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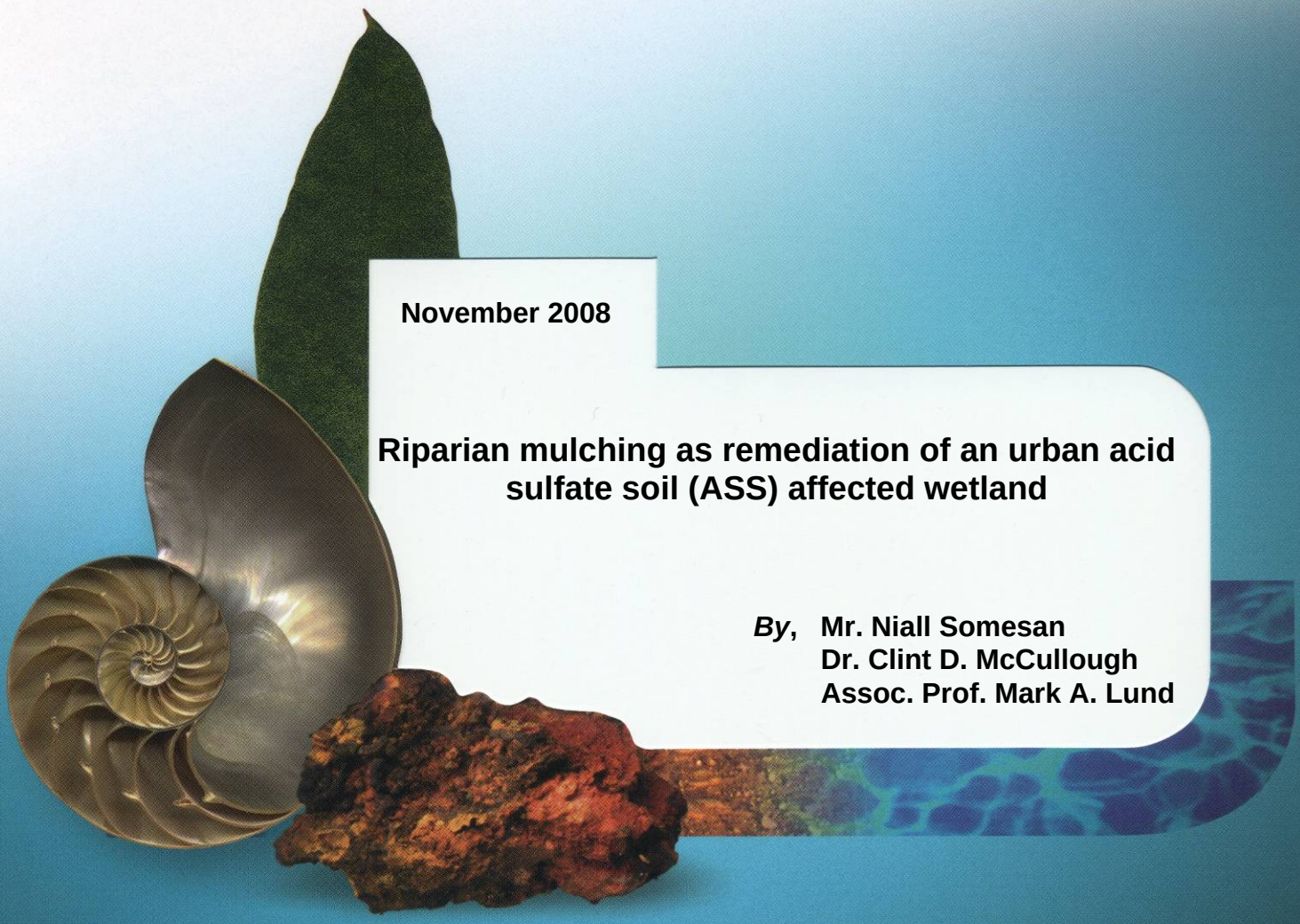

November 2008

Riparian mulching as remediation of an urban acid sulfate soil (ASS) affected wetland

By, Mr. Niall Somesan
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Mine Water and Environment Research/Centre for Ecosystem Management

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Prepared for,

City of Stirling

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1 Executive Summary

- The City of Stirling in Perth, WA, is currently experiencing groundwater acidification as a result of sulfidic peat disturbance caused by dewatering and peat stockpiling to provide residential construction.
- The Spoonbill-Shearwater Reserve is located in the Perth suburb of Stirling and has two lakes excavated to below the water table, with spoil used to create a large island in each lake. The lakes are each approximately 1.4 ha in area and 2–4 m deep.
- The lakes have a lack of aquatic biota, scalded shores and decreased vegetation as a result of their acidity. Water levels in the lakes vary seasonally, with drying events contributing to further acidification.
- Following recommendations by Edith Cowan University, mulching of the south lake margins has been made as a treatment to replace littoral habitat lost to scalding and to promote sulfur reduction by increasing the level of labile carbon.
- Previous studies have shown that mulching in the Spoonbill Lakes initially appeared to have a negative effect on water quality, with increased metal concentrations.
- The aim of this study was to determine if the addition of mulch to an acid sulfate affected wetland would affect its ecological values of water quality and biota over a longer period of treatment.
- Each lake was sampled at various points on September 5 2008. Sampling was made for water quality parameters of general physico-chemical and for metal and nutrient concentrations. Biotic samples of macroinvertebrates, zooplankton and periphyton and phytoplankton were also made.
- This study found that mulching has substantially improved the physical and chemical water quality in the Spoonbill treatment lake over this longer term. Significant decreases in metal concentrations, nutrient loads and increased or moderated pH suggest that mulching may form the basis of a cheap and easy remediation for other acid sulfate affected wetlands.

- Addition of organic material to stimulate carbon and nutrient cycling in ASS affected lakes may lead to long-term increases in water quality and more natural and ecological communities to develop.
- Further monitoring of these lakes is recommended in order to ascertain how long the benefits of mulching as remediation may last.

Frontispiece

Figure 1. Riparian vegetation may struggle to survive acidified conditions in the Spoonbill Lakes.

This document should be referenced as follows.

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2 Contents

3 Background

Acid Sulfate Soils (ASS) and human-induced acidification are recognised as a serious problem in many countries including Australia (White *et al.*, 1997; Green *et al.*, 2006; Hinwood *et al.*, 2006) ASS sediments commonly underlie coastal floodplains (Sammut *et al.*, 1996; Tulau & Naylor, 1999), where increasing human demands for land and water result in their exposure or disturbance, often leading to acidification. ASS contain reduced forms of sulfur (typically as pyrite - FeS_2) which are stable under anoxic conditions but oxidise to produce sulfuric acid if exposed to air. The oxidation process is strongly catalysed by the presence of Sulfate Oxidising Bacteria (SOB) and occurs in a series of stages. The overall process of sulfate oxidation can be represented by Equation 1 below (White *et al.*, 1997).

Equation 1: $FeS_{2\,(s)} + 15/4\ O_{2\,(g)} + 7/2\ H_2O_{(l)} \rightarrow Fe(OH)_{3(s)} + 2SO_4^{2-}{}_{(aq)} + 4H^+{}_{(aq)}$

Subsequent rewetting of the oxidised sediment results in low pH water that dissolves metals such as iron and aluminium as it moves through surrounding soils. The resulting discharge is acidic, sulfate and metal-rich and can cause severe environmental problems when it enters surface or groundwater (Green *et al.*, 2006; Hinwood *et al.*, 2006). Consumption of groundwater contaminated with metals may represent a significant public health risk (Appleyard *et al.*, 2004; Appleyard *et al.*, 2006; Hinwood *et al.*, 2006). The effects of ASS drainage on wetlands and waterways include: mass kills of biota, changes in aquatic macrophyte community composition, smothering the benthos with iron and aluminium flocs, algal blooms and invasion by acid-tolerant species (Sammut *et al.*, 1996; Tulau & Naylor, 1999). The problems associated with acidification are more severe in freshwater systems than in marine systems due to their generally lower buffering capacity (Tulau & Naylor, 1999).

Acid Sulfate Soils are widespread in Australia and their effects are well documented in eastern Australia, where widespread drainage engineering has resulted in acidification (Sammut *et al.*, 1996; White *et al.*, 1997). ASS are present in most

estuarine lowlands on the eastern and northern coasts of Australia as well as in parts of Western Australia (WA), South Australia and Victoria (Sammut *et al.*, 1996). Australia-wide there are approximately 1.2 million hectares of coastal land that contain ASS sediments (Sammut *et al.*, 1996).

In WA, ASS have been largely overlooked until recent drying events at Lake Jandabup in the northern suburbs of Perth were followed by a severe acidification event (Lund *et al.*, in press). The Swan Coastal Plain (SCP), which contains the city of Perth, is considered to have minimal acid generating potential due to its well-drained sandy soils (Appleyard *et al.*, 2004). While marine or estuarine ASS do occur in WA their distribution is limited to the immediate vicinity of major estuaries. However, ASS do occur commonly in WA as sulfidic peats in groundwater-fed wetlands (Appleyard *et al.*, 2004). Many such wetlands were previously considered marginal for development but are now under increasing pressure for residential expansion (Appleyard *et al.*, 2004). Urban development of sulfidic peats can result in oxidation of peats and subsequent contamination of groundwater. Groundwater contamination by acidic, metal-rich drainage has significant implications for public health since WA is heavily reliant on groundwater for both drinking and agriculture (Appleyard *et al.*, 2004; Hinwood *et al.*, 2006). The problem of acidification is compounded by the decreasing trend in rainfall, which increases demand on groundwater as well as reducing the rate of aquifer recharge. Declining groundwater further contributes to the exposure of ASS to oxidation (Hinwood *et al.*, 2006; Lund *et al.*, in press). These conditions increase the likelihood of future acidification events in wetlands and groundwater (Lund *et al.*, in press). As such, it has become increasingly necessary to investigate and develop techniques for the remediation of acid-affected wetlands.

There are a range of techniques available to remediate acidified and metal enriched waters, most of which have been developed as treatments for Acid Mine Drainage (AMD), a similar problem to ASS. The cost of active treatments such as liming can be prohibitive and treatment may be limited by supply of alkaline materials

(McCullough, 2007). Alternative treatments such as bioremediation are considered preferable since they require less in the way of maintenance.

The basis of bioremediation in ASS affected water is the activity of Sulfate Reducing Bacteria (SRB). These bacteria are able to produce alkalinity and immobilise metals, effectively reversing the acidification process (Tuttle *et al.*, 1969; Johnson & Hallberg, 2005). SRB increase pH by converting sulfate into hydrogen sulfide as shown in Equation 2 (Johnson & Hallberg, 2005).

Equation 2: $SO_4^{2-} + 2CH_2O + 2H^+ \rightarrow H_2S + 2H_2CO_3$

Sulfate reduction also removes iron and heavy metals from solution since many metals will form highly insoluble sulfides that precipitate and remain inert in benthic sediments as long as they are not aerated again (e.g., Equation 3, (Johnson & Hallberg, 2005; McCullough, 2007)).

