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Sediment Quality Triad (SQT) assessment of surface sediments in the Lower Lakes

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January 2015

Interim Report-Commercial in confidence

Interim report in fulfilment of project milestones 1.3 and 2.3 of project 298/0586 for:

Ann Marie Jolley and Liz Barnett; Department of Environment, Water and Natural Resources, as part of the Coorong, Lower Lakes and Murray Mouth Recovery Program

Land and Water Flagship Report series ISSN:

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Citation: Kumar A, Corbin T, Doan H, Grocke S, Gonzago D, Kirby J, Mosley L. and Shand P. (2015). Sediment Quality Triad (SQT) assessment of surface sediments in the Lower Lakes

CSIRO Technical report: Land and Water Flagship.

Site Photographs by Hai Doan and Sonia Grocke

Contents

Figures

Tables

Acknowledgements

We would like to acknowledge the support, comments and project management provided by Ann Marie Jolley and Dr Liz Barnett for this project. This work was funded by DEWNR and CSIRO. The authors acknowledge Dr Andrew Baker for his advice on site selection in the Lower Lakes area and for sharing monitoring data from the previous studies conducted in this area. We thank the following CSIRO staff for analytical and logistical support: Claire Wright, Rory O' Brien and Lintern Fairbrother. CSIRO would also like to thank Graham Lancaster (Director, Environmental Analysis Laboratory) and his team at Southern Cross University for completing the acid-base accounting analysis. Paul McEvoy from DEWNR is thanked for his detailed review of earlier draft of this report.

This work was funded through the South Australian Government's *Murray Futures* initiative and the Australian Government. This interim report satisfies Milestones 1.3 and 2.3 of SA DEWNR Project 298/0586 -The research of ecotoxicological effects of acidification on key aquatic organisms in the Lower Lakes.

Australian Government

Government of South Australia

Department of Environment, Water and Natural Resources

Executive Summary

The Millennium Drought (2000-2010) caused large expanses of previously long-inundated sediments and subaqueous soils to be exposed around the margins of lakes Alexandrina and Albert in South Australia from 2007 to mid-2010. This exposed acid sulfate soil (ASS) material became progressively oxidised over increasing depths in the soil profiles. The resultant formation of sulfuric materials (pH < 4) produced significant water quality and ecological problems.

The main focus of earlier field and laboratory investigations of ASS environments in lakes Alexandrina and Albert was measuring and analysing the physico-chemical parameters at the two lakes. Based on the seven years of monitoring (2007 to 2014), the subaqueous soils in the Lower Lakes are in a transient state and the build-up of sulfide is likely to continue under saturated conditions. In the present study, the Sediment Quality Triad (SQT) was used to investigate the extent and significance of sediment pollution, and of pollution-induced degradation, within sediments that are transitioning through the recovery from drought-induced acidification. Three lines of evidence (chemistry, ecotoxicity and benthic community) were integrated to compare and classify sites in lakes Alexandrina and Albert according to their degrees of potential hazard and impact.

Surface sediment samples were collected in March and November 2013 from seventeen Lower Lake sites which were impacted during a severe drought three years earlier. Sediments were analysed for a variety of metals, ions and acidity metrics. Bioassays were conducted on *Chironomus tepperi* using both acute and chronic endpoints. The benthic macroinvertebrate community at each site was also investigated for those species considered to be either acid tolerant or acid sensitive. The three lines of evidence were investigated using a scoring technique for the likelihood of potential impacts due to the sediment quality. Of the 17 sites monitored in the region, those which were acidic (pH<6.5) totalled seven in March and five in November. Only three sites (Milang, Poltalloch and Dog Lake) had acidic surface sediment during both times. In the present study, a mixture of metals including Al, Cu, Zn, As, Mn and Co were above ANZECC guideline values in the pore water collected from the sediments at some sites. Most sites, particularly in March 2013, had quite low SQT scores (<2) suggesting surface sediments at these sites were not impacted by acidification. However, two sites may be of concern; Boggy Creek on Hindmarsh Island and Boggy Lake on the north-western side of Lake Alexandrina, which received the highest score in March and November 2013 respectively. Results from the SQT suggest that while most surface sediments studied in the Lower Lakes region in 2013 were likely unimpacted from acidification, Boggy Creek and Boggy Lake could be considered to be hot spot sites and may remain impacted from ecological issues relating to ASS. A comparison between pre-drought and post-drought benthic communities was not possible due to limited data on the pre-drought baseline. We found the SQT approach to be a useful tool to assess the impact of ASS in a lake environment, despite some of the current effects from acidification being only minor. If this SQT assessment had been undertaken during or straight after the drought ended, there may have been more obvious impacts in the lakes due to ASS.

This is an interim report and part of a three year investigation. A final synthesis report will be written which will include all investigations of the Lower Lakes Ecotoxicology project.

Recommendations:

- The current investigation was carried out using surface sediments from the sediment-water interface (up to 10 cm sediment depth) of the lakes. An upward flux of contaminants may pose a moderate to high level risk to the biota inhabiting ASS impacted sites. This should be investigated at selected sites.
- Due to heterogeneity of the soils and seasonal/flow changes, further ecotoxicological monitoring is recommended to assess the spatial and temporal variation in the toxicity at selected Lower Lakes sites.
- The development of rapid monitoring tools and modelling approaches should be considered. They would utilise chemical, physical and microbial parameters to enable assessment of sediment health and impacts of stress-induced changes.
- In the present study, a mixture of metals including Al, Cu, Zn, As, Mn and Co were above ANZECC guideline values in the pore water collected from the sediments at some sites. As the potential for multiple chemical exposure increases, the question raised is whether the toxicity of mixtures of chemicals is simply additive or whether there is potentiation of toxicity. The general consensus has been that chemicals interact by concentration addition, however past studies have demonstrated that concentration addition of the components of a mixture does not always reflect the overall interaction of a mixture. The risk assessment procedures should account for mixtures of contaminants present in a given system.
- Pore water Al had the highest hazard quotient. Thus, it becomes obvious that Al is a significant hazard associated with ASS. Unfortunately, there is a notable shortage of literature on the biological response of the aquatic biota to Al released from ASS. The locations and soil characteristics of ASS are well defined throughout the literature. Similarly, Al is recognised as a highly toxic element when bioavailable. However, Al forms a range of chemical species, little is known about speciation, bioavailability and toxicity when it comes to a system dominated by ASS. The current Al guideline is applicable at pH <6.6. The ANZECC/ARMCANZ water quality guidelines require review for aluminium, particularly in relation to deriving guideline value(s) for aluminium toxicity in lower pH water. The sediment guidelines for aluminium should also be reviewed.

1 Introduction

The Millennium Drought (2000-2010) resulted in unprecedented lower water levels in lakes Alexandrina and Albert (the Lower Lakes) during 2006-2010 with extensive areas of acid sulfate soils (ASS) being exposed. This caused soil acidification (pH<4) over areas around the margins of this Ramsar listed site with acidification of surface waters in some localised areas (Figure 1). During the drought in the Murray-Darling Basin, water flow to the Lower Lakes was reduced to unprecedented levels (as low as -1 m AHD, Figure 2). This resulted in drying out of the lake margins, exposure of pyrite-containing ASS to oxidation, resulting in severe acidification (Fitzpatrick *et al.* 2008). Sediments in the Lower Lakes region that were exposed in the drought have since been re-inundated. Monitoring since then has shown that these sediments pose a hazard to aquatic biota due to their persistent acidification and the presence of acidification products in the sediments (Fitzpatrick *et al.* 2011). Fitzpatrick *et al.* (2011) found that 75% of test sites in Currency Creek and Finniss River in the west of the region had a pH <7 and 21% had a pH <4. Hicks *et al.* (2009) studied pore waters in field mesocosms, concluding that metal mobilisation is a significant risk. Pore water studies using peeper sampling devices noted that only the top 5 cm or less of the sediment profile had been neutralised (to a pH >4) following 24 months of inundation of lake water (Creeper *et al*. 2015). ANZECC guideline trigger values for many metals such as aluminium, arsenic, boron, cadmium, chromium, copper, manganese, nickel, lead and zinc were exceeded in the pore water at these sites. A detailed study at selected sites using sequential extractions (Shand *et al*. 2012) highlighted a wide range of contaminants present in pore water, and also complex associations with a range of different solid phases.

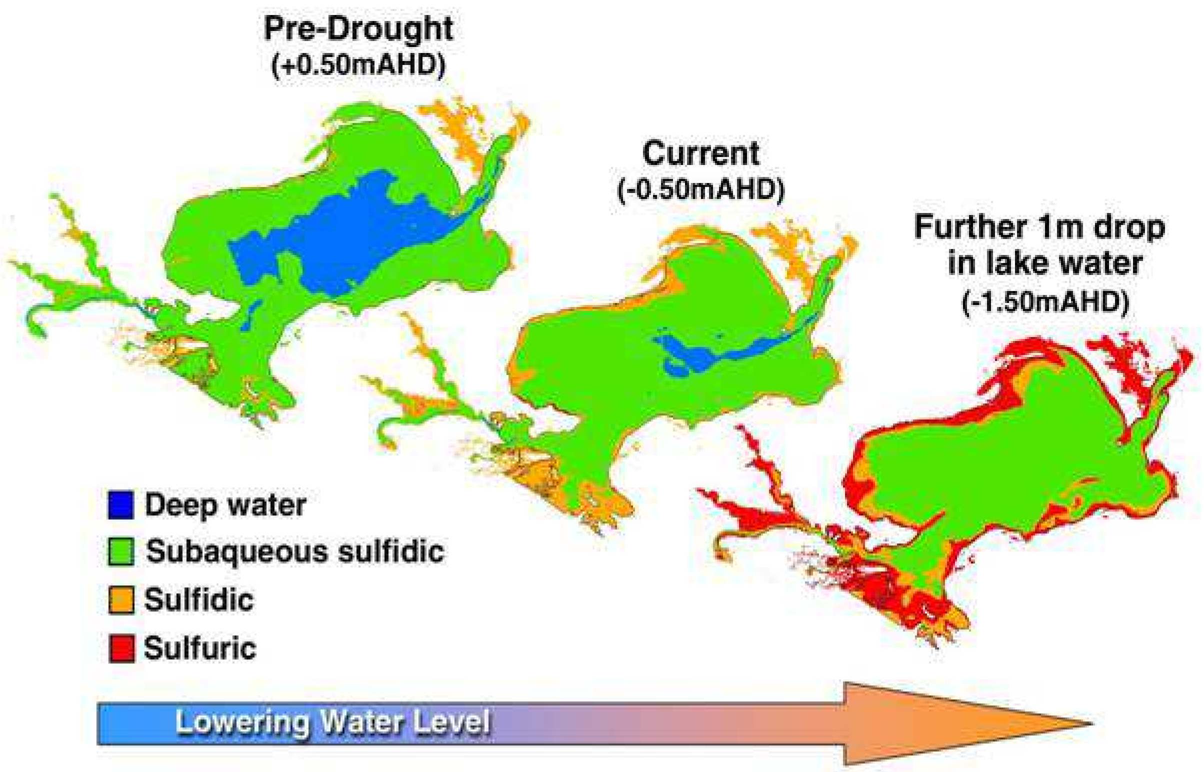


Figure 1 Predictive scenario maps depicting changes in ASS materials at different levels in Lake Albert (+0.5 m AHD, -0.5 m AHD and -1.5 m AHD; adapted from Fitzpatrick *et al.* 2008b)

