Environmental Assessment of the Effects of Leachate Irrigation and Seepage from the Paokahu Landfill, Gisborne, New Zealand

A thesis
submitted in partial fulfillment
of the requirement for the degree
of
Master of Science in Earth and Ocean Sciences
at
The University of Waikato
by

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2010

Abstract

Paokahu landfill operated near Gisborne from 1977 to 2002. About 1 million cubic metres of mixed domestic and industrial (predominantly food processing) wastes were disposed of at the site. The Paokahu landfill is located on the Poverty Bay Flats on low lying flat ground, which was originally the base of a tidal lagoon. A 300m wide band of costal dunes separates the landfill from the Poverty Bay coast. The landfill covers an area of approximately 20 Ha and is unlined but fully capped. Leachate is collected in a cut-off drain which surrounds three quarters of the landfill. Disposal of the leachate is by spray irrigation onto the landfill cap with 13,000 – 15,000 m^3 of leachate irrigated annually. The site is currently used for grazing sheep.

The overall aim of this study was to improve our understanding of the effect that Paokahu landfill is having on the environment and to determine if the current management practices are sustainable. Specific objectives were to collate and review all the groundwater and leachate monitoring data held by Gisborne District Council to determine if the landfill was affecting the local groundwater and to investigate the effect of leachate irrigation on the landfill cap's soil and vegetation.

The groundwater monitoring data showed the local groundwater was generally affected by salt water intrusions giving the groundwater high anion and cation concentrations. There was no evidence of a leachate plume originating from the landfill and no conclusive evidence of leachate contamination in any of the groundwater monitoring bores.

The leachate had a high electrical conductivity (mean = 9350 μScm^{-1}) and high soluble salts content (mean values, Na = 845 gm^{-3}, K= 496 gm^{-3}, Ca = 240 gm^{-3}, Mg = 127 gm^{-3}, Cl =1346 gm^{-3}). Heavy metals were present in low concentrations and Semi Volatile Organic Compounds were no longer present in the leachate. Cation and anion concentrations were generally higher in the groundwater than in the leachate. The main risk to groundwater quality from leachate contamination was from ammonical nitrogen and nitrate.

The leachate irrigation had caused an increase in soil cation concentrations particularly Na in the landfill cap. However, the soil Exchangeable Sodium Percentage of 1.5 - 2.9, leachate Sodium Adsorption Ratio (6.5) and Electrical Conductivity of (4317 $\mu S\ cm^{-1}$) and silty soil material of the landfill cap indicated that soil swelling and dispersion was unlikely to occur in irrigated areas of the landfill cap. Dispersion index testing supported this conclusion with no significant increase in aggregate dispersion under the irrigated areas of the landfill compared to the non-irrigated areas. Soil metal concentrations were low and there was no significant difference in soil heavy metal concentrations between the irrigated and non-irrigated areas of the landfill cap. Leachate Mn concentrations (mean = 1.39 $g\ m^{-3}$) were high enough to be potentially toxic to plants, but the soil pH (7.4) and Ca

(46.7 me/100g) content mean that Mn toxicity is unlikely to occur. Leachate irrigation appears sustainable under current conditions.

Acknowledgements

I would like to extend my sincere thanks to the following people and organisations for the help and support they gave me during the course of this thesis.

My first thanks go to my supervisor, Megan Balks, for her support, guidance, assistance and most of all her belief in me. Her enthusiasm has been of invaluable help to keep me on track to complete this thesis.

Thank you to my employer, Gisborne District Council, without whose support and financial assistance I could not have undertaken this project. In particular, I would like to thank: Dave Hadfield for his support and allowing me time to work on this thesis; Dwayne Pomana for all his help; Ian Hughes for assisting me with my field work; Dave Hemmington and Wayne Edwards for drafting all the plans and maps.

Thank you to Jacinta Parenzee and Annette Rodgers of the University of Waikato's soils laboratory who performed the soil particle size analysis.

Last but not least thanks to my wife Janet who spent hours proof reading my work and for all the support she has given me without which I would have been unable to complete this thesis.

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Chapter 1:

Introduction

1.1 Landfills in New Zealand

Landfills are the conventional method of disposal of solid wastes in New Zealand. Until the 1980's most New Zealand landfills were no more than tip/dump sites and often poorly sited, designed and managed (Ministry for the Environment 2001). Many older landfills are now closed; the number of operating landfills in New Zealand decreased from 327 in 1995 to 60 in 2006 (Ministry for the Environment 2008) . This reduction has been partly the result of a drive to improve landfills and the development of national environment standards and landfill management guidelines (Ministry for the Environment 2008). The Ministry for the Environment (2000) estimated that there are over 1000 closed landfills in New Zealand.

Once closed, landfills still have the potential to cause significant environmental damage. The natural decomposition processes taking place while the landfill is operational, continue on long after closure. Continued production of leachate and landfill gas can cause problems such as:

- Leachate escaping from the landfill contaminating groundwater and surface water;
- Landfill gas is flammable, potentially explosive and can cause odour problems and damage vegetation.

Other concerns regarding closed landfills have been identified by the Ministry for the Environment (2001) and include:

- emission of green house gases;
- subsidence;
- land stability, particularly with respect to erosion;
- the unknown location or nature of some of the landfill constituents; and

- health and safety aspects relating to subsequent land use.

Therefore, continued site management is needed to prevent problems occurring in the future.

Management of closed landfills is generally site specific but typical aftercare operations include:

- leachate collection and treatment or disposal;
- landfill gas control;
- monitoring of site integrity;
- repairs to the final cover system;
- maintenance and control of vegetation;
- stormwater and sediment control and
- monitoring of groundwater, surface water and landfill gas (Centre for Advanced Engineering, 2000).

The aftercare should continue until the landfill no longer has the potential to adversely affect the environment, which can be in excess of 60 years after final closure.

1.2 The Paokahu Landfill

One New Zealand example of a closed landfill is Paokahu landfill near Gisborne. The site is privately owned but the aftercare of the site is the responsibility of Gisborne District Council. The Paokahu landfill served as Gisborne District's disposal site for solid waste from 1977 to 2002 with approximately 1 million cubic metres of mixed domestic, agricultural and industrial wastes disposed of at the site. The landfill covers an area of approximately 20 hectares and has been filled to 9m above original ground level.

The site is located 5 km east of Gisborne City, immediately behind coastal dunes on a reclaimed tidal lagoon (Figure 1.1). The majority of the site is unlined, which was typical of landfills from that era. Prior to 1998 only 4% of New Zealand landfills had an engineered liner (Ministry for the Environment, 2003).

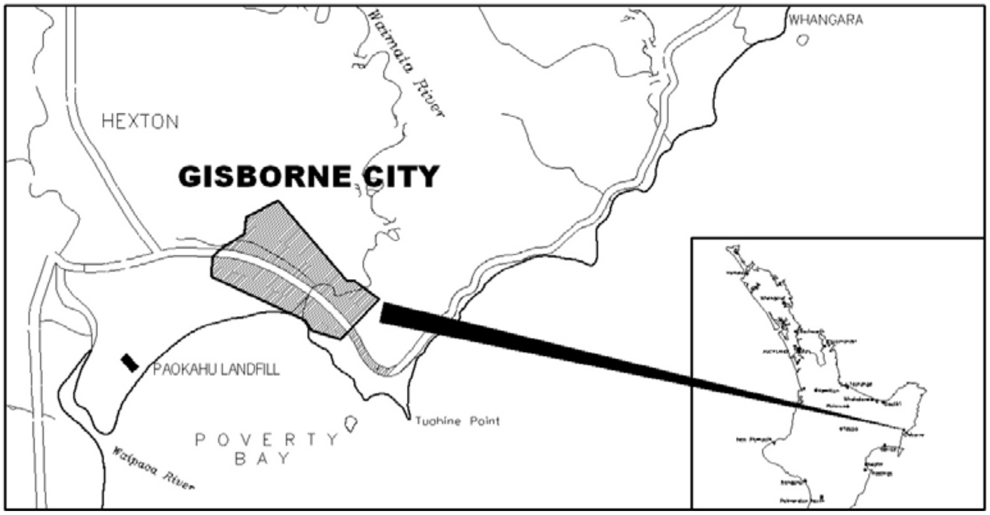


Figure 1.1: Location of Paokahu Landfill.

There is a leachate collection system which covers three quarters of the site. No physical treatment of the leachate takes place; instead it is irrigated back onto the landfill through a series of sprinklers (Figure 1.2). The site is fully capped and is currently used for grazing sheep.

Figure 1.2: Leachate irrigation at Paokahu Landfill.

Part of the aftercare of the site has involved the monitoring of nearby groundwater and surface waters and leachate concentrations. Monitoring has been for general chemical parameters, heavy metals, and volatile and semi volatile organic compounds. As a result Gisborne District Council currently holds data taken from the site and surrounding locations. Much of the collected data is disjointed and needs to be collated into a useable format. No detailed analysis of the monitoring data has been undertaken.

1.3 Objectives

The overall aim of this study is to improve our understanding of the effect that Paokahu landfill is having on the environment and to determine if the current management practices are sustainable. More specifically the objectives of the research contained in this thesis are to:

1. Collate and review the leachate and groundwater monitoring data from Paokahu landfill to determine if the landfill is having any adverse effects on the environment, particularly the local groundwater.

2. Investigate the effects of leachate irrigation on the landfill cap's soil and vegetation.

Chapter 2. Literature Review

2.1 Introduction

This chapter reviews some of the current knowledge on landfill leachate, especially leachate generation, composition and methods of disposal. The effects of leachate irrigation on soil and plants are examined, with emphasis on the effects of high salinity levels on soil structural stability and plant growth.

2.2 Leachate Generation and Composition

2.2.1 Introduction

Landfill leachate is the liquid waste that forms within a landfill. Leachate forms when the soluble components present in the waste, dissolve and leach out as water moves through the landfill. The soluble components can be leached directly from the refuse or form as a result of the physical, chemical and biological processes that take place during decomposition. Rainfall is the main source of water in leachate but surface water, groundwater and the moisture in the waste can also be sources.

Landfill leachate is a water based solution containing dissolved organic matter, inorganic components, heavy metals and artificial organic compounds (Kjeldsen *et al.*, 2002). The exact composition of a leachate is a function of the type, age, and stabilisation of the landfill waste, the prevailing physiochemical conditions, microbiology and water content of the landfill.

2.2.2 Stabilisation Processes in Landfills

Leachate composition is primarily a function of the age and degree of stabilisation of the waste within a landfill (Reinhart and Grosh, 1998). Once buried, a complex series of chemical and biological reactions occur within a landfill as the refuse decomposes. Chemical and biological reactions stabilise the waste and proceed in a distinct series of phases. The idea of refuse decomposing in phases was first described by Farquhar and Rovers (1973) and has been the subject of several other studies (Barlaz *et al.,* 1989; Bozkurt *et al.*, 1999; Kjeldsen *et al.,* 2002). In the literature the exact number of phases reported range from three (Centre for Advanced Engineering, 2000) to seven (Kjeldsen *et al.,* 2002). However, it is commonly accepted that in the short term four distinct phases of decomposition take place (Barlaz *et al.*, 1989; Bozkurt *et al.*, 1999; Kjeldsen *et al.,* 2002; Lisk, 1991). The rate of production and characteristics of the leachate produced vary from one phase to the next. Descriptions of the first four phases of refuse decomposition are given below:

Phase 1: Aerobic Phase

The aerobic phase begins when the refuse is first landfilled and only lasts a couple of days as oxygen is not replenished once the refuse is covered (Kjeldsen *et al.*, 2002). Aerobic organisms break down the degradable material such as sugars present in the fresh refuse to produce carbon dioxide, organic compounds, heat and water (Barlaz *et al.*, 1989; Centre for Advanced Engineering, 2000). Small amounts of leachate are produced during this phase as the refuse is not typically at a moisture content at or above field capacity. Most of the leachate produced is the result of moisture being released during compaction of the refuse (Kjeldsen *et al.*, 2002). Phase 1 is characterised by a neutral pH and an increase in the Biochemical Oxygen Demand (BOD) and Chemical Oxygen Demand (COD) (Taulis, 2005).

Phase 2: Acid Phase

After burial of the refuse, oxygen is not replenished and anaerobic conditions dominate. Hydrolitic, fermentative and acetogenic bacteria take over the decomposition process in the absence of oxygen. During this phase complex organic material is degraded by hydrolysis to smaller organic molecules, such as carboxylic acids, alcohols and carbon dioxide (Bozkurt *et al.,* 2000). The formation of organic acid results in a drop in pH (Kjeldsen *et al.,* 2002). The acidic leachate is chemically aggressive and will increase the solubility of many compounds including metals and other inorganic ions (Kjeldsen *et al.,* 2002; Taulis, 2005). Carbon dioxide concentrations reach their maximum values during the acid phase and values of over 90% CO_2 have been observed in the landfill gas (Barlaz *et al.*, 1989; Centre for Advanced Engineering, 2000). The leachate from the acid phase is characterised by a pH between 5 and 6, high ammonia and BOD concentrations and a high BOD/COD ratio (Robinson & Barr, 1989).

Phase 3: Initial Methanogenic Phase

The onset of the third phase begins when measurable quantities of methane are produced (Kjeldsen *et al.,* 2002). Acetogenic bacteria convert the carboxylic acids produced in the acid phase to acetate, hydrogen and carbon dioxide which raises the pH (Barlaz *et al.*, 1989; Kjeldsen *et al.,* 2002). The increase in pH allows the growth of methanogenic bacteria whose growth is limited in the acidic conditions of the acidic phase (Kjeldsen *et al*., 2002). The degradation products act as substrate for the methanogenic bacteria which convert it to methane and carbon dioxide (Bozkurt *et al.,* 2000). It is during the initial methanogenic phase that the decomposition of cellulose and hemi cellulose begins (Barlaz *et al.*, 1989; Kjeldsen *et al.,* 2002).

Phase 4: Stable Methanogenic Phase

In the stable methanogenic phase methane production reaches its maximum and decreases thereafter. Carboxylic acids are consumed at the same rate at which they are produced and methane production is dependent on the rate of cellulose and hemicellulose hydrolysis (Kjeldsen *et al.,* 2002; Taulis, 2005). The rate of pH increase declines and the BOD and COD concentrations decrease.

2.2.2.1 Long term Stabilisation Effects

In general, landfills have only been controlled and monitored for 30 - 40 years and the four phases referred to above are based on observations of landfill processes (Kjeldsen *et al.,* 2002). Recent studies have proposed a subsequent phase in which the landfill turns aerobic again over time and has been referred to as the Humic phase (Bozkurt, *et al.,* 1999). The humic phase is based on theory and is somewhat speculative as there is no field data available to document the onset of aerobic conditions. Most well monitored landfills are less than 40 years old and have not progressed past the methanogenic phase (Kjeldsen *et al* 2002).

The humic phase has been described by Bozkurt *et al.* (2000) as occurring after all methane production has stopped. The time needed to reach the humic phase is unknown but may be in the region of 100 years and could last up to many thousands of years. As the methanogenic phase progresses the remaining substrate becomes more resistant to degradation and microbial activity slows down. The organic matter which is still present has been converted to much more stable compounds, mainly humic substances. During the first four phases any oxygen entering the landfill will readily react with simple organic compounds. During the humic phase the readily degradable organic matter is either depleted or the reactions are too slow to consume all the oxygen entering the landfill. When this occurs the landfill may become aerobic and acidic due to the carbon dioxide formed (Bozkurt *et al.*, 2000).

Kjeldsen *et al.* (2002) examined the long term stabilization process in landfills and have suggested an additional two intermediate phases occur before the humic phase, which is outlined below.

Phase 5: Methane Oxidation

This phase refers to very localised oxygen intrusion in the upper layer of the landfill or around any gas vents used for methane extraction and is relatively minor. The oxygen entering the landfill will promote the oxidation of methane still being produced (Kjeldsen *et al.* 2002).

Phase 6: Air Intrusion

As methane production decreases, air will diffuse through the landfill cover and enter the mass waste. Eventually all of the methane produced will be oxidised in the mass waste. Methane emissions will cease and carbon dioxide concentrations will increase as a result of the methane oxidation reaction. The nitrogen content will increase due to air intrusion; oxygen will be almost undetectable due to rapid consumption (Kjeldsen *et al.* 2002).

Phase 7: Carbon Dioxide

Methane production will be insignificant during the final phase and the oxygen entering the landfill will be consumed by the oxidation of any residual methane, organic matter and reduced inorganic species such as sulphur, nitrogen and iron containing compounds. Additional refuse oxidation will also occur as some of the organic material is more degradable under aerobic conditions compared to anaerobic conditions. The pH may decrease due to the presence of carbon dioxide and the oxidation of reduced sulphur, nitrogen and iron compounds. As the pH decreases, metal carbonate precipitates may dissolve and the carbonates released are predicted to buffer the pH, although this has not been demonstrated experimentally (Kjeldsen *et al*. 2002). The carbon dioxide phase is the equivalent to the humic phase described by Bozkert *et al*., (2000).

2.2.3 Leachate Composition

Household waste is reasonably consistent in composition so landfills that accept predominantly municipal solid waste and operate under anaerobic conditions tend to produce leachates with similar constituents, although concentrations vary between landfills (Centre for Advanced Engineering, 2000). The major components in landfill leachate can be divided into four groups:

1. Dissolved organic matter such as volatile fatty acids, and humic and fulvic compounds. These are usually measured as total organic carbon (TOC) or chemical oxygen demand.

2. Inorganic macro components such as calcium, magnesium, sodium, potassium, ammonium, sulphate, chloride, iron and hydrogen carbonate.

3. Heavy metals like cadmium, chromium, copper, lead nickel and zinc.

4. Xenobiotic organic compounds which can include aromatic hydrocarbons, phenols and pesticides (Christensen *et al.,* 2001; Kjeldsen *et al.,* 2002).

2.2.3.1 Dissolved Organic Matter

Dissolved organic matter is a bulk parameter covering a wide range of organic compounds including volatile fatty acids, and refractory products such as fulvic and humic like compounds. The dissolved organic matter content of landfill leachate is usually expressed as biological oxygen demand (BOD), chemical oxygen demand (COD) or total organic carbon (TOC) (Kjeldsen *et al*., 2002).

BOD and COD concentrations are highest during the acid phase and decrease over time (Table 2.1). The easily oxidised compounds, mainly volatile fatty acids, produced in the acid phase are converted to methane and carbon dioxide, leaving the more stable fulvic and humic compounds (Bilgili *et. al*, 2006; Lisk, 1991).

Table 2.1: BOD and COD Leachate Concentrations.

Parameter	Acid Phase		Methanogenic Phase	
	Mean	Range	Mean	Range
BOD_5 (g m^{-3})	13000	4000-40000	180	20-550
COD (g m^{-3})	22000	6000-60000	3000	500-4500
BOD_5/COD ratio	0.58		0.06	

(after Kjeldsen *et al*., 2002)

The BOD_5/COD ratio can be used as a measure of leachate stability (Kjeldsen & Christophersen, 2001) and the degree of anaerobic degradation of the organic material (Ehrig, 1983). During the acid phase the BOD_5/COD ratio can be in the order of 0.4 to 0.8 (Ehrig, 1983) but can drop to below 0.1 during the methanogenic phase (Kjeldsen *et al*., 2002). A high BOD_5/COD ratio (>0.4) indicates the presence of high concentrations of biodegradable material, typically volatile fatty acids and that the landfill is likely to be in the acid phase. A low BOD_5/COD ratio (<0.1) suggests a leachate with low concentrations of volatile fatty acids and high concentrations of relatively biologically inert material such as the fulvic and humic compounds which occur during the methanogenic phase (Kjeldsen *et al*.,2002; Kulikowska & Klimiuk, 2008).

2.2.3.2 Inorganic Macro-components

Inorganic macro-components are the inorganic constituents present in the leachate (Christensen *et al.*, 2001). The common cations and anions found in leachate include sodium, potassium, sulphate, chloride, and ammonia. The concentration of the inorganic macro-components can vary over time depending on the degree of stabilisation in the landfill.

In the methanogenic phase calcium, magnesium, iron and manganese concentrations decrease. The higher pH enhances sorption and precipitation and the lower organic matter content allows the formation of complexes with the cations. Sulphate concentrations are also lower due to the microbial reduction of sulphate to sulphide (Kjeldsen *et al* 2002).

Chloride, sodium, and potassium concentrations are not influenced by the landfill phase as the effects of sorption complexation and precipitation are minor for these ions (Centre for Advanced Engineering, 2000; Kjeldsen *et al.*, 2002).

Nitrogen

Most of the nitrogen present in leachate is in the form of ammonia which is produced by the degradation of proteins and amino acids. Ammonia concentrations are typically between 500 to 2000 g m^{-3} (Kjeldsen *et al.*, 2002) but concentrations in excess of 10 000 g m^{-3} have been recorded (Tatsi & Zouboulis, 2002).

In the literature it has been reported that ammonia concentrations tend not to decrease over time. In a study of 50 German landfills Kruempelbeck and Ehrig (1999) found that there was no significant change in ammonia concentrations over a 30 year period. Ehrig (1988) found no significant change in ammonia concentrations between the acidic and methanogenic phases. Hartmann and Hoffmann (1990) reported that after a period of 3 to 8 years ammonia concentrations reached a mean value of between 500 and 1500 g/m^3 and would remain at this level for at least 50 years.

Ammonia concentrations do not decline over time as there is no mechanism for its degradation under anaerobic conditions; therefore any ammonia loss is a result of leaching out of the landfill. As a result ammonia is considered by many researchers to be the most significant pollutant present in leachate (Bilgili *et al.* 2006; Kulikowska & Klimiuk, 2008).

2.2.3.3 Heavy Metals

Heavy metals are defined as metals with a density greater than 5 g cm^{-3} (McLaren & Cameron, 1990). Heavy metals present in leachate include arsenic (As), cadmium (Cd), chromium (Cr), cobalt (Co), copper (Cu), lead (Pb), mercury (Hg)), nickel (Ni), and zinc (Zn) (Kjeldsen *et al.*, 2002; Taulis, 2005). Heavy metals can be detrimental to the environment and can cause metal contamination in waterways and land. For example cadmium, chromium and some nickel compounds are highly toxic. Arsenic is poisonous at high concentrations and can cause skin cancer even at low concentrations (Taulis, 2005). Some heavy metals such as copper and zinc are essential for growth but can become toxic at high concentrations (McLaren & Cameron, 1990).

Although heavy metal can be toxic, leachate heavy metals generally do not pose a groundwater pollution problem as average metal concentrations are typically low and only a small proportion of the metals that are disposed of in a landfill are leached out (Kjeldsen *et al*., 2002; Taulis, 2005). Mass balance studies have shown that less than 0.002% of heavy metals are leached from a landfill over the first 30 years of operation (Kjeldsen *et al*., 2002). Qu *et al*. (2008) reported that total heavy metals concentration was less than 1% of that deposited in a Chinese Bioreactor landfill during a 20 month study period.

Heavy metal concentrations are typically highest during the acid phase when the low pH leachate is chemically aggressive and can increase the solubility of metals (Kjeldsen *et al*., 2002). As the pH increases and the landfill moves to the methanogenic phase, leachate heavy metal concentrations decrease. Studies conducted on German landfills (Ehrig 1983) and in the USA by (Krug and Ham, 1991) showed that magnesium, iron, zinc and manganese concentrations are higher in the acid phase (Taulis, 2005). Qu *et al*. (2008) examined leachate from a full scale bioreactor landfill in China over a 20 month period and reported that cadmium, chromium, copper, nickel, lead, and zinc concentrations were initially high but after 5 months the methanogenic stage had been reached and the heavy metal concentration had dropped below the Chinese national standards.

The relatively low leachate metal concentrations during the methanogenic phase are commonly attributed to adsorption and precipitation processes, binding and immobilising the metals within the landfill. Landfills will typically contain significant amounts of soil and organic matter, which absorb the metals, reducing their solubility and mobility (Kjeldsen *et al*., 2002; McLaren and Cameron, 1990; Taulis, 2005). Absorption of metals by soil and organic matter occurs more readily at neutral to high pH values, so occurs during a landfill's methanogenic phase (Taulis, 2005).

Precipitation is the other mechanism which immobilises metals. Sulphides and carbonates which are common in leachate especially during the methanogenic phase readily form precipitates with cadmium, copper, lead, nickel and zinc (Kjeldsen *et al*., 2002; Taulis, 2005). Sulphite precipitation is the dominant metal attenuation process as the solubility of metal sulphides are higher than of metal carbonates and other precipitating agents (Reinhart & Grosh,1998). Erses & Onay (2003) experimenting with a landfill simulator, reported that 90% of all heavy metals were precipitated out of the reactor due to sulphide precipitation.

Leachate does not contain enough sulphide ions to bind all the heavy metals present (Kjeldsen *et al*., 2002; Martenesson *et al.,* 1999). Aulin and Neretnieks (1995) reported that in the Högbytorp landfill in Sweden the sulphide present was only sufficient to precipitate 5% of the metals present. Chromium does not precipitate out with sulphides and carbonates, but forms insoluble precipitates with hydroxides at neutral to high pH values (Taulis, 2005).

Heavy metals are not always immobilised within the landfill; there are also processes that are capable of increasing the concentrations of metals in the leachate. Complexation of metals to organic ligands, and adsorption onto colloids can increase the concentrations of heavy metals in leachate (Kjeldsen *et al*., 2002; Taulis, 2005).

Long term landfill processes may also lead to heavy metals being mobilised from the landfill into the leachate. If a landfill progresses to the humic phase, air can penetrate in to the landfill, changing conditions from anaerobic to aerobic. Under aerobic conditions oxidation of sulphur, nitrogen, iron containing compounds and residual organic matter can occur lowering the pH and changing the redox potential, producing a more chemically aggressive leachate which can result in the mobilisation of heavy metals (Taulis, 2005). Metal sulphides will oxidise to metal sulphates which are considerably more soluble, for example the pK_{so} of PbS is 27.6 compared to 7.73 for $PbSO_4$ (Kjeldsen *et al*., 2002).

2.2.3.4 Xenobiotic Organic Compounds

Xenobiotic Organic Compounds (XOCs) are substances derived from anthropogenic sources (Baun *et al.* 2004) and include aromatic hydrocarbons, phenols and chlorinated alipatics (Slack *et al.* 2007) with the most frequently found XOCs being monoaromatic hydrocarbons and halogenated hydrocarbons (Kjeldsen *et al*., 2002). XOCs are commonly associated with industrial or hazardous waste but a large number occur in municipal and domestic waste. Paint, garden chemicals, household cleaning agents, motor vehicle products, batteries, waste electrical and electronic equipment are all sources of XOCs in municipal solid waste (Slack *et al.* 2007).

The number and concentration of XOCs present in leachate can be quite variable between landfills. Baun *et al.* (2004) monitored 10 different Danish landfills and reported the presence of 55 different XOCs plus 10 degradation products of XOCs with concentrations ranging from <0.1 μg/l to 2220 μg/l. Paxeus (2000) identified more than 200 individual compounds or groups of compounds in a screening of leachates from three Swedish landfills, with concentrations ranging from less than one µg/l to several hundred µg/l. The compounds common to all three landfills included plasticizers, phosphate esters, substituted phenols, chlorinated phenols and phenolic antioxidants (Paxeus, 2000).

The concentrations of XOCs in landfill leachate should decrease over time, depending on how quickly the compound degrades in the landfill (Christensen *et al*., 2001). Kjeldsen *et al*., 2002 reported that concentrations of XOCs are higher in older municipal landfills compared to younger landfills. This was attributed to lower acceptance rates of XOCs in the newer landfills rather than any landfill ageing process.

2.3 Leachate Treatment

2.3.1 Introduction

Conventional landfill leachate treatments can be classified into three broad groups: biodegradation, physical and chemical methods and leachate transfer (Renou *et al.* 2008). The three groups of treatment processes will be discussed briefly and leachate disposal through irrigation will be examined in more detail.

2.3.2 Biological Treatments

Biological treatments include aerobic treatment in aerated ponds, activated sludge processes, trickling filters and anaerobic digesters. Micro organisms break down the organic compounds present in the leachate to carbon dioxide and sludge under aerobic conditions or to carbon dioxide and methane under anaerobic conditions (Renou *et al*., 2008). The biological treatments are commonly used to remove BOD from the leachate and are most effective when the BOD/COD ratio is high (>0.5); therefore not suited to landfills in the methanogenic phase (Renou *et al*., 2008: Robinson & Barr, 1999).

2.3.3 Physical and Chemical Treatment

Physical and chemical treatments include: flotation, coagulation-flocculation, chemical precipitation, oxidation and air stripping of the leachate. Physical and chemical treatments are designed to remove either colloidal particles or other suspended solids, or to treat specific components such as ammonia.

Physical/chemical treatments are typically used in conjunction with biological treatment. Chemical precipitation and air stripping are common treatments used when the leachate has high ammonia concentrations (Renou *et al*., 2008).

2.3.4 Leachate Transfer

Leachate transfer refers to systems where leachate is transported off site for disposal and includes: (1) disposal of leachate into the municipal sewage system and treatment through a sewage treatment plant, (2) recirculation of leachate back into the landfill and (3) irrigation of leachate onto the land.

Disposal of leachate through the sewage system is commonly used where there is ready access to a sewage system as the operating and maintenance costs are low. However, the presence of non biodegradable organic compounds and heavy metal in the leachate can reduce the efficiency of some treatment plants (Renou *et al.* 2008).

Recirculation of leachate typically involves pumping leachate back into the landfill. The advantage of this system is that it is one of the least expensive treatment options available (Renou *et al.* 2008). The other advantage is the increased moisture content within the landfill speeds up the decomposition of organic waste and improves the quality of the leachate (Reinhart *et al.* 2002).

2.3.5 Leachate Irrigation

Leachate can be disposed of by spray or trickle irrigation onto vegetated land. Leachate irrigation is best suited to areas where: (1) high rainfall leads to the production of large volumes of dilute leachate, (2) there is enough suitable land available for irrigation, and (3) the landfill is too far away from any sewer network to make off site disposal economic (Gray *et al.* 2005). An advantage of leachate irrigation is that leachate treatment/disposal can be accomplished on site, particularly on closed landfills where irrigation can take place over the capped landfill surface. The other advantage of irrigation is that the nutrients present in the leachate can be recycled and used for plant growth.

Treatment of the leachate is achieved through a combination of evaporation, infiltration, microbial degradation, retention in the soil, and plant uptake of nutrients (MacDonald *et al.* 2008). Jones *et al.* (2006) divided the treatment processes into above ground processes and below ground processes (Figure 2.1). The above ground processes include: (1) the uptake of soluble nutrients and metals by plants and their use in growth (e.g. nitrate and zinc) or storage in leaves (e.g. lead); and (2) evaporation which reduces the volume of leachate. Below ground processes include: (1) uptake by plant roots of inorganic nutrients, metals and organic compounds which can be used for growth, transported to the shoots, or stored in the leaves; (2) rizosphere micro organisms reduce the BOD load, detoxify organic pollutants and render some metals non-toxic (e.g. copper); (3) sorption, complexation and precipitation of metals onto soil organic matter and mineral particles; (4) evaportranspiration reduces leachate volumes; and (5) improved root growth improves soil structure, which enhances infiltration and reduces the risk of surface run-off.

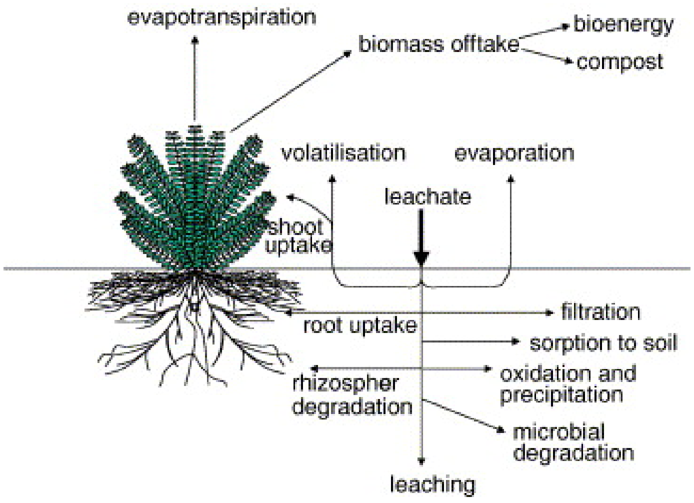


Figure 2.1: Above and below ground soil-plant leachate treatment processes (Jones *et al.*, 2006).

Improvements in leachate quality (Table 2.2) were reported by Harrington & Maris (1986) in a land based treatment scheme where leachate was irrigated onto pasture at a rate of 50 m^3 ha^{-1} d^{-1}. Similar results (Table 2.3) were reported by

Menser *et al.* (1979) where leachate quality was shown to improve when irrigated onto a forest covered soil over a five year period.

Table 2.2: Leachate quality before and after irrigation on pasture.

Constituent	Leachate (g m^{-3}) (Pre irrigation)	Leachate (g m^{-3}) (Post irrigation)
Suspended Solids	70	25
NH_4^+	15	5
COD	400	60
BOD	300	15
Fe	12	2

(Adopted from Harrington & Maris,1986)

Table 2.3: Leachate quality before and after irrigation on pasture.

Constituent	Leachate (g m^{-3}) (Pre treatment)	Leachate (g m^{-3}) (Post treatment)
pH	7.25	6.84
EC	1.9	0.2
COD	1878	239
Na	194	32
K	80	7
Zn	0.89	0.09
Fe	77	4

(Adopted from Menser *et al.*, 1979)

2.3.5.1 Effects on Plant Growth

Leachate is rich in nutrients, including nitrogen needed for plant growth, and leachate irrigation has been shown to improve plant growth when managed correctly (Bowman *et al.* 2002; Maurice *et al.* 1999; MacDonald *et al.* 2008; Revel *et al.* 1999; Shrive *et al.* 1994). Revel *et al.* (1999) irrigated leachate with high concentrations of NH_4^+ (1520 g m^{-3}), Na^+ (1920 g m^{-3}), K^+ (2200 g m^{-3}) and Cl^- (2130 g m^{-3}) onto pots containing ryegrass (*Lolium* sp.). Increased grass growth was shown when the pots were irrigated with solutions containing up to 400 g m^{-3} of leachate. Above 400 g m^{-3} the leachate had a detrimental effect on plant growth which was attributed to sodium toxicity.

Shrive *et al.* (1994) found that irrigation of a high ionic strength leachate significantly increased stem growth in hybrid poplar samplings (*Populas* ssp.

Nigra x maximowiczii). They also determined that direct exposure of the leaves to the potentially phytotoxic compounds (volatile organics compounds and inorganic substances including metals) present in the leachate did not induce phytotoxic reactions in the plants.

2.3.5.2 Effects of Leachate Salts

When discussing the influence of the soluble salts on leachate irrigation the terms salinity and sodicity are commonly used. Salinity refers to the presence of soluble salts in the irrigated water or soil, while sodicity refers to the proportion of available sodium ions relative to the available calcium, magnesium, potassium and aluminium ions present in the soil or irrigated water.

Salinity and sodicity are common problems in areas irrigated for agricultural use, especially if rainfall is low and evaporation high (Rietz & Haynes, 2003). Soluble salts present in the irrigation water can accumulate in the soil when the rainfall is insufficient to leach the salts from the soil profile (Blaylock, 1994). Increased soil sodicity and salinity can cause a reduction in soil hydraulic conductivity and reduced plant productivity.

The sodium adsorption ratio (SAR) is used to quantify the salinity of the irrigating water and the exchangeable sodium percentage (ESP) is used to quantify the soil salinity. The SAR describes the level of sodium relative to other cations present in the irrigation water and is defined by:

$$SAR = \left[Na^{+}\right] / \left(\left[Ca^{2+}\right] + \left[Mg^{2+}\right]\right)^{\frac{1}{2}}$$

where $[Na^{+}]$, $[Ca^{2+}]$ and $[Mg^{2+}]$ are the concentrations of sodium, calcium and magnesium in mmol l^{-1} (Balks *et al.*, 1998). The ESP is the percentage of the cation exchange capacity occupied by sodium (Balks *et al.*, 1998) and is calculated as:

$$ESP = \frac{100 \times echangeable\ \ Na^{+}}{CEC}$$

where Na^+, is the concentrations of sodium in $cmol_c$ kg-^{1} and CEC is the cation exchange capacity in $cmol_c$ kg^{-1} (Halliwell *et al.*, 2001).

The detrimental effects of high sodium concentrations on soils are exhibited through swelling and dispersion of clay minerals (Halliwell *et al.* 2001). Clays are generally negatively charged plate shape particles which are often found in parallel alignments called domains (McLaren & Cameron, 1990; Quirk 2001). When the ESP is below 15-25% sodium ions are preferentially adsorbed on to the outer layer of the clay domains and swelling doesn't occur. As the ESP increases above 15-25%, sodium replaces calcium in between the individual clay platelets of the clay domains causing microscopic swelling between the clay platelets (Halliwell *et al.* 2001), making swelling of clay minerals the dominant mechanism for clay aggregate degradation when the ESP is high.

Clay dispersion is influenced by the electrolyte concentration (E_c) of the soil. As the E_c reaches a critical concentration known as the coagulation value, clay domains form (Halliwell *et al.* 2001). If the E_c drops below the critical coagulation value clay domains can disperse. At high ESPs clay dispersion can occur spontaneously, whereas at lower ESP levels energy inputs are required for dispersion to occur. As the ESP of a soil increases there has to be a corresponding increase in E_c in order to keep the E_c above the critical coagulation value and prevent the soils from dispersing. A soil with a very low ESP can still be affected by clay dispersion provided the E_c is below the critical coagulation value (Sumner, 1993). Therefore dispersion of clay particles can occur throughout the range of ESPs and will be the dominant mechanism for clay aggregate degradation at low ESP values.

The value at which the ESP will become hazardous to a soil will vary between soils. Soils with identical sodicities and ionic strengths will not always exhibit similar clay dispersion characteristics; similarly a soil with a high ESP may not necessarily disperse when the soil water has a low E_c (Balks *et al.*, 1998; Halliwell *et al.*, 2001). Factors which account for the differences include soil texture and mineralogy, bulk density, pH, aggregate binding agents such as iron and aluminium oxides and organic matter, and the mechanical stresses applied to the soil (Balks *et al*., 1998). For example, Cameron *et al.* (2003) found that the critical ESP value required for clay dispersion increased from 3 to 13 after 4 to 10 years of irrigation with dairy factory effluent containing high concentrations of sodium. The increase in the critical ESP was attributed to the high organic matter content (lactose) in the effluent, increasing the soil wet aggregate stability.

Dispersed clay particles can block soil pores reducing hydraulic conductivity and infiltration rate. In sandy soils clay dispersion generally does not cause any problems and the clay particles can pass directly through the larger soil pores (Halliwell *et al*., 2001). The infiltration rate is more sensitive to changes in ESP and SAR than hydraulic conductivity, as falling water drops transfer energy to the soil surface allowing dispersion to take place at lower ESP levels than further down the soil profile (Sumner, 1993) where the impact of water drops is not felt. The reduction of hydraulic conductivity and infiltration rate at the soil surface is largely irreversible due to the disproportionate breakdown of the clay structure (Halliwell *et al*., 2001).

At an ESP of ≥15-25 swelling of clays can reduce the size of soil pores and therefore reduce the hydraulic conductivity (Halliwell *et al*., 2001). The swelling process can generally be reversed by increasing the E_c of the infiltrating water through the addition of divalent cations (Sumner, 1993).

Dispersion and swelling will not be a problem in wastewater irrigation schemes as long as the E_c of the irrigating water remains high (above the critical coagulation value). However, the application of low E_c water inevitably occurs, either through rainwater or variations in composition of the waste water (Halliwell *et al*., 2001; Sumner, 1993). The reduction in infiltration rate and hydraulic conductivity can be reduced by the addition of chemical ameliorates such as gypsum or lime to the soil or by irrigating wastewater during rain events to increase the E_C of the rainwater above the critical coagulation value (Halliwell *et al*., 2001).

Several studies have shown a relationship between soil ESP and the SAR (Rengasamy *et al*., 1984; Seilsepour *et al*., 2009; United States Salinity Laboratory, 1954) and the SAR can be used to estimate the soil ESP. Several models have been developed to estimate soil ESP using the SAR, however, the models can not be applied from one soil to the next with any consistency as factors such as the E_c, clay mineralogy, soil texture etc all affect the relationship. For example, the United States Salinity Laboratory staff (1954) suggested the soil ESP could be estimated from the SAR of the soil water using the following relationship:

$$ESP = \frac{1.475 \times SAR}{1 + 0.0147 \times SAR}$$

The above relationship holds for soil solutions obtained from saturated paste extracts. In an Australian study, Rengasamy *et al.* (1984) used a 1:5 soil: water extract and developed a different relationship:

$$ESP = 1.95 \times SAR + 1.8$$

Therefore, the relationship between soil ESP and SAR should be determined directly for different soils. In a recent study Seilsepour *et al*., (2009) developed the following model to estimate the ESP of soils in the Varamin region of Iran.

$$ESP = 1.95 + 1.03\ SAR$$

Developing a model can be advantageous when numerous ESP measurements are required from a particular soil type as it eliminates the requirement to perform repeated soil cation analysis tests (Seilsepour *et al*., 2009).

Analysis of the SAR and electrical conductivity (EC) of applied wastewater can also be used to predict if dispersion or swelling of clay is likely to occur. The following table (Table 2.4) can be used to estimate if soil structural problems are likely given a known SAR and EC combination.

A relationship between SAR and EC has been proposed (Figure 2.2) in the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (2000) to estimate structural stability using the SAR and EC of irrigation water.

Table 2.4: Guidelines for the interpretation of water quality for irrigation.

	EC (dSm^{-1})		
SAR	No problem expected	Slight to moderate problem expected	Severe problem expected
0-3	>0.9	0.2 – 0.9	<0.2
3-6	>1.3	0.25 – 1.3	<0.25
6-12	>2.0	0.35 -2.0	<0.35
12-20	>3.1	0.9 – 3.1	<0.9
20+	>5.6	1.8 – 5.6	<1.8

(after Ayers & Tanji (1981) cited in Halliwell *et al.*, 2001)

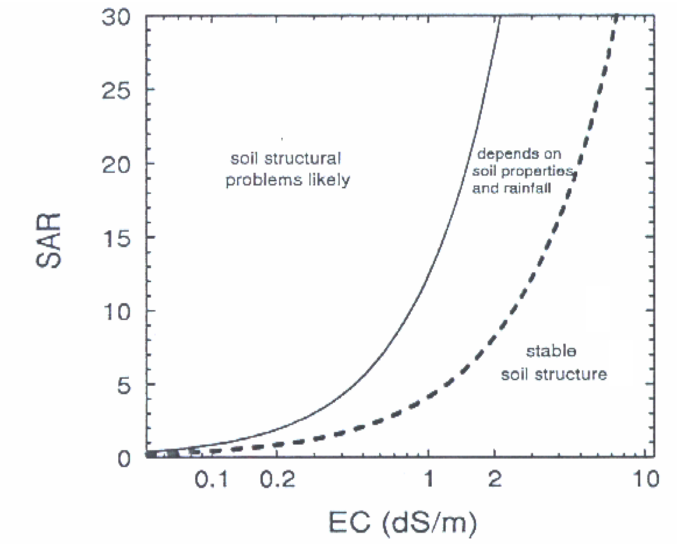


Figure 2.2: Relationship between SAR and EC of irrigation water to estimate structural stability, (ANZECC & ARMCANZ 2000).

2.3.5.3 Effects of leachate salts on plants

Excessive salt concentrations can affect plants in three ways (ANZECC & ARMCANZ, 2000; Blaylock, 1994; Parida & Das, 2005):

1. *Salinity*: Excessive salt concentrations affect a plant's ability to osmotically take up water. As the salt concentrations increase, the water potential between the plant and soil increases, reducing the plant-available water and making it harder for plants to take up water. If soil salinity exceeds a plant's tolerance, growth reductions occur and in extreme cases can cause the plant's death (ANZECC & ARMCANZ, 2000; Blaylock, 1994; Parida & Das, 2005).

2. *Specific ion toxicity:* Excessive concentrations of specific salts can be toxic to some plants particularly sodium, chloride and boron. Plants sensitive to these elements can be affected at relatively low levels if the soil concentrations are high enough (Blaylock, 1994). The effects of ion toxicities are noticeable in the leaves, particularly the leaf margins where systems include necrotic spots, leaf bronzing and, in the worst case, defoliation (ANZECC & ARMCANZ, 2000).

3. *Nutritional disorders:* High salt concentrations can cause nutritional imbalances in plants. Many salts are essential plant nutrients and high soil salt concentrations can upset the nutrient balance in plants or affect the uptake of some nutrients (Blaylock, 1994).

Plants vary in their response to high soil salinity; generally there will be no reduction in yield up to a threshold level, which varies for different plant species. Beyond the threshold limit, yields reduce at an approximately linear rate with increasing soil salinity (Table 2.5).

Table 2.5: Plant salt tolerance data.

Common name	Scientific name	Salinity threshold (soil EC, dSm^{-1})	Productivity decrease per dSm^{-1} increase (%)
White clover	*Trifolium reperis*	1.0	9.6
Paspalum	*Paspalum Dilatatum*	1.8	9.0
Kikuyu grass	*Pennisetum clandestinum*	3.0	3.0
Couch grass	*Cynodon dactylon*	6.9	6.4
Wheat	*Triticum aestivum*	6.0	7.1
Barley	*Hordeum vulgare*	8.0	5.0
Grapefruit	*Citrus paradise*	1.8	16.1
Strawberry	*Fragaria*	1.0	33.3

(after ANZECC & ARMCANZ, 2000)

Plant species that are able to grow and complete their life cycle in saline conditions are often termed ‘salt tolerant’. Salt tolerance can be defined as the ability to survive and produce economic yields under adverse conditions caused by salinity (ANZECC & ARMCANZ, 2000). Salt tolerant plants are known as halophytes and can be divided into two groups; obligate halophytes which require saline conditions and facultative halophytes which can survive in saline or freshwater conditions.

2.3.5.4 Salt Stress Symptoms in Plants

Symptoms of salt stress in plants resemble those of plants subjected to drought conditions and are characterised by wilting and reduced growth (Blaylock, 1994). Salt stress affects all major processes in plants such as growth, photosynthesis, protein synthesis, and energy and lipid metabolism (Parida *et al.,* 2005). As a result plants affected by salt stress grow more slowly and are smaller than unaffected plants. One cause of growth rate reduction is reduced photosynthesis caused by stomatal closure limiting carbon dioxide uptake (Zhu, 2001).

The leaves of salt stressed plants are smaller, but may be thicker as salinity can cause increases in epidermal and mesophyll thickness, and elongation of the palisade cells (Bernstein, 1975; Parida & Das, 2005). The leaves can be a different colour to those of unaffected plants as generally the chlorophyll and carotenoid content of leaves decrease under salt stress (Bernstein, 1975, Parida & Das, 2005).

2.3.5.5 Examples of Leachate Irrigation Schemes

There are many examples of laboratory based studies of the effects of leachate irrigation on plants and the soil, whilst there are relatively few examples of leachate irrigation studies at actual landfills in the literature.

Hernández *et al.* (1999) investigated the effects of leachate irrigation on four wild herbaceous species in a greenhouse pot study. Three different leachates (Table 2.6) were applied to pots containing one of four plant species; two legumes (*Trifolium glomeratum*. and *Trifolium tomentosum*) and two grasses (*Hordeum maurinum* and *Bromus hordaceus*).

Table 2.6: Chemical analysis of the Leachate.

Leachate Constituents	Leachate A	Leachate B	Leachate C
pH	8.3	8.0	9.1
EC (μSm^{-1})	4480	16,060	29,960
Cl^- (gm^{-3})	748	1928	4402
Ca^{2+} (gm^{-3})	215	780	39
Mg^{2+} (gm^{-3})	60	2040	755
K^+ (gm^{-3})	82	70	2350
Na^+ (gm^{-3})	530	2075	3825
NO^{3-} (gm^{-3})	10	82	64

(after Hernández *et al.*, 1999)

Leachate irrigation significantly increased soil salinity (Table 2.7). The plant response to the increase in soil salinity differed for each species with the legumes more sensitive to the increase in salinity than the grasses. The dry weight of both the legumes and grasses decreased with increasing electrical conductivity of the leachate.

Table 2.7: Soil analysis after leachate irrigation.

Soil Constituents	Soil before irrigation	Soil with Leachate A	Soil with Leachate B	Soil with Leachate C
pH	7.1	7.4	7.5	7.3
EC (µSm^{-1})	1430	6225	12500	342
Cl^- (g m^{-3})	4	95	176	4402
Ca^{2+} (g m^{-3})	275	552	583	562
Mg^{2+} (g m^{-3})	69	90	213	126
K^+ (g m^{-3})	44	57	53	234
Na^+ (g m^{-3})	6	53	183	275
NO^{3-} (g m^{-3})	3	91	60	557

(after Hernández *et al.*, 1999)

Bowman *et. al* (2002) reported on a 2 year landfill leachate irrigation trial held at the Newington Landfill, Sydney. The leachate had high salinity (17,600 µS cm^{-1}) and high concentrations of sodium (3000-4000 g m^{-3}), chloride (6700-8000 g m^{-3}) and ammonium (250-330 g m^{-3}). Plots vegetated with Couch grass (*Cynodon dactylon*) and Kikuyu grass (*Pennisetum clandestinum*) was irrigated with town supply water and leachate diluted to 20%, 50% and 100% of its original concentration with town supply water. In the plots irrigated with 50% and 100% leachate an increase in soil salinity and bulk density was observed along with a decrease in porosity, aggregate stability and biomass production. The degradation of the soil physical properties was attributed to an increase in the sodium adsorption ratio in the soil. In the 20% leachate plot, biomass production increased while there was no significant change in the soil's physical properties compared to the control plot. Nitrogen losses through leaching were significantly less in the 20% leachate plot compared to the 50% and 100% leachate plots. It was concluded that leachate irrigation is sustainable provided that management strategies, such as dilution of the leachate to reduce the electrical conductivity down to 3600 µS cm^{-1} (20% of original concentration), are adopted. Furthermore, pollution due to leaching of nitrogen to groundwater could be minimised by appropriate management of the soil to enhance denitrification and if application rates of 1400 kg NH_4 ha^{-1} yr^{-1} were achieved. It was suggested that denitrification rates of 3500 kg NH_4 ha^{-1} yr^{-1} may be achievable if the salinity hazard can be effectively managed (Bowman *et. al*, 2002).

MacDonald *et al.* (2008) investigated the effects of leachate irrigation on the capped area of an active landfill in Michigan, USA. Six experimental plots were established and three were spray irrigated with leachate and left three as a control. The leachate had high mean values for electrical conductivity (6000-7000 µS cm^{-1}), chloride (760-900 g m^{-3}) and ammonical nitrogen (290-390 g m^{-3}) but was low in metals and volatile organic compounds. During the first year of irrigation 320 mm of leachate was applied causing high rates of nitrate leaching and the soil electrical conductivities to rise from 500 µS cm^{-1} to 2800 cm^{-1} in the irrigated plots compared to 700 cm^{-1} in the control plots. In the subsequent year leachate applications rates were limited to <96 mm y^{-1} to reduce nitrate leaching and to keep soil electrical conductivities at a level which wouldn't affect plant growth or soil structural stability. After leachate application rates were reduced soil electrical conductivities reduced to 1400 µS cm^{-1} in 2004 and 1600 µS cm^{-1} in 2005. Leachate irrigation did not significantly affect soil metal concentrations and plant metal concentrations were within normal ranges.

2.4 Leachate Plumes

2.4.1 Introduction

A potential environmental problem associated with landfills is contamination of ground and surface waters by leachate (Kjeldsen *et al.*, 2002). Historically most landfills, including Paokahu, were constructed without engineered liners or leachate collection systems. In the older unlined landfills, leachate can accumulate and then percolate through the landfill base and enter local groundwater where it can cause significant changes in water quality. An area of groundwater contaminated by leachate is known as a leachate plume.

2.4.2 Leachate Plume Generation

As leachate enters an aquifer it mixes with the groundwater and dilution reduces the strength of the leachate. As a result most cases of leachate plumes are relatively small with lengths of less than 1000m although in some cases they have been detected up to 2000m from the landfill. Widths are also relatively narrow and typically limited to the width of the landfill (Christensen, *et al.* 2001). The shape of a leachate plume is primarily determined by flow directions of the groundwater. However, leachate flow patterns may differ from that of the local groundwater, which can affect the shape of a leachate plume. Leachate moving down into an aquifer can cause a groundwater mound (a localised elevation in the water table) in the vicinity of the landfill which can enhance lateral spreading and create downward flows of leachate. The viscosity and density of leachate can also influence the shape of a leachate plume, with density shown to be able to significantly affect the vertical positioning of a leachate plume beneath the landfill (Christensen, *et al.* 2000).

2.4.3 Effects of Leachate on Groundwater

The leachate present in leachate plumes often originates from methanogenic phase landfills and is often in a reduced state and when it mixes with groundwater a sequence of redox zones are created within the leachate plume. When leachate is continuously released into the groundwater a methanogenic zone evolves close to the landfill. Sulphate reduction can take place within and around the methanogenic zone. As the plume moves down gradient from the landfill, the redox potential decreases and other elements are reduced. As a result a series of reducing zones can be observed typically in the order of sulphate closest to the landfill followed by zones of iron, manganese and nitrate reducing conditions to aerobic conditions at the edge of the plume. Field investigations have shown that the redox zones don’t occur as distinct, separate zones, but rather overlap, allowing simultaneous reduction of several species, although one process may dominate (Christensen, *et al.* 2001).

2.5 Conclusion

A review of the literature has shown that leachate composition is dependent on the age and decomposition phase of the landfill. It is generally accepted that in the short term refuse decomposition takes place in four distinct phases and the concentrations of many of the leachate constituents vary between phases.

Therefore, Paokahu landfill's current decomposition phase will have an influence on it's leachate quality. The review has shown that the concentrations of some leachate species do not decline over time. Nitrate and sodium are two examples of leachate constituents which tend not to decline over time and could significantly impact on the aftercare of Paokahu landfill; nitrate, because it can cause deterioration in water quality if leachate enters the surrounding groundwater; and sodium, because it can affect soil structural properties when irrigated onto the landfill cap. In the literature it has been suggested that there are additional long term phases in which mobilisation of heavy metals can occur. Mobilisation of heavy metals within the landfill will have implications for the future long term aftercare of Paokahu landfill.

A review of the literature has also shown that leachate irrigation can be used successfully as a treatment or disposal method for landfill leachate. Leachate irrigation can reduce leachate volumes through evaporation and evaportranspiration, and improve leachate quality through the uptake of nutrients by plants, retentions in the soil and microbial degradation of the leachate. However, the literature has also shown that in some instances leachate irrigation can be detrimental. High sodium concentrations in irrigated leachate can affect soil structure, reducing hydraulic conductivities and infiltration rates. High sodium leachate concentration can also stress the plants, reducing plant growth, or in extreme conditions cause plant death.

Chapter 3. History, Geology and Management Practices

3.1 Introduction

This chapter gives a brief overview of the history of Paokahu landfill from its inception in the late 1960's to its eventual closure in 2002. The area around the landfill is located in the former Awapuni Lagoon, for which a brief description of the geology, hydrology and soils is given. Finally, the current management practices used at the landfill are summarised.

Much of the information contained in this chapter, particularly relating to the history, was obtained by reviewing the Gisborne District Council's files held on Paokahu Landfill, while the information of the management practices comes from personal experience gained during my time managing the site as Solid Waste Engineer, for Gisborne District Council.

3.2 Site History

Gisborne City Council (which amalgamated to become Gisborne District Council in 1989) first became interested in developing a landfill in the Awapuni Lagoon area in the late 1960's early 1970's. The Council was interested in this area because of its proximity to the city, lack of residential housing, large potential capacity and plentiful supply of sand dunes which it intended to 'mine' for use as cover material. The proposed site was not within the boundaries of Gisborne City, but in the adjacent Cook County. The site was on a 522 acre block of Maori Freehold land, locally known as the Paokahu Block. At the time the land was under lease to the Economic Butchery Ltd who used the land for grazing. The area was zoned as Rural under the Cook County Council's District Scheme, which prohibited development of a landfill.

Gisborne City Council formally announced its interest in the site in June 1970. The City Council applied under Section 35 of the Town and Planning Act (1953) for consent for a specified departure from the Cook County District Scheme on 20 March 1972, for 321 acres of the Paokahu Block to be rezoned as a refuse disposal site. Thirteen objections to the departure notice were lodged which included objections from the landowners, the trustees of the land and the Poverty Bay Catchment Board. A hearing took place on 6 July 1972 which resulted in "consent in part" being given. Sixteen conditions were imposed on the development by the Cook County Council, most notable were:

- Consent was only given for 50 acres of the 321 applied for.
- The dune section of the block was not included.
- The GCC had to develop a comprehensive site plan detailing what the site would look like once the landfill had reached the end of its life.

Gisborne City Council appealed the decision under section 35(5) of the Town and Country Planning Act on the grounds that the whole site was needed to give some long term certainty to the City's waste disposal operations. Gisborne City Council also stated that using the sand from the sand dunes for landfill cover was essential to the site operating successfully. The Town and Country Appeal Board heard the appeal in May 1973. The appeal was dismissed and the conditions imposed by the Cook County Council were upheld.

Although their appeal was dismissed the Gisborne City Council decided to continue with its plans to develop a landfill at Paokahu but in accordance with the conditions of the consent. As only 50 acres were consented, Gisborne City Council did not pursue acquisition of the land through the Public Works Act. Instead it was decided to try and obtain a lease for the site. After some negotiation the Council and Mungatu Blocks (the trustees of the Paokahu Block at the time) agreed terms on a 15 year lease. The lease was signed on 4 September 1975 upon which the Council took possession of the site.

The landfill opened 14 February 1977 and received a mixture of domestic, agricultural and industrial waste. Approximately 100 million cubic metres of waste was disposed of in the landfill during its life, averaging approximately 4 000 000 cubic metres per annum. Disposal peaked at 70 000 tonnes per annum in the1980/90's of which approximately 60% was organic waste produced in the city's vegetable processing plants. Tip fees were phased in during the 1990's after which the total tonnages began to decrease and at closure the landfill was receiving less than 20,000 tonnes per annum.

An engineered liner was never installed beneath the landfill; instead the designers relied on the low permeable nature of marine sediments located under the site to prevent leachate migration into the surrounding ground water.

Only a portion of the site was ever used for dumping at any one time, and the active tip face moved across the site over time (Figure 3.1). The landfill typically practiced an accept, compact, and cover method of landfilling. Waste was tipped at the tip face (Figure 3.2), then compacted using a caterpillar compactor and the tip face covered at the end of each day. Initially sand from the adjacent beach was used for daily cover, but this practice was stopped in the1990's and replaced by bark, a waste product from the Port of Gisborne log export operation. The exact volume of sand used in the landfill is unknown, but up to 8,000 m^3 of sand per year was extracted between 1977 and 1997 for capping purposes.

In 1997 Gisborne District Council applied for consents to extend the landfill into adjacent land, known locally as the Gavins Block. The consent application was unsuccessful and the landfill closed on 31st December 2002; after which it was used as a temporary transfer station while a more permanent refuse transfer station was constructed in Gisborne.

Figure 3.1: Paokahu Landfill December 2000, showing the active tip face in the northern corner.

The landfill was finally completely capped in 2003. The cap is made up of 100 mm layer of bark placed directly over the refuse, overlaid with a 250mm thick layer of clay, 150mm of beach sands and finally a 200mm thick layer of top soil. Some older areas which had previously been capped had an additional cap placed on top consisting of 250mm layer of clay, 150mm of beach sands and a 200mm thick layer of top soil. The landfill cap is classified as a Buried Refuse Anthropic Soil under the New Zealand Soil Classification (Hewitt, 1992).

Figure 3.2: Paokahu tip face 1996.

3.3 Hydrology, Geology and Soils of the Paokahu Area

3.3.1 Introduction

Paokahu landfill is located in the south-western corner of the Poverty Bay Flats, which consists of a prograding beach backed by sand dunes and beach sand ridges enclosing estuaries and lagoons. Behind the dune system is an alluvial plain extending inland for approximately 20 km and to 30 m above sea level (Brown and Elmsly, 1987). The Waipaoa River meanders across the plains and is tidal inland of the landfill.

Paokahu landfill is located on flat ground, which was originally the base of the Awapuni Lagoon. The lagoon once covered more than 300 Ha between the Waipaoa River mouth and Awapuni Rd (Figure 3.3). The lagoon was drained in the 1950's for farm development, with the installation of the Awapuni drain; a tide gate controlled drain. Coastal dunes which are 3-4 metres in height are located between the landfill and the beach (Nelson, 1994).

Figure 3.3: Awapuni Lagoon in 1942.

Basement rock consists of Tertiary age mudstone, siltstones and sandstones, which are expected to lie at a depth of 50 to 100m below the landfill. A series of gravels, sands, silts intermixed with volcanic ash showers form a complex series of soils between the surface and the bedrock (Nelson, 1994).

3.3.2 Ground Water

At least six aquifers have been identified within the Poverty Bay Flats (Brown and Elmsly, 1987). The six aquifers being the: Matokitoki Gravel, Makauri Gravel, Waipoa Gravels, Te Harapa Sands, shallow fluvial deposits and a number of localised aquifers. Generally ground water flows towards the coast, with salt water intrusions in some costal areas.

Three aquifers are found in the vicinity of the landfill (Figure 3.4); Shallow Fluvial deposits, Te Hapara Sands Aquifer and the Makauri Gravels Aquifer (Brown and Elmsly, 1987).

3.3.2.1 Shallow Fluvial Deposits

The Shallow Fluvial deposits are shallow ground water aquifers that occur near the surface through out the Poverty Bay Flats. They consist of silt layers inter-bedded with pumice sands and can be up to 10m thick. The groundwater is recharged by rainfall infiltration and from the underlying aquifers (Brown and Elmsly, 1987). In the area of the Paokahu landfill the shallow aquifers occur in the shallow estuarine sand and silt deposits found on the base of the former Awapuni Lagoon.

3.3.2.2 Te Hapara Sands Aquifer

The Te Hapara Sands Aquifer lies beneath the landfill and extends approximately 5km inland from the present day coast. It has formed over the last 4000 years as a result of coastal progradation infilling lagoons and establishing sand dunes. The groundwater is derived from rain and interconnections with the shallow fluvial deposits and the Waipaoa Gravels. The Te Hapara sands are up to 20 m thick and form a shallow semi confined aquifer. Permeability decreases as the aquifer nears the Waipaoa River in the south west due to an increasing silt content (Brown and Elmsly, 1987).

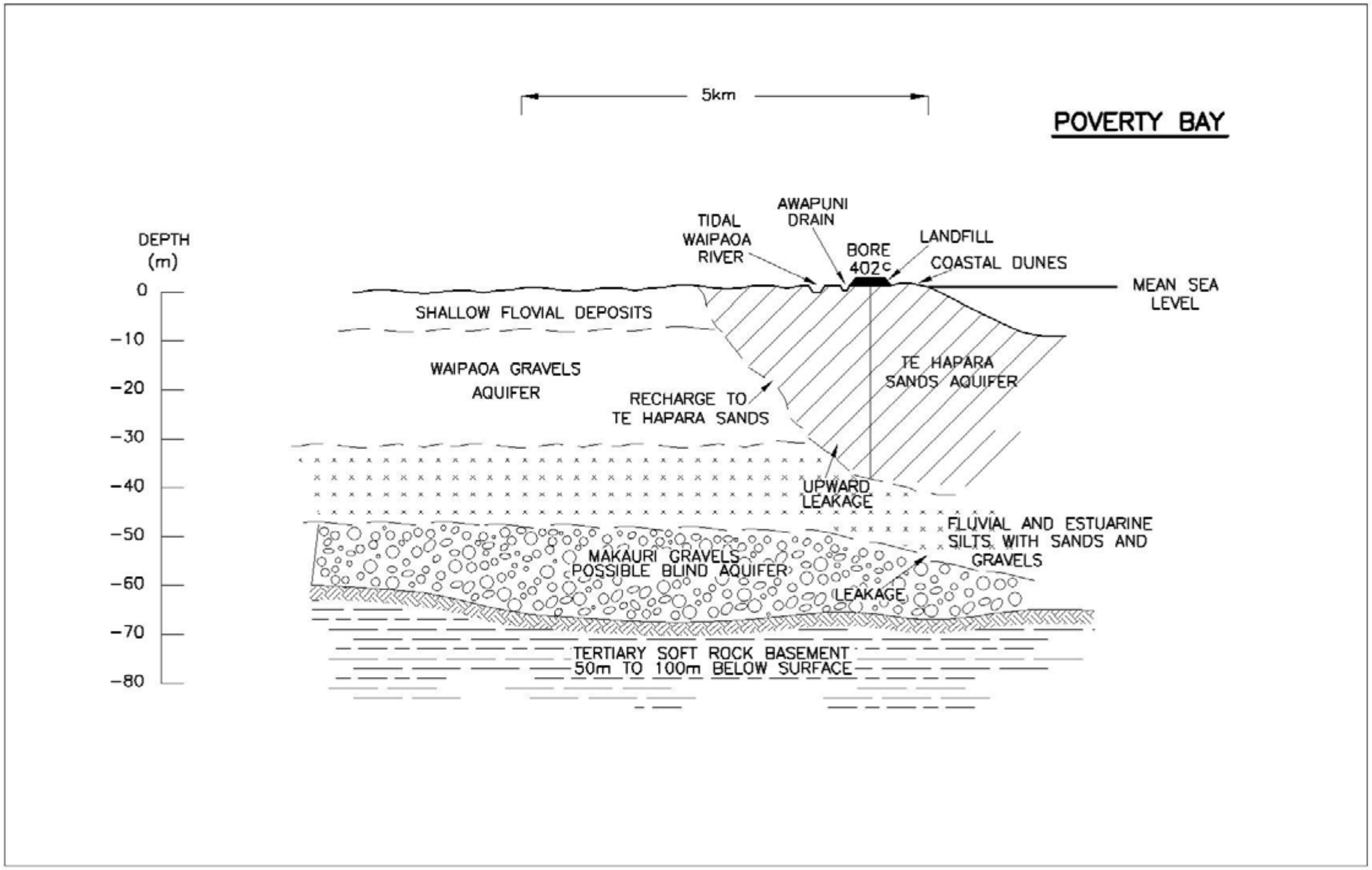


Figure 3.4: Cross Section showing the aquifers under Paokahu Landfill (after Nelson, 1994).

The water quality can be variable and susceptible to bacteriological and chemical pollution from industrial, horticultural, pastoral and domestic sources due to its shallow nature. Salinity, chloride and conductivity increase toward the coast due to salt water intrusion (Brown and Elmsly, 1987).

3.3.2.3 Makauri Gravels Aquifer

The Makauri Gravels Aquifer is the deepest and most extensive aquifer on the Poverty Bay Flats (Brown and Elmsly, 1987). The aquifer is thought to have been deposited by the Waipaoa River early in the post glacial period that occurred 14,000 years ago (Brown, 1984). Gravel and sand deposits form a confined aquifer at a depth of 40 to 60 metres below the ground surface. The aquifer is confined by overlying silt layers and it is likely that slow upward leakage occurs into the base of the Te Hapara Sands Aquifer in the vicinity of the Paokahu landfill (Nelson, 1994). The water quality of the Makauri Gravels aquifer is generally poor. Iron and hardness content exceed the World Health Organisation’s allowable limits for drinking water and chloride concentrations can exceed 28 mmol kg^{-1} in some locations (Taylor, 1994).

3.3.2.4 Landfill Groundwater Flow Charteristics

A groundwater study was undertaken at Paokahu Landfill during the summer of 1993/94 by Nelson as part of the resource consent application to extend the landfill. The report identified that the ground water flow under the landfill is directed inland towards the Awapuni drain, as opposed to the regional flow which is towards the coast. The reason for the inland groundwater flow was attributed to the low water levels maintained in the Awapuni drain which has a level equivalent to mean low tide (Nelson, 1994).

Along with an inland flow of ground water, salt water intrusions occur from the coast in the direction of the Awapuni drain as indicated by increased salinity and conductivity in the groundwater monitoring bores. Nelson suggested an upward leakage of ground water occurs from the Makauri Gravels Aquifer into the Te Hapara Sands aquifer. As a result there are very shallow zones of fresh water near the surface, a deeper mixed zone dominated by salt water and trend to fresh water with depth, possibly due to leakage from the Makauri Gravels Aquifer (Nelson, 1994).

Hydraulic conductivities under the landfill were calculated using constant head drawdown tests with the mean value for the landfill being 1.9 x 10^{-5} m/s over a range of 1.9x10^{-4} to 1x10^{-6} m/s. There is no difference in conductivity between the dune sands and the Te Hapara Sands unit, possibly due to the presence of a clay layer at the base of the dunes acting as an aquitard (Nelson, 1994).

3.3.3 Soils of the Paokahu Area

Three soil types are found in the vicinity of the landfill (Figure 3.5); Opoutama Sand, Muriwai Clay, and the Makaraka Clay Loam, Saline Phase (Pullar, 1962).

