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Use of flow management to mitigate cyanobacterial blooms in the Lower Darling River, Australia

SIMON M. MITROVIC [1,2]*, LORRAINE HARDWICK [1] AND FOROUGH DORANI [1]

[1]NSW OFFICE OF WATER, PO BOX 3720, PARRAMATTA 2124, AUSTRALIA AND [2]CENTRE FOR ENVIRONMENTAL SUSTAINABILITY, DEPARTMENT OF ENVIRONMENTAL SCIENCES, UNIVERSITY OF TECHNOLOGY, SYDNEY PO BOX 123, BROADWAY, NSW 2007, AUSTRALIA

*CORRESPONDING AUTHOR: simon.mitrovic@water.nsw.gov.au

The growth of planktonic cyanobacteria in a weir pool on the Lower Darling River, Australia, downstream of the major regulated Menindee Lake system was examined. Blooms of the saxitoxin producing freshwater cyanobacterium *Anabaena circinalis* occurred for two summers out of four studied. Large cell numbers of other cyanobacteria including *Aphanizomenon*, *Planktolyngbya* and *Merismopedia* also occurred during the same summer periods as the *Anabaena* blooms. The growth events also coincided with periods of improved light climate. Flow releases from the regulated Menindee Lakes System were assessed for their ability to either suppress bloom development or to mitigate pre-existing blooms over this period. A discharge of 300 ML/day (flow velocity of 0.03 m/s) was found to be sufficient to prevent prolonged periods of persistent thermal stratification, which also suppressed the development of *A. circinalis* blooms. A flow release of 3000 ML/day was effective at removing an established cyanobacterial bloom, and total cyanobacterial numbers declined from over 100 000 to <1000 cells/mL within a week. In two summers without blooms, higher flows and decreased light availability prevented the development of cyanobacterial blooms. Flow releases were effective at mitigating cyanobacterial growth through either the suppression of persistent thermal stratification or through dilution and translocation of cells. Greater discharges also increased turbidity, which diminished the growth of cyanobacteria through reduced light availability under the mixed conditions, which also reduced the ability for surface migration through buoyancy regulation. The volume of water required for different management strategies varied and is considered in terms of environmental allocations.

KEYWORDS: flow management; environmental flows; cyanobacteria; *Anabaena circinalis*

INTRODUCTION

Cyanobacterial blooms, and in particular toxic blooms, are a major problem in many rivers around the world. Discharge (Marshall and Burchardt, 1998; Ha *et al.*, 1999), flow velocity (Lung and Paerl, 1988) and turbulence (Steinberg and Hartman, 1988; Eldridge and Sieracki, 1993) have been identified as important factors influencing the development of cyanobacterial blooms.

Cyanobacteria are generally advantaged under scenarios of reduced discharge and flow velocity due to increased retention time and decreased washout of cells (Oliver and Ganf, 2000). These conditions can help compensate for the uncompetitive growth rates of *A. circinalis* in comparison to many chlorophytes and diatoms, due to its colonial habit diminishing nutrient acquisition efficiency (Reynolds, 1994) and for the high energy requirements of nitrogen fixation (Smith, 1990). Water column stability may also favour the buoyancy regulating gas vacuolate species which may attain a competitive advantage over other phytoplankton through vertical migration to gain increased quanta of light relative to non-buoyant or non-motile species (Walsby *et al.*, 1997; Oliver and Ganf, 2000; Mitrovic *et al.*, 2001).

In lowland rivers of south-eastern Australia such as the Barwon-Darling (Oliver *et al.*, 2000; Mitrovic *et al.*, 2003), the Murray (Bormans *et al.*, 1997) and the Murrumbidgee (Webster *et al.*, 1995; Sherman *et al.*, 1998), the proliferation of *Anabaena* and other cyanobacteria has been linked to low discharge. Where this occurs, there is scope for flow management to be used to manage nuisance blooms. The manipulation of discharge is one of the few ways in which phytoplankton growth can be controlled in a river (Webster *et al.*, 2000). One approach is to suppress the formation of blooms by maintaining a minimum flow velocity that will prevent the occurrence of nuisance species (Mitrovic *et al.*, 2003). Another approach is to release water from an upstream storage to flush and dilute a downstream bloom after its development (Berger *et al.*, 2008). This may either move the bloom downstream or may change the conditions in the river so that these are no longer conducive to the growth of the problem species. There is a need to test the effectiveness of flow management strategies to determine which can be successfully implemented under various river conditions and in different locations (Webster *et al.*, 2000; Berger *et al.*, 2008).

The Lower Darling River in the semi-arid south-west region of New South Wales, Australia, is a lowland river regulated by the large Menindee Lakes system. The Menindee Lakes Storage Scheme, completed in 1968, regulates a series of terminal lunette lakes formed beside the Lower Darling River. The regulated lakes capture water passing down the Barwon-Darling River, storing it in a large lake on the main river (Lake Wetherell). Water can then be further stored into three other lakes (Pamamaroo, Menindee and Cawndilla). The Lakes can store 1794 GL and release water downstream to the Lower Darling River and the Great Darling Anabranch. Due to the sporadic nature of inflow events to the lakes, the volume stored is usually much lower than this with a mean supply volume of

approximately 50%. As a result, during droughts, water for release can be in short supply.

Past phytoplankton sampling of the Lower Darling River has detected blooms of cyanobacteria (Hötzel and Croome, 1994), including *Anabaena circinalis* Rabenhorst ex Bornet and Flahault 1888 (recently renamed *Dolichospermum circinalis*—see Wacklin *et al.*, 2009). This species is known to produce the neurotoxin saxitoxin and its analogues and is therefore a risk to drinking water supplies for both human consumption and stock watering (Humpage *et al.*, 1994). Recreational swimming and fishing also occur along the river and are disrupted when blooms occur.

Due to high levels of extraction of water for irrigation, the New South Wales State Government has developed water sharing rules to reduce impacts on river ecology. Environmental flows have been provided to many rivers in Australia for purposes such as wetland replenishment, facilitation of fish and bird breeding and the control of cyanobacterial blooms (Chessman, 2003). Environmental flows have been allocated from Menindee Lakes to manage problematic cyanobacterial blooms in the Lower Darling River. A water reserve of 30 GL has been allocated each year that can be used when blooms are detected. The water is released to disperse and remove blooms and create conditions that are less conducive to their growth. However, the causes of *A. circinalis* blooms in the river and the effectiveness of the flow management strategy are currently unknown.

