School of Environmental Biology

The Classification of Inland Salt Lakes in Western Australia

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Declaration

To the best of my knowledge and belief this thesis contains no material previously published by any other person except where due acknowledgment has been made.

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university.

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Abstract

Inland salt lakes in Western Australia have been used by the mining industry for the disposal of excess water generated during the mining process. However, the impact of these operations on the salt lakes is poorly understood. This is mainly due to the lack of information on the biota and chemistry for the lakes. The main aim of this project was to develop a classification system for inland salt lakes of Western Australia based on abiotic and biotic factors such as sediment and water quality, invertebrates and algae to determine lakes with unique or significant features.

Water and sediments collected from the salt lakes were generally characterised by an alkaline pH, high salinity and the majority of lakes being dominated by sodium and chloride. Concentrations of some metals were also high, particularly in surface water. A high degree of variation in water and sediment quality was demonstrated both within and between the study lakes. In addition, these parameters were shown to be influenced by geography, geology, stage of the hydrocycle within which the lake was sampled and the occurrence of dewatering discharge.

Biota in the salt lakes must be able to cope in a harsh environment, adjusting to temporary water regime, high temperature, and high salinity. As such, the species richness of these systems is generally low. Diatoms (a group of algae) and invertebrates were investigated among the biota. A total of 56 diatom species were recorded from 24 lakes. The most common species were *Amphora coffeaeformis*, *Hantzschia* aff. *baltica* and *Navicula* aff. *incertata.* These species were shown to have broad tolerances to environmental variations. Sediment chemistry explained variations in diatom community structure, with zinc, moisture content and cobalt having the greatest and negative influence.

In terms of invertebrates, a total of 101 invertebrate taxa were recorded from 13 lakes in this study. Crustacea dominated and the greatest number of taxa was from the genus *Parartemia*. There were some differences in invertebrate community structure between lakes, most likely reflecting the high degree of speciation, and poor dispersal mechanisms of certain key species. Community structure was influenced by water quality, with phosphorus, bicarbonate and magnesium contributing to the variations in community structure.

Among the 43 lakes chosen for this study a total of 17 lakes had received, or are currently receiving dewatering discharge. Sites receiving dewatering discharge generally reported higher concentrations of salts, nutrients and some metals in both water and sediments compared to natural lakes. Species richness of biota such as diatoms and invertebrates was lower at the lakes receiving dewatering discharge. However, the impact was generally localized within the pooled area of dewatering discharge. Also, despite these impacts, there appears to be signs of amelioration by flushing events.

Currently there are no guidelines for water and sediment chemistry for inland salt lakes in Western Australia. Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines are the most relevant available. Concentrations of cadmium, cobalt, chromium, copper, lead, nickel and zinc in surface water of the natural inland salt lakes were shown to exceed ANZECC guideline values. Comparison with the relevant ANZECC sediment guidelines showed that they were applicable to the salt lakes, with the exception of nickel and chromium which were naturally high in the salt lake sediments.

Classification of data using multivariate analysis was done for both dry and wet phases of the hydroperiod. Six groups were delineated for the sediment and diatom data, and four groups were defined for the water quality and invertebrate data. It was common for sites from particular lakes to fall in more than one group as a result of the variability in these systems. There are a number of practical applications of this system for the mining industry and it may be used as a predictive tool for determining the impact of dewatering discharge and highlighting unique salt lakes within the Goldfields of Western Australia.

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1 INTRODUCTION

In Western Australia most of the inland salt lakes occur in association with rich mineral deposits (De Deckker 1983). The need to collect and compile information on both the biotic and abiotic components of salt lakes in inland Western Australia has become increasingly urgent in recent years, due to the level of mining development in the vicinity and the potential threat of these activities to the lakes. In addition, a number of companies use the salt lakes for the disposal of saline to hypersaline water from their mining operations ('dewatering discharge'), and little is known about the impact of dewatering discharge on the carrying capacity of the receiving lakes. While most of these operations monitor the impact of their activities on a regular basis, results are often stored in unpublished company reports. As a result, our knowledge of the salt lake systems as a whole has been compromised. The main aim of this project was to develop a classification system for inland salt lakes of Western Australia based on data collected to date. It was envisaged that this would contribute to management objectives, aid in determining the significance of specific water bodies and further clarify the impacts of dewatering discharge on the lakes.

This thesis consists of a total of eight chapters including this introduction and literature review. Chapter 2 describes the location, characteristics, land use and brief history of the lakes addressed in the study. The water chemistry of the lakes is described in detail in Chapter 3. Chapter 4 consists of a discussion of the sediment quality including salinity, pH and metal concentrations. Chapters 5 and 6 address the biota in terms of algae and invertebrates in the lakes. In addition to describing the condition and biota of the inland salt lakes, chapters 3 – 6 also focus on the impacts of dewatering discharge to these factors. Chapter 7 proposes a classification system for the study lakes based on the results of the preceding four chapters and Chapter 8 presents the major conclusions of the study. The methods used in each chapter were combined and a sampling protocol for field data collection is presented in Appendix A.

1.1 Inland Salt lakes

Salt lakes are extensive throughout the world, with occurrences of these lakes on most continents (Williams 1981a). As a rule, most are shallow and temporary systems in areas subject to high evaporation rates (Hammer 1986). While numerous, these lakes have been shown to be fairly dissimilar throughout the world in terms of chemistry and biota (Williams 1981a; De Deckker 1983; Hammer 1986). The biota occurring in these systems must have the ability to survive the 'harsh' conditions of salt lakes such as temporary water regime and high salinity. In addition, most of the biota have the ability to survive long periods without freshwater inputs when the lakes are dry (Williams 2002). The production of desiccation resistant cysts, as seen in the anostracans, ostracods, cladocerans, notostracans and conchostracans are common in these lake types (Williams 1985).

1.2 Inland Salt Lakes in Australia

De Deckker (1983) has described four main salt lake types in Australia, including; large and small closed inland basins, crater lakes and coastal lakes. This study is concerned with the inland salt lakes which are not connected with the sea and for the most part are defined as large closed basins (De Deckker 1983). Inland salt lakes in Australia are widespread, with high concentrations of lakes occurring in the semi arid areas of Western Australia, South Australia and Victoria (Hammer 1986; Hart and McKelvie 1986). The semi arid zone of Western Australia has the greatest concentration of salt lakes on the Australian continent (Gentilli 1979), with most located in what is known as the "Salinaland" (Jutson 1934; Gentilli 1979). These are remnants of the palaeodrainage systems forming to provide drainage to the surrounding areas (Mann 1983; Commander 1999; Williams 2002). Most of the Western Australian salt lakes are therefore endorheic – that is internally draining (Williams 1998b). In addition, they are characterized by flat playas, generally in areas which have little topographic relief. They are shallow when filled, with most lakes rarely exceeding a metre in depth (John 1999).

1.2.1 Water Quality of Inland Salt Lakes in Western Australia

While elevated salinity is not the only distinguishing feature of the inland salt lakes of Western Australia, it is widely accepted that salt lakes are water bodies with surface water salinity of 3 g/L or greater (Geddes *et al.* 1981; Williams 1998b; 2002). These lakes fill infrequently, usually in response to heavy rainfall events (Timms *et al.* 2006) and it is possible for salt lakes to remain dry for long periods of time, even decades (Roshier and Rumbachs 2004; Timms 2005a). When they do fill, the drying period is relatively rapid due to the high evaporation rates in these semi arid areas, lasting a couple of months at the most (Timms *et al.* 2006).

Typically, salt lakes exhibit a high degree of variation in water quality over their hydrocycle (Williams 1985; Roshier and Rumbachs 2004). For example the salinity range of the salt lakes in the Goldfields has been reported to be 5 to 300 g/L (Williams and Buckney 1976). Sodium (Na) and chloride (Cl) tend to dominate the salts (Williams and Buckney 1976), and generally the cationic balance of surface water remains seasonally constant and follows a gradient of Na>magnesium (Mg)>calcium (Ca)> potassium (K), while the anionic gradient typically follows Cl>bicarbonate (HCO_3)>sulphate (SO_4) (Williams 1985). The pH of the waters is usually alkaline, exceeding eight (Williams and Buckney 1976). Owing to the shallow nature of the salt lakes, levels of oxygen and light are generally high in surface water and do not hinder productivity (Williams 1985). However, high salinity can reduce the solubility of oxygen (Borowitzka 1981). Stratification in the water column does not usually occur, as mixing is frequently a result of wind movement, partly facilitated by the shallow nature of the lakes (Williams 1985). While there is a considerable amount of information on the salt load and basic chemistry of these systems, there is little published information on the concentrations of metals in surface water of Western Australian salt lakes and this will be addressed in this study.

1.2.2 Biota of Inland Salt Lakes in Western Australia

Salt lakes in Western Australia display one of the highest levels of endemism and biodiversity in Australia (Hebert and Wilson 2000; Halse and McRae 2001; Remegio *et al.* 2001). The primary producers of the salt lakes include bacteria, cyanobacteria (blue green algae) and eukaryotic algae (Borowitzka 1981). Most of the major algal groups present in saline waters are also represented in freshwater systems (Williams 1998b). Cyanobacteria that are common to salt lakes in Western Australia include *Phormidium*, *Oscillatoria*, *Microcoleus* and *Schizothrix* species (Borowitzka 1981; Williams 1985). These species often form thick mats on the lake sediments in combination with the diatoms (Bauld 1981). The diatoms are the most numerous primary producers in these systems, with common species including *Navicula*, *Nitzschia* and *Amphora* (Williams 1998b). One charophyte species, *Lamprothamnium*, also occurs in salt lakes (Williams 1985). While macrophytes can be uncommon in the lakes, beds of *Ruppia* are quite often observed during filling events and are known to survive in salinities greater than 100 g/L (Williams 1985).

In terms of algae this study focuses primarily on the diatoms for a number of reasons. They are unicellular and are composed of silica and are relatively easy to identify, hence their extensive use in limnological studies (John 2007). Diatoms occur in a wide range of habitats and can be used to indicate a change in water quality within a system (Gell and Gasse 1990; Blinn and Bailey 2001; Cook and Coleman 2007; John 2007). Also, they are particularly sensitive and respond to changes in pH, salinity and nutrients (Biggs 1995; Cook and Coleman 2007; Tibby *et al.* 2007). In addition, they have been expansively studied for the purposes of paleolimnological and climate change studies (Gasse *et al.* 1995; Gell 1997).

Invertebrate communities occurring in the inland salt lakes of Western Australia are poorly studied (Timms 2007). Adding to the lack of knowledge, generally the salt lake invertebrates present in the West Australian salt lakes have a fairly restricted distribution and there are numerous species which are undescribed (Halse 1999). Early studies indicate that the most commonly found groups include the Crustacea, Gastropoda, and Rotifera (Brock and Shiel 1983; De Deckker 1983). Individuals from the genus *Parartemia* are often the most common species encountered (John 2001), with some of the larger playas supporting a large biomass of these species (Coleman *et al.* 2004; McMaster *et al.* 2007).

The salt lakes support large numbers of waterbirds which utilize the lakes for feeding when full. Chapman and Lane (1997) identified 35 avian species during a filling event of salt lakes of the Goldfields in the early 90s. The most common species recorded included the Black Swan (*Cygnus atratus*), Australian Shelduck (*Tadorna tadornoides*), Grey Teal (*Anas gracilis*) and the Hoary-Headed Grebe (*Poliocephalus poliocephalus*) (Chapman and Lane 1997).

1.3 Threats to Inland Salt Lakes

The inland salt lakes of Western Australia are threatened by a number of processes including salinisation, climate change, pastoral activities, invasion by exotic species and mining (Williams 2002; Timms 2005a; McMaster *et al.* 2007). Salinisation, particularly in the Wheatbelt of Western Australia has contributed to an increase in salinity in most of the water bodies in this region (Strehlow *et al*. 2005). This in turn has had major implications for the biota which exist in these lakes (Halse *et al.* 2003). Another threat to the salt lake ecosystem is the effects of climate change. While the impact of this phenomenon on these systems is relatively unknown, early studies indicate that substantial changes in water regime are likely (Williams 2002; Timms 2005a). An example of the invasion by exotic species to inland salt lakes in Western Australia has been presented by McMaster *et al*. (2007). The authors have noted the spread of *Artemia parthenogenetica* into inland salt lakes in Western Australia. It is most likely that this species was introduced to coastal salt lakes and are spreading by vectors such as avian fauna. The impact of mining to the salt lake ecosystem is known to be substantial and is therefore the major focus of this study.

Mining in and around Western Australian salt lakes has resulted in considerable impact to the hydrology of the lakes. The construction of causeways onto the lake is common during exploration activities (Williams 2002) and can result in changes in surface hydrology and prevention of the movement of biota throughout the lake (Timms 2005a). Also, the construction of pits in the lakes is becoming more common and occurs in some of the larger lakes such as Lake Way, Lake Carey and Lake Lefroy. Pit dewatering, which is required for mining pits that intercept the water table, results in a 'cone of depression' or drawdown of groundwater in the vicinity of the pit (Datson 2007). Drawdown often causes significant drying of the playa (personal observation). In association, waste rock landforms and bund walls are also constructed on the side and within the lakes which results in altered drainage and in some cases increased erosion.

Excess surface water generated from mine dewatering is used in the processing of minerals and in some instances the volume of water is too great, and/or the quality is not suitable for re-use. In these circumstances, there is a need for disposal of excess water. There are a number of options for disposal of surplus water. On occasion, and if feasible, old mine voids are used for water storage. This option is limited by the occurrence of mine voids close to the active void that is being dewatered. Aquifer re-injection is another possible method for disposing of the excess water, however this may be cost prohibitive. Traditionally, salt lakes have been the preferable disposal option for excess water produced during mining. Their large flat surfaces optimize evaporation, and in the past, this was probably seen as the most 'environmentally friendly' option for discharge. In that sense the playas are usually not vegetated and are devoid of fauna during dry conditions. Also it is often the cheapest and most effective disposal method.

Currently, discharge to salt lakes occurs within nine salt lakes within the Goldfields (OES unpublished data), however an additional eight lakes have received dewatering discharge. The lakes currently receiving dewatering discharge include Lake Way, Lake Carey, White Flag Lake, Lake Lefroy, Lake Hope North, Lake Cowan, Lake Wownaminya, Yarra Yarra Lakes and Lake Raeside North. Lakes historically receiving dewatering discharge include Lake Miranda, Lake Austin, Kurrawang White Lake, Banker Lake, Southern Star Lake, Lake Fore, Lake Tee and Lake Koorkoordine. This disposal of surplus water ('dewatering discharge') is controlled by a site licence issued by the Department of Environment and Conservation (DEC). The licence usually specifies the maximum volume of water that can be discharged per annum, the water quality parameters to be tested, the sampling frequency and the reporting requirements. Parameters to be tested typically include pH, total dissolved solids (TDS) and electrical conductivity (EC). In some cases, concentrations of metals are also required to be tested. While these reports are being prepared on an annual basis, the impact of these operations as a whole is poorly known.

1.4 Gaps in knowledge

There have been numerous studies of wetlands on the Swan Coastal Plain and south-west, while in comparison, little focus has been placed on the wetlands of the Goldfields region of Western Australia (Williams and Buckney 1976; Geddes *et al.* 1981). Recently threats to Goldfields wetlands such as impacts from mining have led to an increase in research on salt lakes (URS 2003; Cale *et al.* 2004; Vellekoop and van Etten 2004). Despite this, the uniqueness and ecology of the salt lakes in Western Australia is poorly understood (Timms 2007). The lakes as a whole have never been compared or examined and little is known about the significance of each of the water bodies. Obtaining data for these ecosystems is also hampered by the frequency with which the lakes fill. Due to this, information on water quality, algae and invertebrates which are present in the lakes is poorly known.

While approximately 17 lakes in Western Australia have received dewatering discharge from nearby mining activities, the impact of these operations has not been defined (Timms 2005a). In addition it is not known if the lakes receiving dewatering discharge are unique or contain attributes dissimilar to other inland salt lakes. This has identified the need for a classification system for the inland salt lakes in Western Australia. This need has been raised on numerous occasions at recent conferences relating to salt lakes (Jasper 1999; John 2001). Classification systems are used to simplify large data sets, and allow groups of data to be formed according to similar attributes (Semeniuk and Semeniuk 1997). Most importantly, they allow informed decisions to be made for the protection, recognition of unique attributes and management of aquatic systems (Department of Environment 2005). Classification systems can be based on numerous attributes such as water permanence, geomorphology, water quality, biota and size.

One of the most widely accepted classification systems world wide is that defined by the Ramsar Convention (Semeniuk and Semeniuk 1997). The Ramsar classification system recognizes three main types of wetlands; Marine/Coastal wetlands, Inland Wetlands and Human-made Wetlands. These are further divided in 42 types according to parameters such as salinity, water flow, size and water regime (Ramsar 2006). This system is loosely based on the classification system presented by Cowardin *et al*. (1998) (Semeniuk and Semeniuk 1997). Cowardin's system is widely used throughout the United States and recognizes five main system types; estuarine, marine lacustrine, riverine and palustrine (Cowardin *et al.* 1998). Within the five system types, further classification occurs according to water regime and sediment type. A number of other descriptors such as salinity, pH and soil type can also be added. While efficient for the wide range of wetland types worldwide, the Ramsar classification system is too broad for the purposes of this study, in that all salt lakes considered in this study are defined as Seasonal/Intermittent Lakes (Category R).

A number of Western Australian classification systems have been described for freshwater systems to date but their use is limited to wetlands of the Swan Coastal Plain, South-west and the Wheatbelt (Davis *et al.* 1993; Semeniuk and Semeniuk 1997; Cale *et al.* 2004). The Department of Environment (2005) (now DEC) supports the use of the geomorphic classification system proposed by Semeniuk and Semeniuk (1995), however they also state that systems based on biological data are acceptable. The classification system proposed by Semeniuk and Semeniuk (1995) is primarily based on the landform setting and hydroperiod of a particular wetland. Other factors or 'descriptors' such as vegetation, water chemistry, size and landforms can be added to the system for greater definition. Within the vegetation descriptor, further classification is possible according to plant distribution patterns, for example Zoniform (zones of vegetation on the periphery) or Bacataform (vegetation types in no particular pattern but on the periphery) classifications can be applied. According to Semeniuk's classification system, the lakes of the Goldfields are considered similar. However, this system does not consider aquatic biota, including invertebrates and algae. In that context, the presence of unique, ubiquitous and important species, and the management measures required to protect them, is not addressed.

1.5 Significance of this study

The need to combine all available data on the lakes is apparent and the importance of a classification system for practical use, particularly within the mining industry, was a major concern raised at the close of the 2003 Australian Centre for Minerals Extension and Research (ACMER) workshop on Water Quality Issues in Final Voids, Salt Lakes and Ephemeral Streams (John 2003b). The need for the accumulation and integration of data for a range of different water bodies was also expressed by Batley *et al.* (2003) in their review of the current ANZECC guidelines (Batley *et al.* 2003).

The regulation of water disposal to the salt lakes has been inadequate, because current licence and guidelines are not based on existing knowledge. Currently the Australian and New Zealand Environment Conservation Council (ANZECC) guidelines (2000) are applied to some of these lakes. The ANZECC guidelines are inadequate for salt lake systems and do not help to identify impacts associated with the mining that occurs in Western Australia. There is a need for regulators and users of the lakes to have a set of guidelines to maintain and follow.

The use of inland salt lakes should be viewed and regulated from the perspective of the resilience of the lakes and the impact of the dewatering discharge practices to the lakes.. The data collected and compiled in this study will significantly contribute to the understanding of salt lake ecology in Western Australia, while also considering the impacts of dewatering discharge, and the assimilative capacity of the lakes.

1.6 Objectives

The primary objectives of the project were;

1. To gather and validate data on inland salt lakes, with reference to water and sediment quality and aquatic biota
2. To identify the gaps in knowledge related to inland salt lakes
3. To develop a field protocol for sampling salt lake ecosystems
4. To develop a classification system of inland salt lakes with predictive capability, taking into consideration their ecological sustainability.

2 STUDY LAKES

2.1 Introduction

The majority of the lakes in this study are located in the Goldfields of Western Australia and fall in the Murchison, Coolgardie, Yalgoo and Avon Wheatbelt Bioregions (Figure 1). Of the 43 lakes in this study, 17 are currently or have in the past received dewatering discharge. The lakes range from Lake Hope in the south-west, Lake Cowan in the south east, Lake Way in the north and Lake Wownaminya in the north-west. All lakes in the current study have been grouped according to the Interim Biogeographic Regionalisation for Australia (IBRA) Bioregion in which they occur (Figure 1). The IBRA regions are grouped according to major ecosystem characteristics such as vegetation, geology and fauna (Thackway and Cresswall 1995).

The climate of the Goldfields region is described as semi-arid, characterized by hot dry summers and mild winters. The annual average rainfall for the region is around 270 mm per year with evaporation far exceeding this, ranging between 2000 and 3000 mm per year (BOM 2007). Most of the heavy rainfall events occurring in this region are as a result of cyclonic activity on the coast. There has been a considerable level of variability in rainfall at Kalgoorlie particularly since 2000 (Figure 2). For example, during tropical cyclone John, 169 mm of rainfall was recorded in Kalgoorlie during January 2000 (BOM. 2000).

A large proportion of the salt lakes in this study overlay the Yilgarn Craton. The Yilgarn block consists of highly weathered granite, gneisses and greenstone belts (Lyons *et al.* 1990). Lacustrine sediments present in the lakes can be up to 100 m thick, particularly in the larger playas (McArthur *et al.* 1991). These lakes provide drainage to the surrounding areas and are remnants of the palaeodrainage systems (Mann 1983; Turner *et al.* 1993; Commander 1999). In terms of groundwater, the high salinities in the Goldfields are primarily due to evapo-concentration in salt lake waters (Turner *et al.* 1993; Commander 1999). The ground waters of the region are typically hypersaline, recording values exceeding 200 g/L.

Figure 1. Major IBRA Bioregions in Western Australia, with study area represented by the black square. Map extracted from CALM, 2002.

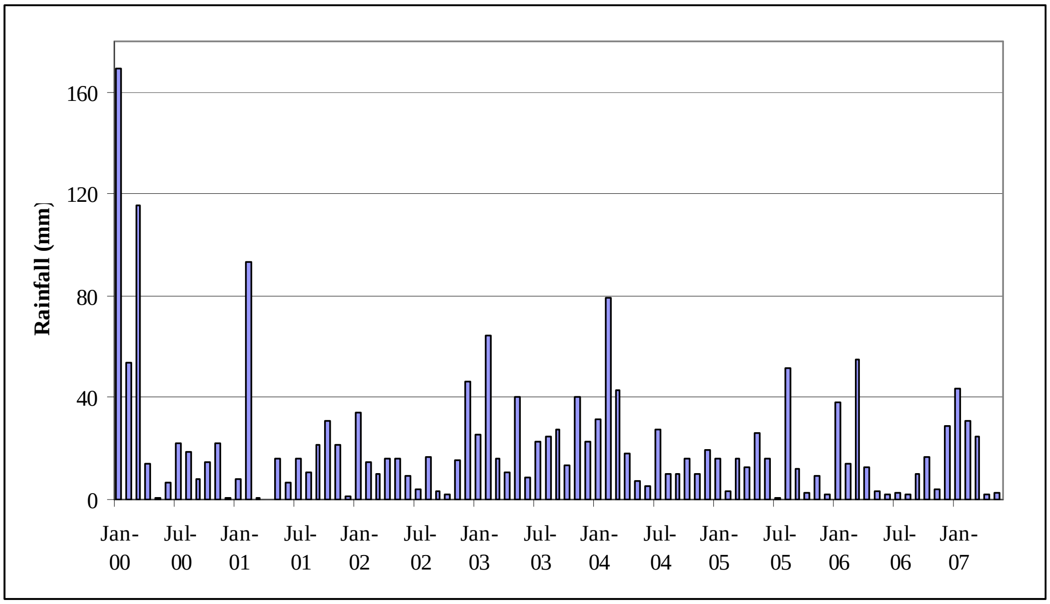

Figure 2. Total monthly rainfall (mm) recorded at the Kalgoorlie Airport from 2000 to May 2007.

Vegetation surrounding the lakes is dominated by Samphires, such as *Halosarcia* species, close to the playa, with other chenopods such as *Atriplex* and *Frankenia* being recorded further up the dune. It is fairly common for *Halosarcia* to encroach the playa, depending on the dampness and the salinity of the playa. Often, they tend to form distinct zones around the lake, depending on each particular species preference for certain conditions such as salinity and moisture (English *et al.* 1999; Datson 2002). For example species such as *Halosarcia halocnemoides* and *H. peltata* are common on the fringes of the playa preferring moist conditions (Datson 2002).

The sampling effort in each lake differed with some lakes being sampled over a range of seasons in both the wet and dry phases of the hydrocycle in comparison to some lakes which were only sampled once. This was related to the lack of filling events, the length of the project and the occurrence of monitoring programs within specific lakes. In addition the sites were grouped according to the influence of dewatering discharge. Those not impacted by dewatering discharge were named natural lakes (or sites). It should be noted that in some of the larger lakes it was possible for portions of the lakes to remain free from dewatering discharge and retain the classification of 'natural site.'

2.2 Murchison Bioregion

A total of eight salt lakes in this study occur within the eastern portion Murchison Bioregion (Table 1, Figure 3 and Plate 1). The catchments of these lakes are generally endorheic (internally draining) and the lakes follow the major palaeodrainages in the region, namely the Raeside, Carey and Yindarlgooda Palaeorivers (Timms 1992). Soils of the bioregion are classified as red-brown soils, red sand plains or breakaways (Beard 1990). The vegetation of this region is characterized as 'Mulga shrubland' (CALM 2002). *Halosarcia, Atriplex, Maireana* and *Frankenia* species are all common surrounding the low dune areas of the salt lakes. Of the eight lakes within this study five are or have received dewatering discharge. With the exceptions of Lake Maitland and Lake Penny, the majority of lakes in this region exceeded 200 km^2 in size.

Table 1. Area, catchment size and land use of salt lakes included in this study which occur in the Murchison Bioregion. The lakes are arranged from largest to smallest lake.

Lake	Size (km^2)	Catchment size	Major Land Use
Lake Carey[1.]	1000	9000	Pastoral, mining, dewatering discharge
Lake Austin[2.]	770	-	Dewatering discharge
Lake Raeside North[3.]	300 km long	-	Dewatering discharge
Lake Way[4.]	270	-	Dewatering discharge, mining, exploration and pastoral
Lake Rebecca[5.]	270	2500	Pastoral, exploration
Lake Miranda[6.]	200	1400	Pastoral, dewatering discharge
Lake Maitland[7.]	60	-	Pastoral and exploration
Lake Penny[8.]	7.9	-	Pastoral

[1.](John 1999) [2.] (Vellekoop and van Etten 2004) [3.](Outback Ecology 2006b) [4.] (Outback Ecology 2006d)[5.](Turner 1999) [6.] (URS 2003) [7.] (Outback Ecology 2007) [8,] (Campagna and John 2003)

a.

b.

c.

d.

e.

f.

Plate 1. Lakes of the Murchison Bioregion, specifically a. and b. Lake Carey, c. Lake Way, d. Lake Maitland, e. Lake Raeside, f. Lake Rebecca.

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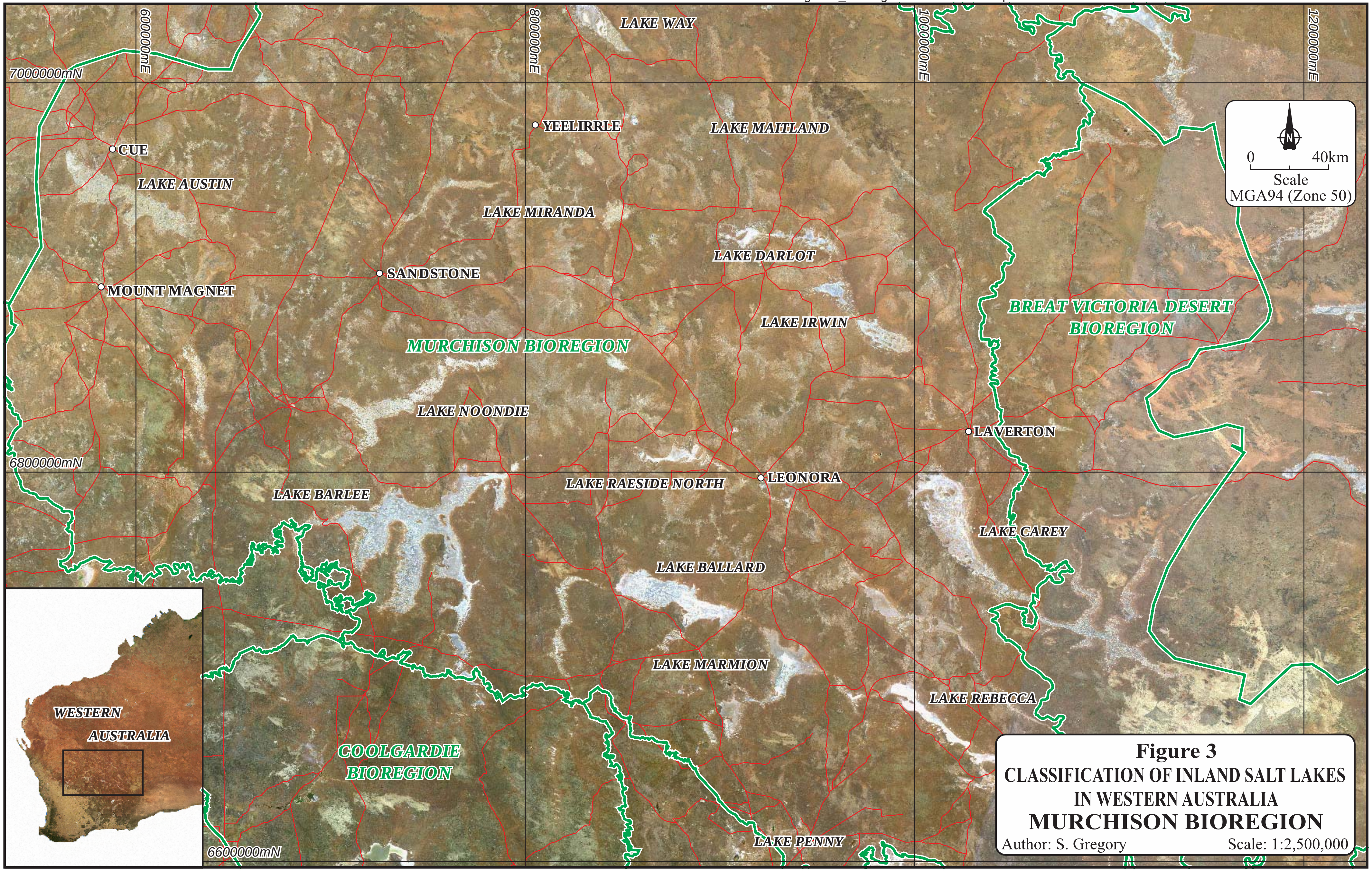

Figure 3
CLASSIFICATION OF INLAND SALT LAKES IN WESTERN AUSTRALIA
MURCHISON BIOREGION
Author: S. Gregory Scale: 1:2,500,000

2.3 Coolgardie Bioregion

There are 30 salt lakes included in this study which are located in the Coolgardie Bioregion (Figure 4, Table 2 and Plate 2). Similarly to the Murchison this Bioregion is underlain by the Yilgarn Craton (CALM 2002). The Coolgardie Bioregion consists of three sub regions; the Southern Cross, Eastern Goldfields and the Mardabilla. The vegetation of this district is characterized by eucalypt woodland, with a chenopod understorey. The soils are described as brown calcareous soils. Large playas are a characteristic of soils in the area (Beard 1990). Of the 30 lakes in this study, 11 have received or are receiving dewatering discharge.

Table 2. Area, catchment size and land use of salt lakes in the Coolgardie Bioregion, arranged from largest to smallest in terms of size.

Lake	Size (km^2)	Catchment size (km^2)	Major Land Use
Lake Cowan[1.]	1780	4460	Dewatering discharge, mining , causeway construction
Lake Lefroy[.2]	570	4560	Dewatering discharge, mining , causeway construction
Lake Yindarlgooda[2]	420	3864	Pastoral, mining on fringes, mining-related evaporations ponds
Lake Hope	382	-	Recreation
Lake Johnston[3.]	175	-	Recreation
Lake Hope North	175	400	Mining, receives dewatering discharge
Baladjie Lake[4.]	89	-	Recreation
Black Flag Lake[2]	7	730	Pastoral, some exploration on fringes of lake
White Flag Lake[2.]	32	200	Mining to the south, has received dewatering discharge
Lake Zot	21	-	Mining in the vicinity of the shore
Lake Koorkoordine[5.]	4.5	-	Mining, Tailings Storage Facility located on the shore
Lake Eaton North	3.4	-	Pastoral
Lake Tee[6.]	3.3	0.16	Dewatering discharge
Binneridgie Road Marsh	3.2	-	Causeway constructions
Kurrawang White Lake[7.]	1.75	-	Received dewatering discharge
Lake Fore[6.]	1.2	3.35	Dewatering discharge
Lake Josh[6.]	0.85	0.15	Pastoral

Lake	Size (km²)	Catchment size (km²)	Major Land Use
Greta Lake[7].	0.75	-	Mining activities south of the lake
Kopai Lake[2]	0.5	-	Mining activities south of the lake
Southern Star Lake[8.]	0.5	450	Mining, has received dewatering discharge historically.
Victory Lake	<0.5	-	Mining in the vicinity of the shore
Lake Polaris[5.].	<0.5	1207	Recreation, township
Swan Refuge	<0.5	-	Pastoral
Lake Why	<0.5	-	Pastoral, used for stock watering
Golf Lake	<0.5	-	Pastoral
Airstrip Lake	<0.5	-	No direct mining impacts
Creek Lake	<0.5	-	No direct mining impacts
South West Lake	<0.5	-	No direct mining impacts
Un-named Lake	<0.5	-	No direct mining impacts

[1.] (Aquaterra. 2006)[2] (Turner 1999) [3.](Cowan *et al.* 2001) [4] (Department of Environment and Heritage 1999) [5.].(Coleman 2003) [6.] (Foster 2004) [7.] (Outback Ecology 2006a) [8.] (Barrett 2003)

NB – where no source was given, lake size was estimated using specific mapping tools.

Note – the last four lakes names are not formally recognized and have been named by site personnel.

a.

b.

c.

d.

e.

f.

Plate 2. Selected lakes within the Coolgardie Bioregion, including; a. White Flag Lake, b. Black Flag Lake, c. Lake Lefroy, d. Baladjie Lake, e. Lake Hope, F. Lake Johnston.

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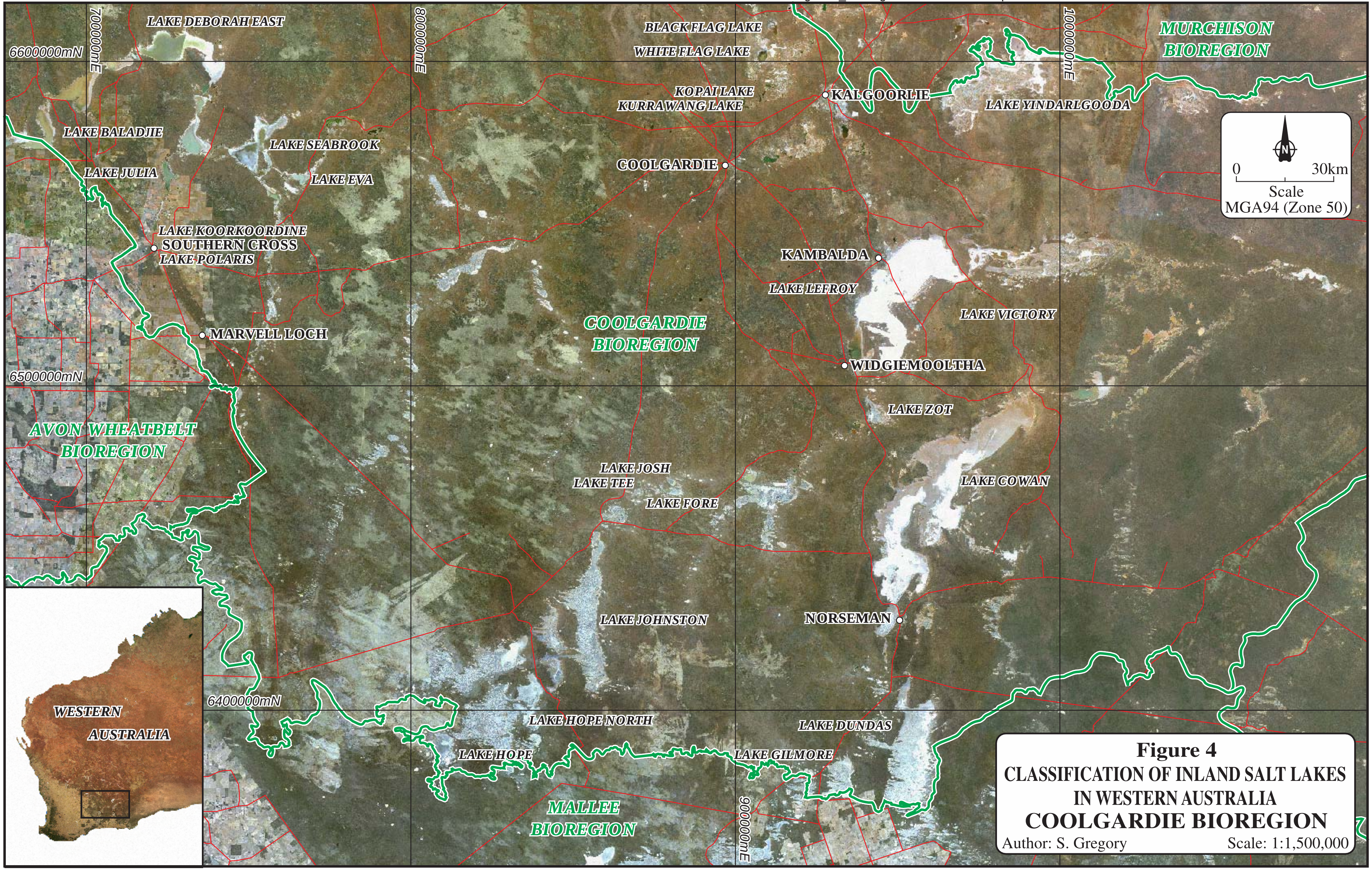

Figure 4
CLASSIFICATION OF INLAND SALT LAKES IN WESTERN AUSTRALIA
COOLGARDIE BIOREGION
Author: S. Gregory Scale: 1:1,500,000

2.4 Avon Wheatbelt Bioregion

The Yarra Yarra Lakes are situated within the Avon Wheatbelt Bioregion of Western Australia (Figure 5). This Bioregion consists of two subregions, the eastern and western, and the Yarra Yarra lakes are situated in the eastern subregion. This is a region of ancient drainage, underlain by the Yilgarn Craton (CALM 2002). Vegetation surrounding the clay pans typically consists of *Halosarcia doleiformis* and *H. halocnemoides*. The Yarra Yarra catchment covers approximately 42 600 km^2, with a predominantly agricultural land use (Boggs *et al.* 2006). The Yarra Yarra lakes encompass almost 300 km and contain approximately 4 500 lakes and clay pans (Boggs *et al.* 2006). The three lakes included in this study are located in the northern sections of the Yarra Yarra drainage system, and are unnamed (Outback Ecology 2005). The lakes are small, less than <0.5 km^2 in size (Plate 3). One of the lakes receives dewatering discharge, however the other lakes are located immediately upstream and downstream. While not impacted by discharge it is likely that they are impacted by pastoral activities.