3.3.3.1 Opoutama Sand

Opoutama Sand is formed from the beach deposits nearest the shore line. In its natural state the sands are stabilised by a close cover of weeds and coastal grasses. A typical profile was described by Pullar (1962) as consisting of:

0-15 mm	Layer of dead roots.
15-65 mm	Pale brown sand held together by a mat of living and dead roots.
>65 mm	Grey loose sand with roots penetrating to 255 mm.

New Zealand Soil Classification: Typic Sandy Recent Soil (Hewitt 1992).

In a typical profile the top 150 mm has a near neutral pH (pH 6.8). Below 380 mm the subsoil is moderately alkaline. The Opoutama sands dry out badly in summer which limits productivity (Pullar, 1962).

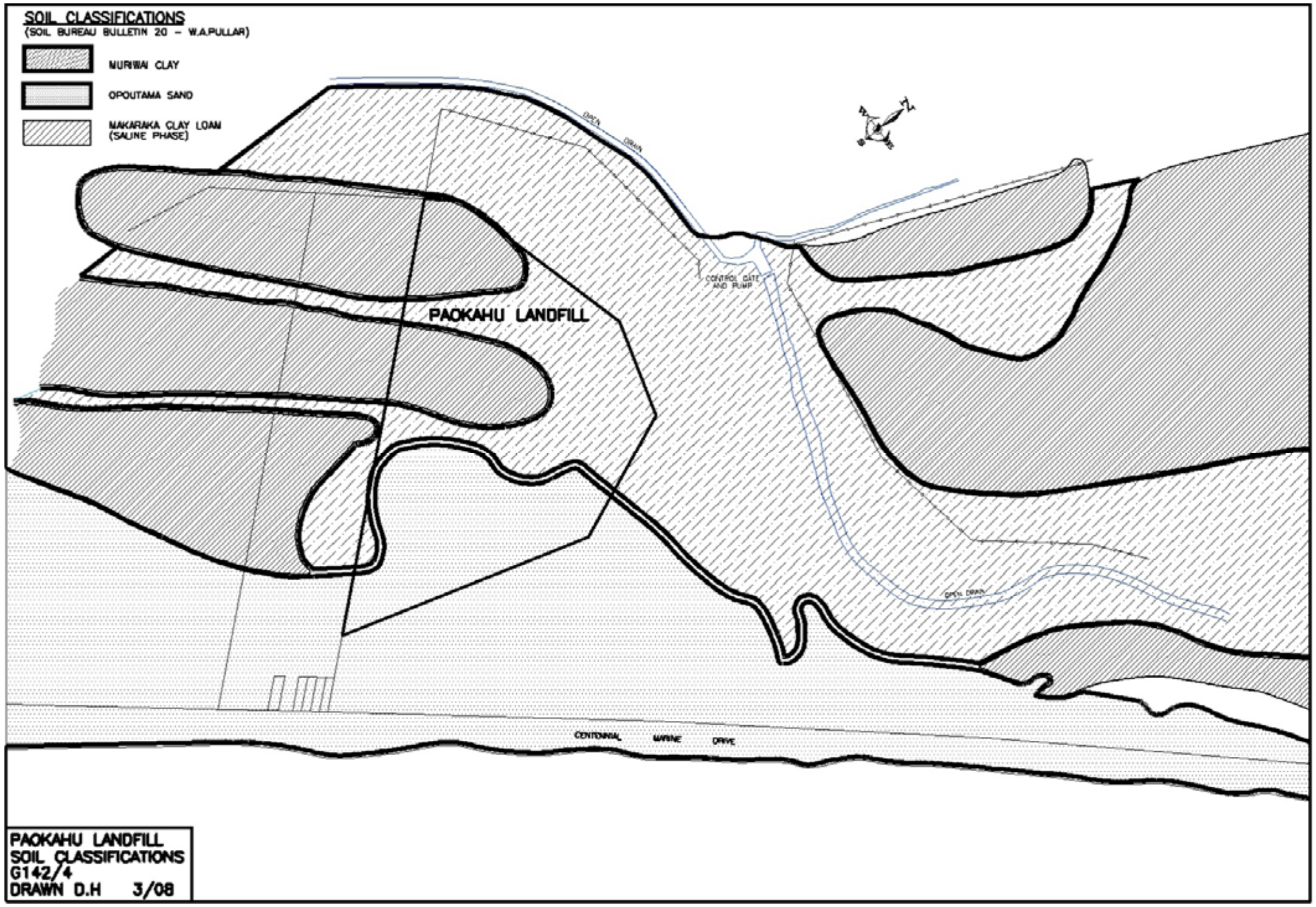


Figure 3.5: Soil Map of the Paokahu Area (after Pullar, 1962).

3.3.3.2 Muriwai Clay

There are approximately 140 hectares of the Muriwai clay in what was the Awapuni Lagoon. The lower lying areas were flooded daily with sea water prior to the installation of the tide control gate. The soil is moderately alkaline (pH 7.8) and is extremely high in soluble salts, particularly sodium and magnesium chlorides and the organic matter content is low (Pullar, 1962). A typical profile was described by Pullar (1962) as consisting of:

0-100 mm	Grey laminated clay with a coarse prismatic structure, breaking into coarse blocks. Many yellowish brown mottles are present on the faces of blocks indicating periodic wetting and drying.
100-400 mm	Greyish brown sticky plastic clay with a weak coarse black structure. Many fine roots are present which are enclosed in bluish grey sheaths indicating permanent wetness.
400-500 mm	Greyish sticky massive clay.
>500 mm	Wet viscous greenish grey clay.

New Zealand Soil Classification: Saline Recent Gley Soil (Hewitt 1992).

3.3.3.3 Makaraka Clay Loam Saline Phase

The Saline phase of the Makaraka Clay Loam occupies 170 Hectares of the Poverty Bay Flats' and is restricted to lower lying areas bordering Awapuni Lagoon and at Muriwai. The profile of the Makaraka clay loam saline phase in the Awapuni Lagoon was described by Pullar (1962) as being:

0-100 mm	Grey clay loam, impregnated by roots.

100-250 mm	Very dark greyish brown clay loam. An abundance of organic matter is present with weakly decayed roots and stems that impart a peatiness to the soil.
250-630 mm	Grey clay loam with distinct fine yellowish brown mottles.
>630 mm	Sand.

New Zealand Soil Classification: Typic Sandy Recent Soils (Hewitt 1992).

3.3.4 Geology under the landfill.

The geology beneath the landfill has been determined by analysis of the bore logs of the bores which have been dug in the area. Figure 3.6 gives the location of two cross sections through the landfill. A north-south (A-A) cross section is shown in Figure 3.7 and an east-west cross section (B-B) is shown in Figure 3.8. The geology beneath the landfill can be summarised as:

Recent estuarine silts

Recent estuarine silts are located immediately beneath the landfill. The silts are of varying thickness but generally inland of the landfill. Bore log data indicates the estuarine silts are absent beneath the sand dunes south of the landfill but increase to 2-3 m thick 150m northwest of the landfill (Nelson, 1994).

Recent marine sands

Recent marine sands underlie the seaward (southern) end of the landfill.

Te Hapara Sands

The Te Hapara Sands consist of fine to coarse grained pumiceous sands, with scattered shells and woody fragments. The Te Hapara Sands extend to a depth of at least 20 m below the landfill (Nelson, 1994).

Older estuarine silts

Older estuarine silts consisting of stiff to very fine grained sandy silt, with scattered shells and woody fragments interlined with silty sands. The older silts have been identified in two of the bore logs and underlie the Te Hapara Sands (Nelson, 1994).

The Makauri Gravels

The Makauri Gravels are thought to underlie the Te Hapara Sands but have not been positively identified within a depth of 40 m (the maximum bore depth) under the landfill (Nelson, 1994).

Recent coastal dune sands

Recent coastal dune sands form the higher ground on the seaward side of the landfill.

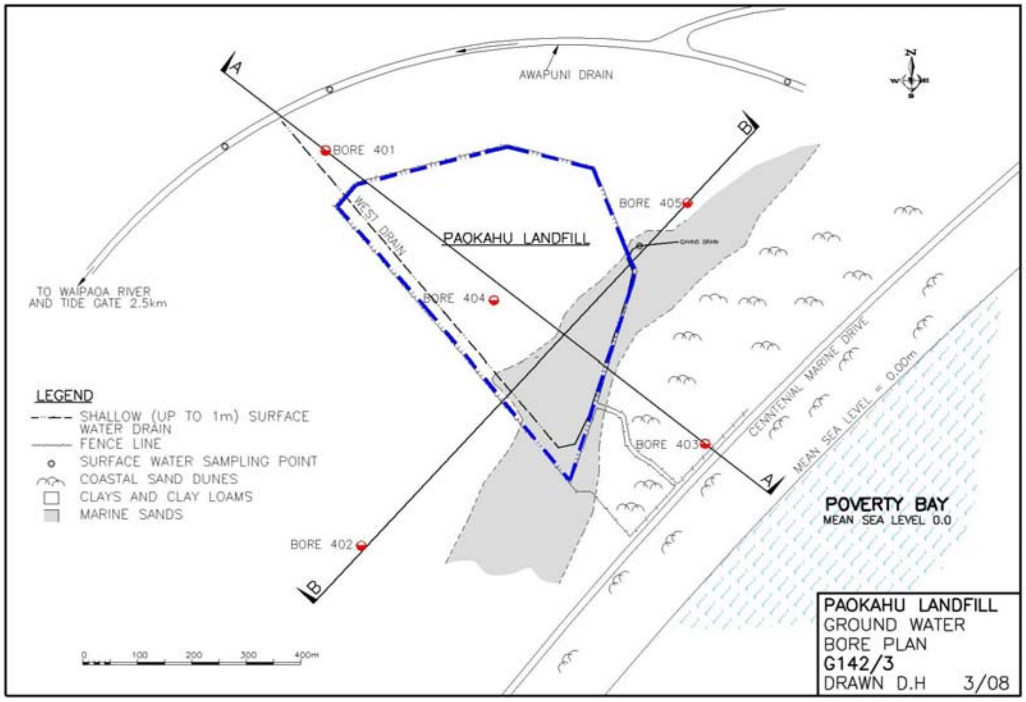


Figure 3.6: Site Plan and Bore Locations (after Nelson, 1994).

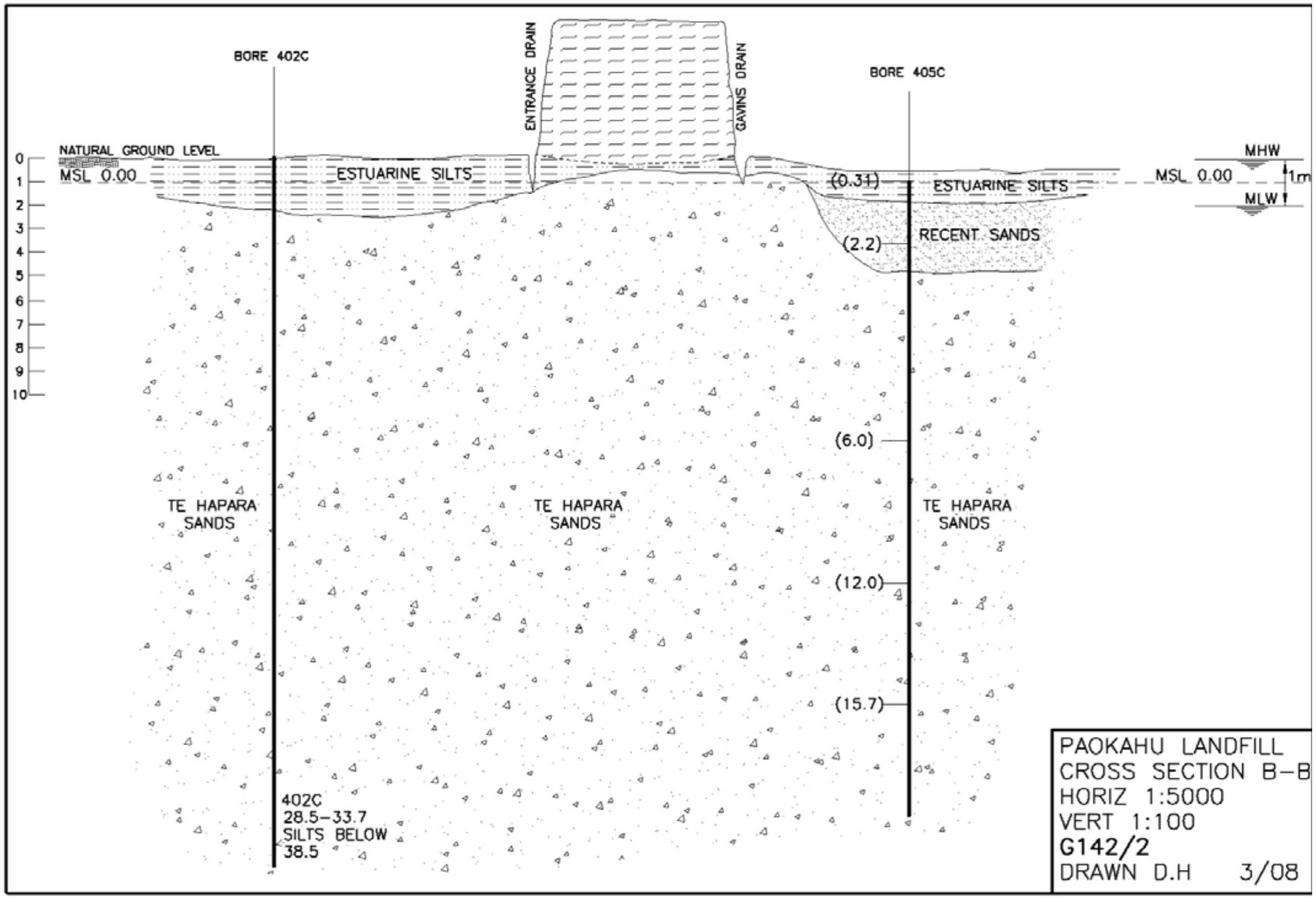


Figure 3.7: Cross section A-A (after Nelson, 1994).

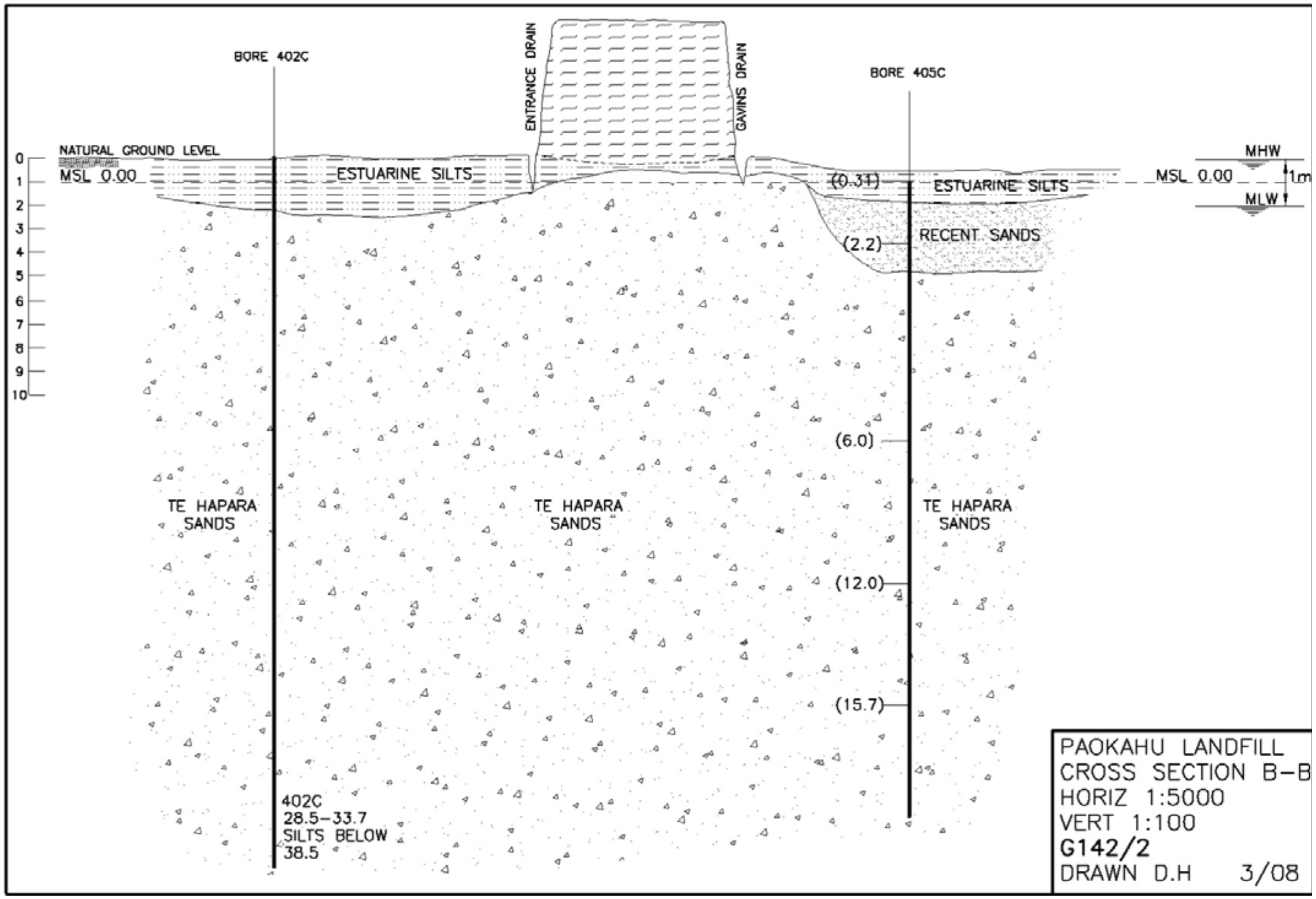


Figure 3.8: Cross section B-B (after Nelson, 1994).

3.4 Management Practices

3.4.1 Introduction

The following sections outline some of the current management practices at Paokahu Landfill.

3.4.2. Landfill Aftercare Management Plan

A management plan for the aftercare of Paokahu Landfill has been in place since 2004. The purpose of the plan was to ensure the closure of the landfill is managed properly and to mitigate any adverse effects on the environment. The plan outlines how the leachate, stormwater, cap maintenance, landfill gas, fencing, monitoring and reporting are to be managed.

3.4.3 Leachate Management

3.4.3.1 Leachate Collection System

A leachate collection system is in place which covers approximately three quarters of the site. There is no leachate collection in the south western face of the landfill, as this is the older original area of the site and contains a much thinner layer of refuse. The leachate collection system comprises an engineered bund with a design permeability of 10^{-9} m s^{-1} at the edge of the landfill to capture the leachate. Inside the bund a 90 mm MDPE perforated pipe collects the leachate. The pipe was laid at a depth of 1m and the trench was back filled with metal and then landfill material.

There are five submersible pumps installed in sumps along the collection line to pump the leachate to a central collection tank located in the south western corner of the landfill (Figure 3.9). The pumps operate automatically with pumping triggered by a float switch inside the well. A telemetry system monitors the leachate collection system and warns of high sump levels, pump failures and power failures. Additional sumps were installed to act as inspection chambers.

Figure 3.9: Leachate collection sump: A) the outside of a leachate collection sump; B) the inside of a collection sump.

3.4.3.2 Leachate Irrigation

Disposal of leachate is by spray irrigation onto the landfill cap. Leachate in the collection tank (Figure 3.10) is pumped through two distribution lines to the sprinklers. Three sprinkler lines, each containing seven "K line" sprinkler heads irrigate the leachate (Figure 3.11). The sprinkler lines are moved regularly to ensure even distribution of the leachate and prevent ponding. Irrigation is fully automated and pumping is triggered by a float switch in the collection tank. No system is in place to stop pumping during times of high rainfall.

Figure 3.10: Leachate collection tank and pump house.

Figure 3.11: Leachate sprinkler line showing "K line" sprinklers.

Approximately 15 ha are available for leachate irrigation. Leachate irrigation is limited to the top of the landfill cap and doesn't takes place on batter slopes. Irrigation rates are measured by a flow meter installed between the irrigation pump and the start of the irrigation line (Figures 3.12 & 3.13). Approximately 13 000 -15 000 m^3 of leachate is irrigated annually and irrigation rates show marked seasonal variations (Figure 3.14), ranging as low as $3m^3$ per month over summer to a $4500m^3$ per month in winter. The current management plan allows for an irrigation rate of 130-150 mm of leachate per year with actual irrigation rates of approximately 100mm per year.


Figure 3.12: Leachate Irrigation Pump.

Figure 3.13: Flow Meter.

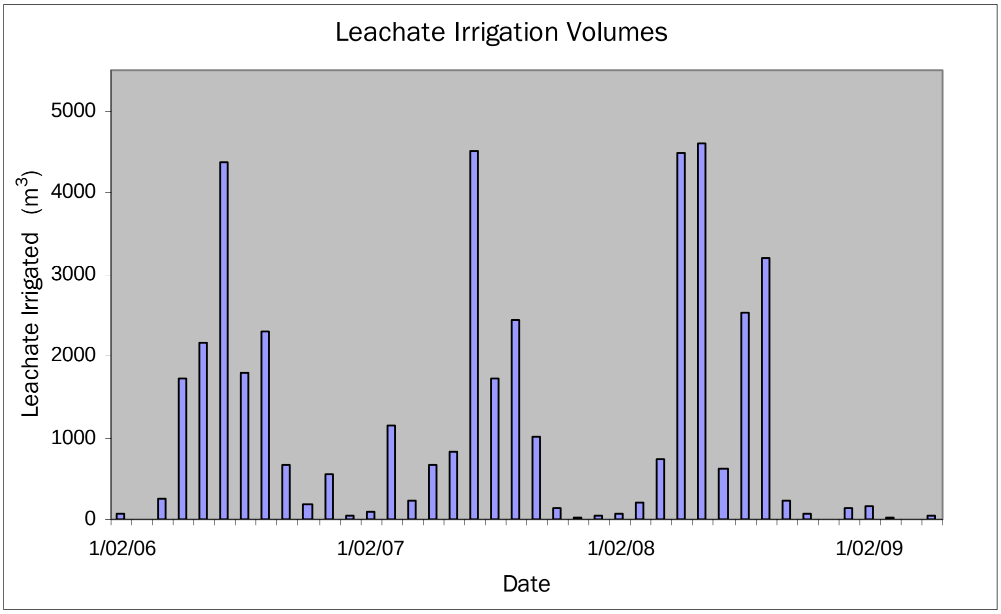


Figure 3.14: Monthly Leachate Irrigation Volumes.

3.4.4 Stormwater Management

With the establishment of pasture on the landfill cap most of the stormwater problems have largely been mitigated. Stormwater quality is now mainly influenced by the agricultural land use (Gisborne District Council, 2007).

Stormwater flows into perimeter drains (Figure 3.15) at the edge of the landfill which discharge into the West Drain and Gavins Drain, which in turn discharge into the Awapuni Drain.

During capping, a stormwater pond was installed in the north western corner of the landfill to catch any sediment coming off the landfill cap during rainfall events. The pond is still in place but is now not used.

Figure 3.15: Perimeter Drain adjacent to the Gavin's Block.

3.4.5 Pasture Management

The landfill is currently used for pastoral farming with grazing limited to sheep only. Large flocks of sheep are periodically brought onto the landfill and it is intensively grazed for short periods. The landfill is not divided into smaller paddocks but has been left as one large block. There is no grazing withholding period after leachate irrigation and irrigation continues while stock are on site. Cattle are not allowed on the landfill because of their potential to damage the landfill cap.

The dominant pasture species are clovers and rye grasses on the top section of the cap, and Indian Doab (Cynodon Dactylon) on the slopes. Indian Doab was planted on the slopes because it is fast growing, drought tolerant and provides a strong dense cover to prevent surface erosion. Indian Doab is not palatable to sheep and so not subjected to grazing pressure. As a result the Indain Doab is spreading out from the slopes and displacing the clovers and ryegrasses on the top of the landfill. To control the spread of the Indian Doab it is mowed to simulate grazing by cattle. It is still to be determined if this is an effective method of control.

Soil and herbage samples are taken annually in autumn from set sampling points across the top of the landfill. The results are used to determine the fertiliser requirements. The results of the soil and herbage testing indicate that the pH is high (7.8); soluble salts concentrations are medium but high enough to affect clover growth; and organic sulphur is low and the application of a slow release sulphur fertiliser is needed.

3.5 Climate

There is no weather station situated at Paokahu Landfill but climate data is recorded at the nearby Gisborne Airport (approximately 3.7 km northeast of the landfill). Climate data can be accessed online through National Climate Database (CliFow) run by Niwa. All climate data in this thesis relating to Paokahu Landfill comes from the Gisborne Airport weather station.

The landfill generally has warm summers and mild winters with a yearly average of 2 200 sunshine hours. Day time temperatures range from a mean of 20°C over summer to a mean of 12°C over winter (Figure 3.16). Warm dry weather predominates in summer, often accompanied by dry north westerly winds and temperatures as high as 38°C have been recorded.

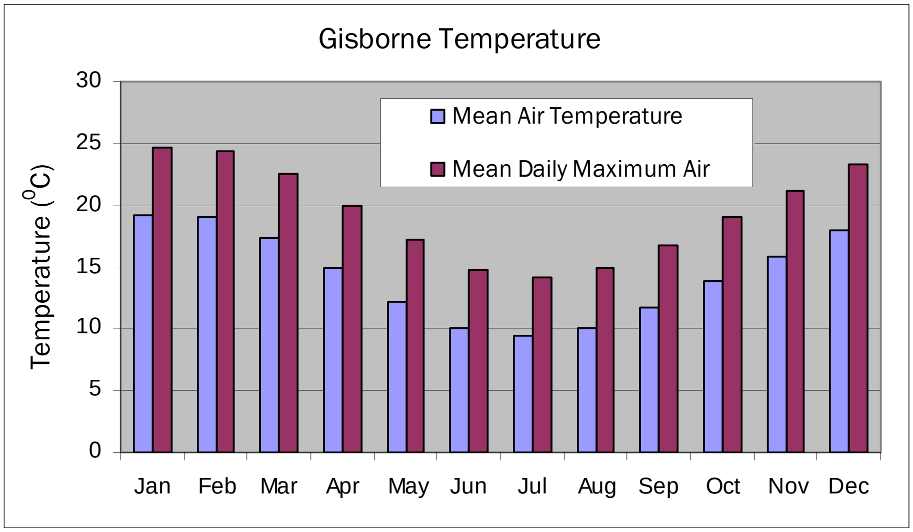


Figure 3.16: Mean monthly temperatures (1999-2009).

Paokahu's mean annual rainfall is approximately 1000mm per year, with less rainfall in summer than winter (Figure 3.17). Evopotranspiration had been estimated and ranges from 140mm per month in summer to as low as 4mm per month in winter (Figure 3.18). There is a monthly soil moisture deficit of approximately 110 mm in the summer which drops as low 6 mm in winter (Figure 3.19).

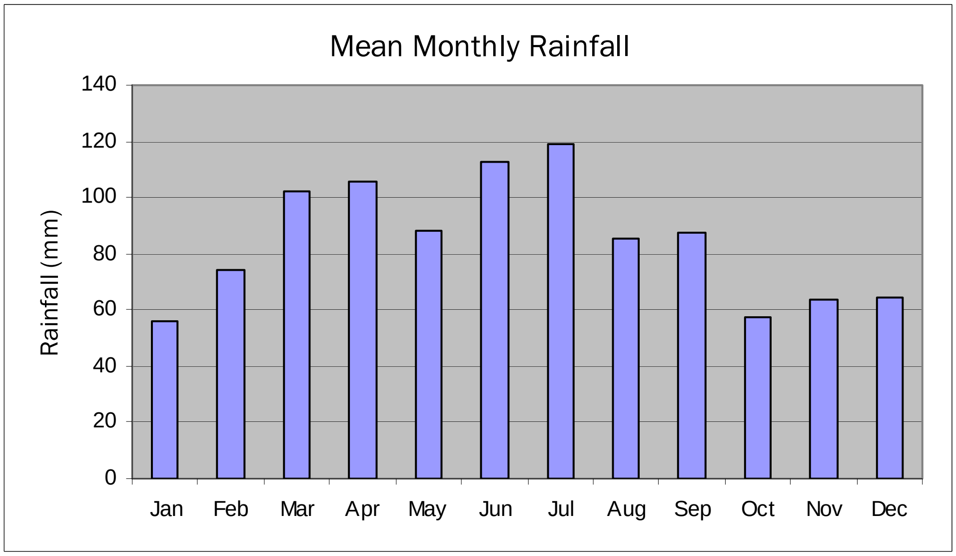


Figure 3.17: Paokahu mean monthly rainfall 1999-2009.

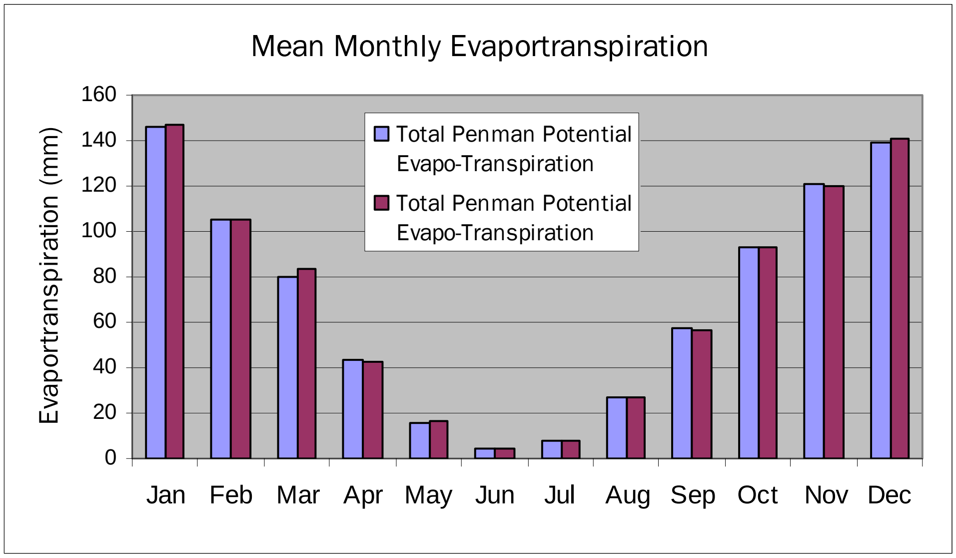


Figure 3.18: Paokahu mean monthly evapotranspiration 1999-2009.

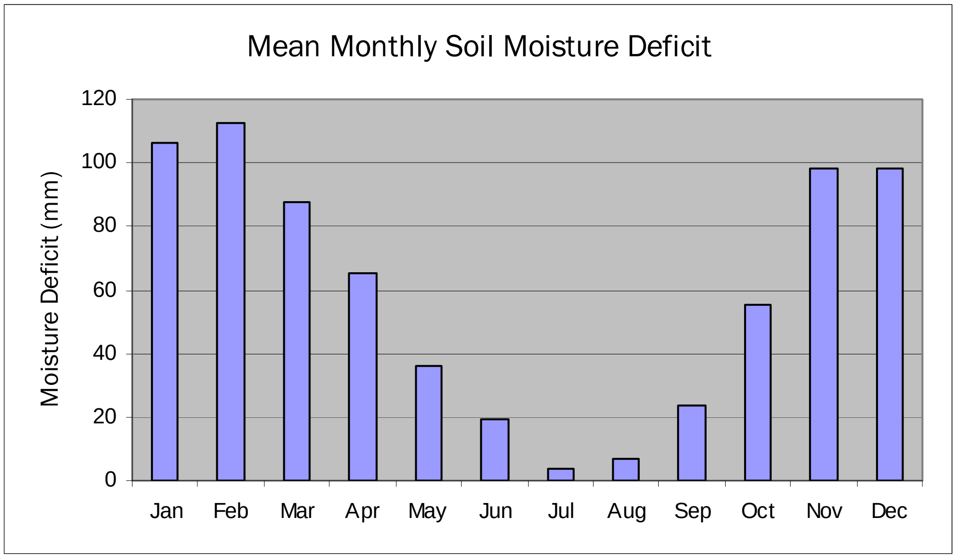


Figure 3.19: Paokahu mean monthly soil moisture deficit 1999-2009.

Chapter 4.
Methods

4.1 Introduction

This chapter describes the methods used in the collection and chemical analysis of the groundwater and leachate bores samples. Both the leachate and groundwater bore analyses used the same sampling and analytical methods.

The statistical methods used to compare the groundwater bores, leachate bores and soil samples are also outlined.

4.2 Bore Sample Collection

Leachate and ground water monitoring began in 1999 and is currently ongoing. Samples were collected from the bores by Hydro-Technologies Limited of Gisborne under their environmental monitoring contract with Gisborne District Council. Samples were collected by a trained field technician using a battery powered 12 volt submersible pump (Figure 4.1). The pump was lowered into the bore head (Figure 4.2) until it reached the water level and a sample was then pumped out of the bore directly into a sample bottle. Sample size was determined by the type of tests; specific sample bottles were used depending on which chemical tests were undertaken. The samples were then taken to the Hytro-Technologies Ltd Laboratory, Awapuni Rd, Gisborne, for analysis.

The frequency of sampling, number of bores being monitored, and the analysis undertaken has varied over the life of the landfill. Generally a full suite of tests was undertaken yearly in both the leachate and groundwater bores. Interim monitoring was carried out over the year but with only a few key indicators being tested. Since the closure of the landfill monitoring frequencies have decreased in both the groundwater and leachate bores and stopped all together in some bores. Currently, monitoring occurs on a yearly frequency in the groundwater bores and two yearly in the leachate bores.

Figure 4.1: One of the pumps used to collect bore samples.

Figure 4.2: Leachate bore 2 and groundwater bores 404A and 404B.

4.3 Bore Sample Analysis

Not all the required tests are performed in Gisborne and so testing was split between Hydro-Technologies Ltd and R J Hill Laboratories Limited, Hamilton. At the Hydro-Technologies Ltd Laboratory the following were measured: pH, sample temperature, electrical conductivity, alkalinity, biological oxygen demand and chemical oxygen demand. The remainder of the tests were performed by R J Hill Laboratories Limited. Both laboratories are IANZ accredited and all testing is performed in accordance with the procedures set out in Clesceri *et al*. (1998); The American Public Health Association (APHA) Standard Methods for the Examination of Water and Wastewater (Table 4.1). Tests performed between 1999 and 2007 were conducted using the 20th edition of Standard Methods for the Examination of Water and Wastewater, while tests carried out between 2008-2009 were conducted using the 21st edition. Semivolitile organic compounds were tested for, using gas chromotagraphy-mass spectrometry (GC-MS). Where a result was returned which was outside the expected range or if a laboratory error was suspected no retesting of the samples took place.

4.4 Statistical Analyses

Statistical analyses were undertaken to compare results between bores and between locations where soil samples were taken. The statistical method used was a Students T Test, and was performed using Microsoft Excel. It was assumed that there was a two tailed distribution and an equal variance between samples.

Table 4.1: Leachate and groundwater analysis methods.

Test	**Method Description**
pH	APHA method 4500H.
Electrical Conductivity	APHA method 2510B.
BOD	APHA method 5210B.
COD	APHA method 5220B.
Alkalinity	APHA method 2320B.
Dissolved Boron	Filtered sample, APHA method 3125B.
Dissolved Cadmium	Filtered sample, APHA method 3125B.
Dissolved Calcium	Filtered sample, APHA method 3125B.
Dissolved Chromium	Filtered sample, APHA method 3125B.
Dissolved Copper	Filtered sample, APHA method 3125B.
Dissolved Iron	Filtered sample, APHA method 3125B.
Dissolved lead	Filtered sample, APHA method 3125B.
Dissolved Magnesium	Filtered sample, APHA method 3125B.
Dissolved Manganese	Filtered sample, APHA method 3125B.
Dissolved Mercury	Filtered sample. Permanganate / Persulphate digestion. Analsis by FINS.
Dissolved Nickel	Filtered sample, APHA method 3125B.
Dissolved Potassium	Filtered sample, APHA method 3125B.
Dissolved Selenium	Filtered sample, APHA method 3125B.
Dissolved Sodium	Filtered sample, APHA method 3125B.
Dissolved Zinc	Filtered sample, APHA method 3125B.
Chloride	Filtered samples. Ferric thiocyanate colorimetry. Discrete Analyser. APHA method 4110 Cl^- E (modified from continuous flow analysis).
Total Ammonical-N	Filtered samples. Phenol/hypochlorite colorimetry. Discrete Analyser. (NH^{4-+}–N + MH3-N). APHA method 4110 Cl- E (modified from manual analysis).
Nitrite-N	Automated Azo dye colorimetry,flow injection analyser. APHA method 4500-NO_3^-.
Nitrate-N	Calculation: (Nitrate-N + Nitrite-N) - NO_2-N.
Nitrite-N + Nitrate-N	Total oxidised nitrogen. Automated cadmium reduction, flow injector analyser. APHA method 4500-NO_3^- I
Dissolved Reactive Phosphorous	Filtered samples. Molybdenum blue colorimetry. Discrete Analyser APHA method 4500-P E (modified from manual analysis).
Sulphate	Filtered samples, Ion Chromatography. APHA 4110 B.
Semivolitile Organic Compounds	
Semi volatile Organic compounds	Solid phase extraction, GC-MS FS anaylsis.
Haloethers	Solid phase extraction, GC-MS FS anaylsis.
Nitrogen containing compounds	Solid phase extraction, GC-MS FS anaylsis.
Organochlorine pesticdes	Solid phase extraction, GC-MS FS anaylsis.
Polycyclic Aromatic hydrocarbons	Solid phase extraction, GC-MS FS anaylsis.
Phenols	Solid phase extraction), GC-MS FS anaylsis.
Plasticisers	Solid phase extraction, GC_MS FS anaylsis.

Halogenated compounds	Solid phase extraction, GC_MS FS anaylsis.

Chapter 5 Leachate Analysis

5.1 Introduction

This chapter presents the leachate monitoring data for Paokahu Landfill from 1999 until 2008. Environmental monitoring in its current form began at Paokahu Landfill in August 1999 and coincided with the site becoming fully consented under the Resource Management Act (1991). Leachate samples were collected from two bores; Leachate Bore Two (LB2) (Figure 5.1) and Leachate Bore Six (LB6). LB6 is located in the north western corner of the landfill while LB2 is located in the centre of the landfill (Figure 5.2). A range of chemical analyses was undertaken on the leachate and the results recorded (Appendix 1). Monitoring frequencies have decreased over time since the landfill closed and are currently carried out on a two yearly cycle.

Figure 5.1: Leachate Bore 2.

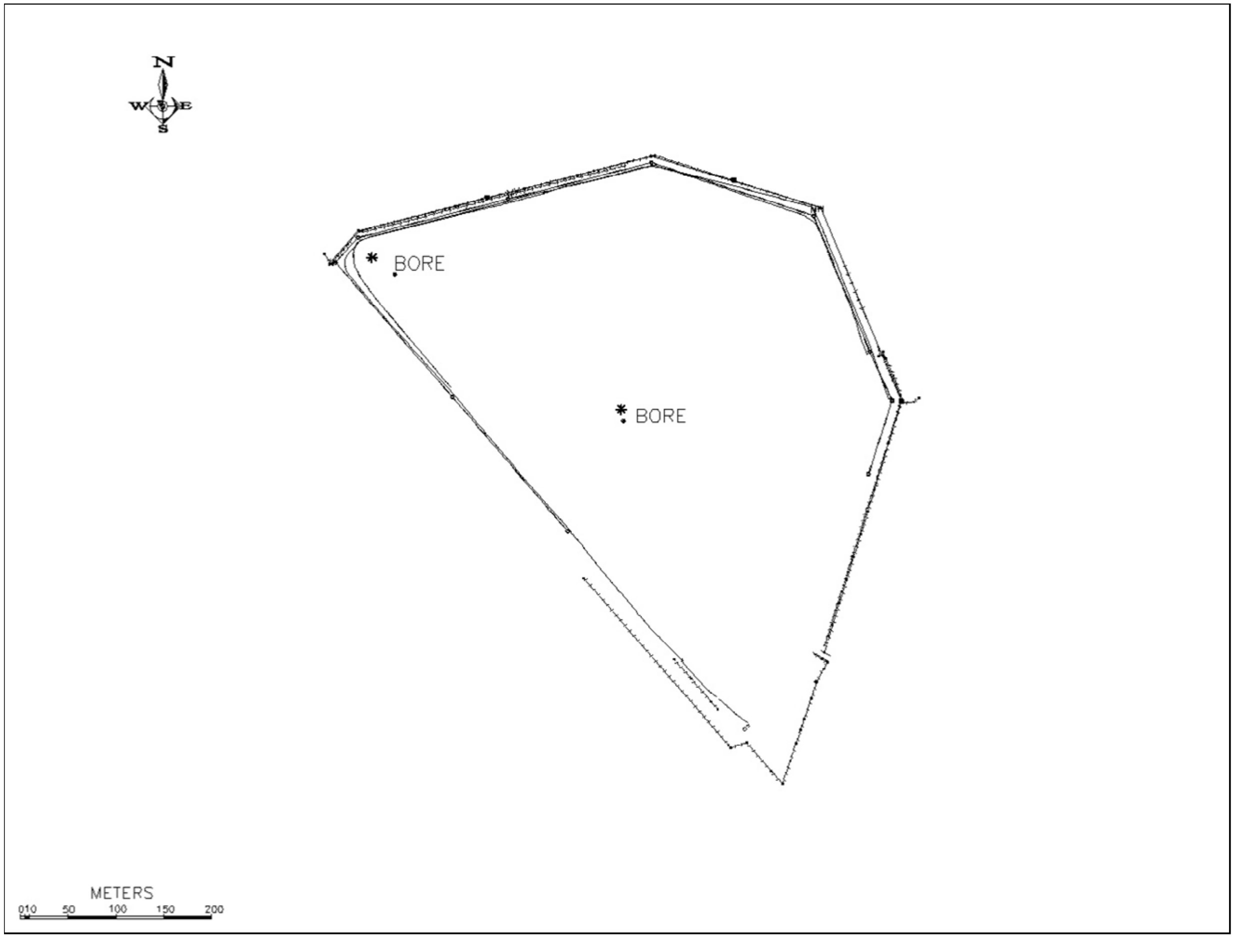


Figure 5.2: Location of Leachate Bores 2 and 6

5.2 Bore Monitoring Data

5.2.1 pH

The leachate pH was quite variable prior to 2002 (Figure 5.3), and became more consistent after the landfill closed in December 2002. Since the closure of the landfill the mean pH has changed little and now averages 7.2 in both bores with a standard deviation of 0.3 in LB2 and 0.1 in LB6

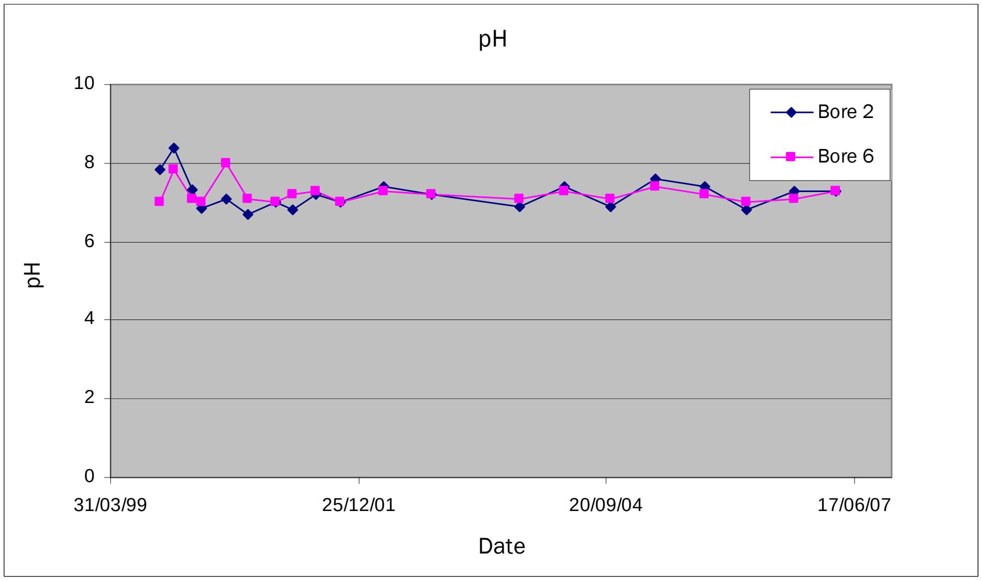


Figure 5.3: pH in LB 2 and LB6.

5.2.2 Electrical Conductivity

Electrical conductivities (Figure 5.4) ranged from 2 750 to 13 370 μScm^{-1} in LB2 and 5 620 to 15 800 μScm^{-1} in LB6. The electrical conductivity (EC) of the leachate in LB6 (mean = 11 100 μScm^{-1}) was generally higher than that of LB2 (mean = 7 600 μScm^{-1}). The EC does not appear to be trending up or down over time.

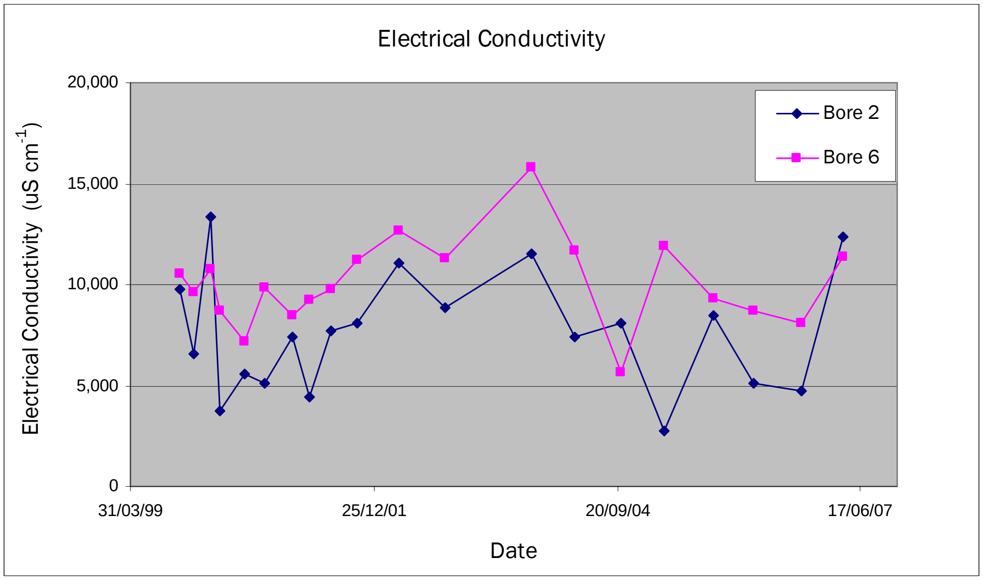


Figure 5.4: Electrical conductivity in LB2 and LB6.

5.2.3 Biological Oxygen Demand and Chemical Oxygen Demand

a. Biological Oxygen Demand

The biological oxygen demand (BOD) ranged from between 13 and 89 g m^{-3} in LB2 and between 22 and 150 g m^{-3} in LB6 (Figure 5.5). The BOD concentrations in both bores appear to be decreasing over time.

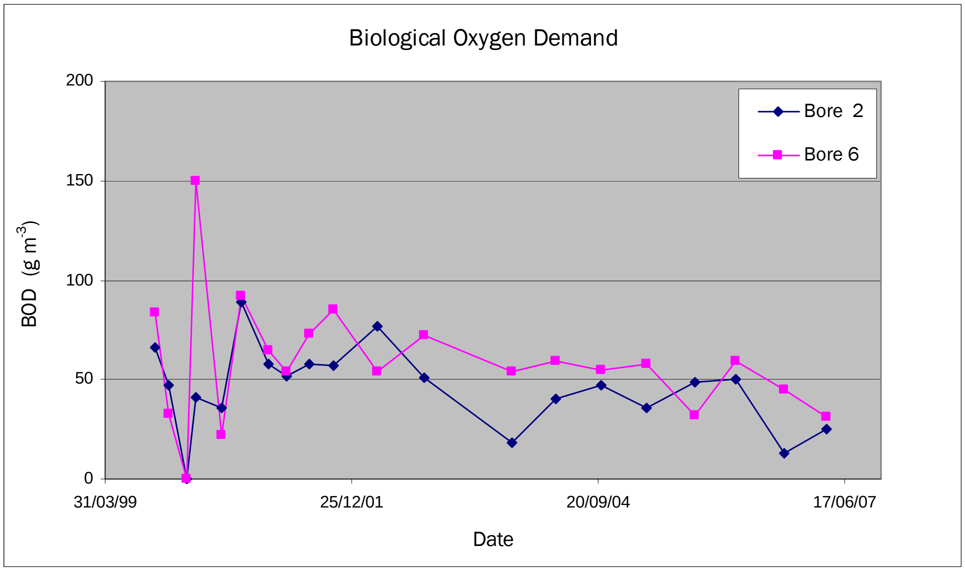


Figure 5.5: Biological oxygen demand concentrations in LB2 and LB6.

b. Chemical Oxygen Demand

Chemical oxygen demand (COD) values ranged between 70 and 1 750 g m^{-3} in LB2 and between 140 and 1 200 g m^{-3} in LB6 (Figure 5.6). The COD concentrations have generally been decreasing over time in both bores.

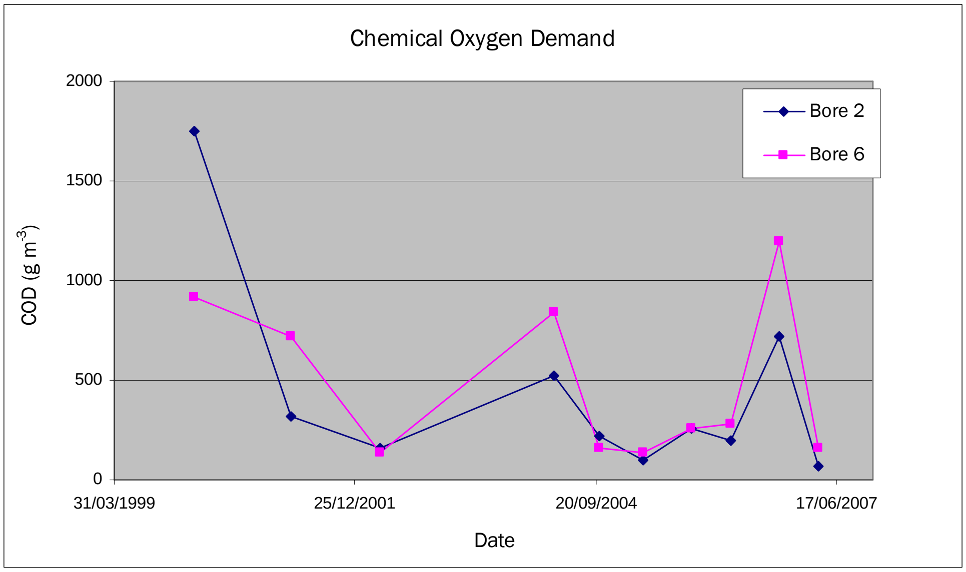


Figure 5.6: Chemical Oxygen Demand concentrations in LB2 and LB6.

5.2.4 Nitrogen

a. Ammonical Nitrogen

The majority of the nitrogen present in the leachate is in the form of ammonical nitrogen. Ammonical nitrogen concentrations ranged from 75 to 770 g m^{-3} in LB2 and 220 to 957 g m^{-3} in LB6 (Figure 5.7). There was no apparent trend in ammonical nitrogen concentrations over time and mean concentrations were 328 g m^{-3} in LB2 and 472 g m^{-3} in LB6.

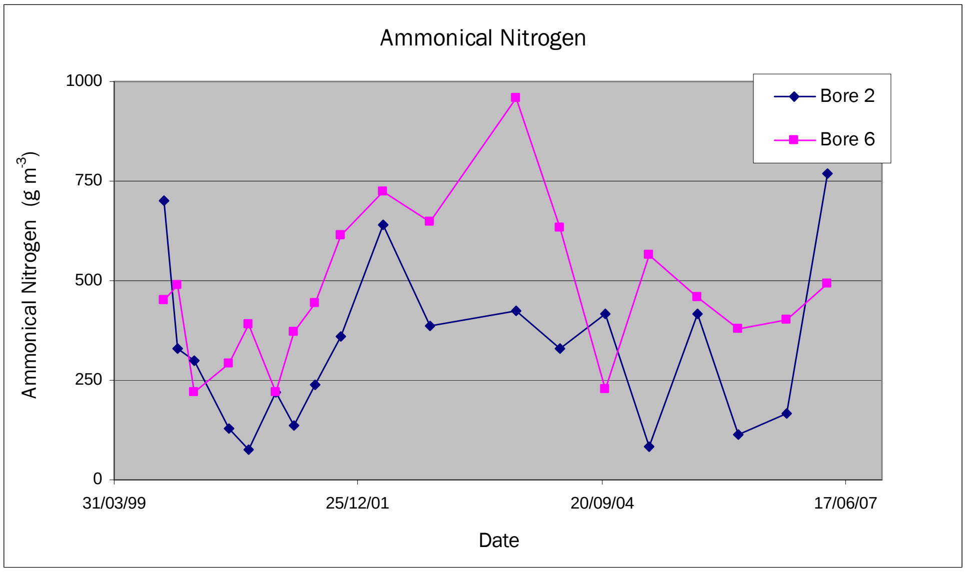


Figure 5.7: Ammonical nitrogen concentrations in LB2 and LB6.

b. Nitrate

Nitrate concentrations (Figure 5.8) ranged from below the detection limit of 0.02 g m^{-3} in both bores to a maximum of 30.3 g m^{-3} in LB2 and 79.9 g m^{-3} in LB6. Since 2005 the nitrate concentrations have been increasing in LB6.

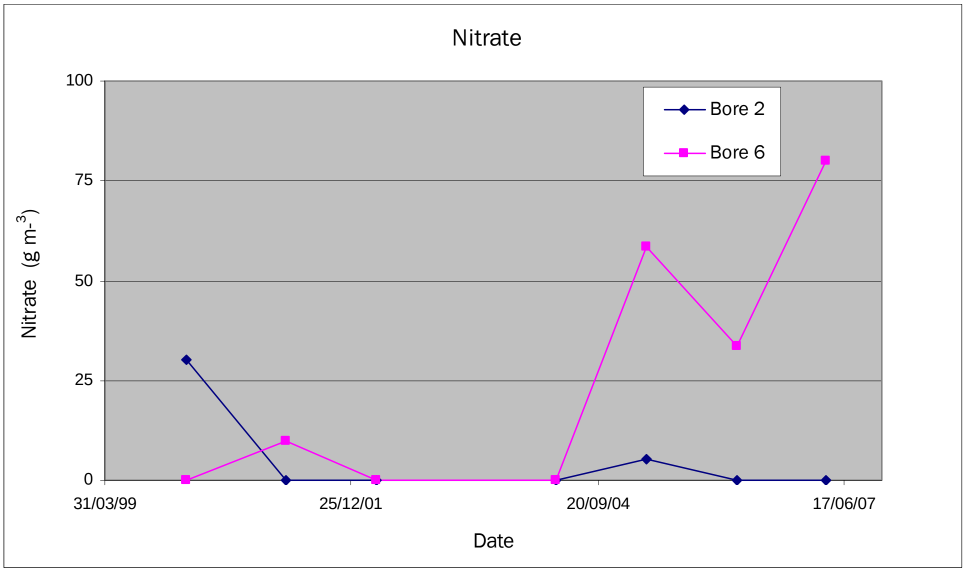


Figure 5.8: Nitrate concentrations in LB2 and LB6.

c. Nitrite

Nitrite concentrations (Figure 5.9) ranged from 0.003 to 3.5 g m^{-3} in LB2 and from 0.009 to 1.35 g m^{-3} in LB6.

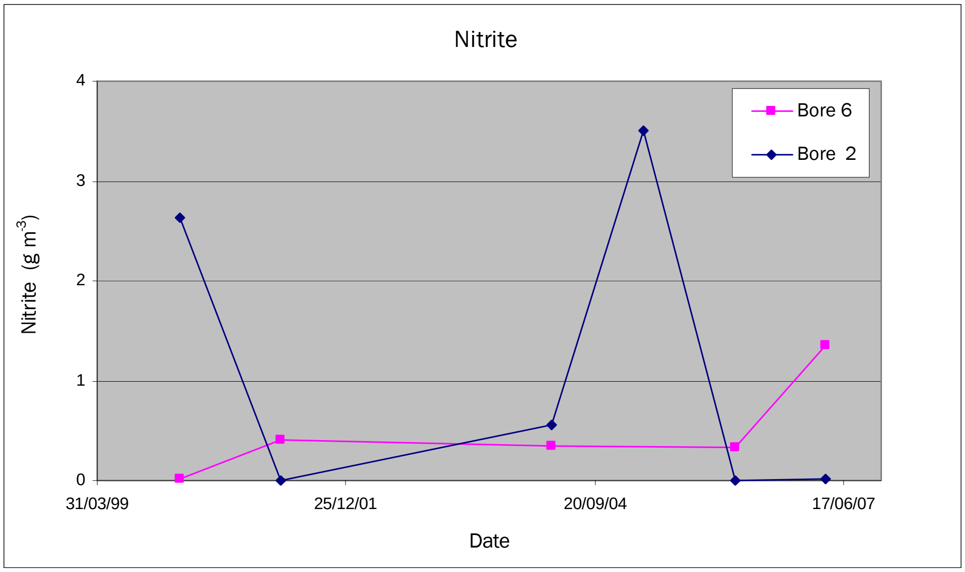


Figure 5.9: Nitrite concentrations in LB2 and LB6.

d. Total Kjeldahl Nitrogen

Total Kjeldahl nitrogen concentrations (Figure 5.10) were in a similar range to those of the ammonical nitrogen and were between 99 – 703 g m^{-3} in LB2 and 347 - 701 g m^{-3} in LB6. The mean concentration was 336 g m^{-3} in LB2 and 532 g m^{-3} in LB6.

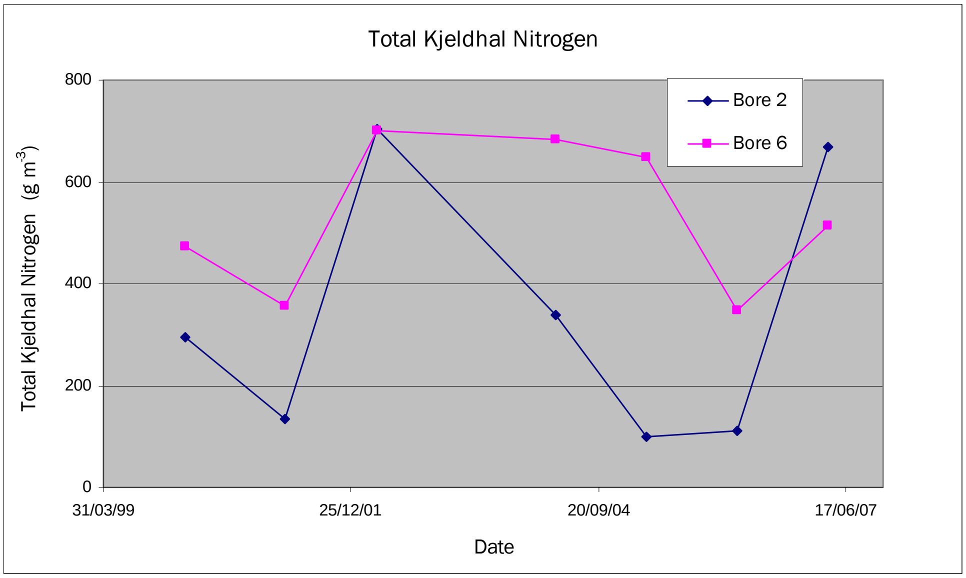


Figure 5.10: Total Kjeldhal Nitrogen concentrations in LB2 and LB6.

5.2.5 Major Cations

a. Sodium

Dissolved sodium concentrations (Figure 5.11) ranged between 229 – 1390 g m^{-3} in LB2 and 773 – 1200 g m^{-3} in LB6. The mean concentration was 743 g m^{-3} in LB2 and 966 g m^{-3} in LB6. The dissolved sodium concentrations do not appear to be decreasing over time.

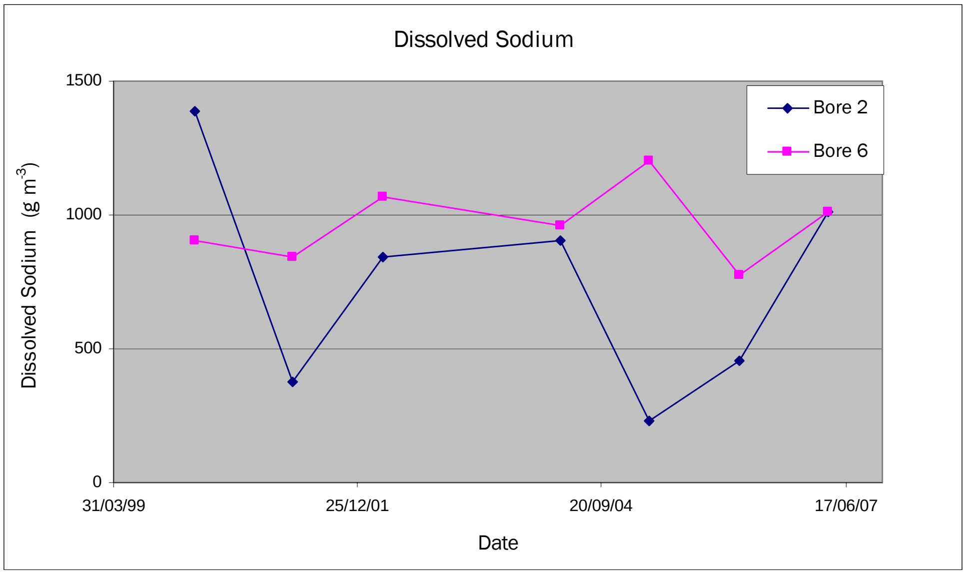


Figure 5.11: Dissolved Sodium concentrations in LB2 and LB6.

b. Potassium

The dissolved potassium concentrations ranged between 141 – 693 g m^{-3} in LB2 and 431 – 776 g m^{-3} in LB6 (Figure 5.12). The mean concentration was 369 g m^{-3} in LB2 and 623 g m^{-3} in LB6.

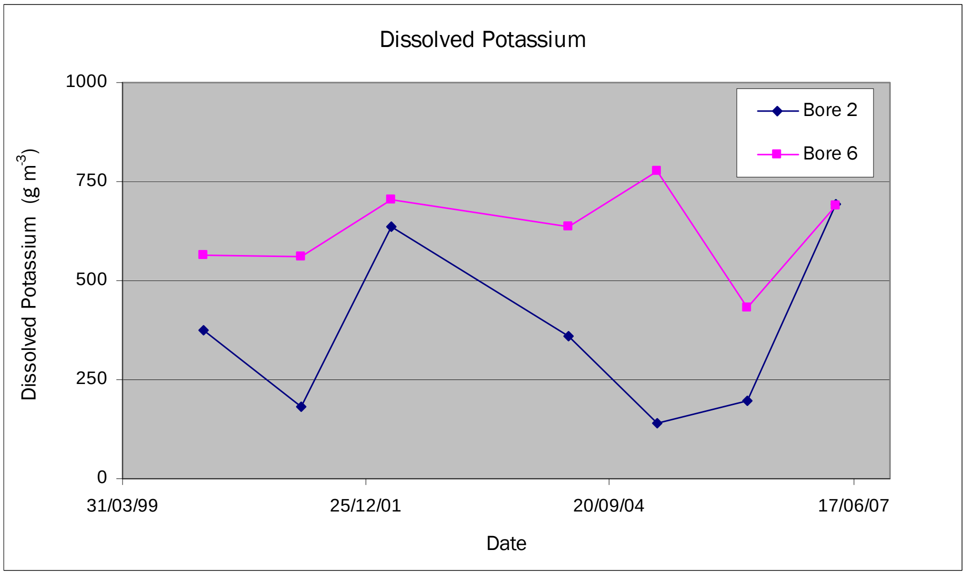


Figure 5.12: Dissolved Potassium concentrations in LB2 and LB6.

c. Calcium

The dissolved calcium concentrations (Figure 5.13) ranged between 104 – 533 g m^{-3} in LB2 and 178 – 323 g m^{-3} in LB6. The mean concentration was 249 g m^{-3} in LB2 and 231 g m^{-3} in LB6.

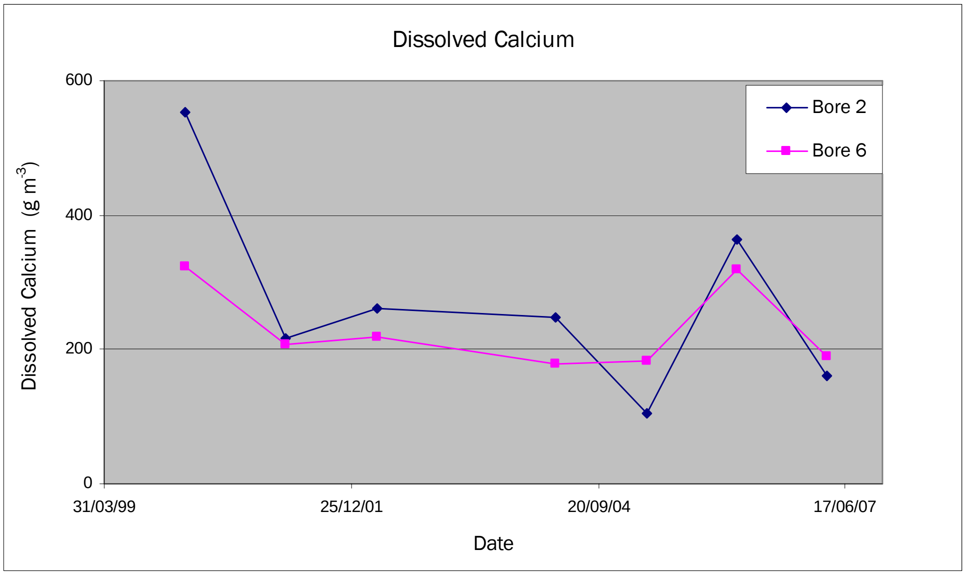


Figure 5.13: Dissolved Calcium concentrations in LB2 and LB6.

d. Magnesium

The dissolved magnesium concentrations (Figure 5.14) ranged between 35 - 238 g m^{-3} in LB2 and 121 – 148 g m^{-3} in LB6. The mean concentration was 120 g m^{-3} in LB2 and 134 g m^{-3} in LB6.

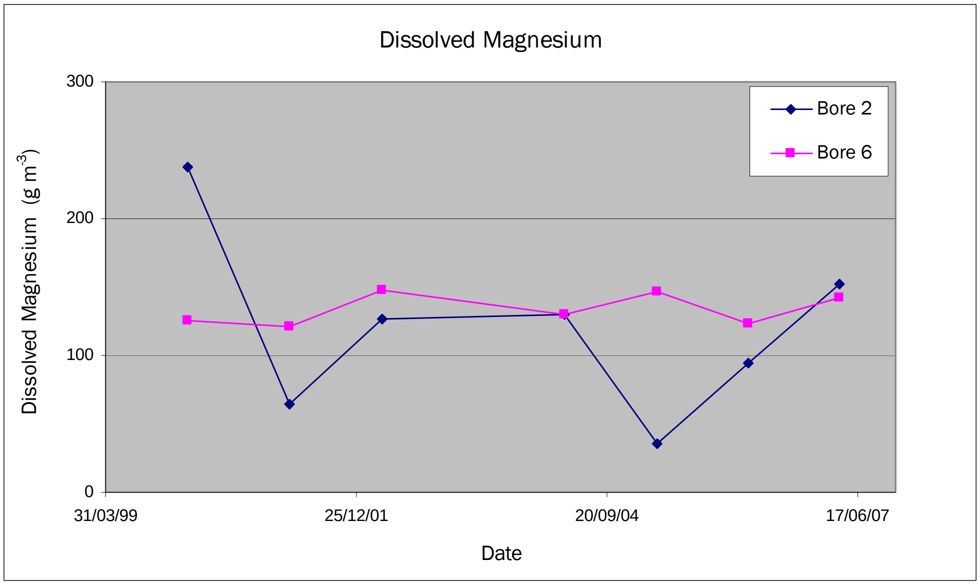


Figure 5.14: Dissolved Magnesium concentrations in LB2 and LB6.

5.2.6 Major Anions

a. Chloride

The chloride concentrations ranged between 330 – 3050 g m^{-3} in LB2 and 1250 – 1550 g m^{-3} in LB6 (Figure 5.15). The chloride concentrations in LB6 have been less variable than those recorded inLB2. The mean chloride concentrations were 1142 g m^{-3} in LB2 and 1319 g m^{-3} in LB6. The April 2007 round of monitoring showed an increase in chloride concentrations in both bores.

