Impacts of a toxic *Microcystis* bloom on the macroinvertebrate fauna of Lake Elphinstone (Central Queensland, Australia).

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Running headline: Impacts of *Microcystis* on macroinvertebrate fauna

Extra keywords: cyanoprokaryotes, macroinvertebrates, *Microcystis*, shallow lakes.

This paper has not been submitted elsewhere in identical or similar form, nor will it be during the first three months after its submission to *Hydrobiologia*.

ABSTRACT

The biological, physical and chemical properties of Lake Elphinstone were studied during a dense, toxic cyanoprokaryote bloom dominated by *Microcystis*. Decreases in total abundance and richness in macroinvertebrate communities coincided with increases in *Microcystis* toxicity. Water quality was characterized by high light attenuation values caused by abiogenic turbidity and shading and absorbance from thick algal scums. The study highlights the potential for multidimensional environmental impacts associated with toxic cyanoprokaryote blooms, and the consequent implications for the management of shallow, inland and tropical lakes that are susceptible to toxic blooms.

INTRODUCTION

Cyanoprokaryote (blue-green algal) blooms are potentially highly toxic to the ecological health of tropical, shallow water bodies. Whilst relationships between environmental conditions and toxic bloom events have been examined (Havens *et al*., 1998; Marinho & Huszar, 2002), hardly any studies have interpreted the effects of these blooms on affected waterways, particularly with respect to macroinvertebrate fauna (Kotak *et al*., 1996). The few that do exist have focussed on the calculation of LC_{50} values (e.g. Metcalf *et al.,* 2002; Hiripi *et al.,* 1998) rather than ecological effects. Uptake, depuration, and mechanism of actions for cyanoprokaryote toxins also remain poorly understood in invertebrates (Vinagre *et al*., 2002).

This paper examines changes in the macroinvertebrate community and physico-chemical parameters of a shallow tropical Australian lake (Lake Elphinstone) during the progression of a toxic cyanoprokaryote bloom, demonstrating the potentially wide-ranging effects of toxic cyanoprokaryote blooms on the macroinvertebrate fauna of such lakes.

MATERIALS AND METHODS

Lake Elphinstone is a tropical, inland freshwater lake located in the far north of the Fitzroy Catchment in Central Queensland (Figure 1a). The lake is a natural, shallow water body (mean depth during this study 1.3m), approximately 3 kilometres long and 600m across, with no significant inflows or outflows. Surrounding land uses include beef cattle grazing and the extraction of nearby coal resources in the Bowen Basin; the lake itself is used recreationally for water sports and fishing. Lake Elphinstone experiences a tropical climate, with highly variable rainfall averaging 666mm annually. Field trips were conducted on 20 - 22 March, 13 - 14 May and 22 - 23 July 2002, to coincide with the expected peak cyanoprokaryote bloom season. Methods relating to the collection and analysis of phytoplankton samples and their toxicity can be found in White *et al.* 2003.

For macroinvertebrate sampling, the lake was divided into ten areas and six were randomly chosen as sampling sites (Figure 1b). These same sites were sampled on each visit; using collection by pond net (32 x 32 x 35cm; 250µm mesh size) for one minute in each of two 1m^2 quadrats; the two samples were combined and immediately preserved in 70% ethanol. Samples were later washed using 100µm mesh sieves to remove fine particles and sorted, identified and counted to family level by randomly subsampling one-tenth of the total sample, and examining under an Olympus dissection microscope. Sorting continued until at least 300 individuals were reached, or the entire sample was sorted. Microcrustacea (cladocerans, copepods and ostracods) were not considered in accordance with the Australian River Assessment Scheme protocol (AUSRIVAS, 2001) however water mites were included due to their conspicuous abundance.

Substrate samples (approximately 300g) were also collected at each macroinvertebrate sampling site; dried to constant weight (105^{0}C), cooled and sieved to obtain particle size distribution (AS/NZ 1141.11). Organic matter content was determined from 20g test portions of unsieved material by ashing (500^{0}C, 8 h).

