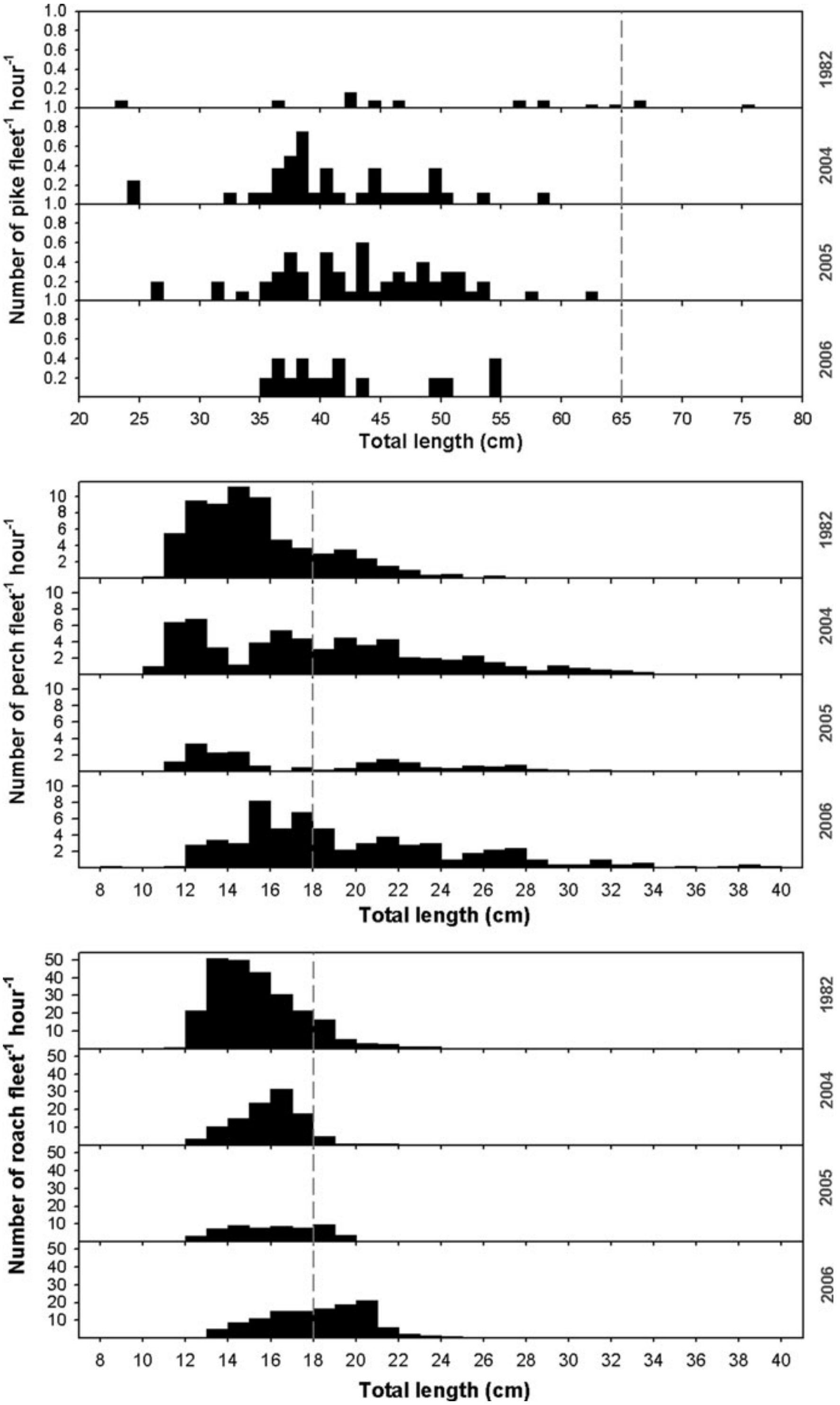

Fig. 15.6 Catch per unit effort (number of fish gillnet fleet^{-1} h^{-1}) before (1982) and after experimental exploitation of large pike (2004, 2005, and 2006) in Lake Årungen. Please note the different scale for each species. *Dotted vertical lines* are drawn to separate small and large fishes (Sharma and Borgstrøm 2008a)

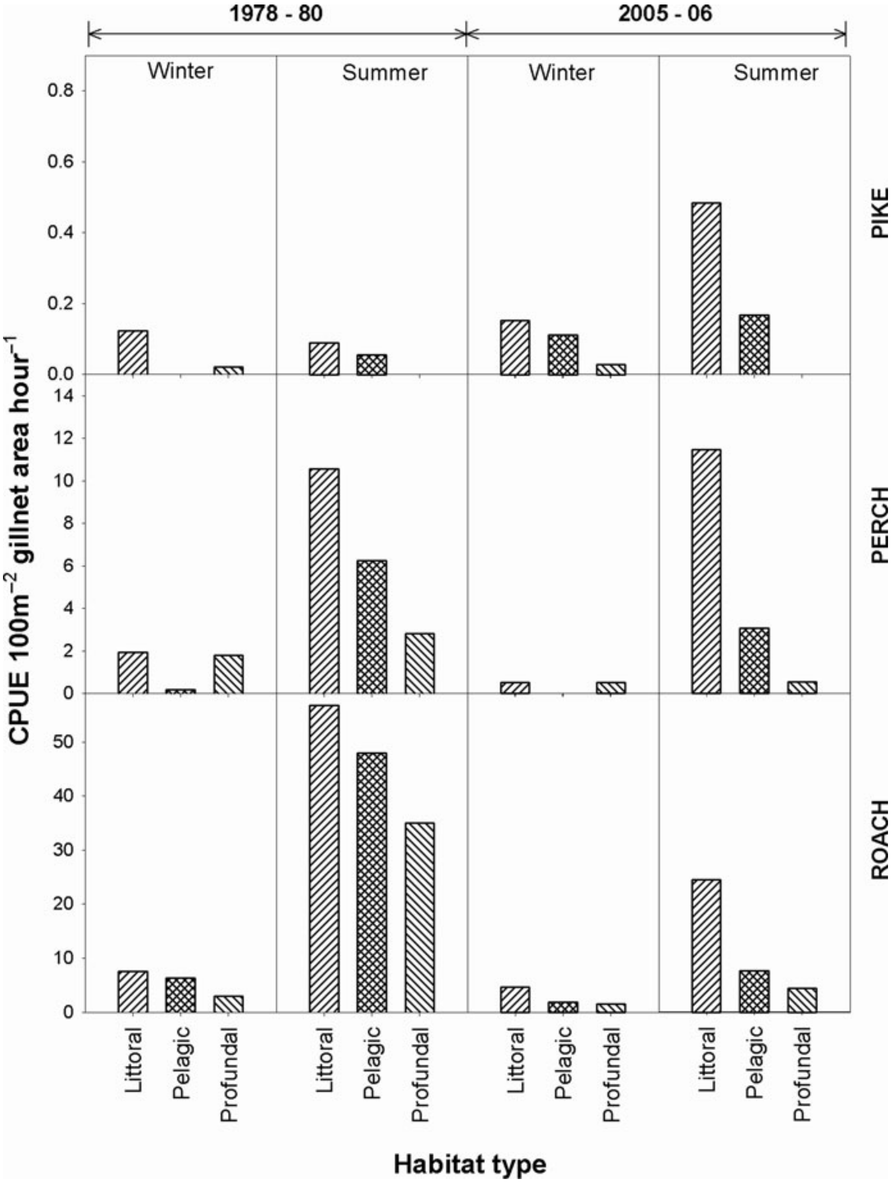

Fig. 15.7 Average CPUE 100 m^{-2} gillnet area ha^{-1} for pike, perch, and roach in 1978–1980 and 2005–2006 in the littoral, pelagic, and profundal habitats of Lake Årungen. Please note the different CPUE scales (Sharma and Borgstrøm 2008a)

(0.105) in roach. A significant increase in burden of THg concentrations with increase in length was observed in pike and perch (Table 15.3). None of the samples analyzed in 2005 had mercury concentrations which would have exceeded the FAO/WHO *provisional tolerable weekly intake* value (1.6 μg kg^{-1}); however, four samples of pike exceeded this limit in 2003.

15.3.5 Organochlorine Compounds and Their Biomagnification

The main OCs detected in fishes from Lake Årungen were DDTs and PCBs. The metabolite p'DDE formed more than 88% of ΣDDTs. All pike and perch samples had pp'DDE metabolite, whereas only two roach

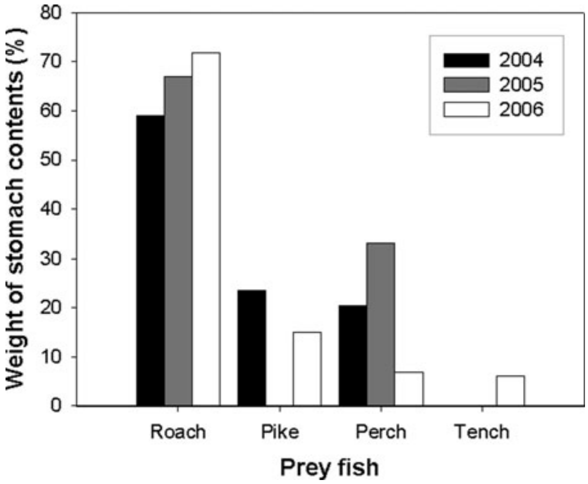

Fig. 15.8 Weight (%) of prey fish in the stomach contents of pike > 65 cm captured in Lake Årungen during 2004–2006 (Sharma et al. 2008)

had detectable levels. The DDT isomers (op'DDT and pp'DDT) and the metabolite pp'DDE were detected only in pike (Table 15.4). Pike had significantly higher concentrations of ΣDDT compared to perch (ANCOVA: $F_{1,17}=14.79$; $p<0.05$), whereas only two samples of roach had detectable levels of DDTs (Sharma et al. 2009).

Three PCB congeners (PCB-153, PCB-138, and PCB-180) were detected in all samples analyzed, and two of them, PCB-153 and PCB-180, formed between 50 and 60% of ΣPCB concentrations. Concentrations of ΣPCB were significantly higher in pike compared to perch (ANCOVA: $F_{1,17=}12.78$; $p<0.01$) and roach (ANCOVA: $F_{1,17}=13.86$; $p<0.01$). However, there was no significant difference of ΣPCB concentrations between perch and roach (ANCOVA: $F_{1,17}=4.31$; $p>0.05$). Hexachlorobenzene (HCB) concentrations were detected in all samples of pike and 30% samples of roach, whereas samples of perch had no detectable levels of HCB. Heptachlor epoxide (HCE) was detected only in pike samples. The pooled data showed a significant correlation of $\delta^{15}N$ with both ΣPCB and HCB ($r_s=0.492$ and 0.676, respectively; $p<0.01$) but not with HCE and ΣDDT ($r_s=0.214$ and 0.226, respectively; Fig. 15.16). Although there is an increasing trend of OCs with age of fishes (Fig. 15.17), a significant relationship was observed only in perch for ΣDDT. Sexwise differences of OCs were detected in pike, with males having significantly higher concentration compared to females (Table 15.5; see details in Sharma et al. 2009).

15.4 Discussion

Depending on the age composition of the population, even a very low rate of cannibalism can have a significant effect upon mortality rate (Le Cren 1965, Fox 1975). This reflects the importance of the cannibalism

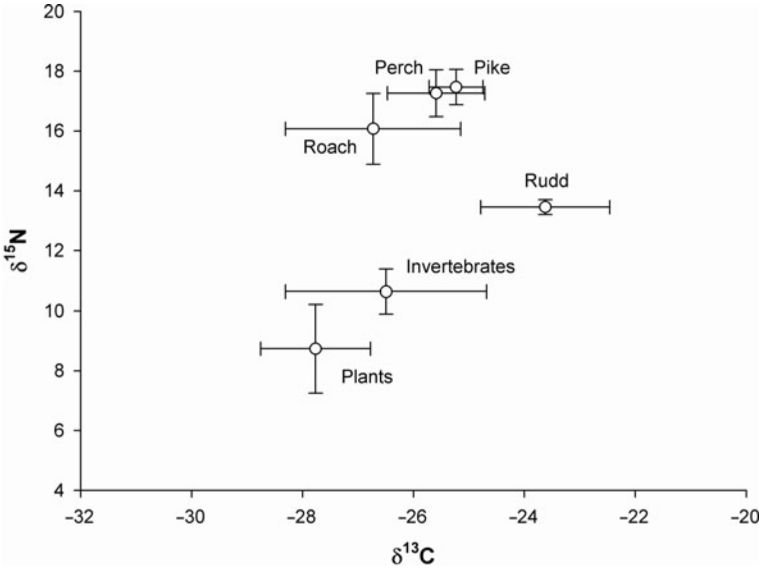

Fig. 15.9 The relationship between $\delta^{15}N$ and $\delta^{13}C$ values of fish; invertebrates and plants in Lake Årungen in 2005–2006. Ranges of error bars indicate standard deviations from the mean, *vertical bars* for $\delta^{15}N$, and *horizontal bars* for $\delta^{13}C$ values (Sharma et al. 2008)

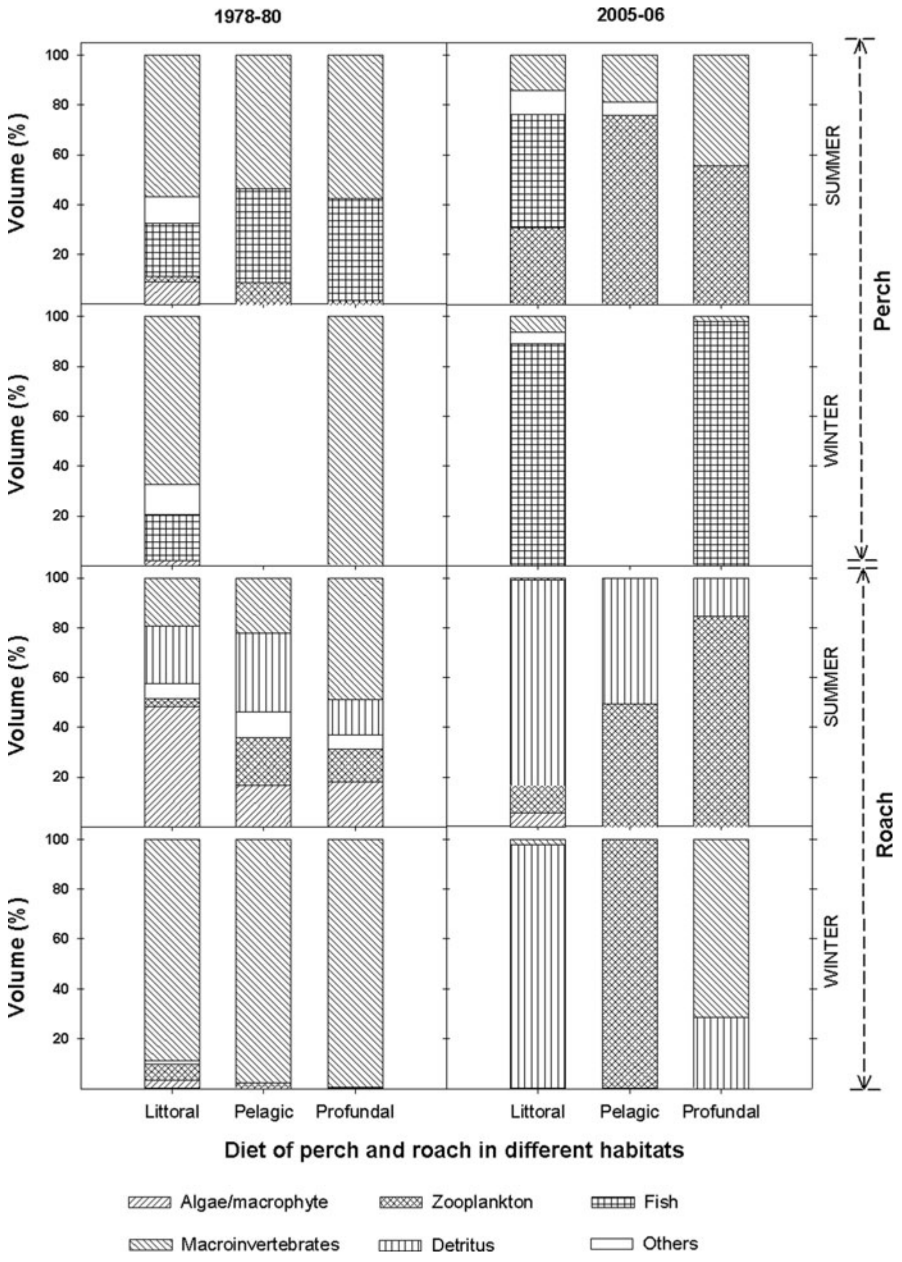

Fig. 15.10 Mean volume percentage of six food categories: algae/macrophytes, zooplankton, fish, macroinvertebrates, detritus, and unidentified items (others) consumed by roach and perch in the littoral, pelagic, and profundal habitats of Lake Årungen during summer and winter 1978–1980 (*left column*) and 2005–2006 (*right column*) (Sharma and Borgstrøm 2008a)

in regulating fish population structures in lakes (Le Cren 1965, Kipling 1984, Svenning and Borgstrøm 2005). The pike consumed by pike cannibals during 2004–2006 were 2-year-olds (Sharma and Borgstrøm 2008b), corresponding to the findings of Le Cren (1965). Therefore, the increase in 3-year-old (3+) pike in Lake Årungen in 2005–2006 could be due to a marked reduction in the consumption of 2-year-old pike as an effect of the heavy removal of large, potential cannibals. This indication of cannibal-driven dynamics within a pike population is in full agreement with results from similar studies, since both recruitment of young of the year (YOY) pike to 1-year-old (Persson et al. 2006) and then to 2-year-old

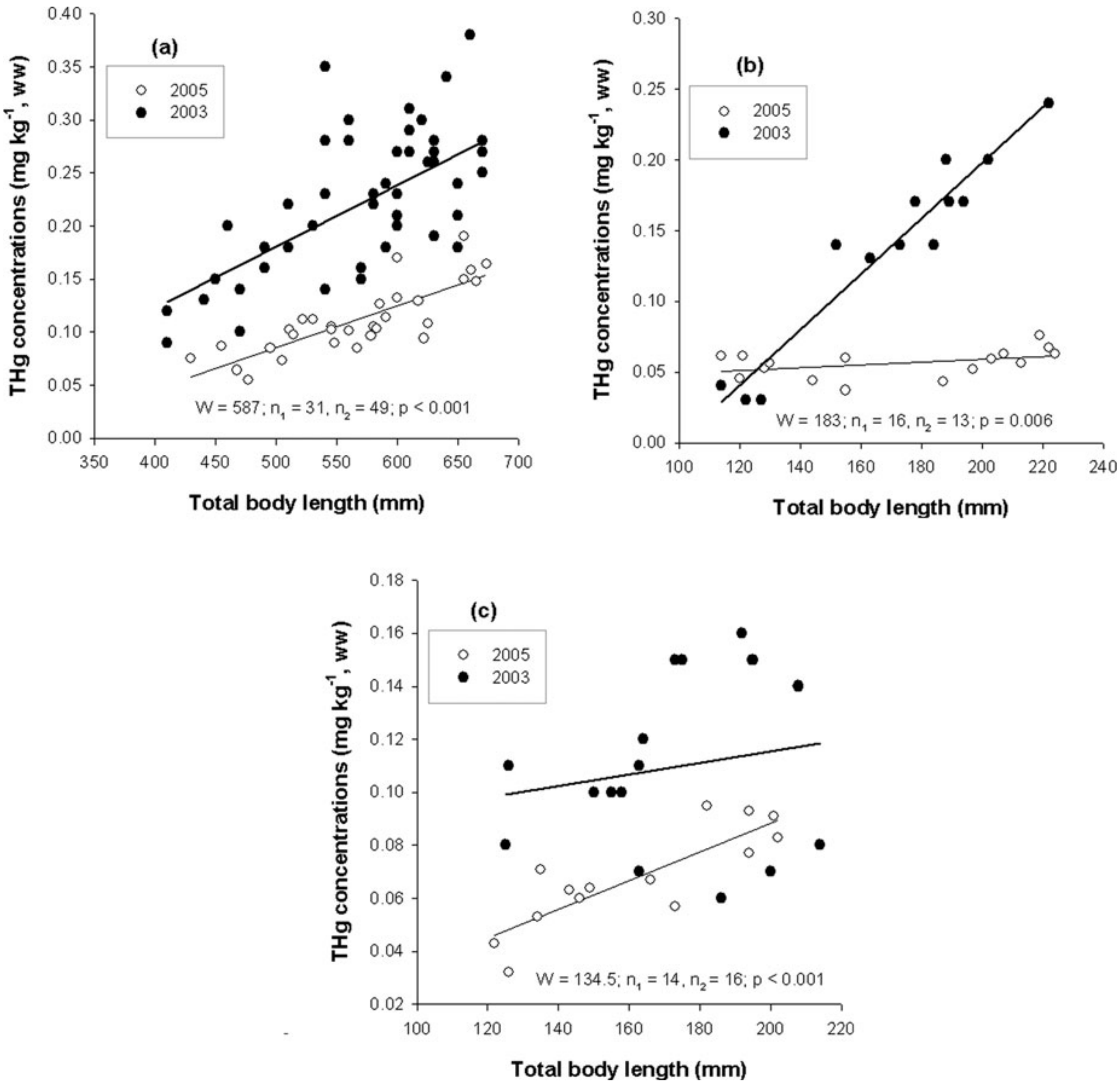

Fig. 15.11 THg concentrations (mg kg^{-1}, ww) in muscles of (**a**) pike, (**b**) perch, and (**c**) roach in 2003 and 2005, plotted against total fish length. The corresponding regression lines are given. Comparisons between the 2003 and 2005 values were performed with analysis of covariance (ANCOVA, *F*-statistic) using length as the covariate. Significance levels are indicated as *p*-values (Sharma et al. 2008)

pike (Le Cren 1965) have been reported to be controlled by cannibalism. Capture probability of fish by gillnetting depends on many factors, including swimming activity, body form, niche shift, mesh size, and position of the nets (Hamley 1975, Rudstam et al. 1984, Borgstrøm 1989, Borgstrøm and Plahte 1992, Finstad and Berg 2004). Accordingly, a change in CPUE does not directly reflect the change in population numbers, but the substantial changes in the CPUE data from Lake Årungen, with a significantly higher CPUE of small pike (<55 cm), large perch (>18 cm), and significantly lower CPUE of small roach (< 18 cm), are indications of marked changes in the densities and structures of all three populations (Sharma and Borgstrøm 2008a). Although pike may utilize the habitat in an ideal free manner (Haugen et al. 2006), individuals of pike spatially avoid larger conspecifics, probably due to predation risks (Nilsson

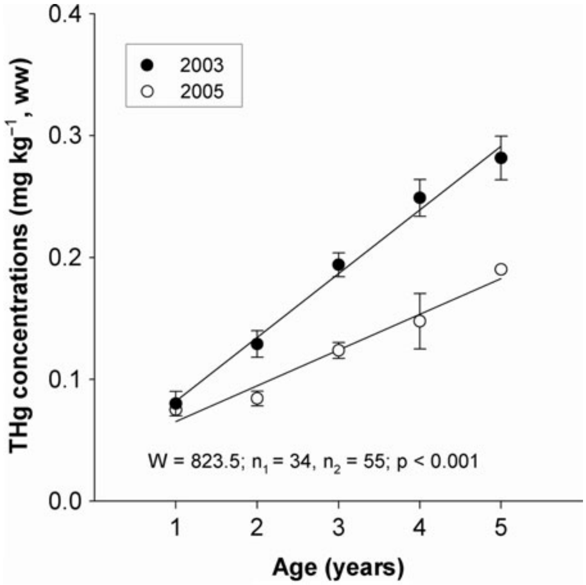

Fig. 15.12 Mean THg (mg kg^{-1}, ww) in pike muscles of age classes 1–5 years in 2003 and 2005. Ranges of error bars (*two horizontal bars*) represent one standard deviation on mean values. Comparisons were performed with Mann–Whitney test: n_1 and n_2 represent sample sizes, and significance levels were indicated as *p*-values (Sharma et al. 2008)

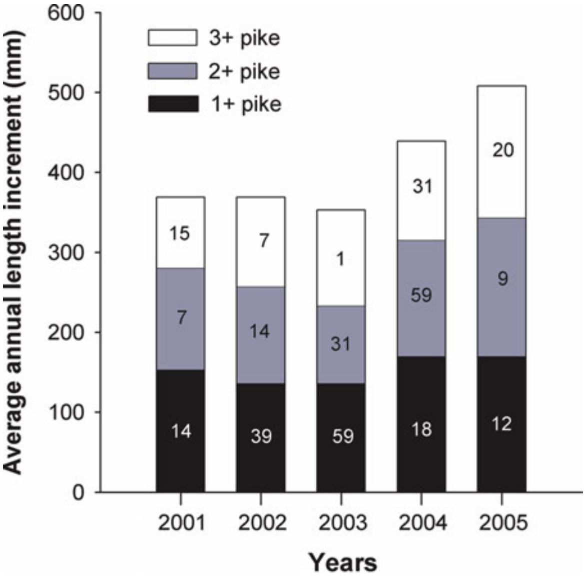

Fig. 15.13 Average annual increments in total length (L_T) of the age classes 1+ to 3+ of pike based on the difference in length in year *t*+1 and year *t* during the period 2001–2005. Number inside bar indicates the sample size for corresponding age classes (Sharma et al. 2008)

2006). The significantly higher CPUE of pike in June 2004–2006 (being several times larger than in June 1982 (Sharma and Borgstrøm 2008a)), although the estimated population numbers were less than doubled (Sharma and Borgstrøm 2008b), indicates that the pike (mainly small- and medium-sized pike < 65 cm in total length) covered larger swimming distances per time in 2004–2006. Indirectly, the elevated CPUE data may thus be a result due to the considerable reduction of large-sized pike (Sharma and Borgstrøm 2008b), i.e., a reduced predation risk, especially in the pelagic habitat, and therefore more cruising of the pike.

Predator–prey relationship in fish is size dependent (Persson 1988, Mittelbach and Persson 1998), with increasing prey size when predator size increases (Diana 1979, Kahilainen and Lehtonen 2003). Since a similar relationship between pike and its prey fish was observed in 1978–1983 in Lake Årungen, but not after the heavy removal of large pike in 2004, the lower occurrence of prey fish within the length class 10–18 cm in the stomachs of pike in 2005–2006 (Sharma and Borgstrøm 2008a) may be explained by a lower availability of prey fish in this length class, as revealed by the CPUE data (Sharma and Borgstrøm 2008a). Alternatively, there could be a marked increase in density of small prey fish (< 10 cm) because fish reduction due to biomanipulation leads to a high recruitment success of young of the year (YOY) fish (Hansson et al. 1998). The higher population density of similar-sized pike in 2005–2006 might also have influenced prey size choices, i.e., resulted in a preference toward smaller prey fish to avoid long handling times and a higher risk of kleptoparasitism (Nilsson and Brönmark 1999, Nilsson 2001). Due to the relative increase in number of piscivores (small pike and large perch) in Lake Årungen after manipulation of the pike population (Sharma and Borgstrøm 2008a, b), the predation pressure on roach might have increased (Eklöv and VanKooten 2001) resulting in a decreased population density of roach. Smaller individuals of prey fish change their habitat utilization to minimize the predation risks (Persson 1986, Romare and Hansson 2003, Schulze et al. 2006), and this could be an explanation for the reduced use of the pelagic habitat by roach as well as by small perch in Lake Årungen in 2005–2006 (Sharma and Borgstrøm 2008a). The experimental removal of large pike in Lake Årungen ended in May 2006, and based on the high number of

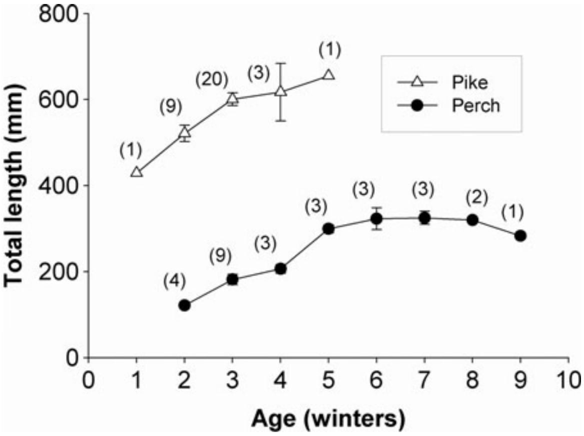

Fig. 15.14 Mean length at age for pike and perch captured in Lake Årungen in 2005. Ranges of error bars represent one standard deviation on mean values (Sharma et al. 2008)

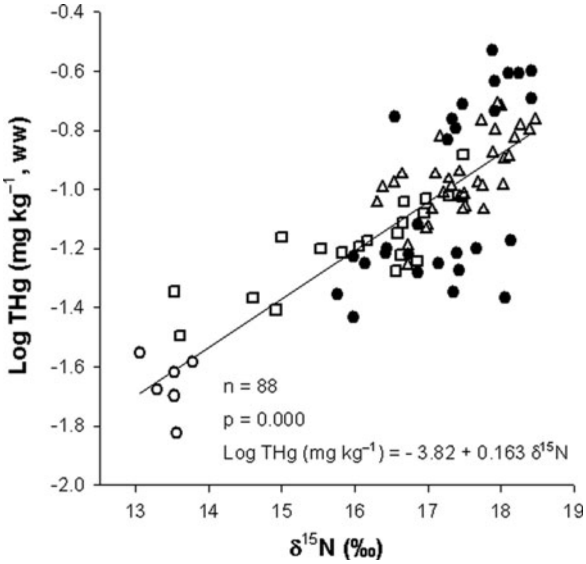

Fig. 15.15 logTHg (mg kg^{-1}, ww) in fish muscles plotted against δ^{15}N (‰) for the four species, pike, perch, roach, and rudd, from Lake Årungen in 2005, showing the biomagnifications rate of THg through the food web. Slope of the regression line (0.163) indicates the biomagnification rate of THg. Symbols used are Δ=pike, •=perch, =roach, and o=rudd (Sharma et al. 2008)

pike in length classes < 65 cm in May 2006 (Sharma and Borgstrøm 2008b), a marked increase in number of large pike in the next years, i.e., 2007–2008 is expected, again resulting in an increased cannibalism in pike, and thus a reduced number of smaller pike, i.e., the predation effect on prey fish will not be as marked as during the years 2004–2006.