This study builds on a simple flow and stratification model developed for *Anabaena circinlais* growth in the unregulated Barwon-Darling River (Mitrovic *et al.*, 2006). Here we examine whether this model holds for the Lower Darling River and extend it to include light availability. The factors that led to the development of blooms of other cyanobacteria and their applicability to flow management were also examined. Importantly, this study fulfils a research gap in assessing the effectiveness of flow releases in mitigating *Anabaena* and other problematic cyanobacterial blooms through suppression of growth or flushing flow releases. Strategies for the best use of the available environmental water allocations for bloom mitigation are also discussed.

METHOD

Study area

The Lower Darling River, New South Wales, Australia runs for ∼300 km from the Menindee Lakes to its confluence with the Murray River at Wentworth (Fig. 1). The river has several weirs along its length, usually near
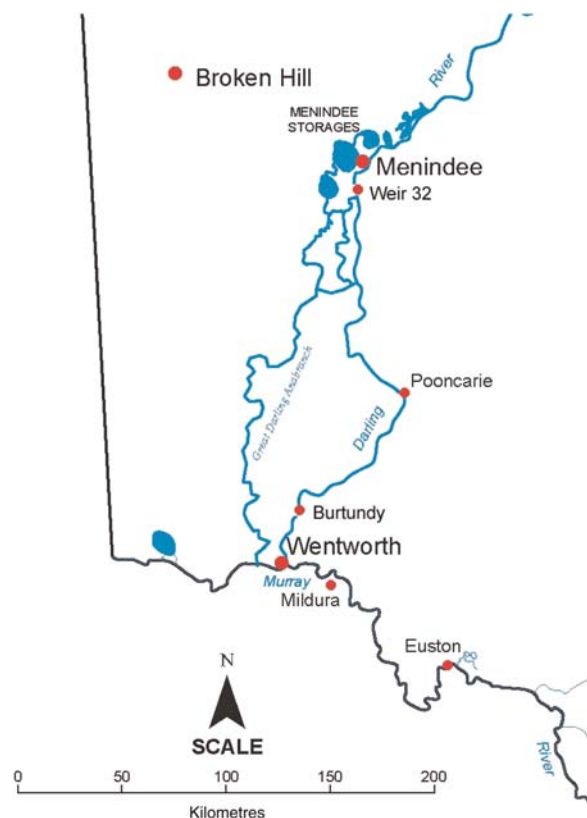
Fig. 1. Location of Weir 32 study site on the Lower Darling River, NSW, Australia.

towns and homesteads, to ensure water availability during low flow periods. The weirs are simple walls that impound a volume of water and cannot be used to regulate flows downstream. They are relatively small and store water within the confines of the river channel, forming weir pools that stretch for ca. 30–50 km upstream. All the weirs discharge by overflow whenever the river level exceeds the weir height.

The site examined in the study was within Weir 32, the weir pool immediately downstream of Menindee Lakes (Fig. 1). Data from other downstream weir pool sites of Pooncarie and Burtundy were also used in developing z_{eu} and turbidity relationships.

Sampling and analysis

Data were collected from September 2005 to May 2009 covering four summer periods. Discharge data were available for the site from routine hydrographic records maintained by the New South Wales Office of Water. Readings are presented as mean daily discharge. Samples for algal enumeration were taken weekly during summer and fortnightly otherwise from the shore using a 3 m long sampling pole with samples

taken 25 cm below the water surface and preserved with Lugol's iodine solution. These samples were subsequently counted using a calibrated Lund cell and compound microscope after concentration by sedimentation in a measuring cylinder (APHA, 1998). Counting precision was $\pm 20\%$ (Hötzel and Croome, 1999). Phytoplankton were identified at genus level using the keys of Prescott (Prescott, 1978), except for the potentially toxic taxa which were identified at species level (Baker, 1990). Samples for turbidity were also taken from 25 cm below the water surface and were immediately analysed using a calibrated HACH nephelometer (APHA, 1998). Samples for nutrient analyses were collected in the same way then immediately filtered through 0.45 µm cellulose acetate membrane filters and kept cool ($<2°C$) until frozen and transported to the laboratory (within 10 h). Samples were analysed for filterable reactive phosphorus (FRP) and oxidized nitrogen (NOx–N). The FRP subsamples were analysed by the ascorbic acid method and the NOx–N subsamples by the automated cadmium reduction method following standard methods (APHA, 1998).

Water temperature, electrical conductivity, pH and dissolved oxygen were measured *in situ* using a precalibrated Hydrolab MS5 water quality probe and Surveyer 4A. Water transparency was measured using a white 20 cm diameter Secchi disk lowered on the unshaded side of a boat. On select quarterly field trips, light penetration and euphotic depth (z_{eu}) were determined from a boat-anchored mid-stream using a Licor LI-185B quantameter coupled to an underwater quantum sensor and reference sensor in air. Measurements of downwelling irradiance were recorded at 10 cm intervals down the water column (Kirk, 1977). The euphotic depth was determined as 1% of the incident surface irradiance (I_o). For concurrent measurements of the euphotic depth and turbidity, the relationship between them was determined to infer euphotic depth for the greater turbidity data set. These data were used to determine the euphotic depth to mixed depth ratio. Two chains of recording thermistors (HOBO^{TM}) were placed at the sites for the duration of the study and recorded temperature ($0.2°C$ detection limit) at approximate depths of surface, 0.4, 1.0, 1.8, 2.6, 3.5 and 4.35 m at 30 min intervals.

Estimation of flow velocities for growth of *A. circinalis*

Fifteen cross-sectional profiles at 0.2 km intervals were measured at the site. Survey sections were referenced to a starting point and to AHD (Australian Height Datum) where possible. Cross-sections were performed

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at between 1 and 2 m from bank height across the river. Pegs marked each cross-section. Survey was performed by Leica Total Station Theodolite Survey methods. Cross-sectional areas were calculated at the river heights equivalent to the critical discharge required to stop the formation of persistent stratification. The average cross-sectional areas were used to convert this critical discharge to mean velocities with the equation $V = Q/A$, where V is the velocity in ms^{-1}, Q the discharge in $\mathrm{m}^3\,\mathrm{s}^{-1}$ and A the cross-sectional area in m^2.

Data analysis

Correlation analysis was applied to the physico-chemical and phytoplankton abundance data. Significant correlations ($P < 0.05$) were analysed by regression to find the line of best fit. The phytoplankton and environmental data were subjected to canonical correspondence analysis (CCA) with the CANOCO program (Ter Braak, 1992). Phytoplankton cell counts were transformed prior to analysis as ln (cells/mL + 1), and environmental variables with a large variance were also converted to logarithms. Forward selection of environmental variables was used to obtain a set maximally related to phytoplankton species. CCA results were displayed as correlation bi-plots from the covariance matrices. The analysis ordinates the species data on axes that are constrained to be linear combinations of the environmental variables. This provides a graphical representation of the relationships between phytoplankton species and the environmental variables measured. Significance was tested with the Monte Carlo permutation test with 499 permutations and α of 0.05.