Equation 3: $Zn^{2+} + H_2S \rightarrow ZnS + 2H^+$

The process of bacterial sulfate reduction requires a source of labile carbon (Tuttle *et al.*, 1969; Kolmert & Johnson, 2001; McCullough, 2007) which is usually supplied as plant material (mulch or green waste) or sewage in treatment areas. This type of *in situ* treatment for acidified waters has potential to be both efficient and effective but further research is necessary to determine optimum organic substrates (McCullough, 2007).

The City of Stirling in Perth, WA, is currently experiencing groundwater acidification as a result of sulfidic peat disturbance (Appleyard *et al.*, 2004; Appleyard *et al.*, 2006). Dewatering and peat stockpiling to provide residential construction space has exposed large amounts of ASS to oxidation. Two lakes were excavated for public amenity in the Spoonbill Reserve. A nearby peat stockpile has oxidised and acidified

and contaminated with high levels of aluminium, iron and arsenic the local groundwater which also feeds the artificial lakes in Spoonbill Reserve (Appleyard *et al.*, 2004; Hinwood *et al.*, 2006). The contamination was noted in December 2001 (WRC, 2002), when domestic gardens downstream of the lakes began to die after being watered with bore water. The City of Stirling investigated bores in the area and found pH as low as 1.9 and levels of metals far exceeding national drinking water guidelines (Appleyard *et al.*, 2004). Water in the excavated lakes was found to have pH as low as 2.4 (Appleyard *et al.*, 2004). Arsenic levels were found to exceed the national drinking water standard of 7 μg L^{-1} (NHMRC, 2004) by orders of magnitude, which is of particular concern for public health since some residents drink the bore water (Appleyard *et al.*, 2004).

The Spoonbill lakes present an opportunity to investigate the effectiveness of bioremediation treatments for acid sulfate affected wetlands. The lakes are undergoing trial bioremediation by the addition of mulch to the banks, and the construction of an aerobic wetland at the north end of the South lake. The aim of this study was to determine if the addition of mulch to an acid sulfate affected wetland would affect its ecological values, specifically:

1: Does mulching affect water quality in an ASS-affected wetland?

2: Does mulching affect aquatic biota in an ASS-affected wetland?

4 Methods

4.1 Study Site

The Spoonbill-Shearwater Reserve is located in the Perth suburb of Stirling. Perth has a Mediterranean climate with hot, dry summers and cool winters. The Spoonbill-Shearwater Reserve has two lakes excavated to below the water table, with spoil used to create a large island in each lake. The lakes are each approximately 1.4 ha in area and 2–4 m deep. Groundwater flow is to the southwest, which puts the lakes in the path of a plume of low pH groundwater generated by a stockpile of oxidised peat to the north of the lakes (Appleyard *et al.*, 2004). The lakes have a lack of aquatic biota, scalded shores and decreased vegetation as a result of their acidity. Water levels in the lakes vary seasonally, with drying events contributing to further acidification. Figure 2 shows the layout and location of the lakes and the sampling locations.

The remediation being applied comprises two parts: a water treatment plant that draws acidic water from the North lake and discharges circum-neutral treated water to the South lake, and mulching and planting of the riparian zone in the South Lake (Lund *et al.*, in press). The purpose of mulching is to replace littoral habitat lost to scalding, and to promote sulfur reduction by increasing the level of labile carbon.

Mulching in the Spoonbill Lakes initially appeared to have a negative effect on water quality, with increased metal levels recorded near the discharge site in South Lake. This was likely the result of accumulated ASS secondary minerals in the sediments being reduced and mobilised (Lund *et al.*, in press). The effect was expected to be limited by the supply of these minerals, and positive effects on water quality were expected in the longer term (Lund *et al.*, in press). Mulching was also found to increase macroinvertebrate abundance in the treated lake, independently of the water treatment plant. Acidification of wetlands can have a severe impact on macroinvertebrate community structure (Sommer & Horwitz, 2001) so it is

anticipated that remediation of acidity and metal contamination will be reflected in improved macroinvertebrate community health and composition.

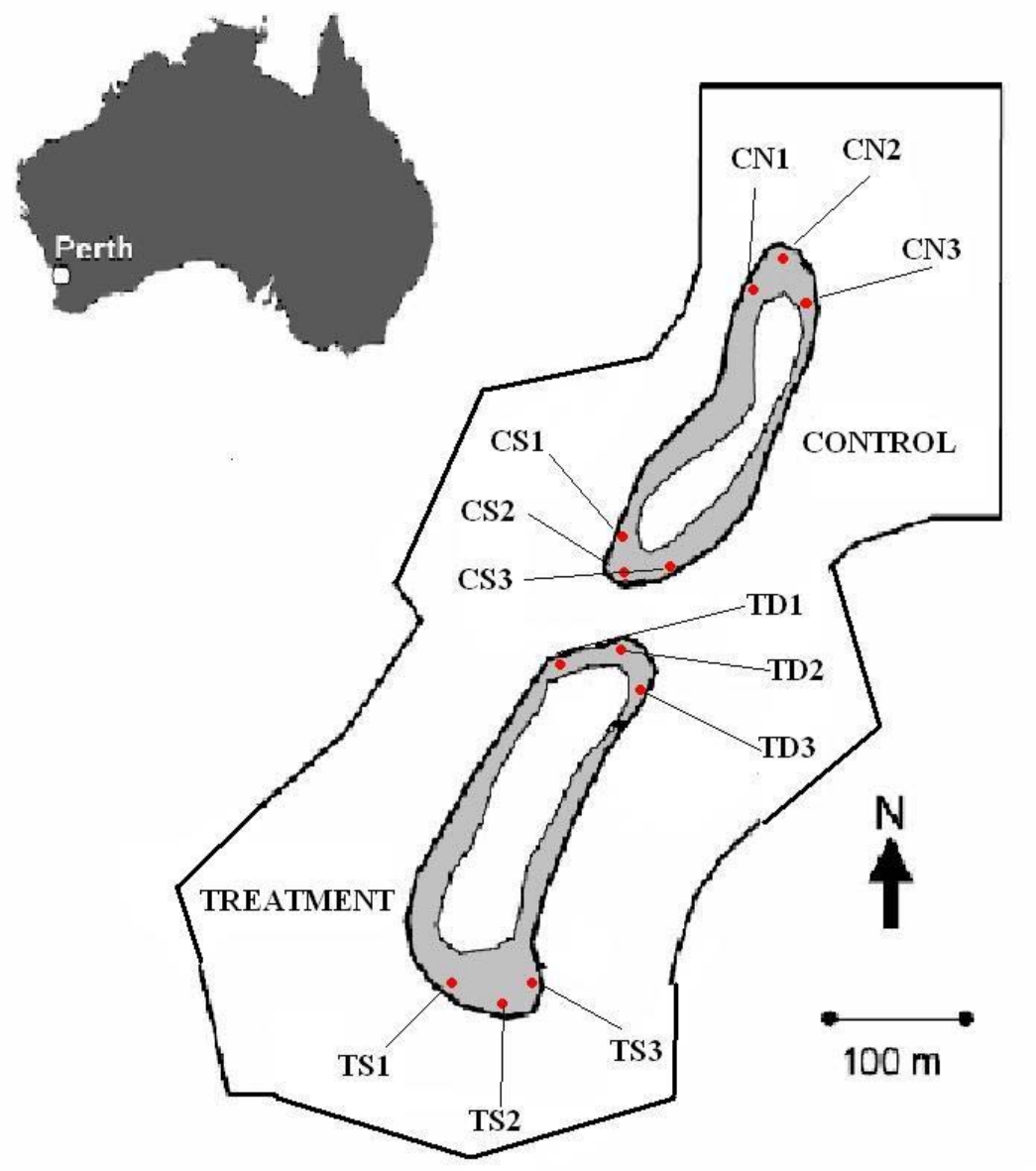

Figure 2. The artificial lakes in Spoonbill Reserve, showing replicate locations at each sampling site CN, CS, TD and TS. (Lund et al, in press)

4.2 Water Quality Sampling

Sampling took place on September 5 2008. Conditions on the day of sampling were fine and clear. Each lake was sampled at its northern and southern ends, with three replicates at each site. Sample sites are referred to in this report as CN (Control Lake

Northern end), CS (Control Lake Southern end), TD (Treatment Lake Discharge end) and TS (Treatment Lake Southern end). Water samples and measurements of water quality were taken prior to macroinvertebrate and sediment sampling to minimise disturbance of sediments that would affect measurements. Physico-chemical parameters were measured using a Hydrolab Datasonde 4a multiparameter probe with measurements displayed on a PDA. Prior to sampling the Datasonde was calibrated in the Wetlands Laboratory at ECU, following calibration methods detailed in the manual. Measurements at each location were taken approximately 1.5 metres from the bank, at a depth of approximately 20cm. Measured parameters included pH, temperature, conductivity, dissolved oxygen, ORP, turbidity and Chlorophyll *a*.

Surface water samples were collected for laboratory analysis of metal concentration, nutrients and Non Purgible Organic Carbon (NPOC). All water samples were collected in acid-washed polyethylene bottles to preserve sample integrity. Nutrient samples were stored on ice until returned to the University and frozen to prevent chemical changes. Measurement of nutrient content was by laboratory determination of total phosphorus (TP), total Kjeldhal nitrogen (TKN), filterable reactive phosphorus (FRP), ammonia (NH_4^+) and oxides of nitrogen (NO_x). Samples for metals content analysis were filtered on-site with a 0.45 μm Glass Fibre – Coarse (GF/C) filter paper, then acidified by addition of 10% HNO_3 and stored on ice until returned to the University and stored at 4 °C. Metal and metalloid concentrations were quantified using Inductively Coupled Plasma Atomic Emission Spectrography (ICP-AES). The samples were analysed for the following metals and metalloids: Al, As, B, Ca, Cd, Co, Cr, Cu, Fe, Hg, K, Mg, Mn, Na, Ni, Pb, S, Se, Zn. All chemical analyses were carried out by Mark Bannister at the School of Natural Sciences Analytical Services laboratories.

4.3 Sediment Sampling

Sediment cores were taken to a depth of 10cm at each replicate location. Sediment samples were analysed for organic content by the loss on ignition (LOI) method. Wet sediment was placed in weighed metal containers and dried at 105 °C for 24 hours. The dried samples were weighed and then placed in a furnace at 550 °C for 24 hours

to burn off all organic material. Burned samples were cooled in a desiccator and reweighed. The difference between dried and burned weights allowed calculation of organic content.