Recreational angling can act as an effective mortality factor (Johnson and Martinez 1995) and can make up a large proportion of the pike harvest from a lake (Pierce et al. 1995, Pierce and Cook 2000). No direct studies of recreational fishing activity in Lake Årungen have been carried out, but based on single day observations; the recreational angling seems to have increased in Lake Årungen recently compared to 1980s

Table 15.3 Regression of log-transformed mercury (logTHg) against total length (L_T), δ^{15}N, and δ^{13}C and regression of δ^{15}N and δ^{13}C against L_T for pike, perch, and roach from Lake Årungen in 2005

Description	Regression	*n*	Intercept	Slope	$r_{adj}{}^2$	*p*-value
Pike	logTHg vs. L_T	**34**	**−1.79**	**0.001**	**0.728**	**0.000**
	logTHg vs. δ^{15}N	**34**	**−3.59**	**0.151**	**0.419**	**0.000**
	logTHg vs. δ^{13}C	**34**	**−2.62**	**−0.065**	**0.027**	**0.174**
	δ^{15}N vs. L_T	**34**	**14.7**	**0.004**	**0.388**	**0.000**
	δ^{13}C vs. L_T	**34**	**−24.1**	**−0.002**	**0.070**	**0.174**
Perch	logTHg vs. L_T	**28**	**−1.53**	**0.003**	**0.559**	**0.000**
	logTHg vs. δ^{15}N	**28**	**−5.01**	**0.232**	**0.345**	**0.000**
	logTHg vs. δ^{13}C	**28**	**2.41**	**0.134**	**0.123**	**0.038**
	δ^{15}N vs. L_T	**28**	**16.0**	**0.005**	**0.288**	**0.002**
	δ^{13}C vs. L_T	**28**	**−26.9**	**0.005**	**0.221**	**0.007**
Roach	logTHg vs. δ^{15}N	**20**	**−2.87**	**0.105**	**0.676**	**0.000**
	logTHg vs. δ^{13}C	**20**	**−3.32**	**−0.08**	**0.694**	**0.000**
	logTHg vs. L_T	20	−1.28	0.001	0.07	0.05
	δ^{15}N vs. L_T	20	15.9	0.001	0.00	0.903
	δ^{13}C vs. L_T	20	−26.1	−0.004	0.00	0.64

The intercept, slope, adjusted r^2, and *p*-values of the regressions are shown. The sample size is indicated by '*n*'. Results with $p < 0.05$ are written in bold
Source: Sharma et al. (2008)

Table 15.4 Organochlorine concentrations in fish liver from Lake Årungen expressed as mean ±SD (ng g^{-1}, ww)

Compounds	Pike ($n = 10$)	Perch ($n = 10$)	Roach ($n = 10$)
pp'DDE	408.01±225.32	42.35±39.16	65.96±24.75
pp'DDD	31.49±16.84	nd	nd
op'DDT	2.83±0.90	nd	nd
pp'DDT	17.07±7.51	nd	nd
∑DDT	444.82±246.63	42.35±39.16	65.96±24.75
PCB 28	1.49±0.98	nd	nd
PCB 52	4.60±5.53	2.30±1.86	0.58±0.28
PCB 101	17.26±11.55	4.14±2.75	0.81±0.45
PCB 118	17.05±18.61	3.84±2.67	1.84±0.74
PCB 138	38.43±28.60	6.21±5.07	3.62±3.22
PCB 153	41.18±25.44	7.38±6.35	4.46±3.48
PCB 180	14.91±8.60	1.97±1.68	1.95±1.96
∑PCB	135.44±97.54	24.91±20.07	11.28±8.62
Hexachlorobenzene	2.23±1.04	nd	1.05±0.14
Heptachlor epoxide	0.79±0.37	nd	nd

'*n*' indicates the number of fish analyzed and 'nd' stands for 'not detected'
Source: Sharma et al. (2009)

and 1990s. Because of this, the achieved increase in number of pike due to the manipulation of large pike may, therefore, be lower than the potential increase (Sharma and Borgstrøm 2008b). The increased use of zooplankton by both roach and perch during summer (Sharma and Borgstrøm 2008a) may be a result of high zooplankton abundance, as also explained as a cause for high food (niche) overlap in roach and bleak (*Alburnus alburnus*) (Vøllestad 1985). Higher densities of Cladocera species, especially *Daphnia* spp., may increase grazing on phytoplankton (Søndergaard et al. 1990, Pires et al. 2005), thus leading to a higher transparency of the lake water (Lathrop et al. 2002). The contribution of the phosphorus load from the sediment disturbance may also be reduced due to the reduction in roach density. With a range in δ^{15}N values of approximately 12‰ from producers to top predator fish in Lake Årungen and with at least two trophic levels (δ^{15}N>5.4‰) in the fish community (Vander Zanden et al. 1997, Post 2002), the biomagnification rate of THg through the fish community (‰ δ^{15}N) is within the worldwide range (0.11–0.35) including tropical, temperate, alpine, and arctic lakes (Power et al. 2002, Rognerud et al. 2002, Kidd et al. 2003, Campbell et al. 2005, 2006, Desta et al. 2006). However, low uptake rate of THg in pike compared to perch, although occupying nearly the same trophic position according to the δ^{15}N data, could be caused by growth biodilution (Campbell et al. 2003, Desta et al. 2007) as a result of a faster growth rate in pike compared to perch (Sharma et al. 2008), and vice versa; a higher uptake rate of THg in perch could be due to the slow growth rate in that species compared to pike, resulting in more concentrated THg per unit tissue mass, as also demonstrated by Campbell et al. (2006) in Nile perch (*Lates niloticus*) from Lake Saka. Similarly, the faster growth rate of pike in 2005 compared to 2003 may also be the main reason for the reduced THg concentrations in 2005 (Sharma et al. 2008), due to a similar growth biodilution of THg, as has been documented in many other fast growing fishes (Verta 1990, Kidd et al. 1999, Campbell et al. 2003, Simoneau et al. 2005, Syväranta et al. 2006, Desta et al. 2007). Since diet is an important component in explaining THg concentrations in fish (Bull et al. 1981, Sandlund et al. 1987, Stafford and Haines 2001), an omnivorous feeding habit with a diet specialization of individual fish even including fish eggs and fry (Brabrand 1985) could be an explanation for the large variation in THg concentrations in roach, as has also been observed in cyprinid fishes in East Africa in which fish formed part of the diet in some individuals (Desta et al. 2006).

Göthberg (1983) and Surette et al. (2006) reported that a mass removal of fish biomass reduced the mercury concentration in the remaining biota of Lake Hacksjön, southern Sweden, and in three lakes in northern Québec. Although a large amount of fish biomass has been removed from Lake Årungen by the experimental exploitation of pike during 2004–2006, together with the recreational fishing, the removed fish biomass is probably a small fraction only of the total

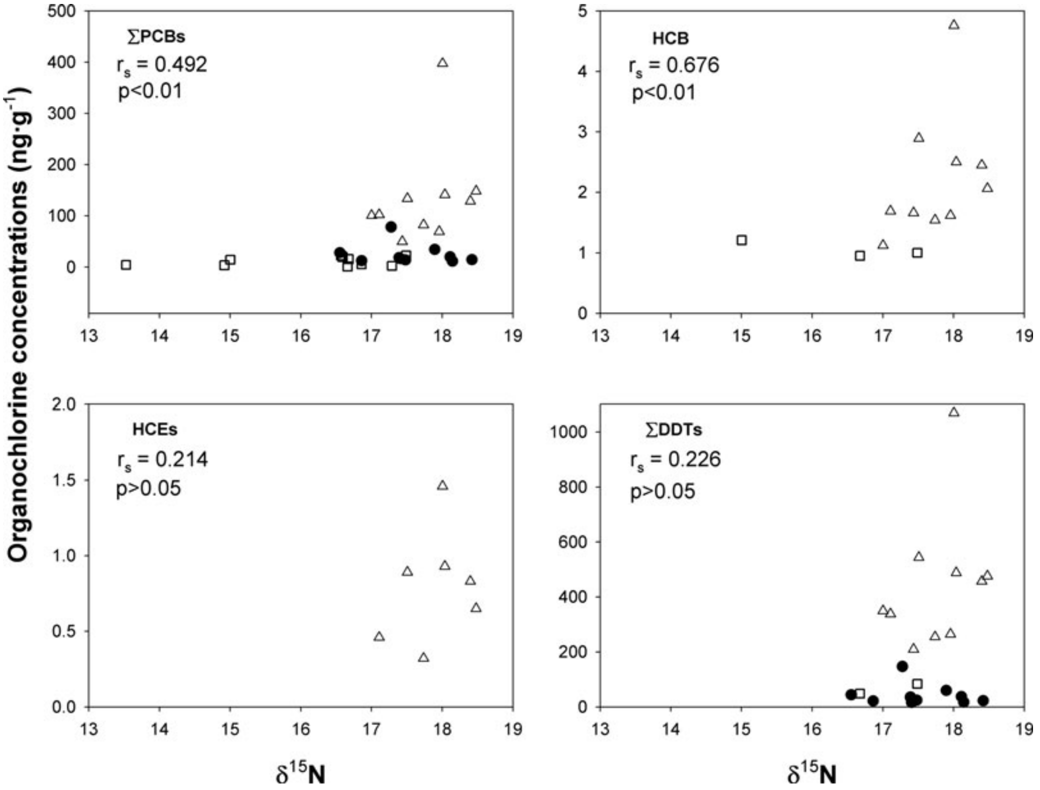

Fig. 15.16 Correlations of organochlorine compounds (ng g^{-1}, ww) in liver with δ^{15}N signatures of individual fish from Lake Årungen. Symbols are Δ=pike, •=perch, and =roach (Sharma et al. 2009)

remaining fish biomass of the lake. Therefore, it is not very probable that such a removal contributes much to reduce the contamination levels in the fish. The general dominance of DDTs among organochlorines in Lake Årungen has also been documented in many other studies (Erdogrul et al. 2005, Covaci et al. 2006, Rosseland et al. 2007), and the metabolite *pp*'DDT alone has contributed more than 88% in Lake Årungen. The detectable limits of most of the metabolites of DDTs in pike, but not in perch and roach, might have been due to the biomagnification of these compounds in pike, which is an exclusively piscivorous fish. A significant relationship between organochlorines and age has been documented for brown trout (*S. trutta*) in Lake Redo by Vives et al. (2005) and in other salmonides from alpine and Arctic lakes by Rognerud et al. (2002). Similar finding was observed only for ΣDDT in perch in the present study. Nevertheless, there is an increasing trend of all organochlorine compounds with increased age of fishes in Lake Årungen (Sharma et al. 2009). The effectiveness of δ^{15}N in describing accumulation of organochlorines has been a controversial issue (Sharma et al. 2009). Our study indicated that δ^{15}N may describe the accumulation of organochlorines if individual fish which occupy two clearly distinct trophic levels (i.e., if δ^{15}N>3.4‰), for example, ΣPCB and HCB (detected in pike and roach) showed significant correlations with δ^{15}N in the present study. The higher concentrations found in this study of the PCB congeners PCB-153 and PCB-180 have been found as a typical result of biomagnification in salmonides from alpine and Arctic lakes (Rognerud et al. 2002).

The lower concentrations of organochlorines in female pike observed in Lake Årungen were also observed in female brown trout by Rosseland et al. (2003) in Lake Redo. According to Larsson et al. (1993) the lower levels of pollutants in female pike are probably due to the higher elimination of the lipophilic pollutants in roe during spawning. However, this hypothesis has strongly been refuted in other species (e.g., perch) conducted by Olsson et al. (2000).

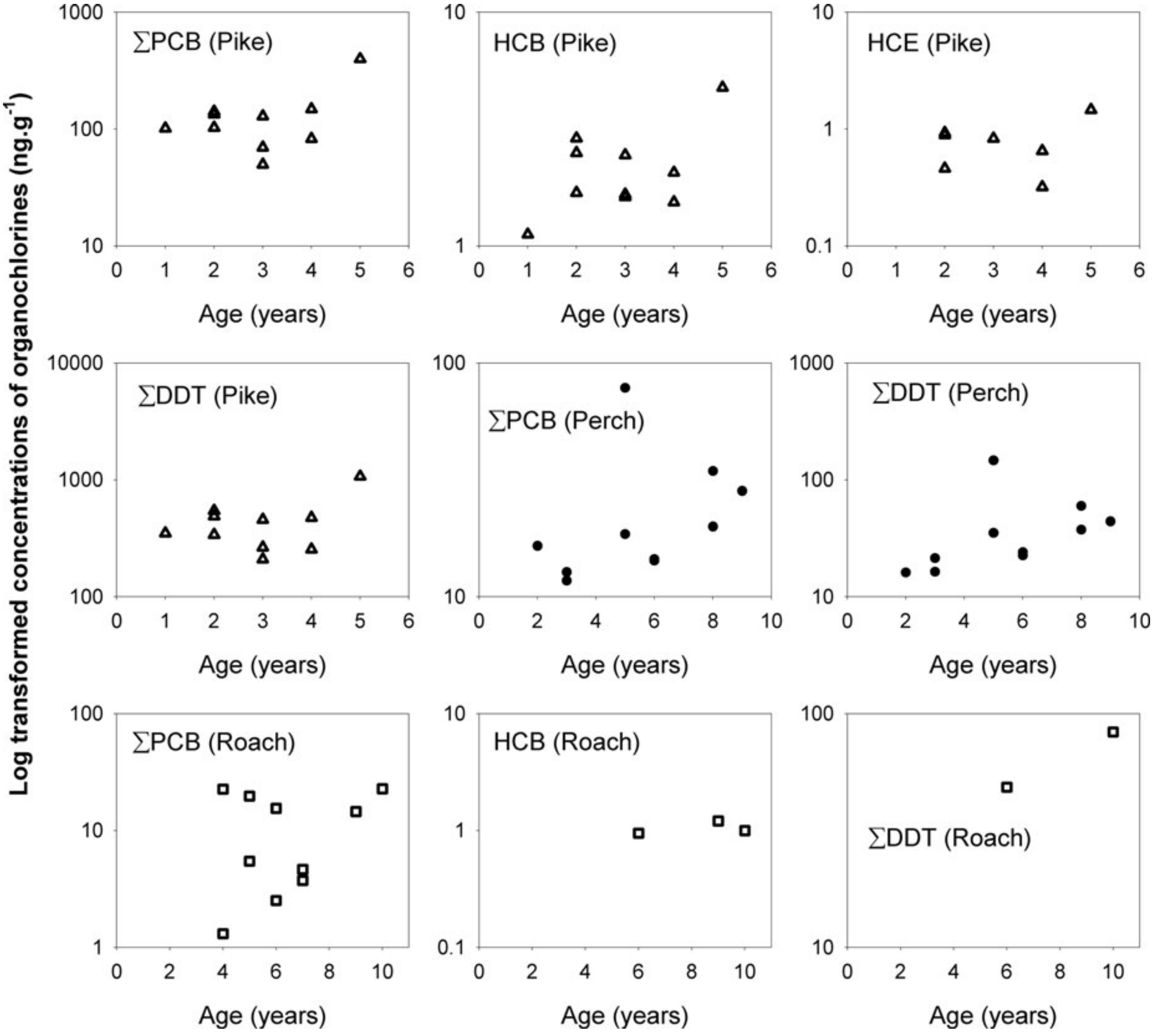

Fig. 15.17 Correlations of log-transformed organochlorine compounds (ng g^{-1}, ww) in liver with age in three fish species in Lake Årungen. Symbols are Δ=pike, •=perch, and =roach (Sharma et al. 2009)

The levels of ΣDDT and ΣPCB in pike and ΣDDT in perch in Lake Årungen were lower in 2005 compared to 2004. This may likewise be a result of the biomanipulation which contributed to faster individual growth rates (Sharma et al. 2008), as well as changes in the diet of perch and roach (Sharma and Borgstrøm 2008a).

15.5 Main Conclusions

Selective exploitation of large pike in Lake Årungen may have favored increased survival of young pike (age 2+) and thus increased the number of 3+ pike in the following years, resulting in a marked increase in number of pike in length class 45–65 cm. However, the increase in population density of pike would probably have been even larger without the recreational fishing in Lake Årungen. This fishing activity may have reduced the potential predation effect on the prey fish populations, i.e., roach and small perch. To obtain an increased mortality of these prey fishes, a restriction limit on the recreational fishing might be suggested, for example, by defining a minimum capture size (around 65 cm) of pike. Furthermore, to obtain a continuous higher survival of small pike, the large pike have to be extensively exploited annually. There

Table 15.5 Fish characteristics based on mean, minimum, and maximum total length (L_{T}), weight (*W*), and age (years) collected for analysis of persistent organochlorine compounds in the liver (ng g^{-1}, ww). Sexwise differences in OC pollutants were also tabulated for each species

Species	Sex	*n*	L_{T} (mm) mean (min–max)	Weight (g) mean (min–max)	Age (years) mean (min–max)	$\sum$PCB (ng g^{-1}) mean (min–max)	HCB (ng g^{-1}) mean (min–max)	HCE (ng $g^{-1)}$ mean (min–max)	$\sum$DDT (ng g^{-1}) mean (min–max)
Pike	♀	6	625 (429–741)	1,628 (420–2340)	3 (1–4)	96.64 (49.68–148.43)	1.74 (1.12–2.45)	0.60 (0.32–0.83)	334.80 (209.1–475.49)
	♂	4	554 (455–655)	1,111 (550–1,750)	2.75 (2–5)	193.64 (102.61–397.37)	2.96 (1.69–4.76)	0.94 (0.46–1.46)	609.85 (338.54–1069.19)
Perch	♀	6	268 (121–324)	297 (15–460)	5.33 (2–8)	19.24 (11.9–34.49)	nd	nd	31.47 (16.1–59.71)
	♂	4	260 (197–283)	242 (81–325)	5.75 (3–9)	33.43 (12.74–78.19)	nd	nd	58.68 (21.38–146.7)
Roach	♀	9	210 (134–282)	130 (20–284)	6.44 (4–10)	10.34 (1.31–22.74)	1.05 (0.95–1.21)	nd	65.96 (48.46–83.46)
	♂	1	135	20	5	19.7	nd	nd	nd

‘nd’ indicates not detected

Source: Sharma et al. (2009)

has been a considerable shift in density, habitat use, and diet of roach and perch in Lake Årungen, at the same time the number of pike was nearly doubled after the experimental exploitation of the large pike. The increased importance of zooplankton in the diet of roach and perch, mainly during summer, indicates an increased biomass of zooplankton and a higher potential for phytoplankton consumption in Lake Årungen. The mercury and some organochlorine concentrations increase as the trophic level increases in the fish community of Lake Årungen. The reduced THg and organochlorine concentrations in 2005 compared to 2003/2004 might have occurred due to growth biodilution, as a result of faster growth rates after the removal of large pike from the lake. Therefore, the substantial removal of a top predator fish may be an important tool to reduce concentrations of mercury and organic pollutants in the remaining fish community. The overall effect of the biomanipulation may therefore give a cascading influence on the biotic community. However, a sustained positive effect demands a continuous effort to keep the density of large, cannibalistic pike at a low level.

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Chapter 16

Reasons and Control of Eutrophication in New Reservoirs

Cuiling Jiang, Liqin Zhu, Xiaoqin Hu, Junyu Cheng and Minghua Xie

Abstract According to the survey, most of the reservoirs and lakes in China are eutrophicated, which have affected urban water supply security and sustainable development of local social economy. Due to the rapid social and economic development and growing population, a large number of pollutants flow into the lakes and reservoirs, leading to nutrients enrichment in different levels. New reservoirs cover generally large submerged areas of jungles, villages, farmlands, and so on. With a lot of pollutants dissolved from the submerged soil and the continuously importing from the basin, a new reservoir is easily eutrophicated because of the low biodiversity and self-purification capacity before the formation of a mature aquatic ecosystem. Meanwhile, it is also significant to establish a network of environmental monitoring in main inflow rivers and reservoirs, so as to observe the variation of water quality in time and provide accurate information for the local authority to formulate practical management policy or control strategies.

Keywords Eutrophication · New reservoirs · Pollutant · Control

16.1 Introduction

Eutrophication is a phenomenon of phytoplankton bloom caused by excessive nutrients (N, P) in water. Due to the rapid development, great harm, difficult control, and slowly restoration, it has gradually become a global water pollution problem (Jin 1990). According to the survey, most of the reservoirs and lakes in China are eutrophicated, which have affected urban water supply security and sustainable development of local social economy.

There are a lot of lakes and reservoirs in china. More than 2,700 lakes including artificial lakes (reserviors) have an area beyond 1 km^2 and the total area is up to 90,000 km^2. About one third is shallow lakes located mostly in Qinghai-Tibet Plateau and plain area of middle and lower Yangtze River. (Chinese lake database, CAS Nanjing Institute of Geography and Limnology, searched in April 2009). Due to the rapid social and economic development and growing population, a large number of pollutants flow into the lakes and reservoirs, leading to nutrients enrichment in different levels. Based on “2008 Environmental Condition Bulletin of China” published by Ministry of Environmental Protection, PRC, among 26 state-controlled lakes and reservoirs (Table 16.1), only 1 lakes met Class II water quality standard, accounting for 3.9%, 5 lakes accounting for 19.2% met Class III, 6 lakes occupying 23.1% satisfied Class IV, 5 lakes occupying 19.2% satisfied Class V, and 9 lakes occupying 34.6% met inferior to Class V. In 26 lakes and reservoirs monitored the nutritional status (Table 16.2), 1 lake accounting for 3.8% was severely eutrophicated; 2 lakes accounting for 7.7% were moderate eutrophicated; 8 lakes were mildly eutrophicated, accounting for 30.8% and the rest were mesotrophic. The main

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Table 16.1 Water quality of state-controlled lakes and reservoirs in China

Water quality standard	Number	Class I	Class II	Class III	Class IV	Class V	Inferior Class V
Three lakes[a]	3					1	2
Large freshwater lakes	9			3	2	1	3
Urban lakes	5				2	1	2
Large reservoirs	9		1	2	2	2	2
Total	26		1	5	6	5	9
Proportion (%)		0	3.9	19.2	23.1	19.2	34.6

[a] Three lakes include Taihu Lake, Dianchi Lake and Chaohu Lake

Table 16.2 Nutritional statuses of 26 lakes in China

Name	Nutritional status index	Nutritional status	Name	Nutritional status index	Nutritional status
Taihu Lake	60	Mild eutrophication	Chaohu Lake	60	Mild eutrophication
Dianchi Lake	71	Severe eutrophication	Daming Lake	55.7	Mild eutrophication
Dalai Lake	67.7	Moderate eutrophication	West Lake	50.5	Mild eutrophication
Baiyang Lake	59.5	Mild eutrophication	Kunming Lake	45.2	Mesotrophy
Hongze Lake	58.0	Mild eutrophication	Laoshan Reservoir	48.6	Mesotrophy
Nansi Lake	49.6	Mesotrophy	Yuqiao Reservoir	47.1	Mesotrophy
Bositeng Lake	38.8	Mesotrophy	Songhua Lake	45.9	Mesotrophy
Poyang Lake	50.6	Mild eutuophication	Dongpu Reservoir	44.6	Mesotrophy
Dongting Lake	48.1	Mesotrophy	Menlou Reservoir	41.7	Mesotrophy
Jingpo Lake	40.2	Mesotrophy	Dahuofang Reservoir	34.9	Mesotrophy
Erhai Lake	41.0	Mesotrophy	Qiandao Lake	34.0	Mesotrophy
Donghu Lake	60.3	Moderate eutrophication	Miyun Reservoir	32.7	Mesotrophy
Xuanwu Lake	58.8	Mild eutrophication	Danjiangkou Reservoir	35.6	Mesotrophy

pollution indicators of these lakes were TN (total nitrogen) and TP (total phosphorus). With the continuous input and accumulation of pollutants, natural lakes and reservoirs still can not avoid eutrophication, although they have a complete aquatic ecosystem structure and strong self-purification or water environmental bearing capacities. As to new reservoirs, which were usually built by submerging large areas of untreated soil, with the internal sources of pollutants dissolved and external sources of pollution input, they are easily eutrophicated especially in the part for their low biodiversity and poor self-purification capacity. Eutrophication in new reservoirs becomes a hot issue in the world. Study on the process of eutrophication and reasons of first cyanobacteria bloom in new reservoir can provide scientific bases for control and management of entrophication in lakes and reservoirs.

16.2 Reasons of Eutrophication Occurring in New Built Reservoirs

Factors leading to eutrophication can be divided into three: (1) physical factors are mainly natural conditions (specifically appropriate temperature and light) and the hydrodynamic conditions. (2) Chemical factors refer to the nutrient concentrations (especially N and P) in water. (3) Biological factors specifically mean composition and structure of aquatic ecosystems. Besides, the occurrence of eutrophication also related to other factors such as the basin topography, geographical location, lake morphology, and regional industrial structure, which are not independent but influence each other. In different circumstances, the impact of these factors varies in different degrees.

In the early stage of water collection, nutrients in new reservoirs are mainly from the inflow water and submerged soil release. In this process, biocoenosis have not yet completely formed and aquatic biomass is low, which makes the reservoirs in an unstable state and eutrophication and cyanobacteria blooms take place easily in local region. Cyanobacteria blooms are not only caused by physical and chemical factors but also affected by biological factors such as competition of other phytoplankton or predation of zooplankton and filter-feeding fish and so on. In the stable lakes and reservoirs, the outbreak of algae will not take place when nutrient concentrations are high but other conditions (such as temperature and light) are unsuitable. For the aquatic communities of new reservoirs are in early stage of succession, with the poor capacities of self-purification and resistance, it is easy to break out water bloom even in the low nutrient content. Following are the specific analysis of eutrophication conditions in new reservoirs.

16.2.1 Natural Factors and the Hydrodynamic Conditions

One phenomenon of eutrophication is the rapid growing of algae. The correlations between algae growth rate and light, temperatures, nutrient concentrations, and hydrodynamic conditions are showed by the following formula:

$$\mu = f(T) \times \min[f(P), f(N)] \times f(L) \times f(u)$$

$$f(T) = \mu_{\max} \times \theta^{T - T_{\max}}, f(P) = \frac{T_{\mathrm{P}}}{T_{\mathrm{P}} + K_{\mathrm{P}}}$$

$$f(N) = \frac{T_{\mathrm{N}}}{T_{\mathrm{N}} + K_{\mathrm{N}}}, f(L) = \frac{1}{ah} \ln \frac{I_0 + I_s}{I_0 e^{-2\alpha h} + I_s}, f(u) = v^{ru}$$

where μ is the actual growth rate of algae, L s^{-1}; $\mu_{\max}$ the maximum growth rate of algae, L s^{-1}; T the actual temperature of the water body, °C; $T_{\max}$ the optimum temperature required for algae growth, °C; θ temperature correction factor; I_0 water surface radiation intensity, kcal m^{-2} d^{-1}; I_s light half-saturation constant, kcal m^{-2} d^{-1}; h water depth, m; α water comprehensive extinction coefficient, L m^{-1}; T_{P} total phosphorus concentration, g m^{-3}; K_{P} phosphorus half-saturation constant, g m^{-3}; T_{N} total nitrogen concentration, g m^{-3}; K_{N} nitrogen half-saturation constant, g m^{-3}; υ, γ undetermined coefficient (Li et al. 2006).

The formula demonstrates that temperature, light, and hydrodynamic conditions have a great impact on the algae growth and are closely related to eutrophication.

Algae especially cyanobacteria are found often in summer with abundant light and high temperature. Water blooms usually break out when water temperature is between 25 and 35°C, and they seldom occur when the temperature is below 20°C. Due to sufficient light and relatively high temperature in the surface layer of reservoirs, phytoplankton such as diatoms, green algae, cyanobacteria, and other algae grow easily. While in deep layer, less light leads to slow growth of submerged plants, and only benthic animals and anaerobic bacteria can survive. Changes in the hydrodynamic conditions also have a significant effect. In some shallow lakes and reservoirs, after a large waves or heavy rain, hydrodynamic actions lead to sediment suspension and nitrogen, phosphorus, or other nutrients release. At the same time, algae cells released from the sediment make the water bloom formed (Xing and Hu 2006). Generally speaking, with the lower water flow speed, the possibility of algae blooms is higher.

