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Remediation of water quality and macroinvertebrate communities in an urban artificial lake affected by acid sulphate soils

Mark A. Lund, Clinton D. McCullough and David Galeotti

Introduction

Worldwide human-induced acidification and the commonly associated metal–metalloid toxicity is one of the top issues affecting inland waterbodies (e.g., DENT & PONS 1995). Three key processes are responsible for the acidification: acid rain, acid mine drainage (AMD), and exposure of acid sulphate soils. Reduced forms of sulphate are oxidised to produce sulphuric acid in all processes, and the resulting low pH mobilises metals and metalloids. The extent and significance of acid sulphate soils are only now starting to be recognised in Australia (SAMMUT & LINES-KELLY 2000, RUSSELL & HELMKE 2002), and particularly in Western Australia (WA; SOMMER & HORWITZ 2001, APPLEYARD et al. 2004, 2006). In Perth (state capital of WA), the appearance of a number of localised acid sulphate soil issues in the City of Stirling, around canal developments along the Peel Harvey estuary and in Lake Jandabup have largely caught government agencies by surprise. Although acid sulphate soil issues associated with developments around estuaries are relatively common in Australia (SAMMUT et al. 1995, 1996), the acidification of a large conservation status lake north of Perth following a drying event highlighted risks associated with drought-induced acidification in the State. Perth has experienced a trend of declining rainfall since the 1970s, coupled with increasing demand for scheme water that cannot be met through dams, and has seen substantive use of groundwater resources (domestically for watering gardens, drinking via scheme water, and agriculture). Declining groundwater tables expose potential acid sulphate soils to oxidation, a problem compounded by urban development, which is also dewatering areas and excavating potential acid sulphate soils in areas where oxidation can occur. These acid sulphate soil issues have been initially presented as largely contamination of groundwater, which is then expressed in groundwater-dependant wetlands, in drains, or in groundwater wells used to water gardens. This latter use is of particular concern because in many areas the acidity is generated from arsenopyrites, exposing residents to arsenic through consumption of home-grown food crops (HINWOOD et al. 2006).

A number of techniques have been developed to remediate anthropogenic acidification, AMD in particular (see MCCULLOUGH 2007). Many AMD treatment technologies are now starting to be transferred to acid sulphate soil issues, including water management approaches to prevent further oxidation, neutralising technologies (typically requiring active management), and "passive remediation" technologies that require limited maintenance. Nevertheless, the nature of these technologies has generally seen them applied in agricultural rather than urban settings. Perth is situated on the Swan Coastal Plain (SCP), which consists of a series of sand dune systems of varying ages running parallel to the coastline. This situation poses some unusual problems for acidification problems because frequently groundwater rather than surface water is contaminated, and groundwater is not suited to most of the standard acid sulphate soil remediation techniques.

We reported the impact of discharges from a low-cost pilot treatment system for treating acid sulphate soil caused acidification in an urban area, using a range of technologies commonly used to treat AMD. The impact of treated waters on water quality and macroinvertebrate communities were used to assess the success of the system.

Key words: acid sulphate soils, acidity, arsenic, bioremediation, Western Australia

Study site

The Spoonbill-Shearwater Reserve is located on the SCP in a residential area within the City of Stirling in Perth (31°52'S; 115°48'E; Fig. 1). Two small similar lakes with large central islands in the reserve (~1.4 ha, 2–4 m deep) have been excavated to intercept water table. Groundwater flows in a southwesterly direction from control to treatment lake. Peat excavated during the development of the suburb was placed upstream of the groundwater flow through these lakes. In the late 1990s, low pH (< 3) was recorded in nearby groundwater bores and in these reserve lakes (APPLEYARD et al. 2004).

