Report on the Monitoring of Lake Austin Fringing Vegetation following 3 Years Dewatering Discharge from the Cuddingwarra Prospect.

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Executive Summary

I. This report assesses the impacts of approximately three years of discharge to Lake Austin of hypersaline water collected from the Cuddingwarra mine pits. The field sampling was done in April and May of 2002 and because of the lack of surface waters at this time, only the fringing vegetation was monitored. By October 2002 discharge has ceased as mining activities at Cuddingwarra were scaled back.

II. The BACI ('Before – After – Control – Impact) design of the monitoring program revealed that discharge has increased the topsoil pH within the vegetation immediately fringing the lake, particularly in areas where discharge waters have been in direct contact with the vegetation. There is strong evidence that the salinity of these soils has also been increased by discharge.

III. Despite changes in the topsoil of fringing vegetation in the vicinity of the discharge points, no impact on the vegetation was detected by the monitoring program.

IV. Possible reasons why no impact on the health, growth and recruitment of plants in the fringing vegetation was detected are: a) lack of power (post-hoc power tests demonstrated that, due to the high variability, around 30-50 sampling points would be required to detect any differences in the degree of change between 'control' and 'impact' zones); b) flooding of lake and fringing vegetation, due to abundant rains in summer and autumn 2000, has masked impacts by diluting and mixing discharge waters; and c) the inherent ability of fringing samphire plants to survive and grow in extreme salt levels has meant they can tolerate an increase in salinity (although there was some evidence that seedlings may be more vulnerable to the increase in salt levels of the topsoil).

V. Given Lake Austin is an internal drainage system, the half a million or so tonnes of salt added through discharge should be regarded as a long-term addition to the system. Rather than being more-or-less evenly deposited on the lake surface following evaporation of the 2000 floodwaters, salt was preferentially deposited in the lowest part of the system. These were mainly inlet channels which now contain up to 1m thick deposits of salt. This means the next major in-flows into the lake should carry exceedingly high salt loads. Depending on where this water goes, future impacts on fringing vegetation may occur.

VI. This study, due to the fact that monitoring has occurred both before and after discharge, and across drought and flood periods, has given us a far better understanding of the how salt lakes function

and change over time, and of how they respond to added hyperaline water. Some important message for improved monitoring and management of discharge to salt lakes, now a common practice in the gold mining areas of Western Australia, include:

- **Extension of pipelines towards the middle of lakebed, such as occurred at Lake Austin early in the discharge period, is highly effective in minimising impacts on fringing vegetation;**
- **Sampling points for monitoring of impacts should ideally exceed 30 due to the high variability in the response of vegetation to discharge;**
- **Although fringing plants are typically very tolerant of high levels of salinity in both soil and floodwater, they require flooding by fresh to brackish water to simulate mass germination and recruitment; germination and seedling growth are likely to be vulnerable to flooding by discharge waters of high salinity as well as enhanced salt levels from previous contact with saline discharge waters. In other words fringing plants are far more sensitive at their seed and seedling life stages.**

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1. Introduction

1.1 History & Background

A new gold mine was commissioned at Cuddingwarra (located between Cue and Big Bell in the Murchison district of Western Australia) in 1999 to provide supplementary ore to the nearby gold extraction facilities at Big Bell. Constant dewatering of the mine pit has been required at Cuddingwarra due to shallow groundwaters. Environmental approval was obtained to discharge this hypersaline water to the northern end of Lake Austin, an extremely large, flat and mostly unvegetated salt lake. The various licences to discharge this water into the lake issued by the Department of Environmental Protection has allowed up to 6000 kL/day (or 180,000 kL/month) of water between 100,000 to 130,000 mg/L total dissolved solids. Actual discharge commenced in May 1999 and continued at a rate of between 3000-5000 kL /day (averaging 4200 KL/day) until early 2002 (see Figure 1). Since that time, discharge volumes have declined as mining activities were scaled back, and from October 2002 discharge to Lake Austin ceased in favour of disposal to mine pits at Cuddingwarra. Total dissolved solids of the discharge have averaged 112000 mg/L, with electrical conductivity averaging around 150000 uS/cm (see Appendix 1). Monitoring has been a condition of environmental approvals and this study represents the third visit to the discharge area by the Centre for Ecosystem Management at ECU, and was conducted during April 2002 some 3 years following commencement of discharge. The first visit was to record baseline (pre-impact) data and establish monitoring plots and protocol for both fringing vegetation and aquatic biota (Horwitz *et al.* 1999); the second was to monitor any changes some 14 months following commencement of discharge (van Etten *et al.* 2000). Due to lack of rains in 12 months prior to the most recent trip, it has not been possible to study aquatic biota as per previous visits. Consequently, this report concentrates on the impact of discharge on the fringing vegetation and soils.

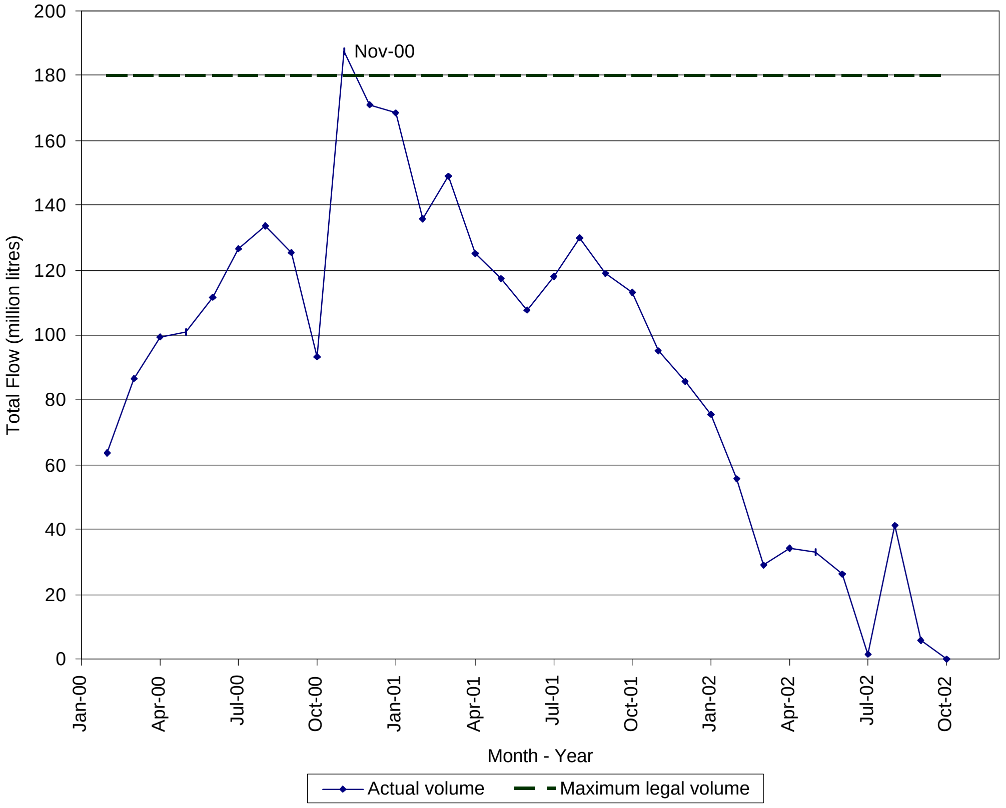

Figure 1. Monthly discharge volumes from the Cuddingwarra mines to Lake Austin from February 2000 to October 2002. Actual monthly volumes shown in solid line with maximum monthly limit in broken line. (Source: BBGO Environmental Department, 2002).

Due to concerns expressed in previous studies that discharge was potentially damaging fringing vegetation and preferentially entering adjacent inlet channels, the pipeline was extended in March 2000 from the lake edge some 600 m out onto the lake-bed. This was to encourage discharge flow towards the middle of the lake. In February 2001, floodwaters moved the pipeline several hundred metres so that it again discharged close to the fringing vegetation. The pipeline was secured back into its original location on around August of 2001.

1.2 Previous Findings

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 macro-invertebrate 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.

Monitoring sites so established were revisited in June 2000, some 14 months following commencement of discharge, and re-measured (van Etten *et al.* 2000). Lake and inlets were full of water at this time due, predominantly, to well above average rainfall in the year previous (see section 1.3). Consequently, considerable changes were detected in water levels and ionic concentrations between sampling periods, as well as in the richness, abundance and composition of aquatic and fringing plant species, and in the salinity and pH of soils. 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 2000 sampling. The health and cover of species closest to the lake declined markedly due flooding and/or smothering by *Ruppia* and macro-algae; abundance of these species generally increased however due to seedling recruitment. The condition of perennial plant species did not differ between ‘control’ and ‘impact’ sites, 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 were possible.

1.3 Rainfall, Hydrology and Lake Levels

Monthly rainfall for Cue, located some 25 km east of the discharge point, is shown in Figure 2 for the period January 1996 to August 2002. 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 was received 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. Since the flood of 2000, above rainfall was received during January-February 2001 and October 2001, with most other months receiving below average rain. Summer and autumn of 2002 were particularly dry. In summary, monitoring has been conducted across both fluvial and drought periods.

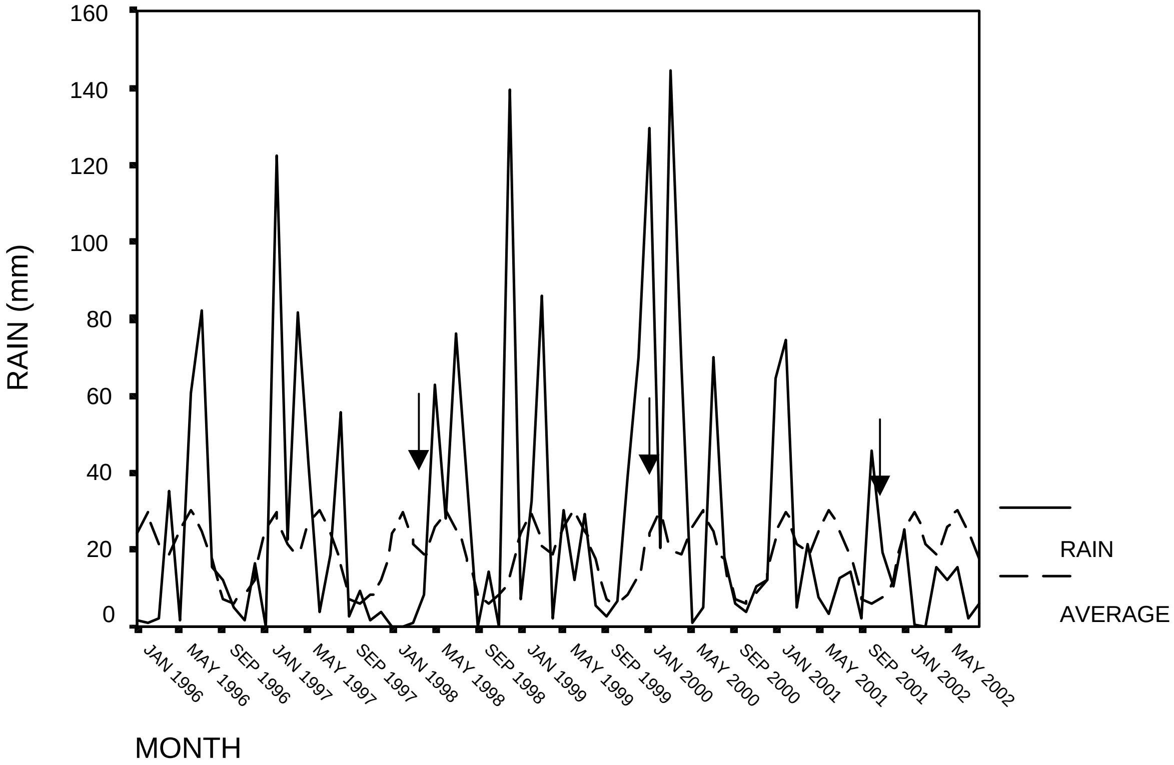

Figure 2. Monthly rainfall for Cue from January 1996 to August 2002 (solid line). Monthly averages are also shown (broken line). Arrows indicate monitoring dates.

