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CHAPTER 4 MANAGEMENT AND CONTROL IN SOURCE WATERS (LEVEL 1)

BACKGROUND

In this chapter we discuss management strategies that can be applied within the water body for the control of cyanobacteria, assuming that, where possible, efforts have been undertaken to address any external nutrient inputs from the catchment (Chapter 2).

There are a number of techniques to control or minimise the growth of cyanobacteria in reservoirs. They are represented by a range of:

- Physical controls
- Chemical controls
- Biological controls

In essence management strategies focus on either controlling factors that influence growth, or damage or destroy the cyanobacteria. Management strategies have recently been comprehensively summarised and reviewed by Cooke et al. [1].

A summary of measures that can be applied in lakes and rivers for the management of cyanobacteria is given in Table 4-1. The most commonly utilised techniques are described in more detail in the following sections.

Table 4-1 Techniques for the management of cyanobacteria.

Control method	**Technique**
Physical	
	Artificial destratification, aeration, mixing
	Dilution to decrease retention time
	Scraping of sediments to remove benthic algae
	Drawdown and desiccation to remove benthic algae
	Sediment removal to reduce nutrient release
Chemical	
	Sediment “capping” with P-binding agents
	Algicides, algistats
	Coagulation
	Hypolimnetic oxygenation
Biological	
	Virus, bacterial infection
	Biomanipulation, increasing grazing or competition for available light and nutrients

PHYSICAL CONTROLS

MIXING TECHNIQUES

A major problem in reservoirs experiencing periods of warm stable conditions is the warming of the upper layer of water; one effect of this is reduction in the mixing of the water column, resulting in stratification (see Chapter 1). During stratification the water stratum adjoining the bottom sediments, the hypolimnion, becomes depleted of oxygen, and contaminants such as ammonia, phosphorus, iron and manganese can be released from the sediment in a soluble form. This increase in nutrient levels can lead to the uncontrolled growth of cyanobacteria. Species such *Microcystis* and *Anabaena* are susceptible to this effect as they exhibit buoyancy due to internal gas vacuoles, and can migrate vertically within the water column, taking advantage of both the light near the surface and increased nutrient levels near the sediment of the water body. Mixing of the water column will disrupt this behaviour and limit the accessibility of nutrients, and thus limit cyanobacterial growth. It may also introduce oxygen to the hypolimnion, preventing further release of nutrients, and possibly increasing the oxidising conditions sufficiently to induce precipitation of the nutrients back to the sediments. In some cases this can prevent the formation of surface scums of toxic cyanobacteria. The mixing regime may also provide more favourable conditions for growth of competing organisms such as diatoms. Artificial mixing has been shown to be effective in many situations e.g. [2, 3, 4].

The two most commonly used methods of artificial destratification are bubble plume aerators and mechanical mixers.

AERATORS

Bubble plume aerators operate by pumping air through a diffuser hose near the bottom of the reservoir. As the small bubbles rise to the surface they entrain water and a rising plume develops. This plume will rise to the surface and then the water will plunge back to the level of equivalent density. An intrusion will then propagate horizontally away from the aerator plume at that depth. As the intrusion moves through the reservoir there is return flow above and below the intrusion and these circulation cells cause mixing between the surface layer and the deeper water or hypolimnion. An illustration of this effect is given in Figure 4-1(a).

The efficiency of a bubble plume is determined by the depth of the water column, the degree of stratification and the air flow rate. The number of plumes, plume interaction and the feasible length of aerator hose required to destratify a particular water body must also be considered in aerator design. As a general rule, bubble plumes are more efficient in deeper water columns. In shallow water columns (<5.0m depth) the individual air flow rates of the plumes must be very small to maintain efficiency.

Level 2 link to more detail about aerators

MECHANICAL MIXERS

Mechanical mixers are usually surface-mounted and pump water from the surface layer downwards towards the hypolimnion, or draw water from the bottom to the surface. This produces a simple mixing effect that is illustrated in Figure 4-1(b).

Both types of destratifiers have been shown to mix the surface layers close to the mixing device but areas of the water body further away from the immediate influence of the mixing may remain stratified and provide a suitable environment for cyanobacterial growth. One approach to consider is the use of both mixing techniques in the same water body, where the aerator generates basin-wide circulation cells and the mixer targets the surface stratification outside the direct influence of the aerator plume. This has been used with some success at the Myponga Reservoir in South Australia.

Links to Myponga Reservoir case studies

Effect of mixing on stratification and the phytoplankton community

Effect of mixing on nutrient release and algal biomass

Using mathematical models to predict cyanobacterial growth

Simulation of various management strategies

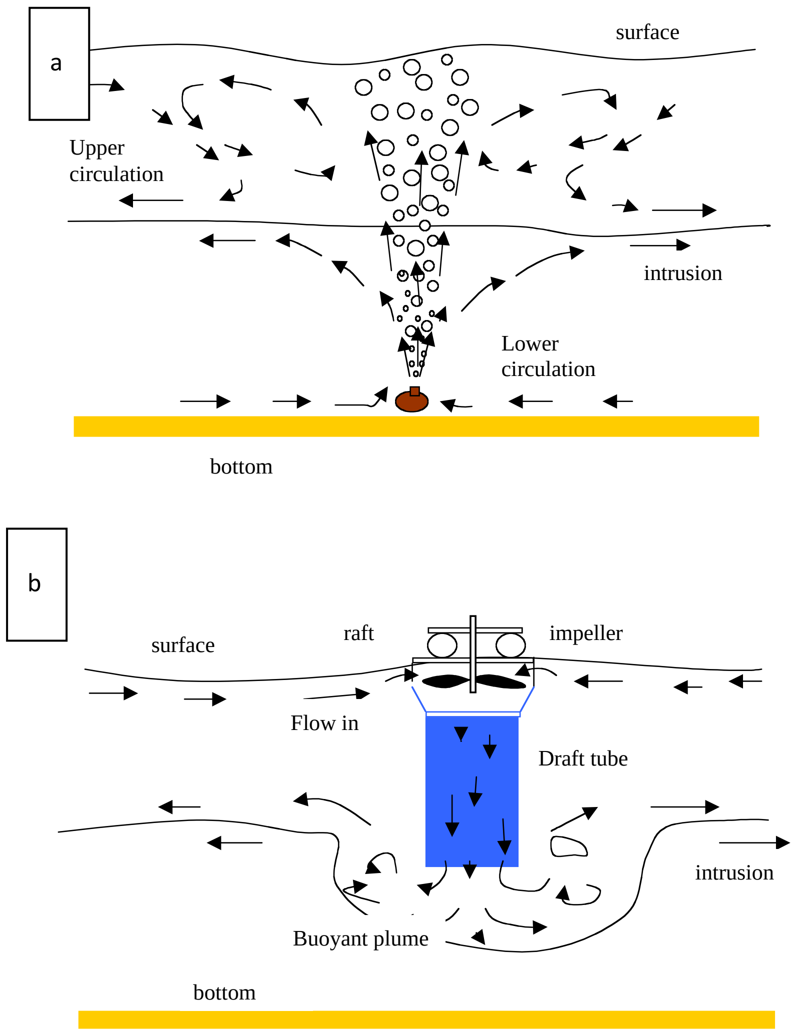

Figure 4-1 Flow and circulation fields created by a bubble plume aerator a) and a surface-mounted mechanical mixer b) in reservoirs

For the successful application of artificial destratification the water body must be sufficiently deep for efficient mixing of at least 80% of the volume. If a larger percentage of the water lies in shallow regions cyanobacteria may accumulate and multiply in these favourable stratified conditions [5]. It is therefore important to apply the appropriate mixing processes for a particular water body. Schladow [6] describes in detail a method for the design of destratification systems for water bodies impacted by cyanobacteria blooms.

Figure 4-2 shows the implementation of mechanical mixing and aeration at Myponga Reservoir, South Australia.

Figure 4-2 Mechanical mixer (left) and aerator (right) at Myponga Reservoir

Destratification is normally employed during late spring, summer and autumn depending upon the amount of surface water heating experienced during those periods. Historical records of temperature would give a guide to when destratifiers should be used. Regular temperature profiles will provide information on how well mixed the reservoir is. The most sophisticated destratification systems automatically adjust the compressor flow rate based upon data from on-line thermistor strings.

A summary of some factors influencing the application of destratification can be found here

MANIPULATION OF RIVER FLOWS

Low flow conditions in rivers can lead to stratification and cyanobacterial growth. In regulated rivers the magnitude and timing of discharge can be manipulated to disrupt stratification every few days thereby controlling cyanobacterial growth. Bormans and Webster [7] reported the development of criteria for flow manipulation that may result in destratification sufficient to disrupt cyanobacterial growth. Sufficient water must be available for the application of this management strategy and consideration should also be given to the impact of a variation of flows on other aquatic organisms.

OTHER PHYSICAL METHODS

As many problem cyanobacteria can form scums at the surface of a water body, oil-spill skimmers have been used to remove the cyanobacteria, usually to sewer or landfill. Figure 4-3 shows the use of a skimmer to remove surface scum in a recreational lake in South Australia. Atkins et al [8] reported the effective use of coagulation with polyaluminium chloride combined with the removal of surface scum with an oil spill skimmer to treat a severe cyanobacteria bloom in the Swan River in Perth, Australia.