RESULTS

Bloom occurrence, discharge and thermal stratification

During late 2005 and early 2006, while discharge generally remained around 400 ML/day, only small numbers of *A. circinalis* (<2000 cells/mL) were present (Fig. 2). During this period, diel stratification was
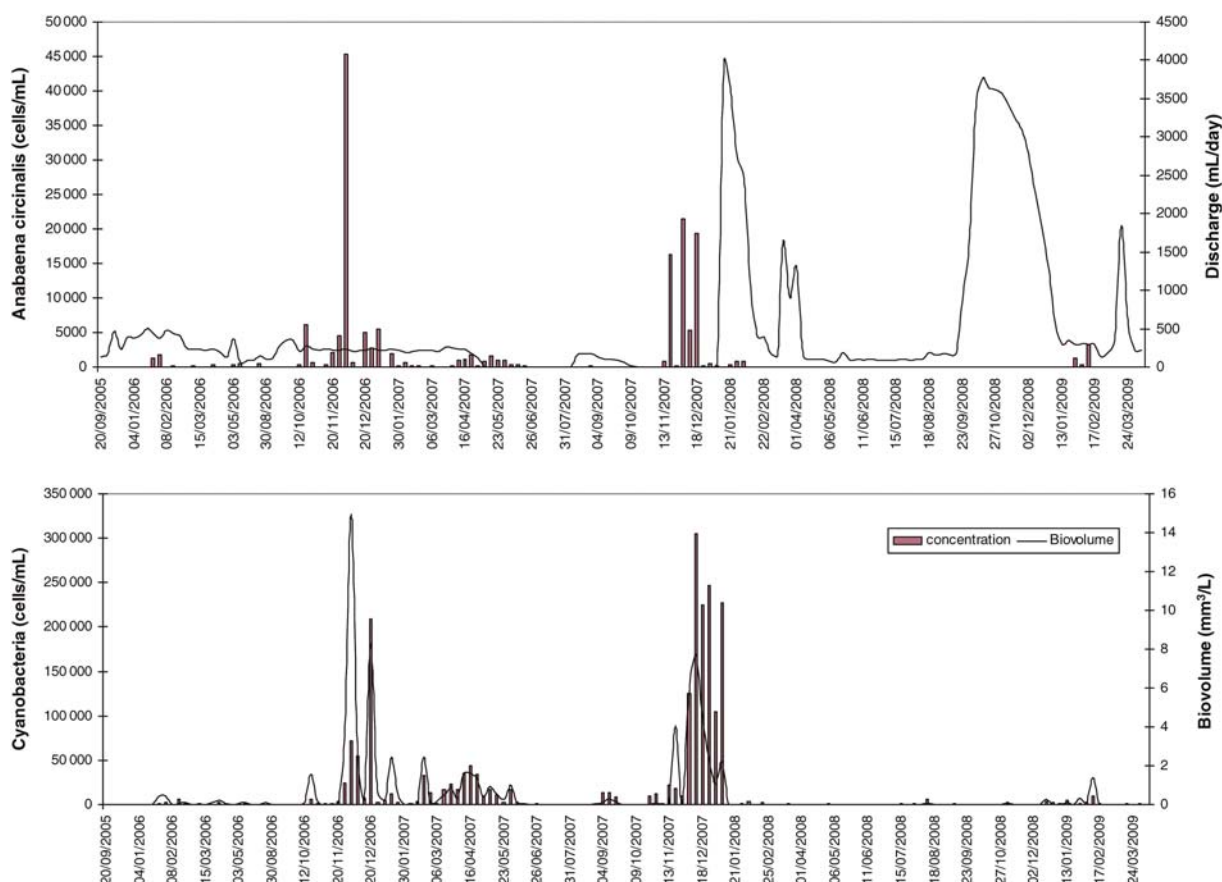
Fig. 2. *Anabaena circinalis* numbers (bars) and discharge (top) and total cyanobacterial numbers and biovolume (bottom) in the Lower Darling River at Weir 32 over 2005 to 2009.

dominant, with frequent overnight mixing and only short periods of a few days of continual thermal stratification occurring (Fig. 3). As discharge decreased after 20 February 2006, thermal stratification persisted for more extended periods. However, mixing still occurred frequently during this period. During this summer, total cyanobacterial concentration, dominated by the genera *Aphanocapsa* and *Oscillatoria*, remained low at <7000 cells/mL with a biovolume of less than 0.05 mm^3/L (Fig. 2).

A bloom of *A. circinalis* occurred during November and December 2006 reaching a density of over 40 000 cells/mL (Fig. 2). Flows decreased prior to this bloom, from 450 ML/day in early October to <275 ML/day for most of November and December. During November and December, periods of thermal stratification persisted for up to 15 days at a time without overnight mixing (Fig. 4). Persistent stratification recommenced after a brief mixing event on the 25 to 27 November. This period coincided with the greatest density of *A. circinalis* of 45 000 cells/mL (Fig. 2). Although some further overnight mixing occurred disrupting persistent stratification around the 9 December 2006 and 22 December 2006 (Fig. 4), *A. circinalis* concentrations of between 2000 and 5000 cells/mL occurred through till the end of January 2007 (Fig. 2). The cyanobacterial community, although dominated in early summer by *Anabaena*, was dominated by *Planktolyngbya*, *Merismopedia* and *Aphanocapsa* from late December onwards and numbers exceeded 200 000 cells/mL with a biovolume of up to 14 mm^3/L (Fig. 2). Although numbers subsided briefly in January 2007, numbers of *Planktolyngbya* and *Merismopedia* remained above 20 000 cells/mL through to the end of April.

