ASSESSMENT OF THE IMPACT OF AGRICULTURAL PRACTICES ON THE QUALITY OF GROUNDWATER RESOURCES IN SOUTH AFRICA

by

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Report to the Water Research Commission on the Project
"Assessment of the Impact of Agricultural
Practices on the Quality of Groundwater
Resources in South Africa"

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WRC Report No	:	641/1/99
CSIR Report No	:	ENV/S-C 98035

This report is dedicated to the memory of Tony Reynders, who as Groundwater Research Manager at the WRC, provided outstanding guidance and enthusiasm for this project.

ABBREVIATIONS

BMP	Best management practice
BPEO	Best practicable environmental option
Cd	Cadmium
cm	Centimetre
Co	Cobalt
COD	Chemical Oxygen Demand
DCD	Dicyandiamide
DNH&PD	Department of Nation Health and Population Development
DOC	Dissolved organic carbon
DWA&F	Department of Water Affairs and Forestry
EFMA	European Fertilizer Manufacturers Association
EPA	(United States) Environmental Protection Agency
FAO	(United Nations) Food and Agriculture Organisation
FREP	Fertilizer Research and Education Programme (California Department of Food and Agriculture)
g	Gram
GFP	Good Farming Practice
ha	Hectare
IAH	Intensive animal husbandry
ID	Internal diameter
IFA	International Fertilizer Industry Association
K	Potassium
kg	Kilogram
LAN	Limestone Ammonia Nitrogen
m	Metre
Mo	Molybdenum
N	Nitrogen
NH_4-N	Ammonia (concentration as Nitrogen)
NO_3-N	Nitrate (concentration as Nitrogen)
NSA	Nitrate Sensitive Area
p/c	Plate count
P	Phosphorus
PE	Potential evaporation
TDS	Total dissolved solids
TKN	Total Kjeldahl nitrogen
TMG	Table Mountain Group
$t.ha^{-1}.a^{-1}$	Tons per hectare per annum
WHO	World Health Organisation
WWTP	Waste water treatment plant.
Zn	Zinc

Executive Summary

1. BACKGROUND TO THE STUDY

1.1 Introduction

The agricultural sector has been identified as the biggest user of groundwater in South Africa (DWA&F, 1992). It is estimated that 78% of all groundwater abstracted is used for irrigation, 7% for rural domestic purposes and 6% for stock watering. Only 4% of groundwater abstracted is used in the urban environment. Traditionally, however, the urban users have received the most groundwater research and investigation attention. As a result, little is known about the status of groundwater used by the agricultural sector in South Africa.

2. PROBLEM STATEMENT AND STUDY OBJECTIVES

It is recognised worldwide that the agricultural sector contributes to diffuse contamination (Foster and Bath, 1986; Gui, 1987; Pionke *et al.*, 1988; Sommerfeldt *et al.*, 1988). Problems include irrigation practices, animal feed lots and the use of fertilizers, pesticides and herbicides. Point sources of pollution which exacerbate the problem include septic tanks and pit latrines and the disposal of household and agricultural waste products.

Research has been undertaken in specific areas where the quality of groundwater has had an impact on agricultural activities. Areas such as the Sprinkbok Flats, Breede River, Fish River (Middelton) and the Hex River Valley are some examples. On the whole, however, little quantified information is available as to the impact of agricultural practices on groundwater resources and the status of such resources.

Further, it was recognised by the Ground Water Quality Task Group that much could be achieved in terms of protecting groundwater from contamination by agricultural activities by education (Braune *et al.*, 1991). The transfer of information from the groundwater community to the agricultural sector is urgently required. In addition to quantifying the extent of the problem in South Africa, the research effort can be used as a means of preparing and transferring this sort of information. Approximately 90% (by volume) of groundwater users will thus benefit directly from this work.

The initial research objectives of the research programme were as follows:

a. To quantify the impact that agricultural practices have on the quality of groundwater resources in South Africa.
b. To identify which practices have the most serious impacts.
c. To identify practical and easy-to-implement strategies and practices which can be used to prevent or reduce contamination.
d. To prepare a non-technical booklet, in four languages, which transfers the information to farmers and members of the public associated with the agricultural sector.

A one year literature study was undertaken in order to provide some general information concerning the impact of agriculture on groundwater and present known cases of such contamination in South Africa. The study showed that little quantified information regarding known cases of groundwater contamination resulting from agriculture exists. It was, however, possible to highlight those agricultural activities which posed the most serious threat to the quality of South African groundwater resources. Based on the results of the literature study, the project Steering Committee proposed that the research objectives be revised.

The revised research objectives were thus as follows:

a. To confirm the contamination impact on groundwater quality resulting from those agricultural activities identified as posing the most serious threat, namely

    i. Intensive animal husbandry (IAH)
    ii. Use of sewage sludge as a fertilizer
    iii. Use of inorganic fertilizers
    iv. Irrigation
    v. Use of pesticides

b. To identify practical and easy-to-implement strategies and practices which can be used to **prevent** or **reduce** contamination resulting from agricultural activities.

c. To prepare a non-technical booklet, which transfers the information to farmers and members of the public associated with the agricultural sector.

Two research products resulted namely:

a. A *non-technical handbook* - which sets out practices that may be used to prevent or reduce groundwater contamination by agricultural activities ("Handbook of Groundwater Quality Protection for Farmers" - available from the WRC); and

b. A *scientific report* - which presents the findings of the literature study as well as the results of the individual investigations into the five most serious threats listed above (this report).

3. METHODOLOGY AND REPORT LAYOUT

Of the top five potentially contaminating activities, three were identified as not having been sufficiently studied in the field. These activities were intensive animal husbandry (IAH), use of inorganic fertilizers and sludge application to land. Three field study sites were chosen across the country for each activity and studied in detail. Several studies have already been conducted on the use of pesticides and the impacts of irrigation on groundwater quality in South Africa. It was not thought necessary to conduct further field research and this information was collated in the form of essays and is presented in this report.

Details of the methods, results and conclusions of the field investigations and summaries of the information collated in essays are presented in this report. The supporting data from the field studies are contained in an Appendices Volume (available from the CSIR). One of the main aims of this study was to inform farmers about contaminating activities and provide guidance on Good Farming Practices (GFPs). Information on GFPs was collated during course of the specialist studies and presented in an accessible format in a handbook. In addition to GFPs relating to the five activities researched in detail, a chapter on pollution at the farmstead (septic tanks, underground fuel tanks, etc.) has been included. Background information on the occurrence and vulnerability of groundwater and nitrate, the most common contaminant, has also been included in the handbook.

4. SUMMARY OF MAJOR RESULTS AND CONCLUSIONS

An impact on groundwater quality was seen at all the field study sites with vulnerable, shallow, unconfined or semi-confined conditions. Nitrate was the most common agricultural contaminant evident in groundwater sampled. Nitrate distribution and isotopic analyses of NO_3-N indicated the most important sources to be sludge, manure and soil biota. Elevated DOC levels were associated with sludge application and IAH. Potassium, ortho-phosphate and microbiological indicators of faecal pollution, contaminated groundwater as a result of IAH practices.

The greatest impact was seen at a site where **sludge** was applied to agricultural land. Nitrate levels were elevated to 268 $mg.l^{-1}$ in the dolomitic aquifer directly beneath the sludge applied area and were persistent to a level of 30 $mg.l^{-1}$ to a distance of greater than 1 km. However it should be noted that sludge was applied at this site primarily as a means of disposal and application rates were not tied to crop requirements. There was therefore a long history (15 years) of over application. At the other sludge sites, where sludge was used to condition sandy soils, limited increases in nitrate levels were seen. At one site, shallow groundwater (< 3mbgl) was contaminated to a maximum of 38 $mg.l^{-1}$ but this was not persistent with depth. This site also had a long history of application. At the other site, only a 2 $mg.l^{-1}$ increase in nitrate levels was noted during the first season of application. Cumulative impacts of sludge application therefore appear to be significant.

Increased DOC levels were seen at 2 of the sludge field sites in shallow groundwater samples:
to 30 $mg.l^{-1}$ at site C and to 10 $mg.l^{-1}$ at site A.

Significant impacts on groundwater quality were seen at **IAH** site B, overlying a shallow alluvial aquifer. Pollution was associated with livestock concentration in pasture and irrigation of effluent. The highest nitrate levels seen at this site were 156 $mg.l^{-1}$. Increased levels of DOC, potassium, ortho-phosphate and faecal coliforms were also seen. IAH site A showed some contamination by nitrate and faecal streptococci. Site C showed no contamination as a result of natural aquifer protection and good practices.

Limited impacts on groundwater quality were found to be associated with the application of inorganic **fertilizers**. High nitrate levels (>10 $mg.l^{-1}$) were seen at all the fertilized sites, however, the source of this nitrate was not exclusively inorganic fertilizers. Fertilizers are believed to contribute to high nitrate levels but other activities, such as the application of organic fertilizers and deep rip ploughing are also thought to play a significant role.

The literature reviewed in the essay on the impact of **irrigation** showed that percolates from the root zone and irrigation return flow can cause the salinisation of groundwater in irrigated aquifer systems. Significant salinisation has been reported in groundwater underlying the irrigated lands of the Great Fish-Sundays River basin (TDS increased from 2 000 to 3 400 $mg.l^{-1}$). At the Vaalharts irrigation scheme it is estimated that between 17 and 63 million cubic metres percolate to the water table annually carrying nearly 30 000 tons of dissolved salts. This has resulted in increased groundwater salinity, a rise in the water table and some water logging of soils.

The review of **pesticide** contamination of groundwater showed that limited, scattered information is available. In the early 1990s, a study showed that the herbicide atrazine was present in most rivers and dams in the maize producing areas of South Africa. These included the Olifants, Vals, Vaal, and Renoster rivers. This river water is used in many areas for irrigation and therefore where aquifers are recharged, acts as a source of pesticide contamination. Tests of the herbicides methochlor and terbuthylazine have shown that they leach beneath the root zone in a wide variety of South African soils.

5. RECOMMENDATIONS FOR FURTHER RESEARCH

Details of further scientific work related to the specialist studies are given in those chapters. It is recommended that longer term monitoring of groundwater quality and agricultural activities at the study sites listed below would improve the understanding of the processes at work.

- Sludge site B.
- IAH site B
- Fertilizer sites B and C.

As nitrate is the main contaminant of concern, it is recommended that further investigations into the impact of elevated nitrate levels on the environment and human health should be conducted. More detailed studies are required on the processes of nitrate accumulation in groundwater from anthropogenic and natural sources.

New areas of study warranting further research are:

- The impact on groundwater quality of deep 'rip' ploughing new areas for cultivation. This is thought to release nitrates derived from soil biomass decaying as a result of exposure, but the level or persistence of contamination is not well documented.

- Methods to determine the rate of bioavailable nitrogen release from organic sources such as sludge, different soil types and manures.
- The mobility and persistence of hormones and steroids used in IAH is a new area of research that has not received attention in South Africa. If effective, analytical techniques are available, this area should be investigated.

At the moment knowledge and expertise of pesticide impact on the environment and groundwater in particular is widely scattered in South Africa. Contact between the various persons is informal and occasional. It is strongly recommended that a vehicle be established that will encourage focussing of this expertise. This will enable increased public and/or government interaction with more effective results. The ideal vehicle would be electronic interlinking of the expertise and opening the group to interested parties. The first step would be to identify the role players and establish a working group.

Acknowledgements

The research in this report emanated from a project funded by the Water Research Commission and entitled:

"ASSESSMENT OF THE IMPACT OF AGRICULTURAL PRACTICES ON THE QUALITY OF GROUNDWATER RESOURCES IN SOUTH AFRICA"

The Steering Committee actively responsible for this project, consisted of the following persons :

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Mr A C Fritz	Water Research Commission (Committee Secretary)
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The financing of the project by the Water Research Commission and the contribution of the members of the Steering Committee is acknowledged gratefully.

This project was only possible with the co-operation of many individuals and institutions.

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CHAPTER 1

INTRODUCTION

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1.1 INTRODUCTION

The agricultural sector has been identified as the biggest user of groundwater in South Africa (DWA&F, 1992). It is estimated that 78% of all groundwater abstracted is used for irrigation, 7% for rural domestic purposes and 6% for stock watering. Only 4% of groundwater abstracted is used in the urban environment. Traditionally, however, the urban users have received the most groundwater research and investigation attention. As a result, little is known about the status of groundwater used by the agricultural sector in South Africa. By means of a questionnaire, Braune and Coetzer (1992) investigated some problems associated with groundwater abstraction for irrigation. They found that, on a national scale, little groundwater quality deterioration was reported. However, 30% or more of the respondents in 20% of the districts surveyed report a deterioration in quality. Braune and Coetzer (1992) recognised that the groundwater quality data collected was problematic.

It is recognised worldwide that the agricultural sector contributes to diffuse contamination (Foster and Bath, 1986; Gui, 1987; Pionke *et al.*, 1988; Sommerfeldt *et al.*, 1988). Problems include irrigation practices, animal feedlots and the use of fertilizers, pesticides and herbicides. Point sources of pollution which exacerbate the problem include septic tanks and pit latrines and the disposal of household and agricultural waste products.

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Further, it was recognised by the Groundwater Quality Task Group that much could be achieved in terms of protecting groundwater from contamination by agricultural activities through a process of education (Braune *et al.*, 1991). The transfer of information from the groundwater community to the agricultural sector is urgently required. In addition to quantifying the extent of the problem in South Africa, the research effort can be used as a means of preparing and transferring this sort of information. Approximately 90 % (by volume) of groundwater users will thus benefit directly from this work.

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A one year literature study was undertaken in order to provide some general information concerning the impact of agriculture on groundwater and present known cases of such contamination in South Africa. The study showed that little quantified information regarding known cases of groundwater contamination resulting from agriculture exists. It was, however, possible to highlight those agricultural activities which posed the most serious threat to the quality of South African groundwater resources. After completion of the literature study, a workshop was held with the Steering Committee and the research objectives were revised.

The revised research objectives were thus as follows:

a. To confirm the contamination impact on groundwater quality resulting from those agricultural activities identified as posing the most serious threat, namely

    i. Intensive animal feedlots
    ii. Use of sewage sludge for preparing land for crop production
    iii. Use of fertilizers
    iv. Irrigation
    v. Use of pesticides

b. To identify practical and easy-to-implement strategies and practices which can be used to prevent or reduce contamination resulting from agricultural activities.

c. To prepare a non-technical booklet, which transfers the information to farmers and members of the public associated with the agricultural sector.

Two research products resulted from the research study, namely:

a. A *non-technical handbook* - which sets out practices that may be used to prevent or reduce groundwater contamination by agricultural activities; and

b. A *scientific report* - which presents the findings of the literature study as well as the results of the individual investigations into the five most serious threats listed above.

CHAPTER 2

LITERATURE REVIEW

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2.1 INTRODUCTION

In order to satisfy the research objectives, the project comprised a number of distinct components. One of the initial components included a literature study which:

- provided general information concerning the impact of agriculture on groundwater,
- presented known South African cases of groundwater contamination resulting from agricultural activities, and
- identified areas requiring further research.

A summary of the literature survey has been listed in Tables 2.1, 2.2, 2.3 and 2.4. This provided a preliminary assessment of the level of information on the impact of agricultural practices on groundwater quality in South Africa. This assessment was further developed through a workshop attended by various parties active in the geohydrological and agricultural fields.

The results of the literature study and workshop (held to reassess the project objectives) are summarised in Table 2.5. This led to the decision that the main agricultural activities to be studied in the field are intensive animal husbandry, the use of inorganic fertilizers and the application of sewage sludge to land. Irrigation practices and the use of pesticides are also known to impact groundwater quality and several studies have been conducted in South Africa. This information has been collated in the form of essays and it was not thought necessary to conduct further field research.

Table 2.1: Groundwater quality problems reported in the literature which are associated with crop farming

Area	Agricultural Activity and Preliminary Finding	Impact	Reference
Vaal Harts Irrigation Scheme Jan Kempdorp	▸ Intensive irrigation (mainly flood) of 1 200 farms. ▸ Main crops are cotton, maize, ground-nuts and lucerne. ▸ Main source of water is from surface water, but groundwater is used to make up the shortfall when available surface water is limited. ▸ Pesticides and fertilizers are used intensively - it was estimated that 120 000 kg of pesticide is used per annum over an area of 32 000 ha. ▸ Area is characterised by poor drainage, resulting in a rise of the water table following the implementation of the irrigation scheme in the 1930s. ▸ Area underlain by tillite.	▸ **Increased groundwater salinity:** increased salinity in the groundwater has been ascribed to irrigation practices and an associated rise in the water table. ▸ **Increased nitrate concentrations:** increased nitrate levels in both surface and groundwater were confirmed and attributed to irrigation and fertilizer application. ▸ **Pesticide contamination:** 7 of the main pesticides used were analysed from samples collected at 11 surface monitoring stations and 6 boreholes, atrazine contamination detected regularly in surface water while other pesticides detected periodically. After initial sampling problems, no pesticides detected in groundwater after 4 runs.	Weaver (1993) Badenhorst (1994, pers.comm.)
Springbok Flats area Potgietersrus	▸ Intensive crop production area. ▸ Area had to be cleared of natural vegetation before agriculture could be developed. ▸ Area of cultivation increased from 20 % in 1920 to 64 % in 1970. ▸ Area underlain by Jurassic basalts of the Letaba Formation.	▸ **Increased nitrate concentrations:** extremely high levels of nitrate are found in the groundwater (NO_x-N > 100 mg.l^{-1}), this has been ascribed to the cultivation and ploughing of the basaltic soils which caused mobilisation and leaching of nitrate and the subsequent accumulation in the subsurface beyond the root zone, no significant increase in nitrate levels detected in last few years thereby suggesting that levels have stabilized.	Heaton (1985) Van der Merwe (1990) Orpen and Fayazi (1991) Tredoux (1993)

Area	Agricultural Activity and Preliminary Finding	Impact	Reference
Cape Flats aquifer Phillippi, Cape Town	- Intensive market gardening carried out in the area. - Sewage sludge and manure used to add nutrients to the sandy soil.	- **Increased nitrate concentrations:** relatively small increases in the level of NO_3 were found - 2.6 mg.1^{-1} (N) while ambient levels are in the 0.1 mg.1^{-1} range. The presence of S^{2-} suggests that the sewage waste causes the water to become anaerobic water leading to denitrification.	Pombo *et al.* (1978)
Elspark/Rondebult area Alberton	- Main agricultural activities are intensive market gardens (vegetables) and turf (roll-on lawn) product. - Groundwater from the dolomitic aquifers used for irrigation while effluent from a nearby sewage works is used to irrigate some of the turf fields as well as cattle fodder crops. - Fertilizers used extensively on the market gardens.	- **Increased nitrate concentrations:** the contaminated groundwater is easily identified as it differs markedly from ambient nature of the groundwater, ambient NO_3 levels are low while highest polluted level recorded to be 48.9 mg.1^{-1}, application of fertilizers found to be a more important contributor than use of sewage effluent.	Walton *et al.* (1993)
Venda Kutama District near Louis Trichart	- Found high nitrate levels associated with maize growing and allowing land to stand fallow for two thirds of the year. - Dry-land maize production most important crop. - Underlying geology mainly Baviaanskloof Gneiss of Archean age.	- **Increased nitrate concentrations:** No_x-N levels ranged between 45 mg.1^{-1} and 70 mg.1^{-1}, found levels low during winter but increased rapidly after first major summer ranges - ascribed this to nitrogen build-up during fallow periods and flushing during summer.	Connelly and Taussig (1991) Walton *et al.* (1993)
Fish River Middleton	- Irrigation scheme using water form the Orange - Fish River Scheme.	- **Increased salinity:** irrigation return flow is extremely saline, probably due to irrigation in an arid environment and owing to saline soils.	Reynders (1984)

Area	Agricultural Activity and Preliminary Finding	Impact	Reference
Fish River Golden Valley	▸ Irrigation using water from the Orange - Fish Rivers scheme.	▸ **Increased salinity:** due to poor drainage and a rise of the water table to surface, similar to that described by Reynders (1988).	Veenstra (1994, pers.comm.)
Hex River Valley De Doorns	▸ A major fruit farming area with intensive summer irrigation using mainly river water. ▸ Groundwater used to make up shortfalls during dry periods. ▸ Pesticides and fertilizers widely applied. ▸ Hydrogeology of the area well understood. ▸ Has 12 groundwater monitoring stations with 14 pesticides being tested for in each sample. ▸ 3 sampling runs were carried out in a year. ▸ No pesticides were detected (detection level of 0.005 mg.1^{-1}).	▸ No pesticide contamination detected. ▸ **Increased nitrate concentrations:** increased levels of both nitrate (up to 150 mg.1^{-1}) and potassium (up to 25 mg.1^{-1}) detected, ascribed to fertilizer application.	Rosewarne (1981) Weaver (1993)
Poesjenels River Breede River Valley	▸ Irrigation using surface water supplies to irrigate vineyards, orchards and lucerne. ▸ Have recorded increasing levels of salinity in the river.	▸ **Increased surface water salinity:** saline irrigation return flow resulting in a deterioration of water quality, is controlled by bedrock and irrigation practice. Most of the salts are released from Bokkeveld shales, also most of the salts are leached out during the early part of the irrigation season, impact on groundwater quality not known owing to ambient saline groundwater conditions.	Moolman *et al.* (1983) Greeff (1990, 1991) Kienzle (1990)
Breede River Valley Robertson	▸ Irrigation using surface water supplies. ▸ Have recorded increasing levels of salinity in the river, with EC increasing from 26 mS/m to 82 mS/m over a 40 km length of the river.	▸ **Increased surface water salinity:** irrigation water is leaching salts out of the naturally saline bedrock (Witteberg and Bokkeveld Formations).	Jolly (1990)

Area	Agricultural Activity and Preliminary Finding	Impact	Reference
National study	▸ Looked at problems in irrigation using groundwater in South Africa. ▸ Identified problems included dropping water levels, declining yields, impacting neighbouring boreholes and declining water quality.	▸ On a regional scale, deteriorating groundwater quality as a result of irrigation found to be a minor problem.	Braune and Coetzer (1992)
Kamdeboo Aquifer Unit Graaff-Reinet District	▸ Irrigation of lucerne fields using borehole and surface water.	▸ **Increased salinity:** this was attributed to the continual use of brackish water for irrigation and the build up of salt in the soil zone as a result of high evapotranspiration losses.	Parsons (1986)
Hartebeespoort Dam Pretoria	▸ Intensive tobacco producing area. ▸ Irrigation water from Hartbeespoort Dam.	▸ **Increased salinity:** Cl levels in the dam have risen which impacts on the tobacco crops which is sensitive to Cl concentration, water in the dam and irrigation return flow probably infiltrates into the underlying groundwater body resulting in Cl contamination.	??
Dendron area	▸ Irrigation using groundwater. ▸ Main crops include vegetables, citrus and tobacco. ▸ Water table has dropped as a result of abstraction by as much as 40 m in places.	▸ Impact on groundwater quality is not know (DWAF regularly sample groundwater, since 1988).	Orpen (1994, pers.comm.) Syfferdt (1993)
Witfinger	▸ Irrigation using groundwater. ▸ Water abstracted from norite aquifer.	▸ **Increased salinity:** this has been attributed to over-abstraction of groundwater for irrigation purposes.	Dziembowski (1994, pers.comm.)

Area	Agricultural Activity and Preliminary Finding	Impact	Reference
North-western Orange Free State	▸ Dry-land maize growing area. ▸ Relatively permeable sandy soils with water table 10 m below surface. ▸ NO_x-N levels found to be greater than 10 $mg.1^{-1}$.	▸ **Increased nitrate concentrations:** increased nitrate levels attributed to the use of fertilizers, but published data is lacking.	Tredoux (1993)
Crocodile River	▸ Large irrigation extending 80 km along the Crocodile River. ▸ NO_x-N levels exceeding 20 $mg.1^{-1}$ were found to be common.	▸ **Increased salinity:** EC levels ranging from 100 mS/m to greater than 250 mS/m were ascribed to irrigation. ▸ **Increased nitrate concentrations:** increased nitrate levels attributed to irrigation (based on data stored on the National Hydrochemical Database.	Tredoux (1993)
Verlorenvlei West Coast	▸ Groundwater is used extensively to grow potatoes in the sandy soils of the West Coast. ▸ Fertilization takes place to supplement nutrients in the infertile sandy soils.	▸ **Increased nitrate concentrations:** increased nitrate levels were detected in the aquifer, these aquifers play an important role in the dynamics of rivers and vleis in the area.	Harck (1994)
Selected plot studies (7) (Location not given)	▸ Investigated the impact on soil of spreading sewage sludge onto land. ▸ Compared soil in veld to that of lands spread with sewage sludge. ▸ Found some positive impacts on the soil (particularly the increased content of nutrients such as N, P, K, Ca and Mg). ▸ Found negative impacts to be increase soil salinity, increased toxicity (Zn and Cd and other trace metals) and leaching of No_x and trace elements which *could* impact on groundwater quality.	▸ No impact on groundwater quality established, but potential to pollute groundwater recognised.	Steyn and Du Toit (1994)

Area	Agricultural Activity and Preliminary Finding	Impact	Reference
Eastern Transvaal Loskop Dam catchment Selected plot studies	▸ Investigated the Zn, Cu and Mn content of 10 agricultural soils. ▸ In general, found higher concentrations in irrigated lands than in rain-fed lands. ▸ Found significant increase in Zn levels in both rain-fed and irrigated soils, but little change in Cu and Mn levels.	▸ No impact on groundwater quality established, but potential to pollute groundwater recognised.	Bienkowski and Kidson (1994)
Makatini Flats	▸ Irrigation area.	▸ Possible groundwater quality degradation, but not yet investigated.	Dziembowski (1994, pers.comm.)
Coetzersdam Vryburg	▸ Intensive production of seed-maize, water level is shallow (8 m).	▸ **Increase nitrate concentrations:** due to fertilizer application.	Dziembowski (1994, pers.comm.)
Limpopo - Shashi River	▸ Irrigation area.	▸ **Increased salinity:** irrigation return flow is extremely saline while the water table is shallow.	Dziembowski (1994, pers.comm.)
Orange River	▸ Irrigation area. ▸ Most water obtained from river and dams.	▸ **Increased salinity:** levels appear to be low because of use of good quality surface water for irrigation.	Orpen (1994, pers.comm.)
Pongola Sugar Cane Area	▸ Major sugar cane growing area.	▸ Suspected salinisation, but not yet studied.	Dziembowski (1994, pers.comm.)
Tala Valley Pietermaritzburg	▸ Irrigation area. ▸ Main crops are cash crops such as vegetables and potatoes.	▸ Suspected pesticide contamination but not yet studied.	Dziembowski (1994, pers.comm.)
Lamberts Bay area	▸ Irrigation of crops along the West Coast on sandy soils.	▸ Suspected salinisation problems, but not yet studied.	Taylor (1994, pers.comm.)

Table 2.2: Groundwater quality problems reported in the literature which are associated with stock farming

Area	Agricultural Activity	Impact	Reference
Springbok Flats area Pietersburg	▸ The release of nitrates from the soil into groundwater by ploughing is well recognised. ▸ Using statistical methods, 4 cases of groundwater pollution caused by animal wastes or fertilization could be identified.	▸ **Increased nitrate concentrations:** 4 cases of extremely high nitrate levels in the groundwater were found (NO_x-N between 300 and 700 $mg.l^{-1}$), suspected to result from fertilizer application and animal waste.	Van der Merwe (1990)
Venda Kutama District near Louis Trichart	▸ Noted that over-grazing reduce plant growth, thereby interrupting the natural nitrogen cycle and resulting in less nitrogen being taken up by the plants.	▸ **Increase nitrate concentration:** noted that overgrazing may contribute to the build-up of nitrogen, but impact not quantified.	Connelly and Taussig (1991)
Carmine Estates Middelburg (Tvl)	▸ Intensive feedlot with about 30 000 head of cattle on 10 ha of ground. ▸ Had a major problem with contaminated surface water from the feedlots entering the river and ultimately the Middelburg town water supply dam. ▸ Contaminants which need to be look at include Cu, Zn and Mo levels in food.	▸ Impact on groundwater not known.	Veenstra (1994, pers.comm.) Vosloo (1994, pers.comm)
Swartwater - Mallabas area	▸ Rural area with extensive stock farming.	▸ **Increased nitrate concentrations:** localised elevation of nitrate levels in boreholes near to kraals and watering points.	Bush (1989)
North West Region	▸ Identified stock-farming practices as a source of high nitrate concentrations in groundwater.	▸ **Increased nitrate concentrations.**	Xu *et al.* (1991)

Area	Agricultural Activity	Impact	Reference
Durbanville - Milnerton area	▸ Intensive stock farming area with both extensive farming and feedlots. ▸ Run-off from pens into river has resulted a major river pollution problem.	▸ Suspected increases in nitrate concentration, but groundwater impact not established.	Taylor (1994, pers.comm.)
Western Kalahari	▸ Area has a high ambient background nitrate concentration. ▸ Borehole located near to livestock enclosures.	▸ **Increased nitrate concentrations:** localised contamination due to livestock with NO_x-N exceeding 56 mg.1^{-1}.	Adam (1982), as reported by Tredoux (1993) Tredoux and Du Plessis (1993)
Selected case studies (Location not given)	▸ Investigated the impact of two piggeries and one feedlot on the soil and water sources. ▸ Found that the spreading of animal waste could improve some soil physical characteristics, but higher levels of toxic elements such as Cu, Zn and Hg were recorded. ▸ The sludge dams posed the biggest pollution threat with raised salinity levels and higher concentration of Al, Co, Cr, Ni, Se and Mo - recommended limited use of water for irrigation and strict control on sludge dam construction. ▸ From single samples, no chemical contamination of surface or groundwater detected.	▸ Based on very limited study, no impact on groundwater quality established.	Steyn (1994)

Area	Agricultural Activity	Impact	Reference
Kwa-Zulu Natal Nagal Dam	▸ Feedlots suspected of leading to water quality problem. ▸ Samples collected from boreholes in the area, but results not published.	▸ **Increased nitrate concentrations:** Umgeni Water sampled boreholes around feedlots, recorded increased nitrate concentrations.	Howard (1995, pers.comm.)
Hartswater	▸ Feedlots suspected of causing nitrate contamination.	▸ Suspected increases in nitrate concentration, but groundwater impact not established.	Taylor (1994, pers.comm.)
Carolina	▸ Feedlots suspected of causing nitrate contamination.	▸ Suspected increases in nitrate concentration, but groundwater impact not established.	Dziembowski (1994, pers.comm.)

Table 2.3: Groundwater quality problems reported in the literature which are associated with farming infrastructure

Area	Agricultural Activity	Impact	Reference
Kuruman	▸ The placement of houses and boreholes is strongly controlled by geology. ▸ House are built on calcrete mounds associated with dolerite dyke structures as opposed to building on the lower lying dolomite which is regularly covered by shifting sand. ▸ This has lead to sewage disposal (using pit latrines) taking place on the dyke contact zone, which is also the best drilling target zone.	▸ **Increase nitrate concentrations:** elevated nitrate levels were detected in groundwater.	Weaver (1995, pers.comm.)

Table 2.4: Groundwater quality problems reported in the literature which are associated with other farming activities

Area	Agricultural Activity	Impact	Reference
Richards Bay	▸ Land-farming of oily waste after the Petingo oil spill. ▸ Spread waste to promote weathering and biodegradation.	▸ No impact on groundwater quality recorded.	Dehrmann (1991)

Table 2.5 : Summary of literature study and workshop findings regarding the impact of agricultural activity on groundwater quality in South Africa

AGRICULTURAL ACTIVITY	IMPACT ON GROUNDWATER QUALITY				NATURE OF FURTHER INVESTIGATION REQUIRED			
	High	Medium	Low	Unknown	National	Site Specific	BMP*	Priority**
Crop Farming								
land clearing			●					4
ploughing			(●)					4
fertilizer application	●				●		●	1
sludge application to land	●					●	●	2
sowing			●					4
dry-land crop farming			●				●	3
irrigation-induced salinity	●				●			1
land-treatment systems				●		●	●	2
pesticide application	●					●	●	2
harvesting			●					4
fallow			(●)				●	3
Stock Farming								
feedlots	●					●	●	1
ranching or extensive farming			(●)				●	2
dips			(*)				●	3
Infrastructure								
damming good quality water								4
damming poor quality water				●		●		1
waste disposal			●				●	3
sewage disposal			(*)				●	3
cemeteries			(*)				●	3
storage			(*)				●	3

NOTES: (●) significant local impact recorded
* Best Management Practice
(*) significant local impact possible / probable
** priority ratings: 1 = top, 4 = low.

CHAPTER 3

METHODOLOGY

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3. METHODOLOGY

A phased approach was used to meet the aims of the project. This was as follows:

- Review the literature to identify agricultural activities with the greatest potential to contaminate South African groundwater resources.
- Hold a workshop in order to prioritise the main polluting activities.
- Collate information from previous studies on the impact of these activities.
- Conduct field investigations of those activities where there is little or no field data for South Africa.
- Assess the impact on groundwater quality of these activities and those where previous investigations were carried out.
- Review good farming practices that help to minimise the impact of these activities.
- Present these Good Farming Practices in an information booklet for farmers.

3.1 FIELD INVESTIGATIONS

Potentially contaminating activities which had not been studied sufficiently in the field in South Africa were identified as:

- Intensive animal husbandry
- Use of inorganic fertilizers.
- Sludge application to agricultural land

Three field study sites were chosen for each activity. These were located around the country and chosen based on the following criteria:

- The land owners and/or operators fully supported the project and were willing to supply supporting information.

- The sites overlaid potentially vulnerable aquifers.
- Groundwater sampling could be carried out at a minimal cost (ie. a sufficient density of boreholes existed or could be installed relatively cheaply).

Groundwater sampling

Groundwater monitoring points were established, where necessary, to ensure that representative samples of potentially contaminated groundwater could be collected and that one or more control sampling points were available. New boreholes were constructed using techniques to ensure that no potentially contaminating substances were introduced into the aquifer with experienced technicians carrying out the installation under the supervision of a hydrogeologist.

Where existing boreholes were used as monitoring points, information on their construction, history and performance was gathered from the owners to enable as complete an understanding as possible of the sampling conditions.

Groundwater sampling protocol was followed (as outlined in Weaver, 1992) to ensure that samples representative of water in the aquifer were analysed. This included careful use of sterile sample containers, purging at least 3 well volumes prior to sampling and storage for minimal periods of time (less than 48 hours) in the dark at temperatures of less than 4°C.

At most field sites, more than one phase of sampling was carried out to give an indication of variation of groundwater with time.

Determinands

Field measurements of groundwater pH, EC and temperature were carried out at each sampling point. The colour and smell of the groundwater was also noted.

A suite of determinands for laboratory analysis was selected to include major ions usually found in groundwater and substances known to contaminate groundwater as a result of the agricultural activity being studies. These included:

Hydrochemical parameters:	pH, EC, alkalinity, temperature, COD.
Major ions:	Ca, Mg, K, Na, HCO_3, SO_4, Cl.
Common potential contaminants:	NO_3, NH_4, TKN, DOC.
Heavy metals:	Fe, Mn, Al, Zn, Cu, Cd, Ni, Pb.
Pathogens:	Faecal coliforms, faecal streptococci, heterotrophic plate count, *Clostridia*.

The hydrochemical analyses were carried out at accredited CSIR laboratories using standard methods.

Samples were collected for nitrogen isotope analysis using recommended procedures including mercuric chloride preservation (Weaver, 1992; Talma, pers. comm.). The analyses were conducted at the CSIR isotope laboratory using standard methods as described in Heaton and Collett (1985).

Data quality

The quality of hydrochemical data was analysed to ensure that the results were meaningful and representative of water in the aquifer. Quality control samples were routinely run during analysis. The ionic balance of the hydrochemical results was determined to confirm that the results were within an acceptable margin of error (<5%) and a comparison of pH and EC measured in the field and in the laboratory was made.

Hydrochemical data was analysed in the context of other data on the field sites (soil analyses, nitrogen isotope analyses, application rates, crop types, etc) to assess the impact of the activity on groundwater quality.

3.2 ESSAYS

The use of pesticides and irrigation are known to impact groundwater quality worldwide and several studies have been conducted in South Africa. This information was collated in the form of essays and is presented in this report. It was not thought necessary to conduct further field research.

3.3 TECHNICAL REPORT

Details of the methods, results and conclusions of the field investigations and summaries of the information collated in essays are presented in Volume 1 of this report. The supporting data from the field studies are contained in Volume 2.

3.4 GOOD FARMING PRACTICES

One of the main aims of this study was to inform farmers about contaminating activities and provide guidance on Good Farming Practices (GFPs). Information on GFPs was collated during the course of the specialist studies and presented in an accessible format in a handbook. In addition to GFPs relating to the five activities researched in detail, a chapter on pollution at the farmstead (septic tanks, underground fuel tanks, etc) has been included. Background information on the occurrence and vulnerability of groundwater and that of nitrate, the most common contaminant, has also been included in the handbook.

3.5 INFORMATION TRANSFER

The handbook for farmers aims to make the scientific background to agricultural contamination of groundwater accessible to commercial farmers. It is hoped that if there is a greater understanding of the processes resulting in a deterioration in groundwater quality, the implementation of remedial measures will be more effective. The handbook also attempts to point out the cost effectiveness of many of the GFPs, to encourage farmers from an economic as well as environmental perspective.

The initial model for the handbook was the Code of Good Agricultural Practice for the Protection of Water produced by the Ministry of Agriculture, Fisheries and Food (MAFF) in the UK (1991). The approach for the farmers handbook has been to offer the same balance of practical advice and technical information. The main differences between the MAFF Code and our handbook are as follows:

- As a Code, the MAFF publication is legally defensible. This means that although not the law, it may be considered in the legal defence of a farmer prosecuted for water pollution.
- As a Code, the MAFF publication provides a greater level of detail in its recommendations,

particularly for engineering specifications (eg - walls of slurry storage facilities should be constructed to withstand loads as specified in British Standards 5502).

- The MAFF publication covers all forms of water contamination and is focussed primarily towards preventing increased levels of BOD in surface water.

CHAPTER 4

INTENSIVE ANIMAL HUSBANDRY

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EXECUTIVE SUMMARY

The objectives of this specialist study are

- to collect field data from 3 study sites to determine whether groundwater contamination has occurred as a result of intensive animal husbandry operations in South Africa
- to collect data and information which can be used to support the non-technical booklet.

Intensive animal husbandry includes industries which entail farming with substantial capital and labour inputs per unit area, namely aquaculture, dairy farming, poultry, feedlots, pig farming and ostriches in breeding and feeding camps. This specialist study has focused on the impact on groundwater quality of a feedlot, a piggery and two dairies.

The findings of the field investigations are summarised in the table below.

Summary of findings of the specialist field study on the impact of sludge application to agricultural land on groundwater quality.

	Site A	Site B	Site C
IAH activities	Dairy (1200 cows) Covered stalls and pasture.	Dairy (800 cows) Piggery (6000 pigs) Covered stalls and pasture.	Feedlot (4000 cattle) Open pens
Waste management practices	Irregular flushing of stalls & manual solids clearance. Solid separation (dysfunctional), manure stockpiling, effluent lagoons.	Regular flushing & manual solids clearance. Solid separation, manure stockpiling, effluent lagoon, effluent irrigation.	Regular scraping of pens, run-off capture, effluent lagoons, effluent irrigation.
Field data collected as part of this project	Groundwater hydrochem. from est. boreholes & spring. Soil nitrogen profiles	Shallow groundwater hydrochem from temp well points. Soil nitrogen profiles	Groundwater hydrochem from est. & new boreholes. Drilling data Soil nitrogen profiles
Climate	500 mm p.a winter rain	900 mm p.a winter rain	700 mm p.a summer rain
Unsaturated zone	> 2 m thick, sandy soil with clay layers	<1.5 m thick, sandy alluvium.	>30 m thick, clayey.
Aquifer type	Surficial weathered zone of granite, possibly fractures.	Unconsol. alluvium.	Fractured shales, possibly some quartzites and dolomite.
Hydrochemistry	NaCl type TDS 600 - 1500 $mg.l^{-1}$	Variable, NaCl type TDS 100 - 2000 $mg.l^{-1}$.	CaCO3 type TDS < 40 $mg.l^{-1}$
Potential contamination from non-IAH sources.	Fertilizers and pesticides applied to wheat and vines. Pit latrines.	Effluent from food processing. Fertilizers and pesticides applied to orchards and vines. Pit latrines.	Fertilizers and pesticides applied to maize. Pit latrines.

continued....

	Site A	Site B	Site C
Probable IAH groundwater contamination:			
Nitrate	Yes - to a max of 27 mg.l^{-1} assoc. with the dairy operations (?). Also assoc. with waste slurry flooding.	Yes - to max of 156 mg.l^{-1} assoc. with cow conc. at feed trough in pasture. Also in effluent irrigated pasture.	No
DOC	No	Yes - to max of 55 mg.l^{-1} next to waste channel. Also in effluent irrigated pasture and at feed trough in pasture.	No
Potassium	No	Yes - to max of 259 mg.l^{-1} at feed trough in pasture. Also in effluent irrigated pasture and next to waste channel	No
Ortho-phosphate	No	Yes - to max of 6 mg.l^{-1} in effluent irrigated pasture. Also next to waste channel.	No
Microbiological indicators	Yes - to a max of 2 faecal Strep. per 100 ml assoc. with the dairy operations (?)	Yes - to max of 37500 faecal coliform per 100ml in effluent irrigated pasture. Also close to waste channel.	ND
Impact of IAH activities on groundwater quality.	Significant w.r.t nitrate and microbiological indicators, localised.	Significant w.r.t a wide range of determinands, variable, apparently localised.	None

(?) Further research required to confirm.
ND Not determined

The case studies therefore show an impact on the following aspects of groundwater quality as a result of IAH operations: nitrate, DOC, ortho-phosphate, potassium, microbial indicators of faecal pollution.

The most important factor influencing the degree of contamination at the sites studied appears to be the natural hydrogeological controls on groundwater vulnerability.

The results of these case studies will be incorporated into the non-technical booklet to show examples of groundwater contamination in South Africa. In addition, information gathered as part of this specialist study will be used to discuss pollution prevention and best agricultural management practice. Good management practices illustrated by the study include:

- run-off control
- maintenance of a feedpen interface layer
- storage of solid waste on constructed areas
- siting of feedpens on a slope with areas of cattle congregation around the feed troughs up-gradient.

Poor practice seen as part of this study mainly concerns management of animal waste. The following activities have been shown to be associated with groundwater contamination:

- irrigation of effluent onto pasture
- concentration of livestock into limited areas of pasture intermittently (as opposed to permanent high densities in feedpens which results in the formation of an interface layer)
- leakage from constructed effluent channels
- leakage (to the unsaturated zone) from effluent lagoons where the self-sealing layer is removed.

4.1 INTRODUCTION

4.1.1 Motivation

The literature study carried out as part of this project identified intensive animal husbandry (IAH), and particularly feedlots, as having significant potential to contaminate groundwater (CSIR, 1995). Little information regarding the impact in South Africa was found therefore it was decided to conduct a specialist study investigating the impact of IAH on groundwater quality at 3 study sites.

4.1.2 Intensive animal husbandry

Farmers have become increasingly reliant on intensive agricultural practices in order to reduce costs and increase yields. This is the case in livestock farming where the practice of intensive animal husbandry, using feeding stalls and battery farming, is becoming more widespread worldwide.

The draft of new regulations for Pollution of Natural Agricultural Resources Caused by Farming Practices defines intensive farming enterprises as: '*An industry which entails farming with the following, namely*

- *aquaculture*
- *dairy farming*
- *poultry*
- *feedlots*
- *pig farming*
- *ostriches in breeding and feeding camps*

which entails substantial capital and labour inputs per unit area.'

The case studies in this investigation focus on a piggery, dairies and a feedlot therefore these areas of IAH receive the main focus in this introduction.

The main environmental impact of IAH is the large volume of animal waste that is produced concentrated in a limited area. IAH waste management is therefore an area of concern.

Commonly livestock waste is disposed of by selling manure as fertilizer or on-site beneficial application to pasture or other cropped land.

Environmental concerns associated with IAH include contamination of surface water by effluent run-off, contamination of groundwater by infiltration from animal waste and problems with fly and odour nuisances.

4.1.2.1 IAH in South Africa

IAH is practised in many areas of South Africa. The Directorate of Resource Conservation (Department of Agriculture) is currently surveying IAH farms around the country and this is approximately 40% complete.

Feedlots vary in size from 300 to 40 000 head of cattle, although there are thought to be only 5 feedlots operating at the top end of the range (SA Feedlot Association pers. comm.). Total feedlot capacity in South Africa is estimated to be around 360 000 cattle, but it is thought that around 280 000 cattle are currently being reared. The feedlot area usually covers one to a few hectares, depending on the number of cattle. Fodder crops are often cultivated on peripheral land. Feedlots tend to be located close to the major markets and sources of feed. Waste management presents the most significant environmental problem. Solid waste is often used by the feedlot farmer to fertilize fodder crops or sold to other farmers. Effluent is usually stored temporarily in dams and then used by the feedlot farmer to irrigate crops or applied to pasture.

Dairies have a wide distribution around the country and contain between 5 and 1 800 cows (Directorate of Resource Conservation). In 1996 there were 1.25 million dairy cows recorded in South Africa over two years of age, 81 617 excluding the former homeland areas (Directorate of Agricultural Statistics and Management Information pers. comm.).

Piggeries tend to be located close to main markets. There were 1.6 million pigs recorded in South Africa in 1996 (including former homeland areas) but it is not known how many of these were farmed intensively (Directorate of Agricultural Statistics and Management Information pers. comm.). Effluent from piggeries is thought to have greater potential to contaminate soil and water resources as pig feed, and therefore pig effluent, contains higher levels of metals such as copper (Directorate of Resource Conservation).

4.1.2.2 South African legislation aimed at preventing groundwater contamination from IAH practices

Prevention of groundwater contamination from activities carried out at IAH farms (also known as intensive farming enterprises) is covered in the Conservation of Agricultural Resources Act and the Water Act.

The Conservation of Agricultural Resources Act No. 43 of 1983 simply states that intensive farmers should not pollute the environment (Article 6(2)N). This Act has recently been updated and a draft of the new control measures is presented in Appendix 1.A. New recommendations include the use of seepage-free storage dams for effluent, the establishment of wetland areas for disposal of effluent and beneficial use of separated solids in such a way that will not contaminate the soil, groundwater or surface water (Directorate of Resource Conservation pers. comm.). Effluent should comply with maximum recommended limits for certain water quality parameters before disposal to supplement surface or subterranean water sources. These include the following maxima:

- EC of 250 $mS.m^{-1}$
- free and bound ammonium-nitrogen concentration of 10 $mg.l^{-1}$
- soluble orthophosphate concentration of 1 $mg.l^{-1}$
- sodium concentration of 200 $mg.l^{-1}$
- residual chlorine concentration of 0.1 $mg.l^{-1}$
- suspended solids concentration of 25 $mg.l^{-1}$

In terms of the Water Act, potential contamination of the water environment, including groundwater, is covered in Sections 21 to 26. The current approach to water pollution prevention by non-hazardous substances is outlined in the DWA&F's Receiving Water Quality Objectives. This approach "*involves specification of the desired quality of the receiving water environment and the control of sources. The approach takes into account non-point and point sources of pollution to the degree necessary to maintain the desired quality. This approach recognises that the receiving water has a certain capacity to assimilate pollution without serious detriment to quality requirements of recognized users.*" (DWA&F, 1991).

4.1.3 Impact of IAH operations on ground water quality

Research has been carried out, primarily in the US, Europe and Australia on the impact on groundwater quality of IAH and a summary of important findings is outlined below. The main risks of groundwater contamination from IAH are usually as a result of inadequate management of the following practices:

- *Storage of liquid and solid waste* - unsealed effluent dams and manure heaps with uncontrolled run-off (see below).
- *Distribution and disposal of liquid and solid waste* - particularly where this is carried out without considering crop/pasture nutrient requirements (Pavelic *et al.*, 1992).
- *Cleaning stalls and pens and disposal of waste water* - particular care must be taken to leave the interface layer intact (see below).
- *Abandonment of stalls and pens* - shrinkage cracks in drying manure provide pathways for vertical migration (see below).
- *Periodic concentration of livestock in pasture* - high levels of urine and manure accumulation in areas with limited compaction such as around feeding troughs and gates.
- *Burial of livestock carcasses* - can lead to contamination in areas with vulnerable shallow groundwater.
- *Storage of feed* - contamination of groundwater may be a risk if feed is not stored under cover and if drainage from the feed area is not managed.

Effluent lagoons play an important role in waste management at many IAH farms. They act as waste storage facilities and allow the anaerobic degradation of organic matter. This usually proceeds as a three phase process (Funke *et al.*, 1984):

(i) Hydrolysis of high molecular compounds like carbohydrates, fats and proteins to low molecular components.

(ii) Break down of these components to volatile organic acids, alcohols, ketones, carbon dioxide, water, ammonia and hydrogen sulphide by acid-forming bacteria.
(iii) Methane-forming bacteria further degrade these components (except hydrogen sulphide) into methane and carbon dioxide.

The potential for seepage of nutrients to the groundwater below effluent lagoons and retention basins is reduced due to the accumulation of solids and clogging by bacterial cells and fine organic matter (Mazzone *et al.*, 1992). Infiltration may occur from new unlined ponds but with several months accumulation many become self-sealing. Davis *et al.* (1973) reported that the permeability beneath a dairy retention pond decreased from 1220 $mm.d^{-1}$ to 5 $mm.d^{-1}$ after four months operation.

Self-sealing may not be established in areas with coarse sands or well developed secondary permeability. In these areas artificial pond lining is required. Cases of groundwater contamination from effluent lagoons are also associated with the rupture of the lagoon seal by seasonal drying of the base and by gas release from microbial activity in the soil beneath the seal (Tellez *et al.*, 1993).

In anaerobic effluent lagoons nitrogen, potassium and sodium tend to partition into the liquid phase and heavy metals, magnesium and phosphorus tend to partition into the sludge at the base of the lagoon (Funke *et al.*, 1984). The anaerobic conditions in the ponds lead to the reduction and denitrification of nitrate. This chemical reduction combined with low permeability at the base of the lagoon tend to mitigate nitrate leaching.

The literature shows that nitrate leaching to groundwater from areas of **land disposal** of waste and effluent is significant and widespread. Application to land is widely practised as it provides an economic means of disposing of effluent and manure. Other forms of disposal, such as discharge to a surface water body, would require expensive treatment. Disposal to land is a beneficial use option where fodder crops are grown.

Problems of contamination occur due to the high levels of contaminants in the waste, high application rates and aerobic conditions in the disposal area which enhance nitrification and subsequent leaching. The risk of leaching is greatest where high volumes of effluent are applied or irrigation is carried out in addition to waste application.

In addition to problems associated with groundwater quality, soil salinisation and polluted run-off to streams during concurrent effluent application and precipitation are reported.

Feedlot pads receive a high loading of bovine waste but generally low levels of infiltration and contaminant transport are recorded in the literature for well managed pads. The concentration of cattle in the pad area, whilst resulting in high levels of manure deposition, also acts to compact manure forming a low permeability, anaerobic layer at the manure - soil interface. The dense, poorly aerated layer which forms the interface layer is between 2 and 10 cm thick, depending on the stock holding period (Mazzone *et al.*, 1992). Microbial decomposition of manure in this layer produces organic gels and polysaccharides that further reduce infiltration by plugging the soil pores (Mazzone *et al.*, 1992). Care should be taken when removing surface manure from pads not to remove the interface layer.

The risk of ground and surface water contamination from a feedlot pad is greatest during the first precipitation event of a wet season. The highest concentrations of potential contaminants are usually found in the initial run-off from uncovered stalls and pads (Mazzone *et al.*, 1992). This is known as the first flush effect. Careful management of this run-off is required to prevent run-off reaching surface water or areas where it may infiltrate to groundwater. High contaminant concentrations occur during warm weather and when the components of manure dissolve slowly under saturated conditions. Continual wetting with fresh manure and urine and compaction in an active pad act to reduce permeability in the interface layer. However, when a pad is abandoned the manure dries and shrinkage cracks form (Mazzone *et al.*, 1992). The permeability of the pad base is then increased and the risk of infiltration increases.

Significant leaching of nitrate to groundwater has been observed beneath dry leguminous **pastures** (Dillon, 1993). The leaching is mainly associated with urine patches. Nitrogen in dung is predominantly in the organic form and contributes gradually to nitrate leaching whereas urine nitrogen is predominantly urea and this converts rapidly to ammonium, contributing more significantly to nitrate leaching in the short term (Sherwood and Ryan, 1990). The very irregular distribution of livestock wastes across a paddock results in many small patches where extremely high nitrogen concentrations exceed the uptake and attenuation capacity of the plant-soil system (Dillon, 1993). Irregular patches occur due to excretion during grazing in pasture and heavier manure loads occur due to periodic livestock concentration such as at feeding or water troughs and gates.

4.1.3.1 Contaminants

The contaminants associated with the potential pollution sources discussed above are:

- Nitrate - found in a range of hydrogeological conditions associated with leaking retention ponds, abandoned feedpens and over-application of animal wastes to crop land (US EPA, 1992).
- Total dissolved solids (TDS)
- Dissolved organic carbon (DOC)
- Chemical oxygen demand (COD)
- Potassium
- Microbes - primarily where microorganisms can migrate to groundwater through fractures and macropores (US EPA, 1992)
- Heavy metals

4.1.3.1.1 Nitrogen

Of all the potentially contaminating substances associated with IAH, **nitrate** has been shown to have the most significant impact on groundwater quality. It is the contaminant with the greatest risk of reaching groundwater at potentially harmful concentrations, particularly where effluent is applied to land .

The US EPA Feedlot Workgroup (1992) found that nitrate, bacteria and salts from feedlots had contaminated groundwater in 17 states. In Australia the single largest source of nitrate in groundwater is that leached from livestock urine on grazed leguminous pastures (Dillon, 1988). The Darling Downs, where many feedlots are situated, overlie a fractured basalt aquifer with nitrate nitrogen levels of 60 $mg.l^{-1}$ (Lawrence, 1983). Nitrate pollution of groundwater has also been associated with dairies in south Australia.

Piggery effluent in an anaerobic lagoon typically contains 440 to 840 $mg.l^{-1}$ total Kjeldahl nitrogen (TKN) and 360 to 550 $mg.l^{-1}$ ammonia nitrogen whereas effluent lagoons at dairies contain 220 to 430 $mg.l^{-1}$ TKN and 175 to 330 $mg.l^{-1}$ ammonia nitrogen (Funk *et al.*, 1984). Feedlot wastes have been reported to contain between 10 and 500 $mg.l^{-1}$ TKN, up to 150 $mg.l^{-1}$ ammonium and up to 200 $mg.l^{-1}$ nitrate (Mazzone *et al.*, 1992).

Nitrate is a cause for concern in drinking water at levels greater than 10 $mg.l^{-1}$ (NO_3- N). At concentrations greater than this there is an increased risk to humans, particularly infants, of

methaemoglobinaemia, commonly known as Blue Baby Syndrome (Freshwater Foundation, 1988). The mortality rate among affected infants is thought to be around 10% (Freshwater Foundation, 1988). Nitrate is converted by microbial activity in the low pH conditions of the stomach to nitrite. Once nitrite enters the blood stream it changes haemoglobin to methaemoglobin which reduces the capacity of the blood to carry oxygen.

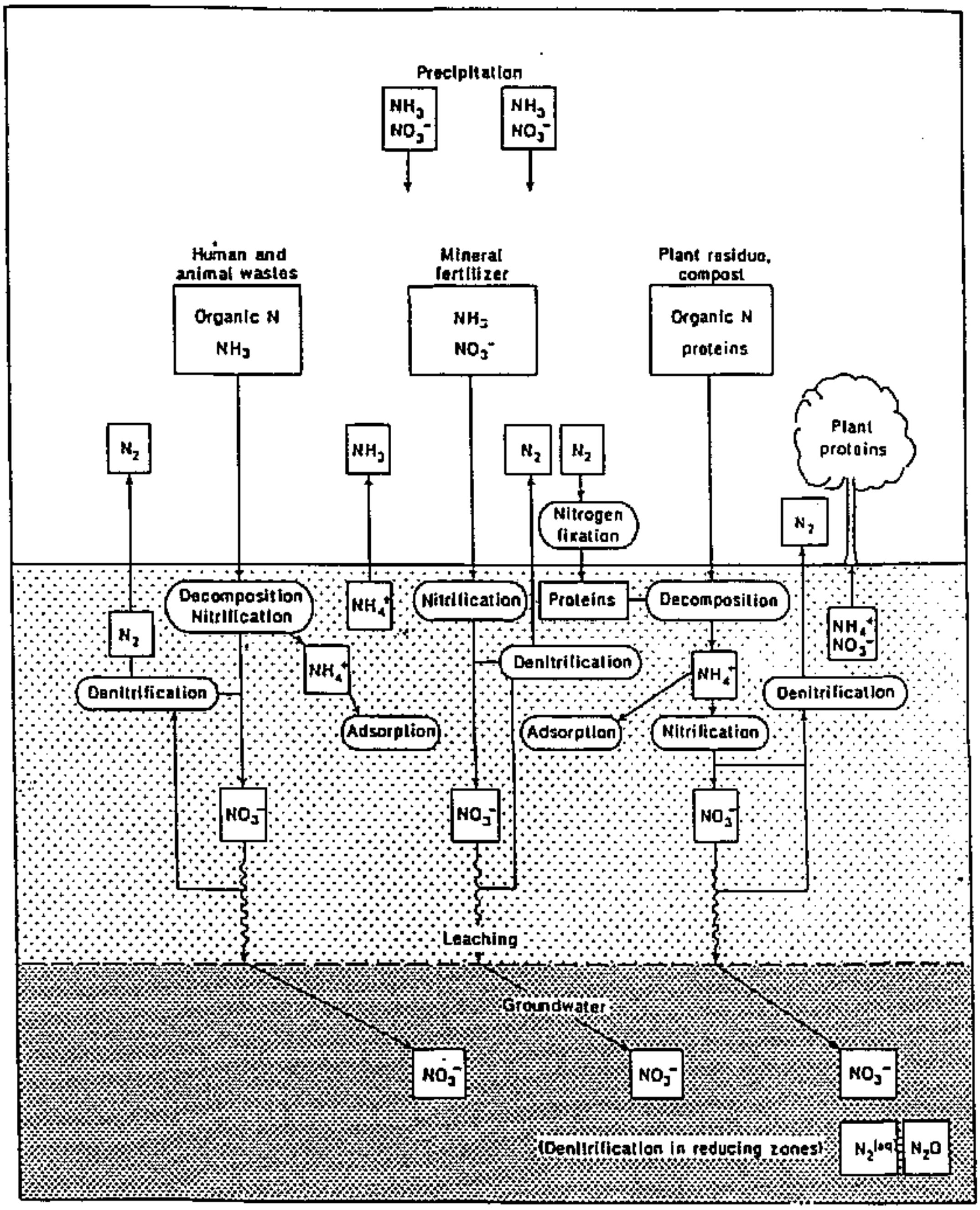
Livestock has been found to be similarly effected, but at higher nitrate concentrations. Nitrate-nitrogen concentrations in livestock water of 100 to 300 mg.l^{-1} may cause problems if the feed is high in nitrate. Water with concentrations above 300 mg.l^{-1} should not be consumed by livestock.

Nitrogen is found in all soils. Usually in excess of 90% of soil nitrogen occurs in organic forms (Vinten and Smith, 1993). Nitrogen from the atmosphere is naturally fixed (to organic nitrogen) by soil microfauna and leguminous plants at a rate which may be as high as 40 kg N.ha^{-1}.yr^{-1} (Goulding, 1990).

Organic nitrogen is not available for plant uptake and is essentially immobile. However, once applied to the soil, microbial action mineralizes organic nitrogen to ammonia, which is fixed by microbial action to ammonium and then nitrified to nitrite and nitrate (see Figure 4.1). This is the process of slow release of immobile organic nitrogen to that which is available for plant uptake, leaching to groundwater and run-off or release to the atmosphere.

Mineralization of organic nitrogen to inorganic ammonium form is carried out under a wide variety of environmental and edaphic conditions by a large diversity of soil micro-organisms (Wild, 1988). An important control on the rate of ammonium mineralization is the ratio of C:N in the organic matter (Smith, 1996). Nitrogen is more mineralizable with lower ratios of C:N. Animal waste with a ratio of 5:1 is highly mineralizable. Organic matter such as straw with a ratio of >15 is very slow to mineralize and immobilizes residual soil N until some C has been released from the soil environment, usually respired as CO_2 (Smith, 1996). Urea nitrogen is more rapidly mineralized to ammonium than organic nitrogen found in dung (Hutchings and Kristensen, 1995).

The rate of mineralization of organic nitrogen to ammonium increases with soil moisture content between wilting point and field capacity (Vinten and Smith, 1993). However, once field capacity is exceeded the rate falls due to restricted aeration of the soil. Mineralization increases in soils with high pH values (Vinten and Smith, 1993).



Chemical symbols: N, elemental nitrogen;
N_2, nitrogen gas;
N_2O, nitrous oxide;
NO_2^-, nitrite;
NO_3^-, nitrate;
NH_3, ammonia;
NH_4^+, ammonium;
N_2(aq), nitrogen gas; dissolved in water.

Figure 4.1 : The nitrogen cycle *(adapted from Freeze and Cherry, 1979)*

Once mineralized, ammonium may volatilise or remain in the soil, possibly to be nitrified. Nitrogen losses by **volatilization** will proceed more rapidly at higher temperatures, lower soil moisture and higher air speeds (Kreitler, 1975).

Nitrification is carried out by nitrifying bacteria, commonly *Nitrosomonas* and *Nitrobacter*. The rate of nitrification is more sensitive to environmental factors than that of ammonium mineralization and is influenced by the following (from Smith, 1996; Krumbein, 1983):

- Soil temperature. Generally the rate of nitrification increases with increasing temperature to an optimal rate at 30 - 35° C. Very little nitrification occurs in soils below 5°C.
- Aeration. Nitrification is an oxidising reaction and requires aerobic conditions.
- Soil moisture content
- pH. Nitrifying bacteria are inhibited by the acid they produce and therefore the rate is optimised in neutral to alkaline soils.

Subsequent **leaching** is affected by the relative rates of plant uptake, denitrification, losses of N_2 to the atmosphere and percolation rates (dependent on rainfall or irrigation). Nitrogen is more likely to leach to groundwater under the following conditions:

- High rates of nitrogen loading.
- Low ratios of C:N increasing the availability of nitrogen for mineralization.
- High soil temperatures increasing the rate of nitrification (and inhibitor breakdown if these are applied).
- Well aerated soil encouraging nitrification. (Denitrification is likely to occur in anoxic conditions.)
- Low levels of plant uptake due to bare ground, low crop requirements or seasonally variable requirements.
- High levels of precipitation or irrigation
- High vertical permeability (including macro-pores in clayey soils allowing rapid downward migration).
- Shallow, unconfined water table.

Nitrate in groundwater and the unsaturated zone is subject to **reduction** (primarily by heterotrophic bacteria) under anoxic or very low oxygen conditions in the presence of an electron donor such as organic carbon (Spalding *et al.*, 1993). Maximum levels of dissolved oxygen of between 2 $mg.l^{-1}$ and 0.3 $mg.l^{-1}$ have been quoted in the literature (Edmunds, 1972; Postma, 1991). Chemodenitrification may occur if the redox potential of the groundwater is low enough (Freeze and Cherry, 1979). The reduction process is defined as **denitrification** when nitrogenous gas is produced (Krumbein, 1983).

In addition to redox potential and the availability of carbon, the rate of denitrification is also dependent on the concentration of nitrate. High rates of unsaturated zone denitrification, >0.2 $kg.ha^{-1}.d^{-1}$ are recorded where soil nitrate concentrations exceed 5 $mg.kg^{-1}$. (Vinten and Smith, 1993).

The concentration of nitrate in groundwater is primarily dependent on the following factors (Burt *et al*, 1993; Krumbein, 1983; Freeze and Cherry, 1979):

- The concentration of nitrate leached to the water table
- The relative proportions of contaminated leachate and uncontaminated lateral inflow
- The rate of convective groundwater flow
- The rate of hydrodynamic dispersion
- The rates of reduction and denitrification (also dependent on redox conditions in the saturated zone)

Nitrate is rarely reported in fully confined aquifers due to the reducing conditions which tend to exist.

Although nitrate is the more soluble form of nitrogen, plumes of organic nitrogen and ammonium contamination due to effluent application have been reported (Dillon, 1993). Plumes may be predominantly reducing due to the high organic concentrations with nitrification occurring only on the edges where labile organic carbon is depleted.

4.1.3.1.2 Other major ions and hydrochemical parameters

Pig, poultry and dairy manures contain particularly high levels of **total dissolved solids** (TDS) due to the salts added to the feed. Of potential concern to groundwater are high levels of sodium, chloride and bicarbonate. Typical anaerobic effluent from a piggery has an EC of around 550 $mS.m^{-1}$ (approximately 4 000 $mg.l^{-1}$) (Funke *et al.*, 1984). Sweeton (1989) reports TDS levels in feedlot waste to vary from 2 700 to 15 000 $mg.l^{-1}$. Higher levels of most major ions occur in effluent applied to land compared to recharge percolating from precipitation or 'clean' irrigation recharge.

The degree to which **pH** is affected depends on the buffering capacity of groundwater and soil water at the site.

Phosphorus is found in feedlot waste at between 1 and 400 mg.l^{-1} (Mazzone *et al.*, 1992). It occurs mainly in the faeces of livestock rather than urine (Funke *et al.*, 1984). Excess phosphorus presents an environmental problem as its presence in bioavailable forms may result in eutrophication. Phosphorus in soil is relatively immobile and insoluble. South African soils are typically deficient in phosphates therefore the likelihood of exceeding the phosphate retention capacity of soils here is even lower.

Potassium is reported in feedlot wastes at between 445 and 1320 mg.l^{-1} (Mazzone *et al.*, 1992) and has been found in elevated levels in groundwater.

Elevated levels of **biological** or **chemical oxygen demand** (BOD/COD) are often a good indication of nutrient rich wastes. BOD levels of 7500 mg.l^{-1} in run-off from feedlots are reported (Mazzone *et al.*, 1992).

Dissolved organic carbon (DOC) is expected to be leached from the organic rich wastes of IAH.

4.1.3.1.3 Heavy metals

Copper and zinc are present as growth stimulants in feed for pigs, dairy cows and poultry (Funke, 1984). These are found in the solid waste but are not thought to pose a threat to groundwater quality.

Metals have been shown to be highly spatially immobile, even under conditions which increase their solubility and bioavailability to crops (Smith, 1996). So while they may pose a problem as soil contaminants the literature suggests that impact on groundwater quality is minimal.

4.1.3.1.4 Microbiological Pathogenic Indicators

Viruses, bacteria, protozoa and helminth eggs are present in animal excreta (Funke *et al.*, 1984). Pathogens are not thought to represent a major risk to groundwater quality, however, as most are effectively retained in the surficial few centimetres of soil by colloidal matter and do not remain viable for the time that it would take to transport them to the groundwater table (Smith, 1996). The unsaturated zone acts as a mechanical strainer, sieving out the larger pathogens such as protozoa and helminths, as well as providing adsorbtion surface for viruses (Smith, 1996).

Certain hydrogeological conditions cause groundwater to be more susceptible to microbiological contamination. Microbes remain viable for longer in soils which are moist, rich in organic matter, have a relatively high pH and a low temperature and where they are out of direct sunlight (Gerber *et al.*, 1975). There is greater risk of migration to groundwater where preferential pathways for vertical mass flow of effluent to the water table exist such as fissured rock or desiccation cracks.

Microbes are more likely to reach the water table if, at some stage in their life cycle, they are persistent and small. A good example of this is *Clostridia* which forms spores.

4.1.4 Objectives of this specialist study

The main objective of each specialist investigation carried out as part of this study is to collect data and information in order to confirm that groundwater contamination occurs as a result of the agricultural practice of concern and to collect evidence which can be used to support the non-technical booklet.

The literature review carried out as part of this project identified the potential impact from feedlots on groundwater quality as the most significant current agricultural contamination issue in South Africa. No data on the impact of IAH on groundwater quality in South Africa were found to be available. It was therefore decided that this sector should be investigated in the field to determine whether groundwater contamination due to IAH in South Africa concurred with results reported from other countries.

The objectives of the IAH specialist study were as follows:

- to select 3 sites for investigation where potentially vulnerable aquifers underlie IAH farms.
- to determine whether contamination of the groundwater had occurred as a result of IAH
- to determine, insofar as is possible within the budget and time constraints of the project, the nature and extent (spatially and temporally) of groundwater contamination
- to recommend to IAH farmers how they can best avoid groundwater contamination and protect their resource.

4.1.5 Approach

4.1.5.1 Field Site Selection

The Department of Agriculture, Department of Water Affairs, the Feedlot Association of South Africa and individual farmers were contacted to assess potential IAH sites. Three sites were selected for groundwater investigation based on the following criteria:

- the sites should be underlain by groundwater vulnerable to contamination from the surface
- sites should be representative of a range of IAH farming practices and hydrogeological settings
- the land owner(s) should be supportive of the study and willing to provide relevant information and access
- groundwater monitoring points should be established in the area (where possible) to minimise drilling costs

The aim was therefore to select sites where risk of groundwater contamination existed and where goodwill enabled the study to have access to relevant available data.

Figure 4.2 shows the location of sites investigated as part of this study.

4.1.5.2 Data acquisition

Where available, the following sources of data were acquired to give as complete an understanding as possible of the hydrogeology and contamination pathways at each site:

- Geological and hydrogeological literature on the locality
- Engineering/ geotechnical/ geophysical/ drilling reports on the locality
- Local data from the National Groundwater Database and National Water Quality Database
- Information on the local climate and precipitation
- Results of soil and water analyses carried out by previous researchers and land owners in the locality
- Information on site specific farm management practice, particularly with respect to effluent disposal and treatment.

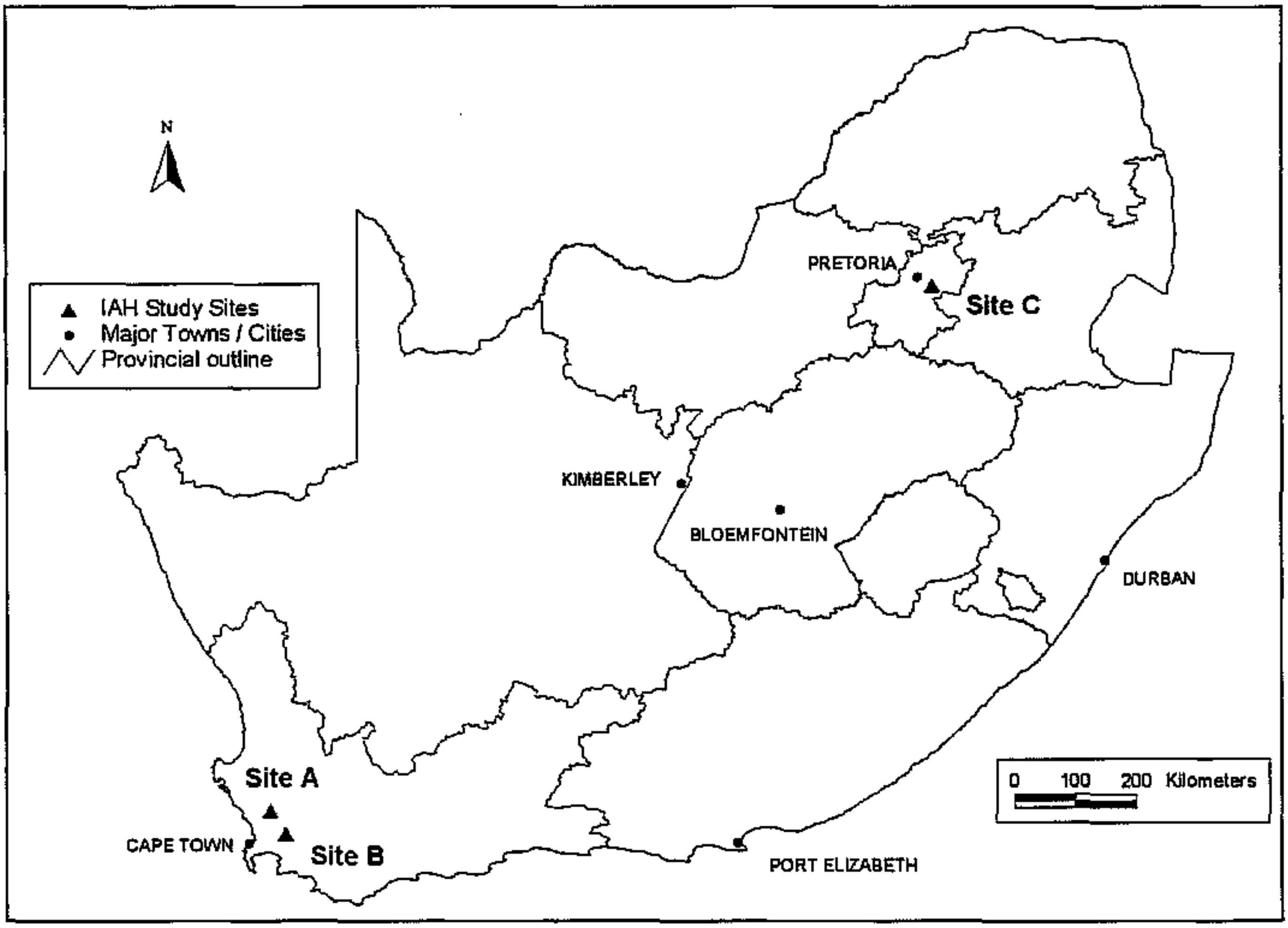


Figure 4.2 : Location of study sites

4.1.5.3 Sampling strategy

Groundwater monitoring points were established, where necessary, to ensure that representative samples of potentially contaminated groundwater could be collected and that one or more control sampling points were available. Where new boreholes were constructed, techniques were used to ensure that no potentially contaminating substances were introduced into the aquifer with experienced technicians or drillers carrying out the installation under the supervision of a hydrogeologist.

Where existing boreholes were used as monitoring points, information on their construction, history and performance was gathered from the owners to enable as complete an understanding as possible of the sampling conditions.

Groundwater sampling protocol was followed (as outlined in Weaver, 1992) to ensure that samples representative of water in the aquifer were analysed. This included careful use of sterile sample containers, purging at least 3 well volumes prior to sampling and storage for minimal periods of time (less than 48 hours) in the dark at temperatures of less than 4°C.

Soil samples were also collected at specific depths in the unsaturated zone and analysed for water soluble and potassium chloride extractable nitrate and ammonium by the Agricultural Research Council's Infruitek laboratory using standard methods.

4.1.5.4 Determinands

Field measurements of groundwater pH, EC and temperature were carried out at each sampling point. The colour and smell of the groundwater was also noted.

A suite of determinands for laboratory analysis was selected to include major ions usually found in groundwater and substances known to contaminate groundwater from livestock. These are outlined below:

Hydrochemical parameters: pH, EC, alkalinity, temperature, COD.
Major ions: Ca, Mg, K, Na, HCO_3, SO_4, Cl.
Common potential contaminants: NO_3, NH_4, DOC.
Pathogens: Faecal coliform, faecal streptococci, heterotrophic plate count, *Clostridia.*

The hydrochemical analyses were carried out at accredited CSIR laboratories using standard methods.

4.1.5.5 *Data quality*

The quality of hydrochemical data was analysed to ensure that the results were meaningful and representative of water in the aquifer. Quality control samples were routinely run during analysis. The ionic balance of the hydrochemical results was determined to confirm that the results were within an acceptable margin of error (<5%) and a comparison of pH and EC measured in the field and in the laboratory was made.

4.1.5.6 *Interpretation*

Results of hydrochemical analyses were interpreted in conjunction with information from literature, soil analyses, drilling records and anecdotal information on the local hydrogeology to determine the impact of IAH on groundwater quality at each site.

4.1.5.7 *Limits of the project*

The study aims to examine examples of the impact on groundwater quality in working agricultural situations. In looking at examples from 'the real world' an element of scientific control is sacrificed. Often the exact history of the site is not known and the influence of other potential sources of groundwater contamination is difficult to account for.

Budget, time and access constraints limit the density of sampling points at each study site and data must be extrapolated, often over tens to hundreds of metres, between data points. The degree of hydrogeological heterogeneity at each site is not fully accounted for in the simplified summaries of results.

In many cases where existing boreholes are used to sample, the depth and thickness of aquifer sampled are not known. This makes comparison between sampling points difficult.

Soil sampling has been carried out to determine nitrate and ammonium profiles in the unsaturated zone. This sampling gives a snap-shot with no indication of spatial or temporal variability. Detailed data on the soil moisture, grain size distribution and redox conditions would enable a better understanding of unsaturated zone processes affecting leaching of nitrate. However, the primary aim of this project is to determine whether groundwater contamination has occurred, rather than the detailed modelling of its occurrence.

4.2 SITE A

4.2.1 Site background

This case study examines the impact on local groundwater of a dairy operation in the Western Cape. The site was selected for the following reasons:

- several boreholes and a spring are established at the farm, providing groundwater sampling points
- problems with contaminated run-off were known to have occurred at the site and the potential for contamination of the subsurface due to poor waste management was high
- shallow, unconfined groundwater vulnerable to contamination occurs in the weathered zone of the granite
- the farmer was initially willing to cooperate with the study

The farm covers an estimated area of 1000 ha. It includes vineyards, wheat cultivation and a dairy with pasture. The dairy and pasture area covers approximately 50 ha, with intensive livestock concentration in about 6 ha.

Several problems were encountered during the field investigation which hampered data acquisition. Boreholes, originally described as accessible, were found to be blocked or dry. Prolonged water logging of some pasture areas delayed investigation of the shallow subsurface and the farmer became less available and willing to give information and access to the researchers once the project was under way.

It had been hoped to drill several research boreholes at the farm to gain a better understanding of the local hydrogeology. Drilling assistance was offered by DWA&F, however, it was not possible to undertake the drilling within the time available.

4.2.2 Operations at the dairy

The dairy herd consists of approximately 1200 cows, around 450 of which are producing milk at any one time. The non-producing cows are put to pasture on high ground to the south east of the dairy. Those that are being milked are kept in 6 stalls with concrete bases. There are small paddock areas (< 1 ha) adjacent to the stalls, unlined with limited grass cover, to which milking cows have occasional access. The route to the dairy is covered with concrete and waste is collected manually or flushed from this area and channelled to the solid separator.

Waste from the stalls is either removed manually or flushed with water. There is apparently no routine as to the removal method used. Flushed waste is channelled to the solid separator. Liquid effluent from the solid separator is then gravity fed to nearby effluent lagoons. Solid waste is temporarily stored in an uncovered concrete lined area near the separator and then sold or used on the farm as manure.

The solid separator was not running efficiently for the duration of this project. This resulted in the following problems:

- serious overflow of slurry waste in the area of the separator which then flooded the field down-slope
- effluent fed into the lagoon had a high proportion of solids
- stalls were flushed irregularly resulting in waste build up

This problem has been affecting waste management at the farm for some time and resulted in the farmer having to construct a dam at the down-slope boundary of his farm to prevent run-off contaminating his neighbour's land (apparently at a cost of R250 000).

The effluent lagoons are not artificially lined. The first lagoon acts a settling pond and, following the long period of inefficient solid separator operation, was almost full of settled waste. Liquid overflows from the first lagoon into a second lagoon where a floating pump draws it off for irrigation.

4.2.3 Land-use and potential groundwater contamination in the study area

Wheat cultivation and viticulture is carried out to the west and up gradient of the dairy area of the farm. Inorganic fertilizers and pesticides are applied here. Feed for the animals is stored under cover on concrete lined areas. Pit latrines are located around the site at labourers houses to the north east of the dairy.

Figure 4.3 shows land use on the farm and the position of groundwater and soil sampling points. The area of uncontrolled effluent run-off is also shown. The effluent lagoons are located immediately down-slope and to the north the dairy.

Potential sources of groundwater contamination associated with the dairy are identified as:

i. Infiltration from areas in the pasture where cows congregate (at feeding troughs and near gates)
ii. Infiltration from the paddock areas next to the stalls
iii. Infiltration from the liquid waste channels running to the solid separator and effluent lagoons
iv. Infiltration from the slurry flooded area down-slope of the solid separator
v. Infiltration from the effluent dams
vi. Infiltration from areas of temporary manure storage (if the base is cracked).

Of these, contamination from the flooded area down-slope of the solid separator was thought to pose the greatest risk of contamination to the subsurface as a high loading of waste occurred over an extensive area for a prolonged period of time with repeated periods of drying out. This meant that, unlike in the effluent lagoons where the surface is permanently saturated, there is less opportunity for an anaerobic sealing layer to form and limit soluble contaminant (particularly nitrate) downward migration.

Other potential sources of groundwater contamination on the site are as follows:

i. Use of inorganic fertilizer in the wheat fields and vineyards
ii. Infiltration from cultivated areas irrigated with effluent
iii. Infiltration from pit latrines and septic systems.

4.2.4 Hydrogeology of the site

The farm is situated on granites of the Cape Granite Suite. These intrude greywackes and phyllites of the Malmesbury formation.

Groundwater is used on the farm, abstracted from several boreholes and a spring. The granites in this area do not form high yielding aquifers due to the limited lateral extent and thickness of surficial weathered material, the presence of clays as weathering products and the limited extent of fractures. Most groundwater abstraction in the wider area is from fractures in shales and other metasediments of the Malmesbury formation.

Drilling records for boreholes established at the site are not available. The pump installation depths, rate of abstraction and draw-down are also apparently not known. The boreholes are installed with 6" diameter steel casing at the top but are thought to be open in the aquifer. The thicknesses of aquifer intercepted and strike depths are not recorded. Only one of the boreholes sampled was accessible to a water level meter (borehole 1) and the RWL here, in the upper pasture field, was approximately 5 mbgl.

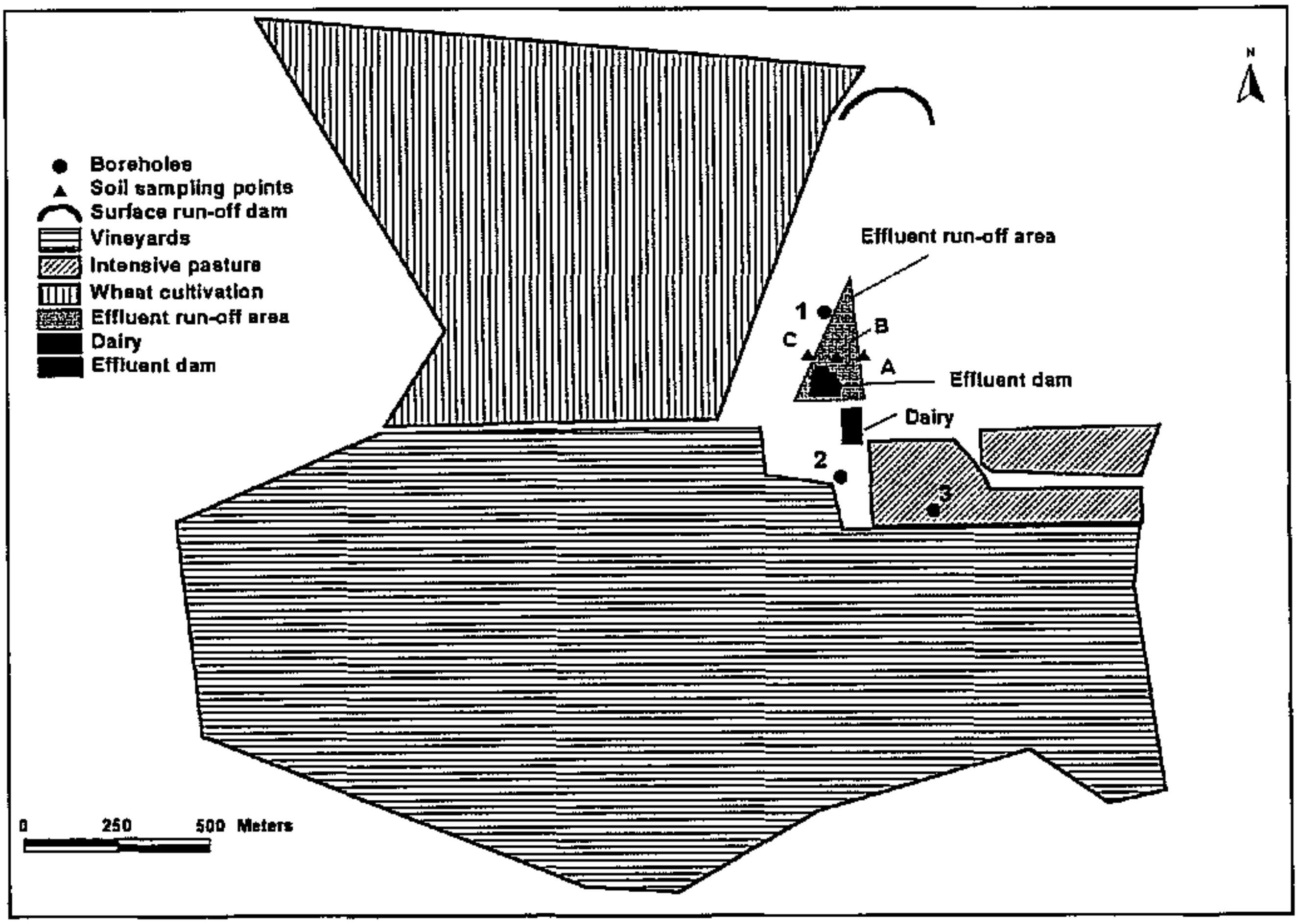


Figure 4.3 : Site A : Location of sampling points and land-use

Saturated subsurface conditions were not encountered during augering at the site. However, the occurrence of an ephemeral spring at the site indicates the presence of limited shallow groundwater. The spring is located on the edge of a slope (but not the lowest topographic point in the area), possibly at the contact between the fine to medium even-grained granite and coarse porphyritic granite. Its location and the surrounding surficial cover indicate coarse to medium grained, weathered regolith aquifer, probably with limited thickness, overlying less permeable unweathered granite.

The annual **rainfall** in this area is approximately 500 mm and occurs mainly in the winter months (Midgley *et al.*, 1994). Potential **evaporation** is around 1500 mm p.a. (Midgley *et al.*, 1994).

4.2.5 Field investigation

The field investigation was carried out in June 1996 and March 1997. Figure 4.3 shows the position of sampling points in the dairy area of the farm. All boreholes which were either operational or accessible to a bailer were sampled. These were as follows:

(1) Production borehole in a field down-slope of the effluent dam, pumped daily to supply the dairy. Sampling point at the dairy outlet.
(2) Production borehole near the dairy office, pumped daily to supply the dairy. Sampling point at the well head.
(3) Disused, uninstalled borehole in the up-slope pasture close to a feeding trough and surrounded by an area with a high excrement loading. Sampled by bailing.

In addition, a sample was taken from the spring at a point were groundwater appeared to be seeping in through weathered material. Groundwater samples were analysed for major ions and selected samples for microbial faecal indicators (faecal coliforms, faecal Streptococci and Clostridium).

Soil samples were obtained by hand augering to between 0.6 and 2.0 m where the ground became impenetrable due to the presence of clay. The sites for soil sampling were chosen to represent a control area (A), an area flooded with effluent and solid waste from the dysfunctional solid separator (B) and an area down slope of the first effluent dam (C). Nitrate and ammonium-nitrogen analyses were carried out on the samples to give an indication of different nitrogen soil profiles with depth.

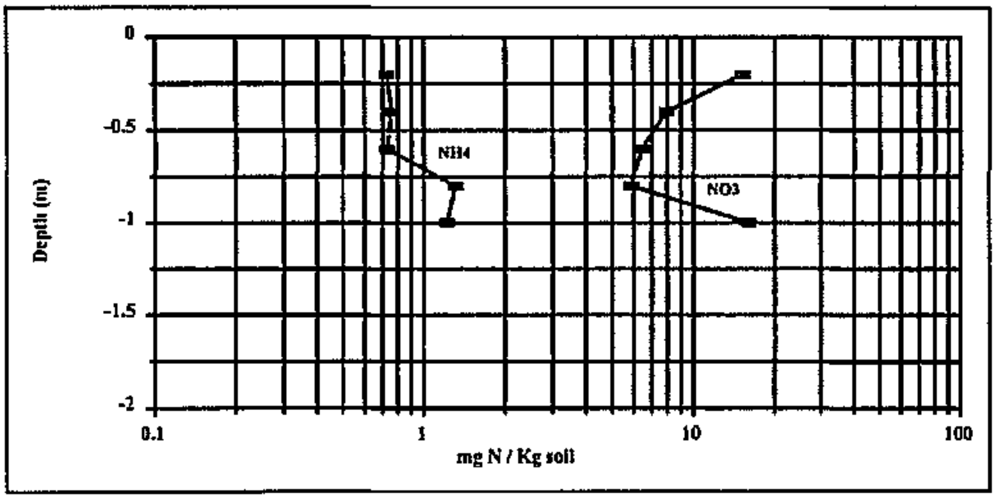


Figure 4.4: Site A: Sampling point A: Nitrogen soil profile of NO_3 - N and NH_4 - N (mg N/kg soil)

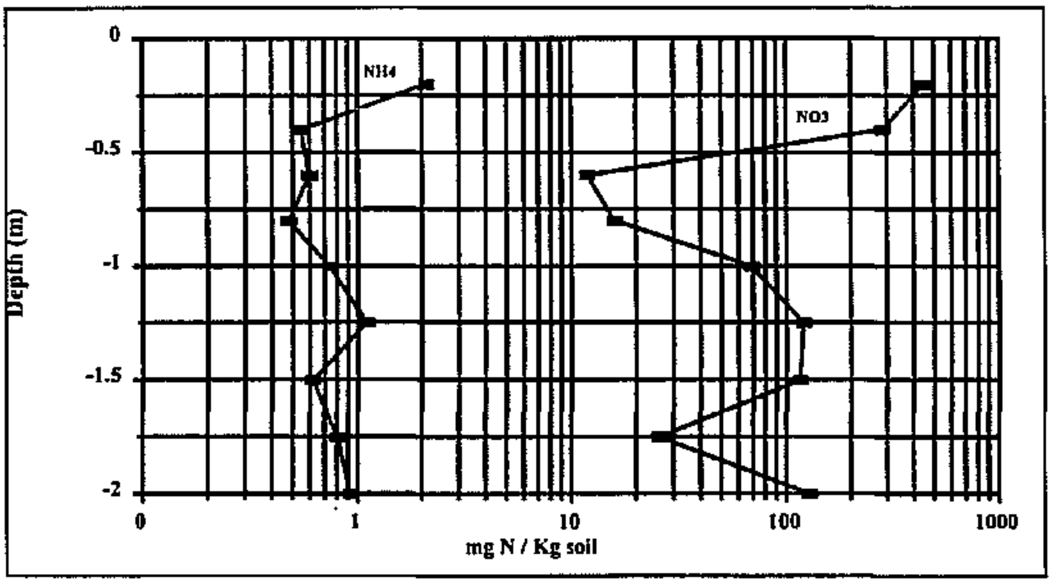


Figure 4.5: Site A : Sampling point B: Nitrogen soil profile of NO_3 - N and NH_4 - N (mg N/kg soil)

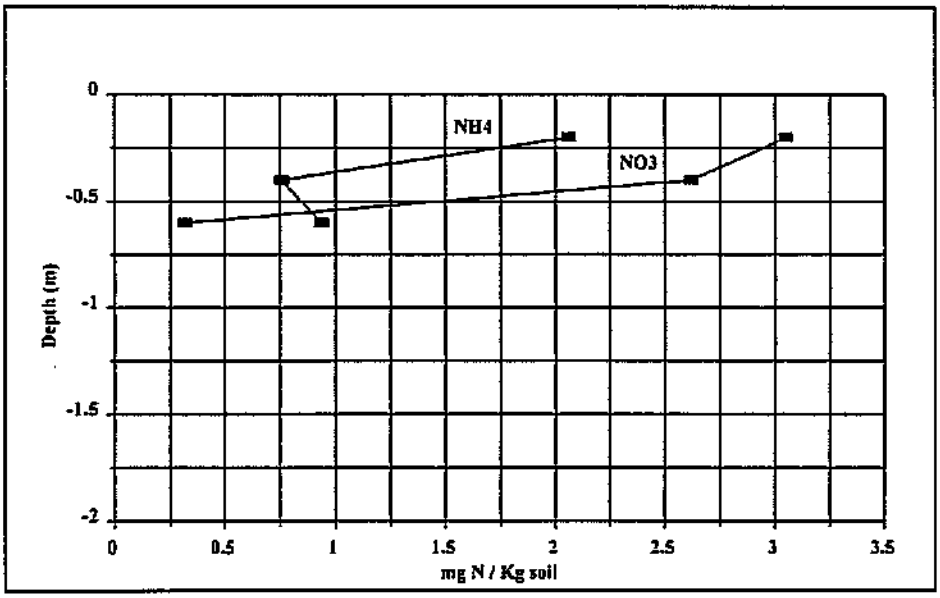


Figure 4.6 : Site A : Sampling point C: Nitrogen soil profile of NO_3 - N and NH_4 - N (mg N/kg soil)

4.2.6 Results of the field investigation

4.2.6.1 The unsaturated zone in the effluent/waste flooded area

The water table was not encountered during hand augering in the field down-slope of the solid separator and the effluent dams. The unsaturated zone in this area was found to comprise fine to coarse quartz sand, becoming clayey with depth until, at between 0.6 and 2.0 m, hard clay dominated.

