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PRIMARY RESEARCH PAPER

Development of blooms of *Cyclotella meneghiniana* and *Nitzschia* spp. (Bacillariophyceae) in a shallow river and estimation of effective suppression flows

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Abstract Diatom blooms in the middle reaches of the shallow, freshwater, Hunter River, Australia, are a frequent nuisance to river users. During a 4-year study, blooms of *Cyclotella meneghiniana* and *Nitzschia* spp. coincided with water temperatures above 23°C and flows below 400 Ml d^{-1} that lasted for more than 12 days. Redundancy analysis showed that water temperature was positively related, and antecedent flow was negatively related, to the abundance of both taxa. Addition experiments indicated that nutrients are seldom limiting to growth. It is suggested that a combination of faster growth rates at higher temperatures and longer retention times at low flows allows bloom populations to develop. Simulation modelling showed that flow regulation and water extraction have decreased flows in the river during summer, and consequently have probably increased the number of diatom blooms. Environmental flows have been provided to the river, but are not sufficient to prevent blooms. Discharges required for bloom suppression are described.

Keywords *Nitzschia* · *Cyclotella* · Diatoms · Flow management · Growth suppression

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Introduction

Eutrophication and consequent algal and cyanobacterial blooms are major problems worldwide. Much of the focus on phytoplankton blooms in freshwater rivers relates to toxic cyanobacteria (Chorus & Bartram, 1999). Bacillariophyceae (diatoms) can also bloom in freshwater rivers (de Sève, 1993; Gosselain et al., 1994; Garnier et al., 1995), but are not known to produce toxins in fresh waters. Excessive growths of freshwater diatoms can taint and discolour water, increase water purification costs at treatment plants, and block pump filters and drip-irrigation pipes (Gorczyca & London, 2003). Studies of the underlying causes of freshwater riverine diatom blooms are rare except for lower sections of rivers (Admiraal et al., 1990; de Sève, 1993; Kiss & Genkal, 1993; Ha et al., 2003).

Freshwater planktonic blooms of *Cyclotella meneghiniana* and *Nitzschia* spp. have been recorded frequently in the middle section of the Hunter River, New South Wales, Australia, since a large outbreak was reported in 1994. The middle Hunter River is relied on heavily for irrigation, domestic water

supply, stock watering, and cooling water for power generation, and its flows are highly modified by impoundment, flow regulation and water extraction (Ainsworth, 1996). Total discharge, the smaller floods and summer flows have been substantially reduced, and these changes are suspected of increasing the prevalence of diatom blooms.

Environmental flows have been provided to many rivers in New South Wales for such purposes as wetland replenishment, facilitation of fish and bird breeding, and suppression and flushing of algal and cyanobacterial blooms (Chessman, 2003). Manipulation of flow is one of only a few ways in which phytoplankton growth can be controlled in a river (Webster et al., 1995; Mitrovic et al., 2003). In the Barwon-Darling River in New South Wales, flow protection through restrictions on pumping has been predicted to suppress the toxic cyanophyte *Anabaena circinalis* (Mitrovic et al., 2006). Environmental flow provisions in the Hunter River include protection of low flows through limits on water extraction and an environmental contingency allowance held in storage. However, the effectiveness of these provisions in suppression and flushing of phytoplankton blooms is not known.

To our knowledge, no previous studies have reported on the development of planktonic diatom blooms in shallow (average depth <1 m) freshwater rivers. The objective of this study was to understand the environmental factors that lead to the development of blooms of planktonic diatoms in the Hunter River, and to evaluate the potential of environmental flows for bloom suppression. As flow manipulation may be a useful amelioration technique, flows required to prevent the occurrence of blooms are proposed.

Methods

Study area

The Hunter River rises on the Liverpool Range and flows for 470 km to the Pacific Ocean (Fig. 1). Terrain in the river basin (22,000 km^2) ranges from steep and dissected in some headwater areas to flat along the lower Hunter River where the floodplain is up to 40 km wide. Native woody vegetation has been removed from most of the valley floor. Major industries are coal mining, power generation, heavy