Little is known of the hydrology of the lake and no detailed measurement of lake levels has occurred. It is known that the lake is usually dry, but fills in response to large rainfall episodes in the surrounding catchment. To what degree water entering the lake is derived from surface run-off via drainage lines, as opposed to surface expression of rising groundwater, is unknown. The Lake Austin catchment is known be endorheic – that is it represents an internal drainage system with Lake Austin, being at the lowest point in the catchment, the ultimate source of much of the surface drainage and groundwater discharge (Curry et al. 1994). The size of the catchment is approximately 13,750 km^2.

At the time of initial (baseline) survey in September 1998, the lake contained a reasonable 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 contributed to extremely high lake levels which inundated much of the lower parts of the fringing vegetation around the lake and inlet channels. At the time of the June 2000 survey, the edges of fringing vegetation were still flooded in many places although the floodwaters had receded from their peak of April that year by several centimetres (A. Wilkeis, pers. comm.). Since the flood of 2000, below average rainfall has meant that lake levels have slowly decline through evaporation. During March 2001, there was some discrete ponds of water remaining on the lake bed, but at the time of most recent field visit (April 2002), water remained only in the deeper drainage lines entering the lake and in the area immediately around the discharge point.

The water in these deeper drainage lines is all that is left of the huge volume of water from the 2000 flood and as this now salt-saturated water has slowly evaporated, a thick layer of salt crystals has been deposited in these drainage lines. The fact that the lake seems to have drained to these drainage lines is somewhat counter-intuitive, but their incised nature probably means they lie lower than the flat lake bed. The thick salt deposits would be expected to be mobilised with the next major rainfall and inflow episode, resulting in a profound flush of salt and other ions into the lake. The lake bed is covered with only a very thin layer of salt except in slightly lower areas, such as where vehicles or large fauna have left impressions in the muddy surface. This pattern again shows the salt mainly precipitates from solution once it become concentrated at the lowest parts of the landscape, albeit the difference in micro-relief may only be a matter on centimetres.

1.4 Monitoring Approach & Review of Monitoring Design

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 channels which enters the lake near the discharge point. These were referred to as ‘impact’ sites and transects in previous reports. 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. These were known as ‘controls’ in previous reports. Transect locations are shown in the Figure 8, together with extra sampling points conducted in April and May 2002.

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 was argued, indicated 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.

Two events have occurred to force a re-assessment of the rationale for the monitoring design. The first of these was the extension of the pipeline from the lake edge to 600 m or so into the interior of the lake in March 2000. This effectively confined discharge water to the lake bed, although the plume of water around the discharge point effectively moves with the wind. The second was the flooding event of March to July 2000. Measurements during this period showed that there was a high degree of mixing and homogenisation of salinity and other water quality parameter. It is argued that there is still justification for examining differences between the so-called control and impact sites of the fringing vegetation for the following reasons. Firstly, the historic impact from the period discharge was at the lake edge may still be detectable. Secondly, impacts from this period may be slow to develop, such as effects on recruitment processes and population parameters. Thirdly, flooding should be considered a rare departure from the usual condition of a dry lake with discharge water mostly forming a discrete pond of water which occasionally gets close to fringing vegetation and inlet channels as staining of the lake bed surface and observations have shown. Lastly, the pipe bent due to arrival of flood waters from the inlet channel in February 2001 and for a period discharge was again close to lake edge. All-in-all, although the above events have made it less likely that impacts will be found, there is still justification in investigating differences between sites close to and distant from the discharge point as evidence for impacts of discharge. In this report the terms ‘discharge’ and ‘non-discharge’ sites are used to describe these two groups of sites and are used in place of ‘impact’ and ‘control’ sites respectively.

1.5 Taxonomic Review of Samphires

The difficulty in correctly identifying samphires (species of *Halosarcia, Sclerostegia, Tecticornia* and other succulent Chenopods of the tribe *Salicornieae*) collected in the field has been mentioned in previous reports. Some of the reasons for this difficulty relate to: 1) lack of characters on which to base classification and identification; 2) small size and general unavailability of flowering and fruiting parts; 3) phenotypic plasticity, especially in response to rainfall in the months previous to sampling; 4) extensive hydridisation between species; 5) variation within species (with many subspecies, varieties and forms recognised); 6) difficulty in pressing and preserving specimens; and 7) lack of taxonomic work for some of the species groups. These genera are, not surprisingly, considered to be taxonomically difficult. During the 2002 field trip, various collections proved difficult to identify and it was obvious that some species demonstrated considerable variation across the study sites. This warranted a taxonomic review of species and confirmation of identifications by Paul Wilson of the WA Herbarium (now retired but regarded as the world authority on the *Salicornieae*). This revision revealed that a number of species identified from previous trips should be split into two or more forms, that some species names have been changed, and that some species have been mis-identified in the past. A summary of the taxonomic changes from previous reports is shown below:

Previous Name	**Correct Name**
Halosarcia pergranulata	*Halosarcia pruinosa* (form 'a')
Sclerostegia disarticulata	*Sclerostegia tenuis*
Halosarcia indica subsp. *leiostachya*	*Halosarcia fimbriata*
Sclerostegia tenuis	*Halosarcia halocnemoides* (form 'b')

In addition, two species referred to in previous reports are markedly variable and complex across the study area and should be split into a number of taxa. The first of these is *Halosarcia halocnemodies.* The single-stemmed and spreading shrubs on the crests of the sandy fringing banks, identified in previous reports as *H. halocnemodies* subsp. *catenulata* in fact are form 'a' of this species. *H. halocnemodies* subsp. *catenulata* itself occurs occasionally on heavy soils of the study area. Form 'b' of this species was previously identified as *Sclerostegia tenuis* (see table above). Hybrids of this species and *H. fimbriata* were also found. It is obvious a number of different taxa in the *H. halocnemodies* group are found around Lake Austin and the group shows considerable variation and even some inter-gradation within the species and with other species. More taxonomic work on this group is urgently required. The second splitting was for the species previously identified as *H. doleiformis* – noted as highly variable in previous reports. Mostly this species was correctly identified, but at some sites should be changed to *H. pergranulata* (subsp. *pergranulata*) or *H. pterygosperma* subsp. *denticulata*. A full list of species at each monitoring site is shown in Appendix 2.

The classification of the salt marsh and surrounding vegetation of Lake Austin, as provided in the original report (Horwitz *et al.* 1999), requires an update given these name changes. The new names of plant communities and their descriptions are as follows:

1. *Acacia sclerosperma – Eremophila miniata* woodland on sandy dune systems.

This community is found on the red sand dunes that surround the lake bed and salt-marshes. The dunes are gently undulating with between 1 to 5m relief above the lake bed. Soils are red, deep, earthy sands with a pH of 7-8 and a low salinity (30-50 µS/cm). The vegetation of this community is dominated by *Acacia sclerosperma* (limestone wattle) with *Eremophia minimata* also common on higher parts of the dune. Other acacias (eg *A. tetragonophylla, A. anuera, A. xiphophylla*) and *Hakea preissii* (needlebush) also occur on a regular basis. Common understorey shrubs and subshrubs are typical of mulga woodlands/shrublands (eg *Rhagodia eremea, Ptilotus obovatus, Solanum lasiophyllum, S. orbiculatum, Enchylaena tomentosa, Cheopodium gaudichaudianum*). More salt tolerant shrubs such as *Didymanthus roei, Frankenia pauciflora* , *Maireana* spp. and *Sclerolaena* spp. are found on the lower slopes of dune where they begin to give way to salt-marshes and other low lying vegetation. Annual grasses and daisies are common at ground level during winter/spring. Some perennial grasses are also found (*eg Eriachne helmsii, Eragostis* spp., *Stipa scabra*) although their abundances would be expected to vary with grazing history and climate. The species richness of this community far exceeds that of the others. [sites: 1-1, 4-4]

2. Mixed chenopod low shrubland on low banks.

In some places a series of elliptical banks running parallel to the lake shore is found between the dunes and the saltmarsh fringing vegetation. The banks, which are particularly widespread above the north-eastern shore of the lake, appear to be aeolian origin. The banks are around 1-1.5m above lake level and have intervening interbank plains . The surface soils are fine sands variously colored from brown - red to grey – brown and appear to be siliaceous and gypseous in places. Clay often appears below the surface soil. pH is slightly alkaline (7-8.5) and conductivity is variable but relatively low (50-300 µS/cm). The vegetation consists of low shrublands of chenopods. The species composition varies considerably from place to place with common species *being Atriplex vesicaria, A.* ?*acutibractea, Scleroleana eurotioides, Maireana amoena, M. atkinsiana* and *M. pyramidata*. The interbank plains often have different composition to the banks and sometimes have samphires (*Halosarcia* spp.) on low lying areas. [sites: 3-3, 2-1, 2-2]

3. *Halosarcia pruinosa – Sclerostegia tenuis* saltmarsh on low lying saline plains.

Just before the inlet channels enter the north-easterrn part of Lake Austin they spread out into low lying flats. Here a community dominated by *Halosarcia pruinosa* (form 'a') and related chenopod *Sclerostegia tenuis* is found. These low shrubs form dense thickets of up to 85% cover. Few other species are found between the shrubs apart from the odd halophytic annual (eg *Atriplex holocarpa* (pop saltbush), *Angianthus* spp. and *Caladrinia* spp.). The surface soil is a silty clay which cracks when dry and is a sticky mud when wet. The pH of this soil is extremely alkaline (9.5-10.5) and is moderately saline (500-1500 µS/cm). Seasonal waterlogging appears to be the norm with occasional flooding possible. The community is low lying and appears to be at or just below the level of the inlet channel floor, being separated from the channels by a low bank of only 0.2-0.4m height (which has communities 4 and 5). Consequently, the irregular waterlogging and possible flooding comes from either: a) run-off from surrounding slopes; b) surface flow from the inlet channel either over the top of the protective bank or through some minor channel; c) groundwater rise following major rainfall events; or d) some combination of these water sources. [sites: 1-3. 1-2, 5-1, 5-2]

4. *Halosarcia halocnemoides* saltmarsh on crests and upper slopes of fringing banks.

A single or small number of low banks (0.2-0.5m height) typically run parallel to the shores of the main inlet channels and lake bed. On the crest to upper slopes of these banks is found a community dominated by *Halosarcia halocnemoides* (form 'a') (10-30% cover). On the midslope, *H. doleformis* mixes with this dominant. On lower slopes, *H. fimbriata* takes over to form community 5 down to the water's edge. Therefore a zonation of three dominant *Halosarcia* species can be observed from the crest of the low banks to the shore of the lake/channels and this zonation is linked to subtle change in slope and microrelief. Community 4 also features halophytic annuals, small subshrubs and creepers such as *Gunniopsis* spp., *Atriplex holocarpa, Swainsona* sp., *Maireana amoena*, *Dysphania* spp., *Frankenia* sp. and *Caladrinia eremaea*. Annual daisies are particularly common. The soils on these bank crests are typically fine sands of aeolian gypsiferous material. Conductivity is moderate (typically 50-100 µS/cm) and pH alkaline (mostly 8-9). [*H. halocnemoides* dominant sites: 1-4, 2-3, 3-2, 5-5, 5-6, 5-7; *H. h.* mixed with *H. doleformis* sites: 4-3, 2-4, 2-5, 6-2; 7-1]

5. *Halosarcia fimbriata* saltmarsh on lower slopes of fringing banks.

On the lower slopes of the fringing banks as they gradually approach the lake, soil salinities increase dramatically. The surface of the soil on these gradual slopes consists of a light grey, silty clay which forms a salt-encrusted, cracking surface. Soil salinity here is extremely high at around 10-20 mS/cm. Below this crust is a light brown silt which was moist at the time of the field visit (September 1998); salinities here are between 4-11 mS/cm. On these lower slopes down to the lake bed itself *Halosarcia fimbriata* is dominant. It sometimes co-occurs with small *H. halocnemodies* subsp. *catenulata* mainly on the inlet banks. Other species are uncommon but are mainly annual halophytes such as *Gunniopsis septifraga*, *Swainsona* sp., *Atriplex holocarpa* and several species of daisy. [sites: 4-1, 7-2, 3-1, 3-4; 4-2, 5-3, 1-5, 7-3 (all with some form of *H.h*); 5-8, 6-3; 6-4 (*H.i.* only)]

N.B. Much of this community was inundated in the first half of 2000 and is referred to in this report as fringing community or vegetation to distinguish it from other communities which weren't inundated and are less likely to be impacted by discharge waters (see Table 5).