The results of soil analyses are shown in Figures 4.4 to 4.6. The levels of nitrate are low in the control sample (A), as would be expected, at around 3 mg N. kg soil $^{-1}$. Nitrate levels are dramatically higher in the sample taken from the waste flooded area (B) at 200 to 400 mg N. kg soil $^{-1}$ in the samples from <0.5 mbgl. Nitrate - nitrogen levels decline slightly but remain high with depth at this sample site varying from 10 to 100 mg N. kg soil $^{-1}$. Ammonium-nitrogen concentrations are not significantly elevated compared to the control sample.

Similarly at sampling point C, on the edge of the waste flooded area and down-slope of the effluent dam, ammonium levels are comparable to the control site. The concentration of nitrate-nitrogen is elevated to 15 mg N. kg soil $^{-1}$ in the upper most and deepest samples and is slightly higher in the rest of the profile. The levels are not as significantly elevated as those seen in the centre of the flooded area.

The occurrence of sporadic flooding of waste slurry from the solid separator in this area therefore has a significant effect on nitrate levels in the unsaturated zone. Ammonium levels are not seen to be affected. Nitrogen is likely to be present mainly in organic form in the slurry. Over time organic nitrogen will be mineralized to ammonium and some losses to the atmosphere due to volatilization are likely to occur. Drying out of the slurry on the soil allows nitrification to occur in aerobic conditions and a proportion of the nitrate produced will then infiltrate the permeable unsaturated zone, probably mobilized during periods of precipitation or the next flooding of slurry. The degree of mobility of nitrate in the less permeable clay layers is not known. These layers may attenuate nitrate contamination, particularly if reducing conditions exist. The results indicate that there is certainly potential for nitrate contamination to occur, however, the presence of a sufficiently thick, low permeability layer may mitigate any impact on groundwater quality.

4.2.6.2 Description and discussion of hydrochemical data

The results of all hydrochemical analyses are presented in full in Appendix 1.B. The hydrochemical data is deemed to be of good quality as the charge balance error is within acceptable limits (<5%) and parameters measured in the field correlate well with those measured in the laboratory.

The percentage difference between the results of the two sampling runs (also shown in Appendix 1.B) is low and does not exceed 4% for any results where determinands were detected in both runs. This indicates a fairly consistent groundwater hydrochemistry at individual sampling points.

Hydrochemical interpretation of results is made difficult by the fact that so little data are available on the depth and lithologies encountered during construction of the boreholes sampled. It is not known whether the boreholes intercept the same aquifer or even the same aquifer lithologies and what the relative flow directions are.

The EC of the groundwater sampled varies from 99 to 278 $mS.m^{-1}$. EC appears to increase upslope and southwards. This is the opposite of what would be expected if groundwater flow follows the topography. However, these boreholes may be intercepting deep fractures in which flow is not related to the local scale topography or intercepting aquifers of limited extent which are not in hydraulic continuity.

Figure 4.7 shows a **Piper plot** of the results of hydrochemical analysis. The samples plot fairly close together and are all of approximately NaCl type. This indicates that the boreholes and the spring are fed by groundwater from the same aquifer type. It can be seen in the field that the spring is fed from the weathered zone of the granite, therefore it is likely that the boreholes also receive significant groundwater contribution from this lithology. They may also intercept fractures in the granite at depth.

The **stiff diagrams** presented in Figure 4.8 show data from all the groundwater sampling points. Their similar hydrochemical nature is obvious on these diagrams. The highest TDS sample from borehole 3 has a higher level of $CaCO_3$.

Elevated **nitrate-nitrogen** values are seen at boreholes 1 and 2 and at the spring. The borehole intercepting groundwater identified as being potentially most at risk, down-slope of the effluent dams and the flooded area around the solid separator (borehole 2), has nitrate levels elevated to 13 $mg.l^{-1}$. The closest source of contamination at this borehole is from slurry flooded onto the nearby field. Hard clay layers intercepted during augering in this area may not act as a complete barrier to contaminant migration from the surface due to their limited extent or the existence of preferential pathways for flow. In addition to contamination from the waste flooded area, there may be some infiltration from the effluent dams.

The highest levels of nitrate-nitrogen, 27 $mg.l^{-1}$, were detected in samples from borehole 2 which is located near the dairy and offices. The area immediately around the well head is paved with concrete and hard surfaces are constructed for a minimum of 100 m in all directions. The recharge area for this borehole is not obvious from the data available and therefore the source of nitrate contamination is difficult to define.

Moderate nitrate levels (4 $mg.l^{-1}$) were seen in samples from the borehole in the upper pasture. This level is likely to be higher than ambient levels and may originate from the area close to the well head which was seen to have a high loading of cow excrement as it is close to a feeding trough. However, the hydrogeology intercepted by this borehole is not known therefore there is significant uncertainty as to the source. Fertilizer applied in the vineyards up-slope of the borehole may have also contributed to nitrate in the groundwater here.

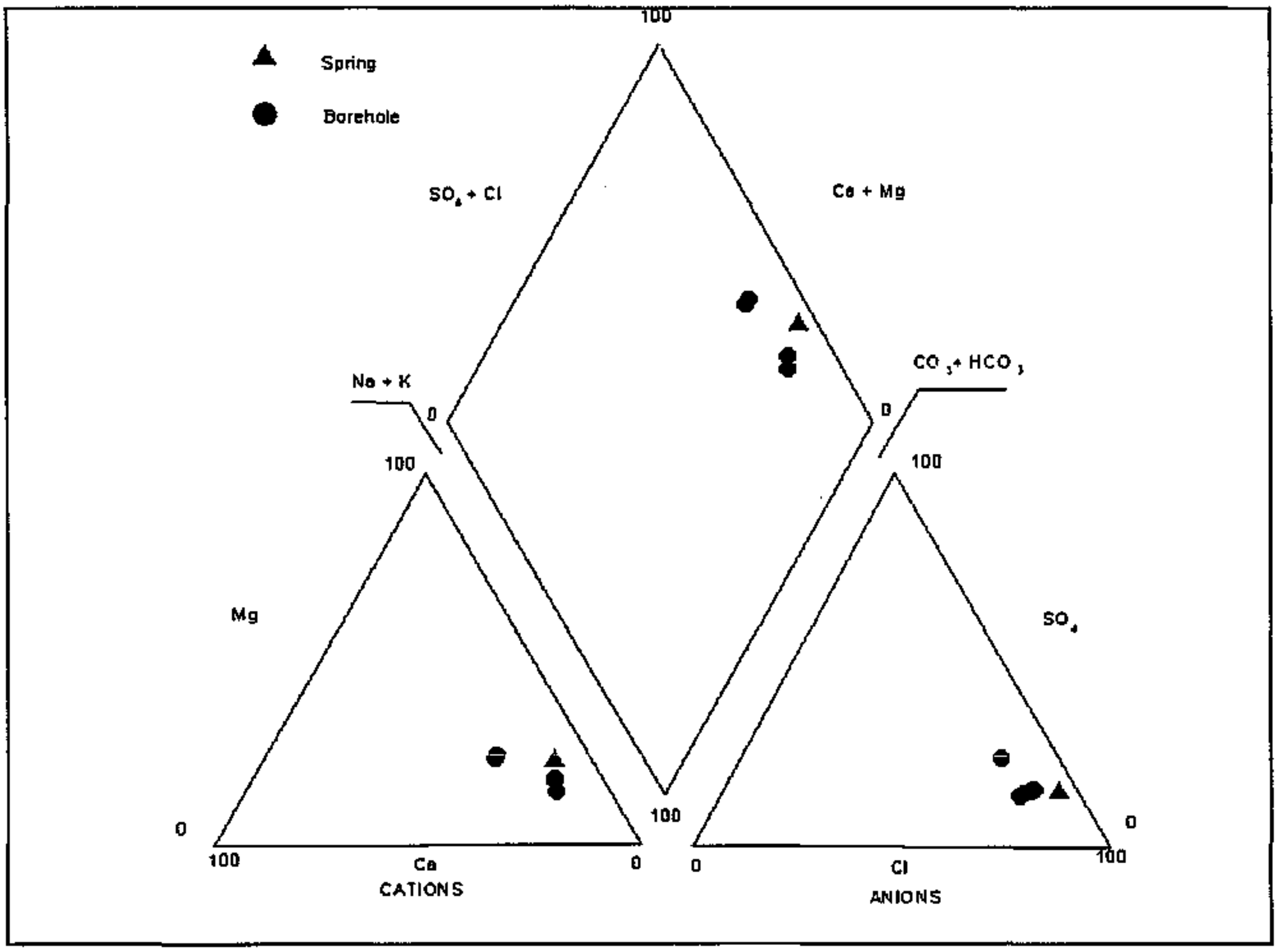


Figure 4.7 : Site A : Piper plot of hydrochemical results.

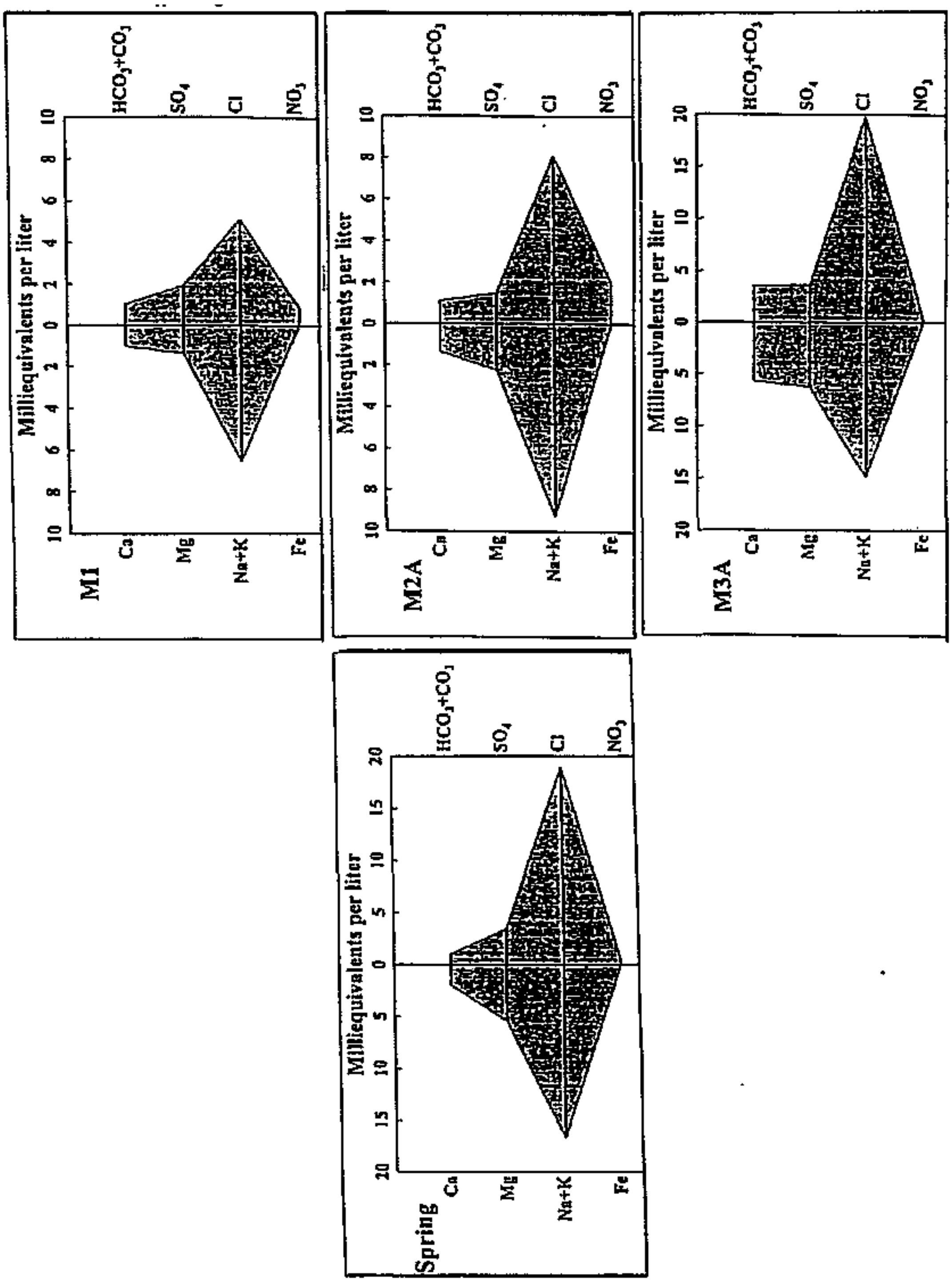


Figure 4.8 : Site A : Stiff diagrams

Water sampled from the spring contained levels of nitrate elevated to 8 mg.l^{-1}. Possible sources close to the spring and up slope of it are the paddock areas outside the stalls, which receive a heavy loading of cow manure, or fertilizer applied to the area of wheat cultivation.

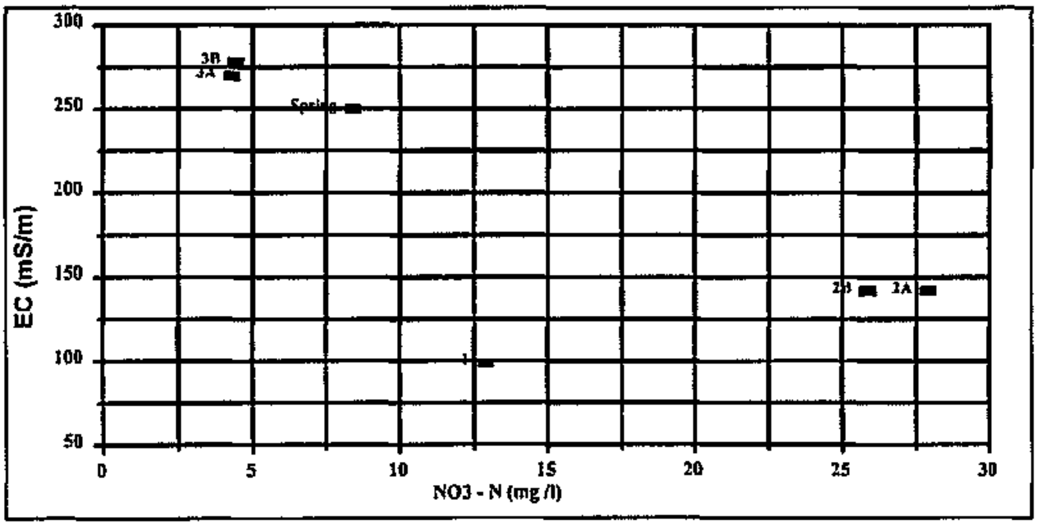


Figure 4.9 : Site A : NO_3-N versus EC

Figure 4.9 shows an approximately negative correlation between nitrate levels and EC of the groundwater. This indicates that nitrate is not a dominant ion in the ground water samples and possibly that groundwater that has been contaminated with nitrate is younger (shallower) water.

No significant correlation between nitrate levels and **potassium** can be determined from the data available, although there may be a negative correlation, as shown in Figure 4.10. Water sampled from the spring has the highest potassium levels at 10 mg.l^{-1}. This may be the result of natural hydrochemical variation in a more potassium rich zone of the weathered granite aquifer or indicate contamination from N:P:K inorganic fertilizers used in the wheat cultivation area up slope of the spring.

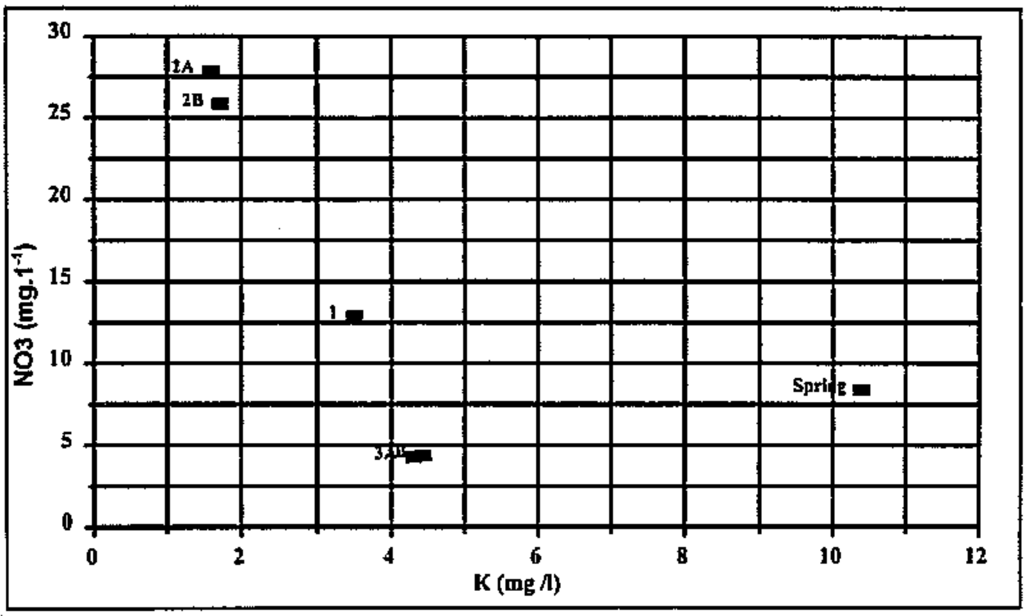


Figure 4.10 : Site A : NO_3 - N versus K

Ammonium and **orthophosphate** were not detected in any of the samples. **DOC**, another potential contaminant associated with animal waste, was detected in only one sample, at borehole 2 where the highest nitrate levels were also seen, but at an insignificant level of 1.5 mg.l^{-1}.

4.2.6.3 Microbiological analysis

Samples from boreholes 1 and 2, which showed the highest nitrate levels, were analysed for microbial indicators of faecal pollution. Indicators were not detected in the sample from borehole 1. Low levels of faecal Streptococci (2 per 100ml) and Clostridium (1 per 100 ml) were detected in the sample from borehole 2. This indicates a faecal source of contamination of groundwater from this borehole, possibly from the dairy.

The lack of microbial indicators in groundwater from borehole 1 may be due to microbe attenuation across the greater distance from faecal source to groundwater.

4.2.7 Summary of impact of the IAH operations on groundwater quality

Groundwater was found to be contaminated with nitrate and microbial indicators at levels of concern at some sampling points. No other potential contaminants were detected.

High levels of nitrate contamination at 27 mg.l^{-1} were detected at one groundwater sampling point. The presence of microbial faecal indicators in groundwater from this borehole indicates a faecal source for the nitrate. However, the borehole is located in a paved and constructed area, up-slope of the cow stalls and close to the dairy, and the location of a contamination source is not evident. The depth and thickness of aquifer intercepted, abstraction rate and aquifer parameters are not known therefore the borehole capture zone cannot be determined.

Elevated nitrate levels (12 mg.l^{-1}) were also detected in groundwater from borehole 1, down-slope of the effluent dam and close to an area flooded with animal waste slurry from the dysfunctional solid separator. Samples from the unsaturated zone showed high nitrate levels associated with the slurry flood therefore this is thought to be a likely source. Potentially attenuating clay layers were identified in the unsaturated zone in this area but their extent is not known.

Water from the spring had nitrate levels of 8 mg.l^{-1}. It is not known if this represents contamination from operations associated with the dairy of if it is due to fertilizer application to the wheat area.

Groundwater from borehole 3 in the pasture, which had a high excrement loading close to the well head, contained nitrate at 4 mg.l^{-1}.

In summary, possible sources of groundwater contamination are thought to be:

- the area flooded with animal waste slurry down-slope of the solid separator
- animal waste in the dairy
- pasture and paddock areas outside the stalls
- fertilizer applied to wheat

4.2.8 Synopsis

This investigation looks at the impact of a moderately sized dairy on local groundwater. The site is thought to be underlain in places by shallow groundwater hosted in weathered granite of the Cape Granite Suite, and possibly deeper groundwater in fractures in the granite.

Waste management is not practised efficiently at this site. In particular, repeated breakdown of the solid separator meant that cow stalls were not flushed regularly, excess solids were fed into the effluent dam and waste slurry periodically flooded the field down-slope.

All available groundwater sampling points, 3 boreholes and a spring, were sampled. Soil samples were also collected to gauge the impact on the unsaturated zone of the waste slurry flood.

Significant nitrate contamination (to 400 mg N. Kg $soil^{-1}$) of the unsaturated zone was found beneath the waste slurry flooded area.

The quality of groundwater is variable across the site with EC between 99 and 278 $mS.m^{-1}$. The spatial variation in quality does not correlate to expected flow directions and may indicate restricted hydraulic continuity across the site. All the groundwater sampled is approximately of NaCl type.

The only potential contaminants detected at levels of concern were nitrate, at boreholes 1 and 2, and faecal indicators, at borehole 2. Slurry waste flooded onto the field is thought to be the most important source of contamination at borehole 1, with possible additional contribution from the effluent dams. The dairy operation is thought to be responsible for nitrate and microbial contamination at borehole 2 but the specific sources are not known. Groundwater from the spring and borehole 3 also contained nitrate.

4.2.9 Conclusions

This study has shown contamination of the unsaturated zone and possibly groundwater to have occurred as a result of animal waste slurry flooding a field area over the course of at least several months. The flooding is a preventable aspect of operations at this dairy. Better waste management practices and attention to the proper working of the solid separator would result in more efficient handling of waste in the cow stalls, improve the quality of effluent fed to the lagoon and prevent slurry overspill.

It is not known what causes the highest level of nitrate contamination seen at borehole 2. The presence of microbes indicates a faecal source which is probably associated with the dairy rather than other agricultural operations at the site. Detailed inventorying of the surrounding area (with the open cooperation of the farmer) and possibly nitrogen isotope analysis would help to determine the source.

Insufficient data for this site has limited the interpretation of that information which is available. The quality of understanding of contaminant processes in operation would have been significantly improved by more cooperation from the farmer, access to drilling records and investigative drilling at the site.

4.2.10 Recommendations for further investigations

It would be interesting to obtain soil nitrogen profiles to the water table to assess the attenuation potential of clay layers in the unsaturated zone. Mechanical augering would be necessary to obtain samples in and below the hard clays.

It had been hoped to drill new boreholes at this site to gain a better understanding of the hydrogeology. This was not possible within the time available but should be carried out if investigations are continued and if the farmer is willing to cooperate.

Isotope analyses of nitrate in groundwater samples would help to determine the source(s) of the nitrogen.

4.3 SITE B

4.3.1 Site background

This case study examines the impact on local groundwater of an intensive piggery and dairy operation in the Western Cape. The site was selected for the following reasons:

- the site is underlain by shallow, unconfined groundwater vulnerable to contamination
- the shallow depth of the groundwater meant that samples could be obtained cheaply and easily
- previous investigation of shallow groundwater and surface water at the site showed that contamination had occurred
- the farmer was willing to cooperate with the study

The farm covers 3 600 ha in total and includes vineyards, orchards, a fruit cannery and frozen food processing unit, a piggery and a dairy with pasture.

4.3.2 Waste management at the dairy and piggery

Approximately 6000 pigs are being reared in the piggery. The dairy herd consists of around 800 Ayrshire cows, around half of which are producing milk at any one time.

The pigs are reared intensively in stalls with concrete flooring. These are cleaned by flushing with water up to three times per day. Some solid waste is removed manually.

Dairy cows are kept in stalls with concrete bases and on pasture land. In the stalls, most of their waste, which is mixed with straw, is removed manually. The stalls are also flushed with water occasionally.

Solid waste is stored temporarily in an uncovered concrete lined area near the dairy and then sold or used on the farm as manure. Liquid slurry waste is gravity fed by concrete and brick lined channels from the stalls to a solid separator. Solids are added to the manure stockpile and the liquid phase is pumped to the effluent lagoon. The lagoon also receives effluent from the cannery and food freezing plants. This type of effluent typically has a high COD of several thousand $mg.l^{-1}$ and a similar level of dissolved organic solids. Total dissolved inorganic solids are normally of the order of several thousand $mg.l^{-1}$ and TKN (mainly organic) is typically from 20 to 100 $mg.l^{-1}$. These levels are similar to those which would be expected in piggery and dairy effluent, except for TKN which may be higher in IAH waste streams.

The table below shows the results of analyses of the different effluent streams entering the lagoon which were carried out in 1995 (Adams, 1995). The sample from the dam is probably most representative of the effluent that is used to irrigate pasture. The character of the two liquid waste streams is similar for the determinands measured, although the piggery and dairy stalls stream appears to have a higher level of ammonia contamination. The reported concentration of phosphate is regarded with some suspicion as it is significantly higher than either of the two waste streams feeding the dam. It may be that this value is for total phosphate rather than soluble phosphate. It should be noted that these are spot samples and give no indication of the range of values that may occur.

Table 4.1: Results of effluent analysis

Sample source	Phosphate (ppm)	Nitrate (ppm)	Ammonia (ppm)	Conductivity (mS/m)
Dam	129	<0.1	189.6	23
Effluent stream from piggery and dairy stalls	8.3	<0.1	201.2	32
Effluent stream from food processing units and dairy	10.9	<0.1	138.0	16

Source: Adams, 1995.

The lagoon is not artificially lined. It acts as a temporary balancing pond for effluent prior to disposal by irrigation onto pasture. A floating pump draws off effluent which is then piped through the sprinkler irrigation system and applied to approximately 30 ha of pasture.

4.3.3 Land-use and potential groundwater contamination in the study area

Fruit cultivation (mainly peaches and apples) and viticulture are carried out to the south and up gradient of the IAH area of the farm. Inorganic fertilizers and pesticides are applied here. Feed for the animals is stored under cover on concrete lined areas. Pit latrines are located around the site at labourers cottages.

Figure 4.11 shows land use in the area of the farm concerned with IAH. Effluent is applied to flat and gently sloping pasture either side of the river that runs northwards across the site. The rate of application is determined by disposal needs rather than agronomic requirements. Different pasture areas are irrigated on a fairly ad hoc basis. The effluent lagoon is located approximately in the centre of the area with the dairy and piggery stalls in the south west of the area.

Potential sources of groundwater contamination associated with IAH in the study area are thought to be:

i. Infiltration from unconstructed areas where cows congregate - at feeding troughs and near gates in the pasture and around access points to the dairy.
ii. Infiltration from the liquid waste channels running to the solid separator
iii. Infiltration from the effluent dam
iv. Infiltration from pasture irrigated with effluent
v. Infiltration from areas of temporary manure storage (if the base is cracked).

Other potential sources of groundwater contamination on the site are as follows:
vi. use of inorganic fertilizer in the orchards and vineyards
vii. contamination with effluent from the food processing and canning operation
viii. infiltration from pit latrines and septic systems.

Previous investigations detected elevated levels of some contaminants in the Keurbos river, which flows through the farm, and in some groundwater samples (Adams, 1995). Nitrate levels are fairly stable along the length of the river at 5 $mg.l^{-1}$. Phosphate concentrations appear to increase downstream, from below the detection limit of 0.1 $mg.l^{-1}$ to 1.9 $mg.l^{-1}$. There is a very slight increase in ammonia levels in the river to just above the detection limit (0.19 $mg.l^{-1}$) and conductivity of the surface water increases from 1.8 to 3.0 $mS.m^{-1}$. Faecal coliform counts increase along the length of the river in the farm, from > 1000 to too numerous to count.

Waste water from the farm is not discharged directly into the river therefore elevated levels of contaminants indicate either contribution from contaminated baseflow or run-off of irrigated effluent into the river.

4.3.4 Hydrogeology of the site

The farm is situated on unconsolidated recent deposits of thick alluvium containing large pebbles and boulders. The alluvium is underlain by an intrusive granite of the Cape Granite series, the Stellenbosch pluton (Adams, 1995). The granites intrude sandstones of the Table Mountain and Malmesbury groups.

Groundwater is not used on the farm but boreholes are known to exist in the area. Augering at the site in November (a few months after the late winter rains) intercepted saturated conditions at between 0.5 and 1.5 mbgl. It was not possible to determine relative RWLs at the sampling points as they were not accurately surveyed. It is assumed that shallow **groundwater flow** approximately follows the topography (northwards) at the site. Shallow groundwater is likely to be in continuity with the Keurbos River which flows approximately northwards through the farm.

The annual **rainfall** in this area is approximately 900 mm and occurs mainly in the winter months (Midgley *et al.*, 1994). Potential **evaporation** is around 1400 mm p.a. (Midgley *et al.*, 1994).

4.3.5 Field investigation

The field investigation was carried out in November 1996. Figure 4.11 shows the position of sampling points in the IAH area of the farm. Sampling points were chosen to target different potential sources of groundwater contamination associated with the feedlot as outlined below:

(i) An area in pasture with low grass cover and heavy cow manure loading where cows congregate close to a gate and feeding trough.
(ii) Pasture directly down slope of the effluent dam and on the edge of an effluent irrigated area.
(iii) A low lying pasture irrigated with effluent close to the river.
(iv) Pasture on higher ground irrigated with effluent.
(v) Control sample from non-irrigated, non-cultivated bush area.
(vi) Close to the manure stockpile and liquid waste channel.

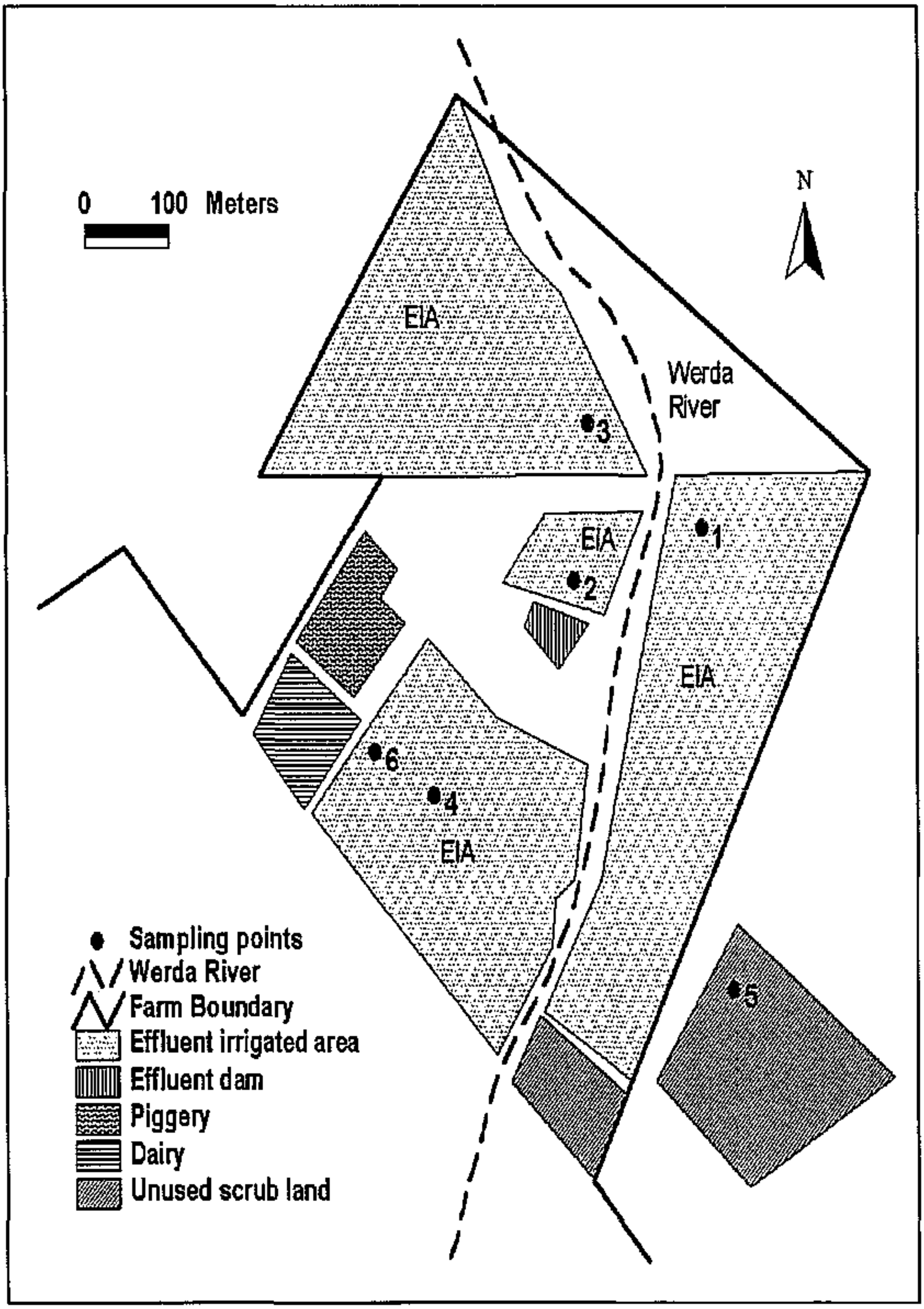


Figure 4.11 : Site B : Position of groundwater sampling points and land-use

Groundwater and soil samples were obtained by hand augering to approximately 0.5m below the water table. uPVC, 50 mm ID casing with 0.5m of screens was temporarily emplaced at each site. The casing was cleaned with distilled water between sampling. The installation of permanent well points was not permitted as this would interfere with the IAH operations. Temporary well points were purged with a battery operated pump for about half an hour to obtain a sample of groundwater representative of the aquifer.

Nitrate and ammonium-nitrogen analyses were carried out on soil samples from augering sites 1, 3 and 4 to give an indication of different nitrogen soil profiles with depth.

4.3.6 Results of the field investigation

4.3.6.1 Augering

A description of lithologies encountered during augering successful boreholes is given on the sampling sheets in Appendix 1.C. Several unsuccessful augering sites were abandoned, where impenetrable clays were encountered. In particular to the north east of the effluent lagoon.

The depth to the water table is less in the topographically higher parts of the study area and away from the river (< 1 m) as is shown in Table 4.2. The lithologies and variable depths to the water table encountered during augering indicate that the study area is underlain by a multiple surficial aquifer system, typical of alluvial deposits, with variable hydraulic continuity across the site and with depth. These results are summarised in the table below.

Table 4.2: Depth and lithology at water table in relation to topographic height and distance from the river

Sampling point number	Depth to WT (mbgl)	Approx. topo. height (mamsl)	Approx. distance from river (m)	Lithology at WT
1	2	147	70	Sand
2	1.2	150	100	Sandy clay
3	1.2	145	100	Sandy clay
4	0.5	163	200	Sand & boulders
5	0.8	167	220	Sand & boulders
6	0.5	163	300	Sand & boulders

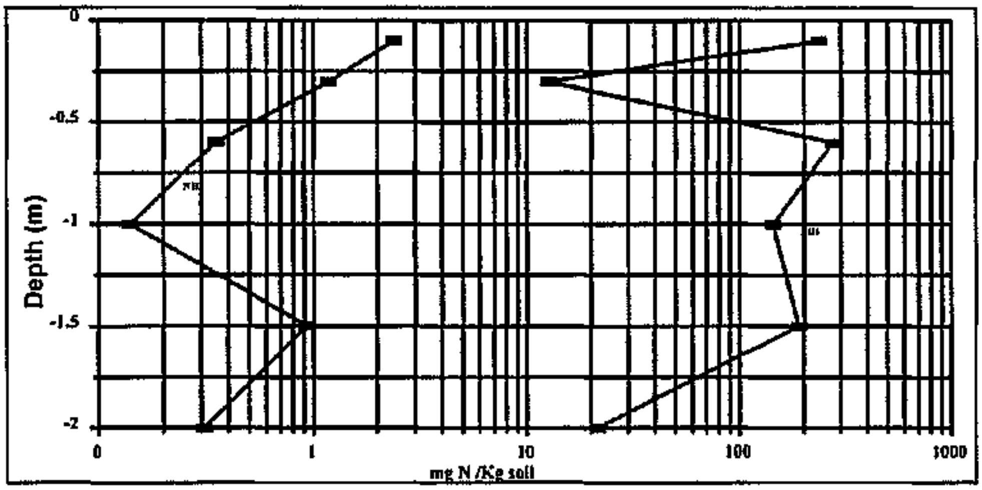


Figure 4.12 : Site B : Soil nitrogen profile: sampling point 1

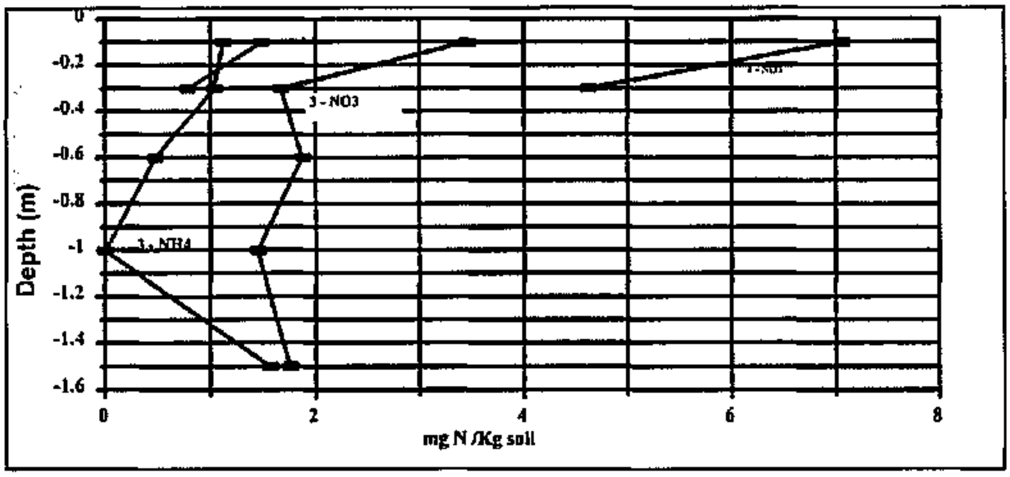


Figure 4.13 : Site B : Soil nitrogen profile: sampling points 3 and 4.

The results of nitrate and ammonium analyses of soil samples from selected sites are shown in Figures 4.12 and 4.13. Sites 3 and 4 are in pasture irrigated with effluent and site 1 is in an area of pasture where cows congregate close to a gate and feeding trough with a heavy loading of excrement and no grass cover.

Levels of ammonium nitrogen are similarly low at all sites and remain low throughout the profile. Ammonium levels in the effluent are high at around 180 $mg.l^{-1}$, therefore it appears that it is rapidly nitrified and/or volatilizes in this environment and is not mobile in the subsurface.

Nitrate levels are very high at the surface and with depth in samples from site 1, averaging 200 $mg.l^{-1}$. This shows that the high nitrate levels detected in groundwater at this site are the result of contamination from the surface in the immediate locality and that leaching is currently occurring in the permeable sandy alluvium.

The shallow samples analysed at site 4 showed moderately elevated nitrate levels at 4 to 7 $mg.l^{-1}$. Nitrate detected in groundwater here is therefore likely to originate from the effluent irrigated to the pasture. The alluvium encountered here is permeable, comprised of sand with boulders at depth. Nitrate levels in the unsaturated zone may have been higher following previously higher levels of application.

No nitrate contamination was detected in groundwater from site 3, and the nitrate profile shows that levels are low at 3.5 $mg.l^{-1}$ and decline with depth to < 2 $mg.l^{-1}$. The application of ammonium rich effluent at this site has not resulted in nitrate contamination of the subsurface. Assuming that rates of nitrification, nitrate uptake and volatilization will be approximately the same at surface in pasture across the study area, lower nitrate levels here may be the result of a lower rate of application to this pasture than at site 4 and/or the less permeable subsurface. A greater proportion of clay was seen in samples at site 3.

4.3.6.2 Description and discussion of hydrochemical data

The results of all hydrochemical analyses are presented in full in Appendix 1.C. The hydrochemical data is deemed to be of good quality as the charge balance error is within acceptable limits (<5%) and parameters measured in the field correlate well with those measured in the laboratory. A summary of the results for impact related determinands is presented in Table 4.3 below.

The **EC** of the groundwater sampled is variable across the site. The value for the control sample (19 $mS.m^{-1}$) is approximately an order of magnitude lower than at sites with potential contamination. The highest EC (366 $mS.m^{-1}$) is seen at site 1 where groundwater occurs in permeable sand beneath a thick excrement layer. Elevated EC is therefore likely to be an indication of levels of contamination.

Figure 4.14 shows a **Piper plot** of the results of hydrochemical analysis. The plot shows a wide spread in the hydrochemistry of groundwater sampled at the site. Bicarbonate and chloride are the dominant anions. Variation in their relative proportions does not appear to correlate to different intensities of contamination as the control sample and the sample with the highest nitrate level both have the highest proportion of chloride. Sodium and potassium are the dominant cations.

The **stiff diagrams** presented in Figure 4.15 show data from the control area and 3 samples from the area impacted by the IAH activities. The control sample (B5) has a sodium chloride hydrochemical signature with minor bicarbonate. The low TDS nature of this groundwater means that the milliequivalent scale is different to the other samples shown. The sample B1, from the manure covered trough/gate area in the pasture is a similar sodium/potassium chloride type with significant nitrate and calcium and magnesium. The samples from the effluent applied pastures are dominated by sodium and potassium with dominant bicarbonate at B3 but approximately equal anionic proportions of bicarbonate, chloride and nitrate at B4. Chloride and nitrate are therefore major ions whose concentration is influenced by effluent contamination.

All of the samples analysed, including the control, show some hydrochemical and /or microbiological indication of contamination from animal waste, as indicated in table 4.3. The only indication of contamination in the control is the presence of faecal coliforms at 47 per 100 ml (the lowest value where faecal coliforms are detected). The source of faecal contamination is not known. The most likely source is from informal grazing of livestock in this bush area. Other determinands do not indicate any impact on groundwater quality from faecal contamination.

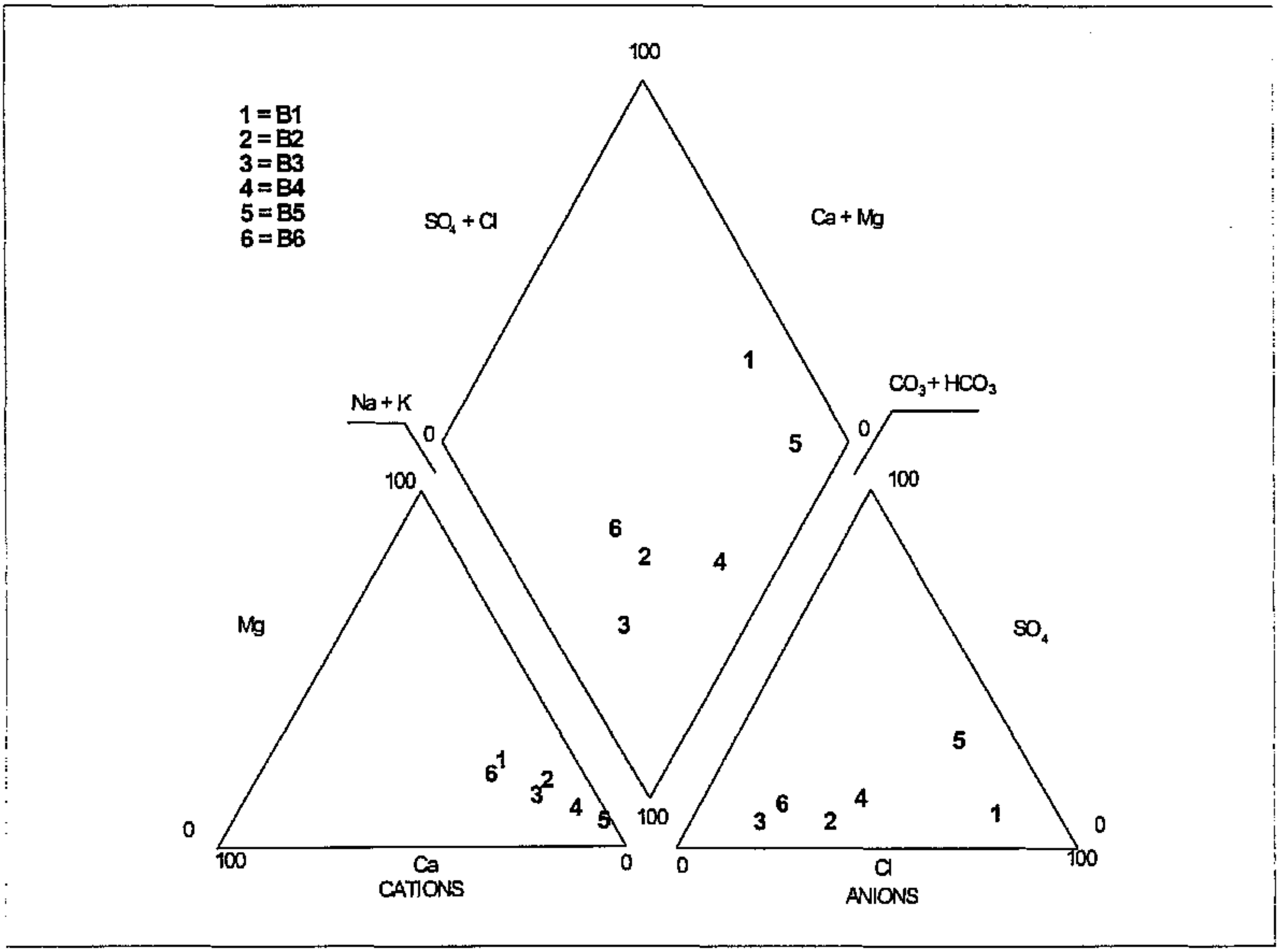


Figure 4.14 : Site B : Piper plot of hydrochemical results.

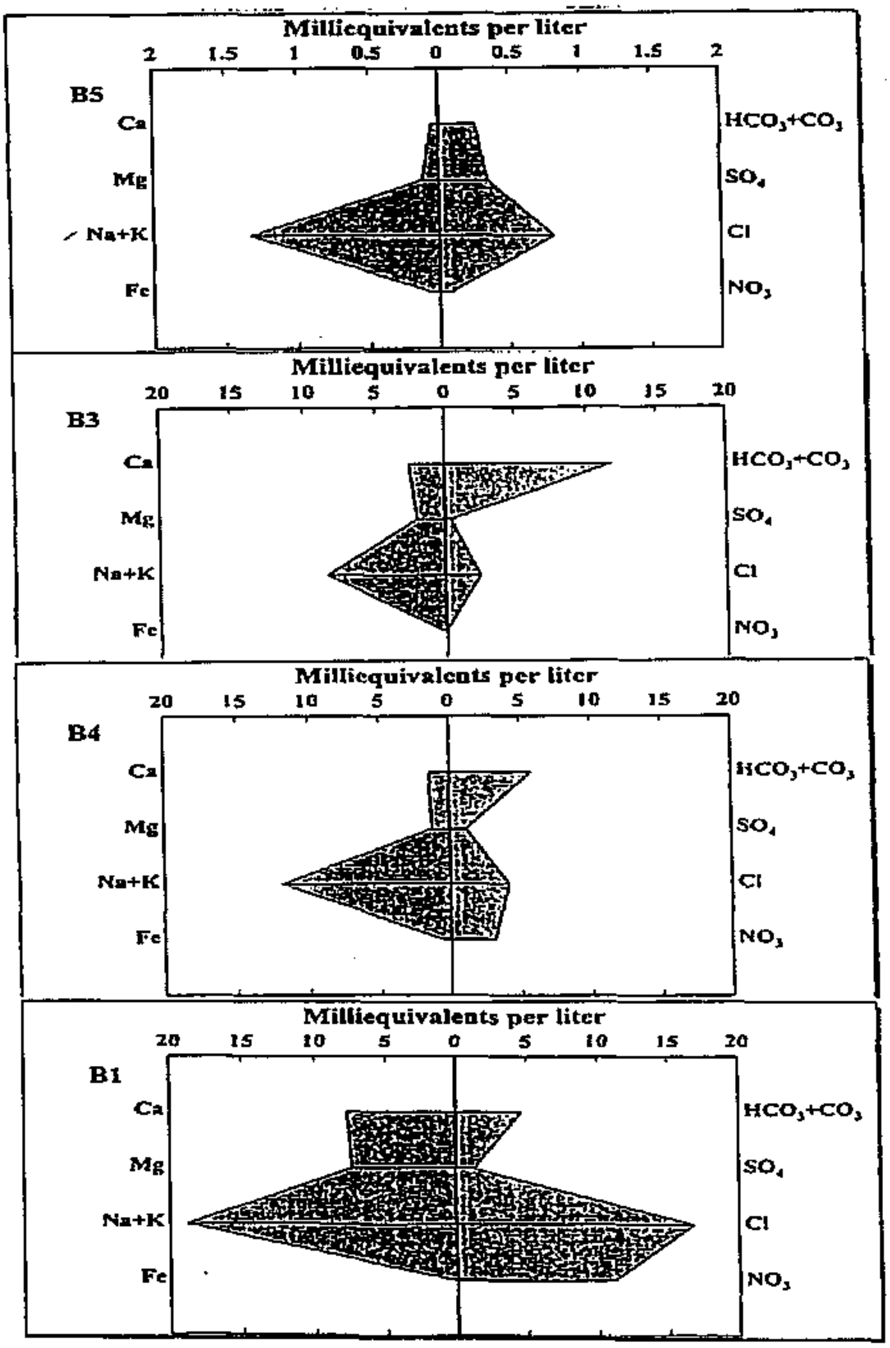


Figure 4.15 : Site B : Stiff diagrams of hydrochemistry from sampling sites 5, 3, 4 and 1.

Table 4.3: **Summary of hydrochemical and microbiological results**

SAMPLE ID:	**Bosch 1**	**Bosch 2**	**Bosch 3**	**Bosch 4**	**Bosch 5**	**Bosch 6**
Depth of borehole	2	1.2	1.2	0.5	0.8	0.5
Lithology at WT	Sand	Sandy clay	Sandy clay	Sand & boulders	Sand & boulders	Sand & boulders
Location	Gate/trough	Dam	EIA	EIA	Control	Manure/ channel
K mg.l^{-1}	259	31	161	126	2.5	221
NH4-N mg.l^{-1}	0.2	0.53	0.19	1.17	0.21	1.85
NOx-N mg.l^{-1}	156	<0.1	<0.1	44.9	1.23	<0.1
P mg.l^{-1}	<0.1	<0.1	<0.1	6.59	<0.1	3.68
C O D mg.l^{-1}	56	73	117	169	36	169
D O C mg.l^{-1}	10	17.6	36	30	3	55
E.C. mS/m	366	76	126	154	19	160
T D S (Calc) mg.l^{-1}	2342	486	806	986	122	1024
Faecal coliforms per 100ml	NA	5027	0	37500	47	340

WT - Water table EIA - Effluent irrigated area NA - Not analysed

Shaded values represent contaminated levels.

All the sampling sites targeted on the farm as being vulnerable to contamination indicate some hydrochemical impact of contaminant infiltration from animal waste sources. However, there is no positive correlation between the degree of impact on the different determinands (apart from COD and DOC which are always closely related).

The lack of correlation between the different contaminant indicators may be due to the following:

- the different composition of the contaminating waste in different areas
- different attenuation processes in operation in different areas due to the variable alluvial lithologies
- variable grass cover in the different areas resulting in different rates of nutrient uptake

The sample most contaminated with respect to nitrate and potassium is from beneath the manure covered area close to a gate and feeding trough in a pasture (B1). This area is underlain by permeable sand and has no grass cover which probably explains the high leaching rate of these nutrients.

The highest DOC and COD levels are seen in sample B6 taken close to a channel feeding flushed waste from the cow stalls near the dairy to the solid separator and close to the manure stockpile (on a concrete base). Effluent coming from the dairy and piggery would have higher levels of DOC and COD than that applied to pasture which has undergone some anaerobic decomposition in the lagoon. It is therefore likely that this contamination represents some leakage from this effluent channel. Concentrations of phosphate, potassium and faecal coliforms are also elevated in this sample.

Sample B2 was taken on the edge of an effluent irrigated area where cows graze the pasture. It is immediately down gradient of the effluent lagoon. It is contaminated with respect to potassium, DOC, COD and faecal coliforms, but generally to a lesser extent than that seen in the other effluent irrigated areas. It is not know whether the source of the contamination is from the irrigation activities, cows grazing or leakage from the dam.

Samples B3 and B4 are taken from effluent irrigated areas. The overall impact of contamination on groundwater quality is less at B3, with no nitrate, phosphate or faecal coliforms detected (although the level of potassium is 12% higher). The relative rates of effluent application are not recorded for the two different areas therefore this influence on the degree of contamination cannot be estimated. However, it is likely that the less permeable subsurface at B3, which has a proportion of clay, significantly contributes to attenuating the impact of contamination from effluent applied at surface.

4.3.7 Summary of impact of the IAH operations on groundwater quality

Analyses of all the groundwater samples indicated some degree of contamination from animal waste. The control sample, taken from an area of bush which is reportedly not formally used for agricultural purposes, contained faecal coliforms. This most likely to be the result of informal livestock grazing in the area. There was no apparent impact of the concentrations of hydrochemical determinands of this sample from animal waste.

The highest levels of **nitrate** and **potassium** contamination were found in groundwater underlying an area close to an intensively used trough and gate which has a high loading of manure and no grass cover.

The **greatest number** of determinands showing an impact and the highest levels of **faecal coliforms** were detected in groundwater from a shallow sandy aquifer with boulders beneath a pasture irrigated with effluent from the lagoon.

The highest values for **DOC** and **COD** were detected in groundwater sampled close to a constructed channel feeding flushed waste from the cow stalls near the dairy to the solid separator. Lower levels of impact were seen in samples from clayey sand aquifer layers. In particular, no nitrate was detected.

4.3.8 Synopsis

This case study investigates the impact of a piggery and dairy operation on groundwater quality in a shallow alluvial aquifer.

Waste is removed manually and by flushing from the IAH areas. After passing through a solid separator, solid waste is stockpiled before being used as an organic fertilizer and liquid waste is pumped to an effluent lagoon. Effluent is irrigated onto pasture which is grazed by the cows.

The site is underlain by a shallow, unconfined, alluvial aquifer which varies lithologically from clayey sand to sand and boulders. Groundwater samples were obtained from temporarily installed well-points in areas where contamination from different sources was likely and from an uncultivated control area.

Contamination was evident in all samples but varied in type and intensity. Groundwater from a sandy aquifer layer underlying effluent irrigated pasture appeared to be impacted by the widest range of contaminants. The presence of sandy clay beneath effluent irrigated pasture appeared to attenuate the impact on groundwater quality of nitrate and faecal coliforms. The highest nitrate and potassium contamination occurred beneath an ungrassed area with a heavy manure loading. The highest DOC and COD contamination occurred close to a constructed channel feeding flushed waste from the cow stalls to the solid separator.

4.3.9 Conclusions

Groundwater quality was seen to be impacted by the following activities at the site:

- irrigation of effluent to pasture
- congregation of cows near a feeding trough and gate in pasture resulting in high loadings of manure and destruction of grass cover
- leakage of flushed animal waste from constructed channel.

Infiltration of contaminants from the effluent lagoon may have occurred but this cannot be determined with reasonable certainty from the data available.

In all cases, improved management of waste and livestock behaviour at the site could mitigate groundwater contamination, however, the cost implications of improvements may make them infeasible. The following steps could be taken:

- more effective treatment of effluent in the lagoon prior to irrigation
- application of effluent at agronomic rates
- mobilise feeding troughs around the pasture to limit the manure loading in any one area
- construction of impermeable channels conducting liquid/ slurry animal waste.

The type and degree of contamination across the site is indicated by the data available to be very variable. This is likely to be due partly to variations in the composition of animal waste applied to the surface (manure, pre-lagoon flushed waste, post-lagoon effluent, etc). In addition, the range of alluvial lithologies encountered results in different attenuation processes occurring in different parts of the aquifer and variable hydraulic continuity with depth and laterally.

4.3.10 Recommendations for further investigations

It would have been useful to install permanent (or semi-permanent) well points so that samples could be obtained over a period of time to gain confirmation of the spatial variations in quality and an indication of temporal variation at the sampling points. This may have helped indicate periods of peak contamination which could then be related to agricultural practice.

A higher density of sampling points would give a better indication of spatial variation. More data may indicate a degree of correlation between different contaminant indicators.

A new effluent dam is being constructed at the site, which, it is hoped, will allow a greater degree of anaerobic degradation of the effluent prior to irrigation. It would be interesting to continue monitoring shallow groundwater quality and effluent quality to see if the new dam is effective.

4.4 SITE C

4.4.1 Site background

This case study examines the impact on local groundwater of a feedlot operation in Gauteng. The site was selected for the following reasons:

- the farmer was willing to cooperate fully with the study
- the site is underlain by a fractured aquifer(s) which was thought to be vulnerable to contamination
- groundwater is used on and around the farm and therefore many borehole sampling points were available
- the feedlot has been operating for over 20 years therefore if contamination of groundwater is likely there is a good chance of detecting it
- it is an example of generally good management practice where the farmer is environmentally aware and up to date with recommended procedures for minimising the impact of the operation.

4.4.2 The feedlot

The feedlot was established in 1973. It currently has a total capacity of 10 000 cattle but was operating at about 40% capacity during the period of the study. Cattle are kept at the feedlot for between 100 and 120 days and receive approximately 8.5 kg of feed and 35 litres of water per day.

The farm is dependent on groundwater and approximately 200 $m^3.d^{-1}$ are estimated to be abstracted from the 8 operating boreholes.

4.4.3 Land-use and potential groundwater contamination in the study area.

The feedpens cover an area of approximately 7 ha. The only artificial lining in the pens is a concrete apron around the feeding troughs. The pens are cleaned approximately 3 times in the dry winter months and often once during a drier period of the summer. They are cleaned by scraping with a bulldozer, and care is taken to remove only surficial manure and not to disturb the compacted interface layer which is well developed. Most of the manure is temporarily stored outside the pens in an access area before removal for sale. Some is used to construct manure mounds in the pens to allow an elevated area for the cattle to congregate away from the saturated surface during rain.

The pens are sited in a naturally sloping area with feeding troughs located at the up-gradient end of each pen. During summer rains, run-off from the pens drains into channels which gravity feed into 2 effluent lagoons down-slope of the pens. Following a period of settlement in the lagoons, an outflow pipe feeds effluent to the pasture area between the pens and the river.

The lagoons are emptied during the dry winter months by disposal to pasture and evaporation. Pasture is irrigated with effluent by flood and overhead methods. The bottom sludge layer is mechanically removed from the lagoons each winter and temporarily stored on nearby grassland before being mixed with manure for sale to farmers and application to maize crops.

Cattle carcasses are sold to a local zoo. However, they were previously buried on the farm.

Feed for the cattle is stored under cover on concrete lined areas.

Maize is cultivated in the area to the west of the feedpens. Manure from the feedpens is applied once every three years at a loading rate of 56 m^3. ha^{-1}, which is estimated to be equivalent to around 26 t dry mass.ha^{-1}. Inorganic LAN fertilizers are applied in this area in the years when manure is not applied at a rate of 150 kg.ha^{-1}.

Pit latrines are located around the site at labourers cottages. The offices and main farm house, on top of the ridge, have septic systems.

In summary, potential sources of groundwater contamination associated with the feedlot operation are thought to be:

i. Infiltration from the feedlot pens
ii. Infiltration from the effluent run-off channels
iii. Infiltration from the effluent dam
iv. Infiltration from pasture irrigated with effluent
v. Infiltration from areas of temporary manure and lagoon sludge storage.

Of these, the most likely source is thought to be the effluent lagoons. Any sealing layer that may form while they are full may be ruptured during removal of the sludge and shrinkage cracks are likely to form when they dry out in the winter allowing vertical migration of the heavily contaminated first flush run-off.

Other potential sources of groundwater contamination on the site are as follows:

vi. use of manure as fertilizer in the maize cultivated area
vii. use of inorganic fertilizer in the maize cultivated area
viii. seepage from pit latrines and septic tanks.

4.4.4 Hydrogeology of the site

Boreholes on the farm are reported to intercept approximately 30 m of aquifer thickness from 30 mbgl in fractured shale and quartzites. Abstraction rates are generally low at around 40 $m^3.d^{-1}$.

The farm is situated mainly on shales of the Pretoria Series. Timeball Hill Stage quartzites with alternating shale, siltstone and greywackes form the ridge in the centre of the farm. The feedlot pads are located on this ridge. Shales and siltstones of the Daspoort Stage dominate the lithology to the east of the ridge as the ground slopes down to the Kafferspruit River. The gradient between the ridge and the river is approximately 1 in 20. This land is set to pasture and periodically irrigated with effluent. To the west of the ridge, shale with breccia, conglomerate and quartzite crops out. Maize is cultivated here and the topography has a similar slope. Drilling records show that the area is underlain at between 10 and 70 mbgl by chert and dolomites.

Boreholes in the area are recorded to have been drilled to between 40 and 350 mbgl. Information from the National Groundwater Database shows that groundwater quality is generally good with levels of TDS recorded at between 100 and 200 $mg.l^{-1}$. Groundwater is widely used for domestic, irrigation and stock watering purposes.

Most groundwater in the area is thought to be abstracted from the dolomites which are recharged to the west of the area. The quartzites, shown to comprise the ridge underlying the feedpens, may also form an aquifer unit locally. If it is recharged in this area of outcrop it is vulnerable to contamination from the feedlot operation.

Drilling records show clayey soil covering much of the area to a variable depth averaging 15 mbgl (National Groundwater Database). This low permeability surficial cover, combined with the sloping topography of the site, is expected to result in high rates of run-off and limit infiltration.

It was not possible to determine relative RWLs at the sampling points as most were inaccessible to a water level metre and they had not been accurately surveyed. It is assumed that **groundwater flow** is down-dip at the site, ie. approximately eastwards towards the river. RWLs that were measured varied from 3 to 7 m below the top of the casing.

The annual **rainfall** in this area is approximately 700 mm and occurs mainly in the summer months (Midgley *et al.*, 1994). Potential **evaporation** is around 1700 mm p.a. (Midgley *et al.*, 1994).

4.4.5 Field investigation

The field investigation was carried out in May and August 1996 as follows:

- In May, 15 existing boreholes on and around the farm were sampled to give an indication of the groundwater quality at the site.
- In August electromagnetic (EM) geophysics was carried out to determine optimum sites for new boreholes in an area where risk of contamination was thought to be high (immediately down gradient of the effluent dam and lagoon sludge pile).

- Two new boreholes were drilled in August and groundwater samples taken. This gave more data about the nature of the subsurface at the site and additional groundwater samples at locations of high contamination risk.

The EM survey was carried out using horizontal dipoles (to maximise the sensitivity to vertical fissures and faults) and a 20 m spacing giving a penetration depth of approximately 30 m.

Figure 4.16 shows the layout of groundwater sampling points.

Existing boreholes at the site were drilled between 1975 and 1985 to approximately 60 mbgl. They are reported to be open in fractured hard rock (shales and quartzite) below approximately 30 mbgl.

The project dedicated boreholes were drilled to 60 mbgl and are open in fractured shale below 42 m.

4.4.6 Results of the field investigation

4.4.6.1 Drilling

Drilling was carried out using a down the hole hammer. Two locations were chosen down dip of the effluent lagoon and lagoon sludge stockpile. These locations are also down dip of the feedpens and could therefore detect contamination from all the potential sources associated with the feedlot operations.

The boreholes were drilled to 60 m. The sites are less than 150 m apart and the lithologies encountered were very similar. Drilling returns indicated the following rock types:

0 - 17 mbgl	Orange-brown clayey soil.
18 - 42 mbgl	Orange-brown clayey soil with some fine grained, brown-grey shale chips. Occasional Fe stained quartzite.
42 - 60 mbgl	Blue-grey shale chips.

Several moderate and minor water strikes were encountered between 47 and 55 mbgl giving a total blow yield of approximately 1m^3.hr^{-1}. The RWL was measured at approximately 7 mbgl in both boreholes therefore the groundwater intercepted appears to be confined and is not likely to be affected by contamination from the surface in the immediate area. It may still show signs of contamination if is in continuity with groundwater recharged up-dip in the feedpen area, but it is likely that there is a similarly thick, low permeability soil cover beneath the pens.

4.4.6.2 *Soil analyses*

Soil samples were taken from the base of the dry effluent lagoon for nitrate and ammonium analyses. Figure 4.16 shows the resulting soil nitrogen profile to 0.50 mbgl.

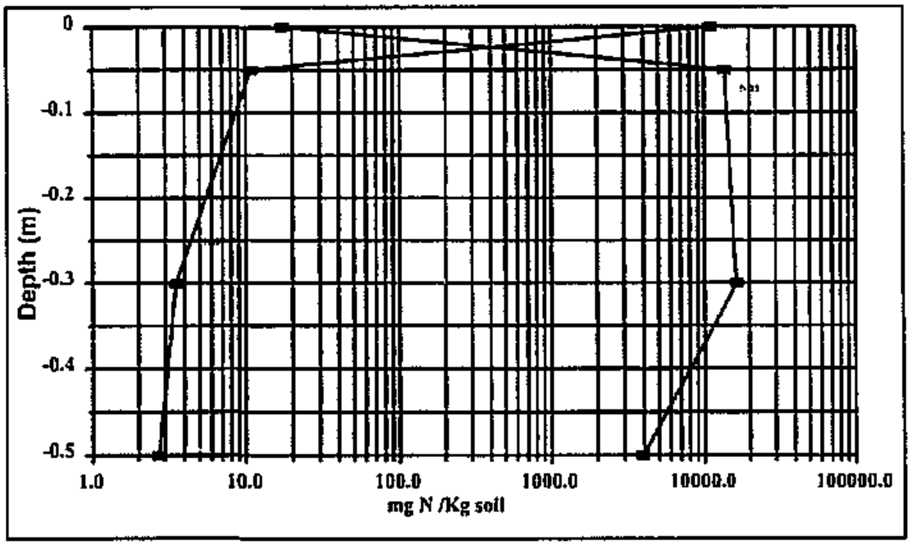


Figure 4.16 : Site C : Soil nitrogen profile beneath the effluent lagoon

The soil underlying the lagoon was found to be semi-saturated, dense clay with occasional angular blocks (2 to 30 cm) of iron stained quartzite. The clay will have very low bulk permeability of the order of 10^{-5} m.d^{-1}, but pathways for preferential migration may exist along the contacts with the angular clasts and, to a limited depth, in desiccation cracks.

The results obtained for nitrate and ammonium nitrogen concentrations are not easily explained. Nitrate levels are moderate in the surface dry layer at 17 mg N. kg^{-1} soil and increase rapidly to between 1 000 and 10 000 mg N. kg^{-1} soil with depth. Conversely, ammonium was detected at 1100 mg N. kg^{-1} soil at the surface and declines rapidly with depth to between 1 and 10 mg

N. kg^{-1} soil. The opposite relationship would be expected in this low permeability, damp environment. The data has been double checked and found to be correct but it would be useful to collect several other sets of samples from the lagoon base for confirmation.

Although there is confusion about the speciation of nitrogen with depth, it can be concluded (assuming the results to be correct) that there are high levels of nitrogen in the subsurface and these are not significantly attenuated within the depths sampled.

4.4.6.3 Description and discussion of hydrochemical data

The results of all hydrochemical analyses are presented in full in Appendix 1.D. The hydrochemical data is deemed to be of good quality as the charge balance error is within acceptable limits (<5%).

The **EC** of the groundwater sampled is generally low at less than 4 mS.m^{-1} (<40 mg.l^{-1} TDS) for most samples. Three samples have TDS levels in excess of 100 mg.l^{-1} but none is greater than 130 mg.l^{-1}. Two of the higher EC boreholes are at sampling points CBC5 and CB3, as shown on Figure 4.17. The third is at sampling point CBC 14, which is shown on Figure 4.16, approximately 1 km to the south west of the site.

There is no obvious spatial pattern to the variability of EC values. Sampling points which gave samples with TDS values above 100 mg.l^{-1} are not located close together in the study area. The relative sampling depths and aquifer lithologies intercepted at the boreholes are not known, except in the case of those drilled specifically for this project (Chalma 1 and Chalma 2). It may be that the boreholes with higher TDS intercept a different aquifer, for instance the dolomite rather than the shales and quartzites, or abstract from a different depth.

Figure 4.18 shows a **Piper plot** of the results of hydrochemical analysis. The samples are all of (Ca, Mg)CO_3 type. The higher TDS samples (plotted as B, C and 4 on the Piper) plot close together and have the highest cationic proportion of Ca and Mg. This may indicate that they are intercepting a dolomitic aquifer. Samples from boreholes drilled near to the effluent dam for this study and CB4 and CB6, also in the southern half of the farm, have a higher cationic proportion of Na and K. It is known from the drilling returns that Chalma 1 and 2 intercept a fractured aquifer in the shales and it is likely that CB 4 and 6 intercept similar aquifer lithology.

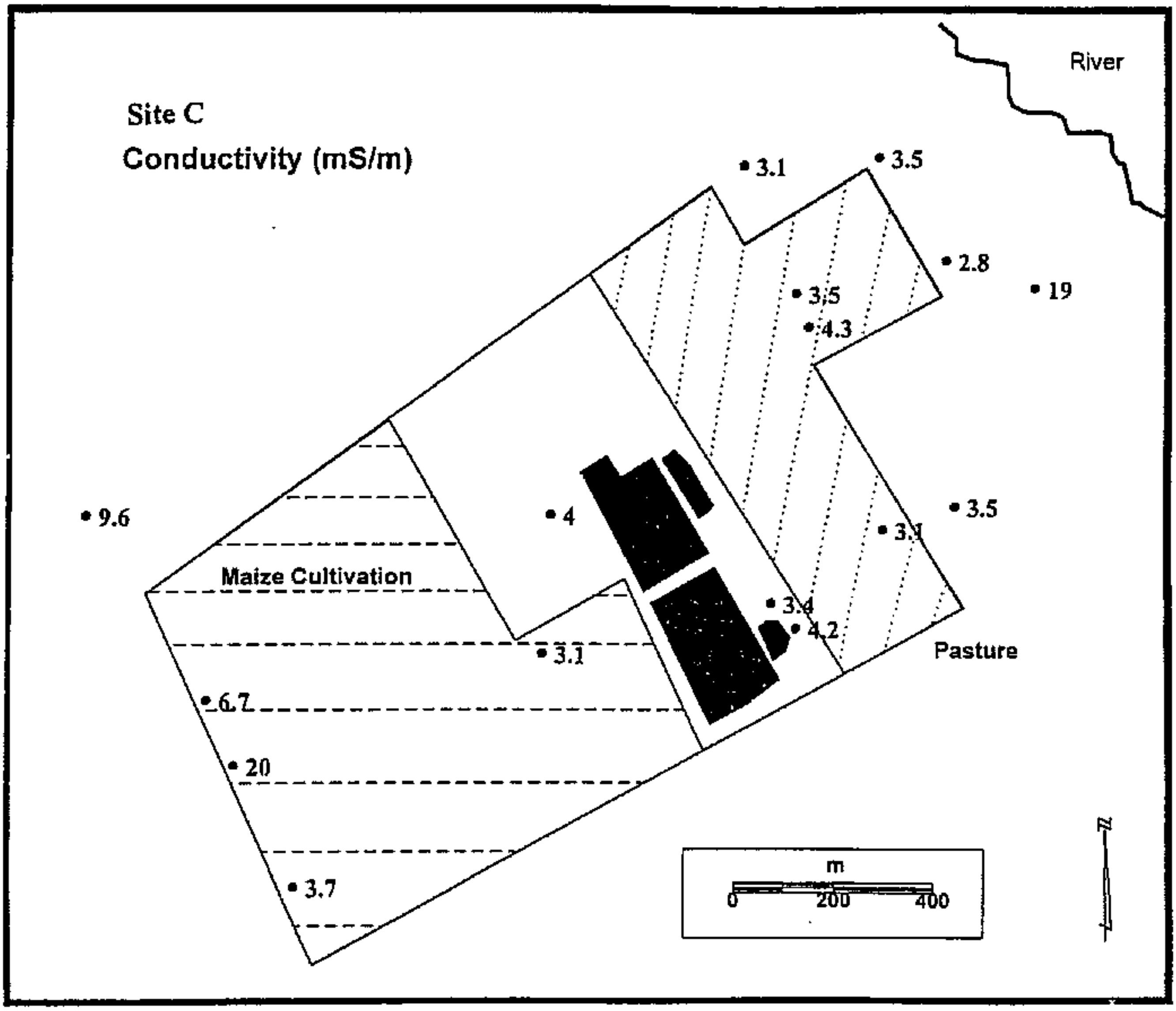


Figure 4.17 : Site C : Spatial distribution of conductivity values

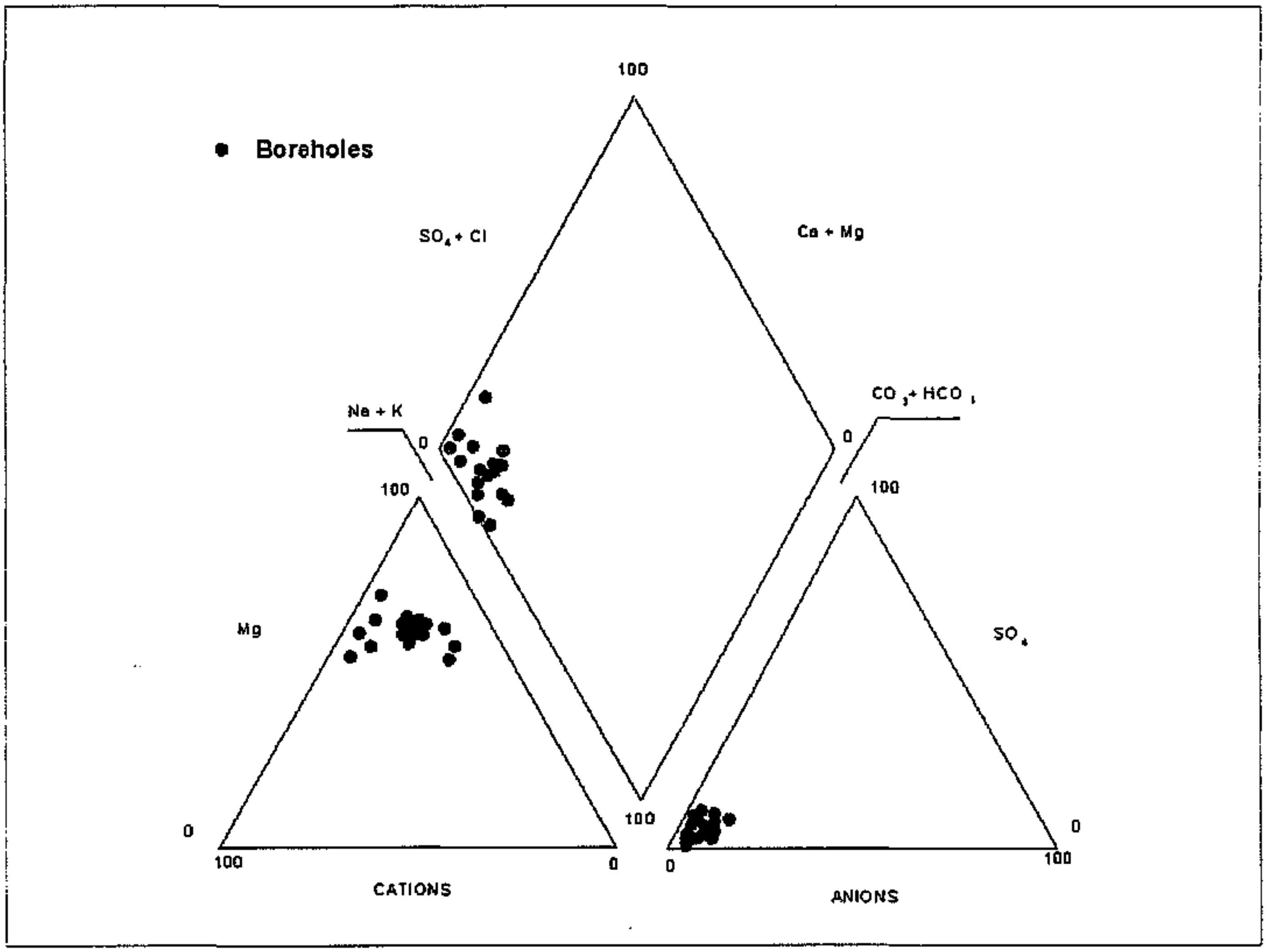


Figure 4.18 : Site C : Piper plot of hydrochemical results

The **stiff diagrams** presented in Figure 4.19 represent data from selected boreholes along a southwest-northeast cross-strike traverse of the site. The CB3 sample has a higher TDS of 128 $mg.l^{-1}$ (note the higher milliequivalent scale) and is comprised predominantly of Ca, Mg and HCO_3. The lower TDS boreholes have HCO_3 as their dominant anion but a more even spread of cationic strength between Ca, Mg and Na. None of the samples has significant NO_3. There is no obvious evolution of groundwater hydrochemistry down-dip along this traverse.

Nitrate-nitrogen values are generally low, averaging 0.15 $mg.l^{-1}$, with no nitrate detected in 2 samples at the southwest edge of the site. The highest value was 1.7 $mg.l^{-1}$ in the sample from CBC5, down-dip of the site, close to the river. The spatial distribution of the nitrate results, as shown in Figure 4.20, does not appear to vary as a result of land use at the site. There is no significant correlation of nitrate-nitrogen values with EC. These low values indicate that only insignificant nitrate is present in the groundwater and they do not indicate contamination.

Ammonium was detected in only one sample, from CB2, at the very low concentration of 0.13 $mg.l^{-1}$.

Other potential contaminants analysed for were either not detected in any samples, such as **dissolved organic carbon** and **orthophosphate**, or were detected at insignificant levels, such as **potassium**.

4.4.7 Summary of impact of the feedlot operations on groundwater quality

There is no evidence of contamination of groundwater from the feedlot operation. This is thought to be due to a combination of hydrogeological and management factors. The initial design of the feedpens on sloping ground with run-off channels and careful maintenance of the interface layer has contributed to limiting contaminant infiltration. The potential contaminant of most concern, nitrogen, is thought to be removed from the site in organic form in the manures and to volatilize from the feedlot surface during winter when conditions are relatively dry. Therefore the potential nitrogen pool is reduced at the same time as nitrification is limited by maintaining anaerobic conditions in at the base of the feedlots.

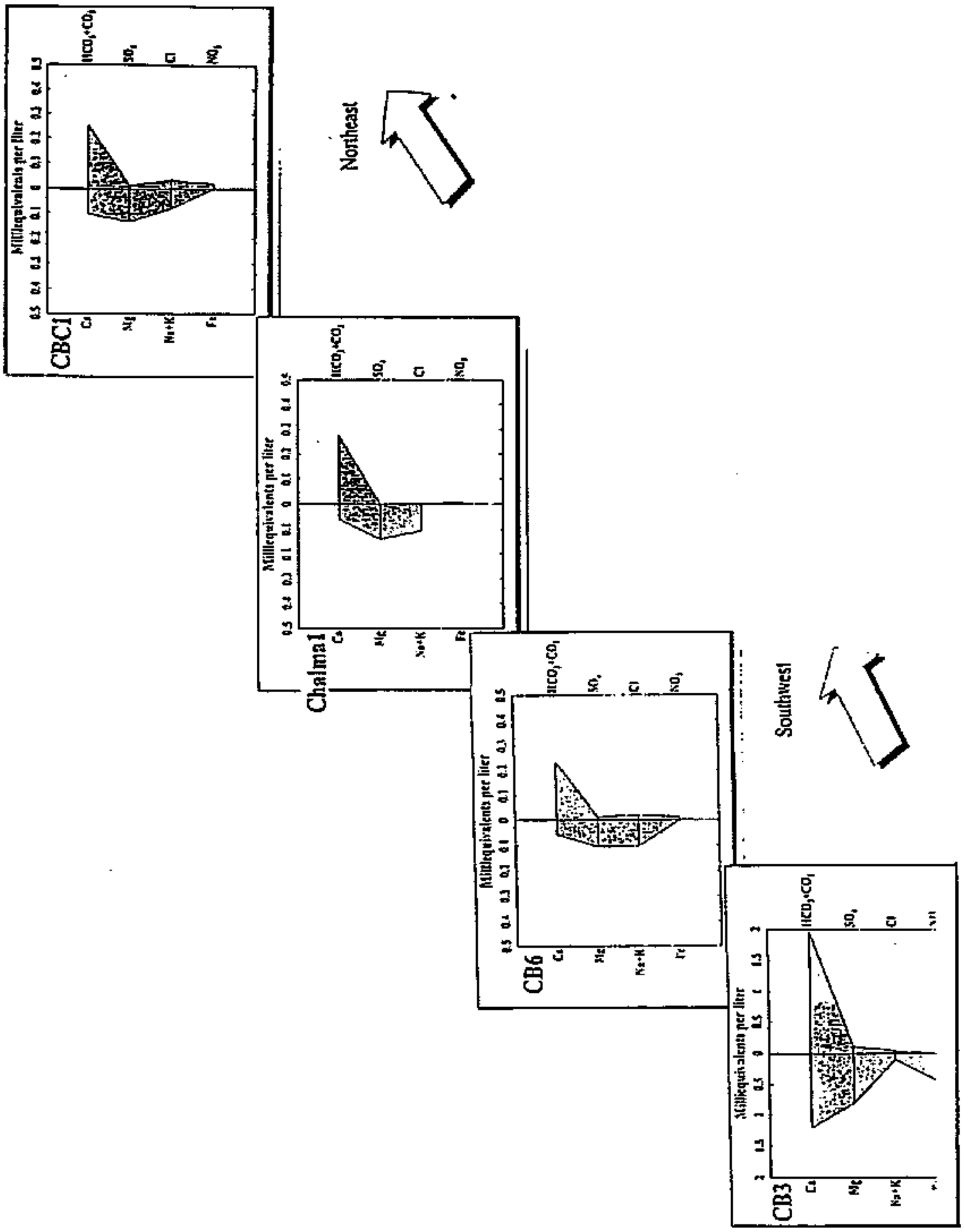


Figure 4.19 : Site C : Stiff diagrams of hydrochemistry of selected samples along a southwest - northeast traverse of the site

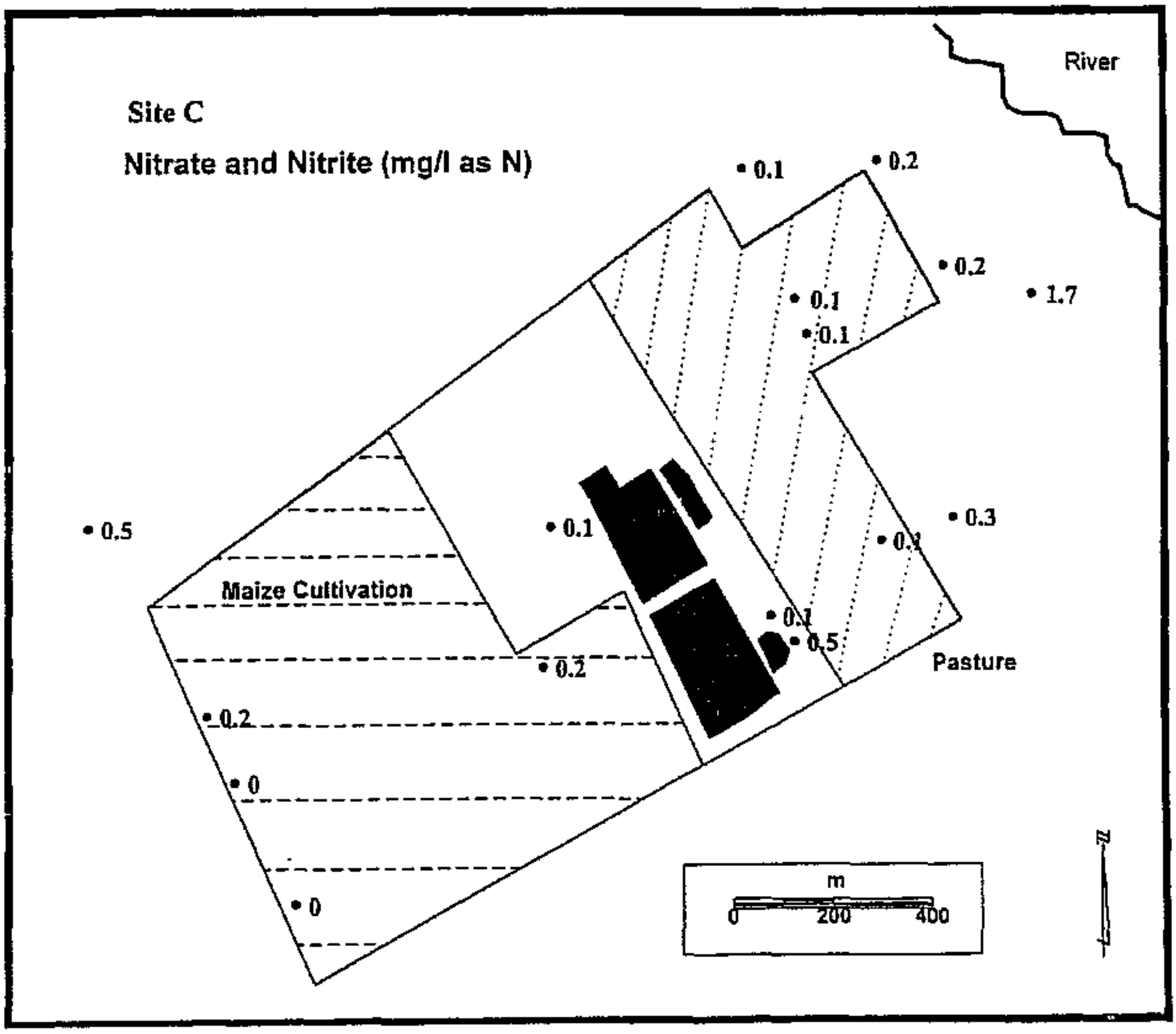


Figure 4.20 : Site C :Spatial distribution of nitrate-nitrogen values

The naturally low permeability and thickness of the unsaturated zone at the site means that potential contaminants infiltrating from the effluent dams, the effluent irrigated pasture and beneath sludge and manure stockpiles, are attenuated before they reach groundwater.

Nitrogen is present at significantly elevated concentrations in the surficial unsaturated zone beneath the effluent lagoon. This is likely to result from the following factors:

- very high loadings of organic nitrogen in the lagoon effluent and sludge
- periodic drying out of the lagoon destroying any sealing layer that may develop and allowing nitrification to occur in aerobic conditions
- the presence of preferential pathways for vertical contaminant migration along quartzite clast boundaries and desiccation cracks (despite the very low bulk permeability of the clayey soil).

The total depth of unsaturated zone contamination is not known. It should be noted that this source is likely to have resulted in groundwater contamination if the water table were shallow and unconfined.

4.4.8 Synopsis

The feedlot has been established at the site for over 20 years. It currently has approximately 4000 cattle, but has capacity for 10 000. The operation is designed and managed to minimize environmental impact.

Potential sources for groundwater contamination associated with the feedlot were identified at the feedpens, effluent drainage system, effluent lagoons, effluent irrigated pasture and manure and sludge stockpiles.

The aquifer underlying the formation is thought to occur mainly in fractured shales and quartzites of the Pretoria Series. Some boreholes with slightly higher TDS groundwater may intercept a dolomitic aquifer.

Drilling showed groundwater at the site to be confined beneath approximately 20 m of clayey soil. Groundwater was intercepted in low to moderate transmissivity fractures in shale.

Soil profiles beneath the effluent lagoon showed high levels of nitrogen contamination in the unsaturated zone to at least 0.5 mbgl despite the low bulk permeability nature of the subsurface. However, the unsaturated zone is apparently thick enough to attenuate any contaminants from the feedlot operation before they reach groundwater.

There was no evidence of groundwater contamination as a result of the feedlot operations. The groundwater quality was generally very good with low levels of TDS.

4.4.9 Conclusions

Nitrogen (principally nitrate) is the contaminant of greatest potential concern at this site due to the high loadings of animal waste concentrated in a limited area. There is no evidence of groundwater contamination. This is the result of the relatively invulnerable hydrogeological environment and careful management of the waste at the site. The main factors controlling the mitigation of groundwater contamination are as follows:

- low permeability, thick (> 20 m) unsaturated zone
- confined (and possibly semi-confined) groundwater at the site
- removal of manure 3 to 4 times a year from the feedpens
- efficient channelling of effluent to lagoons and irrigation over a wide area
- careful preservation of the interface layer in the feedpens
- high losses of nitrogen by volatilization in the feedpens which are situated on an exposed ridge, particularly in dry periods.

4.4.10 Recommendations for further investigations

It would be interesting to compare the relative depths of nitrogen infiltration in the unsaturated zone in different areas of potential contamination. These could include soil nitrogen profiles in the pasture irrigated with effluent, below the feed pens, beneath sludge and manure piles and to a greater depth in the base of the effluent lagoons.

4.5 SUMMARY OF THE FIELD INVESTIGATIONS

The findings of the field investigations are summarised in the table below.

Summary of findings of the specialist field study on the impact of sludge application to agricultural land on groundwater quality.

	Site A	**Site B**	**Site C**
IAH activities	Dairy (1200 cows) Covered stalls and pasture.	Dairy (800 cows) Piggery (6000 pigs) Covered stalls and pasture.	Feedlot (4000 cattle) Open pens
Waste management practices	Irregular flushing of stalls & manual solids clearance. Solid separation (dysfunctional), manure stockpiling, effluent lagoons.	Regular flushing & manual solids clearance. Solid separation, manure stockpiling, effluent lagoon, effluent irrigation.	Regular scraping of pens, run-off capture, effluent lagoons, effluent irrigation.
Field data collected as part of this project	Groundwater hydrochem. from est. boreholes & spring. Soil nitrogen profiles	Shallow groundwater hydrochem from temp well points. Soil nitrogen profiles	Groundwater hydrochem from est. & new boreholes. Drilling data Soil nitrogen profiles
Climate	500 mm p.a winter rain	900 mm p.a winter rain	700 mm p.a summer rain
Unsaturated zone	> 2 m thick, sandy soil with clay layers	<1.5 m thick, sandy alluvium.	>30 m thick, clayey.
Aquifer type	Surficial weathered zone of granite, possibly fractures.	Unconsol. alluvium.	Fractured shales, possibly some quartzites and dolomite.
Hydrochemistry	NaCl type TDS 600 - 1500 $mg.l^{-1}$	Variable, NaCl type TDS 100 - 2000 $mg.l^{-1}$.	CaCO3 type TDS < 40 $mg.l^{-1}$
Potential contamination from non-IAH sources.	Fertilizers and pesticides applied to wheat and vines. Pit latrines.	Effluent from food processing. Fertilizers and pesticides applied to orchards and vines. Pit latrines.	Fertilizers and pesticides applied to maize. Pit latrines.

	Site A	Site B	Site C
Probable IAH groundwater contamination:			
Nitrate	Yes - to a max of 27 mg.l^{-1} assoc. with the dairy operations (?). Also assoc. with waste slurry flooding.	Yes - to max of 156 mg.l^{-1} assoc. with cow conc. at feed trough in pasture. Also in effluent irrigated pasture.	No
DOC	No	Yes - to max of 55 mg.l^{-1} next to waste channel. Also in effluent irrigated pasture and at feed trough in pasture.	No
Potassium	No	Yes - to max of 259 mg.l^{-1} at feed trough in pasture. Also in effluent irrigated pasture and next to waste channel	No
Ortho-phosphate	No	Yes - to max of 6 mg.l^{-1} in effluent irrigated pasture. Also next to waste channel.	No
Microbiological indicators	Yes - to a max of 2 faecal Strep. per 100 ml assoc. with the dairy operations (?)	Yes - to max of 37500 faecal coliform per 100ml in effluent irrigated pasture. Also close to waste channel.	ND
Impact of IAH activities on groundwater quality.	Significant w.r.t nitrate and microbiological indicators, localised.	Significant w.r.t a wide range of determinands, variable, apparently localised.	None

(?) Further research required to confirm.
ND Not determined

The case studies therefore show an impact on the following aspects of groundwater quality as a result of IAH operations: nitrate, DOC, ortho-phosphate, potassium, microbial indicators of faecal pollution.

The most important factor influencing the degree of contamination at the sites studied appears to be the natural hydrogeological controls on groundwater vulnerability.

No contamination of groundwater was evident at site C, despite this site having the highest density of livestock (4000 cattle in feedpens covering 7 hectares) and the fact that it has waste management practices in common with other sites (effluent lagoons, effluent irrigation to pasture, etc.). The groundwater in this study area is naturally protected by:

- the topographic gradient
- the low bulk permeability of the clay rich unsaturated zone
- the thickness of the unsaturated zone

It is also protected by the following good management practices:

- siting of feedpens on a exposed slope (maximising run-off and volatilization and minimising infiltration)
- good run-off control
- careful maintenance of an interface layer in the feedpens

In contrast, site B has experienced a wide range of significant groundwater contamination. The groundwater at this site is vulnerable due to the following hydrogeological conditions:

- very shallow (<1.5m) depth to the water table
- high permeability sandy unsaturated zone

In addition, the following management practices may have increased the levels of contamination experienced:

- allowing livestock to repeatedly concentrate in pasture areas around a feed trough until it was denuded of grass and thickly covered in an excrement layer
- application of very poor quality effluent to pasture
- over- application of effluent determined by disposal needs rather than acceptable agronomic rates.

The hydrogeology of site A is poorly understood due to insufficient data, but ineffective waste management at this site, particularly around the dairy, and failure to maintain efficient operation of the solid separator have resulted in contamination of the unsaturated zone and groundwater.