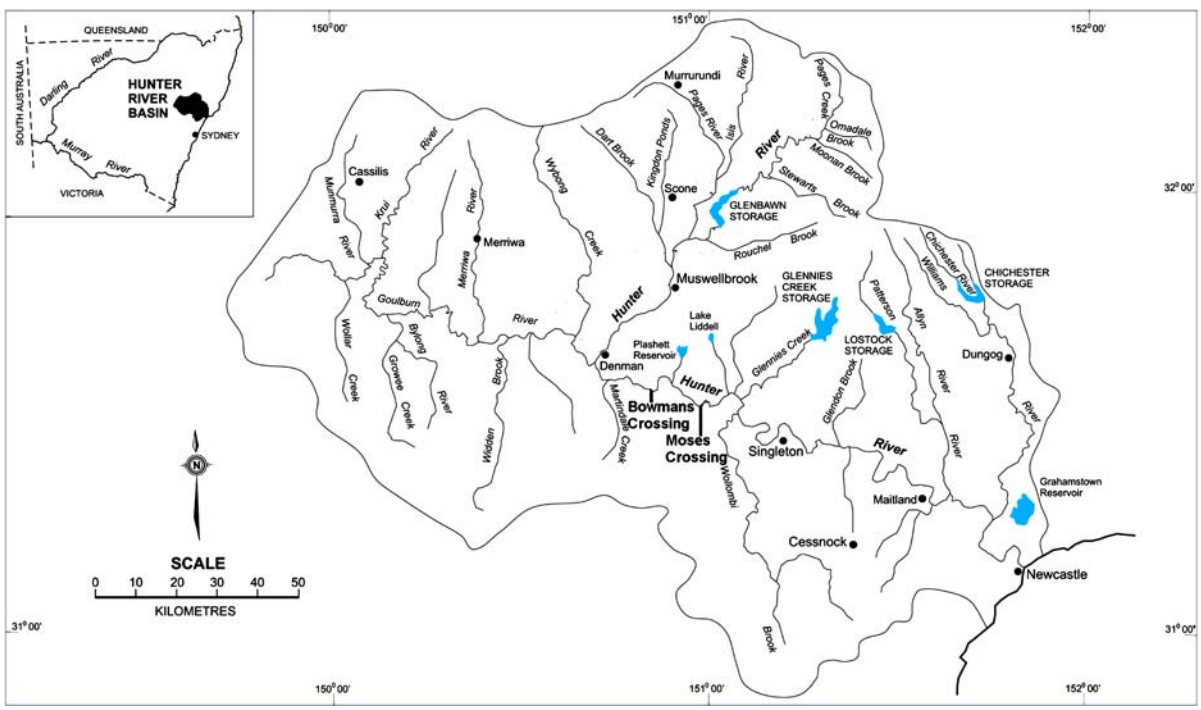

Fig. 1 Location of the study sites on the Hunter River

industry, grazing and cropping, and there are several large urban areas. Seven major impoundments occur in the basin with three (Glenbawn Dam, Glennies Creek Dam and Lostock Dam) providing regulated flow for downstream irrigation as well as urban and industrial supplies. Most abstraction of water from the Hunter River is in the middle reaches between Muswellbrook and Singleton. The effect of Glenbawn Dam (capacity 869,000 Ml) is progressively moderated by downstream inflows from unregulated tributaries, but upstream of Muswellbrook the natural seasonal pattern of high winter flows and lower summer flows has been reversed. Farther downstream, flows tend to be below natural levels in both winter and summer because of extraction, except in very dry periods when minimum flows are maintained above natural levels.

Data were obtained from two sites with frequent diatom blooms: the Hunter River at Bowmans Crossing and the Hunter River at Moses Crossing (Fig. 1). At these sites the river is wide, free flowing, generally well mixed, and shallow (usually less than 0.5 m at lower flows), although occasional deeper pools exist.

Algal and environmental measurements

The phytoplankton community and water quality variables were measured fortnightly at both sites from August 1998 to 2002. Samples for algal enumeration were taken in a 200 ml polyethylene bottle from 0.25 m below the surface and approximately 2 m from the river bank. They were preserved with Lugol's iodine solution and kept in the dark in transit to the laboratory, where cells were counted with a calibrated Lund cell and compound microscope after concentration by sedimentation in a measuring cylinder (APHA, 1998). Counting precision was ±20% (Hötzel & Croome, 1999).

Samples for nutrient analyses were collected fortnightly in the same way and kept cool (<2°C) until frozen and transported to the laboratory (within 48 h). All analyses followed standard methods (APHA, 1998). Samples were filtered immediately after collection through 0.45 μm cellulose acetate membrane filters and analysed for filterable reactive phosphorus (FRP), oxidised nitrogen (NO_x–N) and silica. The FRP sub-samples were analysed by the ascorbic acid method, the NO_x–N subsamples by the automated cadmium reduction method, and silica was determined with the ascorbic acid method. Samples were also taken for determination of turbidity and stored cool (4°C) during transport to the laboratory where they were analysed with a calibrated Hach nephelometer. Water temperature and dissolved oxygen were measured in-situ with a WTW temperature/dissolved oxygen meter. Surface pH and conductivity were recorded with a TPS Pty Ltd digital pH meter and conductivity meter, respectively.

Hydrographic data for the study sites were obtained from gauging stations operated by the New South Wales Department of Water and Energy. Stage was recorded every 15 min and converted to mean daily flow. Average daily flows were calculated for the week before each phytoplankton sampling occasion (prior-week flow).

Data analysis

The phytoplankton and environmental data were subjected to redundancy analysis (RDA) with the CANOCO program (Ter Braak, 1992). Phytoplankton cell counts were transformed prior to analysis as log (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. RDA 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 $\alpha = 0.05$.

Nutrient limitation of phytoplankton

From January to August 2005, six in-situ nutrient manipulation experiments were conducted at the two sites to test whether phosphorus, nitrogen, or silicon was limiting phytoplankton growth. Methods were as in Mitrovic et al. (2001a) except that zooplankton were removed by filtration through a net with 63-μm mesh. Also, bottles were attached by nylon rope to steel posts anchored in the river, which allowed movement within the water column

and approximately natural light exposure. In short, tests were performed in clear 1.5-l PET bottles that were either unamended or amended with 0.5 mg l^{-1} NO_3^-, 0.2 mg l^{-1} PO_4^{3-} or 5 mg l^{-1} silicate; three replicates per treatment were used at each site for each experiment. Algal cell counts were measured initially and after 4 days in each replicate. Differences in growth response among treatments were tested with ANOVA and treatment means were compared with Tukey's test. Results were considered significant at $\alpha = 0.05$.

Within-site phytoplankton homogeneity

The river channel at the two sites was examined for homogeneity of chlorophyll *a* and phytoplankton during a low-flow period (<200 Ml d^{-1}) in the summer of 2005–2006. Water samples were taken at each site from at least seven locations in mid stream and in backwaters and other areas of reduced current, at 25 cm below the water surface if depth permitted, and otherwise at the mid-point between surface and bottom. Phytoplankton was preserved and counted as previously described. Samples for chlorophyll *a* analysis were kept cool until filtration (within 24 h) through GFC filters in a pre-washed Millipore filtration unit, and then frozen. Chlorophyll *a* was determined by standard methods (APHA, 1998) with the grinding technique and acetone as a solute. Absorbance was measured at wavelengths 630, 647, 664 and 750 nm, then at 665 and 750 nm after acidification. Significant differences between midstream samples and samples from areas of reduced velocity were determined with a two-tailed *t*-test. Results were considered significant at $\alpha = 0.05$.

Modelling of discharge scenarios

A hydrological model (Simons et al., 1996) simulated daily discharge over the period 1998–2000 in the absence of major water impoundment and major extraction (low development). These discharges approximate natural flows, but are not identical because the model cannot remove the effects of minor impoundment and extraction, changes in runoff patterns caused by clearing of catchment vegetation, or flow changes caused by anthropogenic impacts on the morphology of channels and drainage connections. Simulated flows were compared with measured flows.