2. Methods

Monitoring sites established during the initial survey of flora and vegetation in September 1998 were revisited during one of two field trips, either between 4-8th April or 1st – 3rd May 2002, 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

Altogether 30 (out of 33) sites were remeasured along 7 transect lines; those omitted were mainly upland sites dominated by *Acacia* and other non-halophytic species. Soils were also collected (from the top 1-4 cm) at each site 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, some of which are reproduced in this report, with others available on request. All of the above measurements were performed here using the same methods as the initial study and overseen by the chief investigator (Dr van Etten) to improve consistency.

These data were used to calculate change in the above measured parameters from both the pre-impact (Sept 1998) and post-impact (June 2000) states – that is change across either a 43 month or 22 month period respectively. The degree of change was expressed both in absolute terms and in terms of percentage of initial values.

Mean change in each parameter was calculated for sites located close to the discharge point ('discharge' sites; transects 1, 2, 3 & 6) and for sites located some distance from this point ('non-discharge' 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. This was done for all sites and for a subset of sites located within the fringing vegetation community. As some species 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 (disharge vs.non-discharge). In other words individual plants were replicates rather than sites. In addition to t-tests comparing change between discharge and non-discharge sites, two-way ANOVA was used to compare overall differences across monitoring periods ('time'), discharge vs on-discharge ('zone'), and interactions between time and zone.

3. Results

3.1. Changes in Species Composition

The below average summer and autumn rainfall of 2002 meant there was no annual flora present in monitoring plots, which contrasts with previous visits. Many short-lived perennials, which were present on previous monitoring visits, were also no longer evident at sites. Therefore, most sites were floristically simple compared to previous visits, with little more than long-lived halophytes present in and surrounding monitoring sites. Within plant communities, no differences in species richness were found between discharge and non-discharge sites.

3.2. Analysis of BACI Design

The standard statistical analysis for Before – After – Control – Impact (BACI) experiments are two-way ANOVA with time (‘before’ vs ‘after’) and zone (‘impact’ vs ‘control’) being the two factors (Underwood 1999). Statistically significant interactions between time and zone are of interest here as they disprove beyond a reasonable doubt (here less than 5% chance) that there has been no impact due to an experimentally imposed treatment, and that therefore we should accept the an impact due to treatment has occurred over time. In our case, time refers to the time of each monitoring study, zone refers to either the discharge area or areas distant from it (although see rationale under 2.4), and the treatment is the discharge.

With one exception, there were no statistically significant interaction between time and zone (Table 1), demonstrating that almost none of the changes in vegetation and soil parameters measured over time could be attributable to discharge. The exception was for soil pH in fringing vegetation communities which was significantly different between discharge and non-discharge zones over time. Several parameters not surprisingly showed significant differences over time alone (such as health of *H. fimbriata* and *H. halocnemoides* (form ‘a’), and soil pH and conductivity; Table 1), whilst others showed significant differences between zone only (eg abundance of *H. halocnemoides* (form ‘a’), cover and abundance of *H. pruinosa* and soil pH; Table 1).

Table 1. Results of two-way ANOVA showing *F* values, observed power in parentheses, and levels of probability of a type I error (*** denotes $p<0.001$, ** denotes $p<0.01$, * denotes $p< 0.05$). $^{\#}$ denotes $p<0.05$ for Levene’s test of equality of error variances.

Parameter		Time *F* value	Zone (*F* value)	Time X Zone Interaction (*F* value)
H. fimbriata	Height	0.5 (.13)	0.7 (.13)	0.3 (.54)
	Width	0.1 (.06)	2.6 (.35)	0.1 (.61)
	Health	5.5 (.80) **	0.1 (.06)	0.1 (.06)
	Cover	0.8 (.17)	0.04 (.05)	0.03 (.05)
	Abundance	0.3 (.09)	1.1 (.18)	0.5 (.12)
H. halocnemoides (form a)	Height $^{\#}$	0.04 (.05)	0.6 (.12)	0.01 (.05)
	Width	0.3 (.09)	2.7 (.35)	0.1 (.06)
	Health	12.6 (.99) ***	2.4 (.32)	1.5 (.29)
	Cover	0.02 (.05)	0.8 (.14)	0.4 (.11)
	Abundance	0.1 (.06)	5.5 (.61)*	0.5 (.12)
H. pruinosa (form a)	Height $^{\#}$	0.5 (.11)	2.4 (.28)	0.6 (.13)
	Width $^{\#}$	0.05 (.06)	1.1 (.15)	0.1 (.06)
	Health	0.05 (.05)	3.4 (.38)	0.02 (.05)
	Cover	0.2 (.07)	6.8 (.64)*	0.06 (.06)
	Abundance	0.04 (.05)	9.0 (.76)*	0.1 (.06)
Soil electrical conductivity	All sites $^{\#}$	3.6 (.65)*	0.1 (.07)	0.25 (.05)
	Fringing sites	4.3 (.70)*	1.1 (.17)	0.7 (.16)
Soil pH	All sites $^{\#}$	16.6 (1.0)***	5.9 (.67)*	2.9 (.56)
	Fringing sites	4.4 (.70)*	4.3 (.50)	4.3 (.68)*

To explore actual trends and degrees of difference, the mean change in soil and vegetation parameters over time are compared between discharge zone and non-discharge zone in the next three sections.

3.3 Change in Vegetation Structure

No significant difference was found in the change to cover and abundance of perennial species between 'discharge' and 'non-discharge' sites for both periods 1998-2002 and 2000-2002 (Table 2). This is despite the fact that mean cover increased by around 5% (relative to initial values) across both periods at discharge sites compared to a drop of 3-8% in non-discharge sites (Figure 3). Very high site to site variation in the degree of change at least partly explains the lack of statistically significant results. There were also no significant differences between 'discharge' and 'non-discharge' sites in the areas immediately fringing the lake and inlet channels (Table 4). These fringing areas were almost completely inundated for several months in the first half of 2000. This flooding resulted in a decline in cover of perennial species at both discharge and non-discharge zones, although to a far greater degree at non-discharge sites (1-2% compared to 22-28%; Table 4). However the decline in cover was in some ways compensated for by an increase in the abundance of perennial plants in these areas following inundation (Table 4).

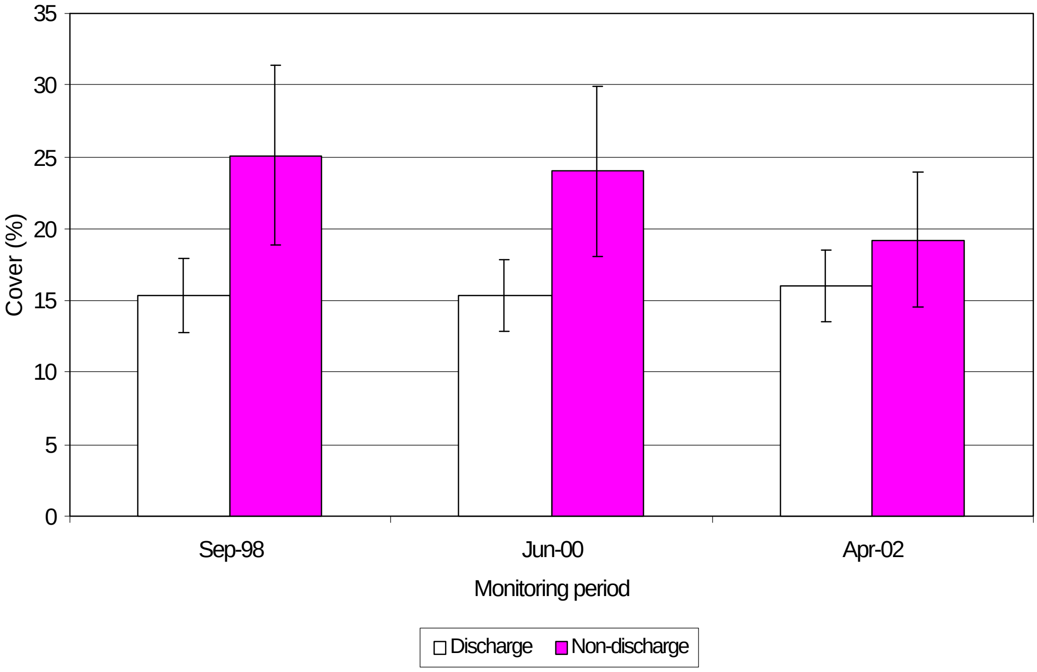

Figure 3. Mean and standard error of perennial species cover over all sites. The change in cover between the discharge and non-discharge sites was not significantly different over time.

3.4 Change in Plant Condition

Following the taxonomic revision, adequate replication was available for only three *Halosarcia* taxa to compare growth and health status in areas near the discharge to areas distant from the discharge. These three species dominated each of the three distinct saltmarsh communities (no. 3 to 5 as described in section 1.5) of Lake Austin and are outlined in turn.

H. fimbriata dominates the areas immediately fringing the lake and lower reaches of the inlet channels and other low lying areas. These areas have extremely high soil salinities (10-80 mS/cm) and were generally inundated for several months in 2000. This species seems to be very slow growing and have, in absolute terms, grown only 4-5 cm in height and less than 1 cm in width, on average, across almost four years of monitoring (Table 3). Relative to its initial height however this species has declined in height by around 10% on average with most of this decline occurring following the inundation in 2000 (Tables 2 & 3). In contrast plants have, on average, increased their width by around 7% from their initial size, again mostly following inundation. Health of plants has declined overall by around 30% across both monitoring periods. Most of this change can be attributed to death and damage of growing tips following flooding (perhaps due to environmental stress and/or the smothering of plants with *Ruppia* and macro-algae) and subsequent lateral regrowth of plants. Growth however varies widely from site to site and from plant to plant (Tables 2 & 3). No significant differences were found in the growth and change of health of this species

Table 2. Comparison of mean % changes in vegetation attributes between discharge and non-discharge monitoring sites from September 1998 to April 2002. (NB Degrees of freedom have been modified where variances were unequal at p<0.05). Standard errors are given in parentheses.

Attribute	September 1998 to April 2002					Jun	
	Mean % change for Discharge sites	Mean % change for Non-discharge sites	df	|t|	Prob	Mean % change for Discharge sites	M N
Total perennial cover	5.52 (13.8)	-3.33 (14.4)	44	.442	.660	5.89 (12.1)	
Total perennial abundance	31.93 (16.1)	13.95 (24.1)	44	.638	.527	45.39 (20.0)	
H. fimbriata cover	-22.62 (17.0)	-35.38 (13.5)	10	.550	.595	-20.98 (15.7)	
H. fimbriata abundance	59.98 (45.3)	-25.23 (36.1)	10	1.373	.200	55.48 (58.5)	
H. fimbriata height	-4.21 (5.5)	-12.39 (6.5)	10	.965	.357	3.56 (3.0)	
H. fimbriata width	5.48 (7.1)	7.99 (9.8)	10	.214	.835	-2.16 (3.4)	
H. fimbriata health	-34.36 (7.3)	-28.21 (11.0)	10	.489	.635	-21.94 (5.7)	
H. halocnemoides (a) height	0.93 (8.9)	13.28 (8.9)	7	.866	.415	-6.17 (8.8)	
H. halocnemoides (a) width	8.72 (11.8)	46.56 (8.3)	7	2.099	.074	-0.55 (11.5)	
H. halocnemoides (a) health	-28.52 (12.8)	-5.85 (21.1)	7	.975	.362	-37.64 (10.8)	

Table 3. Comparison of absolute and percentage changes in mean plant condition between plants in discharge and non-discharge zones from September 1998 to April 2002. (NB Degrees of freedom have been modified where variances were unequal at p<0.05). Standard errors are given in parentheses.

