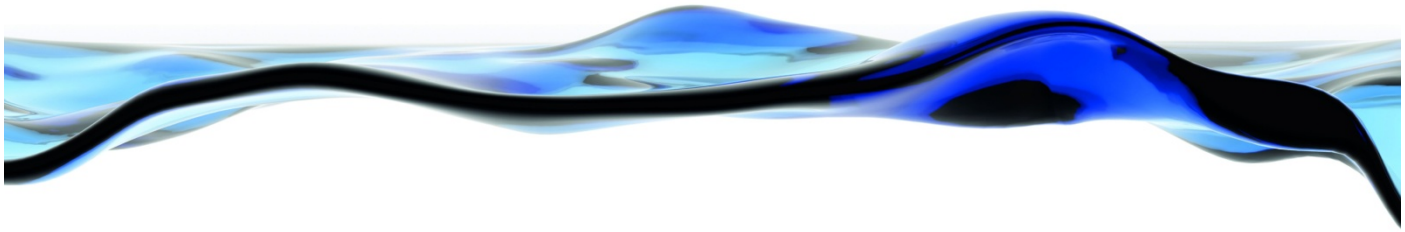
River Torrens Water Quality Improvement Trial – Summer 2011/2012



Goyder Institute for Water Research
Technical Report Series No. 12/4

GOYDER INSTITUTE FOR WATER RESEARCH

www.goyderinstitute.org

Goyder Institute for Water Research Technical Report Series ISSN: 1839-2725

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Citation

Brookes JD (ed) (2012) *River Torrens Water Quality Improvement Trial - Summer 2011/12,* Goyder Institute for Water Research Technical Report Series No. 12/4

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TABLE OF CONTENTS

LIST OF FIGURES

CHAPTER TWO

CHAPTER THREE

CHAPTER FOUR

CHAPTER FIVE

LIST OF TABLES

CHAPTER ONE

CHAPTER TWO

CHAPTER THREE

CHAPTER FOUR

CHAPTER FIVE

Executive Summary

The management of rivers, lakes and reservoirs is a difficult task which demands considerable effort to restore deteriorating catchments, water habitats and water quality. The Torrens Lake, in Adelaide South Australia, has similar problems to many lakes worldwide including catchment clearing, nutrient focussing, flow regulation, water extraction for irrigation and cyanobacterial blooms.

It is unlikely that any management strategy will reduce cyanobacterial numbers to zero and therefore the most realistic strategy would be one that controls the cyanobacterial population below the guideline concentration for recreational exposure. Artificial destratification is likely to be ineffective at controlling cyanobacteria in the Torrens, as the system is shallow and mixing inefficient. Nutrient control will constrain cyanobacterial biomass but the Torrens catchment delivers high loads of nutrients to the lake and it will be decades before control measures limit growth to the desired thresholds.

The aim of this study was to investigate whether a controlled upstream release of water could effectively dilute the population and control cyanobacteria numbers below a threshold.
For a growth rate of 0.4/day, which is a typical exponential growth rate of cyanobacteria in the Torrens Lake, it can be concluded that a diluting flow of at least 10% per day would be required to have noticeable impact on the cyanobacteria population. With a starting cell concentration of 100 cells/mL, a growth rate of 0.4/day and a diluting flow of 10%, the cell concentration after 20 days would be 74,420 cells/mL, which is below the critical threshold for cell numbers.

A trial was conducted in the summer of 2011/12. Flows were released from Hope Valley Reservoir in response to the level of cyanobacteria present in the Torrens Lake with the aim of keeping cyanobacterial numbers in check and avoiding lake closure. During a period when upstream water was unavailable the cyanobacteria grew exponentially and the population exceeded the threshold concentration. Under similar meteorological conditions later in summer the population was maintained by a combination of controlled amenity flows and rain event inflows.

The study concluded that:

- The simple model of growth and dilution is a reasonable predictor of biomass change at low cell density, however population decrease is underestimated by the predictive model
- Dilution is possibly greater than predicted because high concentrations of cells near the surface are washed out of the lake with the overflow of water over the weir
- Nutrient or light limitation possibly contributed to the observed decrease in growth rates; nutrient limitation as the population expands and consumes the available resources during periods of rapid growth, and light limitation following rain events as coloured material is washed into the lake.
- The results of the trial suggest that if dilution flows are released early enough, the size of the cyanobacterial population can be controlled. However, there is a reliance on rain events to flush the system and dilute the resident cyanobacterial population. On average, the flow

return interval analysis suggests that rain events occur frequently enough in summer for this strategy to be effective. However, in a variable climate like that observed in Adelaide, there may be incidences of very long periods between significant rainfall events. This may reduce the confidence in rain events to reset the population.

- The dilution flow was not observed to have any impact on the freshwater fish community.

The use of flows to control cyanobacterial growth shows promise as an event management technique to control the growth of cyanobacterial biomass. In the absence of any other acceptable strategy to control the development of a cyanobacteria bloom during an event in the Torrens Lake, it is recommended that the dilution flow trial be repeated to conclusively determine the effectiveness of this control strategy.

If a second trial were to be conducted, the findings of this study recommend that dilution flows should commence when cell concentrations approach 1000 cells/mL. If dilution flows commence at higher cell concentrations, it is difficult for dilution to adequately mitigate the exponentially expanding population. The coordination of the 2011/12 trial demonstrated that it was feasible to deliver flow in response to cell counts two days prior, which was observed to be appropriate considering the rate of growth, and this should again be considered in the operational planning of flow releases.

At a catchment scale, the nutrient concentrations of water entering the lake are still high and so can support growth of high cyanobacterial biomass. Nutrient control is considered the best long-term solution to the problems associated with eutrophication and it is recommended that there is a continuation of the remedial works for nutrient control in both the urbanised and peri-urban catchments.

Acknowledgements

The interdisciplinary nature of this work and the logistics and teamwork required for sampling and flow delivery means there are many people to thank.

Alan Ockenden conceived this project and was instrumental in securing funding, coordinating the logistics, and managing the interaction with various government agencies and councils. The Adelaide and Mount Lofty NRM Board (AMLRNRMB) also provided support for the assessment of transmission losses and flow travel times for controlled releases to the River from Gorge Weir and Hope Valley Reservoir for the River Torrens Water Quality Improvement Flow Trial in the summer of 2011/12 by Richard Clark and Associates. The findings of this assessment are reported in a Technical Paper at the AMLRNRMB (Clark, 2012).

Kym Bowden from SA Water was a critical part of the project team coordinating the release of water from upstream and pumping water downstream.

Neil Power from the Department for Water provided excellent support in project development and his enthusiasm for the work is appreciated.

Funding and in kind support from the Goyder Institute, SA Water, The Adelaide City Council, The Adelaide and Mount Lofty Natural Resource Management Board,

Sampling, cell counts and nutrient determination were undertaken by the Australian Water Quality Centre.

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Part A: Synthesis of the Torrens Lake Water Quality Improvement Trial

1. Introduction

The management of rivers, lakes and reservoirs is a difficult task which demands considerable effort to restore deteriorating catchments, water habitats and water quality. The Torrens Lake, in Adelaide South Australia, has similar problems to many lakes worldwide due to the impacts of urban catchments, nutrient loads, flow regulation, water extraction for irrigation and episodes of toxic algal blooms, known as cyanobacteria. Torrens Lake is used predominantly for recreational activities such as fishing, rowing and paddle boating, and so maintaining the lake 'open' for these activities is a priority.

A combination of warm temperatures, nutrients, light and stratification are ideal conditions for algal blooms to occur, and these are characteristic of the Torrens Lake. The ongoing occurrence of algal blooms in the Torrens Lake has prompted exploration into numerous management techniques to control these blooms. Long-term strategies, such as catchment management interventions, are ongoing but may take decades for the full benefits of these activities to be realised in improving lake water quality. The assessment of the various in-lake techniques also revealed that it would be challenging to control the growth of cyanobacteria. Artificial destratification is challenging in shallow environments where cyanobacteria can still access adequate light. Nutrient control is challenging when there is continuous renewal from the catchment.

One strategy that is being considered is the use of controlled releases from upstream catchments. This strategy relies upon dilution flows flushing the cyanobacteria from the system and thereby limiting the accumulation of cyanobacteria to form a bloom. There are several challenges with this strategy, not least of which is achieving the desired dilution given the likely density difference between the inflow and lake water.

A trial was undertaken during the summer of 2011/12 to determine the effectiveness of using dilution flows as a strategy to control cyanobacteria in the Torrens Lake. The aim of this dilution flow trial was to monitor the release of water from an upstream storage to dilute cyanobacteria in the Torrens Lake. In addition, a fish monitoring component was established to evaluate the effect of flow management on fish populations within the lower Torrens and Breakout Creek. This component was developed to provide a baseline survey of fish communities, against which post-

flow patterns could be compared and to conduct a survey during flow releases to determine any short-term responses or impacts.

2. Methods

The methods described below are a summary of the detailed methodology contained in Part B of The Goyder Institute Technical Report 4/12 '*River Torrens Water Quality Improvement Flow Trial – Summer 2011/2012*'.

2.1 Site description

The sampling sites used in this monitoring program were the same as those used in the regular monitoring program for algal counts by the City of Adelaide. This enabled consistent datasets, comparison with historical records and a representative coverage of the lake.

Location of monitoring sites 1-7 on the Torrens Lake, Adelaide. Site 1 – Torrens Lake weir, Site 2 – Morphett Street Bridge, Site 3 – Elder Park, Site 4 – King William Road Bridge, Site 5 – University Footbridge, Site 6 Frome Road Bridge, Site 7 – Hackney Road Bridge.

2.2 Hydrodynamics

The premise of controlled upstream water releases to control cyanobacteria in the Torrens Lake is that there is sufficient dilution and loss of cells downstream to overcome growth and biomass expansion in the lake. To measure mixing processes in the Torrens Lake to inform the volume of dilution required, a vertical profile of temperature was monitored throughout the depth of the water column in the lake and upstream at Hackney Road Bridge.

2.3 Algal Biomass and nutrients

Algal biomass was sampled twice a week at all seven sites. Samples for nutrient analysis were collected weekly at four sites: Site 1 - Torrens Lake Weir, Site 2 – Morphett Street Bridge, Site 4 – King William Road Bridge, Site 6 - Frome Road Bridge. All laboratory analyses were performed by the Australian Water Quality Centre, a NATA accredited laboratory.

2.4 Predicting Cyanobacteria Growth Rates

To aid in evaluating the flow trial, a predictive model of cyanobacterial growth and dilution was applied to conditions that occurred in the Torrens Lake in January and February 2012. One species of cyanobacteria that is of concern is *Anabaena circinalis,* with the predicted growth rate of this species being compared against the measured population at Site 1 (Torrens Lake) and the average cell concentration as measured at all seven sites.

2.5 Fish Survey

A baseline survey of fish populations on a transect between the city weir and Breakout Creek was undertaken during December 2011. Surveys were then undertaken in response to flow releases to identify any accumulations of fish below barriers, or to detect any directional movement of fish in response to flows and any signs of toxic stress.

A VEMCO acoustic tracking array and twenty-five carp implanted with VEMCO acoustic transmitters were used to monitor the response of carp to the dilution flow within Torrens Lake. The carp response to the dilution flow was compared to carp behavioural patterns recorded during periods of natural flow and no-flow to determine if there were any differences. These patterns have been captured as part of a broader tracking study of invasive species being conducted by SARDI in collaboration with the Adelaide City Council.

3. Results

The results described below are a summary of those provided in detail in Part B of The Goyder Institute Technical Report 4/12 '*River Torrens Water Quality Improvement Flow Trial – Summer 2011/2012*'.

3.1 Hydrodynamics

Temperature conditions in the Torrens Lake are ideal for cyanobacterial growth. During the period from mid December 2011 to 30 March, 2012 the water temperature at Site 2 (Morphett St Bridge)

was less than 20°C for only six days. Water temperatures at the surface reached as high as 31°C and the diurnal heating of the surface water led to regular stratification. Stratification was not evident on days when the water column was cooling. Similar temperature patterns were observed at all sites, but there were differences in the maximum and minimum temperature and the magnitude of daily stratification. The differences in temperature will drive currents (Monismith et al., 1990; Wells and Sherman, 2001) as warmer, less dense water floats over cooler water, and eventually sinks.

The nocturnal mixing will entrain cells and distribute the phytoplankton community throughout the water column. This will be counteracted by the buoyancy of cyanobacteria which generally float. However, they tend to mix back into the water column when wind speeds exceed $3ms^{-1}$ (Webster and Hutchinson, 1994).

3.2 Flow

Flow in the River Torrens was recorded at two sites, upstream and downstream of the Torrens Weir. The upstream site was situated immediately downstream of the confluence of Second Creek and the Torrens River. The site downstream of the Torrens Lake was at Holbrooks Road. Flow until 13 January 2012 was a function of rainfall and catchment runoff. On 13 January 2012 a 20 ML/day flow release from Hope Valley Reservoir commenced in an attempt to dilute increasing cyanobacterial numbers in the Torrens Lake. Flow releases had been requested earlier, when cyanobacteria numbers approached 1,000 cells/mL but logistical issues with water delivery meant this was not possible. Flow was increased to 40 ML/day on 18 January 2012 but was terminated on 20 January 2012 because of logistic difficulties with flow delivery.

In response to development of cyanobacterial numbers again in early February 2012, flow was increased to 20 ML/day and then maintained at 40 ML/day from 15 to 29 February 2012. The controlled release of water from Hope Valley was ceased when large summer rain events washed out the cyanobacteria from the lake.

3.3 Algal Biomass

Algal biomass was determined twice weekly with a particular focus on cyanobacteria. The total cyanobacteria cell counts were dominated by *Anabaena circinalis*, which remained at low concentrations through December 2011. Water temperature began to increase in January 2012

and was accompanied by rapid growth of *Anabaena circinalis,* which peaked at 161,000 cells/mL. Additional water was not available at this time to control the population and it remained at fairly high concentrations until a stormwater inflow of approximately 80 ML on 29 January 2012 followed by a significant stormwater inflow of 200-250 ML on 5 February 2012. The inflow resulting from this rain event dramatically decreased the *Anabaena circinalis* population. Similar temperature conditions occurred again in February 2012. The combination of flow releases and rain events during this latter period tended to dilute the cyanobacterial population and total abundance was maintained below 20,000 cells/mL.

The cyanobacterial population was dominated by *Anabaena circinalis* with relatively minor contributions from other species such as *Microcystis aeruginosa*, *Microcystis flos-aquae*, and *Planktothrix*. The distribution of cells in the lake was biased towards the downstream-weir end. This could be due to a combination of preferential growth in the broad main body of the lake and the general transport of cells from the upstream end of the lake and their accumulation downstream.

3.4 Evaluation of the flow trial against predicted growth rate and dilution

The predictive model of cyanobacterial growth and dilution, described in Chapter 2 of the report, was applied to conditions that occurred in the Torrens Lake in January and February 2012. Two growth rates were simulated with three flow scenarios, including no dilution, 10% dilution and the actual flow rate. Cells were initially observed at a concentration of 235 cells/mL and increased exponentially over a three weeks period. The growth and dilution model scenarios that best describe the observed population increase are a growth rate of 0.3 day^{-1} and a 10% dilution of the actual flow.

The same modelling approach was applied to predict growth commencing 19 January 2012 with a starting cell concentration of 82,900 cells/mL. The trajectory of growth in the model predictions was consistently increasing with minor decreases in the population size associated with inflows. It was observed that the large inflow event on 5 February 2012 significantly decreased the population size. It is possible that there are other factors influencing the population size other than homogeneous dilution and a constant growth rate. For example, availability of nutrients has previously been observed to limit growth of cyanobacterial blooms in the Torrens Lake.

3.5 Efficiency of diluting flows

A concern with releasing water for dilution is that density differences between the inflowing water and the ambient lake water will prevent complete mixing and thus provide poor dilution. The Estuary and Lake Computer Model (ELCOM) results presented in Chapter 2 suggests that complete mixing occurs as the river flow enters the lake. The observed temperature in the water profile do not show indications of a persistent underflow or overflow and that the lake completely mixes every night.

Observed variability in flow velocity throughout the system is described in more detail in Chapter 2 of Part B of this report.

3.6 Frequency of flushing events

The flow strategy for managing cyanobacteria in the Torrens Lake is able to dilute the population but at the flow volumes available, there is still a reliance on periodic natural high flow from rain events. To investigate how often adequate diluting rain events occur, the historical flow record at Holbrooks Road gauging station was analysed. Although it receives inputs downstream of the weir, this gauging station was considered most representative of flow through the Torrens Lake. Flow magnitude is generally higher in winter and spring than in summer and autumn. Flow is less than 10 ML/day for about half of the year.

The return period between flows of different magnitude is of interest as the natural runoff is required to reset the system and provide a major dilution event to control the cyanobacterial population. The entire available flow record (1978 – 2011) was analysed to calculate the annual average return interval for different flow magnitude and the maximum return interval over the entire period. Summer flows (Dec, Jan, Feb) in the order of 80 ML/day would be required for a 20% dilution and 100 ML/day for a 25% dilution. These flows have an average summer return interval of 24.6 days and 27 days for 80 ML/day and 100 ML/day, respectively. Modelling presented in Chapter 2 suggests that a diluting flow of 40 ML/day can control cyanobacterial populations for about 3 weeks before the biomass threshold is exceeded. Therefore, rain event inflows approximately every 21 days could help reset the system. This is the same order of magnitude as the return interval of 80-100 ML/day flows and so it appears feasible that this would work synergistically with controlled diluting flows.

3.7 Nutrient concentrations and growth

Nutrient concentrations give a good indicator of the potential of the waterbody to support algal growth. Phosphorus, in particular, often has a strong correlation with phytoplankton biomass (Vollenweider, 1975; Sherman et al., 2000; Linden et al., 2004). Nutrients were sampled and measured at three sites but only the Weir site is presented in Part B, Chapter Three of this report. The dissolved nutrients (oxidised nitrogen, filterable reactive phosphorus and ammonium) were typically in excess to demand. A decrease in concentrations of these nutrient forms was evident on 16 January and 23 January 2012 as the cyanobacterial community grew exponentially and utilised the available nutrient resources. Nitrate and nitrite dropped from 0.23 mg/L to 0.007 mg/L, Ammonia from 0.265 mg/L to 0.005 mg/L and filterable reactive phosphorus from 0.04 to 0.008 mg/L. These nutrient concentrations recovered with a rain event inflow in late January.

The water quality of inputs and outputs from the Torrens Lake is also of interest for management. A transect of sites from Hope Valley Reservoir to the sea was established for regular monitoring of total phosphorus. The phosphorus concentration increased with downstream travel from the Hope Valley Reservoir Outlet to the gauging station at second creek and Site 6 (Frome Road Bridge). There was a gradual attenuation of phosphorus downstream of the weir except at the Grange Lake, which showed the highest total phosphorus concentrations.

3.8 Fish Baseline Survey

The Torrens fish fauna is composed of endemic, River Murray translocations and exotic species. Part B, Chapter Five provides a summary list of total species catches and lengths, with the observed spawning status, disease and other ecologically relevant information also noted.

All endemic species captured had been previously recorded in this reach (McNeil *et al* 2011a). Of the species observed, four are considered endemic to the Torrens; flathead gudgeon, jollytail, congolli and western blue spot goby. The flathead gudgeon was the most abundant and ubiquitous fish of the study. It appeared in multiple size classes with high rates of gravid and spawning individuals noted. These are a hardy and versatile fish species that thrive throughout their range.

The remaining three species share an ability to utilise oceanic life stages (diadromy and amphidromy). Whilst the common jollytail and congolli are widely distributed throughout the

Western Mount Lofty Ranges they were almost entirely absent from the River Torrens during this survey. These diadromous fishes require oceanic connectivity and juveniles have previously been recorded accessing the river reach using the Torrens Fishway at the mouth of Breakout Creek (McNeil *et al* 2010). Currently, however, these diadromous species appear unable to pass the two recently constructed weirs between Breakout Creek and Henley Beach Road (Schmarr *et al* 2011). Three Murray-Darling species that have been translocated into the reach (McNeil and Hammer 2007) were identified during the survey, consistent with those captured previously in the reach (McNeil *et al* 2011a). Carp gudgeon occurred along the entire length of the lower Torrens in large numbers with high proportions of gravid and spawning individuals. This species is doing well under the current regime.

The Murray rainbowfish was observed in the larger and deeper pools of the Torrens in large numbers. Several individuals were in spawning condition. This species was commonly observed suffering from anchor worm (*Lernia sp.*) which is not uncommon. This species are doing well under the current regime.

The eel-tailed catfish is protected within the Murray River, but has adopted the role of top-predator in the Torrens and appears to be thriving. Multiple size classes were observed including a cohort of juveniles (20-40mm) and 25 individuals greater than 300mm.

Three exotic species were identified. European carp were observed in unexpectedly low numbers during the survey, although very large numbers of juvenile carp have been previously recorded in the reach, and adult carp are commonly captured (McNeil et al. 2011a). Goldfish were only identified in the stretch below the Breakout Creek Weirs. In this small stretch of river a single cohort was observed in remarkable densities (5344 individuals). In this instance the barriers appear to be playing a positive role, isolating this population from invading further upstream. The response of goldfish to strong flow is unclear. One hypothesis is that strong winter flows may wash a large proportion of this population out to sea. Gambusia are a remarkably tolerant and ubiquitous fish across their introduced range. Numbers were steady across most stretches of the river with a slight preference for larger, calmer pools. Gambusia were regularly observed to be pregnant with dissection showing well developed young within female fish.

4. Discussion

The Torrens Lake amenity flow trial was conducted over a range of meteorological and hydrological conditions. Even though the trial period was of limited duration, it allowed the cyanobacterial growth to be measured under low and high flow conditions. Without dilution flows in early January 2012, the population rose to 'bloom' proportions and forced the closure of the lake to recreational users. Similar climatic conditions to those that gave rise to this 'bloom' were experienced in late January 2012 but the population growth was controlled with flow releases commencing in early February 2012.

The explosive growth without dilution in late January 2012 decreased due to nutrient concentration reductions that were considered limiting. The diluting flows in February 2012 under similar meteorological and nutrient conditions seemed to control the phytoplankton growth, but it is unclear whether other factors may have assisted in controlling the growth rate. The simple model presented in Part B, Chapter 2 suggests that growth can only be controlled for about 3 weeks. Rain event inflows are required to significantly dilute the population and reset the population size to a level that can be maintained with diluting flow.

Natural inflows from rain events are a critical part of the strategy for cyanobacterial control by dilution. The dilution flow with programmed releases from Hope Valley Reservoir appear to exert some control on the size of the cyanobacterial population but another season of trials would aid in conclusively determining how successful dilution flows are likely to be in controlling cyanobacterial biomass in the long-term. The dilution flow strategy as an option for cyanobacterial control is contingent upon water availability from Hope Valley Reservoir or Kangaroo Creek Dam.

The simple model of growth and dilution predicted the biomass change well at low cell density but the population decrease was underestimated in late January and early February 2012. There are several possible reasons for this:

- Dilution was greater than predicted by the simple model
- Growth rate was lower than predicted.
- Another unanticipated feature (eg. Cyanophages) was contributing to population decline.

Greater than expected dilution could occur due to the fact that the flow over the top of the weir is disproportionately diluting cyanobacteria, which are buoyant and so are at higher concentration near the surface.

Typically algal growth may be limited by light or nutrients. Nutrient concentrations decreased as the population expanded and had reached concentrations that could be considered limiting to the growth rate. Light limitation also affects algal growth in the Torrens Lake, particularly after rain events that tend to wash in a significant amount of black material from the urbanised catchment.

Concerns about the flow acting as a discrete riverine intrusion and short-circuiting the lake were not supported by the temperatures in the water profile, or the apparent dilution rates.

5. Conclusions

The main findings of the Torrens Lake amenity flow trial during the summer of 2011/12 were:

- The simple model of growth and dilution is a reasonable predictor of biomass change at low cell density, however population decrease is underestimated by the predictive model
- Dilution is possibly greater than predicted because high concentrations of cells near the surface are washed out of the lake with the overflow of water over the weir
- Nutrient or light limitation is a possible contributor to the observed decrease in growth rates; nutrient limitation as the population expands and consumes the available resources during periods of rapid growth, and light limitation following rain events as coloured material is washed into the lake.
- The results of the trial suggest that if dilution flows are released early enough, the size of the cyanobacterial population can be controlled. However, there is a reliance on rain events to flush the system and dilute the resident cyanobacterial population. On average, the flow return interval analysis suggests that rain events occur frequently enough in summer for this strategy to be effective. However, in a variable climate like that observed in Adelaide, there may be incidences of very long periods between significant rainfall events. This may reduce the confidence in rain events to reset the population.
- The dilution flow was not observed to have any impact on the freshwater fish community.

6. Recommendations

The use of flows to control cyanobacterial growth shows promise as an event management technique to control the growth of cyanobacterial biomass. This is supported by both the modelling of growth and dilution, and the results of the field trial. However, this project was conducted for on a very short period of time and dilution flows were only possible in February 2012.

In the absence of any other acceptable strategy to control the development of a cyanobacteria bloom during an event in the Torrens Lake, it is recommended that the dilution flow trial be repeated to conclusively determine the effectiveness of this control strategy.

If a second trial were to be conducted, the findings of this study recommend that dilution flows should commence when cell concentrations approach 1000 cells/mL. If dilution flows commence at higher cell concentrations, it is difficult for dilution to adequately mitigate the exponentially expanding population. The coordination of the 2011/12 trial demonstrated that it was feasible to deliver flow in response to cell counts two days prior, which was observed to be appropriate considering the rate of growth, and this should again be considered in the operational planning of flow releases.

At a catchment scale, the nutrient concentrations of water entering the lake are still high and so can support growth of high cyanobacterial biomass. Nutrient control is considered the best long-term solution to the problems associated with eutrophication and it is recommended that there is a continuation of the remedial works for nutrient control in both the urbanised and peri-urban catchments.

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Part B: Research Methodology, Results and Key Findings

Chapter 1: Options for cyanobacterial management in the Torrens Lake, Justin D. Brookes

Chapter 2: The Feasibility of dilution as a solution, Justin D. Brookes

Chapter 3: Torrens Lake Water Quality Improvement Trial, Justin D. Brookes, Alex Payne, Kane T. Aldridge, Kym Bowden, Alan Ockenden

Chapter 4: Hydrodynamic Modelling of amenity flows, Robert Daly, Justin D. Brookes,

Chapter 5: Flow trial – Fish Monitoring, Dale McNeil, David Schmarr, Rupert Mathwin, Leigh Thwaite

Chapter One: Options for cyanobacterial management in the Torrens Lake

1.1 Introduction

The management of rivers, lakes and reservoirs is a difficult task which demands considerable effort to restore deteriorating catchments, water habitats and water quality. Water quality problems have been detailed for decades by scientists, managers and governments, however, the solutions to these problems are not as forthcoming. Major water problems still to be addressed include eutrophication, salinity, pesticide contamination, pathogens, cyanobacteria and the maintenance of environmental flows.

Catchment and river authorities are becoming increasingly aware that rivers, lakes and reservoirs cannot be managed in isolation; rather, these features require management within the context of a landscape. The landscape is invariably composed of a variety of land-types and sustains a variety of land-uses, which contribute different water quality.

There is considerable risk that land management will impact water quality and aquatic habitats. If viewed in an economic context, these impacts could be termed 'externalities', which are things that influence the welfare of individuals or a community through a non-market process. There is no market feedback from the person who experiences the loss or gain to the person who creates it (Young, 2000). Externalities in water are often difficult to manage as there may be considerable temporal and spatial separation between the 'event' and the 'impact'. Additionally water is often the jurisdiction of several different individuals or agencies as it travels from catchment to tap; the same parcel of water may at some stage be the responsibility of land-holders, catchment management boards, shire councils, irrigators, water supply and distribution companies, and several state government agencies.

The management of catchment and externalities requires a duty of care and the adoption of policy to enforce sustainable land-use. However, water quality problems require immediate action and managers are forced to attempt remedial strategies at the lake level to improve water quality for recreation or potable supply.

The Torrens Lake, South Australia, has similar problems to many lakes worldwide including catchment clearing, nutrient focussing, flow regulation, water extraction for irrigation and cyanobacterial blooms. In this chapter water quality problems are highlighted, management options to improve water quality are summarised, and the 'in lake' strategies to control cyanobacteria are critically reviewed for the Torrens Lake. Several of the options have been trialled and the relative success of these is assessed. The aim is to review options for cyanobacterial control and assess what is possible in the shallow, urban Torrens Lake.

1.2 Study site

The Torrens Lake is an ornamental urban lake situated on the River Torrens and marks the northern boundary of the Adelaide Central Business District, South Australia. The lake was formed by the construction of a weir across the River Torrens in 1881. It has a surface area of 15 ha and a capacity of 400 ML. The mean water depth is 2 m but ranges from approximately 1.5–6 metres. Twenty kilometres upstream of the Torrens Lake is Kangaroo Creek Dam, a large storage reservoir which receives water from the local catchment and is supplemented by water pumped from the River Murray. The River Torrens, downstream of Kangaroo Creek Dam, is also fed by six creeks that originate in the Mt. Lofty Ranges. Water is extracted from the River Torrens at the Gorge Weir and piped to Hope Valley Reservoir. The overflow or scour water from Hope Valley Reservoir is returned to the River Torrens via a shallow overflow creek. Clearing of vegetation and urbanisation of the catchment has meant that storm-water, which historically was intercepted by vegetation, drains rapidly into the river during storm events considerably increasing flow and disrupting thermal stratification. As a consequence the Torrens Lake can switch from operating like a lake to a river and back to a lake again within 24 hours (Ganf *et al*., 2000).

1.3 History of cyanobacteria and management of the Torrens Lake

Microcystis aeruginosa was first observed in the Torrens Lake in 1998 and developed into an extensive 'bloom', forcing the closure of the lake. A water release of 525 ML (at 150 ML/day) in January 1998 was used to disperse the cyanobacterial bloom. However, there were concerns regarding the fate of toxins transported downstream, the environmental impact of such a 'flush' and the cost of water which was diverted from potable supply. The water release appeared to flush the cyanobacteria from the system and was considered successful. A dredging project to

remove contaminated sediment was undertaken in 1997/98 but this ceased during the period of the algal bloom.

In January 1999 a similar cyanobacterial bloom developed but there was no management intervention. At the Torrens weir site, permanent stratification was a common feature, with a thermal gradient persisting between 2.5 and 5m for the periods January 1-7 and 9-18. Thermal stratification was punctuated on January 8 with isothermal conditions associated with a 100 ML flow from a rainfall event The rainfall event increased filterable reactive phosphorus (FRP) from <0.005 mg/L to between 0.036 and 0.089 mg/L. The nutrient load from the storm event, coupled with nutrient release from sediment, supported the *Microcystis* bloom, which reached a maximum of 420,000 cells/mL (Ganf *et al*., 2000). Diurnal stratification ensued and there was a decline in *Microcystis* cell numbers over the following two weeks to < 10,000 cells/mL. Ganf *et al*., 2000 calculated the internal nutrient load to be in the order of 4.2–33 mg P/m^2/day. The *M. aeruginosa* numbers rose to high levels and the lake was closed but the population 'crashed' rapidly when it had exhausted its nutrient stores (Ganf *et al*., 2000).

In January and February 2000 artificial destratification and controlled water releases were trialled and the physical, biological and chemical properties of the lake were closely monitored. The following section theoretically considers the applicability of each strategy to control thermal stratification and cyanobacteria and the results from the trial are analysed. Since this initial study numerous mixing devices have been installed in the lake and extensive upstream works carried out, including constructed wetlands and catchment management.

1.4 Assessing the management options

It is unlikely that any management strategy will reduce cyanobacterial numbers to zero, therefore the most realistic strategy would be one which controls the cyanobacterial population below the guideline concentration for recreational exposure; 20,000 cells/mL (WHO, 1999).

Manipulation of the physical environment is the most widely used and effective 'in lake' strategy to control cyanobacteria. In rivers and weir-pools the physical environment can be manipulated by artificial destratification or flow regulation. The two mechanisms by which destratification can limit cyanobacterial growth are:

- increasing the depth to which algae circulate, and inducing light-limitation, or
- reducing the internal nutrient load and the nutrient pool available for growth.

1.5 Artificial destratification: Mixing, light penetration and cyanobacterial growth

The light harvesting potential of pelagic cyanobacteria and eukaryotic micro-algae is determined by their photo-physiology and the hydrodynamics and optical properties of the water-body. Typically, the bloom-forming species such as *Anabaena* spp and *Microcystis aeruginosa* are confined to water bodies where the euphotic depth (1% surface irradiance) is 0.5-3.5 times the mixed depth (Reynolds and Walsby, 1975).

The light dose algae would receive whilst suspended in a water column with a mixed depth (Zmix) can be calculated and the growth rate inferred from growth irradiance curves.

The daily solar radiation was measured at Kent Town, approximately 2 km from the Torrens Lake. The average daily short wave radiation for January 1999 was 335 W/m^2 (maximum 393 W/m^2 on 12 January 1999), which is 58 mol photons/m^2/d of photosynthetically available radiation (400-700nm). This period was used for assessment because quality meteorological data was available for the lake, which was not available in 2012 due to failure of the meteorological station.
The instantaneous solar irradiance at the water surface (Io) can be expressed as a function of the maximum irradiance at solar noon (Imax) and the daylength (DL)

$$\mathbf{Io = Imax \times \sin(\pi/DL)}$$

The sum of Io calculated at ten minute increments for a day length of 15 hours and Imax of 1750 µmol/m^2/s gives a daily average PAR of 57.55 mol/m^2/d which is approximately the measured monthly average.

The amount of light which is reflected from the water surface is dependent upon the solar elevation which is a function of the day of the year, the time of day and the geographical latitude (Kirk, 1994; Walsby, 1997). The equations to determine the reflection (r) and sub-surface irradiance are given in Walsby (1997). Further corrections for reflectance can made to account for wind roughening of the water surface (Walsby, 1997), however, the smooth water reflectance

value was considered satisfactory for this model of light dose in different Zeu/Zmix conditions. Having accounted for reflection from the water surface the average sub-surface daily light dose for January was 47.8 mol/m^2/day.

The daily light dose an alga would experience is also determined by the mixing characteristics of the waterbody and can be calculated using the equation of Riley (1957):

$$I_{ave.} = I_u (1-e^{(-kd*Zmix)})/(kd*Z_{mix})$$

where I_u is the sub-surface irradiance, kd-the attenuation coefficient and Z_{mix} is the mix depth.

For the Torrens Lake in January 1999 the mean light dose experienced by algae circulating within the euphotic zone was 10.28 mol/m^2/day. The daily light dose exposure under different physical conditions of light penetration and mixed depth is given in Table 1.

Table 1. Modelled mean daily light dose (mol/m^2/day) experienced by phytoplankton circulating through various mixed depths in water-bodies with different euphotic depth and a sub-surface light intensity of 47.8 mol/m^2/day.

	Euphotic depth (m)			
Mixed depth (m)	**1**	**2**	**3**	**4**
1	10.28	18.68	24.43	28.39
2	5.19	10.28	14.85	18.68
3	3.46	6.91	10.28	13.40
4	2.60	5.19	7.77	10.28
5	2.08	4.15	6.23	8.28
6	1.73	3.46	5.19	6.91
7	1.48	2.97	4.45	5.93
8	1.30	2.60	3.89	5.19
9	1.15	2.31	3.46	4.61
10	1.04	2.08	3.11	4.15

The light attenuation coefficient and euphotic depth of the Torrens Lake are presented in Table 2. In January the euphotic depth was 1.85 m. The water depth of the Torrens Lake ranges between 2-6m and therefore phytoplankton circulating through the entire water column would have experienced a light dose of at least 4 mol/m^2/day and about 9 mol/m^2/day in much of the lake.

Table 2. Monthly average attenuation coefficient (Kd ± SD, n=5-6) at Zoo site, Torrens Lake, 1998/99.

Month	Kd (m^{-1})	Euphotic depth (m)
October	2.46 ± 0.74	1.87 ± 0.56
November	1.23 ± 0.65	3.74 ± 1.98
December	1.86 ± 0.31	2.48 ± 0.41
January	2.49 ± 0.24	1.85 ± 0.18
February	2.32 ± 0.27	1.98 ± 0.23
March	5.78 ± 2.21	0.80 ± 0.3

1.6 Light dose and growth rate of *Microcystis aeruginosa*

The growth of *Microcystis aeruginosa* in response to a series of light doses is given by Reynolds (1997; fig 26). The minimum light dose required to maintain a population of *Microcystis aeruginosa* is 1 mol photon/m^2/d. The growth rate increases with increasing light dose until maximum growth at about 7 mol photon/m^2/d.

These results suggest that it is unlikely that artificial destratification would inhibit algal growth in the Torrens Lake due to light limitation, although it may reduce P-fluxes from the sediments and prevent or reduce the severity of sustained cyanobacterial blooms. Destratification using a bubble plume aerator would only be necessary in the deep part of the lake as the shallow sections typically mix through the entire water column nocturnally.