Plate 3. Lakes of the Yarra Yarra Lakes catchment.

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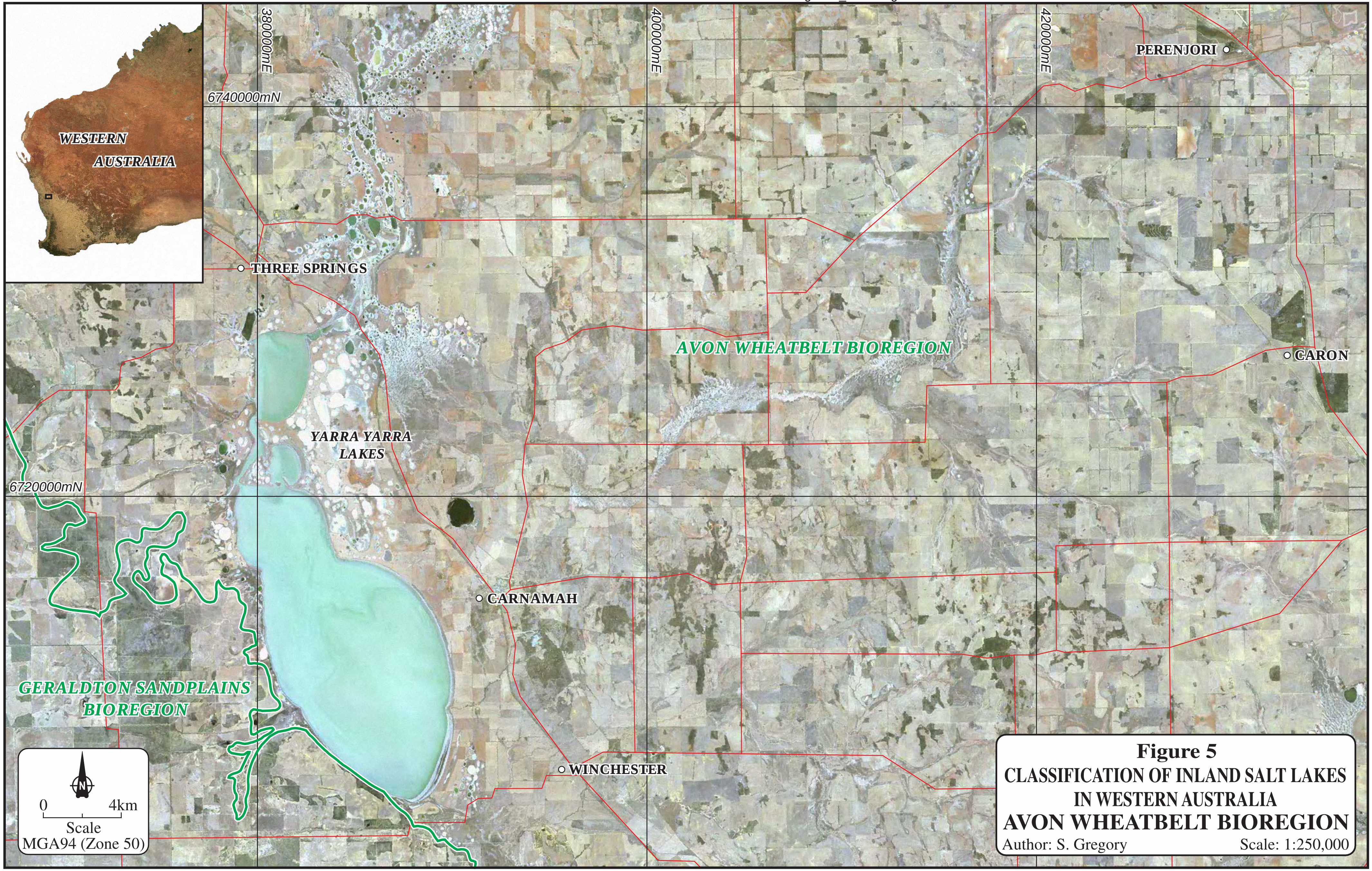

Figure 5
CLASSIFICATION OF INLAND SALT LAKES
IN WESTERN AUSTRALIA
AVON WHEATBELT BIOREGION
Author: S. Gregory
Scale: 1:250,000

2.5 Yalgoo Bioregion

One lake, Lake Wownaminya is situated in the Yalgoo Bioregion (Figure 6 and Plate 4). This Bioregion is the transition zone between the Murchison and the South west Bioregions. *Eucalyptus salubris* and *Acacia aneura* are common over-story vegetation. Pastoral activities are common in the region and are thought to impact the lake. Lake Wownaminya is located approximately 225 km east of Geraldton. At approximately 150 km^2 in size, it is one of a series of lakes located in a large drainage system with a catchment in excess of 700 km^2 (Outback Ecology 2006d). Copper, zinc gold and silver are mined near the lake. Dewatering discharge has been deposited on the playa since 1997 and unlike many other discharges occurring in the Goldfields, this discharge ranges from meso to hypo saline. Lake Wownaminya is characteristically dry for most of the year (Outback Ecology 2006d).

Plate 4. Lake Wownaminya during the dry phase of its hydrocycle.

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Figure 6
CLASSIFICATION OF INLAND SALT LAKES IN WESTERN AUSTRALIA
YALGOO BIOREGION
Author: S. Gregory Scale: 1:400,000

3 SURFACE WATER CHEMISTRY

3.1 Abstract

Sporadic rainfall in recent times has contributed to the lack of knowledge with regards to water chemistry of the inland salt lakes in Western Australia. This was mainly due to limited opportunities to sample these ecosystems. The principal objectives of this chapter were to describe the water quality of the lakes, assess the differences in water quality between both lakes and Bioregions and to determine the potential impacts of dewatering discharge to water quality. In addition a comparison was made between water quality of the lakes and guidelines considered most relevant to these systems. The results indicated that highly variable water quality was characteristic of the lakes sampled and this was related to a combination of factors including the stage of the hydrocycle sampled, local geology and the input of dewatering discharge. Lakes such as Carey, Way and Miranda showed the greatest variation in most parameters and this was attributed to greater sampling effort at these particular lakes combined with the input of dewatering discharge. Differences in water quality between Bioregions were apparent, particularly in regards to metals. In terms of the impact of dewatering discharge on water quality, most lakes influenced by dewatering discharge have shown changes in pH and higher concentrations of salts, nutrients and some metals compared to natural lakes. Comparison of concentrations of metals in natural lakes and relevant ANZECC guidelines showed that cadmium, cobalt, chromium, copper, lead, nickel and zinc exceeded ANZECC guidelines indicating that concentrations of most metals are naturally high in surface waters of these lakes.

3.2 Introduction

The opportunity to collect surface water from the salt lakes of the study area has been limited due to sporadic rainfall and subsequent lack of filling events. In particular, the majority of studies conducted since the 1990s have been funded by the mining industry, with data being predominantly published as internal company reports, which are generally not publicly available. The purpose of this chapter is to compare and contrast the current water quality in terms of pH, salts, nutrients and dissolved metals of salt lakes in Western Australia. In total surface water data was available for 24 of the 42 lakes considered in this study. This represents the largest data set of its kind for water quality of inland salt lakes in Western Australia, and will contribute greatly to the knowledge of the water quality attributes of these systems.

The quality of surface water in these temporary systems is influenced by climate, geology and changes over the hydrocycle (Bayly and Williams 1966; Hammer 1986). The climate or rainfall patterns of the region determine how often the lakes fill and are important in the context of water quality (Biggs 1995). Upon filling catchment inputs such as salts from weathering of parent rock and input of organic material can influence the water quality of the lakes (De Deckker 1983). Internally as the lakes fill a pulse of nutrients occurs in the water column as they are released from the sediments (McComb and Qui 1998; Boulton and Brock 1999). As the lake completes the hydrocycle and starts to dry most parameters evapo-concentrate, resulting in increased concentrations of most parameters, particularly the salts (Khan 2003b). This results in a considerable variation in water quality over the hydrocycle, much greater than would be expected in permanent water bodies.

The earliest, most-comprehensive studies of surface water in the salt lakes occurred in the 1980s and compared the lakes of the Goldfields (Williams and Buckney 1976; Geddes *et al.* 1981), however these wetlands as a group have not been considered. Since these initial studies, most limnological studies have been restricted to lakes that have been impacted by mining, either in the lake itself, or on the periphery (Finucane 2004; van Etten 2004; Timms 2005a). This has led to a paucity of information for water quality given the large amount of work completed. In comparison, water quality of wetlands in other regions of Western Australia has been comprehensively studied particularly in response to impacts by secondary salinisation and urban development (Davis *et al.* 1993; Pinder *et al.* 2005; Boggs *et al.* 2007).

A number of the salt lakes included in this study have received inputs of saline to hypersaline groundwater, produced by the dewatering of nearby mines. While the influence of dewatering discharge to salt lakes are relatively unknown, it is thought that this activity contributes to the salt, metal and nutrient loads of these wetlands (van Etten *et al.* 2000; URS 2003; Timms 2005a; Outback Ecology 2006c). In some cases, dewatering discharge may be the only surface water present in these lakes for the majority of any given year. Therefore, while the results of this study reflect the water chemistry of these lakes, a number of them have been impacted by dewatering discharge and this is reflected in the water chemistry data.

The majority of studies tended to focus on salinity and concentrations of major anions and cations in the lake (Williams and Buckney 1976; Geddes *et al.* 1981). To date little information exists for metals, however anecdotal evidence suggest that because these lakes are located in naturally mineralized zones it would be expected that concentrations of some metals would be high in surface waters (Mann 1983). However, this has not been confirmed. This lack of knowledge has posed problems to both the mining industry and regulators of these systems, particularly in response to determining the impacts of dewatering discharge and mining operations on the water quality of these lakes.

In addition to the lack of knowledge with regard to the impacts of these activities there are no specific guidelines for assessment of these impacts to salt lakes in Western Australia. Traditionally, the ANZECC guidelines have been used in impact assessment to determine concentrations of parameters which may negatively impact the biota (ANZECC. 2000b). Although water quality of salt lakes in the Goldfields is not fully understood, the suitability of the guidelines in regards to these systems has been questioned in recent times (Smith *et al.* 2004).

The specific objectives of this chapter were to;

- describe the water quality of the salt lakes within the study area
- determine the similarities and differences between Bioregions and Lakes in terms of water chemistry
- comment on the impact of dewatering discharge to the surface water chemistry of the lakes
- compare and contrast the water quality of the lakes in comparison to relevant guideline values

3.3 Materials and Methods

Surface water samples for this study have been collected from 24 salt lakes in the period from 1998 and 2007. Due to the lack of filling events during the study water chemistry results were obtained from a number of unpublished sources. While the majority of data was collected by the author and staff and students of Curtin University, data collected for Lake Austin was supplied by Edith Cowan University (van Etten *et al.* 2000).

Each of the lakes was grouped according to Bioregion and whether the lakes had received dewatering discharge. For the purposes of this study those lakes receiving dewatering discharge were represented by a D, while those not receiving dewatering discharge were named natural lakes (N). It is unknown if natural lakes were impacted by another source, and generally this was not quantified in this study. Also, even though some of the larger lakes are classified as discharge (D), given the size of the lakes it is likely that some of the sites had not been impacted by the dewatering discharge. Those lakes with only baseline data collected, but have been impacted by discharge since were classified as natural lakes.

3.3.1 Field Methods

Surface water grab samples were taken in the littoral zone, using sampling containers relevant to the different analyses. Samples to be analysed for major cations, anions, pH and salinity were collected in 1 L plastic bottles with no preservative added. Samples to be analysed for nutrients such as phosphorus and nitrogen were collected in 250 mL plastic bottles and preserved. Samples analysed for dissolved metals were filtered within a water filter unit through 0.45 µm Millipore glass filter paper, and placed in a 125 mL plastic vessel with nitric acid added. Samples were stored in insulated containers with ice and sent to a NATA-accredited laboratory as soon as possible after sampling. Analysis of metals by the laboratory was conducted by Inductively Couple Plasma Mass Spectrometry (ICP-MS) (Appendix B).

In most cases field measurements of pH, electrical conductivity, dissolved oxygen and temperature were taken using a hand held meter, generally however most data used in this study are laboratory based. Water depth was measured using a standard ruler.

3.3.2 Data Compilation and Statistical Analysis

Results for water quality parameters were tabulated in excel spreadsheets and analysed using the PRIMER (Plymouth Routines in Multivariate Ecological Research) (version 6) and Minitab (version 14) software packages (Minitab Incorporated 2003; Clarke and Gorley 2006).

Ordination of the data was performed using PRIMER's Principal Components Analysis (PCA) to assess similarities between the lakes in terms of concentrations of certain parameters in surface water. When compiling the data, results that were below detection were altered to absolute values and halved (Lim *et al.* 2005). PCA produces a plot on which sites with similar water quality are located close together, while those with different chemistries are located further apart. Vectors on the plot represent the influence of the different parameters on the data set. The longer the vectors the greater influence on the data set. The strength of the PCA results are explained in terms of percentage variation, a value that should exceed 60 % over the first two axes, in order to adequately represent the data set (Clarke and Gorley 2006).

Minitab was used to perform a One-way Analysis of Variance (ANOVA) on data to determine if selected water quality parameters were significantly different between the lakes and for comparison natural and discharge sites (p values of <0.05 were considered significant, at a confidence interval of $\alpha = 0.05$).

Additionally, to determine the salinity range of all the wetlands, electrical conductivity (EC) data were converted to g/L (This was completed using the conversion factor presented in Williams, 1998a).

For the compilation of the dissolved metal concentrations in surface water, the table of ranges details the maximum and minimum concentration for each parameter. This was due to at least 80% of the data being below detection values. This limited the statistical analysis which could be performed on the data. The current ANZECC guidelines were also included in this table and these represent the level for which 80% of species are protected in a moderately disturbed marine ecosystem (ANZECC. 2000a). These guidelines were chosen for comparison on the basis that the study lakes are saline, and slightly too moderately affected by human activity.

3.4 Results

The collated water quality data for 24 inland waters showed that the parameters tested were not always consistent between lakes, and/or between sites. Therefore the data sets for each of the parameters tested differed in size, which limited statistical analysis in some cases.

3.4.1 pH

The pH of the lakes ranged between 4.6 and 10.6, however the majority of sites exceeded 7 (Appendix C – Table 1). Some lakes, such as Binneridgie Road Marsh and Lake Why, were alkaline recording pH values ranging between 9 and 10. In contrast, Lake Yindarlgooda recorded the most acidic surface water at 4.6. Lakes Austin, Yindarlgooda and Wownaminya showed the greatest variation in pH over time. Surface water pH was significantly different between Bioregions ($p<0.05$), with those lakes situated in the Coolgardie Bioregion tending to be more acidic, while those situated in the Avon Wheatbelt Bioregion were more alkaline. While pH was not significantly different between the discharge and natural lakes within the Avon Wheatbelt Bioregion, it was within the Coolgardie and Murchison lakes (Table 3). In the Coolgardie Bioregion pH was significantly higher in the discharge lakes in comparison to the natural lakes. In the Murchison Bioregion, this trend was reversed with the natural lakes reporting a significantly higher pH than the discharge lakes.

Table 3. Results of the ANOVA analysis showing the differences between pH in the natural and discharge lakes in each of the Bioregions. * represents a significant relationship.

Code	Avon Wheatbelt			Coolgardie			Murchison		
	Mean	SD	n	Mean	SD	n	Mean	SD	n
Discharge	7.77	0.33	43	7.65	0.64	300	6.76	0.64	32
Natural	7.93	0.38	3	6.71	0.97	46	7.48	1.28	76
P value		0.405			0.000*			0.003*	

3.4.2 Salinity

In terms of salinity (measured as TDS), a wide range of values were recorded (Appendix C – Table 2). The salinity of the lakes ranged from 1 to 390 g/L. The lakes with the highest salinity included Lakes Cowan, Tee and Fore, and those with the lowest salinity were Binneridgie, Lake Why and Swan Refuge. The data set for each of these lakes was small, which may have influenced results. Salinity was significantly different between Bioregions, with the salt lakes of the Avon Wheatbelt recording a significantly higher salinity than those of the Coolgardie and Murchison ($p<0.05$) (Appendix D). The salinity of the lakes within the Coolgardie Bioregion was intermediate to the Avon Wheatbelt and Murchison Bioregions.

Salinity was significantly different between the natural lakes and those lakes receiving dewatering discharge in both the Murchison and Coolgardie Bioregions ($p<0.05$ respectively) (Table 4). In both these Bioregions, lakes receiving dewatering discharge recorded significantly higher TDS values. There was no relationship between discharge and natural sites within the Avon Wheatbelt Bioregion.

Table 4. Results of ANOVA analysis including Mean, standard deviation and number of records for TDS (g/L) in surface water from each Bioregion. Significantly different relationships between discharge and natural lakes are highlighted by a *.

Code	Avon Wheatbelt			Coolgardie			Murchison		
	Mean	SD	n	Mean	SD	n	Mean	SD	n
Discharge	191	100	40	272	65	26	140	122	320
Natural	227	107	6	85	74	41	27	9	40
P value	0.421			0.000*			0.000*		

3.4.3 Anions and Cations

Nine different cationic sequences were reported from the surface water of the lakes in this study (from 311 records). The most common sequence was Na>Mg>K>Ca, followed by Na>Ca>Mg>K and Na>Mg>Ca>K. The percentage of sites reporting each sequence was 46%, 24% and 22% respectively. Results of a one-way ANOVA using salinity of each of these sequences showed that they reported significantly different salinity ($p=0.000$) and was highest for Na>Mg>K>Ca and lowest for Na>Mg>Ca>K, indicating the possibility that ionic balance was related to salinity.

For the cationic sequences, most sites in Lake Miranda, Lake Lefroy, Lake Fore, Creek Lake, Yarra Yarra Lakes, Lake Way, Lake Raeside and Lake Carey generally recorded an ionic balance of Na>Mg>K>Ca. Most sites in White Flag Lake, South-west Lake, Lake Zot, Lake Austin and Black Flag Lake reported a cationic sequence of Na>Mg>Ca>K. The cationic sequence Na>Ca>Mg>K was also reported at a number of sites in both Lake Carey and Lake Miranda.

The number of ionic patterns for each lake was calculated and ANOVA performed to determine whether there were any differences between natural and discharge lakes (Table 5). The number of different ionic balances was significantly higher ($p<0.05$) in lakes receiving dewatering discharge (mean of 2.7) in comparison to the natural lakes (mean of 1.1).

Table 5. Results of the ANOVA for number of different ionic balances in either the discharge or natural lakes. Mean, standard deviation (SD) and number of records for each group are presented.

Code	Mean	SD	n
Discharge	2.70	1.95	10
Natural	1.13	0.35	8
P value		0.039	

The size of the lake and the number of different ionic patterns was also investigated using ANOVA analysis (Table 6). The lakes greater than 50 km^2 in size had a significantly higher ($p<0.05$ number of ionic balances than the lakes less than 50 km^2.

Table 6. Results of the ANOVA for number of different ionic balances in each size group. Mean, standard deviation (SD) and number of records for each group are presented.

Size	Mean	SD	n
<10 km^2	1.40	0.70	10
10 – 50 km^2	1.00	0.00	2
>50 km^2	3.33	2.25	6
P value		0.038	

Of the 295 records of anionic sequences, 293 reported an anionic sequence of $Cl>SO_4>HCO_3$, while two lakes reported a sequence of $Cl>HCO_3>SO_4$. These two lakes were Lake Eaton North and Lake Why, which only had data from one sampling date. It is therefore unknown whether this result is representative of these lakes.

3.4.4 Dissolved Metals and Metalloids

The range of values for most metals and metalloids recorded in the Goldfields was large, however there were a high number of below detection values within the data set (Table 7). Two sets of ranges are presented, those from lakes receiving dewatering discharge and natural lakes. Concentrations of beryllium, tellurium, tin, tungsten, hydrocarbons (C10-14, C15-28, C29-36 and C6-9) and vanadium remained below detection in both lake types and are not presented in Table 7. Data sets were small for each of these parameters (±10 records each) with the exception of vanadium (32 records).

Generally the upper range of dissolved metals in the discharge lakes was higher than that recorded in the natural lakes (Table 7). This was not able to be statistically proven given the high number of below detection values in the data set. The exceptions to this were the upper range of aluminum, gold and iron at the natural lakes.

The upper range of concentrations of cadmium, chromium, cobalt, copper, lead, nickel and zinc in the natural lakes all exceeded the ANZECC guideline value for protection of 80% of species in marine water (Table 7). For the lakes receiving discharge the upper range of values, exceeded all ANZECC guideline values.

Table 7. Range of metal concentrations recorded in the salt lakes included in this study.

Parameter	Discharge Lakes			Natural Lakes			ANZECC Guidelines
	Min	Max	n	Min	Max	n	
Aluminum	BD	4.7	107	BD	44	48	
Antimony	BD	2.5	149		NT		
Arsenic	BD	2.8	294	BD	0.1	50	
Barium	BD	2	37	0.1	0.1	1	
Boron	0.1	37.2	130	0.8	4.4	5	
Bromine	14.3	83.4	8		NT		
Cadmium	BD	3.6	298	BD	0.4	50	0.04
Chromium	BD	0.5	283	BD	0.5	10	0.09
Cobalt	BD	2.5	252	BD	0.5	9	0.15
Copper	BD	1	299	BD	0.8	54	0.008
Gold	BD	0.003	73	BD	0.004	30	
Iron	BD	99	143	BD	120	19	
Lead	BD	13	294	BD	1.9	53	0.01
Mercury	BD	0.002	262	BD		46	0.001
Nickel	BD	4.4	259	BD	3	50	0.56
Selenium	BD	1.7	230	BD		8	
Silica	BD	54	41		1.4	1	
Silver	BD	0.01	9	BD		5	0.003
Strontium	0.16	21	18	14	21	2	
Titanium	BD	2.8	8		NT		
Uranium	0.015	0.023	9		NT		
Zinc	BD	6.7	279	BD	0.6	50	0.04

All values reported in mg/L

BD = below detection, NT= not tested

***ANZECC guidelines are trigger values for protection of 80% of species in marine water.**

The PCA plot of metals in surface water was limited by an incomplete data set, with many lakes not tested for the full suite of parameters (Figure 7). Available data shows that there was a high degree of variation within some salt lakes, as shown by the spread of points on the plot, particularly for Lake Way and Lake Carey. In terms of between-lake differences, Lake Way generally had high concentrations of most metals in comparison to the other inland salt lakes. In contrast, Lake Miranda reported the lowest concentrations of mercury and arsenic in comparison to the other lakes. South West Lake and Creek Lake reported higher concentrations of nickel, zinc and copper in comparison to the other lakes. Concentrations of most of these metals were comparatively low in Lake Raeside, Black Flag Lake, Lake Lefroy and White Flag Lake.

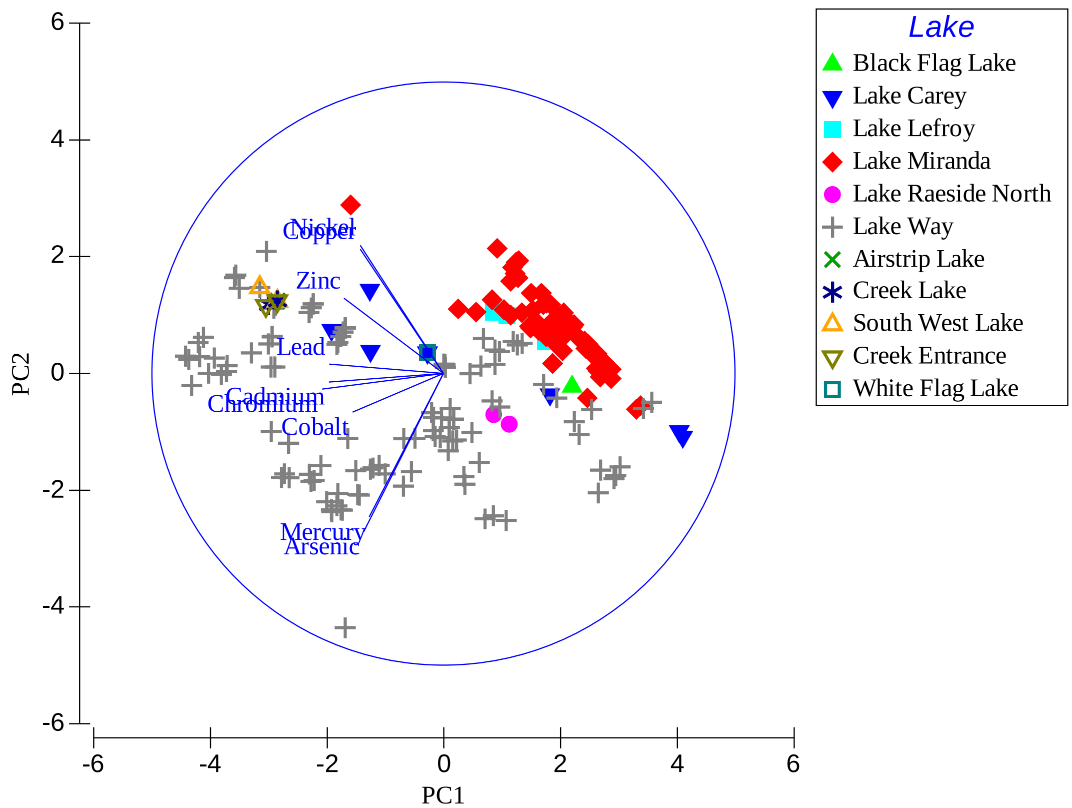

Figure 7. PCA plot of dissolved metal concentrations in surface water. A total of 68.4% of variation in the plot was explained by first two axes.

3.4.5 Nutrients

Total nitrogen ranged from 0.27 to 136 mg/L within the lake waters. The highest average concentrations was recorded in Creek Lake, while the lowest average total nitrogen concentrations was recorded in Swan Refuge and Black Flag Lake (Appendix C – Table 3). All lakes displayed a high degree of variation in total nitrogen values. With the exception of Creek Lake, those lakes impacted by dewatering discharge generally recorded higher concentrations of total nitrogen. Total phosphorus values were generally much lower than that of total nitrogen ranging from below detection to 42.3 mg/L. On average the highest concentrations of total phosphorus were recoded in White Flag and Lake Carey. However, both of these lakes recorded a high degree of variation in the data sets. The average N:P ratios ranged from 7:1 to 1000:1, with most sites demonstrating phosphorus limiting conditions within the surface waters. Total nitrogen and total phosphorus were not significantly different between natural lakes and those receiving dewatering discharge, or between Bioregions.

3.4.6 Combined Water Quality

The PCA plot of water quality for selected parameters demonstrated the high degree of variability between the lakes particularly Lake Miranda, Lake Way and Lake Carey (Figure 8). In terms of differences between the lakes, Lake Cowan recorded higher concentrations of calcium in comparison to all the other lakes and Creek Lake recorded high concentrations of total nitrogen and lower acidity than the other lakes.

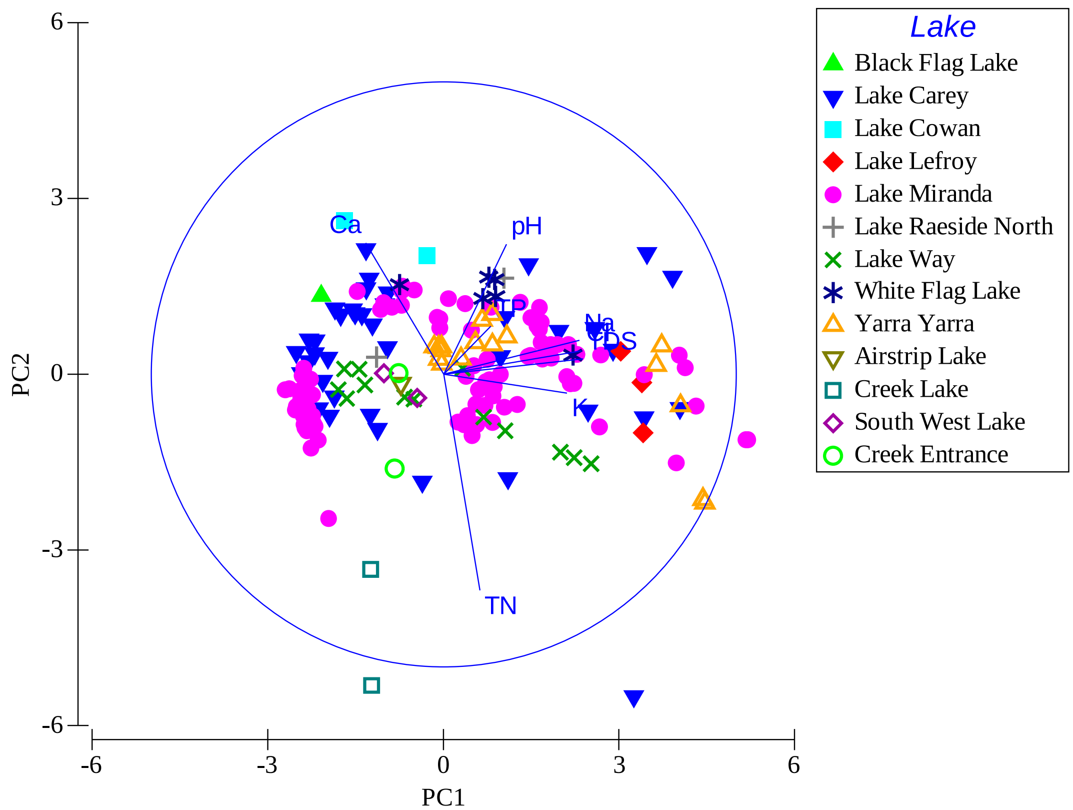

Figure 8. PCA plot of pH, TDS, major anions and cations and total nitrogen (TN) in the salt lakes of Western Australia. A total of 66.8% of variation is explained by the first two axes.

3.4 Discussion

Water quality of these salt lakes is highly dependant on the stage of the hydro-period (the cycle of wetting and drying) that the lake is sampled in (Boulton and Brock 1999). This, along with other factors results in a high degree of within-lake variation of water chemistry within these lakes. Sampling in different stages of this cycle results in a high variation in water quality (Williams and Buckney 1976; Williams 1998b; Boggs *et al.* 2007). Other factors which have the potential to influence the water chemistry of these systems include local geology, inputs from groundwater, and the potential of impact from mining (Boulton and Brock 1999).

3.4.1 pH

The pH of inland waters is generally alkaline (Williams and Buckney 1976; Geddes *et al.* 1981; Hammer 1986), as was the case for most of the lakes in this study. Overall, differences in pH of surface water occurred according to a combination of variables including geographic location, dewatering discharge inputs and groundwater chemistry. Within some of the larger lakes, such as Austin, Yindarlgooda and Wownaminya pH ranges were up to one pH unit, showing a considerable degree of variation within these lakes. This within lake variation was considered to be a result of the distance between sites which resulted in differences in geology and different catchment inputs at each site. Between lakes, the pH of the lakes appeared to be related to geographical location as pH was significantly different between the Bioregions, with the Coolgardie Bioregion lakes reporting significantly lower pH than all other regions. Acidic surface waters are common within the lakes of this region (Mann 1983; Douglas and Degens 2006). On average, lakes in the Yalgoo Bioregion recorded the more alkaline pH.

Within each Bioregion comparison of pH data between natural and discharge lakes showed some interesting trends. While in the Avon Wheatbelt Bioregion pH was similar between the two lake types, within the Coolgardie Bioregion discharge was significantly higher while in the Murchison discharge lakes reported significantly lower pH. Low pH values for surface waters are common within the Coolgardie region, which may explain the trend within this region (Douglas and Degens 2006). The trend of changing pH within lakes affected by dewatering discharge is common and is dependant on the quality of the dewatering discharge water entering the lake

(URS. 2001; Foster 2004). The implications of this for the mining industry is that changes in pH while directly affecting biota may also result in some metals becoming more soluble (ANZECC. 2000b).

3.4.2 Salinity

Surface water salinity (measured as TDS) of the lakes ranged from 3 to 390 g/L. This corresponds with results from other studies which include data from West Australian salt lakes, however the top end of the range recorded in this study is higher than other reported ranges (290 g/L) (Williams and Buckney 1976; Geddes *et al.* 1981; Boggs *et al.* 2007). Salinity was most likely related to the position in each of the lakes hydrocycle that was sampled (Brock and Shiel 1983; John 2003b; Boggs *et al.* 2007) and the occurrence of dewatering discharge. The lakes recording the highest salinities were those that had been impacted by dewatering discharge such as Lake Cowan and Tee. In contrast natural lakes such as Swan Refuge and Lake Why reported much lower salinities. It should be noted that these lakes were only sampled once, most likely corresponding with the start of the hydrocycle.

Salinity within the Avon Wheatbelt Bioregion was higher than that of the Coolgardie and Murchison Bioregions. While the source of salts within these lakes is contentious (De Deckker 1983), the Avon Wheatbelt is closest to the coast and oceanic salts may be the origin and therefore cause of higher salt loads in this region.

3.4.3 Major anions and cations

In terms of salinity, surface water in most lakes were dominated by Na and Cl, as commonly occurs in salt lakes in other regions of Australia (De Deckker 1983). The most common pattern of ionic dominance for the cations was Na>Mg>K>Ca and $Cl>SO_4>HCO_3$ for the anions in the lakes of this study. Australian salt lakes typically follow this sequence for both cations and anions however Ca and K are often interchangeable (Williams and Buckney 1976; Geddes *et al.* 1981; Radke *et al.* 2003). These two cationic balances accounted for approximately 68% of all sites tested. It is likely that the change in Ca and K was related to salinity as sites with the K>Ca balance reported significantly higher salinity than those with Ca>K. This trend has also been described in studies of other Australian inland waters (Hart and McKelvie 1986; Williams 1998b). The ionic sequence of Na>Ca>Mg>K was also relatively common in some lakes (24% of sites recorded this balance) indicating that the concentration of Ca was sometimes higher in some of the Western Australian lakes. This trend is common in water of low salinities (Williams and Buckney 1976). The concentrations of each particular ion is governed by inputs from the atmosphere by precipitation, the presence of marine derived salts and leaching from minerals in the area and typically displays little seasonal variation (Williams 1998b). Geddes et al. (1981) reported a typical anionic dominance of Cl>SO_4>HCO_3 in inland salt lakes and this trend was also reported in the majority of lakes in this study (only two sites recorded a different anionic balance).

Three lakes reported five different ionic balances during the course of this study. These were some of the larger lakes and included Lake Austin, Lake Carey and Lake Miranda. Having such variation in ionic balances may be due in part to the large size of the lake and therefore the number of different habitat and geological types present (Geddes *et al.* 1981; Williams 1998b). However it is also likely that the inputs of dewatering discharge may have contributed to the change in ionic dominance of some sites. In Lake Austin the sites classified as natural sites reported a unique ionic balance, which indicated that Ca was of greater concentration in the water than both Mg and Ca (i.e. Na>Ca>Mg>K). This trend was also recorded at Lake Miranda where K was of greater concentration in comparison to both Mg and Ca and the ionic balance Na>Mg>K>Ca was unique to the sites which were influenced by dewatering discharge. The displacement of Mg and Ca usually occurs in freshwaters (Geddes *et al*. 1981), and this may explain why the sites that were not impacted by dewatering discharge (which tend to be of lower salinity) recorded different ionic balances than the dewatering discharge sites.

3.4.4 Metals and metalloids

Assessment of dissolved metal and metalloid concentrations of surface waters in salt lakes in Western Australia has been limited over time to site-specific studies and generally, most of the data has been collected from dewatering discharge outfalls. Very little dissolved metals data has been collected from natural salt lakes. It has been reported that concentrations of lead, uranium, copper, and zinc are high in groundwaters within the Yilgarn Block (Mann 1983) and as a rule these parameters were higher in the lakes which received the dewatering discharge (sourced from groundwater), the exception being uranium which has not been tested in the natural lakes surface water. There were no clear trends in terms of differences in metal concentrations in surface water between Bioregions as the majority of data collected was from the Murchison Bioregion.

The use of the ANZECC guidelines for comparison with data from discharge-affected salt lakes is a common occurrence in Western Australia due to the lack of guidelines for metal concentrations within salt lakes (URS 2003). Concentrations of cadmium, cobalt, chromium, copper, lead, nickel and zinc in natural lakes exceeded ANZECC guidelines for protection of 80% of marine species. This indicates that concentrations of most metals are naturally high in surface waters of salt lakes in the Goldfields, regardless of dewatering discharge influence.

