MICROCOSM TESTING OF MUNICIPAL SEWAGE AND GREEN WASTE FOR FULL-SCALE REMEDIATION OF AN ACID COAL PIT LAKE, IN SEMI-ARID TROPICAL AUSTRALIA[1]

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Abstract: Pit lakes (abandoned flooded mine pits) represent a potentially valuable water resource to mining companies, the environment and regional communities across arid inland Australia. However, the water is often of low pH with high dissolved metal concentrations.

The addition of organic matter to the pit lakes to enhance microbial sulfate reduction is potentially a cost effective and sustainable remediation strategy for these acid waters. However, the cost and availability of sufficient quantities of suitable organic substrates is typically limiting in these remote regions. Nevertheless, small quantities of sewage and green waste (organic garden waste) are often available in these areas from the regional towns which support the mines. This paper reports on preliminary microcosm laboratory experiments in preparation for the treatment of an acid (pH 2.2) coal mine pit lake in semi-arid tropical, inland north Queensland, Australia with municipal treated sewage and green waste.

A laboratory experiment using microcosms (acrylic tubes) containing acid pit lake water and sediment were treated as follows; controls (untreated), sewage, green waste and sewage and green waste. The pH increased to a maximum of 5.5 in 145 days in the green waste and sewage treatment, with notable decreases of iron, aluminium and toxic heavy metals. Our results indicated that the green waste was a key component in alkalinity production and heavy metal removal.

Additional Key Words: acid mine drainage, pit lakes, sulfate reduction, wetland, organic matter.

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Introduction

Microbial sulfate ($SO_4^{-2)}$ reduction is considered to be an efficient and effective remediation for the treatment of acid mine drainage (AMD), through alkalinity production and precipitation of metals as sulfides through reaction with sulfide cations (Equation 1) and as carbonates through increased pH (Equation 2) (Benner et al., 2002; Dillon et al., 1997; Küsel and Dorsch, 2000; Praharaj and Fortin, 2004). In-lake neutralisation via SO_4^{-2} reduction is expected to play a key-role in the remediation of acidic mining pit lakes (Kleeberg, 1998). However, since Tuttle et al. (1969) first suggested the use of SO_4^{2-} reducing bacteria (SRB) in the treatment of AMD, treatment has largely focused on *ex-situ* treatment in bioreactors. What *in-situ* treatment relevant experiments have occurred have generally been performed at microcosm (e.g., carboy vessels; (Castro et al., 1999; Frömmichen et al., 2004; Fyson et al., 1998)) or at best macrocosm (e.g., limnocorrals) scales (Martin et al., 2003). It is largely recently that attention has switched to *in-situ* remediation systems at actual field treatment scales (Gibert *et al.*, 2002).

Equation 1 $\text{Metal}^{+2} + SO_4^{-2} + 2C_{(\text{organic})} \rightarrow \text{MetalS} + 2CO_2$

Equation 2 $\text{Metal}^{+2} + CO_3^{-2} \rightarrow \text{MetalCO}_3$

A review by Gibert et al. (2002) found that the nature of the organic matter was a prime determinator of the efficacy of the passive treatment system. For example, the availability of carbon from plant matter is dependent upon decomposition, which is extremely limited in acidic and anoxic conditions (Harris and Ragusa, 2001). To this end, Waybrant et al. (1998) found mixtures containing multiple sources of organic matter demonstrated higher SO_4^{2-} reduction rates than those of single sources. Castro and Moore (1997) observed that, “Therefore, for economic feasibility the added organic matter must be cheap and locally available.” Hard et al. (2003) echoed this view noting that, “For a microbial process to be economically feasible, the carbon (C) and energy source should be cheap, widely available and highly effective.” However, most experiments into the utility of SO_4^{-2} reduction processes for ameliorating AMD have instead focused upon highly labile but expensive C substrates such as ethanol (Kolmert and Johnson, 2001; Martin et al., 2003; McNee et al., 2003), sugar (Frömmichen et al., 2003; Frömmichen et al., 2004; Geller et al., 2003; Pöhler et al., 2002; Wendt-Potthoff et al., 2002), cow manure (Drury, 1999; Drury, 2000), etc. For many remote mining locations, these materials are unviable for practical and economic reasons. The only bulk C sources likely to be available in these areas will be municipal sewage, and green waste (including a broad range of plant material collected by local government from domestic and municipal lawns and gardens).

Using readily-available and economically-viable sources of organic materials, Waybrant et al. (1998) found all of the eight organic matter types they tested reduced SO_4^{-2}, with sewage sludge the fastest to achieve high levels of sulfate reduction. The mixture of sewage sludge and green waste (leaf mulch, woodchips and sawdust) reduced 4,500 mg L^{-1} of SO_4^{-2} to <25 mg L^{-1} in only 35 days. Again using locally available bulk materials, a laboratory experiment by Harris and Ragusa (2000) also found a mixture of sewage sludge and plant material (fresh rye grass) was effective in initiating amelioration of acidity and metal concentrations of acid mine waters through sulfate reduction. The pH levels increased significantly from 2.3 to >3 within 30 days of SRB activation and simultaneously decreased divalent metal concentrations. This mixture proved more effective in ameliorating pH and metal concentrations than either sewage sludge (little response) or plant material (nil response). Gusek (2002) even suggested injecting sewage sludge into mine shafts and adits to remove and prevent production of acidity from these sources.

More specific data for our particular study area was reported in a 1993 study at the Collinsville Coal Project Mine in North Queensland, Australia (Fallon, 1994). Although unquantified, this report details findings that the extremely acid waters of Blake A-cut had "notably improved" (i.e. reduced acidity, SO_4^{-2} and metal concentrations) and suggested that one mechanism may be bacterially-mediated SO_4^{-2} reduction occurring with the sewage effluent discharged into this pit being used as a source of carbon. Rates of SRB activity may also be greater in a tropical area such as this mining location, than in the other published research, much of it originating from temperate Europe and North America.

