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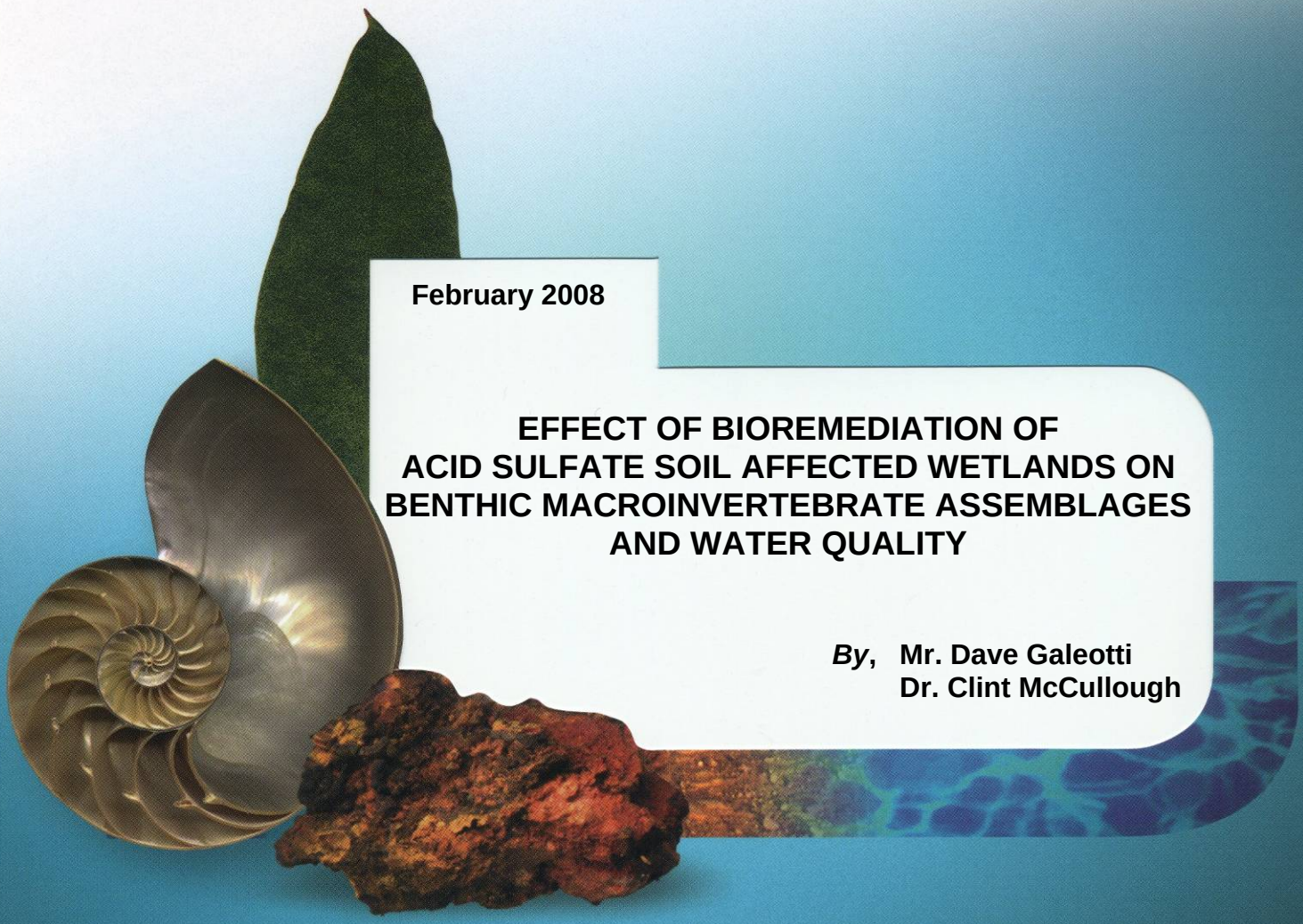

February 2008

EFFECT OF BIOREMEDIATION OF ACID SULFATE SOIL AFFECTED WETLANDS ON BENTHIC MACROINVERTEBRATE ASSEMBLAGES AND WATER QUALITY

By, **Mr. Dave Galeotti**
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Prepared for,
City of Stirling

Mr. Dave Galeotti BEnvSc,

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Frontispiece

Mr. Dave Galeotti sampling water chemistry in North Lake, Spoonbill-Shearwater Reserve, City of Stirling.

This document should be referenced as follows.

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1. Contents

2. Background

Acid sulfate soils[1] (ASS) are a global land management problem, and have been described as 'the nastiest soils on earth' (Dent and Pons 1995). Their causes and consequences have been researched for hundreds of years (Dent and Pons 1995; Teakle and Southern 1937). The resulting land management options have been studied and advised (with advice frequently unheeded) to those with the power to prevent environmental disaster (Gilkes 2004; Lines-Kelly 2004). ASS can cause groundwater contamination (Hinwood, Horwitz *et al.* 2006), ruin important marine breeding grounds (Russell and Helmke 2002) and valuable agricultural land (Lin, Melville *et al.* 1995), and cause general environment damage (Sammut, Melville *et al.* 1995).

ASS are normally found in low lying coastal areas which indirectly have had an affect on millions of people in countries including China, the Netherlands and Australia (Lin, Melville *et al.* 1995; van Duinen, Brock *et al.* 2003). Tens of thousands of years ago iron sulfide minerals were deposited in mainly coastal marine environments. These minerals were then overlain by alluvial runoff containing high amounts of organic matter. The sulfide rich soils then generally remain at depth, saturated and in an anoxic state, below the groundwater level. If groundwater levels fall or the soil is disturbed the sulfides oxidise and release sulfuric acid, iron and toxic heavy metals into the surrounding soils and groundwater (Sammut and Lines-Kelly 2000). The effects from this can be devastating to the urban and natural environment for a very long time, possibly in excess of 100 years (Dent & Pons 1995).

In Western Australia, since European arrival, around 70% of wetlands on the Swan Coastal Plain (SCP) have been filled or drained for agricultural or urban development (Davis and Froend 1999). It is estimated that 10% of the SCP may be affected by ASS, especially wetland areas around Perth and the Peel-Harvey estuary (DoE 2006). When coupled with declining rainfall and hot/dry summers, continuing population growth and growing dependency on groundwater resources (Appleyard, Angeloni *et al.* 2006), further problems may arise if identification and management advice for ASS is not heeded.

[1] The use of the word sulfate within this paper is synonymous with the word sulphate.

