Monitoring Study of Lake Austin Wetlands following 14 Months of Dewatering Discharge from the Cuddingwarra Mine.

A report prepared for Newhampton Goldfields P/L

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Executive Summary

I. This report outlines the condition of the Lake Austin biota and its fringing vegetation during June 2000, some 14 months following commencement of saline water discharge from the Cuddingwarra mine. Changes from pre-discharge (September 1998) and post-discharge (June 1999; aquatic only) condition are reported. The rainfall in the six month period before June 2000 was well above average; this resulted in rare occurrence of a lake full of water, as well as considerable inundation of fringing vegetation.

II. Broad-scale changes were detected in water levels and ionic concentrations between sampling periods, as well as in the richness, abundance and composition of aquatic species. As no differences were detected between sites close and distant from discharge, none of these changes could be attributable to discharge. Rather changes were generally attributable to the high rainfall and flooding preceding June 2001 sampling.

III. Changes were also detected in the fringing vegetation between sampling periods: annual/ephemeral species composition altered; overall cover of perennial species increased (particularly those not-flooded); and slight changes in perennial species composition occurred. These changes are believed due to differences in the season and amount of pre-sampling rainfall. The health and cover of species closest to the lake declined due flooding and/or smothering by *Ruppia* and macro-algae; abundance of these species generally increased however due to seedling recruitment. No structural or composition differences in the vegetation were detected between impact (close to discharge) and control (distant from discharge) sites, suggesting no impact due to discharge. Similarly, the condition of perennial plant species did not differ, except for two species: *Sclerostegia tenuis* and *H. halocnemoides,* which were less healthy at impact sites. This suggests discharge is impacting these species, although alternative explanations are discussed. More comparisons and further monitoring are recommended to either confirm or reject the notion of impact due to discharge.

IV. Despite very few, if any, impacts being detected due to discharge of saline waters into the lake, the fact that sampling has occurred during extensive flooding of the lake should be taken into account. This rare event has, to probably a strong degree, diluted ionic concentrations and homogenised physico-chemical properties of water and soil. Therefore, impacts may be masked or more slow to develop compared to situations where there are appreciable differences between waters and soils close to discharge and some distant from it (ie the control and impact groupings of sites as established during pre-impact sampling).

V. To more firmly gauge the degree and nature of impacts due to discharge, further monitoring over time is recommended, particularly as impacts may take some time to develop. Additional studies recommended include: tracing ions concentrations and ratios from discharge to gauge zone of impact; obtaining greater understanding of hydrology; and measuring the recovery of flood damaged vegetation.

VI. This report constitutes ECU's Centre for Ecosystem Management Report **2000 – 12.**

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1. Introduction

1.1 History & Background

A new gold mine was commissioned at Cuddingwarra (between Cue and Big Bell) in 1999. Shallow groundwater layers mean constant dewatering of the mine pit is required. Environmental approval was obtained to discharge this hypersaline water to Lake Austin. This discharge commenced in May 1999 and continues at a rate of between 3000-5000 KL /day (averaging 4200 KL/day). Total dissolved solids of the discharge average 112000 mg/L, with electrical conductivity averaging around 150000 uS/cm (see Appendix 1). Monitoring was a condition of environmental approval; this study represents the first of such monitoring episodes, some 14 months following commencement of discharge.

Due to concerns expressed in previous studies that discharge was preferentially entering adjacent inlet channels, the pipeline was extended in April 2000 from the lake edge to 900 m out onto the lake-bed (see cover photo). This was to encourage discharge flow towards the middle of the lake.

1.2 Previous Findings - September 1998

A pre-impact, baseline study was conducted in September 1998 (Horwitz et al. 1999). This study described faunal and floristic communities in and around the discharge point and established monitoring sites and protocol. Sixteen macroinvertebrate taxa were found in surface water samples, several of which were undescribed species, such as the abundant brine shrimp (*Parartemia* sp.) and an ostracod (possibly new genus). Rehydration of sediments yielded 13 macrovertebrate taxa, including 9 not found in surface water samples. Eight plant communities were delineated and mapped in the area around the discharge point. Four of these communities were dominated by samphires and fringed the lake and inlet channels, whereas the other four communities were found on higher ground further back from the lake. Major floristic changes were linked to steep salinity gradients from lake bed to dune and to subtle changes in microrelief.

1.3 Rainfall and Lake Levels

Monthly rainfall for Cue, located some 25 km east of the discharge point, is shown in Figure 1 for the period December 1995 to June 2000. This graph illustrates the highly variable distribution of rain in this warm to hot, arid climate. It is quite common for no or negligible amounts of rain to be received in any given month. In contrast, monthly rainfall several times the average is also a regular feature - this occurred in: June and July of 1996; February, April and August of 1997; May, July, August & December of 1998; March & December of 1999; and January, March and April of 2000. Overall, 1700 mm of rain has fallen between December 1995 and June 2000 which is 37% above the average expected for this period. The period between December 1999 and April 2000, which saw a number of cyclonic, low pressure systems move inland from the north-west coast, was clearly the wettest 5 months of recent times.

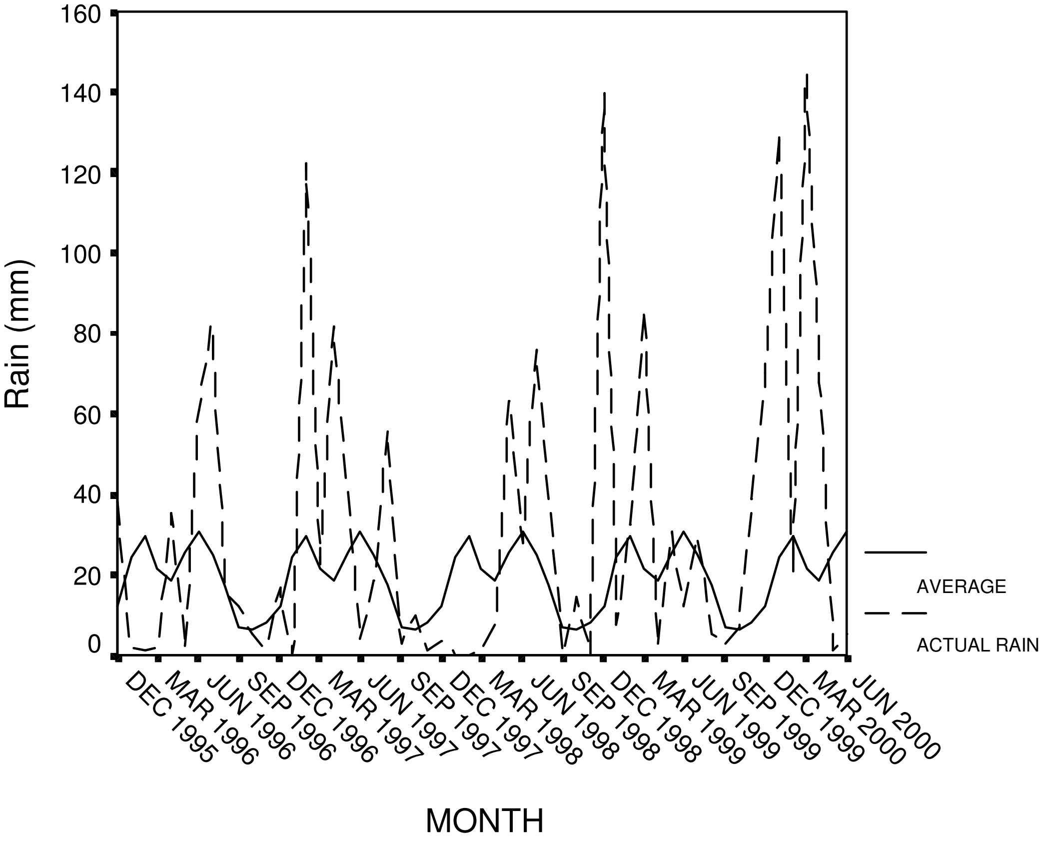

Figure 1. Monthly rainfall for Cue from December 1995 to June 2000 (hatched line). Monthly averages are also shown (solid line).

No detailed monitoring of lake levels has occurred. It is known that the lake is usually dry. At the time of initial survey in September 1998, the lake contained an amount of water, contributed by above average rainfall during winter of that year, but was not near full. Lake levels remained below fringing salt-marsh vegetation. The substantial summer-autumn rains of 1999/2000 have contributed to extremely high lake levels which have inundated much of the edge of the fringing vegetation around the lake and inlet channels. At the time of this survey (June 2000), lake levels had receded from their high-point of April this year by several centimetres (A. Wilkeis, pers. comm.); high water marks were also evident by the deposition of *Ruppia* and macroalgae and by watermarks on the soil.

Water in the main part of the lake shifts with the direction of the wind. Therefore lake levels at survey sites can vary daily by several centimetres depending on whether winds are off-shore or on-shore. During the period of the June 2000 survey, wind speeds were moderate and still conditions often prevailed.

Discharge water is regularly monitored for water quality (ions, salinity, EC, total dissolved salts, pH). These data are shown in Appendix 1. NaCl makes up the vast majority of dissolved solids.

1.4 Monitoring Approach

During September 1998, permanent monitoring sites were established and measured along seven transects placed more or less perpendicular to the shore-line from lake bed to dune systems above fringing salt-marsh. Four of these transects were located close to the discharge point, two leading from the lake-bed, and two across the major inlet channel which enters the lake near the discharge point. These are hereafter identified as ‘impact’ sites and transects. The other three transects were located some distance from the mine discharge point, one across an inlet channel several kilometres to the north-east of the discharge point, and the other two on the other side of the lake. They are hereafter known as ‘controls’. Transect locations are shown in the Appendix 2.

Sites were re-measured in June 2000. This included the measurement of vegetation characteristics, soil parameters and tagged plants within permanent plots, as well as sampling of water quality and fauna at inundated sites. Change in condition from September 1998 was calculated. Statistical tests to compare the average change in the impact zone to that in the control zone were performed. Significant differences, it is argued, indicate either positive or negative impact due to discharge. The underlying assumption here is that the discharge of large volumes of hypersaline water leads to greater water volumes and water/soil salinity in the area immediately surrounding the discharge point than elsewhere. Discharge water has been observed to persist close to drainage point when the lake is dry or contains small amounts of water; it has also been observed to move up the adjacent major inlet channel under certain wind directions. The fate of discharge water when the lake contains substantial amounts of water and the rate at which it mixes with lake water is however unknown. If discharge water disperses efficiently and rapidly into lake waters, the argument for distinct impact and control zones is weakened.

Where appropriate, comparisons are made to monitoring work undertaken during 1999 (see Sommer *et al. 2000).*

2. Changes to Fauna

2.1 Methodology

2.1.1 Rationale

The aim of the fauna survey carried out between 23.6. and 25.6.2000 was to establish whether any changes in aquatic macroinvertebrate diversity have occurred since the 1998 baseline survey and commencement of discharging of hyper-saline water into Lake Austin, and if so, whether this can be attributed to the discharge. The same transects and sites (descriptions outlined in Horwitz *et al.* 1999) were sampled as during the baseline survey, although some minor discrepancies occurred due to differences in the water level of the lake (Table 1). The locations of the seven transect lines are shown in Appendix 2.