Consequently, there is a large body of published data to suggest that bacterially-mediated sulfate reduction processes can ameliorate acid mine drainage waters. Published literature from laboratory and mesocosm experiments also indicates that sewage is suitable for use as an organic substrate stimulating SO_4^{-2} reduction. However, there are few published reports of field scale attempts of AMD lake remediation. Furthermore, very little bioremediation work has been carried out on the mining pit lakes of Australia using these methods, especially in semi-arid tropical Australia where a significant number of acid pit lakes historically occur and are still being developed of increasing sizes (Harries, 1997). As an arid continent with increasing pressure on water resources, there is growing demand for new sources of water to meet a variety of end uses (Doupé and Lymbery, 2005).

The end use of many of these pit lakes may also often be only for slightly remediated water quality of lower salinity for mining operation dust suppression or similar industrial use. Consequently, financially viable treatment to this lower standard may still be very achievable in even remote mining areas. Consequently, this research project intended to test the potential of this biotechnology to remediate a representative extremely low pH, high-acidity and high solute concentration mining lake in a typical remote Australian mining community.

Methods

Study area

Collinsville is a typical small inland Australian mining town located approximately 70 km from the coast of North Queensland, Australia (Fig. 1). Together with the mining town of Scottville, Collinsville is located within a kilometre of the active mining Collinsville Coal Project coal mining lease (Fig. 2). Collinsville was underground (bord and pillar) mined for its coal as early as 1919 at the State Mine working the Bowen Seam, and began open-cast mining in the mid 1950s. Mining operations are expected to continue until 2013. The combined population of Collinsville and nearby Scottville is currently around 2,021 people (1996 census) and is undergoing a decline as mining activity declines.

Collinsville has a semi-arid tropical climate with a rainfall regime that falls into a transition between sub-humid and semi-arid. The climate is dominated by a moderately low, highly episodic and unreliable summer rainfall (707.9 mm/annum) and a very high evaporation rate (1,860 mm/annum) (Commonwealth of Australia Bureau of Meteorology, 09/02/2005; Davies and Willcocks, 1992) (Fig. 2).

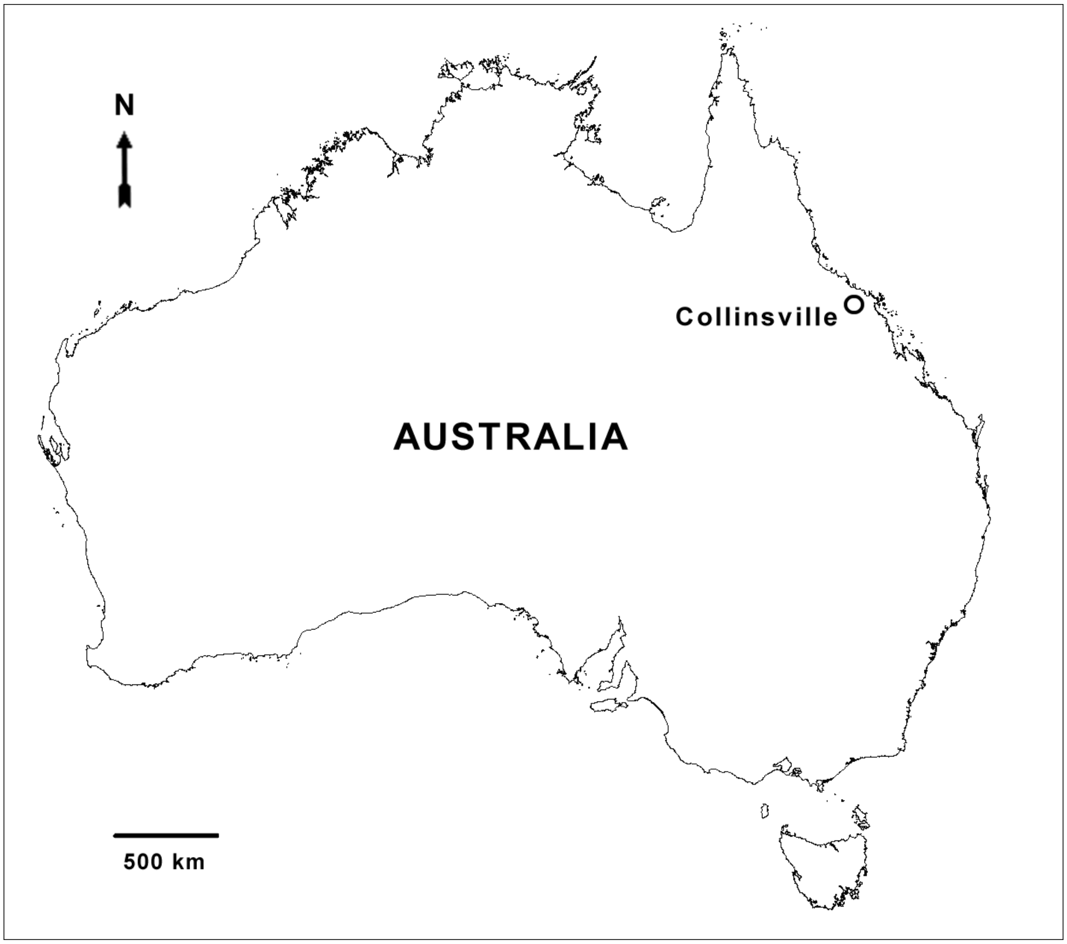

Figure 1 Location of the study site in Collinsville, North Queensland, Australia.

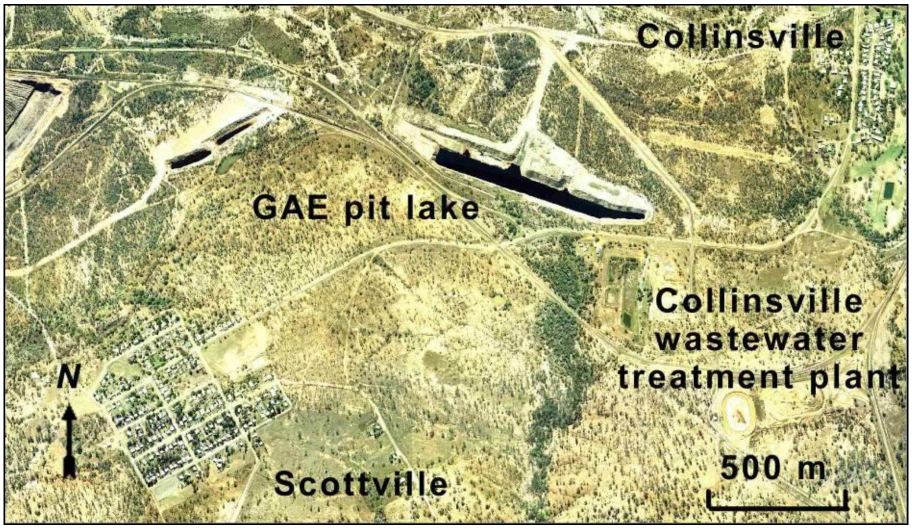

Figure 2. GAE pit lake; the source of water and sediment for this and another laboratory microcosm study, and the site of a current pit lake bioremediation field trial.

