Toxic Cyanobacteria in Water: A guide to their public health consequences, monitoring and management
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Chapter 8. PREVENTATIVE MEASURES

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Cyanobacterial bloom formation can be avoided by measures which address their growth requirements, i.e. plant nutrients and light. The basis for excessive growth of cyanobacteria and other phytoplankton organisms (planktonic algae) is enrichment of aquatic ecosystems with plant nutrients. This process is termed eutrophication. The key nutrient in many cases is phosphate. In some systems, not all of the phosphate available is actually used for phytoplankton growth because other resources limit the maximum possible biomass. These may be light intensity or availability of nitrogen. Furthermore, other biota can affect the growth of cyanobacteria and phytoplankton organisms: submerged aquatic plants may compete for nutrients, and grazing by zooplankton may reduce the stock of many phytoplankton organisms and (to a lesser extent) also of some cyanobacteria.

The key management action for abatement of Cyanobacterial blooms is to address the source of the problem by control and reduction of external nutrient loading to the water body, and thus of the concentrations within it. Measures addressing light availability directly (e.g. artificial mixing) or targeting the community structure of the biocoenosis (e.g. biomanipulation) have been successful chiefly in less eutrophic situations. For highly eutrophic water bodies under restoration by reduction of nutrient loading, such measures may accelerate and enhance success.

Control and reduction of nutrient loading usually focuses on phosphorus (for the reasons discussed in section 8.1), but measures addressing phosphorus may be designed to reduce nitrogen input simultaneously. Targets for nutrient concentrations can be achieved by following basic principles of good catchment management with respect to agriculture and sewage treatment. This chapter aims to assist decision making by giving information on:

- Target values for phosphorus concentrations likely to control cyanobacterial blooms in a given water body.

- The potential impact of hydrophysical and biological methods for control of specific Cyanobacterial ecostrategists (see section 2.3) in a given water body.

• The likelihood of success of poorly substantiated methods, sometimes propagated with remarkably effective marketing strategies.

A decision tree for application of different management approaches in order to control phytoplankton growth is given in Reynolds (1997). It leads from restriction of all sources of phosphorus enrichment to enhancing flushing or sediment removal, biomanipulation and artificial destratification. Reynolds (1997) points out that decision points for application of measures are still being quantified by current research.

8.1 Carrying capacity

The concept of the carrying capacity of the resources in a given ecosystem to sustain a population has proved very helpful in planning measures to control the size of that population; Applied to cyanobacteria, this means asking questions such as:

• How much biomass can be sustained on the basis of the amount of nitrogen available?
• How much biomass can be sustained on the basis of the amount of phosphorus available?
• How much biomass can be sustained with the amount of light that penetrates into the water?

At any one point in time, it is likely that one of these three resources will limit the possible amount of biomass at a lower biomass level than the others. However, the limiting resource may change seasonally, for example at higher latitudes it changes in relation to the angular height of the sun and day length, and in tropical climates it frequently changes in relation to turbidity changes caused by seasonality of the flow regime. During winter (even in clear water) or in turbid situations, light is usually the limiting factor, whereas the available nitrogen and phosphorus could have allowed a higher level of biomass. As light intensity increases in spring or as water becomes clearer, phytoplankton organisms begin to multiply and incorporate available nitrogen and phosphorus into their biomass, often up to the point where either of these resources is depleted in the environment and further growth is not possible. If nutrient concentrations are excessively high, phytoplankton may reach a density that causes such a high level of turbidity that light availability limits any further growth, and in these situations populations will be light- rather than nutrient-limited.

For planning and management, it is important to be able to estimate which of the key resources (light, nitrogen or phosphorus) is likely to control phytoplankton biomass in any given system. In other words, the questions are:

• Which resource determines the carrying capacity for phytoplankton?
• How high is the carrying capacity?

For a first step in answering this, it is not important to differentiate between cyanobacteria and other phytoplankton, because the maximum amount of phytoplankton possible can be equal to the maximum amount of cyanobacteria possible, once cyanobacteria have become dominant. One approach at estimating the carrying capacity in relation to nitrogen and phosphorus is to look at the relative amounts of these nutrients in phytoplankton biomass, known as Redfield Ratio by mass (Round, 1965). These ratios are:

C	H	O	N	P
42	8.5	57	7	1

Among these components, hydrogen and oxygen are never limiting in aquatic environments. Carbon is available as carbon dioxide (CO_2) and is consumed by photosynthesis. Consumption of dissolved CO_2 enhances diffusion of atmospheric CO_2 into the water, a process which takes time. Carbon limitation has been extensively investigated and has been summarised by Reynolds (1997). He concluded that CO_2 limitation can occasionally have an impact, particularly in soft-water lakes with low bicarbonate alkalinity, but that these situations are generally brief and do not substantially limit the maximal amount of biomass possible.

8.1.1 Nitrogen

Nitrogen may enter water bodies as leachate from soils, as run-off from animal feedlots, and from untreated or biologically-treated sewage, unless treatment includes nitrification and denitrification. Phytoplankton can take up inorganic dissolved nitrogen in the form of nitrate, nitrite and ammonia. In some arid continental regions, nitrogen is found to be the chief factor limiting phytoplankton growth (Reynolds, 1997). The relevance of nitrogen to limitation of cyanobacterial biomass is under debate, because a number of cyanobacterial taxa can compensate for its lack by fixing atmospheric nitrogen at rates of up to 175 kg $ha^{-1}a^{-1}$ (Rönicke, 1986). Thus, lack of dissolved inorganic nitrogen may actually support the dominance of such species as *Anabaena* and *Aphanizomenon.* However, these taxa also occur under conditions of surplus inorganic nitrogen. More importantly, nitrogen fixation is a process requiring high amounts of light energy and will not be effective in very turbid waters (as is the case during dense algal blooms). Thus, in a given water body the maximum amount of biomass that can grow, in addition to the biomass already present, can be estimated from the Redfield Ratio (see above) on the basis of the concentrations of dissolved inorganic nitrogen.

8.1.2 Phosphorus

Phosphorus, like nitrogen, enters water bodies from untreated and from biologically treated sewage, and further treatment steps are required to eliminate it. Phosphorus is biologically available as phosphate, which binds to soil particles more effectively than nitrate. Thus, the main entry route into water bodies from land areas is as surface run-off and with erosion. Although biomass needs only about one seventh of the amount of phosphorus as it needs of nitrogen, phosphorus is the resource which most frequently limits phytoplankton growth in aquatic environments. Cyanobacteria and many other phytoplankton organisms have developed storage mechanisms for phosphate (known as luxury uptake). These enable them to store enough phosphate for 3-4 cell divisions. As a consequence, one cell can multiply into 8-16 cells without requiring any further phosphate uptake, and biomass can increase by a factor of 10 or more even when dissolved phosphate is entirely depleted. For this reason, the amount of biomass that can grow in addition to the biomass present cannot be predicted from the concentrations of dissolved phosphate (see Box 8.1).

Phosphorus is naturally abundant only in very few aquatic ecosystems (such as some lowland estuaries, some volcanic lakes and some ground-water-fed lakes). Furthermore, its inputs to aquatic environments are often easier to control than nitrogen inputs.

Methods for elimination of phosphorus from domestic sewage are well developed and currently more cost-effective than nitrification and denitrification (although current developments may provide better approaches to combined elimination of both nutrients). Measures to protect soils from erosion can also be very effective against loss of phosphorus, whereas control of nitrate leachate from over-fertilised soils may be more difficult. Nitrogen limitation may be to some extent compensated by fixation of atmospheric nitrogen by cyanobacteria, whereas there is no comparable compensation mechanism for phosphorus.

8.1.3 Available light energy

Light energy is a critical resource indirectly affected by nutrient concentrations. Light arriving at a water surface is partly reflected, and the remainder is very quickly absorbed by the water itself as well as by the dissolved substances and by the suspended particles in the water. An upper limit of phytoplankton cell density is reached when the cells shade each other to such an extent that further growth is no longer possible because the individual cells do not receive enough light. This level can be estimated following Lambert-Beer's basic law of exponential extinction with increasing thickness of the water layer. This law can be expressed as:

$$I_z = I_o' e^{-z\varepsilon}$$

where I_Z is the intensity at depth Z

I'_o is the surface intensity

ε is the vertical extinction coefficient, which in turn is the sum of extinction by the water itself and the substances dissolved in it such as humic acids ε_w, the algae suspended in the water ε_a, and other particles suspended in the water ε_p.

The average amount of light I^* available to a phytoplankton organism entrained in vertical mixing of the entire water body or (under conditions of thermal stratification) within the upper, warm water layer (the epilimnion) is the square root of the intensity at the surface I_o and at the bottom of the mixed layer or of the water body I_m (Reynolds, 1997). This relationship illustrates the decisive influence of depth on light availability and enables estimation of the carrying capacity for phytoplankton biomass. For otherwise clear water with the sum of ε_w and ε_p being only 0.2, Reynolds (1997) uses chlorophyll as a measure of phytoplankton biomass and demonstrates that at 1 m depth and a daily insulation of 10^{-3} mol photons m^{-1} s^{-1} a maximum of 670 µg l^{-1} of chlorophyll may be sustained, whereas at a mixed depth of 10 m, only 49 µg l^{-1} are possible, and if mixing occurs down to 30 m, carrying capacity declines to only 3 µg l^{-1} chlorophyll. At these phytoplankton biomass levels, turbidity has also increased (expressed as increase of the term εa), and not enough light can penetrate to enable further growth.

Nutrient availability often influences light limitation. If nutrients are limiting, phytoplankton cannot grow to density levels that reach the light-determined carrying capacity. If nutrient concentrations are excessive, phytoplankton will reach the biomass limit determined by light (unless other factors such as hydrological flushing prevent growth). Further increase of nutrient concentrations will then have no further effect on phytoplankton biomass. This is often the case in hypertrophic water bodies. Turbid situations where the light-determined carrying capacity has been reached are often dominated by cyanobacteria, because at low light intensity these have a higher growth rate than many other phytoplankton organisms (see section 2.2).