In December 2001, a problem with ASS contaminated groundwater arose in the Perth (Western Australia) suburb of Stirling, which includes the Spoonbill Lakes within Spoonbill–Shearwater Reserve. Commercial and domestic bores were sampled in Stirling and surrounding the lakes and those that were less than pH 5.5 were found to have high levels of arsenic and heavy metals. Many samples had levels around pH 3 (WRC 2002). There were a number of factors thought to cause this ASS event including lowering of the water table due to low rainfall, excessive groundwater extraction and excavation and stockpiling of soils for residential development (WRC 2002).

The Stirling ASS groundwater contamination resulted in some residents' gardens dying. The Department of Environment and Department of Health advised residents in affected areas not to drink the bore water or eat home-grown fruit and vegetables (irrigated by bore water) because of the potential health risks associated with the arsenic and heavy metals present (Halloran 2004; WRC 2002). The effect that ASS has had on the environment is also observable at the Spoonbill–Shearwater Reserve itself. There are many trees dying and orange/brown staining (iron hydroxide) around the lakes edge and on trees and playground equipment from bore water reticulation. This can be contributed to ASS, as the iron hydroxide is released due to the higher acidity in the groundwater. Spoonbill Lake has become ultra-oligotrophic, causing a lack of aquatic and terrestrial wildlife (Melissa Mazzella, Department of Environment, unpublished data).

A water quality treatment system was constructed at the northern end of the island in the south Spoonbill Lake in mid-2006. It was designed to reverse the effects of ASS water contamination by reducing sulfate, metals and arsenic concentrations, and to produce slightly anaerobic pH-neutral water (McCullough 2007). During construction about one tonne of potatoes were scattered along the banks and littoral zone of the northern side of the south lake directly opposite the treatment system. In August 2006 mulch was spread over about 100 m of the banks opposite the treatment system. These organic materials were used to amend the low organic carbon content of the water and sediments of the northern section of this lake. The treatment system became fully operational in November 2006. To further enhance the treatment system about 2 500

wetland plants of *Baumea articulata* and *Schoenoplectus validus* were planted around the margin of the northern lake in January and March 2007.

The aim of this project was to assess the effect of ASS on water quality and benthic macroinvertebrate communities, before and after establishment of the water quality treatment system. This was achieved by comparing data from the north Spoonbill Lake and southern section of the south Spoonbill Lake as reference lakes against the treatment lake, and with other relevant research (including Appleyard, Wong *et al.* 2004; Balla and Davis 1995; Sommer and Horwitz 2001). The data obtained is used to discuss the changes in the lakes' physico-chemical properties and macroinvertebrate communities.

3. Site Description & Methods

The Spoonbill–Shearwater Reserve consists of a wide range of vegetation including native species such as *Melaleuca, Acacia* and *Eucalyptus,* a number of introduced species and vast areas of grass turf up to the edges of the lakes. The northern end of the north lake (study site A) is surrounded by a moderate density of trees, shrubs and rushes up to the waters edge, has loose grey soil on the banks and contains a large stormwater drain outlet (see Figure 1 for vegetation cover and site locations). Site B at the southern end of the north lake has sparse vegetation and the banks consist of orange brown soil. The treatment site C, at the northern end of the south lake, is similar to site A in vegetation cover, although it has more compacted grey-orange soil. At the southern end of the south lake site D has moderate vegetative cover with orange soils. There are another two stormwater drain outlets at the southern end of the south lake.

Sampling took place at mid-morning on the 25th of October 2006 (mid spring) and the 9th of March 2007 (late summer). There were 12 sites with three replicates at the north and south of each of the two lakes, as shown in Figure 1. The known poor water quality and apparent homogeneity within the sites allowed the three samples of macroinvertebrates and on site water testing from the four ends of the lakes to be combined in the laboratory. Water samples for laboratory analysis were taken at the same four sites at the northern and southern ends of both lakes.

When sampled in October 2006 the water level was almost at its peak with the maximum water depth around 2 m near site B. By comparison, when sampling occurred in March 2007 the water level had fallen approximately 1 m and was very shallow, <30 cm at site C. All study sites at both sampling dates had large amounts of filamentous algae present.

The Spoonbill Lakes are a shallow surface expression of an unconfined aquifer, the Gnangara Mound (WRC 2002), set in well-draining sandy soils. The groundwater flows in a west-south-west direction, so the north lake will not reflect a change in water quality attributed to the south lake. For the purpose of this study the north lake

at Spoonbill-Shearwater Reserve was used as a reference and the south lake the treatment lake. The reference lake was chosen to reduce variables recorded in the treatment lake attributed to changes in natural or seasonal processes, including inflow from stormwater drains.