The geology of the region is of highly weathered hard rocks with soils of very low organic carbon content. Surrounding vegetation is predominantly *Eucalyptus* and *Acacia* spp. dominated open woodland with an annual grass understorey. There are some 20 pit lakes in the Collinsville Coal Project lease, all of which are acidic with high concentrations of dissolved solutes (Fig. 3). All of these lakes are of extremely low pH (*ca*. pH 2) and contain high concentrations of dissolved solutes (electrical conductivity = 9–19 mS cm^{-1}). One of these acid pit lakes, "Garrick East" (GAE), is the focus of this study due to its proximity to the Collinsville waste water treatment plant and green waste dump (<500 m away). This lake is of maximum depth 12 m, surface area 1.3 ha and volume 120 ML (Fig. 2, Fig. 4).

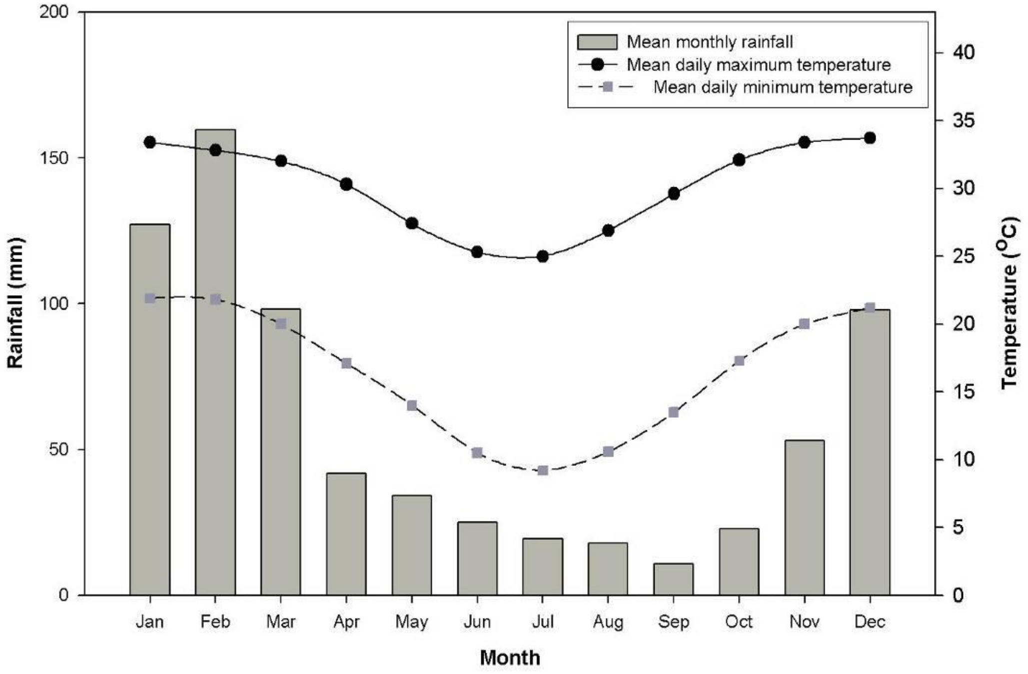

Figure 3. Mean temperature and rainfall climate of Collinsville (Commonwealth of Australia Bureau of Meteorology, 09/02/2005).

Experimental design

Microcosm experiments were designed to mimic the hypolimnetic water column and sediment regions of a typical strongly stratified Collinsville pit lake. Twelve clean 100 mm diameter and 600 mm long (4.5 L) acrylic tubes were set up in an uninsulated laboratory on the mine site as microcosms containing 140 mm of sediment and 440 mm of GAE pit lake water. Cores were pushed into the littoral sediment of GAE at 50 cm water depth and sealed with rubber bungs at the top and bottom. The height of sediment in the bottom of the core was then adjusted by sliding it downwards out the bottom of the core to a final depth of 140 mm across all cores. Three replicate cores were then allocated to each of the following; control (untreated), green waste only, sewage, or green waste and sewage at dosing rates realistic of that able to be achieved in a typical CCP pit lake using regional sources over a few month's filling duration.

Figure 4. "Garrick East" Collinsville Coal Project mining pit lake. Note very dark orange colour of water due to high dissolved Fe concentrations (pH = *ca*.2, electrical conductivity = *ca*.9 mS cm^{-1}), and efflorescence of elemental sulfur, gypsum and epsomite around littoral fringe.

Green waste was sourced from the Collinsville shire green waste dump nearby the Collinsville Coal Project lease. Greenwaste consisted of a wide range of garden clippings from both woody and herbaceous species that had been exposed to Dry season drying climatic conditions for some weeks. Primary-treated sewage sludge was sourced from the Collinsville Municipal Wastewater Treatment Plant. Due to the over-capacity of the plant for the declining town size, the sewage sludge had been exposed to the sun in drying beds for around 12 weeks.

The cores were then filled with GAE water to within 100 mm of their brim, green waste was then added where appropriate, and then sewage in the following scheme (Table 1). Microcosm core water levels were topped-up to with 20 mm of the core's brim and a loose-fitting rubber bung was applied to the top (not airtight) to reduce air infiltration into the core water as would a strongly stratified epilimnion (Fig. 4). The entire suite of microcosm cores were then placed in a 500 mm high opaque black plastic planter tub which was filled with water to evenly distribute ambient temperatures between cores and to reduce the incidence of leakage. The lower water-filled tub was then capped with an identical inverted planter tub to occlude light, as PAR levels are extremely low in the hypolimnion of these lakes (author's unpublished data).