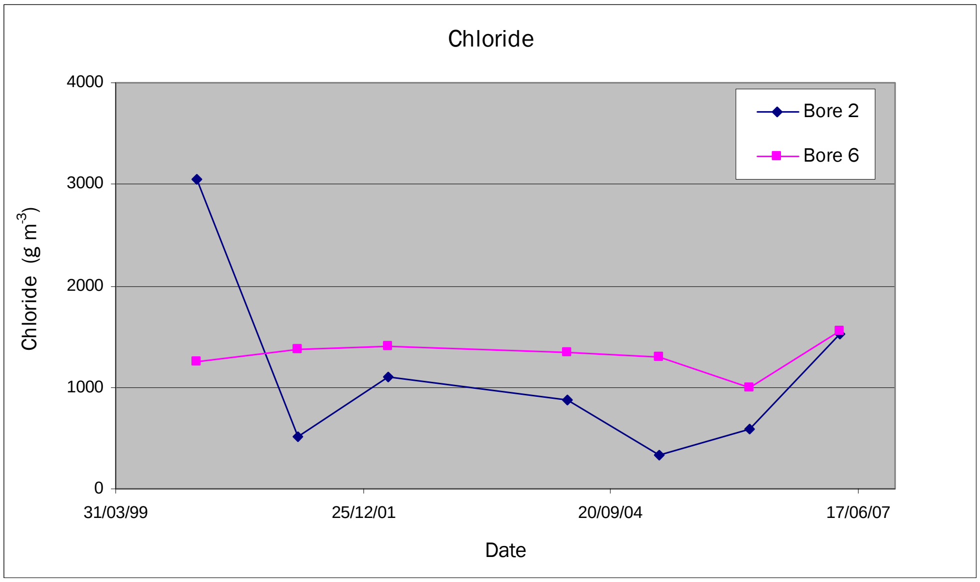


Figure 5.15: Chloride concentrations in LB2 and LB6.

b. Sulphate

The sulphate concentrations ranged between 1.0 - 728 g m^{-3} in LB2 and 1.0 - 216 g m^{-3} in LB6 (Figure 5.16) with a mean of 162 g m^{-3} in LB2 and 51 g m^{-3} in LB6. Both bores showed an increase in leachate concentrations in 2005/06.

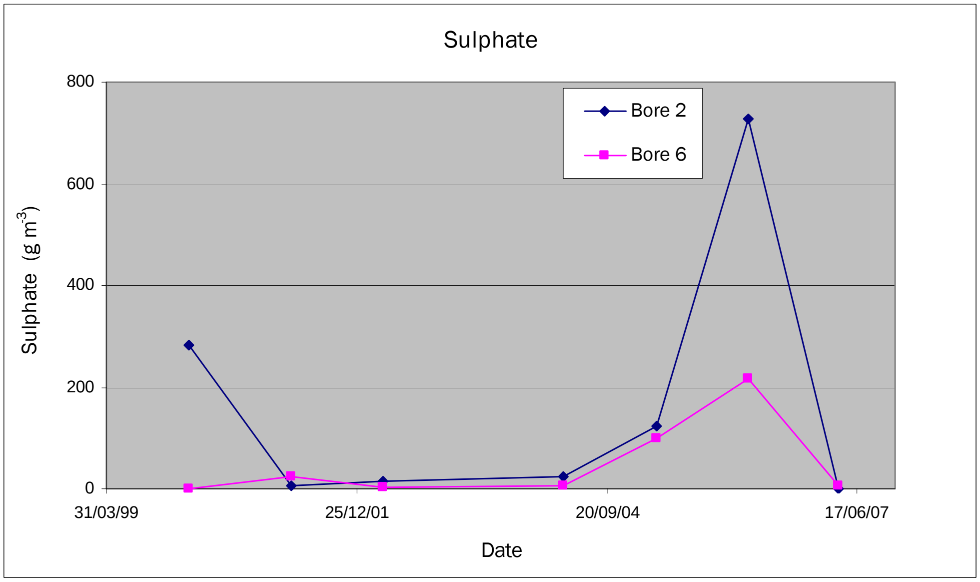


Figure 5.16: Sulphate concentrations in leachate LB2 and LB6.

c. Alkalinity

Alkalinity concentrations ranged between 810 – 5000 g m^{-3} in LB2 and 2300 – 4100 g m^{-3} in LB6 (Figure 5.17) with a mean of 2951 g m^{-3} in LB2 and 3786 g m^{-3} in LB6.

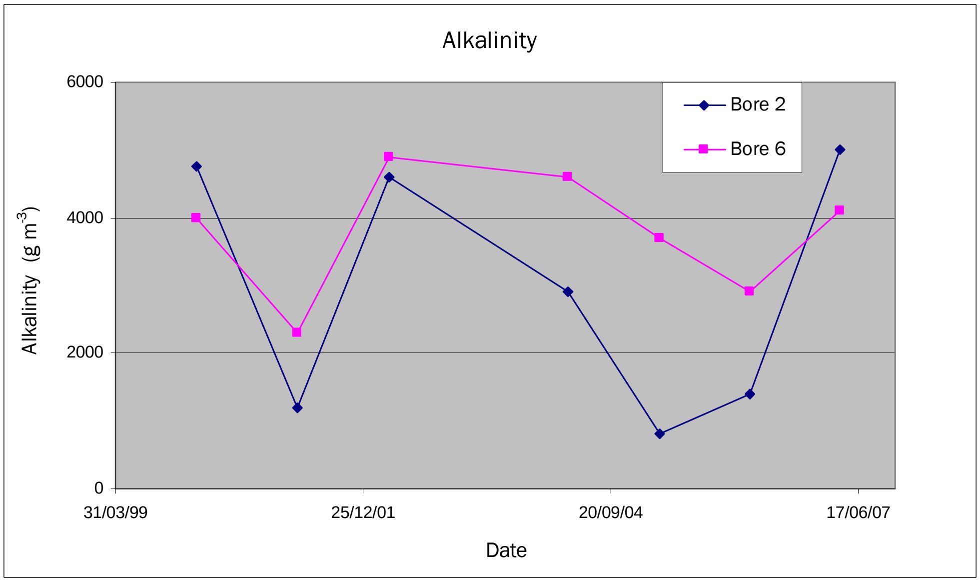


Figure 5.17: Alkalinity in LB2 and LB6.

5.2.7 Metals and Metalloids

a. Zinc

Dissolved zinc concentrations ranged from between 0.01 – 0.142 g m^{-3} in LB2 and 0.01 – 0.521 g m^{-3} in LB6 (Figure 5.18). A large increase in the zinc leachate concentrations was recorded in LB6 in 2005.

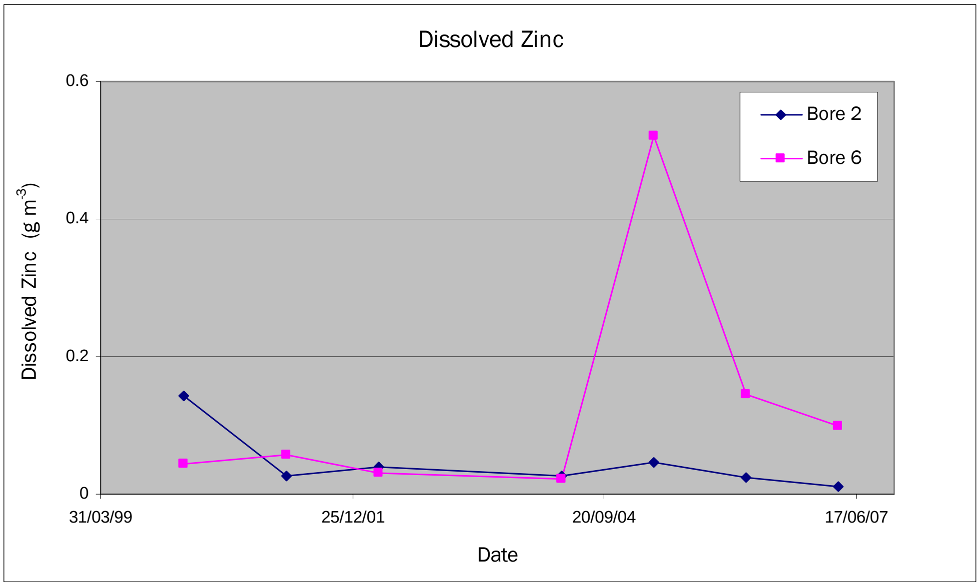


Figure 5.18: Dissolved zinc concentrations in LB2 and LB6.

b. Copper

Dissolved copper concentrations (Figure 5.19) ranged from below 0.005 g m^{-3} in both bores to a maximum of 0.02 g m^{-3} in LB2 and 0.026 g m^{-3} in LB6.

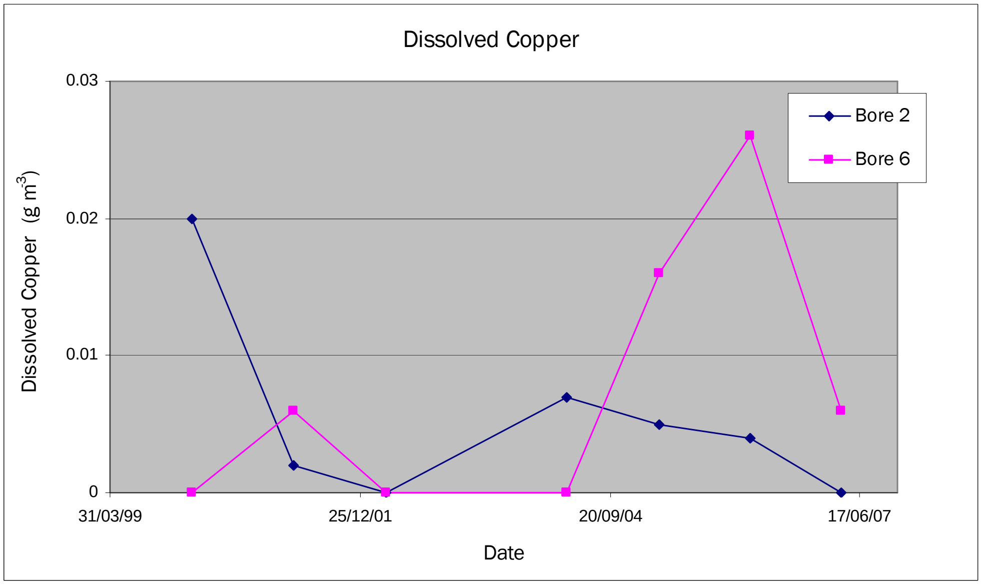


Figure 5.19: Dissolved copper concentrations in LB2 and LB6.

c. Manganese

Dissolved manganese concentrations ranged between 0.41 – 2.45 g m^{-3} in LB2 and 0.66 – 2.06 g m^{-3} in LB6 (Figure 5.20). Since closure the concentrations in LB2 have continued to decrease over time while those in LB6 have been consistently close to the mean value of 0.78 g m^{-3}.

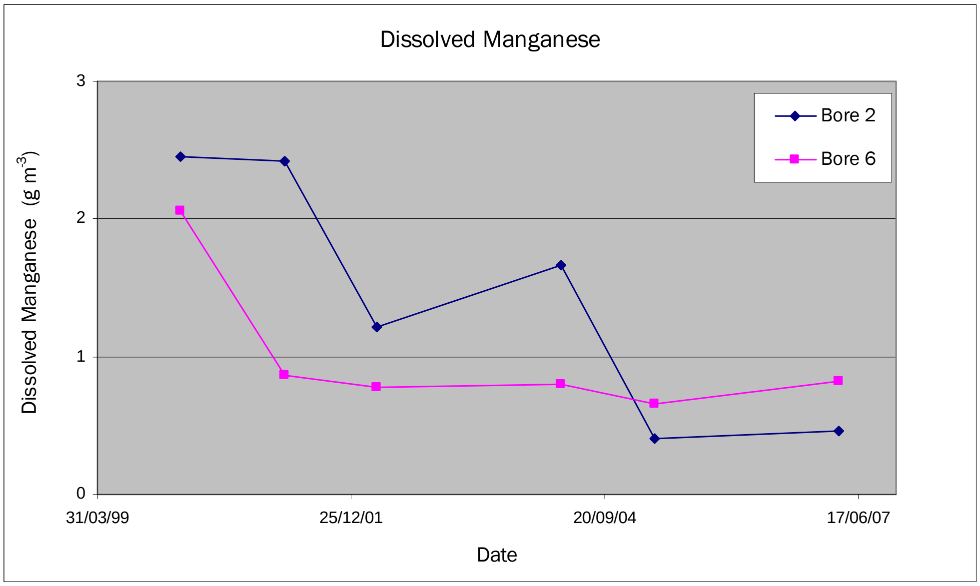


Figure 5.20: Dissolved manganese concentrations LB2 and LB6.

d. Nickel

Dissolved nickel concentrations ranged from between 0.018 – 1.11 g m^{-3} in LB2 and 0.062– 0.096 g m^{-3} in LB6 (Figure 5.21) with a mean of 0.088 g m^{-3} in LB2 and 0.0735 g m^{-3} in LB6.

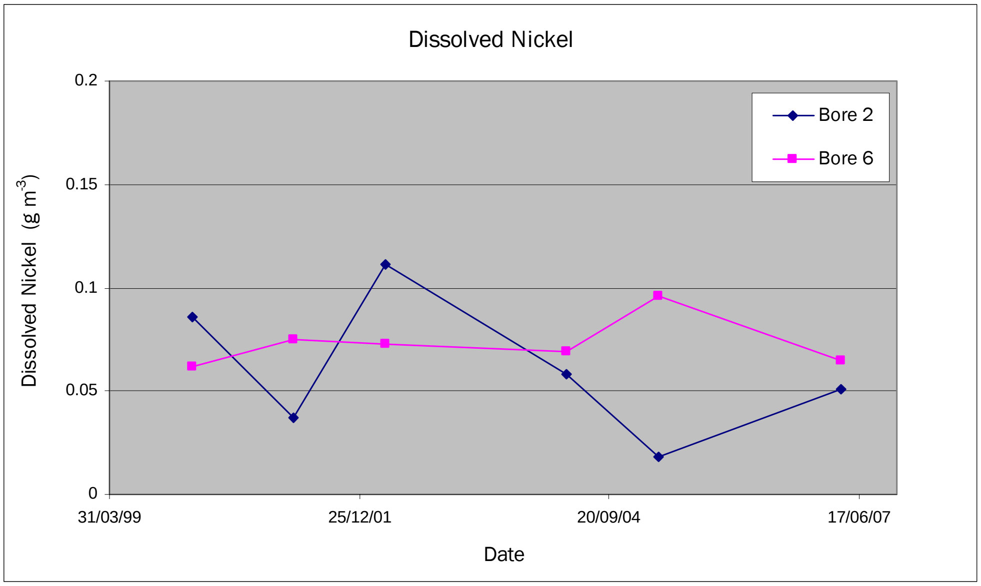


Figure 5.21: Dissolved nickel in LB2 and LB6.

e. Lead

Dissolved lead concentrations ranged from below 0.001 g m^{-3} in both bores to highs of 0.004 g m^{-3} in LB2 and 0.0031 g m^{-3} in LB6 (Figure 5.22).

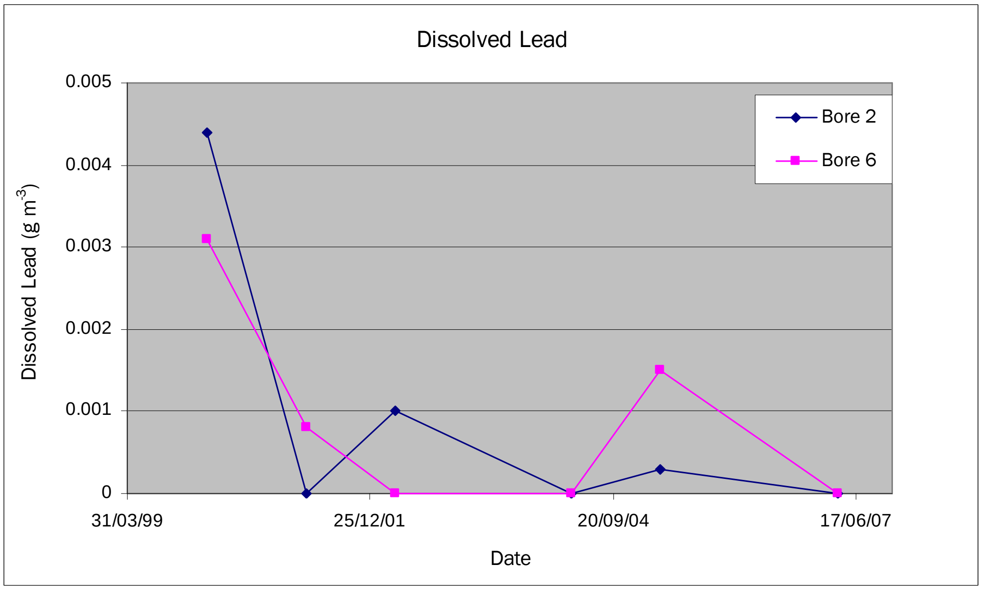


Figure 5.22: Dissolved lead in LB2 and LB6.

f. Chromium

Dissolved chromium concentrations (Figure 5.23) ranged from between 0.001 – 0.040 g m^{-3} in LB2 and 0.016 – 0.036 g m^{-3} in LB6. The mean concentration was 0.016 g m^{-3} in LB2 and 0.021 g m^{-3} in LB6.

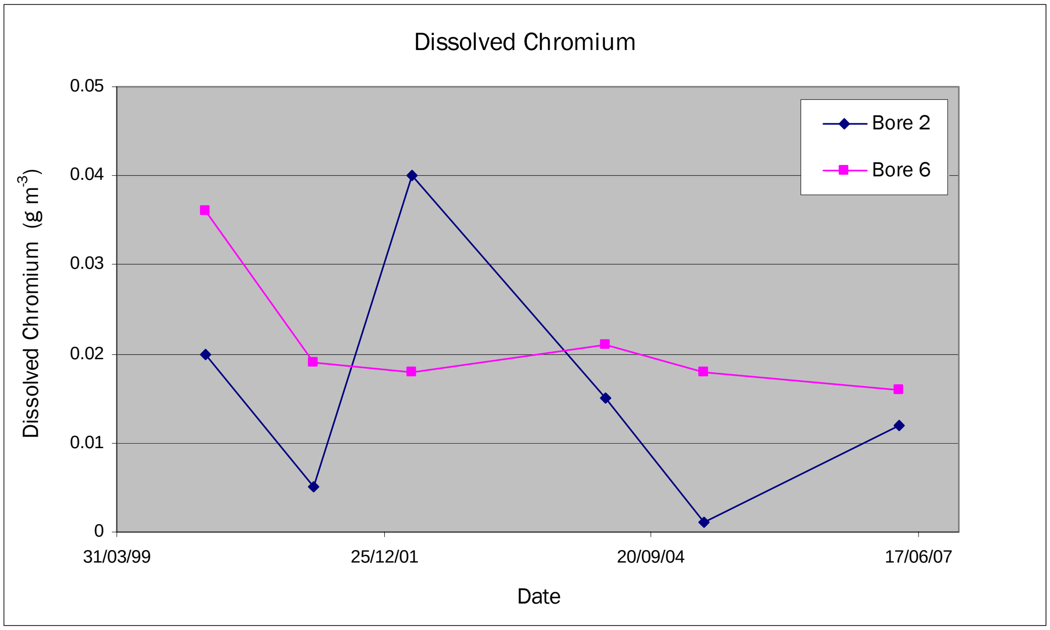


Figure 5.23: Dissolved chromium in LB2 and LB6.

g. Arsenic

Dissolved arsenic concentrations (Figure 5.24) ranged between 0.004 – 0.02 g m^{-3} in LB2 and <0.01 – 0.03 g m^{-3} in LB6.

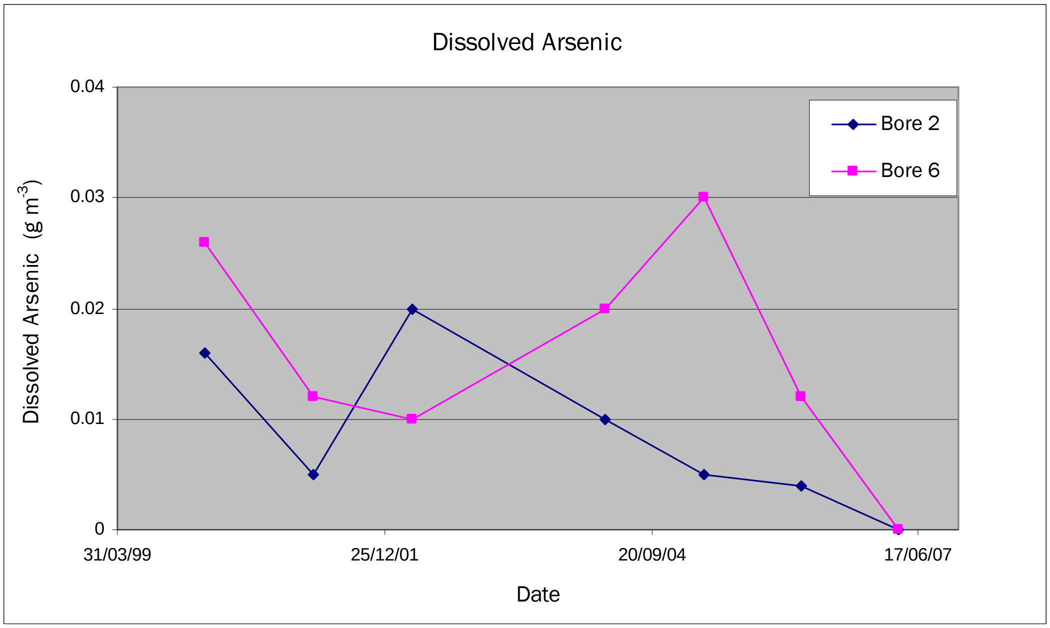


Figure 5.24: Dissolved arsenic in LB2 and LB6.

h. Cadmium, Mercury & Selenium

Dissolved cadmium was only detected once, in LB2 in February 2002. The mercury concentrations in both bores have not exceeded the detection limit of 1×10^{-4} g m^{-3} in any of the samples. The selenium concentrations in both bores have not exceeded the detection limit in any of the samples. The detection limit ranged between 0.005 - 0.01 g m^{-3}.

5.2.8 Other Elements

a. Boron

Dissolved boron concentrations (Figure 5.25) ranged between 0.50 – 1.76 g m^{-3} in LB2 and 1.30 – 2.27 g m^{-3} in LB6. The mean concentration was 1.56 g m^{-3} in LB2 and 1.67 g m^{-3} in LB6. The dissolved boron concentrations do not appear to be decreasing over time.

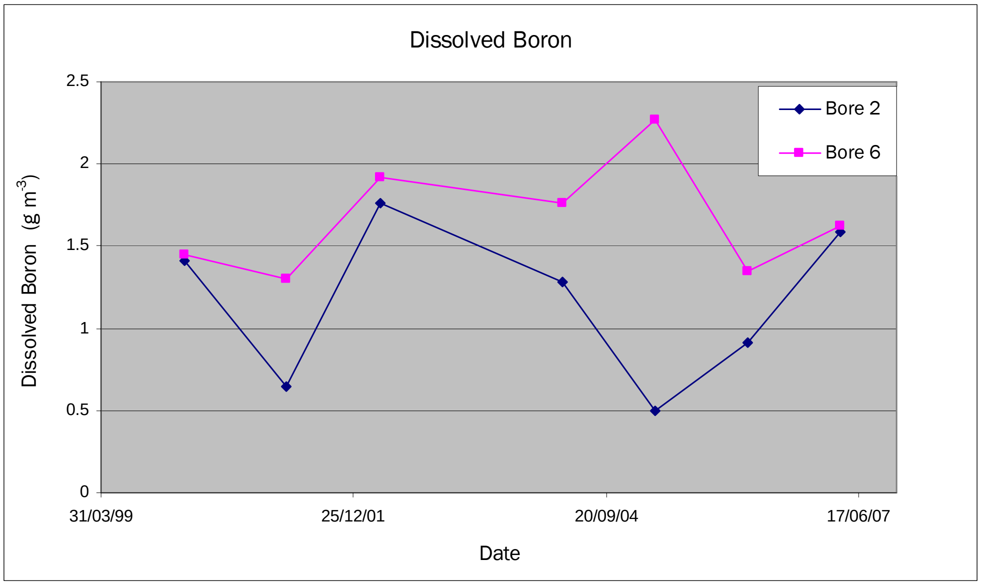


Figure 5.25: Dissolved boron in LB2 and LB6.

b. Phosphorous

The dissolved reactive phosphorous (DRP) concentrations ranged from between 0.021 – 0.50 g m^{-3} in LB2 and 0.016 – 0.32 g m^{-3} in LB6 (Figure 5.26) with a mean of 0.15 g m^{-3} in LB2 and 0.16 g m^{-3} in LB6.

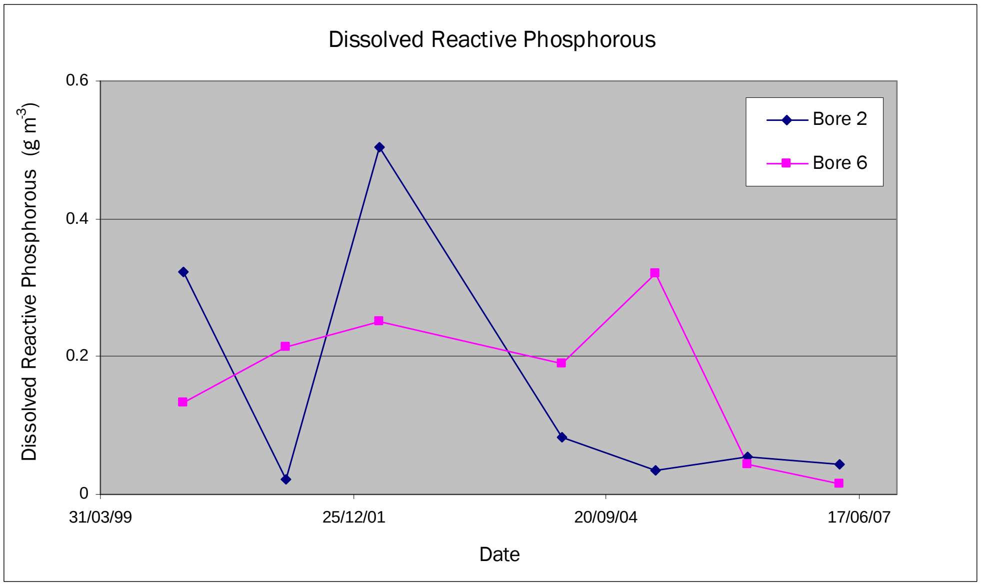


Figure 5.26: Dissolved reactive phosphorous in LB2 and LB6.

5.2.9 Semi Volatile Organic Compounds

The leachate in both bores was tested for a wide range of Semi Volatile Organic Compounds (SVOC) and phenols (Appendix 2). The detection limits varied for each compound but were all less than 0.0004 g m^{-3}. Typically SVOC and phenols concentrations were below the detection limits of the tests; the exceptions are given in Table 5.1. The SVOC compounds can be found in a range of products and have varying effects on humans and other animals. For example naphthalene can be found in a wide range of products, including moth balls, plasterboard, pesticides and dyes and exposure to large amounts can destroy red blood cells and it is a suspected carcinogen.

Table 5.1: Semi Volatile Organic Compound concentrations.

	Bore 2 (g m^{-3})		Bore 6 (g m^{-3})	
Date	Compound	Concentration (g m^{-3})	Compound	Concentration (g m^{-3})
24/02/00	Naphthalene	0.00011	Naphthalene	0.00047
	Di-(2 ethyhexyl) phthalate	0.0022		
05/04/01	2-Metthylnaphthatene	0.00176	1-4 Dichlorbenzene	0.0005
	Dimethyolphenol	0.001		

5.2.10 Comparison between Bores

The monitoring data showed that LB6 generally had higher mean concentrations than those found in LB2 (Figure 5.27, 5.28 & 5.29), the exceptions being calcium, sulphate, iron and manganese. The differences in concentrations between the two bores were generally not statistically significant (P<0.1). The parameters where there was a significant difference were EC (P=0.0045), ammonical nitrogen (P=0.03), BOD (P=0.068), potassium (P=0.036) and boron (P=0.04) with all the elements higher in LB6.

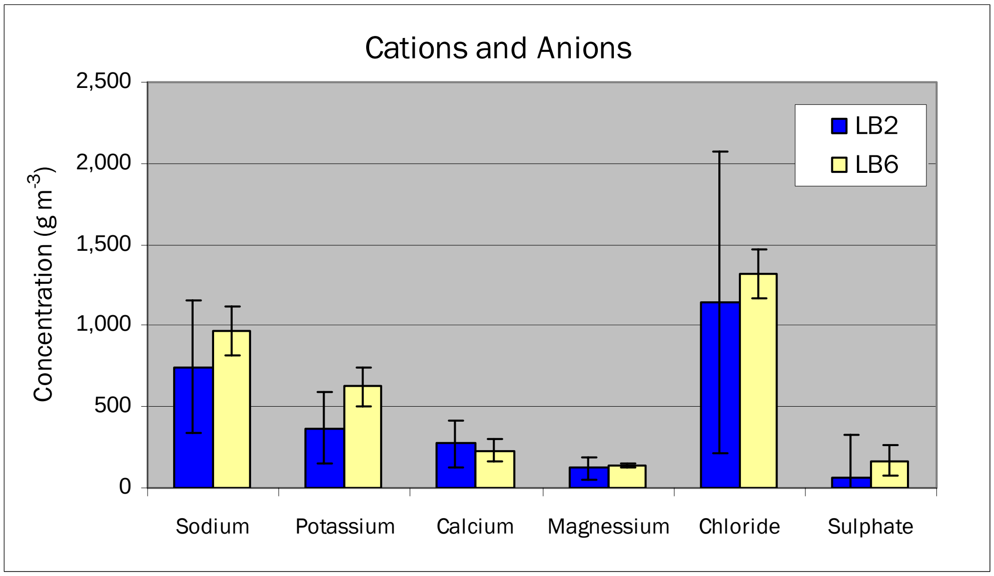


Figure 5.27: Mean cation and anion concentrations in LB2 and LB6 (error bars = 1 standard deviation of the mean).

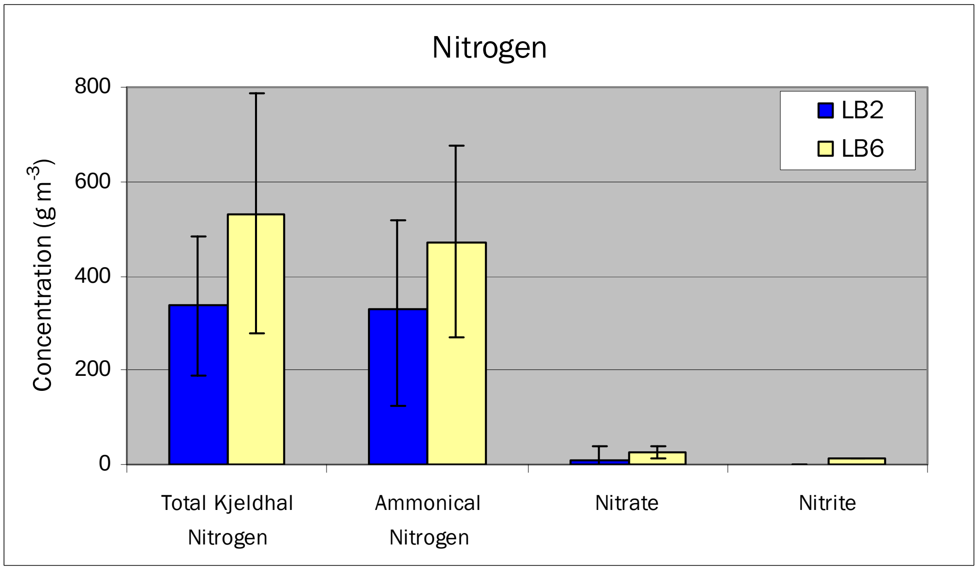


Figure 5.28: Nitrogen concentrations in LB2 and LB6 (error bars = 1 standard deviation of the mean).

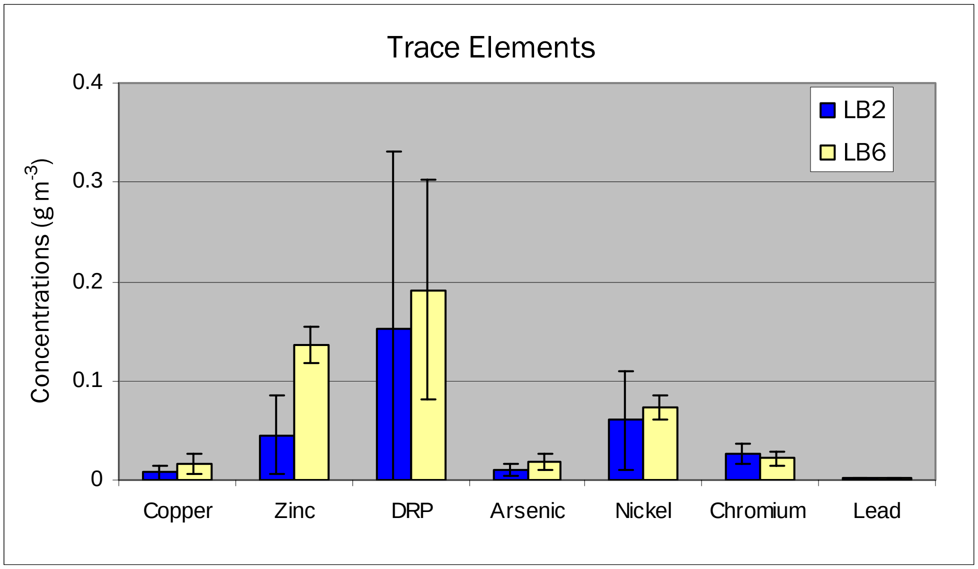


Figure 5.29: Trace elements concentrations in LB2 and LB6 (error bars = 1 standard deviation of the mean).

5.3 Leachate Irrigation Tank Analysis

5.3.1 Introduction

When the review of the monitoring data began there was a large volume of data on the composition of the leachate in LB2 and LB6 but there was no data available on the composition of the leachate in the irrigation tank where the leachate is stored just prior to irrigation. Analysis of the leachate in the irrigation tank was required to determine if there was any variation between the leachate in the bores and the leachate that is irrigated onto the landfill cap.

5.3.2 Sampling Procedure

Three samples were taken from each of LB2 and LB6 and the irrigation tank at the same times over a six week period in November/December 2008 and again in June/July 2009 to allow a comparison between the leachate in the landfill and the leachate irrigated on to the landfill cap. As the initial analysis of the leachate data showed the high salt content of the leachate to be of concern, the leachate sampling was limited to pH, EC, and the soluble salts to keep costs within the allocated budget.

5.3.3 Results

The mean results from the samples taken from LB2 and LB6 and the irrigation tank (Figure 5.30, 5.31 and Table 5.2) showed that there were large variations between the pH, EC and ion concentrations, in the leachate bores and the irrigation tank. The results also showed that the differences in EC and ion concentrations were greater in the winter (June /July) samples compared to the summer samples (November/December).

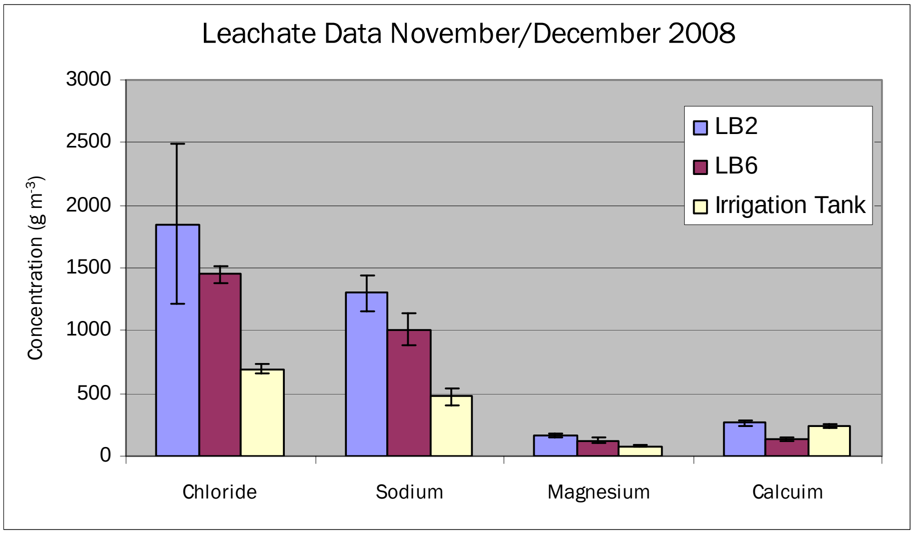


Figure 5.30: Leachate data from LB2 and LB6 and the IrrigationTank, November/December 2008 (error bars = 1 standard deviation of the mean).

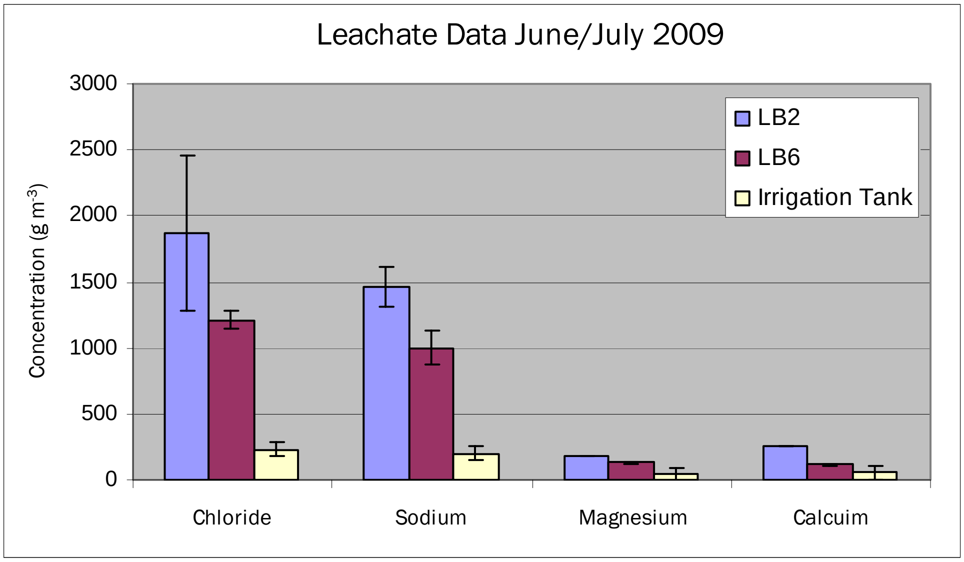


Figure 5.31: Leachate data from bores LB2 and LB6 and the Irrigation Tank, June/July 2009 (error bars = 1 standard deviation of the mean).

Table 5.2: Leachate Data from bores LB2 and LB6 and the Irrigation Tank**.**

	pH	EC (μS cm^{-1})	Cl (g m^{-3})	Na (g m^{-3})	Ca (g m^{-3})	Mg (g m^{-3})
Nov/Dec 2008						
LB 2	7.1	12000	1850	1300	265	165
LB 6	7.3	13350	1450	1010	135	125
Irrigation Tank	7.5	6285	695	475	155	81
June/July 2009						
LB 2	7.1	13033	1867	1467	257	180
LB 6	7.2	13733	1213	1000	113	130
Irrigation Tank	7.5	2310	232	202	57	38

In the June/July samples EC, Cl, Na, Mg and Ca were all significantly higher in LB6 ($P<0.01$) and LB2 ($P<0.1$) compared to the leachate irrigation tank. LB2 also had significantly higher ($P<0.1$) Na, Cl and Ca concentrations compared to LB6.

The November/December sampling round produced similar results to the June/July samples. The EC, Cl, Na, Mg concentrations were all significantly higher ($P<0.1$) in both LB2 and LB6 compared to the leachate irrigation tank.

The EC and ion concentrations were higher in LB6 compared to LB2 but were only significantly different ($P<0.1$) for EC and Ca.

5.3.4. Sodium Adsorption Ratio

The SAR was determined using the mean values for sodium, calcium and magnesium recorded in LB2 and LB6 and the mean of the samples collected from the leachate irrigation tank. SAR is determined to allow the effects of the leachate salinity on the soil structure to be estimated. The SAR was calculated as:

$$SAR = \left[Na^{+}\right] / \left(\left[Ca^{2+}\right] + \left[Mg^{2+}\right]\right)^{\frac{1}{2}}$$

where $[Na^{+}]$, $[Ca^{2+}]$ and $[Mg^{2+}]$ are the concentrations of sodium, calcium and magnesium in mmol l^{-1}. The highest SAR was found in LB2 (Table 5.3) while the lowest SAR was found in the leachate irrigation tank.

Table 5.3: Leachate SAR Data.

	EC μScm^{-1}	Ca ($mmol^{-1}$)	Mg ($mmol^{-1}$)	Na ($mmol^{-1}$)	SAR
Leachate Bore 2	7600	7.0	7.4	60.9	16.1
Leachate Bore 6	11 100	6.2	5.3	47.8	14.1
Irrigation Tank	4317	2.6	2.4	14.7	6.5

5.4 Discussion

5.4.1 Variation between Leachate Bores

As the landfill was not divided into separate engineered cells, samples taken from LB2 and LB6 would have been a mixture of leachate from different areas of the landfill and as such have varying ages and slightly different compositions due to variations in the refuse deposited. Other researchers have reported similar leachate variations within a landfill. For example, significant spatial variations in leachate composition were reported by Assumuth (1992) in a 5 year study of 43 Finnish landfills. Also, Kjeldsen *et al.* (1998) researched the Grindsted Landfill in Denmark and found a pronounced variability in leachate composition in the 31 leachate wells installed through the site.

Leachate produced in the areas of freshly tipped refuse mixing with leachate from the older parts of the landfill and rainwater infiltrating through the active tip face is the likely cause of the variations between bores which was observed while the landfill was open.

5.4.2 pH

Typical pH values for other landfills range from 4.5 -7.5 with a mean of 6.1 for landfills in the acid phase, to 7.5- 9.0 with a mean of 8.0 in the methanogenic phase (Centre for Advanced Engineering, 2000; Kjeldsen *et al.,* 2002) which would put Paokahu Landfill (pH of 7.2) in the late acid to early methanogenic phase. Other literature reviews and studies, (Griffith and Trois, 2006; Heyer & Stegman, 2001; Vavilin *et al.,* 2006) have reported pH to be stable and near neutral during the methanogenic phase. Due to the landfill's age it would be expected that the landfill would be in the methanogenic phase.

The pH does not appear to have increased over time as would be expected as the landfill moves from the acid phase to the methanogenic phase (Griffith & Trois, 2006; Heyer & Stegmann, 2001; Kjeldsen *et al.,* 2002). Monitoring only began at the landfill in 1999, by which time large areas of the landfill should already have been in the methanogenic phase. It is a possibility that the leachate in LB2 and LB6 predominantly came from the larger, older, methanogenic areas of the landfill which would explain why an increase in pH was not observed during the monitoring period.

5.4.3 Electrical Conductivity

The EC gives an estimate of the total dissolved salts present in the leachate. Excessive salt concentrations can affect plants ability to uptake water. Therefore EC can be used to predict if the irrigated leachate is likely to affect the pasture on the landfill cap (Table 5.4).

Table 5.4: Irrigation water ratings based on electrical conductivity (Adapted from ANZECC & ARMCANZ; 2000).

EC µS cm^{-1}	Water Salinity Rating	Plants Suitability (based on salt tolerance)
< 650	Very low	Sensitive
650 – 1300	Low	Moderately sensitive
1300 – 2900	Medium	Moderately tolerant
2900 - 5200	High	Tolerant
5200 – 8100	Very High	Very tolerant
>8100	Extreme	Generally too Saline

(Adapted from ANZECC & ARMCANZ; 2000).

When the mean EC from both leachate bores (9350 µS cm^{-1}) was compared to Table 5.4, the Salinity Rating was very high, meaning only very salt tolerant plants would be unaffected by the high salt content present in the leachate. However, when the leachate from the irrigation tank (4317 µS cm^{-1}) was compared, the Salinity Rating was medium meaning only salt sensitive plants are likely be affected by the salts present in the leachate. The two dominant pasture species are white clover and rye grass. White clover is moderately salt sensitive and is affected by salt when the EC is over 1000 µS cm^{-1} (ANZECC & ARMCANZ; 2000), therefore may be affected by the salts present in the leachate. Rye grasses are generally moderately tolerant to saline conditions (Marcar, 1987), so are unlikely to be affected by the leachate salts.

5.4.4 BOD and COD

As with pH, the BOD and COD concentrations were more variable when the landfill was operational which is likely to be the result of fresh refuse mixing with leachate from the older parts of the landfill and rain infiltration through the open tip face.

The BOD, COD and BOD/COD ratio of the leachate is comparable to that reported in other similar New Zealand landfills (Table 5.5). An average BOD/COD ratio of 0.11 has also been reported by Kjeldsen and Christophersen (2001) in an investigation of 106 old Danish landfills and by Reinhart and Grosh (1998) in a study of Florida landfills. The BOD_5, COD and BOD/COD ratio were all consistent with a landfill in the methanogenic phase.

Table 5.5: Typical landfill BOD, COD and BOD:COD ratio concentrations, (Centre for Advanced Engineering; 2000).

Parameter	Typical Leachate				Horotui[1]	Redvale[2]	Paokahu[3]
	Acid Phase		Methogenic Phase				
	Mean	Range	Mean	Range			
BOD_5 g m^{-3}	13000	4000-40 000	180	20-550	14	50	51
COD g m^{-3}	22000	6000-60 000	3000	500-4500	213	84	457
BOD_5/COD	0.58	_	0.06	_	0.07	0.63	0.11

[1] leachate from a cell closed for more than 10 years
[2] composite leachate from the whole site
[3] mean of Bores LB2 and LB 6

BOD_5 concentrations appear to be decreasing slowly over time in both bores indicating that there is still a small amount of readily biodegradable material present in the landfill.

There was a spike in COD concentrations in the samples taken on the 19 October 2006 observed in both bores. The spike was not associated with a change in pH or any of the other recorded parameters, and is unable to be explained so could be due to a laboratory error as no retesting of samples took place when an unusual result was returned or a laboratory error suspected.

5.4.5 Nitrogen

Ammonical nitrogen (NH_3-N) is a measure of the amount of ammonia present in the leachate. The Paokahu landfill ammonical nitrogen concentrations were within the range expected and were comparable with those recorded in other landfills both internationally and in New Zealand (Table 5.6).

Table 5.6: Typical landfill Ammonical Nitrogen concentrations.

NH_3-N (g m^{-3})	Landfill
328	**Paokahu** – mean of Bore LB 2
472	**Paokahu** – mean of Bore LB 6
67	**Horotiu** - leachate from a cell closed for more than 10 years[1]
178	**Rosedale** - composite leachate from the whole site[1]
428	**Omarunui** [1]
110	Average from 104 year old Danish Landfill[2]
445	Average from 21-30 year old landfill in Germany[3]

Sources: [1] Centre for Advanced Engineering, (2000), [2] Kjeldsen and Chrisophersen (2001), [3] Krumplebeck & Ehrig (1999).

Ammonia is one of the compounds present in leachate whose concentrations do not decrease with time (Chu et al., 1994; Kjeldsen *et al*., 2002; Kruempelbeck and Ehrig, 1999) as there is no mechanism for its degradation under anaerobic conditions. Chu et al. (1994) reported that after a period of 3 to 8 years ammonia concentrations reached a mean value and would remain at this level for at least 50 years. Therefore, it can be expected that the ammonical nitrogen concentrations will be consistently around the mean of 400 g m^{-3} for the next 20 – 50 years. If ammonical nitrogen were to suddenly start to significantly decrease over time, it could indicate that leachate was escaping from the landfill.

On six occasions the recorded TKN concentrations in the leachate bores were lower than the ammonical nitrogen concentrations. This is a physical impossibility as TKN is the sum of ammonical and organic nitrogen concentrations, so must be a laboratory error. TKN concentrations lower than ammonical nitrogen have also been reported by Bone *et al.* (2003) and Griffith and Trois (2006), who were both unable to explain why this occurred.

Analysis of the leachate data revealed that the EC and ammonical nitrogen concentrations followed a similar trend over time (Figures 5.32 and 5.33). Due to the higher mean sodium and chloride concentrations it is unlikely that the ammonical nitrogen concentrations alone were affecting the EC of the leachate.

The relationship is more likely associated with either rainwater or groundwater entering the landfill during rainfall events, diluting the leachate. However, when the EC and ammonical nitrogen concentrations were compared to the rainfall data there doesn't appear to be any clear trend linking them together. When the ammonical nitrogen and EC were compared to other leachate constituents such as chloride and the sodium, it is not possible to see a trend over time as there were insufficient data points. A review of the literature was unable to identify any other reports of a relationship between EC and ammonical nitrogen in landfill leachate.

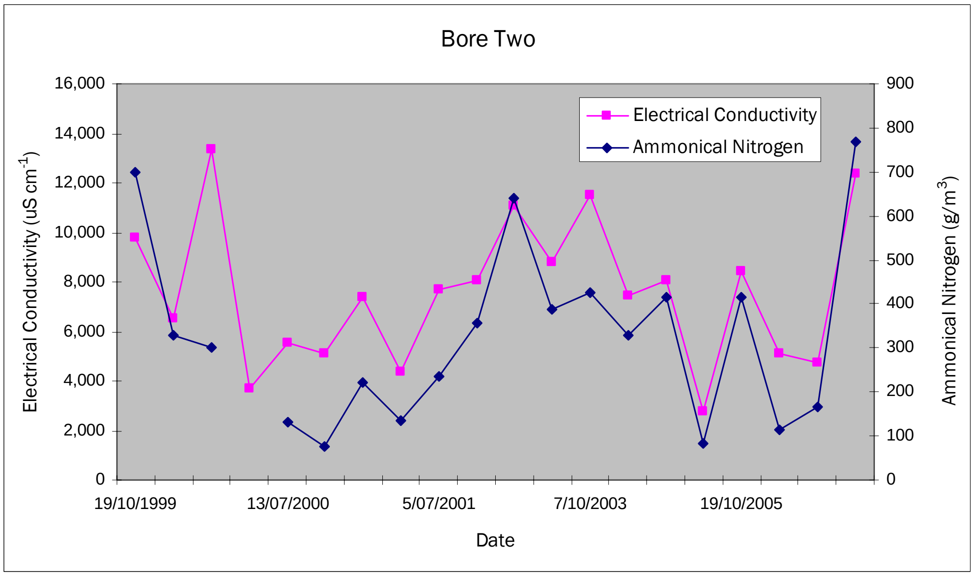


Figure 5.32: EC and ammonical nitrogen concentrations in LB2.

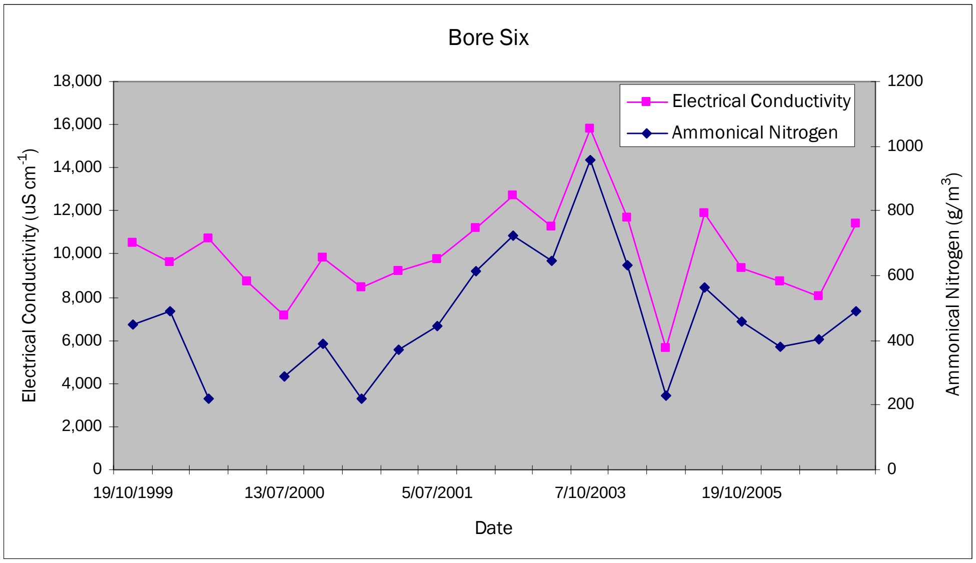


Figure 5.33 EC and ammonical nitrogen concentrations in LB6.

5.4.6 Major Cations

Sodium and potassium concentrations are not influenced by changes in the landfill phase; however, calcium and magnesium concentrations are affected by changes in the landfill phase. The sodium and potassium concentrations were within the ranges reported in other leachates (Table 5.7) but were a lot higher than those reported in similar New Zealand landfills. Calcium and magnesium concentrations were closer to concentrations generally found in a methanogenic phase landfill and were also a lot higher than those reported in similar New Zealand landfills. The most likely source for the high cation concentrations compared to the other New Zealand landfills could be the use of beach sand for daily cover when the landfill was operational and the use of a beach sand layer in the landfill cap construction.

Table 5.7: Typical leachate cation concentrations (Centre for Advanced Engineering; 2000).

Parameter	Typical Leachate				Horotui[1]	Rosedale[2]	Paokahu[3]
(g m^{-1})	Mean		Range				
Na	1350		50-4000		57	222	544
K	1100		10-2500		80	131	469
	Acid Phase		Methanogenic Phase				
	Mean	Range	Mean	Range			
Ca	1200	10 - 2500	60	20 - 600	85	69	240
Mg	470	50 - 1150	180	40 - 350	20	36.5	127

[1] leachate from a cell closed for more than 10 years
[2] composite samples from the whole site prior to discharge
[3] mean of LB2 and LB6

Another possible source of soluble salts could be infiltration of the local groundwater into the landfill. The groundwater around the landfill generally has much a higher EC, Na, K, Ca, and Mg content than that of the leachate. If ground water was infiltrating into the landfill the effects should be more noticeable in winter when the groundwater levels are higher. When the seasonal EC was compared there was no significant difference (LB2, P=0.44; LB6, P=0.9131) between the spring and autumn. However, if summer and winter measurements were available for comparison rather than spring and autumn a seasonal difference might be detectable as the difference should be more pronounced.

a. Sodium

Irrigation water high in sodium relative to the other cations can cause problems in the receiving soil, such as loss of structure and decreased permeability. The potential for sodium to cause problems is expressed in terms of the sodium adsorption ratio and this will be examined later in this chapter.

b. Potassium

Heavy applications of potassium fertilizers may induce magnesium deficiency in grazing animals, called hypomagnesaemia or more commonly ‘grass staggers’ (McLaren & Cameron, 1990). The potassium concentrations recorded in the leachate bores would give potassium loadings of approximately 459 kg/ha/yr, which is high enough to cause grass staggers. However, the analysis of the leachate in the irrigation tank showed that the actual potassium loading rates were likely to be lower than 459 kg/ha/yr.

5.4.7 Major Anions

Chloride and alkalinity ($CaCO_3$) concentrations are not influenced by changes in the landfill phase, while sulphate concentrations are influenced by changes in the landfill phase. The chloride and alkalinity concentrations were within the ranges reported in other leachates (Table 5.8) but were a lot higher than those reported in similar New Zealand landfills. Sulphate concentrations were more representative of the lower concentrations found in a methanogenic phase landfill but were higher than those reported in similar New Zealand landfills. One possible reason for the high anion concentrations compared to the other New Zealand landfills, may be the use of beach sand for daily cover, infiltration of saltwater influenced groundwater or salt inputs into the landfill from sea spray drift.

Table 5.8: Typical leachate anion concentrations (after, Centre for Advanced Engineering; 2000).

Parameter	Typical Leachate				Horotui[1]	Rosedale[2]	Paokahu[3]
(g m^{-1})	Mean		Range				
Cl	2100		100-5000		74.9	294	1231
Alkalinity	6700		300-11500		264	1260	3369
	Acid Phase		Methanogenic Phase				
	Mean	Range	Mean	Range			
SO_4	500	70-1750	80	10-420	1	19	107

[1] leachate from a cell closed for more than 10 years
[2] composite samples from the whole site prior to discharge
[3] mean of LB2 and LB6

The high chloride concentrations of the leachate have the potential to cause problems when irrigated onto the landfill cap, as high levels of chloride in the soil solution can lead to reduced plant growth due to the osmotic effect, in a similar way to which high sodium concentrations can affect plant growth (ANZECC & ARMCANZ, 2000).

High chloride concentrations in irrigation waters can also cause an increase in the uptake of cadmium by plants. Chloride forms a series of complexes with cadmium when there are high chloride concentrations in the soil solution. As the chloride concentrations increase above 400 g m^{-3}, $CdCl^+$ becomes more abundant in solution than Cd^{2+}. Cadmium-chloride complexes raise total cadmium concentrations in the soil solution and can lead to enhanced cadmium uptake by plants (ANZECC & ARMCANZ, 2000; Mc Laughlin *et al*., 1997). For example, Mc Laughlin *et al*. (1994) found a good relationship (R^2= 0.65) between cadmium concentrations in potato tubers and water-extractable chloride in a wide range of soils. The cadmium concentrations in the Paokahu leachate were low so enhanced cadmium uptake by plants caused by high chloride concentrations is unlikely to occur.

5.4.8 Heavy Metals and Metalloids

The leachate heavy metal concentrations were low (Table 5.9) which is the same as reported in other landfill studies (Ehrig, 1983; Krug & Ham, 1995 & Qu *et al.* 2008). The Paokahu heavy metal concentrations were all at the lower end of the concentrations reported in Table 5.8, which may be attributed to the high organic matter content from the vegetable processing waste deposited in the landfill. Organic matter present in the landfill can absorb significant amounts of metals, reducing their solubility (Kjeldsen *et al.* 2002). The mean pH of 7.2 will also aid the adsorption of heavy metals as the adsorption process occurs more readily at neutral to high pH values (Taulis, 2005). The sulphate present in the leachate will also contribute to the relatively low metal concentrations as metal-sulphite precipitates form readily during the methanogenic phase.

Table 5.9: Typical leachate metal and metalloid concentrations (after Centre for Advanced Engineering, 2000).

Parameter	Typical Leachate		Horotui[1]	Rosedale[2]	Paokahu[3]
(g m^{-1})	Mean	Range			
Arsenic	0.16	0.005-1.6	0.006	0.0073	0.012
Copper	0.08	0.004 -1.4	-	-	0.07
Nickel	0.2	0.02 – 2.05	0.012	0.034	0.08
Lead	0.09	0.008 – 1.02	0.001	<0.02	0.0008
Chromium	0.3	0.03 – 1.6	0.012	0.015	0.06
Cadmium	0.006	0.0005 - 0.14	<0.001	<0.005	<0.0004
Mercury	0.01	0.0002 - 0.05	-	-	<0.0004
Zinc	0.6*	0.03 - 4.0*	0.015	0.145	0.08
Manganese	0.7*	0.03 – 45*	0.67	-	1.4
Selenium	-	-	-	-	<0.01

*Methanogenic phase leachate

[1] leachate from a cell closed for more than 10 years

[2] composite samples from the whole site prior to discharge

[3] mean of LB2 and LB6

When the Paokahu leachate was compared to the recommended metal concentrations for irrigation waters (Table 5.10) only manganese is above the long-term (up to 100 years) recommended levels (ANZECC & ARMCANZ; 2000).

Manganese in high concentrations can be toxic to plants, with solution concentrations as low as 0.75 g m^{-3} being toxic to plants (ANZECC & ARMCANZ; 2000). The concentration of the manganese in the irrigated leachate is likely to be lower than 1.2 g m^{-3} as the leachate in the irrigation tank had lower concentrations for the elements tested compared to the leachate bores.

Table 5.10: Maximum short and long-term recommended irrigation levels (after ANZECC & ARMCANZ, 2000).

Parameter ($g\ m^{-3}$)	Maximum recommended short-term irrigation concentrations[1]	Maximum recommended long-term irrigation concentrations[2]	Paokahu Landfill Leachate Bores
Arsenic	0.2	0.1	0.012
Cadmium	0.05	0.01	<0.0004
Chromium	1.0	0.1	0.06
Copper	5.0	0.2	0.07
Lead	5.0	2.0	0.0008
Manganese	10	0.2	1.2
Mercury	0.002	0.002	<0.0004
Nickel	0.2	2.0	0.08
Selenium	0.05	0.02	<0.01
Zinc	5.0	2.0	0.08

[1] Up to 20 years

[2] Up to 100 years

Soil pH strongly influences manganese availability to plants. When the pH is above 5.0-5.5 manganese is adsorbed onto the soil organic matter and is less available to plants (El-Jaoual & Cox, 1998). In the literature there are several examples of manganese toxicity being influenced by soil pH. Vega et al (1992) found that a combination of low pH and high application rates of sewage sludge caused severe manganese toxicity in cow pea (*Vigna unguiculata).* Hue *et al.* (2001) found that for each unit increase in pH there was a 100 fold decrease in manganese concentration in an Oxisol soil and that a combination of gypsum and lime was effective at correcting manganese toxicity in soybean. Davis (1996) reported that increasing the soil pH reduced the toxicity rating in peanut plants and application of lime was the most practical method of increasing the soil pH.

High calcium availability has also been shown to reduce manganese toxicity in plants. High Ca^{2+} competes with Mn^{2+}, so higher calcium concentrations reduce manganese uptake by plants and high plant calcium contents also increase plant tolerances to manganese (El-Jaoual & Cox, 1998; Hue *et al.*, 2001). Hue *et al.* (2001) found that manganese toxicity was correlated to the leaf Ca/Mn ratio and that a Ca/Mn ratio $\geq$ 50:1 was required to prevent toxicity. The calcium content of the leachate (240 g m^{-3}) was also considerably higher than that of the manganese (1.2 g m^{-3}) so the high Ca:Mn ratio present in the leachate should reduce the risk of manganese toxicity occurring.

5.4.9 Trace Elements

a. Boron

Mean boron concentrations were within the range of those found at similar New Zealand landfills (Table 5.11), but were above the ANZECC & ARMCANZ (2000) long-term guideline.

Table 5.11: Typical leachate boron concentrations (after ANZECC & ARMCANZ, 2000 and Centre for Advanced Engineering, 2000).

Parameter (g m^{-3})	Horotui[1]	Rosedale[2]	Paokahu[3]	ANZECC & ARMCANZ long-term Guideline
Boron	0.52	2.1	1.62	0.5

[1] leachate from a cell closed for more than 10 years
[2] composite samples from the whole site prior to discharge
[3] mean of LB2 and LB6

The main concern associated with high boron concentrations is that it can be toxic to plants. The ANZECC & ARMCANZ (2000) long-term guideline is set to protect the most boron sensitive species such as blackberry and lemons. Justin & Zupancic (2007) investigated the effects of leachate boron irrigated onto a landfill cap planted in a mixture of grass and white clover (*Trifolium repens*). Boron concentrations ranged from 0.8-3.83 g m^{-3} and no toxic effects were observed in the leaves of the pasture species although the soil boron concentrations increased over the 10 month trial period. Therefore, the grass species at Poakahu landfill are unlikely to be influenced by boron toxicity.

b. Phosphorous

The dissolved reactive phosphorous concentrations (DRP) were higher than those reported in other similar New Zealand landfills (Table 5.12). High DRP concentrations shouldn't cause any problems as long as the leachate is contained within the landfill and doesn't enter the surrounding waterways.

Table 5.12: Typical leachate Dissolved Reactive Phosphorous concentrations (after Centre for Advanced Engineering, 2000).

Parameter (g m^{-3})	Horotui[1]	Rosedale[2]	Paokahu[3]
Dissolved Reactive Phosphorous	0.012	0.040	0.155

[1] leachate from a cell closed for more than 10 years
[2] composite samples from the whole site prior to discharge
[3] mean of LB2 and LB6

5.4.10 Semi Volatile Organic Compounds

The concentrations of all semi volatile organic compounds and phenols dropped below the detection limit since the landfill closed and have not been detected since. The concentrations of semi volatile organic compounds and phenols decrease over time in landfill leachate (Kjeldsen *et al.* 2002) as the compounds degrade. Therefore the already very low levels of semi volatile organic compounds and phenols should continue to diminish making it unlikely that they will affect the pasture and grazing animals on the landfill cap.

5.4.11 Leachate Irrigation Tank

The samples collected from the leachate irrigation tank in June/July had lower concentrations than the November/December samples with EC, chloride, and calcium values being significantly lower ($P<0.1$) in the June/July samples compared to the November/December. The seasonal variation in the leachate concentrations was likely to be caused by rainfall, with 47 mm falling in November/December compared to 199 mm in June/July. The influence of rainfall on leachate irrigation rates can also be seen by comparing the leachate irrigation rates to rainfall over time, which shows that times of high rainfall coincide with times of high leachate irrigation (Figure 5.34).

When the June/July samples were compared to the November/December samples for LB2 and LB6, the only parameters in which there was a statistically significant difference were the EC in LB2 ($P=0.02$) and calcium in LB6 ($P= 0.03$). The similarity in concentrations between LB2 and LB6 indicates that any seasonal variations in leachate concentrations were much less pronounced in the centre of the landfill compared to the leachate irrigation tank.

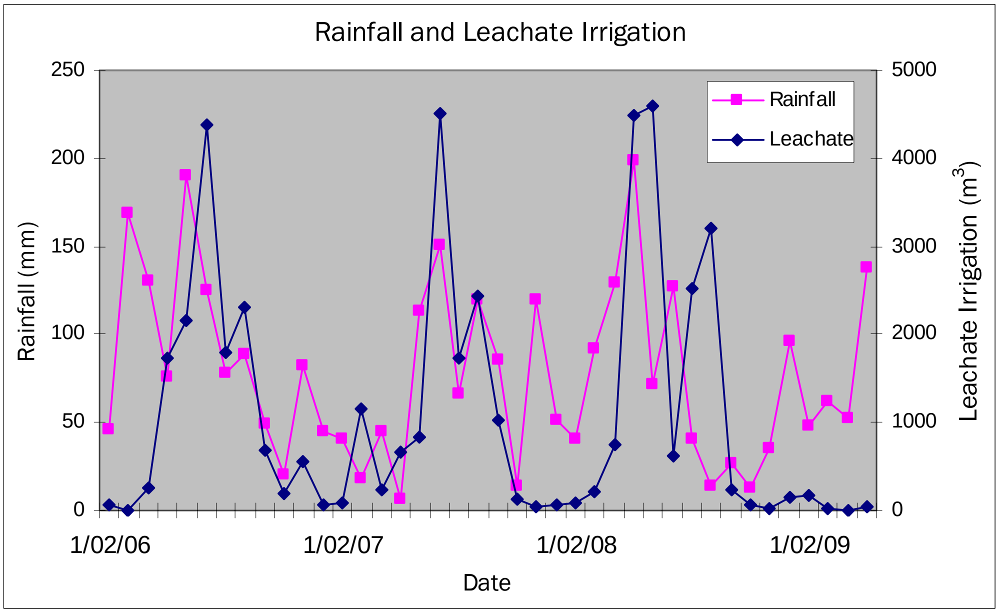


Figure 5.34: Rainfall and leachate irrigation data.

When the samples taken from LB2 and LB6 were compared to the samples taken from the Irrigation Tank they show the leachate also undergoes a reduction in strength between the centre of the landfill towards and the leachate irrigation tank. The highest EC and ion concentrations were observed in LB2 which is located approximately in the centre of the landfill. The EC and ion concentrations decreased in LB6 which is located approximately 40m from the leachate collection drain and decreases still further in the leachate collection drain.

The reduction in leachate strength that occured between the leachate bores and the irrigation tank was most likely caused by water infiltrating the landfill diluting the leachate. Water could be entering the landfill through:

- infiltration of rainwater through the landfill cap,
- a raised ground water level,
- surface water runoff from the cap infiltrating into the leachate collection system, at the outside edge of the landfill,
- a combination of some or all of the above processes.

The pattern shown in the November/December 08 and June/July 09 of the leachate in LB2 having the highest concentrations is the opposite of the long term trend shown in the monitoring data where LB6 generally had higher constituent concentrations than those found in LB2. LB6 is located in the northwest corner of the landfill which was the site of the active tip face from approximately 1999 to 2001 and the last area in which refuse was buried in any great volumes. Therefore, the leachate in the area of LB 6 will be younger compared to LB 2 and is the likely reason for the higher mean concentrations over time.

As the majority of the leachate constituents had lower concentrations in the leachate tank than in the leachate bores, the historical data from the leachate bores alone shouldn't be used to calculate the leachate irrigation nutrient loading rates. Use of the leachate bore data can only be used to give an indication of nutrient loading and will result in an over estimation of the actual loading rates. Future routine monitoring should be altered to include the leachate irrigation tank to enable a more accurate estimate of the leachate constituent loading rates.

