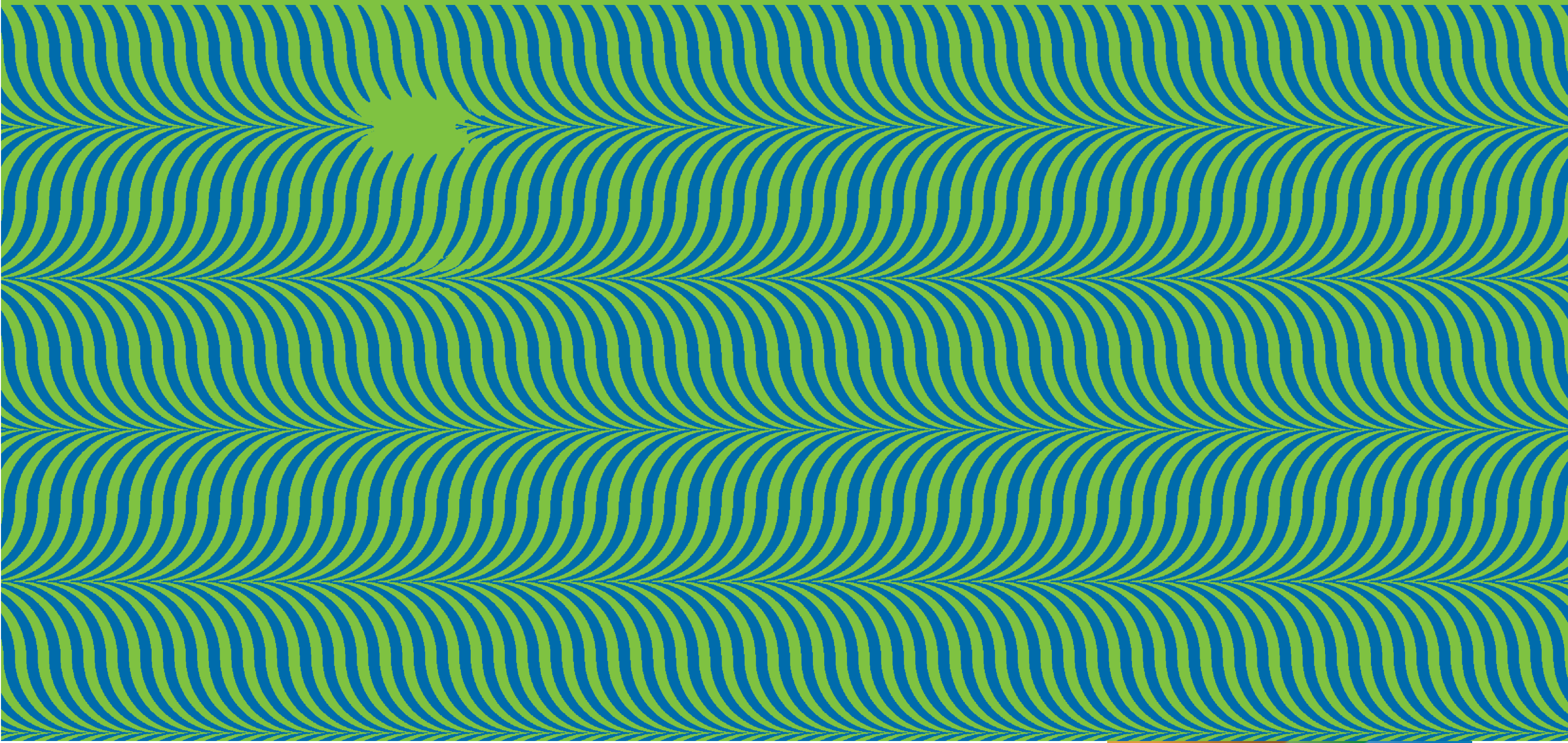
Urban Lakes Discussion Paper

Managing the Risk of Cyanobacterial Blooms

water by design



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Project Manager: Sarah Jones (Water by Design, HWL)

Authoring Team: Jason Sonneman, Andrew O'Neill and Shaun Leinster.

Water by Design

Water by Design is a program of Healthy Waterways Limited. Water by Design builds the capacity of the water and urban development sectors to help successfully implement sustainable urban water management

Healthy Waterways

Healthy Waterways is a collaboration between government, industry, researchers and the community. It was created as a Partnership in 2001. The partners work together to improve catchment management and waterway health in Moreton Bay and the rivers of South East Queensland between Noosa and the Queensland–New South Wales border. Healthy Waterways developed and implemented the *South East Queensland Regional Water Quality Management Strategy* (2001) and its successor, the *South East Queensland Healthy Waterways Strategy 2007–2012* (2008). Healthy Waterways also manages the Ecosystem Health Monitoring Program, which produces an annual report card on the health of the region's waterways, estuaries and bays.

Further information on Healthy Waterways and the Water by Design Program is available from www.healthywaterways.org and www.waterbydesign.com.au.

Contents

Definitions

For the purpose of this review, some definitions are provided to assist the reader:

Alternative states model – the potential existence of different biological communities within lakes under similar environmental conditions.
Anoxic – without oxygen.
Arfiticial lake or waterbody – see ‘Urban lake’**.**
Biofilms – living layer of microorganisms, generally dominated by microscopic algae, that covers the surfaces of sediment, rocks or plants.
Buffer – factors within a lake system that resist a switch between alternative states.
Constructed lake – see ‘Urban lake’.
Cyanobacteria – a group of single-celled organisms with a bacterial structure that contain chlorophyll and photosynthetic biochemistry similar to that of algae and higher plants.
Detention time – see ‘Retention time’.
Epilimnion – the warmer and well mixed upper waters of a stratified lake (i.e. the water layer above the thermocline in a stratified lake).
Euphotic zone – the part of the water column that receives sufficient sunlight for photosynthetic organisms and plants to make energy.
Eutrophication – process where there is an increase in the rate of addition of nutrients (nitrogen and phosphorus) to a system, typically resulting in excessive growth of algae or plants in response.
Hydraulic retention time – see ‘Retention time’.
Hypolimnion – the cooler and denser part of the water column below the thermocline in a stratified lake.
Internal Loading – accumulation of a pollutant in the lake sediments from an external source (e.g. untreated stormwater, wastewater, bird faeces).
Macrophytes – large, submerged or emergent, aquatic plants.
Photosynthesis – process where plants, algae and cyanobacteria use sunlight and carbon dioxide to generate carbohydrates and oxygen.
Phytoplankton – microscopic algae that are either suspended or floating within the water column.
Retention time – average time taken for the entire lake volume to be displaced via lake inflows and outflows. Commonly referred to as Hydraulic Retention Time (HRT), lake detention or residence time. The term ‘lake retention time’ has been used in this document as it is commonly used within the scientific literature.
Stratification – the separation of different layers of water due to different densities (in constructed lakes this is typically caused by a temperature gradient with cooler water being heavier and sinking to the bottom).
Switches – mechanisms which determine the ‘switch’ between alternative states.
Thermocline – the thin transition layer in a stratified lake which delineates the epilimnion and hypolimnion. The thermocline decreases in temperature rapidly and creates a physical barrier between the warmer mixed waters of the epilimnion and the cooler denser waters of the hypolimnion.
Urban lake - is defined as an artificial body of permanent water that may be fringed with emergent macrophytes or a formal edge such as a revetment wall or turf. While submerged macrophytes may occur throughout the water column, the dominant feature is open water (adapted from - Mackay City Council, 2008).

1 INTRODUCTION

South-East Queensland (SEQ) now has a large number of artificial waterbodies or “urban lakes” which were designed and built to support new urban development. The drivers for constructing lakes in urban areas are:

- Fill generation: Often very large amounts of fill are required to facilitate drainage on flat sites. Construction costs may therefore be reduced for the developer by excavating a lake on the development site.
- Amenity: It is perceived that developments with open water will attract higher land prices including premium prices for dwellings which have views of open water.

A growing body of knowledge amongst local government authorities (LGAs) in SEQ indicates that urban lakes experience water quality problems often leading to excessive algal growth or blooms, particularly of cyanobacteria (also referred to as blue-green algae). This is often a major management issue for constructed urban lakes and maintenance and rectification places a significant burden on LGAs in SEQ.

The rapid growth of cyanobacteria under optimal growth conditions means that cyanobacterial populations can rapidly reach bloom concentrations. Cyanobacterial blooms are often associated with eutrophication, however the development of blooms is a natural phenomenon and blooms can also occur in relatively undisturbed lakes (Moss, 2010). Urban lakes typically lack the resilience of natural systems, and tend to bloom more frequently and for longer periods of time. Urban lakes that regularly suffer persistent cyanobacterial blooms generally reflect an imbalance amongst the various hydrologic, chemical and biological processes occurring within the lake.

Cyanobacterial blooms adversely impact lake ecosystems and reduce public amenity and aesthetics. Negative ecological impacts may include shading, increased pH, decreased dissolved oxygen due to excessive respiration or degradation, and the production of cyanotoxins that are a risk to public health (Chorus and Bartrum, 1999). Cyanotoxins are produced by various cyanobacterial species mainly within the genera *Microcystis*, *Anabaena*, *Oscillatoria* and *Nostoc* (Sivonen, 1996). The release of cyanotoxins during and immediately following cyanobacterial blooms further degrades lake water quality which may lead to fish kills, animal death, and risks to public health. The cyanobacterial species of most concern in Australia are *M. aeruginosa*, *Nodularia spumigena*, *A. circinalis* and *Cylindrospermopsis raciborskii* (Steffensen *et al*., 1999). All of these species are capable of producing cyanotoxins and have been observed to regularly bloom in freshwater ecosystems within Queensland (L. Fabbro, pers. comm.).

A wide range of management strategies are currently used to control cyanobacterial blooms in lake ecosystems, such as in-lake sediment treatments (Cooke *et al*., 2005; Hart *et al*., 2003), physical in-lake measures (Nurberg, 1987; Reynolds *et al.*, 1987), biological control (Reynolds, 1994) and algicides (Burge *et al.,* 2002). However none of these strategies offer a cost-effective or sustainable option for managing cyanobacterial blooms in constructed urban lakes.

More recently, the management of lake retention times has been proposed as a means of reducing the risk of cyanobacterial growth in freshwater constructed urban lakes. This document provides a review of the potential effectiveness of manipulating lake retention times to reduce the risk of cyanobacterial growth in freshwater urban lakes. The review is broken into the following sections:

- **Section 1**. Introduction
- **Section 2.** Urban lake health: A summary of the issues that affect urban lakes.
- **Section 3.** Alternative states model: A discussion of a model that improves our understanding of how urban lakes operate.
- **Section 4.** Factors that influence cyanobacterial growth: A brief review of the key factors that influence the growth of cyanobacteria.
- **Section 5.** Retention time design criteria for managing cyanobacterial growth: Investigation into the role of retention time in the deterioration of urban lake health.
- **Section 6.** Use of constructed wetlands to manage lake retention times: A review of the strategy adopted in several key guidelines in Australia to recirculate lake water through constructed wetlands.
- **Section 7.** Alternative strategies for managing cyanobacterial growth: A range of existing and alternative strategies for managing cyanobacterial growth in lake systems will be reviewed.
- **Section 8.** Conclusions.

2 URBAN LAKE HEALTH

Constructed lakes within urban catchments are subject to a large number of stressors which ultimately impact the health of the lake ecosystems. Stormwater runoff from urban catchments delivers increased pollutant (nutrient and sediment) loads resulting in poor water quality, increased turbidity, increased siltation and highly organic sediments. These factors combined with inappropriate hydrologic regimes often result in the disappearance of aquatic vegetation from urban lakes (i.e. submerged macrophytes) and an increase in algal blooms.

A large number of urban lakes have been constructed in SEQ, the great majority of which have been poorly designed, with excessive depth (generally > 3 m), low hydraulic efficiency and long hydraulic retention times resulting in poorly mixed waters and often, the establishment of stratified conditions throughout the year. When stratification occurs, mixing of the warmer, well-oxygenated waters nearer the surface (epilimnion) and the cooler, poorly-oxygenated waters deeper in the water column (hypolimnion) is prevented by a strong temperature gradient (thermocline). Anoxic conditions develop within the hypolimnion, resulting in the release of soluble nutrients from the sediments and via anaerobic decomposition of organic carbon. This often leads to the development of phytoplankton blooms, particularly cyanobacteria, many of which are able to regulate their buoyancy and gain access to both the nutrient-rich hypolimnion and the well-lit upper waters (euphotic zone) of the urban lake. The development of phytoplankton biomass associated with increased nutrient loading can be exacerbated in warm temperate and tropical areas where an extended growing season and higher water temperatures prevail.