In the Torrens Lake the manipulation of flow is an alternative means to disrupt stratification, limit nutrient resolubilisation and reduce the residence time of the lake.

1.7 Manipulation of flow

Management of cyanobacterial blooms could be achieved by manipulating the size and timing of river flow in such a way that thermal stratification is destroyed or the cyanobacterial population is flushed from the system. Natural flow through the Torrens River is minimal during summer and therefore flow would need to originate from Kangaroo Creek Dam.

Bormans and Webster (1997) developed a mixing criterion for turbid rivers which is applicable to the Torrens Lake. In summary, the degree of stratification in a water column is determined by the

relative supply rates of stratifying thermal energy and destratifying turbulent kinetic energy (TKE). The rate of change of potential energy (PE) required to mix the heat input and maintain a well-mixed water column of depth H, heated by a net surface heat flux Qnet is given by

$$D(PE)/dt = \alpha gH/2Cp(Qnet-2Q_I/KdH)$$

Where α is the thermal expansion coefficient, g is the gravity acceleration, Cp is the specific heat capacity of water, Q_I is the short wave radiation and Kd is the attenuation coefficient. (See Bormans and Webster (1997) for more details.) In this model, wind-mixing is not considered and flow over the bottom is assumed to be the only source of turbulent kinetic energy. The rate of generation of turbulent kinetic energy by working against the bottom is given by

$$d(TKE)/dt = c_D\rho_w U^3$$

where c_D is the bottom friction coefficient which applies to the depth-average velocity U, and ρ_w is the density of water. A fraction $\in$ of this TKE is available for increasing the PE of the water-column. The water column will destratify if the rate of increase of PE due to turbulent mixing exceeds the rate of decrease of PE due to surface heating. When the two competing processes are equal the transition between mixed and stratified conditions can be expressed as

$$R=U^3/(H(Q_{net}-2Q_I/k_dH)\ \alpha gH/2Cp) = 1/2 \in c_D$$

The criterion is not valid when $Q_{net}-2Q_I/k_dH$ is negative, which is when the water-column is cooling and the water-column will be well mixed regardless of discharge (Bormans and Webster, 1997). Well mixed conditions in weir pools, such as the Torrens Lake, correspond to R > 55,000, while stratified conditions correspond to R < 35,000. The transition between mixed and stratified conditions occurs at about R = 45,000.

1.8 Calculation of the flow required to destratify the Torrens Lake

For flow regulation to be an effective strategy to disrupt thermal stratification the flow must be sufficiently large to destratify on the days of greatest insolation and low wind speed. A meteorological station was deployed in the Torrens Lake in January 2000. The met station logged

solar insolation, wind speed and direction, air temperature, relative humidity and water temperature at 20 depths. The flow required to destratify the Torrens Lake on 11 January, 2000, was calculated at hourly intervals using data acquired by the met station.

The net surface heat flux (Qnet) into the water is given by

$$\mathbf{Q_{net} = Q_I - Q_B - Q_S - Q_E}$$

Where QI is the net downward short wave radiation, QB is the net upward long wave radiation, QS is the upward sensible heat flux and QE is the heat flux of evaporation (Henderson-Sellers, 1986: Bormans and Webster, 1997). QI was measured on 11 January at 13:00 was 1082 W/m^2 and QB was 218.7 W/m^2. QI and QB can be calculated if meteorological data is unavailable (see Bormans and Webster, 1997).

The upward sensible heat flux is given by

$$\mathbf{Q_S = C_p \rho_a C_H W(T_S - T_a)}$$

Where C_p is the specific heat capacity of air (1.01 x 10^3 J/kg.C), C_H is a coefficient for sensible heat exchange (1.5 x 10^{-3}; Fischer *et al*., 1979), ρ_a is the density of air (1.2 kg/m^3), Ts and Ta are the temperature of the water surface and air. W is the wind speed at 10m above the water surface, which was calculated from the wind speed at 2m height using the equation:

$$\mathbf{W10 = (W2 \times \ln (10/Zo))/\ln(Zm/Zo)}$$

Where W2 is the measured wind speed at 2m above the surface (m/s), Zm is the height of the anemometer (2m) and Zo = 0.000115 (CWR, 1998).

Heat flux due to evaporation is given by

$$\mathbf{QE = Lv \rho_a C_E W(q_s - q_a)}$$

Where Lv is the latent heat flux of evaporation (2.5 x 10^6 J/kg) CE is the coefficient for evaporative heat flux (1.5 x 10^{-3}; Fischer *et al*., 1979), q_s and q_a are the specific humidities estimated from temperature, relative humidity (0.15) using formula from Kimball *et al*. (1982).

Solving the equations for R = 45,000 yields maximum flow velocity of U = 0.22 m/s at 13:00 would be necessary for destratification. Flow required to destratify the lake (R>45000) varied with time of day, and ranged from 0 when the lake was naturally cooling to a maximum flow of 81.5 ML/hour, given an average width of 50m and depth of 2m. A daily flow of at least 297 ML/day would be required to maintain mixed conditions, however, the majority of this flow would need to be delivered between 10:00 and 15:00 daily. This equates to the entire lake volume being replaced within five hours every day.

Table 3. Input data used to calculate heat flux.

Time	Ta (ºC)	Ts (ºC)	RH	ea	es	qa	qs
7:00	24.6	22.14	0.33	1020.383	2665.704	0.006347	0.016581
8:00	25.4	22.39	0.32	1037.754	2706.55	0.006455	0.016835
9:00	17.6	23.79	0.28	563.3565	2945.548	0.003504	0.018321
10:00	29.9	26.83	0.23	970.0036	3528.78	0.006033	0.021949
11:00	31.3	29.7	0.21	959.3636	4169.232	0.005967	0.025933
12:00	32.8	32.2	0.18	895.0254	4807.203	0.005567	0.029901
13:00	33.8	34.72	0.15	788.7884	5534.435	0.004906	0.034424
14:00	34.6	36.58	0.12	659.7309	6130.676	0.004104	0.038133
15:00	34.6	37.71	0.12	659.7309	6519.441	0.004104	0.040551
16:00	35.2	37.3	0.11	625.1599	6375.987	0.003888	0.039659
17:00	34.3	34.69	0.11	594.7685	5525.246	0.003699	0.034367
18:00	34.6	34.93	0.10	549.7758	5599.13	0.00342	0.034827
19:00	33.3	33.55	0.14	715.9234	5185.725	0.004453	0.032255
20:00	31.5	31.98	0.17	785.4944	4747.849	0.004886	0.029532

Table 4. Input variables used to calculate the flow necessary to destratify the Torrens lake.

Time	Ws	W10	CN1R (W m^{-2})	EP16 E (W m^{-2})	Qb	Qe	Qs	Qnet	U (ms^{-1})	flow (m^3/h)
7:00	1.37	1.596	-43.5	18	61.5	73.492	-7.151	-109.841	NA	
8:00	2.3	2.679	-29.7	202	231.7	125.141	-14.690	-140.151	NA	
9:00	1.33	1.549	324.6	516	191.4	103.299	17.469	203.832	NA	
10:00	1.74	2.027	492.2	709	216.8	145.161	-11.334	358.374	0.149	53722.94
11:00	1.3	1.514	609.3	857	247.7	136.050	-4.413	477.663	0.182	65394.58
12:00	1.31	1.526	705.9	966	260.1	167.093	-1.668	540.475	0.190	68362.98
13:00	1.08	1.258	863.3	1082	218.7	167.104	2.108	694.087	0.226	81523.59
14:00	1.07	1.246	624.3	1047	422.7	190.860	4.495	428.945	0.077	27605.14
15:00	1.01	1.176	-116.7	471	587.7	192.960	6.665	-316.324	NA	
16:00	0.86	1.002	339	654	315	161.249	3.832	173.919	NA	
17:00	0.88	1.025	118.1	325	206.9	141.462	0.728	-24.090	NA	
18:00	1.07	1.246	-60.3	499	559.3	176.152	0.749	-237.201	NA	
19:00	1.12	1.305	153.5	146	-7.5	163.221	0.594	-10.315	NA	
20:00	0.97	1.130	-80.1	21	101.1	125.312	0.988	-206.400	NA	

1.9 Aerator and flushing trial, January – February, 2000

Both artificial destratification and controlled flushing were trialled in Summer 2000 to determine what impact these had on the physical environment, nutrients, dissolved oxygen and cell counts.

Four significant 'events' affected the physical environment in the Torrens Lake during the monitoring period, January-February 2000. These consisted of:

1. A controlled release of water from Kangaroo Creek Dam of 365 ML over 66 hours, starting on 31 January, 2000. Gauging at a station just downstream of the Torrens Weir revealed that the release took approximately 30 hours to reach the Lake and the outflow volume was 210 ML. 155 ML were lost filling up weir pools upstream of the Torrens Lake which were overflow weirs at low summer levels.
2. An aerator, powered by a portable compressor, was installed at the western end of the Lake on 12 February, 2000. The aerator delivered air at a rate of 56 L s^{-1} through a 100 m hose.
3. A second water release of 125 ML occurred on 16 February and approximately 90 ML reached the lake on 17 January.
4. A significant rainfall event increased flow through the Torrens Lake on 19 February 2000. Twenty-four hour accumulated rainfall to 0900 was recorded at Kent Town, 2 km from the Lake; 27 mm recorded on 20 February, 11 mm on 21 February, 3 mm on 22 February and 4 mm on 23 February.

1.10 Monitoring Program and summary of outcomes

Monitoring of physical variables was undertaken at a meteorological station permanently deployed at the western (weir) end of the Torrens Lake. Meteorological variables were measured at one minute intervals and the 15 minute mean recorded. Measurements consisted of wind speed and direction (Climatronics WM-111) measured 2m above the water surface, downwelling shortwave radiation (300-3000 nm, Middleton EP08) and upwelling and downwelling longwave radiation (0.3-60 µm, Middleton CN1-R). Thermistors (Betatherm, accuracy ± 0.01°C, resolution 0.01°C) were deployed at 20 depths over the six metre water column and logged water temperature at two minute intervals. Dissolved oxygen was measured continuously with Greenspan DO300 sensors at 0.5m and 5m depth.

Samples for phytoplankton identification and enumeration were collected with a hose pipe integrating the top 2m of the water column. *M. aeruginosa* abundance was determined on samples which were preserved in Lugol's iodine, pressurised to 1000 kPa to collapse gas vesicles and counted in a Sedgewick Rafter chamber. Samples for nutrient analysis were collected at discrete depths and chemical analysis performed by the Australian Water Quality Centre.

1.10.1 Thermal stratification

Thermal stratification was a common feature of the Torrens Lake during the study period, which was a function of high solar radiation, warm air temperatures and low wind speeds. Persistent thermal stratification was generally associated with high nocturnal air temperature. The depth of the surface mixed layer was defined as the shallowest thermistor depth at which the temperature difference between the surface and the thermistor was 0.05 °C or greater. Cool air temperatures from January 24-28 allowed cooling and mixing of the entire water column. As air temperatures rose in late January surface water temperatures increased, and the surface mixed layer was persistently shallow. The first release of water, which arrived at the lake on 1 February, had no impact on the depth of the surface mixed layer. This was because the release was not a plug flow but intruded into the lake at a depth with equivalent temperature and density to the inflowing water, which at 13:00, February 1 was 1-1.5 m depth.

The aerator deepened the surface mixed layer for longer periods close to the aerator deployment but had little impact at sites further from the aerator. The arrival of large flows from rainfall in the catchment mixed the lake late in February.

Dissolved oxygen (DO) was measured continuously and the low concentrations highlight the severely degraded state of the lake. The dissolved oxygen concentrations at both at 0.5m and 5m depth decreased in mid January until, by 26 January, concentrations were less than 2 mg/L. Although there were mixed conditions from January 24-30 this could not satisfy the high oxygen demand. The DO increased in the epilimnion in late January due to increased algal photosynthesis and separation of the epilimnion from the sediment, which had a high oxygen demand. Following the first release of water the DO plummeted and the hypolimnion remained anoxic until the aerator was installed. The bubble plume aerator increased the DO concentration at 5m depth, but was unable to oxygenate the water column sufficiently to cope with the large organic load, from the catchment, during the rain events in mid February.

1.10.2 *Microcystis aeruginosa abundance*

Cool conditions from mid to late January did not favour *M. aeruginosa* growth, however, as the surface mixed layer depth became shallower *M. aeruginosa* numbers increased but remained below 10,000 cells/mL until 14 January. A maximum concentration of 48,000 cells/mL was recorded on 18 February. The accelerated growth that occurred after the aerator was turned on may be attributable to phosphorus-rich hypolimnetic waters mixing with surface water and relieving phosphorus limitation. *M. aeruginosa* abundance decreased following the second water release and the rain event, however, lake-wide average *M. aeruginosa* counts suggest that the population was in decline prior to the flushing (data not shown).

1.10.3 *Nutrients*

The filterable reactive phosphorus (FRP) concentrations in the lake were highly dependent upon the degree of stratification and rainfall. Between January 7 and 11 there was strong stratification and the FRP at 5m increased by 0.06 mg/L, which, if integrated over a 2m hypolimnion, equates to an internal nutrient load of 30 mg P/m^2/day. Similarly, between February 4 and 11 thermal stratification persisted and the internal load to the three metre hypolimnion was approximately 17 mg P/m^2/day. The FRP decreased by 17 February and could be attributable to uptake by the expanding *M. aeruginosa* population or complexation with iron. The rain event on 19 February imported a large, catchment derived nutrient load to the lake.

Ammonia concentrations in the hypolimnion on 14 January were greater than 1 mg/L which represents an enormous bioavailable nitrogen source. Ammonia concentrations remained high until 12 February, when the aerator was switched on and demand for nitrogen from the cyanobacterial bloom was high. Ammonia, nitrate and total phosphorus all increased with the rain event on 19 February.

1.11 Discussion

There have been several initiatives to improve water quality in the Torrens Lake, however, it has been identified that shallow urban lakes are extremely difficult to manage. The dredging that took place in 1998 was largely ineffective at removing the sediment phosphorus and a significant internal load was observed in January 2000 during periods of persistent thermal stratification.

1.11.1 Artificial destratification

The model of *M. aeruginosa* growth in different euphotic depth/mixed depth waterbodies revealed that destratification was unlikely to control growth of cyanobacteria due to light limitation in most of the Torrens Lake. However, the aerator deepened the surface mixed layer and increased the dissolved oxygen concentration when the aerator was first switched on. Unfortunately the aerator could not cope with the enormous oxygen demand associated with the catchment-derived organic load, and destratification did not decrease cyanobacterial numbers in the short term. Because the aerator could not cope with BOD associated with rainfall events, and there is an average of four rain events each January (Australian Meteorological Bureau), it was concluded that artificial destratification was of limited value in the Torrens Lake for cyanobacterial control but would probably help with managing the deoxygenation events.

1.11.2 Increased flow

The model of destratifying flows (Bormans and Webster, 1997) correctly predicted that the volume of water released (210 ML) would not destratify the river. Using the mixing criterion for the Torrens Lake, the minimum flow required for destratification was calculated to be at least 297 ML day^{-1}. In large river systems the maximum velocity would be required continuously in order to ensure the entire reach, which may have a travel time of weeks, was destratified. The dilution of the lake population with a single large flushing flow to push algae downstream may be inefficient if the river inflow water is cooler than the lake water and slides under the warm surface layer containing the cyanobacteria.

To maintain a flow large enough to permanently destratify the Torrens Lake would be expensive and not sustainable. This option may be suitable in systems where it is just a matter of scaling up base flow to ensure total or periodic destratification, such as in the Murrumbidgee and Murray Rivers in Australia (Bormans *et al*., 1997; Maier *et al*., 2000). Pulsed flow to disrupt stratification and control cyanobacteria in weir pools on highly regulated rivers has proven to be successful (Sherman *et al*., 1998; Webster *et al*., 2000). Reducing the residence time of water in the Torrens Lake by releasing a pulsed flow appears to be a useful, albeit expensive, management option that does not guarantee complete bloom dispersal.

Several potential problems with using a release from an upstream storage to flush the lake require consideration. For example, the quality of the water used to increase flow needs to be reasonable.

At Kangaroo Creek Dam, upstream of the Torrens Lake, release is from the bottom of the reservoir and consequently the water may be enriched in phosphorus due to sediment phosphorus resolubilisation during stratification. Pulsing with phosphorus-rich water would have a similar effect on the phytoplankton as the rain event, allowing the luxury uptake of phosphorus by the cyanobacteria and extending the duration of the bloom. The use of a bottom release weir at the lake will further exacerbate the problem and the flow may have little effect at disrupting stratification or flushing the population downstream. This may be overcome, to some extent, by closing the bottom release gate and using siphons to create an overflow weir. Siphons could be used several days before a pulsed flow to reduce cyanobacterial abundance in the lake and the flow used to refill the lake, rather than flushing.

Of the cyanobacterial control strategies trialled to this point, none have avoided the formation of a bloom. In 1999 when no management intervention was employed the bloom lasted for a few weeks but starved itself to death when phosphorus became limiting (Ganf *et al*., 2000). Although the blooms were unsightly and forced the closure of the lake the impact was short-lived and appeared more successful than manipulation of the physical environment by either pulsed flow or artificial destratification.

It is apparent that any 'in lake' strategy to control cyanobacteria or water quality will be compromised if catchment work does not address water quality issues such as nutrient and organic runoff. Another issue of concern is the internal nutrient load, which appears greatest during the periods of persistent thermal stratification. There are methods to control the internal nutrient load that show promise however, these require trialling before they can be recommended for shallow lakes.

1.12 Reducing the internal nutrient load

In a typical phosphorus cycle, phosphorus is remobilised from sediment or decaying organic matter and entrained into the water column, where it is taken up by algae. From there the phosphorus is either passed on to higher levels of the food web or lost to the bottom as the algae sediment. In deep lakes the resolubilisation of phosphorus at the sediment is vertically separated from the algae and so each phosphorus molecule can only be accessed with entrainment from the hypolimnion to the epilimnion. In strongly stratified deep lakes this may happen only once or

twice a year during significant 'over-turn' events. In contrast, in shallow lakes the zone of phosphorus resolubilisation is much closer to the zone of greatest productivity and a single molecule may be recycled a number of times during the growing season (Reynolds, 1997 and thereby sustain a high algal biomass for longer. This is why shallow lakes prove so much more difficult to restore than deep ones (Sas, 1989: Reynolds, 1997).

A lake remediation strategy that has been used extensively to decrease nutrient concentrations in waste treatment ponds and some natural lakes is sediment capping with phosphorus binding agents. However, some of the chemicals used to flocculate phosphorus in waste treatment facilities, such as alum, iron salts and lime, are not suitable for application in natural systems. Recently there have been advances in the use of modified clays to bind phosphorus and reduce the filterable reactive phosphorus in water-bodies. However, the most common commercial activated clay contains Lanthanum, which is not suitable for application in natural waters.

Another means of decreasing the internal nutrient load is by hypolimnetic oxygen injection. However, given that anoxic conditions persisted 0.5m below the surface suggests that the oxygen demand is too great to be overcome by oxygen injection or any destratification device.

There is evidence emerging that desiccation/oxidation of sediments can significantly reduce the release of phosphorus from lake sediments upon re-wetting (Baldwin, 1996; Baldwin and Mitchell, 2000). This has been attributed to a number of interrelated factors including a shift in the bacterial community structure (loss of viable sulfate-reducing bacteria), increased carbon limitation in the dried sediments and aging of minerals with which phosphorus can be associated (Mitchell and Baldwin, 1998). Whilst it is not known how long the effects of desiccation will last, drying the lake sediments may be a suitable strategy to reduce phosphorus release in some shallow lakes. Many natural shallow lakes may have undergone seasonal wetting and drying before modification of the hydrology to maintain unnaturally high water levels.

The first recorded cyanobacterial bloom in the Torrens Lake occurred in January 1998, which coincided with a large-scale dredging project in the lake. There has been considerable debate regarding the possible impact of the dredging on initiating a cyanobacterial bloom. The phosphorus binding capacity of sediments depends upon the sediment type and the number of sites available to absorb phosphorus. If the sediment has received phosphorus-rich waters for a

considerable period then the binding sites may be saturated and removal of this sediment may be an effective strategy to reduce phosphorus concentrations in the receiving water.

Fortunately there are extensive catchment works in the Torrens catchment to intercept gross pollutants, and an increasing awareness among the public about their impact on the Torrens waterway. However, there is still a need to identify the sources of the high nutrient and organic load so that management can be implemented to improve this. This paper has identified the difficulties associated with cyanobacterial control in an urban lake and highlights the limitations of 'in-lake' strategies when catchment-derived factors interfere.

The strategy that was not trialled, and has been proposed since the trials a decade ago, is dilution by amenity flows. The major aim of this work supported by the Goyder Institute is to determine whether amenity flows and dilution of the cyanobacterial populations in the lake is a feasible management strategy.

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Chapter Two: The Feasibility of dilution as a solution

Recurring cyanobacterial blooms in the Torrens Lake, Adelaide, compromise the amenity of the lake. The proposed development of Adelaide Oval and the riverside precinct has renewed efforts to find a sustainable solution to algal issues. A range of options have been trialled with varying levels of success. These include artificial destratification with bubble plume aerators, venturi pumps and fountains, the application of a Lanthanum based activated clay, flushing flows and biofiltration.

The Adelaide and Mt Lofty Ranges Natural Resource Management (NRM) Board and the City of Adelaide are seeking a long-term sustainable option for cyanobacterial management in the Torrens Lake. One option being considered is the delivery of diluting flows to flush algae from the system and limit the accumulation of high biomass.

The Adelaide and Mt Lofty Ranges NRM Board proposed a trial to examine the impact of dilution flows on cyanobacterial growth in the Torrens Lake. Before this could be evaluated in the field a desktop feasibility assessment was required.

The feasibility assessment included:

- A risk matrix of cyanobacterial growth at different growth rates and dilution flows
- Analysis of historical data to determine the growth rate of *Microcystis* in the Torrens Lake to apply in a risk assessment of growth vs dilution
- A review of the modelling of the lake hydrodynamics and utilisation of this to determine the feasibility of dilution flows adequately mixing and diluting cyanobacteria from the lake
- Development of a flow strategy that would document the required dilution flows at different cyanobacterial biomass and risk.
- Assessment of the likelihood of the various flow strategies actually achieving the reduction in biomass based on the stratification, the flow rates and density differences between inflow and the lake water.

- Determination of the appropriateness of various source water to provide diluting flows based on nutrient concentrations

2.1 A risk matrix of cyanobacterial growth at different growth rates and dilution flows

2.1.1 Approach and methodology

A risk assessment framework is used to determine whether dilution can effectively control cyanobacterial blooms. This considers the range of likely growth rates of *Microcystis* and *Anabaena* and how the provision of diluting flows would affect the magnitude of the cyanobacterial population.

A simple model was developed that considers the rate of growth of cyanobacteria and dilutes the population at various rates representing various flows. The major assumption with this model is that the entire population in the lake is completely mixed and the inflow of diluting water mixes evenly across the entire lake thereby diluting evenly. In reality this is probably not the case, but this will be explored later in the document. Regardless of this caveat, the model provides insight into the feasibility of using diluting flows to control cyanobacterial biomass in the Torrens Lake.

In the model the growth rate of phytoplankton (algae or cyanobacteria) is described by the equation

$$N_t = N_0^{kt}$$

Where N is the population size at time t or time 0 (initial population size) and K is the growth rate per day. Population growth over 20 days is simulated. The base case has no dilution over the 20 day period. Eight population dilutions are simulated which correspond to different diluting flows (Table 5). The population is diluted at daily time-steps by the set rate. In this case we assume a total lake volume of 400 ML.

Table 5. Dilution rates modeled and the corresponding lake volume required to achieve this dilution assuming a total lake volume of 400 ML.

Dilution rate (% of lake volume replaced)	Dilution volume for 400 ML Lake
5	20
10	40
15	60
20	80
25	100
40	160
50	200
75	300

The range of modelled growth rates were determined from previous light-limited growth rate experiments with monospecific laboratory cultures of *Anabaena circinalis* and *Microcystis aeruginosa*, modified from the thesis of Brookes (1997). *Anabaena circinalis* had a maximum growth rate of approximately 0.6/day over the six day experiment (Figure 1). *Microcystis aeruginosa* grew at a slightly lower rate with a maximum rate of 0.45/day (Figure 2). Exponential growth rates for *Anabaena* species may be as high as 0.8/day (Reynolds, 1997) and similarly high rates have been observed for *Microcystis* (Brookes, 1997). The modelled growth rates were 0.1, 0.2, 0.3, 0.4, 0.5, 0.6, 0.7, 0.8/day.

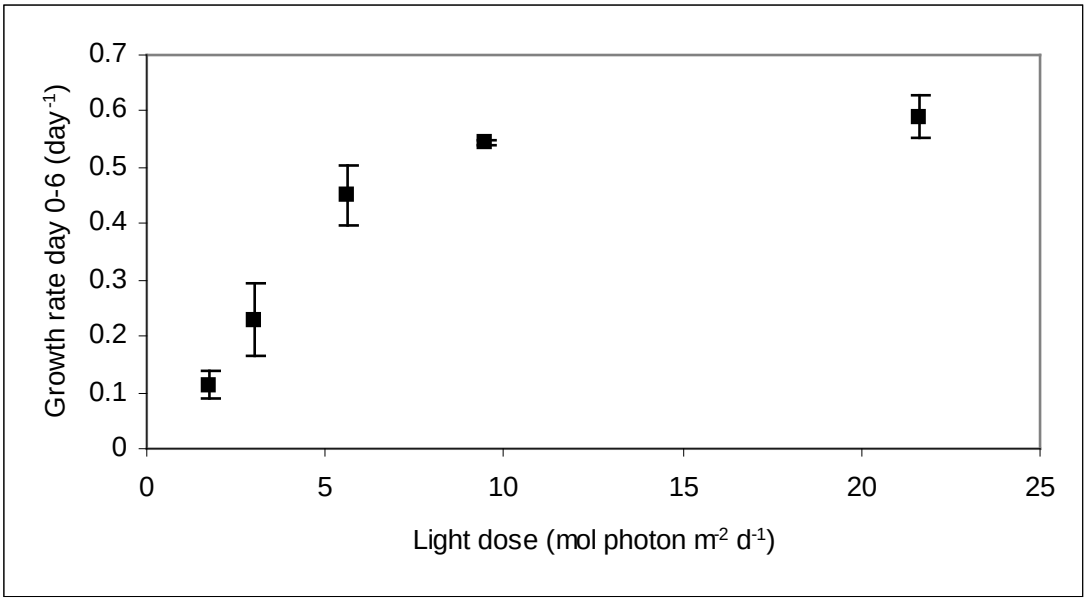


Figure 1. Light dependent growth of *Anabaena circinalis*.

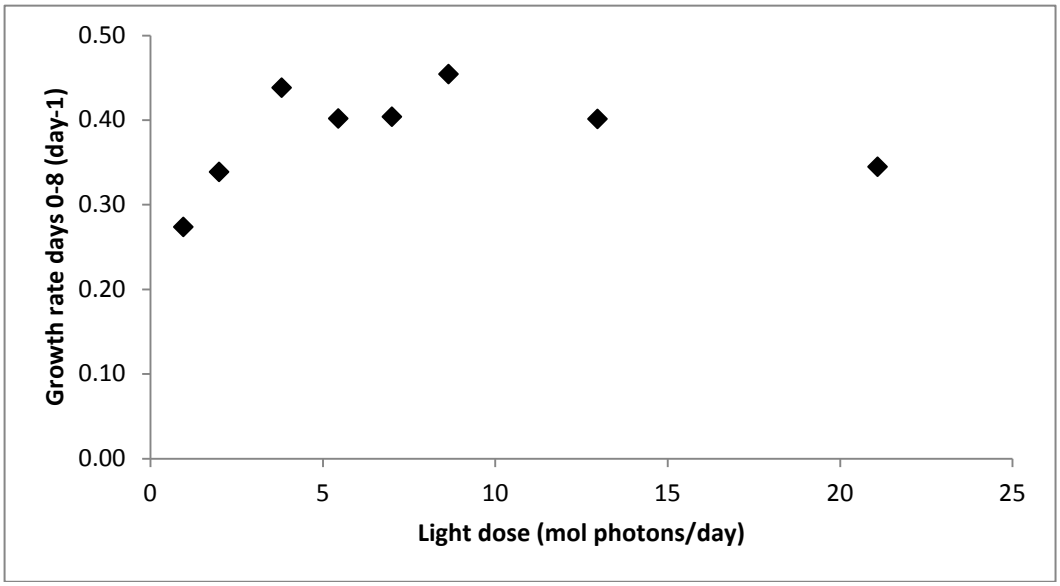


Figure 2. Light dependent growth of *Microcystis aeruginosa.*

2.1.2 Population size at different growth and dilution rates

The population size of phytoplankton growing at different rates was predicted with continual washout or dilution. This enables a risk assessment of cyanobacterial blooms under different condition of growth and river inflow/outflow. High flows were also considered to provide context for what dilution flows are required when growth rates are high.

The initial inoculum was set as 100 cells/mL. The critical level for primary contact in recreational water is 50,000 cells/mL. Recent internal correspondence at the City of Adelaide highlight important details in shifting the trigger levels from cell counts to biovolume (email 5 January 2010). The critical cyanobacteria biovolume for recreational waters is 10 mm^3/L. Using cell count - biovolume data from the Australian Water Quality Centre the critical cell count data for this threshold were calculated for the dominant cyanobacteria species in the Torrens Lake (Table 6).

Table 6. Cell count concentrations that reach the recreational guideline for cyanobacteria in the Torrens Lake.

Cyanobacteria	Cell count (cells/mL)	Biovolume equivalent (mm^3/L)
Anabaena circinalis	40,000	10
Microcystis aeruginosa	115,000	10.005
Microcystis flos-aquae	455,000	10.01
Planktothrix mougeotii	156,500	10.016

In this risk assessment the level of 100,000 cell/mL was considered an inability of that flow to control the cyanobacterial biomass in the Torrens Lake. This concentration is less than the biovolume critical level for most species but would be high enough to cause unsightly scums.

With no diluting flows a growth rate of greater than 0.4/day will exceed the 100,000 cells/mL threshold after 20 days (Figure 3). The situation does not improve much with a dilution flow of 5%; it takes 18 days to reach 100,000 cells/mL with no dilution and 19 days with 5% diluting flow (Figure 4). With a dilution flow of 10% this threshold is not exceeded in the 20 day simulation with a start concentration of 100 cells/mL and growth rate of 0.4/day (Figure 5).

As the diluting volumes increase, the tolerable growth rate increases before the threshold is exceeded (Figures 4 – 11). A diluting flow of 15% presents a reasonable risk reduction from the base case as it takes 17 days to exceed 100,000 cells/mL at a growth rate of 0.5/day (Figure 6).

The risk assessment is expanded in the next section with inclusion of historical data from the Torrens Lake and estimated growth rates.

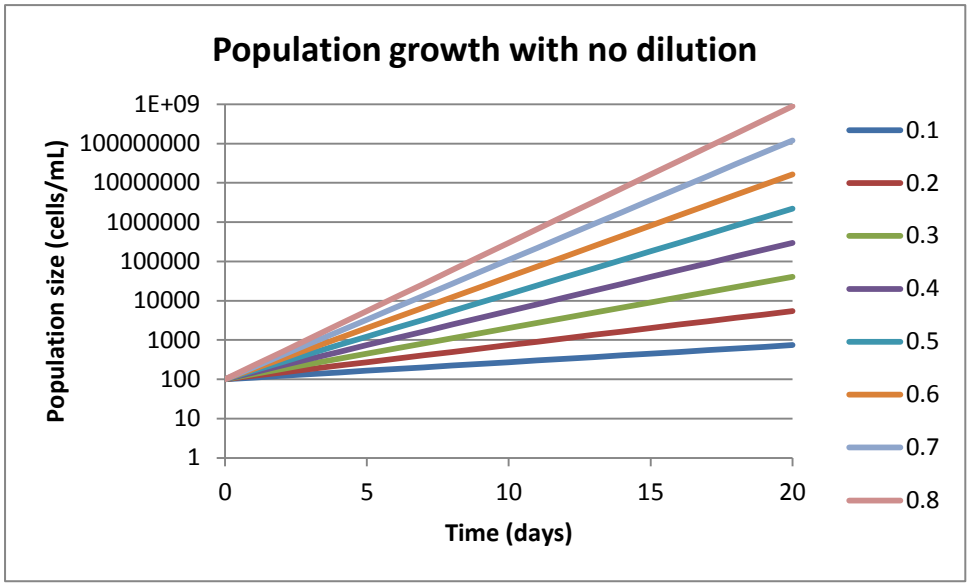


Figure 3. Population size of algae growing at eight different growth rates.

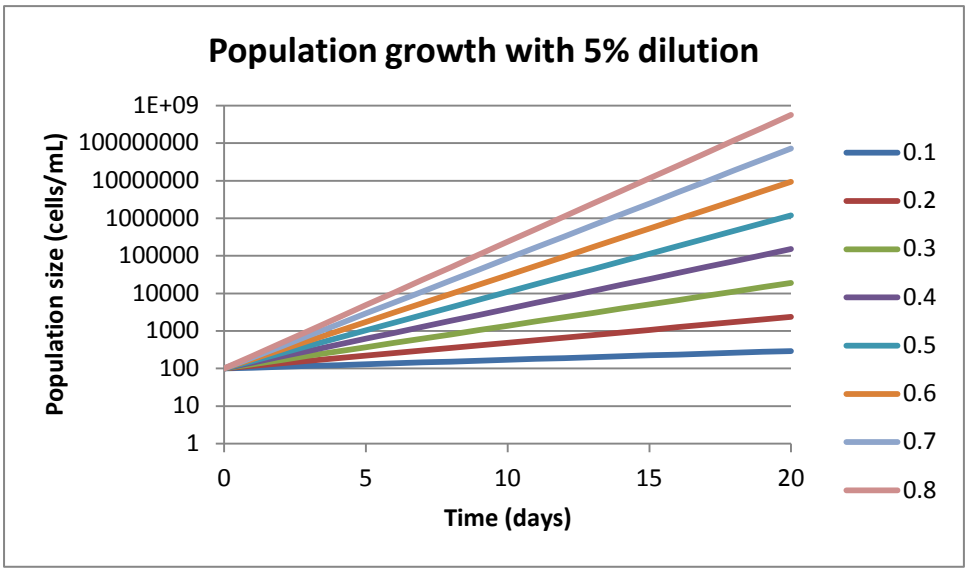


Figure 4. Population size of algae growing at eight different growth rates but with a daily dilution rate of 5%.

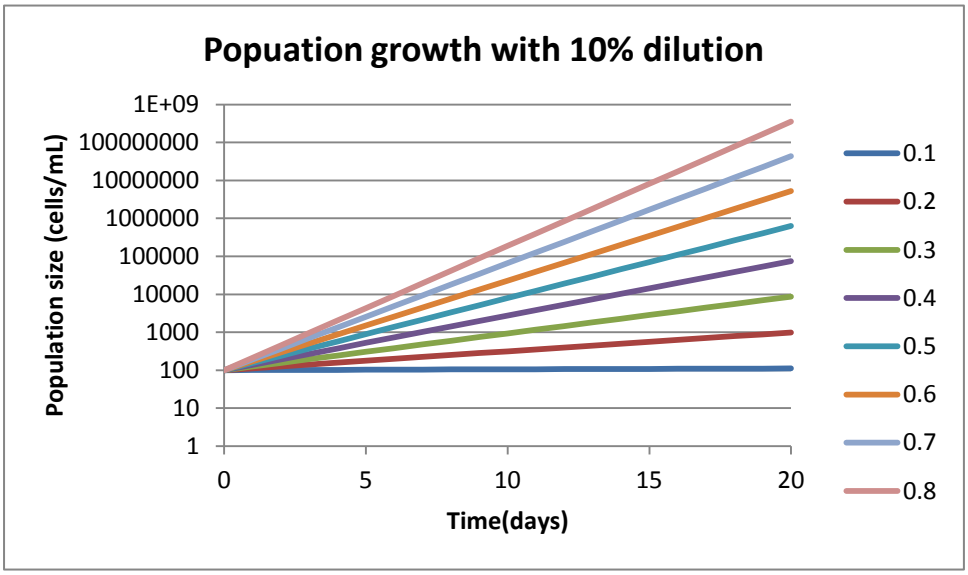


Figure 5. Population size of algae growing at eight different growth rates but with a daily dilution rate of 10%.