The cores were then sampled for the physico-chemical variables; temperature, pH specific conductance, oxidation-reduction potential and dissolved oxygen (% saturation and mg/L) with a Hydrolab Quanta multiparameter meter.

Table 1. Experimental design for organic dosing of pit lake cores.

Treatment level	Organic dosing mass (g)	Water:green waste:sewage ratio	Number of replicates
Control	0	1:0:0	3
Green waste	200	16:1:0	3
Green waste and sewage	100 and 200	32:1:2	3
Sewage	200	16:0:1	3

Physico-chemical measurements were taken at one day after filling, two days after, and then at approximately two-weekly intervals thereafter for 145 days.

A 100 mL water sample was taken after 82 days and analysed for concentrations of solutes. Cations (Al, As, Br, Ca, Cd, Cr, Co, Cu, Fe, Pb, Mg, Hg, SO_4^{-2}, Se, Si, Sn, Zn) were analysed via ICP-AES. Ammonium and NOx (nitrate and nitrite) anions were analysed on an auto-analyser using the Berthelot and persulfate digestion methods respectively, with subtraction (APHA, 1998).

Primary treated municipal sewage was collected from drying beds of the Collinsville Water Treatment Plant. Due to a very low output from the plant, this material had been dried for around 12 weeks and was thus of an extremely low water content (sewage density = 1.35 kg/L). A sample from this sewage collection was taken for analysis.

Green waste was collected from the Collinsville green waste tip. This material was largely representative of Australian garden waste and included lawn clippings, palm fronds and other leafy and woody material. Green waste was chopped into approximately 50 mm sections to fit into the cores. As Western Australia Custom's horticultural restrictions prevent importation of green waste from Queensland, a representative sample of green waste was prepared in Perth from similar local plants and was dried at 25ºC for one week to simulate ambient conditions at Collinsville municipal green waste dump. The sewage and green waste samples were then analysed for total N, total P and total organic C content as well as for the same suite of cations as the solid sewage sample via ICP-AES.

Statistical analyses

Hypothesis testing of univariate data was carried out in SPSS (2000) by one way Analysis of Variance (ANOVA) following $\log_{10}$ transformation and testing for normality of distribution and homogeneity of variance. If a significant difference was detected at $\alpha = 0.05$ level, *post-hoc* tests were made with Dunnett's multiple comparison test against the control group.

Multivariate wet chemistry data and physico-chemical data from day 86 was analysed by Principal Components Analysis (PCA) in the PRIMER software package (PRIMER-E Ltd, 2001). Prior to analysis, draftsman plots were created to determine which variables were highly (i.e., 95% or greater) Spearman rank-correlated with each other. These variables were then deleted to leave only a single dummy variable which then represented all of these correlated variables. For example, the two variables dissolved O_2 (%) and dissolved O_2 (mg/L) were reduced to just dissolved O_2 (%) to form a dummy variable "dissolved O_2" representing an index of the availability of dissolved O_2. Data were then $\log_{10}$ transformed to enhance a linear relationship between variables and finally normalised to account for different variable scales (Clarke and Warwick, 2001).

Results

Garrick East pit lake had a very low pH and high total Fe concentrations indicative of an Fe buffered system (Table 2), with heavy metals at environmentally toxic concentrations (ANZECC/ARMCANZ, 2000). Nutrient levels were also very low in this pit lake; especially for P as is typical of AMD waters (Borg and Holm, 2001; Lessmann et al., 2000).

Collinsville sewage sludge was high in S (10,000 mg kg^{-1}) and also contained notable levels of Al (2,700 mg kg^{-1}), Fe, Ca and Mg. However, concentrations of some heavy metals including Zn (1,500 mg kg^{-1}) and Ni (39 mg kg^{-1}) were also surprisingly high. Sewage was moderately high in N (31 mg kg^{-1}) and P (12 mg kg^{-1}), had relatively high organic C content (29%) and contributed significant alkalinity to the dosed AMD water (Table 2).

Although green waste contributed less alkalinity to the AMD water, it contained the highest percentage of organic C (31%). Green waste also contained moderate amounts of S (1,700 mg kg^{-1}), N (20 mg kg^{-1}) and P (2.3 mg kg^{-1}). Although Ca and Mg levels were higher in green waste compared to sewage, heavy metal concentrations were very low.

Although the control treatment looked largely unchanged over the course of the experiment, the green waste treatment had darkened considerably; probably through leaching of labile organic material and with a sweet odour indicative of fermentation processes. The presence of this process was further reinforced by the development of a blue-green coloured mould on the surface of both the green waste treatment and on the green waste and sewage treatment throughout the experiment (Fig. 5). The sewage treatment became more orange in colour, although when inspected from above this was found to be due to a precipitate (probably Fe) settling on the inside of the acrylic tube. The water in the sewage treatment appeared to have lost the orange tinge which the control cores still retained. The space between the greenwaste and sewage and the sediment surface began to blacken in the green waste and sewage treatment after only a few weeks of the experiment beginning. After Castro et al. (1999), this black material was assumed to be Fe^{+2} sulfides from the activities of SO_4^{-2} reducing bacteria, as the space above the mesocosm sediments of this treatment which were likely to be highly anaerobic. A strong sulfide smell, presumed to be hydrogen sulfide, also evolved as the black precipitate extended upwards through the mesocosm above the green waste and sewage solids. This sulfide smell and the commiserate presence of a black precipitate indicated that SO_4^{-2} reduction was occurring in these green waste treatments.

Two days after addition of organic substrates to AMD cores there was a statistically significant mean increase in pH above the control for the treatments of green waste and sewage, and sewage ($F_{3,8} = 23.060$, $p<0.001$) (Fig. 6). Green waste alone also showed a slight increase in pH, although this was not significant ($p>0.05$). All treatments also showed a significant decrease in redox compared to the control ($F_{3,8} = 16.673$, $p = 0.001$).

Table 2. Chemistry of GAE pit lake water and organic materials used in core experiments.