5.4.12 Sodium Adsorption Ratio

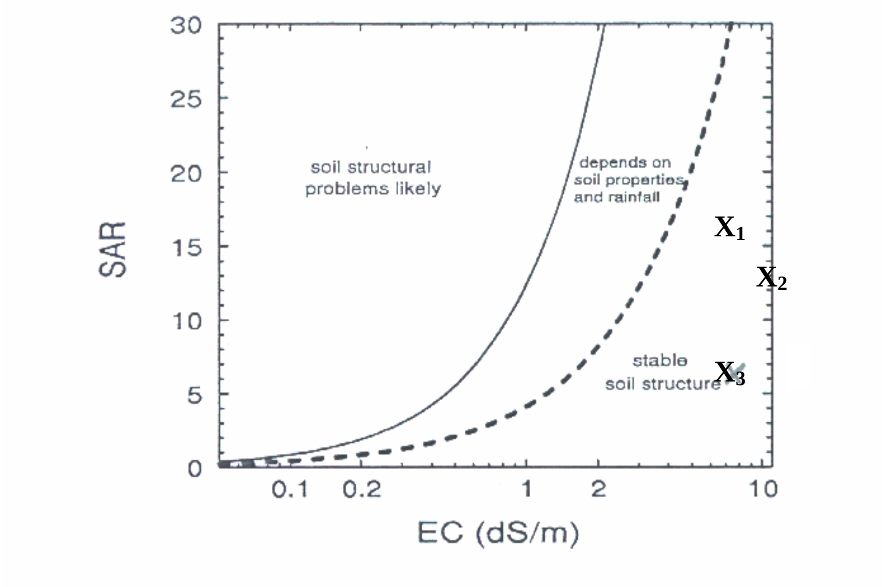
Analysis of the SAR and EC of the leachate can be used to predict whether dispersion or swelling of clays is likely to occur when the leachate is irrigated onto the landfill cap (Table 5.13). The SAR and EC of the leachate in the irrigation collection tank were used to assess the salinity risk for the leachate that was irrigated over the landfill cap. When the SAR and EC of the leachate in the irrigation tank (Table 5.3) were compared with guidelines (Table 5.13) it is apparent that soil structural problems are not expected.

The plot of the relationship between SAR and EC (Figure 5.35) suggests that soil structural problems are unlikely. Although the SAR of the leachate is relatively high, the EC is great enough to be above the "critical coagulation value" needed for clay dispersion to occur.

Table 5.13: Guidelines for the interpretation of water quality for irrigation (after Ayers & Tanji (1981) cited in Halliwell *et al*., 2001).

	EC (dSm^{-1})*		
SAR	No problem expected	Slight to moderate problem expected	Severe problem expected
0-3	>0.9	0.2 – 0.9	<0.2
3-6	>1.3	0.25 – 1.3	<0.25
6-12	>2.0	0.35 -2.0	<0.35
12-20	>3.1	0.9 – 3.1	<0.9
20+	>5.6	1.8 – 1.8	<1.8

*note that 1 dSm^{-1} = 1000 μScm^{-1}



X_1 = Mean EC and SAR for LB2
X_2 = Mean EC and SAR for LB6
X_3 = Mean EC and SAR for Irrigation tank

Figure 5.35: Relationship between SAR and EC of irrigation water to estimate structural stability. Note that 1 dSm^{-1} = 1000 μScm^{-1} (source, ANZECC & ARMCANZ 2000).

When the SAR and EC of Leachate LB2 and LB6 were analysed the results indicated that the leachate would not cause structural problems if it was irrigated onto the landfill cap. Although the SAR is higher in both LB2 and LB6 than that of the leachate in the irrigation tank, the EC is also higher which counters the effect of the higher SAR.

5.4.13 Nutrient Value of the Leachate

The nutrient loading of the irrigated leachate was calculated using the leachate bore data for four key plant nutrients (Table 5.14) and compared to the fertiliser requirements for pasture (based on an estimate of 10 stock units per hectare, using the methods of Cornforth; 1998). The irrigated leachate is likely to have lower concentrations of potassium and sulphate than that found in the two leachate bores as shown by the leachate tank analysis. The results still indicate that no potassium fertiliser would be needed to support pasture growth as the leachate loadings were above the recommended fertilizer requirement. Additional phosphate fertiliser will be needed as the leachate phosphate levels were lower than the other nutrients. No method to estimate nitrogen fertilisers requirements was given as there is no reliable soil test in common use in New Zealand to test for plant available nitrogen (Cornforth; 1998).

Table: 5.14: Leachate Nutrient Values.

	Leachate Bores (kg/ha/yr)	Recommended fertilizer Requirements[1] (kg/ha/yr)
Potassium	459	0
Sulphate	104	19
Phosphate	0.15	8
Nitrogen	784	_

[1] Based on Cornforth 1998

The leachate nutrient estimates were compared to the latest soil nutrient tests (Appendix 3) and both methods agreed that no potassium fertilizer was required but phosphate fertilizer was. The soil fertility tests did recommend the addition of nitrogen due to the low clover and organic matter content present in the landfill cap.

5.5 Conclusion

Analysis of the leachate data indicates that the landfill is in the methanogenic phase. The leachate constituents were all within the ranges expected of a methanogenic landfill, even though the soluble salts content was higher than that found in some similar New Zealand landfills. Leachate metals and metalloid concentrations were present at low levels and were generally below the mean found in other landfills.

Comparison of the leachate in the irrigation tank with the leachate in LB2 and LB6 showed that the parameters tested were generally at lower levels in the leachate irrigation tank. The lower leachate concentrations in the irrigation tank indicate that some dilution occurred between the centre of the landfill and the leachate irrigation tank. Use of the leachate bore data to assess any effects of leachate irrigation may lead to an over estimation of any potential effects. It is recommended that the leachate irrigation tank be included in the future routine monitoring programme to allow the leachate which is actually irrigated onto the landfill cap to be monitored.

Analysis of the leachate indicated that no significant detrimental effects can be expected in the long term. Analysis of the SAR and EC indicate that soil structural problems caused by the high sodium concentrations are unlikely to occur, however the soluble salt content of the leachate may be high enough to affect salt sensitive plants. With the exception of manganese, the metal and metalloid concentrations were below the long term recommended irrigation limits. Leachate manganese concentrations were high enough to cause toxicity in plants, however the high soil pH and calcium content was likely to counter the high manganese concentrations. Boron concentrations were also high enough to be toxic to boron sensitive plants. It is recommended that herbage tests be carried out on the landfill cap pasture to confirm that manganese or boron toxicity is not affecting pasture growth. Alternatively manganese or boron concentrations could be determined and the loading rates calculated to determine if toxicity is likely to occur.

Chapter 6. Ground Water Analysis

6.1 Introduction

This Chapter presents the ground water monitoring data obtained from Paokahu Landfill between 1999 and 2008. Ground water quality data was collected from two series of bores known as the 'Shallow Ground Water Bores' and the 'Deep Ground Water Bores' (Appendix 4) beginning in August 1999 and is currently ongoing. In this chapter the results for the shallow groundwater monitoring bores are presented first followed by the deep groundwater monitoring bore results. Monitoring frequencies have decreased over time since the landfill closed and are now currently carried out on a two yearly cycle.

6.2 Groundwater Bore Locations and Descriptions

The groundwater monitoring bores are located in seven locations around the Paokahu site (Figure 6.1), extending over a range of depths and geological units (Table 6.1). The bores are numbered on the Gisborne District Council's database as Bore 401 through to Bore 407. The bores also have a letter suffix which gives an approximate indication of the bore depth as follows:

- A = shallow bores (depth generally less than 5m below mean sea level),
- B = deep bores between (depth between 5m to 20m below mean sea level),
- C = the deepest bores (depth greater than 20m below mean sea level).

At most of the bore locations there is more than one bore, with each bore a different depth, e.g. north of the landfill at site 401 there are three bores in close proximity; Bores 401A, 401B and 401C, which range in depth from 5m to 26m below mean sea level (MSL). The Gisborne District Council bore names have been used in this study to allow future monitoring results to be easily compared with the results of this study.

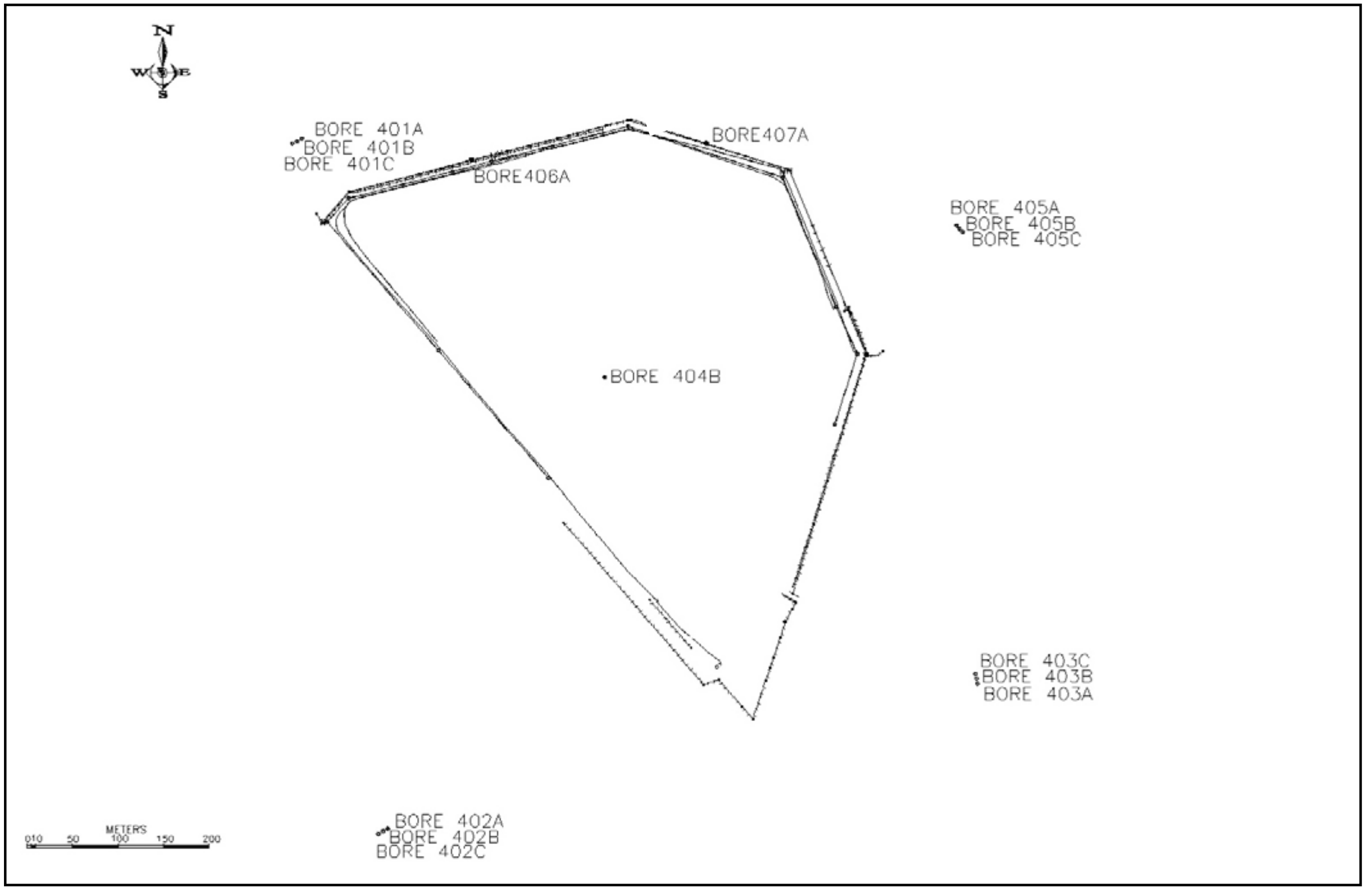


Figure 6.1: Groundwater monitoring bore locations.

The monitoring bores are predominantly located in the Te Hapara Sands aquifer, so the water quality data of the shallow ground water bores was also compared to two other nearby bores in the Te Hapara Sands aquifer. The bores were chosen after discussion with Gisborne District Council's Senior Water Conservator, Dennis Crone, as being representative of the general water quality in the Te Hapara Sands aquifer in the Awapuni area. Monitoring data for the two background bores was downloaded from the Gisborne District Council database (Appendix 5). The two bores chosen GPA036 (Background Bore 1), located at the Poverty Bay Golf Club, Lytton Rd and GPC045 (Background Bore 2), located at G & E Louden's property, Willows Rd, Gisborne (Figure 6.2).

Table 6.1: Bore lengths and depths.

Bore	Location	Bore Length (m)	Bore Depth[1] (m)	Geological Unit at Base of Bore[2]
Shallow Bores				
401A	North of landfill	6.0	-4.9	Shallow Estuarine Silts
402A	Southwest of landfill	6.0	-4.7	Te Hapara Sands
403A	South of Landfill	5.7	-1.0	Recent Sands
404B	Centre of Landfill	10.0	+1.9	Te Hapara Sands
405A	East of Landfill	6.0	-5.1	Te Hapara Sands
406A	Northwest Boundary of landfill	4.7	-3.5	Te Hapara Sands
407A	Southwest Boundary of Landfill	5.5	-4.3	Te Hapara Sands
Deep Bores				
401B	North of Landfill	12.0	-10.96	Te Hapara Sands
401C	North of Landfill	27.3	-26.18	Te Hapara Sands
402B	Southwest of Landfill	17.3	-15.9	Te Hapara Sands
402C	Southwest of Landfill	40.0	-38.7	Te Hapara Sands
403B	South of landfill	13.0	-8.25	Te Hapara Sands
403C	South of Landfill	26.0	-21.33	Te Hapara Sands
405B	East of Landfill	12.0	-11.04	Te Hapara Sands
405C	East of LAndfill	20.0	-19.16	Te Hapara Sands

[1] Depth relative to mean sea level.

[2] Data obtained from bore logs records held at Gisborne District Council.

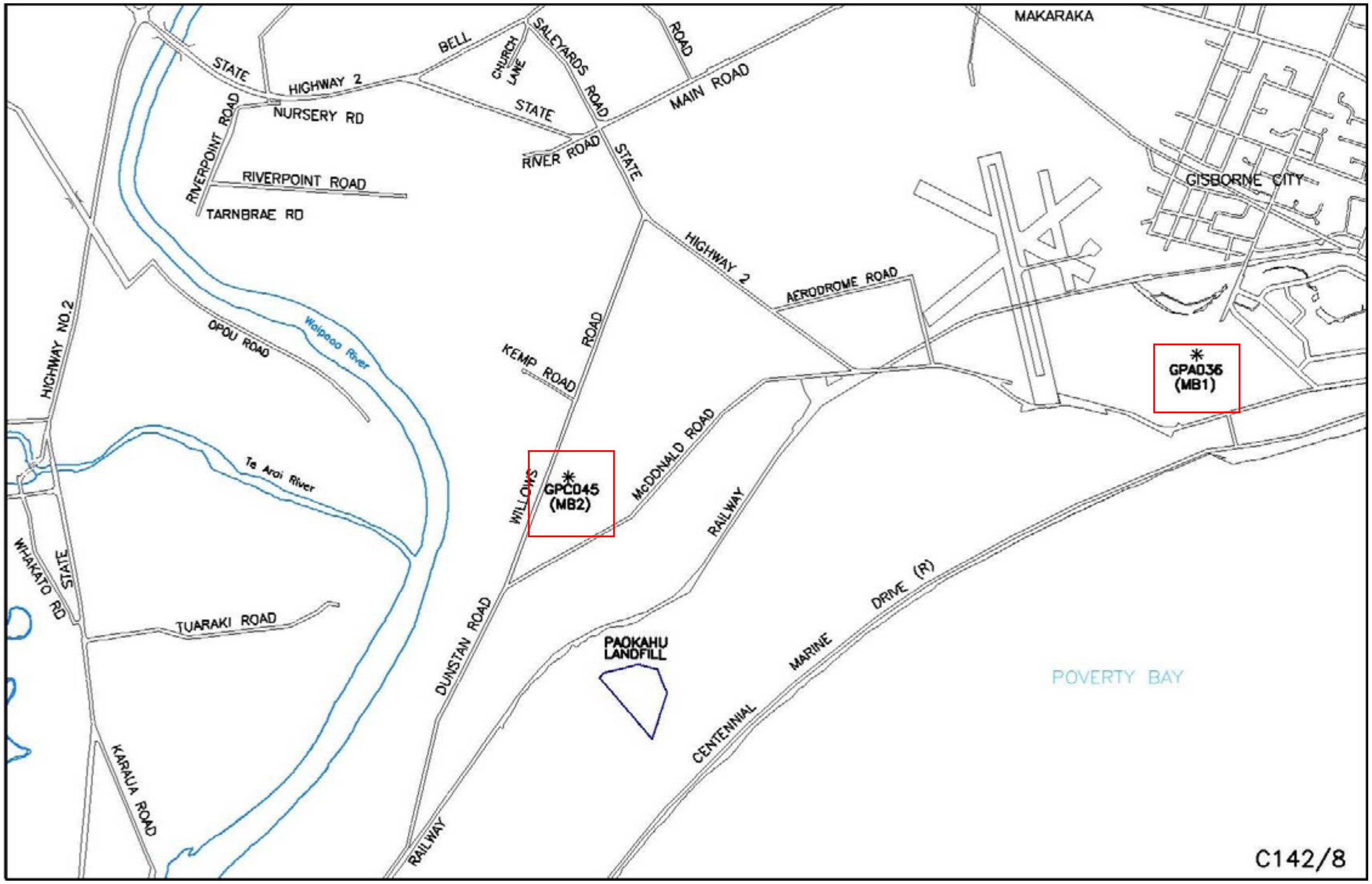


Figure 6.2: Location of Background Bore 1 and Background Bore 2.

6.3 Previous Groundwater Studies

Prior to Gisborne District Council applying for resource consent to extend the landfill in 1997 a study of the groundwater in the area of the Paokahu landfill was undertaken as part of the assessment of environmental effects. An initial study conducted in 1993 found that leachate was present in the groundwater at shallow depths (around 2m below MSL) in bores close to the landfill. The leachate contamination was attributed to leachate seeps observed coming from the sides of the landfill. There was no direct evidence of leachate in any of the other groundwater samples including those taken from directly beneath the landfill (Taylor 1997).

A more detailed study was conducted in the summer (Nelson, 1994). The study found that:

- Groundwater was dominated by seawater intrusions during the summer.
- All deep bores recorded increasing head with depth, indicating the landfill was located in a groundwater discharge zone in the summer, probably caused by upward leakage from the Makauri Gravels Aquifer.
- The direction of groundwater flow was away from the coast towards the Awapuni Drain.
- Leachate was only detected in the very shallow bores located very close to the toe of the landfill and in the site drains immediately adjacent to the landfill and was attributed to leachate seeping from the landfill side slopes. No Leachate was detected in any of the groundwater monitoring bores, including Bore 404B under the centre of the landfill.

The report also recommended that further monitoring and investigations be undertaken to confirm the results of the 1994 study and that a detailed study be undertaken in winter to provide information on both the summer and winter groundwater conditions, but was never undertaken.

6.4 Analytical Results and Preliminary Discussion

6.4.1 Shallow Groundwater Bores

6.4.1.1 pH

The pH observed in the shallow bores was generally slightly alkaline (Figure 6.3). The highest pH was in Bore 401A which is located in the Shallow Estuarine . The pH in the Te Hapara Sands aquifer bores was generally slightly lower and ranged between 7.1 and 7.6. The pH of the Te Hapara Sands aquifer has been reported as typically alkaline and ranging between 7.2 – 7.9 (Brown & Elmsly, 1987). The changes in pH in the Te Hapara Sands aquifer generally correlate with changes in CO_2 and bicarbonate content of the groundwater (Brown & Elmsly, 1987). The fluctuations in pH in the shallow ground water bores were not correlated to any fluctuations in the alkalinity concentrations.

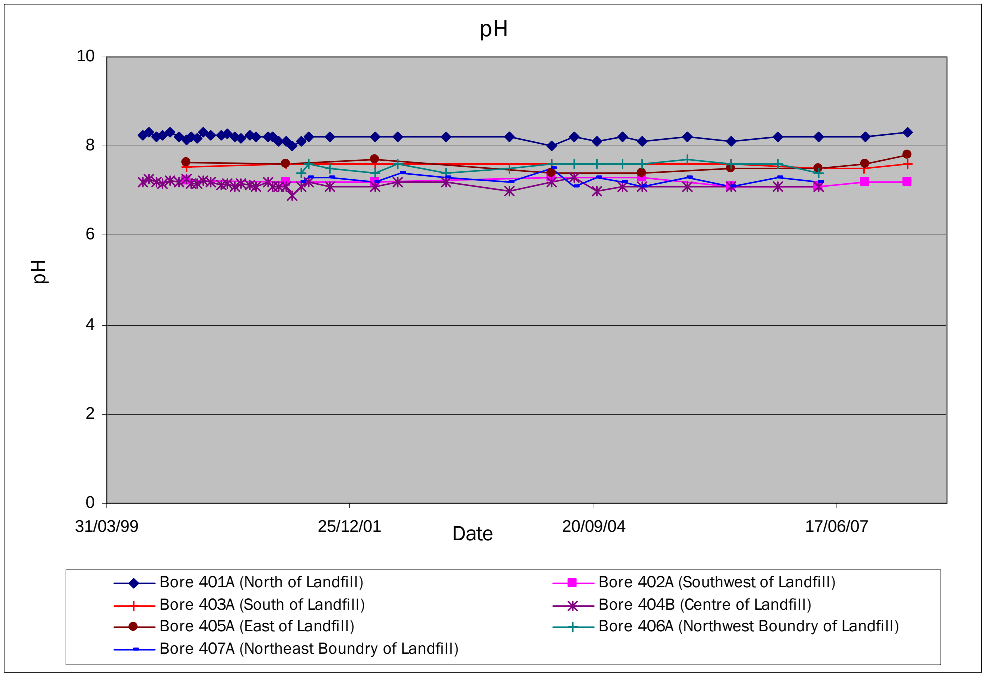


Figure 6.3: pH in shallow ground water bores.

6.4.1.2 Electrical Conductivity

A range of electrical conductivities (EC) were observed in the shallow ground water bores (Figure 6.4). The EC in Bores 401A (Shallow Estuarine Silts) and 403A (Recent Sands) were lower than the ECs of the bores located in the Te

Hapara Sands aquifer. There was no relationship between bore depth or static water levels to the EC found within the Te Hapara Sands aquifer. For example Bores 402B and 405A were both located at a depth of approximately 5m below mean sea level and have similar mean static water levels but the mean EC in the bores ranged from 4547 to 34 667 µScm^{-1}. There also does not appear to be any correlation between the locations of the bores and the EC, with the highest EC observed in Bore 402A located to the west of the landfill.

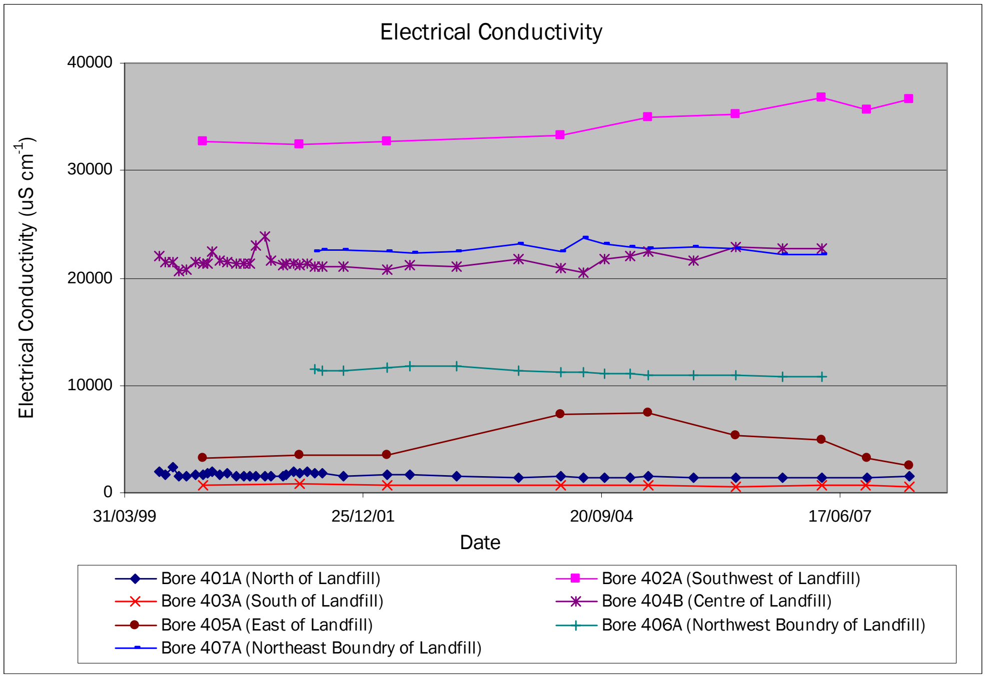


Figure 6.4: Electrical conductivity in shallow ground water bores.

Brown & Elmsly (1987) reported the EC of the Te Hapara Sands aquifer typically ranged between 870 – 1160 µScm^{-1} and generally increased towards the coast with ECs of up to 8780 µScm^{-1} recorded in the Matawhero/Awapuni area. Only the ECs reported in Bore 405A (east of the landfill) were within the range reported by Brown & Elmsly (1987) for the Te Hapara Sands aquifer in the Awapuni area. The EC in the nearby by Background Bore 1 (BG1) (mean 702 g m^{-3}) and Background Bore 2 (BG2) (mean 898 g m^{-3}) were lower than the ECs in the shallow ground water bores within the Te Hapara Sands aquifer.

6.4.1.3 Biological and Chemical Oxygen Demand

a. Biological Oxygen Demand

The biological oxygen demand (BOD_5) ranged from a low of <0.5 g m^{-3} in Bore 403A (Recent Sands) to a maximum of 6.3 g m^{-3} in Bore 407A (Te Hapara Sands aquifer) (Figure 6.5). BOD_5 concentrations in Bore 407A showed a steady increase up to 2006 when concentrations peaked at 6.3 g m^{-3}. The BOD_5 concentrations in the other bores were typically below 2 g m^{-3}.

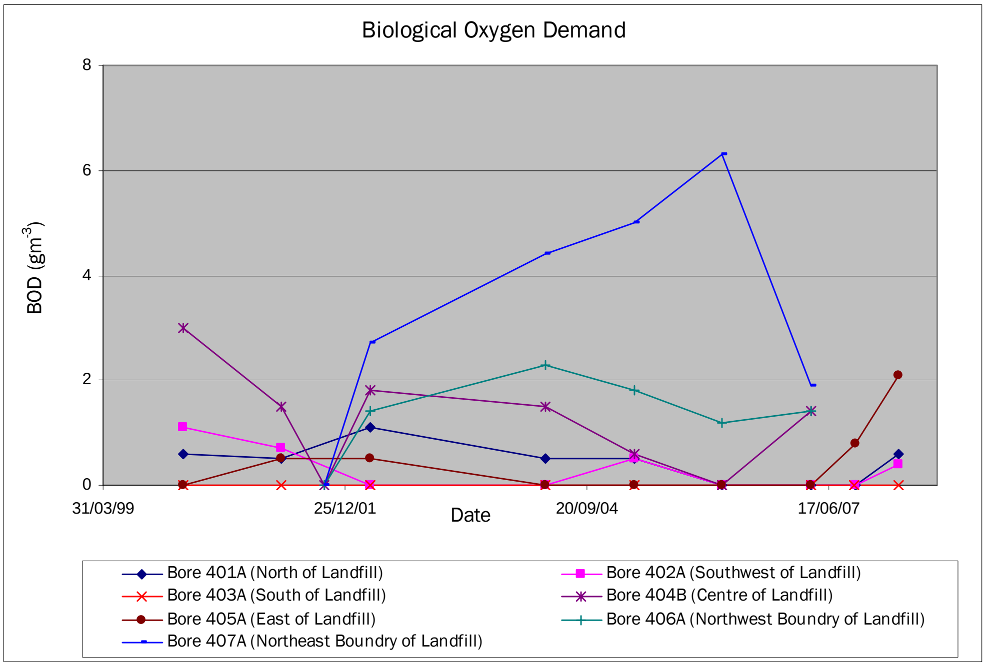


Figure 6.5: Biological oxygen demand in shallow groundwater bores.

There are no generic BOD and COD values available for the Te Hapara Sands aquifer to compare against shallow bores. Bore BG1 had a mean BOD_5 of 1.2 g m^{-3} and a mean COD of 21.3 g m^{-3}. In Bore BG2 the mean BOD_5 was 1.1 g m^{-3} and the mean COD was 18.4 g m^{-3}.

The BOD levels in the shallow bores were generally low, with four of the bores having a mean of 0.5 g m^{-3} or less. There was no obvious reason for the steady increase in BOD_5 in Bore 407A between 2002 and 2006, other than variations in ground water temperature at the time of sampling. The ground water temperatures (Appendix 4) ranged from 15.4°C to 19.3°C and the lowest temperatures coincided with the lowest BOD values.

b. Chemical Oxygen Demand

Chemical oxygen demand (COD) values (Figure 6.6) were lowest in Bore 403A (Recent Sands deposits) where the concentrations were typically less than 5 g m^{-3}. The highest COD concentrations were observed in Bores 402A and 404B (Te Hapara Sands aquifer). The maximum COD concentration of 400 g m^{-3} was recorded in Bore 407A (Figure 6.6). The Te Hapara Sands aquifer had mean COD concentrations higher than that of the two background bores.

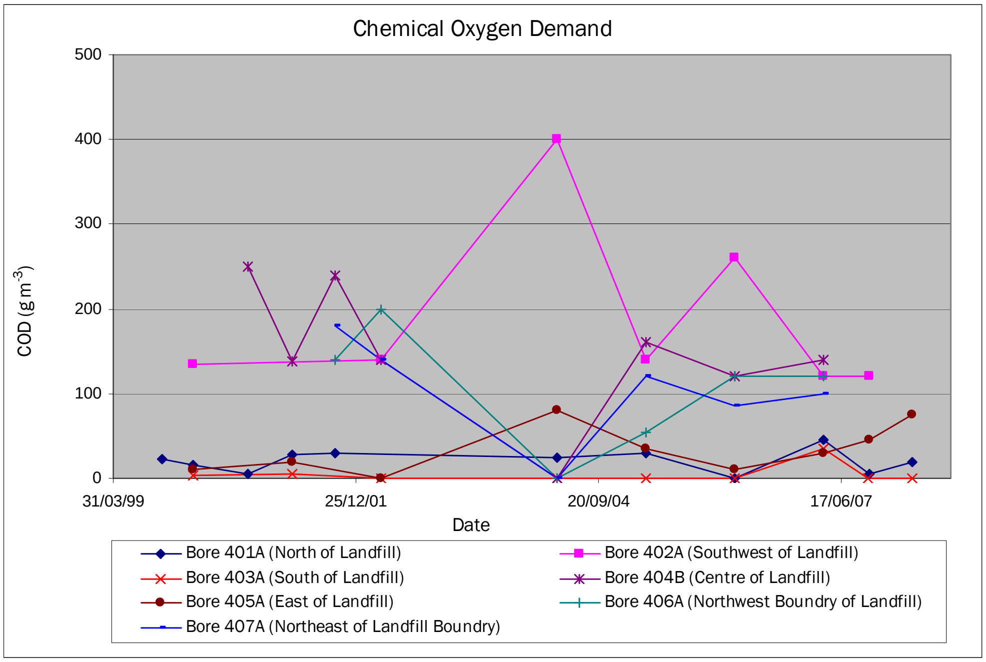


Figure 6.6: Chemical oxygen demand in shallow groundwater bores.

6.4.1.4 Nitrogen

a. Ammonical Nitrogen

Ammonical Nitrogen (Figure 6.7) concentrations were lowest in Bore 403A (Recent Sands) where they were below the detection limit of <0.1 g m^{-3}. The highest concentrations were recorded in Bore 404B (Te Hapara Sands aquifer) with a mean of 2.59 g m^{-3}. The ammonical nitrogen concentration in Bore 407A showed large fluctuations ranging from 0.06 – 2.9 g m^{-3}.

The bores in the Te Hapara Sands aquifer had higher ammonical nitrogen concentrations than the two background bores (Bore BG1 mean = 0.29 g m^{-3} 0.25 g m^{-3}; Bore BG2 mean = 0.29 g m^{-3}). There was no distinguishable pattern between the bore locations or depths and the ammonical nitrogen concentrations.

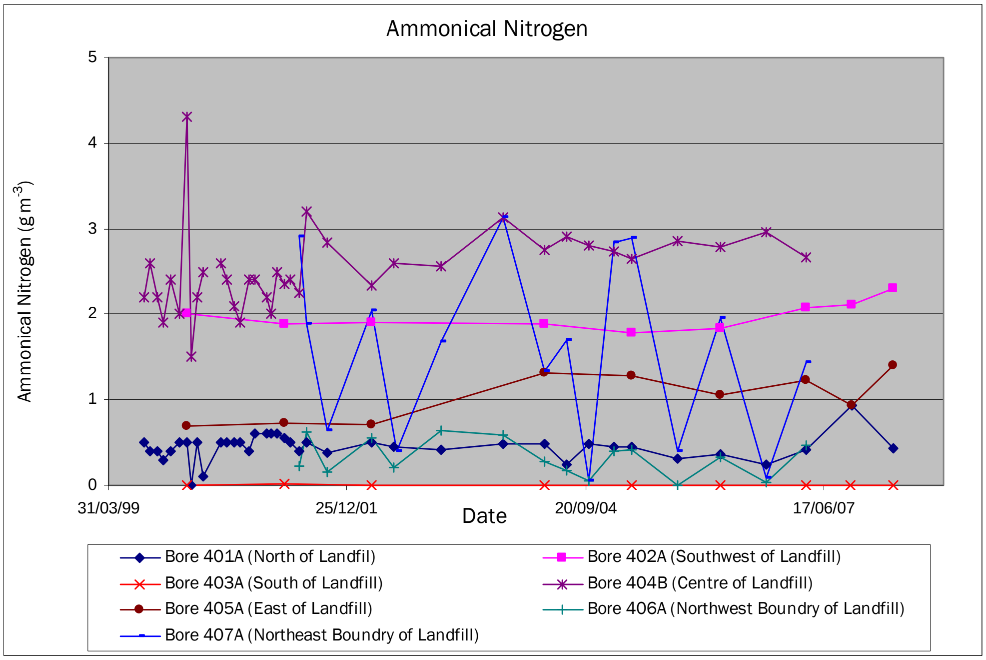


Figure 6.7: Ammonical nitrogen in shallow ground water bores.

The ammonical nitrogen and nitrate concentrations in Bore 407A (north east boundary, Te Hapara Sands aquifer) showed large fluctuations over time compared to the other bores. When the ammonical nitrogen and nitrate concentrations in Bore 407A were compared (Figure 6.8), the decreases in ammonical nitrogen concentrations were accompanied by an increase in the nitrate concentrations. The pattern observed in Bore 407A indicates that nitrogen oxidation was occurring in the groundwater. The fluctuations in the ammonical nitrogen and nitrate roughly follow a seasonal pattern with the highest ammonical nitrogen concentrations generally occurring in the summer.

Ammonical nitrogen concentrations were highest in Bore 404B (centre of the landfill, Te Hapara Sands aquifer) which could indicate leachate in the aquifer, as the high ammonical nitrogen concentrations in the leachate would increase the ground water ammonical nitrogen concentrations. Alternatively the higher ammonical nitrogen concentrations could just be due to ground water variations between bores.

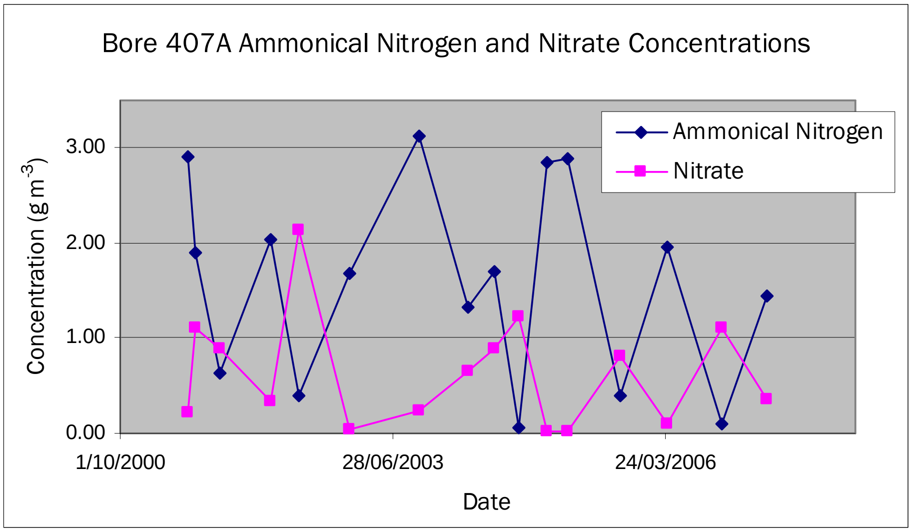


Figure 6.8: Ammonical nitrogen and nitrate concentrations in Bore 407A.

The lowest ammonical nitrogen concentrations occurred in Bore 403A (south of the landfill, Recent Sands) where all but one sample was below the detection limit of the test. The low ammonical nitrogen concentrations were accompanied by relatively high nitrate concentrations compared to the other bores, indicating that the nitrogen present is in the form of nitrate.

b. Nitrate

Nitrate concentrations (Figure 6.9) were highest in Bores 403A and 407A. Nitrate concentrations in Bores 401A, 402A, 404B and 405B were consistently below or near the detection limit of the test which ranged between 0.02-0.05 g m^{-3}.

Brown & Elmsly (1987) reported nitrate concentrations in the Te Hapara Sands aquifer typically ranged between 0.0-0.3 g m^{-3} with concentrations increasing up to 0.5 – 5.5 g m^{-3} in the Awapuni area. The background bore BG1 had a mean nitrate concentration of 1.21 g m^{-3} and BG2 had a mean of 0.012 g m^{-3}. The nitrate concentrations in the Paokahu shallow ground water bores were all within the range reported for the Awapuni area by Brown & Elmsly (1987).

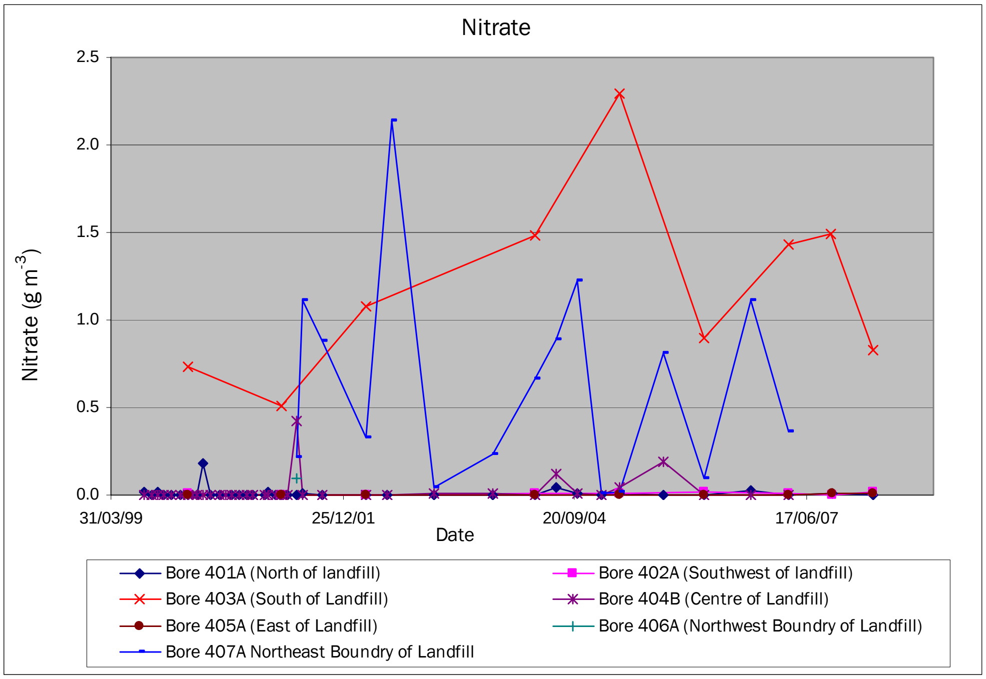


Figure 6.9: Nitrate in shallow ground water bores.

c. Nitrite

Nitrite concentrations (Figure 6.10) showed a similar trend to that of nitrate with the highest concentrations observed in Bores 403A and 407A. Nitrite concentrations in Bore 405A were all below the detection limit of 0.002 g m^{-3}. The background Bores BG1 and BG2 both had mean concentrations below the detection limit of the test (0.002 g m^{-3}). The nitrite levels in the groundwater bores were generally higher than that of the background bores, but were still low with mean concentrations all below 0.025 g m^{-3}.

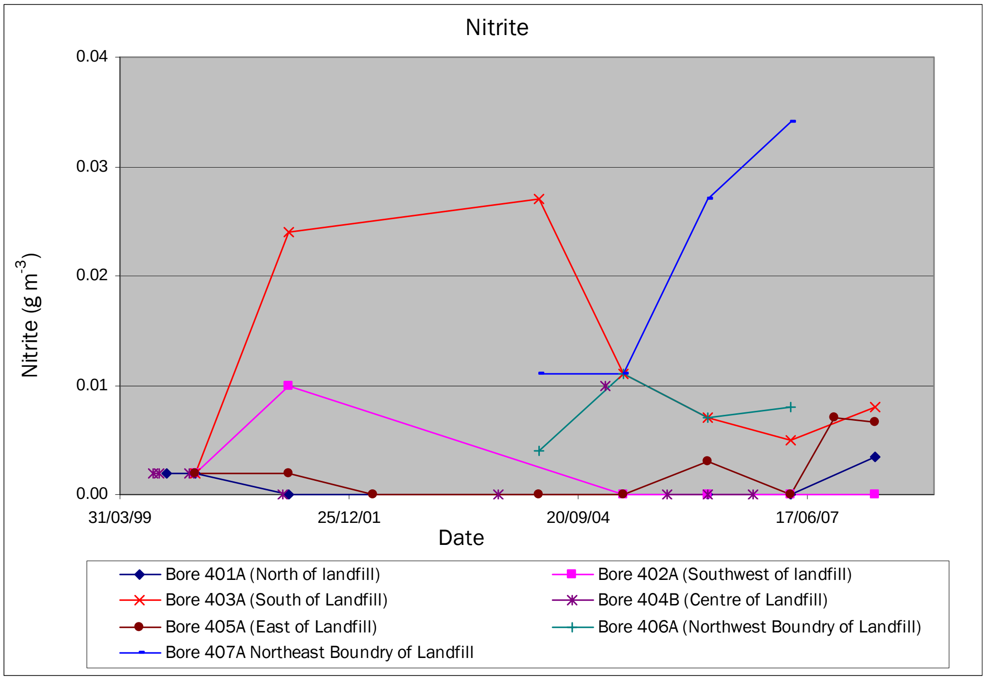


Figure 6.10: Nitrite in shallow ground water bores.

6.4.1.5 Major Cations

a. Sodium

A range of sodium concentrations (Figure 6.11) were observed in the seven shallow ground water bores. The lowest concentrations were observed in bore 403A which had a mean of 33 g m^{-3}, while Bore 402A had the highest concentrations with a mean of 6708 g m^{-3}.

The plot of dissolved sodium (Figure 6.11) shows a similar trend to the EC plot (Figure 6.4) for the shallow bores with the highest concentration in Bore 402A and the lowest concentrations in Bore 403A indicating a relationship between EC and sodium concentrations. The lowest sodium concentrations were observed in the two bores located in the Recent Sands and shallow Estuarine Silts deposits. The Te Hapara Sands aquifer shallow water bores had significantly higher (P<0.001) sodium concentrations than the two back ground bores (BG1 mean = 36 g m^{-3}; BG2 mean = 32 g m^{-3}).

Sodium concentrations have been reported to increase towards the coast in the Te Hapara Sands aquifer due to residual saline soil conditions and from salt water intrusions (Brown & Elmsly, 1987). Typical seawater has a sodium concentration of 10 752 g m^{-3} (Anderson, 2003), so salt water intrusion into the aquifer may be the cause of the high sodium concentrations in the Te Hapara Sands aquifer bores.

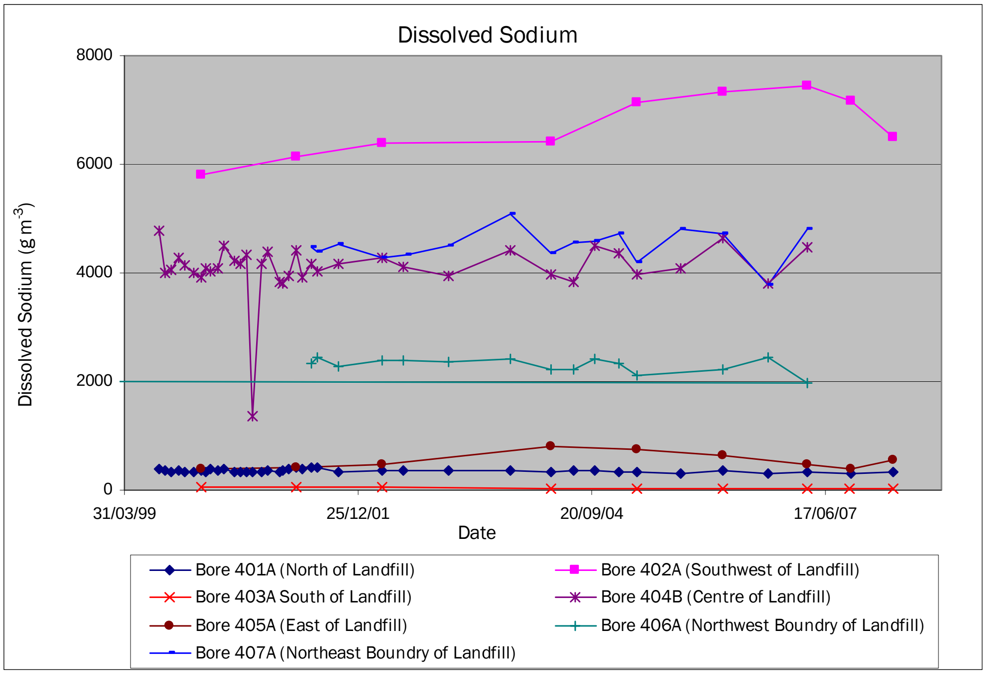


Figure 6.11: Sodium concentrations in the shallow ground water bores.

b. Potassium

The lowest potassium concentrations (Figure 6.12) were observed in Bores 401A (Shallow Estuarine Silts) and 403A (Recent Sands). Potassium concentrations were high compared to the back ground bores (BG1 mean 7.3 g m^{-3} and BG2 mean = 6.2 g m^{-3}).

Potassium concentrations also follow a similar trend to the EC and sodium concentrations with the highest concentrations in Bore 402A and the lowest concentrations in Bore 403A. The potassium concentrations in Bore 404B were higher relative to the sodium and EC when compared to the other shallow ground water bores. The shallower depth of bore 404B (1.9m above MSL) compared to the other bores may have had some influence. Alternatively, as Bore 404B is below the centre of the landfill the relative increase in potassium could be due to leachate infiltration. The leachate potassium concentrations were higher than those of the surrounding groundwater so any leachate entering the groundwater would increase the groundwater potassium concentrations.

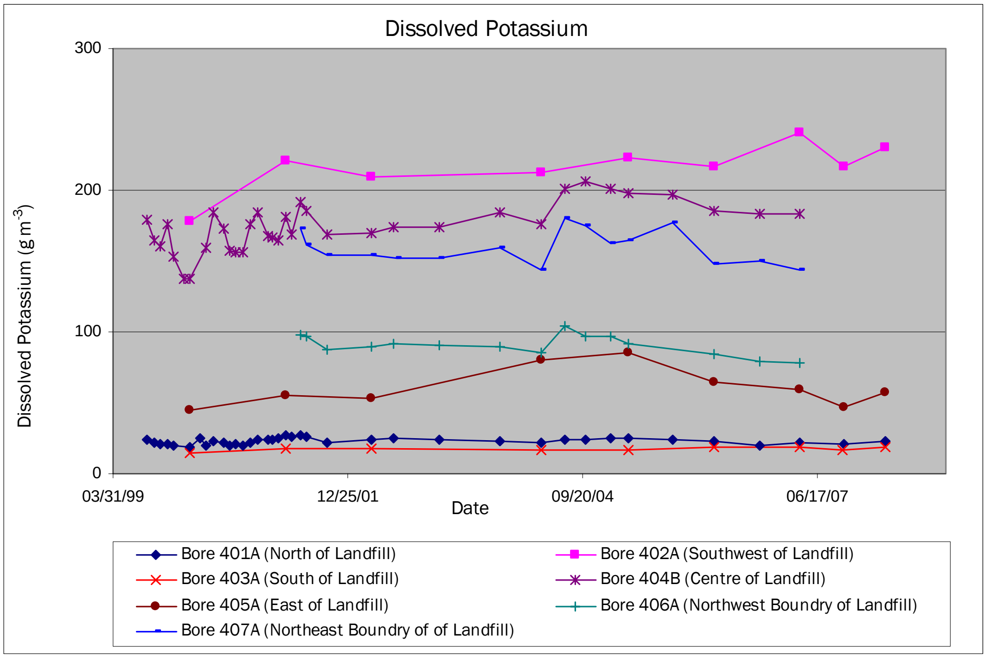


Figure 6.12: Potassium concentrations in the shallow ground water bores.

c. Calcium and Magnesium

The highest calcium concentrations (Figure 6.13) were observed in Bore 402A (Te Hapara Sands aquifer), but differ slightly to the trend observed with sodium and potassium concentrations as the lowest concentrations occur in Bore 401A (Shallow Estuarine Silts).

d. Magnesium

Magnesium concentrations (Figure 6.14) followed a similar pattern to that of calcium. The lowest concentrations were in Bore 401A (Shallow Estuarine Silts) and the highest concentrations in Bore 402A (Te Hapara Sands aquifer). The calcium and magnesium concentrations in the groundwater bores generally follow a similar trend to the other cations and EC. No dissolved calcium and magnesium concentrations were recorded for the two background bores so no comparison between the bores can be made.

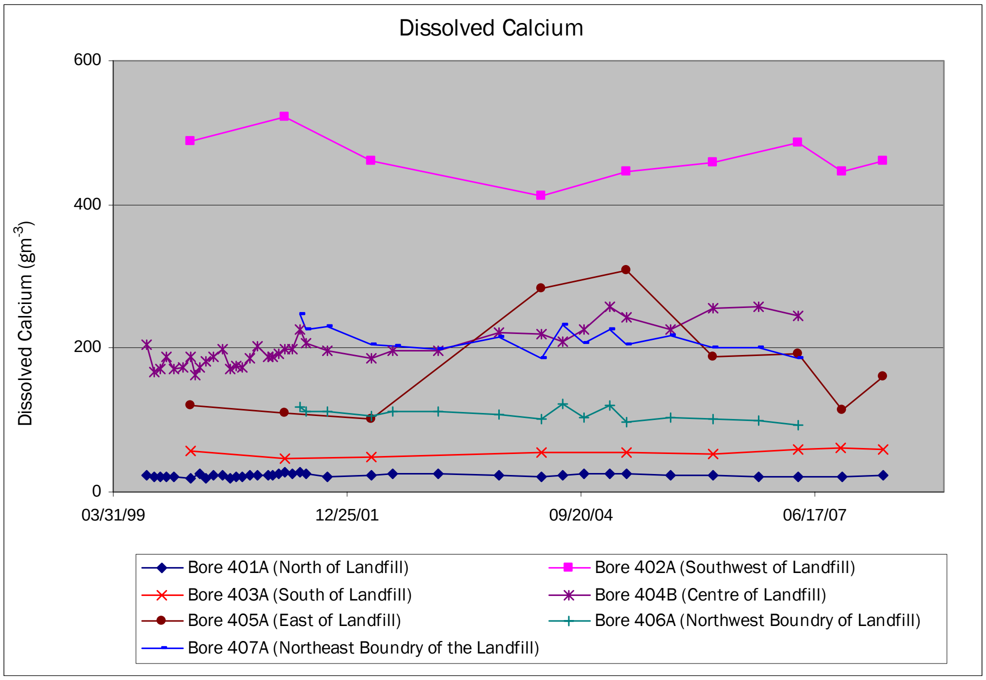


Figure 6.13: Calcium concentrations in the shallow ground water bores.

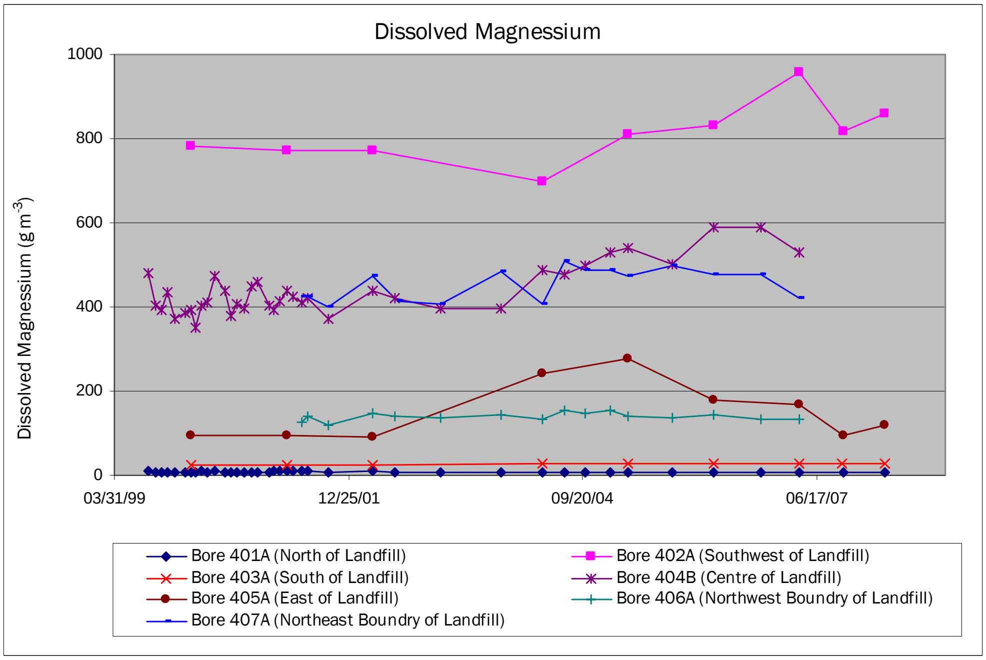


Figure 6.14: Magnesium concentrations in the shallow ground water bores.

The cation concentrations in Bore 405A (east of landfill) showed a steady increase between 2002 and 2005 before gradually declining again between 2005 and 2008. The increase was more pronounced for calcium and magnesium. The reason for the increase is unknown, but could be due to a temporary increase in saltwater intrusions. The increase in cation concentrations was unlikely to be associated with a change in groundwater levels as the static water levels in Bore 405A followed a similar trend to the static water levels in the other bores during this period (Section 6.5).

6.4.1.6 Major Anions

a. Chloride

A range of chloride concentrations (Figure 6.15) were observed in the seven shallow ground water bores. The lowest concentrations were observed in Bore 403A (Recent Sands) which had a mean of 36 g m^{-3} and 401A (Shallow Estuarine Silts) which had a mean of 204 g m^{-3}. Bore 402A (Te Hapara Sands aquifer) had the highest chloride concentrations with a mean of 12 133 g m^{-3}.

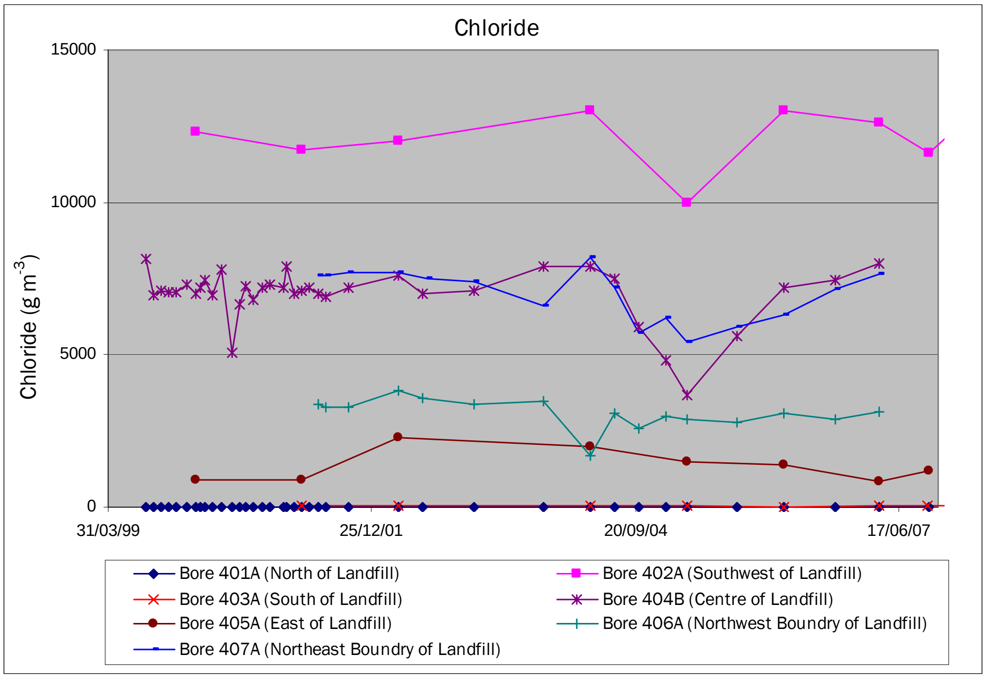


Figure 6.15: Chloride concentrations in the shallow ground water bores.

The plot of dissolved chloride concentrations (Figure 6.15) show a similar trend to the EC and dissolved sodium plots (Figures 6.4 & 6.11) for the shallow bores indicating the high EC, sodium and chloride concentrations in some of the bores is caused by sea water infiltration into the aquifer. Sea water typically has a Na:Cl ratio of 0.56, and a K:Cl ratio of 0.020. Bore 402A (Te Hapara Sands aquifer), which had the highest chloride concentrations, had a Na:Cl ratio of 0.55 and a K:Cl ratio of 0.021 indicating sea water is the dominant factor influencing water quality. The Na:Cl and K:Cl ratios in the other bores ranged from 0.39-0.74 and 0.017-0.044 respectively which indicates that some sea water is present in the groundwater.

Bore 401 (Shallow Estuarine Silts) had a Na:Cl ratio of 1.78, and a K:Cl ratio of 0.11 while Bore 403A (Recent Sands Deposits) had a Na:Cl ration of 0.040, and a K:Cl ratio of 0.21 which indicates that the composition of the water in the two bores is different from each other and is also different to the Te Hapara Sands aquifer .

Brown & Elmsly (1987) reported chloride concentrations in the Te Hapara Sands aquifer typically ranged between 25-80 g m^{-3} with concentrations increasing up to 2750 g m^{-3} in the Matawhero/Awapuni areas. The two background bores (BG1 mean = 59 g m^{-3} and BG2 mean = 48 g m^{-3}) were within the range reported as "typical" by Brown & Elmsly (1987). The shallow groundwater bores in the Te Hapara Sands aquifer had chloride concentrations higher than that reported in the background bores and only Bore 406A was within the range reported by Brown & Elmsly (1987) for the Te Hapara Sands aquifer in the Matawhero/Awapuni area. The chloride concentrations in Bore 402A, 404B and 407A were more comparable to the chloride concentration of seawater (19 345 g m^{-3}) than to the concentrations typically found in the Te Hapara Sands aquifer.

b. Sulphate

The lowest Sulphate concentrations (Figure 6.16) were also observed in bores 401A (mean = 5.2 g m^{-3}) and 403A (mean = 39 g m^{-3}) and the highest concentrations in Bore 402A which had a mean of 1911 g m^{-3}.

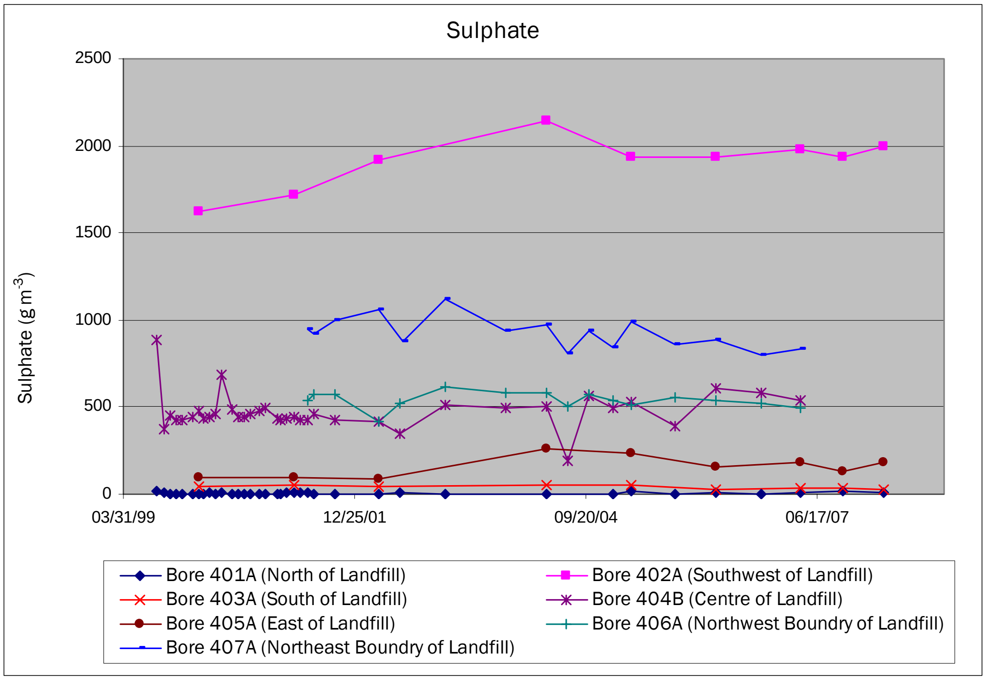


Figure 6.16: Sulphate concentrations in the shallow ground water bores.

In the Te Hapara Sands aquifer the sulphate concentrations in bores 402A, 404B, 406A and 407A were high compared to the concentrations reported by Brown & Elmsly (1987) and those recorded in the two background bores. Brown & Elmsly (1987) reported that sulphate concentrations in the Te Hapara Sands aquifer typically ranged between 25 - 63 g m^{-3} increasing to 129 - 170 g m^{-3} in the Awapuni area. Bore BG1 had a mean concentration 15 g m^{-3} and BG2 a mean of 28 g m^{-3}. Sulphate concentrations in bores 402A, 404B, 406A and 407A were closer to the concentrations typically found in sea water (approximately 2700 g m^{-3}) than the concentrations typically reported for the Te Hapara Sands aquifer.

c. Alkalinity

Alkalinity concentrations (Figure 6.17) ranged from a low of 205 – 290 g m^{-3} in Bore 403A (Recent Sands deposits) to a high of 1400 g m^{-3} in Bore 404B (Te Hapara Sands aquifer).

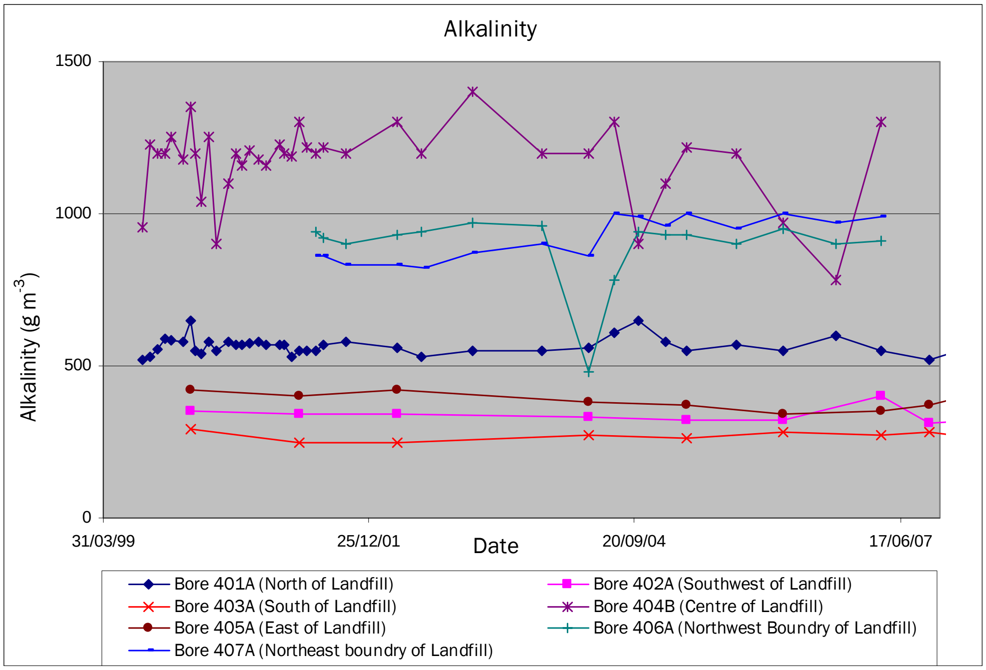


Figure 6.17: Alkalinity concentrations in the shallow ground water bores.

Brown & Elmsly (1987) reported that alkalinity typically ranged between 300-500 g m^{-3} in the Te Hapara Sands aquifer and was slightly higher in the Awapuni area at approximately 700 g m^{-3}. Alkalinity in the background bore BG1 had a mean of 229 g m^{-3} and BG2 had a mean of 350 g m^{-3} which was lower than the Paokahu shallow bores located in the Te Hapara Sands aquifer. The mean alkalinity in Bores 404B, 406A and 407A was high compared to the other reported alkalinity concentrations.

6.4.1.7 Trace Elements

a. Dissolved Reactive Phosphorous

The dissolved reactive phosphorous (DRP) concentrations (Figure 6.18) in Bore 401A were higher than those observed in the other shallow ground water bores.

The mean DRP concentration in Bore 401A was 2.2 g m^{-3} compared to the other shallow ground water bores which ranged from a mean of 0.04 g m^{-3} in Bore 402A to 0.02 g m^{-3} in Bores 405A and 406A.

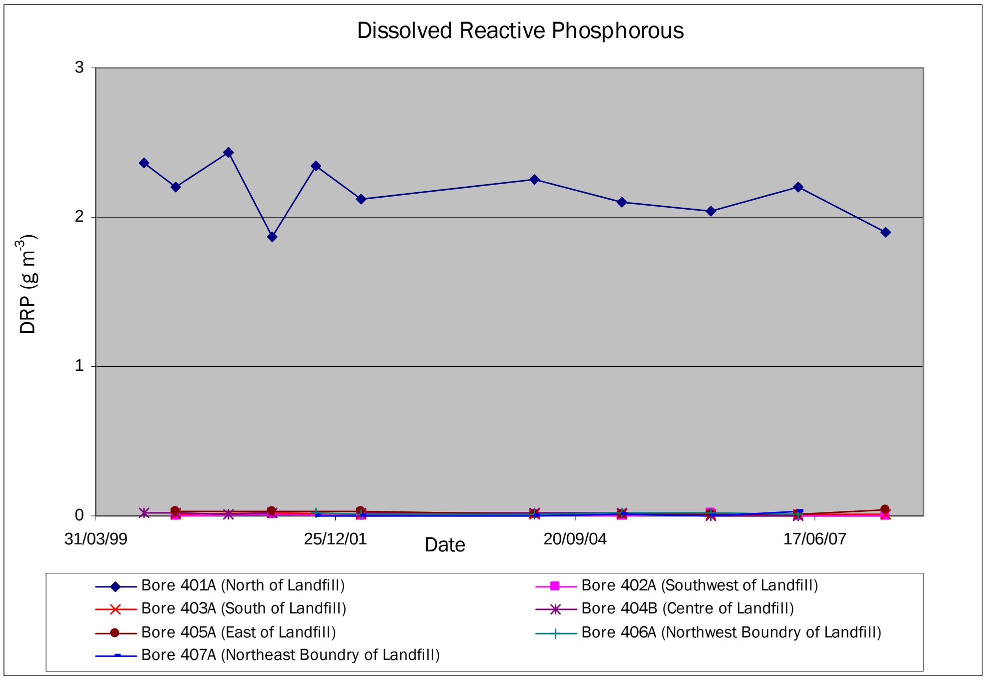


Figure 6.18: Dissolved Reactive Phosphorous concentrations in the shallow ground water bores.

Brown & Elmsly (1987) reported phosphorous was generally present in low levels in the Te Hapara Sands aquifer typically ranging between 0.0-0.15 g m^{-3} and slightly higher in the Awapuni area ranging from 0.16-0.20 g m^{-3}. Bore BG1 had a mean of 0.2 g m^{-3} and Bore BG2 had a mean of 0.01 g m^{-3} which was similar to the phosphate concentrations reported in the Paokahu shallow groundwater bores (with the exception of Bore 401A).

Bore 401A is located in farmland to the north of the landfill in the Shallow Estuarine Silts. The land surrounding the bore was under different management to the landfill and surrounding areas and was used for grazing cattle and sheep and cropping in the summer months. The high DRP concentrations can't be the result of a leachate in the aquifer as the leachate DRP concentrations were lower than those found in Bore 401A. The high DRP concentrations in Bore 401A could be caused by higher application rates of phosphate fertilizers leaching into the shallower Estuarine Silts.

b. Boron

The boron concentrations (Figure 6.19) in Bores 404B, 406A and 407A were very similar and all follow the same trend of minor fluctuations with concentration ranging between 1.8 and 2.68 g m^{-3}. The boron concentrations in Bores 401A, 403A and 405A were also similar and ranged from 0.069 – 0.5 g m^{-3}. The Boron concentrations in Bore 402A appear to be increasing over time.