4.4 Aquatic Biota Sampling

Sampling for aquatic biota was conducted in early September as this is thought to be the peak time for macroinvertebrate abundance. Macroinvertebrate samples were obtained by sweeping 1 m^2 quadrats in the littoral area with a 250 μm dip net. Sweeps were towards the shore and each quadrat was swept twice, with a maximum of four strokes per sweep. Macroinvertebrate samples were preserved on-site with 100% ethanol and returned to the laboratory for sorting, counting and identification. Several samples required sub-sampling due to large amounts of mulch present, as well as high numbers of Chironomid larvae. Samples were separated into coarse and fine fractions using a set of nested sieves (2 mm and 500 μm) with material below 500 μm discarded. The coarse fraction was placed in a photographic developing tray with water and examined for macroinvertebrates. The fine fraction was transferred into a Bogarov tray and examined for macroinvertebrates using a 7-70X magnification stereo dissecting microscope. All macroinvertebrates found were stored in 100% ethanol for identification. Macroinvertebrates were identified using taxonomic keys, to species level wherever possible. For each sample the particulate matter was separated into fine particulate organic matter (FPOM, 500 μm–2 mm) and coarse particulate organic matter (CPOM, >2mm). These fractions were analysed for organic content by the loss-on-ignition (LOI) method as described in section 3.3.

Zooplankton and phytoplankton were collected by tows through measured distances with a 153 μm plankton net. Zooplankton samples were preserved by addition of 100% ethanol and phytoplankton samples were preserved with Lugol's iodine solution. A subsample from each zooplankton sample was examined under the stereo dissecting microscope to determine if zooplankton were present. A subsample from each phytoplankton sample was examined under a compound microscope to determine if phytoplankton were present.

Benthic primary production was measured with periphytometers made of sections of jarrah plank (125 x 175 mm) to simulate conditions in the mulch. These were deployed at each replicate site one week prior to sampling and retrieved one week after sampling, to maximise the opportunity for periphyton growth. As a control, glass periphytometers (203 x 105 mm) were deployed in tandem with the wooden periphytometers. Upon return to the lab, periphyton was scraped into labelled centrifuge tubes and frozen prior to processing. Processing followed APHA (1998) methods: samples were thawed then centrifuged at 3,000 rpm for 30 s and as much water as possible pipetted out. Ten mL of N-N Dimethylformamide (DMF) was added to each sample and the samples were shaken then left in the dark at 4°C for approximately 24 h. The samples were centrifuged again at 3,000 rpm for 30 s and the DMF solution removed by glass pipette. A spectrophotometer was zeroed using pure DMF and readings taken of each sample's absorbance at 750 nm and 665 nm. For the 750 nm absorbance a 40 mm quartz cuvette was used but at 665 nm this gave unacceptably high absorbance and so a 10 mm cuvette was substituted. Next the samples were acidified by addition of 200 µl of 10% HCl solution and after 60 s the 665nm absorbance was measured again. Calculations based on these absorbances allowed the derivation of chlorophyll a and phaeophytin densities at each replicate location. Following spectrometry the samples were recombined in pre-weighed 25mL crucibles and dried at 50°C to remove the DMF, then dried at 105°C overnight and weighed for dry mass. The dried samples were placed in a furnace at 550°C for 24 h to burn off all organic material, then cooled in a desiccator and weighed again. Calculations then determined the organic content of each sample (APHA, 1998).

5 Results

All data were processed and graphed using Microsoft Excel. Certain elements of the raw data were excluded from analysis as outliers. These are noted in the appendices. At two sites replicate periphytometers were lost, thus for these sites the means are calculated using only two replicates.

5.1 Physico-chemical parameters

Measurements of physico-chemical properties showed that pH was low at all sampling locations, with mean values ranging from 2.4–3.1. Values for pH were lower at both control sites than at either treatment site. Variation in pH levels was comparable at all sites except TD, which had much lower variation than the others. Water temperature ranged from around 16.6°C in the control lake to around 17.4°C in the treatment lake. Chlorophyll a was present at the northern end of each lake, with the highest value recorded at TD although at TD the variation between replicates was high. Site CS recorded no chlorophyll a and site TS recorded a low value compared to CN and TD. Turbidity was low in general as the water was clear. The highest value (0.67 NTU) was recorded at site TS. Sites CS and TD had comparable low values and site CN recorded zero turbidity. For all sites with measured values for turbidity the variation between replicates was high. ORP was high at all sites, with values at CN, CS and TS consistently around 500 mV. Site TD had significantly lower ORP at 407 mV, and variation between replicates was considerably higher than for any other site. Conductivity was marginally lower in the treated lake than in the control, with low variation. The lowest value was recorded at site TD. Dissolved oxygen levels were high in both lakes (>70%), with each lake showing higher oxygen at its south end than its north end. Figure 3 illustrates the differences between sites for physico-chemical properties.

Eight metals were present at concentrations greater than detection limits. Concentrations for these metals are shown in Figure 4. Levels of Al, Fe and Zn were significantly lower in the treatment lake, with Al and Fe showing an almost threefold reduction. Conversely, levels of K and Mg were notably higher in the treated lake, and levels of Ca, Mn and Na were marginally higher in the treated lake.

Concentrations of Fe, Al and Zn exceeded ANZECC/ARMCANZ guideline 95% protection values (ANZECC/ARMCANZ, 2000) by orders of magnitude, even in the treatment lake.

Measured nitrogen content in water samples showed a strong decrease between control and treatment lakes. The majority of N was present as NH_4^+ and for this species the difference between control and treatment lakes was most pronounced, with concentrations in the control lake exceeding those in the treated lake by six times. Concentration of nitrogen oxides (NO_x) was lower in the treatment lake than the control lake, with a decreasing trend from north to south. Variability between replicates was significantly higher in the treatment lake than the control lake.

Overall concentrations of non-purgible organic carbon (NPOC) decreased from control to treatment lake, as did phosphate concentration. In both lakes the concentration of NPOC and PO_4 decreased from north to south. Sulfur concentrations were lower in the treatment lake than in the control. Total phosphorus was high in general, with CN recording the lowest value and CS the highest.

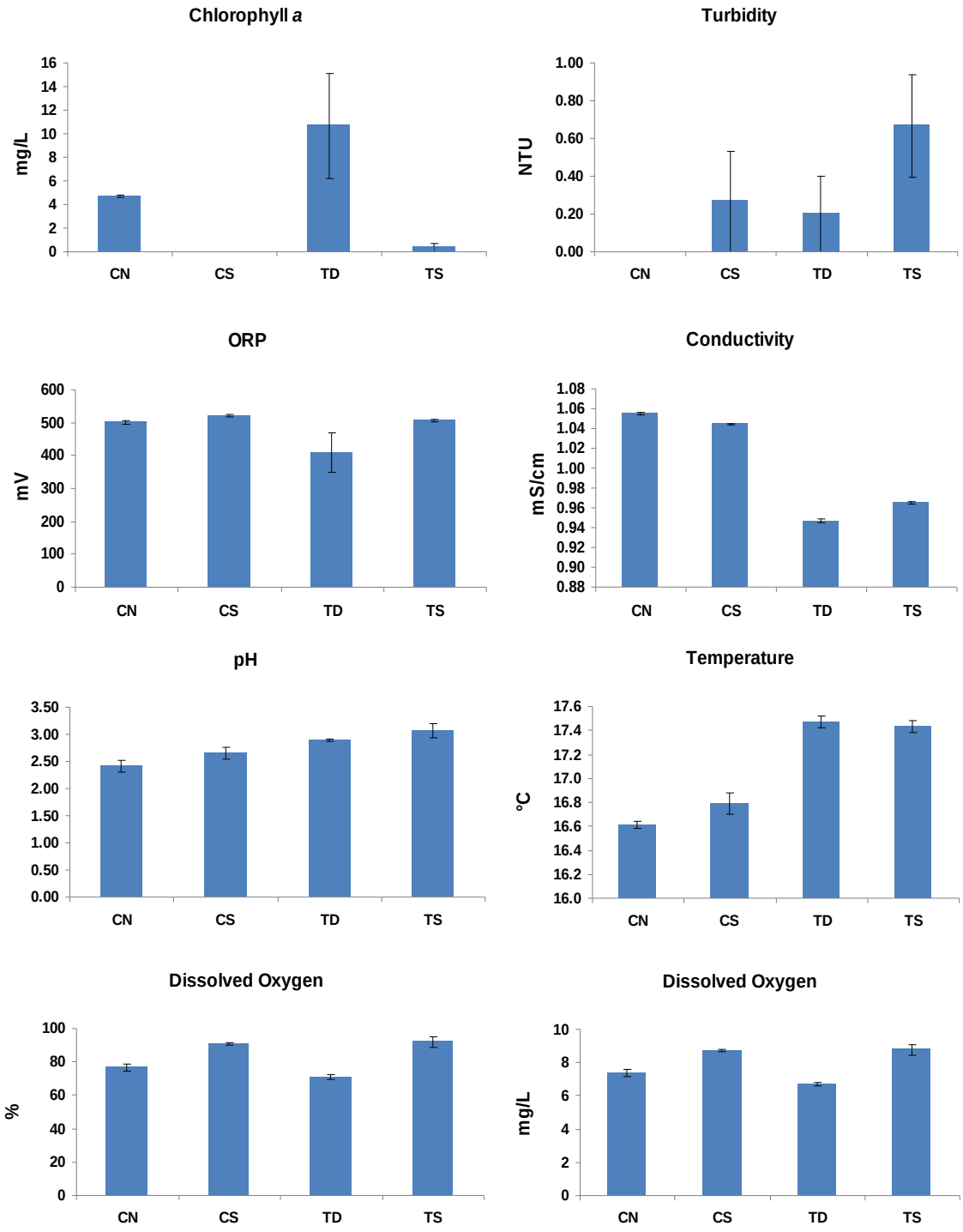

Figure 3. Variation of physico-chemical parameters between control and treatment sample locations in the Spoonbill Lakes. Note: not all axes begin at zero.