3.4.5 Nutrients

Concentrations of nitrogen were high in salt lakes of the Goldfields, a trend also reported in Victorian salt lakes (Khan 2003b). With the exception of one of the natural lakes (Creek Lake) total nitrogen remained below 2.5 mg/L. At the lakes impacted by discharge total nitrogen ranged from 1 to 10 mg/L and the general trend was for these lakes to have greater concentrations of nitrogen. This trend has been reported in individual lakes receiving discharge (Gregory in press) and anecdotal evidence suggests that blast chemicals used in the pit may be the source of nitrogen in the dewatering discharge water (L. Hill pers comm.).

Concentrations of total phosphorus within the lakes consistently remained below total nitrogen, a trend also common in other regions and freshwater systems (Boulton and Brock 1999; John 2003b). There were no differences in total phosphorus values between dewatering discharge and natural lakes or Bioregions.

3.5 Conclusions

Water quality in salt lakes of Western Australia was highly variable both within and between lakes. In addition, water quality was dependant on the stage of the hydrocycle, local geology and the input of dewatering discharge. The larger lakes such as Lake Carey, Way and Miranda showed a considerable degree of variation in water quality in comparison to the smaller lakes, particularly in regards to metals and major anions and cations. This was most likely due to greater monitoring effort at these lakes and greater variability in geology throughout the lake. These lakes have also received dewatering discharge which also appears to have affected the water quality of these lakes. Generally, those lakes receiving dewatering discharge have shown differences in pH, higher concentrations of salts, nutrients and some metals compared to natural lakes. Concentrations of cadmium, cobalt, chromium, copper, lead, nickel and zinc in natural lakes were elevated and exceeded ANZECC guidelines. This indicated that concentrations of most metals are naturally high in surface waters, regardless of dewatering discharge influence. In terms of differences between Bioregions, while some parameters such as nutrients and salinity were apparent, particularly in the Avon Wheatbelt, however there were insufficient data for metals within regions to determine differences in these parameters.

4 SEDIMENT CHEMISTRY

4.1 Abstract

Sediment parameters of salt lakes in terms of pH, salinity, metals, major salts and nutrients as a whole is poorly known in Western Australia. Within this chapter particular focus was placed on the impact of dewatering discharge on the sediments and the differences in sediment chemistry between both lakes and Bioregions. As shown in the surface water chapter, there was a high degree of variation in sediment chemistry both within and between salt lakes. This variation was generally related to geological features of the lakes and in some cases, the additional influence of dewatering discharge. The sediments of the Murchison Bioregion reported high concentrations of zinc, arsenic and copper, while the Coolgardie region sediments were typified by higher concentrations of lead, cadmium, chromium and nickel. The use of ANZECC Interim Sediment Quality Guidelines for impact assessment of dewatering discharge was also investigated. Generally, sediment values recorded in the natural lakes were lower than guideline values. The only exceptions were the trigger values of nickel and chromium which were exceeded. In regards to the uniqueness of the lakes, Lakes Maitland, Black Flag and Miranda were relatively distinctive in terms of the concentrations of metals in sediment, compared to the other lakes in this study.

4.2 Introduction

While water quality of the salt lakes was presented in the previous chapter, it was common for sediment to be the only media available to be sampled within the salt lakes of semi-arid regions, due to their episodic nature. Despite the fact that they are the media most often sampled, the level of published information on the sediments of the Western Australian salt lakes is minimal, particularly in regard to metals and nutrients. Published information to date is limited to early studies based on one-off sampling events, conducted in 1977 and 1990 (Förstner 1977; Arakel *et al.* 1990; Lyons *et al.* 1990). A number of mining companies sample sediments of the lakes on a regular basis, both due to the lack of water, and as a means of determining the impact of their operations on the lakes given that sediment chemistry can directly influence water chemistry (ANZECC. 2000b; Simpson *et al.* 2005).

Sedimentary processes are both complex and numerous within the inland salt lakes. Sediments of the lakes are composed of particles sourced from the weathering of materials and matter produced from organic processes such as decay within the lakes (Clark and Wasson 1986). The bedrock or underlying geology is the main influence on sediment chemistry within the natural salt lake environment (Förstner 1977; Lyons *et al.* 1990). Groundwater quality also has a considerable influence on sediment quality in many lakes (De Deckker 1988).

In addition to groundwater and weathering inputs the wetting and drying of the water-body can result in profound changes to the sediment properties and chemistry (McComb and Qui 1998). For example, when sediments are submerged the levels of oxygen in the sediment are drastically reduced, resulting in several changes (Ponnamperuma 1972). As sediments are wetted, the surface layer (<2 mm) is oxygenated, and overlies an anoxic layer (Ponnamperuma 1972). In this layer, the solubility of metals can change, either binding to sediment or releasing them into the overlying water (Boulton and Brock 1999). Upon drying, sediments of the salt lakes are influenced by erosion and depositional forces mainly as a result of windblown movement (Clarke 1991).

The sedimentary processes in Western Australian salt lakes are poorly known, particularly in response to dewatering discharge to the playas. This chapter serves to compare and contrast the data collected in regards to pH, salinity, major anions and cations, nutrients and metals between lakes and Bioregions. The effects of dewatering discharge on these sedimentary parameters and the effectiveness of the ANZECC interim sediment quality guidelines in impact assessment will also be discussed.

4.3 Materials and Methods

4.3.1 Field Methods

Sediment samples were collected using 250 mL sterilized glass jars on the fringes of each lake. The top 1-5 cm of sediment was scraped into the sample vessel. Samples were stored in insulated containers with ice bricks and transferred to the laboratory for analysis. Except where specifically mentioned, total metal values have been presented as mg/kg. Laboratory methods for sediment analysis are presented in Appendix B.

While the parameters tested differed between sites, generally sediments were tested for the following;

- Basic parameters – pH, total soluble salts (TSS), total organic carbon (TOC), total nitrogen (TN), total phosphorus (TP)
- Major Cations and Anions – sodium (Na^+), calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+), chloride (Cl^-), sulphate (SO_4^{2-}), bicarbonate (HCO_3^-)
- Total Metals – aluminum (Al), arsenic (As), barium (Ba), boron (B), cadmium (Cd), chromium (Cr), cobalt (Co), copper (Cu), iron (Fe), lead (Pb), manganese (Mn), mercury (Hg), nickel (Ni), selenium (Se), strontium (Sr), sulfur (S), uranium (U), vanadium (V) and zinc (Zn)

For data comparison, lakes receiving dewatering discharge were separated from lakes not receiving discharge (referred to as natural lakes hereafter). The lakes were also grouped according to the Bioregion in which they occur, as described in Chapter 2.

Sediments were additionally classified according to texture (McDonald *et al.* 1998). Generally the three types of sediments were present within these lakes, falling into three categories; sand, sandy clay and clay.

Sediments collected from lakes within the Avon Wheatbelt region were only tested for basic parameters, and have not been assessed for major anions and cations or metals during this study.

4.3.2 Data Compilation and Statistical Analysis

Results for sediment quality parameters were tabulated in Excel spreadsheets and analysed using the PRIMER (version 6) and Minitab (version 14) software packages.

PRIMER's Principal Components Analysis (PCA) was used to assess similarities between the lakes in terms of concentrations of total metals in sediment (Clarke and Gorley 2006). When compiling the data, results that were below detection were altered to absolute values and halved (Lim *et al.* 2005). PCA analysis produces a plot on which sites with similar sediment quality are located close together, while those with different chemistries are located further apart. Vectors were also included on the plot and they represent the influence of the different parameters. Longer vectors indicate a greater influence on the data set. The strength of the PCA results are explained in terms of percentage variation, a value that should exceed 60 % over the first two axes, in order to adequately represent the data set (Clarke and Gorley 2006). Sites included in the PCA plot were grouped according to lake and Bioregion.

Minitab was used to perform a One-way Analysis of Variance (ANOVA) to determine if sediment parameters were significantly different between the lakes (p values of <0.05 were considered significant, at a confidence interval of $\alpha = 0.05$).

For total metal concentrations within sediments, the maximum and minimum concentration for each parameter was calculated. The third quartile value was also included and represents the value that 75% of data is below. The current Australian and New Zealand Environment Conservation Council (ANZECC) Interim Sediment Quality Guidelines were also incorporated for comparison with natural lake values (ANZECC. 2000a).

4.4 Results

4.4.1 pH

The pH of sediments within the salt lakes ranged from 4.0 – 9.4, however most lakes exceeded 7 (Appendix E –Table 1). The exceptions to this were Creek Lake, Lake Rebecca and South-West Lake which generally recorded a pH value below 7. The pH of sediments was significantly different between Bioregions ($p<0.05$) with sediments of lakes in the Coolgardie Bioregion recording a more acidic pH while those of the Yalgoo recording a more alkaline pH. There was a high degree of within-lake variation in sediment pH in a number of lakes, particularly Lake Rebecca, Lake Miranda and Lake Johnston, which recorded up to 1 pH unit variation between sites.

Comparison of pH in sediments between natural lakes and those receiving dewatering discharge between Bioregions showed that in both the Coolgardie and Murchison regions pH was significantly higher in lakes receiving dewatering discharge ($p<0.05$) (Table 8). This was not the case within the lakes of the Avon Wheatbelt bioregion with sediment being similar between lake types.

Table 8. Results of the ANOVA, including mean, standard deviation (SD), number of records (n) and p value of pH within sediments of the Bioregions. Significant values are represented by a *.

Code	Avon Wheatbelt			Coolgardie			Murchison		
	Mean	SD	n	Mean	SD	n	Mean	SD	n
Discharge	8.1	0.4	38	7.7	0.5	139	7.8	0.6	391
Natural	8.1	0.5	29	7.4	0.9	71	7.5	0.9	43
P value	0.688			0.004*			0.008*		

4.4.2 Salinity

Salinity (measured as TSS) in sediments displayed a high degree of variation both within and between lakes (Appendix E – Table 2). Sediment salinity in all lakes ranged from 1.6 to 514 g/kg. Salinity was significantly different between Bioregions with the Yalgoo recording the lowest salinity in sediments ($p<0.05$). In contrast, the sediments of the Avon Wheatbelt region recorded the highest salinity in comparison to the other Bioregions. In terms of mean sediment salinity the Un-named Lake recorded the highest mean salinity while Lake Wownaminya reported the lowest mean sediment salinity. Lakes which recorded a considerable range of sediment salinity included Lake Cowan, Lake Carey and Lake Lefroy.

In the Coolgardie Bioregion sediment salinity of the discharge lakes was significantly higher than that of the natural lakes ($p<0.05$) (Table 9). In the Murchison Bioregion while the mean value of salinity in sediments was higher in the discharge, this was not significant ($p>0.05$). For the lakes of the Avon Wheatbelt, sediment salinity was significantly lower at the lakes receiving dewatering discharge ($p<0.05$).

Table 9. Results of ANOVA including, mean, standard deviation of TSS (g/kg) in salt lake sediments from each bioregion. Significant relationships are highlighted by a *.

Code	Avon Wheatbelt			Coolgardie			Murchison		
	Mean	SD	n	Mean	SD	n	Mean	SD	n
Discharge	125	95	36	112	84	314	124	101	131
Natural	192	121	27	89	50	27	96	67	50
P value	0.021*			0.152			0.034*		

4.4.3 Major Anions and Cations

A PCA plot of major cations and anions in sediment, Na^+, Ca^{2+}, Mg^{2+}, K^+, Cl^-, SO_4^{2-}, HCO_3^- showed some differences between the lakes (Figure 9). CO_3^{2-} was not included in this assessment as the majority of lakes recorded concentrations of less than 1 mg/kg. Lakes Maitland and Way reported a high degree of within-lake variation in comparison to the other lakes as shown by the spread of sites on the plot. Also higher concentrations of Ca^{2+}, $Mg^{2+}$and HCO_3^- were reported in these lakes in contrast to the other lakes. Lakes Raeside North and the Un-named Lake reported higher concentrations of Na^+ and Cl^- in comparison to the others. Concentrations of most parameters in Kopai Lake and Southern Star were low compared to other lakes. Lakes such as Lake Rebecca, Lake Johnston and White Flag Lake showed little within-lake variation, and were also similar to each other as indicated by their position on the plot.

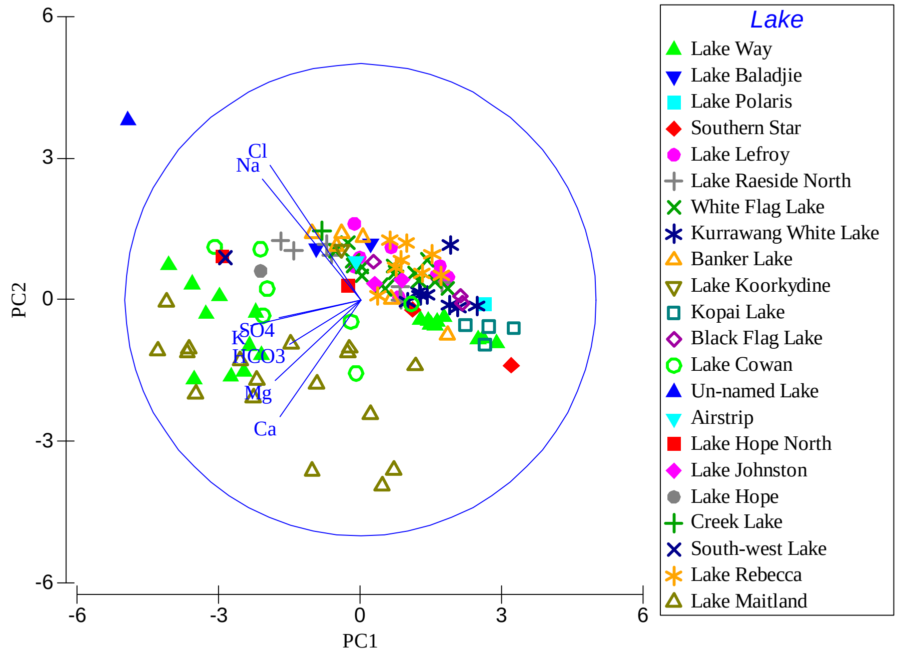

Figure 9. PCA plot of major anions and cations in WA salt lake sediments (all Bioregions). A total of 70% of variation explained by the first two axes.

For the cations within the sediments, the majority of sites were dominated by sodium, however there were some lakes dominated by calcium and magnesium (Table 10). In contrast anions, recorded the ionic balance of Cl>SO_4>HCO_3, in all but two sites.

Table 10. The ionic gradient of major anions and cations recorded in Western Australian salt lake sediments.

Ionic Balance	Proportion of Sites (%)
Na>Ca>Mg>K	33
Na>Mg>Ca>K	26
Ca>Na>Mg>K	15
Na>Mg>K>Ca	14
Na>Ca>K>Mg	5
Ca>Mg>Na>K	4
Mg>Ca>Na>K	2
Mg>Ca>K>Na	1

4.4.4 Total Metals and Metalloids

The mean total metal value at the discharge lakes from all Bioregions was generally higher in comparison to the mean concentrations recorded in the natural lakes (Table

11). The exceptions to this were boron, strontium and uranium which were higher in the natural lakes. Minimum and maximum ranges of metals within the lakes are presented in Appendix E. Maximum concentrations of chromium and nickel in the natural lake sediments exceeded the ANZECC interim sediment quality guideline values (Table 5 - Appendix E).

Concentrations of tellurium, tin, tungsten and silver were consistently below detection in both discharge and natural lakes, so data were not included in Table 11.

Table 11. Mean, standard deviation (Std dev) and number of records (n) for total metal and metalloid concentrations recorded in Western Australian salt lake sediments of natural and discharge lakes.

Parameter	Discharge Lakes			Natural Lakes			ANZECC Guidelines
	Mean	Std Dev	n	Mean	Std Dev	n	
Aluminum	9843	5587	182	5744	3809	25	
Arsenic	52	191	553	3	2	49	
Barium	41	48	148	35	39	42	
Boron	106	95	209	159	105	17	
Cadmium	2	10	563	0.5	0.5	49	10
Chromium	132	172	563	118	140	51	350
Cobalt	22	45	392	7	7	44	
Copper	59	147	574	16	14	51	270
Iron	34855	30591	146	12437	9631	26	
Lead	10	17	562	7	10	49	220
Manganese	368	471	483	179	171	42	
Mercury	0.1	0.1	515	0.1	0.1	42	1
Nickel	46	136	521	31	39	51	52
Selenium	3	4	251	1.2	0.9	25	
Strontium	496	347	18	886	985	17	
Sulfur	8055	2555	20	4110	4231	3	
Uranium	5	5	10	16	14	17	
Vanadium	64	38	138	62	55	25	
Zinc	36	36	550	17	15	51	

All values reported in mg/kg, BD = below detection, NT= not tested

***ANZECC guidelines are ISQG-High Values**

The PCA plot of total metal concentrations showed that some lakes, such as Lake Maitland, Lake Miranda and Black Flag Lake, reported distinct concentrations of total metals in comparison to the other lakes included in this study (Figure 10). For example, concentrations of chromium, nickel and lead were highest in Black Flag Lake in comparison to the other salt lakes. Concentrations of arsenic, copper and zinc were highest in Lake Miranda and Lake Raeside North. Concentrations of most other metals in sediment were low in Lake Maitland and Lake Rebecca in association to the other lakes.

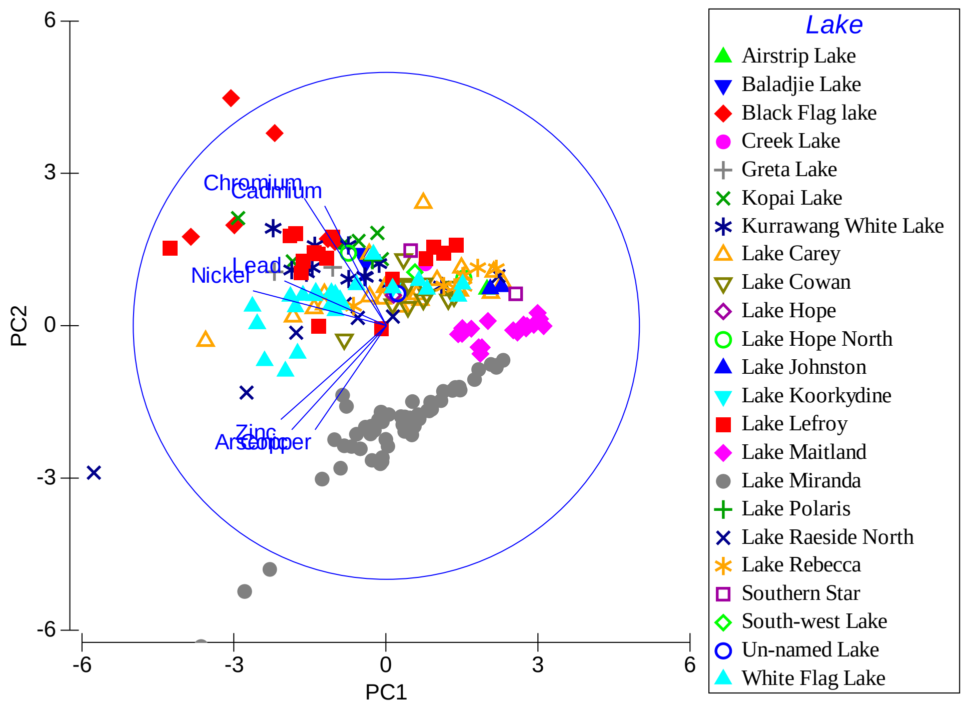

Figure 10. PCA plot of total metal concentrations in sediment (all Bioregions). A total of 67.6% of variation was explained by first two axes.

There were some clear differences in terms of concentrations of metals between the Bioregions (Figure 11). For example the lakes in the Coolgardie region were characterized by higher concentrations of cadmium, chromium, nickel and lead. In contrast lakes in the Murchison Bioregion were generally characterized by higher concentrations of zinc, arsenic and copper and low concentrations of chromium, cadmium, nickel and lead.

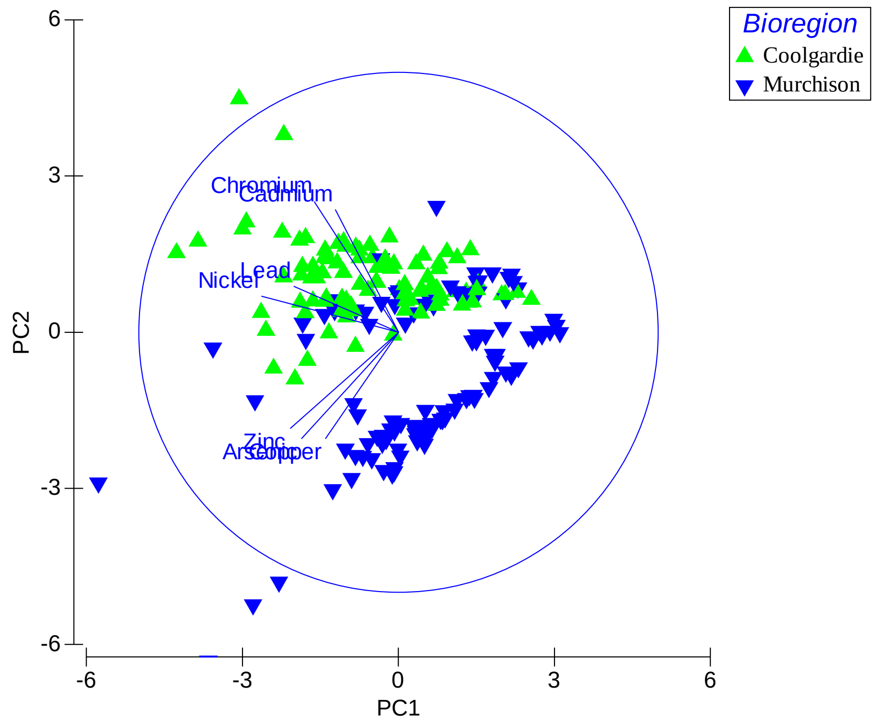

Figure 11. PCA plot of metals in Western Australian salt lake sediments, grouped according to Bioregion. A total of 67.6% of variation was explained by first two axes.

4.4.5 Nutrients

Of all the lakes, sediments at Lake Hope reported the lowest average concentration of total nitrogen (Appendix E – Table 3). In contrast total nitrogen concentrations were highest in Kopai North. While total nitrogen was high in Kopai North, the lake recorded the lowest mean value for total phosphorus. The highest total phosphorus average value was recorded in the Un-named Lake.

In terms of differences in nutrients between natural and lakes receiving dewatering discharge in Bioregions, total nitrogen was significantly higher in natural lakes in the Coolgardie Bioregion ($p<0.05$). There were no other significant relationships between the nutrients and their concentrations in natural lakes or those lakes receiving dewatering discharge.

Total nitrogen was also significantly different between Bioregions, with the lakes of the Avon Wheatbelt reporting significantly higher total nitrogen in sediments than the other regions ($p<0.05$). Lakes of the Yalgoo Bioregion reported the lowest total nitrogen concentrations. Total phosphorus concentrations were also significantly different between Bioregions and were significantly higher in the Yalgoo sediments and lowest in the sediments of the Coolgardie lakes ($p<0.05$). These trends differed from that recorded in the surface water of these lakes, therefore may not be related to dewatering discharge.

4.4.6 Total Organic Carbon

Total Organic Carbon (TOC) in sediments ranged from below detection to 6.9 mg/kg. Creek Lake, Un-named Lake, South-west Lake and Lake Johnston recorded high average concentrations of TOC (Appendix E – Table 4). In contrast, concentrations of TOC in sediments of Lakes Kopai and Lake Wownaminya were below detection. TOC was significantly higher in the sediments of the natural lakes within the Coolgardie and the Avon Wheatbelt Bioregions ($p<0.05$) (Table 12). TOC in sediments of the natural lakes of the Coolgardie Bioregion was significantly higher than that of the lakes receiving dewatering discharge.

Table 12. Mean, standard deviation and number of records used in the ANOVA analysis for TOC (%) between discharge and natural lakes within each Bioregion.

Code	Avon Wheatbelt			Coolgardie			Murchison		
	Mean	**SD**	**n**	**Mean**	**SD**	**n**	**Mean**	**SD**	**n**
Discharge	0.38	0.18	26	1.05	1.32	156	0.56	0.59	96
Natural	0.27	0.14	15	1.28	1.12	28	1.36	1.72	35
P value		0.027*			0.397			0.000*	

4.4.7 Sediment type

Three sediment types were present the in the lakes of the study; sandy clay, clay and sand. The majority of sites were sandy clay or clay, with only a couple of sand sites occurring within the lakes. The following parameters were significantly higher in the clay sediments; aluminum, antimony, moisture content, nitrite, potassium, sulphate, tin and total soluble salts. The following parameters were significantly higher in the sandy clay sediments; boron, nitrate + nitrite and total nitrogen. The following parameters were significantly higher in the sandy sediments; beryllium, cadmium, copper, iron, lead, total organic carbon and zinc.

4.5 Discussion

The analysis of sediments in the absence of surface water may allow assumptions to be made with regard to both water chemistry and biological communities that may be present in the particular water body (Clark and Wasson 1986). This is particularly useful given the extended length of time in which these systems remain dry. Mining companies using salt lakes in Western Australia collect most of their data during the dry phase of the hydrocycle and it is common practice in Western Australia for salt lake sediments to be sampled to determine the impact of dewatering discharge. Also numerous baseline studies occur when the lake is dry, as the likelihood of wet sampling for the purposes of baseline or impact studies is low due to the episodic nature of the lakes (Gregory *et al.* 2006).

Like water quality, the sediments of salt lakes in Western Australia showed a large degree of variation in terms of chemistry both within and between the lakes. Sediment quality appeared to be related to the sites location, its associated geology and the occurrence of dewatering discharge. While not specifically addressed in this study, there are a number of changes in sedimentary processes which occur over the wetting or drying cycle which may have also contributed to differences in sediment chemistry between lakes (Boulton and Brock 1999). Also, in aquatic systems there is a considerable degree of heterogeneity within sediments (Simpson *et al.* 2005).

4.5.1 pH

Within the sediments of these lakes, pH ranged from 4 to 10.6, however most of the sediments exceeded pH seven. The pH of sediments in temporary systems such as these are controlled by a number of factors including the hydrocycle, inputs from groundwater recharge, redox reactions, carbonate and organic matter concentrations (Ponnamperuma 1972; Commander 1999). The lakes of the Coolgardie Bioregion, such as Creek Lake, South-west Lake were the most acidic in this study and this may be related to the influence of acidic groundwater in the south western section of the Yilgarn block (Mann 1983). It is common for lakes in this region to receive discharges of acidic groundwater on the periphery of the water body (Commander 1999). In addition a number of lakes in this region reported high concentrations of organic matter which may have contributed to the acidic conditions.

Sediment pH was significantly higher in sediments of the discharge lakes in comparison to the natural lakes. Changes in sediment pH are important in the context of determining impact of dewatering discharge as pH plays an important role in the availability of some metals (Miao *et al.* 2006). Metals likely to be made more soluble and toxic by a higher sediment pH include metals such as aluminum. In contrast, the toxicity of lead, zinc and nickel and numerous other metals decreases as pH increases (ANZECC. 2000b).

4.5.2 Salinity

The sources of salts in sediments of salt lakes can result from the weathering of minerals, marine sources and rainfall (Arakel *et al.* 1990; Chivas *et al.* 1991; Clarke 1994). In conjunction to these sources, dewatering discharge in a number of the Western Australian salt lakes has contributed to the salt load of the sediment (Finucane 2004; Foster 2004). Sediment salinity was significantly different between Bioregions, with the lakes of the Murchison recording the highest salinities, followed by Coolgardie, and then the lakes of the Yalgoo. The lower salinity of the Yalgoo Bioregion was most likely related to the fresh dewatering discharge received by the solitary lake studied in the region, which most likely contributed to flushing salts from the sediment of these lakes (Outback Ecology 2006d). The higher salinity recorded in sediments of the Murchison may be related to greater proportion of sites receiving discharge in that region in comparison to that of the Coolgardie region (i.e. 75% of sites received discharge in the Coolgardie region compared to 92% of sites in the Murchison region). In the Coolgardie region the sediment salinity was significantly higher in the discharge lakes in comparison to the natural lakes. This was not the case in the Murchison, where the discharge lakes recorded a larger range of sediment salinities. This was particularly prevalent in the larger lakes such as Carey, Miranda and Way where the dewatering discharge did not impact the whole lake (Outback Ecology 2006c; e).

4.5.2 Major anions and cations

In terms of major anions and cations, the sediments of the salt lakes included in this study were dominated by sodium and chloride in line with other studies in the region (Mann and Deutscher 1978; Arakel *et al.* 1990). Generally, the cations followed the order of dominance of Na>Ca>Mg>K at the majority of sites. For the anions Cl^- was dominant, followed by SO_4^{2-} and HCO_3^-. The majority of lakes sediments in this study were characterised by low concentrations of CO_3^{2-}, similarly to other lakes in the region (Förstner 1977). There were however some deviations from this trend. For example, higher concentrations of calcium and magnesium within Lakes Way and Maitland may be related to the presence of calcrete within the vicinity of these lakes (Mann and Deutscher 1978).

4.5.3 Total metals and metalloids

The acidic waters in the south western section of the Yilgarn block contain high concentrations of lead (Förstner 1977; Mann 1983), as was observed within the lake sediments of the Coolgardie Bioregion. It has been suggested that the source of the lead in these systems is most likely related to the occurrence of granite within these regions (Mann 1983). Mann (1983) indicates that the more acidic and saline groundwater conditions in the south west of the Yilgarn plateau contributes to the remobilization of lead. Other studies indicate that lead concentrations are highest in lakes receiving discharge from acid groundwaters (Lyons *et al.* 1990). In relation to this, the more acidic sediments in this study were also reported within the Coolgardie bioregion. Concentrations of cadmium, chromium and nickel were also higher in lakes of this Bioregion, a characteristic reported by Förstner (1977). In contrast, lakes within the Murchison region generally recorded higher concentrations of zinc, arsenic and copper.

It appears that dewatering discharge has been a source of arsenic, cadmium, chromium, cobalt, copper, iron, lead, mercury, nickel and zinc, with elevated levels of these parameters being recorded in some of the lakes. In contrast in the natural lakes concentrations of most metals were lower than ANZECC guidelines except for concentrations of nickel and chromium. It appears that sediments of the study area are characteristically high in these parameters. This was particularly prevalent within sediments of the Coolgardie Bioregion.

Lakes Maitland, Black Flag and Miranda all reported fairly unique sediment chemistry in terms of metals. Sediments of Lake Maitland were characterized by low concentration of parameters such as arsenic, zinc and copper in comparison to the other lakes in this study. Higher concentrations of cadmium and chromium reported at Black Flag may be related to local geology. Concentrations of arsenic, zinc and copper were high in the Lake Miranda sediments in comparison to the other lakes and this was related to the presence of dolomite within sediments of this lake (Arakel *et al.* 1990).

While some studies have reported high concentrations of uranium in salt lakes in the northern Yilgarn (Mann 1983), only two lakes in this study have been tested for uranium. Lakes Way and Maitland are in areas which contain uranium bearing calcrete and are therefore expected to have high concentrations of this parameter (Mann 1983; White 2000). Uranium concentrations in these lakes ranged from 1 to 45 mg/kg.

In terms of metals concentrations of cobalt, nickel and chromium are relatively high in sediments of Western Australian salt lakes in comparison to other Australian salt lakes (Förstner 1977). Also there is some evidence of relatively high lead levels in sediments of in Wheatbelt salt lakes (Lyons *et al.* 1990). While concentrations of some metals may be high in these systems, the solubility and therefore the impact of most metals on biota is affected by pH and redox potential (Boulton and Brock 1999).

4.5.4 Nutrients

Concentrations of nutrients such as total nitrogen, ranged from below detection to 4 300 mg/kg in the salt lakes sediments of this study. The availability of nitrogen is governed by a number of processes such as rewetting of sediments which results in a pulse of nutrients (McComb and Qui 1998) and nitrogen fixing bacteria (Ponnamperuma 1972). There were no relationships between natural lakes and discharge lakes, indicating little influence of dewatering discharge to total nitrogen in sediments. In terms of differences in Bioregions, the Avon Wheatbelt generally recorded the highest total nitrogen values in sediment and this is most likely related to the farming activities and use of fertilizers prevalent in the Bioregion in comparison to the other regions (Boggs *et al.* 2007).

Total phosphorus in sediment ranged from below detection to 970 mg/kg, with concentrations being recorded lower than total nitrogen in all sediments.

Concentrations of phosphorus in sediments are influenced by the concentration of organic matter and interactions with iron hydroxides (Barbanti *et al.* 1995). There appeared to be no influence by dewatering discharge to concentrations of total phosphorus. Sediments of the Yalgoo Bioregion reported the highest concentrations of total phosphorus within sediments, however data for this region was only reported from one lake and this most likely influenced the results within this Bioregion.

4.5.5 Total Organic Carbon

The concentration of organic matter can control a number of processes within the sediments such as concentrations of phosphorus, changes in pH and availability of some metals (Golterman 2004). Also, total organic carbon (TOC) can be used to indicate primary production (Wang and Williams 2001). The lakes recording the highest TOC values were located in close proximity to each other and were in the Johnston lakes system. The high level of organic matter present in these lakes may be related to the much denser riparian vegetation systems of these lake and the presence of thick microbial mats in some of these lakes (Chaplin 1998).

4.6 Conclusions

There was a high degree of variation in sediment chemistry both within and between salt lakes included in this study. This variation was generally related to geological features of the lakes and in some cases, the additional influence of dewatering discharge. The impact of dewatering discharge was linked to local geology. Generally impacts were different between Bioregions with changes in sediment pH, salt load and metals noted at lakes receiving dewatering discharge. Within the Murchison Bioregion, sediments had characteristically high concentrations of zinc, arsenic and copper. In contrast sediments of the Coolgardie Bioregion were typified by high concentrations of lead, cadmium, chromium and nickel. In terms of comparison with ANZECC guidelines, generally the Interim Sediment Quality Guidelines for protection of biota were applicable to the salt lakes. The only exceptions were the trigger values of nickel and chromium which were exceeded by concentrations in the natural lakes. Lakes Maitland, Black Flag and Miranda were relatively unique in terms of the concentrations of metals in sediment, compared to the other lakes in this study, most likely related to a unique local geology.

5 ALGAE

5.1 Abstract

Biota present in the inland salt lakes of Western Australia is controlled by the changing water regime as the lake dries. Diatoms are the only biota present during all stages of the hydrocycle and are the focus of this chapter. The objectives of this chapter were to describe the diatom community structure of the lakes, with particular emphasis on the relationship to the physico-chemical parameters and dewatering discharge. Community structure of diatom species generally differed between Bioregions, however there was a considerable degree of overlap in species. Dominant species included common and salt tolerant taxa such as; *Amphora coffeaeformis*, *Hantzschia* aff. *baltica* and *Navicula* aff. *incertata*. Being indicators of environmental change the relationship between sediment chemistry and diatom community structure was explored due to the lack of surface water. Concentrations of zinc, moisture content and cobalt in sediments displayed the greatest influence on the diatom data set. In terms of the impact of dewatering discharge on diatom community structure, sites receiving dewatering discharge reported significantly lower species richness. Despite this, there appears to be a capacity for impacts to be ameliorated by flushing events. This was highlighted by historical dewatering discharge sites showing little impact compared to current dewatering discharge sites.

5.2 Introduction

The previous two chapters established the physico-chemical condition of the salt lakes in Western Australia in terms of sediment and water chemistry. These chapters described a considerable variability both within and between salt lakes in this study in terms of environmental variables. This chapter introduces the primary producers of the lakes, with particular emphasis on the diatoms. The primary producers of inland salt lakes commonly includes bacteria, cyanobacteria and algae (Borowitzka 1981; Lewis 2007). For the most part, the primary producers in salt lakes are usually limited to benthic mats, which are composed of cyanobacteria and diatoms (Williams 1998b; John 2003b). However, cyanobacteria are not always present and diatoms are often the only biota present, particularly as the lakes enter the dry phase of the hydrocycle (Saros and Fritz 2000). Biota such as algae present in these systems must have the ability to cope with extreme conditions such as a lack of water, high temperature, high salinity and high levels of light (Flechtner 2007). These conditions tend to limit photosynthesis and growth of algae (Karsten *et al.* 2007).

The use of diatoms for monitoring change in aquatic systems has occurred extensively throughout Australia (Gell *et al.* 2002; Philibert *et al.* 2006; Tibby *et al.* 2007). Diatoms are highly sensitive to changes in environmental variables and have been used in a number of studies throughout Western Australian in baseline studies and investigations on inland salt lakes (Brearley *et al.* 1999; URS 2003). Previous studies have investigated diatom distribution in relation to secondary salinity within the Wheatbelt (Blinn *et al.* 2004; Taukulis and John 2006). They have also been used to determine the impacts of catchment degradation within the northern Wheatbelt region (Boggs *et al.* 2007). In addition, in the Goldfields of Western Australia, diatoms have been used to indicate the impact of dewatering discharge on salt lakes located in close vicinities of mining operations (Batley *et al.* 2003). While numerous studies have been carried out, the community structure of the inland salt lakes of the Goldfields as a collective has not been analysed. Also, this study will identify changes in community structure within the lakes and the tolerances of species to certain environmental variables within the lakes.

The impacts of dewatering discharge to this community will also be investigated. Frequently, these are the only biota present within the lakes and given their sensitivity to changes in environmental conditions were considered likely to be useful indicators of impact from these activities. As the lakes were mostly dry, diatom community structure was examined in response to changing sediment chemistry within the lakes. During the dry phase of the hydrocycle the lake sediments provide a medium for periphytic and benthic diatoms (Krejci and Lowe 1986). Therefore the composition of sediments can influence the community structure and result in considerable spatial variability within diatoms in the benthos of the saline lakes (Chessman 1986; Herbst 1988).

Specifically, the objectives of this chapter were to;

- describe the diatom communities from salt lakes in the Murchison, Yalgoo, Coolgardie and Avon Wheatbelt Bioregions of Western Australia

- determine the likely impacts of dewatering discharge on the diatom communities
- explore the environmental preferences of diatom species in relation to sediment chemistry

5.3 Materials and Methods

5.3.1 Diatom identification and analysis

Sediment cores were taken for digestion of diatoms and observation of microbial mats (John 2000b). Plastic vials (70 mL in volume) were pushed into the sediment to a depth of approximately 2 cm. The cores were then frozen and transferred to laboratory for analysis. On return to the laboratory, the top 5 mm of the sample was removed from the core surface. This was boiled in 50% nitric acid for four to six hours to clean the silicon frustules and remove the organic matter from the sample (John 2000b). These samples were then washed five times with distilled water and centrifuged.