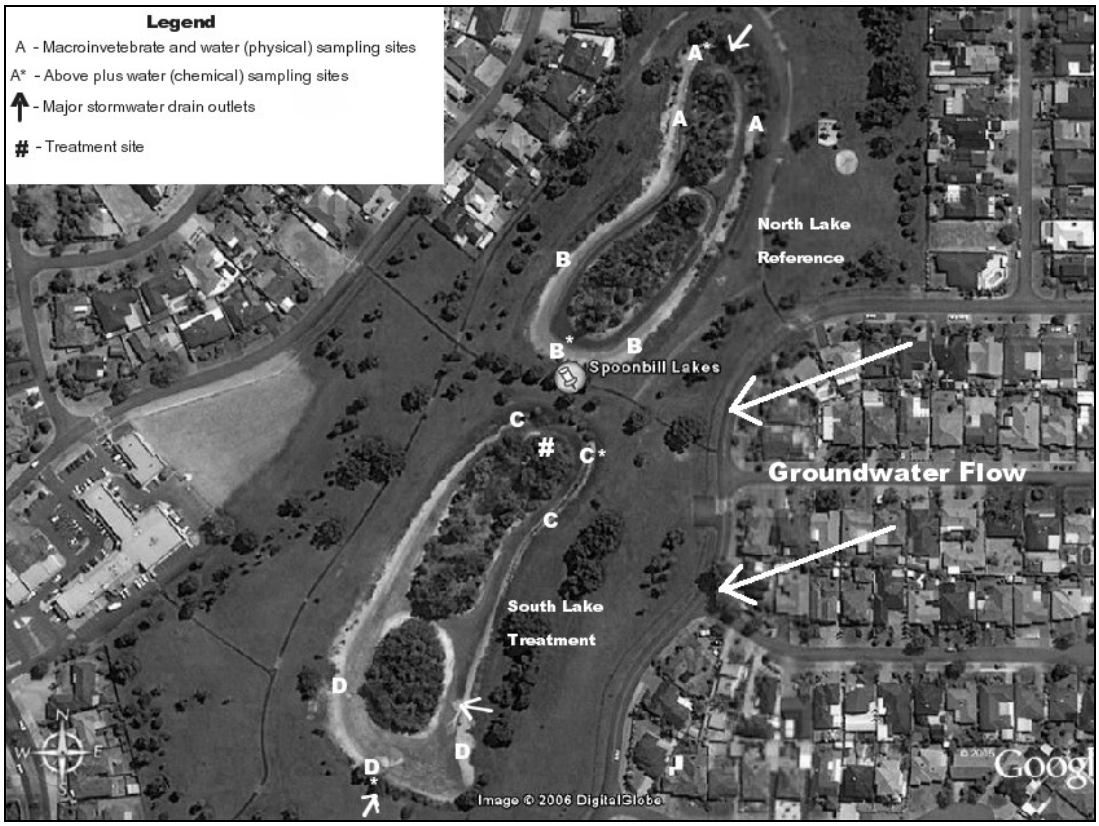

Figure 1. Satellite image of Spoonbill-Shearwater reserve, Stirling, Western Australia, with sampling sites and groundwater movement indicated (adapted from Google Inc 2006).

3.1 Water quality sampling

3.1.1 Physical sampling

The on-site water testing and collection of water samples took place immediately before macroinvertebrate sampling to minimise disturbance to the sediments which would give inaccurate results. A Hydrolab DataSonde 4a multiprobe was used to take on site measurements, with data recorded to a connected Personal Digital Assistant (PDA). Readings were taken approximately one metre from the water's edge at a depth of 15–30 cm at all sites. The parameters recorded on site were: pH, water temperature, conductivity, dissolved oxygen, oxygen-reduction potential, turbidity and chlorophyll a. The multiprobe was calibrated for pH and dissolved oxygen in the Wetlands Laboratory prior to going on site. For pH the built-in auto-calibration two-point method was used with pH 4 and 7 standards; dissolved oxygen was also calibrated using the auto-calibration in the multiprobe.

3.1.2 Chemical sampling

The water samples for the laboratory analysis were collected in detergent washed and acid-rinsed 250 mL polyethylene bottles. The samples were filtered with a 0.45 µm Glass Fibre – Coarse (GF/C) filter paper. The filtering took place as collected on site to eliminate changes to the chemical structure, for example arsenic can be lost as it coprecipitates with iron oxyhydroxides (Appleyard, Angeloni *et al.* 2006). The samples were then taken to a 3^{o}C coolroom at Edith Cowan University (ECU) Joondalup campus.

Prior to analysis the filtered samples were acidified with 1% analytical grade HCl. The samples were analysed for metal concentrations at the ECU Joondalup's School of Natural Sciences Analytical Services using Inductively Coupled Plasma Atomic Emission Spectrophotometry (ICP-AES). The water samples were analysed for 19 metals and metalloids: Al, As, B, Ca, Cd, Co, Cr, Cu, Fe, Hg, K, Mg, Mn, Na, Ni, Pb, S, Se, Zn.

3.2 Macroinvertebrate sampling

For the macroinvertebrate samples, an area of one square metre within the littoral margin of the lakes was swept through twice towards the shore, using up to four strokes each time. Due to the sweep net size (250 µm) and the amount of filamentous algae and leaf litter in some areas, some macroinvertebrate sampling was restricted to one sweep. Once most of the water had drained from the net, the contents were transferred to a labelled plastic bag and 100% ethanol added to preserve the organisms until sorting and identification was carried out. Bags were then placed into an esky and taken to ECU Joondalup campus for storage in a coolroom at ~3^{o}C.

In the Wetlands Laboratory at ECU the combined samples, from each end of the two lakes, were placed into a photographic developer's tray to remove large objects and to elutriate the sand, which was discarded. The fine particulate organic matter (FPOM, 500 µm to 2 mm) and coarse particulate organic matter (CPOM, >2 mm) was separated using a set of nested sieves. The amount of filamentous algae meant that water needed to be sprayed through under pressure to break it down. This also released any macroinvertebrates caught within the algae and organic material and reduce the amount of suspended material that would otherwise interfere with sorting macroinvertebrates.

Once sieved, the CPOM was returned to the cleaned tray with water and examined for macroinvertebrates. Macroinvertebrates that were found were placed into small vials containing 70% ethanol, to be formally identified when the sorting had finished. Some samples were sub-sampled for Chironomidae larvae (non-biting midges) due to the high numbers found. The FPOM was examined in a Bogarov tray using a dissecting microscope. Some samples were also sub-sampled for chironomid larvae. The sieving and sorting of CPOM and FPOM was repeated for samples from each end of both of the lakes. When sorting had finished identification took place by using taxonomic keys, initially with some help from Wetlands Laboratory Research Assistants.

3.3 Quantifying organic carbon content

After sieving and sorting, the remaining material was returned to their respectively labelled plastic bags, separated as CPOM and FPOM and stored in the coolroom. To

estimate the amount of potential food available to benthic organisms, the organic carbon content was measured using the loss on ignition (LOI) technique on both the CPOM and FPOM. The sample remnants were individually placed into clean and labelled pre-weighed foil trays and dried in an oven at 105^{o}C for 24 hours. Once oven-dried the samples were weighed, and then placed into a furnace oven at 550^{o}C for 24 hours to be reduced to ash. After the samples cooled sufficiently, they were weighed once more. Thus the weight of the organic carbon was the difference of the dry sample and the ashed sample, minus tray weights.

4. Results

The results from data collected were tabulated and graphed using Microsoft Excel spreadsheets. Where mentioned, water quality guidelines are derived from ANZECC & ARMCANZ (2000) which are aimed at sustaining environmental values. Macroinvertebrate abundance between the different sites was compared using Chi-Square with a probability (*P*) factor of >95% or <0.05. Statistical formulae and methods followed Brower, Zar *et al* (1998). The hypothesis tested was that the sites were homogenous in respect to habitat and water quality. Macroinvertebrate diversity was calculated with the Shannon Diversity Index[2] using logarithmic base 2. Results of 0 implied little or no taxonomic diversity to >7 implying high diversity.

Results

For the remainder of the paper the sampling dates will be referred to as SD1, Sampling Date One 25th October 2006, and SD2, 9th March 2007. The sites referred to as 'Sites A, B, & D' are reference sites as shown in Figure 1, and 'Site C' is the treatment site.