The new reservoirs' characteristics of long and narrow shape, large slope in the river bed, heavy fluctuation in water level, and instability show a strong influence in distribution of water temperature, dissolved oxygen, nutrients, and suspended solids, which affect the growth of algae further. Algal growth is controlled by the hydraulic conditions and other natural conditions. In the river-like reservoirs, even the higher nutrient concentrations are not easy to lead to algae excessive growth due to high water flow speed. In contrast, slow flow velocity in the transition water or lake-like reservoirs, the same nutrient concentrations may cause water bloom. "Environmental Impact Report on the Three Gorges Water Control Project" (Dai 2007) written in 1991 by Environmental Assessment Department of the Chinese Academy of Sciences and the Yangtze River Water Resources Protection Science Research Institute forecasted the water quality variation and eutrophication trends of the Three Gorges Reservoir. First, in backwater fluctuation zones of the small tributary and still water areas of bays or reservoirs, the secondary pollution caused by severe hypoxia or partial eutrophication pollution would happen after the first heavy rain each year. Second, eutrophication would not occur in the areas of main streams, for the Three Gorges Reservoir is a canyon-type reservoir and the average depth is about 70 m. Third, the formation of reservoirs would increase the pollution of surrounding sub-river. For the rising of water level in the main stream, the widening of the water surface, the decreasing of the flow velocity, and the blocking of the pollutants transportation make nitrogen, phosphorus, and organic matter enriched in the backwater area and the estuary surrounding. Under certain conditions, eutrophication may be emergent in the local reservoirs branch and bays of the river tributaries. Wind direction and hydrodynamic conditions can change the distribution of algal and also play a certain role in the algal migration process. Temperature and rainfall affect the type of algal and bloom duration. Therefore, in appropriate time, effective measures should be taken to avoid the adverse effects of natural and hydrodynamic conditions in control water bloom, so that the water qualities develop in a good direction (Zhou et al. 2006).

16.2.2 The Nutrient Concentrations in Reservoirs

Eutrophication in lakes and reservoirs are resulted from nitrogen, phosphorus, and other nutrients enrichment, which brings out the rapid reproduction of algae and other plankton, the decline of dissolved oxygen concentration, the death of a large number of fish or other organisms, and the deterioration of water quality. Nitrogen and phosphorus are the main factors leading to water blooms and also the important indices for evaluation reservoirs' degrees of eutrophication. In general, when TP and TN concentrations in water reach 0.02 and 0.2 mg L^{-1}, respectively, from a single nutrient factor view, algae blooms may occur (Lin and Liao 2003). What is more, the rate between nitrogen and phosphorus (TN:TP) in water also significantly affects the composition of phytoplankton populations. A study about 17 lakes around the world showed that when the rate is less than 29, cyanobacteria will become the dominant algae in lakes. And the ratio does not only impact on population composition, but also closely related with the algae proliferation (Smith 1983). Nutrients in new reservoirs are mainly from the exogenous inputs and endogenous release. Exogenous inputs refer to pollutants in the rainfall and upstream water. And the endogenous release is pollutants from the submerged soil. The upstream water quality of reservoirs is mainly up to industrial structure and lifestyle in the basin, including industrial production, the loss of agricultural fertilizers, rural residents, livestock and poultry breeding, land erosion and fishery. Pollutants from upstream basin discharging into the reservoir directly or indirectly through rivers have an important effect on the reservoir water quality. Long-term exogenous nutrient inputs and aquatic organisms' residue deposition lead to the accumulation of a large number of nutrients from sediment in rivers and lakes. In particular conditions, these nutrients are easily released by molecular diffusion (it mainly depends on the concentration differences between nutrients in the sediment pore water and overlying water), wind power, disturbance of benthonic organisms, bubbles overflow, phytoplankton floating, and of the transfer among aquatic plants roots (Qing et al. 2006). That results in the increasing of nutrients content in water and promoting water blooms. It is one of the most important reasons of eutrophication appearance in most domestic lakes and reservoirs.

Phosphorus concentration is an important indicator often used to measure the productivity levels and eutrophication degrees of lakes. In natural lakes, the majority of phosphorus exists in organic-P. Furthermore, about 70% of organic phosphorus is particulate, which makes phosphorus easy settle in the bottom after entering water. Then, nutrients in sediments are much higher than in the overlying water. Under certain conditions, they will be released into the overlying water and become the endogenous load. As to new reservoirs, dredging soils with high nutrients and cleaning waste left by emigration should be done before water storage. In the beginning of water storage, the influences of soil to water quality are mostly concentrated in bottom water layer. The strength of the nutrients release is relatively weak and the impact on the water quality of entire reservoir is not distinct. Cui Lei and Hao Fanghua (2003) did a laboratory simulation study on influences of soil to water quality in initial impoundment period of a regulation and storage reservoir – Baohe reservoir in South–North Water Diversion Project. Because of the interfacial action in water and soil, exchange between soil pore water and overlying water and reactions among main components in the water, the concentration of different forms of nitrogen, total phosphorus, total nitrogen, COD, and heavy metals changed in some extent. However, from the overall trends view, nutrients released from submerged soil are low, which could not have a significant adverse impact on water quality. In a word, compared to the nutrients released from the reservoir sediments, the water quality of importing river for new reservoirs plays the decisive role in outbreak of cyanobacteria blooms in initial water storage period. Therefore, more attention should be paid to the quality of inflow water.

16.2.3 The Structure of the Ecosystem in Reservoir

The fundamental mode of the freshwater ecosystem's succession is a process from the oligotrophy to the eutrophication and from the water body to the land. In the initial stage of the reservoir formation, only phytoplankton can be found because of the low-nutrition concentration. With the importing of exogenous substance, organic matters increase and sediments deposit on the bank, aquatic higher plants emerge, and then followed with aquatic insects, annelida, fish and other kinds of animals. In the end, coastal plants develop slowly from edge to center. The integrated aquatic ecosystem comes into being, which can restrict the over growth of phytoplankton by the food net and maintain a relative balance. The occurrence of water eutrophication is closely and inseparably related to the changes of food web and nutritive structure. The food web based on the relationship between food compositions in the aquatic ecosystems has always been a focus as a driving force to the migration and transformation of biogenic elements. Pomeroy discovered the protozoa's predation to the bacteria, emphasized the importance of micro-plankton in the aquatic food web, and linked it to the biogenic elements' transformation behavior in the water and sediments (Fan and Wang 2007). In the process of reservoir impoundment, with the great fluctuations and changes of environmental factors, there is no sufficient time for organisms to grow and reproduce. Food net and aquatic ecosystem have not been formed, so the biodiversity is low. (Liu and Xie 2001). After the detailed analysis of many reservoirs around the world Straskraba found that the mean water residence time is directly related to the hydrodynamic, chemical, and biological processes of reservoir. First of all, the water residence time has an effect on the stratification of reservoirs. When the residence time is less than 10 days, it is difficult for reservoirs to stratify. If the water residence time is more than 100 days, reservoirs will appear obvious stratification (Straskraba et al. 1999). When the concentrations of pollutants flowing in reservoirs keep steady, the nutrients load will tend to decrease with the residence time increase, which meet the exponential function (Styczen and Storm 1993). Retention time directly affects the population dynamics of the aquatic organisms. If it is too short, the population of zooplankton is difficult to develop due to the lack of sufficient reproduction time. After the water storage of the reservoir, phytoplankton as primary organism will firstly emerge. When the stored organic matters are consumed, the aquatic ecosystem will become stable with low productivity. If the soil erosion in the basin is not serious, and the amount of nutrients imported from the upstream is not large, the stable phase can last for a long time. Otherwise, because of the inputs of a large number of nutrients, the balance will be destroyed soon, which results in reservoirs eutrophication and algae blooms. The eutrophication caused by human activities is obviously different from that in natural state, which is a gradual and prolonged process, dominated by itself and reflecting a normal trend of ecological succession. However, the eutrophication of most water bodies is generated and influenced by human activities (for example, the importing of nutrients and the slow flow velocity caused by the reservoir construction). As the biological process is interfered or destructed, ecological succession is changed. During the water impounding period, nitrogen and phosphorus, etc., came from the submerged farmland, soil, plants residue, and the upper reaches accelerate the algae growth and increase the organic matters in water. Because the community formation of the aquatic animals are slow in the new reservoir (for the fish mainly relying on artificial stocking), the algae blooms with the surplus nutrients and the absence of predation pressure. The organic matters start to decompose after the death of algae. The consumption of oxygen causes severe hypoxia in the lower layer of reservoirs. Then the water quality deteriorates and the growth of the other aquatic organisms becomes difficult. The unstable structure of community limits the abilities of self-adjustment and self-rehabilitation of reservoirs, which leads to the imbalance of aquatic ecosystem and reservoirs eutrophication.

16.3 Water Quality Variation and Eutrophication in New Reservoirs (Take the Three Gorges Reservoir and Laohutan Reservoir as an Illustration)

16.3.1 The Three Gorges Reservoir

The Three Gorges Reservoir is one of the most super-sized water conservancy and hydropower projects in the world. It has great comprehensive benefits in flood controlling, power generation, shipping, and water supply, etc. The Three Gorges Project is located in Sandouping Town of Yichang County in Hubei Province, about 40 km to the Gezhouba Water Control Project downstream and it's drainage impacting area is 1,000,000 km^2. The Three Gorges Reservoir was built in December 1994 and dammed Yangtze River in November 8, 1997. The Three Gorges Project began to play the benefit role of power generation and navigation in June 2003 with the water level to 135 m. In October 2006, the reservoir started running at the water level of 156 m. This strategy will be lasted until September 2009, and then it will raise to the final normal water level of 175 m (Yue 2003). The water area of the Three Gorges reservoir was divided into three parts. One is in front of the dam, the second is the main stream water area, and the third is the backwater reach of the tributary. After the Three Gorges reservoir began to store water, the water levels rose, rate of flow slowed down, and the exchange capacity of water weakened. The residence time of pollutants in front of the dam and in backwater of the tributary was prolonged and the nutrient contents increased. The Cyanobacteria, Dinoflagellate and Diatom were outbreak many times in these areas. The water bodies have been in the situation of eutrophication. Ecological environment of the Three Gorges Reservoir is concerned strongly by ecologists after water storage. A large number of studies concerned about how water quality changes after water storage, how the aquatic community succession is in progress, and how the eutrophication in front of the dam and in backwater of the tributary have developed.

16.3.1.1 Changes of Hydrodynamic Character After the Water Storage in the Three Gorges Reservoir

The water hydrodynamic conditions have changed significantly after the second phase of the reservoir running (with the water level of 156 m). After water storage the average flow velocity in front of the dam is 10-fold less than natural rivers before. Surface flow velocity of the Yangtze River was generally 1.0–3.0 m s^{-1} and the maximum was up to 8.8 m s^{-1} in rapid reach of tributary before the water storage. Parts of the water areas become slack water and some was basically static after water storage of the Three Gorges Reservoir. In the flood season (about 4 months), the surface flow velocity is less than 1.0 m s^{-1} along the bank of the reservoir and only in the main stream near the dam it is 6 m s^{-1}. Also, there are slow areas with the velocity less than 0.1 m s^{-1} or the relatively calm water in the large- and medium-sized bays formed by tributaries (Dai et al. 2004). In general, the reservoirs are divided into three types according to the flow velocity, they are river-like (velocity> 0.2 m s^{-1}), transitional (0.05–0.2 m s^{-1}), and lake-like (velocity <0.05 m s^{-1}) types, respectively. Even though the composition and density of the phytoplankton population vary considerably, eutrophication may occur in every case in the backwater area of tributaries and stagnant inlet area.

16.3.1.2 The Change of Water Quality in Three Gorges Reservoir Before and After Impounding

The water quality was generally good before the construction of the Three Gorges Reservoir. The evaluation of water quality from 1996 to 2001 showed that the average concentrations of the indicators in the main stream and tributaries met to the national standards of drinking water source [according to the Grade II of "Environmental Quality Standards for Surface Water"

(GB3838-2002)]. Only a few parameters such as potassium permanganate index, total lead, and ammonia nitrogen in part-time were over Class II water standards. The water quality was better in non-flood season than that in flood season and had being kept the stability during 6 years.

If only nutrients were considered, when TN concentration was no less than 0.20 mg L^{-1} and TP was no less than 0.02 mg L^{-1}, the reproduction of algae was in a high speed and eutrophication would occur. According to the survey of Meng Chunhong and Zhao Bing (2007) finished after the first stage of water impounding in October 2003 and April 2004, total nitrogen and total phosphorus concentrations in the Three Gorge Reservoir area were between 1.10–1.85 and 0.063–0.098 mg L^{-1}, respectively. It indicated that the maximum TN concentration was almost eight times higher than the eutrophication condition and TP was about four times higher. Therefore, the nutrients were an important condition for eutrophication emergence in the reservoir. In fact, as a result of high concentrations of nitrogen and phosphorus, water blooms had broken out in the front of dam and in the backwater area with the appropriate light and temperature conditions after the second phase of water impoundment.

The impact of the same importing nutrients on levels of eutrophication was different in each area of the reservoir. After water storage, the main stream was oligotrophy and mesotrophic, while some backwaters of the tributaries had reached eutrophication. The Three Gorges Reservoir area is one of the most serious soil erosion areas in China basically due to the loss of soil and water from the sloping fields with high fertility. After impounding by the normal water level of 175 m in the reservoir, the flooded arable land reached to 237.8 km^2 (including 73.8 km^2 of paddy field and 164 km^2 of dry land), accounting for 37.6% of the total flooded land area (632 km^2). There were many subtributaries in the reservoir area, which received nonpoint source pollutants first from the eroded soil and submerged farmlands. Zhang Sheng, Li Chongming, and Fu Yongchuan, etc., studied on the nutrients in the backwater areas of 12 main tributaries affected by the reservoir (Zhang et al. 2008) and found that these tributaries had been polluted in different degrees. TN, TP, and chlorophyll-*a* concentrations were in the range of 0.535–7.47 mg L^{-1}, 0.016–0.835 mg L^{-1}, and 1.38–23.7 mg m^{-3}, respectively, which were much higher than the outbreak condition of "water bloom." In addition, the flow velocities of these tributaries became slow down due to the influence of the main stream lockup. It became a favorable hydrological condition for the algal growth and the water bloom.

16.3.1.3 The Dynamic Variation of the Aquatic Community

Hu Zhengyu and Cai Qinghua (2006) studied on the changes of aquatic ecosystem before and after impoundment of the Three Gorges Reservoir and found the significant differences in the population structures and cell densities of algae in the main stream and tributary (the Xiangxi River) before and after the impoundment of the Three Gorges Reservoir. The algae increased from 7 phyla, 66 genera, and 79 species before water storage to 7 phyla, 85 genera and 151 species after. All the algae species except diatoms increased after the water impoundment, especially the green algae which had an evident rise. The average cell densities of algae in the main stream and tributary had an increase of 41.2 and 92.6%, respectively, than before. It showed that the construction of Three Gorges Reservoir had an impact on the algae community, especially on aquatic organism in the tributaries. On the river of Three Gorges reservoir, the Cladocera species collected near the sluggish shore increased from 2 to 20 just 1 week after the water storage and copepod species rose from 5 to 11. In the backwater area of the Xiangxi River, 26 species of zoobenthos were collected which were attached to 4 phyla and 6 families, including 2 species of Bivalve, 2 families and 13 species of Oligochaeta, 10 species of Chironomidae, and 1 species of Nematodes. The numbers of zoobenthos increased gradually with the running of the reservoir. The total of 21 kinds of fish were found in the backwater area of the Xiangxi River, most of them were fit for still water environment and lived in the middle and bottom layers of the reservoir. In species composition, the types of omnivorous fish (47.62%) were on the top, followed by carnivores (38.95%). As a result of rich organic detritus and humus in the submerged area, the omnivorous fishes were dominant, but the herbivorous species were few due to the deficiency of plant food. The intensive variations of the water environment after water storage had not formed suitable circumstances for the growth of filter-feeding fish. Due to the composition and construction of aquatic organisms in large size such as fish as "consumers" at the top of the food chain were not steady, the controlling capacity to algae as "producer" at the first food chain was weak. Therefore, the eutrophication and algae bloom break out easily in a new reservoir.

16.3.2 The Laohutan Reservoir

The Laohutan Reservoir is located in the upper reaches of Daixi, a branch river of Dongtiaoxi River, in Huzhou City of Zhejiang Province. It is used to drinking water supply and flood control. The reservoir catchments area is about 110 km^2 with 8.62×10^7 m^3 of the annual normal flow. The reservoir volume is 9.97×10^7 m^3, the regulating storage is 6.81×10^7 m^3, and the normal water level is 49 m.

Laohutan Reservoir started to collect water in May of 2008. It is going to supply drinking water in the year of 2009. In the first month of water storage, the reservoir's water was green and had a lot of floaters on the surface. From July to September, part of the water was exchanged due to water quality improvement and flood control. After several days of fine weather in late of September, the Cyanophyceae bloom occurred. The nutrients cycle is more simply in a new reservoir than in steady one because there are little aquatic plants and animals grown in it. The analyses of water quality variation in the first period of water storage explain the process of eutrophication and the reason of cyanobacteria bloom in a reservoir.

16.3.2.1 Assessment of Inflow Water Quality and Soil Before Water Storage

In September 11th and 12th (rainy period) and November 10th and 11th (drought period) of 2007, water samples in the entrance of 1# inflow stream were collected to analyze the concentrations of dissolved oxygen (DO), total nitrogen (TN), total phosphorus (TP), ammonia nitrogen (NH_3–N), nitrate nitrogen (NO_3^-N), permanganate value (COD_{Mn}), and chlorophyll-*a* (Chl-*a*). The concentrations of DO were high and NO_3^-N were low in rainy and drought periods as showed in Table 16.3. The concentrations of TP, TN, NH_3–N, and COD_{Mn} were all lower than the national standards of drinking water source [according to the Grade III of "Environmental Quality Standards for Surface Water" (GB3838-2002)].

The concentrations of TP were low in drought period, but exceeded the standards in rainy time. The concentrations of chlorophyll-*a* were 0.018~0.022 mg L^{-1} in September and 0.006~0.009 mg L^{-1} in November. It reached to eutrophic (≥ 0.01 mg L^{-1}) and mesotrophic state, respectively (Meng and Zhang 2007, Su et al. 1996, Wu et al. 2001, Xu et al. 2003). TP and chlorophyll-*a* are two dominant factors and indictors of most enriched lakes and reservoirs (Cai 1997, Han et al. 2003, Shang and Shang 2007). Therefore, the high concentrations of TP and chlorophyll-*a* in rainy period in the inflow stream indicated the eutrophication risk of the Laohutan Reservoir.

Before water storage of the reservoir, soil samples from the surface layer of the main used land of the submersed area such as mountain forest, bamboo land, paddy filed, and residential area were taken to analyze the pH value, organic matters, total nitrogen, total phosphorus, and ammonia nitrogen. The monitoring result in Table 16.4 showed that the bamboo soil had the highest contents of organic matters, total phosphorus, and ammonia nitrogen. The paddy soil had high contents of total nitrogen and the other nutrients. That was caused by the utilization of fertilizers and pesticides. The Laohutan Reservoir belongs to the Taihu Basin. Compared with the environmental background value of the paddy soil in Taihu Basin, total phosphorus and organic matters in the bamboo soil, total nitrogen in the paddy soil 36.7, 13.4, and 10 times, respectively, higher than the background values. The surface layers of bamboo and paddy soils in high contents of nutrients were cleaned out before the reservoir water storage to reduce the nutrients release from sediments and to prevent the reservoir eutrophication.

16.3.2.2 Water Quality Variation and Eutrophication Mechanism of Laohutan Reservoir

After water storage in May 30 of 2008, water qualities in the reservoir and the inflow stream were being detected continuously from June to September. Sample stations were located in the entrance of 1# inflow stream (S1), the front of dam (S2), and bridge (S3)

Table 16.3 Water quality of 1# inflow stream mg L^{-1}

Sampling Time Parameters	Sep.11th	Sep.12th	Nov.10th	Nov.11th	Standard
DO	9.40	10.07	9.89	10.81	5
TN	0.58	0.65	0.88	0.96	1.0
TP	0.07	0.06	0.023	0.024	0.05
NH_3-N	0.28	0.59	0.278	0.16	1.0
NO_3^--N	0.57	0.08	1.23	1.19	10
COD_{Mn}	2.10	2.41	1.26	1.15	6
Chl-a	0.018	0.022	0.009	0.006	

Table 16.4 Nutrient contents in the submersed soil of the Laohutan Reservoir

Sampling Parameters	Mountain Forest	Bamboo Land	Paddy Field	Residential Area	Background value
pH	4.97	6.31	5.32	6.76	
Organic Matters (g/kg)	9.2	36	17.3	9.0	2.68±0.667
TN (g/kg)	0.13	0.52	1.42	0.43	0.141±0.032
TP(g/kg)	0.32	1.95	0.86	1.20	0.053±0.006
NH_3-N(mg/kg)	17.38	44.56	27.76	21.26	

in the reservoir (Fig. 16.1). Samples were taken from the surface layer (0.5 m below surface), middle layer (1/2 depth), and bottom layer (0.5 m above bottom), respectively, in two stations in the reservoir. The analyses parameters were the same as Table 16.1 except pH value.

The serial monitoring results showed that the variation trends of TN, TP, NH_3–N, NO_3^-N, COD_{Mn}, and Chl-*a* concentrations in Laohutan Reservoir (mean results from the surface to bottom layer in two stations) were similar to those in the 1# flowing stream (Figs. 16.2, 16.3, 16.4, 16.5, 16.6, and 16.7). It meant that the branch water quality affected reservoir. In June, the concentrations of TN were 1.64 mg L^{-1} in the stream, and 1.49 mg L^{-1} in the reservoir, both of them exceeded the national standards. From July to September, the mean concentrations of TN in reservoir went down continuously. It was a little high in August in the stream, but not exceeded the national standard in reservoir due to the self-purification ability. The variation trends of TP were different from TN. TP in 1# stream and reservoir were low in June, but increased in July. It was 0.052 mg L^{-1} in 1# stream in July, a little higher than the national standard, but in the reservoir, it was below the standard. This meant that the reservoir also had the self-purification ability to TP. However, the ability was limited. The high concentration of TP in 1# stream (0.12 mg L^{-1}) in September led to the self-increasing in the reservoir (0.073 mg L^{-1}). It exceeded the level of an enriched reservoir according to the grading criterion of eutrophic lakes in China (Xing and Hu 2006, Qing et al. 2006). The concentrations of chlorophyll-*a* were rising rapidly and evidently with 0.012 mg L^{-1} in the surface layer of S2 station. Cyanobacteria bloomed first in Laohutan Reservoir. The concentrations of pollutants such as TN, NH_3–N, NO_3^-N, and COD_{Mn} were all low in September. TP was the only high nutrient in reservoir. This led to the rise of chlorophyll-*a*, and the mass growth and reproduction of cyanobacteria. Therefore, the high concentration of TP from inflow stream caused the reservoir eutrophication (Jiang and Pei 2007, Salvia et al. 2001,

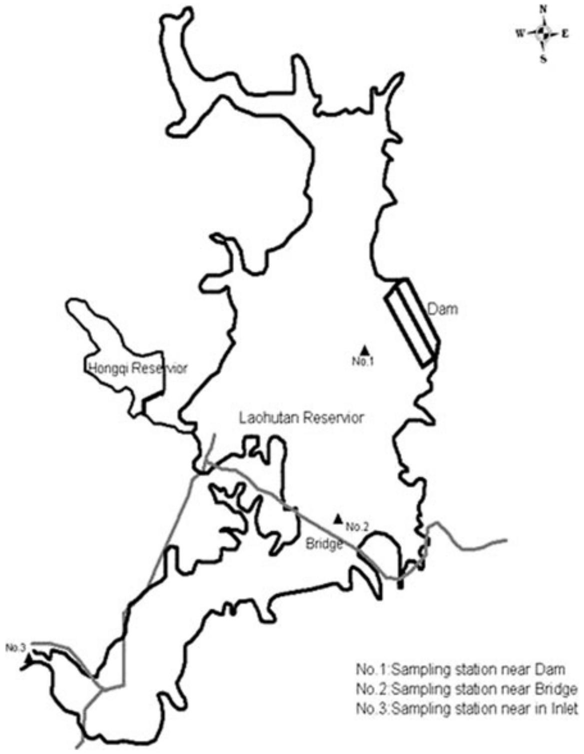

Fig. 16.1 Monitoring stations of water quality in the area of Laohutan Reservoir

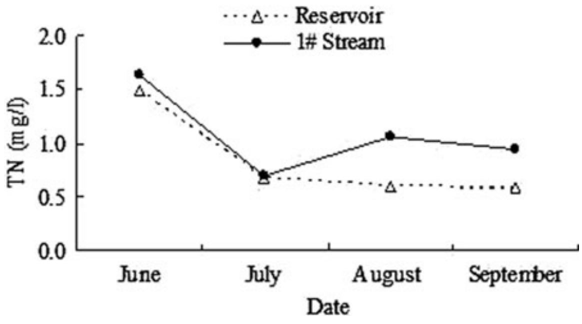

Fig. 16.2 Variation of TN concentrations in Laohutan Reservoir and 1# stream

Zhao and Han 2005) and cyanobacteria bloom. The changing processes of NH_3–N and COD_{Mn} in the reservoir were also similar to those in the stream, but they were higher in the reservoir than those in the stream (Figs. 16.4 and 16.6). This meant that part of NH_3–N and COD_{Mn} in the reservoir came from the release of submerged soil.

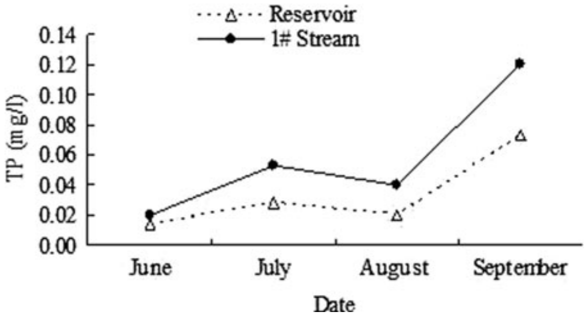

Fig. 16.3 Variation of TP concentrations in Laohutan Reservoir and 1# stream

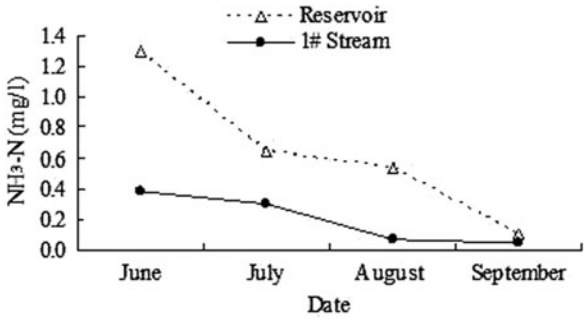

Fig. 16.4 Variation of NH_3–N concentrations in Laohutan Reservoir and 1# stream

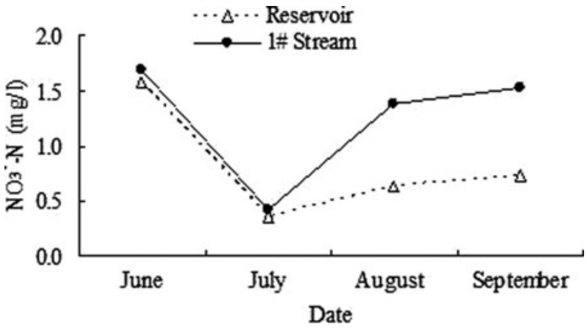

Fig. 16.5 Variation of NO_3–N concentrations in Laohutan Reservoir and 1# stream

The land utilization types before water storage of the reservoir included resident area, paddy field, bamboo field, and forest. The pollutants would be released from soil after water impounding. The release intensity was large in the first period of water storage, but started decreasing gradually with the reservoir operation until a balance was reached (Cui and Hao 2003,

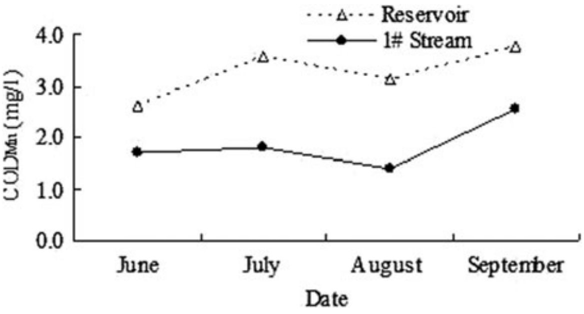

Fig. 16.6 Variation of COD_{Mn} concentrations in Laohutan Reservoir and 1# stream

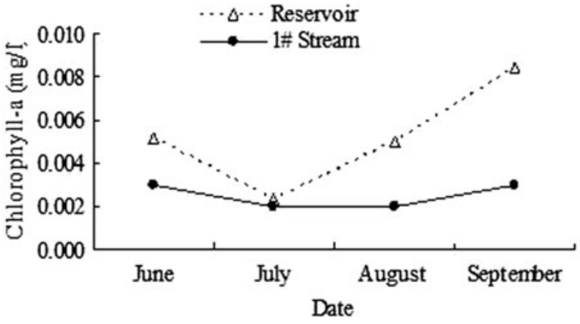

Fig. 16.7 Variation of chl-*a* concentrations in Laohutan Reservoir and 1# stream

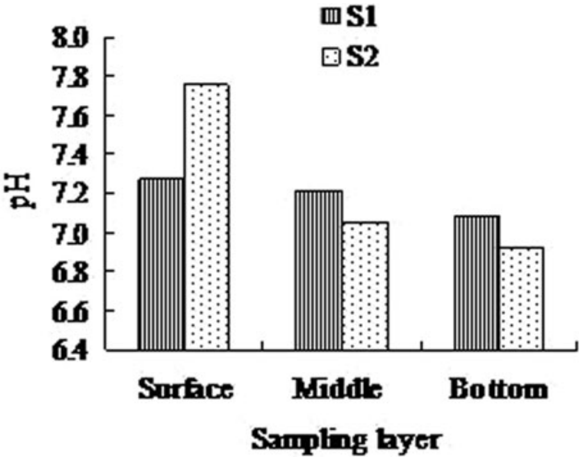

Fig. 16.8 Vertical distribution of pH in the reservoir in June

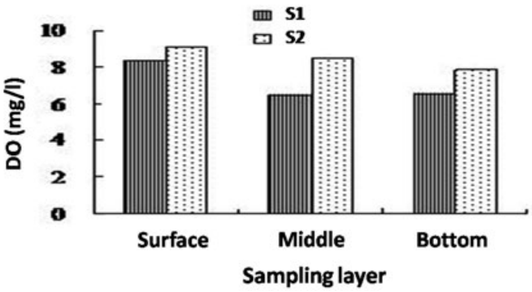

Fig. 16.9 Vertical distribution of DO in the reservoir in June

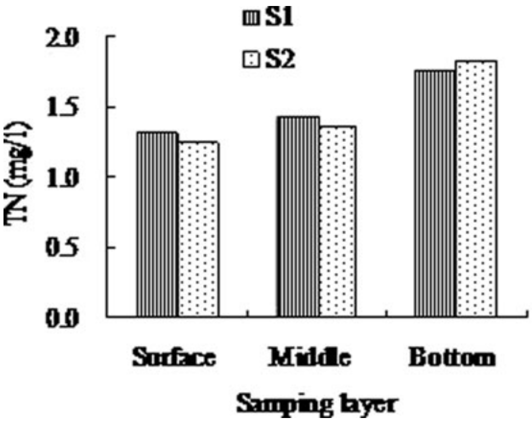

Fig. 16.10 Vertical distribution of TN in the reservoir in June

Yuan et al. 2008, Serena et al. 2007). Study on the vertical distributions of pollutants in June, just 1 month after water storage, was helpful to analyze the impact of pollutants released from submerged soil on water quality, because the release had been in steady status. The distributions of pH value, DO, TP, TN, and NH_3–N concentrations from the surface layer to the bottom in June were showed in Figs. 16.8, 16.9, 16.10, 16.11, and 16.12.

