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Eutrophication Management and Ecotoxicology

With 75 Figures

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1 Eutrophication and the Ecosystem

1.1 Background

Eutrophication of surface waters is generally recognised as a matter for environmental concern. Eutrophication is characterised by increased algal growth, with an increased incidence of toxic cyanobacteria blooms and a decrease in the abundance of species.

Some of the manifest problems brought about by prolific algal biomass include: turbid waters; anoxic conditions; bad smell and chironomid and Culex midge plagues (Vollenweider 1990; Moss et al. 1996a; Carpenter et al. 1998). Such eutrophication problems ("eutrophication" *sensu lato*) are generally considered to be the consequence of enhanced nutrient loadings ("eutrophication" *sensu stricto*) (Likens 1972; Vollenweider 1990; Reynolds 1992; Moss et al. 1996a; Carpenter et al. 1998). Therefore, the management of eutrophicated water bodies is usually primarily focused on the reduction of nutrient loading, supported by a policy of reduced environmental releases of phosphorus from laundry detergents, sewage and agriculture.

However, it became apparent over the past decade, that reduced grazing of algae by daphnids can be a crucial factor determining whether or not nutrient enrichment will lead to eutrophication problems (Moss et al. 1991; Moss et al. 1996b; Reynolds 1994). Biomanipulation of eutrophicated shallow water bodies, thereby improving ecological conditions for daphnids, became a regular tool applied in eutrophication management practice (Benndorf 1990; McQueen 1998; Harper et al. 1999).

Biomanipulation is mainly focussed on the improvement of biological conditions leading to a higher survival rate for daphnids as part of the aquatic foodweb. Examples of biomanipulation measures include: reduction of predation by planktivorous fish and improvement to the submerged vegetation as a shelter for daphnids against predation. More recently, the palatability of suspended particles as a factor determining the grazing efficiency of daphnids has become a topic of interest. High concentrations of resuspended inorganic particles hamper daphnid grazing, while at the same time stimulating algal growth due to increased nutrient releases (Kirk and Gilbert 1990; Ogilvie and Mitchell 1998). Top-down control by daphnids under eutrophicated conditions may also be reduced by the presence of unpalatable algal species (mainly cyanobacteria) that may gain competitive advantage over the heavily grazed palatable algal species.

This book covers another important aspect regarding the improvement of environmental conditions for daphnids, which is necessary for successful eutrophication management, i.e. optimalisation of the abiotic water conditions. Bales et al.

(1993) suggested that the sensitivity of daphnids to saline conditions may be a significant reason for the higher susceptibility of brackish waters to eutrophication. From this perspective, ecotoxicologically reduced daphnid grazing due to micropollutant loadings could be a crucial factor leading to problems associated with eutrophication (Hurlbert et al. 1972; Hurlbert, 1975; Gliwicz and Sieniawska, 1986).

The toxic effects of pesticides and other chemicals on the viability of cladoceran populations reduce their capacity to graze the surplus algal growth caused by increased nutrient availability. In ecotoxicological semi-field studies, it has been observed that cladocerans are amongst the most sensitive species when it comes to toxicant exposure, consequently resulting in a reduction in the top-down control of the algal growth (Day 1989; Yasuno et al. 1993). Based upon the analysis of sediment cores, Stansfield et al. (1989) argued that a switch from submerged plant dominance to phytoplankton dominance (eutrophication) in a series of shallow lakes, i.e. the Norfolk Broads, U.K, during the 1950s and 1960s, was likely to have been due to the poisoning of cladocerans (viz. *Daphnia*) as a result of liberal organochlorine pesticide use.

The aim of this book is to provide a better understanding of the ecotoxicological aspects of eutrophication processes in shallow, temperate fresh waters, so that these processes may become a recognized factor in the restoration of eutrophicated water bodies.

Some basic limnological ecology, which is fundamental to the further contents of this book, is presented in the first chapter. Chapter 2 provides more information on daphnids, encompassing their ecology, grazing efficiency and any subsequent consequences for the control of algal densities. Chapter 3 concerns daphnid ecotoxicology, and provides information on toxicity induced reduction of daphnid grazing effectiveness (so called "toxic anorexia") in experimental settings. Variation in daphnid grazing effectiveness in the field situation is described for two Dutch lakes in Chap. 4. The applicability of an ecotoxicological assessment of eutrophicated water bodies is discussed, and practical tips given, in Chap. 5.