8.2 Target values for total phosphorus within water bodies

In determining target values of phosphorus within water bodies to control cyanobacterial blooms, two questions are important:

• What phytoplankton biomass density can be expected at a given concentration of total phosphorus?

• At what threshold concentration of total phosphorus does phytoplankton density create a turbidity level high enough to reach the light-determined carrying capacity, and thus switch a water body from total phosphorus control of biomass to control by light limitation?

Finding answers to these questions requires a clear definition of total phosphorus. In the past, the soluble phosphate fractions have frequently been addressed when dealing with eutrophication issues. This has some predictive value if it can be measured in seasons where very little phytoplankton is present to consume dissolved phosphate (e.g. during severe light limitation in winter), and if inputs are fairly constant throughout the year. However, the carrying capacity for phytoplankton biomass is more reliably analysed in terms of the total amount of phosphate, i.e. the sum of phosphate bound in biomass and phosphate dissolved in the water, known as total phosphorus (Box 8.1).

Box 8.1 Monitoring total phosphorus as opposed to soluble phosphate fractions

Considerable confusion prevails in the use of the term "phosphate". Historically, soluble reactive phosphate (SRP) or orthophosphate has been measured and addressed when dealing with phytoplankton growth, because this is the fraction of total phosphate which is directly available for uptake by cyanobacteria and algae. However, recycling of phosphate molecules within the plankton communities has proved to be extremely rapid (within 5-100 minutes) (Wetzel, 1983), and phosphate liberated by degradation of organic material will be taken up by bacteria and algae faster than scientists can sample and measure it. Furthermore, cyanobacteria and algae can store enough phosphate for up to four cell divisions and increase 16-fold, even if no soluble reactive phosphate could be measured. If SRP is found above detection limits, this means that it is surplus to the requirements of the cyanobacteria and algae. The only informational value of such a finding is that growth is limited by some factor other than phosphate. The upper limit of the biomass of cyanobacteria and/or algae that can develop in a given water body is, therefore, often largely determined by the amount of phosphate bound within the cells, and total phosphate phosphorus is the variable that should be studied for biomass management. This variable is not equivalent to total phosphorus, which includes the mineral form (such as apatite) unavailable for biological uptake. However, mineral forms are of quantitative importance only in some water bodies (e.g. with high silt loading) and, for the sake of simplification, total phosphorus has become widely used to represent total phosphate phosphorus.

The term total phosphorus is preferable to the term total phosphate, because results are reported in terms of phosphorus rather than phosphate. This is important because the weight of the PO_4 molecule is about three times that of its central P atom, and lack of specification in reporting results as to whether they refer to µg PO_4 or µg P has caused considerable confusion in the literature.

For predicting phytoplankton density from total phosphorus concentrations several models have been developed. The most comprehensive statistical model was established through an international (largely European and North American) co-operative study organised by the Organisation for Economic Co-operation and Development (OECD) (Vollenweider and Kerekes, 1982). The concentration of chlorophyll *a* was used as an easy-to-analyse measure for phytoplankton density. Data for annual mean values of chlorophyll *a* and for maxima of chlorophyll *a* were compiled from a wide variety of phosphorus-limited lakes (77 for annual means and 50 for maxima) and related to annual mean concentrations of total phosphorus. The resulting regressions were almost linear and highly significant (Figure 8.1).

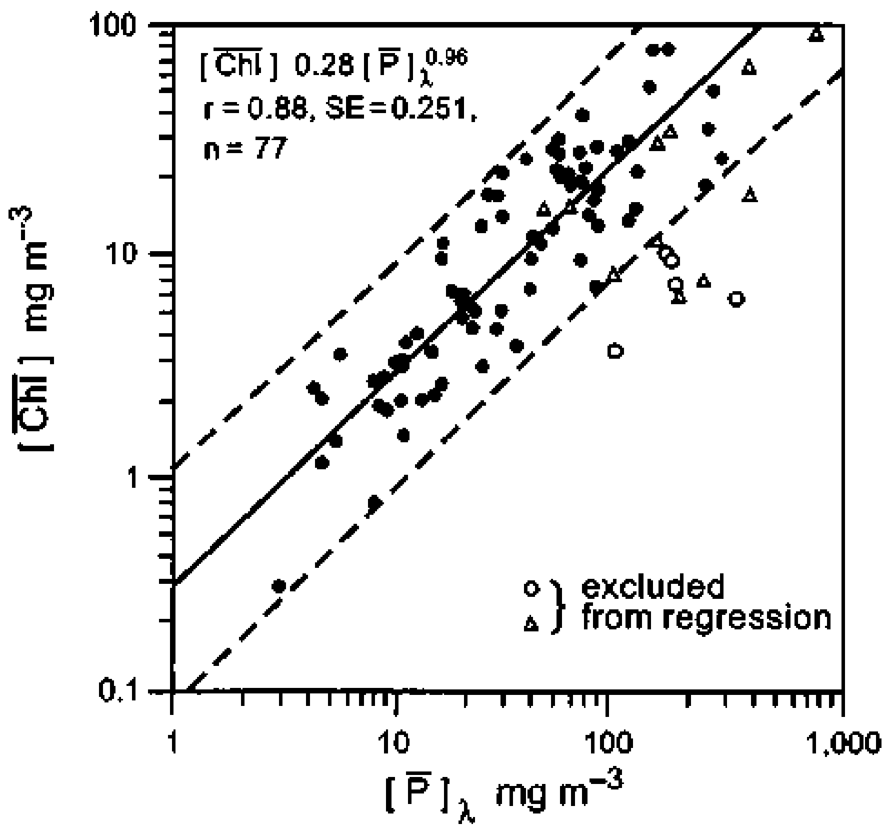
The result roughly means that per microgram of total phosphorus, an annual mean phytoplankton biomass corresponding to 0.25 µg of chlorophyll *a,* and a maximum of up to 1 µg of chlorophyll *a,* may be expected. These results, together with results on the occurrence of cyanotoxins given in Chapter 3, can be used as a basis for rough guide values for estimation of maximum cyanobacterial bloom biomass and toxin concentrations. In natural ecosystems, 1 µg of total phosphorus can support a biomass up to 100 µg of organic substance (corresponding to approximately 1 µg of chlorophyll *a),* which in turn may contain up to 1 µg microcystin. Substantially higher biomass and higher microcystin content are possible, but occur only through accumulation of cells in surface scums, or in some laboratory cultures.

Regressions, such as the OECD model illustrated in Figure 8.1, can be applied as management tools to predict the average and the maximum phytoplankton biomass range likely at a given concentration of total phosphorus. However, this approach has been criticised because these models integrate the behaviour of a number of lakes rather than the response of any one lake to changes in phosphorus concentrations. It must be emphasised that regression B is useful to estimate the maximum phytoplankton biomass at a given phosphorus concentration. However, the estimate given by the double logarithmic regression is only rough, the scatter of points within the 95 per cent confidence limits covers a factor of 10. Maximum ratios of chlorophyll to total phosphorus of 2 are still within this limit. This scatter reflects the effects of other environmental factors controlling phytoplankton biomass, particularly depth and mixing conditions, and losses due to grazing of algae and some cyanobacteria by zooplankton. The carrying capacity for phosphorus will not have been reached in all of the lakes among the wide variety used for regression B. Further management actions in addition to phosphorus control may be useful to avoid this carrying capacity being reached, i.e. to move the vertical position of a water body downwards in Figure 8.1.

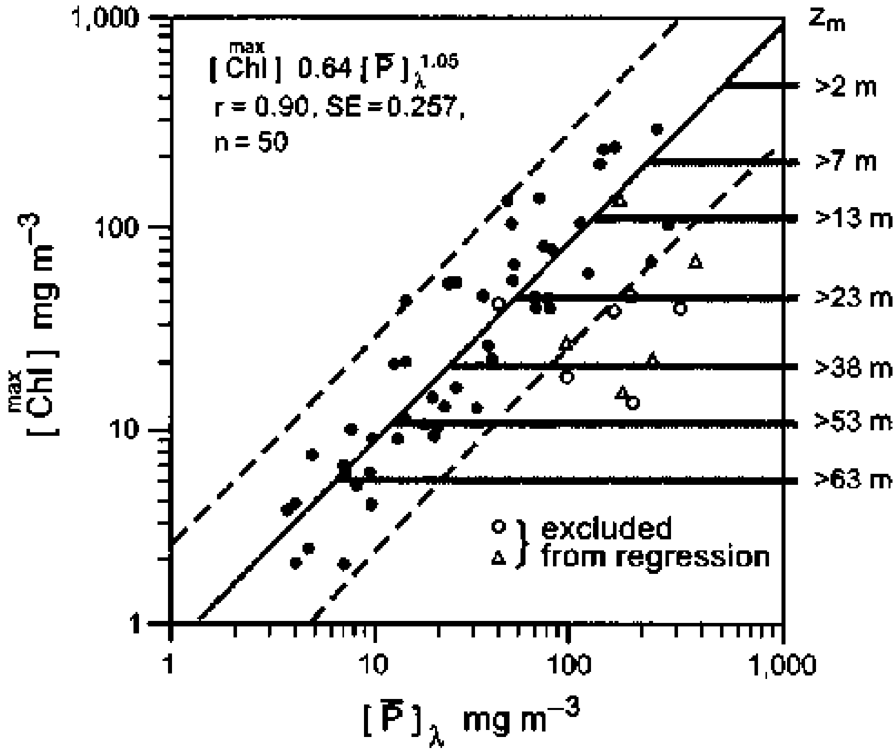
Predictability of maximum biomass levels is enhanced by combining this model of phosphorus-determined carrying capacity with a measure for light-determined carrying capacity by introducing the threshold concentrations where phosphorus limitation switches to light limitation. This requires knowledge of the depth of the water body and, if it is thermally stratified, knowledge of the depth of the warm upper mixed layer (epilimnion), also termed "mixing depth" Z_m. If data on light extinction are available as outlined above, average light intensity through the mixing depth I^* can be calculated. In absence of such data, light availability can be estimated by regarding Z_m in relation to the depth of light penetration (euphotic depth, Z_{eu}). If mixing is deep in relation to light penetration, cell or colony densities cannot become very high, because the deeply entrained cells would be spending too much time in the dark. If mixing is shallow, cells are frequently moved near the surface, where enough light can penetrate, even through a dense suspension, to enable extensive proliferation.