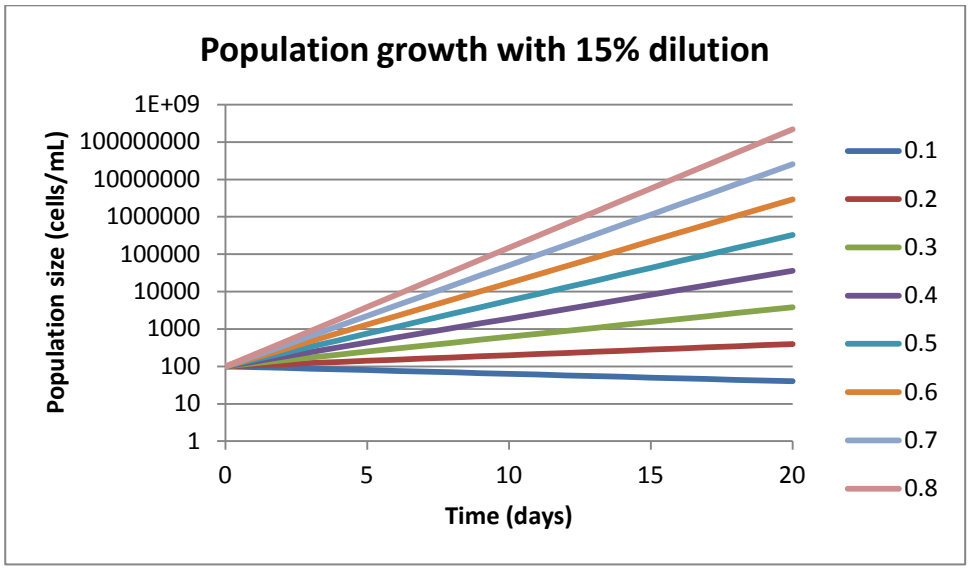


Figure 6. Population size of algae growing at eight different growth rates but with a daily dilution rate of 15%.

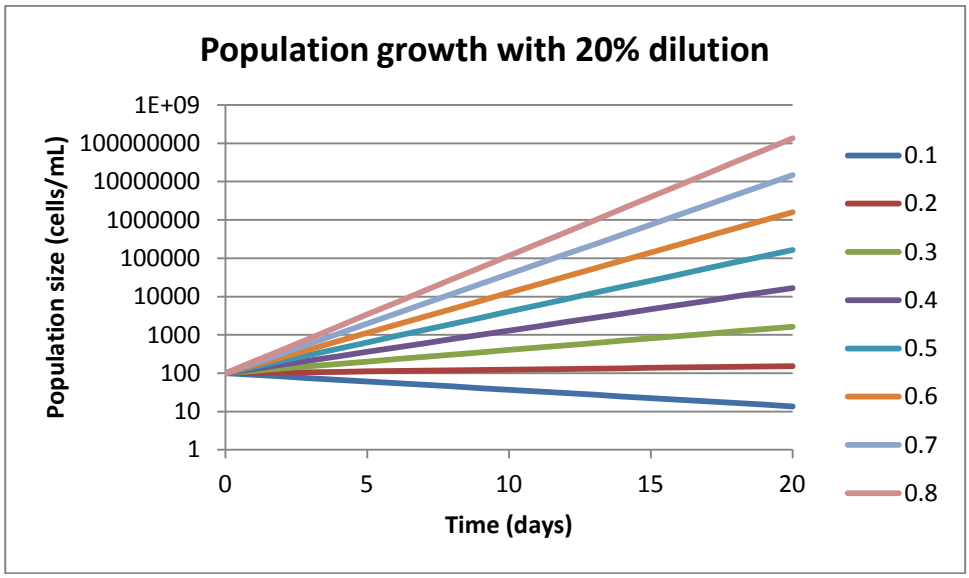


Figure 7. Population size of algae growing at eight different growth rates but with a daily dilution rate of 20%.

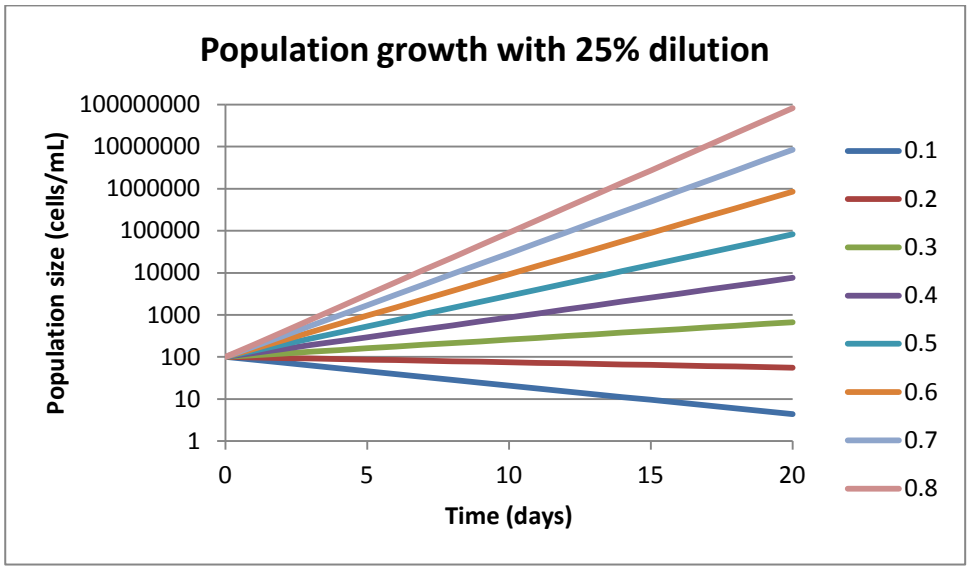


Figure 8. Population size of algae growing at eight different growth rates but with a daily dilution rate of 25%.

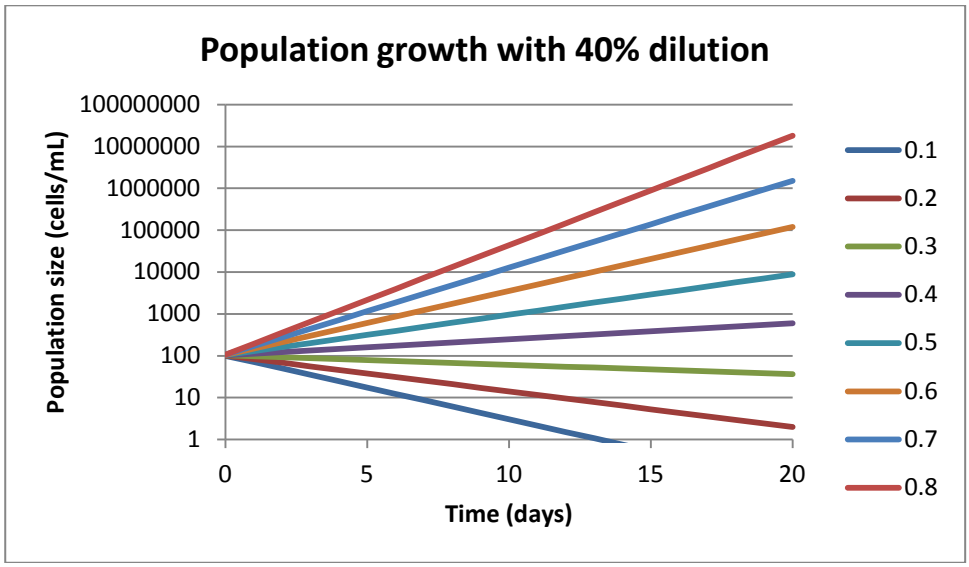


Figure 9. Population size of algae growing at eight different growth rates but with a daily dilution rate of 40%.

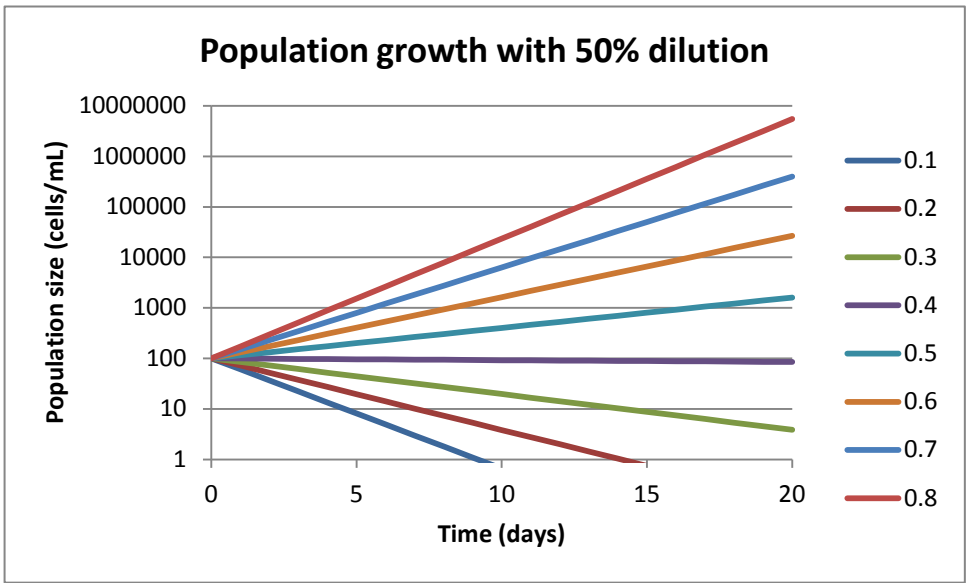


Figure 10. Population size of algae growing at eight different growth rates but with a daily dilution rate of 50%

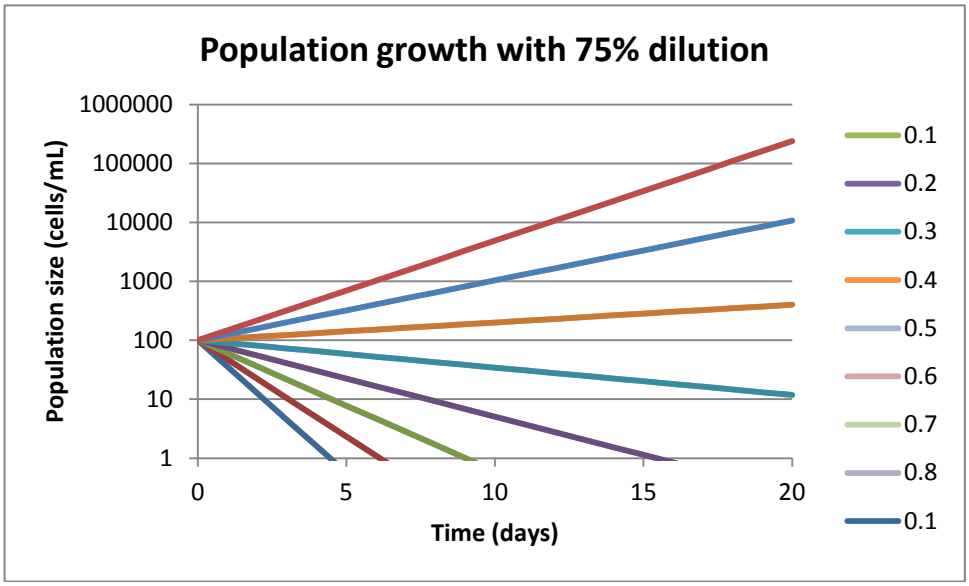


Figure 11 Population size of algae growing at eight different growth rates but with a daily dilution rate of 75%.

2.2 Analysis of historical data to determine the growth rate of cyanobacteria in the Torrens Lake to apply in a risk assessment of growth vs dilution

The abundance of the four dominant cyanobacteria was plotted by site for the years 2007 to 2011 (Figures 12-15). Typically the problem months are mid-late January and February, although positive growth rates were observed as late as April. Some blooms appear to last for a long time but this is usually a function of the graph recognizing a continuous sequence of data, which are generally not continuous, or the time-steps evenly spaced. This data was used to determine the number of sampling occasions that the dominant species exceed the critical threshold. Two thresholds were used; the 10 mm^3/L biovolume and the cell concentration of 100,000 cells/mL. Note that these thresholds are not equivalent. *Anabaena circinalis* was the species that most frequently exceeded the critical threshold when biovolume was used as the parameter for determining the critical threshold (Table 7). When cell number was used as the critical threshold it was *Microcystis flos-aquae* that most frequently triggered a response (Table 8) in the period 2007-2011.

Some of the threshold exceedences are on consecutive sampling occasions. This is important as it highlights that although cyanobacteria are present every summer they do not always grow to concentrations that exceed the threshold. Dilution and washout will be species non-specific but it is clear that *Anabaena circinalis* and *Microcystis flos-aquae* are the major problem species. Diluting flows may not be required every year but it is probably prudent to commence in early January for a duration of about 2 months.

The growth rate of each of the dominant cyanobacteria was calculated to allow comparison with the model of growth and dilution described in the above section. Only positive growth rates are presented as these represent an escalating risk (Figure 16, Figure 17, Figure 18). *Planktothrix mougeotii* growth is not presented as this poses a low risk and offers little more to the risk assessment. The Bonython Park site is not included in the figures either as this site is downstream of the weir and subject to different recreational use.

Anabaena circinalis growth rates in the Torrens Lake are typically up to 0.6/day, although there are occasions when growth is greater than this (Figure 16). The very high growth rates observed

in early 2011 are likely to be a combination of both high growth but also concentration from floating to the surface or wind-blown surface accumulations. *Microcystis aeruginosa* had growth rates typically up to 0.3/day, although it was higher on occasions. The majority of growth rates for *Microcystis aeruginosa* were below 0.4/day, although this species too displayed high growth on occasions.

The most appropriate growth rate to model in the risk assessment depends on the species. *Anabaena circinalis* has the highest growth rate, and exceeds the critical biovolume threshold most often. It is also appropriate to consider *Microcystis flos-aquae* as this species also tends to have high biomass.

For a growth rate of 0.4/day, which is a typical exponential growth rate of cyanobacteria in the Torrens Lake, it can be concluded that a diluting flow of at least 10% per day would be required to have noticeable impact on the cyanobacteria population. With a starting cell concentration of 100 cells/mL, a growth rate of 0.4/day and a diluting flow of 10% the cell concentration after 20 days would be 74,420 cells/mL, which is below the critical threshold for cell numbers.

Further to this, the most suitable flow strategy to trial would be a relatively high flow during the months of January and February.

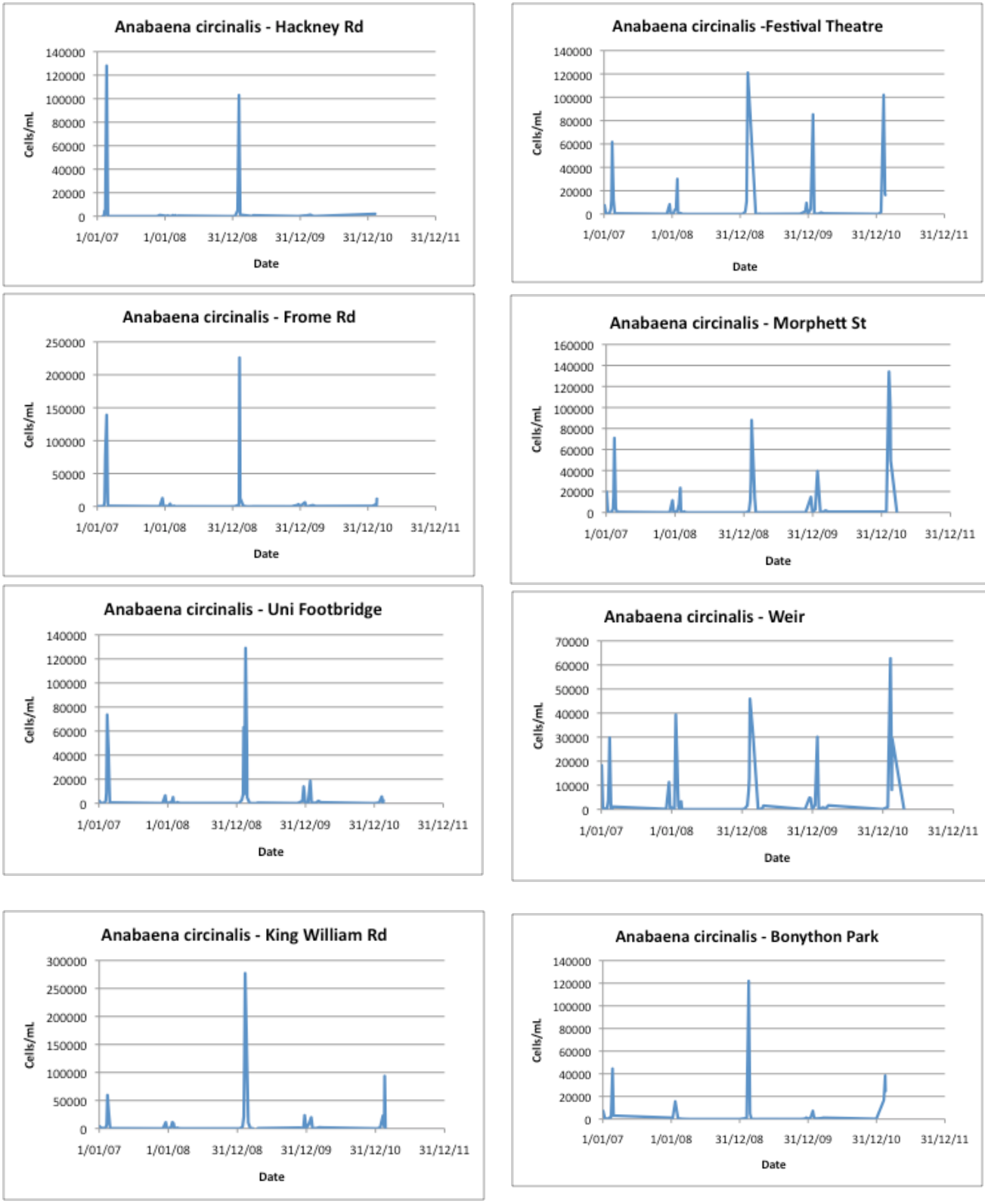


Figure 12. Anabaena circinalis cell numbers at sites in the Torrens Lake.

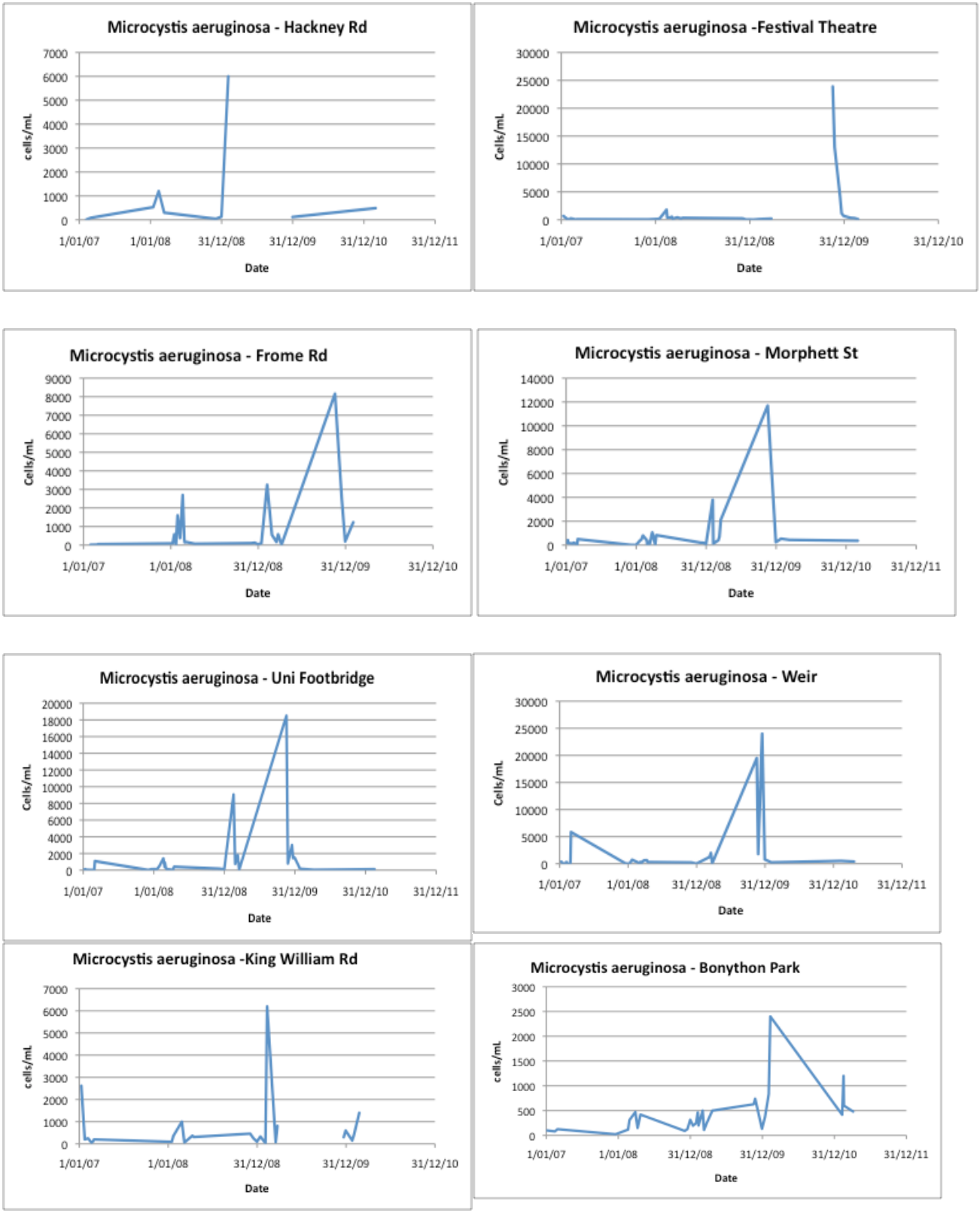
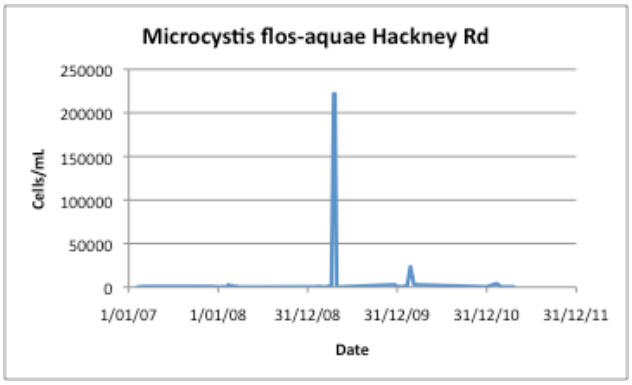
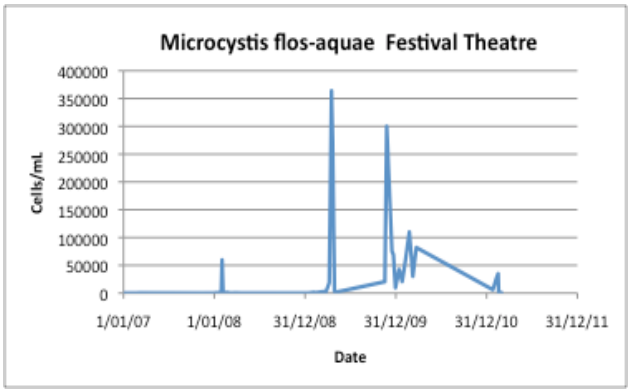
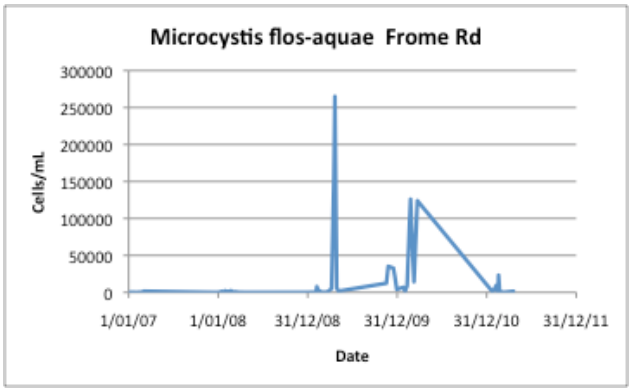
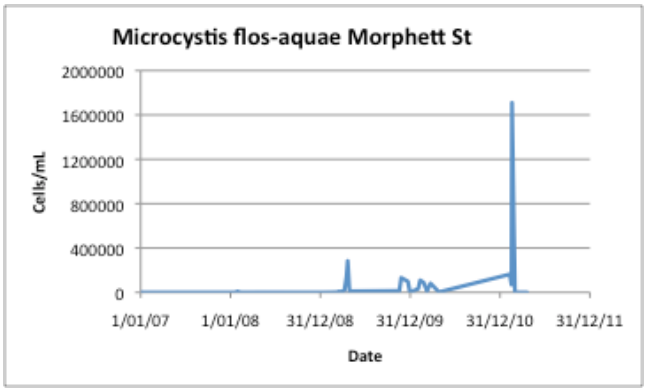


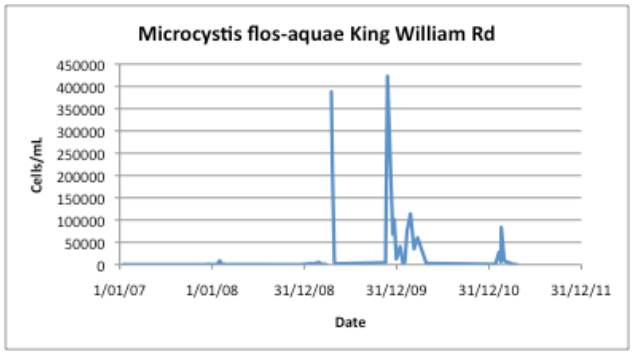
Figure 13. ***Microcystis aeruginosa*** **cell numbers at sites in the Torrens Lake.**

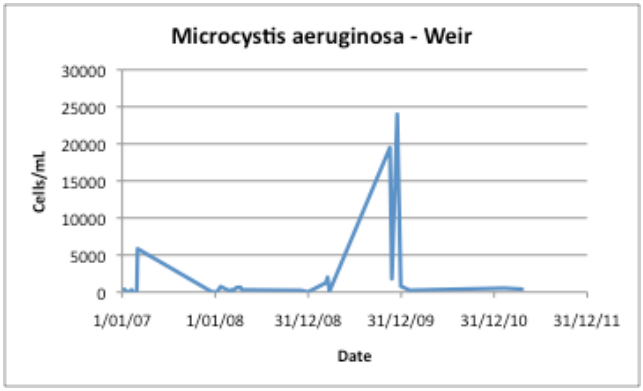


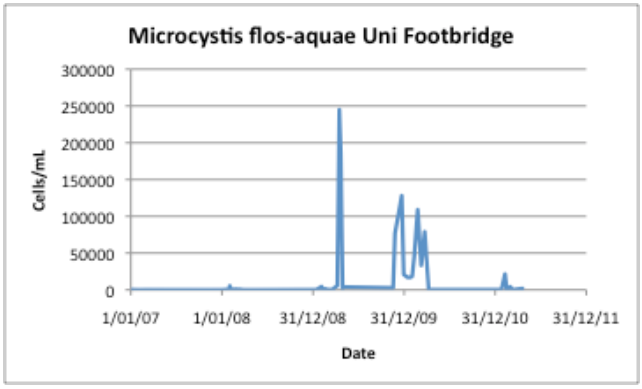












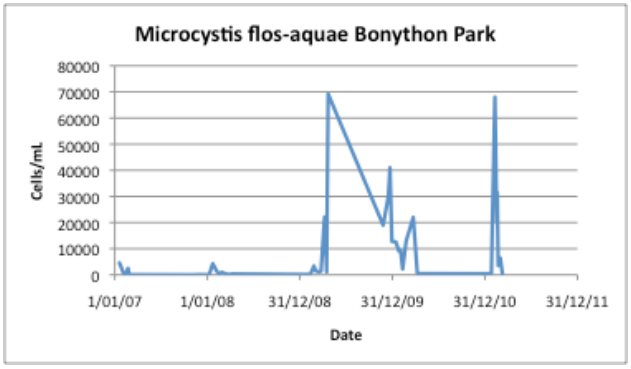
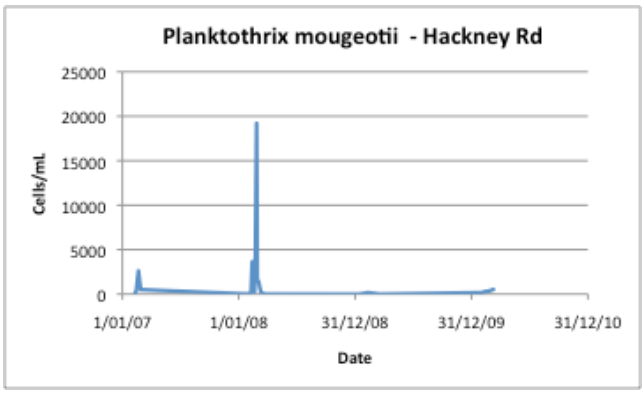
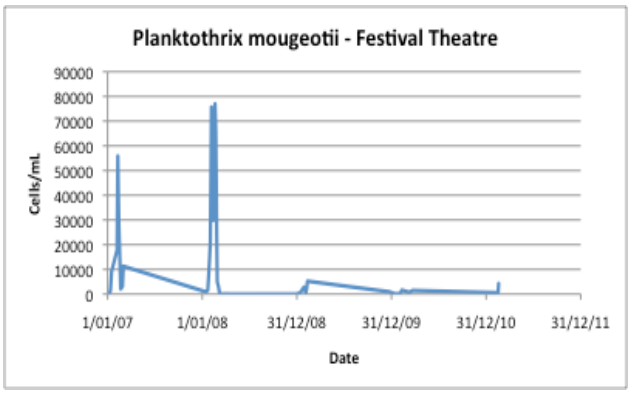
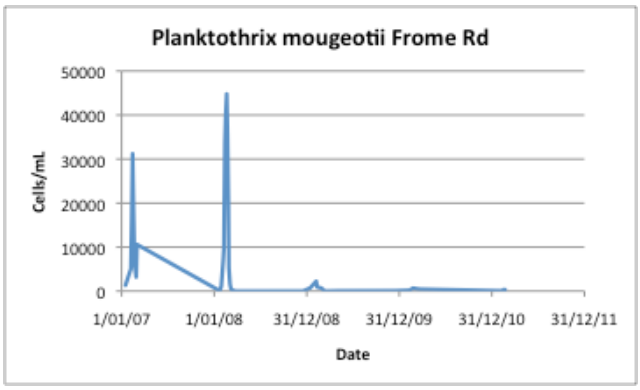
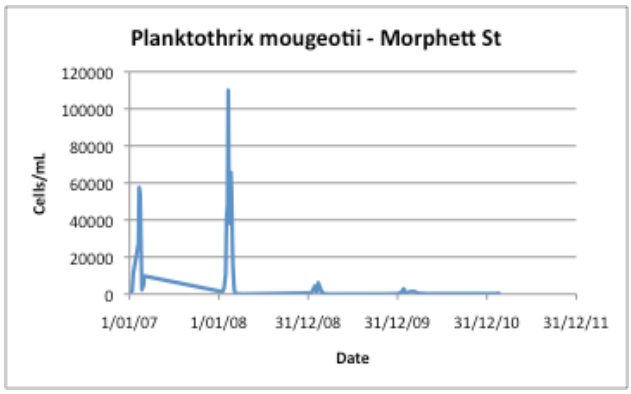


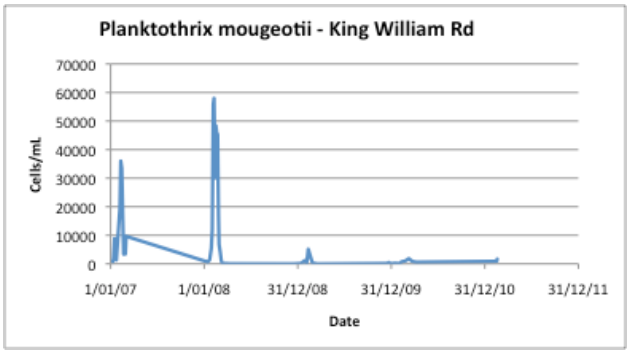
Figure 14. *Microcystis flos-aquae* cell numbers in the Torrens Lake.

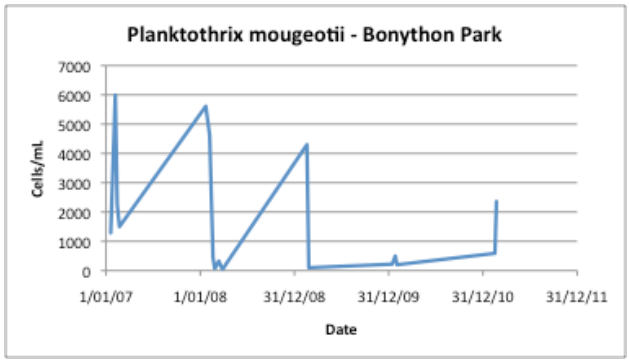


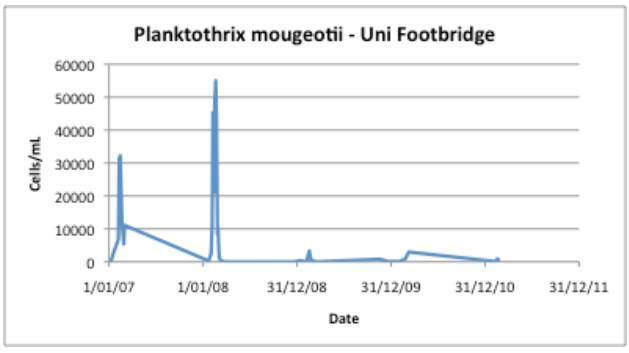












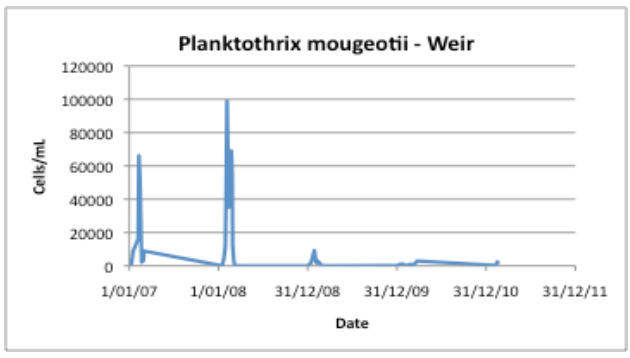


Figure 15. *Planktothrix mougeotii* cell numbers in the Torrens Lake.

Table 7. Number of sampling occasions when the biovolume of cyanobacteria populations exceeded the 10 mm^3/L guideline between 2007 and 2011.

	Number of sampling occasions when the biovolume of cyanobacteria populations exceeded the 10 mm^3/L guideline			
Site	***Anabaena circinalis***	***Microcystis aeruginosa***	***Microcystis flos-aquae***	***Planktothrix mougeotii***
Hackney Rd	3	0	0	0
Frome Rd	3	0	0	0
Uni Footbridge	5	0	0	0
King William Rd	4	0	0	0
Festival Theatre	5	0	0	0
Morphett St	5	0	1	0
Weir	2	0	1	0
Bonython Park	2	0	0	0

Table 8. Number of sampling occasions when the cell concentration of cyanobacteria populations exceeded 100,000 cells/mL between 2007 and 2011.

	Number of sampling occasions when the cell concentration of cyanobacteria populations exceeded 100,000 cells/mL			
Site	***Anabaena circinalis***	***Microcystis aeruginosa***	***Microcystis flos-aquae***	***Planktothrix mougeotii***
Hackney Rd	2	0	2	0
Frome Rd	2	0	4	0
Uni Footbridge	1	0	5	0
King William Rd	2	0	5	0
Festival Theatre	2	0	4	0
Morphett St	1	0	6	1
Weir	0	0	6	0
Bonython Park	1	0	0	0

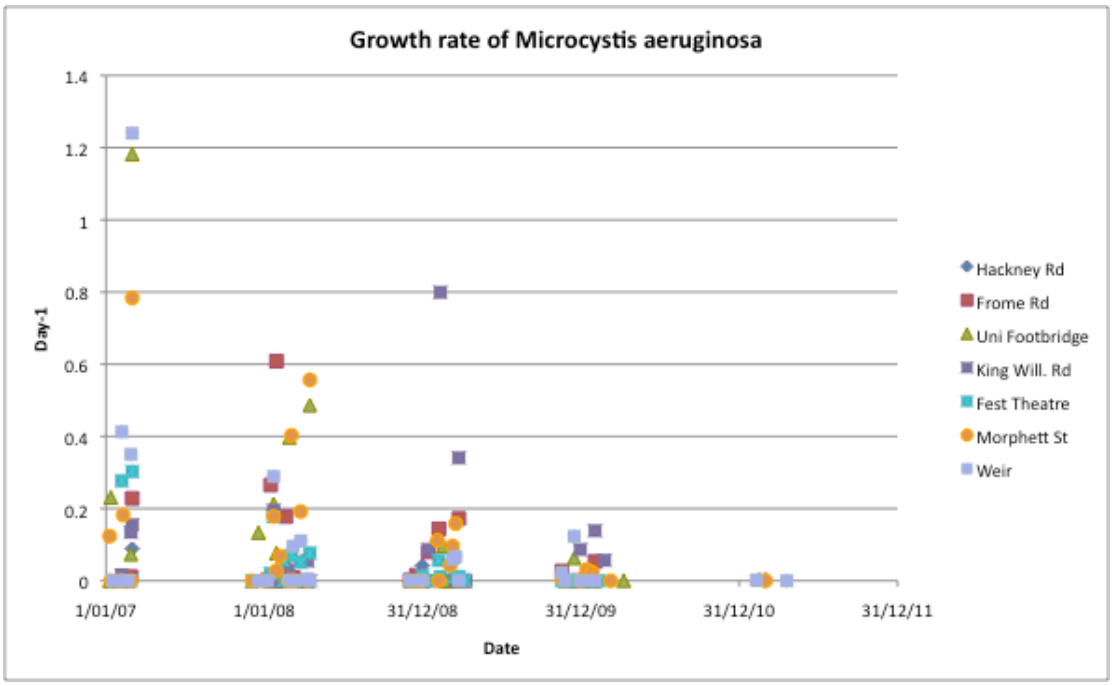


Figure 16. Growth rates of *Anabaena circinalis* in the Torrens Lake. Only positive growth is shown as this represents escalating risk.

