National Program for Irrigation Research and Development

Ecological Risk Associated with Irrigation Systems in the Goulburn-Broken Catchment – Phase II: Priority Risk – Blue Green Algal Blooms

Final Report

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Abstract

The occurrence of blue-green algal blooms within the Goulburn-Broken catchment has been identified as a priority ecological issue of concern to stakeholders. This study reports the findings of an ecological risk assessment designed to assess the risks associated with algal blooms within the catchment, and how those risks might be affected by factors associated with irrigation. The likelihood of bloom occurrence was assessed through numerical modelling of four ecosystems. Phase 1 of this project identified two case study systems: Lake Nagambie / Goulburn Weir, and the lower Goulburn River. In addition, another two systems (East Goulburn Main Channel, Lake Mokoan) were modelled to assist in validation of the assessment technique. Modelling enabled integrated examination of a number of environmental stressors, including nutrients, temperature, light, and flow regime within each system. The potential impact of these stressors on the likelihood of algal blooms was investigated. Level of filterable reactive phosphorus (FRP) was identified as the primary factor influencing the likelihood of algal blooms, and any future or current irrigation developments that increase the levels of FRP should be assessed carefully as they may increase the likelihood of bloom formation. As current concentrations are relatively low, the magnitude of environmental perturbation required to increase bloom risk is quite large. However, it should also be noted that a lack of site-specific data makes risk prediction very uncertain. Further monitoring, and possibly modelling, of the case study sites and alternative proxy sites may assist in reducing this uncertainty.

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1. Introduction

1.1 Preamble

During Phase I of the project entitled "Assessment of Ecological Risk Associated with Irrigation Systems in the Goulburn Broken Catchment" (Cottingham *et al.*, 2001) the project team, in conjunction with stakeholders from the catchment, identified the occurrence of cyanobacterial (or blue-green algal) blooms as a priority ecological issue to be further investigated in Phase II of the project. This report contains information on investigations carried out to assess the risks associated with cyanobacterial blooms, and how these risks may be affected by irrigation in the region.

1.2 Ecological Risk Assessment

Ecological risk assessment (ERA) is designed to assess the level of risk to ecosystems posed by either single or multiple stressors (e.g. salinity, toxicants, nutrients, temperature, flow, habitat, exotic species). Risk is often defined as the product of the *probability or likelihood* of a hazard occurring, and the *consequence* if that hazard occurs. Thus we have the concept:

Risk = Likelihood x Consequence

It is important to realize that risk represents a tradeoff between the likelihood of the hazard occurring, and the resulting consequences. Thus, an unlikely event may still represent an unacceptable risk due to its severe consequences. Conversely, a hazard that has relatively minor consequences may still present an unacceptable risk because it is likely to occur frequently. Thus, ERA provides a basis for comparing and ranking risks, so that natural resource managers can focus attention on the most severe risks first. Ideally, the ERA process should be iterative, allowing new information to be incorporated into the risk assessment as it becomes available.

ERA is not an especially new technique, but the majority of risk assessments done to date tend to be qualitative (or semi-quantitative at best) with descriptive ratings (e.g. unlikely, possible, likely, highly likely) being used to describe the risk. This approach has been criticised for its failure to account explicitly for subjectivity and uncertainty (Burgman, 2001). The aim always should be to use numerical data for both the consequence and likelihood components, and to use methods that make these assessments transparent and internally consistent (Burgman, 2001; Hart *et al.*, 2001). Current research in ERA is focusing particularly on the use of various forms of models to provide more objective estimates of the likelihood of events occurring. This report largely describes the development of a quantitative model, and its application to the problem of assessing the risks associated with cyanobacterial blooms.

While the majority of ecological risk assessments still focus on single issues and a limited number of stressors, there are now a number of initiatives aimed at further developing the ERA technique to provide a framework for considering a wider number of interacting stressors within a catchment or river basin context (Cormier *et al.*, 2000; Hart *et al.*, 2001; Leuven & Poudevigne, 2002).

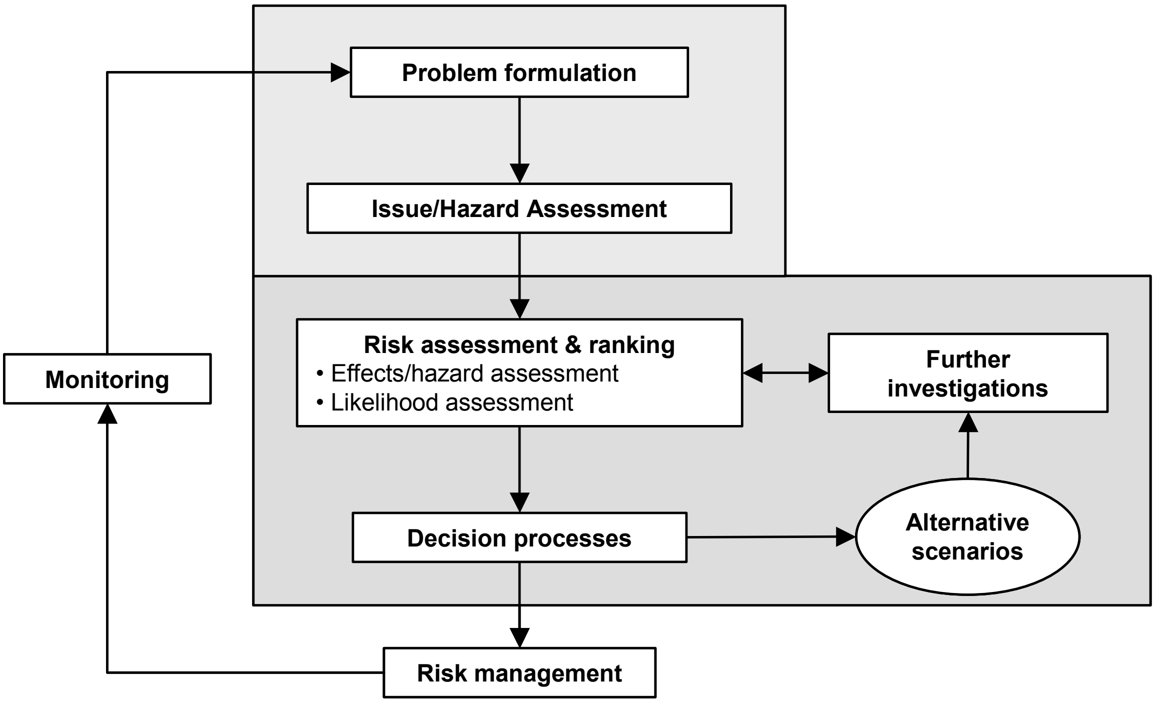

Figure 1: Conceptual diagram of the risk assessment / risk management process. The process begins with problem formulation followed by an assessment of the existing data. Assessing the risks to the system may require further data, and decision processes will be informed by examining multiple scenarios. Once risk management has been undertaken, results of the intervention will inform further iterations of the risk assessment process, which should benefit from the knowledge generated.

The main steps involved in an ecological risk assessment (problem formulation, risk analysis and risk characterisation) and the subsequent management and monitoring programs are shown in Figure 1. These are discussed below using examples from the current project - assessing the risks of cyanobacterial blooms within the Goulburn-Broken Catchment – to illustrate the key processes.

1.3 Problem formulation

This is the planning phase that establishes the goals, breadth and focus of the risk assessment. Key outputs from this stage are identification of the ecosystem(s) to be considered, their ecological values and the stressors that threaten these values. It is highly desirable that this step be undertaken with stakeholder involvement (Borsuk *et al.*, 2001).

The first stages of problem formulation were completed during Phase I of this project. See Cottingham et al. (2001) for a detailed description of the process. During this phase of the project it was determined that we should perform a specific risk assessment that focused on the likelihood of cyanobacterial blooms. No specific temporal boundaries were specified, and the study focussed on current conditions, as well as scenarios that altered one or more of the variables that affect the occurrence of cyanobacterial blooms. In terms of spatial boundaries, we were later directed to assess the risks of algal blooms for two specific areas within the catchment: Goulburn Weir / Lake Nagambie and the lower Goulburn River (Figure 2).

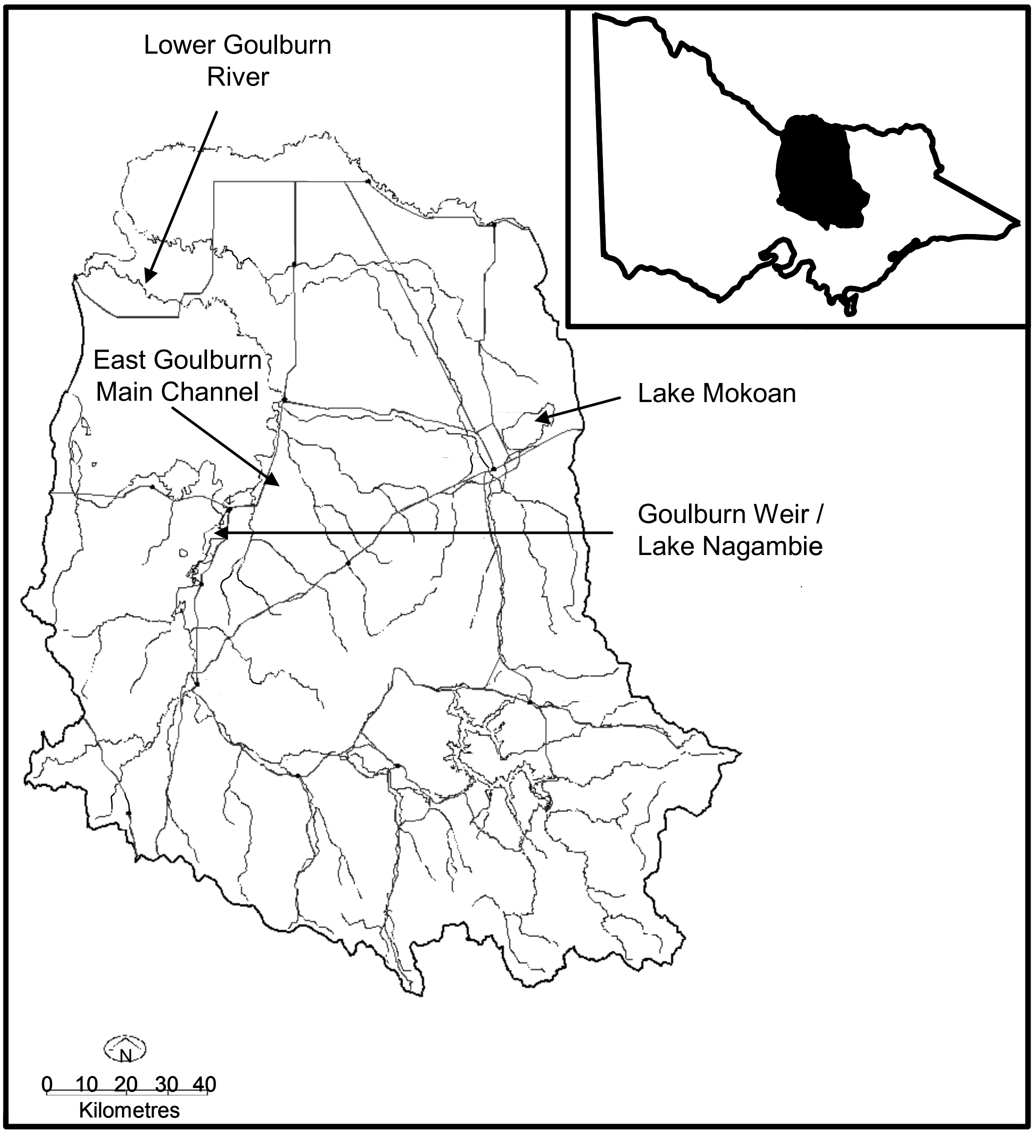

Figure 2: Map of the Goulburn and Broken catchments in Victoria, Australia. The two case study sites Goulburn Weir / Lake Nagambie and the lower Goulburn River are marked, as are the two analogue sites Lake Mokoan and the East Goulburn Main Channel that were used to validate the model used.