Figure 8.1 Vollenweider/OECD regressions for phytoplankton biomass (as chlorophyll *a*).

A. Annual means;



B. Maxima in relation to total phosphorus, amended with threshold levels for different mixing depths (Z_m) taken from Reynolds (1997) at which carrying capacity is limited by light ($\varepsilon_w + \varepsilon_p$ $m^{-1} = 0.2$). Dotted lines represent 95% confidence intervals of the regressions



The maximum biomass density which can be reached is higher in lakes with shallow mixing depths than in deeply mixed lakes, because the latter reach their light-determined carrying capacity at a much lower level of biomass. Figure 8.1B gives the chlorophyll levels at which light limitation truncates phosphorus limitation for different mixed depths in otherwise clear water (εw+ εp = 0.2 m^{-1}), and further phosphorus input will not result in a further biomass increase. In turbid waters (e.g. due to silt loads), this level will be reached at much lower phosphorus concentrations (see Reynolds (1997) for chlorophyll capacities at higher light extinction coefficients).

When designing programmes to reduce cyanobacterial proliferation in highly eutrophic lakes, total phosphorus concentrations must be reduced below the threshold value for phosphate limitation of biomass in a given lake in order to have an effect; this threshold value will depend upon the depth of the lake. However, once phosphorus levels have been brought below the threshold value the OECD model can be applied. For example, if a shallow drinking water resource has total phosphorus levels of 600 µg l^{-1} and cyanobacterial densities corresponding to 200 µg l^{-1} chlorophyll *a,* and if restoration measures are applied that achieve total phosphorus levels of 200 µg l^{-1}, the annual mean chlorophyll *a* concentration is likely to decline to 70 µg l^{-1}. Although this is a step in the right direction, cyanobacterial biomass is still very high and problems of bloom formation are not yet resolved.

Experience collected during the past two decades with restoration of water ecosystems shows that phosphorus control for abatement of cyanobacterial blooms should target concentrations at least as low as 30-50 µg l^{-1} total phosphorus (Cooke *et al.,* 1993). In many water bodies, substantial reduction of cyanobacterial and algal population density can be expected at these concentrations when compared with higher concentrations. However, significantly lower total phosphorus concentrations (less than 10 µg l^{-1}) may be required, particularly in deep lakes, in order to prevent blooms of some stratifying ecostrategists in the long term. Information on the prevalent ecostrategists within the cyanobacterial population will help to predict the success of management measures (see section 2.3 and Box 8.2).

Box 8.2 Thresholds for phosphorus control of different cyanobacterial ecostrategists

Knowledge of the prevalent ecotypes in a given water body leads to the following consequences for total phosphorus management:

• *If scum-forming ecostrategists prevail* (such as *Microcystis* spp. or *Anabaena* spp.) cell numbers and biomass are likely to decline if total phosphorus concentrations can be brought well below 50 µg l^{-1} P. This will also reduce scum formation, because less cells and colonies will be available to concentrate into scums. Nonetheless, some scums will probably continue to occur until phosphorus limitation becomes so severe that cell density (and therefore turbidity) decreases to the point where the depth of light penetration is as deep as the depth of large areas of the water body ($Z_{eu} = Z_m$). Under these conditions, vertical migration of these taxa is less effective because their buoyancy regulating mechanism requires some time in the dark (see section 2.2). Therefore, they lose their competitive advantage over other phytoplankton.

• *If dispersed ecostrategists prevail* (such as the filamentous species *Planktothrix agardhii,* formerly named *Oscillatoria agardhii)* very pronounced "switches" may be expected. As phosphorus limitation reduces filament density, and thus turbidity, to the point where the relationship of the depth of light penetration to the depth of the mixed layer (Z_{eu}/Z_m) is greater than 0.4, these species are likely to disappear quite abruptly, and turbidity will increase even further, thus stabilising the result.

• *If metalimnetic ecostrategists prevail* (such as *Planktothrix rubescens*), the water layer above these cells is usually quite clear. Very low concentrations of total phosphorus (often below 10 µg l^{-1} P) are necessary to decrease turbidity further and thus increase light intensity down to the depth inhabited by these species. If this can be achieved, metalimnetic ecotypes may disappear. If not, hydrophysical measures may be more successful in controlling their density.

• *If nitrogen fixing ecotypes prevail* (such as *Anabaena* spp.), reduction of total phosphorus down to concentrations effectively limiting biomass will cause dissolved nitrogen concentrations in excess of uptake by phytoplankton. Nitrogen fixation is then no longer an advantage in competition over other cyanobacteria and algae. This may induce disappearance of the nitrogen fixing species.

8.3 Target values for total phosphorus inputs to water bodies

The OECD study, which provided the regression shown in Figure 8.1, evaluated data from 87 lakes with respect to the relationship between total phosphorus concentrations in the lake (annual means) and the external load (input) (Vollenweider and Kerekes, 1982). The regression showed a close correlation between annual means of in-lake concentrations and the annual means of inlet concentrations in relation to the residence time of the water:

$$TP = 1.55[P_{inlet} / (1 + \sqrt{\text{residence time}})]^{0.82}; r = 0.93, n = 87$$

Table 8.1 Permissible and dangerous inputs for phosphorus and nitrogen for different depths and for a renewal time of 2 $m^3\ m^{-2}\ a^{-1}$

Mean depth (m)	**Permissible inputs**[1]		**Dangerous inputs**	
	P (g $m^{-2}\ a^{-1}$)	**N (g $m^{-2}\ a^{-1}$)**	**P (g $m^{-2}\ a^{-1}$)**	**N (g $m^{-2}\ a^{-1}$)**
<5	<0.07	<1.0	>0.13	>2.0
<10	<0.1	<1.5	>0.2	>3.0
<50	<0.25	<4.0	>0.5	>8.0
<100	<0.4	<6.0	>0.8	>12.0
<150	<0.5	<7.5	>1.0	>15.0
<200	<0.6	<9.0	>1.2	>18.0

[1] Permissible inputs increases with residence time; a doubling of the residence time increases the permissible inputs by a factor of 1.6

Source: Harper, 1992

The "safe loadings" given in Table 8.1 were derived from this model. The model may serve for preliminary scaling of measures to reduce inputs of phosphorus. For prediction of the total phosphorus concentrations in a given water body, estimates of inputs from inlets, from surface run-off (especially from agricultural areas with tillage, fertilisation and erosion), from urban storm-water outfalls and from atmospheric precipitation are needed. Acquiring all of the necessary data may be difficult, and approaches to estimates are discussed in the context of lake and reservoir diagnosis by Cooke *et al.* (1993).

Currently, very little information on relationships between inputs and in-lake concentrations is available from tropical and subtropical aquatic ecosystems. Future research must investigate whether relationships established for water bodies in temperate climates apply, or whether changes are necessary. Differences may be expected, especially with respect to sediment-water interactions and mineralisation rates, because these depend strongly upon temperature and upon thermal stratification.

If inputs exceed critical values for a given system, increase of the concentrations of total phosphorus within that system are likely. In turn, cyanobacterial proliferation is likely, and management measures are then needed to reduce phosphorous inputs.

8.4 Sources and reduction of external nutrient inputs

In most cases, eutrophication is enhanced by anthropogenic activities. The three major sources of external nutrient inputs are run-off and erosion from fertilised agricultural areas, erosion resulting from deforestation, and sewage. Exceptions may occur and are illustrated by the example of Lake Victoria. This large lake has an area of 68, 000 km^2. Tributaries supply about 15 per cent of its water and 85 per cent originates from precipitation. Burning of field stubble is widely practised and leads to substantial air pollution. Thus, 60 per cent of the phosphorus load is estimated to enter Lake Victoria through precipitation. Only 40 per cent originates from sewage and run-off into the tributaries (Lindenschmidt *et al.,* 1998). Replacement of the practice of burning stubble would substantially reduce this load.

Sustainable approaches aim at reducing nutrient loads at their source or as close to the source as possible (sections 8.4.1 and 8.4.2). If this is not feasible, approaches to reducing inputs from a main tributary (i.e. treating a main tributary as if it were a sewage channel) may be considered (sections 8.4.3). Reduction of external inputs beneath the threshold expected to be effective is an important basis for the success of further in-lake restoration measures (which may address internal nutrient loads or ecosystem structure, see section 8.5).

The first questions to ask in designing programmes for abatement of eutrophication by improving agricultural practices and/or by introducing or improving sewage treatment are:

• How high is the phosphorus input from wastewater (sewerage outfalls or diverse small sewage inlets) to a given water body (excretion of 2-4 g P per person per day may be assumed) (Siegrist and Boiler, 1996)?

• How high is the input from agriculture and run-off from other surfaces?

• Down to what concentrations must phosphorus input be reduced in order to reduce concentrations in the receiving water body beneath a total P threshold likely to be effective in the given water body (see section 8.2)?

Answering these questions requires specific evaluation of the resources to be protected, of their catchment land use, and of the water and effluent drainage network. Ideally, nutrient inputs and the relative share of different nutrient sources should be estimated and catchment characteristics, such as soil type, run-off potential and vegetation cover should be considered. In a region with nutrient-rich, erodible soils and reduced vegetation cover or natural eutrophication (e.g. river deltas or some tropical areas), reductions in inputs will not be possible to the same extent as in a region with sandy soils, flat relief and dense tree cover (Cooke *et al.,* 1993). Thus, the same measures and similar investments into reducing inputs are likely to be more successful in reducing eutrophication in a potentially oligotrophic ecosystem than in a naturally eutrophic one.