Table 1. Sites sampled (x) during the 1998 and 1999 fauna surveys

	Transect 1		*Transect 2*		*Transect 3*		*Transect 4*		*Transect 5*		*Transect 7*	Total
Site	LA1 'shallow'	LA1 'deep'	LA3 'Pool1'	LA3 'Pool 2'	LA5-1	LA5-2	LA7-1	LA7-2	LA9-1	LA9-2	LA13	no. of sites
1998	X	X	X	X		X		X	X		X	8
2000	X	X	X		X	X	X	X	X	X	X	10

All of the sites had substantially more water in June 2000 than in September 1998. At Transect 2 half way between Transect 1 and the mouth of the inlet, there were two distinct shallow pools in 1998, whereas in 2000 the entire area was inundated forming just one deeper body of water (see photo No. 2, Appendix 3). At transect 3 (location of discharge pipe at the mouth of the inlet) the lake proper was dry in 1998 (LA5-1), and so only the inlet channel (LA5-2) could be sampled, whereas in 2000 both sites were sampled. At Transect 4 there is a lagoon-like channel separated from the lake proper by a small sand bar (photo no. 7, Appendix 3). This "lagoon" (LA7-1) was dry in 1998, while the lake proper (LA7-2) contained very shallow water which was sampled at the time. In 2000 both of these sites could be sampled. At Transect 5, the inlet channel (LA9-1) was sampled in 1998 and 2000, while the lake proper (LA9-2) could only be sampled in 2000. Photos of all of the sites sampled in June 2000 are included in Appendix 3.

2.1.2 Sampling method

Along each of the transects selected and sampled during the baseline survey where surface water was found, aquatic macroinvertebrates were sampled using a 250 µm meshed D-framed pond net by vigorous sweeping over a distance of 20m (where possible). Macroinvertebrate samples were sorted alive in the field for one person hour, and preserved immediately in ethanol. They were later identified in the laboratory to species wherever possible using taxonomic keys and voucher specimens from the 1998 survey.

Along each transect, and in the location where macroinvertebrate samples were collected, water depth, pH, electrical conductivity, temperature, redox potential, dissolved oxygen and turbidity were measured with a portable electronic meter (Yeokal model). Physico-chemical parameters were measured at the top and bottom of the water column if the water depth was greater than 20 cm. A surface water sample was collected from below the surface of the water in an acid-washed polyethylene bottle (rinsed once prior to filling), for subsequent measurement of total dissolved solids (TDS), ionic composition, available nutrients, colour and alkalinity. Water samples were filtered in the field where required, and filtered and non-filtered samples were frozen within 5 hours of collection, and analysed following thawing. Analysis of the water samples was organised by Newhampton Goldfields Ltd.

2.1.3 Data analyses

Macroinvertebrate species lists were constructed for each sampling site and each sampling occasion (i.e. September 1998 and June 2000), as were lists of physico-chemical data. Data analyses consisted of comparing macroinvertebrate and physico-chemical data from the baseline survey and the June 2000 survey. Particular attention was given to relationships between salinity/electrical conductivity (EC) and macroinvertebrate species richness and community structure. These relationships were established by creating descriptive graphs and carrying out a Pearsons correlation between species richness and EC. Qualitative analyses of species composition were also made.

Macroinvertebrate presence/absence and physico-chemical data were then classified and ordinated using multivariate techniques from the CSIRO statistical package PATN. Analysis of the association between sites was carried out using the Bray and Curtis dissimilarity measure for the faunal data, and the Gower Metric measure for the physico-chemical data. Sites were classified using flexible UPGMA, an hierarchical polythetic agglomerative method of cluster analysis, and dendrograms were produced. The dissimilarity matrices were then ordinated using semi-strong hybrid (SSH in PATN) multidimensional scaling to show relationships between sites in two or three dimensional space. Principal axis correlations (PCC in PATN) were performed to see if there was any linear relationship between the macroinvertebrate gradients and the physico-chemical variables. Correlation coefficients are given for each physico-chemical attribute and these were used as an indicator of significance, or of the degree to which each variable explains variation in the macroinvertebrate data.

2.2 Results

2.2.1 Aquatic macroinvertebrates

Aquatic macroinvertebrates collected during the 1998 baseline survey and the 2000 survey are listed in Table 2. Many more taxa were collected during the 2000 survey than during the 1998 survey. In 1998, a total of 16 macroinvertebrate taxa were collected, while in 2000, 31 were collected. In addition, two fish species were collected in 2000 (one of them is *Craterocephalus cuneiceps*, the other is an unidentified species from the grunter Family Terapodinae). Large numbers of fish hatchlings were observed close to the shoreline in 2000, especially in the vicinity of Transect 1. Three additional ostracod species were collected from surface waters in 2000, *Reticypris pinguis*, *Heterocypris n. sp.* and *Cyprinotus edwardi*. These species were hatched from sediment rehydration trials conducted in 1998. However, not all of the ostracod species hatched from the rehydration experiments in 1998 were found in surface waters in 2000. *Bennelongia australis* and *Ilyocypris australiensis* were only present in the rehydration experiments. The as yet undescribed cypridid ostracod was the only invertebrate species to be present at all sites in the 2000 survey. In 1998 only shells of the gastropod *Coxiella gilesi* were found, while in 2000 complete animals were collected in small numbers from every site except LA9-2 (Transect 5).

As in 1998, only the shells of Conchostraca were collected in June 2000. The as yet undescribed species of *Parartemia* which was found at every site, at times in large numbers, during the 1998 survey was absent from surface waters in June 2000. The most conspicuous difference between the invertebrate fauna of Lake Austin in September 1998 and in June 2000 is the large number of aquatic insects present during the latter survey. In 1998 only six taxa/species of insects were collected, while during June 2000 eighteen were collected. Of these the larvae of chironomid midges, ceratopogonid flies and the damselfly *Austrolestes annulosus* were found in numbers greater than 1000 at some sites. In addition to the insects, a freshwater mite (family Arrenuridae) was sampled at LA9-1 in June 2000.

These faunistic differences are also evident when summary occurrences of taxa (presence or absence of all taxa collected between 1998, 1999 and 2000) are presented (Table 3). Results for rehydration experiments in 1998 revealed mainly crustacean taxa with dessication resistant life stages (principally ostracods), and richness is limited for these life history reasons (see Horwitz et al. 1998). Results for the survey in September 1998 revealed substantially lower richness than either monitoring during 1999, or June 2000 survey. The monitoring results for 1999 were taken over a five-six month sampling schedule, over a period when the lake was full to when it had dried completely, at inlet sites for Transects 1 and 5 (Sommer et al. 2000). The monitoring was undertaken by relatively untrained personnel, and normally it would be expected that such detailed sampling would detect a greater diversity of target taxa as water chemistry changes. Nevertheless, the 1999 monitoring also detected a higher diversity of aquatic insect faunae.

Figure 2 shows a comparison of macroinvertebrate taxa richness at each site during September 1998 and during June 2000. The highest taxa richness in 2000 was at site LA5-2 (Transect 3) where 19 invertebrate and 1 fish species were found. The lowest number of species (6) was collected from site LA9-2 (lake proper at the end of Transect 5). Site LA9-1 (inlet on Transect 5) had the second highest taxa richness (18), and this site also had the highest taxa richness during the 1998 survey (13). Figure 3 shows the classification of sites based on macroinvertebrate presence/absence data. There are two clear site groupings. LA1-shallow, LA3-1, LA5-2, LA1-deep, LA9-1 and LA5-1 form one group, and LA7-2, LA13, LA7-1 and LA9-2 and form another. All of the sites from the first group are located in inlet channels (see Appendix 2), while those from the second group are located in the lake proper, indicating that these two habitat types have a characteristically different set of invertebrate fauna.

Table 2. Macroinvertebrates and fish sampled at Lake Austin during the September 1998 and June 2000 surveys

SITE				LA1 'shallow'		LA1 'deep'		LA3 'Pool1'		LA3 'Pool 2'	LA5-2		LA5-1
Higher Taxa	**Family**	**Genus**	**Species**	**Sept. 98**	**Jun-00**	**Sept. 98**	**Jun-00**	**Sept. 98**	**Jun-00**	**Sept. 98**	**Sept. 98**	**Jun-00**	**Jun-00**
Cladocera	Daphnidae	*Daphniopsis*	*Pusilla ?*	2		4				3		1	2
Ostracoda	Cyprididae	*New genus*	*new sp.*	rare	1	3	rare		3	2		3	3
		Diacypris	*Dictyote*	2	4?	4	?4	2	?3		rare	?4	
		Diacypris	*Whitei*	2	2	4	rare	4	3	4	4	4	
		Reticypris	*Pinguis*		3		rare		1			2	3
		Heterocypris	*n. sp.*		rare								
		Cyprinotus	*Edwardi*				rare					1	
Copepoda	Cyclopoida	*Microcyclops ?*	*Platypus ?*	rare		2			rare	rare			
Anostraca	Branchipodidae	*Parartemia*	*n. sp.*	1		2		3		2	2		
Gastropoda (shells only 9/98)	Pomatopsidae	*Coxiella*	*Gilesi*		1	1	rare	2	rare	2	1	rare	rare
Conchostraca (shells only)										1	rare	1	
Diptera	Chironomidae				2	3	3	3	2	2	1	4	4
	Ceratopogonidae				2	rare		2	1	1		1	
	Stratiomydae												
	Dolichopodidae							rare			1		
Hemiptera	Corixidae									rare			
Lepidoptera	Pyralidae				2		2		1			2	
Zygoptera	Lestidae	*Austrolestes*	*Annulosus*		1				2			2	
		Austrolestes	*sp.2*				rare		2			2	
Anisoptera	Corduliidae	*Hemicordulia*	*Australiae*										
	Libellullidae	*Diplacodes*	*Bipunctata?*										
Coleoptera	Dytiscidae	*Necterosoma*	*Darwini adults*	2	1		2		2			2	
		Necterosoma	*Darwini (larvae)*	2	2		2		2			2	rare
		Rhantus										rare	
		Allodessus (adult)											
	Haliplidae (larvae)												
	Hydrophilidae	*Berosus*	*sp. (adults)*									1	
		Berosus	*sp. (larvae)*			rare		rare	1	1		1	rare
	Limnichidae?											rare	
Trichoptera	Leptoceridae	*Oecetis*											
Hydracarina	Arrenuridae												
Fish		Craterocephalus	*Cuneiceps ?*		rare							1	1
	Sp. 2				rare				1				

Species richness				7	14	10	11	8	15	11	7	20	8

(Cont. Table 2)

SITE				**LA7-2**		**LA7-1**	**LA9-1**		**LA9-2**	**LA13**	
Higher Taxa	**Family**	**Genus**	**Species**	**Sept. 98**	**Jun-00**	**Jun-00**	**Sept. 98**	**Jun-00**	**Jun-00**	**Sept. 98**	**Jun-00**
Cladocera	Daphnidae	*Daphniopsis*	*pusilla*	2	4	2	3	1	4		4
Ostracoda	Cyprididae	*New genus*	*new sp.*		4	3	4	4	4		4
		Diacypris	*dictyote*	4		4	4	4?	4	rare	
		Diacypris	*whitei*	4	4	4	4	4	4	4	4
		Reticypris	*pinguis*		2			4			
		Cyprinotus	*edwardi*					3			
Copepoda	Cyclopoida	*Microcyclops ?*	*platypus ?*	rare			rare	4			
Anostraca	Branchipodidae	*Parartemia*		4			3			1	
Gastropoda (shells only 9/98)	Pomatopsidae	*Coxiella*	*gilesi*	1	2	1	2	2		2	2
Conchostraca							1				
Diptera	Chironomidae			rare	1	2	2	2	4		2
	Ceratopogonidae			rare		2					
	Stratiomydae					rare					
	Dolichopodidae										
Hemiptera	Corixidae										
Lepidoptera	Pyralidae					rare		1			rare
Zygoptera	Lestidae	*Austrolestes*	*annulosus*					4			
		Austrolestes	*sp.2*			rare		3			1
Anisoptera	Corduliidae	*Hemicordulia*	*australiae*		1			rare	1		
	Libellullidae	*Diplacodes*	*bipunctata?*					1			
Coleoptera	Dytiscidae	*Necterosoma*	*darwini adults*			3	3				
		Necterosoma	*Sp. 2 (adult)*					rare			
		Necterosoma	*darwini (larvae)*			3	3	2			rare
		Rhantus									
		Allodessus (adult)				rare					
	Haliplidae (larvae)				rare						
	Hydrophilidae	*Berosus*	*sp. (adults)*			1	rare				
		Berosus	*sp. (larvae)*	1	2	1	rare				2
	Limnichidae?										
Trichoptera	Leptoceridae	*Oecetis*						rare			
Hydracarina	Arrenuridae							rare			
Fish		*Craterocephalus*	*cuneiceps ?*		3						
	Sp. 2										1
Species richness				9	10	15	13	18	6	4	10

Abundance in sample:	rare = (1-2 individuals)
	1 = uncommon (3-10 individuals)
	2 = mod. common (10-100 individuals)
	3 = common (100-1000 individuals)
	4 = abundant (1000+ individuals)

Table 3. Summary table for presence absence of all known taxa (macroinvertebrates and fish) from rehydration samples and sampled at the lake during the September 1998 survey, monitoring during 1999, and June 2000 survey.