[2] The Shannon Diversity Index is sometimes (imprecisely) referred to as the Shannon-Weaver or Shannon-Wiener Index Brower JE, Zar JH, von Ende CN (Eds) (1998) 'Field and laboratory methods for general ecology (4th edn).' (WCB/ McGraw - Hill: Boston) .

4.1 Physical water properties

Table 1 lists the full range of data recorded for all sites and at both sampling dates. The physical water properties showed that pH was consistently low at around pH 3 throughout the lakes, except for Site C at SD2 which was considerably less acidic. Site C at SD2 also had comparatively lower conductivity and ORP readings and had a high turbidity reading. Overall, there was less variation for dissolved oxygen concentration at SD2 than SD1. Chlorophyll a concentration was low in general, although slightly higher levels were found at SD2. Interestingly, temperature readings generally increased from north to south, even though there was only about 45 minutes between first and last sampling times.

Table 1. The physical condition of the water at the end of each lake at Spoonbill – Shearwater Reserve. Averages given for three measurements taken at each end of the lakes on two sampling dates. The treatment site (C) is shaded grey, the other sites (A,B,D) are reference sites.

Date	Site	Temperature °C	pH	Conductivity mS/cm	DO %	DO mg/L	ORP mV	Turbidity NTU	Chlorophyll a ug/L
25/10/2006	A	21.0	3.20	1.27	53.1	4.71	457	16.5	2.20
25/10/2006	B	21.5	3.20	1.27	72.2	6.33	458	16.1	0.72
25/10/2006	C	22.9	3.15	1.56	97.0	8.29	476	18.7	0.73
25/10/2006	D	22.1	3.03	1.64	99.6	8.64	506	16.2	0.00
9/03/2007	A	23.0	3.10	1.43	79.9	6.82	426	32.5	5.75
9/03/2007	B	25.1	2.95	1.52	85.6	7.02	461	19.3	1.45
9/03/2007	C	25.9	4.90	1.22	64.6	5.22	76	299.5	2.81
9/03/2007	D	27.0	2.81	1.81	91.8	7.28	462	66.3	4.52

4.2 Water chemistry

All metals that were found above detectable limits are shown in alphabetical order in Figure 2, except for Pb and Zn. Pb was only found at Site C SD2 and was nearly 5 times the guideline limit (ANZECC & ARMCANZ 2000), while Zn was found only at Site A on both occasions, but within the ANZECC/ARMCANZ guidelines. Over both sampling dates, Al concentrations were between 5 and 50 times higher than 95% environmental protection trigger values, and Fe concentrations were 30 to 90 times higher (ANZECC & ARMCANZ 2000). K concentrations at Site C were higher than the others at SD1 and lowest at SD2. The south lake consistently had levels of Mn just under the guideline limit, while the north lake was just over. Full details for all results are in **Error! Reference source not found.**.

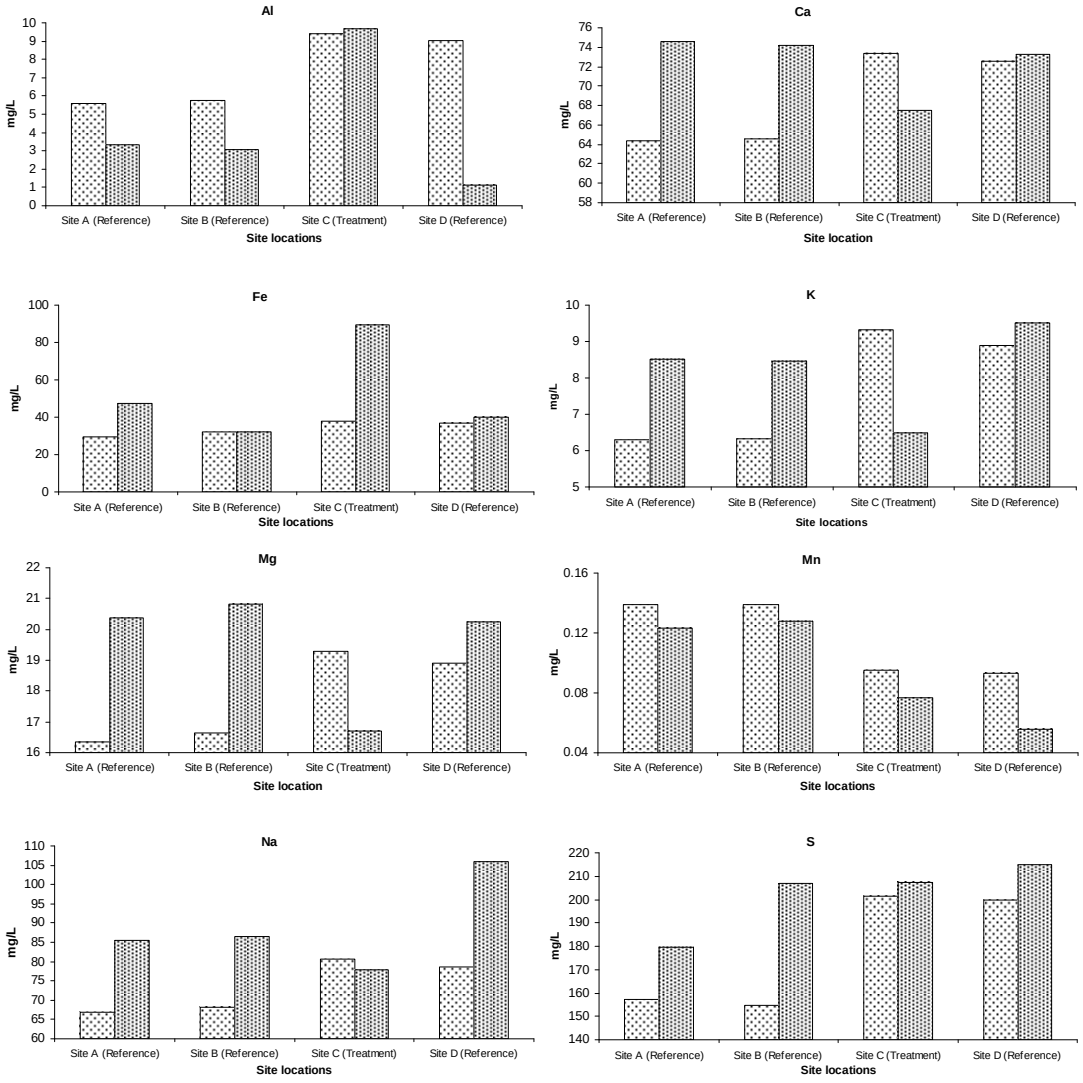

Figure 2. Water chemistry results of metal concentrations found above detectable levels from Spoonbill Lakes on both sampling dates. Note: Scales vary for each graph and not all y-axis start at zero. The legend at right applies to each graph above.