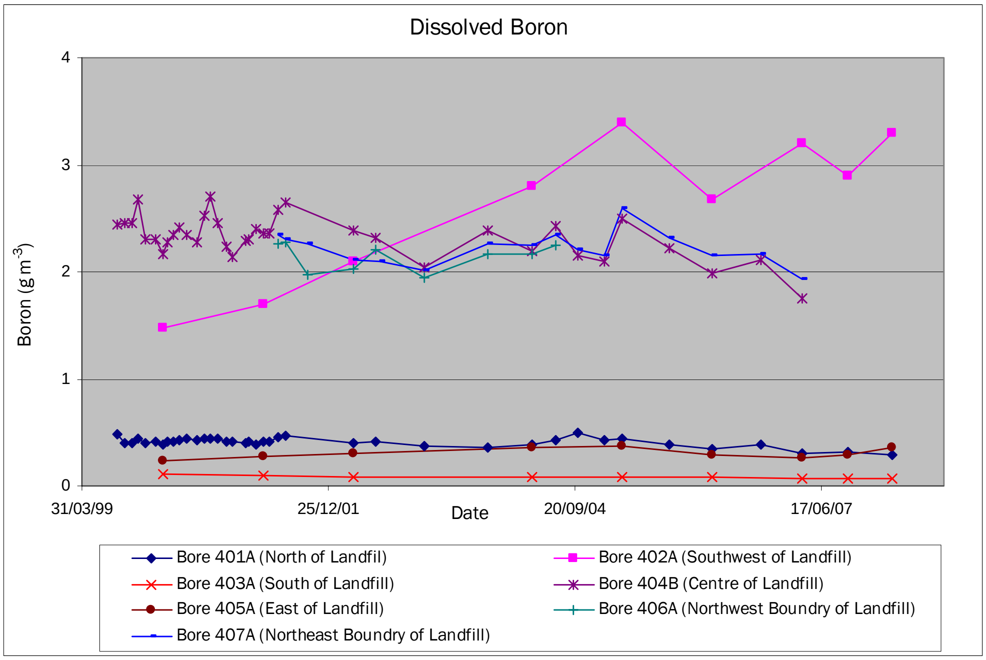


Figure 6.19: Dissolved boron concentrations in the shallow ground water bores.

There were no reported boron concentrations available for the Te Hapara Sands aquifer or the background bores to compare against the Paokahu shallow water bores. The boron concentrations in Bore 402A increased over time and with a mean of over 3 g m^{-3} since September 2004, which is higher than the boron concentrations found in both leachate bores. Typical sea water has a boron concentration of 4.5 g m^{-3} (Anderson; 2003) so the high concentration may be due to sea water infiltration as Bore 402A also had high EC, Na, K and Cl concentrations

c. Iron

The highest iron concentrations (Figure 6.20) were observed in Bore 401A which had a mean value of 0.78 g m^{-3}. Apart from the samples taken from Bore 405A between 2000-2001 all other recorded concentrations were below the detection limit of the test.

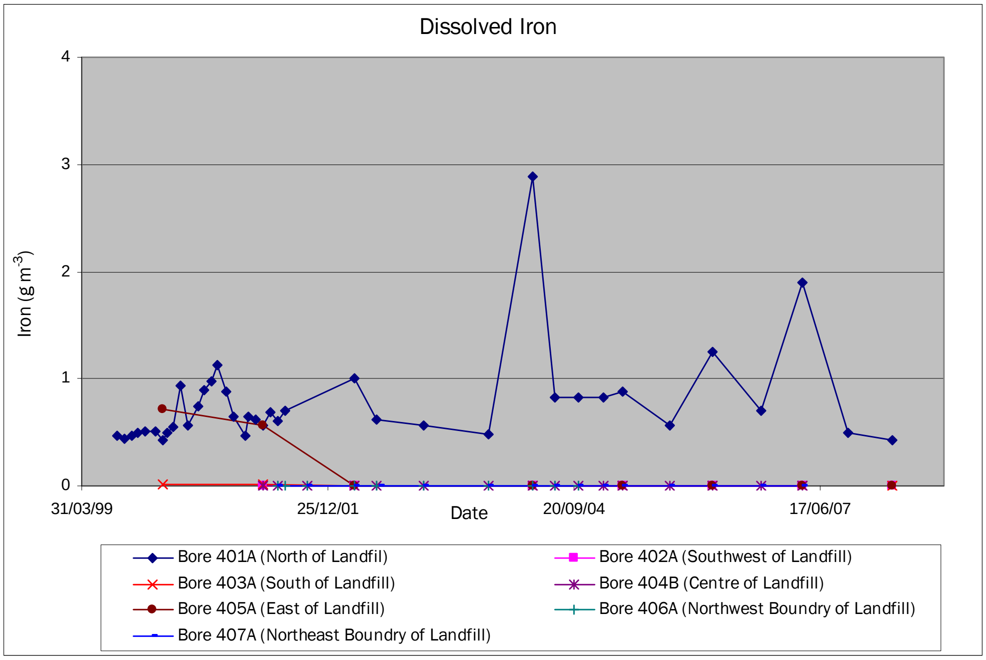


Figure 6.20: Dissolved iron concentrations in the shallow ground water bores.

Brown & Elmsly (1987) reported iron concentrations in the Te Hapara Sands aquifer were variable but typically ranged from 0-7 g m^{-3} and have been reported to increase to 8-16 g m^{-3} in the Matawhero/Awapuni areas. The mean iron concentration in bore BG1 was 4.3 g m^{-3} and 12.4 g m^{-3} in bore BG2. The iron concentrations in the Paokahu shallow bores (typically below detection limits in most bores) were generally low compared to those typically reported for the Te Hapara Sands aquifer.

The only bore where iron was consistently detected was Bore 401A (north of the landfill) which is located in the Shallow Estuarine Silts.

d. Manganese

The lowest dissolved manganese concentrations (Figure 6.21) were observed in Bore 403A where the concentrations ranged from <0.0005 – 0.004 g m^{-3}. The highest concentrations were recorded in Bore 407A where the concentrations ranged from 1.03 – 1.35 g m^{-3}.

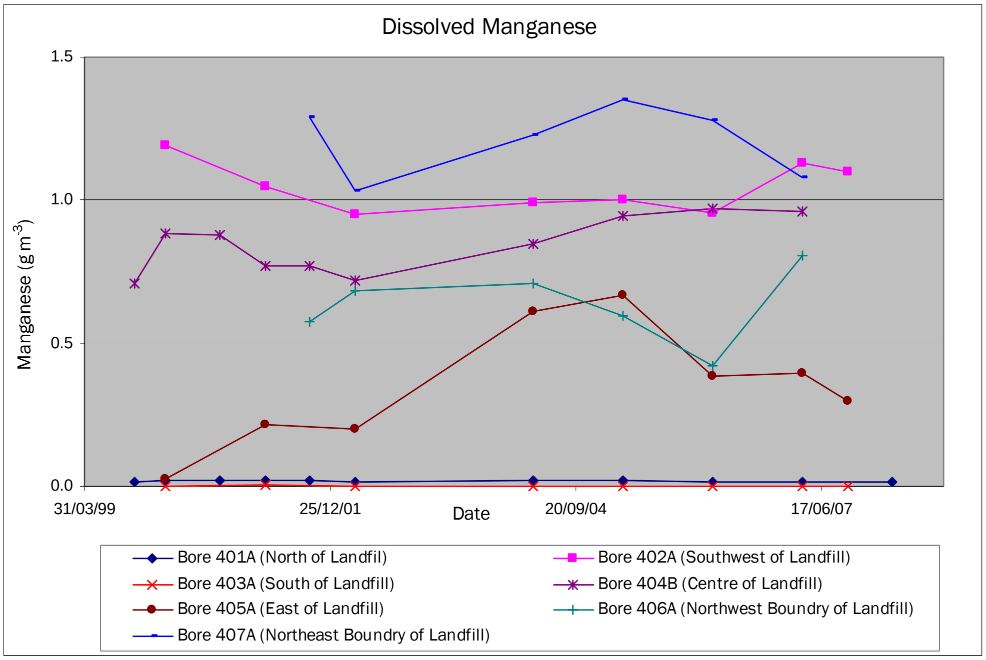


Figure 6.21: Dissolved manganese concentrations in the shallow ground water bores.

There were no generic manganese concentrations available for the Te Hapara Sands aquifer but data is available for Bores BG1 (mean 0.04 g m^{-3}) and BG2 (mean 1.16 g m^{-3}). The manganese concentrations in the Paokahu shallow ground water bores were generally within the range shown in the two background bores.

The two bores not located in the Te Hapara Sands aquifer (Bores 401A and 403A), had the lowest manganese concentrations. The high manganese concentrations in the Te Hapara Sands aquifer were not likely to be the result of sea water infiltration as typical sea water has a manganese concentration of 0.004 g m^{-3} (Anderson, 2003) which is lower than found in the Te Hapara Sands bores.

e. Copper

Dissolved copper concentrations (Figure 6.22) were below the detection limit of 0.005 g m^{-3} in bores 402A, 403A, 404B, 405A and 407A. Bore 401A (Shallow Estuarine Silts) was the only bore where the copper concentrations were consistently above the detection limit of the test.

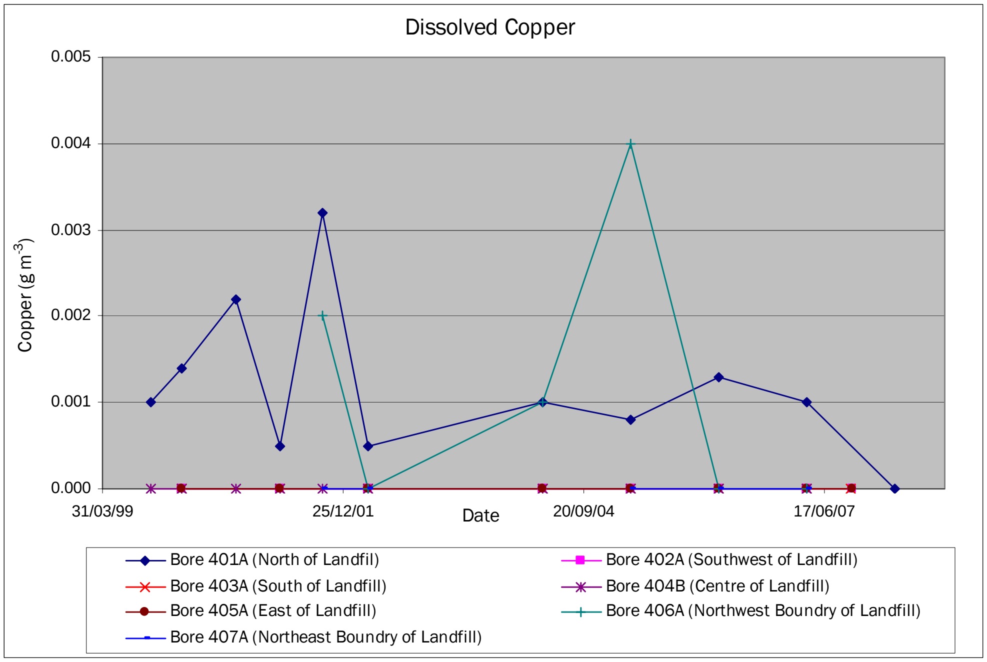


Figure 6.22: Dissolved copper concentrations in the shallow ground water bores.

f. Zinc.

Dissolved zinc concentrations (Figure 6.23) above the detection limit were only observed in Bores 401A, 405A and 406A. Bore 401A (Shallow Estuarine Silts) was the only bore where zinc concentrations were consistently above the detection limits of the test.

Zinc has previously been detected in the Te Hapara Sands aquifer at concentrations of <0.02 – 0.12 g m^{-3} (Brown & Elmsly, 1987). The zinc concentrations in the shallow ground water bores of the Te Hapara Sands aquifer were all lower than the zinc concentrations reported by Brown & Elmsly (1987).

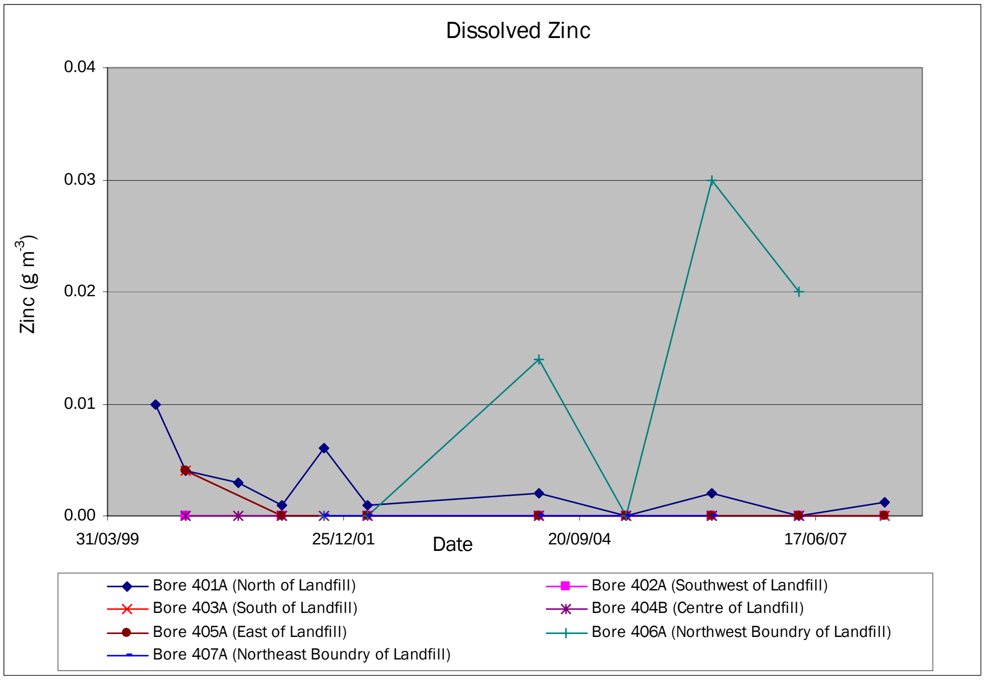


Figure 6.23: Dissolved zinc concentrations in the shallow ground water bores.

g. *Other Trace Elements*

Dissolved cadmium, chromium, lead, mercury, nickel and selenium were also measured. Selenium (detection limit <0.001 g m^{-3}) and mercury (detection limit <0.00008 g m^{-3}) concentrations were below the detection limit in all bores throughout the sampling period. Dissolved cadmium, chromium, lead and nickel were below the detection limit with a few exceptions (Figures 6.24, 6.25, & 6.26).

The lead detected in Bore 404B corresponds to the time when the leachate lead concentrations were greatest (1999/2000) and could indicate leachate entering the ground water under the landfill.

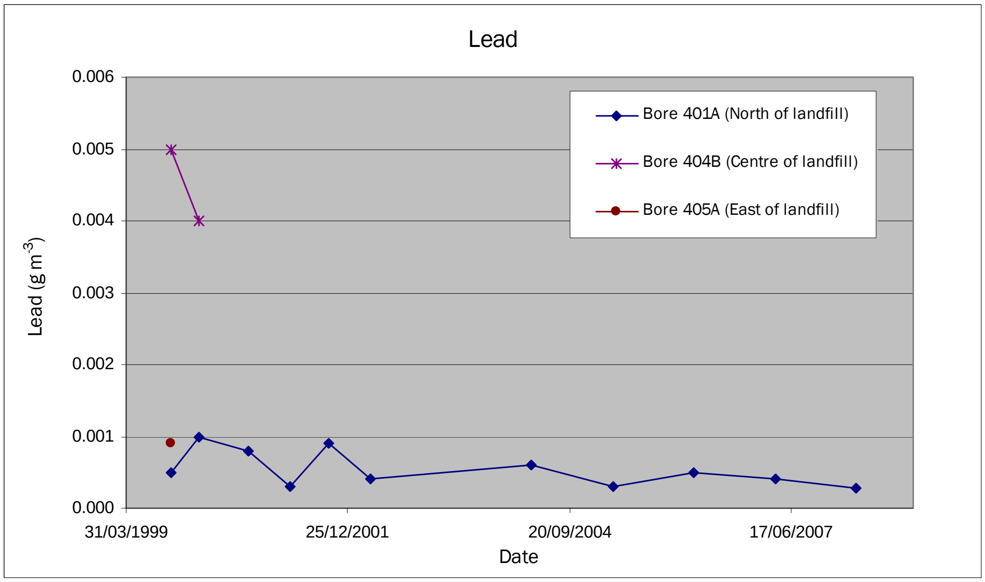


Figure 6.24: Dissolved lead concentrations in the shallow ground water bores.

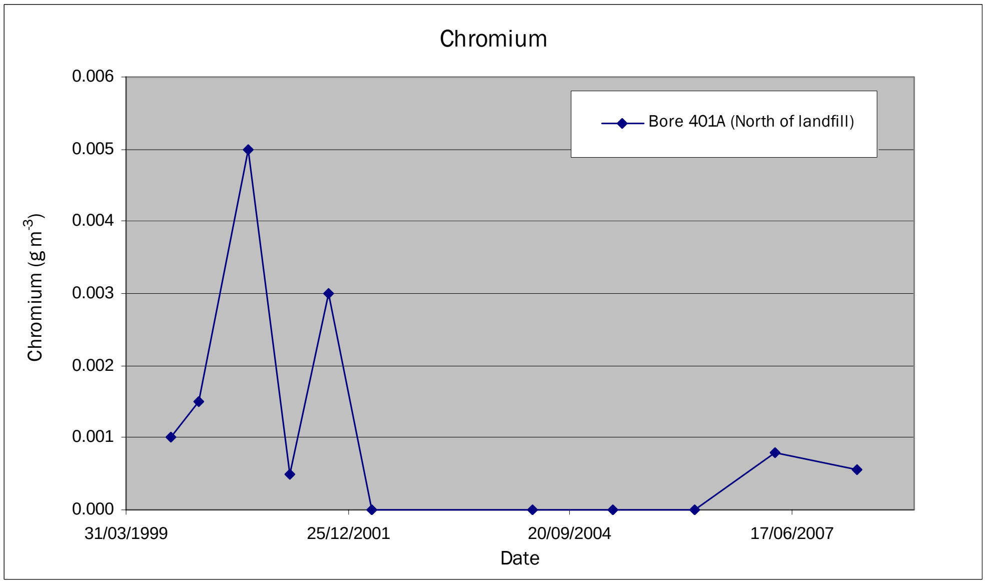


Figure 6.25: Dissolved chromium concentrations in the shallow ground water bores.

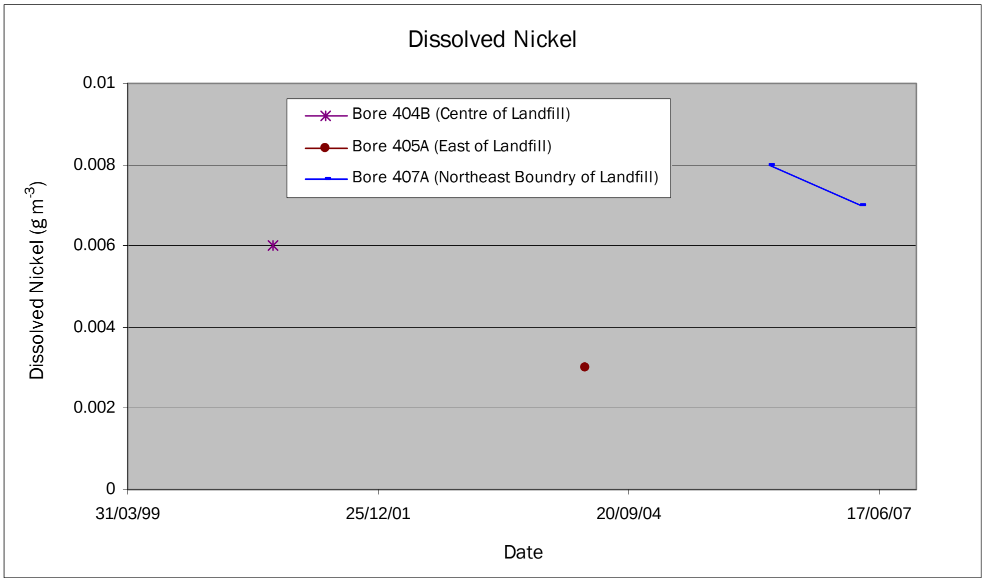


Figure 6.26: Dissolved nickel concentrations in the shallow ground water bores.

6.4.1.8 Semi Volatile Organic Compounds

No SVOCs were detected in any of the groundwater samples.

6.4.2 Deep Groundwater Bores

6.4.2.1 pH

The mean pH in the deep ground water bores was slightly alkaline and ranged from 7.1 in Bore 401B to 7.4 in Bore 403B (Figure 6.27). The pH was within the range typically found in the Te Hapara Sands aquifer, but overall the mean pH was lower than the pH in the shallow ground water bores.

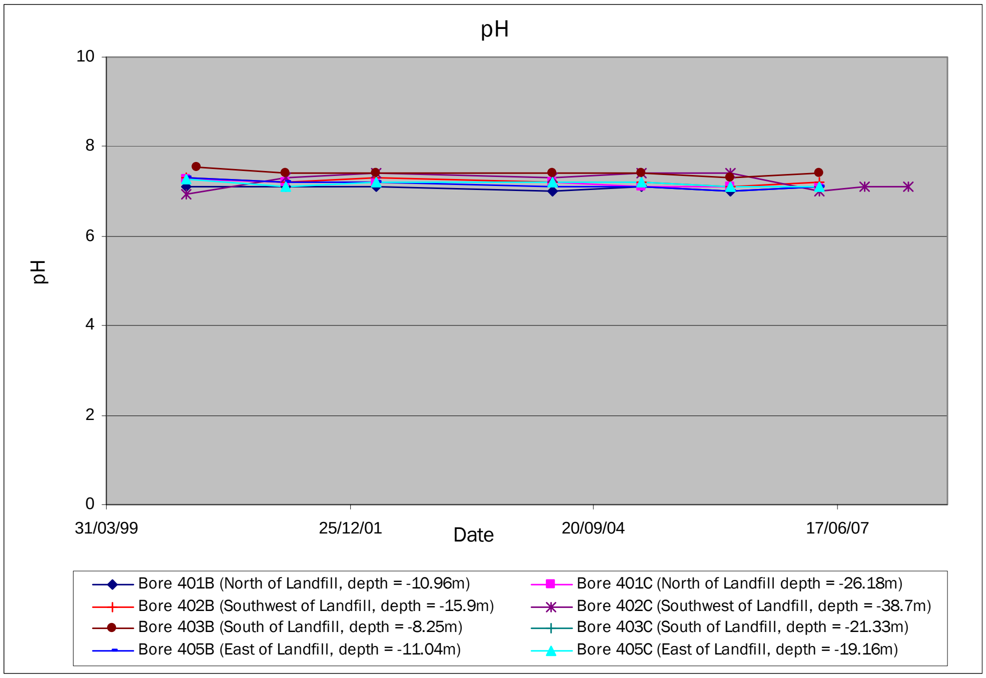


Figure 6.27: pH in deep ground water bores.

6.4.2.2 Electrical Conductivity

The mean EC in the deep ground water bores (Figure 6.28) typically ranged between 40 000 and 50 000 µScm^{-1} with the exception of Bore 402C which had a mean EC of 21 478 µScm^{-1}. The deep groundwater bores generally had a higher EC than the shallow water bores and the EC in Bores 403B and 403C was close to the EC found in sea water of 52 000 µScm^{-1} (Anderson, 2003).

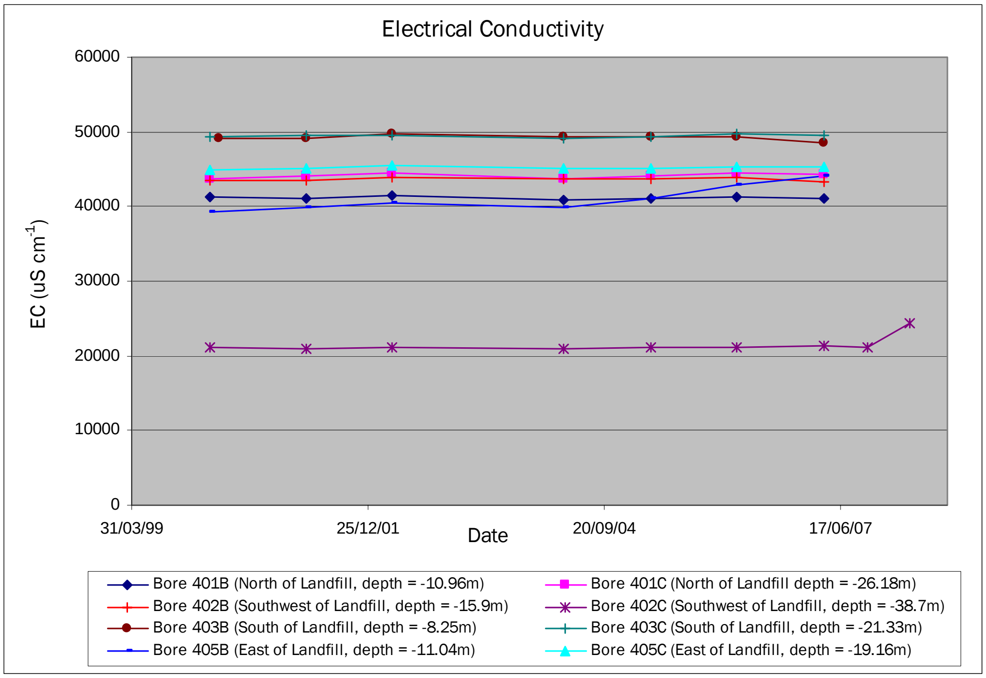


Figure 6.28: Electrical conductivity in deep groundwater bores.

6.4.2.3 Nitrogen

a. Ammonical Nitrogen

The ammonical nitrogen concentrations (Figure 6.29) were highest in Bore 402C which had a mean concentration of 31 g m^{-3}. The lowest ammonical nitrogen concentrations were recorded in Bore 403C which had a mean of 0.20 g m^{-3}. The mean ammonical nitrogen concentrations in the other deep ground water bores were all between 1 – 3 g m^{-3}. Generally the ammonical nitrogen concentrations in the deep groundwater bores were similar to ammonical nitrogen levels in the shallow groundwater bores.

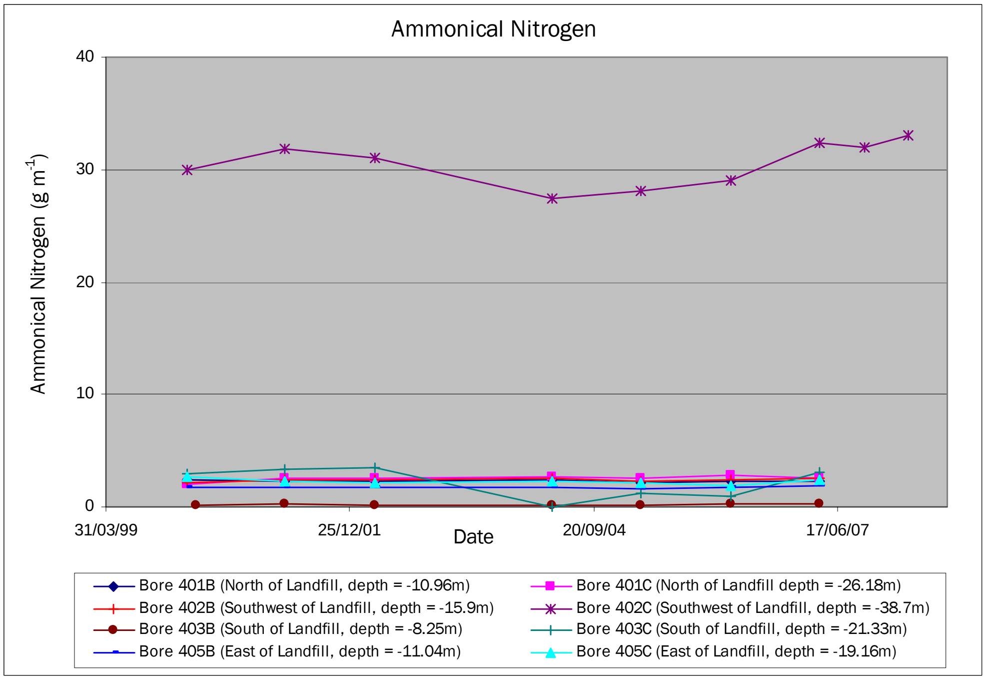


Figure 6.29: Ammonical nitrogen in deep ground water bores.

b. Nitrate

Dissolved nitrate concentrations (Figure 6.30) were generally between the detection limit of the test (0.002-0.05 g m^{-3}) to 0.3 g m^{-3}. The nitrate concentrations were generally low and similar to the nitrate concentrations in the shallow ground water bores. High nitrate concentrations were found in Bore 403B on three occasions. Bore 403B was the shallowest of the deep ground water bores with a depth of 8.25m below sea level. The high nitrate concentrations observed on 4/04/04 and 5/04/05 coincided with the two lowest recorded static water levels (no static water level was available for 05/04/06) indicating that the high nitrate concentrations were possibly associated with low ground water levels.

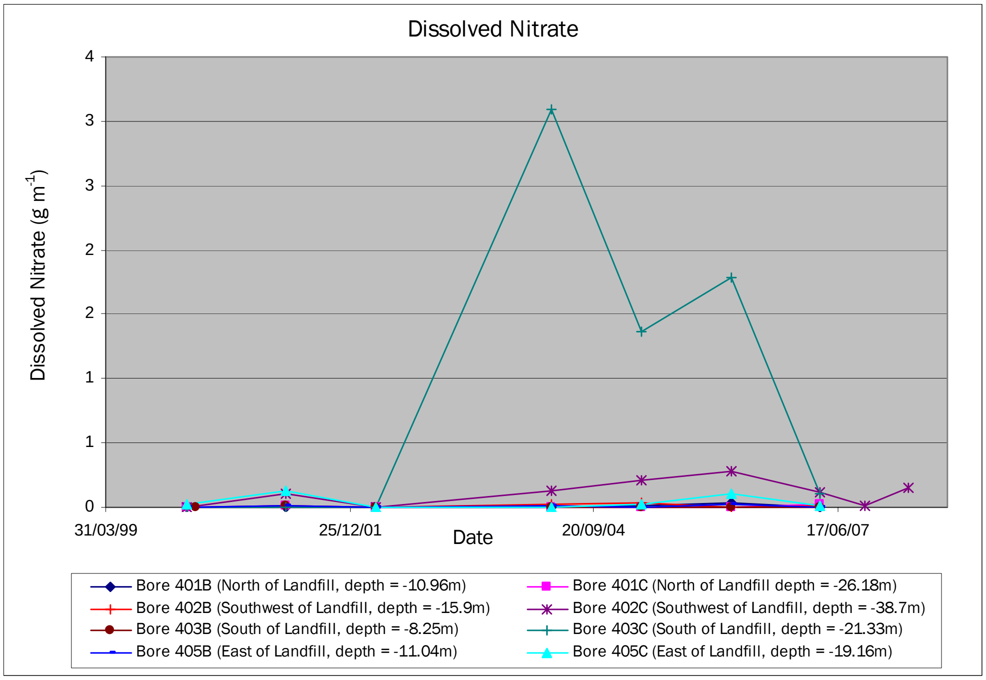


Figure 6.30: Nitrate in deep ground water bores.

c. Nitrite

Dissolved nitrite concentrations (Figure 6.31) were all between the detection limits (<0.002 to <0.01 g m^{-3}) and 0.1 g m^{-3}.

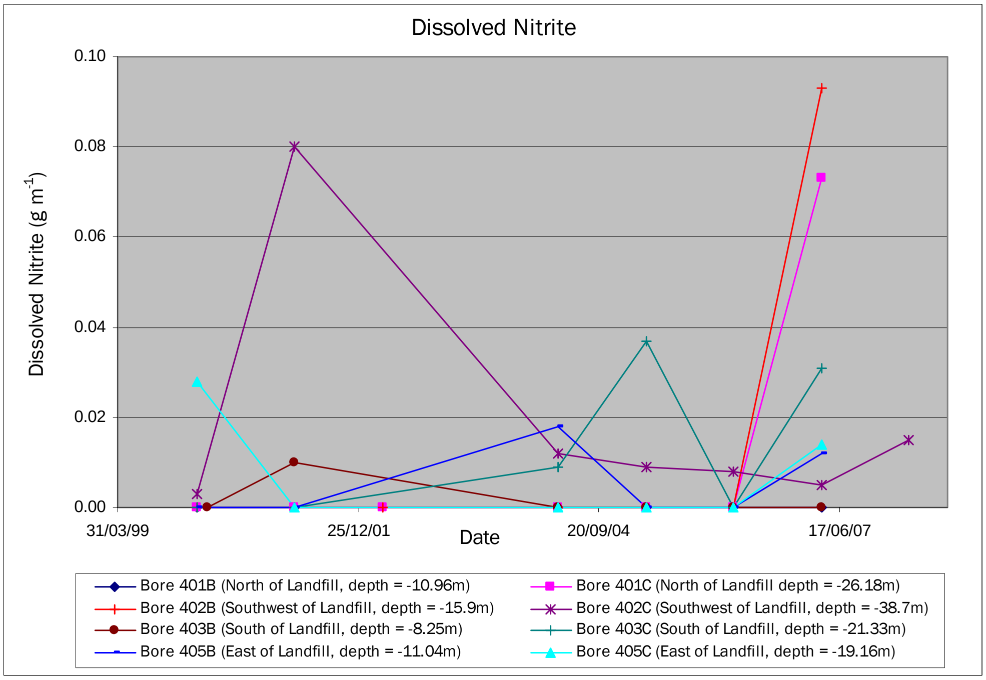


Figure 6.31: Nitrite concentrations in deep ground water bores.

6.4.2.4 Major Cations

a. Sodium

The dissolved sodium concentrations (Figure 6.32) in Bore 402C were lower than the other deep ground water bores. Bore 402C had a mean dissolved sodium concentration of 4358 g m^{-3} compared to means between 8200 and 10 285 g m^{-3} in the other deep ground water bores.

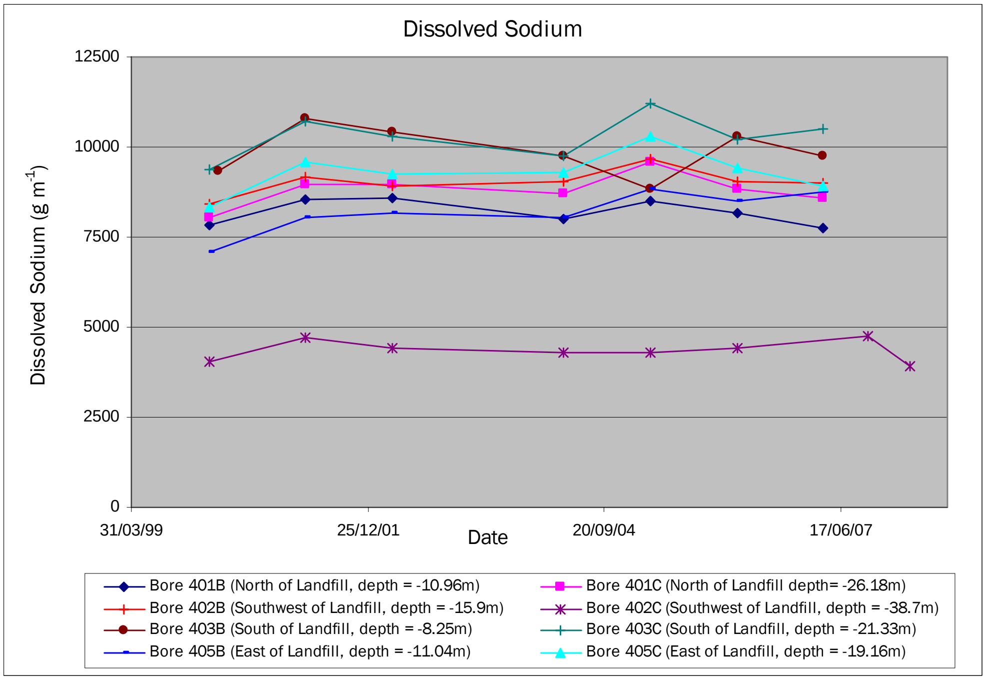


Figure 6.32: Dissolved sodium concentrations in the deep ground water bores.

The dissolved sodium concentrations in the bores were generally high, for example Bore 403C had a sodium concentration of 10 285 g m^{-3}, comparable to sea water which has a typical sodium concentration of 10 760 g m^{-3} (Anderson, 2003). The sodium concentrations in the deep ground water bores were higher than the shallow ground water and leachate bores. The plot of dissolved sodium shows a similar trend to the EC plot (Figure 6.28), with the concentrations in Bore 402C much lower than the other deep ground water bores but still high compared to the shallow ground water bores. The high sodium concentrations in the deep ground water bores indicate that sea water was the predominant influence on the water quality in the deep ground water bores, but is not as pronounced at greater depths in Bore 402C. Sea water typically has a Na:Cl ration of 0.56 and in the deep ground water bores (excluding Bore 402C) the Na:Cl ratios ranged from 0.53 – 0.56 which supports the theory that the high sodium and chloride concentrations were the result of sea water infiltrating into the aquifer. Bore 402C had a Na:Cl ratio of 0.74 which shows that the water in the bore was influenced by a different water source.

b. Potassium

The dissolved potassium concentrations (Figure 6.33) were also generally high; bores 403B and 403C had mean concentrations of about 350 g m^{-3} compared to 399 g m^{-3} (Anderson, 2003) typically found in sea water. The potassium concentrations in the deep groundwater bores were higher than the concentrations in the shallow ground water and leachate bores. The potassium concentrations in Bore 402C were lower than in the other deep groundwater bores which was consistent with the EC cation and anion concentrations in the deep ground water bores. The high potassium concentrations were likely to be from the infiltration of sea water. Sea water typically has a K:Cl ratio of 0.020 and with the exception of Bore 402C all the deep ground water bores had a K:Cl ratio between 0.015 – 0.019 indicating that sea water is present in the aquifer. Bore 402C had a K:Cl ratio of 0.025 indicating that the water contained less sea water.

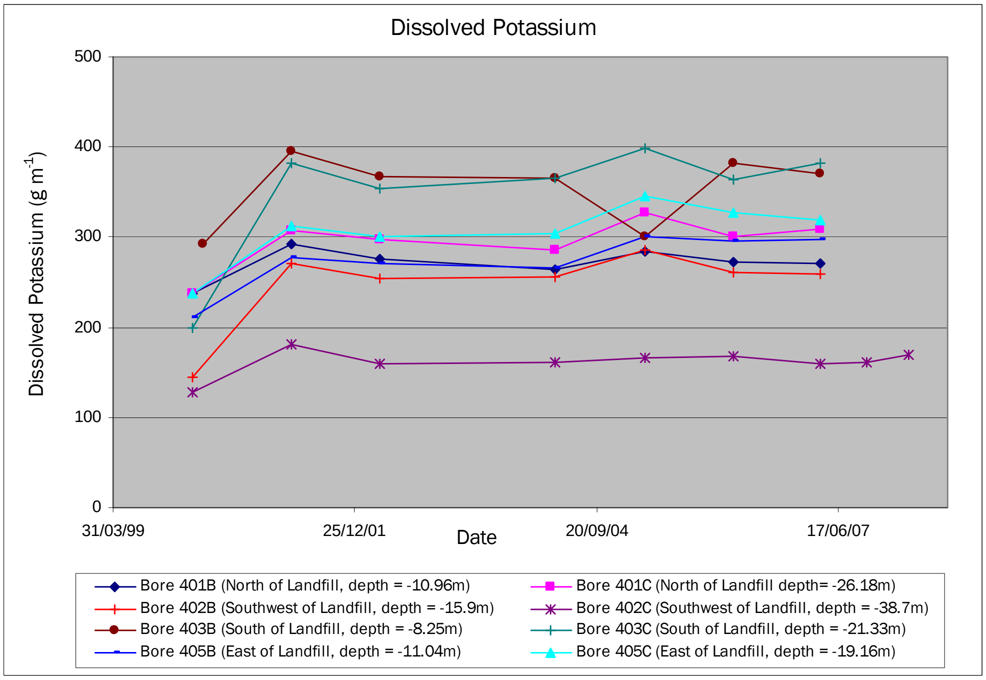


Figure 6.33: Potassium concentrations in the deep ground water bores.

c. Calcium and Magnesium

Dissolved calcium concentrations (Figure 6.34) were highest in Bore 401C which had a mean of 535 g m^{-3}. Bore 402C once again had the lowest mean concentration of 122 g m^{-3}.

Dissolved magnesium concentrations (Figure 6.35) were lowest in Bore 402C (mean = 364 g m^{-3}) compared to the other deep ground water bores (means between 998 and 1303 g m^{-3}).

The calcium and magnesium concentrations follow a similar trend to the other cations with Bore 402C having a lower mean concentration than the other deep ground water bores. The calcium and magnesium concentrations in the deep ground water bores were generally higher than in the shallow groundwater bores.

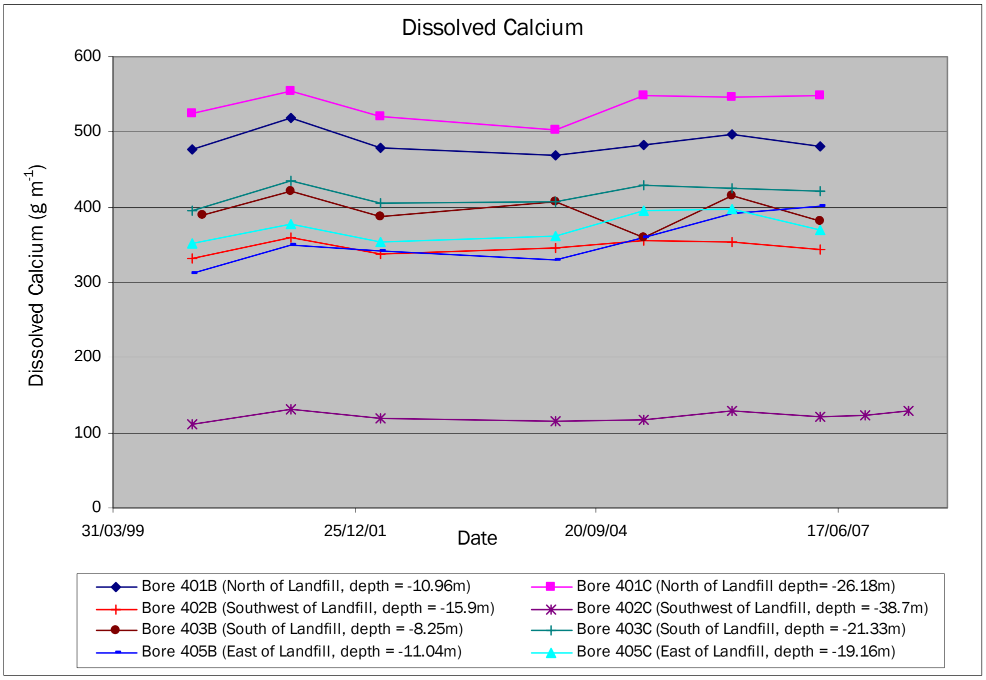


Figure 6.34: Dissolved calcium concentrations in the deep ground water bores.

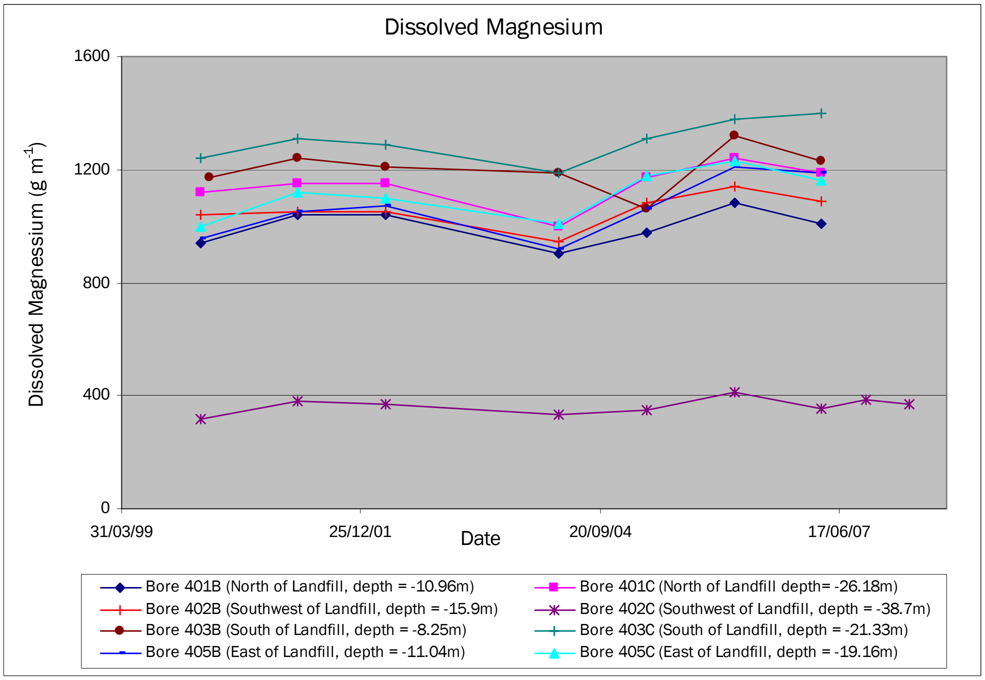


Figure 6.35: Dissolved magnesium concentrations in the deep ground water bores.

The mean calcium concentrations in bores 401C and 401B were higher than the concentrations found in typical sea water, which had a mean concentration of 416 g m^{-3} (Anderson, 2003). A calcium concentration higher than that of sea water indicates that not all of the calcium in the groundwater was due to the influence of seawater.

6.4.2.5 Major Anions

a. Chloride

Bore 402C had the lowest chloride concentrations (mean of 6536 g m^{-3}) and the other bores mean chloride concentrations ranged from 14 971 g m^{-3} in Bores 401B and 405B, to 18 600 g m^{-3} in Bore 403B (Figure 6.36).

The chloride concentrations follow the same trend as observed with the EC and cation concentrations. The chloride concentrations in the deep ground water bores were generally higher than those in the leachate bores and shallow groundwater bores. Sea water typically has a chloride concentration of 19 345 g m^{-3} (Anderson, 2003). All the bores had chloride concentrations below that of sea water, but were still comparatively high, for example Bores 403C and 403B had concentrations above 18 300 g m^{-3}. The high chloride concentrations were likely to be the result of sea water intrusion into the aquifer as shown by the similarity in the Na:Cl and K:Cl ratios between the deep ground water bores and that of sea water.

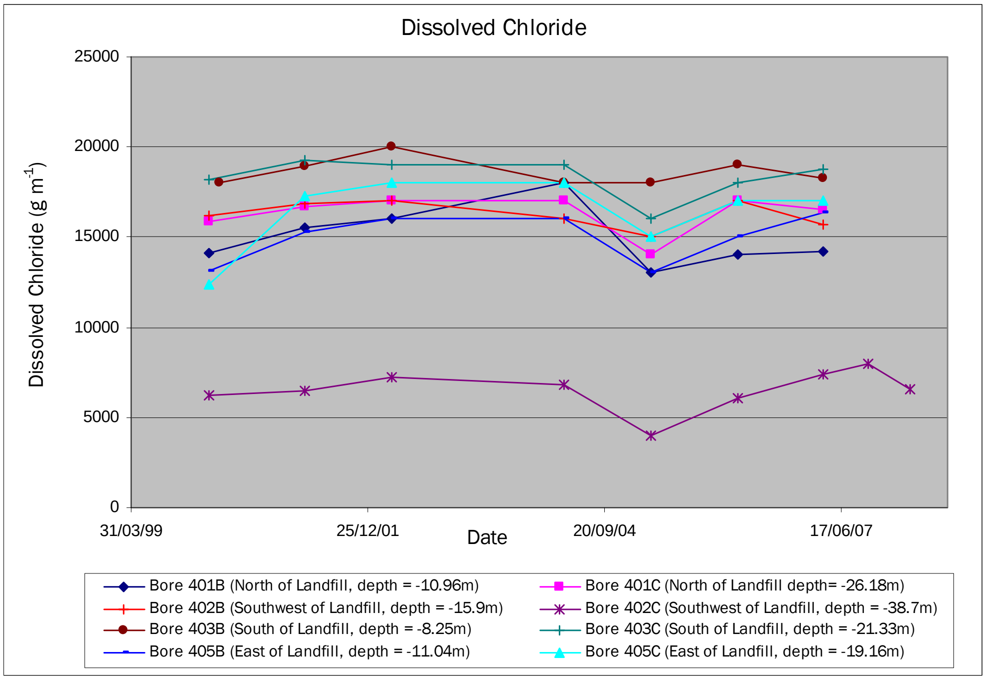


Figure 6.36: Chloride concentrations in the deep ground water bores.

b. Sulphate

On all but one occasion the dissolved sulphate concentrations (Figure 6.37) in Bore 402C were below the detection limit. The mean sulphate concentrations in the other bores ranged between 2094 g m^{-3} (Bore 401B) to 2710 g m^{-3} (Bore 403B).

The sulphate concentrations in the deep groundwater bores were generally high compared to the leachate and shallow ground water bores, with the exception of Bore 402C. The sulphate concentrations were generally similar to sea water (2700 g m^{-3}) (Anderson, 2003)) indicating sea water intrusion into the Te Hapara Sands Aquifer. The sulphate concentrations in Bore 402C were lower than the other deep ground water bores and were generally below the detection limit of the test. The low sulphate concentrations followed the trend observed in Bore 402C with the bore having lower cation and anion concentrations compared to the other deep ground water bores, indicating that sea water was not the dominant factor in the water quality in this bore.

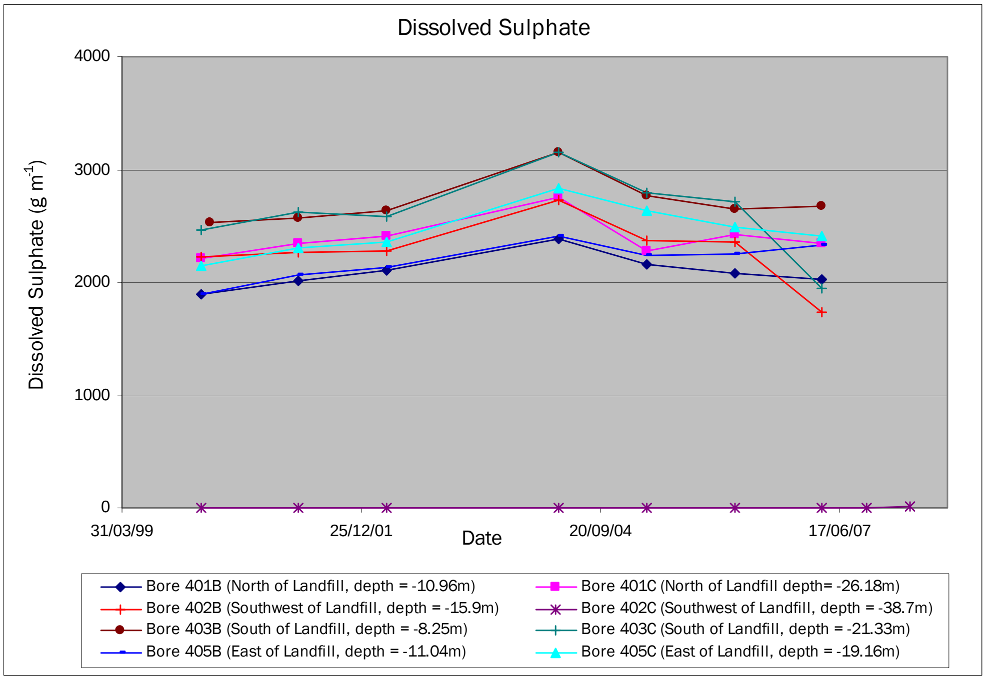


Figure 6.37: Sulphate concentrations in the deep ground water bores.

c. Alkalinity

The highest alkalinity concentrations were observed in Bore 402C (Figure 6.38) and ranged from 2250 to 2600 g m^{-3}. The alkalinity in the other bores was lower and ranged between 130 – 530 g m^{-3}.

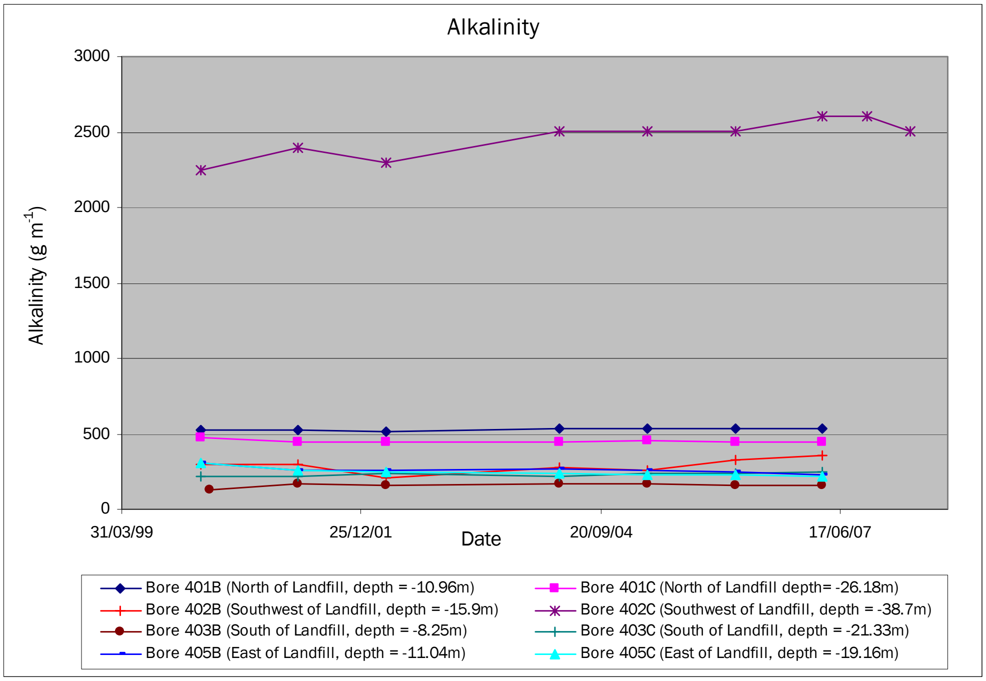


Figure 6.38: Alkalinity concentrations in the deep ground water bores.

6.4.2.6 Trace Elements

a. Boron

Mean boron concentrations (Figure 6.39) ranged from 2.65 g m^{-3} in bores 401B and 401C to 4.0 g m^{-3} in bores 403B. The boron concentrations in the deep ground water bores were higher than those recorded in the leachate bores, but not as high as typical sea water (4.5 g m^{-3} (Anderson, 2003)). The highest boron concentrations were observed in Bores 403B and 403C, which are the closest bores to the coast and also had the highest sodium and chloride concentrations. Bore 402C which had lower EC, cation and anion concentrations than the other bores had a mean concentration within the range of the other deep groundwater bores, indicating that not all of the boron content of the groundwater comes from sea water.

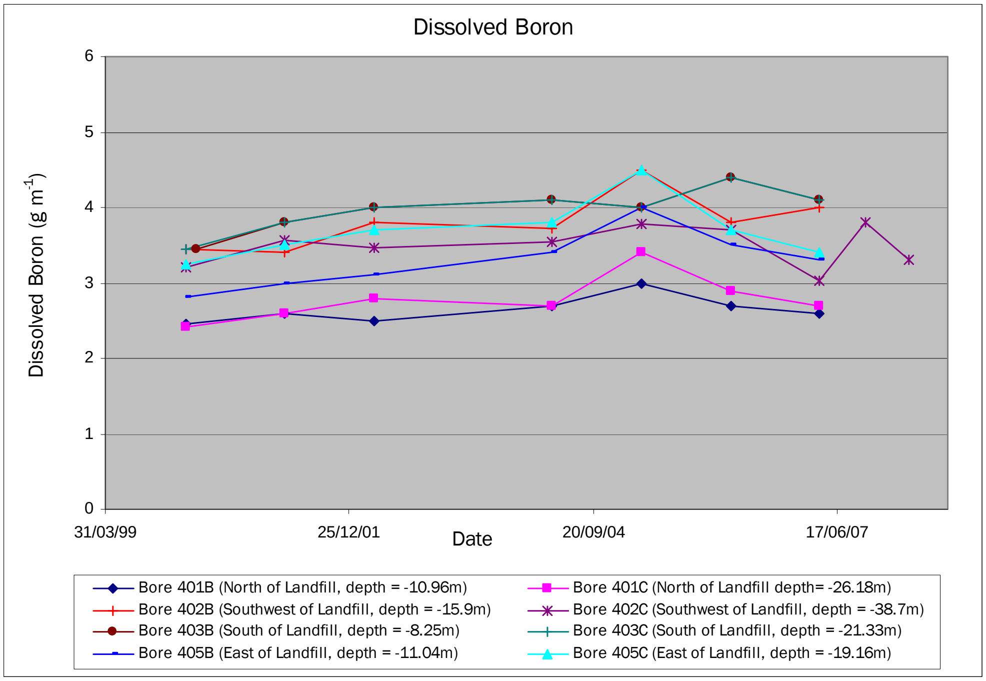


Figure 6.39: Dissolved boron concentrations in the deep ground water bores.

c. Iron

Dissolved iron concentrations (Figure 6.40) all peaked with the samples taken on the 01/04/04. In the majority of the other samples taken, iron concentrations were below the detection limits of the tests.

The iron concentrations in the deep groundwater bores were generally low and below the detection limit in over 50% of the samples. There was an increase in iron concentrations in all samples taken on 01/04/04, which doesn’t fit the general trend for bores. The same sampling and testing procedures were used to calculate the dissolved iron concentrations as on previous occasions. The increase was not observed in the shallow groundwater bore data which was sampled and analysed at the same time using the same procedure which would indicate the unusually high iron concentrations were not due to a laboratory error. The static water levels measured in April 04 were all below the mean levels, but lower static water levels had occurred on other occasions without any corresponding increase in iron concentrations.

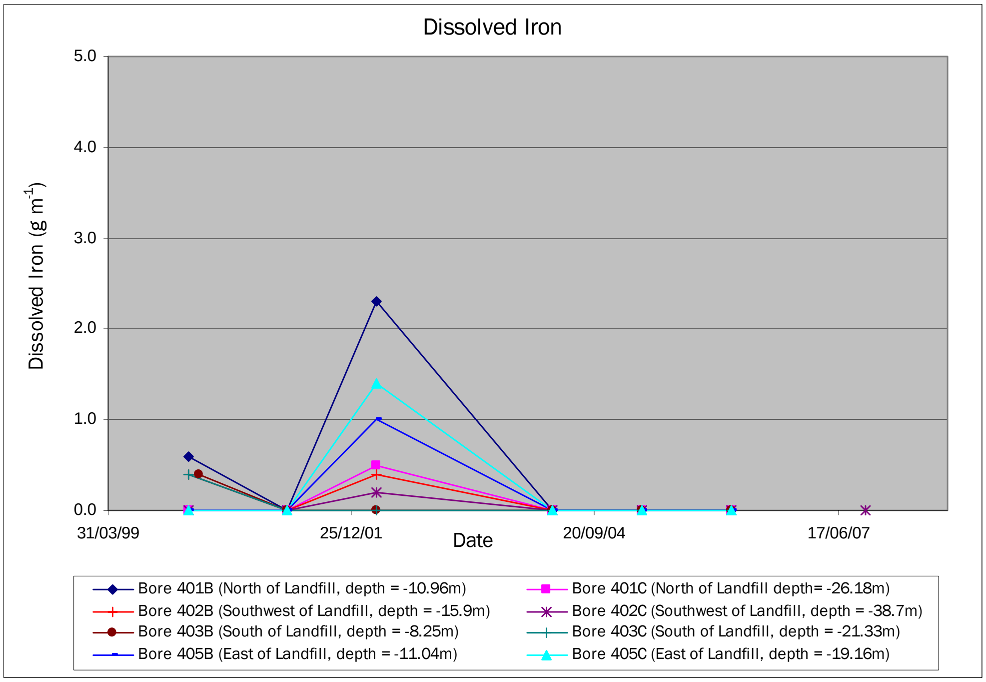


Figure 6.40: Dissolved iron concentrations in the deep ground water bores.

6.5 Bore Static Water Levels

The static water levels in both shallow (Figure 6.41) and deep groundwater (Figure 6.42) bores show a seasonal pattern with higher static water levels in the winter months and lower levels in the summer months. Monthly monitoring stopped in 2005 and has been carried out at irregular intervals since, which has made it more difficult to determine any long term trends.

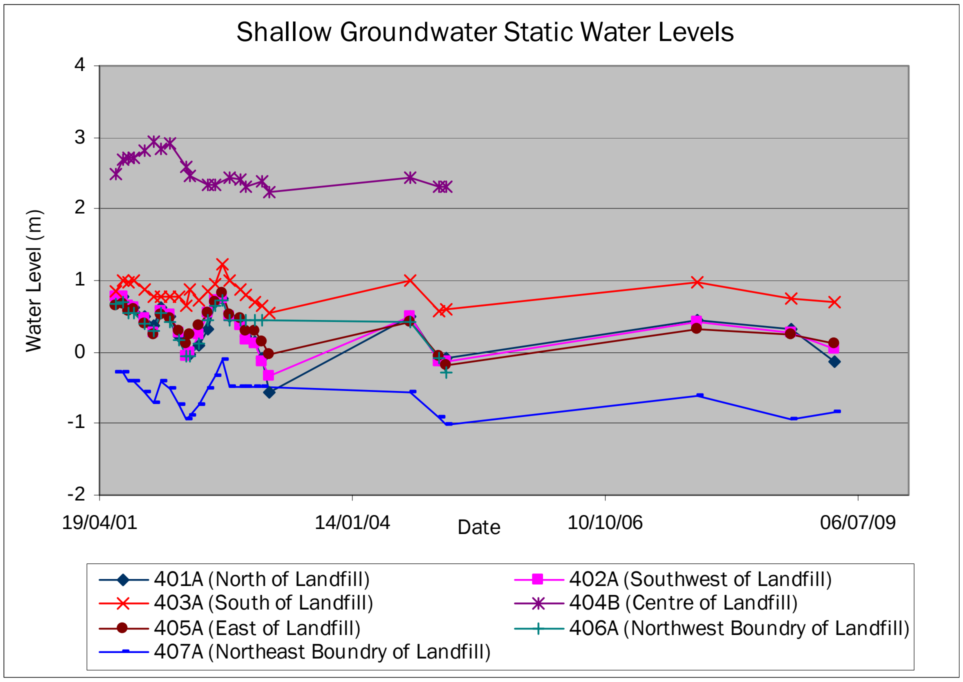


Figure 6.41: Shallow ground water bore static water levels.

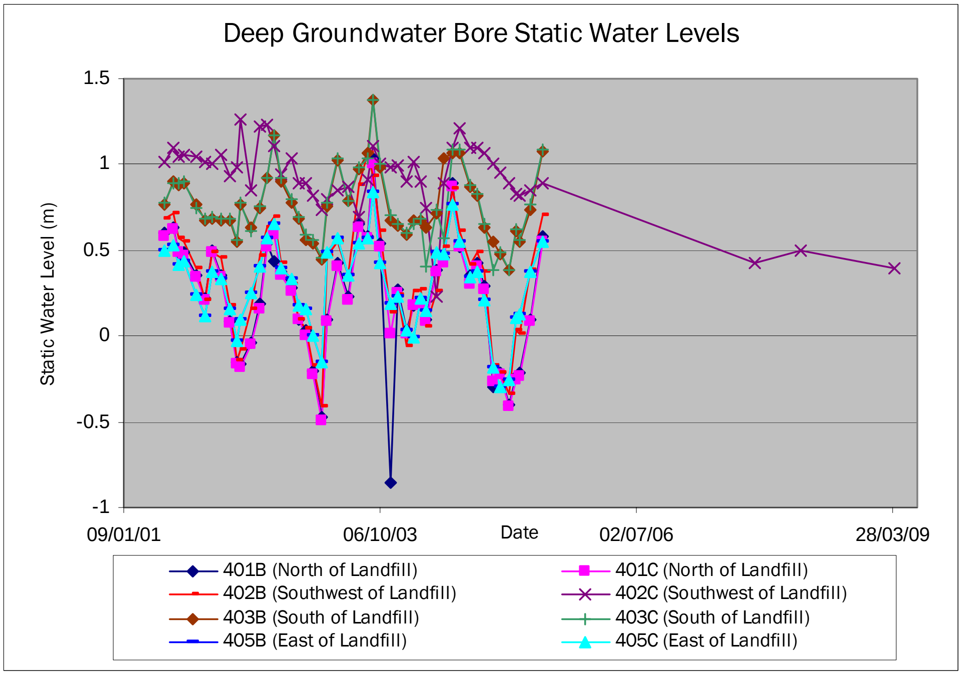


Figure 6.42: Deep ground water bore static water levels.

6.6 Comparison to Drinking Water Standards

The ground water present in the monitoring bores was compared to the Maximum Acceptable Values and Guideline values set out in the New Zealand Drinking Water Standard (Ministry of Health, 2005) and the Live Stock Drinking Water Standards (ANZECC & ARMCANZ, 2000) (Table 6.2).

When the bore water is compared to the maximum acceptable values set out in the New Zealand Drinking Water Standard (Ministry of Health, 2005) it was generally poor with only three bores, 401A (Shallow Estuarine Silts) and 403A (Recent Sands deposits) and 405A (shallow bore, Te Hapara Sands aquifer) meeting all standards. In most instances the boron and manganese concentrations were above the maximum acceptable value.

As well as the maximum acceptable values the New Zealand Drinking Water Standards (Ministry of Health, 2005) also gives guideline (non-regulatory) values. When the bore water was compared to the guideline values only bore 403 met the guideline values. The sodium, chloride and iron concentrations were generally too high to meet the guidelines. The high sodium and chloride concentrations would make the water taste salty, while the high iron concentrations can lead to staining of laundry and sanitary ware (Ministry of Health, 2005).

All the shallow bores except Bore 401A meet the live stock drinking water guidelines for the parameters tested, while only bore 402C met the Live Stock Drinking Water guidelines in the deep groundwater bores. The bores which failed to meet the standard did so because the sulphate concentrations were too high. The ANZECC & ARMCANZ (2000) Stock Drinking Water guidelines recommend sulphate concentrations of less than 1000 g m^{-3} as concentrations in excess of 1000 g m^{-3} can cause diarrhoea in livestock.

Table 6.2: Comparison between groundwater bores and the New Zealand Drinking Water Standard (2005), and the Live Stock Drinking Water Standards.

Element	**Drinking Water Standards[1] (DWS)**		**Livestock Drinking Water Standards[2] (LDWS)**	**Comments**
	Maximum acceptable value (g m^{-3})	Guideline Value (g m^{-3})	Guideline Value (g m^{-3})	
boron	1.4	-	5.0	Bores 401A, 403A & 405A were below the DWS guideline
cadmium	0.004	-	0.01	All Bores tested were below limits
chloride	-	250	-	Bores 401A & 403A were below the DWS guideline value
chromium	0.05	-	1.0	All Bores tested were below limits
copper	2.0	1	0.5	All Bores tested were below limits
iron		0.2	-	Bore 401A above limit
lead	0.01	-	0.1	All Bores tested were below limits
manganese	0.4	0.04	-	Bores 401A, 403A & 405A were below the DWS guideline value
mercury	0.007	-	0.002	All Bores tested were below limits
nickel	0.08	-	1.0	All Bores tested were below limits
nitrate (short term)	50	-	400	All Bores tested were below limits
nitrite (long-term)	0.2	-	-	All Bores tested were below limits
nitrite (short-term)	3.0	-	-	All Bores tested were below limits
sodium	-	200	-	Bore 403A was below the DWS guideline value
sulphate	-	250	1000	Bores 401A, 403A & 405A were below the DWS guideline. Bores 402A, 403A,404B,405A, 406A & 407A were below the LDWS guideline
zinc	-	1.5	20.0	All Bores tested were below limits

[1] New Zealand Drinking Water Standard, 2005

[2] ANZECC & ARMCANZ; 2000

6.7 Discussion

6.7.1 Groundwater Composition

The analysis of the groundwater monitoring bores showed there was considerable variation in water composition between bores, with no obvious pattern between bore location and bore water composition. However when the bore water composition was examined in conjunction with the bore depth and aquifer material, the following pattern was observed:

1. The shallow aquifers (Bores 401A and 403A) located in the Recent Sands and Shallow Estuarine Silts deposits contained 'fresh water', shown by their low EC and low cation, chloride, sulphate and concentrations. Bores 401A and 403A appear to be located in two separate aquifers, as there was a difference in water composition between the bores.