1.2 Eutrophic and Eutrophicated Waters

Nutrients or Algae

In order to acquire a better understanding of the causes of eutrophication in fresh water ecosystems, it is helpful to make a distinction between **"eutrophic" waters** (classification of water according to its intrinsic nutrient status, eutrophication *sensu stricto*) and **"eutrophicated"** (or "eutrophied"**) waters** (perception of manifest water quality problems related to ecological malfunctioning such as turbid water, bad smell and high algal density, eutrophication *sensu lato*).

The total phosphorus concentration of fresh surface waters is generally used as an indicator for the trophic status of that water body. Phosphorus is one of the essential nutrients for algal growth. It is considered to be the prime limiting element determining the biological productivity (algal productivity, and subsequent higher order productivity) in many freshwater aquatic systems.

Probability distribution for trophic categories

mg [P]/m^3

Probability distribution for trophic categories
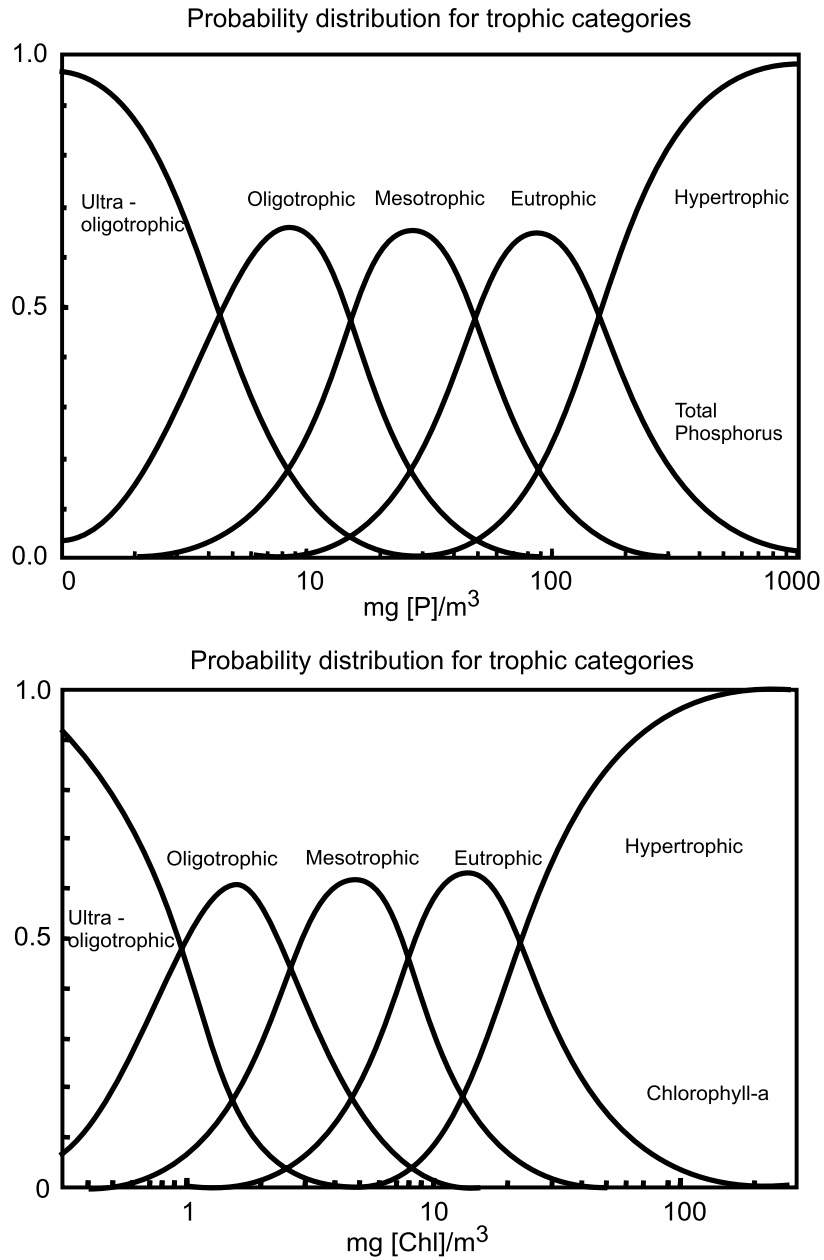