Permanent slides were made with the resultant sample. Coverslips were placed on a hotplate with between 50 and 100 µL of sample and enough distilled water to equate to a total of 1000 µL. The amount of sample added was dependant on the density of the sample. The coverslip was allowed to dry, and then inverted onto a slide which contained Napthrax. These slides were then heated for approximately 30 minutes until all the air bubbles had been excluded and the Napthrax has set.

The slides were then examined under a compound microscope at (1000 x) using oil immersion. Transects of the slide were observed with any diatoms recorded and identified. The number of diatoms counted varied between 100 and 300 frustules depending on the density of diatoms within the slide. The percentage composition of each species was then calculated for each species within each sample.

The majority of diatoms in this study were identified by a specialist taxonomist.

5.3.2 Statistical Analysis

Minitab (version 14) was used to run an Analysis Of Variance (ANOVA) (Minitab Incorporated 2003) on the data to determine whether there were any significant differences between species richness in sites receiving dewatering discharge and the natural lakes. Data was tested to determine if it was normally distributed and the difference between the sites was considered significant at $p<0.05$.

Ordination of the data was performed using PRIMER (version 6) (Clarke 1993). Multi-dimensional scaling (MDS) was analysed on the full data set to determine if there were any differences between Bioregion and to observe the differences in community structure of diatoms between lakes. MDS was also completed on each Bioregion to analysis the differences between lakes within each region. On all occasions data was transformed using a square root transformation. Resemblance was calculated using the Bray Curtis similarity method and the MDS was considered a useful indication of community structure when the 2-dimensional stress of the plot was <0.2 (Clarke 1993; Clarke *et al.* 2006) .

ANOSIM (Analysis Of Similarities) was performed to determine if there were any significant differences in community structure between the Bioregions. Data was averaged over time to produce one result per sampling site. One-way design was chosen, with Bioregion used as the factor. For ANOSIM, R statistic values close to one indicated that the groups were different, while those with values centered around 0 were considered to be similar (Platell *et al.* 1998; Clarke and Gorley 2006). To determine which species contributed to the average similarity within each Bioregion, SIMPER (Similarity Percentages) analysis was used.

CANOCO (version 4.5) was used to complete detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA) (ter Braak and Smilauer 2002). Prior to the analysis in CANOCO, rare data (i.e. <1% abundance) was identified and deleted using PC-ORD (Version 4). DCA analysis was completed to determine if the data set had a unimodal response. CCA was used to examine the effects of environmental parameters on diatom community structure. Monte carlo permutation tests were used to determine the significance of the CCA.

5.4 Results

A total of 56 diatom species were recorded within the salt lakes included this study (Appendix F – species list, including authorities). The most commonly occurring species in order were *Navicula* aff. *incertata, Amphora coffeaeformis* and *Hantzschia* aff. *baltica* (Table 13). The number of diatom species (i.e. species richness) recorded from each site ranged from 0 to 14 species.

Table 13. The ten most commonly occurring diatom species within the salt lakes of this study.

Species	Number of Occurrences
Navicula aff. *incertata*	152
Amphora coffeaeformis	143
Hantzschia aff. *baltica*	135
Luticola mutica	57
Navicella pusilla	52
Hantzschia amphioxys	50
Navicula aff. *salinicola*	47
Nitzschia punctata	28
Navicula elegans	22
Pinnularia borealis	21

5.4.1 Comparison of diatom community structure between Bioregions

The MDS plot of diatom community structure shows some slight differences between the three Bioregions (Figure 12). Community structure of the Murchison Bioregion showed some affinities to both the Coolgardie and the Yalgoo Bioregions. In contrast the Coolgardie and Yalgoo Bioregions were fairly dissimilar as depicted by the distance of sites on the plot.

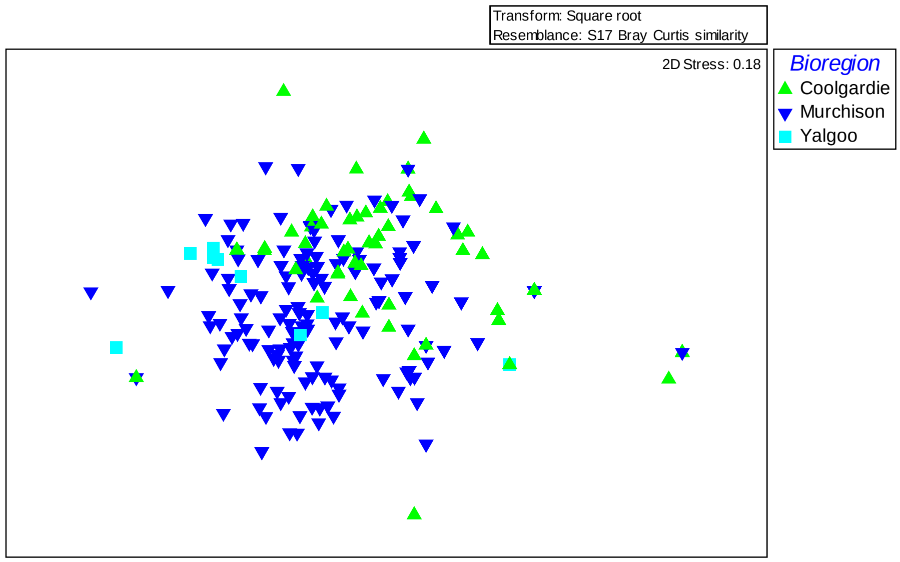

Figure 12. 2-dimensional MDS plot of percentage abundance of diatoms within each Bioregion.

The ANOSIM analysis showed that community structure was significantly different between Bioregions, however there was a considerable overlap in species as indicated by the low global R statistic (R=0.233 and p=0.001). In particular there was a high degree of overlap in community structure in the Coolgardie and Murchison (R=0.145), and the Coolgardie and Yalgoo bioregions (R=0.286) (Table 14). In contrast, community structure between the Murchison and Yalgoo regions was significantly different with less species overlap as indicated by the higher R statistic value (0.476).

Table 14. Results of ANOSIM analysis between Bioregions based on percentage abundance diatom data.

Bioregion	R statistic	Significance level
Coolgardie, Murchison	0.145	0.002
Coolgardie, Yalgoo	0.286	0.006
Murchison Yalgoo	0.476	0.001

Comparison between the Coolgardie and Murchison Bioregions showed that the first two dominant species in these regions, *Navicula* aff *incertata* and *Amphora coffeaeformis* were the same (Table 15). However, while *Navicula* aff *incertata* was the most dominant within these Bioregions it was less dominant within the Yalgoo Bioregion which was dominated by *Amphora coffeaeformis* and *Synedra* cf. *acus.*

Table 15. SIMPER analysis of percentage abundance of diatoms within each Bioregion. Only the species which made up 60% of the cumulative community structure were included.

Species	Mean Abundance		Consistency	Percent	Cum.
	Region 1	Region 2	Ratio		%
	Coolgardie	Murchison	Mean Dissimilarity = 70.6		
Navicula aff. *incertata*	4.5	4.6	1.2	17.4	17.4
Amphora coffeaeformis	3.0	3.5	1.1	15.3	32.7
Hantzschia aff. *baltica*	1.2	3.4	1.1	13.5	46.2
Luticola mutica	1.5	0.6	0.6	7.0	53.2
Navicula aff. *salinicola*	0.5	1.2	0.6	5.9	59.1
	Coolgardie	Yalgoo	Mean Dissimilarity = 84.8		
Navicula aff. *incertata*	4.5	1.1	1.2	14.8	14.8
Amphora coffeaeformis	3.0	4.1	1.2	14.0	28.8
Luticola mutica	1.5	0.8	0.6	7.7	36.5
Synedra cf. *acus*	0.0	1.9	0.6	6.5	42.9
Navicella pusilla	0.5	1.4	0.8	5.7	48.6
Hantzschia aff. *baltica*	1.2	0.9	0.8	5.6	54.2
Amphora sp. 2	0.0	1.5	0.7	5.0	59.2
	Murchison	Yalgoo	Mean Dissimilarity = 82.1		
Navicula aff. *incertata*	4.6	1.1	1.2	15.1	15.1
Amphora coffeaeformis	3.5	4.1	1.2	13.4	28.6
Hantzschia aff. *baltica*	3.4	0.9	1.1	11.2	39.8
Synedra cf. *acus*	0.0	1.9	0.6	6.3	46.1
Navicella pusilla	0.7	1.4	0.9	5.3	51.4
Luticola mutica	0.6	0.8	0.4	5.1	56.5

5.4.2 Comparison of diatom community structure between lakes

MDS plots were used to analyse the differences between lakes in each of the Bioregions (Figure 13 and Figure 14). It should be noted that sites recording no diatoms were excluded from this analysis. The MDS plot of diatoms within the Coolgardie Bioregion shows that most sites reported similar species as depicted by the grouping on the plot (Figure 13). Most lakes had 40% similarity in terms of species. A number of sites in Banker Lake, Lake Lefroy, White Flag, Lake Cowan and Greta Lake were located proximal to the main grouping of sites indicating some differences in community structure.

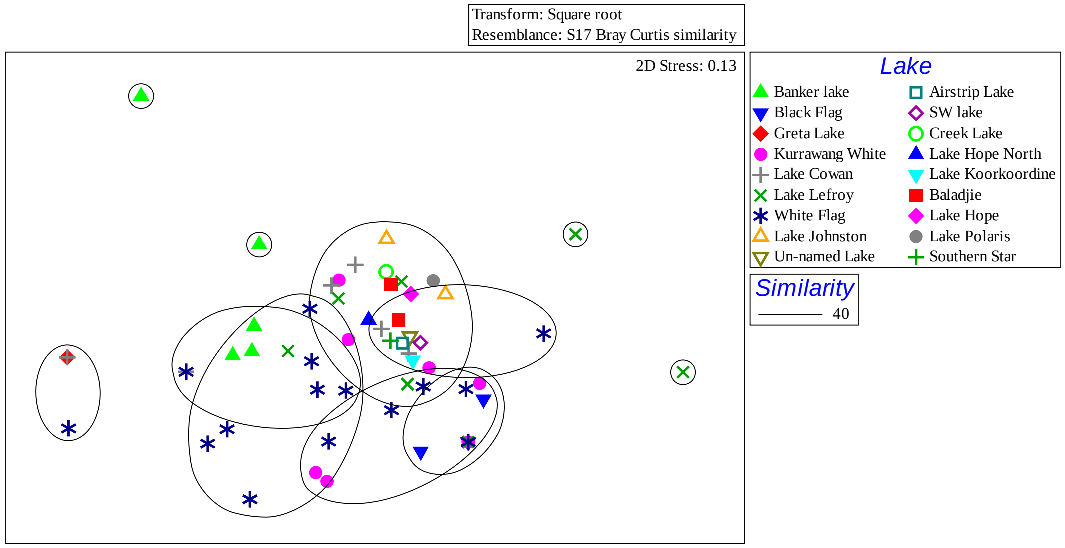

Figure 13. 2 dimensional MDS plot of diatoms within the Coolgardie region based on percentage abundance data.

Diatom community structure within the Murchison lakes also appeared to be similar, with most lakes reporting at least 40% of species in common to all lakes (Figure 14). With the exception of sites in Lake Carey, most sites formed groups, indicating some differences in community structure between lakes. Lake Carey, showed a high degree of variation in community structure.

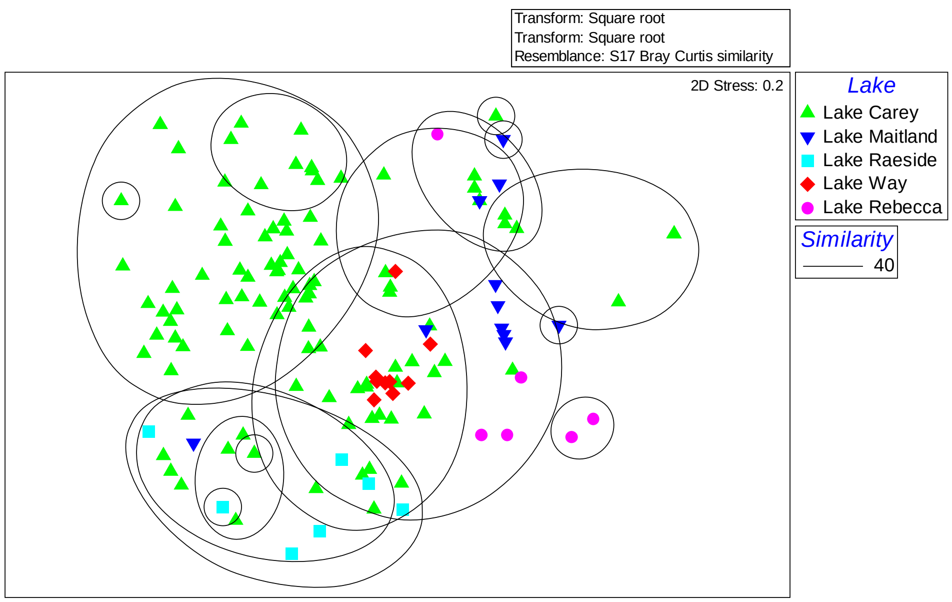

Figure 14. 2-dimensional MDS plot of diatoms within the Murchison Bioregion based on percentage abundance data. Sites recording no species were removed.

5.4.3 Impact of dewatering discharge on diatoms

Diatom species richness was significantly lower at sites receiving dewatering discharge in comparison to natural sites or those that had been subjected historically to discharge ($p<0.05$) for the lakes as a group (Table 16). However there were no significant trends between Bioregion and site classification (i.e. Natural, Discharge and Historical Discharge). The general trend was for the lakes receiving dewatering discharge to have lower species richness in comparison to natural lakes and lakes historically receiving dewatering discharge (Figure 15).

Table 16. Results of ANOVA for species richness between sites impacted by discharge (Discharge), sites not impacted by discharge (Natural) and sites historically impacted by dewatering discharge (Historical Discharge).

Code	Mean	SD	n
Discharge	3.24	2.51	131
Natural	3.96	2.22	184
Historical Discharge	4.63	2.56	16
P value		0.015	

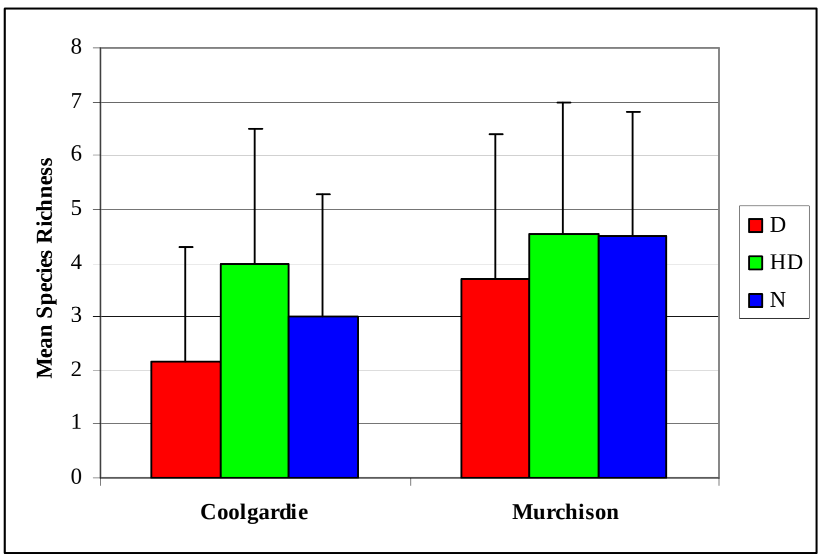

Figure 15. Mean species richness with sites receiving dewatering discharge (D), historically impacted dewatering sites (HD) and natural sites (N) within the Coolgardie and Murchison Bioregions.

5.4.4 Diatom community structure versus sediment chemistry

CCA axes 1 (0.44) and 2 (0.37) explained 18.9% of variation in diatom community structure. A low value such as this is common in large data sets with lots of zero values as was the case in this instance (Gasse *et al*. 1995). Even so the Monte Carlo permutation test indicated that the axes 1 and 2 were significant (p=0.03 in both cases, 999 permutations). The first CCA axis was correlated with nickel (r=0.45), in contrast the second axis was negatively correlated with zinc (r=-0.58), moisture content (r=-0.53) and cobalt (r=-0.52).

Species such as *Chaetoceros muelleri* (*chamue*), *Entomoneis paludosa (entpal)* were associated with higher concentrations of total organic carbon (Figure 16). In contrast *Hantzschia virgata (hanvir), Caloneis* aff *bacillum (calaba), Pinnularia borealis (pinbor) and Luticola mutica (lutmut)* were common in sediments containing high concentrations of nickel, chromium and lead. Species such as *Nitzschia aff rostellata*, *Proschkinia aff complanata* were associated with higher concentrations of most salts, copper, moisture content and cobalt.

The most commonly recorded species in this study *Amphora coffeaeformis* and *Hantzschia* aff. *baltica* were located in a similar location to each other on the CCA plot (Figure 16). They were situated in the mid section of the plot, and were commonly associated with moderate concentrations of all of the parameters tested. *Navicula* aff. *incertata* was common in sediments with higher concentrations of barium and total organic carbon and lower concentrations of most other parameters.

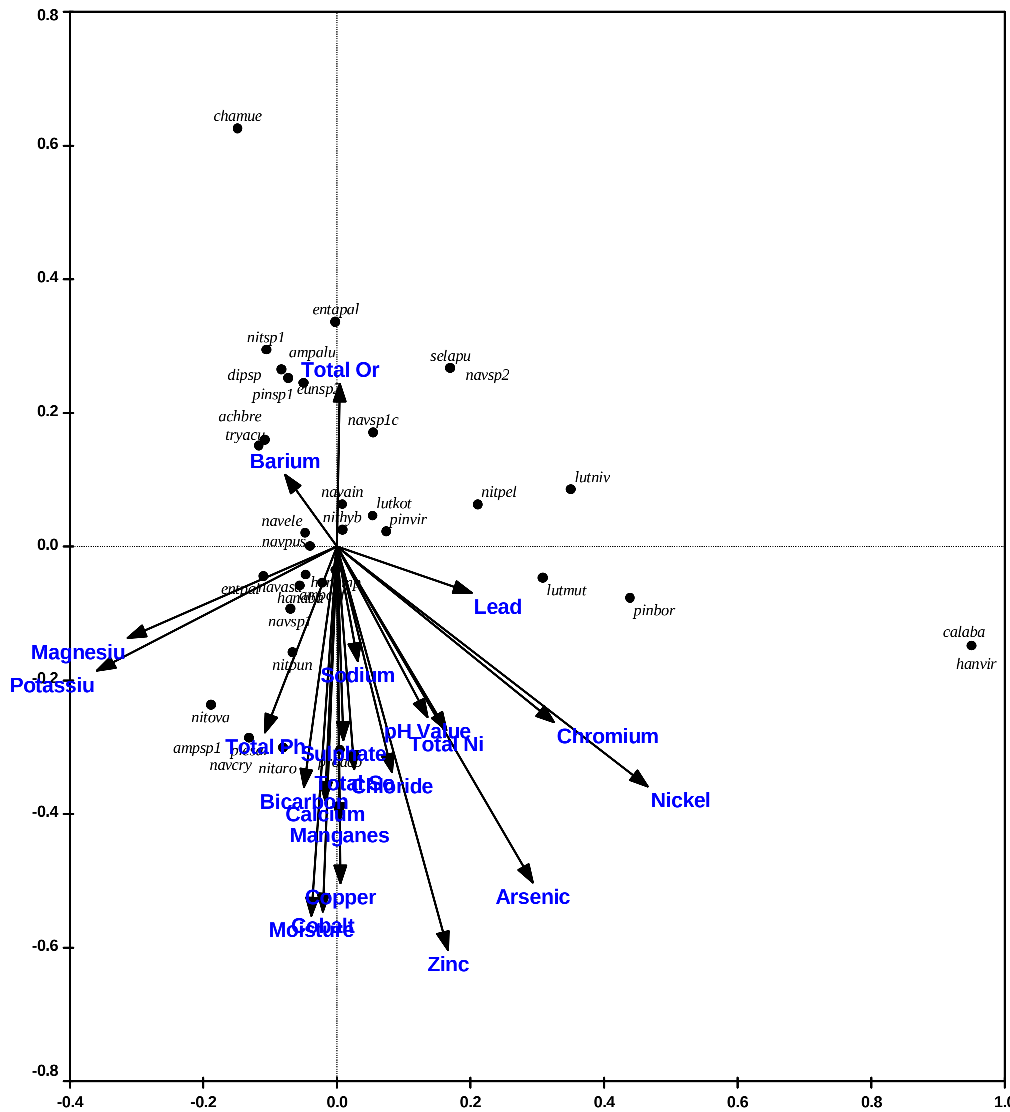

Figure 16. CCA of diatom species versus sediment parameters, based on 121 samples. Only species with a relative abundance of 1% were included. Species codes are presented in Appendix G.

5.5 Discussion

Inland salt lakes are considered to be extreme environments for the biota which exists in them (Gilmour 1990). They must cope with a range of conditions including high salinity and heat, temporary water regime and evapo-concentration of parameters as the lakes dry (Gilmour 1990). A total of 56 diatom species from 24 salt lakes has been recorded in this study with species richness of sites ranging between zero and 14 species. This figures are consistent with other studies of benthic algae in salt lakes of Western Australia (Brearley *et al.* 1999; URS. 2001). Common salt tolerant species were dominant throughout the study and included; *Amphora coffeaeformis*, *Hantzschia* aff. *baltica* and *Navicula* aff. *incertata* (Gell and Gasse 1990; Blinn *et al.* 2004; Taukulis and John 2006).

5.5.1 Community structure between Bioregions

While diatom community structure was significantly different between Bioregions, there was a considerable degree of overlap, with Bioregions reporting similar dominant species. Given the degree in variation reported in the chemistry of the lakes this indicates that the diatoms present in these extreme environments can tolerate a varying range conditions (Philibert *et al.* 2006; John 2007). While the Murchison and Coolgardie Bioregions reported similar diatom community structure, the diatom community structure between the Murchison and Yalgoo Bioregions was fairly dissimilar. This may be due in part to the small data set from the Yalgoo Bioregion (i.e. one lake). In addition, the lake in this region receives a fresher dewatering discharge unlike most of the other lakes which receive hypersaline dewatering discharges. This has most likely resulted in the differing community structure, with species preferring lower salinities becoming more dominant as diatom species can be highly sensitive to changes in salinity (Kashima 2003; Philibert *et al.* 2006; Taukulis and John 2006).

Three species (*Navicula* aff. *incertata*, *Amphora coffeaeformis* and *Hantzschia* aff. *baltica*) were common in the salt lakes of the Coolgardie and Murchison Bioregions, however only *Amphora coffeaeformis* was dominant within the samples from the Yalgoo bioregion. In addition to this species, *Navicella pusilla* and *Cocconeis* *placentula* were apparent in that region. These two species prefer lower salinity (Blinn 1993) hence their increased dominance at the Yalgoo sites.

5.5.2 Community structure between Lakes

Within the Coolgardie region diatom community structure was fairly similar between lakes, the exceptions being Banker Lake, Lake Lefroy, White Flag Lake, Lake Cowan and Greta Lake. This was mainly associated with low species richness within the sites of these lakes. With the exception of Greta Lake, these lakes receive dewatering discharge, with salt crusts in sections of the lakes indicating high salinities. It is likely that the extreme salinity and unfavorable conditions at these sites has resulted in the low species richness (Blinn 1993; Gasse *et al.* 1995).

The diatom community structure of each lake within the Murchison Bioregion was fairly unique with the exception of Lake Carey. This lake showed a community structure at particular sites, similar to that of the other lakes. The occurrence of the three dominant species, *Navicula* aff. *incertata*, *Amphora coffeaeformis* and *Hantzschia* aff. *baltica* within the lake may explain the similarities between Lake Carey and the other lakes. A combination of factors such as a greater sampling effort, the large number of habitat types and the variation in environmental parameters within this lake may have resulted in the occurrence of a wide range of species reported (John *et al.* 2002; Saros and Fritz 2002; Townsend and Gell 2005).

5.5.3 Impact of dewatering discharge on diatoms

Sites that were directly influenced by dewatering discharge reported significantly lower species richness than sites historically receiving dewatering discharge and natural lakes. This was most likely related to the extreme salinity, increased water flow, thick salt crust and high concentrations of metals at these sites. In addition the trend of lower diatom abundances is common and has been reported in areas impacted by heavy metals contamination (Pérès *et al.* 1997). The impact of the dewatering discharge appeared to be localized, and restricted to the footprint of the discharge. This was particularly the case in the larger lakes where the dewatering discharge footprint was small compared to the large, unimpacted areas (John *et al.* 2002). In contrast, the smaller lakes which were wholly impacted by dewatering discharge tended to show poor species diversity throughout the lake, a trend reported in other studies (Timms 2005a). Those sites which had received historical dewatering discharge that had ceased at least one year prior to assessment showed a significantly higher species richness than current discharge sites, indicating that the impacts of dewatering discharge can be ameliorated by considerable rainfall events (URS 2003; Gregory *et al.* 2006).

Once surface water salinity exceeded 50 g/L in the salt lakes it would be expected that there will be little variation in community structure due to the narrow range of species which can inhabit the lakes in these conditions (Hammer *et al.* 1983; Williams 1998b). This was demonstrated by the dominance of only three species across all Bioregions, even thought the sediment chemistry of these sites was markedly different. However, the absence of species from the dewatering discharge sites, and the low species richness at these sites indicates that the dewatering discharge is having some impact on the diatom community.

5.5.4 Diatom community structure versus sediment chemistry

Owing to the absence of surface water, diatom community structure was correlated with sediment chemistry. The effect of sediment chemistry on diatom community structure has been scarcely studied, although the limited studies have found some correlation (van Kerckvoorde *et al.* 2000; Zalat and Vildary 2005). Of the 22 parameters investigated in this study, the three parameters with an apparent negative influence on diatom community structure were zinc, nickel and moisture content. In addition, a number of species such as, *Caloneis* aff. *bacillum, Proschkinia aff complanata* and *Hantzschia virgata* showed an affinity for concentrations of metals such as nickel, chromium, zinc, lead and cobalt. While the effects of metals on diatom community structure is poorly understood, some studies indicate tolerance of certain species to some metals (Morin *et al.* 2007b), however there was no information available on the effects of these metals in sediment on these particular species (Hirst *et al.* 2002). Some studies of periphyton have shown some deformity in structure and reduced size related to high concentrations of certain metals in water and this may be used as an indicator of metal toxicity (Morin and Coste 2006; Morin *et al.* 2007a). Although beyond the scope of this study, this is something that should be considered in future sampling of these systems.

While the influence of major anions and cations on diatom community structure has been reported in a number of other studies this was not evident in this study (Blinn 1993; Gasse *et al.* 1995; Gell 1997; Laing and Smol 2000). This may be due to the high concentrations of salts within these lakes, and species living here are adapted to these conditions, in this regard, other parameters such as metals and moisture content were more likely to influence diatom community structure as was shown in these results.

Moisture is considered to be a controlling factor to the growth of algae in temporary lakes (Flechtner 2007). However, in this study moisture content was shown to have a negative impact of the diatom community. It is likely that this was associated with the presence of dewatering discharge, in that sites receiving dewatering discharge recorded higher moisture content than natural sites. The effects of moisture content on diatoms has been considered in a number of studies with relationships between moisture content and certain species such as *Hantzschia amphioxys* and *Pinnularia borealis* being reported (Van de Vijver and Beyens 1997; van Kerckvoorde *et al.* 2000). Moisture content has been correlated with valve size in these particular species, with low moisture content resulting in reduced valve size (Van de Vijver and Beyens 1997). It is likely that the diatoms at the discharge site would be impacted in a similar way to sites with low moisture content.

5.6 Conclusions

Community structure of diatom species differed between Bioregions, however there was a considerable degree of overlap. Species richness overall was low, with three species being dominant: *Amphora coffeaeformis*, *Hantzschia* aff. *baltica* and *Navicula* aff. *incertata*. Sediment chemistry explained some variation in community structure with zinc, moisture content and cobalt displaying the greatest and negative influence on the diatom data set. Sites receiving dewatering discharge reported significantly lower species richness, although there appears to be capacity for impacts to be ameliorated by flushing events. This was highlighted by historical dewatering discharge sites showing little impact compared to current dewatering discharge sites.

6 AQUATIC INVERTEBRATES

6.1 Abstract

The invertebrate fauna of the Western Australian salt lakes has been poorly documented to date, most likely related to the limited opportunities to collect live invertebrates during the filing events. The relationship between invertebrates and water quality in relation to Bioregion and dewatering discharge within the salt lakes of Western Australia was considered in this chapter. The Crustacea dominated, with the most taxa recorded from the *Parartemia* genus. There were some differences in community structure between Bioregions, indicating that invertebrates species presence was somewhat related to geography and therefore speciation. Community structure was found to be influenced by water quality parameters, with phosphorus and bicarbonate contributing mostly to variation. Species richness was minimal at dewatering discharge lakes, which reported significantly lower species richness in comparison to the natural lakes.

6.2 Introduction

The diatom community structure of salt lakes in Western Australia was discussed in the previous chapter with an emphasis on the influence of sediment chemistry and dewatering discharge. The current chapter explores the relationship between invertebrates and water quality in Bioregions and the influence of dewatering discharge within the salt lakes of Western Australia. While not as diverse in taxa as their freshwater counterparts, typically salt lakes such as those situated in the Goldfields of Western Australia can be rich in invertebrate fauna (Timms 2007). In Australia, the invertebrate communities of salt lakes within some regions have been extensively studied, providing an insight into the potential diversity and speciation of taxa within these salt lakes (Timms 2007). In contrast, the invertebrate fauna of the Western Australian salt lakes to date has been poorly documented, most likely related to the limited opportunities to collect live invertebrates during the filing events (Roshier and Rumbachs 2004; Timms 2005a). Also, when the lakes do fill, access is often difficult and the studies are usually limited to one-off opportunistic sampling strategies, with no consideration of the various changes in community structure over the hydrocycle (De Deckker and Geddes 1980; Timms *et al.* 2006).

Studies to date indicate that invertebrate communities of the Western Australian salt lakes are dominated by the Crustacea, typically consisting of taxa from the following orders: Anostraca, Ostracoda, Cladocera and Copepoda (Geddes *et al.* 1981; Brock and Shiel 1983; De Deckker 1983). In addition, representatives from the Insecta, Rotifera and Gastropoda are also common within the lakes (Geddes *et al.* 1981; Brock and Shiel 1983; John 2003b).

It is widely accepted that species richness of invertebrates is inversely proportional to surface water salinity. However at the high salinities (over 150 g/L), typically reported within the Western Australian salt lakes, it is expected that species richness will be low, given the limited numbers and types of organisms which can survive these conditions (Williams 1981b; Williams *et al.* 1990; Timms 2007). For example, in lakes where salinity exceeds 150 g/L it is likely only species of *Parartemia* and Ostracoda will be present (Timms 2007). While salinity can influence certain species, the survival of some invertebrates can also be related to pH, ionic composition, light, temperature, dissolved oxygen and nutrients (Bayly and Williams 1966; De Deckker 1983; Timms *et al.* 2006). In addition to water quality parameters, geographic location appears to have a considerable influence on invertebrate communities and speciation within the salt lakes (Geddes *et al.* 1981).

Dispersal mechanisms within the salt lake fauna are generally poor, accounting for the large amount of speciation between regions of Australia (Finston 2002; Timms 2007). In addition, a number of survival mechanisms are present in salt lake fauna to ensure the species remain within the lakes even as they dry (Williams 1981a; Williams 1998b). One such method is observed in *Coxiella* which are able to seal their shells to protect them from the surrounding environment. They can survive for a number of months in this state (Williams 1985). Other species such as *Parartemia*, ostracod and *Daphniopsis* produce desiccation-resistant eggs or cysts. These cysts, particularly those of the *Parartemia* may resist desiccation and remain viable for a number of years (Thiéry 1997).

The impact of dewatering discharge on invertebrate communities has been considered in some internal consultant reports, however, there is little published information on the impact of dewatering discharge on the invertebrate fauna of these lakes (Timms 2005a). While a number of studies have indicated that the effects of dewatering discharge are minimal, some have found a decline in species richness (Chaplin *et al.* 1999; van Etten *et al.* 2000; John *et al.* 2002). The lack of a hydrocycle during the study period has hampered the determination of the impacts of dewatering discharge.

Given the lack of published information, this chapter serves as a comparison of invertebrate community structure between the salt lakes in this study. Specifically the objectives are to;

- describe the invertebrate communities present in the salt lakes of the Murchison, Yalgoo, Coolgardie and Avon Wheatbelt Bioregions in Western Australia
- explore the environmental preferences of invertebrate species in relation to water chemistry
- determine the potential impacts of dewatering discharge on the invertebrate communities
- determine lakes which have unique invertebrate communities

6.3 Materials and Methods

A considerable portion of the data for this study was collected from unpublished sources such as unpublished company reports, due to the lack of filling events and therefore opportunity to collect live invertebrate data from the Goldfields since 2000. Generally most of the data was in presence/absence format. The identification of specimens has occurred using a variety of sources, however the majority of identification occurred prior to specialist keys being available.

In all cases, methods were similar a known volume of water was isolated by a Perspex tube. All invertebrates were removed from the water column using a 50 μm zooplankton net and fixed in ethanol.

6.3.1 Statistical Analysis

Analysis Of variance (ANOVA) was conducted using Minitab (version 14) (Minitab Incorporated 2003). The primary aim of this analysis was to determine whether there were any significant differences between invertebrate species richness in sites receiving dewatering discharge and ‘natural sites’. Data was tested to determine if it was normally distributed and the difference between the sites was considered significant at $p<0.05$.

Ordination of the presence/absence data was performed using PRIMER (version 6) (Clarke 1993). Multi-dimensional scaling (MDS) was used to determine if there were any differences between Bioregions in terms of invertebrate community structure for presence/absence data. Resemblance was calculated using the Bray Curtis similarity method and the MDS was considered a useful indication of actual community structure when the stress of the plot was <0.2 (Clarke 1993; Clarke *et al.* 2006).

To ascertain if there were any significant differences in community structure between the Bioregions, ANOSIM (analysis of similarities) was performed. The mean of the data over time at each site was calculated for the analysis to produce only one replicate for each site. One-way design was chosen, with Bioregion chosen as factor. For ANOSIM, R statistic values close to one indicate that groups were different. Those with values centered around 0 were considered to be similar (Platell *et al.* 1998; Clarke and Gorley 2006). To determine which species contributed to the average similarity within each Bioregion SIMPER (similarity percentages) analysis was used.

CANOCO (version 4.5) was used to complete detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA) on the invertebrate presence/absence data and water quality data (ter Braak and Smilauer 2002). Prior to analysis using CANOCO, rare data (i.e. <1% abundance) was identified and deleted using PC-ORD (Version 4). DCA analysis was completed to determine if the data set had a unimodal response. CCA was then used to examine the effects of environmental parameters in surface water on community structure. Monte carlo permutation tests (999 permutations) were used to determine the significance of the CCA. Water data that was used to understand relationships with macro-invertebrates, was transformed and normalised, prior to analysis of CCA.

6.4 Results

6.4.1 Invertebrate taxa

A total of 101 taxa have been recorded within the salt lakes of this study (13 lakes in total) since 1998 (Appendix H). This frequency is comparable to species recorded in the Western Victorian regions, and is higher than that documented from the Paroo, Eyre and Coorong regions (Table 17). Within the Western Australian salt lakes, individuals were recorded from a total of six classes, representing the Crustacea, Arachnida, Rotifera, Insecta, Gastropoda and Nematoda.

Table 17. Number of invertebrate taxa recorded in inland regions of Australia.

Region	No. of species	No. of wetlands
Western Victoria[1, 2]	89	18
Paroo Region[3]	25	74
Eyre Peninsula[4]	84	40
Wheatbelt of WA[5]	957	230
Coorong[6]	38	23
Goldfields of WA	101	13

[1] (Timms 2007) [2] (Williams 1981b) [3].(Timms 1993) [4] (Timms 2005b) [5] (Pinder *et al*. 2005) [6.] (De Deckker and Geddes 1980)

The Crustacea were the most commonly occurring of the six classes. The most commonly reported species throughout the lakes was a Chironomidae species, which was recorded on 34 occasions (Table 15). Other commonly occurring taxa were the Ostracoda with genera such as *Diacypris* and *Reticypris* being recorded.

Table 18. The ten most commonly occurring invertebrate taxa recorded within this study.

Taxa	No. of Occurrences
Chironomidae	34
Reticypris sp.	33
Cyclopoida	30
Daphniopsis sp.	21
Diacypris whitei	20
Coxiella gilesi	16
Diacypris dictyote	14
Branchionus sp.	13
Diacypris sp.	12
Parartemia sp.	11

6.4.2 Comparison of community structure between Bioregions

The global R statistic for observing differences in community structure between Bioregion was 0.328 at a significance level of 0.001. This signifies that while the community structure of invertebrates was significantly different between Bioregions, there is some overlap in species. Invertebrate community structure between the Avon Wheatbelt and Yalgoo Bioregions was most dissimilar, recording a R statistic value of 0.749 (Table 19). In contrast the community structure of the Yalgoo and Murchison Bioregions was similar and not significantly different.

Table 19. Results of the ANOSIM analysis of presence/absence data for invertebrates between Bioregions, including the R statistic and significance level.

Bioregions	R Statistic	Significance Level
Avon Wheatbelt, Yalgoo	0.749	0.002
Avon Wheatbelt, Coolgardie	0.233	0.001
Avon Wheatbelt, Murchison	0.444	0.001
Yalgoo, Coolgardie	0.208	0.001
Yalgoo, Murchison	0.125	0.063
Coolgardie, Murchison	0.363	0.001

For the SIMPER analysis, the mean dissimilarity was lowest between the Yalgoo and Murchison Bioregions (85.3%), indicating that these were the most similar for the community structure of invertebrates (Table 20). While both Bioregions were dominated by Cyclopoida, the proportion of other species differed. In contrast the most dissimilar Bioregions were the Coolgardie and Avon Wheatbelt Bioregions (99.7%). *Parartemia nova* sp and the Ceratopogonidae species were most common within the Coolgardie Bioregion and were not represented in the Avon Wheatbelt Bioregion.