	Transect 1				*Transect 2*			*Transect 3*			*Transect 4*			*Transect 5*		
	LA1 'shallow'		LA1 'deep'		LA3 'Pool1'		LA3 'Pool 2'	LA5-2		LA5-1	LA7-2		LA7-1	LA9-1		LA9-2
	Sept. 98	Jun-00	Sept. 98	Jun-00	Sept. 98	Jun-00	Sept. 98	Sept. 98	Jun-00	Jun-00	Sept. 98	Jun-00	Jun-00	Sept. 98	Jun-00	Jun-00
Depth (cm)	30	30	100	85	7	40	3	5	33	11	5	18	12	5	85	5
pH	9.32	9.75	9.42/6.52	9.69/10.07	8.25	9.85/9.85	8.07	7.89	9.38/9.35	9.2	8.0	9.17	9.53	9.35/8.61	9.69/9.24	9.13
Redox potential	57	317	59/-80	320/302	1	309/307	13	17	285/289	309	67	336	338	8/17	320/322	324
Conductivity (mS/cm)	81.5	49.4	81.5/281.6	49.2/56.9	160.3	46.9/46.9	158.1	161.6	45.8/47.8	49.4	86.5	37.2	54.1	36.5/49.8	29.8/30.2	39.7
Temperature (℃)	17.35	18.84	16.51/43.7	17.11/22.5	14.2	15.16/15.28	17.8	23.7	13/16.5	12.41	21.79	18.37	20.84	16.76/21.7	15.06/15.17	13.7
Dissolved oxygen (mg/l)	8.50	10.5	7.8/5.4	9.4/13.4	7.6	8.5/8.4	7.8	7.3	8.2/5.2	6.8	7.4	10.4	12	9.7/12.4	11.6/8.8	6.6
Dis. oxygen (% sat.)	126.70	133.9	114.7/124.3	117.3/190.1	106.5	101.7/101.5	117.6	121.0	92.6/64.9	74.8	107.5	129.6	164.7	115/176	128.6/101.2	74.6
Turbidity (NTU)	16.8	2.2	10.8/59	1.5/7	129	0/0	600	599.3	0/64.8	0	590	0	3.6	8.9/91.5	0/0	0
Phosphate (mg/L)	0.05	0.03	0.13	0	0.045	0.02	0.06	0.07	0.02	0.02	0.11	0.01	0.05	0.04	0.1	0
Sulfate (mg/L)	4100	3500	3000	5000	5600	3700	5300	8000	3700	5100	5300	2700	3500	4600	1900	2800
TDS (evap) (mg/L)	67500	46000	210000	45000	11700	37000	114000	145000	39000	44000	58500	25000	39000	86600	21000	34000
Sodium (mg/L)	23000	11000	71000	11000	40000	11000	39000	51000	8900	9100	20000	7700	9400	29000	5000	6800
Potassium (mg/L)	780	350	1600	380	1400	380	1300	1600	320	290	670	290	290	1100	190	400
Calcium (mg/L)	1100	610	770	690	1500	730	1300	1400	630	550	980	600	720	1400	220	590
Magnesium (mg/L)	1300	880	3500	850	2300	850	2200	2700	730	710	940	500	560	1500	270	350
Iron (total) (mg/L)	0	0.38	99.0	0.41	3.3	0.39	17.0	1.3	0.34	0.33	8.1	0.28	0.26	6.5	0.12	0.16
Chloride (mg/L)	37000	19000	120000	19000	66000	18000	64000	79000	18000	18000	31000	12000	21000	48000	10000	14000
Nitrate as NO3 (mg/L)	10	0	10	0	10	0	10	20	0	0	10	0	0	10	0	0
Nitrite as NO2 (mg/L)	10	0	10	0	10	0	10	20	0	0	10	0	0	10	0	0
Carbonate as CaCO3 (mg/L)	1	6	1	18	1	12	1	1	15	18	1	0.1	0.1	1	27	0.1
Ammonia as NH3 (mg/L)	0.50	0.19	2.2	0.21	0.38	0.17	0.32	1.7	0.16	0.18	0.5	0.16	0	0.28	0.04	0.09
Silica as SiO2 (mg/L)	3.7	1	4.5	1	0.78	1	0.31	0.41	2	1	0.91	1	0.1	3.4	1	1

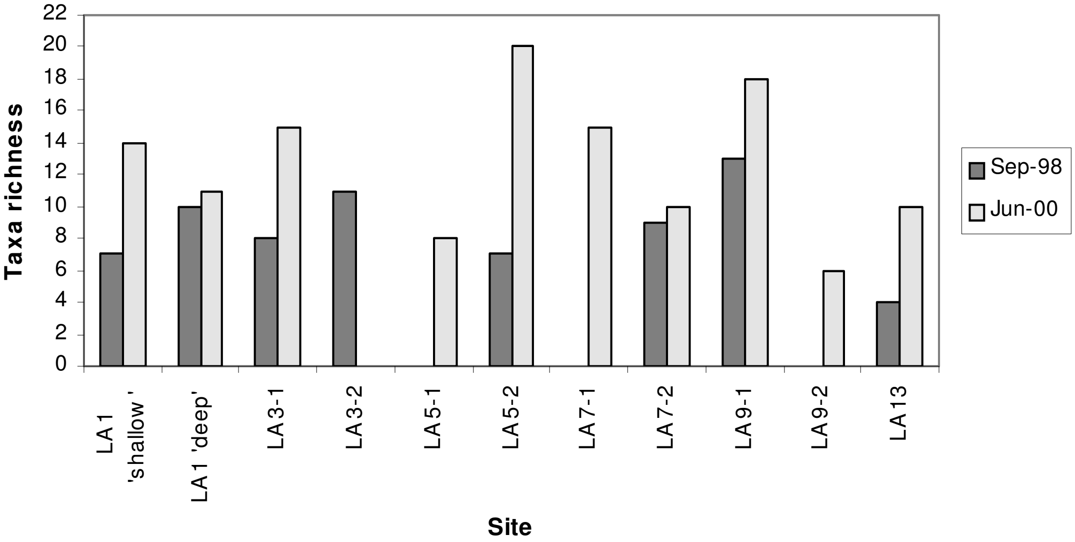

Figure 2. Number of macroinvertebrate taxa (plus fish) sampled at the different sites during the 1998 and 2000 surveys.