mg [Chl]/m^3

Fig. 1.1. Classification of waters according to the OECD (1982). Above: on the basis of P-loading, and below: on the basis of average algal density
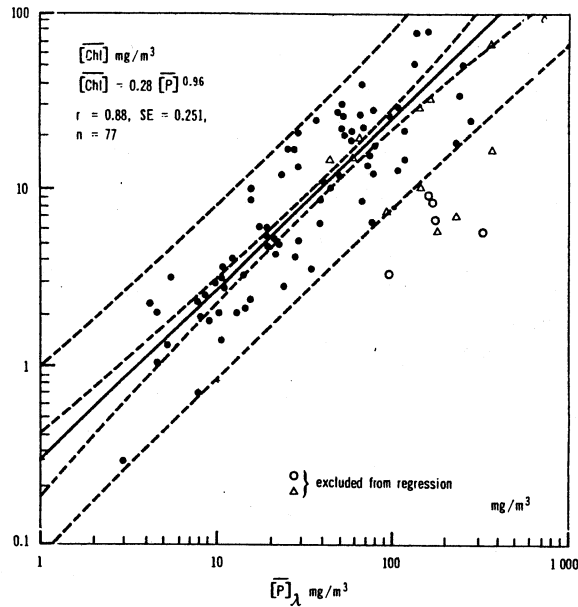
Fig. 1.2. The "Vollenweider-model" (OECD 1982) illustrating large variation in the P:chl-a ratio

Other main nutrient elements present in potential limiting concentrations are nitrogen and silicate. Nitrogen is not a suitable indicator, because it is poorly retained by soils and leaches easily to aquatic systems. Additionally, transformations between various fractions of particulate (organic) nitrogen, dissolved (organic/inorganic) nitrogen and atmospheric (inorganic) nitrogen occur at high rates, ensuring a continuous supply of available nitrogen under most circumstances.

Silicate (ortho-silicic acid) is only essential for the growth of diatoms, which incorporate silicic acid into their frustulers. Therefore, it is not a suitable indicator for total system productivity.

In the classification of the trophic state of waters according to the OECD (1982), no distinction was made between eutrophic and eutrophicated waters. Waters are referred to as mesotrophic, eutrophic or hypertrophic on the basis of their P-loading as well as on the basis of algal densities (Fig. 1.1), with the underlying assumption that nutrient status (P loading) is positively and causally correlated with algal density (Fig. 1.2).

The Vollenweider Regression

The OECD classification was based upon a regression model (known as the "Vollenweider" model) in which the phosphorus concentration (as indicator for the trophic status) of surface waters and mean algal density (as indicator for manifest "eutrophication" problems) were related to each other. The regression model was based upon observed relationships between phosphorus and algal densities in vari-

ous deep upland lakes during the nineteen seventies (Vollenweider and Kerekes 1980). The model has been used often and validated in many situations (CUWVO 1976, 1980, 1988; OECD 1982; Klein 1989). This has led to the perception that the ecological water quality problems related to high algal densities were solely due to "eutrophication" (i.e., nutritional or, more specifically, phosphoric loading of waters).

It should, however, be noted that the "model" demonstrates a broad variation in the relationship between the two indicator parameters (a factor of 10 at 99% confidence), even after plotting on a log-log scale (Fig 1.2). This means that many eutrophic waters exist with relatively low algal densities; and oligotrophic/ mesotrophic waters with relatively high algal densities. Additionally, it should be noted that examples of eutrophic lakes with a relatively low algal density were excluded from the regression (open symbols in Fig 1.2).

Beyond Nutrients

Although phosphorus is a suitable rough indicator for the trophic status of a water body, this does not imply that it is a reliable (causal) predictor of manifest eutrophication problems (Reynolds 1992). The relationship between phosphorus content and eutrophication problems is not fixed, due to the fact that there are many other variables, in addition to phosphorus concentrations, that influence the manifestation of eutrophication problems. In certain circumstances, eutrophic waters are not eutrophicated, while mesotrophic waters sometimes appear to be eutrophicated.