Table 20. SIMPER analysis of invertebrate taxa within each Bioregion based on presence/absence data. Only the top five species were included (where applicable).

Species	Frequency		Consistency Ratio	%	Cum. %
	Region 1	Region 2			
	Coolgardie	Avon	Mean Dissimilarity = 99.7		
Parartemia informis	0.0	1.0	2.2	26.1	26.1
Parartemia nova sp.	0.2	0.0	0.5	6.8	32.9
Ceratopogonidae sp. 1	0.2	0.0	0.5	6.8	39.6
Cyclopoida	0.03	0.2	0.5	4.7	44.3
	Coolgardie	Yalgoo	Mean Dissimilarity = 98.4		
Cyclopoida	0.03	0.8	1.2	11.6	11.6
Diacypris sp	0.0	0.6	1.2	8.8	20.3
Cyprinotus sp	0.0	0.5	0.9	7.4	27.7
Tanypodinae sp.	0.0	0.4	0.7	6.0	33.7
	Avon	Yalgoo	Mean Dissimilarity = 94.5		
Parartemia informis	1.0	0.0	2.1	24.5	24.5
Cyclopoida	0.2	0.8	1.0	15.9	40.4
Diacypris sp	0.0	0.6	1.2	12.3	52.7
Cyprinotus sp	0.0	0.5	1.0	10.4	63.1
	Coolgardie	Murchison	Mean Dissimilarity = 97.1		
Cyclopoida	0.03	0.5	0.9	6.8	6.8
Reticypris sp.	0.1	0.4	0.8	6.3	13.0
Daphniopsis sp.	0.0	0.5	0.7	4.0	17.1
Chironomidae sp.	0.1	0.5	0.8	3.8	20.9
	Avon	Murchison	Mean Dissimilarity = 93.0		
Parartemia informis	1.0	0.1	1.2	14.8	14.8
Cyclopoida	0.2	0.5	0.8	9.3	24.1
Reticypris sp.	0.0	0.4	0.8	8.9	33.0
Daphniopsis sp.	0.0	0.5	0.7	5.3	38.3
	Yalgoo	Murchison	Mean Dissimilarity = 85.3		
Diacypris sp	0.6	0.1	1.0	6.9	6.9
Reticypris sp.	0.0	0.4	0.8	6.7	13.6
Cyprinotus sp	0.5	0.0	0.9	6.0	19.5
Cyclopoida	0.8	0.5	0.7	5.9	25.5

6.4.3 Comparison of community structure between lakes

An MDS plot of invertebrate presence/absence data shows that there is some variation in the study lakes in terms of the species composition (Figure 17). For example, the invertebrate community structure of both Lake Lefroy and Lake Zot was fairly unique in comparison to the other lakes. The community structure of White Flag Lake was also fairly dissimilar to the other lakes, however this lake was only sampled on one occasion. Species richness within the lakes ranged from two species at Black Flag Lake and the Yarra Yarra Lakes to 25 species within Lake Lefroy (Table 21). There was a high degree in uniformity between sites in Lake Austin, as depicted by the tight cluster of sites on the MDS plot in comparison to the other lakes.

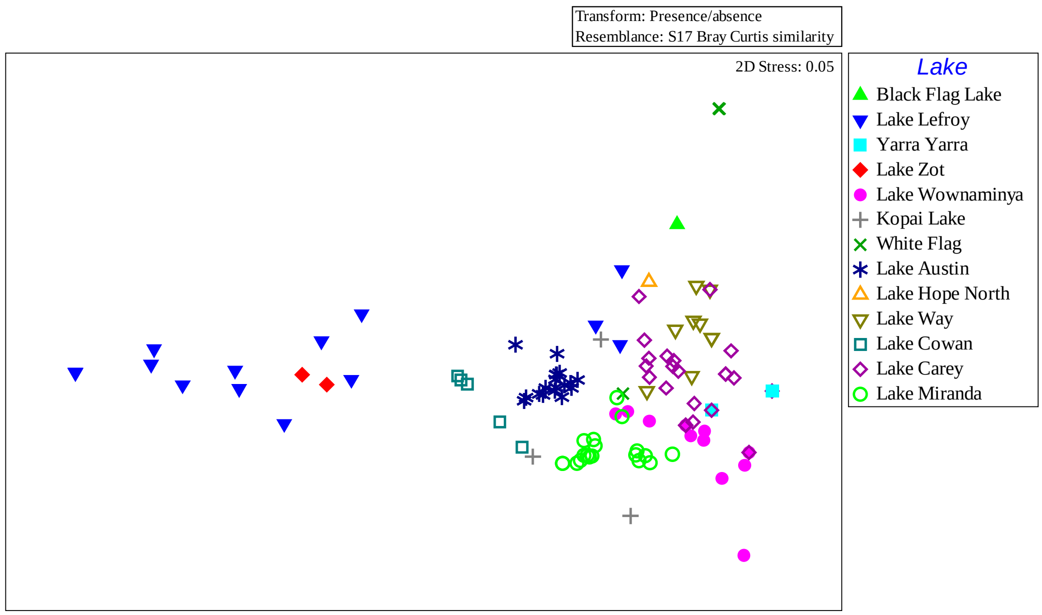

Figure 17. 2-dimensional MDS plot of invertebrate communities within lakes of this study. This plot is based on presence/absence data. Data was transformed and the resemblance calculated using the Bray Curtis method.

Table 21. Number of invertebrate taxa recorded in each of the lakes from 2000 to 2007.

Lake	Bioregion	No. Species Recorded
Black Flag Lake	Coolgardie	2
Kopai Lake	Coolgardie	10
Lake Austin	Murchison	24
Lake Carey	Murchison	12
Lake Cowan	Coolgardie	14
Lake Hope North	Coolgardie	3
Lake Lefroy	Coolgardie	25
Lake Miranda	Murchison	18
Lake Way	Murchison	7
Lake Wownaminya	Yalgoo	11
Lake Zot	Coolgardie	4
White Flag Lake	Coolgardie	3
Yarra Yarra Lakes	Avon Wheatbelt	2

6.4.4 Impact of dewatering discharge on invertebrates

Species richness at the dewatering discharge sites was significantly lower than that of the 'natural' sites overall (p=0.037) (Table 22). Also, differences between the species richness at discharge and natural sites were apparent in each of the Bioregions (Figure 18). Within the Avon Wheatbelt region mean species richness was slightly lower at the natural sites. In contrast mean species richness was lower at the discharge sites within the Coolgardie and Yalgoo Bioregions. Within the Murchison Bioregion mean species richness was similar between both site types.

Table 22 Results of ANOVA for species richness of invertebrates between sites impacted by discharge (Discharge), sites not impacted by discharge (Natural).

Code	Mean	SD	n
Discharge	2.72	1.93	57
Natural	3.59	2.24	46
P value		0.037	

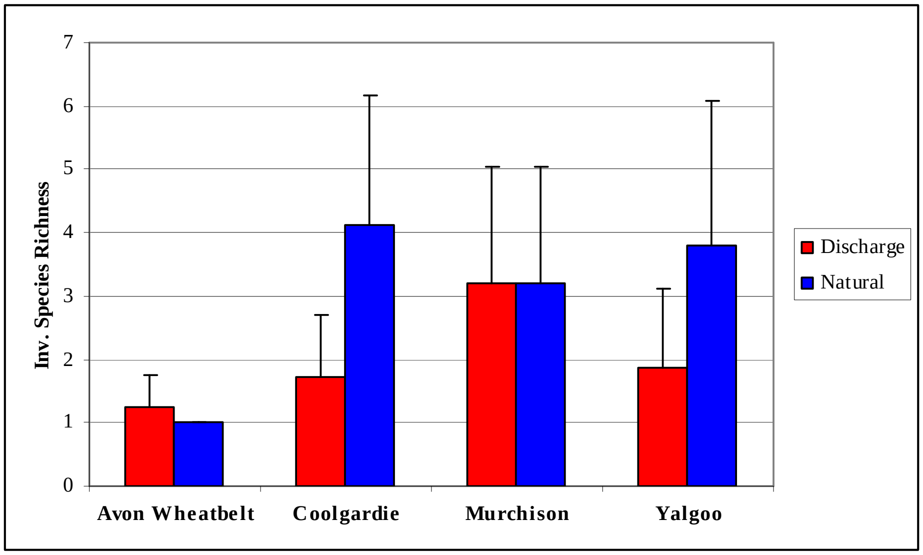

Figure 18. Mean species richness of invertebrates within sites receiving dewatering discharge and natural sites within the four Bioregions.

6.4.5 Invertebrate community structure versus water chemistry

The environmental variables; electrical conductivity, pH, total nitrogen, total phosphorus and major anions and cations were used for comparison with invertebrate community for the CCA analysis. CCA axes 1 (0.67) and 2 (0.61) explain a total of 44.6% of variation in invertebrate community structure. Also, the Monte Carlo permutation test indicated that the axes 1 and 2 were significant (p=0.012 and p=0.001 respectively in both cases, 999 permutations). The first CCA axis was correlated with total phosphorus (TP) (r=0.66), while the second axis was correlated with bicarbonate (HCO_3) (r=0.55). Total phosphorus and bicarbonate had the greatest influence on the invertebrate communities as depicted by the longer vectors on the CCA plot in comparison to the other parameters (Figure 19). Taxa such as *Parartemia informis* and Chironomidae species were associated with high concentrations of bicarbonate and subsequently higher pH in comparison to the other taxa. In contrast, *Parartemia* species 2 and *Daphniopsis* species were related to higher concentrations of potassium (K).

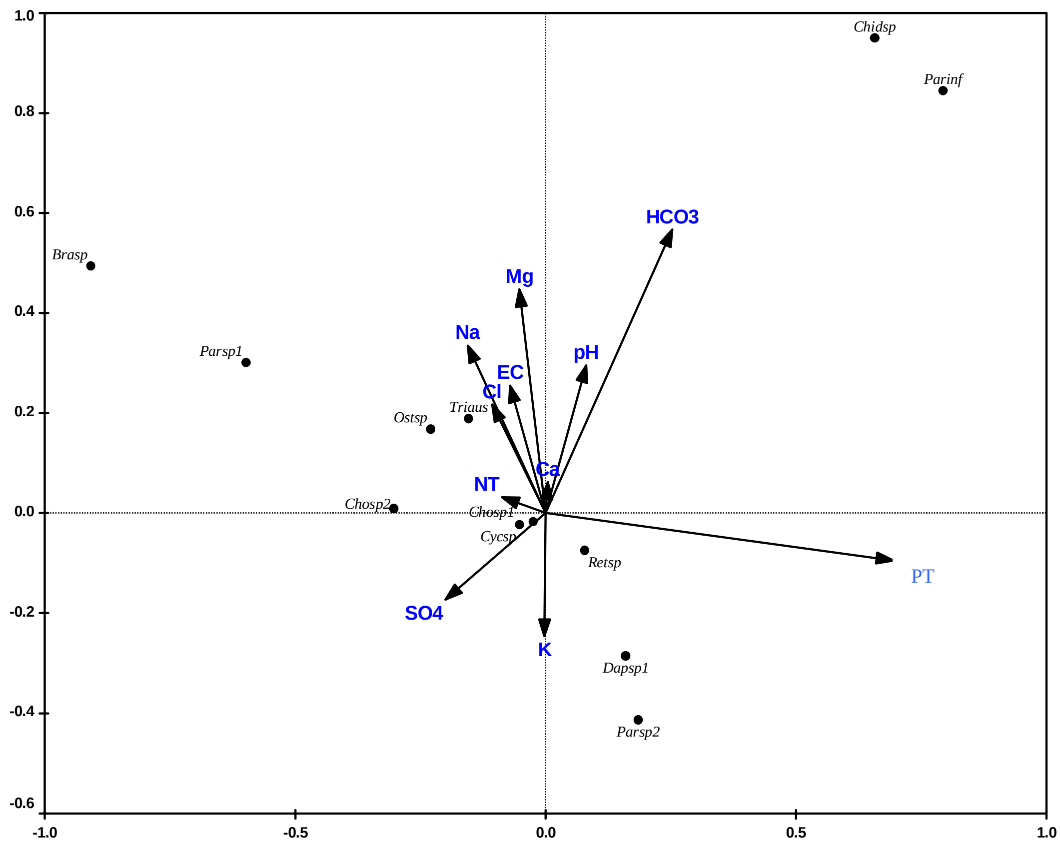

Figure 19. CCA of invertebrate species for water quality and invertebrates based on presence absence data. Only species with a relative abundance greater than 1% were included in the analysis. Species codes are presented in Appendix I.

6.5 Discussion

6.5.1 Invertebrate taxa

A total of 101 taxa have been reported in the 13 wetlands investigated in this chapter. The Crustacea were the most commonly represented group, a trend reported in other Australian salt lakes (De Deckker and Geddes 1980; De Deckker 1988; Halse *et al.* 1998). This is a similar frequency to the number of invertebrates recorded in the Western Victorian region and greater than recorded in other salt lakes studies in Australia, particularly from the Coorong (South Australia), Paroo (north western New South Wales), Eyre peninsula and the Tasmanian Midlands (Timms 2007). In this study, the Chironomid species, *Daphniopsis* species and *Diacypris whitei* were most abundant. The most commonly occurring genus within the lakes was the *Parartemia* with 10 species recorded. Most of the taxa recorded are commonly represented within salt lakes of Australia (Timms 1993; Williams 1998b; Timms 2007).

6.5.2 Comparison of community structure between Bioregions

The data from this study indicated that the occurrence of certain species in the lakes was related to geographic location, with significant differences reported between Bioregions. This may be explained by the high degree of speciation in Australian salt lakes, attributed to the large distances between the lakes (Williams and Kokkinn 1988; Williams 1998b; Timms 2002; Timms 2007). The speciation between these lakes is most likely related to the poor dispersal mechanisms of some species such as the *Parartemia* whose eggs tend to stay where deposited (Williams 1985).

In terms of dominant species in each Bioregion, *Parartemia informis* was the only species reported from the lakes in the Avon Wheatbelt Bioregion. This species is widespread throughout the Wheatbelt of Western Australia, preferring a salinity of less than 100 g/L (Timms 2004). In contrast lakes in the Coolgardie Bioregion were dominated by *Parartemia* species and a Ceratopogonidae species. Both species were recorded within Lake Lefroy and its associated wetlands (Brearley *et al*. 1999). Specimens from the group Ceratopogonidae have been reported historically in studies of the Western Australian salt lakes, and are fairly salt tolerant (Geddes *et al.* 1981; Pinder *et al.* 2005). Representatives from the Cyclopoida were dominant in both the Murchison and Yalgoo Bioregions. Several species of Cyclopoida have been reported from Western Australian salt lakes and include representatives from the genera *Microcyclops* and *Apocyclops* (Geddes *et al.* 1981; Halse *et al.* 1998). Two ostracods species, *Reticypris* and *Diacypris* were also common within these Bioregions. Both of these species are common and widespread throughout Western Australia, reporting wide salinity tolerances (Geddes *et al.* 1981).

In addition to geographical affinities of certain species, changes in invertebrate communities are highly dependant on the stage of the hydrocycle, with community structure changing greatly over this cycle (Williams 1998a; John 2003b). Most of the sampling events in this study are one off events which may have skewed the data sets as has been reported in other studies (De Deckker and Geddes 1980; Timms 1998). For example *Branchinella* are often present upon filling when the salinity is low, but were rarely observed in this study. As the lakes dry, and the salinity increases, *Parartemia* species tend to become more prevalent (Timms 1993; 1998). This is important as some species present in the lakes may have been missed due to the stage of the hydrocycle which was sampled.

Another limitation of the study is that some differences between lakes and Bioregions may be related to differences in opinion regarding the identification of taxa. For example while some sources may identify species as Ostracoda, others identified to species level i.e. *Reticypris*. Also, *Parartemia* species reported as a separate species, may actually be the same, due to limited description of this group when the species were identified. It is impossible in these cases to verify the data, and for the purposes of this study, the species have been separated. Therefore it is possible that Bioregions are actually more similar than reported in this study.

6.5.3 Comparison of community structure between lakes

Lakes Zot, Lefroy and White Flag lakes recorded a fairly unique community structure in terms of invertebrates. Data collected at Lefroy and Zot was collected in a baseline study in 1999 during a considerable filling event (Brearley *et al.* 1999). Data from White Flag Lake was collected in 2006, when the lake was only partly inundated. This lake had been impacted by dewatering discharge, with very high salinities recorded during the assessment. As a result of conditions at the time of sampling, samples collected from the White Flag sites were fairly sparse, consisting of one *Parartemia* species, hence the difference in community structure.

In terms of species richness, Lakes Austin and Lake Lefroy have reported the greatest species richness over time, with 24 and 15 species respectively recorded from each lake. Lakes such as Black Flag and the Yarra Yarra have reported low species richness, however this is most likely related to poor sampling effort compared to other lakes. Also the lack of filling events, particularly at Black Flag Lake, has resulted in minimal opportunities to sample invertebrate fauna at these lakes.

6.5.4 Impact of dewatering discharge on invertebrates

Dewatering discharge appears to be contributing to lower invertebrate species richness, within the immediate vicinity of the discharge location. Even so, community structure between discharge and natural sites is similar, with comparable species dominating. In the larger lakes, the impact of dewatering discharge during filling events is not as noticeable in the dry phase of the hydrocycle, as the salt crust and other contaminants are diluted and mixed throughout the lakes (Gregory *et al.* 2006). However in the smaller lakes and clay pans, the impacts of dewatering discharge on the invertebrate community are likely to be much greater, due to the smaller dilution factor and therefore greater concentrations of metals and salts likely in these lakes (Foster 2004). In some cases this may prevent certain species from hatching due to the optimum thresholds not being reached, particularly in relation to salinity (Timms 2005a).

6.5.5 Invertebrate community structure versus water chemistry

While salinity has been widely reported as influencing diversity of invertebrates (Williams 1998b), total phosphorus and bicarbonate had the greatest influence on community structure in this study. The influence of parameters other than salinity on community structure is common in the hypersaline lakes as the biota present in salt lakes are able to tolerate a wide range of salinities, and it is therefore likely that other parameters are going to have more of an influence on community structure (Williams *et al.* 1990; Williams 1998a). Taxa such as *Parartemia informis* and Chironomidae species were associated with high concentrations of bicarbonate and higher pH in comparison to the other taxa. Bicarbonate has been found to play a role in community structure of invertebrates in numerous studies (Halse *et al.* 1998; Radke *et al.* 2003; Zhang *et al*. 2007). The contribution of phosphorus to community structure is probably related to the link between the growth of algae and levels of phosphorus (Boulton and Brock 1999), with the algae providing a food source applicable to certain types of species (Khan 2003a).

6.6 Conclusions

The species richness of invertebrates in the salt lakes of this study was high in comparison to other regions in Australia. The Crustacea were dominant, with the most number of taxa recorded from the *Parartemia* genus. This was a reflection of the broad salinity tolerance and proliferation of this group compared to others. There were some differences in community structure between Bioregions, indicating that invertebrates species presence was somewhat related to geography. Community structure was also influenced by water quality parameters, with phosphorus and bicarbonate contributing mostly to variation between and within Bioregions. Species richness was lower at dewatering discharge sites, reporting significantly lower species richness in comparison to 'natural lakes.' There was a strong relationship between water quality and invertebrate community structure within this study. This indicates that invertebrate communities within these lakes are fairly sensitive to change in environmental parameters within surface water.

7 CLASSIFICATION SYSTEM

7.1 Abstract

Classification and ordination of the data presented in the preceding four chapters was considered necessary to simplify the large data set. The objectives of this chapter were firstly to combine and classify data for water quality, sediment chemistry, invertebrates and diatom community structure. Secondly, was to discuss the practical application of the classification system within the mining industry. Two classifications were completed using LINKTREE analysis, one for sediment and diatoms and one for water quality and invertebrates. This was to take into consideration the temporary nature of the water bodies in this study. Six groups were delineated for the sediment and diatom data, each with differing community structure and sediment characteristics. In contrast four groups were distinguished for the water quality and invertebrate data. One of the major findings of the study was that lakes often have sites which fall into more than one grouping, further displaying the high level of heterogeneity within the lakes. It is possible that the resultant groupings may be used to predict the likely impact of dewatering discharge to a specific lake and to determine lakes which contain unique characteristics.

7.2 Introduction

The chemistry and biological characteristics of salt lakes in Western Australia have been described in the previous four chapters of this thesis. This chapter serves to apply these characteristics to develop a classification system for salt lakes in Western Australia. Classification is a tool used to simplify large data sets, grouping objects with similar characteristics (Leps and Smilauer 2003). Worldwide, the most commonly known classification system for wetlands is defined by the Ramsar Convention (Kim *et al.* 2006). In this system, attributes of the wetlands such as water regime, wetland size, vegetation and salinity may be used. This system was considered too coarse for delineating the salt lakes in the study region.

The development of a classification system for salt lakes in the Goldfields was considered critical given the large amount of data that had been collected, but not compiled. The completed classification system should allow for identification of lakes with unique characteristics (Department of Environment 2005). In turn, this has implications for the management objectives for the lakes, which can then be targeted towards the preservation of the lakes unique characteristics (Leathwick *et al.* 2003; Snelder *et al.* 2007).

In Western Australia, classification of wetlands by geomorphic attributes as described by Semeniuk and Semeniuk (1995) is the preferred method of classification (Department of Environment 2005). While this system has been used successfully within the Swan Coastal Plain and the South-West of Western Australia (V. & C. Semeniuk Research Group 1997), it has shown limited application to the lakes in this study (Gregory *et al.* 2006). Using the Semeniuk system, all lakes in the study were classified as playas (Semeniuk and Semeniuk 1995). Even the addition of descriptors to the lakes did little to distinguish particular lakes. While systems such as this are important in classifying wetlands in large regions, the use of geomorphic systems can result in wetlands containing sensitive and unique biotic assemblages to be overlooked (Snelder *et al.* 2006).

As a group, the lakes in this study share similar flooding regime and vegetation characteristics, features which are used to delineate in both the Semeniuk and Ramsar classification systems (Semeniuk and Semeniuk 1997; Ramsar 2006). For this study, it is proposed that these features are used as broad filters to classify the lakes, and that lake classification will be further divided on the basis of chemistry and ecological parameters such as algae and invertebrates as discussed in the previous chapters. While most classification systems are based on qualitative data, classification using multivariate statistics is becoming more commonplace (Snelder *et al.* 2007) and is proposed for this chapter. Classification systems using these methods are repeatable, and not based on qualitative judgments and are therefore easily replicated (Hargrove and Hoffman 2005). A number of classification systems have been designed using biological data for a variety of different aquatic systems using multivariate statistics (Marchant *et al.* 1994; John 2000a; Snelder *et al.* 2006). John (2000) used diatoms and water quality parameters to classify urban streams on the Swan Coastal Plain to determine the impact of urbanization. The use of macroinvertebrates for classification has also occurred extensively throughout Australia (Marchant 1990; Marchant *et al.* 1994; Newall *et al.* 2006).

The primary objective of this chapter was to devise a classification system based on data collected for surface water, sediment chemistry, invertebrate and diatom communities within salt lakes of Western Australia. The practical application of the classification system within the mining industry will also be discussed.

7.3 Materials and Methods

Multivariate analysis of the data was performed using PRIMER (Version 6). The analysis LINKTREE (Linkage Tree's) was used to produce classification and regression trees (Clarke and Gorley 2006). These tools are useful for classifying complex ecological and environmental data (De'ath and Fabricius 2000). Two data sets were compiled for the classification analysis. The diatoms and sediment were combined as was the water quality and invertebrate data. LINKTREE analysis produces a tree like structure, which groups sites on the basis of abiotic and biotic parameters (Clarke and Gorley 2006). The analysis calculates an Analysis of Similarities (ANOSIM) R value which indicates how much overlap the groups have in terms of species, a B% value which represents the difference between groups. Similarity Profiles (SIMPROF) were also calculated to test whether the resultant groups should be further divided. The results table displays environmental parameters which contribute the most to the split in the data. Within the LINKTREE table produced, the first value is for the group on the left of the tree. The values within parenthesis describe the parameters for the groups on the right side of the tree.

Once LINKTREE produced groups based on the two data sets Principal Components Analysis (PCA), Analysis of Variance (ANOVA), ANOSIM and Similarity Percentages (SIMPER) were used to test the differences between the groups.

PCA was used to assess similarities between groups observed in LINKTREE analysis in terms of sediment and water chemistry (Clarke and Gorley 2006). The PCA analysis produces a plot on which sites with similar chemistry are located close together, while those with different chemistries are located further apart. Vectors represent the influence of the different parameters on the data set. The strength of the PCA is explained in terms of percentage variation, a value that should exceed 60 % over the first two axes, in order to adequately represent the data set (Clarke and Gorley 2006).

Minitab (Version 14) was used to perform a ANOVA to determine if environmental parameters were significantly different between groups for both water and sediment quality (p values of <0.050 were considered significant, at a confidence interval of $\alpha = 0.05$) (Minitab Incorporated 2003).

ANOSIM was performed to determine if there were ay significant differences in community structure between groups. One-way design was chosen, with group used as the factor. For ANOSIM, R statistic values close to one indicated that the groups were different, while those close to zero were considered to be similar (Platell *et al.* 1998; Clarke and Gorley 2006). Groups were considered significantly different at $p<0.05$. To determine which species contributed to the average similarity within each group SIMPER analysis was used. The dissimilarity between each group was presented as percentage dissimilarity. Values close to 100% were considered to be completely different with lower values recording more similar species abundances between groups.

7.4 Results

Two classification analyses were completed for biotic and abiotic data, one for diatoms and sediment data and one for live aquatic invertebrates and water chemistry. The data sets for each varied in size and were defined by the amount of 'complete' data available for the environmental parameters.

7.4.1 Classification of sites based on sediment and diatom data

The LINKTREE analysis for diatom and sediment chemistry was based on 22 environmental parameters and diatom data from 86 sites (Figure 20). This analysis produced 6 groups based on these properties. All groups were significantly different from each other. Group 1 was characterized from the rest of the sites by calcium values lower than 270 mg/kg (Figure 20 and Table 23). Although this group was significantly different from the rest of the groups, there was some overlap in diatom community structure with other groups as depicted by the low R statistic (R=0.36) (Table 23).

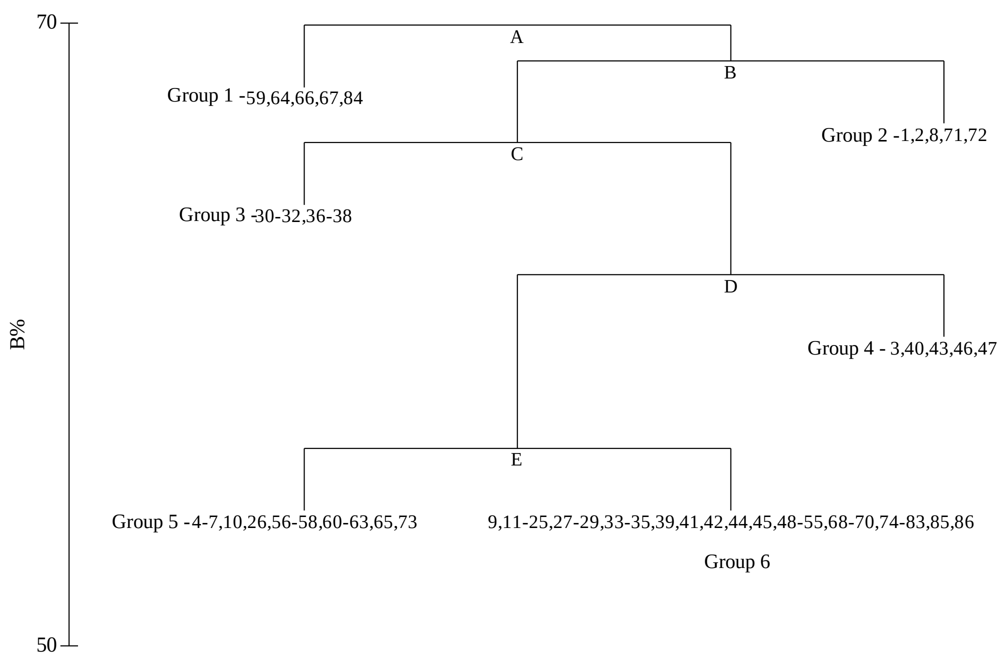

Figure 20. Results of LINKTREE analysis using sediment data and diatom data. Note site names are presented in Appendix J. Diatom data was transformed using a square root transformation and similarity calculated using the Bray Curtis method.

Table 23. Data produced from the LINKTREE analysis of sediment (mg/kg) and diatom community structure, parameters are listed according to the in descending influence that they had on the splits in the data. The concentration of parameters is each group is presented along with the SIMPROF π and p values, the ANOSIM R value and the difference of the groups as a % (B).

Group	Variable	LHS (RHS) split	π	Significance (%)	R	B%
1 → 2	Calcium	< 270 (> 370)	2.34	0.1	0.36	69.9
2 → 3	Magnesium	> 1 120 (< 960)	2.51	0.1	0.38	68.8
3 → 4	Lead	< 1.6 (> 1.7)	2.22	0.2	0.4	66.2
4 → 5	Arsenic	< 38 (> 50)	2.16	0.2	0.39	61.9
5 → 6	Potassium	< 420 (> 430)	2.35	0.2	0.41	56.3

Group 2 differed from group 1, with sites characterized by a calcium value exceeding 370 mg/kg and magnesium concentrations greater than 1 120 mg/kg (Table 23 and Figure 20). Like group 1, this group was significantly different from the rest however there was a considerable degree of overlap in species (R=0.38). Group 3 was separated on the plot, recording values of calcium > 370 mg/kg, magnesium < 960 mg/kg and lead < 1.6 mg/kg. The R statistic (0.4) was slightly higher than the other groups indicating less overlap with other groups. Group 4, recorded calcium > 370 mg/kg, magnesium < 960 mg/kg, lead > 1.7 mg/kg and arsenic > 38 mg/kg, with a R statistic of 0.39 and a group difference of 61.9% (Table 23). Group 5 was characterised by calcium > 370 mg/kg, magnesium < 960 mg/kg, lead > 1.7 mg/kg, arsenic > 50 mg/kg and potassium < 420 mg/kg, and comparatively less overlap in species with the other groups (R=0.41). The largest group was group 6, in terms of the number of sites, it recorded calcium > 370 mg/kg, magnesium < 960 mg/kg, lead > 1.7 mg/kg, arsenic > 50 mg/kg and potassium > 430 mg/kg. Groups 5 and 6 were most dissimilar in terms of species (R=0.41).

It was common for some lakes such as Lake Carey, White Flag Lake and Lake Way to have sites in more than one group (Table 24). In contrast sites within Lake Maitland all fell within group 3. Lakes Johnston, Un-named Lake, South-west Lake, Creek Lake, Lake Hope North, Baladjie Lake, Lake Koorkoordine, Southern Star Lake also fell into one group, however there was only one sampling event for each of these lakes in contrast to the other lakes.

Table 24. Lakes occurring in each of the specific groups as defined by the LINKTREE analysis.

Group	Lakes
Group 1	White Flag Lake, Lake Rebecca, Lake Polaris
Group 2	Banker Lake, Lake Carey, Kurrawang White Lake
Group 3	Lake Maitland,
Group 4	Banker Lake, Lake Raeside, Lake Way,
Group 5	Black Flag, Kurrawang White Lake, Lake Carey, White Flag Lake, Lake Rebecca
Group 6	Lake Carey, Lake Cowan, Lake Lefroy, Lake Way, White Flag Lake, Lake Rebecca, Lake Johnston, Un-named Lake, South-west Lake, Creek Lake, Lake Hope North, Baladjie Lake, Lake Koorkoordine, Southern Star Lake

7.4.2 Testing of diatom/sediment classification

Groups produced using LINKTREE were overlaid on the PCA plot of sediment chemistry for metals, salts, nutrients and pH (Figure 21). Although there was considerable variation between some of groups in terms of sediment chemistry, some differences between groups were obvious. For example group 6 sites generally reported the greatest concentrations of salinity, major anions and cations and nutrients. A large proportion of group 5 sites were characterised by high concentrations of most metals in comparison to the other groups. Group 3 sites were characterised by low concentrations of most metals and higher concentrations of calcium. Groups 2 and 1 had similar chemistry and low concentrations of salts and nutrients and intermediate concentrations of most metals were a feature of this group in comparison to the other groups (although group 1 had comparatively lower calcium). Group 4 was characterised by high concentrations of most metals, particularly arsenic, zinc, manganese, nickel and lead in comparison to the other groups.

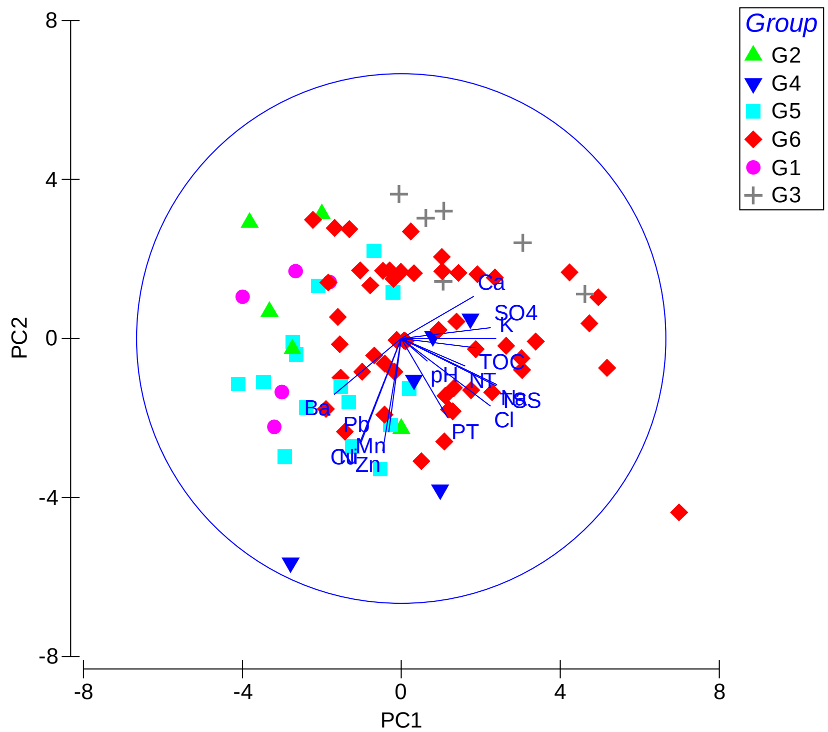

Figure 21. PCA analysis of sediment according to group. 55.4% of variation explained by the first two PCA axes. Concentrations of metals such as arsenic, cobalt and chromium were removed from the analysis as they were collinear with zinc. Also the following parameters were collinear with TSS and were removed; bicarbonate, moisture content and magnesium. All sediment data was transformed and normalized prior to analysis.

The mean and standard deviation of each sediment parameter within each group was calculated (Table 25). While there was some overlap in parameters, in most cases each parameter was significantly different between groups ($p<0.050$ – Appendix K). The exceptions were concentrations of lead, TOC and total phosphorus, which were not significantly different between groups.

Table 25. Mean and standard deviation (SD) of selected sediment parameters of each group. All values are in mg/kg except where stated.

Parameter	G1		G2		G3		G4		G5		G6	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Arsenic	16.6	16.0	16.2	26.8	1.2	2.2	111.4	63.8	6.9	3.1	6.0	7.1
Barium	50.0	25.5	20.8	14.5	12.7	25.1	27.0	30.1	60.0	45.5	24.8	24.2
Bicarbonate	1.1	0.8	21.8	35.5	7.8	4.7	25.0	23.6	6.8	7.2	18.7	40.7
Calcium	198	66	5,490	1,782	145,933	53,125	45,716	89,097	4,624	2,392	28,320	52,861
Chloride	34,120	14,247	29,184	34,063	47,417	18,468	77,940	26,939	54,347	21,067	78,646	45,777
Chromium	115.0	52.9	61.6	53.0	15.5	5.2	88.4	59.0	127.9	70.4	63.4	51.9
Cobalt	6.8	4.1	6.9	5.1	1.3	0.4	95.0	87.1	14.6	14.1	7.8	5.8
Copper	19.2	15.2	13.1	9.2	4.0	1.6	27.1	24.6	22.9	10.6	14.2	12.5
Lead	5.4	2.7	10.3	6.7	1.1	0.3	5.3	2.0	10.0	10.8	5.6	6.2
Magnesium	3,886	1,874	696	307	16,205	15,191	10,974	9,223	4,131	2,300	8,700	7,124
Manganese	191.2	94.6	212.6	148.0	61.7	33.2	895.2	1069.4	442.5	330.9	306.6	387.2
Moisture	10.6	5.7	18.4	9.2	21.4	7.4	38.1	8.1	17.2	5.9	23.7	9.2
Nickel	27.4	17.6	55.2	83.9	2.6	1.1	77.4	55.1	59.3	36.9	22.9	20.7
pH (pH units)	7.3	0.4	7.4	0.4	8.3	0.2	7.8	0.5	7.6	0.6	7.5	0.9
Potassium	224	172	190	120	3,520	1,674	2,792	2,406	249	119	2,697	2,792
Sodium	21,138	9,249	16,844	19,745	35,833	21,744	49,400	15,506	29,700	10,569	57,356	40,815
Sulphate	7,248	3,452	15,310	4,401	31,767	12,437	28,100	12,193	17,643	7,537	28,266	14,690
Total Nitrogen	134.0	87.3	558.0	566.7	611.7	496.6	612.0	569.8	229.3	287.5	544.8	570.7
TOC (%)	0.37	0.24	0.64	0.49	1.73	1.13	0.69	0.16	1.11	0.96	1.51	1.54
Phosphorus	90.8	31.2	64.6	23.5	107.8	84.0	107.6	75.4	101.7	53.6	106.5	90.2
TSS	59,500	17,898	50,240	35,527	83,533	23,330	106,600	29,021	82,100	24,003	118,410	40,308
Zinc	28.0	27.8	15.3	14.3	8.7	3.8	55.4	45.0	22.9	14.8	14.8	14.7

ANOSIM analysis according to group for the diatom data resulted in a global R statistic of 0.527 at p=0.001, indicating that the 6 groups are significantly different from each. Further interpretation of the relationship between groups, showed that some groups were fairly similar and not significantly different (Table 26). For example groups 1 and 2, and groups 1 and 5, were not significantly different from each other. In contrast groups 1 and 6 were significantly different with a high R statistic (0.701) indicating that there was little overlap in species occurring in these groups.

