GROUND WATER POLLUTION AT SANITARY LANDFILL SITES: GEOHYDROLOGICAL, ENVIRONMENTAL ISOTOPE AND HYDROCHEMICAL STUDIES.

MICHAEL JOHN BUTLER

JOHANNESBURG, 1998

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A dissertation submitted to the Faculty of Science,
University of the Witwatersrand, Johannesburg,
in fulfilment of the requirements of
the degree of Master of Science in Geology

JOHANNESBURG, 1998

DECLARATION

I declare that this dissertation is my own, unaided work. It is being submitted for the Degree of Master of Science in Geology in the University of the Witwatersrand, Johannesburg. It has not been submitted before for any other degree or examination in any other university.

ABSTRACT

This study determines the potential of predicting pollution to ground water by sanitary landfills. The tracing capabilities of both stable and radioactive environmental isotopes are also evaluated. Four landfills were selected, the Linbro Park and Waterval sites in Johannesburg, and the Bloemfontein northern and southern landfill sites. The sites all differ in geological environment, size, age and physiographic setting.

Well-head measurements and chemical and environmental isotope samples from both boreholes and surface water bodies were collected at each site. In most cases, the sampling was repeated after a period of time.

It was established that the ground water monitoring network at the Linbro Park landfill site was inadequate. Several of the monitoring boreholes had collapsed or had been penetrated by tree roots. Interpretation of the data from the remaining boreholes indicated no evidence of leachate pollution to ground water, even though the turnover time is rapid. An enrichment in the δD composition, possibly due to gaseous diffusion, was observed in the down-gradient boreholes. This shift is a potential tracer of any future pollution.

Observation boreholes at the Waterval landfill site had all collapsed with the exception of one up-gradient borehole. Surface water and a number of private boreholes down-gradient of the site were sampled. Surface water from the culvert beneath the landfill was found to contain artificial tritium. Tritium values slightly higher than environmental values were observed in a private borehole. Sources other than the landfill, ie. sewage, were also found to be polluting the stream.

The ground water below and within the Bloemfontein northern site is extremely polluted. The two pollution monitoring boreholes down-gradient of the site respond

rapidly to nearby surface water. Artificial tritium, and possibly carbon-14, are present in the leachate and observed in the ground water.

Ground water at the Bloemfontein southern site is severely polluted due to infiltration of run-off from the landfill into cracks in the surface clay material. Both artificial tritium and carbon-14 are present in a monitoring borehole. Artificial tritium and similar ionic ratios are seen in nearby farm boreholes, probably due to flow overland after a major rainfall event.

The main conclusions from the study were:

1. Artificial tritium was discovered in leachate from some landfill sites.
2. The artificial tritium can be exploited as a tracer of leachates from such sites into the surrounding ground and surface water.
3. There is a general enrichment of the stable isotopes down-gradient of a landfill site.
4. It is possible to establish incipient pollution using isotope data, enabling the early detection of pollution sources.
5. It is essential to combine environmental isotope data with hydrochemistry to fully understand results from a ground water monitoring system.
6. Ground water monitoring networks are generally inadequate for present and future monitoring of any pollutants being emitted from landfills.

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INTRODUCTION

The steady increase in the population and the resulting urban development and industrialisation in the major cities of South Africa generate a constant rise in the production of solid and liquid wastes, of which 95% is disposed of in sanitary landfill sites (City Council of Johannesburg, 1989). A large percentage of the urban water supplies in this country is derived from surface impoundments, with less than 12% being obtained from boreholes (Theron *et al.*, 1982; Slim, 1987). In South Africa, ground water as a resource is becoming increasingly important as the fresh water surface impoundments become overutilised. However, ground water is highly susceptible to pollution from the surface, particularly in the form of waste disposal sites. The constant threat of long periods of drought conditions, places further strain on surface water supplies. It is therefore important to protect our ground water from potential and prevailing pollution as this resource will conceivably be harnessed for domestic supply in the foreseeable future.

The primary aim of this study has been to determine the possibility of predicting the potential pollution of ground water by sanitary landfill sites. The study has placed particular emphasis on the utilisation of the environmental isotopes of carbon, oxygen and hydrogen which, together with hydrochemistry and hydrogeology, provide information on the residence times of the ground water, the recharge rates, as well as the source or mode of recharge, with reference to any potential pollutants. The tracing of polluted ground and surface water from the landfill is also an important component of this study. The conservative nature of the environmental isotopes, as well as the mobility of dissolved ions in water, makes this feasible.

Rainfall will seep through any waste on top of, or buried in the unsaturated zone, extracting dissolved and suspended solids, creating leachates (Farvolden, 1975). This leachate may also contain bacteria and viruses. Once leachates reach the saturated zone, the contaminated solution migrates, following the local ground water gradient, slowly mixing with, and dispersing and diffusing, into uncontaminated ground water beneath the site. A pollution plume may result, moving through the aquifer, contaminating the water as well as the aquifer, as compounds are adsorbed and desorbed to and from the solid matrix (Holm *et al.*, 1986).

Landfills situated in favourable hydrogeological settings should produce a minimal amount of both ground water and surface water pollution. However, land of this type may not be available within acceptable transportation distances, or the situation may not be in an area that is publicly acceptable for landfilling. It is for these reasons and others that landfills are located on terrain that often has some adverse hydrogeological features.

Four landfill sites, all with some pollution potential, were considered for this investigation. These sites are the Linbro Park landfill site and the Waterval landfill site in the Johannesburg area, and the Northern and Southern landfill sites on the outskirts of Bloemfontein (Figure 1). The sites differ in geological environment, size and age. This study was able to make use of previous geohydrological information was available, and the results obtained enhanced and confirmed previous reports on the local geology, hydrochemistry and water levels.

The Linbro Park landfill site to the north-east of Johannesburg is located on granites of the Johannesburg-Pretoria dome, and has been in operation since 1969. The Waterval landfill site, situated in the western suburbs of Johannesburg, is situated within schists of the ancient Swaziland Supergroup. The site, which has been closed since 1978, is the oldest and largest landfill in the Johannesburg area. The Bloemfontein northern

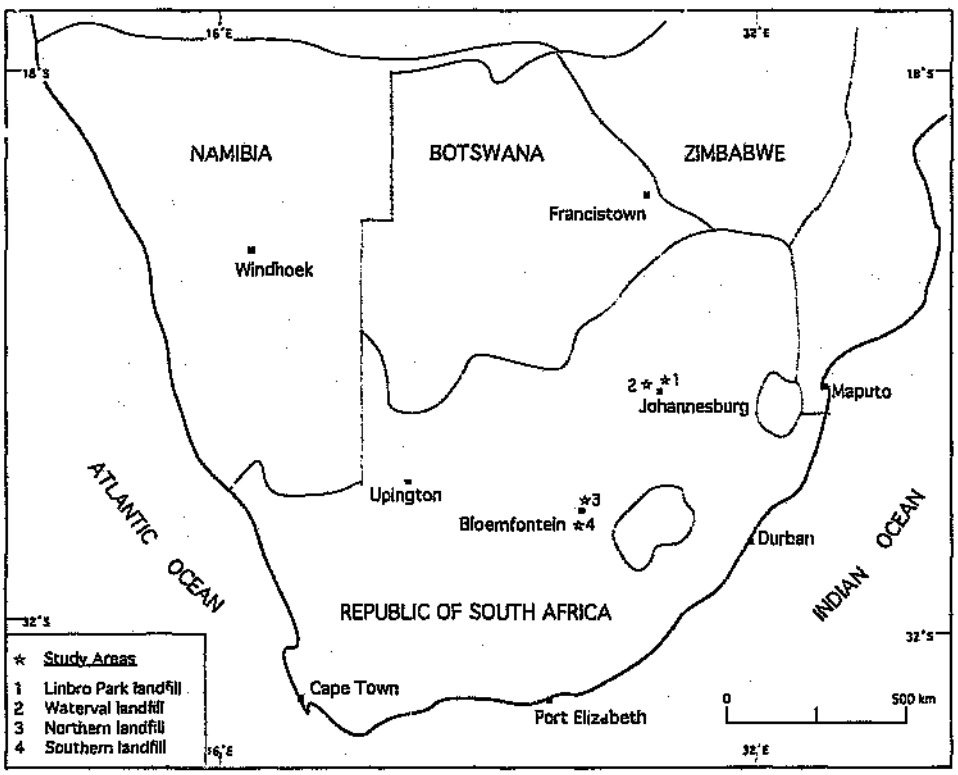

Figure 1. Location map of the four landfill sites in this study

landfill site, situated to the north of the city, is underlain by dolerite, while the southern landfill site to the south of Bloemfontein is situated in Karoo sediments of the Beaufort Group, with a thin sloping dolerite dyke below the site.

CHAPTER 1

1 WASTE MANAGEMENT

1.1 Sanitary Landfills

Although landfilling is not the most space-efficient method of solid waste disposal, it is considered economical and an environmentally viable solution to the waste disposal problem. That is, provided the correct procedures are followed during and long after its operation. The United States Environmental Protection Agency (USEPA, 1977) defines a sanitary landfill as follows:

> "An engineered method of disposing waste on land in a manner that protects the environment by spreading the waste in thin layers, compacting it to the smallest practicable volume and covering it with soil at the end of each working day."

According to the Landfill Classification System, landfills are divided into ten different classes (DWAF, 1994a). They are first divided by types of waste, namely general waste (G) or hazardous waste (H). General waste is then subdivided into four groups, namely communal landfills (C), small landfills (S), medium landfills (M) and large landfills (L). Each of these groups is further subdivided into whether significant leachate will be generated in terms of the Climatic Water Balance (B) calculation, which is equal to the rainfall in mm of water minus the evaporation from a soil surface in mm of water

(DWAF, 1994a). If B is positive for less than one year in five, no significant leachate generation will occur (B^-). If B is positive for more than one year in five, significant leachate generation will occur (B^+), and leachate management is a minimum requirement. Hazardous waste is divided into hazard ratings 3 and 4 (h) which are moderate and low hazard (DWAF, 1994b), and hazard ratings 1 to 4 (H), hazard ratings 1 and 2 being extreme and high hazards respectively.

Refuse in landfills is generally compacted and completely covered by a continuous layer of compacted soil. The primary function of this soil cover is to minimise the amount of water entering the waste, therefore decreasing the amount of leachate generated, as well as any threat of pollution to the ground water (Blight, 1992). Other functions of soil covers are to control the emission of gases, to stop refuse being washed or blown away from the site, to provide a growing medium for a vegetation cover, and to reduce smells and flies.

Options of solid waste disposal, other than in landfills, include incineration which efficiently reduces the refuse to one tenth of its initial volume, but is expensive, noxious gases are emitted and the final residue of this process has to be disposed of in some other manner; recycling, although labour intensive, holds great potential for the future and is currently being practised at Robinson Deep; and shredding or pulverising, which although reducing volumes, still requires disposal by some other process.

1.2 Leachates

"The production of leachate and its migration into usable water supplies is one of the significant environmental problems associated with solid waste disposal by sanitary landfilling methods" (Pavoni *et al.*, 1975). Leachate is produced when the field capacity of the waste is exceeded, either by extensive infiltration of rain water, or by ground water encroachment into the landfill. It results from complex decomposition processes

which involve simultaneous physical, chemical and biological interactions (Hojem, 1988). The volume of leachate produced is largely dependent on the degree of compaction and the nature of the waste, as well as the local climatic conditions which include rainfall, humidity and evapotranspiration rates (Marsh, 1980).

One of the primary aims when disposing waste in a landfill is to minimise the generation of any leachate. Leachate production may be decreased by selecting what goes into the landfill site, as a large portion of domestic waste decomposes to form highly toxic waste substances, as well as by controlling the amount of water which either infiltrates from the surface into the landfill site or moves laterally as ground water into the deposited refuse (Harrington *et al.*, 1986). The most practised method to control leachate migration, although labour intensive, is to provide impenetrable liners in the bottom and on the sides of the landfill, and to collect any fluids, preventing their movement into the ground water. These plastic liners, as well as multiple liners, are important, especially where leachate production is a problem, as in Natal. Natural liners such as clays and bentonites are also used (Blight, 1992), which may act as natural biological filters, removing bacteria and other contaminants from the leachate (Hagerty *et al*, 1973).

When ground water infiltrates through refuse, increased recharge causes mounding of the water table within or below the landfill. This results in the downward and outward flow of leachates from the landfill, forming a pollution plume, which is a potential threat to the ground water resources, as well as to streams and other surface water bodies. Landfills situated on relatively permeable materials may have leachate migration contaminating areas many times the size of the actual landfill site, as was shown in a study by Wallick *et al.* (1976). If the water table is in hydraulic continuity with the landfill, ground water may supply a major quantity of water into the system, which could be converted into leachate (Farvolden, 1975).

The composition of leachate is dependant on the stage of decomposition of the waste and the materials that are contained within the landfill (O'Leary *et al.*, 1986). Therefore leachate compositions tend to vary significantly between landfills, and also at individual landfills over time. In general, ground water polluted by leachates from landfills has significantly higher concentrations of the ions and organics. As the major ions are generally considerably higher than background ground water concentrations, it is often possible to detect the presence of leachates in the ground water, as well as in surface water, thus enabling relatively early detection of pollution. Pohland (1986) illustrates the value of certain constituents of leachates in identifying the stage of development of a landfill until it finally stabilises (see Chapter 3). Throughout the various stages of development of the landfill, processes can be followed by isotopic tracers as they are conservative, and unaffected by the changing leachate chemistry.

1.3 Ground Water Monitoring

It is important to carry out a hydrogeological investigation of the potential site prior to the development of a landfill in order to determine the location and history of the ground water levels, the direction and rate of flow of the ground water, and the recharge potential of the aquifer. Sites which should be avoided for landfill development are 3000m from the end of any runway or within 500m of an airport boundary, areas below the 1 in 50 year floodline; areas in close proximity to significant surface water bodies; unstable areas; sensitive ecological and/or historical areas; catchment areas for important water resource areas; areas characterised by flat gradients, shallow or emergent ground water; areas of ground water recharge on account of topography and/or highly permeable soils; areas overlying or adjacent to important aquifers; areas characterised by shallow bedrock; areas in close proximity to land uses which are incompatible with landfilling; areas upwind of a residential area. (DWAF, 1994a). The topography and soil characteristics of the site must be such that the amount of rain and surface water infiltrating the landfill will be kept to a minimum. Hoeks (1976) suggests locating the

site close to the ground water divide, preferably in areas with thick aquifers. This is intended to ensure that the transport of the leachate through the aquifer to any surface water will take a long time.

As landfill sites are considered to be final disposal methods, they are often not sufficiently monitored, particularly after closure. It is essential that ground water in the vicinity of the site is monitored in order to assess the success or failure of the landfill in retaining the waste. It also allows the opportunity to detect contaminants escaping the landfill, providing an early warning before contaminants enter other water resources. The American Society of Civil Engineers (ASCE, 1976) recommend that the quality of both the groundwater and surface water in the immediate vicinity of the site and at some reasonable distance around the site be monitored. For the purpose of sampling ground water, pollution monitoring boreholes are drilled in the vicinity of the landfills and are usually covered with manhole covers to prevent vandalism and any surface water or foreign material entering the hole.

The monitoring boreholes utilised by the Johannesburg Municipality are generally drilled by air-percussion drilling, without the addition of chemicals (DWAF, 1994c). Slotted casing is used to ensure lateral ground water flow, and usually extends a couple of metres into the solid rock. The casing installed in the monitoring boreholes in the two Johannesburg Municipality landfills addressed in this study is 110 mm un-plasticised Poly Vinyl Chloride (UPVC), which is the minimum size recommended (DWAF, 1994c). At the two Bloemfontein landfill sites in this study, steel casing is used which is considerably more durable than UPVC casing, although susceptible to corrosion, and has a larger diameter, making the boreholes accessible to most pumps.

The Johannesburg City Council operates three landfills, each serving a different geographical region. They are the Linbro Park, Robinson Deep and Goudkoppies landfill sites. The Meredale Refuse disposal site was recently closed as it had reached its

capacity. The frequency with which boreholes are monitored at these landfills is determined by the City Council's Pollution Monitoring (Leachates) Committee once laboratory results have been analysed. At present, boreholes are claimed to be sampled every two months using the open-bailer method, where a sampling tube is lowered down the borehole by means of a string or rope, and a sterile glass bottle and chemical-sample bottle are filled. The depth to the water table is measured along the string using a tape measure. This study clearly established that this sampling method is inadequate as only a fraction of the entire water column is sampled (see Section 6.4.2). As no water is extracted prior to sampling, *in situ* ground water will not have infiltrated into the borehole, resulting in a sample which has been standing in the borehole for an unknown period of time, and is not representative of the ground water at the time of sampling. The casing, constructed from UPVC, is not sufficiently durable to monitor the ground water for the life of the landfill, as well as after the site is closed (see Sections 4.3.2 and 5.4.1).

To determine how long monitoring should be continued once landfill operations have ceased depends on a number of factors, which include the amount of waste deposited, the geology of the site, the depth to the ground water table, amount of precipitation, the proximity of residences, and the eventual use of the completed site (ASCE, 1976). Ongoing inspections and maintenance required after site closure will be determined by the Department of Water Affairs (DWAF, 1994a). It is important to continue monitoring until conditions around the site stabilise. The ASCE (1976) suggest a period of at least 5 to 10 years, although it is estimated that a landfill may take centuries before it ceases to emit pollution in harmful concentrations (Blight, 1992). The Bloemfontein University recommend at least 20 years for the two Class 2 sites in Bloemfontein (Bekker, 1992). Farvolden (1975) established that the leachate of a landfill improved in quality with the age of the landfill, while Fritz *et al.* (1979) found an improvement in the quality of water 7 years after the closure of a landfill.

1.4 Sampling Methods

Sampling is an important component of studies involving ground and surface water. It is vitally important when sampling a natural water body that the sample taken represents the body of water being sampled. The representativeness of a sample will depend on the homogeneity of the water body, the number of samples taken, the manner of collection, as well as the size of the samples (Hem, 1975).

When ground water is sampled from boreholes, the volume of water standing in the hole should be pumped out or purged, in order to remove any stagnant water so that the sample is truly representative of the ground water at the time. Purging involves the removal of sufficient water until parameters such as conductivity and pH are stable. Weaver (1992) suggests the removal of three times the volume of water standing in the hole. This study has shown that this practise is not always possible, however, as it is often the case that the borehole takes a substantial amount of time to recover once the first standing water volume has been removed. The recovery time generally depends on the aquifer type, fractured aquifers being somewhat slower than sandy aquifers, as well as the borehole construction and maintenance.

The level of the water standing in the borehole is usually the first measurement taken when arriving at the borehole. These measurements on a set of observation boreholes are useful to determine the direction of water flow. Along with topographic maps, a better interpretation of sampling results can be achieved. This is generally followed by measuring the depth of the borehole. Once the depth and standing level of the water are known, it is then possible to determine to what level the pump should be lowered, as well as the volume of water which must be purged before a representative sample may be taken. Well-head measurements of temperature, alkalinity, electrical conductivity, pH and dissolved oxygen are essential while still in the field in order to obtain a meaningful reading and a clearer understanding of the nature of the ground water. This information

is also vital when a more in-depth interpretation of the hydrochemistry is undertaken. It is important that the samples are clearly labelled with all the appropriate information.

A further requirement in hydrogeological studies is an adequate sampling network. This should, where possible, include samples of ground and surface water both up-gradient and down-gradient of the study area, which will monitor any alteration to the chemistry of the ground water entering or leaving the site.

1.5 Environmental Isotope Application

Environmental isotopes are important and extremely useful tools in the study of landfills and their effects on the neighbouring surface and ground water. The stable isotopes of hydrogen and oxygen reflect the source of water entering the refuse, whether it be rainfall, mains water or from a surface water body. Fritz *et al.* (1976) found that deuterium and oxygen-18 in the ground water leaving a solid waste disposal site were enriched relative to the surrounding ground water, and could therefore be used as a tracer of this water. A more identifiable tracer of the leachate movement in surface and ground water is artificial tritium, which often has been found to be in leachates from South African (Verhagen *et al.*, 1998) and British sites (Robinson *et al.*, 1995) well above natural environmental concentrations. Environmental tritium can also be used to assess the turnover time of the monitoring boreholes, ensuring representative samples of the ground water at that time. Tritium and carbon-14, are also useful in determining the recharge potential of an aquifer prior to the development of a landfill site, thus assessing potential pollution of the aquifer.

Environmental isotopes together with chemical data have proven to be useful in assessing incipient pollution, even when levels were not yet unacceptable. In a study in the Midrand area (Verhagen *et al.*, 1995), the sulphate, chloride and nitrate concentrations, which were not high enough to be considered anthropogenic, were

correlated with tritium and stable isotope data. From these correlations it was established that chloride and sulphate, which were associated with active recharge, were introduced into the ground water by sources other than rainfall recharge, probably industrial, while the nitrate, which was associated with rainfall and active recharge, was primarily the result of agricultural activities.

CHAPTER 2

2 ENVIRONMENTAL ISOTOPES : PRINCIPLES IN APPLICATION

2.1 Introduction

Isotopes are atoms of the same element which have the same atomic number, but different mass numbers. An atom is characterised by its atomic number (number of protons) and the mass number (sum of protons and neutrons). The mass number (A) and atomic number (Z) appear in front of the chemical symbol of the element (X) as superscript and subscript respectively:

$$^{A}_{Z}X$$

The chemical symbol defines the atomic number and generally only the mass number is given. When the nucleus of an isotope is unstable and disintegrates spontaneously, radiation is emitted and another isotope is formed.

For several decades, naturally occurring isotopes, or environmental isotopes, in the hydrological cycle have been used in ground and surface water studies. Environmental isotopes are defined by Payne (1983) as "those isotopes, both stable and radioactive, which occur in the environment in varying concentrations over which the investigator has no direct control". They are valuable tools and complement the wide range of techniques used by hydrologists. The radioactive isotopes used in this study are tritium

(^{3}H) and radiocarbon (^{14}C) with which the mean residence time of ground water can be assessed. The stable isotopes, or non-radioactive isotopes used, oxygen-18 (^{18}O), deuterium (^{2}H) and carbon-13 (^{13}C), indicate source areas and mechanisms of recharge.

Environmental isotopes are assumed to be conservative and, once introduced into the ground water, will not react chemically or be gained or lost by radioactive decay. They can be applied to the investigation of ground water recharge, the study of the saturated and unsaturated zones, dating of ground water, distinguishing between different sources of ground water, the velocity of ground water movement, and quantitatively estimating proportions of the mixing of water of differing ages and origins.

2.2 Environmental Isotope Sampling

2.2.1 Deuterium (D) and Oxygen-18

A 50 ml sample is usually adequate for the stable isotope analyses. The sample bottles should be rinsed several times with water from the site, almost completely filled allowing for expansion due to temperature changes, and then tightly sealed. Precautions must be taken to ensure that evaporation is kept to a minimum.

2.2.2 Tritium

One litre samples are usually taken to allow for the enrichment of low tritium concentrations, as well as for repeated measurements. However, this will depend on the laboratory requirements.

2.2.3 Carbon-14

The precipitation and extraction of the total dissolved inorganic carbon (TDIC) species for ^{14}C analysis is performed in the field. Depending on the TDIC concentration and the laboratory requirements, samples of between 50 and 200 litres are required. Various methods of extraction can be employed at the sampling point. For this study, the samples were collected in large polyethylene bags.

The sample must be collected with minimum exposure to air. If ground water is in contact with the atmosphere for a prolonged time, isotope exchange may take place between the carbon dioxide in the air and the dissolved bicarbonate, and the ^{14}C-content of the water will be increased. It is therefore necessary to pump each well long enough to replace the free-standing water before taking a sample (see also: 1.4 Sampling Methods). Vogel (1967) found that contact with the atmosphere for a short time does not noticeably alter the ^{14}C-content of most ground water.

Saturated sodium hydroxide is added to the sample to raise the pH well above 8.2 in order to dissociate carbonate ions from bicarbonate ions (Figure 3). Barium chloride is added to the sample and as a result the carbonate dissolved in the water precipitates out as barium carbonate ($BaCO_3$). The clear supernatant is then drained and the $BaCO_3$ precipitate is removed from the field. It is left to settle and is then transferred to a bottle and taken to the laboratory. The precipitate is treated in the laboratory with orthophosphoric acid to liberate CO_2, which is then used to produce samples for radioactivity measurements. Aliquots of gas for ^{13}C measurements is derived from the bulk CO_2.

2.3 Stable Isotopes

2.3.1 Introduction

Stable isotopes are important tools in hydrology, and are usually utilised as indicators of evaporation, to determine the genesis and mixing of ground water, or to determine the source area of ground water (Verhagen *et al.*, 1991).

The isotopic composition of water, which is measured with the use of a mass spectrometer, is expressed in the delta notation (δ), which is calculated as follows:

$$\delta = \{(R_s - R_{std.}) / R_{std.}\} \times 1000\ ‰$$

where R_s is the isotope ratio of the sample, and $R_{std.}$ is the isotope ratio of the standard. Water with a higher concentration of the heavy isotope species relative to the standard will have a positive δ value, and water with a lower concentration of the heavy isotope species relative to the standard will have a negative δ value.

2.3.2 Deuterium and Oxygen-18

The isotope ratios for deuterium (D) and oxygen-18 are $D/^1H$ or $^{18}O/^{16}O$ respectively. The delta values are expressed in parts per thousand or per mil (‰) deviation from a standard. High accuracies of about 0.1‰ for $\delta^{18}O$ and 1.0‰ for δD are readily obtained. The international reference standard for deuterium and oxygen-18 is SMOW (Standard Mean Ocean Water), which corresponds to a hypothetical water with hydrogen and oxygen isotope ratios close to the mean isotope ratios of ocean water, the largest water reservoir. As SMOW does not actually exist, a water sample having an isotopic composition as close as possible to that of SMOW was prepared, and is available to calibrate laboratory measurements. This water sample is called VIENNA-SMOW, abbreviated V-SMOW.

Hydrogen and oxygen isotopes form water molecules of differing mass, the most common having molecular masses of 18, 19 and 20 ($^1H_2{}^{16}O$, $^1HD^{16}O$ and $^1H_2{}^{18}O$ respectively) (Drever, 1982). These water molecules have differing vapour pressures and, during a change of phase such as evaporation, condensation or sublimation, they undergo a process known as isotope fractionation which involves the enrichment of the lighter isotope in the more volatile phase. For instance, when water undergoes evaporation, the lighter isotopic species preferentially leave the surface, and the remaining body of water is enriched in the heavy isotope species or more positive δ values (Payne, 1972).

The altitude effect was demonstrated by Gonfiantini *et al.* (1976) on the volcanic island Gran Canaria, where these isotopes were used to distinguish the various altitudes of recharge, and by Payne *et al.* (1979) who associated the deep ground water in Ecuador, which has a depleted mean stable isotopic composition, with the Chimbo River.

The Kalahari ground water investigated by Mazor *et al.* (1977) was found to have a stable isotope composition lighter than the annual average rains because only intensive rains are effectively recharging the ground water, which have an isotopic composition lighter than the average annual rainfall composition. In a recharge study in Gordonia, Verhagen (1990) noted a dissimilarity in the stable isotope data between ground water alongside the river valley, and ground water a few kilometres south of the river valley which had an isotopic signal closer to that of precipitation in the area. It is therefore evident that rainfall, and not river water, is the source of the fresh ground water found further to the south. In studies on sanitary landfill sites in Germany, Fritz *et al.* (1976) found the polluted ground water to be enriched in $\delta^{18}O$ in the range of 1.5 to 2 ‰ with respect to the adjacent unpolluted ground water. A similar enrichment was measured for deuterium. Assuming that deuterium and oxygen-18 behave as conservative tracers in leachates, provided the leachate does not seep to the surface, it was suggested that the enrichment is possibly due to evaporation within the landfill site, decomposition of

organic matter, the exchange of oxygen between water and carbon dioxide, or differing isotopic compositions of recharge water. The principal aim of the study of Fritz *et al.* (1976) was to distinguish between the removal and dilution of pollution, and was accomplished using mixing diagrams of oxygen-18 and chemical pollution indicators. It was found that in most polluted boreholes, over a period of 3 years, a decrease in oxygen-18 and a decrease in the chemical constituents occurred, indicating an improvement in the quality of the water.

The Meteoric Water Line (MWL)

It is necessary in environmental isotope studies to determine the isotope compositions of oxygen-18 and deuterium in precipitation. Precipitation forms under virtually equilibrium conditions between the condensed and vapour phases in the atmosphere, and results in a linear relationship between the isotopic compositions of deuterium and oxygen-18 in meteoric water as follows:

$$\delta D = s\ \delta^{18}O + d$$

The δ-values are in parts per thousand (‰) relative to SMOW. This line, called the meteoric water line (MWL), has a slope (s) of 8 for water which has not undergone evaporation (Figure 2)(Craig, 1961). The slope of the MWL will decrease as a result of kinetic evaporation at the surface of a water body as this is a non-equilibrium process. As the fractionation for oxygen-18 is greater than that for deuterium, the slope for evaporation usually ranges from 4 to 6, depending on meteorological conditions (Wallick *et al.*, 1976). The average global value for the deuterium excess (d) for precipitation is 10, although it may vary slightly according to region. Extreme non-equilibrium conditions result due to evaporation from warm, land-locked seas into very dry air masses, resulting in deuterium excesses of 20 ‰ and more (Yurtsever and Gat, 1981). Examples are the Persian Gulf, the Red Sea, the Black Sea and the Mediterranean Sea, particularly the eastern parts.

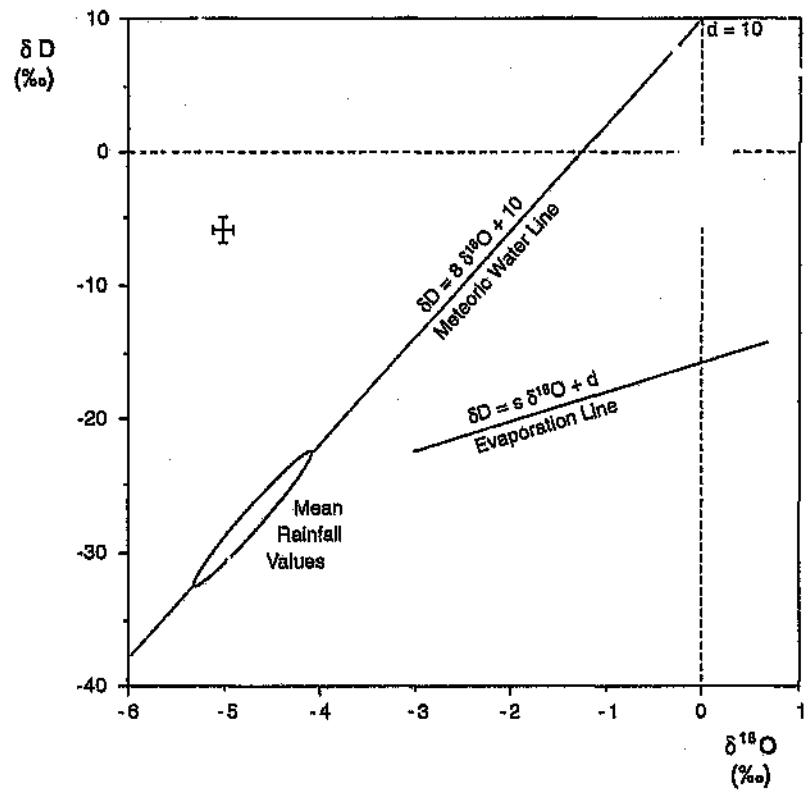

Figure 2. Typical δD - $\delta^{18}O$ diagram, showing the world meteoric water line, a typical spread of mean rainfall compositions for a specific area, and an evaporation line ($s < 8$; $d < 10$). A typical analytical error for the data is also shown

Temperature, altitude and the amount effect result in a variation in the isotopic composition of precipitation along the meteoric water line. The temperature effect depends on the temperature at which the vapour leaves the ocean water, forms clouds and rain or snow, higher temperatures resulting in heavier isotopic values. This effect is a useful tool in distinguishing between summer and winter rainfall recharge. The altitude effect is caused by heavier isotopes in precipitation being depleted as a cloud rises up a mountain slope, resulting in isotopically lighter residual precipitation. This contrast can therefore be useful in tracing groundwater recharge in mountainous terrain. The amount effect depends on the amount of precipitation that occurs between the time the air mass left the ocean and the time of the particular rainfall. The rain becomes progressively isotopically lighter from the coast inland, and from the equator towards the poles (Mazor, 1991).