Physical and chemical sampling was conducted at four study sites, chosen to reflect the most within-lake variability based on earlier studies in 2001 (Figure 1b); including measurement of Secchi disc extinction depths (30cm diameter Secchi disc), downwelling photosynthetically active radiation (PAR; LI-COR LI-189 meter fitted with underwater sensor S/N UWQ 450) and water column profiles (pH, conductivity, temperature and dissolved oxygen; YSI-650 MDS meter). All light data were recorded from the sunny side of the boat in similar weather conditions. The vertical attenuation coefficient of downwelling irradiance (K_d) was calculated using linear regression of the natural log of PAR according to Kirk (1994).

Statistical analyses were carried out using SigmaStat (analysis of variance, post-hoc Tukey testing; Pearson Product Moment analyses) or the PRIMER suite (Bray-Curtis similarity matrices, Shannon Diversity Index (H′) and Principal Components Analysis) (Carr, 1996).

RESULTS

Phytoplankton and Cyanotoxicity

A dense cyanoprokaryote bloom was recorded during March, with peak concentrations following in May, before collapse occurred prior to sampling in July (White *et al.* 2003). Toxin concentrations were variable throughout sampling. Microcystin (MC) levels were below the limit of detection (< 0.5 μg L^{-1}) in March, followed by an extreme toxicity peak in May (2500 μg L^{-1}), corresponding to bloom dominance by *Microcystis panniformis*, and finally a decline to 1000 μg L^{-1} measured from a scum sample collected in July (White *et al.,* 2003). Cylindrospermopsin was detected in small concentrations (0.3 μg L^{-1} or less) during May (White *et al.* 2003).

Macroinvertebrate community

Mass deaths of aquatic snails were a striking feature observed during sampling at Lake Elphinstone: concentrated masses of carcasses washed onto the western shore covered several metres square in some areas. The snails ranged in size (approximately 5 – 20mm) and consisted of five families (Thiaridae, Physidae, Sphaeriidae, Glacorbiidae and Planorbidae).

A poor remaining macroinvertebrate population was indicated by low macroinvertebrate family diversity and very low abundances. A total of 24 macroinvertebrate families were recorded from the lake (Table 1; Figure 2), with the total number of families per site significantly decreasing over the study, particularly between March and July (One-way ANOVA; $p = 0.011$, $F = 6.165$, $df = 2$). Macroinvertebrate abundance also varied strongly between sampling sites and dates (Figure 2, Table 1). Abundance was especially low during May and July, averaging < 75 individuals per sample. Water mites (particularly *Piona* and *Arrenurus*) and dipterans (almost entirely Chironomidae) were the most strongly represented orders, though hemipterans became relatively abundant in July (Table 1). Significant differences between abundance in March and both of the other sampling months were detected (Tukey testing; $p = < 0.001$). Also, a Bray-Curtis similarity matrix performed on log-transformed abundance data showed samples to be grouped generally according to sampling date rather than location (Figure 3).

A Shannon Diversity Index calculated on the macroinvertebrate family data indicated that diversities during May (averaging 1.64) were higher than those in March (1.30) and July (1.48) (data not shown). Significant differences in index values were not found between sampling dates ($p = 0.384$, $F = 1.020$, $df = 2$).

Physico-chemical conditions

Lake Elphinstone was characterized by wide variations in water quality throughout sampling. Both supersaturation of surface layers (up to 227% saturation of dissolved oxygen, Figure 4a) and severe benthic oxygen depletion (< 10% saturation, Figure 4b) were recorded. During March, mild to strong stratification was evidenced by an oxycline apparent between approximately 1.2m and 1.4m, together with surface to bottom temperature gradients of 0.2 – 2.0^{0}C (Figure 4a, b). Surface oxygen concentrations were highly positively correlated with temperature data (Spearman rank-order analysis; $p = < 0.001$).

Low Secchi disk extinction depths, averaging 25, 14 and 18cm in March, May and July, respectively, were recorded along with shallow euphotic depths ranging from 33 – 54cm (Table 2). High light attenuation values of 8.9 m^{-1} to 14.0 m^{-1} were also noted (Table 2).

Average surface pH was highest during March (9.0) and decreased in May (8.7) and July (8.4) (Table 2). At some sites pH values decreased with depth, especially where dramatic differences in surface and bottom oxygen concentrations were evident. Conductivity became progressively higher as mean lake depth decreased, ranging from 1372 µS cm^{-1} in March to 1857 µS cm^{-1} by July (Table 2).