Another bloom of *A. circinalis* occurred during November and December 2007, reaching a cell density of ~20 000 cells/mL (Fig. 2). This growth coincided with very low discharges which were at zero for much of the time and with river height below weir height. During this period, persistent stratification started from 14 October 2007 and was maintained, apart from several brief mixing events, until 9 January 2008. The most notable of these mixing events occurred over 6 days between 22 and 28 December 2007 (Fig. 5). A decrease in *A. circinalis* numbers coincided with this mixing event (Fig. 2). Increased flows arrived into the weir on 6 January 2008 (Fig. 2) and by 9 January 2008 thermal stratification had broken down and the weir pool was isothermal (Fig. 5). The total cell numbers of cyanobacteria were often in excess of 200 000 cells/mL with a biovolume >7 mm^3/L through much of this summer period (Fig. 2) and comprised, along with the *Anabaena*, the genera *Planktolyngbya*, *Aphanizomenon*, *Merismopedia* and *Aphanocapsa*. The flow release from Menindee Lakes in early January 2008 reduced the
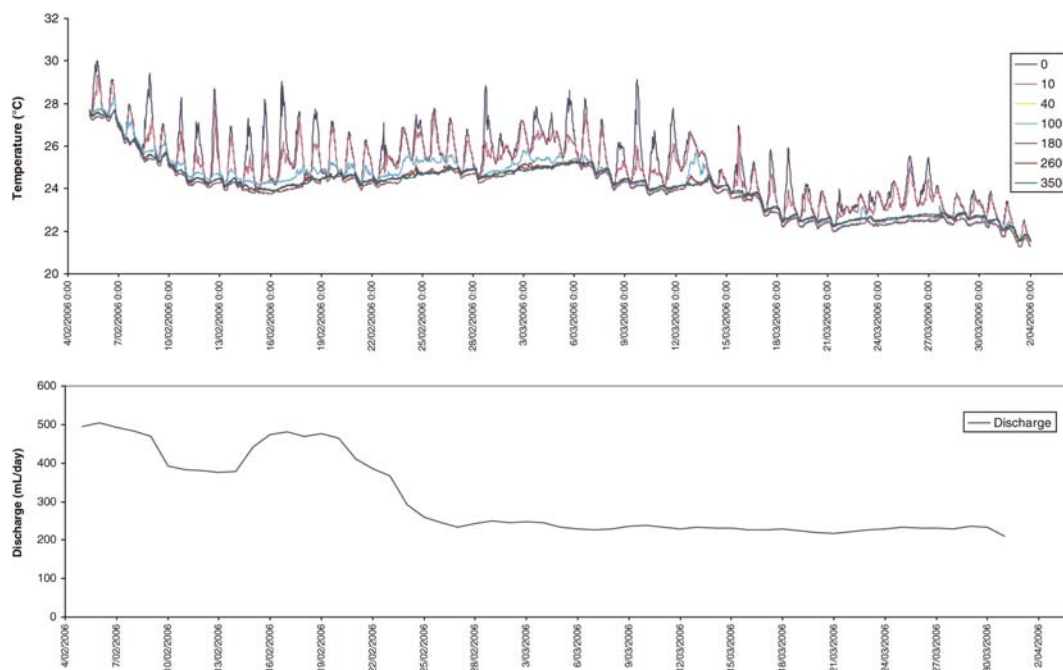
Fig. 3. Thermal stratification (top) and discharge (bottom) in the Lower Darling River at Weir 32 from 5 February until 24 April 2006. Legend represents approximate thermistor depths in centimetre.
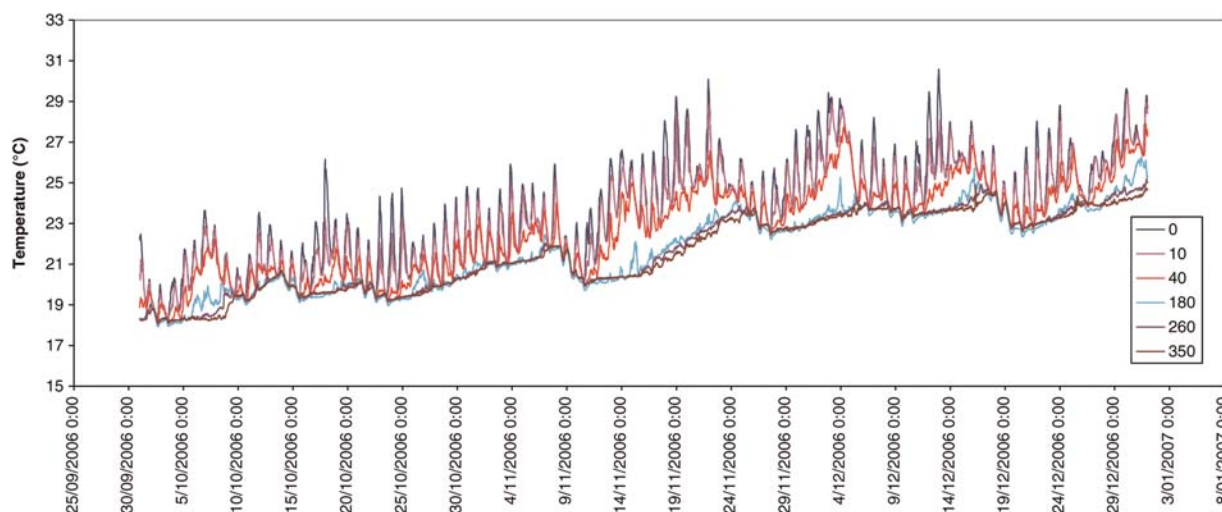
Fig. 4. Thermal stratification in the Lower Darling River at Weir 32 from 1 October 2006 until 31 December 2006. Legend represents approximate thermistor depths in centimetre.

numbers of all cyanobacteria to very low levels (<1000 cells/mL) within a week (Fig. 2) probably due to dilution and cell wash out from the weir pool.

The summer of 2008/2009 was characterized by higher discharge of ~3500 ML/day in September 2008, which decreased steadily to below 300 ML/day during February 2009. No *A. circinalis* blooms occurred during this period (Fig. 2). As discharge dropped to below 300 ML/day, a small increase in *A. circinalis* cell numbers to ~2000 cells/mL occurred as persistent thermal stratification again commenced after 9 February 2009 (Fig. 6). The total number of cyanobacteria also remained low during this entire summer period and did

not generally exceed 5000 cells/mL with a biovolume less than 1.5 mm^3/L (Fig. 2). *Planktolyngbya, Microcystis, Aphanocaspa* and species of *Anabaena* other than *A. circinalis* were dominant.