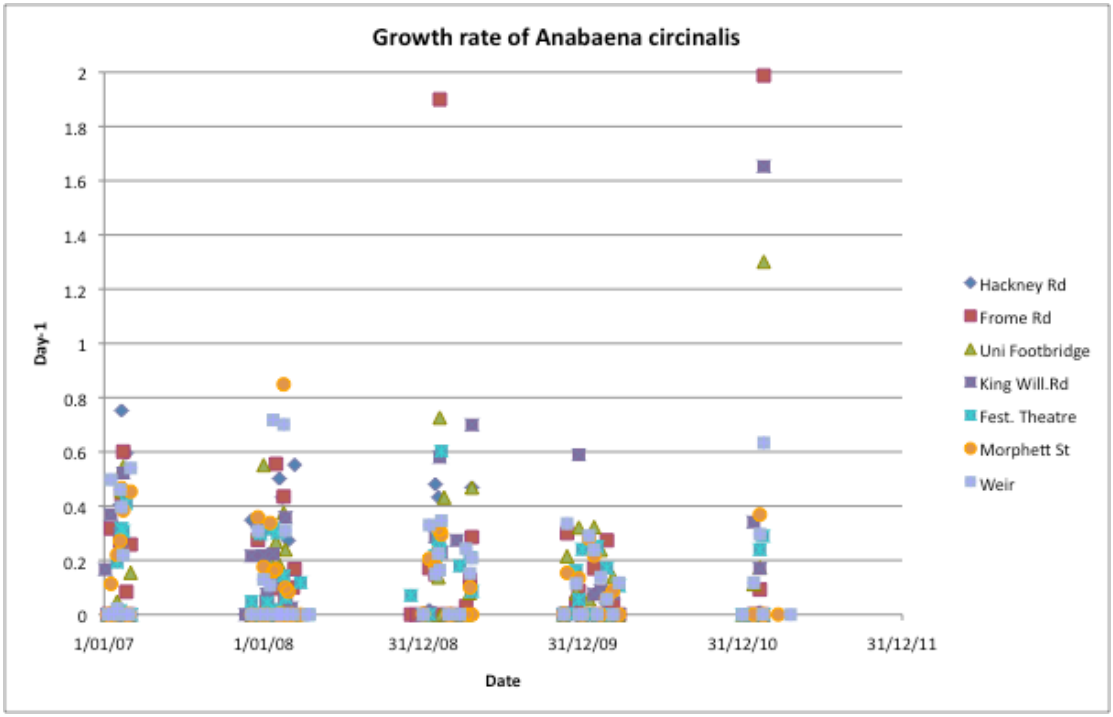


Figure 17. Growth rates of *Microcystis aeruginosa* in the Torrens Lake. Only positive growth is shown as this represents escalating risk.

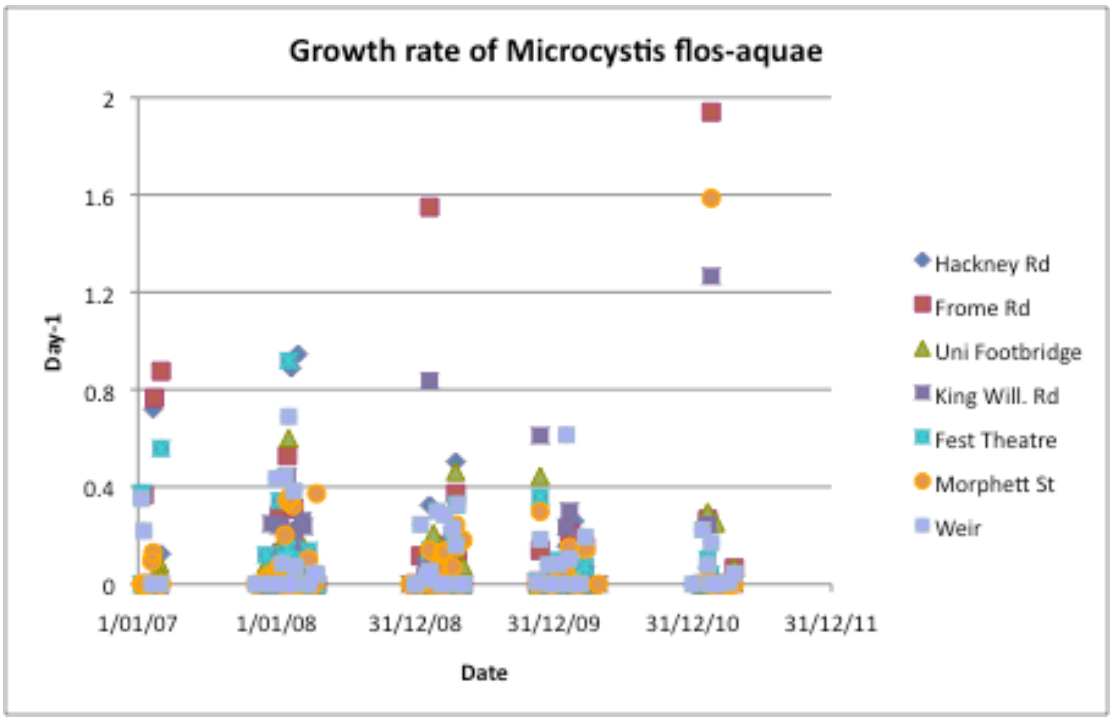


Figure 18. Growth rates of *Microcystis flos-aquae* in the Torrens Lake. Only positive growth is shown as this represents escalating risk.

2.3 Review of the hydrodynamic modelling and utilisation of this to determine the feasibility of dilution flows adequately mixing and diluting cyanobacteria from the lake

A three dimensional hydrodynamic model (Estuary and Lake Computer Model - ELCOM , Hodges et al., 2000)of the Torrens Lake was constructed by David Lewis from the University of Adelaide. This model considers meteorological inputs, inflows, outflows and predicts the temperature profile of the lake. The simulations were run for January 2000 meteorological conditions with seven different inflow rates (Table 9. Flow scenarios simulated with ELCOM for January 2000 meteorological conditions). Four flow volumes are presented below to assess how stratification may be affected by the inflow and how this may in turn affect the efficiency of dilution.

Two time periods are modelled; 12:35 hours – representing a mid-day period with strong stratification, and a morning period at 08:59 hours – when stratification is weaker. Figure 19 presents the temperature structure of the water body at four cross-sections across the lake at different locations.

It is evident that at the weir site the inflows of 0.3 m/s and greater erode the stratification in the hypolimnion. This weakening of stratification makes it easier for wind mixing and nocturnal cooling to overcome the buoyancy flux and mix the water column.

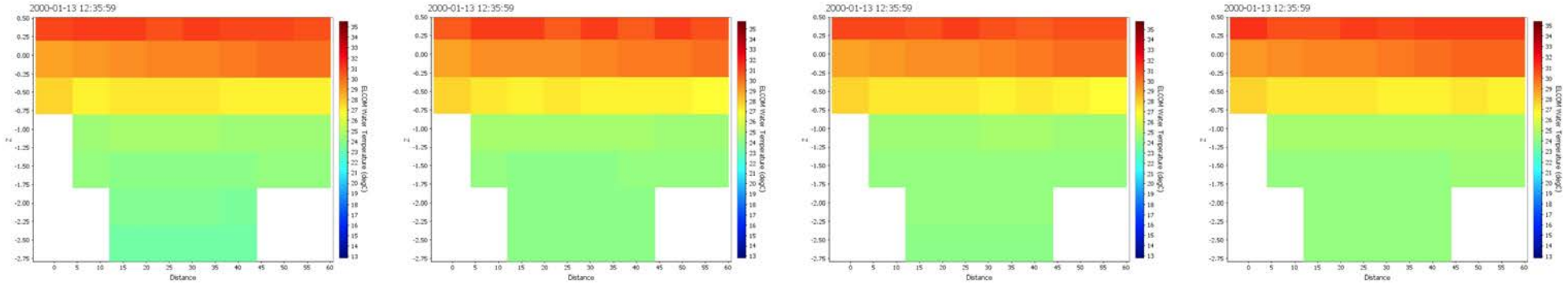
It can be concluded from this that the low flows will erode the stratification but not completely, mixing increases as the intrusion moves in to the lake and dilution will occur but not at 100% efficiency as predicted by the growth rate – dilution model presented previously. This inefficient dilution because of stratification needs to be accounted for when considering dilution as a mechanism for controlling cyanobacterial biomass in the Torrens Lake.

Table 9. Flow scenarios simulated with ELCOM for January 2000 meteorological conditions.

Simulation	Flow rate (m^3/s)	Daily volume (ML)	Torrens Lake volume (400ML) replacement (d)
1	0.1	8.64	46.3
2	0.2	17.28	23.1
3	0.3	25.92	15.4
4	0.5	43.2	9.3
5	1.0	86.4	4.6
6	1.2	103.68	3.8
7	1.5	129.6	3.1

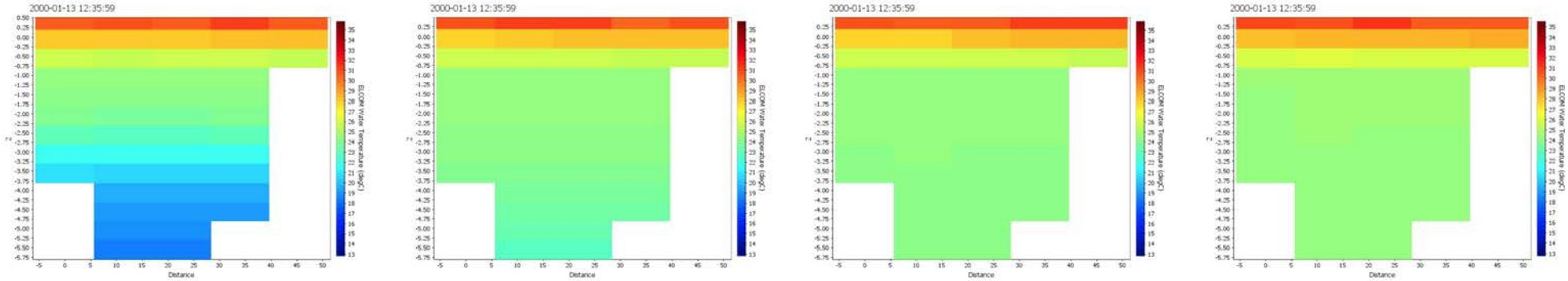
Figure 19 Temperature structure in cross-sections across the Torrens Lake with varying flow conditions

Victoria Bridge 13-Jan-2000 12.35 PM



0.1 m/s 0.3 m/s 1.0 m/s 1.5 m/s

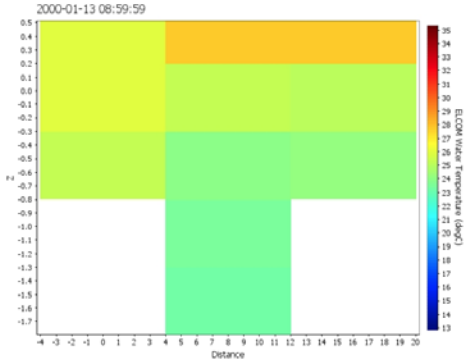
Weir 13-Jan-2000 12.35 PM



0.1 m/s 0.3 m/s 1.0 m/s 1.5 m/s

Results – Onset of stratification

Albert Bridge 13-Jan-2000 08:59AM



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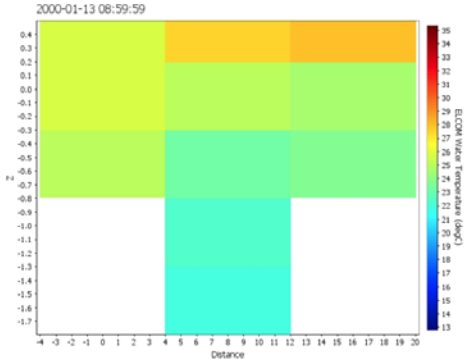

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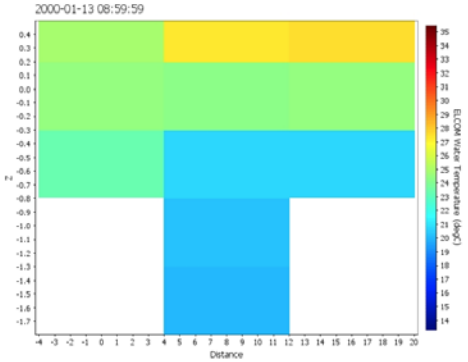

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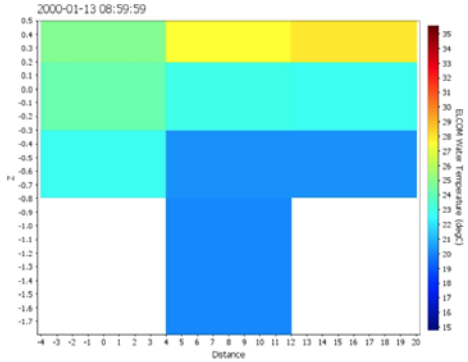

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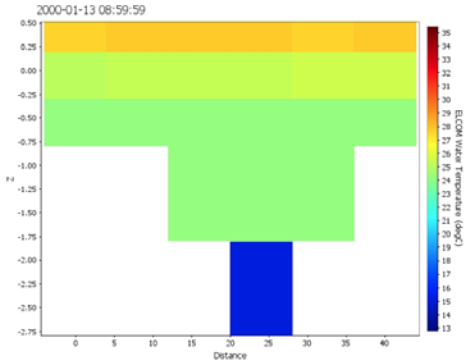

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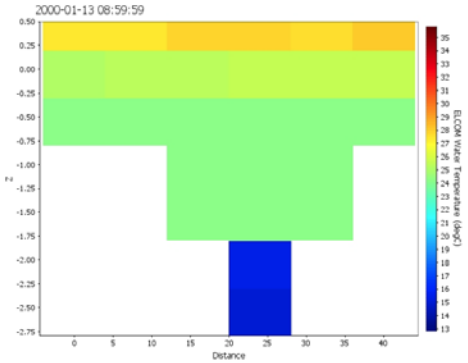

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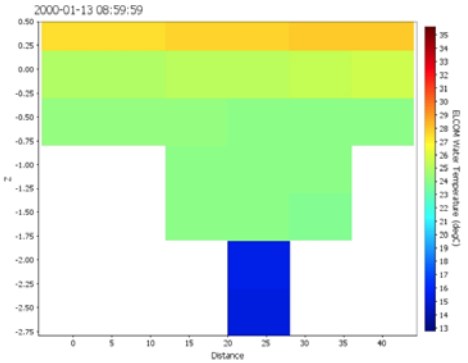

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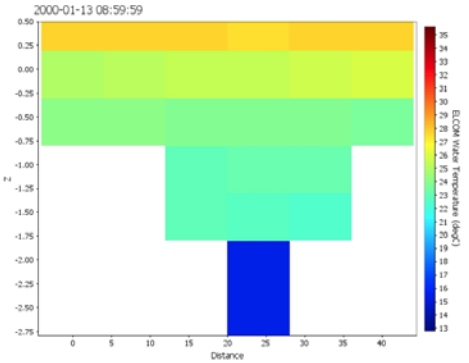

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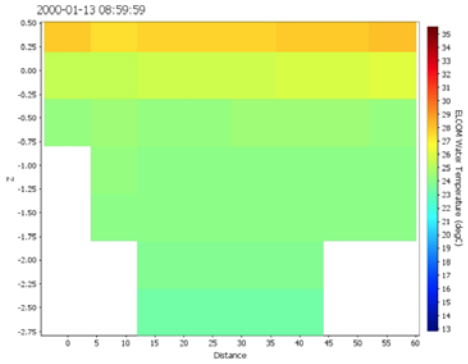

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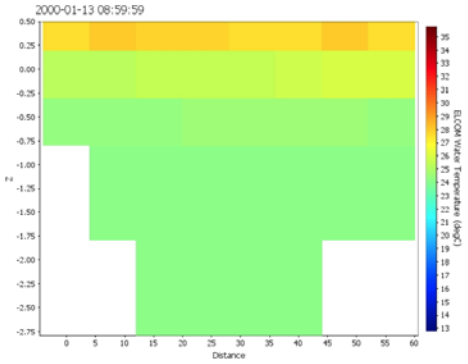

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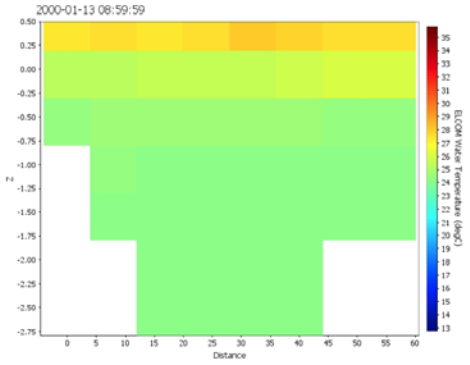

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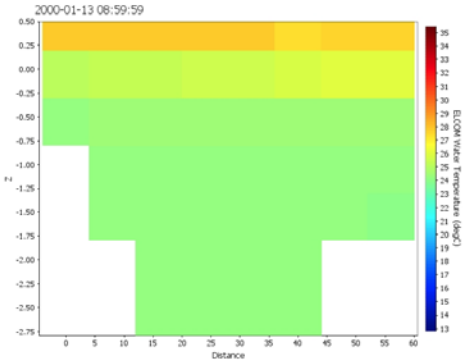

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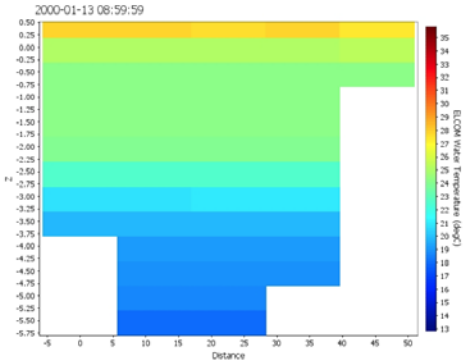

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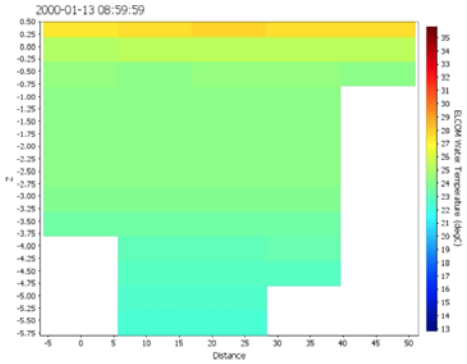

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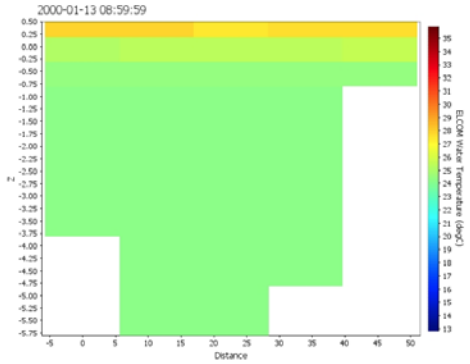

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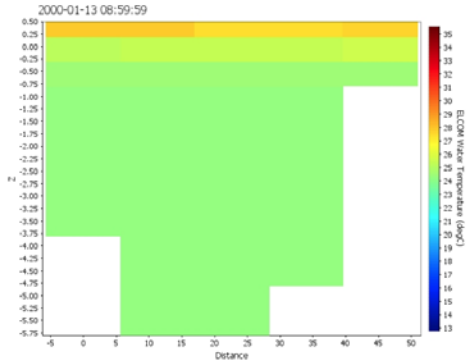

1.5 m/s

2.4 Development of a flow strategy that would document the required dilution flows at different cyanobacterial biomass and risk.

For a dilution flow strategy to be implemented there are a number of things to consider. The first is how much flow and for how long. This will change in response to risk, unless downstream purchasers require a reliable water source.

Assuming a growth rate of 0.4/day, which is a typical exponential growth rate of cyanobacteria in the Torrens Lake, it can be concluded that a diluting flow of at least 10% per day, 40 ML, would be required to have noticeable impact on the cyanobacteria population. With a starting cell concentration of 100 cells/mL, a growth rate of 0.4/day and a diluting flow of 10% the cell concentration after 20 days would be 74,420 cells/mL, which is below the critical threshold for cell numbers.

Further to this, the most suitable flow strategy to trial would be a relatively high flow during the months of January and February. It may be commenced earlier upon detection of cyanobacteria with concentrations exceeding 500 cells/mL at more than one site. The diluting flow should be continued at least until the end of February and possibly extended if cell concentrations exceed 500 cells/mL.

The two major limitations to this strategy being successful are:

- the ability to deliver flow adequate to sufficiently dilute the cyanobacterial population.
- the density difference that would occur between the inflowing water and the lake. The lake stratification would inhibit mixing and reduce the effectiveness of the dilution process.

2.5 Assess the likelihood of the various flow strategies actually achieving the reduction in biomass based on the stratification, the flow rates and density differences between inflow and the lake water.

Without adequate information on inflow temperature, it is difficult to precisely predict where the inflow will intrude into the lake. At low flow, little mixing will be generated and the inflow is likely to maintain its integrity a fair way into the lake. ELCOM modelling shows the epilimnion is eroded as the inflow moves further into the lake. This will increase mixing and dilution.

The depth of extraction from Kangaroo Creek Dam could be manipulated to select water from a preferred depth and temperature/density. The difficulty with this is that the water's temperature will change as it travels down the River Torrens so will be challenging to manage. There is usually temperature stratification in Kangaroo Creek Dam (Figure 20) but the temperature in the River Torrens at Gorge Weir (Figure 21) is usually a couple of degrees cooler than the surface temperature in the upstream reservoir.

A field program trialling dilution flows could be designed to adequately assess where the inflow is going and how much dilution is achieved. This would require thermistors at several sites and gauging of flow.

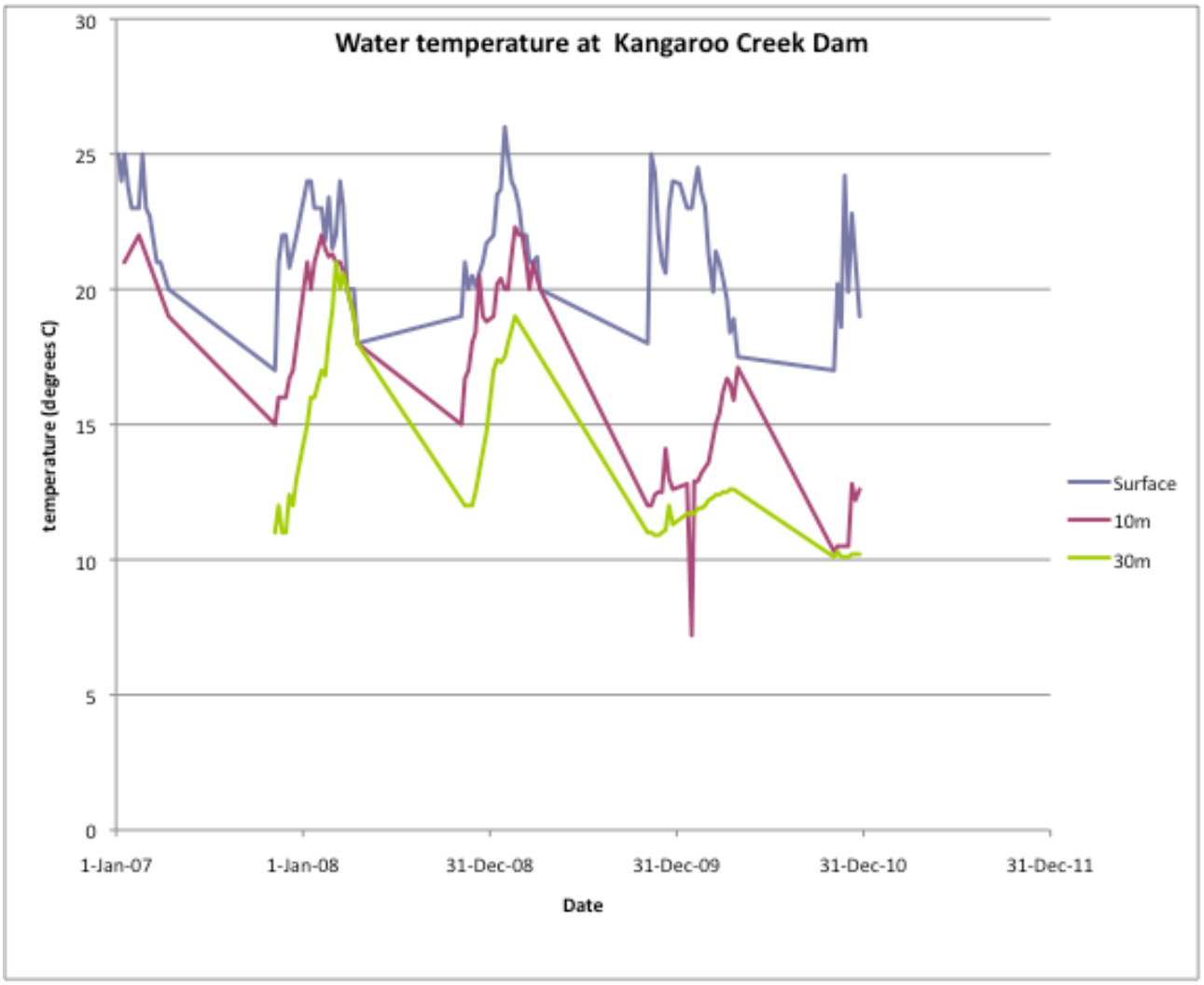

Figure 20. Water temperature at three depths in Kangaroo Creek Dam.

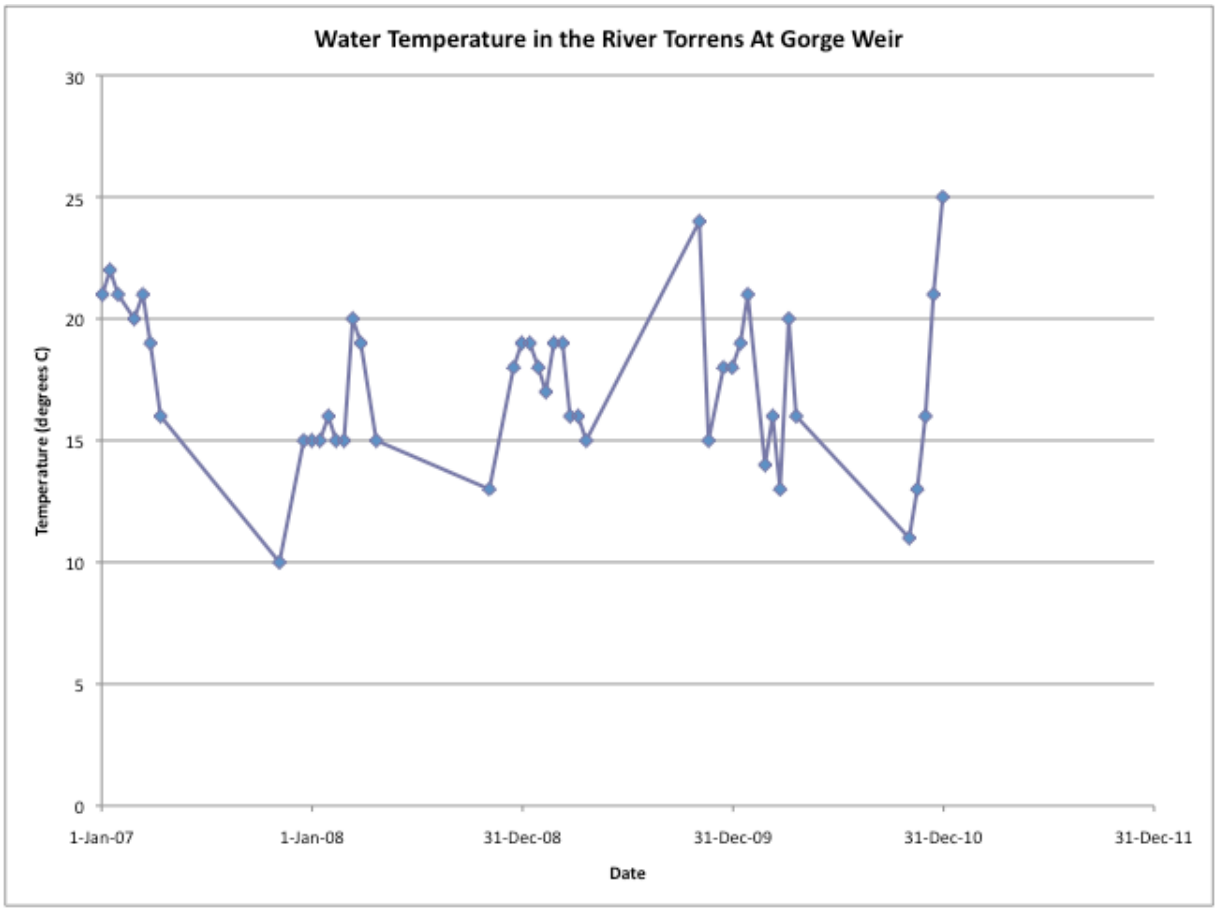

Figure 21. Water temperature in the River Torrens at Gorge Weir.

2.6 Determining the appropriateness of various source water to provide diluting flows based on nutrient concentrations

Source water quality needs consideration when delivering diluting flows. Water accessed from either the hypolimnion of Kangaroo Creek Dam, or from reuse would tend to be high in phosphorus and nitrogen. Water with low nutrients would tend to limit the rate of growth and reduce the maximum possible algal biomass, and should be selected in preference to other sources.

Anabaena circinalis is able to fix atmospheric nitrogen and so the focus should be on selecting water low in phosphorus. Total phosphorus concentrations less than 40 μg/L and filterable reactive phosphorus concentrations less than 10 μg/L would be preferable but these may be hard to obtain in the Torrens Catchment, without additional phosphorus stripping.

The nutrient concentrations and water quality of various source waters were examined to determine the range of water quality expected in the diluting flow. Reuse water could be available

from the Glenelg-Adelaide Pipeline (GAP). Water quality data was supplied by the South Australian Water Corporation. The data presented was for water from the sampling location in North Adelaide Parkland 100m from Main North Rd/O'Connell St.

The GAP water has relatively high conductivity, typically between 1600 µS/cm and 2500 µS/cm (Figure 22). As a short-term diluting flow this high conductivity may be of minor environmental consequence but this may become problematic as the water replaces most of the lake volume. Some lake species and species downstream could be under salinity stress at concentrations of 2500 µS/cm.

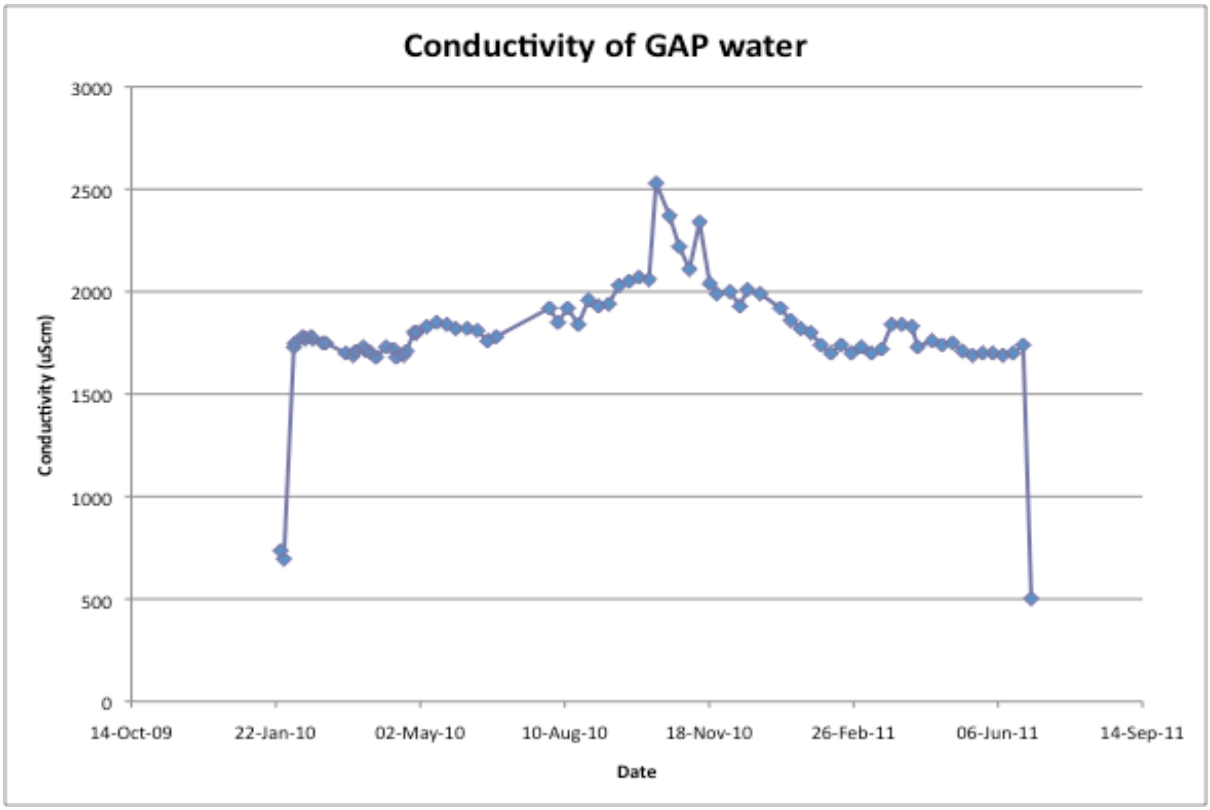

Figure 22. Conductivity of reuse water from the Glenelg-Adelaide Pipeline.

Of greater concern than the conductivity are the high nutrient concentrations in the GAP reuse water. Filterable reactive phosphorus concentrations were between 5 and 8 mg/L (Figure 23). This concentration is three orders of magnitude higher than the ideal concentrations for filterable reactive phosphorus, which is the most bioavailable form of phosphorus. These concentrations would likely lead to excessive algal growth, regardless of the dilution caused by the programmed diluting inflow. Ammonia concentrations (Figure 24) are high but not excessive, however, this is because most of the nitrogen is present in oxidized form; nitrate concentrations range from 4-11 mg/L (Figure 25).

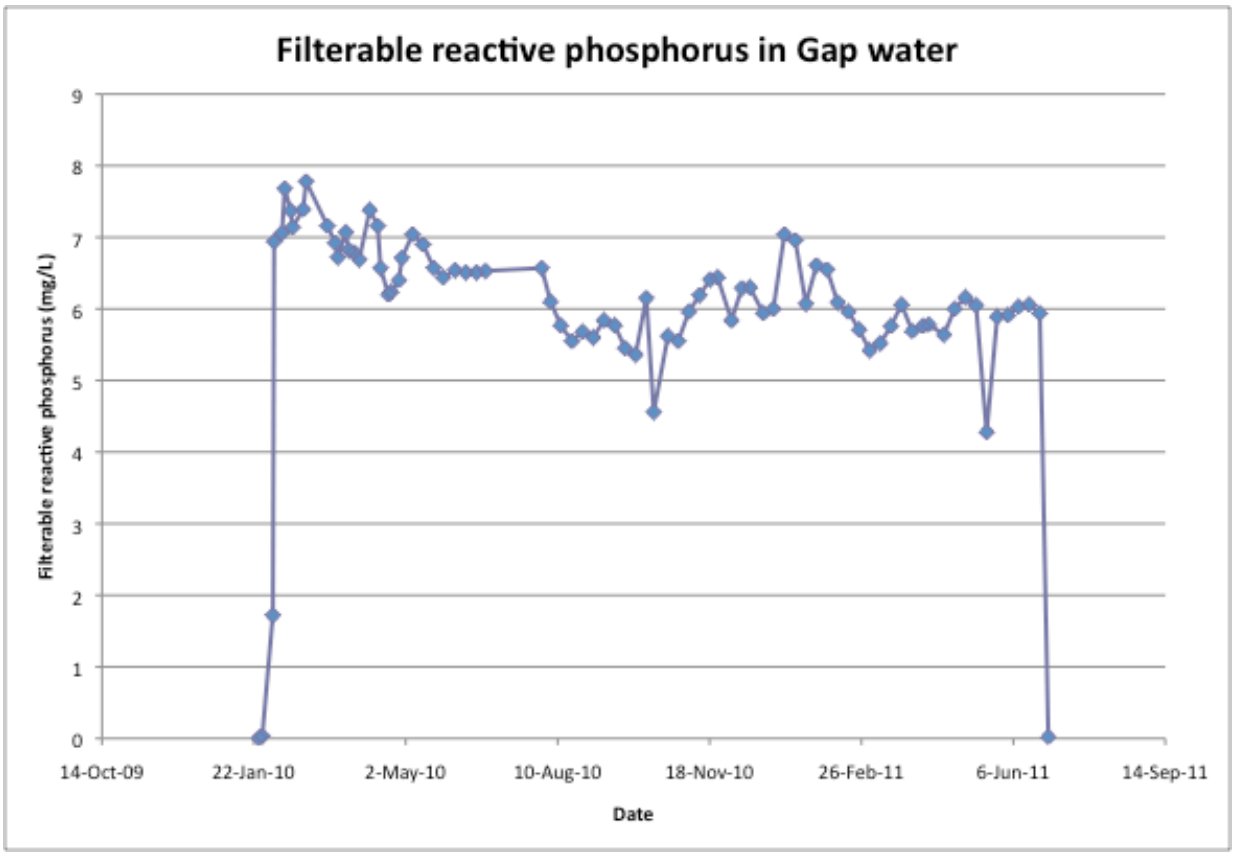

Figure 23. Filterable reactive phosphorus of reuse water from the Glenelg-Adelaide Pipeline.