Substrate conditions

Most sites were composed predominantly of fine sand or coarse silt, with, on average, almost 45% of mass represented by particles < 300µm. Particle size composition did not change significantly over the sampling period (One-way ANOVA; $p = 0.246$, $F = 1.545$, $df = 2$). Organic matter represented on average 1.42% dry weight of substrate samples; again, no significant differences were detected between sampling occasions (One-way ANOVA; $p = 0.425$, $F = 0.006$, $df = 2$).

Examining changes in the macroinvertebrate community: water quality, substrate conditions, or cyanotoxicity?

Macroinvertebrate abundance and richness were both significantly negatively correlated with MC concentration ($p = 0.00000428$ and $p = 0.0249$, respectively) based on a Pearson Product Moment Correlation of natural-log transformed macroinvertebrate data and MC values. No significant correlation existed between MC and diversity (H').

No significant relationships ($p < 0.050$) were found between macroinvertebrates (abundance, richness) and substrate conditions (particle size, organic matter) (Pearson Product Moment correlation; calculated on macroinvertebrate data and substrate characteristics for 13 of 18 sampling occasions (five samples omitted due to lost labels)). However, significant, positive relationships were evident between Shannon diversity and particle size ($p = 0.0403$, $r= 0.574$, $n = 13$), richness and abundance ($p = 0.03158$, $r= 0.596$, $n = 13$) and richness and diversity ($p = 0.04426$, $r= 0.5649$, $n = 13$).

Finally, PCA was used to examine the strength of influence of environmental parameters on the macroinvertebrate community, using water quality data including microcystin toxicity. Selected parameters were log-transformed prior to analysis. Temperature, pH, light, conductivity and microcystin concentration were primarily responsible for placement of sampling sites according to sample date, with the first two components explaining 77% of variation (Table 3, Figure 5).

DISCUSSION

Macroinvertebrate community

The resident macroinvertebrate population of Lake Elphinstone was low in both abundance and richness, particularly during the last sampling occasion. Low richness in macroinvertebrate fauna is claimed to be typical of Australian lakes, at least for benthic fauna (Timms, 1979), especially for small, shallow and periodically dry water bodies. Shallow lakes (< 4m mean depth) in the Upper Burdekin region of Queensland, for example, have recorded fewer than five macroinvertebrate species in total, probably as a result of substratum hypoxia (Timms, 1979). In contrast, between 70 – 86 macroinvertebrate taxa have been recorded from riverine sites in the Fitzroy Catchment to which Lake Elphinstone belongs (Duivenvoorden *et al.*, 2000; Duivenvoorden & Roberts, 1997).

The dominance of Acarina, particularly *Piona* and *Arrenurus,* is not unexpected given these genera are usually associated with nutrient enriched ponds (Di Sabatino *et al*., 2000). High chironomid abundance is also characteristic of poor water quality (Williams, 1980), and the domination of this taxon in Lake Elphinstone is probably linked to eutrophy and substratum hypoxia.

Physical and chemical conditions

A toxic cyanoprokaryote bloom coincided with major changes in the water quality of Lake Elphinstone. Reduced light availability resulted principally from thick cyanoprokaryote scums. High abiogenic turbidity was associated with resuspension of bottom sediments, especially during the very shallow (< 1.5m) depth in July. Benthic anoxia in the morning, coupled with high surface oxygenation after midday reflected the early morning onset of photosynthesis coinciding with an increase in oxygen concentrations. Senescing phytoplankton cells contributed to high oxygen demands in the subsurface waters.

Conductivity values recorded from Lake Elphinstone were far higher in comparison to those recorded from the lower Fitzroy River (150 - 900 $\mu S\ cm^{-1}$) (Fabbro, 1999), and of other Queensland lakes (100 – 300 µS/cm) (Finlayson & Gillies, 1982; Farrell *et al*., 1979), probably as a result of evaporative concentration. High conductivities have been reported in conjunction with cyanoprokaryote blooms in other Australian systems, such as the Barwon-Darling, which peaked above 1400 $\mu S\ cm^{-1}$ during dominance by *Anabaena circinalis* (Bowling & Baker, 1996).