The "Vollenweider-model" had its function in clarifying the role of excessive phosphorus loads in the manifestation of severe eutrophication problems, but for successful eutrophication management it is necessary to look beyond nutrient loads in order to arrive at reliable and sustainable water management options (Golterman 1991; Moss et al. 1994). Nutrient-algal relationships are only a fraction of the complete aquatic food web (Hosper et al. 1992; Scheffer et al. 1993; Leibold and Wilbur 1992; Vijverberg et al. 1993). An extensive statistical data analysis of 231 lakes in the Netherlands (Portielje and van de Molen 1997a; 1997b; 1998) has made it clear that top-down control of algal density development by submerged vegetation (competition) and daphnids (grazing) is a significant factor explaining variation in chlorophyll nutrient ratios. In eutrophication management it is important to understand what has caused the eutrophicated state of the waters, and to recognise the bottleneck situations inhibiting their restoration.

1.3 Trophic Cascades in Freshwaters

The Foodweb

The availability of nutrients is a basic requirement for the development of organic biomass. Algae can respond quickly and opportunistically to increases in nutrient availability. However, the capacity of individual algae to store nutrients is limited. It is the population that retains the nutrients, but only during the growing season.

Decaying algae release nutrients and the following season all nutrients must be re-acquired.

Macrophytes on the other hand, cannot respond as quickly as algae to changing nutrient levels, but do retain nutrients individually within storage organs. Macrophytes are also able to acquire nutrients, both actively and passively, from the substrate on which they grow. This gives them a competitive advantage over algae. The shade created by macrophyte leaves may inhibit algal growth. Some macrophytes can even suppress algal growth by releasing allelochemicals (Mjelde and Faafeng 1997). By preventing algal growth, macrophytes may also gain access to nutrients in the water that might otherwise be utilised by the more rapidly responding algae. As a result, macrophytes are the predominant primary producers in shallow waters where sunlight can reach through to the sediment, whereas algae are the primary producers in deeper waters.

The primary production of algae and macrophytes is the basis of aquatic food webs (see Fig 1.3). The main groups of algal consuming secondary producers in fresh water ecosystems are cladocerans and other zooplankton groups (viz. copepods and rotifers) and filter feeding benthic species (viz. bivalves). Zooplankton (especially cladoceran water fleas) plays a key role in aquatic ecosystems by effectively responding to variations in algal production through its opportunistic population dynamics. Bivalves, though individually capable of filtering large volumes of water, do not play such a central role, since they cannot multiply at the same rate as their algal food source. Therefore, bivalve populations are limited by the minimum amount of food available.

The algal biomass density is the nett result of algal production and algal losses due to sedimentation and grazing by zooplankton or other secondary producers. The production of herbivore biomass in aquatic ecosystems is approximately 2–6% of the primary (i.e., algal) production, which is extremely high compared to terrestrial ecosystems (<< 1%) (Barnes and Mann 1993). Accounting for respiration and defaecation losses, this means that in a healthy aquatic ecosystem that 30–90% of the primary production is consumed and ends up in the aquatic food chain. In plankton dominated systems this is at the higher end of the scale (up to 90–95%). Only a small fraction of the primary production is allocated for the establishment of a standing vegetation (e.g., reed and rush marshes) and temporary algal blooms. Sedimentation and shore deposition of unconsumed algal and plant debris (detritus) brings about decay by benthic detrivores.

Zooplankton is consumed by small fish (e.g., roach and smelt, and the juveniles of larger species) and amphibians, which, in turn, are food for predatory fish (e.g., pike and perch). Zooplankton predation, especially of the larger cladoceran species, may be very high. Therefore, cladocerans usually seek refuge in the macrophyte vegetation and migrate out into open water only during the night when the risk of predation is low (Timms and Moss 1984; Lauridsen and Lodge 1996; Stephen et al. 1998). Zooplanktivorous fish, on the other hand, avoid macrophyte vegetation due to the fact that predators such as pike may be concealed there.

The aquatic food chain provides a food source for many birds (e.g., waterfowl) and various mammals (e.g., otters).