Figure 4-3 The use of a skimmer to remove surface scum in a recreational lake in South Australia. Toxic material was collected and disposed to sewer

Benthic cyanobacteria can be treated using physical methods such as reservoir draw down, followed by desiccation and/or scraping to remove the layer of algae attached to sediments or rocks. However, these methods may not have the desired outcome. A recent study has shown that benthic cyanobacteria can be tolerant to desiccation [9], and scraping or other physical removal can generate turbidity and localised spikes in odour compounds or toxins, which may be an issue depending upon the proximity of the supply offtake.

Figure 4-4 shows the exposure of benthic cyanobacteria after draw-down of a reservoir aimed at control by desiccation.

Figure 4-4 Benthic cyanobacteria exposed after reservoir draw down

If a high nutrient level is due to sediment release it is possible to physically remove sediments. However this is a labour intensive process with implications for short term water quality, and should only be applied if external nutrient input has been significantly reduced.

CHEMICAL CONTROLS

CHEMICAL CONTROL OF NUTRIENTS

HYPOLIMNETIC OXYGENATION

The main aim of hypolimnetic oxygenation is to increase the oxygen concentration in the hypolimnion to prevent or reduce the release of nutrients from the sediment without disrupting the existing stratification of the water body. In this way the nutrient levels in the upper layers of the water body may become limiting to cyanobacterial growth. Techniques used to achieve hypolimnetic oxygenation include airlift pumps, side stream oxygenation and direct oxygen injection [10]. These techniques are relatively expensive, so an extensive understanding of lake hydrodynamics, sediment nutrient release rates and the internal and external contributions to the total nutrient load is necessary to determine whether this would be the most effective management option.

PHOSPHORUS PRECIPITATION AND CAPPING

Precipitation of phosphorus from the water body to the sediment, and treating the sediment to prevent phosphorus release, sometimes called sediment capping, are two methods that have been applied with mixed success.

Reports in the literature show that precipitation of phosphorus can be accomplished with aluminium sulphate, ferric chloride, ferric sulphate, clay particles and lime. The effectiveness of these treatments is highly dependent on the hydrodynamics, water quality and chemistry of the system as the phosphorus can become resuspended or/and resolubilised, depending on the turbulence of the water and the oxidising conditions near the sediments.

Treatments to prevent phosphorus release by applying a layer on the top of the sediment to adsorb or precipitate the nutrient have included oxidation to insoluble iron compounds or adsorption onto zeolites, bauxite refinery residuals, lanthanum modified bentonite clay, clay particles and calcite. Once again, the chemistry and other conditions can have an important effect on the success of these methods [5].

The use of commercial products for this purpose has recently become more widespread. The best known product is a lanthanum modified bentonite clay (‘Phoslock’) which was specifically designed to bind phosphorus in the clay and maintain it under most conditions encountered in aquatic systems [11]. Limited published results seem to indicate that Phoslock is effective under a range of environmental conditions including under reducing conditions. Issues to consider are dose rates and longevity of treatment depending upon local water chemistry conditions.

Link to phosphorus precipitation case study

CHEMICAL CONTROL OF CYANOBACTERIA

COAGULANTS

Coagulants can be used to facilitate the sedimentation of the cyanobacteria cells to the floor of the water body. Unable to access light, the cells do not continue to multiply, and eventually die. Some coagulants that may be used to coagulate cells include aluminium sulphate, ferric salts (chloride or sulphate), lime, or a combination of lime and metal coagulants. Although it has been reported that cells can be coagulated without damage, over a period of time the coagulated cells will become stressed and unhealthy, break open, or lyse, and release cyanobacterial metabolites [12]. Therefore, unless the coagulated cells are removed from the water body, this process will increase the dissolved toxins present in the water.

ALGICIDES

Algicides are compounds applied to the water body to kill cyanobacteria. As the injured or dead cells will rapidly lyse and release cyanotoxins into the water, this method is most often used at the early stages of a bloom, where numbers are low, and the toxic compounds released into the water can be removed effectively during the treatment process (see Chapter 5, removal of dissolved toxins). As with the application of any chemical to water destined for human consumption, there are a number of issues to be considered, including:

- Calculation of the required concentration to ensure the destruction of the cyanobacteria, with minimal residual of the chemical
- Effective application in terms of location and mode of dosing (e.g. from a boat, aerial spraying)
- The effect of dosing a potent chemical on the existing ecosystem in the water body
- Accumulation of the algicide in sediments
- Implications in the treatment plant of residual algicide (e.g. copper is coagulated in conventional treatment and may contaminate waste streams)

Chemicals that have been utilised as algicides are shown in Table 4-2 , along with key references which describe their properties and effectiveness.

Table 4-2 Algicides, their formulations and key references (after [13])

Compound	Formulation	References
Copper sulphate	$CuSO_4.5H_2O$	14, 15,16,17
Copper II alkanolamine	Cu alkanolamine.$3H_2O$ $^{++}$	18
Copper-ethylenediamine complex	$[Cu(H_2NCH_2CH_2NH_2)_2(H_2O)_2]^{++}SO_4$	18
Copper-triethanolamine complex	$Cu\ N(CH_2CH_2OH)_3.H_2O$	18
Copper citrate	$Cu_3[(COOCH_2)_2C(OH)COO]_2$	19,20
Potassium permanganate	$KMnO_4$	21 22
Chlorine	Cl_2	21
Lime	$Ca(OH)_2$	23
Barley straw		24, 25

COPPER BASED ALGICIDES

Copper based compounds are often used for chemical control of cyanobacteria. It is believed that the oxidative potential of the copper ion at high concentrations causes the cell membrane to rupture, thus lysing and destroying, the cyanobacteria cell. The effectiveness of copper as an algicide is determined by a combination of factors. Chemical parameters such as pH, alkalinity and dissolved organic carbon (DOC) of the receiving water control copper speciation and complexation, which affects copper toxicity. Thermal stratification affects the distribution of copper after application, which may then affect contact with the algae.

It is important to note the environmental impacts that copper dosing may have. Copper is known to be toxic to non-target organisms such as zooplankton, other invertebrates and fish [26]. It is also a bactericide, and may result in the destruction of various beneficial bacteria, including those that participate in the degradation of the cyanotoxins, once they are released. It is also known to accumulate in lake sediments and treatment plant sludge [27, 28]. In many countries there are national or local regulations to control the use of algicides due to their adverse environmental impacts.

Copper sulphate is the most commonly used of the copper-based algicides. Table 4-3 shows the relative toxicity of copper sulphate to several species of cyanobacteria.

Table 4-3 Relative toxicity of copper sulphate to cyanobacteria. Modified after Palmer [16].

Group	Very Susceptible	Susceptible	Resistant
Cyanobacteria	*Anabaena, Microcystis (Anacystis), Aphanizomenon, Gomphosphaeria, Rivularia*	*Cylindrospermum, Planktothrix (Oscillatoria), Plectonema*	*Nostoc, Phormidium*

A range of methods is available for copper sulphate dosing. The commonly used method involves applying dry granular copper sulphate alongside or behind powerboats. Copper sulphate can also be dosed by conventional aerial application similar to other agricultural chemicals. The method of application of copper sulphate may have important effects on copper dispersal and ultimately the toxicity and success of treatment. It is important to try to achieve the best possible coverage of the reservoir surface and avoid missing shallow, difficult to access, zones where cyanobacteria can accumulate. Figure 4-5 (a-c) shows copper sulphate dosing by boat.

Copper sulphate can also be used to manage benthic cyanobacteria once reservoir draw-down has occurred (Figure 4-5 (d)).

Figure 4-5 Copper sulphate dosing of a reservoir (a-c) and benthic cyanobacteria after reservoir draw-down d)

Recommendations for copper sulphate dosing techniques, including dose rate and application

The toxic component of copper sulphate is the cupric ion (Cu^{2+}). After dosing the effective concentration of the active component will depend on the water quality parameters mentioned above. For example, Cu^{2+} complexes readily with natural organic material present in all water bodies, which renders it much less effective as an algicide.

The problem of the reduced effectiveness of copper sulphate treatment in hard alkaline water has long been recognised [16]. Chelated copper algicides were developed to overcome the problems of the complexation and loss by precipitation of toxic copper under these circumstances. Examples of copper chelate algicides include copper ethanolamine, copper ethylene-diamine and copper-citrate (Table 4-2). The chemical properties and application rates for these algicides are given by Humberg *et al.* [18]. These chelated algicides are available as liquid formulations, and in some cases a granular form is also manufactured.

Copper citrate has been used as an algicide in the U.S. [19]. It is available either as a commercial preparation [29] or by simultaneously dosing copper sulphate and citric acid [19]. It is claimed that the use of citric acid as a chelating agent enhances the solubility of copper allowing it to remain in solution longer under alkaline conditions [30].

The chelated copper compounds are often more expensive than copper sulphate, however they may be more effective as they maintain Cu^{2+} in solution longer than copper sulphate. As with any chemical, the efficiency is highly dependent on the mode of application and the water quality conditions. Unfortunately, despite the relatively widespread use of chelated copper algicides the effect of water chemistry on their efficacy is poorly understood.

