3 Nutrients and eutrophication in rivers

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3.1 Introduction

The impacts of inorganic nutrient enrichment on river ecosystems are seen in many rivers and have been intensively studied. For the implementation of the WFD it is equally important to be able to also describe quantitatively the ecological consequences of reduced nutrient concentrations in the different types of rivers to be able to decide on nutrient reductions necessary to meet WFD quality objectives. Enrichment in inorganic phosphorus and nitrogen concentrations in rivers can cause the following impacts, as summarised in Nijboer and Verdonschot (2004):

- Increase in primary production rate and mass growth of algae and plants. Increase in nutrients can potentially increase the growth of algae and macrophytes if these nutrients were initially limiting and if the other in-stream conditions allow more growth (e.g. light).
- Change in communities. A change in nutrients can lead to a shift from nutrient-sensitive slow-growing species to nutrient-tolerant fast-growing species. This may result in competition for light between epiphytes, macrophytes and phytoplankton, modifying the balance of these algae and plants. This may have impacts on macroinvertebrates and fish through changes to habitats and the food web.
- Reduction in oxygen. High nutrient levels favour a high biomass of algae and plants. Consequently, larger amounts of biomass are decomposed under oxygen consumption at a later stage. A high autotrophic production of oxygen during the day is usually followed by a high rate of respiration and often low oxygen concentration during the night. These processes can lead to oxygen depletion in the water or in the sediment.

Nitrogen and phosphorus are naturally present in streams through transport of terrestrial organic matter, leaching and runoff of terrestrial inorganic nutrients and deposition from the atmosphere. Mainstone and Parr (2002) estimated that the background export rates of total phosphorus in four UK rivers would be between 0.1 and 0.2 kg TP $ha^{-1}year^{1}$, which means a total phosphorus concentration level of 5-25 $\mu g\ l^{-1}$. All anthropogenic activities that release organic matter and inorganic nutrients in the environment may cause in-stream enrichment in inorganic nutrients. The predominant sources are population, industry, agriculture, transport and power plants.

In western countries the wastewater contribution from each person amounts to 1-1.5 kg P $capita^{-1}.year^{1}$ and 2.2 to 5.5 kg N $capita^{-1}.year^{1}$. The main sources are excretion from the human body and use of detergents (European Environment Agency, 1999). The wastewater nutrient loadings to rivers depend mainly on the population density in the river catchment and the level of nutrient removal at wastewater treatment plants. Depending on the treatment level, up to 95% of the phosphorus can be removed from wastewaters (Brett *et al.*, 1997).

Industries, primarily the food and drink industry, the industries producing phosphorus-based fertilisers, cleaning materials, pesticides and/or metal finishing (Morse *et al.*, 1993), can discharge significant amounts of phosphorus to surface waters. Emissions from industries are driven by the technology used, the level of wastewater treatment demanded (EC Urban Waste Water Treatment) and regulations on discharges (discharge consents) and emissions (UNECE Gothenburg Protocol to the Convention on the Long Range Transboundary Emissions, EU National Emission Ceiling Directive and EC Integrated Pollution, Prevention and Control Directive).

Agriculture releases nutrients to the environment through different pathways: emissions of ammonia to the air and leaching and runoff of nitrogen and phosphorus that reach the river. The export rate of phosphorus and nitrogen are affected by the amount of fertiliser applied, cultivation practice, climate, geology, topography, soils hydrology, land management and the spatial location of nutrient sources in the catchment (Johnes *et al.*, 1996). The transport and power production sectors combust fossil fuels. By-products of these combustions are nitrogen oxides. Once deposited, they can be transported to the rivers. 60% of the nitrogen oxides in 1995 came from transport, 21% from the energy sector and 14% from other industries (European Environment Agency, 1999).

Relative importance of the different sources

Nitrogen load. Nitrogen within rivers is derived predominantly from agricultural sources (e.g. Jarvie *et al.*, 1998). In Europe, agricultural sources can represent between 46 and 87% of the total nitrogen load. Agricultural N losses depend on a range of factors, including the area of land in agricultural use, intensity of agricultural production and fertilizer application rates. Point sources (principally sewage discharges) can also play an important role accounting for up to 35 to 43% of N loads in some catchments (e.g. Boorman, 2003). Nitrogen from the atmosphere can also represent an important source of N, for example up to 35% for a Swedish catchment between 1982 and 1987 (European Environment Agency, 1999) whereas the atmospheric deposition of N in the Humber catchment (UK) reached values comparable with river output (Smith *et al.*, 1997).