Conceptual models are an extremely useful part of the problem formulation process, and provide a basis for discussion about particular issues with stakeholders. Such models are representations of our present understanding of the system, and force researchers to clearly state their beliefs about the way in which system components interact. System-specific conceptual models are useful in forming the basis for quantitative ecological models in systems where there is both sufficient knowledge about the linkages and sufficient data to quantify these linkages (Borsuk *et al.*, 2001). They are also powerful tools for identifying substantial knowledge gaps in ecosystem understanding, and can thus be useful for directing specific investigations that may take part as part of the risk assessment process (Figure 1). A conceptual model of the processes governing cyanobacterial population levels in the two case study areas was prepared. This conceptual model, and the numerical model subsequently derived are fully explained in the case study methods below.

1.4 Risk Analysis

During this phase, information relevant to the key issue – in this case the occurrence cyanobacterial blooms – is gathered for the two components of risk: likelihood and consequence of the effects.

1.4.1 Effects Characterization

For many types of ecological risk assessments, multiple levels of effects will be considered. For example, Probabilistic Risk Assessment (sensu Solomon, Giesy & Jones, 2000) can consider the likelihood of an effect on 1%, 5%, 10% or indeed any percentage of species under consideration. In this case we are attempting to look at the likelihood of algal blooms. There are various management-oriented definitions of blooms around Australia. In Victoria, several different levels of blooms are recognized. A concentration of 2000 cells mL^{-1} is defined as a low alert level bloom, and triggers the introduction of drinking water treatment procedures. Subsequently, a concentration of 15000 cells mL^{-1} is defined as a high alert level bloom, and leads to the banning of recreational activities such as swimming and boating from affected water. Thus it would be possible to simultaneously consider risks associated with high and low alert level blooms. Unfortunately, our knowledge of the dynamics of cyanobacterial populations within the Goulburn-Broken Catchment (see below and in case study methods and results) does not allow us to quantify numbers of cells that will be produced with any confidence. Thus, we confined the risk assessment to the single effect of "algal bloom", without specifying how severe the effect might be.

1.4.2 Likelihood Characterization

The focus of the work undertaken during this phase of the project has been the development of models that can be used to simulate the development or otherwise of cyanobacterial blooms under various conditions. A relatively simple process-based model was applied to the case study sites as well as two other sites within the catchment (see below). Use of the model allowed 'experiments' to investigate the resistance of the system to algal blooms.

1.4.3 Further investigations

In undertaking a risk assessment, it is often found that further investigations are needed to provide more information before the detailed assessment can be undertaken (Figure 1). For this study, specific investigations were carried out on the relationship between turbidity and light attenuation. These studies were necessary in order to parameterize the likelihood model, and thereby provide estimates for the risk characterizations. The investigations are described fully below.

1.4.4 Feedback

Figure 1 shows that the risk assessment and management process is iterative in nature. Knowledge gaps identified during a risk assessment may not be able to be filled within the time lines of the initial project. Alternatively, a risk assessment may not be able to quantify risk with desired precision or certainty. The knowledge generated during a risk assessment provides a starting point for subsequent investigations, which will then show an iterative improvement in the quality of the risk estimates provided. For this case study, we have been unable to provide quantitative estimates of the risk of algal blooms, but have been able to show that major environmental perturbations would need to occur before blooms become a likely event in either Lake Nagambie or the lower Goulburn. The results of this study will point researchers towards more rigorous validation of the algal growth models, such that quantitative estimates of algal numbers / biomass can be forecast, rather than a qualitative recapitulation of biomass dynamics, as was achieved in this case. The first step in such a process would be the continued monitoring of the case study sites so that the performance of the models applied during this study could be better assessed.

1.4.5 Risk Management

In this case little needs to be done to manage the risks of blue green algal blooms at the case study sites. In terms of long-term catchment management, the only recommendation of this risk assessment would be to limit the development of agricultural or urban development upstream of Goulburn Weir, as such development could lead to the types of increases in phosphorus that are likely to lead to algal blooms in the system during some years. Of course, if such development became an economic imperative, managers could look at other ways to reduce the likelihood of blooms in the impoundment, such as increased flow rates with resultant flushing, or to introduce planning controls to ensure phosphorus was trapped onsite or mitigated in transit rather than being efficiently delivered to the river through effective drainage networks.

1.5 Cyanobacterial (Blue-Green Algal) Blooms

Cyanobacterial blooms represent an enormous challenge for water quality management the world over, and reports of these events have increased in frequency in recent years (Whitton & Potts, 2000). Consequences of water contaminated by cyanobacteria include, but are not necessarily restricted to: death of stock and wild animals after drinking contaminated water (Beasley *et al.*, 1989), human health effects such as liver failure (Jochimsen *et al.*, 1998) and increased cancer rates (Yu, 1994), loss of tourism revenue due to reduced aesthetic amenity of waterways and the banning of water-based activities (e.g. Walker & Greer, 1992), and costs associated with the treatment of drinking water supplies or the supply of alternative drinking water (e.g. Alaouze, 1999). Public concern about cyanobacterial blooms in Australia was first aroused by a massive bloom of *Anabaena circinalis* in 1991 that affected over 1,000 km of the Darling River, New South Wales (Bowling & Baker, 1996). At the time, this was the largest cyanobacterial bloom ever recorded in inland waterways.

The prevalence of cyanobacterial blooms has led to a great deal of research into the factors that cause them. Early research indicated the importance of phosphorus availability in determining average algal biomass (Dillon & Rigler, 1974), and as a consequence, anthropogenic eutrophication is often seen as the root cause of excessive algal growth (e.g. Smith, Tilman & Nekola, 1999). However, other factors have also been shown to be important in mediating bloom formation. These include a stable water column, warm weather, high incident irradiance, enhanced allochthonous organic matter loading, and selective activity of grazers (Paerl, 1988).

Minimization of the severity and frequency of cyanobacterial blooms is a key target for many 'whole-of-catchment' management strategies (e.g. GBCMA, 2002). Effective management of cyanobacterial blooms requires an understanding of the local conditions that favour bloom formation. However, since many factors determine whether or not a bloom will form, methods are required that are able to integrate the effects of various environmental variables. Modelling is one technique that can provide such integration.

A number of models already exist that seek to simulate the dynamics of cyanobacterial populations (e.g. Whitehead & Hornberger, 1984; Lung & Paerl, 1988; French & Recknagel, 1994; Howard *et al.*, 1995; Easthope & Howard, 1999; Everbecq *et al.*, 2001; Maier, Sayed & Lence, 2001; Reynolds, Irish & Elliott, 2001). Some of these models perform remarkably well, and can simulate the population dynamics of multiple species of competing algae and cyanobacteria. However, many of these models utilize environmental data for a large range of parameters (e.g. Yabunaka, Hosomi & Murakami, 1997) and/or at a fairly fine spatial / temporal scale (e.g. Lewis *et al.*, 2002). Such data are quite rare, and do not exist for the case study areas in this investigation. Thus, a model that can be driven by standard monitoring data is required.

In addition to the requirements for more comprehensive driving data sets, models of various aspects of cyanobacterial biology or population dynamics are also tending towards greater and greater complexity (e.g. Laws & Chalup, 1990; Belov & Giles, 1997). The volume of work done to elucidate cyanobacterial biology and the characteristics of the physical environment they inhabit has facilitated this. Such models lead to detailed explanations of observed system behaviour, but will generally be of less use in predicting future states. Conversely, simpler models cannot explain as much, but may be more useful for prediction (Scheffer & Beets, 1994; Steel, 1997). Simple models usually require less in the way of driving data and parameterisation, and so future scenarios can be modelled with greater confidence (Scheffer & Beets, 1994; Breiman, 1995). One must also consider the aim of the model. For an ecological risk assessment, being able to predict the onset of cyanobacterial blooms is of prime importance. Due to the fact that management decisions concerning reactions to algal blooms are based on the exceedence of 'alert levels' or thresholds, modelling threshold exceedence is of prime importance for an ecological risk assessment. Managers need to be able to know what combinations of conditions will lead to an unacceptably high likelihood of an algal bloom, so that preventative measures can be taken.

1.6 Case study sites

During the problem formulation stage (see above) stakeholders decided that the study should focus on the risks of algal blooms in two areas within the Goulburn-Broken catchment where the consequences of an algal bloom would be of concern. The two areas chosen were the Goulburn Weir / Lake Nagambie system, and the Lower

Goulburn River (Figure 2). Another impoundment in the Goulburn-Broken catchment, Lake Mokoan, has been subjected to regular algal blooms since the mid 1980s. However, Lake Mokoan has been extensively studied, and further specific study was not deemed necessary within the context of this ecological risk assessment. Nonetheless, in validating the modelling approach used for Lake Nagambie, we utilized data from Lake Mokoan (see below).

1.6.1 Area 1: Goulburn Weir / Lake Nagambie impoundment

This area includes waters whose water surface levels are dictated by Goulburn Weir. This includes small stretches of river, and three relatively large open areas: Goulburn Weir, Richard's Flats and Lake Nagambie. The areas are collectively referred to as Lake Nagambie throughout this report. The impoundment serves to guarantee the flow of water to the East Goulburn Main Channel to the east, and to the Cattanach and Stuart Murray Canals to the west. Lake Nagambie also has social value as an area for water sports and recreational fishing.