		September 1998 to April 2002					June 2000 to April 2002		
	Attribute	Change in discharge zone	Change in non-discharge zone	Df	|t|	Prob	Change in discharge zone	Change in non-discharge zone	df
ta	Height (m)	0.05 (0.03)	0.04 (0.04)	33	.069	.945	0.005 (0.01)	-0.02 (0.04)	13
	% Height	-7.38 (6.6)	-12.40 (14.1)	33	.369	.714	3.23 (2.9)	0.14 (14.7)	11
	Width (m)	0.0004 (0.04)	0.004 (0.05)	33	.583	.564	-0.04 (0.04)	0.03 (0.05)	33
	% Width	4.41 (8.9)	9.25 (18.9)	33	.264	.793	-2.47 (4.6)	13.27 (16.1)	13
	Health (% living)	-23.26 (4.2)	-22.50 (8.3)	33	.091	.928	-12.22 (3.3)	-4.17 (5.5)	33
	% Health	-36.31 (6.6)	-30.40 (12.3)	33	.465	.645	-22.14 (5.6)	-7.76 (17.0)	33
	Cover (% area)	-4.93 (3.8)	-7.00 (2.6)	10	.412	.689	-4.43 (3.1)	-4.6 (1.9)	10
	Abundance (no. plants)	42.71 (30.3)	-15.8 (22.0)	10	1.439	.181	31.29 (33.2)	-16.00 (24.2)	10
emoides	Height (m)	0.005 (0.03)	0.06 (0.03)	24	1.300	.206	-0.02 (0.02)	0.01 (0.02)	24
	% Height	1.45 (8.0)	13.28 (8.5)	24	.934	.359	-6.57 (6.6)	3.91 (3.1)	24
	Width (m)	0.07 (0.03)	0.37 (0.1)	10	2.711	**.022**	0.02 (0.04)	0.12 (0.07)	24
	% Width	8.75 (10.2)	46.57 (10.1)	24	2.387	**.025**	-0.70 (8.5)	19.17 (7.4)	24
	Health (% living)	-20.88 (6.0)	-5.78 (7.6)	24	1.521	.141	-28.24 (6.0)	-23.56 (6.5)	24

	% Health	-30.20 (9.2)	-5.85 (12.2)	24	1.576	.128	-39.50 (7.6)	-29.54 (7.0)	24
	Cover (% area)	3.50 (3.7)	-1.00 (6.7)	7	.650	.537	4.08 (3.7)	-2.00 (7.1)	7
	Abundance (no. plants)	16.67 (11.7)	4.67 (8.2)	7	.671	.524	19.83 (12.1)	5.33 (7.3)	7
sa (a)	Height (m)	-0.18 (0.2)	-0.006 (0.03)	8	1.005	.344	-0.03 (0.02)	-0.02 (0.02)	8
	% Height	-22.41 (19.6)	-0.03 (5.9)	8	1.090	.307	-5.33 (2.8)	-2.71 (4.9)	8
	Width (m)	-0.13 (0.07)	-0.03 (0.08)	8	.878	.406	-0.05 (0.03)	-0.08 (0.09)	8
	% Width	-30.54 (19.1)	-6.83 (16.5)	8	.939	.375	-8.35 (4.4)	-12.56 (15.4)	8
	Health (% living)	-9.00 (8.27)	-9.00 (13.0)	8	.000	1.00	-4.00 (2.4)	-4.00 (5.3)	8
	% Health	-28.99 (24.0)	-20.48 (26.4)	8	.238	.817	-8.89 (5.4)	-23.08 (17.8)	5
	Cover (% area)	-4.50 (3.5)	-16.67 (10.1)	3	.912	.429	-4.00 (4.0)	-11.67 (6.0)	3
	Abundance (no. plants)	5.50 (6.5)	-21.33 (16.2)	3	1.251	.299	7.50 (7.5)	-10.67 (7.9)	3

Table 4. Comparison of mean % changes in vegetation and soil attributes between discharge and non-discharge FRINGING monitoring sites from September 1998 to April 2002. (NB Degrees of freedom have been modified where variances were unequal at p<0.05). Standard errors are given in parentheses.

Attribute	September 1998 to April 2002					
	Mean % change for Discharge sites	Mean % change for Non-discharge sites	df	|t|	Prob	Mean % change Discharge site
Total perennial cover	-2.33 (17.6)	-27.82 (12.6)	14	1.021	.324	-1.32 (16.3)
Total perennial abundance	56.51 (33.3)	18.12 (80.2)	14	.514	.615	45.27 (42.1)
H. fimbriata height	-7.03 (5.6)	-11.65 (8.3)	8	.483	.642	2.36 (3.3)
H. fimbriata width	3.89 (8.2)	11.38 (11.8)	8	.539	.605	-4.12 (3.3)
H. fimbriata health	-37.19 (7.9)	-26.93 (14.1)	8	.689	.510	-21.22 (6.6)
H. fimbriata cover	-16.39 (18.7)	-36.72 (17.3)	8	.752	.474	-14.48 (16.9)
H. fimbriata abundance	65.44 (53.3)	-47.8 (36.4)	8	1.562	.157	58.06 (69.1)
Soil pH	5.27 (4.6)	-9.95 (2.1)	12.1	3.013	**.011**	9.95 (0.7)
Soil electrical conductivity	116.50 (38.5)	41.77 (32.8)	14	1.329	.205	435.25 (179.5
Soil % particle size >2mm	-78.13 (8.2)	15.83 (65.9)	5.2	1.414	.215	-72.4 (7.6)
Soil moisture	-82.37 (8.0)	-56.2 (9.7)	12	2.101	.057	

Table 5. Changes in vegetation and soil attributes between fringing and non-fringing monitoring sites from September 1998 to April 2002. (NB Degrees of freedom have been modified where variances were unequal at p<0.05). Standard errors are given in parentheses.

Attribute	September 1998 to April 2002					
	Mean % change for Fringing sites	Mean % change for Non-Fringing sites	df	|t|	Prob	Mean % change fo Fringing sites
Total perennial cover	-11.89 (12.1)	8.61 (13.7)	44	.988	.328	-9.23 (11.2)
Total perennial abundance	42.11 (35.3)	13.91 (10.5)	44	.961	.342	35.40 (38.2)
H. fimbriata cover	-24.5 (12.9)	-45.0 (15.0)	10	.672	.517	-18.80 (11.9)
H. fimbriata abundance	20.13 (38.3)	46.19 (18.9)	10	.292	.776	16.21 (44.8)
H. fimbriata height	-8.88 (4.5)	-1.35 (14.0)	10	.656	.527	2.88 (3.6)
H. fimbriata width	6.89 (6.5)	4.72 (10.3)	10	.139	.892	3.70 (5.0)
H. fimbriata health	-33.09 (7.1)	-25.37 (8.0)	10	.463	.653	-14.72 (7.0)
H. halocnemoides (a) height	-1.29 (32.8)	6.86 (4.6)	7	.487	.641	-24.22 (25.0)
H. halocnemoides (a) width	-13.05 (18.6)	31.16 (9.4)	7	2.206	.063	-18.88 (36.7)
H. halocnemoides (a) health	-55.85 (13.3)	-10.99 (11.0)	7	2.000	.086	-60.20 (12.5)
Soil pH	-0.44 (3.5)	-9.58 (1.3)	19.0	2.469	**.023**	5.77 (1.7)
Soil electrical conductivity	88.47 (27.9)	477.96 (121.6)	28.6	3.123	**.004**	299.50 (111.5)
Soil % particle size >2mm	-42.89 (26.6)	30.63 (57.2)	41	.950	.348	-72.40 (3.6)
Soil moisture	-71.14 (6.9)	-45.89 (19.8)	31.8	1.202	.238	

between discharge and non-discharge zones. However when comparing these zones in terms of sites immediately fringing the lake only (Table 4), in the period 2000-2002, the mean relative growth in width within the non-discharge zone (+15.4%) was significantly greater (p=0.048) than in the discharge zone where a mean decline of 4.1% was recorded. The average cover of *H. fimbriata* has declined across both zones, particular following inundation, whereas the abundance of individual plants has generally increased, particularly in the discharge zone (Tables 1 & 2). This reflects death and dieback of plants following flooding and subsequent recruitment of new individuals. No significant difference in the change in cover and abundance across monitoring periods was detected between discharge and non-discharge zones (Tables 1 & 2), even when restricting the analysis to fringing lake sites only (Table 4). The response in terms of recruitment and death was highly patchy across the study area, which no doubt contributed to the high standard errors measured for these parameters (Table 1). Photo 1 shows of the plant deaths and decline, and subsequent recruitment, of fringing *H. fimbriata* following flooding.

H. halocnemoides (form ‘a’) was mainly found as single-stemmed plant around 1 m high atop of the low banks fringing the lake and inlet channels. These banks mainly had coarse sandy soils of moderate salinity and were not flooded in 2000. These mostly large plants grew only by 3 cm in height on average (but highly variable) despite the above average rainfall received across much of the monitoring period. Growth was greater in the non-discharge zones compared to the discharge zone (Table 3), with the mean percentage change in width of plants in the non-discharge zones (9%) significantly less than in the non-discharge zone (47%). No such difference was found for the 2000-2002 period, which suggests that the differences detected mainly relate to the period before inundation. The health of this species has declined across all monitoring periods, whilst cover has increased slightly on average in the discharge zone but decreased in the non-discharge zone (differences however are not significant).

The third taxa compared is *H. pruinosa* (form ‘a’). This taxa dominates the low-lying clay flats (with their highly alkaline soils) adjacent to inlet channels. This taxa has declined in size, health and cover from 1998 to 2002 and 2000 to 2002 (Table 3). No significant differences in the mean level of decline were detected between plant located in discharge zone compared to plant distant from it.

3.5 Change in Soil Parameters

The pH of the topsoil decreased following discharge to a greater degree (in both relative and absolute terms) in the non-discharge zone compared to the discharge zone (Table 6; Figure 4). Between 1998 and 2002, the pH declined by around 2% on average in the discharge zone, whilst in the non-discharge zone it was almost 11%; this difference was statistically significant ($p = 0.06$; Table 6). Between 2000 and 2002, the pH actually increased in the discharge zone by 5% whilst it declined by almost 3% in the non-discharge zone; again this difference was significant ($p=0.001$).

The trend in electrical conductivity (EC) over time is similar but far more pronounced than that of pH. Indeed the two factors are highly correlated to one another ($r=0.76$ using Pearson’s correlation). However site-to-site variability in the change in EC was very high and no significant differences in the average change between discharge and non-discharge zones were found. In terms of average level of change across all sites, there was a three times increase in EC from 1998-2002, whereas it increased by almost six times in the non-discharge zone (Table 6). There is no doubt that massive changes in EC recorded at two sites outside the fringing vegetation contributed to the large overall increase away from the discharge zone (Table 5). Indeed when looking at fringing vegetation only, there was an 117% increase in the discharge zone on average compared to an increase of 42% in the discharge zone over the same period (Table 4; Figure 6). EC values were lowest at June 2000 when much of the fringing vegetation was inundated or recently flooded. Since that time evaporation of waters has led to several fold increases in EC in both discharge and non-discharge zones. No significant difference in he degree of this change from 2000 to 2002 were found despite the

Photo 1. Fringing vegetation dominated by *Halosarcia fimbriata* showing dead and unhealthy mature plants. Also shown in the foreground are seedlings which have arisen following the 2000 flood.

Table 6. Changes in soil attributes (top 1 cm) between discharge and non-discharge monitoring sites from September 1998 to April 2002. (NB Degrees of freedom have been modified where variances were unequal at p<0.05). Standard errors are given in parentheses. ** denotes p<.01, * denotes p<.05.