Substrate conditions

Low organic matter recorded from Lake Elphinstone's substratum potentially indicates low food availability for sediment- and detritus-feeding macroinvertebrate fauna, with <1.5 % organic matter recorded in comparison to values of 11.7 – 75.8 % recorded from the deepest sections of lakes of Northeastern Queensland (Timms, 1979). In turn, predator and grazer groups may also have suffered flow-on effects including lack of available prey (other macroinvertebrates, zooplankton).

Examining changes in the macroinvertebrate community

Significant decreases in both macroinvertebrate richness and abundance were recorded from Lake Elphinstone during the transition from a non-toxic to highly toxic *Microcystis* bloom (Figure 2). Extreme microcystin toxicity (maximum concentration 2,500 $\mu g\ L^{-1}$) is considered likely to have strongly impacted the macroinvertebrate community: both macroinvertebrate abundance and richness were significantly negatively correlated with high microcystin values ($p = < 0.05$). Furthermore, in the absence of cyanoprokaryote toxicity during March, macroinvertebrate abundance was comparable to other Central Queensland riverine and wetland sites unaffected by algal blooms (Duivenvoorden & Roberts, 1997).

It is difficult to pinpoint the exact nature of the effects of MC exposure, given the limited information regarding MC toxicity to macroinvertebrates. Circumstances observed at the lake during sampling are consistent to those reported by Krzyżanek *et al*. (1993), where dying out of bottom fauna (particularly chironomids and decapods) was observed during a dense cyanoprokaryote bloom. Krzyżanek *et al.* (1993) linked blooms of *Aphanizomenon flos-aquae* with mortality of benthic fauna including Chironomidae, Oligochaeta, Ceratopogonidae, Decapoda and Bivalvia in the Goczalkowice Reservoir, although the toxicity of this bloom was not examined.

Equally, it is likely that the overall paucity in macroinvertebrate fauna at Lake Elphinstone resulted from the challenging environmental conditions associated with the dense *Microcystis* bloom. The PCA analysis of water quality data (Figure 5) demonstrated that some variables impacted by algal blooms (e.g. pH, light conditions) were influential in the differentiation of sampling sites. High abiotic turbidity may also have affected the macroinvertebrate population via smothering, clogging of feeding apparatus and alteration of photosynthetically derived food availability (Rempel *et al.,* 2000; Hogg & Norris, 1991). Reduced food availability was also likely, given the low organic matter content and fine silt layer, which may have prevented food access (Rempel *et al*., 2000).

The contributions of other conditions in impacting the macroinvertebrate community are considered to be minimal. For example, seasonal changes have not been associated with richness and abundance losses in other central Queensland studies (Duivenvoorden *et al*., 2000; Duivenvoorden & Roberts, 1997). Furthermore, although interactions between macroinvertebrates and substrate conditions have been demonstrated by other authors (Rempel *et al*., 2000; Hogg and Norris, 1991), no such relationships were evident at Lake Elphinstone.

Conspicuous mass snail deaths observed at the lake may have resulted from unsuitable habitat conditions (e.g., loss of aquatic macrophytes from March to July, benthic anoxia, and highly turbid water conditions) and/or from cyanotoxicity. Microcystin toxicity specifically for freshwater gastropods has received limited study: both Zurawell *et al.* (1999) and Kotak *et al.* (1996) reported MC bioaccumulated in freshwater snails, although neither documented the effects associated with such contamination. MC bioaccumulation has also been demonstrated in freshwater bivalves (Yokoyama & Park, 2002), mussels (Williams *et al.*, 1997), clams (Prepas *et al.* 1997) and crayfish (Lirås *et al.*, 1998). Mass snail deaths at Lake Elphinstone may signal a concern for the possibility of bioaccumulation and trophic transfer.

CONCLUSIONS

This paper demonstrates the potential for multidimensional impacts associated with toxic cyanoprokaryote blooms, with poor water quality, extreme microcystin concentrations and decline in macroinvertebrate communities all recorded in conjunction with a *Microcystis* bloom. Macroinvertebrate family richness and abundance were both negatively correlated with microcystin concentrations. The toxic cyanoprokaryote bloom clearly resulted in deleterious effects on the macroinvertebrate community, either directly, via microcystin toxicity; or indirectly, via change in water quality characteristics of the lake mediated by the bloom.