In many cases, quantitative assessments of inputs will not readily be available because this requires detailed analysis of hydrological conditions (e.g. assessment of stream flow rates and water retention times), as well as nutrient concentrations and their variations over time in all of the main tributaries. Such investigations require time and resources. Whereas inputs from point sources like sewage effluents are relatively easy to assess, diffuse inputs from agriculture are often very difficult to quantify. Managers are likely to be confronted with water bodies for which almost no limnological data are available, and perhaps not even the depth contours of the water body are known. Planning and implementation of resource protection measures may be delayed for several years before reliable data become available. The dilemma for managers is whether to begin with measures to reduce obviously substantial inputs, without having the data basis to predict whether the measures taken will reduce nutrient concentrations below the threshold effective for controlling cyanobacteria, or whether to delay planning and decision-making until data become available.

In some countries, user friendly "decision support" software programs (see Box 8.3) have been compiled for use by managers and community groups in identifying the main sources of nutrients from a catchment, and for identifying possible actions e.g. off-river disposal of wastewaters or (re) construction of riparian buffer strips to protect against inputs from erosion. There is an array of computer models that may be used to simulate the hydrodynamic and transport conditions in a catchment system. Trudgill (1995) gives examples of available models and profound discussions of various processes that may be included in the compilation. Since numerous parameters are required for calibration of these models, the complexity of the model should be tailored to the extent of the data base available. For applying these models, the study should include the entire watershed (and, in some cases, the airshed as well) and not just the lake water body.

No general recommendation can be given to resolve the dilemma between the necessity of adequate planning data, and the need to implement obvious measures without delay. Although numerous restoration measures around the world have proved to be ineffective due to insufficient diagnosis and evaluation by scientists and managers, most of these were measures addressing ecological balances within the water body (see section 8.5). In contrast, measures addressing reduction of external nutrient inputs are not likely to be applied mistakenly. In the worst case they may prove to be insufficient and require further action - either further reduction of inputs, or in-lake action.

An effective alternative to quantitative assessment of loading, is a "common-sense" or qualitative approach. This begins with studying maps and geographical information to identify main tributaries, slopes critical for erosion, precipitation patterns and land use. Detailed and critical inspection of the catchment area may provide an excellent basis for recognising priority actions, some of which may be implemented at low cost. Such inspection is generally of underestimated value. Qualitative assessments should include identification of sewage outfalls (possibly illegal or unregistered), land use, vegetation cover, agricultural practices (e.g. soil tilling supportive of erosion, lack of protective riparian buffer strips with dense vegetation cover as a barrier between surface runoff and water body, and stubble burning).

Box 8.3 Testing the Catchment Management Support System (CMSS) in the Murrumbidgee River Catchment, New South Wales, Australia

The Catchment Management Support System (CMSS) is a simple computerised decision support system developed by CSIRO Land and Water, Australia. It combines land use, nutrient generation and land management information into a single model which can predict the impacts and costs of different land management practices in a catchment. As the set-up and use of CMSS does not require a technical or computing background, it is highly suited to use by catchment management committees and other community or management groups (Davis and Farley, 1997).

In a recent application (Cuddy *et al.,* 1997), CMSS was developed for the Murrumbidgee River Catchment, New South Wales (NSW) with the intention that it should be provided to local catchment committees as a tool in the development of nutrient management plans. This pilot study was used to demonstrate the application of CMSS to a specific area, as an initial phase of its application to major catchments in NSW. In applying CMSS to the Murrumbidgee Catchment, significant inputs were required from relevant management organisations and individuals. The necessary data, ranging from soil types and rainfall distribution to the initial and ongoing costs of changed sewage treatment practices, were available from these organisations. The Murrumbidgee CMSS was developed within the timeframe of the pilot study and handed over to the local agencies. Although the pilot study successfully met its objectives and the program has now been widely adopted, CMSS has yet to be influential in the production of nutrient management plans. This reflects both the prolonged nature of developing plans through community consultation and the lack of involvement by local catchment committee members in the pilot study.

While investments in internal or "in-lake" control measures can be wasted without sufficient pre-restoration evaluation, reduction of external nutrient inputs will be at least a first step in the right direction. Managers are encouraged to implement nutrient control measures, even if the data base is not sufficient to predict the quantitative impact on concentrations within the water body.

8.4.1 Domestic wastewater

Wastewater emissions have frequently been managed by a philosophy of using water as a medium for transporting wastes out of the locality without considering the impairment of the function of the water as a resource further downstream, or of the possible enrichment of lakes with pollutants and nutrients.

Many developing and least developed countries are extending their coverage of drinking water supplies in order to improve human health. Evaluation of experience shows that this usually leads to a several-fold per capita increase in water consumption and thus also in the amount of waste-water generated. Improvement of human health therefore also requires development of wastewater collection and treatment, as highlighted by the World Bank:

"For urban water supply, experience indicates that the collection and proper treatment of sewage must be an integral part of water supply projects. Bringing water into a city without taking sewage out exposes the population - and particularly the poor -to increased pollution." (World Bank, 1993)
Possible exposure to pathogens is a major reason why collection and treatment of wastewater is important. Another reason is that cyanobacteria proliferate in eutrophic lakes and rivers fertilised by wastewaters.

Replacement of phosphorus in laundry detergents can typically reduce phosphate loads in sewage by 50 per cent at best, the remaining 50 per cent being inevitable because it originates from human excreta. Wherever sewage outfalls are considered to be a significant input of phosphorus to a water resource, phosphate elimination, alternative treatment approaches (see Box 8.4), or sewage diversion is necessary. Criteria for adequate technology largely depend upon population structure and on geographic conditions.

If population density is low, and the flushing rates of water bodies or phosphorus absorption capacity of the soils is high, nutrient elimination from sewage may not be necessary - high nutrient concentrations in water resources in such situations are more likely to originate from agriculture. Treatment methods adequate for protection from infectious agents may also be sufficient in these circumstances (on-site treatment such as properly constructed latrines, septic tanks and sewage lagoons). If domestic waste-water is used in agriculture, health risks should be avoided by following the WHO guidelines for the use of wastewater in agriculture and aquaculture (Mara and Cairncross, 1989). A special aspect of such situations is tourism. Low population areas affected by tourism may need special consideration because the population may increase temporarily several-fold and overload sewage treatment capacities. In temperate regions, the tourism season may coincide with the cyanobacterial growth season.

Box 8.4 Nutrient retention using low- and medium-technology approaches

Alternatives to "high-tech" nutrient stripping methods in sewage treatment (involving an array of methods using lagoons or land treatment) have been in use for more than a century, and their advantages as well as their drawbacks are well established. Many land treatment systems require large areas (up to 10 m^2 per population equivalent), and infiltration of wastewater into the ground below has often occurred without control for hazardous substances or pathogens. Modern artificial wetland systems are being developed to overcome these shortcomings. These systems combine mineralisation processes in the water body with filtration through the soil substrate; they are sealed towards the bottom and they have controlled effluents. As with "high-tech" treatment plants, artificial wetland systems require careful maintenance operation, and control. They are by no means a solution for "letting nature do the job alone". Poor maintenance and overloading rapidly lead to malfunctioning and to poor hygienic conditions in the system. In temperate regions, performance during the cold season may be less effective, but as a means for handling additional sewage loads caused by tourism during the warm season, such systems may be excellent supplements to year-round treatment technologies. Other approaches comprise hygienic methods of collection of excreta, re-use of waste in agriculture and regular emptying of septic tanks. Care must be taken particularly with respect to occupational hazards, and with the designing and siting of alternative treatment methods in order to avoid relocation of the wastewater problem to another site (e.g. causing groundwater contamination).

Sparsely populated regions with water bodies highly susceptible to eutrophication, e.g. because of long water retention times (such as many lakes in Sweden), will require more carefully designed sanitation systems to protect these resources. Supplementary treatment techniques may be necessary, particularly during the tourist season (see Box 8.5).

In many densely populated areas, municipal sewage is the source of at least half of the total phosphorus inputs to rivers and lakes. In large urban areas, treatment of wastewater collected in sewerage systems requires industrial-scale plants for the protection of surface waters. Untreated municipal sewage contains more than 10 mg 1^{-1} of phosphorus. Biological (secondary) treatment oxidises organic matter, but does not substantially reduce phosphorus content. Where municipal sewage constitutes a significant source of phosphorus pollution, removal of phosphorus at treatment plants is necessary (see Box 8.5). Heavy seasonal tourism may also be a problem in such situations because it increases the demand on the capacity of treatment plants and sewerage and also causes substantial fluctuations in the sewage load. Sewerage and treatment for fluctuating amounts of sewage present specific technical difficulties. Lake Balaton is an example of such a situation, where the number of tourists during July and August is twice that of the local population (Somlyody and van Straten, 1986).

Box 8.5 Two well-established and widely used technologies for phosphorus removal in treatment plants

Chemical precipitation with ferric or aluminium salts, often performed simultaneously with biological treatment, can reduce phosphorus concentrations by an order of magnitude to values around 1 mg l^{-1}.

Biological phosphorus removal ("bio-P") can be achieved by alternating aerobic and anaerobic steps in biological treatment and thus substantially enhancing P-uptake by bacteria. This method saves flocculation chemicals and produces less sludge, but requires an adequate design of basins and careful operation of the process. In large treatment plants (> 100, 000 population equivalents) it is more economic than chemical flocculation, because operation costs are lower (Gleisberg *et al.,* 1995). It can reduce treatment plant effluent concentrations down to 0.2-0.5 mg l^{-1} P. Although the method has been known for several decades, experience has only recently accumulated to a level of understanding which allows stable and reliable performance, and it is advised to maintain chemical precipitation facilities as a back-up (Harremoes, 1997), especially for treatment plants discharging into water bodies with critical phosphorus concentrations.