Table 26. Results of ANOSIM analysis between groups based on percentage composition of diatoms.

Group	R statistic	Significance Level
G2, G4	0.028	0.365
G2,G5	0.041	0.332
G2,G6	0.642	0.001
G2,G3	0.452	0.006
G2,G1	-0.216	0.944
G4,G5	0.024	0.364
G4,G6	0.598	0.001
G4,G3	0.336	0.004
G4,G1	0.264	0.127
G5,G6	0.408	0.001
G5,G3	0.222	0.032
G5,G1	-0.029	0.561
G6,G3	0.605	0.002
G6,G1	0.701	0.001
G3,G1	0.608	0.002

In terms of the species contributing to the differences between groups, groups 3 and 1 reported the most dissimilar species abundance (84.7% dissimilar) (Table 27). This is illustrated by a difference in the most abundant species in each group; *Hantzschia* aff. *baltica* was highest in group 3, *Hantzschia amphioxys* was most abundant in group 1, but did not occur in group 3. In contrast abundance of species in groups 5 and 6 were most similar, with *Navicula* aff. *incertata* being dominant in both groups. In addition other abundant species within the groups included, *Luticola mutica* in group 2 and *Amphora coffeaeforms* was in group 4.

Table 27. Results of SIMPER analysis for diatoms according to group.

Species	Mean Abundance Region 1	Region 2	Consistency Ratio	Percent	Cum. %
	G2	**G4**	Mean Dissimilarity= 74.7		
Amphora coffeaeformis	2.4	5.4	1.3	18.5	18.5
Hantzschia aff. *baltica*	2.4	4.1	1.5	16.0	34.5
Luticola mutica	2.9	2.0	1.2	13	47.4
	G2	**G5**	Mean Dissimilarity = 77.0		
Navicula aff. *incertata*	2.6	3.7	1.2	14.4	14.4
Amphora coffeaeformis	2.4	3.1	1.0	14.3	28.7
Hantzschia aff. *baltica*	2.4	1.6	0.8	12.3	41
	G4	**G5**	Mean Dissimilarity = 74.6		
Amphora coffeaeformis	5.4	3.1	1.2	19.1	19.1
Hantzschia aff. *baltica*	4.1	1.6	1.5	14.3	33.4
Navicula aff. *incertata*	1.2	3.7	1.1	14.0	47.4
	G2	**G6**	Mean Dissimilarity = 73.5		
Navicula aff. *incertata*	2.6	6.5	1.5	19.1	19.1
Amphora coffeaeformis	2.4	3.4	1.3	13.4	32.5
Hantzschia aff. *baltica*	2.4	2.6	1.1	13.1	45.6
	G4	**G6**	Mean Dissimilarity = 68.6		
Navicula aff. *incertata*	1.2	6.5	1.9	23.0	23.0
Amphora coffeaeformis	5.4	3.4	1.3	17.6	40.6
Hantzschia aff. *baltica*	4.1	2.6	1.5	12.4	53.0
	G5	**G6**	Mean Dissimilarity = 66.5		
Navicula aff. *incertata*	3.7	6.5	1.4	19.1	19.1
Amphora coffeaeformis	3.1	3.4	1.3	16.8	35.8
Hantzschia aff. *baltica*	1.6	2.6	1.3	12.1	47.9
	G2	**G3**	Mean Dissimilarity = 80.0		
Hantzschia aff. *baltica*	2.4	7.3	1.6	23.4	23.4
Navicula aff. *incertata*	2.6	2.3	1.3	10.8	34.2
Luticola mutica	2.9	0.3	1.1	10.5	44.6
	G4	**G3**	Mean Dissimilarity = 71.3		
Amphora coffeaeformis	5.4	0.3	1.3	23.8	23.8
Hantzschia aff. *baltica*	4.1	7.3	1.4	19.5	43.3
Navicula sp. (Maitland)	0.0	2.2	0.6	10.2	53.5
	G5	**G3**	Mean Dissimilarity = 80.6		
Hantzschia aff. *baltica*	1.6	7.3	1.8	25.4	25.4
Navicula aff. *incertata*	3.7	2.3	1.2	14.1	39.5
Amphora coffeaeformis	3.1	0.3	0.8	12.9	52.4

Species	Mean Abundance		Consistency	Percent	Cum. %
	Region 1	Region 2	Ratio		
	G6	**G3**	Mean Dissimilarity = 71.1		
Hantzschia aff. *baltica*	2.6	7.3	1.7	23.0	23.0
Navicula aff. *incertata*	6.5	2.3	1.6	21.1	44.1
Amphora coffeaeformis	3.4	0.3	1.2	13.8	57.9
	G2	**G1**	Mean Dissimilarity = 71.6		
Hantzschia amphioxys	1.1	3.7	1.3	13.9	13.9
Navicula aff. *incertata*	2.6	2.6	1.2	12.3	26.3
Hantzschia aff. *baltica*	2.4	1.8	0.9	11.9	38.2
	G4	**G1**	Mean Dissimilarity = 79.3		
Amphora coffeaeformis	5.4	1.7	1.3	16.6	16.6
Hantzschia amphioxys	0.2	3.7	1.1	13.7	30.3
Luticola mutica	2.0	3.2	1.3	12.9	43.2
	G5	**G1**	Mean Dissimilarity = 73.9		
Navicula aff. *incertata*	3.7	2.6	1.1	14.7	14.7
Hantzschia amphioxys	2.1	3.7	1.2	14.1	28.7
Amphora coffeaeformis	3.1	1.7	1.0	13.1	41.8
	G6	**G1**	Mean Dissimilarity = 75.8		
Navicula aff. *incertata*	6.5	2.6	1.4	17.6	17.6
Hantzschia amphioxys	0.4	3.7	1.2	13.7	31.3
Luticola mutica	0.5	3.2	1.3	11.6	42.9
	G3	**G1**	Mean Dissimilarity = 84.7		
Hantzschia aff. *baltica*	7.3	1.8	1.7	21.4	21.4
Hantzschia amphioxys	0.0	3.7	1.1	14.2	35.6
Luticola mutica	0.3	3.2	1.2	11.5	47.1

7.4.3 Classification of sites based on water quality and invertebrates

LINKTREE analysis of water and sediment chemistry resulted in the production of four groups (Figure 22). Group 1 differed from the other sites, recording concentrations of sodium > 85 100 mg/L, chloride > 125 000 mg/L and total phosphorus > 1.01 mg/L (Table 28). In contrast concentrations of all these parameters were much lower in all of the other groups. Group 3 sites were distinguished from group 2 sites on the basis of bicarbonate which was > 136 mg/L in group 2 sites and < 120 mg/L in groups 3 and 4. Group 4 differed from group 3 sites, reporting higher concentrations of total nitrogen and potassium in comparison to group 3 sites. In terms of differences between groups, groups 3 and 4 reported the least overlap in species as shown by the highest R statistic of the groups (R=0.56).

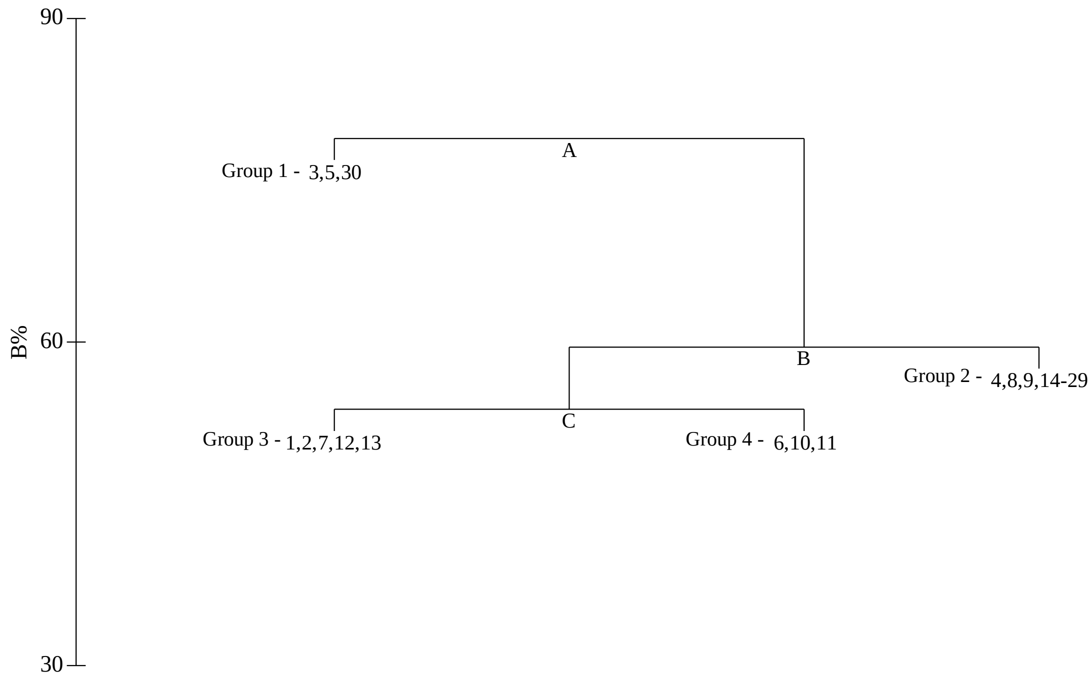

Figure 22. Results of LINKTREE analysis using water quality and presence/absence invertebrate data. Note site names are presented in Appendix L. Similarity was calculated using the Bray Curtis method.

Table 28. Data produced from the LINKTREE analysis of invertebrate presence absence data and water quality. Parameters are listed according to the in descending influence that they had on the splits in the data. The concentration of parameters is each group is presented along with the SIMPROF π and p values, the ANOSIM R value and the difference of the groups as a % (B).

Group	Variable	LHS (RHS) split	π	Significance (%)	R	B%
1 → 2	Sodium	>85 100 (<63 000)	1.52	3.7	0.53	78.9
	Chloride	>125 000 (<98 000)				
	Total Phosphorus	>1.01 (<0.7)				
2 → 3	Bicarbonate	>136 (<120)	1.84	2.8	0.33	59.5
3 → 4	Total Nitrogen	<6.6 (>7.7)	10.92	1.1	0.56	53.8
	Potassium	<1 100 (>1 100)				

Lake Carey was the only lake with sites falling exclusively in one group (Table 29). Sites within White Flag Lake occurred in groups 1 and 2, while Lake Way sites occurred in groups 2 and 4. Group 3 was composed of sites from Yarra Yarra Lakes, which also occurred in group 4.

Table 29. Lakes occurring in each group

Group	Lakes
Group 1	White Flag Lake
Group 2	White Flag Lake, Lake Way, Lake Carey
Group 3	Yarra Yarra Lakes
Group 4	Lake Way, Yarra Yarra

7.4.4 Testing of invertebrate/water quality classification

The PCA plot shows some distinction between water quality in terms of salinity, major anions and cations and pH between groups (Figure 23). Within groups 1, 3 and 4, water quality was similar, but in contrast there was a substantial variation within water quality within group 2. Group 1 was characterised by higher concentrations of calcium, sodium, chloride and total phosphorus in comparison to the other groups. Groups 3 and 4 were dissimilar to the other groups reporting higher concentrations of pH, bicarbonate, total nitrogen and potassium. However, group 4 sites generally recorded higher concentrations of these parameters in comparison to group 3 sites. Concentrations of all parameters were low in group 2 sites in comparison to the other groups.

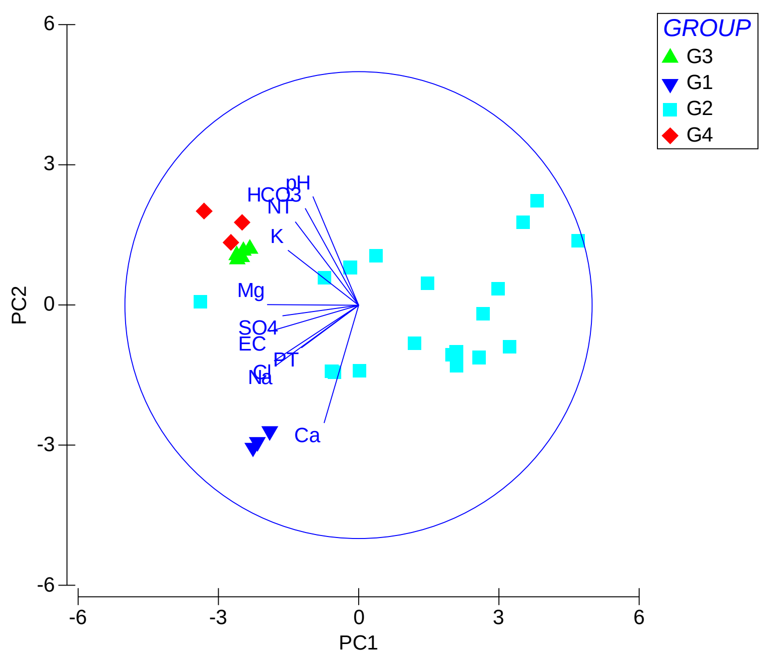

Figure 23. PCA analysis of water chemistry including pH, salinity, major anions and cations according to group. A total of 76.9% variation explained by the first two PCA axes. All data was transformed and normalised prior to analysis.

Results of the ANOVA analysis showed that most water quality parameters were significantly different between groups ($p<0.05$) (Appendix M), even though some overlap in particular variables was evident (Table 30). The exception was calcium which was not significantly different between groups ($p>0.05$).

Table 30. Mean and standard deviation of parameters recorded in each group. All values are reported in mg/L except where mentioned.

Parameter	G1		G2		G3		G4	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
EC (µS/cm)	122,000	1,000	60,516	52,802	130,000	0	127,667	6,807
pH (pH units)	7.1	0.05	7.1	0.45	8.1	0.00	8.1	0.04
Total Nitrogen	1.4	0.2	3.1	4.9	5.4	0.8	10.0	2.6
Total Phosphorus	1.45	0.38	0.07	0.08	0.57	0.14	0.25	0.30
Sodium	86,433	1,193	14,906	14,353	35,800	447	38,700	4,677
Potassium	210	10	494	600	1,080	45	1,850	681
Magnesium	2,843	135	908	878	3,660	89	3,217	536
Calcium	1,370	125	887	499	916	21	592	293
Chloride	128,000	3,000	23,821	22,560	56,600	1,817	61,467	4,606
Sulphate	7,247	445	5,364	3,797	6,360	219	13,100	6,188
Bicarbonate	49	2	72	25	258	19	196	83

The ANOSIM analysis according to group resulted in a global R statistic of 0.435 ($p=0.001$), indicating that the species composition between groups was significantly different, there was however some degree of overlap between species (Table 31). Groups 3 and 4 were completely different as the R statistic reached the maximum for these groups. In contrast groups 2 and 4 were not significantly different and reported a substantial overlap in species ($R=0.28$).

Table 31. Results of ANOSIM analysis between groups based on percentage composition of invertebrates.

Group	R statistic	Significance Level
G3, G4	1	0.018
G3, G2	0.38	0.001
G3,G4	0.56	0.036
G1,G2	0.62	0.002
G1,G4	0.67	0.10
G2,G4	0.28	0.69

In terms of the contribution of particular species, abundance of particular species in groups 3 and 4, 1 and 2, and 1 and 4 were completely different (i.e. 100% dissimilarity) (Table 32). Groups 3 and 4 were most similar in terms of species abundances, with *Parartemia informis* featuring in both sites. *Parartemia informis* was most abundant in group 3 sites, while *Parartemia* species (WF) was most abundance in group 1. Cyclopoida species dominated group 2 and *Parartemia* species 1 dominated group 4 sites.

Table 32. Results of simper analysis for invertebrates according to group. Only the top three species were included (where applicable).

Species	Mean Abundance Region 1	Region 2	Consistency Ratio	Percent	Cum. %
	G3	G1	Mean Dissimilarity= 100		
Parartemia informis	21.4	0.0	1.9	52.7	52.7
Parartemia species (WF)	0.0	43.7	1.4	43.4	96.1
	G3	G2	Mean Dissimilarity = 94.7		
Parartemia informis	21.4	2.3	1.4	33.1	33.1
Cyclopoida species	0.0	31.4	1.1	22.1	55.3
Reticypris species	1.0	25.3	0.8	14.5	69.8
	G1	G2	Mean Dissimilarity = 100.0		
Parartemia species (WF)	43.7	0.00	1.1	34.2	34.2
Cyclopoida species	0.00	31.4	1.0	21.2	55.4
Reticypris species	0.00	25.3	0.7	12.4	67.8
	G3	G4	Mean Dissimilarity = 75.4		
Parartemia informis	21.4	4.6	1.2	48.3	48.3
Parartemia species 1	0.0	36.2	0.9	37.3	85.6
Branchinella species	0.0	9.0	0.7	7.7	93.3
	G1	G4	Mean Dissimilarity = 100		
Parartemia species (WF)	43.7	0.0	1.4	46.3	46.3
Parartemia species 1	0.0	36.2	1.0	30.8	77.0
Parartemia informis	0.0	4.6	0.6	14.0	91.0
	G2	G4	Mean Dissimilarity = 93.2		
Parartemia species 1	5.5	36.2	0.90	24.7	24.7
Cyclopoida species	31.4	0.5	1.0	21.7	46.3
Reticypris species	25.3	0.7	0.8	13.6	60.0

7.5 Discussion

The production of two classification systems was considered necessary given the temporary nature of salt lake systems, and the absence of water for long periods (Roshier and Rumbachs 2004). Sampling of these systems can occur when the lake is dry and in that case only the diatoms and sediment can be sampled. Alternatively, when the lake is in the wet stage of a hydrocycle, a range of biota can be sampled. In this scenario, water quality and invertebrates can be used to classify the lakes. The practical application of each of the systems and the limitations of the classification systems are discussed.

7.5.1 Diatom and sediment classification

The classification of the diatom and sediment data from 86 sites in the study area resulted in the separation of data into six groups that were defined by similar sediment characteristics and diatom community structure (Table 33). A number of the larger lakes, such as Lake Carey, White Flag Lake and Lake Way had sites falling into a number of different groups. This was due to the high degree of variability in sediment chemistry within the lakes, a characteristic of the larger playas in the study area (Arakel *et al.* 1990; John 2003a). Also it is likely that dewatering discharge has contributed to the variability within the sediments in these particular lakes (Finucane 2004; Foster 2004). In addition to highly variable sediments, there was some overlap in species between the groups. This was particularly apparent for groups 5 and group 1, and groups 2 and group 1, which recorded similar species. Despite this overlap, the abundance of these particular species was generally different between groups. The overlap of species between groups indicates that the diatom species occurring within the lakes were able to tolerate a wide range of sediment chemistries. This is a feature of biota that commonly occur in these particular systems (Williams 1998b; Taukulis and John 2006).

Table 33. Summary of lake classification in dry conditions according to sediment chemistry, showing dominant diatom species and lakes in each group.

Group	Sediment Chemistry	Dominant Diatom Species	Lakes
Group 1	Calcium < 270 mg/kg	*Hantzschia amphioxys* *Luticola mutica* *Pinnularia borealis*	White Flag Lake Lake Rebecca Lake Polaris
Group 2	Calcium > 370 mg/kg Magnesium > 1 120 mg/kg	*Luticola mutica Navicula* aff *incertata* *Hantzschia amphioxys*	Banker Lake Lake Carey Kurrawang White Lake
Group 3	Calcium > 370 mg/kg Magnesium < 960 mg/kg Lead < 1.6 mg/kg	*Hantzschia* aff. *baltica* *Navicula* aff. *incertata*	Lake Maitland
Group 4	Calcium > 370mg/kg Magnesium < 960 mg/kg Lead > 1.7mg/kg Arsenic < 38 mg/kg	*Amphora coffeaeformis* *Hantzschia* aff. *baltica* *Nitzschia* aff. *rostellata*	Banker Lake Lake Raeside Lake Way
Group 5	Calcium > 370mg/kg Magnesium < 960 mg/kg Lead > 1.7 mg/kg Arsenic > 50 mg/kg Potassium < 420 mg/kg	*Navicula* aff. *incertata* *Amphora coffeaeformis* *Hantzschia amphioxys*	Black Flag Lake Kurrawang White Lake Lake Carey White Flag Lake Lake Rebecca
Group 6	Calcium > 370mg/kg Magnesium < 960 mg/kg Lead > 1.7 mg/kg Arsenic > 50 mg/kg Potassium > 430 mg/kg	*Navicula* aff. *incertata* *Amphora coffeaeformis* *Hantzschia* aff. *baltica*	Lake Carey, Lake Cowan Lake Lefroy, Lake Way, White Flag Lake, Lake Rebecca, Lake Johnston, Un-named Lake, South-west Lake, Creek Lake, Lake Hope North, Baladjie Lake, Lake Koorkoordine, Southern Star Lake

7.5.2 Invertebrate and water classification

The LINKTREE analysis of 30 sites for invertebrate and water quality resulted in the production of four groups (Table 34). This analysis was only completed on four lakes which fitted the criteria for the analysis (i.e. complete data set), due to the limited opportunities to collect data of this type in recent times and the length of time that the lakes have remained dry (Roshier and Rumbachs 2004). Taxonomic issues also occurred within this data set, with some specimens identified to species level, and others only to genera or family level (Pinder *et al.* 2005). These type of issues are a common trait of classification systems of this type (Herbst 2001; Newall *et al.* 2006). An implication of taxonomic issues is that some lakes may actually have more similar species composition than the analysis suggests.

Table 34. Summary of lake classification in wet conditions according to water chemistry, showing dominant invertebrate species and lakes in each group.

Group	Water Chemistry	Dominant Invertebrate Species	Lakes
Group 1	Sodium > 85 100 mg/L Chloride > 125 000 mg/L Total Phosphorus > 1.0 mg/L	*Parartemia* species (White Flag)	White Flag Lake
Group 2	Sodium < 63 000 mg/L Chloride < 98 000 mg/L Total Phosphorus < 0.7 mg/L Bicarbonate > 136 mg/L	Cyclopoida species *Reticypris* species	White Flag Lake, Lake Way, Lake Carey
Group 3	Sodium < 63 000 mg/L Chloride < 98 000 mg/L Total Phosphorus < 0.7 mg/L Bicarbonate < 120 mg/L Total Nitrogen < 6.6 mg/L Potassium <1 100 mg/L	*Parartemia informis*	Yarra Yarra Lakes
Group 4	Sodium < 63 000 mg/L Chloride < 98 000 mg/L Total Phosphorus < 0.7 mg/L Bicarbonate < 120 mg/L Total Nitrogen > 7.7 mg/L Potassium > 1 100 mg/L	*Parartemia* species 1.	Lake Way, Yarra Yarra Lakes

7.5.3 Practical Applications of the Classification Systems

Both of these systems can potentially be utilized within the mining industry in a number of ways;

1. To predict the likely diatom and invertebrate species within the particular lakes given a specific sediment or water chemistry
2. To consider the impacts of dewatering discharge on a particular lake, on the basis of differing chemistry or biotic data
3. To determine lakes with unique characteristics
4. To consider the management options for the lakes

The results from multivariate programs such as LINKTREE may be used to predict which biota are likely to occur within the particular set of environmental parameters (Clarke and Gorley 2006). For example, if using the classification system for sediment and diatoms, a site that has a concentration of calcium less than 270 mg/kg would fall into group 1. Diatom species such as *Hantzschia amphioxys*, *Luticola mutica* and *Pinnularia borealis* may be common within this particular lake. In contrast, although the LINKTREE analysis produced positive results for the classification system used for identifying likely invertebrates within a given water quality, the system should be used with care. This is due to the high degree of speciation within the lakes and the poor dispersal mechanisms of the invertebrates which inhabit them (Williams 1985; Timms 2007), unlike the diatoms which are fairly widespread (Gell *et al.* 2002).

The system may be used to predict the potential impact of dewatering discharge for a particular lake. Firstly it would be imperative to decide which group the lake fell into, prior to impact. Also the dewatering discharge quality would also need to be known. If the lake fell within group 1 within the sediment/diatom classification system prior to discharge, and the discharge water was high in calcium, it could be assumed that the classification of the lake would change, with species dominant in group 1 such as *Luticola mutica* and *Pinnularia borealis* becoming less abundant. The ongoing collection of data from more lakes will increase the predictive power and robustness of the classification system.

Another application of the classification system for the mining industry is to identify lakes with similar characteristics to the lake they may be potentially impacting. This may help to determine the significance of the particular water body within the region, which is currently a requirement of the annual dewatering discharge licencing report, as set out by the Department of Environment and Conservation (DEC). The classification system indicates that groups 1 – 4 are fairly unique in terms of their sediment and diatom chemistry, and less than six lakes occurred in each group. These results should be viewed in the context that in some cases, the presence of dewatering discharge may have altered the grouping of lakes and therefore contributed to their uniqueness. This was evident in group 2 sites such as Lake Carey, Banker Lake and Kurrawang White Lake which reported high concentrations of magnesium most likely associated with the input of dewatering discharge. However, the other groups contain sites which are not impacted by dewatering discharge and are likely to be relatively unique. These sites are within Lake Rebecca, White Flag Lake, Lake Polaris, Lake Way, Lake Raeside and Lake Maitland. While White Flag Lake, Lake Way and Lake Raeside receive dewatering discharge, there are large areas of these lakes which remain free from the impacts of this practice.

Once the regional significance of a lake is determined, it is easier to identify lakes (or sites within lakes) with unique characteristics which should be preserved or effectively managed to avoid substantial impacts. This may include engineering options with regard to dewatering discharge to preserve particular areas of the lake. Options such as bunding dewatering discharge areas may limit the impact of dewatering discharge on these systems.

7.5.4 Limitations

There were a number of limitations within the production of both of the classification systems. For the diatom/sediment classification the data sets had to be reduced considerably as the analysis would be weakened by missing data (Clarke and Gorley 2006), therefore even though diatom data existed for 350 sites, only 86 of these sites had enough sediment data to run this analysis (using 22 parameters). Similarly the invertebrate/water chemistry classification was also based on a restricted data set, 30 records were used in this instance to run the LINKTREE analysis. Even though the data sets were considerably reduced, there were some trends and some useful conclusions which could be gleaned from the classification system. In addition, classification systems such as these hinge on the assumption that environmental parameters alone are contributing to community structure of biota, which may not always be the case (Snelder *et al.* 2006). In this instance there is a possibility that some other factor not tested in this study may be influencing the data. Both systems would benefit from the ongoing collection of data to enhance the data sets and strengthen the classification system as sampling effort can effect the classification results (Marchant 1990; Gell 1997). The establishment of a protocol for data collection will be essential to ensure uniformity in sampling methods and analysis.

Poor identification of invertebrate species may have also contributed to some error in the classification system. The majority of invertebrate data was collected prior to specialist guides being published. Also, there appears to be some discrepancy in nomenclature. With regards to classification using invertebrate and water quality data, also the stage in which the lake is sampled would influence the lakes classification. The changes over the hydrocycle of these lakes is great and this can also result in major changes in the biotic community.

7.6 Conclusions

Classification and regression tree analysis, using the LINKTREE analysis, were applied to data collected from salt lakes within the study area. Two analyses were completed, one for sediment and diatoms and one for water quality and invertebrates to cater for the temporary nature of the water bodies. Six groups were delineated for the sediment and diatom data, each with differing community structure and sediment characteristics. In contrast four groups were found for the water quality and invertebrate data. These groupings are useful to the mining industry and can help to determine the significance of a particular lake, the impacts of dewatering discharge and management of these systems.

8 CONCLUSIONS

This study represents the first of its kind for salt lakes in the Goldfields region of Western Australia and it has contributed to an increased understanding of the ecological functioning of salt lakes in Western Australia. This goal was considered important, given the number of salt lakes that are being threatened by mining processes in the region. The main parameters of the ecosystem addressed in this study were water and sediment chemistry, diatoms and aquatic invertebrates. Classification of this data was performed using multivariate statistical analysis to produce functional groups with similar abiotic and biotic characteristics.

8.1 Western Australian Salt Lakes

Historically early studies focused on the aquatic invertebrates and basic water chemistry parameters of inland salt lakes in Western Australia (Geddes *et al.* 1981; Brock and Shiel 1983). Recently however, the focus has shifted from studies with a purely ecological purpose to studies researching the potential impact of mining on these lakes (Finucane *et al.* 2001; Foster 2004). Most of this information is restricted to single lakes with very minimal comparison being made between a number of lakes. In addition the episodic nature of the hydrocycle of the lakes in this study has caused some difficulties in determining both the ecological significance of the lakes, and the impacts of mining processes. Comparison and contrast of the data collected in this study has contributed to identifying unique characteristics of the salt lakes and increasing the knowledge base relating to them.

The water chemistry of salt lakes in this study was consistent with that of other inland waters in Australia, specifically relating to pH, salinity and nutrients. The pH was alkaline, salinity ranged from 3 to 390 g/L and the majority of lakes followed the ionic gradient in dominance of Na>Mg>K>Ca (and Ca>K) for the cations and $Cl>SO_4>HCO_3$ for the anions. While these results were fairly consistent, concentrations of metals in surface waters have been scarcely studied in other regions in contrast to the present study. Concentrations of metals such as chromium, cobalt, copper, lead, nickel and zinc exceeded 0.5 mg/L in the lakes and this was considered elevated in comparison to other freshwater and marine systems. Water quality fluctuated substantially over time within the lakes and was influenced by geography, geology, stage of the hydrocycle at the time of sampling and the occurrence of dewatering discharge, as has been reported in other studies (Williams 1998b; Boulton and Brock 1999; Finucane *et al.* 2003; Foster 2004). The larger lakes in this study, such as Lakes Carey, Way and Miranda, reported the greatest variation in water quality, most likely related to the wide range of geological strata that occurred in these lakes (Förstner 1977) in addition to increased sampling effort in these lakes.

The majority of lakes remained dry during the study, and sediments became the focus of the study. Similarly to water quality, there was a high degree of variation in sediment chemistry both within and between the lakes. This variation was generally associated with geological features of the lakes and in some cases, dewatering discharge (Förstner 1977; Foster 2004). Sediments of the lakes were in the most part alkaline, and sodium and chloride dominated the major anions and cations. Concentrations of metals in sediments were a feature of the Bioregion that they occurred in, for example Coolgardie sediments were characterised by elevated concentrations of lead, cadmium, chromium and nickel. Elevated concentrations of zinc, arsenic and copper were common in lakes in the Murchison Bioregion.

A number of the lakes included in this study contained unique attributes in terms of sediment chemistry. For example sediments of Lakes Way and Maitland contained greater concentrations of calcium in comparison to the other lakes, most likely related to the presence of calcretes in these lakes (Mann and Deutscher 1978). In addition Lakes Maitland, Black Flag and Miranda were relatively distinctive in terms of metal concentrations in sediments. While in Lake Miranda sediments were influenced by dewatering discharge, the other two lakes were not impacted by mining activities. Concentrations of all metals in Lake Maitland were low, while concentrations of chromium and cadmium were elevated in Black Flag and were distinct from other lakes in same the region.

Biota present in salt lakes of Western Australia is limited by extreme conditions such as temporary water regime, high salinity and intense temperature. With regard to algae, the focus of this study was on the diatoms as they were often the only biota present in the lakes. A total of 56 diatom species was reported and the most common species were; *Amphora coffeaeformis*, *Hantzschia* aff. *baltica* and *Navicula* aff.

incertata. All these species are widespread and common in inland hypersaline systems in Western Australia (Taukulis and John 2006). Generally, broad tolerances to variation in environmental parameters were evident within the diatoms with species such as *Amphora coffeaeformis* occurring in the majority of the lakes. As would be expected in this type of environment, species richness was compromised as the number of species with the ability to exist in these conditions is greatly reduced. Due to the absence of water, the relationship between diatom and sediment chemistry was explored. In the absence of water, sediments of these lakes provide a medium for periphytic and benthic algae, and it is therefore expected that the chemistry of these sediments will influence the community structure. In this study, zinc, moisture content and cobalt reported the greatest negative influence on the diatoms. While the influence of metals on these particular species is poorly understood the relationship between diatoms and moisture content is well known and expected in the drier environments such as these.

A total of 101 aquatic invertebrate taxa have been recorded from 13 lakes in this study. The Crustacea were dominant, with the greatest number of species recorded from the *Parartemia* genus. A high level of speciation was evident between Bioregions and was most likely related to poor dispersal mechanisms of the species and the large distances between the lakes (Williams 1985; Williams and Kokkinn 1988; Timms 2007). Variability in taxonomic resolution also contributed to differences between lakes and Bioregions with data being identified to species level in some cases, but only genus or family level in other cases. In addition most lakes were only sampled once during the hydrocycle. This would have excluded species which may have been present at other stages of the hydrocycle. Community structure was influenced mostly by phosphorus which may be related to the link between the growth of algae and phosphorus concentrations.

A field sampling protocol has been presented in Appendix A, detailing the type of abiotic and biotic factors which can be sampled in these systems. It addresses parameters which can be sampled in the wet and dry phases of the hydrocycle and site selection. While not an exhaustive manual, it describes the main methods used within this study.

8.2 Dewatering discharge and salt lakes in Western Australia

A total of nine salt lakes (in this study) are currently receiving dewatering discharge from mining operations in Western Australia. An additional eight lakes have received dewatering discharge in the past. Differences between natural lakes and lakes receiving dewatering discharge were considered in this study. Sites receiving dewatering discharge generally reported higher concentrations of salts, nutrients and some metals in both water and sediments compared to natural lakes. Additionally, some discharge lakes reported changes in pH in surface waters. Ultimately the impact of dewatering discharge to the environment is controlled by the local geology within the mining void.

Species richness of biota such as diatoms and invertebrates was generally lower at dewatering discharge sites. Although surface water was generally permanent at each of these sites, the quality was poor and biota was impacted. In the larger lakes, the impact was generally localized within the pooled area of dewatering discharge. In addition, despite these impacts, there appears to be great capacity for amelioration by flushing events, with historical dewatering discharge sites showing minimal impact compared to current dewatering discharge sites. This is a significant finding from this study that requires further investigation. Also, consideration should be given to the fact that the majority of impacts found in this study occurred when the lakes were mostly dry. The impact of dewatering discharge during a filling event or the wet stage of the hydrocycle is yet to be investigated, but on the basis of preliminary data, impact in these conditions appears to be less defined. Also, it is likely that the smaller lakes are more likely to show signs of impact as the discharge plume generally impact the whole lake and the chance of recovery is limited in this instance.

8.3 Comparison of collected data with relevant guidelines

Currently there are no guidelines for comparison with water and sediment chemistry for inland salt lakes in Western Australia. For water quality, the ‘default’ guidelines that are most commonly used for these systems are the Australian and New Zealand Environment Conservation Council (ANZECC) guidelines for marine water. Given that most of the salt lake ecosystems in Western Australia are not pristine and have been impacted by mining, recreation and farming activities they were considered to be slightly to moderately disturbed as defined by the ANZECC guidelines. In this case values for protection of 80% of species in marine water are commonly used (Batley *et al.* 2003). These figures are based on chronic toxicology tests and are aimed to trigger further investigation if exceeded (Batley *et al.* 2003). A comparison of water chemistry from natural waters within this study and ANZECC guidelines was completed. Concentrations of cadmium, cobalt, chromium, copper, lead, nickel and zinc of the natural inland waters all exceeded ANZECC guideline values. This indicates that these parameters are naturally much higher in the inland salt lakes than their freshwater and marine counterparts.

The ANZECC Interim Sediment Guidelines are based on toxicity tests from American aquatic systems (Batley *et al.* 2003; Simpson *et al.* 2005). Therefore it could be argued that they may not be applicable to the inland salt lakes of Western Australia. The trigger values within the guidelines are values for which further investigation should be carried out to determine the impact to biota (Simpson *et al.* 2005). Comparison of sediment from lakes not impacted by dewatering discharge with ANZECC guidelines (ANZECC. 2000b) showed that generally the guidelines were applicable to the salt lakes, with most values in the lake sediments being below trigger values. The exceptions were the trigger values of nickel and chromium which were exceeded by concentrations of these parameters in the natural lakes a reflection of the local geology of the region.

As a rough guide, metal concentrations presented for surface water and sediment quality in the natural lakes (Appendix C and E) could be used for comparison or as a target for data collected from impacted discharge sites.

8.4 Classification System

Although there are a number of widely accepted classification systems within Australia (Pressey and Adam 1995), it was found that none effectively distinguished the differences between salt lakes in the Goldfields. Two analyses were completed for both dry and wet phases of the hydroperiod; one for sediment and diatoms, and one for water quality and invertebrates. Six groups were delineated for the sediment and diatom data, each with differing community structure and sediment characteristics. In contrast four groups were defined for the water quality and invertebrate data. It was common for sites from particular lakes to fall in more than one group, further demonstrating the variability within these ecosystems.

There are a number of practical applications of this system for the mining industry. For example it is possible that biota of a particular lake may be predicted by examining the chemistry of the wetland. While this application is less applicable to the water and invertebrate classification due to speciation and poor dispersal mechanisms (and lack of data), it is possible and more practical for the diatom community. The classification system may also be used to determine lakes that have unique features in comparison to the lakes that were used in this classification system. In addition it may be used for the prediction of impacts based on changes in community structure, if a shift in water or sediment chemistry was observed.

8.5 Research Direction

This study has filled a number of gaps in knowledge on the inland salt lakes of Western Australia. It is the first study of its kind, examining sediment and water chemistry, diatoms and aquatic invertebrates in these systems. The development of a classification system has helped to identify lakes which are considered to be unique or have unique features. Both of these classification systems would benefit from the collection of more data to consolidate statistical relationships. This is particularly important in the case of water quality and invertebrates. A standardized method for sampling and sample analysis has been prepared to ensure consistency for future sampling programs across the state.