Phosphorus load. Source apportionment of phosphorus can highly vary between catchments. It is usually proportional to population density. Phosphorus present naturally would, in many catchments, represent only 3 to 15% of the current levels (Morse *et al.*, 1993; Farmer, 2004). In densely populated catchments, 50 to 76% of the phosphorus load was found to come from point sources, mainly from sewage discharges in highly industrial catchments, and 20 to 40% from agricultural diffuse sources (European Environment Agency, 1999). The percentage contribution of non-point sources of phosphorus could vary between 2 and 60% for European rivers (Farmer, 2004).

Quantification of diffuse losses of nitrogen N and phosphorus P. Several quantification tools exist to estimate the diffuse nutrient losses to surface freshwater systems. An EU research project EuroHARP[1] is evaluating nine different methodologies for quantifying diffuse losses of nitrogen and phosphorus (AMINO, REALTA, N-LES, MONERIS, TRK, SWAT, EVENFLOW, Nopolu and Source apportionment methodology) on 17 catchments.

Nutrient concentrations in rivers

The level and fluctuation of nutrient concentrations in rivers is determined mainly by natural and anthropogenic inputs. Catchment behaviour contrasts between urban/industrial catchments and

[1] EuroHARP is a research project under the 5th Framework program. It started on the 1st of January 2002 and is running for 4 years. For information see www.euroharp.org

agricultural catchments. In the urban/industrial catchments, dilution effects can mean that nutrients peak at low flows and are potentially nitrogen limited. In agricultural catchments, runoff can lead to a nutrient peak at high flow with potentially phosphorus limitation (Jarvie *et al.*, 1998; Mainstone and Parr, 2002). Variation of the ratio between nitrogen and phosphorus in time also leads to a variation in potential nutrient limitation. However the concentration levels are also influenced by plant assimilation and sedimentation (Young *et al.,* 1999).

Phosphorus can be adsorbed or released from sediments. Interactions between river water and sediments are particularly important for phosphorus (House, 2003). Soluble reactive phosphorus can be adsorbed or released from sediments as a result of a combination of physico-chemical and biochemical processes as well as hydrodynamic effects and bioturbation (House and Denison, 1998; House and Warwick, 1999; House, 2003; Jarvie *et al.*, 2005). Particulate phosphorus is also subject to deposition on the river bed and resuspension according to hydrodynamic factors and the influence of biofilms in stabilizing the sediment surface (Battin and Sengshmitt, 1999). The biogeochemical cycling of phosphorus in rivers is therefore complex and influenced by a wide range of factors such as the composition and reactivity of the sediments, sediment redox conditions, growth of biofilms, hydrology and antecedent conditions (House, 2003). Nitrogen transformations include e.g. nitrification of ammonium into nitrate and denitrification of nitrate into atmospheric nitrogen also depending on the redox potential (Hamilton *et al.*, 2001; Peterson *et al.*, 2001). Decomposition of organic matter releases plant-available inorganic nutrients to the water column.

A high population of macrophytes, phytobenthos and phytoplankton can strip soluble reactive phosphorus (Mainstone and Parr, 2002) and ammonium (Balbi, 2000) out of the water column, even at high nutrient levels. Effects of nutrients on biota also vary in time: greater impacts by nutrients during the active season (light and temperature driven) and time to uptake the nutrients (water residence times, flow driven). Increase in biomass leads to an increase in organic matter. This organic matter can either decompose *in situ* (very slow flowing rivers) or be carried away, increasing the nutrient levels of the river either locally or downstream when organic matter is mineralised.

Models of water quality

A wide range of models has been developed. They all include a hydrological compartment and can represent the in-stream processes and/or the land processes with varying degrees of complexity. Benchmarking Models for the Water Framework Directive" (BMW) created a database of water quality models that include their main characteristics and their domain of applicability. The BMW database[2] includes models of in-stream processes for the river domain, for example QUESTOR, QUAL-2E, RIVQM, SOBEK and HERMES, and models of terrestrial processes for nutrients, for the diffuse domain, for example CREAMS, CASCADE, MIKE SHE, and SWAT.

[2] This database is available on Internet, after a free registration to the BMW website: http://www.rbm-toolbox.net/toolbox3/index.php.

Effects of reduction in nutrient inputs

It is expected that a reduction in nutrient inputs to a river leads to a reduction in nutrient concentrations. However, few studies have explored the effect of reduction in nutrient inputs. There is a potential for a release of phosphorus stored in soft river sediments and thus delaying the reduction in river concentration (House, 2003). The catchment as a whole can also buffer the effect of a reduction in nutrient use: in Denmark and Estonia, no evidence of reduction in river nutrient concentration was found after 15 to 20 years of reduction in fertilizer use (Stalnacke *et al.*, 2003; Stalnacke *et al.*, 2004; Kronvang *et al.*, 2005).