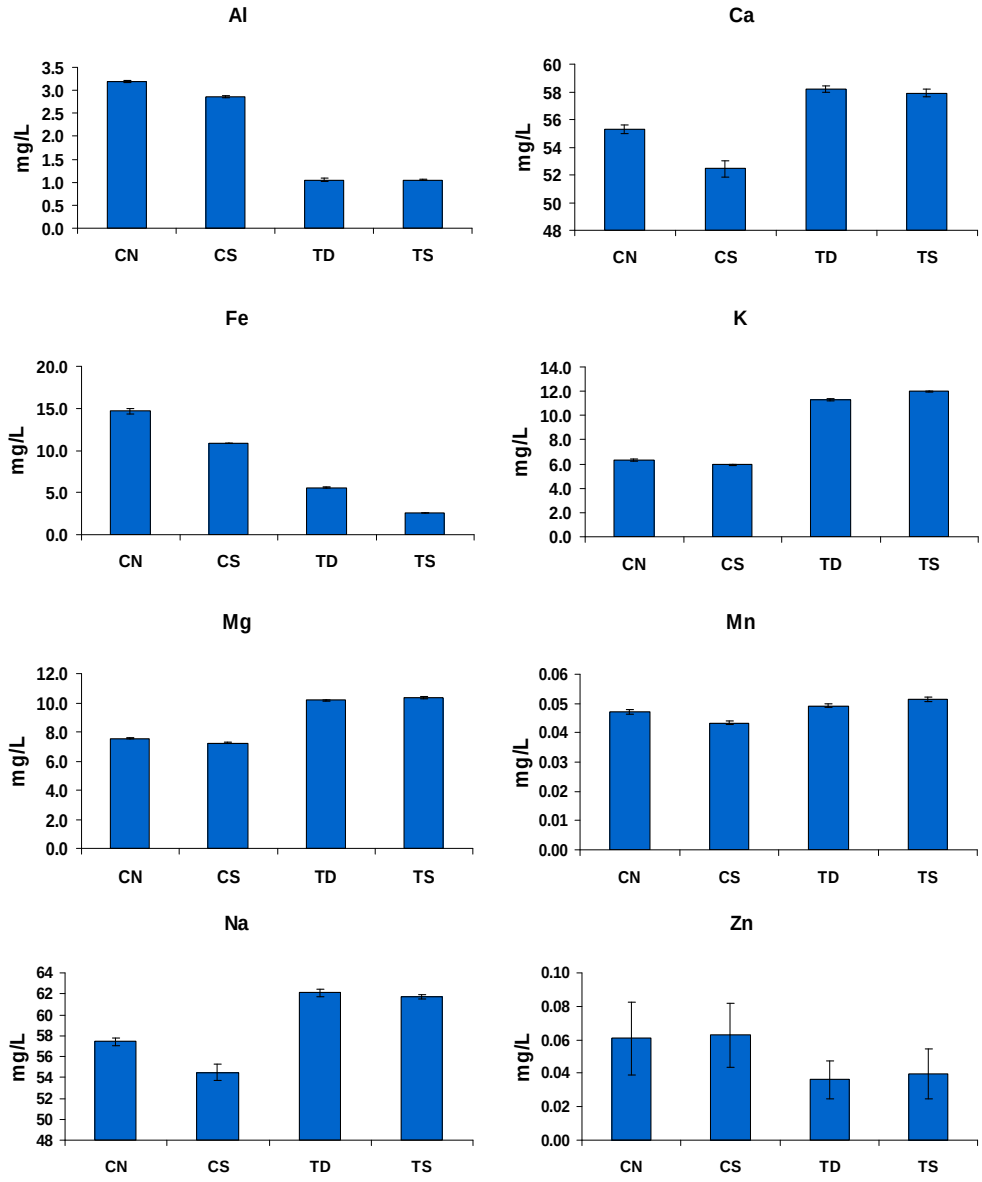

Figure 4. Concentrations of metals in water samples from the Spoonbill Lakes. Only those metals that were above detection limits are shown. Note that for some graphs the y-axis does not begin at zero.

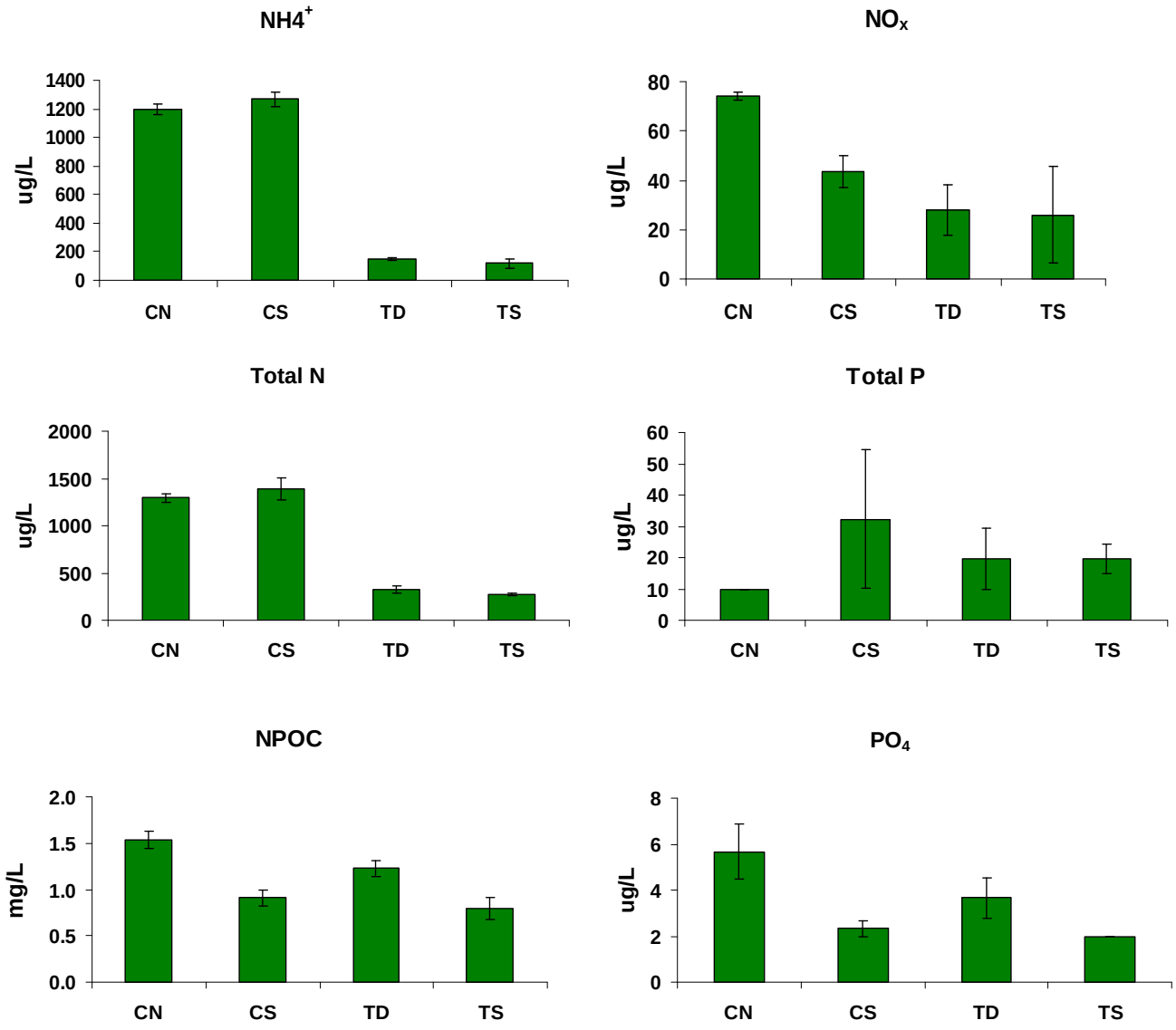

Figure 5. Nutrient concentrations in water at each sampling site in the Spoonbill Lakes.

5.2 Aquatic Biota

A total of 42 taxa and 5,490 individuals m^{-2} were found between the two lakes, representing 18 families. Interestingly, the treatment lake actually had less species richness than the control lake, with site CN having by far the greatest taxa richness. For both lakes the northern end showed higher species richness than the southern end, although the difference was more pronounced in the control lake. Macroinvertebrate abundance showed an opposite trend to taxa richness, with the treatment lake far exceeding the control. It should be noted, however, that while site TD had by far the

highest mean abundance its variability was also very high. Figure 6 illustrates the trends in richness and abundance between sampling locations.

Two macroinvertebrate families, Chironomidae (midges) and Dytiscidae (diving beetles) accounted for almost all of the macroinvertebrate abundance in the Spoonbill Lakes. The vast majority of macroinvertebrate abundance at all sites was contributed by larval Chironomidae, although the proportions of genera and species varied between sites. For example, both control sites (CN and CS) were dominated by *Chironomus occidentalis* but had a significant presence of *Chironomus alternans*, and while neither of the treatment sites (TD and TS) recorded any presence of *C .alternans* site TD was dominated by *C. occidentalis* and site TS was dominated by another Chironomid, *Tanytarsus fuscithorax*. Also present in significant numbers were larvae of the Dytiscidae genus *Necterosoma,* for which the treatment lake showed considerably higher abundance than the control lake. Adult *Necterosoma* were only found in the treatment lake. Larval Culicidae (mosquitoes) of the genus *Culex* were present in significant numbers at site TD but not at any other site. Larvae and adults of the large Dytiscid genera *Rhantus* and *Lancetes* were found at site CN but nowhere else. Adults and larvae of the Hydrophilidae (scavenger beetle) genus *Berosus* were present at site TS but at no other site. Several genera of Corixidae (waterboatmen) were present, almost exclusively in the control lake, with the exception of the genus *Agraptocorixa* being present at site TS. Full details of macroinvertebrate results are shown in Appendix 1.

Benthic primary production as indicated by periphyton chlorophyll *a* content was greatest at site TD and lowest at site CS. In both lakes the northern sites had significantly higher production of chlorophyll *a* than the southern sites. Generally the jarrah blocks had higher levels of primary production than the glass controls. Particularly at CS and TS, large numbers of sand tubes constructed by Chironomid larvae were present on the periphytometers at the time of collection. Figure 7 shows the calculated densities of chlorophyll *a* at each sampling site, for both the jarrah and glass periphytometers. It should be noted that during the spectrometric determination of chlorophyll *a* content the samples from site TD consistently exceeded

recommended absorbances at 665 nm, even when the smallest cuvette was used. For this reason the chlorophyll *a* result at TD, while undoubtedly high, cannot be considered accurately quantified and is therefore considered an outlier for the purpose of further analyses.

No zooplankton or phytoplankton were observed in any of the samples taken.

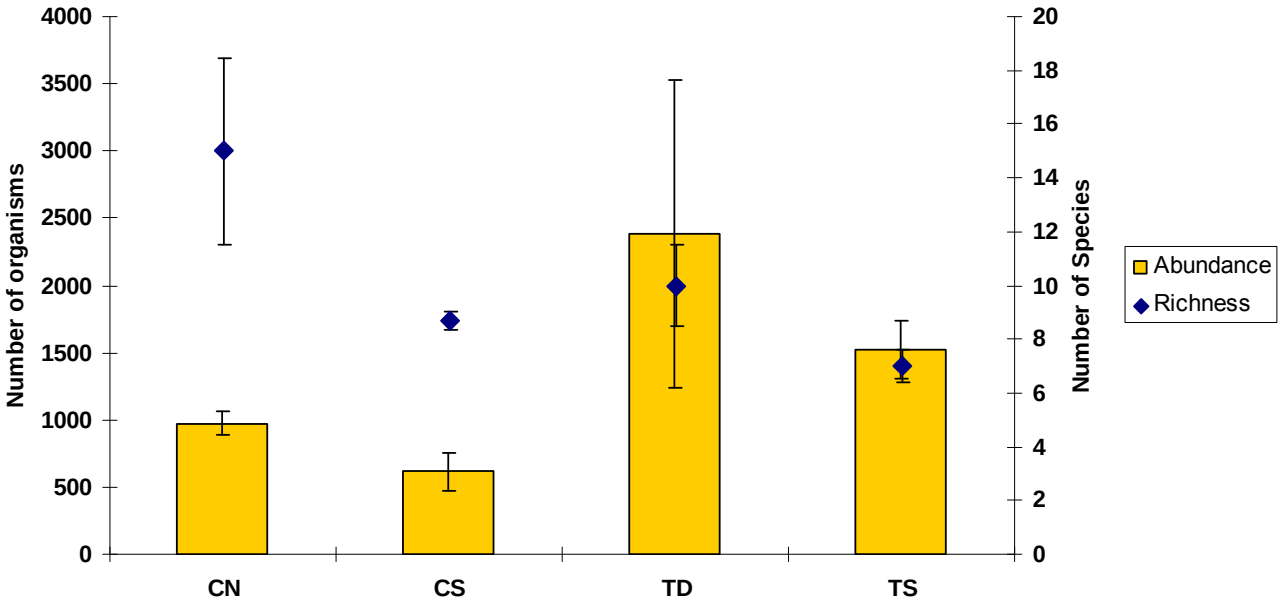

Figure 6. Macroinvertebrate species richness and abundance at each sampling location in the Spoonbill Lakes. Abundance is measured on the left y-axis and richness on the right. Note that the scales are different.