Results

Phytoplankton growth

Concentrations of phytoplankton were low during the cooler period (April to September) in all years, except for a slight rise in 2002 to above 5000 cells ml^{-1} for the Chlorophyceae. The community at the two sites was primarily composed of species of Bacillariophyceae, Chlorophyceae and Cryptophyceae, with low numbers of Cyanophyceae, Euglenophyceae, Chrysophyceae, Dinophyceae and Xanthophyceae. Several periods had high concentrations of diatoms, often over 30,000 cells ml^{-1} (Fig. 2). Blooms at Bowmans Crossing were dominated by *Nitzschia* spp. in January 1999 and *C. meneghiniana* in February 2000 and January 2001. At Moses Crossing, *Nitzschia* blooms occurred in January–March 1999 and March 2002, and *C. meneghiniana* blooms in January and March 2001 (Fig. 3). Other diatoms found at moderate levels at both sites included species of *Achnanthidium* and *Fragilaria, Skeletonema potamos* and *Aulacoseira granulata.* The diatoms behaved consistently at the two sites, with blooms occurring at similar times and with the same dominant genus (Fig. 3). However, the bloom in February 2000 appeared to be less intense at Moses Crossing, where counts reached only approximately 10,000 cells ml^{-1}.

On one occasion (January 2001) a bloom of Chlorophyceae reached over 30,000 cells ml^{-1} and was dominated by the genera *Planctonema* and *Oocystis*, which were the most common chlorophytes. Other common chlorophyte genera included *Scenedesmus, Ankistrodesmus, Chodatella* and *Dictyosphaerium*. Cryptophytes comprised the genera *Rhodomonas, Cryptomonas* and *Chroomonas*. The other phytoplankton groups were at concentrations lower than 400 cells ml^{-1} on every sampling occasion. Cyanophytes, although few, were dominated by the genera *Anabaena, Microcystis, Aphanocapsa* and *Oscillatoria*.

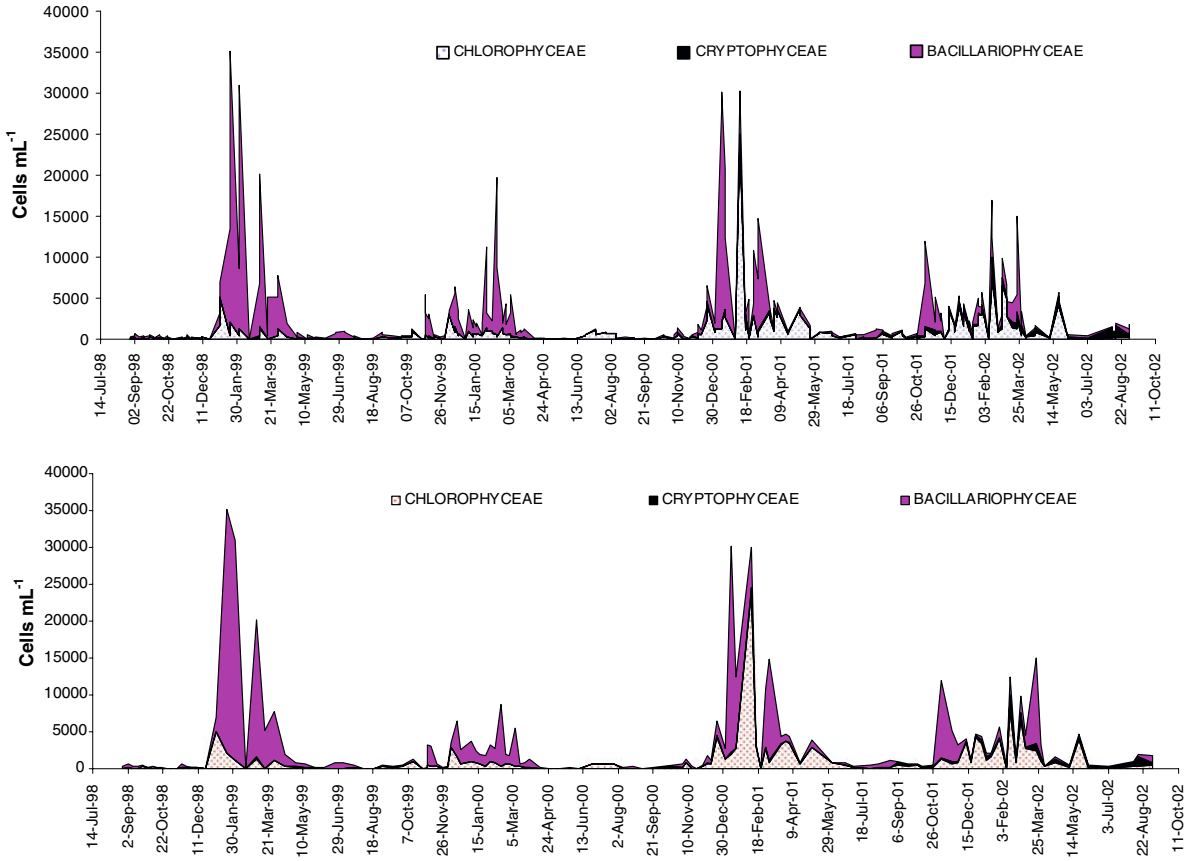

Fig. 2 Cell concentration of main phytoplankton groups over time for the Hunter River at Bowmans Crossing (top) and Moses Crossing (bottom)

Environmental conditions before and during diatom growth

Growth of *C. meneghiniana* and *Nitzschia* spp. occurred only through the hotter months (water temperatures above 23°C: Fig. 4) and in periods of low flow. Discharge had decreased to below 400 Ml d^{-1} in every diatom bloom incident (defined as greater than 10,000 cells ml^{-1}), and below 300 Ml d^{-1} for three incidents (Fig. 3). For example, the bloom of *Nitzschia* spp. at Moses Crossing in January 1999 was not recorded until flows were below 300 Ml d^{-1} for more than 13 days (Fig. 5, top). The pattern was similar during January and February 2000 at Bowmans Crossing, with growth of *C. meneghiniana* occurring after flows dropped below 300 Ml d^{-1} for 12 days (Fig. 5, bottom). Between 7 and 25 December 1999, flows at Bowmans Crossing fluctuated above and below 300 Ml d^{-1}, temperature was around 23°C and *Cyclotella* did not reach bloom levels but did grow to 4300 cells ml^{-1}. After 13 January and in early February 2000 flows were constantly below 300 Ml d^{-1} and *Cyclotella* increased again (Fig. 3). During November 2001 flows varied considerably between 700 and 150 Ml d^{-1} while water temperature was above 23°C, and in this period *Cyclotella* and *Nitzschia* numbers remained below 1000 cells ml^{-1}. Flows were consistently below 400 Ml d^{-1} after 13 January until mid February 2002, when *Cyclotella* increased to over 6000 cells ml^{-1} (Figs. 3 & 4). After flows dropped below 400 Ml d^{-1} in early March 2002, a diatom bloom of 12,000 cells ml^{-1} was dominated by *Nitzschia* with *Cyclotella* numbers around 3000 cells ml^{-1}. Flow again dropped below 400 Ml d^{-1} after 5 April and remained low till June. Temperatures generally stayed below 22°C during this period and diatom numbers remained low. The log abundance data of both *Cyclotella* and *Nitzschia* combined, over the warmer water temperature periods of November