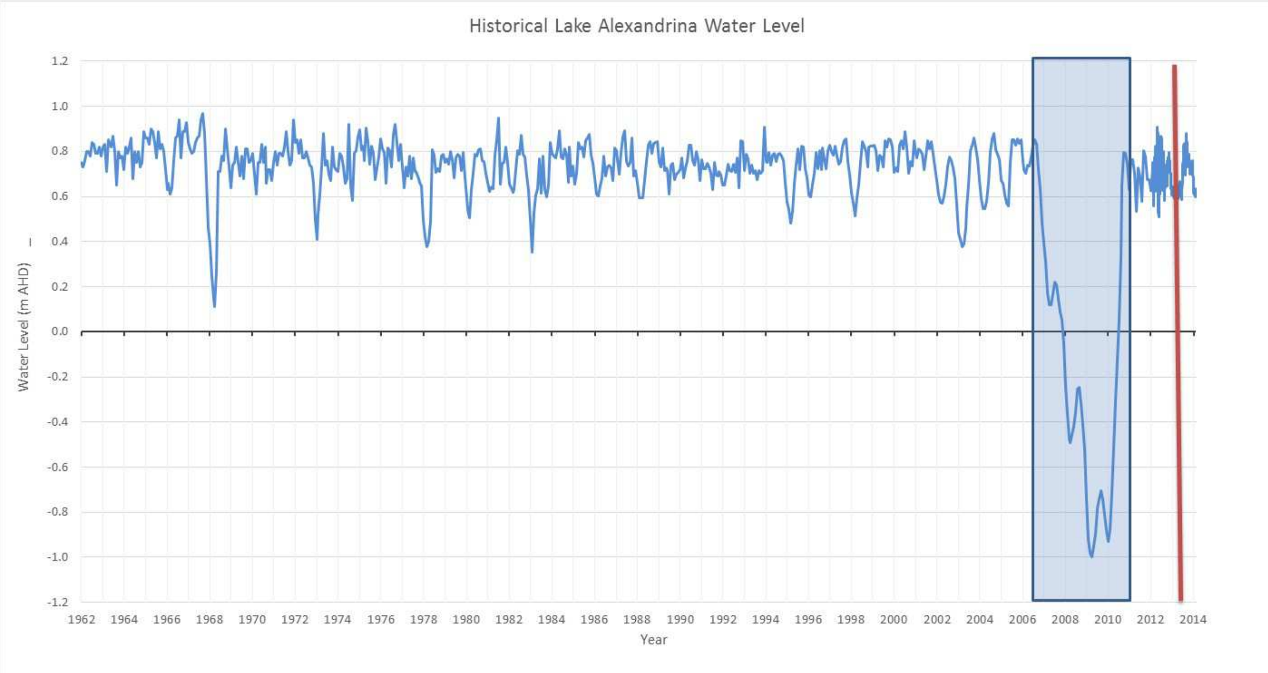


Figure 2 Historical Lake Alexandrina water levels. The blue rectangle indicates the drought period and the red line indicates when the sampling for this project was undertaken

Sediments constitute an association between organic and inorganic particles and living organisms. They provide a habitat for many aquatic organisms (bacteria to macroinvertebrates) and are a major repository for many of the more persistent contaminants that are introduced into the water environment. Contaminants in sediments may be directly toxic to aquatic life or can be a source of chemicals for bioaccumulation in the food chain. Although certain contaminants are highly sorbed to sediment, these may still be available to the biota. The sediment-bound contaminants may become resuspended from wetland beds into the water column by physical (water currents and wind) or biological (bioturbation) processes. (Rainbow *et al*., 2007; Simpson and Batley, 2007; Proux and Hare, 2014). Bioturbation involves redistribution of sediment particles within the sediment column and sediment resuspension in the overlying water. These processes are a consequence of burrowing, feeding and defecation, and tube-building activities by benthic organisms, fish and other wetland biota (e.g waterbirds, water rats, turtles). Additionally, contaminated sediments may be directly toxic to the benthic organisms that inhabit them or be a source of contaminants for bioaccumulation in the food chain. Toxicity tests measure the cumulative effects of all bioavailable contaminants and their interactions as mixtures. Different benthic species have different sensitivities to different toxicants, and organisms possess differing exposure pathways (feeding strategies and behaviours); thus, a suite of test endpoints (e.g. survival, development, growth, reproduction) should be used for ecotoxicological assessment of contaminated sediments.

Macroinvertebrates play an important part in the aquatic food web, being the major consumers of all type of organic materials (Bunn *et al*. 1999). They, in turn, are a major food source for many vertebrates (e.g. fish, birds, tortoises, water rats and frogs). Awareness of the groups that are sensitive or tolerant to changing environmental conditions can assist in understanding the changes that are occurring within the aquatic environment. Research in other locations has shown that acid and metals from oxidised ASS can lead to severe ecological effects (Sammut *et al.* 1995). Macroinvertebrates have also been well studied in South Australia and are therefore suitable for studies on aquatic ecosystem condition. The impact of ASS on the ecosystems and key aquatic organisms of the Lower Lakes and Coorong is not well understood but is crucial in understanding the ecological significance of the risks posed by acid sulfate soils in the region. As there are still acidic materials and pore water present in previously exposed sediments, there is potential for the recovery of the ecosystem, in particular its benthic organisms, to be hindered. Monitoring of the macroinvertebrate community in the surface sediment of the Lower Lakes from May 2009–May 2011 found that while improvements in macroinvertebrate health occurred as water levels returned to pre-drought levels, molluscs and some crustaceans had still not returned (Giglio 2011). Results from sampling in February 2012 and 2013 conducted by the AWQC Aquatic Ecology & Biomonitoring team indicated some return of macro invertebrate fauna toward pre-drought levels as described in the SA EPA/AWQC Report (2013). In May 2012 a Benthic Ecosystem Toxicity Assessment (BETA) pilot study (Corbin *et al.* 2012) was conducted which investigated the pore water chemistry, benthic composition and chironomid deformity rates at two sites that were impacted by ASS (at lower Currency Creek and the lower Finniss River) and one site not affected by ASS on the Finniss River.

The pilot study found:

- Low pH levels (less that the ANZECC trigger value of 6.5) below a depth of 8 cm in the sediment profile at the sites known to be impacted by acid sulfate soils
- High ammonia concentrations at the impacted sites, although, other toxicants such as metals and metalloids were below the ANZECC trigger values suitable to protect 95% of species (except for boron, chromium and manganese below a depth of 15 cm at Currency Creek).
- Low overall deformity rates in chironomids, although the deformities of more major structures in larvae from Currency Creek in comparison with the reference site may suggest sub-lethal impacts to biota at this site
- The sediment cores showed that most organisms living in the sediment were found in the top 5 cm, with only a few individuals found deeper. However, very few organisms were collected from sediment in Currency Creek below 2 cm.

This pilot study was conducted on a very limited number of sites. Hence, further investigations are required to confirm if the above findings are representative across the Lower Lakes region.

The recommendations from this report included the use of the Sediment Quality Triad approach (henceforth referred to as the SQT); a three-pronged approach to determine the extent and significance of sediment pollution and pollution-induced degradation. The SQT involved assessment of the sediment chemistry, benthic community assemblage and sediment toxicity through bioassays (Figure 3, Long and Chapman 1985). The SQT approach integrates data from: a) chemical analyses - to determine potential exposure of benthic organisms to toxicants; b) laboratory toxicity tests - to determine relative response, and c) analyses of the composition of resident benthic assemblages - to determine possible *in-situ* adverse effects. The data from the components of the SQT are both treated separately and also integrated for an overall assessment of sediment quality.

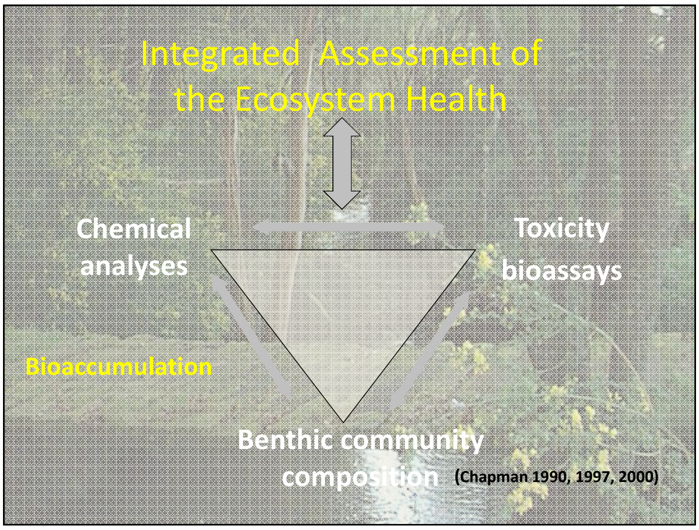


Figure 3 Sediment Quality Triad approach based on multiple lines of evidence

The combination of the potential cause (chemistry) and effect (biology) makes the SQT a powerful tool when investigating sediment pollution and potential impacts (Chapman 1990). The SQT approach has been used overseas, particularly in the U.S. for the past few decades, in studies on estuaries and marine environments (e.g. Chapman *et al.* 1987, Riba *et al.* 2004, McDonald 2005, Pinkney *et al.* 2005, Lee *et al.* 2006, Hartwell *et al.* 2009), freshwater environments (Canfield *et al*. 1994, Soucek *et al.* 2000) and has also been trialled on groundwater environments near industrial sites (Crevecoeur *et al.* 2011). While the basic SQT uses three lines of evidence to determine the extent and significance of sediment pollution, more lines of evidence can also be incorporated generating a 'weight of evidence' approach. Since the SQT was initially proposed in the 1980s, researchers have established different methods of interpreting and combining these lines of evidence to categorise study sites or regions as unimpacted or impacted (Chapman *et al.* 1997, Long *et al.* 2004, Bay and Weisberg 2012).

The aim of the present study was to use the SQT approach to assess the potential hazard and impact of drought induced acidification on surface sediments in the Lower Lakes region that are transitioning through recovery; This assessment will inform management decisions to mitigate ASS risks and promote the continued recovery of this important Ramsar site.

Specific objectives included:

1. Assessment of spatial and temporal variation of ecotoxicological effects of drought-induced acidification in the sediments of the Lower Lakes that are transitioning through recovery;
2. Identification of hot-spots in the Lower Lakes region based on the SQT approach;
3. Assessment of bioavailability of contaminants from ASS under *in-situ* conditions; and
4. Evaluation of toxic and synergistic effects of key contaminants to selected key aquatic organisms.

In the current study, the SQT approach was used to address the following guiding questions:

1. Are contaminants present in sediments at concentrations considered a potential hazard?
2. Are contaminants bioavailable?
3. Is there a measurable biological response?
4. Are the contaminants causing this biological response?

This is an interim report and part of a three year investigation. A final synthesis report will be written which will include all investigations of the Lower Lakes Ecotoxicology project.

2 Approaches and methodology

The sampling locations for this study were chosen to coincide with a Lower Lakes region monitoring project performed by CSIRO (Baker *et al.* 2014) (Figure 4; Table 1).