The concentrations of pH value and DO had decreased gradually (Figs. 16.8 and 16.9). It meant the decomposition of chemical matters would consume oxygen and produce acid in the bottom. The concentrations of TP, TN and NH_3–N all increased from the upper layer to the bottom (Figs. 16.10, 16.11, and 16.12). NH_3–N concentrations in all of the monitoring points and TN, TP concentrations in a part of sampling points were higher than those in 1# inflow stream. The phenomena above indicate to the effect of nutrients release from soil on water quality of the reservoir at the first period of water storage. But the influence will weaken with the storage and discharge alternation. The vertical distributions of pH value, TP, TN, and NH_3–N were not change significantly after June except DO.

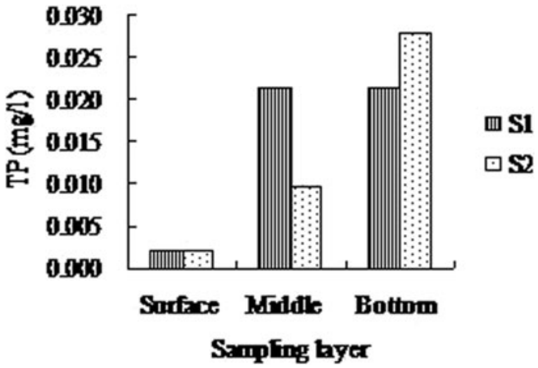

Fig. 16.11 Vertical distribution of TP in the reservoir in June

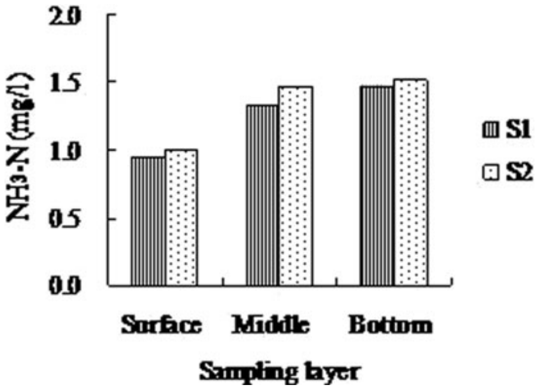

Fig. 16.12 Vertical distribution of NH_3–N in the reservoir in June

16.3.2.3 Result and Discussion

The variation trends of TN, TP, NH_3–N, NO_3–N, COD_{Mn}, and Chl-*a* in the reservoir were similar to those in the 1# flowing stream. It showed that the branch water quality affected the reservoir. However, NH_3–N concentrations exceeded the national limits in the reservoir in June, but it was low in 1# flowing stream. Also the concentrations of COD_{Mn} were always higher in the reservoir than those in the stream. The vertical distributions trends of TP, TN, and NH_3–N all exhibited an upward trends from the upper layer to the under layer in the reservoir in June. The phenomena above meant that the nutrients released from the soil affecting the reservoir water quality. But the release effect weakened with the operation of the reservoir and about 1 month later, the inflow stream was the main factor to affect the water quality. High concentrations of TP from the stream resulted in the rapid rise of chlorophyll-*a*. Mass of cyanobacteria occurred in September and led to the deterioration of the reservoir water quality. Therefore, Laohutan Reservoir had been in an unhealthy eutrophic state caused by the high concentration of TP coming from the stream. The runoff sources of Laohutan reservoir were the inflow stream and rainfall. There were about 8,000 residents, many small enterprises and poultry cultivation in the upper reaches area. They were the sources of pollutants. Also, the non-point pollutants from fields influenced the reservoir water quality. Therefore, treatment of point and non-point source pollutants and improvement of water quality in drainage basin are the key to control eutrophication of Laohutan Reservoir.

16.3.3 Comparison of the Similar Reservoirs

16.3.3.1 Comparison of the New Reservoir with an Old One

Hongqi Reservoir, one of the water supply sources for the Laohutan Reservoir, was constructed in the 1960s of last century. The reservoir has established a steady ecosystem after a longtime operation. The Hongqi Reservoir is about 500 m far from the Laohutan Reservoir (Fig. 16.1). They are in the same geographical conditions such as the landform, hydrology, soil, and climate. The water qualities of the Hongqi Reservoir were monitored in September of 2007 and 2008 (Table 16.5). All of the parameters except TP met the national standards of drinking water source [according to the Grade III of "Environmental Quality Standards for Surface Water" (GB3838-2002)]. The concentration of TP exceeded the national standards, and the concentrations of chlorophyll-*a* were all higher in the Hongqi and Laohutan reservoirs in September of 2008. Compared with the nutrients in the Laohutan Reservoir, the concentrations of organic matters, TN, TP, NH3–N and chlorophyll-*a* were all higher in the Hongqi Reservoir. The Laohutan Reservoir was mesotrophic while the Hongqi Reservoir was eutrophic according to the chlorophyll-*a* concentrations (Table 16.6). No Cyanobacteria bloom occurred in the Hongqi Reservoir, however, it took place in the bay and in the front of the dam in the Laohutan Reservoir after only 4 months running. The phenomena above showed that the simple and unsteady ecosystem of a new reservoir is weakened to resist nutrients enrichment and Cyanobacteria development.

Table 16.5 Water quality in Hongqi and Laohutan Reservoirs

Sampling station	Sampling time	TN (mg L^{-1})	TP (mg L^{-1})	NH_3–N (mg L^{-1})	COD_{Mn} (mg L^{-1})	BOD_5 (mg L^{-1})	Chl-*a* (mg m^{-3})
Hongqi	2007.9	0.45	0.06	0.48	3.38	0.26	
Reservoir	2008.9	0.736	0.109	0.338	4.49	1.57	28
Laohutan Reservoir	2008.9	0.687	0.073	0.117	3.78	1.54	9
National standard		≤1.0	≤0.05	≤1.0	≤6	≤4	

Table 16.6 Concentrations of Chl-*a* in each nutrients enrichment degree (mg m^{-3})

Nutrients enrichment type	Dystrophic	Oligotrophic	Mesotrophic	Eutrophic
Yoshimura (1937)		<4	4–10	>10
Seirgensew (1980)	0.01–0.5	0.3–3	2–15	10–500

16.3.3.2 Comparison of Two New Reservoirs

The Three Gorges Reservoir is a famous, large, and new reservoir. Compared with the lake-like reservoir of the Laohutan reservoir, the Three Gorges Reservoir is a river-like one. However, with the rising of water level, the broadening of water surface and the slowing down of the flow velocity after water storage, nutrients such as organic matters, nitrogen, and phosphorus accumulated in the entrance of river and backwater zone, water eutrophication and Cyanobacteria bloom occurred in these areas. There are different effects of nutrients on eutrophic degree of different water body. The main stream of the Yangze River is in oligotrophic and mesotrophic state, but, some estuaries and backwater areas have become eutrophicated. Therefore, Cyanophyceae bloom and eutrophication take place easily in a new reservoir, even with the little nutrients accumulation, and no matter the reservoir is like a lake or like a river. Because a new reservoir has low biodiversity and the ecosystem is sensitive. With the suitable temperature, sunlight, hydrodynamic force, and nutrients, the Cyanobacteria bloom will occur in part or the whole reservoir.

16.4 Control Methods of Eutrophication

Lakes and reservoirs are used to store and supply water for industry, agriculture, domestic, and fishery which also have the functions of maintaining biodiversity, regulating regional climate, storing flood, modulating surface water runoff, and purifying water. However, eutrophication has been one of the focus problems for environmental protection.

16.4.1 Reducing the Importing Nutrients

The importing nutrients from the drainage basin is the main reason for eutrophication in a new reservoir. The first and important measure to control the nutrients enrichment is to reduce the importing nutrients.

16.4.1.1 Industrial Pollution Control

Cleaner production strategies are adopted by industrial enterprises who implemented effluent standards and total amount control of pollutants. What's more, authority concerned should shut down small enterprises with serious pollution, strengthen the management of water abstraction licensing, restrict on water consumption in high quantity, and serious pollution and carry out strict management of drainage water at the same time.

16.4.1.2 Agricultural Pollution Control

With the development of modern agriculture, in order to increase crop yield, the amount of fertilizer and pesticides application is constantly increasing. The improper mode of application has led to inefficient use of fertilizers, of which a large number of N, P was taken into the water body by surface runoff, resulting in an increasing storage of N and P in water (Jiang and Pei 2007, Salvia et al. 2001, Zhao and Han 2005). Fertilizer, pesticides used in reservoir areas increase the potential risks of eutrophication. Therefore, the rational application of fertilizer can decrease the loss of nutrients. Vigorous efforts should be made to develop eco-agriculture, and new agricultural technologies should be adopted in order to control nutrients sources. Farmers should be guided and encouraged to improve the measures of fertilization and irrigation, to cultivate crops reasonably, to minimize the amount of pesticide and chemical fertilizers, and to reduce the impact on water quality brought by agricultural production.

16.4.1.3 Domestic Pollution Control

The main measure is to control waste water from urban residents around reservoirs. Moderate-scale sewage collection and treatment facilities should be built to purified the domestic waste water centrally, removing suspended solid, oil, nitrogen, and phosphorus. At the same time, rural feces, domestic garbage, and other solid wastes should be treated carefully. In addition, the water polluted from tourists should also be collected and treated before discharging.

16.4.2 Endogenous Nutrients Control

New reservoirs are built by flooding a large area of vegetations, villages, fertilized agricultural lands, and so on. The flooded area soil rich in nitrogen, phosphorus, and other nutrients should be cleaned up in order to reduce risks of water quality deterioration and possibilities of the nutrients release. The means of endogenous nutrient control include biological measures and engineering measures.

16.4.2.1 Biological Measures

In accordance with the order of nature, water self-purification abilities can be strengthened to remove pollutants by biological measures. Some studies have shown that the existence of submerged vegetation can effectively preclude the flourish of algae and control the further eutrophication development. Moreover, it can improve the living environment for other organisms, increase biological diversity, and maintain the ecological balance of water bodies (Ma and Li 2002, Serena et al. 2007, Yuan 2004, Yuan et al. 2008).

16.4.2.2 Engineering Measures

At present, the engineering measures mainly include sediments dredging, deep aeration, and water diversion. For example, Xuanwu Lake, Dianchi Lake, and others in China successfully used sediments dredging to improve water quality. Deep aeration is used to supply oxygen through aeration, maintaining an aerobic state between water and sediments, so as to suppress phosphorus release from sediments. Water diversion is a method used to dilute the eutrophicated lakes in order to reduce the pollutants concentrations (Chen et al. 2004, Zhong 2005, Feng and Wu 2006). For a new reservoir, dredging rich nutrient soil before water impoundment and increasing water exchange frequency are effective methods to control water eutrophication.

16.4.3 Construction of a Stable Ecosystem

In the early stage of new reservoirs, types and biomass of large aquatic plants are scarce, which are extremely unfavorable for pollutants purification, water ecosystem balance, and suppression of algae blooms. Establishing and perfecting the submerged aquatic plants are significant to form a healthy and stable water ecosystem. Through ecological construction projects to achieve the continuity of near-shore plant communities, a good living environment can be created for aquatic animals. After water storage, fish fry, snail, mussels, or other benthic organisms are raised at a certain amount in reservoirs (bait feeding must be forbidden and aquatic creatures grow naturally). Phytoplankton is controlled through filter feeding of fish and benthic organisms so as to lower eutrophication indicators in part and improve water quality. Construction of the water conservation forest and vegetation around reservoirs, by afforestation, returning farmland to forests and grasslands, as well as bank collapse control, grazing prohibition and other ecological environment restoration projects. Ecological protection zones and buffer strips of vegetation are gradually built to reduce non-point source pollutants from surface runoff.

16.4.4 Ecological Scheduling of Reservoir

Main factors leading to eutrophication in reservoir can be divided into nutritional factors (N, P, etc.), environmental factors (temperature, light, transparency, etc.) and hydrological factors. Studies have shown that the area, volume, depth, shoreline, water exchange period, water level variation, runoff flow, and velocity are closely related with eutrophication (Qing 2007, Wagner 2000, Zhao and Cai 2004). So it is necessary to study the impact of hydrological, meteorological and other processes on algae and other aquatic life in spatio-temporal patterns further. According to different hydrological and ecological characteristics, ecological operation of reservoirs should be done combined with flood control, power generation, management of downstream channel, and environmental protection to control algae bloom (Tang et al. 2004, Cai and Hu 2006).

16.4.5 Water Quality Monitoring

The water quality monitoring of reservoirs and rivers must be strengthened after water storage. It is very important to establish a network of environmental monitoring in the main rivers inflow and reservoir, so as to observe the variation of water quality in time. The monitoring result can provide accurate information and scientific basis for prevention of water bodies eutrophication, so that the local authority formulates practical management policy and control strategies.

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Chapter 17

Plant Nutrient Phytoremediation Using Duckweed

Louis Landesman, Clifford Fedler, and Runbin Duan

Abstract Over the last 40 years a great deal of research has been published on the use of duckweed to treat wastewater both from point sources (feedlots, food processing plants) and from non-point sources. These plants can recover nutrients such as nitrogen and phosphorus from contaminated waters in those agricultural practices. They can also remove or accumulate metals, radionuclides, and other pollutants in their tissues. In addition, the duckweed can be used as a feed source for livestock and poultry as well as an energy source for biofuel production. A summary of some of the published work done using duckweed species to phytoremediate natural, domestic, industrial, and agricultural wastewaters is presented.

Keywords Duckweed · Plant nutrients · Phytoremediation · Lemnaceae · *Lemna* · *Wolffia*

17.1 Introduction and Background of Duckweed

Duckweeds belong to the arum family Araceae, subfamily Lemnoideae, a family of floating, aquatic plants. This family consists of five genera with at least 40 species identified as of 1997 (Les et al. 2002). Duckweeds are among the smallest and simplest flowering plants, consisting of an ovoid frond a few millimeters in diameter and a short root usually less than 1 – cm long (Fig. 17.1). The frond represents a fusion of leaves and stems and represents the maximum reduction of an entire vascular plant (Landolt 1986). Some species of the genus *Wolffia* are only 2 mm or less in diameter; other *Lemna* spp. have frond diameters of about 5 – 8 mm. The largest species of Lemnaceae have fronds measuring up to 20 mm in diameter (*Spirodela* sp.). The minute flowers are rarely found in most species. Under adverse conditions such as low temperatures or desiccation, modified fronds called turions appear which sink to the bottom of the water body. These turions can resurface at the onset of favorable conditions of light, moisture and temperature to start new generations of duckweed plants (Hillman 1961, Perry 1968). Because flowering in Lemnaceae is rare, reproduction normally occurs by budding from mature fronds. The tolerance of Lemnaceae fronds and turions to desiccation allows a wide dispersal of Lemnaceae species. This low level of gene flow and infrequent sexual reproduction has produced substantial levels of genetic divergence among populations, despite an absence of morphological differentiation (Cole and Voskuil 1996). However, asexual reproduction in Lemnaceae allows for rapid reproduction in this family. Occasionally extreme weather events, such as unusually high summer temperatures, can cause mass flowering (Bramley 1996). Usually flowering has to be induced with plant hormones or photoperiod manipulation (Cleland and Tanaka 1979). All Lemnaceae flowers are minute and barely discernable without magnification (Landolt 1986).

Due to its ease of culture and worldwide distribution, a tremendous literature exists on duckweed ecology, physiology, production, and systematics. Landolt and Kandeler's two monographs on Lemnaceae are the

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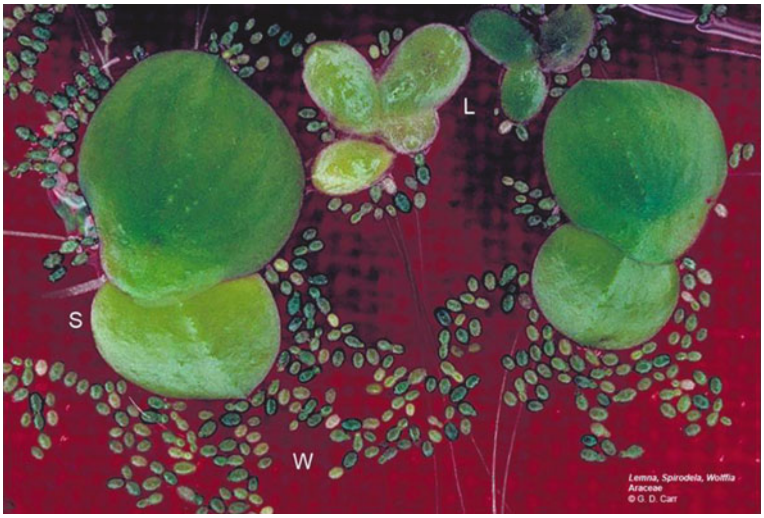

Fig. 17.1 *Spirodela* (*large*), *Wolffia* (*small*), and *Lemna* (*intermediate*)

Lemna, Spirodela, Wolffia
Araceae
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most comprehensive works on Lemnaceae and list virtually all published works up to 1986 (Landolt 1986, Landolt and Kandeler 1987). In addition there are several web sites that have more updated information on duckweed biology and applications (Cross 2007, Landesman 2008).

The genera *Lemna*, *Spirodela*, and *Wolffia* of the family Lemnaceae play an important ecological role in lakes, ponds, and wetlands. They often are an important source of food for waterfowl (Krull 1970) and aquatic invertebrates. The outer margins of duckweed fronds (phyllosphere) support dense populations of diatoms, green algae, rotifers, and bacteria (Coler and Gunner 1969). Associated with this epiphytic community is an assortment of insects, including beetles, flies, weevils, aphids, and water striders (Scotland 1940). Some of these insects may become abundant enough to affect the duckweed population. Together with the frond biomass this microfauna enhances the nutritive value of duckweed to grazing animals such as ducks, geese, nutria, turtles, coots, fish, and snails, all of which have been recorded as feeding on duckweed.

Duckweed populations are limited mostly by light, nutrients, and temperature (Hillman 1961). Duckweed populations can grow very densely in nutrient-rich environments, so much so that layers of fronds grow one on top of another to form a mat that can be up to 6 – cm thick. This thick mat creates an anaerobic environment in the water body on which this mat floats, thus promoting anaerobic digestion and denitrification of the water body in which the duckweed grows. Since duckweed floats freely on water surfaces, strong winds can sweep fronds from the water surface.

The presence of duckweed in an aquatic environment has both direct and indirect effects on that environment. When duckweed is abundant enough to completely cover a pond, ditch, or canal, this layer of opaque fronds can shade out rooted aquatic macrophytes (Janes et al. 1996) as well as reduce phytoplankton abundance. In eutrophic environments such as the polders of Holland, *Lemna* sp. can form a climax community that prevents *Chara* and other submerged macrophytes from getting established (Portielje and Roijackers 1994). A complete cover of duckweed on the water surface can lead to the creation of an anaerobic environment in the water column, which in turn can make that water body inhospitable to fish and aquatic insects (Pokorny and Rejmankova 1983, Leng et al. 2004).

The presence of duckweed can contribute to the organic matter present in a water body. Layers of *Lemna minor* L. excrete amino acids and humic substances into the aquatic environment which can provide nutrients to other organisms such as bacteria, epiphytic algae, and indirectly to snails, springtails, isopods (*Asellus* sp.), and other microdetrivores (Thomas and Eaton 1996). Dead and dying duckweed fronds fall to the bottom of the water column where their decay contributes organic matter, nitrogen, phosphorus, and other minerals to the benthos (Laube and Wohler 1973). In addition cyanobacteria residing in the phyllosphere of duckweed fronds can fix atmospheric nitrogen, providing a nitrogen input in oligotrophic environments (Tran and Tiedje 1985). This can be an important source of nutrients in aquatic environments.

Duckweeds are among the fastest growing aquatic angiosperms in the world, frequently doubling their biomass under optimum conditions in 2 days or less (Culley et al. 1981). Based on growth rates recorded in the literature, duckweeds can grow at least twice as fast as other higher plants (Hillman 1978). Depending on the genus, duckweed daughter fronds are produced vegetatively in pairs (*Lemna* and *Spirodela*) or as a daughter frond from the basal end of the mother frond (*Wolffia*). Each daughter frond repeats the budding history of its clonal parents, resulting in exponential growth (Landolt 1987). *Lemna*, *Spirodela*, and *Wolffia*, three important genera of Lemnaceae, are all subject to self-shading (intra-specific competition) and reach a steady – state condition where frond death equals frond multiplication. Hence Lemnaceae is subject to density-dependent growth (Ikusima 1955, Ikusima et al. 1955). Once essential nutrients are depleted or waste products build up, the growth rate declines.

When duckweed was cultured in axenic (sterile) conditions using chemically defined media under artificial lights, growth rates were recorded that far exceeded growth rates measured under natural conditions (Hillman 1961). Excessively high light levels (more than 200 $\mathrm{Wm^{-2}}$), nutrient shortages, and the presence of herbivores, parasites, and commensal organisms antagonistic to duckweed populations greatly reduce the growth rates of duckweeds in natural environments (Landesman 2000). Duckweed growing in wastewater treatment plants, however, is under less pressure from herbivores because the high ammonia and low dissolved oxygen levels prevalent in wastewater may exclude potential grazers such as fish and turtles. Wastewater environments also have abundant supplies of nitrogen and phosphorus as compared to natural aquatic environments.

17.2 Duckweed for Phytoremediation of Contaminated Waters

Phytoremediation is defined as the method to utilize higher plants to alter contaminated environments. It is a cost-effective, low-impact, and environmentally sound remediation technology (Cunningham and Ow 1996). And phytoremediation includes five different mechanisms, which are rhizofiltration, phytostabilization, phytoextraction, phytovolatilization, and phytotransformation (Ghosh and Singh 2005). Rhizofiltration is that plants are used to absorb, concentrate, and precipitate contaminants from polluted aquatic environment by their roots; phytostabilization involves the stabilization of contaminated soils by sorption, precipitation, complexation, or metal valence reduction rather than the removal of contaminants; phytoextraction, also referred as phytoaccumulation, is the process that plants absorb, concentrate, and precipitate the contaminants in the biomass; phytovolatilization is the mechanism that plants extract certain contaminants in nearby roots and then transpire them into the atmosphere; phytotransformation, also referred as phytodegradation, is the process that plants remove contaminants from environment by their metabolism. More detailed information on these five different mechanisms is listed into Table 17.1.

17.2.1 As an Alternative Means of Wastewater Treatment

Duckweed has been utilized in the treatment of municipal and industrial wastewaters for more than two decades, which can be traced back to before 1990 (Oron et al. 1988). Duckweed is widely and effectively used for phytoremediation of contaminated water due to its ability to grow at wide ranges of temperature, pH, and nutrient level (Landolt and Kandeler 1987) in areas where land is available for its application (Krishna and Polprasert 2008). Considerable work was done in the 1970s and 1980s on the use of duckweed genera, especially *Lemna*, as a means of treating wastewater of both agricultural and domestic origin. When *Lemna* is grown in wastewater treatment ponds the floating mat of fronds is held in place by partitions and baffles that prevent wind from blowing fronds to one side off or completely off the surface of the treatment pond. These partitions and baffles are usually made of polyethylene in industrialized countries but may be made of bamboo or other natural materials in developing countries.

Table 17.1 Contaminant removal processes and mechanisms by phytoremediation

	Rhizofiltration	Phytostabilization	Phytoextraction	Phytovolatilization	Phytotransformation
Mechanism	Rhizosphere accumulation, absorption, concentration, precipitation	Complexation, sorption, precipitation, metal valence reduction	Hyper-accumulation, absorption, concentration, precipitation	Volatilization	Degradation by plant metabolism
Contaminant	Organics/inorganics, Pb, Cd, Cu, Zn, Cr, Ni	Inorganics, heavy metals	Inorganics, heavy metals	Organics/inorganics, Hg, Se	Organics, ammunition wastes, chlorinated solvents, herbicides
Environment	Industrial discharge, agricultural runoff, acid mine drainage	Soil, sediment, sludge	Diffusely polluted areas	Soil, water, sediment	Soil, water, groundwater
Reference	Chaudhry et al. (1998), USEPA (2000), Ghosh and Singh (2005)	Mueller et al. (1999), USEPA (2000), Ghosh and Singh (2005)	Rulkens et al. (1998), USEPA (2000), Ghosh and Singh (2005)	Bañuelos (2000), Henry (2000), Ghosh and Singh (2005)	Black (1995), Ghosh and Singh (2005)

As part of a facultative treatment system, duckweed can cover treatment ponds and reduce the growth of algae in these ponds as well as reduce nitrogen in the effluent from these ponds through ammonia uptake and denitrification (Alaerts et al. 1996; Hammouda et al. 1995). Duckweed can also be part of constructed wetland systems, either as a component of a wetland receiving wastewater or as plants that polish nutrients from wetland-treated effluents (Ancell 1998, Fedler et al. 1999, WEF 2001).

Harvesting wastewater-grown duckweed helps to remove surplus nutrients, which might otherwise be released into aquatic environments by wastewater treatment plants (Harvey and Fox 1973, Oron et al. 1988). Duckweeds, like other plants, take up nutrients from their surrounding environment (Landesman 2000). This ability has been exploited to remove surplus nutrients from swine lagoon effluents (Cheng et al. 2002b). The growing plants can then be harvested to remove surplus nitrogen and phosphorus. However, the application of duckweed in recovery (Cheng et al. 2002a) and removal of nitrogen and phosphorus in swine lagoon water was found to be subject to the water concentrations and seasonal climate since the primary mechanism is assimilation of those nutrients in environment; therefore, the appropriate light intensity and preferable temperature are key parameters for duckweed in removal of surplus nutrients (Cheng et al. 2002b), and duckweed prefers to take up NH_4^+ than NO_3^- by both roots and fronds (Fang et al. 2007).

Duckweed populations can remove nutrients from stormwater ponds. A monoculture of *L. minor* consistently removed a large amount of ammonia from stormwater while a mixture of *L. minor* and *Spirodela polyrhiza* removed the largest amount of phosphorus from stormwater within 8 weeks of treatment (Perniel et al. 1998). Recently, Drenner et al. (1997) have described a system for culturing periphyton on eutrophic effluents and raising fish that graze on this wastewater-grown periphyton. In this way, surplus nutrients are concentrated in fish flesh. A similar system could be designed using duckweed as the nutrient stripping plant (van der Steen et al. 1998).

Duckweed systems can remove 50–60% of nitrogen and phosphorus (Vatta et al. 1994) from domestic wastewater or even 73 – 97% of total Kjeldahl nitrogen and 63–99% of total phosphorus in duckweed-covered domestic wastewater (Körner and Vermaat 1998). The removal of chemical oxygen demand (COD) is faster in duckweed-covered domestic wastewater than uncovered wastewater, and organic degradation can be improved by additional oxygen supply and additional surface in duckweed-covered domestic wastewater (Körner et al. 1998). The removal efficiencies can be reached at high to 84, 88, 68, 58, and 87%

for COD, BOD5, NH_3^-N, TN, and TSS, respectively, in duckweed-based wastewater treatment system under optimum operating and environmental conditions (Krishna and Polprasert 2008). Furthermore duckweed systems evaporate 20% less water compared to other open water wastewater treatment systems (Oron et al. 1986, Borrelli et al. 1998). The reduced evaporation of duckweed-covered surfaces in wastewater treatment is an asset in arid climates.