OTHER ALGICIDES

Potassium permanganate: A survey of North American utilities in the 1980s, indicated that a small number used potassium permanganate as an algicide in reservoirs [22]. Fitzgerald [22] found that the dose range required to control algae and cyanobacteria was in the range 1 - 8 mg L^{-1}.

Chlorine: Chlorine is used mainly for control of algae in water treatment works but has also been employed in reservoir situations [15]. The effective dose rates would obviously be dependent on the chlorine demand of the water, but most algae are reportedly controlled by doses of free chlorine between 0.25 and 2.0 mg L^{-1} [15].

Hydrogen peroxide: Hydrogen peroxide has been shown to selectively damage cyanobacteria over other planktonic species such as green algae [31]. Recently a range of stabilised hydrogen peroxide compounds have been developed in the US specifically to provide an alternative to overcome the environmental issues associated with copper algicides. Several manufacturers have now had these formulations added to the list of USEPA registered pesticides as algicides for use in drinking water reservoirs. The formulations contain solid granules of sodium carbonate peroxyhydrate which are directly applied to a water body releasing sodium carbonate and hydrogen peroxide. The hydrogen peroxide then degrades further into hydroxyl free radicals which are claimed to cause oxidative damage to cell membranes and to cell physiological processes.

ISSUES ASSOCIATED WITH ALGICIDES AND OTHER CHEMICAL CONTROLS

Before applying chemical controls against toxic cyanobacteria it is important to be fully aware of both the environmental and practical problems with their use.

The most commonly used algicide - copper sulphate, has a significant ecological impact. It should be used only in dedicated water supply reservoirs, and even then it is an unsatisfactory long-term solution. In many countries there are national or local environmental regulations which prohibit or limit the use of algicides due to their adverse environmental impact. This should be taken into consideration when developing management strategies for water sources.

As mentioned earlier, the disruption to the cell walls produced by algicides leads to the rapid release of the intracellular cyanobacterial metabolites. This can result in the diffusion of algal toxins throughout the water body within hours. Additional measures must then be applied within the treatment plant to remove these dissolved metabolites (See Chapter 5, removal of dissolved cyanotoxins). If possible, after algicide treatment, the reservoir should be isolated for a period to allow the toxins and odours to degrade. This is particularly important if the treatment is applied during bloom conditions. Unfortunately, it is difficult to advocate a minimum withholding period prior to recommencing use of the water body as the degradation of the toxin will depend upon local conditions (i.e. temperature, microbial activity); however, it could be in excess of 14 days [32]. A range of microorganisms have been shown to very effectively degrade several of the major cyanotoxins, including microcystins and cylindrospermopsin [33, 34]. However, the time taken for total toxin degradation varies widely from 3-4 days to weeks or months depending upon the circumstances [35]. Therefore, it is recommended that monitoring be undertaken to determine the amount of toxin remaining in the waterbody after treatment with an algicide.

Generally, microcystins are known to degrade readily in a few days to several weeks [33, 36]. Cylindrospermopsin has been shown to persist in the waterbody for extended periods and its degradation is dependent upon the presence in the reservoir of the microorganisms with the necessary enzymes for cylindrospermopsin degradation [34]. However, in water bodies where the cylindrospermopsin is found regularly degradation has been shown to occur relatively rapidly [37].

Saxitoxins have not been shown to be degraded by bacteria so, if a toxic bloom of *Anabaena circinalis* is dosed, it may be necessary to have water treatment strategies for dissolved toxin removal [38]. In addition, although saxitoxin appears to be non-biodegradable, it can undergo biotransformations involving conversion from less toxic forms to more toxic variants [39].

BIOLOGICAL CONTROLS

Cyanobacterial growth can be moderated by manipulation of the existing ecosystem in a reservoir or lake. Important aims can be to:

- Increase the numbers of organisms that graze on the cyanobacteria
- Increase competition for nutrients to limit the growth of cyanobacteria

Biomanipulation is often described as either “bottom up” (nutrient control) or “top-down” (increased grazing).

INCREASING GRAZING PRESSURE

The introduction of measures to encourage the growth of zooplankton and benthic fauna that feed on cyanobacteria can be effective in limiting cyanobacterial proliferation. Methods reported in the literature include:

- Removal of fish that feed on zooplankton and other benthic fauna, or introduction of predators to these fish
- Development of refuges to encourage the growth of the beneficial organisms [5]

ENHANCING COMPETITION BY INTRODUCING MACROPHYTES

In relatively shallow water bodies with moderate phosphorus concentrations the introduction of macrophytes can limit available phosphorus and therefore limit cyanobacterial growth. When other measures are also taken such as the control of fish types and numbers, the introduction of macrophytes to a water body may result in improved turbidity and lower cyanobacteria growth [5]. Figure 4-6 shows the introduction of water plants into a heavily contaminated water body in an effort to reduce nutrient levels and improve water quality.

Figure 4-6 Introduction of water plants into a heavily contaminated water body in an effort to reduce nutrient levels and improve water quality

OTHER BIOLOGICAL STRATEGIES

The potential of microorganisms such as bacteria, viruses, protozoa and fungi to control cyanobacteria has been studied on a laboratory scale. Although successful on a small scale, the full scale use of such measures has not been attempted due to a range of problems such as the difficulty culturing large numbers of microorganisms, and the ability of the cyanobacteria to become immune to infection [5].

ISSUES ASSOCIATED WITH IMPLEMENTATION

Biomanipulation is a very difficult management practice to implement, as many interacting factors influence the ecology of a water body. The deliberate modification of the biodiversity of the system may have unintended consequences for other organisms and water quality parameters. In addition, the ongoing implementation of such a strategy will require constant monitoring and adjustment, as it is likely that the system will tend to readjust to the original biological structure [5].

Click here for more detailed information on the manipulation of the foodweb to improve water quality

Click here to read a case study of biomanipulation

ALTERNATIVE METHODS

BARLEY STRAW

The use of decomposing barley straw for the control of algae and cyanobacteria has been the subject of considerable interest and investigation since the early '90s [24, 25, 40, 41]. Laboratory studies have suggested algistatic effects on both green algae and cyanobacteria. Several causes have been suggested for the observed effects, including the production of antibiotics by the fungal flora responsible for the decomposition, or the release of phenolic compounds such as ferulic acid and *p* - coumaric acid from the decomposition of straw cell walls [25]. While reservoir trials with barley straw appeared to confirm these laboratory observations [41, 42], other trials resulted in no observable effect [43, 44].

Because of its affordability and ease of use, barley straw is used in many reservoirs and dams in the United Kingdom with positive results. A fact sheet prepared by the Centre for Hydrology and Ecology, Natural Environment Research Council and the Centre for Aquatic Plant Management in the UK, details the application and mechanism of the effect of barley straw for the control of algae in a range of water bodies [45].

Although some water authorities have applied this method due to the low cost and appeal as a natural treatment, Chorus and Mur [5] do not recommend its use due to the possibility of the production of unknown compounds (possibly toxic, or odour-producing) and consumption of dissolved oxygen during the decomposition process.

ULTRASOUND

Ultrasound has been the focus of several studies. It has been found to limit the growth of cyanobacteria [46] as well as causing sedimentation due to disruption of the gas vesicles [47] depending on the energy and length of time of application. The observed effects are also dependent on the species of cyanobacteria [48]. The application of ultrasound was reported to successfully reduce the proliferation of cyanobacteria in a treated pond compared with a similar pond that was not exposed [49]. The study of ultrasound as a method of control for cyanobacteria is still in its infancy, and the technical hurdles involved in the application of this technology in a large water body are clear; however, further work may reveal it to be an effective, non-chemical control strategy.

CHAPTER 4 MANAGEMENT AND CONTROL IN SOURCE WATERS (LEVEL 2)

PHYSICAL CONTROLS

POSITIONING OF AERATORS IN A RESERVOIR

Destratification devices are usually placed near the offtake or dam wall in a deep area of the reservoir. It is possible to simulate reservoir destratification using a hydrodynamic-ecological numerical model to determine whether the destratifier will maintain cyanobacterial growth below 2000 cells mL^{-1} (for geosmin producing *Anabaena circinalis*) and dissolved oxygen (DO) at greater than 4mg L^{-1}. The one-dimensional hydrodynamic ecological model, DYRESM-CAEDYM, is ideal for this type of modelling. DYRESM-CAEDYM was developed by the Centre for Water Research and is available as free-ware from www.cwr.uwa.edu.au. Combinations of the various management options (e.g. no artificial intervention, aerator operating, surface mixers etc) can be simulated to determine which operational strategies would give the desired result of low cell numbers and increased DO. Informed operational strategies can then be implemented according to the results of the simulation.

Return to level 1

MIXING CASE STUDY – MYPONGA RESERVOIR

THE PHYTOPLANKTON COMMUNITY

This case study was derived from [50 and 58].

Current management at Myponga Reservoir in South Australia includes both artificial destratification and chemical algicides to control cyanobacteria. Although there are two different destratifying systems in Myponga Reservoir, there is still strong persistent stratification in the surface layer at particular times when high nocturnal temperatures and low wind speed inhibit cooling (Figure 4-1 (L2)). However, modelling studies have shown that the destratifiers have significantly reduced the period over which *Anabaena* can grow. The phytoplankton community in Myponga Reservoir is dominated by green algae and diatoms, which rely on turbulence to remain entrained, and the conditions when cyanobacteria grow have been narrowed to short periods each year (Figure 4-2(L2)).