The possibility of using diatoms to predict impact of dewatering discharge may be implemented once further data is collected. This would involve using models and transfer functions to predict the optima for species to certain parameters. Transfer functions have been used successfully to predict past hydrochemical features and for assessing water quality in a number of aquatic ecosystems (Gasse *et al.* 1995; Philibert *et al.* 2006).

This study has addressed the impacts of dewatering discharge and initial indications are that these impacts are temporal – that is, they can be ameliorated by large rainfall events. This is an important finding for the mining industry, and it would therefore be advantageous to further investigate this trend.

8.6 Recommendations

For the users of the inland salt lakes of Western Australia, the following approaches are recommended;

- Ongoing collection of natural lake data to create a robust set of target values for Western Australian salt lakes, in preference to the more-generic ANZECC guidelines
- Collection of opportunistic data during flooding events to increase knowledge of ecology and recovery during these conditions
- Classification system data should be updated yearly and sites reclassified, particularly for invertebrates and water quality
- The development of a predictability model for diatoms using transfer function to predict the impact of dewatering discharge.

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Appendix A

Field Sampling Manual

FIELD SAMPLING PROTOCOL

1. Purpose

Sampling of the salt lake systems are required to;

1. Determine whether dewatering discharge from mining operations are impacting the salt lake ecosystem.
2. Assess the recovery of the lake after the discharge is terminated
3. Predict the possible impact, prior to start of discharge

Standard sampling techniques have been outlined within this protocol. Three sampling scenarios are presented; Baseline data collection and Operational Monitoring and Post Closure Monitoring.

2.1 Baseline Data Collection

It is important to collect as much information as possible prior to the commencement of operations for a number of reasons;

1. To determine whether there are unique or protected flora/fauna within the salt lake
2. To establish the baseline conditions of the lake, prior to any discharge, against which future data can be compared
3. To aid in deriving objectives and targets for managing the lakes involved

Baseline data allow the quantification of the potential impacts of dewatering discharge.

When collecting baseline data, the following should be considered;

1. Collection of data over a number of seasons, both in the wet and dry phases of the hydrocycle
2. Chemical parameters that are likely to be affected in the region, and a focus on these parameters in the suite of analysis of the baseline studies
3. Sampling of both the zone of potential impact and comparable control sites, to determine the heterogeneity of the lake
4. Sampling sites with similar geology and geomorphic characteristics, as these parameters can have a profound affect on chemical properties of the lakes

Knowing the chemistry of the water body will contribute greatly to determining the impact of the dewatering discharge on these factors. In addition, the type of aquatic invertebrates which inhabit the wetland should be assessed at various stages of the hydrocycle to determine the full suite of species which occur within the wetland. In regards to algae, phytoplankton (floating algae) should be sampled when wet. When dry, samples of the top sediments (i.e. 5cm) containing diatoms and dehydrated algal mats may be collected. Biota play an important role in the functioning of the wetland and should be considered fully prior to commencement of operations. The baseline condition of riparian vegetation should also be assessed over different seasons to determine the changes in their dynamics over the season. The presence of avian fauna should also be noted, particularly during wet conditions. For comparison with other lakes within the guidelines of the classification system presented in this thesis, diatoms and sediment should be collected as a minimum requirement. Collection methods for these parameters are presented in this manual.

2.2 Operational Monitoring

It is not always possible to collect baseline data due to the long history of mining within some of the salt lake environments. In this case every effort should be made to find control or reference sites within the lake which are not affected by dewatering discharge. If the whole lake is affected by dewatering discharge, then similar lakes within the vicinity may be used as a proxy. Similar lakes should be defined on the basis of sediment chemistry, size and vegetation. The data collected during operational monitoring should be similar to those collected during the baseline studies, except that the operational monitoring sites should be representative of both natural and impacted sites within the lake.

2.3 Post Closure Monitoring

This type of monitoring occurs on cessation of dewatering discharge. Recovery or amelioration of the salt lakes can be monitored over time and should be completed to determine whether the effects of the operation are long-term or not. Changes that would be expected over time may include a reduction in salinity, size of salt crusts, major anions and cations, metals and nutrients. Also, increases in species richness of invertebrates and algae would indicate recovery of the system.

3. Field Sampling Methods

There are a number of methods that should be followed to standardise the collection of data from salt lakes. Table 1 presents the ecosystem components and identifies the parameters that can be analysed in different stages of the hydrocycle. Sample sites should be chosen in areas with adequate access during wet conditions, areas which have a similar geology to the area of impact and areas with similar site morphologies (i.e. embayment's and creek lines).

Table 1. Ecosystem components which can be sampled in certain phases of the hydrocycle

Ecosystem Component	Stage of Hydrocycle	
	Wet Phase	**Dry Phase**
Water	✔	
Sediment	✔	✔
Invertebrates		
Live	✔	✔ (hatching trial)
Cysts/Eggs	✔	✔
Algae		
Phytoplankton	✔	
Benthic Microbial Community	✔	✔
Seeds/Oospores	✔	✔
Vegetation	✔	✔
Avian Fauna	✔	✔

3.1 Surface Water

Sampling procedures from relevant guidelines have been adapted for sampling of these systems (Batley *et al.* 2003; Smith *et al.* 2004). For surface water, take grab samples within the littoral zone, using sampling vessels relevant to the type of analysis to be done. Samples to be analysed for major cations, anions, pH and salinity are to be collected in plastic bottles with no preservative added. Samples to be analysed for nutrients such as phosphorus and nitrogen should be collected in 250 mL plastic bottles (these may contain preservative however this is dependant on the laboratories requirements). Samples to be analysed for dissolved metals must be filtered within a water filter unit through 0.45 µm Millipore glass filters. Samples should be stored in insulated containers with ice bricks and sent to a NATA-accredited laboratory as soon as possible after sampling to meet holding times of certain parameters. Approach the chosen laboratory for holdings times for parameters being tested.

Some parameters such as pH have short holding times and should be sampled *insitu*. Field measurements should be taken for pH, electrical conductivity, dissolved oxygen and temperature using hand held meters. Water depth should also be assessed using a standard ruler as in most cases water depth is less than 1 m.

For further information on the sampling of aquatic systems in regards to ANZECC guidelines see Bately *et al*., 2003 and Smith *et al*., 2004.

3.2 Sediment

Sediment samples for this study were collected using 250mL sterilized glass jars provided by the laboratory. The top 5 cm of sediment should be scraped, with all air excluded from the sample vessel. This depth was chosen for sediments as it was correlated with biota which usually occurs in the top few millimeters of the sediment. Samples should be stored in eskis with ice bricks and transferred to the laboratory for analysis as soon as possible.

For further information on sampling of sediments in relation to guidelines and holding times see Simpson *et al*., 2005.

3.3 Algae

Only diatoms from the benthic microbial communities (BMCs) were assessed for this study. For the analysis, sediment cores were taken using 70mL plastic vials with holes drilled in the bottom of the container to allow air to escape (John 2000) (Plate 1). The vials are to be pushed into the sediment to a depth of approximately 2 cm. The cores are then frozen and transferred to laboratory for analysis. In the laboratory, the top 5 to 10 mm of the sample should be removed from the core surface. The resultant sub-sample should be boiled in 50% nitric acid for four to six hours (‘digested’) to clean the silicon frustules and remove the organic matter from the sample (John 2000). Once digested the samples should be centrifuged five times with the supernatant water removed and replaced with distilled water between each treatment.

Permanent slides can then be made with the sample. Coverslips should be placed on a hotplate with between 50 and 100 μl of sample and enough distilled water to equate to a total of 1000 μl added to the coverslip. The amount of sample added can be dependant on the density of the sample. The coverslip should be allowed to dry, and then inverted onto a permanent slide containing Nathrax. These slides are then heated for approximately 30 minutes until all the air bubbles had been excluded and the Napthrax has set.

Plate 1. Example of a sediment core taken for analysis of diatoms.

The slides should be examined under a compound microscope at (1000x) using oil immersion. Transects of the slide should be observed with any diatoms recorded and identified. The number of diatoms counted may vary between 100 and 300 frustules depending on the density of diatoms within the slide. A number of guides for the identification of diatoms are available (John 1983; Gasse 1986; Cox 1996; John 1998).

If water is present phytoplankton (floating algae) should be collected. This can be achieved by sampling a known volume of water (1 L) and preserved using a few drops of Lugol's solution. The sample should be allowed to settle overnight and the supernatant water siphoned off. The remaining sample can be analysed, with species identified under a compound microscope. It is also possible to sample phytoplankton, using a plankton net of 10 – 40 μm mesh size (John *et al.* 2002). This should be dragged through the water on a transect of known distance. Similar to other methods described, these should be preserved in Lugols solution and observed under a compound microscope. Common salt lake green and blue green algae can be identified using appropriate guides (e.g. Entwisle *et al.* 1997; Baker and Fabbro 1999).

3.4 Invertebrates

There are a number of methods available for collecting salt lake invertebrates. In this study a known volume of water was isolated by a Perspex tube. All invertebrates were removed from the water column using a zooplankton net made of 50µm mesh, and were fixed in 70% ethanol. Other studies use zooplankton nets of varying sizes over transects of known distance (Geddes *et al.* 1981; Timms *et al.* 2006), and this method can be used when the habitats within which sampling is occurring are not ubiquitous. These samples should also be fixed in 70% ethanol.

Specimens can be identified using the appropriate guides (e.g. Williams 1980; Davis and Christidis 1999; Gooderham and Tsyrlin 2002; Timms 2004).

3.4.1 Resting stages

Within the dry phase of the hydrocycle, it is possible to observe the cysts or eggs present in the sediment. Surface soil scraps (to a depth of 1 cm) are taken from a certain area (Plate 2). The samples can be oven dried at 40 °C and sub-samples are sieved through Endecotte® sieves, first 500 µm, followed by 106 µm. Sediment larger than 106 µm is retained for examination. The presence of cysts can be observed under a dissecting microscope (Plate 3)

Plate 2. Collection of sediments for analysis of resting stages

Plate 3 *Parartemia* and *Branchinella* eggs commonly found within sediments of salt lakes. Photo care of Veronica Campagna.

Hatching trials may also be used to determine the species present within a particular lake. While not all species will hatch, some indication of species and environmental preferences for hatching will be gleaned from a study of this nature. To set up a hatching trial, put a known volume of sediment within a clear contain and add distilled water. These samples should be left in a location with a lot of natural light. Invertebrates can hatch within 24-48 hours or later as the water conditions change.

Further information relating to hatching trials and resting stages can be found in Campagna, 2007.

3.5 Vegetation

While not addressed within the framework of this study, dewatering discharge and mining operations can influence the riparian vegetation of the salt lake, particularly if vegetation is inundated. Species should be identified, and estimates of plant cover, density and health should be calculated as a minimum requirement. The establishment of photo points can also aid in determining impact of mining activities to vegetation.

3.6 Avian Fauna

Avian fauna can be prolific within the wet phase of the hydrocycle and it is important to understand what type of fauna is using the lake. When the lakes are inundated, observations should be made on the fauna and their behavior (i.e. breeding and feeding).

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Appendix B

Laboratory methods for assessing sediments and water

Methods for assessing water and sediment quality at ALS

SEDIMENT

Preparation Methods

EK061/EK067 : TKN/TP Digestion - APHA 21st ed., 4500 Norg- D; APHA 21st ed., 4500 P - H. Macro Kjeldahl digestion.

EN34 : 1:5 solid / water leach for soluble analytes - 10 g of soil is mixed with 50 mL of distilled water and tumbled end over end for 1 hour. Water soluble salts are leached from the soil by the continuous suspension. Samples are settled and the water filtered off for analysis.

EN69 : Hot Block Digest for metals in soils sediments and sludges - USEPA 200.2 Mod. Hot Block Acid Digestion 1.0g of sample is heated with Nitric and Hydrochloric acids, then cooled. Peroxide is added and samples heated and cooled again before being filtered and bulked to volume for analysis. Digest is appropriate for determination of selected metals in sludge, sediments and soils. This method is compliant with NEPM (1999) Schedule B(3) (Method 202)

Analytical Methods

EA002 : pH (1:5) - (APHA 21st ed., 4500H+) pH is determined on soil samples after a 1:5 soil/water leach. This method is compliant with NEPM (1999) Schedule B(3) (Method 103)

EA010 : Electrical Conductivity (1:5) - (APHA 21st ed., 2510) Conductivity is determined on soil samples using a 1:5 soil/water leach. This method is compliant with NEPM (1999) Schedule B(3) (Method 104)

EA014 : Total Soluble Salts - In-house. The concentration of TSS in a soil is calculated from the Electrical conductivity of a water extract. This method is compliant with NEPM (1999) Schedule B(3) (Method 104)

EA055-103 : Moisture Content - A gravimetric procedure based on weight loss over a 12 hour drying period at 103-105 degrees C. This method is compliant with NEPM (1999) Schedule B(3) (Method 102)

EG005T : Total Metals by ICP-AES - (APHA 21st ed., 3120; USEPA SW 846 - 6010) (ICPAES) Metals are determined following an appropriate acid digestion of the soil. The ICPAES technique ionises samples in a plasma, emitting a characteristic spectrum based on metals present. Intensities at selected wavelengths are compared against those of matrix matched standards. This method is compliant with NEPM (1999) Schedule B(3)

EG035T : Total Mercury by FIMS - AS 3550, APHA 21st ed., 3112 Hg - B (Flow-injection (SnCl2)(Cold Vapour generation) AAS) FIM-AAS is an automated flameless atomic absorption technique. Mercury in solids are determined following an appropriate acid digestion. Ionic mercury is reduced online to atomic mercury vapour by SnCl2 which is then purged into a heated quartz cell. Quantification is by comparing absorbance against a calibration curve. This method is compliant with NEPM (1999) Schedule B(3)

EK059G : Nitrite and Nitrate as N (NOx)- Soluble by Discrete Analyser - APHA 21st ed., 4500 NO3- F. SEAL Method 2-018-1-L February 2003. Combined oxidised Nitrogen

(NO2+NO3) in a water extract is determined by Cadmium Reduction, and direct colourimetry by SEAL.

EK061G : TKN as N By Discrete Analyser - APHA 21st ed., 4500-Norg-D Soil samples are digested using Kjeldahl digestion followed by determination by Seal Discrete Analyser.

EK062G : Total Nitrogen as N (TKN + NOx) By Discrete Analyser - APHA 21st ed., 4500 Norg/NO3- Total Nitrogen is determined as the sum of TKN and Oxidised Nitrrogen, each determined

seperately as N.

EK067G : Total Phosporous By Discrete Analyser - APHA 21st ed., 4500 P-B&F This procedure involves sulfuric acid digestion and quantification using Seal.

EP004 : Organic Matter - AS1289.4.4.4 - 1997., Dichromate oxidation method after Walkley and Black. This method is compliant with NEPM (1999) Schedule B(3) (Method 105)

WATER

Analytical Methods

EA005-P : pH by PC Titrator - APHA 21st ed. 4500 H+ B. This procedure determines pH of water samples by automated ISE. This method is compliant with NEPM (1999) Schedule B(3) (Appdx. 2)

EA010-P : Conductivity by PC Titrator - APHA 21st ed., 2510 This procedure determines conductivity by automated ISE. This method is compliant with NEPM (1999) Schedule B(3) (Appdx. 2)

EA015 : Total Dissolved Solids - APHA 21st ed., 2540C A gravimetric procedure that determines the amount of `filterable` residue in an aqueous sample. A well-mixed sample is filtered through a glass fibre filter (1.2um). The filtrate is evaporated to dryness and dried to constant weight at 180+5C. This method is compliant with NEPM (1999) Schedule B(3) (Appdx. 2)

EG020A-F : Dissolved Metals by ICP-MS - Suite A - (APHA 21st ed., 3125; USEPA SW846 - 6020, ALS QWI-EN/EG020): The ICPMS technique utilizes a highly efficient argon plasma to ionize selected elements. Ions are then passed into a high vacuum mass spectrometer, which separates the analytes based on their distinct mass to charge ratios prior to their measurement by a discrete dynode ion detector.

EG035F : Dissolved Mercury by FIMS - AS 3550, APHA 21st ed. 3112 Hg - B (Flow-injection (SnCl2)(Cold Vapour generation) AAS) FIM-AAS is an automated flameless atomic absorption technique. A bromate/bromide reagent is used to oxidise any organic mercury compounds in the filtered sample. The ionic mercury is reduced online to atomic mercury vapour by SnCl2 which is then purged into a heated quartz cell. Quantification is by comparing absorbance against a calibration curve. This method is compliant with NEPM (1999) Schedule B(3) (Appdx. 2)

Appendix C

Water Quality Results

Table 1 Minimum and maximum values and number of records for each of the lakes included in the study.

Bioregion	Lake	Discharge/ Natural lake	Min	Max	n
Murchison	Lake Austin	D	6.7	9.9	33
	Lake Carey	D	6.4	8.4	68
	Lake Miranda	D	6.0	8.3	112
	Lake Penny	N	7.6	7.6	1
	Lake Raeside North	D	7.6	7.8	4
	Lake Way	D	5.6	9.6	142
Coolgardie	Binneridgie Road Marsh	N	9.0	10.1	4
	Black Flag Lake	N	8.0	8.7	5
	Golf Lake	N	6.5	7.7	5
	Lake Cowan	D	7.1	8.4	6
	Lake Eaton North	N	7.2	8.1	5
	Lake Fore	D	6.8	7.4	4
	Lake Hope North	D	6.8	7.2	20
	Lake Lefroy	D	6.2	7.1	19
	Lake Tee	D	7.0	7.2	3
	Lake Why	N	9.0	10.6	6
	Lake Zot	N	7.1	8.2	5
	Swan Refuge	N	8.9	8.9	1
	Victory Lake	N	6.3	7.4	5
	White Flag Lake	D	7.3	7.8	8
	Yindarlgooda	D	4.6	7.3	7
Avon/Wheatbelt	Yarra Yarra	D	7.1	8.3	46
Yalgoo	Lake Wownaminya	D	6.4	9.6	10

N = lake not receiving dewatering discharge, D=lake receiving dewatering discharge, n=number of records for each lake

Table 2 Mean TDS (g/L), standard deviation and number of records for each of the lakes included in the study.

Bioregion	Lake	Discharge/ Natural	Mean	Standard deviation	n
Avon/Wheatbelt	Yarra Yarra	D	196	100	46
Coolgardie	Binneridgie Road Marsh	N	3	3	4
	Black Flag Lake	N	22	4	5
	Golf Lake	N	196	35	5
	Lake Cowan	D	329	13	2
	Lake Eaton North	N	81	43	5
	Lake Fore	D	285	33	5
	Lake Hope North	D	206	69	20
	Lake Lefroy	D	258	75	12
	Lake Tee	D	309	30	4
	Lake Why	N	7	3	6
	Lake Zot	N	104	14	5
	Swan Refuge	N	14		1
	Victory Lake	N	189	6	4
	White Flag Lake	D	136	70	8
	Yindarlgooda	D	112	44	7
Murchison	Lake Austin	D	89	50	32
	Lake Carey	D	163	119	70
	Lake Miranda	D	123	99	112
	Lake Penny	N	84		1
	Lake Raeside North	D	155	87	4
	Lake Way	D	111	68	141

N = lake not receiving dewatering discharge, D=lake receiving dewatering discharge, n=number of records for each lake

Table 3 Mean concentrations of total nitrogen and total phosphorus, standard deviation and number of records for each of the lakes included in the study

Bioregion	Lake	Discharge/ Natural	Total Nitrogen			Total Phosphorus			Average N:P ratio
			Mean	Standard dev.	n	Mean	Standard dev.	n	
Avon Wheatbelt	Yarra Yarra	D	9.2	12.9	17	0.38	0.36	17	31
Coolgardie	Black Flag Lake	N	1.2		1	0.18		1	7
	Lake Cowan	D	5.8	6.3	4	0.03	0.03	4	14
	Lake Lefroy	D	4.8	4.7	3	0.07	0.04	6	49
	Swan Refuge	N	0.7		1	0.04		1	18
	White Flag Lake	D	1.8	0.9	6	0.96	0.73	6	15
	Yindarlgooda	D	1.0	0.2	3	0.04	0.03	7	18
	Airstrip Lake	N	1.7		1	0.03		1	
	Creek Lake	N	25.0	19.8	2	0.03	0	2	1000
	South West Lake	N	2.5	0.6	2	0.05	0.03	2	29
Murchison	Lake Austin	D				0.05	0.04	14	
	Lake Carey	D	7.2	21.2	42	0.87	3.90	44	73
	Lake Miranda	D	6.9	4.8	107	0.12	0.24	105	159
	Lake Penny	N				0.09		1	
	Lake Raeside North	D	6.0	5.1	2	0.07	0.04	2	133
	Lake Way	D	6.3	6.3	13	0.24	0.60	18	45
Yalgoo	Lake Wownaminya	D				0.04	0.02	10	

N = lake not receiving dewatering discharge, D=lake receiving dewatering discharge, n=number of records for each lake

Appendix D

Water Quality Statistics

One way ANOVA based on lakes influence by dewatering discharge

One-way ANOVA: pH versus Discharge or Natural

```
Source            DF      SS     MS     F      P
Discharge or nat   1   3.220  3.220  4.84  0.039
Error             21  13.980  0.666
Total             22  17.200

S = 0.8159   R-Sq = 18.72%   R-Sq(adj) = 14.85%


                           Individual 95% CIs For Mean Based on
                           Pooled StDev
Level   N    Mean   StDev   -+---------+---------+---------+--------
D      14  7.4000  0.6000   (--------*--------)
N       9  8.1667  1.0782                (----------*-----------)
                            -+---------+---------+---------+--------
                           7.00      7.50      8.00      8.50

Pooled StDev = 0.8159
```

One-way ANOVA: TDS versus Discharge/ Natural

```
Source            DF      SS     MS      F      P
Discharge/ Natur   1   67160  67160  10.82  0.004
Error             20  124171   6209
Total             21  191331

S = 78.79   R-Sq = 35.10%   R-Sq(adj) = 31.86%


                          Individual 95% CIs For Mean Based on
                          Pooled StDev
Level   N    Mean  StDev  ------+---------+---------+---------+---
D      13  190.15  81.30                      (-------*------)
N       9   77.78  74.89  (--------*--------)
                          ------+---------+---------+---------+---
                               60       120       180       240

Pooled StDev = 78.79
```

One-way ANOVA's according to Bioregion

One-way ANOVA: pH versus Location Code

```
Source          DF       SS     MS     F      P
Location Code    3   18.996  6.332  8.77  0.000
Error          507  365.924  0.722
Total          510  384.920

S = 0.8496   R-Sq = 4.94%   R-Sq(adj) = 4.37%


                                     Individual 95% CIs For Mean Based on
                                     Pooled StDev
Level             N    Mean   StDev  --------+---------+---------+---------
+-
Avon/Wheatbelt   46  7.7826  0.3288           (----*----)
Coolgardie      108  7.2699  1.1735  (--*---)
Murchison       347  7.5207  0.7641         (*-)
Yalgoo           10  8.4750  1.1402                  (---------*----------)
                                     --------+---------+---------+---------
+-
                                           7.50      8.00      8.50
9.00

Pooled StDev = 0.8496
```

One-way ANOVA: TDS g/L versus Location

```
Source     DF       SS      MS      F      P
Location    2   264565  132283  14.56  0.000
Error     496  4504832    9082
Total     498  4769397

S = 95.30   R-Sq = 5.55%   R-Sq(adj) = 5.17%


                                     Individual 95% CIs For Mean Based on
                                     Pooled StDev
Level             N    Mean   StDev  --+---------+---------+---------+------
-
Avon/Wheatbelt   46  195.59  100.02                    (--------*--------)
Coolgardie       91  160.33  104.25           (-----*------)
Murchison       362  124.42   92.32  (--*---)
                                     --+---------+---------+---------+------
-
                                     120       150       180       210

Pooled StDev = 95.30
```

Appendix E

Sediment Quality Results

Table 1 Mean sediment pH (pH units), standard deviation and number of records for each of the lakes included in the study

Bioregion	Lake	Discharge/ Natural	Mean	Standard deviation	N
Avon Wheatbelt	Yarra Yarra Lakes	Both	8.0	0.5	67
Coolgardie	Airstrip Lake	N	7.8	0.5	2
	Baladjie	N	7.9	0.0	2
	Banker Lake	D	7.5	0.5	6
	Black Flag Lake	N	7.7	0.3	9
	Creek Lake	N	4.0		1
	Greta Lake	N	7.5	0.2	6
	Kopai Lake	N	7.7	0.3	18
	Kopai North	N	7.5	0.1	6
	Kurrawang North	N	7.8	0.1	4
	Kurrawang White Lake	D	7.8	0.5	21
	Lake Cowan	D	7.8	0.3	17
	Lake Fore	D	5.3	0.3	7
	Lake Greta	N	7.9	0.2	7
	Lake Hope	N	7.4		1
	Lake Hope North	D	7.8	0.3	5
	Lake Johnston	N	7.3	0.6	2
	Lake Koorkydine	D	8.1		1
	Lake Lefroy	D	8.0	0.6	31
	Lake Polaris	N	7.8		1
	Southern Star	D	7.0	0.5	1
	South-west Lake	N	6.8		2
	Un-named Lake	N	7.8		1
	White Flag Lake	D	7.5	0.4	59
Murchison	Lake Carey	D	7.8	0.6	292
	Lake Maitland	N	8.4	0.2	17
	Lake Miranda	D	7.4	0.6	90
	Lake Raeside North	D	7.2	0.4	7
	Lake Rebecca	N	6.4	1.0	8
	Lake Way	D	8.2	0.3	20
Yalgoo	Lake Wownaminya	D	8.4	0.3	12

N = lake not receiving dewatering discharge, D=lake receiving dewatering discharge

Table 2 Mean total soluble salts (TSS) (g/kg), standard deviation and number of records for WA salt lake sediments.

Bioregion	Lake	Discharge/ Natural	Mean	Standard deviation	n
Avon Wheatbelt	Yarra yarra	Both	154	111	63
Coolgardie	Airstrip Lake	N	104	0	1
	Baladjie Lake	N	124	18	2
	Banker Lake	D	90	40	6
	Black Flag Lake	N	82	54	9
	Creek Lake	N	116	0	1
	Greta Lake	N	88	50	4
	Kopai Lake	N	42	56	10
	Kopai North	N	53	17	2
	Kurrawang White Lake	D	68	28	15
	Lake Cowan	D	143	96	17
	Lake Greta	N	175	61	7
	Lake Hope	N	134	0	1
	Lake Hope North	D	128	70	2
	Lake Johnston	N	79	0	2
	Lake Koorkydine	D	119	0	1
	Lake Lefroy	D	160	150	31
	Lake Polaris	N	30	0	1
	Southern Star	D	33	35	2
	South-west Lake	N	139	0	1
	Swan Refuge	N	126	62	2
	Un-named Lake	N	288	0	1
	White Flag Lake	D	117	84	41
	Yindarlgooda	D	114	66	22
Murchison	Lake Carey	D	114	87	287
	Lake Maitland	N	79	48	17
	Lake Penny	N	200	28	
	Lake Raeside North	D	115	41	7
	Lake Rebecca	N	83	15	8
	Lake Way	D	90	40	20
Yalgoo	Lake Wownaminya	D	14	10	9

Table 3 Mean concentrations of total nitrogen and total phosphorus, standard deviation and number of records for each of the lakes included in the study

Bioregion	Lake	Discharge/ Natural	Total Nitrogen			Total Phosphorus		
			Mean	Standard deviation	n	Mean	Standard deviation	n
Avon Wheatbelt	Yarra Yarra	Both	912	699	66	196	158	66
Coolgardie	Airstrip	N	480		1	41		1
	Baladjie	N	640	594	2	160	42	2
	Banker Lake	D	1312	740	6	149	152	6
	Black Flag Lake	N	368	283	9	136	106	9
	Creek Lake	N	330		1	129		1
	Greta Lake	N	1756	1079	13	114	69	13
	Kopai Lake	N	1758	1498	18	54	61	18
	Kopai North	N	2500	1479	6	23	7	6
	Kurrawang White Lake	D	130	87	21	58	46	21
	Kurrawang North	N	250	35	4	6	4	4
	Lake Cowan	D	273	295	17	63	52	17
	Lake Hope	N	80		1	248		1
	Lake Hope North	D	1770	184	2	186	53	2
	Lake Johnston	N	625	587	2	27	4	2
	Lake Koorkydine	D	2120		1	162		1
	Lake Lefroy	D	126	213	16	34	39	31
	Lake Polaris	N	180		1	57		1
	Southern Star	D	275	82		72	21	2
	South-west Lake	N	770		1	122		1
	Un-named Lake	N	1710		1	560		1
	White Flag Lake	D	529	446	59	94	82	59
Murchison	Lake Carey	D	408	635	180	86	72	179
	Lake Maitland	N	549	404	17	115	65	17
	Lake Miranda	D	564	443	92	264	137	92
	Lake Raeside North	D	1059	638	7	142	79	7
	Lake Rebecca	D	190	104	8	86	27	8
	Lake Way	D	224	171	20	76	31	20
Yalgoo	Lake Wownaminya	D				215	124	12

Table 4 Minimum and maximum total organic carbon (%) in sediments

Bioregion	Lake	Discharge/ Natural	Minimum	Maximum	n
Avon Wheatbelt	Yarra Yarra	Both	0.1	0.83	41
Coolgardie	Airstrip Lake	N	0.52	2.3	2
	Lake Baladjie	N	0.25	0.78	2
	Banker Lake	D	0.53	1.77	6
	Black Flag Lake	N	BD	2.5	8
	Creek Lake	N	5		1
	Greta Lake	N	0.8	0.9	2
	Kopai Lake	N	BD		6
	Kurrawang White Lake	D	BD	1.8	12
	Lake Cowan	D	0.08	2.9	12
	Lake Hope	N	3		1
	Lake Hope North	N	BD	4.4	5
	Lake Johnston	N	3.4	3.5	2
	Lake Koorkydine	D	1.2		1
	Lake Lefroy	D	0.05	0.49	9
	Lake Polaris	N	0.31		1
	Southern Star Lake	D	0.29	0.4	2
	South-west Lake	N	3.8		1
	Swan Refuge	N	0.17	0.28	3
	Un-named Lake	N	6.9		1
	White Flag Lake	D	BD	2.6	32
	Lake Yindarlgooda	D	0.05	0.98	22
Murchison	Lake Carey	D	BD	6.2	129
	Lake Maitland	N	BD	4.2	17
	Lake Penny	N	0.56	0.99	3
	Lake Raeside North	D	0.04	0.96	7
	Lake Rebecca	N	BD	1.3	8
	Lake Way	D	BD	2.3	20
Yalgoo	Lake Wownaminya	D	BD		12

Table 5 – Minimum, maximum and 3rd quartile values for total metals and metalloids in Western Australia salt lake sediments of natural and discharge lakes.

Parameter	Discharge Lakes				Natural Lakes				ANZECC Guideline
	Min	Max	3rd Quartile	N	Min	Max	3rd Quartile	N	
Aluminum	900	31,000	12,425	182	1,360	12,700	8,630	25	
Arsenic	BD	2,100	31	553	BD	11		49	
Barium	BD	290	60	148	BD	200	50	42	
Boron	BD	630		209	50	400	210	17	
Cadmium	BD	160		563	BD	3		49	10
Chromium	BD	1,690	180	563	7	690	225	51	350
Cobalt	BD	368	21	392	BD	34	10	44	
Copper	BD	2,300	44	574	BD	61	23	51	270
Iron	3,900	310,000	43,000	146	4,200	50,400	14,325	26	
Lead	BD	230	16	562	BD	58	9	49	220
Manganese	BD	5,250	400	485	9	650	204	42	
Mercury	BD	1		515	BD	1		42	1
Nickel	BD	2,200	44	521	1	172	47	51	52
Selenium	BD	27		253	BD	2		25	
Strontium	31	1,250	661	27	39	4,360	1,100	17	
Sulfur	3,010	13,800	9,607	20	810	8,880	5,760	3	
Uranium	2	18	4	10	1	45	27	17	
Vanadium	BD	209	88	138	11	238	71	25	
Zinc	BD	20,000	47	571	BD	88	23	51	410

Appendix F

Diatom Species List (including authorities)

Lake	Authority	Airstrip	Baladjie	Banker	Black Flag	Creek	Greta	Kurrawang	Carey	Cowan	Hope	Hope North	Johnston	Johnston	Koorkoordine	Lefroy	Maitland	Polaris	Raeside	Rebecca	Way	Wownaminya	Southern Star	SW lake	Un-named	White Flag
Achnanthes aff. *fogedii*	Hakansson			v																						
Achnanthes brevipes	Agardh													v												
Achnanthes coarctata	Hustedt													v												
Achnanthidium sp. 1									v																	
Amphora aff. *luciae*	Cholnoky																	v	v			v				
Amphora coffeaeformis	(Agardh) Kützing	v	v		v	v		v	v	v	v	v	v	v	v	v	v	v	v		v	v	v	v	v	v
Amphora luciae	Cholnoky								v																	
Amphora sp. 1																			v							
Amphora sp. 1 (Lake Carey)									v																	
Amphora sp. 3									v													v				
Amphora sp. *4*																						v				
Brachysira aff. *manfredii*	Lange-Bertalot								v																	
Caloneis aff. *bacillum*	(Grunow) Cleve			v																						
Caloneis bacillum	(Grunow) Cleve								v																	
Chaetoceros muelleri	Hustedt												v													
Cocconeis placentula	(Ehrenberg) Hustedt																					v				
Craticula cuspidata	(Kütz.) Mann								v																	
Cymbella microcephala	Grunow																	v								
Diploneis sp.			v								v							v								
Diploneis subovalis	Cleve								v																	
Encyonema gracile	Ehrenberg																v					v				
Entomoneis aff. *paludosa*	(W.Smith) Reimer					v																				
Entomoneis paludosa	(W.Smith) Reimer								v																	
Entomoneis tenuistriata	John								v																	
Eunotia sp. 1																		v								
Eunotia sp. 2																		v								
Frustulia magaliesmontana	Cholnoky																v									
Gomphonema sp.																								v		
Hantzschia aff. *baltica*	Simonsen		v			v		v	v	v	v			v	v	v	v	v	v	v	v	v	v		v	v
Hantzschia amphioxys	(Ehrenberg) Grunow			v	v			v	v	v		v				v			v	v	v		v			v
Hantzschia virgata	(Roper) Grunow			v					v																	
Luticola aff. *kotschyi*	Grunow								v																	

Lake	Authority	Airstrip	Baladjie	Banker	Black Flag	Creek	Greta	Kurrawang	Carey	Cowan	Hope	Hope North	Johnston	Johnston	Koorkoordine	Lefroy	Maitland	Polaris	Raeside	Rebecca	Way	Wownaminya	Southern Star	SW lake	Un-named	White Flag
Luticola cohnii	(Hilse) Mann								v																	
Luticola kotschyi	(Grunow) Mann								v																	
Luticola mutica	Kütz.			v				v	v	v						v	v		v	v	v	v				v
Luticola nivalis	(Ehrenb)								v											v						v
Mastogloia halophila	John								v																	
Navicella pusilla	(Grunow ex. A. Schmidt) Krammer		v					v	v					v		v			v		v	v			v	v
Navicula aff. *arvensis*	Hustedt								v																	
Navicula aff. *incertata*	Lange-Bertalot	v	v	v		v		v	v	v	v	v	v	v	v	v	v	v	v	v	v	v	v	v	v	v
Navicula aff. *nivalis*	Ehrenberg			v					v																	
Navicula aff. *salinicola*	Hustedt				v			v	v							v			v	v	v					v
Navicula cruciculoides	Brockman																				v					
Navicula cryptocephala	Kützing								v										v			v				
Navicula cryptocephala var.*exilis*	Kützing								v																	
Navicula elegans	W. Smith			v				v	v				v						v							
Navicula ergadensis	Gregory								v										v							
Navicula halophila	(Grunow) Cleve													v												
Navicula rhynchocephala	Kuetz																v									
Navicula salinicola	Hustedt								v																	
Navicula sp. 1				v					v							v			v		v					
Navicula sp. 1 (Lake Carey)									v																	
Navicula sp. 1 (Lake Maitland)																	v									
Navicula sp. 2																v										
Navicula sp. 2 (lake Carey)									v																	
Navicula sp. 3 (Lake Carey)									v																	
Nitzschia aff. *microcephala*	Grunow																					v				
Nitzschia aff. *rostellata*	Hustedt								v										v		v	v				
Nitzschia aff. *scalpelliformis*	(Grunow) Grunow								v																	
Nitzschia hybrida	Grunow								v																	
Nitzschia linearis	W. Smith																									
Nitzschia ovalis	Arnott								v	v									v	v			v	v		v
Nitzschia pellucida	Grunow								v																	v

Lake	Authority	Airstrip	Baladjie	Banker	Black Flag	Creek	Greta	Kurrawang	Carey	Cowan	Hope	Hope North	Johnston	Johnston	Koorkoordine	Lefroy	Maitland	Polaris	Raeside	Rebecca	Way	Wownaminya	Southern Star	SW lake	Un-named	White Flag
Nitzschia punctata	(W. Smith) Grunow								v										v							
Nitzschia rostellata	Hustedt								v																	
Nitzschia sigma	(Kützing) W. Smith								v																	
Nitzschia sp. 1											v							v								
Pinnularia aff. *borealis*	Ehrenberg								v																	
Pinnularia aff. *subcapitata*	Gregory																					v				
Pinnularia borealis	Ehrenberg			v			v	v	v	v						v	v		v	v	v					v
Pinnularia sp. 1			v	v		v			v			v	v			v		v		v		v	v			
Pinnularia viridis	(Nitzsch) Ehrenberg															v										
Pleurosigma salinarum	Grunow																		v			v				
Proschkinia sp. aff. *complanata*	(Grunow) Mann								v										v			v				
Rhopalodia gibberula	(Ehrenberg) O. Müller																									v
Sellaphora aff. *pupula*	Kützing															v										
Surirella sp. (Lake Carey)									v																	
Surirella sp. 1									v																	v
Surirella sp. 1 (White Flag)																										v
Synedra aff. *radians*	Kützing								v																	
Synedra aff. *vaucheriae*	Kützing																					v				
Synedra cf. *acus*	Kützing																					v				
Thalassionema aff. *nitzschiodes*	Grunow								v																	
Tryblionella acuminata	W. Smith										v			v										v		