Guidelines for the use of duckweed to remove ammonia and phosphorus from effluent from an algae culture system were given by Koles et al. (1987). Researchers at the Politecnico di Milano, Italy, have developed models for duckweed-based wastewater treatment plants (Boniardi et al. 1994, Rota et al. 1995). These models will greatly assist in the design and management of duckweed-based wastewater treatment systems (Landesman et al. 2005). Duckweed-based treatment systems have their limitations. They require large areas of land that may not be available near urban areas. In temperate climates duckweed growth slows in the winter. This may restrict the use of such treatment systems in cooler climates unless a greenhouse system is utilized. Duckweedbased treatment systems may be most useful in treating secondary effluents from small communities where land costs are low (Bonomo et al. 1997).

A series of investigations on duckweed application in restoration of eutrophic water were done in the past decades. Eutrophic water is associated with excessive nitrogen and phosphorus in water input by discharge from agricultural wastewater, industrial water, and domestic water. Eutrophic water had the risk of eutrophication defined as the negative effects of the excessive growth of phytoplanktons (Khan and Ansari 2005), degradation of water ecosystems, or even disappearance of the water body involved in. Duckweed was used to remove the targeted nutrients in eutrophic water due to its ability to survive in nutrient-laden environments and its rapid growth (Li et al. 2009) so that those nutrients can be removed by harvesting duckweed biomass (Li et al. 2007) and eutrophic water can be recovered by combining other technologies. The duckweed *L. minor* is suitable for phytoremediation of eutrophic waters at acidic pH and at temperature from 20 to 30°C (Ansari and Khan 2008); however, the duckweed *S. polyrhiza* cannot be used to recover the eutrophic waters at low temperature of 10–12°C (Song et al. 2006). Many mathematical models have been developed for duckweed systems to describe its phytoremediation of eutrophic waters (Frédéric et al. 2006); those models incorporated duckweed growth parameters including temperature, photoperiod, nitrogen concentration, phosphorus concentration, and mat density (Lasfar et al. 2007).

17.2.2 As a Means of Removing Heavy Metals and Other Toxic Elements in Waters

Heavy metals are readily accumulated and transported in aquatic environment in the form of dissolved or solid wastes from domestic, industrial, and agricultural runoff (Megateli et al. 2009). Heavy metal contamination in environment can be cost-effectively removed by phytoremediation. Such a technology is most suitable for developing countries (Ghosh and Singh 2005). Generally, heavy metal cannot degrade or decompose as other contaminants; therefore, their removal by phytoremediation mainly depends on phytoextraction mechanism. In recent years, there were many findings reported on the removal of heavy metals by duckweed phytoremediation.

Khellaf and Zerdaoui (2009) addressed that the duckweed *Lemna gibba* L. can be successfully employed to remove Zn from contaminated water by 61 – 71%. Another research found that the duckweed *L. gibba* could remove Zn and Cu rapidly in the first 2 days with concentration reduction higher than 60% and then slowly in the following 8 days with reduction of 10 – 20%; however, the removal of Cd was linear and determined by initial Cd concentration and the removal was about 90% after 6 or 8 days with initial concentrations of 0.1 or 0.001 mgL^{-1} (Megateli et al. 2009). Duckweed phytoremediation has its limitation in heavy metal removal due to heavy metal's toxicity. Hou et al. (2007) stated that Cd^{2+} was more toxic than Cu^{2+} for the duckweed *L. minor*; the tolerance levels of Cd and Cu were smaller than 0.5 and 10 mgL^{-1}, respectively, and *L. minor* was recommended to phytoremediate low-level contaminated waterbody by Cu and Cd.

S. polyrhiza was found to have a large capability for the uptake and accumulation of heavy metals, surpassing that of algae and other angiosperms. For example, the zinc concentration in frond tissue was 2,700 times higher than that of its medium (Sharma and Gaur 1995). Under experimental conditions *L. minor* proved to be a good accumulator of cadmium and copper and a moderately good accumulator of chromium. Duckweed can accumulate other toxic elements such as selenium (Ornes et al. 1991), technetium (Hattink 2000), lead (Jain et al. 1990, Kruatrachue et al. 2002), uranium, and arsenic (Mkandawire et al. 2004). The growth rates and ease of harvest make duckweed species useful for phytoremediation of certain heavy elements as compared to many algal species that require much more extensive harvesting equipment (Zayed et al. 1998). Duckweed can therefore prove useful in treating effluents from mining operations. However, heavy metal concentrations can depress duckweed growth reducing its effectiveness in removing toxic elements from the water body in which it grows (Boniardi et al. 1999). The duckweeds *L. minor* (Alvarado et al. 2008), *L. gibba* (Marín and Oron 2007, Sasmaz and Obek 2009), and *S. polyrhiza* L. (Rahman et al. 2007) investigated for their phytoremediation ability to remove arsenic, boron, and uranium in water; *L. gibba* was found to be a suitable candidate used for the treatment of water containing boron with concentration lower than 2 mgL^{-1} (Marín and Oron 2007) and to accumulated arsenic (133%), uranium (122%), and boron (40%) (Sasmaz and Obek 2009); *L. minor* had good treatment of water with arsenic lower than 0.15 mgL^{-1} (Alvarado et al. 2008); *Spirodela polyrhiza* L. was identified as a good arsenic phytofiltrator by physico-chemical adsorption mechanism (Rahman et al. 2007).

17.2.3 As a Means of Removing Toxic Organic Compounds from Wastewater

Duckweed species can accumulate toxic organic compounds such as phenols, chlorinated phenols, pharmaceuticals, and surfactants. Duckweed species can do this directly or indirectly through microbiota living on frond surfaces. For example, surfactants like alkylbenzene sulfonate and alcohol ethoxylate are mineralized by duckweed microbiota (Federle et al. 1989). Duckweed can take up fluorinated agricultural chemicals (Reinhold 2006) and detoxify chlorinated phenols (Barber et al. 1995). The duckweed *S. oligorrhiza* L. wash proven to have the ability to uptake and transform DDT and organophosphorus pesticides (Gao et al. 2000a,b). The ability of duckweed to perform reductive dechlorination can be used in phytoremediation of industrial wastewaters (Ensley et al. 1997). Duckweed species definitely have the potential to contribute to natural systems of bioremediation.

17.3 Duckweed's Other Practical Application

In addition to the application for phytoremediation of contaminated waters, duckweed has been developed for other applications. Duckweed can be used as livestock food, toxicity testing, and raw material for biofuel production.

17.3.1 As a Source of Livestock Feed

The value of duckweed as a source of feed for fish and poultry has been promoted by the World Bank, especially in developing countries (Skillicorn et al. 1993). Research at Louisiana State University demonstrated the value of using dried duckweed fronds as a feed source for dairy cattle and poultry (Culley et al. 1981). Research at Texas Tech University has shown that duckweed species have potential as a feed ingredient for cattle, sheep, and pigs (Johnson 1998, Moss 1999). Duckweed also has potential as a feed ingredient in fish farming (Gaigher et al. 1984).

A great deal of work has been done on the nutritional value (Table 17.2) of species of Lemnaceae, especially *Lemna*, *Spirodela*, and *Wolffia* (Rusoff et al. 1980, Landesman et al. 2004). Duckweed has

Table 17.2 Chemical composition of *L. gibba* meal (% dry matter)

Chemical composition	Dry matter (%)
Dry matter	3.5
Crude protein	41.7
Crude fat	4.4
Acid detergent fiber	15.6
Non-fiber carbohydrate	17.6
Ash	16.2

Table 17.3 Amino acid composition of dried *L. gibba* (g amino acid/100 g dry *L. gibba*)

Amino acid	g amino acid/100 g dry *L. gibba*	Amino acid	g amino acid per/100 g dry *L. gibba*
Taurine	0.03	Methionine	0.64
Aspartic acid	3.51	Isoleucine	1.66
Threonine	1.68	Leucine	2.89
Serine	1.39	Tyrosine	1.27
Glutamic acid	3.67	Phenylalanine	1.75
Proline	1.42	Histidine	0.73
Glycine	1.93	Ornithine	0.05
Alanine	2.30	Lysine	1.85
Cysteine	0.44	Arginine	2.14
Valine	2.12	Tryptophan	0.40

been fed to pigs, cattle, sheep, chickens, ducks, and fish and can substitute for soybean meal in animal feed rations (Robinette et al. 1980, Haustein et al. 1994, Bell 1998, Moss 1999, Johnson 1999, Leng 2004). *Wolffia arrhiza* is collected for human food in Thailand and Laos and is sold at local markets in these countries (Bhanthumnavin and McGarry 1971). Its amino acid composition (Tables 17.3 and 17.4) is more like that of animal protein than plant protein having a high lysine and methionine content, two amino acids normally deficient in plant products (Dewanji 1993). Finally, dried duckweed can provide vitamins, minerals, and pigments such as beta-carotene in livestock diets, reducing the need to add these compounds to rations and thus reducing the cost of producing feed.

Research was conducted at Texas Tech University to utilize duckweed species as part of a system for recycling cattle wastes from feedlots (Fedler and Parker 1998). Duckweed growing in a series of ponds receiving wastewater from a cattle feedlot concentrated nitrogen, phosphorus, and other elements, both purifying this wastewater and providing an ingredient for cattle feed. Since the protein content of duckweed was found to be almost as high as that of soybean meal, duckweed production provided both a means of water purification and a source of livestock feed as well (Allen 1997, Johnson 1998, Moss 1999). It was found that a level of up to 11% of the protein requirements for cattle could be supplied by duckweed and provide added growth benefits as compared to soybean meal as the protein source (Johnson 1998).

Mature poultry can utilize dried duckweed as a partial substitute for vegetable protein such as soybean meal in cereal grain-based diets (Islam et al. 1997). Duckweed used at a level of up to 15% in broiler diets can represent an important alternative source of protein for poultry feeds in countries where soybean or fish meal is unavailable (Haustein 1994). When dried *Lemna* spp. Nex fed to crossbred meat ducks as a substitute for soybean meal there was no significant difference in the carcass traits between treatments (Bui et al. 1995). The protein from duckweed has a biological value equivalent to that of soya beans in diets formulated for ducklings (Nguyen et al. 1997). Duckweed has a high organic matter and protein content but has a low digestibility for ducks. When duckweed was used to replace half the ration in diets for ducks resulted in a reduced feeding costs by up to half (Khanum et al. 2005).

Diets formulated for pigs can substitute duckweed for soybean meal (Leng et al. 1995). Duckweed has

Table 17.4 Essential amino acid composition of dried *L. gibba* meal (g amino acid/100 g dry *L. gibba*)

Essential amino acid	g amino acid/100 g dry *L. gibba*
Leucine	2.89
Arginine	2.14
Valine	2.12
Lysine	1.85
Phenylalanine	1.75
Threonine	1.68
Isoleucine	1.66
Tyrosine	1.27
Histidine	0.73
Methionine	0.64
Cysteine	0.44
Tryptophan	0.40

been ensiled with other feed crops such as corn or cassava leaves to produce an alternative diet for pigs raised on small farms in Vietnam and that fresh duckweed (providing 5% of the diet dry matter) has a stimulating effect on weight gain (Du 1998). The addition of duckweed (*Spirodela* sp.) to corn significantly increased both the pre-ensiled and the postensiled protein content of the silage (Eversull 1982).

What has not been found are articles published on the effect of incorporating duckweed meal into penaeid shrimp diets. Fresh and decomposed duckweed (*Spirodela* sp.) has been used as feed for the Australian red claw crayfish (*Cherax quadricarinatus*) (Fletcher and Warburton 1997). They found that decomposed *Spirodela* species supported crayfish growth as well as commercial pellets did. The abundance of carotenoids and pigments can stimulate crustacean growth (Hertampf and Piedad-Pascual 2000).

Perhaps the most promising use of duckweed is as a feed for pond fish such as carp and tilapia (Landesman et al. 2002). Ponds for duckweed production can be located next to fish culture ponds, eliminating the need for expensive drying to produce a dried feed. Nile tilapia and a polyculture of Chinese carps fed readily on fresh duckweed added to their ponds, and the nutritional requirements of tilapia appear to be met by duckweed (Saber 2004). *W. arrhiza* L. alone supported the growth of two species of Indian carp and four species of Chinese carp as well as one species of barb *Puntius javanicus* (Bikr.) (Naskar 1986). The herbivorous grass carp (*Ctenopharyngodon idella*) digests duckweed species such as *Lemna* and *Wolffia* quite well and it could, by itself, support production of this fish (Cassani et al. 1982, Van Dyke and Sutton 1977). Duckweed has also been tested as a component in the diet of catfish (Robinette et al. 1980), silver barb (Azim et al. 2003), and tilapia (Hassan and Edwards 1992; Fasakin et al. 1999) where it was also able to be substituted for soybean meal. A system for combining duckweed and fish culture was developed in Bangladesh for use by small farmers in developing countries by the non-governmental organization PRISM (Skillicorn et al. 1993). This system could sustain a dry weight production of duckweed in excess of 20 – 35 metric tons a year, (Leng 1999). Hence, duckweed can become a competitive source of plant protein especially in tropical countries.

17.3.2 As an Inexpensive and Accurate Way of Toxicity Testing

Due to its small size and ease of growth, duckweed species make useful organisms for toxicity testing (Lakatos et al. 1993). Duckweed species offer many advantages for the testing of toxic compounds. Duckweed fronds assimilate chemicals directly from their aquatic media into their leaf tissue, allowing for toxicant application in a controlled manner. The growth assay for toxicant assessment is rapid and can be performed without special equipment by counting leaves. Since *Lemna* and *Spirodela* are inexpensive to maintain and the fronds are small, multiple treatments are easy to do simultaneously (Greenberg et al. 1992). Duckweed species have been used to test the toxicity of oils (King and Coley 1985), herbicides (Nitschke et al. 1999), phenol (Barber et al. 1995), and polycyclic aromatic hydrocarbons (Huang et al. 1992), among other toxicants.

A new company in Germany has devised a *Lemna* toxicity test that has been approved by the European Commission (*Lemna*Tec 1999), and the use of duckweed for toxicity testing is mentioned in Standard Methods (1995). Duckweed can be used in both static and the dynamic test procedures (Davis 1981, Wang 1990, Taraldsen and Norberg-King 1990).

17.3.3 Miscellaneous Uses

The ease and convenience of culturing duckweed species under both natural and artificial lights make this species an ideal teaching tool, both at the university and at the primary school level. An example of an experiment using duckweed that can be performed by elementary school students was published in the *Journal of Biological Education* by a Japanese teacher and two research workers (Kawakami et al. 1997). Since duckweed is so quick and easy to grow, students can learn how to study concepts of exponential growth, heavy metal toxicity, photosynthesis, and asexual reproduction. The effect of environmental variables like light and temperature can also be studied using duckweed (Robinson 1988).

An allelopathic effect of duckweed on mosquito larvae may have public health significance. Extracts of *L. minor* caused significant mortality in the larvae of

Aedes aegypti L., a known vector of human diseases such as malaria. The presence of *L. minor* interfered with egg oviposition by *Culex pipiens pipiens* L. and was lethal to *C. p. pipiens* larvae at the first instar stage (Eid et al. 1992). Duckweed may provide a source of mosquito anti-larval compounds that could have commercial significance. Another use for duckweed is as fertilizer. In developing countries like India and Bangladesh where fertilizer is scarce and expensive for the small farmer, duckweed collected from local ponds and wetlands can provide a cheap and effective fertilizer for rice and other crops (Ahmad et al. 1990). It also makes an excellent compost and much of the duckweed harvested from Louisiana wastewater treatment ponds is used for this purpose. Finally a new use for duckweed biomass as a cell-structured support material has emerged as a new technology for yeast fermentation. *W. arrhiza* biomass was extracted with ethanol and loaded with yeast cells. This yeast-impregnated *W. arrhiza* was placed in a semicontinuous fluid-bed fermenter for the production of beer (Richter et al. 1995). New uses for duckweed species will doubtless arise as more researchers learn to appreciate the versatility and potential of Lemnaceae.

From an energy standpoint, most terrestrial plants vary from about 14.8 to 18.4 kJ/g while aquatic plants vary from 10.0 to 21.5 kJ/g. Duckweeds average about 13.5 kJ/g. When you consider the production levels of the various plants, duckweeds can produce from 122×10^6 to 539×10^6 kJ/ha annually, yet the range for many species of aquatic plants considered varies from a low of 12×10^6 to a high of $2{,}900 \times 10^6$ kJ/ha annually (Fedler et al. 2007). Table 17.5 shows the production level of duckweeds at various locations around the world.

17.4 Summary

Duckweeds of the family Lemnaceae are small, floating, aquatic plants with a worldwide distribution. They are one of the fastest growing angiosperms and can double their biomass within 2 days under optimal conditions. They have a high protein content (10 – 40% protein on a dry weight basis) although the moisture content (95%) of fresh duckweed biomass is quite high as well. Potentially, members of the Lemnaceae (of the genera *Lemna*, *Spirodela*, and *Wolffia*) can produce edible protein six to ten times as fast as an equivalent area planted with soybeans. Therefore species of Lemnaceae potentially have a great value in agriculture.

A great deal of work has been done on the nutritional value of species of Lemnaceae, especially *Lemna*, *Spirodela*, and *Wolffia*. Duckweed has been fed to pigs, cattle, sheep, chickens, ducks, and fish and can substitute for soybean meal in animal feed rations. Its amino acid composition is similar to that of other plant proteins except for having a higher lysine and methionine content, two amino acids normally deficient in plant products. Finally, dried duckweed can provide vitamins, minerals, and pigments, such as beta-carotene in livestock diets, reducing the need to add these compounds to rations and thus saving the producer money while having a higher quality feed as compared to the normal basal diet usually fed.

Much research has been done on the use of duckweed in wastewater treatment systems. As part of a facultative treatment system, duckweed can cover treatment ponds and reduce the growth of algae in these ponds as well as reduce nitrogen in the effluent

Table 17.5 Annual worldwide duckweed growth rates (Leng et al. 1995)

Location	Yield (tons/acre)	Yield (metric ton/ha)	Source
Thailand	4.5 – 4.9	10 – 11	Hassan and Edwards (1992), Landolt and Kandeler (1987)
Israel	5 – 8	10 – 17	Porath et al. (1979)
Russia	3.1 – 3.6	7 – 8	Landolt and Kandeler (1987)
Uzbekistan	3 – 7	7 – 15	Landolt and Kandeler (1987)
Germany	7– 10	16 – 22	Landolt and Kandeler (1987)
India	10	22	Landolt and Kandeler (1987)
Egypt	5	10	Landolt and Kandeler (1987)
Louisiana	1 – 10	2 – 23	Culley and Epps (1973), Rusoff et al. (1980),
	12 – 35	27 – 79	Mestayer et al. (1984)
Israel	16 – 23	36 – 51	Oron et al. (1984)

from these ponds through nitrogen uptake and denitrification. Duckweed can also be a part of constructed wetland systems, either as a component of a wetland receiving wastewater or as plants that polish nutrients from wetland-treated effluents.

Due to their small size and ease of growth, duckweed species make ideal organisms for toxicity testing. A new company in Germany has devised a *Lemna* toxicity test that has been approved by the European Commission, and the use of duckweed for toxicity testing is mentioned in Standard Methods. Duckweed can be used in both the static and the dynamic test procedures.

Duckweed plays an important role in the ecology of wetland environments by providing a substrate for the growth of diatoms, protozoa, and bacteria. This phyllosphere (microorganisms living on the outer frond surface) in turn supports insect life as well as enhancing the nutritive value of duckweed for waterfowl and wetland animals such as nutria and turtles. In addition cyanophytes residing in the phyllosphere of duckweed fronds can fix atmospheric nitrogen, providing nitrogen input in oligotrophic aquatic environments.

The three dominant duckweed genera (*Lemna*, *Wolffia*, and *Spirodela*) will all grow on organic (for example, wastewater) as well as an inorganic media (for example, Hoagland's medium). All three species grow faster on organic as opposed to inorganic media with equivalent amounts of nitrogen and phosphorus. This may be due to the ability of duckweed species to take up organic molecules directly from the media in which they grow. Even inorganic media supplemented with glucose will support faster duckweed growth than media without glucose.

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Chapter 18

Nitrogen Removal from Eutrophicated Water by Aquatic Plants

Olga Babourina and Zed Rengel

Abstract Increased N concentration in an aquatic ecosystem is one of important causes of eutrophication. Aquatic higher plants in eutrophic water bodies can remove N by direct uptake and by being a carbon source and a surface substrate for microorganisms and algae, which also use N as a nutrient and can transform N into gaseous forms that escape from the ecosystem. Rooted submerged higher plants can transport oxygen to inundated soils, which increases microbial nitrification, thus providing NO_3^- to the water column. Factors influencing N removal efficiency by aquatic plants include N and hydraulic loading, light, temperature, pH, season, plant harvesting, and availability of other nutrients. These factors differ for different plant species, therefore, plant selection for a particular wastewater treatment should be considered in an integrated approach.

Keywords Ammonium · Duckweed · Eutrophication · Nitrate · Seagrass · Wetlands

Abbreviations

N	nitrogen
P	phosphorus
C	carbon
TN	total nitrogen
TKN	total Kjeldahl nitrogen
TIN	total inorganic nitrogen
TP	total phosphorus
HRT	hydraulic retention time
DNRA	dissimilatory nitrate reduction to ammonium
K_m	the half saturation constant
V_{max}	the maximum rate
FWS	free water surface constructed wetlands
SSF	subsurface flow constructed wetlands

18.1 Introduction

An increase of primary biomass production in an ecosystem caused by an increase of nutrient supply in a growth medium is named 'eutrophication'. It can happen due to natural processes of nutrient wash-out from floodplains or from mineral-rich rocks. However, human activities are considered as a major contributor to eutrophication of ecosystems. Main problems linked to eutrophication of water bodies are increased algae and cyanobacteria production, which leads to high concentration of toxins and anaerobic conditions in water, decreased biodiversity (extinction and/or changes in ecosystem community composition, including vascular plants), changes in sediment biogeochemistry, and nutrient cycling and nutrient ratios (Khan and Ansari 2005). Despite numerous studies, it is still unclear which factors are the primary ones in eutrophication. Currently, there is a controversy about N or P being the main regulators of eutrophication of aquatic ecosystems. Also, limited information is available on how nutrient enrichment interacts with other stressors (toxic contaminants, fishing harvest, aquaculture, non-indigenous species, habitat loss, climate change, etc.) in changing aquatic ecosystems (Cloern 2001). In addition to increasing scientific knowledge, answers to these questions can influence strategies for

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ecosystem restoration or rehabilitation, and save funds which would be spend on reducing a factor, which may turn out not to be important (Schindler and Hecky 2009).

Initially, P and C attracted major attention of limnologists and ecologists as primary agents causing eutrophication. N influence on eutrophication was considered only for estuaries in which N is a limiting factor in normal conditions. However, recent research demonstrated N as the primary cause of eutrophication in many coastal ecosystems as well (Paerl 2009). Indeed, a widespread seagrass loss results from coastal eutrophication (Duarte 2002), with N entering the coastal zone via rivers, groundwater, and the atmosphere (Jickells 1998). Even in freshwater lakes where N_2-fixing bacteria can contribute to N pools (Schindler et al. 2008, Schindler and Hecky 2009), growth of algae was N limited under low atmospheric N deposition, whereas in lakes with high deposition, it was P limited. In experiments investigating the effects of increased NO_3^- supply alone and in combination with P in shallow lakes, N controlled productivity and community composition of phytoplankton (Nydick et al. 2004). High-deposition lakes had higher NO_3^- and TN concentrations and higher TN:TP ratios. High-deposition lakes had a decreased frequency and magnitude of response to N and to combined N and P enrichment. This adds to accumulating evidence that sustained N deposition may have important effects on a lake ecosystem (Elser et al. 2009).

In contrast to naturally eutrophic aquatic ecosystems, constructed wetlands are marshes or swamps created for treatment of domestic and industrial wastewater. Aquatic plants in constructed wetlands have two functions. First, they remove excess of nutrients and pollutants. Second they provide a carbon source and a surface (roots, stems, and leaves) upon which periphyton (microorganisms) can grow as they break down organic materials. Generally, constructed wetlands can be described as an artificial ecosystem where (i) water is present; (ii) soils are different from terrestrial soils; (iii) plants are adapted to inundation (Scholz and Lee 2005). Constructed wetlands are of two basic types: subsurface-flow, SSF (horizontal flow, HSF, and vertical flow, VSF), and free surface-flow wetlands, FWS. Subsurface-flow wetlands move wastewater through a gravel or sand on which plants are rooted; in surface-flow wetlands effluent moves above the soil in a planted marsh or swamp. Subsurface-flow systems have the advantage of requiring less land area for water treatment, but are not generally as suitable for wildlife habitat as are surface-flow wetlands.

Generally, N removal ability of FWS and SSF wetlands is similar, and preferences for a particular type of wetland depend on availability of surface area, a particular nutrient to be removed, or recreational purpose. For instance, during 1 year of treatment of the eutrophicated lake waters, it was found that average removal rates of NH_4^+, NO_3^-, and TN were 23–46, 34–65, and 20–52% in VSF, HSF, and FWS wetlands, respectively. The VSF and HSF showed a statistically similar high potential for removal of nutrients, except NH_4^+. However, the FWS wetland showed the least effect compared to the VSF and HSF at the high hydraulic loading rate (Li et al. 2008). In contrast to wetland types, there is substantial difference in capacity of various plant species to remove N from the eutrophic water. Hence, the current chapter is aimed at analysing the ability of aquatic plants to decrease N content in aquatic ecosystems.

18.2 Sources of N in Natural Aquatic Ecosystems

Coastal waters and inland freshwater bodies receive elevated inputs of anthropogenic N largely from atmospheric emission and deposition, livestock feeding, agricultural runoff, timber harvesting practices, and domestic/industrial affluent discharge. Also, bedrock containing appreciable concentrations of fixed N can contribute a large amount of NO_3^- to surface waters (Holloway et al. 1998). Interestingly, TN concentrations in US streams and rivers currently exceed natural background levels by 6.4 times (Smith et al. 2003).

N input in natural ecosystems is seasonal and often depends on agricultural activity, leaf fall in autumn, and/or plant growth in spring and summer. For example, seasonal NO_3^- patterns showed a summer peak (14–96 μmol L^{-1}) and an autumn decline (< 1 μmol L^{-1}). Stream delta $^{18}O_3^-$ values indicated microbial nitrification as the primary source of stream NO_3^-, although snowmelt or atmospheric sources contributed up to 47% of stream NO_3^- in March. The summer NO_3^- peaks were likely due to a delayed release of

NO_3^- previously flushed to groundwater, weathering of geologic N, and summer increases in net NO_3^- production (Goodale et al. 2009). The autumn NO_3^- decline coincided with leaf fall. The quality of the water flowing from the watercourses to the salt marsh showed a seasonal pattern, with higher contents of NO_3^- (> 200 mg NO_3^- L^{-1}) in periods of maximum agricultural activities, as well as a higher content of NH_4^+ (> 30 mg NH_4^+ L^{-1}) when the human population increased in the zone as a result of tourism (Alvarez-Rogel et al. 2006). N input from natural sources (like rocks) plays an important part in N supply in natural aquatic systems as well. Studies of groundwater N fluxes in Grape Vine Canyon Stream (USA) demonstrated that various groundwater sources may differ in N concentration by more than an order of magnitude. In the surface water, NO_3^- concentration was twofold greater than the concentration predicted from groundwater input, indicating that in-stream processes generated NO_3^- from the mineralization of dissolved organic N to NH_4^+ and subsequent transformation to NO_3^- via nitrification (Jones 2002).

18.3 N Uptake by Aquatic Plants

18.3.1 NO_3^- Uptake

As for all anions, NO_3^- uptake into plant cells requires energy, even when the external NO_3^- concentration is in the millimolar range. The energy for ion uptake is considered to be provided by the proton gradient. Roots possess at least three NO_3^- transport systems that are kinetically distinct. Constitutive high-affinity transport systems are characterized by low values of both K_m and V_{max} (6–20 μM and 0.3–0.82 μmol g h^{-1}, respectively). High-affinity transporters with higher K_m and V_{max} values (20–100 μM and 3–8 μmol g h^{-1}, respectively) are induced within hours to days of exposure to NO_3^-. Low-affinity transporters do not saturate at high NO_3^- (Crawford and Glass 1998).

Generally, kinetics parameters of NO_3^- uptake systems in aquatic plants do not differ from those found in terrestrial plants (Table 18.1). There are some factors, which affect NO_3^- uptake, such as light, a certain pH range for pmf in freshwater plants and Na^+ gradient in seawater plants. In *Pistia stratiotes*, the maximum rate of NO_3^- uptake (V_{max}) and the saturation constant (K_m) were found higher in the light (3.9 μmol NO_3^- g DW^{-1} h^{-1} and 70 μM) than in the dark (1.3 μmol NO_3^- g DW^{-1} h^{-1} and 11.9 μM) (Nelson et al. 1981). The maximum NO_3^- uptake rate in *Typha latifolia* was highest at pH 5.0 (31.7 μmol h^{-1} g^{-1} root DW) (Brix et al. 2002). For the combined NO_3^- and NO_2^- uptake by *Phyllospadix torreyi* leaves, K_m and V_{max} values were 10.1 μM and 54.5 μmol N g DW^{-1} h^{-1}, respectively (Terrados and Williams 1997). In a seagrass *Zostera marina* it was found that NO_3^- uptake can be fuelled by energy provided by the Na^+ gradient that is available in the seawater environment (Rubio et al. 2005).