The main impacts of irrigation on these waters are expected to be flow related. The creation of Goulburn Weir created a largely static water environment, whereas previously flow was continuous. There is now little fluctuation in water levels within the system. There has also been a reversal of seasonal flow patterns through the system with higher flows occurring during the summer irrigation season than during winter.

Monitoring data are collected as part of the Major Storages Operational Monitoring Program (MSOMP). Physico-chemical parameters regularly measured include temperature, EC, turbidity, FRP, TP, TP, TKN andNOx. Algal count data are also available, with identification of major groups including cyanobacteria, chlorophytes, diatoms and dinoflagellates. Cyanobacteria have further been identified and enumerated to genus level, including *Microcystis* and *Anabaena*. Monitoring data are collected from close to Goulburn Weir, and for the purposes of this exercise, must be extrapolated to the entire volume of the impoundment.

The weir has a volume of approximately 25 x 10^6 m^3 (25,000 ML). Thermal stratification of the water column does not appear to occur within the impoundment. SKM (2002) found some diurnal stratification, but waters generally mixed overnight. This may be a function of the relatively shallow water depth. The area is also exposed to some wind disturbance, which would facilitate mixing of the water column.

1.6.2 Area 2: Lower Goulburn River

We define this area as running from the entrance of the Rodney main drain to the confluence with the Murray River, although for practical purposes (availability of monitoring data), this stretch will have to be considered as a homogenous section of river.

The main impacts of irrigation on the Lower Goulburn are expected to be associated with irrigation returns to the river, which will introduce substantial quantities of nutrients, sediments, and possibly salt to the river system. In terms of flow, regulation of the river above the lower stretches means that there has been a reduction in the frequency of floods, along with a reduction in the total amount of water passing to the Murray. Stratification is also unlikely in the lower Goulburn due to the amount of flow.

Physico-chemical data on a wide range of parameters are available from the McCoys Bridge monitoring station that forms part of the Victorian Water Quality Monitoring Network (VWQMN). Monitoring data are also available for the irrigation returns exiting the Rodney main drain as part of Goulburn Murray Water's drain monitoring program. Data on algal abundances essentially do not exist, but anecdotally, there have been no algal blooms within this section of the river within the last ten years.

1.6.3 Analogue sites

Importantly, neither of the case study sites has a documented history of algal blooms. This lack of blooms means that it is not possible to validate bloom models using monitoring data for the sites. Moreover, to our knowledge, there have been no attempts in the past to model cyanobacterial populations at either of the two sites.

In order to give an indication of how well models were working for the case study sites, the cyanobacterial model was applied to data from two nearby systems that have a history of cyanobacterial blooms. Lake Mokoan has been well studied and is the subject of several detailed models of cyanobacterial population dynamics. The East Goulburn Main Channel has not been studied in the past, but elevated cyanobacterial numbers were recorded over the summer of 2002/03. The channel is used to convey irrigation water from the Lake Nagambie impoundment to various areas, and flows over the irrigation period (August-May). During the rest of the year, there is little flow within the channel, and it essentially dries to a series of pools. The two systems provided rough analogues of the Lake Nagambie, and lower Goulburn river sites respectively (Figure 2).

2. Bloom Modelling

2.1 Rationale

The data available for the two case study sites (and for the East Goulburn Main Channel) do not cover the range of parameters necessary to drive some of the more detailed algal models that have been produced. Moreover, the specific knowledge of the systems that would be required to parameterize the models is also unavailable. As such we are unable to apply models that can give a detailed representation of algal population dynamics. Lake Mokoan has been the subject of several detailed models of cyanobacterial population dynamics, but for the reasons above, these models cannot be applied to the case study sites.

However, as stated above, the lack of a detailed population dynamics model is a minor concern for an ERA-based modelling exercise. Our primary aim is to be able to qualitatively track biomass dynamics, and identify the types of conditions that could lead to blooms in the systems.

We applied a model originally developed for the weir pool at Bourke, N.S.W. (Webb, Linacre & Grace, submitted) to the case study and analogue sites. In its original application, this model successfully recreated all the blooms that occurred in Bourke Weir over a 10-year period. It also falsely predicted one other bloom. Thus, we had an *a priori* expectation that this model would provide a conservative estimate of the conditions likely to lead to bloom formation in the Goulburn-Broken sites. Some modifications were carried out compared to the model described in Webb et al. (submitted). The model is described below.

The model was developed for the bloom-forming cyanobacterium *Anabaena* sp. *Anabaena* is the most common bloom forming cyanobacterium in Australian inland waters (Jones & Orr, 2000), and members of the genus are common throughout the world. This genus is also common in blooms within the Goulburn Broken catchment. The cells form filaments that can control their buoyancy (Oliver & Ganf, 2000), and can thus float into well-lit surface waters during periods of reduced water circulation. *Anabaena* species also have the ability to fix atmospheric nitrogen (Oliver & Ganf, 2000). The dominant bloom forming taxon in Lake Mokoan is *Microcystis aeruginosa*, and so several parameters in the model specific to *Anabaena* sp. had to be changed. These parameters were the cell quota of phosphorus (P_{cell}), the half saturation rate of phosphorus uptake (K_S), the temperature-growth multiplier (Q_{10}), and the basal growth rate (r_{20}). Values for the parameters for both the Lake Mokoan model and the *Anabaena*-based models are given in Table 1. Unlike *Anabaena*, *Microcystis aeruginosa* cannot fix atmospheric nitrogen. Strictly speaking, the model for Lake Mokoan should have included an opportunity for Nitrogen limitation. However, the additional effort entailed in such modelling was not justified, given that Lake Mokoan is designated as an analogous site with which we can test the model developed for Lake Nagambie.

2.2 Conceptual model

The numeric model is based on the conceptual model pictured in Figure 3. In the model, either light or phosphorus limitation, together with the effect of water temperature, regulate the daily rate of algal reproduction. Mortality reduces population numbers, as does a component representing the net number of cells washed from the weir each day. Day length combines with turbidity to determine the extent of light limitation on population growth. The amount of filterable reactive phosphorus determines the extent of phosphorus limitation on cell production.

2.3 Implementation

The model was implemented in Mathcad Professional 2001 (MathSoft Inc., 2000). It was driven by a Microsoft Excel 2000 (Microsoft ®, 1999) spreadsheet of monitoring data (see below), and delivered results to a separate sheet. The algal population was projected via a difference equation that calculated the number of cells mL^{-1} within the modelled water body each day.

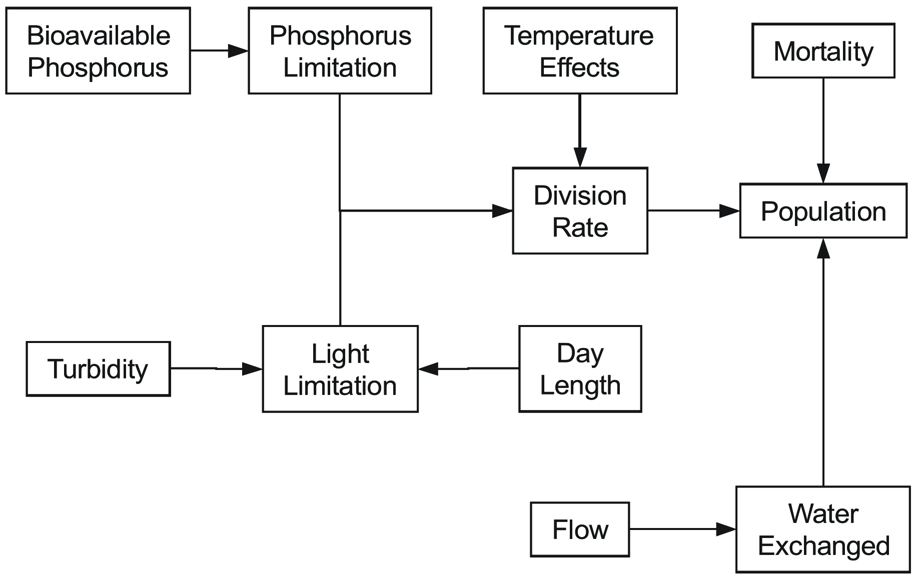

Figure 3: Conceptual model of cyanobacterial population dynamics. The population is governed by the rate of mortality, the rate at which cells are washed from the system, and the rate at which cells are produced. Reproductive rate is governed by either light or phosphorus limitation, and is scaled according to temperature. Phosphorus limitation is dictated by the amount of filterable reactive phosphorus available, and light limitation by the combination of day length and water turbidity. Strictly speaking, this model is only appropriate for N-fixing cyanobacteria.

2.4 Driving data

The model was driven by monitoring data for filterable reactive phosphorus (P_{frp}), turbidity (Tu), temperature (Te), and flow through the system (F). Flow data were generally available daily. Data for P_{frp}, Tu and Te had been collected as single sub-surface samples, and were available at varying time intervals from weekly up to approximately monthly. We used linear interpolation to provide a 'daily' data set that could drive the models. Day length (N) was approximated using a sine function that added (or subtracted) a proportion of a day length range to an average day length, depending on the date.

2.5 Model Functions

2.5.1 Parameter values

The functions, and the rationale behind each of them, are outlined below. The majority of calculations were performed for each day, but for ease of reading, most equations are presented without the $_t$ subscript. Parameter values and the sources of these data are as in Webb et al. (submitted), with the exceptions as outlined in Table 1 and Table 2.

Table 1: Model coefficients for algal cells parameters, where they differ from those used in Webb et al. (submitted)

Parameter	Temperature adjustment for 10°C	Growth rate at 20°C	**Half saturation coefficient for P uptake**	**Phosphorus quota per algal cell**
	Q_{10}	r_{20} [day^{-1}]	K_s [μmol L^{-1}]	P_{cell} [μmol]
Default values (*Anabaena* sp.)	2.1 [a]	0.58 [b]	0.21 [c]	$1.42 \cdot 10^{-8}$ [d]
Lake Mokoan (*Microcystis* sp.)	1.8 [e]	0.63 [f]	0.18 [g]	$8.87 \cdot 10^{-9}$ [h]

[a] Healey (1973)
[b] McCausland et al. (2002)
[c] Cembella et al. (1984)
[d] Thompson *et al*. (1994)
[d] 1.4-3.0 in Robarts and Zohary (1987) for *M. aeruginosa.*
[e] 0.1-0.8 d^{-1} in Oh et al (2000) for *M. aeruginosa.*
[f] Holm and Armstrong (1981) for *M. aeruginosa.*
[h] note: default actually used, this value estimated from Vollenweider (1968) and Berman and Pollingher (1974) for *Microcystis* sp. cell data.