Figure 4 Lower Lakes sediment sampling locations for March and November 2013 (from Baker *et al*., 2014)

Table 1 SQT project surface sediment sample locations within each sampling site

CODE	WATERBODY	SITE NAME	EASTING [A]	NORTHING [A]	SEDIMENT ACIDITY [B]	DOMINANT SUBSTRATE [B]
LF01	Finniss River	Wally's Landing	303198	6079714	Neutral, Neutral	Fine, Fine
LF02	Lake Alexandrina	Point Sturt North	321247	6070294	Acidic, Neutral	Coarse, Coarse
LF03	Lake Alexandrina	Milang	316106 (316024)	6079440 (6079440)	Acidic, Acidic	Coarse, Coarse
LF04	Lake Alexandrina	Tolderol	331889	6083697	Neutral, Neutral	Coarse, Coarse
LF06	Lake Alexandrina	Poltalloch	339011 (338928)	6070334 (6070125)	Acidic, Acidic	Coarse, Coarse
LF07	Lake Albert	Waltowa	352376	6059074	Neutral, Neutral	Medium, Fine
LF08	Lake Albert	Meningie	349125	6049311	Neutral, Neutral	Medium, Medium
LF10	Lake Albert	Campbell Park	341261	6056503	Neutral, Acidic	Medium, Medium
LF12	Lake Alexandrina	Loveday Bay	326379	6061724	Acidic, Neutral	Coarse, Coarse
LF13	Lake Alexandrina	Tauwitcherie	319050 (319092)	6060550 (6060494)	Neutral, Neutral	Medium, Medium
LF15	Lake Alexandrina	Boggy Creek	311139	6065855	Acidic, Neutral	Fine, Medium
LF17	Lake Alexandrina	Point Sturt South	314849	6069780	Neutral, Neutral	Coarse, Coarse
LF19	Lake Alexandrina	Dog Lake	331551	6086656	Acidic, Acidic	Fine, Medium
LF20	Lake Alexandrina	Boggy Lake	334997	6089162	Neutral, Acidic	Fine, Fine
LF21	Lake Albert	Windmill Site	345597	6064184	Neutral, Neutral	Coarse, Coarse
LF23	Currency Creek	Currency Creek	301055	6072892	Acidic, Neutral	Coarse, Coarse
LF24	Finniss River	Finniss River South	305763	6073896	Neutral, Neutral	Fine, Fine

[A] = co-ordinates listed are the locations of the sediment core samples collected in March 2013, co-ordinates in brackets are the locations sampled in November, if they differed substantially from the March samples.

[B] = listed as ‘results for March 2013, results for November 2013’. Measurements were recorded from the top 10 cm of sediment in March and the top 5 cm in November.

Sediment samples for chemical and ecotoxicological analyses were collected to a depth of 10 cm in March 2013. However, as most benthic organisms collected in March were found in the top 2 cm of sediment, this was reduced to 5 cm for ecotoxicological studies and 2 cm for benthic assessments in November 2013.

Samples were collected at sites (n=3 replicates for each of the three lines of evidence) with a handheld spade that was cleaned between samples with surface water. Samples were collected at a water depth between 50 and 120 cm.

Samples were placed into large sealable plastic bags for chemical analyses, and glass jars (approximately 4 L) for toxicity testing. All samples had a minimum head space and were stored on ice for transportation to the laboratory. The samples were stored in a laboratory refrigerator at 4^{o}C until analysis. At the time of the benthic fauna sampling, 250 mL sediment samples were also collected. The sediment was washed through a net with a mesh size of 250 µm in the field to remove fine sediment before being placed in a plastic screw top container and preserved with methylated spirits.

2.1 Benthic samples

Rose Bengal solution was added to each sample to increase the visibility of the invertebrates they contained. All samples were inspected under a dissection microscope and macroinvertebrates and microcrustacea were identified to the lowest taxonomic level possible and enumerated. Whole samples for each replicate were inspected (i.e. no sub-sampling occurred).

A list of taxa considered tolerant to acidity [acid tolerant taxa (ATT)] and those considered sensitive to acidity [acid sensitive taxa (AST)] were compiled using South Australian EPA data (unpublished) and findings from studies conducted on acidified water (Anthony 1999, Orendt 1999, Gerhardt *et al.* 2004, Tripole *et al.* 2008, Sommer and Horwitz 2009, Hogsden and Harding 2012). The abundance of ATT and AST identified from each site were recorded.

Taxa considered acid tolerant for the purposes of this study included nematodes, the cladocerans *Macrothrix* and *Neothrix*, and the midges *Chironomus*, *Tanytarsus* sp. and *Tanytarsus barbitarsis*. Taxa considered acid sensitive included hydra, turbellaria, snails, amphipods, shrimp, ostracods, mayflies, caddisflies and odonates. The Benthic Reponse Index (Smith *et al.* 2001) was adapted for assessing the acid tolerance of the benthic community. Taxa in the sample for which acid tolerance was not determined were excluded from the analysis. All other taxa were assigned either a value of 1 if the species is considered sensitive to acidity or a value of 5 if the species was considered acid tolerant.

2.2 Midge survival and larval development test – sediment toxicity test

Sediment toxicity was assessed in the laboratory using the first instar larvae of *Chironomus tepperi* midges. For the assay, ten 5-day old midge larvae (first instar) were added to beakers containing ca. 140 g (wet weight) of 425 µm sieved field collected sediment and 200 mL moderately hard water, with eight replicate beakers per sediment. After a 5-d incubation at 21°C (16:8 light:dark cycle) and prior to pupation, midge larvae are removed from 4 replicate beakers and their lengths measured. Survival rates are also determined. The sediment exposures to midge larvae were continued for another four replicate beakers to measure emergence and sex ratios after 12 days exposure. The test methods are summarised in Table 2.

Larvae from aquaria-raised midges were used for the toxicity tests. Five days prior to testing, egg masses were collected from cultures maintained at CSIRO, Adelaide, and placed in 1 L beakers (3-4 egg masses/beaker) with 800 mL of moderately hard water (MHW). As an artificial substrate, 7.5 g of shredded tissue was also added to each beaker. The water quality parameters of MHW was as follows: electrical conductivity (EC) 220 – 300 µS/cm, pH 6.9 to 7.9, dissolved oxygen (DO) >60%. Over the next 5 days, egg masses in these beakers were aerated continuously, fed twice with ground fish flakes (4 g/100 mL), and incubated under constant temperature conditions (21 ± 1°C) with a 16:8 h light:dark photo period using cool-white fluorescent lamps (10-20 µmol photons/s/m^2). Five-day-old larvae (first instar) were used for testing. The cultures were considered suitable for use in toxicity tests if they provided a constant supply of larvae, if the larvae were healthy and behaved normally, and if mortality was $\leq$ 10%.

For the survival and growth bioassay, ten 5-day old midge larvae were added to beakers containing 140 g pre-weighed (wet weight) of 1 mm sieved sediment and 250 mL MHW, with 4 replicate beakers per sediment. Each beaker was incubated under the same conditions described above. After 5 days, and prior to pupation, midge larvae from each replicate beaker were removed by sieving the sediments and collecting live midge larvae. These larvae are fixed in 10% buffered formalin and processed for their length measurements using the Stereomicroscope and the image analyses system. Survival of the midge larvae in each beaker was also determined.

Development (emergence of the larvae from sediment) was determined as outlined in Table 2. The number of emerging adult *C. tepperi,* and their sex, were measured daily. Males are easily distinguished from females because males have large, plumose antennae and a much thinner abdomen with visible genitalia. The pH, EC, DO and temperature of the surface waters were measured at the beginning, end and every day in which surface waters were renewed.

In order to determine if the reference cultures had an appropriate sensitivity, a reference toxicant test, using copper, was also carried out using *C. tepperi* larvae from the same batch of cultures used in the sediment bioassay. A water-only reference toxicant test was used in preference to a whole-sediment test due to the greater ease and reduced time spent in preparing and conducting the test. For the reference toxicant test, a 48 h survival test was conducted using copper at concentrations ranging from 16-512 µg/L.

Table 2 Summary of the test conditions for the midge *Chironomus tepperi* bioassays

TEST PARAMETER	TEST CONDITION
Test type	Static renewal
Test duration	Survival and growth: 5 d
	Larval development: 12-14 d
Temperature	21 ± 1˚C
Light quality	Cool-white fluorescent tube lighting
Light intensity	10-20 µmol photons s^{-1} m^{-2}
Photoperiod	16 h light : 8 h dark
Test chamber size	400 mL
Test solution volume	140 g sediment plus 250 mL MHW
Age of test organisms	2nd Instar larvae, 5 days
No. of organisms per replicate	10
No. of replicates per treatment	8
No. of organisms per treatment	80
Feeding regime	Midges fed during exposure period
Test chamber aeration	Aeration provided
Dilution water/overlying water	Moderately hard water (MHW, 230 mg $CaCO_3$/L)
Endpoint	5 day: Survival and growth 12-14 days: larval development (emergence) and sex ratio
Test acceptability criterion	≥80% survival in controls; Reference toxicant LC50 within Cusum limits

2.3 Laboratory chemical analysis methods

2.3.1 Pore water analysis

Pore water was collected from sediments following centrifugation. Sediments were transferred into 50 mL centrifuge tubes and centrifuged for 25 minutes at 3500 rpm (Sorvall R3C3 Plus). The supernatant fluid was then removed and filtered using 0.45 µm syringe filters (Millex GV Durapore PVDF) into 50mL tubes. The pore water samples were filtered through 0.45 µm filters and divided into two subsamples. One sub-sample was, as previously described, used for the measurement of pH, EC, DO, and major anions, by a Hach HQd water quality meter and Eh using a TPS WP81 meter and Ionode IJ64 Redox electrode. The second was acidified to pH~2 and kept at 4^{o}C until chemical analysis by ICP-AES and ICP-MS.The pore water samples were analysed for chemistry as follows: (i) alkalinity/acidity (ii) TOC, (iii) major anions (Cl, NO_3, NH_4, PO_4, SO_4), (iv) major cations (Al, Fe, Mn, Na, K, Ca, Mg), and (v) trace elements (As, Cd, Co, Cr, Cu, Ni, Zn). Various instrumentation methods were used for pore water analyses as shown in Table 3.

Table 3 Instrumental methods used for analyses of pore-water samples

ANALYTE	METHOD
Dissolved metals by ICP-AES	Dissolved metals were measured by ICP-AES (CIROS, SPECTRO). The sample is converted to an aerosol and transported into the plasma. Atoms and ions of the plasma are excited and emit light at characteristic wavelengths. The light emitted by the sample passes through the entrance slit of the spectrometer. The different wavelengths are measured and converted to a signal and quantified by comparison with standards.
Dissolved metals by ICP-MS	Dissolved metals were measured by ICP-MS (Agilent 7500 CE). Analyte species originating in a liquid are nebulised by a Micromist nebuliser and a cooled double-pass spray chamber. The ions are detected by an electron multiplier. The ions are quantified by comparison with prepared standards.
Alkalinity and Acidity as calcium carbonate	APHA 21st ed., 2320 B. This procedure determines alkalinity by both manual measurement and automated measurement (PC Titrate) using pH 4.5 for indicating the total alkalinity end-point. Acidity is determined by titration with a standardised alkali to an end-point pH of 8.3.
Major anions – filtered	APHA 21st ed., 4500 Cl - B. Automated Silver Nitrate titration.
Chloride	APHA 21st ed., 3120; USEPA SW 846 - 6010 The ICP-AES technique ionises filtered sample atoms emitting a characteristic spectrum. This spectrum is then compared against matrix matched standards for quantification.
Nitrite and nitrate as N	APHA 21st ed., 4500 NO_3^- I. Nitrate is reduced to nitrite by way of a cadmium reduction column followed by quantification by flow injection analyser (FIA). Nitrite is determined separately by direct colourimetry and result for Nitrate calculated as the difference between the two results.
Reactive phosphorus – filtered	APHA 21st ed., 4500 P-E. Water samples are filtered through a 0.45 µm filter prior to analysis. Ammonium molybdate and potassium antimonyl tartrate reacts in acid medium with othophosphate to form a heteropoly acid -phosphomolybdic acid - which is reduced to intensely coloured molybdenum blue by ascorbic acid. Quantification is achieved by FIA.
Total organic carbon (TOC)	APHA 21st ed., 5310 B. The automated total organic carbon (TOC) analyzer determines Total and Inorganic Carbon by IR cell. TOC is calculated as the difference.
Moisture content	A gravimetric procedure based on weight loss over a 12-24 h drying period at 110±5ºC.
Paste pH, conductivity	Paste pH (USEPA 600/2-78-054): pH determined on a saturated paste by ISE. Electrical Conductivity of Saturated Paste (USEPA 600/2-78-054) - conductivity determined on a saturated paste by ISE.