As for impacts on the biota, Jarvie *et al.* (2002) indicate that implementation of phosphorus-stripping before the discharge of wastewater did not lead to an improvement of macrophytes and phytobenthos. Young *et al.* (1999) examine this situation resulting from an insufficient reduction of in-stream nutrients that can maintain a eutrophic status. On the other hand, Sosiak (2002) found a decrease in phytobenthic biomass after reduction of phosphorus in a wastewater discharge (even if not directly downstream) and in macrophyte biomass following reduction in nitrogen. These differing conclusions indicate the lack of knowledge of the effect of nutrient stripping and the recovery that can be expected from the biota.

3.2 Macrophytes

Relationships between nutrients and macrophytes

Macrophytes are macroscopic aquatic plants. They are important primary producers in rivers. Their growth is partly conditioned by the uptake of inorganic nutrients by their roots and their shoots. Inorganic nutrients can be taken up either from the water column or the sediments, or both in varying proportion (Clarke and Wharton, 2001). In nutrient-limited rivers enrichment in inorganic nutrients may lead to an increase in biomass and species richness up to a point when competitive exclusion occurs (Mesters, 1995).

However, these effects of nutrients are contested in several studies. Demars and Harper (1998) underlined the difficulty of identifying the specific effect of nutrients because of the synergistic effect of a number of physical factors. Holmes *et al.* (1999) indicated that species diversity in itself is not an indicator of trophic status of rivers as a wide range of diversity can occur at any phosphorus concentration and low to intermediate diversity occur at any phosphorus concentration as well. The Environment Agency (2002) also pointed out that macrophytes were not an effective indicator of the level of nutrients in lowland rivers and Madsen and Cedergreen (2002) did not find an effect of enrichment on the growth rate of two macrophyte species, suggesting that nutrients were not limiting for their growth.

Other studies confirmed the prominence of physical characteristics in determining macrophyte biomass and species composition (Demars and Harper, 1998; Sand-Jensen, 1998; Schorer *et al.,* 2000; Baattrup-Pedersen *et al.*, 2003; Barendregt and Bio, 2003). They underlined the effects of flow and its variation, light (water depth, shading by riparian vegetation, suspended solids, phytoplankton and epiphytes), substrata and weed cutting.

Existing methods to investigate the relationships

Macrophytes communities can be assessed by their biomass and their composition. Direct measures of macrophyte biomass are not widely used. Macrophyte biomass is more often assessed using measures of coverage or abundance. Macrophytes species composition can be assessed by recording the species/taxa present at a site and possibly the species abundance. Several indices for species richness and evenness have been developed. Average Score Per Taxon (H.M.S.O., 1987) based on number of taxa, surface cover class and biomass, distribution factor based on species abundance per section, Shannon-Wiener functions (Shannon and Weawer, 1963), Sorensen index and the Simpson index (Simpson, 1952). The relationships between inorganic nutrients and macrophyte biomass and composition can potentially be studied through 1) laboratory bioassays, 2) in-situ enrichment, and 3) analysis of monitoring data.

Monitoring data have been far more widely used than laboratory or in-situ experiments. Analysis of the data can vary from simple relations to more complex analyses. Regression analysis allows the assessment of linkages between species composition (as indices), abundance or biomass with nutrients or other factors (Flynn *et al.,* 2002; Carr *et al.*, 2003). Correlations are used to analyse interactions with a wider range of variables, including nutrients, as in Mesters (1995), Szoszkiewicz *et al.* (2002) and Thiebaut *et al.* (2002). Examples are shown in Table 3.1.

Macrophyte models and indices

Several dynamic models have been developed; for example Wright *et al.* (1986) presented a model of macrophyte biomass in shallow waters as a function of light and nutrient limitation. A model of macrophyte and epiphyte biomass has been developed for the River Kennet (Southern England), including SRP from sediment pore water (Wade *et al.,* 2002a, b).

The equation of macrophyte biomass in the Kennet model (Wade *et al.,* 2002b) is:

$$d(x_7)/dt = \frac{c_{10}.\theta_M^{(T-20)}.x_7.x_{12}.R.c_{12}}{(c_{11}+x_{12}).(c_{12}+x_7)} - c_{14}.x_7.x_8.x_1$$

where x_7 macrophyte biomass at time t (g C m^{-2}); c_{10} macrophyte growth rate (day^{-1}); $\theta_M^{(T-20)}$ macrophyte temperature dependency; x_{12} SRP in pore water at time t (mg P L^{-1}); R solar radiation at time t (normalised to 0-1 from W.m^{-2}); c_{12} self shading (g C m^{-2}); c_{11} half saturation of P for macrophyte growth (mg P L^{-1}); c_{14} macrophyte death rate (s g C^{-1} day^{-1}); x_8 epiphyte biomass at time t (g C m^{-2}); x_1 flow out of reach at time t (m^3.s^{-1});

An example of a macrophyte index used for classification, the Mean Trophic Rank index (Holmes *et al.,* 1999) is presented in Box 3.1.