The meteoric water line can also act as a mixing line. In the Gauteng area, the tap water plots on the MWL, and it is therefore possible to determine the percentage tap water in the ground water. This was established in a study in the Midrand area where high $\delta^{18}O$ values, as well as high tritium and carbon-14 concentrations, in a shallow borehole, led to the discovery that a highly polluted drainage channel was contaminating the ground water (Verhagen *et al.*, 1995a). The distinctive stable isotopic composition of the tap water has been successfully utilised in detecting and locating leaks from the supply mains in the suburbs of Pretoria (Verhagen *et al.*, 1995b).

2.3.3 Carbon-13

The ratio of the stable isotopes, ^{13}C to ^{12}C, is approximately 1 to 90 and any small variations in this ratio are the result of isotopic fractionation (Smith, 1976). The abundance of ^{13}C is expressed in per mil deviation (δ) of the $^{13}C/^{12}C$ ratio in the sample from a standard (see Section 2.2.1). The standard used for the ^{13}C measurements is the Pee Dee Belemnite (PDB), a particular type of marine limestone. The $\delta^{13}C$ in ground

water is established by the concentration of $\delta^{13}C$ in the inflow water, as well as the supply of carbon to, and removal of carbon from, the water while it is moving through the aquifer (Drever, 1982). These values therefore give an indication of any carbonate chemical interaction between the ground water and the aquifer rock, which must be taken into account as this can affect the ^{14}C values.

The $\delta^{13}C$ values help to determine the hydrochemical source of the carbon in ground water, as well as to distinguish between carbon derived from the oxidation of organic matter (isotopically light carbon), and carbon derived from the dissolution of carbonate minerals (isotopically heavy carbon) (Drever, 1982). $\delta^{13}C$ also gives information about mixing ratios as well as being used as a tracer in pollution studies of any contamination of water by dissolved carbon of suspected organic origin (Letolle *et al.*, 1983). Down-gradient from landfill sites, $\delta^{13}C$ values more positive than -5 ‰ for ground water in equilibrium with atmospheric CO_2 will usually suggest an artificial source of carbon, particularly when ^{14}C values are higher than environmental values.

The $\delta^{13}C$ values obtained by Mazor *et al.* (1977) in ground water studied in the Kalahari range from -13.7 ‰ to -5.3 ‰, and correlate positively with the HCO_3 concentrations, indicating a higher portion of fossil carbon in water with higher bicarbonate concentrations. A similar correlation was found between the $\delta^{13}C$ values and pH values. The wide range of values in the ground water of the Kalahari is thought to be the result of fast reactions under varying environmental conditions of infiltration.

2.4 Radioactive Isotopes

2.4.1 Introduction

Tritium and carbon-14, the radioisotopes applied in this study, emit low energy beta particles. These particles have a continuous range of energies from zero up to a

maximum ($E_{max.}$), and because of their low mass, they are easily scattered. It is therefore essential for the efficient detection and measurement of the samples to be carried out in a specialised laboratory.

The specific activity of the radioisotope starts decreasing (decaying) as soon as the recharge water moves deep enough into the ground water zone and is effectively isolated from the atmospheric source. This decay is exponential and is expressed in terms of the half-life of the source (Smith, 1976). The specific activity decreases according to the radioactive decay law:

$$t = (\tau / \ln 2) \cdot \ln (A_o/A)$$

from which the age of the sample t can be calculated, where τ is the half life of the given radioisotope, and A is the specific activity after time t. This law, together with measurements of the radioisotope content of the ground water, can be used as a guide to the ground water "age" or residence time. When applying radioactive isotopes in hydrological studies, the initial specific activity (A_o) of the radioactive isotope must be known (Payne, 1972). All the processes which change the radioisotope concentration other than decay, such as exchange and dilution, must be taken into account in order to apply the method precisely.

An example of the potential of radioisotopes is discussed in a study of Kalahari ground water by Mazor *et al.*, (1977). Radiocarbon and tritium data provided positive evidence of recent rain recharge within the last two decades in the northern and, in some cases, in the southern Kalahari.

2.4.2 Tritium

Tritium is the only radioisotope which is part of the water molecule, and for this reason is considered an important tool in ground water studies (Fontes, 1983). It is chemically bound in the water molecule and is therefore a conservative radioactive tracer of water

behaviour. The half-life of tritium is 12.43 years with a mean life of 18 years, and the dating range covers 100 years. It is measured in tritium units (TU), and one TU has an isotopic ratio of $^3H/^1H = 10^{-18}$ (Payne, 1972; Torgersen *et al.*, 1979). Low energy beta radiation is emitted, with an energy maximum of $E_{max.} = 18$ keV.

Environmental tritium is produced naturally by the interaction between nitrogen atoms and cosmic ray neutrons in the upper atmosphere ($^{14}N + n \rightarrow {}^3H + {}^{12}C$). The free tritium atoms are oxidised to $^1H^3HO$ which enter the hydrological cycle in precipitation. A second source of tritium in the atmosphere is man-made, and is the result of nuclear weapons tests in the early fifties and in 1961. The tritium values in precipitation in the northern hemisphere increased dramatically by up to 1000 times the natural level, while in the southern hemisphere a slightly delayed increase of some 10 times the natural level was observed. As a result it is no longer possible to assume that the initial specific activity of tritium is constant. Since 1961, the tritium concentration, as well as the stable isotopic composition of precipitation, has been monitored by the IAEA/WMO Global Network of Isotopes in Precipitation (GNIP) (Payne, 1983), from which the tritium input for a particular study area can be estimated. This information is also utilised to determine mean residence times of unconfined ground water in fractured and sedimentary aquifers.

Qualitatively, ground water with a value of greater than 1 TU has a turnover time of less than 30 years. This information can be used to study the storage of a ground water system and determine its vulnerability to short-term pollution sources, such as sanitary landfills and agriculture, which may exist at the surface (Smith, 1976).

A discontinuity in the tritium (or stable isotope) content is often the result of a boundary between two neighbouring aquifers, while comparable tritium and stable isotope compositions in the adjacent water bodies suggest the movement of water through this boundary. It is particularly useful in locating the source of recharge of a water body in

systems with a rapid turnover (Harpaz *et al.*, 1963). It was established in the London Basin that chalk outcrops had tritium values of 10 to 20 TU while the confined aquifer had values close to zero, suggesting the slow movement of the water through the chalk (Smith, 1976).

Tritium analyses provided Mazor *et al.* (1977) with a method to group the ground water of various parts of Botswana into three active recharge areas, depending on tritium content. These areas could then be correlated with the rain distribution and the thickness of the Kalahari beds from north to south. A study by Verhagen *et al.* (1979) using tritium profiles provided evidence of accelerated infiltration in the Kalahari sands as a result of a succession of high rainfall seasons. This was determined by the presence of tritium concentrations greater than 10 TU (bomb produced ^{3}H) at depths of 16 to 23 metres.

Tritium can be used as a tracer of local contamination due to sewage water from industries using radioactive products, as well as the dumping of radioactive products such as gaseous tritium light sources (GTLS) and products with radioluminous paint or plastic, such as watches and instrument dials, in landfill sites (Wehner, 1979).

2.4.3 Carbon-14

The carbon-14 content of ground water measures the time which has elapsed since the ground water passed through the zone of aeration. This radioisotope of carbon emits beta radiation and has an energy maximum ($E_{max.}$) of 156 keV. The specific activity of ^{14}C decreases with a half-life of 5730 years, although the previous figure of 5568 years is still used in order to compare earlier archaeological data (Bowen, 1986). The difference between the half-lives is not of great importance in hydrology.

Environmental carbon-14 is produced in the upper atmosphere by cosmic ray neutrons which react with nitrogen atoms ($^{14}N + n \rightarrow {}^{14}C + {}^{1}H$). Carbon-14 has also been added to the atmosphere as a result of the testing of thermonuclear devices. The carbon-14 oxidises to CO_2, which is then mixed with the atmospheric carbon dioxide reservoir (Payne, 1972). It reaches the ground water indirectly by various interactions of biological material, soil CO_2 and calcite with soil water.

In order to calculate the mean residence time in a given sample, the initial activity of the TDIC before any radioactive decay must be estimated after all chemical and isotopic processes. The residence time produced is only an apparent age, as it is often altered by the carbon exchanging with the rock minerals, and by solution and precipitation of carbon compounds (Bowen, 1986). The ^{14}C concentration is in relation to the $^{14}C/^{12}C$ ratio of wood grown in 1850 AD, in an environment free of fossil carbon dioxide for the last 7000 years, after which it is assumed to have been constant (Drever, 1982). This natural concentration in atmospheric CO_2 is $[^{14}C]/[^{12}C] \approx 10^{-12}$ and is equivalent to 100 pMC (percent modern carbon) (Torgersen *et al* 1979; Verhagen *et al.*, 1991). The sample is measured as a percentage relative to the standard as pMC.

Carbon-14 is of value in ground water studies as it is found in ground water as dissolved CO_2, HCO_3^-, and CO_3^{2-}, which collectively are the total dissolved inorganic carbon (TDIC). In order to more meaningfully interpretate carbon-14 values, complete major ion chemical analyses of the water are required, including field measurements of pH and alkalinity, and $\delta^{13}C$ measurements of the different carbon-containing species (Torgersen *et al.*, 1979). This is necessary as carbon-14 is a non-conservative tracer of the water molecule. The $\delta^{13}C$ of the bicarbonate in the ground water is measured to determine the proportion which originated from modern biogenic material, as well as any exchange with carbon in the lithology (see 2.2.2).

Together with tritium, the assessment of ground water recharge using ^{14}C is possible, as was established in the Kalahari by Mazor *et al.* (1977) and Verhagen (1990), and in the Libyan Desert by Vogel *et al.* (1963).

CHAPTER 3

3 SOME HYDROCHEMICAL PRINCIPLES

3.1 Introduction

The chemistry of ground water is generally dependent on the recharge water, substances from the atmosphere, human influences, and subsurface geochemical processes, which include the interaction of water with vegetation, the interaction between water and minerals by dissolution, hydrolysis and mineral transformations (Matthess *et al.*, 1982). The water quality is controlled by these processes during its course below the surface, changing the concentration and nature of dissolved solids. These are mainly present as dissociated species; cations (positively charged ions) and anions (negatively charged ions). These ions have varying solubilities which are controlled by external factors, such as temperature, pressure, and hydrochemical parameters, such as the pH, redox potential and ionic strength of the solution (Matthess *et al.*, 1982).

Carbonic acid (H_2CO_3) is the most important acid involved in the dissolution of minerals and is the result of dissolved carbon dioxide in the water (Matthess *et al.*, 1982):

$$CO_2 + H_2O \Leftrightarrow H_2CO_3$$

The carbon dioxide content in the unsaturated zone can have concentrations greater than 100 times the atmospheric concentration due to the respiration of the roots of plants, and the bacterial decomposition of plant material (Mazor, 1991).

Hydrolysis is the process whereby minerals are broken down as a result of the presence of H^+ and OH^- ions in water. One of the primary sources of H^+ in water is the dissociation of carbonic acid, H_2CO_3 (Davis *et al.*, 1966):

$$H_2CO_3 \Leftrightarrow H^+ + HCO_3^-$$

$$HCO_3^- \Leftrightarrow H^+ + CO_3^{2-}$$

Hydrolysis is the main process in weathering the surface of relatively insoluble silicate minerals such as feldspars, micas and hornblende. The clay minerals kaolinite, montmorillonite and illite are formed, and Ca^{2+}, Mg^{2+}, Na^+, and K^+ ions are removed into solution. The breaking down of the minerals is largely due to the low pH and redox potential (Eh) of the water (Matthess *et al.*, 1982).

3.2 Dissolved Ion Sampling

Samples for dissolved ions are usually collected in half-litre plastic bottles, although it is often necessary to confirm with the laboratory as to how much sample is needed, as this will depend on the laboratory method of analysis. The sample bottles and caps must be clean, and rinsed at least three times with water from the sampling point. The samples must be kept as cool as possible and out of direct sunlight. In the case of trace and heavy metals, the sample must be maintained at a pH of less than 2 in order to keep the metals from precipitating onto the sample bottle. Suspended solids should be filtered from the sample prior to being acidified otherwise metals will be leached from these solids.

3.3 Field Measurements

A number of chemical and physical observations (minimally temperature, electrical conductivity, pH and alkalinity) are taken in the field at the time of sampling as they give a well-head indication of the water quality and basic chemistry. These parameters may also be altered once the sample is removed from the sampling site. The pH and alkalinity values are likely to shift once the sample is exposed to the atmosphere due to the release of carbon dioxide under pressure in the ground water, causing the HCO_3 to break down. It is also possible for the electrical conductivity of the water to decrease as, once the sample is in a container, certain minerals may precipitate from the water. The temperature is another parameter which will begin to change almost immediately.

3.3.1 Electrical Conductivity

Electrical conductivity (EC) is measured in order to estimate the total dissolved ions (TDI) in a ground water sample, as all individual dissolved species carry a charge. Conductivity is the reciprocal of resistivity, and is expressed in mhos, usually as specific electrical conductivity, µmhos/cm. In the S.I. system of units, conductivity is given as milliSiemens per metre, where 1 mS/m = 10 µmhos/cm. The specific conductivity of water is relative to the temperature, the type of ions present, and the concentration of the various ions (Davis *et al.*, 1966). Since conductivity increases with temperature, it is necessary that measurements be related to a standard temperature, normally taken as 25° C. For most ground water, a quasi-linear relationship exists between the total dissolved solids (TDS) and the electrical conductivity. For a rough estimate of the TDS in ppm in fresh water, the specific conductivity of the water in micromhos should be multiplied by 0.7 (Davis *et al.*, 1966). In many cases, this factor may be as low as 0.6.

3.3.2 Alkalinity

Alkalinity is defined as the amount of strong acid which can be added to a solution without changing the pH of that solution. Alkalinity is produced almost entirely by bicarbonate and carbonate ions, or total dissolved inorganic carbon (TDIC), bicarbonate usually being the dominant ion. It is therefore usually a reliable measure of these ions (Davis *et al.*, 1966). When the standardised strong acid solution is added, these ions are replaced by the strong acid anions:

$$Ca(HCO_3)_2 + 2HCl = CaCl_2 + CO_2 + 2H_2O$$

When replacement is complete and all the alkalinity is accounted for, free strong acid, which will readily release H^+ ions due to high dissociation, appears in the solution and the pH immediately begins to drop:

$$HCl \Leftrightarrow H^+ + Cl^-$$

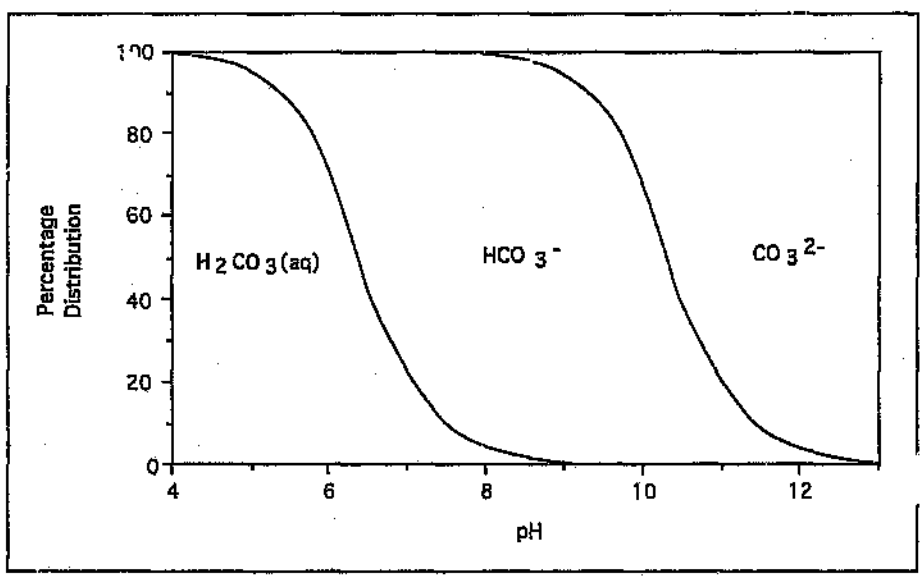

Figure 3. Percentage of total dissolved inorganic carbon (TDIC) in solution as a function of pH, at standard temperature and pressure (Hem, 1975).

A pH meter or an indicator, such as methyl orange is used to determine the titration end-point, achieving a resolution of at least 0.1 meq/l. When the change of colour occurs at pH 4.5, the number of milliequivalents of acid added equals the number of milliequivalents of total alkalinity (or bicarbonate) in the unknown solution (Figure 3). Water with pH values greater than 8.5 may first be titrated using phenolphthalein indicator to determine carbonate alkalinity.

Other sources of alkalinity include weak acids such as silicic, phosphoric and boric, as well as organic acids, usually produced in landfills (Hem, 1975).

3.3.3 pH

The pH of a solution is a measure of the concentration of the hydrogen ion, H^+, and is calculated from the negative logarithm of the hydrogen ion concentrations, expressed as moles per litre.

$$pH = -\log_{10}H^+$$

A small proportion of pure water is dissociated into H^+ and OH^- ions, written as:

$$H_2O(l) = H^+ + OH^-$$

In mass law form, this equation can be expressed as:

$$\frac{[H^+][OH^-]}{[H_2O]} = K_w$$

where K_w is the equilibrium constant and, at 25°C, is equal to 10^{-14}. As the activity of $H_2O = 1$, a neutral solution is therefore $[H^+] = [OH^-] = 10^{-7}$. The pH of pure water is therefore 7, which is equivalent to 1×10^{-7} moles per litre of hydrogen ion, a neutral solution. Any increase in the hydrogen ion concentration will decrease the pH, while a decrease in the hydrogen ion concentration will increase the pH. Natural water contains dissolved carbon dioxide and hydrogen carbonate ions which buffers the pH in the range of 5.0 to 8.0. Other reactions which affect the pH is the dissociation of acidic

solutes, hydrolysis reactions, certain reactions where precipitates are formed and most oxidation reactions (Hem, 1975).

To measure the pH, a colour indicator may be used, but usually a meter with a glass hydrogen ion electrode is used, reading to at least 0.1 pH resolution. The meter is first calibrated in buffers, and the readings are repeated at least once.

3.3.4 Temperature

Temperature, in degrees Centigrade (°C), is taken during or immediately after sampling for at least 30 seconds until a steady value is obtained, preferably at the pump outlet. Temperature measurements are useful in identifying different groups of water, determining water end-member properties, and investigating the depth of water circulation. They are also essential in hydrochemical calculations, eg. saturation indices. If the temperature of shallow ground water is equal to that of the rainy season it is usually an indication of rapid recharge, and if it equals the average annual temperature, it indicates retardation in the soil, probably in the aerated zone (Mazor, 1991).

3.3.5 Appearance and Taste

Taste and aroma result from the presence of bacteria, dissolved gases or minerals. Unpleasant smells are often the result of sulphur compounds in the water, usually H_2S, which are due to either reducing conditions (as is most often the case within sanitary landfills), bacteria, sewage or other pollutants (Mazor, 1991). By tasting relatively unpolluted water, the salinity may be estimated, and some major ions and pollutants may be identified. Colour should also be noted and can be compared with certain standards. The various colours are usually due to mineral or organic material in solution. The yellowish-brown colour of natural stream water is usually the result of

leaching of organic debris. Colour in natural water may also be due to inorganic and organic waste.

3.4 Dissolved Constituents

3.4.1 Anions

Sulphate

The initial source of most sulphates in the hydrosphere is from the oxidation of sulphides from igneous rocks and volcanic sources. Sulphates are also produced in various bacterial decay processes, as well as being recycled from the atmosphere. The sulphate in the atmosphere is derived from dust particles, oxidised sulphur dioxide and hydrogen sulphide gas (Davis *et al.*, 1966). The oxidised form is then returned to the ground water from the soil or in rain. In moist environments, sulphate is usually at low concentrations, but in more arid conditions, higher concentrations are often found. In arid environments, gypsum is often formed through evaporation, and then re-mobilised in rare rainfall events. Sulphur compounds are introduced into the sulphur cycle by man's activities, particularly SO_2 in flue gases and sulphate in fertilisers (Matthess *et al.*, 1982).

The most common natural process for the removal of sulphates from water is the reduction of sulphate by bacteria in anaerobic conditions, which produces hydrogen sulphide gas as a by-product.

The sulphate concentration of landfill leachate tends to increase soon after the refuse is deposited in the site as a result of aerobic oxidation, reaching values of as high as 3240 mg/l (Pohland, 1986). Once anaerobic conditions are established the sulphate concentration decreases and, after some time, is absent, as it undergoes reduction to

form sulphides. Once the landfill stabilises, and aerobic conditions return, the sulphate concentration will generally be in the range of 5 to 40 mg/l.

Nitrate

Nitrate is the most common form of nitrogen in ground water, where the concentration generally ranges from 0.1 to 10 mg/l. Most of the nitrate found in ground water is produced by the oxidation of organic nitrogen in the soil, where it acts as an important nutrient to plants. Some of this nitrate is transported to the saturated zone where it remains in oxidising conditions. A second source is from igneous rocks, which contain a small amount of soluble nitrate or ammonia (Matthess *et al.*, 1982). Fertiliser application, animal husbandry, and liquid and solid wastes usually also increase the concentration of nitrates in ground water.

If reducing conditions arise, as is often the case within landfills, it is possible for the nitrate to be reduced to nitrogen by denitrifying bacteria. This is the only method whereby the highly soluble nitrate compounds are taken out of natural water (Davis *et al.*, 1966).

In the early stages of a landfill, the nitrate concentration in leachates increases to values as high as 19 mg/l due to the oxidation of ammonia (Pohland, 1986). Once the landfill begins to stabilise, the nitrate concentration will decrease as a result of reducing conditions, forming ammonia or N_2 gas, and will eventually be absent due to complete conversion. Once the landfill has completely stabilised, concentrations of between 0.5 and 0.6 are generally found.

Chloride

The bulk of natural chloride present in ground water is most likely the result of airborne transport from the oceans, carried to earth by rain or snow, as chloride is a minor constituent of the Earth's crust. It is also present in igneous and metamorphic (Matthess *et al.*, 1982). Ground water in an igneous terrain tends to have low Cl^- values of less than 30 mg/l. Higher values would therefore indicate either the presence of more than one mineralised water, surface enrichment as a result of evaporites, or man-made pollution as liquid and solid waste material and fertilisers (Matthess *et al.*, 1982).

The chloride concentration in rain is as low as 1 mg/l, and may exceed 100 000 mg/l in brines. All chloride salts are highly soluble, so chloride is rarely removed from water by precipitation except under the influence of freezing or evaporation. Chloride is also relatively unreactive, as well as being free from effects of exchange, adsorption, and biological activity (Davis *et al.*, 1966). Due to chloride being biologically stable, it is a good tracer of leachates from landfills, and concentrations tend to range between 30 and 5000 mg/l (Pohland, 1986). Maps showing lines of equal chloride content, or isochlors, can be very useful in hydrochemistry studies.

Fluorine

Fluorine occurs more abundantly than chlorine in igneous rocks but is very much scarcer in the hydrosphere as fluorine compounds have a low solubility (Matthess *et al.*, 1982). The natural concentration of fluorine in ground water ranges from 0.01 to 10.0 mg/l, although in most fresh water the fluorine content is below 1 mg/l. Fluoride concentrations in landfill leachates tend to range between 0.1 and 1.3 mg/l (O'Leary *et al.*, 1986).

Phosphate

Apatite is the mineral with the largest phosphorous content, which is released on weathering, with the bulk of it being adsorbed on clay minerals. Generally, the concentration of phosphate in ground water is in the range of hundredths of a milligram per litre. However, in landfill leachates the total phosphate concentration can be as high as 120 mg/l, and even in a stabilised landfill the concentration can be as high as 14 mg/l (Pohland, 1986). The lower phosphate concentrations in the stabilised landfill are due to biological assimilation and the complexation of metal species (resulting from a decrease in the pH).

3.4.2 Cations

Calcium

Calcium is one of the most common cations in fresh water as it is abundant in the earth's crust and is very mobile in the hydrosphere. The concentration of calcium in potable water generally ranges between 10 and 100 mg/l, its main source being from limestone rocks, the dissolution of soil carbonate or gypsum, or the weathering of feldspathoids. Calcium is also found in fertilisers, and is frequently a component in man-made pollution. Concentrations in landfill leachates generally range between 200 and 2100 mg/l (O'Leary *et al.*, 1986), but decrease to values of around 9 mg/l in a stabilised landfill (Pavoni *et al.*, 1975)

Calcium carbonate is easily soluble in water which has a high concentration of H^+ (Davis *et al.*, 1966):

$$CaCO_3 + H^+ \Leftrightarrow Ca^{2+} + HCO_3^-$$

The primary process responsible for the removal of Ca^{2+} ions from the ground water is by ion exchange for sodium and other ions.

Magnesium

Th concentration of magnesium in fresh water is usually less than 40 mg/l. Concentrations of up to 1140 mg/l have been found in landfill leachates (Pohland, 1986). During the production of methane the pH decreases, causing the complexation of metal species, which results in a decrease in the magnesium concentration. Once the landfill stabilises, magnesium concentrations generally range between 81 and 190 mg/l.

In dolomitic environments, ground water usually contains about equal amounts of calcium and magnesium. However, the solubility of calcium is closely controlled by the

which can result in calcium precipitating out of solution more rapidly, leaving a predominance of magnesium. When dolomitic ground water is used for irrigation, high magnesium concentrations may build up in the soil.

Magnesium is generally associated with calcium in the hardness of ground water. Low values of magnesium and calcium are sometimes found in water which has undergone natural softening by cation exchange. Most commonly, clays will exchange sodium, if available, for both magnesium and calcium ions (Davis *et al.*, 1966).

Sodium

Concentrations of sodium vary from around 1 mg/l in rain water to 10 500 mg/l in sea water, with even higher concentrations in brines. NaCl is the most soluble salt, followed by Na_2CO_3 and $NaHCO_3$. Sodium is usually the major cation in mineralised ground water with the exception of gypsiferous and Ca^{2+}-HCO_3^- water (Matthess *et al.*, 1982).

In sandstones, sodium is found in either the unweathered mineral grains, the cement, or as a crystalline residue in the pores (Davis *et al.*, 1966). Ground water from anhydrite

deposits shows high concentrations of SO_4^{2-} in combination with Ca^{2+} or Mg^{2+}, which are also found in gypsum. Sodium is also found in various chemicals used by man, and is often present in the ground water as a result of man-made pollution. Concentrations in landfill leachates range from 20 to 7600 mg/l (Pohland, 1986).

Potassium

Most potable ground water has potassium concentrations of less than 1 mg/l, usually ranging between 0.1 and 0.5 mg/l. The potassium content of igneous and metamorphic rocks is only slightly less than that of sodium. Sandstones have a relatively high potassium content, as the ions are adsorbed in the cementing material, as well as being present in the unweathered potassium feldspars and micas. Even though potassium silicates are fairly resistant to weathering, potassium ions are released by weathering processes. Potassium is also present in agricultural fertilisers, and has occasionally been observed as a result of man-made pollution (Davis *et al.*, 1966).

Although potassium salts are very soluble, the ions are relatively immobile as they often become fixed, either by sorption on clay minerals or in the formation of secondary minerals (Matthess *et al.*, 1982). For this reason, potassium should be avoided as an indicator of ground water pollution in sanitary landfills as it does not migrate with the polluted ground water. However, the potassium concentration of leachates within a landfill remain roughly constant throughout the life of a landfill, ranging in concentration from 35 to as high as 2300 mg/l (Pohland, 1986).

Iron

Iron is present in various minerals and mineral groups, including pyroxenes, amphiboles, magnetite, pyrite, biotite and garnets. The bulk of the iron released during

weathering is converted to iron oxides, which are relatively insoluble and stable (Matthess *et al.*, 1982), with only a small amount being transported by surface and ground water. The most common type of iron in solution is ferrous iron (Fe^{2+}) which is usually found only in minor concentrations. When the ground water comes into contact with the atmosphere, most of the iron precipitates out:

$$2Fe^{2+} + 4HCO_3^- + H_2O + {}^1/_2O_2 \Leftrightarrow 2Fe(OH)_3 + 4CO_2$$

Therefore, high dissolved iron concentrations, of between 1 and 10 mg/l, are usually found under reducing conditions. In landfills, where reducing conditions dominate, iron concentrations range between 90 and 2200 mg/l in the early stages of a landfill, but decrease due to the complexation of metal species (Pohland, 1986). Once a landfill has stabilised, concentrations of between 4 and 20 mg/l can be expected.

At pH values less than 3, the predominant type of iron is ferric iron (Fe^{3+}). At pH values greater than 3, ferric iron often forms complex ions with chloride, fluoride, sulphate and phosphorous ions, as well as with organic substances (Davis *et al.*, 1966).

Aluminium

Although aluminium is one of the most abundant elements in the Earth's crust, it is one of the least mobile in the hydrosphere. Water with a pH range of between 5.0 to 9.0 will generally have an aluminium concentration less than 1.0 mg/l, with normal ground water ranging between 0.005 and 0.3 mg/l. In landfill leachates, concentrations range from values too low to determine, up to values as high as 85 mg/l (O'Leary *et al.*, 1986).

Lead

Lead is found at very low concentrations in rocks of the Earth's crust, except in lead-rich mineral deposits where it is present in the minerals galena, cerussite and anglesite. It has also been found to replace the potassium in silicates, mostly in feldspars and phosphates (Matthess *et al.*, 1982). Lead has a low solubility and geochemical mobility, and is usually found in concentrations less than 20 µg/l. It is used in both pesticides and fertilisers, and is often present in solid and liquid waste, where concentrations of up to 1.44 mg/l have been found in the landfill leachates, although these values tend to decrease with time as they form complex metal species (Pohland, 1986).

Manganese

Manganese is geochemically similar to iron although it is usually found at lower concentrations due to its low geochemical abundance. Manganese encourages the growth of bacteria which tend to stain bathroom fixtures and laundry (Davis *et al.*, 1966). It is a relatively minor constituent in igneous rocks, but is widely distributed in sediments and soils as oxides and hydroxides, and is derived mainly from the weathering of these minerals. At a pH of around 7, the dominant manganese ion is Mn^{2+}, which is present in concentrations of less than 0.2 mg/l in most natural water. Values in landfill leachates are generally around 0.6 mg/l, although concentrations do reach values of over 40 mg/l (Pohland, 1986).

3.4.3 Graphical Techniques

The graphical techniques used to plot the chemical data in this study are the Schoeller diagram and the Piper diagram.

The Schoeller diagram is a plot of the major ion components of Ca, Mg, Na+K, Cl, SO_4 and HCO_3+CO_3, which are plotted on individual vertical logarithmic scales (Brassington, 1990). The concentrations of each ion are plotted along the logarithmic scale in milliequivalents per litre (meq/l). The Schoeller diagram is used to compare the absolute major ion concentrations. Although the diagrams in this study have been referred to as Schoeller diagrams, the sequence of ions has been somewhat modified in order to clearly represent the primary tracers of any potential pollution (see eg. Figures 9a and b).

The Piper diagram is a plot of the ionic ratios on a trilinear graph. This allows for more than two parameters to be plotted as one point in a field once the data has been converted to meq/l (see eg. Figure 8). However, these are only relative concentrations and not absolute concentrations. The individual ionic groups are expressed as a percentage of the sum of the cations and anions respectively, and then plotted in there respective ionic fields. the points in the two fields are then transferred to the diamond field, where they are plotted at the point of intersection. This characterises the whole analysis by one point. This graphical method can indicate trends in ground water quality.