	Soil attribute	September 1998 to April 2002					June 2000 to April 2002				
		Discharge sites	Non-discharge sites	df	|t|	Prob	Discharge sites	Non-discharge sites	df	|t|	Prob
Absolute change	PH	-0.249 (0.19)	-0.997 (0.13)	41	3.024	**.004**	0.396 (0.12)	-0.239 (0.51)	38	3.705	**.001**
	Electrical conductivity (ì S/cm)	7851.0 (3266)	15277.0 (7132)	27	.947	.352	8370.4 (3944)	12897.0 (6658)	38	.598	.553
	Particle size >2mm (%)	-2.309 (0.56)	-1.003 (0.31)	33	2.033	**.050**	-1.415 (0.59)	-4.520 (2.00)	21	1.491	.151
	Moisture content (%)	-9.262 (2.12)	-9.115 (2.67)	39	.043	.966					
Percent change	pH	-2.12 (2.5)	-10.85 (1.4)	41	2.874	**.006**	4.99 (1.5)	-2.78 (1.4)	38	3.679	**.001**
	Electrical conductivity (ì S/cm)	194.09 (46.6)	492.83 (162.5)	22.1	1.767	.091	231.62 (91.3)	215.63 (105.3)	38	.115	.909
	Particle size >2mm (%)	-43.4 (15.1)	56.94 (77.8)	41	1.352	.184	-4.89 (22.6)	-52.01 (7.6)	20.8	1.978	.061
	Moisture content (%)	-83.81 (4.2)	-23.75 (25.4)	20.1	2.328	**.030**					

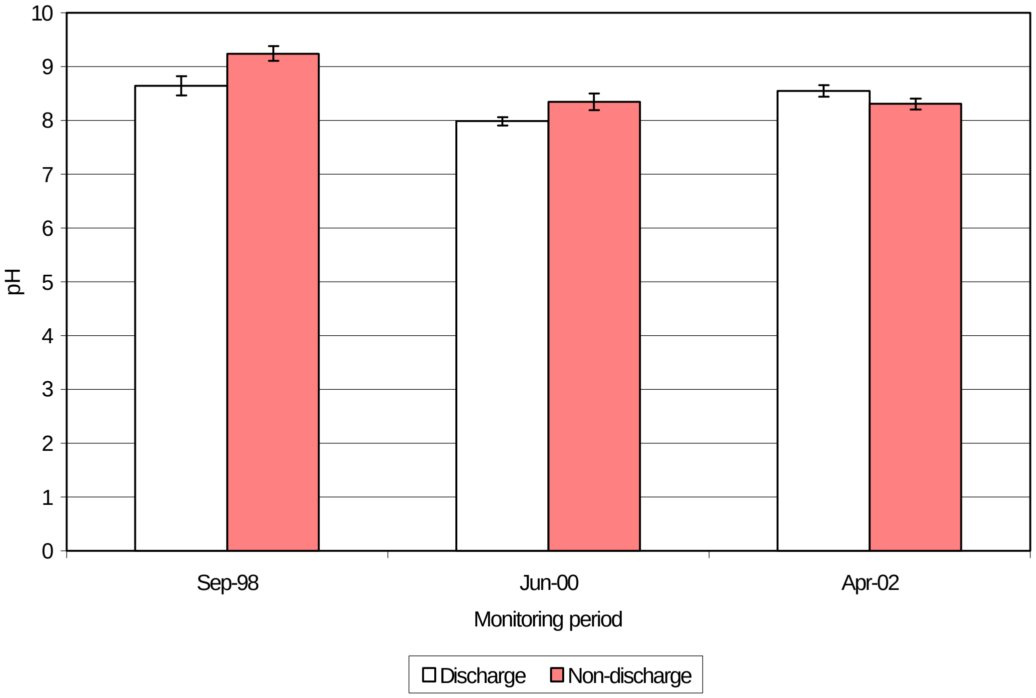

Figure 4. Mean absolute values and standard error of soil pH. The change in pH between the discharge and non-discharge sites was significantly different over 1998-2002 and 2000-2002 (p=.006 & p=.001 respectively).

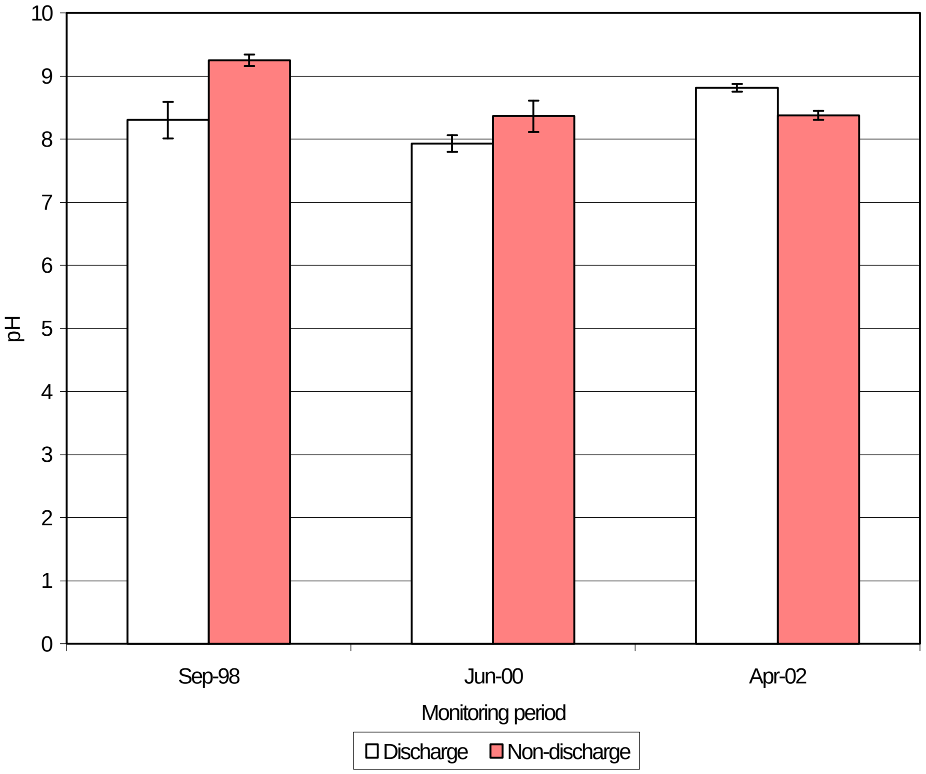

Figure 5. Mean absolute values and standard error of soil pH at 'fringing' sites. The change in pH between the discharge and non-discharge sites was significantly different over 2000-2002 (p=.001).

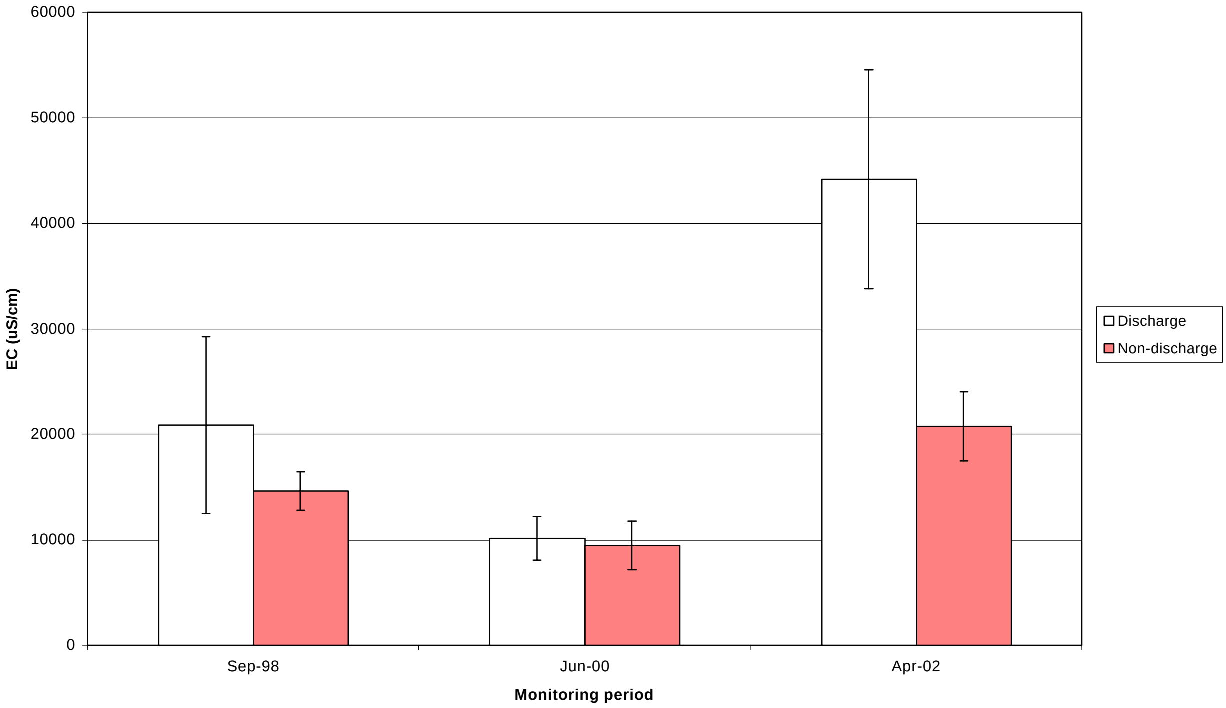

Figure 6. Mean value of electrical conductivity at the three monitoring periods for fringing vegetation sites only. Standard errors are indicated on bars.

discharge zone increasing to a far greater degree on average (Figure 6). Extra sampling in the fringing vegetation during May 2002 demonstrated that the EC of topsoil in the non-discharge zone was generally lower and more consistent spatially compared with that of the discharge zone (Figure 7). EC values varied widely in the fringing vegetation of the discharge zone, but were clearly higher at sampling points immediately around the discharge point (Figure 7).

Moisture of the topsoil declined from September 1998 to April 2002; this is not surprising given the substantial lower rainfall in the months preceding sampling in April 2002. The mean change in moisture content (relative to initial values) was significantly lower in the discharge zone compared to the non-discharge zone (Table 3.5). Similarly, the decline in course particles (>2 mm) in the topsoil was significantly greater in the discharge zone (at p=0.50) than in the non-discharge areas.

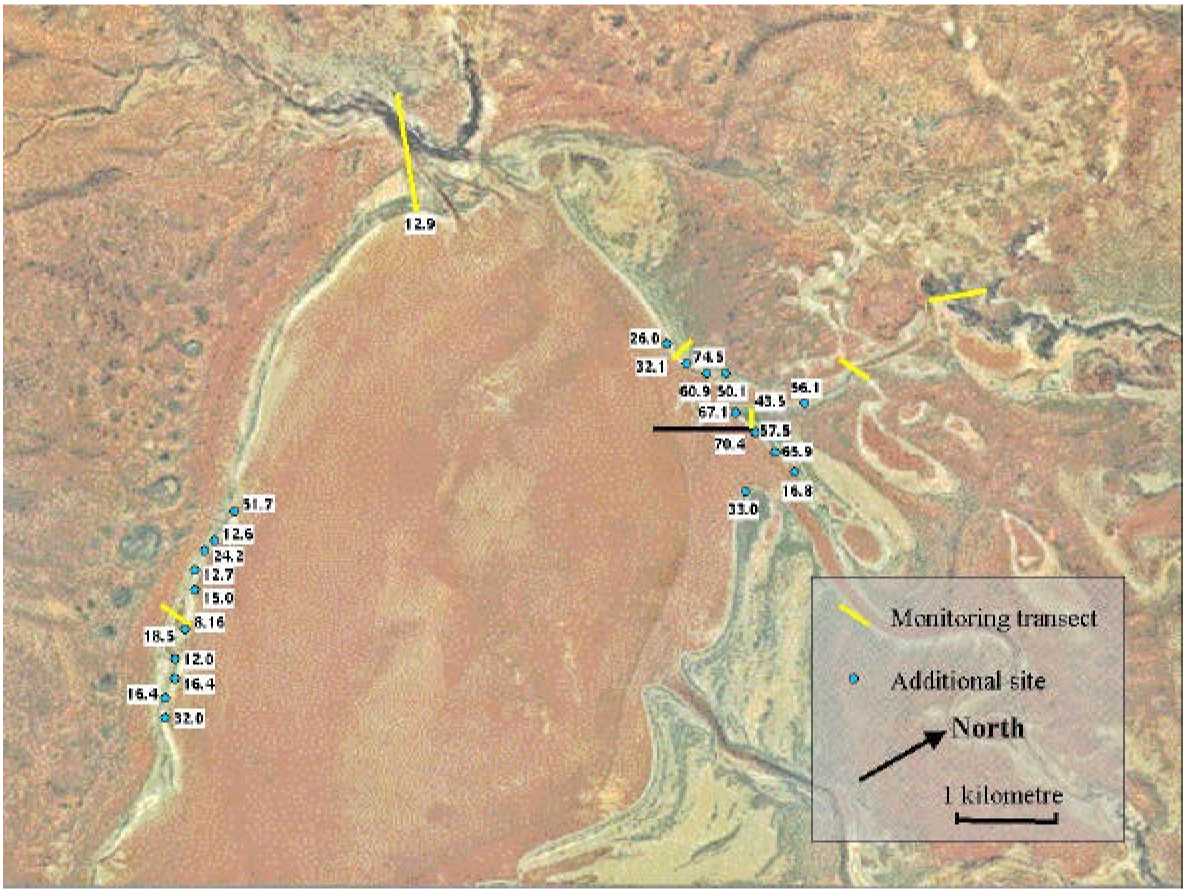

Figure 7. Soil electrical conductivity values at 'fringing' sites in April 2002. Aerial photo taken September 1997. (Image courtesy of Harmony Gold BBGO).