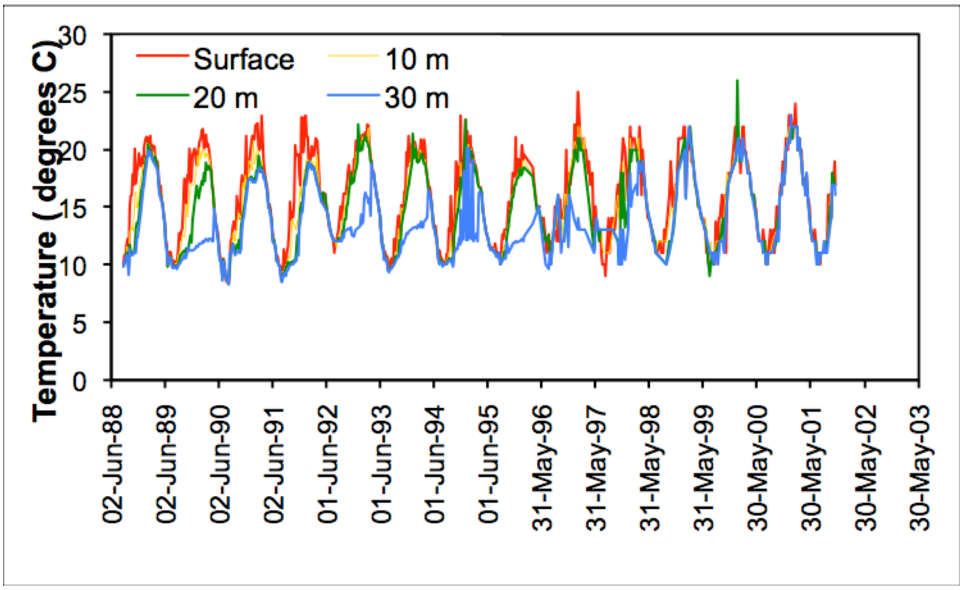

Figure 4-1(L2) Temperature measured weekly at the surface, 10 m, 20 m and 30m depth adjacent to the off-take point at Myponga Reservoir. The aerator was installed in 1994.

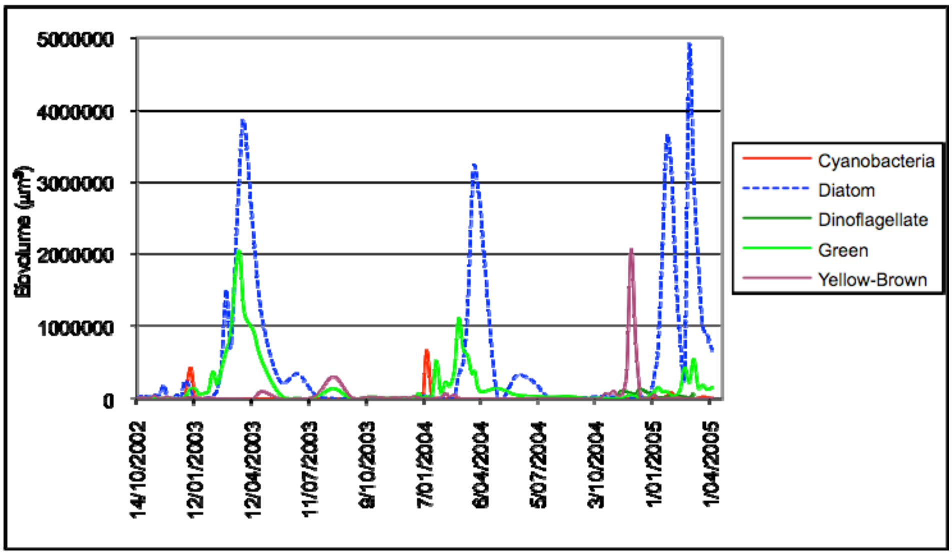

Figure 4-2(L2) Relative abundance of the different phytoplankton groups in Myponga Reservoir

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ARTIFICIAL DESTRATIFICATION TO CONTROL THE NUTRIENT LOAD

Seasonal temperature stratification was evident at Myponga Reservoir during summer from 1984 until 1994. Since installation of the aerator in 1994, close to isothermal conditions have been maintained at the sampling site (Figure 4-1(L2)). However, surface layer heating is evident at other sites in the reservoir outside of the immediate bubble plume, which is consistent with other reservoirs where bubble plume aerators are operating [56,52]. Dissolved oxygen concentrations were below 4 mg L^{-1} for extended periods during 1992/93 and 1993/94, which provided conditions suitable for contaminant resolubilisation. Since aerator operation in 1994 the dissolved oxygen concentration at 30 m has been maintained above 4 mg L^{-1}.

Prior to 1994 the concentration of filterable reactive phosphorus (FRP) at 30 m depth was consistently higher than the surface concentrations during summer and autumn (Figure 4-3(L2)). This coincides with the periods of extreme temperature stratification and low dissolved oxygen in the hypolimnion. Filterable reactive phosphorus at 30 m depth reached a maximum concentration of 0.259 mg L^{-1} in April 1986. The vertical gradient in FRP concentration has decreased since deployment of the bubble plume aerator and the large flux events have been eliminated.

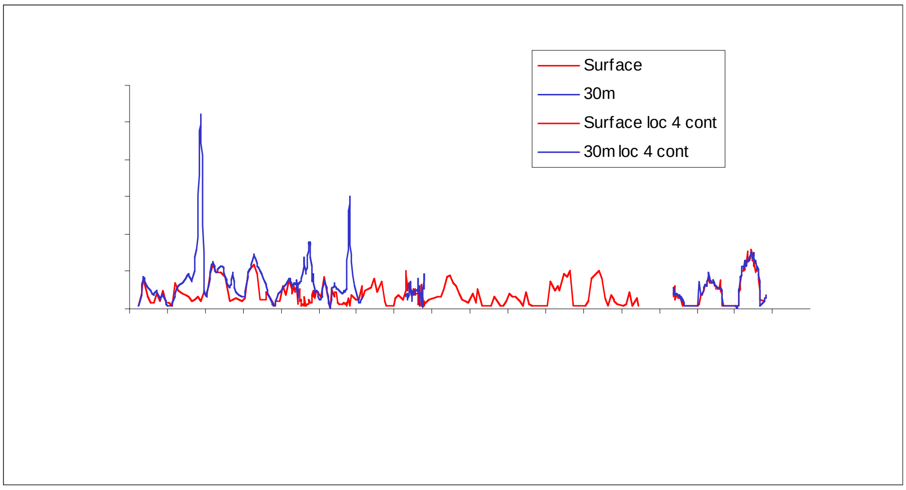

Figure 4-3(L2) Filterable reactive phosphorus at the surface and 30 m at Location 1 near the dam wall and from Location 4 from October 1998. Aerator installation decreased the internal nutrient load and high concentrations in the hypolimnion were not observed following aerator deployment.

RELATING NUTRIENTS TO ALGAL BIOMASS

In Myponga Reservoir the nutrient loading from the catchment occurs predominantly during winter and early spring. The nutrient pool is not utilised immediately as phytoplankton growth is limited by cool water temperatures and grazing pressure. As water temperature increases the phytoplankton grow rapidly and chlorophyll-a concentration increases with an associated decrease in FRP (Figure 4-4(L2)). FRP concentrations decrease to below the detection limit (0.005 mgL^{-1}) and the chlorophyll decreases some time later and the seasonal cycle is repeated.

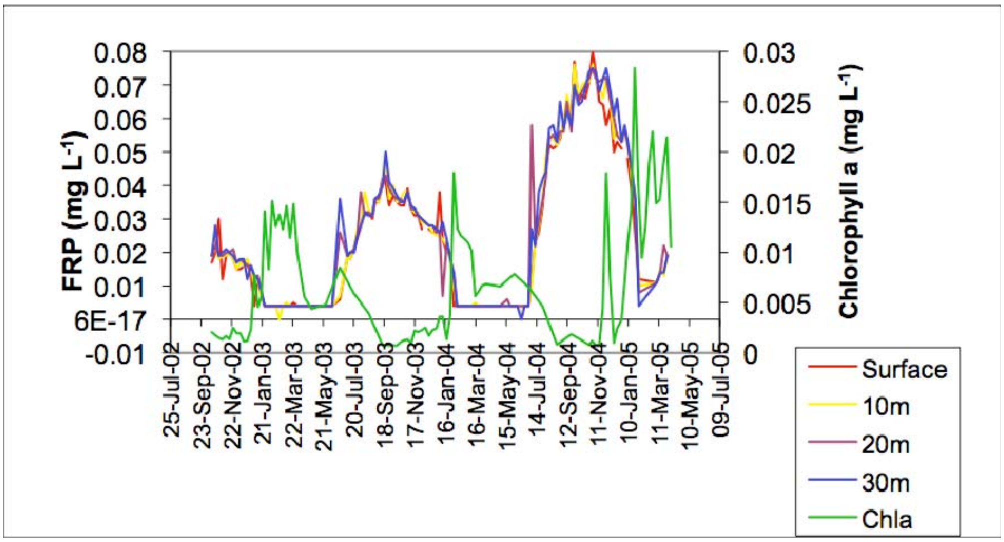

Figure 4-4(L2) Filterable reactive phosphorus at four depths and chlorophyll concentration integrated over the top 5m.