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            0.1430      0.2468      0.3506      0.4544      0.5582      0.6620
                 |           |           |           |           |           |
LA1-shall (    1)_
LA3-1     (    3)|____________
LA5-2     (    4)____________|__
LA1 deep  (    2)______________|______________________
LA9-1     (    8)____________________________________|_________
LA5-1     (    5)_____________________________________________|_______________
LA7-2     (    6)___________________                                         |
LA13      (   10)__________________|________________                         |
LA7-1     (    7)__________________________________|______                   |
LA9-2     (    9)________________________________________|___________________|
                 |           |           |           |           |           |
            0.1430      0.2468      0.3506      0.4544      0.5582      0.6620
```

Figure 3. UPGMA dendrogram showing the clustering of sites (2000 survey) based on macroinvertebrate presence/absence data.

2.2.2 *Water physico-chemistry*

Table 4 shows the physico-chemical data collected during the 1998 and 2000 surveys. The salinities (measured as electrical conductivity) were between 1.22 and 7.96 times lower in 2000 (29.8-56.9 mS/cm) than in 1998 (36.5-360 mS/cm). Accordingly, total dissolved solids (TDS), and the amounts of ions in the water, were also lower in 2000. The freshest site in 1998 (Transect 5, LA9-1:36.5 mS/cm top measurement) was also the freshest site in 2000 (29.8 mS/cm top measurement), however this site underwent the least dilution from 1998 to 2000 (only 1.22 X).

Table 4. Physico-chemical characteristics measured at sites containing surface water during the 1998 and 2000 surveys.

	Transect 1				*Transect 2*			*Transect 3*			*Transect 4*			*Transect 5*		
	LA1 'shallow'		LA1 'deep'		LA3 'Pool1'		LA3 'Pool 2'	LA5-2		LA5-1	LA7-2		LA7-1	LA9-1		LA9-2
	Sept. 98	Jun-00	Sept. 98	Jun-00	Sept. 98	Jun-00	Sept. 98	Sept. 98	Jun-00	Jun-00	Sept. 98	Jun-00	Jun-00	Sept. 98	Jun-00	Jun-00
Depth (cm)	30	30	100	85	7	40	3	5	33	11	5	18	12	5	85	5
pH	9.32	9.75	9.42/6.52	9.69/10.07	8.25	9.85/9.85	8.07	7.89	9.38/9.35	9.2	8.0	9.17	9.53	9.35/8.61	9.69/9.24	9.13
Redox potential	57	317	59/-80	320/302	1	309/307	13	17	285/289	309	67	336	338	8/17	320/322	324
Conductivity (mS/cm)	81.5	49.4	81.5/281.6	49.2/56.9	160.3	46.9/46.9	158.1	161.6	45.8/47.8	49.4	86.5	37.2	54.1	36.5/49.8	29.8/30.2	39.7
Temperature (°C)	17.35	18.84	16.51/43.7	17.11/22.5	14.2	15.16/15.28	17.8	23.7	13/16.5	12.41	21.79	18.37	20.84	16.76/21.7	15.06/15.17	13.7
Dissolved oxygen (mg/l)	8.50	10.5	7.8/5.4	9.4/13.4	7.6	8.5/8.4	7.8	7.3	8.2/5.2	6.8	7.4	10.4	12	9.7/12.4	11.6/8.8	6.6
Dis. oxygen (% sat.)	126.70	133.9	114.7/124.3	117.3/190.1	106.5	101.7/101.5	117.6	121.0	92.6/64.9	74.8	107.5	129.6	164.7	115/176	128.6/101.2	74.6
Turbidity (NTU)	16.8	2.2	10.8/59	1.5/7	129	0/0	600	599.3	0/64.8	0	590	0	3.6	8.9/91.5	0/0	0
Phosphate (mg/L)	0.05	0.03	0.13	0	0.045	0.02	0.06	0.07	0.02	0.02	0.11	0.01	0.05	0.04	0.1	0
Sulfate (mg/L)	4100	3500	3000	5000	5600	3700	5300	8000	3700	5100	5300	2700	3500	4600	1900	2800
TDS (evap) (mg/L)	67500	46000	210000	45000	11700	37000	114000	145000	39000	44000	58500	25000	39000	86600	21000	34000
Sodium (mg/L)	23000	11000	71000	11000	40000	11000	39000	51000	8900	9100	20000	7700	9400	29000	5000	6800
Potassium (mg/L)	780	350	1600	380	1400	380	1300	1600	320	290	670	290	290	1100	190	400
Calcium (mg/L)	1100	610	770	690	1500	730	1300	1400	630	550	980	600	720	1400	220	590
Magnesium (mg/L)	1300	880	3500	850	2300	850	2200	2700	730	710	940	500	560	1500	270	350
Iron (total) (mg/L)	0	0.38	99.0	0.41	3.3	0.39	17.0	1.3	0.34	0.33	8.1	0.28	0.26	6.5	0.12	0.16
Chloride (mg/L)	37000	19000	120000	19000	66000	18000	64000	79000	18000	18000	31000	12000	21000	48000	10000	14000
Nitrate as NO3 (mg/L)	10	0	10	0	10	0	10	20	0	0	10	0	0	10	0	0
Nitrite as NO2 (mg/L)	10	0	10	0	10	0	10	20	0	0	10	0	0	10	0	0
Carbonate as CaCO3 (mg/L)	1	6	1	18	1	12	1	1	15	18	1	0.1	0.1	1	27	0.1
Ammonia as NH3 (mg/L)	0.50	0.19	2.2	0.21	0.38	0.17	0.32	1.7	0.16	0.18	0.5	0.16	0	0.28	0.04	0.09
Silica as SiO2 (mg/L)	3.7	1	4.5	1	0.78	1	0.31	0.41	2	1	0.91	1	0.1	3.4	1	1

LA13 (Transect 7) underwent the greatest dilution from 1998 to 2000, going from 360 mS/cm in 1998 to 45.2 mS/cm in 2000 (a dilution factor of almost 8). The saltiest site in the 2000 survey was the hypolimnion of LA1-deep (Transect 1, 56.9 mS/cm), followed by LA7-1 ("lagoon" Transect 4, 54.1 mS/cm). The surface waters sampled in 2000 were also on average deeper, colder, more oxygenated and less turbid than in 1998. The pattern of ionic dominance, however, generally remained unchanged, with Na>Mg>Ca>K in the inlet channels and Na>Ca>Mg>K in the lake proper for the cations, and $Cl>SO_4>CO_3$ for the anions at all sites. Exceptions to the general cation dominance pattern were LA13 in Sept. 1998 where Na>Mg>K>Ca, and LA9-2 (lake proper) in June 2000 where Na>Ca>K>Mg. Carbonates (as CaCO3) were generally much more prevalent in June 2000 despite more dilute waters. Sulphate: chloride ratios were greater in June 2000 compared to September 1998. Both these trends occurred at all sites. The latter could be attributed to more oxidizing conditions reflected by larger (more positive) redox values (see Table 4). Available nutrients (PO_4, NO_3/NO_2, NH_4), particularly nitrogen, were very low in 2000 compared to 1998. At Site LA9-1, however, there was more available phosphorus in 2000 (0.1 mg/L) than in 1998 (0.04 mg/L). Accordingly, the site was very productive with dense and long *Ruppia* and much fresh and decaying filamentous green algae. There was also evidence of an algal bloom along the shoreline (see photo no. 5 in Appendix 3).

Figure 4 shows the UPGMA classification of sites from both surveys based on physico-chemical characteristics. In the first instance, there is a clear grouping of sites between the 1998 and 2000 surveys, with the extremely saline sites LA13 (1998) and LA1-deep (1998) forming a third group. The 1998 sites (apart from LA13 and LA1-deep) are further grouped into LA1-shallow and LA9-1 on the one hand (sites where *Ruppia* occurred), and LA3-1, LA3-2, LA5-2 and LA7-2 on the other. Among the 2000 sites, three sub-groupings can be recognised: 1) the channel sites LA1-shallow, LA3-1, LA5-2 and LA5-1; 2) the 'lake-proper' sites LA7-2, LA7-1, LA9-2 and LA13; and 3) LA1-deep and LA9-1. LA1-deep and LA9-1 are very much deeper than the remainder of the 2000 sites, and they also have similar pH and redox potentials. These groupings are very similar to the clusters formed in the classification of sites based on faunal assemblages (see below).

2.2.3 Relationship between macroinvertebrates and water physico-chemistry

As outlined in Horwitz *et al.* (1999), the variability of salinities in the surface waters of the Lake Austin wetland system produce identifiable fauna assemblages. In 1998 there was a slight significant inverse relationship between invertebrate taxa richness and electrical conductivity ($p<0.10$), while in June 2000 this relationship was still negative, and not significant. This reflects the smaller variability in invertebrate richness and EC between sites in 2000, when compared to 1998. However, if the data from both surveys are combined, there is a much stronger relationship ($r=0.55$; $p<0.02$ df=16) between EC and invertebrate taxa richness (Figure 5).

Figure 6 shows the SSH ordination of sites sampled in September 1998 and June 2000 based on faunal assemblages and the principal axis correlations with significant physico-chemical parameters. The only insignificant parameters were oxygen saturation, temperature, iron and silica. This means that these parameters poorly explain the distribution of the faunal assemblages. The sites are clearly distributed along a salinity gradient (i.e. EC, sodium, chloride, TDS, potassium, magnesium and sulfate), and similar to the classification of sites based on physico-chemistry, the 1998 and 2000 sites form two separate clusters, reflecting the link between water quality and the distribution of the faunal assemblages. Within the 2000 cluster, there is again a sub-grouping into lake-proper sites and channel sites.