2.3.2 Elemental analyses of surface sediments

Strong acid microwave digestion of surface sediments

Total metal analyses of soil sub-layers were conducted following acid digestion using the US EPA method 3051A (revised version 2007) microwave assisted acid digestion of sediments, sludges, soils and oils (US Environmental Protection Agency, Washington, DC.) The dried sample was digested in a microwave oven

(MARS CEM) using a mixture of concentrated nitric acid and hydrochloric acid (3:1 (v/v) respectively). Approximately 0.25 g dry soil was weighed into Teflon digest vessels with 2.5 mL HCl and 7.5 mL HNO_3 and left overnight to cold digest. After cold digestion, the microwave vessels were sealed and microwave digested using the following time and temperature program: ramp to 110 ^{o}C in 10 min, ramp to 180^{o}C in 10 min and maintain temperature at 180°C for 60 min. After cooling, the digest solutions were filtered through filters with pore size 0.45 µm and analysed for total elements by ICP-AES and ICP-MS. The digest solutions were analysed for a range of elements (Al, As, B, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Mo, Na, Ni, P, Pb, S, Sb, Se and Zn) by ICP-AES and ICP-MS using the method described in Table 3.

A few refractory sample matrix compounds, such as quartz, silicates, titanium dioxide, alumina, and other oxides may not have been totally digested using this strong acid digestion procedure. In this study, elemental concentrations using microwave strong acid digestion are considered the total pool of elements potentially bioavailable to biota.

1M hydrochloric acid extraction

The potential bioavailable or mobile pool of elements in surface sediments was assessed using 1M HCL. The concentration of elements in 1M HCL extracts will (in general) be lower than total elemental concentrations determined using strong acid digestion/extraction because elements often present in fixed pools associated with organic matter or a range of mineral precipitates are not readily mobilized by weak acids.

Hydrochloric acid (1M, 40 mL) was added to 1 g field wet soil (± 0.1 g) in a 50 mL centrifuge tube and extracted for 4 hours on an end over shaker. Samples were centrifuged at 3500 rpm for 30 min and the supernatants removed. The samples were filtered using Millex Nylon 0.45 µm syringe filters and analyzed for a range of elements (As, B, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Mo, Na, Ni, P, Pb, S, Sb, Se and Zn) by ICP-AES and ICP-MS.

2.4 SQT methodology

The possible scenarios using three components in the SQT approach are outlined in Table 4.

The chemical, toxicological and biological variables were each categorised using a scoring factor between 0 and 3 (Table 5) to indicate the likelihood of potential impacts. A total score for each site was calculated as the sum of the scores for each of the variables. Standardised scores for each site were calculated by dividing the sum of the score of each line of evidence (LOE; chemistry, toxicity and biota) by the number of variables used within each LOE and these fractions were then summed together. The standardised approach was used to account for the unequal number of variables that were used in each LOE (Table 5).

Table 4 Possible scenarios from three components of SQT approach

Situation	*Chemistry*	*Toxicity*	*Community*	*Biomagnific*	*Conclusion* *(Note + different to reference, - equivalent to reference)*
1	+	+	+	+	Strong evidence for sediment contamination
2	-	-	-	-	Strong evidence for no contamination
3	+	-	-	-	Contaminants unavailable
4	-	+	-	-	Unmeasured chemicals or conditions exist with potential to cause contamination and resistance may have developed
5	-	-	+	-	Alteration not due to chemical contamination
6	+	+	-	-	Toxic chemicals stressing system but resistance has developed
7	-	+	+	-	Unmeasured toxic chemicals causing contamination
8	+	-	+	-	Chemicals are not bioavailable or alteration is not due to toxic chemicals

Table 5 Table of variables used in the assessment and the scores given to each category

	SCORE 0	1	2	3
Chemistry				
Sediment metals	No metals exceed sediment trigger values[a]	ND	ND	ND
Pore water analytes	No analytes exceed trigger values[b]	1 analytes exceed trigger values[b]	2 or 3 analytes exceed trigger value[b]	ND
Sediment pH (pH units)	>6.5	6.0-6.5	ND	ND
TAA (mol H^+/tonne)	0	1 or 2	3-5	>5
AVS (mol H^+/tonne DW)[c]	<18	18-62	>62	ND
Toxicity to midge larvae				
% survival	>80	60-79	45-59	<45
% growth inhibition	<10	10-24	25-34	>35
% emergence	>80	60-79	45-59	<45
Sex ratio	No skewed ratio	ND	Skewed ratio	ND
Biota				
ATT/AST	No ATT	AST>ATT	ATT>AST	ND

ND = Not determined as scores not required for data collected. ATT = acid tolerant taxa, AST = acid sensitive taxa

[a] = trigger values provided in the ANZECC/ARMCANZ (2000) Sediment Quality Guidelines
[b] = trigger values set for the protection of 95% of species in freshwater systems (ANZECC/ARMCANZ 2000), which were adjusted for hardness of Lake Alexandrina and Lake Albert.
[c] = AVS only measured in November survey.

3 Results and discussion

3.1 Sediment toxicity assessment

No acute toxic effects were observed in either sampling occasion, with ≥90% survival of *Chironomus tepperi* larvae exposed to sediments collected from the 17 Lower Lake sites. However, sub-lethal effects were seen in the form of growth inhibition, emergence success and skewed sex ratio of the adults (Figures 5-8). The results in the March 2013 survey were different to those seen in the November survey, indicating the possibility of temporal differences in toxicity of sediment, possibly due to changes in chemistry.

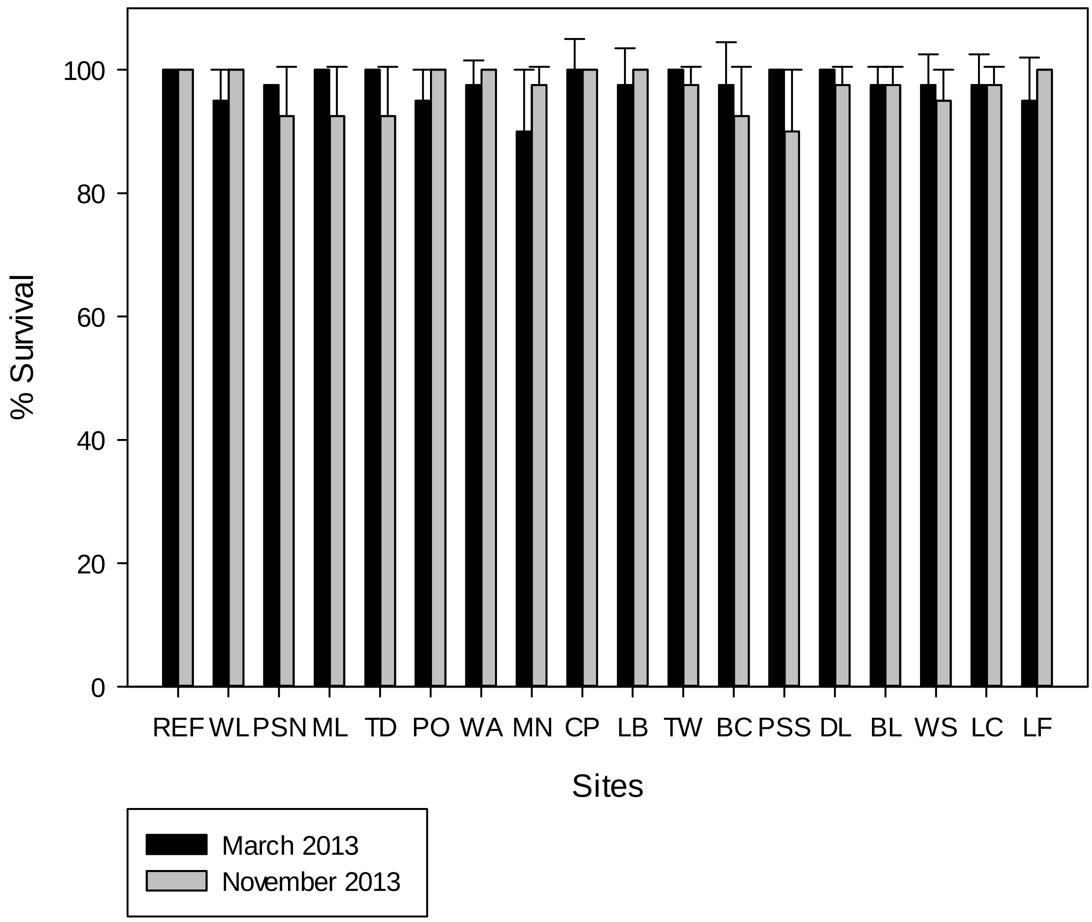


Figure 5 Survival of midge larvae exposed to sediments collected from 17 sites in March and November 2013

REF-Reference site; WL-Wallys Landing; PSN- Point Sturt North; ML- Milang; TD- Tolderol; PO- Poltalloch; WA- Waltowa; MN-Meningie; CP- Campbell Park; LB- Loveday Bay; TW- Tauwitcherie; BC- Boggy Creek; PSS- Point Sturt South; DL- Dog Lake; BL- Boggy Lake; WS- Windmill Site; LC- Lower Currency; LF- Lower Finniss

Error bars represent one standard deviation.

Midge growth mean data results were subjected to a Shapiro –Wilk normality test and passed as normally distributed to validate the use of linear statistics to analyse for site variability versus the reference site using SigmaPlot 12.5 statistical software. Inhibition to midge growth was evident at five sites (Point Sturt North, Milang, Tolderol, Poltalloch and Lower Finniss) in March but no sites had statistically significant (α =0.05) differences from the reference site in November, although some greater variability was present at Loveday Bay and Campbell Park (Figure 6). Emergence success was analysed by one way ANOVA (α =0.05) using SigmaPlot 12.5 statistical software, and three sites (Meningie, Lower Finniss, and Windmill Site) had the most significantly reduced emergence, with Boggy Lake also showing a reduction in emergence success in March 2013. In contrast, emergence of midge larvae exposed to sediments collected in November2013 was reduced at four sites (Wally's Landing,, Meningie, Campbell Park and Point Sturt South), and the most significant reduction was at one site (Boggy Lake; Figure 7).

A skewed sex ratio (<40% or >60% of one sex) occurred at only two sites (Loveday Bay and Lower Finniss) in March 2013 but six sites (Waltowa, Campbell Park, Boggy Lake, Point Sturt South, Dog Lake and Lower Currency Creek) in November (Figure 8). For all three chronic endpoints, differences due to sampling time were apparent (Figures 6-8).