In lowland regions, discharge of domestic and industrial wastewater may amount to 50 per cent and in dry seasons to almost 100 per cent of the total flow of the recipient river, e.g. River Thames in London (Gray, 1994) and Havel River in Berlin (Köhler and Klein, 1997). Such slow flowing rivers are suitable habitats for cyanobacterial growth but the river water may also be needed for production of drinking water. In such situations, phosphorus elimination by simultaneous chemical precipitation or "bio-P" is not adequate, because concentrations in the recipient water body will almost equal the outfall concentrations. Treatment objectives therefore are 0.03-0.05 mg l^{-1} P and this can be attained by adding a further treatment step (filtration) for removal of phosphorus in small, slowly settling floes. The removal of pathogens is usually also a further important objective of filtration treatment in regions with such intensive water use. Various methods of filtration over sand or gravel and pumice beds are available, and new methods of membrane filtration are also being developed.

The costs of such treatment technology may appear intimidating at first glance because of the necessary investments. However, even sophisticated procedures involving filtration need not cost more than US$ 0.15-0.30 per m^3 of treated water (Heinzmann and Chorus, 1994), and this is usually only a small fraction of the costs of drinking water. Effective resource protection in such densely populated areas will save the cost of drinking water treatment, for example by saving the necessity of activated carbon filtration.

Eutrophication due to sewage outfalls has been recognised as a widespread problem since the 1970s. Abatement was begun in that decade with several isolated projects, such as diversion of sewage around lakes with specific touristic value (e.g. at Lake Constance in the 1970s and at major parts of Lake Balaton in the 1980s) and the introduction of phosphorus precipitation in a few treatment plants. Comprehensive programmes began in the mid 1980s. In Europe, an international convention for the protection of the North Sea triggered introduction of phosphorus and nitrogen elimination in larger sewage treatment plants (those treating more than 10,000 population equivalents). Recently, the implementation of measures to eliminate nutrients in sewage has started to show substantial success:

• Denmark achieved a 79 per cent reduction of phosphorus inputs from sewage between 1985 and 1995 and further reduction is expected in 1997 when two plants in Copenhagen go into full operation (Harremoes, 1997).

• Switzerland achieved a 60 per cent decline in total phosphorus inputs from municipal wastewater (Siegrist and Boiler, 1996).

• The USA and Canada together achieved a 50 per cent reduction in phosphorus concentrations in Lake Ontario and the west basin of Lake Erie, with significant impact upon "algal blooms" and accumulations of filamentous cyanobacteria on shorelines, by the construction and upgrading of sewage treatment plants along the Great Lakes (Charlton, 1997). However, in order to maintain the quality now achieved despite the expected increase of population density by the year 2011, and in order to meet the quality targets for Hamilton Harbour, tertiary sewage treatment (effluent filtration) is considered necessary.

Charlton (1997) emphasises the importance of reliable performance of sewage treatment plants: *"Part of the difficulty in understanding sewage problems is the mistaken belief that sewage treatment plants, once built, will perform as planned, with no operational problems"* (Charlton, 1997). Steady degradation of performance during critical summer months has led to considerable phosphorus inputs, the highest being 2.7 times the target. Internal assessment, optimisation of performance, and identification of staff with the job and the achievement of effluent target concentrations, are crucial for reducing effluent loads.

8.4.2 Agriculture and erosion

Losses of phosphorus and nitrogen from deforested, agriculturally-used areas into surface waters are the other major factor enhancing eutrophication. As with phosphorus pollution from wastewater, this problem has increased exponentially in some parts of the world since the 1950s. The causes are structural changes in agriculture involving intensification by tillage of larger plots, extensive application of mineral fertilisers, and the establishment of large-scale animal husbandry. Simultaneously, other major parts of the world suffer substantial lack of phosphorus in topsoils. Zehnder (1996) points out a striking global imbalance between surplus phosphorus in most of the industrialised regions and a lack of phosphorus in most developing regions. On a global scale, phosphorus must be perceived as a limited resource. In regions with phosphorus deficiency, controlled fertilisation in combination with adequate protection from erosion, would help maintain fertility of tropical and subtropical soils, so that further deforestation would no longer be necessary in order to gain new (only transiently fertile) farmland. Further, development of sewage treatment methods that allow reclamation of phosphorus for reuse as fertiliser should be a long-term target for sustainable handling of this resource.

Excessive use of fertilisers and manure has created eutrophication problems in lakes and rivers. In regions with intensive agriculture and slowly flowing rivers with little discharge, e.g. in much of north-western Europe, water bodies without cyanobacterial problems have become scarce. Heavy surplus fertilisation has been enhanced by the widespread trust in phosphorus retention in soils (soils as "savings banks" for phosphorus), but soil erosion and surface runoff have proved to be major pathways into surface waters, particularly through storms and intensive rainfall shortly after application of fertiliser or manure. The extent of these losses is site-specific and largely depends upon geographic and hydrological conditions: "It *has been suggested that up to 90 per cent of the annual phosphorus losses occur from only 5 per cent of the land during only one or two storms, especially in areas where surface runoff and erosion are the dominant routes for phosphorus losses"* (Oenema and Roest, 1997). The importance of leaching, first recognised as a pathway for nitrate inputs, is being recognised increasingly for phosphate in some types of soils, e.g. sandy, acidic soils with a high degree of saturation of their phosphorus adsorption capacity (Oenema and Roest, 1997). For the Netherlands, Oenema and Roest (1977) estimate 300,000-400,000 ha of phosphorus-leaking sandy soils to be pollution "hot spots" requiring high priority in identification and remediation.

In tropical and subtropical regions, eutrophication of lakes and reservoirs due to inputs associated with erosion is greatly enhanced.

Closing cycles by reuse of manure as a nutrient resource in agriculture, especially if combined with changing practices of land tillage and deforestation, can contribute to reduction of nutrient pollution. Such approaches are sustainable alternatives to considering animal slurry as waste, and reduce the costs of purchase of fertilisers. In this context, performance of large-scale livestock farming in industrial dimensions requires regulation just as for industry:

"The output of waste from many cattle and pig units measures up to that of a large town and needs to be managed with at least the same care as that accorded to human waste. At one time the only environmental requirement placed on farmers was that they should observe good agricultural practice. This is no longer a satisfactory basis for environmental protection. Agriculture must be put on the same basis as other major industries with strict controls on the quality and quantity of effluent discharges."
(Packham, 1994)

Box 8.6 Good agricultural practices - best management practices

For fertilisation

- Planning land use, choice of crops and crop rotation to minimise erosion losses.

- Structuring the farmland to minimise erosion by measures such as terracing, interruption of large areas with shrub hedges, and buffer strips planted with shrubs along river banks and lake shores.

- Planning nutrient management in order to avoid losses from the farm by closing nutrient cycles, using manure as fertiliser and avoiding phosphorus import.

- Measuring current fertiliser content of soils and dosing according to the demand calculated for the crop.

- Timing application of fertilisers according to the growth of the crop.

• Cover crops to reduce erosion from bare soil.

• Managing irrigation and groundwater levels.

• Using animal slurry as fertiliser according to the demand of the crop, rather than misusing crops or grasslands as a deposit site for animal slurry.

For animal husbandry

• Reduction of livestock density to 1.5-2 cattle units per hectare.

• Closing nutrient cycles by limiting stock numbers to the fertilisation requirements of the area used for growing crops.

• Placing feedlots and watering sites away from surface waters.

• Protecting river banks and lake shores with fences to keep out livestock in order to reduce both direct pollution by excreta and increased erosion by treading.

The alternative to Packham's request is to redefine "good agricultural practice" to include sustainable resource use. Criteria for "best management practices" or "good agricultural practices" are listed in Box 8.6.

Apart from closing nutrient cycles, sustainable biological methods of production have been developed in many pilot projects and are becoming increasingly popular. Model projects have demonstrated that productivity of biological or "organic" farming methods is not substantially lower than that of conventional farming, provided methods are adapted adequately to the given geographic conditions. The economic balance of "biological" or "organic" farms is frequently equally good because of reduced expenditure for agrochemicals and, in some cases, better prices for the product. Co-operation between water supply agencies and farmers has supported this development in Germany (see Box 8.7) and has shown success in improving resource quality.

In drinking water catchments, it is particularly important that agriculture follows "best agricultural practices". This can be encouraged by designating protection zones around the drinking water source and regulating practices allowed or prohibited within these protection zones. Sophisticated models distinguish two to three degrees of protection, depending upon the relative impact of the respective part of the catchment upon the water quality. In addition to agriculture, other activities which impact water quality, such as forestry, fisheries and tourism, may be regulated in drinking water protection zones.

Countries with a traditional rural society may have options for reclaiming historical agricultural experience and combining it with modern approaches to sustainable "best agricultural practice" in order to attain high outputs of high quality products at low environmental and health impact. Such an integrated approach requires continuous development, evaluation of experience and training.

Countries with large-scale industrialised agriculture may have problems in implementing change. In contrast to the success achieved in phosphorus elimination by wastewater treatment, progress in abatement of agricultural phosphorus pollution has at best been modest. For Switzerland, Wehrli *et al.* (1996) estimate that, while phosphorus emissions from sewage have been reduced by 60 per cent during the past 15 years, losses from agricultural areas into water bodies have rather increased and the need for a new agricultural policy is just beginning to be widely perceived. Some of the reasons for this delay in awareness and action are:

• Losses of phosphorus from farmland are rarely perceived as economic losses.

• Effective measures will differ regionally or even from farm to farm, and management practices must be optimised locally rather than administered generally. This requires shifts of attitudes.

• Even where programmes and regulations exist, their implementation and control may be difficult.

A basic change in attitude is required. In some countries this appears to be developing slowly as a new generation of farmers with better training (including education on sustainable farm management and ecological impacts) takes over, and as consumer awareness for quality criteria and the ecological impact of products is growing. A wide array of measures can be used by government authorities to support such developments. Examples are training and advice to farmers, eco-audits on products, subsidies for setting land aside from use, subsidies or tax redemptions during periods of transition to organic farming methods, pollution taxes and legislation to enforce water protection.

Box 8.7 Co-operation between water suppliers and agriculture for sustainable provision of healthy drinking water

Legislation should include the principle that use of land and water must occur in such a sustainable way that subsequent use by others Is not hampered. This provides a legal basis for requiring co-operation of agriculture with water supply agencies. Furthermore, protection zones above aquifers or around reservoirs must be staked out so that they cover the actual "intake" of the respective resource. Where protection zones already exist, new hydrological understanding often shows that in many cases these areas are much larger than previously presumed, and that protection zones must be expanded.