```
             0.0465      0.1758      0.3051      0.4344      0.5637      0.6930
               |           |           |           |           |           |
1-shall (    1)_______
9-1 98  (   14)______|____________
3-1 98  (    5)_____             |
3-2 98  (    7)____|________     |
5-2 98  (    8)____________|___  |
7-2 98  (   11)_______________|__|_____________________
1-shall (    2)____                                   |
3-1 00  (    6)_  |                                   |
5-2 00  (    9)|__|___                                |
5-1 00  (   10)______|____                            |
7-2 00  (   12)____      |                            |
7-1 00  (   13)___|___   |                            |
9-2 00  (   16)______|__ |                            |
13 00   (   18)________|_|______                      |
1-deep00(    4)_______________ |                      |
9-100   (5    ) ______________|_|______________________|_____________________
1-deep98(    3)_____________________________________                        |
13 98   (   17)____________________________________|____________________ ___|
               |           |           |           |           |           |
          0.0465      0.1758      0.3051      0.4344      0.5637      0.6930
```

Figure 4. UPGMA dendrogram showing the clustering of sites based on physico-chemical attributes of surface waters (combined 1998 and 2000 data).

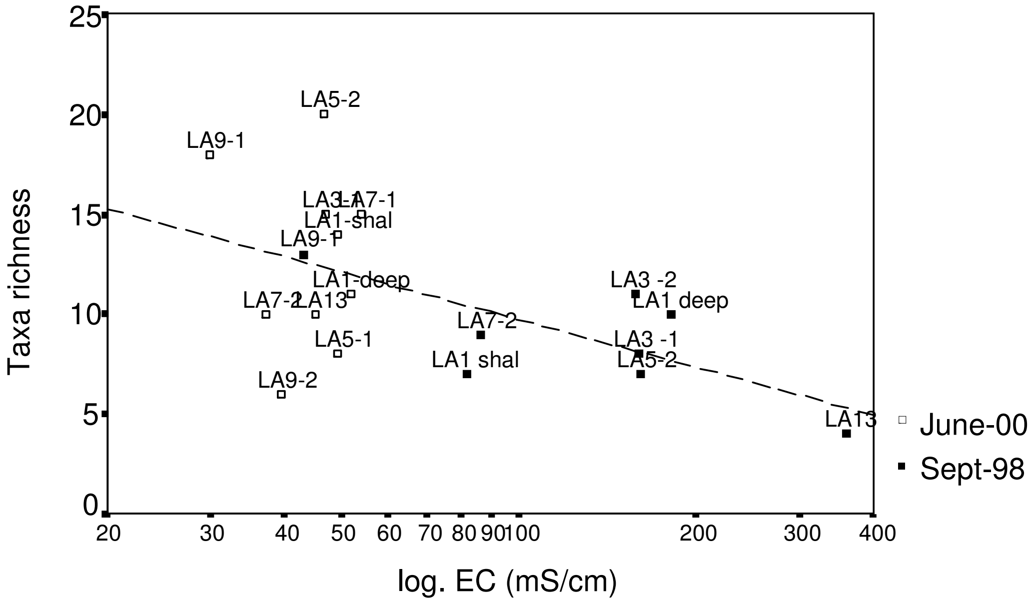


Figure 5. Correlation between invertebrate taxa richness and salinity (expressed as electrical conductivity) during the September 1998 and June 2000 surveys, showing also the linear trendline.

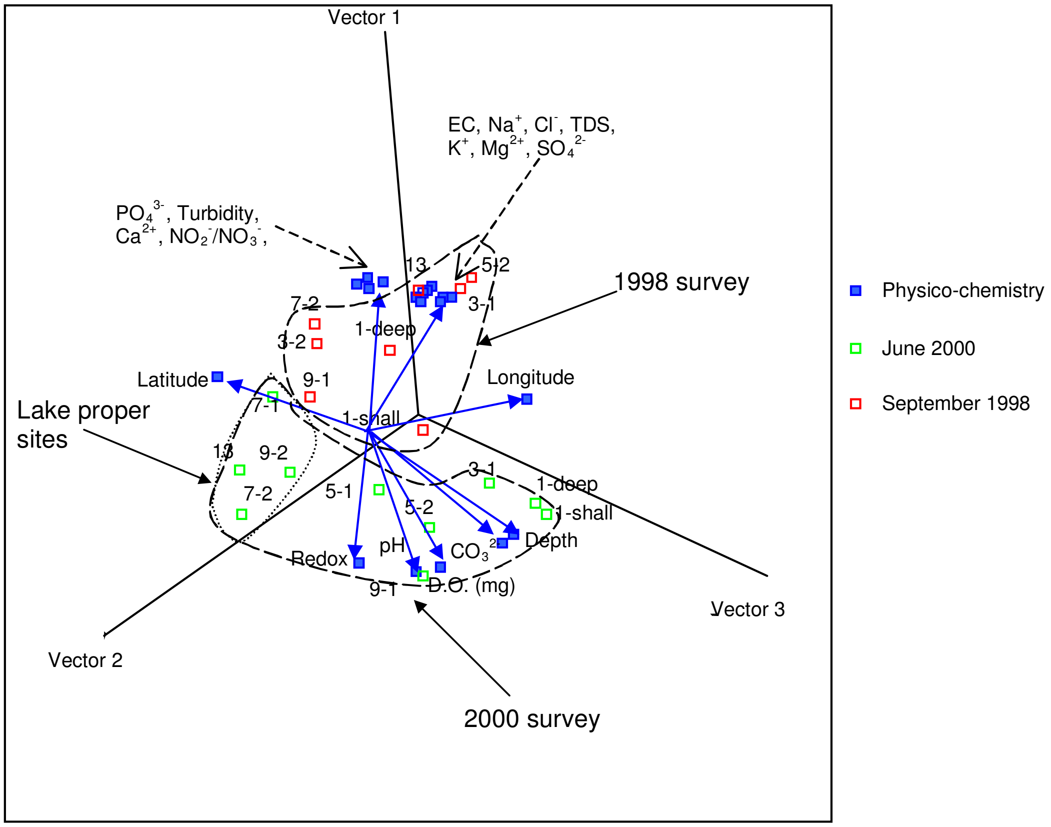


Figure 6. SSH ordination of sites based on Bray-Curtis dissimilarities showing the direction of significant correlations with physico-chemical attributes of surface waters. The long-dashed line indicates major clusters, the short-dashed line indicates the sub-cluster of lake-proper sites.

2.2.5 Comparison of June 2000 data with June 1999 data (Transects 1 and 5)

Table 5 lists macroinvertebrate species collected during June 1999 (Sommer *et al.* 2000) and June 2000 from Transects 1 and 5. Even though samples were collected for the period April to October in 1999, the June 1999 sample was selected as the month for comparison since it would be the best seasonal match for the June 2000 survey sample. (Another form of analysis is presented in Table 3 where total richness for 1999 are presented for all transect sites combined.)

Species richness was higher at both transects during the 2000 survey (Transect 1: 16 vs. 10; Transect 5: 18 vs. 12), and during both sampling occasions only two more species were found at Transect 5 than at Transect 1). At Transect 1, Daphnidae and Cyclopoida were found only in the June 1999 samples, while the ostracods *Reticypris pinguis*, *Heterocypris* (n. sp.)and *Cyprinotus edwardi*, the snail *Coxiella*, sandfly larvae Ceratopogonidae, caterpillars Pyralidae, damselfly nymphs *Austrolestes* (sp. 2), and a second species of fish were found only in the June 2000 samples. At Transect 5, *N. darwini* adult beetles and an unidentified species of fish were found only in the June 1999 samples, while the ostracod *Diacypris dictyote*, caterpillars Pyralidae, damselfly nymphs *Austrolestes* (sp. 2), the dragonflies *Hemicordulia australiae* and *Diplacodes bipunctata*(?), *Nectersoma* beetles (sp. 2), caddisfly larvae Leptoceridae, and a freshwater mite, Arrenuridae were found in June 2000 samples only. *Parartemia* were not found in any of the June 2000 samples, nor the June 1999 samples.

Table 5: Comparison of macroinvertebrates and fish collected during June 1999 and June 2000 from Transects 1 and 5

				Transect 1		*Transect 5*	
SITE				**LA1**		**LA9-1**	
Higher Taxa	**Family**	**Genus**	**Species**	**Jun-99**	**Jun-00**	**Jun-99**	**Jun-00**
Cladocera	Daphnidae	*Daphniopsis*	*pusilla ?*	x		x	x
Ostracoda	Cyprididae	*New genus*	*new sp.*	x	X	x	x
		Diacypris	*dictyote*	x	x		x
		Diacypris	*whitei*	x	x	x	x
		Reticypris	*pinguis*		x	x	x
		Heterocypris	*n. sp.*		x		
		Cyprinotus	*edwardi*		x	x	x
Copepoda	Cyclopoida	*Microcyclops ?*	*platypus ?*	x		x	x
Anostraca	Branchipodidae	*Parartemia*	*n. sp.*				
Gastropoda	Pomatopsidae	*Coxiella*	*gilesi*		x	x	x
Conchostraca							
Diptera	Chironomidae			x	x	x	x
	Ceratopogonidae				x		
	Stratiomydae						
	Dolichopodidae						
Hemiptera	Corixidae						
Lepidoptera	Pyralidae				x		x
Zygoptera	Lestidae	*Austrolestes*	*annulosus*	x	x	x	x
		Austrolestes	*sp.2*		x		x
Anisoptera	Corduliidae	*Hemicordulia*	*australiae*				x
	Libellullidae	*Diplacodes*	*bipunctata?*				x
Coleoptera	Dytiscidae	*Necterosoma*	*darwini adults*	x	x	x	
		Necterosoma	*Sp. 2 (adults)*				x
		Necterosoma	*darwini (larvae)*	x	x	x	x
		Rhantus					
		Allodessus (adult)					
	Haliplidae (larvae)						
	Hydrophilidae	*Berosus*	*sp. (adults)*				
		Berosus	*sp. (larvae)*				
	Limnichidae?						
Trichoptera	Leptoceridae	*Oecetis*					x
Hydracarina	Arrenuridae						x
Fish		Craterocephalus	*cuneiceps ?*	x	x		
	Sp. 2				x	x	
Species richness				10	16	12	18

2.3 Discussion

Major findings of the June 2000 survey (with comparisons to September 1998 baseline survey, see Horwitz *et al*. 1999, or monitoring data for 1999, see Sommer *et al.* 2000) are presented below in the light of potential impacts of the dewatering discharge. The critical features of the design of the sampling were that impacts of the discharge would be evidenced at Transects 3, 2, and 1 (in this order, since it reflects proximity, and therefore likelihood and severity of dewatering impacts). Transect 5 with its inlet channel and lake proper, and Transect 4 with its lagoon and lake proper, were designed to act as controls, supposedly too far away from dewatering to be influenced by it.

Therefore, patterns detected for observations and measurements for Transects 3, 2 and/or 1 between June 2000 and September 1998 (numerated below) might be attributed to dewatering discharge if the same pattern did not occur at Transects 4 and 5.

1. All transects were substantially wetter in June 2000 compared to other sampling periods.
2. All waters sampled along transects in June 2000 were more dilute than the September 1998 sampling period.

Neither of these two differences, as stated, could be attributed to the dewatering discharge; both are likely to be the result of substantially wetter (rainfall) phases for the region between September 1998 and June 2000. Both patterns were observed at Transects 4 and 5 as much as anywhere else.

3. Ionic concentrations differed between June 2000 and September 1998 despite more dilute waters: for example carbonate values (as $CaCO^3$*) increased and sulphate concentrations increased.*

Neither of these differences can be unequivocally attributed to dewatering discharge. Carbonate values are elevated in the discharge (Appendix 1), but an anomalously high reading at Transect 5 in the channel suggests that there might be other reasons for elevated levels. Similarly, elevated levels of sulphate at Transects 4 and 5 were found and match elevated (strongly positive) levels of redox at all sites, suggesting that the elevated levels were due to changes in oxidizing conditions in the waters rather than dewatering discharge *per se*.

4. For the aquatic fauna, significant differences occurred between the two sampling periods:

- *species richness was greatly enhanced in all June 2000 samples, predominantly the result of increased numbers of aquatic insect species, but also the result of finding ostracod species which had previously (1998) only been found hatching from rehydration trials.*
- *no Parartemia shrimp were collected from June 2000, while they were abundant in September 1998.*

These differences are difficult to attribute to the impacts of dewatering discharge because increased species richness occurred at all sites on all transects, with no differences detected between sites likely to be impacted, and controls. Similarly, *Parartemia* were absent from all sites, and also absent from June 1999 monitoring data, suggesting that its absence may be a seasonal factor. In addition, strong similarities exist for faunal assemblages between 1998 and 2000: the faunal assemblage in inlet channels is identifiably different to that assemblage in the lake proper, for both sampling periods (in terms of both water chemistry and aquatic fauna). If dewatering discharge were impacting on proximal sites, this relationship would not be expected in June 2000 samples.

The failure of the snapshot survey in June 2000 to detect impacts as a result of dewatering discharge could be attributed to three causes (or combinations of them):

- there was no impact;
- the experimental design was inadequate to detect changes as a result of the discharge;
- there was no measurable impact because unpredictably wet seasons had diluted the discharge by the time monitoring had occurred in 1999, and by the time the snapshot sampling was undertaken in June 2000.

We reject the first mentioned possibility because changes arising from a dewatering discharge of this magnitude should have been detectable, at least in some way.

Aspects of the sampling/monitoring design can be subject to scrutiny if it can be demonstrated that dewatering discharge has reached Transects 4 and 5. If so, all transects will show the same trends and patterns. Indeed it is quite possible that over time, and particularly in wetter years when the lake waters are mixing well, Transect 4 and possibly even the inlet channel of Transect 5 might be influenced by dewatering discharge. A detailed hydrological study of the lake will be able to resolve this question. In addition, this survey has demonstrated that the ability to trace discharge waters chemically would be a valuable adjunct to existing monitoring protocols. At present ionic analyses may be too crude to typify the discharge. Other minor and trace ions, and their ratios, may be able to distinguish between lake waters and discharge waters.

We consider that the most obvious effects of the discharge on the lake system will occur when the discharge is the significant component of the surface waters of the lake, i.e. at times of low waters in the inlets and lake proper. At these times it will provide relatively constant hypersaline aquatic conditions, rather than fluctuating salinities according to rainfall and evapoconcentration. Under these circumstances it is anticipated that Transect 5 will play an adequate role as a control site.