	GAE water	Sewage	Green waste*
Parameter	(mg/L)	(mg/Kg)	(mg/Kg)
Total sulfur	No data	10,000	1,700
Sulfate	2,610	No data	No data
Total nitrogen	0.51	[Π]31	[Π]20
Total phosphorus	<0.5	12	2.3
Total organic carbon	No data	29%	39%
pH	2.4	[Ψ]6.1	[ψ]5.6
Aluminium	204	17,000	1,000
Arsenic	0.013	7	1
Cadmium	0.019	3.2	<0.06
Calcium	519	12,000	26,000
Chromium	0.080	49	7.0
Cobalt	2.61	4.6	11
Copper	0.331	580	0.4
Iron	689	14,000	670
Lead	0.331	120	<1
Magnesium	726	2,700	2,800
Nickel	5.18	39	1.3
Zinc	16.0	1,500	47

*Green waste data from representative Perth sample; [Π]as Kjeldahl nitrogen; [ψ]1 (sample solid):10 (distilled water) paste test.

Temperature was very even within treatments averaging a coefficient of variation of 1% between all treatment replicates at each sampling date and ranging from 19.5–22°C. The pH of the control treatment level changed little (0.05 pH units) over the course of the experiment. Similarly, apart from an initial 0.4 pH unit increase in the sewage treatment, pH was not seen to increase over the course of the experiment in this treatment either. However, green waste, and green waste and sewage treatments both showed a dramatic increase in pH after day 37, to around 5.5 by day 145. Nevertheless, there appeared to be greater variation in the pH response with the green waste treatment than for either sewage or even green waste and sewage. Variability within treatments appeared to be primarily due to differences in the date at which probable SO_4^{-2} reduction and hence alkalinity production became established.

Specific conductance appeared to increase in all treatments following addition of organic material, with the greatest increase in the green waste treatment (Fig. 7b). Although there was a slight increase in control specific conductance over the experiment from 9.5–10.5 mS cm^{-1}, specific conductance appeared to decline in all treatments at around the same as alkalinity increased (i.e., day 37). Although treatments containing sewage were very variable between replicates, both treatments containing sewage appeared to decline in solute concentrations greater than did green waste alone.

Figure 5. Microcosm cores on day 0. Treatment allocations from left to right are control, green waste, sewage and green waste and sewage.

Figure 6. Mould growing on the surface of green waste-only cores at day 72.

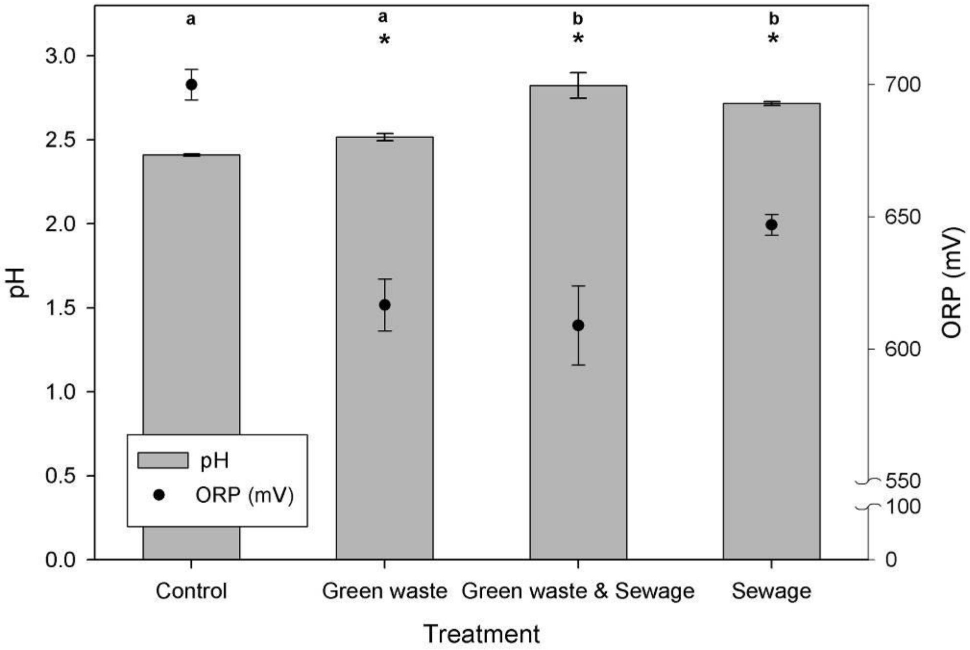

Figure 7. Microcosm core mean pH and redox potential on day 2 after addition of organic substrates. Error bars indicate single standard errors of the mean. Letters indicates statistically significant differences ($p<0.05$) between controls and treatments for pH, and asterisks indicate statistically significant differences ($p<0.05$) between controls and treatments for redox. Redox measured with a platinum reference electrode.

Aside from Fe and Mg which increased in the green waste treatment; and Cd, Ca, Cu, Pb, and Zn which increased in the sewage treatment, other solutes had decreased in all treatments by day 86. However the only statistically significant difference in solute concentrations at day 86 were where the Fe concentrations in sewage treatment was significantly lower than in the control; Cr, where both green waste and sewage, and sewage treatment concentrations were lower than control; and Cu and Se, where both green waste and green waste and sewage treatment concentrations were lower than control (Table 3).