Such areas or zones are especially suitable for developing models of co-operation. One such model is the foundation of a voluntary association of those concerned, namely farmers, water and health administrations, representatives of agricultural associations and, amongst others, the water supply organisation. A steering committee or executive board should be elected, in which the water supply organisation should not take the lead. The guiding principle is "co-operation instead of confrontation". An alternative may be to have direct contracts between the farmers and the water supply agency. Such contracts bind the farmers to certain methods of production in return for some financial support, especially during the years of transition from intensive farming to sustainable methods. Tasks for such associations and their leadership are:

• Issuing regular advice, e.g. for suitable situations for applying manure, fertiliser or well-targeted pesticides, and issuing prohibited periods (e.g. "no liquid manure on frozen ground"), or computer-supported fertilisation schedules.

• Regulating the maximum density of livestock tolerable without risking pollution of the aquifer.

- Organising advice and training for farmers, e.g. for measuring soil content of fertilisers.

- Establishing time schedules for changes in land use (e.g. four years transition time for extensivation, eight years of use as pasture, but then use only for forestry). Often, lenient time schedules will be fulfilled much more quickly than required.

- Provide seeds for intercropping or keeping the ground covered to protect against erosion.

- Purchase suitable machines for demonstration or for communal use.

The general emphasis is on advice; on using, developing, publishing and making available local experience and expertise; and on expressing concern and requesting responsible co-operation rather than on issuing prohibitions. A very successful tool has been the installation of counsellors for sustainable farming whose salaries are paid by the water supply agency, but who work within the agricultural authorities and organise courses as well as giving individual advice and training.

The economic aim is to market products which have been audited in relation to health and sustainable land use. Farmers can join an organisation for organic production methods and sell their products under the name of the association; this should be encouraged by the association. Such organisations then take over the responsibility for checking that members comply with the rules and the methods of production; this helps to enforce sustainable production methods. Such organisations can be encouraged to advertise for membership in the region.

The costs of such models of co-operation vary considerably, depending on whether farmers must be supported during phases of transition and on the services provided. Nevertheless, experience shows that these measures increase the price of water only by a few cents per cubic metre (Such, 1996).

Establishing such co-operations is easiest if the land in the protection zone belongs to the water supply company or agency and is only leased to the farmers. Furthermore, employing an expert on agriculture may help a water supply agency considerably in negotiating with farmers. Success with this approach has been reported by Such (1996), Höllein (1996) and Fleischer (1996).

8.4.3 Treatment of drinking water reservoir inlets

Where drinking water reservoirs with one major inflow have a large share of diffuse, non-point source inputs and a strong need for rapid remediation, reduction of nutrients in the inflow may be the most effective option. Pre-reservoirs with retention times of at least several days can reduce total phosphorus inputs by 50-65 per cent (Klapper, 1992). Retention times should allow incorporation of phosphorus into algal biomass and sedimentation of that biomass, but should not be large enough for slow growing taxa, such as cyanobacteria, to establish dominance. Sediment dredging may be necessary at intervals of several years in order to counteract the re-release of phosphorus.

The Kis-Balaton reservoir in Hungary is an example of a special wetland and shallow reservoir system of 60 km^2 designed to retain phosphorus and reduce inputs to the tenfold larger Lake Balaton. Water is retained for one month (mean value) in an intricate system between coffer dams and reed zones. Phosphorus retention has been successful, but flooding with stormwater from the Zala River has caused pulsed phosphorus inputs to the lake (Padisák, Pers. Comm.).

If the largest nutrient share originates from a single major inflow, phosphorus stripping facilities can be very effective in reducing inputs. Successful examples, with different degrees of technological sophistication, are the Wahnbach Reservoir, Lake Schlachtensee, Lake Tegel (Sas, 1989), and the Haltern Reservoir (Paetsch and Kötter, 1980).

8.5 Internal measures for nutrient and cyanobacterial control

In planning restoration measures for lakes and reservoirs, it is important to realise that substantial time lags may occur between measures to reduce external inputs and the results achieved in the water body. Feedback mechanisms within the ecosystem (e.g. sediment-water interactions or the establishment of new dominant species) require time to reach a new equilibrium. Hypertrophic aquatic ecosystems have specific positive feedback mechanisms which stabilise trophic state and cyanobacterial dominance and therefore resilience effects are not uncommon, even after substantial reduction of inputs below thresholds calculated to be effective. Sas (1989) pointed out that resilience patterns occur on two levels:

- Delayed response of in-lake total phosphorus concentrations to a reduction of input, due to the time required for flushing phosphorus out of the water body, and the time required for establishment of new sediment-water equilibria.

- Delayed response of phytoplankton biomass and species composition to reductions of in-lake total phosphorus concentrations, due to stability of prevailing biocoenosis structures and/or biotic enhancement of internal load.

Experience shows that several years (up to 10) may be necessary between the implementation of a restoration measure that substantially decreases inputs, and visible success in terms of reduction of phytoplankton biomass and cyanobacterial blooms. Monitoring of phosphorus inputs and phosphorus concentrations in the recipient water body during this time is recommended. Usually, a declining trend in total phosphorus concentration will be the first detectable response to a reduction in inputs and will indicate whether a particular measure can be expected to be successful, but it may take years for phosphorus concentrations to decline below the threshold effective for controlling phytoplankton biomass. Often, such time lags are due to the (sometimes substantial) phosphate storage capacity of anoxic sediments typical in hypertrophic waters. Flushing rates (i.e. the inverse of retention times) strongly influence the time necessary to reach a new equilibrium. In some cases, particularly in water bodies with low water exchange rates, supplementary "internal" measures may be advisable in order to accelerate a response. The following sections briefly introduce and evaluate a number of such measures for which experience is available.

With very few exceptions, internal measures are appropriate only after, or in combination with, an effective reduction of external inputs. In principle, reduction of inputs should be the actual restoration or resource protection measure, and internal measures should serve as a further boost to switch the ecosystem out of resilience and into a new balance. If possible, a few years of patience and observation of nutrient concentration trends within the lake will show whether internal measures are necessary. Only rarely are internal measures without adequate reductions of inputs justified as an emergency approach; usually such measures require continuous operation (such as aeration, see section 8.5.2) or repeated application (such as in-lake phosphate precipitation, see section 8.5.1). Medium- to long-term success of restoration investments is at stake if this principle is not considered.

8.5.1 In-lake phosphorus precipitation

In lakes and reservoirs with high water retention times, decline of phosphorus concentrations may be very slow, even after external inputs have been reduced to levels which should ensure a mesotrophic or oligotrophic state. If the water body has a high phosphorus content that is flushed out only slowly, only some of the phosphorus within the biomass will settle to the sediments. Much of it is released from decaying organic material, entrained back into the water body by water circulation, taken up by cyanobacteria or algae and, in part, passed on to higher levels of the food web. Degradation of organic material at the sediment surface has often led to anoxic conditions which may accelerate phosphorus release rates dramatically. Thus, phosphorus within a lake can be recycled many times, and no decline of cyanobacterial biomass can be achieved without reducing this in-lake phosphorus pool. Sometimes, this situation also applies to lakes which are naturally eutrophic, such as lakes in western Canada situated on phosphorus-rich glacial till (Prepas *et al.,* 1997). Precipitation of phosphorus from the water body to the sediment can be a successful measure, if it is undertaken so that phosphorus remains permanently bound in the sediment.

Prerequisites for lasting success are low external loading, sufficient depth to prevent sediment resuspension due to wind events, and adequate choice of flocculants. Experiments with precipitation of phosphorus have been undertaken with aluminium sulphate, ferric salts (chlorides, sulphates), ferric aluminium sulphate, clay particles and lime (as $Ca(OH)_2$ and as $CaCO_3$).

Ferric salts are effective in precipitating phosphorus, but difficult to handle because of their aggressive acidity. Furthermore, the iron-phosphorus complex is stable only under oxic conditions. Thus application of ferric salts usually requires subsequent continuous aeration to avoid re-dissolution of phosphorus under anoxic conditions. Due to the high mobility of iron ions, addition of iron frequently often has to be repeated at regular intervals. In addition, Prepas *et al.* (1997) point out that iron may be a limiting micro-nutrient in some systems and, in such situations, treatment with ferric salts may actually stimulate growth of cyanobacteria and algae.

Aluminium sulphate is poorly soluble under neutral and high pH conditions, but may decrease pH in waters with low buffering capacity, which leads to solubilisation and problems of alum toxicity.

Lime (both $Ca(OH)_2$ and $CaCO_3$) has been used as an algicide to coagulate and precipitate phytoplankton cells out of the water column (Murphy *et al.,* 1990; Zhang and Prepas, 1996). It is non-toxic, usually fairly inexpensive, and the pH-shock for the aquatic biota can be minimised by careful dosing over an extended time span. Unlike treatment with copper sulphate, the precipitation of cyanobacterial cells with $Ca(OH)_2$ does not appear to cause cell lysis and toxin release into the water (Kenefick *et al.,* 1993; Lam *et al.,* 1995). Lime also functions, to some extent, as a longer-term algal inhibitor, reducing eutrophication by precipitating phosphorus from the water (Murphy *et al.,* 1990). It appears that $Ca(OH)_2$ is more effective than $CaCO_3$ in precipitating phosphorus

(Murphy *et al.,* 1990). Many of the studies of both the mechanism and effects of liming for algal control have been carried out in eutrophic, hard water lakes or farm dugouts (dams) in Alberta, Canada (*Murphy et* al., 1990; Zhang and Prepas, 1996). It is possible that the technique may be more effective in these conditions than in soft water. The dose rates used are also quite high (e.g. 50-250 mg l^{-1} $Ca(OH)_2$) (Zhang and Prepas, 1996) which would make the technique prohibitive for large lakes. Techniques for the application of lime, which involve pumping or spraying of a slurry, are described by Prepas *et al.* (1990b).