ACKNOWLEDGEMENTS

The authors thank Anita Dale, Heather Smyth and Graham Boyle; Noel Sawtell for his assistance in the field; and Mark Harvey and Michael Price for assistance in macroinvertebrate identification. Trevor Kavanagh and Wendy Western (Nebo Shire Council) also provided assistance. Nebo Shire Council (Lake Elphinstone Alliance) provided partial funding for this project and kindly provided accommodation during field trips.

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Table 1. Average macroinvertebrate abundance recorded from Lake Elphinstone during sampling in 2002 (*n* = 6). –; not present.

	March	May	July
Hemiptera			
Corixidae	42	4	28
Pleidae	5	1	2
Mesovelidae	-	1	-
Coleoptera			
Hydrophilidae	2	1	-
Dytiscidae	1	1	-
Diptera			
Chironomidae	127	15	10
Ceratopogonidae	3	3	7
Gastropoda			
Thiaridae	6	4	3
Glacorbidae	2	-	-
Physidae	2	2	2
Planorbidae	2	3	6
Sphaeriidae	-	1	-
Acarina			
Arrenuridae	14	12	1
Pezidae	3	-	-
Pionidae	67	26	5
Trichoptera			
Hydroptilidae	1	-	-
Leptoceridae	1	1	5
Hydropsychidae	1	1	2
Odonata			
Coenagrionidae	2	-	-
Libellulidae	1	1	-
Corduliidae	-	-	1
Oligochaeta			
Tubificidae	1	-	1
Decapoda			
Atyidae	-	2	-
Palaemonidae	-	1	1
TOTAL NO. FAMILIES	19	18	14

Table 2. Physical and chemical conditions in Lake Elphinstone during March - July 2002. Values are the average of four sampling sites except ionic composition, where only one sample was collected per trip.

	March (Mean ± SD)	May (Mean ± SD)	July (Mean ± SD)
Mean depth (m)	1.58 ± 0.25	1.19 ± 0.19	1.16 ± 0.09
Surface temperature (^{0}C)	27.8 ± 1	22.3 ± 1	18.2 ± 0.8
Surface pH	9.0 ± 0.3	8.7 ± 0.1	8.4 ± 0.1
Surface Dissolved Oxygen (mg L^{-1})	141 ± 57	58 ± 10	97 ± 5
Secchi depth (cm)	25 ± 9	14 ± 1	18 ± 6
Euphotic depth (cm)	54 ± 14	33 ± 3	38 ± 9
Light attenuation K_d (m^{-1})	8.9 ± 2.5	14.0 ± 1.3	12.6 ± 3.0
Conductivity (µS cm^{-1})	1372 ± 45	1622 ± 0.2	1857 ± 1.7

Table 3. Eigenvalues and eigenvectors for a Principal Components Analysis of environmental data recorded at Lake Elphinstone.

Eigenvalues

PC	Eigenvalues	%Variation	Cum.%Variation
1	4.86	60.7	60.7
2	1.33	16.6	77.3
3	0.89	11.1	88.4
4	0.60	7.6	96.0
5	0.19	2.4	98.4

Eigenvectors (Coefficients in the linear combinations of variables making up PC's)

Variable	PC1	PC2	PC3	PC4	PC5
Depth	-0.316	0.421	0.471	-0.338	-0.037
Temperature	-0.403	0.379	-0.137	0.012	0.055
DO	-0.343	-0.105	0.335	0.695	0.119
pH	-0.363	0.111	-0.491	0.283	-0.643
Conductivity	0.388	-0.325	0.238	0.216	-0.399
Secchi depth	-0.297	-0.517	-0.204	-0.498	-0.150
Euphotic depth	-0.361	-0.440	-0.166	0.104	0.522
Microcystin	0.345	0.299	-0.530	0.129	0.339