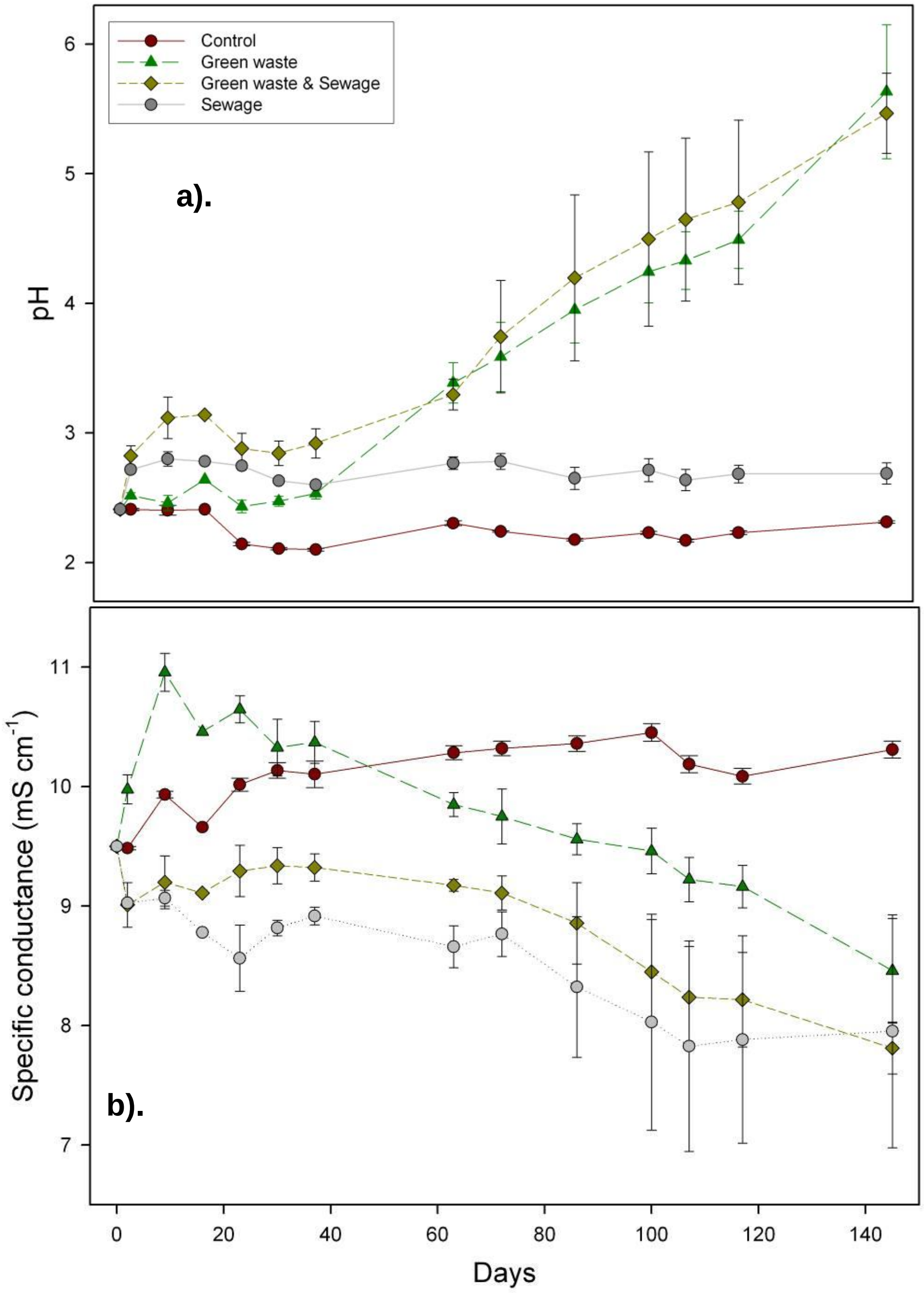

Figure 8. Change in **a).** pH, and **b).** specific conductance, of core microcosm treatment levels over time. Error bars indicate single standard errors of the mean.