The elevated nutrient concentrations and stable water conditions found in many urban lakes also favours the growth of floating aquatic weeds such as salvinia (*Salvinia molesta*), water hyacinth (*Eichhornia crassipes*) and water lettuce (*Pistia stratiotes*) (Scheffer *et al.,* 2003). The presence of floating aquatic vegetation (including native and introduced waterlilies) provides a continual supply of carbon to the lake sediments (Plate 1). This results in a strong feedback loop whereby the decomposition of organic carbon within the lake sediments can deplete oxygen within the water column and sediments, resulting in the release of soluble nutrients back to the water column.

Plate 1 Extensive floating aquatic weed coverage (water lettuce & water hyacinth) of an urban lake in Mackay

In addition to high nutrient concentrations, the water column of many shallow urban lakes is characterised by high turbidity. The concentration of suspended solids within the water column is dependent upon inputs of suspended solids and the dynamic relationship between the processes of sedimentation and resuspension. Where submerged or emergent macrophytes dominate the water column, sedimentation is the dominant physical process affecting sediment particles in the water column. In systems where there are insufficient submerged or emergent vegetation or where a thermocline is lacking to trap sinking particles, the balance is shifted toward resuspension. A shift towards resuspension of sediments results in an additional negative feedback loop for the establishment of submerged macrophytes through inappropriate conditions in the sediments for seed germination and root anchorage. Consequently, the addition of suspended solids from stormwater runoff, the associated accumulation of nutrients and organic carbon within lake sediments coupled to a shift away from sedimentation has resulted in the gradual decline in health of many urban lakes within SEQ.

The concept that the health of shallow lakes is dependent upon turbidity and the associated health of submerged macrophytes has been the focus of a number of eminent researchers. This research suggests that the health of shallow lake systems is characterised by two alternative states:

1. Clear water - dominated by submerged macrophytes (hereon referred to as macrophytes), or
2. Turbid water - dominated by mixed phytoplankton (hereon referred to as phytoplankton).

Moss (1998) proposed an alternative states model for shallow lakes whereby nutrient-poor, shallow lakes are expected to be dominated by macrophytes, and nutrient-rich shallow lakes are expected to be dominated by phytoplankton (Figure 1). The model suggests that both macrophyte-dominated and phytoplankton-dominated states can exist as alternatives over a wide range of nutrient concentrations. Each alternative state is stabilised by a range of buffer mechanisms, however 'switches' are required to trigger a shift from one state to the other (Moss, 1998). More recent work has shown that the phytoplankton-dominated state may also exist as two alternative states: a cyanobacterial-dominated state versus a phytoplankton- (non-cynaobacterial-) dominated state (Scheffer, 1998).

Scheffer (1998) proposed that light availability is one of the major determinants (switches) governing whether shallow lakes switch between macrophyte- and phytoplankton-dominated states, or between phytoplankton- and cyanobacterial-dominated states. A detailed review of the alternative states model for shallow lakes, including the dominant switching mechanisms is provided in Section 3.

It should be noted that the presence of floating aquatic plants (ie. *Salvinia molesta*) represents a third alternative state to macrophyte- or phytoplankton-dominance (Scheffer *et al*., 2003, Figure 1). Generally, the dominance of floating plants is related to the presence of high nutrient levels and their ability to out compete macrophytes for light resources (Scheffer *et al.,* 2003). Floating plants depend upon the availability of high nutrient concentrations in the water column as they have no direct access to the sediment pool of nutrients.

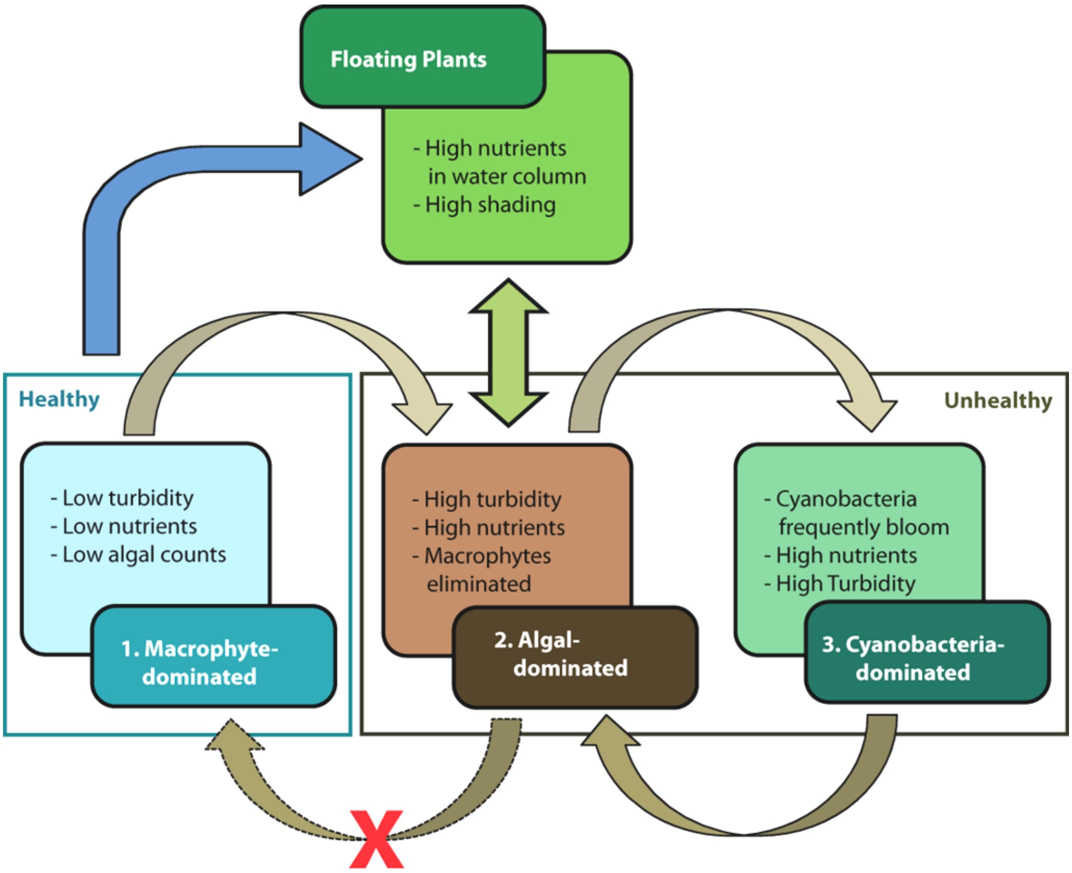

Figure 1 Conceptual model of the alternative states observed in shallow lakes.

Scheffer *et al.* (2003) proposed that submerged macrophytes and free floating plants are able to co-exist under low nutrient concentrations. Submerged plants may influence the growth of floating plants by competing for and reducing the available nutrients (especially nitrogen) within the water column.

As nutrient concentrations within the water column increase, a threshold is reached where the system moves to a floating plant-dominated state. Once developed, the floating plant-dominated state is able to persist as the severe light attenuation in the water column enables the floating plants to out-compete the submerged macrophytes and phytoplankton.

Under floating plant-dominance, the water column stabilises and anoxic conditions develop as oxygen is prevented from penetrating the surface and the dead floating plant biomass is decomposed by bacteria. The development of anoxic conditions results in the further release of nutrients from the sediments and the continual supply of nutrients to the water column.

Owing to the reasons outlined above the switch to floating plant-dominance is a largely irreversible process (Scheffer and van Nes, 2007). The key message is that the urban lake "health continuum" is heavily weighted towards being unhealthy. As the focus of this review is upon the management of cyanobacterial blooms within urban lakes, the role of floating plants within the alternative states model will not be discussed any further.

3 ALTERNATIVE STATES MODEL

3.1 MACROPHYTE – PHYTOPLANKTON

The alternative states model suggests that under certain conditions, shallow lakes can be in either two states: a macrophyte-dominated state with clear water and a turbid state dominated by phytoplankton (Moss, 1998, Scheffer, 1990). According to the alternative states model, the two states represent alternative equilibria that may exist over an intermediate range of nutrient conditions.

Scheffer (1998) proposed a conceptual model (derived from the field observations) that can be used to illustrate the shift between macrophyte-dominated to phytoplankton-dominated states (Figure 2). Scheffer's conceptual model is based upon three assumptions:

1. The turbidity of lake water increases with increasing nutrient concentrations
2. Macrophytes reduce turbidity, and
3. Macrophytes disappear entirely when a critical turbidity threshold is exceeded.

As nutrient concentrations increase within a lake, macrophyte biomass increases and the plants start to fill the water column or concentrate their biomass within the upper water layers. The presence of macrophytes provides a stabilising mechanism that helps to maintain low turbidity, thereby providing a positive feedback and enhancing their own growing conditions (Scheffer *et al.*, 2001). Scheffer (2001) showed that in the macrophyte dominated state, low turbidity levels are maintained in lakes due to):

- prevention of sediment resuspension (see above),
- uptake of nutrients from the water column by the macrophytes (high competition for nutrients), and
- provision of refugia for zooplankton against predators.

It has been noted that macrophyte-dominated lake systems often remain extremely stable when exposed to substantially increased nutrient concentrations (Morris, *et al*., 2003a, 2003b; Moss, 1998; Scheffer, 1998). Macrophyte-dominated lakes are therefore considered be reasonably robust to moderate eutrophication.

The conceptual model depicted in Figure 2 clearly indicates that the two alternative states can exist across a range of nutrient (or turbidity) concentrations (P1 – P2). Increases in nutrient concentrations can lead to both a gradual increase in phytoplankton biomass and an increasein algal biofilms growing upon the macrophytes. This may initially occur slowly as the submerged macrophytes buffer small increases in nutrients through uptake. Ultimately, as nutrients and turbidity (suspended solids, phytoplankton) increase or biofilm growth exceeds a critical threshold (E_{crit}), the macrophytes collapse due to decreasing light availability. Once E_{crit} is exceeded, the switch from a macrophyte- to phytoplankton-dominated state due to eutrophication occurs rapidly and tends to be catastrophic (Scheffer, 2001).

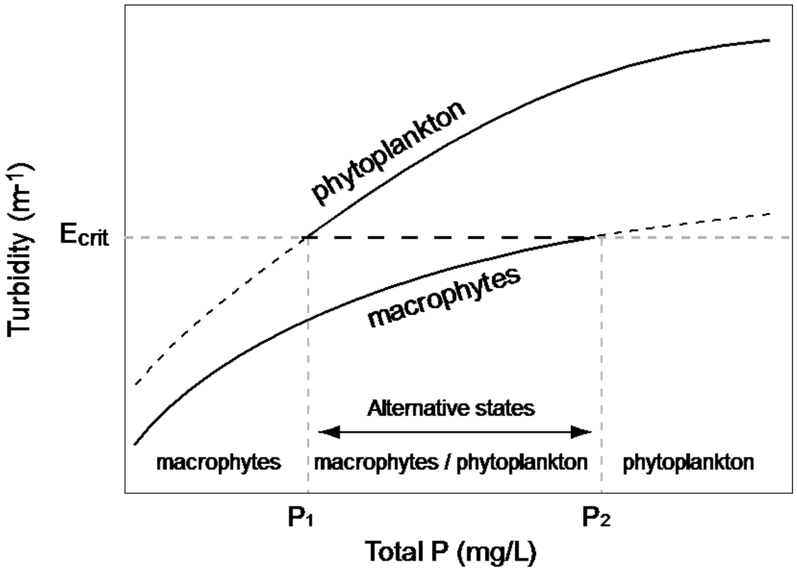

Figure 2 Conceptual model illustrating the transition from macrophyte to phytoplankton dominated states (adapted from Scheffer 1998).

Therefore, at lower nutrient concentrations, only a macrophyte-dominated equilibrium exists, whereas at the highest nutrient concentrations, there is only a phytoplankton equilibrium (Figure 2).

The establishment of a phytoplankton-dominated state is also characterised by a distinct feedback mechanism, whereby the recovery of submerged macrophytes is inhibited due to turbidity-mediated light limitation

(Scheffer, 2001). In the phytoplankton-dominated state, turbidity levels are maintained due to (Hargeby *et al*., 2007; Moss, 1998; Moss *et al*., 1996; Scheffer, 2001):

- Wave resuspension of unprotected sediments
- Maintenance of structureless habitat with no refuge for zooplankton
- High production of small algal species with high capacity for light absorption
- Anaerobic sediments that are no longer stabilized by plant roots
- Production of amorphous, high water content sediment that is unsuitable for plant regeneration.

The recovery of submerged macrophytes following a phytoplankton-dominated state is highly problematic due to the continued maintenance of high turbidity within the lake (Scheffer, 2001). Lakes that have turned turbid are often observed to resist restoration efforts (Scheffer and van Nes, 2007).