With the internal nutrient load largely controlled in Myponga Reservoir by the aeration system, the catchment is the dominant source of nutrients. In Myponga Reservoir two tributaries contribute the majority of the nutrient load, but loading is both seasonally and inter-annually variable. High inflow to the reservoir results in high total phosphorus (TP) loads and reservoir concentrations. A high maximum TP concentration in Myponga Reservoir results in a high chlorophyll-a concentration. Figure 4-5(L2) shows the relationship between the maximum annual TP concentration and the maximum chlorophyll-a found in the following growth period in the years between 1985 and 2000. Two outlier years, 1988 and 1993, are excluded from the regression in the figure. 1988 was an unusual year in that rains were early, and consequently there was a 6 month interval between the TP and Chl-a maximum. In 1993, hypolimnetic anoxia caused by thermal stratification, released higher than usual FRP concentrations from the sediments, sustaining high algal biomass and resulted in a high maximum chlorophyll-a concentration. The operation of the bubble plume aeration system since 1994 has most likely prevented this situation from recurring [53].

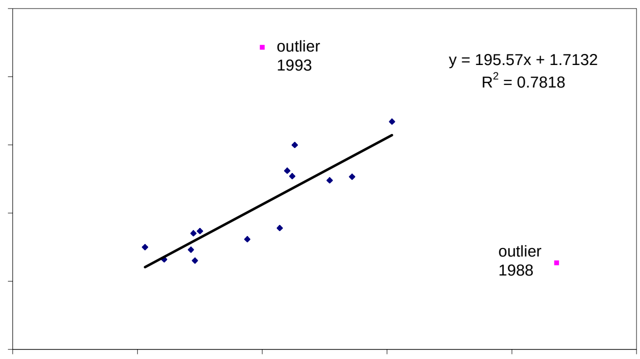

Figure 4-5(L2) Relationship between maximum total phosphorus and maximum chlorophyll *a* in the following growing period

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MODELLING ALGAL GROWTH

Because weather and limnological conditions are never constant it is difficult to determine whether destratification has an impact on cyanobacterial growth without very extensive historical data sets. An alternative approach is the use of numerical models to simulate the hydrodynamics and cyanobacterial growth. DYRESM-CAEDYM is a coupled hydrodynamic, water quality and algal growth model available as free-ware from the Centre for Water Research, The University of Western Australia (http://www.cwr.uwa.edu.au/). The modelling approach has been used in these studies to evaluate destratification in Myponga Reservoir. Meteorological variables measured at the stations on the reservoir were used for model inputs. Algal growth was simulated using equations describing nutrient and light-limited growth of *Anabaena circinalis* and floating velocity.

The DYRESM-CAEDYM simulation of the phytoplankton community was undertaken for the period September 2000 to March 2001. The observed and simulated total Chl-a concentrations are shown in Figure 4-6(L2). The simulated biomass captures the timing of the summer peak in the field data, but did not simulate the unseasonal peak that occurred in December 2000. This peak was attributed to the excessive growth of *Chroomonas* sp., a species which was not included in the model. The simulated growth of *Anabaena circinalis* from September-2000 to March-2001 produced a reasonable match with the observed field data (Figure 4-7(L2)), although the simulated growth started earlier in the season than observed in the field.

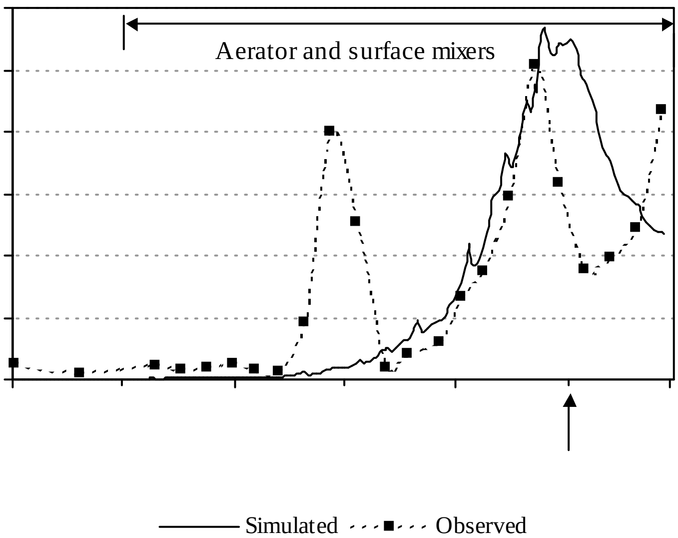

Figure 4-6(L2) Observed and simulated total Chl-a concentration (µg Chl-a L^{-1}), with simulated $CuSO_4$ dosing on 31-January-2000, and surface mixers and aerator operating between 1-October-2000 and 28-February-2001.

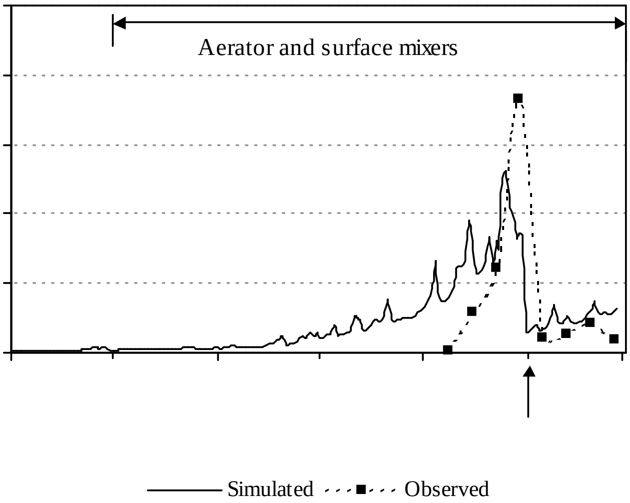

Figure 4-7(L2) Observed and simulated *Anabaena circinalis* concentration (µg Chl-a L^{-1}) from 1-September-2000 to 1-March-2001.

The simulation of the 3 types of phytoplankton that were representative of the assemblage in Myponga Reservoir from September 1999 to March 2001 produced reasonable results considering the limitations of the model. The observed phytoplankton community consisted of more than the three species simulated in this model. Other species will dominate with changes in nutrients, light and temperature as highlighted by the excessive growth of *Chroomonas*. An improvement to the CAEDYM model would be to increase the number of species simulated, although this would require intensive calibration. A trial and error approach as used in this study would be insufficient.

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SIMULATION OF VARIOUS MANAGEMENT STRATEGIES

The CAEDYM model output compared with observed field data gave a reasonable representation of phytoplankton biomass (as total Chl-a) for three species in Myponga Reservoir. The comparison between observed and simulated for *Scenedesmus* showed a strong correlation whereas a moderate correlation was observed with *Anabaena circinalis*. The next step involved using the model to determine the individual and combined impact of the surface mixers and the aerator for destratification and control of cyanobacteria. The following strategies were investigated for their ability to maintain DO greater than 4 mg L^{-1} and to limit *Anabaena circinalis* below 2,000 cells mL^{-1}.

1. No artificial intervention
2. Aerator and surface mixers with no $CuSO_4$ dosing
3. Aerator only
4. Surface mixers at measured flow rate (3.5 $m^3 s^{-1}$)
5. Surface mixers at design flow rate (5 $m^3 s^{-1}$)
6. Surface mixers at increased flow rate (8 $m^3 s^{-1}$)
7. Intermittent operation
8. Equivalent aerator energy input using surface mixers

Detailed results of this modelling can be found in [58].

The effectiveness of the various operational strategies used to limit the growth of *Anabaena circinalis* and maintain DO concentration in the water column is summarised in Table 4-1(L2). The simulation employed for validation, including the surface mixer, bubble plume aerator and $CuSO_4$ dosing algorithms, produced similar results to the observed field data. If no artificial mixing or $CuSO_4$ dosing were employed, excessive growth of *Anabaena circinalis* would occur and permanent stratification would lead to the presence of anoxic conditions. The use of the aerator without $CuSO_4$ dosing adequately maintained well-mixed conditions and DO throughout the water column. However, the growth of *Anabaena circinalis* could exceed 1,000 cells mL^{-1} (for a total of 16 days) but would not reach the threshold of 2,000 cells mL^{-1}.

When the aerator is coupled with the surface mixers (at 3.5 $m^3 s^{-1}$), the growth of *Anabaena circinalis* was further reduced with the peak concentration falling from ~ 1,400 cells mL^{-1} to ~ 1,000 cells mL^{-1}. The operation of the surface mixers (3.5 $m^3 s^{-1}$) alone would not be able to destratify the water column and maintain DO at acceptable levels, and importantly the growth of *Anabaena circinalis* would exceed 2,000 cells mL^{-1}. Increasing the flow rates of the surface mixers improves their destratification ability and reduces the growth of *Anabaena circinalis*. With a surface mixer flow-rate of 8 m^3s^{-1}, optimal results were achieved maintaining DO above 4 mg L^{-1} and limiting the maximum concentration of *Anabaena circinalis* to ~ 1,000 cells mL^{-1}.