18.3.2 NH_4^+ Uptake

NH_4^+ uptake by plants involves at least two types of transport systems: high-affinity transporters (HATs) for low external concentrations and low-affinity transporters for high concentrations (Wang et al. 1993). The source of the energy HATs use for NH_4^+ transport is still unclear. HATs have been proposed to function at less than 500 μM or 1 mM NH_4^+ in a media (Britto et al. 2001, Wang et al. 1993). Considering the NH_4^+ concentration outside the plant cell and the plasma membrane potential (E_m), and if NH_4^+ in the cytoplasm goes as low as 10 μM [as suggested by Howitt and Udvardi (2000)], active NH_4^+ transporters would be unnecessary above 1 μM NH_4^+ in media, because the electrochemical potential gradient would be directed into the cell, allowing for passive transport (Britto et al. 2001). However, Britto et al. (2001) suggested the NH_4^+ concentration in the cytoplasm is in the millimolar range. This millimolar range would require active NH_4^+ transport for the outside NH_4^+ concentration below 500 μM with $E_m = -100$ mV.

From Table 18.1, it can be seen that high-affinity NH_4^+ transporters in aquatic plants fall into two groups: with K_m around 10 μM and around 30 μM as was found in a detailed single study on *Z. marina* (Rubio et al. 2007). Interestingly, kinetics parameters of NH_4^+ uptake systems are more dependent on pH than NO_3^- uptake systems (Brix et al. 2002). Comparison of NH_4^+ uptake rates and K_m in roots and shoots allowed authors to make conclusions on plant's preferences for taking NH_4^+ up by roots or shoots. For instance, much lower NH_4^+ uptake rates by roots than

Table 18.1 Kinetics parameters for NO_3^- and NH_4^+ uptake by aquatic plants

Species	V_{max}	K_m µM	Comments	References
NO_3^-				
Pistia stratiotes				
Light	3.9 µmol g^{-1} DW h^{-1}	70		Nelson et al. (1981)
Dark	1.3 µmol g^{-1} DW h^{-1}	11.9		
Typha latifolia	4–31.7 µmol g^{-1} root DW h^{-1}	2–14	Highest V_{max} at pH 5.0 Highest K_m at pH 7.0	Brix et al. (2002)
Phragmites australis	34.5 µmol g^{-1} root DW h^{-1}	4.5		Tylova-Munzarova et al. (2005)
Glyceria maxima	34.5 µmol g^{-1} root DW h^{-1}	6.3		Tylova-Munzarova et al. (2005)
Phyllospadix torreyi	54.5 µmol g^{-1} DW h^{-1}	10.1		Terrados and Williams (1997)
Zostera marina				
Roots		8.9	In sea water	Rubio et al. (2005)
Leaves		2.3	In sea water	Rubio et al. (2005)
NH_4^+				
P. torreyi				
Leaves	125.1 µmol g^{-1} DW h^{-1}	17.4		Terrados and Williams (1997)
Roots	<0.2 µmol g^{-1} DW h^{-1}			
Thalassia hemprichii				
Leaves	35 µmol g^{-1} DW h^{-1}	21–60		Stapel et al. (1996)
Roots	22 µmol g^{-1} DW h^{-1}			
T. latifolia	14–30.9 µmol g^{-1} root DW h^{-1}	1–5	Highest V_{max} at pH 6.5 Highest K_m at pH 5.0	Brix et al. (2002).
P. australis	190 µmol g^{-1} root DW h^{-1}	21.8		Romero et al. (1999)
	56.5 µmol g^{-1} root DW h^{-1}	4.5		Tylova-Munzarova et al. (2005)
G. maxima	56.5 µmol g^{-1} root DW h^{-1}	6.3		Tylova-Munzarova et al. (2005)
Z. marina				
Leaves		2.2 and 23.2		Rubio et al. (2007)
Roots		11.2		

shoots in *P. torreyi* suggest that this surfgrass acquires most of its N via leaves (Terrados and Williams 1997). In contrast, with similar kinetics in roots and leaves of the tropical seagrass *Thalassia hemprichii*, any differences in root versus shoot uptake were rather due to availability of nutrients in the root zone than to physiological differences between organs (Stapel et al. 1996). High NH_4^+ uptake rates found in *Phragmites australis* were consistent with its reputation as a well-adapted species for growth in nutrient-rich habitats (Romero et al. 1999). In *Z. marina*, the higher affinity of leaf cells for NH_4^+, in comparison with root, was consistent with the higher uptake rates observed in leaves (Rubio et al. 2007).

18.3.3 NH_x Toxicity

Ammonia is one of the major constituents of domestic wastewater and concentrations commonly range from 10 to 50 mg L^{-1} N, but might be as high as 200 mg L^{-1} N in industrial or domestic wastewater (Korner et al. 2001). Total ammonia in aqueous solution consists of two principal forms: the ammonium ion (NH_4^+) at pH lower than 8 and un-ionized ammonia (NH_3) at pH higher than 8. Generally, it is impossible to determine the toxicity of pH, NH_4^+, and NH_3 independently from each other as well as the additive effects of NH_4^+ and NH_3. A number of studies suggested the toxicity of total ammonia might be due to the effect of NH_3 only (Wang 1991, Clement and Merlin 1995). In other studies, both forms were suggested as toxic at higher concentrations (Monselise and Kost 1993, Litav and Lehrer 1978). In laboratory experiments, it has been shown that the toxicity of total ammonia on the duckweed species *Lemna gibba* can be attributed only to the effect of un-ionized NH_3 at concentrations higher than 1 mg L^{-1}. The maximum tolerance level for un-ionized ammonia was detected around 8 mg NH_3 L^{-1} (Korner et al. 2001). In an earlier study, no growth of *Lemna minor* was observed at concentrations of un-ionized NH_3 above 10.5 mg L^{-1} (Clement and Merlin 1995). Growth of *Spirodela polyrhiza* was inhibited more than 30% at total ammonia concentrations above 50 mg L^{-1} and pH above 8 (Caicedo et al. 2000). However, growth of *L. gibba* could also be limited by high NH_4^+ concentration in a sewage lagoon [eg., 100 mg N L^{-1} (Al-Nozaily and Alaerts 2002)]. In analysing toxicity limits of NH_x, the method of application of chemicals should be considered. For example, without media renewal, duckweed could grow in concentrations of up to 8.9 mg L^{-1}, whereas with regular media renewal, un-ionized ammonia at concentrations greater than 3.0 mg L^{-1} depressed duckweed growth noticeably (20% or more). Un-ionized ammonia concentration of 7.2 mg L^{-1} was calculated to cause 50% duckweed growth inhibition (Wang 1991).

Ammonia toxicity has been reported in seagrasses *Ruppia drepanensis* and *Z. marina* at 125 μM water column NH_4^+ applied for 5 weeks (Touchette and Burkholder 2000). However, in another study growth of *Z. marina* was not inhibited at pH 8 and NH_x concentration of up to 250 μM, whereas at pH 9 leaf survival rate was drastically reduced to ~56% at 250 μM NH_x (van der Heide et al. 2008). Ammonia concentrations in excess of 200 mg L^{-1} inhibited growth for *Juncus effusus*, *Sagittaria latifolia*, and *T. latifolia* after a period of weeks and concentrations in excess of 100 mg L^{-1} inhibited growth of *Schoenoplectus tabernaemontani*. Ammonia concentrations in the range of 0–400 mg L^{-1} had an ambiguous effect on *Typha angustifolia* (Clarke and Baldwin 2002). In addition, there is substantial ecophysiological evidence for NH_4^+ stress to submerged aquatic plants and indicates that NH_4^+ toxicity arising from eutrophication probably plays a key role in the deterioration of submerged aquatic plants like *Vallisneria natans*, whose growth was severely inhibited by the NH_4^+ supplement in the water column. At high external NH_4^+ (0.81 mg L^{-1}), *V. natans* failed to propagate (Cao et al. 2007, 2009).

NH_4^+ is the preferred N source of duckweed but at certain levels it may become inhibitory to the plant. The inhibition of *S. polyrhiza* duckweed growth by NH_4^+ was found to be due to a combined effect of NH_4^+ and NH_3, the importance of each one depending on the pH (Caicedo et al. 2000). In case of duckweed, it was found that the toxicity of NH_x was a result of the effect of both ionized and un-ionized forms at low NH_3 concentrations (<1 mg N L^{-1}). Relative growth rates of *L. gibba* decreased linearly with increasing NH_3 concentrations up to a maximum level (8 mg N L^{-1}), above which duckweed died. At higher NH_3 concentrations, the toxic effect of the ionized form was considered insufficient (Korner et al. 2001, 2003). NH_4^+ concentrations greater than or equal to 5 mg L^{-1} caused significant acute biochemical changes in *Potamogeton crispus*, which potentially could lead to significant biochemical damage (Cao et al. 2004). It has been also suggested that toxic effect of NH_4^+ on the growth of *V. natans* is caused by induction of oxidative stress and inhibition of photosynthesis (Wang et al. 2008). Studies of the effect of four NH_4^+ concentrations (20.5, 41.1, 61.6, and 82.4 mg NH_3 L^{-1}) on the biomass production of *S. latifolia*, *P. australis*, *Scirpus acutus*, *T. latifolia*, and *Juncus roemerianus* showed that the only species affected by ammonia concentration was *S. acutus* (Hill et al. 1997). NH_4^+ in excess of 0.50 mg L^{-1} inhibited the growth of *Potamogeton maackianus*. The relationships between external NH_4^+ availability and total TN, protein, free amino acid, and NH_4^+ in plant tissues conformed to a logarithmic model, suggesting that a feedback inhibition mechanism may exist for NH_4^+ uptake. Detailed analysis revealed that NH_4^+ was efficiently incorporated into biomass and eventually stored as protein at the expense of starch accumulation. Hence, it has been suggested that this species may tolerate high levels of NH_4^+ when dissolved oxygen is sufficient (Li et al. 2007).

18.3.4 Aquatic Plants Preferences in Taking up NO_3^- or NH_4^+

Plants are taking up N in inorganic (nitrate and ammonium) and organic (urea and amino acids) forms, but the response to a particular form of nitrogen varies from species to species (Crawford and Glass 1998). For instance, both species *Elodea nuttallii* and *Elodea canadensis* prefer NH_4^+ over NO_3^- when both ions were present in water in equal concentrations (Ozimek et al. 1993). Biomass of *Elodea densa* was positively correlated with NH_4^+ in the stream water and with TN in the sediments (Feijoo et al. 1996). In an electrophysiological study on four aquatic plants, it has been concluded that *Bacopa monnieri* had preference for NO_3^- uptake, whereas both N forms were required by *Ludwigia repens* (Fang et al. 2007b). *T. latifolia* was able to grow with both N sources at near neutral pH levels, but the plants had higher relative growth rates, higher tissue concentrations of the major nutrients, higher contents of adenine nucleotides, and higher affinity for uptake of inorganic N when grown on NH_4^+ (Brix et al. 2002). *Littorella uniflora*, *Lobelia dortmanna*, *Luronium natans*, and *Echinodorus ranunculoides* showed NO_3^--dominated (63–73%) N utilization, whereas *Juncus bulbosus* and *Agrostis canina* had NH_4^+-dominated (85–90%) N utilization (Schuurkes et al. 1986). In seedlings of the seagrass *Amphibolis antarctica*, the uptake of NO_3^- (0–200 $\mu g\ NO_3^-\ g\ DW^{-1}\ h^{-1}$) was significantly lower than the uptake of NH_4^+ (0–500 $\mu g\ NH_4^+\ g\ DW^{-1}\ h^{-1}$), suggesting that the seedlings have a higher affinity for NH_4^+ in the water column (Paling and McComb 1994). Growth and organic N content in *S. polyrhiza* were higher in NH_4NO_3 than in KNO_3 (Jayashree et al. 1996). At a steady-state N availability of 34 μM, the growth rate of *Glyceria* was 16% higher in NH_4^+ than in NO_3^- cultures (Tylova-Munzarova et al. 2005).

In addition to species peculiarities, plants' preferences for different forms of N might be influenced by environmental factors such as root or air temperature, aeration, solution pH, composition of the culture solution, water stress and high concentration of salts in the root zone, and by the plant growth stage (cfs. Fang et al. 2007b). In addition, NH_4^+ or NO_3^- uptake could have a seasonal pattern. In *Thalassia testudinum*, leaf NH_4^+ uptake showed clear seasonal variation: V_{max} was highest in summer and autumn, but K_m was highest in winter. V_{max} of leaf NO_3^- uptake did not change with season, but K_m decreased with increasing incubation temperature. There were no clear differences in leaf NH_4^+ and NO_3^- uptake rates between study sites, although leaf NH_4^+ uptake affinity was higher than that of NO_3^-. Root NH_4^+ uptake was variable with season and did not saturate at the experimental NH_4^+ concentrations (0–300 μM) at either site (Lee and Dunton 1999). There could be changes in NO_3^- uptake during the 24-h day. In *P. stratiotes*, the maximum rate of NO_3^- uptake was higher in the light than in the dark (Nelson et al. 1981), whereas the submerged aquatic plant *Ceratophyllum* takes up NO_3^- only in the light (Toetz 1971). In ecological studies, species preferences for NO_3^- or NH_4^+ uptake can sometimes explain species distribution in nature and why one species is replaced by the other. For example, in some eutrophic inland waters *E. canadensis* has been displaced by the morphologically similar species *E. nuttallii*. Several studies have been undertaken in an attempt to explain this succession. In studies where N was as assessed as TN, the authors could not suggest differences in nutrient enrichment as a reason for the species displacements observed in the field (James et al. 2006). However, in experiments where the NO_3^- concentration was maintained constant at 2 mg L^{-1}, high concentrations of NH_4^+ stimulated photosynthesis in *E. nuttallii* and inhibited it in *E. canadensis*. Also, NH_4^+ caused a distinct inhibition of nitrate reductase (NR) activity (determined by in vivo assay) of *E. canadensis,* whereas the NR activity of *E. nuttallii* was greatly enhanced by concentrations of NH_4^+ up to 0.5 mg L^{-1}. These interspecific differences in response to NH_4^+ intoxication allowed the authors to suggest explanation of the trophic status of these two species in situ (Rolland and Tremolieres 1995). Another example would be acid-tolerant species, with an NH_4^+-dominated N utilization, profiting from increased NH_4^+ concentrations in acidic waters. Therefore, in the case of acidification, increased NH_4^+ concentrations can contribute to the suppression of typical soft-water communities by acid-tolerant species (Schuurkes et al. 1986).

18.3.5 Root Versus Shoot N Uptake

In earlier experiments, Gorham (1941) demonstrated that covering the undersides of fronds with lanolin decreased the growth of duckweed plants, although he noticed that the root length was increased. This observation allowed Hillman (1961) to suggest that roots of floating macrophytes function mostly as anchors, whereas fronds and leaves are the main organs involved in nutrient uptake. In addition, covering fronds and roots of *S. polyrhiza* and *L. minor* with paraffin in one set of experiments, and removing roots in another set, researchers concluded that the roots of duckweeds had a relatively small role in nutrient uptake (Muhonen et al. 1983, Ice and Couch 1987). In experiments on the relative contribution of roots and leaves to nutrient uptake by submerged aquatic plants (*E. canadensis*, *Callitriche cophocarpa*, *Ranunculus aquatilis* and *P. crispus*), Madsen and Cedergreen (2002) concluded that removal of roots had no negative impact on the relative plant growth rate. Further, the organic N concentration of the plant tissue was constant with time for the de-rooted plants. However, the conclusion that roots are not important for N uptake in Lemnaceae was opposed by other studies from the same laboratory (Oscarson et al. 1988, Cedergreen and Madsen 2002, 2003). At low external concentration (5 μM NH_4NO_3), roots of *L. minor* had a higher rate of uptake of both NH_4^+ and NO_3^- than fronds, whereas higher NH_4NO_3 supply (250 μM) reduced root uptake rates for both ions. This decreased uptake rate in roots at high NH_4NO_3 supply was compensated for by higher uptake rates in fronds (Cedergreen and Madsen 2002). In an electrophysiological study, roots of *Landoltia punctata* contributed to N uptake at the same level as fronds (Fang et al. 2007a). Even though the magnitude of ion fluxes in roots was lower than in fronds, the root surface was twofold larger than the frond surface, and ratio of fronds/roots in N uptake was close to 1:1.09 for NH_4^+ and 1:0.79 for NO_3^-. Therefore, plants have equal capacity to use fronds and roots for NO_3^- and NH_4^+ uptake (Fang et al. 2007a). In agreement with this, ^{15}N studies demonstrated that *Myriophyllum spicatum* was capable of taking up inorganic N through both roots and shoots. Plant N requirements can apparently be met by root uptake alone. However, when about 0.1 mg L^{-1} of NH_4^+ was present in water, foliar N uptake was higher than the root uptake. NH_4^+ uptake by leaves was found to be several times faster than that of NO_3 when both forms of N were present in water (Nichols and Keeney 1976). Plant preferences for one or the other N form might depend on a season. For example, in studies on dissolved inorganic N uptake by leaves and roots of surfgrass *Phyllospadix iwatensis*, it was found that both leaves and roots contributed to N acquisition. However, leaves of *P. iwatensis* contributed to total N uptake mainly in autumn and winter when N concentration in the water column was high. Contributions of roots were higher than those of leaves in spring and summer when dissolved inorganic N (DIN) in the water column was low (Hasegawa et al. 2005).

The leaves of tropical seagrass *T. hemprichii* showed a clear capacity for NH_4^+ uptake. Evidence was found that the availability of nutrients in the root zone relative to the leaf zone affects the uptake affinity of leaves. It was concluded that even in the tropics, where water column nutrient concentrations are often very low, leaves clearly have a significant ability for NH_4^+ uptake and that in some situations nutrient uptake by the leaves may even be essential in meeting plant nutrient demands (Stapel et al. 1996). In transplant experiments conducted in a river, aquatic plant *P. crispus* obtained most of its nutrients through the roots (Chambers et al. 1989). When plants from different ecological background were compared, it was found that the soft-water species had NO_3^--dominated (63–73%) N utilization, with roots as the major (83%) uptake organ. At the same time, the acid-tolerant species have an NH_4^+-dominated (85–90%) N utilization, having leaves as the major uptake site (71–82%) because of increased NH_4^+ concentrations in acid waters (Schuurkes et al. 1986). Aquatic plants grow in an environment where NO_3^- may have higher concentration in the bulk water compared to the sediment. Given that there is energy advantage of reducing NO_3^- in shoots compared to roots, aquatic plants were expected primarily to reduce NO_3^- in shoots. In 11 out of 17 plants, the shoot:root NR activity ratio was <1, showing that despite the proposed advantages of shoot

NO_3^- reduction, root reduction is apparently still of importance in aquatic plants (Cedergreen and Madsen 2003). In addition to a question about which organ is more important in nutrient uptake, there is another question which is not clear: whether there is an interaction between roots and leaves in regulation of N uptake. Leafy shoots of seagrass *Z. marina* with roots attached were incubated in two-compartment chambers, with the roots in one compartment and the leaves in the other. Rates of ammonia uptake were compared when roots and/or leaves were supplied with ammonia. Root uptake of ammonia had no influence on the rate of ammonia uptake by leaves. However, leaf uptake of ammonia caused a 77% decrease in the maximum rate of ammonia uptake by roots. The K_m values for ammonia uptake by leaves and roots were 9.2 and 104 μM, respectively, showing that leaves had a greater affinity for ammonia than roots (Thursby and Harlin 1982).

It has been found that increased ratio of the root surface to the frond surface led to increased NH_4^+ uptake rate in *L. minor* (Cedergreen and Madsen 2002). These studies indicate that duckweed plants can regulate their lifecycle, such as increased multiplication rate, and the surface area for the nutrient absorption, at the level of frond–root interactions.

18.4 Aquatic Plants and N Removal Efficiency in Eutrophic Aquatic Ecosystems

18.4.1 Contribution of Aquatic Plants to N Removal

N removal capacity of plants and microorganisms in natural and man-made eutrophicated aquatic ecosystems depends on many factors, including retention time, season, temperature, pH, diversity of species, N loading, hydraulic regimes, plant harvesting, light intensity. With respect to plant species, N removal efficiency by duckweeds varied from up to 20% (Ran et al. 2004) to 98% (El-Shafai et al. 2007) (Table 18.2). Some of the important factors (e.g. N and hydraulic loading) can be regulated in constructed wetlands. For example, TN loading usually does not exceed 50 mg N L^{-1}, although sometimes in primary sewage N concentration can reach 200 mg N L^{-1} (Table 18.2).

A model of an enclosed submerged aquatic plant ecosystem revealed the high importance of the initial conditions or by biomass changes during the experiment. For instance, high initial aquatic plant biomass reduced light and nutrient availability for algae, which prevented the algal overgrowth, even at high nutrient concentrations. On the other hand, epiphytic algae biomass increased with higher water exchange rate regardless of nutrient concentration. Submerged plants grew best at lower water exchange rates that allowed more efficient nutrient uptake and consequently slowing algal growth with depleted nutrients. Model analysis also revealed that it is essential to consider nutrient loading rates per unit of aquatic plant biomass, not just nutrient concentration. The model demonstrated that water exchange rates, which can depend on bed size, could affect eutrophication responses of submerged aquatic plant ecosystems based on *Potamogeton perfoliatus* (Bartleson et al. 2005). At low N loading, TN removal efficiency could be completely attributed to plant uptake, whereas at high N loading, N uptake by plants was inhibited (Al-Nozaily et al. 2000). *P. australis* planted to the wetland tubs removed > 96% TN over the 19-month-study period. Plant nutrient uptake accounted for most (76%) of the N removed. In carbon-limited low nutrient nursery runoff, plants in a gravel-based wetland were essential in achieving efficient nutrient removal (Huett et al. 2005). In a laboratory-scale duckweed-covered system, *L. gibba* plants were directly responsible for 3/4 of the total N removal in 3 days (Korner and Vermaat 1998). As a rule, N removal efficiency in the wetlands increased exponentially with increased residence time (Huang et al. 2000).

In addition to plant species' peculiarities in N uptake, including their preferential form of N, there is also a phenological factor to be considered. Species achieve their intensive growth at different times in a season. Also, N removal rate in a wetland is lower during the establishment stage compared with later stages. Therefore, a choice of a particular plant species for a particular eutrophic ecosystem should be dependent on phenology as well as maturity of the whole system.

18.4.1.1 Temperature Effect

Optimum growth was recorded at between 20 and 30°C for *L. minor* and between 25 and 30°C for *S. polyrhiza*. This temperature interval was found

Table 18.2 N removal efficiencies of aquatic plants

	Nitrogen loading			Nitrogen removal ability					
Species	TN	NO_3^-	NH_x	TN	NO_3^-	NH_x	Type of treatment	Comments	References
Floating plants									
Lemna minor				83–87%			Municipal and industrial WW	Laboratory	Ozengin and Elmaci (2007)
L. minor 8627				3.36 g m^{-2} day^{-1}				Laboratory	Cheng et al. (2002a)
				2.11 g m^{-2} day^{-1}				Field	
Lemna gibba				10–20%			Domestic primary effluent	Wetland	Ran et al. (2004)
				30–47%				Laboratory	Korner and Vermaat (1998)
Spirodella punctata 7776			240 mg NH_4^+ L^{-1}	0.955 mg L^{-1} h^{-1}					Cheng et al. (2002b)
Duckweed				38.8%				Pond, high-temperature periods	Peng et al. (2007)
			100 mg L^{-1}				Abattoir effluent	Viable	Goopy et al. (2004)
			56 mg L^{-1}				$\frac{1}{4}$ dilution, pH 7, 0.5% bentonite	Grew well	
Eichhornia crassipes	7–56 mg L^{-1}		0.2 mg L^{-1}			97.8%	Lower bounds		Zimmels et al. (2007)
	0.5–50.5 mg N L^{-1}					100%	Outdoor tanks	The optimum time for harvesting is 6–9 weeks	Jayaweera and Kasturiarachchi (2004)
								21 days hydraulic retention time	
								Maximum N storage measured in plants cultured at 50.5 mg N L^{-1}	Reddy et al. (1989)
P. stratiotes			0.2 mg L^{-1}				Lower bounds		Zimmels et al. (2007)

Table 18.2 (continued)

Species	Nitrogen loading			Nitrogen removal ability			Type of treatment	Comments	References
	TN	NO_3^-	NH_x	TN	NO_3^-	NH_x			
Salvinia rotundifolia									Zimmels et al. (2007)
Ludwigia palustris									Zimmels et al. (2007)
Emergent plants									
P. australis	10.1 mg L^{-1}		7.4 mg L^{-1}	96%			SSF	19 months	Huett et al. (2005)
Phragmites communis	121 g m^{-2} $year^{-1}$			92.42%					Yang et al. (2007)
Scirpus cyperinus and *T. latifolia*	205 mg L^{-1}			31–45% or 44–73% depending on site	18–39% or 46–67% depending on site		VSF and HSF	Both NH_4^+ and TKN concentrations in the wetlands decreased exponentially with increased residence time	Huang et al. (2000)
Phragmites spp. and *T. latifolia*				38%				Hydraulic retention time was 2 d and 3.6 d	Kouki et al. (2009)
T. latifolia	121 g m^{-2} $year^{-1}$			79%					Yang et al. (2007)
Canna indica	121 g m^{-2} $year^{-1}$			92%					Yang et al. (2007, 2007)
Pennisetum purpureum	120.82 g m^{-2} $year^{-1}$			85%					Yang et al. (2007)
Vetiveria zizanioides	120.82 g m^{-2} $year^{-1}$			78%					Yang et al. (2007)
Submerged plants									
Myriophyllum verticillatum									
Elodea canadensis		2 mg NH_4^+ L^{-1}	2 mg NO_3^- L^{-1}			75–90%		Higher N concentration than 4 mg L^{-1} had a negative effect on growth of both species	Ozimek et al. (1993)

the most suitable for remediation of eutrophic water (Ansari and Khan 2008, 2009). Experiments on warming mesocosms of shallow communities by 4°C demonstrated changes in concentrations of soluble phosphate, total N and conductivity, increased total plant biomass, and decreased algae production due to shading by floating plants. Therefore, it is highly likely that the global temperature increase will intensify eutrophication in shallow lakes. Although it will not cause a switch from plant-dominated to algal-dominated systems, it may result in more active growth of floating aquatic plants. It has been also shown that N enrichment combined with elevated temperatures increased floating plant biomass and decreased plant species richness (Feuchtmayr et al. 2009).

18.4.1.2 Light Effect

It was found that in eutrophicated coastal waters seagrass decline is largely due to light reduction because of high biomass of algal overgrowth as epiphytes and macroalgae in shallow coastal areas and phytoplankton in deeper coastal waters (Burkholder et al. 2007). Recently, it has been shown that light is the significant factor controlling plant growth in a eutrophic lake, largely due to the input of highly coloured dissolved organic matter that enters lakes from surrounding forests (Karlsson et al. 2009).

18.4.1.3 Seasonality

In addition to more favourable temperatures during a particular season, there is some phenological specificity in developing plant biomass, which is contributing to higher plant growth during the season and therefore contributing to higher N removal efficiency. Nutrient removal exhibited a typical seasonal pattern in microcosms with planted *Carex lacustris*, *Scirpus validus*, *Phalaris arundinacea,* and *T. latifolia*, with higher removal rates in the growing season and lower rates in the winter months (Picard et al. 2005). During the autumn season, *Phragmites* spp. and *T. latifolia* reached their optimum growth rate, followed by a dormant phase, which affected their N removal efficiency (Kouki et al. 2009). Wetlands vegetated by *Pennisetum purpureum* significantly out-performed wetlands with other plants in May and June, whereas wetlands vegetated by *Phragmites communis* and *Canna indica* demonstrated higher removal efficiency from August to December. Hence, it has been suggested that a plant community consisting of multiple plant species with different seasonal growth patterns and root characteristics may be able to enhance wetland performance over long periods of time (Yang et al. 2007).

18.4.1.4 N Loading

There is some evidence that N removal can be increased with increased N loading in the wastewater (Tripathi et al. 1991, Cheng et al. 2002b). For instance, whole plant biomass positively correlated with N supplied and therefore had higher depletion rate of N from a nursery runoff (Polomski et al. 2009). In addition, low N removal efficiency slightly increased with higher N loads (Ran et al. 2004). When the growth of *Eichhornia crassipes* was assessed in the culture medium with N concentrations between 0.5 and 50.5 mg N L^{-1}, it was found that the net productivity increased with N supply rate of up to 5.5 mg N L^{-1}. The net productivity increased until plant tissue N content reached 16 mg N g^{-1} DW, but additional increase in tissue N content did not improve yield. However, N storage in the plant tissue increased in response to increased N supply rate, with maximum N storage (80 g N m^{-2}) measured in plants cultured at 50.5 mg N L^{-1} (Reddy et al. 1989).