Persistent thermal stratification and discharge

As discharge decreased, the degree of thermal stratification increased. Figure 7 shows the relationship between discharge and the maximum daily thermal stratification found for several summer stratification periods. As discharge reduced to below 500 ML/day, the differential
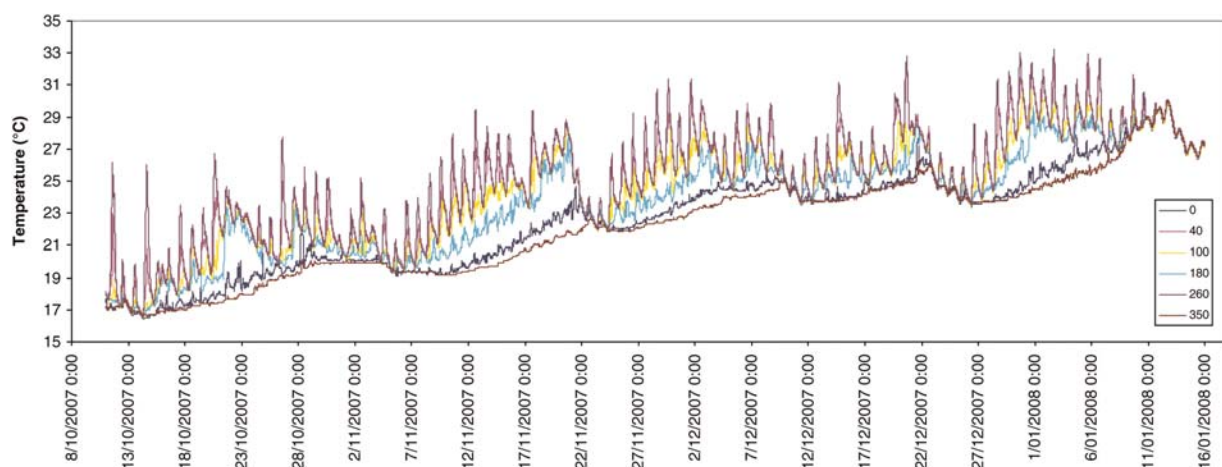
Fig. 5. Thermal stratification in the Lower Darling River at Weir 32 from 11 October 2007 until 16 January 2008. Legend represents approximate thermistor depths in centimetre.
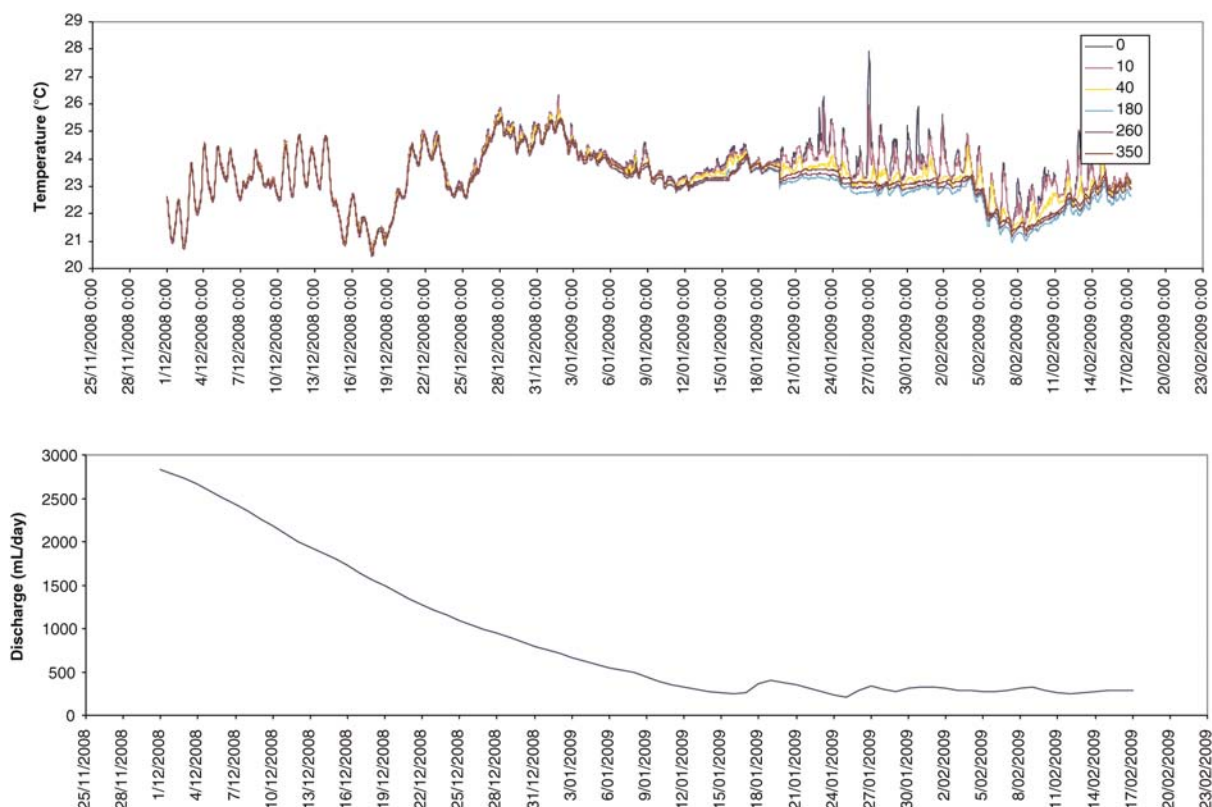
Fig. 6. Thermal stratification (top) and discharge (bottom) in the Lower Darling River at Weir 32 from 1 December 2008 to 17 February 2009. Legend represents approximate thermistor depths in centimetre.

increases substantially to about 2.5°C and below 300 ML/day is greater than 5°C. Periods of persistent stratification extended for longer periods (weeks) at discharges below 300 ML/day where the temperature differential approached about 5°C. Discharge above this did not allow extended periods (>4 or 5 days) of persistent stratification to occur even during the hottest parts of the year (Figs 3 and 6). Diel thermal stratification occurred at other times of the year when discharge was below 300 ML/day, although periods of a few days of persistent thermal stratification could occur at times. The mean cross-sectional area was 114 m^2 based on 15 profiles. Using this and the critical discharge of 300 ML/day gave a critical flow velocity of 0.03 m/s to suppress the formation of persistent thermal stratification in the weir pool.

Light climate and cyanobacteria

z_{eu} values and turbidity were measured concurrently on at least 10 sampling occasions. z_{eu} ranged between 0.5 and 3.5 m, and turbidity from 10 to 400 NTU. To be able to utilize the greater number of turbidity data collected during the study, the linear
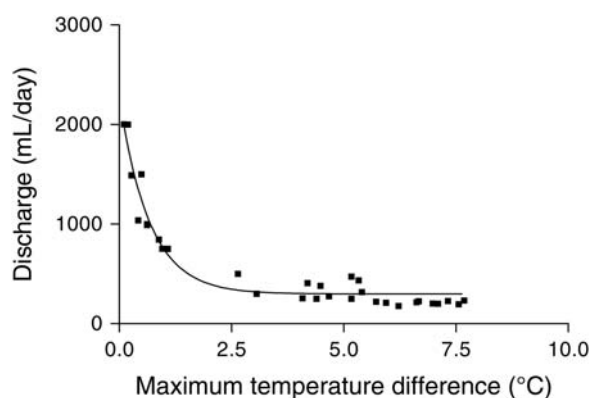
Fig. 7. Relationship between discharge and maximum daily change in temperature stratification from surface waters to bottom waters.

relationship between z_{eu} and turbidity was determined (Fig. 8) with the equation $z_{eu} = -1.97*\log\text{Turbidity} + 5.11$ ($r^2 = 0.94$) describing the relationship. This expression was used to estimate z_{eu} from turbidity measurements gathered throughout the project. Figure 9 shows the z_{eu} to z_{mix} ratio and turbidity during the study. z_{mix} was taken as the depth of the water column and did not consider periods of

persistent thermal stratification when mixing depth may have been shallower. It is likely that the mixing depth would be considerably reduced during these periods, leading to a higher proportion of time spent in the euphotic zone by the phytoplankton. However, during these stratified periods, the Z_{eu} to Z_{mix} ratio was considerably greater relative to other times, indicating an improved light climate without taking the stratified mixing depth into consideration.