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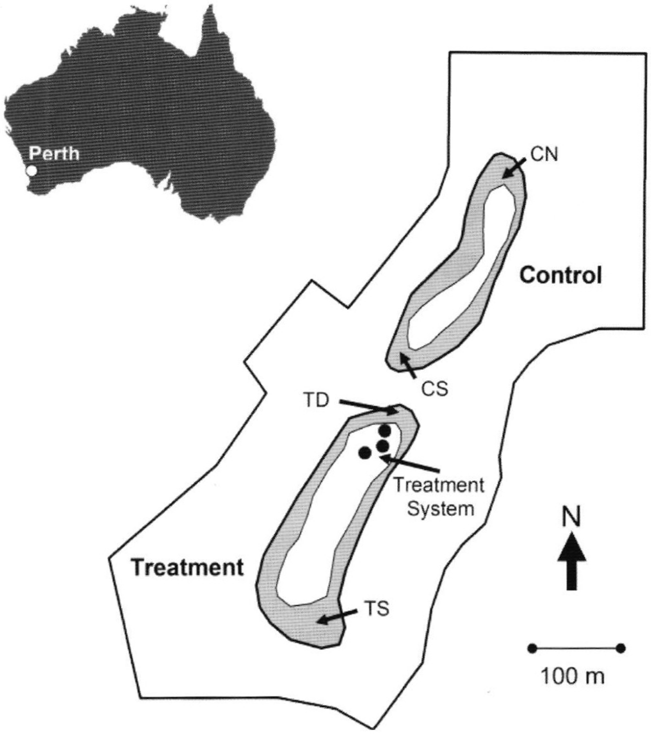

Fig. 1. Spoonbill-Shearwater Lakes (highlighted in grey), showing treatment and control lakes, location of sample sites (CN = Control North; CS = Control South; TD = Treatment Discharge; TS = Treatment South) and treatment system within the public park.

The lake's primary function was aesthetics and wildlife habitat, but they presented an unattractive iron-ochre-scalded shoreline, dying fringe vegetation, and loss of waterfowl.

Methods

The north lake was used as a control while treated water was discharged into the northern end of the southern lake. The treatment system was designed to treat 10 L min^{-1} of acidic water drawn from the control lake. Acidic water was then neutralised with NaOH to pH 6–7, followed by settling and removal of iron and aluminium hydroxide sludge. Clarified water is then gravity fed into the bottom of a 23 000-L bioreactor tank filled with 10 t of potatoes (to rapidly lower oxidation reduction potential [ORP]) covered by a recalcitrant wood mulch (coarsely chopped eucalypt tree) and then onto a second similar bioreactor containing mulch only. The low ORP facilitates sulphate-reducing bacterial processes within the wood mulch. Hydraulic residence times of the tanks are estimated to be 48 hr. Water leaving the final tank is then cascaded down a concrete riffle for re-aeration prior to discharge. To remediate littoral habitat and low lake carbon, the discharge area in the lake was further treated with approximately 1 t of potatoes, with the previously scalded banks covered in mulch to a depth of ~0.1 m. The discharge area was planted with rushes (*Schoenoplectus validus* and *Baumea articulata*) in September 2006 and March 2007 to create an aerobic polishing wetland. The system began operation in November 2006.

Four sites were sampled: control lake north (CN), control lake south (CS), treatment discharge area (TD), and treatment lake south (TS). Three replicate macroinvertebrate samples were collected and pooled at each site (approximately 40 m apart) in October 2006 and March 2007. Each sample consisted of a 1-m^2 area sweep of the littoral margin using a 250-µm mesh dip net. Samples were preserved with 100 % histo-ethanol prior to sorting, counting, and identification to species level, when possible. Conductivity (EC), pH, dissolved oxygen (DO), ORP, turbidity, and chlorophyll *a* were measured at the surface on each occasion at each site using a Hydrolab Datasonde 4a multiprobe. A surface water sample was also collected. One aliquot was frozen for later determination of total phosphorus (TP) and total Kjeldahl nitrogen (TKN; per APHA 1998); another aliquot was then filtered (0.5 µm GF Pall Metrigard) and acidified with HCl. Concentrations of select metals-metalloids were then measured in the laboratory with Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES).

All the physico-chemical, metal, and nutrient data were combined (parameters under detection limits at all times were excluded) and analysed in a principal component analysis PCA with Primer 6 software (PRIMER-E LTD 2006). A nonmetric multidimensional scaling (nMDS) ordination was made of untransformed macroinvertebrate data. The Bio-Env routine was then used to examine relationships with parameters that best explained macroinvertebrate community structure.

Results and discussion

We did not report efficacy of the treatment system, other than it discharged treated waters of circum-neutral water, low in metals and anions to the lake system.