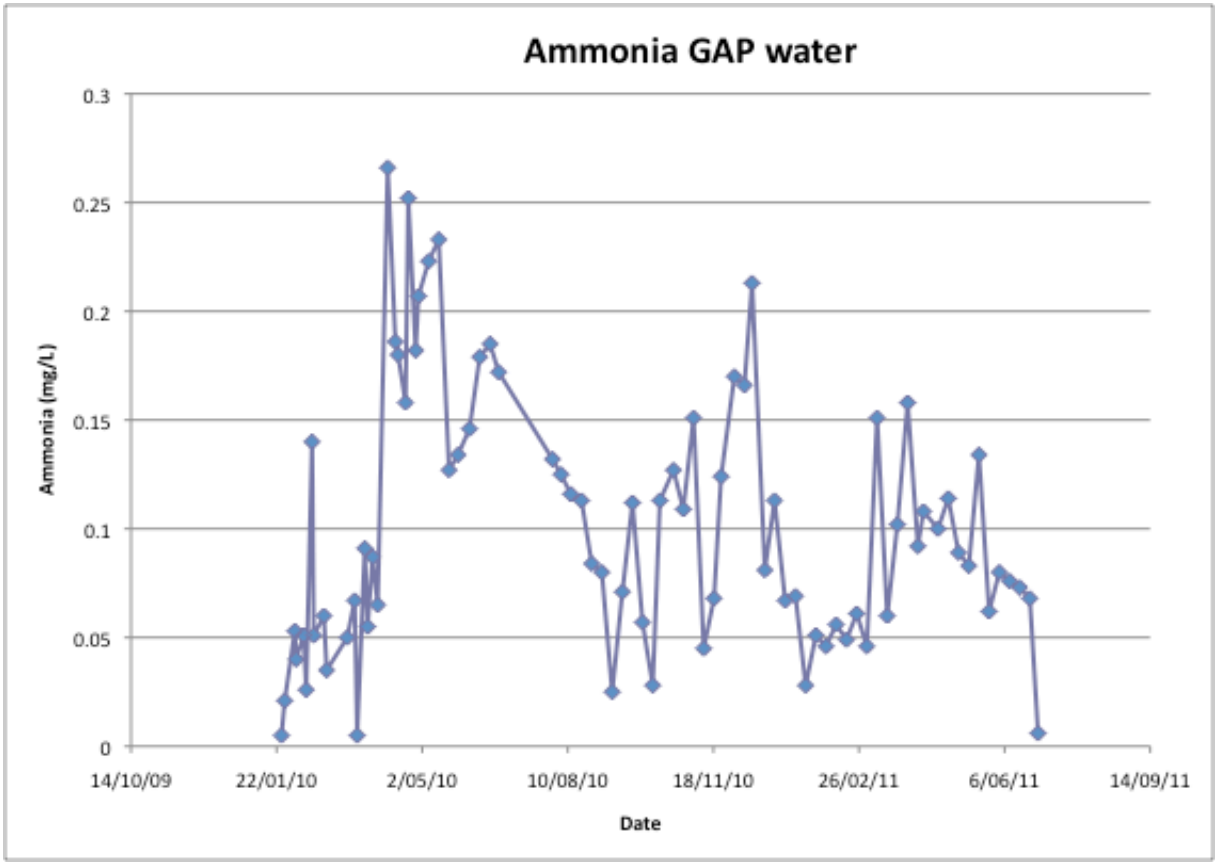

Figure 24. Ammonia concentration of reuse water from the Glenelg-Adelaide Pipeline.

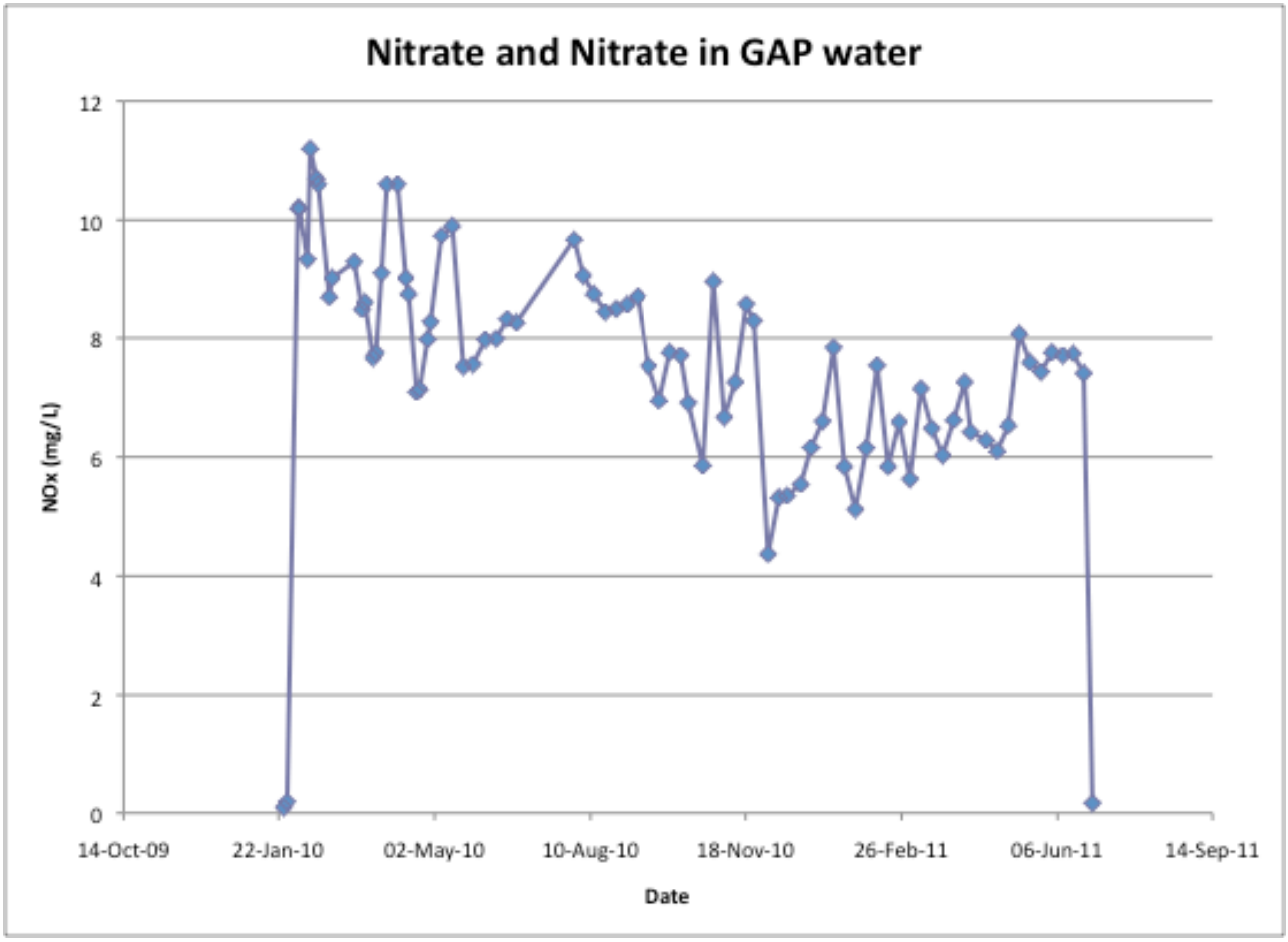

Figure 25. Nitrate and nitrite concentration of reuse water from the Glenelg-Adelaide Pipeline.

Water in Kangaroo Creek Dam had considerably lower nutrient concentrations than the GAP water. Data presented are for Kangaroo Creek Dam and were collected at the surface at location 1. Where concentrations were less than the minimum detectable level (<0.005 mg/L) the concentration was set to 0.005 to enable plotting. Ammonia concentrations ranged between the minimum level of detection (0.005 mg/L) and 0.03 mg/L with one higher concentration at the end of the period (Figure 26). Nitrate and nitrite concentrations were often at or below the minimum level of detection (Figure 27) while filterable reactive phosphorus concentrations were similarly low and approaching or below the minimum level of detection (Figure 28). Total phosphorus typically ranged between 0.02 and 0.045 mg/L with one high concentration at the end of the period (Figure 29).

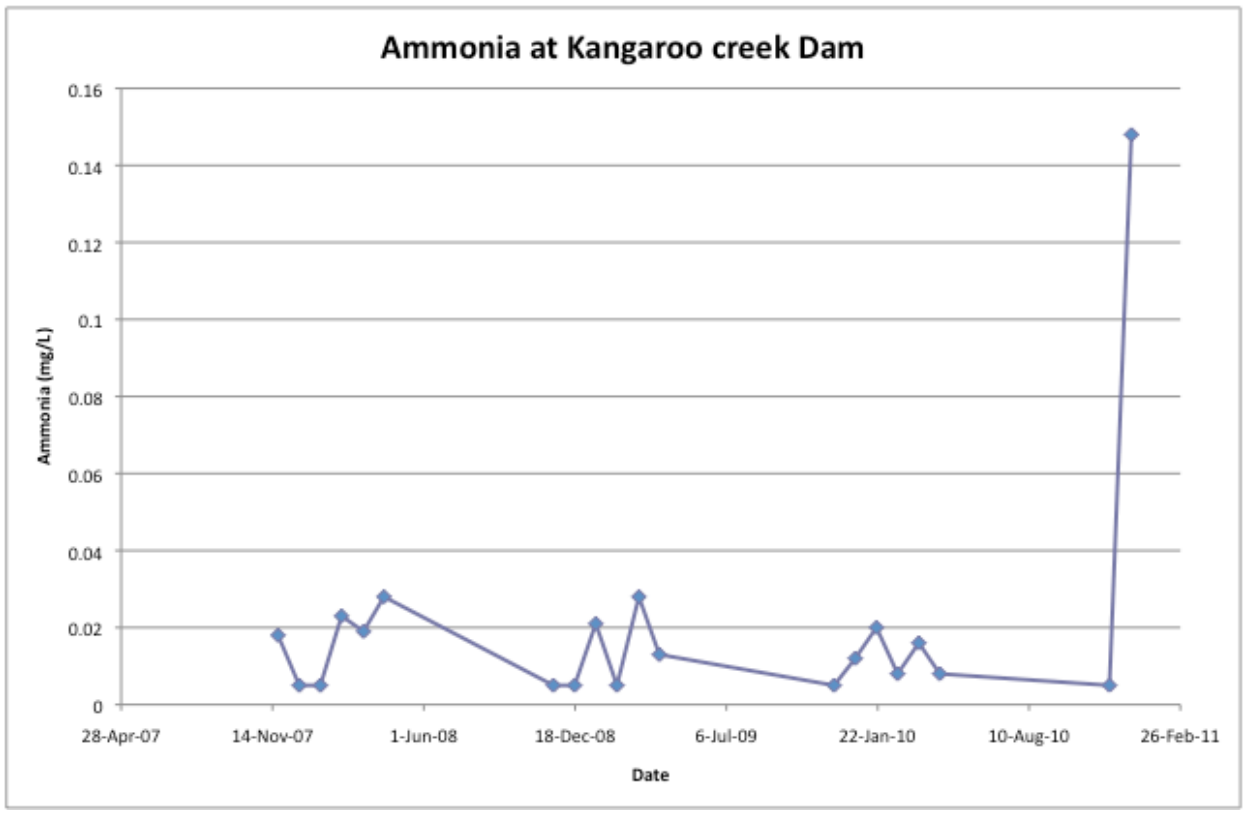

Figure 26. Ammonia concentrations in Kangaroo Creek Dam.

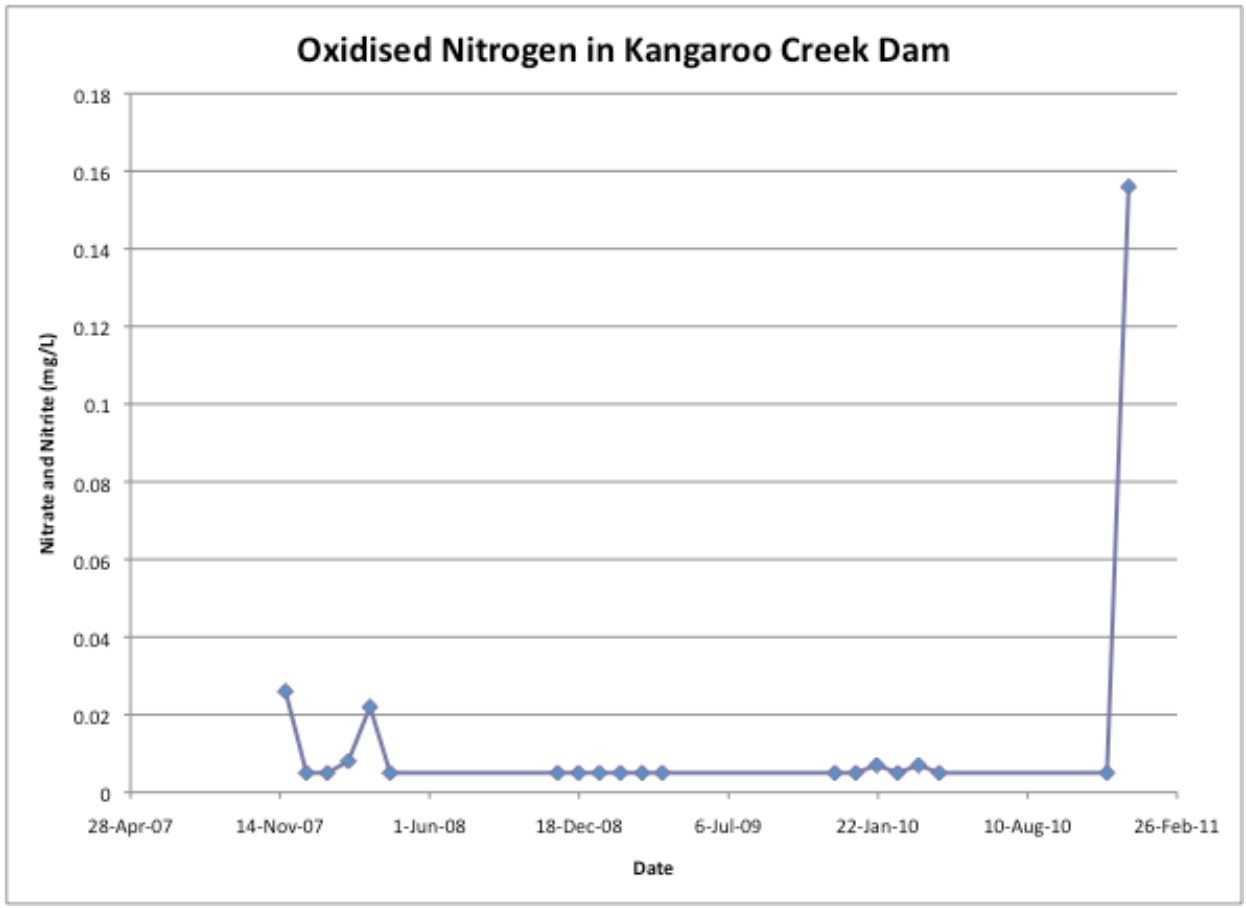

Figure 27. Oxidised nitrogen concentration in Kangaroo Creek Dam.

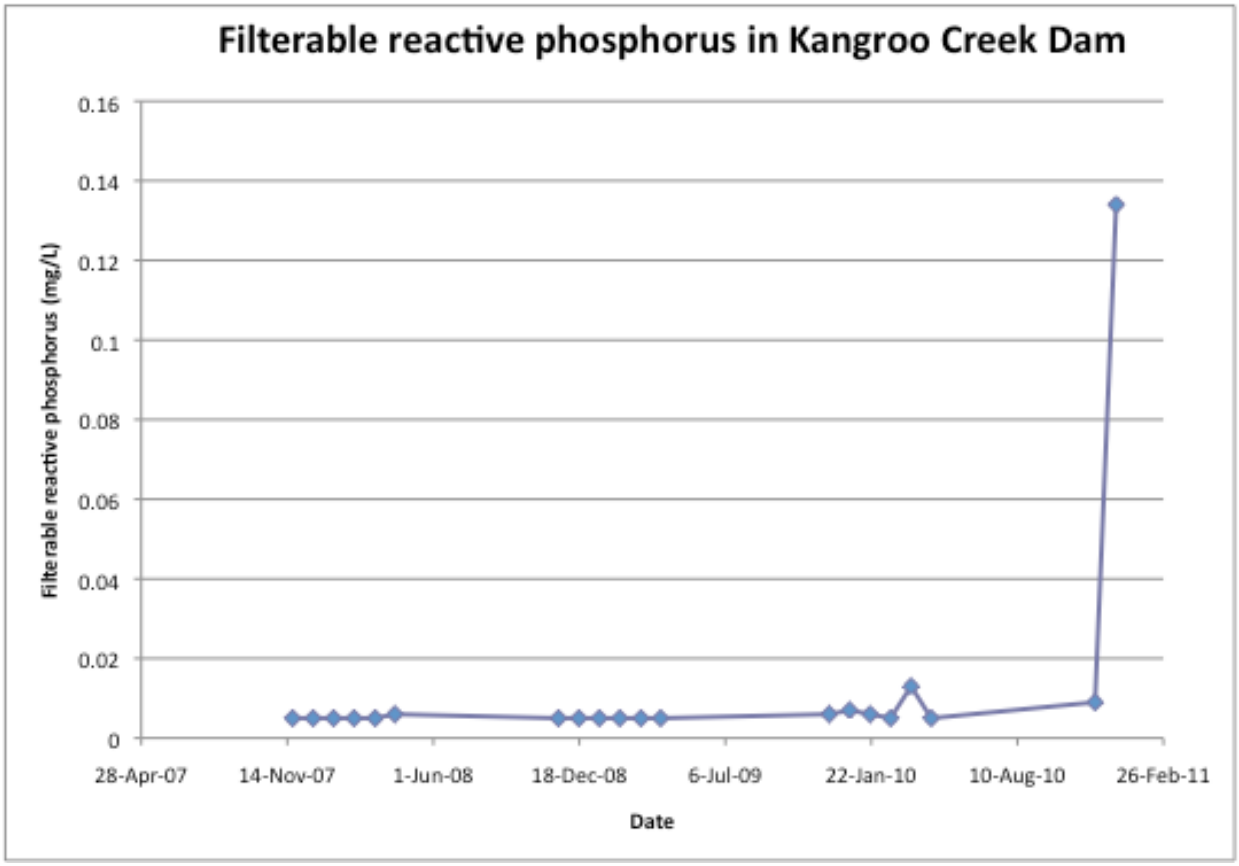

Figure 28. Filterable reactive phosphorus concentrations in Kangaroo Creek Dam.

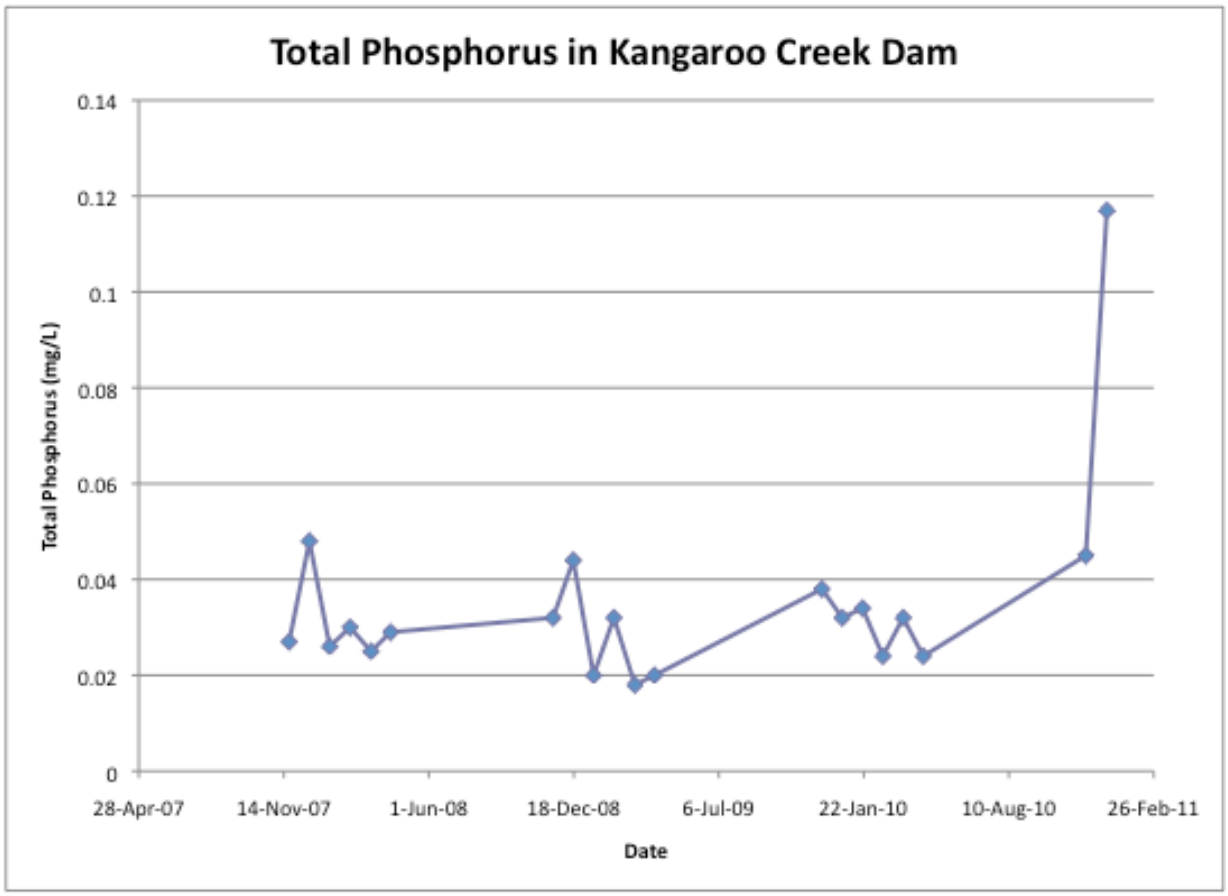

Figure 29. Total phosphorus concentrations in Kangaroo Creek Dam.

Based on the nutrient concentrations it is apparent that reuse water from the GAP scheme are too high to discharge into a natural stream. The high nutrient loads are highly likely to cause excessive algal growth in the short term. These algae would tend to sediment and accumulate high phosphorus concentrations in the sediment, which could become problematic in the long term. Using high nutrient water for dilution flows may require EPA permits. On the other hand, the concentrations observed at Kangaroo Creek Dam are sufficiently low that they would contribute little additional phosphorus to the system and would be the most suitable choice for dilution flows.

2.7 Conclusions

For a growth rate of 0.4 day^{-1}, which is a typical exponential growth rate of cyanobacteria in the Torrens Lake, it can be concluded that a diluting flow of at least 10% per day would be required to have noticeable impact on the cyanobacteria population. With a starting cell concentration of 100 cells/mL, a growth rate of 0.4 day^{-1} and a diluting flow of 10% the cell concentration after 20 days would be 74,420 cells/mL, which is below the critical threshold for cell numbers.

Stratification will decrease the efficiency of dilution, but there is evidence from the ELCOM hydrodynamic model that flows greater than 0.3 m/s (6.5%) dilution erode stratification particularly as the intrusion moves further into the lake. This will act to make the lake more vulnerable to natural wind mixing and complete mixing from nocturnal cooling. These processes will tend to make the cyanobacterial population more homogeneously mixed and dilution more efficient. If the weir is allowed to be an over-top spill weir and not operated as an underflow structure, this will also maximize cyanobacteria loss from the system.

There is potential for dilution from inflows to reduce cyanobacterial biomass in the Torrens Lake, but inflow must be sufficiently high to overcome the growth rate and also account for inefficiencies in dilution due to stratification. A base flow in summer of 6.5% dilution would be recommended but this may need to increase to over 10% when the algal population begins to show exponential growth.

2.8 References

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Chapter Three: Torrens Lake Water Quality Improvement Trial

3.1. Introduction

Ongoing cyanobacterial blooms in the Torrens Lake have prompted exploration into numerous management techniques to control the blooms. Catchment management is ongoing but it may take decades for the full benefits of this to be realised in lake water quality. The assessment of the various in-lake techniques also revealed that it would be challenging to control the growth of cyanobacteria; artificial destratification is challenging in shallow environments where phytoplankton can still access adequate light, nutrient control is challenging when there is continuous renewal from the catchment.

One strategy that has been instigated is the use of controlled releases from upstream catchment or from reuse water. This strategy relies upon the diluting flows flushing the cyanobacteria from the system and thereby limiting the accumulation of high biomass. There are several challenges with this strategy, not least of which is achieving the desired dilution given the likely density difference between the inflow and lake water.

The previous chapter, considering the feasibility of dilution flows for cyanobacteria management in the Torrens Lake, concluded that for a growth rate of 0.4 day^{-1}, a diluting flow of at least 10% per day would be required to have noticeable impact on the cyanobacteria population. With a starting cell concentration of 100 cells/mL, a growth rate of 0.4 day^{-1} and a diluting flow of 10% the cell concentration after 20 days would be 74,420 cells/mL, which is below the critical threshold for cell numbers. Higher rates of growth may reach the threshold concentration in shorter timeframes.

A trial was proposed to determine how effective a flow release management strategy is at controlling cyanobacteria in the Torrens Lake to below the threshold of 40,000 cells/mL of *Anabaena*. It is noted that even below this concentration the cyanobacteria can accumulate at the surface and form unsightly surface scums. Because there is uncertainty of how well the inflows will mix the lake water and dilute the population it will be necessary to monitor the inflows and their impact upon lake hydrodynamics and cyanobacterial abundance. The criteria for the monitoring are to confidently know that the flow dilution releases are stopping algal blooms and poor water quality episodes.

The premise for this trial is that rather than controlling growth, which is proving difficult, the population size could be controlled by continual dilution. The lake is used predominantly for recreational activities such as fishing, rowing and paddle boating, and so maintaining the lake 'open' for these activities is a priority. The recreational guideline for cyanobacteria is a biovolume equivalent of 10 mm^3/L. There is an interspecies difference in the number of cells per millilitre to achieve the biovolume threshold (Table 10).

Table 10. Cyanobacteria cell concentrations that reach the recreational guideline for cyanobacteria in the Torrens Lake.

Cyanobacteria	Cell count (cells/mL)	Biovolume equivalent (mm^3/L)
Anabaena circinalis	40,000	10
Microcystis aeruginosa	115,000	10.005
Microcystis flos-aquae	455,000	10.01
Planktothrix mougeotti	156,500	10.016

The aim of the amenity flow trial was to use flow from an upstream storage to dilute the cyanobacterial populations in the Torrens Lake before they reach large numbers. Due to another environmental flows trial immediately downstream of Gorge Weir it was decided to take water from Hope Valley Reservoir.

The challenging aspect of this type of trial is that it is difficult to prove success (i.e. no cyanobacterial blooms) but it is easy to prove failure (i.e. cyanobacterial numbers exceed the threshold of 40,000 cells/mL of *Anabaena circinalis*). This is because there may be numerous reasons, in addition to the diluting flows, why cyanobacteria do not reach problematic concentrations. These may include nutrient-limitation, low inoculum, natural washout, unsuitable hydrodynamic conditions and competition for resources by other algae.

3.2. Methods

3.2.1. Site description

The sampling sites used in this proposed monitoring program (Table 11) were the same as those used in the regular monitoring program for algal counts by the City of Adelaide. This will enable consistent datasets, comparison with historical records and a representative coverage of the lake.

Figure 30. Location of monitoring sites 1-7 on the Torrens Lake, Adelaide.

Table 11. Description of the monitoring sites and a summary of the monitoring program.

Site #	Site Description	Temperature	Velocity	Cell counts	Nutrients
1	Torrens Lake Weir		2 day intensive sampling every two hours	Twice Weekly	Weekly
2	Morphett Street Bridge	Thermistor chain	2 day intensive sampling every two hours	Twice Weekly	Weekly
3	Elder Park	Thermistor chain	2 day intensive sampling every two hours	Twice Weekly	
4	King William Road Bridge			Twice Weekly	Weekly
5	University Footbridge			Twice Weekly	
6	Frome Road Bridge	Thermistor chain	2 day intensive sampling every two hours	Twice Weekly	Weekly
7	Hackney Road Bridge	2 thermistors	2 day intensive sampling every two hours	Twice Weekly	
7b	Hackney Road Bridge	1 thermistor			

3.2.2. Hydrodynamics

The premise of controlled upstream water releases to control cyanobacteria in the Torrens Lake is that there is sufficient dilution and loss of cells downstream to overcome growth and biomass expansion in the lake. Stratification will act to decrease the efficiency of dilution but there is evidence from the ELCOM hydrodynamic model that flows greater than 0.3 m/s (6.5%) dilution erode stratification particularly as the intrusion moves further into the lake (see Chapter 2). This will act to make the lake more vulnerable to natural wind mixing and complete mixing from nocturnal cooling. These processes will tend to make the cyanobacterial population homogeneously mixed and dilution more efficient. If the weir is allowed to be an over-top spill weir and not operated as an underflow structure, this will also maximise cyanobacteria loss from the system.

Four thermistor chains were installed in the lake with an additional thermistor installed in the river upstream of the lake. The upstream thermistor (site 7) was 'lost' and no data was available for this site. The meteorological station and thermistor chain owned and operated by the Adelaide City Council was also not operational during the study period and data was unavailable.

RBR TR-1050 thermistors were checked for calibration prior to deployment and thermistors found to diverge by more than 0.1°C when suspended in the same water were not used. Thermistor chains at sites 2,3 and 6 each had thermistors deployed at 0.1m, 0.5m, 1.0m, 1.5m and 2.0m. Site 7 had thermistors at 0.1m and 1.3m.

3.2.3. Phytoplankton and nutrients

Phytoplankton was sampled twice a week at sites 1-7. Sampling was undertaken with a 2m hose pipe integrating over the water column. Cell counts were performed by the Australian Water Quality centre, a NATA accredited laboratory.

Samples for nutrient analysis were collected weekly at sites 1, 2, 4, and 6 and were stored on ice prior to analysis for Total phosphorus (TP), Total Kjeldahl Nitrogen (TKN), Filterable Reactive Phosphorus (FRP), Ammonium (NH4) and oxidised nitrogen (nitrate and nitrite, NOx). All chemical analysis was undertaken by the Australian Water Quality Centre.

3.2.4. Evaluation of the flow trial against predicted growth rate and dilution

To aid in evaluating the flow trial the predictive model of cyanobacterial growth and dilution, described in Chapter 2, was applied to conditions that occurred in the Torrens Lake in January and February 2012. The increase in cell concentration was determined for growth rates of 0.25/day and 0.3/day. The flow scenarios were: no dilution, 10% dilution (40 ML/day) and the actual flow measured at Holbrooks Road. The predicted *Anabaena circinalis* population was compared against the population at Site 1 and the average cell concentration for all seven sites.

3.3. Results

3.3.1. Hydrodynamics

Temperature conditions in the Torrens Lake were suitable for cyanobacterial growth. During the period from mid December 2011 to 30 March, 2012 the water temperature at site 2 was less than 20°C for only six days. Water temperatures at the surface reached as high as 31°C and the diurnal heating of the surface water led to regular stratification (Figures 31 - 34). Stratification was not evident on days when the water column was cooling. Thermistors chains at all sites recorded similar patterns of temperature change but there were differences in the maximum and minimum temperature and the magnitude of daily stratification. The differences in temperature will drive currents (Monismith et al., 1990; Wells and Sherman, 2001) as warmer, less dense water floats over cooler water, which sinks.

The nocturnal mixing will entrain cells and distribute the phytoplankton community throughout the water column. This will be counteracted by the buoyancy of cyanobacteria which float, however they tend to be entrained from the surface when wind speeds exceed $3ms^{-1}$ (Webster and Hutchinson, 1994).

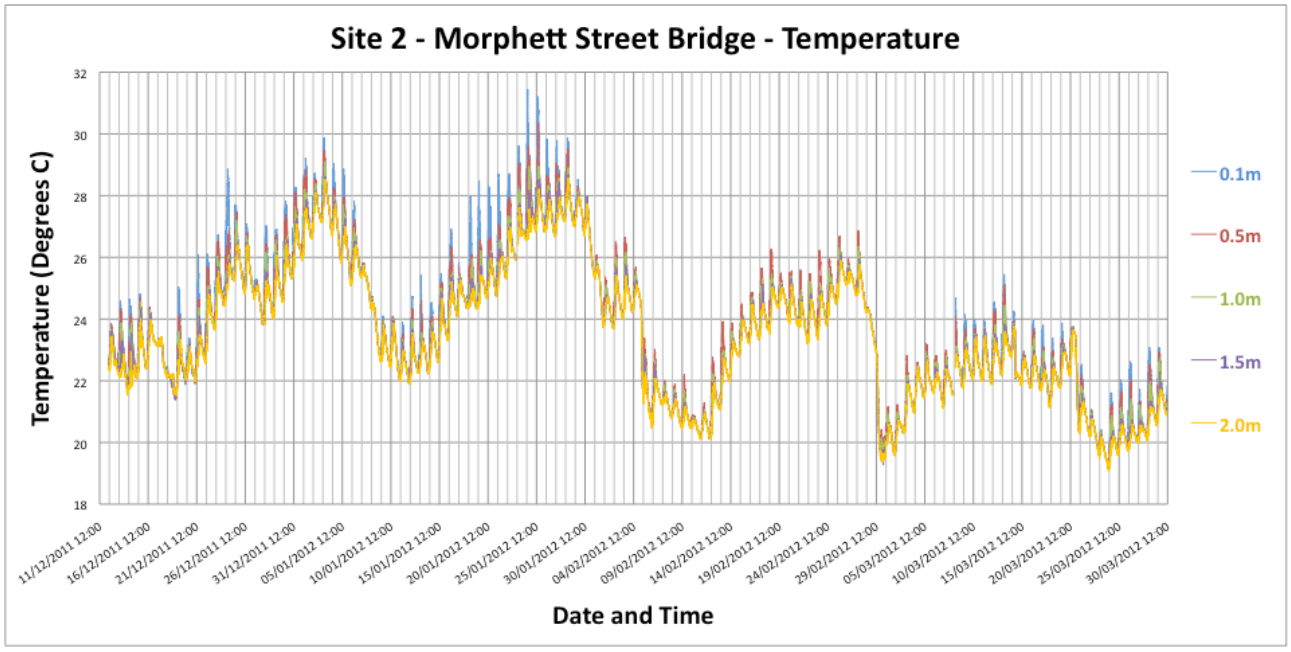

Figure 31. Water temperature measured with a thermistor chain at Morphett Street Bridge.