Finally, we consider that the best way to detect changes associated with dewatering discharge is to monitor the water chemistry and aquatic fauna over time (as recommended by Horwitz *et al*. 1998), rather than at one snapshot in time. Monitoring along these lines was conducted in 1999 (see Sommer *et al.* 2000) but some shortfalls in recording water quality have made these data of limited value. Studies conducted between 1998 and 2000 now provide an excellent set of data for interpreting changes in faunal assemblages and water chemistry.

2.3.1 Conclusion

It seems likely that the impacts of dewatering discharge are yet to be felt in Lake Austin and its surrounding wetand systems, mainly because of the coincidental large rainfall events and wet seasons in the two years since discharge has commenced. There is no doubt that close attention will need to be paid to water chemistry and aquatic invertebrate assemblages in future monitoring programs.

In order to trace the discharge more effectively, a more detailed hydrological study of the lake is required, and more complex analyses of ionic concentrations in the discharge waters themselves, and inlet and lake waters, will give greater power to existing monitoring outcomes.

3. Changes to Vegetation & Flora

3.1 Methods

3.1.1 Field Survey & Laboratory Analysis

Monitoring sites were established during the initial survey of flora and vegetation in September 1998. Altogether 33 sites were established along 7 transect lines.

Sites were revisited in June 2000 and the following re-measured at most sites:

- Cover and abundance of perennial plant species
- Height, width and % volume of plant that is living (% health) for tagged perennial shrubs
- Field pH

Soils were also collected (from the top 1-4 cm) to enable laboratory measurements of conductivity and pH using appropriate probes place in a 1:5 mix of soil and de-ionised water. The % weight of > 2mm particle size was also measured following sieving. Photographs were taken of each site - print and digital copies have been forwarded to the Environmental Section at Big Bell Mine.

All of the above measurements were performed here using the same methods and by the same personnel as the initial study.

3.1.2 Data Analysis

Change in the above measured parameters was calculated and expressed in absolute terms and in terms of percentage of initial values. As the degree of change in soil conductivity and particle size was extremely variable, it was expressed as a percentage of final values to reduce this variability.

Mean change in each parameter was calculated for sites located close to the discharge point ('impact' sites; transects 1, 2, 3 & 6) and for sites located some distance from this point ('control' sites; transects 4, 5 & 7). Student t-tests were then performed to test for significant differences between these two groups of sites, with percentage data arcsine transformed before testing. As some species of shrubs were absent at several sites, t-tests could not be performed on average site data on plant condition, so individual plant measurements were pooled for each of the two groups of sites (control vs. impact). In other words individual plants were replicates rather than sites.

3.2 Results

3.2.1 Changes in Species Composition

Most of the change in plant species composition has been in terms of annuals and short-lived perennials, rather than woody perennials. Daisies (family Asteraceae) dominated the short-lived flora during September 1988, whereas grasses (family Poaceae) were the most common component in June 2000. This is to be expected as September 1988 followed good winter rains, whereas the sampling this year followed on from well above average summer and autumn rains (Figure 1). Compositional differences in the short-lived flora in response to season of deluge are well known for arid areas. Common grass species recorded during the June 2000 survey were: *Aristida contorta, Dactyloctenium radulans, Enneapogon carerulescens, Eragrostis lanipes, E. helmsii, E. dielsii*, and *Setaria dielsii*.

There was very little change in perennial species composition at sites, although the abundances of many species changed to varying degrees (see 2.2 & 2.3 below). Previously unsampled perennial species were found in plots at sites 1-3 (*Sida* sp.), 2-4 (*Maireana amoena*), 2-5 (*M. villosa*), 3-2 (*Maireana* sp.), 3-3 (*M. amoena*), 5-1 (unknown species; seedling), 5-2 (*Acacia victoriae*), and 5-7 (*M. amoena*). Previously sampled species were unable to be found within plots at sites 3-1 (*Atriplex acutibractea*), 5-2 (*A. amnicola*), 5-8 (*Halosarcia doleiformis*), and 6-1 (*M. amoena*). These species were consistently of low relative abundance; they never comprised dominant or common species within plots.

3.2.2 Changes in Vegetation Structure

No significant difference was found in the change to cover and abundance of perennial species between 'control' and 'impact' sites (Table 6). On average, the cover of perennial species increased by 15% at 'impact' sites, but decreased by 2.5% at 'control' sites. Variation in the change of cover was highly variable between sites, with standard error equal or exceeding the means (Table 6). There was also no significant difference in the overall change to samphire species (*Halosarcia*, *Sclerostegia* & *Sarcocornia* spp.) between 'control' and 'impact' sites (Table 6).

Table 6. Changes in vegetation and environmental attributes from September 1998 to June 2000 at sites located in impact zone and those located some distance from discharge (controls). Comparisons between these two groups of sites are also shown. Change is measured in terms of mean % decrease or increase from the original (i.e. Sept. 1998) , except where indicated by ^ when it is expressed in terms of change relative to the final (i.e. June 2000) measurements.

Attribute	Mean % Change for Impact Sites	Mean %Change for Control Sites	df	$\lvert t \rvert$	prob
Total perennial cover	14.7 (13.6)	-2.5 (4.7)	27	1.09	0.28
Total perennial abundance	27.6 (14.1)	20.6 (7.5)	22	0.44	0.67
Cover of Samphires	4.1 (3.9)	-4.1 (5.2)	26	1.28	0.21
Abundance of Samphire	17.2 (8.7)	16.1 (10.3)	26	0.09	0.93
H.indica height	-9.6 (4.0)	-9.9 (7.7)	$8^{\#}$	0.03	0.98
H.indica width	6.6 (7.4)	1.5 (8.2)	11	0.46	0.66
H.indica volume	19.2 (16.3)	10.2 (15.4)	11	0.40	0.70
H.indica health	-16.6 (6.6)	-11.2 (15.7)	$7^{\#}$	0.32	0.76
H. halocnemoides height	6.8 (5.1)	13.0 (1.9)	9	0.88	0.40
H. halocnemoides width	13.6 (7.8)	19.7 (4.7)	9	0.54	0.60
H. halocnemoides volume	39.8 (25.0)	61.9 (7.6)	9	0.65	0.53
H. halocnemoides health	12.2 (4.1)	30.3 (7.1)	9	2.39	0.041*
Conductivity ^	32.7 (13.8)	57.7 (9.5)	27	1.43	0.17
PH	-8.6 (1.9)	-9.2 (1.9)	27	0.21	0.84
particles > 2mm^	3.9 (13.0)	17.8 (18.5)	27	0.63	0.53

3.2.3 Changes in Plant Condition

With two exceptions, no significant differences were found between the change in plant condition within the impact zone and that in the control zone for each of the 5 most common samphire species (Table 7). Generally, there was wide variation in how plants changed over the 21 month period, with considerable differences between both species and sites (see Appendix 4 for raw data). Change to each species is summarised below in the context of their distribution within the landscape.

Of the 5 species, *Halosarcia indica* subsp. *leiostachya* showed the greatest average decline in height (~11% decline relative to initial measurements). Plant width and volume however increased on average. Plants were less healthy over the 21 month period, with a 14% decline in relative terms, however the number of plants increased, especially seedling numbers. There was considerable variation in these parameters between sites and individual plants, with no significant differences between control and impact sites. This species is most common along shorelines and is presumably tolerant of high salinity and occasional inundation. Much of the community dominated by this species was inundated during the early part of 2000, with the lowest lying parts still flooded at the time of the June field visit. Many plants were covered with thick mats of *Ruppia* and macro-algae (see photos 1 to 4). The weight of these deposits is likely to have contributed to the lower heights and greater widths recorded. Many plants were observed to have died recently or had growing tips or lateral branches showing recent death (photo 4). This should be weighed up against large numbers of seedlings at many sites where waters had recently subsided (photo 3).

Sclerostegia tenuis is commonly associated with *H. subsp. leiostachya* around the edge of the lake and inlet channels. As with *H. subsp. leiostachya* it has declined in height and health at many sites. The decline in health within the impact zone (~6% in relative terms) was however significantly greater than in the control zone, where plants actually improved in health substantially. Again plants had been inundated recently and were often covered with *Ruppia* and algae.

Table 7. Changes in mean plant condition for plants in impact and control zones between September 1998 to June 2000 in both absolute and proportional terms. Comparison of means also shown. (NB degrees of freedom have been modified where variances were unequal at $p<0.05$). Standard errors of mean are given in parentheses.

Species	**Parameter**	**Change in Impact Zone**	**Change in Control Zone**	**df**	**|t|**	**Prob.**
H. indica subsp. *Leiostachya*	Height (m)	-0.057 (0.027)	-0.032 (0.0019)	40	0.70	0.49
	% Height	-11.6 (5.7)	-9.9 (6.2)	40	0.21	0.84
	Width (m)	0.036 (0.025)	0.016 (0.022)	40	0.58	0.56
	% Width	5.6 (7.2)	1.5 (7.4)	40	0.39	0.70
	Health (% living)	-12.0 (4.2)	-9.9 (6.5)	40	0.28	0.78
	% Health	-16.7 (6.8)	-11.1 (10.3)	40	0.47	0.64
	Volume (m^3)	0.019 (0.016)	0.0052 (0.0079)	40	0.97	0.34
	% Volume	17.9 (14.2)	10.2 (14.5)	40	0.71	0.38
	Cover (% area)	-0.5 (1.2)	-2.0 (1.6)	10	0.77	0.46
	Abundance (no. plants/plot)	14.0 (8.7)	1.0 (6.9)	10	1.18	0.27
H. halocnemoides	Height (m)	0.019 (0.014)	0.055 (0.015)	31	1.65	0.11
	% Height	6.8 (3.8)	10.2 (4.0)	31	0.59	0.56
	Width (m)	0.037 (0.019)	0.201 (0.068)	13	2.34	0.037*
	% Width	11.1 (5.6)	18.8 (5.5)	31	0.91	0.37
	Health (% living)	6.0 (2.3)	15.8 (3.7)	31	2.38	0.024*
	% Health	12.3 (4.8)	30.3 (7.9)	31	2.06	0.048*
	Volume (m^3)	0.037 (0.020)	0.425 (0.19)	11	2.03	0.067
	% Volume	39.9 (17.6)	62.2 (16.1)	31	0.85	0.40
	Cover (% area)	-0.50 (0.33)	0.75 (0.48)	9	2.22	0.053
	Abundance (no. plants/plot)	-2.6 (3.6)	-1.0 (0.7)	9	0.32	0.76
H. doleiformis	Height (m)	0.007 (0.027)	0.008 (0.018)	11	0.04	0.97
	% Height	5.3 (12.6)	1.9 (4.8)	11	0.32	0.76
	Width (m)	0.053 (0.022)	0.059 (0.026)	11	0.13	0.90
	% Width	24.7 (11.0)	13.3 (4.7)	11	1.11	0.29
	Health (% living)	-13.3 (3.3)	-5.2 (3.0)	11	1.38	0.20
	% Health	-25.0 (4.8)	-9.5 (5.5)	11	1.47	0.17
	Volume (m^3)	0.0069 (0.0035)	0.028 (0.015)	11	0.78	0.45
	% Volume	73.1 (50.9)	34.3 (14.6)	11	1.05	0.32
	Cover (% area)	-5.0	-2.3 (2.5)	3	0.48	0.67
	Abundance (no. plants/plot)	-20.0	-2.8 (2.8)	3	2.81	0.068
H. pergranulata	Height (m)	-0.15 (0.18)	0.009 (0.006)	4	0.87	0.43