As noted above, both clear and turbid states in shallow lakes are stabilized by a number of mechanisms. A reduction in nutrient concentrations is often insufficient to restore macrophytes, and recovery often only occurs at substantially lower nutrient concentrations than those at which the collapse of the macrophyte vegetation occurred (Scheffer *et al*., 2001). This occurs when nutrient concentrations are reduced to a level where phytoplankton growth is limited enough by nutrients alone for turbidity to fall below the critical threshold for macrophytes to re-establish (Scheffer and van Nes, 2007, van Nes *et al*., 2007). Furthermore, it is not possible to restore submerged macrophytes in severely polluted lakes unless the external sources of nutrients are removed and internal nutrient loadings considerably reduced (Qiu *et al*., 2001).

For example, macrophytes disappeared from Lake Veluwe when Total Phosphorus (TP) concentrations exceeded 0.2 mg.l^{-1} , and re-established when TP concentrations decreased below 0.1 m g.l^{-1} (Ibelings *et al*., 2007). In this case the shallowest areas of the lake were initially re-colonised by macrophytes leading to isolated areas of clear water. Ultimately, the area of clear water expanded to the entire lake as the macrophytes re-established including the areas outside the macrophyte beds.

Some shallow lakes are inherently unstable and switch repeatedly back and forth between a vegetated clear-water state and a contrasting phytoplankton-dominated turbid state (Hargeby *et al*., 2007, Zimmer *et al*., 2009, Sayer *et al.*, 2010a, van Nes *et al*., 2007). There are many possible mechanisms by which a lake may switch between states. Some of these include:

- **Macrophyte dominance**. van Nes *et a*l. (2007) suggested that mechanisms associated with internal nutrient processes influences the shift between the two states. Under a macrophyte-dominated state, a net retention of phosphorus occurs due to uptake by macrophytes (and algae in the water column). The gradual accumulation of organic matter (i.e. macrophyte and algae biomass) within the lake sediments may result in high sediment oxygen demand, leading to anaerobic conditions and the subsequent release of phosphorus from the sediments. Thus, it is possible that macrophyte dominance may under certain conditions enhance organic accumulation and promote the release of phosphorus from the sediments.
- **Vegetation type**. The alternation between vegetated and turbid states may also be facilitated by the types of submerged or emergent vegetation present within the lake. Submerged macrophytes such as *Potamogeton spp*. senesce during winter, leading to enhanced organic loading of the sediments.
- **Decreased biodiversity**. Sayer et al. (2010b) observed that eutrophication of shallow lakes often results in decreased macrophyte species diversity. The loss of macrophyte species diversity may have large consequences for lake stability, particularly when there are less plant seasonality strategies present. The loss of a macrophyte species may reduce the effective plant growing season in a lake, and be sufficient to trigger the shift to a phytoplankton-dominated state (Sayer et al., 2010b).
- **Waterbirds**. High abundances of waterfowl may affect the stability of a lake ecosystem substantially by feeding on macrophytes and through deposition of faeces (Rip et al., 2006). Generally, waterfowl grazing does not impact overall macrophyte cover in shallow lakes (Hansson et al., 2010). However, during periods of low macrophyte biomass or when the lake is in a transition between alternative stable states, grazing of the macrophytes by waterfowl may negatively impact the overall macrophyte biomass and trigger the transition to a phytoplankton dominated state (Hansson et al., 2010).
- **Hydrologic influences**. Recent studies have indicated that hydrological factors may regulate the stability of tropical shallow lakes (Loverde-Oliveira et al., 2009). For example, submerged macrophytes were observed to grow rapidly during the high water period (wet season), followed by a sudden collapse of the plant communities and increase in turbidity as the water levels decreased during the dry season. Loverde-Oliveira et al. (2009) suggested that the shift between alternative states was driven by hydrology, whereby water transparency increased with the increased water levels due to flushing of algal biomass from the

lake. As the water levels decreased during the dry season, exposure of the submerged plants to direct sunlight and desiccation then triggered the die-off.

- **Water level variation**. These observations on hydrologic influences contrast with the patterns typically observed in temperate shallow lakes where the reduction of water levels is often accompanied by a shift from turbidity to macrophyte dominance (Scheffer, 1998). Low water levels may stimulate submerged macrophyte expansion owing to increased light penetration to the lake bed, and enhanced germination rates of macrophytes from seed beds. Therefore, the maintenance of artificially stabilised water levels (typical of urban lakes) and lack of alternating high and low water levels may substantially influence the long-term health of submerged macrophytes (Van Geest et al., 2007).

3.2 PHYTOPLANKTON – CYANOBACTERIA

Scheffer (1998) proposed a conceptual model (derived from the empirically-derived patterns observed in the field) that can be used to illustrate the relationship between shading (turbidity), phytoplankton and cyanobacterial dominance in shallow lakes (Figure 3). This model suggests that turbidity in shallow lakes will increase with increasing phosphorus concentrations, starting with a background level (E_b) and levelling off at high phosphorus concentrations when light becomes limiting (Figure 3).

Scheffer et al., (1997) suggested that for a well mixed lake, a simple assumption can be made that a critical turbidity level exists (E_{crit}), above which cyanobacteria will become dominant. The model suggests that there is a range of intermediate nutrient concentrations (P1 – P2) where both cyanobacterial or non-cyanobacterial algal communities may dominate, thus producing what is known as the S-shaped curve of steady states (Scheffer, 1998). The S-shaped pattern (due to the presence of the intermediate segment) is typical of ‘catastrophic’ lakes systems where algal communities can exist in either state (+/- cyanobacterial dominated) (Scheffer et al., 1997). The ability of some shallow lake systems with alternative states to remain in one state over a range of conditions (e.g. nutrient levels) is called hysteresis. Based upon his alternative state model, Scheffer (1998) cited four important characteristics of cyanobacteria that differentiate them from other algae. They:

1. have lower maximum gross productivity
2. experience a lower loss rate
3. exhibit a higher shade tolerance
4. cause a higher turbidity per unit of biomass than other algae.

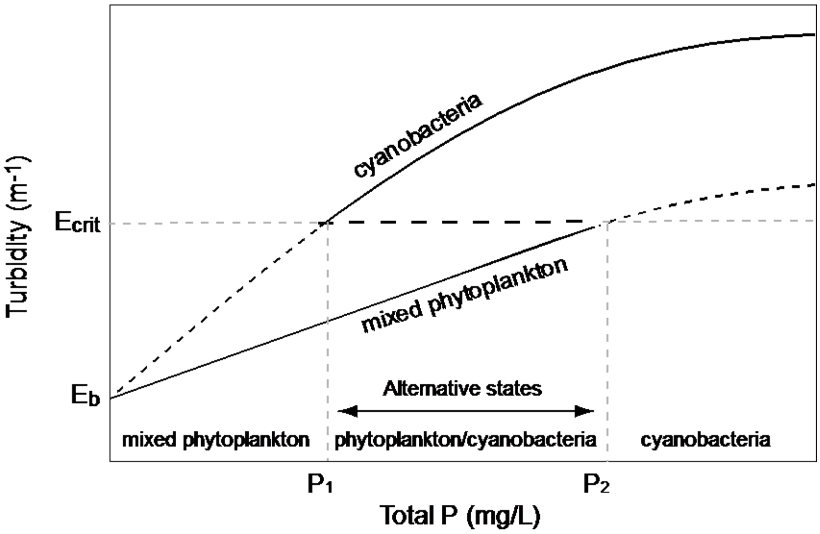

Figure 3 Conceptual model illustrating the equilibrium states of algal communities in shallow lakes (adapted from Scheffer *et al.*, 1997).

The concepts associated with the alternative state model can be extended to considering how turbidity in shallow lakes impacts upon algal community composition. Background levels of turbidity in shallow well mixed lakes can be high due to the frequent re-suspension of particles or from suspended solids contributed by stormwater inflows. In such lakes high turbidity can exist, even at lower nutrient concentrations, and the algal communities are often dominated by cyanobacteria.

As noted above, the regulatory mechanisms involved in the switch between alternative states for both macrophytes and phytoplankton, and phytoplankton and cyanobacteria are quite comparable. Combining the two conceptual models enables a picture to emerge of three alternative equilibria within shallow lakes (Figure 4).

Such a conceptual model has important implications for the potential responses of lake systems to nutrient loading. It can be seen that changes between alternative states is dependent upon the critical turbidity levels at which the

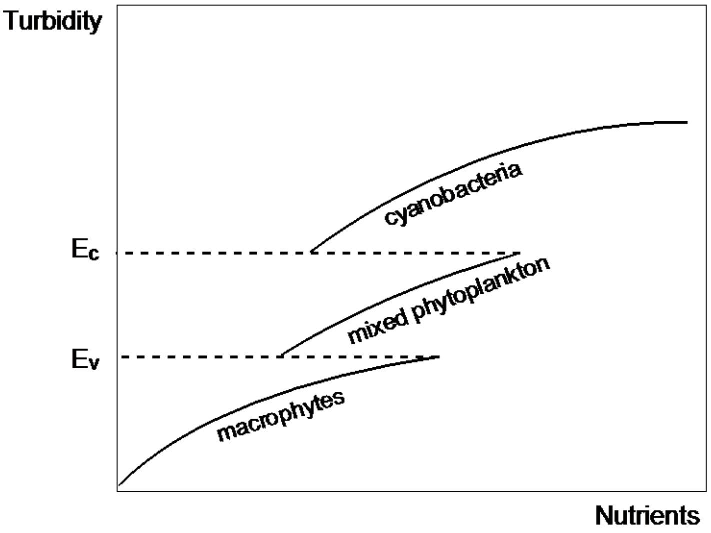

Figure 4 Conceptual model of the three possible equilibria within shallow lakes; macrophytes, a diverse phytoplankton assemblage and cyanobacteria (adapted from Scheffer, 1998).

macrophytes disappear (E_v) and cyanobacteria become dominant (E_c) (Scheffer, 1998). If the thresholds are close together, then the lake system is likely to be dominated by cyanobacteria whenever macrophytes are absent (Scheffer, 1998).

Unfortunately, there are no universally established values for either E_v or E_c, as these 'switching' points are also influenced by many other environmental factors. In the Netherlands, estimates for E_c range are between 4-10 m depth, and E_v between 1-3 m depth (Scheffer, 1998).

Research, from various parts of the world, has found that when phosphorus concentrations exceed levels between 50 and 62 µg.l^{-1}, the system is more likely to shift away from a macrophyte-dominated state:

- Moss et al. (1996) suggested that macrophytes are expected to dominate shallow lakes with <50 ug/L TP, whereas either a macrophyte- or phytoplankton-dominated system can exist as alternative states in lakes with >50 µg.l^{-1} TP.
- Zimmer et al. (2009) found that the majority of lakes with TP concentrations below a threshold of 62 ug/L were dominated by macrophytes. Macrophyte dominated lakes were generally characterised by chlorophyll *a* concentrations less than 22 µg.l^{-1}, whereas phytoplankton-turbid lakes were generally greater than 30 µg.l^{-1} chlorophyll *a*.
- A similar pattern was observed by Sayer et al. (2010), who observed that lakes with <50 µg.l^{-1} TP were generally stable irrespective of macrophyte growth, whereas lakes with higher nutrient levels (>50 µg.l^{-1} TP) were generally unstable, turbid and characterised by phytoplankton blooms.

More recently, nutrient loading rates have been investigated as a more accurate method for assessing the trigger points for switch between macrophytes and phytoplankton. Janse et al. (2010) developed a calibrated model based on data from 40 shallow lakes to predict the critical nutrient loading levels likely to trigger switches between macrophyte and phytoplankton (turbid) dominated states. They estimated that the threshold for the switch from macrophytes to phytoplankton was between 2 – 4 mg mg P.m^{-2}.d^{-1}, and the threshold for restoration of macrophytes was likely to be between 0.75 – 1.25 mg P.m^{-2}.d^{-1}.

It is beyond the scope of this review to provide a more detailed description of the alternative state model for shallow lakes. More comprehensive descriptions of the alternative state model can be found in Scheffer et al. (1997); Scheffer (1998); Scheffer and Jeppesen (2007) & Scheffer and van Nes (2007).

4 FACTORS THAT INFLUENCE CYANOBACTERIAL GROWTH

Section 2 highlighted the decline in the health of urban lakes which typically results in the formation of algal blooms. An understanding of the ecology of shallow lakes is essential if we are to develop strategies for managing algal blooms within urban lakes. There are broadly two types of “algal states” that exist in urban lakes as summarised in Section 3. These are phytoplankton which are closely related to plants and vary widely in form from single cells to filamentous forms, and cyanobacteria (blue-green algae) which are simple organisms that are closely related to bacteria. Whilst cyanobacteria and phytoplankton both obtain energy for growth solely from the sun, cyanobacteria differ from the phytoplankton in that they are prokaryotes, and therefore have no defined nuclei, chloroplasts or organelles.