2. The water in the Te Hapara Sands aquifer, located immediately below the Recent Sands and Shallow Estuarine Silts deposits, generally decreased in quality with depth. The EC, sodium, potassium and chloride concentrations generally increased with depth down to approximately 30-35m below sea level. The high EC, cation and anion concentrations and the Na:Cl and K:Cl ratios indicated sea water intrusion into the aquifer.

3. Below 30-35m (Bore 402C) the water become less saline as indicated by a decrease in EC, sodium, potassium and chloride concentrations and an increase in ammonical nitrogen concentrations.

The results of the groundwater analysis support the results of the earlier study undertaken by Nelson (1994) in which it was suggested that the ground water in the Paokahu Area consists of very shallow zones of fresh water near the surface, a deeper mixed zone dominated by salt water and a trend to fresh water with depth.

Taylor (1994) suggested the Makauri Gravels aquifer discharges occur through slow upward leakage under artesian pressure at its coastal end. Discharge of fresh groundwater from the Makauri aquifer into the base of the Te Hapara Sands aquifer would explain the trend towards fresher water with depth in to the Te Hapara Sands aquifer. Upward movement of groundwater from the Makauri aquifer into the base of the Te Hapara aquifer has also been suggested by Nelson (1994) who found that the head in deep bores in the Paokahu area increased with depth. The increase in head with depth indicated that the landfill was located in a groundwater discharge zone, which was attributed to upward leakage from the Makauri Gravels aquifer.

A second pattern that was distinguished from looking at the bore data, was that at approximately at any given depth the EC, sodium and chloride concentrations were greater in the bores nearer the coast and concentrations gradually decreased in the bores as they progressed inland.

At approximately the same depth (excluding Bore 402C which contains freshwater), EC, sodium and chloride concentrations in the two groups of bores closest to the sea (the 402, and 403 series bores) were significantly higher ($P<0.001$) than the concentrations in the bores further inland (the 401 and 405 series bores). The lower EC, sodium and chloride further inland would indicate in the Paokahu locality that the ground water flow is in a northward direction, away from the coast, and the salt water becomes more dilute as it moves inland. A northward movement of ground water is opposite to the regional flow which is generally southward toward the coast. Nelson (1994) also suggested that the direction of groundwater flow in the Paokahu area was away from the coast towards the Awapuni Drain and was likely to be caused by the low water (water level equals mean low tide level) levels maintained in the Awapuni Drain through use of the tide gate system.

6.7.2 Leachate Contamination of Groundwater

If leachate is escaping from the base of the landfill into the surrounding groundwater it would form a leachate plume in the surrounding groundwater. Leachate plumes are typically small with lengths of less than 1000m and are narrow in width, typically no wider than the landfill (Christensen, *et al.* 2001). The shape of leachate plumes is primarily determined by the direction of flow of the groundwater; therefore if a leachate plume was present it would most likely be detected in the bores north of the landfill (Bores 401A, 401B, 401C, 406A and 407A) as this is the direction of the groundwater flow. However, movement of a leachate plume can differ from the direction of local groundwater (Christensen, *et al.* 2001).

There are several chemical indicators which can be used to detect the presence of leachate in the groundwater. For example, the EC of leachate is typically higher than that of groundwater so an increase in EC in monitoring bores can be used to indicate leachate contamination (Mack, 1993). Chloride is another element which can be used to indicate leachate contamination of groundwater as it doesn't undergo any physical or chemical reactions in groundwater and is considered inert (Christensen, *et al.* 2001). However, the high background EC, cation and anion concentrations found in most of the Paokahu monitoring bores, caused by saltwater intrusion in the groundwater, means that the groundwater EC anion and cation concentrations alone can't be used to determine if leachate is escaping from the landfill and entering the surrounding groundwater. In the two bores which contain 'freshwater' (401A and 403A) the EC, cation and anion concentrations were low and generally within the ranges found in the background bores (BG1 and BG2) which indicate no leachate contamination.

Leachate ammonical nitrogen concentrations (mean = 400 g m^{-3}) were significantly higher (P<0.001) than that present in the groundwater; therefore if leachate contamination had occurred then increased ammonical nitrogen concentrations may be present in the some of the bores, particularly the bores down stream of the landfill (Bores 401A, 401B, 401C, 406A and 407A). However the ammonical nitrogen concentrations in the groundwater bores was generally similar ranging between 1-3 g m^{-3} in all the Te Hapara Sands aquifer bores (except bore 402C) and less in the Recent Sands deposits. The highest ammonical nitrogen concentrations (except bore 402C) were found south (up stream) of the landfill. Although the ammonical nitrogen concentrations were slightly higher than that of the background bores the consistent concentrations across the bores doesn't indicate leachate contamination in any one bore.

The only bore with elevated ammonical nitrogen concentrations is 402C (the deepest bore); the high concentrations can be attributed to the upward movement of groundwater from the Makauri Gravels aquifer into the Te Hapara Sands aquifer. If the elevated ammonical nitrogen concentrations in 402C were caused by leachate contamination, the other two shallower bores at this location (402A and 402B) should also contain elevated ammonical nitrogen concentrations, which is not the case.

The presence of heavy metals in the groundwater can also be used as an indicator to leachate contamination of groundwater. However as the Paokahu leachate contains low metal concentrations any leachate contaminated groundwater may not contain metals at concentrations high enough to be detected after dilution occurs when leachate mixes with groundwater.

Bore 401A (north of the landfill, a shallow bore in the Recent Sands deposits) was the only bore in which metals were consistently detected. The presence of the metals was not associated with an increase in EC, anions, cations or ammonical nitrogen in the bore, which are all present in high concentrations in the leachate. Bore 401A had very high phosphate levels compared to the other bores, higher than that present in the leachate. The presence of metals could be the result of contaminants present in phosphate fertilizer. Although metals were detected in Bore 401A the concentrations were below the limits given in the New Zealand Drinking Water standard (Ministry of Health, 2005).

Lead was detected in low concentrations in Bore 404B (centre of landfill) on two consecutive occasions between1999/2000 and coincided to the time when the leachate lead concentrations were highest. Lead was not detected again after 1999/2000. The presence of lead could indicate leachate contamination of the groundwater in Bore 404B and the location under the centre of the landfill would make this one of the bores most likely to show signs of leachate contamination. Bore 404B also had the highest ammonical nitrogen concentrations of the shallow groundwater bores, and a higher potassium concentration relative to EC, chloride and other cations compared to the other shallow groundwater bores. However, all the elements tested for in Bore 404B were generally within the range found in the other similar bores so the higher concentrations could be due to the variations observed between bores. Therefore it can not be definitively concluded that leachate contamination is occurring in Bore 404B and further investigation may be needed. If leachate is in the groundwater near Bore 404B it could be caused by the bore not being sealed properly during installation which could allow some localised leachate leakage to occur around the bore casing.

In summary, when the chemical composition across all of the groundwater in all the monitoring bores is examined there is no conclusive evidence of a leachate plume being generated from the landfill. There may be some leachate in Bore 404B under the centre of the landfill as it had slightly higher ammonical nitrogen and potassium concentrations, and lead had been detected in the past, but alone this is not enough to confirm the presence of leachate in the groundwater. The slightly higher ammonical nitrogen and potassium concentrations may just be due to the variations between bores. If leachate is present in Bore 404B there is no evidence in the other monitoring bores that leachate is moving away from the landfill forming a leachate plume.

6.7.3 Comparison to the Drinking Water Standard.

When compared to the New Zealand Drinking Water Standards (Ministry of Health, 2005), the water quality in the ground water bores was generally poor. In the shallow bores only Bores 401A, in the Shallow Estuarine Silts, would meet all the maximum acceptable and guideline values. The nitrate concentration in Bore 403A, the other bore not located in the Te Hapara Sands aquifer, is above the long term maximum acceptable value given in the New Zealand Drinking Water

Standard (2005). The boron content in the shallow bores in the Te Hapara Sands aquifer bores make them unacceptable for use as drinking water.

None of the deep bores would meet the criteria set in the New Zealand Drinking Water Standards (Ministry of Health, 2005), as the boron content is also above the maximum acceptable value of 1.4 g m^{-3}. The sodium, chloride and sulphate content in all of the deep bores were also above the guideline values based on the aesthetic qualities category.

6.8 Conclusions

Examination of the groundwater monitoring bores shows that salt water intrusions into the Te Hapara Sands aquifer was the a major influence on the water quality in the Paokahu area. Bore depth and aquifer parent material were the main determinant of water composition within the bores where:

- the shallow aquifers located in the Recent Sands/ Shallow Estuarine Silts deposits contained fresh water,
- the water in the Te Hapara Sands aquifer generally became more saline with depth down to approximately 30-35m below sea level, and
- below 30-35m the water become less saline, probably due to upwelling from the Makarui Gravels aquifer.

The other major factor affecting water composition was the proximity of the bores to the coast, as the bores on the seaward side of the landfill were affected more by salt water intrusion than the bores inland from the landfill.

The groundwater quality in general was poor with only one bore (401A) meeting all the maximum acceptable levels and guidelines values given in the New Zealand drinking Water Standard (Ministry of Health, 2005). Over half of the bores, mainly the deeper ones, also failed to meet the ANZECC & ARMCANZ (2000) Stock Drinking Water guidelines because of high sulphate concentrations.

The results showed that the landfill was not having an effect on the surrounding groundwater. There was no evidence of a recognisable leachate plume being generated from the landfill. Bore 404B was the only bore which could be showing signs of leachate contamination. Bore 404B had slightly higher ammonical nitrogen concentrations compared to the shallow groundwater bores and a higher potassium concentration relative to EC, chloride and other cations. However, the slightly higher ammonical nitrogen and potassium concentrations could be due to the variability in groundwater composition between bores as the ammonical nitrogen and potassium were within the range recorded in the other monitoring bores. Further monitoring of Bore 404B could be undertaken to determine if leachate is contaminating the groundwater. However, if leachate was escaping it is unlikely that anything could be done to prevent future leachate leakage other than keep the leachate levels as low as possible within the landfill, which is currently the practice.

If leachate was getting into the groundwater underneath the landfill it is unlikely that it would have a major effect on the groundwater quality as the groundwater generally already has high background concentrations of cations and anions; in most cases higher concentrations than that present in the leachate bores. The main concern would be elevated nitrogen in the groundwater.

The water present in the Recent Sands and Shallow Estuarine Silts deposits wasn't influenced by salt water intrusions so the water quality in these aquifers would be the most affected by leachate contamination. The water in the Recent Sands and Shallow Silts deposits had shown signs of leachate contamination in the past attributed to leachate seeping from the side slopes of the landfill and flowing into the perimeter stormwater drains from where it drained into the groundwater. Leachate seeps no longer occur as the landfill is capped and there is no sign of leachate contamination in the bores located in the Recent Sands and Shallow Estuarine Silt deposits.

The presence of the landfill should not affect any potential groundwater use as the water would have only limited potential uses. The water quality is already low and generally not suitable for human or livestock consumption. The groundwater would also have limited value for use in irrigation schemes due to the high soluble salts content which could affect soil structure and plant growth. Extensive pumping of the groundwater in the Paokahu area should also be avoided, as it will lower the local water table causing greater seawater intrusions in the Te Hapara Sands aquifer further reducing the ground water quality in the area.

Chapter 7. Soil Analysis-

7.1 Introduction

Before this study was undertaken, the landfill's leachate bore monitoring data had indicated that the sustainability of the leachate irrigation could be at risk due to the high sodium concentrations present in the leachate, as high sodium concentrations can have a detrimental effect on soil structure and plant growth. Therefore it was decided to undertake some soil testing to determine if the high leachate sodium concentrations were negatively impacting on the structure of the soil cap. Another potential problem associated with leachate irrigation schemes is the accumulation of heavy metals in the landfill cap; therefore a heavy metal analysis was also undertaken.

This chapter discusses the results of the soil sampling programme that was undertaken at the landfill. Soil cation and metal concentrations and the dispersion index were determined for sites within the landfill and surrounding area.

7.2 Experimental Design and Methods

7.2.1 Introduction

Fifteen sites were sampled; ten sites on the irrigated area of the landfill cap, three from non-irrigated areas of the landfill cap and two from outside the landfill.

The landfill cap consists of a 10 cm layer of bark placed directly over the refuse, overlaid with a 25 cm thick layer of soil, 15 cm of beach sands and finally a 20 cm thick layer of top soil. On the landfill, samples were taken from the surface topsoil horizon (surface horizon, 0-20 cm) and the subsurface soil horizon (subsurface horizon) at a depth of approximately 40-50 cm. Samples were not taken from the sand horizon as this was unlikely to be affected by soil structural problems. Two sites outside the landfill were sampled to give a comparison between cation concentrations in the landfill and the surrounding landscape. Samples were taken from the sand dunes (Opoutama Sands) and the Gavin's Block, a low lying area to the west of the landfill (Makaraka Clay Loam, Saline Phase). Samples were collected at the soil surface (0–50 cm) and at a depth of between 40-50 cm. Samples were collected in October 2008 and again in July 2009.

7.2.2 Site Selection

To select the sampling locations in the irrigated areas, a grid was drawn over a map of the landfill dividing the top surface into 15 approximately equal sized areas. Ten grids were chosen at random (pulling numbers out of a hat) and samples were taken from within each grid square. The position of each sampling point was recorded using a GPS (Figure 7.1).

Each of the three batter slopes was divided into three equally sized areas in a similar fashion to that used in the irrigated area and one was chosen at random from each of the batter slopes (pulling numbers out of a hat) and samples were taken approximately half way up the slope. The precise locations were recorded using a GPS. In the two areas outside the landfill a site was found that was representative of the surrounding area and samples were taken and the location recorded using a GPS.

7.2.3 Sampling Method

7.2.3.1 Surface Horizon

Seven cores were taken at each location at the soil surface from within a 0.5 metre radius. Cores were cut using a soil core cutter (Figure: 7.2). The core cutter produced a soil core 20 mm in diameter and 50 mm long. All seven cores were placed in a plastic sample bag and then broken up and mixed by hand.

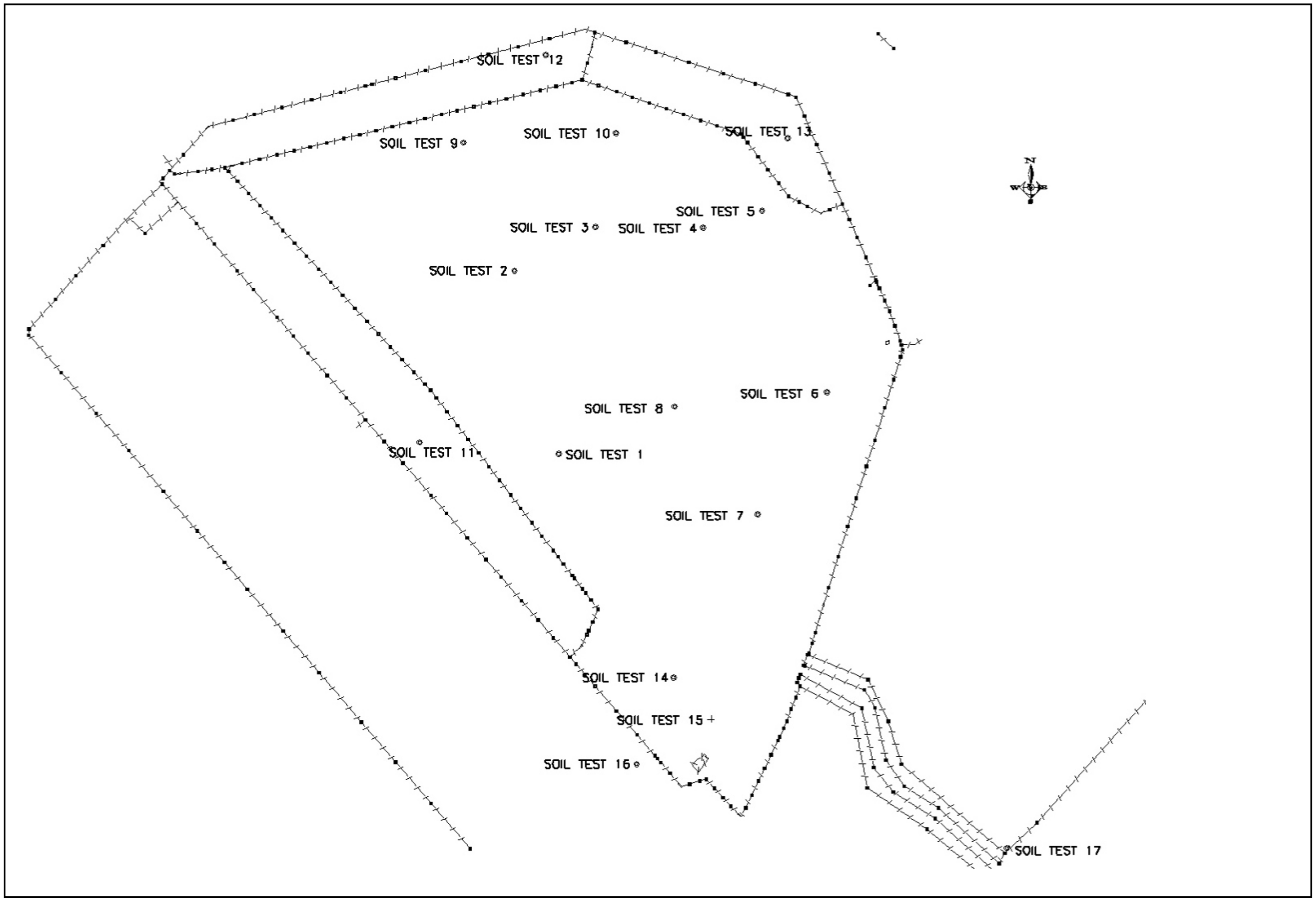


Figure 7.1: Soil sampling sites.

Figure 7.2: Soil core cutter.

7.2.3.2 Subsoil Horizons

After the cores were taken a small pit was dug through the landfill cap (Figure 7.3) until the subsurface layer was reached, approximately 40-50 cm below the soil surface. When the required depth was reached about 1kg of soil was collected. If the subsurface horizon was not found, and in the areas outside the landfill, samples were taken from a depth between 40-50 cm.

Several small aggregates were kept aside from each site to determine the dispersion index and the rest of the sample was sent to R J Hill Laboratories Limited, Hamilton for analysis.

Figure 7.3: Subsurface horizon sample collection in the landfill cap.

7.2.4 Soil Analysis

7.2.4.1 Soil Cation, Anion and Metal Analysis

At R J Hills the Basic Soil Profile, Heavy Metal Screen and Saturated Paste groups of tests were performed on the soil samples as this was the most economic method of obtaining the individual cation and metal concentrations required (Table 7.1).

Table 7.1: Soil Analysis Methods.

Analyte	Method
Basic Soil Profile tests	
Soil Preparation	Air dried at 35-40° (residual moisture typically 4%) overnight and crushed to pass through a 2mm screen.
Potassium Calcium Magnesium Sodium	1 M Neutral ammonium acetate extraction followed by ICP-OES.
Aluminium	1 M Neutral potassium chloride extraction followed by ICP-OES.
Phosphorous	Olsen extractable followed by Molydenum Blue colorimetry.
pH	1:2 (v/v) soil:water slurry followed by potentiometric determination of pH.
Volume Weight	The weight/volume ratio of dried, ground soil.
Soluble Salts (field)	1:5 soil:water extraction followed by ICP-OES.
Cation Exchange Capacity	Summation of extractable cations (K, Ca, Mg, Na) and extractable acidity.
Base Saturation	Calculated from extractable cations and cation exchangeable capacity.
Heavy Metal Screen	
Soil Preparation	Air dried at 35-40° overnight and sieved
Arsenic, Cadmium, Chromium, Copper, Nickel Lead and Zinc	<2mm fraction, Nitric /hydrochloric acid digestion, ICP-MS, screen level.
Saturated Paste tests	
EC	Saturated Paste extraction followed by potentiometric conductivity determination (25°).
Total Soluble Salts	Saturated Paste extraction followed by potentiometric conductivity determination (25°).
Total: Potassium Calcium Magnesium Sodium	Saturated Paste extraction followed by ICP-OES.
C.E.C	Summation of soluble salts from Basic Soil test minus Summation of soluble salts from Saturated paste test.

7.2.4.2 Calculation of exchangeable sodium percentage

The exchangeable sodium percentage was calculated as:

$$ESP = \frac{Exchangeable\ \ Na}{CEC} \times 100$$

where the exchangeable sodium and CEC are expressed as milliequivalents per 100 grams of soil.

7.2.4.3 Dispersion Index

The dispersion index was assessed using a modified emerson dispersion test following the method described by Loveday (1974). This method was used as it provided a rapid, simple and easily repeatable way to assess a soil's tendency to disperse. Measurements were made using distilled water and the degree of dispersion for each aggregate was assessed after two hours.

7.3 Results

The results presented in this section generally summarise the results obtained for each area; the results for individual sites are recorded in Appendix 6.

The terms, "very high, high, medium and low" which are used in this chapter to describe cation concentrations are those defined by Blakemore, 1987 (Table 7.2).

Table 7.2 Ratings for Chemical Properties (after Blakemore, 1987).

	BS[1] (%)	Ca (me/100g)	Mg (me/100g)	K (me/100g)	Na (me/100g)	CEC[2] (me/100g)
Very high	80-100	>20	>7	>1.2	>2	>40
High	60-80	10-20	3-7	0.8-1.2	0.7-2	25-40
Medium	40-60	5-10	1-3	0.5-0.8	0.3-0.7	12-25
Low	20-40	2-5	0.5-1	0.3-0.5	0.1-0.3	6-12

[1] Base Saturation

[2] Cation exchange capacity

7.3.1 Results of Soil Cation Analysis in the Landfill

a. Leachate Irrigated Areas

The leachate irrigated areas were located on top on the landfill cap (Figure 7.4). The samples taken from the surface horizon (Table 7.3) had a vey high calcium content. The base saturation (BS) for each site was 100% which is also very high. The mean sodium and potassium concentrations were high. The pH was slightly alkaline and the mean exchangeable sodium percentage (ESP) was 1.5. There was little difference between the results of the October and July samples results. PH was the only parameter where there was a significant difference ($P<0.1$) between the October and July sample results.

Figure 7.4: Landfill irrigated areas showing soil sampling site.

Table 7.3: Soil cation analysis results for the irrigated area – surface horizon.

Number of Samples		pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP[1] (%)
October 2008 Samples							
10	Mean	7.1	1.0	47.8	2.8	0.8	1.4
	Std	0.2	0.21	3.30	0.69	0.39	0.69
July 2009 Samples							
10	Mean	7.8	1.1	45.7	2.8	0.7	1.3
	Std	1.4	0.46	3.07	0.58	0.24	0.46
Both sampling rounds combined							
20	Mean	7.4	1.1	46.7	2.8	0.7	1.5
	Std	0.4	0.35	3.29	0.62	0.32	0.57

[1] Exchangeable sodium percentage

The samples taken from the subsurface horizon (Table 7.4) had a very high calcium content and a BS of 100%. Potassium concentrations were very high and sodium concentrations high. The pH was slightly alkaline. The ESP was higher than that of the surface horizon with a mean of 2.9. PH was the only parameter where there was a significant difference (*P*<0.1) between the October and July sample results.

Table 7.4: Soil cation analysis results for the irrigated area – subsurface horizon.

Number of Samples		pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP[1] (%)
October 2008 Samples							
9	Mean	7.8	0.7	40.4	2.4	1.1	2.6
	STD	0.3	0.1	5.8	0.6	0.2	0.4
July 2009 Samples							
10	Mean	7.7	0.7	38.5	2.5	1.4	3.2
	STD	0.4	0.2	5.9	0.5	0.9	1.6
Both sampling rounds combined							
19	Mean	7.7	0.7	39.4	2.5	1.3	2.9
	STD	0.3	0.1	5.8	0.5	0.7	1.2
Site 7							
1		7.7	0.2	12.6	0.9	0.08	0.6

[1] Exchangeable sodium percentage

During the October 2008 round of sampling no subsurface layer was found at site seven and the sand horizon extended all the way to the buried refuse. When the second round of sampling was undertaken in July 2009 at site seven about 1 metre from the October 08 sampling site the subsurface layer was intercepted. The sandy October 08 sample from site seven was collected at a depth of approximately 50 cm and is included for comparison. The cation concentrations and ESP recorded in the sand layer were lower than those in the subsurface horizon at a similar depth.

b. Non-Leachate Irrigated Areas

Three sites were sampled in the non-irrigated batter slopes of the landfill cap (Figure 7.5). The cation analysis results (Tables 7.5 and 7.6) showed the calcium contents in both horizons were very high while the other cation concentrations were all within the medium range. The BS for each site was 100% which is also very high. The pH was slightly alkaline in the surface horizon, but increased to moderately alkaline in the subsurface horizon. In both surface and subsurface horizons the pH was the only parameter where there was a significant difference ($P<0.1$) between the October and July sample results.

Figure 7.5: Landfill batter slope showing soil sampling site.

Table 7.5: Soil cation analysis results for the non-irrigated area – surface horizon.

Number of Samples		pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP[1] (%)
October 2008 Samples							
3	Mean	7.1	1.0	33.0	1.8	0.6	1.4
	Std	0.1	0.2	13.4	0.7	0.4	0.8
July 2009 Samples							
3	Mean	7.5	0.9	35.9	1.8	0.3	0.7
	Std	0.2	0.3	6.9	0.3	0.1	0.2
Both sampling rounds combined							
6	Mean	7.3	0.9	34.5	1.8	0.4	1.1
	Std	0.3	0.2	9.7	0.5	0.3	0.6

[1] Exchangeable sodium percentage

Table 7.6: Soil cation analysis results for the non-irrigated area – subsurface horizon.

Number of Samples		pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP[1] (%)
October 2008 Samples							
3	Mean	7.6	0.5	32.8	2.8	0.5	1.5
	Std	0.2	0.0	11.1	1.1	0.1	0.4
July 2009 Samples							
3	Mean	7.9	0.4	30.0	1.9	0.4	1.2
	Std	0.1	0.1	4.6	0.3	0.2	0.5
Both sampling rounds combined							
6	Mean	7.8	0.5	31.4	2.5	0.4	1.3
	Std	0.2	0.1	7.8	0.9	0.1	0.4

[1] Exchangeable sodium percentage

7.3.2 Results of Soil Cation Analysis for areas outside the Landfill

Two sites outside the landfill were sampled to give a comparison between cation concentrations in the landfill and the surrounding landscape. Samples were taken from the Gavin's Block (Figure 7.6), which is a low lying area on the base of the former Awapuni Lagoon approximately 20m from the edge of the landfill. The second site sampled was in the sand dune area located immediately to the south of the landfill (Figure 7.7). The two sites were chosen as they represent the major landforms surrounding the landfill.

Figure 7.6: Gavin s Block viewed from the landfill.

Figure 7.7: Sand Dunes viewed from the landfill.

The samples taken from the Gavin’s Block (Table 7.7) had very high potassium, calcium, magnesium and sodium concentrations. When the October 2008 samples were collected, salt crystals were observed on the soil surface and there were a number of patches where no vegetation was growing (Figures 7.8 and 7.9) which is consistent with the very high sodium concentrations. The ESP was also very high (mean ≈ 27) in both the surface and subsoil horizons.

Table 7.7: Soil cation analysis results for the Gavin's Block.

Number of Samples		pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP[1] (%)
Surface Samples							
3	Mean	7.6	1.9	36.1	13.4	19.9	27.5
	Std	0.4	0.1	1.8	1.6	6.1	6.9
Subsurface Samples							
3	Mean	8.3	1.3	27.7	6.5	13.2	27.2
	Std	0.1	0.5	2.4	1.3	2.4	2.3

[1] Exchangeable sodium percentage

Figure 7.8: The Gavin's Block showing patchy crop growth.

Figure 7.9: Poor crop growth in the Gavin's Block.

The samples taken from the sand dunes (Table 7.8) generally had high calcium, and medium potassium, magnesium and sodium concentrations. The mean ESP changed little with depth.

Table 7.8: Soil cation analysis results for the Sand Dunes.

Number of Samples		pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP[1] (%)
Surface Samples							
2	Mean	7.0	0.6	14.6	1.8	0.4	2.1
	Std	0.5	0.1	1.2	0.4	0.4	1.9
Subsurface Samples							
2	Mean	7.8	0.6	17.4	1.3	0.6	2.7
	Std	0.2	0.3	8.4	0.6	0.6	1.7

[1] Exchangeable sodium percentage

7.3.3 Additional Results

The 'Basic Soil Profile' tests performed by R J Hills calculated the cation exchange capacity (C.E.C) through the summation of extractable cations (K, Ca, Mg, Na). As some of the sites had a high cation concentration, which were unlikely to all be in a freely exchangeable form, the C.E.C was recalculated using the 'Saturated Paste' suite of tests. Additional information was also obtained through the Saturated Paste tests (Table 7.9). Due to financial restraints and as the 'Basic Soil Profile' tests results were consistent between sites; the Saturated Paste test was not performed on all of the samples. The Saturated Paste tests were performed on soil samples from both the surface and subsurface horizons and taken from sites three, seven and ten in the irrigated area of the landfill, site eleven in the non-irrigated areas, the Gavin's Block and the Sand Dunes.

In the landfill cap the CEC was very high in the surface horizon and high in the subsurface horizon for both the irrigated and non-irrigated areas of the landfill. The total soluble salts/EC was also high in the surface and subsurface horizons of both the irrigated and non-irrigated areas; although the EC was lower in the subsurface horizon in the non-irrigated areas of the landfill cap.

Table 7.9: Soil C.E.C, EC and total soluble salts.

	Total Soluble Salts ($g\ m^{-3}$)	EC ($\mu S\ cm^{-1}$)	CEC (me/100g)
Landfill Cap - Irrigated Areas			
Surface Horizon	1317	2000	45
Subsurface Horizon	1250	2000	36
Landfill cap - Non-irrigated Areas			
Surface Horizon	1380	2100	45
Subsurface Horizon	647	1000	26
Gavin's Block			
Surface Horizon	5730	8700	44
Subsurface Horizon	6620	10 000	31
Sand Dunes			
Surface Horizon	752	1100	14
Subsurface Horizon	191	300	14

7.3.4 Comparison of Cation Concentrations Between Sites

As pH was the only parameter where there was a significant difference between the October and July results, therefore the combined mean of both sampling rounds were used for comparison.

a. Irrigated areas of the landfill cap.

When the data from the surface horizon and subsurface horizon were compared, the surface horizon (Figure 7.10) had significantly higher ($P<0.05$) potassium, calcium, magnesium concentrations compared to the subsurface horizon. The pH, sodium and ESP were all significantly lower ($P<0.01$) in the surface horizon compared to the subsurface horizon.

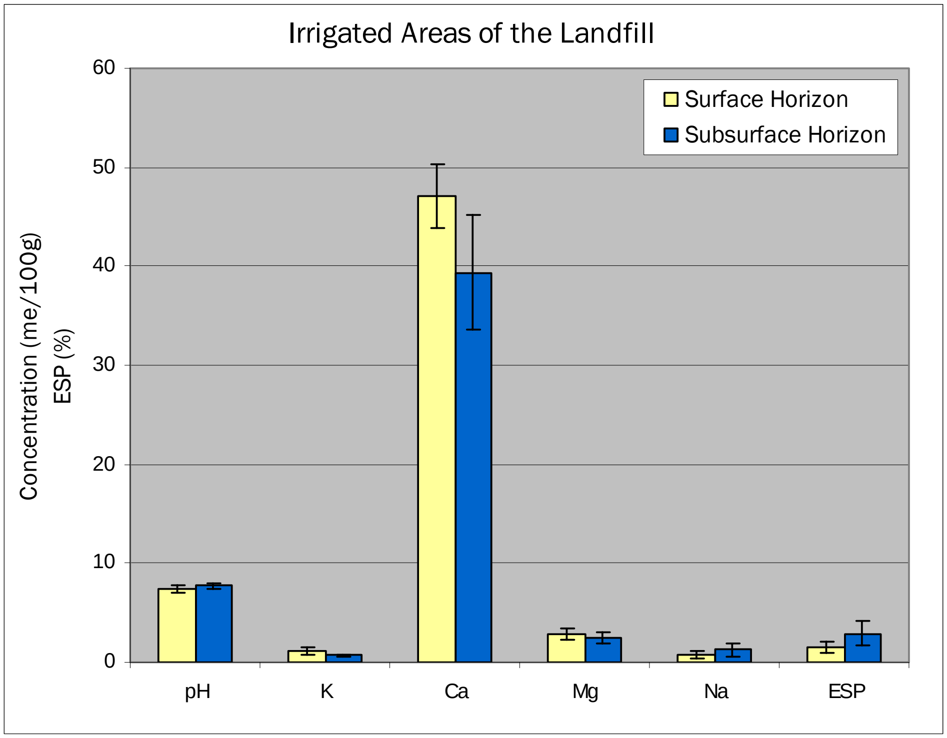


Figure 7.10: Comparison between horizons of the irrigated areas of the landfill (error bars equal 1 standard deviation of the mean).

b. Non-irrigated areas of the landfill cap.

In the non-irrigated areas (Figure 7.11) of the landfill there was generally no significant difference (*P*>0.01) between the surface horizon and the subsurface horizon. The one exception was potassium where the concentrations were significantly higher (*P*<0.001) in the surface horizon compared to the subsurface horizon.

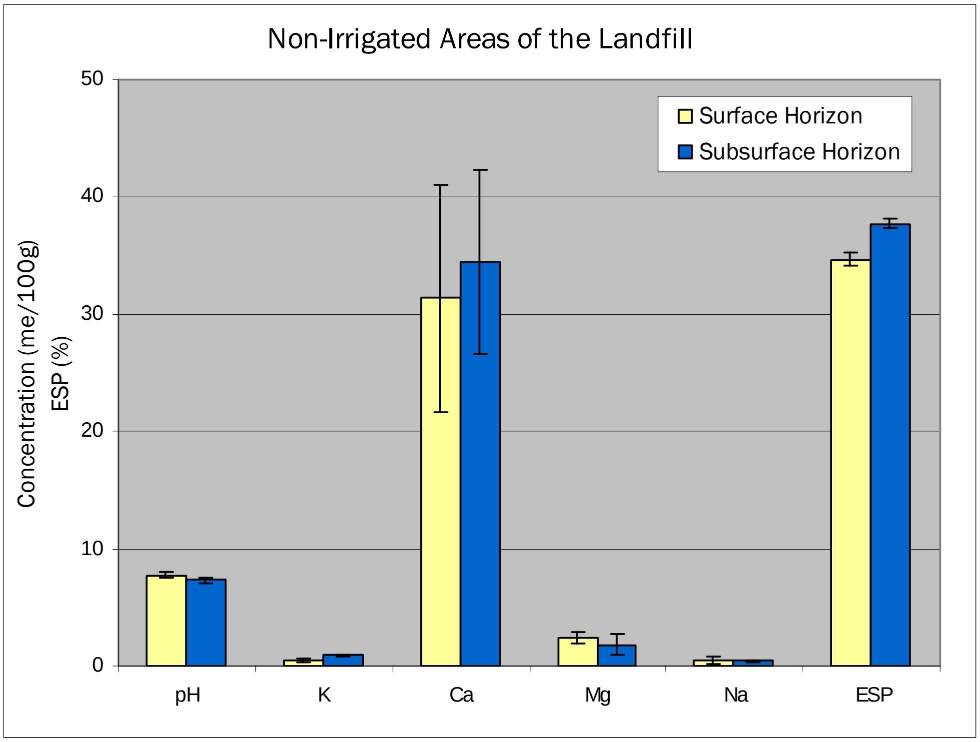


Figure 7.11: Comparison between horizons of the non-irrigated areas of the landfill (error bars equal 1 standard deviation of the mean).

c. Irrigated and Non-irrigated Areas: Surface Horizon

When the surface horizons of the irrigated and non-irrigated areas were compared (Figure 7.12), the calcium, magnesium and sodium were all significantly higher ($P<0.05$) in the irrigated areas of the landfill. There was no significant difference ($P>0.1$) between the potassium and ESP in the irrigated and non-irrigated areas of the landfill cap.

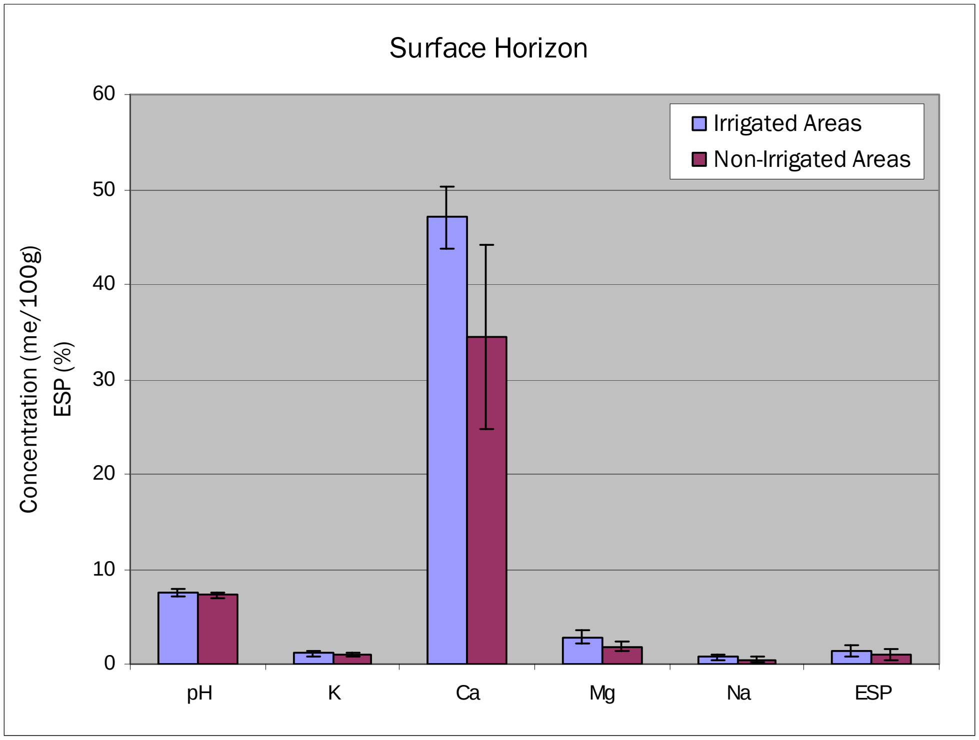


Figure 7.12: Comparison between the surface horizons of the irrigated and non-irrigated areas of the landfill (error bars equal 1 standard deviation of the mean).

d. Irrigated and Non-irrigated Areas: Subsurface Horizon

When the subsurface horizons of the irrigated and non-irrigated areas were compared (Figure 7.13), all the parameters were higher in the irrigated areas of the landfill. Calcium, potassium, sodium and ESP (P<0.01) were all significantly higher in the irrigated areas of the landfill cap compared to the non-irrigated.

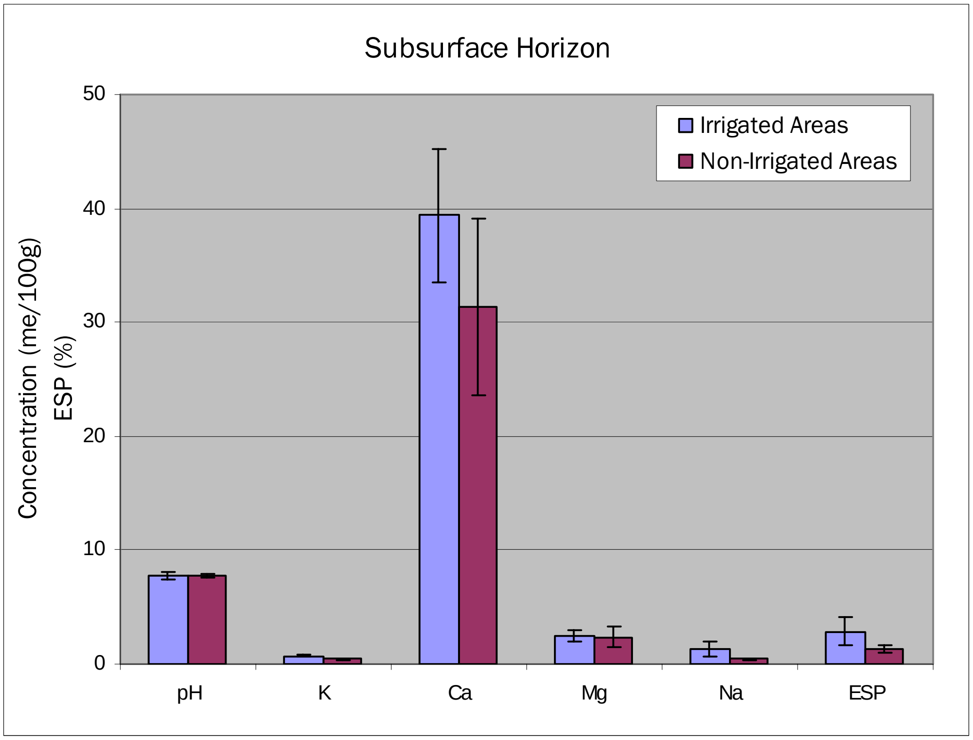


Figure 7.13: Comparison between the subsurface horizons of the irrigated and non-irrigated areas of the landfill (error bars equal 1 standard deviation of the mean).

e. Non-landfill Areas - Surface Horizon

The samples taken from the Gavin's Block generally had the highest cation concentrations out of all the surface horizons sampled (Figure 7.14). Potassium, magnesium, sodium, and ESP were all significantly higher (P<0.001) than in the other sites. The calcium concentration was similar to the non-irrigated areas of the landfill, with no significant difference (P=0.92) between the calcium content in the Gavin's Block and the non-irrigated areas of the landfill. The calcium concentrations were however significantly higher (P<0.001) in the irrigated areas of the landfill compared to the Gavin's Block.

The samples taken from the Sand Dunes generally had the lowest cation concentrations of all the sites sampled. Calcium concentrations were significantly lower (P<0.05) than in the other sites. Sodium concentrations were lower than in the samples taken from within the landfill but not significantly (P>0.01).

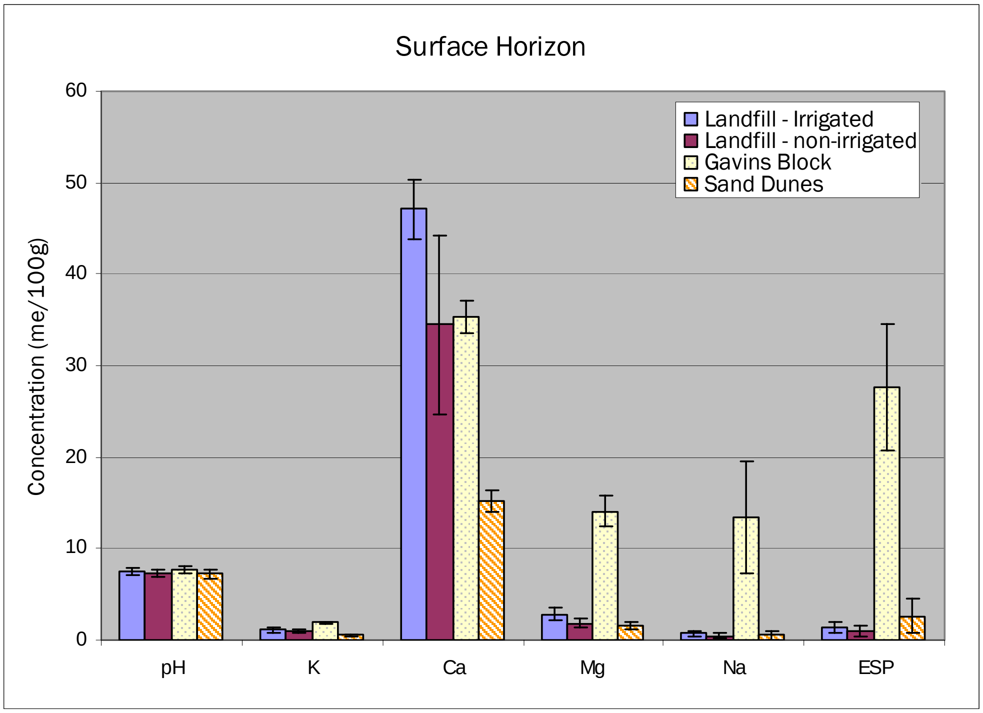


Figure 7.14: Comparison between the surface horizons for all samples (error bars equal 1 standard deviation of the mean).

f. Non Landfill Areas - Subsurface Horizon

The samples taken from the subsurface horizons followed a similar trend to those taken from the surface horizon (Figure 7.15) with the highest cation concentrations generally found in the Gavin's Block samples. Potassium, magnesium, sodium and ESP were all significantly higher ($P<0.001$) in the Gavin's Block compared to the other sites. The calcium concentration was similar to the non-irrigated areas of the landfill with no significant difference ($P=0.64$) between the two sites. The calcium in the irrigated areas of the landfill cap was significantly higher ($P<0.05$) than in the Gavin's Block.

The samples taken from the Sand Dunes generally had the lowest cation concentrations of all the subsoil horizons sampled. In the sand dune samples the sodium concentrations were significantly lower than in the irrigated landfill sites ($P=0.05$) and the Gavin's Block ($P=0.03$). The calcium concentrations were significantly lower ($P<0.05$) in the sand dunes compared to the other sites.

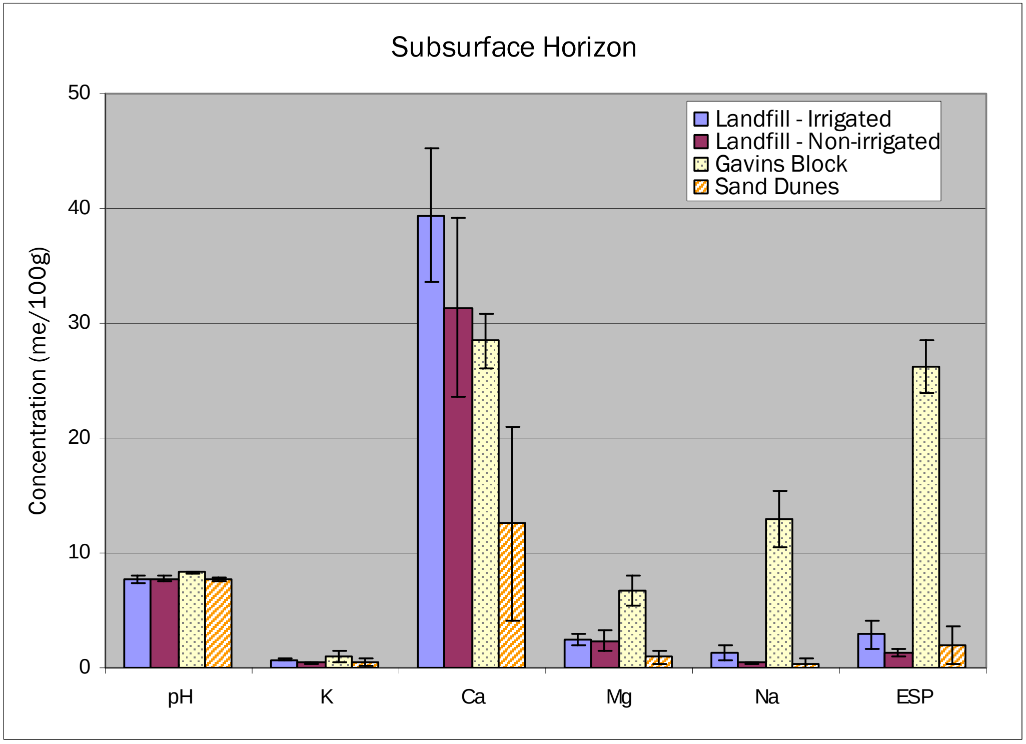


Figure 7.15: Comparison between the subsurface horizons for all samples (error bars equal 1 standard deviation of the mean).

7.3.5 Results of Soil Aggregate Stability Analysis

At each of the sampling sites soil aggregates were collected and tested for aggregate stability. The mean Dispersion Index (DI) for the surface horizon at each site within the irrigated areas of the landfill (Table 7.10) ranged from 0 (no dispersion) (Figure 7.16) to 2.0 (moderate dispersion) with a mean score of 1.0 (slight dispersion). The mean DI results for the subsurface horizon ranged from 0 (no dispersion) to 4 (complete dispersion) (Figure 7.17) with a mean score of 1.9 (slight dispersion). The aggregates which completely dispersed did so quickly, within 2 minutes of being submerged. There was no significant difference (*P*>0.1) between the DI in the surface and subsurface horizons.

Figure 7.16: Aggregate from the surface horizon of site 5 showing no dispersion after two hours.

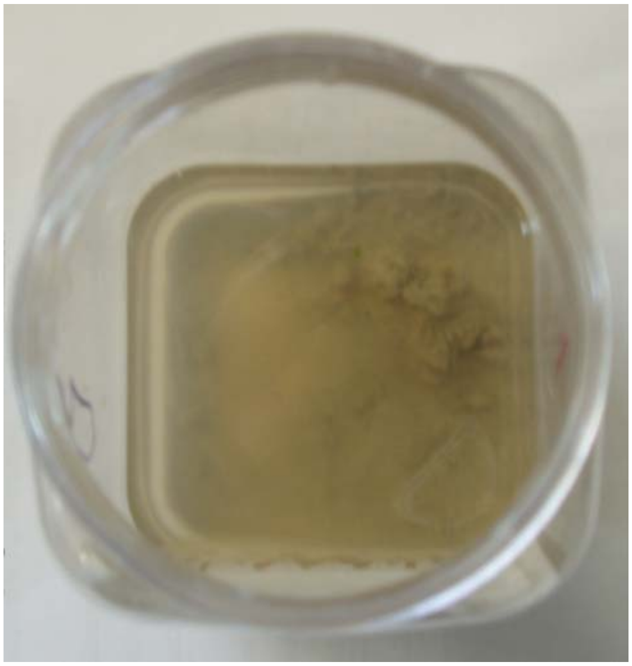


Figure 7.17: Aggregate from the subsurface horizon of site 7 showing complete dispersion after two hours.

Table 7.10: Aggregate stability results for the irrigated areas of the landfill.

	Mean Aggregate Stability in the Irrigated Area	
	Surface Horizon	Subsurface Horizon
Site 1	1	2
Site 2	2	1
Site 3	1	0
Site 4	2	3
Site 5	0	4
Site 6	1	1
Site 7	2	4
Site 8	0	0
Site 9	0	1
Site 10	1	3
Mean	**1.0**	**1.9**
STD	**0.82**	**1.52**

The surface horizon of the non-irrigated area of the landfill showed no dispersion (Table 7.11). In the subsurface horizon Site 10 showed slight dispersion while the other two sites showed no dispersion.

Table 7.11: Aggregate stability results for the non-irrigated areas.

	Mean Aggregate Stability in the Non-Irrigated Area	
	Surface Horizon	Subsurface Horizon
Site 10	0	1
Site 11	0	0
Site 12	0	0
Mean	**0**	**0.33**
STD	**0**	**0.6**

The greatest instability was observed in the aggregates obtained from the Gavin's Block (Table 7.12) with both the soil surface and subsurface horizons showing total dispersion. The dispersion did not occur as quickly as in the irrigated landfill but still occurred within two hours of the aggregates being submerged.

There were very few aggregates near the soil surface at the sand dune site and no aggregates at all in the subsoil samples which consisted of loose fine grained beach sands. The DI for the soil surface samples had a mean of value of 1 (slight dispersion).

Table 7.12: Aggregate stability results for the non-landfill areas.

	Mean Aggregate Stability	
	Soil Surface	Subsurface
Gavin's Block	4	4
Sand Dune	1	no aggregates

7.3.6 Results of Soil Heavy Metal Analysis

The samples taken from the soil surface and subsurface horizon of the irrigated area of the landfill (Table 7.13) generally had similar metal concentrations.

Table 7.13: Soil metal concentrations for the irrigated area – soil surface.

	As (mg kg)	Cd (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Pb (mg/kg)	Ni (mg/kg)	Zn (mg/kg)
Surface Horizon							
Mean	4.8	0.14	17.6	13.5	10.1	22.1	65.4
STD	0.80	0.07	3.66	3.78	1.78	5.07	12.55
Subsurface Horizon							
Mean	4.86	0.15	18	14	9.5	23	66
STD	0.39	0.03	2.42	1.35	0.87	4.06	3.55

The non-irrigated areas of the landfill had similar metal concentrations (Table 7.14) to the irrigated areas of the landfill. There was also very little difference between the mean concentrations with changes in depth.

Table 7.14: Soil metal concentrations for the non-irrigated areas of the landfill.

	As (mg kg)	Cd (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Pb (mg/kg)	Ni (mg/kg)	Zn (mg/kg)
Surface Horizon							
Mean	4.9	0.2	22.0	16.3	8.5	24.7	55.3
STD	0.42	0.03	4.58	1.53	0.64	5.51	4.51
Subsurface Horizon							
Mean	5.2	0.1	23.0	15.7	16.0	26.0	86.7
STD	1.11	0.09	4.36	2.08	10.58	6.08	46.58

Only one sample was taken at each depth from the adjacent Gavin's Block and Sand dunes (Table 7.15). The metal concentrations in the samples from the

Gavin's Block were generally slightly higher than the mean values found inside the landfill. The lowest metal concentrations were generally found in the samples collected in the sand dunes.

Table 7.15: Soil metal concentrations for the areas outside of the landfill.

	As (mg/kg)	Cd (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Pb (mg/kg)	Ni (mg/kg)	Zn (mg/kg)
Gavin's Block							
Surface	5.8	0.15	35	21	13	36	86
Subsurface	5.1	<0.10	29	16	10	33	64
Sand Dunes							
Surface	4.4	<0.10	13	5.2	9.4	16	41
Subsurface	4.4	<0.10	11	4	6.3	16	30

7.3.7 Comparison of Metal Concentrations Between Sites

The metal concentrations in the surface horizon and the subsurface layer were similar (Figure 7.18), with no significant differences ($P>0.1$) between the two horizons. The results indicate that the metals present in the landfill cap were evenly distributed between the two layers and metals were not accumulating in any one horizon of the landfill cap.

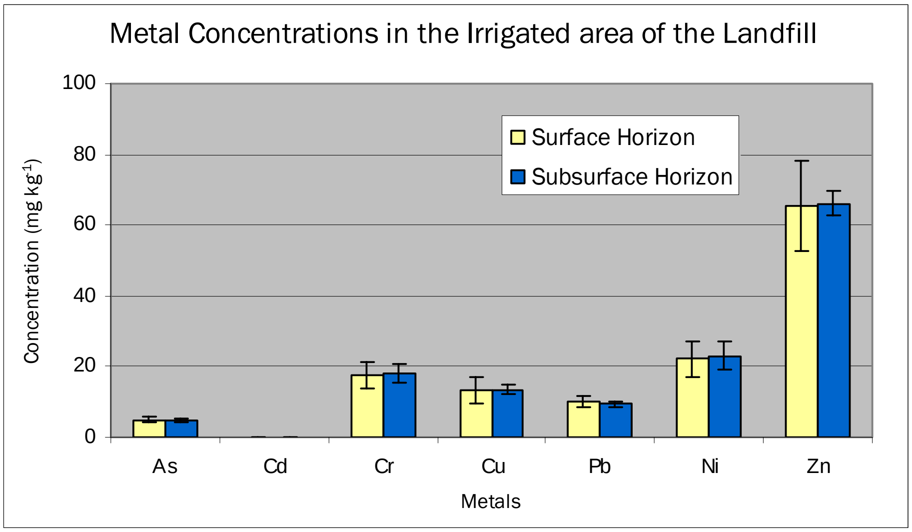


Figure 7.18: Soil metal concentrations in the irrigated area of the landfill (error bars equal 1 standard deviation of the mean).

When the surface horizons of the irrigated and non-irrigated areas were compared (Figure 7.19), there was no significant difference (*P*>0.1) between the metal concentrations.

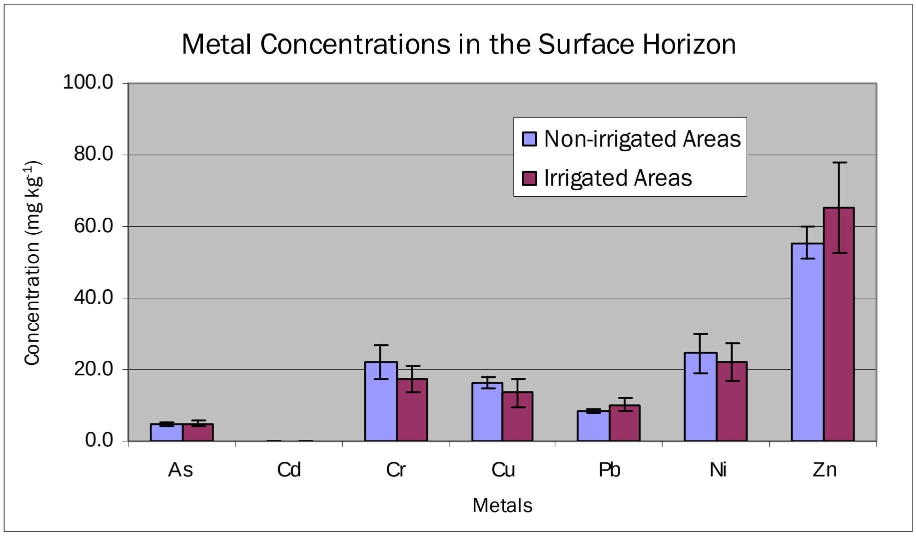


Figure 7.19: Metal concentrations in the surface horizon of the landfill cap (error bars equal 1 standard deviation of the mean).

When the metal concentrations from the subsurface horizon were compared (Figure 7.20) there were no significant differences (*P*>0.1) in the arsenic, cadmium, nickel and zinc concentrations between the irrigated and non-irrigated areas. Chromium (*P*=0.03), copper (*P*=0.05) and lead (*P*=0.05) were all significantly higher in the non-irrigated areas compared to the irrigated areas of the landfill cap.

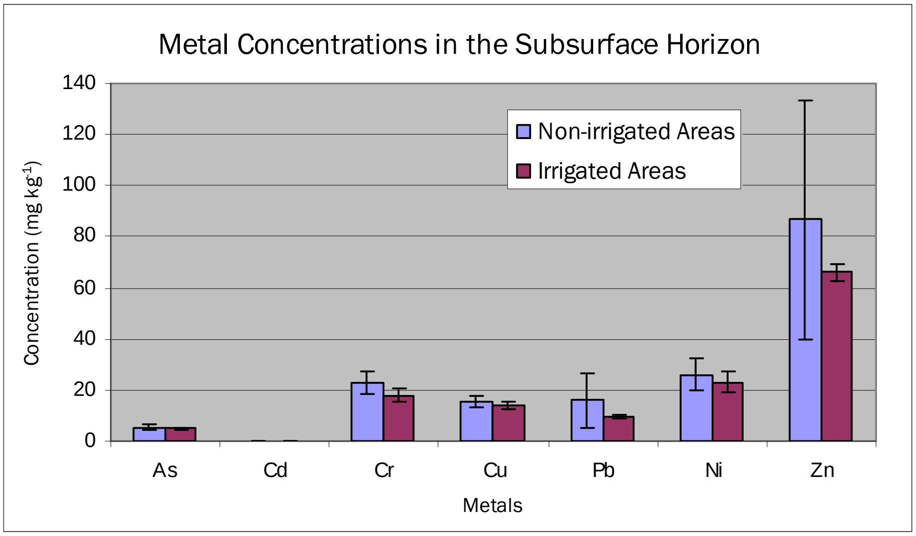


Figure 7.20: Metal concentrations in the subsurface horizon of the landfill cap (error bars equal 1 standard deviation of the mean).

7.3.8 Soil Particle Size Analysis

The soil particle size analysis showed the material in the landfill cap comprises predominantly silt and sand sized material (Figure 7.21). The clay content in the landfill cap was approximately 3 - 4 % by volume. There was no significant difference in particle size between the irrigated and non-irrigated areas of the landfill cap, so the mean particle size calculation used the results from both the irrigated and non-irrigated areas of the landfill.

The soil from the Gavin's Block contained the highest clay content; 19% in the surface horizon and 5% in the subsurface horizon (Figure 7.22). The Sand Dunes contained little fine material, comprising mainly sand sized particles.

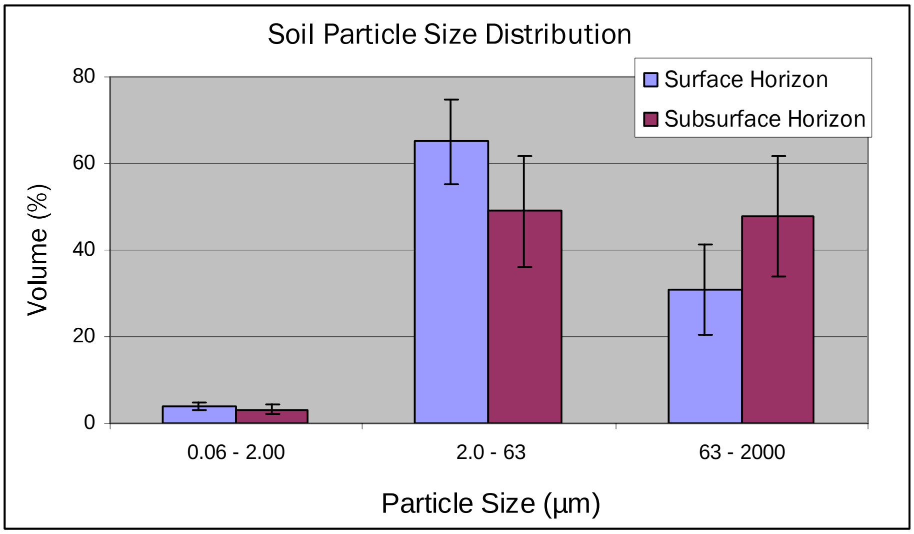


Figure 7.21: Soil particle size distribution in the landfill cap (error bars equal 1 standard deviation of the mean).

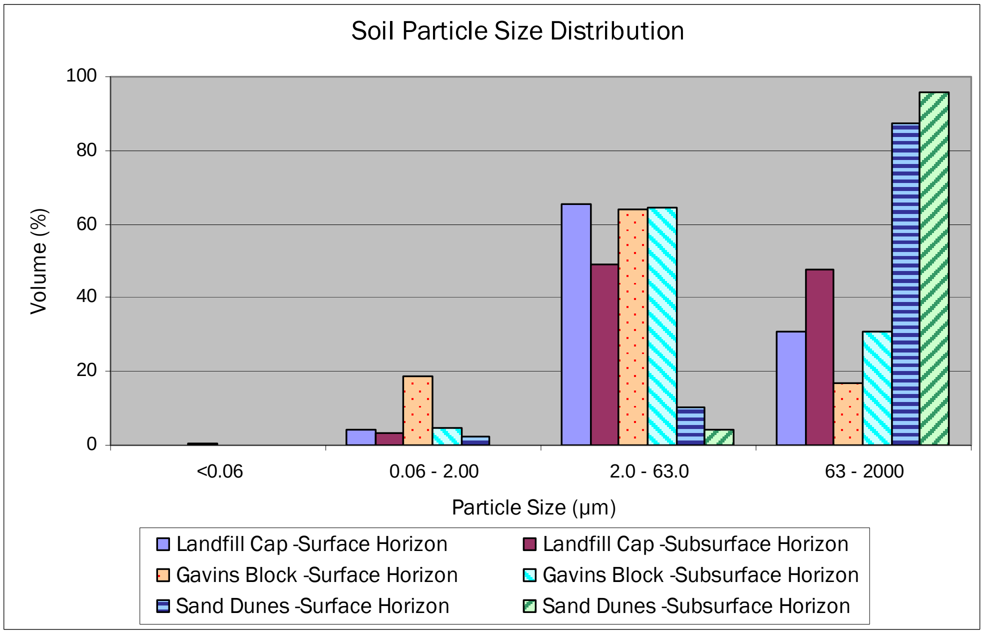


Figure 7.22: Soil particle size distribution for all soil horizons.

7.4 Discussion.

7.4.1 Soil Particle Analysis

The soil particles size analysis showed the surface horizon of the landfill cap would be classified as a silt while the subsurface horizon a silt loam. The landfill cap design called for the subsurface layer to be constructed from a clay material.

The results show that clay wasn't used in the construction of the subsurface horizon. The two horizons of the landfill cap were most likely constructed from the same material which was sourced from inside the stop banks of the Waipaoa River. The soil metal analysis supports this theory as the metal concentrations were very similar between horizons (Figure 7.18). The higher volume of sand sized particles in the subsurface layer is likely due to contamination from the sand layer of the cap. Although care was taken when collecting the soil samples some sand from above the subsurface horizon fell into the sampling pit creating some contamination of the subsurface material. Therefore both the surface and subsurface horizons were likely to be constructed from a silt dominated material (flood deposits from the Waipaoa River).

The landfill cap design specified that the subsurface horizon be constructed from a clay material with permeability between 1×10^{-8} m s^{-1} - 1×10^{-6} m s^{-1}. As the subsurface horizon was constructed from a silt dominated material (3 – 4 % clay) the design permeability may not have been achieved. If design permeability was not achieved, it could result in increased rainfall and leachate infiltration through the landfill cap leading to increased leachate production.

7.4.2 Soil Cation Analysis for the Landfill Areas

The results from the irrigated and non-irrigated areas of the landfill showed that some sodium accumulation had occurred in the irrigated areas of the landfill. When the irrigated areas were compared to the non-irrigated there was a significant increase in sodium concentrations at the soil surface (P=0.03) and in the subsurface horizon (P=0.005).

The increase in sodium concentrations in the irrigated areas was more pronounced in the subsurface horizon of the landfill cap. Comparison of the surface horizon and subsurface horizon of the landfill cap also showed that sodium accumulation was greater in the subsurface horizon. Sodium concentrations were significantly higher (P=0.003) in the subsurface horizon compared to the surface horizon in the irrigated areas while there was no significant difference (*P*=0.89) between the surface and subsurface horizons in the non-irrigated areas.

The ESP tended to follow a similar trend to that of the sodium concentrations with the ESP increasing as sodium concentrations increased. The mean ESP of the surface horizons was slightly higher (*P*=0.15) in the irrigated areas compared to the non-irrigated areas. In the subsurface horizons the ESP was significantly higher (*P*=0.005) in the irrigated areas than in the non-irrigated areas. The ESP was also significantly ($P=3.9 \times 10^{-5}$) higher in the subsurface horizon compared to the surface horizon in the irrigated areas, while there was no significant difference (*P*=0.15) between the two horizons in the non irrigated areas.

In the landfill cap the sodium and ESP also increased with depth, with a significant increase ($P<0.01$) in sodium concentration and ESP in the subsurface horizon compared to the surface horizon. In effluent irrigation schemes sodium concentrations and ESP are generally lowest at the soil surface and increase with depth, due to uptake of water by plants concentrating cations in the remaining soil water (Halliwell *et al.*, 2001), which is the most likely explanation for the increased sodium and ESP with depth in the landfill cap. The increase in sodium and ESP is unlikely to be due to a change in texture as both the surface and subsurface horizons have similar soil texture and were probably constructed from the same soil material. However, the effect of soil texture can be seen at Site 7 where no subsurface horizon was present and the ESP decreased with depth.

The sodium concentrations in both the surface and subsurface horizons were high in the irrigated areas of the landfill. The detrimental effects of high sodium concentrations on soils are exhibited through swelling and dispersion of clay minerals (Halliwell *et al.* 2001). The silty texture of the landfill cap reduces the risk of soil structural problems occurring. The landfill cap soils contain only a small percentage of clay (approximately 3% - 4% by volume) so swelling and dispersion of clay particles is not likely to cause major soil structural problems.

Swelling generally occurs at high ESP levels while dispersion can occur through out the range of ESP. Halliwell *et al.* (2001) suggested that when the ESP is above 15-25 swelling will occur. In Australia, an ESP of 6 has been suggested as the value at which swelling can occur (Sumner, 1993) and critical ESP value of 6 is often used in New Zealand in resource consent applications (Cameron *et al.*, 2003). The mean ESPs of the soils present at the Paokahu landfill generally ranged from 1–3 so it is unlikely that swelling of the clays present in the cap will occur.

At low ESP values, dispersion of the clay particles can occur but in wastewater irrigation, dispersion will not normally occur as long as the EC remains above a critical coagulation value. An estimate of the critical coagulation value was calculated in Chapter 5, from which it can be concluded that soil is not likely to be affected by clay dispersion and will be structurally stable. Currently the irrigated leachate has a high EC likely to be above the critical coagulation value. However, if leachate irrigation was stopped in the future, rainwater which has a low EC would infiltrate into the landfill cap, and dispersion may occur, but is unlikely due to the silt dominate material used in the landfill cap construction. In the landfill situation dispersion in the subsurface horizon of the landfill cap would not necessarily be a bad thing, as dispersion would reduce the permeability, leading to reduced infiltration rates and lower leachate production.

Although the sodium concentrations increased from medium levels in the non-irrigated areas to high levels in the irrigated areas the increase in ESP was not large. The likely reason for this is the very high calcium concentrations in the landfill cap. As sodium made up only a small proportion of the total base saturation (Table 7.16) the ESP did not increase greatly with increases in sodium concentrations.