	% Height	-16.6 (21.2)	2.2 (1.3)	4	0.89	0.43
	Width (m)	-0.074 (0.090)	0.031 (0.012)	4	1.16	0.31
	% Width	-21.8 (21.8)	6.5 (2.5)	4	1.29	0.27
	Health (% living)	-5.0 (6.1)	1.7 (6.0)	12	0.72	0.49
	% Health	-22.6 (23.3)	4.5 (12.7)	12	1.12	0.28
	Volume (m^3)	0.019 (0.061)	0.014 (0.060)	4	0.09	0.94
	% Volume	-13.8 (33.8)	16.8 (6.5)	12	1.19	0.26
	Cover (% area)	-0.5 (0.5)	-5.0 (5.0)	3	0.70	0.54
	Abundance (no. plants/plot)	-2.0 (1.0)	-10.7 (9.7)	3	0.69	0.54
S. tenuis	Height (m)	-0.017 (0.020)	0.032 (0.012)	7	2.17	0.067
	% Height	-7.6 (8.3)	9.8 (3.4)	7	2.34	0.052
	Width (m)	0.003 (0.012)	0.043 (0.025)	7	1.08	0.32
	% Width	3.8 (6.0)	18.1 (10.6)	7	0.89	0.41
	Health (% living)	-3.3 (3.3)	15.0 (4.2)	7	2.81	0.026*
	% Health	-5.7 (5.7)	36.1 (9.8)	7	2.82	0.026*
	Volume (m^3)	-0.0009 (0.0077)	0.0083 (0.0038)	7	1.64	0.15
	% Volume	-0.8 (9.8)	58.7 (31.3)	7	1.29	0.24
	Cover (% area)	2.0	5.0 (0)		na	
	Abundance (no. plants/plot)	10	2.0 (0)		na	

H. doleiformis occurs adjacent to *H. subsp. leiostachya* at slightly higher elevations where soil salinities are appreciably lower and inundation is not known to occur, although recent lake levels have come close to flooding this species. Plants of this species generally have grown over the inter-sampling period, although the health of plants, and overall cover and abundance, have declined. No significant differences between impact and control zones were found.

H. halocnemoides is common on the sandy soils on the tops of the levee banks which rise around 1/2 metre above the lake bed and are above the highest water mark. Plants of this species grew substantially over the inter-sampling period, with volumes increasing by around 50% on average. Plants within the control zone grew significantly wider than in the impact zone. This was found in absolute terms only and probably reflects the fact that larger plants were initially sampled in the control zone. Improvement in the health of plants was significantly greater in the control zone compared to impact zone, both in relative and absolute terms.

H. pergranulata subsp. *pergranulata* is restricted to highly alkaline and heavy soils in low lying areas set back from major drainage lines. Soils of this community were commonly waterlogged or very moist during June 2000, although no evidence of recent inundation was found. How these areas are waterlogged is not clear. It may be due to entry of water from the main inlet channels via small drainage lines. Alternatively, groundwater during wetter periods may rise to the surface. Plants of this species declined on average in the impact zone, whereas there was slight improvement generally in the control zone. Large plant-to-plant variation in the degree of change meant differences between control and impact zones were not significant.

3.2.4 Changes in Environment

Given that ions of lake waters are predominantly Na and Cl (Table 4), electrical conductivity measurements of topsoil are used here as an indicator of soil salinity, although other salts in addition to NaCl would make a small contribution to the levels of conductivity measured. Soil salinities have risen at most sites sampled over the 21 month inter-sampling period. On average, this has been in the order of 50% of final values. Some sites increased several fold in salinity, whilst others had no change or even slight decreases. No significant difference was found between percentage increases in salinity levels of control compared to impact sites (Table 6). Similarly, no significant difference was found between sites which had been inundated or waterlogged in the first half of 2000 and those which had not (df=27; t=0.37; p=0.71). In absolute terms however, salinity increases were significantly greater at flooded sites compared to those not flooded (df=27; t=2.42; p=0.023), although initial salt levels were substantially higher at low-lying sites.

pH levels have declined to a fairly consistent degree across sites. Again no significant difference was found between control and impact sites (Table 6), as well as between flooded and non-flooded sites, for changes in pH in both absolute and relative terms.

The proportion of topsoil with greater than 2 mm particle size also increased on average by ~10% of final values. There was no significant difference between control and impact sites, as well as between flooded and non-flooded sites.

Photo 1. Inundated edge near site 7-3. *Halosarcia indica* is the dominant species.

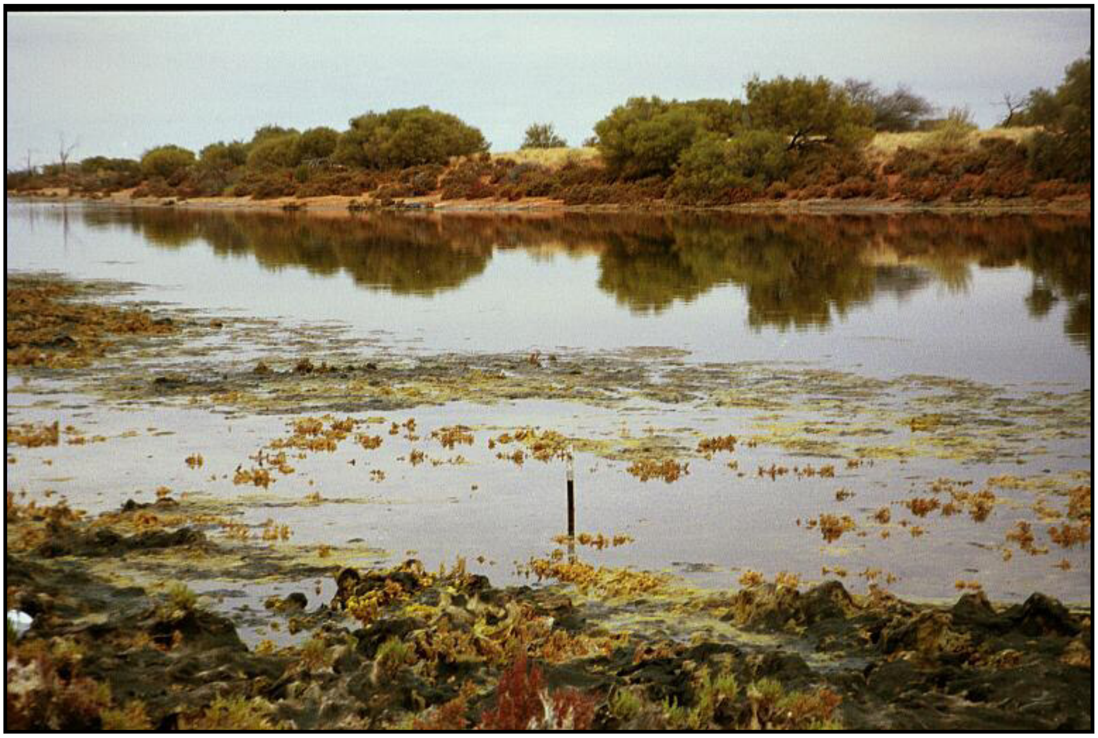


Photo 2. Inundated edge of saltmarsh along transect 5 due to flooded inlet channel. The post in the foreground is the middle of site 5-4. Species flooded are *Halosarcia indica* and *H. doleiformis*. Extensive mats of *Ruppia* sp and macroalgae are evident.

Photo 3. Edge of samphires at site 7-2. Ruppia mats cover plants, especially those closer to the water. High water mark of April 2000 is evident by limit of Ruppia deposits. Several plants have dead or dying branches.

Photo 4. Close up of one plant in photo 3 showing large degree of recent plant death

3.3 Discussion

High lake levels experienced during the first half of 2000 have resulted in the inundation of the outer edge of the extensive saltmarsh vegetation which surrounds the main lake bed and inlet channels. A strip of vegetation between 5-20 m wide has been inundated up to a depth of between 1 to 30 cm. Inundation was at its greatest in April 2000 (A. Wilkeis, pers. comm.), and water levels have gradually receded since. At the time of the June 2000 survey, some saltmarsh plants at the outer edge were still completely submerged; these would have been underwater for several months (photos 1 & 2). Abundant growth of *Ruppia* sp. and macroalgae in shallow waters have covered most inundated plants. This has persisted around plants following recession of water levels (photos 3 & 4). The upper boundary of *Ruppia* covered plants corresponded to high water mark of autumn this year and clearly demarcated the upper limit of the inundation.

A fair degree of plant death and damage was recorded along the inundated edge. This is presumed to be a response to some combination of flooding, water salinity and/or smothering by aquatic plants. It was observed that heavily smothered plants were generally the most unhealthy and damaged; also greater impact was noted on smaller plants compared to plants which remained emerged above the water. The dominant species of the edge community is *Halosarcia indica* subsp. *leiostachya* - this species declined substantially at almost all sites studied. No significant difference between impact and control sites was detected, indicating discharge can not be held responsible for the decline in this species. The secondary and minor species of the edge community, *Sclerostegia tenuis*, also declined in health and height, but decline in health was significantly greater within the impact zone compared to control zone. This indicates that discharge is having a detrimental impact on this species. The lack of statistical power (df=7) however suggests we should be cautious with this finding, and it is recommended that more intensive sampling of this species occur to compare health in the impact versus control zone.

The decline and death of the mature samphire plants of the edge community needs to be weighed up against large numbers of recent germinants, with hundreds of small seedlings found at some sites. It seems likely that inundation of the edge, a rare event as it is, is a cue for extensive germination and recruitment of new individuals into the community. Although most seedlings would be expected to die, there only needs to be a small proportion surviving to maturity to replace those lost as a result of flooding. Seedlings located some distance from mature survivors would more likely survive given reduced competition for resources. The dynamic nature of this outer edge of the saltmarsh can also be envisaged. If sufficient time without flooding occurs, plants may be able to colonise further out into the salt lake and obtain enough height to survive the next major flooding. When high water returns, those too far out or too small will get completely inundated and most likely die. Obtain sufficient height to remain emerged above the next flood and the chances of survival are greater. The position and density of the saltmarsh edge may therefore be the product of recent flooding history.

Just above the inundated edge, *Halosarcia doleiformis* often dominates the midslopes of the small levee banks which surrounds the lake and inlet channels, with *H. halocnemoides* dominant on the sandy crests of the banks. Both these species grew significantly over the inter-sampling period, although there was an average decline in health of the former species compared to an improvement in health of the latter. No differences were detected between control and impact sites, with the exception of width and health in *H. halocnemoides.* This change has been attributed to greater initial size of the plants selected at control sites, rather than being an impact of discharge. Again more intensive monitoring is suggested for these two species.

Inland from the levee banks on low-lying, heavy, alkaline soils, *H. pergranulata* dominates. Compared to initial sampling, there was measurable decline in this species at several sites. A possible cause of this decline is grazing and trampling by sheep given the preponderance of such livestock in this community. Discharge is discounted as a cause as no significant difference in the degree of decline was detected between control and impact zones.

Only minor changes in the composition of perennial species was measured at sites, with a small number of sites either losing or gaining one to several minor species. Dominant species did not change at sites. The substantial change in the short-lived flora was related to seasonal differences in the incidence of saturating rains.

Levels of soil salinity have increased significantly on average. At some sites it has increased several fold. The reason(s) for such an increase is unknown. The fact that no significant difference in the degree of increase was found between control and impact sites suggests that discharge cannot be held responsible. However, if hypersaline discharge water has been effectively dispersed across the whole lake, inundation of the lake edge may be implicated. Indeed, water salinity was reasonable consistent across the lake in June 2000 (Table 4), and was around the level of seawater (~20 mS/cm). This would be expected to raise soil salinity levels when waters recede, especially given that soil salinity was generally below 20 mS/cm at the edge before inundation. In terms of percentage increase in salinity, no significant difference was detected between recently inundated sites and non-inundated sites. Perhaps, mobilisation of salts in the topsoil due to large rainfall events during summer/autumn of this year has occurred at non-inundated sites. In absolute terms, significantly greater increases in salinity occurred at flooded sites compared to those not flooded, although this would be expected as low-lying sites were one to two orders of magnitude greater in terms of initial salinity levels. Causes of the decrease in pH recorded at most sites are unknown - they are especially difficult to understand given that pH tends to increase with salinity.