The Role of Macrophytes

Macrophytes as seen in figure can play substantial role in shallow lakes and ponds as they contribute to primary production of organic material that give organic matter to the bacterial loop and invertebrates. Of course, in high trophic and turbid lakes, this process do not occur as a significant contribution in nutrient cycles as lack of light is limitant factor for their growth.

We must distinguish the different types of macrophytes as shown in the (following) figure:
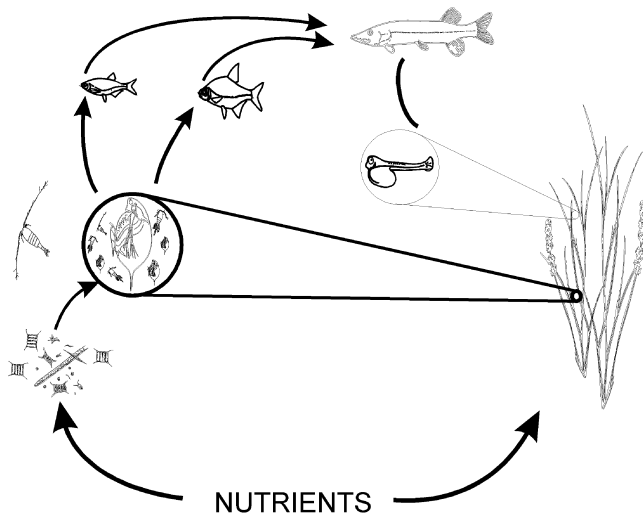

Fig. 1.3. The position of zooplankton (i.e. cladocerans) in the foodweb

As any plants, macrophytes participate to the productivity of the ecosystem. When rooted macrophytes are present, they can export nutrient from sediments and at their decay and senescence contribute to the enrichment of the water column. This phenomenon may be temporarily decreased by periphyton fixed on leaves and stems, nevertheless, N and P are rapidly available for small algae and heterotrophic microflora. Senescing macrophyte tissue and attached algae, as well as organic matter and nutrients retained in them stay in sediments where they collapse in the littoral zone of lakes. The accumulative retention of these inputs of nutrients and carbon can be very high. It should be recalled that most aquatic macrophytes are perennials and much of certain nutrients as phosphorus extracted by plant growth from the sediments is largely translocated back to the rooting tissues at the end of the growing season and taken off from the nutrient pelagic cycle of lakes. In case of shallow lakes and high development of littoral zone, the total nutrient releases to the pelagic zone is reduced. The tendency is for the littoral developments and metabolism to reduce total phytoplankton growth of the lake to levels below the growth potential that would occur if most of the nutrient loads entered the pelagic zone directly. Wetzel (1990) has shown that in these conditions, "in the ecosystem with profundal sedimentation patterns and the converse differ-

ences in system components and their metabolism, one finds a overall decrease in the phytoplanktonic efficiency of nutrient utilization". The relations between P and macrophytes have been discussed also for a long time and it is known that these plants are able to keep quite large quantities as "luxury consumption". In summer, this contribution of macrophytes in the P cycle can participate of decreasing concentrations of the growth capacity of algae in the littoral zones of the lakes, but in autumn, large amounts of P are released. In the same way, the allocation of macrophytes in toxicants explains the leaching of these compounds under specific conditions (pH, redOx...).

Balanced Foodchains

In a healthy ecosystem, a finely tuned balance exists between production and consumption within each link in the food chain (Carpenter et al. 1985). In shallow freshwater bodies, macrophytes are important for the stabilisation of this balance. Their long-term accumulation of nutrients tempers phytoplankton dynamics and at the same time they offer shelter for zooplankton, protecting it from predation. To a certain extent, bivalves also have a stabilising function, as they graze a base level of algae and clear the water of any suspended particles that may interfere with zooplankton grazing.

However, zooplankton, especially cladocerans, controls the short-term variation in phytoplankton dynamics, through their opportunistic response to food availability. The opportunistic response of cladocerans to algal (food) conditions is thus a critical and central step in the trophic cascade of aquatic foodchains. A well-balanced and synchronised coupling of cladoceran development to algal development is, required therefore, especially under mesotrophic or eutrophic conditions.