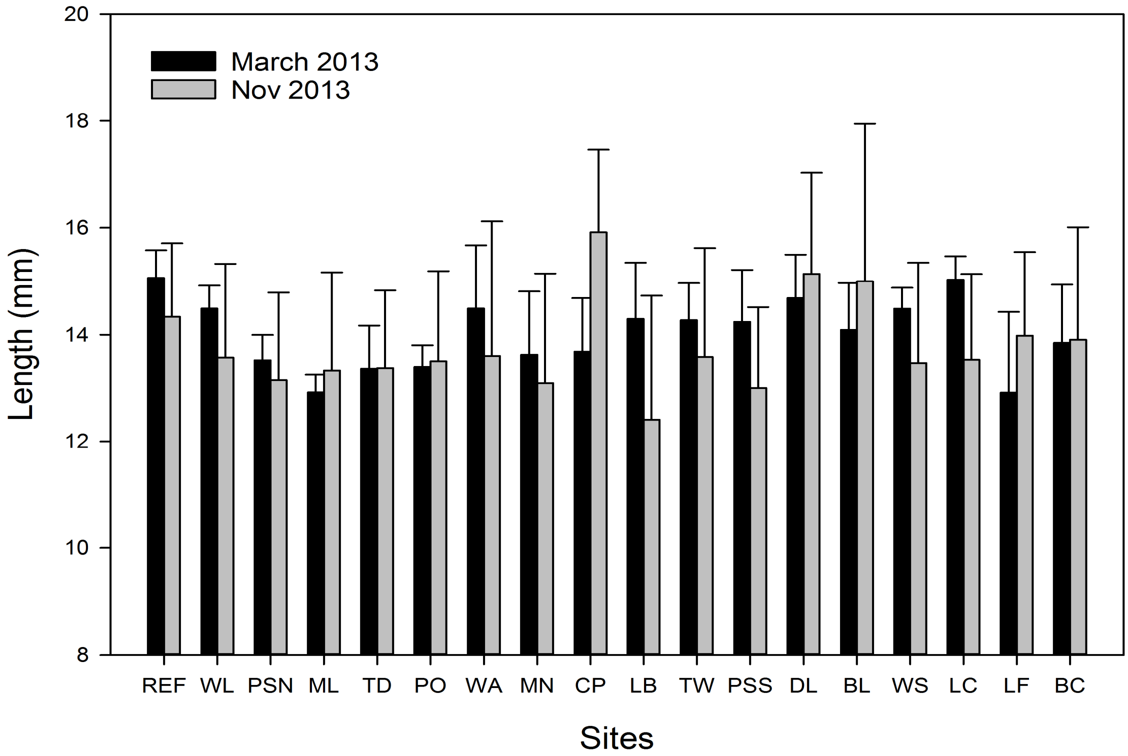


Figure 6 Growth in midge larvae exposed to sediments collected from 17 sites in March and November2013

REF-Reference site; WL-Wallys Landing; PSN- Point Sturt North; ML- Milang; TD- Tolderol; PO- Poltalloch; WA- Waltowa; MN-Meningie; CP- Campbell Park; LB- Loveday Bay; TW- Tauwitcherie; BC- Boggy Creek; PSS- Point Sturt South; DL- Dog Lake; BL- Boggy Lake; WS- Windmill Site; LC- Lower Currency; LF- Lower Finniss

Error bars represent one standard deviation.

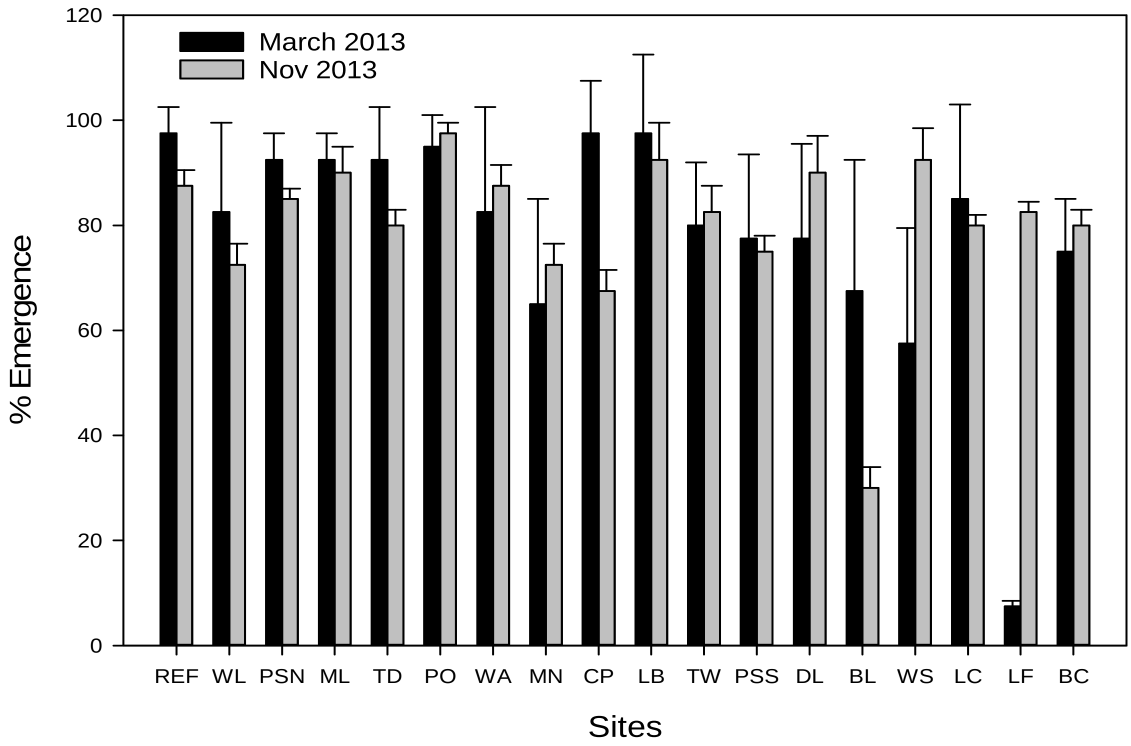
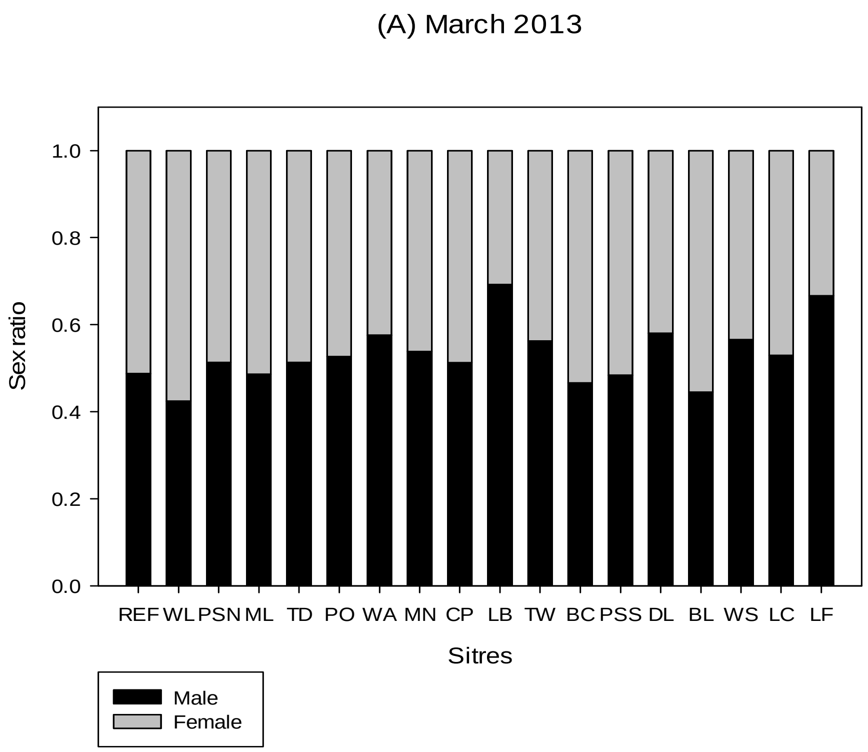


Figure 7 Emergence of midge larvae exposed to sediments collected from 17 sites in March and November 2013

REF-Reference site; WL-Wallys Landing; PSN- Point Sturt North; ML- Milang; TD- Tolderol; PO- Poltalloch; WA- Waltowa; MN-Meningie; CP- Campbell Park; LB- Loveday Bay; TW- Tauwitcherie; BC- Boggy Creek; PSS- Point Sturt South; DL- Dog Lake; BL- Boggy Lake; WS- Windmill Site; LC- Lower Currency; LF- Lower Finniss

Error bars represent one standard deviation.



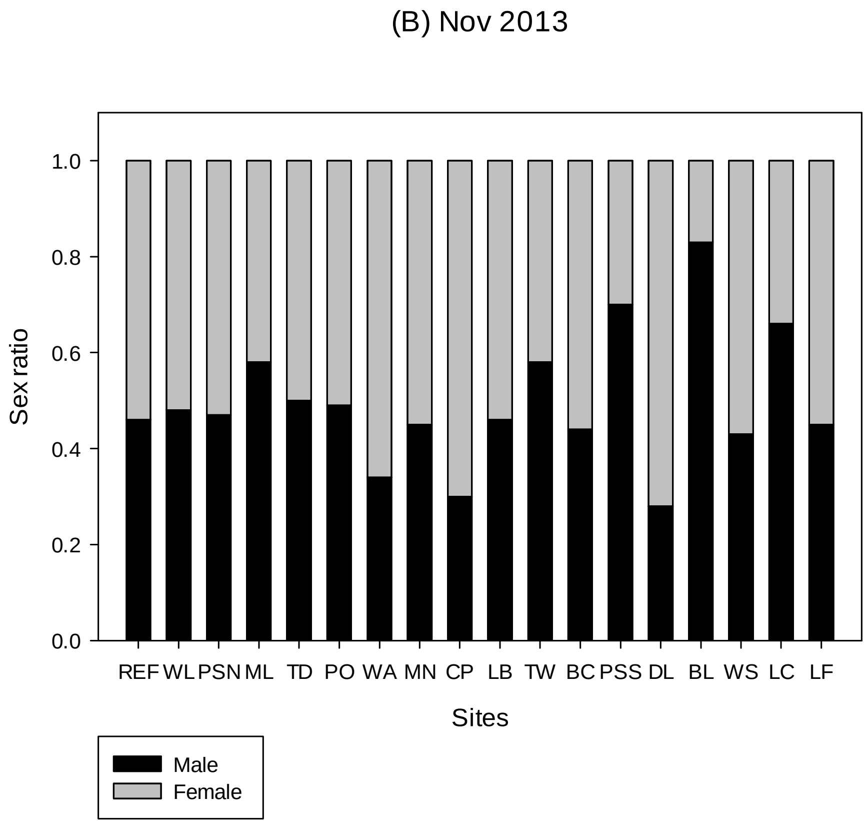


Figure 8 Sex ratio of midge larvae exposed to sediments collected from 17 sites in (A) March and (B) November 2013

REF-Reference site; WL-Wallys Landing; PSN- Point Sturt North; ML- Milang; TD- Tolderol; PO- Poltalloch; WA- Waltowa; MN-Meningie; CP- Campbell Park; LB- Loveday Bay; TW- Tauwitcherie; BC- Boggy Creek; PSS- Point Sturt South; DL- Dog Lake; BL- Boggy Lake; WS- Windmill Site; LC- Lower Currency; LF- Lower Finniss
Data represented as mean values.

3.2 Benthic community characterisation

Benthic species composition, abundance, and biomass are influenced by many habitat conditions including salinity, sediment type, and environmental stressors, both natural and anthropic (Slim *et al.* 1997, Nanami *et al.* 2005). Information on changes in benthic population and community parameters due to habitat change can be useful for separating natural variation from changes associated with human activities. For that purpose, benthic community studies have a long history of use in regional monitoring programs and have been proven to serve as an effective indicator for describing the extent and magnitude of pollution impacts and habitat modification in freshwater and estuarine ecosystems, as well as for assessing the effectiveness of management actions (Llanso *et al*. 2003, Long *et al.* 1985).