In summary, it is difficult to be conclusive with respect to the impact discharge is having on the fringing vegetation. There is a possibility that discharge has contributed to higher than normal water salinity for lake levels as high as they were recorded during the first half of 2000. This should be the time that lake waters are at their least salty, although in the absence of long-term monitoring, expected salinity levels are somewhat speculative. The degree to which higher than expected water salinity may have contributed to the decline in the edge vegetation can also only be speculated upon. Furthermore, the direct impact of water salinity on plants cannnot be readily separated here from that of oxygen depletion and smothering by aquatic plants. Synergistic effects are also quite likely. Nonetheless, the decline in samphire plants as a result, either directly or indirectly, of inundation is cause for concern. On-going monitoring of the recovery of existing plants and recruitment of new individuals at the saltmarsh edge is essential to gauge the long-term impacts of inundation and the degree to which the decline observed this year is part of natural vegetation dynamics.

Although water salinity was fairly uniform across the lake at June 2000, the possibility remains that areas close to discharge point will receive waters of much higher salinity than elsewhere when lake levels are either low or dry, and this may be having a detrimental impact over the longer term. No such impact was detected in the dominant species of the edge, although the subdominant species was more impacted close to the discharge point than further away. The extension of the discharge pipe further from the edge, if it is maintained, is expected to lessen the chance of salty water reaching the saltmarsh vegetation.

4. Conclusions & Recommendations

This study has detected major changes in many biological and physico-chemical parameters of Lake Austin and fringing wetlands between September 1998 (date of baseline study of Horwitz et al. 1999) and June 2000, including changes in:

- Species composition of aquatic fauna and vegetation (particularly annual and short-lived species);

- Cover, abundance and health of dominant plant species, including death and damage to samphires of the lake margins;
- Salinity, pH and other physical and chemical features of lake water and fringing topsoil; and
- Lake levels.

None of these changes can unequivocally be attributable to discharge of hypersaline water into the lake (resulting from dewatering of the mine at Cuddingwarra). In other words no impacts due to discharge were detected. This conclusion is based on the assumption of higher salinities and greater inundation in areas close to the discharge ('impact' sites) compared to areas remote from the discharge (control sites). Substantial, above-average rainfall since the commencement of discharge has filled the lake and may have resulted in discharge waters dispersing rapidly and widely across the lake, such that impacts may be either negligible (because of dilution) or undetectable (due to controls being similar to impact sites).

Recommendation 1. A hydrological study be conducted which quantifies the patterns and rates that discharge disperses and mixes with lake waters. Minor and trace elements which occur in discharge but not lake waters should be appropriate indicators to track discharge movements.

A full lake is a rare phenomenon even with discharge as evaporation rates are many times that of precipitation on average. Impacts around the discharge point are more likely to occur when the lake is dry or low in water volume and, indeed, the monitoring strategy was designed to detect such impacts. Impacts are also likely to be linked to the accumulation of salt load within the lake system and therefore may take several years to manifest.

Recommendation 2. Continue monitoring of lake and lake margin biota and physico-chemistry, as per this study, at least annually for the duration of discharge. Ideally monitoring would be more frequent for water quality and invertebrates, and continue for several years following completion of discharge to incorporate seasonal and inter-annual variations in climate.

Despite no impacts being detected, the authors are concerned about cumulative effects of discharge, particularly given the high volumes and ionic concentrations, as well as the likelihood of increases in these levels due to more water entering the mine pit at Cuddingwarra. Currently, each day sees the discharge of around 4200KL of water containing about 420 tonnes of total dissolved solids and 380 tonnes of NaCl; this amounts to over 150,000 tonnes of salt per year added to what is effectively a closed system. The lake is huge is size however and may be able to cope with these levels. For instance, based on very crude estimates, annual discharge would only amount to about 1% of the total lake volume. Despite this, the pattern of flooding and drying may be altered, given that parts of the lake will effectively never dry completely. Much of the salt is expected to accumulate on the lake floor following evaporation and dissolve again in water when floods arrive. Hence, salt concentrations may be appreciably higher for a given volume of lake water. This may lead to changes in biota, particularly as many species sampled have a documented preference for fresh to brackish water. All this remains speculation however in the absence detailed salt and water budgets for the lake and discharge.

Recommendation 3. A detailed salt and water budget be performed for the lake, particularly with the view to predict cumulative changes in water salinity with given lake volumes.

The pipeline was extended in April of this year to direct discharge towards the centre of the lake and away from the saltmarsh of the lake edge (which is heavily damaged at the discharge point) and the major inlet channel (which was recognised in our previous studies to be vulnerable to discharge). We applaud this move, but suggest that movement of water be monitored, especially when lake levels are low given the water moves freely in response to wind.

Recommendation 4: The pipeline extension be retained as much as possible, with a view to extending further or shifting in another direction should it be shown that discharge is preferentially flowing/moving to fringing vegetation or inlet channels.

Recent flooding of fringing saltmarsh vegetation has resulted in widespread death and damage of plants at this edge. Recruitment of new plants as waters receded was also commonly observed. Although discharge wasn't conclusively shown to be responsible for decline in plants, some evidence exists that a minor species of the edge was impacted to a greater degree close to the discharge.

Recommendation 5: The fringing saltmarsh vegetation be monitored intensively over the next 18 months to gauge the level of decline against the degree of recovery. In particularly differences between impact and control zones should be investigated.

Finally, studies of salt lake biota and ecology remain limited in Australia, and this has made the tasks of predicting and measuring impacts more difficult. Discharge of excess mine-pit water into salt lakes is however becoming popular in the Goldfields and Murchison regions. Consequently, there has been a recent surge in survey and research effort, principally supported by mining companies (eg PhD in salt lake ecology at Curtin University, Kalgoorlie). Critically, environmental researchers, managers and consultants are also beginning to exchange information through such means as workshops, conferences and on-line discussion groups. This will improve our understanding of such systems, and our ability to make predictions.

Recommendation 6: This and other monitoring studies conducted at Lake Austin be communicated to a wider audience of environmental professionals.

References

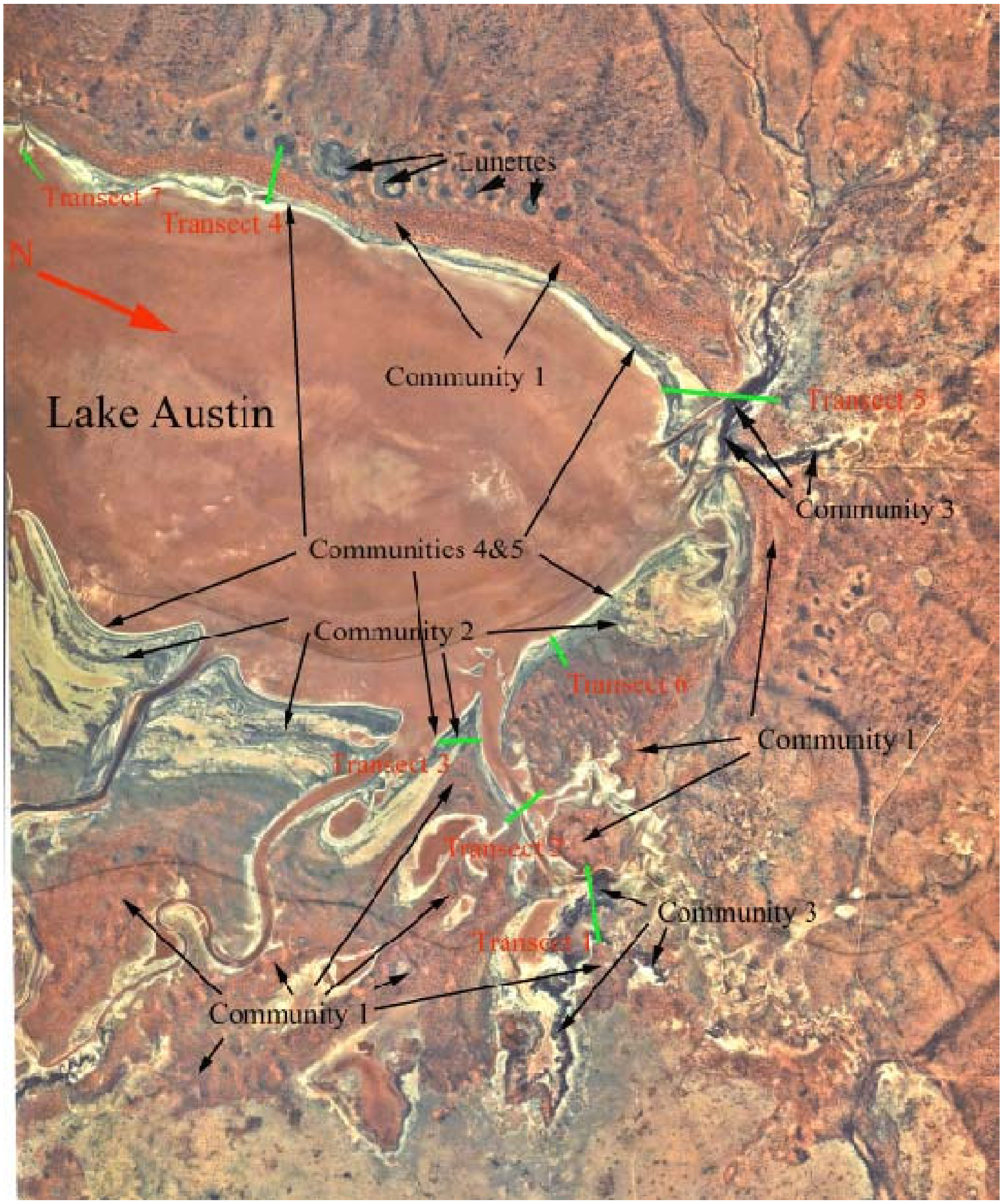
Sommer, B., Harkins, C. and Horwitz, P. (2000). Monitoring of aquatic invertebrates of Lake Austin wetlands receiving dewatering discharge from the Cuddingwarra Prospect: Monitoring period April - October 1999. Interim Report commissioned by Newhampton Goldfields Ltd, July 2000. Report No. 2000-07. Joondalup: Centre for Ecosystem Management, Edith Cowan University.

Horwitz, P.H. and van Etten, E.J. (1999). Survey of receiving wetlands, and impacts of dewatering discharge for the Cuddingwarra prospect. A report prepared for Wirralie Gold Mines Ltd. Joondalup: Centre for Ecosystem Management, Edith Cowan University

Appendix 1. Characteristics of discharge water from the Cuddingwarra mine.

Parameter	Units	24.7.97	24.8.98	31.5.99	21.11.99	18.2.00	14.5.00	22.6.00
Al	mg/L			1.7	<0.1	0.1		<0.1
As	mg/L		<0.085					
Ca	mg/L	890		970	890	780		740
Cd	mg/L		<0.001			<1		<1
Cl	mg/L	62000		56000	53000	51000	54000	59000
CO_3	mg/L			<1	<1	6.7		<1
Cr	mg/L		<0.05			0.01		<0.01
Cu	mg/L					0.31		0.1
EC	uS/cm			190000	171000	173000	126000	125700
Fe	mg/L	1		<0.5	1.37	2.62		2.2
Hardness ($CaCO_3$)	mg/L			4500	21800	9900		19000
HCO_3	mg/L	250		190	247	<1		24
Hg	mg/L		0.0015			<0.1		<0.1
K	mg/L			900	950	780	950	970
Mg	mg/L	4700		500	4750	1950	265	4100
Mn	mg/L		2.2	<0.5	0.37	0.79		0.51
Na	mg/L	39000		37000	32000	30000	30000	33000
Ni	mg/L		0.05			0.03		0.55
NO_3	mg/L			120	94	178		100
Pb	mg/L		<0.05			<1		<1
pH		8	7.2	7.5	7.05	6.9	6.85	7.2
Si	mg/L			42	11	22		9
SO_4	mg/L	14000		13000	13000	15000	16000	16000
TDS	mg/L		23000	100000	112000	118000	118000	120000
Total. Alk. ($CaCO_3$)	mg/L			160	203	198		
WADCN			<0.01			<0.01		<0.01
Zn	mg/L		0.1			0.39		0.11

Appendix 2. Locations of transects and main plant communities of the north-west corner of Lake Austin. Please note: communities 4 and 5 could not be differentiated between at this map scale.



Appendix 3: Photographs of macroinvertebrate sampling sites

1) Transect 1, LA1

2) Transect 2, LA3

3) Transect 3, LA5-1

4) Transect 3, LA5-2

5) Algal bloom near Transect 5

6) Transect 4, LA7-1 and LA7-2

7) Transect 4, LA7-1 in foreground, LA7-2 in background

8) Transect 4, connecting channel between LA7-1 and LA7-2

9) Transect 5, LA9-1 (bands of decaying *Ruppia*)

10) Transect 5, LA9-2

11) Transect 7, LA13