3.4.4 Reaction Errors

In any solution the sum of the cations will equal the sum of the anions. However, this is not always true for the results received from the laboratory. In order to assess the quality of the data and determine the accuracy of the complete chemical analysis, the cation-anion balance should be calculated. This is called the reaction error, and is determined as follows (Mazor, 1991):

$$\text{reaction error} = \frac{\sum_{\text{cations}} - \sum_{\text{anions}}}{\sum_{\text{ions}}} \times 100$$

where the reaction error is expressed as a percentage of the total ion concentration. The cations and anions must first be converted to milliequivalents per litre. A positive reaction error indicates a cation excess, while a negative result indicates an anion excess. The error will generally not exceed 2 %, and reaction errors greater than 5 % are unacceptable. The cause of the errors are usually due to either analytical errors of the individual parameters, or due to the fact that not all the ions are measured. Another reason the results may not balance is that in strongly coloured solutions, as is often the case in samples from landfil sites, organic anions form complexes with metals (Hem, 1975).

CHAPTER 4

4 THE LINBRO PARK LANDFILL SITE

4.1 Introduction

4.1.1 Location

The Linbro Park landfill site is located on the north-eastern boundary of the Johannesburg municipal area, within the Sandton municipal district (Figure 4). It is situated on part of Portion 16 of Lombardy No. 36-IR in a borrow pit from the construction of the N3 highway, which borders the site to the west.

A broad shallow valley separates the northern and southern sections of the landfill, sloping from Third Road, in a westerly direction, down to the N3 highway (Figure 5). The northern section of the site slopes in a south-westerly direction with a slope of about 2.5°, while the southern section slopes in a north-westerly direction with a slope of 4.0°. The landfill lies above the water table on weathered granites of the Johannesburg-Pretoria granite dome.

4.1.2 History and Development

The initial stage of landfill of the Linbro Park landfill site was initiated in 1969 in the northern section of the site. Although a preliminary hydrogeological investigation was

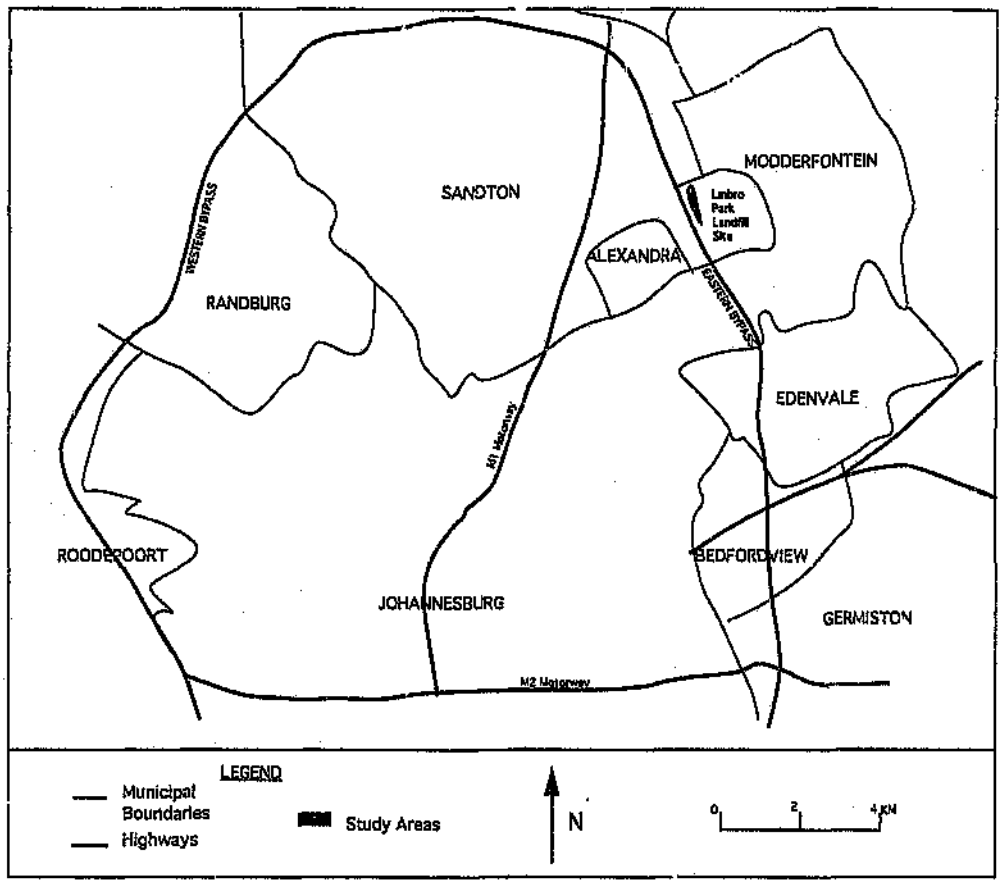

Figure 4. Map showing the locality of the Linbro Park landfill site

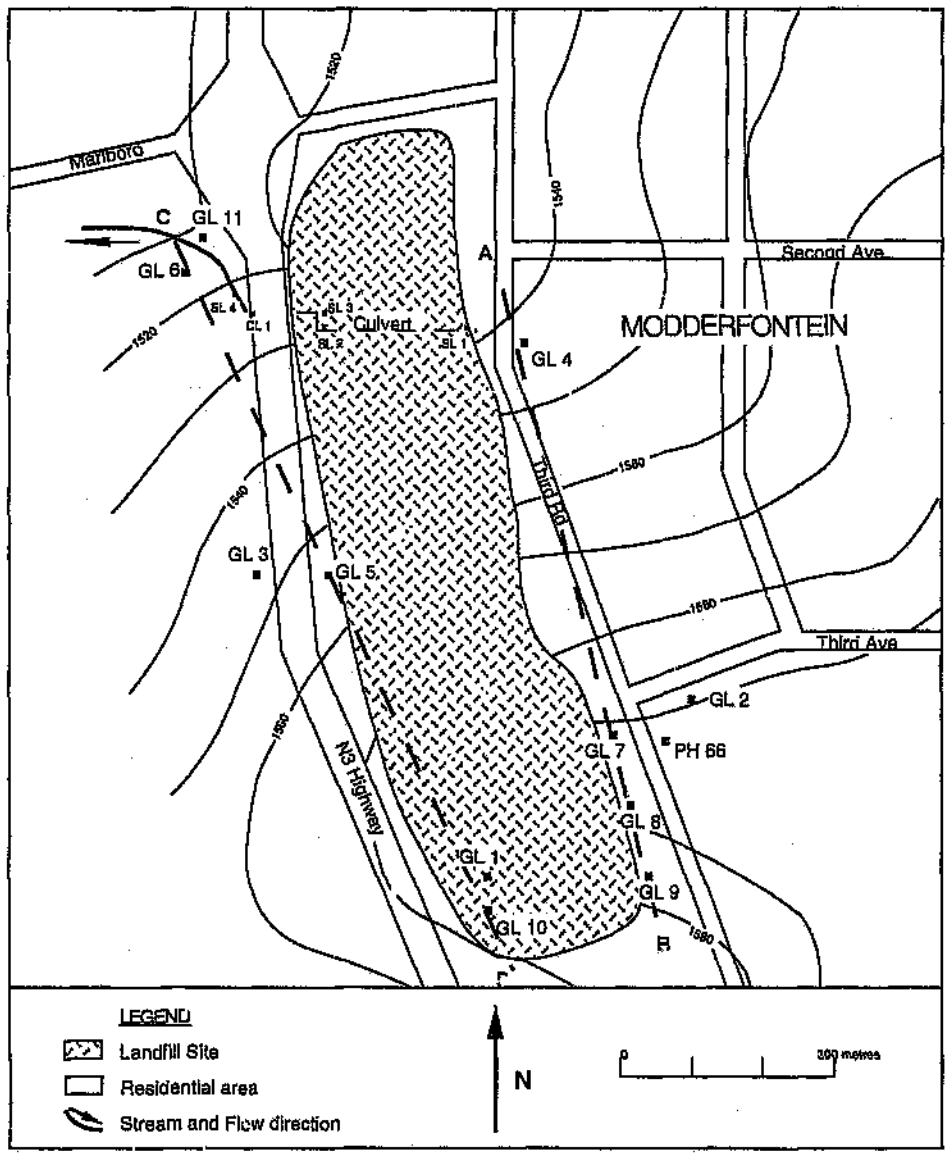

Figure 5. Location of the ground and surface water pollution monitoring points for past and present studies at the Linbro Park landfill site.
Shown also are sections A-B and C-D in Figure 7.
Surface contours in metres above sea level

not carried out in this part of the site, and a lining was not installed (Blight, 1992), the Johannesburg City Council maintain that the sanitary landfill standards were followed. This stage of the landfill site has been completed, and the second stage, in the southern section, was started in 1989 and is currently underway. In this southern section, the water table is approximately 4 metres below the base of the landfill, except in the proximity of the stream where it approaches to within 2 metres of the base (Williams *et al.*, 1983).

The site serves the Johannesburg, Sandton and Alexandra municipalities, with a calculated refuse input of approximately 220 thousand tons per annum. The refuse is mainly urban wastes, as toxic, hazardous and liquid wastes are not accepted at the site which is classified as a G:M site (see Chapter 1.1). The greater part of the waste is obtained from bulk containers, made up of domestic waste, builders rubble and soil, as well as refuse from gardens, streets and stands. A very small percentage of non-hazardous industrial waste is also present (Hojem, 1988). In 1992, it was estimated that the Linbro Park landfill site had another 10 to 15 years of life remaining (Blight, 1992).

The refuse is placed between two parallel sand or rubble berms situated about 30 metres apart, compacted into 2 metre high cells, and covered with a 150 to 300 mm layer of soil at the end of each working day (Williams *et al.*, 1983). The refuse is then compacted using a landfill compactor. The soil which is excavated from the site is used as the cover material. A 600 mm thick soil layer of decomposed granite is placed on top of the cells as a final layer which is then vegetated.

4.2 Geology

4.2.1 Regional Geology

The Linbro Park landfill is situated on weathered granites of the Grey Granodiorite Suite, which are a part of the Johannesburg-Pretoria granite dome (Figure 6). This dome is an ancient granite-greenstone basement exposed on the Kaapvaal Craton, which covers an ovoid area of approximately 700 square kilometres. A number of greenstone remnants are found on the dome, similar to the those of the lower Onverwacht Group of the Swaziland Supergroup in the Barberton Mountain Land (McCarthy, 1977), as well as a wide variety of granitic rocks which cover the greater part of the dome (Anhaeusser *et al.*, 1982; Tankard *et al.*, 1982).

The granite found at the Linbro Park landfill site is a homogeneous, medium-grained grey granodioritic to adamellitic granite which covers most of the centre of the southern half and eastern regions of the dome, and is one of three types of granite in the Grey Granodiorite Suite (Anhaeusser, 1973).

4.2.2 Local Geology

Geological outcrops are not exposed in the Linbro Park landfill site, although large, round boulders of granite are present. Outcrops of the grey granodiorites are found to the west of the N3 highway, as well as along the banks of the Jukskei river. The major minerals of the grey granodiorites are quartz, orthoclase and microcline, with minor amounts of biotite, muscovite, epidote and sphene (Anhaeusser, 1973). On some of the weathered surfaces feldspar phenocrysts are visible, and occasional weak banding and small mafic xenoliths have been observed (Anhaeusser, 1973). Pegmatites occur fairly often in this suite, and are prominent in the granite outcrops in the Jukskei River to the west of the site. These granites also display narrow jointing and cracks.

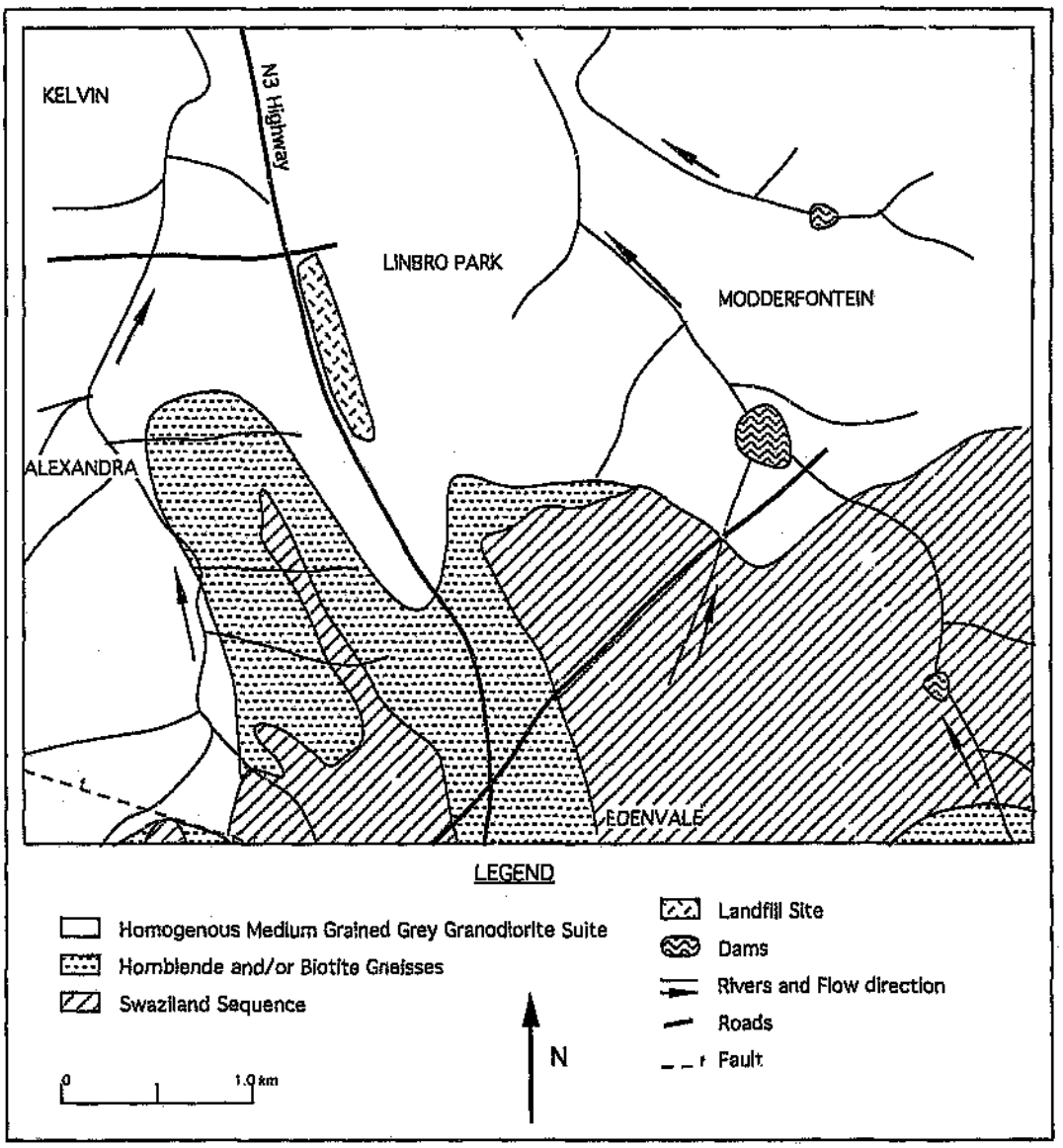

Figure 6. Map of the main geological features of the Linbro Park study area

The City Engineers Department (Williams *et al.*, 1983) located the residual granite in all the trial holes sunk for the proposed southerly section. It ranges in thickness from 2m near the stream up to 12m in the north-western section of the site. This material is silty medium to coarse sand and is a light red colour with mottlings of green, yellow and pink. Some micaceous layers, as well as lenses of grey clayey sand, were also present. Fissures and joints were found to be closed in this residual granite.

Most of the site is covered by a 1 metre layer of orange-red hillwash material, except where it has been replaced by alluvium along the river. The hillwash material is silty sand with fine quartz grains, while the alluvium is red clay derived from the weathered granites and varies between 0,9 and 2,5 metres in thickness (Williams *et al.*, 1983). A hardpan ferricrete layer of about 0,5m was observed in two of the trial holes in the vicinity of the stream. This layer is similar to the alluvium, although it only covers a small portion of the site.

4.3 Geohydrology

Well yields are generally very low in solid, unweathered granitic rock as any pores which are present are small and usually not interconnected, frequently resulting in porosities of less than 1%. However, significant porosities and permeabilities may be developed through weathering and fracturing of the rock. The permeability decreases rapidly with depth as joints, faults and other fractures close due to the weight of the overlying material, and any surface disturbances which may occur only penetrate a short distance into the bedrock.

Differences in well yields in an area are large, and are usually the result of differences in the degree of weathering or fracturing of the rock, rather than differences in

mineralogy or texture and structure within the rock (Davis *et al.*, 1966). Boreholes situated in fracture systems often have high initial yields which decrease rapidly as a result of insufficient storage of ground water near the well. It is therefore likely that monitoring boreholes in fractured rock will not give representative results of the ground water in the area, as only the immediate area will be sampled and very little water will be extracted from the surrounding area. It is therefore important, where possible, to locate boreholes in weathered zones or in alluvium where the greatest permeability is found.

The chemical quality of ground water in granitic terrains is almost always excellent, although it has been found to be slightly acidic (Matthess *et al.*, 1982; Mazor, 1991). The slow breakdown of the major minerals in granitic rocks generally results in a low concentration of solutes in the ground water. The characteristic composition of ground water in granitic rocks is a very low mineralisation of around 300 mg/l, the major anion is bicarbonate, Ca^{2+} and Na^{+} are the major cations, and has a pleasant taste. The typical chemistry of such ground water is given in the following table.

Table 1. Chemistry of typical ground water from granitic terrains (in mg/l) (Matthess *et al.*, 1982)

	Minimum (mg/l)	Maximum (mg/l)	Average (mg/l)
Ca^{2+}	1.8	12	5
Mg^{2+}	0.6	4.5	2
$Na^{+} + K^{+}$	1.3	14	7
HCO_3^{-}	12	80	35
SO_4^{2-}	0.1	23	4
Cl^{-}	1.1	17	4
F^{-}	n.d.	0.9	0.1
NO_3^{-}	n.d.	14	2.9
pH	5.8	7	

Boreholes in the granites of Linbro Park have low water yields, but the water appears to be fairly pure, and has been found to be suitable for most uses (Beaumont *et al.*, 1987). According to Williams *et al.* (1983), the residual granite which overlies the parent rock has a low permeability as it is fairly compact and, as a result, transmits water largely by fissure flow. The fracture systems are fairly complex, and the permeability of the joints and cracks was found to decrease rapidly below 40m.

The water table generally follows the surface topography from the river to the western boundary of the site. Localised, perched water tables were observed in the residual granites in the vicinity of the stream and result from infiltrating water being obstructed either by clay pockets or the hardpan. These perched water tables are not interconnected, and this water often appears on the surface in the form of springs (Williams *et al.*, 1983).

Several holes sunk into alluvial sediments collapsed, namely GL 4, 10 and 11, and it was also established that tree roots pierce the casing, preventing access to the hole, as was the case in GL 3. These casings are obviously not durable enough to outlive the landfill sites, and continued monitoring of the completed site is not possible.

The rest levels and depths of the monitoring boreholes were measured using a twin-core cable with an amp meter, mounted on a hand-winch (commonly known as a "dipper"). Four measurements of the rest levels of both the up-gradient and down-gradient boreholes were taken, the dates and measurements of which are shown in Table 2. Rest level measurements for boreholes GL 3 and 4 were not obtained in 1994 as obstructions prevented the cable from being lowered down the borehole. The data in Table 2 shows that the boreholes generally have lower rest levels in September and October of 1992, after the sampling exercise in June of 1992. This is possibly an

indication of the length of time these boreholes in the granites take to recover after sampling. The rest level of borehole GL 6 remains fairly constant as it is located in alluvium alongside a river, and therefore recovery is rapid. A significant decrease in the volume of standing water is seen in the measurements taken in January of 1994. It is not known whether these boreholes had been sampled just prior to these measurements being taken as it was expected that the rest levels would be at their highest in the middle of the rainy season. The only boreholes where this is the case is GL 6 which is 2 metres higher than in previous measurements, probably due to a higher recharge rate as a result of its location in alluvium, as well as its proximity to the river, and borehole GL 7 which is 1.1 m higher, possibly the result of surface run-off from the landfill.

Table 2. Rest levels and borehole depths of the pollution monitoring boreholes

Borehole Number	Borehole Depths (m)		Rest Levels (below surface) (m)				
	initial	02.06.92	initial	02.06.92	24.09.92	14.10.92	14.01.94
GL 3	30	19.0	9.8	13.0	10.4	14.7	
	60	55.0	6	7.0	7.9	8.0	
GL 5	71	69.8	8	15.5	16.2	16.3	17.3
GL 6	90	85.0	1	3.0	3.1	3.0	1.1
GL 7	44	44.0	22	25.0	28.8	29.0	27.9
GL 8	23	23.2	18	11.7	13.1	13.2	14.3
GL 9	23	23.4	5	6.9	8.2	8.3	8.9

From the borehole depth measurements shown in Table 2 it is clear that by June of 1992 borehole GL 3 had already partially collapsed, and GL 4 and 6 had substantial decreases in their depths, most likely the result of silting. The eventual blockage of GL 4 is believed to be due to vandalism as the borehole was inappropriately covered.

Ground water flows in a westerly direction from the landfill towards the Jukskei River. There is also a possibility that a certain proportion of the ground water flows from the

site in an easterly direction towards some of the privately owned boreholes along Third Road (Williams *et al.*, 1983).

An inferred large drawdown at the highest elevation on the eastern side of the landfill site, in the vicinity of borehole GL 7 and the Linbro Park residences, is depicted in the water table transect in Figure 7a. The rest level of borehole GL 7 is around 29 metres below the surface. The boreholes at a similar elevation on the western side of the site, along the N3 highway (Figure 7b), have rest levels from around 10 metres up to slightly over 16 metres below the surface (Table 2), and follow the surface topography. The large drawdown is most likely the result of the pumping of private boreholes to the east of borehole GL 7, creating a depression cone below the site, possibly reducing the threat of pollution to the ground water by the landfill site. However, this tends to increase the risk of drawing in any contaminated water towards the private boreholes.

As the site was originally traversed by a stream, a culvert was constructed along the length of the channel, which is now beneath the landfill. This stream flows westwards, from the site, into the Jukskei River. The general direction of the surface water runoff is in a north-westerly direction, obliquely across the site towards the stream. This stream also channels water flowing down Third Road from both the north and south.

4.4 Previous Studies

Monthly surface water sampling and analysis undertaken at Linbro Park was initiated in 1978 by the Johannesburg municipality in the culvert beneath the site, at CL 1 (Figure 5), as well as in the stream, at SL 1, SL 2 and SL 3 (Theron, 1986). Discharge from the weepholes in the culvert was sampled from 1980, the analyses of which suggested mineralised leachate from the landfill. In 1989, a further stream sampling point, namely SL 4, was established further down-gradient of the landfill in order to determine

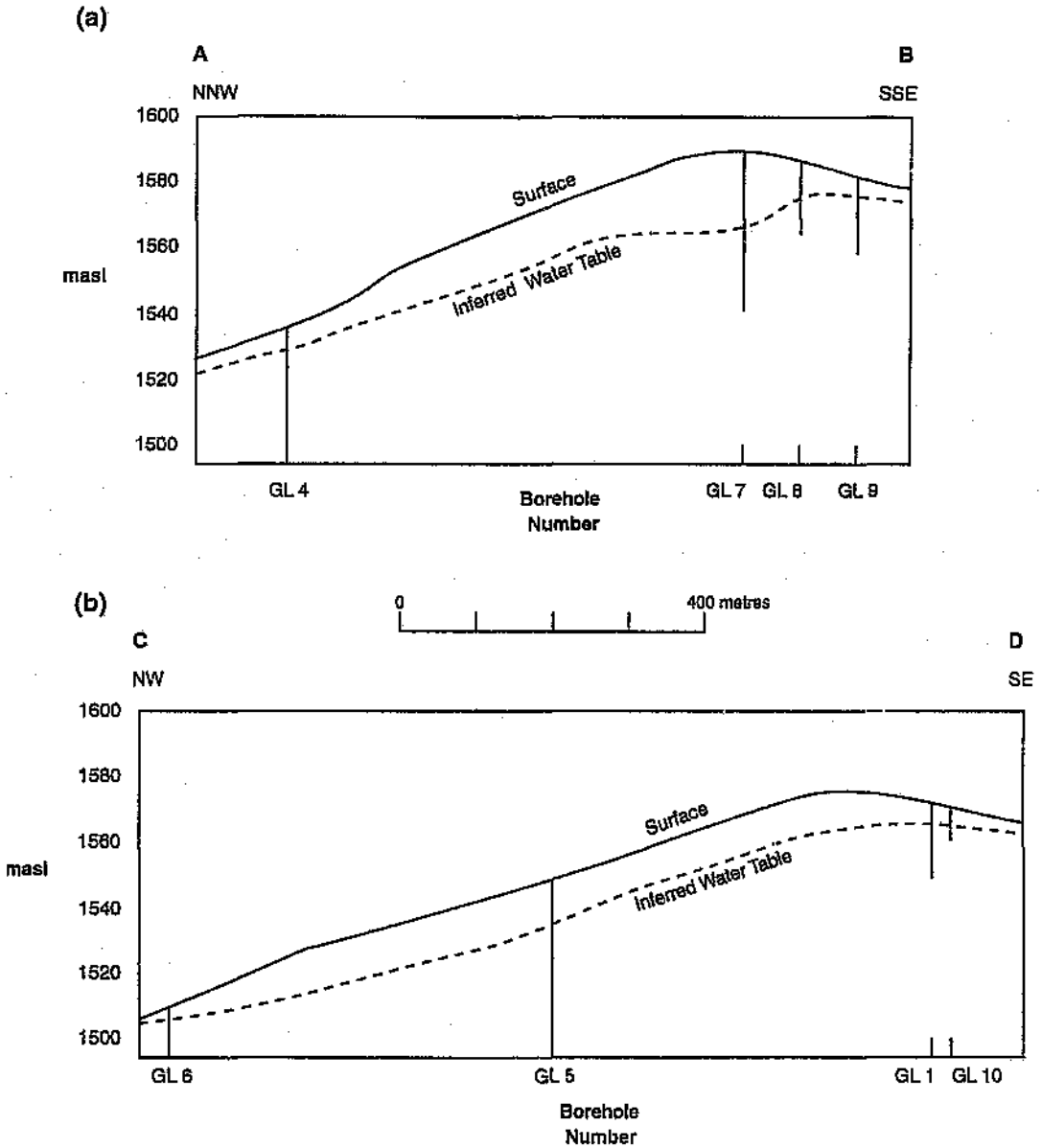

Figure 7. Water table transects, using initial rest levels, for:

a) the eastern section A-B and

b) the western section C-D.

Sections A-B and C-D are shown in Figure 5. Note inferred depression cone at borehole GL 7 in section A-B

whether the quality of the stream improved away from the site. In 1990, sampling of the stream which crosses the new southerly extension of the site was initiated.

Since 1979, a number of boreholes have been drilled at the Linbro Park landfill site for various reasons. These have been summarised in Table 3. The locations of these boreholes are illustrated in Figure 5. The positions of the trial boreholes TP 1 to 14 are not known. The pollution monitoring boreholes, GL 1 to 11 were drilled through alluvium and decomposed granite, and have been cased with UPVC casing.

Williams *et al.* (1983) of the City Engineers Department claim that, because of the low permeability of the residual and hard rock granite, and the high evaporation rates in the area, leachate should not pollute the ground water. This claim is largely based on the following considerations: the excavations have not been made too deep; the floor of the landfill has been compacted and slopes east to west at about 1:30; a subdrain was constructed along the western boundary of the site to drain off any potential leachate to the culvert; and the refuse is adequately covered at the end of each day.

Due to allegations of polluted ground water by the residents of Linbro Park, a study was undertaken by Hill Kaplan Scott Incorporated in 1987 (Beaumont *et al.*, 1987). The tests included chemical and micro-biological analyses of ground water samples taken at 19 borehole sites, and 2 samples of surface water, both up-gradient and down-gradient of the refuse dump. The results showed minor pollution of the stream flowing through the culvert and, as this stream flows away from the residential area, it was argued that it would not present a problem to the private boreholes. Other than high nitrate values attributed to septic tank effluents, no chemical pollution of the ground water at the small holdings was found. Micro-biological tests indicated pollution of the ground water by bacteria, but this was also attributed to septic tanks, as well as to animals from the nearby residences. From this study, McLachlan and Lazar, a firm of

industrial chemists, concluded that there was inadequate data to link any ground water pollution to the site.

Table 3. Summary of the monitoring boreholes drilled at Linbro Park landfill site (Williams *et al.*, 1983; Theron, 1986; Pollution Monitoring (Leachate) Committee, 1990)

Borehole Number	Date Drilled	Initial Depth (m)	Initial Rest Level (m)	Purpose of Boreholes	Current Status
EL 1	1979	11	dry	}	}
EL 2	1979	12	dry	} Depth of	}
EL 3	1979	11	10 m	} decomposed	} Filled
EL 4	1979	6	dry	} granite	}
EL 5	1979	8	dry	}	}
EL 6	1979	4	dry	}	}
EL 7	1979	25	8	}	}
EL 8	1979	19	10	} Monitor water	} Destroyed
EL 9	1979	30	17	} table depths	}
EL 10	1979	35	19	}	}
TP 1	May 82			} Depth of	} Filled
to				} rippable	} except for
TP 14				} material	} TP 8 &14
PL 1	1982	25	14.8	} Monitor perched	} Destroyed
PL 2 (TP 14)	1982	9	dry	} water	}
GL 1 (TP 8)	1982	11	7.8	}	Destroyed
GL 2	1982	30	25.7	}	Dry
GL 3	1982	30	9.8	}	Collapsing
GL 4	1987	60	6	}	Collapsing
GL 5	1987	71	8	} Monitor water	}
GL 6	1987	90	1	} pollution	}
GL 7	1989	44	22	}	} Functional
GL 8	1989	23	18	}	}
GL 9	1989	23	5	}	}
GL 10	1989	9	2	}	Collapsed
GL 11	1989	11	1	}	Collapsed

Hojem (1988) submitted an MSc dissertation dealing with a method for predicting the flow of leachate in a landfill. Rainfall, evapotranspiration, runoff and infiltration were measured, and two 1.2 metre diameter auger holes were drilled through the refuse to check predicted percolation at the Linbro Park landfill site. From the results, Hojem claims that the fill had not reached its field capacity and so no leachate was percolating through the base. He estimated that the landfill would start producing leachate from about the year 2001.

In 1992, Blight completed an MSc dissertation on the feasibility of designing top covers of landfills to eliminate or minimise leachate production at the Linbro Park landfill site. A number of cover designs, cover materials, and their advantages and disadvantages are discussed by Blight in the study. From data gathered in the two auger holes and the monitoring boreholes, Blight was able to conclude that the landfill had not produced any leachate. Lateral movement of moisture within the landfill is expected, as the TDS concentrations similar to those found within the refuse were not found at the base of the landfill. This is thought to result from the compaction of the refuse in sloping layers.

At present, borehole and surface water is monitored every three months by the site staff. Results from the monitoring of the boreholes at Linbro Park site indicate that leachate has not yet entered the ground water system and, according to the City Council, this condition has not changed in the last three years. However, sampling of the pollution monitoring boreholes is carried out using the open-bailer method. Only the top of the standing water column is sampled, and borehole yields are extremely low (see 5.5.1). The results of this operation are therefore at best debatable.

4.5 Results of the Present Study

4.5.1 Introduction

This study has involved the location of both the up-gradient and down-gradient pollution monitoring boreholes at the Linbro Park landfill site, as well as determining whether it was possible to retrieve samples from these boreholes. The functional monitoring boreholes, namely boreholes GL 3 to 9, were sampled twice in 1992. The first set of samples was taken in the dry season in early June, and the second after considerable rain had fallen in October. Boreholes GL 7, 8 and 9 are considered to be up-gradient of the landfill site, while boreholes GL 5 and 6 are down-gradient (Figure 5). Boreholes GL 3 and 4 appear to be in an intermediate position. For comparison with the water within the residential areas, a private borehole situated about 300 metres east and up-gradient of borehole GL 7, at number 66 Third Road, Modderfontein, was sampled (Figure 5). This borehole, designated PH 66, is claimed by the owner to have been drilled to a depth of 115 metres into the granites.