Using intermittent mixing, the growth of cyanobacteria was restricted to a maximum concentration of ~ 700 cells mL^{-1} and well-mixed conditions were maintained. The use of $CuSO_4$ dosing would not be required under this strategy and operational costs would be lower due to the reduced use of the aerator and surface mixers. The use of 25 surface mixers, using the same energy as the existing aerator, adequately destratified Myponga Reservoir and almost completely inhibited the growth of *Anabaena circinalis*.

Table 4-1(L2) Results from existing and simulated water quality management strategies.

Artificial mixing operation	Maximum cyanophyte concentration ($cells.mL^{-1}$)	Days above 1000 $cells.mL^{-1}$	Minimum DO (mgL^{-1})	Simulated phytoplankton assembly composition		
				Chlorophytes	Cyanophytes	Diatoms
Existing - Field	1625	1	~5.00	96.30%	0.50%	3.20%
Existing - Sim	278	0	4.70	96.60%	0.70%	2.70%
Strategy 1	4444	196	1.00	91.30%	6.80%	1.90%
Strategy 2	1069	3	4.70	94.10%	2.90%	3.00%
Strategy 3	1389	16	4.70	92.90%	4.00%	3.10%
Strategy 4	2361	133	1.20	93.90%	4.70%	1.40%
Strategy 5	1556	21	4.70	95.30%	3.40%	1.30%
Strategy 6	1014	1	4.70	96.40%	2.40%	1.20%
Strategy 7	667	0	4.70	97.10%	1.70%	1.20%
Strategy 8	153	196	4.70	98.30%	0.60%	1.10%

The addition of the surface mixer and $CuSO_4$ dosing algorithms to DYRESM-CAEDYM enabled the phytoplankton succession and DO concentration to be adequately simulated and validated against observed field data for the period 1 September 1999 to 1 September 2000. This enabled various management strategies to be investigated. Modelling showed that the potential for growth of *Anabaena circinalis* would occur during periods of thermal stratification and with the presence of a shallow surface mixed layer. This coincided with oxygen depletion in the hypolimnion and adequate levels of nutrients (FRP> 0.01 mg L^{-1} and NO_x > 0.1 mg L^{-1}).

The actual mixing program with an aerator at Myponga Reservoir adequately maintains DO throughout the water column, and coupled with $CuSO_4$ dosing, limits the growth of *Anabaena circinalis* to a maximum concentration of ~ 1,600 cells mL^{-1} or 1.17 µg Chl-a L^{-1} (0.5% of the total biomass as Chl-a). The simulation of the existing aerator, surface mixers and $CuSO_4$ dosing produced similar results, affirming the need for intervention to maintain manageable levels of cyanobacteria and DO concentrations. The simulation showed that when the surface mixers and aerator are used without $CuSO_4$ dosing (strategy 2) the *Anabaena circinalis* would not exceed concentrations that would be of concern for water supply. The sole use of the surface mixers was found to be adequate at maintaining water quality if the flow rate could be increased to 8 m^3s^{-1}. However, at their current flow rate (3.5 $m^3 s^{-1}$) they are unable to fully destratify Myponga Reservoir and limit the growth of *Anabaena circinalis* to below 2,000 cells mL^{-1}.

The use of intermittent artificial mixing would reduce operational costs as the aerator and surface mixers would run at 50% less than the current operational schedule. Using this technique, destratified conditions are maintained, DO concentrations are kept high and the growth of *Anabaena circinalis* is minimal and importantly, the use of $CuSO_4$ dosing is not necessary. Under the current operating conditions, the simulation demonstrated that the use of $CuSO_4$ dosing is not necessary, as *Anabaena circinalis* concentrations did not exceed 2,000 cells mL^{-1}. As demonstrated with DYRESM-CAEDYM, the current nutrient concentrations, light climate, meteorological forcing and artificial mixing operations at Myponga Reservoir do not favour the excessive growth of *Anabaena circinalis*. However, even at these concentrations taste and odours can be problematic and require additional treatment.

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FACTORS INFLUENCING DESTRATIFICATION

The theoretical requirements for mixing to cause destratification and reduce cyanobacterial growth and biomass can be explained as follows. The reduction of cyanobacterial biomass is dependent upon the relationship between the depth to which the water column is mixed (Z_{mix}) and the depth of the penetration of light or photosynthetically active radiation (PAR, 400-700nm) into the water column. Light penetration is often described as the euphotic depth (Z_{eu}) which is the depth to which 1% of the subsurface irradiance penetrates [51]. The ratio between these depths can be used to evaluate the potential for light availability to limit the growth of phytoplankton which are circulating within the surface mixed layer. For example Z_{mix}:Z_{eu} ratios of 2.5 [3] or 3 [52] are regarded as ratios that will not support cyanobacterial growth. This means that the surface layer must mix to much deeper than light penetrates. Therefore, both the mixing and the clarity of the water column determine the Z_{mix}:Z_{eu} ratio. It follows that if a water body is inherently turbid or coloured it is theoretically more suitable to use mixing as a control technique than in clear water because the euphotic depth is shallower.

Artificial destratification has achieved good results in reducing iron and manganese problems for water treatment plants [53, 54], however the results in relation to the control of nuisance algae and cyanobacteria have been more variable [55]. This is most likely due to the complex interaction of the effects of destratification upon the availability of nutrients and light which are both required for the growth of photosynthetic organisms such as algae and cyanobacteria.

Destratification systems operating in deep reservoirs (mean depth >15m) have generally been more successful in changing the composition of the phytoplankton community [56, 3], while studies in shallower water bodies show less impact [57,52]. Even in deep reservoirs destratifiers may not be able to prevent the development of a stratified surface layer, outside of the immediate influence of the plume or mixer, which means that there is still a habitat for buoyant cyanobacteria to exploit [56].

It is likely that in situations where artificial destratification has failed to reduce cyanobacterial growth, neither nutrients nor light were limited sufficiently to impact on growth. Either there was a large enough external load to continue to supply adequate nutrients, and therefore limiting the internal load was inconsequential, or the artificial mixing was not adequate to light-limit the cyanobacteria.

A detailed description and comparison of the use of aerators and mechanical mixers to control cyanobacteria is provided in [58].

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CHEMICAL CONTROLS

PHOSPHORUS PRECIPITATION CASE STUDY

Taken directly from the USEPA website:

http://www.epa.gov/owow/lakes/kezar.html

Watershed Protection: Clean Lakes Case Study

Phosphorus Inactivation and Wetland Manipulation

Improve Kezar Lake, NH

EPA 841-F-95-002

Office of Water (4503F)

Kezar Lake, located in central New Hampshire, has had a long history of water quality problems. Following a major fish kill and persistent algae blooms beginning in the early 1960s, a Diagnostic/Feasibility Study (Phase I of the Clean Lakes Program) was initiated in 1980 under section 314 of the Clean Water Act. The study established that the lake's problems were from internal loading of phosphorus, and outlined a management strategy to restore the lake. Lake sediments, contaminated by years of effluent discharge from a nearby wastewater treatment facility, were the source of this internal loading.

A Restoration/Protection Project (Phase II of the Clean Lakes Program) commenced in 1984 to implement the recommended management strategy for Kezar Lake. Two main approaches were employed to reduce phosphorus concentrations in the lake. First, aluminum salts were injected into the hypolimnion to inactivate sediment phosphorus. The injections were performed using a modified barge system that was an efficient and cost-effective means of aluminum salts application. Second, upstream riparian wetlands were manipulated by elevating water level and planting new species to encourage phosphorus removal by sedimentation and vegetative uptake.

From 1984 to 1994, comprehensive water quality monitoring programs (including part of the Phase II project, a state-assisted volunteer program, and an EPA Phase III Post-Restoration Monitoring Project) were conducted to assess the effects of the restoration activities. Results from these efforts have generally indicated that water quality has improved following aluminum salts injection, although some parameters did worsen during 1988 and 1993. Furthermore, recreational use of Kezar Lake has increased substantially since restoration.

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RECOMMENDATIONS FOR COPPER SULPHATE DOSING

When determining the dose rate it is recommended to obtain the current pH, alkalinity and dissolved organic carbon (DOC) of the water to be dosed as these parameters will influence the action of the copper sulphate in the water as already mentioned. The conditions that will significantly reduce the toxicity of copper sulphate treatment are alkaline pH i.e. >7.5-8.0; high alkalinity i.e. > 40 mg L^{-1} as $CaCO_3$; and moderate to high DOC i.e. > 4 mg L^{-1}. Guidelines for copper sulphate treatment are given by Cooke at al. [1] .

To accurately determine the required dose rate it is useful to do a range-finding bioassay test with the target organism in the reservoir water to be treated. This is like a water treatment 'jar' test where cyanobacterial cells are treated with a range of concentrations of copper sulphate ($CuSO_4.5H_2O$) - for example 6-8 concentrations in the range from 0.01 to 0.5 mg Cu L^{-1}, and maintained at room temperature for either 24 or 48 hours. Subsamples are removed and either stained with cell activity stains and assessed by fluorescence microscopy and/or counted by conventional cell counts. This allows the calculation of the MLD_{100} or "Minimum Lethal Dose to 100% of cells" at the time end point you require – either 24 or 48 hours.