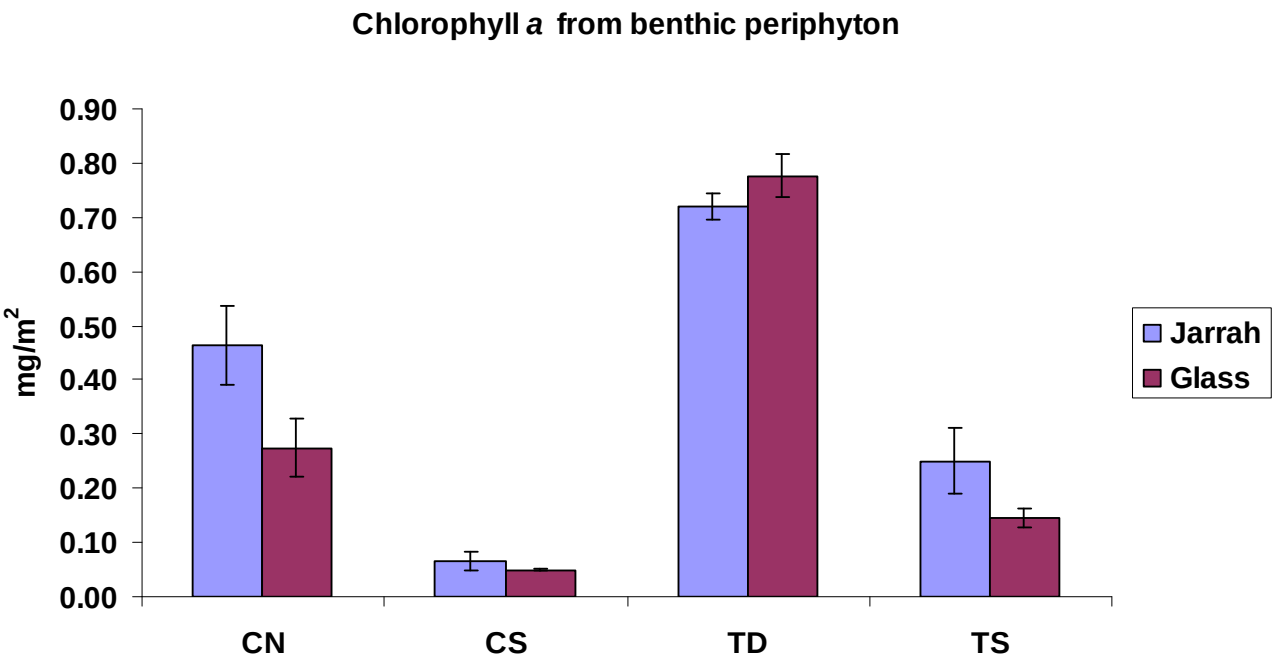

Figure 7. Chlorophyll a measured from periphyton growth on differing substrates in control and treatment lakes at Spoonbill Reserve. Chlorophyll a is a useful indicator of primary production and can be related to carbon and nutrient availability (Carpenter et al., 1998; Rabalais, 2002).

5.3 Organic Content

The organic content of coarse material (CPOM) collected in sweep nets was high for all sites. Site CN had both the lowest organic content and the highest variability. For fine material (FPOM) the organic content was consistently high in the treatment lake, but in the control lake site CN recorded a substantially lower value and site CS showed near zero organic content. Sediment organic contents showed a clear difference between control and treatment lakes, with values near zero for both sites in the control lake and treatment lake values ranging from 20–60% organic content (Figure 8). Organic content of periphyton scrapes was higher in the treatment lake than the control (Figure 8).

Scatter plots were used to investigate the effect of CPOM organic content on dissolved oxygen, macroinvertebrate species richness and abundance in the Spoonbill Lakes. Neither dissolved oxygen nor macroinvertebrate abundance had any significant

correlation to CPOM, but macroinvertebrate species richness showed an apparent decreasing trend with higher CPOM organic content (**Figure 9**).

A positive correlation was found between benthic primary production (as measured by chlorophyll *a*) and macroinvertebrate species richness. The trend was also positive for primary production and macroinvertebrate abundance but the correlation was weak. Chlorophyll *a* production was found to decrease with increasing organic content in CPOM, with a strongly linear relationship (**Figure 10**).

Macroinvertebrate abundance was found to have a strongly linear correlation with the organic content of sediments. There was also a negative correlation between macroinvertebrate abundance and the concentration of aluminium in the water (**Figure 11**).

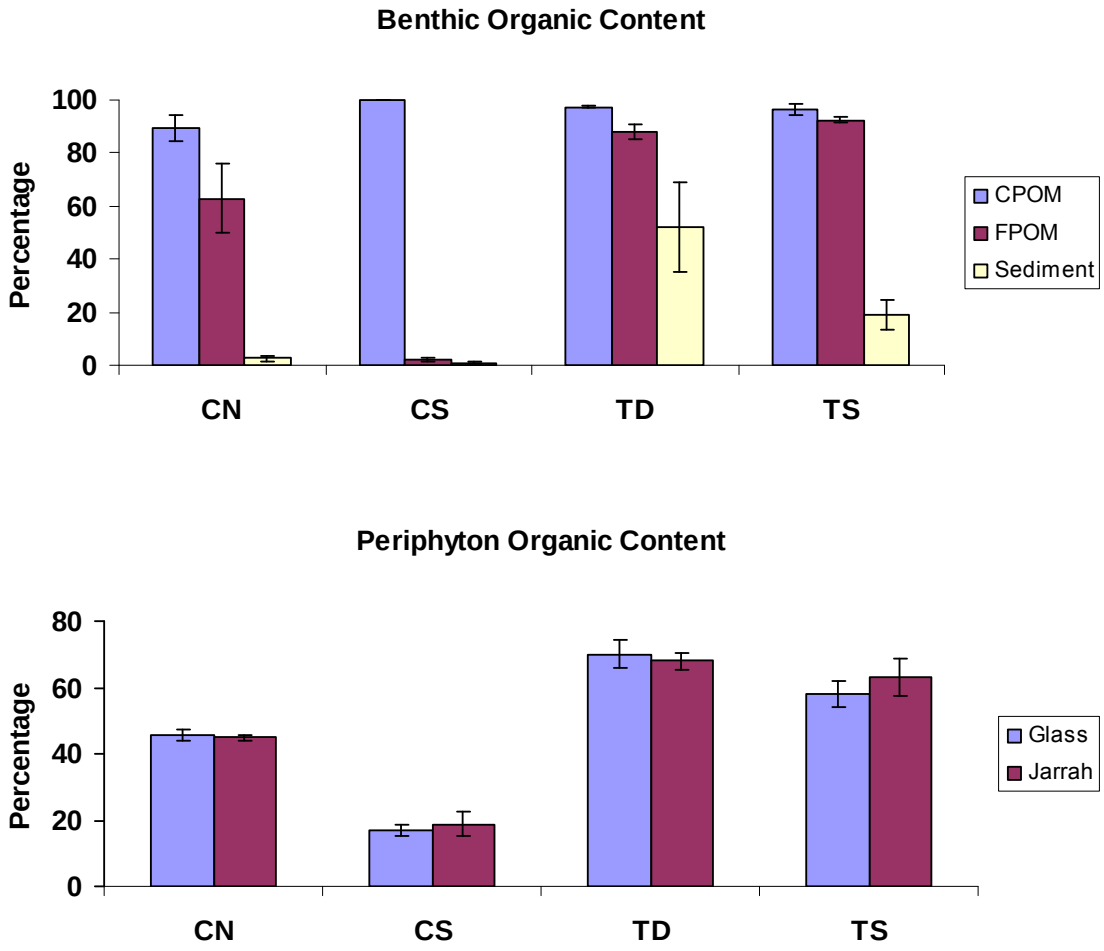

Figure 8. Organic content in sweep net samples, sediment cores and periphyton scrapes at each sampling location in the Spoonbill Lakes.