The pump used to sample these pollution monitoring boreholes is a small diameter submersible pump, made available by the Earth and Environmental Technology Division of the Atomic Energy Corporation. This was the only submersible pump that could easily be lowered down these boreholes as the diameter of the casing was too small to allow a standard submersible pump access. The volume of water standing in each borehole was first purged, and then the samples were taken, which are believed to be the most representative samples of the ground water possible under the given poor conditions. The only borehole not fully purged was borehole GL 3 which had partially collapsed, and only two 1 litre samples could be extracted. The boreholes were resampled around three months later when it was expected that the rest levels would have more or less recovered (Table 2). It was found at this time that borehole GL 3 had collapsed at a depth of around 10 metres and, as a result, no sample could be taken.

When the monitoring boreholes were pumped at around 500 litres an hour, it was found that the water levels were rapidly drawn down. The already low pumping speed had to be drastically reduced, and it was often the case that barely sufficient water was available for carbon-14 sample extraction (~ 100 litres). This is probably due to either the low permeability of the granitic rocks and the residual granite, or is the result of the casing not being slotted or obstructed slots in the casing. However, in spite of poor sampling conditions, it is argued that at least the water sampled had been extracted from outside the casing. The samples were analysed for the major ions and for the stable and radioactive isotopes. Well-head measurements of temperature, pH, conductivity and alkalinity were taken during sampling. The rest levels of each hole, as well as the depths of the holes, were also measured prior to sampling (Table 2).

4.5.2 Hydrochemistry

Well-head Measurements

The well-head measurements for the pollution monitoring boreholes are given in Table 4. Generally, the down-gradient boreholes (D) have higher alkalinity, conductivity, pH and temperature values than the up-gradient boreholes (U).

The up-gradient boreholes are lower in alkalinity, electrical conductivity and pH than the down-gradient boreholes (Table 4). The conductivity values are relatively low, as is expected for ground water from granitic terrains.

A decrease in the alkalinity and pH values over the two sampling dates for each borehole is most likely due to a chemically different water recharging the boreholes.

Table 4. Well-head measurements for the two sampling periods

Borehole	Alk (meq/l)		E.C. (mS/m)		pH		Temp (°C)	
Number	2 Jun	14 Oct	2 Jun	14 Oct	2 Jun	14 Oct	2 Jun	14 Oct
GL 3	1.0		8		6.1		19.0	
GL 4	1.7	1.5	16	22	6.4	6.0	20.0	19.8
GL 5 (D)	2.6	2.1		22	7.4	6.5	20.0	20.6
GL 6 (D)	2.6	2.4	32	32	6.6	6.0	18.5	20.9
GL 7 (U)	1.1	0.7	14	14	5.5	4.2	18.2	19.8
GL 8 (U)	1.2	0.9	9	12	6.3	5.2	19.0	19.9
GL 9 (U)	1.5	1.1	8	11	6.6	5.5	18.9	18.9
PH66 (U)	1.2		16		6.2		17.0	

(D) = Down-gradient boreholes

(U) = Up-gradient boreholes

Ions

The up-gradient boreholes are generally sodium bicarbonate dominant ground water, while the down-gradient boreholes tend to be calcium bicarbonate dominant (Tables 5a and b). The only exception is the October sample of GL 9 which tends towards calcium bicarbonate dominance. These differences between the up-gradient and down-gradient boreholes are further illustrated in the Piper diagram (Figure 8). Minor differentiation is observed in the anion triangle. In the cation triangle, the down-gradient boreholes have calcium percentages greater than 40% and Na + K less than 40%, while the up-gradient boreholes have calcium percentages less than 40% and Na + K greater than 40%. The only exception is the October sample of the up-gradient borehole GL 9, which approaches values of the down-gradient boreholes.

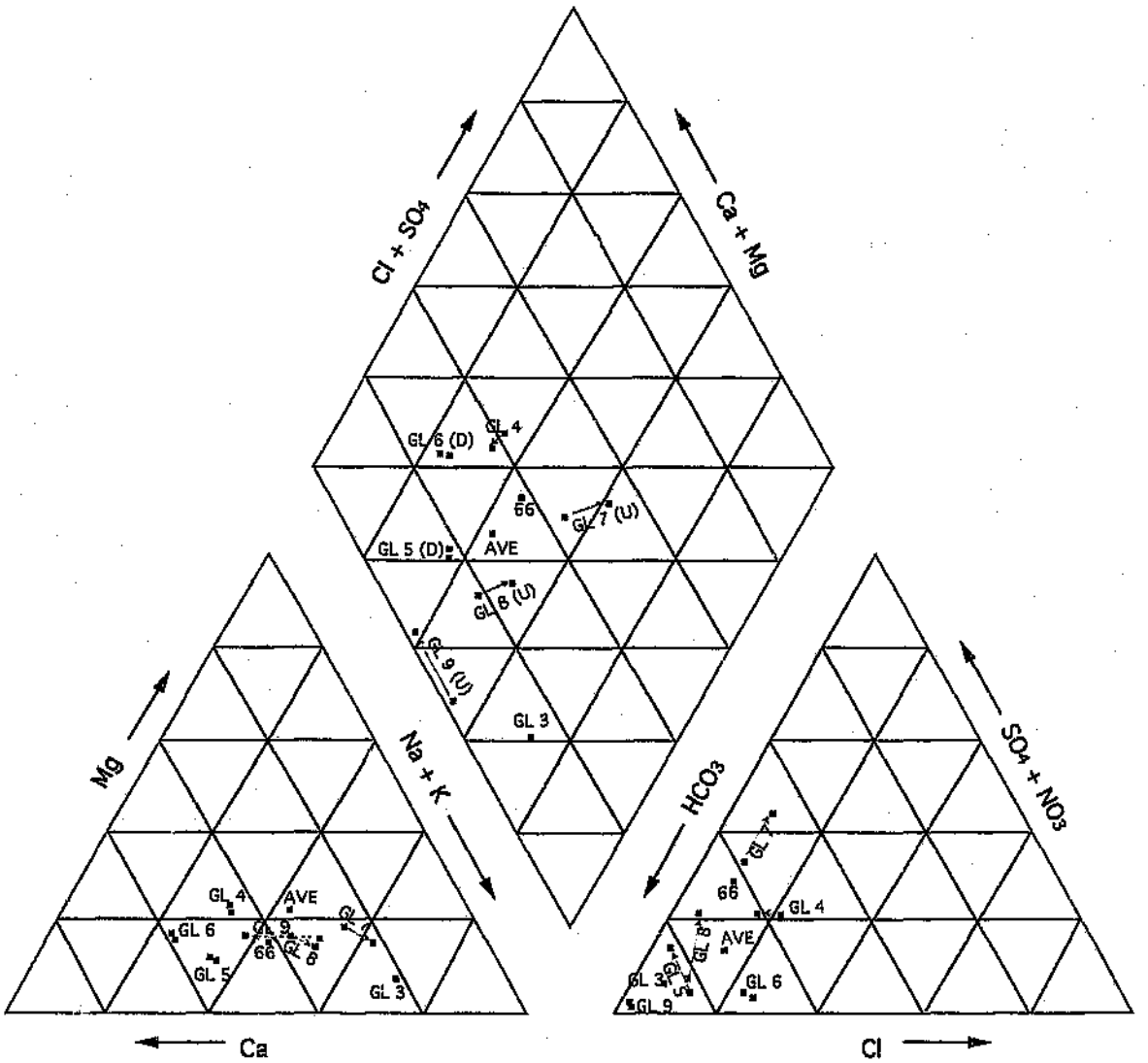

Figure 8. Piper trilinear diagram for the pollution monitoring boreholes at the Linbro Park landfill site. Shown also is the average chemistry of ground water from a granitic terrain (AVE) (Matthess, 1982)

Table 5a. Cation concentrations in meq/l for the two sampling periods

Borehole Number	K (meq/l)		Mg (meq/l)		Na (meq/l)		Ca (meq/l)	
	2 Jun	14 Oct	2 Jun	14 Oct	2 Jun	14 Oct	2 Jun	14 Oct
GL 3	0.03		0.08		0.70		0.2	
GL 4	0.04	0.04	0.51	0.51	0.7[illegible]	0.78	1.1	1.1
GL 5 (D)	0.01	0.01	0.31	0.31	0.9[illegible]	0.96	1.4	1.4
GL 6 (D)	<0.002	0.004	0.50	0.54	0.8[illegible]	0.91	1.9	2.1
GL 7 (U)	0.14	0.12	0.32	0.25	0.87	0.78	0.5	0.3
GL 8 (U)	0.12	0.10	0.23	0.19	0.57	0.52	0.6	0.4
GL 9 (U)	0.06	0.11	0.20	0.33	0.61	0.65	0.4	0.9
PH66 (U)	0.10		0.27		0.65		0.7	

Table 5b. Anion concentrations in meq/l for the two sampling periods

Borehole Number	HCO_3 (meq/l)		SO_4 (meq/l)		Cl (meq/l)		NO_3 (meq/l)	
	2 Jun	14 Oct	2 Jun	14 Oct	2 Jun	14 Oct	2 Jun	14 Oct
GL 3	1.0		0.05		0.09		0.02	
GL 4	1.6	1.7	0.25	0.25	0.54	0.51	0.29	0.31
GL 5 (D)	2.4	2.3	0.15	0.15	0.11	0.11	0.27	0.29
GL 6 (D)	2.6	2.8	0.12	0.12	0.85	0.99	0.06	0.06
GL 7 (U)	1.3	0.8	<0.03	0.01	0.17	0.16	0.69	0.73
GL 8 (U)	1.3	0.9	0.16	0.13	0.11	0.09	0.02	0.13
GL 9 (U)	1.5	1.5	0.01	0.02	0.05	0.04	<0.01	<0.01
PH66 (U)	1.1		0.03		0.15		0.48	

(D) = Down-gradient boreholes

(U) = Up-gradient boreholes

Table 5c. Total dissolved ion (TDI) concentrations in meq/l

Borehole Number	TDI (meq/l)	
	2 Jun	14 Oct
GL 3	2.2	
GL 4	5.1	5.2
GL 5 (D)	5.6	5.6
GL 6 (D)	6.8	7.5
GL 7 (U)	3.9	3.2
GL 8 (U)	3.1	2.6
GL 9 (U)	2.9	3.6
PH 66 (U)	3.5	

(D) = Down-gradient boreholes

(U) = Up-gradient boreholes

The higher calcium concentrations, and to a lesser extent, higher sodium and bicarbonate concentrations in the down-gradient boreholes are illustrated in Figures 9a and b, referred to as modified Schoeller diagrams. In the up-gradient boreholes, an obvious change in the Ca : Na ratio of borehole GL 9 is apparent and closely resembles the down-gradient boreholes.

Borehole GL 6 generally has higher major ion concentrations. Due to its proximity to the stream, it is possible that leachates, seeping through weepholes in the culvert into the stream, are entering the ground water in the vicinity of this borehole. The higher TDI concentrations observed in the down-gradient boreholes correspond to lower potassium concentrations. This could be the result of:

a) adsorption of the potassium ions by clay minerals during the migration of the ground water, or

b) that the downstream boreholes tap somewhat different flow lines.

Generally, the major ion concentrations in both the up-gradient and down-gradient boreholes are well within the recommended limit for drinking water (Table 6). However, the up-gradient borehole GL 7 does exhibit a slightly higher nitrate concentrations. This may be attributed to septic tank effluents from the neighbouring small holdings, although the higher levels are also probably partly due to the up-gradient boreholes being more oxic than those down-gradient. The recommended drinking water standard for nitrate is exceeded in the sample taken from borehole PH 66, with a nitrate (as nitrogen) level of almost 7 mg/l, while the maximum permissible limit of 10 mg/l is reached in borehole GL 7. The iron concentration in all the down-gradient boreholes is less than 0.1 mg/l, and within the standards for drinking water, while the iron concentration in the up-gradient boreholes, with the exception of borehole PH 66 and the June sample of GL 7, exceed the recommended limit. The aluminium concentration in boreholes PH 66, GL 9 and the October sample of GL 8 is higher than the recommended limit. Substantial increases in the phosphorous

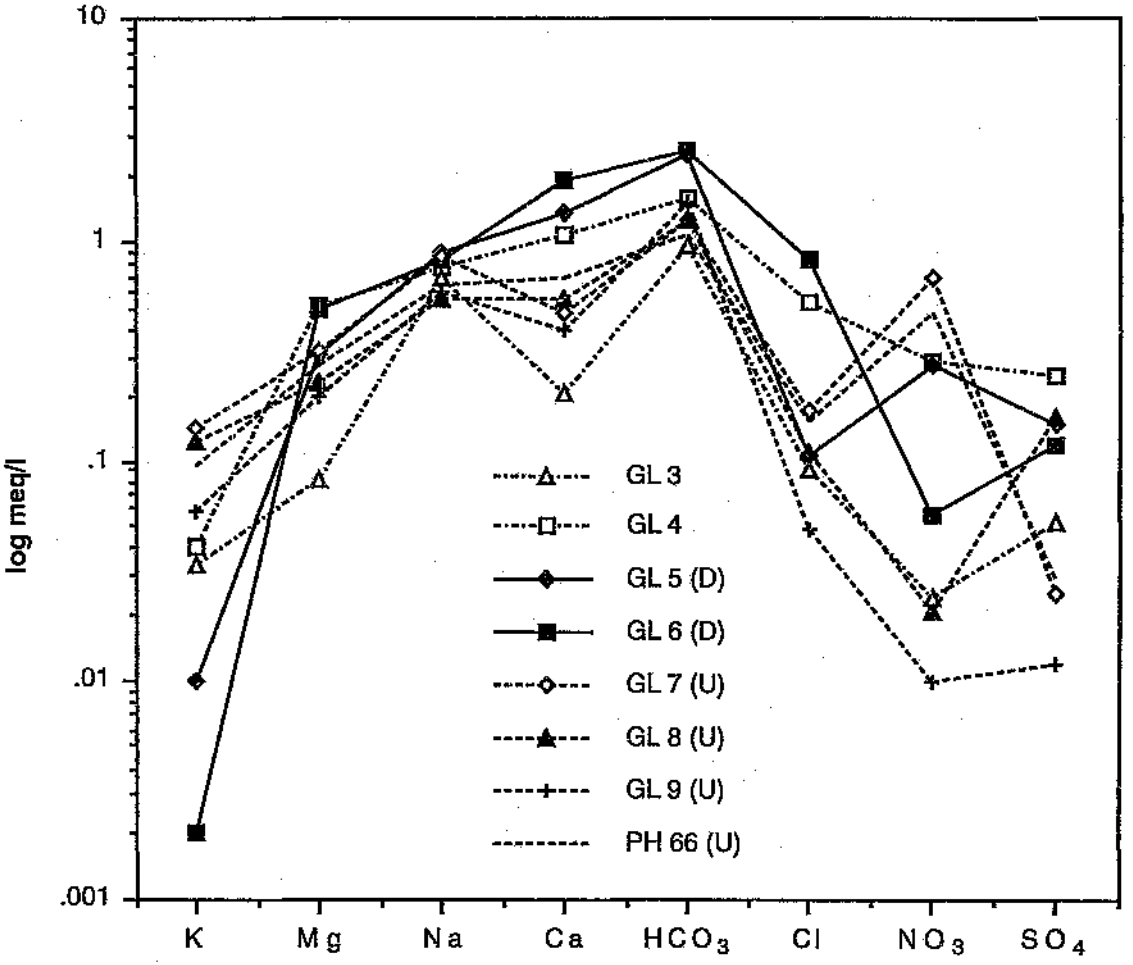

Figure 9a. Modified Schoeller diagram for samples taken on 02/06/1992

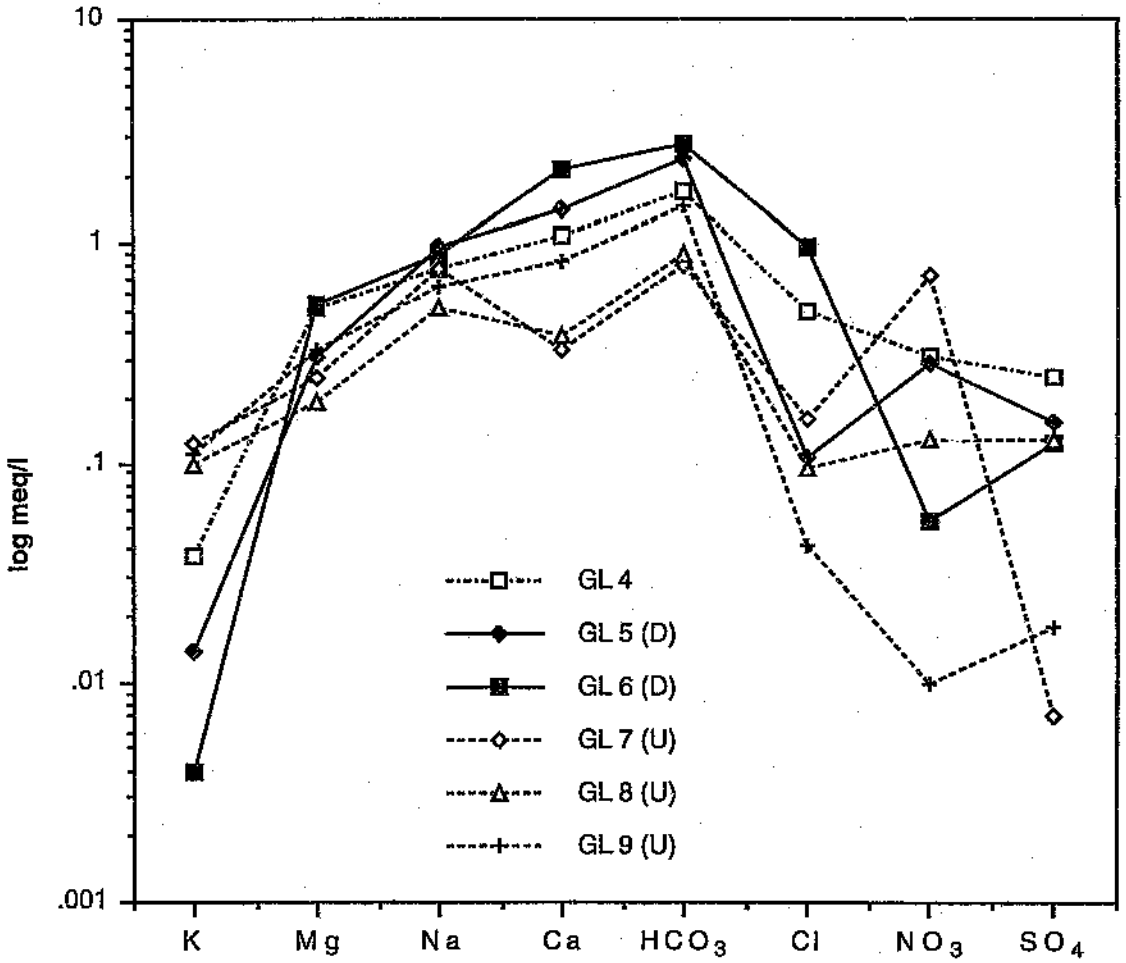

Figure 9b. Modified Schoeller diagram for samples taken on 14/10/1992

concentrations of boreholes GL 8 and 9 are apparent over the sampling period, both increasing from 0.02 to 0.11 and 0.15 meq/l respectively. Borehole GL 9 also shows significant increases in iron, from 0.36 to 21.00 mg/l, silica from 9.90 to 17.00 mg/l and aluminium, from <0.1 to 6.90 mg/l. This is the only instance of the appearance of these elements, assumed to be pollutants observed during this study, the concentrations greatly exceeding the crisis limit specified in Table 6.

Table 6. Drinking water standards as proposed for application in South Africa (Weaver, 1992)

Determinand	Recommended Limit	Maximum Permissible	Crisis Limit
K (meq/l)	5.1	10.2	20.4
Mg (meq/l)	5.8	8.2	16.4
Na (meq/l)	4.3	17.4	34.8
Ca (meq/l)	7.5	10.0	20.0
Cl (meq/l)	7.1	16.9	33.9
NO_3 (meq/l)	0.4	0.7	1.4
SO_4 (meq/l)	4.2	12.5	25.0
F (meq/l)	0.05	0.08	0.16
Fe (meq/l)	0.004	0.04	0.08
Al (meq/l)	0.02	0.06	0.11
E.C. (mS/m)	70	300	400
pH	6.0-9.0	5.5-9.5	<4.0->11.0
Temp (°C)	<25	<30	<40

Ground water chemistry from typical granitic terrain (Matthess, 1982) has been compared to both the up-gradient and down-gradient boreholes, showing the minimum and maximum values for granitic water (Figures 10a and b). The $Na^+ + K^+$ values are consistently higher than the maximum concentration of typical granitic water in both the up-gradient and down-gradient borehole samples, possibly the result of the high percentage of sodium-rich plagioclase (albite) found in the grey homogeneous granites.

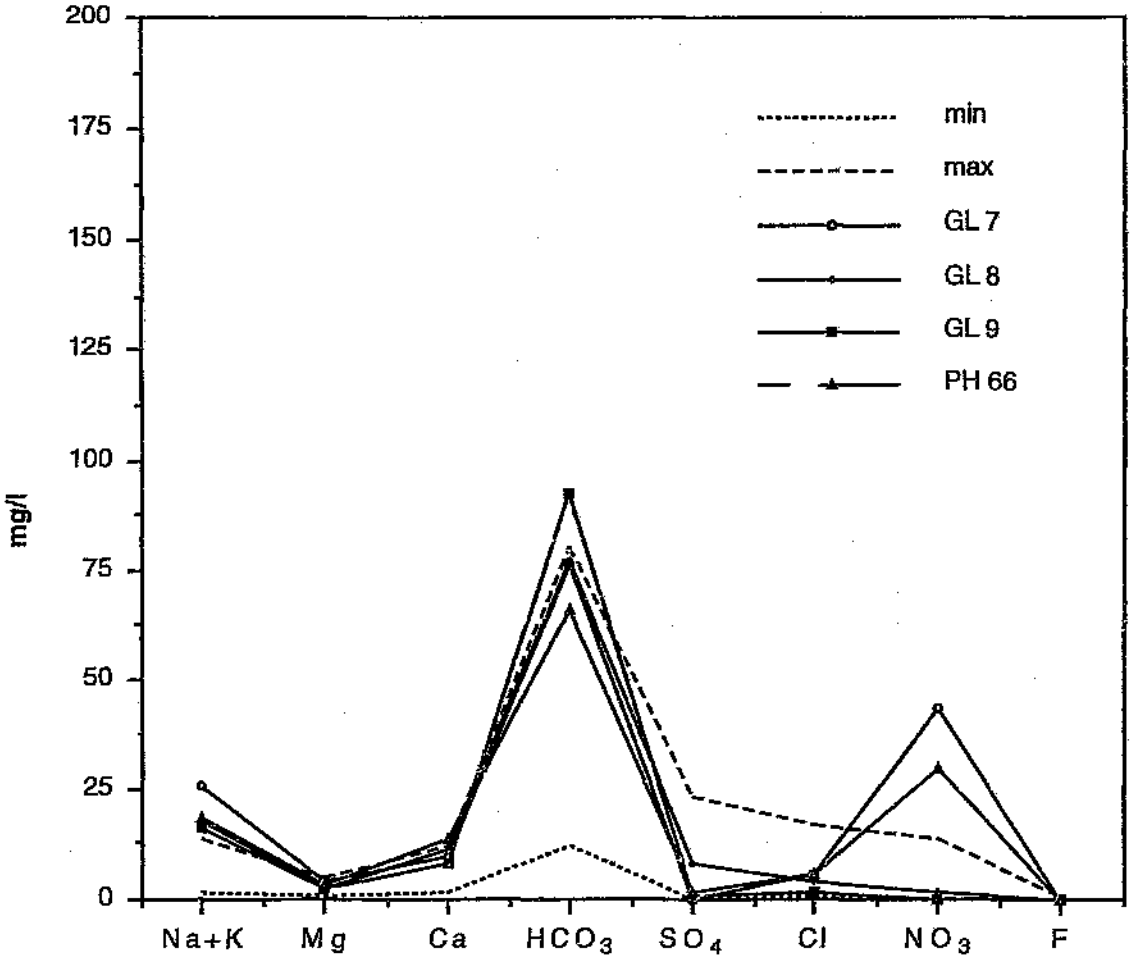

Figure 10a. Ion conceatrations in mg/l of the up-gradient pollution monitoring boreholes (02/06/1992) compared to the minimum and maximum concentrations of ground water in a granitic terrain (Matthess, 1982)

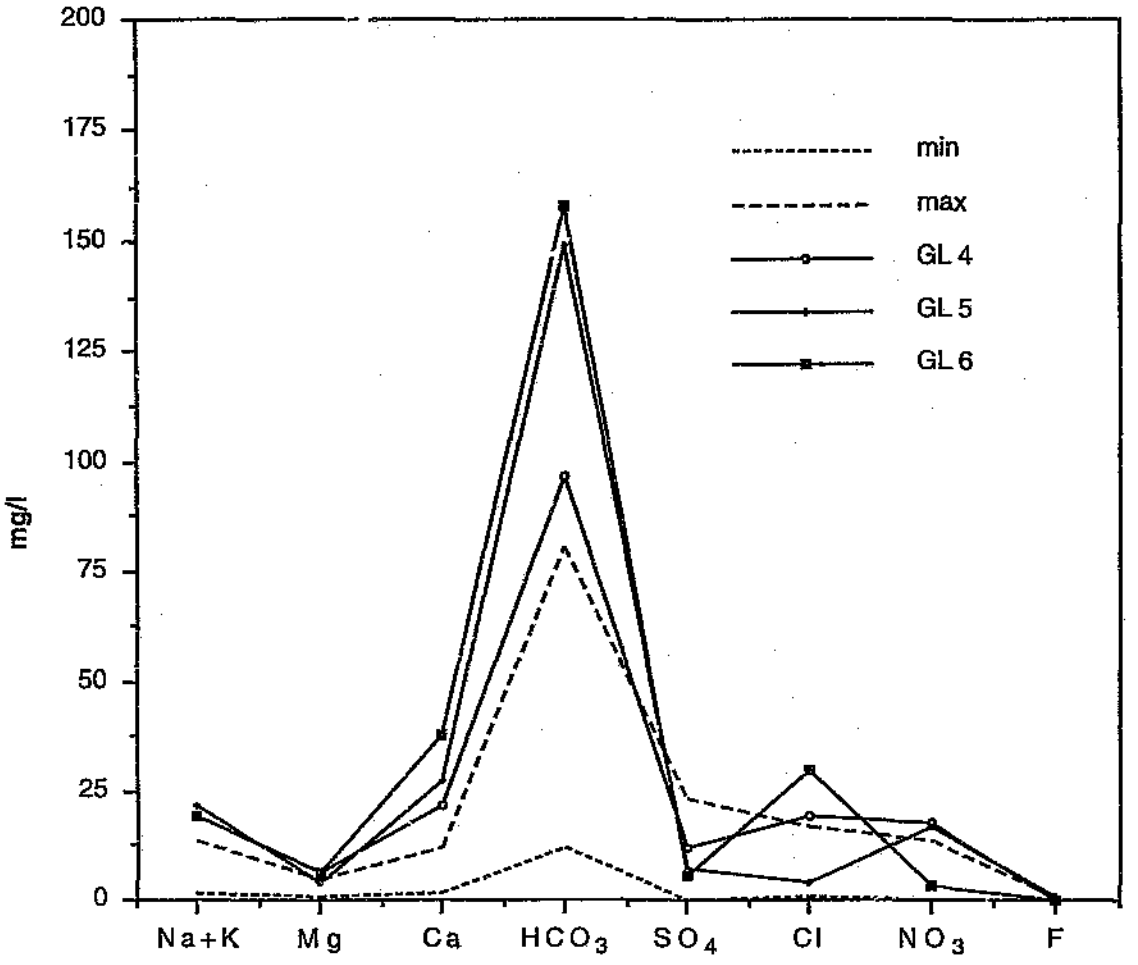

Figure 10b. Ion concentrations in mg/l of the down-gradient pollution monitoring boreholes (02/06/1992) compared to the minimum and maximum concentrations of groun water in a granitic terrain (Matthess, 1982)

as well as a higher degree of weathering of these granites. The chemistry of the pollution monitoring boreholes generally resembles the granitic water chemistry, and the concentrations are seldom higher than that of water solely from a granitic terrain (Figure 10a and b). The most obvious differences are seen in the anion concentrations, mainly in the up-gradient boreholes.

4.5.4 Isotopes

Stable Isotopes

The range of the δD and $\delta^{18}O$ values is fairly small for both the up-gradient and down-gradient boreholes (Table 7).

Table 7. Mean δD and mean $\delta^{18}O$ for the two sampling periods

Borehole Number	δD (‰)	$\delta^{18}O$ (‰)
GL 3	-21.3	-3.95
GL 4	-25.4	-4.45
GL 5 (D)	-24.0	-4.63
GL 6 (D)	-20.2	-4.07
GL 7 (U)	-25.0	-4.50
GL 8 (U)	-25.1	-4.58
GL 9 (U)	-28.3	-4.86
PH 66 (U)	-26.4	-4.50

(D) = Down-gradient boreholes

(U) = Up-gradient boreholes

The values of δD and $\delta^{18}O$ generally plot close to the meteoric water line, indicating that the samples are derived directly from rain, without any major modifications to their stable isotope compositions (Figure 11). As no significant evaporation is evident in both the up-gradient and down-gradient borehole samples, it is possible that a component of Rand Water (RW) mains water is present in the samples of boreholes GL

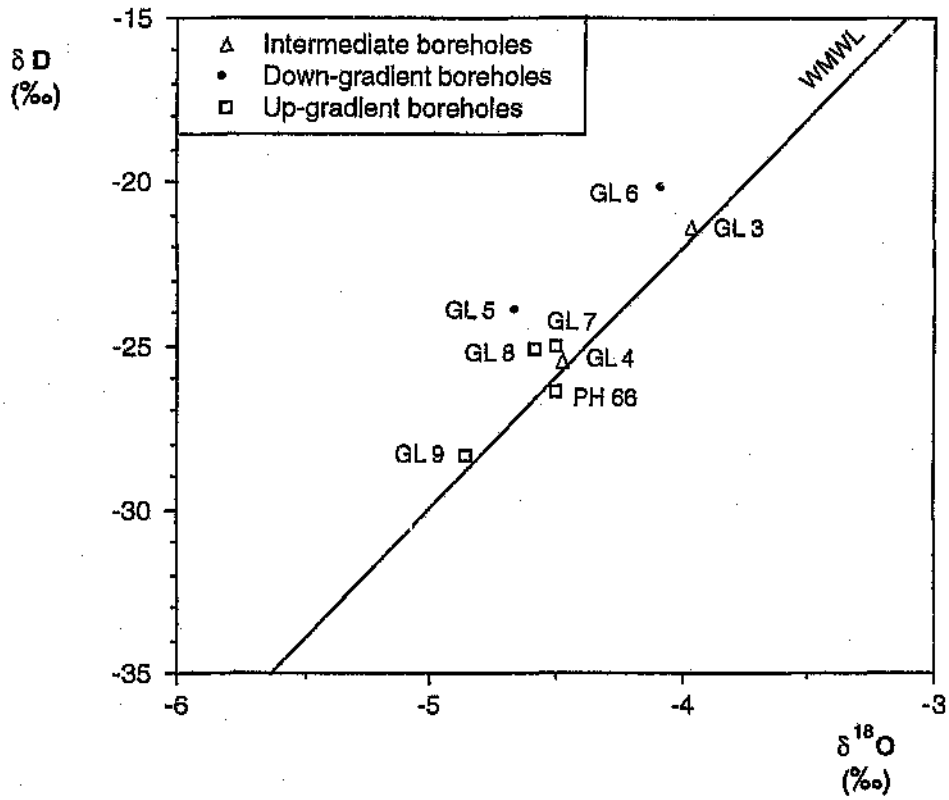

Figure 11. Plot of δ D against $\delta^{18}O$. Also shown is the world meteoric water line (WMWL)

3 and 6, entering the ground water by local infiltration with natural water from the stream (Verhagen *et al.*, 1995b). This would explain the more positive δ values seen in Figure 11.