4.3 Macroinvertebrate sampling

The size of the macroinvertebrates found varied from approximately 0.1 mm to over 5 mm in length. All together nine species and 2 017 individuals were encountered over the four sites and two sampling dates (Figure 3). The most common order across both sampling dates were the Diptera at just over 95%, with over 92% of these belonging to the Chironomidae family (non-biting midges). Coleoptera were the next most abundant with a total of only 83 individuals, the bulk of these were *Necterosoma*

sp. larvae (Dytiscidae). Full macroinvertebrate data is in **Error! Reference source not found.**.

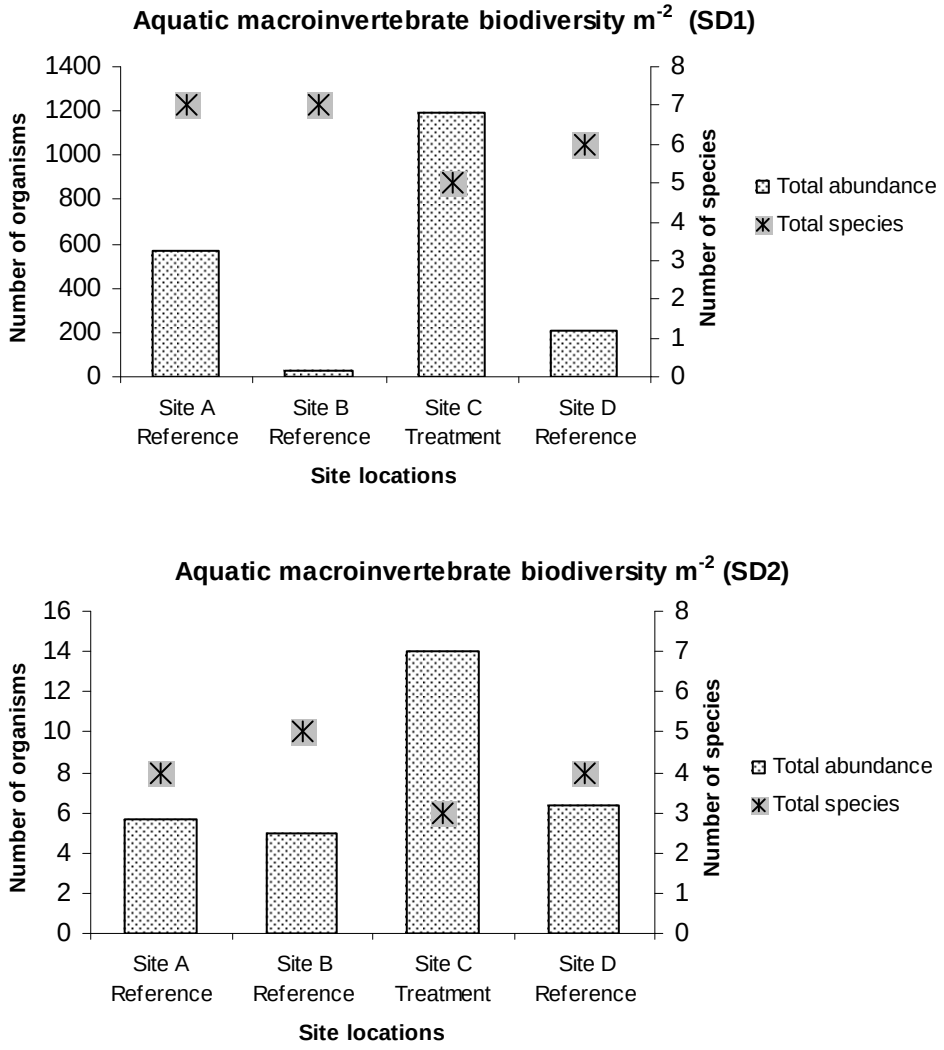

Figure 3. Macroinvertebrate biodiversity measured by abundance and species richness at reference sites (A, B and D), and treatment site (C) at Spoonbill Lakes. Note: there is a big difference in scales for abundance; species richness scale is the same for both graphs.

The resulting statistical analysis figures for abundance were to be expected for SD1, which showed there was a significant difference ($P < 0.05$) in abundance between the sites. Sampling at SD2 was quite different, with comparatively low abundance but similar richness. The low abundance figures were reflected in the statistic results indicating no significant difference between sites.

Figure 4 shows the different proportions of families present at Spoonbill Lakes. There is a noticeable decline in the domination of Chironomidae larvae at SD2, but an increase in Ceratopogonidae larvae (biting midges). There is an overall increase in Coleoptera representation at all sites at SD2. The diversity (Table 2) at SD1 is shown to be relatively low, with a reasonable increase at SD2 for all sites except Site C which had only a small increase.

Table 2. Results of Shannon Diversity Index for macroinvertebrates at Spoonbill Lakes.

	Site A Reference	Site B Reference	Site C Treatment	Site D Reference
SD1 - 25/10/2006	0.8	1.3	0.7	1.2
SD2 - 9/03/2007	1.6	2.1	1.1	1.7