Over-application of effluent in irrigated areas may result in diffuse contamination of vulnerable groundwater. However, most of the sources of groundwater contamination identified in this study are point sources: over-used areas around feed troughs, effluent lagoons, leakage from waste channels, areas of waste spillage/flooding. The contamination detected appeared to be fairly limited in extent. More detailed investigation is required to confirm this.

4.6 RECOMMENDATIONS FOR THE NON-TECHNICAL BOOKLET

One of the objectives of this specialist study is to gather data and information which can be incorporated into a non-technical booklet to inform farmers on how they can avoid contaminating groundwater. The most important factor controlling the degree of contamination of groundwater appears to be the natural vulnerability of the aquifer at the site. It is therefore of prime importance to educate farmers to recognise the on-site factors that influence aquifer vulnerability so that they can minimise the impact of their activities. The booklet should provide an introduction to hydrogeological principles and draw attention to the value of a good quality groundwater resource.

There should be some further contact with IAH organisations worldwide to make sure that the researchers have all the available information and guidelines on good environmental IAH practice. The case studies provide examples of good and bad management practice at IAH operations in South Africa, both with evidence of groundwater contamination and maintenance of quality.

Good management practices illustrated by the study include:

- run-off control
- maintenance of a feedpen interface layer
- storage of solid waste on constructed areas
- siting of feedpens on a slope with areas of cattle congregation around the feed troughs up-gradient

Poor practice seen as part of this study mainly concerns management of animal waste. The following activities have been shown to be associated with groundwater contamination:

- irrigation of effluent onto pasture
- concentration of livestock into limited areas of pasture intermittently (as opposed to permanent high densities in feedpens which results in the formation of an interface layer)
- leakage from constructed effluent channels
- leakage (to the unsaturated zone) from effluent lagoons where the self-sealing layer is removed.

In summary, the non-technical booklet should aim to increase farmers' awareness of groundwater contamination risks. It should adhere to the principles of integrated pollution control and make practical recommendations for best agricultural management practice.

CHAPTER 5

FERTILIZER APPLICATION TO AGRICULTURAL LAND

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EXECUTIVE SUMMARY

The literature study carried out as part of this project identified the application of inorganic commercial fertilizers, and particularly nitrogen fertilizer, to agricultural land as having significant potential to contaminate groundwater (CSIR, 1995). Limited information regarding the impact in South Africa was found, therefore it was decided to conduct a specialist study investigating the impact of agricultural fertilizer application on groundwater quality at study sites throughout the country.

The objectives of this specialist study are:

- to collect field data from 3 study sites to determine whether groundwater contamination has occurred as a result of the surface application of inorganic fertilizers in South Africa
- to collect data and information which can be used to support the non-technical booklet.

Commercial fertilizers have become an effective means of maintaining plant growth, replenishing soil nutrients and ensuring optimum crop yields for farmers. Continuous use of agricultural land for crop production strips the soil of essential plant nutrients such as nitrogen (N), phosphorous (P) and potassium (K) which are most conveniently replaced by the application of commercial inorganic fertilizers. Presently around two million tons of commercial inorganic plant fertilizer are sold annually in South Africa

The findings of the field investigations are summarised in the table below.

Summary of findings of the specialist field study on the impact of fertilizer application to agricultural land on groundwater quality.

	Site A	Site B	Site C
Farming Activity	Export Table Grapes	Potatoes	Wheat
Loading	400 kg.ha^{-1}.a^{-1}	240 kg.ha^{-1}.a^{-1}	100 kg.ha^{-1}.a^{-1}
Irrigation	Yes	Yes	No
Use of organic fertilizers in the area	Yes	Yes	No
Use of inorganic fertilizers in the area	Yes 1:0:1 and LAN	Yes 1:0:1 or 3:0:5	Yes 3:2:0
Field data collected as part of this project	Groundwater levels, Groundwater hydrochemistry, from shallow wells and boreholes	Groundwater levels, Groundwater hydrochemistry, from shallow boreholes	Groundwater hydrochemistry, from boreholes
Climate	529 mm (1996), winter rain	350 mm(1996) winter rain	843 mm (1996) summer rain
Unsaturated zone	2 - 3 m, alluvial soils	2 - 3 m, sandy soils	5 m, clay rich soils
Aquifer Type	Unconfined	Unconfined	Confined
Hydrochemistry	Ca, Mg, Cl and SO_4	Na, Cl	Ca, Mg, CO_3, HCO_3
Potential groundwater contamination from other sources	Pit latrines and septic systems	Pit latrines and septic systems	None
Groundwater contamination:			
Nitrate	Yes 0 to 13.2 mg.1^{-1}	Yes 0 to 23.7 mg.1^{-1}	Yes 6.9 to 20.4 mg.1^{-1}
$\delta^{15}N$	+6 to +16 ‰	+3 to +12 ‰	+6 to +8 ‰
Phosphorous	0 mg.1^{-1}	0 mg.1^{-1}	0 mg.1^{-1}
Potassium	Low 0.2 to 18 mg.1^{-1}	Low to med. 1.3 to 121 mg.1^{-1}	Low 3.2 to 7.1 mg.1^{-1}
EC	Low to med. 2 to 140 mS.m^{-1}	Low to high, 31 to 840 mS.m^{-1}	Med. 61 to 83 mS.m^{-1}
Impact of fertilizers on nitrate concentrations in groundwater	Very low, although higher in localised areas. The % that fertilizer contributes to total nitrate concentration undetermined.	Very low. The % that fertilizer contributes to total nitrate concentration undetermined.	Low. The % that fertilizer contributes to total nitrate concentration undetermined.

All the case study areas show elevated nitrate levels, however, the impact from inorganic fertilizers is indicated by isotope analyses to be minimal in all cases. At site A, 60% of groundwater sampling points showed nitrate levels above 5 mg.l^{-1} and fluctuations in nitrate concentrations with time corresponded with the application of fertilizer. However, the range of $\delta^{15}N$ values and a positive correlation of increasing nitrate concentration to increasing $\delta^{15}N$ suggest that organic fertilizers contributed more significantly to elevated nitrate levels. This assumes that the ranges for $\delta^{15}N$ values correspond to nitrogen sources as outlined in the introduction.

At site B, there was no corresponding increase in nitrate concentrations with the application of fertilizer at surface. The isotope analyses indicated that the nitrate concentrations in groundwater are mainly attributable to natural soil sources with some possible influence from animal waste.

Historical data available for one monitoring point at site C showed a nitrate increase of 5 mg.l^{-1} over a 20 year period. Once again, however, although it seems likely that agricultural activities in the area have affected groundwater quality, the contribution from inorganic fertilizers appears to be minimal. This area has some of the highest nitrate levels but the lowest fertilizer application rate. The range for $\delta^{15}N$ values indicates a natural soil source (this is the one study area where organic fertilizers are not reportedly used). Ploughing practices typical of wheat farming, which leave bare ground in the area exposed during the rains, are thought to be the cause of high nitrate levels in groundwater from this source.

5.1 INTRODUCTION

5.1.1 Motivation

The literature study carried out as part of this project identified the application of inorganic commercial fertilizers, and particularly nitrogen fertilizer, to agricultural land as having significant potential to contaminate groundwater (CSIR, 1995). Limited information regarding the impact in South Africa was found, therefore it was decided to conduct a specialist study investigating the impact of agricultural fertilizer application on groundwater quality at 3 study sites.

5.1.2 Commercial Inorganic Fertilizers

Commercial fertilizers have become for farmers an effective means of maintaining plant growth, replenishing soil nutrients and ensuring optimum crop yields. Continuous use of agricultural land for crop production strips the soil of essential plant nutrients such as nitrogen (N), phosphorous (P) and potassium (K) which are most conveniently replaced by the application of commercial inorganic fertilizers. Presently around two million tons of commercial inorganic plant fertilizer is sold annually in South Africa (H. Venter, Fertilizer Society of S.A., pers. comm. 1997).

Some of the usual components of commercial fertilizers worldwide are listed in Table 5.1. Nitrogen is the most common and widely used fertilizer nutrient (Follet and Walker, 1989).

Table 5.1 : Common components of commercial fertilizers worldwide.

Nitrogenous Fertilizers	Phosphate Fertilizers	Potash Fertilizers
Ammonia (anhydrous)	Ammonium Phosphate	Natural potassic salts
Ammonium Nitrate	Ammonium Polyphosphates	Muriate
Ammonium Phosphate	Basic slag	Potassium sulphate
Ammonium Polyphosphates	Diammonium phosphate	
Ammonium Sulphate	Ground Rock Phosphates	
Ammonium Sulphate Nitrate	Superphosphate	
Calcium Ammonium Nitrate	Triple superphosphate	
Calcium Cyanamide		
Calcium Nitrate		
Diammonium phosphate		
Sodium Nitrate		
Urea		
Urea-formaldehyde		

Sources: FAO (1997), Lee (1991), Peterson and Frye (1989).

Although conclusive scientific evidence is lacking for South Africa, it is considered highly probable that fertilizer application to land constitutes the most widespread diffuse source of nitrate pollution to groundwater as it was identified in other parts of the world (Tredoux, 1993).

5.1.2.1 Application of commercial fertilizers to agricultural land

Benefits

Crops require adequate amounts of plant nutrients to produce optimum yields. Under natural conditions, the nutrients removed from fertile soil by plants are replaced by the **death and decay** of plants and animals, **biological fixing** of atmospheric nitrogen by bacteria and weathering of rocks as part of a **biogeochemical** cycle. However, prolonged and intensive use of land for cultivation disrupts the cycle by removing essential plant nutrients faster than they can naturally be replaced. Besides carbon, hydrogen and oxygen which are usually in excess and can be drawn by the plant from the atmosphere, soil and water; **nitrogen, phosphorus** and **potassium** are the three major elements that are needed for plant growth.

Plant fertilizers normally contain (Lee, 1991):

- *Nitrogen*, as ammonium salts, nitrate salts or urea. Nitrogen is essential for plant growth, particularly of leaves as it is a constituent of amino acids and proteins, which must be made to make new cells.
- *Phosphorus* for root growth. Usually a slightly soluble from of phosphate such as superphosphate or triple superphosphate is applied.
- *Potassium* ions for flowering, often provided as potassium sulphate.

Thus, if correctly applied, commercial fertilizers provide a convenient method of restoring the soil nutrient balance.

There is a large amount of nitrogen in the atmosphere, but in its stable gas form, N_2, it is unavailable to plants. **Industrial fixing** of this nitrogen as ammonia or calcium cyanamide provides important feedstock for the fertilizer industry. Ammonia is used to manufacture fertilizers which contain combined forms of nitrogen, mainly nitrates or ammonium salts which are then easily absorbed by the roots of plants.

A major advantage of inorganic fertilizers is that they can be formulated to provide a balanced supply of the major nutrients, readily available to the plants at a time and in quantities best suited to their requirements. They are relatively easy to transport and can be stored without risk to the environment (IFA - EFMA, 1990).

Hazards

Although the use of fertilizers without question contributes largely to increased food production, some of the distinctive and often serious problems and dangers involved are **soil acidification, soil structure degradation** and **water pollution** (Du Plessis, 1984).

Superphosphate contains the acid salt $Ca(H_2PO_4)$ which is one of the more soluble forms of phosphate and so dissolves in the soil water. Superphosphate has been blamed for the acidification of soils, although this is disputed by McLaughlin (1990) for Australian soils and is possibly incorrect for other areas.

In certain horticultural areas, intensive use of ammonium based fertilizers has resulted in local acidification of soils. The acidification effect of product removal, natural leaching and the use of ammonium based fertilizers has been recognised worldwide for many years (McLaughlin, 1990). This is caused by the hydrogen ions that are released during the nitrification of ammonium to nitrates. As the acidity of the soil increases, aluminium in the soil becomes more freely available for plants to absorb, causing problems with root development and the uptake of other nutrients (Fugle and Rabie, 1994). Corrective applications of lime are often applied to alleviate the problem.

The widespread use of inorganic fertilizers in cultivated areas has resulted in a gradual decrease in the organic content of the soil. In warm countries such as South Africa this is accelerated by the rapid rates of microbial decomposition of organic matter (Korentajer, 1991). The deterioration of the physical properties of soil and irreplaceable loss due to erosion are pressing environmental problems.

Other negative environmental side-effects of fertilizer application usually relate to non-productive losses of plant nutrients from the soil (McLaughlin, 1990). These are:

- *Nitrogen*: leaching of nitrates, production of ammonia and nitrous oxides by volatilisation and denitrification, soil acidification.
- *Phosphorus*: leaching and run-off of soluble P.

These nutrients if leached to groundwater or transported to surface water bodies in run-off, may occur at toxic levels or result in eutrophication.

Enrichment of a surface water body in these plant nutrients can lead to enhanced plant growth (algae and macrophytes). When decaying, this plant material causes the depletion of the oxygen reserves of the water body, along with secondary problems such as the release of corrosive gases and other undesirable substances. Such **eutrophication** is often a sign of phosphorus contamination as a result of runoff from agricultural land, although industrial effluents, detergents and sewage are also sources. **Nitrate-induced eutrophication** has caused extensive algal blooms in coastal waters in the Baltic sea, with consequent deoxygenation of the water and loss of fishing grounds. The suspected cause of this pollution is the seepage of high nitrate groundwater into the sea from the agricultural areas of Germany and Denmark (Croll, 1994).

Over application of fertilizer nitrogen by farmers poses a threat to surface water and groundwater quality. Providing crops with too much nitrogen, or providing the nutrient at a time when the crop is unable to take up all the nitrogen supplied, may result in accumulation of the mineral forms of N (ammonium and nitrate) within the crop root zone. Excessive amounts of accumulated nitrate may then leach out of the soil and eventually contaminate groundwater and surface water supplies (Follet and Walker, 1989).

Most of the incidences of nitrate pollution in drinking water sources are not solely attributable to fertilizer application, but this practice is often a major contributor, and many fertilizer manufacturers in the U.S. and Europe have received poor publicity in this respect in recent years.

Injudicious fertilizer application can even have detrimental effects on the crops themselves. Excessive nitrogen fertilisation can result in excessive vegetative growth which manifests itself as delayed maturity and loss of yield and quality of fruit crops and lodging difficulties with the harvesting of grain crops (Tredoux and Du Plessis, 1993).

Another hazard is related to impurities in the fertilizers themselves and not to fertilizer application. Impurities in fertilizers reflect the composition of the raw materials used, particularly in the case of phosphatic fertilizers made from natural phosphate rock and include cadmium, arsenic, silicon, uranium, chromium, lead, etc. usually in minute proportions. Of these impurities, it is cadmium that has received the most attention (McLaughlin, 1990).

Cadmium, particularly in phosphate fertilizers can accumulate, particularly near the point of addition in soils as it is relatively immobile and there are small losses due to leaching. This could lead to undesirable concentrations of cadmium in plants in the long term. Limits on the allowed concentration of cadmium in fertilizers have been imposed in some places.

Application rates and methods

The amount of fertilizer needed is a function of the type of crop, the season of growth, nutrient levels in the root zone, bacterial activity and the amount of nutrient in the irrigation water. Effective management of nitrogen presents a greater challenge to the farm operator than does that of any other fertilizer nutrient.

Various factors affect the amount, rate and method by which farmers apply fertilizer to crop land and the choice of fertilizer. These include tradition, level of education, and economic considerations such as crop price, fertilizer price and application costs.

In their paper on fertilizer nitrogen management (1989), Peterson and Frye set out a recommended Best Management Practice (BMP) to establish the amount of fertilizer required so that the difference between applied and harvested N will be minimal. The steps involved are:

- Establishment of realistic yield goals based on previous yield records.
- Soil tests to determine the natural capacity of the soil to supply required nutrients.
- Proper soil sampling techniques, including sampling at the correct depth and time of year.
- Tissue tests of N in plants during the growing season, so that application levels can be adjusted accordingly.

Fertilizers can be applied in a number of ways and at various times of the year, the method and timing playing an important part in fertilizer efficiency.

Table 5.2 shows the commonly used methods of application and placement of N fertilizers for various crop stages.

Table 5.2 : **Application and placement methods of N fertilizers for various crop stages.**

Crop Stage	Method of application and placement
Preplant	Broadcast on surface Broadcast, mixed into soil Injection, subsurface
At-plant or preemergence	In-row with seed Banded beside seed Injection subsurface away from seed Broadcast on surface
Postemergence	Side-dress, surface banded Side-dress, injected subsurface Top-dress (broadcast), surface Foliar, sprayed on leaves of crop

Source: Peterson and Frye (1989).

In some cases fertilizers are dissolved and sprayed with irrigation water is a process known as fertigation.

Generally fertilization is more efficient when the application is timed to coincide with the crop's demand for nutrients. Timing of the application of at least most of the N fertilizer to coincide with the beginning of the high N demand period minimizes the N loss effects of leaching, denitrification, and immobilization. The nutrient demand period will be specific to the crop type and climatic factors. A U.S. study, for example, found uptake of nitrogen by maize plants to be low for the first month after planting with a rapid increase between days 31 and 46. High N demand then continued for the next 2 months (Peterson and Frye, 1989).

5.1.2.2 Commercial fertilizer use in South Africa

Fertilizer consumption trends in South Africa since the 1960s are shown on the graph in Figure 5.1.

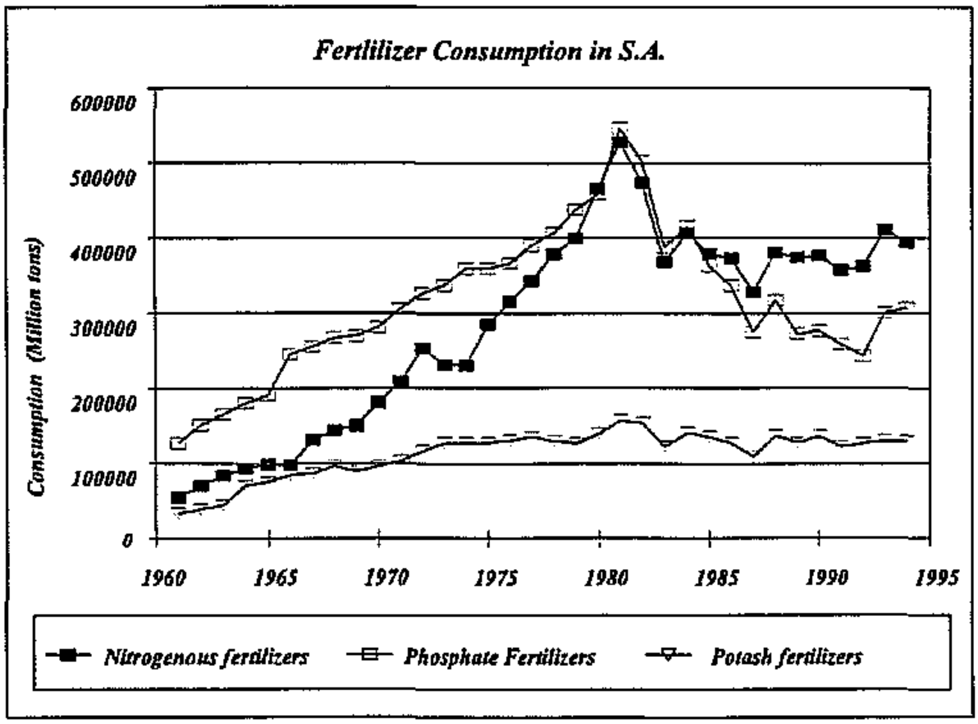


Figure 5.1 : Nitrogenous, phosphate and potash fertilizer consumption values for South Africa. *Source: FAO Fertilizer Database.*

Fertilizer consumption was low until 1966, but the use of nitrogenous and phosphate fertilizers especially rose steadily over the next 15 years. A peak in consumption occurred for all fertilizer types in 1981, following which economic constraints and wider use of alternative nutrient sources have caused the consumption of inorganic fertilizers to level off. Educated farmers have come to realise that efficient management of fertilizer application can ensure the same good crop yields with lower volumes of fertilizer applied.

Unfortunately, crop production rates did not match the rapid fertilizer consumption rates in the years before 1981. This would mean that an increased surplus quantity of nitrogen was available in the soil for oxidation to nitrate and subsequent leaching. Considering the experience of countries abroad, a considerable quantity of this nitrate could still be present in the unsaturated zone (i.e. it has not yet reached the groundwater) in areas where the groundwater level is deeper below surface. In areas where the water table is shallow (<10 m below surface), it is expected that the nitrate applied during the previous two decades has already reached the groundwater (Tredoux, 1993).

As in the rest of the world, nitrogenous fertilizers are the most cause for environmental concern in South Africa. The nitrogenous fertilizer sales by province over the last five years are shown in Figure 5.2.

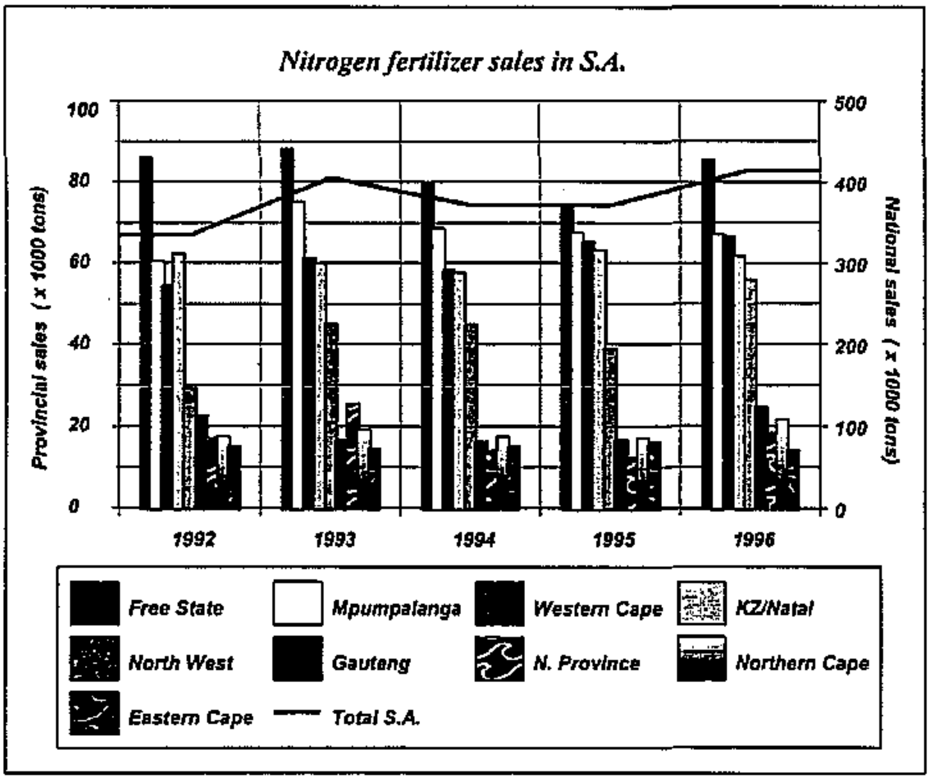


Figure 5.2 : **Nitrogenous fertilizer sales in South Africa, 1992 to 1996.**
Source: Fertilizer Society of South Africa.

5.1.2.3 South African Legislative Controls on Fertilizer Application

Legislation relating to the sale, acquisition disposal and use of fertilizers is covered by the Fertilizers, Farm Feeds, Agricultural Remedies and Stock Remedies Act No. 36 of 1947. After several amendments, the Act now reflects a concern for the protection of the general public and for the environment (Fugle and Rabie, 1994).

Additional legislation relevant to fertilizer application are the Conservation of Agricultural Resources Act 43 of 1983 and the Water Act (DWAF, 1956).

Under section 6 (1) of the Conservation of Agricultural Resources Act, the Minister of Agriculture is authorised to issue environmentally related directions or to prescribe environmentally relevant control measures which must then be complied with by the individual landowners and others to whom they apply (Fugle and Rabie, 1994).

In terms of the Water Act, potential contamination of the water environment, including groundwater, is covered in Sections 21 to 26. The current approach to water pollution prevention by non-hazardous substances is outlined in the DWA&F's Receiving Water Quality Objectives. This approach "*involves specification of the desired quality of the receiving water environment and the control of sources. The approach takes into account non-point and point sources of pollution to the degree necessary to maintain the desired quality. This approach recognises that the receiving water has a certain capacity to assimilate pollution without serious detriment to quality requirements of recognized users.*" (DWA&F, 1991).

Unfortunately nitrate pollution from fertilizer is a non-point source pollution problem and, when it occurs, it is difficult to attribute where the responsibility for solving it should lie.

5.1.3 Impact of fertilizer application on groundwater quality

Research has been carried out, primarily in the US and Europe on the impact on groundwater quality of inorganic commercial fertilizer application and the important findings are outlined here.

The main risks to groundwater contamination from inorganic fertilizers are usually as a result of inadequate management of volume and timing of fertilizer application.

Nitrate is the contaminant with the greatest risk of reaching groundwater at potentially harmful concentrations if fertilizer is applied at above agronomic rates. Potassium from fertilizers may reach the groundwater, but not in quantities sufficient to constitute a health risk. Phosphorus is fairly immobile in the subsurface environment.

5.1.3.1 Nitrogen

Nitrate is the major threat posed to groundwater by the use of fertilizers and is highly soluble in water in a variety of forms. Typically about 20 to 30 % of the applied fertilizer nitrogen leaches out as nitrate regardless of the type of fertilizer used (Bouwer and Bowman, 1989).

Wells and groundwater exhibiting nitrate contamination have been noted in every state in the U.S. Two types of area particularly affected are areas of widespread grain production, marked by heavy fertilization and row cropping, and areas of irrigation and fertilization of vegetable and speciality crops, particularly shallow rooted vegetable crops on sandy soils (Hallberg, 1989).

Recognition of the nitrate problem in groundwater drinking supplies has led to the designation of Nitrate Sensitive Areas (NSA) for special management in the U.K. (Croll, 1994) and California.

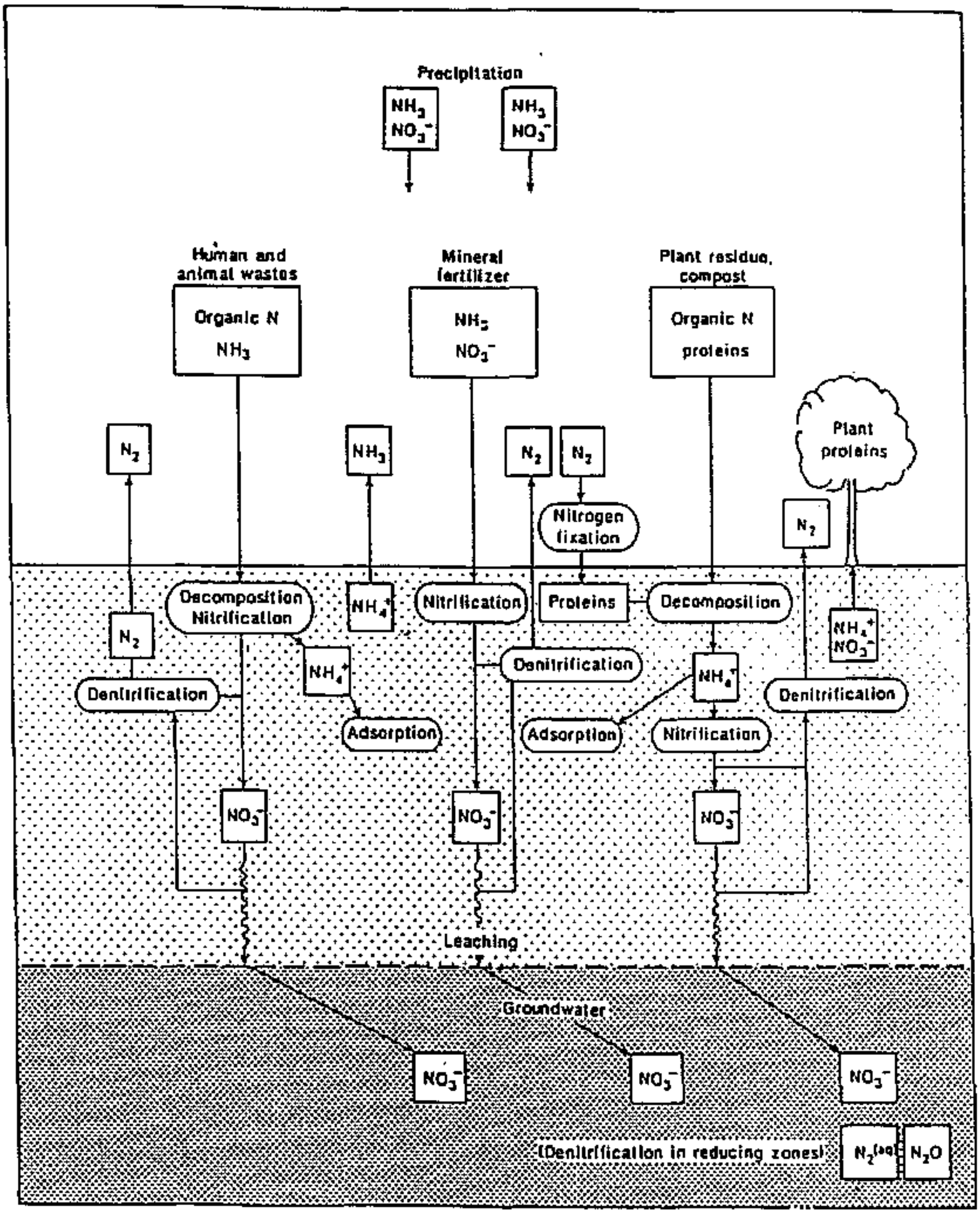
Although little information is available for South Africa, inorganic fertilizer consumption indicates that a situation similar to that in the developed countries could be expected, at least in the areas of intensive agriculture. The highly variable rainfall worsens the situation in the case of fertilized dry land farming due to the associated unpredictable plant cover and fertilizer needs (Tredoux, 1993; Tredoux and du Plessis, 1993).

Nitrate is a cause for concern in drinking water at levels greater than 10 mg.l^{-1} (NO_3- N). At concentrations greater than this there is an increased risk to humans, particularly infants, of methaemoglobinaemia, commonly known as Blue Baby Syndrome (Freshwater Foundation, 1988). The mortality rate among affected infants is thought to be around 10% (Freshwater Foundation, 1988). Nitrate is converted by microbial activity in the low pH conditions of the stomach to nitrite. Once nitrite enters the blood stream it changes haemoglobin to methaemoglobin which reduces the capacity of the blood to carry oxygen.

Livestock has been found to be similarly effected, but at higher nitrate concentrations. Nitrate-nitrogen concentrations in livestock water of 100 to 300 $mg.l^{-1}$ may cause problems if the feed is high in nitrate. Water with concentrations above 300 $mg.l^{-1}$ should not be consumed by livestock.

Nitrogen is found in all soils. Usually in excess of 90% of soil nitrogen occurs in organic forms (Vinten and Smith, 1993). Nitrogen from the atmosphere is naturally fixed (to organic nitrogen) by soil microfauna and leguminous plants at a rate which may be as high as 40 kg $N.ha^{-1}.\ yr^{-1}$ (Goulding, 1990). Commercial fertilizers also achieve this function of fixing atmospheric nitrogen in a plant-available form.

Nitrogen occurs in commercial fertilizers in a variety of forms as nitrate, ammonium, urea or organic nitrogen (Bouwer and Bowman, 1989). As a mobile, highly soluble anion, nitrate moves without adsorbing to soil solids in most soil, and can leach to groundwater whenever inputs to the soil cause drainage which is sufficient to reach the water table (Jury and Nielsen, 1989). Nitrate anions carry a negative charge and are repelled by negatively charged clay mineral surfaces, allowing them to move freely through most soils. Nitrate may be retarded in soils with a significant anion exchange capacity, but these are rare except in tropical areas. Positively charged ammonium ions bind to exchange complexes on clay minerals and organic matter (Smith, 1996) and so have a greater tendency to be retarded in the soil unless nitrified to nitrate. (see Figure 5.3).



Chemical symbols: N, elemental nitrogen;
N_2, nitrogen gas;
N_2O, nitrous oxide;
NO_2^-, nitrite;
NO_3^-, nitrate;
NH_3, ammonia;
NH_4^+, ammonium;
$N_2(aq)$, nitrogen gas; dissolved in water.

Figure 5.3 : The nitrogen cycle (*adapted from Freeze and Cherry, 1979)*.

Ammonium in fertilizers may volatilise or remain in the soil, possibly to be nitrified. Nitrogen losses by **volatilization** will proceed more rapidly at higher temperatures, lower soil moistures and higher air speeds (Kreitler, 1975).

Nitrification is carried out by nitrifying bacteria, commonly *Nitrosomonas* and *Nitrobacter*. The rate of nitrification is more sensitive to environmental factors than that of ammonium mineralization and is influenced by (from Smith, 1996; Krumbein, 1983):

- Soil temperature. Generally the rate of nitrification increases with increasing temperature to an optimal rate at 30 - 35° C. Very little nitrification occurs in soils below 5°C.
- Aeration. Nitrification is an oxidising reaction and requires aerobic conditions.
- Soil moisture content.
- pH. Nitrifying bacteria are inhibited by the acid they produce and therefore the rate is optimised in neutral to alkaline soils.

Nitrification inhibitors, such as dicyandiamide (DCD), are applied in some parts of the world in an attempt to slow down nitrification and consequent nitrate losses. The effectiveness of inhibitors has been shown to be highly variable and sensitive to environmental conditions such as soil temperature (Smith, 1996). DCD degrades more rapidly under warm soil conditions.

Subsequent **leaching** is affected by the relative rates of plant uptake, denitrification, losses of N_2 to the atmosphere and percolation rates (dependent on rainfall or irrigation).

Reported rates of nitrogen leaching are typically between 20 and 30 % of the applied fertilizer nitrogen (Bouwer and Bowman, 1989). Many studies show a direct relationship between nitrate in groundwater and nitrogen fertilization rates and/or fertilization history (Hallberg, 1989). If available nitrogen exceeds crop requirements, then loss of nitrate by leaching generally occurs as a function of the increasing rate of applied fertilizer. In clayey soils this loss occurs above a threshold limit of around 120 kg N ha^{-1}. Nitrate losses from clayey soils are expected to be lower than sandy soils due to the higher soil water retention. The presence of macro-pores in clayey soils can reverse this trend however (Smith, 1996). Several studies have shown that the percentage of nitrate leached increases with increasing total load of nitrogen applied (Vinten and Smith, 1993).

In summary, nitrogen is more likely to leach to groundwater under the following conditions:

- High rates of nitrogen application due to high content in the fertilizer and/or high bulk loadings.
- Application of high N-availability fertilizer or of slow-release N fertilizer at incorrect times to crops with a seasonally limited N demand.
- High soil temperatures increasing the rate of nitrification (and inhibitor breakdown if these are applied).
- Well aerated soil encouraging nitrification. (Denitrification is likely to occur in anoxic conditions.)
- Low levels of plant uptake due to bare ground, low crop requirements or seasonally variable requirements.
- High levels of precipitation or irrigation
- High vertical permeability (including clayey soils with macro-pore channels for rapid downward migration).
- Shallow, unconfined water table.

Nitrate in groundwater and the unsaturated zone is subject to **reduction** (primarily by heterotrophic bacteria) under anoxic or very low oxygen conditions in the presence of an electron donor such as organic carbon (Spalding *et al.*, 1993). Maximum levels of dissolved oxygen of between 2 mg l^{-1} and 0.3 mg l^{-1} have been quoted in the literature (Postma, 1991). Chemodenitrification may occur if the redox potential of the groundwater is low enough (Freeze and Cherry, 1979). The reduction process is defined as **denitrification** when nitrogenous gas is produced (Krumbein, 1983).

In addition to redox potential and the availability of carbon, the rate of denitrification is also dependent upon the concentration of nitrate. High rates of unsaturated zone denitrification, >0.2 kg ha^{-1} d^{-1} are recorded where soil nitrate concentrations exceed 5 mg kg^{-1} (Vinten and Smith, 1993).

High rainfall dilutes nitrates and contributes to denitrification. Nitrate is also less of a problem when water tables are deep. Frequently nitrate concentrations in an aquifer are highest near the top of the water table and decrease with depth. Shallow wells that do not extend far into the groundwater often have the highest nitrate content (Bouwer and Bowman, 1989).

The concentration of nitrate in groundwater is primarily dependent on the following factors (Burt and Trudgill, 1993; Krumbein, 1983; Freeze and Cherry, 1979):

- The concentration of nitrate leached to the water table
- The relative proportions of contaminated leachate and uncontaminated lateral inflow
- The rate of convective groundwater flow
- The rate of hydrodynamic dispersion
- The rates of reduction and denitrification (also dependent on redox conditions in the saturated zone)

Nitrate is rarely reported in fully confined aquifers due to the reducing conditions which tend to exist.

5.1.3.2 Other nutrients and major ions

A review of available literature (CSIR , 1995) found that:

Potassium usually forms a small part of fertilizers. Even though K is very mobile, it is usually only present in low concentrations which do not pose a health risk. K from fertilizers is not regarded as constituting a threat to groundwater quality.

Phosphate is fairly immobile in the subsurface environment as it rapidly adsorbs onto soil particles. Remobilization can occur, but high loads of P, high organic carbon concentrations and warm subsurface conditions are required. As a result, P tends to stay in the upper parts of the root zone and in general does not have an apparent impact on groundwater quality. South African soils are typically deficient in phosphates therefore the likelihood of exceeding the phosphate retention capacity of soils here is even lower.

Inorganic commercial fertilizers comprise soluble salts which produce ions in solution. The level of TDS and electrical conductance of groundwater are therefore parameters that can be influenced by contamination from fertilizer application, if these ions are not immobilized in the unsaturated zone or taken up by plants. The degree to which pH is affected depends on the type of fertilizer applied and the buffering capacity of groundwater and soil water at the site.

5.1.4 Objective of this specialist study

The main objective of each specialist investigation carried out as part of this study is to collect data and information in order to confirm that groundwater contamination occurs as a result of the agricultural practice of concern and to collect evidence which can be used to support the non-technical booklet (CSIR, 1995).

Limited research has been carried out on the impact of fertilizer on groundwater quality within South Africa and very little data has been presented in the literature. It was therefore decided that this practice should be investigated in the field to determine whether the occurrence of groundwater contamination associated with fertilizer application in South Africa concurred with results reported from other countries.

The objectives of the fertilizer specialist study were as follows:

- to select 3 sites for investigation where potentially vulnerable aquifers underlie land exposed to fertilizer application.
- to determine whether contamination of the groundwater had occurred as a result of commercial fertilization.
- to determine, insofar as is possible within the budget and time constraints of the project, the nature and extent (spatially and temporally) of groundwater contamination
- to recommend best management practices to farmers with regard to quantity, type and frequency of fertilizer application so that they can avoid groundwater contamination and protect their resource.

5.1.5 Approach

5.1.5.1 Field site selection

Literature covering similar studies conducted worldwide was reviewed to enable the researcher to conduct the South African study in the most practicable and efficient way.

Three sites were selected for groundwater investigation based on the following criteria:

- the sites should be underlain by a relatively shallow aquifer vulnerable to contamination from the surface
- sites should be representative of a range of fertilization practices, crop types, climatic regions and hydrogeological settings
- the land owner(s) should be supportive of the study and willing to provide relevant information and access
- groundwater monitoring points should preferably exist in the area
- relevant information on the historical and current pattern of surrounding land-use should be available to determine other potential sources of groundwater contamination.

The aim was therefore to select sites where the greatest risk of groundwater contamination existed and where goodwill enabled the study to have access to relevant available data. The study sites were chosen after considerable discussion with hydrogeologists throughout the country who had knowledge of where agricultural activities had impacted on groundwater quality. Many possible areas throughout South Africa were considered for study and, taking into account budget limitations, it was decided that a total of three sites would meet the objectives of the study.

Figure 5.4 shows the location of sites investigated as part of this study.

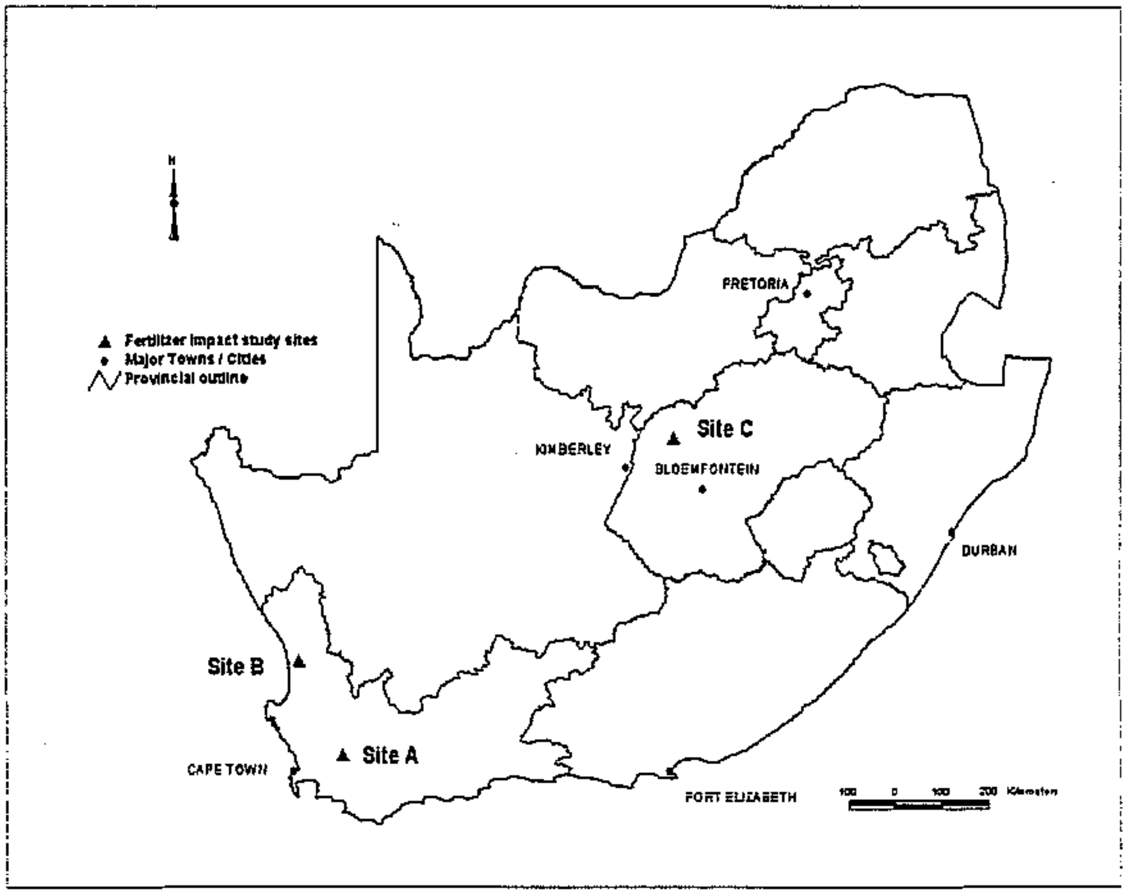


Figure 5.4 : **Location of study sites for investigations of the impact of fertilizer on groundwater quality.**

5.1.5.2 Data acquisition

Where available, the following sources of data were acquired to give as complete an understanding as possible of the hydrogeology and contamination pathways at each site:

- Geological and hydrogeological literature on the locality
- Engineering/ geotechnical/ geophysical/ drilling reports on the locality
- Local data from the National Groundwater Database and National Water Quality Database and Department of Water Affairs and Forestry authorities.

- Information on the local climate and precipitation
- Results of soil and water analyses carried out by previous researchers and land owners in the locality
- Quantities and type of fertilizer used
- Irrigation schedule
- Crop types
- Additional local knowledge from farmers and extension officers

Land-use on the sites and surrounding areas was assessed for other potential sources of groundwater contamination and these were taken into account when interpreting hydrogeological data.

5.1.5.3 Sampling strategy

Groundwater monitoring points were chosen to ensure that representative samples of potentially contaminated groundwater could be collected and that one or more control sampling points were available. Where possible, information on the construction, history and performance of the monitoring boreholes was gathered from the owners and reports from previous studies to enable as complete an understanding as possible of the sampling conditions.

Groundwater sampling protocol was followed, as outlined in Weaver (1992), to ensure that samples representative of water in the aquifer were analysed. This included careful use of sterile sample containers, purging at least 3 well volumes prior to sampling and storage for minimal periods of time (less than 48 hours).

In an attempt to address the objectives of the study, sampling was carried out on a two monthly basis over a period of one year.

Where possible, relevant river water samples were collected in addition to the groundwater samples.

Soil sample results were obtained to assist in understanding the nitrate dynamics.

5.1.5.4 Determinands

Field measurements of groundwater pH, EC and temperature were carried out at each sampling point. The colour and smell of the groundwater was also noted.

A suite of determinands for laboratory analysis was selected to include major ions usually found in groundwater and substances known to occur in commercial fertilizers which have a reasonable degree of subsurface mobility and solubility.

Hydrochemical parameters:	*pH, EC, Alkalinity, temperature.*
Major ions:	*K,* NO_3*-N,* NH_4*-N, P*

On selected occasions, a full set of major cations and anions were analysed for by adding Na, Ca, Mg, SO_4, Cl, DOC, TDS and Hardness to the above.

The hydrochemical analyses were carried out at accredited CSIR laboratories using standard methods.

Soil analyses were obtained from the Department of Agriculture.

5.1.5.5 Nitrogen isotopes

The use of fertilizer was not the only potential source of nitrate identified at the study sites. Nitrogen isotope analyses were conducted in an attempt to determine which source contributed most significantly to any elevated levels of groundwater nitrate.

Seven natural isotopes of nitrogen are know to exist. ^{12}N, ^{13}N, ^{16}N, ^{17}N and ^{18}N are radioactive and have half lives of less than 12 minutes (Fritz and Fontes, 1980). ^{14}N and ^{15}N are stable isotopes. Most nitrogen occurs as ^{14}N with only 0.37% of atmospheric nitrogen occurring as ^{15}N.

The seven percent difference in the atomic masses of the two stable isotopes leads to isotopic fractionation in natural systems. Thus, the isotopic ratio of a nitrogen compound is the result of the following processes (from Kreitler, 1975):

- physical fractionation (diffusion, evaporation and sublimation)
- chemical equilibrium fractionation (equilibrium reactions)
- chemical kinetic fractionation (non-equilibrium reactions)
- the isotopic ratio of the source material.

The ratio of stable nitrogen isotopes in a given sample is compared to a standard (atmospheric nitrogen) as shown below and quoted as $\delta^{15}N$ (parts per thousand, ‰) (Heaton *et al*, 1983) .

$$\delta^{15}N = [\ ({}^{15}N/{}^{14}N \text{ sample} - {}^{15}N/14N \text{ standard}) \,.\, ({}^{15}N/14N \text{ standard})^{-1}\ ] \,.\, 10^{3}$$

A positive $\delta^{15}N$ value indicates ${}^{15}N$ enrichment and a negative value ${}^{15}N$ depletion relative to natural atmospheric proportions of the isotopes.

Variations in natural abundance of stable nitrogen isotopes can semi-quantitatively differentiate agronomic from organic waste sources of nitrate in groundwater (Spalding *et al.*, 1993). As a general rule, manufactured (inorganic) fertilizer materials tend to have $\delta^{15}N$ values close to natural atmospheric abundance (ie. $\delta^{15}N$ is close to zero) as they are manufactured from atmospheric gas. Organic matter in soils is typically enriched in ${}^{15}N$ giving a $\delta^{15}N$ of +3 to +10 (Keeney, 1989; Heaton, 1986). Animal and human wastes have been found to be slightly to highly enriched in ${}^{15}N$ as a result of biochemical fractionation and subsequent ammonia volatilization of the waste. Nitrogen from animal and human wastes typically give $\delta^{15}N$ values of + 7 to +25 (Kreitler, 1975; Heaton, 1986). Therefore isotope data was thought to be useful to this study in differentiating between nitrogen naturally occurring in the soil, nitrogen from sewage contamination and that from inorganic fertilizers applied in agricultural areas.

Samples were collected for nitrogen isotope analysis using recommended procedures including mercuric chloride preservation (Weaver, 1992; Talma, pers. comm.). The analyses were conducted at the CSIR isotope laboratory using standard methods as described in Heaton and Collett (1985).

5.1.5.6 Data quality

The quality of hydrochemical data was analysed to ensure that the results were meaningful and representative of water in the aquifer. Quality control samples were routinely run during analysis. The ionic balance of the hydrochemical results was determined to confirm that the results were within an acceptable margin of error (<5%) and a comparison of pH and EC measured in the field and in the laboratory was made.

5.1.5.7 Interpretation

Results of hydrochemical, microbiological and isotope analyses were analysed in conjunction with information from literature, soil analyses, drilling records and anecdotal information on the local hydrogeology to determine the impact of fertilizer application on groundwater quality at each site.

5.1.5.8 Limits of the project

The study aims to examine examples of the impact on groundwater quality in 'real' situations, ie. where fertilizer is being applied in working agricultural situations. In looking at examples from 'the real world' an element of scientific control is sacrificed.

Often the exact history of the site is not known, the past fertilization practices, fertilizer application rates and so on. The influence of other potential sources of groundwater contamination is also difficult to account for.

Budget, time and access constraints limit the density of sampling points at each study site and data must be extrapolated, often over tens to hundreds of metres, between data points. The degree of hydrogeological heterogeneity at each site is not fully accounted for in the simplified summaries of results. Ideally the monitoring period should have covered more than one crop cycle to assess long-term nitrate accumulation.

A greater density of sampling points would increase the quality of the data, particularly for nitrogen isotopes.

In many cases where existing boreholes are used to sample, the depth and thickness of the aquifer sampled are not known. This makes comparison between sampling points difficult.

The nitrogen isotopes and soil sampling results give a snap-shot with no indication of temporal variability. Detailed data on the nitrate content, soil moisture, grain size distribution, plant N concentrations and demand and redox conditions would enable a better understanding of unsaturated zone processes affecting leaching of nitrate. However the primary aim of this project is to determine whether groundwater contamination has occurred, rather than the detailed modelling of its occurrence.

5.2 SITE A: HEX RIVER VALLEY

5.2.1 Site Background

The Hex River Valley is situated approximately 150 km northeast of Cape Town, in the Worcester district (Figure 5.4). The national road from Cape Town to Johannesburg (N1) passes through this scenic valley. The main town within the valley is De Doorns.

The Hex River Valley is one of the country's main areas for the production of export quality table grapes. It comprises 50 percent of all the table grape growing areas in South Africa.

This site was selected for the following reasons:

- the entire valley is essentially dedicated to the cultivation of table grapes and extensive use is made of inorganic fertilizer;
- it comprises a northeast/southwest trending valley that is 23 km long, has a maximum width of 5 km and is entirely encircled by high mountains;
- there is a long history of table grape cultivation within the valley;
- the valley floor comprises unconsolidated alluvial material with a deep unconfined aquifer and a shallow water table;
- the expected rate of travel within the unsaturated zone is high;
- the aquifer is high yielding and significant groundwater abstraction occurs;
- there are many wells and drains within the area and farmers welcome groundwater investigations on their property;
- a study was carried out in 1989 investigating the impact of pesticides on groundwater quality within the valley. Thus historical data exists within the area for nitrate, potassium and phosphorous concentrations.

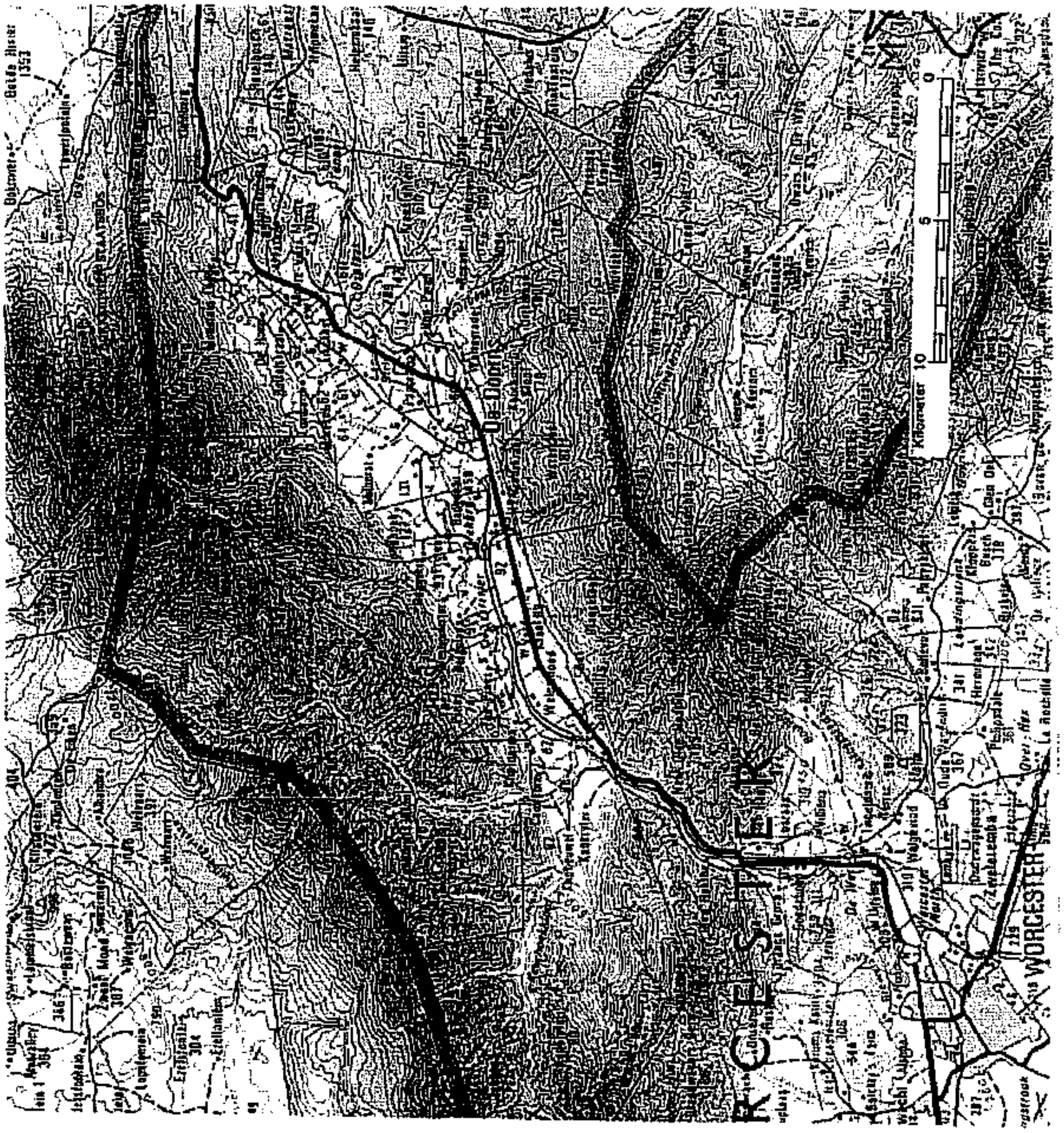


Figure 5.5 : Site A: Locality Map of the Study Area

5.2.2 Application of Fertilizer to the Site

The growing and harvesting cycle of the grapes has a one-year duration. The area has a winter rainfall pattern and during June and July the vines are dormant. During winter, lime and gypsum are applied at 1 ton/ha^{-1}/yr^{-1} to maintain correct soil pH. In winter the vines are also pruned and trimmed. Vines start budding in Spring (August/September) and fertilizer is applied. Fertilizer is typically applied as granules from a tractor drawn calibrated spreader or by hand. Fertigation practices are not commonly used. The fertilizer used is 1:0:1 (NPK). At a shoot length of approximately 20 cm, trace elements are applied to the leaves. Once a shoot length of 35-40cm is attained, bunches start flowering and the flowers transform to "berries". When berry bunches appear (Nov/Dec/Jan) fertilizer is applied again to enhance berry size and the application rate is determined by the appearance of the bunches. For optimum grape quality it is important not to over-fertilize, as too much growth causes a thick canopy of leaves to develop and grapes bunches ideally require 25% direct sunlight. In addition excessive vine growth results in poor grape quality. The vines are also ring-barked to prevent nutrients returning to the ground. The fertilizer commonly used during growth is Limestone Ammonium Nitrate (LAN) (i.e. nitrogen rich). After the harvest (mid April) the vines are fertilized again to carry the plant through winter. Not all farmers fertilize at the start of the growing season, but generally all fertilize after harvest. The fertilizer most commonly used is 1:0:1 which is applied at a rate of 200 kg/ha^{-1}. Soil analyses are carried out every 2 to 3 years and fertilizers applied accordingly. Different types and amounts of organic fertilizer (kraalmis) are used according to the soil type. Water logging of the soils is avoided to prevent root rot and vine die-off. Where required, subsurface drains are installed.

Irrigation:

No irrigation occurs during winter, unless it is very dry, or if it is noticed that the soil is drying from the "bottom up". During growth, irrigation is based on a seven day cycle, i.e. irrigation takes place once per week. Approximately 1m of water is applied per year, but this varies according to soil type, cultivar and root stock. Soil moisture is measured by tensionmeters or neutron probes, typically at depths of 30, 60 and 90 cm. These measurements are also valuable in winter to see if the soil is drying at depth. During irrigation wetting of the leaves is avoided, as this can cause rot. Root stimulation also needs to be maximized, so the water is applied as closely as possible to the roots via a dripper system or ground level sprayer system. Irrigation continues throughout summer and even after picking. On occasion "shock irrigation" is also used. This is usually a 36 hour period of continuous irrigation, for deep wetting of the soil profile. Water for irrigation is abstracted from boreholes, both within the shallow alluvium or the deeper bedrock. These deeper boreholes can be as deep as 100 m. Water is also obtained from the Sanddrif Dam scheme, however, this is an expensive option. Certain farmers have water rights to draw off water from dammed surface water flows.

5.2.3 Geology of the Site

The Hex River Valley is surrounded by rugged mountain ranges. The Hex River Mountains occur to the north and north-west of the valley and the Kwadousberge to the south and south-east. The mountains rise up to 1637 m above the valley floor, with the highest peak, the Matroosberg, being 2249 m above mean sea level. The Hex River Valley is an example of concordant morphology with the valley sides conforming to the limbs of a synclinal structure (Rosewarne, 1984). The surrounding mountains are deeply dissected by tributaries of the Hex River, and the material removed from these steep gorges has been deposited in the valley floor as large alluvial fans. The Hex River flows generally in a south-westerly direction and exits the valley in the south-western corner.

The Hex River Valley has developed along a south-westerly plunging synclinal fold axis. Only the lower two members of the Cape Supergroup are present in the study area, i.e. the Table Mountain Group and the Bokkeveld Group. The Table Mountain Group consists mainly of quartzitic sandstones, whilst shales and mudstones essentially comprise the Bokkeveld Group. Extensive alluvial deposits have been laid down over the valley floor, ranging in lithology from clayey sand to sand and gravel of considerable thickness, particularly along the sides of the valley.

After deposition of sediments of the Cape Supergroup, a period of major tectonic movement occurred and the rugged topography of the Hex River Mountains and Kwadousberge is the result of tectonic conflict at the convergence of two major fold belts.

The lithostratigraphy of the Cape Supergroup in the Hex River Valley and environs is shown in Table 5.3 (SACS, 1980).

Table 5.3 : Lithostratigraphy of the Cape Supergroup in the Hex River Valley and environs (SACS, 1980)

Group	Subgroup	Formation	Thickness (m)	Lithology
Witteberg	Weltevrede			
Bokkeveld	Ceres	Boplaas Sandstone	30	Sandstone
		Tra-Tra Shale	180	Shale
		Hex River Sandstone	70	Sandstone
		Voorstehoek Shale	220	Shale
		*Gamka Sandstone**	*35*	*Argillaceous Sandstone*
		*Gydo Shale**	*140*	*Shale*
Table Mountain		*Nardouw Sandstone**	*500*	*Sandstone*
		*Cederberg Shale**	*120*	*Shale*
		*Pakhuis**	*40*	*Sandstone Diamictite*
		Peninsula Sandstone	1550	Sandstone

* *present in the study area*

Table Mountain Group

The base of the Group is not seen in the Hex River Valley. The Cedarberg Shale and Pakhuis Formation are generally poorly exposed and form a green vegetated swath that contrasts very clearly with the barren rocky surfaces of the quartzites above and below.

In the Hex River Valley the most important member of the Table Mountain Group is the Nardouw Sandstone.

Bokkeveld Group

Conformable with the Table Mountain Group and occupying the core of the syncline is the sequence of alternating shales and sandstones of the Bokkeveld Group. This group is only exposed to the east and north-east of De Doorns. The unweathered shales are typically dark grey in colour, micaceous and locally contain abundant pyrite. The sandstones are impure, micaceous and fine to medium grained. The lowermost sandstone, the Gamka Sandstone, is poorly developed in the Hex River Valley.

Alluvium

The alluvium can be divided according to the source rock, i.e. Table Mountain Group and Bokkeveld, whilst there is alluvium consisting of material derived from both sources. Alluvium derived from the Table Mountain Group is the most extensive of the alluvial deposits and is predominantly quartzitic and typically off-white to light brown in colour. Alluvial deposits derived from the Bokkeveld Group are much less extensive and on average thinner. The alluvium is reddish brown in colour. Mixed alluvium consists typically of a silty and sometimes clayey matrix with sand and gravel up to cobble/boulder size and is buff coloured.

Hydrogeology

The hydrogeology of the Hex River Valley has been investigated by Dziembowski (1970), Vegter (1974), Rosewarne (1981, 1984), and Weaver (1993). There is hydraulic connection between the alluvial deposits, Bokkeveld Group and the Table Mountain Group with aquifer conditions ranging from unconfined to semi-confined with leakage (Rosewarne, 1984). **Recharge** is mainly effected by rainfall infiltration on the surrounding mountainous areas of the Table Mountain Group. Leakage from the alluvial deposits to the underlying bedrock is an important recharge mechanism. Figure 5.6 is a conceptual hydrogeological cross-section.

Groundwater in the Table Mountain Group is of good **quality** whilst that of the Bokkeveld is of poorer quality. The quality of the groundwater improves with proximity to the TMG Nardouw Sandstone.

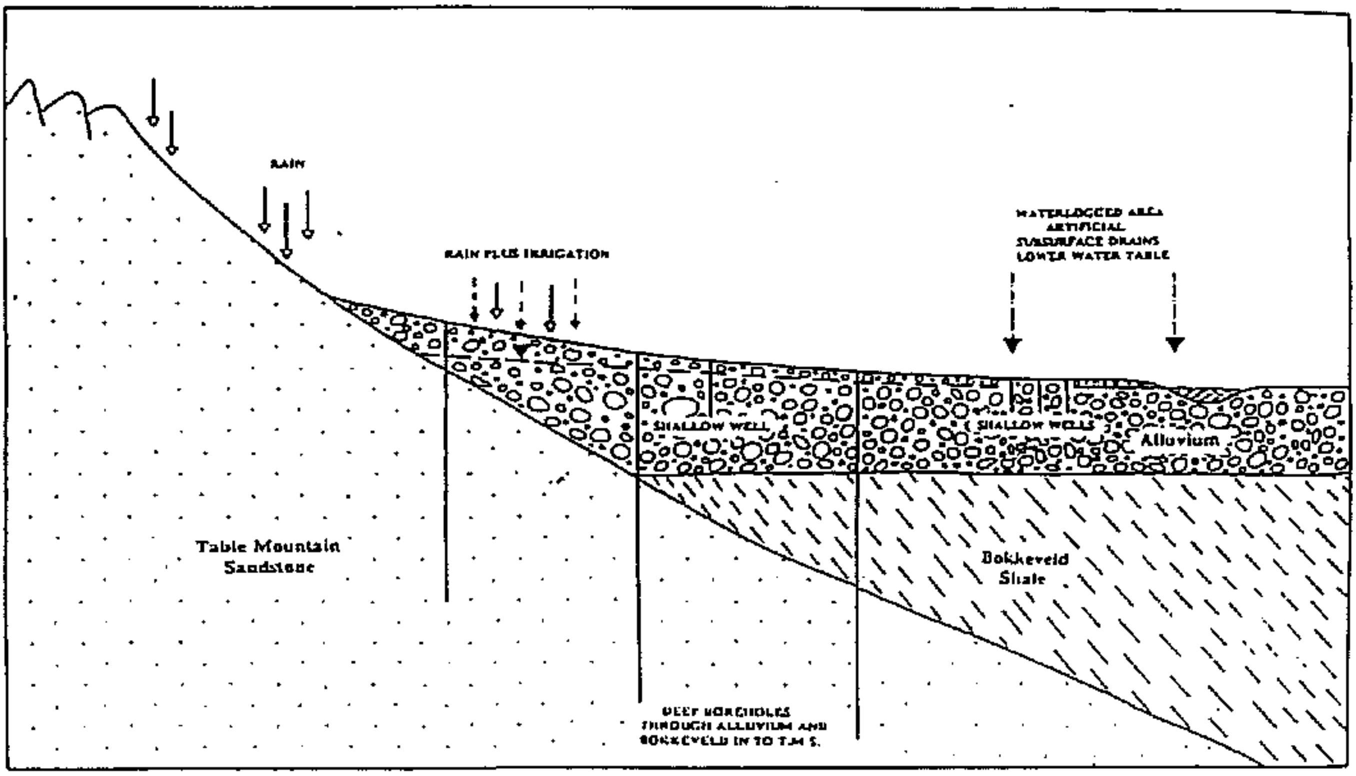


Figure 5.6 : **Site A: Diagrammatic geological cross-section showing the hydrogeology of the Hex River Valley** *(Weaver, 1993).*

The range of transmissivity values in the alluvial deposits is large ranging from 20 to 280 m^2/day, however, saturated thickness of the alluvium varies considerably and a range of 6 m/day to 14 m/day is reported for hydraulic conductivity (Rosewarne, 1984), with the higher values obtained in alluvium derived from the Table Mountain Group. Specific yields range from 1×10^{-3} to 1×10^{-1}, which are representative of unconfined to semi-confined conditions. Yields for boreholes drilled into the mixed alluvium range from 0.5 L/s to 4.0 L/s, whilst boreholes drilled into the Table Mountain Group alluvium range from 2.6 to 7.3 L/s. The hydrogeology of the underlying bedrock is not discussed, as only alluvial water was sampled to meet the objectives of this study.

The Hex River Valley lies within the winter rainfall region. Total rainfall for 1996 measured at De Doorns was 529 mm. The annual evaporation for 1996 at De Doorns was 2155 mm. The driest months are typically December, January and February and the wettest June and July. Only for 2 months of the year did rainfall exceed evaporation (mid-May to mid-July) in 1996 (Figure 5.7). Figure 5.7 also shows the monthly averages and absolute monthly minimum and maximum air temperature.

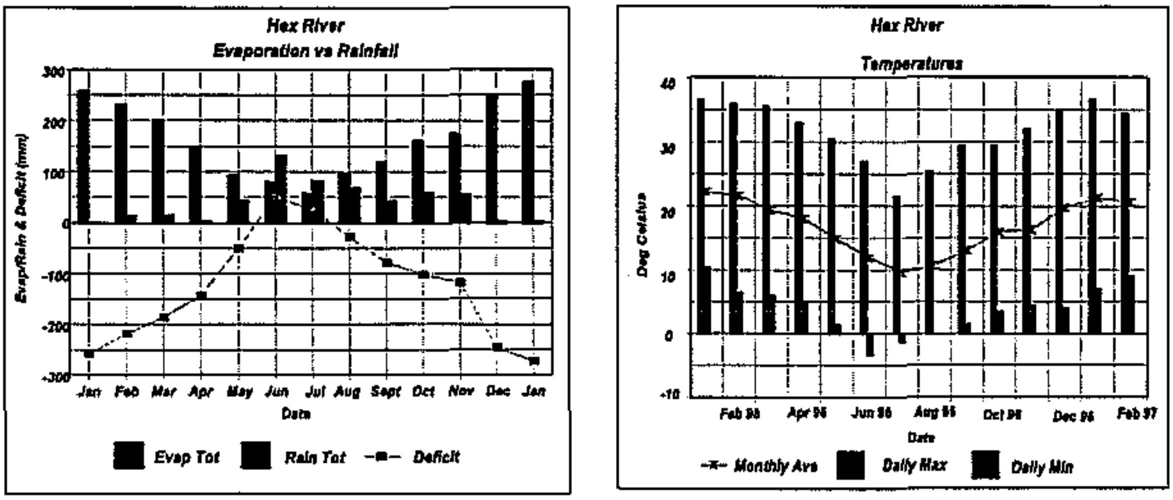


Figure 5.7 : **Site A: Climatic data for the Hex River Valley (De Doorns)**

Static water levels were measured in the wells that were sampled. There is a similarity in water level fluctuations between the measured sites. The water levels are at their lowest in June and their highest in November (Figure 5.8).

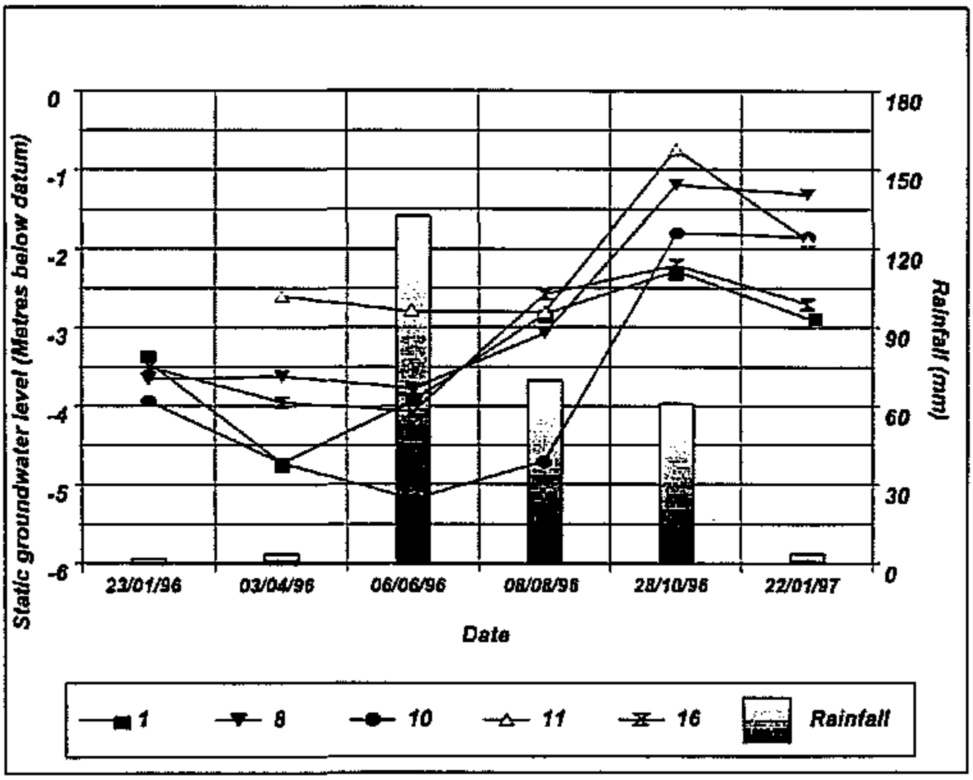


Figure 5.8 : **Site A: Static water level fluctuations**

The low levels in June are attributable both to the low rainfall in the months of January, February, March and April and the high abstraction from the wells for irrigation purposes up until mid-April. The winter rainfall recharges the aquifer and water levels rise until after November when there is a reduction in rainfall. The maximum difference in water levels per well varies from 1.7 m to 3.4 m. Figure 5.9 shows a static water level map, generated from the data obtained for this study.

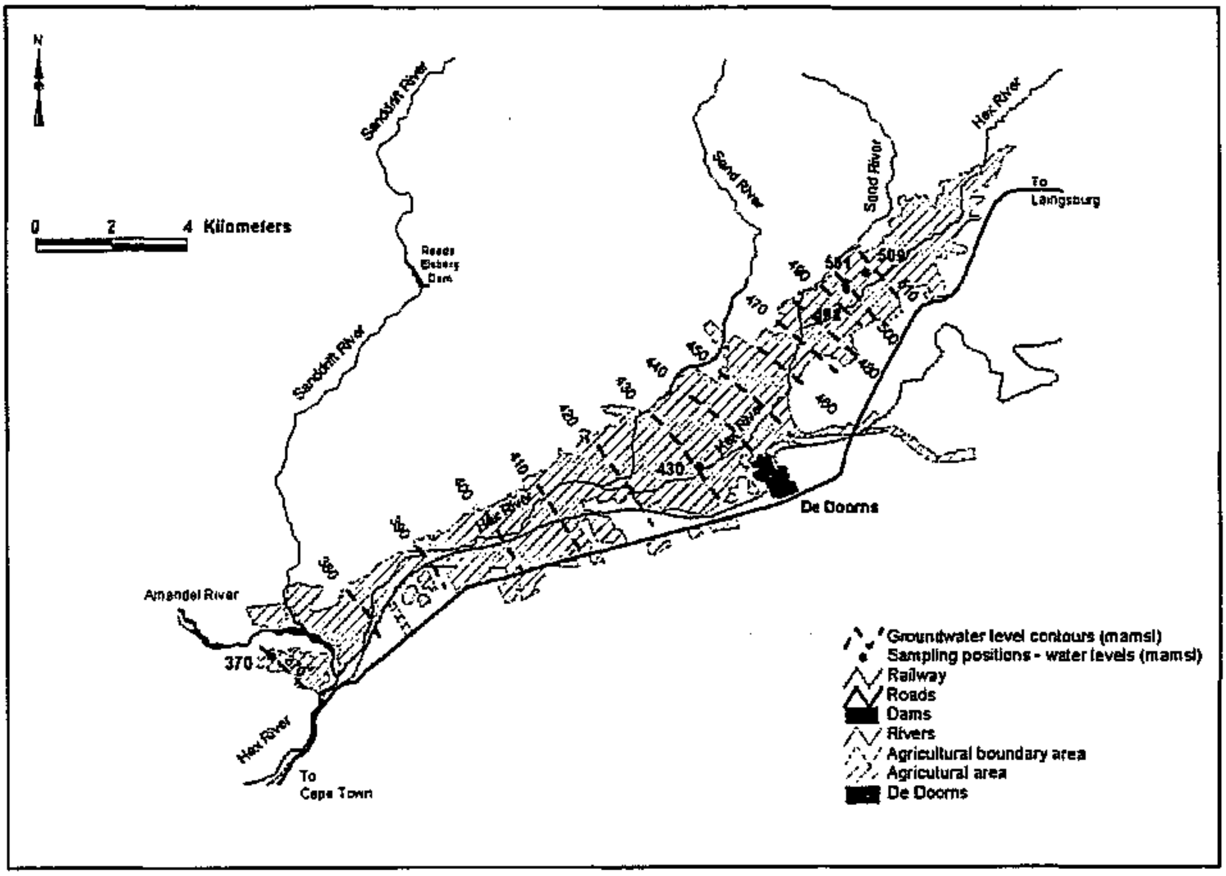


Figure 5.9 : Site A: Generalized Static Water Level Map(metres above mean sea level) The Contour interval is 10 m.

Within the alluvium, the groundwater flows in a southerly direction from the Hex River Mountains; in a westerly direction from the eastern areas and in a south westerly direction along central parts of the valley. The main recharge fronts are located on the northern flanks of the valley adjacent to the Hex River Mountains, where rises of over 30 m occur (Rosewarne, 1984). The alluvial aquifers are in hydraulic connection with the surface water in the Hex River and its tributaries.

Regarding the hydrochemistry of the groundwater within the Hex River Valley, the water with the lowest electrical conductivity (EC) occurs in the alluvial deposits associated with the Nardouw sandstone in the western portion of the valley. The conductivity increases towards the east of the valley as the Table Mountain Group thins and the Bokkeveld Group becomes dominant.

5.2.4 Field Investigation

The field investigation was carried out over the year from January 1996 to January 1997. Figure 5.10 shows the sampling positions for the current study. The sampling points chosen were those already established during an earlier study of pesticides in groundwater (Weaver, 1993).

The sampling was carried out over a year with groundwater sampling and water level measurements being taken every 2 months. The objective was to investigate temporal trends as well as spatial patterns in groundwater quality. The growing and harvesting of grapes occurs on an annual cycle, so it was chosen to monitor groundwater quality and water levels for one year to cover the impact of the whole cycle. As the application of fertilizer to the grapes only occurs at certain times of the year and not on a continuous basis, it was hoped that if nitrate levels in groundwater fluctuated this would follow the same pattern as fertilizer loading, accepting there may be a lag, thereby confirming that the use of fertilizers does have an impact on groundwater quality.

The sampling points are well distributed throughout the valley and as this study was to focus on the impact of fertilizers on nitrate concentrations within groundwater, only shallow alluvial wells were sampled. Samples were collected from wells, drains and rivers. The samples collected from each sampling run were analysed for NO_3-N, NH_4-N, K, P and EC. A full analysis of all major cations and anions was also completed once for each sampling point (Sample Run no. 4, 8 August 1996). In addition samples were also collected for nitrogen isotope analysis (Sample Run no. 6, 22 January 1997). No microbiological or metals analyses were carried out.

A total of 13 sampling localities were used. Five of these were large diameter wells, four were sub-surface drains and four were river samples. For sampling the wells, at least 3 well volumes were pumped out prior to the collection of the samples.

5.2.4.1 Description and discussion of hydrochemical data

The full results of all hydrochemical analyses are presented in Appendix 3.A. Table 5.4 is a listing of selected chemistry and waterlevel data. Phosphate and ammonia data has not been included as in most cases these were below detection limits.

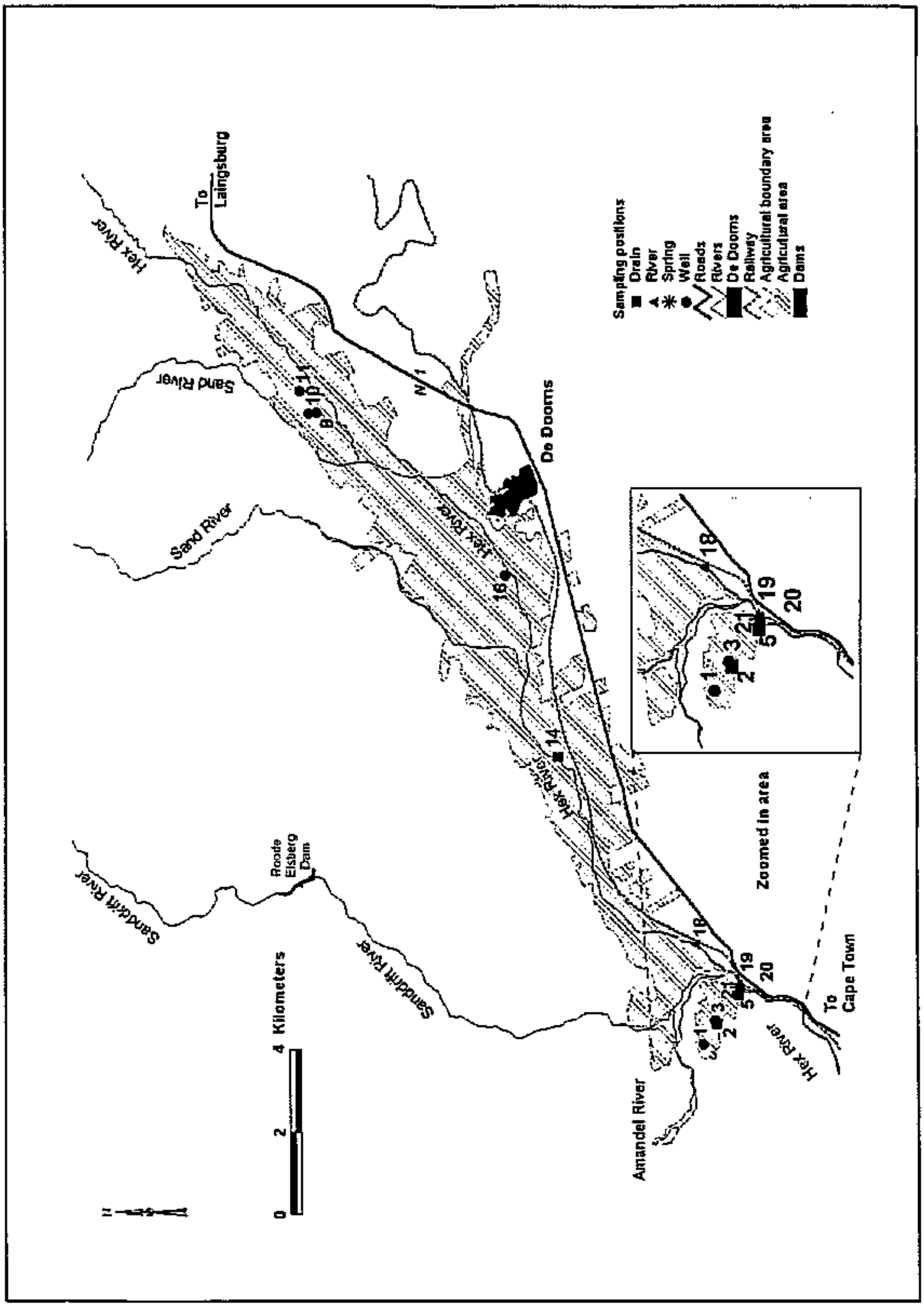


Figure 5.10 : Sampling Localities

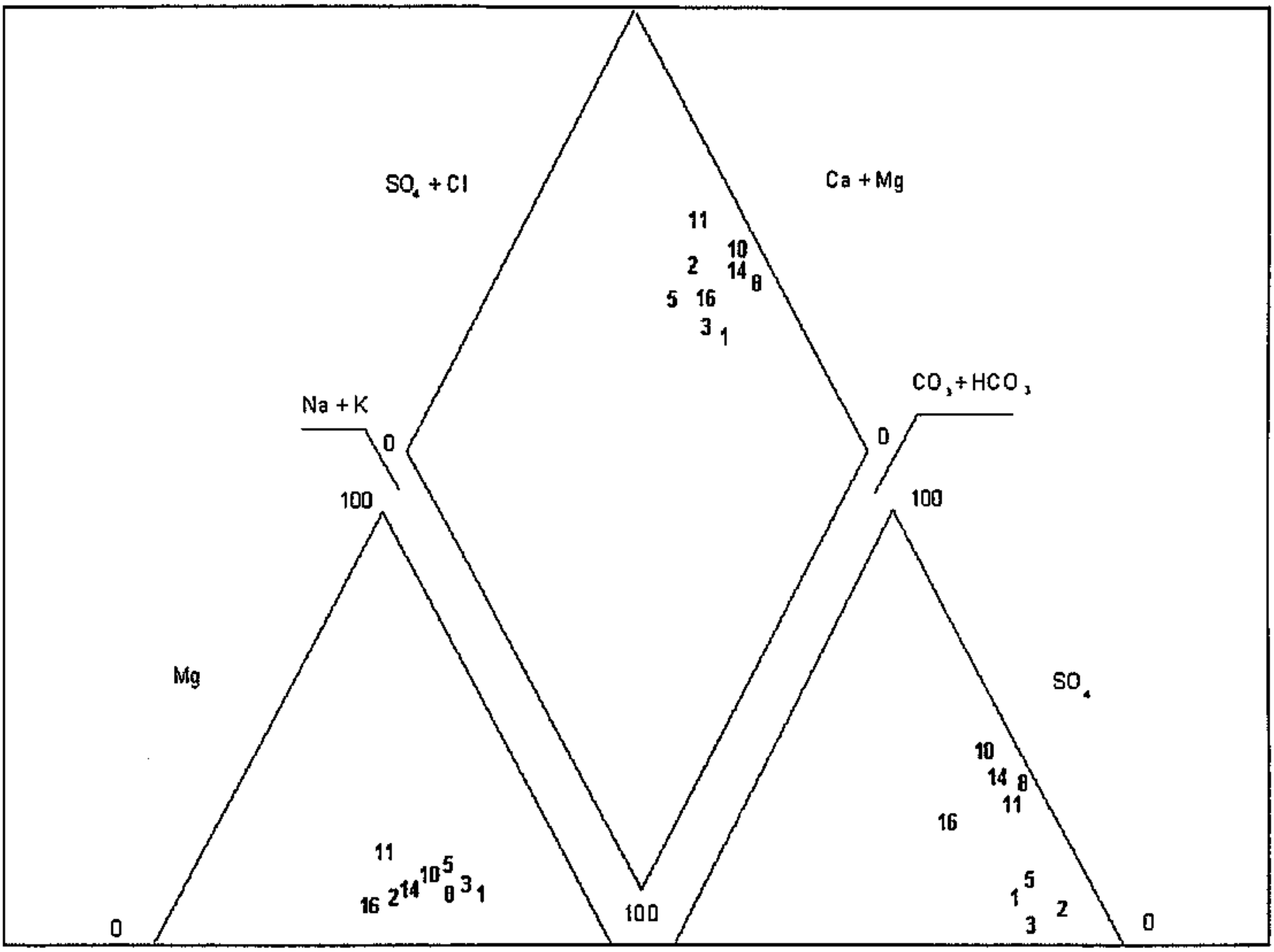


Figure 5.11 :A Piper plot of well and drain samples

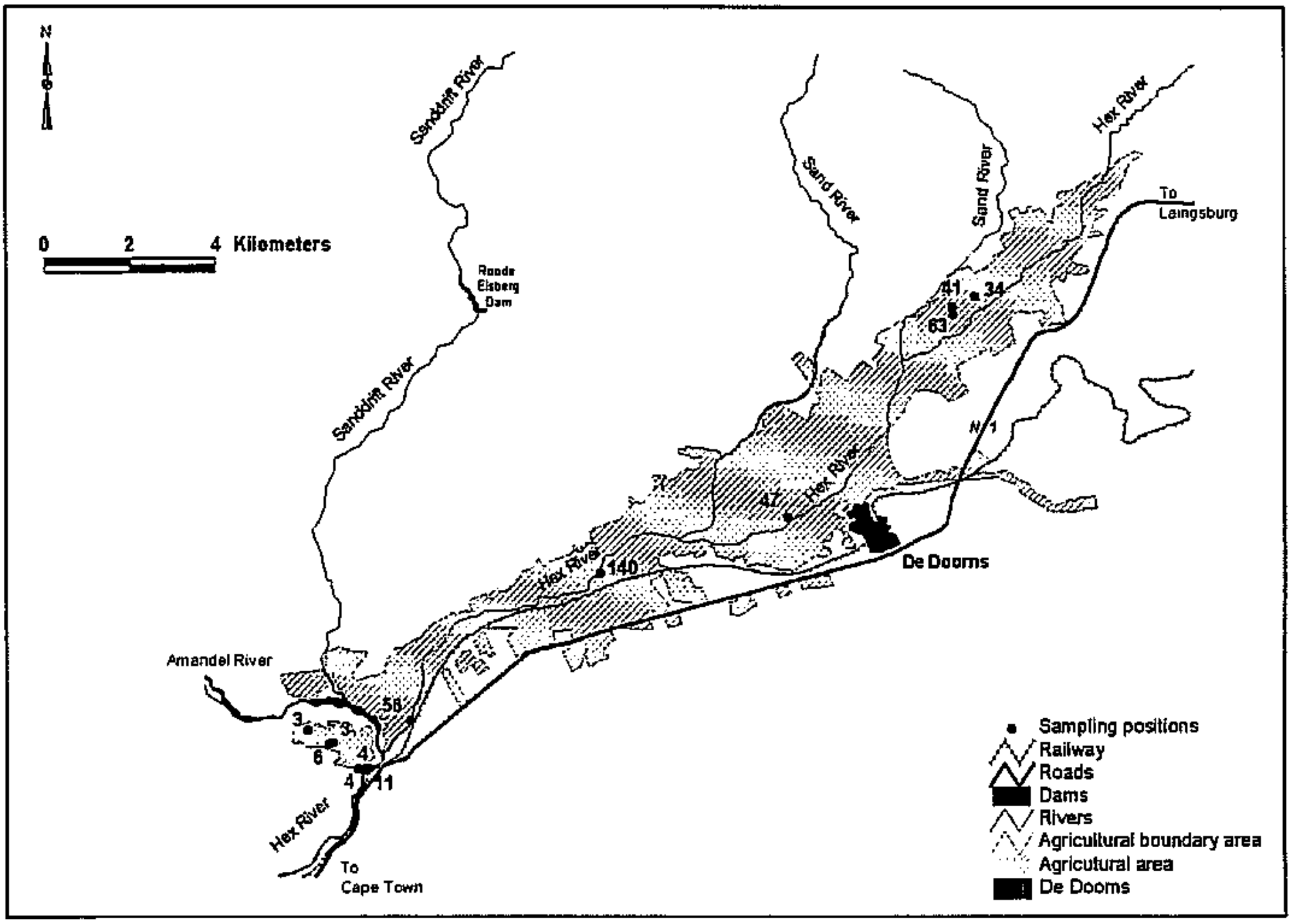


Figure 5.12 : Spatial distribution of electrical conductivity values (mS m^{-1}) (24 October 1996)

Major ions

The results of the full analysis (Sample Run no. 4, 8 August 1996) were plotted on a Piper diagram (Figure 5.11), showing the general hydrochemical character of the samples. There are no cations that are dominant and chloride is the dominant anion. The alluvial groundwaters plot in the upper half of the diamond field and show Ca^{2+}, Mg^{2+}, Cl^- and SO_4^{2-} character.

Electrical Conductivity (EC)

EC ranges from a very low 2 mS.m^{-1} to 140 mS.m^{-1}. Figure 5.12 shows the spatial distribution of EC values through the Hex River Valley, based on the 24 October 1996 sampling run, when the EC values were generally at the range maximum.

The south western corner of the valley (Kanetvlei) has groundwater of outstanding quality, with EC values typically below 5 mS.m^{-1}. The north eastern portion, has poorer quality water, with values ranging between 34 mS.m^{-1} to 63 mS.m^{-1}. The highest value recorded in this study was 140 mS.m^{-1} in the central portion of the valley, measured in an agricultural drain. There are insufficient data points to generate an EC contour map of much value, however, Rosewarne (1984) produced a conductivity contour map using many more data points and there is close agreement between the values found in the two studies. The recharge water flowing in from Hex River Mountains and Kwadousberge is of very high quality, being essentially rain water, and salinity increases toward the central portion of the valley and toward the east of the valley. The deterioration in water quality toward the eastern portion of the valley is attributable to the higher content of soluble ions in the Bokkeveld Group which outcrops in the eastern portion of the valley. The highest EC value recorded is 77 mS.m^{-1} higher than the second highest value. The reason for this high value is not understood at this stage, as the agricultural setting and farming practices are no different from other parts of the valley.

Regarding the temporal variations of EC values over a period of a year, the biggest of 21 mS.m^{-1} was recorded at sample point 8, whilst an adjacent sample point, 180 m away showed the biggest decrease of 16 mS.m^{-1} (Table 5.4). These differences are derived from comparing January 1996 data to January 1997 data. The average change for all 12 sampling points is +0.8 mS.m^{-1} and is considered insignificant.

Table 5.4 : EC differences between January 1996 and January 1997

Sample No	Feature	Sample Run 1 mS.m^{-1} 23/1/96	Sample Run 6 mS.m^{-1} 22/1/97	Difference mS.m^{-1}
1	Well	3.0	3.0	0
2	Drain	3.6	5	+1.4
3	Drain	2.9	3.0	+0.1
5	Drain	3.3	4.0	+0.7
8	Well	32	53	+21
10	Well	45	29	-16
11	Well	26 (1/4/96)	21	-5
14	Drain	136	131	-5
16	Well	52	54	+2
19	River	27	38	+11
20	River	4.0	3.0	-1
21	River	3.0	3.0	0
			Average	+0.8

On inspection of the yearly fluctuations of Electrical Conductivity (Figure 5.13), there is a general increase in EC measured during the 24 October 1996 sampling run, except for Sample 16. This increase will be discussed in more detail when the nitrate and potassium concentrations are investigated.

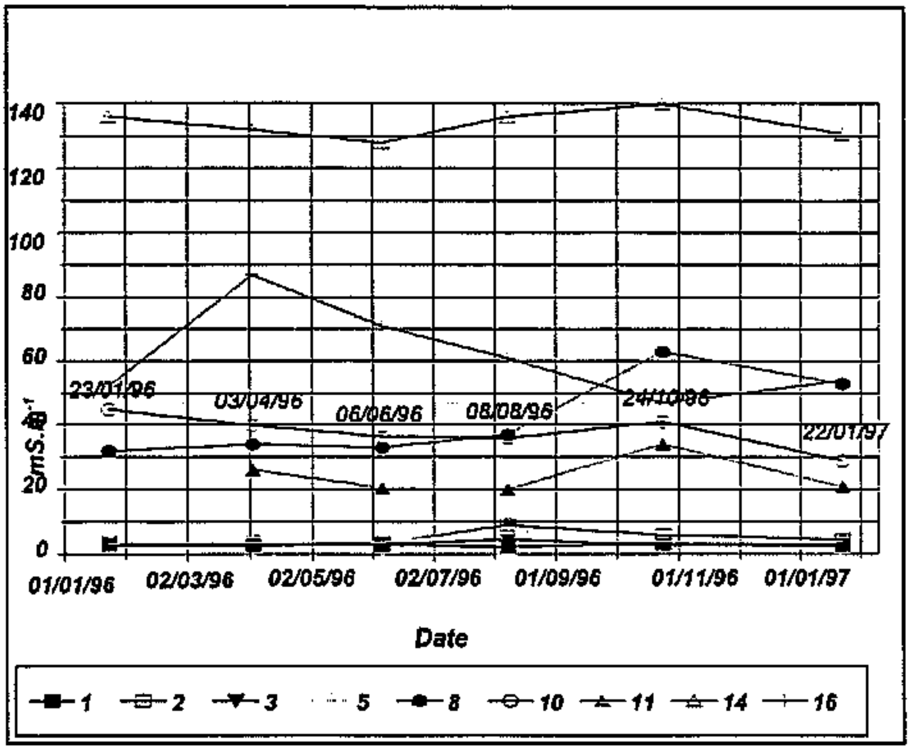


Figure 5.13 : Site A: Electrical Conductivity values for a year of sampling (1996)

Nitrate

There is a large degree of spatial variability with regard to nitrate values. Values range from below detection limit to 13.2 mg.l^{-1} NO_3-N. The nitrate values are lowest in the south-western Kanetvlei area, and highest in the north-eastern valley area. Figure 5.14 shows the spatial distribution of nitrate values throughout the valley for the 24 October 1996 sampling run.