In Figure 11, boreholes GL 5 and 6 show an enrichment in deuterium, but not $\delta^{18}O$, relative to the WMWL. It is clear that the enrichment is not due to evaporation, and therefore is probably the result of gaseous diffusion within the landfill.

Tritium and Carbon-14

Table 8. Radioactive isotope values for the two sampling periods

Borehole Number	^{3}H (TU)		^{14}C (pMC)		$\delta^{13}C$ (‰)	
	2 Jun	14 Oct	2 Jun	14 Oct	2 Jun	14 Oct
GL 3	4.6±0.4					
GL 4	1.2±0.2	1.4±0.3	86.2±0.8	91.3±1.6	-10.5	-10.8
GL 5 (D)	0.6±0.2	0.4±0.2	69.4±0.8	73.8±0.6	-9.1	-9.0
GL 6 (D)	3.4±0.3	2.2±0.3	94.4±0.8	104.9±1.0	-10.9	-10.3
GL 7 (U)	0.8±0.2	0.9±0.2	117.2±0.5	123.8±1.4	-8.5	-8.3
GL 8 (U)	5.8±0.4	4.8±0.4		110.9±0.9	-11.6	-11.5
GL 9 (U)	0.4±0.2	0.4±0.2	97.7±0.2	104.8±2.4	-10.7	-10.0
PH 66 (U)	0.4±0.1		84.0±0.9		-10.2	

(D) = Down-gradient boreholes

(U) = Up-gradient boreholes

Generally, the radiocarbon and tritium concentrations are within the normal environmental range, and indicate a fairly rapid turnover, more apparent in the up-gradient boreholes. An exception is borehole GL 7 which has a carbon-14 concentration well over 100 pMC, and a tritium concentration less than 1 TU (Table 8). A possible mechanism for this combination is the introduction of CO_2 into the ground water through root respiration by the well-developed vegetation (trees) in the vicinity.

Such combinations have been observed in other granitic areas (Verhagen, unpublished data).

An estimate of the mean residence time in years of the ground water (see Table 8a) was calculated utilising the exponential model (see Appendix)

Table 8a. Mean residence times calculated from ^3H values

Borehole Number	^3H (TU)		Mean Residence Times (Years)	
	2 Jun	14 Oct	2 Jun	14 Oct
GL 3	4.6±0.4		15	
GL 4	1.2±0.2	1.4±0.3	150	120
GL 5 (D)	0.6±0.2	0.4±0.2	300	500
GL 6 (D)	3.4±0.3	2.2±0.3	30	60
GL 7 (U)	0.8±0.2	0.9±0.2	230	200
GL 8 (U)	5.8±0.4	4.8±0.4	15	15
GL 9 (U)	0.4±0.2	0.4±0.2	500	500
PH 66 (U)	0.4±0.1		500	

(D) = Down-gradient boreholes

(U) = Up-gradient boreholes

An increase, varying from 4 to 11 pMC, is evident in the carbon-14 concentrations for the different boreholes (Table 8). No consistent change in the tritium data is evident over the sampling period, and the $\delta^{13}C$ data remain constant and within the normal range. This is suggestive of non-steady state conditions and may possibly indicate either poor contact of the boreholes with the aquifer, further illustrating the poor sampling conditions, or may be due to the relatively low natural ground water flow.

4.5.5 Summary and Discussion

In general, the down-gradient pollution monitoring boreholes have higher alkalinity, conductivity, temperature and pH values than the up-gradient boreholes. They also have higher ion concentrations than the up-gradient boreholes. The ion which most clearly distinguishes the up-gradient boreholes from the down-gradient boreholes is calcium and, to a lesser extent, bicarbonate and sodium. These differences are clearly illustrated in the modified Schoeller and Piper trilinear diagrams.

The higher major ion concentrations in the down-gradient boreholes is possibly the result of a more mature hydrochemistry as a result of the dissolution of the silicate minerals in the residual granites, leaving an aluminosilicate residue such as kaolinite, illite or montmorillonite. This is also evident in the higher alkalinity and hydrogen carbonate concentrations.

The δD and $\delta^{18}O$ data plots on, or close to, the meteoric water line, with no sign of evaporation. There is however a shift towards more positive δD in the two down-gradient boreholes, which is comparable to the more heavier stable isotope compositions in down-gradient boreholes of a study by Fritz *et al.* (1976). Although there is some apparent contradiction between the 3H and 14C, one could expect, on the assessed turnover times, that the ground water should, in general, reflect the presence of leachate. On average, the up-gradient boreholes show a shorter turnover time than the down-gradient boreholes. With the exception of the up-gradient boreholes GL 7, 8 and 9, there is no clear evidence of pollution. The pollution (e.g. nitrate) found in these up-gradient boreholes could possibly be derived from the neighbouring small holdings, the nitrate originating from animal husbandry and septic tank effluents, and the phosphate and sulphate from fertilisers.

Within the constraints of the poor sampling conditions, it can therefore be concluded that no significant leachate is entering the down-gradient ground water at present. The consistent increase in the carbon-14 concentrations of the resampled boreholes is a possible indication of the poor sampling conditions which, in turn, suggests that the pollution monitoring boreholes poorly reflect any temporal changes in the hydrochemistry. These boreholes have proved to be inadequate in their construction, and have poor hydraulic connection with the aquifer.

CHAPTER 5

5 WATERVAL LANDFILL SITE

5.1 Introduction

5.1.1 Location

The Waterval landfill site is situated approximately 15 kilometres west-north-west of central Johannesburg between the north-western suburbs of Newlands, Triomf and Albertville (Figure 12), co-ordinates 26° 10' S and 27° 57' E (Ball, 1984), in the centre of a residential area, with a school at the north-west corner (Figure 13).

5.1.2 History and Development

The Waterval landfill site is the oldest and largest in Johannesburg. It was established in 1928, and covers an area of approximately 28 hectares. It has been estimated that the volume of the landfill is between 3.0 and 4.5 x 10^6 cubic metres. It was initially a quarry, where residual clays from weathered schists of the Swaziland Supergroup were used in the making of bricks. These brickworks commenced in 1896 and continued until 1906, after which no further activity took place until the landfill was initiated. The site was closed in 1978, 50 years after landfilling commenced.

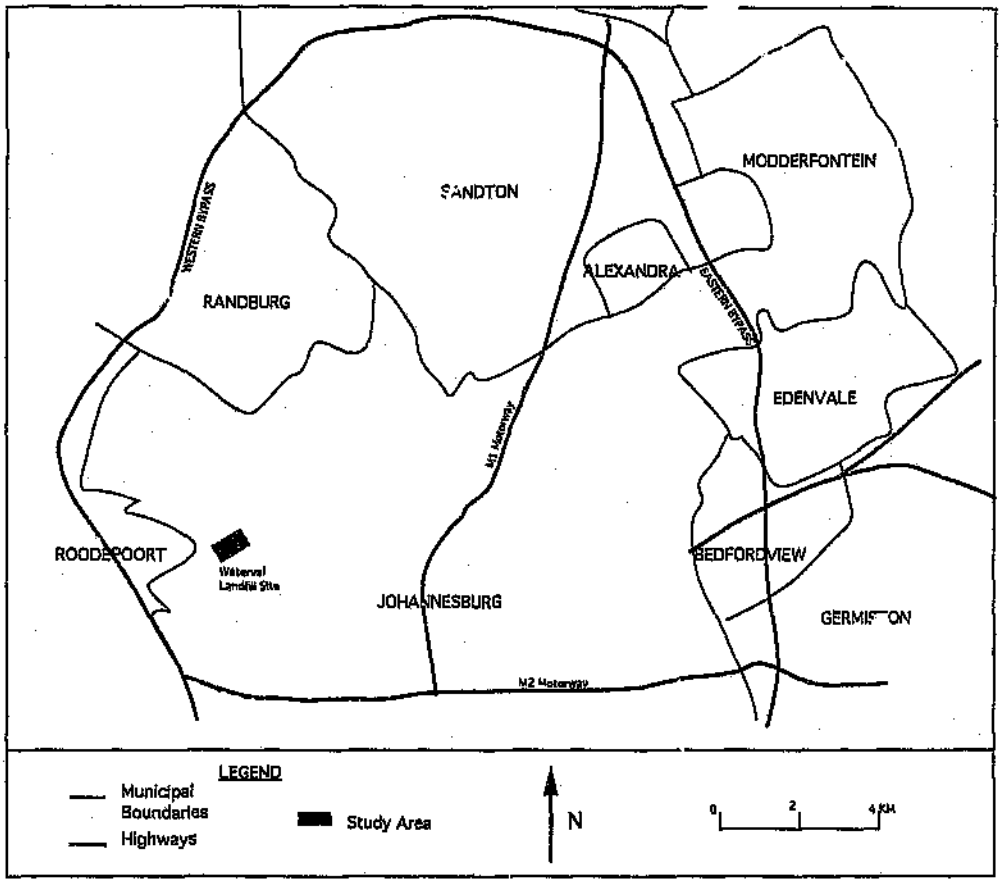

Figure 12. Map showing the locality of the Waterval landfill site

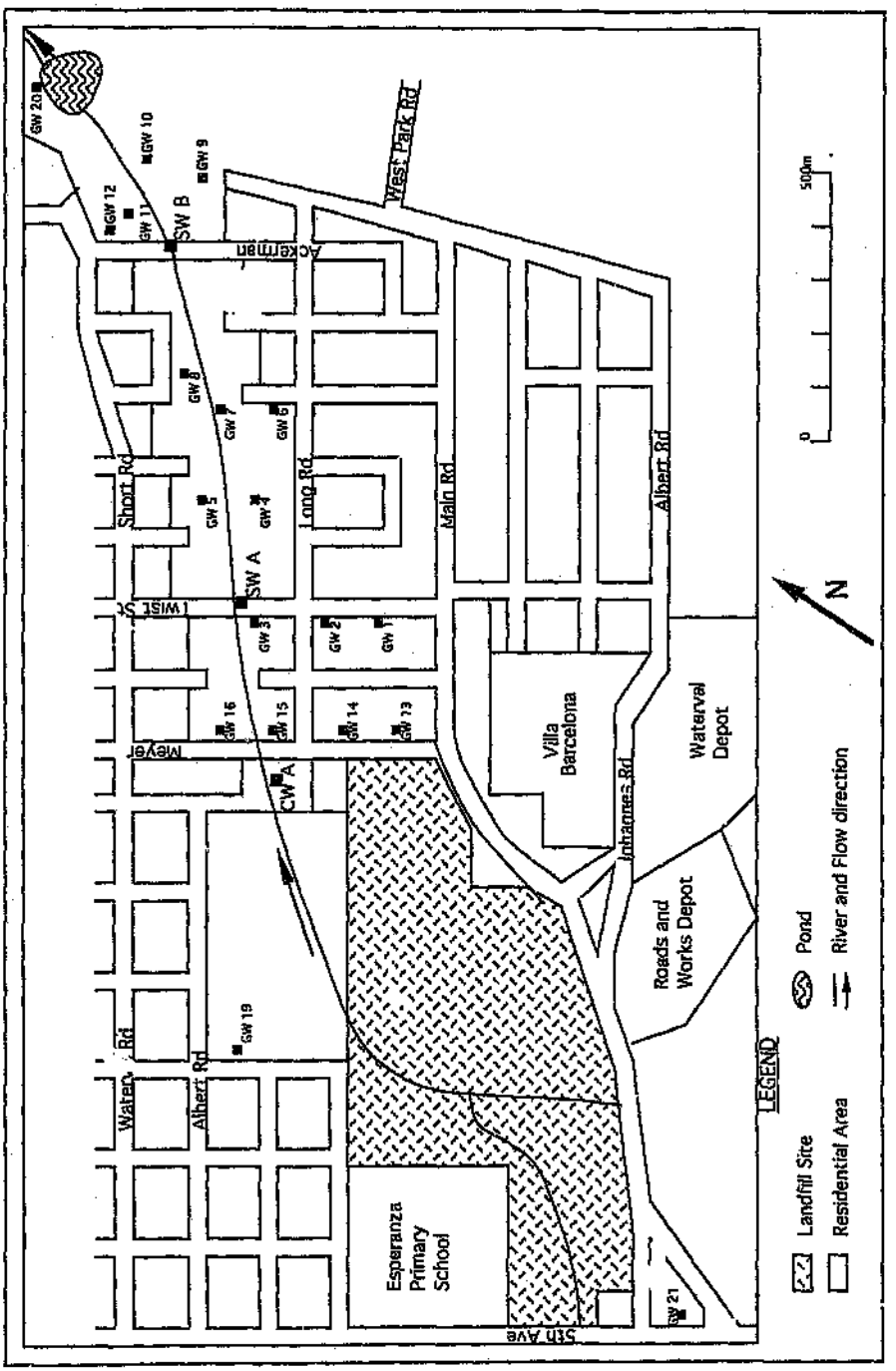

Figure 13. Map of the Waterval landfill site, showing the location of the ground and surface water pollution monitoring points

Dumping initially took place by "wet tipping" into the site of the old brickfield which, at the time, was flooded (Ball, 1984). The site received approximately 100 000 tons of refuse per annum, which was predominantly urban domestic refuse from the north-western suburbs of Johannesburg, as well as garden refuse and minor amounts of business and industrial refuse. The refuse was only covered when odours were being emitted from the landfill site. This continued until the 1970's, when a daily application of soil over the compacted refuse became standard practice. This layer of soil was between 0.15 and 0.30 metres thick, with a final layer of up to a metre in thickness. The operation was by a cut and fill method where refuse was deposited in the void left by the cover excavations and then terraced. In 1970, a culvert was constructed beneath the landfill to contain the natural stream course which originates below the landfill site. Although the culvert was insulated by a metre of clay, ground water seeps in through weepholes and joints below the water table (Ball, 1984).

5.2 Hydrogeology

The Waterval landfill site is situated on the south-south-west margin of the Johannesburg-Pretoria Granite Dome. The rock types on this contact zone are dioritic gneisses and hybrid granites of the dome, and ultrabasic schists from the lower Onverwacht Group of the ancient Swaziland Supergroup (Figure 14) (Ball, 1984; Tankard *et al.*, 1982). The only indication of the Swaziland Supergroup in the vicinity of the landfill is the river bed, which is made up largely of clayey silt, recognised by Brink (1979) as being the remnants of rocks from the Swaziland Supergroup after undergoing a high degree of chemical decomposition. The unweathered schists are expected to have low porosities, as the pores are small and generally not interconnected. This often results in zero permeability except in zones of fracturing and weathering.

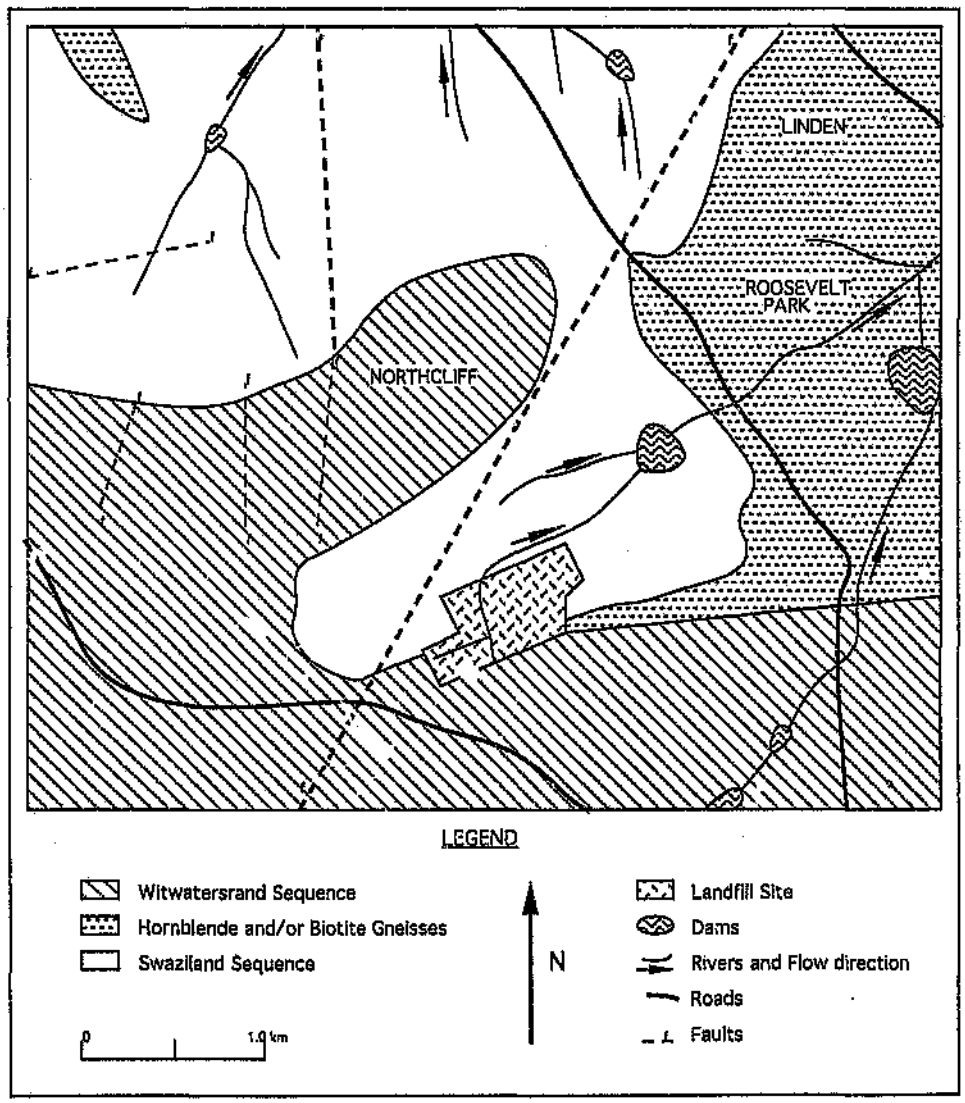

Figure 14. Map of the main geological features of the Waterval study area

To the south of the landfill site, and striking east-west, are shales and quartzites of the Witwatersrand Supergroup (Figure 14). These younger rocks unconformably overlie the rocks of the Swaziland Supergroup (Anhaeusser, 1973). Northcliff Hill, situated to the north of the landfill, is a ridge of Orange Grove Quartzites which was faulted and displaced about two kilometres to the north-east, and slightly rotated. This resulted in the formation of the Waterval Valley which trends in a NE/SW direction, bounded on either side by quartzites, and forms a catchment area which drains to the north-east. The Waterval landfill site is situated within this valley (Figure 14).

5.3 Previous Studies

In September 1976, the Johannesburg City Council initiated monitoring of the Waterval landfill site. Water samples from pollution monitoring boreholes, streams and the culvert were taken during both wet and dry seasons, and analysed for various elements and pollutants. The locations of the pollution monitoring boreholes are depicted in Figure 13. As was the case for the Linbro Park monitoring boreholes, these boreholes were sampled using the open-bailer method, where only the uppermost portion of the standing water column is sampled. It is therefore questionable whether these samples were representative of the ground water at that time.

An MSc thesis (Ball, 1984) gives a fairly extensive report of both the surface and ground water situations at the Waterval site. This project involved the sampling of the surface and ground water in the vicinity of the landfill, at the end of each season, to assess the pollution status and any variation in pollution with time and distance from the landfill. It was concluded, using conductivity and chlorinity as proxy indicators, that significant pollution in the form of a plume in the ground water could be observed in some of the observation boreholes down-gradient of the landfill. A 1.2 metre diameter hole was augered through the landfill which was then sampled and profiled.

5.4 Results of the Present Study

5.4.1 Introduction

An initial attempt to sample the pollution monitoring boreholes early in 1992 proved to be futile as almost all these boreholes were secured with manhole covers, which proved to be impossible to remove without the correct tool. The boreholes that were accessible were dry and appeared to be filled-in with soil which, at the time, was assumed to be the result of vandalism.

At a later date, the correct tool was obtained, and most of the manhole covers were removed. However, all but one of the boreholes were found to be dry and filled-in with soil, which could only have resulted from the casing collapsing, as vandalism would not have been possible. The only pollution monitoring borehole that could be sampled was GW 21, which is situated up-gradient of the landfill site (Figure 13). This borehole had an extremely low yield, although sufficient water (a few litres) was available for stable isotope and tritium samples.

Later in 1992, several private boreholes in the vicinity of the landfill site were sampled for stable isotopes and tritium. Well-head measurements were not recorded for any of these boreholes. The following are the addresses of these private boreholes:

5 Smuts Street,	Waterval Estate
8 Willow Street,	Waterval Estate
Auckland Park Bowling Club, John Adamson Drive,	Montgomery Park
44 Max Michaelis Street,	Montgomery Park
Huis Hoëveld, Main Street,	Albertville

The localities of these borehole are shown in Figures 15 and 17. A greater spread of sampling points was foreseen; however no further private boreholes could be located in the area.

Three surface water samples for both stable isotope and tritium analysis were also taken. One sample was taken from the culvert passing through the landfill slightly downstream of the landfill, namely CW A, and two were from the stream, namely SW A and SW B (Figure 13). At the time, no field measurements were taken. However, in December of 1994, these sampling points were analysed for the various field parameters to clarify certain features observed in the environmental isotope data.

5.4.2 Field Measurements

The conductivity of the surface water samples was found to be higher in the stream water sample, SW A (Table 9), than in the culvert water, CW A, which is assumed to be predominantly seepage from below the landfill site (Ball *et al.*, 1986). This suggests that additional contamination enters the stream from a source situated between the culvert and the SW A sampling point. The estimated flow rate roughly doubles between the culvert and this sampling point, indicating an influx of water, which is also evident in the somewhat higher alkalinity value, and the lower pH. In all four samples, a fairly high nitrate concentration was measured using nitrate test strips; however they are all within the recommended limit for drinking water (Table 9).

Table 9. Field measurements for surface water samples

Sample Number	E.C.	Alk	pH	Estimated Flow Rate	Estimated Nitrate
	(mS/m)	(meq/l)		(l/sec)	(mg/l)
CW A	68.0	4.0	8.15	~ 1	~ 20
SW A	83.5	4.3	7.80	~ 2	~ 20
SW B	58.0	2.8	8.15	~ 4	~ 20
Pond Entry	65.0	3.4	8.30	~ 4	~ 20

Further downstream, at sample point SW B, the conductivity is lower, the pH returns to the previous value found at CW A, and the alkalinity drops considerably. The

estimated flow rate once again increases to roughly double that seen at the SW A sample point, indicating further influx of water into the stream, but apparently dissimilar in solute contents to that seen at SW A. Between S W B and the pond sample, the field chemistry begins to return to the initial values seen in the CW A sample.

5.4.3 Environmental Isotopes

Stable Isotopes

The samples from the culvert and stream have heavier stable isotopic compositions than the surrounding private boreholes (Figure 15), particularly the up-gradient pollution monitoring borehole GW 21. SW A is significantly heavier than the other two surface water samples, further suggesting the introduction of water from a source other than ground water. The decrease in the $\delta^{18}O$ value at the SW B sampling point is an indication that isotopically lighter water, probably ground water, has entered the river, substantially diluting the isotopically heavier stream water. Moisture above the water line along the banks of the stream during low flow conditions, supports the assumption that the isotopically lighter water is ground water. The ground water elevations are higher than the stream bed, with the exception of 44 Max Michaelis, where the stable isotope composition suggests influence from the river.

The sample from the culvert, CW A, plots on the MWL, but significantly heavier than the ground water samples, probably due to contributions of RW drainage water from the higher lying residences and schools. The stream water sample, SW A, plots isotopically considerably heavier than the other samples, and along a mixing line between the local ground water and Rand Water mains (Verhagen *et al.*, 1995b)(Figure 16). This suggests a large percentage of RW mains combined with the stream water, and is more than likely a leakage in the sewer system which runs along the length of, and is occasionally exposed in, the stream bed. The two isotopically lighter samples are

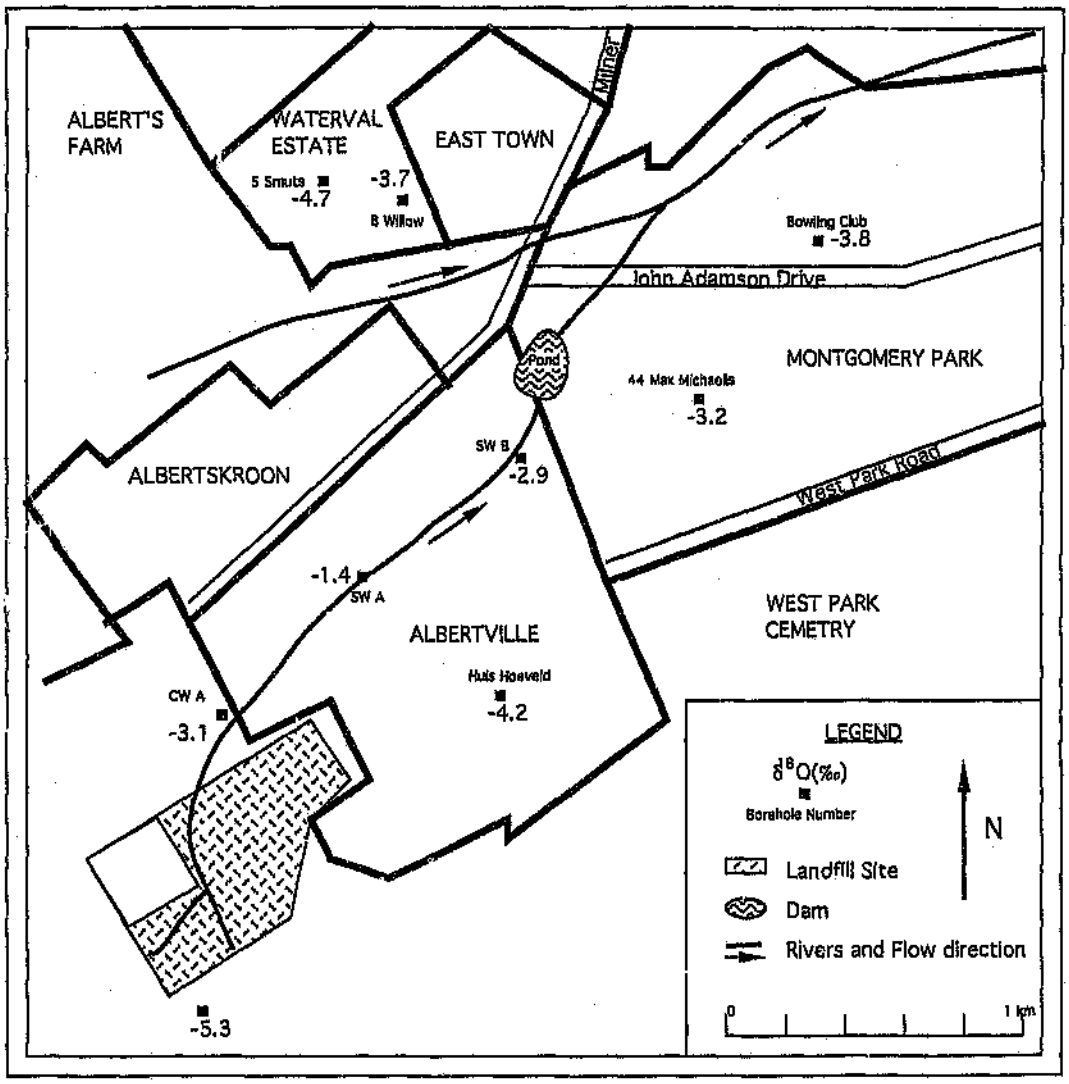

Figure 15. $\delta^{18}O$ values of the ground, culvert and surface water in the vicinity of the Waterval landfill site

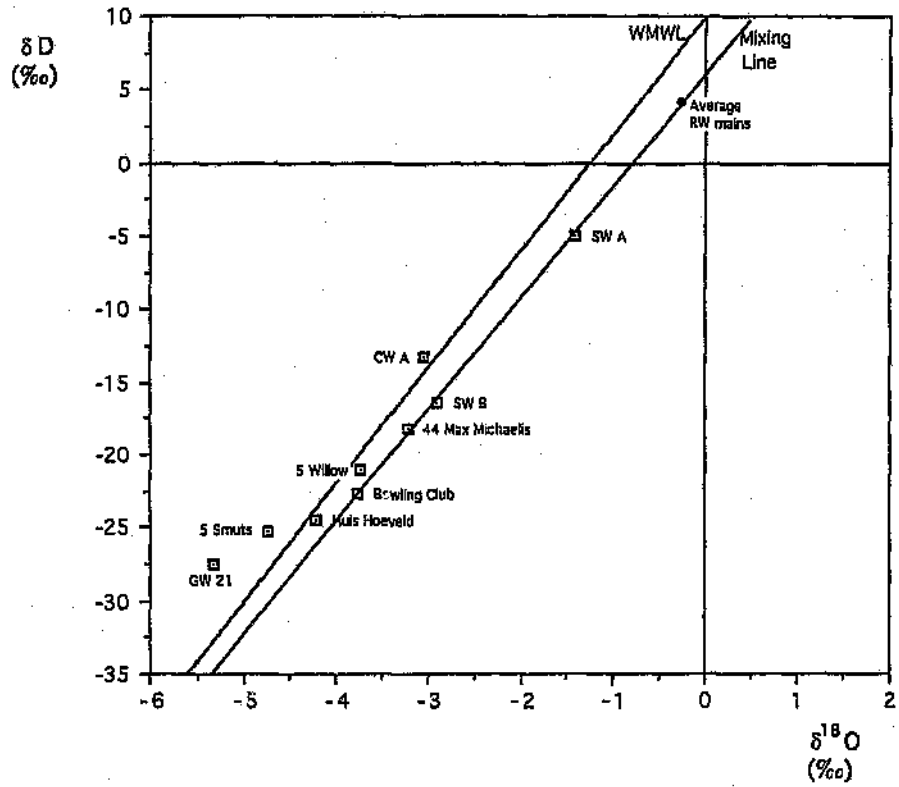

Figure 16. Plot of δD against $\delta^{18}O$. Also shown is the world meteoric water line (WMWL), as well as the mixing line between local ground water and RW water

the two most upstream samples. It therefore appears that the effect of the RW mains water is widespread. The average stable isotope composition of RW mains water for the period 1979 to 1995 within the Gauteng area has been plotted in Figure 16 (Verhagen *et al.*, 1995). It is clear that the SW A sample plots between this average value and the value of uncontaminated ground (GW 21) and surface water in the area, indicative of mixing.

Table 10. Stable and radioactive isotope values for the pollution monitoring boreholes

Borehole Number	δD (‰)	$\delta^{18}O$ (‰)	^{3}H (T.U.)
Huis Hoëveld	-24.5	-4.21	0.7±0.2
CW A	-13.2	-3.05	23.2±1.1
SW A	-5.0	-1.41	12.2±0.6
SW B	-16.4	-2.90	8.3±0.5
5 Smuts	-25.3	-4.73	2.6±0.3
Bowling Club	-22.7	-3.75	4.5±0.3
8 Willow	-21.1	-3.74	4.2±0.4
44 Max Michaelis	-18.2	-3.21	6.6±0.4
GW 21	-27.5	-5.32	3.4±0.3

Tritium

A significantly high tritium concentration of 23.2 TU is found in the culvert water sample, CW A. This is much higher than the normal environmental range (IAEA, 1992; Schonland unpub. data), indicating an artificial tritium source, probably within the landfill. Elevated tritium levels have been identified in other South African landfill sites, the source of which is yet unknown (Verhagen *et al.*, 1998). The tritium concentration decreases considerably downstream (Figure 17). This is probably the result of dilution by two inflows previously identified:

a) channel water, assumed to be Rand Water mains-derived, with an expected value of some 5 TU;

b) local ground water, with at most 4.5 TU.