Appendix G

Diatom Species Codes for CCA

Species	**Species Code**
Achnanthes coarctata	achcoa
Achnanthes brevipes	achbre
Achnanthes aff. *fogedii*	achafo
Amphora aff. *luciae*	ampalu
Amphora coffeaeformis	ampcof
Amphora sp. 1	ampsp1
Caloneis aff. *bacillum*	calaba
Cymbella microcephala	cymmic
Diploneis sp.	dipsp
Encyonema gracile	encgra
Entomoneis paludosa	entpal
Entomoneis aff. *paludosa*	entapal
Frustulia magaliesmontana	frumag
Hantzschia aff. *baltica*	hanaba
Hantzschia amphioxys	hanamp
Hantzschia virgata	hanvir
Luticola mutica	lutmut
Luticola nivalis	lutniv
Luticola kotschyi	lutkot
Navicella pusilla	navpus
Navicula aff. *incertata*	navain
Navicula aff. *salinicola*	navasa
Navicula aff. *nivalis*	navani
Navicula cruciculoides	navcru
Navicula cryptocephala	navcry
Navicula duerrenbergiana	navdur
Navicula elegans	navele
Navicula ergadensis	navera
Navicula rhynchocephala	navrhy
Navicula sp. 1	navsp1
Navicula sp. 1 (Lake Maitland)	navsp1m
Navicula sp. 1 (Lake Carey)	navsp1c
Navicula sp. 2	navsp2
Nitzschia cf dissipata	nitadi
Nitzschia aff. *rostellata*	nitaro
Nitzschia hybrida	nithyb
Nitzschia pellucida	nitpel
Nitzschia ovalis	nitova
Nitzschia punctata	nitpun
Nitzschia sp. 1	nitsp1
Pinnularia borealis	pinbor
Pinnularia viridis	pinvir
Pinnularia sp. 1	pinsp1
Pleurosigma salinarum	plesal
Proschkinia aff. *complanata*	proaco
Rhopalodia gibberula	phogib
Sellaphora aff. *pupula*	selapu
Surirella sp. 1	sursp1
Surirella sp. 1 (White Flag)	sursp1w
Chaetoceros muelleri	chamue
Eunotia sp. 1	eunsp1
Eunotia sp. 2	eunsp2
Gomphonema sp.	gomsp
Navicula halophila	navhal
Nitzschia acicularis	nitaci
Tryblionella acuminata	tryacu

Appendix H

Invertebrate Species List

TAXA	Kopai Lake	Lake Austin	Lake Carey	Lake Cowan	Lake Hope North	Lake Lefroy	Lake Miranda	Lake Way	Lake Wownaminya	Lake Zot	Yarra Yarra
CRUSTACEA											
Anostraca											
Branchipodidae											
Parartemia informis			v								v
Parartemia sp. 1			v					v			
Parartemia sp 2.								v			
Parartemia sp g							v				
Parartemia serventyi				v							
Parartemia n (austin)		v									
Parartemi a sp. nov.						v				v	
Parartemi a sp. nov. a					v						
Parartemia contracta						v					
Thamnocephalidae											
Branchinella sp.						v					
Branchinella								v			
Notostraca							v				
Triops australiensis			v			v					
Lepidurus apus viridis											
Cladocera											
Daphniidae							v				
Daphniopsis sp.		v	v					v	v		
Daphniopsis pusilla	v										
Moinidae							v				
Moina sp.						v					
Bosminidae											
Bosmina meridionalis						v					
Conchostraca											
Conchostraca sp.1			v								
Conchostraca sp.2			v								
Conchostraca (shells only)		v									
Cyzicus sp.						v					
Limnadia sp						v					
Eulimnadia sp.						v					
Ostracoda	v		v		v		v				
Cyprididae											
Australocypris sp.											
Australocypris sp. nova				v							
Cyprinotus edwardi		v									
Mytilocypris sp 1							v				
Cyprinotus sp.									v		
Diacypris dictyote		v									
Diacypris fodiens				v						v	
Diacypris sp.			v						v		
Diacypris whitei		v		v							
Heterocypris n.sp		v									
Heterocypris sp.				v		v					
Mytilocypris sp 2									v		
Reticypris pinguis		v									
Reticypris sp.			v			v	v	v			

TAXA	Kopai Lake	Lake Austin	Lake Carey	Lake Cowan	Lake Hope North	Lake Lefroy	Lake Miranda	Lake Way	Lake Wownaminya	Lake Zot	Yarra Yarra
Saracypridopsis sp. nov				v							
Undescribed species		v									
Copepoda						v					
Calanoida	v										
Centropagidae											
Boeckella sp.									v		
Calamoecia sp.						v				v	
Cyclopoida			v				v	v	v		v
Cyclopidae											
Metacyclops sp.				v		v				v	
Apocyclops dengizicus				v		v					
Microcyclops ? Platypus?		v									
Copepod nauplii									v		
Harpacticoida							v	v			
Nauplii larval form							v				
ARACHNIDA											
Hydracarina											
Arrenuridae		v									
ROTIFERA											
Monogononta											
Brachionidae											
Branchionus							v		v		
Branchionus nilsoni				v							
Asplanchnidae											
Asplanchna sp.				v							
Asplanchna herricki						v					
Flosculariaceae											
Filinidae											
Filinia perjleri						v					
INSECTA											
Diptera					v						
Culicidae				v		v	v				
Chironomidae	v	v		v			v		v		
Tanypodinae									v		
Chironominae	v								v		
Ceratopogonidae		v		v			v				
Ceratopogonidae sp. 1 (black)						v					
Ceratopogonidae sp. 2						v					
Simuliidae			v								
Tabanidae	v										
Stratiomydae		v									
Dolichopodidae		v									
Plecoptera											
Lepidoptera											
Pyralidae		v									

TAXA	Kopai Lake	Lake Austin	Lake Carey	Lake Cowan	Lake Hope North	Lake Lefroy	Lake Miranda	Lake Way	Lake Wownaminya	Lake Zot	Yarra Yarra
Coleoptera							v				
Dytiscidae				v							
Antiporus sp.	v										
Necterosoma darwini (adults)		v									
Necterosoma darwini (juveniles)		v									
Necterosoma darwini species 2 (adults)		v									
Rhanthus		v									
Allodessus		v									
Dytiscidae sp. 1						v					
Dytiscidae sp. 2						v					
Carabidae											
Haliplidae											
larvae		v									
Hydrophilidae											
Berosus sp. (adults)		v									
Berosus sp. (larvae)		v									
Limnichidae?		v									
Trichoptera											
Helicopsychidae	v										
Leptoceridae											
Oecetis		v									
Odonata											
Anisoptera							v				
Corduliidae											
Hemicordulia australiae		v									
Libellullidae											
Diplacodes bipunctata?		v									
Zygoptera							v				
Lestidae											
Austrolestes annulosus		v									
Austrolestes sp.2		v									
Hemiptera											
Corixidae		v				v					
Micronecta	v										
Notonectidae											
Notonectidae sp. 1						v					
Notonectidae sp. 2						v					
GASTROPODA											
Gastropoda											
Hydrobiidae											
Coxiella sp.							v				
Coxiella gilesi		v									
Potamopyrgus sp.	v										
NEMATODA			v								

Appendix I

Invertebrate Species Codes for CCA

TAXA	CODE
Anisoptera	Anisp
Antiporus sp.	Antsp
Boeckella sp.	Boesp
Branchinella	Brasp
Branchionus	Bransp
Mytilocpris sp. 1	Cypmyt
Calanoida	Calsp
Ceratopogonidae	Cersp
Chironomidae	Chidsp
Chironominae	Chinsp
Choncostraca sp.1	Chosp1
Choncostraca sp.2	Chosp2
Coleoptera Larvae	Colsp
Copepod nauplii	Copsp
Corixidae	Corsp
Coxiella sp.	Coxsp
Culicidae	Culsp
Cyclopoida	Cycsp
Cyprididae *reticypris*	Cypret
Cyprinotus sp.	Cypsp
Daphniidae	Dapsp
Daphniopsis pusilla	Dappus
Daphniopsis sp.	Dapsp1
Diacypris sp.	Diasp
Harpacticoida	Harsp
Moinidae	Moisp
Mytilocpris sp. 2	Mytsp
Nauplii larval form	Naupsp
Nematoda	Nemsp
Notostraca	Notsp
Ostracoda	Ostsp
Parartemia sp 2.	Parsp2
Parartemia informis	Parinf
Parartemia sp (white flag)	ParWF
Parartemia sp. 1	Parsp1
Parartemia sp. n g	Parspng
Potamopyrgus sp.	Potsp
Reticypris sp.	Retsp
Simuliidae	Simsp
Tabanidae	Tabsp
Tanypodinae	Tansp
Trichoptera	Trisp
Triops australiensis	Triaus
Zygoptera	Zygsp

Appendix J

Site code names for sediment in LINKTREE

LINKTREE No.	Site Code	Lake
1	BL3 0806	Banker Lake
2	BL4 0806	Banker Lake
3	BL5 0806	Banker Lake
4	BF3 1106	Banker Lake
5	KDKW1 0306	Kurrawang White Lake
6	KDKW2 0306	Kurrawang White Lake
7	KDKW4 0306	Kurrawang White Lake
8	CM1 1106	Lake Cowan
9	CM3 1106	Lake Cowan
10	CM7 1106	Lake Cowan
11	WF1 0306	White Flag Lake
12	WF3 0306	White Flag Lake
13	WF6 0306	White Flag Lake
14	WF7 0306	White Flag Lake
15	WF8 0306	White Flag Lake
16	WF1 1106	White Flag Lake
17	WF2 1106	White Flag Lake
18	WF3 1106	White Flag Lake
19	WF5 1106	White Flag Lake
20	WF8 1106	White Flag Lake
21	NE Ref 0606	Lake Lefroy
22	WP49J 0606	Lake Lefroy
23	RLM8	Lake Maitland
24	RLM9	Lake Maitland
25	RLM10	Lake Maitland
26	RLM11	Lake Maitland
27	RLM12	Lake Maitland
28	RLM13	Lake Maitland
29	RLM14	Lake Maitland
30	RLM15	Lake Maitland
31	RLM16	Lake Maitland
32	LR1	Lake Raeside
33	LR2	Lake Raeside
34	LR3	Lake Raeside
35	LR4	Lake Raeside
36	LR5	Lake Raeside
37	LR6	Lake Raeside
38	LR7	Lake Raeside
39	LWPIPE1 0506	Lake Way
40	LWPIPE2 0506	Lake Way
41	WP2 0506	Lake Way
42	WP3 0506	Lake Way
43	WP4 0506	Lake Way
44	WP6 0506	Lake Way
45	WP7 0506	Lake Way
46	LW3 0506	Lake Way
47	LW9 0506	Lake Way
48	SG1	Lake Rebecca
49	SG3	Lake Rebecca
50	SG5	Lake Rebecca
51	SG6	Lake Rebecca
52	SG7	Lake Rebecca
53	SG8	Lake Rebecca
54	KW1 1106	Kurrawang White Lake
55	KW2 1106	Kurrawang White Lake

LINKTREE No.	Site Code	Lake
56	KW4 1106	Kurrawang White Lake
57	LJN	Lake Johnston
58	Un-named Lake	Un-named Lake
59	Airstrip Lake	Airstrip Lake
60	SW lake	South-west Lake
61	Lake Johnston	Lake Johnston
62	Creek Lake	Creek Lake
63	LHN1	Lake Hope North
64	LK1	Lake Koorkorrdine
65	BAL2	Baladjie Lake
66	Lake Hope	Lake Hope
67	LP1	Lake Polaris
68	BAL1	Baladjie Lake
69	ST1	Southern Star lake
70	SDN 1 0906	Lake Carey
71	SDS 4 0906	Lake Carey
72	SDN1 0407	Lake Carey
73	SDS4 0407	Lake Carey
74	MR2	Lake Carey
75	MR3	Lake Carey
76	MR4	Lake Carey
77	MR5	Lake Carey
78	MR6	Lake Carey
79	MR7	Lake Carey
80	MR8	Lake Carey
81	MR9	Lake Carey
82	MR10	Lake Carey
83	MR11	Lake Carey
84	MR12	Lake Carey
85	Site 15 0606	Lake Carey
86	Site 16 0606	Lake Carey

Appendix K

Minitab results for sediment versus group

One-way ANOVA: As versus Group

```
Source  DF     SS     MS      F      P
Group    5  52399  10480  36.73  0.000
Error   80  22828    285
Total   85  75227

S = 16.89   R-Sq = 69.66%   R-Sq(adj) = 67.76%


                          Individual 95% CIs For Mean Based on
                          Pooled StDev
Level   N    Mean  StDev  ----+---------+---------+---------+-----
G1      5   16.60  16.01      (----*---)
G2      5   16.20  26.79      (----*---)
G3      6    1.16   2.22  (---*---)
G4      5  111.40  63.81                                (---*---)
G5     15    6.93   3.05      (-*-)
G6     50    5.96   7.13      (-*)
                          ----+---------+---------+---------+-----
                              0        35        70       105

Pooled StDev = 16.89
```

One-way ANOVA: Ba versus Group

```
Source  DF     SS    MS     F      P
Group    5  18934  3787  4.46  0.001
Error   80  67852   848
Total   85  86786

S = 29.12   R-Sq = 21.82%   R-Sq(adj) = 16.93%


                         Individual 95% CIs For Mean Based on
                         Pooled StDev
Level   N   Mean  StDev  ----+---------+---------+---------+-----
G1      5  50.00  25.50                (---------*---------)
G2      5  20.80  14.52    (---------*----------)
G3      6  12.65  25.13  (--------*---------)
G4      5  27.00  30.12      (----------*---------)
G5     15  60.00  45.47                        (-----*-----)
G6     50  24.82  24.19            (--*--)
                         ----+---------+---------+---------+-----
                             0        25        50        75

Pooled StDev = 29.12
```

One-way ANOVA: hco3 tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  13.355  2.671  2.74  0.024
Error   80  77.864  0.973
Total   85  91.219

S = 0.9866   R-Sq = 14.64%   R-Sq(adj) = 9.31%


                             Individual 95% CIs For Mean Based on
                             Pooled StDev
Level   N     Mean   StDev    -+---------+---------+---------+--------
G1      5  -1.2649  0.5682    (-------*--------)
G2      5   0.4361  0.9700                     (-------*--------)
G3      6   0.2511  0.3560                   (--------*-------)
G4      5   0.8936  0.6861                         (--------*--------)
```

```
G5      15  -0.0610  0.6461                   (----*----)
G6      50   0.0176  1.1468                      (--*--)
                               -+---------+---------+---------+--------
                             -2.0      -1.0       0.0       1.0

Pooled StDev = 0.9866
```

One-way ANOVA: Ca versus Group

```
Source  DF           SS           MS     F      P
Group    5  99106196512  19821239302  8.67  0.000
Error   80  1.82878E+11   2285978447
Total   85  2.81984E+11

S = 47812   R-Sq = 35.15%   R-Sq(adj) = 31.09%


                          Individual 95% CIs For Mean Based on
                          Pooled StDev
Level   N    Mean  StDev  -------+---------+---------+---------+--
G1      5     198     66  (------*------)
G2      5    5490   1782   (------*------)
G3      6  145933  53125                         (-----*------)
G4      5   45716  89097          (------*------)
G5     15    4624   2392      (---*---)
G6     50   28320  52861           (--*-)
                          -------+---------+---------+---------+--
                                 0     60000    120000    180000

Pooled StDev = 47812
```

One-way ANOVA: cl tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  21.040  4.208  6.70  0.000
Error   80  50.224  0.628
Total   85  71.264

S = 0.7923   R-Sq = 29.52%   R-Sq(adj) = 25.12%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev  -----+---------+---------+---------+----
G1      5  -0.8533  0.5910        (-------*--------)
G2      5  -1.3140  1.2462  (--------*-------)
G3      6  -0.3715  0.5684              (-------*-------)
G4      5   0.4480  0.6444                        (--------*-------)
G5     15  -0.1713  0.6105                    (----*----)
G6     50   0.3685  0.8345                             (--*-)
                            -----+---------+---------+---------+----
                              -1.60     -0.80     -0.00      0.80

Pooled StDev = 0.7923
```

One-way ANOVA: cr tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  23.234  4.647  6.60  0.000
Error   80  56.327  0.704
Total   85  79.561

S = 0.8391   R-Sq = 29.20%   R-Sq(adj) = 24.78%
```

```
                               Individual 95% CIs For Mean Based on Pooled
                               StDev
Level   N     Mean   StDev    +---------+---------+---------+---------
G1      5   0.6758  0.6521                      (-------*------)
G2      5  -0.2834  1.1082             (------*-------)
G3      6  -1.3042  0.2819    (------*------)
G4      5   0.2371  0.9171                  (------*-------)
G5     15   0.7634  0.8756                          (----*---)
G6     50  -0.1640  0.8478                   (-*--)
                              +---------+---------+---------+---------
                           -2.0      -1.0       0.0       1.0

Pooled StDev = 0.8391
```

One-way ANOVA: Co versus Group

```
Source  DF     SS    MS      F      P
Group    5  36208  7242  16.60  0.000
Error   80  34895   436
Total   85  71103

S = 20.89   R-Sq = 50.92%   R-Sq(adj) = 47.86%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N   Mean  StDev  ----+---------+---------+---------+-----
G1      5   6.80   4.09   (----*----)
G2      5   6.90   5.05   (----*----)
G3      6   1.28   0.41  (---*----)
G4      5  95.00  87.07                         (----*----)
G5     15  14.60  14.10       (--*--)
G6     50   7.77   5.75       (*-)
                         ----+---------+---------+---------+-----
                             0        35        70       105

Pooled StDev = 20.89
```

One-way ANOVA: Cu versus Group

```
Source  DF     SS   MS     F      P
Group    5   2469  494  3.05  0.014
Error   80  12967  162
Total   85  15436

S = 12.73   R-Sq = 16.00%   R-Sq(adj) = 10.75%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N   Mean  StDev  -----+---------+---------+---------+----
G1      5  19.20  15.19             (--------*--------)
G2      5  13.10   9.24        (---------*--------)
G3      6   3.95   1.55  (-------*--------)
G4      5  27.10  24.56                   (---------*--------)
G5     15  22.93  10.62                    (----*-----)
G6     50  14.17  12.53              (--*--)
                         -----+---------+---------+---------+----
                              0        12        24        36

Pooled StDev = 12.73
```

One-way ANOVA: Pb versus Group

```
Source  DF      SS    MS     F      P
Group    5   483.0  96.6  2.05  0.080
Error   80  3764.3  47.1
Total   85  4247.4

S = 6.860   R-Sq = 11.37%   R-Sq(adj) = 5.83%


                           Individual 95% CIs For Mean Based on
                           Pooled StDev
Level   N    Mean   StDev  -------+---------+---------+---------+--
G1      5   5.400   2.679        (---------*---------)
G2      5  10.300   6.723                (---------*---------)
G3      6   1.100   0.283  (--------*--------)
G4      5   5.300   1.987        (---------*---------)
G5     15  10.033  10.816                    (-----*-----)
G6     50   5.626   6.228              (--*---)
                           -------+---------+---------+---------+--
                                0.0       6.0      12.0      18.0

Pooled StDev = 6.860
```

One-way ANOVA: mg tran versus Group

```
Source  DF       SS     MS      F      P
Group    5   48.293  9.659  10.49  0.000
Error   80   73.649  0.921
Total   85  121.942

S = 0.9595   R-Sq = 39.60%   R-Sq(adj) = 35.83%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean  StDev   ---+---------+---------+---------+------
G1      5  -0.4826  0.7226                (-----*----)
G2      5  -2.5459  0.7321  (-----*-----)
G3      6   1.0361  1.0688                           (----*----)
G4      5   0.5012  1.2434                       (----*-----)
G5     15  -0.4669  0.7831                   (--*--)
G6     50   0.2740  0.9993                         (-*-)
                            ---+---------+---------+---------+------
                            -3.0      -1.5       0.0       1.5

Pooled StDev = 0.9595
```

One-way ANOVA: mn tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  15.935  3.187  4.00  0.003
Error   80  63.784  0.797
Total   85  79.719

S = 0.8929   R-Sq = 19.99%   R-Sq(adj) = 14.99%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean  StDev   -------+---------+---------+---------+--
G1      5  -0.0925  0.4762          (-------*-------)
G2      5  -0.1112  0.7050          (-------*-------)
G3      6  -1.0139  0.4622  (------*------)
G4      5   1.0019  0.9240                     (-------*-------)
```

```
G5     15   0.5049  0.6601                   (----*----)
G6     50  -0.1288  1.0132               (--*-)
                                 -------+---------+---------+---------+--
                                     -1.0       0.0       1.0       2.0

Pooled StDev = 0.8929
```

One-way ANOVA: mc tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  22.615  4.523  7.26  0.000
Error   80  49.825  0.623
Total   85  72.440

S = 0.7892   R-Sq = 31.22%   R-Sq(adj) = 26.92%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev  -------+---------+---------+---------+--
G1      5  -1.3231  0.9573  (-----*-----)
G2      5  -0.3392  0.9367          (-----*-----)
G3      6   0.0212  0.7202              (----*-----)
G4      5   1.3380  0.5235                        (-----*-----)
G5     15  -0.4072  0.6091            (---*--)
G6     50   0.1949  0.8301                   (-*)
                            -------+---------+---------+---------+--
                                -1.2       0.0       1.2       2.4

Pooled StDev = 0.7892
```

One-way ANOVA: ni tran versus Group

```
Source  DF      SS     MS      F      P
Group    5  33.511  6.702  10.25  0.000
Error   80  52.312  0.654
Total   85  85.823

S = 0.8086   R-Sq = 39.05%   R-Sq(adj) = 35.24%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev  ---------+---------+---------+---------+
G1      5   0.0885  0.6966                (-----*-----)
G2      5   0.3281  1.4944                  (-----*-----)
G3      6  -1.6492  0.2885  (----*-----)
G4      5   1.1546  0.7297                         (-----*-----)
G5     15   0.7563  0.8588                        (--*---)
G6     50  -0.1662  0.7635                  (-*-)
                            ---------+---------+---------+---------+
                                  -1.2       0.0       1.2       2.4

Pooled StDev = 0.8086
```

One-way ANOVA: pH tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  14.552  2.910  3.09  0.013
Error   80  75.405  0.943
Total   85  89.957

S = 0.9709   R-Sq = 16.18%   R-Sq(adj) = 10.94%
```

```
                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev  -----+---------+---------+---------+----
G1      5  -0.6672  0.5620  (-------*--------)
G2      5  -0.5539  0.5955   (-------*--------)
G3      6   1.3122  0.4409                     (-------*-------)
G4      5   0.2009  0.9885          (--------*--------)
G5     15  -0.0111  0.9068            (----*----)
G6     50  -0.0737  1.0722              (-*--)
                            -----+---------+---------+---------+----
                               -1.0       0.0       1.0       2.0

Pooled StDev = 0.9709
```

One-way ANOVA: K versus Group

```
Source  DF         SS        MS     F      P
Group    5  119299372  23859874  4.55  0.001
Error   80  419375981   5242200
Total   85  538675353

S = 2290   R-Sq = 22.15%   R-Sq(adj) = 17.28%


                        Individual 95% CIs For Mean Based on
                        Pooled StDev
Level   N  Mean  StDev  ---------+---------+---------+---------+
G1      5   224    172  (---------*---------)
G2      5   190    120  (---------*---------)
G3      6  3520   1674                   (---------*--------)
G4      5  2792   2406               (---------*---------)
G5     15   249    119      (-----*-----)
G6     50  2697   2792                     (--*---)
                        ---------+---------+---------+---------+
                                 0      2000      4000      6000

Pooled StDev = 2290
```

One-way ANOVA: Na tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  29.681  5.936  7.50  0.000
Error   80  63.282  0.791
Total   85  92.964

S = 0.8894   R-Sq = 31.93%   R-Sq(adj) = 27.67%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev   --+---------+---------+---------+-------
G1      5  -0.9716  0.6361      (-------*-------)
G2      5  -1.4231  1.1374   (-------*-------)
G3      6  -0.2764  0.9209               (------*------)
G4      5   0.3791  0.6605                     (-------*-------)
G5     15  -0.4707  0.5560                (---*----)
G6     50   0.4590  0.9708                           (--*-)
                             --+---------+---------+---------+-------
                            -2.0      -1.0       0.0       1.0

Pooled StDev = 0.8894
```

One-way ANOVA: So4 tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  24.126  4.825  6.04  0.000
Error   80  63.880  0.798
Total   85  88.005

S = 0.8936   R-Sq = 27.41%   R-Sq(adj) = 22.88%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev  ---+---------+---------+---------+------
G1      5  -1.5538  0.4288  (------*-------)
G2      5  -0.6795  0.4025          (-------*-------)
G3      6   0.5670  0.7816                       (-------*------)
G4      5   0.2973  0.9293                    (-------*-------)
G5     15  -0.5122  0.7394               (----*---)
G6     50   0.2626  0.9932                         (--*-)
                            ---+---------+---------+---------+------
                            -2.0      -1.0       0.0       1.0

Pooled StDev = 0.8936
```

One-way ANOVA: Nt tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  11.921  2.384  2.39  0.045
Error   80  79.695  0.996
Total   85  91.615

S = 0.9981   R-Sq = 13.01%   R-Sq(adj) = 7.58%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev    -+---------+---------+---------+--------
G1      5  -0.6095  0.5896    (-----------*------------)
G2      5   0.2471  0.9113                (------------*-----------)
G3      6   0.3883  0.8415                   (-----------*----------)
G4      5   0.4050  0.8585                  (------------*-----------)
G5     15  -0.6633  1.0284        (-------*------)
G6     50   0.1701  1.0467                         (---*---)
                              -+---------+---------+---------+--------
                            -1.40     -0.70      0.00      0.70

Pooled StDev = 0.9981
```

One-way ANOVA: TOC versus Group

```
Source  DF      SS    MS     F      P
Group    5   12.21  2.44  1.43  0.224
Error   80  136.97  1.71
Total   85  149.18

S = 1.308   R-Sq = 8.19%   R-Sq(adj) = 2.45%


                         Individual 95% CIs For Mean Based on
                         Pooled StDev
Level   N   Mean  StDev  --------+---------+---------+---------+-
G1      5  0.372  0.241  (-----------*----------)
G2      5  0.636  0.490     (----------*-----------)
G3      6  1.725  1.133               (---------*----------)
G4      5  0.686  0.157     (-----------*-----------)
G5     15  1.118  0.961           (------*------)
G6     50  1.512  1.541                  (---*---)
                         --------+---------+---------+---------+-
```

```
                              0.0       1.0       2.0       3.0

Pooled StDev = 1.308
```

One-way ANOVA: PT versus Group

```
Source  DF      SS    MS     F      P
Group    5    8986  1797  0.29  0.920
Error   80  503174  6290
Total   85  512159

S = 79.31   R-Sq = 1.75%   R-Sq(adj) = 0.00%


                          Individual 95% CIs For Mean Based on
                          Pooled StDev
Level    N    Mean  StDev -+---------+---------+---------+--------
G1       5   90.80  31.20      (-------------*-------------)
G2       5   64.60  23.46 (-------------*-------------)
G3       6  107.83  84.04           (------------*-----------)
G4       5  107.60  75.35         (--------------*-------------)
G5      15  101.67  53.64              (-------*-------)
G6      50  106.52  90.21                   (---*----)
                          -+---------+---------+---------+--------
                           0        50       100       150

Pooled StDev = 79.31
```

One-way ANOVA: Tss tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  32.290  6.458  9.20  0.000
Error   80  56.171  0.702
Total   85  88.460

S = 0.8379   R-Sq = 36.50%   R-Sq(adj) = 32.53%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level    N     Mean   StDev -------+---------+---------+---------+--
G1       5  -1.1471  0.5425    (---------*--------)
G2       5  -1.4334  1.0522 (--------*--------)
G3       6  -0.4376  0.6719              (--------*-------)
G4       5   0.2033  0.7830                     (---------*--------)
G5      15  -0.4805  0.7026                 (----*----)
G6      50   0.4613  0.8915                               (--*--)
                            -------+---------+---------+---------+--
                               -1.60     -0.80     -0.00      0.80

Pooled StDev = 0.8379
```

One-way ANOVA: Zn tran versus Group

```
Source  DF      SS     MS     F      P
Group    5  14.469  2.894  3.73  0.004
Error   80  62.124  0.777
Total   85  76.594

S = 0.8812   R-Sq = 18.89%   R-Sq(adj) = 13.82%


                            Individual 95% CIs For Mean Based on
```

```
                                 Pooled StDev
Level    N      Mean   StDev   --+---------+---------+---------+-------
G1       5    0.3019  1.1474          (-------*-------)
G2       5   -0.1734  0.9694     (-------*-------)
G3       6   -0.4798  0.4112   (------*------)
G4       5    1.2316  0.8380                  (-------*-------)
G5      15    0.3369  0.7776              (---*----)
G6      50   -0.2602  0.9145          (-*--)
                               --+---------+---------+---------+-------
                               -1.0       0.0       1.0       2.0

Pooled StDev = 0.8812
```

Appendix L

Site code names for water quality versus invertebrates

LINKTREE Site Code	Site Code	Lake
1	DISL1 0906	Yarra Yarra
2	DISL2 0906	Yarra Yarra
3	KDWF2	White Flag Lake
4	KDWF3	White Flag Lake
5	KDWF6	White Flag Lake
6	L1 0906	Yarra Yarra
7	L2 0906	Yarra Yarra
8	LW2 0305	Lake Way
9	LW3 0305	Lake Way
10	LWpipe1 0305	Lake Way
11	LWpipe2 0305	Lake Way
12	R1 0906	Lake Way
13	R2 0906	Lake Way
14	SDDIS 0404	Lake Carey
15	Site 11 0304	Lake Carey
16	Site 12 0304	Lake Carey
17	Site 12 0404	Lake Carey
18	Site 14 0304	Lake Carey
19	Site 14 0404	Lake Carey
20	Site 16 0304	Lake Carey
21	Site 17 0304	Lake Carey
22	Site 17 0404	Lake Carey
23	Site 18 0304	Lake Carey
24	Site 18 0404	Lake Carey
25	Site 3 0404	Lake Carey
26	Site 7 0304	Lake Carey
27	Site 7 0404	Lake Carey
28	WALDIS 0304	Lake Carey
29	WALDIS 0404	Lake Carey
30	WFdischarge site	White Flag Lake

Appendix M

MINITAB results for water quality versus group

One-way ANOVA: EC versus Group

```
Source  DF           SS           MS     F      P
Group    3  31083340070  10361113357  5.36  0.005
Error   26  50279311930   1933819690
Total   29  81362652000

S = 43975   R-Sq = 38.20%   R-Sq(adj) = 31.07%


                          Individual 95% CIs For Mean Based on Pooled StDev
Level   N    Mean  StDev      +---------+---------+---------+---------
G1      3  122000   1000             (-------------*------------)
G2     19   60516  52802      (----*----)
G3      5  130000      0                  (----------*---------)
G4      3  127667   6807               (------------*------------)
                              +---------+---------+---------+---------
                          40000     80000    120000    160000

Pooled StDev = 43975
```

One-way ANOVA: pH versus Group

```
Source  DF     SS     MS      F      P
Group    3  5.755  1.918  13.60  0.000
Error   26  3.668  0.141
Total   29  9.423

S = 0.3756   R-Sq = 61.08%   R-Sq(adj) = 56.59%


                           Individual 95% CIs For Mean Based on
                           Pooled StDev
Level   N    Mean   StDev  -------+---------+---------+---------+--
G1      3  7.0867  0.0503  (--------*--------)
G2     19  7.0947  0.4509       (---*--)
G3      5  8.1000  0.0000                      (------*------)
G4      3  8.0567  0.0379                   (--------*--------)
                           -------+---------+---------+---------+--
                               7.00      7.50      8.00      8.50

Pooled StDev = 0.3756
```

One-way ANOVA: NT tran versus Group

```
Source  DF      SS     MS     F      P
Group    3   9.273  3.091  4.32  0.013
Error   26  18.591  0.715
Total   29  27.864

S = 0.8456   R-Sq = 33.28%   R-Sq(adj) = 25.58%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev  -----+---------+---------+---------+----
G1      3  -0.4796  0.0944  (---------*---------)
G2     19  -0.3436  1.0105          (---*---)
G3      5   0.6427  0.1206                (------*-------)
G4      3   1.1998  0.2614                   (---------*---------)
                            -----+---------+---------+---------+----
                              -1.0       0.0       1.0       2.0

Pooled StDev = 0.8456
```

One-way ANOVA: PT tran versus Group

```
Source  DF      SS     MS      F      P
Group    3  18.757  6.252  13.74  0.000
Error   26  11.834  0.455
Total   29  30.591

S = 0.6747   R-Sq = 61.31%   R-Sq(adj) = 56.85%


                            Individual 95% CIs For Mean Based on Pooled
                            StDev
Level   N     Mean   StDev   -+---------+---------+---------+--------
G1      3   1.5029  0.1371                        (---------*---------)
G2     19  -0.5640  0.7748   (---*---)
G3      5   1.0577  0.1253                    (-------*-------)
G4      3   0.3383  0.6811        (---------*---------)
                             -+---------+---------+---------+--------
                            -0.80      0.00      0.80      1.60

Pooled StDev = 0.6747
```

One-way ANOVA: Na tran versus Group

```
Source  DF      SS     MS      F      P
Group    3  18.738  6.246  13.53  0.000
Error   26  12.005  0.462
Total   29  30.743

S = 0.6795   R-Sq = 60.95%   R-Sq(adj) = 56.44%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev  --------+---------+---------+---------+-
G1      3   1.8725  0.0283                     (-------*-------)
G2     19  -0.5040  0.8154  (--*--)
G3      5   0.6238  0.0138          (-----*-----)
G4      3   0.7073  0.1339         (-------*-------)
                            --------+---------+---------+---------+-
                                  0.0       1.0       2.0       3.0

Pooled StDev = 0.6795
```

One-way ANOVA: K tran versus Group

```
Source  DF     SS    MS     F      P
Group    3  18.20  6.07  5.69  0.004
Error   26  27.71  1.07
Total   29  45.91

S = 1.032   R-Sq = 39.64%   R-Sq(adj) = 32.67%


                          Individual 95% CIs For Mean Based on
                          Pooled StDev
Level   N    Mean  StDev   --+---------+---------+---------+-------
G1      3  -0.578  0.039   (-------*-------)
G2     19  -0.358  1.198         (---*--)
G3      5   0.889  0.049               (-----*-----)
G4      3   1.906  0.971                    (-------*-------)
                           --+---------+---------+---------+-------
                          -1.5       0.0       1.5       3.0

Pooled StDev = 1.032
```

One-way ANOVA: Mg tran versus Group

Source	DF	SS	MS	F	P
Group	3	13.845	4.615	9.23	0.000
Error	26	12.994	0.500		
Total	29	26.839			

S = 0.7069   R-Sq = 51.59%   R-Sq(adj) = 46.00%

```
                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev   --+---------+---------+---------+-------
G1      3   0.3700  0.0655        (-----------*-----------)
G2     19  -0.5343  0.8306   (---*----)
G3      5   1.1423  0.1713                      (--------*---------)
G4      3   0.7149  0.4749             (-----------*-----------)
                             --+---------+---------+---------+-------
                            -0.70      0.00      0.70      1.40
```

Pooled StDev = 0.7069

One-way ANOVA: Ca versus Group

Source	DF	SS	MS	F	P
Group	3	948840	316280	1.75	0.181
Error	26	4688928	180343		
Total	29	5637768			

S = 424.7   R-Sq = 16.83%   R-Sq(adj) = 7.23%

```
                          Individual 95% CIs For Mean Based on
                          Pooled StDev
Level   N    Mean  StDev  --------+---------+---------+---------+-
G1      3  1370.0  125.3                 (---------*---------)
G2     19   887.2  499.1              (---*---)
G3      5   916.0   20.7           (------*-------)
G4      3   591.7  293.2  (---------*---------)
                          --------+---------+---------+---------+-
                                500      1000      1500      2000
```

Pooled StDev = 424.7

One-way ANOVA: Cl tran versus Group

Source	DF	SS	MS	F	P
Group	3	24.717	8.239	13.43	0.000
Error	26	15.955	0.614		
Total	29	40.672			

S = 0.7834   R-Sq = 60.77%   R-Sq(adj) = 56.25%

```
                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev  --------+---------+---------+---------+-
G1      3   2.1818  0.0679                    (-------*-------)
G2     19  -0.5636  0.9404  (--*--)
G3      5   0.7085  0.0419          (-----*-----)
G4      3   0.8173  0.1017         (-------*-------)
                            --------+---------+---------+---------+-
                                  0.0       1.2       2.4       3.6
```

Pooled StDev = 0.7834

One-way ANOVA: SO4 tran versus Group

```
Source  DF      SS     MS     F      P
Group    3   7.973  2.658  3.19  0.040
Error   26  21.640  0.832
Total   29  29.613

S = 0.9123   R-Sq = 26.92%   R-Sq(adj) = 18.49%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev  ---------+---------+---------+---------+
G1      3   0.5586  0.1272    (-------------*-------------)
G2     19  -0.2954  1.0545  (----*-----)
G3      5   0.2761  0.0782    (---------*----------)
G4      3   1.2985  0.8855              (------------*-------------)
                            ---------+---------+---------+---------+
                                  0.00      0.80      1.60      2.40

Pooled StDev = 0.9123
```

One-way ANOVA: HCO3 tran versus Group

```
Source  DF      SS     MS      F      P
Group    3  17.154  5.718  13.94  0.000
Error   26  10.668  0.410
Total   29  27.822

S = 0.6406   R-Sq = 61.66%   R-Sq(adj) = 57.23%


                            Individual 95% CIs For Mean Based on
                            Pooled StDev
Level   N     Mean   StDev  -----+---------+---------+---------+----
G1      3  -0.7868  0.1031  (------*-------)
G2     19  -0.2691  0.7561           (--*--)
G3      5   1.4532  0.0712                        (-----*----)
G4      3   1.1085  0.4105                  (-------*-------)
                            -----+---------+---------+---------+----
                              -1.0       0.0       1.0       2.0

Pooled StDev = 0.6406
```