The 2005/2006 summer was characterized by a diminished light climate with a Z_{eu} to Z_{mix} ratio of less than 0.3 (Fig. 9) and low numbers of both total cyanobacteria and *A. circinalis* (Fig. 2). The *A. circinalis* bloom of 2006/2007 coincided with a Z_{eu} to Z_{mix} ratio of 0.5 to 0.6 indicating a considerably improved light climate. This light climate was not only suitable for the growth of *Anabaena*, but high cell numbers of other cyanobacterial taxa were also present. As low discharge remained throughout the summer, the light climate did not change greatly and cyanobacterial cell

numbers remained high until a gradual decline in late April.

Very low and zero discharge in early November 2007 combined with a more stable water column led to an improved light climate with a Z_{eu} to Z_{mix} ratio increasing from 0.4 in October to close to 1 in December (Fig. 9). This coincided with an *A. circinalis* bloom and high cell concentrations of other cyanobacteria. As a flow event entered the weir pool in late December, turbidity increased to over 500 NTU and the Z_{eu} to Z_{mix} ratio dropped to below 0.1. In the summer of 2008/2009, flows were kept high and the Z_{eu} to Z_{mix} ratio remained at less than 0.2 and cyanobacterial numbers remained low throughout this period. In early February, the Z_{eu} to Z_{mix} ratio increased to close to 0.4 and this coincided with a small increase in *A. circinalis* concentration.

Other factors influencing cyanobacteria

Table I shows a summary of the nutrient and water quality data for the study. Mean filtered oxidized nitrogen and reactive phosphorus levels were at 0.11 and 0.12 mg/L, respectively. Mean total nitrogen and total phosphorus levels were 1.14 and 0.21 mg/L. Mixing events that occurred after extended periods of persistent thermal stratification such as the 28 November 2006, 27 November 2007 and 12 December 2007 were not found to replenish nutrient supply from sediment sources within the weekly sampling regime of this study. CCA of the cyanobacterial taxa with the environmental variables explained 61% of the variability. All environmental variables had a significant influence on the taxa ($P < 0.05$; Monte Carlo permutation test) with turbidity having the most influence followed by discharge, water temperature, inorganic phosphorus and inorganic nitrogen (Fig. 10). Of the species that were found to
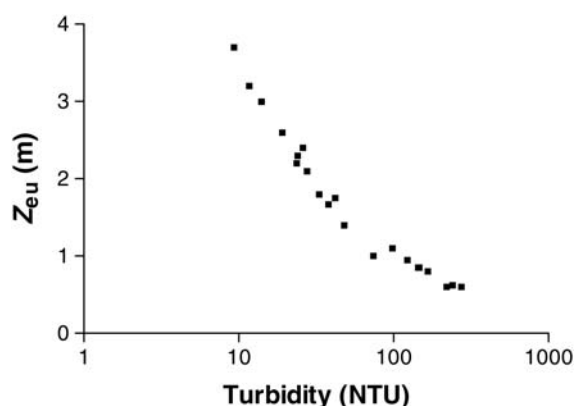

Fig. 8. Relationship between turbidity and euphotic depth (Z_{eu}) determined for the study. Data based on Weir 32 as well as 2 downstream weir pools.
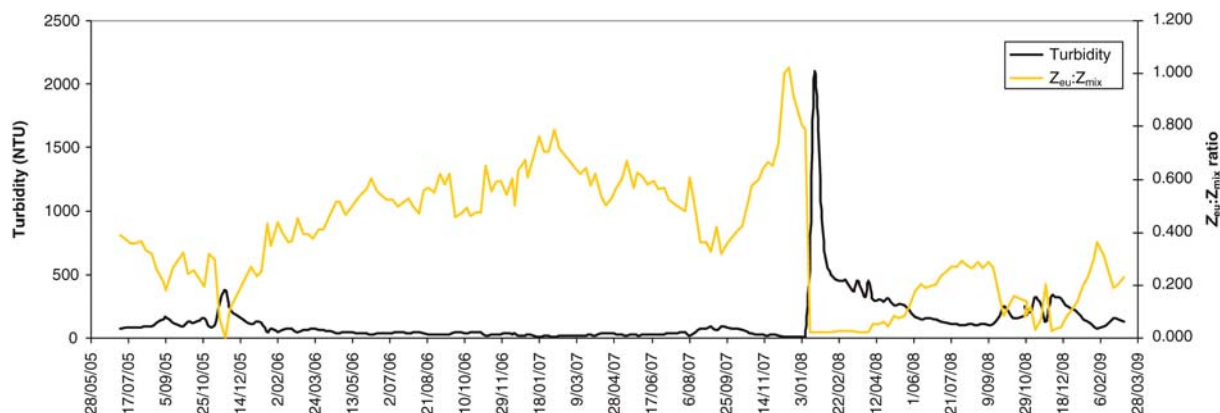

Fig. 9. Time series of changes in the Z_{eu} to Z_{mix} ratio and turbidity in the Lower Darling River at Weir 32.