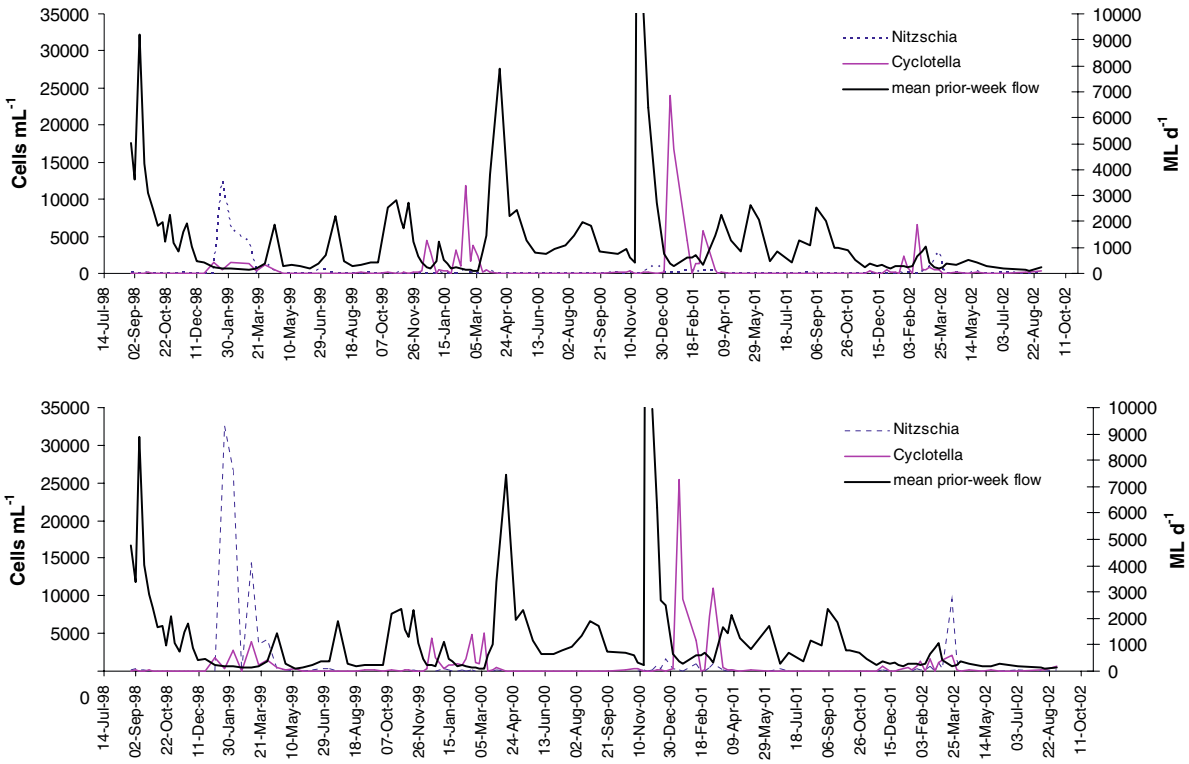

Fig. 3 *Cyclotella* and *Nitzschia* cell concentration and mean prior-week flow for the Hunter River at Bowmans Crossing (top) and Moses Crossing (bottom)

to April, had a significant negative correlation with log flow for both sites ($P < 0.001$, $r^2 = 0.20$). The log abundance data of both *Cyclotella* and *Nitzschia* combined, at flows less than 400 Ml d^{-1}, also had a significant positive correlation with water temperature for both sites ($P < 0.001$, $r^2 = 0.33$).

Nutrient data showed a similar pattern at both sites though the downstream Moses Crossing site had some higher peaks in silica and oxidised nitrogen concentration (Fig. 6). No correlation between silica concentration and *Cyclotella* and *Nitzschia* growth was observed. At times of high diatom abundance such as December 1998 and January 2001, concentrations of NO_x–N and to a lesser extent FRP were sometimes low. This relationship may have been due to nutrient draw-down during the growth period. No other pattern in the nutrient data was obviously linked to the abundance of *Cyclotella* and *Nitzschia*.

Redundancy analysis

In the RDA, three environmental variables had a significant ($P < 0.05$; Monte-Carlo permutation test) relationship with the phytoplankton community (Fig. 7), and these explained 87% of the variability. Water temperature was positively related to abundance of both *Cyclotella* and *Nitzschia*, but more strongly to the former. Prior-week flow was negatively related to these two diatoms, as was nitrogen, but the relationships for silica and phosphorus were not significant. Other dominant phytoplankters, *Oocystis* and *Rhodomonas*, were negatively related to phosphorus, *Melosira* was positively related to high flow and negatively related to temperature, and *Fragilaria* was related to higher silica concentrations. *Aulacoseira* was weakly related to the silica and phosphorus concentrations.

Nutrient limitation of phytoplankton

Diatoms were the most abundant class of algae at both sites on the Hunter River during the six nutrient manipulation experiments. *Cyclotella* spp. (dominated by *C. meneghiniana*) and *Nitzschia* spp. were found in all samples counted. For most experiments at both sites, ambient nutrient concentrations were