The pH in the lakes was < 3.2 at all sites except for TD in March 2007 when the circum neutral discharge increased it to > 5. The pH dropped slightly from control to treatment on both occasions, probably due to ongoing ferrolysis through the wetland (Fig. 2). The EC increased slightly from 1.27 to 1.64 mS cm^{-1} in October 2006 from control to treatment lakes, showing a similar pattern to March 2007, with slightly higher EC reaching 1.81 mS cm^{-1}. The exception was TD (Mar 2007) where low conductivity discharge diluted incoming water to 1.26 mS cm^{-1}. The DO increased from control to treatment on both occasions except at TD (Mar 2007) where discharge from the treatment system did not appear to be as strongly aerated as expected, resulting in a noticeable drop in DO. However, DO remained above 57 % satu-

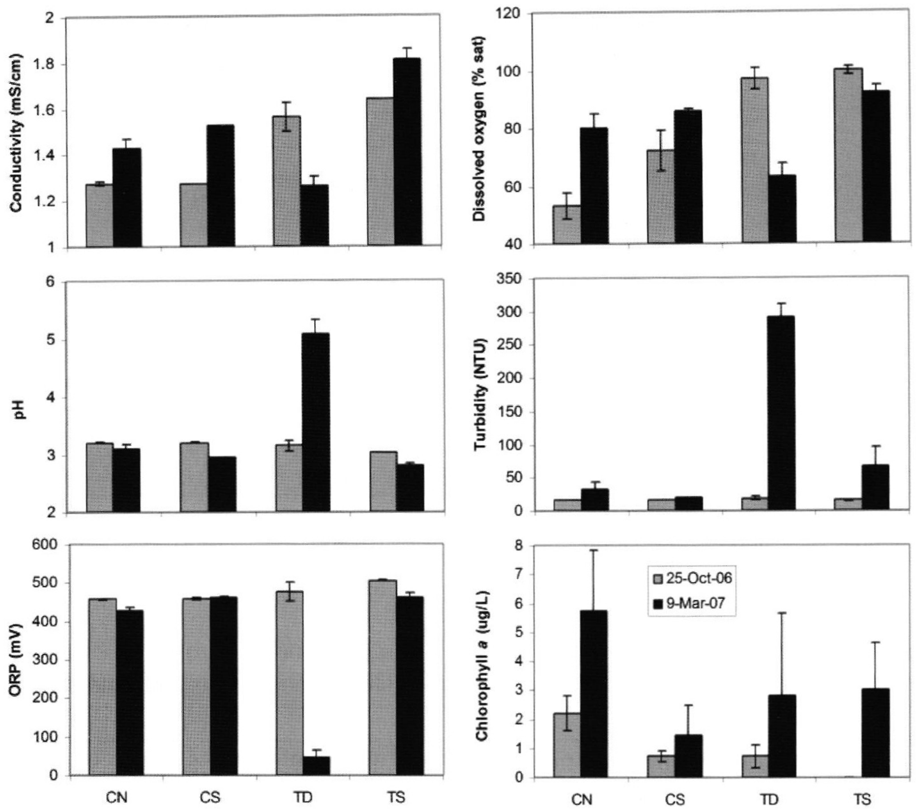

Fig. 2. Mean (±S.E.) for physico-chemical parameters recorded in the Stirling Lakes on 10/06 and 3/07. CN = Control North; CS = Control South; TD = Treatment Discharge; TS = Treatment South.

rated (4.7 mg L^{-1}) on both occasions. The ORP was >450 mV except in TD (Mar 2007) where it dropped to 46 mV, presumably due to low re-aeration of discharge from the treatment system. Chlorophyll *a* was slightly higher at CN compared to the other sites (2.2 and 5.8 μg L^{-1} for October 2006 and March 2007, respectively), which were < 1.5 μg L^{-1}, with the exception of treatment sites on March 2007, which ranged from 2.8 to 3 μg L^{-1}. Higher turbidity was generally associated with higher chlorophyll *a*, although the 290 NTU at TD (Mar 2007) was possibly due to some iron mono-sulphides exiting the treatment system. High concentrations of TP (13–33 μg L^{-1}) and TKN (3500–3900 μg L^{-1}) should have been able to support higher chlorophyll *a* concentrations than experienced. The higher algal biomass at CN suggests that when the groundwater entered the lake it was still anoxic, allowing particularly P to be bioavailable. However P binding to rapidly oxidising iron and complexation with dissolved aluminium is likely to have quickly reduced that availability. The mulch or discharge seems to contain P and reached concentrations of 64 μg L^{-1} at TD on March 2007. Decomposition of the fresh mulch also seemed to have reduced TKN concentrations to <1400 μg L^{-1}, a phenomena also observed by Lund et al. (2006) when using mulch in a similar situation.