Experience with in-lake precipitation of phosphorus is increasingly being compiled. A number of documented case studies show success either in terms of reducing phytoplankton biomass or in terms of shifting species dominance away from cyanobacteria. Nevertheless, numerous unsuccessful cases have also been documented, and further development of these techniques is ongoing (see compilation in Klapper (1992) and in Cooke *et al.* (1981, 1993)). Furthermore, in some water bodies, the concentrations of iron or calcium compounds in the inflow are naturally high and regularly provide sufficient binding sites for phosphate to induce natural phosphorus precipitation. Measures in the catchment area or changes in inflow regime may have considerable impact in either increasing or decreasing this input, and thus may have a significant impact on the trophic state of the water body.

8.5.2 Sediment dredging and phosphorus binding

Release from sediments may be a substantial source of phosphorus (sometimes referred to as internal loading) for many years after external inputs have been minimised. Water exchange rates, sediment chemistry, temperatures, mixing conditions, and bioturbation govern phosphorus release rates. Iron-bound phosphorus is highly sensitive to redox conditions; when sediment surfaces turn anoxic during summer stratification, phosphorus concentrations may increase dramatically, fertilising cyanobacteria in their optimum growing season. Under oxic conditions in shallow, unstratified systems, high pH (> 9.8) may strongly enhance oxic phosphorus release (Ryding, 1979). Because high pH values are a result of intensive photosynthetic activity, this phosphorus release pathway is a positive feedback mechanism in favour of cyanobacterial blooms. Other aerobic phosphorus release mechanisms may also be significant, especially bioturbation by feeding fish and invertebrates (Gardner *et al.,* 1981).

Options for measures to counteract sediment release are removal of sediment (dredging) or treatment to bind phosphorus. Dredging is costly and will reduce release rates only if:

- It is carried out down to sediment layers with a lower or less mobile phosphorus content.

- Phosphorus-rich interstitial water is handled in such a fashion that it does not reach the water body and cause additional inputs.

- Dredged sludge can be deposited where it does not create a new external input with erosion and stormwater runoff into the lake.

In some urban and industrial regions, dredging is precluded or complicated by high concentrations of heavy metals and organic contaminants in the sediments which would then require disposal as hazardous waste. Dredging is particularly recommended for smaller water bodies where the trophic state can be further improved by gaining depth, or which also need to be cleared of dumped rubbish.

Sediment treatments aim at trapping phosphorus in the sediment, either by oxidation to insoluble iron compounds, or by adsorption onto calcium carbonate or clay particles. During the past two decades, broad experience collected with numerous failures (see Box 8.8) and a few successful cases has shown that effective treatment requires careful design on the basis of profound understanding of the sediment chemistry and hydrology of the water body to be treated. Oxidisation may be achieved by aeration, artificial mixing (see also section 8.5.5), or the introduction of pure oxygen. It appears to be most effective if achieved with nitrate, which transports more oxygen and penetrates more readily into sediments. Well-treated sewage effluent (not contaminated with harmful substances, fully nitrified and after phosphorus removal) may be suitable for this purpose, if the process is controlled so that nitrate concentrations are not elevated in drinking water.

Box 8.8 Is aeration effective in binding phosphate in sediments?

The frequent failures in the use of aeration to meet the objective of reducing phosphorus efflux from sediments require critical highlighting. Many aeration projects had several objectives, often not carefully distinguished and planned, such as (i) providing sufficiently high oxygen concentrations for survival offish and fish eggs in deep waters and on the sediment surface, (ii) destratification in order to entrain buoyant cyanobacteria, and (iii) oxidising sediment surfaces. Some of these objectives may be conflicting, e.g. destratification will increase sediment surface temperatures, thus potentially enhancing phosphorus release, and it will transport nutrient-rich, near-surface water into upper strata where these nutrients can be used for growth of cyanobacteria or other phytoplankton. Often, aeration has proved to be insufficient for achieving the aim of reducing phosphorus release. Even the prominent, carefully designed, experiment at the Swiss Baldegger See did not succeed in increasing phosphorus retention of the sediments after 10 years of operation (Wuest and Wehrli, 1996). Energy costs of aeration may be considerable. At the present state of the art, it can be recommended only for increasing the oxygen content of the water (e.g. as a fish habitat), or if artificial mixing is desired - success in increasing phosphorus retention in lake sediments appears doubtful. Injection of pure oxygen appears to be more successful in some cases (Gemza, 1997; Prepas *et al.,* 1997).

8.5.3 Withdrawal of bottom water from the hypolimnion

In thermally stratified eutrophic lakes, phosphorus accumulates in the hypolimnion (cold bottom water layer) during summer stagnation, partly from settled organic material originating in the upper water layers and, in many lakes, largely from the release of sediment-bound phosphorus under anoxic conditions. Although most natural outflows drain surface water, it is often possible to dam the natural outflow and to abstract hypolimnetic water instead (Olzewski, 1961). This is especially easy to apply to reservoirs and can reduce in-lake concentrations significantly. In the Swiss Mauensee the biomass of *Planktothrix rubescens* was reduced from 152 g m^{-3} to 42 g m^{-3} using this approach (Gächter, 1976).

Nürnberg (1997) compiled the advantages of hypolimnetic withdrawal during summer stratification as a method based solely on selective output of total-P rich water. The advantages of the method are:

- It addresses the cause of eutrophication.
- It does not add chemicals.
- It does not necessarily change the water budget.
- It can break the cycle of enhanced sediment accumulation of total phosphorus.
- It can flush more phosphorus out of the system than the sediments accumulate each year.

Hypolimnetic withdrawal is effective only if enough water flows into the lake. Furthermore, some lowering of the water level may be tolerable, but complete destratification by removal of most of the hypolimnion should be avoided, because increasing the contact area between warm surface water and sediments will enhance phosphorus release due to elevated temperatures. In addition, impairment of water quality downstream will require attention if the amount of phosphorus released is high in relation to the total flow. Downstream phosphorus pollution may be avoided by treatment of the hypolimnion outlet with chemical phosphorus precipitation. Nevertheless, the low temperatures of the hypolimnion water may have a substantial impact on downstream biological processes, such as fish breeding.

8.5.4 Reduction by flushing

Flushing with water of low phosphorus concentrations can greatly reduce external inputs and will also accelerate recovery from internal loading by removing in-lake phosphorus which would otherwise be recycled for a number of growing seasons. If suitable water is available in sufficient quantity, flushing can be a very effective tool for reduction of cyanobacterial proliferation. Successful examples are Veluwemeer in the Netherlands (Sas, 1989) and Moses Lake in the USA (Welch *et al.,* 1972). However, this measure also implies a relocation of the phosphorus to another water body, and this impact must also be evaluated.

8.5.5 Hydrophysical measures

Cyanobacteria show different "strategies" of survival in competition against other phytoplankton organisms (section 2.3). Many of these strategies are adapted to specific hydrophysical conditions. Changing these conditions may therefore substantially reduce the success of these cyanobacterial "ecostrategists" and allow other phytoplankton species to become dominant. This approach can be an effective temporary, supplementary measure alongside reduction of external inputs of nutrients, particularly if in-lake nutrient concentrations have declined to values around the threshold where success may be expected. In some cases, where eutrophication levels cannot be decreased, permanent installation of hydrophysical measures can be a solution (see Visser *et al.* (1996) for the example of Nieuwe Meer in Amsterdam).

The mass development of scum-forming species is highly dependant on the stability of the water column. In water without vertical mixing, the colonies of *Microcystis* or other colony-forming taxa can migrate up and down by changing their specific weight (see section 2.3). Interrupting this vertical migration of the colonies by artificial mixing of an otherwise stably stratified water body, can prevent rapid development of surface scums.

Furthermore, disrupting the possibility for these organisms to move into strata with optimum light conditions is likely to reduce their growth rate and thus their efficiency in competing against other phytoplankton. In contrast, mixing improves growth conditions for taxa such as diatoms, which depend on mixing to remain in suspension. Thus, increased mixing may shift species composition from cyanobacteria to, for example, diatoms.

Thermally stratified water bodies naturally have an upper mixed layer known as the epilimnion. If artificial mixing substantially increases the depth of this layer, it reduces the light-determined carrying capacity, or the concentration of phytoplankton biomass possible (see also section 8.1). To be successful, artificial mixing measures must satisfy three conditions (see Visser *et al*., 1996):

- At least 80 per cent of the water volume should be mixed.

- The artificial mixing rate must be higher than the rate of vertical movement of the colonies of cyanobacteria. Rates of colony movement depend on colony size and thus are somewhat variable (see section 2.3), but as a general rule, a mixing rate of 1 m h^{-1} is sufficient to prevent cyanobacterial blooms.

- A large part of the water body must be sufficiently deep. In most cases artificial mixing has been caused by installing aeration tubes which are connected to a compressor on the shore. The aeration tubes are situated in the deeper regions of the water body. Waters with extensive shallow areas have a low circulation rate which can negatively influence the results of artificial mixing. Furthermore, if the water body is too shallow, mixing cannot reduce the light-determined carrying capacity strongly enough to prevent cyanobacterial growth. The example of Nieuwe Meer (Visser *et al.,* 1996) shows that more than 20 m depth may be required.

A number of mixing projects have been unsuccessful because these principles were neglected. Many systems are now on the market, provided by different engineering companies. Engineering expertise is sufficiently developed to design systems that can meet the hydrophysical requirements. Care must be taken, however, to select competent companies, and to plan the measure to meet the ecological targets set in combating cyanobacterial blooms. Furthermore, in tropical and subtropical countries with high and prolonged insulation, the costs of systems are enough for mixing to become prohibitive.

8.5.6 Biomanipulation

Biomanipulation includes a range of techniques that influence algal growth by manipulation of parts of the food web of a lake. Examples are removal of planktivorous and benthivorous fish populations, providing refuges for zooplankton and introducing predatory fish such as pike (*Esox lucius*) in order to decimate planktivorous fish populations, and introducing submerged aquatic plants to compete with phytoplankton in consuming nutrients (Kitchell, 1992). These techniques aim at stimulating the growth or presence of phytoplankton-grazing organisms or of phytoplankton competitors.