The MTR index has been developed for England and Wales to implement the EC Urban Waste Water Directive. It is used to assess the impact of point sources on rivers (Kelly, 1998). It is based on the combination of species at a site and, for each species, its indicator value and its abundance. 128 species have been identified for this index. They can be split into 4 broad groups and 10 River Community Types. These River Community Types are physically described and the mean MTR

Table 3.1 Summary of some of the relations found between macrophytes and inorganic nutrients.

Biological element	Water quality element	Relation	Sites	Nutrient range	References
Total % macrophyte cover	SRP	Regression: $y = -120.37 + 77.1 \cdot \log SRP$ ($R^2 = 0.45$. $p = 0.002$)	River Kennet, UK n = 20	Not indicated	(Flynn *et al.*, 2002)
MTR	Soluble NO_3 Soluble PO_4	Correlation: –0.43 ($p < 0.042$) –0.58 ($p < 0.003$)	River Welland, UK	Not indicated	(Demars and Harper, 1998)
Shift in species	PO_4^{3-}	1st ordination axis correlated among other things with: PO_4^{3-} ($R = 0.74$, $p = 0.0015$, $n = 17$)	28 streams, The Netherlands	Not indicated	(Mesters, 1995)
Species richness	NH_4-N PO_4-P	Coefficient of rank Spearman correlation 0.457 ($p < 0.05$) 0.611 ($p < 0.001$)	30 sites of Vosges/ Alsace floodplain, France	Mean NH_4-N = 31–422 µg.l^{-1} PO_4-P = 6-413 µg/l^{-1}	(Thiebaut *et al.*, 2002)
Abundance	NH_4-N PO_4-P	Coefficient of rank Spearman correlation 0.600 ($p < 0.001$) 0.707 ($p < 0.001$)	30 sites of Vosges/ Alsace floodplain, France	Mean NH_4-N = 31–422 µg.l^{-1} PO_4-P = 6-413 µg/l^{-1}	(Thiebaut *et al.*, 2002)
Species eveness	NH_4-N PO_4-P	Coefficient of rank Spearman correlation –0.238 (non significant) –0.780 ($p < 0.0001$)	30 sites of Vosges/ Alsace floodplain, France	Mean NH_4-N = 31–422 µg.l^{-1} PO_4-P = 6-413 µg.l^{-1}	(Thiebaut *et al.*, 2002)
MTR	NO_3-N NH_4-N N total SRP P total	Pearson linear correlation coefficient 0.156 –0.299 ($p < 0.05$) –0.368 ($p < 0.05$) –0.260 –0.294 ($p < 0.05$)	48 sites (19 rivers) in lowland Poland	Not indicated	(Szoszkiewicz *et al.*, 2002)
6 phyto-sociological groups	NH_4-N NO_3-N PO_4-P	–0.68 + 0.81 (contextual) –0.92	29 sites	NH_4-N = 0–90 µg.l^{-1} PO_4-P = 1–42 µg.l^{-1} NO_3-N = 0.2-7.5 mg.l^{-1}	(Carbiener *et al.*, 1990)

value for each of these groups can be used as a reference for physically similar sites. The MTR has been adapted to several countries, including Poland (Szoszkiewicz *et al.,* 2002).

Other macrophytes indices based on nutrients have been developed in Europe, such as the GIS Macrophytes in France (Haury *et al.*, 1996), the Trophic Makrophyten Index (Schneider and Melzer, 2003) and the Reference Index (Meilinger *et al.*, 2005) in Germany, and the Canonical-Based Assessment System for Northern Ireland (Dodkins *et al.*, 2005).

Box 3.1 Brief description of Mean Trophic Rank (derived from Holmes *et al.*, 1999).

$$MTR = \frac{\sum CVS}{\sum SCV} .10$$

where CVS = SCV x STR
and where SCV = Species Cover Value (cover class per taxon, on a scale 1-9)
STR = Species Trophic Rank (score assigned to species on a scale 1-10: the higher the score, the lower the tolerance to nutrient enrichment)

MTR score	Interpretation
MTR > 65	Unlikely to be eutrophic*
25 - 65	Site likely to be either eutrophic or at risk of becoming eutrophic**
MTR < 25	Site badly damaged by eutrophication, organic pollution, toxicity or is physically damaged

*If the score is less than might be expected for a site of this physical type, the site may be at risk of becoming eutrophic.
**If the score is less than might be expected for a site of this physical type, the site is eutrophic or at risk of becoming eutrophic.