On inspection of the data for temporal variation (Figure 5.15) there are different responses for different geographical areas. In the Kanetvlei area, samples 1, 2, 3 and 5 show a nitrate maximum during the 8 August 1996 sample run. This is the dormant period of the grape season, there are no leaves on the vines, most trees have lost their leaves and the mountain peaks are usually snow covered at this time of year.

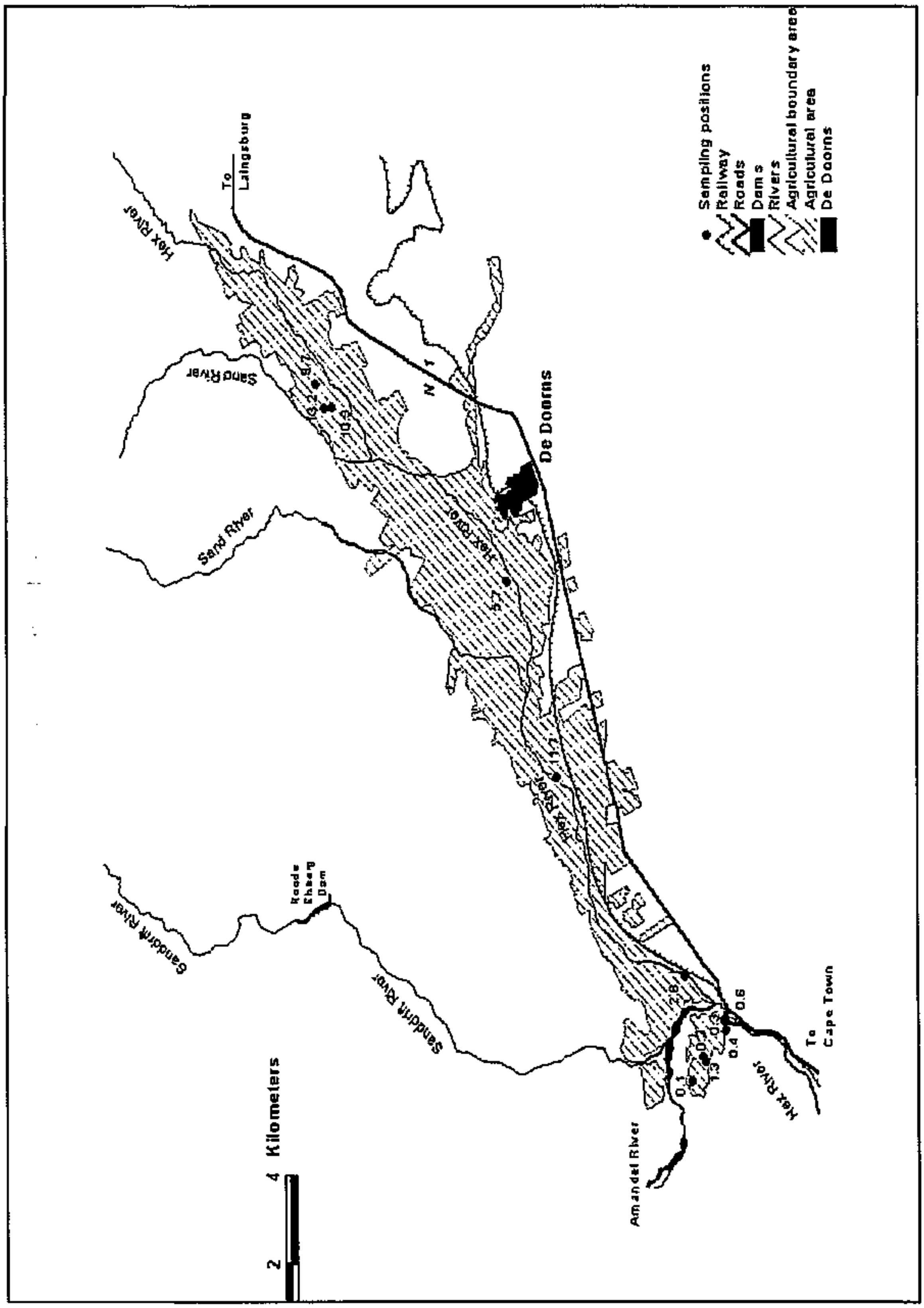


Figure 5.14 : Site A: Spatial distribution of nitrate (NO_3-N) as mg.l^{-1} (24 October 1996)

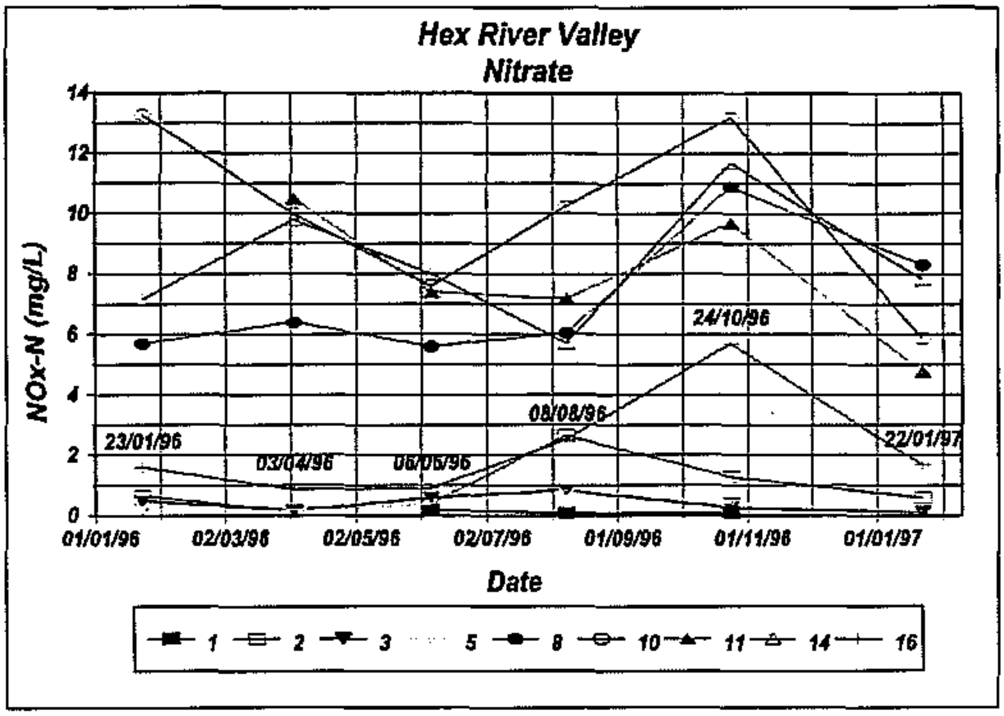


Figure 5.15 : Site A: Nitrate values for a year of sampling (1996)

This is, however, a period of high rainfall and the elevated nitrate levels may be due to the increased downward flow of water mobilizing the nitrate in the soil, with denitrification processes being minimized and the nitrate being flushed down to the water table. However, this pattern is not seen for the samples collected in other portions of the valley. For samples 8, 10, 11 and 14 there appears to be a nitrate concentration peak in early April and again an even higher peak in late October. The nitrate peak in early April coincides well with the post harvest fertilization carried out to sustain the vines through the winter period. The rapid response of nitrate concentrations in groundwater to the surface application of fertilizer is reasonable in that the unsaturated zone comprises alluvial gravels and coarse sands and although not measured, should have a hydraulic conductivity of $\geq 10^{-2}$ m/day. In addition the water table is shallow and the maximum water table depth recorded in the valley was 5.35 m (well 10, August 1989). After this post-harvest fertilizer application, no further fertilizing is carried out during winter and there is a corresponding decrease in nitrate levels in groundwater. The second and higher peak, measured in late October, coincides with the application of fertilizers at the start of the growing season. The slight elevation of nitrate values in early August seen in the Kanetvlei area is not seen in the central and north-eastern parts of the valley.

Regarding the net increase or reduction in nitrate levels over a one year period, the largest increase recorded was 2.62 $mg.l^{-1}$ (Sample 8) and interestingly the largest decrease was 7.40 $mg.l^{-1}$ at the sampling point 180 m away (Sample 10). These differences are derived from comparing January 1996 data to January 1997 data. The reduction in nitrate concentration at Locality 10 is believed to be due to a tract of a vineyard adjacent to the well, being removed, retrellised and re-planted. In addition melons were planted between the vines, just in this area, so nitrate uptake would have been high.

Table 5.5 : Nitrate differences between January 1996 and January 1997

Sample No	Feature	Sample Run 1 $mg.l^{-1}$ 23/1/96	Sample Run 6 $mg.l^{-1}$ 22/1/97	Difference $mg.l^{-1}$
1	Well	<0.1	<0.1	0
2	Drain	0.65	0.59	-0.06
3	Drain	0.49	0.12	-0.37
5	Drain	0.22	0.14	-0.08
8	Well	5.70	8.32	2.62
10	Well	13.28	5.88	-7.4
11	Well	10.5 (1/4/96)	4.78	-5.72
14	Drain	7.15	7.83	+0.68
16	Well	1.60	1.68	+0.08
19	River	1.31	2.10	+0.79
20	River	1.23	<0.1	-1.13
21	River	<0.1	<0.1	0
			Average	-0.88

Table 5.5 shows the changes in nitrate levels over a period of a year. There is a net reduction of 0.88 $mg.l^{-1}$ if the differences are averaged for all sampling points.

Potassium

Potassium values throughout the valley are low, with the lowest values being in the south-western part of the valley. Figure 5.16 shows the potassium values for the 24 October 1997 sampling run.

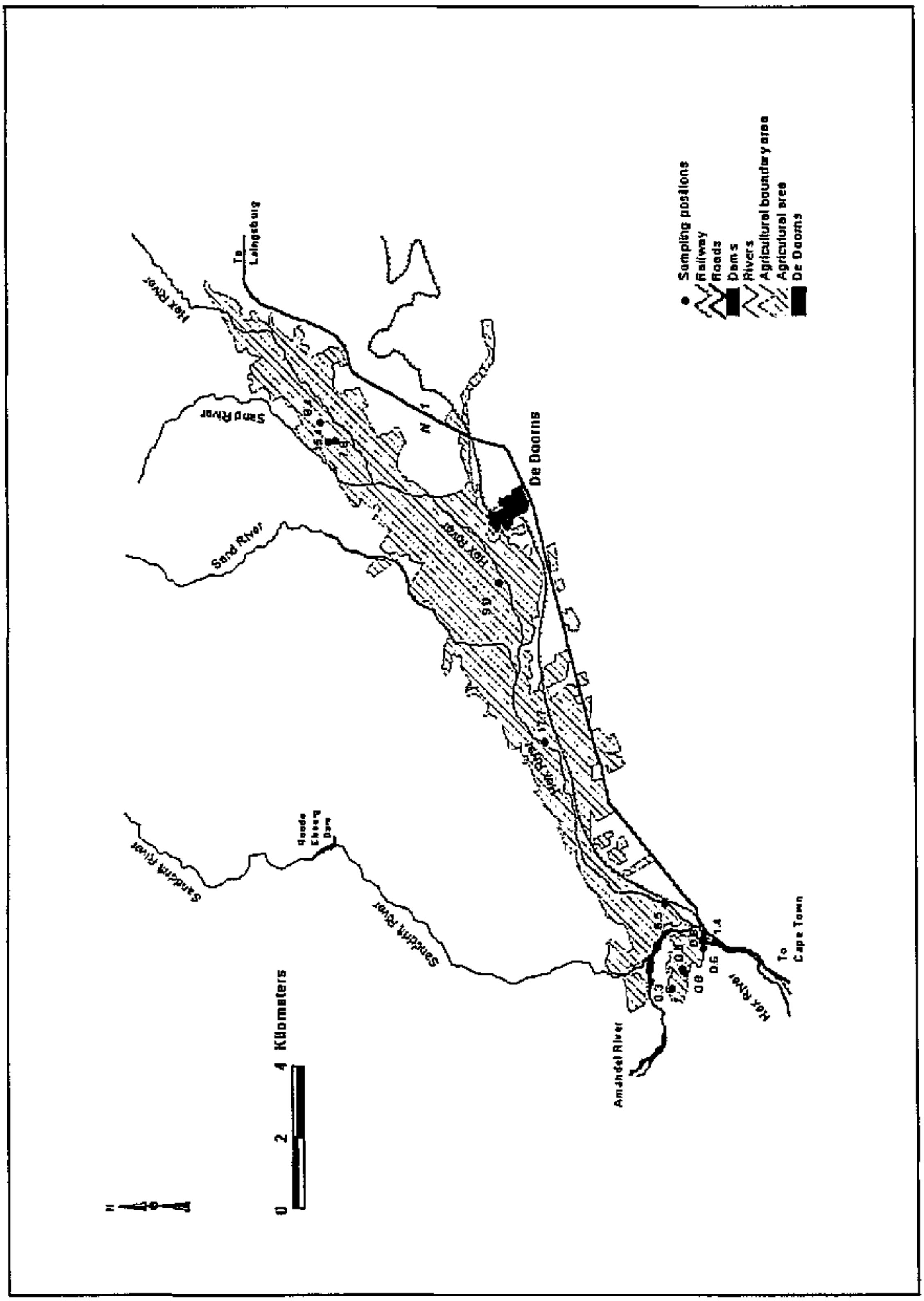


Figure 5.16 : Site A: Spatial Distribution of Potassium (mg.l^{-1}) 24 October 1997

Figure 5.17 shows the potassium concentrations for each sampling locality as measured for one year.

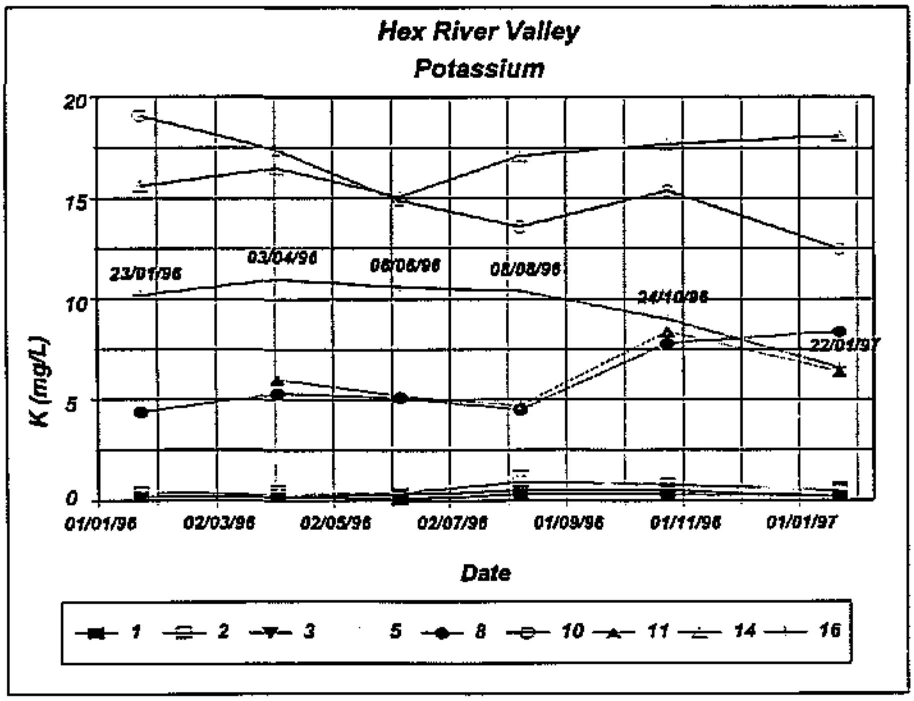


Figure 5.17 :Potassium values for a year of sampling (1996)

The trends apparent from the nitrate values are not that clearly defined from potassium. From the early August to late October sample runs there are four localities that show an increase in potassium levels which may be linked to the application of fertilizers at the start of the growing season. Otherwise the potassium concentrations remain relatively stable, indicating that plant uptake can match the amount being applied or that the soils have a high capacity to retain potassium. Table 5.6 shows the data for assessing whether there was a net gain or decrease in potassium concentrations.

Table 5.6 : **Potassium difference between January 1996 and January 1997**

Sample No	Feature	Sample Run 1 mg.l^{-1} 23/1/96	Sample Run 2 mg.l^{-1} 22/1/97	Difference mg.l^{-1}
1	Well	0.2	0.2	0
2	Drain	0.41	0.5	+0.09
3	Drain	0.19	0.2	+0.01
5	Drain	0.41	0.6	+0.19
8	Well	4.4	8.4	+4
10	Well	19.1	12.5	-6.6
11	Well	6 (1/4/96)	6.4	+2.4
14	Drain	15.6	18.1	+2.5
16	Well	10.2	6.6	-3.6
19	River	3	4.2	1.2
20	River	3	0.6	-2.4
21	River	0.5	0.6	0.1
			Average	-0.19

Phosphorous

Phosphorous was also analysed for, however, in all samples collected and analysed the concentrations were below analytical detection limits. Phosphate fertilizers are not commonly used in the study area.

Nitrogen isotopes

Nitrogen-isotope analyses ($\delta^{15}N$) were carried out in an effort to determine the source of nitrate in the groundwater. Samples were collected during Sample Run No. 6 (22 January 1997) from localities 8, 10, 11 and 14. Not all samples were submitted for isotope analysis due to the low nitrate concentrations of most of the samples. The results were as follows:

Table 5.7 : **δ^{15}N values for four sampling localities**

Locality No	δ^{15}N (%)	Nitrate-N (mg.l^{-1})
8	+15.1	8.32
10	+11.0	5.88
11	+6.6	4.78
14	+14.5	7.83

According to the widely accepted list published by Hübner (1986) commercial fertilizers have a range of -2 to +4% δ^{15}N and human and animal waste has a range of +10 to +20% of δ^{15}N. The dominant land-use practice for localities 8, 10 and 11 however, is grape growing (Figure 5.18). The nearest house or septic system is at least 300 metres up gradient from the sampling sites, and it is highly unlikely that these values are affected by any other anthropogenic activity besides grape farming. At locality 14, which is an agricultural drain, it is difficult to assess the extent of the sub-surface drainage system, however, 50 m away up gradient from the drainage system there are houses with septic systems and it is possible that part of this water is entering the agricultural drains.

The isotope values obtained in this study are more typical of human and animal waste, yet, this is an unlikely source of nitrate, since the dominant land use activity is table grape production. The impact of organic fertilizer (kraalmis) on nitrogen isotope values still needs to be investigated. The higher δ^{15}N values may be caused by isotopic fractionation during the volatile loss of ammonia from nitrogen applied in reduced forms. Kohl *et.al.* (1971) report that soil is generally enriched in ^{15}N. This probably results from the fact that denitrification favours ^{14}N which leaves soil nitrogen enriched in ^{15}N. In laboratory experiments Kohl *et.al.* (1971) produced nitrate from reduced soil nitrogen. The nitrogen from this nitrate was enriched and had a value of $\delta^{15}N = +13$. Nitrogen from anhydrous ammonia fertilizer was found to be considerably less enriched in ^{15}N and a value of +3.7 was reported.

Böttcher, *et.al.* (1990), Hübner (1986) and Freyer and Aly (1974) have noted a negative correlation between nitrate concentrations and δ^{15}N values. Figure 5.19 shows this relationship to have a strong positive correlation ($r^2 = 0.98$) for the samples in this study.

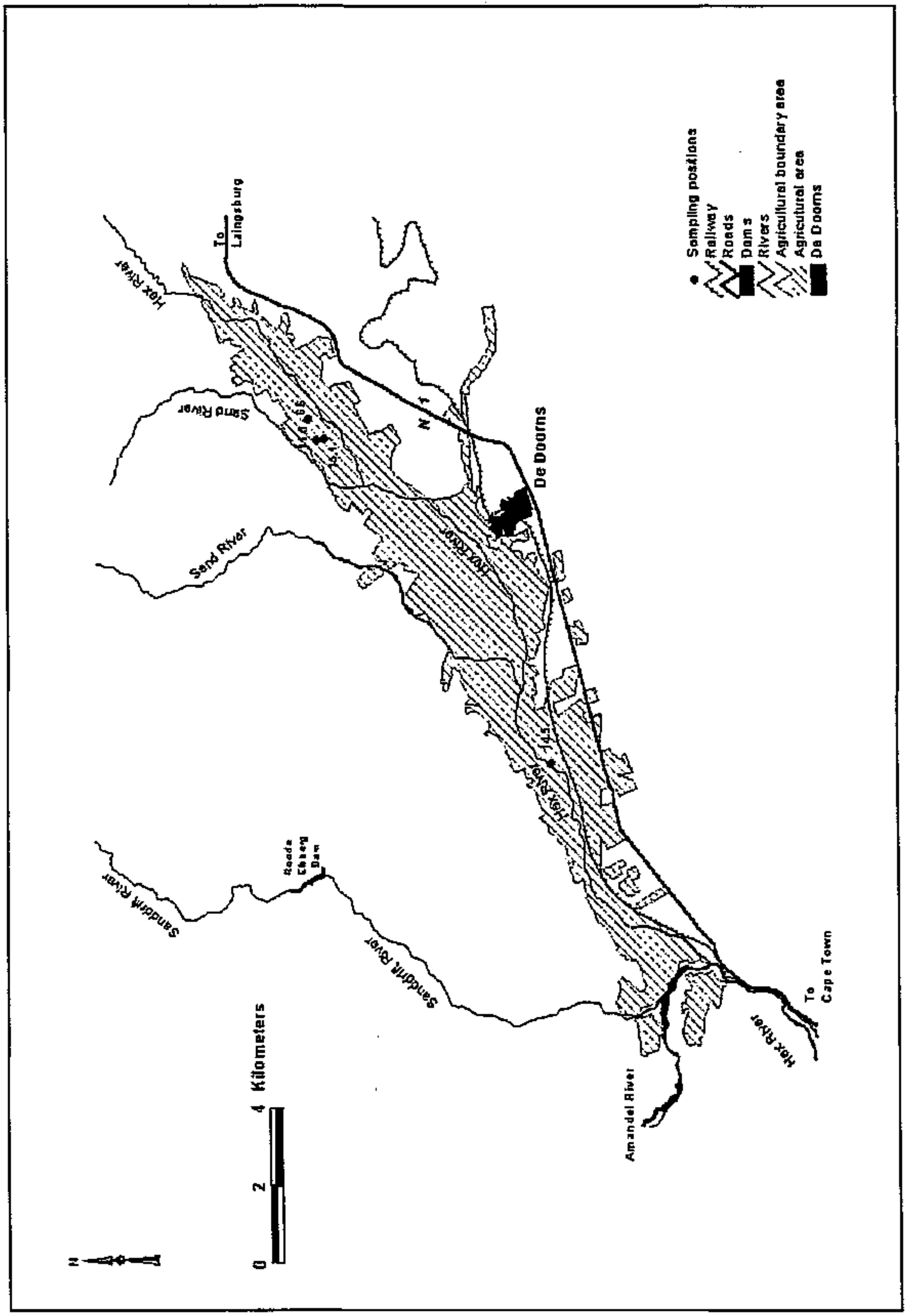


Figure 5.18 : Site A: The spatial distribution of $\delta^{15}N$ values

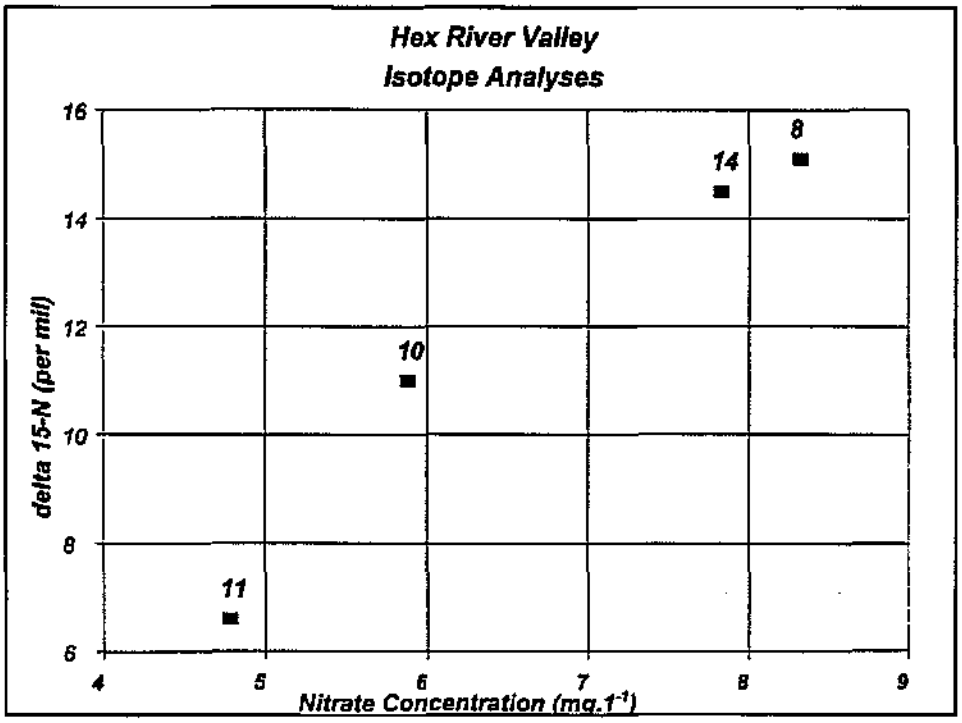


Figure 5.19 : Site A: Nitrate concentrations vs $\delta^{15}N$ values

Interpretation of the nitrogen isotope data for this study area has proved difficult, and future work is needed to assess the value of these measurements.

5.2.4.2 *Description and discussion of soil data*

Soil analyses were conducted for samples from two of the farms in the study area. Seven soil sample analyses were obtained from the farm on which sample Locality 14 is situated. Twelve soil analyses were taken from the farm on which sample location 11 is situated. Both these sets of data are listed on Table 5.9.

Table 5.9 : Soil analyses for sample Locality 14 and 11

Sample		Sunred	A1	A2	B1	B2	C	D
Depth	(cm)	0-60	0-30	30-60	0-30	30-60	0-60	0-60
Texture		sandy	loam	loam	sandy	sandy	sandy	sandy
pH		5.9	5.6	4.9	6.3	6.1	6.0	5.8
Resistivity	(ohm)	350	810	500	620	550	1140	640
P(citric acid)	(mg/kg)	71	100	138	81	75	61	50
K(citric acid)	(mg/kg)	82	82	64	154	61	33	51
Sodium	(me%)	0.59	0.45	0.71	0.32	0.43	0.21	0.36
Potassium	(me%)	0.21	0.21	0.16	0.39	0.16	0.08	0.13
Calcium	(me%)	5.10	6.23	5.46	3.39	3.51	1.78	2.86
Magnesium	(me%)	0.75	1.34	1.05	0.60	0.74	0.37	0.61
Total cations	(me%)	6.65	8.23	8.75	4.70	4.83	2.44	3.96

Recommendations					
Lime (stone)	not required	not required	not required	not required	not required
Gypsum	not required	not required	not required	not required	not required
P	not required	not required	not required	not required	not required
K (Potassium chloride)	392 kg/ha	not required	not required	100 kg	not required
N (L.A.N.)	not required	432 kg/ha	432 kg/ha	432 kg/ha	432 kg/ha

Sample		1A	3A	3E	4A	5A	7	1	4	6	9	10	12
Depth		60	60	60	60	60	60	60	60	60	60	60	60
pH	(KCl)	6.2	6.2	5.5	5.3	5.3	5.9	6.6	5.5	6.4	6.3	6.1	6.0
Resistivity(ohms)		2500	276	202	400	335	325	310	136	163	294	470	142
Clay	(%)	5	0	0	0	0	0	0	0	0	0	0	0
P	(mg/kg)	92	5	5	5	5	5	5	5	5	5	5	5
Ca	(me%)	2.27	99	138	114	131	143	112	68	93	55	23	114
Mg	(me%)	0.53	2.28	2.93	2.46	2.34	3.23	3.38	5.32	6.3	1.07	0.85	2.01
Na	(me%)	0.06	0.59	0.80	0.70	0.53	0.93	0.65	1.13	1.0	0.25	0.18	0.51
K	(mg/kg)	27	0.06	0.12	0.12	0.10	0.07	0.09	0.13	0.1	0.06	0.06	0.07
			38	35	32	51	54	35	121	84	32	18	33

From the soils analyses it can be seen that the soils are generally sandy. Although phosphorous is present in the soil horizon, it is being completely attenuated or retained within the soil horizon as no phosphorous was detected in the groundwater. The potassium concentrations within the soil are higher for the farm with sample Locality 14 (average K concentration is 82 ppm) compared with the farm with sample Locality 11 (average K concentration is 46 ppm). This is also reflected in the groundwater concentrations with Locality 14 having an average K concentration of 20 ppm and Locality 11 an average K concentration of 6 ppm. The potassium is mobile and is being transported to the saturated zone. There are unfortunately no nitrate analyses for the soil horizon.

5.2.5 Summary of impact of fertilizer application on groundwater quality, with particular reference to nitrate levels

There is a large amount of spatial variability with regard to nitrate levels in the groundwater. The groundwater quality in the south-western portion of the valley is excellent, essentially due to the quartzitic nature of the geology and derived soils, which are poorly developed. Nitrate concentrations in this area are also very low or even absent. Toward the east of the valley, the Bokkeveld Group becomes more predominant and the derived soils are thicker and have a more complex composition. The groundwater for this area has higher TDS values. This includes elevated nitrate values. Nitrate concentrations measured over a year fluctuate with the higher values corresponding with the application of fertilizer at surface. This seems to confirm that the nitrate within the groundwater is derived from fertilizer, however, the $\delta^{15}N$ values are high, which could not be explained as coming from fertilizer, but rather from natural processes. After a year of monitoring there is no net gain of nitrate within the groundwater. When comparing the recent sampling results with those collected in 1989, localities 1, 2, 3, 5, 11 and 14 have essentially remained the same in terms of EC. Localities 8, 10 and 16 have shown a significant reduction in EC values and there are no localities that show an increase in EC. With respect to nitrate, localities 1, 2, 3, 5 and 11 have concentrations that have remained constant, whilst localities 10 and 14 show a decrease in nitrate concentrations. Only locality 17 shows increasing nitrate levels. Over a period of 7 years (August 1989 to August 1996) the nitrate concentrations has increased by 2.1 $mg.l^{-1}$. There are no localities that show an increase in potassium concentration between August 1989 and August 1996.

5.2.6 Synopsis

This valley provides an ideal study site as it is completely surrounded by mountains, the geology and hydrogeology are well understood, it is intensively farmed with a single "crop" type and the farming has been going on for many decades.

The level of nitrate contamination varies in the valley. In the south-western portion of the valley contamination is absent, however, in the eastern portion, contamination is present with nitrate values being close to the Maximum Allowable Drinking water limit of 10 mg.l^{-1} (SABS, 1984). It is difficult to attribute the nitrate levels in these areas directly to the application of inorganic fertilizers. Although fluctuations in groundwater nitrate concentrations coincide with the application of fertilizers to the grapes, the nitrate nitrogen isotopic ratios do not confirm this. The ratios point more toward natural soil enrichment processes or human/animal waste. The latter is a possibility in that there is no water borne sewage connections within the Valley.

5.2.7 Conclusions

Although export table grapes have been grown for a long time in this Valley, there is little evidence from this study of fertilizers causing groundwater contamination.

5.2.8 Recommendations for longer term investigations

It would be valuable to obtain additional sampling sites within the valley to improve the spatial distribution of sampling sites. An additional study of the impact of fertilizers on groundwater quality should be conducted for a number of years and needs to be improved in terms of monitoring the unsaturated zone. Characterization of the unsaturated zone should also include studying concentration variations with depth.

5.3 SITE B: VERLORENVLEI

5.3.1 Site Background

The site, which is situated in the Sandveld 200 km north of Cape Town on the West Coast, was selected for the following reasons:

- the geology of the area is uncomplicated
- an extensive unconfined aquifer underlies the agricultural area
- there is significant groundwater abstraction from this aquifer
- the soils within the area are sandy and infertile, so fertilizer is used extensively
- there is concern from a number of organisations, including some of the farmers, about potential groundwater level lowering and deterioration due to the farming, so hydrogeological studies in the area are welcomed
- the farming practices are well established and are thought to have an impact on groundwater quantity and quality

5.3.2 Application of fertilizers to the site

The main farming activity within the area is potato farming. The Sandveld generates 30% of the country's total seed potato production. Typically there are two plantings per year. The first planting occurs in mid-February, March, or early April, although there are some farmers who plant in May and June. The potatoes need to be planted before the air and soil temperatures drop too low. It takes about 3 months before the potatoes are harvested. The second planting starts in late June, July or early August, with the potatoes being lifted in November or December. After the harvest, the land is left fallow for a minimum of four years to prevent the spread of soil borne disease.

At planting, 50 kg N is applied per hectare, mostly as granules. Phosphate (double superphosphate) and potassium are applied, as the soils are deficient in these elements. In addition, 1½ to 3 tons of gypsum is applied to increase calcium and sulphur concentrations. The fertilizer most commonly applied is 1:0:1 (NPK), or 3:0:5 if the potassium concentration is less than 30 $mg.l^{-1}$. As germination starts, 20 kg $N.ha^{-1}$ is applied per week for four weeks. The fertilizer used is typically 1:0:1 and is applied through the central pivot irrigation system by fertigation. During the following four weeks 16 kg $N.ha^{-1}$ is applied per week. For the next four weeks 12 kg $N.ha^{-1}$ is applied. About 350 $kg.ha^{-1}$ of potassium nitrate granules are added to increase potassium concentrations. Granules are used rather than fertigation as potassium does not dissolve readily. Over the entire growing season a total of approximately 240 kg N and 350 kg K per hectare are applied. After 10 to 12 weeks the potatoes are left alone for one month and then they are harvested. If seed potatoes are being planted, the ground is left fallow for four years once the potatoes have been harvested, however, if table potatoes are being grown, two plantings can occur in succession. Once the potatoes have been harvested, a cereal crop is planted to reduce wind erosion of the soil.

Potatoes are a particularly difficult crop to manage with respect to fertilizer and water. They have high N requirements, are grown on coarse-textured soils, and must not be subjected to long periods of moisture stress due to potential formation of misshapen tubers. The roots intercept only a portion of the volume in the root zone, and nitrate leached below about 35 to 45 cm is not recovered by the crop (Van Zyl, pers. Comm., 1997).

Irrigation in this area typically occurs through a central pivot system. Tests of soil moisture have been carried out using tensiometers to control irrigation rates, however, these were unsuccessful. The coarseness of the sand has caused problems and the tensiometers were not reliable. New systems are being tested to overcome these difficulties. Irrigation is essentially based on farmers judgement, and they consider it safer to over-irrigate than to under-irrigate. It is important not to apply excessive nutrients especially nitrogen to the potatoes as this can lead to poor quality potatoes and a condition called "hollow heart".

5.3.3 Land use and potential groundwater contamination within the study area

By far the largest land use activity in the area is potato farming. Very little else is grown besides potatoes. The only other potential source of contamination is from domestic sewage systems.

5.3.4 Hydrogeology of the site

Figure 5.20 shows the surface geology of Verlorenvlei catchment, with 40% of the area comprising Recent to Tertiary sands. Equal proportions of Table Mountain Group (TMG) and Malmesbury Group rocks crop out over the remaining 60% of the catchment (Maclear, 1994). The Table Mountain Group predominates in the north-western portion of the catchment, whilst the Malmesbury Group outcrops predominantly in the south-eastern portion or 'headwaters' section of the catchment.

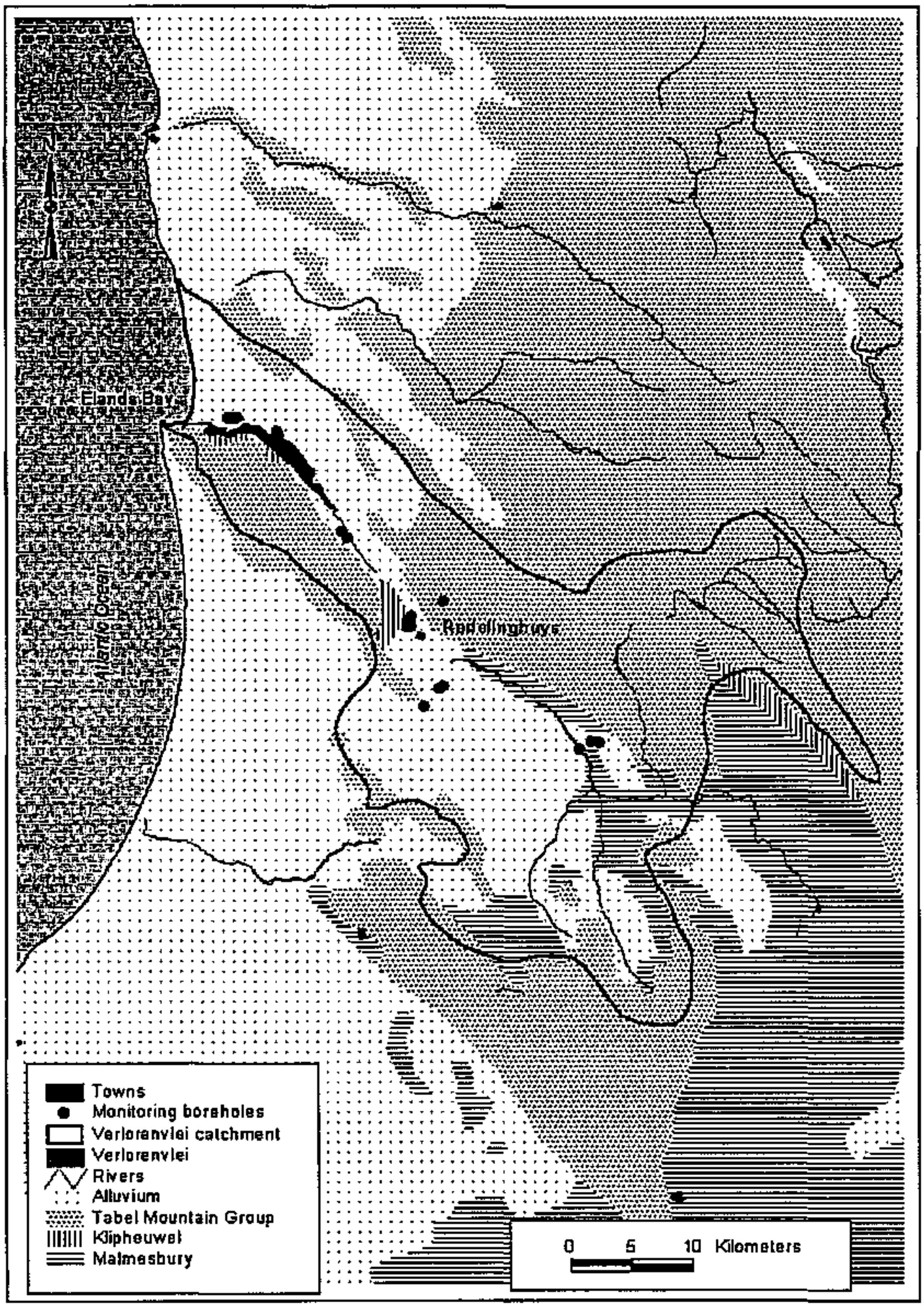


Figure 5.20 : Site B: Geology map of the Verlorenvlei catchment

Maclear (1994) reports that the area is covered by alluvial Tertiary to Quaternary sand deposits of estuarine-lagoonal facies, overlying predominantly horizontally bedded rocks of the TMG and Klipheuwel Formation.

The Graafwater and Piekenierskloof Formations of the TMG are most commonly encountered below the sand deposits. The Graafwater Formation consists of characteristically reddish, thinly bedded sandstone, siltstone, mudstone and shale. The basal Piekenierskloof Formation consists of quartzitic sandstone, with numerous thick lenticular conglomeratic bands of vein quartz pebbles.

The major outcrop in the area is a range of low hills of TMG rocks (Peninsula Formation) to the south and west of Verlorenvlei, comprising white to reddish brown, medium- to coarse-grained massively and commonly cross-bedded sandstone. The Piekenierskloof Formation outcrops as hills to the north along the Leipoldtville road.

The Cambrian Period Klipheuwel Formation, which outcrops in the Verlorenvlei area (e.g. immediately west of the vlei road crossing at Danielsdrif farm), consists of poorly bedded, purple to reddish brown sandy micaceous shales. To the south of Verlorenvlei, the basal Piekenierskloof Formation lies unconformably on the Klipheuwel Formation. A cross-section is shown in Figure 5.21.

There are two main aquifer types in the area, an upper unconfined primary sand aquifer overlying a semi-confined to confined, secondary aquifer hosted in the fractured Table Mountain Group sandstone. The primary aquifer consists of fluviatile deposits ranging from fine sand to sandy gravel, with a permeability (K) of 20 m/day, a specific yield of 0.17 and an average saturated thickness of 15 m. Figure 5.22 shows the geological profiles at 2 boreholes, G44064 and G44065, drilled just to the north of Verlorenvlei.

The generalized groundwater flow direction for the study area is believed to conform closely to the slope of the surface topography. Collar elevations of sampling localities were not measured, due to the costs associated with the accurate measurements required for meaningful values. The groundwater gradient is generally towards the west. The location of the sampling points in relation to the irrigation circles is shown in Figure 5.23.

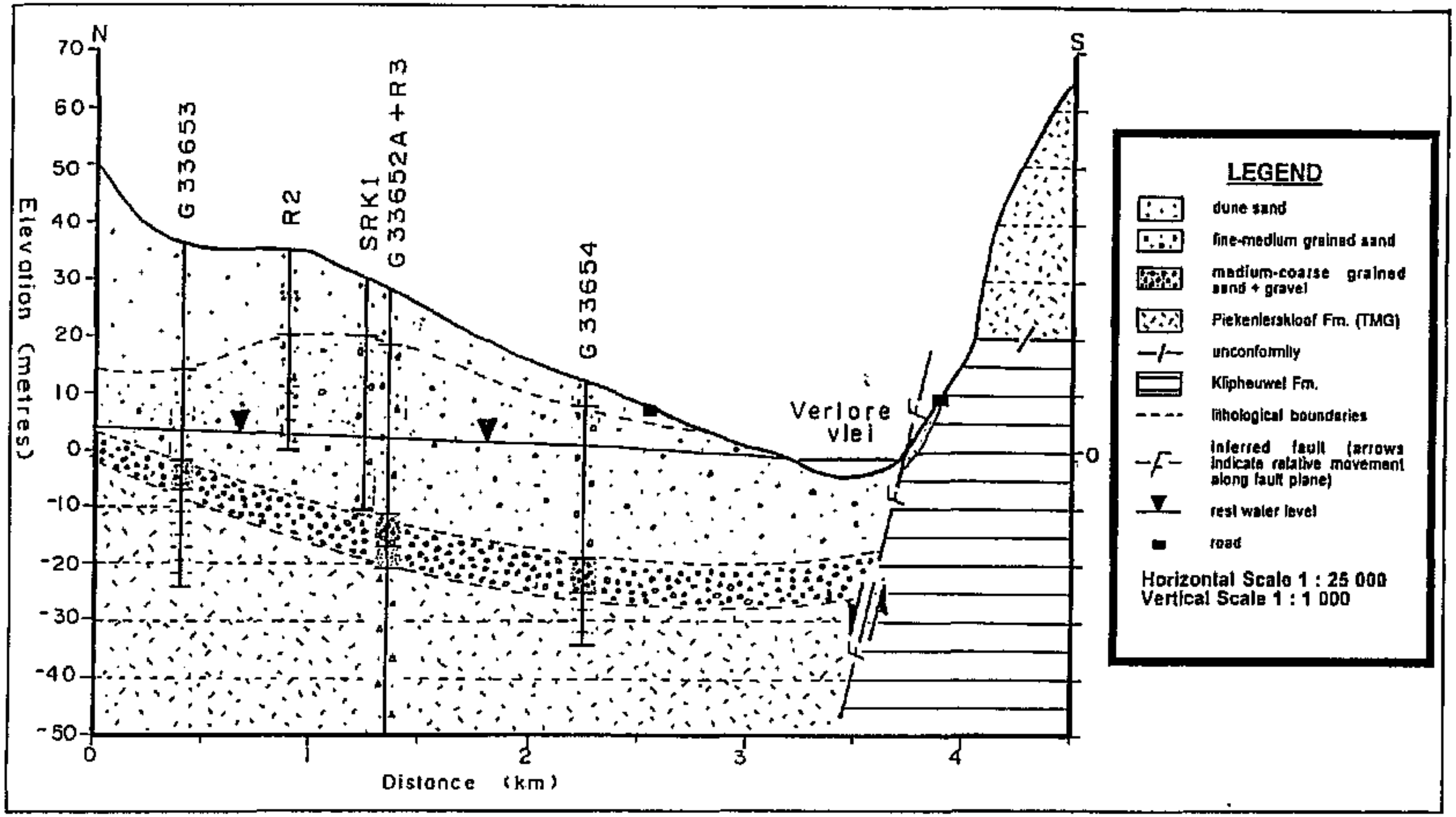


Figure 5.21 : Site B: Geological cross-section *(Maclear, 1994)*

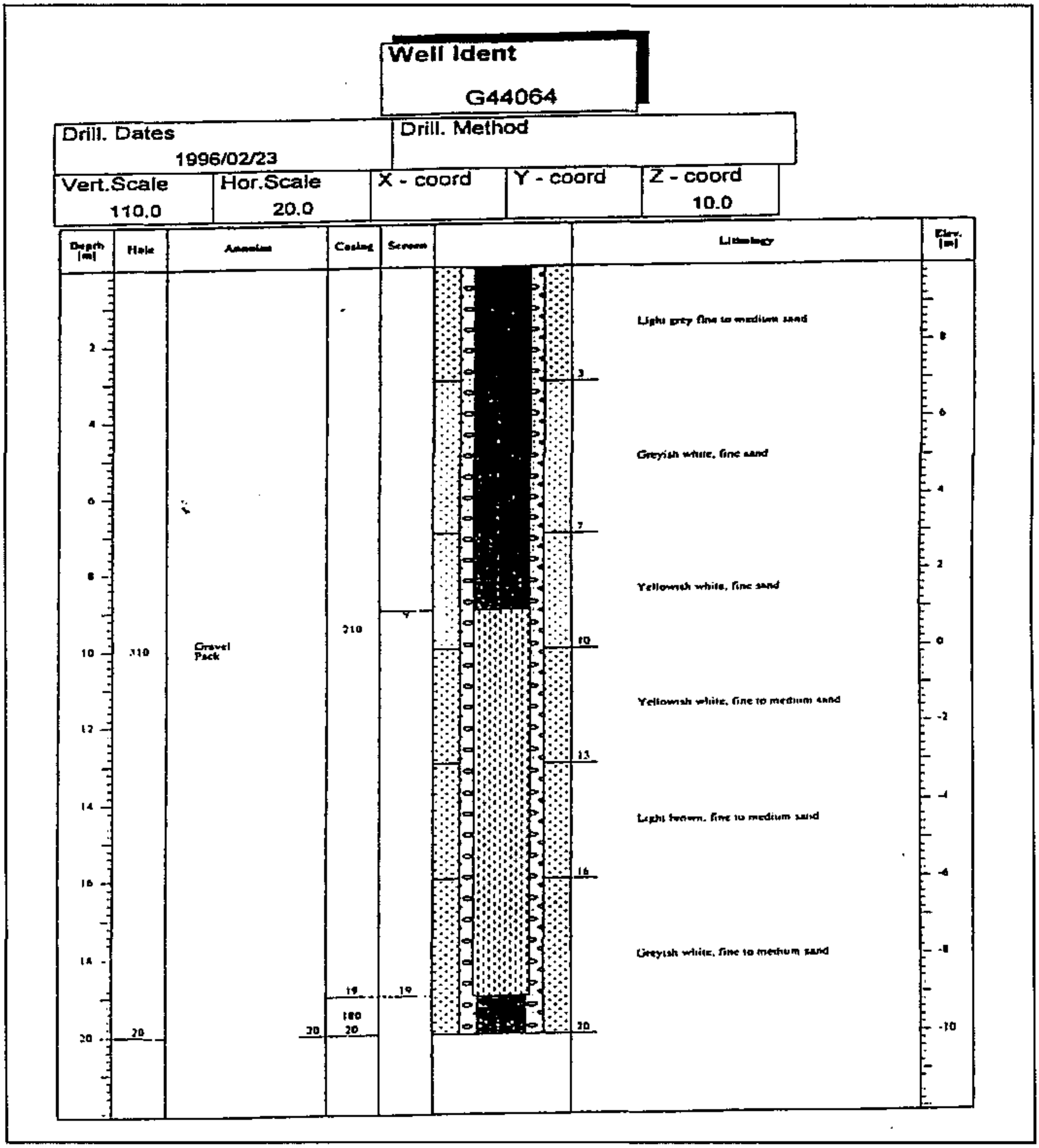


Figure 5.22 : Site B: Geological profile of boreholes G44064 and G44065

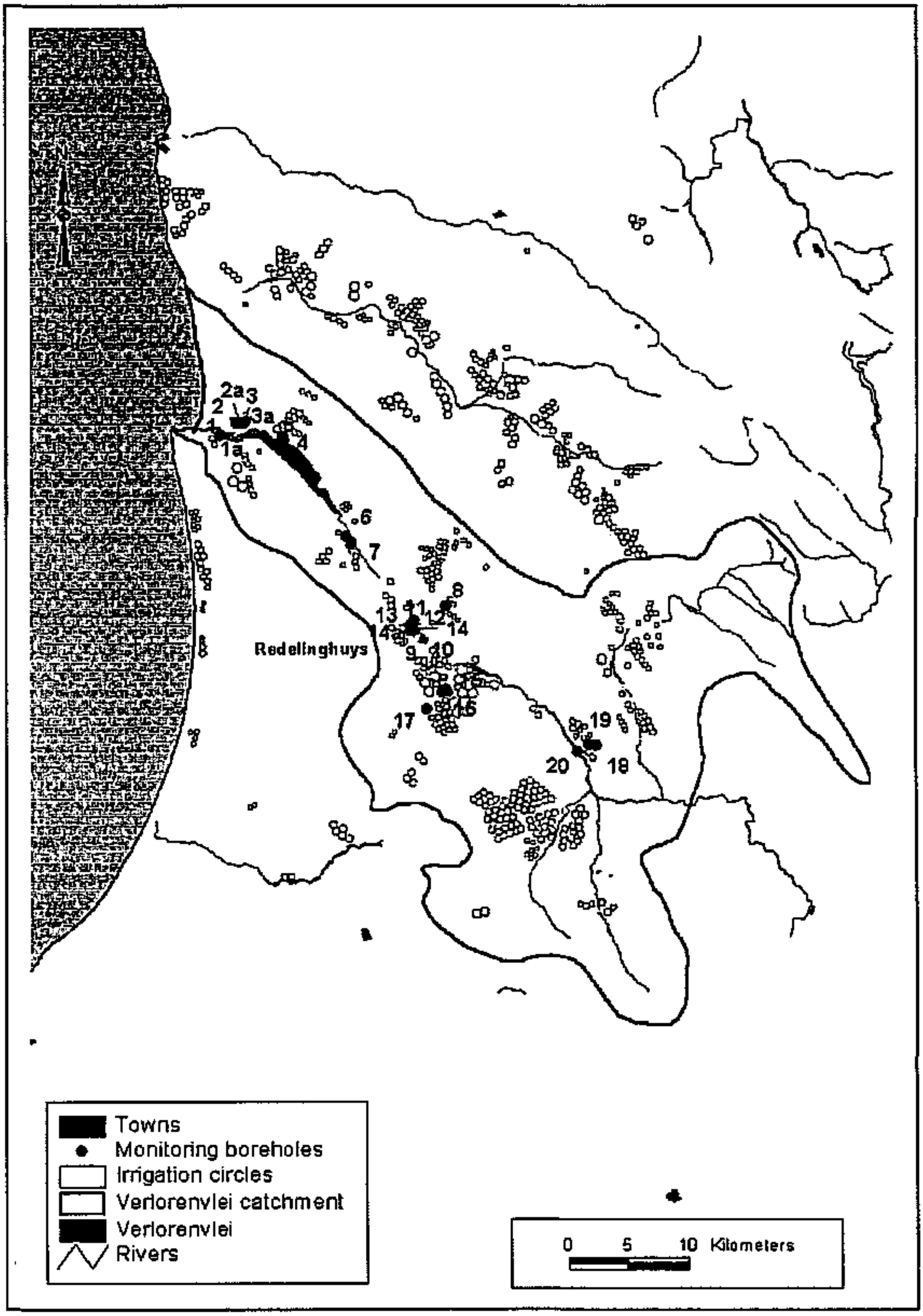


Figure 5.23 : Site B: Sampling localities and irrigation circles.

Recharge to the primary aquifer is from direct precipitation and runoff off the hill slopes to the north and south of the Verlorenvlei river. Figure 5.24 shows the amount of rainfall and evaporation for the duration of the study period.

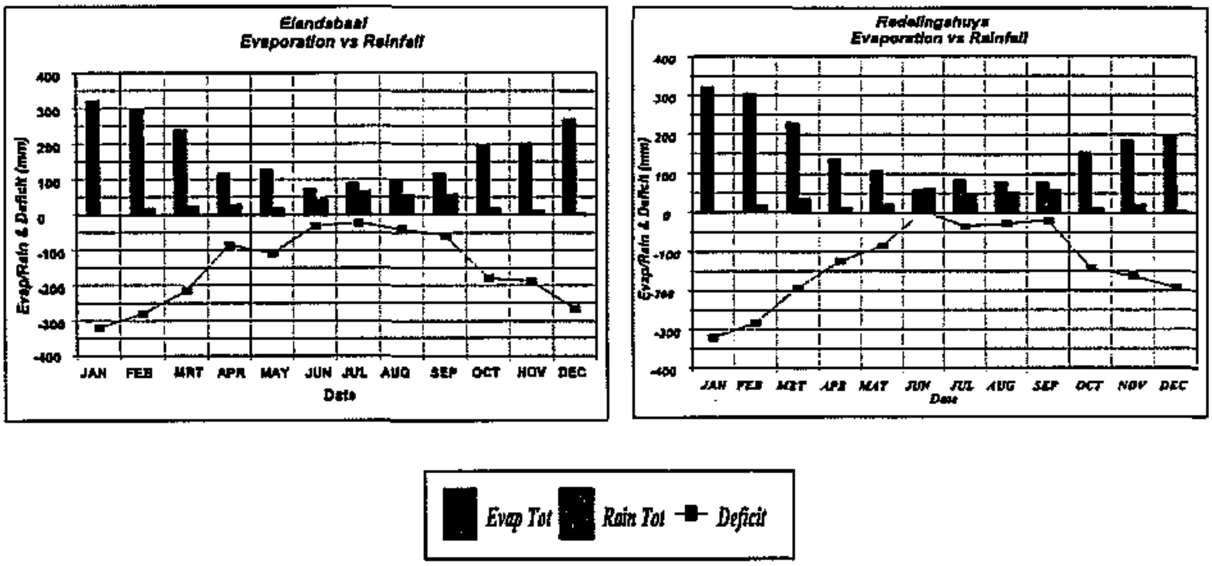


Figure 5.24 : Site B: Climatic data

The Verlorenvlei River is the main river that drains the catchment and its flow is very low during the dry summer months. However, in winter the river flows strongly and after large storm events can burst its banks and cover the flood plain. The river flows into the Verlorenvlei, which is located at the coast. The Verlorenvlei mouth is usually closed by a sandbar formed by wave action. The vlei never dries out and in wet winter months flows into the sea. Surface and groundwater in the study area are in hydraulic continuity.

This study focussed on assessing the impact of fertilizers on groundwater quality and only the most vulnerable primary aquifer was studied in detail. A single secondary aquifer borehole was, however, monitored.

The Verlorenvlei groundwater is sodium-chloride type, (Maclear, 1994) and the TDS values generally fall in the 2000 - 3000 mg.l^{-1} range, classifying the water as brackish.

5.3.5 Field investigation

The field investigation was started in April 1996 and completed in March 1997. Prior to the actual sampling of boreholes, a number of meetings were held with representatives from the Department of Agriculture, Kynoch and Department of Water Affairs and Forestry. A questionnaire was also sent out to all farmers to gauge their level of interest in cooperating with this project. Based on the survey response, farmers were then visited by members of the project team, along with representatives from the Department of Agriculture (including the extension officer) and the Department of Water Affairs and Forestry.

A total of six sampling runs in the area were undertaken on: 11/12 April 1996, 25/26 June 1996, 12/13 September 1996, 28/29 November 1996, 3/4 February 1997 and 24/25 March 1997. The sampling runs included water level measurement, purging of the borehole if it was not being pumped, measurement of temperature, electrical conductivity and pH, and collection of water samples.

The samples were routinely analysed for K, NO_3, P and EC. A full analysis of major cations and anions was carried out on the samples collected during the June 1996 sampling run. Isotope analysis ($\delta^{15}N$) was carried out on a number of samples collected during the November 1996 field trip. A total of 24 boreholes were identified as being useful for the project and 19 of these were sampled, including one borehole drilled into the fractured bedrock (borehole 17). Microbiological analyses were not carried out on the samples, as these were considered irrelevant to a study of the impact of inorganic fertilizers on groundwater.

5.3.5.1 Description and discussion of hydrochemical data

A piper diagram was prepared using the hydrochemical data from June 1996 (Figure 5.25). All the water samples plot in the same sectors of the diagram with the exception of borehole 17 which intercepts a different aquifer in the fractured bedrock. The dominant cation is sodium and the dominant anion chloride except for borehole 17 where CO_3 + HCO_3 also contribute significantly to the anion concentrations. In the diamond section of the diagram, the samples all plot in the field where alkalis and strong acids dominate the water chemistry.

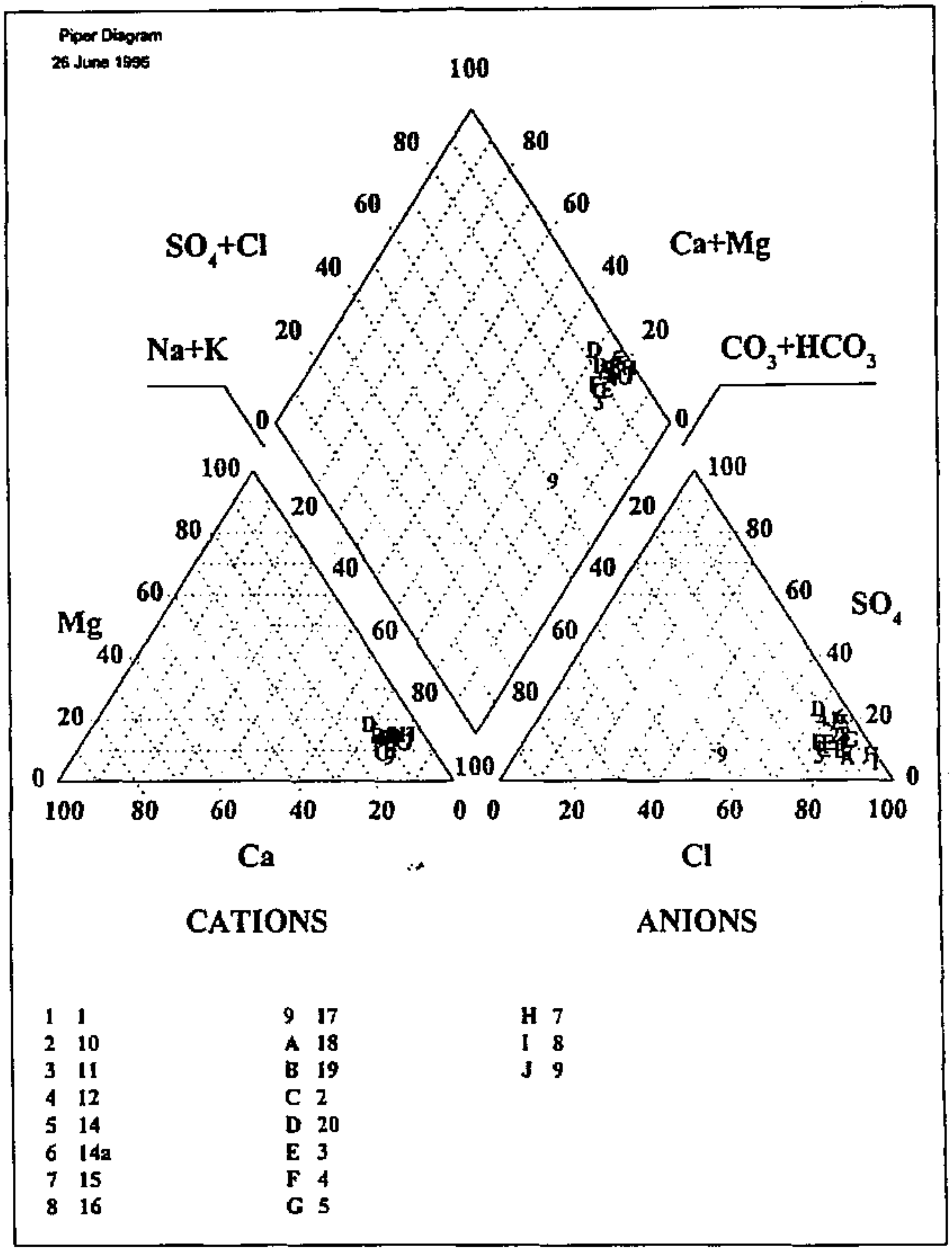


Figure 5.25 : Site B: Piper diagram of hydrochemical data (26 June 1996)

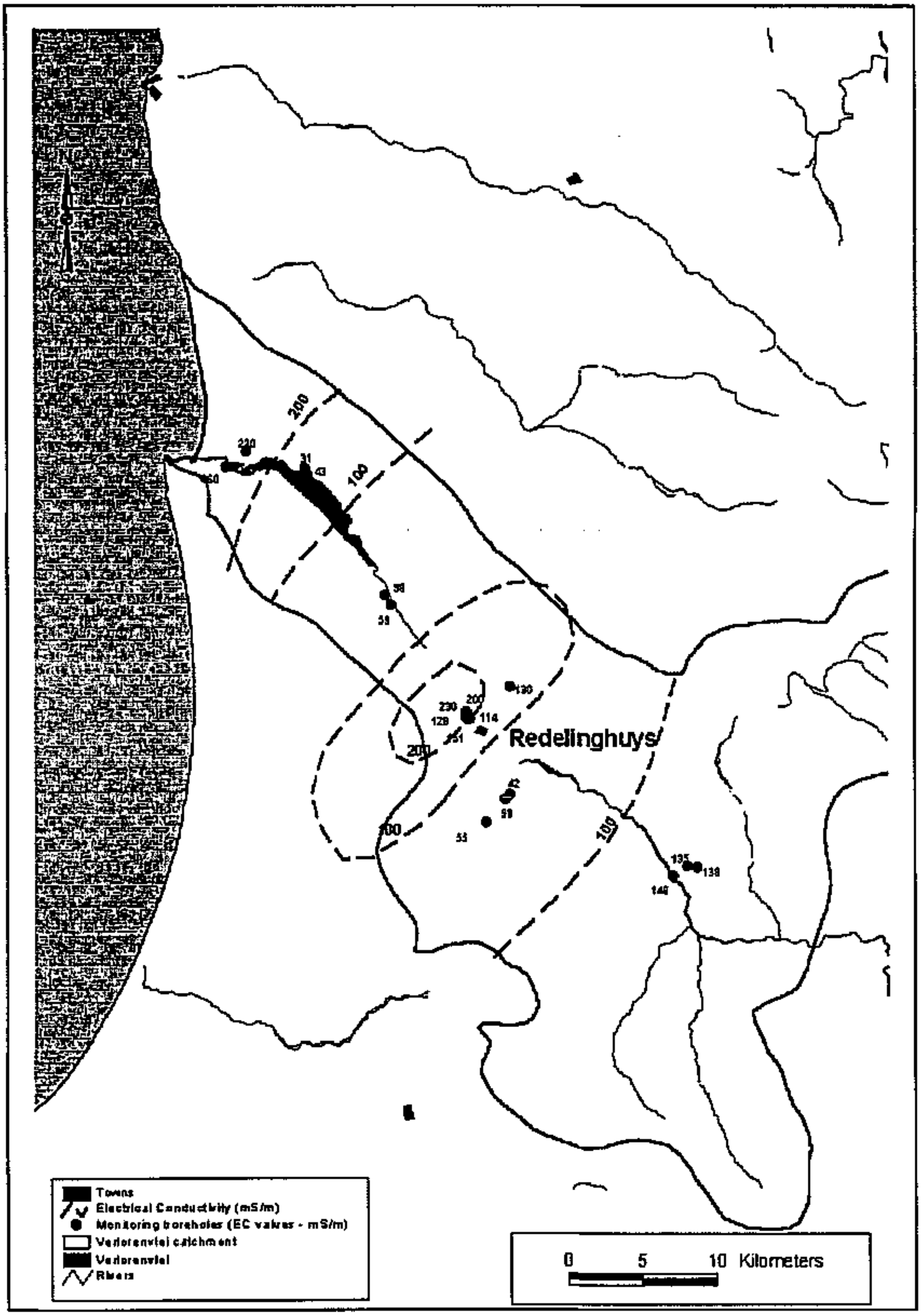


Figure 5.26 : Site B: Spatial distribution of Electrical Conductivity values (mS.m^{-1}) (25 March 1997)

Electrical Conductivity

The spatial distribution of EC values is shown in Figure 5.26. The data plotted is from the March 1997 sampling run. Values range from 31 $mS.m^{-1}$ to 840 $mS.m^{-1}$. The well (borehole 1) with the maximum value is situated on the southern side of the Verlorenvlei estuary and the water is used for domestic use even though the farmer complains that it tastes like "Epsom salts". This water has a very different chemistry from the other samples. Its composition could be linked to the interbedded cretaceous and argillaceous horizons of the Klipheuwel Formation which outcrop in the area. The proximity of the vlei and Klipheuwel Formation rocks to this well, suggests that the groundwater abstracted from the well does not experience the same influences as the water abstracted from the other wells in this study. This well will thus be excluded from further discussion.

The well with groundwater having an EC of 31 $mS.m^{-1}$ (borehole 5) has not been constructed properly and is too shallow to fulfil the objectives of the current study. The well does not intercept the sandy gravel horizon which comprises the aquifer in the area. When it is pumped the water level drops by 8 m and slowly recovers when pumping stops. The groundwater abstraction from this borehole is essentially water most probably derived from vertical recharge, as lateral flow is considered to be very low. This is unfortunate as the borehole is situated in a low lying valley downstream from a very large, potato growing area and would have been ideal for study if it were not for the abovementioned problems.

The sampling point drilled into the bedrock (borehole 17) yields water typical of that from Table Mountain Group aquifers, with an EC of 58 $mS.m^{-1}$. In most cases, there appears to be a relationship between EC values and the underlying geology beneath the alluvium. The groundwater from boreholes that are drilled into alluvium overlying Malmesbury shales has an average EC of 133 $mS.m^{-1}$. Groundwater from alluvium overlying the Table Mountain Group has an average EC of 65 $mS.m^{-1}$ and groundwater in proximity to Klipheuwel Formation rocks has an average EC of 264 $mS.m^{-1}$. Very generalized EC contour lines have been drawn on Figure 5.26 to indicate the variability of EC values within the study area.

An EC profile was determined for two of the boreholes in the study (G44064 and G44065) by lowering an electrical conductivity probe down the borehole and measuring the change in EC with depth. The profiles are plotted in Figure 5.27. It appears that there is very little increase in salinity with increasing depth.

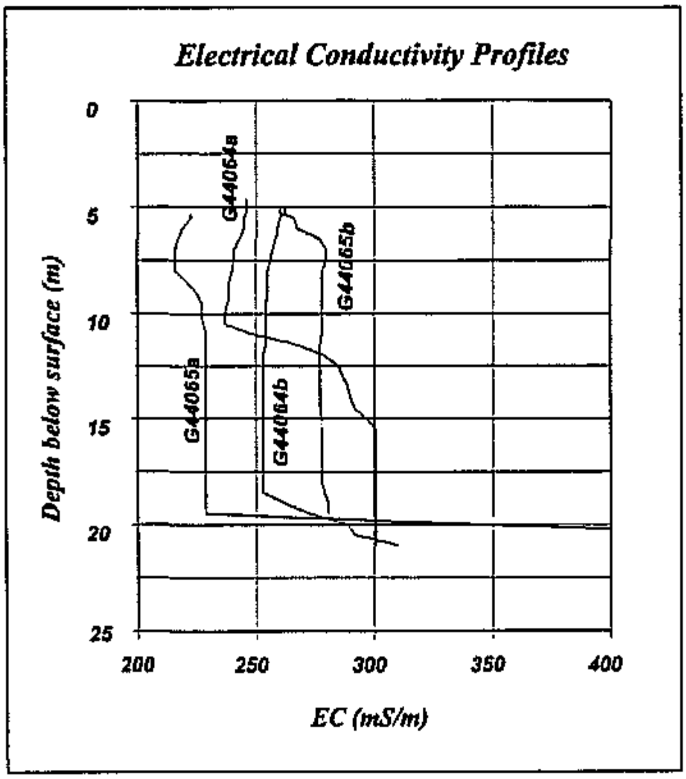


Figure 5.27 : **Site B: Electrical conductivity profiles for G44064 and G44065.**

On inspection of the temporal variation of EC values for all relevant sampling points (Figure 5.28), there is no general pattern of conductivity fluctuations that applies to all samples. This is not surprising due to the large spatial and temporal variability of factors which affect water quality within the study area. Spatial variability, in this case, arises from the differences in proximity of the individual sampling points to active cultivation areas which impact on the water quality. There is also the added complication of temporal variability, in that the groundwater samples may be influenced by nearby cultivation practices which took place any time between 6 months and 4 years previously. A sound knowledge of the field conditions at each sampling point, and when the irrigation circles were active, is therefore necessary before evaluation of any possible EC trends can be attempted. A more thorough description of the temporal trends for each borehole appears in the section below.

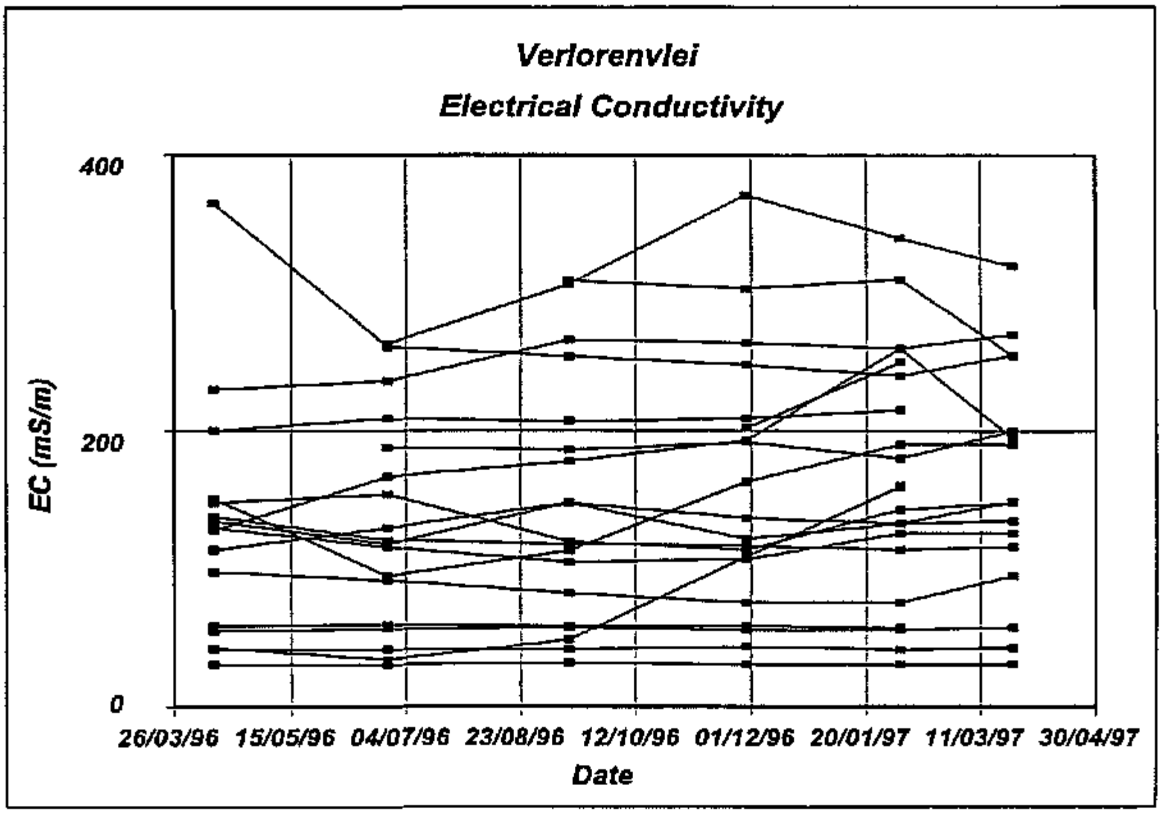


Figure 5.28 : Site B: EC values for a year of sampling

Nitrate, potassium and TDS concentrations

Figure 5.29 shows the spatial distribution of nitrate concentrations. The bedrock borehole shows a nitrate concentration below detection limits (<0.1 mg.l^{-1}). The average nitrate concentration for groundwater abstracted in areas away from housing and septic tank systems is 4.2 mg.l^{-1}. For the cluster of sampling positions near Redelinghuys, there are a number of houses nearby and the higher nitrate values (8 mg.l^{-1} to 23.7 mg.l^{-1}) may be attributable to mixing of groundwater with septic system flow. There is a well and a borehole in close proximity to the Verlorenvlei River, (near Redelinghuys) with nitrate values below detection limits, indicating that nitrate levels in the river itself are probably negligible.

As is the case with EC, there is no clear pattern of nitrate level fluctuations with time (Figure 5.30). Arguably, there may be a slight increase in nitrate levels for most boreholes in late November. This would tie in with the closing of the second annual planting, after which fertilizer would have been applied for at least 12 weeks.

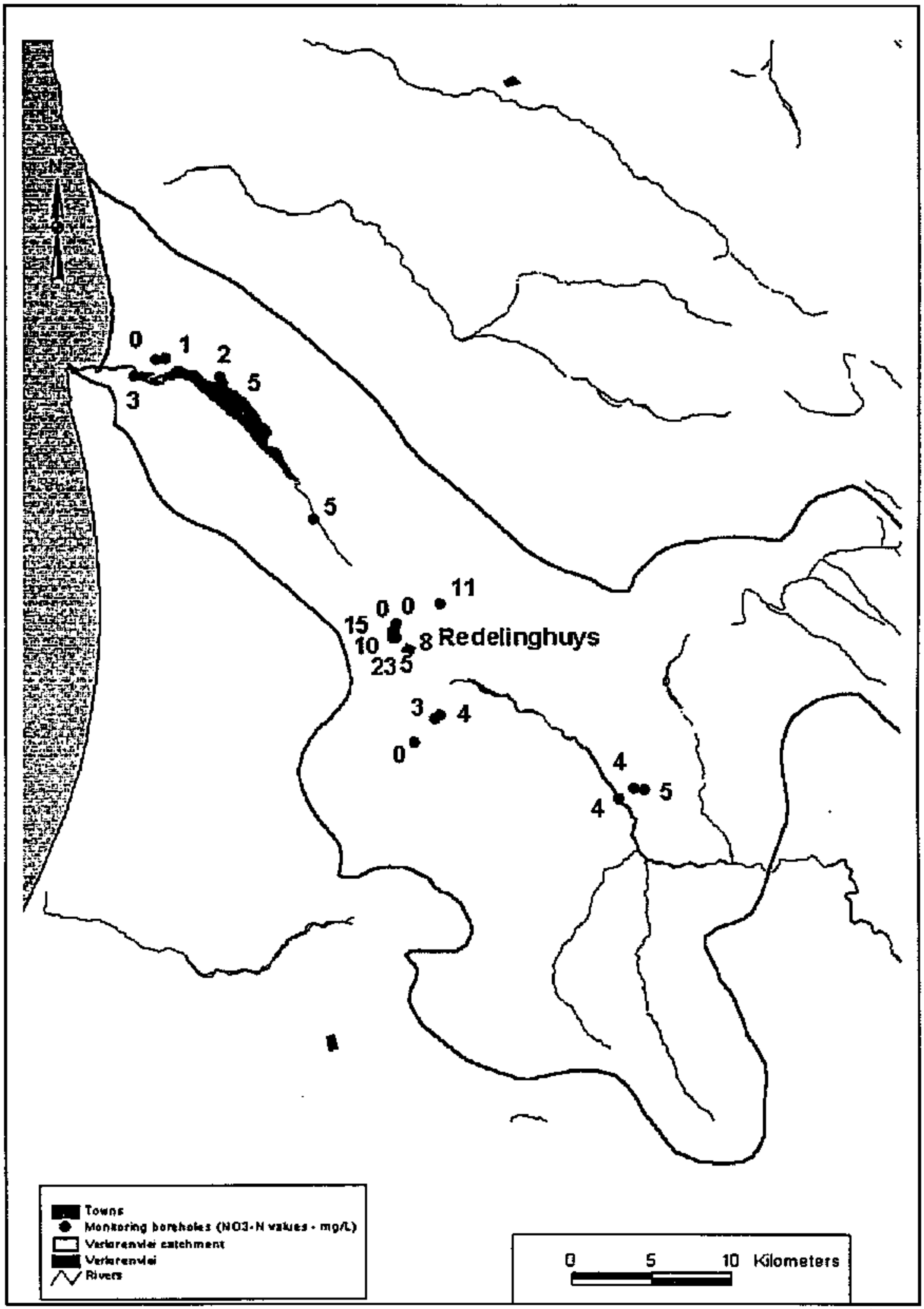


Figure 5.29 : Site B: Spatial distribution of (NO_3-N) nitrate concentrations (mg.l^{-1}) (25 March 1997)

Locality 7 is situated down slope from a large cultivation area which was last planted and fertilized in 1993 with no farming activity since. The nitrate concentration at this locality has remained constant at 5 $mg.l^{-1}$ for the entire year. Unlike the boreholes at locality 2 and 3, the nitrate from the earlier cultivation has not flushed out of the area, but the constant elevated concentration reflects an equilibrium value for nitrate in the soil-water system.

Locality 8 is below a cultivation region, yet no active farming directly up slope and adjacent to the borehole took place during the study period. There were, however, active pivots within the region during this period. The nitrate levels have fluctuated during the year of study, although it is difficult to attribute this with any confidence to the impact of fertilizer application. The potassium levels have remained constant for the entire period. It would be worthwhile to determine the extent of the capture zone of the borehole, to see if this extends far enough to capture groundwater that may have been affected by nearby cultivation practices. TDS trends conform to those of potassium, yet there appears to be an inverse relationship between nitrate and TDS trends.

Localities 9 and 10 are in close proximity to each other and display similar nitrate trends. There are small fluctuations in the nitrate levels, however, these are unlikely to be related to fertilizer application practices since the boreholes are a long way from any active cultivation areas. They are also situated close to the Verlorenvlei River floodplain. Interestingly, the TDS and potassium concentration trends for the two boreholes differ.

Localities 11 and 12 can be grouped together, as both occur on the Verlorenvlei River floodplain. Locality 11 is adjacent to the river channel. Nitrate concentrations are below detection limits and TDS and K concentrations remained constant for both localities throughout the year of sampling.

Localities 13, 14 and 14a can be considered as a group. Although not far from Localities 11 and 12, they are close to farm houses and labourers' cottages which could impact on the water quality. The nitrate concentrations are between 5 $mg.l^{-1}$ and 25 $mg.l^{-1}$ and fluctuate over the year of sampling. The source of this nitrate may well be from septic tank flow. Potassium levels show similar fluctuations to TDS, however, as also seen for some of the previous boreholes, nitrate trends are the inverse of TDS trends.

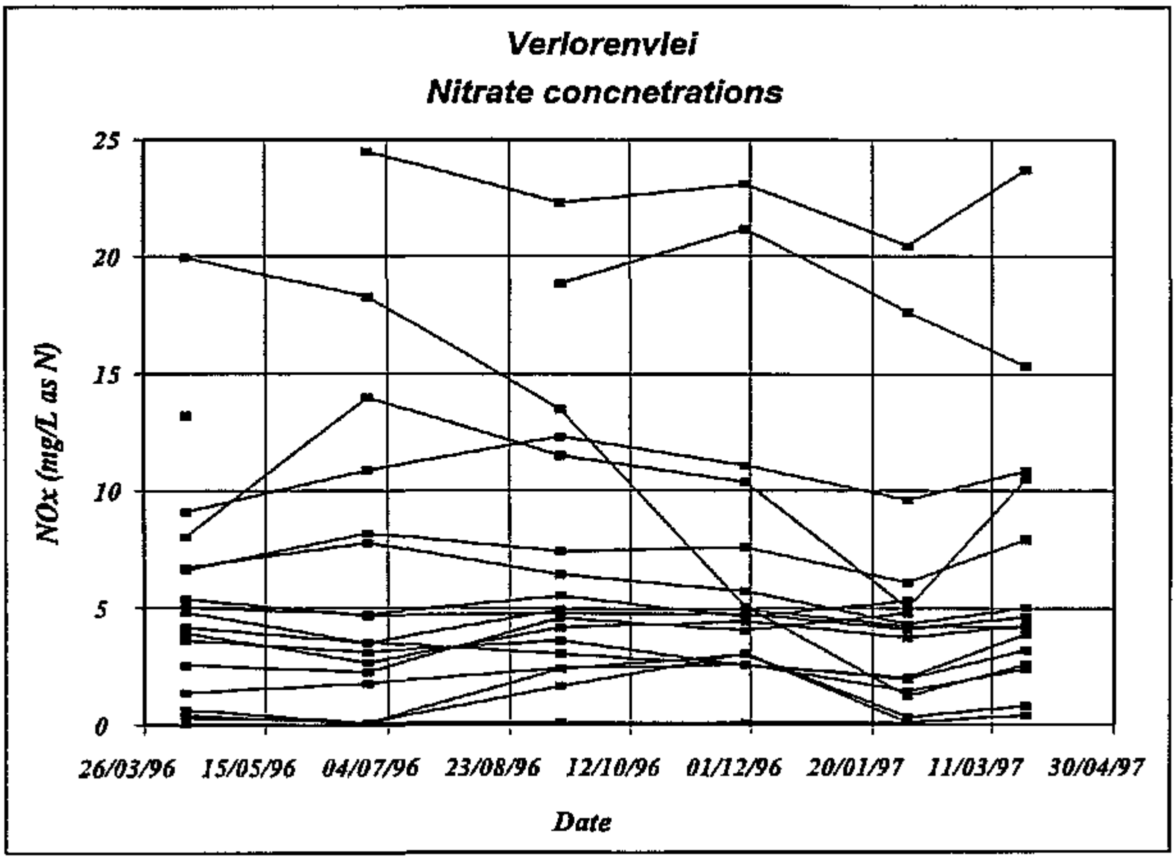


Figure 5.30: Site B: NO_3-N concentrations for a year of sampling

For samples collected from **localities 2 and 3**, which are 400 m apart there is a definite increase in nitrate levels for both samples which corresponds very well with the cultivation periods. The boreholes are located in close proximity to, and down slope from, a cultivation area being irrigated with a central pivot. The potato growing season for this area extended from August to December 1996. The nitrate levels increased by 3 $mg.l^{-1}$ over the season, but values returned to the original concentrations after the cultivation was completed. Potassium concentrations show a similar trend to nitrate. Water from these boreholes does have a slight H_2S odour.

For **locality** 4, which is in close proximity to the vlei, and at least 500 m down slope of a large area of active cultivation, there is a steady increase in nitrate levels. Over a period of a year the levels have increased by 2.2 $mg.l^{-1}$. Potassium and TDS levels show an even more significant increase toward the end of the monitoring period than that of the nitrate concentration.

Localities 15 and 16 are in close proximity (300 m) and have low nitrate concentrations (± 3 $mg.l^{-1}$). There is very little variation in these concentrations with time. The boreholes are located in a river bed, which is a non-perennial tributary of the Verlorenvlei River, and there was no active cultivation in the immediate vicinity of the boreholes, during the study period. However, there was significant farming activity higher up the tributary valley. Five hundred metres up gradient from the boreholes there are farm labourers' cottages, which may be contributing to the nitrate concentrations via their septic systems.

Locality 17 is the bedrock borehole, which is 94 m deep and was drilled through 45 m of sand into a maroon coloured weathered and fractured sandstone. At 94 m the sandstone becomes massive and unfractured. There is no nitrate in this groundwater and the potassium concentration is 5 $mg.l^{-1}$. There is no variation in groundwater quality over the year of monitoring.

Localities 18, 19 and 20 comprise the final group. The boreholes are situated at the base of lengthy slopes and down gradient from farming activities, although they are not in close proximity to cultivation areas. Nitrate concentrations are approximately 4.5 $mg.l^{-1}$ and show very little variation in concentration throughout the study period. The potassium levels are approximately 4 $mg.l^{-1}$ and show greater fluctuation that nitrate. Trends for TDS are very similar to those for potassium. The presence of nitrate in the samples is most likely due to fertilizer application. The lack of temporal variation in nitrate concentrations may well be due to distance effects, since the boreholes are some distance from active cultivation areas.

Table 5.10 summarizes the changes in nitrate concentration over the year of monitoring, as well as average values and standard deviations.

Table 5.10 : **Change in nitrate concentration for a year of monitoring.**

Locality	NO_3-N (mg.l^{-1}) 12/4/96	NO_3-N (mg.l^{-1}) 25/3/97	Difference (mg.l^{-1})	Yearly average (mg.l^{-1})	Std deviation (mg.l^{-1})
1	19.9	2.6	-17.3	10.1	7.5
2*	0.7	0.4	-0.3	1.1	1.2
3*	0.3	0.8	+0.5	1.0	1.0
4	2.6	4.8	+2.2	3.0	1.7
5*	1.4	2.4	+1.0	2.0	0.5
7*#	5.0	5.3	+0.3	4.2	1.9
8*#	9.1	10.9	+1.8	10.7	1.0
9	6.7	5.0	-1.7	6.0	1.1
10	6.6	8.0	+1.4	7.3	0.7
11*	0.1	0.1	0.0	0.0	0.0
12*	0.4	0.1	-0.3	0.2	0.1
13	13.3	15.4	+2.1	17.3	2.8
14	8.1	10.5	+2.4	9.9	2.8
14a	24.5	23.7	-0.8	22.8	1.4
15*	3.6	3.9	+0.3	3.1	0.7
16*#	4.2	3.2	-1.0	3.1	0.7
17	0.1	0.1	0	0.0	0.0
18*	5.4	4.6	-0.8	4.7	0.4
19*	3.9	4.3	+0.4	3.9	0.6
20	4.8	4.2	-0.6	4.4	0.5
Averages:		All samples	-0.5	5.7	
		* Agricultural impact only	+0.2	3.1	

* *only agricultural impacts, no other anthropogenic nitrate sources*

\# *isotope data supports fertilizer as the only nitrate source*

For the entire study area, including locality 1, the average NO_3-N concentration is 5.7 $mg.l^{-1}$. There was an average decrease of 0.5 $mg.l^{-1}$ NO_3-N for the monitoring period 12 April 1996 to 25 March 1997. For boreholes that are only thought to be affected by agricultural practices, where there are no other anthropogenic nitrate sources in close proximity to the boreholes, the average NO_3-N concentration was 3.1 $mg.l^{-1}$. Over the monitoring period the NO_3-N concentration increased 0.2 $mg.l^{-1}$ for these boreholes.

The average concentration of potassium for the entire study area over the monitoring period is 12.5 $mg.l^{-1}$, with an average decrease of 8.5 $mg.l^{-1}$ for the year of monitoring. The average concentration for areas where agriculture is the only anthropogenic activity is lower at 4.5 $mg.l^{-1}$, but there is an increase in potassium levels for these areas over the year of monitoring. All the potassium concentrations are, however, far lower than the recommended maximum limit of 200 $mg.l^{-1}$ proposed in the Drinking Water Guidelines.

The two localities which show the greatest increase in potassium concentrations are localities 3 and 4 which increased by 5.2 and 4.4 $mg.l^{-1}$ respectively. Locality 3 is in close proximity to areas of agricultural activity. The potassium trends mirror those of EC for most of the sampling points.

Table 5.11 : Change in potassium concentration for a year of monitoring.

Locality	K (mg.l^{-1}) 12/4/96	K (mg.l^{-1}) 25/3/97	Difference (mg.l^{-1})	Yearly average (mg.l^{-1})	Std. deviation (mg.l^{-1})
1	147	121	-26.1	125.5	22.1
2*	7.6	7.5	-0.1	8.4	0.9
3*	2.8	8.0	5.2	6.8	1.9
4	2.2	6.6	4.4	3.5	1.8
5*	2.0	1.9	-0.1	1.9	0.1
7*	1.4	1.4	0.0	1.4	0.1
8*	3.3	3.0	-0.3	3.0	0.2
9	9.3	10.6	1.3	8.8	1.3
10	9.5	10.3	0.8	10.1	0.8
11*	4.8	5.0	0.2	4.6	0.3
12*	8.4	11.0	2.6	9.3	1.0
13	10.4	11.2	0.8	12.0	1.4
14	6.1	8.5	2.4	8.1	1.7
14a	11.2	13.4	2.2	13.8	1.8
15*	1.3	1.3	0.1	1.2	0.1
16*	5.8	5.6	-0.2	4.7	0.8
17	4.5	4.7	0.2	4.5	0.1
18*	4.1	3.4	-0.7	3.7	0.2
19*	4.4	3.9	-.05	4.2	0.2
20	8.3	7.6	-0.7	5.8	2.0
Averages:		All samples	-8.5	12.4	
		* Agricultural impact only	6.2	4.5	

* *only agricultural impacts, no other anthropogenic nitrate sources*

With the irrigation circles having been carefully mapped by the Department of Agriculture, it was possible using GIS analysis to determine the distance from the sampling boreholes to the nearest irrigation circle. These distances were then correlated to an annual average nitrate concentration for the sampling points. There is no correlation between nitrate concentration and distance from irrigation circles to sampling localities.

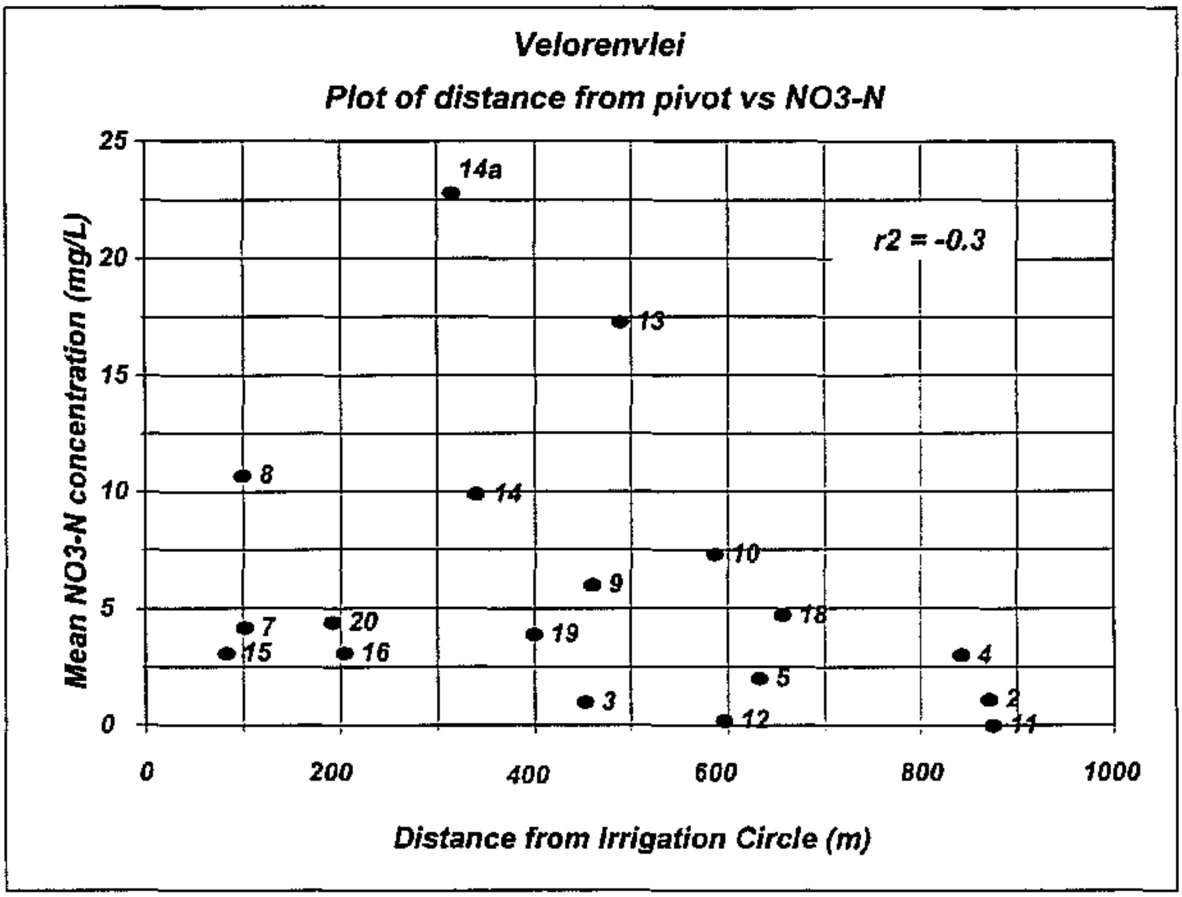


Figure 5.32 : Site B: Graph of nitrate concentration versus the distance between irrigation circle and sampling location.