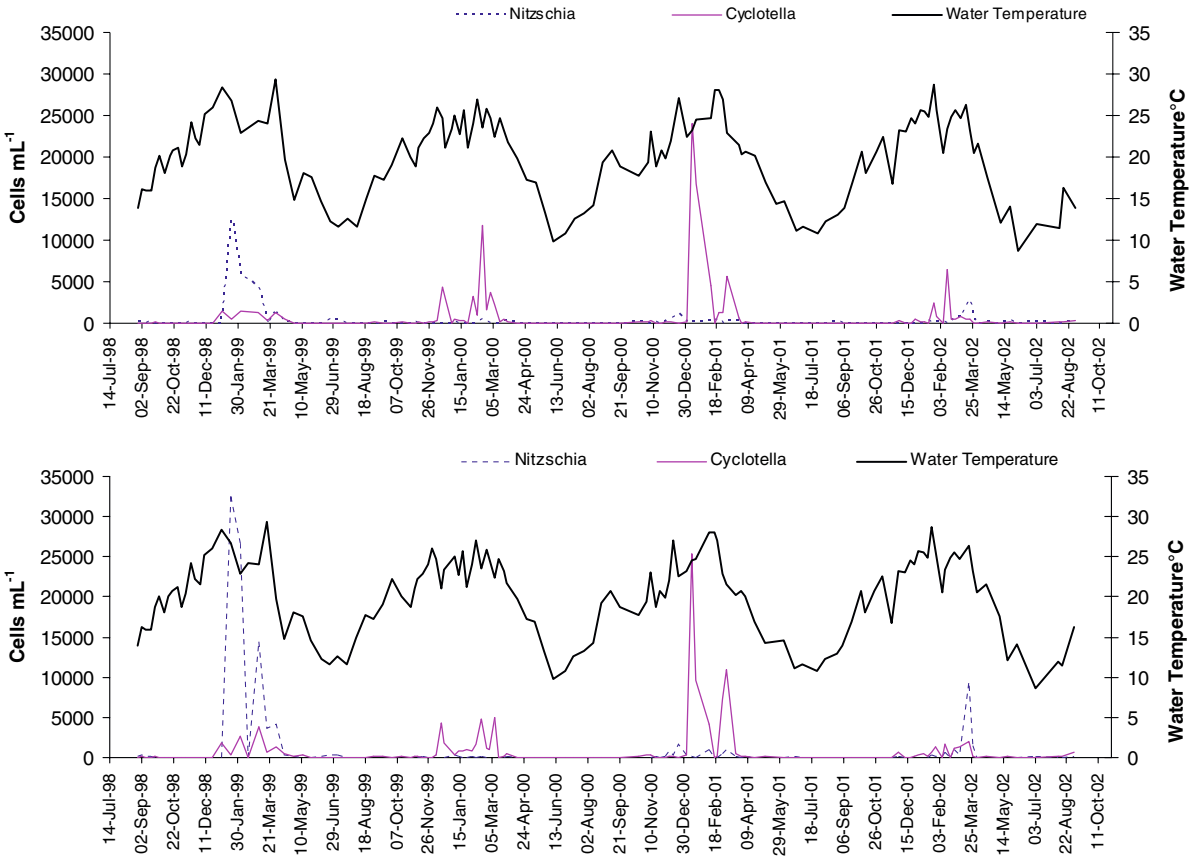

Fig. 4 *Cyclotella* and *Nitzschia* cell concentration and water temperature for the Hunter River at Bowmans Crossing (top) and Moses Crossing (bottom)

sufficient to promote significant ($P < 0.05$) growth of both genera from initial counts in control treatments (Table 1). In most cases, addition of nutrients did not result in significantly increased growth above that of the control, and average cells counts differed little among treatments. Exceptions included the 7 March and 1 August experiments at Bowmans Crossing and the 7 March experiment at Moses Crossing, when all nutrient additions produced a significant ($P < 0.05$) growth response for *C. meneghiniana*, compared to the control treatment. The March experiment possibly coincided with either the start or the finish of a *Cyclotella* spp. growth event, as shown by a high-initial cell concentration of above 2000 cells ml^{-1}.

Phytoplankton homogeneity

Phytoplankton was relatively homogeneous at mid stream at both sites, with a chlorophyll *a* concentration of 4.7 ± 0.6 µg/l (mean ± standard error). The backwater and edge samples had significantly higher ($P < 0.01$; two-tailed *t*-test) chlorophyll *a* concentrations (7.9 ± 0.9 µg/l). Algal cell counts were also significantly lower in mid stream than in the backwaters ($P < 0.05$; two-tailed *t*-test) being 842 ± 182 and 1640 ± 244 cells ml^{-1}, respectively. The composition of the phytoplankton community was similar in both areas.

Modelling of flows

Under the low-development flow scenario, discharges below 200, 300 and 400 Ml d^{-1} would have occurred on only about 5, 11 and 17% of days, respectively, in the period 1998–2000. Measured flows over this period fell under 200, 300 and 400 Ml d^{-1} for approximately 17, 31 and 37% of days, respectively (Fig. 8).