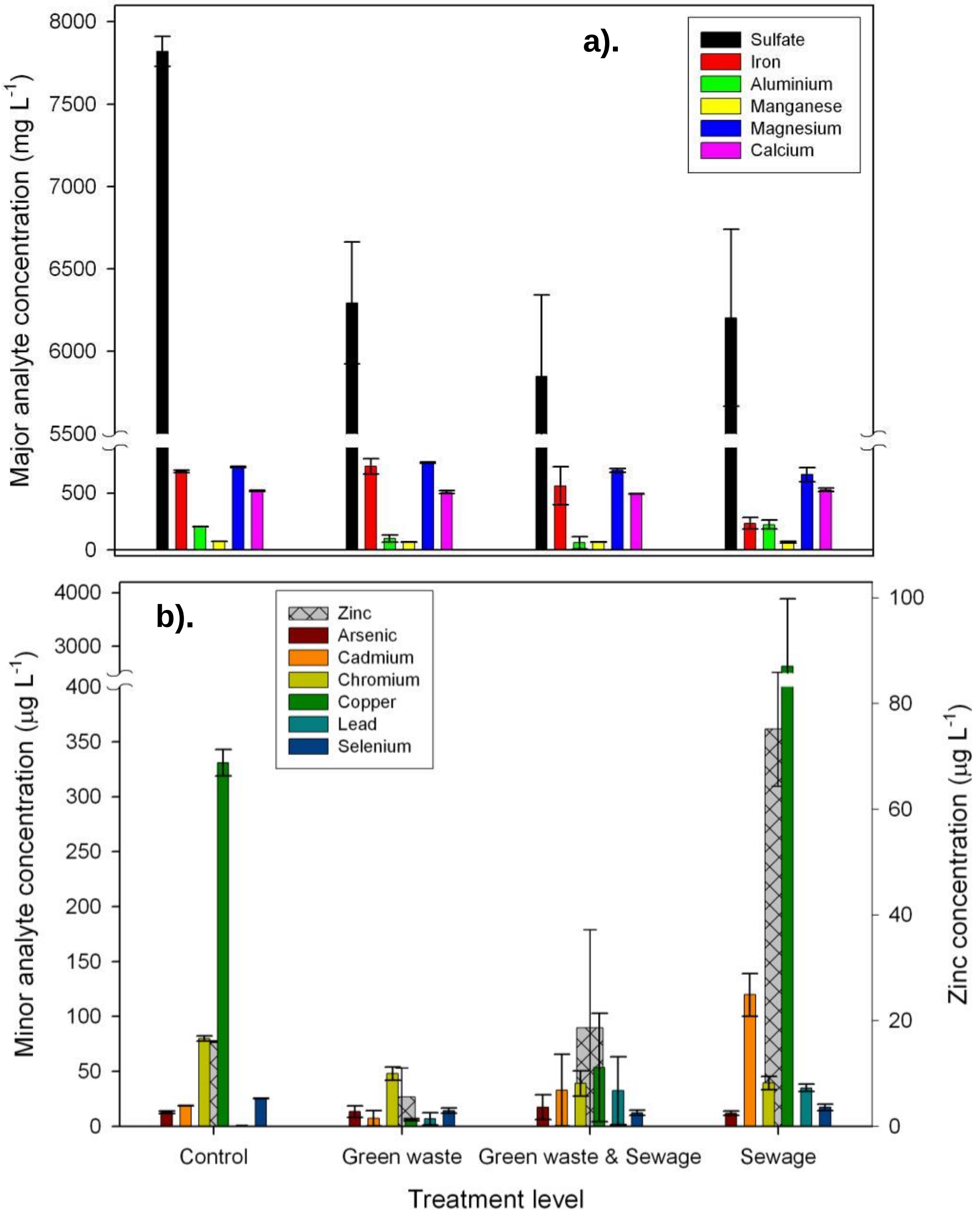

Figure 9. Day 86 concentrations of **a).** major, and **b).** minor, analytes for each treatment level. Error bars indicate single standard errors of the mean.

Table 3. Results of ANOVA test for differences between treatment levels solute concentrations at day 86. Bold type indicates significant difference at $\alpha = 0.05$ level.

Analyte	F-statistic	p-value
Aluminium	$F_{3,11} = 3.487$	0.070
Arsenic	$F_{3,11} = 0.010$	0.998
Cadmium	$F_{3,11} = 3.302$	0.078
Calcium	$F_{3,11} = 2.479$	0.135
Chromium	$\mathbf{F_{3,11} = 4.642}$	**0.037**
Copper	$\mathbf{F_{3,11} = 8.334}$	**0.008**
Iron	$\mathbf{F_{3,11} = 6.819}$	**0.014**
Lead	$F_{3,11} = 2.899$	0.102
Magnesium	$F_{3,11} = 1.635$	0.257
Manganese	$F_{3,11} = 0.532$	0.673
Selenium	$\mathbf{F_{3,11} = 4.936}$	**0.032**
Sulfate	$F_{3,11} = 3.441$	0.072
Zinc	$F_{3,11} = 3.435$	0.072

A Principal Components Analysis (PCA) indicated that addition of organic matter appeared to increase the variability of water chemistry parameters over that of the control (Figure 9). There also appeared to be synergistic effects on variability with the combination of green waste and sewage producing the most varied response between replicates. Addition of green waste moved the water chemistry primarily along the first principal component axis, addition of sewage separated along the second principal component axis. After 86 days, addition of sewage was correlated with increases in dissolved concentrations of the heavy metals Cd, Cu, Pb and Zn; and ammonia. Addition of sewage was correlated with decreases in Fe, Cr, SO_4^{-2} and electrical conductivity. Addition of green waste was correlated with increases in pH and with decreases in dissolved oxygen, redox, Al, Ni, SO_4^{-2} and Se. Nevertheless, although more variable, water chemistry correlations with green waste and sewage were very similar to green waste.

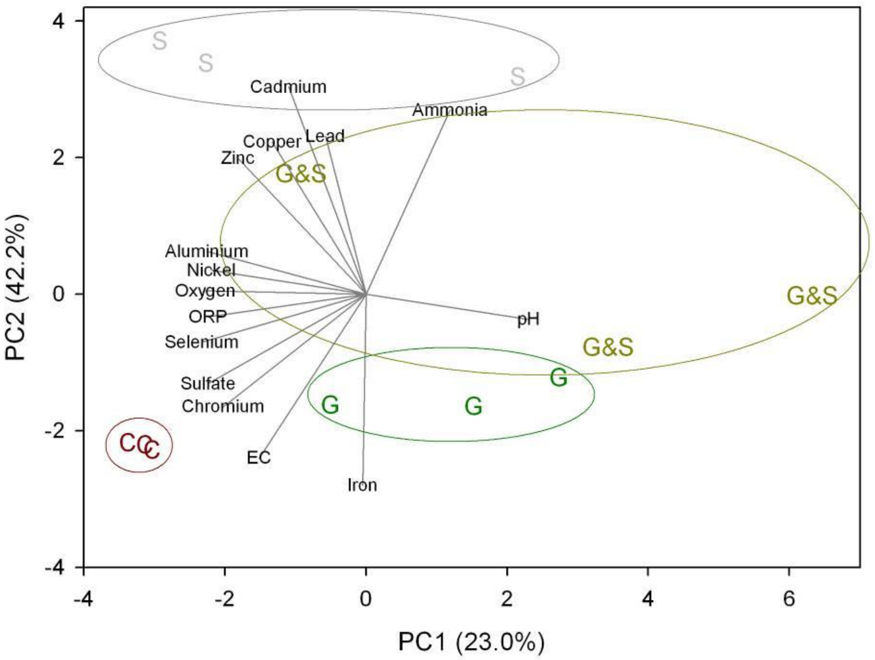

Figure 10. PCA of day 86 microcosm core solute concentrations and physico-chemistry. Vectors indicate variables contributing │>0.25│ to an eigenvector. Ellipses represent range of variation within treatment levels.

Discussion

The decrease in SO_4^{-2} concentrations in all of the treatment microcosms provides further support for biological SO_4^{-2} reduction occurring at varying rates and removing cations as precipitates in the dosed mesocosms. However, the success of SO_4^{-2} reduction in low pH waters appears to be contrary to Postgate's (1984) assertion that SO_4^{-2} reducing bacteria (SRB) activities only occur at pH >5. Nevertheless, published studies some time ago have reported SRB activity at pH values below this threshold. For example, SRB activity in an acid lake has been found by Herhily and Mills (1985) at pH 2.5, and at pH 2.7 by Gyure et al. (1987). In these studies and our own, it appears that slightly-acid microclimates, facilitated in part by alkalinity produced through some SO_4^{-2} reduction, enabled SO_4^{-2} reducing bacteria to survive at and to chemical reduce SO_4^{-2}, producing alkalinity as part of this biological process (Küsel and Dorsch, 2000; Küsel et al., 2001).