FIGURE CAPTIONS

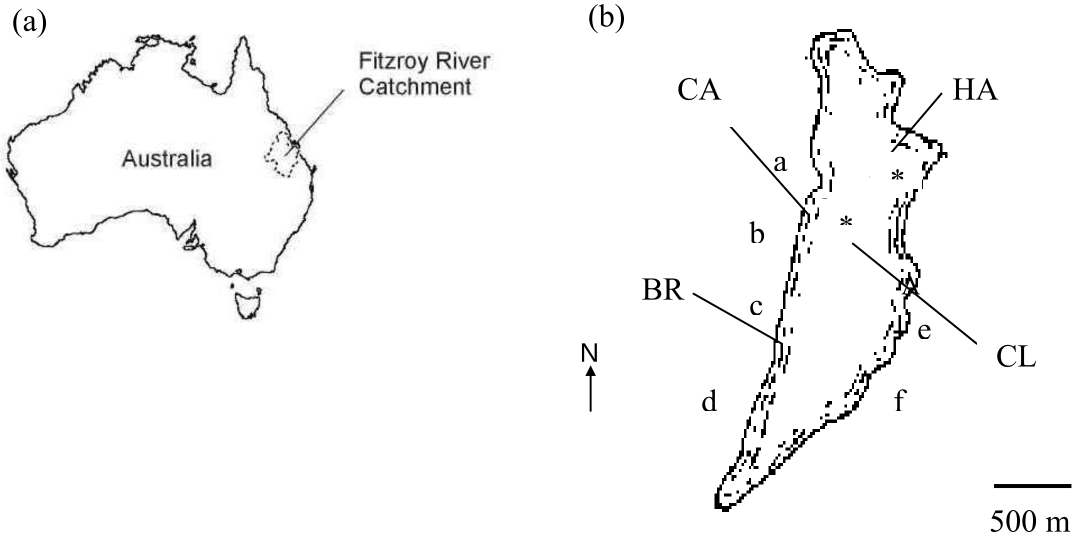
Figure 1. (a) Location of the Fitzroy Catchment in Central Queensland. (b) Approximate position of four phytoplankton/physical-chemical study sites, macroinvertebrate sampling sites (a - f) and placement of buoys with temperature loggers within the lake (*). CA – Camping Area; HA – Homestead Area; BR – Boat Ramp; CL – Centre of Lake.

Figure 2. Average macroinvertebrate family richness and abundance recorded from Lake Elphinstone during 2002, together with the highest recorded toxin concentration for the sampling date.

Figure 3. Dendogram of a Bray-Curtis similarity matrix performed on log-transformed macroinvertebrate abundance data.

Figure 4. Selected water column profiles from Lake Elphinstone, comparing patterns of temperature, dissolved oxygen and PAR (A)Camping area 4:15pm (B) Camping area 7:50am.

Figure 5. Factor scores on the first and second principal components for environmental data measured on three sampling dates at Lake Elphinstone. *Note*: CA – Camping Area; HA – Homestead Area; BR – Boat Ramp; CL – Centre of Lake.


(a)
Fitzroy River
Catchment
Australia
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CA
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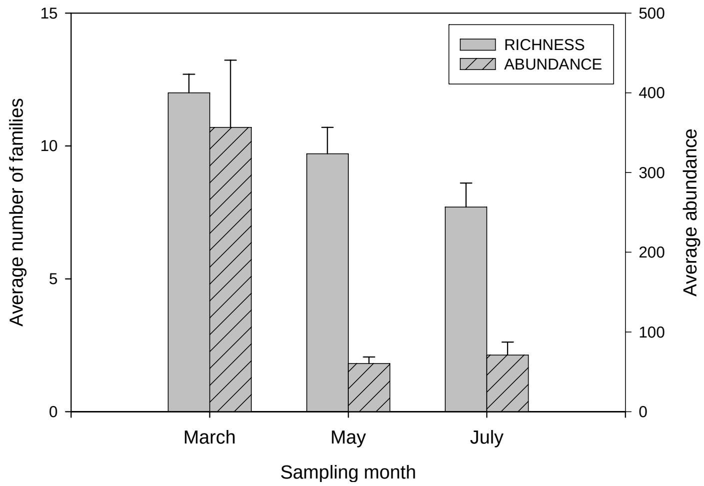
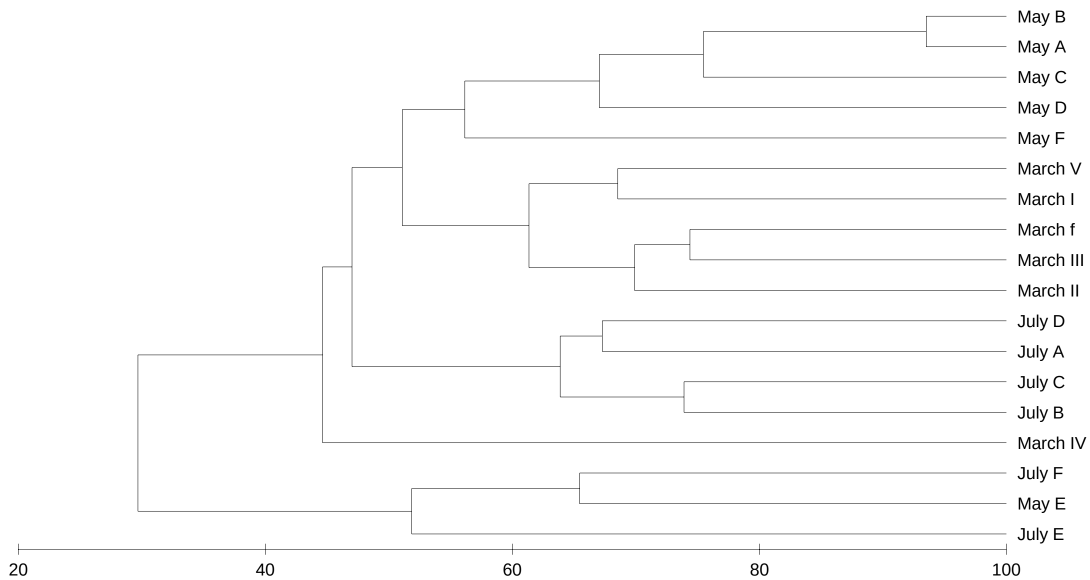