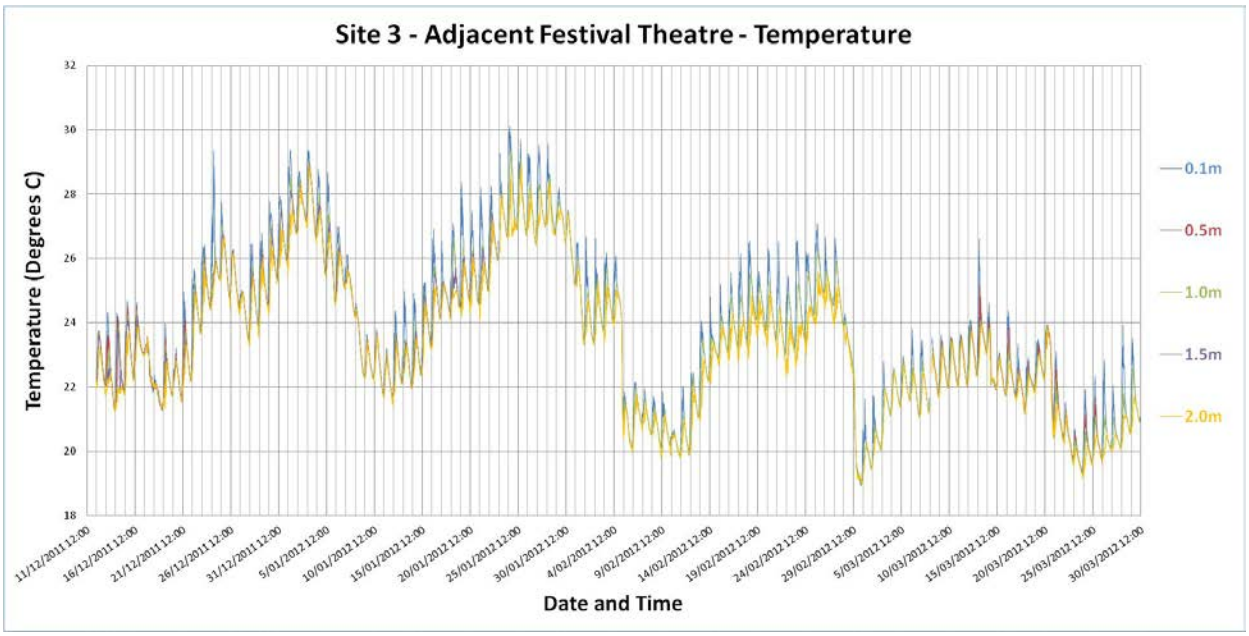

Figure 32. Water temperature measured with a thermistor chain adjacent to the Festival Theatre.

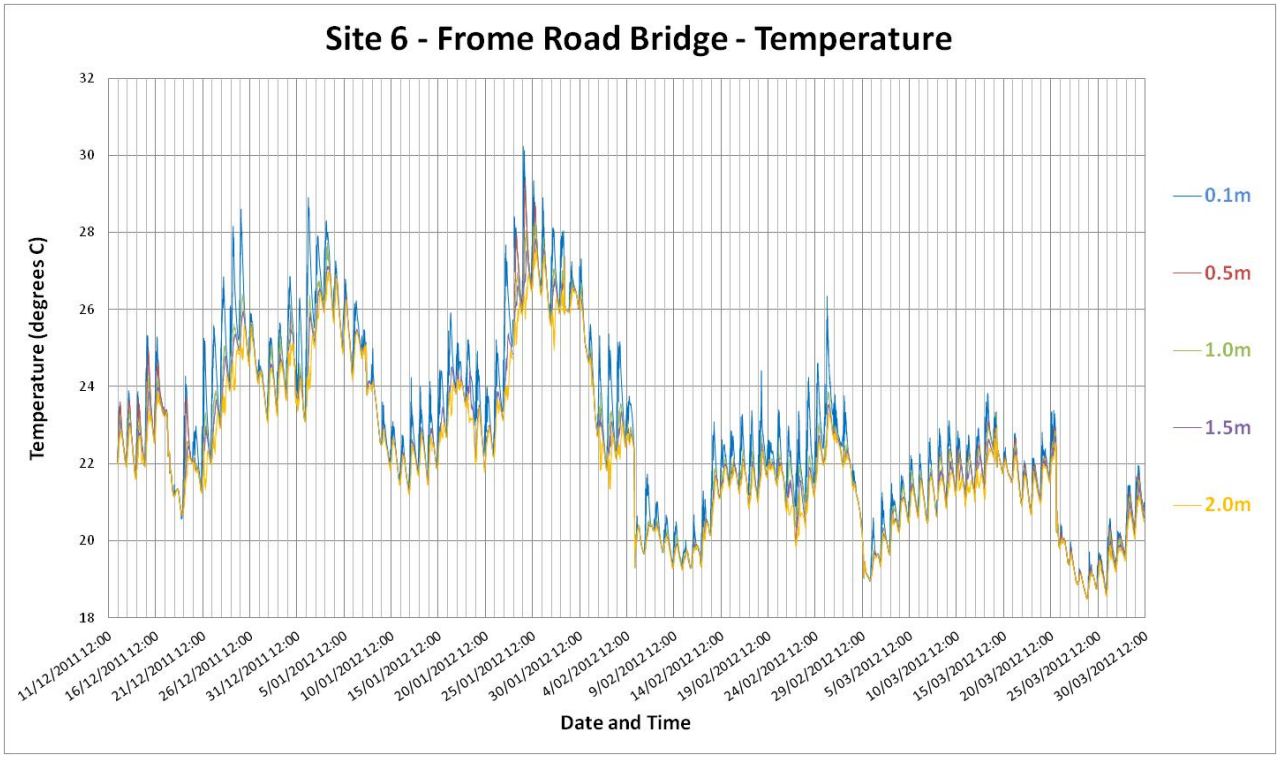

Figure 33.Water temperature measured with a thermistor chain at Frome Road Bridge.

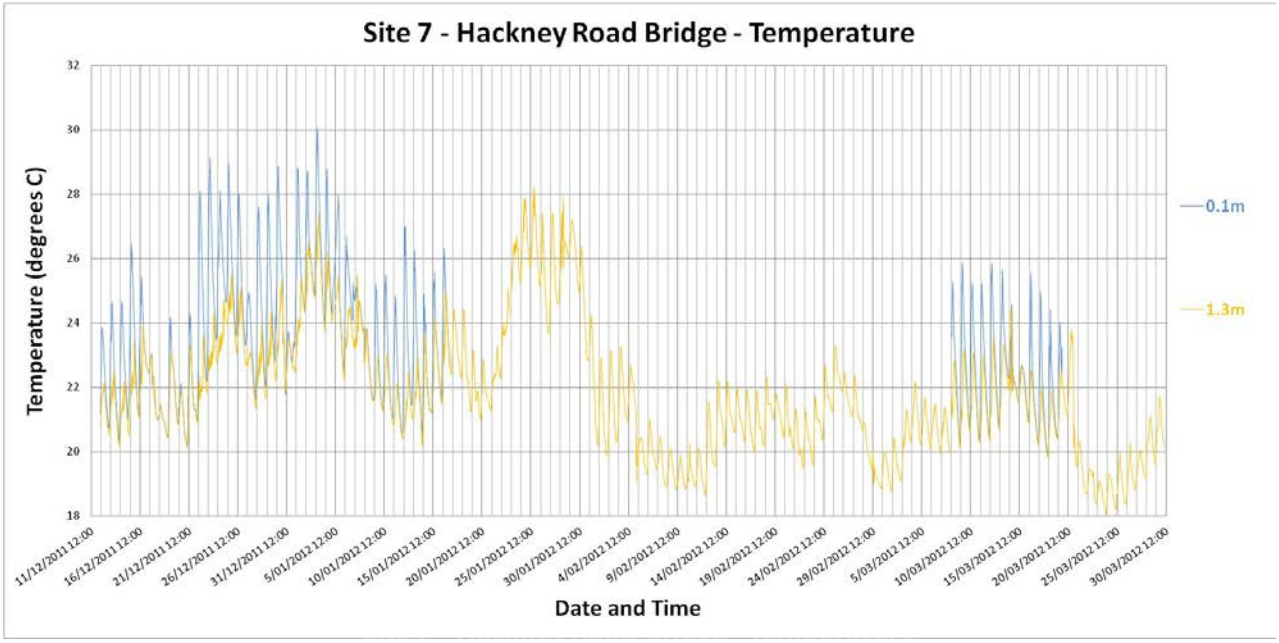

Figure 34. Water temperature measured with a thermistor chain at Hackney Road Bridge.

3.3.2. *Flow*

Flow in the River Torrens was recorded at two sites, upstream and downstream of the Torrens Weir (Figure 35). The upstream site was situated immediately downstream of the confluence of Second Creek and the Torrens River. The site downstream of the Torrens Lake was at Holbrooks Road. Flow until 13 January 2012 was the product of rainfall and catchment runoff. On 13 January a 20 ML flow release from Hope Valley Reservoir commenced in an attempt to dilute increasing cyanobacterial numbers in the Torrens Lake. Flow releases had been requested earlier, when cell numbers approached 1,000 cells/mL but logistical issues with water delivery meant this was not possible. Flow was increased to 40 ML/day on 18 January but was terminated on 20 January because of ongoing logistic difficulties with flow delivery. Flow was manipulated to save water and dilute cells when possible. In response to development of cyanobacterial numbers again in early February the flow was increased to 20 ML/day and then maintained at 40 ML/day from 15 to 29 February to trial the flow strategy over a very warm period. Controlled release of water from Hope Valley was stopped when large summer rain events washed out the remaining cyanobacterial populations from the lake. A timeline of flow releases and cyanobacterial abundance summarising this period is presented in Figure 36.

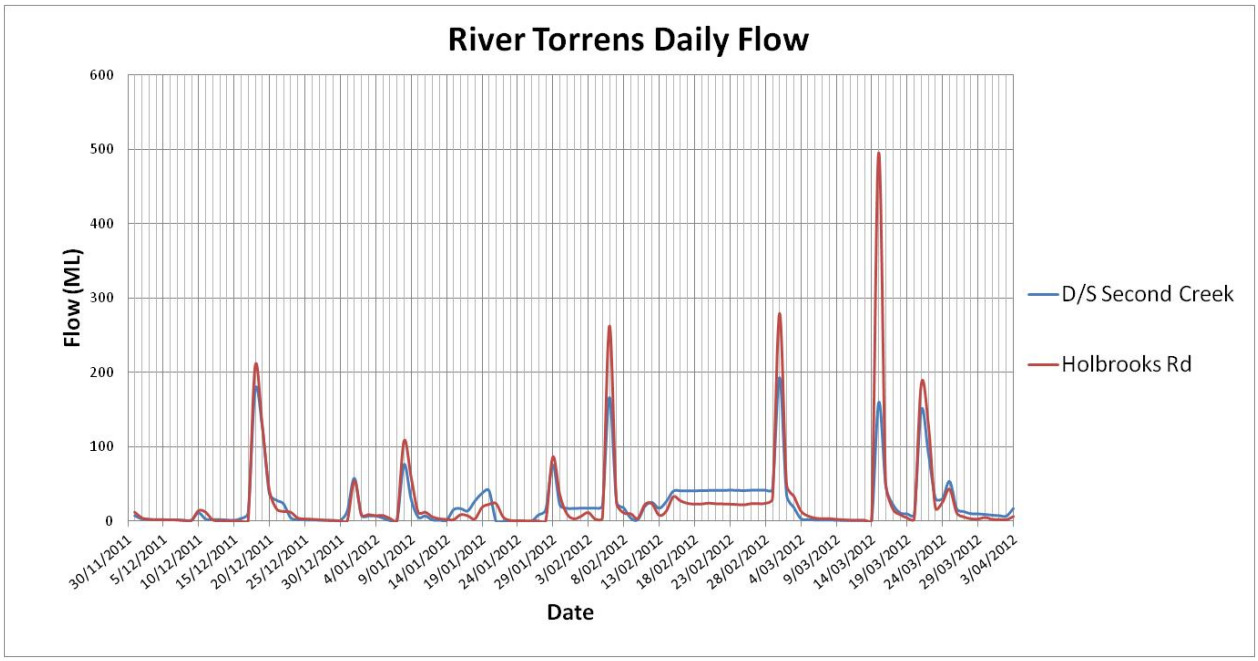

Figure 35. River Torrens Flow measured upstream of the Torrens Lake (D/S Second Creek) and downstream of the Lake (Holbrooks Rd).

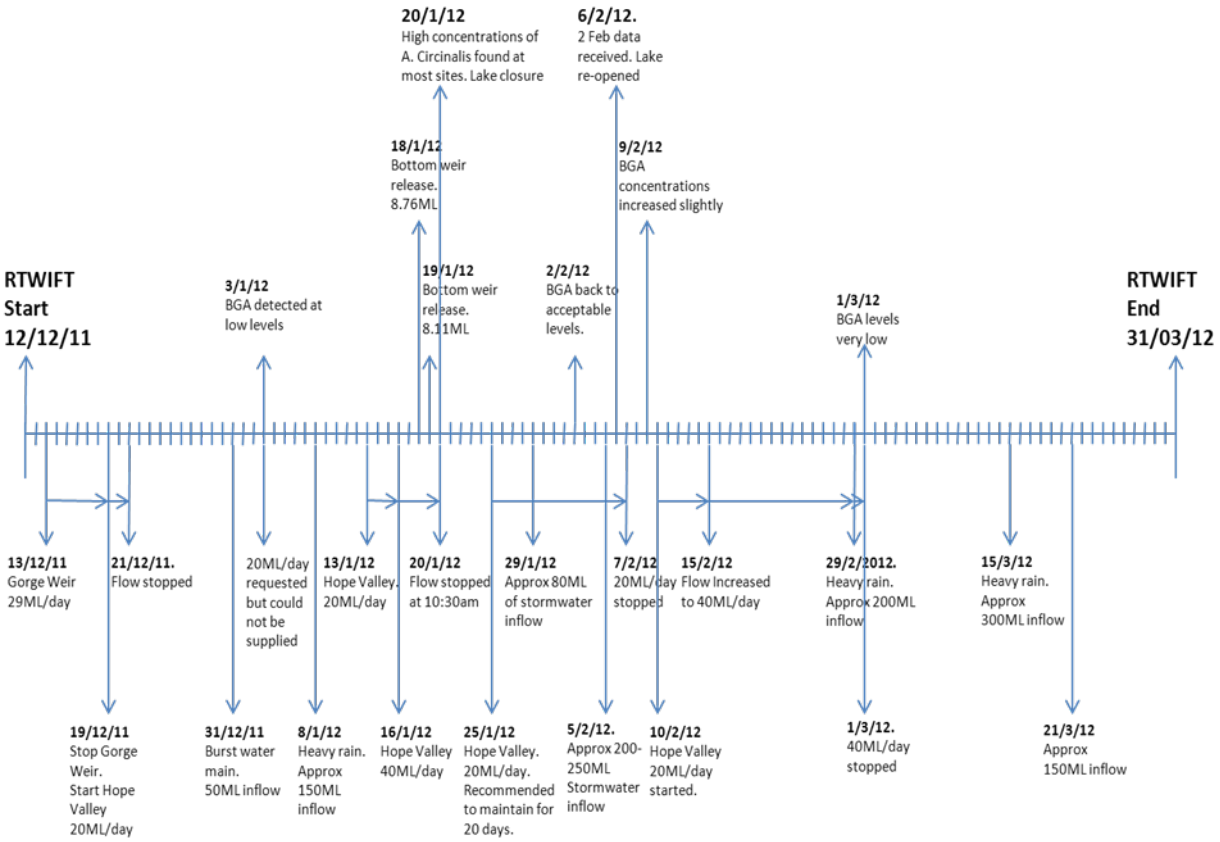

Figure 36. Time line of flow manipulation and cyanobacterial abundance.

3.3.3. Phytoplankton

Phytoplankton cell counts were determined twice weekly with a particular focus on the cyanobacteria. The total cyanobacteria cell counts (Figure 37) were dominated by *Anabaena circinalis* (Figure 38), which, remained at low concentrations through December. Water temperature began to increase in January and was accompanied by rapid growth of *Anabaena circinalis* which peaked at 161,000 cells/mL. Additional water was not available at this time to control the population and it remained at fairly high concentrations until a stormwater inflow of approximately 80 ML on 29 January followed by a significant stormwater inflow of 200-250 ML on 5 February. The rain event inflow dramatically decreased the *Anabaena* population. Although temperature conditions were similar in February as they were in January the flow releases and rain events tended to dilute the cyanobacterial populations and total abundance was maintained below 20,000 cells/mL (Figure 37; Figure 38).

The cyanobacterial component of the community (Table 12) was dominated by *Anabaena circinalis* (Table 13) with relatively minor contributions from *Microcystis aeruginosa* (Table 14),

Microcystis flos-aquae (Table 15), and *Planktothrix* (Table 16). The distribution of cells in the Lake was biased towards the downstream-weir end. This could be due to a combination of preferential growth in the broad main body of the lake and transport of cells from the upstream end and concentration downstream.

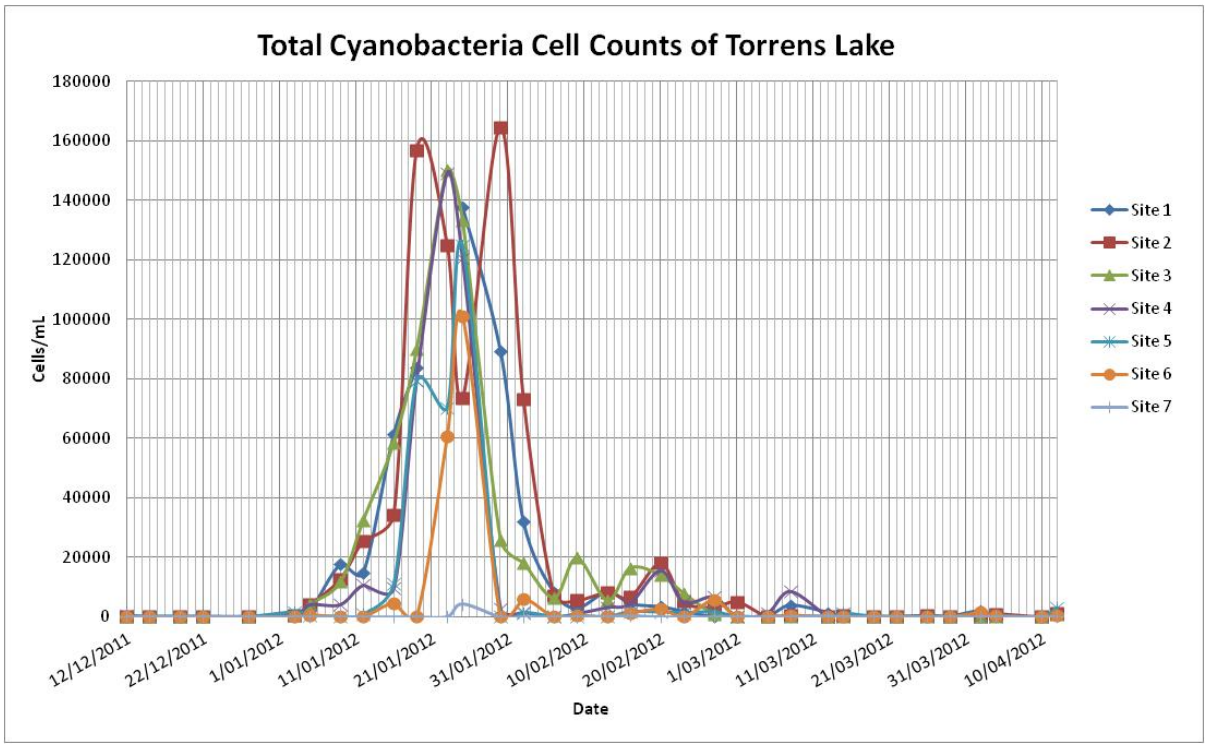

Figure 37. Total cyanobacteria at seven sites in the Torrens Lake, expressed as cells/mL.

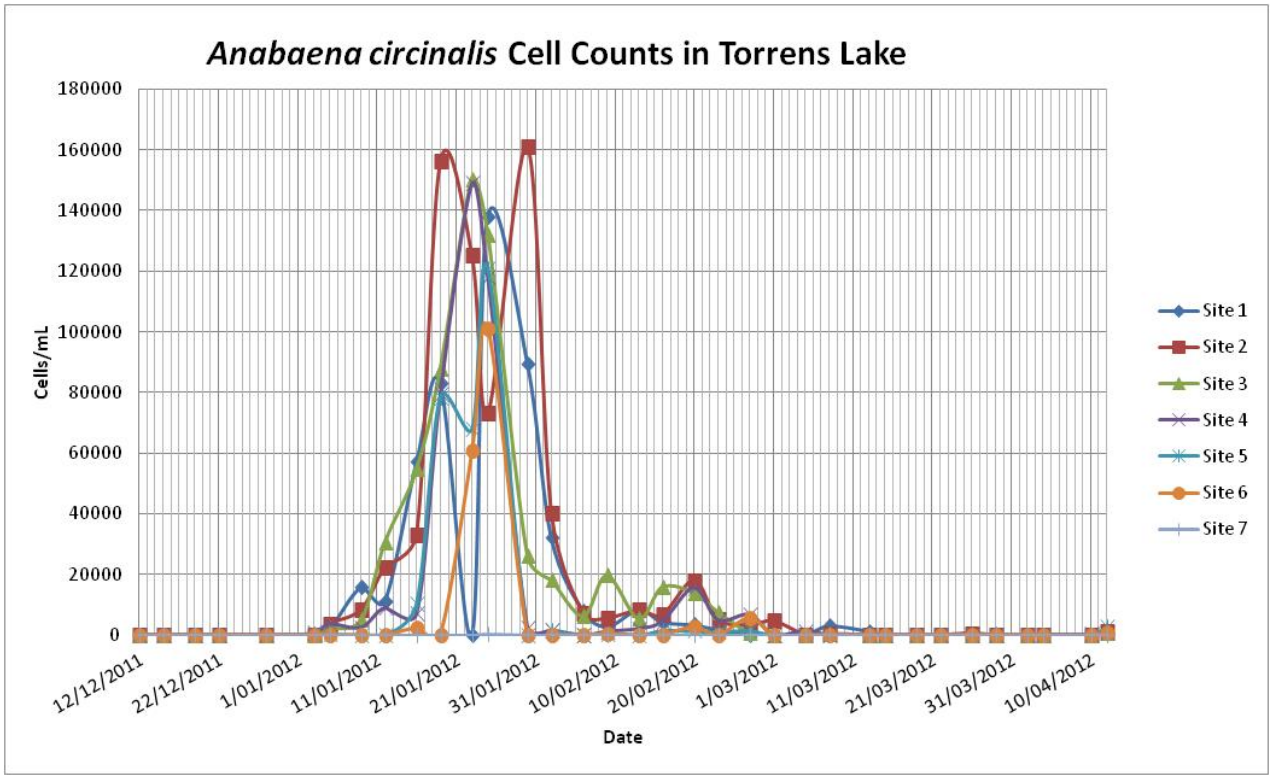

Figure 298. *Anabaena circinalis* in the Torrens Lake.

Table 12. Total cyanobacteria cell counts (cells/mL).

Sample Date	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Average
12/12/2011-28/12/2011	ND	ND	ND	Low	ND	ND	ND	ND
3/1/2012	235	680	590	560	1560	ND	ND	518
5/1/2012	1940	4070	4360	4060	490	670	ND	2227
9/1/2012	17600	12670	11720	4140	ND	ND	ND	6590
12/1/2012	14770	25650	32540	10620	800	ND	ND	12054
16/1/2012	61430	34470	58540	9580	11000	4560	ND	25654
19/1/2012	83900	156900	89920	79400	76540	100	ND	69966
23/1/2012	NS	125000	150200	149200	70320	60600	100	92570
25/1/2012	138000	73540	133460	120620	124820	101000	4300	99391
30/1/2012	89300	164660	26000	2780	ND	ND	ND	40391
2/2/2012	32000	73280	18200	1580	1430	6000	ND	18927
6/2/2012	8090	7120	6420	ND	ND	ND	ND	3090
9/2/2012	2840	5840	19800	1100	250	420	ND	4270
13/2/2012	8360	8160	5550	3170	ND	ND	ND	3606
16/2/2012	4390	6630	16410	4520	1590	1248	380	5024
20/2/2012	3350	18000	14000	15600	1540	2720	ND	7887
23/2/2012	1790	5110	7990	4910	1260	ND	ND	3009
27/2/2012	ND	3190	940	6740	1600	5620	ND	2584
1/3/2012	180	5060	ND	60	ND	ND	ND	757
5/3/2012	130	ND	ND	1350	ND	ND	ND	211
8/3/2012	3860	330	240	8560	ND	580	ND	1939
13/3/2012	1380	ND	90	ND	ND	ND	280	250
15/3/2012	ND	690	ND	ND	1140	ND	ND	261
19/3/2012	ND	ND	260	ND	ND	ND	ND	37
22/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
26/3/2012	ND	530	ND	ND	ND	ND	ND	76
29/3/2012	208	ND	ND	ND	ND	ND	ND	30
2/4/2012	2010	180	ND	ND	ND	1500	ND	527
4/4/2012	ND	990	ND	ND	370	350	ND	244
10/4/2012	ND	ND	ND	170	ND	ND	ND	24
12/4/2012	610	1100	730	880	3100	620	ND	1006

Table 13. *Anabaena circinalis* cell counts (cells/mL).

Sample Date	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Average
12/12/2011-28/12/2011	ND	ND	ND	ND	ND	ND	ND	ND
3/1/2012	235	ND	ND	560	360	ND	ND	165
5/1/2012	1940	3720	2310	3680	490	ND	ND	1734
9/1/2012	15800	8330	5170	2430	ND	ND	ND	4533
12/1/2012	11200	22200	30600	8910	ND	ND	ND	10415
16/1/2012	57100	33100	54800	6960	10400	2170	ND	23504
19/1/2012	82900	156000	87900	79400	78400	ND	ND	69229
23/1/2012	NS	125000	150200	149000	68100	60600	ND	92150
25/1/2012	138000	73100	132000	118000	121000	101000	220	97617
30/1/2012	89300	161000	26000	2410	ND	ND	ND	39816
2/2/2012	32000	40200	18200	1580	1430	ND	ND	13344
6/2/2012	8090	7120	6420	ND	ND	ND	ND	3090
9/2/2012	2840	5840	19800	1100	250	420	ND	4270
13/2/2012	8360	8160	5550	2110	ND	ND	ND	3454
16/2/2012	4390	6630	16000	4520	1590	ND	ND	4733
20/2/2012	3350	18000	14000	15600	1430	2720	ND	7871
23/2/2012	1790	5110	7710	4910	12606	ND	ND	2969
27/2/2012	ND	3190	940	6740	1600	5620	ND	2584
1/3/2012	180	4620	ND	60	ND	ND	ND	694
5/3/2012	130	ND	ND	1350	ND	ND	ND	211
8/3/2012	3070	330	60	ND	ND	ND	ND	494
13/3/2012	1340	ND	90	ND	ND	ND	ND	204
15/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
19/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
22/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
26/3/2012	ND	530	ND	ND	ND	ND	ND	76
29/3/2012	208	ND	ND	ND	ND	ND	ND	30
2/4/2012	ND	ND	ND	ND	ND	ND	ND	ND
4/4/2012	ND	60	ND	ND	80	ND	ND	20
10/4/2012	ND	ND	ND	170	ND	ND	ND	24
12/4/2012	280	1100	610	800	2710	620	ND	874

Table 14. *Microcystis aeruginosa* cell counts (cells/mL).

Sample Date	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Average
12/12/2011-28/12/2011	ND	ND	ND	ND	ND	ND	ND	ND
3/1/2012	ND	ND	ND	ND	ND	ND	ND	ND
5/1/2012	ND	ND	220	ND	ND	ND	ND	31
9/1/2012	ND	260	4200	ND	ND	ND	ND	637
12/1/2012	ND	ND	ND	1710	ND	ND	ND	244
16/1/2012	3120	ND	140	ND	ND	810	ND	581
19/1/2012	ND	ND	2020	ND	1140	ND	ND	451
23/1/2012	NS	ND	ND	ND	1840	ND	ND	307
25/1/2012	ND	ND	1460	2530	3180	ND	2920	1441
30/1/2012	ND	3660	ND	ND	ND	ND	ND	523
2/2/2012	ND	32000	ND	ND	ND	6000	ND	5429
6/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
9/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
13/2/2012	ND	ND	ND	1060	ND	ND	ND	151
16/2/2012	ND	ND	ND	ND	ND	278	ND	40
20/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
23/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
27/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
1/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
5/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
8/3/2012	230	ND	ND	8500	ND	ND	ND	1247
13/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
15/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
19/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
22/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
26/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
29/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
2/4/2012	ND	ND	ND	ND	ND	ND	ND	ND
4/4/2012	ND	ND	ND	ND	ND	ND	ND	ND
10/4/2012	ND	ND	ND	ND	ND	ND	ND	ND
12/4/2012	ND	ND	ND	ND	ND	ND	ND	ND

Table 15. *Microcystis flos-aquae* cell counts (cells/mL).

Sample Date	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Average
12/12/2011-28/12/2011	ND	ND	ND	low	ND	ND	ND	ND
3/1/2012	ND	680	590	ND	1200	ND	ND	353
5/1/2012	ND	350	1830	380	ND	670	ND	461
9/1/2012	1800	4080	2350	1710	ND	ND	ND	1420
12/1/2012	3570	3450	1940	ND	ND	ND	ND	1280
16/1/2012	1210	1370	3600	2620	600	1580	ND	1569
19/1/2012	770	900	ND	ND	ND	100	ND	253
23/1/2012	NS	ND	ND	200	ND	ND	ND	33
25/1/2012	0	440	0	90	640	0	0	167
30/1/2012	ND	ND	ND	ND	ND	ND	ND	ND
2/2/2012	ND	1080	ND	ND	ND	ND	ND	154
6/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
9/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
13/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
16/2/2012	ND	ND	410	ND	ND	970	380	251
20/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
23/2/2012	ND	ND	280	ND	ND	ND	ND	40
27/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
1/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
5/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
8/3/2012	ND	ND	ND	ND	ND	580	ND	83
13/3/2012	40	ND	ND	ND	ND	ND	280	46
15/3/2012	ND	390	ND	ND	1140	ND	ND	219
19/3/2012	ND	ND	260	ND	ND	ND	ND	37
22/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
26/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
29/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
2/4/2012	2010	180	ND	ND	ND	1500	ND	527
4/4/2012	ND	930	ND	ND	290	350	ND	224
10/4/2012	ND	ND	ND	ND	ND	ND	ND	ND
12/4/2012	330	ND	120	80	390	ND	ND	131

Table 16. *Planktothrix* cell Counts (cells/mL).

Sample Date	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Average
12/12/2011-9/1/2012	ND	ND	ND	ND	ND	ND	ND	ND
12/1/2012	ND	ND	ND	ND	800	ND	ND	114
23/1/2012	NS	ND	ND	ND	380	ND	100	80
25/1/2012	ND	ND	ND	ND	ND	ND	1160	166
30/1/2012	ND	ND	ND	360	ND	ND	ND	51
2/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
6/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
9/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
13/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
16/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
20/2/2012	ND	ND	ND	ND	110	ND	ND	16
23/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
27/2/2012	ND	ND	ND	ND	ND	ND	ND	ND
1/3/2012	ND	440	ND	ND	ND	ND	ND	63
5/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
8/3/2012	560	ND	ND	ND	ND	ND	ND	80
13/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
15/3/2012	ND	300	ND	ND	ND	ND	ND	43
19/3/2012	ND	ND	260	ND	ND	ND	ND	37
22/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
26/3/2012	ND	530	ND	ND	ND	ND	ND	76
29/3/2012	ND	ND	ND	ND	ND	ND	ND	ND
2/4/2012	ND	ND	ND	ND	ND	ND	ND	ND
4/4/2012	ND	ND	ND	ND	ND	ND	ND	ND
10/4/2012	ND	ND	ND	ND	ND	ND	ND	ND
12/4/2012	ND	ND	ND	ND	ND	ND	ND	ND

3.3.4. Evaluation of the flow trial against predicted growth rate and dilution

The predictive model of cyanobacterial growth and dilution, described in Chapter 2, was applied to conditions that occurred in the Torrens Lake in January and February 2012. Two growth rates were simulated with three flow scenarios, including no dilution, 10% dilution and the actual flow rate. Cells were initially observed at a concentration of 235 cells/mL and increased exponentially over a three weeks period. The growth and dilution model scenarios that best describe this population increase are a growth rate of 0.3 day^{-1} and a 10% dilution or the actual flow (Figure).

The same modelling approach was taken to predict growth commencing 19 January with a starting cell concentration of 82,900 cells/mL. The trajectory of growth in the model predictions was consistently increasing with a minor perturbation in the population size associated with the large flow (Figure 39). During this period the growth steadily declined until the large inflow event on 5

February, which significantly decreased the population size. It is possible that there are other factors influencing the population size other than homogeneous dilution and a constant growth rate.

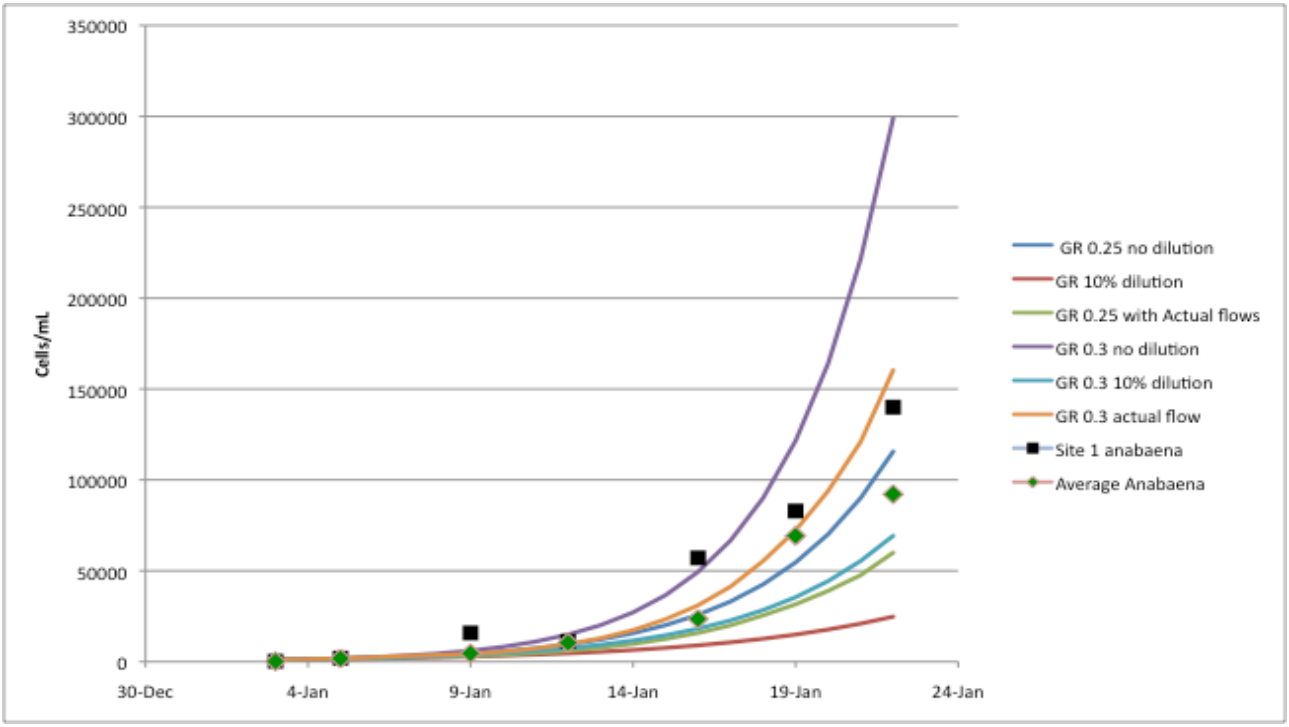

Figure 39. *Anabaena population* size measured at site one and the average of all seven sites, and the predicted population size with a growth rate of either 0.25 or 0.3 and either no dilution, 10% dilution or the actual flows recorded at Holbrooks Road.

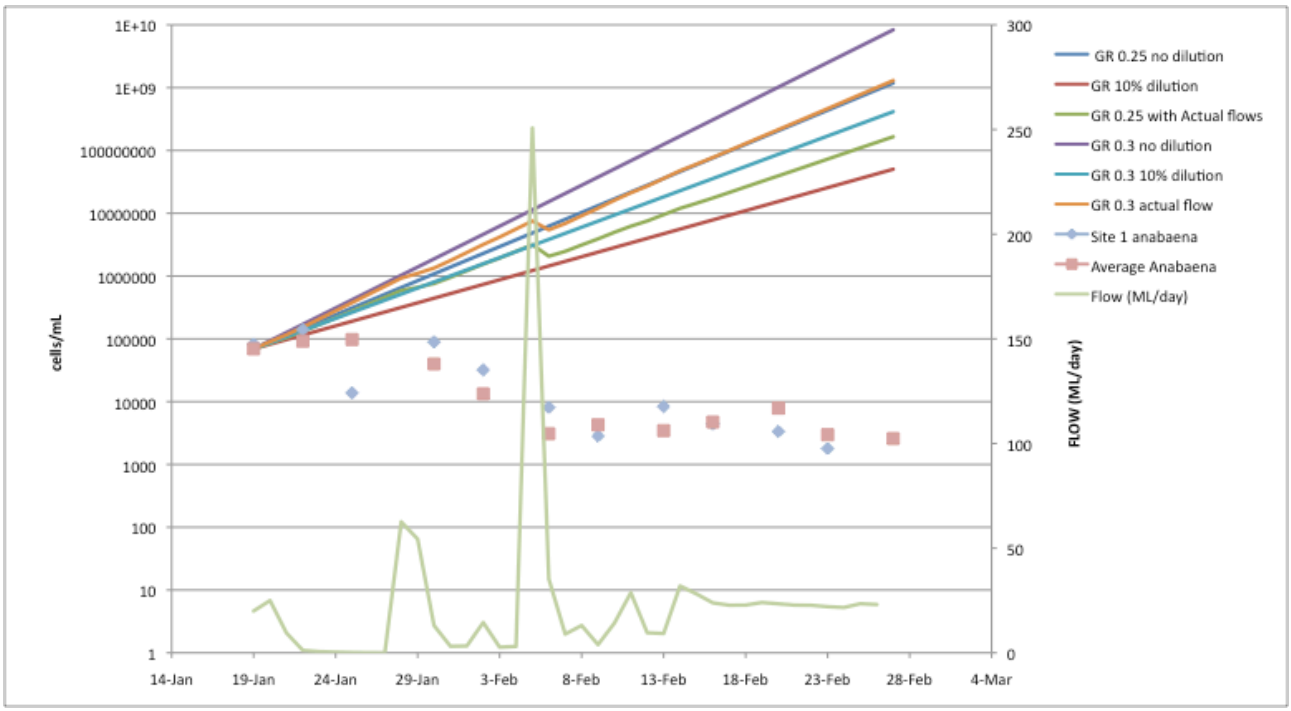

Figure 40. *Anabaena population* predicted and actual for the period 19 January 2012 – 26 February 2012. *Anabaena* size is presented for site one and the average of all seven sites, and the predicted population size with a growth rate of either 0.25 or 0.3 and either no dilution, 10% dilution or the actual flows recorded at Holbrooks Road. Flow at Holbrooks Road is also shown. Note the y-axis is log scale.