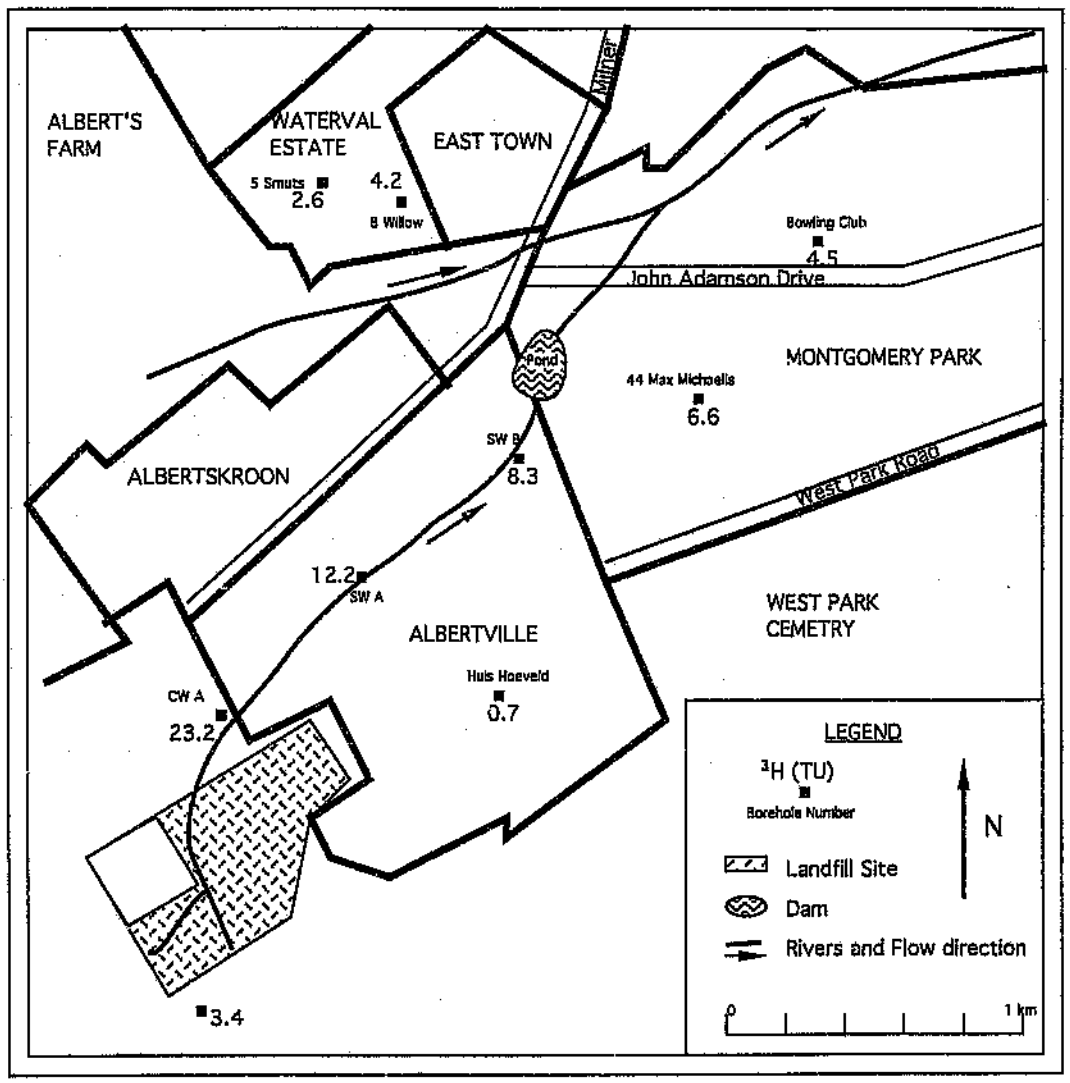

Figure 17. Tritium values of the ground, culvert and surface water in the vicinity of the Waterval landfill site

The tritium concentrations in the stream are higher than the ground water concentrations encountered in the private boreholes, with the exception of 44 Max Michaelis which further suggests contamination from the river.

5.4.4 Summary and Discussion

The study on this landfill site confirms that the UPVC casing, used by the Johannesburg Municipality in the pollution monitoring boreholes, is not sufficiently durable to sustain the extensive period of time required to monitor the landfill during its operation, and after the closure of the site, particularly in loose sediments (see also Chapter 4).

A strong case for the continued monitoring of this landfill site is the fact that private boreholes, down-gradient of the landfill, may be directly recharged by the stream, as is suggested by both the $\delta^{18}O$ and tritium values at 44 Max Michaelis. The work of Ball (1984) indicates that pollution could be observed in the (now defunct) network of pollution monitoring boreholes. This plume of higher concentrations of conductivity and chloride values observed downstream of the landfill site was presumed to indicate the movement of pollution in shallow ground water in the alluvium. The $\delta^{18}O$ values, together with field chemistry, clearly show that further pollution is occurring to the river water downstream of the landfill site. This may be attributed to leakages in the sewage system, which runs along the river bed. It is also evident that this water is diluted further downstream, probably by influent ground water. The higher chloride values seen in the plume in the study by Ball (1984) centred on the stream bed might therefore have been produced by bank infiltration of polluted water from the site, with possible further solute concentration through evapotranspiration from the shallow ground water.

Downstream private boreholes are potentially susceptible to pollution within the stream, which, in turn, is partially derived from leachate seeping through weepholes in the culvert. The University of the Free State recommend a period of 20 years after the closure of the two Bloemfontein landfill sites (Bekker, 1992). The Waterval site has been closed for less than that suggested time which might be regarded as an absolute minimum. From the condition of the boreholes, it appears that monitoring of the ground water had already ceased by the early 1990's.

The significantly high tritium concentration in the culvert water sample, apparently due to artificial tritium within the landfill, is evidence of the continued pollution of the stream, most likely resulting from leachates emanating through the weepholes in the culvert. The source of this artificial tritium is probably from the minor amounts of industrial (and medical?) refuse which were previously deposited at this site. This tritium concentration, which must be assumed to have been introduced pre-1978 when the landfill was still in operation, is considerably diluted as it moves downstream by, what appears to be, both Rand Water mains and ground water. Leachate from this site is therefore uniquely labelled and the artificial tritium can be used as a convenient surface and ground water tracer for pollution from the site. With the detection of elevated tritium levels in other South African landfill sites, tritium is rapidly becoming an extremely useful tool as a tracer of pollution from these sites (Verhagen *et al.*, 1998).

Possible influence of the stream water is seen in the ground water downstream at 44 Max Michaelis. There is also evidence of influent conditions from the ground water in the lower reaches of the stream, up to the dam. However, boreholes situated close to the stream bed might well reflect as much the stream water as they may reflect the ground water, carrying contamination from the landfill.

CHAPTER 6

6 THE BLOEMFONTEIN NORTHERN LANDFILL SITE

6.1 Introduction

6.1.1 Location

The Bloemfontein Northern landfill site is located to the north of the city, and is about 35 hectares in extent. The slope of the general area is to the south-east, ranging from around 40° in the north-west (Petra Groef), down to 2° in the south-east (Farmer's Dam)(Figure 18). The landfill site has a fairly steep slope to the north, and is situated on a slight rise which is controlled by the high resistance to weathering of the underlying dolerite.

Water from the dolerite mine, Petra Groef, located on the north-western boundary and up-gradient of the Northern landfill site (Figure 18), may affect the chemistry of the ground and surface water, as a large percentage of run-off from the mine drains through a significant portion of the landfill. A catchment dam in a streambed is situated on the eastern boundary of the landfill site, and a farmer's dam is positioned about 100 metres further downstream (Figure 18).

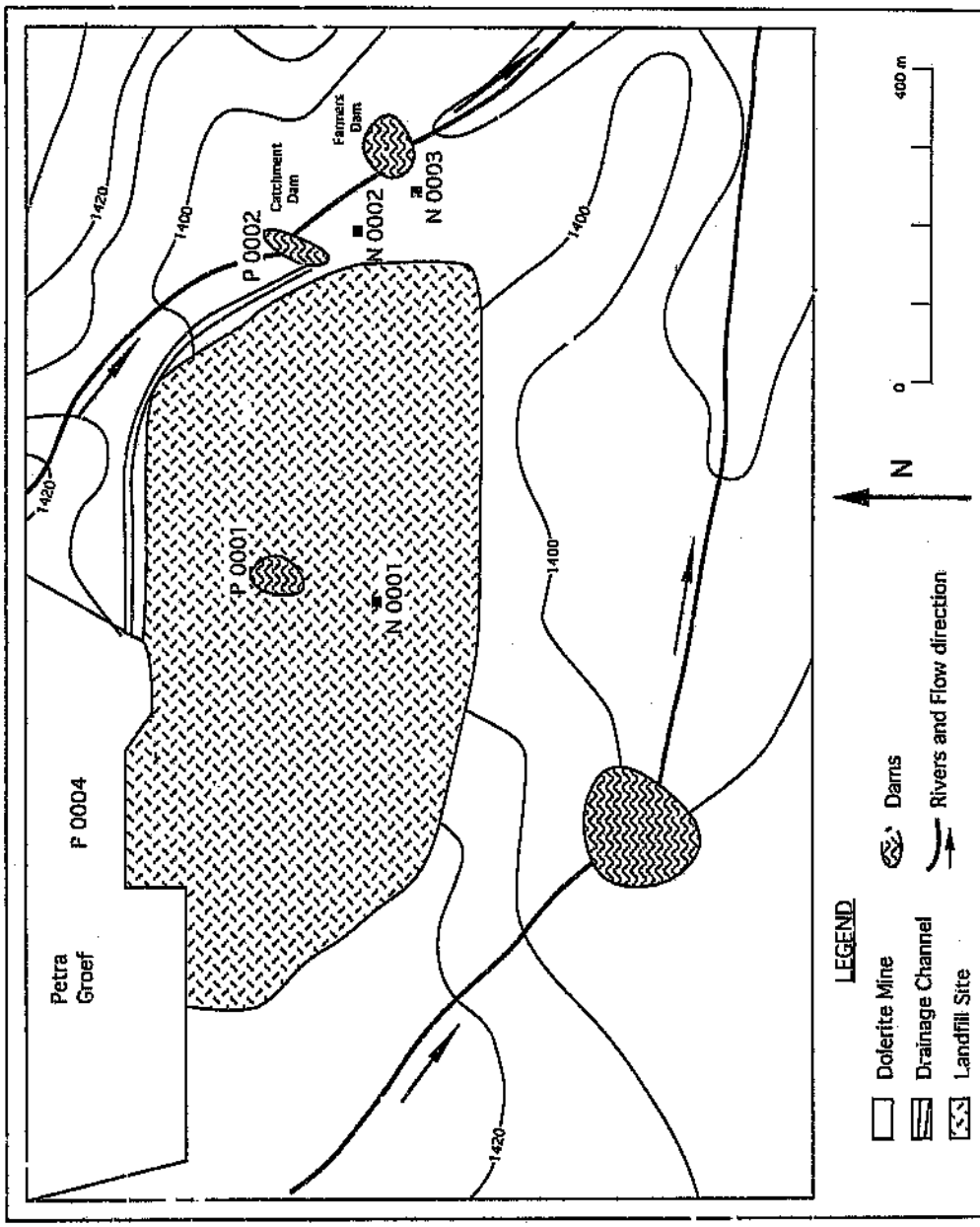

Figure 18. Map of the Bloemfontein Northern landfill site, showing the location of the pollution monitoring boreholes, as well as the Petra Groef dolerite mine

6.1.2 History and Development

The Northern landfill site is classified as a G:M site (Chapter 1.1). The waste material is placed in cells which are then mixed and compacted with clayey soil (Bekker, 1992).

Three boreholes have been drilled to monitor any ground water pollution from the landfill. Borehole N 0001 is situated approximately in the middle of the site, while boreholes N 0002 and N 0003 are both down-gradient of the landfill site, along its eastern boundary (Figure 18). All three boreholes were drilled into dolerite to a depth of about 35 metres (Table 11) (Bekker, 1992). The monitoring boreholes in this site and the Bloemfontein Southern landfill site (see Chapter 7) are much higher yielding, and are constructed with large diameter steel casing which is considerably more durable and accessible, than boreholes previously discussed in Chapters 4 and 5.

Table 11. Rest levels and borehole depths of the pollution monitoring boreholes

Borehole Number	Borehole Depths	Rest Levels (below surface)	
		31/02/92	30/08/93
	(m)	(m)	(m)
N 0001	31.5	3.0	
N 0002	38.5	2.0	
N 0003	31.5	2.0	3.5

6.2 Hydrogeology

The landfill site is underlain by 3-4 metres of very coarse and crystalline, weathered dolerite, followed by fresh dolerite which is estimated to be about 38 metres thick, forming a solid base for the flow of ground water (Bekker, 1992). The yield of the

monitoring boreholes in the weathered dolerite ranges from 0.2 to 0.5 l/s, and the permeability, measured by slug tests, was found to be fairly high, with k-values of between 0.4 and 0.6 metres/day for boreholes N 0002 and N 0003 respectively.

The run-off water from the upper reaches of the area is captured in the catchment dam, and includes drainage from the Petra Groef mine. A large portion of the surface drainage from the mine is diverted along a channel situated to the north of the site to prevent it from flowing through the site (Figure 18). A pond, designated P 0001, is situated on the landfill site, north of, and down-gradient of, borehole N 0001.

The expected ground water flow is in a south-easterly direction, following the topography, towards the two dams. However, as the landfill slopes to the north, ground water within and below the disposal site itself will flow in a northerly direction.

The water levels of the pollution monitoring boreholes at the time of drilling were between 2 and 3 metres below the surface (Bekker, 1992). The only rest level measurement recorded during the sampling in August of 1993, in borehole N 0003, dropped slightly from 2 to 3.5 metres (Table 11), which is still quite shallow.

6.3 Previous Studies

A study of the Northern landfill site, initiated by the Bloemfontein Health Department, is being conducted by the Institute for Ground Water Studies at the University of the Free State. It has involved the sinking of boreholes down-gradient of the landfill, an interpretation of the local geology, as well as sampling and chemical analyses of the ground and surface water. Taking into account the shallow water table, the relatively high permeability of the underlying weathered dolerites, the steep topography and the proximity of the nearest farm, the area was classified as poor for a sanitary landfill (Bekker, 1992).

In 1992, borehole N 0001 was found to be significantly polluted, with high concentrations of organics, calcium, magnesium, chloride (Table 13) and a manganese concentration of 8.0 mg/l. Borehole N 0002, although considerably less polluted than N 0001, had significantly high iron and lead concentrations of 2.37 and 0.63 mg/l respectively, while borehole N 0003 had an iron concentration of 0.64 mg/l .

Aluminium and iron exceeded the maximum acceptable value in a water sample from the Petra Groef dolerite mine, with values of 1.98 and 2.15 mg/l. It therefore appears as if the higher iron and aluminium concentrations in boreholes N 0002 and N 0003 are mainly due to run-off from the dolerite mine, probably introduced into the ground water via the catchment dam.

6.4 Results of the Present Study

6.4.1 Introduction

An environmental isotope and hydrochemical survey of the three pollution monitoring boreholes was conducted in January, 1992 and in August, 1993. The borehole water in all three holes was extracted using a mobile submersible pump, made available by the Institute of Ground Water Studies, University of the Free State. In each case, the pump was lowered to a depth of around 25 metres. The volume of water standing in each hole was purged until a constant conductivity reading was obtained prior to the collection of each sample. Well-head values of conductivity, alkalinity, pH and temperature were recorded. A carbon-14 sample was not taken at N 0001 on account of the extremely poor condition of the water, and as it is believed that the chemistry would be too complex due to significant pollution, including organics.

6.4.2 Hydrochemistry

Well-head Measurements

The well-head measurements for the three pollution monitoring boreholes are given in Table 12. The conductivity of borehole N 0001 during sampling in August of 1993 had an initial value of 190 mS/m with much scum, but stabilised at 550 mS/m once the hole had been sufficiently purged. This value was lower than the value of 620 mS/m obtained during sampling in January of 1992. According to Pohland (1986), the value of 550 mS/m indicates that these leachates are at a well developed stage. The conductivity of borehole N 0002 for August, 1993, remained stable at 120 mS/m throughout the period it was pumped, while N 0003 was initially 80 mS/m, but increased to between 115 and 120 mS/m once a sufficient volume of water had been extracted. All three boreholes have conductivities above the recommended limit for drinking water (Table 6), and borehole N 0001 produced a value above the crisis limit of 400 mS/m, which is the maximum value for low risk (Weaver, 1992).

Table 12. Well-head measurements for the pollution monitoring boreholes (August, 1993)

Borehole Number	Alkalinity (meq/l)	Conductivity (mS/m)	pH	Temperature (°C)
N 0001		550		
N 0002	7.1	120	7.35	19.0
N 0003	6.4	115	7.00	19.0

The alkalinity and pH values for borehole N 0002 during August of 1993 were both slightly higher than the further down-gradient borehole N 0003 (Table 12), both appearing to decrease with distance from the landfill. Although no alkalinity

measurement was obtained for borehole N 0001 during August of 1993, a previous value higher than 25 meq/l was measured at this borehole in January, 1992.

Ionic Concentrations

The major ion concentrations for the various sampling dates are given in Table 13. In the three pollution monitoring boreholes, as well as the sample from the pond on the landfill, P 0001, the dominant cation is magnesium, while the dominant cation in the 1992 samples taken at Petra Groef mine, P 0004, and the catchment dam, P 0002, is sodium. However, the 1994 catchment dam sample becomes calcium dominant. Chloride is the dominant anion in borehole N 0001, the 1993 and 1994 samples of N 0002, and in sample P 0001, while bicarbonate is dominant in the 1992 sample of N 0002, borehole N 0003, and in P 0002 and P 0004. The 1993 and 1994 samples from borehole N 0003 have a high chloride component, almost as high as the bicarbonate. The high sulphate component in 1992 catchment dam sample, P 0002, is possibly associated to run-off from the Petra Groef dolerite mine.

The primary differences between the pollution monitoring boreholes are in the calcium, magnesium and chloride concentrations, and to a lesser extent, sodium and bicarbonate. The high bicarbonate value, inferred from total alkalinity, may be partially generated by CO_2 release in the landfill, and partially from other compounds resulting from pollution, affecting the alkalinity.

From the modified Schoeller diagram in Figure 19, it is clear that borehole N 0001 has a significantly higher ion concentration than the other two down-gradient pollution monitoring boreholes. It is also evident that the ion concentrations of this borehole have increased since the initial sample in early 1992. This increase is also apparent in the other two monitoring boreholes. The occurrence of higher concentrations of ions, particularly chloride, as well as sodium, calcium and magnesium, in boreholes N 0002

Table 13. Major ion concentrations in meq/l for various sampling dates

Borehole Number	Date Sampled	Potassium (meq/l)	Sodium (meq/l)	Calcium (meq/l)	Magnesium (meq/l)	Chloride (meq/l)	Bicarbonate (meq/l)	Sulphate (meq/l)	Nitrate (meq/l)
N 0001	Jan 92	0.06	3.2	18.9	19.0	19.1	18.6	0.57	0.01
N 0001	Aug 93	0.05	9.1	18.5	38.2	31.3	27.8	0.06	0.002
N 0001	Apr 94	0.03	4.1	17.6	31.7	34.6	16.1	1.59	0.002
N 0002	Jan 92	0.02	1.1	3.0	3.2	0.7	4.6	1.29	0.03
N 0002	Aug 93	0.02	3.8	5.6	8.1	9.7	7.3	1.37	0.02
N 0002	Apr 94	0.02	3.1	5.2	7.7	9.0	6.5	0.06	0.01
N 0003	Jan 92	0.05	1.0	3.0	4.2	1.3	6.0	0.92	0.01
N 0003	Aug 93	0.02	2.2	4.9	7.2	6.0	6.1	1.53	0.02
N 0003	Apr 94	0.02	1.8	4.8	6.8	5.8	6.2	1.57	0.02
P 0001	Jan 92	0.56	4.4	10.0	14.2	17.7	10.5	0.20	0.03
P 0001	Apr 94	0.08	6.7	2.8	10.6	11.6	7.3	1.33	0.002
P 0002	Jan 92	0.74	5.5	1.4	0.9	7	4.3	2.22	0.14
P 0002	Apr 94	0.14	2.9	3.9	2.7	2.8	5.3		0.002

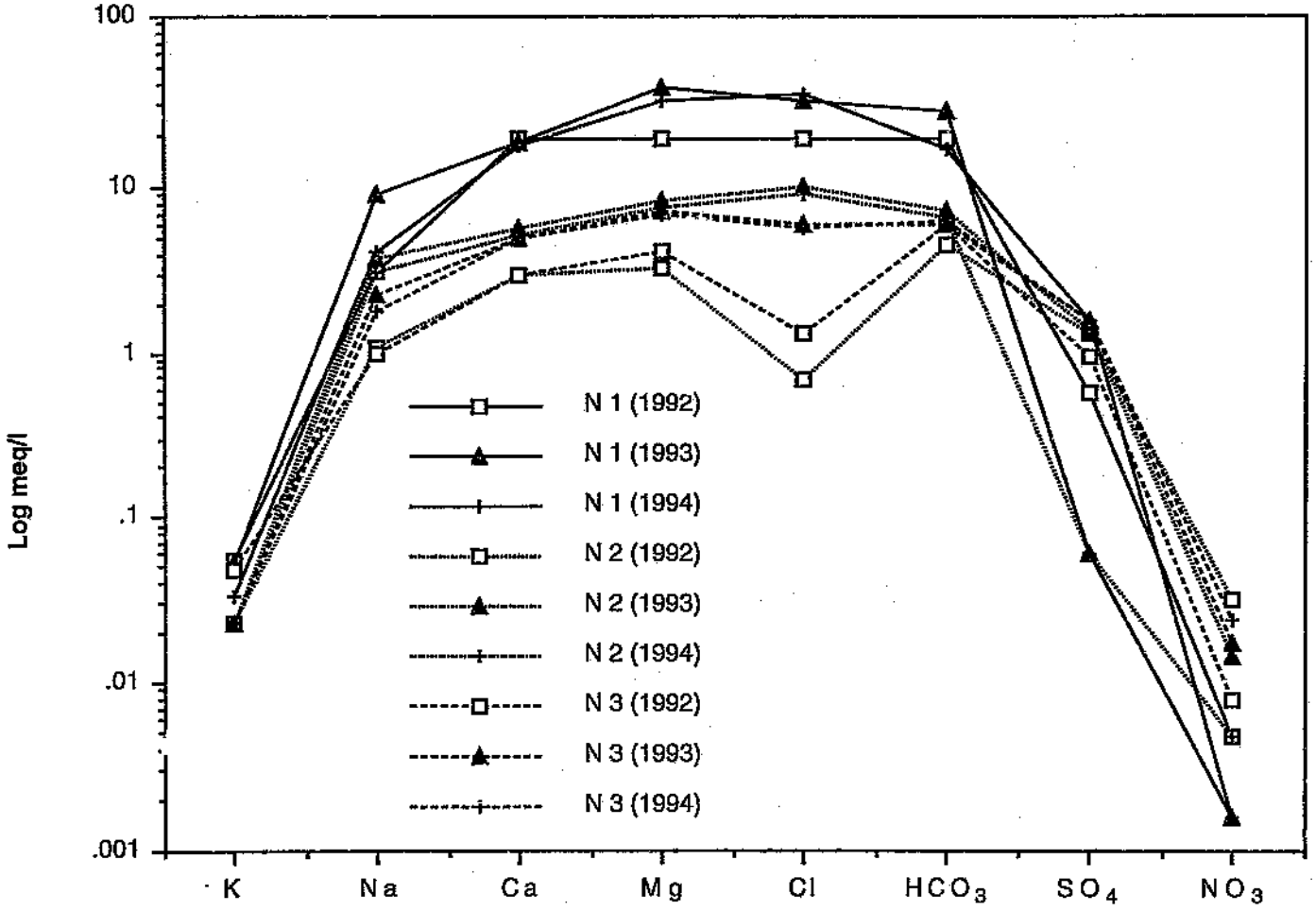

Figure 19. Modified Schoeller diagram for the Bloemfontein Northern landfill monitoring boreholes

and N 0003 subsequent to the 1991 sample, suggests incipient pollution down-gradient of the landfill (Figure 19).

It is possible to compare the 1992 catchment dam sample, P 0002, with the sample from the dolerite mine, P 0004, as well as with the 1992 samples of N 0002 and N 0003 (Figure 20). However, the relative ionic concentrations of the 1994 P 0002 sample are more characteristic of P 0001, the pond sample alongside N 0001, and reasonably similar to N 0002 and N 0003, suggesting leachate transport to the down-gradient boreholes near the catchment dam. The higher concentrations of the major ions in the two down-gradient boreholes is either indirectly due to the enrichment of the ions in the pond as a result of evaporation, or directly due to the encroachment of the polluted ground water from the site.

The three pollution monitoring boreholes plot in the calcium, magnesium dominant field in the cation triangle of the Piper trilinear diagram (Figure 21). These boreholes, as well as the surface water sample alongside N 0001, namely P 0001, show slight increases in percent magnesium. The surface water samples from the Petra Groef mine and the catchment dam both plot well away from the boreholes, but an increase in both magnesium and calcium is evident in the 1994 sample from the catchment dam. In the anion triangle, there is a general increase in the chloride concentration, with both N 0002 and N 0003 trending towards values in the highly polluted borehole N 0001. Sample P 0002 increases in both chloride and bicarbonate, and moves away from the sample from the Petra Groef mine P 0004, towards the samples from the down-gradient boreholes. The diamond field clearly shows the similarity in the chemistries of the 1993 and 1994 samples from the down-gradient boreholes to the chemistry of borehole N 0001. It is also clear that the composition of sample P 0002 trends towards the composition of the monitoring boreholes.

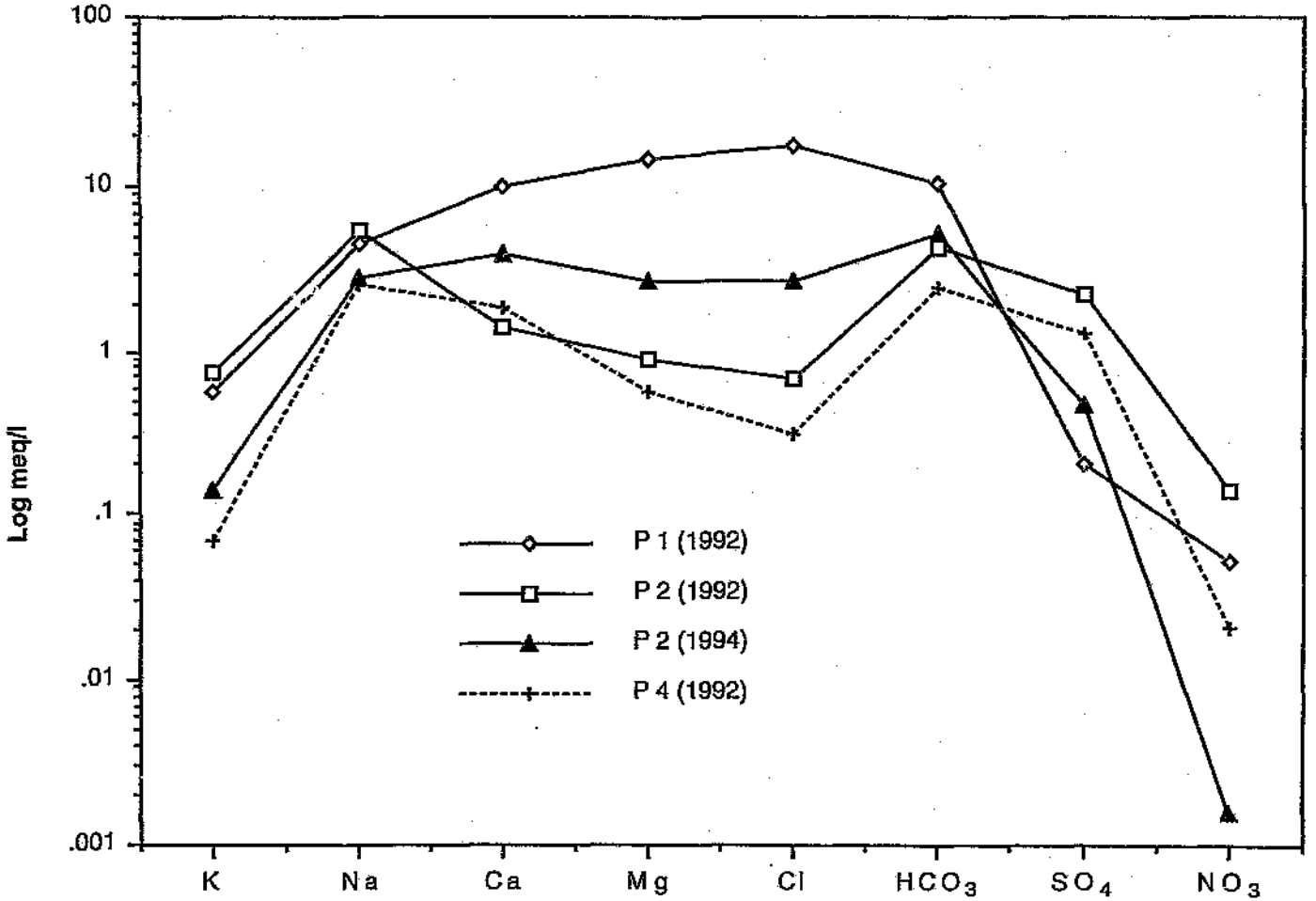

Figure 20. Modified Schoeller diagram for the surface water samples

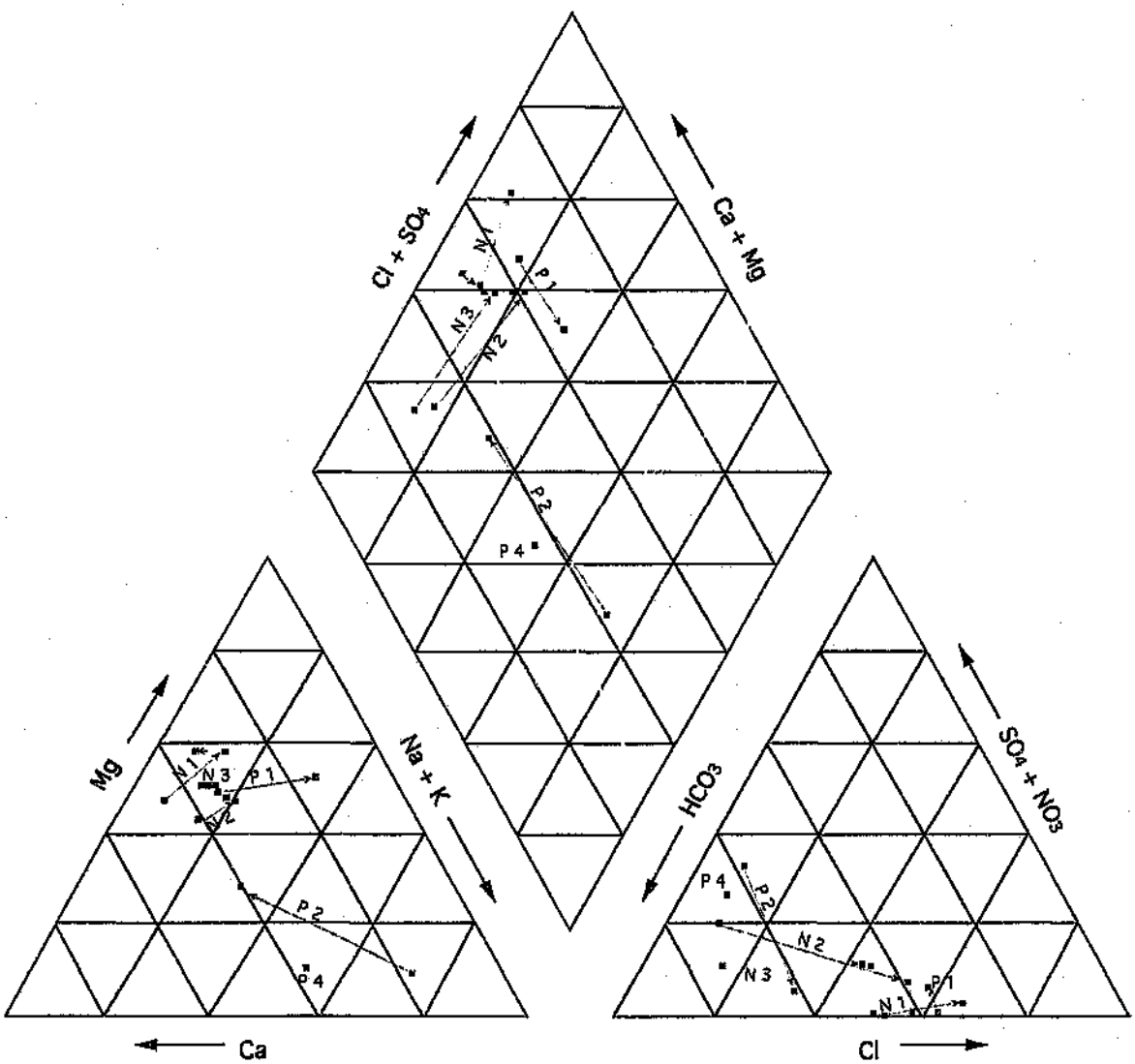

Figure 21. Piper trilinear diagram for the pollution monitoring bc reholes and the surface water samples at the Bloemfontein Northern landfill site, showing temporal trends

As borehole N 0001 is virtually pure leachate, it is possible to approximate the stage of development of the landfill according to Pohland (1986). The only major ion which has a lower concentration than mentioned by Pohland (1986) is potassium. The chloride, sodium, magnesium and calcium indicate that the site is well developed, in the stage where either acids or methane are being formed. The presence of high sulphate concentrations however, indicates that reducing conditions are not yet predominant in the landfill, even though nitrate levels are low, suggesting the site is still in the stage of acid formation with the continuing hydrolysis and fermentation of waste and leachate constituents.