3.3 Phytoplankton

Phytoplankton are microscopic plants suspended in the water column. They can include both 'true' phytoplankton species and phytobenthic species detached by flow, especially after spates (Descy, 1993; Reynolds, 2003). The generation time of phytoplankton is typically several days. This makes them highly reactive to changes in the condition of water and light. It also underlines that a significant phytoplankton biomass cannot be reached unless the residence time of water in the river is greater than the generation time (large rivers, slow-moving rivers, canals and on-river reservoirs).

Nutrient enrichment can lead to an increase in biomass and/or a change in species composition (e.g. Vanni and Findlay, 1990). Algal blooms have been found to occur principally in spring in rivers when increasing light and temperature and decreasing flow are favourable to phytoplankton growth (Marker and Collett, 1997; Balbi, 2000). Species composition is also affected by the way nutrient enrichment occurs: nutrient pulses allow the persistence of species preferring low and high nutrient concentrations, thus leading to increased species richness (Hecky and Kilham, 1988).

The effect of nutrient enrichment will, however, be modulated by the initial nutrient concentrations in the water column. Thus, Mallin *et al.* (2004) showed no impact of phosphorus enrichment on phytoplankton biomass (0-5 mg P/l, stream concentration: 3-143 μg P/l^{-1}) for 2 streams. In this study nitrogen enrichment over 200 μg N/l in nitrate (stream concentration: 5-930 μg N/l^{-1}) led to an increase in phytoplankton, whatever the form of nitrogen. Young *et al.* (1999) also showed that there was little correlation (r = –0.29) between chlorophyll a and phosphorus concentrations (0.1-6.7 mg PO_4-P /l^{-1}) for 3 lowland rivers of the Thames catchment. These studies demonstrate that the relationships between nutrients and phytoplankton in rivers are not simple.

Phytoplankton-nutrient relationships

Commonly-used models are mainly simple regression models (Table 3.2). Hakanson *et al.* (2003) concluded that it is unlikely that a regression model for mean monthly chlorophyll concentrations in rivers could yield R^2 higher than about 0.6 (for all n). Other more complex and dynamic models exist and are described in Recknagel *et al.* (1997).

The dynamic models describe the phytoplankton development and need very substantial data input e.g. algal growth and respiration rates, light attenuation, nutrient concentrations and growth rates as a function of the nutrient concentrations. Some of these models are described below. SWAT2000 (Neitsch *et al.*, 2002): models algal biomass through chlorophyll a, depending on light, nitrogen and phosphorus. AQUATOX (Park and Clough, 2004): models phytoplankton, periphyton, macrophytes, invertebrates and fish depending on nutrients, organic matter, organic toxicant, oxygen, suspended sediments, sediments, temperature and discharges. It is applicable to streams, lakes and reservoirs.

RIVERSTRAHLER (Billen *et al.,* 1994): combines a hydrological component based on the stream order and the AQUAPHY model. The AQUAPHY model models the chlorophyll a concentration and biomass evolution in each stream order depending on light, temperature, flow, ammonium, nitrate and phosphate concentrations. In Garnier *et al.* (2002), this model is further refined to distinguish between diatoms and chlorophycae, to include predation by zooplankton and mussels and to model the interactions of nutrients across the sediment-water interface.

Table 3.2 Summary of some relations found between phytoplankton and inorganic nutrients.

Biological element	Water quality element	Relation	Sites	Nutrient range	References
Chlorophyll-*a*, January to spring chlorophyll maximum	Silicate NH_3 NO_3-N TN SRP N/P ratio	Regression 0.71 ($p < 0.05$) 0.43 ($p < 0.05$) 0.25 ($p < 0.05$) 0.20 ($p < 0.05$) 0.09 ($p < 0.05$) 0.15 ($p < 0.05$)	River Nene (small lowland river), UK 11 sites, up to over 22 years	NH_4–N = 0-5.3 mg.L^{-1} NO_3–N = 0.4-20 mg.L^{-1} NO_2–N = 0-0.4 mg.L^{-1} SRP = 0.1-4.4 mg.L^{-1}	Balbi, 2000
Chlorophyll-*a*, summer period	TP	Quadratic model $\log(Chl) = -1.65 + 1.99 (\log TP) - 0.28(\log TP)^2$ ($r^2 = 0.67$) Chl in mg.m^{-3} and TP in mg.m^{-3}	Temperate streams (mainly North America, but also Europe, Korea, Iraq and Australia) N = 292	TP = 5 – 1030 mg P.m^{-3}	Van Nieuwenhuyse and Jones, 1996
Chlorophyll-*a*	NO_3 SiO_2	Pearson correlation coefficient: 0.91 ($p = 0.003$) 0.87 ($p = 0.006$)	Canojoharie Creek, Hudson River Basin, US	NO_3–N = 0.05-3.3 mg.L^{-1} SiO_2 = 1–5 mg.L^{-1}	Wall *et al.,* 1998