Table I: Summary of water quality data for the Lower Darling River at Weir 32

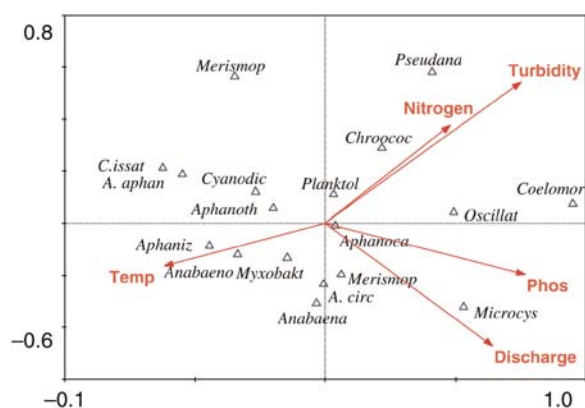|	Filtered oxidized nitrogen (mg/L)	Filtered reactive phosphorus (mg/L)	Total nitrogen (mg/L)	Total phosphorus (mg/L)	Turbidity (NTU)
Mean	0.11	0.12	1.14	0.21	140.2
10th Percentile	0.01	0.06	0.62	0.10	21.2
90th Percentile	0.39	0.18	1.86	0.35	310.0

Fig. 10. CCA ordination of environmental variables and phytoplankton taxa abundances for the Lower Darling River at Weir 32, 2005–2009. All environmental vectors were found to significantly influence the cyanobacterial community ($P < 0.05$; Monte Carlo permutation test). The length of an environmental vector indicates correlation strength and its direction indicates its relationship with the genera. Genera that lie in the same direction as an environmental vector are positively correlated with that variable and genera in the opposite direction are negatively correlated. Genera at right angles to a vector are not correlated with that variable. Environmental vectors are Discharge (mL/day), Temp (water temperature, °C), Turbidity (Turbidity, NTU), N (oxidized nitrogen, mg/L), P (filterable reactive phosphorus, mg/L). Taxa are *Anabaena* (*Anabaena* other species), *S. Aphan* (*Sphaerospermum aphanizomenoides*), *A. circ* (*Anabaena circinalis*), *Anabaeno* (*Anabaenopsis* spp.), *Aphaniz* (*Aphanizomenon* spp.), *A. issat* (*Cuspidothrix issatschenkoi*), *Aphanoca* (*Aphanocapsa* spp.), *Aphanoth* (*Aphanothece* spp.), *Chroococ* (*Chroococcus* spp.), *Coelomor* (*Coelomoron* spp.), *Cyanodic* (*Cyanodictyon* spp.), *Merismop* (*Merismopedia* spp.), *Microcys* (*Microcystis flos-aquae*), *Myxobakt* (*Myxobaktron* spp.), *Oscillat* (*Oscillatoria* spp.), *Planktol* (*Planktolyngbya* spp.), *Pseudana* (*Pseudanabaena* spp.).

form blooms in Weir 32, *Cuspidothrix issatschenkoi* was related to lower discharge and phosphorus while *Aphanizomenon* spp. were found to be related to increased temperature, lowered discharge and decreased turbidity. *Planktolyngbya* was less influenced by the environmental variables but was more positively related to turbidity and nitrogen, and negatively to temperature. *Merismopedia* and *Microcystis* were more positively related to discharge and phosphorus concentrations. *Anabaena* was related to decreased turbidity and nitrogen as well as increased temperature.

Model of cyanobacterial growth

Based on the data gathered in this study, a predictive model for *Anabaena* blooms and for other major cyanobacterial blooms was developed based on flow velocity, time of year, presence of persistent stratification and the Z_{eu} to Z_{mix} ratio (Table II). The model predicts whether a bloom is likely to occur and whether it is likely to be *Anabaena* or other common cyanobacterial genera occurring in the Lower Darling River. The model also suggests monitoring or flow management strategies that may be used to mitigate the blooms by reducing potential for development, or to remove blooms that have occurred.

DISCUSSION

Several authors have suggested strategies for the use of flow management to control problem cyanobacterial blooms (Webster *et al.*, 2000; Mitrovic *et al.*, 2003; Verspagen *et al.*, 2006; Paerl, 2008). The testing of the outcomes of such actions has rarely been reported and Webster *et al.* (Webster *et al.*, 2000) recommend that these strategies need to be field tested to confirm their effectiveness. Our study of the impacts of flow releases from the regulated Menindee Lakes on problem cyanobacterial growths in the Lower Darling River was an opportunity to do this. For flow management to be successfully used to control cyanobacterial growth, a link between growth and flow needs to be established. Where blooms are related to other factors, it needs to be demonstrated that blooms can be mitigated by flows through dilution, advection or other factors disrupting growth such as decreased light availability.

Blooms of *A. circinalis* were found to be related to flow and only occurred within Weir 32 during periods of low discharge. Blooms initiated after periods of extended persistent thermal stratification which only formed during the hotter months of October to March, most likely as a result of increased solar insolation during these periods (Sherman *et al.*, 1998). A time period of approximately 10–14 days after the onset of persistent stratification is considered enough to allow populations to develop concentrations that may pose a risk to human users (Mitrovic *et al.*, 2006; Viney *et al.*, 2007). A discharge of >300 ML/day resulted in diel stratification during this summer period and was sufficient to prevent the development of *A. circinalis* blooms. As the critical discharge is site specific and will vary with river channel size, it was converted to flow velocity to allow comparison to other locations within similar climatic regions,

Table II: Predictive model for cyanobacterial growth in Weir 32 of the Lower Darling River

Flow velocity	Time of year	Persistent stratification	$Z_{eu}:Z_{mix}$	Cyanobacterial bloom development	Flow management action
Flow velocity <0.03 m/s	October to March	Yes	Z_{eu} close to Z_{mix} or lower	*Anabaena* bloom and other cyanobacterial blooms possible	Maintain minimum flow above 0.03 m/s
Flow velocity <0.03 m/s	October to March	Yes	Z_{eu} much > Z_{mix}, close to river depth	No *Anabaena* bloom but likely bloom of other cyanobacteria	Monitor for cyanobacteria, and if bloom occurs send flushing flow
Flow velocity <0.03 m/s	April to September	No	Z_{eu} close to Z_{mix} or lower or Z_{eu} much > Z_{mix}, close to river depth	No cyanobacterial blooms likely	No action required
Flow velocity >0.03 m/s	October to March	No	Above 0.3	No *Anabaena* bloom but possible bloom of other cyanobacteria	Monitor for cyanobacteria, and if bloom occurs send flushing flow
Flow velocity >0.03 m/s	October to March	No	Below 0.2	Cyanobacterial blooms unlikely	No action required
Flow velocity >0.03 m/s	April to September	No	Above 0.2	Cyanobacterial blooms unlikely	No action required

The hotter months are October to March being a southern hemisphere study. Z_{mix} is based on the mixing (thermocline) depth during persistent stratification, and the depth of the river when persistent stratification is absent.

by accounting for differences in cross-sectional area (Mitrovic *et al.*, 2003). This gave a critical velocity of 0.03 m/s to disrupt persistent thermal stratification. Provided flows are maintained above this level, it should create conditions unfavourable to *A. circinalis* growth as witnessed in the summers of 2005/2006 and 2008/2009 when discharge was generally above 400 ML/day (velocity = 0.04 m/s). Similar critical velocities (0.03 m/s) were found for overflow weir pools of the upstream unregulated Barwon-Darling River (Mitrovic *et al.*, 2006) suggesting that this target velocity should have broader application. As most river managers deal with discharge, the critical velocity must be converted to discharge to be applied to other river sites.