RICHNESS
ABUNDANCE
Average number of families
Average abundance
Sampling month
March
May
July
0
5
10
15
0
100
200
300
400
500


May B
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March II
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May E
July E
20
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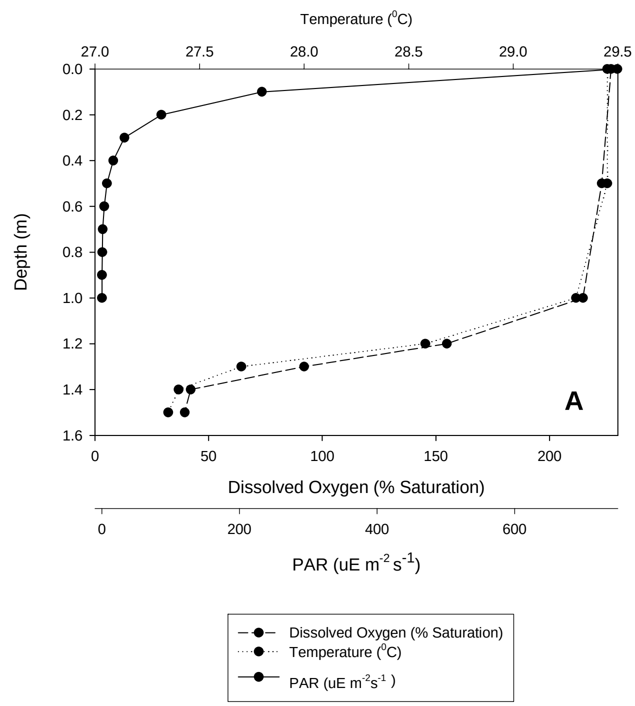
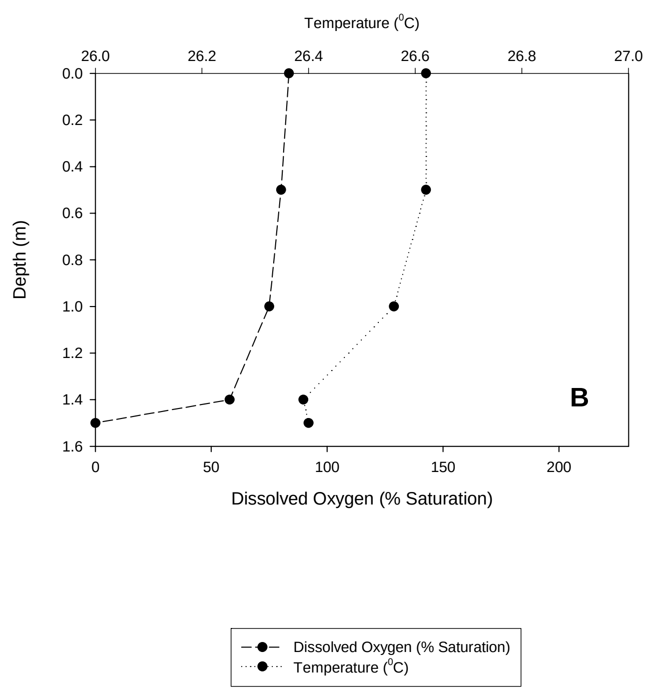
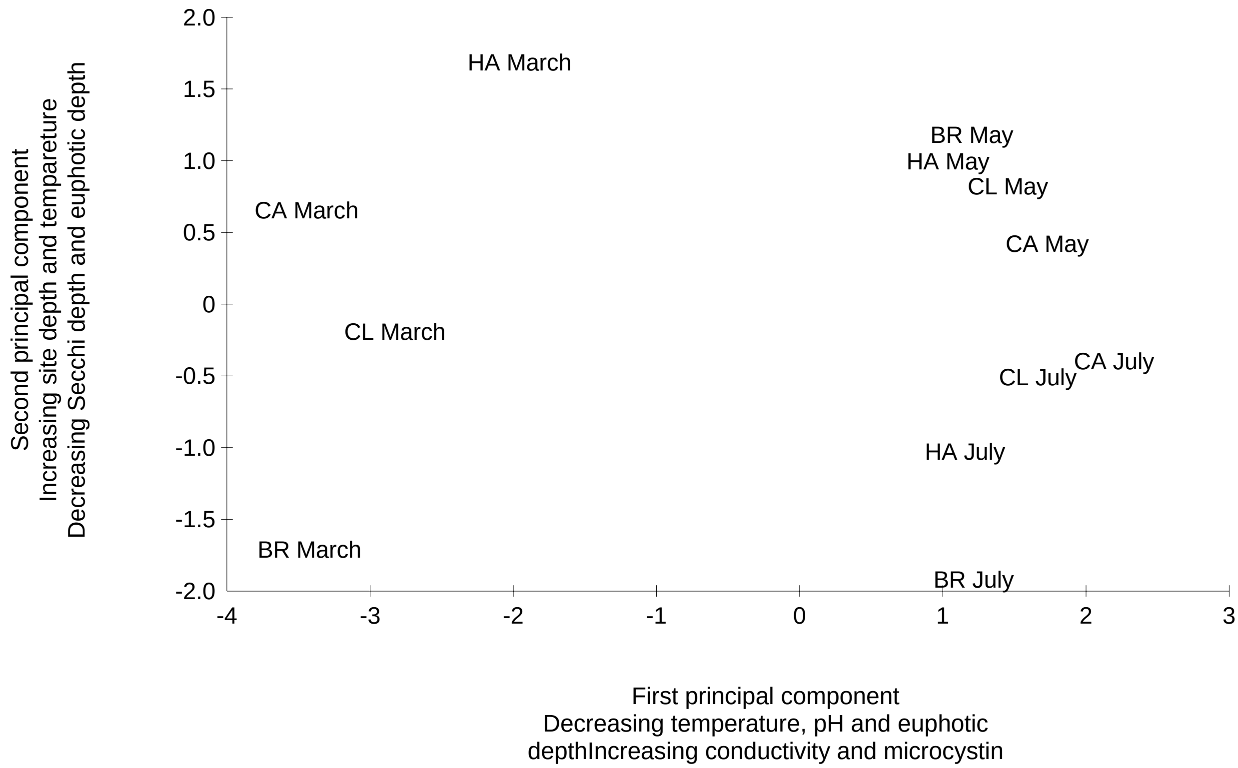
Camping Area: 4:15pm, 21/03/02



Camping Area: 7:50am, 22/03/02




2.0
1.5
1.0
0.5
0
-0.5
-1.0
-1.5
-2.0
-4
-3
-2
-1
0
1
2
3
HA March
CA March
CL March
BR March
BR May
HA May
CL May
CA May
CA July
CL July
HA July
BR July
Second principal component
Increasing site depth and tempareture
Decreasing Secchi depth and euphotic depth
First principal component
Decreasing temperature, pH and euphotic
depthIncreasing conductivity and microcystin