Table 2: Water body volumes used in modelling the rate of cell wash-out from the systems

Volume of East Goulburn Main Channel	$9.9 \cdot 10^4$	m^3
Volume of Lake Mokoan	$4.5 \cdot 10^3$	m^3
Volume of Lake Nagambie	$2.5 \cdot 10^7$	m^3
Volume of lower Goulburn River	$8.0 \cdot 10^6$	m^3

2.5.2 Flow related processes

Flow only affected the population in that cells were washed from the water body by flow out of the system. We assumed that because the systems do not stratify, cells mix over the depth of the water column. Algae are washed from the weir at the ambient cell concentration (cA), and wash into the weir at a small nominal concentration (cA_{wi}). Therefore the proportion of the system to flow out each day was multiplied by the difference between cA and cA_{wi} in the final population projection (Eq. 6) to determine the net number of cells per unit volume washed out of the weir on any given day. This part of the model requires that a system volume be specified, so that cell concentrations can be calculated after the effects of dilution are accounted for.

For the Lake Nagambie and Lake Mokoan systems, volumes are readily available. The East Goulburn Main Channel can also be visualized as something akin to a weir pool environment, due to the low flow rates that were observed through the system, and a volume figure was calculated from bathymetric data and the length of the channel. Conceptually, the lower Goulburn River is more difficult to regard as a weir pool. However, for the purposes of this model, we defined the 'volume' of the system as being for that length of channel between McCoy's Bridge and the confluence with the Murray (as defined above). McCoy's Bridge is shortly downstream of the input of the Rodney Main Drain, and so we might reasonably expect that algal bloom events would be initiated around this point by nutrients introduced to the system by the drain. The volume was calculated using several cross-sectional areas of the channel, multiplied by the length of channel. For the Lake Mokoan system, stratification of the water column is observed. Therefore, we implemented the model as originally described in Webb et al. (submitted), with cells circulating within an upper mixing layer under stratified conditions or within the entire water column under mixed conditions.

2.5.3 Light limitation

Limitation of growth by light is mediated by water turbidity (*Tu*) and day length (*N*). We assume that growth is not limited by light until light falls below a saturating intensity (I_{sat}). Below this figure, growth decreases linearly with light to zero. This simple relationship has previously been applied to the modelling of growth-irradiance relationships (Bannister, 1979), and has also been successfully applied to photosynthesis-irradiance modelling (Jassby & Platt, 1976). In the model, surface irradiance is assumed to be constant throughout daylight hours, ignoring changes across the course of each day and with season. The coefficient of light attenuation (*k*) is expressed as an empirically determined linear function of turbidity, which was derived during specific investigations undertaken for this project (see below).

$$k = 0.047 \cdot Tu + 0.86 \tag{1}$$

The depth at which I_{sat} is reached (Z_{sat}) is calculated using a transposition of the Beer-Lambert equation,

$$Z_{sat} = \frac{\ln(I_{sat})}{-k} \tag{2}$$

where I_{sat} is expressed as a proportion of the surface irradiance. We assume that cyanobacterial cells are evenly distributed throughout the water column (Z_w). Depending on *Tu*, cells are subjected to one of two growth limitation regimens. First, if the cells circulate entirely within the well-lit zone ($Z_w \leq Z_{sat}$), then growth is not limited. If Z_{sat} is shallower than Z_w, then growth is reduced for part of the time.

$$I_{lim} = \frac{N}{N_{\max}} \cdot \begin{cases} 1 & \text{if } Z_w \leq Z_{sat} \\ \left(Z_{sat} + \dfrac{I_{sat} - \mathrm{e}^{-k \cdot Z_w}}{k \cdot I_{sat}} \right) \cdot \dfrac{1}{Z_w} & \text{otherwise} \end{cases} \tag{3}$$

Limitation between the depths Z_{sat} and Z_w is derived from the definite integral of the Beer-Lambert equation within this interval, and calculates the proportion of possible growth occurring over the depth interval. The average limitation of light over the course of each day is scaled for day length so that light can only be truly non-limiting ($I_{lim} = 1$) on the longest day of the year.

2.5.4 Phosphorus availability and uptake rate

We were fortunate that the monitoring data included filterable reactive phosphorus (P_{frp}), rather than total phosphorus. Thus no conversion was required to estimate bioavailability. The proportional extent of phosphorus uptake limitation is described by Michaelis-Menten kinetics.

$$P_{lim} = \frac{P_{frp}}{K_S + P_{frp}} \tag{4}$$

The maximum rate of phosphorus uptake (V_{max}) does not appear in the equation as the model only needs to determine the proportional extent to which the rate of P uptake is limited in order to calculate phosphorus limitation.

2.5.5 Temperature effects

In the model the rate of reproduction increases with temperature via a scalar, S_{Te}. The exponential relationship is modelled by

$$S_{Te} = \left(\sqrt[10]{Q_{10}}\right)^{Te-20} \tag{5}$$

In this case, the tenth root of the Q_{10} (the proportional increase in rate after a 10°C rise) gives the proportional increase for each degree C. The value of S_{Te} is 1 at 20°C, which is the temperature at which the basic rate of reproduction (r_{20}) was measured (see below).

2.5.6 Population projection

The population is projected using a difference equation that follows the basic structure: new pop = starting pop – mortality – net cells washed from weir + cells produced. The rate of reproduction is reduced by either light or phosphorus limitation (whichever is more limiting), and scaled according to temperature.

$$cA_{t+1} = cA_t - q \cdot cA_t - V_{ex_t} \cdot \left(cA_t - \cdot cA_{wi}\right) + \min\left(r_{20} \cdot \min\ (I_{lim_t}, P_{lim_t}) \cdot S_{Te_t} \cdot cA_t,\ \frac{P_{drp_t}}{P_{cell}}\right) \tag{6}$$

The basic rate of reproduction for *Anabaena* sp. (r_{20}) was taken from the study of McCausland *et al.* (2002). The number of cells produced is the product of the modified rate of reproduction and the population. Occasionally, this calculation results in more cells being produced than can be supported by the amount of available phosphorus. In these cases, the number of cells produced is constrained to the number supported by P_{frp} by dividing the phosphorus pool by the amount required for one cell P_{cell}.

2.6 Validation and scenario testing

The model was run for each site for periods where driving data for parameters, and monitoring data for algal abundances were available. On the graphs that follow, these simulations are referred to as “base line”. For each of the sites, we then tested various scenarios in order to identify the combinations of conditions that might lead to blooms at the sites. These scenarios involved artificially altering the temperature, level of FRP, turbidity, flow, and combinations of these factors.

3. Specific Investigations – Light Attenuation

The relationship between turbidity and specific light attenuation depends upon the types of sediments suspended in the water column. For the original application of the model, an empirical relationship developed for the waters of Bourke Weir (Oliver *et al.*, 1999) was used. For this project, we undertook similar investigations. A total of 25 measurements were made of the light attenuation coefficient within the East Goulburn Main Channel and Lake Nagambie. The resulting relationship between light attenuation and turbidity was different for those points taken during Summer and Autumn when compared to those points taken during Winter and Spring. Accordingly, the data were separated, and two relationships were calculated. The measurements resulted in a relatively good linear relation between turbidity and the light attenuation coefficient for both data 'sets' (*k*; Figure 4, Figure 5). The slope of the relationship was very similar for the two data sets, but the intercept varied. Given that the vast majority of algal blooms occur during the warmer months of the year, we used the Summer-Autumn relationship coefficients in the model. We applied this relationship to all of the sites modelled, as there were no other estimates available. Moreover, the relationship was very similar to that originally calculated for Bourke Weir – slope = 0.04, intercept = 0.73, (Oliver *et al.*, 1999), which suggests that site-to-site variation may not be very important.

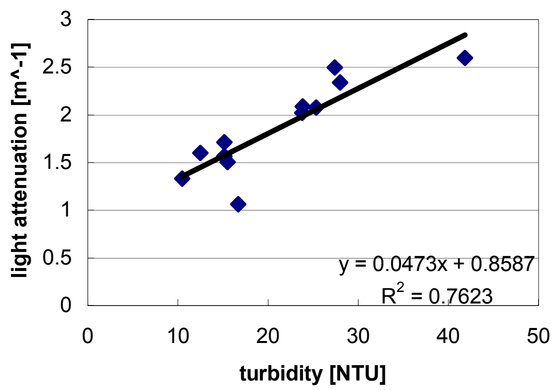

Figure 4: Light attenuation vs turbidity for measurements made during summer-autumn

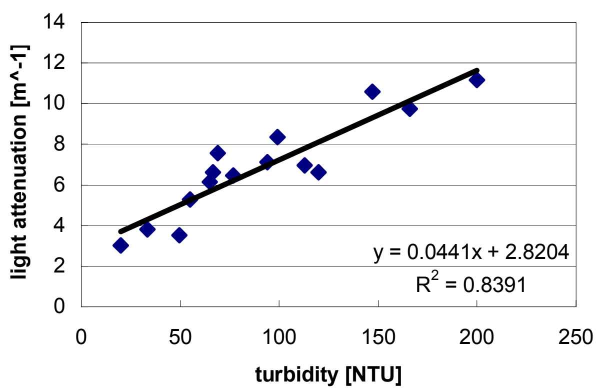

Figure 5: Light attenuation vs turbidity for measurements made during winter-spring

4. Application to the case study and analogue sites

4.1 Site i: East Goulburn Main Channel

4.1.1 Validation

The model was applied to the East Goulburn Main Channel (Figure 2) for 2003 only, as water quality monitoring commenced only when a mild *Anabaena* bloom was noted in March of that year (Figure 6). The modelling simulates the collapse of the bloom from March through May for the field algal counts available. After this period, counts were not performed, as biomass appeared to be low within the channel. During the initial period, the simulations underestimate biomass, but do indicate presence of blue-green algae until the end of May. Further simulations indicated sensitivity of the model to temperature. This sensitivity may account for the discrepancy, as the field data used to drive the model were collected early in the morning. This additionally suggests that elevated temperatures that were most likely experienced in the weeks preceding the observed bloom (it was mid-summer) would have been a significant factor in initiating the bloom. From May to October, the model indicates that significant levels of blue-greens are unlikely. Although cell counts were unavailable, chlorophyll *a* levels during this period were low, reaching a maximum of 15 µg/L (average ~5 µg/L).

4.1.2 Scenarios tested

As mentioned above, temperature (blue lines, Figure 6) had a noticeable influence on biomass. Increasing the temperature (+2°C and +3°C) extended the period of elevated biomass, but peak magnitude remained moderate. Similarly, elevating the available nutrients (FRP, the green lines) and decreasing turbidity (the grey line) also extended the period of risk of an algal bloom, but total magnitude remained moderate. However the most significant effects were seen when these changes to water quality were combined (increased FRP combined with increased temperature and with increased water clarity). Such an effect is perhaps, not surprising, as these scenarios improved two of the three factors that generally limit algal biomass. Due to a lack of data, it is unknown whether such levels of FRP are seen within the channel. Thus it is not possible to predict the likelihood of such scenarios.