From this data the amount of copper required for the dosing can be calculated for the volume to be treated. In some cases for treating buoyant cyanobacteria it may only be necessary to dose a zone of the top 5m, which is approximately equivalent to the surface mixed layer in the reservoir. The majority of cells will be located in this layer if conditions are calm and stable and especially if the reservoir is stratified. It follows that if treatment is done under these conditions there is a greater of chance of achieving the maximum contact of toxic copper with the target cyanobacterium as the copper dissolves and disperses at a high concentration throughout the surface layer. Also when stratification is present, it is recommended to dose early in the day, as buoyant cyanobacteria are more likely to be at the surface of the water column. It is therefore beneficial to turn off any mixing or aerating apparatus prior to dosing with copper sulphate.

If treatment is done on a regular basis it is recommended that a procedure be developed to track and guide the boat using GPS, to move in a systematic pattern to achieve optimum coverage of the reservoir surface with the chemical.

Once a waterbody has been dosed with copper sulphate it is important to monitor the water for copper residuals, to ensure that guidelines for drinking water are not likely to be exceeded. For species of cyanobacteria known to be toxic or taste and odour producers, it may also be necessary to monitor for toxins, tastes and odours.

Figure 4-8(L2) shows a flow diagram of actions recommended for copper sulphate dosing.

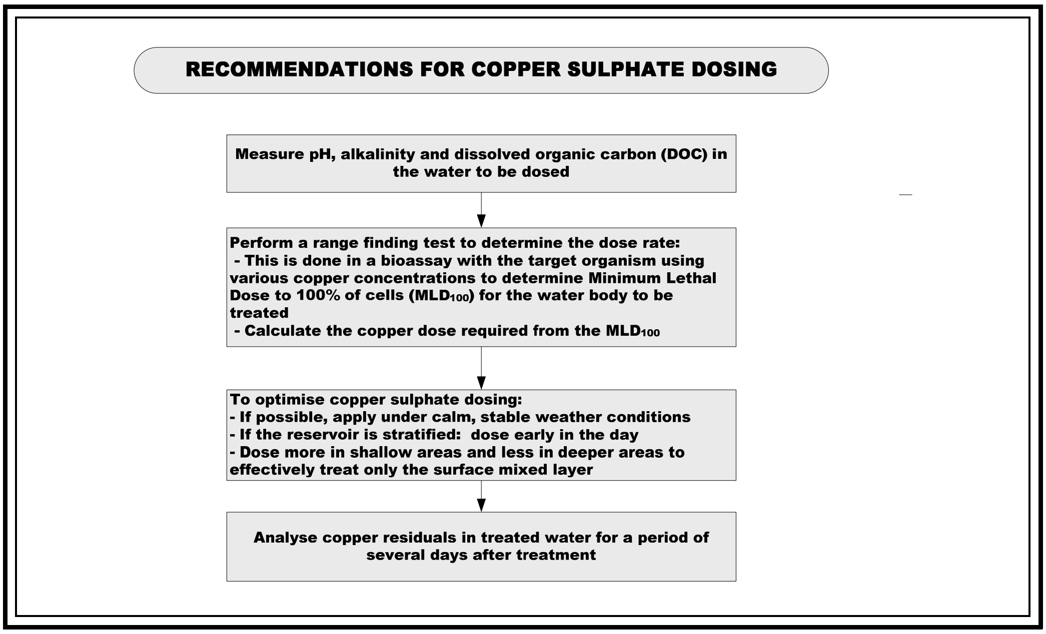

Figure 4-8(L2) Flow diagram for copper sulphate dosing: determining dose rates, application guidelines and follow-up monitoring

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BIOLOGICAL CONTROLS

IMPACTS OF MANIPULATION OF THE FOODWEB

Eutrophication problems that result in algal blooms, although commonly linked to a non-limiting supply of nutrients that support algal growth, may also occur as the result of other factors or a combination of factors. Various biophysical factors exacerbate the degree to which algal blooms occur or the frequency at which they occur. As the ambient concentration of phosphorus in reservoirs increases, so does the total biomass of fish. Research has shown that coarse fish, typically species of benthivorous and/or zooplanktivorous species – such as carp, barbell or canary kurper, tend to become dominant unless actively managed; this is also known as foodweb manipulation. Imbalanced fish populations results in an increased rate of availability of nutrients in the water column, via benthic disturbance and sediment resuspension - which also increases turbidity and decreases light availability- as well as via increased rates of excretion or recycling of nutrients into the water column. The same process also results in the uprooting of submerged macrophytes and hinders, or even precludes, the re-establishment of rooted macrophytes in disturbed sediments. The loss of macrophyte stability can force the system towards dominance by phytoplankton (see Figure 4-9(L2)).

In addition to impacts on the sediments and nutrient availability, high levels of zooplanktivore activity reduces the zooplankton within the reservoir foodweb, leading to destabilisation of the zooplankton-phytoplankton grazing dynamic. Current applied research in South Africa shows that these imbalances can be mitigated via a process of sustained and targeted foodweb management applied to the reservoir fishery [59].

These examples illustrate the value of knowing and understanding the key drivers that may be influencing the conditions in a particular waterbody. Figure 4-10(L2) broadly describes the major interactions occurring in a reservoir foodweb and, importantly, how an increase or decrease in any one or more may occur. This simple flowchart (Figure 4-10(L2)) allows the user to understand and/or determine the consequences of an action directed at one or more aspects of the water body's environment. The (+) or (-) signs on the directional arrows indicate the effect that the component has on the next. The effects are added in a multiplicative fashion – based on the mathematical relationships whereby a (+) multiplied by a (-) = (-) and a (-) x a (-) = (+).

For example: an increase in nutrients will cause algal levels to increase (+); an increase in numbers of benthivorous fish will increase the level of sediment resuspension (+), which in turn will increase nutrients and turbidity, and so on. Increased turbidity will have a negative (-) impact on vegetation, i.e. the increase will result in light reduction and blanketing, and reduce vegetation growth and coverage. So, an increase in turbidity has a negative (-) impact on vegetation, and in turn a negative (-) x (+) impact on the zooplankton which now have less vegetative habitat or cover available.

A second example: what would be the net impact on zooplankton of reducing nutrients: This would be (-) x (+) [effect of nutrients on algae] x (+) [effect of algae on turbidity] x (-) [effect of turbidity on vegetation] x (+) [effect of vegetation on zooplankton] = (-)x(+)x(+)x(-)x(+) = net positive effect on zooplankton. This would lead to more zooplankton which in turn would reduce (-) the levels of algae though grazing. Lastly, creating more aquatic vegetation habitat (the floating wetlands, littoral and riparian reedbeds, or protecting existing stands of pondweed) would support the development of a greater biomass of zooplankton able to graze on and reduce the algae, provided the fish grazing on zooplankton is in balance.

The flowchart includes options for assessing fishery, nutrient loading and bird management, in each case considering the management as reducing the impact caused by one or more. Waterfowl can have a profound and often unnoticed impact on nutrient loading (see chart), especially on smaller systems or on the shallow littoral environments in sheltered bays in large reservoirs.

By taking the time to assess as much information or knowledge about a particular reservoir or waterbody as possible, managers can make reasonable assessments of the likely drivers and knock-on effects using the flowchart. In many cases this will underpin a balanced management approach, as opposed to a single and often unsuccessful approach based on nutrients alone. Importantly, severe foodweb imbalances can produce impacts that have the appearance of nutrient-bolstered eutrophication.