18.4.1.5 pH Effect

It has been found that *L. minor* performs better at low pH for phytoremediation of eutrophic waters (Ansari and Khan 2008) and for increased treatment efficiencies in duckweed-covered wastewater, pH values should be decreased (Korner et al. 2001). The toxicity of total ammonia to duckweed was a result of the effect of both ionized and un-ionized forms at low NH_3 concentrations (< 1 mg N L^{-1}). Relative growth rates of *L. gibba* decreased linearly with increasing NH_3 concentrations up to a maximum level (8 mg N L^{-1}), above which duckweed died. Up to pH of 7.8, a substantial production of 55 kg DW ha^{-1} day^{-1} was achieved. Wastewater treatment using *L. gibba* became impossible at pH levels above approximately 9.8 (Korner et al.

2001, 2003). At higher NH_3 concentrations, the toxic effect of the ionized forms could be disregarded.

18.4.1.6 Hydraulic and Organic Loading and Retention Time

Hydraulic and mass loading parameters were found important in experiments with *L. minor* during the secondary treatment of domestic wastewater. Optimum N removal occurred at 1.2 cm day^{-1} hydraulic loading rate and 90–160 mg TKN m^{-2} day^{-1} mass loading rate. Effluent TKN concentration was around 2.5–3.0 mg L^{-1}, while NH_3 concentration was almost zero at these loading conditions. On the other hand, effluent NO_3^- concentrations varied between 7 and 11 mg L^{-1}. When investigating the longitudinal profile, values were reduced rapidly along the reactors (Nalbur et al. 2003). In addition, N removal efficiency depends on organic loading rates (OLR) and stocking densities (SD). For instance, the maximum NH_3 and TN removal efficiencies of, respectively, 68 and 58% were found at optimum operating conditions comprising retention time of 10 days, organic loading rates of 50 kg COD ha^{-1} day^{-1}, and stocking densities of 0.5 kg mL^{-1} (Krishna and Polprasert 2008). At the hydraulic loading rate 160,000 m^3 day^{-1} *Lemna* sp. showed the average removal efficiency of 56% for NO_3^-, 48% for TN, and 46% for organic N (Shammout et al. 2008).

18.4.1.7 Best/Worst Performers Among Plant Species

Among a range of plant species, highest N recovery rates were exhibited by water hyacinth (*E. crassipes*) and water lettuce (*P. stratiotes*) (Polomski et al. 2009). *E. crassipes* has been proposed as a promising candidate for a batch removal of TN from wastewaters: 100% removal of TN was observed at the end of the ninth week mainly due to assimilation (Jayaweera and Kasturiarachchi 2004). The nutrient removal capacity in tropical freshwater ponds was highest by *Eichhornia*, followed by *Pistia*, then *Lemna*, and lowest by *Salvinia* (Tripathi et al. 1991). Allocation of above- and below-ground growth and nutrient uptake and pollutant removal were compared for *Schoenoplectus validus*, *P. australis*, *Glyceria maxima*, *Baumea articulata*, *Bolboschoenus fluviatilis*, *Cyperus involucratus*, *J. effusus*, and *Zizania latifolia* grown in mesocosms fed with dairy farm wastewaters pre-treated in an anaerobic lagoon. After 124 days, total biomass for all various species ranged from 0.3 to 7.4 kg m^{-2}, with above/below-ground ratios between 0.35 and 3.35. Growth of *Baumea* and *Juncus* was relatively poor. *Zizania* and *Glyceria* showed the highest above-ground biomass values, ranging between 3 and 4 kg m^{-2}. Above-ground tissue concentrations of N ranged from 15 to 32. Maximum plant accumulations of 135 g N m^{-2} accounted for around 30% of the amounts supplied in wastewaters. Mean removal of total N ranged from 65 to 92%, showing a significant positive linear correlation with plant biomass (Tanner 1996). In FWS and one SSF constructed wetlands, reduction in total N concentrations ranged from 18 to 86%, ammonia N from 8 to 95%, and oxidized N from 55 to 98%, producing effluent with total N between 1.6 and 18 mg L^{-1}. Submerged (*Ceratophyllum*) and free floating species (duckweed) had the highest tissue nutrient concentrations, followed by the waterlily (*Nymphoides indica*), aquatic vines (*Ipomoea* spp., *Ludwigia peploides*), and waterferns (*Ceratopteris*, *Marsilea*). Emergent species had lower nutrient concentrations, whereas the highest nutrient concentrations occurred in the exotic sedge *C. involucratus*. Aquatic grasses including *Phragmites* had higher nutrient concentration than the sedges. N concentrations were higher in leaf/stem tissue compared to the root/rhizome. Emergent species had greater biomass than submerged or free floating species and were therefore able to store more nutrients per unit area of wetland. Harvesting the shoots of emergent species increased nutrient content in new shoot growth (Greenway and Woolley 1999). Highest N recovery rates were exhibited by *E. crassipes* and *P. stratiotes* in comparison with *Myriophyllum aquaticum* in a laboratory-scaled SSF system (Polomski et al. 2009). Planted in microcosms, *C. lacustris* was the least efficient plant in comparison with *S. validus*, *P. arundinacea*, and *T. latifolia* (Picard et al. 2005).

Typha significantly out-performed *Juncus* and *Scirpus* both in growth and in effluent quality improvement in small-scale constructed wetlands receiving primary-treated wastewater. There was also some evidence that the species mixture out-performed species monocultures (Coleman et al. 2001). *S. validus* was most effective and *P. arundinacea* was generally least effective at reducing N in monocultures. The four-species mixture of *S. validus*, *C. lacustris*, *P. arundinacea*, and *T. latifolia* was generally highly effective at nutrient removal (Fraser et al. 2004).

In constructed wetlands, *S. validus* and *P. communis* were superior at removing NH_x than *T. latifolia* (Gersberg et al. 1986). In a vegetated, laboratory-scale SSF system, highest N recovery rates were exhibited by *Thalia geniculata* and *Oenenathe javanica* in comparison with *Phyla lanceolata* (Polomski et al. 2008). Nutrient removal capacity of some important aquatic plants, i.e. *E. crassipes*, *L. minor*, and *Azolla pinnata* has been tested individually as well as in combinations in microcosms. Maximum removal was observed in species mixtures, involving *E. crassipes* and *L. minor* (79% N) (Tripathi and Upadhyay 2003).

18.4.1.8 Effect of Other Nutrient on Capacity of Aquatic Plants to Remove N

The concentrations of nutrients in the wastewater and nutrient loading rate may influence the nutrient removal from eutrophicated water or wastewater. Significant interactive effects of N and P loadings were found for the removal efficiencies of NH_4^+ but not that of NO_3^- in secondary-treated wastewater using *S. validus* in the vertical free surface-flow wetland microcosms (Zhang et al. 2008b). Optimal ratios of N and P in water are needed for promoting plant growth, increasing tissue nutrient concentrations, and therefore enhancing nutrient removal from wetlands (e.g. *C. indica*, Zhang et al. 2008a). In a field experiment in the Netherlands, growth of *Elodea* was significantly enhanced by enrichment with N alone and even more so by combining N with P or P and K (Best et al. 1996). The net productivity and nutrient storage of N by *E. crassipes* grown in outdoor tanks increased with increasing K supply up to a point, indicating a saturable type response (Reddy et al. 1991).

18.4.2 *Aquatic Plants Improvement of the Eutrophic Aquatic Ecosystems*

In eutrophic aquatic ecosystems, N undergoes different chemical processes mediated by microorganisms. For example, in nitrification, as a central process in the N cycle, NH_4^+ gets oxidized by bacteria and archaea to NO_2^- and then to NO_3^-. Another process contributing to N removal from wastewater is denitrification, a bacteria-facilitated process of NO_3^- reduction, whereby NO_3^- acts as electron acceptor; denitrification occurs where O_2, a more energetically favourable electron acceptor than NO_3^-, is depleted. Direct reduction from NO_3^- to NH_4^+, a process known as dissimilatory nitrate reduction to ammonium (DNRA), is facilitated by many Enterobacteriaceae, bacilli, and clostridia under anaerobic conditions (Knowles 1982). As mentioned earlier, in addition to taking up nutrients, aquatic plants have an important role in nitrogen utilization in eutrophic waters by acting as a substrate, providing carbon source for microorganisms, supplying oxygen to soil, etc., thus facilitating a recovery of natural eutrophic ecosystems and making constructed wetlands more effective in water purification. In addition, aquatic plants stabilize the surface of the beds, provide good conditions for physical filtration, prevent VFS from clogging, insulate against frost during winter, etc. (Brix 1994). Numerous studies demonstrated that the vegetated wetlands are more efficient in N removal than unvegetated ones, and this difference is not only due to plant contribution to N uptake (Chung et al. 2008, Fraser et al. 2004, Gersberg et al. 1986). ^{15}N studies demonstrated that elevated denitrification in the presence of *Glyceria declinata* was attributed to a higher degree of soil oxidation, which is considered to be the principal regulator of NO_3^- partitioning between denitrification and DNRA. Shoot harvest did not affect the fate of $^{15}NO_3^-$, but inhibited new root production, and increased the NO_3^- assimilation capacity of shoots by a factor of 5 (Matheson et al. 2002). Interestingly, plant-related oxygenation of soils is not only confined to submerged plants, but floating duckweed also enhanced nitrification by providing oxygen and surface for bacterial growth (Korner et al. 2003).

It has been shown that internal N cycling processes in sediments and the overlying water column may contribute to the eutrophication of aquatic ecosystems. In ^{15}N studies, a small constructed wetland removed 45 g N m^{-2} $year^{-1}$, with removal efficiency of 27%. Denitrification contributed 94% to the N removal, while only 6% of the removed N accumulated in the sediments. Denitrification was most efficient during periods in which a water column contained dissolved oxygen while sediments were anoxic, as

NH_4^+ released during mineralization of organic matter in the sediment was completely nitrified and subsequently denitrified at the sediment-water interface. During water column anoxia, NH_4^+ accumulated in the water and was taken up by duckweed (Reinhardt et al. 2006). In nitrification studies in a lake, in aerated conditions, NH_4^+ oxidation to NO_3^- in sediment showed two-phase kinetics. The rapid first phase of nitrification was due to the oxidation of NH_4^+ initially present in the sediment, while the slower second phase was limited by the rate of production of NH_4^+ during ammonification of organic N. Denitrification was found to be limited by NO_3^- availability (Dangelo and Reddy 1993). In both laboratory- and pilot-scale constructed wetlands, sequential nitrification/denitrification reactions and plant uptake were the major mechanisms of N removal, while NH_3 volatilization was considered to be insignificant when pH was lower than 8 (Koottatep and Polprasert 1997). Nitrification rates were higher in SSF than in FWS systems, whereas denitrification and ammonification rates were similar (Tuncsiper et al. 2005). The indirect contribution of duckweed to the total nutrient removal in laboratory-scale experiment was high and included the uptake of NH_4^+ by algae and bacteria in periphyton and the removal of N through nitrification/denitrification by bacteria attached to the duckweed. Together, these accounted for 35–46% of the total N removal (Korner and Vermaat 1998). In a different laboratory-scale experiment, where five duckweed species (*L. gibba*, *L. minor*, *Lemna trisulca*, *S. polyrhiza*, and *Wolffa arrhiza*) were grown on domestic sewage and simulated waste water, denitrification was also the major pathway for N removal (Vermaat and Hanif 1998).

Contribution of nitrification/denitrification depends on a season/temperature in outdoor experiments. In low-temperature periods, nitrification/denitrification accounted for 10% of TN removal, whereas in high-temperature periods it was 39%. The most important function of duckweed for TN removal was not direct uptake, but indirect enhancement of nitrification and denitrification rates (Peng et al. 2007). Depending on initial N concentrations in wastewater, duckweed (*L. gibba*) removed 42–62% of total N and 56–95% of TKN, mostly due to denitrification and ammonia volatilization (Zimmo et al. 2000). In addition to NO_3^- and NH_4^+ as major N sources in eutrophication of aquatic ecosystems, there is a growing contribution of urea, whose use has significantly increased in the past decades. Urea concentrations in aquatic ecosystems can be substantially elevated and can represent a large fraction of the total dissolved organic N pool that can be utilized by phytoplankton (Glibert et al. 2006). In a wetland study, urea transformation was 100 times higher in sediment compared with the epiphytic activity on the surfaces of the submerged plant *E. canadensis*. However, the epiphytic activity responsible for urea transformation differs for different plant species, being lower on leaves of *T. latifolia* than on *E. canadensis*, and was negligible on the submerged leaves of the emergent plant *P. australis* (Thoren 2007).

18.5 Conclusions

- Increased N concentration in an aquatic ecosystem is one of important causes of eutrophication.
- Aquatic higher plants in eutrophicated water bodies can remove N by direct uptake and by being a carbon source and a surface substrate for microorganisms and algae, which also use N as a nutrient and can transform N into gaseous forms that escape from the ecosystem.
- Rooted submerged higher plants transport oxygen to inundated soils, which increases microbial nitrification, thus providing NO_3^- to the water column.
- Factors influencing N removal efficiency by aquatic plants include N and hydraulic loading, light, temperature, pH, season, plant harvesting, and availability of other nutrients; these factors differ for different plant species.

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Chapter 19

Accelerated Eutrophication in the Mekong River Watershed: Hydropower Development, Climate Change, and Waterborne Disease

Guy R. Lanza

Abstract Accelerated eutrophication poses a major threat to global water quality and represents a particular concern in Southeast Asia. The riparian countries of the Mekong River watershed are currently under threat from major episodes of accelerated eutrophication catalyzed by the combined effects of hydropower development and emerging climate change factors that can interact and profoundly affect ecosystem structure and function. Eleven dams are planned for the lower Mekong mainstream with scores more on its tributaries. Dams in the Mekong River watershed will interrupt the natural flow regime and alter the critical stoichiometric relationship of nutrients and other physical, chemical, and biological factors regulating biodiversity in the region. Imbalances in microbial communities can result and contribute to a trophic cascade that can lead to seriously degraded water quality, frequent occurrence of cyanobacterial blooms with lethal cyanotoxins, Trihalomethane (THM) precursors, taste and odor compounds, and changes in aquatic habitat that favor increased waterborne disease transmission including schistosomiasis, fasciolopsis, malaria, and dengue fever.

Keywords Accelerated eutrophication · Schistosoma Mekongi · Neotricula aperta · Cyanotoxins · Hydropower projects · Climate change

19.1 Introduction: A Brief History of Dam Building in Southeast Asia

> You cannot step twice into the same river; for fresh waters are ever flowing in upon you
>
> Heraclitus, ca. 500 B.C.

The Mekong River arises from its source at an elevation of 5,000 m near the Dzanag La pass of the Tanghla Shan mountain range in Tibet and meanders approximately 4,500 km through Southeast Asia along the route to its juncture with the South China Sea. During its travel, the river casts its dynamic influences on six riparian nations, China, Burma, Thailand, Laos, Cambodia, and Viet Nam and falls more than 330 m over 2,400 km of its length providing considerable potential for hydroelectric power generation and irrigation. Average flows calculated at its confluence with the South China Sea are approximately 500 million m^3 with remarkable variation in discharge and velocity over a typical monsoon cycle.

In 1965, during the Vietnam conflict Lyndon Johnson, the President of the United States of America, delivered a televised address entitled "Peace without Conquest" from the campus of Johns Hopkins University. Johnson offered a pledge of $1 billion for a vast program of regional development in return for North Vietnam's acceptance of peace. The program would be modeled after the Tennessee Valley Authority's program in the USA and would feature extensive river damming for hydroelectric power, fisheries, and irrigation (Dieu 1999). Although the project was rejected by Vietnam and never initiated as offered, it clearly set the stage for a new phase of water resource development policy for the Mekong River watershed.

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At that point in history the river meandered from its headwaters to its confluence with the South China Sea without a single dam or bridge across it or the majority of its major tributaries. During 1972–1974, the first benchmark ecological studies were completed as part the initial pre-impoundment studies for Pa Mong, the first dam planned for the Mekong River (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data; Smithsonian Institution – Mahidol University 1974). However, changes in the political atmosphere in the region soon led to a moratorium on plans for dam projects in the Mekong River until the early 1990s.

19.1.1 The Nexus of Hydropower Development, Climate Change, Accelerated Eutrophication, and waterborne disease

In recent decades, a rather ambitious scheme of complex alterations to the Mekong River's lower basin has been either completed or proposed including impoundments on major tributaries and impoundments, dikes, bridges, and various channel modifications in its mainstream. The major impacts of large hydropower dams are presented in Fig. 19.1. Major environmental impacts from the dam structure, the reservoir, and areas of river habitat above and below the dam and reservoir are severe and often cause irreversible ecological damage. The environmental and social implications of the impacts of hydropower development are extremely important and highlight major deficiencies in our current understanding of the human ecology and ecological function of tropical rivers (Imhof and Lanza 2010).

Accelerated Eutrophication remains a major threat to global water quality (Lanza 1995) and represents a particular concern in Southeast Asia. The riparian countries of the Mekong River watershed are currently under threat from major episodes of accelerated eutrophication catalyzed by the combined effects of hydropower development and emerging climate change factors that can interact and profoundly affect ecosystem structure and function. Damming rivers to create large hydropower impoundments disrupts the natural flow regime of rivers (Poff et al. 1997) and upsets the ecological integrity that maintains the physical and biological habitat essential to healthy rivers.

Climate change is expected to dramatically alter the global hydrological cycle and change the dynamics of many rivers, worsening both droughts and floods

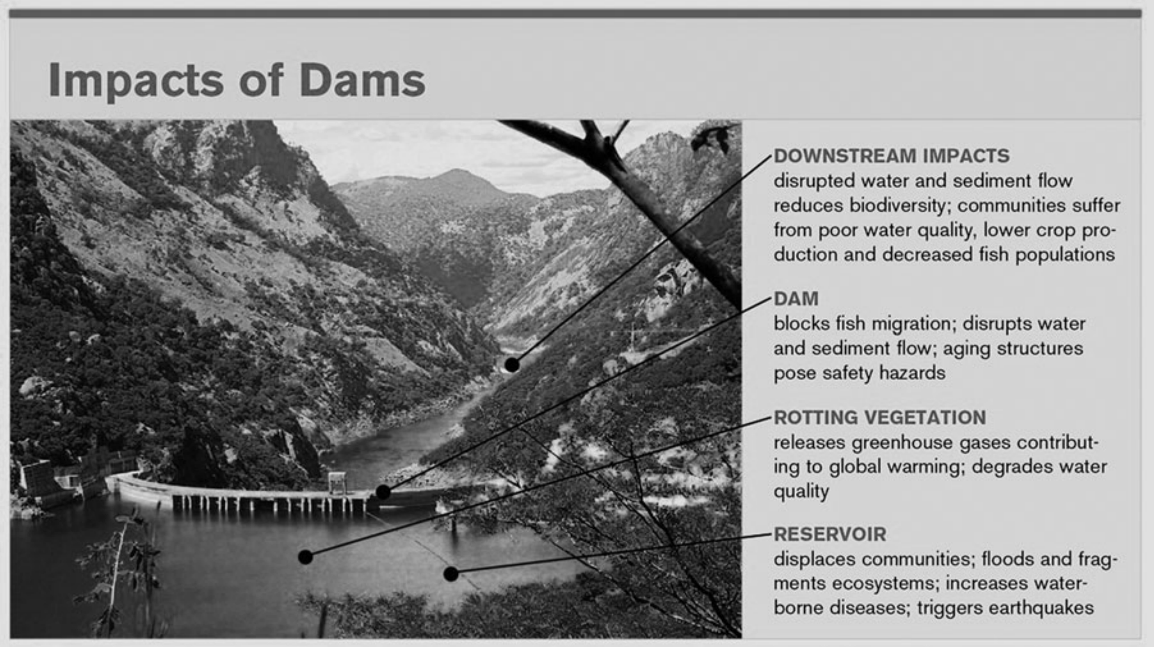

Fig. 19.1 Impacts of dams (International Rivers, Berkeley, CA, with permission)

in areas slated for hydropower development. Other areas of concern are the impacts of land use and climate change on water resources, water allocation, and aquatic and riparian environments (Kite 2001), and the loss of water from altered flow patterns and evapotranspiration from impoundments in the tropics as predicted by the GRACE Project (Ramillien et al. 2005).

Dam building projects will add to the negative impacts of climate change and contribute to accelerated eutrophication with degraded water quality. Unprecedented flooding will hasten the rate at which reservoirs fill with nutrient laden sediment and the new sediment load coupled with changes in the normal stratification patterns of reservoir water due to temperature/density changes will upset the stoichiometric relationships of key chemical species that play a major role in accelerated eutrophication and overall ecosystem balance.

The stoichiometry of different chemical species of nitrogen and phosphorus nutrients is of particular importance because of their critical role in regulating the primary producer communities at the base of aquatic food webs. Damming a river triggers sudden biological, chemical, and physical habitat alterations that produce imbalances in the microbial communities at the base of the river food web and also negatively impact higher trophic levels including fish. Studies have documented lowered fish diversity in tropical lakes caused by eutrophication- induced turbidity that constrained the color recognition essential for sexual selection (Seehausen et al. 1997). Imbalanced microbial communities contribute to a trophic cascade that can lead to seriously degraded water quality, frequent occurrence of cyanobacterial blooms with lethal cyanotoxins, Trihalomethane (THM) precursors, taste and odor compounds, and changes in aquatic habitat that favor increased waterborne disease transmission.

19.2 Mekong River Habitat Ecology – Benchmark Studies of Pre-impoundment Conditions

Accelerated eutrophication and habitat change following impoundment at different sites on the Mekong River and its tributaries in Thailand, Laos, and Cambodia can dramatically impact the trophic status, water quality, biodiversity, and disease transmission patterns in the built reservoirs and the river upstream and downstream of the dam. Detailed information on the sediment, water quality, and biological habitat structure in the Mekong River watershed is essential to accurately monitor river ecological integrity and changes in critical biological habitat over time.

The first ecological studies of Mekong River shoreline habitat included sediment, water quality measurements, and biological surveys completed during 1971–1973 by the Smithsonian Institution (Washington, DC, USA) and the Faculty of Tropical Medicine, Mahidol University (Bangkok, Thailand). Although the studies focused on waterborne disease, they can be used with the more recent detailed data collected by the Mekong River Commission (MRC) from 1985 to the present time. Together, the 1971–1973 benchmark data and the MRC data can serve as a very useful predictive tool to estimate accelerated eutrophication, current and future changes in water quality, and potential waterborne disease transmission resulting from the nexus of hydropower projects, climate change, and accelerated eutrophication.

The 1971–1973 benchmark studies focused on two sites of documented *Schistosoma mekongi* transmission by the aquatic snail vector *Neotricula aperta* (*Lithoglyphopsis aperta; Tricula aperta*), and also provided general baseline information on water quality, sediment characteristics, dominant species of snail populations, aquatic and shoreline plant species, and microbial species in the free-flowing river (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data, Smithsonian Institution – Mahidol University 1974).

19.2.1 Study Areas

Khong Island, Laos (Latitude 14°7′30″N, Longitude 105°51′45″E), is the site of the first confirmed report of the transmission of human schistosomiasis by the snail vector *N. aperta* (Kitikoon et al. 1973) and also the best-documented site of schistosomiasis transmission at that time. Ban Dan represented a typical upstream village in NE Thailand (Latitude 15°19; 15″N, Longitude 105°30′45″E) near the confluence of the Mun and Mekong Rivers with established

Fig. 19.2 Regional map Mekong River Basin with study sites

populations of *N. aperta* and reports of *S. mekongi* transmission (Fig. 19.2).

The studies extended over a complete monsoon cycle punctuated by periods of shoreline instability during the wet season (May to October) and relative stability during the dry season (November to April) and provided (1) a general description of river shoreline habitats during the periods of dynamic change from wet to dry season and (2) a specific characterization of the dry season shoreline habitat supporting both algal and cyanobacterial communities and the snail vector *N. aperta* during active periods of waterborne disease transmission.

The benchmark studies are useful descriptions of the trophic structure and function of the Mekong River prior to any impoundments on the river mainstream or its major tributaries. The data offer useful insights into habitat changes that can result from accelerated eutrophication following the damming process. Along the lower Mekong, the governments of Laos, Thailand, and Cambodia are planning to build a cascade of 11 dams on the river's mainstream and scores of additional dams on its tributaries. The projects are being proposed by Chinese, Vietnamese, Malaysian, and Thai developers, with financing presumably from public and private financial institutions in their home countries. The growth of regional capital has fueled the resurgence of these projects, which have been on the drawing board for decades.

19.2.1.1 Threats to Biological Water Quality – Cyanotoxins and Schistosomiasis

A survey of recent research indicates that accelerated eutrophication continues to produce severe episodes of water quality degradation from cyanobacterial population increases (i.e., blooms) with cyanotoxins, increased concentrations of THM precursors, and taste and odor compounds (Lanza 1995, Conley et al. 2009, Noges et al. 2008). Concerns about the effects of cyanotoxins on overall water quality and human and animal health are a major concern and a recent survey indicated that in the Asia Pacific Region, 54% of lakes are eutrophic (Chorus and Bartram 1999).

Recent estimates indicate that 600 million people are at risk of schistosomiasis and 200 million people worldwide are infected with the snail-vectored waterborne disease. Schistosomiasis is considered to be second only to malaria as a global waterborne disease threat, with approximately 2 million severe infections and 200,000 fatalities each year (WHO 1973). Malaria, dengue fever, and other snail-vectored waterborne diseases are also of concern in the Mekong River watershed (Yossepowitch 2004, Graczyk and Fried 2007).

19.2.2 Hydropower Projects, Accelerated Eutrophication, Water Quality, and Waterborne Disease Transmission

19.2.2.1 General Habitat Dynamics

Sediments and Water: Physical and Chemical Factors

The geomorphological profile of the Mekong River shoreline zone during the dry season offered a wide array of microhabitat types (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data, Smithsonian Institution 1974). However, it appears that the optimal habitat for cyanobacteria and vectors of schistosomes and other snail-vectored waterborne diseases is both patchy and limited (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data). The habitat characteristics supporting cyanobacteria and schistosomes and their snail vectors during the dry season can be viewed as a microcosm of potential habitat types after the impoundment of the free-flowing river.

Sediment substrate dominated by (1) sand and/or mud, (2) rock outcroppings, and (3) islets of vegetation were typical of the Mekong shoreline areas studied. It was not uncommon to note mixtures of the dominant substrate types or shoreline stretches primarily of one particular type. Small islands with similar substrate configurations often augmented the main river shoreline by providing added microhabitat for biological communities.

The shoreline zone of the river was heavily regulated by the extreme changes in water discharge over a typical monsoon cycle (Table 19.2). A marked decrease in discharge and velocity occurred during the transition from rainy season (May to October) to dry season (November to April) producing a dramatic change in the sedimentation character of the littoral habitat. Many of the islands and shoreline terraces submerged during the turbulent discharge of the rainy season re-appeared in the dry season recharged with fresh nutrient-rich deposits (Table 19.1).

Total residue (dissolved and suspended) added to the Mekong River water during the turbulence of the rainy season was deposited as alluvia along the littoral zone during the onset of reduced current velocity accompanying the dry season. Total residue values in rainy season waters increased more than 60% over typical dry season values and total non-filterable values (suspended material) increased 30–70% over the same period.

Turbidity values decreased from 115 to 180 Jackson Turbidity Units (JTU) during the rainy season to clear water values of 30 or below in the dry season, thus permitting increased light penetration to the shoreline water column. Alkalinity and calcium values, both important parameters conditioning the shoreline zone food web, rose from the dilute levels of the early rainy season to more concentrated levels in the late rainy season to early dry season.

The establishment of early dry season conditions between December and February was a relatively abrupt transition marked by dramatic change in the shoreline zone structure and function. Reduced discharge, turbulence, velocity, turbidity, and less dilution of soluble cation and anion biological growth regulators produced a new set of stable conditions suitable for biological development (Table 19.2). Increased substrate availability, light penetration, and nutrient

Table 19.1 Shoreline sediment characteristics Mekong River 1972

	MeB[b]		MeK[c]		
Particle size (%)[a]	Mean	S.D.	Mean	S.D.	Significant $P = 0.05$
Very coarse sand (1.0–2.0)	0.08	0.04	0.05	0.71	
Coarse sand (0.5–1.0)	0.53	0.20	1.33	1.81	
Medium sand (0.25–0.05)	3.20	1.95	1.34	0.85	
Fine sand (0.10–0.25)	29.25	18.8	25.41	10.7	
Very fine sand (0.05–0.10)	16.88	7.79	45.87	7.03	X
Total sand (0.05–2.0)	50.12	27.1	77.94	5.80	
Silt (0.002–0.05)	38.05	21.1	16.36	5.30	
Clay (<0.002)	11.83	6.08	5.70	3.20	
Organic carbon (%)	0.75	0.33	0.25	0.09	X
Available phosphorus (mg L^{-1})	30.43	3.81	31.20	3.82	
Active iron (%)	1.61	0.58	0.99	0.11	X
Saturation extract EC_e @25°C (µmho)	0.62	0.07	0.81	0.44	
Soluble cations (meq 100 g^{-1})					
Ca^{2+}	4.93	0.63	5.80	2.43	
Mg^{2+}	1.15	0.13	1.45	0.82	
Na^+	0.76	0.13	1.21	0.89	
K^+	0.13	0.02	0.33	0.38	
Soluble anions (meq 100 g^{-1})					
Cl^-	0.60	0.16	1.00	0.52	
HCO_3^-	3.87	0.45	3.39	0.44	
CO_3^-	0.00	0.00	0.00	0.00	
Effective exchangeable cations (meq 100 g^{-1})[d]					
Ca^{2+}	13.63	6.20	7.12	1.12	X
Mg^{2+}	1.28	0.75	0.86	0.25	
Na^+	0.02	0.04	0.01	0.01	
K^+	14.93	6.99	8.16	1.20	

[a] Bracketed values are particle size ranges in mm

[b] MeB, Mekong River at Ban Dan, Thailand ($n = 6$)

[c] MeK, Mekong River at Khong Island, Laos ($n = 7$)

[d] Extractable cations (by 1 N KCl) minus soluble cations (in saturation extract)

replenishment were evident. By January or February many shoreline locales had quiescent pool areas sheltered from the flowing current of the river mainstream by adjacent islands.