The effect of decreasing the flow (and thus flushing) of the channel was negligible, and is not plotted. The channel only really flows during the summer irrigation season, but water velocities even during this time are very low. Consequently, the fact that further reducing flow did not greatly affect model output is not surprising.

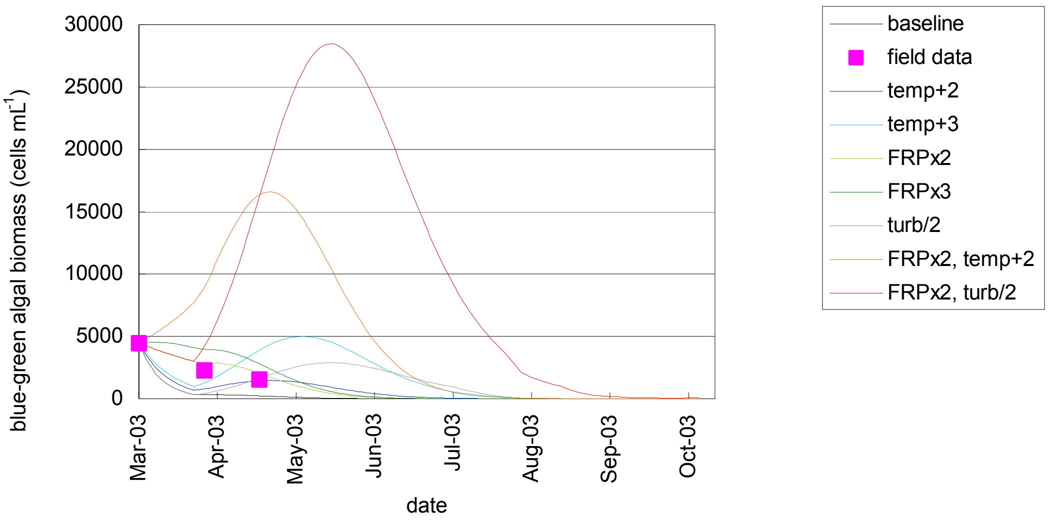

Figure 6: East Goulburn Main Channel, 2003 commencing with bloom collapse, with effects of input data variation

4.2 Site 1: Lake Nagambie / Goulburn Weir

Lake Nagambie was modelled for the end of 1999 through to mid 2001, and for the end of 2002 to mid 2003. The first period showed typical behaviour for the system, with low algal biomasses. The second period was unusual for the system in that cell counts of *Anabaena* sp. reached several thousand per mL.

4.2.1 Modelling results 1999-2001

For the first period, as expected from field algal data, conditions in this system are quite resistant to algal blooms. Despite high clarity and availability of light and moderate temperatures in summer, the very low nutrient levels severely restrict development of algal biomass. This limitation was successfully replicated in the simulations (baseline, Figure 7), with the trace being almost invisible.

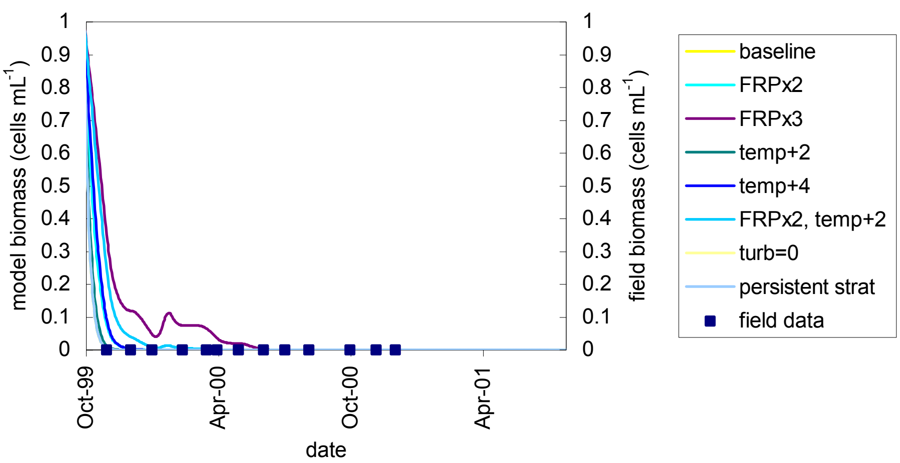

Figure 7: Scenarios tested in Lake Nagambie. Refer to key for combinations of conditions.

4.2.2 *Scenarios 1999-2001*

A number of scenarios were examined. Most cases followed the baseline case very closely (Figure 7, note most data overlap the lowest line, including the baseline). Increases in FRP of two- and three-fold had a minimal effect on the behaviour of the modelled population. Temperature increases (of 2°C and 4°C), and a major increase in clarity (turbidity set to 0 NTU) had a negligible effect. An increase of fourfold FRP appears to cross a threshold, however peak biomass (off axis) remains low (< 100 cells mL^{-1}). Additionally, applying the model with persistent stratification, and with reduced flow had negligible effect, due to the clarity of the water in the former case, and the negligible size of flow compared to Lake volume in the latter case. Only in the most extreme scenarios (Figure 9) did the risk of algal blooms become significant. A fivefold increase in FRP results in blooms during the summers of 2000 and 2001. Combining an elevated FRP (threefold increase) with a temperature increase (4°C) also results in severe seasonal blooms. It should be noted that increasing Lake Nagambie FRP fivefold (~0.003 mg L^{-1} → ~0.015 mg L^{-1}) results in FRP levels that are still below the levels found in Lake Mokoan.

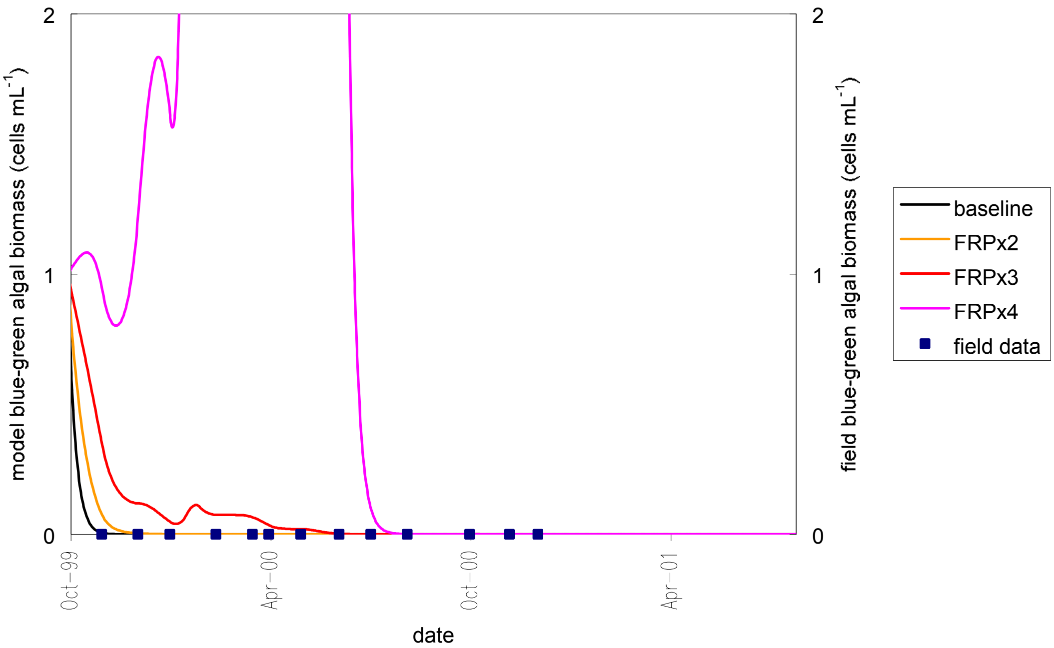

Figure 8: Lake Nagambie algal biomass with FRP multiplied by two, three and four times field concentrations.

4.2.3 *Modelling results 2002-2003*

Results for the 2002-2003 period are presented in Figure 10. Algal biomass in Lake Nagambie during 2003 was significantly higher than observed in previous years (max 2800 cells mL^{-1} vs 470 cells mL^{-1} in previous record), but the model simulated this increase in biomass poorly, with biomass decreasing to less than 1 cell mL^{-1} and staying constant over the simulated period. However, from the end of January 2003, the supplied water quality data had a constant, low value of FRP (0.0015 mg L^{-1}). A constant value such as this seems unlikely in reality, and the magnitude is low in comparison to previous portions of the water quality data. We believe, therefore, that there might be a problem with these FRP data. Replacing the suspect FRP data with phosphorus levels more akin to those normally observed within Lake Nagambie (increasing FRP by a factor of three, four or five), the period of elevated biomass more closely approximates the field counts. As with other cases, the model's main strength appears to be prediction of the timing of onset of elevated blue-green levels. Increasing temperature for this data set only had an effect when done in combination with increased FRP concentrations relative to the suspect data.

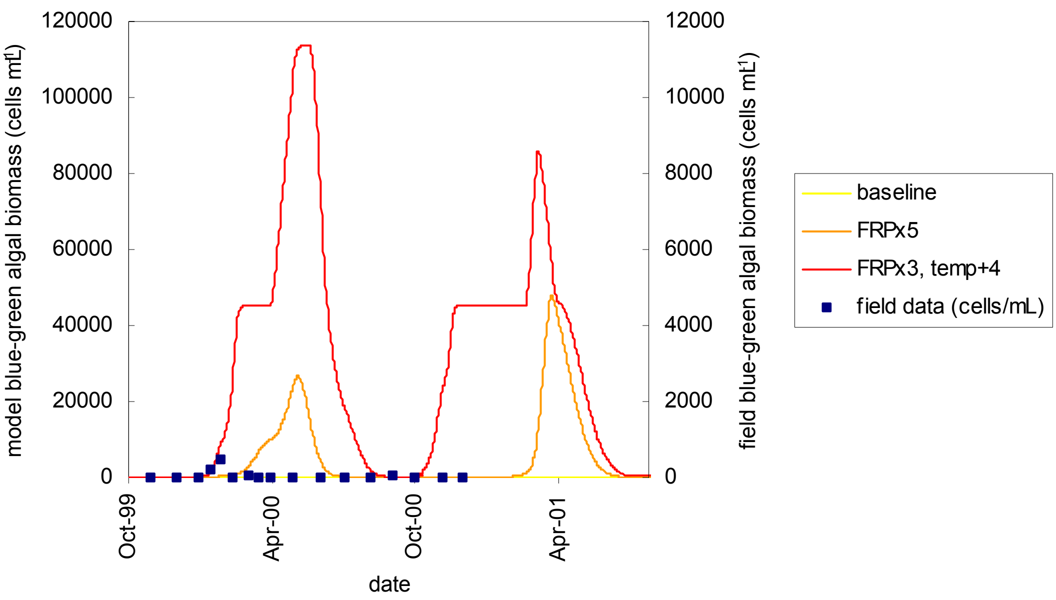

Figure 9: Conditions expected to lead to significant risk algal blooms in Lake Nagambie.