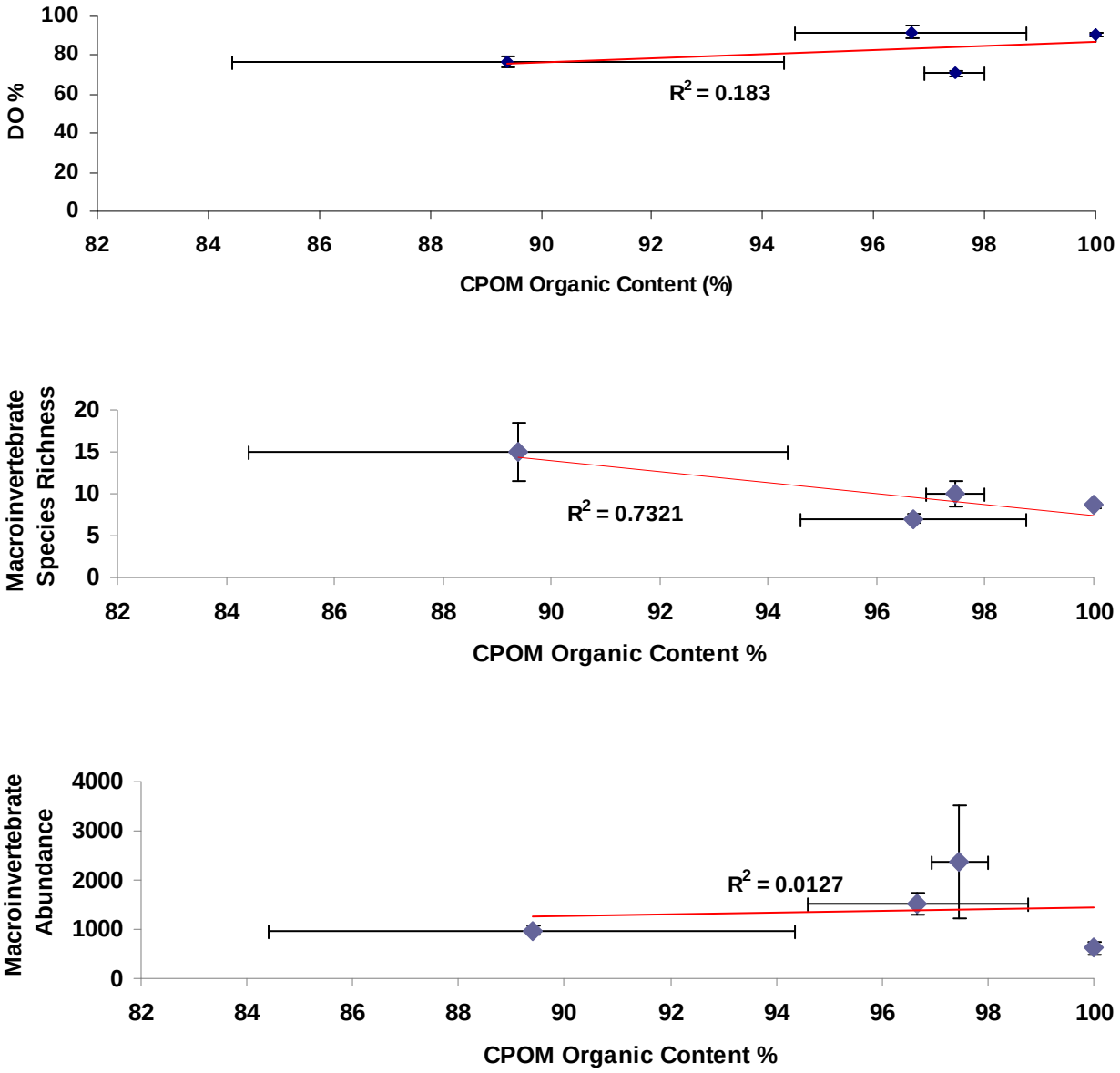

Figure 9. Correlation of CPOM organic content with macroinvertebrate richness and diversity as well as dissolved oxygen.

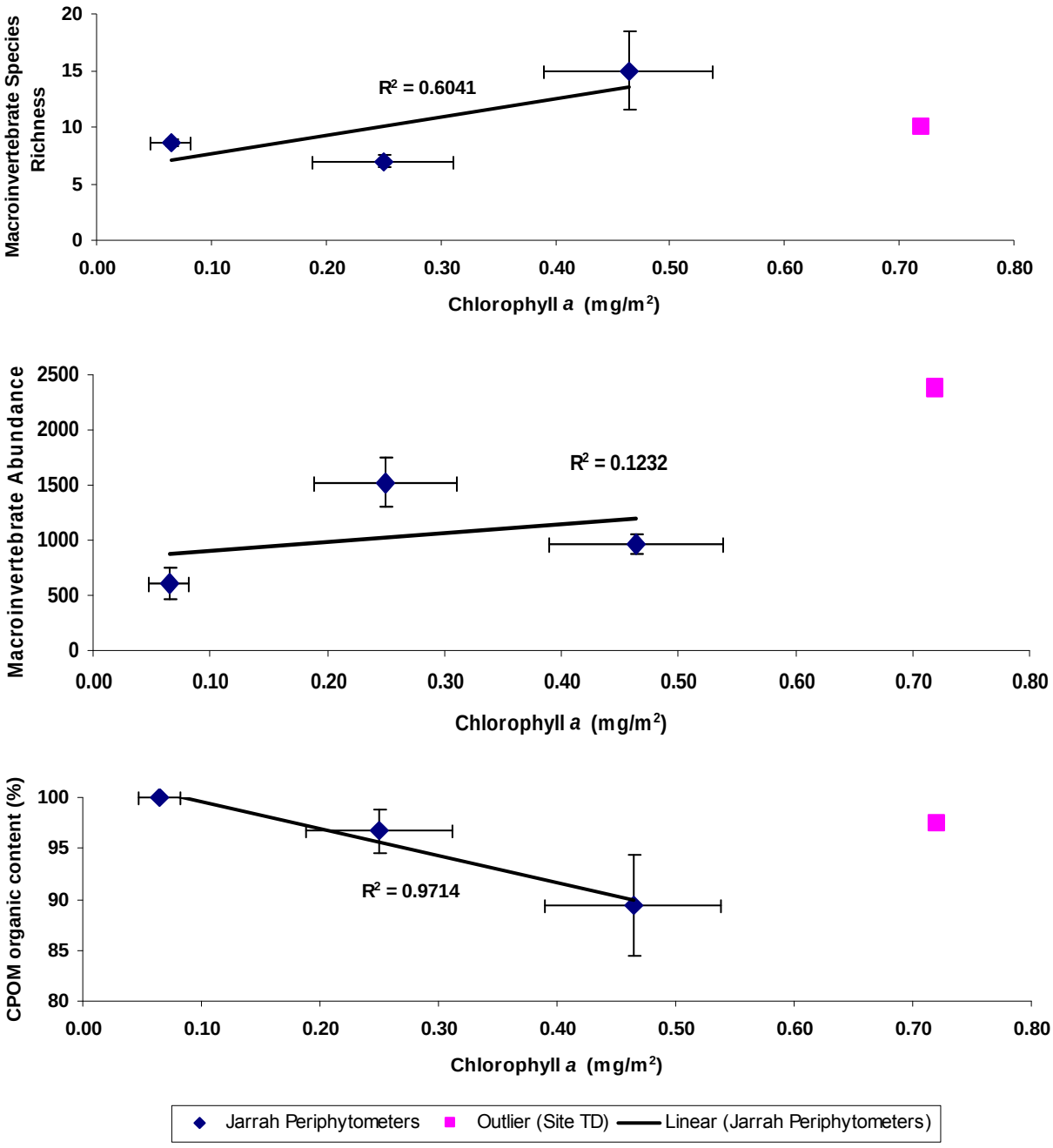

Figure 10. Correlations between benthic chlorophyll a production and macroinvertebrate richness and abundance, as well as the organic content of benthic CPOM.

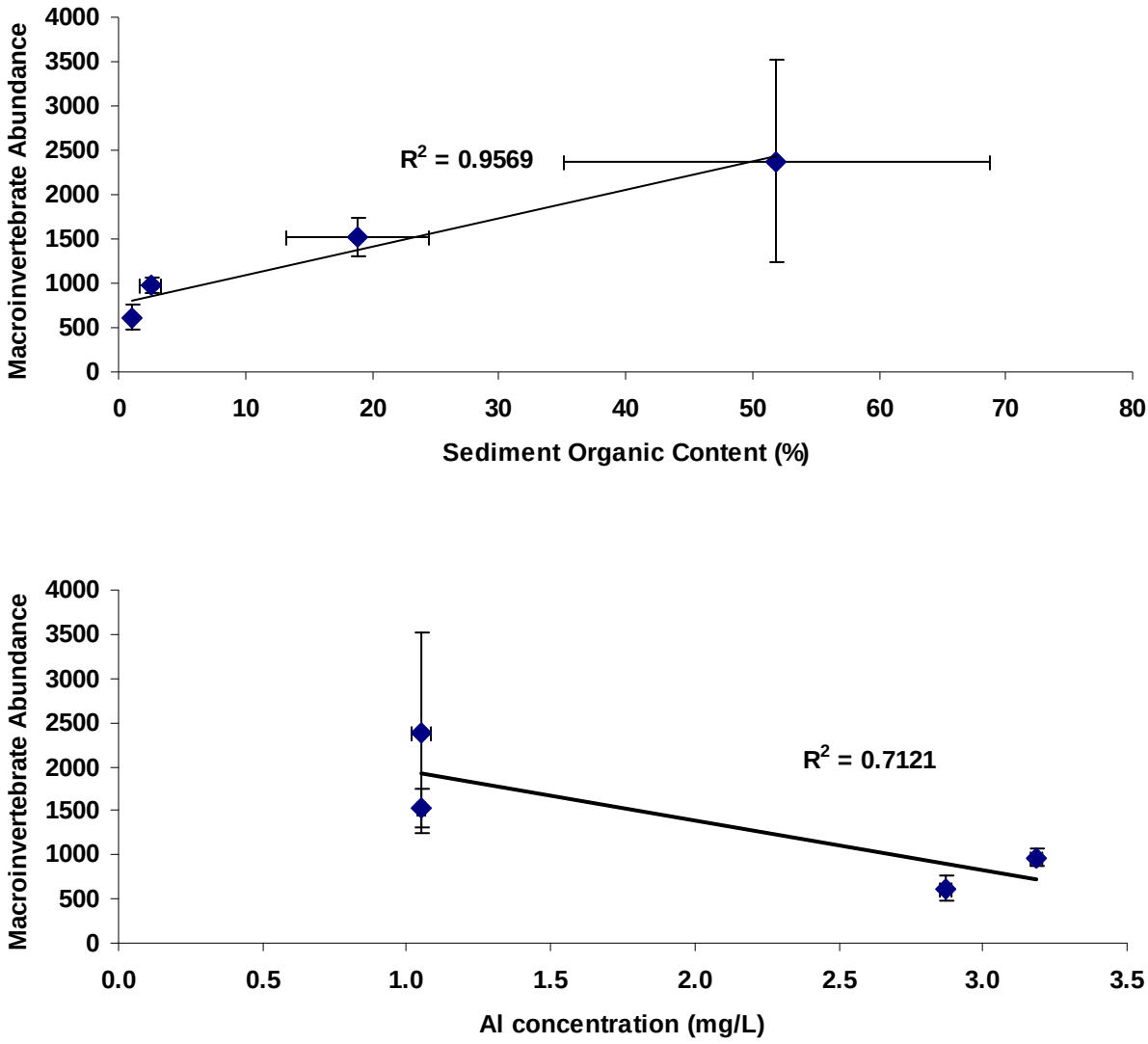

Figure 11. Correlations of macroinvertebrate abundance with sediment organic content and dissolved aluminium concentrations.

6 Discussion

Previous investigations of the Spoonbill Lakes have found lower pH in the treatment lake than in the control lake (Galeotti & McCullough, 2008; Lund *et al.*, in press), which contrasts with the present results. Comparison of current and previous data (Galeotti & McCullough, 2008) shows that the difference is in fact due to a significant decrease of pH in the control lake rather than an increase in the treatment lake. Considering that the lakes are connected via groundwater it would seem likely that this decrease would be reflected in the treatment lake if not for some moderating process. As the primary aim of mulching is to increase pH in acidified systems it seems likely that mulching is responsible for maintaining the pH in the treated lake. The noticeably lower ORP at TD is likely due to the potatoes added during the initial stages of treatment (Lund *et al.*, in press).

Dissolved metals are a typical problem with ASS, particularly Al and Fe (Sammut *et al.*, 1996). The threefold reduction in these species between control and treated lakes, considered with the apparent pH moderation in the treated lake, suggest that Sulfate Reducing Bacteria (SRB) are active in the treated lake. Previous studies (Galeotti & McCullough, 2008; Lund *et al.*, in press) found that concentrations of Fe and Al were actually higher in the treated lake. It was suggested that this was due to anaerobic conditions beneath the mulch causing the reduction and mobilisation of Fe and Al ochres, and that once the limited pool of these materials had been exhausted the effect should cease. It seems likely that this condition has been reached and that future investigations will show further increases of water quality in the treated lake.