The macroinvertebrate fauna collected from the Lower Lakes in March comprised of total identified 68 taxa; 36 genera from 28 families and 17 orders. In November, a total of 70 taxa were identified; 36 genera from 29 families and 15 orders. The benthic community was dominated by nematodes and oligochaetes with these two taxa comprising 93% of the benthic individuals in March and 76% in November. Chironomids comprised a low percentage of the community composition - at just 1.9% in March and 1.2% in November. A greater abundance of zooplankton was seen in November in comparison to the March samples.

Very low abundances of acid tolerant taxa were collected in March (Corbin *et al.*, 2014; Table 6). However, higher abundances were seen in November at most sites, due mainly to the increase in zooplankton in the lakes, and the increase in abundance of the cladocerans *Macrothrix* and *Neothrix*. The abundance of acid sensitive taxa also increased from March to November. These seasonal changes in the benthic community would be expected to occur naturally and are likely not a reflection of impacts due to acid sulfate soils.

Table 6 Abundance of acid tolerant taxa (ATT) and acid sensitive taxa (AST) at each site for both surveys

		MARCH		NOVEMBER	
		ATT	AST	ATT	AST
LF01	Wallys Landing	3	64	73	56
LF02	PtSturt North	0	33	2	30
LF03	Milang	0	11	4	51
LF04	Tolderol	0	11	0	1
LF06	Poltalloch	0	12	0	13
LF07	Waltowa	0	36	29	561
LF08	Meningie	0	7	39	26
LF10	Campbell Park	1	40	207	138
LF12	Loveday Bay	0	4	3	762
LF13	Tauwitcherie	13	25	0	56
LF15	Boggy Creek	14	4	322	58
LF17	PtSturt South	0	2	0	3
LF19	Dog Lake	0	17	54	80
LF20	Boggy Lake	0	12	436	108
LF21	Windmill Site	0	26	3	248
LF23	Lower Currency	0	3	1	68
LF24	Lower Finniss	2	56	3	91

AST is defined as species that have difficulty in maintaining ion regulation at low pH, calcified shells, vulnerable exoskeletons or herbivores susceptible to food scarcity in acidified environments. ATT not impacted. (Unpublished EPA monitoring data)

3.3 Chemical characterisation

Results from pore water analyses are summarised in Tables 7-8. The pore water concentrations for Al and Cu were above guideline (ANZECC/ARMCANZ, 2000) values (Al: 55 µg/L, Cu: 1.4 µg/L) at most sites sampled in March 2013. Arsenic was high at Dog Lake, Campbell Park and Boggy Lake sites in March 2013 and at Tauwitcherie site in November 2013. The pore water concentrations of other metals such as Mn, Zn, Co and Cr were reported to be above guideline levels at some sites (Tables 7-9).

Based on the Hazard Quotient (HQ) calculations from the pore water data in March 2013, Wallys Landing and Point Sturt South were the most contaminated sites with cumulative HQ>100 and Dog Lake, Boggy Lake, Boggy Creek and Lower Finniss were considered moderately contaminated with the cumulative HQ ranging between 0.9-23.3 (Table 10). In contrast, in November 2013 the cumulative HQ values for the 17 sites ranged between 0.2-11.9 with Wally's Landing being the most contaminated site and Point Sturt North being a moderately contaminated site (Table 11).

The surface sediment pH ranged from 6.07-8.27, with most sites found to be circumneutral and within the pH range considered not to cause significant impacts on ecosystems (6.5 to 8.0; freshwater, lakes and reservoirs trigger value; ANZECC/ARMCANZ 2000) during both sampling events. There were seven sites with a pH less than 6.5 in March and five such sites in November. Three sites had a sediment pH less than 6.5 on both occasions (Figure 9).

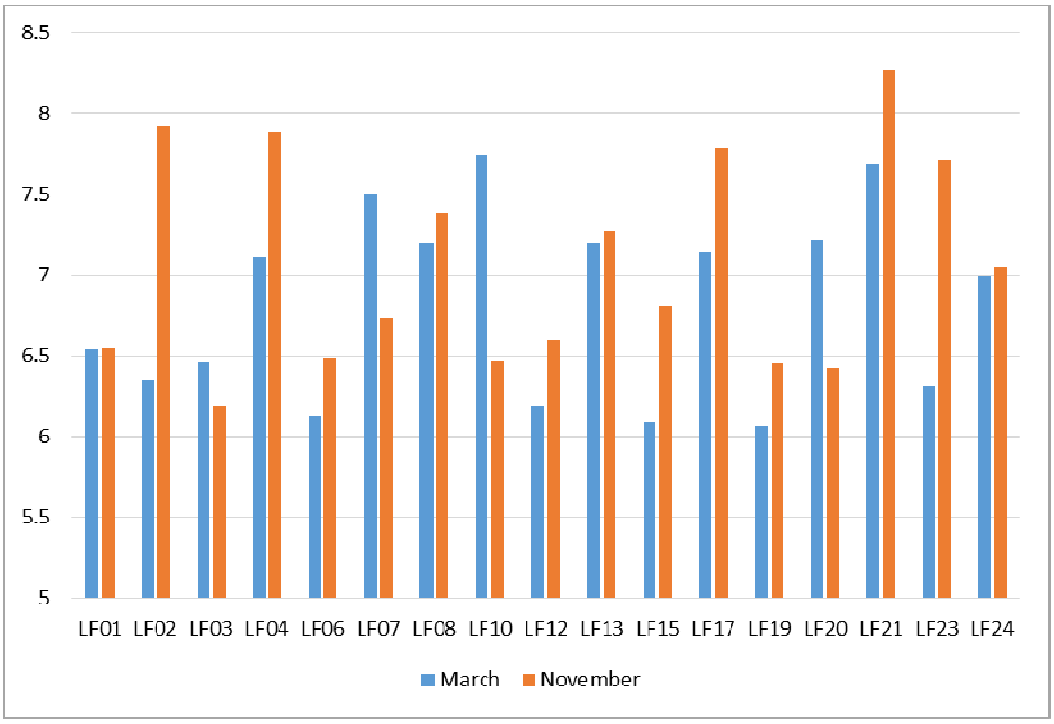


Figure 9. Surface sediment pH values recorded at each of the 17 sampling sites in March and November 2013.

LF01- Wallys Landing; LF02- Point Sturt North; LF03- Milang; LF04- Tolderol; LF06- Poltalloch; LF07- Waltowa; LF08- Meningie; LF10- Campbell Park; LF12- Loveday Bay; LF13- Tauwitcherie; LF15- Boggy Creek; LF17- Point Sturt South; LF19- Dog Lake; LF20- Boggy Lake; LF21- Windmill Site; LF23- Lower Currency; LF24- Lower Finniss

Elemental concentrations in whole sediments and 0.1M HCL extracts were, in the majority of cases, below ISQG-low sediment guideline values (Tables 12-14). These findings suggest that even though total metal concentrations in whole soil layers are below those considered to impact environmental health, the metal concentrations in pore waters (the most bioavailable fraction) may still be present at concentrations that could impact the health of aquatic organisms.

Table 7 Pore water concentrations of elements at 17 sites in March 2013

SITE	Ca (mg/L)	K (mg/L)	Mg (mg/L)	Na (mg/L)	S (mg/L)	Al (µg/L)	As (µg/L)	Co (µg/L)	Cr (µg/L)	Cu (µg/L)	Fe (µg/L)	Mn (µg/L)	Mo (µg/L)	Ni (µg/L)	Pb (µg/L)	U (µg/L)	Zn (µg/L)
Wally's Landing	25	8.0	19	154	9.2	**2270**	7.2	**2.2**	**4.9**	**31**	7170	251	0.1	4.0	**5.9**	**1.4**	**979**
Point Sturt North	16	10	6.1	29	41	**75**	4.2	0.30	0.2	**9.7**	178	202	2.9	7.0	0.05	0.03	6.0
Milang	12	11	7.6	29	21	**144**	2.1	0.1	0.2	**10**	126	2.3	1.7	3.2	0.14	0.03	3.7
Tolderol	16	8.3	7.3	28	17	**59**	1.5	0.1	0.2	**11**	42	1.0	1.2	6.7	0.05	0.03	3.3
Poltalloch	43	14	9.0	31	28	**49**	1.9	0.1	0.2	**13**	10	16	3.0	3.5	0.05	**4.0**	1.8
Waltowa	26	14	21	80	16	54	8.5	0.42	0.2	**11**	83	0.6	7.3	1.8	0.05	**2.0**	1.8
Meningie	53	14	29	69	25	4.0	3.5	0.1	0.2	0.80	5.0	0.1	7.8	1.3	0.05	**4.1**	0.80
Campbell Park	26	13	22	109	27	43	**14**	0.51	0.2	**12**	119	0.43	7.1	2.4	0.05	**2.2**	1.3
Loveday Bay	10	7.5	5.7	27	24	**79**	2.1	0.1	0.2	**11**	29	0.20	5.8	0.2	0.05	0.03	2.0
Tauwitcherie	21	8.3	15	35	18	**51**	2.8	0.1	0.2	**7.9**	149	422	0.80	0.90	0.05	0.10	5.0
Boggy Creek	16	13	11	80	48	**197**	10	0.80	1.6	**16**	6220	239	0.80	4.2	1.9	**0.76**	6.0
Point Sturt South	12	6.6	6.3	36	3.5	**981**	3.6	0.1	1.6	**16**	621	9.2	0.90	3.3	1.5	**0.62**	**402**
Dog Lake	24	10	19	43	30	**324**	**26**	**2.6**	0.50	**11**	5900	878	2.5	5.0	0.80	0.30	**15**
Boggy Lake	13	12	8.8	56	13	**267**	**19**	0.30	1.0	**16**	537	2.3	8.9	4.8	**3.9**	**1.7**	4.0
Windmill Site	23	13	18	46	37	39	7.7	0.1	0.2	**9.0**	1219	212	2.9	0.2	0.05	0.30	0.4
Lower Currency	15	8.2	13	26	35	31	4.1	0.1	0.2	**8.0**	1455	212	2.8	0.2	0.05	0.03	**11**
Lower Finniss	17	10	16	96	1.8	**289**	5.2	0.1	1.2	**12**	912	75	2.1	4.0	0.80	**2.1**	6.0

Values in red are higher than the trigger values obtained from guidelines in Table 9