Biological Factors

Filamentous Chlorophyta sp. dominated by profuse growth of *Ulothrix* sp., *Cladophora* sp., and *Spirogyra* sp. formed dense ubiquitous mats and began a rapid decline induced by crowding effects. In some littoral areas, algal mats growing attached to rock outcroppings had filaments extending to approximately 1 m in length and provided considerable secondary substrate for snails and insects. The mats soon declined in situ or fragmented from their attachment substrate and floated offshore. The dominant genera in the primary producer community at Khong Island, Laos, are provided in Table 19.3. The cyanobacteria *Oscillatoria* sp. and *Spirulina* sp. were common in the microbial community indicating the potential for cyanobacterial water quality problems (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data).

In April 1972, many species of immature snails, including *N. aperta*, were noted in the stable habitat areas in relatively high density on various substrates including submerged rocks, mud, sand, twigs, leaves, shells, and filamentous algae. Laboratory culture studies of *N. aperta* completed subsequent to these observations indicated that *N. aperta* had an ova

Table 19.2 Cation and anion analyses Mekong River 1972–1973[a]

Date		Ca^{2+}	Mg^{2+}	K^+	Na^+	HCO_3^-	CO_3^-	CL^-
01/20/72	meq L^{-1}	1.26	0.48	0.04	0.33	1.13	0.17	0.30
	mg L^{-1}	25.30	5.84	1.56	7.59	68.95	5.10	10.63
03/26/72	meq L^{-1}	1.38	0.53	0.03	0.41	1.86	0.00	0.43
	mg L^{-1}	27.71	6.45	1.17	9.43	113.50	0.00	15.24
04/22/72	meq L^{-1}	1.36	0.58	0.04	0.50	2.01	0.00	0.52
	mg L^{-1}	27.21	7.06	1.56	11.50	122.65	0.00	18.43
06/07/72	meq L^{-1}	0.66	0.29	0.05	0.55	0.98	0.00	0.61
	mg L^{-1}	13.25	3.53	1.96	12.65	59.08	0.00	21.62
08/06/72	meq L^{-1}	0.87	0.26	0.03	0.15	1.24	0.00	0.26
	mg L^{-1}	17.47	3.16	1.17	3.45	75.67	0.00	9.21
09/29/72	meq L^{-1}	1.04	0.32	0.03	0.82	1.90	0.00	0.27
	mg L^{-1}	20.88	3.89	1.17	18.86	115.95	0.00	9.57
10/22/72	meq L^{-1}	0.98	0.35	0.04	1.20	2.07	0.00	0.25
	mg L^{-1}	19.68	4.26	1.56	27.60	126.31	0.00	8.86
11/28/72	meq L^{-1}	0.74	0.29	0.04	0.46	1.18	0.00	0.43
	mg L^{-1}	14.86	3.53	1.56	10.58	72.00	0.00	15.24
01/08/73	meq L^{-1}	0.81	0.46	0.03	0.82	1.74	0.00	0.42
	mg L^{-1}	16.26	5.59	1.17	18.86	106.17	0.00	14.88
02/18/72	meq L^{-1}	1.20	0.55	0.03	0.06	1.86	0.07	0.39
	mg L^{-1}	24.09	6.69	1.17	13.80	113.49	2.10	13.82

[a] Samples collected at Ban Dan, Thailand

Table 19.3 Microbial primary producer community Mekong River, Khong Island, Laos[a]

Organism/assemblage	Microhabitat
Oedogonium sp. *Navicula* sp. *Oscillatoria* sp. *Chlorococcum* sp. *Chlamydononas* sp.	Approximately 1 m offshore, collected on Styrofoam substrates
Ulothrix sp. *Oscillatoria* sp. *Malleodendron* sp. *Navicula* sp.	Associated with floating wood debris and rock outcroppings, approximately 3 m offshore. *Malleodendron* sp. and *Navicula* sp. epiphytic on *Ulothrix* sp.
Azolla sp. *Navicula* sp. *Spirogyra* sp.	Nearshore, free floating
Cladophora sp.	Offshore attached to rock outcroppings and floating wood debris
Synedra sp.	Nearshore, epipelic/epipsammic in mud/sand
Fragilaria sp. *Spiulina* sp.	Nearshore, epiphytic on twigs
Closterium sp. *Navicula* sp.	Nearshore, epiphytic on leaves
Microspora sp.	Nearshore, free floating

[a] Collections from Ban Xieng Wang Village area on Khong Island, Laos, the site of documented schistosomiasis transmission

incubation period of about 4 weeks with the young reaching maturity in 16–20 weeks. Logarithmic growth began between 12 and 14 weeks with egg laying about 6–8 weeks later (Liang and van der Schalie 1975). Assuming a similar time sequence in the natural habitat, *N. aperta* and other Hydrobiidae probably attached ova to suitable substrate in late January or early February.

The benchmark studies documented the establishment of extremely stable shoreline zone habitat between January and May. During this period habitat structure was optimal for both algal and snail logarithmic growth and post-logarithmic development. Conditions were also optimal for the schistosome transmission cycle in littoral areas. A rapid change in habitat stability accompanied the onset of the rainy season in late May to early June during the final development and reproduction period of the snail life cycle. Increased flow and turbulence raised the river levels between 15 and 20 m and total residue levels were elevated to 430–530 mg L^{-1}, producing a massive scouring effect during the freshet.

Adult hydrobiid snails were collected in November immediately after the maximum flow conditions. The snails were found in crevices and holes on the underside of large boulders in the littoral zone indicating the survival of some adults during periods of high instability. Because hydrobiid snails, including

N. aperta, were sometimes associated with algal mats and other unattached floating substrate, the potential for export of the snails to other areas was also present (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data).

Habitat Dynamics During Disease Transmission

Figure 19.3 provides a schematic description of the basic food web supporting the microbial and snail-schistosome community at Khong Island, Laos, during a typical dry season along the shoreline areas of the Mekong River and its tributaries. Allochthonous nutrients in the form of leaf-litter detritus and other organic debris is deposited in shoreline areas with reduced current. Primary substrate existed as submerged brush in the form of twigs, branches, and leaves largely from the euphorb shrub *Homonoia* sp. growing as discontinuous patches along shoreline areas. Additional primary substrate was provided as rock outcroppings and sediment. In addition to providing attachment, primary substrate served as a reservoir of essential nutrients for algal, bacterial, and snail assemblages. Substrate nutrients were produced by the mineralization of organic material during microconsumer decomposition activity.

Attachment and/or association with substrate is critical to the completion of algal, snail, and microconsumer life cycles. Substrate attachment and nutrients allowed in situ photosynthesis and the proliferation of algae including diatoms serving as the snail communities' major food source. Many gastropod snails, e.g., *N. aperta*, require substrate for attachment of their egg masses during incubation and for their relatively dense juvenile populations.

Algal and macrophyte assemblages made up the primary producer segment of the littoral zone food web and provided food, oxygen, and secondary substrate

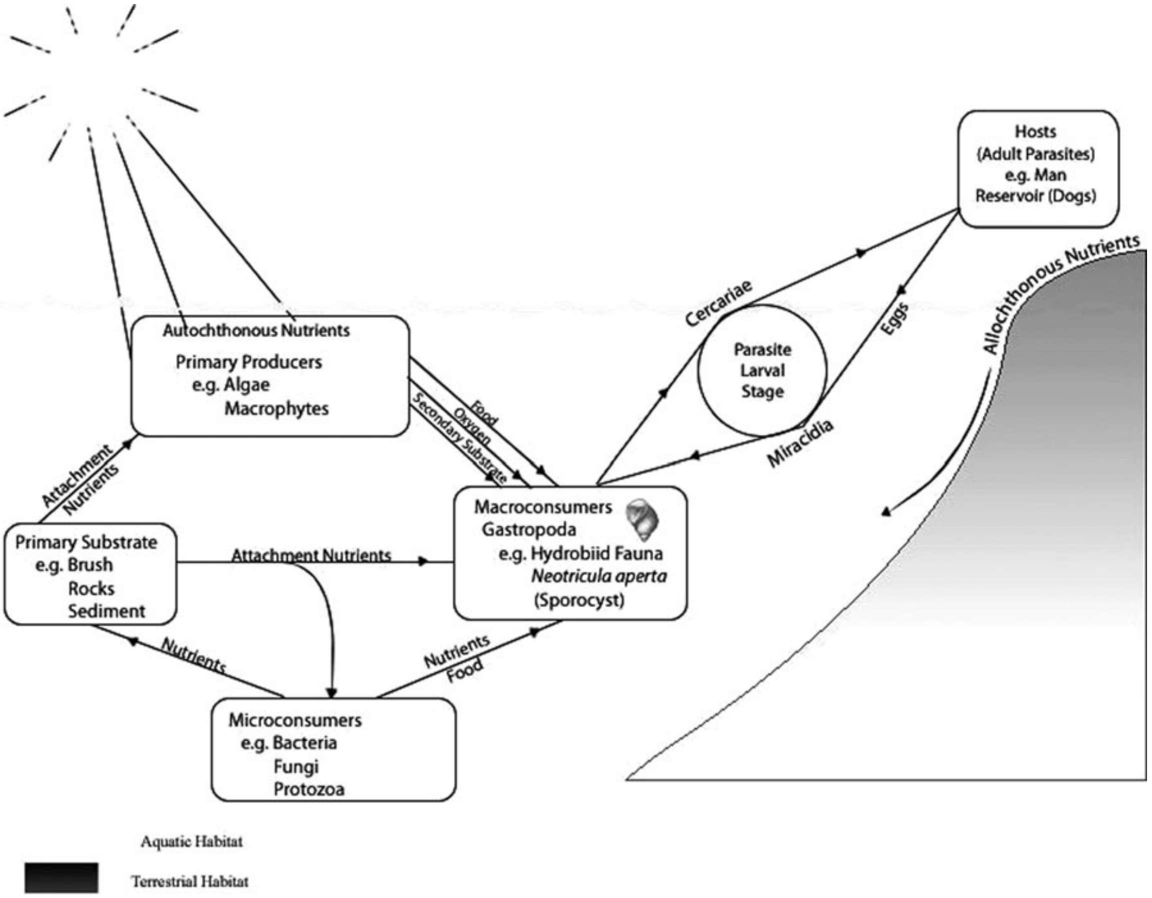

Fig. 19.3 Mekong River Shoreline Habitat and Food Web, Khong Island, Laos 1972

for the macroconsumers including snails (Fig. 19.3; Table 19.3). Supplementary food and nutrients were available to the snail assemblages from the microconsumer component.

In balance, the interaction of the habitat components outlined in Fig. 19.3 supported a healthy, diverse (up to 80 species) hydrobiid snail community. And optimizing the habitat for the snail community supported the transmission cycle of human schistosomiasis spread by contact with water containing infective cercariae. It is important to note that the schistosome larval stages and disease cycle were not separate entities isolated in the shoreline zone community, but were integrated components of the littoral zone ecosystem. A detailed characterization of the major components outlined in Fig. 19.3 follows.

Sediment Characteristics

Table 19.1 summarizes the analysis of littoral zone sediment collected at Ban Dan and Khong Island. Results of the two sample *t*-tests at $p = 0.05$ indicated significant differences in 4 of 23 sediment characteristics; very fine sand, percent organic carbon, active iron, and effective exchangeable Ca^{2+}. Average very fine sand values from Khong Island (45.87 S.D. 7.03) were approximately three times those noted at Ban Dan (16.88 S.D. 7.79). Both sites had greater than 50% total as sand with the balance in silt and clay (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data).

Ban Dan sediment had percent organic carbon values (0.75 S.D. 0.33) three times greater than Khong Island (0.25 S.D. 0.09) and greater percent active iron values (1.61 S.D. 0.58) than Khong Island (0.99 S.D. 0.11). Effective exchangeable Ca^{2+} values in Ban Dan sediment (13.63 S.D. 6.20) were also higher than Khong Island (7.12 S.D. 1.12). Similar trends of soluble cations and anions and effective exchangeable cations were noted in both study areas (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data).

The statistically significant differences in sediment ($p = 0.05$) noted in very fine sand, percent organic carbon, active iron, and effective exchangeable Ca^{2+} can be partially explained by the fact that Khong Island is downstream from Ban Dan. The higher values of percent fine sand at Khong Island and lower values of organic carbon, active iron, and effective exchangeable Ca^{2+} could be associated with sedimentation/precipitation phenomena (Table 19.1).

Water Chemistry

Table 19.2 presents a detailed water chemistry profile of the Mekong River over a typical monsoon cycle. Table 19.2 summarizes cation and anion values measured over one monsoon cycle. Water temperature ranged from 26.5 to 32.0°C with the highest value noted in the dry season. Dissolved oxygen levels ranged between 6.0 and 7.8 mg L^{-1} with percent saturation between 73 and 99 and dry season values between 6.5 and 7.8 mg L^{-1} and 85–99% saturation (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data).

Carbon dioxide levels extended from non-detectable to 12 mg L^{-1} with a very slight trend toward higher dry season values. Alkalinity and total hardness had similar monsoon cycle trends with values between 40 and 80 and 40–75 mg L^{-1}, respectively. Both parameters dropped at the onset of the rainy season and had elevated levels in September and/or October with a general tendency for increased levels during the dry season. Hydrogen ion concentration fell to 7.8–8.6 except for a single observation in October that dropped to 6.5. Turbidity values ranged between 25 and 180 JTU with lower values noted during the dry season.

Total dissolved solids and electroconductivity values followed the same general patterns described for alkalinity and total hardness with initial decreases during the early rainy season and elevated levels in September and October. Total dissolved solids ranged between 60 and 154 mg L^{-1} with the two lowest levels (60 and 68) noted in the late rainy season to early dry season and the two highest levels (148 and 154) recorded in the dry season. Electroconductivity measurements extended from 128 to 240 μmho with the highest levels (>220) noted during the dry season (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data). Cation and anion analyses are summarized in Table 19.2. Ca^{2+}, Na^{+}, and HCO_3^{-} had trends toward initial decreases during the early rainy season and elevated levels in September and October. One exception was noted with Na^+ decreasing from 12.65 mg L^{-1} in June to 3.45 mg L^{-1} in August prior to rising to 18.86 and 27.65 mg L^{-1} in September and October, respectively. Ca^{2+} and HCO_3^- tended to progressively increase during the dry season and had values over a monsoon cycle between 14.86 and 27.71 mg L^{-1} for Ca^{2+} and 68.95–126.31 mg L^{-1} for HCO_3^-. Na^+ values ranged between 3.45 and 27.60 mg L^{-1} over a monsoon cycle. Mg^{2+} had decreased values during the rainy season ranging between 3.16 and 4.26 mg L^{-1} with a trend toward increased concentration during the dry season. Mg^{2+} values fell between 3.16 and 7.06 mg L^{-1} over a monsoon cycle. K^+ remained fairly stable over a monsoon cycle with concentrations between 1.17 and 1.96 mg L^{-1}.

Cl^- levels fluctuated over a monsoon cycle with dry season values showing either little change or progressive increases as the season advanced. Cl^- values ranged from 8.86 to 21.62 mg L^{-1} over a cycle with relatively high values appearing in both rainy and dry seasons. CO_3^- values were only detected during January and February of the dry season when values of 5.10 and 2.10 mg L^{-1} were noted, respectively (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data).

Diel Cycles

Twenty-four hour diel cycle measurements of air temperature, water temperature, dissolved oxygen, and pH were recorded at 4 h intervals at the shoreline and offshore river areas on two dates during the dry season (Fig. 19.4). Similar trends marked by high stability were noted with all parameters at the two stations on both dates. Dissolved oxygen values at both stations ranged between 6.2 and 8.1 mg L^{-1} with 78–107% saturation. Highest values occurred at 12 noon or 4 p.m. while lowest values appeared between 8 p.m. and 8 a.m. The pH levels remained quite stable over a diel cycle with values between 8.00 and 8.25.

Air temperature ranged between 25.0 and 34.5°C and water temperature between 27.5 and 33.0. Samples collected mid-way in a diel cycle were noted as turbidity 10–25 JTU and electroconductivity 245–252 μmho. Milligrams per liter of other mid-cycle parameters were noted as total residue 236–258, total filterable residue (total dissolved solids) 147–170, total nonfilterable residue 70–103, carbon dioxide 4–10, alkalinity 75–85, total hardness 70–75, Ca^{2+} 27.31–28.51, Mg^{2+} 7.06–7.79, K^+ 1.56–1.96, Na^+ 11.50–12.65, HCO_3^- 114.11–122.65, CO_3^- not detectable, and $C1^-$ 16.66–24.10 (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data).

The detailed diel profiles documented a very stable physical–chemical environment for the biological community in the Khong Island area. The shoreline habitat conditions available for schistosome vectors and their algal-bacterial food sources. parasite larval stages (i.e. cercariae), and cyanobacteria are optimal during the dry season. It is possible that many patches of shoreline area upstream, downstream and along the impoundment perimeter of hydropower projects could have the optimal habitat noted in Figure 19.4 and result in increased waterborne disease transmission, cyanobacterial blooms, and other water quality problems.

Primary Producer Community

The dominant components of the shoreline primary producer community included 12 genera of algae, 1 genus of aquatic vascular plants, and 2 genera of cyanobacteria. The dominant algal species represented the Chlorophyta and Chrysophyta (including diatoms) and are listed by organism assemblage and littoral zone microhabitat in Table 19.3. Styrofoam substrates anchored in the water approximately 1 m offshore yielded three green algae, *Oedogonium* sp. a filamentous type, *Chlorococcum* sp. a unicell, and *Chlamydomonas* sp. a flagellate. The filamentous cyanobacterium *Oscillatoria* sp. and the diatom *Navicula* sp. were also part of the artificial substrate community (Lanza 1973, Smithsonian Institution Waterborne Disease Project, 1971–1973, Unpublished data).

Assemblages associated with floating wood and rock outcroppings approximately 3 m offshore included the filamentous *Oscillatoria* sp. and *Ulothrix* sp. with well-developed yellow-brown epiphytic communities of *Navicula* sp. and *Malleodendron* sp. The green filamentous alga *Cladophora* sp. was collected from several offshore sites, also attached to floating wood and rock outcroppings, but without epiphytic

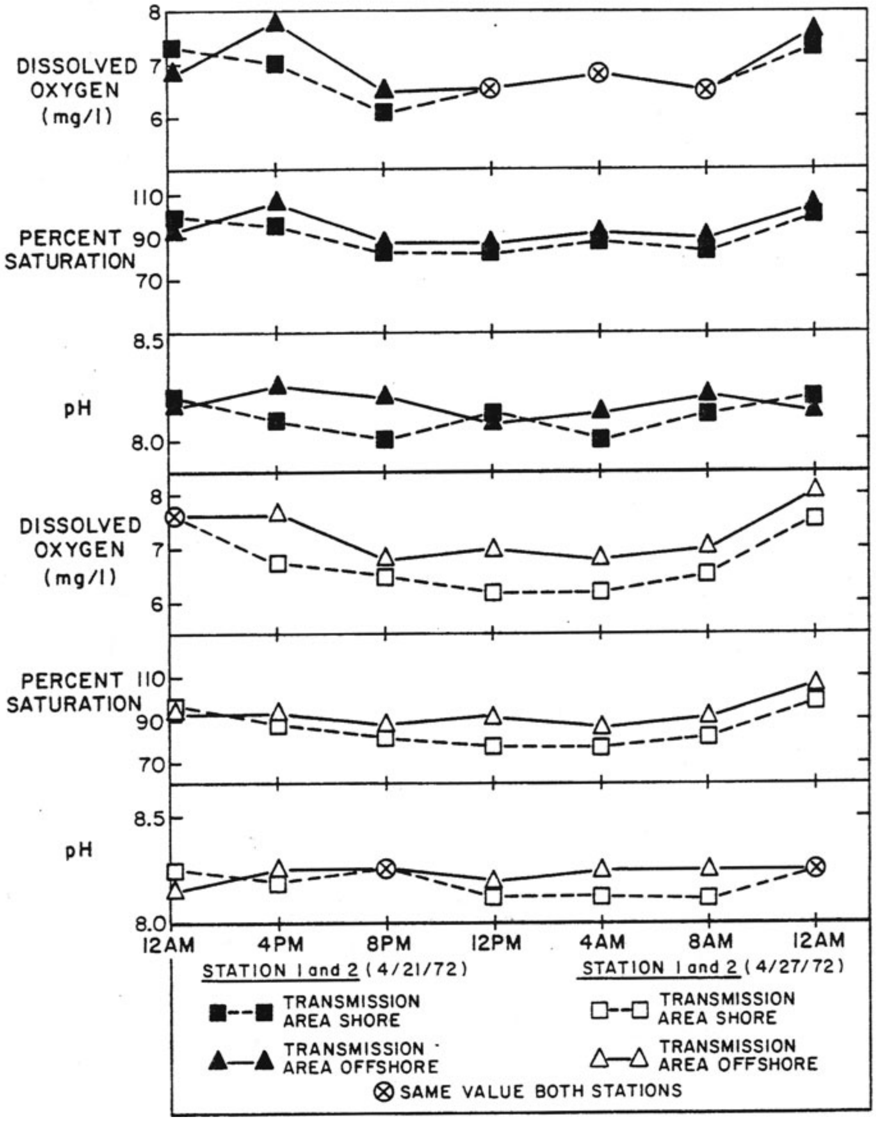

Fig. 19.4 Diel cycles of selected water quality parameters – N.aperta Habitat, Khong Island, Laos 1972

communities. Nearshore free-floating species included mats of predominately *Spirogyra* sp., a common green filamentous alga, *Navicula* sp., and *Microspora* sp., a green filamentous form. *Fragilaria* sp., a diatom, and *Spirulina* sp., a filamentous cyanobacterium, were seen as common epiphytes of twigs, and *Closterium* sp., a green unicell, and *Navicula* sp. were found epiphytic on leaves. *Synedra* sp., a diatom, was epipelic on mud and epipsammic on sand substrate nearshore. The only vascular plant evident in relatively high densities was the free-floating duckweed or water velvet, *Azolla* sp.

19.3 Using the Benchmark Studies to Predict Accelerated Eutrophication Impacts from Dam Impoundments

The 1971–1973 benchmark studies of the Mekong River indicated that the water quality, sediments, and the natural habitat ecology in the Kong Island and Ban Dan areas support a healthy and balanced aquatic food web with two major characteristics: (1) a relatively small number and a patchy distribution of shoreline sites (see Fig. 19.3) that can support algal and snail vector communities and (2) balanced microbial communities at those sites adapted to the monsoon cycles with a dominance of green algae and relatively low density of cyanobacteria (see Table 19.3).

The natural flow regime of rivers organizes and defines the aquatic habitat characteristics that support healthy food webs (Poff et al. 1997). Dams convert lotic systems to lentic systems and completely change the habitat structure and ecological function of the free-flowing river. Major hydropower projects are planned for the Khong Island area and downstream in the Mekong River at Stung Treng and Sambor (see Fig. 19.2). The planned hydropower impoundments coupled with the impacts of climate change can result in a cascade of wide scale water quality degradation punctuated by accelerated eutrophication, biodiversity loss, imbalanced aquatic food webs, water toxicity from sediment releases and cyanotoxins, and increased waterborne disease transmission.

Disrupting the natural flow regime will most likely create increased shoreline area surrounding the planned impoundments. Sections of the new shoreline can create additional optimal habitat for snail vectors of schistosomiasis (see Fig. 19.3) and other major snail-vectored trematode diseases (Yossepowitch 2004, Graczyk and Fried 2007). The absence of sudden high flows during the wet season due to dams can extend the residency time of breeding snail populations and the transmission of disease. Increased average temperature from climate change can add to the problem. Recent research using Growing Degree Day (GDD) models of *Schistosoma japonicum* in China has demonstrated that climate change can further contribute to the transmission of snail-vectored disease by increasing the extent and level of disease transmission (Yang et al. 2006).

Following impoundment of the Mekong River or its tributaries, offshore populations of reservoir cyanobacteria (e.g., *Oscillatoria* sp., see Table 19.3) could proliferate in reservoir water with optimal conditions of temperature, light, and nutrient ratios (N:P). Nutrient imbalances in hydropower project reservoirs are one major factor that can influence accelerated eutrophication and optimize the habitat that supports increased cyanobacterial growth.

A detailed description of the microbial habitat alteration typical of accelerated eutrophication occurring in many lakes or reservoirs is provided in Lanza (1995). Nutrients and nutrient limitation are major factors influencing the habitat preferences of cyanobacteria. Data from Mekong River Commission (MRC) water quality assessments collected between 1985 and 2000 indicate N:P ratios in the Mekong River of 4.5:1 and 13:1 at Pakse, Laos. and Kratie, Cambodia, respectively (MRC 2010). Relatively low ratios of N to P (i.e., ratios below 30:1) often contribute to increased numbers of toxin-producing cyanobacteria (Smith 1983, Chorus and Bartram 1999, Havens et al. 2003). The N:P ratios in the Mekong River at Pakse and Kratie are cause for concern because of the close proximity of Pakse and Kratie to the planned hydropower projects at Stung Treng and Sambor. Increasing temperatures from climate change can also contribute to increased frequency and duration of harmful cyanobacterial blooms and a recent episode in Spain attributed to climate change resulted in the loss of fish and waterfowl due to cyanotoxins from a bloom of *Microcystis aeruginosa* (Dixon 2008).

Based on recent experience in the region, other water quality problems that can be anticipated in Mekong River watershed impoundments undergoing accelerated eutrophication include rapid decrease in dissolved oxygen due to nutrient-stimulated algal growth producing fish kills in the reservoir, anoxic bottom sediments that release toxic materials, nutrients, and greenhouse gases (e.g., hydrogen sulfide, phosphorus, methane), and serious taste and odor problems from volatile organic compounds produced by nuisance microbes (Lanza 1996, 1997, 2005). Phosphorus reserves in Mekong River sediment were very high (i.e., about 30 mg L^{-1}; Table 19.1) and indicate the potential for nutrient additions from sediment releases following dissolved oxygen deficits typical of bottom sediments in tropical impoundments (Lanza 1995, 1996, 1997, 2005).

The 1971–1973 benchmark data on water quality parameters in the Mekong River during a monsoon cycle can be used along with more recent detailed data from the MRC to help to predict water quality degradation, accelerated eutrophication, and the potential for increased waterborne disease during climate change. Recent research on the effect of climate change on a watershed system indicates that variability of flows due to climate change can influence nutrient levels and other contaminants in the watershed (Marshall and Randhir 2007), and that TSS, alkalinity, and conductivity can be used as sensitive water quality parameters for monitoring impacts of changing climate in the lower Mekong River (Prathumratana et al. 2007).

19.4 Summary

Benchmark studies completed during 1971–1973 describe how discontinuous areas of the Mekong River and its major islands naturally evolve into highly stable shoreline and offshore river habitat during the dry season. Water quality profiles recorded over an annual monsoon cycle produced a pattern indicating poor habitat conditions for shoreline algal and snail communities during the turbulent high flow periods of the rainy season. At Khong Island, Laos, a high level of physical and chemical stability evolved during the dry season with enhanced habitat for the primary producer and snail vector communities (Fig. 19.3, Table 19.3). Reduced flow and less dissolved and suspended material eliminated scouring impacts and resulted in lower turbidity and increased light penetration in substrate areas colonized by algae and snails (Fig. 19.4). Greater light penetration from reduced turbidity stimulated primary productivity adding essential food, oxygen, and secondary substrate to the habitat structure and supported a highly diverse community of algae and snails (Fig. 19.3, Table 19.3). During the dry season, shoreline and offshore water quality and habitat structure (Fig. 19.3 and 19.4; Tables 19.1, 19.2 and 19.3) supported a limited and patchy distribution of algae, cyanobacteria, and snail communities including the vectors of schistosomiasis and other waterborne diseases.

In the absence of dams, shoreline, and offshore river habitat in the Mekong River watershed supports a healthy and balanced algal and cyanobacterial community. Hydropower projects in the Mekong River watershed Prathumratana, I, Sthiannopkao, S, and KW Kim The relationship of climatic and hydrological parameters to surface water quality in the lower Mekong River, Environ. Int. (2007), doi:10.1016/j.envint.2007.10.011 will obstruct the natural flow regime and convert free-flowing lotic systems into lentic systems. Damming the Mekong River and its tributaries can mimic many, if not all, of the dry season conditions noted at Khong Island, Laos. The result would be additional patches of year round habitat for snail vectors (e.g. N.aperta) and waterborne disease organisms (e.g. schistosomes and other parasites) in impoundment areas and in other areas above and below the dam. The dams and the resulting impoundments can also upset the nutrient stoichiometry in the reservoir water and sediments producing blooms of toxic cyanobacteria. The negative impacts of climate change add to the problems and can contribute to accelerated eutrophication, water quality degradation and toxicity, loss of biodiversity, and increased waterborne disease transmission.

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Index

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