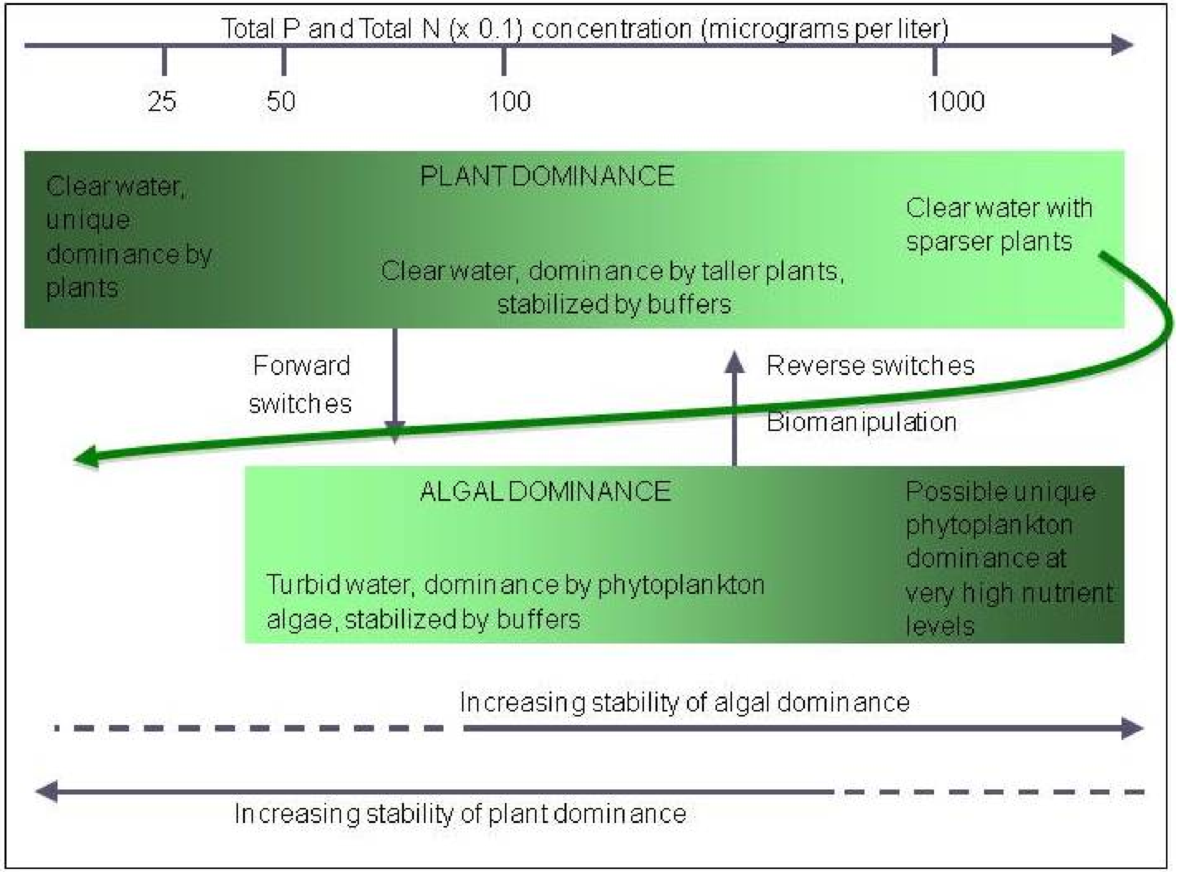

Figure 4-9(L2) Elements of the food web influencing cyanobacterial abundance

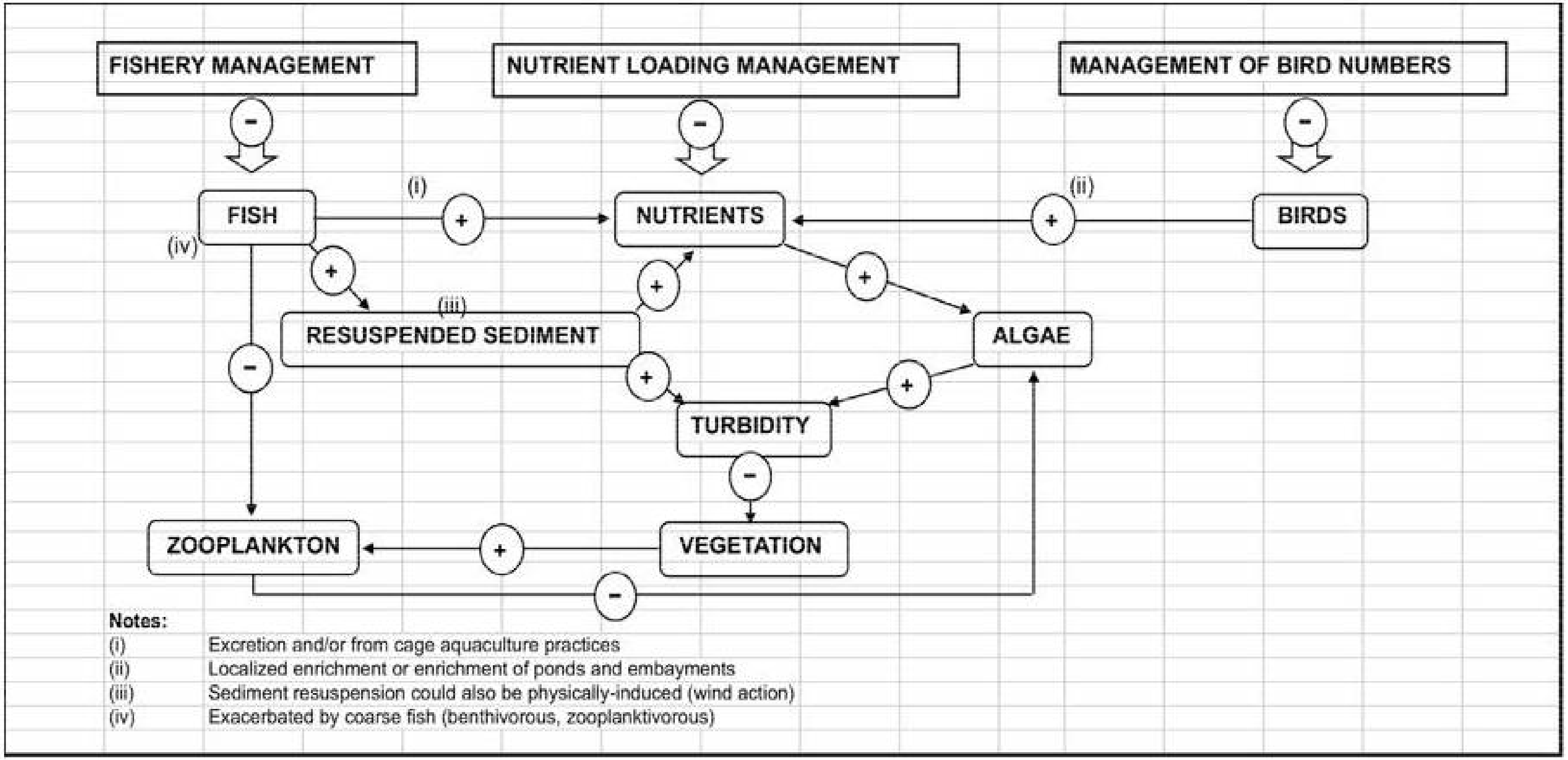

Figure 4-10(L2) Simple flowchart illustrating impacts of variation in various parameters on the aquatic environment

The flowchart is augmented by Figure 4-11(L2) – which broadly divides reservoirs into two types – the classically nutrient-driven case of eutrophication (Group 1) and where the situation is exacerbated by coarse fish dominance. This table broadly divides eutrophication problems into two areas (Group 1) the component that is associated with oversupply of nutrients originating from the catchment on a sustained basis and which is only effectively addressed at the catchment level; and (Group 2) that is associated with the long-term effects of poor impoundment management, with or without nutrient excesses, that has resulted in a gross disturbance in the foodweb - and in particular that associated with the change in the fish population from indigenous to rough fish. There are very few options for effective management available for large waterbodies falling within Group 1 other than short-term attention to the problem of algal blooms. An exception to this would be the case of a waterbody where the loading is primarily internal (external loading curtailed to manageable levels). In the latter case, and depending on the size of the waterbody, bottom sealing (physical or chemical) or dredging would now be a viable option - although perhaps expensive, the benefits would be both immediate and sustained.

Chemical in-lake controls might be effective in Group 1 waters where flushing rates are very low, especially during the summer in winter rainfall (Mediterranean) regions - where low availability of P during the summer, often geological in origin, could be reasonably offset by low level iron dosing during the latter months of the winter. Problems associated with Group 2 waters provide a genuine opportunity for effective in-lake control, even in the face of continuing nutrient loading from external sources. Obviously in such cases where external loading is not - or no longer - a problem, attention to foodweb restructuring offers significant potential for impoundment restoration/rehabilitation.

SYMPTOMS vs CAUSES ANALYSIS FOR IMPOUNDMENTS

COMMON SYMPTOMS		INDICATOR	PRIMARY CAUSES
GROUP 1			
N:P < 10-20	Increased phosphorus availability	Water chemistry	EXCESS NUTRIENT LOADING
Increased algal biomass	Phytoplanktonic or filamentous	Chlorophyll-a Water transparency	
Reduced algal diversity	Sustained dominance by few genera	Algal assemblage	
Cyanobacterial dominance	Colouration Blooms and/or scums	Visual Visual	
Increased frequency of algal blooms	In number per season or duration per event	Monitoring records	
Reduced water clarity	Organic origin (algal biomass)	Secchi depth	
Aquatic macrophyte dominance	Floating and/or rooted aquatic plants	Visual	
GROUP 2			
Increased inorganic turbidity	Sediment resuspension	Water transparency Turbidity measurements	COARSE FISH DOMINANCE
Increased P availability	Resuspension (fish, wind action) Excretion (fish, birds)	Water chemistry	
Reduced zooplankton dominance	Diversity, assemblage, body size	Monitoring	
Cyanobacterial dominance	Colouration Blooms Algal assemblage	Visual Visual Taxonomic	
Decrease in rooted macrophytes	esp. pondweeds	Spatial mapping	

Figure 4-11(L2) Symptoms and causes of various water quality issues in the water body

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CASE STUDY OF BIOMANIPULATION

In the late 1960s, Lake Veluwe, The Netherlands, displayed a transformation in its ecosystem from a macrophyte-dominated state when the total phosphorus levels exceeded 0.20 mg L^{-1}. The water of the Lake became turbid and remained so despite a significant reduction in the external nutrient load due to catchment management strategies. It was found that after these strategies were in place the Chl-a levels decreased, indicating a drop in levels of algal; however the light attenuation due to turbidity remained high due to the interaction of wind and benthivorous fish resuspending fine sediment particles. After several years macrophytes recolonised the shallower parts of the lake, resulting in localised clear water, while the deeper sections remained turbid. Once the causes were identified, a program to reduce the population of benthivorous fish commenced. This resulted in a recolonisation of the lake with zebra mussels, leading to further clarification of the water through filtration by the mussels. Finally this enabled the re-establishment of macrophyte species such as *Chara* and the clarification and rehabilitation of the entire lake [60].

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