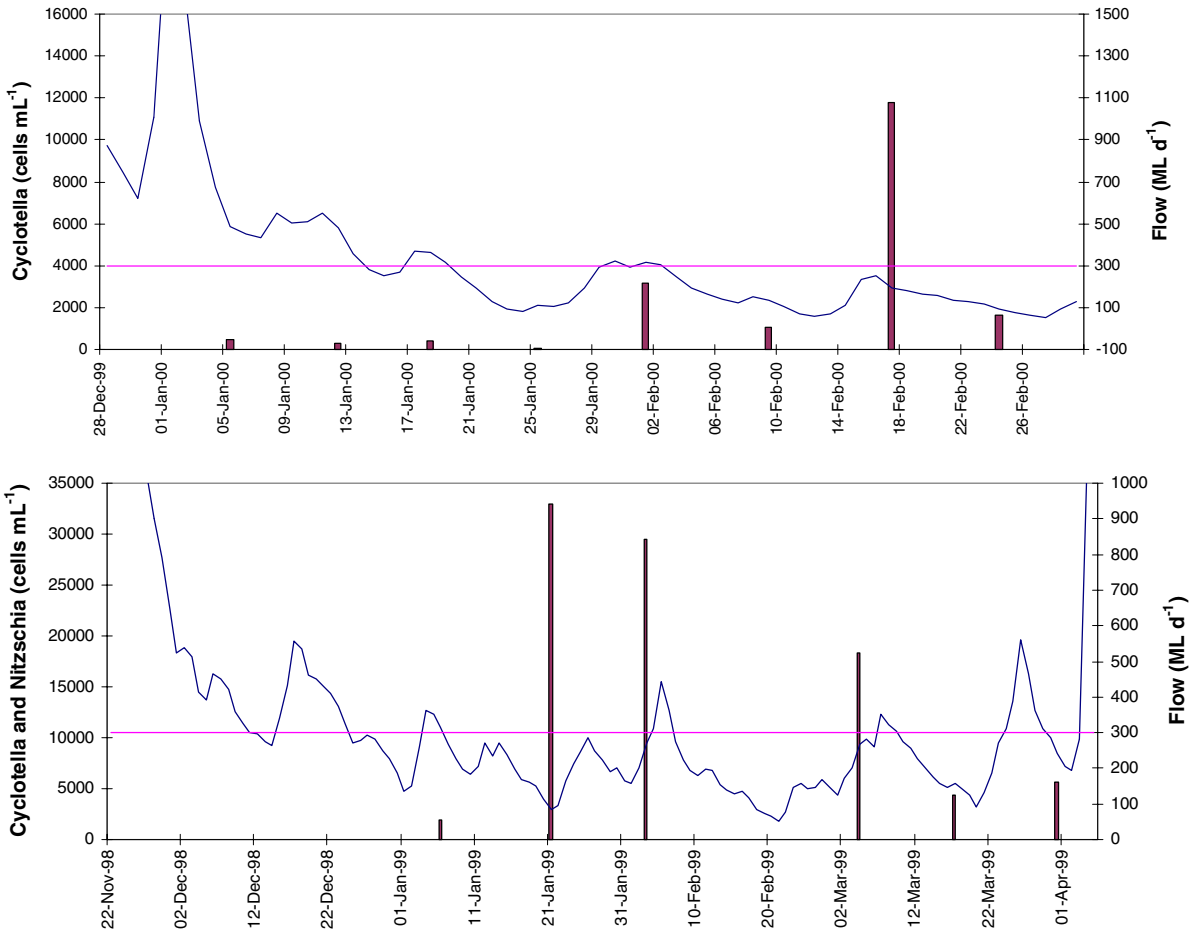

Fig. 5 *Cyclotella* cell concentration and mean daily flow for the Hunter River Bowmans Crossing, summer 1999–2000 (top) and *Cyclotella* and *Nitzschia* cell concentration (combined) and mean daily flow for the Hunter River at Moses Crossing, summer 1998–1999 (bottom). The straight line indicates the flow threshold of 300 Ml d^{-1}

Discussion

Environmental variables and diatom blooms

Blooms of either *C. meneghiniana* or *Nitzschia* spp., or both, occurred on every sampling occasion in the study with flows less than 400 Ml d^{-1} (and usually <300 Ml d^{-1}) and water temperatures greater than 23°C. Flows below 400 Ml d^{-1} did occur at times when water temperatures were lower, but did not lead to large increases in diatom concentrations. Similarly, flows above 400 Ml d^{-1} during the hotter months when water temperatures were above 23°C did not lead to diatom blooms. Redundancy analysis (RDA) revealed that water temperature was positively related to abundance of both *C. meneghiniana* and *Nitzschia* spp. and that mean prior-week flow was negatively related to these two diatoms. Significant positive correlations between these diatoms and temperature, and negative correlations with flow ($P < 0.001$) were also found.

It is probable that high flows increase advection of cells and do not allow large populations to develop. At low flows, retention times are greater and if temperatures are adequate, diatom blooms can occur. During low-flow periods in summer there is a great capacity for solar heating of the middle Hunter River as it is shallow and its width results in little shading. Flow and temperature have been reported to influence the development of diatom blooms in some lowland rivers (Kiss & Genkal, 1993; Ha et al., 2003). De Sève (1993) found a *C. meneghiniana* bloom in the

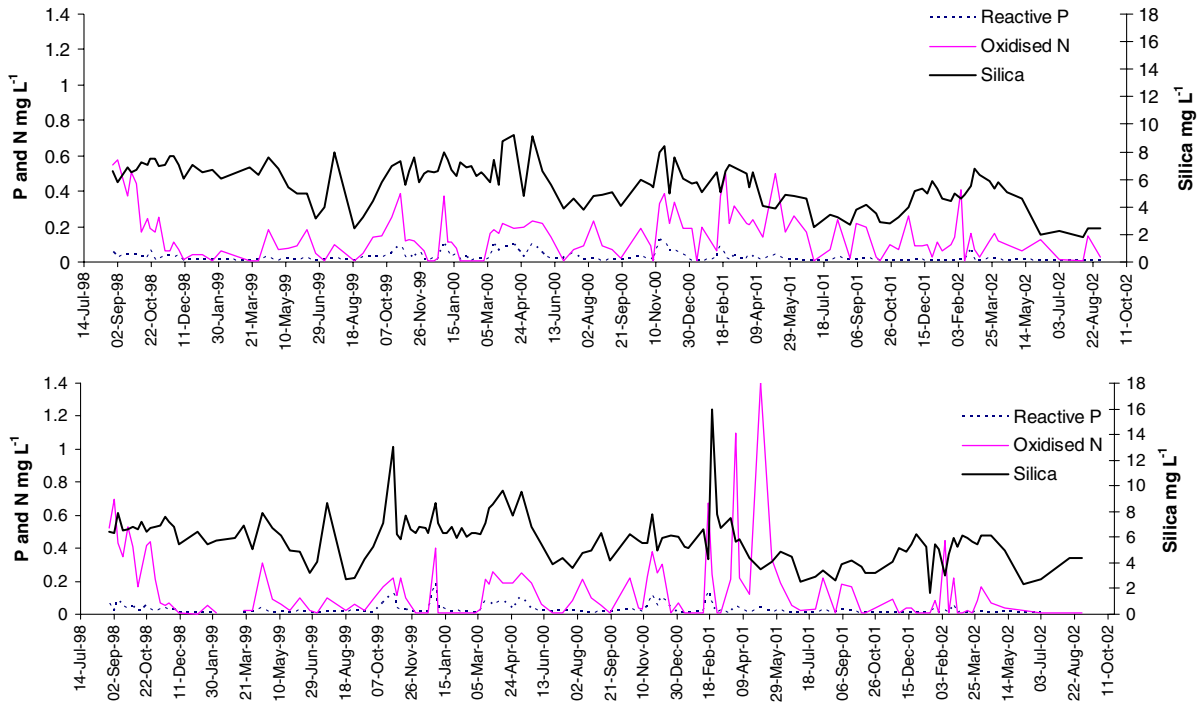

Fig. 6 Filterable reactive phosphorus (P), oxidised nitrogen (N) and silica concentration for the Hunter River at Bowmans Crossing (top) and Moses Crossing (bottom) over the duration of the study

freshwater tidal section of a river to be due to increased residence time during low-flow periods. They found blooms could form within a week with doubling times of 1–3 days. In our study, bloom levels appeared after approximately 12 days of low flow. Growth rates of the diatoms *C. meneghiniana* and *Nitzschia* spp. are at their fastest at higher temperatures (Shafik et al., 1997; Montagnes & Franklin, 2001). A *C. meneghiniana* culture was found to have growth rate maxima at 25–28°C in laboratory growth experiments (Davie & Mitrovic, unpublished data).