6.4.4 Environmental Isotopes

Stable isotopes

The observed changes in the stable isotope compositions between the 1992 and 1993 sampling dates in the three monitoring boreholes (Table 14) are interpreted as follows: The samples from borehole N 0001 plot slightly to the left of the world meteoric water line (Figure 22), possibly due to hydrogen isotopic exchange with organics and methane within the landfill. This was also seen in the down-gradient boreholes of the Linbro Park landfill site (see Section 4.5.4). This distinctive isotope composition can possibly be used as a tracer of landfill leachates, as was achieved by Fritz *et al.*, (1976). The values of both borehole N 0002 and N 0003 indicate significant, but widely different evaporative enrichment, suggesting recharge from evaporating surface water. During the sampling in 1992, the catchment dam was almost dry, probably resulting in the significantly heavy isotopic composition of N 0002. During 1993, this dam was completely dry. It appears as if in 1993 ground water became the main source of recharge for these two boreholes, resulting in the closely similar stable isotope compositions, along the evaporation line (Figure 22).

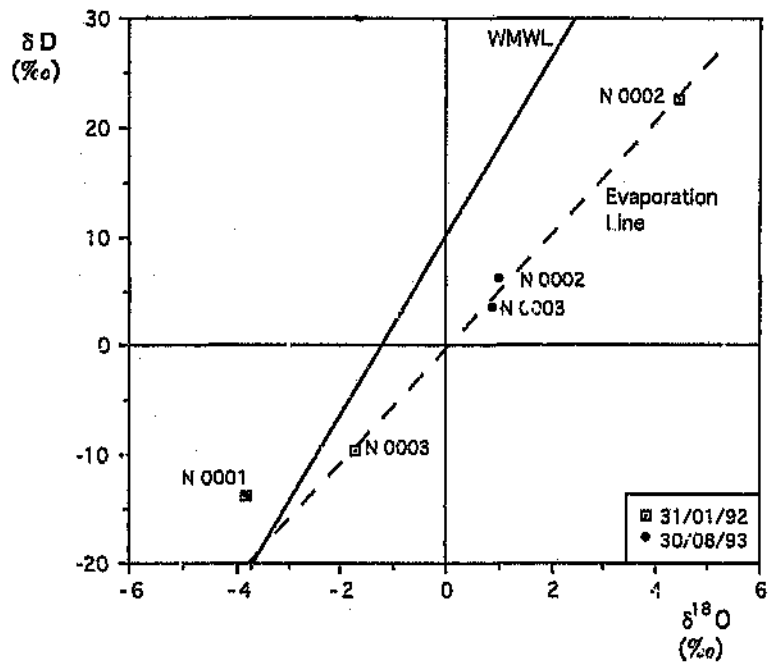

Figure 22. Plot of δD against $\delta^{18}O$. Also shown is the world meteoric water line (WMWL).

Table 14. Stable isotope values for the pollution monitoring boreholes for the 1992 and 1993 sampling periods

Borehole Number	δD (‰)		$\delta^{18}O$ (‰)	
	Jan 92	Aug 93	Jan 92	Aug 93
N 0001	-13.7	-14.1	-3.83	-3.79
N 0002	+22.6	+5.7	+4.44	+0.98
N 0003	-9.5	+3.2	-1.72	+0.88

Tritium

Table 15. Radioactive isotope values for the pollution monitoring boreholes for the 1992 and 1993 sampling periods

Borehole Number	^{3}H (TU)		^{14}C (pMC)		$\delta^{13}C$ (‰)	
	Jan 92	Aug 93	Jan 92	Aug 93	Jan 92	Aug 93
N 0001	26.5±1.5	27.7±1.1				
N 0002	8.0±0.5	11.8±0.8	108.2±0.9	111.4±0.7	-7.9	-10.0
N 0003	8.1±0.6	8.8±0.7	113.8±0.8	111.8±0.7		-10.0

The tritium concentrations in the three pollution monitoring boreholes exceed the present day environmental range in the southern African continental environment (≤ 5 TU). The significantly high tritium concentration found in borehole N 0001 indicates a source of artificial tritium within the landfill (Table 15). The only significant increase in tritium concentration between the two sampling periods is seen in the N 0002 samples, possibly indicating the encroachment of the polluted ground water from the landfill. The relative amounts of tritium in the boreholes will depend on the relative importance of rainfall infiltration and on the levels, and therefore influence, of the dams.

Carbon isotopes

A carbon-14 sample was not taken for borehole N 0001 in view of the complexity of the disturbed ground water chemistry within the actual landfill. The carbon-14 values for the other boreholes lie within the range of present day environmental values for southern Africa, and indicate ground water mean residence times of less than 30 years, confirming the tritium observations and the probability of leachate encroachment.

The $\delta^{13}C$ composition decreased in both borehole N 0002 and N 0003 by around 2 ‰ to -10 ‰, which is about the expected value for ground water. It therefore appears that in 1993, the main source of recharge to these boreholes was ground water.

6.4.5 Summary and Discussion

A decrease in pH, conductivity and alkalinity is found with distance down-gradient from the landfill site. The increase in the conductivity of the 1993 samples of boreholes N 0001 and N 0003 once sufficient water had been removed from the borehole, is further evidence of the need to sufficiently purge a borehole before sampling. It is clear from the extremely high conductivity and the general poor condition of water pumped from borehole N 0001 that the ground water below and within the landfill site is severely polluted. This borehole further indicates that this landfill is in the stage of acid formation, and has not yet completely reached reducing conditions, although it appears as if reducing conditions have been initiated as the nitrate concentration is very low.

In both the environmental isotope and hydrochemical data, boreholes N 0002 and N 0003 are seen to respond quite rapidly to surface water in the catchment and farm dams. In 1992, the catchment dam was drying and the farm dam water stood close to the two boreholes. At that stage, the stable isotope contrast between them was greatest, with N 0002 showing a strong evaporation signal. In 1993, with the catchment dam dry, and

the farm dam level considerably lower, the two boreholes had closely similar, albeit still evaporative, stable isotope signals. At this stage too, the carbon-14 values have become identical, as are the $\delta^{13}C$ values which are considerably lighter than in 1992. These values suggest more uniform ground water in the vicinity of these two boreholes, and that ground water is the principal source of recharge to these two boreholes.

The detail obtainable, and the consistency of the isotope and chemical data, can probably be related to the proper construction of the monitoring boreholes, which allows for sampling according to sound principles, as stated in Section 1.4.

CHAPTER 7

7 THE BLOEMFONTEIN SOUTHERN LANDFILL SITE

7.1 Introduction

7.1.1 Location

The Bloemfontein Southern landfill site is situated to the south of the city, and covers an area of approximately 44 hectares, sloping slightly to the south-west at approximately 4° (Bekker, 1992). It is mainly a G:M site, with a small area to the east designated as a H site (see Chapter 1.1).

The nearest private boreholes in the area are at a distance of 600 to 800 metres in a south-westerly direction from the site (Figure 23), used for domestic supply, poultry and the irrigation of trees and a vegetable garden (Bekker, 1992).

7.1.2 History and Development

The refuse at the Class 2 site is placed in cells and compacted with clayey soil (Bekker, 1992). The smaller Class 1 site is designed for toxic waste which is stored in concrete containers designed by the Department of Water Affairs (Seymour *et al.*, 1991). The

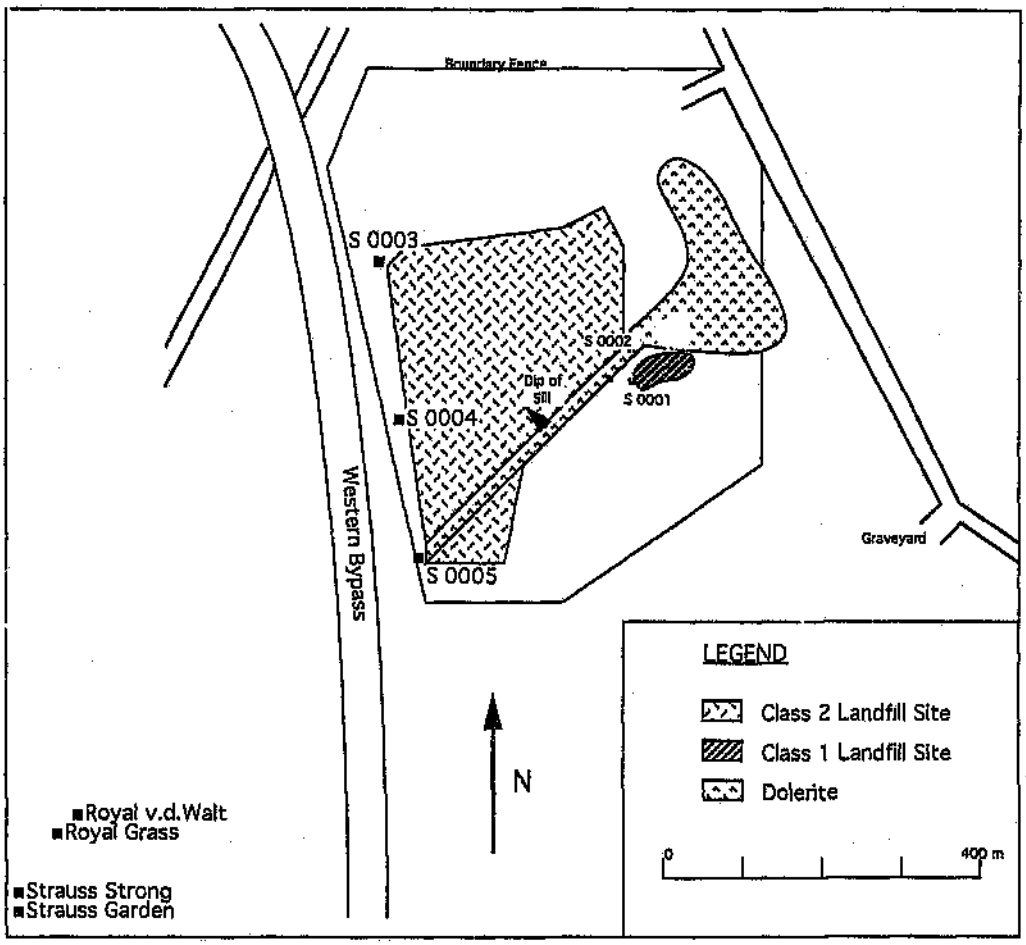

Figure 23. Map of the Bloemfontein Southern landfill site, showing the location of the pollution monitoring boreholes, as well as the farm boreholes

waste is placed in non-corrosive drums within the concret , which are stored in cells and then covered with soil.

Three pollution monitoring boreholes, S 0003 to S 0005, were drilled down-gradient of the Class 2 si. within 15 metres of the western boundary, between the highway and the landfill site (Figure 23). Two more boreholes, S 0001 and S 0002, are situated about 10 metres in an easterly direction and are intended to monitor the Class 1 site. Borehole S 0001 was drilled into sediments, while boreholes S 0002 to S 0005 were drilled into dolerite.

7.2 Hydrogeology

The landfill site is underlain by Karoo sediments of the Beaufort Group, mainly siltstones with lenses of shale and sandstone. A thin dolerite dyke is situated about 15 metres south-east of the landfill and slopes about 25° to the north-west (Figure 23).

Borehole S 0001 was drilled almost entirely through horizontally bedded siltstones interbedded with thin layers of clay, sandstone and shale. A 5 metre layer of dolerite located above these siltstones was encountered in borehole S 0002. In borehole S 0003, weathered dolerite was encountered at a depth of 5 metres, while boreholes S 0004 and S 0005 made contact with the weathered dolerite at a depth of about 35 metres (Bekker, 1992).

Boreholes S 0001 to S 0003 were dry during the drilling operations, although water did seep into the holes subsequent to drilling and a rest level was measured (Table 16). S 0004 had a yield of 0.2 l/s, S 0005 had a yield of 0.3 l/s, while the borehole on the Royal Grass farm had a yield of 0.7 l/s. The expected direction of ground water flow is in a south-westerly direction, following the gentle surface gradient, towards the van der Walt farm, as well as several other farms. From slug tests, it was established that the

permeability of the underlying siltstones and shales is low, with a k-value in borehole S 0005 of 2.76 x 10^{-2} metres/day (Bekker, 1992).

The Southern landfill site has the potential to contaminate the ground water supplies used by the neighbouring farms. A dam at a pleasure resort is situated 1.2 kilometres south west of the site, and any contamination of this dam will most likely be detrimental to the health of the public.

Table 16. Rest levels and borehole depths of the pollution monitoring boreholes and farm boreholes

Borehole Number	Borehole Depths (m)	(below surface) 31/02/92 (m)	30/08/93 (m)
S 0001	46.0	28.5	
S 0002	35.0	28.5	
S 0003	17.0	14.8	
S 0004	40.0	18.1	22.3
S 0005	38.0	21.4	24.7
Royal Grass	36.0	22.0	
Royal v.d. Walt		22.0	

7.3 Previous Studies

A study on this landfill site initiated by the Bloemfontein Health Department was carried out by the Institute for Ground Water Studies at the University of the Free State, and has included the sinking of the pollution monitoring boreholes down-gradient of the

site, as well as an interpretation of the local geology (Bekker, 1992). These boreholes, as well as two farm boreholes, Royal van der Walt and Royal Grass, were sampled for chemical analyses. The area was classified as good for a landfill site, mainly because of the relatively deep water table, the low permeability of the underlying geology, the gentle slope, and the fairly large distance to the nearest farm.

It was found that all five pollution monitoring boreholes had high aluminium and iron concentrations, the iron ranging from 0.16 to 0.27 mg/l, and the aluminium from 0.25 to 0.44 mg/l, all above the recommended limit for drinking water. Borehole S 0005 also had significantly high chloride, magnesium and calcium values (Table 18), and borehole S 0001 had a fluoride concentration of 7.9 mg/l, higher than the crisis limit given in Table 6.

The Royal van der Walt sample had magnesium, chloride and calcium concentrations above the recommended limit for drinking water, while in the Royal Grass sample, iron, manganese and arsenic were higher than the recommended limit, with values of 0.67, 0.055 and 0.204 mg/l respectively.

7.4 Results of the Present Study

7.4.1 Introduction

An environmental isotope and hydrochemical survey of pollution monitoring and farm boreholes was conducted in January, 1992, and again in August, 1993. Two pollution monitoring boreholes, S 0004 and S 0005, as well as 4 private boreholes at farms down-gradient from the landfill site, were sampled for environmental isotopes. The submersible pump was lowered to a depth of 34 metres for both the pollution monitoring boreholes, and the volume of water standing in the holes was purged before

a sample was taken. Boreholes S 0001 and S 0003 had very low yields, and borehole S 0002 was dry.

During the 1992 sampling period, it was observed that a large part of the south section of the landfill had been eroded away due to heavy rains. A substantial amount of surface run-off had occurred, flowing in a south easterly direction towards borehole S 0005, as well as through a culvert under the highway, in the direction of the farms. The clayey soil in the vicinity of borehole S 0005 shows extensive cracks when dry, allowing the run-off easy access below the surface. The damage to the landfill had been repaired by the time the 1993 samples were taken.

The farm boreholes sampled are at the van der Walt farm, the Royal Grass farm and two boreholes at the Strauss farm. One of the Strauss boreholes, Strauss Garden, was lower yielding and used to irrigate the garden, and the other, Strauss Strong, a borehole with a much higher yield, used to irrigate the farm.

7.4.2 Hydrochemistry

Well-Head Measurements

Both pollution monitoring boreholes S 0004 and S 0005 had relatively high alkalinities, conductivities and temperatures in August, 1993 (Table 17). Borehole S 0005 had a significant increase in alkalinity since January 1992, when a value of 12.9 meq/l was recorded. However, there was no change in the conductivity of this borehole between January, 1992 and August, 1993. The alkalinity and conductivity of the farm borehole samples decrease with distance from the landfill site.

Table 17. Well-head measurements for the pollution monitoring boreholes and farm boreholes (August, 1993)

Borehole Number	Alkalinity (meq/l)	Conductivity (mS/m)	pH	Temperature (°C)
S 0004	10	100	6.60	20.5
S 0005	15.5	400	6.80	22.5
Royal v.d.Walt	10.2	200	7.15	18.5
Royal Grass	8.9	145	6.80	19.6
Strauss Garden	6.3	100	6.85	19.4
Strauss Strong	6.6	97	7.05	19.3

Ionic Concentrations

The major ion concentrations for the different sampling dates for both the pollution monitoring boreholes and the farm boreholes are given in Table 18. Calcium is the dominant cation in the pollution monitoring boreholes, as well as in the farm boreholes (Figures 24 and 25), with the exception of boreholes S 0001 and S 0003 which are sodium dominant, and the 1994 sample from the Royal Grass borehole which has magnesium as the dominant cation. Bicarbonate is the dominant anion in boreholes S 0001, S 0003, S 0004, the Strauss borehole, and the 1992 and 1993 samples from the Royal Grass borehole. In boreholes S 0005, Royal van der Walt, and the 1994 sample from Royal Grass, chloride is dominant.

The main differences between the boreholes are apparent in the concentrations of calcium, magnesium and chloride and, to a much lesser extent, in bicarbonate, while the other ionic concentrations are relatively similar. Calcium, magnesium and chloride

Table 18. Major ion concentrations in meq/l for various sampling dates

Borehole Number	Date Sampled	Potassium (meq/l)	Sodium (meq/l)	Calcium (meq/l)	Magnesium (meq/l)	Chloride (meq/l)	Bicarbonate (meq/l)	Sulphate (meq/l)	Nitrate (meq/l)
S 0001	Jul 91	0.02	3.1	0.7	0.2	0.5	2.2	0.31	0.01
S 0001	Jul 91	0.00	3.1	0.6	0.1	0.4	2.2	0.31	0.01
S 0001	Aug 93	0.02	4.9	3.1	0.6	0.4	8.5	0.37	0.002
S 0001	Apr 94	0.02	6.6	3.0	0.6	0.4	9.5	0.14	0.002
S 0003	Jan 92	0.25	9.40	4.8	4.7	3.1	10.2	5.79	0.10
S 0003	Aug 93	0.15	5.6	1.2	5.3	2.1	5.3	4.47	0.002
S 0003	Apr 94	0.15	7.1	5.4	5.4	2.1	11.1	4.38	0.002
S 0004	Jan 92	0.09	2.0	3.0		1.4	5.3	0.61	0.01
S 0004	Aug 93	0.05	2.2	5.1	3.7	1.6	9.8	0.08	0.00
S 0004	Apr 94	0.06	2.1	4.3	3.1	1.4	8.3	0.14	0.002
S 0005	Jan 92	0.21	4.1	24.4	14.8	27.1	11.2	0.37	0.01
S 0005	Aug 93	0.08	3.7	23.2	18.0	27.4	15.4	0.69	0.002
S 0005	Apr 94	0.08	3.6	20.3	14.8	20.7	17.5	0.29	0.002
v/d Walt	Jan 92	0.10	3.1	9.8	8.6	12.8	10.5	0.51	0.01
v/d Walt	Aug 93	0.07	2.8	10.2	9.2	11.2	10.0	0.71	0.002
v/d Walt	Apr 94	0.08	2.8	12.1	9.5	13.8	10.2	0.73	0.002
G. Royal	Jul 91	0.05	2.9	6.2	4.4	4.6	8.4	0.43	0.01
G. Royal	Aug 93	0.07	2.9	7.9	6.3	7.5	8.2		0.01
G. Royal	Apr 94	0.07	3.0	5.4	6.4	8.2	6.0	0.51	0.002
Strauss	Aug 93	0.06	2.7	4.5	3.5	2.9	6.7	0.88	0.04
Strauss	Apr 94	0.07	2.4	4.4	3.7	3.3	6.6	0.71	0.04

also show the greatest variability at the Northern landfill site. These three ions are therefore taken as characterising leachate formed in these landfill sites.

The boreholes situated down-gradient of the landfill site, namely borehole S 0005 and the four farm boreholes, are shown in the Schoeller diagram in Figure 24. From both the well-head measurements and the major ion concentrations, it appears as though borehole S 0005 is the borehole which most prominently reflects pollution from the landfill (Figure 24). Ion concentrations for boreholes up-gradient of the landfill, namely S 0001 to S 0004, are shown in Figure 25. Borehole S 0001, which has a low mineralisation, is a Na-HCO_3 dominant ground water and is, for the purpose of comparison, taken as "natural" ground water (Figure 25). Taking these two cases as end-members, the modified Schoeller diagrams suggest that the farm boreholes, namely Royal van der Walt, Royal Grass and Strauss, show the influence of the polluted end-member in decreasing proportion with their distance from the landfill. This is most clear in the magnesium and chloride concentrations, which have been illustrated in Figure 24. The 1993 and 1994 samples from Royal Grass have significantly higher magnesium and chloride concentrations than in 1992, probably indicating the encroachment of these pollutants. The 1994 sample from the Strauss borehole also displays a slight increase in both magnesium and chloride. Boreholes S 0003 and S 0004 show little effect from the landfill, both with low magnesium and chloride concentrations relative to sodium and bicarbonate.

In the cation triangle in the Piper trilinear diagram (Figure 26), there is a general increase from 1992 to 1994 in the percent magnesium in all the boreholes. In the anion triangle, all the pollution monitoring boreholes increase in percent bicarbonate and decrease in chloride, while the farm boreholes decrease in percent bicarbonate and increase in chloride. The polluted and "natural" ground water end-members can be clearly seen in the diamond triangle, with borehole S 0001 plotting in the sodium

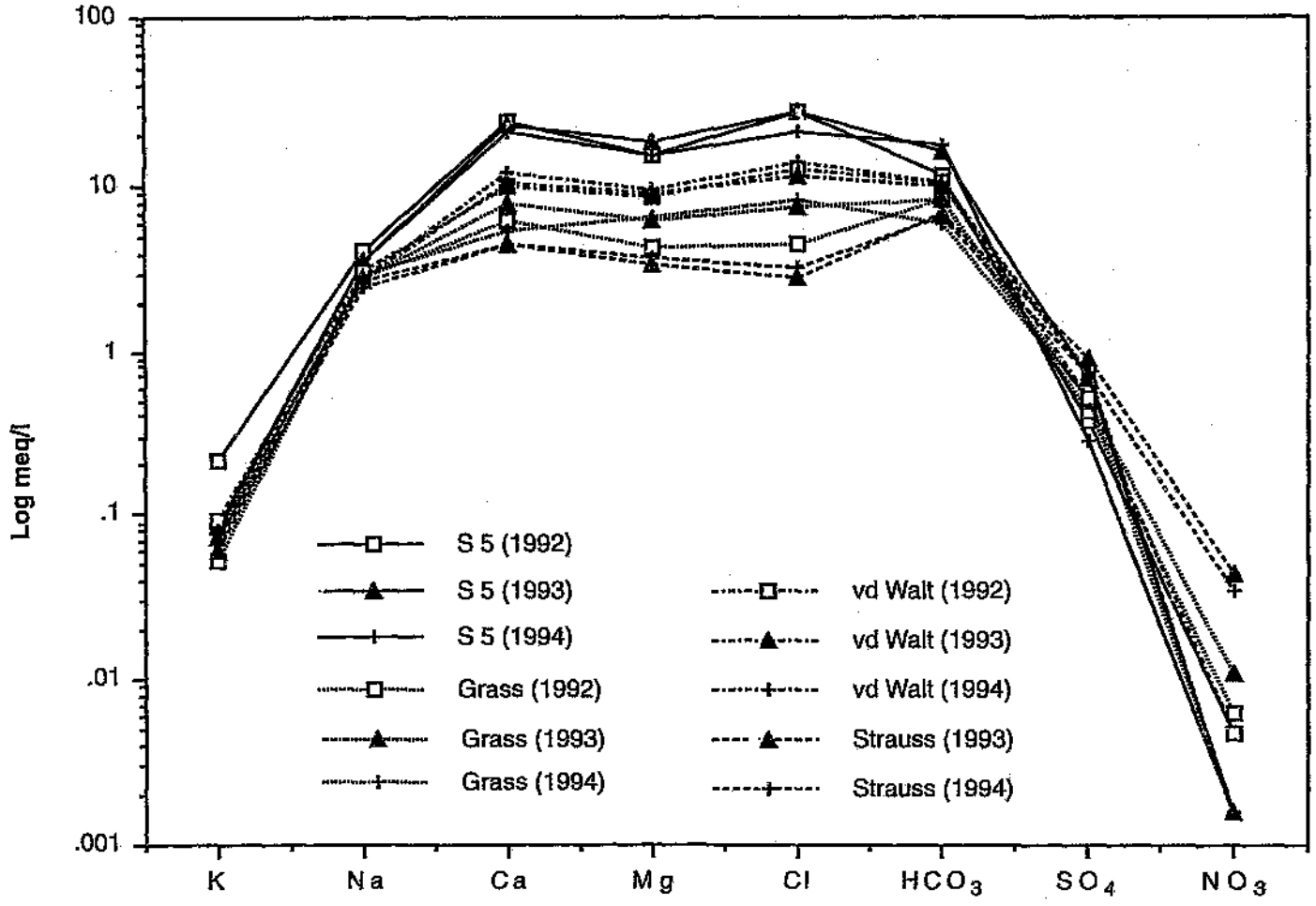

Figure 24. Modified Schoeller diagram for the down-gradient pollution monitoring and farm boreholes

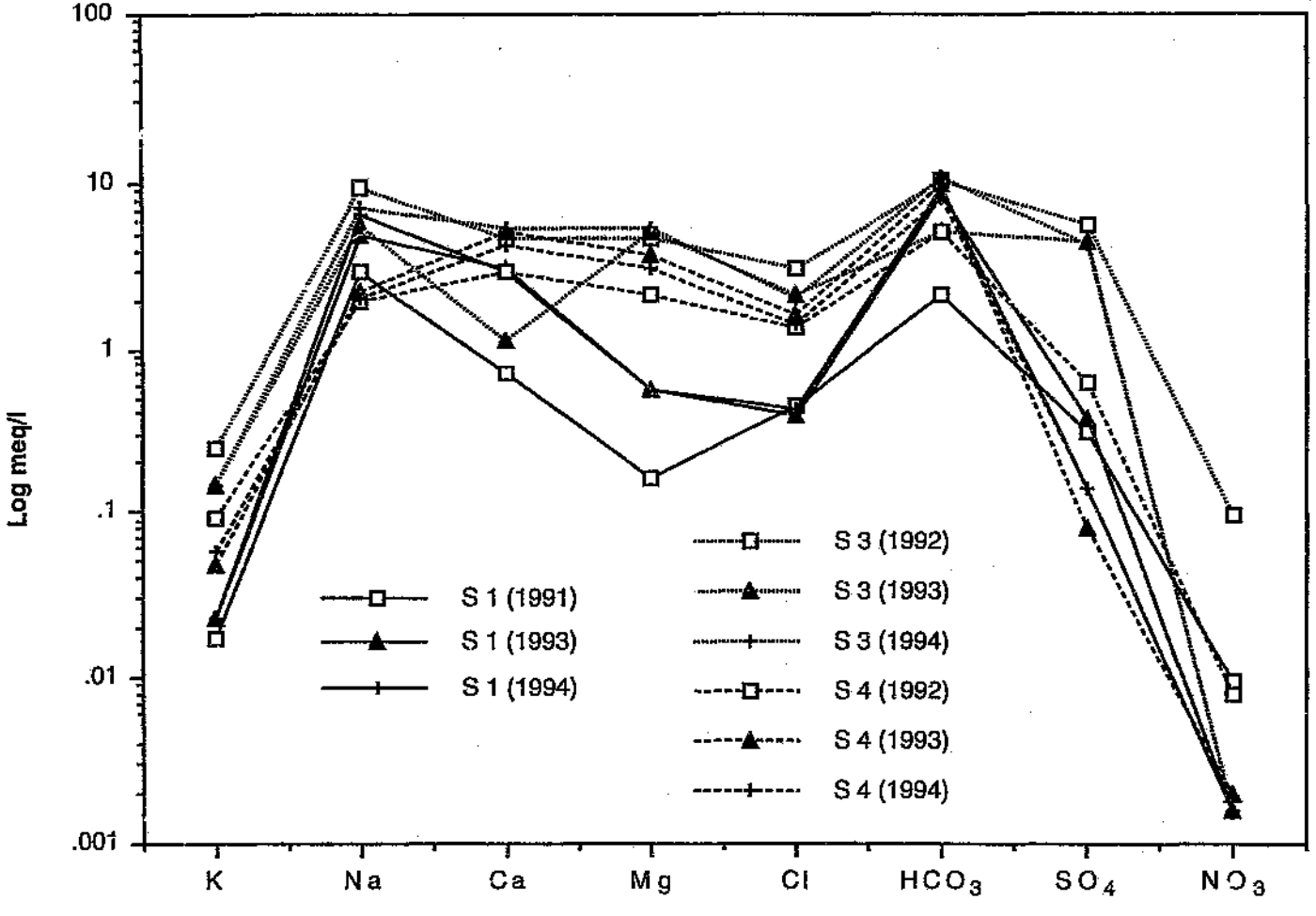

Figure 25. Modified Schoeller diagram for the up-gradient pollution monitoring boreholes

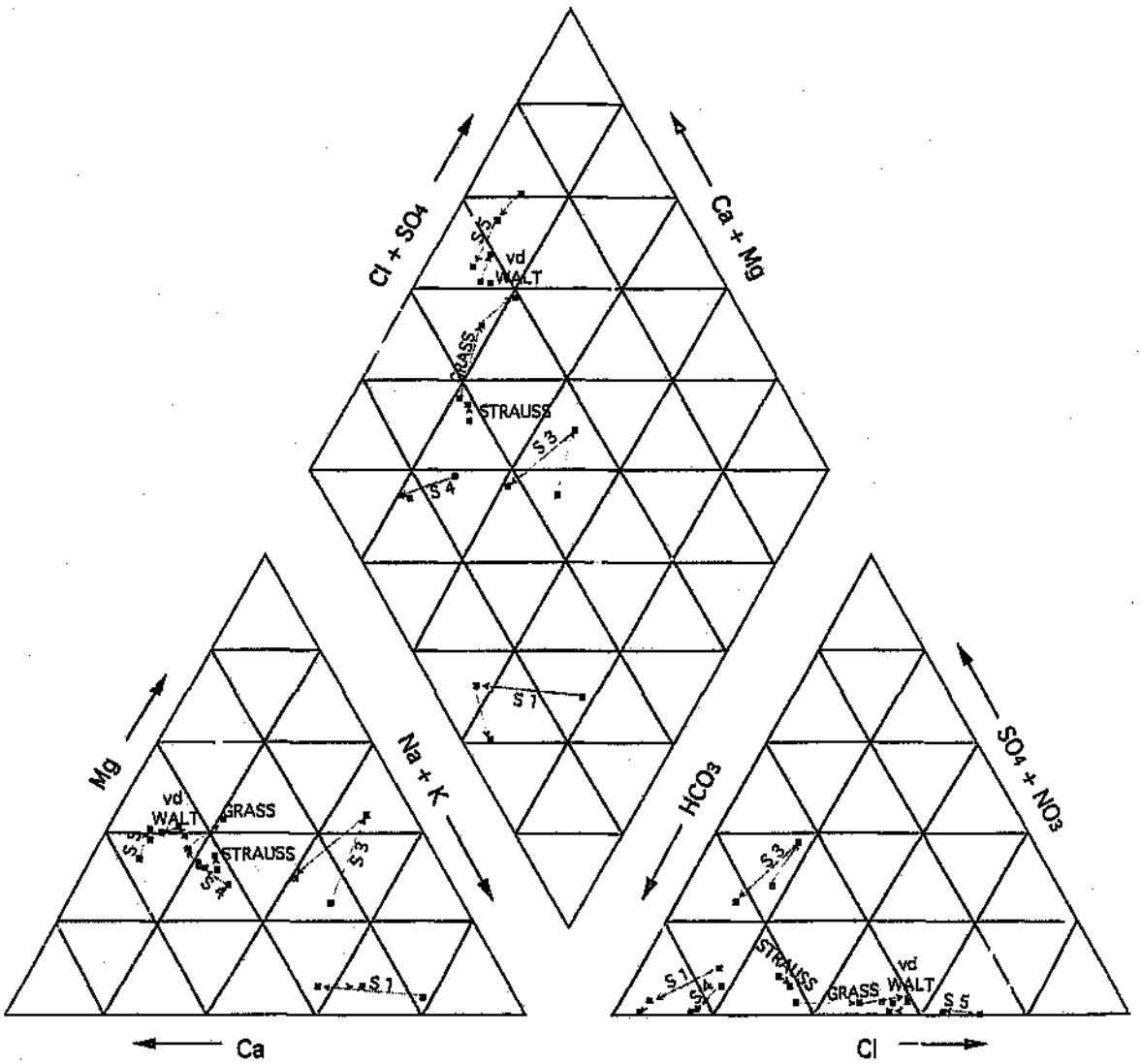

Figure 26. Piper trilinear diagram for the pollution monitoring boreholes and the farm boreholes at the Bloemfontein Southern landfill site

bicarbonate field. The farm boreholes are between the end-members, plotting in sequence from the polluted end-member with distance from the landfill.