4. Discussion

4.1 Impacts Due to Discharge

The monitoring approach established by the Centre for Ecosystem Management was specifically designed to detect impacts (if any occurred) arising from discharge of hypersaline water into Lake Austin from the Cuddingwarra mine. Baseline (i.e. pre-imapct) data was collected at several sites both from near the discharge and in a similar area around 10 km away which was envisaged would be away from the influence of the discharge waters. Two 'post-impact' monitoring studies were done, the latest, in April 2002, occurred some 3 years following discharge commencement. The design therefore followed widely accepted BACI design principles for detecting impact. Despite the best of intentions when the design was established before commencement of discharge, two events occurred during the discharge phase that would be make it less likely that impact would occur in the vegetation surrounding the lake. The first of these was the extension of the pipeline some 600 m from the lake edge which effectively moved the discharge away from the fringing vegetation (at least most of time). The second was the flooding event of 2000 which substanitally decreased and homogenised salt levels in the water around the discharge for a period of at least several months. Despite the decreased likelihood of impacts being detected, as previously argued, investigation of any impacts were still warranted.

An impact of discharge was detected in soil properties of the lake edge. Since the recession of floodwaters from June 2000, the topsoil pH of the fringing vegetation has increased by 10% in the area close to discharge point which was significantly more so than in areas distant from it (0.2%). Salinity of the topsoil in these areas has shown a similar trend and although differences were not statistically significant, it is obvious that certain sites close to the discharge point increased substantially in salinity, and, overall, topsoil salinity is currently much higher in areas close to discharge than in areas remote from it. As pH is strongly related to salinity levels of the soil, it is reasonable to conclude that the hypersaline levels in the discharge water has increased salt and pH levels in the topsoil of the fringing vegetation. How then has this occurred? Firstly, since the recession of floodwaters, it is possible that discharge water has spread up to fringing vegetation nearest to the discharge pipe, particular at times of high discharge volumes. Indeed observations of salt scalds on the surface of the lake at April 2002, both on the ground and from the air (Figure 8), support this contention. In addition the movement of the pipe by incoming floodwaters from the inlet channel in early 2001 moved the pipe to a position quite close to fringing vegetation. The second possibility is that accumulation of salt which occurred in the fringing vegetation prior to the pipe extension in March 2000 is still persisting to at least some degree. Although it would be expected that this added salt would have been brought into solution when flooded, perhaps deeper stored salt would have not and has since risen in response to evaporative rise.

Figure 8. Evidence of surface salt on the lakebed surrounding the end of the discharge pipeline in April 2002, as seen from an aircraft. The visible salt is approx. 2 km in diameter (S. Vellekoop, 2002).

There is little evidence that this enhanced salt loading of fringing soils close to the discharge point has led to a impact on the vegetation of this area. Although the change in width of the dominant fringing species *Halosarcia fimbriata* was significantly less in the discharge zone compared to the non-discharge zone over the period 2000-2002, the fact the probability level of this being true is close to the 95% confidence limit, and the fact that other parameters and other time periods measured for this species did not reveal significant differences, suggests caution in attributing decline in this species to discharge. *H. halocnemoides* (form ‘a’), which dominates the small levee banks surrounding much of the lake, grew significantly less in areas close to discharge compared to areas distant from this for the period September 1998- June 2000. Given this species has not been in direct contact with discharge water, it is difficult to attribute discharge as the reason for the difference. Salt spray is one possibility, but it is more likely that some other difference in disturbance (such as grazing intensity) or in the physical environment (such as depth to groundwater) across zones may be occuring. Non-discharge sites were located on a different pastoral station to that of the discharge sites.

There are a number of possible reasons why impact on the fringing vegetation were not conclusively demonstrated despite the increases in soil salinity around the discharge point. The first is that, as salt tolerant plants, *Halosarcia* and *Sclerostegia* spp. may be withstanding the effects of increased soil salinity and pH. The fringing species *H. fimbriata* typically grows in soil where the surface EC is some 10-20 mS/cm, with values of up to 50 mS/cm recorded before commencement of discharge. These level of soil salinity are typically 2 to 3 orders of magnitude higher than other salt marsh communities on slightly higher ground. This species is therefore in a league on its own in terms of salinity tolerance and it is not unexpected that it may be able to withstand and grow at elevated levels of salt (the highest recorded EC in April 2002 was 75 mS/cm). A range of annual plant species have been found in the fringing community on previous visits (Horwitz et al. 1999; van Etten et al, 2000), but were absent in April 2002 due to lack of preceding rains. These species may be more sensitive to increased salt levels and it is recommended that monitoring following substantial rainfall episodes be conducted to gauge the impact on these species.

Although no impact on adult plants were detected, it is possible that the effects of increased salt levels may arise sometime in the future, particularly on other stages of the life-cycle. It is likely that *Halosarcia* spp. are more sensitive to salt at the seed germination and seedling stages. Vellekoop (2002), as part of her Honours study investigating the saltmarsh vegetation at Lake Austin, demonstrated, using glasshouse flooding experiments, that *H. fimbriata* require a period of flooding with fresh to brackish water to stimulate germination from the soil store and recruit new individuals. This was supported by field observations which showed ample seedling recruitment, primarily of *H. fimbriata*, as the floodwaters of 2000 receded. Despite being extremely patchy, no difference in seedling recruitment was found between discharge and non-discharge zones, as measured from both the field and flooding experiments, demonstrating that enhanced salt levels at points close to the discharge have yet to impact on the potential for seedling recruitment. Observations of seedling survivorship as at April/May 2002 however show that, where recruitment of *H. fimbriata* on the shoreline has been substantial, more seedlings were dead than alive at sites of enhanced soil salinity (i.e. >30 mS/cm) near the discharge point, whereas in areas of lower salinity further away from the discharge point, the opposite trend occurred (difference in ratio of dead to alive seedling abundance was significant using Mann-Whitney U-test; z=-2.2; p=~0.03). Enhanced salt levels seem therefore to be impacting on the survivorship of the seedlings. The study by Vellekoop (2002) also showed that recruitment only occurs when floodwater salinity is less than around 60 mS/cm. As discharge water is around 120 mS/cm in salinity, this suggests that direct inundation by discharge water or where salt from discharge waters have raised salinity levels of lake water, recruitment is unlikely to occur. Fortunately, for extensive areas of fringing vegetation to be inundated, the lake needs to be full, a uncommon phenomena only occurring after sustained high rainfall. Salinity levels at these times are typically around 30 mS/cm (van Etten et al. 2002) and the massive volumes of water in the lake mean discharge waters are effectively diluted and dispersed.

Lack of statistical power is another reason why impacts may not have been detected in this study. Post-hoc power tests routinely showed that there was a high probability that significant differences could not detected, assuming of course they exist (eg Table 1). This was generally due to sampling intensity not being great enough to counter the high variability in many of the parameters measured. In particular, the degree of change in vegetation characteristics and soil parameters like EC demonstrated huge spatial variability, even within plant communities. Power tests reveal that the sampling effort required to detect impacts on fringing communities, if they indeed exist, is in the order of 50-150 sites (depending on the parameter). This contrasts with the 10 monitoring sites established in the fringing community. It is extremely difficult to estimate, pre-impact, the sampling effort required, especially given that a pilot study was not feasible. The findings however have implications for future BACI type monitoring of salt-lakes subject to saline discharge, at Lake Austin and elsewhere.

In summary, the monitoring program conducted over the last four years has revealed that, in areas close to the discharge point, the salinity and pH of the topsoil have been raised, presumably because discharge waters have at times been in contact with or close to fringing vegetation of these areas. This enhanced salt level seems to be having a detrimental impact on the survivorship of seedlings, but otherwise no impacts were detected on soil seed store, recruitment potential, health and size of the main fringing species, *H. fimbriata*. Impacts may occur some time in the future however as Lake Austin is widely believed to be an enclosed drainage system. The half a million tonnes or so of extra salt deposited in the lake system through discharge should in many ways be regarded as a long-term addition to the system. As it did during the floods of 2000, much of this extra salt becomes dissolved during flood events and, given the huge volumes of water at these times, is diluted to a level where it makes only a marginal contribution to total water salinity. However, in contrast to what was anticipated, the deposition of salt as the floodwaters evaporate and recede is not even across the lake-bed. Waters drain to the lowest points in the system which, somewhat counter-intuitively, were observed mainly to be inlet channels. It is only then that waters become saturated with salt and deposition occurs. The fact that inlet channels now contain deposits of crystalline salt up to one metre in depth (see Photo 2), means that with the first large flows into the lake following the end of this and other drought periods is likely to be very high in salinity. The build up of salt in drainage lines is also likely to be a concern for aquatic biota and fringing vegetation in these areas.

Photo 2. Salt tended to accumulate at the lowest elevations including the inlet channels. Lake Austin proper is to the left of the photo. (S. Vellekoop, 2002).

4.2 Dynamics of Fringing Vegetation

The fact that monitoring has occurred over both a flooding and drought period has been fortuitous in some ways as it has enabled measurements and observations of profound changes in the vegetation. Arid environments of inland Australia are known for their exceptional variability, particular in the temporal variability in rainfall, which, in turn, drives dramatic changes in the physical environment and the biota . Rainfall episodes substantial enough to flood the fringing vegetation around inland salt lakes like Lake Austin only occur once every decade or so on average. The monitoring around Lake Austin has shown that flooding resulted in substantial death and damage to perennial shrubs (particularly *Halosarcia fimbriata*) due most likely to a combination of several weeks/months of inundation and smothering by macroalgae and *Ruppia* (an aquatic monocot), with smaller plants and those closer to the lakebed impacted upon to a greater degree due to greater length and depth of inundation. Seed germination and recruitment of new *Halosarcia* individuals was substantial although patchy as floodwaters receded, an event not observed in the fringing vegetation on previous visits. The majority of these seedlings were surviving (although again deaths were patchy) some two years after flooding. Growth rates of seedlings differed substantially with differences observed to be linked to subtle difference in microtopography. On slightly higher ground within the fringing vegetation, groups of seedlings were 10 to 20 cm high, whereas on lower parts, seedlings less than 5 cm in size (which co-incidently often showed symptoms of water-stress) were common. The height plants obtain before their next inundation is likely to play an important role in their ability to survive flood, whenever it arrives. Plants on the slightly lower slopes closer to the lake bed are less likely to survive than on higher parts of the slopes or on slight mounds due to difference in growth rates as well as in the frequency, depth and period of inundation they are likely to experience. This therefore would tend to control the position and, possibly, the density of the edge of the fringing vegetation, which would be expected to fluctuate spatially in response to the flooding regime. Substantial number of years (say 10+) between flooding events could result in a extension of the fringing vegetation toward the lake bed as plants have a chance to get to a reasonable size between floods and disperse seed further afield. By the same principle, a retraction in the edge is possible when flooding frequency is increased.

Within non-flooded saltmarsh communities on higher ground, the flood – drought transition saw a change from modest perennial species growth and high annual/short-lived species richness to a decline in the size of perennial plant species and a virtual absence of annual/short-lived species. This reflects typical temporal patterns seen in (terrestrial) arid lands in response to rainfall fluctuations.

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Appendix 1. Chemical attributes of the discharge water from April 2000 to May 2002 (Courtesy of Harmony Big Bell Gold Pty Ltd).