Recruitment of diatoms from upstream areas was unlikely to have influenced the blooms in the Hunter River, as samples collected at upstream sites and in tributaries had low diatom abundance (unpublished data). However, areas of potential recruitment were found in the low-velocity parts of the channel, such as backwaters, sometimes referred to as dead zones (Reynolds et al., 1994; Mitrovic et al., 2001b). These are likely to retain phytoplankton longer and have higher growth potential than the main stream due to increased retention time. They may increase biomass in the main stream when phytoplankton re-enter the faster flowing sections of river. Their contribution is probably modest because they make up less than 20% of total river area and their phytoplankton abundance was on average only about twice that of the main stream, although in some locations this was greater than 4-fold. Despite the different flow environments, composition of the phytoplankton was similar in both locations.

Other environmental variables did not have a strong relationship to diatom abundance. As the water column was shallow (usually less than 0.5 m deep) and well mixed at flows below 2000 Ml d^{-1}, and turbidity was moderate (median of 14 NTU), light availability is not likely to have limited phytoplankton growth. Irradiance was not measured during the study, but would likely correlate with water temperature, and a higher quantum of light available during summer may have also contributed to increased diatom growth.

Redundancy analysis of the data showed a negative association between nitrogen and *Cyclotella/Nitzschia* growth, with nitrogen and flow vectors in

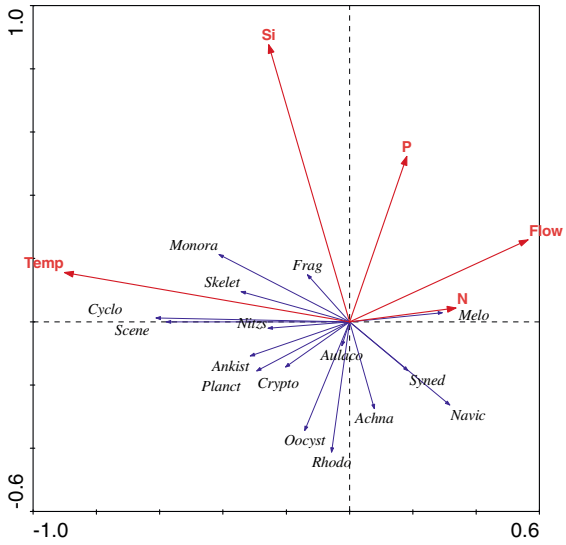

Fig. 7 RDA ordination of environmental variables and phytoplankton genus abundances for the Hunter River at Bowmans Crossing, 1998–2002. The significant ($P < 0.05$; Monte Carlo permutation test) vectors are flow, temperature and nitrogen. The length of an environmental vector indicates correlation strength 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 Flow (Ml d^{-1}), Temp (water temperature, °C), N (oxidised nitrogen, mg l^{-1}), P (filterable reactive phosphorus, mg l^{-1}), and Si (silica, mg l^{-1}). Genera are Cyclo (*Cyclotella*), Nitzs (*Nitzschia*), Skelet (*Skeletonema*), Monora (*Monoraphidium*), Frag (*Fragilaria*), Navic (*Navicula*), Syned (*Synedra*), Achna (*Achnanthidium*), Aulaco (*Aulacoseira*), Oocyst (*Oocystis*), Rhodo (*Rhodomonas*), Melo (*Melosira*), Crypto (*Cryptomonas*), Planct (*Planctonema*), Ankist (*Ankistrodesmus*), Scene (*Scenedesmus*)

the same direction (Fig. 5). It is likely that nitrogen concentration increases with flow because of runoff inputs and increased wetting of terrestrial areas. As the abundance of *Cyclotella* and *Nitzschia* decreases with increased flow, through dilution and translocation effects, the negative relationship with nitrogen is probably due to confounding effects rather than being causal. Silica and phosphorus were not correlated with the abundance of these diatoms. The in-situ nutrient addition experiments revealed few periods of nutrient limitation for *C. meneghiniana* and *Nitzschia* spp., and in most tests, ambient nutrient concentrations were sufficient to promote growth (Table 1). Exceptions such as the March experiment, when all nutrient additions produced a significant ($P < 0.05$) growth response for *C. meneghiniana*, coincided with either the start or the finish of a *Cyclotella* spp. growth event, probably indicative of nutrient drawdown by a larger phytoplankton population. In general, nutrient limitation appears to play a minor role in regulating the growth of planktonic diatoms in the mid Hunter River.

Blooms and environmental flows

As blooms of *Cyclotella* and *Nitzschia* require low flow and water temperatures above 23°C, a continuous flow above 300–400 Ml d^{-1} during summer should be sufficient to prevent blooms. Pulses of flow to above 400 Ml d^{-1} for a day per week may also be effective, as indicated by reduced diatom growth in low-flow periods with spikes in flow to above 400 Ml d^{-1}. Under modelled low-development conditions similar to natural conditions, flows under 400 Ml d^{-1} would have occurred on only about 17% of days while under current water management this has risen to about 37% of days. Consequently, it is likely that water resources development has increased the prevalence and intensity of diatom blooms in the river.

Environmental flows are likely to reduce the frequency and duration of blooms in those rivers where the occurrence of blooms is strongly related to flow, but to our knowledge environmental flows have not been specifically used or proposed to reduce the incidence of diatom blooms in rivers. However, flow management has been suggested as a potential amelioration technique for planktonic cyanobacteria (Webster et al., 1995) and has the potential to be used successfully for this purpose (Mitrovic et al., 2006). Environmental flow provisions, especially the maintenance of minimum flows, do protect flow in the Hunter River during the summer period. However, the minimum flow allocation for the middle river is well below that required to suppress diatom blooms. In the current Water Sharing Plan for the Hunter River, the minimum flow set for Liddell (between the two study sites) is 17 Ml d^{-1} from December to February and 18 Ml d^{-1} from March to May. However, the plan also includes an annual environmental contingency allowance of 20,000 Ml that could be used either to flush blooms or to increase flow during potential growth periods.