The sixfold decrease in total nitrogen from control to treatment lake may be the result of uptake by macrophytes or plankton (Risgaard-Petersen & Jensen, 1997), or the actions of denitrifying bacteria decomposing mulch (Risgaard-Petersen & Jensen, 1997; Lund *et al.*, in press). The presence of the aerobic wetland at TD suggests that macrophytes are responsible for the decrease, particularly since macrophytes are strong competitors for ammonia (Risgaard-Petersen & Jensen, 1997) Denitrification occurs frequently in wetlands with available organic carbon and high inputs of

anthropogenic nitrogen, and is a significant factor in the reduction of eutrophication events (Seitzinger, 1988). It is likely that storm drainage and runoff causes the Spoonbill Lakes to receive significant inputs of nitrogen from fertilisers used on the reserve and surrounding suburbs, and the presence of mulch in the treatment lake makes it an ideal location for denitrification. Phosphorus levels in the lakes are likely to be similarly affected by stormwater runoff.

Macroinvertebrate species richness in the Spoonbill Lakes seems to defy logic, with significantly more species present in the untreated lake than the treated one. In general, species richness decreases as pH decreases (Stokes, 1986) and considering that mulching provides both habitat and food for a range of macroinvertebrates it is unexpected to see higher diversity in the untreated, more acidic lake. It is possible that the heterogeneity of habitat in the control lake, particularly at CN, promotes a higher diversity despite other adverse conditions such as pH and metal levels that would otherwise restrict diversity (Sommer & Horwitz, 2001). It is likely that the taxa present are those most able to cope with low pH conditions and high metal concentrations. This is supported by the presence of dragonfly nymphs (*Orthetrum caledonicum*), known to be sensitive to water quality, in the treated lake but not the control. Mulching the banks of the treated lake may in fact be responsible for the lower diversity of macroinvertebrates, as it may have reduced the complexity of habitat and promoted strong competition from a few species. The abundance of macroinvertebrates seems to follow expectations; there were far more macroinvertebrates in the treated lake than the control. Exactly what is responsible for this is not clear, however. There was no clear link between CPOM and abundance but a strong correlation suggests that sediment organic content is connected to macroinvertebrate abundance. This may be related to the feeding habits of the dominant taxa (Chironomids) which are known to burrow in sediment for organic detritus. It follows that the observed increase in *Necterosoma* abundance would be associated with increased opportunities for predation. However, considering the strong negative correlation between aluminium levels and abundance, it seems likely that the degree of sulfate reduction is indirectly a determining factor in macroinvertebrate abundance. This may in fact be the reason for sediment organic content correlating with increased abundance; high organic content in sediment

provides a rich source of labile carbon for SRB to consume, with the effect of reducing metal concentrations and increasing pH, both of which are probably more directly responsible for affecting abundance.

Benthic primary production was shown to generally increase in the presence of a jarrah substrate, suggesting that mulching should increase primary production. Phosphorus and nitrogen are generally regarded as the primary limiting nutrients in freshwater systems (Rabalais, 2002) and it is unexpected that benthic primary production would be lowest at the location with the highest measured phosphorus and nitrogen levels (**Figure 5**). It is likely that some degree of smothering was responsible for this disparity, since the site CS had virtually no vegetation and a layer of floc was observed on the periphytometers prior to retrieval.

Limitation of the study included that for sites CN, TD and TS there were large quantities of organic debris present that made macroinvertebrate sampling difficult. The net quickly filled with mulch or leaves, resulting in large samples that required subsampling, which is likely to have simplified the results. The periphytometers likely did not give an accurate representation of benthic primary production due to smothering by iron and aluminium flocs, particularly at CS and TS. Their surfaces were also heavily covered with sand tubes constructed by Chironomid larvae, which is likely to have restricted access for periphyton growth. Perhaps the use of in-situ benthic chambers would reduce this effect. It would also have been useful to have data from before any treatment was implemented as this would give a more accurate estimation of the effects of current treatments. It is difficult to define reasonable expectations for an artificial wetland that has become degraded; comparison with natural wetlands is unlikely to be realistic.

7 Conclusions

It seems that the addition of mulch, although initially decreasing water quality, has substantially improved the physical and chemical water quality in the Spoonbill treatment lake. Significant decreases in metal concentrations, nutrient loads and increased or moderated pH suggest that mulching may form the basis of a cheap and easy remediation for other acid sulfate affected wetlands. While further investigation is necessary to understand the apparent anomalies in the macroinvertebrate response there seems to be definite potential for the addition of organic material to stimulate carbon and nutrient cycling and promote a more functional wetland. While it is difficult to say what a constructed wetland should look or behave like in ecological terms, it is certain that in their current state the Spoonbill Lakes do not even meet the amenity requirements they were constructed for. Further increases in water quality may allow the introduction of fish and the return of waterfowl, which should affect the current dominance of Chironomids and incidentally reduce the local midge problem. Hopefully this report will add to the limited body of knowledge available concerning the effects of acid sulfate soils on lentic systems, and contribute to an understanding of how to remediate those effects.

8 Acknowledgements

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10 Appendices

Appendix 1: Water Chemistry 5/9/08

	CN1	CN2	CN3	CS1	CS2	CS3
pH	2.61	2.33	2.30	2.68	2.82	2.44
DO (%)	71.90	79.50	78.50	89.50	92.00	90.80
DO (mg/L)	6.91	7.63	7.57	8.55	8.84	8.78
Temp (°C)	16.55	16.65	16.64	16.96	16.73	16.67
Cond (mS/cm)	1.057	1.054	1.054	1.044	1.044	1.046
ORP (mV)	510	493	499	515	521	525
Turbidity (NTU)	0.00	0.00	0.00	0.00	0.80	0.00
Chlorophyll A (µg/L)	4.82	4.59	4.73	0.00	0.00	0.00

	TD1	TD2	TD3	TS1	TS2	TS3
pH	2.84	2.90	2.94	3.33	3.00	2.86
DO (%)	68.00	73.60	70.80	85.40	94.80	95.40
DO (mg/L)	6.48	6.90	6.69	8.14	9.00	9.14
Temp (°C)	17.56	17.39	17.46	17.50	17.47	17.34
Cond (mS/cm)	0.949	0.948	0.943	0.966	0.962	0.967
ORP (mV)	287	464	471	501	502	515
Turbidity (NTU)	0.00	0.00	0.60	0.30	0.50	1.20
Chlorophyll A (µg/L)	2.00	16.72	13.27	0.00	1.05	0.00

	Mean Values Per Site			
	CN	CS	TD	TS
pH	2.41	2.65	2.89	3.06
DO (%)	76.63	90.77	70.80	91.87
DO (mg/L)	7.37	8.72	6.69	8.76
Temp (°C)	16.61	16.79	17.47	17.44
Cond (mS/cm)	1.06	1.04	0.95	0.97
ORP (mV)	500.67	520.33	407.33	506.00
Turbidity (NTU)	0.00	0.27	0.20	0.67
Chlorophyll A (µg/L)	4.71	0.00	10.66	0.35

	Standard Error Per Site			
	CN	CS	TD	TS
pH	0.10	0.11	0.03	0.14
DO (%)	2.38	0.72	1.62	3.24
DO (mg/L)	0.23	0.09	0.12	0.31
Temp (°C)	0.03	0.09	0.05	0.05
Cond (mS/cm)	0.00	0.00	0.00	0.00
ORP (mV)	4.98	2.91	60.20	4.51
Turbidity (NTU)	0.00	0.27	0.20	0.27
Chlorophyll A (µg/L)	0.07	0.00	4.44	0.35

Appendix 2: Loss on Ignition

Sediments					
Sample	mCan (g)	mCan+Dry (g)	mCan+Burned (g)	LOI	LOI
CN1	8.51	685.57	673.85	11.72	1.73
CN2	8.52	111.19	59.68	51.51	50.17
CN3	8.52	570.24	551.77	18.47	3.29
CS1	8.44	886.15	876.20	9.95	1.13
CS2	8.54	716.41	707.61	8.80	1.24
CS3	8.39	724.46	719.44	5.02	0.70
TD1	8.43	370.90	301.94	68.96	19.03
TD2	8.92	135.10	56.92	78.18	61.96
TD3	8.54	85.39	27.91	57.48	74.80
TS1	8.88	547.75	492.69	55.06	10.22
TS2	8.95	284.10	237.83	46.27	16.82
TS3	8.49	263.75	188.66	75.09	29.42

CPOM from Macroinvertebrate samples					
Sample	mCan (g)	mCan+Dry (g)	mCan+Burned (g)	LOI	LOI
CN1	4.95	20.84	6.99	13.85	87.16
CN2	4.95	37.71	10.81	26.90	82.11
CN3	4.83	7.57	4.86	2.71	98.91
CS1	4.81	5.24	4.81	0.43	100.00
CS2	4.97	7.19	4.97	2.22	100.00
CS3	4.80	5.65	4.80	0.85	100.00
TD1	4.78	53.90	5.72	48.18	98.09
TD2	4.81	18.14	5.29	12.85	96.40
TD3	4.72	12.32	4.88	7.44	97.89
TS1	4.68	22.97	4.92	18.05	98.69
TS2	4.78	51.75	8.30	43.45	92.51
TS3	4.69	22.45	4.90	17.55	98.82

FPOM from Macroinvertebrate samples					
Sample	mCan (g)	mCan+Dry (g)	mCan+Burned (g)	LOI	LOI
CN1	5.04	8.58	7.08	1.50	42.37
CN2	5.05	13.02	8.25	4.77	59.85
CN3	4.95	5.40	5.01	0.39	86.67
CS1	5.00	13.80	13.61	0.19	2.16
CS2	5.01	20.94	20.50	0.44	2.76
CS3	5.05	30.57	30.24	0.33	1.29
TD1	5.05	7.05	5.17	1.88	94.00
TD2	5.09	6.62	5.32	1.30	84.97
TD3	5.03	5.69	5.13	0.56	84.85
TS1	5.07	5.58	5.10	0.48	94.12
TS2	5.05	5.96	5.12	0.84	92.31
TS3	5.04	5.89	5.12	0.77	90.59

Periphyton LOI (Glass Control)

Sample	mCan (g)	mCan+Dry (g)	mCan+Burned (g)	LOI	LOI
CN1	14.58	14.77	14.67	0.09	48.57
CN2	14.07	14.18	14.13	0.05	42.36
CN3	14.13	14.19	14.16	0.03	45.66
CS1	15.26	15.96	15.82	0.13	19.16
CS2	14.29	18.18	17.61	0.57	14.73
TD1	15.65	16.04	15.74	0.30	77.10
TD2	14.28	14.64	14.41	0.23	62.43
TD3	13.74	14.16	13.86	0.30	70.55
TS1	13.83	14.24	13.99	0.25	62.68
TS3	15.24	15.72	15.46	0.26	53.25

Periphyton LOI (Jarrah)

Sample	mCan (g)	mCan+Dry (g)	mCan+Burned (g)	LOI	LOI
CN1	15.22	15.99	15.64	0.35	46.18
CN2	15.40	16.67	16.10	0.57	44.84
CN3	15.11	16.17	15.71	0.46	43.62
CS1	14.41	16.97	16.37	0.59	23.28
CS2	14.80	16.42	16.19	0.23	14.35
TD1	14.53	15.01	14.67	0.35	71.00
TD2	14.24	15.03	14.48	0.55	69.59
TD3	15.19	15.68	15.37	0.31	63.32
TS1	14.35	15.04	14.56	0.48	69.84
TS3	14.07	14.49	14.26	0.23	55.91

Sediments

Sample	Mean LOI (%)	Std
CN	2.51	0.78
CS	1.03	0.17
TD	51.93	16.86
TS	18.82	5.63

Macro CPOM

Sample	Mean LOI (%)	Std
CN	89.39	4.97
CS	100.00	0.00
TD	97.46	0.53
TS	96.67	2.08

Macro FPOM

Sample	Mean LOI (%)	Std
CN	62.96	12.88
CS	2.07	0.43
TD	87.94	3.03
TS	92.34	1.02

Glass	mean	se
CN	45.53	1.80
CS	16.94	1.81
TD	70.03	4.24
TS	57.96	3.85

Jarrah	mean	se
CN	44.88	0.74
CS	18.81	3.64
TD	67.97	2.36
TS	62.87	5.69

Appendix 3: Nutrients and Metals

Table 1. Nutrient and metal concentrations in lake waters. All values mg L^{-1}.