Date	*pH*	*EC*	*TDS*	*Mg*	*Na*	*K*	*Ca*	*Cl*	CO_3	HCO_3	SO_4	*Pb*	*Cu*	*Zn*	*Mn*	*As*	*Fe*	*Cd*	*Si*	*F*	NO_3	NO_2
		US/cm	*mg/L*																			
14-Apr-00	6.85	126000	181000	265	29000	950		54000			16000											
5-Jul-00	7.20	125700	120000	4100	33000	970	740	59000	<1	24	16000	<0.001	0.1	0.11	0.51		2.2	<0.001	9		100	
7-Jan-01	7.20	118000	108000	6200	28000	710	640	51000	<1	250		<0.001	0.13	0.07	0.7	<0.001	0.08		4	0.2	120	<0.1
14-Jan-01	7.25	116000	106000	4000	31000	750	830	51000	<1	250	18000	<0.001	0.02	<0.01	0.44	<0.001	<0.01	<0.001	15	0.2		
1-Feb-01	7.50	120000	110000	4200	33000	770	890	51000	<1	250	12000	<0.001	<0.01	0.07	0.33	<0.001	0.3	<0.001	15	0.2	68	<0.1
19-Apr-01	7.10	120000	110000	3800	28000	750	820	52000	<1	250	1000	<0.001	0.03	0.05	0.47	<0.001	0.36	<0.001	15	0.2	91	<0.1
16-Aug-01	7.15	122000	120000	3800	33000	510	860	47000	<1	240	12000	<0.001	0.06	0.06	0.61	<0.001	0.12	<0.001	14		85	<0.1
13-Nov-01	7.00	104000	104000	3600	25000	460	720	50000	<1	200	6300	<0.001	0.1	0.3	0.1	<0.001	0.1	0.03	20	0.2	38	<0.1
14-Feb-02	7.20	112000	119000	3800	29000	670	770	48000	<1	220	13000	<0.001	<0.1	0.7	0.6	<0.001	<0.1	<0.001	<20	<0.1	62	<0.1
22-May-02	6.70	128000	98000	4730	36000	1020	903		<2	201		<0.050	0.07	0.073			0.59	<0.050	9.3			

Appendix 2 Raw Data showing heights, widths & health of individual plants, as well as cover and abundance of species in monitoring quadrats for 1998, 2000 and 2002 monitoring studies.

Site	Species	Height (m) 1998	Height (m) 2000	Height (m) 2002	Width (m) 1998	Width (m) 2000	Width (m) 2002	Health (%) 1998	Health (%) 2000	Health (%) 2002	Cover (%) 1998	Cover (%) 2000	Cover (%) 2002	Abundance 1998	Abundance 2000	Abundance 2002
1-2	Sclerostegia tenuis	.41	.53	.43	.83	1.00	.90	85	85	35	40	42	30	25	25	56
1-2	Sclerostegia tenuis	.55	.59	.61	1.01	1.15	1.53	80	95	50	.	.	.	.	.	.
1-2	Sclerostegia tenuis	.43	.49	.45	.84	.81	.68	65	63	70	.	.	.	.	.	.
1-2	Halosarcia pruinosa (a)	.38	.39	.38	.69	.49	.47	65	45	35	20	20	12	23	20	35
1-2	Halosarcia pruinosa (a)	.40	.38	.34	.54	.48	.38	35	50	50	.	.	.	.	.	.
1-2	Halosarcia pruinosa (a)	.68	.81	.70	.68	.89	.74	60	45	35	.	.	.	.	.	.
1-3	Sclerostegia tenuis	.59	.62	.67	1.06	1.29	1.08	60	75	50	13	17	40	16	21	31
1-3	Sclerostegia tenuis	.40	.46	.42	.96	1.09	.88	90	90	95	.	.	.	.	.	.
1-3	Sclerostegia tenuis	.54	.56	.70	1.26	1.49	1.33	85	90	80	.	.	.	.	.	.
1-4	H.halocnemoides catenulata	.87	.87	.90	1.24	1.23	1.23	70	70	85	20	20	25	23	24	30
1-4	H.halocnemoides catenulata	.62	.58	.69	.47	.48	.45	40	35	80	.	.	.	.	.	.
1-4	H.halocnemoides catenulata	.57	.58	.66	.67	.70	.65	40	40	85	.	.	.	.	.	.
1-4	Halosarcia pruinosa (a)	.85	.00	.00	.32	.00	.00	5	0	0	1	0	0	1	0	0
1-4	Halosarcia pruinosa (a)	.00	.00	.00	.00	.00	.00	0	0	0	.	.	.	.	.	.
1-5	Halosarcia halocnemoides (b)	.26	.28	.28	.31	.29	.23	50	50	40	20	22	25	240	350	400
1-5	Halosarcia halocnemoides (b)	.21	.19	.19	.14	.16	.16	60	50	50	.	.	.	.	.	.
1-5	Halosarcia halocnemoides (b)	.24	.19		.27	.28		60	60	.	.	.	.	.	.	.
1-5	Halosarcia fimbriata	.22	.21	.21	.08	.11	.11	60	60	50	1	1.3	1	18	22	10
1-5	Halosarcia fimbriata	.14	.14	.17	.10	.09	.12	95	83	80	.	.	.	.	.	.
1-5	Halosarcia fimbriata	.16	.15	.16	.12	.13	.11	90	80	80	.	.	.	.	.	.
2-2	Atriplex vesicaria	.24	.24	.28	.18	.20	.31	70	50	80	1	1	6	8	8	12
2-2	Atriplex vesicaria	.41	.29	.35	.82	.90	.79	60	50	75	.	.	.	.	.	.

Site	Species	Height (m) 1998	Height (m) 2000	Height (m) 2002	Width (m) 1998	Width (m) 2000	Width (m) 2002	Health (%) 1998	Health (%) 2000	Health (%) 2002	Cover (%) 1998	Cover (%) 2000	Cover (%) 2002	Abundance 1998	Abundance 2000	Abundance 2002
2-2	Atriplex vesicaria	.45	.53	.60	.42	.80	.88	75	65	80	.	.	.	.	.	.
2-2	Maireana tomentosa	.24	.24	.32	.35	.39	.58	80	80	5	5	5	3	14	13	11
2-2	Maireana tomentosa	.23	.23		.16	.16		80	80	.	.	.	.	.	.	.
2-2	Maireana tomentosa	.25	.17	.22	.37	.20	.30	40	50	40	.	.	.	.	.	.
2-2	Halosarcia pruinosa (b)	.52	.44	.50	1.04	.99	.85	85	73	65	3	3	4	4	4	4
2-2	Halosarcia pruinosa (b)	.63	.57	.59	.60	.55	.23	50	20	5	.	.	.	.	.	.
2-2	Halosarcia pruinosa (b)	.41	.37	.42	.65	.52	.52	85	75	90	.	.	.	.	.	.
2-3	Halosarcia halocnemoides (a)	.45	.41	.47	.60	.67	.82	65	60	60	10	8.5	20	29	21	26
2-3	Halosarcia halocnemoides (a)	.30	.31	.30	.20	.34	.39	55	75	50	.	.	.	.	.	.
2-3	Halosarcia halocnemoides (a)	.45	.46	.54	.50	.56	.72	70	80	60	.	.	.	.	.	.
2-4	Halosarcia doleformis	.27	.24	.25	.16	.17	.23	60	50	40	20	15	6	80	60	54
2-4	Halosarcia doleformis	.20	.26	.29	.18	.26	.19	40	30	15	.	.	.	.	.	.
2-4	Halosarcia doleformis	.35	.34	.39	.30	.37	.45	60	40	35	.	.	.	.	.	.
2-4	Halosarcia halocnemoides (a)	.24	.38	.40	.25	.20	.20	60	55	30	15	15	30	60	60	134
2-4	Halosarcia halocnemoides (a)	.25	.26	.27	.30	.22	.30	50	70	30	.	.	.	.	.	.
2-4	Halosarcia halocnemoides (a)	.35	.45	.42	.30	.35	.41	40	42	25	.	.	.	.	.	.
2-4	Halosarcia fimbriata	.48	.47	.47	.47	.40	.42	60	35	20	2	1.2	1.5	19	15	14
2-4	Halosarcia fimbriata	.37	.32	.37	.18	.14	.16	65	70	15	.	.	.	.	.	.
2-4	Halosarcia fimbriata	.32	.32	.34	.18	.10	.06	60	55	50	.	.	.	.	.	.
2-5	Halosarcia halocnemoides (a)	.47	.45		.50	.48		75	80	.	20	18	18	80	60	93
2-5	Halosarcia halocnemoides (a)	.46	.40	.40	.50	.54	.69	70	65	70	.	.	.	.	.	.
2-5	Halosarcia halocnemoides (a)	.78	.75	.76	.70	.72	.55	40	50	40	.	.	.	.	.	.
2-5	Halosarcia fimbriata	.37	.38	.52	.20	.21	.24	70	80	85	5	5	2	22	20	28
2-5	Halosarcia fimbriata	.33	.20		.18	.13		80	45	.	.	.	.	.	.	.
2-5	Halosarcia fimbriata	.33	.33	.28	.20	.21	.22	80	85	35	.	.	.	.	.	.
3-1	Halosarcia fimbriata	.50	.47	.51	.36	.45	.39	65	55	45	20	22	14	50	85	100

Site	Species	Height (m) 1998	Height (m) 2000	Height (m) 2002	Width (m) 1998	Width (m) 2000	Width (m) 2002	Health (%) 1998	Health (%) 2000	Health (%) 2002	Cover (%) 1998	Cover (%) 2000	Cover (%) 2002	Abundance 1998	Abundance 2000	Abundance 2002
3-1	Halosarcia fimbriata	.47	.44	.46	.70	.90	.83	65	50	45	.	.	.	.	.	.
3-1	Halosarcia fimbriata	.37	.45	.46	.20	.29	.31	60	75	75	.	.	.	.	.	.
3-2	H.halocnemoides (a)	.68	.79	.85	1.45	1.43	1.45	75	80	25	18	18	25	14	18	21
3-2	H.halocnemoides (a)	.75	.81	.83	1.65	1.56	1.95	80	80	30	.	.	.	.	.	.
3-2	H.halocnemoides (a)	.90	.83	.83	1.10	1.34	1.08	55	70	35	.	.	.	.	.	.
3-3	Atriplex sp.	.50	.45	.00	.42	.70	.00	80	70	0	1	2.5	1	30	4	15
3-3	Atriplex sp.	.45	.59	.00	.30	.85	.00	50	70	0	.	.	.	.	.	.
3-3	Atriplex sp.							.	.	.	.	.	.	.	.	.
3-4	Halosarcia fimbriata	.43	.42	.43	.13	.20	.18	65	55	45	15	17	25	80	120	129
3-4	Halosarcia fimbriata	.65	.15	.15	.30	.20	.20	55	30	15	.	.	.	.	.	.
3-4	Halosarcia fimbriata	.47	.54	.56	.26	.33	.36	45	60	35	.	.	.	.	.	.
3-4	Halosarcia halocnemoides (a)	.32	.33	.00	.19	.24	.00	80	90	0	5	5	6	20	30	23
3-4	Halosarcia halocnemoides (a)	.28	.34	.21	.20	.35	.13	55	60	5	.	.	.	.	.	.
3-4	Halosarcia halocnemoides (a)	.31	.42	.38	.35	.51	.49	60	68	50	.	.	.	.	.	.
4-1	Halosarcia fimbriata	.38	.35		.50	.58		70	75	.	10	9	7	43	40	71
4-1	Halosarcia fimbriata	.40	.29		.24	.29		75	55	.	.	.	.	.	.	.
4-1	Halosarcia fimbriata	.52	.54	.44	.36	.34	.34	75	75	50	.	.	.	.	.	.
4-1	Halosarcia halocnemoides (a)	.35	.43	.48	.62	.82	1.04	65	95	45	5	5	8	17	18	38
4-1	Halosarcia halocnemoides (a)	.28	.24	.24	.28	.27	.43	50	60	30	.	.	.	.	.	.
4-1	Halosarcia halocnemoides (a)	.30	.41	.51	.30	.37	.49	80	90	30	.	.	.	.	.	.
4-2	Halosarcia fimbriata	.43	.44	.46	.54	.54	.53	65	65	50	20	20	13	70	80	112
4-2	Halosarcia fimbriata	.38	.47	.36	.41	.64	.51	70	80	60	.	.	.	.	.	.
4-2	Halosarcia fimbriata	.40	.43	.48	.75	.78	.67	70	75	60	.	.	.	.	.	.
4-2	Halosarcia indica bidens	.24	.23	.19	.30	.34	.32	55	20	15	1	1	1	2	2	10
4-2	Halosarcia indica bidens	.18	.14	.13	.34	.28	.33	45	50	20	.	.	.	.	.	.
4-2	Halosarcia indica bidens	.17	.21	.13	.13	.24	.18	85	50	15	.	.	.	.	.	.