A total of eleven samples were collected for $\delta^{15}N$ isotope analysis during the 29 November 1996 sampling run and the results are shown in Table 5.12.

]

Table 5.12 : δ^{15}N isotope values

Locality No	N^{15} value (%)	NO$_3$-N (mg.l^{-1})
1A	+26.2	5.07
3	+11.9	3.06
4	+8.7	4.59
7*	+4.80	5.52
8*	+5.49	11.1
10	+8.59	7.60
14	+5.48	10.4
13	+6.3	23.1
14A	+5.75	2.57
16*	+3.58	4.42
19	+5.72	

* *Nitrate sources natural or fertilizer derived, no other anthropogenic activities present.*

The spatial distribution of δ^{15}N isotope values is plotted on Figure 5.33.

It is generally accepted that 0 to +4% is fertilizer - , +4 to +8 'natural' nitrate - and >+8 is sewage-derived, although these boundaries are not exact (Heaton, 1986).

According to the isotope data only localities 7, 8 and 16 are in areas that are solely influenced by natural or fertilizer applied nitrate. Considering 7, 8 and 16 the maximum δ^{15}N values ascribable to fertilizer application are then +4.80‰, +5.49‰ and +3.58‰ respectively. All the other samples may have been influenced by other anthropogenic activities. If the nitrate concentrations in the groundwater for these localities (7, 8 and 16) are accepted as being fertilizer-derived, as supported by field evidence and isotope values, then the yearly average NO$_3$-N concentration is 6.0 mg.l^{-1} and over the monitoring period there has been a 1.1 mg.l^{-1} increase in nitrate concentration.

The relationship between nitrate concentration and δ^{15}N value has been plotted (Figures 15 - 34). The anticipated strongly negative correlation is not present and a correlation coefficient of -0.2 was obtained.

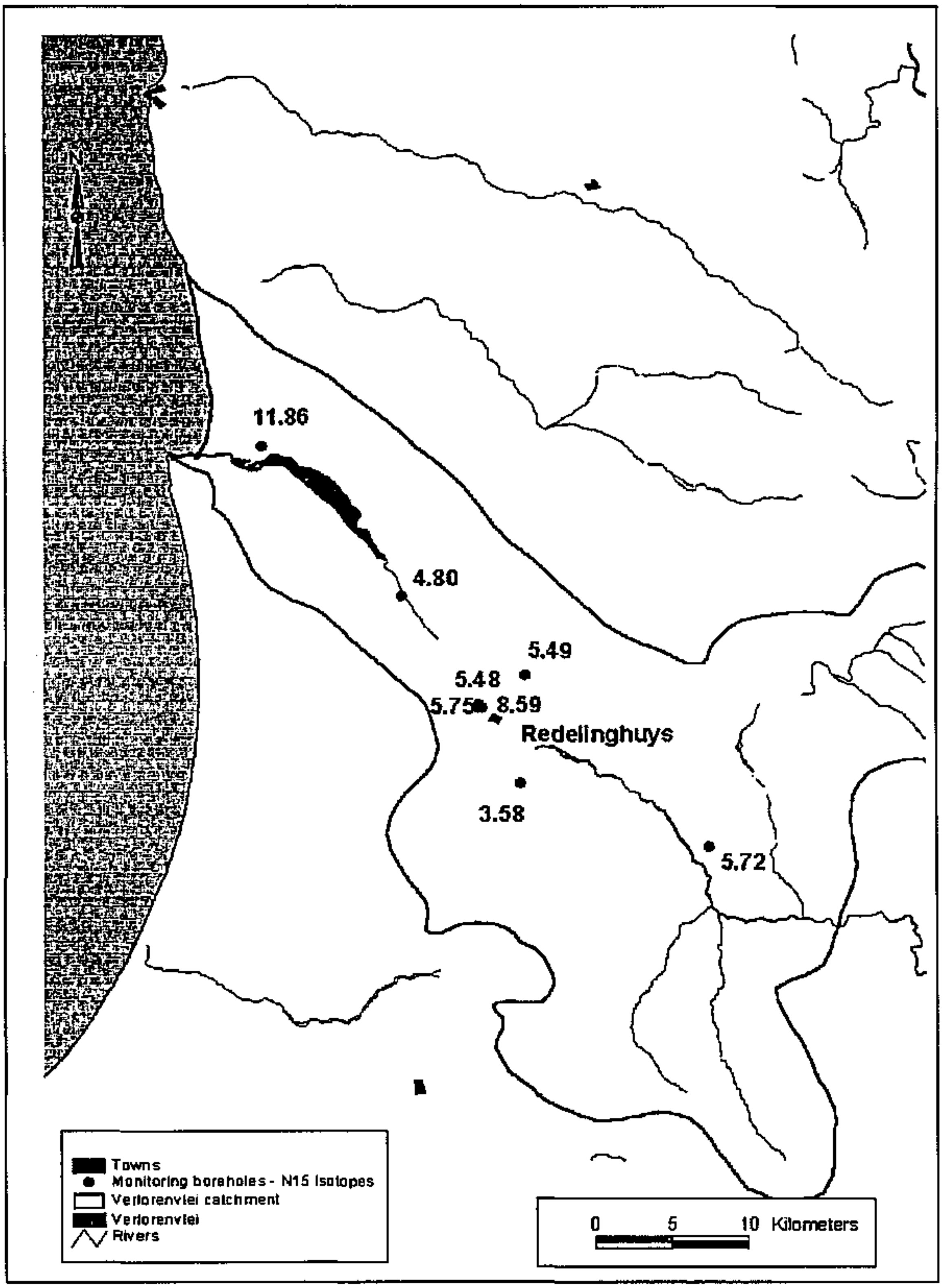


Figure 5.33: Site B: Spatial distribution of $\delta^{15}N$ isotope values (29 November 1996)

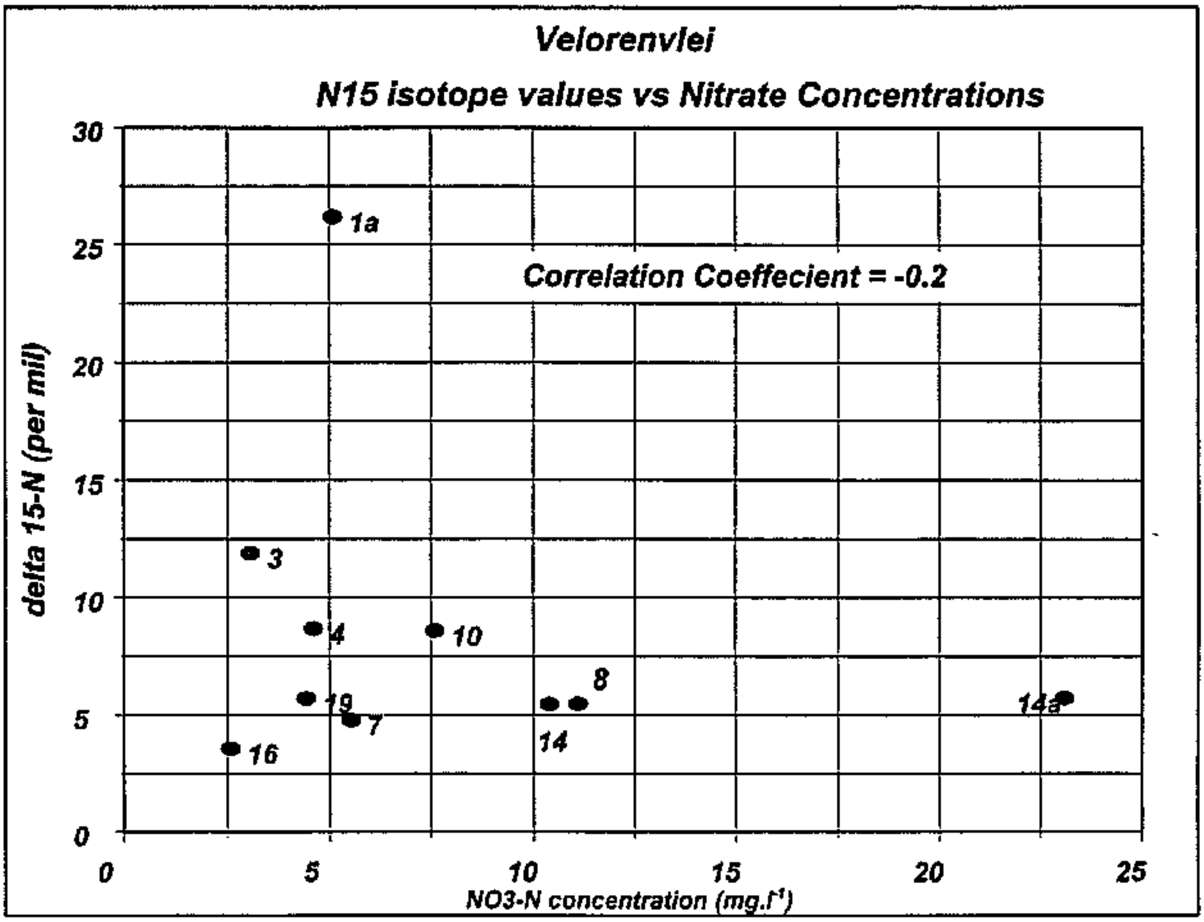


Figure 5.34 :Site B: Graph of NO_3-N versus $\delta^{15}N$ isotope values

5.3.5.2 Description and discussion of soils data

A soils analysis, which was taken from typical sands of the area (32°17' S and 18°33' E), is listed in Table 5.13.

The sodium concentrations are higher in groundwater than measured for the soil sample. The potassium values measured in the groundwater are lower for most samples than the value measured in the soil sample. Thus the potassium is being retained within the soil horizon and is not being flushed by unsaturated flow to the groundwater. For phosphorus, the soil concentration is higher than the very low concentrations detected in the groundwater. From the analysis above, the soil is typically a clean sand and the concentrations of all determinands measured are low. The resistance value is high, indicating a lack of charge-carrying ions in the soil, which confirms the pure nature of the sand.

Table 5.13 : Soils analysis for 32°17' S, 18°33' E (Verlorenvlei area)

Texture		sand
pH (KCl)		4.2
Resistance	ohms	10400
Sodium (Na)	mg/kg	13
Phosphorus (P)	mg/kg	4
Potassium (K)	mg/kg	12
Aluminium (H + Al)	me%	0.25
Calcium (Ca)	me%	0.28
Magnesium (Mg)	me%	0.08
Copper (Cu)	mg/kg	0.07
Zinc (Zn)	mg/kg	0.22
Manganese (Mn)	mg/kg	0.10
Boron (B)	mg/kg	0.11

5.3.6 Summary of impact of fertilizer application on groundwater quality, with particular reference to nitrate levels

The application of fertilizers in the study area is carried out with caution, as the danger of over-fertilization can cause the ruin of a crop. However, the sandy nature of the soils and the shallow water table for the area means that the groundwater is susceptible to contamination. Bearing in mind, that a large area was sampled, with significant spatial variation regarding the location of irrigation circles relative to the sampling localities, and temporal variation regarding when the irrigation circles are actually being used, the impact of inorganic fertilizers in the study area is minimal. On averaging all sampling localities the nitrate values have stayed constant for the year of monitoring. The isotope analyses suggest that the nitrate in groundwater is derived from probably a combination of fertilizer nitrogen and natural soil nitrogen. As of yet it has not been possible to determine the relative contribution of these sources.

5.3.7 Conclusions

This is an important area to study as there is widespread concern about the groundwater quantity and quality. The concern has been expressed by a number of organisations, including fertilizer companies. To date a sampling routine has been established and good data has been gathered for the period of a year. It would be important for the sustainable use of the groundwater if a longer term monitoring programme could be established. From these studies, it appears that besides seasonal fluctuations at the sampling localities there has not been a regional increase in nitrate concentrations. The isotope analyses indicate that, although fertilizer nitrogen may contribute to the groundwater nitrate concentration there are other sources as well.

5.3.8 Recommendations for longer term investigations

In terms of this project, further work is still required in analysing the data and drawing sound conclusions. This study has provided a broad overview of the hydrogeology and hydrgeochemistry of the area, which includes large spatial and temporal variability. The Dept. of Agriculture is currently conducting a detailed study on nitrogen dynamics within an irrigation circle and this should contribute significantly to the understanding of nitrate movement in the Sandveld in a potato crop. It is important to stay in close contact with the people working on this project. In terms of meeting the objectives of this study on-going monitoring is not necessary, however it would be very important for the region that monitoring is continued. It is an area with a sensitive and unique biodiversity and it should be monitored so that a balance can be maintained between farming practices and environmental conservation.

5.4 SITE C: HERTZOGVILLE

5.4.1 Site Background

The site was selected for the following reasons:

- It is an area essentially dedicated to wheat farming, which differs from the crop types in the other two study areas.
- Wheat farming has been carried out for many years in the area, and high groundwater nitrate concentrations are known to occur in the area.
- The geology differs from the other two sites and the aquifer is a fractured rock type aquifer.
- Boreholes are present in the agricultural areas and are pumped on a very regular basis. A number of them are used for town water supply, thus facilitating the ease of sample collection.
- For a number of boreholes in the area, groundwater analyses have been carried out for many years.
- The Hertzogville Town Council had no objection to a study being conducted in the area, and sampling on a two monthly basis was not problematic as the CSIR has staff members located in Bloemfontein.

Hertzogville is situated 100 km northeast of Kimberley in the Orange Free State. Figure 1c shows the position of the boreholes that were sampled, in relation to the agricultural areas and layout of Hertzogville.

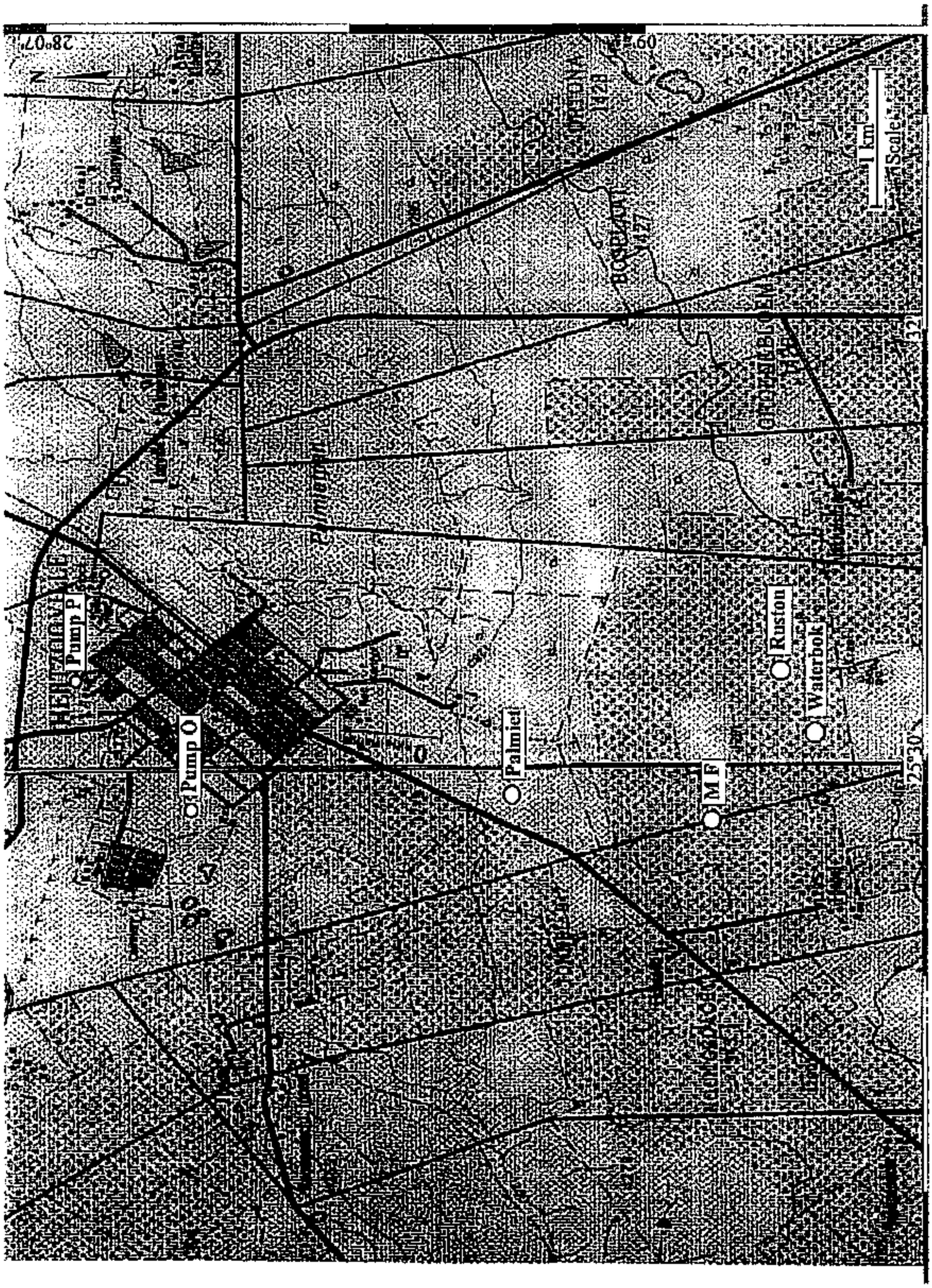


Figure 5.35 : Site C: Location of boreholes in the study area

5.4.2 Application of fertilizers to the site

Annual crop cycle

Wheat is grown on an annual cycle in the area. Towards the end of the rainy season, typically in March or April, fertilizer is applied to the wheat fields as 3:2:0 (N:P:K), with a tractor drawn spreader at 100 $kg.ha^{-1}$. This is the only application of fertilizer during the year. About one month later, when the rains are over, the wheat is planted. Harvesting occurs in November and the wheat plant remnants are ploughed into the ground in December or January.

The Hertzogville Agricultural Officer has expressed surprise at the high nitrate levels found in groundwater in the study area, as he is aware there are other areas that are applying higher fertilizer amounts, but which have recorded lower groundwater nitrate concentrations. In addition, the groundwater strikes in the boreholes are at least 35 metres deep. (B. van der Skyff, pers. comm.) It is unusual for anthropogenic nitrate to reach such depths in a fractured rock type aquifer.

5.4.3 Climatic characteristics

Rainfall data for the area was obtained from the South African Weather Bureau Information Section and the total rainfall recorded for 1996 was 843 mm. The monthly rainfall amounts have been plotted (Figure 5.36) and it can be seen how the rainfall season extends from November through to April. There is very little rainfall during the cold winter months. Evaporation and temperature data was not available for Hertzogville. The reasons for this at the time of writing this report are not clear and will be followed up. However temperature data was provided for the Frankfort area. This data has also been plotted on Figure 5.36. The coldest months are June and July and the warmest January and February.

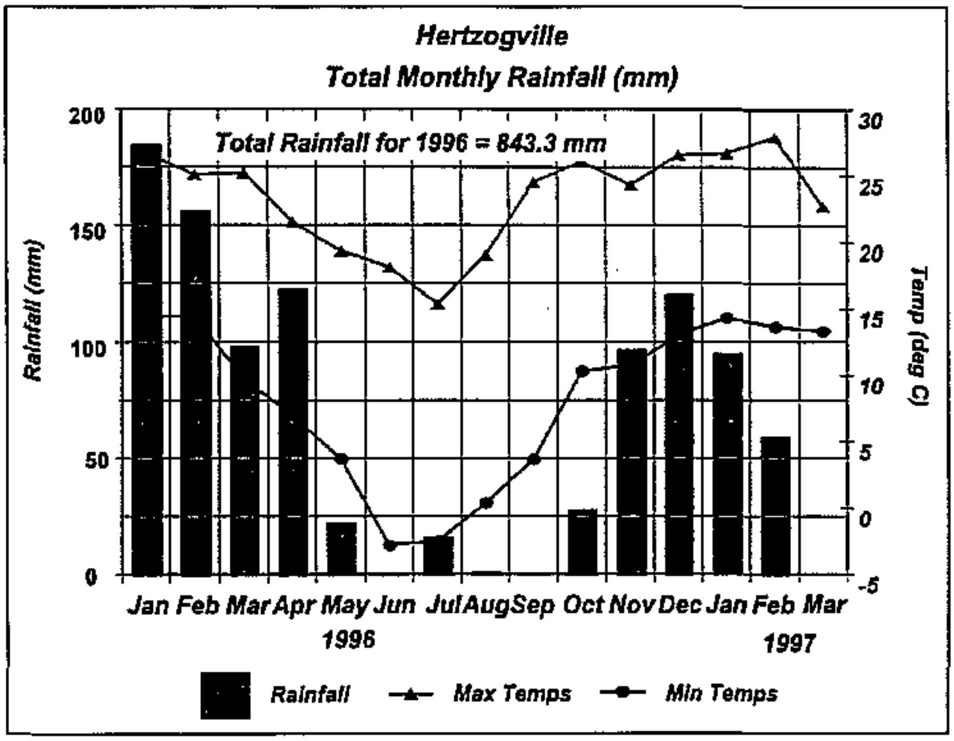


Figure 5.36 :Site C: Climatic data for Hertzogville and Frankfort.

5.4.4 Geology of the site

Two of the boreholes sampled (Borehole P and Borehole O) are drilled directly into the Volksrust Shale Formation. Extensive alluvium, sand and calcrete cover the area and the other four boreholes that were sampled (Palmiet, MF, Waterbok and Ruston) are drilled through this alluvium into the underlying Volksrust Shale Formation. The thickness of the alluvium is undetermined. The Volksrust Shale Formation is part of the Ecca Group within the main Karoo basin. The formation is 150 m to 250 m thick and essentially comprises shale. The Formation overlies the Vryheid Formation, also of the Ecca Group, which consists of sandstone, shale, and subordinate coal beds and has a maximum total thickness of 500 m. Overlying the Volksrust Shale is the Emakwezini Formation of the Beaufort Group which consists of shale, mudstone, sandstone, and minor coal seams and has a reported thickness of approximately 570 m (SACS, 1980). Karoo dolerites exist within the area. These dark coloured crystalline igneous rock abundantly intrude the sediments of the Karoo sequence within the study area environs. The dolerite intrusions give rise to low relief, flat topped hills within the area. The intrusions in the area are generally horizontal, evenly inclined or undulating sheets with a well-marked transgressive tendency. No dolerite dykes occur within the area.

5.4.5 Hydrogeology

The aquifer within the area is a fractured rock type aquifer. Unfortunately no information has been recorded for the boreholes drilled. There are no geological logs of the boreholes, no accurate records of borehole depths, water strike depths or yields.

The alluvium that covers part of the Hertzogville area is reported to be 3 to 4 metres thick (B. Van der Skyff, pers. comm.). The aquifer within the area is a confined fractured rock aquifer. The boreholes in the area are 40 to 50 metres deep with the main water strike being at approximately 35 metres below the surface. The rock types intersected are mainly shale and dolerite. The piezometric water level is approximately 3 metres below surface.

5.4.6 Field investigation

The field investigation was initiated by a site visit on the 25 March 1996. The objectives of the site visit were:

- to meet with relevant Hertzogville Town Council members to gain their approval for this project,
- to establish a sampling protocol with the Bloemfontein based CSIR personnel who would continue the 2-monthly sampling of the boreholes,
- to obtain farmers permission to access available boreholes,
- to investigate the suitability of boreholes in terms of the objectives of this study,
- to determine what information can be obtained from each borehole and
- to inspect the immediate environs of each borehole for possible contributing sources of contaminants.

Listed below are comments made of each borehole during the site visit:

1	**MF borehole:**	Occurs in an area where there is currently no farming, but which has been planted years ago. Nearby fields of fodder (silk grass)have been planted. Water level measurements are not possible.
2	**Ruston borehole:**	Sited in the middle of an agricultural area. The ground is plowed and corn will be planted when the first rains arrive, which is usually in April. Fertilizers have already been applied to the soil in preparation for the planting. A water level measurement is possible.
3	**Waterbok borehole:**	This borehole is situated 100 m away from the Ruston borehole. The comments made for the Ruston borehole are also applicable. Water level measurements are not possible.
4	**Palmiet borehole:**	This borehole is situated within a small game farm and there is no agricultural activity in the area. The borehole is approximately 70 m deep, with a static water level of 9.2 m below ground level. The borehole reportedly yields 5000 gal/hr (H van der Watt pers.comm.).
5	**O borehole:**	This borehole is not situated in an agricultural area, but on the western outskirts of the town. It provides water for the sewage treatment plant. In close proximity to the borehole (500 m away) a low cost housing settlement exists.
6	**P borehole:**	This borehole is situated in close proximity to the town on the northern outskirts. It is situated in a non-agricultural setting. The borehole pumps continuously at 5 to 10 L/sec and supplies the town's main drinking water reservoir.

After this initial sampling run was completed, samples were collected on an approximately 2-monthly basis. Listed below are the sampling dates:

Run #	Date of sampling
1	25 March 1996
2	21 June 1996
3	21 August 1996
4	31 October 1996
5	31 January 1997
6	to be completed

Samples were collected from boreholes MF, Ruston, Waterbok, Palmiet, O and P. Borehole O was out of commission for sample runs 2, 3 and 4. Borehole P could not be sampled on run 5. Data also exists for boreholes A, B, C and S, however these boreholes no longer exist. The water samples collected were analysed for nitrate, potassium, phosphate, electrical conductivity and pH. A full analysis was carried out on the samples collected during the second and fourth sampling runs (21 June 1996 and 31 October 1996) and $\delta^{15}N$ isotope analyses were carried out on the samples collected during the fifth sampling run (31 January 1997). Water levels were not measured in the boreholes as the majority of boreholes were well secured and water levels inaccessible to dip metres. The boreholes that were accessible were pumping at the time of sampling. The pumps are controlled by a reservoir float switch and could not be overridden at the wellhead.

5.4.6.1 Description and discussion of hydrochemical data

From the Piper trilinear plot of the 21 June 1996 sampling results it can be seen that the water chemistry for all sampling points is essentially the same. The water is $CO_3 + HCO_3$, Ca + Mg type water. There is no dominant cation and $CO_3 + HCO_3$ and SO_4 are the dominant anions. (Figure 5.37).

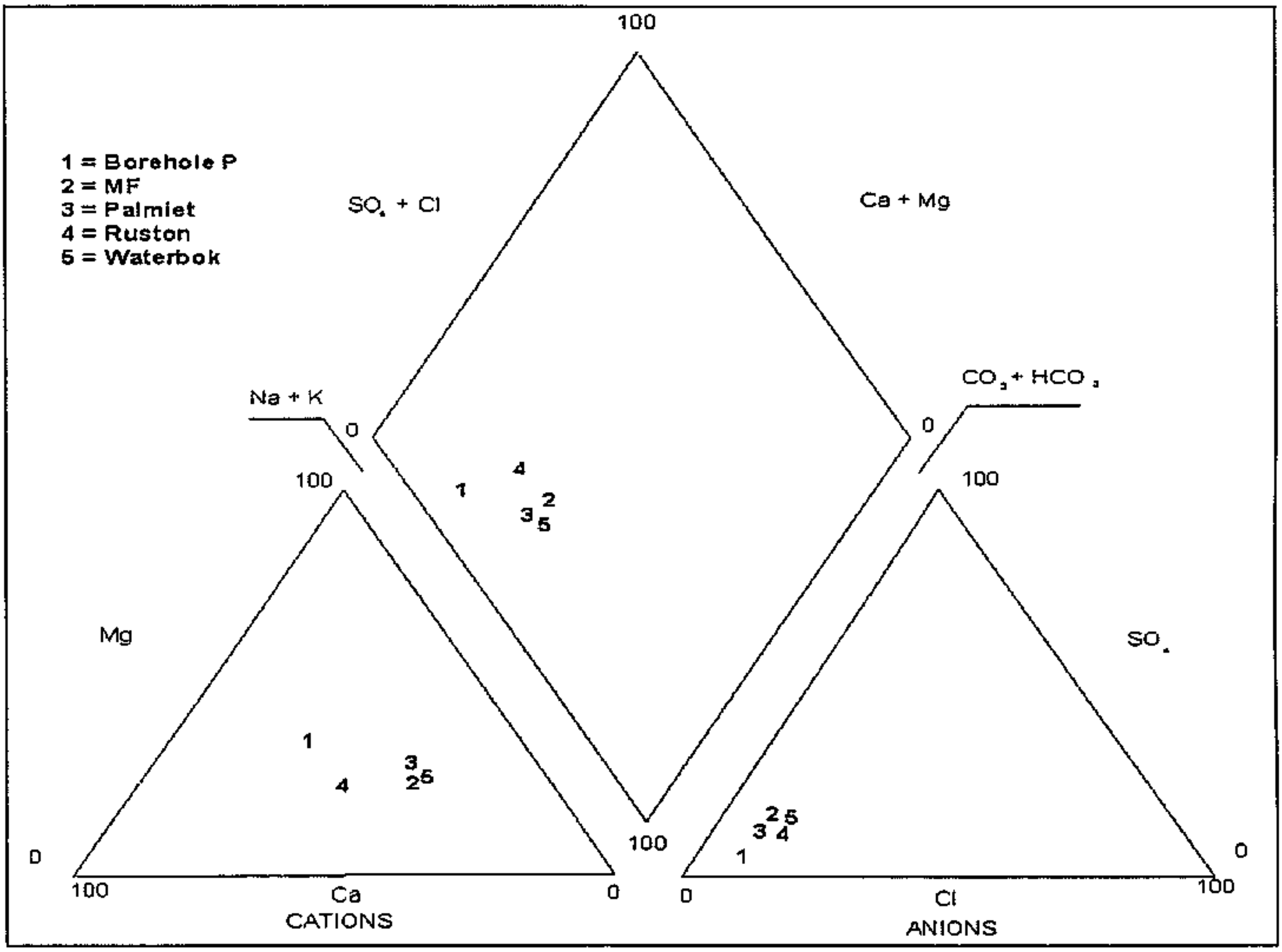


Figure 5.37: Site C: A Piper trilinear plot of the sampling results, collected on 21 June 1996.

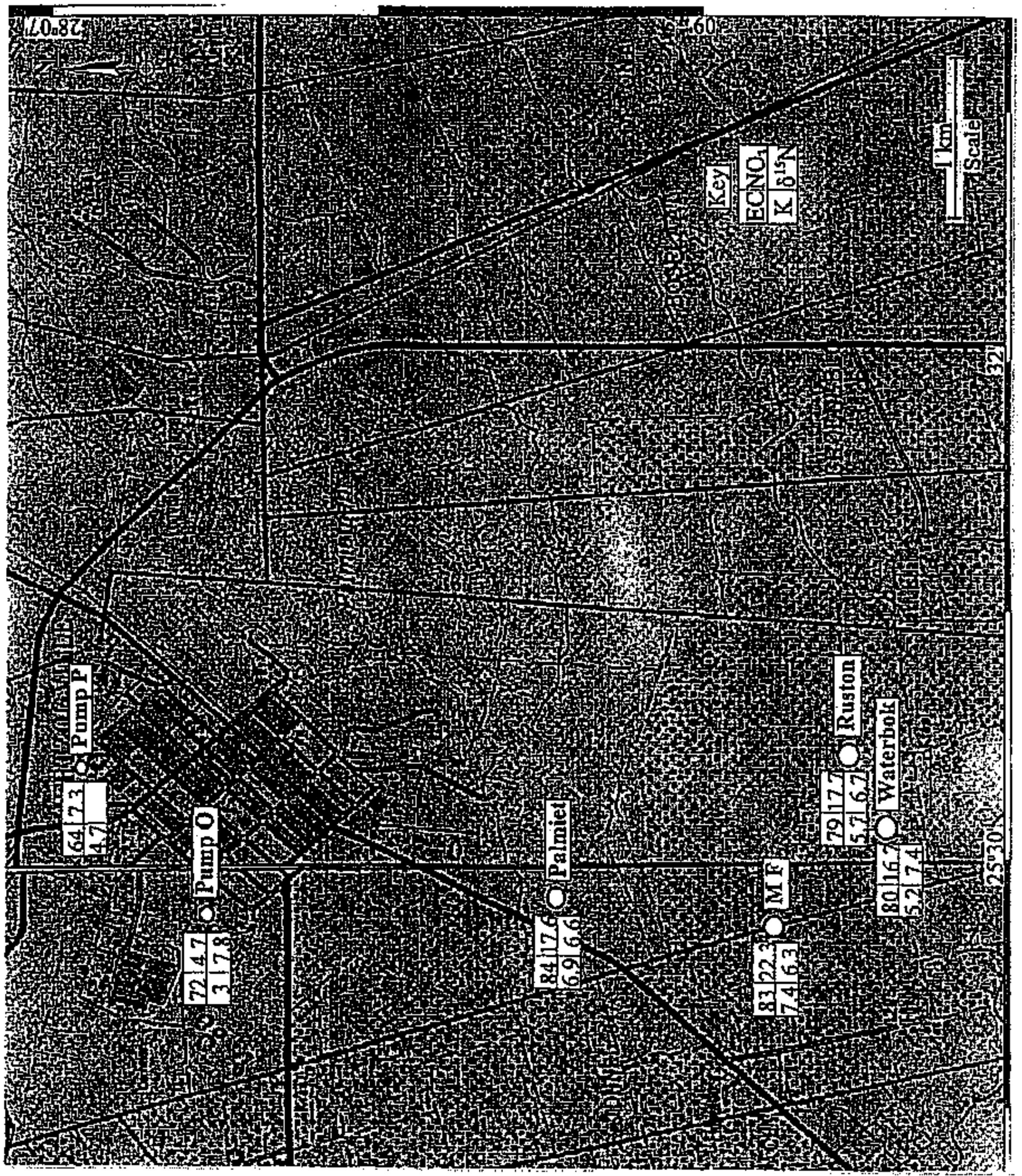


Figure 5.38 : Site C: Spatial distribution of EC ($mS.m^{-1}$), nitrate ($mg.l^{-1}$), potassium ($mg.l^{-1}$) and $\delta^{15}N$ values, based on the 31 October 1996 sampling run

Electrical Conductivity

The spatial distribution of electrical conductivity values, based on the 31 October 1996 sampling run results is shown in Figure 5.38.

The highest EC values recorded for this sampling run are the *MF* and *Palmiet* boreholes (83 and 84 mS.m^{-1} respectively), although the *Waterbok* and *Ruston* boreholes have similar values (80 and 79 mS.m^{-1} respectively). The boreholes in close proximity to the town (O and P) have lower EC values of 72 and 64 mS.m^{-1}. It is interesting to note that the Palmiet borehole, which is within a non-agricultural area has the highest EC values. From the field inspection it was considered that this borehole would provide background chemistry values. However, the values seem to indicate that the aquifer transmissivities are high enough for the borehole to be abstracting water that has recharged through agricultural areas.

Table 5.14 provides a summary of the findings from inspection of the chemistry data.

Table 5.14 : Statistics related to EC, nitrate and potassium concentration changes

Borehole	Sampling Dates	Differences EC (mS.m^{-1})	Differences NO$_3$-N (mg.l^{-1})	Differences EC Best fit (mS.m^{-1})	Differences NO$_3$-N Best fit (mg.l^{-1})	EC mean (mS.m^{-1})	EC SD (mS.m^{-1})	NO$_3$-N mean (mg.l^{-1})	NO$_3$-N SD (mg.l^{-1})	K mean (mg.l^{-1})	K SD (mg.l^{-1})
MF	Oct 76 - Oct 96	+11	+4.3	+7.9	+5.3	79.0	4.7	20.0	3.1	7.1	0.2
Ruston	Oct 76 - Oct 96	+7	-1.3	+7.9	+0.8	76.5	5.0	19.9	3.1	5.5	0.2
Waterbok	Jun 88 - Jun 96	+2	-0.3	+0.8	+0.2	78.6	4.7	20.4	5.0	5.3	0.1
Palmiet	Mar 96 - Jun 97	-1	+0.7	-0.4	+0.8	83.4	0.5	17.3	0.4	6.5	0.3
O	Jan 77 - Jan 97	+16	+0.7	+8.9	+4.8	75.3	7.3	8.9	4.1	3.2	0.2
P	Oct 76 - Oct 96	+14	+3.3	+9.1	+3.1	61.1	6.2	6.9	2.5	4.4	0.2

Borehole *MF* has a long record of sampling results going back more than 20 years. If one plots a straight line through the EC values to smooth out the seasonal fluctuations, there is an increase of 7.9 mS.m^{-1}. Comparing the October 1976 results with those of October 1996 gives an increase of 11 mS.m^{-1} over the 20 year period. The *Ruston* borehole has been monitored for the same length of time and the best fit straight line through the EC values indicates that the values have increased by 7.9 mg.l^{-1}, and comparing October 1976 results with October 1996 results shows a 7 mS.m^{-1} increase in EC. The data for the *Waterbok* borehole does not extend as far back as the previous two boreholes and the first measurements were taken in 1986. The overall

EC values appear to be constant for this borehole and when comparing the October 1986 data to August 1996 (the pump was removed for repairs during the October 1996 sampling run) the EC values have increased only mS.m^{-1}. During the year of monitoring the *Palmiet* borehole's EC value remained constant. This borehole was not monitored prior to the commencement of this study. *Borehole O* has been monitored since August 1976 and the best fit straight line through the data indicates an increase in EC values of 9 mS.m^{-1}. Comparing January 1977 to January 1997 (the borehole was out of commission for the October 1996 sampling run) the EC value has increased by 16 mS.m^{-1}. *Borehole P* has also been monitored for the same period as Borehole O and the best fit line indicates an increase in EC levels of 9 mS.m^{-1} and a comparison between October 1976 and October 1996 data reveals a 14 mS.m^{-1} increase in EC.

Nitrate

Regarding the spatial distribution of nitrate values, the four non-urban boreholes have similar values, ranging between 16.7 mg.l^{-1} and 22.3 mg.l^{-1} from the 31 October 1996 sampling run. The borehole located in fields where no farming is occurring has the highest nitrate values. The *Palmiet* borehole, which is in a non-agricultural area has nitrate values very similar to the ploughed and fertilized areas. Once again this could indicate that the groundwater being abstracted is derived from agricultural areas, or that the natural nitrate values are elevated in the area. The lower nitrate values for *Borehole O* and *Borehole P* suggest that the former scenario is more probable.

The temporal trend for the *MF* borehole indicates a 5.3 mg.l^{-1} increase, according to a best fit line. Comparing October 1976 values with those for October 1996, there has been a 4.3 mg.l^{-1} increase in NO_3-N. The overall trend for the *Ruston* borehole indicates a slight increase (0.8 mg.l^{-1}) in nitrate concentration, and there has been an absolute decrease of 1.3 mg.l^{-1} NO_3-N between October 1976 and October 1996. The *Waterbok* borehole shows a slight increase in nitrate concentrations. However there has been a 14.9 mg.l^{-1} range in values over the time of data collection, reaching a maximum of 31 mg.l^{-1} and a minimum of 16.1 mg.l^{-1}. The *Palmiet* borehole shows slight decrease in nitrate concentrations, and similarly for EC and Potassium. This lack of variability in EC, nitrate and potassium concentrations probably reflects the fact that the borehole is located outside of agricultural areas. There will have been sufficient travel distance and time for the water recharged through the agricultural areas to homogenize and stabilize before being abstracted. After twenty years of monitoring, *Borehole O* and *Borehole P* show increased nitrate concentrations. The best fit lines for the nitrate time series graphs show a 4.8mg.l^{-1} and 3.1 mg.l^{-1} increase for Borehole O and Borehole P respectively. A direct comparison of nitrate concentrations for Borehole O between January 1977 and January 1997 shows a 0.7 $mg.l^{-1}$ increase in nitrate concentration. Between October 1976 and October 1996 there is a 3.3 $mg.l^{-1}$ increase for Borehole P. The position of these boreholes and their nitrate concentration values suggest that temporal trends are not likely to be linked to agricultural activities.

Potassium

Potassium concentrations are slightly higher for the boreholes located away from town in an agricultural setting. These slightly higher concentrations are attributed to agricultural activities. Potassium concentrations are relatively constant over the data collection period for all the boreholes sampled. No phosphorous or NH_4-N was detected in the samples.

It was not possible to determine the location of boreholes A, B, C and S. On inspection of the chemical data the EC values are comparable to what was measured from the boreholes located in the agricultural setting. The nitrate values, however, are four times higher than that for the boreholes sampled during this study.

Nitrogen Isotopes

From the sampling run that was completed on 31 January 1997, five samples were submitted to the CSIR isotope laboratory for analysis. The results are presented in Table 5.15.

Table 5.15 : $\delta^{15}N$ values and nitrate concentrations

Locality	$\delta^{15}N$ (‰)	NO_3-N ($mg.l^{-1}$)
MF	6.3	21.6
Ruston	6.7	17.5
Waterbok	7.4	16.7
Palmiet	6.6	17.5
Borehole O	7.8	4.7

$\delta^{15}N$ values for fertilizer are typically in the range of 0‰ to +4‰, natural nitrate in the range of +4‰ to +8‰ and waste water >+8‰. The highest $\delta^{15}N$ value obtained for the sampling run was for Borehole O which is situated close to the town of Hertzogville. Borehole P, also in close proximity to the town, was not sampled as the pump was out of commission. The $\delta^{15}N$ values for the agricultural boreholes are not much lower than that for Borehole O. One should consider absolute values for $\delta^{15}N$ and not averages.

The relationship between $\delta^{15}N$ and nitrate concentration has also been established from the graph plotted in Figure 5.39. A correlation coefficient of -0.87 exists which suggests that fertilizer nitrogen contributes to the nitrate concentration found in the groundwater.

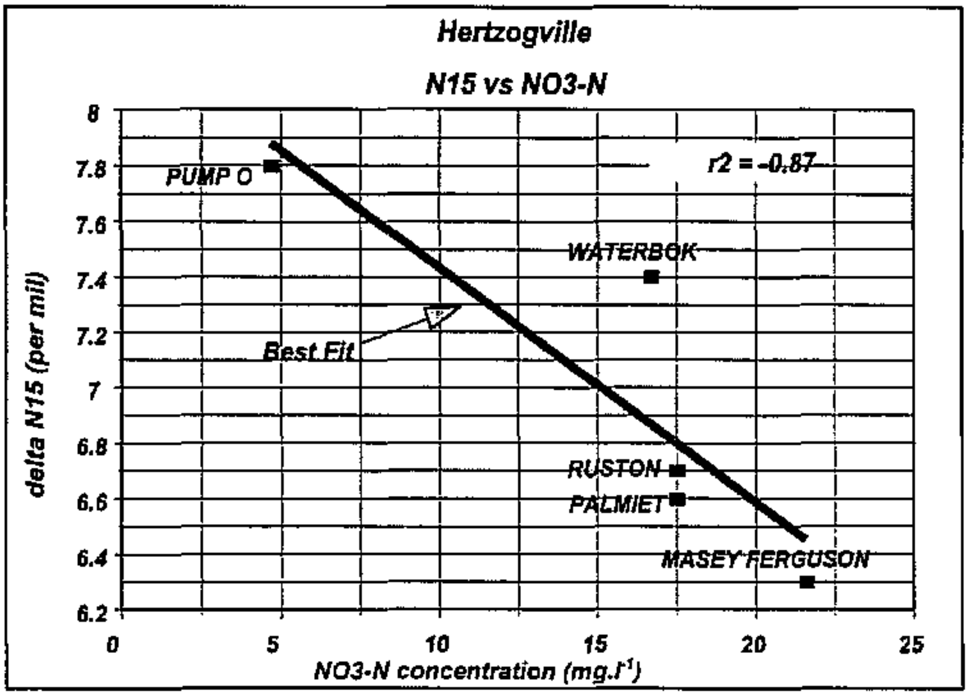


Figure 5.39 :Site C: $\delta^{15}N$ versus NO_3-N concentrations

5.4.6.2 Description and discussion of soil data

No soil analyses could be obtained for the area at the time of writing this report.

5.4.7 Summary of impact of fertilizer application on groundwater quality, with particular reference to nitrate levels

The first consideration is that the nitrate concentrations within the town are one half to two thirds less than the values measured in the agricultural areas. Thus the activities within the agricultural areas are causing nitrate concentrations to be higher. These activities are essentially:

- ploughing, both pre- and post-harvesting,

- application of fertilizers,
- planting of the wheat.

Of these activities the ploughing, both pre- and post- planting, which occurs during the rain season, and the application of fertilizers toward the end of the rain season are the most probable factors that will contribute to raising the nitrate concentrations above those measured in the town. The decomposition of plant residue, would result in the production of NH_4^+, and the soils, being derived from the Ecca Shale would have a high cation exchange capacity. In conjunction with the high cation exchange capacity, nitrification processes and limited denitrification, nitrate would be produced and mobilized down to the water table. Although the aquifer is considered confined, and the unsaturated zone would have a high clay content and thus low hydraulic conductivity, preferential flow path mechanisms would enable rainfall to recharge the aquifer. With the considerable doleritic intrusions in the area, a more semi-confined aquifer is expected rather than it being fully confined. In addition nitrification of the mineral fertilizer, with limited denitrification, would also result in nitrate reaching the groundwater. Further work is required however to determine the ratios that these two sources may be contributing to the total nitrate concentration. The $\delta^{15}N$ values are in the range of natural nitrate, however the $\delta^{15}N$ versus NO_3-N relationship indicates that fertilizer is contributing to the overall nitrate concentration.

5.4.8 Conclusions

The study site has been well chosen, in that it had been generally thought that the nitrate concentrations in groundwater are entirely due to the application of fertilizers for wheat growing enhancement. However, the nature of the aquifer, the application rates, the timing of application, and soil thickness indicates that the nitrate in groundwater is derived from farming practices, which include both ploughing and fertilizer application. At the time of writing this report the relative contributions of these activities to nitrate concentrations had not been determined. Besides seasonal fluctuations, the nitrate levels have remained constant in the longer term for boreholes located in the agricultural areas. Only one borehole showed an increase of 5.3 $mg.l^{-1}$ NO_3-N over a period of twenty years of monitoring.

5.4.9 Recommendations for longer term investigations

More time needs to be spent and expert input is required to gain a better understanding of the nitrate dynamics. Further field sampling is not seen as being necessary for meeting the objectives of this study.

5.5 SUMMARY OF FIELD INVESTIGATIONS

The findings of the field investigations are summarised in the table below:

Summary of findings of the specialist field study on the impact of fertilizer application to agricultural land on groundwater quality.

	Site A	**Site B**	**Site C**
Farming Activity	Export Table Grapes	Potatoes	Wheat
Loading	400 $kg.ha^{-1}.yr^{-1}$	240 $kg.ha^{-1}.yr^{-1}$	100 $kg.ha^{-1}.yr^{-1}$
Irrigation	Yes	Yes	No
Use of organic fertilizers in the area	Yes	Yes	No
Use of inorganic fertilizers in the area	Yes 1:0:1 and LAN	Yes 1:0:1 or 3:0:5	Yes 3:2:0
Field data collected as part of this project	Groundwater levels, Groundwater hydrochemistry, from shallow wells and boreholes	Groundwater levels, Groundwater hydrochemistry, from shallow boreholes	Groundwater hydrochemistry, from boreholes
Climate	529 mm (1996), winter rain	350 mm (1996) winter rain	843 mm (1996) summer rain
Unsaturated zone	2 - 3 m, alluvial soils	2 - 3 m, sandy soils	5 m, clay rich soils
Aquifer Type	Unconfined	Unconfined	Confined
Hydrochemistry	Ca, Mg, Cl and SO_4	Na, Cl	Ca, Mg, CO_3, HCO_3
Potential groundwater contamination from other sources	Pit latrines and septic systems	Pit latrines and septic systems	None
Groundwater contamination:			
Nitrate	Yes 0 to 13.2 $mg.l^{-1}$	Yes 0 to 23.7 $mg.l^{-1}$	Yes 6.9 to 20.4 $mg.l^{-1}$
Phosphorous	No	No	No
Potassium	Low 0.2 to 18 $mg.l^{-1}$	Low to med. 1.3 to 121	Low 3.2 to 7.1
EC	Low to med. 2 to 140 $mS.m^{-1}$	Low to high, 31 to 840 $mS.m^{-1}$	Med. 61 to 83 $mS.m^{-1}$
Impact of fertilizers on nitrate concentrations in groundwater	Essentially minimal, although higher in localised areas. The % that fertilizer contributes to total nitrate concentration undetermined.	Essentially low. The % that fertilizer contributes to total nitrate concentration undetermined.	Potentially moderate. The % that fertilizer contributes to total nitrate concentration undetermined.

The case studies show that the application of fertilizers does have an impact on nitrate concentrations in groundwater, however this impact may be very small. In none of the case studies was it possible to attribute the nitrate concentrations in groundwater directly and entirely to the application of fertilizers. At Site A, the nitrate concentration fluctuations in the groundwater corresponded with the application of fertilizer at the surface, however the isotopic analyses suggest that fertilizer nitrogen may have only contributed minimally to the total nitrate concentration. At Site B, there was no corresponding increase in nitrate concentrations with the application of fertilizer at surface. The large temporal and spatial variability within the study area would mask this possible relationship. The isotope analyses also indicated that the nitrate concentrations in groundwater are mainly attributable to non-fertilizer sources. At Site C, which has the highest nitrate concentrations of all three study sites, the use of fertilizers may have a minimal effect on groundwater nitrate concentrations for a number of reasons. The isotopic data indicates other sources of groundwater nitrate, however there is evidence that fertilizer nitrogen does reach the groundwater. The actual contribution is difficult to quantify.

5.6 RECOMMENDATIONS FOR THE NON-TECHNICAL BOOKLET

One of the objectives of this specialist study is to gather data and information which can be incorporated into a non-technical booklet to inform farmers on how they can avoid contaminating groundwater when applying commercial fertilizers to their land. The booklet should provide an introduction to hydrogeological principles and draw attention to the value of a good quality groundwater resource. Farmers need to be made aware of the fact that they can save money wasted on over application of nitrogenous fertilizers by instituting good management practices which ensure that the level and timing of fertilizer application matches that of crop demand.

It is proposed that the non-technical booklet should attempt to increase farmers' awareness in the following areas in order to mitigate groundwater contamination.

- Introduce farmers to the concept of aquifer vulnerability and the factors which affect contaminant transport to the saturated zone. For example depth to water table, soil permeability, recharge, pathways of preferential vertical migration.

- Discuss the different processes of nitrogen release from fertilizers and highlight the economic waste of inappropriate and over application which lead to leaching losses.
- Publicise common nitrogen crop requirements and sources of information on fertilizer application regimes.
- Introduce farmers to groundwater hydrochemistry and emphasise the importance of keeping ground water quality within drinking water limits.
- The health risks associated with elevated nitrate levels, both to humans and livestock, should be outlined.
- Briefly discuss integrated contamination control with respect to other pathways of environmental pollution. A realistic understanding of farmers' requirements should be taken into account so that recommendations to minimise contamination from fertilizers do not result in pollution from other nutrient sources.

In summary, the non- technical booklet should aim to increase farmers' awareness of groundwater contamination risks. It should adhere to the principles of integrated pollution control and make practical recommendations for best agricultural management practice.

CHAPTER 6

SLUDGE APPLICATION TO AGRICULTURAL LAND

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EXECUTIVE SUMMARY

The objectives of this specialist study are

- to collect field data from 3 study sites to determine whether groundwater contamination has occurred as a result of the application of sewage sludge to agricultural land in South Africa
- to collect data and information which can be used to support the non-technical booklet.

Sewage sludge is applied to agricultural land as a soil conditioner and as a fertilizer, providing nitrogen, phosphorus and micronutrients.

A total of 8% South African sewage sludge is sold to farmers for use on agricultural land and a further 20% is used in beneficial land application (Ekama, 1992). The amount of sludge used in agriculture is likely to increase in the future as more sludge is produced due to improved levels of sanitation and as beneficial disposal options are favoured.

The use of sludge may result in contamination of the soil, crops, atmosphere (nuisance odours), groundwater and surface water. Literature shows potential contaminants of groundwater to be heavy metals, microbes, nitrate, DOC and trace organics. Nitrate is the contaminant with the greatest risk of reaching groundwater at potentially harmful concentrations if sludge is applied at above agronomic rates.

Three sites were selected for groundwater quality investigation where vulnerable aquifers underlie sludge applied agricultural land. A summary of the results is presented in the table below.

Summary of findings of the specialist field study on the impact of sludge application to agricultural land on groundwater quality.

	Site A	Site B	Site C
Sludge application	Liquid sludge ploughed in throughout the year	Sludge cake ploughed in early winter	Sludge cake ploughed in early winter
Loading	High (?)	High, 100 tpa/ha	Low, 3 tpa/ha
Primary purpose	Sludge disposal	Soil conditioning	Soil conditioning
Use of other organic fertilizers in sludge area	No	No	Yes
Use of inorganic fertilizers in sludge area	No	Yes	Yes
Irrigation	Yes	No	Yes
Field data conducted as part of this project	Shallow groundwater Deeper groundwater Soil profiles	Shallow groundwater Deeper groundwater Soil profiles	Shallow groundwater Deeper groundwater Soil profiles
Climate	700 mm p.a summer rain	500 mm p.a winter rain	600 mm p.a winter rain
Unsaturated zone	7 m thick, clayey soil	3 m thick, sandy.	1 m thick, sandy.
Aquifer type	Dolomite with secondary permeability of ≈ 10 m/d	Unconsol. sand with primary permeability of ≈ 20 m/d	Unconsol. sand with primary permeability of ≈ 10 m/d
Hydrochemistry	CaCO3 with elevated nitrate and sulphate. TDS 700 - 1700 mg.l^{-1} Reduced at depth.	NaCl. TDS 100 - 400 mg.l^{-1}.	CaCO3 with NaCl. TDS 540 - 2200 mg.l^{-1} Reduced at depth.

	Site A	**Site B**	**Site C**
Groundwater contamination from other sources	Sulphate (?)	Nitrate - decaying Acacia vegetation	Nitrate and DOC from other fertilizers (?)
Groundwater contamination from sludge:			
Nitrate	Yes - to > 30 mg.l^{-1} at a distance of > 1 km with a peak of 268 mg.l^{-1}.	Yes - seasonal, localised (?), by 2 mg.l^{-1} to 15 mg.l^{-1}.	Yes (?) - contribution to extensive (?) increase to 38 mg.l^{-1} in shallow samples.
EC	Yes - from ≈ 180 to 300 mS/m due to high nitrate levels	No	No
DOC	Yes - from < 2 mg.l^{-1} to ≈ 10 mg.l^{-1} in shallow samples in the field	No	Yes (?)- to ≈ 30 mg.l^{-1} in shallow samples in the field
Microbiological indicators	No	No (?)	No (?)
Metals	Yes (?) - Cd to 0.01mg.l^{-1}	No	ND (?)
Other ions	Yes (?) - K	No	No
Impact of sludge application on groundwater quality.	Long term, significant and extensive.	Temporary, insignificant and localised (?)	Long term (?), significant at shallow depths, extensive (?).

(?) Further research required to confirm.
ND Not determined

Evidence of groundwater contamination at the three sites corresponded well to that reported in the literature.

Nitrate was found to contaminate groundwater at all sites and isotope analyses confirmed the primary source to be sludge at sites A and B.

DOC contamination of shallow groundwater occurred at sites A and C, which have long histories of sludge application.

Most heavy metals, microbes and phosphate were found not to contaminate groundwater as a result of sludge application.

The results of these case studies will be incorporated into the non-technical booklet to show examples of groundwater contamination in South Africa. In addition, information gathered as part of this specialist study will be used to discuss pollution prevention and best agricultural management practice.

6.1 INTRODUCTION

6.1.1 Motivation

The literature study carried out as part of this project identified the application of sludge to agricultural land as having significant potential to contaminate groundwater (CSIR, 1995). Little information regarding the impact in South Africa was found therefore it was decided to conduct a specialist study investigating the impact of agricultural sludge application on groundwater quality at 3 study sites.

6.1.2 Sewage sludge

The treatment of waste water at municipal sewage plants results in the production of large volumes of sludge. The nature of the sludge varies depending on the quality of the water being treated and the type of treatment process used. A study of South African waste water treatment plants (WWTPs) showed that typically 421 kg dry mass sludge are produced for every megalitre of waste water treated (Smith and Vasiloudis, 1989). The subsequent disposal and treatment of sludge is often more costly and complex than that of the waste water treatment that produced it (Ekama, 1992).

Sewage sludge produced at WWTPs is a **putrefactive, concentrated aqueous suspension of particulate organic material containing mainly biodegradable, but also inert substances** (Ekama, 1992). It contains between 3 and 10% solids. Stabilization and dewatering of the sludge are usually carried out prior to disposal (Snyman, 1996).

6.1.2.1 Sludge disposal options

The disposal of sludge presents an increasing problem to waste water treatment operators worldwide. In many countries, including South Africa, the quantity of sludge to be disposed of is increasing as standards of living increase. A greater proportion of the population is served by municipal sewers and more waste water is produced per person. It is therefore of increasing importance to employ effective disposal methods and where possible to derive benefit from disposal, ie. aim at recycling the material rather than eliminating it.

The following disposal options are commonly used:

- beneficial land application
- sacrificial land application
- composting and pelletizing
- disposal to sea
- incineration
- (co)disposal to landfill
- stockpiling at the waste water treatment plant (WWTP) in lagoons or drying beds.

Local legal constraints, quality of the sludge, demand from beneficial users and land availability are factors controlling which disposal option is used. Disposal to sea and to landfills is becoming less popular due to environmental concerns. Beneficial land application is being seen increasingly as the Best Practicable Environmental Option (BPEO).

Thermal conversion of sludge to liquid fuel, in which 50% of the energy of the sludge is recovered in the form of liquid oil, is a relatively new disposal option proven at pilot scale (Lue-Hing, 1996).

Table 6.1: Sludge disposal by different methods in selected countries (%).

Disposal method	**South Africa*(1)* 1989**	**USA*(1)* 1981**	**Portugal*(2)* 1990**	**UK*(2)* 1990**	**France*(2)* 1990**
Beneficial land application	28	42	80	51	27
Other land application	47	-	-	-	-
Incineration	-	27	0	5	20
Disposal to sea*	-	4	8	28	0
Accumulation at WWTP	20.1	12	-	-	-
Landfill	3.2	15	12	16	53

* To be phased out in the European Community by 1998

1- Ekama, 1992.

2- Lue-Hing *et al*, 1996.

Table 6.1 shows that there is a wide variation in levels of sludge recycling, however the trend in developed nations has been towards greater beneficial application to land. In the US beneficial land application is often operated by the municipalities which enables a secure and constant source of land and higher loading rates than would be used by farmers (Lue-Hing, 1996).

The shift in many countries towards greater levels of beneficial recycling of sludge has prompted a change in terminology from 'sludge' to 'biosolids'. The latter is generally more publicly acceptable. The US EPA defines biosolids as the primarily organic solid product produced by waste water treatment processes that can be beneficially recycled (US EPA, 1994). [This definition is therefore dependent on legal constraints on what can be recycled.]

Current developments in sludge research are focussing on minimising the volume of sludge produced and maximising its beneficial use. This is helped in many countries by legislation enforcing greater levels of pretreatment and pollution prevention which contribute towards improving the quality of the sludge (Isaac and Bothroyd, 1996). The approach of the European Community towards sewage sludge disposal is typical of current thinking in the developed world: to ensure that human beings, animals, plants and the environment are fully safeguarded from the uncontrolled spreading of sewage sludge on agricultural land; and to promote the correct use of sewage sludge on such land (EC Directive 86/278).

6.1.2.2 Application of sewage sludge to agricultural land

Benefits

The application of sewage sludge to cultivated land provides the following benefits (from Korentajer, 1991):

- a source of major plant nutrients (nitrogen, phosphorus and to a lesser extent potassium) which are released slowly
- a source of some essential micronutrients (Zn, Cu, Mo, Fe and Mn). These are particularly effectively applied in sludge as the chelating properties of the organic material enhance plant availability. Most are not common in inorganic fertilizers.
- the addition of organic matter improves the physical properties of the soil, in particular its water retention capacity which in turn reduces soil erosion and prevents nutrient leaching.

The composition of sewage sludge is highly variable, but from the point of view of macronutrients it can be considered as a low grade fertilizer. Published analyses of South African sludge show nitrogen content varying from 42 to 29 $g.kg^{-1}$ and phosphorus from 28 to 13.5 $g.kg^{-1}$ (dry weight) as compared to 260 and 110 $g.kg^{-1}$ for the N and P content of common inorganic fertilizers (Snyman, 1996; Korentajer, 1991). The level of potassium in sludge is typically very low at around 2 $g.kg^{-1}$ (Smith and Vasiloudis, 1989) therefore supplementary K sources are usually required. Between 30% and 85% of nitrogen in liquid sludges and 15% to 33% in dried sludge is available to crops within a growing season in soluble form (Smith and Vasiloudis, 1989; Smith, 1996). The remainder exists mainly in organic form therefore it must be mineralised to plant available inorganic forms before the plant requirements are met. The rate of mineralisation is dependent on temperature, pH, soil moisture and organic content and is therefore difficult to predict (see Figure 6.1). Typically the percentage released per year is as follows: 40, 20, 10, 5 and 2.5% for subsequent years (Palmer, 1993). The amount released is also dependent on the type of sludge as shown in the table below.

Sludge is therefore valuable as a slow release source of nitrogen, particularly in areas where the efficiency of inorganic N fertilizers is limited by factors such as high leaching losses (Korentajer, 1991). However, additional inorganic commercial fertilizers are generally required during periods of peak crop nitrogen demand.

Table 6.2: Typical nitrogen and phosphorus content of sewage sludges (UK) (as spread; $kg.m^{-3}$ for liquids; $kg.t^{-1}$ for cakes).

Sludge type	**% Dry matter**	**Total N**	**Available N**[1] *(Predominant form)*		**Total P**	**Available P**[1]
Liquid undigested	5	1.8	0.6	*33% (organic)*	0.6	0.3
Liquid digested	4	2	1.2	*60% (ammonia)*	0.7	0.3
Undigested cake	25	7.5	1.5	*20% (organic)*	2.8	1.4
Digested cake	25	7.5	1.1	*15% (organic)*	3.9	2

(1) Available in first cropping year.

Source: Smith 1996, Agricultural Recycling of Sewage Sludge and the Environment.

Generally a lower proportion of inorganic nitrogen is contained in undigested sludges, therefore less is available within the first cropping year. Sludge cake contains less soluble nitrogen after the dewatering process.

It is thought that between half and two thirds of the total **phosphorus** in sewage sludge is available for crop uptake during the first growing season after application (Smith, 1996; Smith and Vasiloudis, 1989). Calcium phosphates are rather more soluble and available to crops than those of iron, aluminium or heavy metals. Phosphorus is one of the more expensive plant nutrients and as P levels in sludges have been increasing due to more stringent standards placed on effluent treatment works, the contribution from sludge is valuable.

Sewage sludge contributes to the fertilization of cultivated lands as a nutrient source but more importantly as a **soil conditioner**. Test cases have shown that the beneficial application of sludge improves crop yields more than would be expected by the N:P:K values (Borain, 1993). The widespread use of inorganic fertilizers in cultivated areas has resulted in a gradual decrease in the organic content of the soil. In warm countries such as South Africa this is accelerated by the rapid rates of microbial decomposition of organic matter (Korentajer, 1991). The deterioration of the physical properties of soil and irreplaceable loss due to erosion are pressing environmental problems. In view of South Africa's significantly deteriorated soils and growing need for food it is believed that the promotion of sludge use for soil improvement purposes should have a high priority (Murphy, 1997).

Hazards

There are known hazards associated with the application of sewage sludge to cultivated land which have resulted in stringent controls on this practice in many countries. Sewage sludge application to land may result in contamination of the soil, crops, atmosphere (nuisance odours), groundwater and surface water. The occurrence of contaminating substances varies depending upon the health of the population served by the WWTP, the industries occurring within its catchment and the degree of pretreatment operated by industry. Principal hazards are outlined below:

- the introduction of potentially toxic substances to the food chain, soil and water
- the introduction of pathogens
- anaerobic decomposition of putrescible organic matter in the root zone

Raw, or untreated sewage sludge, has a high oxygen demand and creates **anaerobic** conditions in the soil which limit plant growth (Ekama, 1992). Sludge should therefore be stabilized prior to land application. This involves either anaerobic or aerobic controlled degradation or digestion at the WWTP which destroys approximately 40% of the volatile organic solids and reduces the oxygen demand of the sludge to around 1g O_2 / kg total solids/ hour (Ekama, 1992). Over-application of unstabilized and moist organic matter to soil can result in clogging of soil pores and reduced permeability (Myers *et al*, 1995).

Pathogens commonly found in sewage sludge include bacteria, protozoa, helminths and viruses. These present a risk if either transported to surface or groundwater (which is then consumed by animals and humans), if they adhere to crops (which are then consumed) or if they are directly ingested by grazing animals (Korentajer, 1991). Many potential pathogens are killed during digestion treatment of the sludge and many remain viable for only a short period of time in the soil system, particularly if applied to the surface.

Potentially toxic substances commonly found in sludge are as follows:

i. *Nutrients* (N, P, K) which, if leached to groundwater or transported to surface water bodies in run-off, may occur at toxic levels or result in eutrophication.
ii. *Toxic organics* such as PCBs, phenols, dioxins, polycyclic aromatic hydrocarbons, etc. The main risks are thought to lie with the ingestion of sludge adhered to crops or accumulation of toxic organics with low levels of degradability in the soil (Korentajer, 1991).
iii. *Heavy metals* particularly cadmium (which is relatively mobile in the soil environment), zinc, nickel and copper (which are phytotoxic) and molybdenum (toxic to livestock) are frequently found in high concentrations in sludge (Korentajer, 1991). Some heavy metals are preferentially sorbed to particles within the soil while others, such as Cd, Co, Se and Mo are bioaccumulated to toxic levels in plant tissue (Korentajer, 1991). Soil pH is the main factor controlling the solubility of heavy metals in soil (Fetter, 1993) and therefore determines the relative risk of various exposure pathways.

Groundwater may be one of the three important exposure pathways from sludge to humans, the other two being aerosol transmission and food chain contamination (Ekama, 1992).

Application rates and methods

Sludge may be applied as a dried cake or as a slurry (with up to 25% solids). In liquid form it may be irrigated onto the surface of the soil, by flood irrigation or spraying for example, drip irrigated or injected below the surface. Dry sludge cake is usually ploughed into the soil.

Nitrogen concentrations usually determine the season-by-season rate of application as this should correlate to agronomic rates (crop requirement). If excess soluble nitrogen is applied it is at risk of leaching to groundwater. Heavy metal concentrations in the sludge may restrict the cumulative loading from the point of view of soil contamination.

6.1.2.3 Sludge disposal in South Africa

It is estimated that around 288 000 tpa dry mass of domestic sewage sludge is produced each year in South Africa (Ekama, 1993). Total sludge production, including that resulting from industrial waste water treatment is likely to be over 385 000 tpa dry mass (Snyman, 1996). This quantity is expected to increase as the urban population increases, more homes are served with municipal sewers and higher quantities of waste water are produced as living standards increase.

The level of beneficial land application of sludge in South Africa is relatively low at 28%. It includes use in municipal parks and gardens (11% of total disposed dry mass of sludge), brick making (5.3%), on cultivated land (3%) and that sold to farmers (8.4%) (Ekama, 1992). Approximately 11% of sewage sludge can therefore be said to be used in agriculture. A further 11% beneficially used by municipalities may also pose a threat to groundwater quality.

6.1.2.4 South African Legislative Controls on Disposal.

The Department of Water Affairs and Forestry (DWAF) and the Department of National Health and Population Development (DNH&PD) both administer legislation controlling various aspects of the handling and disposal of sludge (McConkey, 1993). The DWA&F is guided by the DNH&PD with respect to disposal of sewage sludge and incorporates their recommendations in exemptions from the Water Act (Section 21) when these are required (McConkey, 1993).

Additional legislation relevant to the agricultural disposal of sewage sludge are the Environment Conservation Act No.73 of 1989, the Fertilizers, Farm Feeds, Agricultural Remedies and Stock

Remedies Act No. 36 of 1947 and the Conservation of Agricultural Resources Act No. 43 of 1983 (Ekama, 1993).

The current approach to water pollution prevention by non-hazardous substances is outlined in the DWA&F's Receiving Water Quality Objectives. This approach "*involves specification of the desired quality of the receiving water environment and the control of sources. The approach takes into account non-point and point sources of pollution to the degree necessary to maintain the desired quality. This approach recognises that the receiving water has a certain capacity to assimilate pollution without serious detriment to quality requirements of recognized users.*" (DWA&F, 1991).

A different approach is adopted for sludges which contain hazardous substances, that of pollution prevention (M^{c}Conkey, 1993).

Sludge is classified into four types by the DNH&PD (1991) depending on its source, how it is treated and the level of potentially harmful substances it contains (see Appendix 2.A for full details):

Type A Sludge from primary settling tanks, possibly mixed with other sludge types but which has received no treatment other than dewatering. It is unstable with a potentially high fly and odour nuisance and high levels of pathogenic organisms. This may be used for composting, landfill and tree plantations with certain restrictions.

Type B Waste activated sludge, humus sludge or sludge which has been digested. Generally some dewatering has taken place. It is stable with low fly and odour nuisance and reduced levels of pathogens. This may be used as above plus on crops (excluding vegetables), grazing, public gardens and parks, etc, and nurseries with certain restrictions.

Type C Sludge to which some specific sterilization treatment has been applied, eg. composting, pasteurisation, lime stabilization. It is stable with insignificant odour and fly nuisance and insignificant levels of pathogens. This may be used as above with certain restrictions.

Type D Tertiary sludge with low heavy metals content. It has a similar hygienic quality to Type C and metal contents are below stated limits. This may be used as above with certain restrictions plus on household vegetables and private gardens with certain recommendations.

The application rate for Type D sludge to agricultural land is restricted to a maximum of 8 tpa. ha^{-1} dry mass (DNH&PD, 1991). Table 4 in Appendix 2.A shows the maximum application rate for selected metals per annum and accumulated in soil. The maximum application rate of other types of sludge is therefore generally limited according to the metals concentrations and is restricted to a maximum of 8 tpa. ha^{-1}. In addition, the DNH&PD recognise that sludge application to agricultural land should not exceed the crop nitrogen requirements to prevent leaching to groundwater. Given the difficulties in accounting for available nitrogen from sludge and any inorganic fertilizers that may be used, they provide only a general guide for crop nitrogen demand.

6.1.3 Impact of agricultural sludge application on ground water quality

Contamination via the food chain is thought to be the pathway with the greatest risk of exposure to humans from beneficially applied sludge (Ekama, 1992). Groundwater contamination is frequently seen as less of a concern because of the low solubility of most sludge constituents and the low risk of leaching if sludge is applied within accepted limits, eg 8 tpa/ha at 5% solids (Ekama, 1992). However, even a moderate impact on groundwater quality may have significant environmental implications.

Research has been carried out, primarily in the US and Europe, on the impact of groundwater quality of sludge application to agricultural land and a summary of important findings is outlined below. (Many studies on the impact on groundwater quality of effluent irrigation of agricultural land are also relevant to sludge application due to the overlap in potential contaminants.)

Nitrate is the contaminant with the greatest risk of reaching groundwater at potentially harmful concentrations if sludge is applied at above agronomic rates.

6.1.3.1 Heavy metals

There is little evidence to suggest that sludge application to agricultural land results in increases in heavy metal concentrations in groundwater to levels which may give concern (Smith, 1996). One study shows an increase in zinc concentrations in groundwater from <40 μg l^{-1} to 500μg l^{-1} following application of sludge to agricultural land at very high loadings (407 kg ha^{-1}) (Grant & Olesen, 1984). However, these levels are still well below those recommended by WHO for aesthetic quality of drinking water at 5000 μg l^{-1} (Smith, 1996).

Metals have been shown to be highly spatially immobile, even under conditions which increase their solubility and bioavailability to crops (Smith, 1996). So while they may pose a problem as soil contaminants, particularly after long periods of sludge application, and there is a risk associated with bioaccumulation in crops (eg. cadmium) the literature suggests that impact on groundwater quality is minimal.

Conditions under which heavy metals are more likely to migrate to the water table (assuming a relatively shallow unconfined aquifer) are as follows:

- application of very high loadings of metals
- low soil and water pH
- low levels of clay, organic matter and sesquioxides (which adsorb metals) in the unsaturated zone (ie. low soil cation exchange capacity)
- the presence of mobile colloids (eg. microorganisms, organic macro molecules)
- the presence of inorganic (eg. ammonia) or organic (eg. fulvic acid) complexing agents in solution.

(From Fetter, 1992; Smith, 1996)

6.1.3.2 Microbiological Pathogenic Indicators

Pathogens are not thought to represent a major risk to groundwater quality as most are effectively retained in the surficial few centimetres of soil by colloidal matter and do not remain viable for the time that it would take to transport them to the groundwater table (Smith, 1996). The unsaturated zone acts as a mechanical strainer, sieving out the larger pathogens such as protozoa and helminths, as well as providing adsorption surface for viruses (Smith, 1996).

Certain hydrogeological conditions cause groundwater to be more susceptible to microbiological contamination. Microbes remain viable for longer in soils which are moist, rich in organic matter, have a relatively high pH and a low temperature and where they are out of direct sunlight (Gerba *et al*, 1975). There is greater risk of migration to groundwater where preferential pathways for vertical mass flow of sludge to the water table exist such as fissured rock or desiccation cracks (Smith, 1996). The composition of the sludge itself influences the degree of risk of pathogen contamination. In South Africa, where diarrhoeal diseases and intestinal parasites are particularly prevalent, waste water and therefore sludge (which has not been disinfected) generally contain large numbers of pathogens (Idema and Kfir, 1994).

Microbes are more likely to reach the water table if, at some stage in their life cycle, they are persistent and small. A good example of this is *Clostridia* which forms spores.

6.1.3.3 Nitrogen

Of all the potentially contaminating substances found within sewage sludge, **nitrate** has been shown to have the most significant impact on groundwater quality (Smith, 1996).

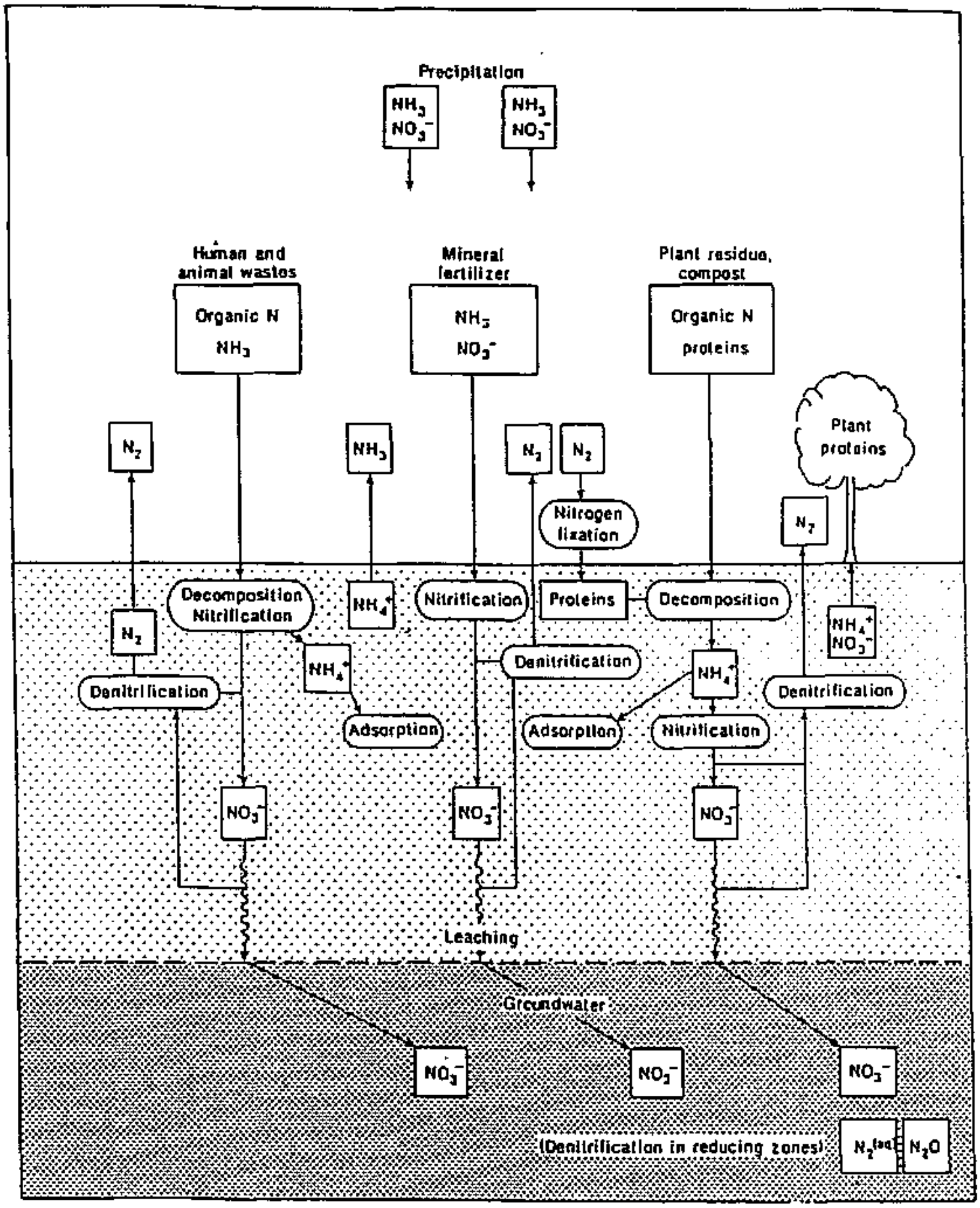
Nitrate is a cause for concern in drinking water at levels greater than 10 mg l^{-1} (NO_3- N). At concentrations greater than this there is an increased risk to humans, particularly infants, of methaemoglobinaemia, commonly known as Blue Baby Syndrome (Freshwater Foundation, 1988). The mortality rate among affected infants is thought to be around 10% (Freshwater Foundation, 1988). Nitrate is converted by microbial activity in the low pH conditions of the stomach to nitrite. Once nitrite enters the blood stream it changes haemoglobin to methaemoglobin which reduces the capacity of the blood to carry oxygen.

Nitrogen is found in all soils. Usually in excess of 90% of soil nitrogen occurs in organic forms (Vinten and Smith, 1993). Nitrogen from the atmosphere is naturally fixed (to organic nitrogen) by soil microfauna and leguminous plants at a rate which may be as high as 40 kg $N.ha^{-1}$. yr^{-1} (Goulding, 1990).

Nitrogen occurs in sludge in organic and ammoniacal form. A characterisation of South African sludges carried out in the late 1980s showed the concentration of total Kjeldahl nitrogen (TKN) (organic and ammoniacal nitrogen) to be between 20 and 45 mg N. Kg^{-1} (Smith and Vasiloudis, 1989). These forms of nitrogen have a relatively low solubility and positively charged ammonium ions bind to exchange complexes on clay minerals and organic matter (Smith, 1996). Organic nitrogen is not available for plant uptake and is essentially immobile. However, once applied to the soil, microbial action mineralizes organic nitrogen to ammonia, which is fixed by microbial action to ammonium and then nitrified to nitrite and nitrate (see Figure 6.1). This is the process of slow release of immobile organic nitrogen to that which is available for plant uptake, leaching to groundwater and run-off or release to the atmosphere.

Mineralization of organic nitrogen to inorganic ammonium form is carried out under a wide variety of environmental and edaphic conditions by a large diversity of soil micro-organisms (Wild, 1988). An important control on the rate of ammonium mineralization is the ratio of C:N in the organic matter (Smith, 1996). Nitrogen is more mineralizable with lower ratios of C:N. Animal waste with a ratio of 5:1 is highly mineralizable, activated sludge with a ratio of >10:1 is mineralized at an average rate and sludges or organic matter such as straw with a ratio of >15 are very slow to mineralize and immobilize residual soil N until some C has been released from the soil environment, usually respired as CO_2 (Smith, 1996).

The rate of mineralization of organic nitrogen to ammonium increases with soil moisture content between wilting point and field capacity (Vinten and Smith, 1993). However, once field capacity is exceeded the rate falls due to restricted aeration of the soil. Mineralization increases in soils with high pH values (Vinten and Smith, 1993).



Chemical symbols: N, elemental nitrogen;
N_2, nitrogen gas;
N_2O, nitrous oxide;
NO_2^-, nitrite;
NO_3^-, nitrate;
NH_3, ammonia;
NH_4^+, ammonium;
$N_2(aq)$, nitrogen gas; dissolved in water.

Figure 6.1: The nitrogen cycle *(adapted from Freeze and Cherry, 1979)*

Once mineralized, ammonium may volatilise or remain in the soil, possibly to be nitrified. Nitrogen losses by **volatilization** will proceed more rapidly at higher temperatures, lower soil moistures and higher air speeds (Kreitler, 1975).

Nitrification is carried out by nitrifying bacteria, commonly *Nitrosomonas* and *Nitrobacter*. The rate of nitrification is more sensitive to environmental factors than that of ammonium mineralization and is influenced by (from Smith, 1996; Krumbein, 1983):

- Soil temperature. Generally the rate of nitrification increases with increasing temperature to an optimal rate at 30 - 35° C. Very little nitrification occurs in soils below 5°C.
- Aeration. Nitrification is an oxidising reaction and requires aerobic conditions.
- Soil moisture content
- pH. Nitrifying bacteria are inhibited by the acid they produce and therefore the rate is optimised in neutral to alkaline soils.

Nitrification inhibitors, such as dicyandiamide (DCD), are applied in some parts of the world in an attempt to slow down nitrification and consequent nitrate losses. The effectiveness of inhibitors has been shown to be highly variable and sensitive to environmental conditions such as soil temperature (Smith, 1996). DCD degrades more rapidly under warm soil conditions.

Subsequent **leaching** is affected by the relative rates of plant uptake, denitrification, losses of N_2 to the atmosphere and percolation rates (dependent on rainfall or irrigation).

Reported rates of nitrogen leaching are very variable from insignificant amounts to 59% of nitrogen applied (Shepherd, 1993). If sludge is applied to uncovered ground or available nitrogen exceeds crop requirements, then loss of nitrate by leaching generally occurs as a linear function of the increasing rate of applied sludge (Smith, 1996). In clayey soils this loss occurs above a threshold limit of around 120 kg N ha^{-1}. Nitrate losses from clayey soils are expected to be lower than sandy soils due to the higher soil water retention. The presence of macro-pores in clayey soils can reverse this trend however (Smith, 1996). Several studies have shown that the percentage of nitrate leached increases with increasing total load of nitrogen applied (Vinten and Smith, 1993).

Table 6.2 shows the predominant forms of nitrogen in different types of sludge and the percentage available during the first cropping season. Liquid digested sludge contains mainly ammoniacal nitrogen and 60% is available during the first cropping season. There would therefore be a higher risk of leaching if this sludge were applied to bare ground during a period of precipitation. However, with slow N-releasing sludges there is a risk of leaching N to groundwater under different conditions. If they are applied to crops that require N for a limited growth period the remaining N will accumulate as nitrate in the soil until leaching or denitrification occurs. One study using a 50 year simulation model compared the relative leaching impacts of different types of sludge used as a fertilizer and found that less stable sludges (undigested liquid) provided the most N for crop uptake and resulted in the highest long term leaching rates (O'Brien and Mitsch, 1980).

In summary, nitrogen is more likely to leach to groundwater under the following conditions:

- High rates of nitrogen application due to high content in the sludge and/or high bulk loadings.
- Application of high N-availability sludge or of low N-availability sludge (digested cake) to crops with a seasonally limited N demand.
- Application of sludge into the soil, reducing the losses of N as ammonia volatilization.
- Low ratios of C:N increasing the availability of nitrogen for mineralization.
- High soil temperatures increasing the rate of nitrification (and inhibitor breakdown if these are applied).
- Well aerated soil encouraging nitrification. (Denitrification is likely to occur in anoxic conditions.)
- Low levels of plant uptake due to bare ground, low crop requirements or seasonally variable requirements.
- High levels of precipitation or irrigation
- High vertical permeability (including clayey soils with macro-pore channels for rapid downward migration).
- Shallow, unconfined water table.

Nitrate in groundwater and the unsaturated zone is subject to **reduction** (primarily by heterotrophic bacteria) under anoxic or very low oxygen conditions in the presence of an electron donor such as organic carbon (Spalding *et al.*, 1993). Maximum levels of dissolved oxygen of between 2 mg l^{-1} and 0.3 mg l^{-1} have been quoted in the literature (Edmunds, 1972; Postma, 1991). Chemodenitrification may occur if the redox potential of the groundwater is low enough

(Freeze & Cherry, 1979). The reduction process is defined as **denitrification** when nitrogenous gas is produced (Krumbein, 1983).

In addition to redox potential and the availability of carbon, the rate of denitrification is also dependent upon the concentration of nitrate. High rates of unsaturated zone denitrification, >0.2 kg ha^{-1} d^{-1} are recorded where soil nitrate concentrations exceed 5 mg kg^{-1}. (Vinten and Smith, 1993).

The concentration of nitrate in groundwater is primarily dependent on the following factors (Burt *et al*, 1993; Krumbein, 1983; Freeze & Cherry, 1979):

- The concentration of nitrate leached to the water table
- The relative proportions of contaminated leachate and uncontaminated lateral inflow
- The rate of convective groundwater flow
- The rate of hydrodynamic dispersion
- The rates of reduction and denitrification (also dependent on redox conditions in the saturated zone)

Nitrate is rarely reported in fully confined aquifers due to the reducing conditions which tend to exist.

6.1.3.4 Other nutrients and major ions

Dissolved organic carbon (DOC) is expected to be leached from sludge amended land as the sludge itself has high levels of organic carbon.

Typically higher levels of most major ions would be expected in leachate from sludge compared to recharge percolating from precipitation or 'clean' irrigation recharge. The level of TDS and electrical conductance of groundwater are therefore parameters that would be influenced by contamination from sludge. The degree to which pH is affected depends on the buffering capacity of groundwater and soil water at the site.

Phosphorus levels in sludge are typically half those of nitrogen, however only 10 to 25% as much phosphorus is required by crops compared to nitrogen (Smith, 1996). Excess phosphorus presents an environmental problem as its presence in bioavailable forms may result in eutrophication. However, there is no evidence in the scientific literature that phosphates are leached to groundwater at significant concentrations following sludge application to agricultural land (Smith, 1996). Phosphorus in soil is relatively immobile and insoluble. South African soils are typically deficient in phosphates therefore the likelihood of exceeding the phosphate retention capacity of soils here is even lower.

6.1.3.5 Trace Organics

In addition to the load of organic waste matter present in sewage sludge, industrially manufactured trace organic chemicals may also occur. The type and concentration of these chemicals depends on the industries operating within the WWTP catchment area, chemicals used domestically and those found in the environment which enter the sewerage system in road run-off. The latter may include organics such as polyaromatic hydrocarbons (PAHs) which are released during combustion of fossil fuels (Halsall *et al.*, 1993). However, in South Africa road run-off is not usually fed into the sewerage system for waste water treatment.

Trace organic pollutants with a moderate to high leaching potential which are commonly found in sludge include: organochlorine pesticides, monocyclic aromatics, chlorobenzenes, short chained halogenated aliphatics, aromatic and alkyl amines, phenols (Smith, 1996). The fate of organic pollutants in sludge amended soils has been investigated only to a limited extent, partly due to the labourious and complex analytical techniques required (Smith, 1996).

A study carried out on a sandy coastal plain aquifer in Israel (Muszkat *et al.*, 1993) found that the aquifer underlying a sewage-effluent irrigated citrus grove was contaminated with various organic pollutants. The pollutants, including toluene and phthalates which are generally considered biodegradable, migrated through 20 m unsaturated zone and their concentrations were found to increase with depth. It is postulated that, under field conditions trace organics are more persistent than would be expected from laboratory data and their mobility may be enhanced by the sewage itself. The presence of detergents, solvents and hydrocarbons and stable colloidal fractions may increase the mobility of certain trace organics and these and other pollutants may reduce the levels of microbial activity in the unsaturated zone thereby limiting biodegradation.

6.1.4 Objectives of this specialist study

The main objective of each specialist investigation carried out as part of this study is to collect data and information in order to confirm that groundwater contamination occurs as a result of the agricultural practice of concern and to collect evidence which can be used to support the non-technical booklet (Progress Report no.3, April 1996).

Limited research has been carried out on the agricultural use of sewage sludge within South Africa (Palmer, 1989; Easton, 1985) and very little data has been presented in the literature on the impact on groundwater quality in this country. It was therefore decided that this practice should be investigated in the field to determine whether the occurrence of groundwater contamination associated with beneficial use of sludge in South Africa concurred with results reported from other countries.

The objectives of the sewage sludge study were as follows:

- to select 3 sites for investigation where vulnerable aquifers (representative of the South African situation) underlie areas where sewage sludge has been applied to agricultural land
- to determine whether contamination of the groundwater had occurred as a result of sewage sludge application
- to determine, insofar as is possible within the budget and time constraints of the project, the nature and extent (spatially and temporally) of groundwater contamination
- to recommend to farmers using sewage sludge as a fertilizer how they can best avoid groundwater contamination and protect their resource.

6.1.5 Approach

6.1.5.1 Field Site Selection

Literature covering similar studies conducted worldwide was reviewed to enable the researcher to conduct the South African study in the most practicable and efficient way. WWTP operators around the country were contacted to determine where sludge was being supplied to the agricultural sector or used on cultivated land operated by the WWTP. Following discussions with farmers and WWTP operators and visits to assess potential sites where sludge was known to be used as a fertilizer, 3 sites were selected for groundwater investigation based on the following criteria:

- the sites should be underlain by a relatively shallow unconfined aquifer vulnerable to contamination from the surface
- sites should be representative of a range of sludge types, crop types and hydrogeological settings
- the land owner(s) should be supportive of the study and willing to provide relevant information and access
- the sludge supplier should be supportive of the study and willing to provide relevant information
- groundwater monitoring points should be established in the area (where possible) to minimise drilling costs
- relevant information on the historical and current pattern of surrounding land-use should be available to determine other potential sources of groundwater contamination.

The aim was therefore to select sites where the greatest risk of groundwater contamination existed and where goodwill enabled the study to have access to relevant available data.

Figure 6.2 shows the location of sites investigated as part of this study.