In the mid Hunter River, which is shallow with the water column well mixed, higher flows appear to

Table 1 Results (mean ± standard error) from nutrient enrichment experiments in the Hunter River

Location	Genus	Date	Treatment response (cells ml^{-1})				
			Initial	Control	+P	+N	+Si
Bowmans crossing	*Cyclotella*	31/01/05	467 ± 56	7113 ± 2921	19262 ± 6502	20423 ± 3695	18879 ± 4510
		07/03/05	2044 ± 109	10320 ± 607[a]	16070 ± 379[b]	44120 ± 1357[b]	14360 ± 1157[b]
		30/03/05	663 ± 127	12307 ± 607[a]	7527 ± 1392	7717 ± 1828	8603 ± 1013
		25/04/05	1395 ± 152	6312 ± 924[a]	10667 ± 910[b]	7632 ± 650	12332 ± 1110[b]
		25/05/05	480 ± 61	4471 ± 143[a]	4628 ± 142	4679 ± 474	4729 ± 150
		01/08/05	154 ± 30	2500 ± 146[a]	3439 ± 179[b]	3517 ± 234[b]	4408 ± 94[b]
	Nitzschia	31/01/05	432 ± 109	2818 ± 435[a]	2324 ± 330	1726 ± 132	1640 ± 208
		07/03/05	242 ± 52	1620 ± 89[a]	1470 ± 130	1580 ± 232	2450 ± 265[b]
		30/03/05	88 ± 7	1242 ± 125[a]	770 ± 164	602 ± 203	607 ± 148
		25/04/05	170 ± 13	204 ± 100	467 ± 49	378 ± 55	855 ± 99[b]
		25/05/05	115 ± 29	248 ± 130	193 ± 74	293 ± 74	500 ± 92
		01/08/05	58 ± 11	684 ± 109[a]	664 ± 97	550 ± 47	816 ± 91
Moses crossing	*Cyclotella*	31/01/05	1009 ± 58	18269 ± 3784[a]	19637 ± 3347	8741 ± 534	3573 ± 2184
		07/03/05	3710 ± 309	12510 ± 642[a]	19700 ± 216[b]	19790 ± 318[b]	21830 ± 625[b]
		30/03/05	1178 ± 53	9225 ± 463[a]	10050 ± 1053	8617 ± 1128	7380 ± 1018
		25/04/05	1010 ± 111	5394 ± 1608	5984 ± 868	12575 ± 2802[b]	7050 ± 310
		25/05/05	763 ± 36	4586 ± 264[a]	5801 ± 594	5985 ± 515	5529 ± 705
		01/08/05	244 ± 36	3483 ± 359[a]	3975 ± 387	2933 ± 862	5132 ± 240[b]
	Nitzschia	31/01/05	483 ± 45	1508 ± 266[a]	1057 ± 221	780 ± 133	913 ± 684
		07/03/05	110 ± 35	2360 ± 583[a]	4510 ± 249[b]	1940 ± 250	3800 ± 555
		30/03/05	136 ± 18	1253 ± 97[a]	1030 ± 91	733 ± 118	538 ± 90
		25/04/05	95 ± 22	325 ± 144	425 ± 28	700 ± 117[b]	409 ± 21
		25/05/05	68 ± 11	243 ± 47[a]	264 ± 58	355 ± 56	257 ± 42
		01/08/05	175 ± 5	1475 ± 106	1100 ± 78	1309 ± 78	1600 ± 83

[a] Denotes control significantly different from the initial population

[b] Denotes significantly different from control

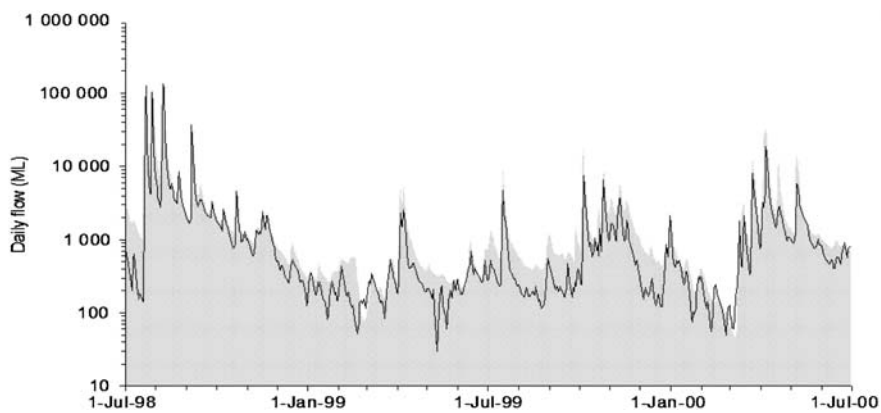

Fig. 8 Modelled low-development flow (shaded area) and measured flow (line) for the Hunter River at Lidell, located between the two study sites

reduce residence time sufficiently to suppress diatom population development. The same may not occur in deeper systems or within weir pools where thermal stratification may occur. Sherman et al. (1998), in their weir pool study, found that high flows were associated with high diatom abundance, due to re-suspension into the euphotic zone, and low flows led to the elimination of diatoms and their replacement by cyanophytes. It is important to qualify the nature of the system being managed as the results of flow management strategies will vary depending on the constraining factors for diatom growth. It should also be considered whether changing the flow regime to control diatom blooms might also change the phytoplankton community balance and potentially allow dominance of a less benign phytoplankton taxon. This is currently unlikely in the mid Hunter River with only low densities of potentially toxic cyanophytes measured at all flow regimes.

Since many rivers worldwide are subject to both eutrophication and hydrological modification (de Jonge et al., 2002; Green et al., 2004), excessive diatom growths and other phytoplankton blooms may become more frequent and intense. Flow management raising discharge at critical periods may be one method to reduce their impact where growth is related to flow. However, the use of water for freshwater diatom suppression in shallow rivers needs to be balanced with other uses of the water, since the deleterious effects of diatoms are much less pronounced than those of toxic cyanobacteria.

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