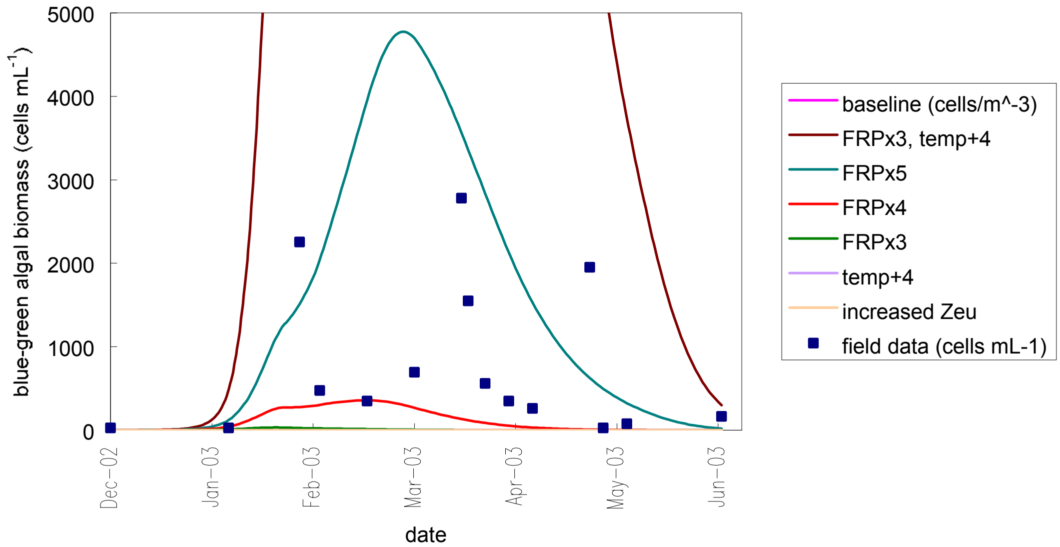

Figure 10: Modelling results for Lake Nagambie during the 2002-2003 period

4.3 Site ii: Lake Mokoan

4.3.1 Validation results

Lake Mokoan was modelled for the 2000 *Microcystis* bloom (Figure 11 and Figure 12). Temperature data were not available for earlier years, and algal data were not made available for later years during the course of this study**.** As Lake Mokoan is the primary site with significant blue-green algal blooms within this region, validation of the simple model at this site was important. Timing of the onset of the bloom in mid-January was predicted quite accurately, and the duration of the period where significant levels of blue-greens occurred was also well simulated. The magnitude of the bloom was overestimated, however, as discussed in the "risk analysis" section of this report, timing and likelihood of the single effect "algal bloom", is the main objective of this study, rather than the severity of this effect.

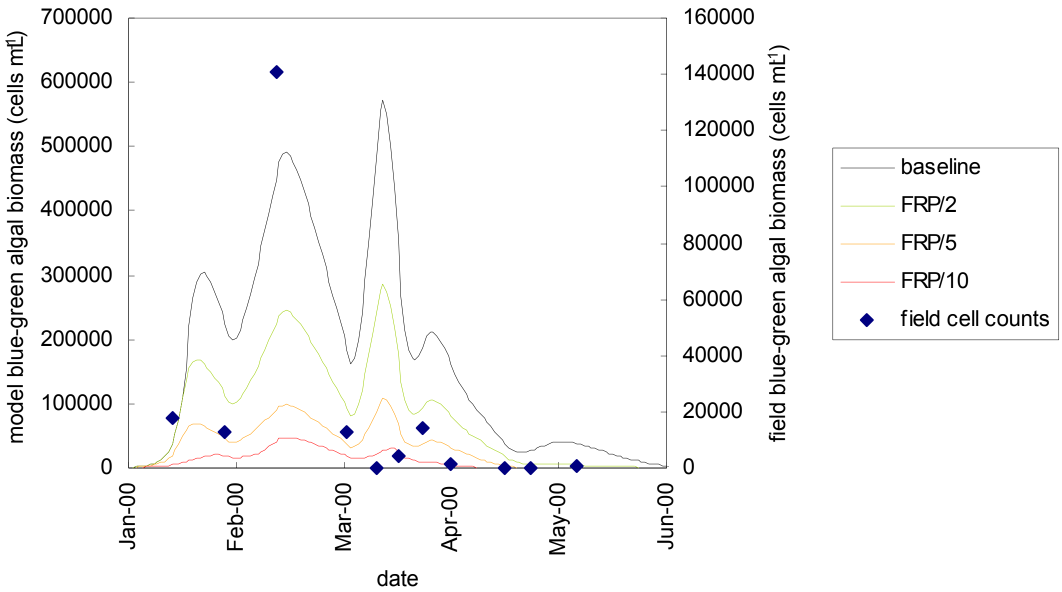

Figure 11: Lake Mokoan modelled algal biomass decreasing with decreasing fractions of field measured FRP.

4.3.2 Scenarios

Scenarios with decreased nutrients, decreased temperature, and decreased turbidity were examined. No variation in biomass was seen with decreased turbidity, whilst the effects of decreased FRP and temperature may be seen in Figure 11 and Figure 12 respectively. Decreased FRP lowers biomass, while decreased temperatures delay formation of the bloom.

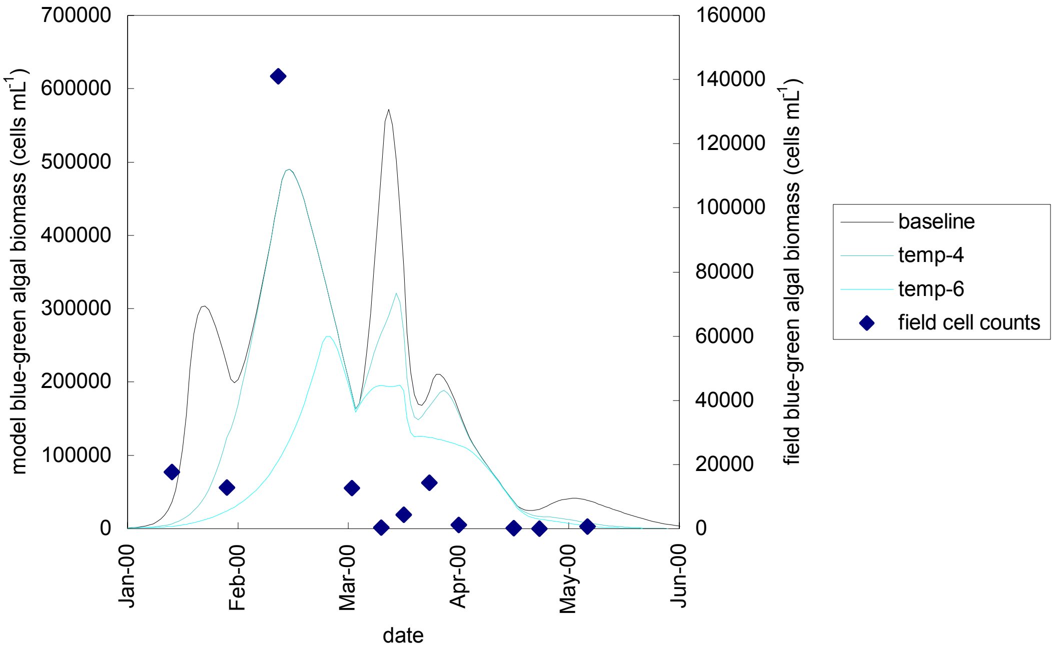

Figure 12: Lake Mokoan algal biomass decreasing with decreasing temperature

4.4 Site 2: Lower Goulburn River

As mentioned above, there are essentially no data on the abundance of cyanobacteria or the concentration of chlorophyll-a for the lower Goulburn River, but anecdotally, there have not been any noticeable bloom events within the last decade. The model results certainly re-create this low-biomass outcome for the period 1994-2003 (Figure 13), with cell concentrations not going above tens of cells per mL. The system also appears to be unusually robust to perturbations of the driving parameters, and scenario modelling (Figure 14) appears to indicate a very low risk of blue-green algal blooms. Significant changes to phosphorus and light availability, temperature and flow rate do not significantly increase the blue-green biomass, and importantly do not lead to peak biomass levels above ~100 cells mL^{-1}. The result for the flow increase scenario was unexpected. Normally, reducing the flow through the model causes an increase in cell concentrations. Why the model is behaving this way for the lower Goulburn River is not known, however given that the effect on biomass is minor, the increased risk associated with this model-uncertainty will also be minor.

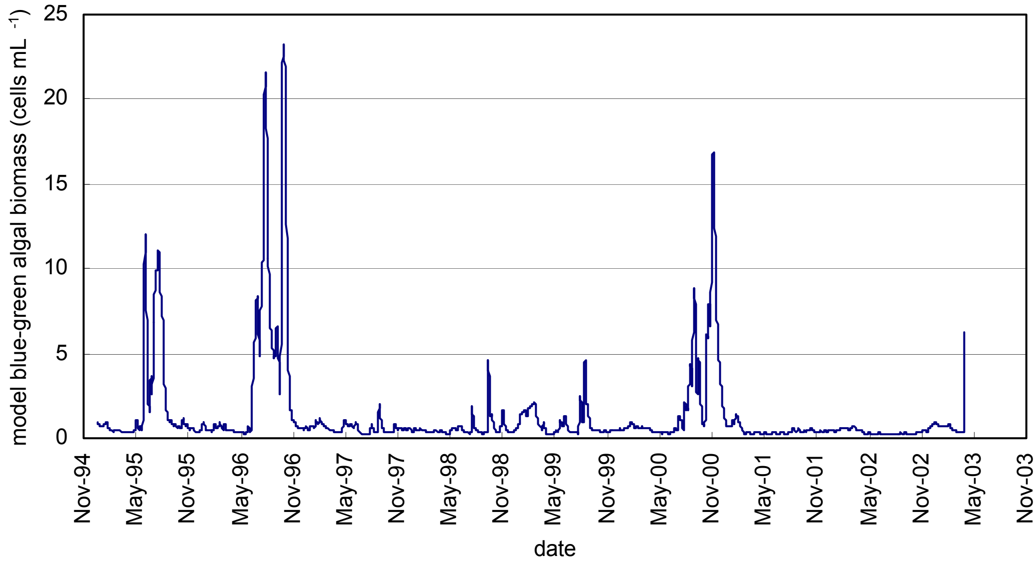

Figure 13: Lower Goulburn blue-green algal biomass modelled over time (no field data available).