The landfill cap was constructed from sediments deposited on the banks of the Waipaoa River, which will be the source of the very high calcium content. The majority of the fine sediments in the Waipaoa River come from the easily eroded

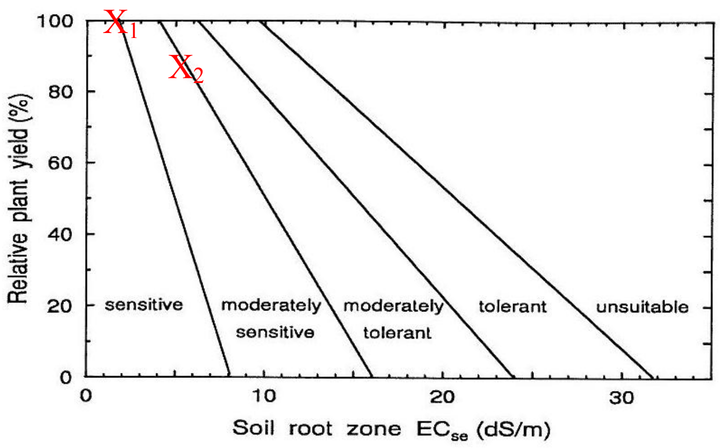
Tertiary sandstone and siltstone, much of which is calcareous, (Taylor, 1994). Claridge (1959) looked at the mineralogy of the Waipaoa river sediments and reported that a considerable amount of the sediment in the Waipaoa River comes from the erosion of calcareous bentonitic mudstones.

Table 7.16. Mean base saturation data for the irrigated and non-irrigated areas.

	Mean Base Saturation (%)				
	K	Ca	Mg	Na	Total
Irrigated Areas					
Surface Horizon	2.1	91.1	5.5	1.4	100
Subsurface Horizon	1.5	91.0	24.2	4.0	100
Non-Irrigated Areas					
Surface Horizon	2.55	92.00	4.58	0.85	100
Subsurface Horizon	1.33	90.83	6.78	1.33	100

Soil calcium concentrations were also higher in the irrigated areas of the landfill, most likely the result of the calcium present in the leachate. The high calcium concentrations in the leachate may increase the soil calcium which would also counter the effects of the high sodium concentrations, reducing the effect on the soil ESP.

The soil EC_{se} (electrical conductivity of soil saturated extract) was determined using the 'Saturated Paste Method' and can be used to determine the effects of soil soluble salts on plant growth (Figure 7.23). In the irrigated and non-irrigated areas of the landfill cap the EC_{se} was less than 2000 $\mu S\ cm^{-1}$ which is not high enough to reduce the yields in plants, therefore the soluble salt content of the soil in the landfill cap is unlikely to affect pasture growth. However, if sodium accumulation continues as a result of leachate irrigation the soil soluble salt content may become high enough to affect plant growth in the future.



X_1 = mean soil EC_{se} of the landfill cap.
X_2 = mean soil EC_{se} of the Gavin's Block.

Figure 7.23: Relative crop yield in relation to soil salinity (EC_{se}) for plant salt tolerance grouping (after Mass and Hoffman, 1977 cited in ANZECC & ARMCANZ; 2000). Note that 1 dS m^{-1} = 1000 µS cm^{-1}.

7.4.3 Soil Cation Analysis for the Areas Outside the Landfill

The highest sodium concentrations and ESP were recorded in the samples taken from the Gavin's Block. All the cations tested for were present in very high concentrations, the source of which was likely to be a combination of the local ground water and the marine environment of the lagoon in which the soil was formed.

The samples were taken from a low lying area, which has a high water table in winter. When the July (winter) round of sampling took place the water table was approximately 40 cm below the soil surface. The samples were taken approximately 450 m from Bores 402A which had the highest mean EC (34 467 µScm^{-1}) and cation concentrations (Na = 6708 g m^{-3}) of all the shallow Paokahu groundwater bores and higher than the concentrations found in the leachate. The ESP of 27 would indicate that swelling of clays is likely to occur. The high clay (19% in the surface horizon) content in the Gavin's Block soils would also mean it is more likely to be affected by swelling and dispersion of clay particles. The soil EC_{se} of 8700 µS cm^{-1} indicates that moderately salt tolerant plants will be affected by the high soluble salts present in the soil (Figure 7.23).

The samples taken from the sand dunes generally had lower cation concentrations than the other areas. The mean ESP of approximately 2 is unlikely to cause any structural problems as the soil has a sand texture so won't be affected by swelling or dispersion of clay particles.

7.4.4 Estimated ESP

In the literature several researchers have produced equations which have been used to predict soil ESP form the irrigated SAR (Rengasamy *et al.*, 1984; Seilsepour *et al.,* 2009 and US Salinity Laboratory Staff, 1954). As texture, mineralogy, bulk density, and organic matter content can all influence soil ESP, the use of SAR to predict ESP doesn't always translate from one location to another. Three published equations were used to predict ESP with the SAR data from Paokahu Landfill (Table 7.17).

Table 7.17: Models used to estimate soil ESP at the Paokahu Landfill.

Equation	SAR	Predicted ESP
$^{1}ESP = \frac{1.475 \times SAR_e}{1 + 0.0147 \times SAR_e}$	7.3	9.7
$^{2}ESP = \frac{100(-0.0126 + 0.01475\ SAR)}{1 + (-0.0126 + 0.01475\ SAR)}$	7.3	8.7
$^{3}ESP = 1.95 + 1.03\ SAR$	7.3	9.4

[1] US Salinity Laboratory Staff (1954)

[2] Austrailian and New Zealand Guidelines for Fresh and Marine Water Quality (2000)

[3] Seilsepour *et al.* (2009)

None of the models used, estimated the ESP correctly, which is not surprising as the models were not designed or calibrated for the specific soil conditions at the landfill. The predicted ESPs from the three chosen models were relatively similar, ranging from 8.7 - 9.7 compared to the actual mean ESPs of 1.5 - 2.6. All three of the chosen models over estimated the actual measured ESP. One possible reason for this could be the high soil calcium content of the original soil material (recently eroded calcareous Tertiary sandstone and siltstone) relative to soil sodium, as seen by looking at the Base Saturation data (Table 7.16).

7.4.5 Dispersion Index

The DI results showed that the soils in the irrigated areas of the landfill had a higher mean DI (*P*>0.01). In the landfill areas the DI corresponded roughly to the ESP (Table 7.18) as the highest mean DI corresponds to the highest mean ESP through to the lowest mean DI corresponding to the lowest mean ESP. However there was not a direct relationship between DI and ESP when each site was looked at individually (Figure 7.24).

Table 7.18: Mean ESP and DI for the landfill site.

	Mean ESP	**Mean DI**
Irrigated Areas		
Surface Horizon	1.5	1.0
Subsurface Horizon	2.6	1.9
Non-Irrigated Areas		
Surface Horizon	1.4	0
Subsurface Horizon	1.5	1.0

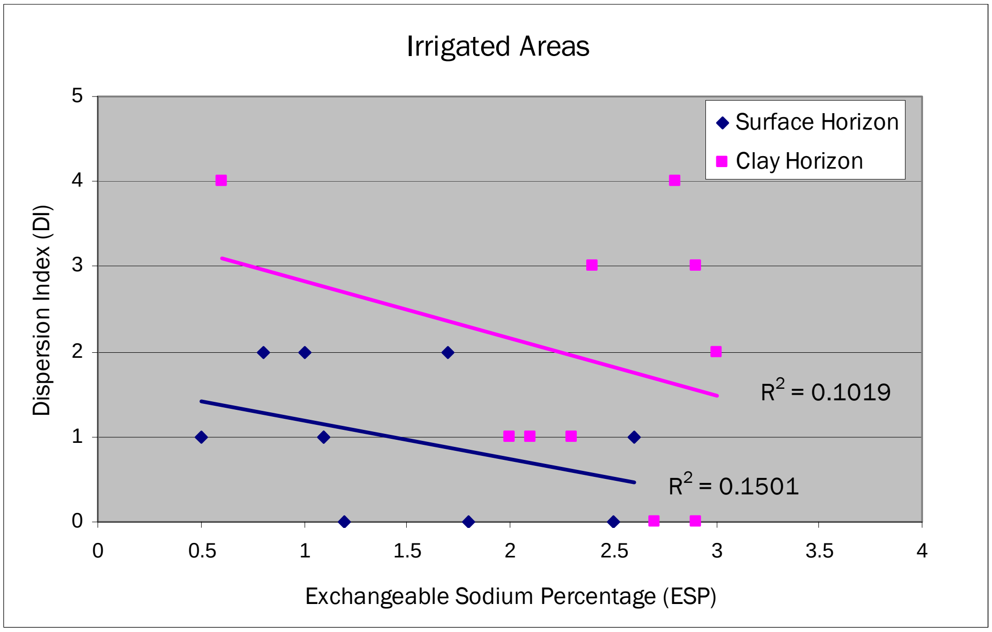


Figure 7.24: ESP versus DI for the irrigated area of the landfill.

Therefore the difference in DI between the irrigated and non-irrigated areas can not unequivocally be attributed to the effects of leachate irrigation or an increase in soil ESP.

The mean DI increased with depth in the irrigated areas of the landfill although the difference in DI between the surface and subsurface horizons was not significant (*P* >0.1). In a study of an effluent irrigation site in Australia, Balks *et al.* (1998) reported a tendency for DI at any given ESP to be greater in fine textured soils (although differences were not significant) compared to coarser textured soils. The increase in DI with depth in the irrigated areas of the landfill cap is unlikely to be associated with the change in the soil texture with depth, as the subsurface horizon and soil horizon had similar textures.

At the Gavin's Block Site all the aggregates sampled dispersed completely, which is likely to be associated with the very high ESP of 27.

7.4.6 Soil Metal Analysis

The results show that there was no build up of heavy metals in the landfill cap caused by leachate irrigation. The low leachate metal concentrations were the likely reason why there were no metal accumulations in the landfill cap.

The results obtained from the Paokahu landfill site were consistent with those reported in previous studies. For example, MacDonald *et al.* (2008) reported no significant (*P*>0.1) effect on metal concentrations after three years of leachate irrigation on a capped landfill in Michigan, USA. Godley et *al.* (2004) and Thorneby *et al.* (2006) both reported minimal accumulation of metals in the soil resulting from leachate irrigation.

The New Zealand Ministry for the Environment currently has no national guideline for soil heavy contents in New Zealand. Gisborne District Council also doesn't currently have any soil heavy metal content guidelines. The landfill metal concentrations have been compared to the New Zealand Waste Water Association (NZWWA) guidelines for the safe application of biosolids to land in New Zealand (Figure 7.25) and the Environment Waikato guidelines for residential soils. The Paokahu heavy metal concentrations were all well below the NZWWA and Environment Waikato guidelines. The guidelines are considered by NZWWA to be conservative and are based on the assumption that all the metals present are in a bio-available form, which will not be the case. Therefore, there should be no ill effects on plants or stock caused by metal accumulation under the current landfill conditions

The samples taken from the Gavin's Block generally had slightly higher metal concentrations, while the dune samples had generally slightly lower metal concentrations compared to the mean metal concentrations found within the landfill. The difference in metal concentrations will be due to the different soil types and parent materials in the three locations.

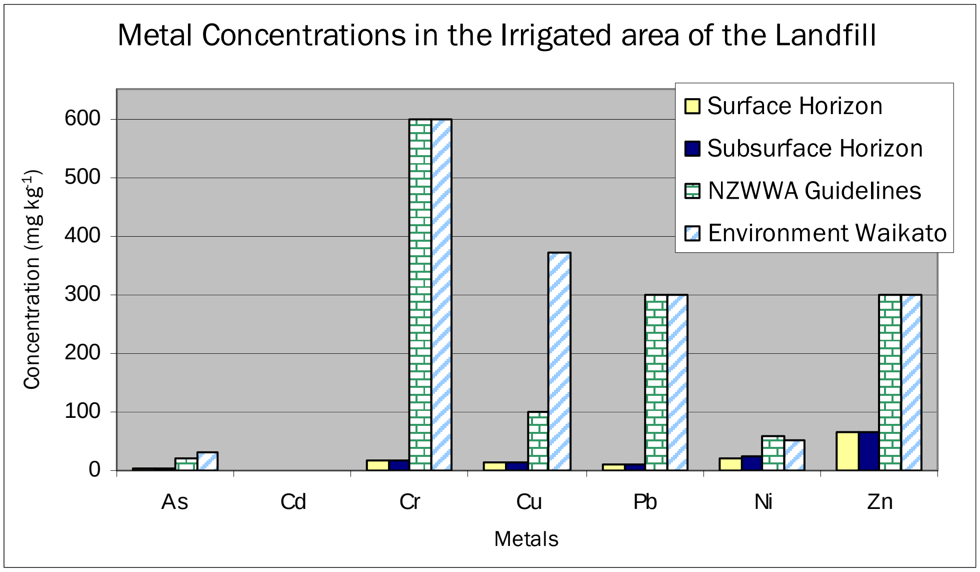


Figure 7.25: Metal concentrations in the irrigated area of the landfill cap compared to NZWWA and Environment Waikato guidelines.

7.5 Conclusions

Analysis of the soil cation data shows that leachate irrigation has caused an increase in soil cation concentrations in irrigated areas of the landfill. Sodium concentrations have significantly increased ($P<0.1$) in both the surface and subsurface horizons under leachate irrigation, but the ESP only increased significantly ($P<0.1$) in the subsurface horizon. The reason the sodium had less of an effect on the ESP of the surface horizon may be due to the uptake of water by plants concentrating cations in the remaining water. Alternatively, the increase in calcium concentrations in the surface horizon caused by the leachate irrigation could be cancelling out the effects of the increase in sodium.

The mean ESP of 1.5 for the surface horizon and 2.9 for the subsurface horizon in the irrigated areas are too low to cause the swelling of clay minerals. The soil ESPs were low enough that dispersion could occur, but an estimation of the critical coagulation value indicated that clay dispersion is unlikely under the current conditions. The silty soil texture also means that swelling and dispersion of clay particles is unlikely to occur.

Several models have been developed to predict soil ESP form the SAR of the irrigated liquid. The three models all over-estimated the soil ESP when using the Paokahu data, which may be due to the very high soil calcium content in the landfill.

The mean DI was higher in the irrigated areas of the landfill compared to the non-irrigated areas but not significantly ($P>0.1$) and when individual sites were compared there was a poor correlation between DI and ESP. Therefore the difference in DI can't unequivocally be attributed to the effects of leachate irrigation or an increase in soil ESP. The mean ESP also increased in the subsurface horizon but not significantly ($P>0.1$).

The soil heavy metal analysis showed that there was no significant difference ($P>0.1$) between the irrigated and non-irrigated areas of the landfill. The result indicates that heavy metals were not accumulating under leachate irrigation, which is consistent with what has been reported in other leachate irrigation schemes. The soil metal concentrations were all below the recommended guidelines; therefore there should be no adverse affects on plants or stock caused by metal accumulation. Therefore, under the current conditions metal accumulation is not likely to be affecting the sustainability of the leachate irrigation at the landfill.

The results show that although the leachate irrigation has increased the soil cation content and ESP, it does not appear to be having a detrimental effect on the landfill cap. Under the current conditions the irrigation of leachate onto the landfill cap appears to be a sustainable practice.

The result from the Gavin’s Block samples show the soil has been detrimentally affected by the saline environment in which it is located and by the high sodium content of the local ground water. The soil had a very high sodium content and mean ESP of 27 indicates that swelling of the clay minerals are likely to occur. The high DI also indicates poor aggregate stability. The combination of these factors indicate that the soil is not well suited for cropping, as witnessed by the poor crop yields observed during the summer of 08/09.

Chapter 8: Summary, Conclusions, and Recommendations

8.1 Introduction

The overall aim of this study was to improve our understanding of the effect that Paokahu landfill is having on the environment and to determine if the current management practices are sustainable. More specific objectives were to collate and review all the groundwater and leachate monitoring data held by Gisborne District Council and to investigate the effect of leachate irrigation on the landfill cap's soil and vegetation.

8.2 Summary

8.2.1 Landfill History

- Paokahu Landfill operated from 1977 to 2002.
- The landfill covers an area of approximately 20 hectares.
- Approximately 1 million cubic metres of waste were deposited in the landfill.
- A large percentage (up to 60% at times) of the waste in the landfill originated from vegetable processing plants.
- The landfill is unlined and fully capped.
- A leachate collection system is in place which covers approximately three quarters of the site and consists of an engineered bund with a design permeability of 10^{-9} m s^{-1} surrounding the landfill's perimeter. Inside the bund a perforated pipe collects the leachate which is then pumped to a central collection tank. Disposal of the leachate is by spray irrigation onto the landfill cap.
- Approximately 13 000 – 15 000 m^3 of leachate is irrigated annually, irrigation rates range from $3m^3$ per month in summer to $4500m^3$ per month in winter.

- The site is currently used for sheep grazing.

8.2.2 Leachate Analysis

The leachate monitoring data for Paokahu Landfill held by Gisborne District Council was examined. Leachate monitoring began in 1999; leachate samples were regularly collected from two leachate monitoring bores located in the landfill and chemical analyses were undertaken including pH, electrical conductivity (EC), and a range of cations, anions, heavy metals and semi volatile organic compounds (SVOC) . Analysis of the leachate data indicated:

- The landfill is in the methanogenic phase.
- The soluble salts content of the leachate was higher than that found in some similar New Zealand landfills (mean values, Na = 845 gm^{-3}, K= 496 gm^{-3}, Ca = 240 gm^{-3}, Mg = 127 gm^{-3}), probably due to the use of beach sands as daily cover and the landfill's coastal location.
- Leachate heavy metals were present in low concentrations (mean values, Cu = 0.07 gm^{-3}, Ni = 0.08 gm^{-3}, Pb = 0.0008 gm^{-3}, Cr = 0.06 gm^{-3}, Cd <0.0004 gm^{-3}, Mercury <0.0004 gm^{-3}, Zn = 0.08 gm^{-3} and Mn = 1.4 gm^{-3}), most likely the result of the landfill being in the methanogenic phase and the high organic matter content in the landfill binding the metals within the landfill.
- A reduction in leachate strength occurred between the leachate bores in the centre of the landfill and the leachate collection tank.
- The reduction in leachate strength was probably due to dilution caused by:
  - infiltration of rainwater through the landfill cap;
  - a raised ground water level;
  - surface water runoff from the cap infiltrating into the leachate collection tank at the outside edge of the landfill;
  - a combination of some or all of the above processes.
- An absence of leachate components in detectable concentrations which could have any major long term detrimental effects on the sustainability of the leachate irrigation, however:

- leachate manganese concentrations (1.4 gm^{-3}), were high enough to cause toxicity in plants;
- leachate boron concentrations (1.6 gm^{-3}) were high enough to be toxic to boron sensitive plants;
- the high leachate chloride (1346 gm^{-3}) concentrations could enhance cadmium uptake by plants.

8.2.3 Groundwater Analysis

There are fifteen groundwater monitoring bores, ranging in depth down to 38 m below mean sea level (MSL), located at seven locations around the Paokahu Landfill. Groundwater monitoring began in 1999, with groundwater pH, electrical conductivity (EC), and a range of cations, anions, heavy metals and semi volatile organic compounds (SVOC) determined. Analysis of the ground water monitoring data showed that:

- The aquifers in the Shallow Estuarine Silts and Recent Sands deposits contained 'fresh water' with a low EC (mean 1150 $\mu S\ cm^{-1}$) and low cation and anion concentrations (mean values, Na = 194 gm^{-3}, K= 20 gm^{-3}, Ca = 31 gm^{-3}, Mg = 18 gm^{-3}, Cl = 117 gm^{-3} and SO_4^{2-} =22 gm^{-3}).
- In the Te Hapara Sands aquifer, immediately beneath the Recent Sands deposits, the groundwater was more saline. EC, cation and anion concentrations tended to increase with depth to approximately 30-35m below sea level, indicating sea water intrusions into the aquifer (means ranged from: EC, 4547 – 49 214 $\mu S\ cm^{-1}$; Na, 542 – 9879 gm^{-3}; K, 61 - 353 gm^{-3}; Cl, 1832 – 18 600 gm^{-3}; and SO_4^{2-}, 158 - 2710 gm^{-3}).
- Below 30-35m the water became less saline and the EC, cation and anion concentrations began to decrease, indicating fresh water was entering the aquifer (mean values, EC = 21 478 $\mu S\ cm^{-1}$, Na = 4860 gm^{-3}, K= 161 gm^{-3}, Ca = 122 gm^{-3}, Mg = 367 gm^{-3}, Cl = 6536 gm^{-3} and SO_4^{2-} = 240 gm^{-3}).
- The majority of groundwater bores had higher EC, anion and cation concentrations than in the leachate bores. Ammonical nitrogen, nitrate and

nitrite concentrations were all higher (*P*<0.001) in the leachate bores compared to the groundwater bores.

- Groundwater quality was generally poor when compared to the New Zealand Drinking Water Standards, with Bore 401A the only bore to meet all the maximum permitted and recommended standards mostly due to seawater intrusions.
- There was no evidence of a leachate plume being generated from the landfill and non conclusive evidence of leachate being present in any of the monitoring bores.

8.2.4 Soil Analysis

- Leachate irrigation caused an increase in soil cation concentrations in the landfill cap compared to the non-irrigated areas.
- Sodium concentrations were significantly (*P*<0.1) higher in the subsoil horizon in the irrigated areas (mean = 1.4 me/100g) compared to the non-irrigated areas of the landfill (mean = 0.4 me/100g).
- The soil exchangeable sodium percentage (ESP) increased significantly (*P*<0.1) with depth in the irrigated areas of the landfill (surface horizon mean = 1.5%, subsurface horizon mean = 2.9%).
- There was no significant difference between the ESP in the surface horizon of the irrigated and non-irrigated areas of the landfill cap.
- In the subsurface horizons the soil ESP was significantly higher (*P*<0.01) in the irrigated areas (mean = 2.9%) compared to the non-irrigated areas (mean = 1.3%) of the landfill.
- Particle size analysis showed the landfill cap was constructed from a silt dominant material (clay < 5% by volume).
- Dispersion index tests performed on soil aggregates sampled from the landfill showed that there was no significant increase (*P*>0.1) in aggregate dispersion under the irrigated areas of the landfill compared to the non-irrigated areas.
- There was no significant difference in soil heavy metal concentrations between the irrigated and non-irrigated areas of the landfill.

- Three models were used to predict the soil ESP from the leachate sodium adsorption ratio (SAR). All the models over-estimated the soil ESP when using the Paokahu data, which is likely to be due to the very high soil calcium content in the landfill.

8.3 Discussion

8.3.1 Groundwater

The collated groundwater data showed that the landfill was not having a major effect on groundwater in the Paokahu area. There was no evidence of any leachate plume originating from the landfill and no conclusive evidence of leachate in the groundwater underneath or surrounding the landfill. Rather than the landfill, the biggest influence on groundwater quality in the Paokahu area was the infiltration of seawater into the Te Hapara Sands aquifer. Seawater infiltration gave the groundwater in the Te Hapara Sands aquifer a high EC and high cation and anion concentrations.

When the groundwater bore data was compared to the leachate bore data any leachate contamination of the groundwater in the Te Hapara Sands aquifer would be difficult to detect due to the high groundwater EC, cation and anion concentrations. In the Te Hapara Sands aquifer most of the shallow groundwater bores and all of the deep groundwater bores had higher mean sodium, calcium, magnesium, chloride and sulphate concentrations than in the leachate. Therefore as the leachate semi volatile organic compounds (SVOC) concentrations were all below the test detection limit and the leachate metal concentrations were low, the major risk to groundwater from leachate contamination would come from ammonical nitrogen and nitrate. Ammonical nitrogen and nitrate concentration were higher in the leachate than in the groundwater so leachate contaminations would increase the groundwater ammonical nitrogen and nitrate concentrations. Nitrate contamination of groundwater can pose a problem where the groundwater is used for drinking as concentrations above 10g m^{-3} may be toxic to humans, or where the contaminated groundwater water enters a surface water body the nitrate can lead to eutrophication of the waterway.

The shallow groundwater present in the Recent Sands and Shallow Estuarine Silts surrounding the landfill was not affected by salt water intrusion; therefore any leachate contamination would have a greater impact on water quality compared to the Te Hapara Sands aquifer. The monitoring data did not show any signs of leachate contamination in the groundwater in the Recent Sands and Shallow Estuarine Silts. However, the groundwater in the Shallow Estuarine Silts showed signs that the agricultural land use was impacting on the groundwater in some areas. For example Bore 401A is located down stream of the landfill and was the only bore where heavy metals were consistently detected. Bore 401A also had the highest dissolved reactive phosphorous (DRP) concentrations of those found in all the groundwater and leachate bores (mean = 2.2 g m^{-3}), indicating that the heavy metals may be coming from the use of phosphate fertilizer. The leachate bores had significantly lower ($P<0.001$) phosphate concentrations (mean = 0.15 g m^{-3}) than in Bore 401A, indicating the high DRP was not the result of a leachate contamination of the ground water.

The presence of the landfill in the Paokahu area shouldn't greatly affect any potential use of the groundwater resources. In general the groundwater quality in the Paokahu area was poor and didn't meet the New Zealand Drinking Water Standard or the Stock Drinking Water guidelines. The high sodium and other soluble salts content of the groundwater would also limit its use in any crop irrigation scheme. The analysis of the leachate data showed that boron and manganese concentrations in the leachate bores were high enough to potentially be toxic to plants when used in crop irrigation. In most of the bores in the Te Hapara Sands aquifer the groundwater had higher boron and manganese concentrations than in the leachate so if used for crop irrigation the groundwater could cause toxicity problems in plants. The high groundwater sodium and chloride concentrations could also be toxic to plants if the groundwater was used for irrigation.

8.3.2 Leachate Irrigation

Analysis of the leachate and soil data showed that leachate irrigation is sustainable under current conditions.

Before this study was undertaken, Gisborne District Council was concerned that the high leachate sodium content could cause soil structural problems affecting the long term sustainability of leachate irrigation onto the landfill cap. This study showed that although the leachate sodium concentrations were high, soil structural problems were unlikely under current conditions. The soil ESP of 1.5 in the surface horizon and 2.9 in the subsurface horizons of the irrigated areas of the landfill cap were too low to cause swelling of clay minerals. Examination of the leachate EC and SAR data showed that dispersion was not likely to occur as the EC and SAR were above the critical coagulation value need for soil dispersion to occur. The dispersion index tests performed on soil aggregates taken from the landfill cap showed only slight dispersion supporting the conclusion that leachate irrigation was not causing soil structural problems in the landfill cap. The silty texture of the soil in the surface and subsurface horizons of the landfill cap soil means swelling and dispersion of clays is unlike to affect the sustainability of the leachate irrigation.

Analysis of the leachate data indicated the EC of the irrigated leachate (mean = 4317 µS cm^{-1}**)** was high enough to affect salt sensitive plants including the white clover (*Trifolium reperis*) present on the landfill cap. However when the soil EC (< 2000 µS cm^{-1}**)** was determined it showed the soluble salts present in the landfill cap were not present in high enough concentrations to affect pasture yields.

Heavy metal accumulation from leachate irrigation was not expected to be occurring in the landfill cap as the leachate metal concentrations were low. The soil metal analysis confirmed that heavy metal accumulation was not occurring in the landfill cap. There were no significant differences between the metal concentrations in the irrigated and non-irrigated areas of the landfill and the soil metal concentrations were well below the recommended guideline (NZWWA, 2003). Therefore under current conditions metal accumulation is not likely to affect the sustainability of the leachate irrigation system. However, this may change in the future as recent research has suggested (Bozkurt *et al.,* 2000; Kjeldsen *et al.,* 2002) that in the long term, upward of 100 years, landfill conditions can change. Methane production can stop, allowing the landfill to become aerobic and acidic, which can increase the mobility of the heavy metals bound in the landfill.

High chloride concentrations in irrigation waters (above 400 g m^{-3}) can cause an increase in the uptake of cadmium by plants. The leachate had a mean chloride concentration of 12 305 g m^{-3}, so is high enough to increase the uptake of cadmium by plants. The low leachate cadmium (<0.0005 g m^{-3}) and low soil cadmium (mean = 0.15 mg kg^{-1}) concentrations mean that plant uptake of cadmium is unlikely to affect the landfill cap pasture as there is little cadmium present in the leachate and landfill cap.

Manganese concentrations above 0.75 g m^{-3} can be toxic to plants. The leachate bore analyses showed that manganese concentrations were above 0.75 g m^{-3} (mean = 1.39) therefore manganese toxicity could occur. Soil pH strongly influences manganese availability to plants. When the soil pH is above 5.0-5.5 manganese is adsorbed onto the soil organic matter, becoming less available to plants (El-Jaoual & Cox, 1998). High calcium availability has also been shown to reduce manganese toxicity in plants (El-Jaoual & Cox, 1998; Hue *et al.*, 2001). The top soil in the irrigated area of Paokahu landfill had a mean pH of 7.1 and a very high exchangeable calcium content (rating based on Blakemore *et al.,* 1987) of 42 me/100g. The calcium content of the leachate (240 g m^{-3}) was also considerably higher than that of the manganese (1.2 g m^{-3}). The combination of high soil pH, available soil calcium and high leachate calcium should reduce the risk of manganese toxicity occurring.

8.3.3 Background Sites

The landfill cap soils showed less signs of structural problems caused by saline conditions than some of the sites outside the landfill. The soil in the Gavin's Block (Makaraka Clay Loam Saline Phase) generally had higher cation concentrations than the soils in the landfill cap. Potassium, magnesium and sodium were all significantly higher ($P<0.001$) in the Gavin's Block soils compared to the landfill cap soils. The soil sampled from the Gavin's Block has a mean ESP of 27 and was the only soil to show signs of soil structural problems. Soil aggregates from the Gavin's Block had a DI of 4 and showed complete dispersion within 2 minutes of being submersed in distilled water. The high soil sodium in the Gavin's Block was the result of the coastal location (a drained tidal lagoon), the high sodium content of the groundwater and the shallow water table. The groundwater in the shallow bore (402A) located approximately 450 metres away from the soil sampling site was affected by salt water intrusions and had an SAR of 42 and an EC of 36,600 μS cm^{-1} indicating that soil structural problems are likely. The soil EC in the surface horizon (8700 μS cm^{-1}) indicated that only salt tolerant plants would be able to grow without reduced yields. During October 2008 salt crystals were observed on the soil surface and pasture growth was patchy with many bare patches also indicating the high salt content was affecting plant growth.

The second background site sampled was in the dunes, south of the landfill (Opoutama Sand). The soil cation concentrations were generally lower than those found in the landfill cap. Soil structural problems were not expected due to the sandy texture of the soil which was confirmed by the DI results. The groundwater underneath the sampling site (close to bore 403A) would be unlikely to cause soil structural problems as it is deeper so not near the soil surface. However the SAR of 0.5 and EC of 616 μS cm^{-1} could cause dispersion of clay minerals in a clay soils.

8.4 Conclusions

This study showed that Paokahu Landfill is not having an adverse effect on the groundwater in the Paokahu Area or surrounding areas as:

- There was no evidence of a leachate plume originating from the landfill.
- There was no conclusive evidence of any leachate contamination in any of the leachate monitoring bores.
- The groundwater in the Paokahu area was generally affected by salt water intrusions giving the groundwater high background cation and anion concentration.
- As the leachate contains low concentrations of heavy metals and semi volatile organic compounds and the groundwater contains high background cation and anion concentrations; the main risk to groundwater quality from leachate contamination is nitrate contamination.

This study also showed that the leachate irrigation was not adversely affecting the landfills cap and vegetation as:

- The leachate irrigation was not affecting soil aggregate stability as:
    - The soil ESP of between 1-3 in the irrigated areas of the landfill means swelling of clay particles was unlikely to occur.
    - Dispersion of soil aggregates is unlikely as the EC and SAR of the leachate are above the critical coagulation value needed for dispersion to occur.
    - Aggregate dispersion index tests showed no significant difference in aggregate stability between the irrigated and non-irrigated areas of the landfill.
    - The silt dominant soil texture (<5% clay by volume) gives the landfill cap a low susceptibility to swelling and dispersion of clays.
- Leachate heavy metal concentrations were low; as a result leachate irrigation is not causing an increase in soil heavy metal concentrations in the irrigated areas of the landfill.

- The leachate doesn't contain any elements or compounds in concentrations high enough to adversely affect the landfill cap's soil or plant life, exceptions were:
    - EC was high enough to indicate that the total soluble salts could affect pasture growth, however the soil EC showed that total soluble salts were unlikely to be affecting pasture growth.
    - Manganese - however the soil pH and calcium content mean that manganese toxicity in plants is unlikely to occur.
    - Boron - the leachate concentrations are high enough to affect boron sensitive plants, however the grass species on the landfill cap are unlikely to be boron sensitive.
- The leachate under-went a reduction in strength between the centre of the landfill and the leachate irrigation tank (probably due to infiltration of water through the base, sides or cap of the landfill), which means the leachate constituents will be present in lower concentrations when irrigated onto the landfill cap than observed in the leachate bores.

8.5 Recommendations

1. The leachate collection tank should be included in the monitoring programme. Comparisons between the leachate bore and the leachate collection tank showed the leachate underwent a reduction in strength between the bores and the collection tank. Regular monitoring of the leachate collection tank is needed to monitor actual chemical loading resulting from the leachate irrigation. Use of the leachate monitoring bore data will produce an over estimation of chemical loading rates on the landfill cap. The monitoring should be conducted at the same time as the leachate bore monitoring and test for the same parameters as used in the leachate bores but excluding the SVOC screen. The leachate collection tank loading rates can then be used to assess the risk to the soil structure and pasture health and aid in the management of the landfill.

2. The semi volatile organic screen (SVOC) should no longer be included in the groundwater monitoring. No SVOCs have been detected in the leachate since 2001 and as SVOC concentrations decrease over time it is unlikely that any SVOC will be detected in future groundwater monitoring. Removing the SVOC screen from the ground water monitoring will reduce the monitoring costs incurred by Gisborne District Council.

3. In 2008 the monitoring frequency was adjusted in the groundwater bores and as a result the only deep groundwater bore which is still monitored is Bore 402C. Bore 402C is the deepest bore in the series and the water has a different composition to the other groundwater bores, probably due to up welling from the Makauri Gravels aquifer. Bore 402C is not really representative of the general water quality at depth in the Te Hapara Sands aquifer. I recommend that rather than to continue to monitor Bore 402C it would better to monitor one of the other deep groundwater bores. My suggestion would be to monitor either bore 401B or 401C. Bores 401B and 401C are more representative of the water quality in the deep groundwater bores and are situated north of the landfill which is down stream from the landfill so would be better positioned to detect any future leachate plume.

4. Bore 404B was the only bore which could have contained leachate contamination. Therefore groundwater monitoring in Bore 404B should be maintained at the current frequency. The groundwater in bore 404B should be closely monitored for signs of leachate contamination particularly the ammonical nitrogen and nitrate concentrations. Ammonical nitrogen and nitrate are the mostly likely compounds to be detected if leachate contamination is occurring and have the greatest potential to affect groundwater quality in the Paokahu area.

5. If leachate contamination of the groundwater is detected in the future the leachate levels inside the landfill should be kept as low as possible by ensuring the leachate collection system is working correctly. Lowering the leachate levels will reduce the volume of leachate escaping the landfill by reducing the leachate pressure head within the landfill.

6. Stock grazing should be continued on the landfill cap as the results have shown that the leachate irrigation is sustainable under current conditions.

7. Extensive pumping of the groundwater in the Paokahu area shouldn't be undertaken, as it will lower the local water table causing greater seawater intrusions in the Te Hapara Sands aquifer further reducing the ground water quality in the area.

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Appendices

Appendix 1: Leachate Analysis Results

Appendix 2: SVOC List and Detection Limits

Appendix 3: Soil Nutrient Test Results

Appendix 4: Ground Water Analysis Results

Appendix 5: Back Ground Bore Results

Appendix 6: Soil Analysis Results

Appendix 1

Table A1.1A: Monitoring Data for Leachate Bore 2

Temperature (°C)	pH	Conductivity (μS cm^{-1})	BOD (g m^{-3})	COD (g m^{-3})	$N-NH_3$ (g m^{-3})	NO_3^- (g m^{-3})	NO_2^- (g m^{-3})	TKN (g m^{-3})	Na (soluble) (g m^{-3})	K (soluble) (g m^{-3})	Ca (soluble) (g m^{-3})	Mg (soluble) (g m^{-3})	Cl^- (g m^{-3})	SO_4^{2-} (g m^{-3})	Alkalinity (g m^{-3})
20.3	7.8	9780	66	-	700	-	-	-	-	-	-	-	-	-	-
17.3	8.4	6530	47	-	330	-	-	-	-	-	-	-	-	-	-
21.2	7.3	13370	30	1,750	300	30.30	2.63	296	1390	374	553	238	3050	283	4750
18	6.9	3710	41	-	-	-	-	-	-	-	-	-	-	-	-
13.2	7.1	5550	36	-	130	-	-	-	-	-	-	-	-	-	-
14.9	6.7	5100	89	-	75	-	-	-	-	-	-	-	-	-	-
20.2	7.0	7380	58	-	220	-	-	-	-	-	-	-	-	-	-
18.5	6.8	4400	52	320	135	<0.02	0.004	135	375	182	216	64	520	5	1200
14.5	7.2	7720	58	-	237	-	-	-	-	-	-	-	-	-	-
15.9	7.0	8080	57	-	358	-	-	-	-	-	-	-	-	-	-
19.2	7.4	11100	77	160	640	<0.05	-	703	841	637	262	127	1100	16	4600
14.5	7.2	8820	51	-	387	-	-	-	-	-	-	-	-	-	-
18.7	6.9	11500	18	-	426	-	-	-	-	-	-	-	-	-	-
18.9	7.4	7430	40	519	328	0.05	0.557	338	903	359	247	130	882	25	2900
14.1	6.9	8080	47	220	415	-	-	-	-	-	-	-	-	-	-
18.8	7.6	2750	36	100	83	5.24	3.500	99	229	141	104	35	330	123	810
16.8	7.4	8440	49	260	415	-	-	-	-	-	-	-	-	-	-
20.2	6.8	5090	50	200	115	0.01	0.003	112	456	198	363	94	590	728	1400
13.1	7.3	4760	13	720	167	-	-	-	-	-	-	-	-	-	-
19.4	7.3	12400	25	70	770	0.01	0.011	668	1010	693	160	152	1520	1	5000

Table A1.1B: Monitoring Data for Leachate Bore 2

DATE	Fe (soluble) (g m^{-3})	B (soluble) (g m^{-3})	DRP (g m^{-3})	Mn (soluble) (g m^{-3})	Cu (soluble) (g m^{-3})	Zn (soluble) (g m^{-3})	Ni (soluble) (g m^{-3})	Pb (soluble) (g m^{-3})	Cr (soluble) (g m^{-3})	Cd (soluble) (g m^{-3})	Hg (soluble) (g m^{-3})	As (soluble) (g m^{-3})	Se (soluble) (g m^{-3})	SVOC (g m^{-3})
19/10/99	-	-	-	-	-	-	-	-	-	-	-	-	-	-
14/12/99	-	-	-	-	-	-	-	-	-	-	-	-	-	-
24/02/00	2.7	1.41	0.323	2.45	0.02	0.142	0.086	0.0031	0.02	0.0004	<0.00008	0.016	<0.005	#
4/04/00	-	-	-	-	-	-	-	-	-	-	-	-	-	-
13/07/00	-	-	-	-	-	-	-	-	-	-	-	-	-	-
3/10/00	-	-	-	-	-	-	-	-	-	-	-	-	-	-
23/01/01	-	-	-	-	-	-	-	-	-	-	-	-	-	-
5/04/01	13.5	0.65	0.021	2.42	0.002	0.027	0.037	0.0002	0.005	<0.0001	<0.00008	0.005	<0.002	##
5/07/01	-	-	-	-	-	-	-	-	-	-	-	-	-	-
16/10/01	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8/04/02	2.1	1.76	0.504	1.21	<0.005	0.04	0.111	0.001	0.04	<0.0005	<0.00008	0.02	<0.01	*
14/10/02	-	-	-	-	-	-	-	-	-	-	-	-	-	-
7/10/03	-	-	-	-	-	-	-	-	-	-	-	-	-	-
1/04/04	9.0	1.28	0.080	1.66	0.007	0.026	0.056	0.000	0.015	<0.0005	<0.00008	0.009	<0.005	*
4/10/04	-	-	-	-	-	-	-	-	-	-	-	-	-	-
4/04/05	0.6	0.50	0.035	0.41	0.005	0.046	0.018	0.0003	0.001	<0.0001	<0.00008	0.005	<0.002	*
19/10/05	-	-	-	-	-	-	-	-	-	-	-	-	-	-
5/04/06	0.4	0.91	0.055	-	0.004	0.025	-	-	-	-	-	0.004	-	-
19/10/06	-	-	-	-	-	-	-	-	-	-	-	-	-	-
4/04/07	2.1	1.59	0.043	0.459	<0.005	0.01	0.051	<0.001	0.012	<0.0005	<0.00008	<0.01	<0.01	*

[#] = Naphthalene = 0.00011g m^{-3}, Di-(2-ethylhexyl)phthalate = 0.0022 g m^{-3}, all other parameters below detection limit.

[##] = 2-Methylnaphthatene = 0.00176 m$^{-3,}$ 2,4-Dimethylphenol = 0.001g m^{-3}, all other parameters below detection limit.

* = all parameters below detection limit.

Table A1.2A: Monitoring Data for Leachate Bore 2

DATE	**Temperature** (°C)	**pH**	**Conductivity** (μS cm^{-1})	**BOD** (g m^{-3})	**COD** (g m^{-3})	**N-NH$_3$** (g m^{-3})	**NO$_3^-$** (g m^{-3})	**NO$_2^-$** (g m^{-3})	**TKN** (g m^{-3})	**Na (soluble)** (g m^{-3})	**K (soluble)** (g m^{-3})	**Ca (soluble)** (g m^{-3})	**Mg (soluble)** (g m^{-3})	**Cl$^-$** (g m^{-3})	**SO$_4^{2-}$** (g m^{-3})	**Alkalinity** (g m^{-3})
19/10/99	17.80	7.0	10500	-	-	450	-	-	-	-	-	-	-	-	-	-
14/12/99	19.20	7.8	9650	33	-	490	-	-	-	-	-	-	-	-	-	-
24/02/00	22.20	7.1	10740	<30	920	220	<0.02	0.009	474	905	565	323	126	1250	<1	4000
4/04/00	20.40	7.0	8700	150	-	-	-	-	-	-	-	-	-	-	-	-
13/07/00	14.50	8.0	7150	22	-	290	-	-	-	-	-	-	-	-	-	-
3/10/00	15.40	7.1	9850	92	-	390	-	-	-	-	-	-	-	-	-	-
23/01/01	22.90	7.0	8450	65	-	220	-	-	-	-	-	-	-	-	-	-
5/04/01	19.00	7.2	9200	54	720	373	9.7	0.408	356	845	560	207	121	1380	25	2300
5/07/01	12.90	7.3	9740	73	-	443	-	-	-	-	-	-	-	-	-	-
16/10/01	17.40	7.0	11200	85	-	612	-	-	-	-	-	-	-	-	-	-
8/04/02	19.00	7.3	12700	54	140	724	<0.05	-	701	1070	706	218	148	1400	5	4900
14/10/02	16.80	7.2	11300	72	-	646	-	-	-	-	-	-	-	-	-	-
7/10/03	19.00	7.1	15800	54	-	957	-	-	-	-	-	-	-	-	-	-
1/04/04	21.10	7.3	11700	59	844	633	0.017	0.349	684	959	635	178	130	1350	5	4600
4/10/04	15.10	7.1	5620	55	160	228	-	-	-	-	-	-	-	-	-	-
4/04/05	19.80	7.4	11900	58	140	565	58.5	-	647	1200	776	184	147	1300	98	3700
19/10/05	16.90	7.2	9350	32	260	457	-	-	-	-	-	-	-	-	-	-
5/04/06	20.40	7.0	8720	59	280	379	33.7	0.0003	347	773	431	320	123	1000	216	2900
19/10/06	12.90	7.1	8060	45	1,200	401	-	-	-	-	-	-	-	-	-	-
4/04/07	20.10	7.3	11400	31	160	492	79.9	1.35	513	1010	688	189	142	1550	6	4100

Table A1.2B: Monitoring Data for Leachate Bore 2

DATE	**Fe (soluble)** (g m^{-3})	**B (soluble)** (g m^{-3})	**DRP** (g m^{-3})	**Mn (soluble)** (g m^{-3})	**Cu (soluble)** (g m^{-3})	**Zn (soluble)** (g m^{-3})	**Ni (soluble)** (g m^{-3})	**Pb (soluble)** (g m^{-3})	**Cr (soluble)** (g m^{-3})	**Cd (soluble)** (g m^{-3})	**Hg (soluble)** (g m^{-3})	**As (soluble)** (g m^{-3})	**Se (soluble)** (g m^{-3})	**SVOC** (g m^{-3})
19/10/99	-	-	-	-	-	-	-	-	-	-	-	-	-	-
14/12/99	-	-	-	-	-	-	-	-	-	-	-	-	-	-
24/02/00	20.4	1.45	0.133	2.06	<0.003	0.044	0.062	0.0031	0.036	<0.0003	<0.00008	0.026	<0.005	*
4/04/00	-	-	-	-	-	-	-	-	-	-	-	-	-	-
13/07/00	-	-	-	-	-	-	-	-	-	-	-	-	-	-
3/10/00	-	-	-	-	-	-	-	-	-	-	-	-	-	-
23/01/01	-	-	-	-	-	-	-	-	-	-	-	-	-	-
5/04/01	1.2	1.3	0.213	0.865	0.006	0.058	0.075	0.0008	0.019	<0.0001	<0.00008	0.012	<0.002	*
5/07/01	-	-	-	-	-	-	-	-	-	-	-	-	-	-
16/10/01	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8/04/02	1.8	1.92	0.251	0.779	<0.005	0.03	0.073	<0.001	0.018	<0.0005	<0.00008	0.01	<0.01	*
14/10/02	-	-	-	-	-	-	-	-	-	-	-	-	-	-
7/10/03	-	-	-	-	-	-	-	-	-	-	-	-	-	-
1/04/04	1.7	1.76	0.190	0.802	<0.003	0.022	0.069	<0.0005	0.021	<0.0003	<0.00008	0.02	<0.005	*
4/10/04	-	-	-	-	-	-	-	-	-	-	-	-	-	-
4/04/05	1.1	2.27	0.320	0.655	0.016	0.521	0.096	0.0015	0.018	<0.0003	<0.00008	0.03	<0.005	*
19/10/05	-	-	-	-	-	-	-	-	-	-	-	-	-	-
5/04/06	0.8	1.35	0.043	-	0.026	0.144	-	-	-	-	-	0.012	-	-
19/10/06	-	-	-	-	-	-	-	-	-	-	-	-	-	-
4/04/07	1.6	1.62	0.016	0.819	0.006	0.016	0.065	<0.001	0.016	<0.0005	<0.00008	<0.01	<0.01	*

[#] = Naphthalene = 0.0047g m^{-3}, all parameters below detection limit.

[##] = 1,4-Dichlorbenzene = 0.0005 g m^{-3}, all parameters below detection limit.

* = all parameters below detection limit.

Table A1.3: Monitoring Data for Leachate Bore 2

Date	**pH**	**Conductivity** (μS cm^{-1})	**Cl$^-$** (g m^{-3})	**Na (soluble)** (g m^{-3})	**Mg (soluble** (g m^{-3})	**Ca (soluble)** (g m^{-3})
29/10/2008	7.1	11900	1400	1400	180	280
27/11/2008	7.1	12100	2300	1200	150	250
21/05/2009	7.1	12800	2300	1500	180	260
4/06/2009	7.1	12900	2100	1600	180	250
16/06/2009	7.0	13400	1200	1300	180	260

Table A1.4: Monitoring Data for Leachate Bore 6

Date	**pH**	**Conductivity** (μS cm^{-1})	**Cl$^-$** (g m^{-3})	**Na (soluble)** (g m^{-3})	**Mg (soluble** (g m^{-3})	**Ca (soluble)** (g m^{-3})
29/10/2008	7.4	12900	1500	1100	130	140
27/11/2008	7.2	13800	1400	920	120	130
21/05/2009	7.2	14300	1500	1100	140	120
4/06/2009	7.2	13600	1500	1000	130	110
16/06/2009	7.1	13300	640	900	120	110

Table A1.5: Monitoring Data for Leachate Irrigation Tank

Date	**pH**	**Conductivity** (μS cm^{-1})	**Cl$^-$** (g m^{-3})	**Na (soluble)** (g m^{-3})	**Mg (soluble** (g m^{-3})	**Ca (soluble)** (g m^{-3})
29/10/2008	7.4	6310	720	430	75	230
27/11/2008	7.5	6260	670	520	87	250
21/05/2009	7.7	2460	260	230	37	110
4/06/2009	7.9	250	17	16	3	30
16/06/2009	6.8	4220	420	360	75	30

Appendix 2

Table A2: Semi Volatile Organic Compounds Screen.

Compound	Detection Limit (g m^{-3})
Bis(2-chloroethyl)ether	<0.002
1,3-Dichlorobenzene	<0.002
1,4- Dichlorobenzene	<0.002
1,2- Dichlorobenzene	<0.002
Bis(2-chloroisopropy)ether	<0.002
N-nitrosodi-n-proyl amine	<0.002
Hexachloroethane	<0.002
Nitrobenzene	<0.002
Isophorone	<0.002
Bis(2-chloroethoxy)methane	<0.002
1,2,4-Trichlorobenzene	<0.002
Napthalene	<0.002
Hexachlorobutadiene	<0.002
2-Methylnapthalene	<0.001
2-Chloronaphthalene	<0.001
Acenaphthylene	<0.001
2,6-Dinitrotoluene	<0.004
Acenaphthene	<0.001
Dibenzofuran	<0.002
2,4-Dinitrotoluene	<0.004
Fluorene	<0.001
4-Chlorophenylphenylether	<0.002
N-Nitrosodiphenylamine	<0.002
Hexachlorobenzene	<0.002
Phenanthrene	<0.001
Anthracene	<0.001
Carbazole	<0.001
Fluoranthene	<0.002
Pyrene	<0.001
Benzo[a]anthracene	<0.001
Chrysene	<0.001
Benzo[b]Fluorathene	<0.001
Benzo[k]Fluorathene	<0.001
Benzo[a]pyrene	<0.001
Indeno(1,2,3-c,d)pyrene	<0.001
Dibenzo[a,h]anthracene	<0.001
Benzo[g,h,i]perylene	<0.001
Dimethylphthalate	<0.004
Diethylphthalate	<0.004
Di-n-butylphthlate	<0.004
Bis(2-ethylhexyl)phthalate	<0.05
Di-n-octyphthalate	<0.004
Butylbenzylphthalate	<0.004

Table A2: Semi Volatile Organic Compounds Screen.

Compond	Detection Limit (g m^{-3})
Di-(ethylhexyl)adipate	<0.004
Alpha BHC	<0.002
Beta BHC	<0.002
Gamma BHC (Lindathe)	<0.002
Delta BHC	<0.002
Heptachior epoxide	<0.002
Endosulfan l	<0.004
4,4-DDE	<0.002
Dieldrin	<0.002
Endrin	<0.002
Endrin Aldehyde	<0.004
Endosulfan 11	<0.004
4,4-DDD	<0.002
Endosulfan sulphate	<0.004
4,4-DDT	<0.004
Phenol	<0.004
2-Chlorophenol	<0.004
2-Methylphenol	<0.004
3 & 4-Methylphenol	<0.004
2-Nitrophenol	<0.004
2,4-Dimethlphenol	<0.007
2,4-Dichlorophenol	<0.004
4-Chloro-3-methylphenol	<0.004
2,4,6-Trichlorophenol	<0.004
2,4,5-Trichlorophenol	<0.004

Appendix 3

Table A3.1: Soil nutrient test results

REPORT OF SOIL ANALYSIS

Paokahu Landfill

Sample Name	Lab Number	pH	Olsen-soluble P ug/mL	Calcium MAF Units	Magnesium MAF Units	Potassium MAF Units	Sodium MAF Units	Sulphate-S ug/g	Ext.Org. Sulphur ug/g	Anaerobic Min. N kg/ha	Soluble Salts % w/w	Org. Matter % w/w	Exch. Al mg/kg
23-Mar-05													
Paokahu Landfill Mp6-MP3	264435	7.5	8	61	51	19	49	44	13	94	--	1.1	< 0.5
Paokahu landfill MP5-Mp6	264436	7.7	9	57	53	17	88	69	17	70	--	1.2	< 0.5
March 2005 Average		**7.6**	**9**	**59**	**52**	**18**	**69**	**57**	**15**	**82**			
23-Dec-05													
178MP6 Landfill MP6-3	325728	7.4	16	54	49	16	83	85	4		0.08		
178MP5 Landfill MP5-6	325727	7.5	20	56	37	16	33	35	4		0.15		
178MP4 Landfil MP4-Rd	325729	7.5	37	57	42	16	45	49	4		0.14		
Dec 2005 Average		**7.5**	**24**	**56**	**43**	**16**	**54**	**56**	**4**		**0.1**		
11-Apr-07													
178MP6 Landfill MP6-3	420859	7.9	26	57	53	22	70	28	4		0.10	3.2	
178MP5 Landfill MP5-6	420858	7.7	29	56	64	18	105	71	3		0.18	3.9	
178MP4 Landfil MP4-Rd	420860	7.8	37	54	51	20	48	26	4		0.09	3.3	
April 2007 Avg		**7.8**	**31**	**56**	**56**	**20**	**74**	**42**	**4**		**0.1**		
Optimum		**6.2**	**25-29**	**4+**	**25+**	**8+**	**<100**	**10+**	**10+**		**<0.15**		

Very High	>0.7	>35
High	0.3-0.	17-35
Medium	0.15-(	7-17
Low	0.05-(	3-7
Very Low	<0.05	<3

Carla Pell, Dip.App.Sci. for ARL Ltd

Enquiries: Michael White M. Hort Sc. (Hons)

Methods of analysis, along with relevant statistical data, are available on request. Results are based on the sample(s) as received.
Note: Cation exchange capacity (CEC) is determined by summation, and may be overestimated in alkaline soils and/or soils with high soluble salts. Contact the laboratory for further information.
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Appendix 4

Table A4:1A Monitoring Data for Bore 401A

DATE	Temperature (°C)	pH	Conductivity (μS cm^{-1})	BOD (g m^{-3})	COD (g m^{-3})	N-NH$_3$ (g m^{-3})	NO$_3^-$ (g m^{-3})	NO$_2^-$ (g m^{-3})
24/08/1999	15.8	8.3	1900	-	-	0.50	0.020	-
21/09/1999	15.5	8.3	1670	-	-	0.40	0.004	-
19/10/1999	15.6	8.2	2400	-	23	0.40	0.020	0.002
16/11/1999	15.6	8.2	1560	-	-	0.30	0.002	-
14/12/1999	15.6	8.3	1600	-	-	0.40	<0.02	-
25/01/2000	15.7	8.2	1640	-	-	0.50	<0.02	-
24/02/2000	16	8.1	1675	0.6	15	0.50	<0.02	0.002
14/03/2000	16	8.2	1770	-	-	<0.1	<0.02	-
4/04/2000	16	8.2	1940	-	-	0.50	<0.002	-
2/05/2000	-	8.3	1700	-	-	0.10	0.180	-
1/06/2000	16.1	8.2	1820	-	-	-	<0.002	-
13/07/2000	15.9	8.2	1550	-	-	0.50	0.004	-
10/08/2000	15.5	8.3	1490	-	-	0.50	-	-
5/09/2000	15.4	8.2	1510	-	-	0.50	<0.02	-
3/10/2000	15.4	8.2	1500	-	5	0.50	<0.02	-
7/11/2000	15.2	8.2	1600	-	-	0.40	<0.02	-
4/12/2000	15.2	8.2	1590	-	-	0.60	<0.02	-
23/01/2001	15.4	8.2	1610	-	-	0.60	0.004	-
7/02/2001	15.4	8.2	1730	-	-	0.60	0.02	-
7/03/2001	15.7	8.1	1940	-	-	0.60	<0.02	-
5/04/2001	16	8.1	1890	0.5	28	0.55	0.002	<0.002
3/05/2001	16.8	8.0	1940	-	-	0.50	0.002	-
6/06/2001	16	8.1	1880	-	-	0.40	<0.002	-
5/07/2001	16	8.2	1790	-	-	0.51	0.005	-
1/10/2001	15.3	8.2	1500	8	30	0.38	<0.02	-
8/04/2002	15	8.2	1680	1.1	35	0.50	<0.05	-
8/07/2002	15.1	8.2	1680	-	-	0.45	<0.002	-
20/01/2003	15.2	8.2	1610	-	-	0.42	<0.002	-
7/10/2003	15.5	8.2	1470	-	-	0.49	<0.002	-
1/04/2004	16.3	8.0	1480	0.5	25	0.49	<0.002	-
5/07/2004	15.5	8.2	1400	-	-	0.25	0.04	-
4/10/2004	15.1	8.1	1440	-	-	0.48	0.01	-
17/01/2005	17	8.2	1450	-	-	0.45	0.002	-
4/04/2005	15.7	8.1	1590	0.5	30	0.45	0.005	-
10/10/2005	15.0	8.2	1360	-	-	0.31	0.004	-
5/04/2006	15.8	8.1	1450	<0.5	<10	0.37	0.004	<0.002
19/10/2006	13.0	8.2	1352	-	-	0.25	0.026	-
4/04/2007	15.8	8.2	1420	<5	45	0.41	0.0	<0.002
8/10/2007	15.2	8.2	1370	<5	5	0.94	0.007	
3/04/2008	15.9	8.3	1480	0.6	20	0.44	<0.0020	0.004

Table A4:1B Monitoring Data for Bore 401A

DATE	**Na (soluble)** (g m^{-3})	**K (soluble)** (g m^{-3})	**Ca (soluble)** (g m^{-3})	**Mg (soluble)** (g m^{-3})	**Cl$^-$** (g m^{-3})	**SO$_4^{2-}$** (g m^{-3})	**Alkalinity** (g m^{-3})
24/08/1999	400	24.2	9.4	10.2	321	16.0	520
21/09/1999	365	22.0	7.1	8.1	270	11.0	530
19/10/1999	334	21.1	7.1	8.1	170	1.0	555
16/11/1999	363	21.2	7.1	8.0	170	1.0	590
14/12/1999	343	20.3	7.0	7.8	72	1.0	585
25/01/2000	346	-	7.4	8.2	200	1.0	580
24/02/2000	368	19.0	8.4	8.4	220	1.0	650
14/03/2000	334	-	8.1	8.6	250	4.0	550
4/04/2000	393	24.9	9.4	10.9	305	13.0	540
2/05/2000	358	19.3	6.8	7.1	190	3.4	580
1/06/2000	381	22.7	7.7	8.9	250	8.2	550
13/07/2000	335	22.2	7.2	7.5	170	0.5	580
10/08/2000	338	19.6	6.4	6.6	115	0.5	570
5/09/2000	324	20.6	6.3	7.3	170	0.5	570
3/10/2000	323	20.2	6.1	7.0	175	0.5	575
7/11/2000	339	22.3	6.4	8.4	190	0.5	580
4/12/2000	364	23.5	6.9	8.0	180	0.5	570
23/01/2001	340	23.5	6.9	7.7	180	0.5	570
7/02/2001	349	23.9	7.5	8.9	270	1.5	570
7/03/2001	401	25.0	10.0	11.0	300	7.6	530
5/04/2001	417	26.7	9.1	10.7	320	11.4	550
3/05/2001	388	26.0	9.1	10.6	320	11.6	550
6/06/2001	406	27.5	9.3	9.2	270	5.0	550
5/07/2001	403	-	8.3	10.5	260	0.5	570
1/10/2001	342	21.9	6.4	6.9	180	<0.5	580
8/04/2002	374	24.0	6.7	8.8	210	2.8	560
8/07/2002	363	24.5	7.0	8.7	260	10.5	530
20/01/2003	357	24.3	6.7	8.0	220	1.0	550
7/10/2003	353	22.4	5.8	7.6	180	-	550
1/04/2004	328	21.5	5.9	7.5	180	2.5	560
5/07/2004	358	24.0	6.0	6.7	120	-	610
4/10/2004	365	24.4	6.1	7.4	110	-	650
17/01/2005	335	24.5	6.8	7.9	140	<0.5	580
4/04/2005	345	24.8	6.5	8.5	200	18.1	550
10/10/2005	319	24.0	5.54	6.76	110	3.1	570
5/04/2006	356	22.8	5.85	8.35	160	8.9	550
19/10/2006	297	20.2	5.48	6.68	101	1.7	600
4/04/2007	325	21.8	5.23	7.36	149	11.4	550
8/10/2007	316	21.0	5.07	6.69	129	15.9	520
3/04/2008	330	23.0	6.10	8.30	170	5.2	570

Table A4:1C Monitoring Data for Bore 401A

DATE	Fe (soluble) (g m^{-3})	B (soluble) (g m^{-3})	DRP (g m^{-3})	Mn (soluble) (g m^{-3})	Cu (soluble) (g m^{-3})	Zn (soluble) (g m^{-3})	Ni (soluble) (g m^{-3})	Pb (soluble) (g m^{-3})
24/08/1999	-	0.480	0.47	-	-	-	-	-
21/09/1999	-	0.400	0.44	-	-	-	-	-
19/10/1999	2.36	0.400	0.47	0.0160	0.0010	0.010	0.001	0.00050
16/11/1999	-	0.440	0.50	-	-	-	-	-
14/12/1999	-	0.402	0.51	-	-	-	-	-
25/01/2000	-	0.420	0.51	-	-	-	-	-
24/02/2000	2.2	0.384	0.42	0.0202	0.0014	0.004	0.0005	0.00100
14/03/2000	-	0.420	0.50	-	-	-	-	-
4/04/2000	-	0.410	0.55	-	-	-	-	-
2/05/2000	-	0.430	0.94	-	-	-	-	-
1/06/2000	-	0.440	0.56	-	-	-	-	-
13/07/2000	-	0.432	0.74	-	-	-	-	-
10/08/2000	-	0.442	0.90	-	-	-	-	-
5/09/2000	-	0.447	0.98	-	-	-	-	-
3/10/2000	2.43	0.435	1.13	0.0182	0.0022	0.003	0.01	0.00080
7/11/2000	-	0.409	0.88	-	-	-	-	-
4/12/2000	-	0.410	0.65	-	-	-	-	-
23/01/2001	-	0.403	0.47	-	-	-	-	-
7/02/2001	-	0.416	0.64	-	-	-	-	-
7/03/2001	-	0.385	0.62	-	-	-	-	-
5/04/2001	1.87	0.408	0.57	0.0216	0.0005	0.001	0.0005	0.00030
3/05/2001	-	0.414	0.69	-	-	-	-	-
6/06/2001	-	0.451	0.60	-	-	-	-	-
5/07/2001	-	0.470	0.70	-	-	-	-	-
1/10/2001	2.34	-	1.01	0.0197	0.0032	0.006	0.0011	0.00090
8/04/2002	2.12	0.404	0.62	0.0170	0.0005	0.001	<0.0005	0.00040
8/07/2002	-	0.412	0.56	-	-	-	-	-
20/01/2003	-	0.366	0.48	-	-	-	-	-
7/10/2003	-	0.360	2.89	-	-	-	-	-
1/04/2004	2.25	0.380	0.83	0.0200	0.0010	0.002	<0.0007	0.00060
5/07/2004	-	0.430	0.82	-	-	-	-	-
4/10/2004	-	0.500	0.82	-	-	-	-	-
17/01/2005	-	0.425	0.88	-	-	-	-	-
4/04/2005	2.10	0.435	0.56	0.0184	0.0008	<0.001	<0.0005	0.00030
10/10/2005	-	0.389	1.25	-	-	-	-	-
5/04/2006	2.0	0.345	0.70	0.0160	0.0013	0.002	<0.001	0.00050
19/10/2006	-	0.387	1.90		-	-	-	-
4/04/2007	2.2	0.307	0.50	0.0142	0.0010	<0.001	<0.0005	0.00040
8/10/2007	-	0.317	-	-	-	-	-	-
3/04/2008	1.90	0.290	0.42	0.0150	<0.0010	0.001	<0.0005	0.00028

Table A4:1D Monitoring Data for Bore 401A

DATE	**Cr (soluble)** (g m^{-3})	**Cd (soluble)** (g m^{-3})	**Hg (soluble)** (g m^{-3})	**Se (soluble)** (g m^{-3})	**SVOC** (g m^{-3})
24/08/1999	-	-	-	-	-
21/09/1999	-	-	-	-	-
19/10/1999	0.0010	<0.0001	<0.00008	<0.002	*
16/11/1999	-	-	-	-	-
14/12/1999	-	-	-	-	-
25/01/2000	-	-	-	-	-
24/02/2000	0.0015	<0.00005	<0.00008	<0.001	*
14/03/2000	-	-	-	-	-
4/04/2000	-	-	-	-	-
2/05/2000	-	-	-	-	-
1/06/2000	-	-	-	-	-
13/07/2000	-	-	-	-	-
10/08/2000	-	-	-	-	-
5/09/2000	-	-	-	-	-
3/10/2000	0.0050	<0.00005	<0.00008	<0.001	*
7/11/2000	-	-	-	-	-
4/12/2000	-	-	-	-	-
23/01/2001	-	-	-	-	-
7/02/2001	-	-	-	-	-
7/03/2001	-	-	-	-	-
5/04/2001	0.0005	<0.00005	<0.00008	<0.001	*
3/05/2001	-	-	-	-	-
6/06/2001	-	-	-	-	-
5/07/2001	-	-	-	-	-
1/10/2001	0.0030	<0.00005	<0.00008	<0.001	-
8/04/2002	<0.001	<0.00005	<0.00008	<0.001	*
8/07/2002	-	-	-	-	-
20/01/2003	-	-	-	-	-
7/10/2003	-	-	-	-	-
1/04/2004	<0.0012	<0.00005	<0.00008	<0.001	*
5/07/2004	-	-	-	-	-
4/10/2004	-	-	-	-	-
17/01/2005	-	-	-	-	-
4/04/2005	<0.001	<0.00005	<0.00008	<0.001	-
10/10/2005	-	-	-	-	-
5/04/2006	<0.001	0.0001	<0.00008	<0.001	*
19/10/2006	-	-	-	-	-
4/04/2007	0.0008	<0.00005	<0.00008	<0.001	*
8/10/2007	-	-	-	-	-
3/04/2008	0.0006	<0.00005	<0.00008	<0.001	*

* = All paramaters below the detection limit of the tests

Table A4.2: Monitoring Data for Bore 401B

DATE	**Temperature** (°C)	**pH**	**Conductivity** (μS cm^{-1})	**N-NH$_3$** (g/m^3)	**NO$_3^-$** (g/m^3)	**NO$_2^-$** (g/m^3)	**Na (soluble)** (g/m^3)	**K (soluble)** (g/m^3)	**Ca (soluble)** (g/m^3)	**Mg (soluble)** (g/m^3)	**Cl$^-$** (g/m^3)	**SO$_4^{2-}$** (g/m^3)	**Alkalinity** (g/m^3)	**Fe (soluble)** (g/m^3)	**B (soluble)** (g/m^3)
24/02/00	15.9	7.1	41300	2.40	<0.002	<0.002	7830	237	476	942	14100	1890	520	-	2.46
5/04/01	15.8	7.1	41000	2.29	<0.01	<0.01	8560	292	519	1040	15500	2010	520	0.6	2.6
8/04/02	15.5	7.1	41400	2.24	<0.05	-	8600	275	479	1040	16000	2110	510	<0.4	2.5
1/04/04	15.7	7.0	40900	2.37	0.002	<0.002	7990	264	468	901	18000	2390	530	2.3	2.7
4/04/05	15.5	7.1	41100	2.15	0.01	<0.001	8490	284	482	976	13000	2160	530	<1.0	3
5/04/06	15.5	7.0	41300	2.22	0.03	<0.01	8180	272	497	1080	14000	2080	530	<2.0	2.7
4/04/07	15.8	7.1	41000	2.27	<0.002	<0.002	7770	270	481	1010	14200	2020	530	<2.0	2.6

Table A4.3: Monitoring Data for Bore 401C

DATE	**Temperature** (°C)	**pH**	**Conductivity** (μS cm^{-1})	**N-NH$_3$** (g/m^3)	**NO$_3^-$** (g/m^3)	**NO$_2^-$** (g/m^3)	**Na (soluble)** (g/m^3)	**K (soluble)** (g/m^3)	**Ca (soluble)** (g/m^3)	**Mg (soluble)** (g/m^3)	**Cl$^-$** (g/m^3)	**SO$_4^{2-}$** (g/m^3)	**Alkalinity** (g/m^3)	**Fe (soluble)** (g/m^3)	**B (soluble)** (g/m^3)
24/02/00	15.8	7.3	43700	2.00	0.002	<0.002	8050	237	525	1120	15900	2210	480	-	2.4
5/04/01	15.5	7.2	44100	2.55	0.01	<0.01	8970	307	555	1150	16700	2350	450	<0.4	2.6
8/04/02	15.3	7.2	44400	2.60	<0.05	-	8960	298	520	1150	17000	2410	450	<1.0	2.8
1/04/04	15.7	7.2	43700	2.70	<0.002	<0.002	8690	286	503	999	17000	2750	450	0.5	2.7
4/04/05	15.6	7.1	44100	2.52	<0.01	<0.01	9570	327	548	1170	14000	2280	460	<1.0	3.4
5/04/06	15.3	7.1	44400	2.81	<0.01	<0.01	8850	300	547	1240	17000	2420	450	<2.0	2.9
4/04/07	15.5	7.1	44200	2.53	0.02	0.073	8580	309	548	1190	16500	2350	450	<2.0	2.7