Site	Species	Height (m) 1998	Height (m) 2000	Height (m) 2002	Width (m) 1998	Width (m) 2000	Width (m) 2002	Health (%) 1998	Health (%) 2000	Health (%) 2002	Cover (%) 1998	Cover (%) 2000	Cover (%) 2002	Abundance 1998	Abundance 2000	Abundance 2002
4-3	Halosarcia indica bidens	.65	.60	1.46	1.25	1.29	.58	55	76	50	3	3	3	5	5	7
4-3	Halosarcia indica bidens	.45	.40	.39	1.00	.96	1.12	60	75	50	.	.	.	.	.	.
4-3	Halosarcia indica bidens	.63	.60	.64	.98	1.02	.93	50	45	30	.	.	.	.	.	.
4-3	Halosarcia halocnemoides (a)	.95	1.03	1.02	.85	1.10	1.20	50	90	80	18	20	4	7	6	4
4-3	Halosarcia halocnemoides (a)	.85	.81	.80	2.10	2.44	2.18	65	80	65	.	.	.	.	.	.
4-3	Halosarcia halocnemoides (a)	.70	.70	.75	.83	1.22	1.58	40	60	50	.	.	.	.	.	.
5-1	Halosarcia pruinosa (a)	.50	.51	.55	.55	.62	.70	50	70	70	70	70	55	103	100	83
5-1	Halosarcia pruinosa (a)	.49	.49		.41	.42		40	50	.	.	.	.	.	.	.
5-1	Halosarcia pruinosa (a)	.46	.48		.33	.36		35	45	.	.	.	.	.	.	.
5-2	Halosarcia pruinosa (a)	.55	.52	.45	.32	.30	.28	45	20	10	85	70	50	150	120	100
5-2	Halosarcia pruinosa (a)	.45	.46		.38	.40		45	60	.	.	.	.	.	.	.
5-2	Halosarcia pruinosa (a)	.49	.52	.44	.50	.60	.18	50	20	5	.	.	.	.	.	.
5-3	Halosarcia halocnemoides (b)	.32	.36	.38	.12	.14	.14	35	47	30	75	80	60	200	200	300
5-3	Halosarcia halocnemoides (b)	.31	.34	.52	.22	.37	.39	35	55	40	.	.	.	.	.	.
5-3	Halosarcia halocnemoides (b)	.39	.47	.44	.24	.25	.31	40	60	35	.	.	.	.	.	.
5-3	Halosarcia pergranulata pergranulata	.38	.40	.41	.48	.58	.66	60	73	45	3	3	3	18	20	11
5-3	Halosarcia pergranulata pergranulata	.49	.50	.54	.49	.70	.66	45	50	40	.	.	.	.	.	.
5-3	Halosarcia pergranulata pergranulata	.38	.42	.45	.29	.36	.32	60	55	70	.	.	.	.	.	.
5-4	Halosarcia pergranulata pergranulata	.39	.35	.32	.36	.38	.18	50	35	15	70	60	60	110	100	160
5-4	Halosarcia pergranulata pergranulata	.34	.30	.00	.38	.38	.00	40	35	0	.	.	.	.	.	.
5-4	Halosarcia pergranulata pergranulata	.29	.27		.28	.26		45	35	.	.	.	.	.	.	.

Site	Species	Height (m) 1998	Height (m) 2000	Height (m) 2002	Width (m) 1998	Width (m) 2000	Width (m) 2002	Health (%) 1998	Health (%) 2000	Health (%) 2002	Cover (%) 1998	Cover (%) 2000	Cover (%) 2002	Abundance 1998	Abundance 2000	Abundance 2002
5-5	Halosarcia halocnemoides catenulata	.49	.54	.57	.58	.55	.53	20	20	20	40	40	30	58	56	47
5-5	Halosarcia halocnemoides catenulata	.40	.50	.48	.60	.57	.67	30	30	40	.	.	.	.	.	.
5-5	Halosarcia halocnemoides catenulata	.62	.71	.71	.75	1.01	.93	40	70	40	.	.	.	.	.	.
5-6	Halosarcia pruinosa (a)	.37	.37		.57	.58		70	75	.	10	10	10	24	25	30
5-6	Halosarcia pruinosa (a)	.39	.40	.42	.37	.40	.46	65	60	75	.	.	.	.	.	.
5-6	Halosarcia pruinosa (a)	.38	.41	.42	.54	.57	.49	50	65	55	.	.	.	.	.	.
5-7	Halosarcia halocnemoides (a)	.91	.96	.98	1.51	2.30	2.42	60	80	68	12	13	20	10	8	6
5-7	Halosarcia halocnemoides (a)	.89	.99	.93	1.21	1.35	1.74	65	80	60	.	.	.	.	.	.
5-7	Halosarcia halocnemoides (a)	1.02	1.10	1.08	1.21	1.25	1.15	50	50	45	.	.	.	.	.	.
5-7	Halosarcia pterygosperma denticulata	.40	.55	.52	.62	.75	.86	90	85	45	1	2	3	1	2	2
5-7	Halosarcia indica bidens	.41	.27	.20	.46	.62	.64	35	15	15	6	6	12	7	9	9
5-7	Halosarcia indica bidens	.59	.54	.53	1.12	1.60	1.77	35	50	65	.	.	.	.	.	.
5-8	Halosarcia fimbriata	.26	.28	.31	.46	.56	.61	60	30	15	18	8	3	55	30	14
5-8	Halosarcia fimbriata	.30	.17	.25	.49	.35	.57	55	30	75	.	.	.	.	.	.
5-8	Halosarcia fimbriata	.28	.00	.00	.21	.00	.00	60	0	0	.	.	.	.	.	.
6-1	Sclerostegia tenuis	.40	.56	.00	1.20	1.73	.00	70	90	0	5	6	3	5	7	3
6-1	Sclerostegia tenuis	.40	.48	.60	.90	1.00	1.36	80	80	35	.	.	.	.	.	.
6-1	Sclerostegia tenuis	.25	.36		.55	.76		90	80	.	.	.	.	.	.	.
6-2	Halosarcia halocnemoides (a)	.65	.58	.64	.60	.55	.58	70	60	40	35	35	25	54	49	60
6-2	Halosarcia halocnemoides (a)	1.00	1.00	.91	1.45	1.55	1.63	30	40	45	.	.	.	.	.	.
6-2	Halosarcia halocnemoides (a)	.65	.75	.69	.75	.72	.82	45	80	50	.	.	.	.	.	.
6-3	Halosarcia fimbriata	.40	.35	.40	.16	.20	.22	65	60	35	25	24	10	160	180	152
6-3	Halosarcia fimbriata	.35	.00	.00	.25	.00	.00	50	0	0	.	.	.	.	.	.

Site	Species	Height (m) 1998	Height (m) 2000	Height (m) 2002	Width (m) 1998	Width (m) 2000	Width (m) 2002	Health (%) 1998	Health (%) 2000	Health (%) 2002	Cover (%) 1998	Cover (%) 2000	Cover (%) 2002	Abundance 1998	Abundance 2000	Abundance 2002
6-3	Halosarcia fimbriata	.40	.35	.41	.18	.19	.16	40	55	40	.	.	.	.	.	.
6-3	Halosarcia fimbriata	.38	.43	.46	.46	.65	.82	50	60	40	.	.	.	.	.	.
6-3	Halosarcia fimbriata	.52	.36	.35	.50	.60	.58	50	30	30	.	.	.	.	.	.
6-3	Halosarcia fimbriata	.68	.60	.49	.56	.58	.46	45	38	20	.	.	.	.	.	.
6-4	Halosarcia fimbriata	.68	.55	.43	1.00	1.33	.35	80	30	15	40	34	20	70	57	285
6-4	Halosarcia fimbriata	.33	.40	.33	.30	.24	.26	85	35	50	.	.	.	.	.	.
6-4	Halosarcia fimbriata	.46	.42	.42	.56	.81	.90	80	45	40	.	.	.	.	.	.
7-1	H.doleformis	.22	.24		.43	.43		60	40	.	18	18	.	85	81	.
7-1	H.doleformis	.40	.35		.35	.41		65	55	.	.	.	.	.	.	.
7-1	H.doleformis	.24	.23		.25	.27		50	50	.	.	.	.	.	.	.
7-2	Halosarcia fimbriata	.60	.58	.46	.34	.45	.61	95	85	75	5	4	5	43	37	10
7-2	Halosarcia fimbriata	.28	.25	.50	.22	.22	.52	95	50	60	.	.	.	.	.	.
7-2	Halosarcia fimbriata	.30	.29	.00	.27	.31	.00	80	5	0	.	.	.	.	.	.
7-3	Halosarcia fimbriata	.35	.35	.34	.40	.36	.52	60	60	70	35	35	25	75	100	.
7-3	Halosarcia fimbriata	.30	.29	.27	.40	.41	.44	50	60	50	.	.	.	.	.	.
7-3	Halosarcia fimbriata	.30	.30		.42	.42		35	60	.	.	.	.	.	.	.
7-3	Halosarcia halocnemoides (b)	.25	.25	.28	.40	.48	.45	50	80	50	25	30	20	58	60	.
7-3	Halosarcia halocnemoides (b)	.25	.29	.22	.35	.34	.43	50	55	55	.	.	.	.	.	.
7-3	Halosarcia halocnemoides (b)	.22	.22	.27	.44	.45	.41	60	63	60	.	.	.	.	.	.

Appendix 3. Photographs of selected monitoring sites taken at April/May 2002.

SITE 1-2

SITE 1-4

SITE 1-5

SITE 3-1

SITE 3-2

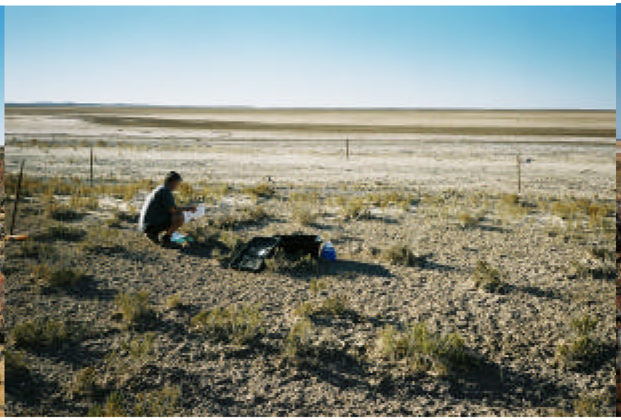
SITE 4-1

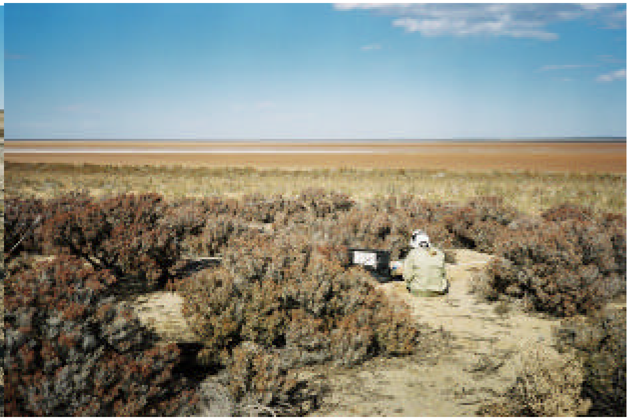

SITE 4-2


SITE 5-3


SITE 5-4


SITE 5-6


SITE 5-8


SITE 6-2