Sample Code	NPOC	NH_4-N	NO_x-N	Total N	PO_4-P	Total P
Reporting Limit	<0.5	<5	<2	<50	<2	<10
CN1	1.4	1255	73	1336	8	<20
CN2	1.7	1207	72	1344	4	<20
CN3	1.5	1125	77	1206	5	<20
CS1	0.8	1171	50	1210	3	<20
CS2	1.1	1358	208	1614	2	<20
CS3	0.8	1269	37	1347	2	77
TD1	1.1	164	8	411	2	<20
TD2	1.3	151	37	288	4	<20
TD3	1.3	138	39	281	5	39
TS1	0.8	74	1	264	2	24
TS2	0.6	104	12	296	2	25
TS3	1.0	182	65	250	2	<20

Sample Code	Al	As	B	Ca	Cd	Co	Cr
Reporting Limit	<0.1	<0.05	<0.05	<0.05	<0.01	<0.01	<0.01
CN1	3.2	<0.05	<0.05	56	<0.01	<0.01	<0.01
CN2	3.2	<0.05	<0.05	55	<0.01	<0.01	<0.01
CN3	3.1	<0.05	<0.05	55	<0.01	<0.01	<0.01
CS1	2.9	<0.05	<0.05	53	<0.01	<0.01	<0.01
CS2	2.9	<0.05	<0.05	53	<0.01	<0.01	<0.01
CS3	2.8	<0.05	<0.05	51	<0.01	<0.01	<0.01
TD1	1.1	<0.05	<0.05	58	<0.01	<0.01	<0.01
TD2	1.0	<0.05	<0.05	58	<0.01	<0.01	<0.01
TD3	1.0	<0.05	<0.05	59	<0.01	<0.01	<0.01
TS1	1.1	<0.05	<0.05	58	<0.01	<0.01	<0.01
TS2	1.0	<0.05	<0.05	58	<0.01	<0.01	<0.01
TS3	1.0	<0.05	<0.05	57	<0.01	<0.01	<0.01
blank acid	<0.1	<0.05	<0.05	<0.05	<0.01	<0.01	<0.01

Sample Code	Cu	Fe	Hg	K	Mg	Mn	Na
Reporting Limit	<0.05	<0.05	<0.1	<0.5	<0.1	<0.5	<0.05
CN1	<0.0	14.7	<0.1	6.5	7.6	0.05	58
CN2	<0.0	15.1	<0.1	6.3	7.7	0.05	57
CN3	<0.0	14.1	<0.1	6.2	7.4	0.05	57
CS1	<0.0	10.9	<0.1	6.0	7.3	0.04	56
CS2	<0.0	10.9	<0.1	5.9	7.3	0.04	55
CS3	<0.0	10.8	<0.1	5.8	7.1	0.04	53
TD1	<0.0	5.64	<0.1	11.1	10.1	0.05	61
TD2	<0.0	5.71	<0.1	11.5	10.2	0.05	62
TD3	<0.0	5.49	<0.1	11.4	10.2	0.05	63
TS1	<0.0	2.60	<0.1	12.0	10.5	0.05	62
TS2	<0.0	2.55	<0.1	12.0	10.4	0.05	62
TS3	<0.0	2.50	<0.1	11.9	10.2	0.05	61
blank acid	<0.0	<0.05	<0.1	<0.5	<0.1	<0.01	<0.5

Sample Code	Ni	Pb	S	Se	Zn
Reporting Limit	<0.1	<0.5	<0.5	<0.2	<0.05
CN1	<0.05	<0.1	74	<0.2	0.06
CN2	<0.05	<0.1	76	<0.2	0.10
CN3	<0.05	<0.1	73	<0.2	0.025
CS1	<0.05	<0.1	70	<0.2	0.08
CS2	<0.05	<0.1	70	<0.2	0.025
CS3	<0.05	<0.1	68	<0.2	0.08
TD1	<0.05	<0.1	66	<0.2	0.025
TD2	<0.05	<0.1	65	<0.2	0.06
TD3	<0.05	<0.1	67	<0.2	0.025
TS1	<0.05	<0.1	67	<0.2	0.025
TS2	<0.05	<0.1	67	<0.2	0.025
TS3	<0.05	<0.1	64	<0.2	0.07
blank acid	<0.05	<0.1	<0.5	<0.2	<0.05

Appendix 4: Macroinvertebrates

Table 2. Raw macroinvertebrate data.

	AgraptocorixaN	AllodessusA	BerosusA	BerosusL	CaenidaeL	CalanoidaN	ChironomidaeP	ChironominaeL	Chironomus alternansL	Chironomus occidentalisL	Coleopynia pruinosaL	CulexL	CulexP
CN1	0	0	0	0	0	0	4	0	0	604	0	0	0
CN2	0	1	0	0	0	2	28	0	80	864	0	1	0
CN3	0	0	0	0	0	0	0	0	256	768	0	0	0
CS1	0	0	0	0	0	0	1	8	53	186	0	0	0
CS2	1	0	0	0	0	0	0	0	196	428	0	1	0
CS3	0	0	0	0	1	0	12	0	56	600	4	0	0
TD1	0	0	0	0	0	0	28	0	0	324	0	6	0
TD2	0	0	0	0	0	0	48	0	0	1,472	0	144	36
TD3	0	0	0	0	0	0	0	0	0	4,080	40	10	0
TS1	0	0	4	16	0	0	112	0	0	96	0	0	0
TS2	0	0	0	12	0	0	172	0	0	168	0	0	0
TS3	4	0	0	12	0	0	112	0	0	384	0	0	0
Mean													
CN	0	0.33	0	0	0	1	11	0	112	745	0	0.3	0
CS	0.33	0	0	0	0	0	4.3	2. 7	102	405	1	0.3	0
TD	0	0	0	0	0	0	25.3	0	0	1,959	13	53	12
TS	1.3	0	1.3	13	0	0	132	0	0	216	0	0	0

	DaphniaN	Diaprepocoris barycephalaN	Diaprepocoris personataN	DipteraP	FerrissiaN	HydracarinaN	IsotomidaeN	KatiannaN	K.martiniL	L.lanceolatusA	L.lanceolatusL	LepidopteraL	MesoveliaN	MicronectaN
CN1	0	31	0	0	0	1	0	1	24	2	4	0	0	20
CN2	0	0	0	4	3	0	0	0	8	2	2	1	0	1
CN3	0	0	0	0	0	0	0	0	16	4	4	0	0	4
CS1	0	0	0	0	0	0	0	0	1	0	0	0	0	0
CS2	0	0	1	0	2	0	0	0	0	0	0	0	0	0
CS3	0	0	0	0	1	0	0	0	0	0	0	0	0	0
TD1	0	0	0	0	0	0	1	0	0	0	0	0	0	0
TD2	0	0	0	0	0	0	4	0	0	0	0	0	4	0
TD3	0	0	0	0	10	0	0	0	0	0	0	0	0	0
TS1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
TS2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
TS3	0	0	0	16	0	0	0	0	0	0	0	0	0	0
Mean														
CN	0	10	0	1.3	1	0	0	0.33	16	2.7	3	0.3	0	8
CS	0	0	0.3	0	1	0	0	0	0.33	0	0	0	0	0
TD	0	0	0	0	3	0	1.67	0	0	0	0	0	1	0
TS	0	0	0	5.3	0	0	0	0	0	0	0	0	0	0

	NecterosomaA	NecterosomaL	Orthetrum caledonicumL	OstracodaN	ParatrichocladiusL	Polypedilum nubiferL	Paralimnophyes pullulusL	RhantusA	RhantusL	ScirtidaeL	SigaraN	TanypodinaeL	T.fuscithoraxL	Unknown	VeliidaeN	**Species Richness**	**Tot6al Abundance**
CN1	1	73	0	0	0	12	0	0	1	0	3	0	12	0	0	**15**	**286**
CN2	0	21	0	1	0	16	4	1	1	5	0	0	0	5	0	**21**	**129**
CN3	0	4	0	0	0	0	0	0	0	0	4	0	0	0	12	**9**	**68**
CS1	0	20	0	0	0	0	1	0	0	0	1	0	60	0	0	**9**	**165**
CS2	0	38	0	0	0	0	0	0	0	0	1	0	92	0	0	**9**	**265**
CS3	0	33	0	0	0	0	0	0	0	0	0	0	44	0	0	**8**	**155**
TD1	1	38	1	0	0	0	4	0	0	1	0	12	100	1	0	**12**	**316**
TD2	4	144	0	0	80	0	32	0	0	0	0	0	192	0	0	**11**	**908**
TD3	20	140	0	0	0	0	0	0	0	0	0	0	160	0	0	**7**	**630**
TS1	24	120	0	0	0	0	0	0	0	0	0	0	768	0	0	**7**	**1,800**
TS2	20	132	0	0	0	0	0	0	0	0	0	0	1,384	0	0	**6**	**3,052**
TS3	0	152	4	0	0	0	0	0	0	0	0	0	864	0	0	**8**	**2,056**
Mean																	
CN	0.3	33	0	0.3	0	9	1	0.3	0.67	1.7	2.3	0	4	2	4		161
CS	0	30	0	0	0	0	0	0	0	0	0.7	0	65.3	0	0		195
TD	8.3	107	0.33	0	27	0	12	0	0	0.333	0	4	151	0	0		618
TS	15	135	1.3	0	0	0	0	0	0	0	0	0	1,005	0	0		2,303