The critical flow velocity can be used to suppress the growth of *A. circinalis*. However, once a bloom has formed, short periods of overnight mixing did not always reduce the blooms as witnessed in November and December 2007. These mixing events were probably due to wind-induced turbulence or to other short-term climatic factors (Bormans *et al.*, 1997). More frequent or longer mixing events may have a negative impact on *A. circinalis* growth as seen in December 2007 (Figs 2 and 5). This suggests that flow velocity may need to be maintained above the critical level for more extended periods to reduce the advantage to *A. circinalis*.

Blooms of other cyanobacterial taxa (mainly *Planktolyngbya*, *Aphanizomenon* and *Merismopedia*) also occurred in the same summers as the *A. circinalis* blooms. CCA analysis showed that some of these were influenced by discharge, while others were less so (Fig. 10). Nutrients did have a significant influence on the cyanobacterial community, although were less important than turbidity, discharge and water temperature. The levels of dissolved nutrients were often above limiting concentrations supporting less of a role in structuring the cyanobacterial community (Oliver *et al.*, 2010). These taxa were still amenable to management by flow releases and in January 2008 their numbers rapidly decreased within a week of increased discharge to 3000 ML/day (Fig. 2). Downstream translocation and dilution would have been the primary factors leading to this rapid decline. The turbidity also greatly increased (to over 2000 from less than 21 NTU) and the Z_{eu} to Z_{mix} ratio decreased from close to 1 to less than 0.05. Severe light limitation and negative growth of phytoplankton would occur at this Z_{eu} to Z_{mix} ratio (Oliver *et al.*, 2010). Isothermal conditions occurred after the flow release entered the weir pool and diel stratification did not start to form until flows dropped to below ~1000 ML/day (Fig. 6). Although cyanobacteria downstream of the weir pool were not monitored, cyanobacterial growth would be negative at this level of light limitation and dilution effects would be considerable.

This suggests that the bloom density was also reduced and not just advected downstream. Flow releases of a smaller magnitude may be less likely to effectively mitigate blooms and may displace the blooms downstream of the weir. This management action, although not ideal, may still be desirable as most water use for the town and recreational activity occurs within the weir pool, and not downstream.

The use of flow management to decrease the incidence of cyanobacterial blooms in rivers is a relatively new practice and the outcomes have seldom been reported (Webster *et al.*, 2000). Berger *et al.* (Berger *et al.*, 2008) also suggest that bloom management techniques have often not been evaluated or optimized. Some studies have modelled the potential value of environmental flows to suppress blooms (Mitrovic *et al.*, 2006) or have used models to predict the development of cyanobacterial growth under different flow management scenarios (Maier *et al.*, 2004; Verspagen *et al.*, 2006). In this study, we have examined the effectiveness of water releases from a large storage at managing nuisance cyanobacteria in the Lower Darling River. These releases were part of river management protocols that dictate when water should be released to control problem algal blooms or for other purposes. An allocation of 30 GL per year is available when there is sufficient volume in the Lakes system. Constraints to the volume of water that can be sent downstream may apply during drought periods. Hence, management strategies that minimize the volume of water used would be most desirable.

Webster *et al.* (Webster *et al.*, 2000) suggested that a strategy for maintaining the discharge through a weir pool above a critical level is possibly the best way of preventing blooms. This appears to be the case in Weir 32 of the Lower Darling River where suppression flows can be used to maintain a critical discharge level above which problematic toxic *A. circinalis* blooms are likely to occur. Releases from Menindee Lakes that maintain a critical discharge in excess of 300 ML/day in Weir 32 (equivalent to a critical flow velocity of 0.03 m/s) during the months that persistent thermal stratification may occur will maintain conditions that should prevent the development of blooms. This strategy would minimize water usage, but at a discharge of 350 ML/day there would only be enough environmental water allocation held in storage for approximately 85 days. A similar strategy in the unregulated Darling River at Wilcannia (~150 km upstream) and Bourke (~300 km upstream) used "cease to pump" flow rules to maintain discharge above the critical levels to reduce the potential for bloom occurrence (Mitrovic *et al.*, 2006). Pulsing the discharge to periodically mix the water column in weir pools to reduce the advantage of buoyancy to cyanobacteria is another option. Webster *et al.* (Webster *et al.*, 2000) suggest a pulse every 3 days to ensure that the cyanobacterial population does not increase by more than 170%. This option would reduce the quantity of water required to suppress *Anabaena* blooms. This management action to our knowledge has not yet been tested. Considerable variability in discharge occurred in the 2005/2006 summer, and this did suppress persistent thermal stratification and the growth of *A. circinalis*. This supports the potential usefulness of such a strategy.

The use of greater discharges of ~3000 ML/day (velocity of 0.31 m/s) was shown to quickly mitigate established blooms of cyanobacteria through translocation, dilution and decreased water column stability and light availability. However, for this management action, greater quantities of water are required. At this discharge, only 10 days of environmental water would be available for use. If the environmental water allocation is used up in a short period of time, there is a risk of cyanobacterial blooms reforming after the discharge decreases. A lesser discharge of 1000 mL/day may also be effective, particularly if it decreases light availability to cyanobacteria through increased turbidity and a mixed water column. This was shown in the summer of 2005/2006 where no cyanobacterial bloom occurred despite reduced flow of ~500 ML/day (still above the critical discharge) as light availability was reduced.

At Weir 32 on the Lower Darling River, the development of *A. circinalis* blooms was manageable through the use of higher discharge to prevent the development of persistent thermal stratification. Higher discharge also increased turbidity and ensured mixing of phytoplankton through the entire water column, which decreased light availability to all cyanobacteria. This appeared to reduce the capacity of bloom populations to develop as the ability to utilize buoyancy regulation to attain greater amounts of light is diminished. Table II suggests a model based on our data for predicting occurrence of blooms to support flow management activities. As many of the problematic cyanobacteria are gas vacuolated and utilize buoyancy as a competitive advantage under conditions of a stable water column (Walsby *et al.*, 1997), the mechanisms described in our study may be applicable to other river and weir pool locations around the world. Global climate change is expected to increase the incidence of blooms worldwide (Viney *et al.*, 2007; Paerl and Huisman, 2009) and water availability for environmental management purposes is likely to become scarcer. Therefore, targeted and effective flow management strategies that maximize benefit while considering water availability will be required. There will also be a need to test their effectiveness and through this gained knowledge, improve techniques.

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