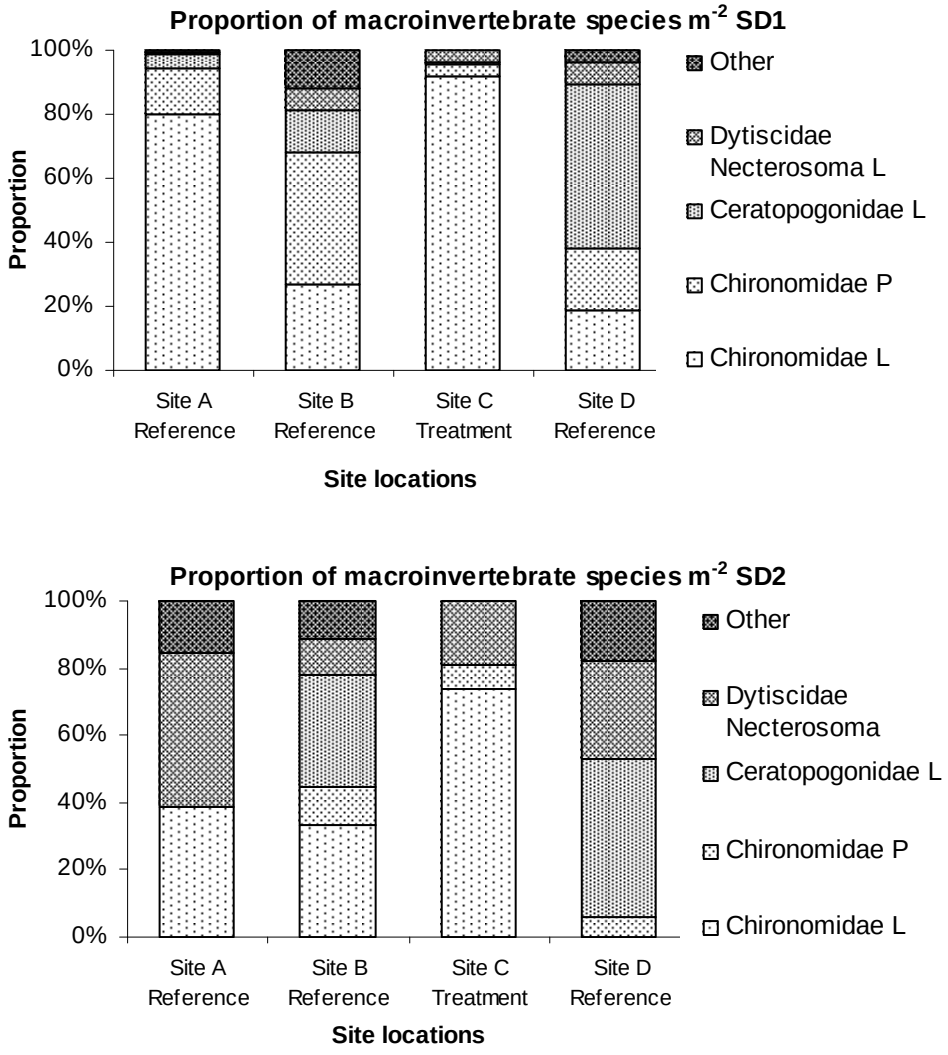

Figure 4. Spoonbill Lakes' macroinvertebrate diversity shown by the proportion of each species found per m^{-2} at two sampling dates. For clarity, species containing <6 individuals each have been grouped into the 'Other' category.

4.4 Carbon content

The effect of the surrounding vegetation corresponds with the amount of carbon found at each site (Figure 5). Site C, with the additional mulch, had the highest amount of total organic carbon content. At SD1 Site C had 65% more than Site A and more than double Sites B and D. At SD2 Site C had 46%, 54% and 62% higher than Sites B, D and A respectively. All LOI results are in Appendix 3.

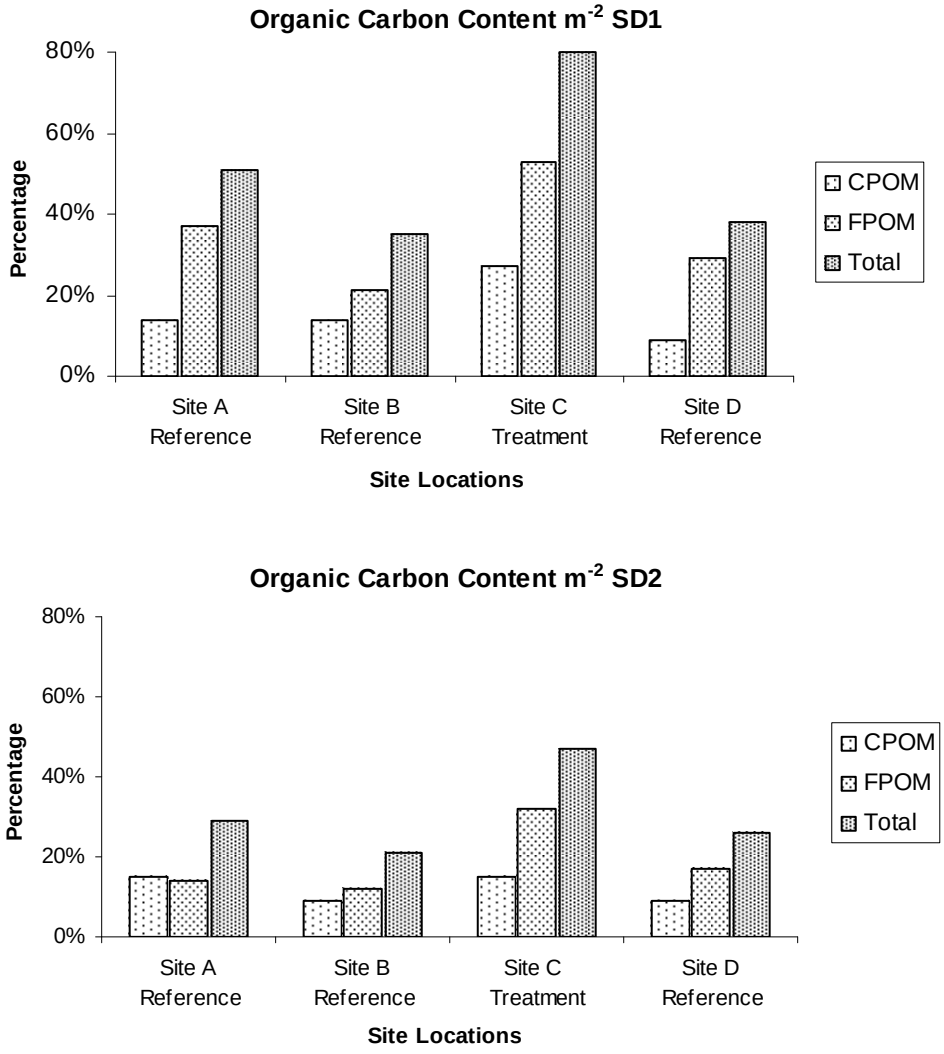

Figure 5. Percentage of organic carbon content in sweep net samples (m^{-2}) for each site at Spoonbill Lakes.

5. Discussion

The pre-treatment results (SD1) reflect water quality measurements from the Spoonbill Lakes in early spring 2001 (Melissa Mazzella, DoE, unpublished data) and during 2002 (Appleyard, Wong *et al.* 2004), showing that water quality has not improved greatly since. The physical properties of the water in the lakes also show only minor difference prior to the treatment system becoming operational. However, the treatment site post-treatment system commencement shows some marked changes, as follows. An increase in pH shows a move towards more typical levels found on the Swan Coastal Plain, such as pre-1996 levels at Lake Jandabup[3] (Balla and Davis 1995). This is supported by lower ORP and conductivity readings, which are sometimes used as a measure of pollution, with higher readings indicating higher levels of pollution (Brower, Zar *et al.* 1998).