3.4 Phytobenthos

Whereas phytoplankton is considered to be a good indicator of eutrophication in rivers with a residence time longer than the generation time of phytoplankton, phytobenthos is a better one for smaller rivers (Piirso *et al.*, 1997). Phytobenthos is a highly diversified biota (Stevenson, 1996) with different types of organisms (diatoms, filamentous algae, blue-green etc), growing on different substrates (rocks, soft riverbed, macrophytes etc) and developing different forms (filamentous, thin or thick mat etc). Most of the studies undertaken have focused on diatoms. Phytobenthos is present in every type of river where light reaches the riverbed or surface of macrophytes.

As for all autotrophs, phytobenthos development depends on the availability of inorganic nutrients. An increase in inorganic nutrients (P and N) potentially leads to an increase in phytobenthos biomass. Simultaneously, a shift in species composition occurs from slow-growing, oligotrophic species at very low nutrient concentrations to fast-growing species with competitive advantages at high nutrient concentrations (Biggs *et al.*, 1998).

The response of phytobenthos to nutrient enrichment depends on whether the nutrient was limiting and the initial nutrient level. Indeed, Perrin and Richardson (1997) obtained little phytobenthic response to P addition, a moderate one when adding N, and an even higher one when adding both N and P (P became limiting once N was added). Stanley *et al.* (1990) demonstrated a rapid response of phytobenthic biomass when the P level was initially low (<0.010 mg.l^{-1}) and only a moderate one when ambient P was higher (0.015-0.025 mg.l^{-1}).

Flow and its variation can also play an important role in determining the response of phytobenthos to nutrient concentrations. The discharge stability over periods of less than a year may govern the average phytobenthic biomass, and nutrients will control the overall biomass only over prolonged periods of stability with moderate to low flow (Biggs, 1996). So, a small increase in dissolved nutrients may greatly increase the frequency of high biomass events if there are infrequent floods and accrual periods over 100 days (Biggs, 2000).

In term of species richness, nitrogen and phosphorus enrichment have been found to increase the species richness in oligotrophic streams up to a point where competitive exclusion occurs and species richness decreases (Marcus, 1980; Snyder *et al.*, 2002). As for phytoplankton (Hecky and Kiham, 1988), the way nutrients are delivered affects how the community assemblage will respond. Nutrient pulses can lead to higher richness, with a mix of species of different nutrient preferences, as found in Finnish rivers (Soininen, 2002).

Kjeldsen (1994) underlined a different response of phytobenthos peak biomass to phosphorus enrichment depending on the riverbed substrate: on fine-grained sediments phosphorus was the main driver of the biomass peak, whereas on stony substrate P concentrations are only linked to the maximum potential biomass and the actual peak biomass is determined by a range of other factors.

Estimations of biomass levels that hampers the aesthetic quality of rivers have been developed. In Welch *et al.* (1988) 100 mg chlorophyll-*a* per square metre is considered to be the threshold over which development of phytobenthos, often as filamentous algae, is perceived unfavourably. Dodds *et al.* (1998) reviewed several studies of such thresholds. They range from 50 to 200 mg chlorophyll-*a* per square metre, and mainly over 100 mg chlorophyll-*a* per square metre.

Kiffney and Bull (2000) showed that light was the single best indicator explaining the variation in peak chlorophyll-*a* biomass. Light varies seasonally and the quantity that reaches the riverbed depends on the water depth and a thick carpet of diatoms covering the sediment is often seen during spring where irradiance is high (Sand-Jensen, 1983). Dense growth of phytobenthos can cause a high diurnal variations in oxygen concentrations (higher during the day, lower during the night) unfavourable for other organisms like macroinvertebrates and fish (ten Cate *et al.*, 1991).

Anderson *et al.* (1999) showed that grazing controlled biomass more than enrichment in soluble reactive phosphate whereas Walton *et al.* (1995) pointed out that grazing lead to a shift in species whatever the enrichment was and grazing is the third dimension in the habitat matrix developed by Biggs *et al.* (1998) with an effect similar to spates.

Methods to investigate the phytobenthos-nutrient relationships

Chlorophyll-*a* per square metre is the most commonly used parameter describing phytobenthic biomass as it is easy to use and not as labour-intensive as species composition surveys. The dry mass, ash-free dry mass, cell density and biovolume can also be used to estimate the biomass of phytobenthic communities (Stevenson, 1996). Species composition is assessed by recording the number of species/taxa present in the community and their relative abundance. Different indices can be calculated from these measures to reflect the species richness, the species evenness and the similarity of species among communities (Zelinka and Marvan, 1961).