Table 8 Pore water concentrations of elements at 17 sites in November 2013

SITE	Ca (mg/L)	K (mg/L)	Mg (mg/L)	Na (mg/L)	S (mg/L)	Al (µg/L)	As (µg/L)	Co (µg/L)	Cr (µg/L)	Cu (µg/L)	Fe (µg/L)	Mn (µg/L)	Mo (µg/L)	Ni (µg/L)	Pb (µg/L)	U (µg/L)	Zn (µg/L)
Wally's Landing	57	8.2	50	284	11	**1128**	2.3	0.60	NA	**2.4**	1458	966	0.40	0.80	0.05	0.60	**601**
Point Sturt North	57	14	19	91	33	6.4	2.6	0.30	NA	**1.4**	10	148	5.2	2.0	0.05	**4.5**	2.3
Milang	23	14	17	126	15	26	12.5	0.1	NA	**2.1**	25	1.7	1.5	0.70	0.05	0.070	1.0
Tolderol	12	6.7	7.8	57	5.1	**152**	2.3	0.1	NA	**1.8**	16	6.0	0.70	1.3	0.05	0.25	0.4
Poltalloch	81	11	25	104	62	0.70	1.5	**5.1**	NA	**1.3**	1269	1387	1.0	4.5	0.05	**1.9**	3.8
Waltowa	104	18	67	356	17	0.90	4.0	**2.2**	NA	1.2	1031	**3260**	0.1	1.8	0.05	0.03	1.1
Meningie	74	26	84	504	49	3.8	5.8	0.1	NA	**1.9**	112	414	9.9	1.6	0.05	**4.6**	1.0
Campbell Park	109	25	85	431	67	2.0	5.7	**6.6**	NA	**1.5**	5162	**2799**	1.1	3.9	0.05	**1.6**	1.5
Loveday Bay	29	16	24	117	23	7.9	2.8	0.40	NA	0.90	55	715	1.1	0.80	0.05	0.03	0.80
Tauwitcherie	18	13	21	155	2.3	39	**14**	0.1	NA	**2.4**	105	742	1.5	1.0	0.05	0.20	1.9
Boggy Creek	31	11	25	138	6.1	**95**	2.6	0.1	NA	**2.0**	924	877	1.4	1.5	2.0	0.30	1.3
Point Sturt South	19	11	7.0	73	7.5	**657**	1.4	0.1	NA	**12**	712	53	2.8	2.0	0.05	**0.96**	**358**
Dog Lake	52	10	39	141	17	6.9	1.8	**3.0**	NA	1.2	794	**3468**	1.3	3.3	0.05	**1.0**	2.2
Boggy Lake	29	10	22	124	14	11	3.4	**2.3**	NA	**6.8**	259	626	1.2	**3.9**	0.05	**0.60**	0.90
Windmill Site	50	27	56	379	43	9.0	3.9	0.1	NA	1.4	12	1.7	7.5	1.3	0.05	**7.1**	1.8
Lower Currency	32	17	28	171	33	3.4	0.90	**1.9**	NA	1.3	169	**1360**	0.1	1.6	0.05	0.03	5.3
Lower Finniss	21	5.7	18	97	8	**57**	1.5	0.40	NA	**1.6**	550	245	2.5	0.80	0.05	**2.9**	2.4

Values in red are higher than the trigger values obtained from guidelines in Table 9

Table 9 Water quality guideline values for elements in freshwater

	Al (µg/L)	Ag (µg/L)	As (µg/L)	Cd (µg/L)	Co (µg/L)	Cr (µg/L)	Cu (µg/L)	Ni (µg/L)	Pb (µg/L)	Zn (µg/L)	Fe (µg/L)	Mn (µg/L)
WQG (95%PC; TV ~30 g $CaCO_3$/L) [a]	55	0.05	13	0.2	1.4	3.3	1.4	11	3.4	8	NV	1900
WQG (hardness=60) [b]	55	0.05	13	0.36	1.4	5.9	2.5	20	8.2	14	NV	1900

[a] WQG (95%PC) = ANZECC/ARMCANZ (2000) WQG trigger value (TV) for 95% species protection applicable to freshwaters of hardness 30 mg $CaCO_3$/L. Values provided are without hardness correction.
As(V) = 13 µg/L / As(III) = 24 µg/L, Cr(VI) = 1 µg/L / Cr(III) = 3.3 µg/L. NV = no guideline value.
[b] Hardness-adjusted WQGs for Ag, Cd, Cr, Cu, Ni, Pb and Zn applicable to fresh waters.

Table 10 Hazard quotient for elements in pore water at 17 sites in March 2013

SITE	Al	Ag	As	Cd	Co	Cr	Cu	Ni	Pb	Zn	Mn	Cumulative HQ
Wally's Landing	**41.27**	0.55	**1.57**	**1.48**	**22.14**	0.36	**1.74**	**122.38**	0.13	**41.27**	0.55	**23.3**
Point Sturt North	**1.36**	0.32	0.21	0.06	**6.93**	0.64	0.01	0.75	0.11	**1.36**	0.32	**1.21**
Milang	**2.62**	0.16	0.07	0.06	**7.14**	0.29	0.04	0.46	0.001	**2.62**	0.16	**1.36**
Tolderol	**1.07**	0.12	0.07	0.06	**7.86**	0.61	0.01	0.41	0.001	**1.07**	0.12	**1.14**
Poltalloch	0.89	0.15	0.07	0.06	**9.29**	0.32	0.01	0.23	0.008	0.89	0.15	**1.21**
Waltowa	0.98	0.65	0.30	0.06	**7.86**	0.16	0.01	0.23	0.0003	0.98	0.65	**1.19**
Meningie	0.07	0.27	0.07	0.06	0.57	0.12	0.01	0.10	0.0001	0.07	0.27	0.16
Campbell Park	0.78	**1.08**	0.36	0.06	**8.57**	0.22	0.01	0.16	0.0002	0.78	**1.08**	**1.20**
Loveday Bay	**1.44**	0.16	0.07	0.06	**7.86**	0.02	0.01	0.25	0.0001	**1.44**	0.16	**1.14**
Tauwitcherie	0.93	0.22	0.07	0.06	**5.64**	0.08	0.01	0.63	0.22	0.93	0.22	0.90
Boggy Creek	**3.58**	0.77	0.57	0.48	**11.43**	0.38	0.56	0.75	0.13	**3.58**	0.77	**2.29**
Point Sturt South	**17.84**	0.28	0.07	0.48	**11.43**	0.30	0.44	**50.25**	0.005	**17.84**	0.28	**9.92**
Dog Lake	**5.89**	2.00	**1.86**	0.15	**7.86**	0.45	0.24	**1.88**	0.46	**5.89**	**2.00**	**2.87**
Boggy Lake	**4.85**	1.46	0.21	0.30	**11.43**	0.44	**1.15**	0.50	0.001	**4.85**	**1.46**	**2.67**
Windmill Site	0.71	0.59	0.07	0.06	**6.43**	0.02	0.01	0.05	0.11	0.71	0.59	0.94
Lower Currency	0.56	0.32	0.07	0.06	**5.71**	0.02	0.01	**1.38**	0.11	0.56	0.32	0.91
Lower Finniss	**5.25**	0.40	0.07	0.36	**8.57**	0.36	0.24	0.75	0.04	**5.25**	0.40	**2.17**

Values in red are higher than 1 and are a potential hazard

Table 11 Hazard quotient for elements in pore water at 17 sites in November 2013

SITE	Al	Ag	As	Co	Cr	Cu	Ni	Pb	Zn	Mn	Cumulative HQ
Wally's Landing	**20.51**	0.18	0.43	**1.71**	0.07	0.01	**75.13**	0.51	**20.51**	0.18	**11.92**
Point Sturt North	0.12	0.20	0.21	**1.00**	0.18	0.01	0.29	0.08	0.12	0.20	0.24
Milang	0.47	0.96	0.07	**1.50**	0.06	0.01	0.13	0.001	0.47	0.96	0.46
Tolderol	**2.76**	0.18	0.07	**1.29**	0.12	0.01	0.05	0.003	**2.76**	0.18	0.74
Poltalloch	0.01	0.12	**3.64**	0.93	0.41	0.01	0.48	0.730	0.01	0.12	0.65
Waltowa	0.02	0.31	**1.57**	0.86	0.16	0.01	0.14	**1.7158**	0.02	0.31	0.51
Meningie	0.07	0.45	0.07	**1.36**	0.15	0.01	0.13	0.2179	0.07	0.45	0.30
Campbell Park	0.04	0.44	**4.71**	**1.07**	0.35	0.01	0.19	**1.4732**	0.04	0.44	0.87
Loveday Bay	0.14	0.22	0.29	0.64	0.07	0.01	0.10	0.3763	0.14	0.22	0.22
Tauwitcherie	0.71	**1.08**	0.07	**1.71**	0.09	0.01	0.24	0.39	0.71	**1.08**	0.61
Boggy Creek	**1.73**	0.20	0.07	**1.43**	0.14	0.59	0.16	0.46	**1.73**	0.20	0.67
Point Sturt South	**11.95**	0.11	0.07	**8.57**	0.18	0.01	**44.75**	0.028	**11.95**	0.11	**7.77**
Dog Lake	0.13	0.14	**2.14**	0.86	0.30	0.01	0.28	**1.83**	0.13	0.14	0.60
Boggy Lake	0.20	0.26	**1.64**	**4.86**	0.35	0.01	0.11	0.329	0.20	0.26	0.82
Windmill Site	0.16	0.30	0.07	**1.00**	0.12	0.01	0.23	0.00	0.16	0.30	0.24
Lower Currency	0.06	0.07	**1.36**	0.93	0.15	0.01	0.66	0.72	0.06	0.07	0.41
Lower Finniss	**1.04**	0.12	0.29	**1.14**	0.07	0.01	0.30	0.13	**1.04**	0.12	0.43

Values in red are higher than 1 and are a potential hazard

Table 12 Total elemental concentrations in 0.1 M HCl extracts of sediment samples at 17 sites in March 2013

SITE	Ag (mg/kg)	Al (mg/kg)	As (mg/kg)	Cd (mg/kg)	Co (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Fe (mg/kg)	Mn (mg/kg)	Ni (mg/kg)	Pb (mg/kg)	U (mg/kg)	Zn (mg/kg)
Wally's Landing	0.004	499	1.5	0.12	5	8	9	2923	186	5	14	0.38	29
Point Sturt North	0.017	50	0.29	0.001	0.053	0.94	0.37	360	3	0.61	0.39	0.034	0.37
Milang	0.004	72	0.16	0.001	0.056	0.18	0.46	137	7	0.20	0.54	0.012	0.58
Tolderol	0.030	47	0.26	0.001	0.14	0.74	0.26	209	8	0.32	1.0	0.013	0.44
Poltalloch	0.021	108	1.0	0.001	0.24	0.29	0.37	865	15	0.44	1.0	0.10	0.44
Waltowa	0.004	234	0.35	0.001	0.27	0.76	1.2	778	14	0.74	1.4	0.020	1.2
Meningie	0.004	141	0.17	0.001	0.065	0.22	0.60	355	8	0.19	1.2	0.010	0.51
Campbell Park	0.004	1027	0.88	0.001	0.31	0.29	1.3	5451	13	0.61	2.1	0.062	0.76
Loveday Bay	0.013	63	0.18	0.001	0.053	0.10	0.35	249	3	0.15	0.29	0.0086	0.36
Tauwitcherie	0.004	387	0.78	0.027	0.19	1.0	1.1	937	39	0.68	2.7	0.037	8.8
Boggy Creek	0.004	904	0.58	0.013	0.54	1.0	3.3	3141	37	1.3	5.3	0.11	5.6
Point Sturt South	0.045	30	0.085	0.001	0.031	0.37	0.16	73	2	0.16	0.19	0.0080	0.18
Dog Lake	0.004	567	0.68	0.013	0.71	0.57	2.1	1876	33	2.2	2.5	0.077	2.1
Boggy Lake	0.004	867	1.1	0.0065	0.37	0.87	3.1	1244	20	1.1	2.7	0.53	1.3
Windmill Site	0.059	88	0.64	0.001	0.12	0.32	0.76	688	10	0.28	1.1	0.052	0.56
Lower Currency	0.004	48	0.10	0.001	0.019	0.20	0.19	164	2	0.14	0.52	0.011	0.31
Lower Finniss	0.004	2008	0.84	0.030	0.45	4.7	1.8	1672	32	4.8	9	3.4	3.6

None of the values are above the freshwater sediment trigger value based on the guidelines in Table 14

Table 13 Total elemental concentrations in 0.1 M HCl extracts of sediment samples at 17 sites in November 2013