6.1.5.2 Data acquisition

Where available, the following sources of data were acquired to give as complete an understanding as possible of the hydrogeology and contamination pathways at each site:

- Geological and hydrogeological literature on the locality
- Engineering/ geotechnical/ geophysical/ drilling reports on the locality
- Local data from the National Groundwater Database and National Water Quality Database
- Information on the local climate and precipitation
- Results of soil and water analyses carried out by previous researchers and land owners in the locality
- Irrigation schedule
- Quantities and type of sludge used (results of sludge analyses)
- Crop types

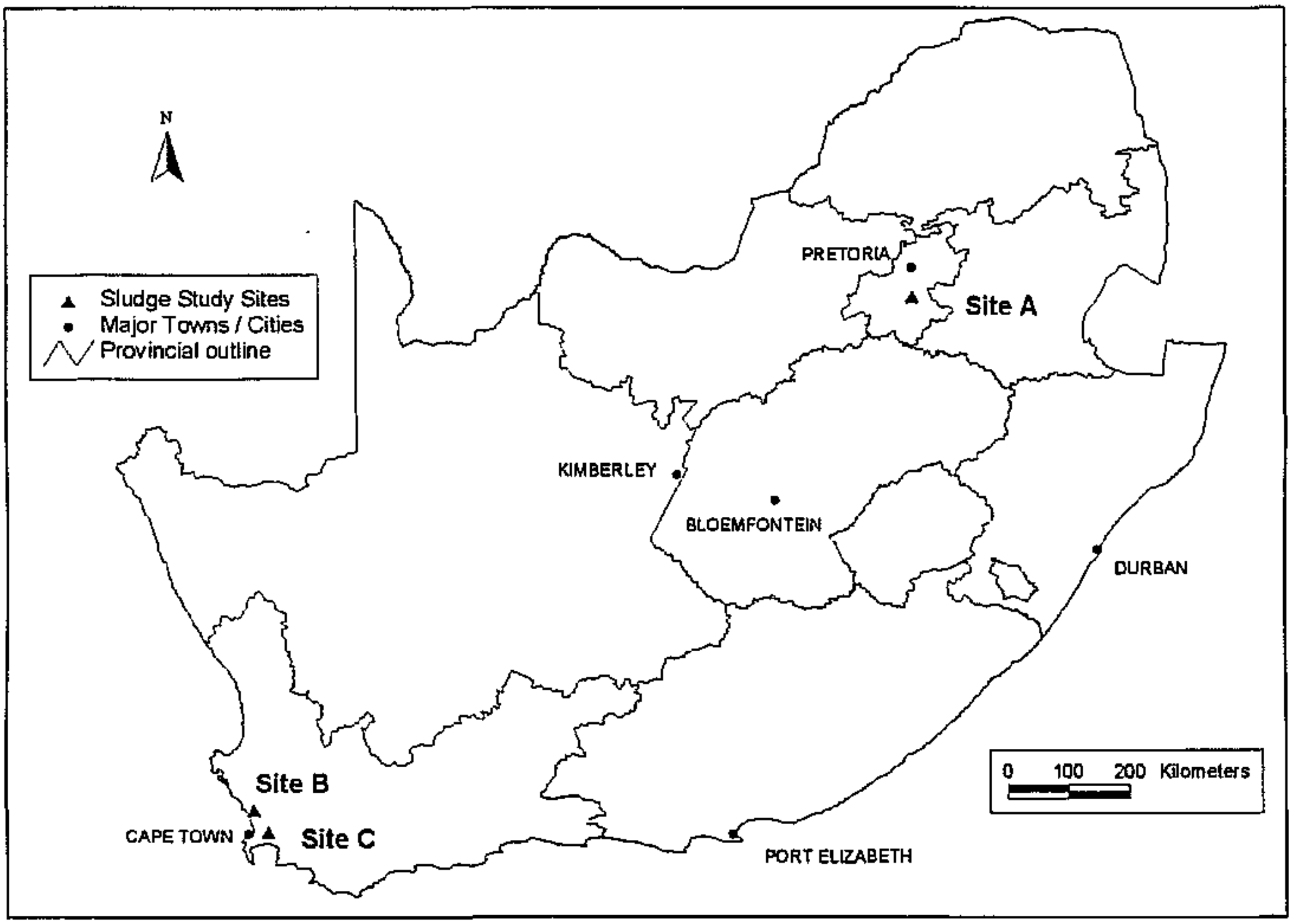


Figure 6.2: Location of study sites

Land-use on the sites and surrounding areas was assessed for other potential sources of groundwater contamination and these were taken into account when interpreting hydrogeological data.

6.1.5.3 *Sampling strategy*

Groundwater monitoring points were established, where necessary, to ensure that representative samples of potentially contaminated groundwater could be collected and that one or more control sampling points were available. New boreholes were constructed using techniques to ensure that no potentially contaminating substances were introduced into the aquifer with experienced technicians carrying out the installation under the supervision of a hydrogeologist.

Where existing boreholes were used as monitoring points, information on their construction, history and performance was gathered from the owners to enable as complete an understanding as possible of the sampling conditions.

Groundwater sampling protocol was followed (as outlined in Weaver, 1992) to ensure that samples representative of water in the aquifer were analysed. This included careful use of sterile sample containers, purging at least 3 well volumes prior to sampling and storage for minimal periods of time (less than 48 hours) in the dark at temperatures of less than 4°C.

More than one phase of sampling was carried out at each of the sites. At sites A and C sludge is applied continuously and there is a long history of application. The first phase of sampling at these sites confirmed contamination of the groundwater and the second phase was carried out over a wider area to determine the extent. Site B had a limited and well documented history of sludge application. Sludge was to be applied over a new area at the beginning of the growing season therefore samples were taken over time to represent groundwater quality prior to sludge application, during and after the growing season (which was coincident with the period of recharge).

Soil samples were also collected at specific depths in the unsaturated zone and analysed for water soluble and potassium chloride extractable nitrate and ammonium by the Agricultural Research Council's Infruitek laboratory using standard methods.

6.1.5.4 Determinands

Field measurements of groundwater pH, EC and temperature were carried out at each sampling point. The colour and smell of the groundwater was also noted.

A suite of determinands for laboratory analysis was selected to include major ions usually found in groundwater and substances known to occur in sludge which have a reasonable degree of subsurface mobility and solubility. A wide range of determinands was analysed in the first instance at each site, including less soluble heavy metals and less mobile microbiological indicators.

Hydrochemical parameters:	pH, EC, alkalinity, temperature, COD.
Major ions:	Ca, Mg, K, Na, HCO_3, SO_4, Cl.
Common potential contaminants:	NO_3, NH_4, TKN, DOC.
Heavy metals:	Fe, Mn, Al, Zn, Cu, Cd, Ni, Pb.
Pathogens:	Faecal coliforms, faecal streptococci, heterotrophic plate count, *Clostridia*.

Subsequent analyses from monitoring points over a wider area or longer period of time were carried out on a more limited range of determinands shown to be relevant in that hydrogeological setting.

The hydrochemical analyses were carried out at accredited CSIR laboratories using standard methods.

Trace organics were not analysed for as data on those contained within the sludge were not available and the analytical process is labourious and costly.

6.1.5.5 Nitrogen Isotopes

The use of sludge as a fertilizer was not the only potential source of nitrate identified at the study sites. Nitrogen isotope analyses were conducted in an attempt to determine which source contributed most significantly to elevated levels of groundwater nitrate.

Seven natural isotopes of nitrogen are know to exist. Five of those, ^{12}N, ^{13}N, ^{16}N, ^{17}N and ^{18}N, are radioactive and have half lives of less than 12 minutes (Fritz and Fontes, 1980). ^{14}N and ^{15}N are stable isotopes. Most nitrogen occurs as ^{14}N with only 0.37% of atmospheric nitrogen occurring as ^{15}N.

The seven percent difference in the atomic masses of the two stable isotopes leads to isotopic fractionation in natural systems. Thus, the isotopic ratio of a nitrogen compound is the result of the following processes (from Kreitler, 1975):

- physical fractionation (diffusion, evaporation and sublimation)
- chemical equilibrium fractionation (equilibrium reactions)
- chemical kinetic fractionation (non-equilibrium reactions)
- the isotopic ratio of the source material.

The ratio of stable nitrogen isotopes in a given sample is compared to a standard (atmospheric nitrogen) as shown below and quoted as $\delta^{15}N$ (parts per thousand) (Heaton *et al*, 1982).

$$\delta^{15}N = [\,(^{15}N/^{14}N\ sample\ -\ ^{15}N/14N\ standard)\ .\ (^{15}N/14N\ standard)^{-1}]\ .\ 10^3$$

A positive $\delta^{15}N$ value indicates ^{15}N enrichment and a negative value ^{15}N depletion relative to natural atmospheric proportions of the isotopes.

Variations in natural abundance of stable nitrogen isotopes can semi-quantitatively differentiate agronomic from organic waste sources of nitrate in groundwater (Spalding *et al.*, 1993). As a general rule, manufactured (inorganic) fertilizer materials tend to have $\delta^{15}N$ values close to natural atmospheric abundance (ie. $\delta^{15}N$ is close to zero) as they are manufactured from atmospheric gas. Organic matter in soils is typically enriched in ^{15}N giving a $\delta^{15}N$ of +3 to +10 (Keeney, 1986; Heaton, 1986). Animal and human wastes have been found to be slightly to highly enriched in ^{15}N as a result of biochemical fractionation and subsequent ammonia volatilization of the waste. Nitrogen from animal and human wastes typically give $\delta^{15}N$ values of + 7 to +25 (Kreitler, 1975; Heaton, 1986). Therefore isotope data was thought to be useful to this study in differentiating between nitrogen naturally occurring in the soil and that from inorganic fertilizers and sewage sludge applied in agricultural areas.

Samples were collected for nitrogen isotope analysis using recommended procedures including mercuric chloride preservation (Weaver, 1992; Talma, pers. comm.). The analyses were conducted at the CSIR isotope laboratory using standard methods as described in Heaton and Collett, 1985.

6.1.5.6 *Data quality*

The quality of hydrochemical data was analysed to ensure that the results were meaningful and representative of water in the aquifer. Quality control samples were routinely run during analysis. The ionic balance of the hydrochemical results was determined to confirm that the results were within an acceptable margin of error (<5%) and a comparison of pH and EC measured in the field and in the laboratory was made.

6.1.5.7 *Interpretation*

Results of hydrochemical, microbiological and isotope analyses were analysed in conjunction with information from literature, soil analyses, drilling records and anecdotal information on the local hydrogeology to determine the impact of sludge application on groundwater quality at each site.

6.1.5.8 *Limits of the project*

The study aims to examine examples of the impact on groundwater quality in 'real' situations, ie. where sludge is being applied in working agricultural situations. In looking at examples from 'the real world' an element of scientific control is sacrificed.

Often the exact history of the site is not known, the degree of variability of the composition of the sewage sludge, the exact loading rates, etc. The influence of other potential sources of groundwater contamination is also difficult to account for.

Budget, time and access constraints limit the density of sampling points at each study site and data must be extrapolated, often over tens to hundreds of metres, between data points. The degree of hydrogeological heterogeneity at each site is not fully accounted for in the simplified summaries of results.

A greater density of sampling points would increase the quality of the data, particularly for nitrogen isotopes.

In many cases where existing boreholes are used to sample, the depth and thickness of the aquifer sampled are not known. This makes comparison between sampling points difficult.

Soil sampling has been carried out to determine nitrate and ammonium profiles in the unsaturated zone. This sampling gives a snap-shot with no indication of spatial or temporal variability. Detailed data on the soil moisture, grain size distribution and redox conditions would enable a better understanding of unsaturated zone processes affecting leaching of nitrate. However the primary aim of this project is to determine whether groundwater contamination has occurred, rather than the detailed modelling of its occurrence.

6.2 SITE A

6.2.1 Site background

This site was selected for the following reasons:

- it has a long history of sludge application and therefore a high potential risk of contamination
- it is an example of the application of liquid sludge at high loadings
- available data indicated the site to be underlain by a shallow dolomitic aquifer with a relatively permeable unsaturated zone
- the aquifer in this area is representative of one of the highest yielding aquifers in South Africa
- groundwater abstraction is extensive in the surrounding area and therefore many sampling points are available
- no other soil additives or fertilizers are reportedly used in the sludge applied area (although they are used extensively in the surrounding agricultural area).

The site is managed by the WWTP operator as a means of beneficial sludge use/disposal. It is located in Gauteng, as shown in Figure 6.2 and is surrounded by predominantly agricultural land with heavy industry and mining established within several kilometres.

The WWTP has been in operation since 1933. It has a capacity of 36 Ml but is currently operating at 10 to 15 Ml. Approximately 90% of waste water treated at the plant comes from industrial sources. These include food factories, metal plating plants and chemical producers. As a consequence the waste water is high in metals, particularly Fe and Mg, and the plant has to cope with frequent industrial spillages into the system.

6.2.2 Application of sewage sludge to the site

Sludge from the primary settling tanks and anaerobic digesters has been flooded onto adjacent agricultural land managed by the WWTP operator for more than 15 years. Sludge with a low percentage of solids is transferred by gravity flow to a storage lagoon and then in open irrigation ditches to the agricultural area neighbouring the WWTP. The total agricultural area is approximately 60 ha. Sludge is flooded onto a 10 ha paddock area daily for a period of several months (typically 3 months) and then ploughed into the soil prior to planting fodder crops (predominantly rye grass with some mealies). A new 10 ha area is then irrigated with sludge for a similar period. During the growing period the crop is irrigated with treated waste water.

An analysis of liquid sludge sampled from an irrigation ditch is presented in full in Appendix 2.B. A comparison of selected constituents found in the sludge and their concentrations in heavily contaminated groundwater (sampled from borehole 18 on the edge of the sludge applied area) and less contaminated groundwater (sampled from borehole 10) is shown in Figure 6.3. Additional potentially contaminating substances from the sludge include heavy metals and microbes. The results of heavy metals analyses carried out by the WWTP operator on sludge samples are discussed in more detail below.

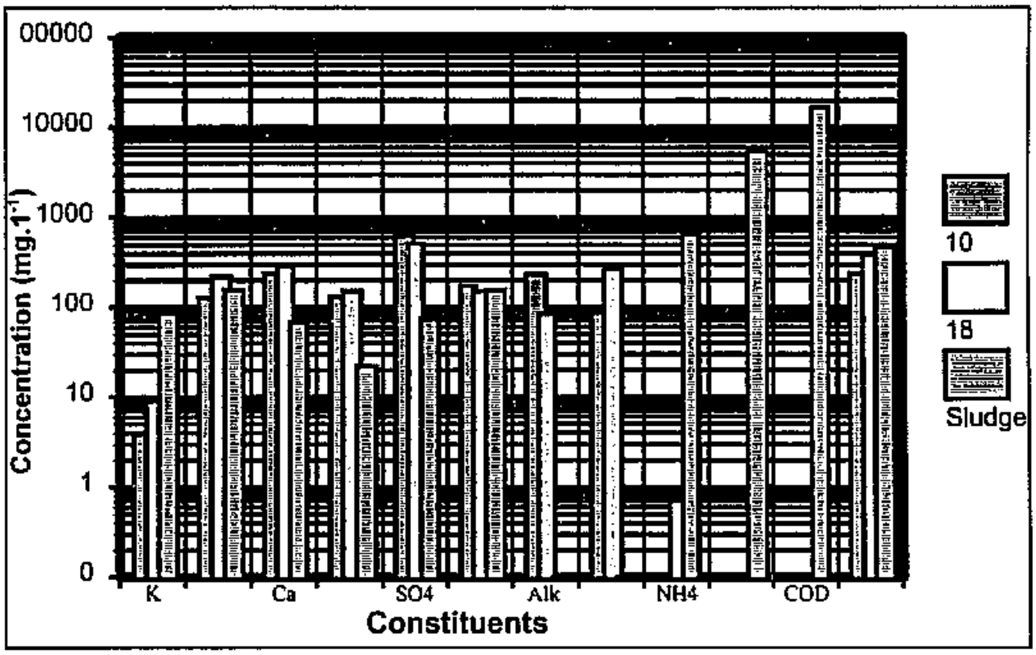


Figure 6.3: Site A: Comparison of selected sludge and groundwater constituents

Of the selected constituents shown, those which could have a significant effect on groundwater quality are those which occur in the sludge at an order of magnitude greater than is seen in groundwater from borehole 10. These constituents are K, NH_4, TKN and COD. The graph shows that COD levels are not increased in groundwater and this parameter is therefore effectively attenuated before reaching the aquifer. TKN and NH_4 are not detected at significant levels in groundwater but are likely to be the source of elevated groundwater nitrate levels following mineralisation and nitrification in the soil. The slightly elevated level of K in the groundwater from borehole 18 may be contributed by the sludge.

6.2.3 Land-use and potential groundwater contamination around the study area.

Figure 6.4 shows the pattern of land-use around the sludge-applied area. Intensive vegetable farming (lettuce, cabbage, beetroot, carrots etc.) is carried out to the north of the sludge applied area. The area is irrigated for between 8 and 12 hours a day with borehole water and intensively fertilized with inorganic fertilizers such as ammonium sulphate, Superphosphate and Limestone Ammonia Nitrogen (LAN) with N:P:K ratios of 2:3:4 and 2:3:2. High nitrogen loadings of up to **750 kg N .ha^{-1}.yr^{-1}** are reportedly applied by the local farmers giving them 2 to 3 crops per annum. No organic fertilizers are used in the vegetable farming area and there is no livestock. Pit latrines are used by some of the farm labourers.

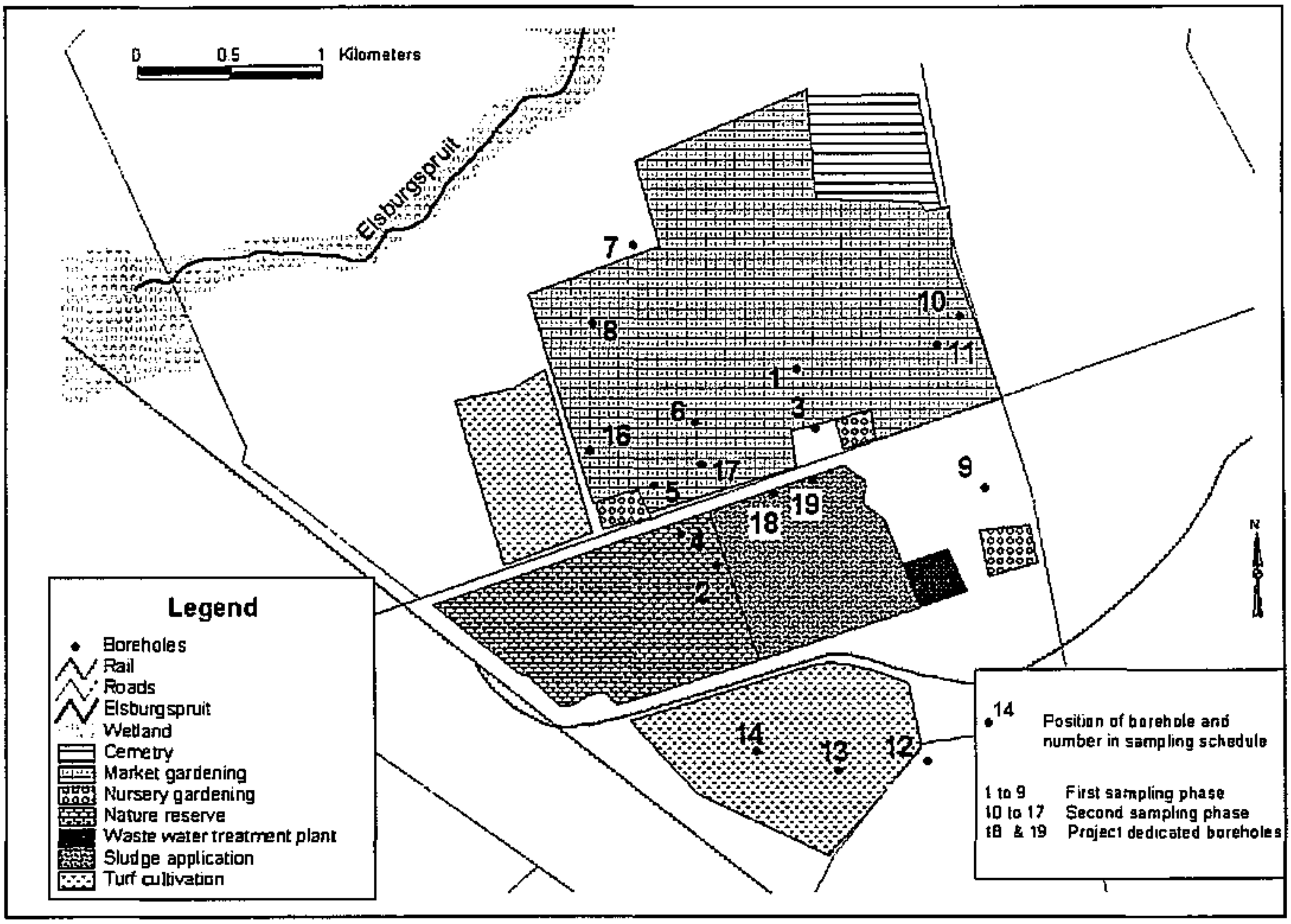


Figure 6.4: Site A: Surrounding land-use and sampling borehole locations.

Nurseries also operate close by cultivating vegetable seedlings, garden plants and trees. They use mainly inorganic with some organic fertilizers and irrigate with borehole water. One nursery to the east of the plant uses some treated water from the WWTP. Most of the cultivation is carried out in raised beds and so there is possibly less risk of infiltration to groundwater of fertilizers compared to the intensive vegetable farming.

Turf is grown in several areas around the plant and irrigated with borehole water. 400 kg.ha^{-1} of inorganic fertilizers are applied three to four times per annum. The most commonly used fertilizers are LAN and N:P:K fertilizers with ratios of 5:1:5 and 2:3:2 respectively.

A bird sanctuary is located to the west of the sludge applied area incorporating two lagoons that receive treated effluent from the WWTP. This treated water is likely to recharge the aquifer. Industry and mines have developed within 4 km of the sludge applied area.

In summary, potential sources of groundwater contamination in the area are likely to include:

iv. inorganic fertilizers
v. storm drains around the WWTP and along the road
vi. airborne pollution from nearby industry (especially SOx)
vii. use of treated waste water for irrigation and at the bird sanctuary lagoons
viii. cemetery
ix. industrial and sewage contamination of the Elsbergspruit river and possible infiltration from slimes dams
x. pit latrines at the farms in the area.

Sources i, to iii and v would result in contamination from inorganic sources whereas sources iv to vii could result in groundwater contamination from organic sources similar in character to the sludge.

6.2.4 Hydrogeology of the site

The sludge applied area is underlain by an unconfined dolomitic aquifer at approximately 7 m depth. The unsaturated zone is comprised of moderately permeable soil with weathered dolomite.

The site is underlain by Malmani subgroup dolomite of the Transvaal sequence (Vaalian age). The geological background to the area given below is taken mainly from Foster, 1986.

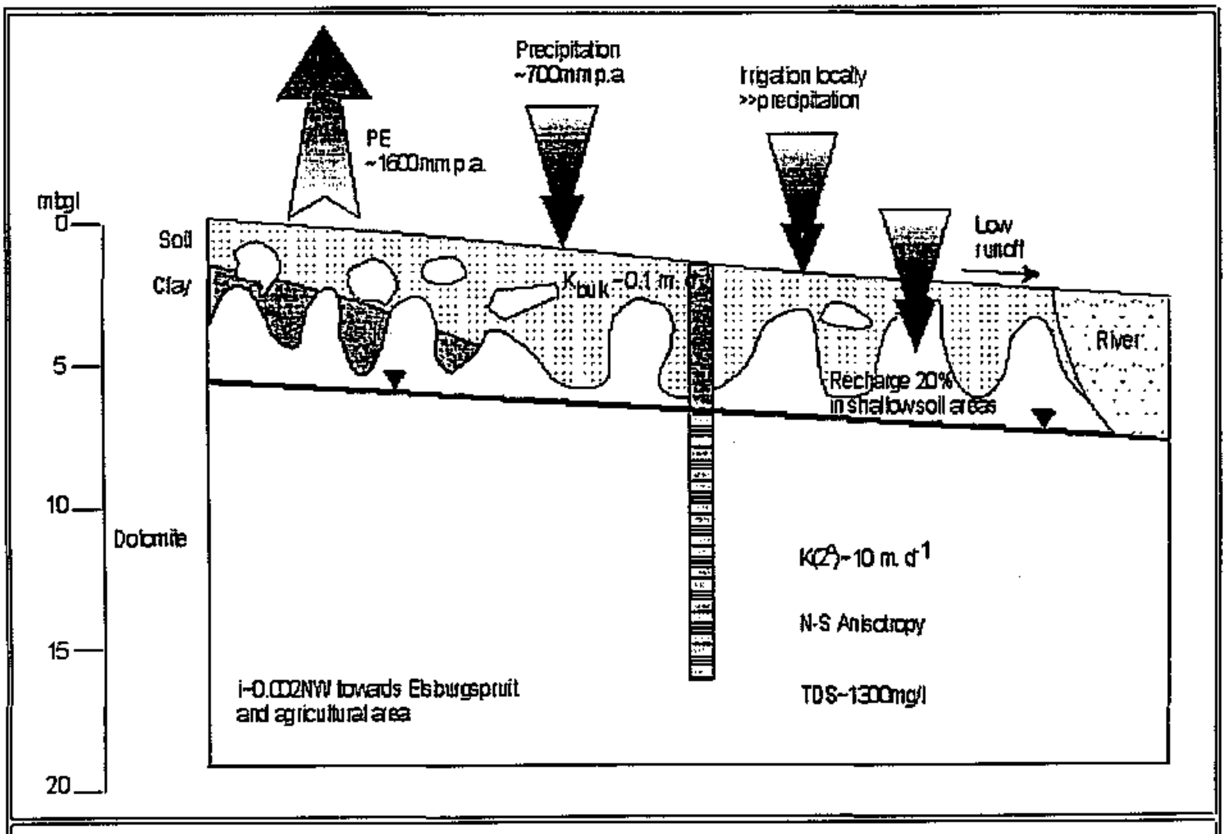
The Malmani dolomite is heterogenous with chert rich bands (creamy white with oxide staining forming the Eccles and Monte Christo formations, and chert poor bands (blue-grey with elephant skin weathering) forming the Lyttleton and Oaktree formations. The chert rich dolomite reportedly forms the best aquifers due to closer joint spacing and a higher proportion of calcite resulting in greater dissolution. Both the Oaktree and Eccles formations contain some pyrite which may be oxidised to sulphate. The average thickness of the Malmani subgroup in this area is around 1000 m. Joint alignment in the Elsbergspruit area is predominantly NNW-SSE.

Figure 6.5 gives a simplified summary of the **hydrogeology** of the site. Geotechnical investigations have been carried out at the site to determine sinkhole risk (Hill, Kaplin & Scot, 1985) therefore reasonably detailed information is available on the shallow subsurface around the WWTP area. The depth to the dolomite in the vicinity of the WWTP varies from 2 to 13 m. Sinkhole related subsidence occurred at various points around the plant and access road in the late 1980s. Records show that rest water levels in on-site geotechnical boreholes around the WWTP occur at between 8 and 18 mbgl and vary seasonally by up to 2 m. Groundwater strikes were encountered at around 7 mbgl during drilling of the sampling boreholes on the edge of the sludge applied area.

Rainfall occurs in the summer months and totals approximately 700 mm p.a. Studies have shown **recharge** to be between 2 and 28% (Foster, 1986). A higher range of 20 to 25% is estimated for areas where the soil cover is thin (<20 cm) overlying chert rich dolomites. An average of 30% of irrigation water is estimated to infiltrate to the water table.

Bulk **permeability** of soils in the area is around 0.01 $m.d^{-1}$ (Foster, 1986). The clay content is between 10 and 30% and the soils are red in colour indicating significant iron. Permeability of the dolomite is thought to be anisotropic and heterogenous with an average value around 10 $m.d^{-1}$ (Foster, 1986).

Groundwater flow directions are difficult to determine from the data available. Rest water level measurements during this study were taken only at the project dedicated boreholes (18 and 19). The other sampling points were not accessible at the well head and, even prior to pumping, may be influenced by other boreholes pumping close by. None of the boreholes sampled had been accurately surveyed. RWL measurements given by DWA&F (Foster, 1986) are only accurate to within 1 m and the collar elevation is frequently estimated from the position of the borehole.



Sources:

DWAF, 1986. Geohydrological Evaluation of the Klip River-Natalspruit Dolomite area

Hill Kaplan Scott, 1985. Dolomite Stability Investigation

Figure 6.5: Conceptual hydrogeology of Site A

Groundwater flow in the study area is thought to be generally north-west, towards the Elsbergspruit, however a cone of depression appears to exist in the high abstraction agricultural area to the north of the WWTP. Flow beneath the sludge applied area is therefore assumed to be northwards, however, during periods of high irrigation and sludge application a recharge mound may exist. Similarly, continued recharge from the lagoons at the bird sanctuary may result in a water table mound locally reversing flow gradients.

Streams in the area are thought to be in hydraulic continuity with groundwater as stream water levels are coincident with RWLs in the vicinity. In addition, levels of groundwater TDS increase towards the Elsbergspruit which has a TDS level in excess of 1000 mg.l^{-1} (Foster, 1986). The cause of the high TDS levels in surface water is reported to be effluent from nearby mining activities (ERPM, ERGO) and sewage effluent (Foster, 1986). However this assumes that the river-aquifer relationship is influent whereas the piezometry of the area indicates that groundwater flow is towards the river. Seasonal infiltration to groundwater may explain the hydrochemical influence of surface water.

The **hydrochemistry** of groundwater in the area around the WWTP and towards the Elsbergspruit is not typical of dolomitic aquifers and indicates pollution in the area. To the south and east of the study area TDS levels are recorded at 300 to 500 mg l^{-1} whereas to the north and west of the study area the results of previous studies show that TDS levels are elevated to around 1000 mg l^{-1} (Foster, 1986; Walton *et al*, 1993).

Previous studies have attributed elevated NO_3 and SO_4 levels in the area to diffuse agricultural sources in the area, particularly the use of fertilizers (Walton *et al*, 1993). Higher concentrations of these contaminants are found in agricultural areas and are not associated with urban or industrial sources. However, contamination of the Elsbergspruit by mine effluent is reported (Foster, 1986) and, if it is in continuity with the dolomite aquifer, this may account for some of the sulphate contamination. Anecdotal evidence also suggests sporadic contamination of the Elsbergspruit with raw sewage.

6.2.5 Field investigation

The field investigation was carried out between March 1996 and January 1997. Figure 6.4 shows the layout of groundwater sampling points. Sampling sheets giving details of the borehole location, sampling methods, borehole design, etc., are included in Appendix 2.B. The sampling programme was carried out as follows:

- an initial phase of groundwater sampling from existing boreholes in March 1996 in the surrounding area, both up and down gradient, to a distance of 2 km to establish the nature and extent of contamination in the area. Boreholes 1-9 were sampled. During this preliminary phase, groundwater samples were analysed for all the determinands listed in section 6.1.5.4 including heavy metals and pathogenic indicators.

- a second phase of groundwater sampling in August 1996 from existing boreholes in the surrounding area, both up and down gradient, to a distance of 2 km to determine more fully the extent of contamination detected during the first phase. A total of 12 boreholes were sampled, of which 8 were new sampling points (boreholes 10 to 17) and 4 were duplicate samples. Heavy metals and pathogenic indicators were not analysed for during this second phase as they were present only as trace contaminants in the first phase. Samples for nitrogen isotope analysis were collected from 2 sampling points.

→ drilling two monitoring boreholes (18 and 19) on the down-gradient edge of the sludge applied area in September 1996 and the collection of soil and groundwater samples from these monitoring points. Groundwater samples were collected for major ion and heavy metals analysis. Soil samples were collected from the unsaturated zone for nitrate and ammonium analysis. A final sampling run was carried out in January 1997 from these two boreholes and groundwater was analysed for major ions and nitrogen isotopes.

A total of 25 groundwater samples were analysed from 19 boreholes, 13 boreholes down-gradient or adjacent to the sludge applied area, 4 up-gradient and 2 in the sludge applied area. Most of the established boreholes are between 5 and 50 years old and no drilling records were available. They are generally around 30 m deep. The boreholes at the bird sanctuary are recently established and borehole 4 is 70 m deep. The boreholes in the sludge applied area were drilled in September 1996 to 12 m.

Reported yields for the boreholes sampled range from 1 to 5 $l.s^{-1}$. The heterogeneity of the aquifer is evident with significantly different yields reported from boreholes drilled to the same depth within 60 m of each other.

6.2.6 Results of the field investigation

6.2.6.1 Description and discussion of microbiological contamination indicators

The results of microbiological analysis for pathogenic indicators are presented in full in Appendix 2.B. Nine groundwater samples and one sludge sample were analysed for faecal coliforms, faecal streptococci and *Clostridia* spores. A further 5 borehole samples were analysed for only faecal coliforms. The results are reported in colony forming units (CFU) per 100 ml of sample.

The sludge sample contained high levels of all the indicators: 30 x 10^{8} per 100 ml faecal coliforms, 45 x 10^{8} per 100 ml faecal streptococci and 52 x 10^{8} per 100 ml *Clostridia*. One of the groundwater samples from borehole 9 contained high levels of all indicators, however, when the borehole owners were questioned about the sampling point it became evident that contamination from treated waste water piped from the WWTP was the most likely source and that this sample was not representative of water in the aquifer. The other groundwater samples contained no faecal coliforms or faecal streptococci (excluding borehole 7 which gave a Streptococci result of 1 which is deemed insignificant). Samples from boreholes 1 to 6 and 8 contained *Clostridia* at counts of 12 or less.

These microbiological indicators, known to be present in the sludge applied to agricultural land in the study area, are therefore not found to contaminate groundwater significantly. It is likely that the bacterial indicators (coliforms and streptococci) do not remain viable for the time taken to infiltrate to the water table and that they are predominantly attenuated in the first few centimetres of soil. Viable *Clostridia* spores from the sludge are more likely to reach the water table as they are relatively small and can survive for months to years. However, the test analyses for several *Clostridia* species, some of which may be present naturally in the soil. The low number encountered in the samples may be naturally present and do not conclusively indicate a sludge source.

6.2.6.2 Description and discussion of hydrochemical data

The results of all hydrochemical analyses are presented in full in Appendix 2.B. The variable thickness and depth of aquifer intercepted by the different boreholes makes comparison between sampling points difficult. High concentrations of contaminants in boreholes 18 and 19 are likely to be due to both the close proximity to the sludge applied area and to sampling from only the top few metres of aquifer.

Heavy Metals

Analyses of the metal content of the sludge are carried out regularly by the WWTP operator and the results of 4 analyses carried out in late 1995/early 1996 were made available to the project and are given in Appendix 2.B. The results show concentrations of Cr, Cu, Ni, Pb and Mn to be of the order of several hundred mg .kg^{-1} (dry mass). Fe concentration is of the order of 10 000 mg .kg^{-1}, Zn 1000 mg.kg^{-1} and Cd less than 30 mg.kg^{-1}. Temporal variations occur for all metals over several factors and differences between the digestor sources are usually within one factor.

Groundwater samples from boreholes 1 to 9 and 18 and 19 were analysed for metals. Concentrations of Al, Cr, Cu and Pb were below the detection limit of the analytical method (usually <0.05 to 0.01 mg .l^{-1}) in all cases. Fe was detected at concentrations of <1.0 mg .l^{-1} in boreholes 1, 2, 4, 5, 9, 19 and 18. Mn and Ni were detected at concentrations lower than 0.1 mg .l^{-1} in samples from boreholes 18 and 19 (on the edge of the sludge applied area) and Zn was detected at 0.06 mg .l^{-1} in borehole 18.

Cd was detected at around 0.01 mg .l^{-1} in samples from boreholes 8, 18 and 19. This is at the maximum level acceptable for drinking water (Jackson, 1980; Fetter, 1993). It is likely that the source of Cd in groundwater sampled from boreholes 18 and 19 is the sludge. Although Cd is reported at lower concentrations than other metals in the sludge and was not detected in the sample of liquid sludge analysed, it is a **relatively mobile heavy metal and is significant at such low concentrations due to its high level of toxicity**. The reported groundwater concentrations are close to detection levels (0.005 mg .l^{-1}) and therefore should be treated with caution, particularly the result for borehole 8 as Cd was not detected at boreholes closer to the likely source of contamination.

In summary, the low levels of Fe detected in samples throughout the study area are thought likely to be ambient levels. Low concentrations of Mn, Ni and Zn in groundwater on the sludge applied area are likely to come from the sludge (as they are not detected elsewhere in the aquifer), however, the impact is not significant as these levels are below maximum allowable limits for domestic consumption (SABS 1984). The limited Cd detected is also suspected to come from the sludge and occurs at significant concentrations. However, these levels are fairly close to the detection limits and should be verified with further sampling.

Major ions

All groundwater samples were analysed for major ions. The quality of the results was checked and in all cases the charge balance error is within acceptable limits and there is a good correlation between field and laboratory measurements. The first sample taken from borehole 9 was discounted due to microbiological indication of contamination by effluent at the sampling point.

Hydrochemical type

Piper plot and Stiff diagrams are shown to give an indication of the general hydrochemical character of the samples.

Figure 6.6 shows a **Piper plot** of the results of hydrochemical analyses of groundwater sampled as part of this project and the results of analyses of groundwater sampled from the dolomite aquifer in an unpolluted area to the south (Foster, 1986). Groundwater sampled in the study area does not plot on the Piper diagram as would be expected for a dolomite aquifer. The DWA&F results are typical of a dolomite aquifer with HCO_3 as the dominant anion and Ca and Mg as the dominant cations. Samples from boreholes 9 and 10 (2 and J on the plot) are closest in character to the typical dolomite samples. While these are not the furthest removed from the sludge applied area, they are both to the east of the area and have low TDS values. If north-south anisotropy is strongly developed these boreholes may have limited continuity with groundwater beneath the sludge applied area.

The relative proportion of Mg is similar in all samples at around 55% of total cations. Samples from the study area have greater Na + K and less Ca than would be expected in a dolomite. The study area samples have significantly more SO_4 than the DWA&F sample and more Cl.

Results for the samples from boreholes drilled on the edge of the sludge applied area, 18 and 19 (A and B on the plot), plot towards the middle of the study results. It would therefore appear that the hydrochemical evolution of groundwater in the area around the sludge applied area is not controlled by the ions which determine the plot position. Nitrate is the major ion which is not represented on the Piper diagram.

A selection of results plotted as **Stiff diagrams** are presented in Figure 6.7 to show the hydrochemical evolution of groundwater in the area including nitrate. It should be noted that the milli-equivalent scale varies in the diagrams.

If regional groundwater flow directions persist in the area, these data sets are representative of a N-S flow path across the study area from borehole 13 → 18→ 17→ 6→ 7. The DWA&F results are also shown as a Stiff diagram to give a comparison with a more typical dolomitic groundwater. Again, the results for the DWA&F sample show the water to be of (Ca, Mg) HCO_3 type.

All the samples from the study area have high levels of SO_4 compared to HCO_3. The level of nitrate increases dramatically at borehole 19 on the up gradient edge of the sludge applied area and gradually appears to decline with distance to the north. The concentration of Ca increases at borehole 19 but is fairly constant up and down gradient. Levels of Na + K are elevated in samples from the south and borehole 19 but decline northwards.

The Stiff diagrams therefore show uncharacteristically high levels of SO_4 in the study area and an increase in NO_3 concentrations associated with the sludge applied area, which appear to decline along the flow path northwards.

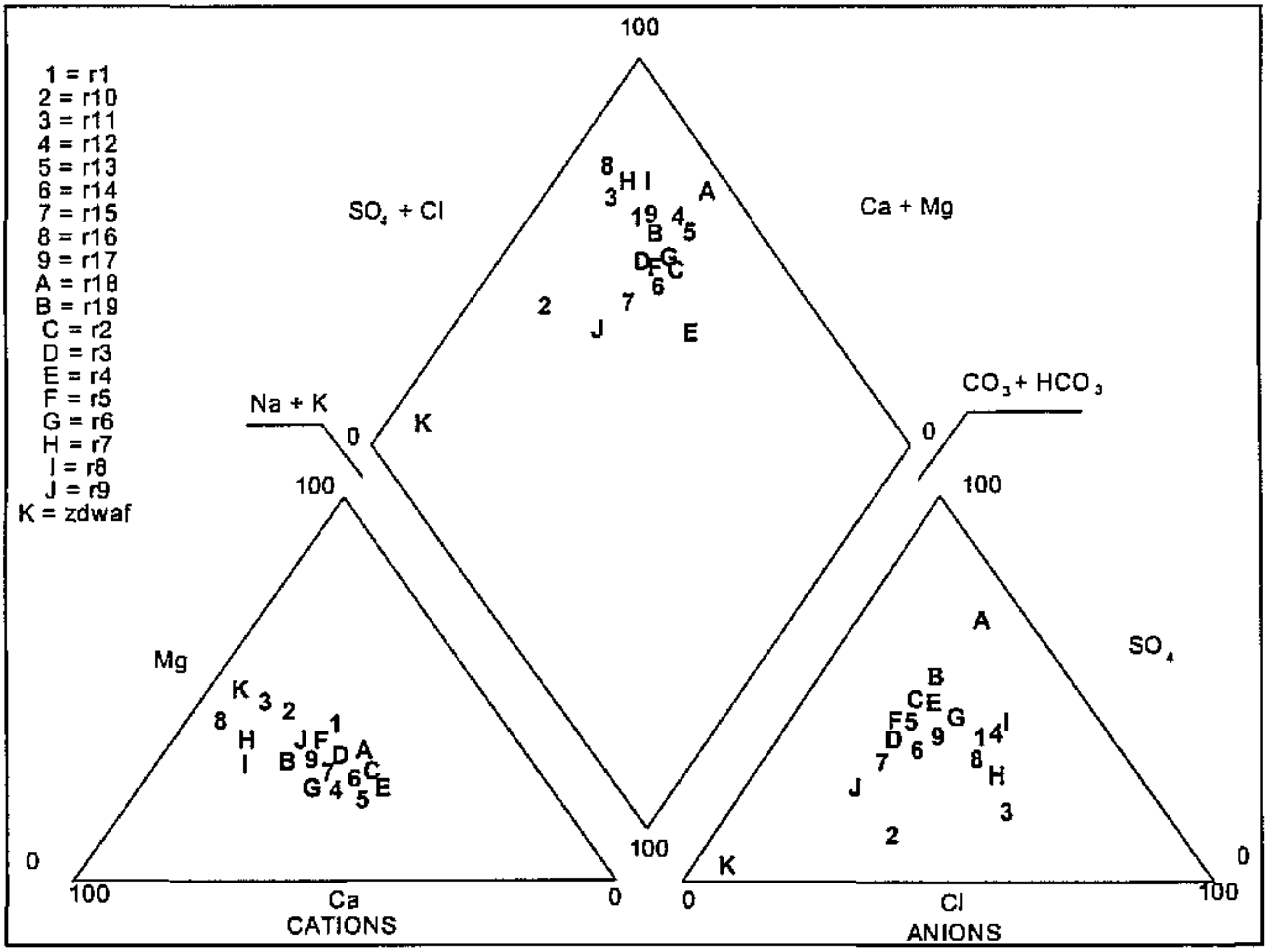


Figure 6.6: Site A: Piper plot

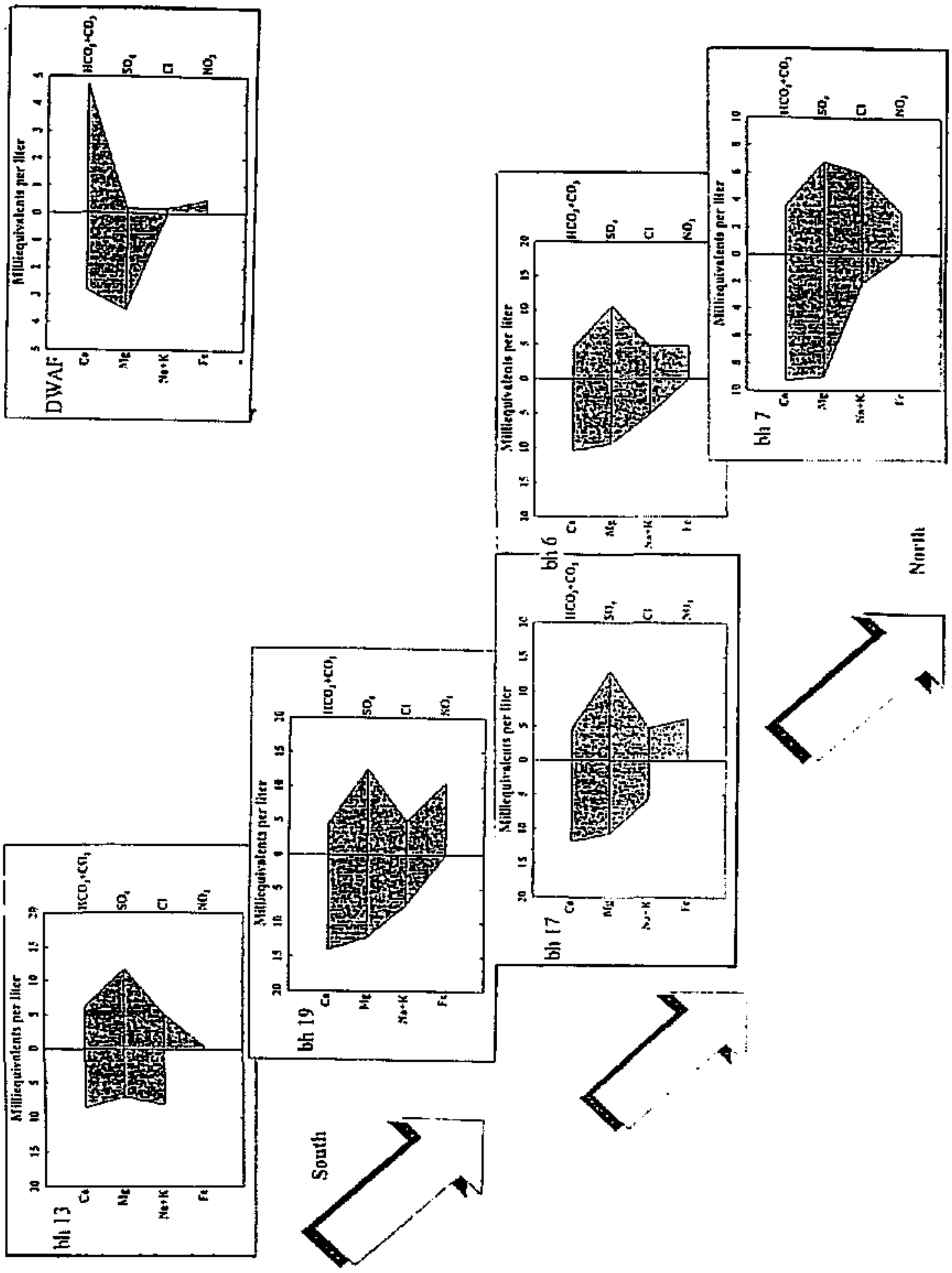


Figure 6.7: Site A: Stiff diagrams along a project flow path

Electrical conductivity

Figure 6.8 shows the spatial variation of EC (electrical conductivity) over the study area. EC varies by a factor of 4 from 105 to 386 mS.m^{-1}. It is highest in samples from the sludge applied area (>300 mS.m^{-1}) but appears to decline fairly rapidly in an approximately east-west direction to < 250 mS.m^{-1} and less rapidly in a north-south direction. Contouring of the values indicates equidistant north-south alignment of elevated EC. If the sludge applied area is the main source of elevated EC values, this may indicate north-south anisotropy in the aquifer coupled with a recharge mound at the sludge applied area.

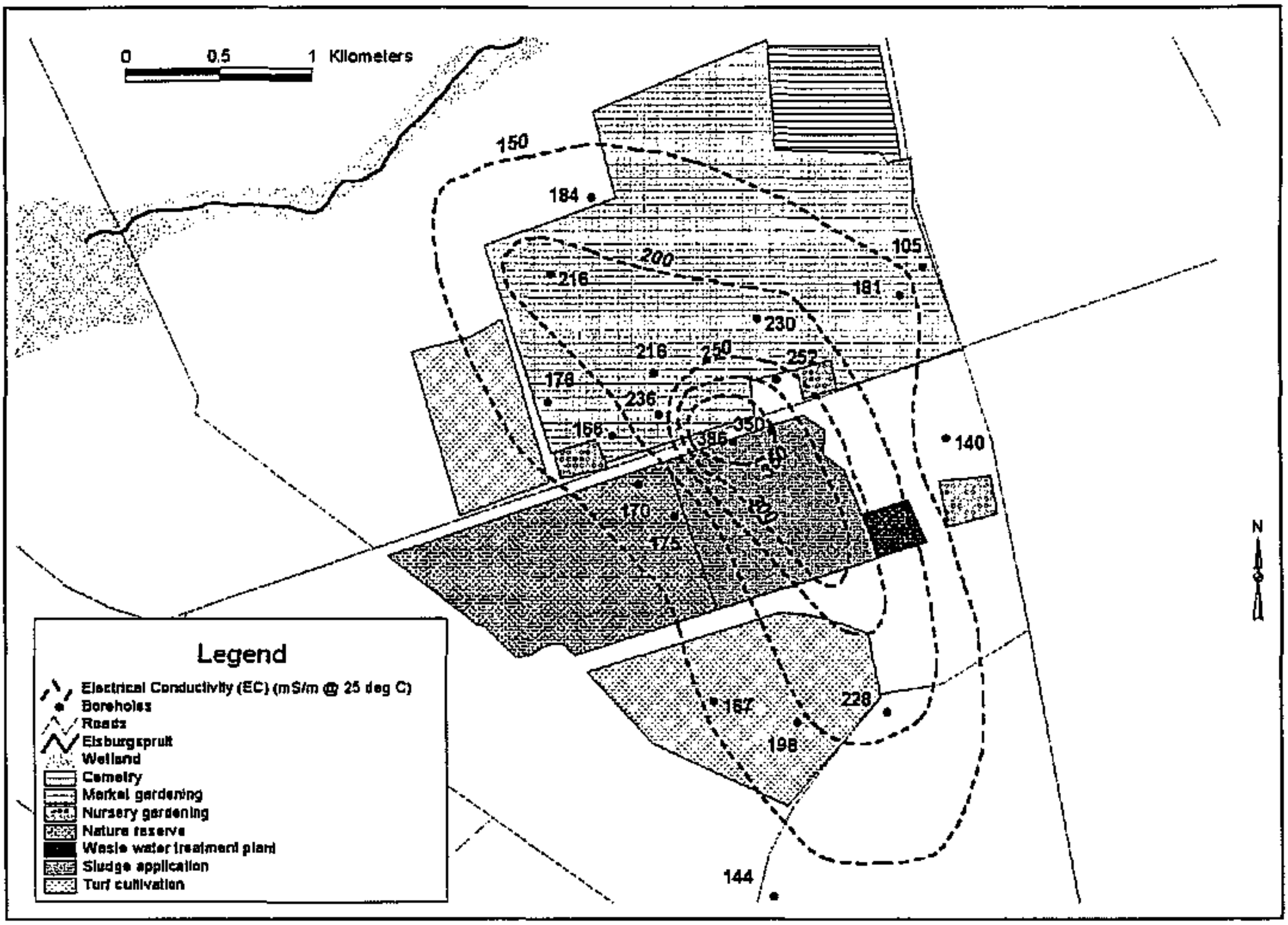


Figure 6.8: Site A: Electrical Conductivity of groundwater samples (mS.m^{-1})

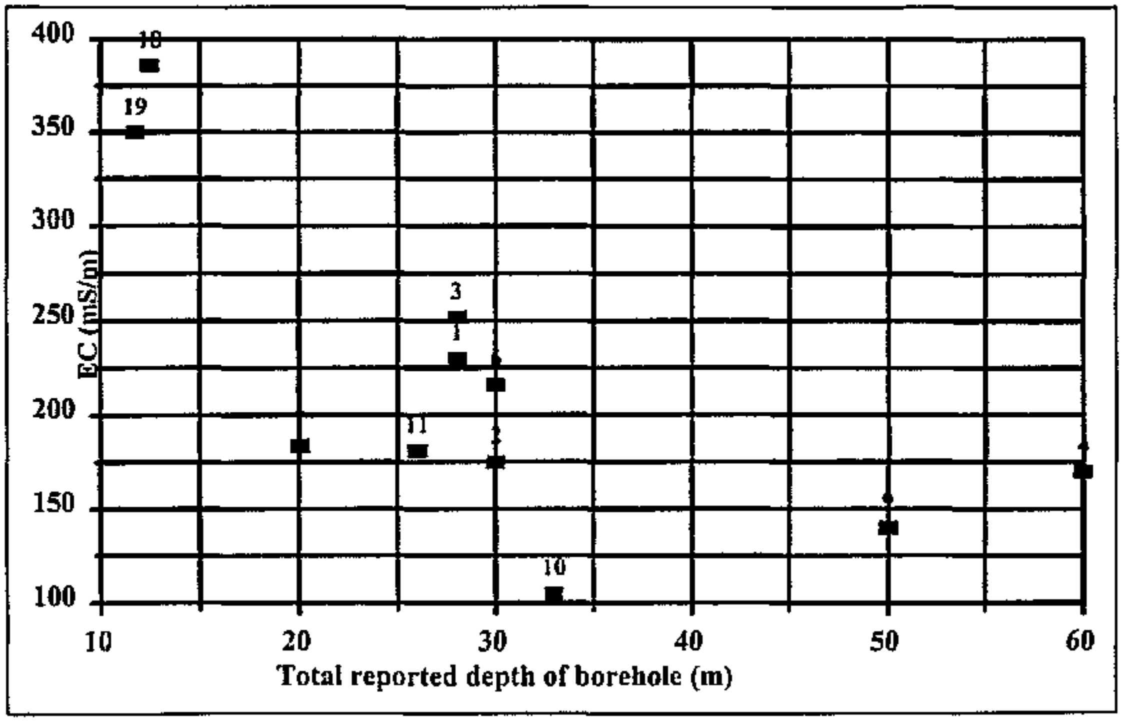


Figure 6.9: Site A: EC versus total reported depth of borehole (borehole numbers indicated)

Consideration should be given to the fact that groundwater has been sampled from different aquifer depths and thicknesses. Figure 6.9 shows a moderate correlation between increasing EC and decreasing total reported borehole depth. These data are not comprehensive for the all boreholes sampled, as in many cases the total depth of the borehole is not known. The fact that the two shallowest boreholes are also those on the edge of the sludge applied area also makes conclusive interpretation difficult. However, taken in conjunction with the polluted hydrochemical signature of the water in the area indicating a surficial source of dissolved solids, it is reasonable to conclude that EC will decrease moderately with depth.

Sulphate

Figure 6.10 shows the spatial distribution of groundwater sulphate values in the study area. High values are seen in the sludge applied area and the agricultural areas to the north and south. The concentration varies by a factor of 2 in most of the area, between approximately 300 and 600 mg.l^{-1}, but is lower in the eastern-most section of the area (38, 109 and 235 mg.l^{-1}). Significant variations in measured concentrations (>100 mg.l^{-1}) occur over short distances between sampling points (200 m).

There is no apparent correlation between sulphate concentrations and total depth of borehole. Lower sulphate concentrations would be expected at depth as aquifer conditions become more reducing, particularly as hydrogen sulphide smells were detected at the some of the deeper sampling points. The excess of nitrate available in the groundwater combined with high levels of sulphate may mean that sulphate reduction does not influence the sulphate concentration until deeper in the aquifer.

No obvious correlation exists between sulphate and nitrate concentrations, indicating that the sources of the two ions are different.

Previous reports have suggested surface water contaminated by industry and slimes dams as possible sources of sulphate contamination in this area (Foster, 1986). If these were the main sources we would expect to see sulphate concentrations increasing northwestwards across the study area towards the Elsbergspruit and the slimes dam north of Elspark. The relatively low values for the eastern boreholes may give some indication of contribution from these sources. However, the occurrence of higher values in the centre of the area away from the river appears to indicate additional contribution from the sludge applied area and the surrounding agricultural land.

The concentration of sulphate in the sludge at the site was measured at 74mg.l^{-1}. As this is lower than groundwater concentrations in the area it indicates that the sludge is not likely to be a principal source.

In summary, these data indicate several possible sources of sulphate input to groundwater, which may include sludge but this is not thought to be the most significant source.

Nitrate

Figure 6.11 shows that the highest values for NOx-N are found in groundwater underlying the sludge applied area and the agricultural area immediately to the north. The lowest values are found along the western boundary of the study area. Contouring of the values indicates greatest input in the vicinity of the sludge applied area with concentrations declining along approximately north and south flow paths.

There is a moderate correlation between decreasing nitrate concentration with increasing depth as is shown in Figure 6.12. This indicates a surficial source of nitrate and is probably combined with reducing/denitrifying conditions at depth in the aquifer. Hydrogen sulphide was detected during sampling at borehole 4, which is reported to be **60 m deep**. This indicates anoxic conditions under which nitrate would have been reduced thus <1 $mg.l^{-1}$ was detected.

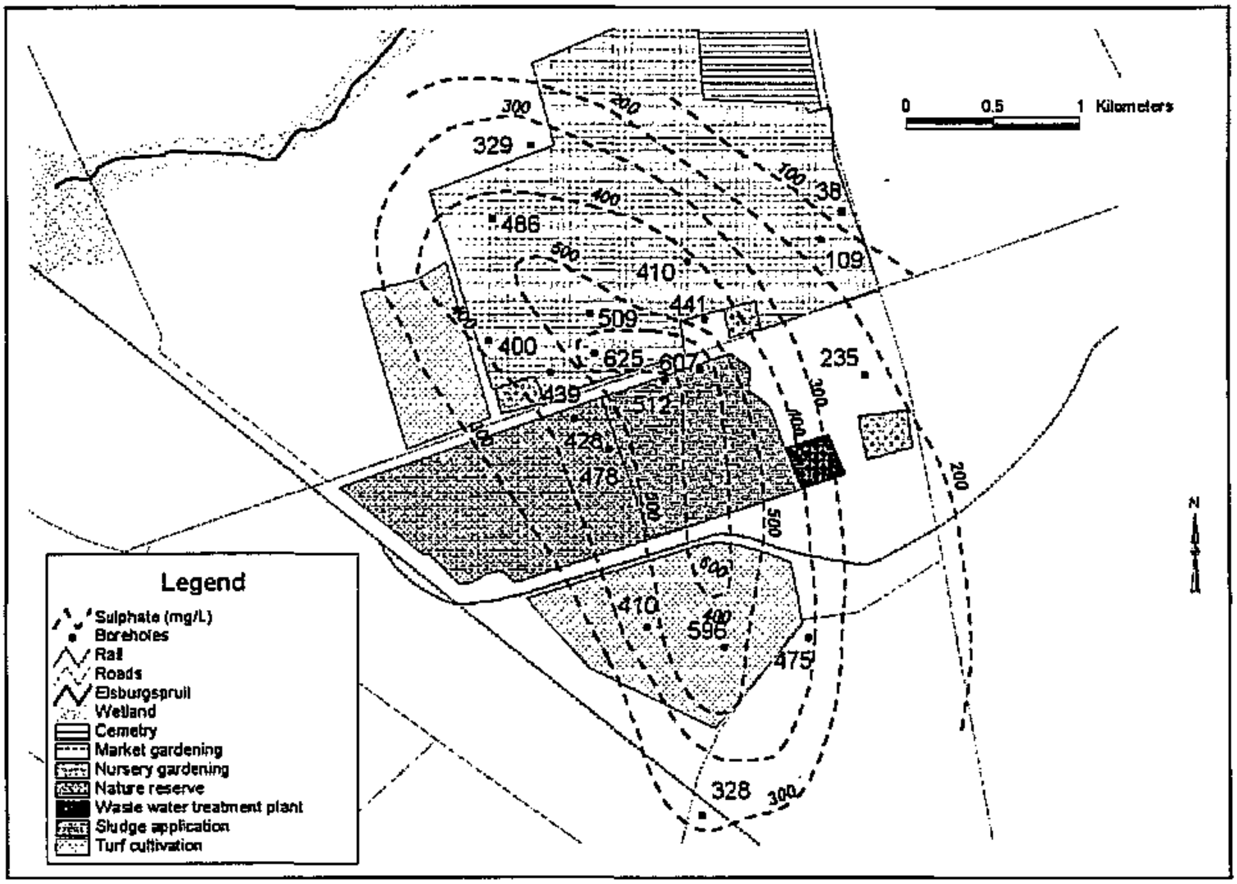


Figure 6.10: Site A: Spatial distribution of SO_4 in groundwater samples

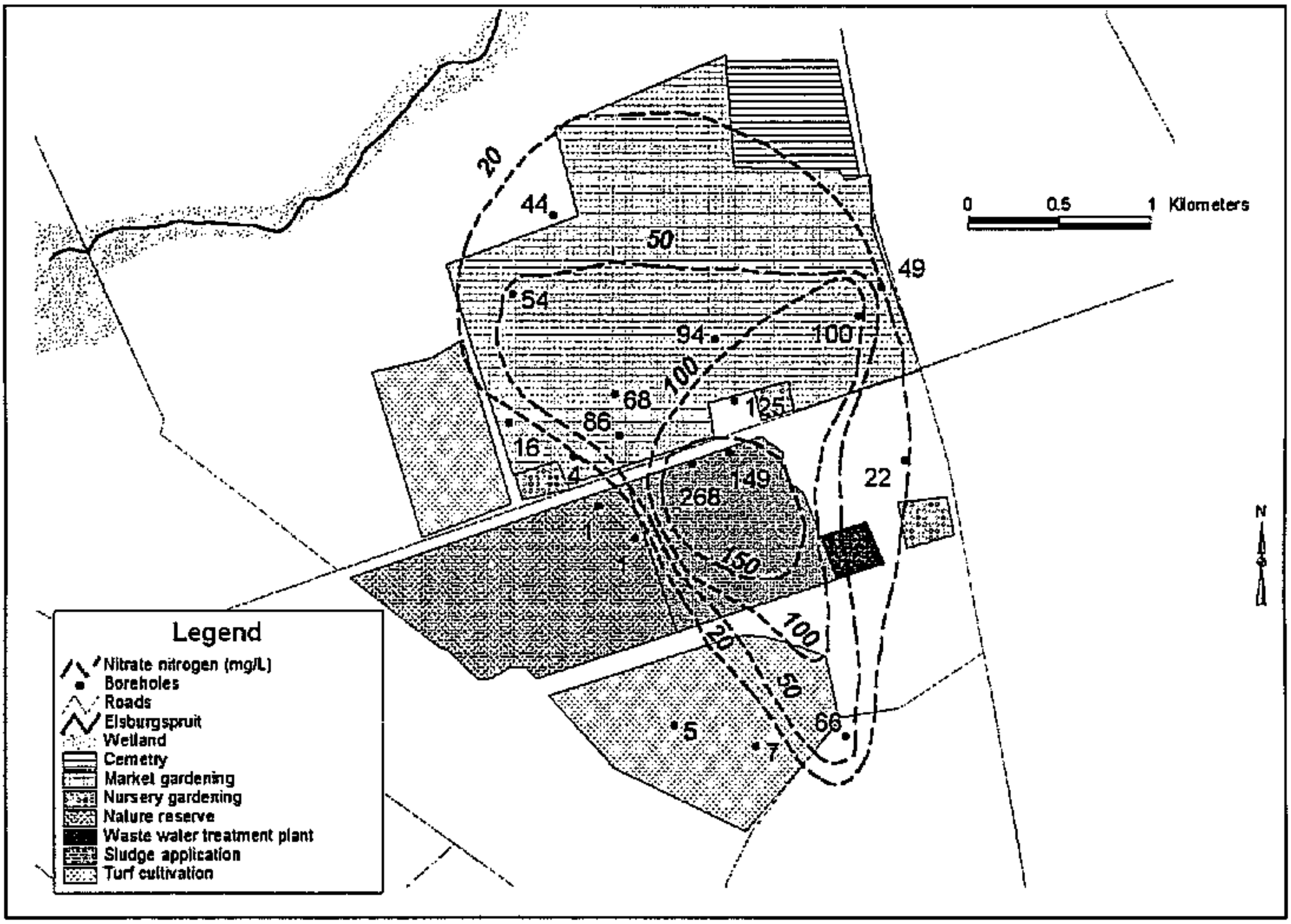


Figure 6.11: Site A: Spatial distribution of NO_3 - N in groundwater samples

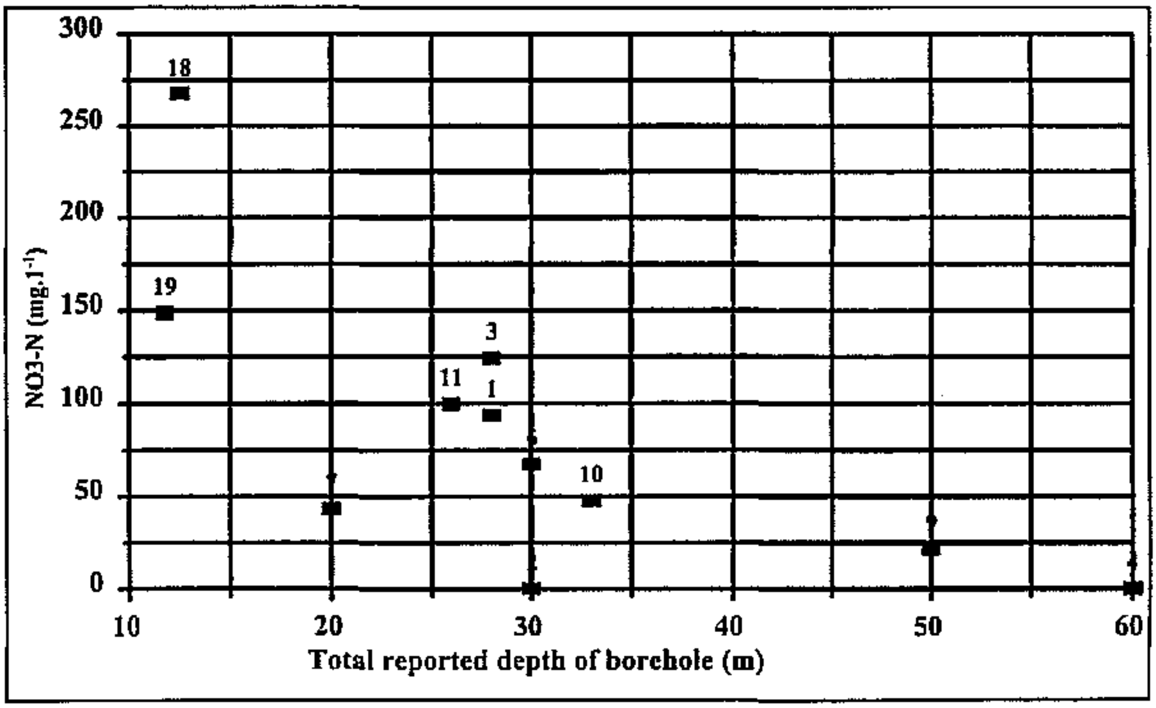


Figure 6.12: Site A: NO_3 - N versus total reported depth of borehole (borehole numbers indicated)

A positive correlation exists between nitrate and EC as is shown in Figure 6.13. This further indicates the significant contribution nitrate makes to the TDS of the groundwater and that the source of nitrate contamination may contribute other ions to solution.

The concentrations of nitrate seen in the study area are very high and assumed to be anthropogenic. Analysis of the sludge showed the concentration of TKN to be 5 328 mg.l^{-1} , ammonium 652 mg.l^{-1} and nitrate below the concentration limit of 0.2 mg.l^{-1}. Sufficient nitrogen is therefore available in the sludge, mainly in organic form, for mineralisation and nitrification in the soil and leaching to groundwater.

Nitrogen is also known to be applied at high loadings in the surrounding agricultural areas in ammoniacal form. Although the contours indicate the sludge applied area to be the main source of nitrate in groundwater in the area they are affected by the relative depth of the sampling points. Further confirmation of the nitrate source was therefore required.

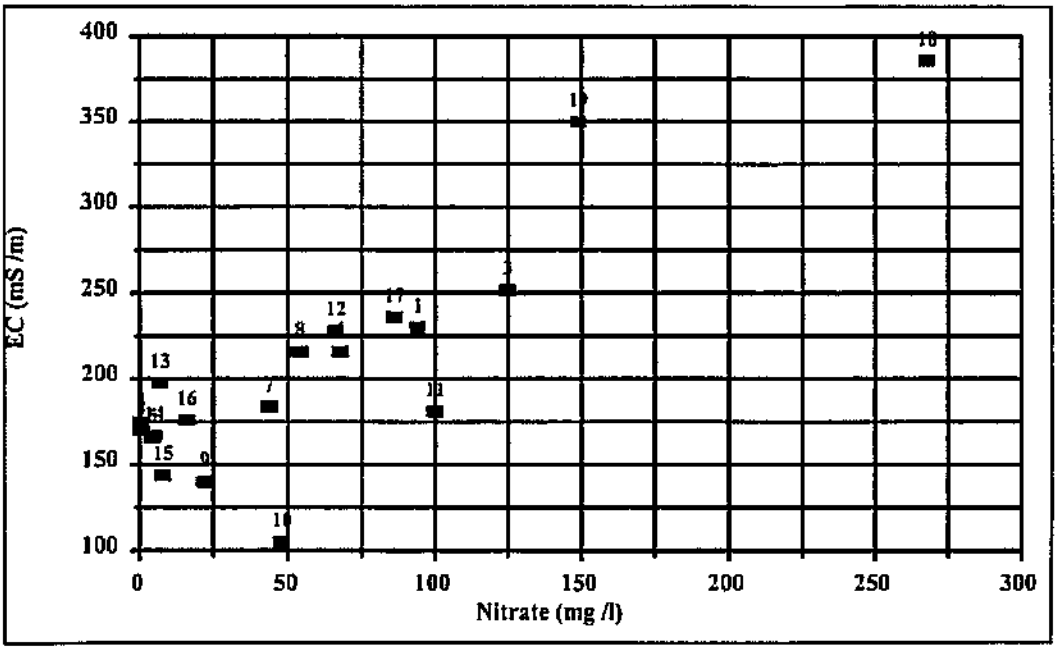


Figure 6.13: Site A: NO_3 - N versus EC (borehole numbers indicated)

Nitrate nitrogen isotopes

Nitrate nitrogen isotope analyses were carried out to try and determine the source of nitrate in groundwater. Samples from boreholes 19, 18, 3, 17 and 8 were analysed. A sample of sludge was also analysed to confirm the nitrogen isotopic signature of this potential source.

Table 6.3: Site A: Results of NO_3-N isotope analyses

Borehole	*NO_3-N ($mg.l^{-1}$)*	*$\delta^{15}N$*	*Approximate distance from sludge applied area (m).*
19	149	+ 18.2	0
18	268	+ 16.1	0
3	125	+ 15.8	200
17	86	+ 17.8	300
8	54	+ 14.2	1200

The sludge sample analysed was in the liquid form as applied to the land. It contained no nitrate but 755 mg.l^{-1} ammonium nitrogen. The ammonium gave an isotopic signature of + 0.9. Conversion of this to nitrate would produce a NO_3 δ^{15} N value of +10 to +20 (Talma, pers.comm).

These results are well within the range of δ^{15} N for organic sludge or animal waste as a nitrate source (+7 to +25 as discussed in section 6.1.5.5). Farmers in the area have reported that they do not use manure as a fertilizer and there is no other evidence to indicate nitrate pollution from an other organic source such as pit latrines (microbiological indicators). Infiltration of contaminated effluent from the bird sanctuary lagoons is another possible source. No hydrochemical data for the effluent are available. However, it is expected that nitrogen levels in the sludge would far exceed those of the effluent, making the sewage sludge the most likely source of organic nitrogen in the area.

Dissolved organic carbon

Figure 6.14 shows the ratio of dissolved organic carbon (DOC) to NO_3-N in groundwater samples. There is no steady increase of DOC with increasing NO_3-N, however, the samples from the edge of the sludge applied field, which have the highest NO_3-N and represent groundwater from the shallowest part of the aquifer, have the highest DOC values at 11.6 and 6.9 mg.l^{-1}. Although elevated DOC levels may be expected in shallower groundwater, the source of DOC at these high levels is likely to be the sludge. Sampling of shallow groundwater at a control point would help to confirm this. The lack of correlation with NO_3-N over greater distances from the sludge applied area indicates that DOC is attenuated much more rapidly than NO_3-N within the aquifer.

Potassium

Potassium occurs in the sludge at 86 mg.l^{-1}, and is considered a potential groundwater contaminant. Figure 6.15 shows little correlation between the levels of potassium and nitrate (a proven sludge contaminant) when all the boreholes are considered. However, if one considers the boreholes with greater than 10 mg.l^{-1} NO_3- N there does appear to be a positive correlation. The boreholes with < 10 mg.l^{-1} NO_3- N are located up-gradient and adjacent to the sludge applied area and show a wide range of K values, including the highest from the deep borehole 4.

Boreholes with elevated nitrate levels show a narrower range of K values but they approximately increase with increasing nitrate indicating that there may be some contribution of K from the sludge.

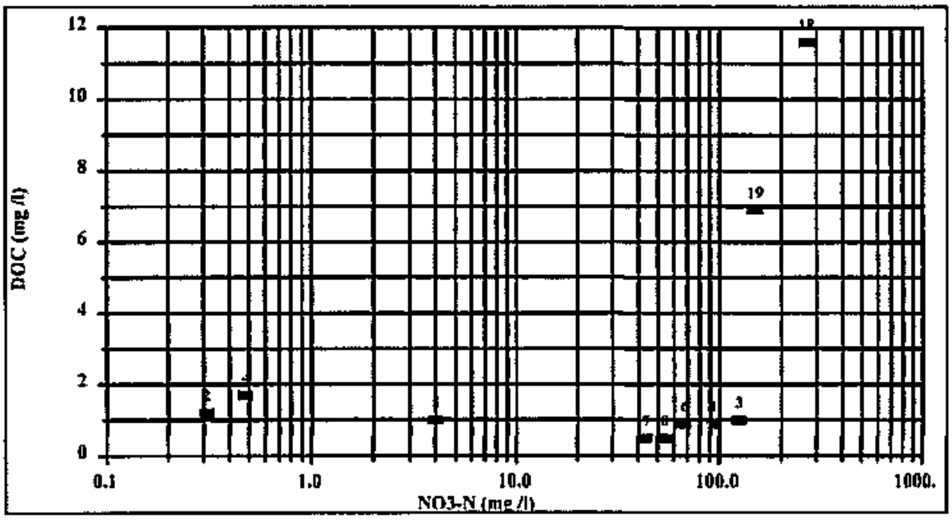


Figure 6.14: Site A: NO_3 - N versus DOC

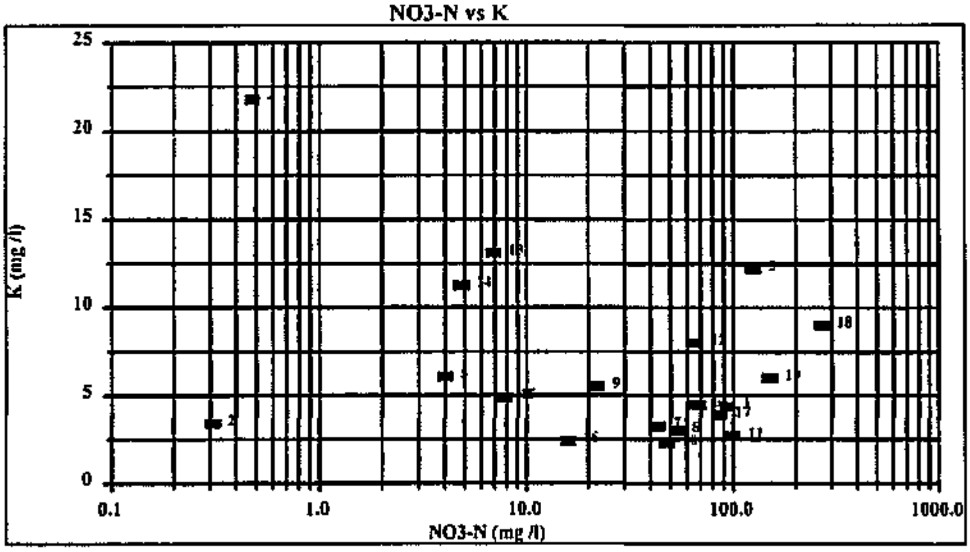


Figure 6.15: Site A: NO_3 - N versus K

6.2.6.3 *Description and discussion of soil data*

Limited soil samples were collected during the drilling phase of the field investigation to give an idea of nitrogen concentrations in the unsaturated zone. Figure 6.16 shows the results of analyses for water soluble and potassium chloride extractable nitrate and ammonium. The deepest sample at 7 mbgl was taken from the first saturated horizon, ie. either at the water table or in the capillary zone.

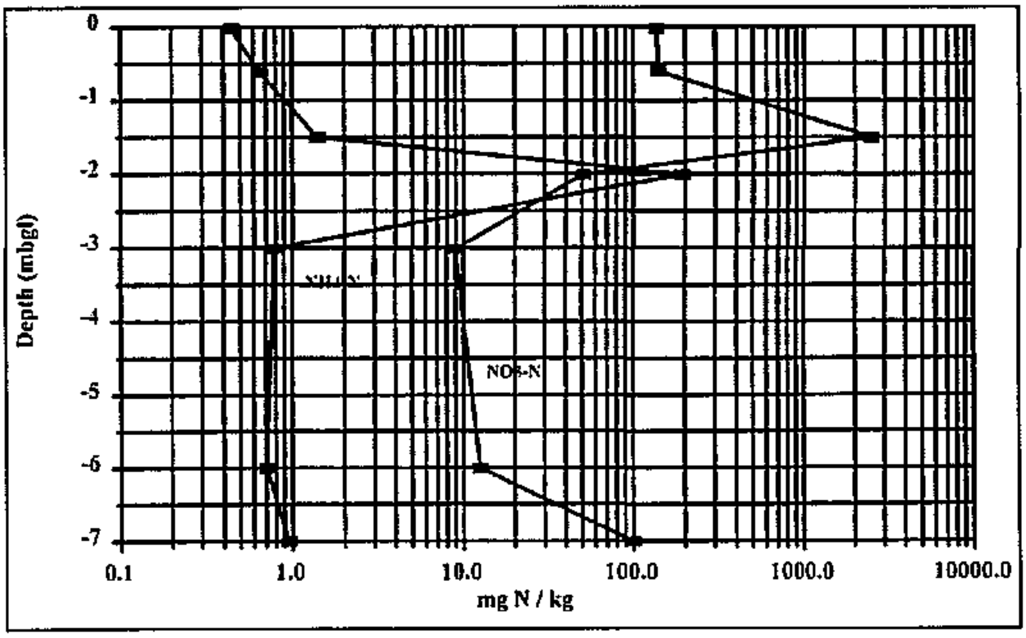


Figure 6.16: Site A: Nitrogen soil profile of NO_3 - N and NH_4 - N.

These results give a snap-shot of nitrogen speciation and concentration in the unsaturated zone but no indication of spatial or temporal variation. Variations in the levels of NO_3-N and NH_4-N with depth are primarily dependent on:

- the input rate of nitrogen to the soil surface
- the rate of mass transport of nitrogen downwards through the soil profile
- relative rates of mineralisation, immobilisation, nitrification and denitrification at different depths
- the degree of saturation (particularly for NO_3-N), permeability and presence of attenuating media (particularly for NH_4-N) in the soil.

Concentrations of NO_3-N are at least an order of magnitude higher than NH_4-N at most sampling depths. NO_3-N levels are relatively high in the shallow subsurface at around 140 mg N. kg^{-1} soil and increase to a peak of 2500 mg N. kg^{-1} soil at 1.5 mbgl. The loading history of this particular area of the sludge applied area is not known therefore we cannot say if this peak represents a period of high N loading at the surface which is now leaching through the unsaturated zone. If the N loading at surface has been approximately constant this peak may represent a more saturated zone in the soil horizon where the higher concentration of NO_3-N is a reflection of greater levels of soil moisture. NO_3 concentrations then decline to around 10 mg N. kg^{-1} soil and increase again at the saturated zone. This indicates an aerobic environment and that the accumulation of leached nitrate at the top of the aquifer occurs at rates exceeding denitrification.

NH_4 occurs at a lower concentration in the near surface samples (< 1 mg N. kg^{-1} soil) but exceeds NO_3 at 2 m depth, slightly deeper than the NO_3 peak. It may be that net mineralization is greater at this depth and that there is some attenuation of NH_4 in a soil layer with a higher cation exchange capacity. NH_4 concentrations in the saturated zone do not increase significantly as is seen with NO_3.

To determine more fully the processes affecting nitrogen transfer to the water table, data on the organic, clay and moisture content of soil horizons are required plus an indication of variations in redox potential and vertical permeability with depth. These data show that a N peak exists at around 1.5 to 2 mbgl and that generally net nitrification exceeds net mineralisation, except at 2 mbgl.

6.2.7 Summary of impact of sludge application on groundwater quality

Potential groundwater contaminants in the sludge were thought to be nitrogen (occurring as nitrate in the groundwater but in organic and ammonaical forms in the sludge), COD, DOC, potassium, heavy metals and microbes.

Interpretation of the results of this study indicates that **nitrate** from the sludge significantly contaminates groundwater below the sludge applied area and at least 1200m down-gradient. This is shown by the spatial distribution of nitrate values in groundwater, the isotopic ratio of nitrate nitrogen in groundwater and evidence of nitrate leaching in the soil.

Limited **DOC** contamination was detected in shallow groundwater immediately next to the sludge applied area. It is not a persistent contaminant in this aquifer.

Some **sulphate** may enter the groundwater from the sludge, however the distribution of sulphate and lack of correlation to nitrate levels indicate the impact of other more significant sources in this area.

The sludge may contribute some **potassium** to groundwater, however, the impact on the local hydrochemistry is not significant.

EC levels in the groundwater are affected by contamination from the sludge, primarily reflecting the high levels of nitrate contamination.

No significant **microbiological** contamination of the groundwater was detected, indicating that microbes are successfully attenuated in this environment before reaching groundwater.

Cadmium was the only **heavy metal** detected in the groundwater at higher than recommended levels. These levels are very low at 0.01 $mg.l^{-1}$ and cannot be conclusively attributed to the sludge.

The application of sewage sludge did not appear to impact the **COD** or **pH** of the groundwater in the area. The nature and thickness of the unsaturated zone and dilution effects in the aquifer are thought to be sufficient to mitigate impact on these parameters.

6.2.8 Synopsis

The site is an example of liquid sludge application to agricultural land where fodder crops are grown. It should be noted that the sludge is put to agricultural use primarily as a means of disposal and the area is managed by the WWTP operator rather than by a commercial farmer for economic returns.

The site is immediately surrounded by agricultural areas where high loadings of inorganic fertilizers are reported. Industrial and mining development occurs within several kilometres. There are several potential sources of groundwater contamination in the surrounding area, primarily inorganic fertilizers, WWTP effluent and industrially contaminated water discharged into the Elsbergspruit river.

Liquid sludge produced from the treatment of industrial waste water has been applied to this site for longer than 15 years. The most significant potential contaminants in this sludge, in terms of health risks, include nitrogen, metals and microbes.

The sludge applied area overlies a shallow, unconfined, dolomitic aquifer. The unsaturated zone contains up to 30% clay and is reported to have a bulk permeability of $0.01 m.d^{-1}$. It is expected that preferential pathways for vertical migration exist in the unsaturated zone due to the nature of the underlying aquifer and the reported risk of sink hole formation. The dolomitic aquifer is heterogenous with north-south anisotropy and has a reported permeability of $10 m.d^{-1}$. The aquifer is reported to be in continuity with the nearby river.

Groundwater quality in the area is not typical of a dolomitic aquifer and is significantly contaminated with sulphate and nitrate. The data do not conclusively indicate a source of sulphate, but it is thought that the sludge is not a principal source.

Isotope ratios of nitrate nitrogen in the groundwater are typical of an organic waste source. These data, combined with the spatial distribution of nitrate values and evidence for leaching from the sludge applied, indicate that nitrate originates mainly from the sewage sludge. The occurrence of nitrate in the aquifer is seen to be influenced by lateral distance from the source, depth and anisotropy. Other potential contaminants from the sludge were not found to impact groundwater quality significantly in the area. They are therefore sufficiently attenuated within the unsaturated environment at the site. Adsorption and biodegradation are likely to be the main processes attenuating contaminants of concern.

6.2.9 Conclusions

The impact of sludge on the groundwater should be considered in terms of the environment and the application practice in order to recommend good practice. At this site there is evidence of significant nitrate contamination to high levels over an extensive area (> 1 km distance from the sludge applied area) and some limited DOC and metal contamination of groundwater. Other potential contaminants such as most heavy metals, microbes and COD are apparently successfully attenuated in the unsaturated zone.

The high cumulative loading of sludge organic matter at the site combined with a vulnerable hydrogeological setting are thought to result in nitrate contamination. Although the reported bulk permeability of the unsaturated zone is moderate at 0.1 $m.d^{-1}$ due to the presence of clays (<30%) in the soil, preferential pathways for infiltration are known to exist as this is a sink hole risk area. In addition, the high secondary permeability of the unconfined dolomite aquifer (approximately 10 $m.d^{-1}$) maintains oxidising conditions away from the water table and nitrate is not reduced along flow paths or until it is relatively deep in the aquifer. Abstraction in the area is from the shallower portion of the aquifer therefore nitrate remains a risk.

In summary, groundwater contamination is thought to be due to the following factors at this site:

- long term application of sludge (>15 years)
- application of sludge at greater than agronomic rates (primarily for disposal rather than agricultural purposes)
- the existence of preferential pathways for infiltration to the water table
- the prevalence of oxidising conditions in the unconfined aquifer.

6.2.10 Recommendations for further investigations

It would be useful to examine shallow groundwater quality in the vicinity of the bird sanctuary lagoons and the quality of the effluent supplied to the lagoons to establish whether they are contributing to groundwater contamination in the area. Similarly the assessment of shallow groundwater quality in the surrounding agricultural area would be valuable.

Multi-level sampling in the unsaturated zone and the aquifer close to the sludge applied area would give a more detailed understanding of attenuation mechanisms (such as adsorption, reduction, biodegradation, etc.) in operation at the site.

6.3 SITE B

6.3.1 Site background

This site was selected for the following reasons:

- sludge was being applied for the first time in the monitored area of the farm therefore any change in groundwater quality over an initial period of application and crop growth could be monitored

- it provides an example of the application of dried sludge cake at significantly higher than recommended loadings

- the site is underlain by a shallow unconfined sandy aquifer and the groundwater is therefore vulnerable to contamination

- the unsaturated zone consists mainly of permeable sands with low attenuation capacity

- the shallow depth of the water table enabled sampling points to be jetted in at low cost and quickly

- the farmer was willing to cooperate fully with the project and keep records of sludge and fertilizer applied.

The site is a small holding operated as an agricultural interest by a family with alternative primary income. It is located near Atlantis in the Western Cape, as shown in Figure 6.2 and is surrounded by other small holdings. To the south there is a stud farm, to the west a poultry farm, and to the east and north (up-gradient) uncultivated land.

A limited number of pigs and cows are reared on the site and fodder crops are grown on 8 ha.

6.3.2 Application of sewage sludge to the site

Dried sewage sludge cake from a nearby WWTP is used as a soil conditioner and fertilizer at the site. It was first used in 1991 on a field in the south-eastern corner of the farm and it has been applied over a gradually wider area since then as a result of improved crop yields. When only inorganic fertilizers were used in the field where the monitoring boreholes are situated, the yield of barley was not sufficient to harvest. When sludge was used in conjunction with limited inorganic fertilizers the 4 ha area yielded 9 t of harvestable barley. This is thought to be mainly as a result of the soil conditioning properties of the sludge. The medium grained sandy soil has very low water retention capacity. Prior to sludge application the farmer reports that the crops would dry out within several days of a rain event. After sludge was applied the crops remained green with sufficient soil water supply for over 2 weeks after a rain event. No irrigation is carried out at the farm.

Approximately 100 $t.ha^{-1}$ was applied to the area used to grow fodder crops, significantly more than current DNP&H guidelines permit (8 $t.ha^{-1}$). The sludge was collected from the WWTP and dumped in 10 t loads over the fields used for fodder cultivation. It was then spread and ploughed in prior to planting in April.

The WWTP has a mixed industrial and domestic catchment, with approximately 30% of waste water treated coming from industrial sources. These include heavy engineering, metal working, fish salting and dog food production. Domestic waste water is treated for nitrogen and phosphorus removal, the industrial waste water for nitrogen removal. Further treatment processes include extended aeration and activated sludge.

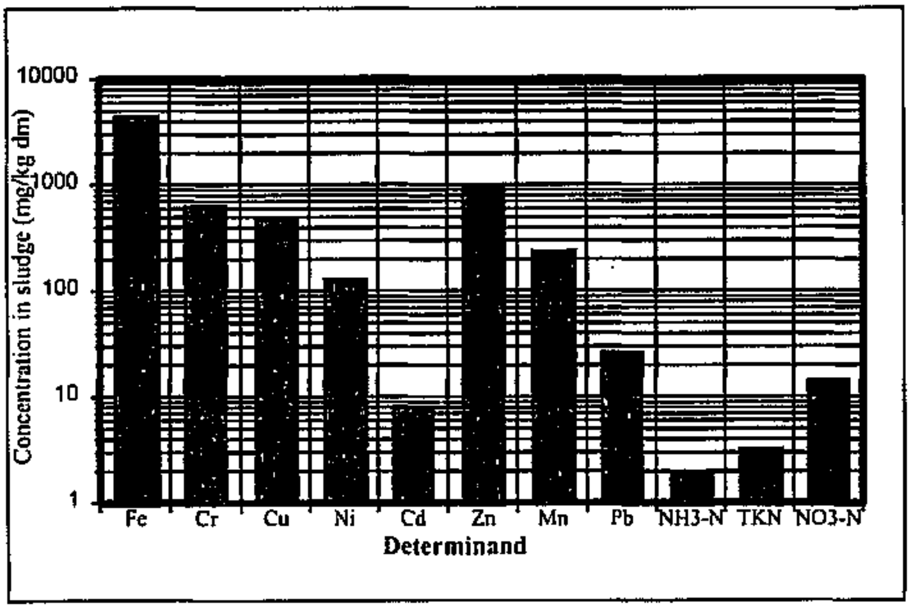


Figure 6.17: Site B: Metals and nitrogen in the sludge

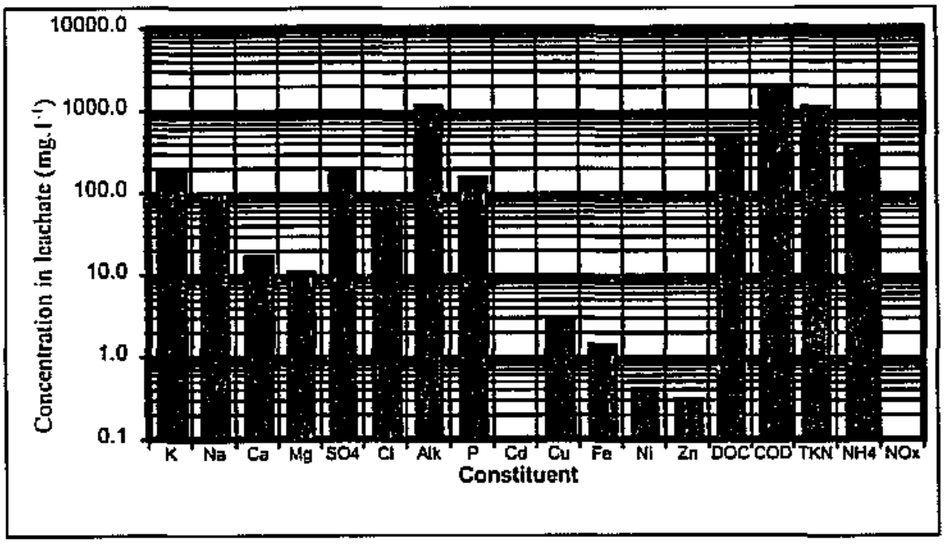


Figure 6.18: Site B: Constituents of the sludge leachate.

Figure 6.17 shows the bulked average results of analyses carried out quarterly during 1995 on the sludge at the WWTP. Sludge used at the site was collected in early 1996 therefore it can be assumed that these analyses are approximately representative of the sludge used. Most metals occur at between 100 and 1000 $mg.kg^{-1}$ (dry mass). Pb and Cd are lower at 26 and 8 $mg.kg^{-1}$ respectively and Fe is higher at 4422 $mg.kg^{-1}$. Nitrogen concentrations are also given and the ratio of NH3-N: TKN: NO3-N is 1: 1.5: 7. The results are given in full in Appendix 2.C.

Figure 6.18 shows the results of analyses carried out on leachate from the sludge cake. Approximately 200 g of sludge cake from the site was allowed to stand in 1 L of distilled water for 48 hours. The analyses were carried out to determine the relative concentrations of readily water soluble potential contaminants. This gives a better indication of the constituents that are likely to infiltrate to groundwater.

Potential soluble contaminants detected in the leachate are P, K, SO_4, NH_4, TKN, DOC and COD. No metals were detected at > 2 $mg.l^{-1}$. No nitrate was detected. This is a result of a lack of nitrate in the sludge sample (although nitrate is reported in analyses of sludge from the WWTP) and a lack of nitrifying bacteria in the water, as would be expected. In a soil system, however, nitrate would be expected to be present in the leachate.

The sludge does not undergo heat treatment disinfection therefore microbes are likely to occur at undesirable levels. All the sludge produced at this WWTP is beneficially used on agricultural land.

The main problems experienced at this site with the application of sludge cake were as follows:

xi. it is difficult to break up into small enough fragments

xii. it remains unevenly distributed in the soil due to the relatively large fragment size

xiii. if spread too long before the first rains it causes a smell nuisance.

6.3.3 Land-use and potential groundwater contamination in and around the study area.

Figure 6.19 shows the layout of the site and the surrounding area. Inorganic fertilizers, in addition to sludge, are applied to the area within the farm boundary where fodder crops are grown. Approximately 50 kg.ha^{-1} of KCl is added during planting and 100 kg.ha^{-1} of LAN during the early growing season to provide an evenly distributed nutrient source.

Cattle and pigs are raised in the areas indicated and some contamination from their waste may occur in these areas. None is directly up gradient of the sampling points however. Pit latrines are established at the farm labourers cottages, near the piggery.

A poultry farm is located to the west of the site (across gradient) and it is likely that poultry manure is stockpiled on the premises.

The area to the north (up-gradient) and east of the site is uncultivated and vegetated by alien Port Jackson (*Acacia saligna*). The farmer reports that during the last 4 years many of the bushes on his uncultivated land (near the control borehole A3) have died, probably as a result of rust fungus introduced to control them. An increasing proportion of dead bushes now occupy the uncultivated land.

In summary, potential anthropogenic sources of groundwater contamination in the area are thought to include:

xiv. inorganic fertilizers
xv. pit latrines
xvi. cattle and pig manure on site
xvii. poultry manure to the north west of the sampling points.