Cyanobacteria may be unicellular or multicellular, the latter comprising of filamentous or colonial aggregations. The formation of colonies (e.g. *Microcystis*) or filamentous aggregates (e.g. *Anabaena*) reduces surface to volume ratio, therefore limiting the uptake rate of nutrients from the water column and potentially reducing growth rates (Reynolds, 1988). It is generally considered that the aggregation of cyanobacteria into colonies also helps to reduce losses to predation, and may be an adaptation to survive high grazing pressures (Scheffer, 1998).

The factors that influence algal growth are multidimensional and involve physical factors (water temperature, light radiation, turbidity, turbulence and flow), chemical factors (pH, nutrients, humic substances, dissolved oxygen) and biotic interactions (competition, grazing and predation).

The following section provides a brief overview of some of the key environmental factors that influence the growth of cyanobacteria in freshwater systems. A basic understanding of cyanobacterial ecology is essential in order to consider potential growth management and control strategies. If the reader requires further information regarding the ecology of cyanobacteria they should refer to the references listed in section 9.

4.1 NUTRIENTS

Cyanobacteria require a number of growth factors including nitrogen, phosphorus, carbon and trace metals. The internal nutrient loading within shallow lakes is important in determining the carrying capacity of the cyanobacteria and their responses. Nutrient management typically focuses on phosphorus and to a lesser extent nitrogen because algal and Cyanobacterial production is generally limited by Phosphororous, but the availability of some trace metals has gained recent attention in the literature. Phosphorus is generally the most important limiting nutrient for algal production in temperate lakes (Correll, 1998; Cunha Pereira, 2010).

4.1.1 Nitrogen

Whilst the vast majority of phytoplankton and cyanobacteria uptake dissolved inorganic nitrogen in the form of nitrate, nitrite and ammonia, many cyanobacterial species have developed mechanisms to cope when nitrogen is scarce, and are able to obtain nitrogen directly from the atmosphere (in much the same way as carbon is obtained from the atmosphere in photosynthesis) using specialised cells called heterocysts. This enables the cyanobacteria to overcome nitrogen-limitation in the water column and confers a direct competitive advantage over other phytoplankton groups.

Whilst it is generally considered that nitrogen-fixing cyanobacteria are more dominant in lake systems with low N:P ratios (Huisman and Hulot, 2005), this has not been found to be the case in most shallow lakes (Jensen et al., 1994, Noges et al., 2008). The management of nitrogen concentrations in lakes is therefore not generally considered a useful strategy for managing cyanobacterial growth.

4.1.2 Phosphorus

The growth of algae in lakes is typically limited by phosphorus. Generally, the phosphorus concentrations within a lake are indicative of the expected mean or maximum algal biomass (Moss, 2010).

In many shallow lakes, mixing and the close interaction of the sediment with the water column results in relatively aerobic sediments, and prevents phosphorus being released from the sediments. Although phosphorus concentrations are generally lower in these well mixed lakes, phosphorus levels are often high enough to trigger algal blooms. Well mixed shallow lakes with high rates of internal loading (i.e. high nutrient content in the sediment) are often characterised by the constant 'leakage' of phosphorus from the sediments and are therefore prone to algal blooms.

Conversely, lakes that seasonally stratify (typically lakes that are deeper than 2-3m) will tend to lose phosphorus from the upper waters (epilimnion) to the deeper waters (hypolimnion) of the lake, as non-motile algal species are denser than water and are gradually lost to the bottom sediments. Cyanobacteria that are confined to the upper waters are typically limited by phosphorus availability at this stage. As the lake destratifies in autumn, the lake 'turns over', releasing the trapped phosphorus into the upper waters, thus relieving cyanobacteria of P-limitation.

Cyanobacteria are particularly efficient at scavenging phosphorus from the water column and have developed specialised storage mechanisms which enable them to store enough phosphate to last for 3-4 cell divisions (Chorus and Mur, 1999). Therefore, a cyanobacterial cell may multiply into 8-16 cells without requiring additional phosphate uptake meaning cyanobacterial biomass can increase by up to 10 times after the water column has been depleted of dissolved phosphates (Chorus and Mur, 1999). The ability of cyanobacteria to 'luxury uptake' and store phosphates has implications for the use of wetland recirculation systems in controlling cyanobacterial biomass. This will be further discussed in Section 6.1.

Regulation of buoyancy

Many cyanobacterial species such as *Anabaena circinalis*, *Microcystis aeruginosa* and *Cylindrospermopsis raciborskii* have gas vesicles which able them to regulate their buoyancy, thereby allowing cells to vertically migrate through the water column. The ability to migrate up and down through the water column enables cyanobacteria to dominate over other phytoplankton groups in lakes with relatively low turbulence, as the cyanobacterial cells are able to optimise their position within the water column. In contrast, many other cyanobacteria and phytoplankton species are heavier than water in which they are dispersed, and are therefore subject to constant sinking (Reynolds, 2006).

In stratified lakes, the ability of cyanobacterial cells to regulate their buoyancy enables them to scavenge nutrients such as phosphorus and trace metals from the non-turbulent hypolimnion layer, and float up to the more mixed epilimnion layer where they have access to better light conditions. During calm conditions, cyanobacterial cells may often float to the surface of the lake, where they are trapped by the surface tension of the water and form visible scums (Ibelings et al., 1991).

4.1.3 Trace metals

It is well understood that extremely low concentrations of the trace metal Molybdenum (Mo) can limit the growth and abundance of nitrogen-fixing cyanobacteria in freshwater environments (e.g. Marino, et. al. 1990). This is because Mo is required to form the nitrogenase enzyme which cyanobacteria use for fixing atmospheric nitrogen (see section 4.1.1 above).

4.2 LIGHT AND TURBIDITY

Whilst the dominance of filamentous cyanobacteria in shallow lakes is generally associated with elevated nutrient concentrations (Healy, 1982), recent studies have shown that the abundance of cyanobacteria is more highly correlated to underwater light conditions (shading) than to phosphorus concentrations (Scheffer et al., 1997).

Plate 2 High turbidity in an urban lake

Scheffer (1998) suggested that the probability that cyanobacteria will dominate the algal community in a shallow lake will depend strongly upon diminishing light levels attributed to increasing turbidity. This is demonstrated by the dominance of cyanobacteria in lakes where the water column is shaded through high turbidity than other lakes with similar nutrient levels. This pattern is particularly evident in lake systems with less than 0.3 $mg.l^{-1}$ TP (Scheffer, 1998). As cyanobacterial populations become more dominant, they also increase shading, suggesting a positive feedback mechanism that further promotes cyanobacterial growth. Similar patterns of cyanobacterial dominance are also evident in other turbid lake systems (e.g. Plate 2) where other factors such as suspended solids influence shade (Scheffer, 1998). This is not surprising as most cyanobacterial species are extremely efficient at scavenging light (Reynolds, 2006), and may well survive and even proliferate within low light conditions for extended periods of time.

4.3 HYDRAULIC RETENTION TIME

Another important factor that may also affect cyanobacterial dominance in shallow lakes is hydraulic retention time. The model developed by Scheffer et al. (1997) (based upon data from 55 Dutch lakes < 3 m depth) demonstrates that a combination of flushing rates (lake retention times) and nutrient concentrations (TP) influence the alternative states model (Figure 5).

From the model it can be seen that by increasing flushing rates, conditions become less favourable for cyanobacteria and phytoplankton. A region exists where the two alternative states may be stable (hysteresis) and the system remains in an equilibrium state (cyanobacteria are dominated by other algae). Thus for Dutch lakes, cyanobacteria are predicted to be absent from the lake irrespective of nutrient concentrations at flushing rates greater than 18% of the total lake volume per day (equivalent to a hydraulic retention time of approximately 5.5 days). Although lake flushing (decreased retention times) is normally contingent upon lower nutrient concentrations within the inflowing water, the model suggests that when the nutrient concentrations are higher, cyanobacterial dominance can be avoided provided the flushing rate is sufficiently high.

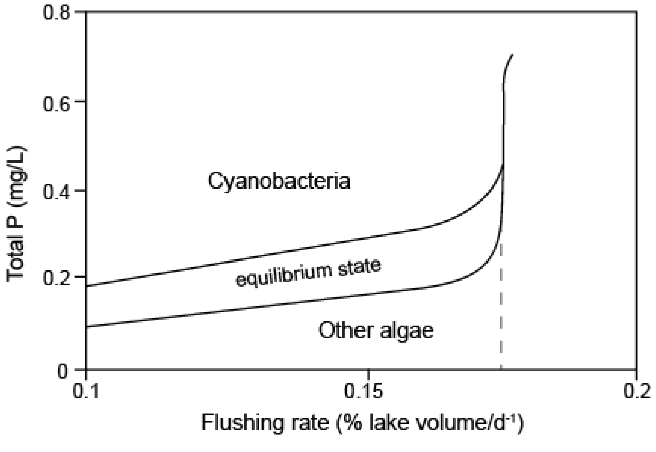

Figure 5 Model showing the response of shallow lake systems to flushing rate and TP concentrations (adapted from Scheffer et al., 1997).

The model presented in Figure 5 illustrates that the effect of flushing to reduce the competitive advantage of cyanobacteria is not necessarily correlated to the reduction of turbidity. Scheffer (1998) postulated that the effect of flushing on cyanobacteria is significant because the comparatively low rates of cyanobacterial productivity are usually compensated by having low loss rates (e.g. grazing). Therefore, the relative impact of loss due to flushing is much higher for the slow growing cyanobacteria than the other algae.

4.4 TEMPERATURE

Cyanobacterial growth rates are strongly regulated by water temperature, and tend to be favoured by warm, stable water columns that experience low flushing rates (Reynolds, 2006). For example, blooms of *Anabaena*, *Aphanizomenon* or *Microcystis* thriving in warm waters with high photosynthetically active radiation, have been known to suddenly terminate in response to a weather related decrease in temperature of 5°C (Paerl, 1988).

Elliott (2010) investigated the sensitivity of cyanobacteria and other phytoplankton to changing flushing rates and water temperature. The study clearly demonstrated that high water temperatures and low flows favoured the dominance and bloom development of cyanobacteria.

There is currently no published data on the relationship between water temperature and Cyanobacterial bloom development in shallow lakes in Australia. Optimal *in situ* growth rates observed for *Microcystis aeruginosa* range between 27.5 – 32 °C, *Anabaena* spp. 24 – 35 °C and *Aphanizomenon* spp. 15 – 28 °C (Robart and Zohary, 1987). Fabbro and Duivenvoorden (1996) suggested that optimal growth rates for *Cylindrospermopsis raciborskii* occur when water temperatures exceed 25 °C. Long term monitoring of a shallow urban lake on the Gold Coast found that the cyanobacterial blooms occurred in the lake during spring each year when water temperature reached 25 °C (DesignFlow, 2009).

Many bloom-forming Cyanobacterial species can withstand extensive daily periods of exposure to high photosynthetic radiation flux and ultraviolet radiation, in contrast to other algae which often experience severe photo-oxidative damage in response to such conditions (Paerl, 1988).

4.5 WATER LEVELS

Water level maintenance in shallow lakes plays an important role in maintaining macrophyte-dominance and therefore the suppression of algal biomass. In shallow water, light conditions at the bottom are more favourable and macrophytes can more easily grow and reach the surface layer where they are relatively immune from the shading impacts of the phytoplankton and suspended solids (Scheffer and van Nes, 2007).

Small changes in water depth may have profound impacts upon critical thresholds for turbidity, thereby resulting in dramatic shifts from one state to another in lakes that are close to the breakpoint already (Scheffer, 2001). Macrophytes are able to resist increased turbidity levels if a lake is shallower Therefore, the critical turbidity threshold required to trigger a transition from a macrophytes-dominated state to a phytoplankton or cyanobacterial dominated state will be higher in shallow lakes (Scheffer, 1998). Conversely, increased water levels may lead to the loss of macrophytes resulting in a dramatic shift to a turbid state (Scheffer and van Nes, 2007).

4.6 MIXING / WIND

Enhanced mixing and turbulence within lake systems limits both the formation and accumulation of cyanobacterial blooms (Wagner-Lotkowska et al., 2004). The growth of cyanobacteria is inhibited by well mixed conditions. Turbulence within lakes may be achieved by decreasing hydraulic retention times, increasing inflows and maximising turbulence created by wind.

4.7 RESTING CELLS

Cyanobacteria reproduce asexually by simple cell division (binary fission), filament fragmentation or germination of specialised resting cells called akinetes. The factors related to cyanobacterial growth have been extensively studied. The development of cyanobacterial populations appear to be intimately related to: a) the growth of *in situ* cell populations within the water column, b) the germination of akinetes from the sediments, and c) growth of overwintering cell populations located within the sediments (Hense and Beckmann, 2006, Tsujimura and Okubo, 2003).