SITE	Ag (mg/kg)	Al (mg/kg)	As (mg/kg)	Cd (mg/kg)	Co (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Fe (mg/kg)	Mn (mg/kg)	Ni (mg/kg)	Pb (mg/kg)	U (mg/kg)	Zn (mg/kg)
Wally's Landing	0.004	2366	1.3	0.001	3	9	7	11916	153	5	16	0.47	26
Point Sturt North	0.004	59	0.16	0.001	0.13	0.63	0.32	217	5	0.51	0.70	0.038	0.31
Milang	0.004	63	0.19	0.001	0.071	0.085	0.46	129	10	0.19	0.58	0.013	0.44
Tolderol	0.004	44	0.26	0.001	0.13	0.84	0.38	184	8	0.53	0.5	0.013	0.48
Poltalloch	0.004	113	0.51	0.001	0.18	0.21	0.37	561	9	0.50	0.66	0.055	0.38
Waltowa	0.004	761	1.3	0.001	0.67	0.86	3.2	2956	57	1.8	3.3	0.023	3.6
Meningie	0.004	242	0.18	0.001	0.26	0.31	0.79	771	15	0.57	1.0	0.016	1.2
Campbell Park	0.004	870	1.9	0.001	0.51	0.70	2.8	3755	33	1.2	3.8	0.18	1.9
Loveday Bay	0.004	86	0.23	0.001	0.089	0.13	0.53	336	8	0.28	0.51	0.012	0.65
Tauwitcherie	0.004	373	0.32	0.001	0.23	0.58	1.1	900	33	0.51	1.5	0.024	7.9
Boggy Creek	0.004	878	0.52	0.001	0.44	1.0	3.0	3300	71	2	3.7	0.084	4.1
Point Sturt South	0.004	24	0.008	0.001	0.004	0.34	0.13	77	2.5	0.008	0.48	0.001	0.20
Dog Lake	0.004	784	1.3	0.001	0.93	0.81	2.9	3195	71	2.2	2.8	0.051	3.0
Boggy Lake	0.004	2320	1.9	0.001	1.0	1.6	7.7	4236	54	3.2	8	1.2	3.4
Windmill Site	0.004	28	0.067	0.001	0.14	0.008	0.17	340	9	0.12	0.37	0.010	0.27
Lower Currency	0.004	123	0.32	0.001	0.066	0.18	0.58	546	8	0.23	0.56	0.026	0.90
Lower Finniss	0.004	2261	2.1	0.001	1.3	15	7.5	1724	42	8	10	2.3	10

None of the values are above the trigger value based on the guidelines in Table 14

Table 14 ISQG-Low and high whole sediment guideline values.

	Al (mg/kg)	As (mg/kg)	Cd (mg/kg)	Co (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Fe (mg/kg)	Mn (mg/kg)	Ni (mg/kg)	Pb (mg/kg)	Sn (mg/kg)	V (mg/kg)	Zn (mg/kg)
Trigger value (TV) [a]	NV	20	1.5	NV	80	65	NV	NV	21	50	5	NV	200
ISQG-High [b]	NV	70	10	NV	370	270	NV	NV	52	220	70	NV	410

[a] **Trigger value (TV)** = ANZECC/ARMCANZ (2000) SQG-low trigger value (TV) for 95% species protection. **Blue** when >SQG trigger value.
[b] **ISQG-High** = ANZECC/ARMCANZ (2000) SQG-high trigger value (TV) for 95% species protection. Red when > SQG-high value.
NV = no guideline value.

3.4 Sediment Quality Triad (SQT)

The surface sediments at 0-10cm at most sites in the Lower Lakes were in the pH 6-7 range, although results at some sites for pH from deeper sections of the sediment profiles were <4 pH units (Baker 2013). It is possible that diffusion and advection of the circumneutral (pH 7-8) lake water is neutralising the top layer of sediment. As the top layer of sediment (top 2 cm) is where the majority of taxa can be found, this neutralising effect could be important in maintaining the invertebrate community of the Lakes post the drought related acidification. In addition, associated processes (e.g reducing bacteria) and actions of bioturbation could play a part in forming/maintaining reductive conditions. The majority of sites were considered to be either unimpacted or possibly unimpacted. However, two locations (Boggy Creek and Boggy Lake), could be considered possibly impacted, suggesting recovery in these areas has still not occurred three years after the drought ended (Tables 15 and 16).

Table 15 Sediment Quality Triad (SQT) results for March 2013

SITE CODE	SITE NAME	CHEMISTRY	TOXICITY	BIOTA	TOTAL	STANDARDISED
LF01	Wallys Landing	5	0	1	6	2.25
LF02	PtSturt North	4	1	0	5	1.33
LF03	Milang	4	1	0	5	1.33
LF04	Tolderol	3	1	0	4	1.08
LF06	Poltalloch	2	1	0	3	0.83
LF07	Waltowa	1	0	0	1	0.25
LF08	Meningie	0	2	0	2	0.67
LF10	Campbell Park	1	0	1	2	1.25
LF12	Loveday Bay	4	2	0	6	1.67
LF13	Tauwitcherie	1	1	1	3	1.58
LF15	Boggy Creek	6	1	2	9	3.83
LF17	PtSturt South	3	1	0	4	1.08
LF19	Dog Lake	6	1	0	7	1.83
LF20	Boggy Lake	5	1	0	6	1.58
LF21	Windmill Site	1	2	0	3	0.92
LF23	Lower Currency	3	0	0	3	0.75
LF24	Lower Finniss	2	6	1	9	3.50

Values in red are considered possibly impacted sites.

Table 16 Sediment Quality Triad (SQT) results for November 2013

SITE CODE	SITE NAME	CHEMISTRY	TOXICITY	BIOTA	TOTAL	STANDARDISED
LF01	Wallys Landing	5	1	2	8	3.25
LF02	PtSturt North	1	0	1	2	1.2
LF03	Milang	2	0	1	3	1.4
LF04	Tolderol	2	0	0	2	0.4
LF06	Poltalloch	2	0	0	2	0.4
LF07	Waltowa	3	2	1	6	2.1
LF08	Meningie	1	1	2	4	2.45
LF10	Campbell Park	3	3	2	8	3.35
LF12	Loveday Bay	1	1	1	3	1.45
LF13	Tauwitcherie	1	0	0	1	0.2
LF15	Boggy Creek	5	0	2	7	3
LF17	PtSturt South	3	3	0	6	1.35
LF19	Dog Lake	4	2	1	7	2.3
LF20	Boggy Lake	**6**	**5**	**2**	**13**	**4.45**
LF21	Windmill Site	2	0	1	3	1.4
LF23	Lower Currency	2	2	1	5	1.9
LF24	Lower Finniss	6	0	1	7	2.2

Values in red are considered possibly impacted sites.

4 Conclusions

The SQT approach used, enabled us to compare the sites and categorise them according to the severity of the impact. Most sites, particularly in March 2013, had quite low scores suggesting that these sites were likely unimpacted from previous drought induced acidification. However, two sites may be of concern; Boggy Creek on Hindmarsh Island and Boggy Lake on the north-western side of Lake Alexandrina, which received the highest score in March and November 2013 (scores: 3.83 and 4.45), respectively. Differences were noted between the two sampling periods in winter and summer, suggesting multiple monitoring events during a year is advisable to assess seasonal effects. We found the SQT approach to be a useful tool to assess the impact of ASS in a lake environment, despite some of the effects from acidification being only minor in surface soils. This monitoring study confirms the recovery of the Lower Lakes three years after inundation. If this SQT assessment was undertaken during or straight after the water had returned, there could have been greater effects observed, relating to the impact of ASS in the lakes. The impact of contaminant fluxes from acidified deeper layers has not been assessed but is more likely to be significant if lake water levels fall. This supports the recommendation of further assessment and monitoring.

The major advantage of the SQT approach is that it integrates the data generated from the three separate measurements and, thereby, facilitates the separation of natural variability in biotic characteristics from variability due to the toxic effects of contaminants. For example, variability in benthic community composition may be due to the presence of contaminants in sediments or it may be related to differences in other aspects of habitat quality (i.e., grain size, depth, etc.). The triad approach provides a basis for distinguishing between these effects; however, it cannot be used to establish cause and effect relationships. Other advantages of this approach are that it may be used for any measured contaminant, it may incorporate information on both acute and chronic effects, and it does not require information on specific processes governing interactions between organisms and toxic contaminants. Integrating the three data types provides a weight of evidence approach to guidelines development.

The major limitations of the SQTA are as follows (Chapman 1990): statistical criteria have not been developed for use with the triad; rigorous criteria for determining single indices for each of the separate measurements have not been developed; a large database is required; it is generally used to develop guidelines for single chemicals, and as such the results can be strongly influenced by the presence of unmeasured toxic contaminants that may or may not co-vary with the measured chemicals; sample collection, analysis, and interpretation is labour intensive and costly; and, the choice of a reference site is often made without adequate information on how degraded the site may be. In addition, the SQTA does not explicitly consider the bioavailability of sediment-associated contaminants. In terms of ERA, the SQT provides a Weight-of-Evidence (WOE) framework comprising a determination related to possible ecological impacts based on multiple lines of evidence (LOE). Over the years, a variety of LOE replacing or additional to the alterations to resident communities LOE have been proposed and applied such as additional LOE incorporating biomagnification, biomarkers, in-situ or sediment contact assays and colonisation experiments. The choice of LOE depends on the stressors of potential concern (which are not limited to chemical contaminants), the receptors of potential concern, the exposure routes, and the type(s) of effects anticipated.

ERA of contaminated sediments need to focus on the 'big picture', considering all relevant stressors and their inter-relationships, and answering the all-important 'So what?' question (Chapman and Guerra, 2005). In the present study, we found the SQT approach to be a useful tool to assess the impact of ASS in a lake environment, despite some of the current effects from acidification being only minor. A comparison between pre-drought and post-drought benthic communities was not possible due to limited data on the pre-drought baseline. If this SQT assessment had been undertaken during or straight after the drought ended, there may have been more obvious impacts in the lakes due to ASS.

5 Recommendations

- The current investigation was carried out using surface sediments from the sediment-water interface (up to 10 cm sediment depth) of the lakes. An upward flux of contaminants may pose a moderate to high level risk to the biota inhabiting ASS impacted sites. This should be investigated at selected sites.
- Due to heterogeneity of the soils and seasonal/flow changes, further ecotoxicological monitoring is recommended to assess the spatial and temporal variation in the toxicity at selected Lower Lakes sites.
- The development of rapid monitoring tools and modelling approaches should be considered. They would utilise chemical, physical and microbial parameters to enable assessment of sediment health and impacts of stress-induced changes.
- In the present study, a mixture of metals including Al, Cu, Zn, As, Mn and Co were above ANZECC guideline values in the pore water collected from the sediments at some sites. As the potential for multiple chemical exposure increases, the question raised is whether the toxicity of mixtures of chemicals is simply additive or whether there is potentiation of toxicity. The general consensus has been that chemicals interact by concentration addition, however past studies have demonstrated that concentration addition of the components of a mixture does not always reflect the overall interaction of a mixture. The risk assessment procedures should account for mixtures of contaminants present in a given system.
- Pore water Al had the highest hazard quotient. Thus, it becomes obvious that Al is a significant hazard associated with ASS. Unfortunately, there is a notable shortage of literature on the biological response of the aquatic biota to Al released from ASS. The locations and soil characteristics of ASS are well defined throughout the literature. Similarly, Al is recognised as a highly toxic element when bioavailable. However, Al forms a range of chemical species, little is known about speciation, bioavailability and toxicity when it comes to a system dominated by ASS. The current Al guideline is applicable at pH <6.6. The ANZECC/ARMCANZ water quality guidelines require review for aluminium, particularly in relation to deriving guideline value(s) for aluminium toxicity in lower pH water. The sediment guidelines for aluminium should also be reviewed.

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