Over both sampling occasions and sites, concentrations of As, Hg, Pb (except TD on Mar 2007 at 0.23 mg L^{-1}) were < 0.1 mg L^{-1}; Se < 0.2 mg L^{-1}; Ni, B, and Cu were < 0.05 mg L^{-1}; and Cd, Co, and Cr were < 0.01 mg L^{-1} (Table 1). Conservative ions such as Ca, Mg, Na, and K increased slightly in concentration from control to treatment lake on both occasions (Table 1) but had higher concentrations on March 2007, probably due to evapo-concentration emphasized by the low water level (dropped from > 1 m in Oct 2006 to < 0.5 m in Mar 2007). The diluting effect of the discharge seen in EC was seen at TD (Mar 2007) for these ions. Interestingly, Mn shows almost the opposite response with no increase between sampling times and a decrease from control to treatment (Table 1). The presence of mulch and lower DO and ORP on March 2007 did not seem to have any influence on S concentrations, which would indicate that ORP in the sediments was probably conducive to sulphate reduction. Fe showed a similar pattern to the conservative ions, although at TD (Mar 2007) it increased from < 50 to 89 mg L^{-1}. Because the concentration of Fe in the discharge was < 1 mg L^{-1}, this should have diluted concentrations at TD; however, the increase was probably from the mulch creating an environment where previous accumulations of oxidised Fe were being reduced and

Table 1. Metal and nutrient concentrations recorded in the Stirling Lakes on 10/06 and 3/07. CN = Control North; CS = Control South; TD = Treatment Discharge; TS = Treatment South; not shown. As & Hg <0.1 mg L^{-1}; Cd & Cr <0.01 mg L^{-1}; Cu & Ni <0.05 mg L^{-1}; Se <0.2 mg L^{-1}.

		CN		CS		TD		TS	
Parameter	Units	10/06	3/07	10/06	3/07	10/06	3/07	10/06	3/07
Al	mg L^{-1}	5.6	3.3	5.8	3.1	9.4	9.7	9.0	1.1
Ca	mg L^{-1}	64	75	65	74	73	67	73	73
Fe	mg L^{-1}	30	47	32	32	38	89	37	40
K	mg L^{-1}	6.3	8.5	6.3	8.5	9.3	6.5	8.9	9.5
Mg	mg L^{-1}	16	20	17	21	19	17	19	20
Mn	mg L^{-1}	0.14	0.12	0.14	0.13	0.1	0.08	0.09	0.06
Na	mg L^{-1}	67	86	68	87	81	78	79	106
Pb	mg L^{-1}	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.23
S	mg L^{-1}	157	180	155	207	201	207	200	215
Zn	mg L^{-1}	0.18	0.08	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
TP	µg L^{-1}	22	31	29	28	33	64	13	24
TKN	µg L^{-1}	3900	3600	3800	3600	1400	1300	1400	3500

Table 2. Abundance of macroinvertebrate taxa (per 3 m^2) recorded in the Stirling Lakes on 10/06 and 3/07. CN = Control North; CS = Control South; TD = Treatment Discharge; TS = Treatment South; space indicates no animals collected.

			CN		CS		TD		TS	
Family	Genus		25/10/06	9/3/07	25/10/06	9/3/07	25/10/06	9/3/07	25/10/06	9/3/07
Chironomidae		L	1,360	5	20	3	3,290	31	113	
		P	244		31	1	120	3	118	1
Ceratopogonidae		L	80		10	3	30		316	8
Dolichopodidae		L			1					
Dytiscidae	*Necterosoma* sp.	L	2	1	5		120		39	
		A	10	6	1	1		8	5	5
	Megaporus sp.	A	1							
Hydrophilidae	*Berosus* sp.	L			1	1	10		19	3
Corixidae	*Sigara* sp.		1	1						
Total abundance			1,698	13	69	9	3,570	42	610	17
Taxa richness			7	4	7	5	5	3	6	4

released, and before reaching TS this additional iron was being reoxidised and settling out (Table 1). Aluminium concentrations were <1 mg L^{-1} in the discharge but reached 9.7 mg L^{-1} at TD (Mar 2007); the source of the additional Al is possibly groundwater (Table 1). The construction of the aerobic treatment wetland *in situ* seemed to have caused a unforeseen consequence in reducing water quality while these materials were reduced; however, as the supply was likely limited, this should disappear with time.