Depending on its source, sewage sludge can contain high concentrations of heavy metals (Berrow and Webber, 1971). This is especially so of Cd, which is often concentrated in sewage sludge from vegetables which have been fertilised with inorganic phosphates; a common application in depauperate Australian soils (Nursita et al., in press). Unexpectedly for a non-industrial content sewage source, the Collinsville sewage also displayed high concentrations for Ni, Pb, and Zn. However, these high heavy metal concentrations are still unlikely to present a problem over a longer remediation time. Along with Fe, these toxic heavy metals are expected to precipitate out of solution by reaction with H_2S, such that their toxic effect is removed from the water column to levels lower than observed at day 86. Further reduction in biological availability is likely to occur through formation of complexes with organic chelators present as components of the refractory green waste, such as organic acids (Tipping and Hurley, 1992). Nevertheless, the contribution that organic materials may make to the heavy metal burden of a pit lake needs to be considered in the choice of organic materials for remediation. This study found a straight green waste treatment performed similarly to that of green waste and sewage, with fewer heavy metals contributed to the water column. In this respect, remediation strategies may be best placed by choosing green waste as the bulk contribution to electron donors over that of sewage. As discussed, green waste also has an additional advantage of providing organic substances such as humic and fulvic acids with which, heavy metals may directly complex to. Ligand formation between heavy metals and refractory organics will further remove these toxic components from biological availability, albeit at a likely reduced capacity to that of SO_4^{-2} reduction processes (Brown Jr., 2001).

The initiation of alkalinity production in green waste and green waste and sewage in only 37 days is likely to be due to a combination of two important factors unique to this study. Firstly, most other research published to date has occurred in cool temperate areas of Europe and North America e.g., (Benner et al., 2002; Frömmichen et al., 2004; Küsel and Dorsch, 2000; Küsel et al., 2001; Tostche et al., 2003). The higher temperatures experienced in Collinsville, even during the middle of the Dry season (19.5–28.2°C) are likely to have exponentially increased biochemical rates of carbon diagenesis and metabolism. Seasonal change in ambient temperatures has also been identified as limiting rates of SO_4^{-2} reduction in some experiments with reduced rates of SO_4^{-2} reduction occurring over cooler seasonal periods (Benner et al., 2002; Gammons et al., 2000).

Secondly, the use of largely fresh green waste as opposed to refractory organic substrates such as straw (Frömmichen et al., 2003; Frömmichen et al., 2004), rye grass (Harris and Ragusa, 2000; Harris and Ragusa, 2001), etc., distinguishes this research from many others in the published literature. The greater labile fraction of organic material available in this fresher material may have contributed directly to electron donors for SO_4^{-2} reduction. For example, chlorophyll was seen to be leached from the green waste and it is likely that highly labile sap sugars would have leached also. Consequently, the use of fresh green waste to acid pit lakes, with or without complementary additions of sewage, may prove to be a novel practicable remediation strategy for AMD issues in remote mining locations.

Although these microcosm results are only indicative of the pit lake's water columns in a stratified form, regular monthly monitoring data of three CCP pit lakes over the last 9 months, including GAE, indicate strong thermal stratification at all times. Although the epilimnion may deepen to a few metres under strong forcing winds, even in the middle of the Dry season they remain thermally stratified within the first 2–3 m due to the rapid heating caused by their high light absorption (photosynthetic coefficient of attenuation of 0.28, author's unpublished data). This heating rate may decrease as Fe precipitates from the water column, however the large mass of woody green waste resting on the benthos in a field-scale trial will nevertheless also provide for hypolimnion stability though increased bed roughness and consequent resistance to wind forcing of the water column.

Mine water research and management is a very new and rapidly developing field (Wolkersdorfer, 2004). The corrective measures for very acidic hardwater mining lakes also greatly differ from those in use for eutrophication control (Klapper, 2003). As a consequence both researchers and regulatory agencies need to maintain an open-mind to treatment solution options for these unique environmental issues. Best practice AMD pit lake treatment is likely to be required to be made on a case-by-case basis where potential pit lake end uses and availability of organic substrates have been identified and a remediation strategy has consequently only then been constructed. Part of the challenge for both mining companies and regulatory waterbody managers with this assessment will be the ability to think laterally as to what the current values of the pit lake are, what treatments may be feasible, and what type of system (water quality and quantity) is desirable as an end result for either social, environmental or other enduses (Doupé and Lymbery, 2005).

Current and future projects

Another microcosm core experiment is also being completed by our CSML research team. This study will focus on the effect of both different organic loading rates and the effects of the addition of small amounts of lime on alkalinity production, and is based in our Perth laboratories for greater ease of experimental control and manipulation. The dosing of organic matter in this lake would be expected to continue over a few months, to be within the capacity of the wastewater treatment plants. For long-term remediation purposes, organic matter additions are expected to be required at an ongoing, albeit lower, dosing rate. The use of refractory organic forms such as woody green waste may also be advantageous to long-term remediation efforts, in that they will continue to degrade into more labile fractions available for SO_4^{-2} reducers over long periods of time.

Water column stratification is expected to remain strong in this remediated pit lake due to even-temperature and hot tropical climate in which it exists. Nonetheless, depth and duration of stratification will remain an important decider in the efficacy and viability of SO_4^{-2} reduction as a remediation measure in any pit lake (Martin et al., 2003; McNee et al., 2003) and to this end thermocouples studies and consequent one-dimensional modelling studies of changes in mixing regimes of a dosed lake with DYRESM software (Imberger and Patterson, 1981) are also planned.

The application of the science now requires on-site experiments at field scale to demonstrate this new technology of remediation with bulk low-grade organic sources. To this end we have begun a full-field scale evaluation of the efficacy of green waste and sewage additions to a section of the GAE pit lake (Fig. 2). This pit lake is typical of those of the CCP mining lease, as it is a highly acidic terminal lake with high concentrations of dissolved solutes. We have sectioned-off approximately ¼ of the western surface area of this lake by pushing overburden along a 100 m wall across the narrowest width of the lake (Fig. 2). This western portion of the lake will form a dosed treatment, receiving sewage and green waste over the next six months. The remaining ¾ section of the lake, along with two other, similar mine pit lakes, will function as control treatments during this time, and in the 12 months monitoring thereafter.

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