Inorganic fertilizers are most likely to impact groundwater quality at the sampling points as the other sources are not close to or directly up gradient of the boreholes.

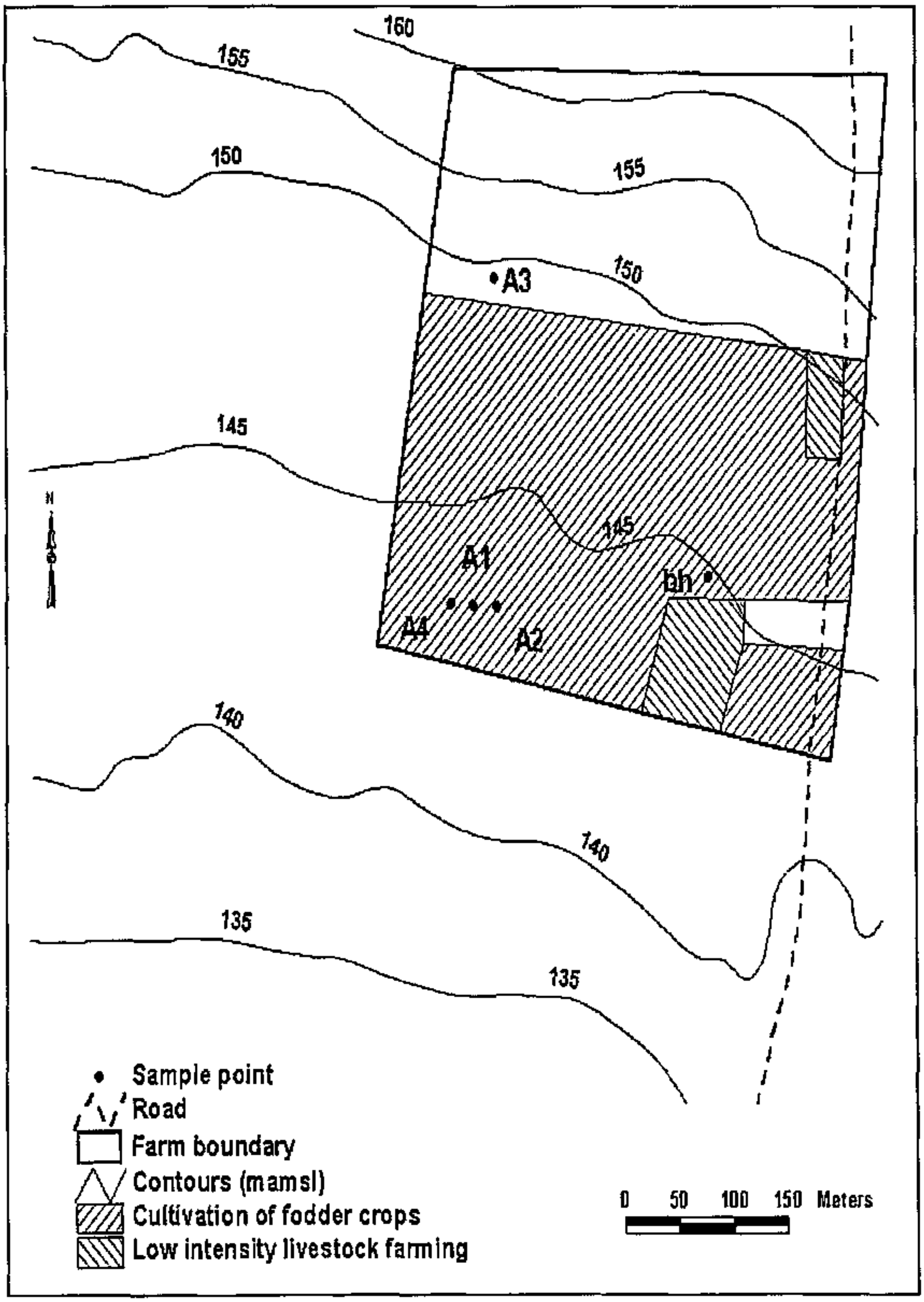


Figure 6.19: Position of sampling points and land-use at Site B

6.3.4 Field investigation

The field investigation was carried out between March 1996 and March 1997. Figure 6.19 shows the layout of groundwater sampling points, one up gradient control point in uncultivated area with alien Acacia vegetation cover, and three points in the down gradient portion of the cultivated fodder field spaced at approximately 30 m intervals. Sampling sheets are included in Appendix 2.C.

The sampling points were installed using a jetting method. A 200 mm borehole was augered to the water table and then 3 m of screen beneath 4 m of plain casing were installed by jetting. uPVC casing with a 63 mm ID was used. The screens had 0.3 mm width slots with an open area of less than 2%. The amount of water introduced during the jetting procedure was noted and in excess of this amount was purged prior to the first sampling run.

In several instances jetting was impeded by the presence of clay rich layers and Fe-rich nodules. These were not laterally extensive and moving the penetration point by a few metres usually avoided these less sandy layers.

During development and purging of the sampling points it was noted that sampling point A1 intercepted less permeable horizons and greater drawdown was experienced.

Soil samples were collected during augering for grain size distribution analysis to give an indication of the bulk permeability of the unsaturated zone.

Soil profile samples were also collected in November in the sludge fertilized field and the control area to determine the concentration of nitrogen in the unsaturated zone. During this sampling run a deeper borehole supplying the farm was also sampled to determine the hydrochemical nature of deeper groundwater in the underlying granite aquifer at the site. The total depth of the borehole is 110 m and the pump is installed at 80 m, however the open/ screened depth and rest water level are not known. The yield is approximately 40 $m^3.d^{-1}$.

6.3.5 Hydrogeology of the site

The site is underlain by an unconfined sandy aquifer at approximately 3 mbgl. The unsaturated zone is mainly comprised of permeable sandy soil with occasional clay rich lenses.

The site occurs on Quaternary Springfontyn sands with an outcrop of Cape Granite forming a topographic high to the north of the farm. Granite is reported to underlie the sands in this area at around 20 to 30 mbgl.

Figure 6.20 gives a simplified summary of the conceptual hydrogeology of the site. Augering of the **unsaturated zone** revealed buff coloured, medium to fine grained, well sorted silica sands. The sand becomes orange to brown close to the watertable, typically at around 2.5 mbgl, and sometimes Fe cemented nodules occur at this depth. Occasional clay rich lenses were intercepted at various depths.

The results of grain size distribution analysis are given in the table below with the calculated bulk saturated **hydraulic conductivity**. The latter has been calculated using Hazen's formula $K = A\ d_{10}^2$, where A is equivalent to 1 for relatively uniformly graded sands (as seen here), d_{10} is the grain size diameter (mm) at which 10% of the soil particles are finer and 90% are coarser and K is the bulk permeability in $cm.s^{-1}$ (Freeze and Cherry, 1979).

Table 6.4: Results of grain size analysis and calculated bulk permeability.

Auger point	A1	A2	A3
Depth	1m	3m	3m
d10 (mm)	0.156	0.162	0.167
d60 (mm)	0.234	0.248	0.251
d90 (mm)	0.282	0.305	0.318
K bulk (cm/s)	0.02	0.03	0.03
K bulk (m/d)	21.03	22.67	24.10

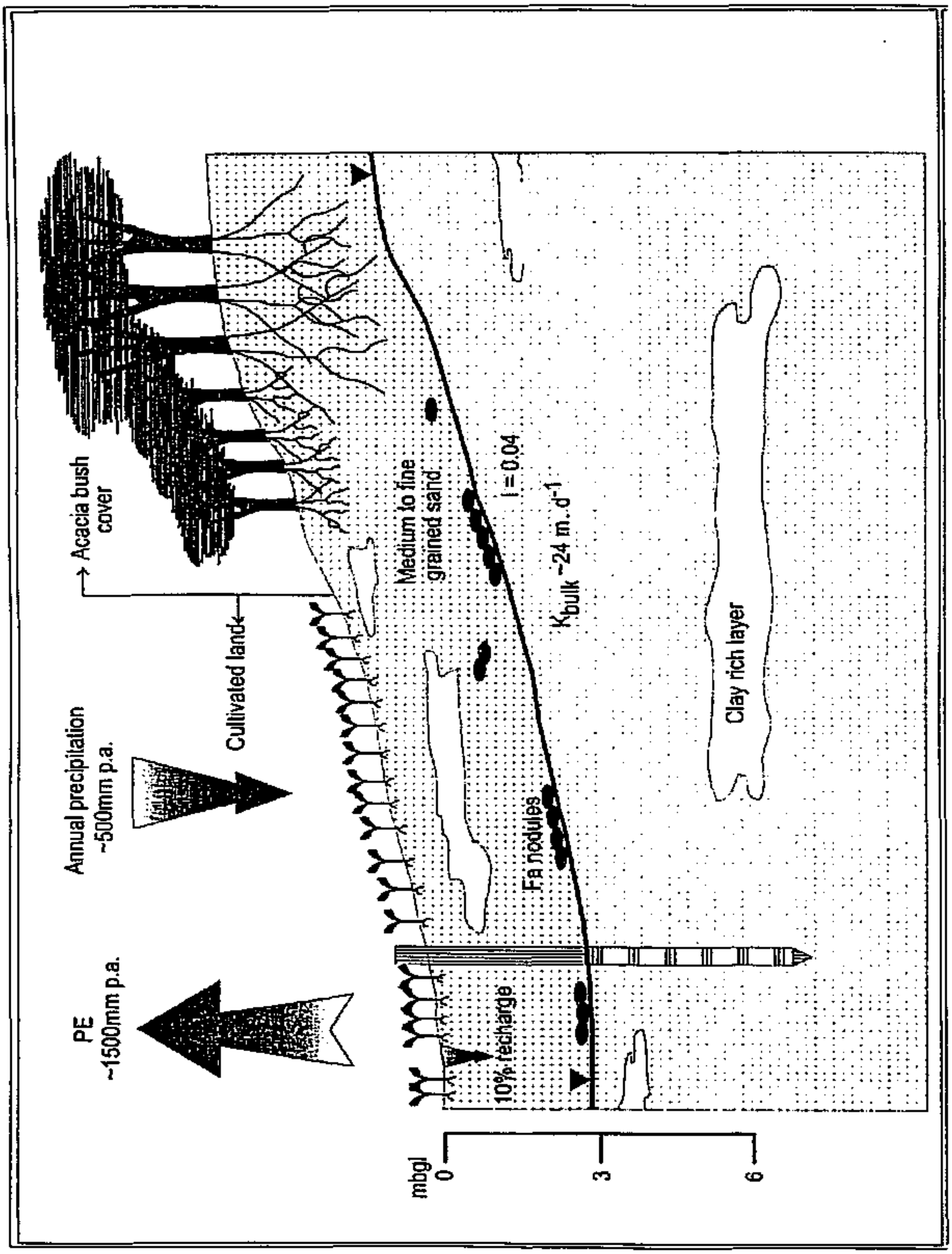


Figure 6.20: Site B: Conceptual Hydrogeology of Site B

The bulk (saturated) permeability is fairly uniform in the samples analysed and averages 22.5 m.d^{-1}. This is comparable to permeability values used for numerical modelling of sandy, unconfined aquifers in the Atlantis area (van der Voort, pers.comm.). It should be noted that the unsaturated permeability will be lower than this value dependent on the water retention characteristics of the soil.

Rainfall in this area averages 500 mm p.a. and potential evaporation 1500 mm p.a. (Midgley, 1994). Studies conducted by the CSIR have shown **recharge** to sandy coastal aquifers in the area to be between 8 and 25% (van der Voort, pers. comm.). A rate of recharge around 10% is expected in vegetated areas such as the study area.

Groundwater flow is thought to follow the local topography and this is borne out by the relative water levels seen in the sampling points. The boreholes have not been surveyed therefore only approximate relative levels can be determined from contour lines on the orthophotograph. This gives a fairly steep gradient of 1 in 26 from the control borehole in the north to the boreholes in the south. It is possible that the control borehole intercepts a perched water table. Boreholes A1, A2 and A4 are located within 50 m of each other on approximately level ground and their rest water levels vary by almost 1 m. This may be a result of interference caused by purging the neighbouring boreholes prior to sampling, or reflect the interception of perched or semi-confined aquifer layers. If due to natural hydrogeological conditions, it indicates limitations to hydraulic continuity within a limited field scale.

Figure 6.22 shows that rest water levels (RWLs) increased at all sampling points by at least 50 cm during the sampling period following the winter rains. March '97 RWLs are slightly higher than those recorded in March '96 (by 10 to 20 cm). This may be due to a higher volume of recharge in the winter of '96 or settling of the casing during the sampling period (the datum used is the top of the casing).

There are no surface water features in the area and run-off is expected to be minimal with this highly permeable cover.

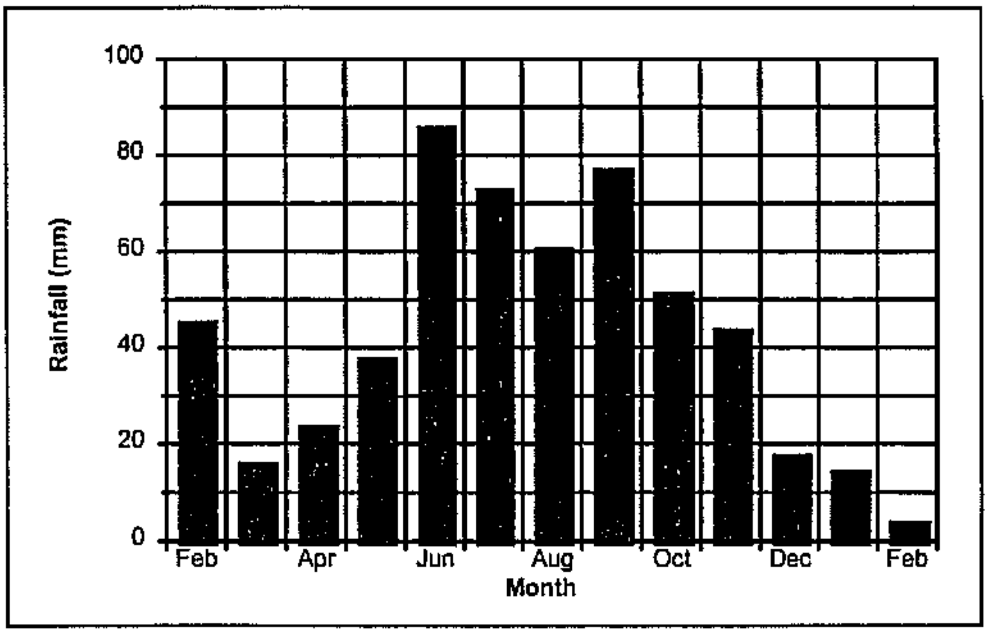


Figure 6.21: Rainfall recorded at Atlantis (Feb'96 to Feb '97)
(Source - Cape Metropolitan Council)

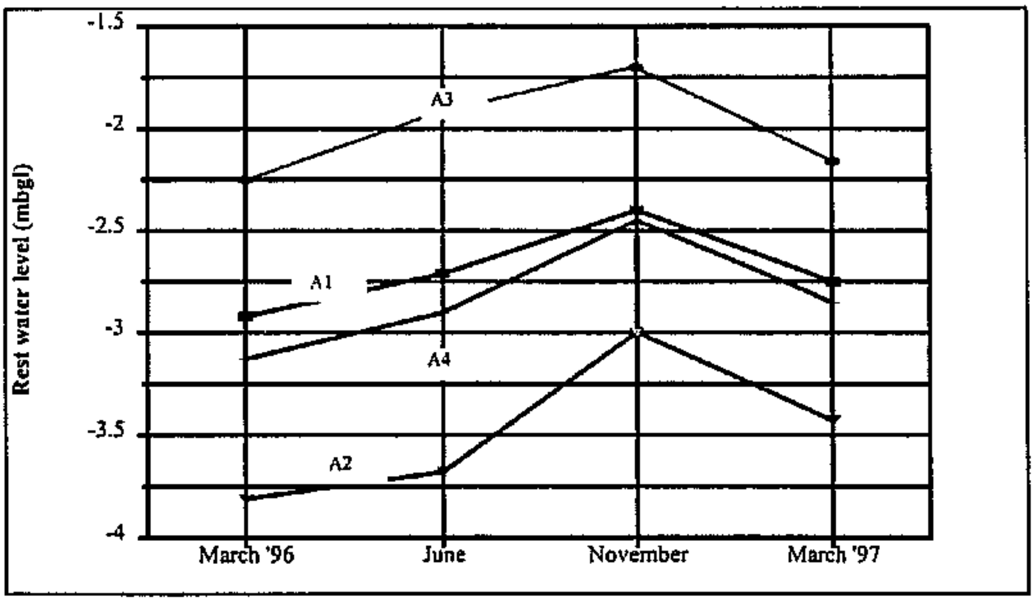


Figure 6.22: Site B: Rest water variations with time (March '96 to March '97)

6.3.6 Results of the field investigation

6.3.6.1 Description and discussion of microbiological contamination indicators

The results of microbiological analysis are presented in full in Appendix 2.C. A heterotrophic plate count was carried out on groundwater samples following the initial sludge application period to gauge the relative changes in microbial activity. The results are shown in Figure 6.23.

There is significant variation in the heterotrophic plate count between groundwater samples during any one sampling run. The level of microbial activity increased significantly in samples from monitoring points A1 and A4 in the sludge applied field during the rains and growing season. The plate count for samples from the control borehole and A2 remained approximately constant between June and November but both increased in March '97. It seems likely that the dramatic increase at A1 in Nov '96 is due to sludge application. However, the data do not conclusively indicate an overall impact as a result of sludge use.

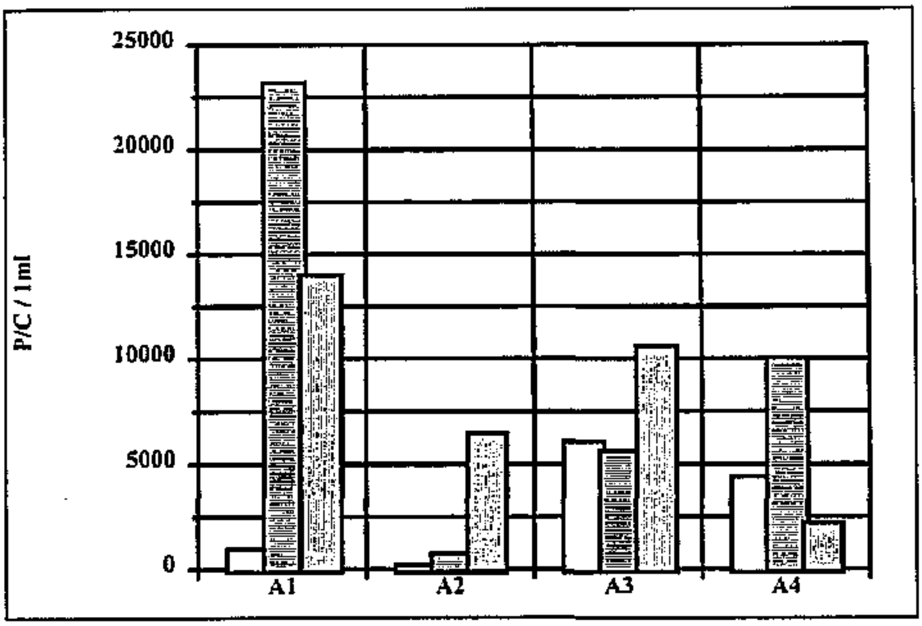


Figure 6.23: Site B: Heterotrophic plate count.

6.3.6.2 *Description and discussion of hydrochemical data*

The results of all hydrochemical analyses are presented in full in Appendix 2.C.

6.3.6.2.1 Metals

Analyses for metals were carried out on the final sampling run in March 1997. Cu was detected at low levels in samples from A1 to A2 (≤ 0.1 mg.l^{-1}). The highest Fe concentration was from the control borehole. Cd, Ni and Zn were not detected. There is therefore no evidence for metal contamination of groundwater as a result of sludge application.

6.3.6.2.2 Major ions

All groundwater samples were analysed for major ions. The quality of the results was checked and in all cases the charge balance error is within acceptable limits. The results of EC field measurements (recorded on the sampling sheets) correlate well with those determined in the laboratory.

Electrical conductivity

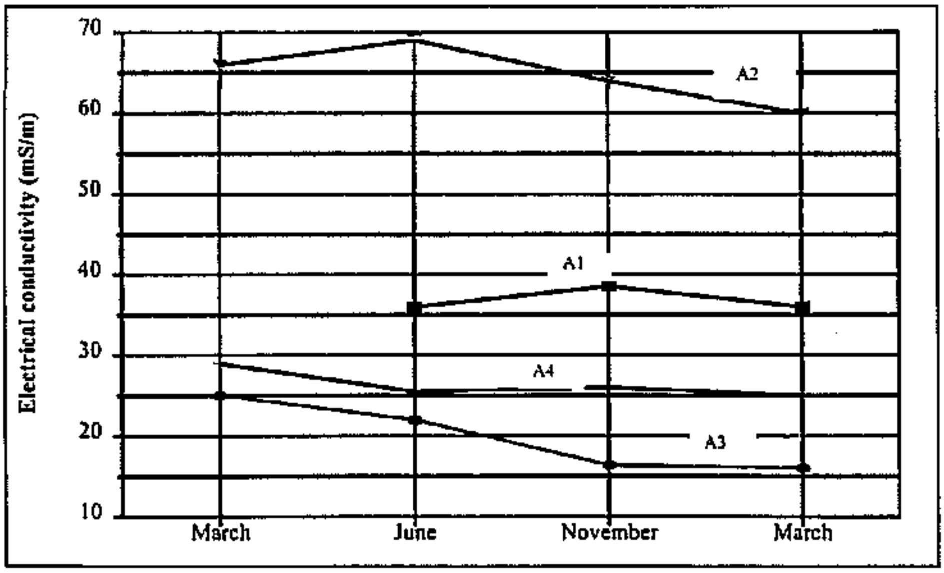


Figure 6.24: Site B: EC variations with time.

EC values are shown on Figure 6.24 to be spatially variable by a factor of 6 but fluctuate insignificantly with time. The variation in values between monitoring points in the cultivated field is greater at 40 mS.m^{-1} than the difference between these and the control monitoring point. This variation is unexpected in a highly permeable, primary porosity aquifer and indicates that there may be localised limits to hydraulic continuity. There appears to be some correlation between the groundwater EC and relative rest water level. The highest EC values are recorded for samples from monitoring point A2 which has the deepest rest water levels below ground level. This may indicate increasing levels of TDS with depth, probably due to residence time and density.

Hydrochemical type.

Piper plot and Stiff diagrams are shown to give an indication of the general hydrochemical character of the samples.

Figure 6.25 shows a **Piper plot** of the results of hydrochemical analyses in the November sampling run. The samples are not tightly clustered but are all of approximately NaCl type. There is no obvious difference between the boreholes from the cultivated field, the uncultivated area and the deep borehole.

The **Stiff diagrams** presented in Figure 6.26 show more clearly the difference in hydrochemistry of groundwater from the different sampling points. It should be noted that the milli-equivalent scale varies in the diagrams. A2 and A4 (with the deepest RWLs below ground level) yielded samples with Na and Cl ions dominant. The sample from A1 had a more significant contribution of nitrate and Ca and Mg, although still of NaCl type. Nitrate is the dominant anion in the A3 November sample. This monitoring point has the lowest EC value and shallowest RWL.

Dissolved organic carbon

DOC levels declined following the first period of recharge and then increased during the summer. Levels measured are low (< 5 mg.l^{-1}) and do not differ significantly between the fertilized field and control monitoring points therefore there is no indication of contamination resulting from the use of sludge.

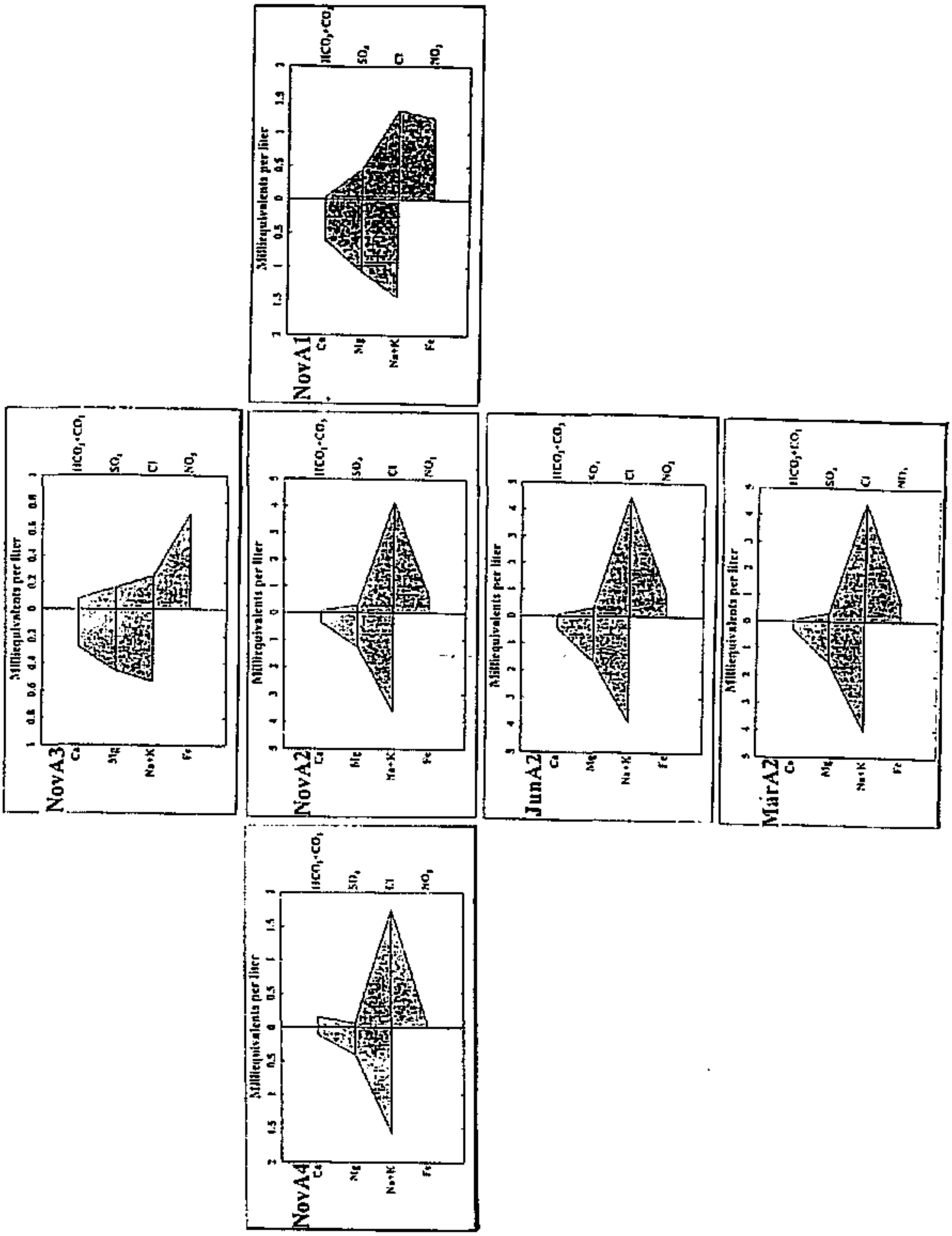


Figure 6.25: Site B: Piper plot of hydrochemical results

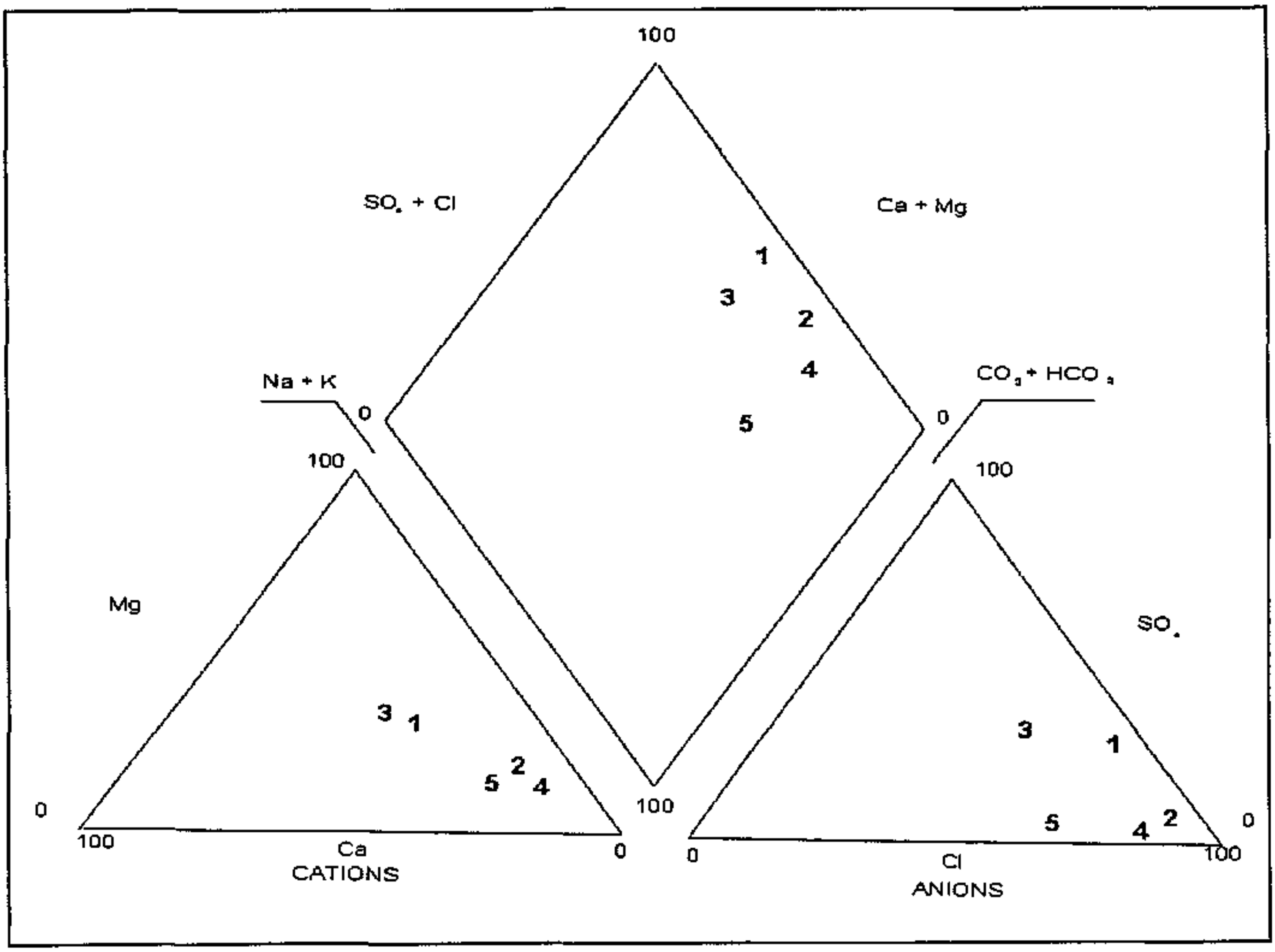


Figure 6.26: Site B: Stiff diagrams of hydrochemistry

Other potential contaminants.

Ortho-phosphate was not detected in groundwater samples at this site. Phosphate is typically attenuated in the first few centimetres of soil.

Potassium levels are low in all samples (< 4 $mg.l^{-1}$) The control samples have the lowest concentration therefore there may be some contribution from fertilizers used in the field but this is more likely to be from the inorganic KCl fertilizer added during planting.

Sulphate concentrations vary from 5 to 20 $mg.l^{-1}$ but are not significantly different in the control and field samples.

6.3.6.2.3 Nitrogen

No significant ammonium was detected in the groundwater samples. TKN was not determined due to the low to negligible ammonium and DOC values obtained.

There is no correlation between EC and NO_3-N concentration, as would be expected where NO_3 is not a dominant anion. However, levels of NO_3 are high and in at least one sample from all monitoring points except A4, exceed acceptable limits for drinking water (10 mg. l^{-1}). Samples from A4 (in the sludge applied field) were low at < 2 mg.l^{-1} and showed no significant variation with time. This may indicate a lower degree of continuity with the surface (although the nature of the unsaturated zone seen during augering did not indicate this to be the case) or local reducing conditions.

Initial NO_3-N levels are between 1 and 10 mg.l^{-1}, as is shown in Figure 6.27, with no significant difference seen between the uncultivated and cultivated land. Results from samples taken in June, after application of sludge, inorganic fertilizer and the first rains, showed an increase in NO_3-N levels in A2 (field) and A3 (control). The increase between samples from A3 on the uncultivated land (to 17.7 mg.l^{-1}) was greater than that seen on the cultivated land. This may be explained by the degradation of Port Jackson in the area. As a nitrogen fixing acacia, Port Jackson has a high proportion of nitrogen in its biomass. The farmer reported that the trees are dying in increasing numbers due to the spread of rust fungus, and the dead wood has been left in the area. If mineralisation and nitrification are proceeding at reasonable rates, the potential for nitrate leaching exists. The soil in the uncultivated area is naturally sandy and has none of the improved water retention characteristics imparted by the addition of sludge to the cultivated soil. The following factors could combine to result in a higher rate of nitrogen leaching from the uncultivated area following the first rains:

- significant nitrogen pool in degrading Port Jackson biomass
- effective mineralisation and nitrification
- low water retention capacity in the sandy soil
- low plant uptake of available nitrogen in uncultivated area.

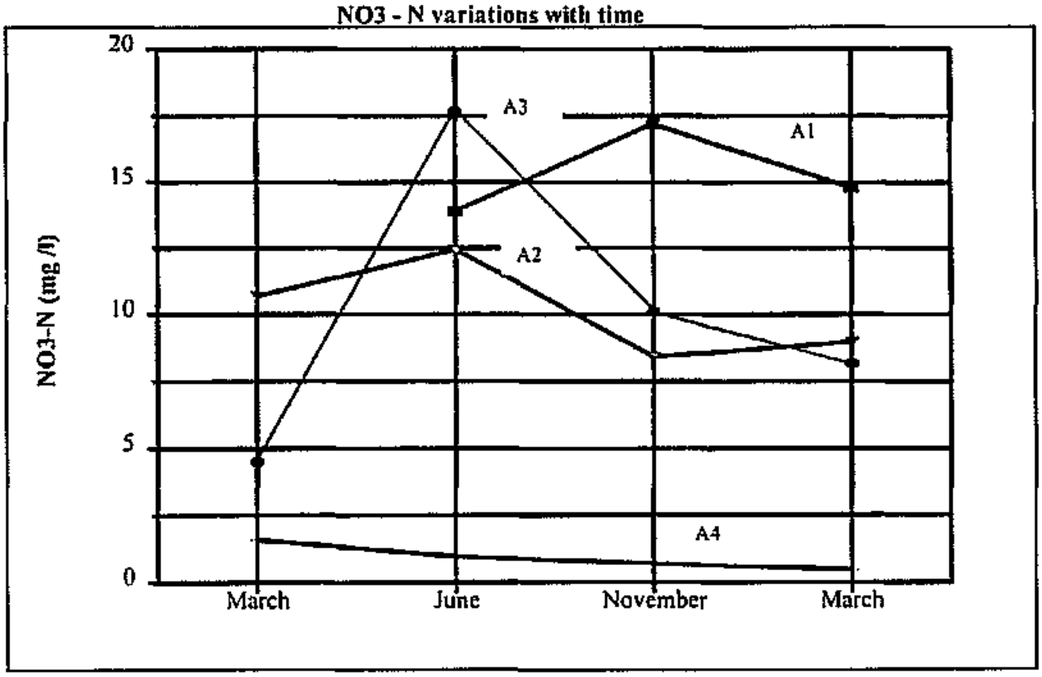


Figure 6.27: Site B: NO3-N variations with time

By contrast, the rate of leaching from the cultivated land could be reduced by the following factors, despite the addition of organic and inorganic nitrogen sources to the soil:

- improved water retention capacity of the soil
- high nitrogen demand and uptake early on in the growing season.

Nitrate levels declined slightly overall between March '96 and '97 at 2 of the 3 boreholes in the sludge applied area. A slight increase from 5 to 8 mg.l^{-1} was seen in the control borehole.
In summary, nitrates in groundwater in the cultivated field and control area show periodic increases following winter recharge. However, the increase in the control area was highest and did not return to pre-winter levels therefore it appears that some accumulation has occurred. This is thought to be due primarily to lower rates of nitrogen uptake in the control area due to the depletion of vegetation.

Nitrate nitrogen isotopes

The nitrogen isotopic ratios of groundwater samples from the uncultivated area, the cultivated area and from the sludge were determined to help identify the sources of elevated nitrate in the study area.

Ground water samples from the November sampling run were analysed. The sample from A3 in the uncultivated area gave a $\delta^{15}N$ value of + 4.8. This is well within the range for nitrogen naturally occurring in soil. The sample from A1 in the sludge applied cultivated area gave a $\delta^{15}N$ value of + 21.9. This indicates a ^{15}N enriched source such as animal or human waste. Animal manure is not used at the site therefore the source is likely to be the sludge.

Two samples of dry sludge from drying beds were analysed for nitrogen isotopes. The isotopic ratios for nitrate nitrogen in the sludge were +15.7 and +43.1, and for ammonium nitrogen they were +27.9 and +23.0. These values are much higher than is found in the groundwater beneath the sludge applied area and is likely to be the result of denitrification having already proceeded to some extent (Talma, pers.comm.). During the winter months, nitrate and ammonia would have been leached from the sludge and migrated to the water table. This leachate produces a nitrogen peak in the groundwater as seen in groundwater in November beneath the sludge applied field. These results therefore confirm that the groundwater nitrate originated from the sludge.

It is interesting to note that inorganic nitrogen applied during planting does not appear to have contributed significantly to the NO_3-N peak seen at A1 in November. This gives the impression that the sludge contributes more significantly to nitrate contamination than inorganic fertilizers. This may be the result of slow prolonged release of nitrogen from sludge as plant uptake wanes. Nitrogen is released more quickly from inorganic fertilizers when the crop requirements are highest. More detailed isotopic analysis with time is necessary to confirm this. Nitrate present in groundwater earlier in the growing season may show an isotopic signature closer to inorganic sources.

6.3.6.3 Description and discussion of soil data

Soil samples taken in November from the control and field area were analysed nitrate and ammonium to give an idea of nitrogen concentrations in the unsaturated zone. Figure 6.28 shows the results which give a snap-shot of nitrogen specification and concentration with depth in the unsaturated zone, but no indication of spatial or temporal variation.

Factors controlling variations in the levels of NO_3-N and NH_4-N with depth are described in section 6.1.3.3.

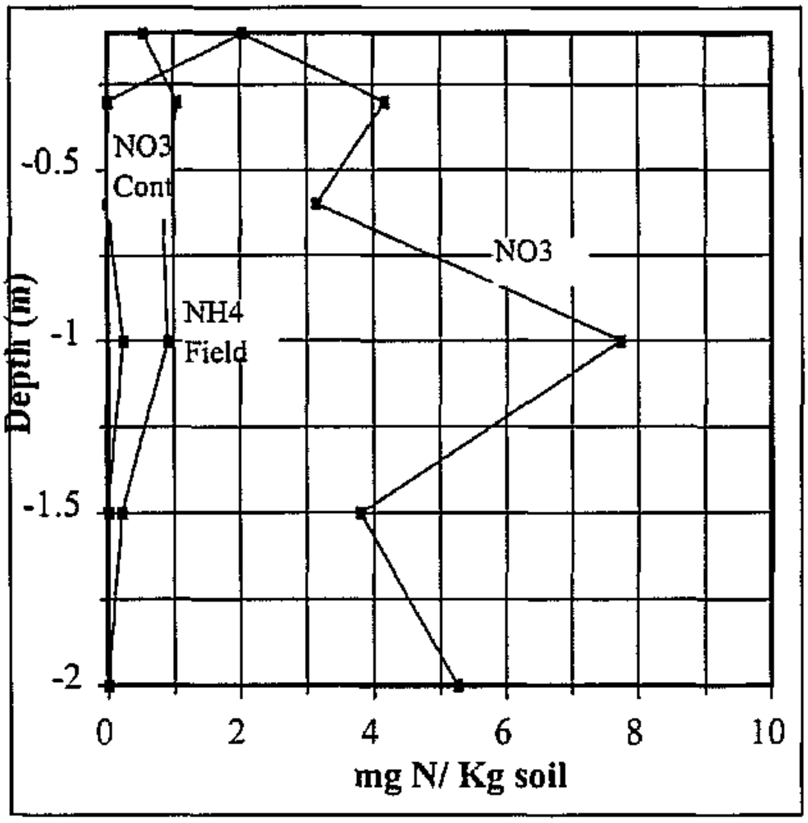


Figure 6.28: Site B: Soil nitrogen profile

No ammonium was detected in samples from the field. This result is regarded with some suspicion and may not reflect field conditions. Nitrate levels are generally higher in the field than in the control area, although results from the surficial root zone are the same at 2 mg.kg^{-1} in each area. Variations in nitrate concentration with depth appear to be erratic and no overall trend is discernible from these data.

The concentration of nitrate in the control area declines with depth. Ammonium concentrations remain slightly higher but also decline with depth.

6.3.7 Summary of impact of sludge application on groundwater quality

The application of sewage sludge to this site appears to have affected groundwater quality at 2 out of 3 monitoring points in the field by temporarily elevating nitrate by 2 to 3 mg.l^{-1} following periods of recharge. This is shown by ion analyses of groundwater from the sludge applied area and the sludge source is validated by N isotope determination. Monitoring over a longer period of time would confirm the degree and periodicity of nitrate variations in groundwater.

It should be noted that at this study site the seasonal increase in nitrate levels in an uncultivated area with decaying Acacia bush was greater than that witnessed as a result of sludge application. This is thought to be due to the lower water (and nutrient) retention capacity of the control area compared to the sludge amended soil and higher crop requirements on the cultivated land during periods of recharge.

During this one year sampling period there was no evidence of significant impact on groundwater quality from other potential contaminants such as phosphate, DOC, COD, EC, metals and microbes. Those shown to have reasonable solubility in analyses of the leachate from the sludge are assumed to be successfully attenuated in the sandy unsaturated zone by a variety of processes (adsorption, biodegradation, volatilization, etc).

6.3.8 Synopsis

Research at this site is focussed on investigating the impact on groundwater quality, over the course of one year, of high loadings of sludge in an area where sludge has not been used before.

Sludge was applied as a dry cake at a loading of around 100 tpa. ha $^{-1}$ before planting barley. Other inorganic fertilizers were applied (in total approximately 150 t.ha^{-1}.ha^{-1}) during planting. The main benefit of the sludge is as a soil conditioner.

This is a winter rainfall area and other studies have shown recharge to be around 10% of precipitation, which is 500 mm pa. The area is not irrigated.

The site is underlain by an unconfined, primary sand aquifer at around 3 mbgl. The unsaturated zone consists of unconsolidated sands with a high bulk permeability of around 22 m.d^{-1} and limited clay rich lenses.

Groundwater hydrochemistry is variable but mainly of NaCl type with TDS concentrations of 100 to 400 mg.l^{-1}.

High nitrate concentrations (> 10 mg.l^{-1}) are found in both the cultivated and uncultivated areas of the site. Temporary increases of 2 to 3 mg.l^{-1} NO_3-N are seen during periods of recharge in the sludge applied area. Nitrogen isotope analyses show the source of the nitrogen to be the sludge. Higher increases of around 12 mg.l^{-1} were seen in samples from the uncultivated control area and the isotopic signature indicates a natural soil source. The levels remained approximately 3 $mg.l^{-1}$ higher than concentrations detected one year earlier at the control monitoring point.

There was no evidence of impact from other potential sludge contaminants in the groundwater.

The results of this investigation show the impact on groundwater quality to be temporary, insignificant and, due to the low level of impact, probably localised. Nitrate levels do exceed recommended limits for domestic consumption beneath the sludge applied area, but ambient levels are also excessive.

6.3.9 Conclusions

The use of sludge as a soil conditioner and fertilizer impacted groundwater quality by providing the source for seasonally elevated nitrate levels. However, a greater seasonal increase was seen in the control area and here nitrate levels increased overall during the year. This is thought to be due to nitrate leaching from decaying Acacia vegetation in an area where the uptake of nitrate is declining. The impact of agricultural sludge use during this one year period is therefore less significant than other nitrate sources found in the environment in this area.

6.3.10 Recommendations for further investigations

Lysimeter sampling of the unsaturated zone during the first rains would give a better understanding of the improved water (and therefore nutrient) retention capacity of sludge amended soil at this site.

It would be interesting to study the site on a more detailed time scale to test the hypotheses put forward in interpreting data collated over one year. Nitrogen isotope analyses of more groundwater samples would also add weight to the ideas discussed and may indicate changes in relative nitrogen source contributions at different times within the growing season and even within individual rainfall events. Levels of dissolved oxygen and Eh-pH should also be monitored.

Sampling over a wider area would give some indication of the extent of elevated nitrate levels associated with both the Acacia decay and sludge and fertilizer use. The establishment of a control site away from both cultivation and Acacia cover would be useful, as would the installation of more monitoring points in the uncultivated Acacia area to give an indication of spatial variability.

6.4 SITE C

6.4.1 Site background

The study area covers 2 commercial vegetable farms in the Cape Flats area. Its location is shown on Figure 6.2. Sewage sludge cake from several WWTPs in the area is used as a fertilizer and soil conditioner. This area was selected for the following reasons:

- it has a long history of sludge application and therefore greater chance of contamination
- the site is underlain by a shallow sandy aquifer with a relatively permeable unsaturated zone which is thought to be vulnerable to contamination
- groundwater is used extensively in the surrounding area and therefore many borehole sampling points are available
- it covers two commercially operated farms and is therefore a good example of economic agricultural use of sewage sludge.

6.4.2 Application of sewage sludge to the site

Sewage sludge has been used at this site since the early 1950s, both as a source of nutrients and to condition the very sandy soil. Dried sludge cake from several WWTPs in the area is mixed with cow and horse manure and applied to the fields once a year. The sludge is usually applied in autumn and early winter at a rate of 2 to 3 $t.ha^{-1}$. It is ploughed into the root zone prior to the planting of potatoes, the main winter crop.

The farmers are aware of the nutrient contribution from the sludge but do not know the range of N:P:K values of either the sludge or other manures. Chicken manure is believed to have the greatest nutrient value, but this is variable depending on the how long it has been standing in the chicken shed. Nutrient contribution from organic sources is taken into account by the farmers qualitatively by adding slightly less inorganic fertilizer than is recommended. The main value of the organic fertilizers is as a soil conditioner. This function is essential for the effective use of inorganic nutrient sources in this very sandy area. Approximately twice as much total organic fertilizer as inorganic fertilizer is applied per year (by weight).

Sludge and manure are stockpiled throughout the year in uncovered areas. Some of the stockpile areas have a concrete base.

6.4.3 Land-use and potential groundwater contamination around the study area

The area is used for intensive vegetable farming. Two to three crops are produced per year, including potatoes, carrots, beetroots, peppers, cauliflowers, etc. Approximately 25 $tpa.ha^{-1}$ of organic fertilizers (including 2-3 $tpa.ha^{-1}$ of sludge) and 13 $tpa.ha^{-1}$ of inorganic fertilizers are used by the farmers. LAN with an N:P:K ratio of 3:1:5/7 and guano are used.

A limited number of houses are situated within the area, some with septic tank systems.

The main potential sources of groundwater contamination in the area, in addition to the use of sludge, are:

- use of manures as fertilizers
- use of inorganic fertilizers
- seepage from septic tanks.

These sources could impact groundwater quality by increasing levels of NO_3, EC, K, P, metals, COD, DOC and microbes.

6.4.4 Hydrogeology of the site

The area is underlain by the Cape Flats aquifer unit. This unit is reported to have a shallow water table at around 4 mbgl, with groundwater of good quality (TDS < 1000 $mg.l^{-1}$) and significant exploitation potential (Maclear, 1995). During augering at the site in December the saturated zone was encountered at approximately 1 mbgl.

The farms are situated on Quaternary sands of the Witzand formation, an unconsolidated sand with a high proportion of shelly fragments. These semi to unconfined sediments are underlain at approximately 40 mbgl by Malmesbury shales and Cape Granite.

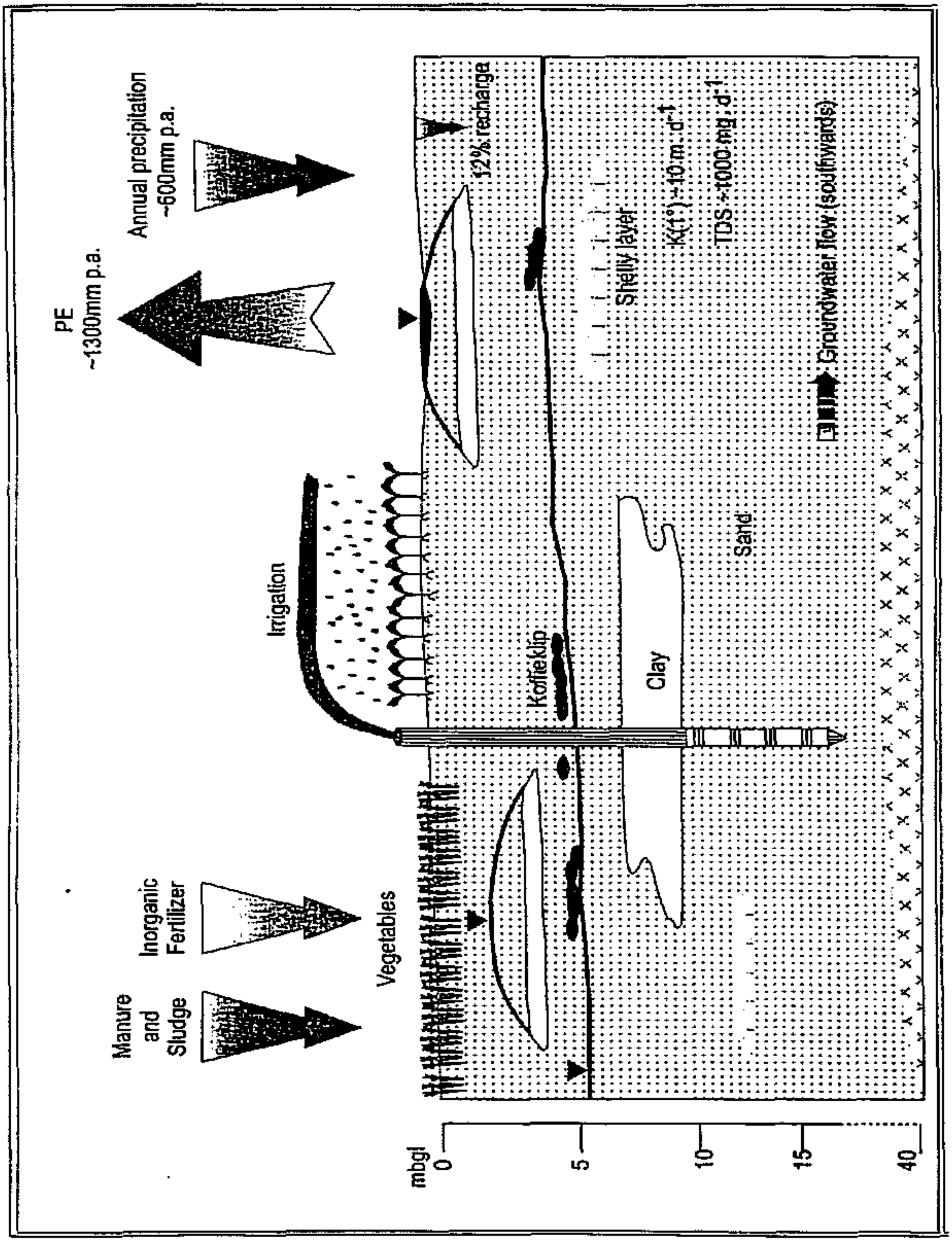


Figure 6.29: Site C: Conceptual hydrogeology at Site C

Figure 6.29 summarises the conceptual understanding of the **hydrogeology** of the area. The unsaturated zone is comprised of permeable well rounded sands with occasional clay rich layers and Fe cemented nodules (koffieklip). Calcretes have formed in the area.

The annual **rainfall** in this area is approximately 600 mm and occurs mainly in the winter months (Midgley *et al.*, 1994). Potential **evaporation** is around 1300 mm p.a. (Midgley *et al.*, 1994). **Recharge** for the Cape Flats unit has been estimated at 12 to 15% of annual precipitation (Gerber, 1980) however this is varies depending on the presence of clay layers.

The transmissivity of the aquifer is reported to be between 50 and 650 $m^2.d^{-1}$ (Tredoux, 1984). This gives a saturated **permeability** of the order of 1 to 20 $m.d^{-1}$.

Groundwater flow directions are assumed to be approximately southwards towards the coast. It was not possible to determine relative RWLs at the sampling points as most were inaccessible to a water level meter and they had not been accurately surveyed.

Seasonal **vleis** are found in this area and mostly represent water tables perched on clay layers (Tredoux pers. comm.). The quality of groundwater above the clay layer is generally more saline than that sampled below as a result of concentration by evaporation. This indicates that although the bulk permeability of the aquifer is high, recharge and therefore hydraulic continuity with the surface is significantly impeded in some areas by relatively thin, laterally limited low permeability layers.

The farmers irrigate with groundwater in the dry summer months. Abstraction rates are not known but the farmers report a decline in borehole yields in recent years which they attribute to increased urbanisation in the surrounding area limiting recharge.

6.4.5 Field investigation

The field investigation was carried out between June and December 1996. Figure 6.30 shows the layout of groundwater and soil sampling points. Sampling sheets giving details of the borehole location, sampling methods, borehole design, etc., are included in Appendix 2.D.

The sampling programme was carried out as follows:

- Borehole 6 is up gradient of the vegetable farming area and was sampled as a control point. Wellhead measurements of EC and pH were made and groundwater samples were analysed for all the determinands listed in section 6.1.5.4, excluding heavy metals. Microbiological analyses were carried out for faecal coliforms and a heterotrophic plate count.
- soil samples were taken from 6 selected areas on the farms for ammonium and nitrate analyses. Sites 1 and 2 are next to the sludge and manure stock pile, site 3 is in an unfertilized area with weed growth, sites 4, 5 and 6 are in fertilized vegetable fields.
- 2 shallow groundwater samples were taken for analysis from augered holes. Hole A was sited next to the uncovered sludge and manure stockpile, in the hope of intercepting any leachate that may have infiltrated, and hole B was augered in a vegetable field where sludge and fertilizers had been applied.

6.4.6 Results of the field investigation

The farmers' boreholes are mainly about 15 years old and cased with steel casing. Borehole 3 is 40 years old and has PVC casing and borehole 2 is only 5 years old. It is not known at what depths the boreholes are screened. The boreholes are drilled to between 30 and 40 m and the pumps are installed at between 25 and 30 m. It can therefore be assumed that samples from the boreholes are generally representative of groundwater from 10 to 30 mbgl. The boreholes are pumped for long periods during the dry months and are reported to deliver between 2 and 7 $L.s^{-1}$.

Boreholes augered to obtain shallow groundwater samples intercepted the water table at around 1 mbgl. Samples are representative of groundwater from the top of the saturated zone at 1 to 2 mbgl.

6.4.6.1 Description and discussion of microbiological contamination indicators

The results of microbiological analysis for pathogenic indicators are presented in full in Appendix 2.D. Deeper groundwater samples were analysed for faecal coliforms and a heterotrophic plate count. Shallow samples were analysed for faecal coliforms.

Figure 6.30: Site C: Position of groundwater and soil sampling

The heterotrophic plate count varies by 3 orders of magnitude from 12 to 1035 p/c per 100 ml across the site. The variation does not correlate to changes in hydrochemical parameters and the value for the control sample is in the middle of the range seen across the fertilized area. Higher levels are not therefore attributable to sludge application with the data available.

Microbial pollution has occurred at borehole 8, where both the heterotrophic plate count (1035 p/c per 100 ml) and level of faecal coliforms (11 per 100 ml) are above the recommended SABS limits, and at borehole 7 where 3 per 100 ml faecal coliforms are recorded. The pollution source is not known.

6.4.6.2 Description and discussion of hydrochemical data

The results of all hydrochemical analyses are presented in full in Appendix 2.D. The hydrochemical data is deemed to be of good quality as there is a good correlation between field and laboratory measurements and the charge balance error is within acceptable limits (<5%).

6.4.6.2.1 Metals

The metal content of the sewage sludge used is not known. Analyses from other sites have shown a very limited impact from metals in sludge, as would be expected due to their relatively low mobility. Sludge at this site is mixed with various animal manures prior to application and the loading is lower than that seen at other study sites. It was decided therefore that the likely impact of metals on groundwater would be negligible and these were not analysed for.

Low levels of iron were present in all groundwater samples excluding the sample from borehole 3 which contained 62 mg.l^{-1} Fe. This is the oldest borehole in the survey (drilled 40 years ago) and is reportedly installed with uPVC casing. The reason for the anomalously high Fe concentration is not clear.

6.4.6.2.2 Major ions

Electrical conductivity

EC of the deeper groundwater samples varies from 106 mS.m^{-1} (TDS 678 mg.l^{-1}) at borehole 1 to 350 mS.m^{-1} (TDS 678 mg.l^{-1}) at borehole 4. There is no spatial trend in the variation. The EC of the shallow samples corresponds to lower values for deeper groundwater at 84 (A) and 143 (B) $mS.m^{-1}$.

Hydrochemical type.

Piper plot and Stiff diagrams are shown to give an indication of the general hydrochemical character of the samples.

Figure 6.31 shows a **Piper plot** of the results of hydrochemical analysis. Calcium is the dominant cation in all samples. The shallow samples (which plot as 1 and 2 on the diagram) have HCO_3 as the dominant anion, while samples from the deeper boreholes do not have a dominant anion type. The spread of the data points along a line of increasing HCO_3 does not correlate to spatial distribution along a flow path or to increasing TDS.

A selection of results plotted as **Stiff diagrams** are presented in Figure 6.32. It should be noted that the milli-equivalent scale varies in the diagrams. P1 represents a low TDS sample (678 $mg.l^{-1}$) and P6 is the control (TDS 986 $mg.l^{-1}$). These samples show dominant $CaCO_3$ with NaCl, possibly indicating mixing of groundwater types. P4 represents a high TDS sample (2240 $mg.l^{-1}$) with $CaCO_3$ and NaCl more evenly balanced and significant contribution from SO_4. SB is the shallow sample which has a TDS of 915 $mg.l^{-1}$. It is similar to the lower TDS and control well samples with dominant $CaCO_3$ and NaCl, but also has significant NO_3.

Figures 6.33 and 6.34 show that Cl and SO_4 concentrations show a positive correlation to EC. Variations in salinity are therefore thought to be related to different degrees of mixing with more saline groundwater (probably connate water) at depth.

Potassium is also found at high concentrations (around 60 $mg.l^{-1}$ compared to 6 $mg.l^{-1}$) in high TDS groundwater.

A rotten egg smell was noted at most of the deeper boreholes during sampling. This is probably due to hydrogen sulphide gas escaping at the well head during pumping and indicates reducing conditions in the aquifer.

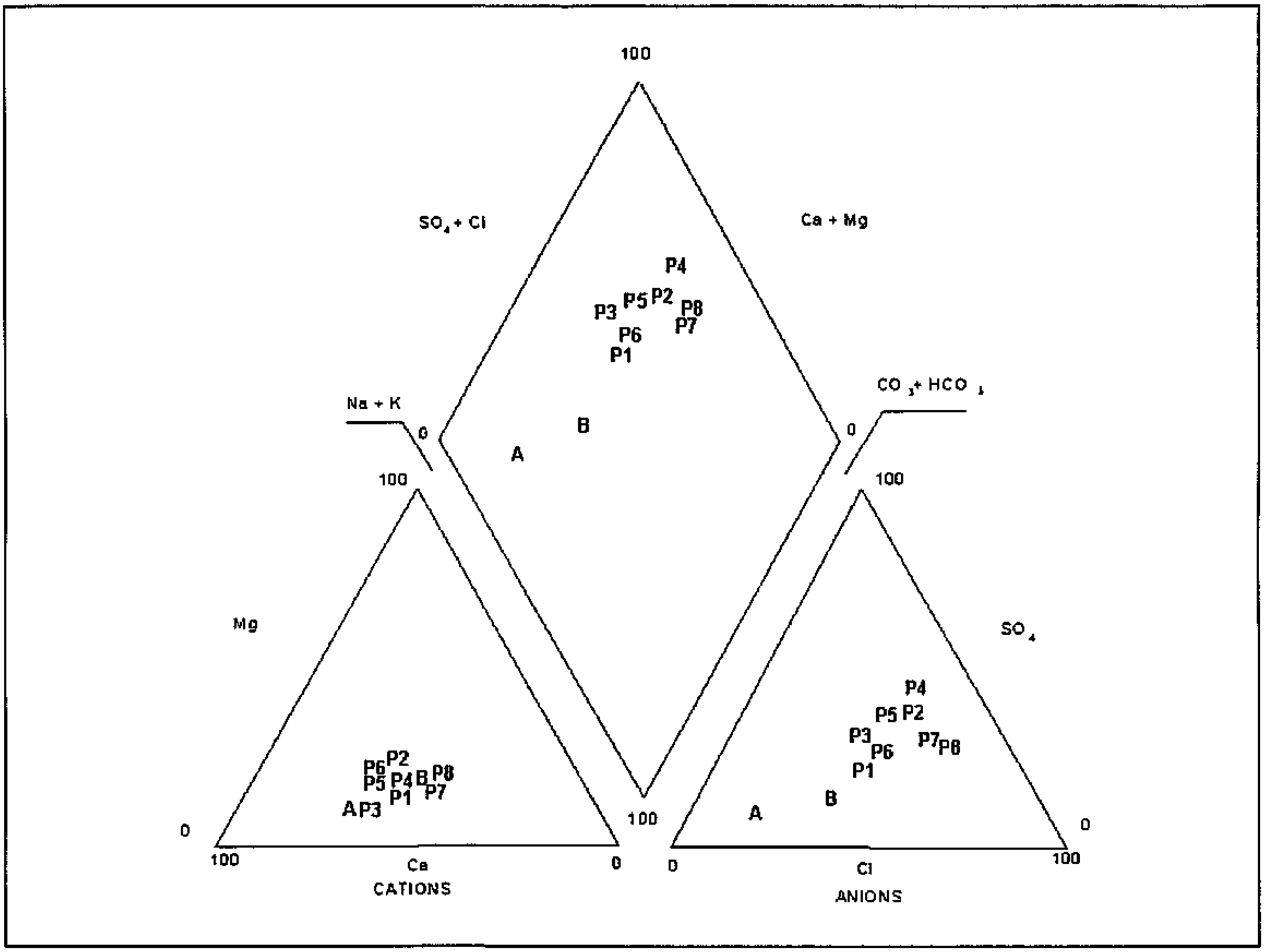


Figure 6.31: Site C: Piper plot of hydrochemical results.

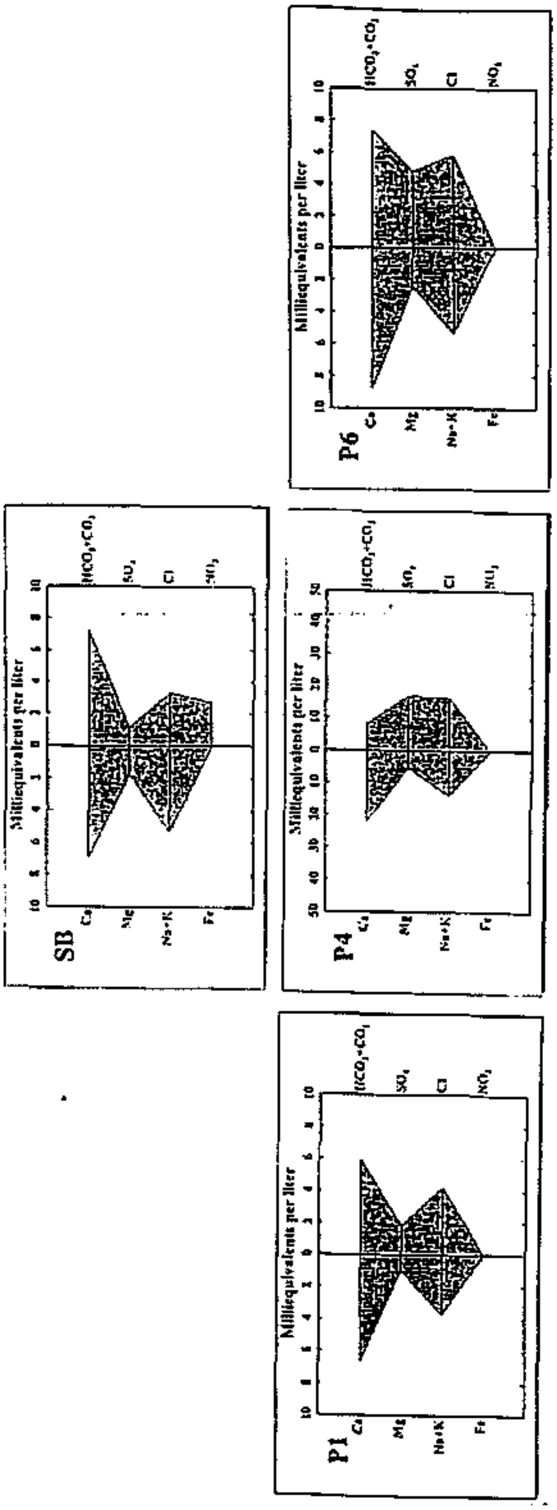


Figure 6.32: Site C: Stiff diagrams of hydrochemistry

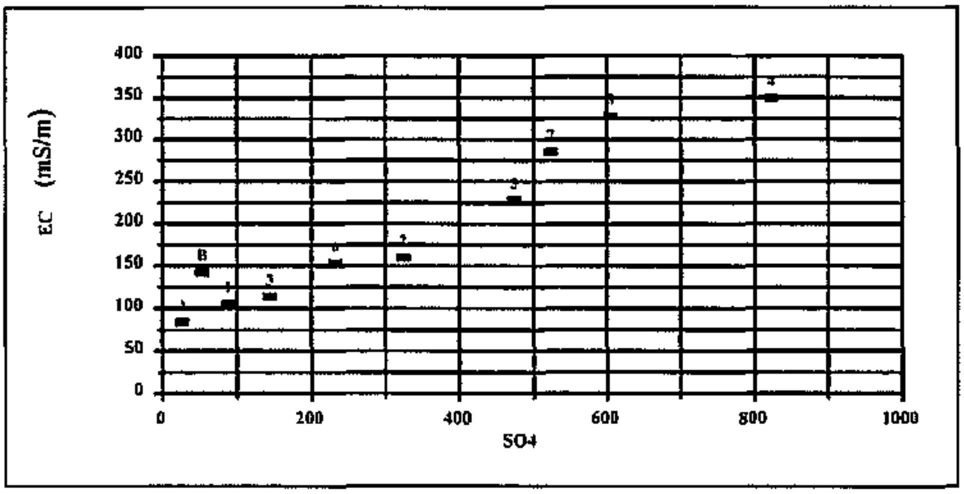


Figure 6.33: Site C: EC versus SO_4 (borehole numbers indicated)

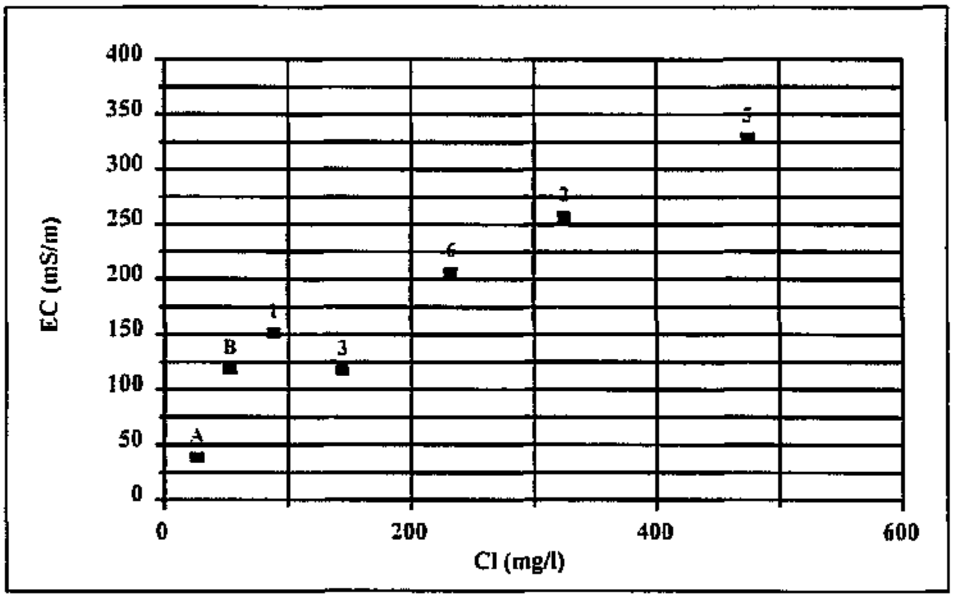


Figure 6.34: Site C: EC versus Cl (borehole numbers indicated)

Nitrogen

Very low levels of **ammonium** were detected in all the groundwater samples. The mean value for the shallow samples was 0.21 mg.l^{-1} and for the deeper samples was 0.71 mg.l^{-1}. The concentration for the control sample was the lowest of the deeper groundwater samples at 0.41 mg.l^{-1}. There is no correlation between ammonium values and EC.

Nitrate was detected in 2 of the 8 samples from deeper boreholes at less than 1 mg.l^{-1}. High concentrations of nitrate were found in the shallow samples: 9.7 mg.l^{-1} in the sample taken close to the manure and sludge stockpile and 38 mg.l^{-1} in the sample from the fertilized field.

Shallow groundwater at the site is therefore contaminated with respect to nitrates. It is likely that this is as a result of fertilizer application at higher than agronomic rates and leaching losses in this highly permeable environment. The available data do not indicate whether one fertilizer source contributes more to nitrate leaching than another. Nitrogen isotope analysis would differentiate between the inorganic and organic sources but would not separate the relative contribution from sludge and manures. It is thought likely that all the nitrogen sources contribute to some degree to groundwater contamination. The use of organic fertilizers enables a reduction in the rate of inorganic nitrogen application and improves the water retention capacity of the sandy soil. Their use may result in an overall reduction in the total leaching losses of nitrate compared to the application of inorganic fertilizers alone. However, the application of inorganic fertilizers is timed more precisely to crop requirements. Organic fertilizers continue releasing nitrogen at times when plant uptake is low, therefore more of the total nitrogen released from sludge and manure may be lost to leaching.

Nitrate contamination of groundwater is not persistent with depth in this environment. This is thought to be primarily as result of reducing/ denitrifying conditions in the aquifer, indicated by the slightly higher ammonium concentrations and presence of hydrogen sulphide at the well heads during pumping.

Potassium

Higher levels of potassium are seen in the shallow groundwater samples and the sample from borehole 4, the only deeper groundwater sample with detectable nitrate (0.57 NO_3-N mg.l^{-1}). Shallow samples contain 60 mg.l^{-1} compared to between 2 and 15 mg.l^{-1} for the deeper samples. The source of the potassium may be either the organic fertilizers or the inorganic N:P:K fertilizers. The latter has the higher concentration.

Dissolved organic carbon

Dissolved organic carbon levels are elevated in the shallower groundwater samples with values of 28 and 29 mg.l^{-1}, compared to 5 to 17 mg.l^{-1} in the deeper samples. All these concentrations are higher than would be expected due to natural organic matter in the soil.

The higher levels in the shallow samples indicate both a surface source of DOC and moderate degree of attenuation, probably by microbial breakdown, in the aquifer with depth. The most likely source of shallow DOC is the sludge and manures. It should be noted that limited peat deposits are known to occur in this area and may also contribute to DOC in the deeper groundwater.

The deeper groundwater still contains DOC at levels which exceed the SABS recommended limit for drinking water of 5 mg.l^{-1}. The lowest level is seen in the control sample, further indicating a fertilizer source. COD and DOC show a strong positive correlation as would be expected.

6.4.6.3 Description and discussion of soil data

Figures 6.35 to 6.37 show the results of analyses of augered soil samples for water soluble and potassium chloride extractable nitrate and (in two cases) ammonium. Samples 1 and 6 were taken in December and samples 2 to 5 were taken in July.

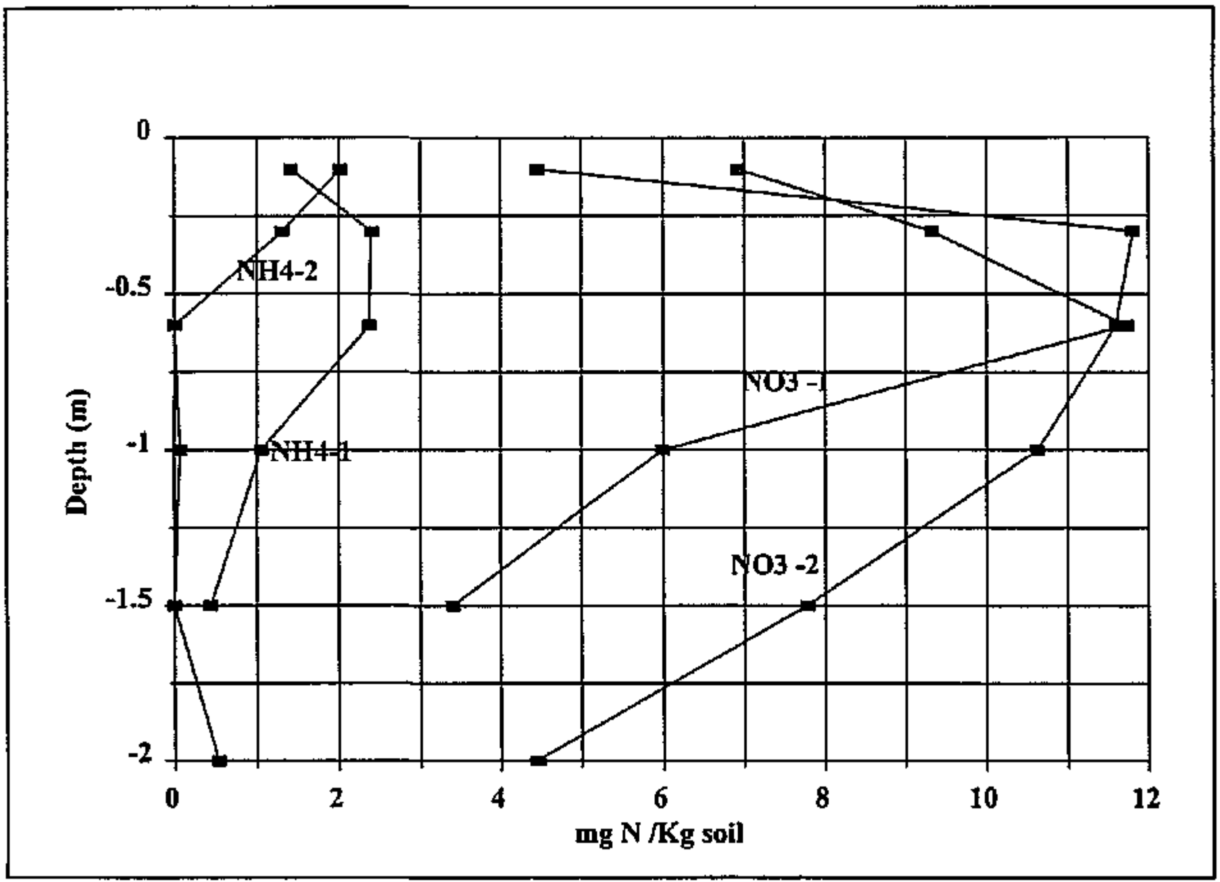


Figure 6.35: Site A: Nitrogen soil profile of sampling points 1 and 2

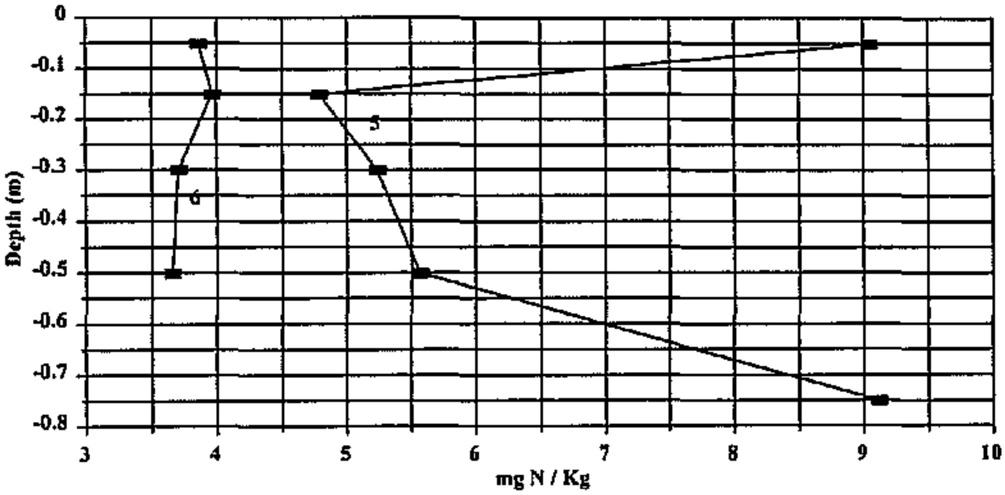


Figure 6.36: Site A: Nitrogen soil profile of sampling points 6 and 5

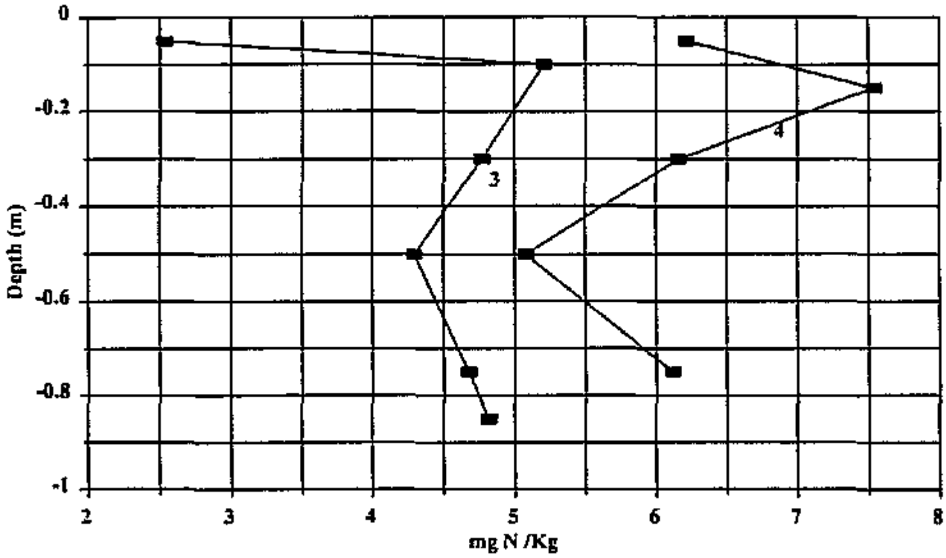


Figure 6.37: Site A: Nitrogen soil profile of sampling points 3 and 4

Sampling sites 1 and 2 are close to sludge and manure stockpiles which are apparently sited on concrete slabs. The extent and integrity of the slabs is not known but they are more than ten years old and therefore cracks allowing vertical migration of leachate may exist. Site 2 shows fairly low concentrations of nitrate (around 3.7 $mg.l^{-1}$) from 0.05 to 0.5m. The levels of nitrate at site 1 are around 4 $mg.l^{-1}$ near to surface and at 1.5 mbgl at the water table, but peak to almost 12 $mg.l^{-1}$ at approximately 0.5 mbgl. Site 2 represents a profile from early in the winter rains and site 1 was augered in summer after the rains. The nitrate peak at site 1 may indicate a leaching front.

Sites 4 and 5 were augered in fertilized areas in winter and show nitrate levels declining in the unsaturated zone and then increasing towards the water table. This may indicate relatively low rates of mineralisation and nitrification in the unsaturated zone at this time of year. Site 6 was augered in the summer and is also from a field fertilized with both organic and inorganic nitrogen sources. The concentration of nitrate nitrogen increases to a peak of around 11 $mg.l^{-1}$ at 0.5 to 1 mbgl and then declines with depth to 4.5 $mg.l^{-1}$ at the water table. This peak may also represent a leaching front similar to that seen at the stockpile at the same time of year.

Site 3 is unfertilized and was sampled in July. Nitrate levels in the root zone are the lowest seen at all the sites (2.5 $mg.l^{-1}$) and levels in the unsaturated zone correspond with the non-peak levels seen in the unsaturated zone beneath fertilized fields.

6.4.7 Summary of impact of sludge application on groundwater quality

From the data available it is not possible to differentiate between fertilizer sources of nitrogen. However, it is considered likely that sludge may have contributed to elevated levels of nitrate and dissolved organic carbon in the groundwater.

Nitrate was detected in shallow groundwater at harmful levels of 10 to 38 $mg.l^{-1}$. A higher degree of contamination appears to have occurred in the vegetable fields than around the sludge stockpile. This may be the result of irrigation in the fields, indicating that the rate of infiltration has a greater affect than loading. More shallow sampling points are needed to determine the degree of spatial variation. Analysis of soil samples showed evidence of leaching of nitrate from fertilized fields and the stockpile area.

No significant nitrate is detected in deeper groundwater samples. This may be due to either a lack of hydraulic continuity with depth or reduction of nitrate with depth. Slightly elevated levels of ammonium and the presence of hydrogen sulphide indicate the latter.

The application of sludge and other fertilizers does not appear to have significantly affected the levels of EC, major ions or microbial indicators at this site.

6.4.8 Synopsis

Sludge cake has apparently been used as a soil conditioner and fertilizer in this area for approximately 40 years. It is currently applied at low loadings of around 3 $tpa.ha^{-1}$ but with high levels of other organic (22 $tpa.ha^{-1}$) and inorganic (13 $tpa.ha^{-1}$) fertilizers.

The site overlies the Cape Flats aquifer unit. This is a sandy, unconfined aquifer with a bulk permeability of between 1 and 20 $m.d^{-1}$. The water table was intercepted at approximately 1 mbgl.

Other potential sources of contamination identified at the site include manures, inorganic fertilizers and pit latrines.

Multilevel groundwater and soil samples were analysed for potential contaminants. Most groundwater samples were abstracted from established farmers' boreholes which intercept the aquifer at between 10 and 30 mbgl.

EC varies by a factor of 3 across the site in samples from the deeper boreholes. Shallow groundwater has a lower TDS. The hydrochemical type is $CaCO_3$ with NaCl. High TDS samples have a significant NaCl component with SO_4 and K, possibly indicating mixing.

Shallow groundwater samples are contaminated with respect to DOC and nitrate. The data indicate that this is due to fertilizer application but the degree of contribution from the sludge cannot be ascertained from the information available.

Shallow groundwater samples have significantly elevated nitrate at 10 to 38 $mg.l^{-1}$. These levels exceed drinking water recommended limits. Nitrate contamination is not persistent with depth, probably due to reducing conditions in the aquifer.

Shallow groundwater samples have high levels of DOC at around 29 $mg.l^{-1}$. Many of the deeper samples from the fertilized area have elevated DOC concentrations at 10 to 17 $mg.l^{-1}$.

6.4.9 Conclusions

Sludge and manures are used in this sandy area primarily as soil conditioners. Inorganic fertilizers are also used at slightly less than the recommended agronomic rate. The nutrient contribution from the organic fertilizers is not fully quantified and the total nitrogen loading in the area is likely to exceed crop requirements.

Although the water retention capacity of the soil is undoubtedly improved by the addition of sludge and manures, leaching of excess nitrate occurs. Soil profiles indicate that this was more pronounced in December than July, possibly as a result of increased nitrification rates due to higher temperatures and optimal soil moisture conditions combined with lower crop requirements.

The source of the nitrate is not proven. Assuming that mineralisation of organic nitrogen proceeds unimpeded, the relative contribution to groundwater from organic sources may be higher as a result of continual slow release . Inorganic nitrogen release is more rapid and applications are timed more precisely to crop requirements.

Nitrate contamination of groundwater is not persistent with depth in this aquifer. Groundwater abstracted from 10 to 30 m contains negligible nitrate, probably as a result of denitrification to nitrogenous gas (ammonium levels are also low).

DOC contamination of groundwater is thought to occur as a result of sludge and manure application. There is some attenuation with depth, from approximately 28 to 19 $mg.l^{-1}$, probably by microbial action.

Other potential contaminants are either applied at insufficient loadings or are attenuated successfully in the 1 m thick sandy unsaturated zone.

6.4.10 Recommendations for further investigations

Further shallow groundwater sampling in areas of varied land use may show which practices lead to the highest levels of DOC and nitrate contamination.

More detailed multi-depth sampling will give a clearer picture of the processes operating in this aquifer system. These should include sampling for major ions, nitrate, ammonium, dissolved oxygen and Eh-pH conditions. Detailed lithological logging to 35 m will give a better idea of the hydraulic continuity with depth in this section of the aquifer.

Nitrogen isotope analysis of shallow groundwater samples in the area will give an indication of the relative contribution from inorganic and organic fertilizers. Isotopic characterisation of the different organic sources (if evident) could be carried out to try and differentiate between nitrate contributions. However, it is likely that there would not be a significant difference in the isotopic signatures of sludge and the various manures.

6.5 SUMMARY OF THE FIELD INVESTIGATIONS

The findings of the field investigations are summarised in the table below.

Summary of findings of the specialist field study on the impact of sludge application to agricultural land on groundwater quality.

	Site A	Site B	Site C
Sludge application	Liquid sludge ploughed in throughout the year	Sludge cake ploughed in early winter	Sludge cake ploughed in early winter
Loading	High (?)	High, 100 $t.ha^{-1}.a^{-1}$	Low, 3 $t.ha^{-1}.a^{-1}$
Primary purpose	Sludge disposal	Soil conditioning	Soil conditioning
Use of other organic fertilizers in sludge area	No	No	Yes
Use of inorganic fertilizers in sludge area	No	Yes	Yes
Irrigation	Yes	No	Yes
Field data conducted as part of this project	Shallow groundwater Deeper groundwater Soil profiles	Shallow groundwater Deeper groundwater Soil profiles	Shallow groundwater Deeper groundwater Soil profiles
Climate	700 mm p.a summer rain	500 mm p.a winter rain	600 mm p.a winter rain
Unsaturated zone	7 m thick, clayey soil	3 m thick, sandy.	1 m thick, sandy.
Aquifer type	Dolomite with secondary permeability of ≈ 10 m/d	Unconsol. sand with primary permeability of ≈ 20 m/d	Unconsol. sand with primary permeability of ≈ 10 m/d

	Site A	**Site B**	**Site C**
Hydrochemistry	CaCO3 with elevated nitrate and sulphate. TDS 700 - 1700 mg.l^{-1} Reduced at depth.	NaCl. TDS 100 - 400 mg.l^{-1}.	CaCO3 with NaCl. TDS 540 - 2200 mg.l^{-1} Reduced at depth.
Groundwater contamination from other sources	Sulphate (?)	Nitrate - decaying Acacia vegetation	Nitrate and DOC from other fertilizers (?)
Groundwater contamination from sludge:			
Nitrate	Yes - to > 30 mg.l^{-1} at a distance of > 1 km with a peak of 268 mg.l^{-1}.	Yes - seasonal, localised (?), by 2 mg.l^{-1} to 15 mg.l^{-1}.	Yes (?) - contribution to extensive (?) increase to 38 mg.l^{-1} in shallow samples.
EC	Yes - from ≈ 180 to 300 mS/m due to high nitrate levels	No	No
DOC	Yes - from < 2 mg.l^{-1} to ≈ 10 mg.l^{-1} in shallow samples in the field	No	Yes (?)- to ≈ 30 mg.l^{-1} in shallow samples in the field
Microbiological indicators	No	No (?)	No (?)
Metals	Yes (?) - Cd to 0.01mg.l^{-1}	No	ND (?)
Other ions	Yes (?) - K	No	No
Impact of sludge application on groundwater quality.	Long term, significant and extensive.	Temporary, insignificant and localised (?)	Long term (?), significant at shallow depths, extensive (?).

(?)	Further research required to confirm.
ND	Not determined

The hydrochemical results correspond well to those reported in the literature. Nitrate is the principal contaminant of concern at all the sites. At sites A and B the source of nitrate in the groundwater was shown by isotope analyses to originate from the sludge. This could not be shown at site C as other organic sources were also used as a fertilizer and soil conditioner.

The extent of nitrate contamination is variable and appears to be controlled the hydrogeological conditions (relative rates of nitrification, leaching and reduction) as well as the cumulative nitrogen loading. At site A nitrate contamination was laterally extensive in the direction of anisotropic flow. This is thought to be mainly due to the high cumulative loading of nitrogen at the site, high permeability in the aquifer and continuous unconfined conditions. Nitrate was not detected in boreholes that intercepted deeper sections (>40 m) of the aquifer, probably as a result of reducing conditions. At site C nitrate was only seen at concentrations of concern in very shallow groundwater and appears to have been attenuated at between 10 and 30 mbgl by reducing conditions in the aquifer. The spatial extent of elevated nitrate levels at site B cannot be determined from the data available.

DOC also contaminated groundwater to levels above those recommended for drinking water at sites A and C in shallow groundwater close to the sites. The source of DOC at site C is not confirmed to be the sludge and is more likely to be manures used with the sludge. This contaminant is not persistent with distance from the sludge applied area.

Other contaminants were detected only at site A, therefore their occurrence may be linked to the long term impact of liquid sludge at high loadings. TDS levels are elevated close to the sludge applied area, primarily reflecting the high contribution of nitrate to total dissolved solids. A slight increase in cadmium and potassium levels may also be attributable to sludge application but further data are required to confirm this.

There was no indication of contamination from microbes, most heavy metals and phosphate, indicating that these potential contaminants are successfully attenuated in the unsaturated zone.

It is not possible to compare the degree of contamination at the different sites and relate these to different controlling factors due to the number of variables. The hydrogeological setting at sites B and C are similar (unconfined sandy aquifer) but the rate of sludge application, crop requirements, loading of other fertilizers used, irrigation regime, etc differ between the sites and the relative controls on the rate of nitrate and DOC leaching to groundwater cannot be quantified with the data available. The aim of this study is to determine if groundwater is contaminated as a result of the agricultural use of sludge rather than to quantify the hydrogeological and agronomic conditions under which contamination occurs.

The studies indicate that cumulative loading may have a strong influence on the degree of contamination. It should be noted that at the 2 sites where elevated nitrate levels in the groundwater were shown by isotopic analysis to be due to sludge application, the rates of application far exceed those recommended in the DNH&PD guidelines. Site C provides an example of loadings within the recommended limits but the impact here is influenced by high manure loading rates.

6.6 RECOMMENDATIONS FOR THE NON-TECHNICAL BOOKLET

One of the objectives of this study is to gather data and information which can be incorporated into a non-technical booklet to inform farmers on how they can avoid contaminating ground water when applying sludge to their land. It is proposed that the non-technical booklet should attempt to increase farmers' awareness in the following areas in order to mitigate groundwater contamination.

- Introduce farmers to the concept of aquifer vulnerability and the factors which affect contaminant transport to the saturated zone. Eg. depth to water table, soil permeability, recharge, pathways of preferential vertical migration.
- Discuss the different processes of nitrogen release from sludge and other fertilizers and highlight the economic waste of inappropriate and over application which lead to leaching losses.
- Publicise common nitrogen crop requirements and sources of information on fertilizer application regimes.
- Introduce farmers to groundwater hydrochemistry and the emphasise the importance of keeping ground water quality within drinking water limits. If ambient nitrate levels are already high, as was seen at site B, farmers will need to take extra care.
- The health risks associated with elevated nitrate levels, both to humans and livestock, should be outlined.
- Introduce the guidelines given by the DNH&PD on the agricultural use of sewage sludge.
- Briefly discuss integrated contamination control with respect to the use of other fertilizers and other pathways of environmental pollution. A realistic understanding of farmers' requirements should be taken into account so that recommendations to minimise contamination from sludge do not result in pollution from other fertilizer or soil conditioner source.

In summary, the non- technical booklet should aim to increase farmers' awareness of groundwater contamination risks. It should adhere to the principles of integrated pollution control and make practical recommendations for best agricultural management practice.

CHAPTER 7

THE EFFECT OF IRRIGATION ON GROUNDWATER SALINITY IN SOUTH AFRICA

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7.1 INTRODUCTION

In arid and semi-arid regions, where agricultural productivity is largely dependent on the availability of streamflow or groundwater, precipitation is usually deficient to meet crop demands. Historically, irrigation has played a vital role in agricultural production and communal growth. Civilizations have risen and fallen with the growth and decline of irrigation systems (Schilfgaarde, 1994). Although irrigation has enormous benefits, there are also associated disadvantages, one of which is chemical deterioration of downstream water resources and sometimes of soils. Irrigation water always contains some salt. Evapotranspiration will consume some of the water but not the salt, thereby concentrating salt in the remaining soil water. Irrigation water can also help mobilize soluble salts present in soils. In this regard, irrigation water can accelerate the release of salts from natural soils. The enriched soil water may travel downwards to underlying groundwater, causing salinities there to increase.

7.1.1 Aims

The main aims of this essay are

- to give examples of the effect of irrigation on groundwater salinity in South Africa.
- to outline management options that can be utilized to minimize groundwater salinity arising from irrigation.