Unfortunately pH and ORP do not tell the complete story when it comes to overall water quality. Physically the water may seem to have had some improvement, but chemically it is not so obvious. High concentrations of Al and Fe are typical with ASS (Sammut and Lines-Kelly 2000) and were found at each site, yet the treatment site (Site C) actually had substantially higher concentrations on both sampling occasions. This may be explained in part by the presence of the mulch. The mulch may be covering the benthos forming an anaerobic layer in between. This layer, along with low pH and a benthos already coated in iron flocculent allows metals to be reduced and mobilised into the water column (DLWC 1998). It is expected that as the habitat improves, a further increase in pH will cause the metals to bind again with organic soil sediments and the increased biological activity will mean the remaining metals will be consumed by bacteria, algae and plants (DLWC 1998).

The mulch around the treatment site has had an obvious impact on organic carbon content, which provides food and shelter for periphyton, zooplankton,

[3] Lake Jandabup was chosen as a comparison due to it being described as representative of shallow lakes on the Swan Coastal Plain Balla SA, Davis JA (1995) Seasonal variation in the macroinvertebrate fauna of wetlands of differing water regime and nutrient status on the Swan Coastal Plain, Western Australia. *Hydrobiologia* **299**, 147 - 161. prior to recent acidification events Sommer B, Horwitz P (2001) Water quality and macroinvertebrate response to acidification following intensified summer droughts in a Western Australian wetland. *Marine and Freshwater Research* **52**, 1015 - 1021..

macroinvertebrates and other benthic fauna and flora. High organic carbon levels can increase the buffering capacity of lakes (Slattery, Edwards *et al.* 1998) allowing pH levels to rise. Similarly Sommer and Horwitz (2001) suggest that extensive littoral vegetation can have the same effect. The rushes that were planted were not expected to have an immediate effect, however once established they should improve conditions by providing habitat, organic material and oxygenating the sediments. With water depth several metres lower at SD2 compared to SD1, it was not unexpected to see lower carbon levels throughout the sites. It is interesting to note that after the water level had subsided >1 m below the mulch that there were still high levels of FPOM, although it may have come from the decomposing mulch as water levels have fallen.

The presence of high levels of Al and Fe does not seem to have had an effect on macroinvertebrate abundance, however their numbers do seem to correspond with organic carbon content. The availability of a food source may encourage recruitment of macroinvertebrates into the lake. The large variation between sampling dates of macroinvertebrate abundance found at the Spoonbill Lakes is consistent with seasonal variation generally found in wetlands on the Swan Coastal Plain (Balla and Davis 1995). It appears the treatment system has not had an effect on macroinvertebrate abundance as the relative numbers at each site were similar before and after it became operational.

The low macroinvertebrate diversity throughout the sites may be due to the low pH levels, as shown by Sommer and Horwitz (2001) at Lake Jandabup. The macroinvertebrate species present may also be more resilient to the metals in the water than other species. It may be for these reasons that relatively sensitive species such as mayflies (Ephemeroptera) or dragonflies (Odonata) were not found at either sampling time (Davis and Christidis 1997). An increase in diversity at the treatment site may not be realised for some time, and may never return to 'normal', as shown by van Duinen, Brock *et al* (2003) in a comparative rehabilitation site[4]. The lack of increase in diversity could also be attributed to the spatial positioning of other

[4] Even though the comparative site was in the Netherlands, their research looked at macroinvertebrate diversity return after the rehabilitation of acidic, nutrient poor wetlands.

wetlands. New macroinvertebrate species have a long way to travel to disperse to the Spoonbill Lakes.

6. Limitations

All sites had large amounts of filamentous algae present, which caused the sweep net to fill rapidly. This may have resulted in the loss of macroinvertebrates due to two effects: the 'bow-wave' created pushed potential captures out of the mouth of the net, or the pressure required to disperse the algae through the sieves broke them apart and forced smaller macroinvertebrates through the mesh.

In hindsight, had sampling occurred before treatment started, including mulch and potatoes, a more subjective analysis could have been provided. This paper does not give a true before and after picture as the mulch had impacted on the results, however, when compared to the reference sites it gives an indication of pre-mulch conditions.

With the large variation of macroinvertebrate abundances seen in this paper more regular sampling would be required to define the association between seasonal and physico-chemical changes and macroinvertebrate community structure. To fully asses the impact on macroinvertebrate communities from the treatment system, a continued long-term bio-monitoring program would need to be established. The addition of nutrient analysis of water or sediment samples may also assist in assessing the efficacy of the treatment system.

7. Conclusions

While there has been much research conducted into the effects of ASS on rivers and estuarine systems (Lin, Melville *et al.* 1995; Sammut, Melville *et al.* 1995; Sammut, White *et al.* 1996), research on the impacts of ASS on lentic systems is limited (Appleyard, Wong *et al.* 2004; Sommer and Horwitz 2001). There is even less research available on the effect of ASS on macroinvertebrates (van Duinen, Brock *et al.* 2003). This paper is unique in that it has examined many facets of the aquatic habitat in a lake (wetland) influenced by the ongoing effects of ASS.

Despite the limitations mentioned here, the results showed that improving water quality and increasing the organic material does have a beneficial effect on macroinvertebrate abundance within ASS affected wetlands. In the long term this may provide suitable habitat for the basis for a food chain not currently present in the lakes and prevent the release of metals into groundwater supplies. This research suggests that the bioremediation treatment system and associated wetland rehabilitation may have provided improved habitat for aquatic macroinvertebrates. Although without further ongoing sampling, determining which remediation treatment has more effect is unknown.

8. Acknowledgements

First and foremost I would like to thank Dr Clint McCullough for his unwavering patience, bountiful knowledge and for comments and advice while reviewing this paper. For kindly allowing the use of the Wetlands Laboratory and all that it contains I must thank Dr Mark Lund, and the Wetland Laboratory Research Assistants for their assistance in techniques in identifying macroinvertebrates and their taxonomy, Kelly and Quinton Burnham. Brad Mettam helped to make the experience as safe as possible, and Mark Bannister for the ICP-AES water analyses.