A PCA of the combined physico-chemical parameters, metals, and nutrients shows that TD was similar to other sites on October 2006, while on March 2007 it was distinct from other sites. The vectors show the contribution of each variable to the eigenvectors; the length of the vector reflects the importance of that variable to the eigenvectors (longer = more important). Chlorophyll *a* is the main factor separating the 2 sampling occasions, while TD on March 2007 is separated by its high turbidity (Fig. 3).

Nine macroinvertebrate taxa (6 families) were recorded from the lakes, although identification to species level of chironomids and ceratopogonids might have improved this slightly (Table 2). Linking of adult, larval, and pupae, however, could have reduced apparent taxa richness. September-October is the time of peak diversity on the Swan Coastal Plain (Davis et al. 1993); this was reflected in greater abundance and diversity on October 2006 than March 2007. The addition of mulch seems to have increased abundance of chironomids rather than the effects of the discharge because abundances were higher on both sampling occasions. The seasonal differences are reflected in the ordination (Fig. 3); location in the lake seems more important than whether it was treated or not

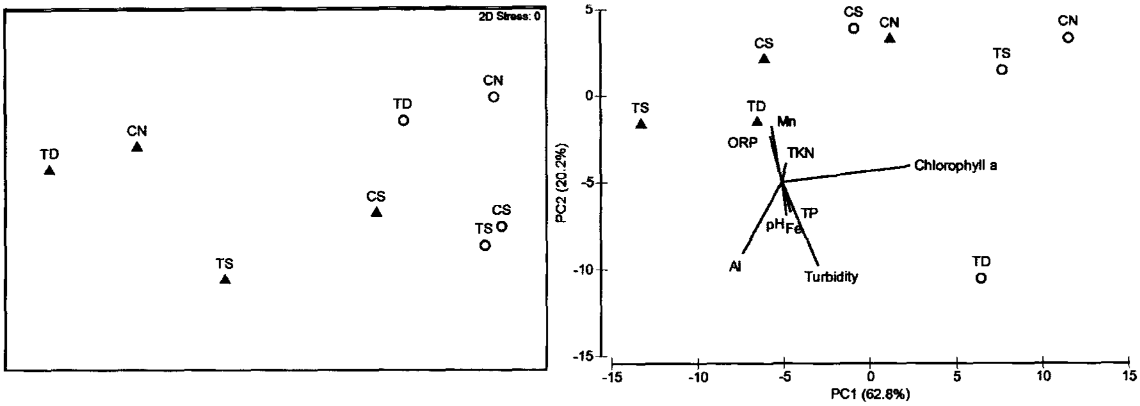

Fig. 3. nMDS ordination of macroinvertebrate communities (left) and a PCA of physico-chemical, metal and nutrient data (variable correlates to the PCA are shown as vectors). CN = Control North; CS = Control South; TD = Treatment Discharge; TS = Treatment South: ▲ = 10/06; O = 3/07.

because TD and CN, and TS and CS group together with in each season. This suggests that the community is responding more to increased chlorophyll *a* in CN and increased food resources in the mulched area (TD) rather than water quality. The low pH and generally low food resources seem to have resulted in depauperate macroinvertebrate communities dominated by mobile or tolerant taxa. Using Bio-Env water quality parameters did not seem to explain the macroinvertebrate community ordination, although DO correlated highest with macroinvertebrate community structure ($\rho = 0.2$).

In conclusion, in the short time of operation, the treatment discharge had not improved water quality (other than through dilution) or increased biodiversity. The construction of an aerobic wetland within the lake had created an environment in which accumulated oxidized iron was reduced and mobilised back into the lake. This source is likely limited, and the benefits of the mulch for enhancing biodiversity and polishing should become more important with time. Better aeration of the discharge water would be beneficial, and a redesign of the concrete riffle would overcome this problem. The treatment system seems to offer potential for improving water quality in the long term, although in the short term it has proven problematic.

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