3.3.5. *Efficiency of diluting flows*

A concern with releasing water for dilution is that due to density differences between the inflowing water and the ambient lake water there may not be complete mixing and poor dilution. If the water was colder and denser than the lake water then it would travel as an underflow but if it was equivalent density to some depth in the water column it would intrude into the lake at that depth. The ELCOM modelling work by David Lewis presented in Chapter 2 suggests that mixing occurs as the riverine intrusion enters the lake and a distinct intrusion is not evident within the main body of the lake. The thermistor plots also don't show indications of a persistent underflow or overflow as the lake completely mixes every night.

Water velocity measurements were taken with an acoustic Doppler velocity meter at sites within the lake to see whether the riverine intrusion could be detected and determine at what depth it was intruding. The instrument was deployed from a moored boat and measurements taken on the upstream side of the boat during a time when a planned flow release was occurring. The negative velocity in the plots is a flow in the downstream direction.

Flow at Hackney Road Bridge was approximately 5.5 cm/s near the surface but decreased with depth (Figure 441). The shallowness at this site meant water was not remaining here but was flowing downstream. The flow appeared to have inserted as an intrusion by the time it had reached the Zoo site. This is evident as the highest velocities were found mid water column but the surface and bottom water velocity are low. At Frome Road Bridge the riverine instrusion is still evident at approximately 1.5 m depth. Within the main body of the lake (Sites 3, 2 and 1) there is a fairly high surface current flowing downstream but the deeper currents at each site vary. At Elder Park (site 3) the bulk of the water is moving downstream but a return current is evident at Morphett Street Bridge (Site 2). At the Torrens Weir (Site 1) the flow is dominated by the surface flow, which is not unexpected given the weir acts as a barrier, halting flow and sheltering from wind. The surface flow is evident as water was overtopping the weir.

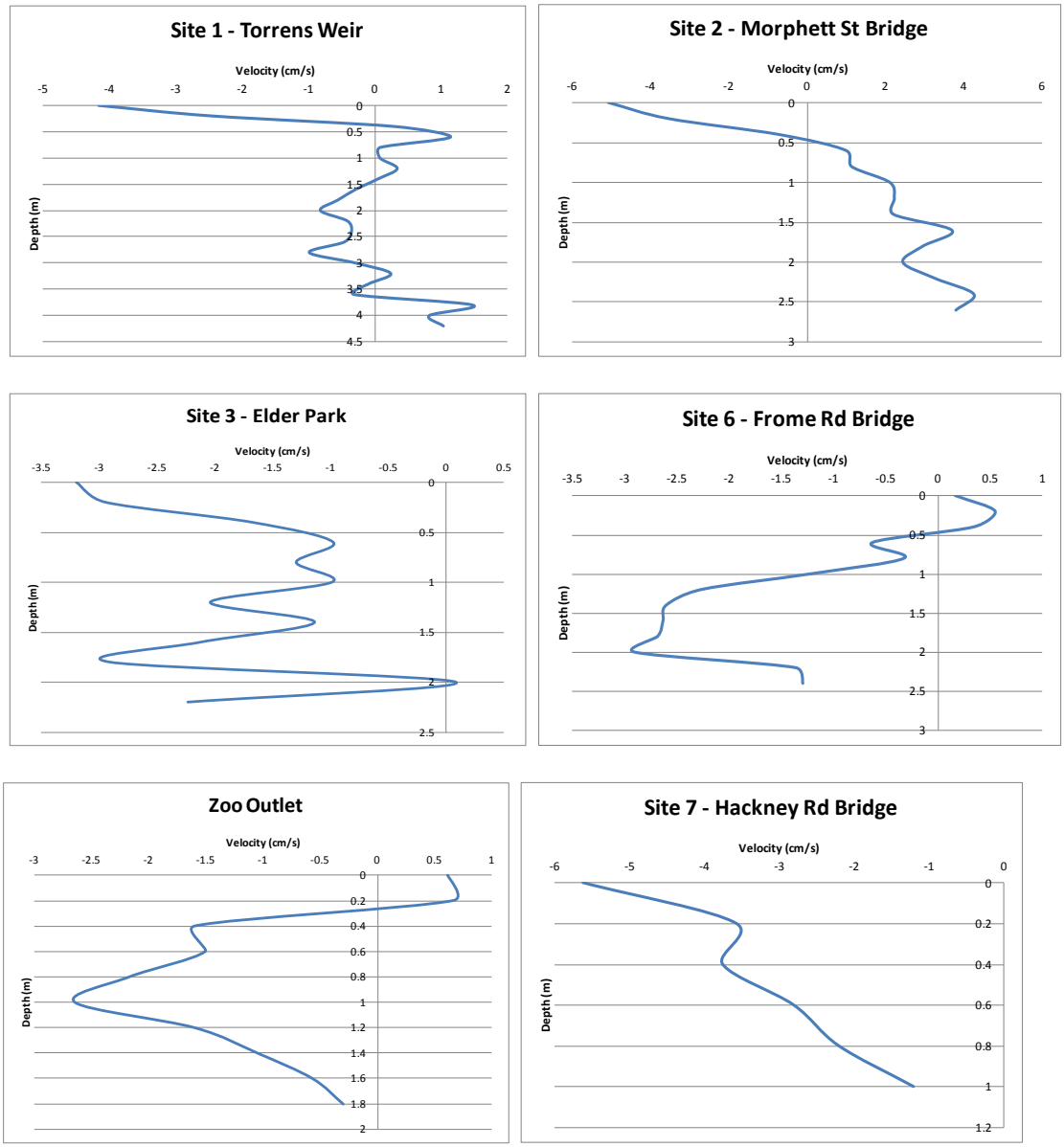

Figure 41. Water velocity measured at six sites in the Torrens Lake. Negative velocity indicates flow in the downstream direction.

3.3.6. Frequency of flushing events

The flow strategy for managing cyanobacteria in the Torrens Lake can dilute the population but at the flow volumes available there is still a reliance on periodic natural high flow from rain events. To investigate how often adequate diluting rain events occur the hydrograph was analysed. The historical flow at Holbrooks Road gauging station was analysed, as although it receives inputs downstream of the weir it was considered most representative of flow through the Torrens Lake. Flow magnitude is generally higher in winter and spring than summer and autumn (Figure 42). Flow is less than 10 ML/day for about half of the year (in the bands 0-5 ML/day and 5-10 ML/day; Table 17).

The return period between flows of different magnitude is of interest as the natural runoff is required to reset the system and provide a major dilution event to control the cyanobacterial population. The entire available flow record (1978 – 2011) was analysed to calculate the annual average return interval for different flow magnitude and the maximum return interval over the entire period (Table 18). Summer flows (Dec, Jan, Feb) in the order of 80ML per day would be required for a 20% dilution and 100 ML for a 25% dilution. These flows have an average summer return interval of 24.6 days and 27 days for 80ML/day and 100 ML/day, respectively. Modelling presented in chapter 2 suggests that diluting flow of 40 ML/day can control cyanobacterial populations for about 3 weeks before the biomass threshold is exceed. Therefore rain event inflows approximately every 21 days could help reset the system. This is in the order of the return interval of 80-100ML/day flows and so it appears feasible that this would work synergistically with controlled diluting flows.

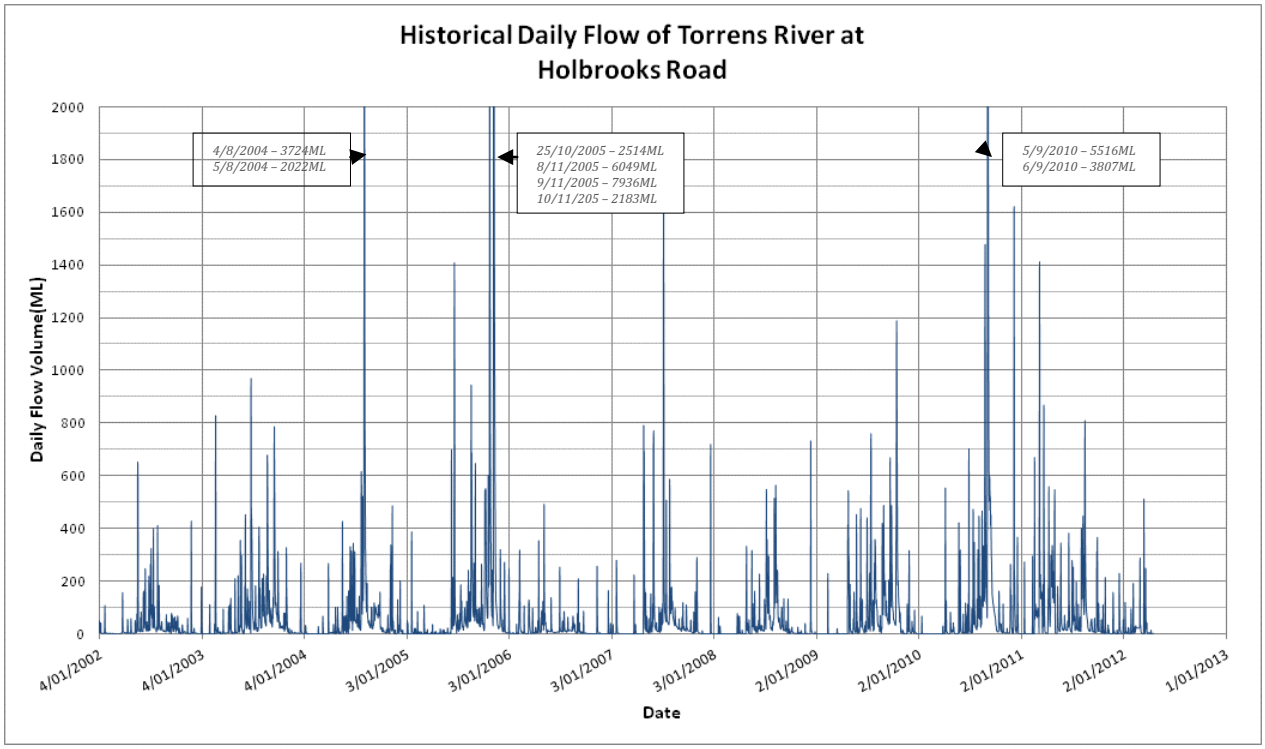

Figure 42. Historical daily flows in the Torrens River at Holbrooks Road. Not all this water passes through the Torrens Lake as there are downstream stormwater inputs, however, it is considered a reasonable indicator of the Torrens Lake hydrology.

Table 17. Number of days in each flow band for years 1991 – 2011 for River Torrens flow measured at Holbrooks Road.

Year	0-5 ML/day	5-10 ML/day	10-20 ML/day	20-40 ML/day	40-60 ML/day	60-80 ML/day	80-100 ML/day	100-200 ML/day	200-400 ML/day	400-800 ML/day	800-1600 ML day	>1600 ML/day
1991	173	19	32	34	11	10	9	20	21	6	3	4
1992	69	12	32	51	34	16	14	52	33	20	16	13
1993	65	36	93	63	24	29	14	27	11	2	1	0
1994	135	46	63	39	15	9	11	22	12	6	7	0
1995	182	27	71	28	12	6	6	17	12	2	1	1
1996	144	19	39	25	10	17	13	42	29	17	8	1
1997	162	51	58	31	14	11	10	15	9	3	0	1
1998	135	34	45	54	29	17	11	20	14	5	1	0
1999	138	41	51	49	25	11	4	21	15	10	0	0
2000	109	47	41	48	28	21	14	26	17	11	4	0
2001	82	30	31	52	35	17	13	54	29	18	4	0
2002	148	53	73	32	19	12	4	13	8	3	0	0
2003	121	34	34	56	32	15	15	29	21	5	3	0
2004	137	33	34	50	24	23	15	26	13	8	1	2
2005	121	29	28	50	32	24	7	33	15	15	6	4
2006	158	87	48	25	14	10	2	14	6	1	0	0
2007	145	37	52	39	22	14	11	20	5	7	3	1
2008	186	24	46	36	22	13	3	22	9	5	0	0
2009	150	29	35	30	23	13	11	35	25	12	2	0
2010	155	46	29	27	10	12	10	33	21	14	4	3
2011	109	34	60	31	18	19	16	42	25	9	1	1

Table 18. Return period of each daily flow magnitude for River Torrens Flow measured at Holbrooks Road determined for flow records from 1978 – 2011.

Flow	10 ML	20 ML	40 ML	60 ML	80 ML	100 ML	200 ML	400 ML	600 ML
Max interval	92	128	128	153	153	153	202	353	530
Average interval	9.5	11.0	12.3	13.9	24.1	55.6	101.8	9.9	34.1
Max summer interval	83	128	128	146	148	149	153	206	276
Average summer interval	9.9	14.5	19.6	22.8	24.6	27.0	40.1	69.7	73.5

3.3.7. *Nutrient concentrations and growth*

Nutrient concentrations give a good indicator of the phytoplankton biomass potential of the waterbody. Phosphorus, in particular, often has a strong correlation with phytoplankton biomass (Vollenweider, 1975; Sherman et al., 2000; Linden et al., 2004). Nutrients were sampled and measured at three sites but only the Weir site is presented in this report. The dissolved nutrients (oxidised nitrogen, filterable reactive phosphorus and ammonium) were all at fairly high concentrations (Table 19) and typically excess to demand. A decrease in concentrations of these nutrient forms was evident on 16 January and 23 January 2012 as the cyanobacterial community grew exponentially and utilised the available nutrient resources. Nitrate and nitrite dropped from 0.23 mg/L to 0.007 mg/L, Ammonia from 0.265 mg/L to 0.005 mg/L and filterable reactive phosphorus from 0.04 to 0.008 mg/L (Table 19). These nutrient concentrations recovered with a rain event inflow in late January.

Table 19. Nutrient concentrations at Site 1, The Weir site in the Torrens Lake.

Date	NOx (mg/L)	Total Phosphorus (mg/L)	TKN (mg/L)	Filterable Reactive P (mg/L)	Ammonia as N – total (mg/L)
12/12/11 9:30	0.3	0.1	1.6	0.038	0.709
19/12/11 9:15	0.527	0.107	1.58	0.01	0.205
28/12/11 9:15	0.167	0.14	1.27	0.06	0.523
3/01/12 9:05	0.27	0.112	1.51	0.035	0.486
9/01/12 10:05	0.23	0.144	1.82	0.04	0.265
16/01/12 9:20	0.007	0.181	2.05	0.019	0.036
23/01/12 9:10	0.009	0.193	2.58	0.008	0.005
30/01/12 9:10	0.056	0.193	2.62	0.015	0.164
6/02/12 9:50	0.343	0.07	0.44	0.03	0.247
13/02/12 9:15	0.188	0.12	1.46	0.041	0.439
20/02/12 9:15	0.372	0.098	1.32	0.028	0.348
27/02/12 9:15	0.353	0.09	1.58	0.02	0.314
5/03/12 9:20	0.213	0.112	1.37	0.028	0.225
13/03/12 9:45	0.212	0.103	1.18	0.023	0.368
19/03/12 9:30	0.345	0.117	1.3	0.028	0.112
26/03/12 10:15	0.273	0.107	1.02	0.033	0.203

The water quality of inputs and outputs from the Torrens Lake is also of interest for management. A transect of sites from Hope Valley Reservoir to the sea was established for regular monitoring of total phosphorus (Figure 43). The phosphorus concentration increased with downstream travel

from The Hope Valley Reservoir Outlet to the gauging station at second creek and site 6. There was a gradual attenuation of phosphorus downstream of the weir except at the Grange Lake which showed the highest total phosphorus concentrations.

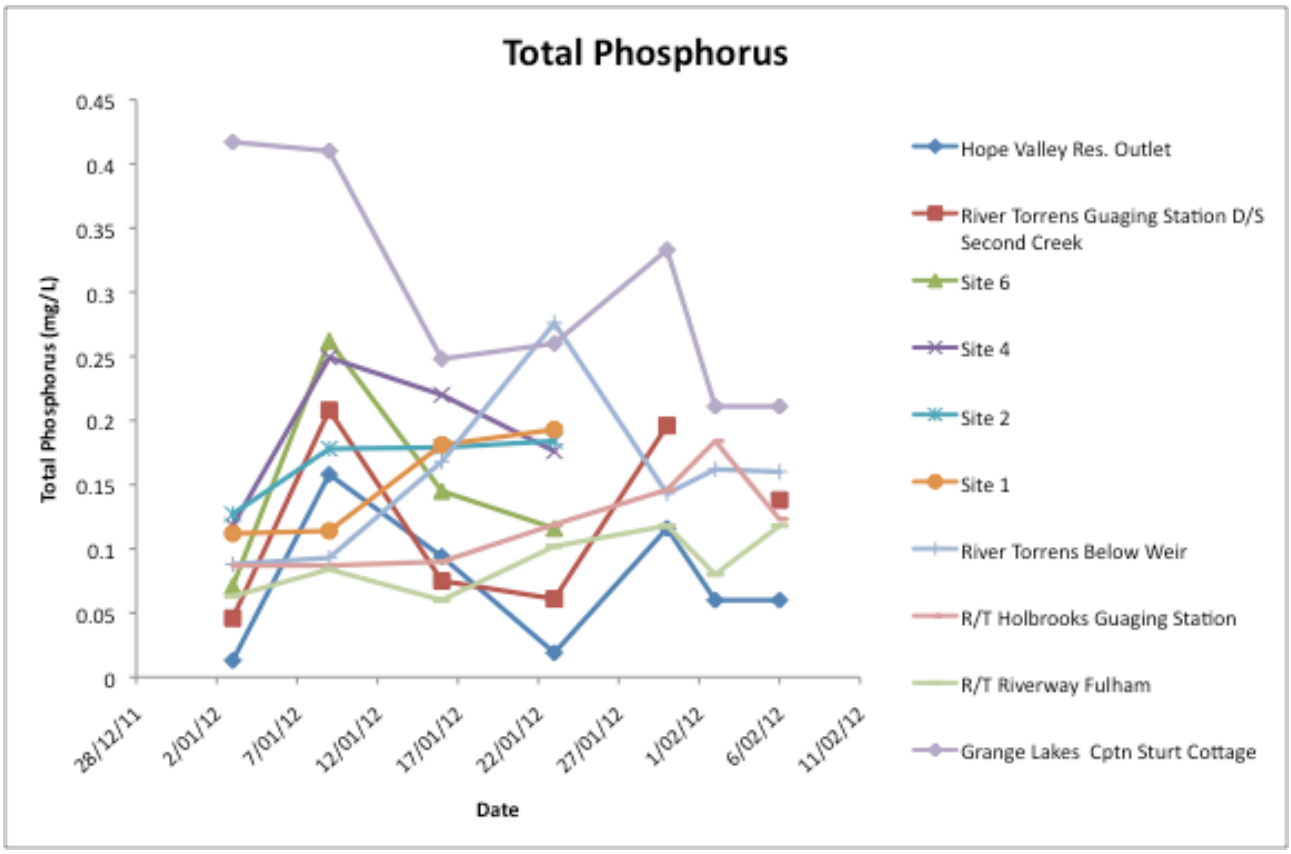

Figure 43. Total phosphorus at sites ranging from the upstream source (Hope Valley Reservoir), in the lake (sites 1, 2, 4, 6) and sites downstream of the weir.

3.4. Discussion

The Torrens Lake amenity flow trial was conducted over a range of meteorological and hydrological conditions. Even though the trial period was of limited duration, it allowed the cyanobacterial growth to be measured under low and high flow conditions. Without diluting flows in early January the population rose to 'bloom' proportions and forced the closure of the lake to recreational users. Similar climatic conditions to those that gave rise to the 'bloom' were experienced in late January but the population growth was controlled.

The explosive growth without dilution in January consumed nutrient resources, which decreased the nutrients to concentrations that are considered limiting. It is unclear whether this controlled the growth rate. The diluting flows in February under similar meteorological and nutrient condition seemed to control the phytoplankton growth. The simple model present in Chapter 2

suggests that growth can be controlled for about three weeks but rain event inflows are required to significantly dilute the population and reset the population size to a level that can be maintained with diluting flow. Natural flows from rain event inflows are a critical part of the strategy for cyanobacterial control by dilution. The dilution flow with programmed releases from Hope Valley Reservoir appear to exert some control on the size of the cyanobacterial population but another season of trials would aid in conclusively determining how successful amenity flows are likely to be in controlling cyanobacterial biomass in the long-term. The dilution flow strategy as an option for cyanobacterial control is contingent upon water availability from Hope Valley Reservoir or Kangaroo Creek Dam.

The simple model of growth and dilution predicted the biomass change well at low cell density (Figure 39) but the population decrease was not well predicted in late January and early February (Figure 40). There are several possible reasons for this:

- Dilution was greater than predicted by the simple model
- Growth was lower than predicted.
- Another unanticipated feature such as cyanophages was contributing to population decline.

Of the possible explanations, greater than predicted dilution or slower growth are considered the most plausible. Greater than expected dilution could occur from an error in the flow measured, or an error in the estimate in lake volume, however, both of these are considered unlikely. A more likely explanation is that the flow over the top of the weir is disproportionately diluting cyanobacteria, which are buoyant and so are at higher concentration near the surface.

The decrease in population size could also be explained by a decrease in the growth rate. Typically phytoplankton growth is limited by light or nutrients. Nutrient concentrations decreased as the population expanded and had reached concentrations that could be considered limiting to the growth rate. Light limitation is also a plausible control on phytoplankton growth in the Torrens Lake, particularly after rain events that tend to wash in a significant amount of black material from the urbanised catchment.

Concerns about the flow acting as a discrete riverine intrusion and short-circuiting the lake were not supported by the thermistor data, or the apparent dilution rates.

3.5. Conclusions

- The simple model of growth and dilution is reasonable predictor of biomass change at low cell density but population decrease is greater than predicted
- Dilution is possibly greater than predicted because high concentrations of cells near surface are washed out with the overflow
- Nutrient or light limitation possibly contributes to a decrease in growth rates; nutrient limitation as the population expands and consumes the available resources during periods of rapid growth, and light limitation following rain events as coloured material is washed into the lake.
- If dilution flows start early enough the control population size can be controlled, however, there is a reliance on rain events to flush the system and dilute the resident cyanobacterial population. The flow return interval analysis suggests that rain events occur frequently enough in summer for this strategy to be effective. However, there are long periods between flows of sufficient magnitude to dilute the population on occasions which may reduce the confidence in rain events to reset the population.

3.6. Recommendations

The use of flows to control cyanobacterial growth shows promise as a technique to control the biomass which is supported by both the modelling of growth and dilution, and the field trial. However, this project was conducted for on a very short period of time and dilution flows were only possible in February. In the absence of any other control strategy for cyanobacteria in the Torrens Lake it is recommended that the amenity flow trial be repeated to conclusively determine the effectiveness of this control strategy.

Should the trial be repeated the diluting flows should commence when cell concentrations approach 1000 cells/mL. If diluting flows commence at higher cell concentrations it is difficult for dilution to adequately dilute the exponentially expanding population. This work is time consuming and demands a high level of logistical support for sampling, reporting, analysis, flow releases and pumping. The coordination of this work enabled the delivery of flow in response to cell counts on samples collected two days prior, which is considered appropriate considering the rate of growth.

The nutrient concentrations of water entering the lake are still high and so can support high biomass. Nutrient control is still the best long-term solution to the problems associated with eutrophication it is recommended that there is a continuation of the remedial works for nutrient control in both the urbanised and peri-urban catchments

3.7. References

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Chapter Four: Hydrodynamic Modelling of amenity flows

Hydrodynamics modelling was undertaken with ELCOM, a three dimensional hydrodynamic model to determine what dilution could be expected in the Torrens Lake under various inflow volumes. A tracer was added to the inflow in the model and the concentration of the tracer was tracked through the Lake.

4.1. Modelling Input Data

- Due to the unavailability of Torrens Lake meteorological data alternative sources of data were used.
- Net and Shortwave solar radiation data from an SA Water monitoring station located at Myponga was used.
- All other meteorological data was used was sourced from the Bureau of Meteorology weather station at Kent Town.
- A scaling of Kent Town wind velocity was needed to correct poor correlation between modelled and observed water temperatures. This was achieved to by using 2010/2011 Torrens wind data (which was available) and comparing it to Kent Town data from the same period. The result was that Kent Town was 'windier' – so a downscaling was done on the data to apply it to the Torrens Lake.
- The gauged flow downstream of Second Creek was used as the inflow to the lake.
- The water temperature measured at Hackney Rd was assumed to be the inflow temperature; data was only available from the sensor at the bottom.
- Outflows from the lake were estimated based on inflows and calculated evaporation rate.

The previous hydrodynamic model for the Torrens Lake was modified with a new boundary condition at the surface adjacent to the weir to simulate weir overtopping. The previous model only simulated outflow from under the weir gates.

4.2. Validation of Torrens Lake model output during trial period

Modelled temperature was compared to observed temperature data for the time period 12 December 2011 to 05 March 2012. The model results follow the diurnal/weather/seasonal trends in temperature well (Figure 44). In terms of absolute temperature, the model results were typically 0.5ºC cooler than observed (Figure45), this temperature difference is more pronounced during the month of February.

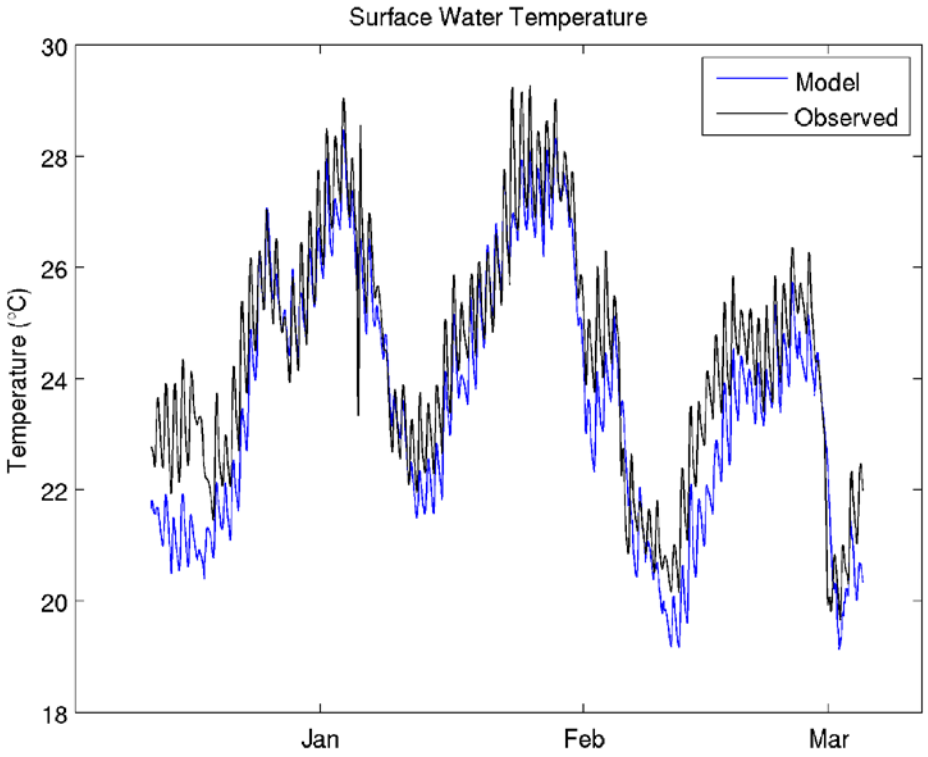

Figure 44. Comparison of modelled and observed surface water temperature at Morphett St Bridge.

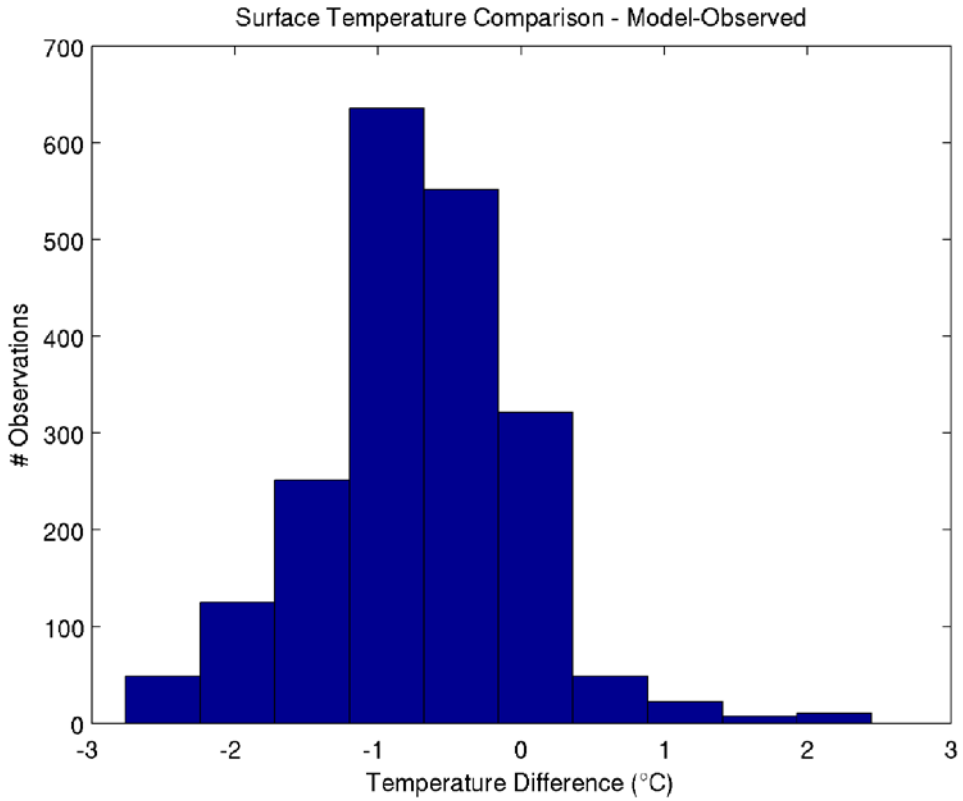

Figure 45. Histogram of temperature difference between modelled and observed surface water temperature at Morphett St Bridge.

4.3. Tracer study of inflow events

To examine the dilution effects of water flowing into the Torrens Lake the addition of conservative tracers during five inflow events was simulated during the period of the trial (Figure 46). The release duration for each tracer was two days. The five flow events were:

- Base Flow (24-Dec-2011)
- Small rain event (31-Dec-2011)
- Large rain event (05-Feb-2012)
- 20ML/day release (31-Jan-2012)
- 40ML/day release (17-Feb-2012)

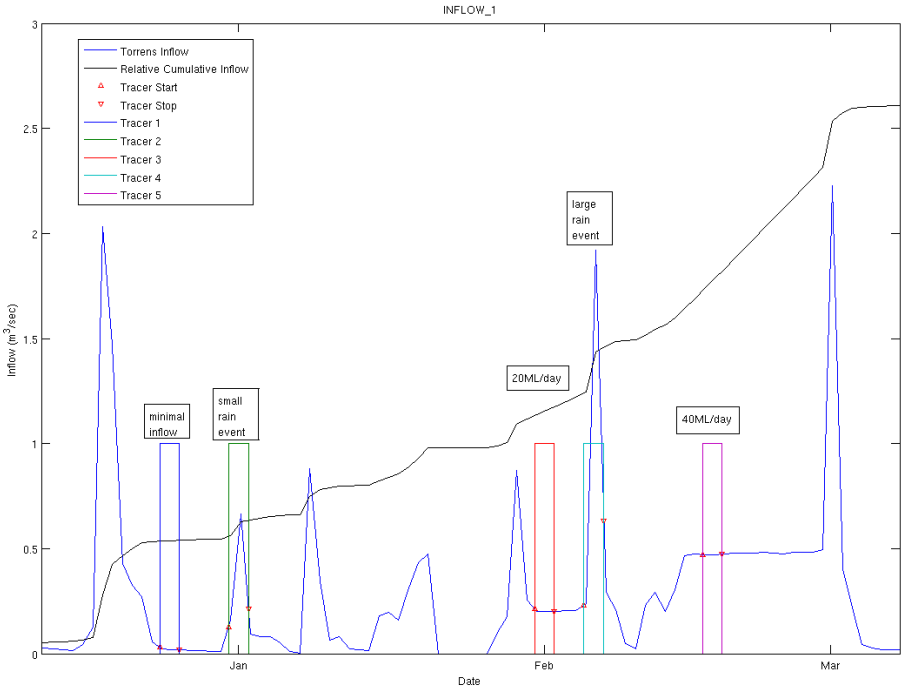

Figure 46. River Torrens hydrograph overlaid with simulated tracer releases.

The results of the model output for Elder Park and the Torrens Weir are shown in Figure 47 to Figure 50. Each plot represents the average concentration in a transect across the river. In many cases the movement of tracer to downstream location was strongly influenced by the flows that occurred after the tracer release. In all cases the water with tracer added behaved as a "plug flow" with no significant intrusions or high density under flows occurring. This means that the water flows through the lake as "parcel" of water from the inflow to the outflow with similar concentrations at the surface and bottom of the lake. Lateral dispersion spreads the leading and trailing edges of the water and is dependent on the time taken for the parcel of water to move through the lake.

4.4. Conclusions

The movement of tracer through the lake is strongly dependant on the flow conditions that follow the initial release. For example the trend in "base flow" tracer closely mirrors the tracer added during the following "small rain event" (Figure 51 v Figure 52). This is because the base flow sits in the upstream reach of the lake until the rain event occurs.

For all simulations the tracer appears homogeneously distributed through the water column when the peak tracer concentration passes Elder Park. This suggests that the inflow will dilute the water column adequately without bias from a riverine intrusion short-circuiting at a discrete depth.

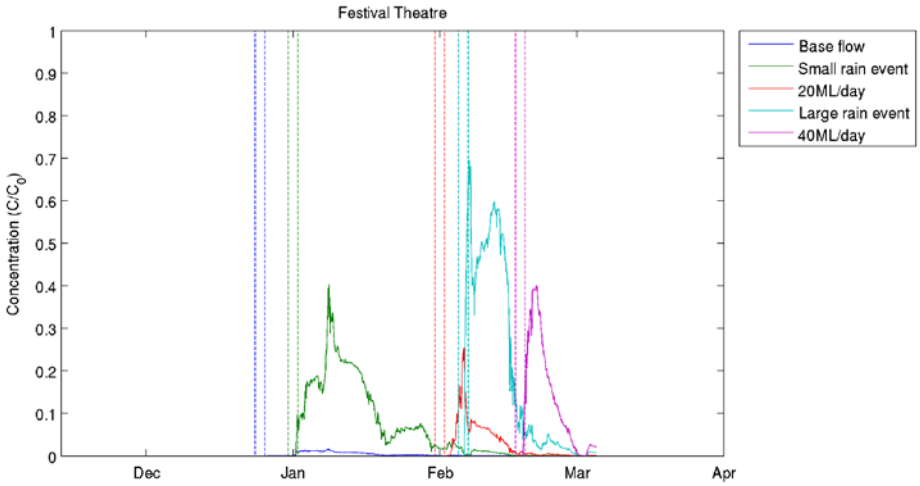

Figure 47. Simulated concentration of tracers at Elder Park relative to the initial concentration.