Increasing grazing pressure

In shallow lakes, the removal of a large proportion of benthic and planktivorous fish can be helpful to diminish algal growth. Without this predation pressure, zooplankton and benthic fauna can develop and feed on algae and some species of cyanobacteria (e.g. early stages of *Microcystis* population growth when colonies are still very small). Selective removal of benthic fish reduces resuspension of sediments and thus mobilisation rates of sediment phosphorus (in deep lakes this is difficult). To stimulate these effects, predatory fish fingerlings can be introduced to diminish the population growth of the planktonic and benthic fish. Artificial refuges can be placed to provide habitats for zooplankton and pike. The artificial refuges are important when the development of submerged aquatic plants (macrophytes) is insufficient to serve this purpose.

The introduction of predatory fish can be effective. Interventions into established hypertrophic ecosystem structures by fish stock management techniques have proved successful in smaller ponds and lakes over shorter periods of time (Hrbácek *et al.,* 1978). If successful the water may become clearer due to a reduction of algal and/or cyanobacterial turbidity, the zooplankton populations increase, and fields of macrophytes may develop which compete for phosphate with the phytoplankton (thus reducing their capacity for growth) (see below). However, the breeding success of the remaining planktivorous fish stock in the lake will be high if insufficient predatory fish are present. Continued control of the development of the fish stocks is required, and the removal of planktivorous fish must be repeated regularly. Depending upon local salary levels, this may be expensive in terms of personnel. Biomanipulation is by no means a cheap method because of the continuous monitoring and management requirements. It is also unlikely that the technique will work naturally and unaided once the change in biological structure has been introduced.

A disadvantage of biomanipulation is that not all phytoplankton species are eaten efficiently by zooplankton. Stimulating the zooplankton without reducing concentrations of phytoplankton nutrients may stimulate dominance of inedible phytoplankton species, such as colony-forming (*Microcystis, Aphanizomenon*) or filamentous cyanobacteria *(Planktothrix agardhii*), or the green alga, *Enteromorpha.* High nutrient levels may also stimulate the growth of epiphytic algal species which grow on the surfaces of macrophytes and suppress their development.

Box 8.9 Ecosystem theory to explain how biomanipulation works

As indicated in section 2.2, eutrophic systems with blooms of *Planktothrix* (formerly *Oscillatoria*) *can* show enormous stability, with uninterrupted dominance over years. May (1977) indicated that multispecies assemblages of plants and animals can have several different equilibrium states. Scheffer (1990) highlighted this for shallow eutrophic lakes. He concluded that during the process of lake restoration, two different equilibria are possible at a state of moderate eutrophication - one with large populations of phytoplankton and planktivorous and benthic fish, and another in which nutrients are incorporated more evenly distributed among zooplankton, fish and macrophytes. The first system is turbid, the second system is clear. Resilience of the ecosystem during restoration maintains the turbid state over long periods, particularly if phosphate concentrations oscillate around the threshold effective for reducing phytoplankton biomass, but do not substantially decline below this level. In such situations, biomanipulation can help to switch the ecosystem from a turbid phytoplankton community to a clear macrophyte community.

Enhancing competition by introducing macrophytes

The introduction of macrophytes has the best chance of success in water bodies with a relatively large shallow littoral area (< 2-3 m deep) and at moderate concentrations of total phosphorus. Reynolds (1997) points out that if the areal nutrient input rate leads to phosphorus concentrations above 0.15 mg 1^{-1}, phytoplankton density can readily reach 0.15 mg 1^{-1} of chlorophyll *a,* and then submerged macrophytes will simply be "shaded out". However, at lower concentrations of phosphorus, particularly in spring, macrophytes have the chance to begin to grow and to incorporate enough of the available phosphorus to achieve substantial phosphorus limitation of phytoplankton biomass. Particularly if combined with the management of fish stocks, measures to support macrophytes may switch an aquatic ecosystem into a different, sometimes rather stable, biological structure resulting in clear water and low cyanobacterial biomass (Box 8.9).

General assessment

Reynolds (1997) summarises current knowledge on mechanisms of biomanipulation under the heading "bottom-up or top-down control"; "bottom up" implies control by nutrients and "top-down" implies control by the top end of the food chain, i.e. by consumers. Reynolds (1997) lists arguments against biomanipulation, such as self-starvation of consumers by outstripping the availability of phytoplankton or consequences for other components of the food web, and points out that the conceptual flaw is thinking in terms of "steady states". He comments:

"the state that generally attains is a lurching alteration between responses to plenty and responses to over-consumption.... When tropically-related organisms, with lives measured in hours to years, inhabit environments fluctuating with periods of days to months, their interactions are certainly likely sometimes to result in strong top-down pressures on producer biomass. At other times, however, the trophic cascade slows down to barely a trickle" (Reynolds, 1997).
These dynamic fluctuations are most likely to occur, such that they affect phytoplankton populations, in situations which are not extremely biased by total lack of nutrient limitation. Biomanipulation as a management tool to reduce algal or cyanobacterial growth is most likely to be successful in situations of moderate nutrient concentrations and in combination with reductions in inputs. Experience shows that, as long as the trophic level of the systems remains high, the risk that the ecosystem switches back into its original composition is also higher. For reviews of biomanipulation see Demerol *et al.* (1992), Carpenter and Kitchell (1992) and Moss *et al.* (1994).

8.5.7 Algicides

Algicides, especially copper sulphate, have been used rather widely in some regions to kill prevailing cyanobacterial blooms. As a result of the Palm Island catastrophe (see section 4.1) it was established that lysine a bloom may exacerbate problems because toxins previously contained within the cells are liberated and pass through drinking water filters far more readily than toxins within intact cells (see also Lam *et al.,* 1995). Nevertheless, preventative treatment at the beginning of bloom development has been widely used (Cameron, 1989) and may be necessary (see section 9.2).

Algicide treatment of water bodies is best considered as an emergency measure and may involve ecological risks. Toxic copper deposits may accumulate in the sediments (Prepas and Murphy, 1988). Repeated treatment may induce shifts in species composition towards more copper-resistant, but not necessarily more pleasant, species. This was the case in Lake Matthews, a drinking water reservoir for California, where taste-and-odour problems caused by *Oscillatoria* spp. were handled by copper sulphate application. Within very few years, the dosage needed to combat these species had to be increased from 27 to 400 t. The treatment led to the replacement *of Oscillatoria* by a more copper-resistant cyanobacterium, *Phormidium* sp., which prevailed for longer time spans and caused almost all year-round off-flavour problems (Izaguirre, 1992). Other undesirable ecosystem impacts of algicide treatment cannot be excluded. Wherever possible, it is preferable to choose abatement measures which address the source of the problem (i.e. growth conditions for cyanobacteria) because such solutions may be effective in the long term and actually safeguard human health by improving environmental quality.

8.5.8 Barley straw

The use of decomposing barley straw for the control of cyanobacteria and microalgae has been investigated recently (Welch *et al.,* 1990; Jelbart, 1993; Newman and Barrett, 1993; Everall and Lees, 1996). The effect of rotting barley straw in reducing filamentous green algal growth was reported by Welch *et al.* (1990) and algistatic effects were shown in laboratory cultures of the cyanobacterium *Microcystis aeruginosa* by Newman and Barrett (1993). The inhibitory effects were suggested to be due to antibiotic production by the fungal flora or to the release of phenolic compounds such as ferulic acid and f-coumaric acid from the decomposition of the straw cell walls.

A reduction in cyanobacterial populations has also been reported in reservoir trials after applying barley straw (Everall and Lees, 1996). These authors suggested that phytotoxic compounds released from decomposing straw inhibited the cyanobacterial populations, but that further chemical identification, and risk and environmental assessment were required prior to use in water supply reservoirs. However, a recent full-scale field trial has been carried out in a potable supply reservoir and was credited with reducing regular summer cyanobacterial populations (Barrett *et al.,* 1996). Although these trials gave apparently favourable algistatic results, they were carried out without replication or control trials and, as such, the study design cannot account for the influence of other factors (such as impact of weather conditions) on phytoplankton development and succession. An earlier trial which did include the dosing with barley straw of one of a pair of closely adjacent lakes (with similar hydrology and biology) resulted in a decrease in the cyanobacterial population in the straw-dosed lake as compared with the non-dosed lake, throughout the two years of post-dose monitoring (Harriman *et al.,* 1997).

There are conflicting data from Australia on the effects of barley straw. Jelbart (1993) failed to find any inhibitory effects with extracts of rotting straw on *Microcystis aeruginosa* isolates. Cheng *et al.* (1995) also found no algicidal or algistatic effects from barley straw over a six month period in a comprehensive field trial in six experimental ponds. The ponds were fertilised to encourage cyanobacterial growth and there were no differences in species composition or final standing crop between control and straw-dosed ponds.

These contradictory findings and the unknown identity of the inhibitory factors in rotting barley straw indicate that straw-dosing is still too poorly understood to recommend for reliable use as a cyanobacterial control measure, particularly in potable water supply reservoirs. Whether barley-straw dosing influences the composition and size of toxin pools in cyanobacterial populations still needs to be determined. Dosing with barley straw has gained unwarranted popularity and notoriety because it is an apparently simple procedure which is relatively inexpensive and highly visible. It is being used in freshwaters for cyanobacterial control in some countries even though the benefits are dubious (e.g. the introduction of rotting, oxygen-consuming organic matter needs consideration).

8.5.9 Other approaches

A market for rapid and cheap water resource protection and restoration methods is evolving. In Europe, poorly validated methods for reduction of cyanobacterial and phytoplankton growth are being advertised, some of which are questionable. In some cases, transient success was actually due to natural seasonal "clear water" phenomena. Although new approaches require field testing as part of development, independent verification of their success can reasonably be requested of their promoters prior to marketing. Public health officers and other public authorities are trained in other fields than environmental sciences and rarely have the expertise to judge restoration proposals. Review by environmental authorities or experts is therefore desirable prior to investment.

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