Akinetes are thick-walled reproductive structures which serve as a resting stage and ensure survival during adverse growth conditions (Baker and Bellifermine, 2000). Akinetes are produced by many Cyanobacterial species, particularly noxious bloom forming species such as *Anabaena circinalis* and *Cylindrospermopsis raciborski*, in response to unfavourable growth conditions, e.g. low water temperatures, phosphorus depletion, iron deficiency or light limitation (Paerl, 1988). Akinetes can reside in sediments for many years before germinating upon the restoration of favourable conditions (Paerl, 1988). For example, Livingstone and Jaworski (1980) successfully germinated *Aphanizomenon* and *Anabaena* akinetes from sediments that were 18 and 64 years old respectively.

In 2009, *Cylindrospermopsis raciborskii* was observed to increase from almost zero cell concentrations to nearly 200,000 cells.ml^{-1} over the course of a week in the Fitzroy River following a long period of stable, favourable conditions (L. Fabbro, pers. comm.). In this case, it was suggested that rapid re-establishment of the cyanobacterial population was facilitated by the mass germination of akinetes within the sediments.

Recent evidence points to the rapid development of cyanobacterial blooms due to the massive recruitment of cells from the germination of akinetes and growth of overwintering cells within the sediments. Baker (1999) observed akinete densities up 70,000 akinetes.g^{-1} sediment for *Anabaena* within the Murray River. Similar akinete densities have been observed in lakes, e.g. 26,100 akinetes.ml^{-1}, Faithful and Burns (2006), and 9,000- 15,000 akinetes.ml^{-1}, Tsujimura (2004), Tsujimura and Okubo (2003).

Faithful and Burns (2006) investigated the germination of *Anabaena* akinetes from a shallow lake. They observed the development of cell concentrations of 5,000 cells.ml^{-1} within 5 days of incubating the akinetes (using diluted sediment inoculum). Cyanobacterial species such as *Microcystis aeruginosa* utilise an alternative strategy, whereby the vegetative cells may over-winter on the bottom sediments of lakes (Reynolds, 2006). Photochemically-active *Microcystis* populations may be found in sediments throughout the year, particularly sediments that are present within the euphotic zone (Verspagen et al., 2004). The overwintering population inoculates the water column in spring and promotes the development of *Microcystis* populations during summer.

Factors relating to the germination of akinetes:

- Exposure to light is considered to be the principal requirement for akinete germination; however other factors such as temperature, mixing, nutrients, salinity and desiccation may also be important (Baker and Bellifermine, 2000). The germination of akinetes may occur over a wide range of water temperatures. For example, Baker and Bellifermine (2000) found that the optimum temperature for the germination of *Anabaena circinalis* akinetes was between 20-25 °C, however akinetes were observed to germinate at temperatures as low as 10 °C and as high as 38 °C.
- Salinity appears to have minimal effect upon the germination of akinetes in freshwater ecosystems. Baker and Bellifermine (2000) observed that *Anabaena circinalis* akinetes were able to germinate in salinities up to 5000 µS.cm^{-1}. Similar observations have been made for *Cylindrospermopsis raciborski* akinetes which were able tolerate salinities up to 6 g.l^{-1} (Moisander et al., 2002).
- Aquatic vegetation may play an important role in preventing the germination of akinetes through shading, competition for nutrients (McQueen, 1990), providing refuge for predators and also the production of allelopathic compounds which are inhibitors of cyanobacterial growth (Nakai et al., 2000; Nakai et al., 2005).

5 RETENTION TIME CRITERIA FOR MANAGING CYANOBACTERIAL GROWTH

Managing the risk of algal bloom development is one of the most important considerations when designing lakes in urban areas. Burge and Breen (2006) proposed an approach that uses the design/management of lake retention times to manage the risk of cyanobacterial growth (and subsequent possible blooms) in constructed urban lakes.

In this section, the application of the proposed retention time design criteria approach for managing cyanobacterial growth in urban lakes will be reviewed.

5.1 USE OF LAKE RETENTION TIMES TO MANAGE CYANOBACTERIAL GROWTH

The relationship between lake retention times and lake water nutrient concentrations (e.g. potential algal biomass) was originally recognised by Schindler and Nighswander (1970) who suggested that lake surface area and ‘water loading’ should also be considered when using the Vollenweider relationship.

Vollenweider (1976) subsequently refined the empirical relationship between nutrient inputs and equilibrium concentrations in lakes (commonly referred to as the Vollenweider equation) to include water lake retention times. Subsequent experimentation, observation and monitoring of lake ecosystems have enabled a progressive understanding as to how lake retention times influence cyanobacterial growth.

The influence of a lake’s retention times can be simply considered in terms of the hydraulic displacement and dispersion of Cyanobacterial cells. The hydraulic displacement of phytoplankton cells occurs when the lake’s inflow is exchanged with the instantaneous lake volume and embedded planktonic cells are removed in the outflowing displaced volume (Reynolds, 2006). When hydraulic displacement occurs, the residual population is now smaller and on average, less concentrated, however this may be offset or compensated by the simultaneous rate of cell replication (Reynolds, 2006). Hydraulic displacement of phytoplankton cells occurs via two mechanisms:

1. Wash-out: the replacement of the original lake volume by direct displacement
2. Flushing: by dilution, where the inflow volume mixes extensively with the original volume, thereby displacing an equivalent volume of well-mixed water.

The impacts of both wash-out and flushing have profound impacts upon algal biomass and composition within lakes. Wash-out is primarily mediated by episodic runoff events and may result in the complete displacement of algal biomass from the lake system, and assumes that inflowing water moves through the lake system as a plug flow. In contrast, flushing relies upon a constant inflow to the lake (often provided through natural or diverted stream flows) and results in a less complete depletion of the algal biomass. The biological mechanisms associated with flushing that can be used to manage cyanobacterial growth in lake systems are detailed in section 1.

There is little or no information within the literature that details the relationship between the hydraulic displacement of cyanobacterial biomass via wash-out and lake retention times. Reynolds and Lund (1988) observed that episodic flushing of a small lake (average lake retention times between 8-20 days) substantially reduced algal biomass to the extent that a long dry summer was required for *Anabaena* to re-establish.

In contrast, almost all of the contemporary research associated with the use of hydraulic displacement to manage cyanobacterial growth in lakes has involved the use of flushing to manage lake retention times. Welch (1988) reported that the addition of low nutrient river water significantly reduced both nutrient and chlorophyll *a* concentrations in two large eutrophic lakes. The dilution of water through the lakes resulted in lake retention times ranging between 12–28 days and 153-416 days respectively (Welch, 1988). Similar observations were also made in Cameron Lake, Ontario, where low chlorophyll a concentrations of less than 5 ug/L were maintained with flushing rates ranging between 19-26 days (Dillon, 1975).

Reynolds (2003) proposed that the responsiveness of lakes (referring to the potential algal biomass development) to phosphorus loads decreased with high fluvial flushing. It was proposed that lakes with >30 days retention were ‘very’ sensitive to altered phosphorus loads, whereas lakes with retention times between 3-30 days were only ‘slightly’ sensitive. Lakes with retention times <3 days were not considered sensitive at all.

The sensitivity of cyanobacterial biomass to increased flushing may depend upon *in situ* growth rates. For example, Reynolds (2006) noted that in lakes with average retention times >100 days, increasing flushing rates from as little as 0.01 d-1 to 0.02 d-1 (i.e. from >100days to 50 days retention time) can significantly reduce cyanobacterial biomass if cyanobacterial growth is limited by other environmental conditions (such as temperature, nutrient or light availability – refer Section 2).

Elliot et al. (2009) found that the response of chlorophyll to changes in lake retention time (i.e. altered flushing rates) was seasonally mediated and dependent upon nutrient sources. Increased flushing rates combined with nutrient inputs from point sources (such as stormwater inflows) resulted in a reduction of chlorophyll concentrations, however chlorophyll concentrations remained unchanged regardless of flushing rate when nutrient inputs were associated with diffuse sources (such as river discharges), or during the winter period due to the low algal growth rates. The authors contended that increased flushing associated with nutrients from point sourced mediated algal biomass was due to the flushing loss of both nutrients and algae (Elliott et al., 2009).

A number of generalisations concerning lake retention times have been made from European lake research. Tarczynska et. al. (2001) found that lake retention times of more than 60 days usually lead to the formation of high algal biomass (considered to be >4 ug/L). Generally, the average cyanobacterial biomass during retention times <60 days were observed to be less than half that observed when retentions were >60 days.

Tarczynska et al. (2002) predicted that algal succession in through-flow reservoirs is expected to be similar to that observed in rivers, as phytoplankton growth is limited by intensive mixing as a result of water exchange and short lake retention times. Therefore, decreasing lake retention times in times when the risk of cyanobacterial growth is expected can effectively reduce or eliminate the occurrence of blooms (Tarczynska et al., 2002). Based upon the findings of Tarczynska et al., (2001), it is recommended that lake retention times should not exceed 30 days for medium sized through-flow reservoirs (Wagner-Lotkowska et al., 2004).

5.1.1 There are currently no recommendations for lake retention times reagrding urban lakes which experience episodical wash-out. RETENTION TIME - ALGAL GROWTH THRESHOLD APPROACH

Burge and Breen (2006) proposed a probabilistic approach based upon lake residence (retention) time criteria which can be used to minimise the risk of cyanobacterial growth in urban lakes. This approach is based upon a theoretical model of cyanobacterial growth that incorporates a range of parameters including temperature, hydrodynamics and *in situ* algal growth rates.

The probabilistic approach to lake retention time design criteria has now been adopted in Queensland as part of the Mackay Planning Scheme Policy 15.15 - Constructed Lakes (Mackay City Council, 2008).

The lake retention time approach is based on a series of assumptions listed below:

- A cell concentration of 15,000 cells.ml^{-1} is an acceptable threshold for cyanobacterial blooms
- An initial cyanobacterial concentration of 50 mg.l^{-1} of algal cells is representative of cell concentrations prior to the peak cyanobacterial growth period
- Lake retention time thresholds based upon episodic wash-out events can provide an effective mechanism for the control of cyanobacterial growth, and
- Exceedance of the lake retention time threshold for an average of 20% of days in a year is acceptable from an algal risk management perspective.

Each of these assumptions will be discussed in more detail through the remainder of this section.

5.1.2 Threshold concentrations for cyanobaterial blooms

- A wide range of critical thresholds exist relating to acceptable cyanobacterial levels in lakes.The World Health Organisation (WHO) recommends that notification to inform the public of potential health risks is required when cyanobacterial cell counts exceed 20,000 cells.ml^{-1} (WHO Level 1: Relatively low probability of adverse health effects, WHO, 2003).
- In contrast, the Australian guidelines recommend that Alert Level 1 is declared when *Microcystis aeruginosa* concentrations range between 5,000 - 50,000 cells.ml^{-1}, or the combined biovolume of all other cyanobacteria falls between 0.4 – 4 mm^3.l^{-1} (NHMRC, 2008).

The relatively higher threshold for cyanobacterial concentrations within Australia is based upon the risk of exposure to cyanotoxins via ingestion rather than skin contact (NHMRC, 2008). Based upon the average biovolume of *Anabaena circinalis* cell (2.2 x 10-4 mm^3, Baker, 1991), a biovolume of 4 mm^3.l^{-1} is equivalent to approximately 18,000 cells.ml^{-1}. This is commensurate with the definition of bloom levels of *Anabaena circinalis* (defined as > 15,000 cells.ml^{-1}) noted for the Barwon-Darling River system (Mitrovic et al., 2006).

Based upon the thresholds cited above for cyanobacterial cell densities in recreational waters, the critical cyanobacterial cell threshold adopted by Burge and Breen (2006) to derive the probabilistic approach to lake retention criteria is considered appropriate.

5.1.3 Initial cyanobacteria concentrations and growth rates

The growth rate model developed by Burge and Breen (2006) to estimate the times required for cyanobacterial populations to reach bloom proportions under different temperature and mixing conditions were based upon reported *in situ* growth rates of *Anabaena circinalis* and assumed a starting concentration of 50 cells.ml^{-1}.

A major limitation associated with such a theoretical approach to modelling cyanobacterial growth is that the model is highly sensitive to the initial cyanobacterial cell concentrations. Background cyanobacterial cell concentrations may vary greatly (between 0-5000 cells.ml$^{-1)}$ during the peak cyanobacterial growth season in Queensland (typically late spring to early autumn). Therefore, the time taken for cyanobacteria to develop to bloom proportions (15,000 cells.ml^{-1}) may be significantly shorter depending upon the in situ cell concentration.