7.4.3 Environmental Isotopes

Stable Isotopes

Table 19. Stable isotope values for the pollution monitoring boreholes and farm boreholes for 1992 and 1993

Borehole Number	δD (‰)		$\delta^{18}O$ (‰)	
	Jan 92	Aug 93	Jan 92	Aug 93
S 0004	-36.9	-36.6	-5.87	-5.73
S 0005	-31.2	-31.5	-5.08	-4.90
Royal v.d. Walt	-36.1	-35.8	-5.88	-5.65
Royal Grass	-41.0	-43.7	-6.24	-5.95
Strauss Garden		-42.0		-6.02
Strauss Strong		-41.8		-6.10
S 0001	-37.1		-5.21	

The stable isotope values for both the pollution monitoring boreholes and the farm boreholes are given in Table 19. Three separate groups plot along the meteoric water line (Figure 27). The more polluted monitoring borehole sample, S 0005, is the isotopically heaviest of the samples, followed by S 0004 and the Royal van der Walt sample. The third group, which has the lightest isotopic composition, consists of the Royal Grass and the Strauss samples. This same trend was found in the well-head

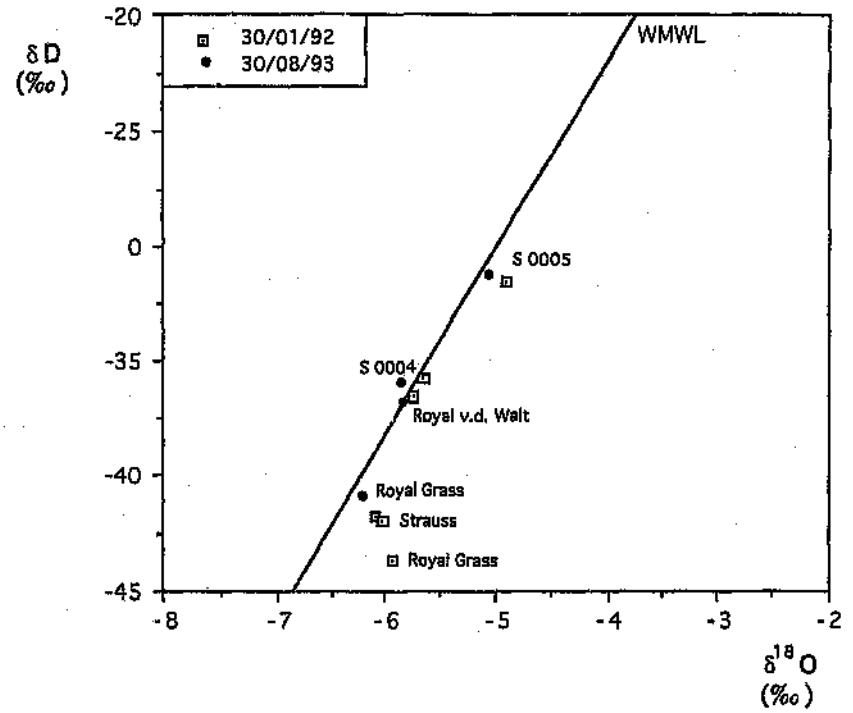

Figure 27. Plot of δD against $\delta^{18}O$. Also shown is the world meteoric water line (WMWL)

measurements, the higher alkalinity and conductivity values corresponding to the heavier stable isotope compositions. No evaporation is evident in any of the samples at this landfill site. Findings in this study were similar to those of Fritz *et al.* (1976), where the more polluted ground water samples have slightly more positive δD and $\delta^{18}O$ values, probably due to gas exchange within the landfill. The Royal Grass sample has a significant shift in both the δD and $\delta^{18}O$ composition, the δD decreases, while the $\delta^{18}O$ increases. This is possibly the result of mixing between water recently infiltrated and older, isotopically lighter ground water.

Tritium

Table 20. Radioactive isotope values for the pollution monitoring boreholes and farm boreholes for 1992 and 1993

Borehole	^{3}H (TU)		^{14}C (pMC)		$\delta^{13}C$ (‰)	
Number	Jan 92	Aug 93	Jan 92	Aug 93	Jan 92	Aug 93
S 0004	0.9±0.2	1.2±0.3	92.8±1.1	112.2±0.7	-6.0	-17.4
S 0005	28.1±1.5	40.1±1.9	123.5±1.0	125.5±0.7	+0.9	+0.1
Royal v.d. Walt	5.7±0.6	5.8±0.5	117.3±1.1	118.2±0.7	-2.5	-2.3
Royal Grass	1.9±0.3		107.7±0.8		-4.2	
Strauss Garden		0.8±0.3				
Strauss Strong		1.0±0.2				
S 0001	1.2±0.2					

Borehole S 0005, which has a tritium concentration considerably higher than the normal environmental values expected in the southern African environment, increases between 1992 and 1993 by 12 TU, indicating an ongoing influx of artificial tritium into

the ground water (Table 20). The tritium values of the other boreholes remain virtually unchanged. Borehole S 0004, the Strauss boreholes, and the Royal Grass sample, have tritium concentrations in the range of 0.8 to 1.9 TU, indicating mean residence times of 25 to 50 years. However, the Royal van der Walt sample has a tritium content at the upper limit of the normal environmental range, suggesting incipient contamination by the landfill.

Carbon-14

The highest carbon-14 concentration is found in the polluted borehole S 0005. The value of 125 pMC is out of normal environmental range, and could be ascribed to artificial carbon-14 leached from the landfill. The positive $\delta^{13}C$ and high alkalinity of the ground water supports this interpretation.

The increase of some 20 pMC over 18 months observed in the nearby borehole S 0004, appears to occur due to a quite different mechanism. It is accompanied by a decrease in the $\delta^{13}C$ value by some 10 ‰, which suggests a natural, biogenic source. It is not accompanied by any indication of pollution. This unequipped borehole was only pumped for an hour or so during sampling. This does not induce much displacement of the surrounding ground water. It is possible that during the widely spaced pumping periods, very recent ground water was only gradually drawn into the borehole. Although the ground water is being fairly rapidly recharged at this borehole position (mean residence time < 30 years), it would appear that no significant leachate from the landfill is entering the ground water here some 200 metres N of polluted borehole S 0005.

The Royal van der Walt borehole results remained unchanged. The high ^{14}C, ^{3}H and only slightly negative $\delta^{13}C$ values, may all be indicative of influence from the landfill.

7.4.4 Summary and Discussion

Borehole S 0005 is the most polluted borehole at the Southern landfill site, with the highest alkalinity, conductivity and temperature of all the boreholes. This borehole is significantly more polluted than the other boreholes. The reason may be that at this point substantial run-off from the landfill, due to earlier subsidence and erosion, seeped through drying cracks in the surface clay material.

The three separate groups of stable isotope values which plot along the meteoric water line appear to correspond to the level of pollution found in each borehole. Borehole S 0005, which has the heaviest isotopic composition, is the most polluted, followed by S 0004 and Royal van der Walt. The other farm boreholes, namely Royal Grass, Strauss Strong and Strauss Garden, with lighter isotopic compositions, appear to have much lower levels of pollution. The latter values, which plot to the right of the MWL suggest a degree of evaporation before infiltration of rain water.

The extremely high tritium concentration measured in S 0005 is more than likely the result of artificial tritium within the landfill site, and is a useful tracer when following the movement of any pollution from the site. The Royal van der Walt borehole has a significantly higher tritium concentration than the neighbouring farm boreholes, and could either indicate the encroachment of the pollution from the landfill, or could be an indication of a very localised recharge source. The ionic ratios of this borehole show similarities with S 0005. It is estimated, taking into account the distance, the hydraulic gradient, and the low permeability of the underlying siltstones, that a pollution front in ground water would take about 200 years to cover this distance. Should these higher values in Royal van der Walt be due to pollution from the landfill, the polluted water would have to be transported on the surface. Such surface run-off was observed following a major rain storm in 1993.

The highest carbon-14 value and most positive $\delta^{13}C$ are found in S 0005. This suggests a source of artificial carbon-14, as well as tritium, in the landfill. The large increase in the carbon-14 concentration in borehole S 0004, accompanied by a just significant rise in tritium, is most likely the result of younger but unpolluted water being drawn into the borehole after pumping, due to natural recharge.

CHAPTER 8

8 SUMMARY AND CONCLUSIONS

8.1 Summary

8.1.1 Linbro Park Landfill Site

The down gradient boreholes in the Linbro Park landfill site were distinguished from the up-gradient boreholes by higher alkalinity, conductivity, temperature and pH, as well as higher ion concentration. The calcium ion concentration showed the greatest contrast between the up-gradient and the down-gradient boreholes, followed by the bicarbonate and sodium ion concentrations. These differences are attributed to a more mature hydrochemistry in the down-gradient boreholes.

Although there are no signs of evaporation from the stable isotope data, the two down-gradient boreholes have a more positive δD composition than the up-gradient boreholes, a phenomenon also seen by Fritz *et al.* (1976), and ascribed to gaseous diffusion. The tritium and ^{14}C data suggest recent ground water. The up-gradient boreholes, where pollution is evident, had shorter turnover times than the down-gradient boreholes, probably the result of activities on the neighbouring farms. There is no significant pollution evident in the down-gradient boreholes.

8.1.2 Waterval Landfill Site

$\delta^{18}O$ and ^{3}H elucidate various inputs into the streambed. Artificial ^{3}H appears to enter from leachate which seeps through weepholes in the culvert beneath the site, while more severe pollution from a leaking sewer in the streambed is indicated by . ^{S}O and field chemistry. Tritium levels slightly higher than environmental levels were seen in one of the private boreholes. It appears as if a source of recharge to this private borehole is by bank infiltration from the stream.

8.1.3 Bloemfontein Northern Landfill Site

A decrease in the pH, conductivity and alkalinity was found with distance from the landfill site. Severe pollution was found in ground water below the site, and the chemical composition indicated that the landfill is in the stage of acid formation, at the onset of reducing conditions.

The extremely polluted borehole has a stable isotope composition which plo' to the left of the MWL, probably the result of hydrogen isotopic exchange. The two down-gradient boreholes both respond rapidly to the surface water in the nearby dams. This is clear from the evaporative signal shifts observed in the stable isotope data, and is also seen in the hydrochemical data. Artificial tritium is evident in all three monitoring boreholes, and an increase in the levels of the two down-gradient boreholes over the sampling period indicates pollution encroachment from the landfill. The carbon-14 results suggest mean residence times of less than 30 years, and the shift towards more negative $\delta^{13}C$ values in these two boreholes suggests that ground water is the principal source of recharge in these two monitoring boreholes.

8.1.4 Bloemfontein Southern Landfill Site

The most polluted borehole was found to have the highest alkalinity, conductivity and temperature. Surface water run-off from the landfill site seeped into cracks in the clay material in the area around this borehole, causing considerable pollution to the ground water.

The stable isotope compositions of the various boreholes correlate well with the level of pollution, the more polluted boreholes having a heavier stable isotope composition. The higher tritium concentration found in the polluted monitoring borehole acts as a useful tracer of pollution migration towards the farms. One of the farm boreholes has a tritium concentration higher than environmental values, and a similar ionic ratio to the polluted borehole. The only logical mechanism which could transport the polluted water over the distance between the landfill and the farm borehole is by surface run-off. High ^{14}C and $\delta^{13}C$ values were also found in the highly polluted monitoring borehole.

8.1.5 Pollution Monitoring

A number of the pollution monitoring boreholes in the Linbro Park and Waterval landfill sites could not be sampled as the casing, constructed from UPVC, is not durable enough, particularly in unconsolidated material, as well as in close proximity to trees where roots have been observed to puncture the casing. The slow recovery in all the boreholes also suggests poor hydraulic connection with the aquifer, with only the ground water in close proximity to the boreholes being sampled.

The bailing method used by the Johannesburg Municipality to sample these boreholes is believed to be inadequate. Only a small portion at the top of the standing water column is sampled, and this is in no way representative of the ground water at that time. The importance of purging a borehole before a sample is taken is clear from eg. the

increases in conductivity seen when the samples from the Bloemfontein northern landfill site were taken.

The significance of boreholes situated in mostly alluvial material alongside streams, rivers and catchment dams in the Linbro Park, Waterval and northern landfill sites is questionable as they most likely largely reflect changes in the surface water bodies, and not be representative of the ground water.

8.2 Conclusions

The ability to predict the potential pollution from a landfill site, as well as to trace the leachate in ground water from the site, has been illustrated using environmental isotopes and the hydrochemistry of the water. The isotopes have been essential in understanding and interpreting the hydrochemistry. Valuable information was provided for each landfill site, providing a clearer understanding of the movement of pollutants in the sub-surface.

In the case of the Linbro Park landfill site, it is possible to identify ground water down-gradient of the landfill site by its more positive δD isotope composition than the surrounding ground water, even though there is no sign of pollution. It is therefore possible to trace this water should pollution ever emanate from the site. As the stable isotope composition in the Bloemfontein southern landfill site correlates well with levels of pollution, and the isotopes are part of the water molecule, it is possible to detect incipient pollution before pollution levels are critical. It is often possible to identify sources of recharge when $\delta^{18}O$ and deuterium are plotted on the MWL. Sources of pollution other than from the landfill may also be detected using stable isotopes, provided the isotope compositions significantly contrast one another. Boreholes with evaporative stable isotope signals will suggest recharge from surface water bodies, while stable isotope compositions which are considerably heavier than surrounding

ground water, but plot along the MWL, indicate that a percentage of pre-1996 RW mains water is present in the ground water. The RW mains water signature clearly illustrated the ingress of mains water into the stream bed down-gradient of the Waterval landfill site. The field chemistry supported the increase in the level of pollution, and it was therefore possible to identify the sewage mains situated along the stream bed.

The elevated artificial tritium levels in the Waterval and Bloemfontein northern and southern landfill sites have been shown as the ideal tracer of leachate migration. This may be true of many other landfill sites, provided such a signal exists. In all three cases, pollution from the landfill was traced down-gradient of the site using tritium, even when the ionic concentrations were not at critical levels. A private borehole downstream of the Waterval landfill site was found to have slightly higher than environmental levels of tritium. A farm borehole down-gradient of the Bloemfontein southern landfill site which had slightly elevated tritium levels, also had a similar ionic ratio to the polluted monitoring borehole, clearly identifying incipient pollution. The hydrochemistry was also useful to differentiate between ground water pollution from the Bloemfontein northern landfill site, and pollution from a neighbouring mine, which had a contrasting ionic signal.

Several of the existing pollution monitoring networks were no longer functioning. Boreholes were often inadequately cased, or were poorly situated and therefore did not provide samples which were a true representation of the local ground water conditions at the time. It was also found that these boreholes had been inappropriately sampled.

In the absence of artificial tritium, environmental (rain) tritium is an excellent tool to determine the turnover times of ground water obtained from monitoring boreholes. This is important in assessing the representativeness of the sample of the ground water at the time of sampling. Even when recent ground water turnover times are observed, as in the case of the Linbro Park landfill site, it is also important to assess the

hydrochemistry, as it was expected that leachate would be entering the ground water. This remains uncertain, however, because of the poor sampling conditions encountered at this site.

Carbon-14 is a further means of determining turnover times in the absence of an artificial source once the limit of tritium is exceeded. Changes in the $\delta^{13}C$ are useful in determining the source of carbon in ^{14}C samples. Values more positive than -5 ‰ for ground water in equilibrium with atmospheric CO_2 suggest the influence of an artificial carbon source.

CHAPTER 9

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APPENDIX

The mean residence time (MRT) of the ground water has been determined by applying the exponential model. The aquifers in this study may all be regarded as phreatic, and therefore receive diffuse recharge. The exponential model, also known as the pure mixing model, describes the movement of ground water in an unconfined isotropic aquifer which receives uniform recharge over its entire area. In fact, aquifers are not isotropic and water components from different recharge events do not mix in the aquifer (see Figure A). When the aquifer is penetrated by a borehole, it is assumed that the aquifer contributes water from different levels into the borehole during sampling, and produces complete mixing from the different flow (IAEA, 1996). The isotopic concentration c(t) of the radionuclide introduced with recharge is given by (Verhagen *et al.* 1991):

$$c(t) = e^{-\lambda t} \cdot \int_0^{\infty} c_0(t - t_u - t')e^{-\lambda t'} f(t', P_i)dt'$$

where: t = time of sampling

λ = decay constant of the environmental isotope

t_u = transit time (time span between the entry of the isotope from the atmosphere into the unsaturated zone and reaching the saturated zone)

$c_o(t - t_u - t')$ = the input function which represents the concentration at which the relevant environmental isotope is present in the ground water recharge.

$f(t', P_i)$ = the system response function which describes the behaviour of the ground water where t' is the transit time of a ground water volume element since its entry into the saturated zone and P_i is one or more hydrogeologically relevant parameters. For the exponential model:

$$f(t', P_i) = \frac{1}{\overline{T}} e^{\frac{t'}{\overline{T}}}$$

$\overline{T}$ = the mean residence time of the mixed water

Therefore, the expression for the exponential model for the concentration of a radioactive tracer is as follows:

$$c(t) = \frac{e^{-\lambda t_u}}{\overline{T}} \int_0^{\infty} c_o(t - t_u - t') e^{-(\lambda + \frac{1}{\overline{T}})t'} dt'$$

The input concentration c_o, in precipitation in the case of tritium, or from atmospheric CO_2 in the case of carbon-14, varies with time, particularly since thermonuclear testing was carried out in the mid-fifties (Figures B(i) and (ii)). Therefore, to determine the input function for tritium, the distribution in precipitation must be taken into account. Applying the average tritium concentrations measured in precipitation in Pretoria since 1958 (IAEA, 1992), the mean residence times in this study were calculated. Figure C illustrates the variation in tritium concentration relative to the mean residence times.

Ambiguities may arise in the interpretation of the tritium data, as in the range between 2.5 and 4 T.U., two mean residence times are possible, as the peak passes through these values (see Figure C). Also, in deeper boreholes the resulting mixing of the water often results in tritium values close to zero, with a corresponding mean residence values much higher than is feasible. Provided the initial carbon-14 concentration is known, the

input functions for the exponential model of carbon-14 and tritium may be plotted against each other, resulting in a curve illustrating the relationship between the two isotopes for different residence 'es. The carbon-14 data will therefore constrain any ambiguous tritium data.

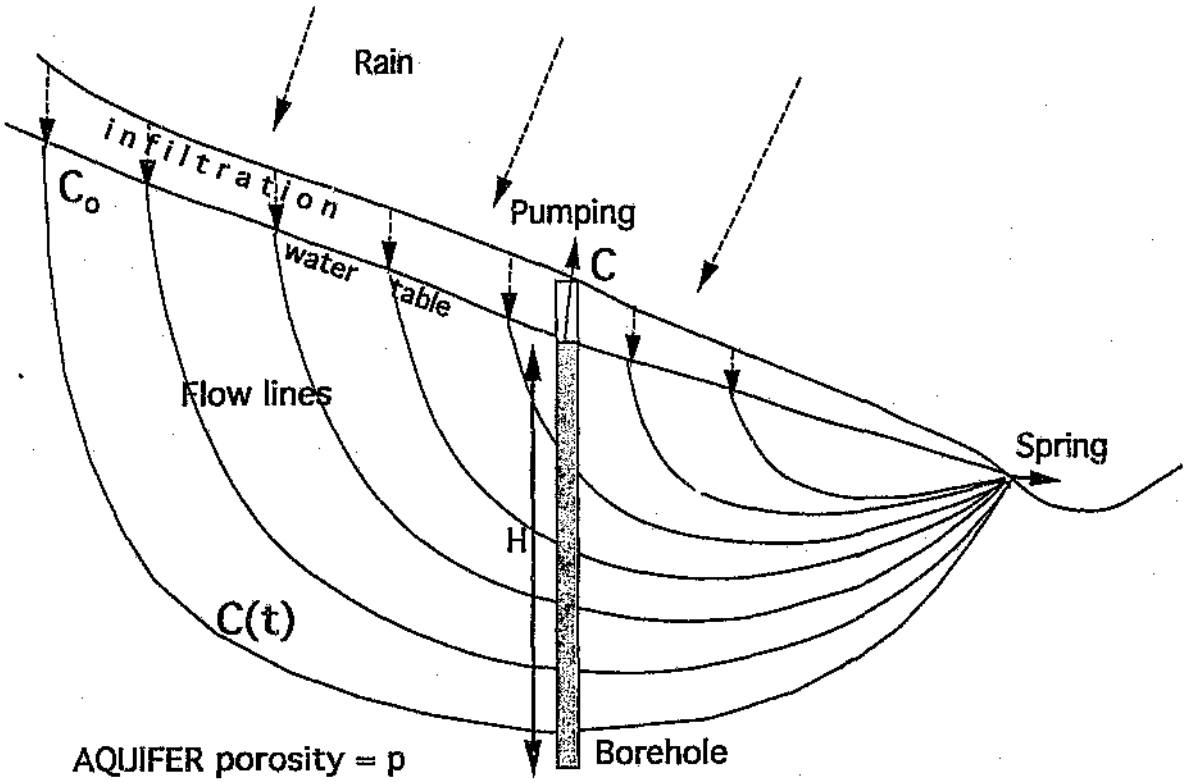

Figure A. Flow diagram through an unconfined aquifer illustrating mixing of flow lines at point of abstraction.

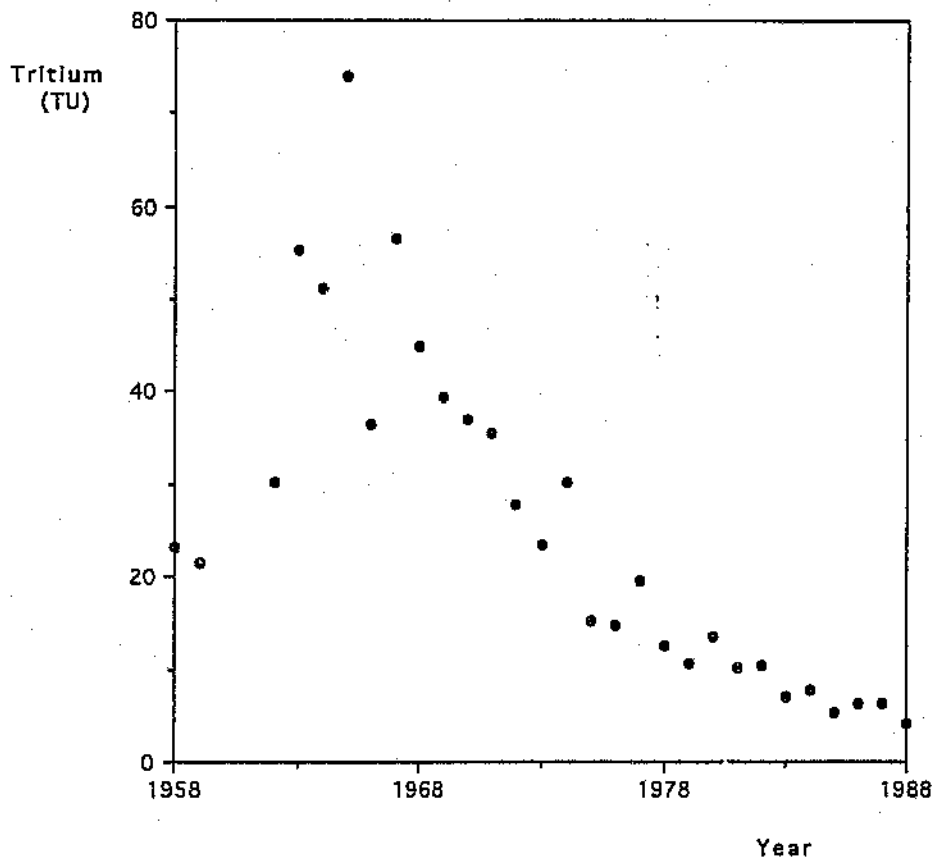

Figure B (i). Tritium concentration in South African precipitation since 1958 (IAEA, 1992)

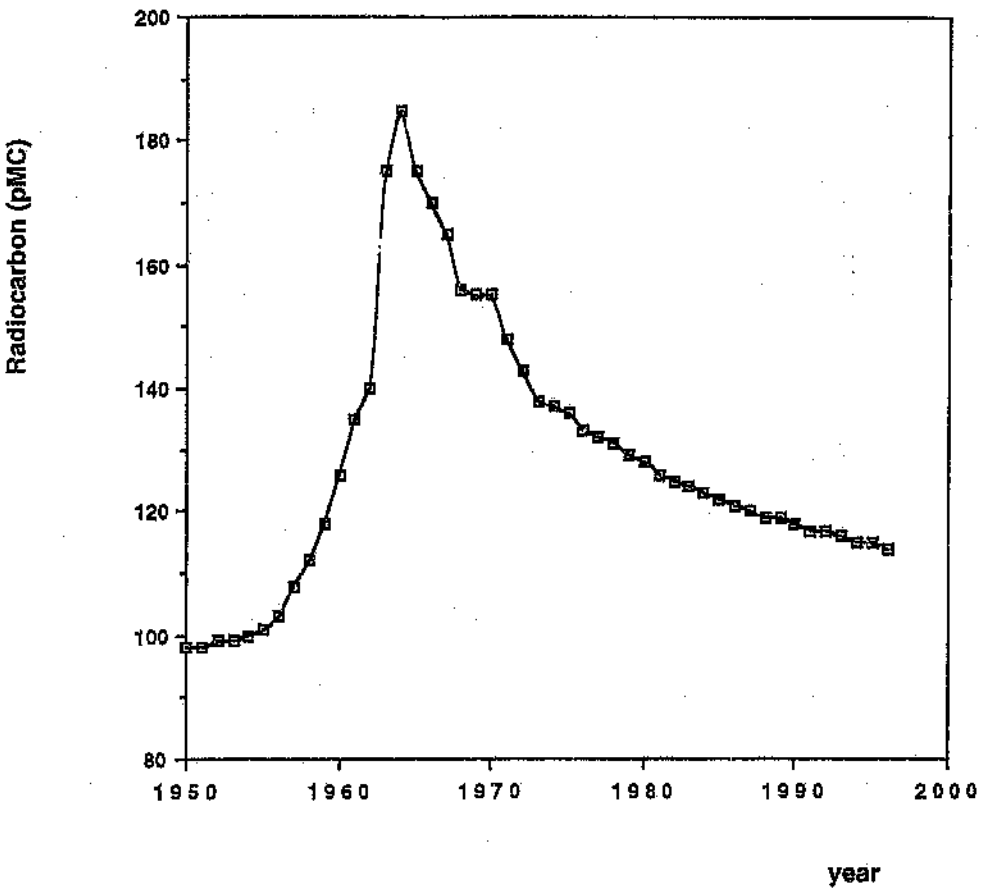

Figure B (ii). Atmospheric carbon-14 concentration in South Africa since 1950 (Zuber, 1986).

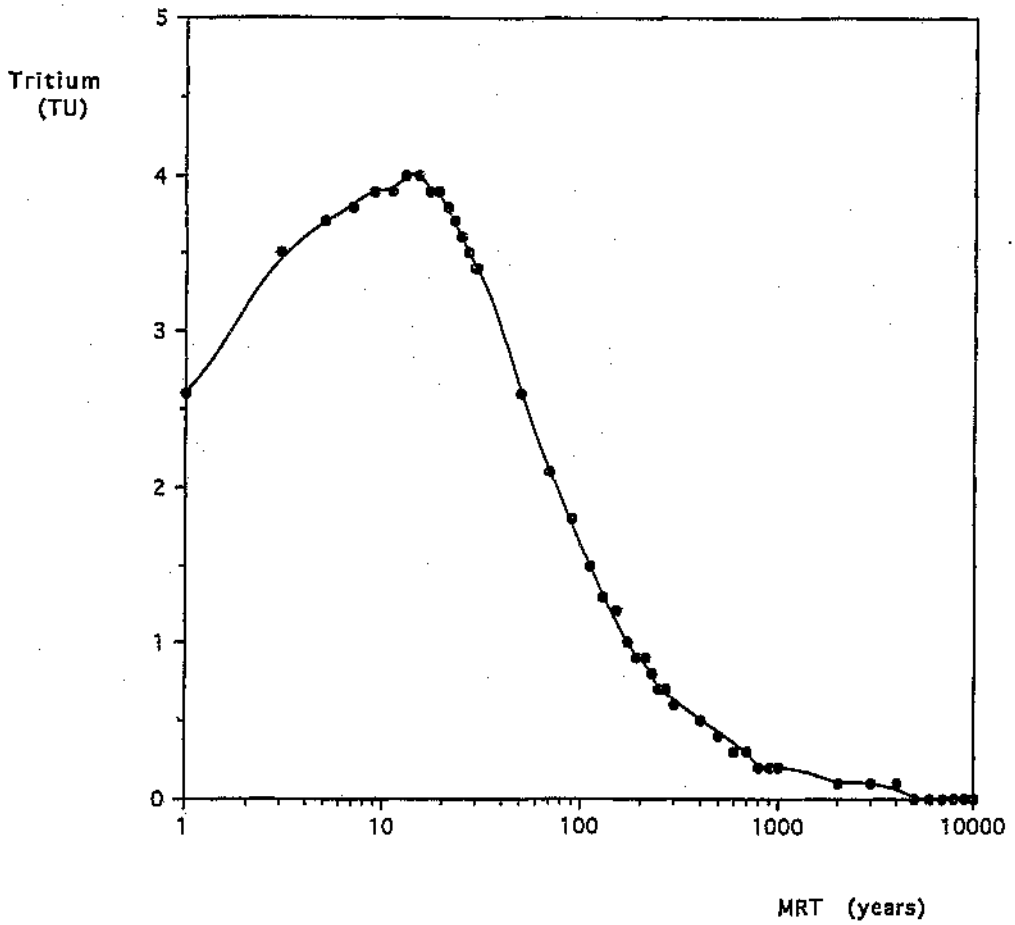

Figure C. Plot of tritium concentrations expected in a well-mixed isotropic, phreatic aquifer as a function of mean residence time. Concentration calculated using the exponential model and values of tritium in Pretoria rain (Figure B) as the input function.