The Trophic Basis of the Cascade

Nutrients for algal growth become available through the mineralisation of organic materials, organism excretions and leaching from sediments. The cycles of the macronutrients C, N and P are the best understood and are briefly described below. Trace elements may limit primary or secondary production both spatially and temporally. This may, for instance, be the case for Si, which is an important element for diatoms (Hecky and Kilham 1988; Moss and Balls 1989). The cycles and importance of trace elements are less well understood and will, therefore, not be further considered here.

The *carbon cycle* is the basis of all biogenic processes. Atmospheric CO_2 is fixed by autotrophs and enters the food chain, where a major portion of it is rereleased by means of heterotrophic metabolism. Heterotrophic consumers rely, both directly and indirectly, on this autotrophic fixation of CO_2 for their carbon supply. Carbon is, therefore, often used to quantify energy fluxes in ecosystems. Generally, carbon is not considered to be a limiting nutrient for primary production, due to the large available pool of atmospheric CO_2. In the aquatic environment, however, C may become temporarily limiting, especially during summer stratification, when the rate of autotrophic CO_2 fixation exceeds the combined fluxes of heterotrophic CO_2 production and the transport of atmospheric CO_2 into the water.

The *nitrogen cycle* is very dynamic and is characterised by many transformations, in which (micro)biological processes play an important role. Nitrogen primarily enters the biogenic cycles by microbial nitrogen fixation of atmospheric N_2 (Vitousek et al. 1997; Galloway 1998; Pastor and Binkley 1998). Nitrogen compounds are not absorbed by soil, clay or ferric hydroxide, etc. A large fraction of the N, therefore, dissolves in either inorganic (ammonia, nitrate, nitrite) or organic (amino acids, proteins, nucleotides) forms and easily leaches to surface waters (Overbeck 1989). Part of the N is lost to the atmosphere due to denitrification. In contrast to terrestrial ecosystems, which are often characterised by N limitation, most freshwater systems have a sufficient supply of N from terrestrial run off. Within aquatic systems, the N cycle is primarily controlled by microbial processes (nitrification, denitrification, ammonification), which are strongly dependent on the redox status of the system (Overbeck 1989; Stumm and Morgan 1996).

The *phosphorus cycle* is mainly driven by physico-chemical processes. P readily absorbs to soil and, thus, most P is particle bound and only a fraction is directly available to biota (Sharpley and Rekolainen 1997). The most important natural route from terrestrial to aquatic ecosystems is terrestrial run off and erosion of particulate P (Sharpley et al. 1995). The bulk of the P in water is in particulate form and much P is bound to the sediment. The bioavailable fraction, the soluble orthophosphate (SRP), is very small and has an extremely high turnover rate (minutes). The P cycle in lakes is heavily dependant on the redox conditions of the system. Under anoxic conditions, the soluble P fraction drastically increases and substantial amounts of P may be released from the sediment. Under oxygenated conditions, soluble inorganic P is readily bound by adsorption to ferric hydroxide and $CaCO_3$ and by precipitation as ferric phosphate. The oxygenated surface layer of the sediment acts as an efficient P trap (Overbeck 1989; Grobbelaar and House 1995). Under certain conditions, P bound to sediment may be utilised by phytoplankton (Golterman 1977; Grobbelaar 1983; Grobbelaar and House 1995). However, this is negligible in clear shallow water due to the presence of a layer of benthic diatoms, which stimulate an oxygenated sediment surface layer and arrest nutrient fluxes from deeper sediment layers. The remaining P fraction is used by benthic diatoms for growth (Van Luijn et al. 1995). Additionally, the layer of benthic diatoms stabilises the sediment surface, thereby reducing the amount of resuspended sediment particles. Bioturbation by sediment-dwelling organisms may increase P-fluxes from the sediment, but this effect is counteracted by the increased oxygenation of the surface layer of the sediment (Andersson et al. 1988). Assimilation in the biomass (e.g., submersed and shoreline vegetation, aquatic food chain) is an important factor in the P-cycle.

Hydrological Aspects

Resuspension of inorganic sediment particles is mainly caused by wave action and, occasionally, by foraging fish. These resuspended particles can act as an internal source of phosphate (Ogilvie and Mitchell 1998). The dynamics of nutrients is also determined by the depth and stratification of lakes. In deep lakes, thermal stratification occurs during winter and summer. Nutrients present in the epilimnion (upper water layer) will be transported to the hypolimnion (bottom wa-