Background cyanobacterial cell densities of at least 250 cells.ml^{-1}could develop and persist in urban lakes throughout the peak cyanobacterial growth season.

Re-modelling of the growth curves developed by Burge and Breen (2006), indicates that the estimated time for cyanobacterial populations to reach bloom proportions is highly sensitive to the initial cell concentration used. For example, an *Anabaena circinalis* population would take only 13 days to reach bloom proportions based upon a starting concentration of 250 cells.ml^{-1} (Figure 6).

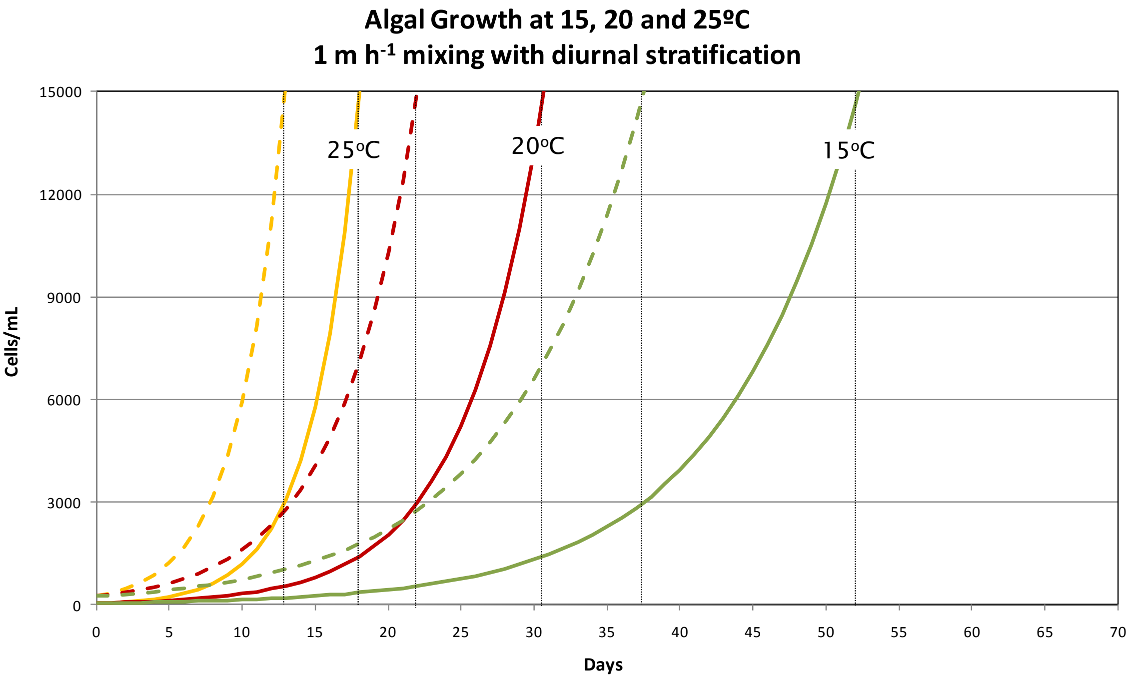

Figure 6 Growth curves indicating times for cyanobacterial populations to reach bloom proportions based upon partially mixed conditions (1 m/h mixing conditions with diurnal stratification) and starting concentrations of 50 cells.ml^{-1} (solid lines) and 250 cells.ml^{-1} (dashed lines). (Figure and modelling adapted from Burge and Breen, 2006).

The modelling undertaken by Burge and Breen (2006) did not incorporate several key elements associated with the establishment of cyanobacterial blooms. As noted in Section 4.7, there are three potential sources of inoculum for cyanobacterial blooms:

1. *In situ* cell populations
2. Akinetes within the sediments, and
3. Overwintering cell populations within the sediments.

Recent observations within Australia have shown that under non-limiting conditions, cyanobacterial populations are able to rapidly develop to bloom proportions.

In February 2005, a population of > 30,000 cells.ml^{-1} of *Anabaena circinalis* developed in the Torrens Lake, Adelaide within the three weeks following a rain event in January which resulted in the complete turnover of the lake volume (Peter Baker, pers. comm.). Long residence times in the lake (influenced by the frequency of rainfall events and discharge through the lake), appear to be a significant factor in the development of cyanobacterial blooms (Peter Baker, pers. comm.).

Other examples which illustrate how quickly cyanobacterial blooms can develop include:

- Chan et al. (2004) observed *Anabaena* spp. populations reaching peak bloom densities of >100,000 cells.ml^{-1} in less than four weeks during in situ enrichment experiments using freshwater ponds
- As discussed in Section 4.7, cyanobacterial blooms can rapidly develop from resting cell (akinete) populations in the sediments which have been measured between 20,000 and 70,000 cells.ml^{-1} sediment
- Sediment incubation studies of akinetes showed > 5,000 cells.ml^{-1} after 5 days (Faithful and Burns, 2006). In comparison, the modelling undertaken by Burge and Breen (2006) predicted cell concentrations of less than 1,000 cells.ml^{-1} within 5 days.

5.1.4 Retention time exceedance criteria (20th percentile)

Based upon the use of the probabilistic approach (lake retention time criteria) for managing the risk of cyanobacterial growth in lakes, the Mackay Lake Guidelines recommend that the 20th percentile residence time for constructed lakes should not exceed 20 days (assuming average water temperatures exceed 25 °C) (Mackay City Council, 2008). Therefore, under the existing guidelines, the probabilistic approach to lake retention time is applicable to approximately 80 % of the year.

Considering the typical climatic conditions for SEQ, cyanobacterial growth occurs within lake systems throughout the year, particularly during the wet season from spring through to autumn. Cyanobacterial blooms may also occur during winter when there is less rain resulting in longer residence times in lakes. Maximum air temperatures at this time of year rarely drop below 20 °C, meaning that sufficiently high water temperatures are maintained throughout winter that may potentially stimulate cyanobacterial growth.

Under the existing guidelines, the probabilistic approach to lake retention time is applicable to the entire year. The climatic conditions in SEQ means there are periods during the warmer months (spring, summer autumn) in almost every year when lake retention times are long (>20 days). Therefore it is possible that some urban lakes may indeed conform to the 20th percentile criteria, but experience far greater retention times (e.g. up to 55 days or two periods of 25+ days) during the dry season.

If the probabilistic approach is adopted, then it is recommended that:

- the retention time criteria are applied to the entire year, as cyanobacterial growth is possible in SEQ throughout the year due to suitable water temperatures
- the exceedance criteria is set to 95 % to limit the occurrences of extended lake retention to a single 20 day period per year.

5.1.5 Lake retention times

It is generally well established that many incidences of algal blooms in lake ecosystems are preceded by periods of minimal inflows, and that the lake retention times play an important role in determining the algal biomass characteristics of lake ecosystems (Reynolds, 2006; Scheffer, 1998). Welch (1981) detailed how the manipulation of lake retention (turnover) times could be used as a strategy for managing algal biomass, whereby substantial reductions in algal biomass were achieved by flushing river water through two large lakes.

As outlined in Section 5.1, cyanobacterial growth in lakes is mediated by two mechanisms that are intimately associated with lake retention times, wash-out and flushing (Reynolds, 2006). Flushing occurs when baseflows or minor events enter the lake system and mix with the existing water, whereas wash-out occurs when large episodic events enter the lake system and a given volume of the lake is displaced.

Algal groups can be eliminated from a lake if algal biomass losses due to flushing exceed the growth rate (Cooke et al., 2005); and all algal biomass may be eliminated when more than one third of the lake volume per day is flushed

(Scheffer, 1998). Small decreases in lake retention times may lead to the disappearance of cyanobacteria in situations where the competitive balance with other algae is already close to shifting (Scheffer, 1998).

Burge and Breen (2006) and the Mackay Lake Guidelines recommend that the 20th percentile residence time for urban lakes should not exceed 20 days. The probabilistic approach to managing cyanobacterial growth is clearly based upon the reliance of episodically driven wash-out events to minimise the opportunity for cyanobacterial populations to reach bloom concentrations (15,000 cells.ml^{-1}). The following concerns are raised regarding this approach:

- The proposed use of lake retention times to manage the risk of cyanobacterial growth is not commensurate with the intent of lake retention times as used elsewhere, which rely upon constant inflows to provide flushing and removal of cyanobacterial cells. There is little evidence available that suggests that episodic wash-out driven flushing of cyanobacterial cells will offer a sufficient level of long-term risk protection for cyanobacterial growth in constructed urban lakes.
- Considering the propensity for cyanobacterial populations to rapidly reach bloom concentrations (depending upon *in situ* concentrations, akinete germination and over-wintering cell populations), the 20 day criteria appears too short. Refer Section 5.2.2 for further discussion of growth rates.

Considering the above, the control of cyanobacterial biomass via lake retention time in SEQ will be most likely facilitated by constant flushing of cyanobacterial biomass from the lake system and adopting a new retention time criteria that is 10-15 days rather than 20 days.

6 USE OF CONSTRUCTED WETLANDS TO MANAGE LAKE RETENTION TIMES

6.1 RECIRCULATION TO MANAGE CYANOBACTERIA

The *Mackay Lake Guidelines* adopt the probabilistic approach to lake retention times, and in addition, recommend that if a lake does not meet the residence time criteria, then water can be pumped from the lake and recirculated through a wetland in order to effectively manage lake retention times (Mackay City Council, 2008).

The concept of using recirculation wetlands to treat lake water quality has been used in other lake systems. For example, a large constructed wetland (3.65 km^2) was used to treat lake water from Lake Apopka, a large shallow lake in Florida (Lowe, et al., 1992). In this case, the aim of the wetland was to reduce phosphorus concentrations in the lake in order to control algal biomass (mean chlorophyll *a* = 60 $\mu g.l^{-1}$). Rodriguez-Gallego,et. al. (2004) evaluated the performance of a wetland recirculation system comprising of a small hypertrophic urban lake and constructed wetlands. They found that the recirculation wetland successfully reduced suspended solids and nutrients within the lake, leading to a noticeable decrease in algal biomass. Recirculation wetlands are also widely used by the aquaculture industry to manage water quality, as the flushing of growing ponds using river water is no longer permitted in many countries.

The use of recirculation wetlands to treat lake water quality is similar to constant flushing of the lake system. It is therefore conceivable that recirculation wetlands could be used to manage lake hydraulic retention times and therefore influence the development of cyanobacterial biomass.

A number of criteria need to be satisfied in order for constant flushing via a recirculation wetland to be an effective strategy for managing cyanobacterial growth:

- The majority of cyanobacterial cells (biomass) must be removed from the lake water whilst passing through the wetland
- The 'treated' water re-entering the lake must have turbidity and nutrient concentrations less than or equal to the *in-situ* lake water
- The lake water should be fully mixed (i.e. have high hydraulic effectiveness).

The *Engineering Design Guidelines*: *Constructed Lakes* (Mackay City Council, 2008) stipulate that recirculated lake water should take five days to pass through the wetland. It is understood that the five day residence time within the wetland is intended to span at least two cyanobacterial growth cycles (Peter Breen, pers. comm.). The combination of high shading and subsequent energy loss via cellular respiration is expected to severely diminish the viability of cyanobacterial cells as they pass through the wetland.

The ability of cyanobacteria to 'luxury' store phosphates (enough to undergo 3-4 cell divisions, see Section 4.1.2) means that simply passing cyanobacterial cells through a heavily shaded wetland over 5 days is unlikely to substantially diminish cell health, particularly as most cyanobacteria are able to tolerate low light conditions (Reynolds, 2006). Therefore, other physical and biological mechanisms occurring within the wetland will need to facilitate the removal of cyanobacterial cells from the water column.

The effectiveness of surface flow wetlands for eliminating cyanobacterial biomass from lake water is largely unknown, however the potential mechanisms by which cyanobacterial cells passing through wetlands may be reduced include:

- herbivory by zooplankton
- entrapment within the epiphytic and benthic biofilms that are attached to plant stems and sediments, respectively
- macrophyte allelopathy (chemicals produced by some macrophytes interfere with cyanobacterial growth)
- sinking/sedimentation processes.

Rodriguez-Gallego, et.al. (2004) noted that the recirculation of lake water (average residence time >70 days) through a constructed wetland promoted algal biomass removal (as opposed to cyanobacterial biomass). The authors attributed the reduction in algal biomass to filtration processes occurring within the wetland rather than reductions to nutrient concentrations within the lake. The constructed wetland was dominated by the free-floating water plants (water hyacinth and duckweed), and it is more likely that heavy shading of the water column was a contributing factor to the decline in algal biomass.