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10. Appendices

School of Natural Sciences
Analytical Services

Customer	Dave Galeotti
Date of Issue:	22/05/2007
Reference:	DG07-01
Your Reference:	Spoonbill Lake

ICP -OES Analysis Report

Sample Code		Al	As	B	Ca	Cd	Co	Cr	Cu	Fe	Hg
		mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Reporting Limit		<0.1	<0.1	<0.05	<0.05	<0.01	<0.01	<0.01	<0.05	<0.05	<0.1
Bottle label	Date										
SPOONBILL NORTH NTHL 25/10	25/10/2006	5.6	<0.1	<0.05	64	<0.01	<0.01	<0.01	<0.05	30	<0.1
SPOONBILL SOUTH NTHL 25/10	25/10/2006	5.8	<0.1	<0.05	65	<0.01	<0.01	<0.01	<0.05	32	<0.1
SPOONBILL NORTH STHL 25/10	25/10/2006	9.4	<0.1	<0.05	73	<0.01	<0.01	<0.01	<0.05	38	<0.1
SPOONBILL SOUTH STHL 25/10	25/10/2006	9.0	<0.1	<0.05	73	<0.01	<0.01	<0.01	<0.05	37	<0.1
SPOONBILL NORTH NTHL 9/03	9/03/2007	3.3	<0.1	<0.05	75	<0.01	<0.01	<0.01	<0.05	47	<0.1
SPOONBILL SOUTH NTHL 9/03	9/03/2007	3.1	<0.1	<0.05	74	<0.01	<0.01	<0.01	<0.05	32	<0.1
SPOONBILL NORTH STHL 9/03	9/03/2007	9.7	<0.1	<0.05	67	<0.01	<0.01	<0.01	<0.05	89	<0.1
SPOONBILL SOUTH STHL 9/03	9/03/2007	1.1	<0.1	<0.05	73	<0.01	<0.01	<0.01	<0.05	40	<0.1

Analysis Date: 22/05/2007

ICP -OES Analysis Report

Sample Labels		K	Mg	Mn	Na	Ni	Pb	S	Se	Zn
		mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Reporting Limit		<0.5	<0.1	<0.01	<0.5	<0.05	<0.1	<0.5	<0.2	<0.05
Bottle label	Date									
SPOONBILL NORTH NTHL 25/10	25/10/2006	6.3	16	0.14	67	<0.05	<0.1	157	<0.2	0.18
SPOONBILL SOUTH NTHL 25/10	25/10/2006	6.3	17	0.14	68	<0.05	<0.1	155	<0.2	<0.05
SPOONBILL NORTH STHL 25/10	25/10/2006	9.3	19	0.10	81	<0.05	<0.1	201	<0.2	<0.05
SPOONBILL SOUTH STHL 25/10	25/10/2006	8.9	19	0.09	79	<0.05	<0.1	200	<0.2	<0.05
SPOONBILL NORTH NTHL 9/03	9/03/2007	8.5	20	0.12	86	<0.05	<0.1	180	<0.2	0.08
SPOONBILL SOUTH NTHL 9/03	9/03/2007	8.5	21	0.13	87	<0.05	<0.1	207	<0.2	<0.05
SPOONBILL NORTH STHL 9/03	9/03/2007	6.5	17	0.08	78	<0.05	0.23	207	<0.2	<0.05
SPOONBILL SOUTH STHL 9/03	9/03/2007	9.5	20	0.06	106	<0.05	<0.1	215	<0.2	<0.05

Analysis Date: 22/05/2007

Macroinvertebrate data m^{-2} at Spoonbill Lakes before and after the treatment system became operational.

			25/10/2006 Pre-Remediation				9/03/2007 Post-Remediation			
Order	Family Name*	Genus (if known)	Site A	Site B	Site C	Site D	Site A	Site B	Site C	Site D
Diptera	Chironomidae L		453	7	1097	38	2	1	10	
	Chironomidae P		81	10	40	39		1	1	1
	Ceratopogonidae L		27	3	10	105		1		3
	Dolichopodidae L			1						
Coleoptera	Dytiscidae L	Necterosoma	1	2	40	13	1			
	Dytiscidae	Necterosoma	3	1		2	2	1	3	2
	Dytisidae	Megasporus	1							
	Hydrophilidae L	Berosus		1	3	6		1		1
Hemiptera	Corixidae N	Sigara	1				1			

	Site A	Site B	Site C	Site D	Site A	Site B	Site C	Site D
Total abundance	567	25	1190	203	6	5	14	6
Total species	7	7	5	6	4	5	3	4

* L - Larvae, P - Pupae, N - Nymph

Organic carbon content of CPOM and FPOM samples taken from Spoonbill Lakes

25/10/2006	dry CPOM + tray weight	tray weight	dry CPOM weight	CPOM ash + tray weight	tray weight	CPOM ash weight	% ash weight
Site A	45.12	8.60	36.52	14.05	8.60	5.45	15
Site B	18.85	9.03	9.82	10.43	9.02	1.41	14
Site C	350.10	8.88	341.22	102.20	8.90	93.30	27
Site D	44.87	8.72	36.15	12.05	8.73	3.32	9
9/03/2007							
Site A	38.02	8.95	29.07	13.60	9.00	4.60	16
Site B	24.60	8.36	16.24	9.99	8.40	1.59	10
Site C	23.32	8.73	14.59	10.98	8.70	2.28	16
Site D	19.44	8.58	10.86	9.59	8.60	0.99	9

25/10/2006	dry FPOM + tray weight	tray weight	dry FPOM weight	FPOM ash + tray weight	tray weight	FPOM ash weight	% ash weight
Site A	8.66	1.53	7.13	4.18	1.53	2.65	37
Site B	3.57	1.53	2.04	1.96	1.53	0.43	21
Site C	3.45	1.53	1.92	2.55	1.53	1.02	53
Site D	8.67	1.53	7.14	3.65	1.53	2.12	30
9/03/2007							
Site A	5.25	1.53	3.72	2.07	1.53	0.54	15
Site B	3.07	1.53	1.54	1.72	1.53	0.19	12
Site C	4.36	1.53	2.83	2.46	1.53	0.93	33
Site D	4.82	1.53	3.29	2.10	1.53	0.57	17

Appendix 3

	% Organic Carbon Weight		
25.10.06	CPOM	FPOM	Total
Site A Reference	15	37	52
Site B Reference	14	21	35
Site C Treatment	27	53	80
Site D Reference	9	30	39

	% Organic Carbon Weight		
9.03.07	CPOM	FPOM	Total
Site A Reference	16	15	30
Site B Reference	10	12	22
Site C Treatment	16	33	48
Site D Reference	9	17	26