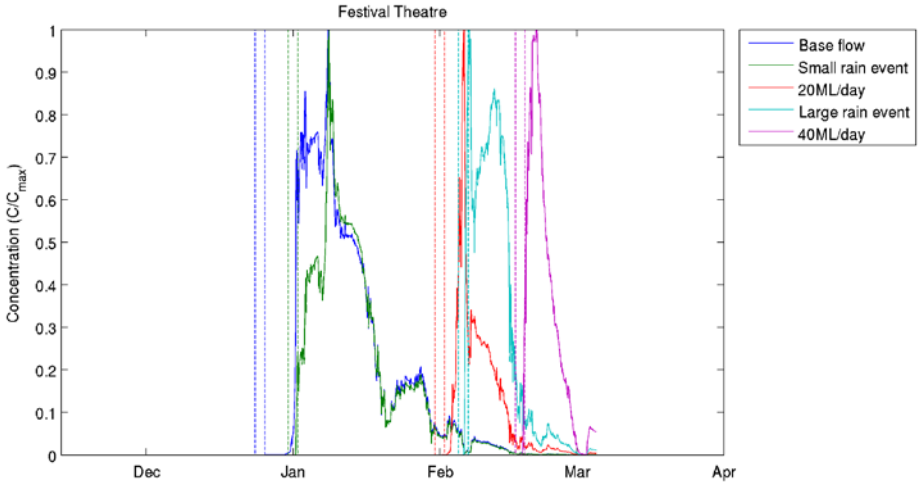

Figure 48. Simulated concentration of tracers at Elder Park relative to the maximum concentration (normalised).

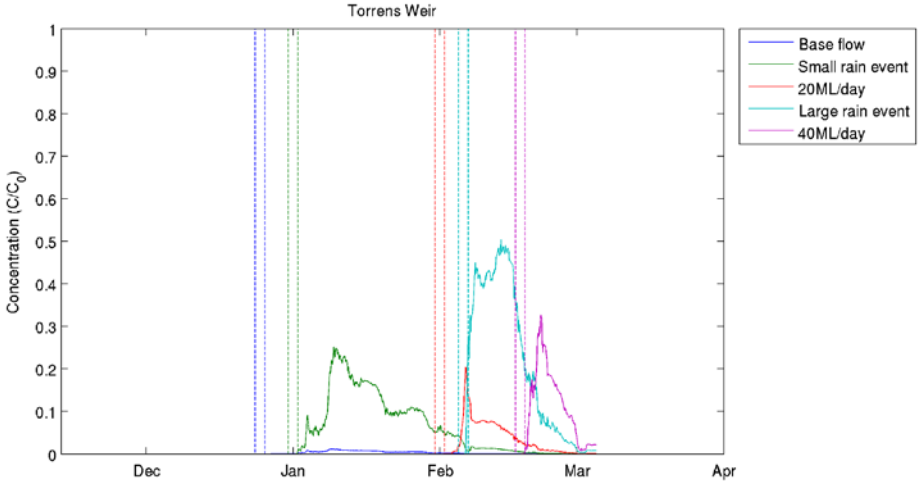

Figure 49. Simulated concentration of tracers at the Torrens Weir relative to the initial concentration.

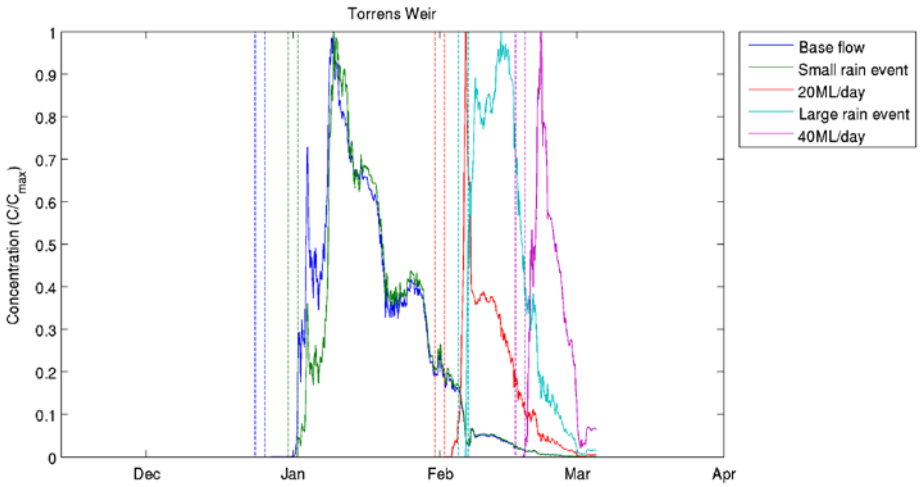

Figure 50. Simulated concentration of tracers at the Torrens Weir relative to the maximum concentration (normalised).

Table 20. Summary of tracer release model experiments presented as travel times.

	Time to first appearance (days)	Time to Increase to 10% C_{max} (days)	Time to Maximum Concentration (C_{max})	Decrease to 10% C_{max} (days)
Elder Park				
Base flow	2.1	8.3	15.5	26.7
Small rain event	1.3	1.7	8.6	19.7
20ML/day release	1.5	3.7	6.1	15.3
Large rain event	0.8	1.3	2.2	12.4
40ML/day release	0.7	1.8	4.4	11.8
Morphett St Bridge				
Base flow	3.1	9	16.5	44
Small rain event	1.6	3.6	9.6	37
20ML/day release	1.8	5.2	6.5	19.2
Large rain event	1.1	1.6	8.9	21.1
40ML/day release	1.1	2.6	5.4	13
Torrens Weir				
Base flow	3.7	12.1	25.4	46.6
Small rain event	1.8	5.8	19.2	43.6
20ML/day release	2.2	6.3	7.8	28.4
Large rain event	1.6	2.4	11.7	25.3
40ML/day release	1.6	3.7	7.3	15.5

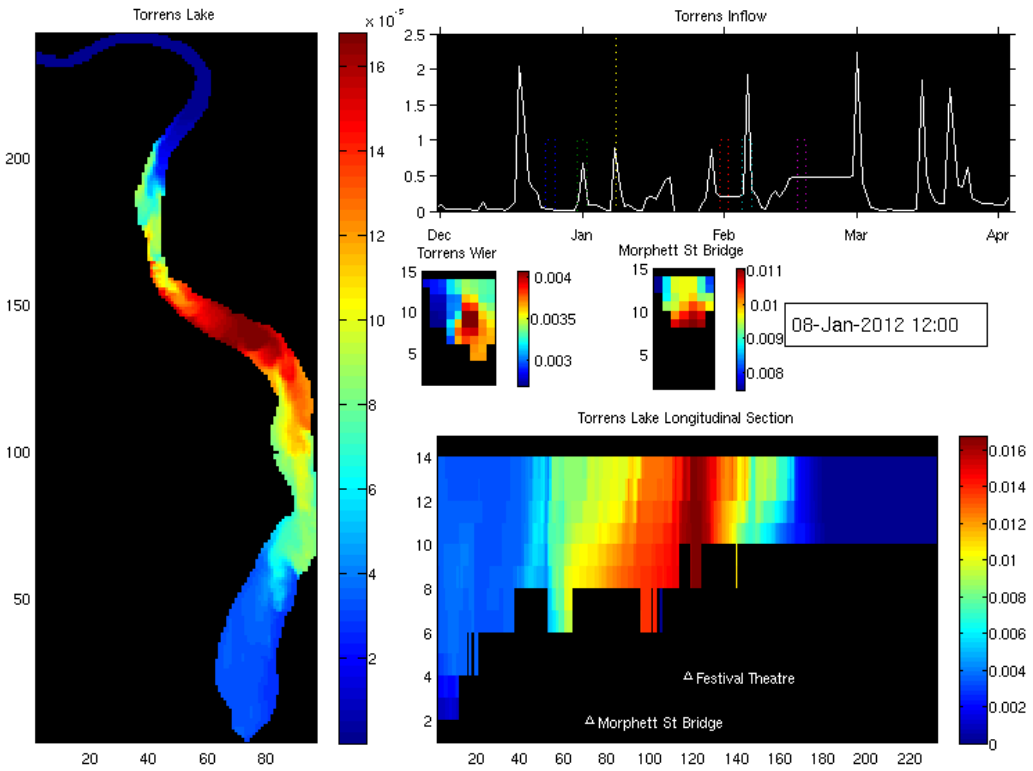

Figure 51. "snapshot" of base flow tracer release when peak concentration is passing Elder Park.

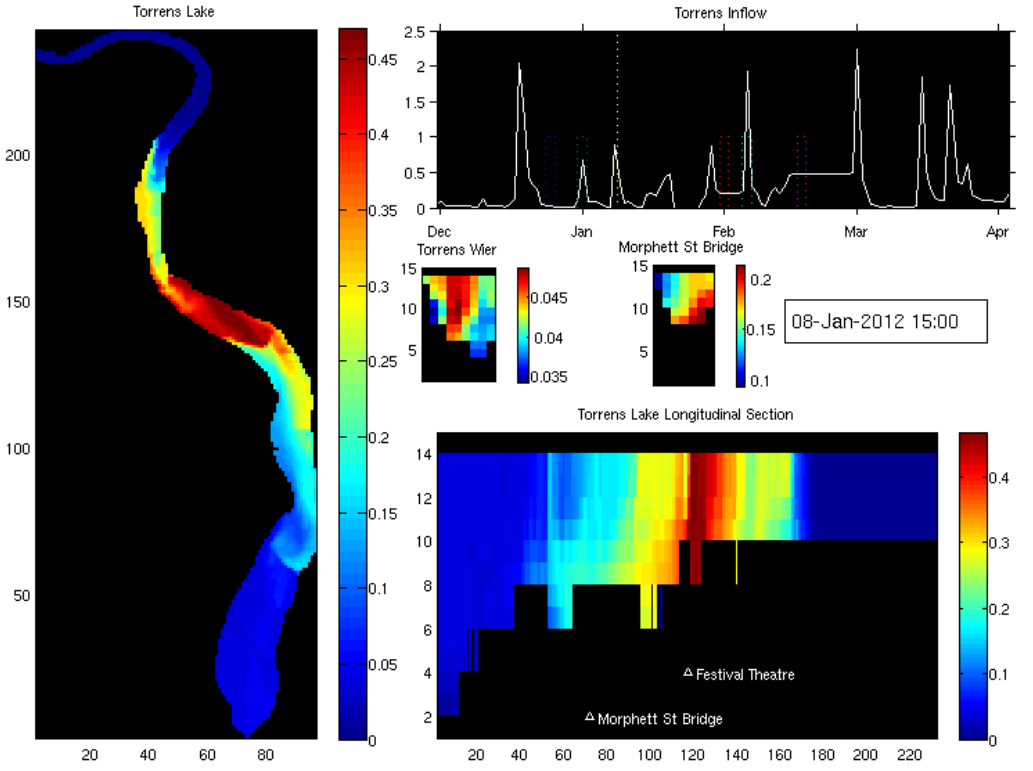

Figure 52. "snapshot" of small rain event tracer release when peak concentration is passing Elder Park.

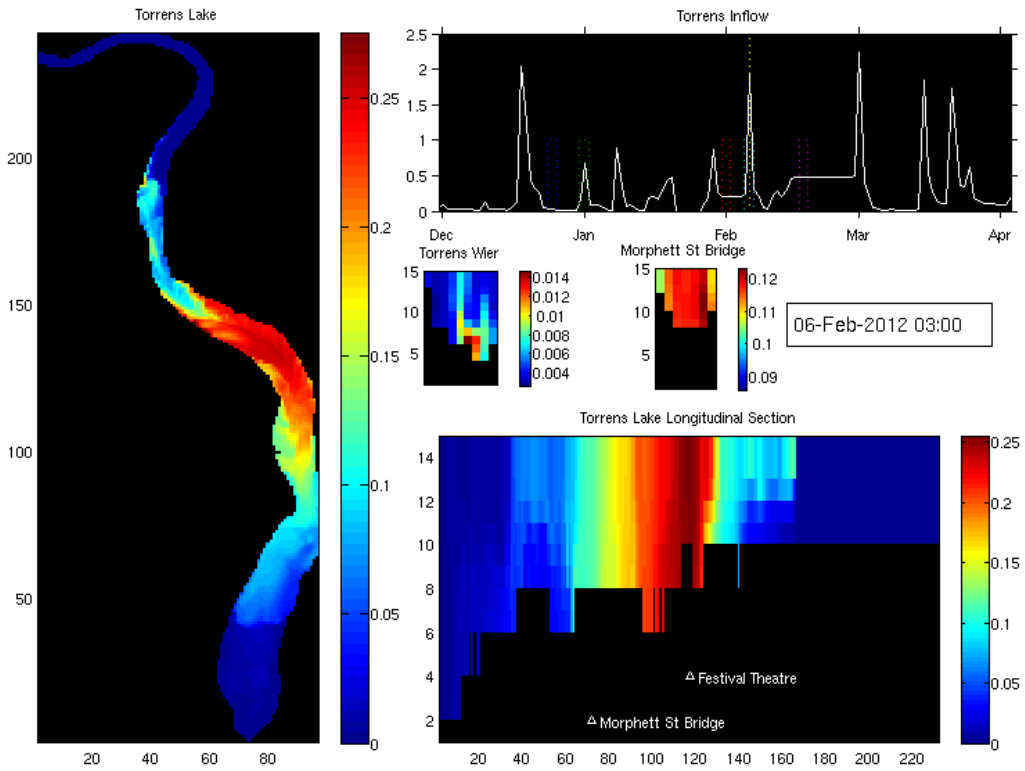

Figure 53. "snapshot" of 20ML/day release tracer when peak concentration is passing Elder Park.

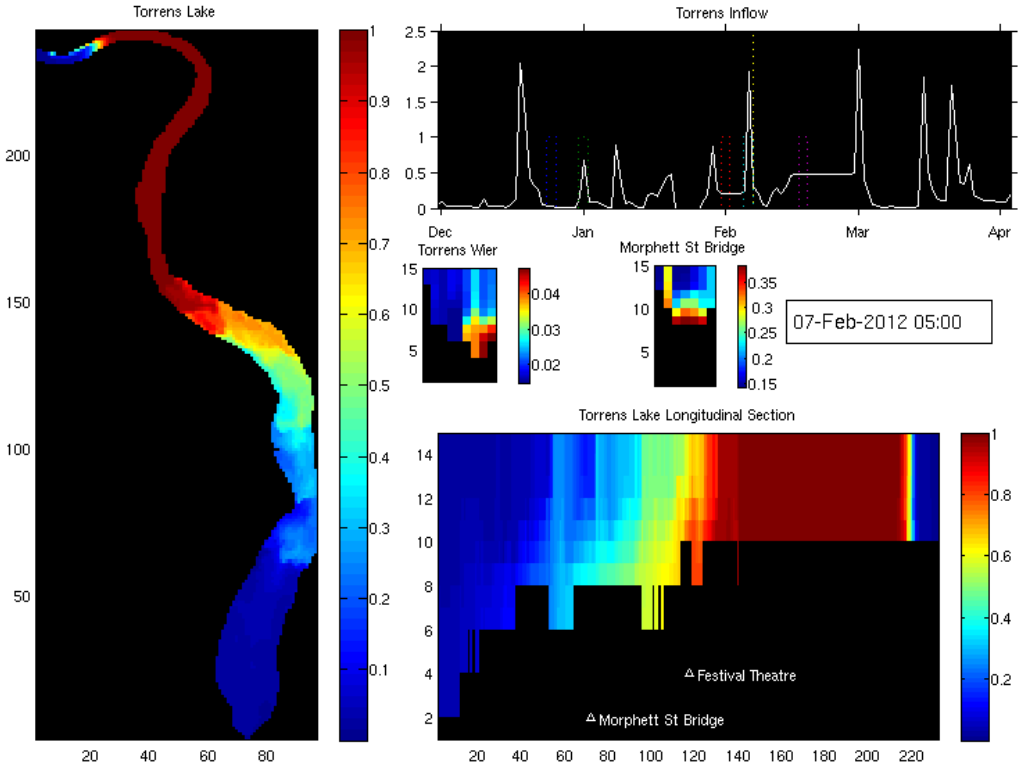

Figure 54. "snapshot" of large rain event tracer release when peak concentration is passing Elder Park.

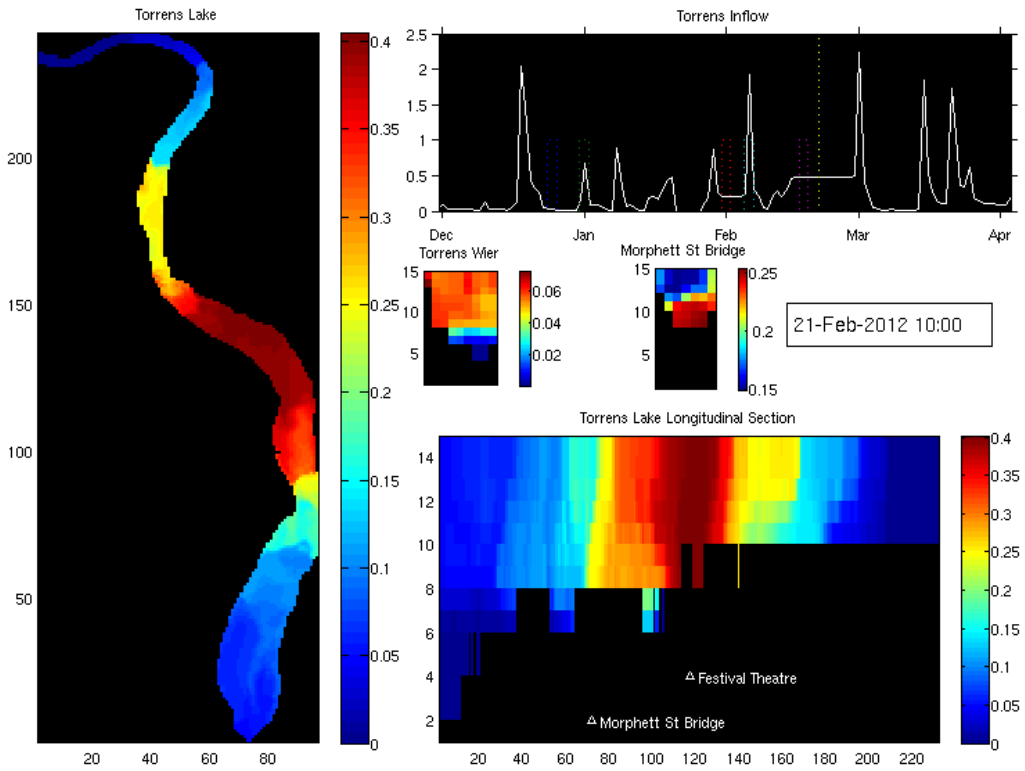

Figure 55 "snapshot" of 40ML/day release when peak concentration is passing Elder Park.

Chapter Five: Flow trial – Fish Monitoring

5.1 Introduction

The fish component of the Torrens Water Quality Improvement Flow Trial monitored the River Torrens river fish population to test the assumptions made regarding fish response to the 2012 trial release of an Environmental Water Provision (EWP) as outlined by Hammer (2011).

The aims of the fish monitoring component was to evaluate whether the flow release could be determined to have any positive or negative impacts on the fish population within the lower Torrens and Breakout Creek. The ID 6 project was developed to provide a baseline survey of fish communities, against which post-flow patterns (to be funded separately following the current project) could be compared and to conduct a survey during flow releases to determine any short term responses or impacts. In response to the flow release it was predicted that:

- Diadromous fishes would be facilitated in moving upstream of the Breakout Creek fish ladder towards the city weir;
- In-channel barriers between Breakout Creek and the city weir (Identified in Schmarr *et al* 2011), may serve as barriers to upstream fish movements resulting in aggregations below barriers.
- Changes in Water Quality resulting from the flows may impact negatively on fish populations where tolerance thresholds were breached.
- Flows may induce spawning and or recruitment responses measurable in fish population structure.
- Freshwater inflows into the relatively poorer water quality environment of the Torrens Lake may result in aggregations of spawning carp and lead to increases in carp abundance, biomass and distribution.

5.2 Baseline Survey

The baseline survey consisted of a five day survey conducted between 12 and 16 December 2011. Ecologists used a combination of small fyke nets (3m wing, 4mm mesh, 3m funnel, 0.6m high), double-winged fyke nets (2 x 5m wing, 4mm mesh, 3m funnels, 0.6m high) and big fyke nets (2 x

10m wing, 12mm mesh, 5 m funnels, 1.2m high) to survey fish populations on a transect between the city weir and Breakout Creek.

Nets were set to target the widest variety of instream microhabitat types, whilst being spaced somewhat homogenously along the reach. Nets were left for approximately 17 hours to harness both dusk and dawn feeding periods. From each net, the first 100 fish of each species were measured (total length), with subsequent fish counted prior to release. Mammals, birds, invertebrates and amphibians that were encountered in the nets were also identified and counted. Vertical water quality profiles were taken at regular points along the reach.

5.2.1 Findings of Baseline Survey

The Torrens fish fauna is composed of endemic, River Murray translocations and exotic species. Table 21 provides a summary list of total species catches and lengths. Observed spawning status, disease and other ecologically relevant information are also noted.

Table 21. Baseline Survey Summary Data for total catch, length range and observational notes.

Common Name	Scientific Name	Total Catch	Min TL (mm)	Max TL (mm)	Spawning Noted	Notes
Common Jollytail	*Galaxias maculatus*	32	45	113	No	Diadromous
Carp Gudgeon	*Hypseleotris spp.*	5360	22	62	Yes	High rates of gravid and spawning fish
Murray Rainbowfish	*Melanotaenia fluviatilis*	2931	19	81	Yes	High parasite prevalence
Flathead Gudgeon	*Philypnodon grandiceps*	7712	20	90	Yes	Most common and widespread fish in the study
Western Bluespot Goby	*Pseudogobius olorum*	1	52	NA	No	Amphidromous
Congolli	*Pseudaphritis urvilli*	1	247	NA	No	Diadromous
Eel-tailed Catfish	*Tandanus tandanus*	64	27	432	No	'Protected' in the Murray and thriving in the Torrens
European Carp	*Cyprinus carpio*	4	26	45	No	Unexpectedly low numbers
Goldfish	*Carassius auratus*	5344	35	62	No	Restricted to below Henley Beach Rd.
Gambusia	*Gambusia holbrooki*	1205	18	55	Yes	Low numbers in this study

5.2.2 Torrens Endemic Species

All species captured were previously recorded from the reach (McNeil *et al* 2011a). Of the species observed, four are considered endemic to the Torrens; flathead gudgeon, jollytail, congolli and western blue spot goby. The flathead gudgeon was the most abundant and ubiquitous fish of the study. It appeared in multiple size classes with high rates of gravid and spawning individuals noted. These are a hardy and versatile fish species that thrive throughout their range.

The remaining three species share an ability to utilise oceanic life stages (diadromy and amphidromy). Whilst the common jollytail and congolli are widely distributed throughout the Western Mount Lofty Ranges they were almost entirely absent from the River Torrens during this survey. These diadromous fishes require oceanic connectivity and juveniles have previously been recorded accessing the river reach using the Torrens Fishway at the mouth of Breakout Creek (McNeil *et al* 2010). Currently, however, these diadromous species appear unable to pass the two recently constructed weirs between Breakout Creek and Henley Beach Road (Schmarr *et al* 2011).

The amphidromous western blue spot goby is a rarer fish in freshwater reaches of the Western Mount Lofty Ranges but one whose life history would be supported by oceanic connectivity. Until fish passage is reinstated throughout the Torrens, environmental water provisions are unlikely to result in increased abundance of diadromous fishes above the lowest barrier, except under very high flow conditions.

The freshwater eel, pouched lamprey and short headed lamprey are rarely observed in the Western Mount Lofty Ranges and were not detected in this survey although all have been recorded from the lower Torrens/Breakout Creek Reach (McNeil *et al* 2011a).They display a strong reliance on diadromous life histories and anthropogenic alterations to riverine environments have been implicated in their demise.

Removing or altering barriers to provide passage and supplying adequate and timely EWPs will support the life histories of most endemics in the Lower Torrens and the importance of these interventions cannot be understated (Schmarr et al. 2011).

5.2.3 River Murray Translocated Species

Three Murray-Darling species that have been translocated into the reach (McNeil and Hammer 2007) were identified during the survey, consistent with those captured previously from the reach (McNeil *et al* 2011a). Carp gudgeon occurred along the entire length of the lower Torrens in large numbers with high proportions of gravid and spawning individuals. This species is doing well under the current regime.

The Murray rainbowfish was observed in the larger and deeper pools of the Torrens in large numbers. Several individuals were in spawning condition. This species was commonly observed suffering from anchor worm (*Lernia sp.*) which is not uncommon. This species is doing well under the current regime.

The eel-tailed catfish is protected within the Murray River but has adopted the role of top-predator in the Torrens and appears to be thriving. Multiple size classes were observed including a cohort of juveniles (20-40mm) and 25 individuals greater than 300mm.

5.2.4 Exotic Species

Three exotic species were identified. European carp were observed in unexpectedly low numbers during the survey, although very large numbers of juvenile carp have been previously recorded from the reach, and adult carp are commonly captured (McNeil et al. 2011a). Four small individuals were caught in nets and a further 3 large adults observed from the Kayak. It is anticipated that further disruptions will occur with EWPs.

Goldfish were only identified in the stretch below the Breakout Creek Weirs. In this small stretch of river a single cohort was observed in remarkable densities (5344 individuals). In this instance the barriers appear to be playing a positive role, isolating this population from invading further upstream. The response of goldfish to strong flow is unclear but our hypothesis is that strong winter flows may wash a large proportion of this population out to sea.

Gambusia are a remarkably tolerant and ubiquitous fish across their introduced range. Numbers were steady across most stretches of the river with a slight preference for larger, calmer pools. Gambusia were regularly observed to be pregnant with dissection showing well developed young within female fish.

5.3 Cyanobacterial Impact Survey

Surveys were planned in response to flow releases to identify any accumulations of fish below barriers, or to detect any directional movement of fish in response to flows. At the time of the survey, however, high cyanobacterial levels developed, making it impossible for the field team to set nets and process fish. As a result, visual transect was conducted to collect observational data to address key aims, regarding accumulations of fish below barriers and behavioural indications of impacts of poor water quality.

On 24 and 25 January 2012, two SARDI ecologists undertook a visual survey of the lower Torrens River from the Morphett Street Bridge to the Port Road Bridge. Further examination was made at the Holbrooks Road gauging station, Henley Beach Rd and Breakout Creek. Fish observations focused on signs of toxic stress. These could include; aquatic surface respiration (ASR), loss of equilibrium and aggregations at inlet and outlet points, as well as obvious mortalities and large fish kills.

Maturity, number and behaviour of bird species were recorded along with turtle presence, behaviour and mortality. These taxa have the capacity to relocate from toxic areas and are unlikely to linger in areas that cause stress. Special note was made of piscivorous bird aggregations which may have reflected a struggling fish population. Invertebrate observations looked for yabby mortality or emergence from water (indicative of intolerable water quality) or unusual densities of macroinvertebrates at the water surface of littoral margins. Vertical profiles of water quality were recorded at strategic locations.

5.3.1 Findings of Cyanobacterial Impact Survey

Visual observations verified the bloom of blue-green algae, with colouration and surface scum indicating strong algal blooms throughout the reach (Figure 56), predominantly at the upstream section below the city weir. The observations made during the visual survey revealed no indications of any negative biotic impacts of the flows on the fish community. During the Baseline survey, carried out in December 2011, one dead carp and two dead birds were observed. This is comparable to the one dead carp and three dead birds observed in the Impact Survey.

There were no mortalities visible in any sheltered inlets along the bank or above dam walls where we would expect mortalities to accumulate. Observations revealed larval fish in high numbers

schooling in shallow littoral areas. Carp displayed no change in behaviour even in the most impacted pool immediately below the city weir. There were no abnormal numbers of predatory birds observed in any pool. All fish observations were consistent with those made at these sites in November.

Observations at the highly impacted area immediately below the Torrens Lake provide the best overall evidence that vertebrates were unaffected. Here animals capable of relocation remained *in situ* despite the bloom. This included long-necked turtles of multiple age classes, a family of black swans and several other species of bird. These animals displayed no abnormal behaviour and no apparent distress. While it is possible that these animals had accumulated to take advantage of mortalities from Torrens Lake flowing over the weir, this seems unlikely as no mortalities were present in the above-dam survey and animals were not actively hunting. Numbers of all Taxa were consistent with observations from previous surveys.

Figure 56. Local blooms lead to an intense milky green colouring of the water with a bright flocculate scum across the river's surface (Left). Immediately below the city weir a family of black swans makes no effort to relocate despite the obvious bloom conditions (Right).

Water quality parameters were within the range suitable for maintaining native fish populations (Table 22). Weakly stratified ODO levels were observed, but no water quality parameters were recorded at high enough levels to be likely to impact on aquatic biota.

Table 22. Water quality parameters for the River Torrens cyanobacterial impact survey in January 2012.

Site Name	Max ODO (mg/L)	Min ODO (mg/L)	Salinity (ppt)	Temperature (°C)	pH	Turbidity (NTU)
Torrens Dam	11.03	7.46	0.61	28.09	9.26	46.83
Railway Bridge	9.87	2.94	0.68	25.56	9.01	14.18
Bonython Park	17.96	8.57	0.39	27.68	8.80	6.90
Holbrooks Rd Upstream	12.73	12.48	0.44	33.61	9.41	12.65
Holbrooks Rd Upstream	28.58	13.57	0.39	28.15	9.52	23.10
Breakout Creek	11.31	10.36	0.46	26.35	9.36	10.17

5.4 Carp Tracking Study

A VEMCO acoustic tracking array and twenty-five carp implanted with VEMCO acoustic transmitters (V9-2X; nominal ping delay 90 ± 40 sec) were used to monitor the response of carp to the amenity flow within Torrens Lake (see Figure 57). The tracking array consisted of six VR2W receivers which were systematically placed to provide 90% coverage of the Lake. Eonfusion geospatial models were generated to visualize the response of carp to the flow (Figure 58). The carp response to amenity flow was compared to carp behavioural patterns recorded during periods of natural flow and no-flow to determine if there were any differences. These patterns have been captured as part of a broader tracking study being conducted by SARDIs Invasive Species sub-program in collaboration with the Adelaide City Council.

Figure 57. Common carp being implanted with VEMCO acoustic transmitters (left) and a VEMCO VR2W receiver (right).

5.4.1 Findings of the Carp Tracking Study

Given that the amenity flow coincided with the known carp spawning period (spring/summer), it was anticipated that carp might display a large upstream spawning migration toward the source of flow in search of potential spawning grounds. This behavioural pattern is observed on an annual basis throughout the Murray-Darling Basin and is particularly strong during spawning periods and periods of alleviated flows (Smith 2005, Conallin et al. In press). However, no migration was observed and carp behavioural patterns did not appear to differ from normal patterns observed during low-flow or no-flow conditions. During the amenity flow, carp were observed to move throughout the Torrens Lake following no fixed pattern. Movements were not in a consistent direction and while some carp appeared to aggregate around the central receivers (behind the Adelaide Zoo) this behavioural pattern was also observed during other periods. While it appears that the amenity flow did not trigger a large scale carp spawning event, there is potential that it may occur during subsequent flow events and this will require ongoing monitoring and careful management. To verify these findings, it is recommended that a carp population survey be conducted within the Torrens Lake System.

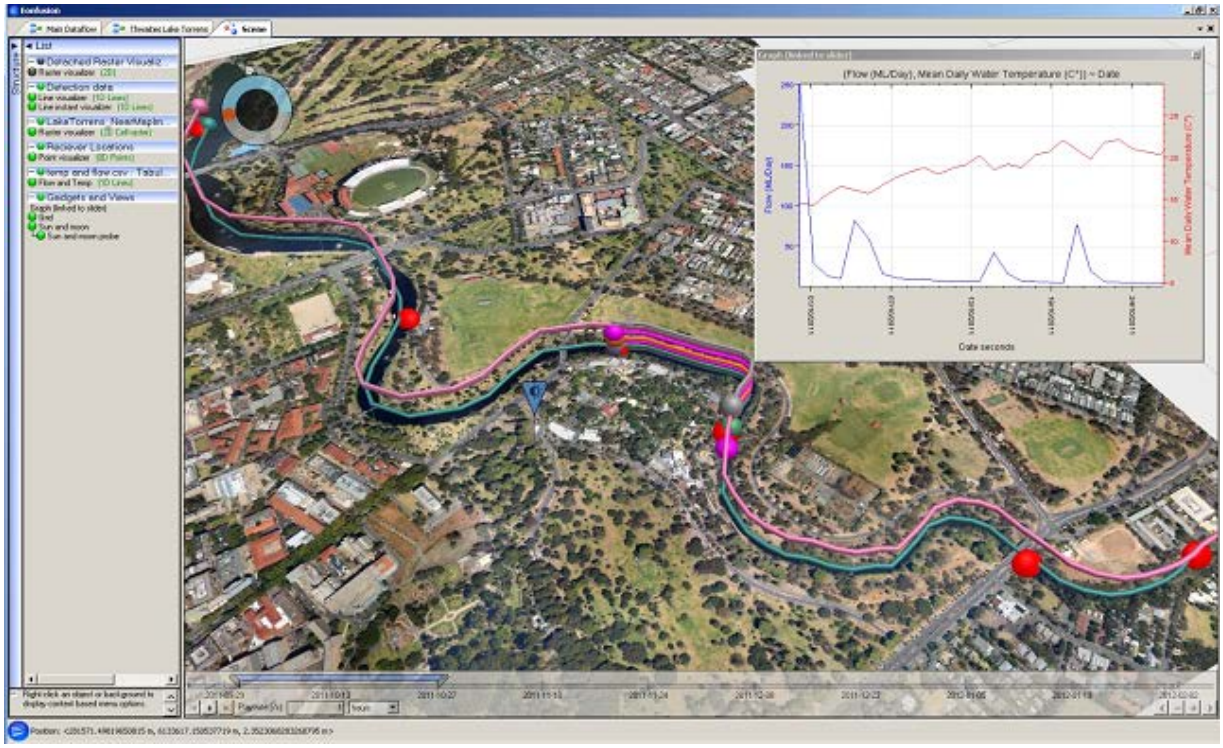

Figure 58. Eonfusion geospatial model showing placement of VR2W receivers (red) and typical carp movement throughout the Torrens Lake system.

5.5 Summary

The outcomes of the fish monitoring task found diadromous fish species did not appear to benefit from the flow. Fish numbers were largely consistent with previous surveys within the reach. There was no adverse biotic response to the release of cyanobacteria laden water provisions. There were no widespread fish or bird deaths. Pest carp spawning and aggregation did not occur in the Lake in response to the flow.

To assess the longer-term benefits of the release, there would need to be a post-flow survey which was not incorporated into the current study scope. In light of the environmental flows trial being conducted in the western Mount Lofty region, monitoring impacts of flows below the city weir should be reconsidered.

The initial survey found extremely low numbers of diadromous fish species throughout the reach suggesting that the beneficial impacts of the Torrens Fish Ladder at Breakout Creek (McNeil *et al* 2010) is not transforming into abundant populations of these fish throughout the lower Torrens. Post Flow sampling is required to determine whether the amenity flow provided an opportunity for migration and establishment of viable populations. The survey revealed the reach to be heavily

dominated by exotic fishes and translocated Murray-Darling species, not naturally present in the Torrens. Endemic fish numbers were extremely low suggesting the reach is in a very poor condition in terms of native fish values. Similarly, no observations of responses to toxic stress were observed, suggesting that fish within the river were not being negatively impacted by the flows, even during periods of high cyanobacterial load.

Whilst spawning was observed during the baseline survey, all species observed in spawning condition were summer low-flow spawners, with no spawning observed for flow responders. Post flow surveys may reveal recent recruitment in populations of flow responsive species such as galaxiids that would indicate a possible benefit of the amenity flow for stimulating native fish spawning.

Carp movement data above the city weir revealed no coordinated movements or aggregations of carp in response to the fresh inflows, despite the seasonal, temperature and habitat characteristics being amenable to carp spawning aggregations in response to small flow increases (McNeil et al. 2011b).

5.6 Conclusions

In summary, the amenity flow was not observed to have any impact on the freshwater fish community in either a positive or negative manner. The reach remains littered with migrational barriers, and the native endemic fish fauna remains depauperate and dominated by introduced species.

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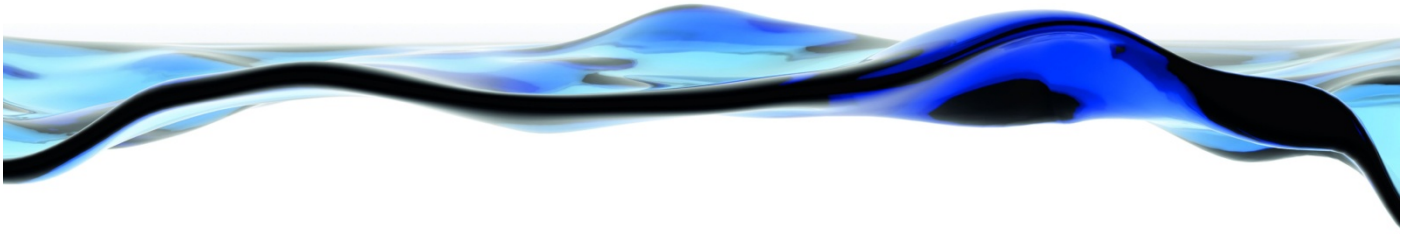

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