Cyanobacteria are commonly found growing within wetlands, and cyanobacterial blooms have been known to occur within stormwater treatment wetlands (L. Bowling, pers. comm.). This is not surprising as most cyanobacterial species are extremely efficient at scavenging light (Reynolds, 2006), and may well survive and even proliferate within wetlands for extended periods of time.

Baker and Belliferme (2000) suggested that shallow wetlands provide a conducive environment for akinete germination due to the continual resuspension of the akinetes from the sediments to the euphotic zone, and the penetration of light to the sediments. It has even been speculated that wetlands may in fact act as a seeding source for cyanobacteria in rivers that experience regular cyanobacterial blooms (such as the Murray River, Baker, 1999).

The flushing rates required through a constructed urban lake in order to manage cyanobacterial growth are currently unknown. Experience from overseas studies suggests that flushing rates (replacement of lake volume) vary widely from one in seven to one in one hundred days. According to the alternative state model for shallow lakes, the required flushing rates are dependent upon threshold lake turbidity levels and phosphorus concentrations (see Section 4.3), although this is also likely to be influenced by climatic region, lake size and catchment characteristics.

Further research is required in order to verify the effectiveness of wetlands in removing cyanobacterial biomass from freshwater lakes. Questions that need to be addressed include elucidating the mechanisms responsible for cyanobacterial cell decline in wetlands and determination of the minimum retention times required for cyanobacterial biomass reduction.

6.2 RECIRCULATION TO IMPROVE LAKE WATER QUALITY

An additional benefit from recirculating lake water through a wetland is the removal of nitrogen, phosphorus and suspended solids. This is extremely important, as the quality of the water re-entering the lake needs to be of equal or better quality than the in situ lake water for constant flushing to be effective.

It is well established that a reduction in phosphorus loadings is an essential element of lake restoration (Moss, et al., 1996). The aim of reducing internal nutrient loads is to enable a transition of the lake system to a stable submerged macrophyte-dominated state. As described above, the alternative states model for shallow lakes indicates that a substantial reduction in phosphorus concentrations is also required to cause a switch from phytoplankton/cyanobacteria to macrophyte dominance (Scheffer, 1998).

The use of wetlands to remove phosphorus loads from lake ecosystems is cost effective. For example, the net cost of operating the recirculation wetland at Lake Apopka was estimated to cost less than 3% of the estimated cost of sediment dredging (Lowe, et al., 1992). Recirculation wetlands therefore offer a particularly attractive method for reducing phosphorus loads within lake systems. Recirculation is also a long-term strategy as the phosphorus load within the lake water will be related to internal nutrient cycling processes and the rate of phosphorus release from the sediments. Lake sediments are variable and complex, and most lake sediments will release phosphorus under both aerobic and anaerobic conditions (Moss, 2010). Some studies have concluded that internal loading and recycling of sediment P stores can be so substantial that P concentrations in the water column may not respond to reduced external P loading, and that recovery is a slow process and may even require decades (Marsden, 1989; Jeppesen et al., 1991, Sondergaard et al., 2001).

There have been many recent attempts to restore algal dominated eutrophic lakes by reducing internal nutrient loads (for example, Kohler et al., 2005; Romo et al., 2005; Jeppersen et al., 2005; Sondergaard et al., 2005). The initial reduction in nutrient loads within eutrophic lakes are generally accompanied by a commensurate decrease in phytoplankton-dominance and turbidity levels (Sondergaard et al., 2005).

In shallow lakes, small improvements in turbidity exposes large areas of the lake bed to sufficient light to facilitate the gradual restoration of submerged macrophytes, enhanced primary production and oxidation of the sediment surface. It should be re-iterated that the re-establishment of submerged macrophytes usually requires significant intervention to restore sediments.

Internal phosphorus recycling is often increased in shallow lakes due to wind-induced sediment re-suspension and higher water temperatures (Sondergaard et al., 2001). Phillips et al. (2005) studied the long-term recovery of a shallow eutrophic lake following reduced nutrient loading. They observed that chlorophyll concentrations decreased during spring and early summer after 5 years, however it took almost 15 years for chlorophyll concentrations to decrease during summer due to sustained phosphorus remobilisation from the sediments.

Jeppersen et al. (2005) analysed 35 shallow lakes which had undergone major reductions in nutrient loading and found that the equilibria for both internal phosphorus and nitrogen concentrations were notably delayed (10-15 years for phosphorus and 0-5 years for nitrogen) irrespective of hydraulic retention time. The study concluded that phosphorus loads present within lake sediments can delay lake recovery (both phosphorus concentration and algal biomass) for many years following nutrient input reduction.

In warmer regions (such as warm temperate and tropical regions), the recovery of eutrophic lakes is often less successful due to increased internal phosphorus cycling from the sediment (Genkai-Kato and Carpenter, 2005). Gradual increases in lake water temperatures may also reduce the critical phosphorus loading required to initiate a regime shift from macrophytes to phytoplankton dominance (Genkai-Kato and Carpenter, 2005).

Romo et al. (2005) investigated the response of Lake Albufera, a warm temperate shallow lake to reduced nutrient loading. Despite a 30% reduction in phosphorus concentrations (after nine years following reduced nutrient loadings) and an average hydraulic retention time of 36 days, TP remained high (0.34 mg.l^{-1}) and improvements in water quality were insufficient to allow macrophyte colonisation.

In summary, reducing internal phosphorus loads within shallow lakes has been demonstrated to be a very important part in any successful strategy for reducing lake phosphorus concentrations, albeit over long periods of time. A reduction in phosphorus concentrations often results in a reduction to algal biomass, however there is often a substantial lag period for phosphorus concentrations to reach a lower equilibrium. Recent studies have shown that the phosphorus equilibrium required to trigger a switch from a phytoplankton/cyanobacterial state to a macrophyte state requires lower in situ phosphorus concentrations in warm temperate and tropical regions compared to temperate regions. Therefore, the management of cyanobacterial growth via in-lake phosphorus reduction and return to a macrophyte-dominated state may be more difficult to achieve in warm temperate and tropical regions.

7 ALTERNATIVE STRATEGIES FOR MANAGING CYANOBACTERIAL GROWTH

A number of alternative strategies have been identified which may be considered for managing cyanobacterial growth in urban lakes.

7.1 MECHANICAL REMOVAL

Cyanobacterial surface scums may be mechanically removed using booms dragged behind boats coupled with pumping the concentrated cyanobacterial scum from the surface. This technique is not widely used for the management of cyanobacterial blooms as cyanobacterial biomass will be present throughout the water column, and therefore only a small proportion of the algal biomass will be removed.

Atkins et al. (2001) reported the use of flocculation and mechanical removal techniques to manage a *Microcystis aeruginosa* bloom in the Swan River, Perth. A mixture of bentonite clay and polyaluminium chloride was used to flocculate and sink the *Microcystis* cells. Mechanical removal of surface scums was also trialled using oil spill equipment.

The mechanical removal of cyanobacterial biomass offers the advantage of also removing nutrient loads from the waterbody. However, the disposal of the odorous and often toxic biomass may be problematic.

7.2 ULTRASONIC TREATMENT

Ultrasonic irradiation of cyanobacterial cells is a relatively new strategy that has been proposed for managing cyanobacterial growth in lakes. Many of the common bloom forming cyanobacteria are characterised by the presence of gas vacuoles which enable the cyanobacterial cells to regulate their position within the water column. The presence of gas vacuoles provides a competitive advantage to cyanobacteria over the majority of other algal groups, as it enables the cyanobacteria to scavenge nutrients from the hypolimnion (lower strata of the water column) and to optimise their exposure to light within the epilimnion (upper strata of the water column) (refer Section 4.1 above).

Lee et al., (2001) investigated the use of ultrasonic irradiation as an alternative strategy for controlling cyanobacteria, and found that the gas vacuoles of cyanobacteria (*Microcystis viridis* and *M. aeruginosa*) were disrupted following brief exposure to ultrasonic irradiation, resulting in about 80% of the cyanobacterial cells settling out. Decreased photosynthetic activity was observed following irradiation, suggesting that the ultrasonic treatment also inflicted damage to the photosynthetic apparatus of the cyanobacteria.

A potential risk associated with exposing cyanobacterial cells to ultrasonic irradiation is that the cell membranes may rupture, potentially releasing cyanotoxins into the water column. Interestingly, Lee et al., (2001) observed that ultrasonication of the *Microcystis* cells did not increase the concentration of cyanotoxins in the water column. More recent research has shown that ultrasonication of cyanobacterial cells (for periods less than 5 mins) does not increase cyanotoxins (in this case Microcystins) as cyanotoxins may also degrade in the ultrasonic fields (Ma et al., 2005). The effects of ultrasonic irradiation upon other cyanotoxins such as cylindrospermopsin (*Cylindrospermopsis* and *Aphanizomenon*) and saxitoxins (*Anabaena*) have not been tested.

The evaluation of ultrasonic irradiation to control cyanobacterial growth in lakes thus far has resulted in mixed outcomes. Nakano, et al. (2001) trialled the use of ultrasonic irradiation coupled with flushing in a large shallow hyper-eutrophic lake (1 m depth, 33 Ha) that is plagued by cyanobacterial blooms. Whilst the use of the ultrasonic irradiation reduced peak concentrations for chlorophyll *a* from 200 $\mu g.l^{-1}$ to 130 $\mu g.l^{-1}$ (Lee, et al. 2002), the study was unable to conclusively demonstrate the effectiveness of the ultrasonic treatment upon cyanobacterial growth.

Ahn, et al. (2007) used ultrasonic irradiation to control cyanobacteria in a small eutrophic lake (2 m depth, 0.9 Ha). During the period of the year when cyanobacterial blooms dominated the lake, chlorophyll *a* concentrations remained low at approximately 10 $\mu g.l^{-1}$ compared to an adjacent control lake where chlorophyll *a* concentrations varied between 20-87 $\mu g.l^{-1}$.

The potential impacts of utilising ultrasonic irradiation to control cyanobacteria upon other aquatic organisms are hitherto largely unknown. Ahn et al. (2007) reported that larger organisms (fish and aquatic insects) did not seem to be affected by ultrasonication. Kotopoulis et al. (2009) found that acoustic pressures produced during ultrasonication surpassed the NATO Undersea Research Centre (NURC) rules and procedures by 35 dB, and cautioned the use of ultrasonication in aquatic environments.

The use of ultrasonic irradiation of cyanobacterial cells appears to offer a potential strategy for control of cyanobacterial growth in small lakes where lake water can be readily recirculated through an ultrasonic unit. Further research is required to determine how this strategy might best be applied to small lakes, and to the effects of ultrasonication on aquatic ecosystems and cyanotoxin release and decay.

7.3 SEDIMENT REMOVAL

The removal of the upper sediment layers from the lake bottom is commonly used strategy for reducing phosphorus loads from the sediments of internally-loaded, highly eutrophic lakes (Plate 3). One of the advantages in removing the upper sediment layer is that any cyanobacterial resting cells (e.g. akinetes) will be also be removed.

Plate 3 Sediment removal from an urban lake in Brisbane

Sediment removal should be considered a ‘last resort’ option for reducing the risk of cyanobacterial growth in urban lakes. Apart from being an extremely costly exercise, the removal of sediments from a lake may have a profound influence upon benthic organisms, including the removal of macrophytes and fauna from the lake. In addition, sediment removal is generally confined to small lakes due to the costs associated with the removal and disposal of the contaminated sediments.

The removal of upper sediment layers may not be an effective strategy for managing cyanobacteria in urban lakes, particularly where stormwater inputs contribute nutrient loads sufficient for cyanobacterial growth to occur. Lake systems with high submerged macrophyte cover are particularly susceptible to sediment removal, as the ensuing turbidity following the sediment removal may inhibit macrophyte recolonisation, which results in the reversion of the lake to an algal/cyanobacterial-dominated state.

7.4 SEDIMENT CAPPING

Sediment capping of *in situ* sediments involves the placement of a cover over the sediment to seal it off, thereby minimising the release of contaminants (nutrients) to the water column (Hart et al., 2003). Sediment capping may involve placing a physical barrier over the sediment (e.g.3 sand or gravel) or providing an active barrier (Hart et al., 2003).

Active barrier systems are generally pervious geochemical materials that are capable of actively demobilising contaminants in the sediment pore water by adsorption or precipitation processes. A number of active barrier systems have been tested in lakes including calcium carbonate, zeolites, modified clays and kaolin amorphous derivatives (Jacobs and Forstner, 1999; Hart et al., 2003).

Both sediment capping methods offer effective strategies for reducing phosphorus release from sediments. Generally, physical barriers need to be placed during construction or following drawdown of lake water levels. Active barrier systems can be very expensive to use, and there is little information available detailing the long-term effectiveness of these systems. The use of sediment capping in constructed urban lakes is unlikely to be a successful long-term strategy for managing cyanobacterial growth due to the influence of stormwater pollutant loads upon water quality through internal loading of sediments.