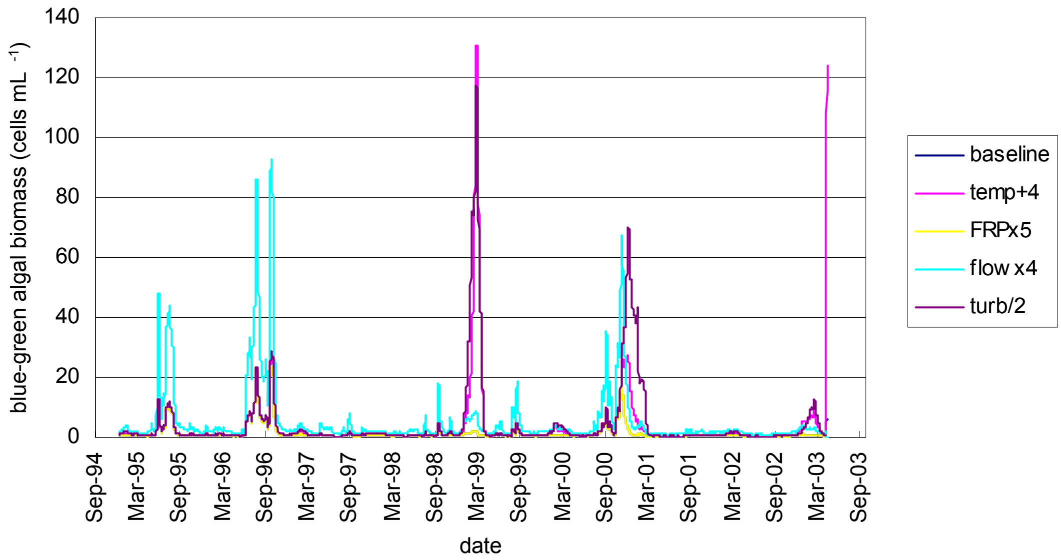

Figure 14: Lower Goulburn blue-green algal biomass scenarios.

4.5 Sensitivity

As found previously (Webb *et al.*, submitted), the model is relatively robust with respect to phytoplankton parameters. The effects of changes to the parameters was as expected, with changes in r_{20} changing peak height, changes in Q_{10} having the most effect during the highest temperatures, etc. The comprehensive sensitivity analysis conducted in Webb et al. (submitted) demonstrates that altering model parameter values, only really affects peak heights. As can be seen for many of the graphs above, there is a poor fit between modelled population heights and those recorded in the field. This is to be expected, given that the majority of analyses have been done for systems for which we have relatively poor information. What is more important is that the gross dynamics of the cyanobacterial populations (increase, decrease etc.) have been re-created with a reasonable degree of concordance with the field data. Future attempts to model cyanobacterial populations at the case study sites could invest more time in attempting to better parameterize the model for these systems, or refining equations used in the model to better reflect the processes occurring at the case-study sites. The role of phosphorus limitation, in particular, is an aspect of the model that received little critical attention when the model was first developed. This is because Bourke Weir pool is almost never subject to phosphorus limitation (Webb *et al.*, submitted). Results of this study show that phosphorus limitation is important for at least some of the sites investigated, and so further research along these lines could be profitable in terms of model performance.

5. Discussion

5.1 General

Modelling successfully replicated periods of bloom activity, however as stated above, there were discrepancies between modelled biomasses and those observed in the field. As previous studies have indicated, a number of factors may contribute to this. The importance of model input data such as water temperatures and nutrient concentrations have been outlined above. Moreover, the degree to which how well these data represent larger spatial and temporal scales, is unknown. Extrapolating point source data to large water bodies is a necessary step when modelling with sparse monitoring data, but its effect on the outcomes of modelling can be great (Kneale & Howland, 1997). Carefully designed additional field monitoring at a range of scales may help to resolve differences in measured data and field variability (Richey, Horner & Mar, 1985). The spatial heterogeneity of algal blooms is another source of uncertainty (Webster, 1990). Moreover, the sampling error for the cell counts will be great at low cell counts (Guillard, 1973). For example, Bormans and Condie (1998) assumed a ±100% error for counts of *Anabaena* that were below 1000 cells mL^{-1}.

Overall, simulations indicate the case study systems identified during the problem formulation stages, Lake Nagambie and the lower Goulburn, are relatively robust. Significant changes in physico-chemical conditions are required before cyanobacterial populations are likely to reach levels of concern in Lake Nagambie, and the lower Goulburn did not show elevated biomass in any of the scenarios modelled.

5.2 Lake Nagambie/Goulburn Weir

The primary risk factor for algal blooms in Lake Nagambie is the level of FRP. The modelling indicates FRP would have to increase substantially (more than four-fold) before algal populations might begin to reach bloom proportions. Modelling indicates FRP increases below this threshold would be a problem only if combined with a significantly elevated temperature regime (Figure 9; importantly, FRPx3, with a temperature increase of 2°C, did not result in a bloom). The lake is unlikely to experience a change in temperature regime of this magnitude. The thermal 'signature' of Lake Eildon does not extend to Lake Nagambie (C.J. Gippel, pers. comm.), and so amelioration of cold-water pollution from Eildon would not be expected to lead to elevated temperatures in Lake Nagambie. Agricultural development upstream of the lake – including irrigated agriculture – could present a risk to its nutrient status, and such scenarios should be investigated more fully if such agriculture were proposed.

It should be noted that the requisite high-nutrient, high-temperature conditions that appear conducive to blooms might sometimes be found in the shallow backwaters of Lake Nagambie. These areas may suffer from locally increased temperatures, and be subject to release of nutrients from sediments, or localized nutrient runoff events following rainfall events. Monitoring of cyanobacterial populations within the impoundment does not usually extend to these areas, and so whether or not they are more bloom-prone is not known. Moreover, there are insufficient physico-chemical monitoring data to run models of these areas, and the bathymetric data required to parameterize the models are also lacking.

As the level of light attenuation within the lake is already quite low (turbidity ~ 15 NTU), the lack of impact of decreasing turbidity was expected. However, one set of circumstances beyond the scope of the algal population model used is the interaction of *Anabaena* with other algal groups. Studies have found that elevated turbidities may encourage dominance of blue-greens over other taxa (such as green algae), due to their ability to vertically migrate from nutrient rich, hypolimnetic (near sediment) waters to the light replete surface waters (Ganf & Oliver, 1982). However, overall low levels of all algal groups in this system indicate that this is a low risk, if other factors (e.g. FRP concentrations) remain constant.

5.3 Lower Goulburn

The lower Goulburn River might be regarded as qualitatively different from the other modelled sites. It experiences much lower residence times, with high flushing effects. Significant turbulence within the water column is likely to have wide-ranging ecological effects.

There is also a greater likelihood that this area will experience changes to the nutrient and light regimes, due to the effects of irrigation returns entering the Goulburn River from the Rodney main drain and other irrigation return drains.

Irrigation channels connected to the river are likely to stagnate occasionally, and experience somewhat different conditions. This phenomenon has been observed in monitoring data for the Rodney main drain. Although there is no reversal of seasonal flow conditions via irrigation, if this were to occur, it would be likely to increase resistance of the system to algal blooms. Higher flushing in summer and lowered residence times in this reach would act to reduce the risk of blooms during the warmer months. Lowered flow in winter is unlikely to result in blooms, due to lower temperature during this period, and the lowered availability of light caused by shorter day lengths and cloudier skies. The irrigation returns that currently enter the system during summer appear to add significant amounts of nutrients to the system (Webb *et al.*, 2003), which may increase the risk of blooms. However, the vast majority of nutrients are not in a form that is likely to be readily bioavailable (reactive phosphorus comprises ~10% of total phosphorus, nitrates and nitrites ~20% of total Kjeldahl nitrogen; Webb *et al.*, 2003). This lack of availability is probably a major reason why blooms generally do not occur within the river. Increased suspended solids and increased salinity (resulting from irrigation returns) are likely to decrease risk of blooms, as they lead to a reduction in light availability, and direct toxicity due to the low salinity tolerances for *Anabaena* and *Microcystis* (Robson & Hamilton, 2003), although increased salinity will tend to enhance particle aggregation/settling and lead to more rapid water column clarification.

Modelling results support the above discussion. The modelled population remained very low for the baseline scenario, and was not really affected by any of the scenarios designed to increase the likelihood of blooms within the model. Substantial increases in FRP levels and temperature, and reductions of turbidity all influenced the population projection, but the cell counts were still uniformly low. Increasing flow through the system increased the biomass projected by the model, but again only by a minor amount. This was an unexpected result from the model, given that increased flow usually leads to decreased biomass projections. It may be that because of the very different conditions experienced in the lower Goulburn River when compared to the other sites, some part of the model is dysfunctional this site. Unfortunately, data required to parameterize the model for the lower Goulburn River was not supplied until very late in the project. Thus, when the unexpected model behaviour was identified, there was insufficient time to experiment with model structure to see if the output could be improved. As a result, we do now know which component/s of the model is/are not working properly, and it would require additional fieldwork to characterize the physical and chemical environment of the lower Goulburn River in order to identify where the model is lacking. Such work is beyond the scope of the current project, but may be considered as a knowledge gap that should be filled by future investigations if deemed to be important enough. This last qualification is important. Because the lower Goulburn is not subject to regular bloom events, and because other scenarios do not appear to increase the risk of a bloom substantially, it may be decided that the expenditure required to explain a relatively minor failure in model behaviour is not justified.

5.4 The Model

The lack of appropriate validation data, in the form of actual algal blooms in Lake Nagambie and the lower Goulburn River, adds considerable uncertainty to the results of the models. Although proxy systems were available, site specificity limits how far these may be used. In addition, even the proxy sites (the East Goulburn channel in particular) had far less data than required to properly validate the model.

An additional caveat was the dominance of *Microcystis* rather than *Anabaena* within Lake Mokoan. Differences in the biology of these taxa may mean significant differences in final bloom-forming dynamics. In particular, as mentioned above, *Microcystis* is not able to fix atmospheric nitrogen. Therefore, in applying this model to Lake Mokoan, we are assuming that adequate nitrogen will be available for cell growth. Re-designing the model to include nitrogen limitation for lake Mokoan was not justified within the scope of this study, given that Lake Mokoan was never meant to be the subject of an ecological risk assessment. However, the results of the simulation appear to indicate that the model may be useful for the broad simulation of blooms of taxa other than *Anabaena.* In addition, Lake Mokoan experiences much higher turbidities than are seen in the other systems

The differences between the lower Goulburn River and the other sites at which this model has been applied, coupled with the unusual behaviour of the model under a flow manipulation scenario, implies that caution should be used when interpreting results. However, it is likely that the primary difference of high flows and high turbulence at this site means that predictions by the model are likely to overestimate the risks of blooms, since blue-green algal species generally favour more lentic environments.