The relationships between inorganic nutrients and phytobenthos can be studied via three different methods: laboratory bioassays (Horner *et al.*, 1990; Humphrey and Stevenson, 1992), in-situ enrichment experiments and empirical methods from monitoring data. Whole river fertilisation has mainly been used in several artic streams to study the long-term effect of fertilisation on phytobenthos and other biota (Harvey *et al.*, 1998, Slavik *et al.*, 2004) and to other streams (Biggs *et al.*, 2000; Bernhardt and Likens, 2004). Monitoring of both phytobenthos and water quality and then using empirical methods to form relationships is the most commonly used method to analyse the link between phytobenthos biomass and composition and environmental conditions (including nutrient levels). Once the phytobenthos and the water quality have been measured, similar techniques as for phytoplankton and macrophytes can be used, from the simplest to the most complex.

Simple regressions have been used in Biggs (2000) to link the mean monthly chlorophyll a and nutrients and days of accrual (reflects the effect of flow variation). Multiple regressions, sometimes with stepwise selection, have been used as well in Biggs (2000) and Kiffney and Bull (2000). Regressions are usually used to link biomass with nutrients.

Relationships, models and indices

Regression models have been developed between phytobenthic biomass and nutrient concentration, for example in Biggs (2000) and Winter and Duthie (2000) and examples given in Table 3.3.

The peak biomass of benthic algae in 10 Danish lowland streams followed the equation: $Y=929.3.x/(49.2+x)$ ($r^2=0.61$), with Y the maximum biomass (mg chlorophyll.m^{-2}) and x the dissolved inorganic phosphorus ($\mu g\ P\ l^{-1}$, 1 - 273 $\mu g\ P\ l^{-1}$; Kjeldsen, 1994).

Table 3.3 Some examples of relations found between phytbenthos and inorganic nutrients.

Biological element	Water quality element	Relation	Sites	Nutrient range	References
Chlorophyll-*a* (in $mg.m^{-2}$)	TP, in $\mu g.l^{-1}$ TN, in $\mu g.l^{-1}$	log Chl a = 0.905 log TP + 0.490 ($r^2 = 0.56$, $p < 0.001$) log Chl a = 0.984 log TN – 0.935 ($r^2 = 0.50$, $p < 0.001$)	33 sites, 13 rivers of Ontario and Quebec (Canada)	TP = 6 - 130 $\mu g.l^{-1}$ TN = 179 - 2873 $\mu g.l^{-1}$	(Chetelat *et al.*, 1999)
Mean monthly benthic algal biomass In mg/m^2 chlorophyll-*a*	SIN SRP	Log_{10} (chla) = 0.109 + 0.483.log_{10} SIN ($r^2 = 0.122$, $p = 0.057$) Log_{10} (chla) = 0.468 + 0.697 log_{10} SRP ($r^2 = 0.226$, $p = 0.008$)	30 sites, 25 streams, New Zealand	SIN = 6.2 – 232 $mg.m^{-3}$ SRP = 1.3 – 31.6 $mg.m^{-3}$	(Biggs, 2000)
Max monthly benthic algal biomass In mg/m^2 chlorophyll-*a*	SIN SRP	Log_{10} (chla) = 0.711 + 0.688.log_{10} SIN ($r^2 = 0.325$, $p = 0.001$) Log_{10} (chla) = 1.400 + 0.797 log_{10} SRP ($r^2 = 0.295$, $p = 0.002$)	30 sites, 25 streams, New Zealand	SIN = 6.2 – 232 $mg.m^{-3}$ SRP = 1.3 – 31.6 $mg.m^{-3}$	(Biggs, 2000)
Diatom taxa richness	TN	Spearman correlation: 0.28 ($p < 0.01$, diatoms at genus level) 0.42 ($p < 0.001$, diatoms at species level)	199 streams, Mid-Appalachian streams, USA	Not indicated	(Hill *et al.*, 2001)
% dominance (diatoms)	TP	Spearman correlation: –0.41 ($p < 0.01$, diatoms at genus level) –0.45 ($p < 0.001$, diatoms at species level)	199 streams, Mid-Appalachian streams, USA	Not indicated	(Hill *et al.*, 2001)
TDI	NH_4-N	Pearson correlation coefficient 0.430 ($p < 0.001$)	9 sites, for Vistula river, Poland	NH_4-N = 0.2 - 3.0 $mg.l^{-1}$ NO_3-N = 0.2 – 3.0 $mg.l^{-1}$ PO_4-P = 0.05-0.55 $mg.l^{-1}$	(Kwandrans, 2002)
Diatom richness	NH_4-N TP	Spearman correlation coefficient 0.55 0.64	11 rivers in Idaho, USA	NO_3-N + NO_2-N = 0.002-1.61 $mg.l^{-1}$ NH_4-N = 0.005-0.037 $mg.l^{-1}$	(Snyder *et al.*, 2002)