Table A4.4: Monitoring Data for Bore 402A

DATE	**Temperature** (°C)	**pH**	**Conductivity** (μS cm^{-1})	**BOD** (g m^{-3})	**COD** (g m^{-3})	**N-NH$_3$** (g m^{-3})	**NO$_3^-$** (g m^{-3})	**NO$_2^-$** (g m^{-3})	**Na (soluble)** (g m^{-3})	**K (soluble)** (g m^{-3})	**Ca (soluble)** (g m^{-3})	**Mg (soluble)** (g m^{-3})	**Cl$^-$** (g m^{-3})	**SO$_4^{2-}$** (g m^{-3})	**Alkalinity** (g m^{-3})
24-Feb-00	15.6	7.2	32700	1.1	134	2.00	0.008	0.002	5810	178	488	783	12300	1620	350
5-Apr-01	15.3	7.2	32400	0.7	-	1.89	<0.01	0.010	6150	221	522	772	11700	1720	340
8-Apr-02	15.3	7.2	32700	<0.5	140	1.9	<0.05	-	6380	209	461	772	12000	1920	340
1-Apr-04	15.7	7.3	33200	<0.5	400	1.88	0.011	-	6430	212	411	700	13000	2140	330
4/04/2005	15.5	7.3	34900	0.5	140	1.78	0.010	<0.1	7140	223	445	811	10000	1940	320
5/04/2006	15.5	7.1	35200	<0.5	260	1.83	0.020	<0.1	7330	217	458	832	13000	1940	320
4/04/2007	15.5	7.1	36800	<0.5	120	2.08	0.009	<0.002	7450	241	485	957	12600	1980	400
8/10/2007	15.2	7.2	35700	<0.5	120	2.11	0.002	-	7180	217	446	816	11600	1940	310
3/04/2008	15.2	7.2	36600	0.4	120	2.30	0.014	<0.01	6500	230	460	860	13000	2000	320

DATE	**Fe (soluble)** DATE	**B (soluble)** (g m^{-3})	**DRP** (g m^{-3})	**Mn (soluble)** (g m^{-3})	**Cu (soluble)** (g m^{-3})	**Zn (soluble)** (g m^{-3})	**Ni (soluble)** (g m^{-3})	**Pb (soluble)** (g m^{-3})	**Cr (soluble)** (g m^{-3})	**Cd (soluble)** (g m^{-3})	**Hg (soluble)** (g m^{-3})	**Se (soluble)** (g m^{-3})	**SVOC** (g m^{-3})
24-Feb-00	-	1.5	0.004	1.19	<0.01	<0.02	<0.01	<0.00008	<0.01	-	<0.001	*	-
5-Apr-01	<0.4	1.7	0.009	1.05	<0.01	<0.02	<0.01	<0.00008	<0.01	<0.002	<0.001	*	<0.02
8-Apr-02	<0.4	2.1	<0.02	0.95	<0.01	<0.02	<0.01	<0.00008	<0.01	<0.002	<0.001	*	<0.02
1-Apr-04	<0.4	2.8	0.007	0.99	<0.01	<0.02	<0.01	<0.00008	0.001	-	<0.001	*	<0.02
4/04/2005	<1	3.4	<0.02	1.00	<0.03	<0.05	<0.03	<0.00008	<0.03	<0.005	<0.003	*	<0.05
5/04/2006	<0.2	2.7	0.018	0.95	<0.005	<0.01	<0.01	<0.00008	<0.005	<0.001	<0.0005	*	<0.01
4/04/2007	<2	3.2	<0.004	1.13	<0.05	<0.1	<0.05	<0.00008	<0.05	<0.01	<0.005	*	<0.1
8/10/2007	-	2.9	-	-	-	-	-	-	-	-	-	-	-
3/04/2008	<0.2	3.3	<0.004	1.10	<0.005	<0.02	<0.05	<0.00008	<0.05	<0.01	<0.005	*	<0.01

Table A4.5: Monitoring Data for Bore 402B

DATE	**Temperature** (°C)	**pH**	**Conductivity** (µS cm^{-1})	**$N\text{-}NH_3$** (g/m^3)	**NO_3^-** (g/m^3)	**NO_2^-** (g/m^3)	**Na (soluble)** (g/m^3)	**K (soluble)** (g/m^3)	**Ca (soluble)** (g/m^3)	**Mg (soluble)** (g/m^3)	**Cl^-** (g/m^3)	**SO_4^{2-}** (g/m^3)	**Alkalinity** (g/m^3)	**Fe (soluble)** (g/m^3)	**B (soluble)** (g/m^3)
24/02/00	15.2	7.3	43500	2.10	<0.002	<0.002	8430	145	332	1040	16200	2220	300	-	3.5
5/04/01	15.0	7.2	43500	2.41	<0.02	<0.01	9150	270	359	1050	16900	2260	300	0.4	3.4
8/04/02	15.1	7.3	43900	2.35	<0.05	-	8910	254	337	1050	17000	2280	210	<1.0	3.8
1/04/04	16.3	7.2	43600	2.56	0.023	<0.002	9040	255	345	943	16000	2730	280	0.4	3.7
4/04/05	15.2	7.2	43700	2.24	0.03	<0.01	9670	285	356	1080	15000	2370	260	<1.0	4.5
5/04/06	15.0	7.1	43800	2.35	<0.01	<0.01	9050	261	354	1140	17000	2360	330	<2	3.8
4/04/07	15.3	7.2	43300	2.53	0.017	0.093	9010	259	343	1090	15700	1740	360	<2	4

Table A4.6: Monitoring Data for Bore 402C

DATE	**Temperature** (°C)	**pH**	**Conductivity** (µS cm^{-1})	**$N\text{-}NH_3$** (g/m^3)	**NO_3^-** (g/m^3)	**NO_2^-** (g/m^3)	**Na (soluble)** (g/m^3)	**K (soluble)** (g/m^3)	**Ca (soluble)** (g/m^3)	**Mg (soluble)** (g/m^3)	**Cl^-** (g/m^3)	**SO_4^{2-}** (g/m^3)	**Alkalinity** (g/m^3)	**Fe (soluble)** (g/m^3)	**B (soluble)** (g/m^3)
24/02/00	15.2	7.0	21100	30.0	<0.02	0.003	4040	128	112	319	6250	2250	2250	-	3.2
5/04/01	15.6	7.3	21000	31.8	0.1	0.08	4720	181	131	380	6500	2400	2400	<0.2	3.6
8/04/02	16.3	7.4	21200	31.0	<0.05	-	4420	160	119	370	7200	2300	2300	<0.2	3.5
1/04/04	15.8	7.3	20900	27.4	0.127	0.012	4300	161	116	335	6800	2500	2500	0.2	3.6
4/04/05	16.2	7.4	21100	28.1	0.212	0.009	4310	166	117	347	4000	2500	2500	<0.20	3.8
5/04/06	15.5	7.4	21200	29.0	0.276	0.008	4410	168	130	412	6100	2500	2500	<0.4	3.7
4/04/07	15.8	7.0	21300	32.4	0.117	0.005	8880	160	121	356	7400	2600	2600	<0.2	3.0
8/10/07	15.1	7.1	21200	32.0	0.008	-	4760	161	124	384	7970	2600	2600	-	3.8
3/04/2008	15.7	7.1	24300	33.0	0.15	0.015	3900	170	130	370	6600	2500	2500	<0.2	3.3

Table A4.7: Monitoring Data for Bore 403A

DATE	**Temperature** (°C)	**pH**	**Conductivity** (μS cm^{-1})	**BOD** (g m^{-3})	**COD** (g m^{-3})	**N-NH_3** (g m^{-3})	**NO_3^-** (g m^{-3})	**NO_2^-** (g m^{-3})	**Na (soluble)** (g m^{-3})	**K (soluble)** (g m^{-3})	**Ca (soluble)** (g m^{-3})	**Mg (soluble)** (g m^{-3})	**Cl^-** (g m^{-3})	**SO_4^{2-}** (g m^{-3})	**Alkalinity** (g m^{-3})
24-Feb-00	16.4	7.6	700	3	<0.5	<0.1	0.73	0.002	45	15	57	26	-	43	290
5-Apr-01	16.3	7.6	775	5	<0.5	0.02	0.51	0.024	54	18	46	24	53	52	250
8-Apr-02	16.4	7.6	697	<5	<0.5	<0.01	1.08	-	43	17	48	26	33	44	250
1-Apr-04	16.3	7.6	652	<5	<0.5	<0.01	1.48	0.027	30	16	54	27	33	49	270
4-Apr-05	16.0	7.6	649	<5	<0.5	<0.01	2.29	0.011	29	16	54	30	38	48	260
5/04/2006	16.4	7.6	576	<5	<0.5	<0.01	0.898	0.007	27	18	53	29	13	25	280
4/04/2007	16.1	7.5	675	35	<0.5	<0.01	1.43	0.005	26	19	58	29	44	36	270
4/10/2007	15.7	7.5	647	<5	<0.5	<0.01	1.49	-	24	17	60	27	30	32	280
3/04/2008	18.7	7.6	616	<5	<0.5	<0.01	0.83	0.008	20	19	60	27	41	26	260

DATE	**Fe (soluble)** (g m^{-3})	**B (soluble)** (g m^{-3})	**DRP** (g m^{-3})	**Mn (soluble)** (g m^{-3})	**Cu (soluble)** (g m^{-3})	**Zn (soluble)** (g m^{-3})	**Ni (soluble)** (g m^{-3})	**Pb (soluble)** (g m^{-3})	**Cr (soluble)** (g m^{-3})	**Cd (soluble)** (g m^{-3})	**Hg (soluble)** (g m^{-3})	**Se (soluble)** (g m^{-3})	**SVOC** (g m^{-3})
24-Feb-00	0.02	0.111	0.015	0.0021	<0.0005	0.004	<0.0005	-	<0.0005	<0.00005	<0.00008	<0.001	*
5-Apr-01	0.02	0.097	0.016	0.0046	<0.0005	<0.001	<0.0005	<0.0001	<0.0005	<0.00005	<0.00008	<0.001	*
8-Apr-02	<0.02	0.087	0.013	0.0011	<0.0005	<0.001	<0.0005	<0.0001	<0.0005	<0.00005	<0.00008	<0.0001	*
1-Apr-04	<0.02	0.087	0.013	<0.005	<0.0005	<0.001	<0.0005	<0.0001	<0.0005	<0.00005	<0.00008	<0.0001	*
4-Apr-05	<0.02	0.085	0.015	0.0007	<0.0005	<0.001	<0.0005	<0.0001	<0.0005	0.00005	<0.00008	<0.001	*
5/04/2006	<0.02	0.077	0.012	<0.0005	<0.0005	<0.001	<0.0005	<0.0001	<0.0005	<0.00005	<0.00008	<0.001	*
4/04/2007	<0.02	0.064	0.007	<0.0005	<0.0005	<0.001	<0.0005	<0.0001	<0.0005	<0.00005	<0.00008	<0.001	*
4/10/2007	-	0.07	-	-	-	-	-	-	-	-	-	-	-
3/04/2008	<0.02	0.069	0.014	0.0007	<0.0005	<0.001	<0.0005	<0.0001	<0.0005	<0.00005	<0.00008	<0.001	*

* = All paramaters below the detection limit of the tests

Table A4.8: Monitoring Data for Bore 403B

DATE	Temperature (°C)	pH	Conductivity (µS cm^{-1})	N-NH$_3$ (g/m^3)	NO$_3^-$ (g/m^3)	NO$_2^-$ (g/m^3)	Na (soluble) (g/m^3)	K (soluble) (g/m^3)	Ca (soluble) (g/m^3)	Mg (soluble) (g/m^3)	Cl$^-$ (g/m^3)	SO$_4^{2-}$ (g/m^3)	Alkalinity (g/m^3)	Fe (soluble) (g/m^3)	B (soluble) (g/m^3)
5/04/00	16.1	7.6	49100	0.10	<0.002	<0.002	9330	292	389	1170	18000	2530	130	-	3.4
5/04/01	16.0	7.4	49100	0.23	0.010	0.01	10800	395	422	1240	18900	2570	170	0.4	3.8
8/04/02	15.8	7.4	49700	0.16	<0.05	-	10400	367	388	1210	20000	2630	160	<1.0	4.0
1/04/04	15.8	7.4	49300	0.20	<0.002	<0.002	9740	365	407	1190	18000	3150	170	<0.4	4.1
4/04/05	15.6	7.4	49300	0.16	<0.01	<0.01	8820	300	359	1060	18000	2770	170	<1.0	4.0
5/04/06	15.7	7.3	49400	0.26	<0.01	<0.01	10300	382	415	1320	19000	2650	160	<2.0	4.4
4/04/07	15.7	7.4	48600	0.26	<0.002	<0.002	9760	370	381	1230	18300	2670	160	<2.0	4.1

Table A4.9: Monitoring Data for Bore 403B

DATE	Temperature (°C)	pH	Conductivity (µS cm^{-1})	N-NH$_3$ (g/m^3)	NO$_3^-$ (g/m^3)	NO$_2^-$ (g/m^3)	Na (soluble) (g/m^3)	K (soluble) (g/m^3)	Ca (soluble) (g/m^3)	Mg (soluble) (g/m^3)	Cl$^-$ (g/m^3)	SO$_4^{2-}$ (g/m^3)	Alkalinity (g/m^3)	Fe (soluble) (g/m^3)	B (soluble) (g/m^3)
24/02/00	16.2	7.4	49400	3.00	<0.002	<0.002	9360	199	396	1240	18200	2460	220	-	3.2
5/04/01	15.7	7.2	49500	3.28	<0.01	<0.01	10700	382	436	1310	19300	2620	220	<0.4	3.8
8/04/02	15.8	7.3	49600	3.44	<0.05	-	10300	353	405	1290	19000	2580	240	<1.0	3.8
1/04/04	16.2	7.4	49100	<.05	3.09	0.009	9740	365	407	1190	19000	3150	220	<1.0	3.9
4/04/05	15.7	7.2	49300	1.25	1.37	0.037	11200	399	430	1310	16000	2800	240	<1.0	4.9
5/04/06	16.0	7.4	49800	0.95	1.78	<0.01	10200	364	426	1380	18000	2710	240	<2.0	3.9
4/04/07	15.7	7.2	49600	3.04	0.109	0.031	10500	382	422	1400	18800	1950	250	<2.0	3.8

Table A4.10A: Monitoring Data for Bore 404B

DATE	**Temperature** (°C)	**pH**	**Conductivity** (µS cm^{-1})	**BOD** (g m^{-3})	**COD** (g m^{-3})	**N-NH$_3$** (g m^{-3})	**NO$_3^-$** (g m^{-3})	**NO$_2^-$** (g m^{-3})
24/08/1999	18.7	7.2	22000	-	-	2.20	0.002	-
21/09/1999	18.7	7.3	21500	-	-	2.60	0.002	0.002
19/10/1999	18.7	7.2	21500	-	-	2.20	0.002	0.002
16/11/1999	18.9	7.2	20700	-	-	1.90	<0.002	-
14/12/1999	18.9	7.2	20800	-	-	2.40	<0.02	-
25/01/2000	19.0	7.2	21500	-	-	2.00	0.003	-
24/02/2000	19.1	7.3	21400	3	-	4.30	<0.002	0.002
14/03/2000	20.0	7.2	21400	-	-	1.50	<0.02	-
4/04/2000	19.1	7.2	22400	-	-	2.20	<0.002	-
2/05/2000	19.0	7.2	21600	-	-	2.50	<0.002	-
1/06/2000	19.1	7.2	21500	-	-	-	<0.002	-
13/07/2000	19.1	7.1	21400	-	-	2.60	<0.002	-
10/08/2000	19.1	7.2	21300	-	-	2.40	0.002	-
5/09/2000	19.3	7.1	21400	-	-	2.10	<0.002	-
3/10/2000	19.3	7.2	23000	-	250	1.90	<0.01	-
7/11/2000	19.4	7.1	23900	-	-	2.40	0.002	-
4/12/2000	19.3	7.1	21600	-	-	2.40	<0.002	-
23/01/2001	19.5	7.2	21200	-	-	2.20	0.002	-
7/02/2001	19.1	7.1	21300	-	-	2.00	<0.02	-
7/03/2001	19.4	7.1	21400	-	-	2.50	<0.002	-
5/04/2001	19.3	7.1	21200	1.5	138	2.36	<0.01	<0.01
3/05/2001	19.3	6.9	21300	-	-	2.40	<0.002	-
6/06/2001	19.2	7.1	21100	-	-	2.25	0.42	-
5/07/2001	19.0	7.2	21100	-	-	3.20	<0.002	-
1/10/2001	19.5	7.1	21000	<5	240	2.84	<0.02	-
8/04/2002	19.5	7.1	20800	1.8	140	2.34	<0.05	-
8/07/2002	19.1	7.2	21200	-	-	2.60	<0.002	-
20/01/2003	19.9	7.2	21100	-	-	2.56	0.01	-
7/10/2003	18.9	7.0	21800	-	-	3.13	0.01	-
1/04/2004	19.8	7.2	20900	1.5	<200	2.75	<0.01	<0.02
5/07/2004	16.9	7.3	20500	-	-	2.90	0.12	-
4/10/2004	19.1	7.0	21800	-	-	2.80	0.01	-
17/01/2005	21.5	7.1	22100	-	-	2.74	<0.002	-
4/04/2005	19.4	7.1	22500	0.6	160	2.65	0.040	0.010
10/10/2005	18.3	7.1	21600	-	-	2.85	0.190	-
5/04/2006	19.3	7.1	22900	<0.5	120	2.78	<0.01	<0.01
19/10/2006	17.7	7.1	22800	-	-	2.95	<0.01	-
4/04/2007	19.5	7.1	22800	1.4	140	2.66	0.002	<0.002

Table A4.10B: Monitoring Data for Bore 404B

DATE	**Na (soluble)** (g m^{-3})	**K (soluble)** (g m^{-3})	**Ca (soluble)** (g m^{-3})	**Mg (soluble)** (g m^{-3})	**Cl$^-$** (g m^{-3})	**SO$_4^{2-}$** (g m^{-3})	**Alkalinity** (g m^{-3})
24/08/1999	4780	179	479	205	8124	887	955
21/09/1999	4000	165	404	167	6950	369	1230
19/10/1999	4060	160	394	171	7100	450	1200
16/11/1999	4270	176	435	189	7050	429	1200
14/12/1999	4130	153	371	171	7050	428	1250
25/01/2000	4000	138	387	173	7300	443	1180
24/02/2000	3920	138	393	189	7000	474	1350
14/03/2000	4070	-	352	163	7200	430	1200
4/04/2000	4020	-	404	174	7450	444	1040
2/05/2000	4090	159	410	182	6950	459	1250
1/06/2000	4500	184	474	188	7800	683	900
13/07/2000	4210	173	440	199	5050	486	1100
10/08/2000	4180	157	378	171	6650	440	1200
5/09/2000	4320	156	407	176	7250	447	1160
3/10/2000	1350	156	395	173	6800	462	1210
7/11/2000	4180	176	450	186	7200	477	1180
4/12/2000	4390	184	460	203	7300	495	1160
23/01/2001	3840	168	404	187	7200	431	1230
7/02/2001	3810	167	393	188	7900	429	1200
7/03/2001	3950	165	414	192	7000	436	1190
5/04/2001	4430	181	438	198	7100	442	1300
3/05/2001	3930	169	426	198	7200	427	1220
6/06/2001	4160	192	411	226	7000	428	1200
5/07/2001	4020	185	422	207	6900	456	1220
1/10/2001	4170	169	371	197	7200	427	1200
8/04/2002	4280	170	440	186	7600	421	1300
8/07/2002	4120	174	422	196	7000	351	1200
20/01/2003	3940	174	395	197	7100	508	1400
7/10/2003	4420	184	395	221	7900	495	1200
1/04/2004	3960	176	487	220	7900	500	1200
5/07/2004	3840	201	477	210	7500	192	1300
4/10/2004	4490	206	497	226	5900	562	900
17/01/2005	4350	201	529	257	4800	495	1100
4/04/2005	3970	198	542	243	3700	530	1220
10/10/2005	4090	197	503	226	5600	389	1200
5/04/2006	4640	185	591	256	7200	609	970
19/10/2006	3810	183	589	257	7470	580	780
4/04/2007	4480	183	530	245	7990	542	1300

Table A4.10C: Monitoring Data for Bore 404B

DATE	**Fe (soluble)** (g m^{-3})	**B (soluble)** (g m^{-3})	**DRP** (g m^{-3})	**Mn (soluble)** (g m^{-3})	**Cu (soluble)** (g m^{-3})	**Zn (soluble)** (g m^{-3})
24/08/1999	-	2.44	-	-	-	-
21/09/1999	-	2.46	-	-	-	-
19/10/1999	-	2.46	0.018	0.71	<0.005	0.02
16/11/1999	-	2.68	-	-	-	-
14/12/1999	-	2.31	-	-	-	-
25/01/2000	-	2.31	-	-	-	-
24/02/2000	-	2.16	0.022	0.884	<0.005	<0.01
14/03/2000	-	2.27	-	-	-	-
4/04/2000	-	2.34	-	-	-	-
2/05/2000	-	2.41	-	-	-	-
1/06/2000	-	2.34	-	-	-	-
13/07/2000	-	2.28	-	-	-	-
10/08/2000	-	2.53	-	-	-	-
5/09/2000	-	2.71	-	-	-	-
3/10/2000	-	2.46	0.011	0.879	<0.005	<0.01
7/11/2000	-	2.23	-	-	-	-
4/12/2000	-	2.14	-	-	-	-
23/01/2001	-	2.29	-	-	-	-
7/02/2001	-	2.31	-	-	-	-
7/03/2001	-	2.4	-	-	-	-
5/04/2001	-	2.36	0.021	0.772	<0.005	<0.01
3/05/2001	-	2.36	-	-	-	-
6/06/2001	-	2.58	-	-	-	-
5/07/2001	-	2.65	-	-	-	-
1/10/2001	<0.2	?	-	0.772	<0.005	<0.01
8/04/2002	<0.2	2.39	0.02	0.717	<0.005	<0.01
8/07/2002	<0.2	2.32	-	-	-	-
20/01/2003	<0.2	2.04	-	-	-	-
7/10/2003	<0.2	2.38	-	-	-	-
1/04/2004	<0.2	2.19	0.02	0.85	<0.005	<0.01
5/07/2004	<0.2	2.43	-	-	-	-
4/10/2004	<0.2	2.15	-	-	-	-
17/01/2005	<0.2	2.1	-	-	-	-
4/04/2005	<0.2	2.50	0.02	0.94	<0.005	<0.01
10/10/2005	<0.2	2.2	-	-	-	-
5/04/2006	<0.2	2.0	<0.004	0.01	<0.005	<0.01
19/10/2006	<0.2	2.11	-	-	-	-
4/04/2007	<0.2	1.8	0.0	1.0	<0.005	<0.01

Table A4.10D: Monitoring Data for Bore 404B

DATE	Ni (soluble) (g m^{-3})	Pb (soluble) (g m^{-3})	Cr (soluble) (g m^{-3})	Cd (soluble) (g m^{-3})	Hg (soluble) (g m^{-3})	Se (soluble) (g m^{-3})	SVOC (g m^{-3})
24/08/1999	-	-	-	-	-	-	-
21/09/1999	-	-	-	-	-	-	-
19/10/1999	<0.005	0.005	<0.005	<0.0005	<0.00008	<0.02	*
16/11/1999	-	-	-	-	-	-	-
14/12/1999	-	-	-	-	-	-	-
25/01/2000	-	-	-	-	-	-	-
24/02/2000	<0.005	0.004	<0.005	<0.0005	<0.00008	<0.01	*
14/03/2000	-	-	-	-	-	-	-
4/04/2000	-	-	-	-	-	-	-
2/05/2000	-	-	-	-	-	-	-
1/06/2000	-	-	-	-	-	-	-
13/07/2000	-	-	-	-	-	-	-
10/08/2000	-	-	-	-	-	-	-
5/09/2000	-	-	-	-	-	-	-
3/10/2000	0.006	<0.001	<0.005	<0.0005	<0.00008	<0.001	*
7/11/2000	-	-	-	-	-	-	-
4/12/2000	-	-	-	-	-	-	-
23/01/2001	-	-	-	-	-	-	-
7/02/2001	-	-	-	-	-	-	-
7/03/2001	-	-	-	-	-	-	-
5/04/2001	<0.005	<0.001	<0.005	<0.0005	<0.00008	<0.01	*
3/05/2001	-	-	-	-	-	-	-
6/06/2001	-	-	-	-	-	-	-
5/07/2001	-	-	-	-	-	-	-
1/10/2001	<0.005	<0.001	<0.005	<0.0005	<0.00008		
8/04/2002	<0.005	<0.001	<0.005	<0.0005	<0.00008	<0.01	*
8/07/2002	-	-	-	-	-	-	-
20/01/2003	-	-	-	-	-	-	-
7/10/2003	-	-	-	-	-	-	-
1/04/2004	<0.005	<0.001	<0.005	<0.0005	<0.00008	<0.01	*
5/07/2004	-	-	-	-	-	-	-
4/10/2004	-	-	-	-	-	-	-
17/01/2005	-	-	-	-	-	-	-
4/04/2005	<0.005	<0.001	<0.005	<0.0005	<0.00008	<0.01	
10/10/2005	-	-	-	-	-	-	-
5/04/2006	<0.0005	<0.001	<0.005	<0.00005	<0.00008	<0.001	-
19/10/2006	-	-	-	-	-	-	-
4/04/2007	<0.005	<0.001	<0.005	<0.0005	<0.00008	<0.01	-

* = All paramaters below the detection limit of the tests

Table A4.11: Monitoring Data for Bore 405A

DATE	**Temperature** (°C)	**pH**	**Conductivity** (μS cm^{-1})	**BOD** (g m^{-3})	**COD** (g m^{-3})	**N-NH$_3$** (g m^{-3})	**NO$_3^-$** (g m^{-3})	**NO$_2^-$** (g m^{-3})	**Na (soluble)** (g m^{-3})	**K (soluble)** (g m^{-3})	**Ca (soluble)** (g m^{-3})	**Mg (soluble)** (g m^{-3})	**Cl$^-$** (g m^{-3})	**SO$_4^{2-}$** (g m^{-3})	**Alkalinity** (g m^{-3})
24-Feb-00	15.8	7.7	3260	<0.5	10	0.7	0.002	<0.002	378	45.2	120	94.6	-	92	420
05-Apr-01	15.8	7.6	3520	0.5	19	0.73	0.002	<0.002	418	55.1	110	95.5	880	92.4	400
08-Apr-02	15.8	7.7	3550	0.5	<5	0.7	<0.05	-	482	53.4	102	92.7	900	89.2	420
01-Apr-04	15.9	7.4	7310	<0.5	80	1.32	<0.002	<0.002	801	80.6	283	242	2300	261	380
04-Apr-05	15.8	7.4	7370	<0.5	35	1.28	<0.002	<0.002	750	85.5	309	278	2000	232	370
5/04/2006	16.1	7.5	5280	<0.5	10	1.06	0.003	<0.002	638	64.7	189	178	1500	160	340
4/04/2007	15.6	7.5	4870	<0.5	30	1.23	<0.002	<0.002	472	59.1	193	167	1410	186	350
8/10/2007	14.8	7.6	3230	0.8	45	0.94	0.007	-	385	47.0	115	93.7	863	129	370
03-Apr-08	15.7	7.8	2530	2.1	75	1.4	0.0066	<0.002	550	57.0	160	120	1200	180	410

DATE	**Fe (soluble)** (g m^{-3})	**B (soluble)** (g m^{-3})	**DRP** (g m^{-3})	**Mn (soluble)** (g m^{-3})	**Cu (soluble)** (g m^{-3})	**Zn (soluble)** (g m^{-3})	**Ni (soluble)** (g m^{-3})	**Pb (soluble)** (g m^{-3})	**Cr (soluble)** (g m^{-3})	**Cd (soluble)** (g m^{-3})	**Hg (soluble)** (g m^{-3})	**Se (soluble)** (g m^{-3})	**SVOC** (g m^{-3})
24-Feb-00	0.71	0.238	0.031	0.028	<0.001	0.004	<0.001	0.0009	<0.001	<0.0001	<0.00008	<0.002	-
05-Apr-01	0.57	0.27	0.026	0.218	<0.001	0.002	<0.001	<0.0002	<0.001	<0.0001	<0.00008	<0.002	*
08-Apr-02	<0.04	0.31	0.035	0.198	<0.001	<0.002	<0.01	<0.0002	<0.001	<0.0001	<0.00008	<0.002	*
01-Apr-04	<0.01	0.36	0.009	0.611	<0.003	<0.005	0.003	<0.0005	<0.003	<0.003	<0.00008	<0.005	*
04-Apr-05	<0.1	0.37	0.009	0.668	<0.003	<0.005	<0.003	<0.0005	<0.003	<0.0003	<0.00008	<0.005	*
5/04/2006	<0.04	0.29	0.006	0.387	<0.001	<0.002	<0.001	<0.0002	<0.001	<0.0001	<0.00008	<0.002	*
4/04/2007	<0.1	0.26	0.006	0.394	<0.001	<0.002	0.003	<0.0002	<0.003	<0.0001	<0.00008	<0.005	*
8/10/2007	-	0.29	-	-	-	-	-	-	-	-	-	-	-
03-Apr-08	<0.02	0.36	0.040	0.300	<0.001	<0.005	<0.0010	<0.0002	<0.003	<0.00010	<0.00008	<0.0020	*

* = All paramaters below the detection limit of the tests

Table A4.12: Monitoring Data for Bore 405B

DATE	**Temperature** (°C)	**pH**	**Conductivity** (µS cm^{-1})	**N-NH$_3$** (g/m^3)	**NO$_3^-$** (g/m^3)	**NO$_2^-$** (g/m^3)	**Na (soluble)** (g/m^3)	**K (soluble)** (g/m^3)	**Ca (soluble)** (g/m^3)	**Mg (soluble)** (g/m^3)	**Cl$^-$** (g/m^3)	**SO$_4^{2-}$** (g/m^3)	**Alkalinity** (g/m^3)	**Fe (soluble)** (g/m^3)	**B (soluble)** (g/m^3)
24/02/00	15.4	7.3	39300	1.80	<0.002	<0.002	7100	211	312	954	13100	1900	310	-	2.82
5/04/01	15.3	7.2	39900	1.73	0.01	<0.001	8030	278	349	1050	15300	2060	260	<0.4	3
8/04/02	15.4	7.2	40500	1.71	<0.05	-	8170	270	341	1070	16000	2130	260	<0.4	3.1
1/04/04	15.6	7.1	39900	1.73	0.006	0.018	8050	265	329	920	16000	2410	270	1.00	3.4
4/04/05	15.5	7.1	41100	1.60	<0.001	<0.01	8820	300	359	1060	13000	2240	260	<1.0	4
5/04/06	16.2	7.0	42900	1.68	0.02	<0.01	8480	296	392	1210	15000	2250	250	<2.0	3.5
4/04/07	15.5	7.1	44100	1.93	<0.002	0.012	8750	298	401	1190	16400	2330	230	<1.0	3.3

Table A4.13: Monitoring Data for Bore 405C

DATE	**Temperature** (°C)	**pH**	**Conductivity** (µS cm^{-1})	**N-NH$_3$** (g/m^3)	**NO$_3^-$** (g/m^3)	**NO$_2^-$** (g/m^3)	**Na (soluble)** (g/m^3)	**K (soluble)** (g/m^3)	**Ca (soluble)** (g/m^3)	**Mg (soluble)** (g/m^3)	**Cl$^-$** (g/m^3)	**SO$_4^{2-}$** (g/m^3)	**Alkalinity** (g/m^3)	**Fe (soluble)** (g/m^3)	**B (soluble)** (g/m^3)
24/02/00	17.0	7.3	45000	2.7	0.026	0.028	8350	237	352	1000	12400	2140	310	-	3.2
5/04/01	15.3	7.1	45200	2.29	0.125	<0.002	9600	312	378	1120	17300	2310	260	<0.4	3.5
8/04/02	15.4	7.2	45500	2.18	<0.05	-	9230	300	353	1100	18000	2360	250	<1.0	3.7
1/04/04	15.8	7.2	45100	2.29	<0.01	<0.01	9290	304	361	1010	18000	2840	240	1.4	3.8
4/04/05	15.7	7.2	45200	2.12	0.022	<0.002	10300	345	396	1180	15000	2630	230	<1.0	4.5
5/04/06	15.9	7.1	45400	1.89	0.11	<0.01	9400	327	397	1230	17000	2490	230	<2.0	3.7
4/04/07	15.6	7.1	45300	2.35	0.008	0.014	8930	319	370	1160	17000	2410	220	<2.0	3.4

Table A4.14A: Monitoring Data for Bore 406A

DATE	**Temperature** (°C)	**pH**	**Conductivity** (μS cm^{-1})	**BOD** (g m^{-3})	**COD** (g m^{-3})	**N-NH$_3$** (g m^{-3})	**NO$_3^-$** (g m^{-3})	**NO$_2^-$** (g m^{-3})	**Na (soluble)** (g m^{-3})	**K (soluble)** (g m^{-3})	**Ca (soluble)** (g m^{-3})	**Mg (soluble)** (g m^{-3})	**Cl$^-$** (g m^{-3})	**SO$_4^{2-}$** [g/m^3]	**Alkalinity** [g/m^3]
6/06/2001	15.3	7.4	11500	-	-	0.23	0.45	-	2340	98	118	126	3400	534	940
5/07/2001	14.2	7.6	11400	-	-	0.62	0.53	-	2450	97	112	139	3300	571	920
1/10/2001	15.2	7.5	11400	<5	140	0.15	0.21	-	2290	88	112	119	3300	569	900
8/04/2002	18.6	7.4	11700	1.4	200	0.55	<0.05	-	2390	89	105	146	3800	421	930
8/07/2002	15.6	7.6	11800	-	-	0.2	0.22	-	2380	92	112	140	3600	521	940
20/01/2003	17.9	7.4	11800	-	-	0.64	0.01	-	2360	91	112	136	3400	612	970
7/10/2003	16	7.5	11400	-	-	0.59	0.03	-	2430	89	107	145	3500	578	960
1/04/2004	19	7.6	11200	2.3	<200	0.28	0.33	0.004	2210	85	101	134	1700	585	480
5/07/2004	15.1	7.6	11200	-	-	0.18	0.15	-	2230	104	122	155	3100	503	780
4/10/2004	14	7.6	11100	-	-	0.06	0.11	-	2420	96	103	146	2600	574	940
17/01/2005	17.2	7.6	11100	-	-	0.4	0.06	-	2320	97	121	155	3000	538	930
4/04/2005	18.0	7.6	11000	1.8	55	0.4	0.17	0.011	2110	91	98	141	2900	515	930
10/10/2005	14.8	7.7	10900	-	-	<0.1	0.10	-	2210	197	103	137	2800	555	900
5/04/2006	18.5	7.6	10900	1.2	120	0.3	0.20	0.007	2440	84	102	144	3100	534	950
19/10/2006	13.1	7.6	10800	-	-	0.0	0.13	-	1960	79	100	132	2890	517	900
4/04/2007	16.2	7.4	10800	1.4	120	0.5	0.10	0.008	2340	78	94	134	3150	495	910

Table A4.14B: Monitoring Data for Bore 406A

DATE	**Temperature** (°C)	**pH**	**Conductivity** (μS cm^{-1})	**BOD** (g m^{-3})	**COD** (g m^{-3})	**N-NH$_3$** (g m^{-3})	**NO$_3^-$** (g m^{-3})	**NO$_2^-$** (g m^{-3})	**Na (soluble)** (g m^{-3})	**K (soluble)** (g m^{-3})	**Ca (soluble)** (g m^{-3})	**Mg (soluble)** (g m^{-3})	**Cl$^-$** (g m^{-3})	**SO$_4^{2-}$** (g m^{-3})	**Alkalinity** (g m^{-3})
6/06/2001	15.2	7.2	22500	-	-	2.90	0.219	-	4460	173	248	425	7600	946	860
5/07/2001	14.8	7.3	22600	-	-	1.89	1.110	-	4390	161	226	425	7600	918	860
1/10/2001	15.4	7.3	22600	<5	180	0.64	0.880	-	4530	154	230	400	7700	1000	830
8/04/2002	18.6	7.2	22500	2.7	140	2.04	0.330	-	4290	154	205	473	7700	1060	830
22/07/2002	16.1	7.4	22300	-	-	0.40	2.140	-	4340	152	202	414	7500	875	820
20/01/2003	17.9	7.3	22400	-	-	1.68	0.039	-	4510	152	198	408	7400	1120	870
7/10/2003	15.1	7.2	23200	-	-	3.13	0.230	-	5080	159	216	485	6600	939	900
1/04/2004	19.3	7.5	22400	4.4	<400	1.33	0.660	0.011	4350	144	186	407	8200	973	860
5/07/2004	14.9	7.1	23700	-	-	1.70	0.890	-	4550	180	233	510	7200	803	1000
4/10/2004	15.5	7.3	23200	-	-	0.06	1.220	-	4570	175	206	488	5700	935	990
17/01/2005	18.4	7.2	22900	-	-	2.84	0.010	-	4730	163	227	487	6200	843	960
4/04/2005	19.0	7.1	22800	5	120	2.89	0.016	0.011	4200	165	205	473	5400	991	1000
10/10/2005	15.5	7.3	22900	-	-	0.40	0.808	-	4810	177	218	497	5900	857	950
5/04/2006	19.3	7.1	22700	6.3	85	1.95	0.092	0.027	4720	148	201	476	6300	884	1000
19/10/2006	13.1	7.3	22200	-	-	0.09	1.110	-	3780	150	201	478	7130	800	970
4/04/2007	19.3	7.2	22200	1.9	100	1.44	0.358	0.034	4800	144	185	420	7660	835	990

Table A4.15A: Monitoring Data for Bore 407A

DATE	**Temperature** (°C)	**pH**	**Conductivity** (μS cm^{-1})	**BOD** (g m^{-3})	**COD** (g m^{-3})	**N-NH$_3$** (g m^{-3})	**NO$_3^-$** (g m^{-3})	**NO$_2^-$** (g m^{-3})	**Na (soluble)** (g m^{-3})	**K (soluble)** (g m^{-3})	**Ca (soluble)** (g m^{-3})	**Mg (soluble)** (g m^{-3})	**Cl$^-$** (g m^{-3})	**SO$_4^{2-}$** (g m^{-3})	**Alkalinity** (g m^{-3})
6/06/2001	15.2	7.2	22500	-	-	2.90	0.219	-	4460	173	248	425	7600	946	860
5/07/2001	14.8	7.3	22600	-	-	1.89	1.110	-	4390	161	226	425	7600	918	860
1/10/2001	15.4	7.3	22600	<5	180	0.64	0.880	-	4530	154	230	400	7700	1000	830
8/04/2002	18.6	7.2	22500	2.7	140	2.04	0.330	-	4290	154	205	473	7700	1060	830
22/07/2002	16.1	7.4	22300	-	-	0.40	2.140	-	4340	152	202	414	7500	875	820
20/01/2003	17.9	7.3	22400	-	-	1.68	0.039	-	4510	152	198	408	7400	1120	870
7/10/2003	15.1	7.2	23200	-	-	3.13	0.230	-	5080	159	216	485	6600	939	900
1/04/2004	19.3	7.5	22400	4.4	<400	1.33	0.660	0.011	4350	144	186	407	8200	973	860
5/07/2004	14.9	7.1	23700	-	-	1.70	0.890	-	4550	180	233	510	7200	803	1000
4/10/2004	15.5	7.3	23200	-	-	0.06	1.220	-	4570	175	206	488	5700	935	990
17/01/2005	18.4	7.2	22900	-	-	2.84	0.010	-	4730	163	227	487	6200	843	960
4/04/2005	19.0	7.1	22800	5	120	2.89	0.016	0.011	4200	165	205	473	5400	991	1000
10/10/2005	15.5	7.3	22900	-	-	0.40	0.808	-	4810	177	218	497	5900	857	950
5/04/2006	19.3	7.1	22700	6.3	85	1.95	0.092	0.027	4720	148	201	476	6300	884	1000
19/10/2006	13.1	7.3	22200	-	-	0.09	1.110	-	3780	150	201	478	7130	800	970
4/04/2007	19.3	7.2	22200	1.9	100	1.44	0.358	0.034	4800	144	185	420	7660	835	990

Table A4.15B: Monitoring Data for Bore 407A

DATE	**Fe (soluble)** (g m^{-3})	**B (soluble)** (g m^{-3})	**DRP** (g m^{-3})	**Mn (soluble)** (g m^{-3})	**Cu (soluble)** (g m^{-3})	**Zn (soluble)** (g m^{-3})	**Ni (soluble)** (g m^{-3})	**Pb (soluble)** (g m^{-3})	**Cr (soluble)** (g m^{-3})	**Cd (soluble)** (g m^{-3})	**Hg (soluble)** (g m^{-3})	**Se (soluble)** (g m^{-3})	**SVOC** (g m^{-3})
6/06/2001	<0.2	2.35	-	-	-	-	-	-	-	-	-	-	-
5/07/2001	-	2.31	-	-	-	-	-	-	-	-	-	-	-
1/10/2001	<0.2	2.26	<0.004	1.29	<0.005	<0.02	<0.005	<0.003	<0.005	<0.0005	<0.00008	<0.01	*
8/04/2002	<0.2	2.11	<0.02	1.03	<0.005	<0.01	<0.005	<0.001	<0.005	<0.0005	<0.00008	<0.01	*
22/07/2002	<0.4	2.1	-	-	-	-	-	-	-	-	-	-	-
20/01/2003	<0.2	2.01	-	-	-	-	-	-	-	-	-	-	-
7/10/2003	<0.2	2.26	-	-	-	-	-	-	-	-	-	-	-
1/04/2004	<0.2	2.25	<0.02	1.23	-	<0.01	<0.005	<0.001	<0.005	<0.0005	<0.00008	<0.01	*
5/07/2004	<0.2	2.34	-	-	-	-	-	-	-	-	-	-	-
4/10/2004	<0.2	2.21	-	-	-	-	-	-	-	-	-	-	-
17/01/2005	<0.2	2.15	-	-	-	-	-	-	-	-	-	-	-
4/04/2005	<0.2	2.60	0.01	1.35	<0.005	<0.01	<0.005	<0.001	<0.005	<0.0005	<0.00008	<0.01	*
10/10/2005	<0.2	2.3	-	-	-	-	-	-	-	-	-	-	-
5/04/2006	<0.2	2.15	0.01	1.3	<0.005	<0.01	0.008	<0.001	<0.005	<0.0005	<0.00008	<0.01	*
19/10/2006	<0.2	2.17	-	-	-	-	-	-	-	-	-	-	-
4/04/2007	<0.2	1.9	0.03	1.1	<0.005	-	0.007	<0.001	<0.005	<0.0005	<0.00008	<0.01	*

* = All paramaters below the detection limit of the tests

Appendix 5

Table A5.1A: Data for Background Bore 1 (BG1)

DATE	**pH**	**Conductivity** (μS cm^{-1})	**BOD** (g m^{-3})	**COD** (g m^{-3})	**N-NH$_3$** (g m^{-3})	**NO$_3^-$** (g m^{-3})	**NO$_2^-$** (g m^{-3})
13/03/1991	7.7	1162	-	-	0.90	-	-
20/03/1991	7.8	1095	-	-	0.70	0.24	-
4/09/1991	7.5	908	2	-	0.30	-	-
19/02/1992	7.6	1076	2.9	-	0.30	-	-
19/02/1992	7.6	1076	2.9	-	1.10	-	-
13/05/1992	7.6	1033	3.1	-	1.10	-	-
13/05/1992	7.6	1033	3.1	-	0.50	-	-
21/10/1992	7.6	801	2.2	-	0.50	-	-
21/10/1992	7.6	801	2.2	-	0.10	-	-
20/01/1993	7.6	961	0.7	-	0.60	-	-
15/04/1993	7.7	750	1.2	-	0.40	-	-
7/07/1993	7.7	834	1.4	-	0.32	-	-
13/10/1993	7.7	810	0.4	-	0.50	-	-
22/12/1993	7.7	880	0.9	-	0.40	-	-
19/01/1994	7.7	970	3.8	20	0.40	0.33	-
22/06/1994	7.8	640	1.2	6	1.30	0.05	-
15/03/1995	7.8	785	6.6	37	0.30	-	-
30/08/1995	7.6	700	3.3	<2	0.60	<0.02	-
17/01/1996	7.7	995	3.5	9	0.50	0.54	-
22/05/1996	7.7	770	0.1	13	0.07	<0.02	-
25/09/1996	7.7	815	0.9	7	0.20	0.13	-
5/02/1997	7.9	685	1.2	<2	0.09	<0.02	-
11/06/1997	7.8	645	1.1	20	<0.05	<0.02	-
15/10/1997	7.7	740	1	7	0.07	0.35	0.026
17/02/1998	7.2	700	0.2	<2	0.05	0.27	0.011
17/06/1998	7.6	655	0.6	10	0.05	0.007	<0.002
13/10/1998	7.7	670	0.5	12	<0.05	0.185	<0.002
17/02/1999	7.9	575	1	4	0.09	<0.002	<0.002
1/09/1999	7.8	570	0.9	7	0.07	<0.002	<0.002
6/01/2000	7.8	505	1.1	5	0.08	0.003	<0.002
9/05/2000	7.8	495	3.7	7	0.05	0.007	<0.002
12/09/2000	7.7	500	3.9	7	0.06	<0.002	-
25/01/2001	7.8	538	1.1	<5	0.03	<0.002	-
22/05/2001	7.6	560	0.8	<5	0.03	0.003	-
2/10/2001	7.6	584	0.5	<5	0.01	<0.002	-
4/02/2002	7.6	541	0.7	<5	0.03	0.003	-
11/06/2002	7.7	600	<0.5	<5	0.02	0.09	-
15/10/2002	7.6	506	1	<5	0.04	<0.002	-
18/02/2003	7.8	588	0.5	10	0.05	0.022	-
24/06/2003	7.9	563	0.5	<5	0.04	<0.002	<0.002
28/10/2003	7.8	639	<0.5	5	<0.01	0.057	0.004
10/03/2004	7.9	491	<0.5	<5	0.03	<0.002	<0.002
6/07/2004	7.8	551	0.5	20	0.05	0.003	<0.002
9/11/2004	7.9	609	<0.5	20	0.05	0.003	<0.002
15/03/2005	7.9	474	<0.5	<5	0.04	<0.002	<0.002
19/07/2005	7.9	485	<0.5	<5	0.04	<0.002	<0.002
23/11/2005	7.9	581	<0.5	20	0.04	<0.002	<0.002
1/08/2006	7.8	530	<0.5	5	0.05	<0.002	<0.002
4/12/2006	7.9	639	<0.5	15	0.04	<0.002	<0.002
11/04/2007	7.9	502	<0.5	15	0.05	<0.002	<0.002
14/08/2007	7.9	540	<0.5	10	0.03	<0.002	<0.002
19/12/2007	7.9	480	<0.5	<5	0.03	<0.002	<0.002
21/04/2008	8.0	456	<0.5	10	<0.010	0.0028	<0.002
26/08/2008	7.8	463	<0.50	5	0.01	<0.0020	<0.0020
8/01/2009	7.9	544	0.5	25	0.09	<0.0020	<0.0020
12/05/2009	7.9	488	0.6	<5	0.25	<0.002	<0.002

Table A5.1B: Data for Background Bore 1 (BG1)

DATE	Na (soluble) (g m^{-3})	K (soluble) (g m^{-3})	Cl$^-$ (g m^{-3})	SO$_4^{2-}$ (g m^{-3})	Alkalinity (g m^{-3})	Fe (soluble) (g m^{-3})	DRP (g m^{-3})	Mn (soluble) (g m^{-3})
13/03/1991	125	13	124	-	410	1	-	0.00
20/03/1991	93	12	132	-	270	0.5	-	0.10
4/09/1991	143	12.8	95	19	310	0.4	0.302	0.00
19/02/1992	113	11.2	118	30	360	0.4	-	<0.025
19/02/1992	113	11.2	118	30	360	0.4	0.128	<0.1
13/05/1992	84	10.1	58	25	380	1.6	-	0.10
13/05/1992	84	10.1	58	25	380	1.6	0.25	0.10
21/10/1992	51	10	75	19	285	1.8	-	0.10
21/10/1992	51	10	75	19	285	1.8	0.122	0.10
20/01/1993	73	10	105	25	330	0.2	0.14	<0.1
15/04/1993	55	8.8	62	20	250	1.1	0.119	<0.1
7/07/1993	-	-	80	40	580	0.4	0.093	0.10
13/10/1993	-	-	77	28	265	0.5	0.069	-
22/12/1993	64	10.2	88	27	285	0.1	0.099	<0.1
19/01/1994	84	11	95	23	343	0.2	0.146	<0.1
22/06/1994	43	7.9	41	17	245	0.5	0.159	<0.1
15/03/1995	70.3	11.8	81	3	286	1.8	0.192	0.05
30/08/1995	31.7	7	31	5	240	0.4	0.151	0.02
17/01/1996	93	12.4	110	10	315		0.21	<0.05
22/05/1996	29.3	6.7	89	17	785	0.3	0.059	0.08
25/09/1996	39.2	7.7	95	23	255	0.3	0.159	<0.05
5/02/1997	29	5.9	53	15	235	0.2	0.127	0.04
11/06/1997	23.9	5.5	46	20	220	0.2	0.077	<0.05
15/10/1997	38.2	7	76	29	270	0.2	0.199	<0.05
17/02/1998	33.9	7.1	63	31	249	0.1	0.19	0.04
17/06/1998	29	6.4	30	21	264	0.1	0.068	0.04
13/10/1998	33.3	6.8	43	32	240	0.14	0.101	0.03
17/02/1999	29.6	5.9	47	11	219	<0.05	0.22	0.03
1/09/1999	26.4	5.9	41	4	240	0.2	0.137	0.03
6/01/2000	23.5	5.3	38	3	210	0.2	0.226	0.03
9/05/2000	23.2	5.4	38	<0.5	220	0.25	0.243	0.03
12/09/2000	22.2	5.2	87	<0.5	160	0.2	-	0.03
25/01/2001	22.1	5.3	52	2.1	200	0.36	-	-
22/05/2001	28.6	5.76	49	6.9	220	0.17	-	0.05
2/10/2001	26.2	5.56	54	10.4	240	0.2	-	0.05
4/02/2002	26.1	5.44	39	7.5	230	0.2	-	0.05
11/06/2002	24.8	5.54	36	<0.5	220	0.27	-	0.06
15/10/2002	26.8	5.55	34	<0.5	220	0.21	0.224	0.05
18/02/2003	29.2	5.9	54	7.3	240	0.16	0.248	0.04
24/06/2003	27.6	5.87	50	0.9	230	0.21	0.29	0.03
28/10/2003	31.7	5.88	67	8.4	240	0.17	0.276	0.03
10/03/2004	22.6	5.21	32	<0.5	230	0.21	0.277	0.03
6/07/2004	25.6	5.61	42	2.3	240	0.25	0.279	0.03
9/11/2004	23.1	5.66	45	7.9	260	0.27	0.263	0.03
15/03/2005	21.7	4.86	23	0.8	230	0.41	0.286	0.03
19/07/2005	21.6	5.11	25	<0.5	240	0.27	0.273	0.03
23/11/2005	23.7	5.8	39	7.3	260	0.35	0.24	0.04
1/08/2006	25.7	5.75	35.8	6.7	230	0.23	0.27	0.03
4/12/2006	24.4	5.56	52.3	27.9	230	0.29	0.23	0.04
11/04/2007	19.8	4.84	21.9	1.4	250	0.27	0.254	0.03
14/08/2007	19.9	4.88	26.9	5.9	260	0.24	0.267	0.03
19/12/2007	23	5.3	22	<0.5	240	0.23	0.25	0.03
21/04/2008	22	5	23	<0.5	230	0.43	0.25	0.03
26/08/2008	17	4.4	24	<0.50	230	0.22	0.27	0.03
8/01/2009	26	5.5	43	1.6	230	0.45	0.25	0.04
12/05/2009	29	8.5	28	<0.5	230	0.29	0.29	0.04

Table A5.2A: Data for Background Bore 2 (BG2)

DATE	**pH**	**Conductivity** (μS cm^{-1})	**BOD** (g m^{-3})	**COD** (g m^{-3})	**N-NH$_3$** (g m^{-3})	**NO_3^-** (g m^{-3})	**NO_2^-** (g m^{-3})
21/07/1983	7.3	968	-	-	0.3	0.02	-
26/06/1991	7.1	910	-	-	0.3	-	-
4/09/1991	7.0	859	1.5	-	0.3	-	-
27/11/1991	7.1	933	0.5	-	0.3	-	-
27/11/1991	7.1	933	0.5	-	0.3	-	-
19/02/1992	7.1	1043	2	-	0.3	-	-
19/02/1992	7.1	1043	2	-	0.3	-	-
13/05/1992	7.1	942	2.9	-	0.3	-	-
13/05/1992	7.1	942	2.9	-	0.3	-	-
19/08/1992	7.1	920	0.6	-	0.3	-	-
19/08/1992	7.1	920	0.6	-	0.4	-	-
21/10/1992	7.2	886	1.9	-	0.4	-	-
21/10/1992	7.2	886	1.9	-	0.3	-	-
20/01/1993	7.2	882	0.7	-	0.3	-	-
15/04/1993	7.1	883	1.6	-	0.3	-	-
15/04/1993	7.1	883	1.6	-	0.3	-	-
7/07/1993	7.2	867	1.8	-	0.27	-	-
13/10/1993	7.2	880	0.5	-	0.3	-	-
22/12/1993	7.3	910	0.3	-	0.2	-	-
19/01/1994	7.1	935	1.7	20	0.3	<0.02	-
22/06/1994	7.3	895	0.8	8	0.5	0.04	-
15/03/1995	7.3	910	1.2	39	0.3	-	-
19/07/1995	7.2	810	1.2	9	0.3	<0.02	-
22/11/1995	7.2	905	0.7	9	0.6	<0.05	-
10/04/1996	7.1	900	1	-	0.3	<0.02	-
14/08/1996	7.4	855	1.4	8	0.3	<0.02	-
11/12/1996	7.2	915	2	6	0.3	<0.02	-
30/04/1997	7.2	885	1.6	5	0.2	<0.02	-
3/09/1997	7.1	900	0.8	10	0.4	0.05	-
7/01/1998	7.0	930	2.2	37	0.5	-	-
5/05/1998	6.8	935	2.2	3	0.4	0.004	<0.002
2/09/1998	7.2	875	1.9	3	-	<0.002	<0.002
6/01/1999	7.1	965	1.8	8	0.3	<0.002	<0.002
12/05/1999	7.1	935	1.9	6	0.2	<0.002	<0.002
21/07/1999	7.1	925	0.8	9	0.2	0.003	<0.002
25/11/1999	7.1	945	1.7	13	0.3	<0.002	<0.002
29/03/2000	7.1	940	1.1	20	0.2	<0.002	<0.002
2/08/2000	7.1	900	2	<5	0.3	<0.002	-
18/12/2000	7.1	892	1.4	<5	0.23	<0.002	-
12/04/2001	7.0	888	0.9	<5	0.21	<0.002	-
22/08/2001	7.0	865	0.9	10	0.22	0.002	-
27/12/2001	7.0	884	0.9	5	0.24	<0.002	-
30/04/2002	7.2	872	1.2	10	0.34	<0.002	-
3/09/2002	7.1	846	1.6	5	0.25	<0.002	-
7/01/2003	7.1	900	2	<5	0.24	<0.002	-
13/05/2003	7.1	880	1.6	<5	0.22	0.002	<0.002
16/09/2003	7.1	876	0.9	65	0.24	<0.002	<0.002
20/01/2004	7.0	916	0.9	<5	0.25	<0.002	<0.002
25/05/2004	7.1	896	<0.5	25	0.24	<0.002	<0.002
28/09/2004	7.1	863	<0.5	<5	0.25	<0.002	<0.002
1/02/2005	7.1	911	<0.5	<5	0.28	<0.002	<0.002
7/06/2005	7.0	861	<0.5	15	0.23	<0.002	<0.002
11/10/2005	7.1	851	0.5	<5	0.22	0.002	<0.002
14/02/2006	7.1	876	<0.5	<5	0.24	<0.002	<0.002
20/06/2006	7.1	830	0.8	10	0.26	<0.002	<0.002
24/10/2006	7.1	799	<0.5	20	0.25	<0.002	<0.002
27/02/2007	7.0	911	<0.5	20	0.23	<0.002	<0.002
3/07/2007	7.0	861	0.7	<5	0.25	<0.002	<0.002
6/11/2007	7.1	850	<0.5	<5	0.28	0.0023	<0.002
11/03/2008	7.1	904	1.2	15	0.22	<0.002	<0.002
15/07/2008	7.1	837	0.6	<5	0.25	<0.002	<0.002
18/11/2008	7.1	830	<0.5	<5	0.23	0.0041	<0.0020
30/03/2009	7.0	925	<0.5	120	-	0	<0.0020

Table A5.2B: Data for Background Bore 2 (BG2)

DATE	**Na (soluble)** (g m^{-3})	**K (soluble)** (g m^{-3})	**Cl^-** (g m^{-3})	**SO_4^{2-}** (g m^{-3})	**Alkalinity** (g m^{-3})	**Fe (soluble)** (g m^{-3})	**DRP** (g m^{-3})	**Mn (soluble)** (g m^{-3})
21/07/1983	-	-	67	12	410	5.56	0.02	-
26/06/1991	31	5.4	52	-	400	14	-	1.1
4/09/1991	45	6.8	46	14	380	12.8	0.036	1.1
27/11/1991	34	6.5	54	20	400	12.2	-	1.3
27/11/1991	34	6.5	54	20	400	12.2	0.005	1.3
19/02/1992	49	7.1	73	17	435	12.7	-	1.5
19/02/1992	49	7.1	73	17	435	12.7	0.008	1.5
13/05/1992	33	6.4	47	17	410	13.7	-	1.2
13/05/1992	33	6.4	47	17	410	13.7	0.005	1.2
19/08/1992	32	6.4	55	19	396	13	-	1.3
19/08/1992	32	6.4	55	19	396	13	0.005	1.3
21/10/1992	31	6.5	57	24	365	13	-	1.2
21/10/1992	31	6.5	57	24	365	13	0.005	1.2
20/01/1993	28	6.1	56	28	373	13.1	<0.005	1.5
15/04/1993	33	6.6	46	24	375	26	<0.005	1.3
15/04/1993	33	6.6	46	24	375	26	<0.005	1.3
7/07/1993	-	-	39	29	405	12.2	<0.005	1.2
13/10/1993	-	-	50	30	375	12.9	<0.005	-
22/12/1993	31	6.4	43	27	410	11.9	0.005	1.3
19/01/1994	32	6.5	53	26	403	13.6	0.009	1.4
22/06/1994	32	6.4	36	25	385	6.1	0.007	1.3
15/03/1995	30.9	6.6	50	22	405	13.6	0.01	1.34
19/07/1995	29.8	6	47	5	385	14	<0.005	0.8
22/11/1995	33.2	6.4	59	23	400	12.4	<0.004	1.3
10/04/1996	29	6	42	21	395	11.6	<0.004	1.17
14/08/1996	29.3	6.5	24	23	356	11.9	0.012	1.23
11/12/1996	29.6	6.3	52	26	400	12.7	<0.005	1.28
30/04/1997	29.4	6.3	38	24	390	14	<0.004	1.28
3/09/1997	31.8	6.3	51	33	395	12.1	0.005	1.23
7/01/1998	32.7	6.7	59	28	410	12	<0.004	1.17
5/05/1998	31.9	7	40	35	400	12.7	<0.004	1.39
2/09/1998	28.7	6	44	38	366	12.3	<0.004	1.07
6/01/1999	34.9	7.3	59	29	403	13.2	0.005	1.36
12/05/1999	32.7	7.2	24	37	380	12.5	0.009	1.29
21/07/1999	34	7.4	42	41	380	12.2	0.007	1.21
25/11/1999	33	7.1	55	33	410	12.9	0.01	1.37
29/03/2000	32.1	7.2	52	39	390	12.6	0.009	1.29
2/08/2000	34.6	6.4	20	42	380	11.1	0.009	1.11
18/12/2000	29.7	6.3	180	34	400	10.8	-	0.992
12/04/2001	34.5	6.66	38	28	82	9.07	-	0.92
22/08/2001	33.1	6.3	38	34	400	4.31	-	0.946
27/12/2001	29	5.9	42	30	410	9.31	-	0.717
30/04/2002	31.9	6.4	41	33	480	10.1	-	0.985
3/09/2002	33	6.54	24	37	400	9.95	-	0.843
7/01/2003	34.2	6.64	42	34	420	12.8	0.011	1.11
13/05/2003	34.8	6.67	44	31	400	10.3	<0.004	0.935
16/09/2003	32.4	6.61	49	38	400	10.4	0.011	0.91
20/01/2004	32.5	6.86	59	30	400	8.47	0.011	1.12
25/05/2004	37.6	6.63	45	26	400	10.5	0.006	0.942
28/09/2004	34.5	7.06	43	31	390	10.6	0.011	0.841
1/02/2005	34.2	6.85	44	32	400	14.3	0.014	1.21
7/06/2005	33.8	7.04	41	29	390	12.1	<0.004	1.1
11/10/2005	33.3	7.15	33	32	380	12.1	<0.004	0.977
14/02/2006	34.8	6.76	38	33	390	10.7	0.01	1.03
20/06/2006	32.5	6.72	35.4	35	380	11.1	<0.004	1.05
24/10/2006	33.6	6.33	30.1	24	380	10.1	<0.004	0.916
27/02/2007	31.3	6.55	-	32	380	13.7	<0.004	1.31
3/07/2007	30.3	6.48	-	36	360	20.2	<0.004	1.44
6/11/2007	24	6.5	-	31	370	13	<0.004	1.2
11/03/2008	26	6.1	-	29	400	14	<0.004	1.3
15/07/2008	29	6.3	-	35	370	12	<0.004	1.1
18/11/2008	28	5.6	19	30	360	12	<0.0040	0.98
30/03/2009	28	6.4	49	27	420	13	<0.0040	0.25

Appendix 6

Table A6.1: Soil cation analysis results for the irrigated area – surface horizon.

	pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP
October 2008						
Site 1	7.5	1.10	48.2	3.35	1.42	2.6
Site 2	7.0	0.81	48.3	1.69	0.44	0.8
Site 3	7.3	1.01	46.1	3.13	0.60	1.1
Site 4	7.2	1.13	44.4	3.**50**	0.86	1.7
Site 5	7.0	1.07	50.5	3.12	1.04	1.8
Site 6	7.0	0.92	50.2	2.30	0.64	1.1
Site 7	7.2	1.42	50.4	3.53	0.61	1.0
Site 8	7.4	0.83	49.2	2.83	0.66	1.2
Site 9	6.8	1.06	50.0	2.95	1.38	2.5
Site 10	7.0	0.66	40.3	1.67	0.23	0.5
July 2009						
Site 1	7.6	1.07	47.1	2.68	0.78	1.50
Site 2	7.6	0.71	45.6	2.05	0.56	1.14
Site 3	7.9	0.93	47.9	3.06	0.34	0.65
Site 4	7.6	1.27	46.7	3.26	0.68	1.31
Site 5	7.8	0.7	47.8	2.6	1.15	2.21
Site 6	7.8	0.84	44	2.33	0.69	1.44
Site 7	7.6	1.18	39.7	3.1	0.6	1.33
Site 8	7.9	2.3	47.7	4.03	0.68	1.24
Site 9	7.8	1.18	41.3	2.35	0.37	0.82
Site 10	7.9	1.22	48.9	2.49	0.9	1.67

Table A6.12: Soil cation analysis results for the non-irrigated areas- surface horizon.

	pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP
October 2008						
Site 10	7.0	0.74	28.7	1.59	0.57	1.9
Site 11	7.0	1.10	22.3	1.2	0.13	0.5
Site 12	7.2	1.08	48.0	2.62	1.02	1.8
July 2009						
Site 10	7.5	0.91	42.8	2.05	0.35	0.76
Site 11	7.7	0.54	29	1.43	0.27	0.87
Site 12	7.4	1.14	35.9	1.84	0.2	0.51

Table A6.3: Soil cation analysis results for the irrigated areas – subsurface horizon.

	pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP
October 2008						
Site 1	7.9	0.76	41.9	3.32	1.4	3.0
Site 2	7.9	0.70	49.3	2.06	1.22	2.3
Site 3	7.9	0.59	39.0	2.27	1.26	2.9
Site 4	8.0	0.72	41.8	2.63	1.11	2.4
Site 5	7.8	0.60	30.7	1.89	0.94	2.8
Site 6	7.1	0.66	33.0	1.69	0.73	2.0
Site 7	7.7	0.24	12.6	0.85	0.08	0.6
Site 8	7.7	0.59	39.0	2.14	1.15	2.7
Site 9	7.9	0.60	44.8	2.44	1.05	2.1
Site 10	7.7	0.68	43.7	3.29	1.41	2.9
July 2009						
Site 1	7.4	0.72	43.9	2	1.05	2.19
Site 2	6.9	0.55	46.9	2.88	1.29	2.48
Site 3	8.0	0.58	38.3	2.09	1.44	3.43
Site 4	7.9	0.63	41.9	2.39	1.13	2.46
Site 5	7.8	0.7	42	3.56	3.76	7.52
Site 6	7.9	0.69	30	2.32	0.77	2.26
Site 7	7.8	1.19	30.6	2.75	1.1	3.06
Site 8	7.8	0.72	40.5	2.46	1.31	2.91
Site 9	7.2	0.81	31.4	2.02	0.61	1.74
Site 10	7.8	0.6	39.5	2.88	1.64	0.6

Table A6.4: Soil cation analysis results for the non-irrigated areas- subsurface horizon.

	pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP
October 2008						
Site 10	7.6	0.47	22.2	1.52	0.42	1.0
Site 11	7.8	0.46	31.7	3.28	0.62	1.7
Site 12	7.4	0.55	44.4	3.62	0.50	1.7
July 2009						
Site 10	7.9	0.35	25.4	2.26	0.34	1.21
Site 11	7.9	0.47	30	1.78	0.54	1.64
Site 12	8.0	0.41	34.5	1.68	0.24	0.65

Table A6.5: Soil cation analysis results for the Gavins Block.

	pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP
Surface Horizon						
Oct 08	7.2	1.82	38.2	14.5	20.7	27.2
Jul 09	7.6	1.96	35.3	14.1	13.5	20.8
Subsurface Horizon						
Oct 08	7.9	0.92	27	1.99	1.25	20.8
Jul 09	8.4	26.5	5.44	10.6	0.83	24.7

Table A6.6: Soil cation analysis results for the Sand Dunes.

	pH	K (me/100g)	Ca (me/100g)	Mg (me/100g)	Na (me/100g)	ESP
Surface Horizon						
Oct 08	6.4	0.51	13.3	2.17	0.19	1.1
Jul 09	7.4	0.44	14.8	1.43	0.16	0.94
Subsurface Horizon						
Oct 08	6.4	0.51	13.3	2.17	0.19	1.1
Jul 09	7.9	0.31	12.2	0.78	0.09	0.94

Table A6.7: Saturated Paste Results – surface horizon

Site	CEC (me/100g)	Total Soluble Salts (g m^{-3})	EC (µS cm^{-1})	K (g m^{-3})	Ca (g m^{-3})	Mg (g m^{-3})	Na (g m^{-3})	ESP
3	46	482	0.7	30	82	12	28	0.8
7	40	2020	3.1	51	383	41	77	1
10	49	1450	2.2	50	222	22	121	2.1
11	45	1380	2.1	26	319	24	44	0.6
Gavin's Block	44	5730	8.7	64	239	143	1440	18.2
Sand Dunes	15	14	752	1.1	22	143	15	26

Table A6.8: Saturated Paste Results – subsurface horizon

Site	CEC (me/100g)	Total Soluble Salts (g m^{-3})	EC (µS cm^{-1})	K (g m^{-3})	Ca (g m^{-3})	Mg (g m^{-3})	Na (g m^{-3})	ESP
3	37	1290	2	22	169	20	213	4.1
7	30	1050	1.6	30	111	17	135	3.2
10	26	647	1	12	124	16	58	1.3
11	37	1290	2	22	169	20	213	4.1
Gavin Block's	41	6620	10	40	166	109	19000	28.1
Sand Dunes	14	191	0.3	6	39	3	10	0.4