Perhaps the biggest shortcoming of the model used, from the point of view of quantitative ecological risk assessment, is that it is deterministic rather than stochastic. As a result, given a certain set of environmental conditions, the same algal population trace will always be produced. The consequence of this model behaviour is that it does not lead to an estimate the likelihood of an algal bloom given a certain combination of driving data – the model says that either a bloom will happen, or it will not. However, given the amount of uncertainty associated with applying a model that has only been extensively validated for a system in northern NSW to systems that do not have a record of algal blooms, we felt that introducing stochastic components to the model, and thereby producing likelihood estimates of blooms would have been placing too much faith in the model output. As such, a qualitative discussion of the factors that may lead to increased bloom likelihood was used, rather than a quantitative estimate of bloom likelihood under various scenarios

Despite the concerns noted above, it is apparent that the formation of blue-green algal blooms can be understood and simulated in terms of well-founded physical and biological relationships that utilize a relatively small set of variables. Theoretical research into the practices of modelling has also found that simpler models may be more useful for prediction than more complex models (Scheffer & Beets, 1994; Steel, 1997), with the added benefit of requiring less input data and parameterization. The fact that the model presented in this report can be driven by standard monitoring data is of prime importance to management of these systems.

5.5 Ecological Effects Tables and the effects of Irrigation

The scenario modeling and existing knowledge of algal processes for the case study systems only allowed for the formation of qualitative ecological effects tables for the two systems of interest (Table 3 and Table 4). These present the environmental stressors of importance to risk of algal blooms in each system, along with a categorical estimate of the probability of an increase in the risk of algal blooms given the stressor, plus an estimate of the uncertainty in these findings. As stated above, the lack of validating data mean that our estimates must be considered preliminary, and thus the uncertainty is high. Moreover, we cannot place quantitative estimates on either the categorical boundaries, or the uncertainties associated with these categories. This may be considered as a major shortcoming of this risk assessment exercise. As stated in the introduction, progress in ecological risk assessment is towards more quantitative estimates of risk, with the majority of ERAs done to date being qualitative. Nevertheless, the construction of the qualitative tables below is an advance over what was previously known for the case-study systems, and clearly identifies possible avenues for further research – i.e. work that would allow quantitative estimates (including uncertainty) to be made.

Table 3: Lake Nagambie/Goulburn Weir.

Parameter	Bloom Likelihood	Uncertainty
Increased FRP	Medium	High
Increased temperature	Low	High
Decreased turbidity	Low	Low
Decreased flow	Low	High

Table 4: Lower Goulburn River – Rodney Main Drain to the Murray.

Parameter	Bloom Likelihood	Uncertainty
Increased FRP	Low	High
Increased temperature	Low	High
Decreased turbidity	Low	High
Decreased flow	Low	High
Increased salinity	Low	High

All of the parameters listed in the tables can be affected to a degree by the presence of irrigated agriculture. As has been discussed above, the primary factor that would increase the risk of algal blooms appears to be elevated levels of filterable reactive phosphorus. Phosphorus is introduced to the lower Goulburn via irrigation returns, and total phosphorus levels for this stretch of river are very high (Webb *et al.*, 2003). However, the majority of this phosphorus is not in the immediately bioavailable form (as estimated by FRP), and thus does not lead to excessive algal growth. It is important to understand why so little of the phosphorus is available as FRP, and further research in this direction may be required. As stated above, any agricultural development upstream of Lake Nagambie that acts to increase phosphorus concentrations in the lake could increase the likelihood of algal blooms. Irrigation is one such form of agriculture. Irrigation will act to reduce the total amount of water in the Lower Goulburn, and is responsible for the existence of the semi-static water body of Lake Nagambie. Both these flow-related factors increase the likelihood of bloom events, although we believe that risk associated with reduction of flows is quite minor. The seasonal reversal of flows that is seen in the Lake Nagambie system, but not in the lower Goulburn, will act to reduce the likelihood of blooms by providing more flushing flows during the warmer months of the year. The thermal signature of Lake Eildon is another major ecological effect of irrigation. However, this effect is not manifested at either of the case study sites, and would act to reduce the likelihood of blooms in any case. Similarly, increased turbidity and salinity that may result from the introduction of irrigation returns to the river would also generally act to reduce primary productivity (and hence algal blooms) in the systems, although the tendency for increased salinity to lead to water column clarification complicates this behaviour to some degree, and will probably favour cyanobacterial growth at slightly raised EC values.

5.6 Refining the Ecological Risk Assessment

As stated above, the ecological risk assessment of algal blooms for the two case-study sites is weakened by a lack of confidence in the modelled results. Principally, there is a high degree of model uncertainty. Due to a lack of validating data, in the form of records of blooms for the case-study sites, we are unable to validate the model being used. It is unfortunate, in particular, that the only elevated cell concentrations observed for the case study sites coincided with an FRP record that appeared to be in error. As a result of this lack of validation data, we do not know whether some of the parameter values that produced well-validated results for the model's initial development for the Bourke Weir Pool may not be appropriate for the case study sites. Moreover, the apparent importance of phosphorus limitation in the case-study systems means that the model output is being largely directed by a subroutine within the model that was of little importance in Bourke Weir. These concerns, notwithstanding the model does appear to recapitulate the basic shape of algal population dynamics for the two analogous systems that were modelled, even if population numbers are only poorly represented. Given this basic uncertainty, introducing stochastic elements into the model, and thereby producing quantitative likelihood estimates did not seem to be justified at this time.

Risk estimates would be improved by increased effort in the modelling of cyanobacterial populations for the case study sites. In particular, further development of the phosphorus limitation routine could be expected to lead to improved model outputs in terms of the quantitative estimates of cell concentrations. Other modifications to model structure may be required to adequately model the lower Goulburn River, which cannot really be approximated by a weir pool (an implicit assumption within the current model).

In addition to refining the models, an algal monitoring program within the Lower Goulburn River would provide at least some validating data for the models, even if the cell concentrations were consistently low. Such a monitoring program could be based on a monthly sampling protocol, with the facility for more frequent sampling if elevated cell concentrations were picked up (or if anecdotal information between monthly sampling dates seems to indicate that cell concentrations have risen).

The possibility of increased bloom likelihood within the backwaters of Lake Nagambie was also identified. In order to model the likelihood of such events, location specific monitoring for both physico-chemical driving data and cyanobacterial abundance for model validation would be required.

All of the above pre-supposes that the necessary expenditure of time and money is justified. Although the risk assessment presented in this report falls a long way short of a quantitative risk assessment with estimates of uncertainty, the modelling exercise has indicated that likelihood of algal blooms within the systems are low. These findings back up the monitoring and anecdotal information available for the two case-study sites. Thus, it would not be unreasonable to allocate funding towards assessing risks of algal blooms in other areas, or assessing the risks associated with another aspect of irrigation within the Goulburn-Broken catchment. Although we consider risk assessment to be an iterative process, the return on investment will be limited for substantial further work on an issue that has been found to pose a low environmental risk by an initial assessment. A possible exception to this argument would be if development were proposed that would substantially increase the levels of FRP in Lake Nagambie. Given the apparent increase in risk posed by such a scenario within this risk assessment, further work to refine and improve the risk estimate would be totally justified in this case.

5.7 Summary

- The two case study systems identified during the problem formulation stage of this project appear to be relatively resistant to formation of algal blooms.
- Within the Lake Nagambie system, the main factor that could increase the likelihood of bloom events appears to be an increase in the level of filterable reactive phosphorus (FRP). The model indicated an increased likelihood of blooms at a level of FRP that is still less than that commonly observed in Lake Mokoan. Thus management of nutrient inputs to the systems should be considered as a priority, although this is already acknowledged in GBCMA (2002).
- Increased temperatures will also increase the likelihood of blooms, but this parameter is effectively beyond the control of managers. Nevertheless, unusually warm summers will lead to an increased likelihood of blooms.
- No factors were identified that could significantly increase the likelihood of blooms in the Lower Goulburn River. However, a lack of validation data, and the qualitatively different nature of this system compared to the other modelled systems, means that there is a high degree of uncertainty associated with all conclusions for this site.
- The conclusions of the modelling are associated with a high level of uncertainty. This derives from three principal sources:
    1. A lack of blooms within the study systems, requiring models to be validated on analogous systems that will behave in slightly different ways.
    2. A relative paucity of data on algal blooms at the analogue sites, meaning that the models are not properly validated for these sites either.
    3. The apparent importance of phosphorus limitation within the case-study systems, whereas the system for which the model was originally developed (Bourke Weir) is almost never phosphorus limited.
- The uncertainty of the modelling results means that the ecological risk assessment can only deliver a qualitative assessment of the risks associated with various scenarios, and the uncertainty associated with these predictions.
- Despite the uncertainty associated with the biomass of algae produced, the models were able to recapitulate monitored biomass dynamics with a reasonable level of concordance.

5.8 Recommendations

- Monitoring of the backwaters in Lake Nagambie will be of use in quantifying additional risks of blooms from heterogeneity and localised conditions. Such a monitoring program could be based upon the one that already exists within Goulburn Weir itself.
- Any proposed agricultural development upstream of the Lake Nagambie impoundment should be assessed in terms of its likely effects on nutrients within the lake.
- For the lower Goulburn River, basic information on the river channel, and the physico-chemical conditions within the river stretch would improve our confidence in the modelled results, or direct us to modify the model appropriately.
- Similarly, a basic algal monitoring program would provide better validation of the modelled results. Given the low risks identified, this program should not need to take samples more than monthly, with the caveat that more frequent sampling occur when elevated cell concentrations are observed in the river (either anecdotally or by the sampling program).
- In order to proceed from this qualitative risk assessment to a fully quantified risk assessment, further research is required into the processes governing algal dynamics within the case study systems. In particular, further development of the phosphorus limitation routine within the cyanobacterial population model is likely to lead to improved simulations of cyanobacterial populations. Once confidence in the output of the deterministic models has been improved, introducing stochastic elements to the models will allow a quantitative estimate of the likelihood of bloom occurrence.
- A decision as to whether to proceed with the above work should be made in light of the low algal abundances in the system, and the low apparent risks of blooms under various perturbation scenarios that have been identified in this report.

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