Indices and classification

A large number of phytobenthos indices (mainly diatoms) have been proposed, e.g.:

- Specific Pollution Sensitivity Index (Cemagref, 1982)
- Generic Diatom Index (Coste and Ayphasshorho, 1991)
- Descy's Index (Descy, 1979)
- Sládeček's index (Sládeček, 1986)
- Leclerq & Maquet's index (Leclercq and Maquet, 1987)
- Trophic Diatom Index (Kelly and Whitton, 1998; Holmes *et al.*, 1999; Kelly *et al.,* 2001)
- EPI-D (Dell'Uomo, 1997; Dell'Uomo *et al.,* 1999).

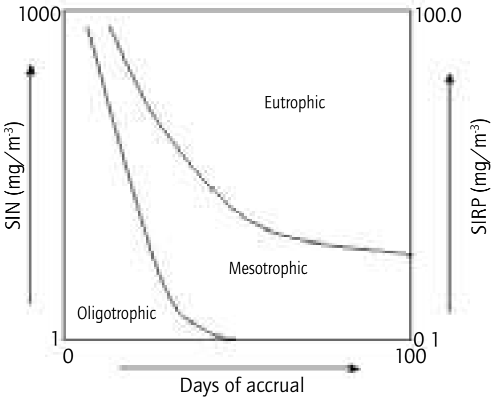

Figure 3.1 Simplified schematic classification as developed by Biggs (2000). The boundary between oligotrophic and mesotrophic is set at 60 mg m^{-2} chlorophyll-*a* and the boundary between mesotrophic and eutrophic is set at 200 mg m^{-2} chlorophyll-*a*.

A classification of river sites has been developed for the UK based on the Trophic Diatom Index (TDI) as an indicator of eutrophication (Kelly, 1998). In Italy, the EPI-D is considered suitable to diagnose inorganic nutrients, organic matter and minerals impacts on phytobenthos (Dell'Uomo *et al.,* 1999). Lastly, Biggs (2000) developed a classification depending on soluble reactive phosphorus and soluble inorganic nitrogen, and the number of days of accrual in gravel/cobble-bed streams, the boundaries being determined by 2 nuisance levels of chlorophyll a: 60 and 200 mg chlorophyll a per square metre (Figure 3.1).

3.5 Summary

An increase in plant available phosphorus and nitrogen concentrations in rivers potentially leads to an overall increase in the biomass of phytobenthos, phytoplankton and macrophytes and to changes in species composition, favouring nutrient tolerant species. Large-scale phytoplankton growth is generally limited to slow-flowing rivers and macrophytes and phytobenthos tend to occur in rivers where light penetration reaches the river bed.

Nutrient enrichment often leads to an increase in biomass and a change in species from nutrient-sensitive toward more nutrient-tolerant species. In term of species richness, enrichment of oligotrophic streams can lead to an increase in species richness until the enrichment reaches a threshold after which competitive exclusion of nutrient-sensitive species decreases the overall species richness. The effects of nutrient enrichment depend on the initial nutrient level. In oligotrophic streams, even a slight enrichment can lead to a high increase in biomass and alter the species assemblage, whereas it may not cause any change at higher nutrient levels. However, no normative thresholds for inorganic nutrients, applicable beyond the individual rivers of the studies, have been found in the literature.

Inorganic nutrients usually do not directly influence heterotrophic organisms such as invertebrates and fish, but it is important to consider also the indirect effects from the changes in the plant communities when assessing the overall impacts of nutrient in stream ecology. Further, the autotrophic communities are influenced by heterotrophic activities (grazing) and the quality of the river sediment, which must therefore be considered in the assessment of river eutrophication.

Numerous studies have been undertaken to examine the relationships between macrophytes and benthic diatoms and nutrient status in rivers. However, the relationships identified are not sufficiently detailed or targeted to meet the needs in the implementation of the WFD. For setting the criteria for ecological classification and the environmental objectives, and for planning the program of measures thereafter, it is necessary to know quantitative relationships between nutrient concentrations and biological quality elements (periphyton, macrophytes and phytoplankton) in the different types of rivers. On-going work in REBECCA and other projects such as DARES (http://craticula.ncl.ac.uk/Dares/index.htm) are exploring these relationships.

Major needs for the implementation of the WFD include clarification of the conditions where the models and relationships established are operational and valid. The conditions to be clarified are related to the type of river and sediment, the type of vegetation and the concentration level of the nutrients. Few studies have focussed on the recovery of a river after reduction in either phosphorus or nitrogen (or both) sources. This aspect should be considered fundamental in determining the measures needed to be implemented to achieve good chemical and ecological status.

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