7.5 PHOSPHORUS PRECIPITATION AND INACTIVATION

The strategy involves the use coagulants to reduce the overall available load of phosphorus available for cyanobacterial growth. The coagulants are added to the lake water, where the coagulant binds to the bio-available phosphorus, then precipitates out of the water column as ‘flocs’. The flocs settle down to the sediments where the phosphorus is effectively removed from the water column and locked up in a form that is unavailable to phytoplankton. Coagulants commonly used to precipitate phosphorus within lakes include aluminium-sulphate, zirconium, iron and calcium salts, and some clay materials (Cooke, et al., 2005; Scheffer, 1998),

The effectiveness of coagulants in shallow lakes is generally low where macrophytes are present, or when the external loading of phosphorus (via stormwater) exceeds the phosphorus binding capacity of the coagulate (Welch and Cooke, 1999). The constant re-suspension of sediments due to wind action in shallow lakes also renders the use of coagulants ineffective.

7.6 ARTIFICIAL DESTRATIFICATION

Artificial destratification refers to the mixing of the hypolimnetic and epilimnetic layers within stratified lakes. Seasonal stratification generally occurs within deep lakes >5 m, although diurnal stratification will often occur in much shallower lakes.

Stratification occurs when a density gradient develops within the water column between the warmer upper layer (epilimnion) and the colder and denser bottom layer (hypolimnion). When lakes are stratified, the hypolimnion becomes depleted of oxygen which promotes reducing conditions and hence the release of nutrients such as ammonia and phosphorus from the sediments. Under stratified conditions, cyanobacterial species that are able to regulate their buoyancy are able to access both nutrients in the nutrient-rich hypolimnion and sunlight for photosynthesis in the epilimnion.

Aerators are commonly used in stratified lakes to continually mix the water column, thus eliminating the competitive advantage of cyanobacterial buoyancy over other non-buoyant algae (Reynolds et al., 1984) and reducing anoxic conditions at the sediment water interface. Whilst artificial destratification has been used successfully in Australia for controlling the release of nutrients from lake sediments, only limited success has been achieved controlling cyanobacteria (McAuliffe and Rosich, 1990). It is expected that improvements in destratification technology (e.g. mixer design) will increase mixing efficiency and improve the control of cyanobacterial growth (Brookes et al., 2002).

7.7 HYPOLIMNETIC WITHDRAWAL

Hypolimnetic withdrawal involves the removal of nutrient-rich hypolimnetic water from stratified lakes (Nurnberg, 1987). The removal of hypolimnetic water effectively decreases phosphorus concentrations in the epilimnetic layers of stratified lake systems, thereby limiting the potential for excessive cyanobacterial growth to occur (Nurnberg, 1987).

Hypolimnetic withdrawal is a widely used strategy for managing water quality (risk of cyanobacterial growth) in a range of typically deep lakes, varying in depth from 4.5 – 25 m (Cooke et al., 2005). Hypolimnetic withdrawal is only applicable to lake systems where the amount of discharged water can be replaced with sufficient inflow, so that the water level remains constant (Cooke et al., 2005).

Whilst hypolimnetic withdrawal is a relatively simple and cost effective strategy, it is unlikely that this technique could be used to effectively manage water quality within shallow urban lakes. Most urban lakes have a shallow depth profile and are relatively well mixed (in some cases shallow lakes may also exhibit diurnal stratification). Furthermore, hydraulic retention times in urban lakes are usually high, and sufficient inflow is unlikely to be available in order to maintain constant water levels.

Constructed urban lakes which regularly exhibit stratification may benefit from hypolimnetic withdrawal However, due consideration should be given to the disposal of the poor quality hypolimnetic water downstream or to the recirculation of the poor quality hypolimnetic water through a treatment system (wetland) to reduce nutrient levels.

7.8 ALGICIDES

Chemical control of algal and cyanobacterial growth within reservoirs has been a widespread water quality management practice for over 100 years (Burge et al., 2002).

The most common algicides used to control cyanobacteria in reservoirs are copper sulphate or chelated forms of copper. These are extremely effective as they act rapidly and are toxic to phytoplankton (and cyanobacteria) at concentrations that do not represent a health risk to humans (Steffensen et al., 1999). The chemical composition of the receiving water, particularly pH, alkalinity and dissolved organic carbon, influence copper speciation and complexation, thus greatly influencing copper toxicity. Copper algicides are economical, easy to apply and are generally considered to not to cause extensive environmental damage (Burge et al., 2002).

However, recent studies have shown that sediment-bound copper and other heavy metals may impact benthic communities, zooplankton, fish and invertebrates. Copper is not biodegradable and once it has entered the environment its potential toxicity is controlled by its speciation or physico-chemical form (Burge et al., 2002). One of the major problems associated with the use of copper-based algicides to control cyanobacterial growth is that copper lyses the algal cells, resulting in the release of intracellular toxins and other odorous compounds (if present) into the surrounding water. This can exacerbate health risks to humans and other animals, and further reduce amenity. Currently, there is little information on the minimum withholding period in relation to toxin degradation (Burge et al., 2002).

Other algicides used to control cyanobacterial growth in lakes include potassium permanganate, simazine compounds, chlorine, lime and barley straw (Burge et al., 2002; House and Burge, 2002). Potassium permanganate, simazine and chlorine are lethal to algal cells and may result in the release of cyanotoxins. Lime is used as an algicide to coagulate and precipitate algal cells from the water column, and does not appear to cause cell lysis and toxin release (Burge et al., 2002). Recent studies have shown that the use of lime is most effective in hard water lakes (Murphy and Prepas, 1990) However, lime dosing rates are quite high and this technique has not been extensively tested within Australian waters which are considered soft waters (Burge et al., 2002).

Considering the potential environmental and human health risks, the use of algicides are not recommended for the sustainable long-term management of cyanobacterial growth in urban lakes.

7.9 BIOMANIPULATION

The biomanipulation method is often referred to as “top-down control”, and refers to the management of higher trophic organisms to maximise grazing of algae, and therefore a reduction in cyanobacterial biomass in the lake system. There is no information on the use of biomanipulation to manage cyanobacteria in constructed lakes in Australia. In a review of 33 whole-lake biomanipulation experiments, Reynolds (1994) concluded that top-down effects on the lower trophic levels (phytoplankton) were not conclusive. Reynolds (2006) postulated that the manipulable linkages between phytoplankton, zooplankton, planktivorous fish and piscivores apply only to truly pelagic systems.

Boon et al. (1994) reviewed the potential use of top-down control of cyanobacteria and concluded that zooplankton assemblages in Australian waters are quite different in composition from those of the Northern Hemisphere, and are therefore unlikely to respond to manipulations of trophic structure in a similar way. The study concluded that the manipulation of trophic structure in Australian fresh waters to control cyanobacterial blooms is unlikely to be effective.

7.10 FLOATING MACROPHYTE BEDS

Floating macrophyte beds have been increasingly used to manage water quality in lakes where other methods of water quality treatment may be constrained. This is particularly the case in many golf courses, where high nutrient concentrations in the stormwater runoff often stimulate excessive algal growth in the waterways.

The use of floating macrophytes has been widely used within the United States for well over 20 years to treat wastewater and stormwater runoff (Reddy and DeBusk, 1987). Floating macrophyte beds can be effective for managing algal growth in lakes as they remove bioavailable nutrients from the water column, therefore reducing their availability to cyanobacteria or other algae. Floating macrophyte beds are relatively simple and cost effective, and regularly harvesting of the macrophytes serves to further reduce nutrient loads from the lake system.

Basil, tomato and silver beet plants growing in aquaponic units (floating plant beds) were recently trialled at Rio Tinto’s Kestrel Mine (Central Queensland) in order to control blue-green algal growth in a number of lake systems (Fabbro and Duivenvoorden, 2004; L. Fabbro pers. comm.). The research showed that the basil, tomato and silver beet plants were particularly effective at removing phosphorus and molybdenum from the water, which are required for cyanobacterial growth (L. Fabbro pers. comm.).

The use of floating macrophyte beds offers a range of potential options for managing water quality in constructed urban lakes. Floating macrophyte beds could be used *in situ* or within dedicated bays as part of a lake recirculation system.

7.11 SUB-SURFACE WETLANDS

Recirculating lake water through sub-surface flow wetlands may be an alternative option for removing cyanobacterial biomass from lake systems. Sub-surface wetlands are most commonly used to treat municipal sewage, and also wastewaters from agriculture, industry and landfill, and less commonly to treat stormwater runoff.

Sub-surface wetlands remove a wide range of pollutants including nutrients, suspended solids and organic matter from the water by fine filtration, adsorption and biological processes (uptake, aerobic/anaerobic degradation, and nitrification/denitrification) (Vymazal, 2005). More recently, hybrid systems comprising of free-water surface flow, vertical flow and sub-surface wetlands have been used in a staged manner to achieve higher treatment efficiency (Vymazal, 2005). No research has been undertaken on sub-surface wetlands and their influence on cyanobacterial biomass.

8 CONCLUSIONS

The alternative states model for shallow lakes provides a conceptual framework which may be used to develop a better understanding on the function of constructed urban lakes:

- Shallow lakes may be characterised by the presence of three alternative states:
    1. clear water state dominated by submerged macrophytes
    2. turbid water state dominated by mixed phytoplankton
    3. turbid water state dominated by cyanobacteria.
- The model suggests that the alternative states can exist over a wide range of nutrient concentrations, and that switches are required to trigger a shift from one state to another
- It is proposed that light availability mediated through water turbidity is one of the major determinants governing whether shallow lakes switch between macrophyte, phytoplankton and cyanobacterial dominated states
- The presence of alternative equilibria states in shallow lakes may potentially have profound implications for management strategies used to manage the development of cyanobacterial blooms
- Lake retention times may be used to control algal biomass via the physical displacement of algal cells from the lake system. The hydraulic displacement of phytoplankton cells occurs via two mechanisms:
    1. dilution known as flushing, and
    2. direct displacement known as wash-out.
- The alternative states model predicts that cyanobacterial control in shallow lakes may be achieved by increasing lake flushing rates (i.e. decreasing lake retention times), thus maintaining a stable phytoplankton/macrophyte equilibrium state
- There is little or no information within the literature that details the relationship between the hydraulic displacement of cyanobacterial biomass via episodic wash-out and lake retention times. In contrast, almost all contemporary research associated with the use of hydraulic displacement to manage cyanobacterial growth in lakes has involved the use of flushing to manage lake retention times.

A review of the proposed probabilistic approach (Burge and Breen, 2006) based upon lake retention (residence) times to manage cyanobacteria indicated:

- The critical cyanobacterial cell threshold adopted to derive the probabilistic approach is appropriate
- A major limitation to such a theoretical approach to modelling cyanobacterial growth is that the model is highly sensitive to the initial cell concentrations
- Other mechanisms intimately involved in the development of cyanobacterial blooms include the germination of akinetes (resting cells) and the growth of over-wintering cells. Therefore, the time required for cyanobacterial to rapidly establish bloom concentrations may be substantially shorter than predicted by the probabilistic approach
- The use of lake retention time criteria to manage cyanobacterial biomass within lakes is appropriate. However, it should be noted that the probabilistic approach represents a precautionary approach at best, and that no strategy will guarantee full protection against cyanobacterial blooms.

If the probabilistic approach is adopted, then it is recommended that:

- The retention time criteria are applied to the period of the year when the risk of cyanobacterial growth is considered high. In SEQ, cyanobacterial growth is possible throughout the year due to suitable water temperatures, and therefore retention time criteria should be applied for the entire year
- An exceedance of 95th percentile is adopted rather than the current 80th percentile
- The control of cyanobacterial biomass via lake retention time design criteria will be most likely facilitated by constant flushing of cyanobacterial biomass from the lake system.
- The control of cyanobacterial biomass via episodic washout (stormwater runoff) is unlikely to provide acceptable management of cyanobacterial blooms
- It is likely that hydraulic retention times of less than 20 days will be required for flushing to be effective at controlling cyanobacterial biomass in SEQ.

- The use of surface flow wetland systems to manage lake hydraulic retention times via constant flushing is deemed to be a feasible alternative, however a number of criteria must be satisfied in order for this strategy to be effective:
    - The majority of cyanobacterial cells (biomass) must be removed from the lake water whilst passing through the wetland
    - The ’treated’ water re-entering the lake must have turbidity and nutrient concentrations less than or equal to the *in situ* lake water
    - The lake water should be fully mixed (i.e. have high hydraulic effectiveness).

Further research is required to verify the effectiveness of wetlands in removing cyanobacterial biomass and to determine the minimum hydraulic retention times required to manage cyanobacterial biomass in urban lakes.

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