7.1.2 Report layout

In the next section, a summary of irrigation from groundwater in South Africa is given. This is followed by a description of the processes that are likely to operate in irrigated aquifer systems. Example case studies of three irrigated areas, (i) The Vaalharts basin (ii) the Great Fish and Sundays River basin and (iii) the Breede River basin, are then given. These intensely irrigated areas are used here to highlight the effect irrigation may have on groundwater quality. The last section describes management options that can be used to minimize groundwater salinity arising from irrigation.

7.2 GROUNDWATER AND IRRIGATION

Braune and Coetzer (1992) conducted a study on the problems of irrigation from groundwater. They subdivided the country into seven Agricultural Districts. Table 7.1, 7.2 and 7.3 summarize data on irrigation from groundwater in the districts. This information should be seen in the context of a total area under irrigation of 1.3 million ha (Department of Water Affairs and Forestry, pers. comm.1997)

Table 7.1: Irrigation from groundwater in Agricultural Districts.

Agricultural district	1976 Census			1981 Census			1990 Census	
	Total* Irrigation (ha)	Irrigation* from ground-water (ha)	Portion of irrigation from ground-water (%)	Total* Irrigation (ha)	Irrigation* from ground-water (ha)	Portion of irrigation from ground-water (%)	Irrigation* from ground-water (ha)	1990 Census results as percentage of 1981 census (%)
Highveld	34208	7256	21	26519	3987	15	2620	66
Karoo	133440	20475	15	70582	13519	19	24184**	178
Natal	18199	501	3	14398	383	3	170	44
Eastern Cape	34408	3592	10	23431	2191	9	2066	94
Gauteng	148684	51346	35	126927	39823	31	19785	50
Free State	127952	19511	15	108951	10494	9	8011	76
Winter Rainfall	27138	4142	15	20790	2278	11	1205	53
Total	524029	106823	20	391598	62675	16	58041**	-**

* Only in districts where information in the 1990 census were received.

** Possibly includes surface water irrigation

Modified after Braune and Coetzer (1992)

Table 7.2: **Districts with highest irrigation from groundwater***

District	No of farms using groundwater for irrigation	Area irrigated with groundwater (ha)	Average irrigated area per farm (ha)
HIGHVELD			
Klerksdorp	25	594	24
Potchefstroom	51	1896	37
District total	101	2630	26
KAROO			
Aberdeen	72	4605	64
Beaufort West	65	673	10
Calvinia	25	190	8
Carnarvon	26	326	13
Cradock	95	2523	27
Fraserburg	23	193	8
Graaff Reinet	127	8551	67
Hofmeyer	23	513	22
Laingsburg	13	426	32
Middelburg	15	802	53
Murraysburg	21	597	28
Pearston	43	2203	51
Prince Albert	24	200	8
Sutherland	38	724	19
Willowmore	56	595	11
District total	755	24184	32
EASTERN CAPE			
Komga	11	484	44
Queenstown	19	382	20
Sterkstroom	21	104	5
Uitenhage	9	774	86
District total	85	2066	24

Continued ....

Table 7.2 continued ....

GAUTENG & NORTHERN PROVINCE			
Brits	5	1575	315
Coligny	26	301	12
Delareyville	66	1434	22
Groblersdal	41	1450	35
Koster	10	1300	130
Lichtenburg	117	4352	37
Marico	13	1400	108
Pietersburg	20	1635	82
Potgietersrus	37	2102	57
Rustenburg	15	1900	126
Thabazimbi	8	1087	136
Ventersdorp	23	474	21
Warmbad	9	415	46
District total	408	19785	48
FREE STATE			
Bloemfontein	24	732	31
Boshoff	17	304	18
Brandtfort	7	345	49
Edenburg	19	235	12
Fauresmith	32	477	15
Herbert	14	202	14
Jacobsdal	21	217	10
Kuruman	8	208	26
Petrusburg	103	2081	20
Phillipolis	20	320	16
Postmasburg	18	252	14
Theunissen	10	477	48
Vryburg	14	585	42
Warrenton	19	212	11
District total	443	8011	18
WINTER RAINFALL			
Piketberg	20	760	38
Clanwilliam	1	280	280
District total	32	1205	38

* Districts with 20 or more farms and/or 200 ha or more irrigation with groundwater

Table 7.3: **Irrigation with groundwater as only source**

Agricultural district	Total no of farms using groundwater for irrigation	Percentage of farms with groundwater as only water resource (%)
Highveld	101	65
Karoo	755	57
Natal	27	11
Eastern Cape	85	62
Transvaal	408	83
Free State	443	86
Winter Rain	32	41

Modified after Braune and Coetzer (1992)

Braune and Coetzer found that the most commonly reported problem in irrigated areas is dropping water levels. Deterioration of groundwater quality was difficult to determine, but in general it was found to be a minor problem, although for 25 farms in the Natal district, 20 farms in the Eastern Cape district and 56 farms in the Karoo district, quality problems were reported.

Table 7.4: **Problems per district**

No. of farms using groundwater for irrigation purposes		Percentage of total no. of farms experiencing problems (%)				
		No problems (%)	Decreasing water levels (%)	Decreasing supplies (%)	Influence of adjacent farms (%)	Deterioration of water quality (%)
Highveld	101	15	62	27	9	1
Karoo	755	50	40	19	12	7
Natal	27	11	0	37	0	93
Eastern Cape	85	15	9	32	5	24
Gauteng	408	6	51	22	47	8
Free State	443	74	18	3	11	0
Winter Rain	32	69	28	3	0	0

Modified after Braune and Coetzer (1992)

7.3 SALINITY PROCESSES IN IRRIGATED SOILS AND UNDERLYING AQUIFER SYSTEMS

The primary processes that determine the salt content of irrigation water that percolates from the root zone and recharges underlying groundwater are as follows:

- physical movement of salt with or in soil water
- concentration of salt in solution by evaporation and consumptive use by plants of soil water
- chemical reactions with the soil/geological strata or in the soil solution.

It is important to recognise that the only commonality in the salinisation of different irrigation schemes is the concentration factor - typically 3-6 times the irrigation water concentration for single use, but in situations where irrigation return flow is sequentially re-used in a downstream direction along a river, the effective factor can be higher than 10 times. This may be evident in a progressive worsening of underlying groundwater salinity in a downstream direction.

The physical properties of the soil, such as texture, structure, layering and aggregation, as well as the hydraulic properties of the underlying formations, play a determining role in the salinisation process and are usually more important than the chemical properties of the medium. Each particular aquifer can therefore be expected to have unique aspects to its salinisation patterns that stem from the physical properties of its overburden and its strata.

7.3.1 The General Situation in South Africa

In South Africa irrigation farming has developed over a range of underlying geological strata, leading to a range of underlying aquifer types. These vary among four broad classes:

i. Unconsolidated material ranging from sands to boulders from reasonable (2 m+) to significant (10 m+) depth at very low gradients acting as unconfined or semi-confined primary aquifers and underlain by either secondary aquifers (fractured rock), or impermeable rock. The following variations exist:

   - riparian alluvium (e.g. Great Fish, Crocodile)
   - flood-plain alluvium (e.g. Gamtoos, Limpopo)

- ⇨ alluvial boulder terraces (e.g. Olifants, Breede)
- ⇨ boulder valley-fill (e.g. Hex)
- ⇨ aeolian (wind-blown) sands (e.g. Cape Flats)
- ⇨ sands formed by in-situ weathering (e.g. Vaalharts, Springbok Flats)
- ⇨ combinations of these (e.g. Lower Sundays, Loskop)

ii. Relatively deep (1m - 2m) soils of in-situ or colluvial origin at medium gradients on either secondary fractured rock aquifers, or relatively impermeable strata. This is often the case in the vegetable and pineapple irrigation districts of the Southern and Eastern Cape, sugar cane areas of KwaZulu-Natal, grain growing areas on the Highveld and certain sub-tropical fruit areas of the Lowveld.

iii. Relatively thin (< 0,5m) soils of in-situ or colluvial origin on medium to steep slopes on either secondary fractured rock aquifers, or impermeable strata. These conditions are common in the vine and deciduous fruit irrigation areas of the Western and Southern Cape and certain sub-tropical fruit areas of the Escarpment.

iv. Soils overlying dolomitic areas. Extensive irrigation farming, primarily grain and cotton, is sourced from dolomitic water on the central and western parts of the Highveld.

Clearly, in all cases the potential salinisation history will be a function of the physical properties of the different media the irrigation percolate encounters on its way to the groundwater table, or in the groundwater flow system after mixing with the resident groundwater.

The following few sections will now focus on aspects of salinity-controlling processes in irrigated soils, which are primarily also applicable to the underlying aquifer strata. The descriptions are generalised from many of the sources listed at the end of this report and recognition of each one is not regarded as necessary for the purposes of this overview. For readers who require detailed physicochemical descriptions of these processes, the following two references are suggested: Shainberg and Shalhevet (1984) and Tanji (1990).

7.3.2 Water and Salt Movement through Soil

Proper management of salt in soil-water systems depends on an understanding of the physico-chemical processes occurring during solute (salt) transport. It has long been known that water and salt do not move at the same rate through soil. The basic processes are as follows:

- Addition of quantities of water at the soil surface which vary in chemical composition, such as occurs with alternating cycles of irrigation or precipitation. Here the so-called leaching fraction is relevant, i.e. the volume of water supplied by irrigation and rainfall in excess of that required to satisfy consumptive use of crops and leaching efficiency.

- Convective solute transport due to the mass flow of water under influence of gravity. The vast majority of instances of soil water flow in the field occur under unsaturated conditions, in which the soil porosity is partially filled with both water and gas.

- Dispersive solute transport due to localised velocity variations in water flow caused by the tortuosity of soil pores.

- Diffusion of salt in response to a concentration gradient.

- Diffusion and convective solute transport that occur simultaneously - known as "miscible displacement".

- "Preferred path" flow, also known as macro-pore flow. This is flow that occurs in cracks in shrinking-swelling clays, or in large pores or inter-pedal voids in well-aggregated soils, or in channels formed by earthworms and decayed roots. A large fraction of the total soil water movement may occur in the macro-pores, while the soil solution in the normal smaller pores will not be as mobile, but can actually act as a distributed source or sink for solutes.

- Capillary flow. This form of water and solute movement is driven by adhesion between water and soil molecules and is supported by surface tension across small pores. Capillary flow may carry salts from the groundwater table into the root zone and is the cause of salt accumulation near the ground surface.

7.3.3 Solute-Soil Surface Interaction

Both cations and anions interact with the electrically charged soil surface during the flow process, as follows:

- Adsorption-Desorption: In salt-affected soils the soil particle surface is usually negatively charged, resulting in retardation of the movement of positively charged ions. The process of a single cation interacting with the soil surface is known as **adsorption-desorption.**

- Anion exclusion: Simultaneously with the above, anions are effectively excluded from a small volume of the pore near the negatively charged surface, with displacement thereby accelerated due to higher concentrations of anions in the relatively more mobile central volumes of the pores. This is known as **anion exclusion.**

- Cation exchange: A third type of interaction involves the simultaneous multiple interaction of several cations with the soil surface, known as **cation exchange**. This is probably the most common process and represents the situation where at least three (Ca,Mg,Na) or four (Ca,Mg,Na,K) cations compete at once for adsorption in the region of a negatively charged mineral surface.

7.3.4 Chemical Reactions in Solution

Prime determinands of the aquifer salinisation process are the ionic composition and concentration of the irrigation water and of the percolate from the root zone, and how these fluctuate. The presence of ions (such as Ca, HCO_3, SO_4), with a tendency to precipitate on the soil complex during concentration by evapotranspiration, is particularly of interest when chemical reactions in solution are considered. When the solubility limits of the particular constituent is exceeded, **precipitation** may temporarily or near-permanently remove part of the salt load from the percolate. The counterpart of precipitation is known as **dissolution** and represents the reverse process during "leaching" with fresher water. Cation exchange and these two processes make up the three most important chemical interactions in the soil in terms of salinisation.

7.3.5 Salt and Sodium Effects on Salt Movement

The hydraulic properties of most soils composed of any amount of layered silicate clays are dependent upon both the total salt present and the proportion of that salinity constituted by sodium (exchangeable sodium percentage or ESP). Changes in hydraulic properties such as reduced effective permeability have been suggested to be the result of swelling of clays (high ESP), or of the blocking of soil pores as a result of clay dispersion (low ESP). Such time-dependent variations in hydraulic properties will obviously result in time-variable impacts on the underlying aquifer.

7.3.6 Aquifer Processes

As the irrigation percolate migrates through the vadose zone, it encounters three potential states of chemical interaction:

i. Unconsolidated soils below the root zone where similar interactions to those described above take place.

ii. The contact between soil and rock where a zone of decomposition and weathering usually exists. This zone may be a natural source of solutes because of the many potentially active surfaces on the weathered fragments. Especially over marine-sedimentary formations, such as shales, is this zone a strong salt reservoir. In shallower soils this zone is often "deepened" by rip-ploughing, as farmers attempt to create greater moisture-holding capacity. If such soils are irrigated, increased leaching of salts into the underlying groundwater can be expected. This effect should recede over many years as a new natural equilibrium sets in, but repeated ripping may maintain the impact.

iii. Bedrock formations with possibly low primary (matrix) porosity, but higher secondary porosity through cracks and fissures, as well as fault lines. Chemical interactions between the solutes in the percolate that enters the groundwater body and the mineral complex through which it now moves, are similar to those described above, but differences arise because of hydraulic factors. These factors determine the migration rate of groundwater, which in turn, determines the contact time of the solutes with the rock strata. In fractured rock aquifers and along fault lines, groundwater flow can be expected to be significant only along fractures with good cross-connectivity with other fractures. The mineral complexes along the rock surfaces of such preferred path fractures are most prominent in chemical interactions and over many years such chemical changes of the migrating irrigation percolate should recede as the original natural equilibrium is gradually replaced by a new one. Blind or poorly connected fractures act as sinks or sources of percolate solutes and over long periods of time may yield increased concentrations.

Other aquifer salinisation-related processes are as follows:

- Perched water tables caused by layers of low permeability (e.g. calcrete) in unconfined aquifers may also cause spatially variable salinisation impacts on deeper groundwater. Such water tables are usually quite dynamic, often reach the root zone of the irrigated crops, and can also relatively easily be drained artificially.

- Artesian groundwater flow towards main-stem river channels (as the lowest points in the landscape), along which riparian alluvial groundwater tables have developed through irrigation, may cause upwelling in the riparian zone. Even normal gradient-controlled rainwater-derived groundwater flow from "upland" rock formations will have to move through the riparian alluvial zone to the river channel. These "regional" groundwater intrusions will obviously affect the salinisation processes in the riparian aquifers.

- A further aquifer-related process that may play a role in longitudinal increases in the salinity of effluent rivers is the mobilisation of ancient (and therefore more saline) groundwater by the increased hydraulic head caused by percolates from new irrigation. Such effects should recede after many years as fresher percolate gradually replaces the older groundwater.

7.3.7 A Footnote on Nutrients

As the potential impact of irrigation on groundwater quality not only relates to increases in total salinity, but also to individual ionic constituent concentrations, it is necessary to accentuate the importance of the particular constituents of phosphorous and nitrogen in groundwater quality. Though the concentrations of these two constituents are generally more than one order of magnitude smaller than those of conservative (inert) ions (such as Na or Cl, etc.), their impacts on the fitness-for-use of groundwater or seepage from groundwater are significant. The foregoing impacts usually relate to compounds of phosphate and nitrate, the so-called nutrients, because of their nutritional value to algae and diatoms in receiving surface waters. These nutrients are the prime cause of "green" water, a state known as eutrophication. The prime source of nutrients is fertilizer that moves with the percolate from the root zone, but in some areas of intensive informal settlements, the nutrients may derive from poor sanitation. Nitrates in groundwater are particularly harmful to infants and pregnant mothers.

7.4 THE VAALHARTS IRRIGATION SCHEME

7.4.1 Introduction

The Vaalharts Catchment is situated between 27° 35' and 28° 05' latitude and 24° 30' and 24° 55' longitude. It lies in the region between the Harts and Vaal Rivers. The climate in the area is semi-arid with the average rainfall of 450 mm per year of which 80% falls within the summer months.

7.4.2 Irrigation and agriculture

The Vaalharts irrigation scheme is the largest irrigation scheme in South Africa. The main crops irrigated in the area include cotton, groundnuts, wheat, lucerne, maize, sunflower, citrus, peas and other vegetables. Vineyards are also irrigated.

The history of the Vaalharts Scheme goes back to 1881-1882 when the area was surveyed for irrigation purposes. Between 1932 and 1934, 36 000 ha of the total 74 000 ha was classified as irrigable (van Garderen, Louw and Rosenstrauch, 1934). In 1933, the Vaalharts Government Water Scheme was put into effect and the first plots allocated in 1938. A number of canal developments were undertaken with the North Canal development completed in 1945. The Western Canal area was developed in the 1950s and the first plots were allocated in 1957/1958 and the last few in 1965/1966. The Barkly West Canal area was also developed in the 1950s.

The Vaalharts Government Water Scheme is divided into the West Canal area comprising about 5000 ha and the North Canal area consisting of about 24 000 ha of irrigated land. The Taung irrigation scheme in Bophuthatswana has an additional 3570 ha. In the North Canal and the West Canal areas, about 97% of the land available for irrigation is under flood irrigation while about 2.5% is under overhead spray irrigation and the remaining 0.5% under drip irrigation. In the Taung area, only about 10 ha of flood irrigation remains from 3570 ha, the rest having been upgraded to centre pivot irrigation (Herold and Bailey, 1996).

Irrigation in the area resulted in waterlogging problems. To overcome the latter, a comprehensive system of about 240 subsurface drainage systems were installed between 1976 and 1979.

7.4.3 Geology and hydrogeology

The geological profile of the Harts River Valley (Figure 7.1) shows four distinct horizontal profiles (Temperley, 1967):

- A top layer of sandy soil between 0 and 5 m thick
- Beneath the soil there is a calcrete layer with a maximum thickness of 15 m.
- Underlying the calcrete is a layer of Dwyka shale and tillite.
- The bedrock consists of Ventersdorp lava of the Ventersdorp system.

The permeability of the formations is very limited and variable both in the horizontal and vertical direction. For example, the Kalahari sand becomes locally loamy or clayey, the calcrete varies from impermeable rock to permeable soft or spongy material, and the old alluvial gravels are only permeable where they have escaped cementation by calcrete while the weathered bedrock is only permeable where it does not consist of clay. The 15 m thick "permeable" zone is therefore highly obstructed with impermeable material, the distribution of which is highly heterogeneous. The depth to the calcrete layer varies from only a few centimetres to about 12 m in places. On average it is found up to a depth of 5 m.

Before the start of intensive irrigation, the groundwater table was at about 24 m depth (Streutker, 1977). Over the years, flood irrigation has raised the water table to about 1 m below the surface. In areas where the calcrete is impermeable, a local perched water table may also occur.

7.4.4 Water quality data and salt balance

A salt balance carried out by Streutker (1977) of the North Canal area gave the following mean annual results for the period 1959 to 1966:

Salt in irrigation water = 40924t (V =142.1, C=288)
Salt in fertilizers = 8 184 t
Salt in drainage water = 7 322 t (V = 10.4, C=704)
Salt removed by plants = 6 480 t

Where V = water volume (million cubic metres) and C = TDS concentration (mg/ℓ). The above figures imply an annual TDS accumulation of 35 306 tons.

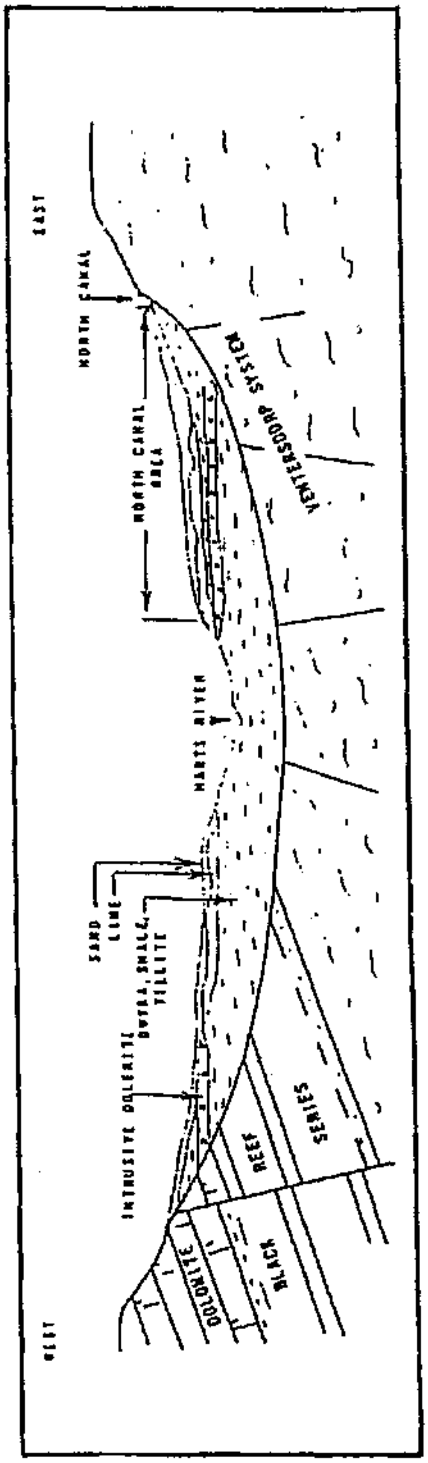

Figure 7.1: A geological profile of the Harts River Valley (after Temperley, 1967)

In 1985, Ninham Shand Inc. used a computer simulation model, IRRSCHEME, to perform a monthly crop water balance on Vaalharts. This provided an annual estimate for deep percolation of 17.10^6 m^3 for the period 1975 to 1982 (Jonker, 1996). As part of the same project, the ACRU model was combined with the Oster and Rhoades model (1975), to estimate the TDS load in the deep percolation and found the mean monthly deep percolate load for the period 1975 to 1980 equal to 2 408 tons (28 896 tons per annum), ie similar to Streutker's estimation (Jonker, 1996).

Stewart Sviridov & Oliver (1987) estimated that in the period 1935 and 1984 the TDS load returned to the Harts River was less than 20% of the TDS load contained in the irrigation water supplied to Vaalharts, ignoring the net effect of the salt input via fertilizer application and that removed by plants. This compares closely with Streutker's estimates for the period 1959 to 1966 above, which gives an equivalent salt return flow of 18%.

Herold and Bailey (1996) estimated the amounts of water percolating to deep groundwater storage. They compared their estimates with those made by other researchers using four alternative methods of calculation. The alternative estimates of mean annual deep percolation ranged between $33x10^6$ m^3 and $63x10^6$ m^3. These values provide an indication of range of possible deep percolation rates.

Herold and Bailey (1996) concluded from their model calibration results that about 65% of the TDS load contained in the irrigation water supplied to the Vaalharts irrigation scheme appears to have been retained in the groundwater store. Their model had a monthly time-step and treated the whole North Canal area as one unit.

Using the DISA model, Jonker (1996) estimated that 75% of the TDS load in the irrigation water supplied to Vaalharts between 1978 and 1983, and 83% between 1989 and 1991 is retained in the groundwater zone. The DISA model has a daily time-step and Jonker treated the North Canal area as a distributed system consisting of 36 modelling units divided according to soil type and soil depth.

7.4.5 Hypothesis of the processes

A hypothesis of the various processes taking place in the Vaalharts irrigation scheme has been given by Herold and Bailey (1996). A summary is given here. Figure 7.2 is a schematic representation of the various flows for the irrigation scheme. Herold and Bailey gave a summary of the sketch as follows: a dam supplies water to a canal which conveys water to the irrigation area. Secondary and tertiary canals distribute this water to the various plots and farm dams. A certain volume of this supply water is lost prior to reaching the irrigation area due to evaporation and seepage from the canals. The farm dams will also be subject to evaporation and seepage. Water is also received from rainfall. The total water supply will be divided into evaporation, surface runoff, infiltration into the soil to be used by the crop (evapotranspiration), lateral seepage towards the veld transport zone and deep percolation into groundwater.

The drains, which were installed to maintain the water level below the crop root zone, intercept a certain amount of groundwater. This water is collected by drainage canals and transported towards the Harts River. Some of these drainage canals also collect upstream catchment runoff which is conveyed over the main supply canal by super passages. The drainage water, together with surface seepage, flows through a veld area towards the river. Some of this water is lost to evaporation or re-used to irrigate pasture land within the veld area.

Drainage below the root zone of the soil that is not intercepted by the sub-surface drains will enter the groundwater (which may comprise both an upper perched water table and a deeper water table). From the perched water table, part of the water and its solutes can gradually flow laterally towards either the nearest river or the nearest main drain. The remainder could percolate through permeable portions of the strata immediately below the perched water table, resulting in gradual filling of the deeper water table. A substantial amount of the TDS load contained in irrigation water would be trapped in the deeper groundwater zone. Once the lower water table is filled there would be a rapid increase in the rate at which water discharges towards the river via direct seepage and the sub-surface drains. In the absence of adequate drainage, the cessation (or severe reduction) of the loss of surplus water to the deeper groundwater would result in waterlogging.

If the flow of the lower groundwater towards the Harts River is unimpeded, then over a long period of time the accumulated TDS load could begin to return to the streamflow. However, in the case of the Vaalharts irrigation scheme, if any such pathways exist, the permeability of the formations is likely to be so low as to restrict the flow rate to such an extent that the contribution to the TDS loading of the Harts River will be small compared with the direct drainage from the upper zone.

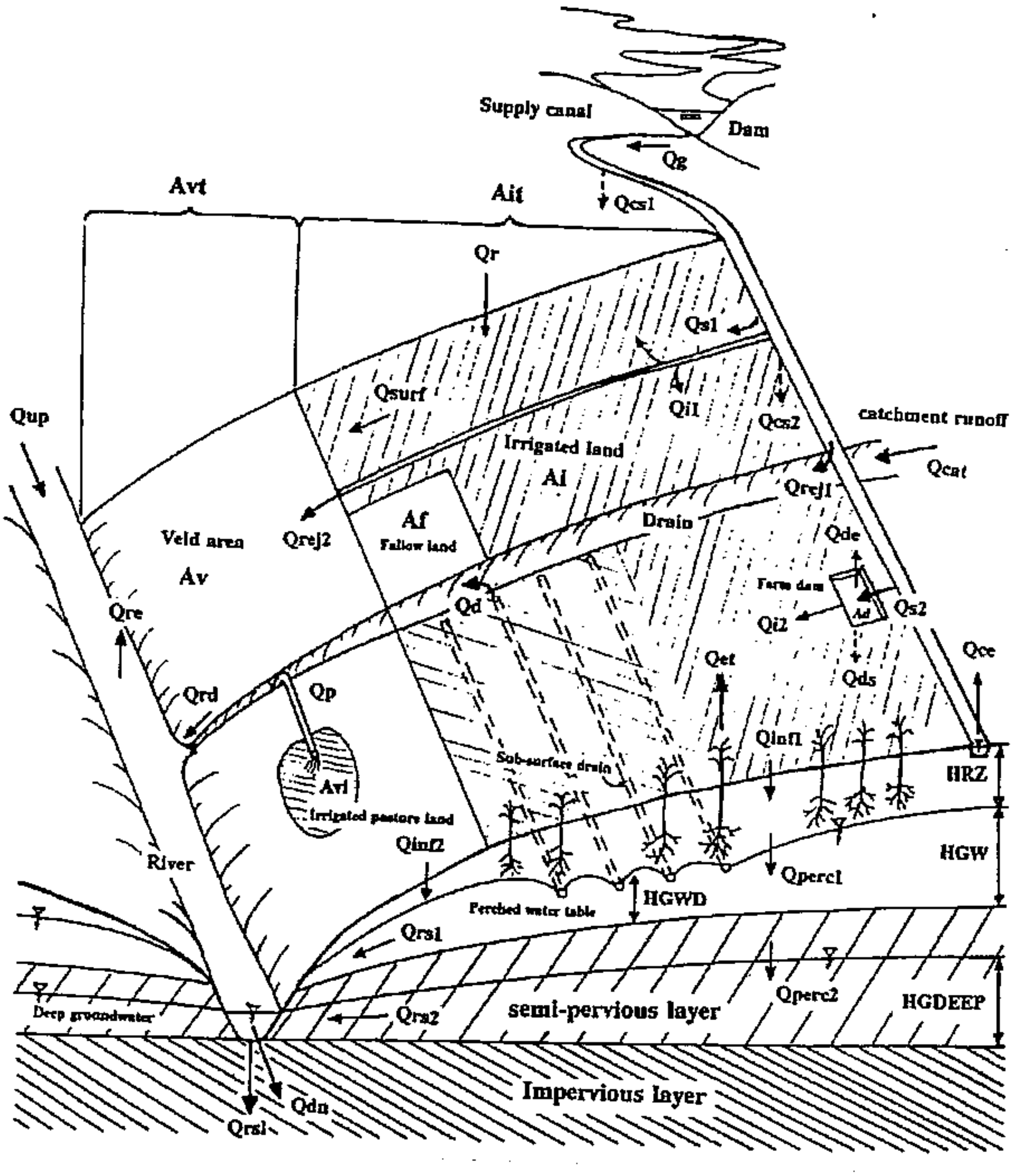

Figure 7.2: Hypothesised flow processes in the Vaalharts irrigation area

Description of terms

Ait = Total irrigation area (= Ai + Af + Ad)
Ai = Irrigated area
Af = Fallow land area (temporary fallow area)
Ad = Full surface area of farm dams
Avt = Total downstream veld area (= Avi + Av)
Av = Veld area
Avi = Pasture land area in transport zone irrigated from reject water
Qp = Water pumped from drains for re-use on lands in transport zone
Qd = Sub-surface flow to drains
Qg = Gross canal supply
Qr = Rainfall
Qs1 = Canal supply to field edge
Qs2 = Canal supply to farm dam
Qce = Canal evaporation
Qcs1 = Canal seepage loss remote from irrigation area
Qcs2 = Canal seepage loss in irrigated area
Qi1 = Direct irrigation from supply canal
Qi2 = Irrigation supply from farm dams
Qcat = Storm water rainoff from upstream catchment (passed through main drains)
Qsurf = Surface water runoff
Qup = Upstream flow in river
Qdn = Downstream flow in river
Qde = Evaporation from farm dams
Qrej1 = Reject water direct to drains
Qrej2 = Reject water to veld/pasture land
Qrd = Return flow to river via drains
Qrs1 = Return flow to river via seepage from perched ground water table
Qrs2 = Return flow to river via seepage from deep ground water
Qperc1 = Percolation to perched groundwater
Qperc2 = Percolation to deep groundwater
Qinfl1 = Infiltration to irrigated area
Qinfl2 = Infiltration in downstream transport zone
Qet = Evapotranspiration loss
Qre = Evaporation from river
Qrsl = Seepage loss from river
HRZ = Root zone storage capacity
HGW = Groundwater storage above perched water table
HGWD = Groundwater storage in drained areas
GHDEEP = Deep groundwater storage

Another process that could play a role is the lateral movement of water (and its associated TDS load) beyond the boundaries of the irrigated area, thereby effectively increasing the aerial extent of the groundwater reservoir.

7.4.6 Future implications

The accumulation of about two-thirds or more of the applied TDS load at Vaalharts implies that the salinity of the groundwater and that of the irrigation return flows is still a long way from a state of equilibrium with the supply water salinity. Eventual filling of the available groundwater storage is expected to result in a state of equilibrium, with the mean annual TDS load returned to the Harts River matching that contained in the irrigation water. The system response to these changes will be complicated further by the effect of the installation during the 1970's of irrigation sub-surface drains, which can be expected to intercept part of the water and salt load draining past the bottom of the crop root zone. This should serve to slow the rate at which salinity levels build up in the groundwater zone. However, it will also ensure that a larger proportion of the salt associated with the applied irrigation water will be returned directly to the River. This implies a more rapid rise in return flow salt loads after the groundwater zone fills.

7.5 THE BREEDE RIVER VALLEY

7.5.1 Introduction

The Breede River and its tributaries comprise a major river system in the Western Cape Province. The river originates in the Ceres Valley, approximately 100 km NW of Cape Town and flows in a south-easterly direction to the Indian Ocean (Figure 7.3).

The catchment is situated in the central part of the Cape Fold Belt and covers an area of 12 596 km^2. Generally, the valley flats and fans formed by the Breede River and its tributaries are surrounded by rugged Table Mountain Sandstone.

The catchment can conveniently be subdivided into three subcatchments, viz upper, middle and lower. The upper catchment stretches down to Brandvlei Dam, while the middle catchment stretches from the Brandvlei Dam to the Secunda weir (H5 M004). The lower catchment stretches from the Secunda weir to the river mouth.

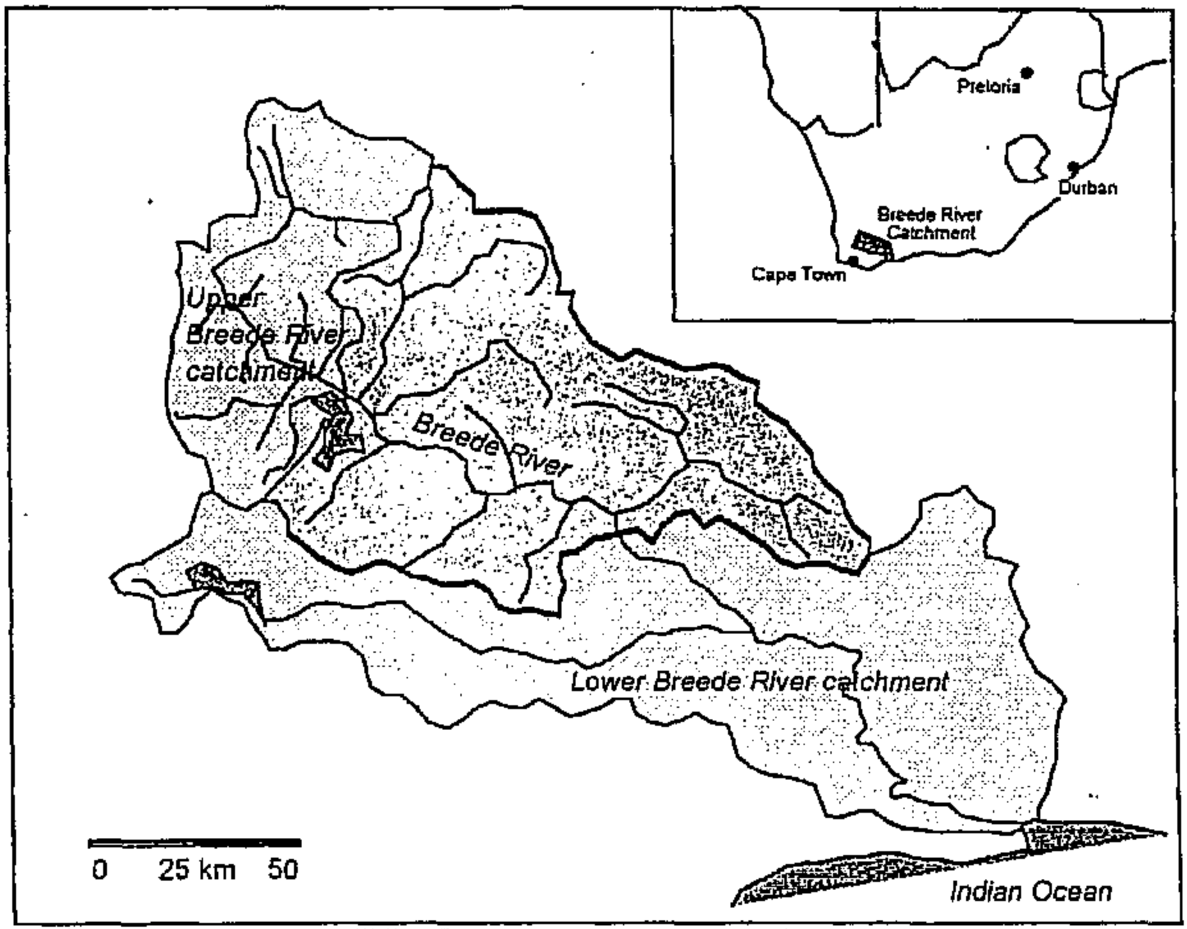

Figure 7.3 : Locality map of the upper, middle and lower Breede River catchments

The Breede River Catchment has a typical Mediterranean climate with a dry, hot summer and moderately warm but wet winters. Typical mean annual values of rainfall that have been recorded in the area vary from less than 300 mm in the valley bottom to greater than 1000 mm in the mountains. In rainfall terms, the valley areas fall under semi-arid conditions, with potential evapotranspiration always higher than the rainfall.

7.5.2 Irrigation and agriculture

The Breede River Catchment is the fourth largest irrigation area in South Africa, with approximately 100 000 ha under irrigation. Viticulture is the main agricultural activity with the growing of fruits and vegetables taking second and third place respectively. Irrigation in the valley began in the 18th century and is mainly needed during the dry season between October and April. The biggest portion of the irrigated lands, comprising some 45 000 ha, is situated in the middle part of the Breede Valley between Worcester and Bonnievale. The irrigation history in the middle Breede River Valley can be divided into four overlapping phases (Kirchner, 1995):

(i) pre 1889: small scale development of the tributary areas
(ii) 1898-1918: development of the main river flood plain
(iii) 1918-1970: intensification of the tributary irrigation
(iv) since 1970: development of the Greater Brandvlei Dam

Farms in the upper reaches of the tributaries receive their irrigation water from farm dams. Supplies from the dams are often augmented by water from fountains and boreholes. Irrigation water can also be obtained from canals. In addition, there are pumps along the river that withdraw water for pumping schemes and riparian owners.

The biggest dam in the catchment, the Brandvlei Dam, together with the Breede River, supplies all the irrigation water for the canals and pumps. The Greater Brandvlei Dam, which has a full supply capacity of 474,7 x 10^6 m^3, is an off-channel storage dam which receives most of its inflow from its own catchment and feeder canals from tributaries of the Breede River. Water can also be pumped from the Breede River into the Dam. The Breede River is not only the main supplier for irrigation water, but also receives irrigation return flow. Over the years, this has resulted in increase in salinity in the river during times of low flow. Water from the Greater Brandvlei Dam is not only released for irrigation but also for dilution if the electrical conductivity exceeds +70 mS/m at the Zanddrift Canal Off-take.

7.5.3 Geology

The geological description given here is a summary mainly from Kienzle (1990). A generalized cross section of the geology is shown in Figure 7.4. The mountain ranges on either side of the valley consist of formations of the Table Mountain Group. The dip of the formations is predominantly in a northerly direction. Formations of the younger Bokkeveld and Witteberg Groups are found south of the river. Rocks of the Karoo Sequence (Dwyka and Ecca formations) occur in the north western part of the area and in a few patches south of the Worcester Fault. North of the fault, the Malmesbury Group is found underlying the Table Mountain Group. The Enon formation occurs east of Robertson, between the Worcester Fault and the Breede River. Alluvium is found in the valley of the Breede River and alongside its tributaries.

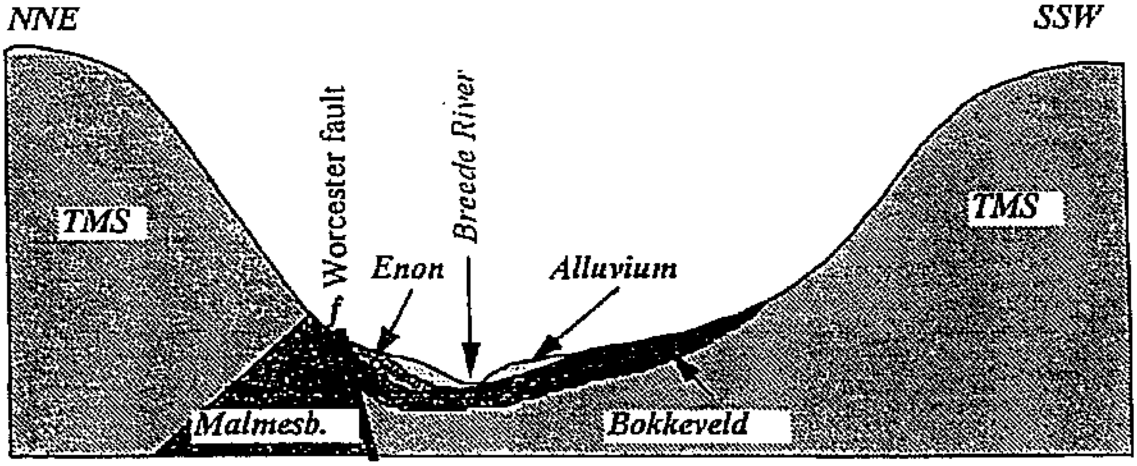

Figure 7.4 : **Generalised cross-section through the Breede River Valley viewing downstream and showing the northerly dip of Table Mountain and Bokkeveld Groups (not to scale).**

7.5.4 Hydrogeology and hydrochemistry

Apart from the alluvium, all aquifers in the area are secondary aquifers. The main recharge areas are the mountainous areas on either side of the Breede River Valley where rainfall is high. Groundwater flows from these mountains towards the tributaries and eventually into the Breede River.

Table Mountain Sandstone (TMS) which forms the mountain ranges is probably the most important aquifer in the area. The aquifer has comparatively high transmissivities. Fountains may be found in areas where the aquifer is underlain by less permeable formations. In certain areas, the aquifer is confined and water may flow into overlying aquifers along joints, fractures and faults. The TMS water has a low salt content. Over large areas, the TMS is overlain by sediments of the Bokkeveld Group which consists of four sandstone and shale formations. The shale contains considerable amounts of salt. The yield of boreholes in these formations depend on the degree of fracturing. It appears that water circulating in more fractured zones is less mineralized (Kirchner, 1995). Water in the Enon formation is usually of very poor quality. Water quality in the alluvial sediments varies. Bertram (1989) found conductivities between 10 and 87 mS/m in production boreholes used for irrigation. Flugel (1989a) measured EC values ranging between 100 and 600 mS/m in test boreholes, probably because most of them were near irrigated fields. The shallow groundwater in the alluvium is influenced by irrigation return flow and/or by evaporation. Aquifers in the Malmesbury formation are also mineralized, although astonishingly low salinities (+5 mS/m) have been recorded near the Worcester fault.

In 1989, the Directorate of Geohydrology of DWAF conducted a hydrocencus between the Greater Brandvlei Dam and the intake of Zanddrift Canal. Information was collected on the occurrence, quality, availability and use of groundwater. It was found that a total of 10,8 million cubic metres is abstracted anually from this area (Bertram, 1987). The average electrical conductivity of the groundwater used for irrigation is 70 mS/m.

Jolly (1990) summarized the quality information from Wittingham (1976), Greeff (1978) and Bertram (1989) together with his own data and presented TDS values for different aquifers (Table 7.5).

Table 7.5: TDS values for different formations (after Jolly, 1990)

Formation	TDS (mg/ℓ)
Alluvium	423
Ecca Group	892
Dwyka Group	1475
Witteberg Group	1753
Bokkeveld Group	586
Table Mountain Group	92
Malmesbury Group	688

With the exception of the TMS, the variability of water quality in different aquifers is high. The major factors which seem to influence the quality of water within aquifers include (Kirchner, 1995):

- the borehole position within the aquifer and relative to recharge areas
- the degree to which it is fed by major fractures
- whether saline irrigation return flow occurs

7.5.5 Irrigation and the salinity of groundwater and return flows

A number of studies have been conducted to determine whether part of the salt load in the Breede River is derived from irrigation return flow. Irrigation return flow could contribute to river salinisation by

- salt in irrigation water concentrated as a result of evapotranspiration.
- the mobilization of the salt stored in the unsaturated zone
- the mobilization of the salts which become available for dissolution during land preparation
- the displacement of saline groundwater

Howard (1986) recorded an increase in salinity in the Breede River. This deterioration was mainly ascribed to inflow of saline irrigation return flow and groundwater seepage into the tributaries and the main river.

Kienzle and Flugel (1988) found that the salinity of three monitored tributaries in the Middle Breede River increased significantly probably as a result of an extension of irrigated areas and changes in irrigation techniques. Fluctuations in stream salinity were found to be closely related to the irrigation season.

Several studies have been conducted in an effort to quantify the salt and water budget for a representative portion of the irrigation scheme at irrigation land and experimental plot level. These studies have shown that a large percentage of the water diverted to irrigation schemes is not used by plants and ends up in the groundwater store (Kirchner *et al.*, 1997).

Moolman and De Clercq (1992) used estimated evapotranspiration data and measured irrigation quantities on individual farm plots to calculate the water balance of a drip irrigated vineyard and inferred that on average, 10% of the applied water would percolate through the root zone. In contrast, using quantitative measurements of soil water content before and immediately after an irrigation event in the same vineyard, it was found that the actual leaching fraction can exceed 40%. The difference between the two estimates is probably related to deep percolation losses due to macropore and preferential flow (Kirchner *et al.*, 1997). Similar observations were made in another vineyard on a stony sandy loam soil irrigated with micro-sprinklers (Kirchner *et al.*, 1997).

Murray, Biesenbach & Badenhorst (1993) calculated the theoretical drainage depths for 36 different irrigated fields in the Breede River Valley using measured irrigation quantities and calculated crop water requirements. Over-irrigation resulting in large potential drainage losses was predicted. In a majority of cases, deep percolation losses reached maximum levels at the beginning of the irrigation season (July to October) for the vineyards. From November to March, very little of the applied water was calculated to percolate to below the root zone. In certain cases, deficit irrigation, as opposed to over-irrigation, was observed during the period of high evaporative demand.

A similar observation was made by Moolman *et al.* (1983) who used a conceptual computer model to simulate the movement of water and solutes in irrigated vineyard soils along a tributary of the Breede River. In their study, it was found that large amounts of water applied as a prebud-burst irrigation in late August and early September, followed by more or less fixed amounts of water being applied at set frequencies during the rest of the irrigation season, resulted in deep percolation losses and accompanying salt loads which are much larger during the last six months of the year than during the first six months. The results further indicated that from January to June, especially during dry winters, capillary rise exceeded deep percolation. The computer predictions were in accordance with the observed TDS content of the receiving river, which for three consecutive years from 1978 to 1980, were substantially higher during the period July to December than from January to June.

Greeff (1990) gave an example of how the establishment of a newly developed vineyard of 25 ha on soil derived from shale increased the salinity of a tertiary tributary of the Breede River . Prior to planting, the soil was deep ploughed (ripped) to a depth of ca. 1,2m. After the first winter rains, base flow salinity levels in the river upstream of the vineyard ranged from 336 mg/ℓ to 475 mg/ℓ. Salinities in the river next to and downstream from the vineyard ranged from 4470 to 7655 mg/ℓ. Greeff (1990) argued that the increase in salinity was caused by disturbance of the partly decomposed shale in the subsoil during soil preparation. Deep ploughing increased the porosity which enhanced percolation of rain and irrigation to much greater depths than before with a concomitant increase in mobilised salt. Note that Greeff (1990) found that in leaching experiments, a maximum of 5658 mg/ℓ of salt was leached from one kilogram of source rock in one litre of water.

Murray *et al.* (1989) reported on a pilot study which was aimed at monitoring the volumes of water and salt loads moving through an irrigation scheme. For this purpose the irrigation canal inflow, volumes extracted from the canal and other sources as well as farm reservoir storage and electrical conductivity of the water were monitored during an irrigation season. The study was conducted in an area of 2 081 ha of the middle Breede River catchment of which 1 241 ha was irrigated. They found that the total volume of water irrigated agreed with the theoretical irrigation requirement and that only about 11% of applied water percolates to below the root zone. Although the irrigation canal was lined, approximately 14 % of water transported in it was lost as seepage. Their calculations showed that more salt was being leached than applied to irrigated lands.

Lautner (1989) investigated the salt- and water balance of 17 drains in the area during a single season. She determined the average irrigation requirement as 34 m^3/ha/d and regarded a leaching fraction of between 10 and 15 % as representative.

From a run of the daily time resolution DISA model spanning three years, Beuster *et al.* (1990) calculated irrigation return flow as between 57 330 000 m^3/season without, or 74 991 000 m^3/season including, canal losses. This is equivalent to a mean return flow of 16,5 or 21,6 % respectively.

Based on salt-balance calculations and assumed irrigation efficiency, Flugel (1989b) concluded that all tributaries show a dominant influence of groundwater seepage from the deep aquifer. He found that in boreholes monitored by the HRI, the groundwater level was always deeper than 15m and the EC higher than 1500 mS/m, while under irrigated land, the water level was less than 3m deep and the conductivity varied generally between 250 and 600 mS/m.

7.6 THE GREAT FISH AND SUNDAYS RIVER BASIN

7.6.1 Introduction

The Great Fish River and Sundays River basins are situated in the Eastern Cape province of South Africa (Figure 7.5). The Great Fish River flows some 630 km before joining the sea north-east of Port Alfred. The Sundays River joins the sea north-east of Port Elizabeth.

The Great Fish River and Sundays River basins have a typical arid to semi-arid climate with hot summers and cold winters. Most of the precipitation (350-450mm/a) occurs between February and March when evapotranspiration is at its highest. The annual evapotranspiration exceeds the annual precipitation with the result that very little water is available for recharge.

7.6.2 Irrigation and agriculture

Irrigation in the valley began just after the turn of the century when stormwater runoff was diverted onto lands for irrigation purposes to supplement livestock farming, which consisted mainly of sheep. Ostriches were also introduced on a fairly large scale round about the turn of the century. The main crop irrigated was lucerne. Vineyards were also introduced along the Sundays River. As intensive farming practices became common, the need to conserve water for irrigation became essential.

Increased demand for water in the early 1920's led the Department of Irrigation to construct the

Grassridge and Lake Arthur dams in the Great Fish River catchment and the Van Ryneveld's Pass Dam and Lake Mentz on the Sundays River. In both these catchments, serious siltation was experienced. During the early 1950's, the Kommandodrift Dam was constructed upstream of Lake Arthur. The wall of Lake Arthur dam was also raised on two occasions. Rapid siltation was also experienced in Lake Mentz and, to counter declining storage capacity, the wall had to be raised in 1934 and again in 1951. Lake Mentz has now been renamed Darlington Dam.

The siltation problem and the erosion of the veld led to the formulation and implementation of various soil conservation measures. The latter were quite effective in arresting soil erosion but unfortunately may have led to a decrease in catchment runoff. This reduced flow of water in the Great Fish River towards the late 1950s placed many of the irrigators in financial difficulties. In an effort to reduce the demand on the dwindling water supplies, the government purchased land from some of the farmers. This resulted in 6000 ha being withdrawn from irrigation. This land was later rescheduled for irrigation after the Orange Fish Tunnel became operative during the mid-1970s. This tunnel, as well as diversion works, tunnels and canals in the Middle-Fish region, allowed irrigation farming in the Fish and lower Sundays valleys to be stabilized and extended by transfer of about $500.10^6 m^3$ of water per year from Gariep Dam on the Orange River.

Geology

The Great Fish and Sundays River basins are underlain by sedimentary succession of the Karoo Sequence. The sedimentary rocks represented include tillite of the Dwyka Group, alternating cycles of deltaic mudstones and sandstone of the Koonap Formation and alternating cycles of fluvial mudstone and sandstone of the Beaufort Group. A complex network of dolerite dykes and sills has intruded into the sedimentary strata.

7.6.3 Hydrogeology

Groundwater occurs under unconfined conditions in the unconsolidated alluvial sediments. The porosity and permeability of the consolidated sedimentary rocks in the Karoo Basin are extremely low due to compaction. Groundwater in these rocks occurs mainly under confined conditions. The water occurs in joints within the sedimentary rocks and also in joints and fracture zones caused by the intrusion of dolerite. The latter is responsible for the formation of groundwater compartments in the basin.

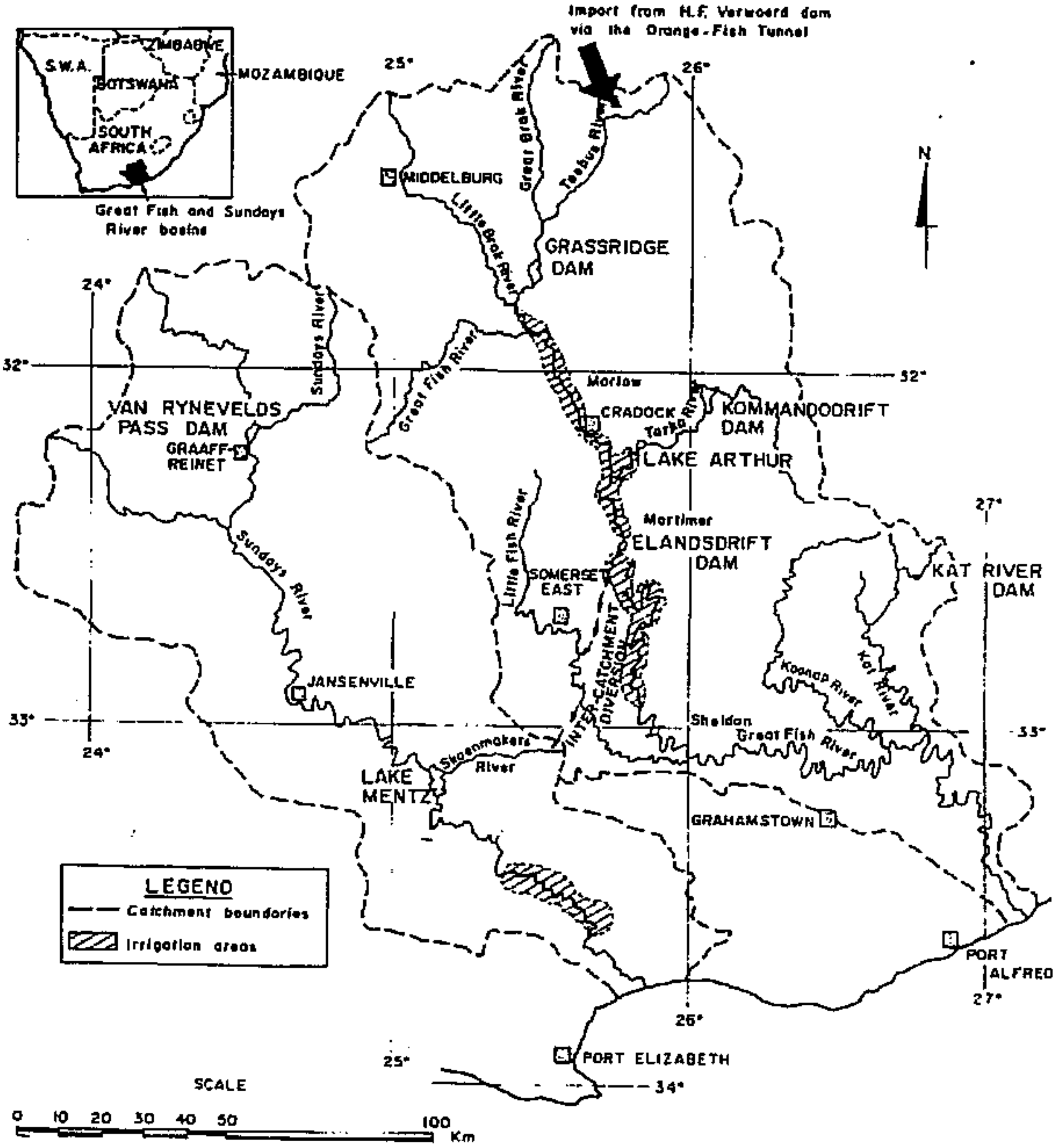

Figure 7.5: Great Fish and Sundays River basins showing features of the Orange River project

Low mean annual rainfall (350-400 mm) and high evaporation losses mean that only a small percentage of the rainfall recharges groundwater storage. Recharge also takes place through irrigation and leakage from canals and farm reservoirs. Reynders (1984) estimated that 34 % of the volume of irrigation water applied to the lands in the Middleton area reaches the groundwater table. A correlation between irrigation and groundwater levels has been observed, both in the unconfined alluvial aquifer and the semi-confined /confined fractured rock aquifer. During the irrigation season from late August to April, water levels show a substantial rise, while a corresponding decline is observed from May to July when no water is lead onto the lands (Reynders, 1984).

7.6.4 Hydrochemistry

The groundwater chemistry in the Great Fish-Sundays River basins is greatly influenced by the different geological formations of the Beaufort Group and underlying Ecca Group. Groundwater associated with shale (mudrock) generally possesses the highest concentrations of salts. Tordiffe (1978) pointed out that topography and climate may also influence the chemistry. Low salt concentrations are generally encountered in the groundwater of the higher lying interior plateau where relatively free circulation and leaching occurs while higher salt concentrations are encountered in the lower lying areas where stagnant conditions prevail (Tordiffe and Botha, 1981).

The water chemistry in non-irrigated areas is characteristic of a stagnant environment. Only during saturating rains, which occur infrequently, are salts in the soil dissolved and transported in bulk down to the groundwater.

7.6.5 Irrigation and groundwater salinity

The chemistry in irrigated lands is representative of a more dynamic environment. The average salinity of water in aquifers underlying irrigated lands is considerably higher than below non-irrigated lands. Reynders (1984) reported that the average TDS content of aquifers underlying irrigated lands was 3385 mg/ℓ as opposed to 1959 mg/ℓ for aquifers below non-irrigated veld. Reynders, Moolman and Stone (1985) estimated that 97% of the deep percolate that recharges the aquifer in the Middleton area, comes from irrigated lands. Their calculations showed that the annual deep percolate from irrigation lands represented a leaching fraction of 0.27 and 0.23 for 1982 and 1983, respectively. This over-irrigation was reflected in the seasonal recharge pattern observed in borehole water level fluctuations measured in 1983. Based on the results of their hydrogeological studies, Reynders, Moolman and Stone (1985) concluded that the residence time of the irrigation deep percolate within the aquifers was short.

For the Middleton area, Reynders (1984) estimated that dissolved salts increased more than five fold (from 820,5 mg/ℓ to 4550 mg/ℓ) between zones of recharge and discharge. This increase in salinity was attributed to evapotranspiration and mobilization of salts already present in the system.

Several studies have been conducted on the source of salinity found in the Great Fish and Sundays rivers. Although most of the salt originates from natural sources, much of it passes through the irrigation cycle. Viljoen and Liebenberg (1974) found a positive correlation between the amount of seepage at a weir and the size of the irrigated area within that particular reach of the weir. From an environmental isotope survey, Verhagen (1979) concluded that extreme groundwater salinities are due to leaching of the soils on infiltration. They found that tritium measurements of saline seepages from the river bank suggested shallow cycling waters.

Reynders (1984) gave a summary of the processes the Middleton area as follows: Irrigation water and to a lesser extent precipitation and leakage from canals and ponds, form the input into the system. Evapotranspiration results in 73 % of the applied water being lost to the atmosphere. The remaining 27 % passes below the root zone and joins the groundwater reservoir. Besides the concentrating effect of evapotranspiration, mobilization of salts already present in the soils further increase the TDS content of the percolate. Poor soil texture results in salts being precipitated in the soils which may be mobilized during future irrigation leads. Macropore flow through solution channels in the calcretes and fractures and joints in the consolidated formations, results in the vertical percolation to the deeper groundwater predominating over lateral flow through the soils. Lateral flow within the transmissive fractured rock aquifers returns mineralized seepage water to the river via joints and fractures.

7.7 SALINITY MANAGEMENT

The best solution to the mineralization problem of the groundwater is to minimize saline percolates from the root zone. This can be achieved by

- reduction in the leaching fraction and over-irrigation.
- reduction in the leakage from canals and dams.
- application of a better quality irrigation water in order to reduce the input of water borne soluble salts.
- artificial drainage to remove percolates rapidly from the soil body and to prevent build up of perched water tables.
- choice of area and timing of new land development.

7.7.1 Reduction in the leaching fraction

Murray, Biesenbach and Badenhorst (1993) investigated the effect of irrigation management on the control of potential irrigation losses and the resultant salinisation of the Breede River. A representative sample of irrigation plots were selected for the study. The hydraulic design of each irrigation system was evaluated, with the IDES computer program. During the evaluation, the coefficient of uniformity of water distribution at both the theoretical and measured inlet pressures were calculated. Each research plot was fitted with a BERMAD water meter in order to measure the volume of irrigation water applied. Three mini weather stations were erected in the area to measure rainfall and A-pan evaporation. Crop factors were used with A-pan evaporation to calculate daily water balances for each individual plot. The results of these calculations were verified qualitatively against results obtained from five sets of tensiometers in conjunction with changes in soil salinity.

The results showed that large potential drainage losses occurred as a result of over-irrigation. At the same time low mean levels of water use efficiency were calculated. The average recorded irrigation depth (after allowing for system losses), and calculated mean drainage depth and water use efficiency (irrigation demand as a percentage of measured irrigation, effective rainfall and available soil water) for each combination of crop and irrigation method, are shown in Table 7.6.

Table 7.6: Observed irrigation efficiency as calculated for 12 month period.

Crop/irrigation method combination	Irrigation depth (mm)	Potential drainage depth		Water use efficiency (%)
		mm	% of irrig	
Vines/drip	435	181	41.6	56.8
Vines/micro	665	260	39.1	66.5
Vines/sprinkler	699	313	44.8	67.8
Vines/flood	507	371	73.2	64.1
Cash crop/sprinkler	852	117	13.2	106.8

As a result of the calculation method used, a higher than 100 % water use efficiency was calculated when "deficit" irrigation is practised. Murray, Biesenbach and Badenhorst (1993) concluded that the average water use efficiency could be improved substantially. High water use efficiencies could be obtained in practice as was demonstrated by some of the individual farmers. Their results showed that , in spite of its low water use efficiency, the lowest potential drainage depth was calculated for drip irrigation, followed by micro-irrigation.

From interviews with the farmers, it was found that they relied mainly on their experience when taking any decisions relating to irrigation management. It also emerged that as a group, the farmers had a low theoretical knowledge about irrigation scheduling and the effects of over irrigation.

From the results of the soil water balance calculations, it was concluded that large potential drainage losses occurred with the present standard of irrigation scheduling at every plot of the valley. The authors wrote: "This fact is attributed to a low level of theoretical knowledge about irrigation scheduling, ignorance about the undesirable consequences of over-irrigation and a dearth of measures to encourage better scheduling. Should these measures be addressed successfully, the potential exists to achieve a significant reduction in drainage and the ensuing salinisation (of the Breede River) associated with irrigation return flow".

7.7.2 Reduction in leakage from canals and dams

Leakage water from canals and dams will have a similar effect to over-irrigation. Murray, Biesenbach and Badenhorst (1989) reported on a pilot study that was aimed at monitoring the water volumes and salt loads moving through an irrigation scheme. For this purpose, monitoring for an irrigation season was done on the irrigation canal inflow, volumes extracted from the canal and other sources, farm reservoir storage, and electrical conductivity of the water. The study was conducted in an area of 2,081 ha of the middle Breede River catchment, of which 1,241 ha was irrigated. Although the irrigation canal was lined and visually in excellent condition, approximately 14 % of water transported in it was lost as seepage. Canal seepage accounted for 23 % of the total water intake of the study area. An additional 13 % was attributed to canal rejects, evaporation, and farm-reservoir seepage.

7.7.3 Application of a better quality irrigation water

Applying better quality irrigation water means that the input of water borne soluble salts to the groundwater store will be reduced. Hall and Du Plessis (1979) used a computer based system model to simulate river flow and salinity in the Great Fish - Sundays River system. General conclusions drawn on future salinity levels in the Great Fish River included the following: as more water is brought in from the Orange River to meet the growing irrigation demands in the Great Fish River valley, the salinity levels in supply water will fall. Diverting more Orange River water via the Great Fish River to the Sundays River will lower salinity levels even further. There are two main reasons for the expected drop in TDS levels. First, the increased flux of low-salinity Orange River water through the system, will dilute naturally saline runoff and saline irrigation return flow. Secondly, as supply water for irrigation becomes less saline, so also will be the resultant irrigation return flow.

Greeff (1990) recommended the exploitation of high quality water stored in the sandstones in the Breede river mountain watershed. The high quality water so produced could be used for "low TDS irrigation", thereby minimizing the salinity of the resultant irrigation return flow.

7.7.4 Artificial drainage

Artificial drainage can be used to remove percolates rapidly from the soil body and to prevent build up of perched water tables. In the Vaalharts basin, a system of subsurface drains were installed between 1976 and 1979, to maintain the water level below the crop root zone (Herold and Bailey, 1996). The groundwater slowly drains into the perforated pipes in this way leaching the dissolved salts out of the root zone. A system of branch drainage canals then collects seepage and stormwater from the plots and carries it out to the main drainage canals.

7.7.5 Choice of area and timing of new land development

In areas where the soils or the underlying geological strata are quite saline, eg soils on sedimentary rock of marine origin, development of new irrigation areas must be minimized. For the Breede River Catchment, Greeff (1990) recommended that "uncontrolled development of new irrigation areas which entail the deep ripping of thin Bokkeveld soils must be restricted if the salinity levels in the catchment drainage are to be kept within present limits".

7.8 CONCLUSION

The three case studies above have provided evidence that percolates from the root zone and irrigation return flow can cause the salinisation of groundwater in irrigated aquifer systems. Irrigation return flow can cause salinisation by

- salt in irrigation water concentrated as a result of evapotranspiration.
- the mobilization of the salt stored in the unsaturated zone
- the mobilization of the salts which become available for dissolution during land preparation
- the development of suspended water tables that accelerate recharge of underlying strata

The best solution to the mineralization problem of the groundwater is to minimize saline irrigation return flow. This can be achieved by

- reduction in the leaching fraction and over-irrigation.
- reduction in the leakage from canals and dams.
- application of a better quality irrigation water in order to reduce the input of water borne soluble salts.
- artificial drainage to remove percolates rapidly from the soil body and to prevent build up of perched water tables.
- choice of area and timing of new land development.

Further Research

Effect of groundwater abstraction for irrigation on the quality and water levels below irrigated lands.

CHAPTER 8

GROUNDWATER CONTAMINATION BY PESTICIDES

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8.1 AIMS

The aims of this essay are:

- To give the reader a brief overview regarding the concerns about the occurrence of pesticides in groundwater.
- To describe the ways by which pesticides reach the watertable and the processes that prevent this from occurring (fate and transport).
- To describe the results of some USA and European investigations and all the currently (1996) known South African examples.
- To propose directions for future research efforts.
- To test some "best farming practices" which are aimed to limit the impact of pesticides on groundwater and surface water.
- To provide a list of recommended reading for giving further insight.

8.2 PESTICIDES AND THEIR IMPACT ON GROUNDWATER

- *(John Weaver)*

8.2.1 Discussion

Rachel Carson in her book **Silent Spring** published almost 30 years ago was the first person to identify the potential impact of pesticides on groundwater contamination - "Soil becomes poisoned as a result of heavy application of arsenical pesticides. Rain then carries part of the arsenic into streams, rivers, reservoirs as well as into the vast subterranean seas of groundwater" (Carson, 1962).

However, it was only 17 years later in 1979 that the first two cases of widespread contamination by pesticides in the United States were detected. Temik aldicarb was discovered in groundwater on Long Island, New York and 1,2 dibromo-3-chloropropane (DBCP) was discovered in groundwater in Lathrop, California (Wartenberg, 1988).

Aharonson (1987) reports that in 1985 a total of 17 different pesticides had been recorded in 23 different states in the USA. He further states that 201 wells have been closed due to groundwater contamination by pesticides and estimates that roughly 6 500 wells contain pesticide residues arising from **normal** pesticide usage, with resultant exposure to roughly 800 000 people in the USA.

Flury (1996) prepared a review paper which focussed on field experiments where pesticide transport through soil was studied. Laboratory studies, lysimeter studies and monitoring studies were not included. A total of 55 pesticides were included in the experiments that were reviewed. In order of number of experiments the 11 most studied are detailed in Table 8.1 below.

8.2.2 Atrazine

It is noted from Table 8.1 that atrazine is the most studied pesticide. This combined with the very large tonnages of atrazine that are applied internationally results in atrazine usually being chosen for monitoring. The corollary is that if it is monitored for, then it can be detected. This does not mean that atrazine is more of a threat to groundwater than any other pesticide which is not as extensively monitored for. In fact, for the statistically designed nation-wide survey of 1300 boreholes in the USA (US EPA, 1990) the herbicide Dacthal was detected in 6,4% of the 783 private boreholes and atrazine in 1,7% of the boreholes, yet atrazine consumption is 10 times that of Dacthal.

Atrazine is a pre-emergence herbicide used internationally for mainly corn (maize) production. Annually large tonnages are applied world-wide and in South Africa. A very similar herbicide simazine is used in grape production. The main difference with respect to groundwater is that atrazine has a water solubility of 33 ppm. Furthermore, the latter herbicide is more strongly held by soil particles than the former compound. Therefore, simazine is less likely to impact on groundwater than is the case for atrazine.

Atrazine is the pesticide which is commonly found in groundwater contamination surveys. For example a groundwater survey conducted in Nebraska involving 2263 wells had 305 wells with atrazine detects of which 22 exceeded the health advisory of 3,0 ppb (Spalding *et al.*, 1989).

Table 8.1: **The 11 most studied pesticides in terms of transport through soils (From Flury 1996)**

Common name	No. of experiments	Pesticide type
Atrazine	58	Herbicide
Alachlor	22	Herbicide
Aldicarb	20	Insecticide, Nematicide
Cyanazine	14	Herbicide
Simazine	10	Herbicide
Metolachlor	9	Herbicide
Metribuzin	8	Herbicide
Isoproturon	7	Herbicide
Carbofuran	6	Insecticide, Nenoticide
Picloram	6	Herbicide
Terbuthylazine	6	Herbicide

8.2.3 Aldicarb

Aldicarb was the first pesticide that was detected over a widespread area. This occurred in Long Island in 1979 (Wartenberg, 1988). After the initial monitoring programme of 270 wells identified aldicarb as a problem, an intensive monitoring programme conducted from April to June 1980 included the collection of water samples from nearly 8 000 wells.

Of the 7 809 wells sampled during this survey, 5 745 (73,6%) had non-detectable concentrations of aldicarb; 1 025 (13,1%) had concentrations over the recommended standard of 7 ppb; and 1 032 (13,3%) contained traces of the pesticide. The mean concentration of aldicarb in all samples in which it was detected was 23,5 ppb.

Data collected from this survey confirmed that the highest concentrations of aldicarb were found closest to potato fields (Holden, 1986). On Long island aldicarb was used as a nematicide on potato fields to control golden nematode.

Aldicarb is a systemic carbamate insecticide that also has nematocidal properties. Aldicarb is the common name for the active ingredient 2-methyl-2-(methylthio) propionaldehyde 0-(methylcarbamoyl) oxime. It is manufactured by Union Carbide Agricultural Products Co., Inc., and is sold in granular form under the trade name Temik. Because of its high acute mammalian toxicity, aldicarb is available only in granular formulations (5-15 percent active ingredients) for soil incorporation. Under field conditions, the parent material is quickly transformed to aldicarb sulfoxide and aldicarb sulfone. The characteristics of aldicarb and its metabolites indicate that the pesticide is a potential groundwater contaminant. The general character of residues in groundwater is 0 percent aldicarb, 40-60 percent aldicarb sulfoxide, and 60-40 percent aldicarb sulfone. This ratio remains essentially stable as the aldicarb metabolites degrade to biologically inactive compounds. The acute mammalian toxicity of aldicarb sulfoxide is similar to that of aldicarb, whereas that of aldicarb sulfone is considerably lower. Aldicarb and its metabolites have been relatively well tested and are not known to be carcinogenic or teratogenic (Holden, 1986).

Aldicarb is the pesticide which has a "bad boy" reputation and is usually included as a target pesticide in groundwater contamination surveys e.g. in Minnesota (Klaseus *et al.*, 1988) and Arkansas (Cavalier *et al.* 1989). This reputation is due in large part to aldicarb being the first pesticide detected on a wide scale. In general, carbamates pose a threat to groundwater as they have a high solubility and are applied to the ground. For example vydate oxamyl was the state-recommended pesticide to replace aldicarb for use on potato fields on Long Island. Vydate oxamyl has similar physical features to aldicarb, namely solubility in ppm of 2,8 x 10^5 versus 6 x 10^3 of aldicarb, partition coefficient 1,0-4,0 versus 15 and toxicity 5,4 versus 0,93. Not unexpectedly within 4 years of being used on Long Island potato fields oxamyl residues were detected in groundwater and subsequently banned in 1984 (Wartenberg, 1988).

Despite carbamates having similar behaviour the only carbamates pesticide which is regularly analysed for is aldicarb. On occasion carbofuran is analysed for. The corollary to this observation is - aldicarb is thus likely to be the only carbamate detected.

8.2.4 Pesticides and the South African Water Law (extracted from Vermeulen 1996)

Section 22 of the Water Act, Act no. 54 of 1956 deals with the prevention of water pollution. Any person who has control over land on which any action was or is taken which involved or involves a substance capable of causing water pollution, whether such substance is a solid, liquid, vapour or gas or a combination thereof, shall take such steps as may be prescribed by regulation under section 26 to prevent any public or private water on or beneath that land, including rainwater which falls on or flows over or penetrates such land, from being polluted by that substance.

There are presently, however, no standards for pesticide levels in water. The Department of Water Affairs and Forestry uses the 1993 World Health Organization Guidelines for Drinking-Water Quality. These guidelines have been used in South African court cases concerning water pollution and the courts have based their decisions on them. The list of pesticides in these guidelines is, however, very limited.

8.3 PESTICIDE FATE AND BEHAVIOUR IN THE SUBSURFACE OF SOILS - *(Charlie Reinhardt)*

8.3.1 Introduction

Pollutants are transported in water from the soil surface to ground- and surface water. It is the supreme capability of water as a solvent which makes the carrying of significant quantities of dissolved material possible. In agricultural situations the most important solutes are nutrients (principally nitrogen compounds) and pesticides which can be found in water resources due to surface run-off and/or leaching from both agricultural and non-agricultural land. Pollution of surface waters is chiefly the result of erosive surface run-off that is associated with intense rainfall. Over-irrigation in summer months could also contribute to movement of pesticides across the soil surface. Obviously, application of pesticides on or near surface water poses a more immediate threat. Groundwater pollution by pesticides is strongly linked to the rates at which the compounds are leached through the soil. Leaching rate is governed by the adsorptive capacity of the soil, the amount of overhead water, and the solubility of the compound leached.

From an agricultural viewpoint, pesticide dissipation refer to losses that occur in the target area and which relate to decreased biological activity on pests. However, pesticides that move from the target area present particular problems in terms of potential environmental impacts. There are many factors that determine the rate of dissipation (=loss) of pesticides from soil. Basically, two distinct groups of factors are involved, namely transfer and transformation processes. Typically, transfer processes involve the transferal of a compound from point A to point B, without its biological activity necessarily being affected. Transformation, on the other hand, entails the changing of the chemical structure of molecules, which usually results in loss of biological activity. Transformation of a compound may occur at any stage during its transfer from one location to another.

The fate of a pesticide applied to soil is governed to a large extent by its adsorption on soil colloids, desorption of the adsorbed fraction, its degradation and persistence. Part of the amount of a pesticide that is sprayed onto plants can be expected to eventually reach the soil through wash-off by rain, or simply, through the spray solution missing the target (the plant) and landing directly on the soil surface. Pesticides dissipate from soil in many ways. By definition, dissipation infers that the chemical is transformed into by-products, which are further transformed, and that the parent molecule and its by-products are also transferred to other places and transformed until none exists, or in most cases none is detectable. Hence the many processes in the dissipation of pesticides are categorized as transfer processes or transformation processes. Some of the more important avenues of dissipation are chemical degradation, leaching, microbial degradation, uptake by living organisms, photo decomposition and volatilization. There are many excellent reviews on the many chemical, physical and biological processes involved in the dissipation of pesticides in the environment (Bollag & Liu, 1990; Wolf, Mingelgrin & Miller, 1990; Weber, 1991a, b; Leeds-Harrison, 1995; Walker, Welch & Turner, 1995).

The physics of soil water and solute movement can be used to determine the fate of these components, although the physico-chemical interactions of the solute with soil particle surfaces will vary depending on the nature of the solute as well as the solid surfaces it encounters. Partitioning of solutes between the solid and water fractions of soil, thus their relative affinities for those entities, will determine the extent and duration of their presence in one or the other soil compartment. In general, solutes occurring mainly in the water phase of the soil would be prone to leach, whereas those compounds that are associated with the solid phase (soil particles) will be less exposed to this form of dissipation. There are excellent review articles on the mechanisms of water and solute movement to surface and groundwaters (Enfield & Yates, 1990; Leonard, 1990; Leeds-Harrison, 1995; Leake & Gatzweiler, 1995), and time-dependent sorption processes in soil (Koskinen & Harper, 1990; Walker *et al.*, 1995).

Logically, compounds that are chemically stable, and also resistant to microbial degradation, will tend to persist for a long time in soil. Hence the risk that a persistent compound may leach and eventually contaminate groundwater is high. Persistence and mobility in soil are therefore important predictors for estimating the potential risks for the principal users of such waters, i.e. plants, animals and humans. Consider that risk is a function of both the inherent toxicity of a compound and a sensitive organism's exposure to it:

Risk (Hazard) = Toxicity x Exposure

What this means is that an organism may be harmed when exposed for only a brief period to a highly toxic compound, and that a low toxic product is unlikely to cause injury in the short term, but possibly in the long-term.

The risk posed by pesticides in terms of contamination of groundwater resources can be described as follows:

Risk = Persistence x Mobility

Reviews on the impact of pesticides on the environment (Madhun & Freed, 1990), and on risk assessments/interpretation (Severn & Ballard, 1990; Zimdahl, 1993) exist.

The principles of solute movement through soils are discussed next with special reference to factors affecting pesticide mobility and persistence. An overview of the basic factors controlling adsorption, leaching and the persistence of pesticides is given, and the complications introduced by either poor or strong adsorption/leaching, and short or long persistence, are discussed.

8.3.2 Transfer Processes

Transfer processes involve the following mechanisms:

- adsorption on, and desorption from, soil colloids;
- uptake by plants (absorption);
- exudation and retention by living organisms;
- movement on the soil surface (run-off);
- volatilization;
- leaching (downward movement in soil), or
- capillary movement (upward transport in soil).

The most commonly detected pesticide in groundwater, the herbicide atrazine, will often be referred to in this chapter. There is considerable interest in atrazine and other *s*-triazine herbicides because of their extensive use in agriculture, their persistence in soil, effects on succeeding crops, and movement into surface and groundwater. The herbicidal activity, mode of action, and fate of atrazine in biological and soil systems have been extensively reviewed by Esser, Dupuis, Ebert, Marco & Vogel (1975). Erickson & Lee (1989) reviewed factors involved in the degradation of atrazine and related *s*-triazines. Nel & Reinhardt (1984) reviewed factors which affect the activity of atrazine in plants and soil, and also summarized local research findings at that stage. Later in this chapter the focus is on locally conducted research on the mobility and persistence of several herbicides. Comprehensive reviews on the effects of soil factors on herbicide activity have been presented by Riley & Morrod (1976), Eagle (1976), Walker (1980), Hance (1983) and Weber (1991a, b). Soil factors could be expected to have similar effects on pesticides with the same chemo-physical characteristics, because their mobility in and uptake from the soil depend on the soil/pesticide interaction. Differential biological activity (bioactivity) of pesticides is only expressed once the chemical reaches the active site in sensitive organisms.

8.3.2.1 Absorption, exudation and retention by plants

The amount of pesticide required to affect (harm) target species is determined by:

- The rate of uptake (absorption) by the organism.
- The release of absorbed compounds through exudation.
- The rate of metabolism (inactivation) of a compound inside the organism.
- The inherent sensitivity of the organism towards a pesticide.

Absorption of pesticides by plant roots reduces the amount that is available to leach and possibly reach free water beyond the rooting zone. The extent to which pesticides are removed from soil by plant roots is dependent on the physico-chemical characteristics of the compounds and the soil. Pesticides that are mobile in both the soil and the plant system will get to the target sites in the plant quickest. This trait is of particular importance for herbicides. A high degree of water solubility will assist in the fast transport of compounds inside plants. The many soil factors that determine the availability of compounds for absorption by roots are dealt with in subsequent sections.

It is virtually impossible to predict how much of an applied pesticide is actually absorbed by target organisms. According to Weber (1991a, b) the amount of a mobile herbicide that is normally absorbed and retained by plants ranges from one-tenth to five per cent of the total amount applied. High plant numbers may result in much bigger amounts being removed from the root zone of soil, but amounts rarely exceed 10% of that applied, and that fraction which is absorbed is sooner or later detoxified in the plant. Thus, absorption of pesticides from the root zone contributes little to their dissipation from the soil subsurface.

8.3.2.2 Adsorption on soil colloids

Pesticide adsorption on soil particles, i.e. the strength of pesticide retention in soil, is dependent on soil chemophysical (chemical and physical) properties, the chemical structure of the compound, and environmental factors such as temperature and soil water content. Adsorption, and not the water solubility of a compound *per se*, is the major factor determining its mobility in soil, including leaching to groundwater. Adsorption also reduces the availability of compounds for absorption by plants for instance, thereby reducing their immediate bioactivity.

The availability of pesticides in soil for absorption by plant roots or other organisms occurring in the soil is dependent on the adsorption-desorption equilibrium being shifted towards the desorption side. This equilibrium must be expected to be quite variable due to the involvement of the different, often interrelated, constituents of the solid soil phase. These soil constituents include clays, hydrous oxides of iron and aluminium, organic material and various other organic substances. It is likely that due to the dynamic and open nature of soil, absorption by plants, insects and microorganisms, and degradation processes ensure that equilibrium with respect to pesticide concentration between the adsorbed and solution phases is never reached.

It has been shown that herbicides generally, and the triazines particularly, are reversibly adsorbed on the organic matter and clay mineral fractions in soils (Frissel, 1961; Talbert & Fletchall, 1965; Weber, 1970a,b; Hayes, 1970; Schiavon, 1988). Several studies revealed that soil organic matter is the most important soil factor as far as atrazine adsorption or bioactivity is concerned (Talbert & Fletchall, 1965; Weber, Weed & Ward, 1969; Harrison, Weber & Baird, 1976; Anderson, Stephenson & Corke, 1980). In spite of relatively low organic matter contents in South African soils, Smit, Nel & Fölscher (1980), Nel & Reinhardt (1984) and Ehlers, Reinhardt & Nel (1987, 1988) found that the organic carbon content (% C) of soil was strongly and negatively correlated with atrazine bioactivity. In all the above-mentioned studies, organic matter content consistently predicted atrazine availability better than either cation exchange capacity (CEC), total clay content or individual clay minerals. According to Eagle (1983b), a positive relationship usually exists between the clay and organic matter content of soils, but problems tend to occur when this is not the case, and the recommendations for herbicide rates to be applied are based solely on the clay content of soils. In these situations, either excessive bioactivity, which may lead to crop damage, or low bioactivity resulting in poor weed control may be expected.

The organic matter, clay minerals and sesquioxides represent the active fraction of soil, and silt and sand the inactive fraction. The active fraction, which is involved in chemical reactions in the soil, is colloidal in nature, i.e. the constituents representing this fraction are organic and inorganic substances with a small particle size (< 0.001 mm) and a resulting large surface area per unit mass. According to Weber (1991a), it can be expected that the tilled surface soil is a relatively homogeneous mixture of the active and inactive fractions. Koskinen & Harper (1990) and Weber (1991a, b) review the characteristics of soil colloids which are related to their ability to adsorb pesticides. Properties of soil colloids involved in adsorption of organic pesticides are briefly discussed next.

8.3.2.2.1 Organic matter

Organic matter is more flexible than clays with respect to the ability to adsorb (=retain) a wide assortment of pesticides. Well-decomposed organic matter have functional groups and ring structures that may carry either positive, negative or no charge, thus allowing the binding of both anions and cations of pesticides on a single organic matter molecule. Some degree of solubility is required of pesticides to become retained at the highly hydrated hydrophilic surfaces (${}^{(-)}$charge) of clay particles, whereas organic matter has both hydrophilic ("water-loving") and lipophilic ("fat-loving") characteristics which enables it to react with pesticides of wide ranging solubility and ionizability. According to Kononova (1966), and Weber, Swain, Strek & Sartori (1986), humified substances (humus) make up 17 to 97% of the total organic carbon in a soil. Humus consists predominantly of humic acids which are high molecular mass compounds with functional groups and aromatic rings which are lipophylic in nature and which possess numerous ionizable carboxyl and hydroxyl groups that give the polymer pH-dependent exchange properties (Kononova, 1966).

8.3.2.2.2 Clay minerals

Bohn, McNeal & O'Connor (1985), Koskinen & Harper (1990) and Weber (1991a, b) list some important characteristics of certain clay minerals which adsorb pesticides. The two major types of clay minerals present in soils are the 1:1 and 2:1 (silica:alumina) types (Brady, 1974). Kaolinite (1:1 clay mineral) does not exhibit high-intensity colloidal properties because of its limited adsorptive capacity for cations and a relatively low surface area. The intensive study of Frissel (1961) focussed on the important role of montmorillonite and to a lesser extent illite and kaolinite in the adsorption of triazines in soil. Talbert & Fletchall (1965) reported that kaolinite at pH 5 and pH 7 did not adsorb either atrazine or simazine. In the 2:1 expanding type of clay, montmorillonite, there is a very large internal surface which far exceeds its external surface area. The combined internal and external surfaces (specific surface area) of montmorillonite greatly exceeds the surface area of kaolinite. Negative charge predominates on the clay mineral surfaces, thus affording many adsorption sites for positively charged ions (cations) of pesticides.

8.3.2.2.3 Oxides and hydrous oxides of iron and aluminium

These colloidal fractions are intermixed with silicate clays in temperate regions. They are prominent in tropical and sub-tropical soils in several mineralogical forms, including boehmite, gibbsite, goethite and hematite (Brady, 1974). These minerals have a pH-dependent charge, and therefore may exist as positive, neutral, or negatively charged particles. The hydrous oxides have a much smaller cation exchange capacity (CEC) than kaolinite. In South Africa the amorphous Fe-Al-OH component is commonly found in acid soils in which kaolinite predominates. The Fe subfraction tends to be bonded strongly to crystalline minerals such as kaolinite, whilst the more weakly bonded Al fraction carry active positive charge (Fouché & Brandt, 1973). In acidic soil, anions of pesticides will be attracted to and retained at the positive charges on the Fe-Al-OH fraction.

8.3.2.2.4 pH-dependent charge on soil colloids

There are two general sources of CEC for inorganic colloids: (a) permanent negative charge that results from isomorphic substitution, and (b) pH dependent charge which results from the ionization of hydrogens from hydroxyl (OH) groups located at the edges of micelles and from iron and aluminium hydroxides (Anderson, 1983). The greater surface area of the expanding type clays, combined with a much greater CEC, endows them with a much greater propensity for binding pesticides than the non-expanding clay types (Weber, 1972; Weber, Shea & Weed, 1986). Organic colloids will carry mainly negative charge at high soil pH, whilst positive charge will develop under acidic conditions. Pesticide molecules that do not ionize in water are also retained by organic colloids, and therefore, this colloidal fraction has the capacity to bind pesticides that occur in the anion, cation or neutral form.

8.3.2.3 Influence of soil pH, water content, electrolyte concentration, temperature and pesticide "age" on adsorption

The primary roles of the physico-chemical properties of both the pesticide and the soil colloids in the adsorption-desorption phenomenon have been discussed above, but the availability of, e.g., herbicides for absorption by plant roots is known to be directly or indirectly influenced by several other factors. Although identification of the individual effects of soil factors involved with pesticide availability for uptake is often complicated by interrelationships between them, several factors which affect pesticide adsorption have been documented.

8.3.2.3.1 Soil pH

As alluded to earlier, soil pH influences pesticide adsorption through its effect on the charge characteristics of both the chemicals and soil adsorption sites. Weber & Whitacre (1982) found that atrazine is a weak base in aqueous solution and exists as a molecular species (zero charge) at high pH, and cations at low pH. Only atrazine in the cation form would be subject to adsorption on soil colloids with negative charge, thus explaining its increased adsorption and lower leaching/bioactivity at low pH in most soils. It was shown by Colbert, Volk & Appleby (1975) that adsorption of atrazine and terbutryn decreased on natural and limed soils as the soil pH increased to pH 8. Best & Weber (1974) found that the total amount of atrazine and prometryn applied was absorbed by plants over a five-month period ranged from 0.6 to 4.3% and was closely linked to the pH of the soil, with higher herbicide concentrations occurring in the plants at the higher soil pH level (7.7) compared to the lower pH (5.5). Consistent with the reported decrease in *s*-triazine adsorption with increased pH, Smit *et al.* (1979, 1980) showed that the phytotoxicity of atrazine in some soils was increased by increases in the soil pH from below pH 5 to about pH 6. They attributed this effect to increased availability of atrazine for uptake by plants, and also to increased chemical stability of atrazine molecules at the higher pH levels.

Several studies revealed that adsorption of *s*-triazines increased with a concomitant decrease in soil pH (McGlamery & Slife, 1965; Talbert & Fletchall, 1965; Weber, 1970b; Yamane & Green, 1972; Marshall, Nel & Smit, 1982). Protonation of atrazine molecules, and subsequent adsorption at negatively charged sites on soil colloids, progressively increases as soil pH decreases. Weber (1970b) suggested that atrazine may be bonded by complexation with protons on clay surfaces under acid conditions. Maximum adsorption of atrazine may be expected to occur near the pKa value (1.68) for this herbicide (McGlamery & Slife, 1965; Weber, 1970b). Reduced sorption is possible at pH levels lower than the pKa value, because of increased competition between hydronium ions (H_3O^+) and atrazine (cations$^{(+)}$) for negatively charged sites on soil colloids.

Harris & Hurle (1979) showed that atrazine and simazine adsorption onto clay colloids are highly sensitive to minute changes of soil solution pH and that plants can play an important role here by bringing about such changes in pH through ion exchange. They theorized that since triazine adsorption is so highly sensitive to pH changes, any change in rhizosphere pH through ion exchange or microbial activity must affect the adsorbed/solution herbicide equilibrium and, therefore, the amount of herbicide immediately available to the plant.

8.3.2.3.2 Soil water

The availability of pesticides for uptake by underground plant parts is influenced directly by soil water content, since water is the medium in which chemicals are transported in soil, and from which it is either adsorbed or absorbed. Green & Obien (1969), Bailey & White (1970) and Appleby (1985) found an inverse relationship between atrazine adsorption and soil water content. Dao & Lavy (1978) suggested that soil water depletion would increase the atrazine concentration in soil solution, allowing polar atrazine molecules to compete favourably with a reduced number of water molecules for binding sites on soil colloids. Green & Obien (1969) were of the opinion that atrazine mobility in soil, and not fluctuations in atrazine concentration in soil solution *per se*, governed herbicide availability at the root surface. Several studies indicated that atrazine phytotoxicity was linked to soil water content, with increases in bioactivity as soil water content increased (Lavy, 1968; Dao & Lavy, 1978; Nel & Reinhardt, 1984).

The depth to which a pesticide leaches in soil is a function of its mobility, persistence, weather factors and percolating water. These parameters need to be considered in estimations of the potential of a pesticide to contaminate groundwater. Leaching of a compound beyond the root zone is dependent on the amount, frequency and intensity of water received. Apart from temperature, which has a significant effect on microbial activity and hence the persistence of compounds, the most important climatic characteristics relating to pesticide leaching are the duration of the "field capacity period" and the amount of "excess rain" during this period (Koskinen & Harper, 1990; Hollis, 1991). According to Talbert & Fletchall (1965) and Nel (1975), most of the atrazine applied at normal rates can be leached out of the root zone, but those fractions adsorbed on the internal surfaces of 2:1 swelling type clays, such as montmorillonite, would be least subject to leaching. Based on K_{oc} values, atrazine (K_{oc} = 102 to 163) is classified as moderately to slightly mobile (Leonard *et al.*, 1988). Because of the pH-dependent adsorption of atrazine on soil colloids, the herbicide is more susceptible to leaching under neutral and alkaline than under acid conditions (McGlamery & Slife, 1965; Smit & Nel, 1977).

Coarse-textured soils, which are low in organic matter and clay, would be particularly susceptible to pesticide leaching because these soils have limited sorption capacity. Fleming, Wax & Simmons (1992) contend that continued registration of atrazine for use in these soils may depend on development of formulation or application methods that reduce leaching potential. Controlled release formulations may have promise for reducing the mobility of atrazine. Fleming *et al.* (1992) showed that a starch-encapsulated atrazine formulation (granules 20-40 mesh) should result in relatively fast atrazine release (compared to release from large granules of 14-20 mesh) and improved weed control (compared to weed control provided by the large granules and a commercial dry flowable formulation), while still reducing the leaching potential of atrazine.

According to Spencer & Cliath (1973), herbicides in the unsaturated soil solution may undergo some net upward movement with water as it moves from high to low potential via evaporation from the soil surface. Hubbs & Lavy (1990) states that the magnitude of this effect is proportional to the adsorbed and soil solution amounts of a compound. These researchers found that evaporation of water from the soil surface, and consequent upward movement of atrazine with capillary water, may be important in the dissipation of the herbicide.

Water solubility is often regarded as an indicator of adsorption and leaching. Retention of urea and triazine herbicides in the upper layers of the soil profile was originally attributed to their low water solubility. As discussed by Hartley (1976), it was subsequently recognized that adsorption by soil, not insolubility, was the most important factor retarding the movement of chemicals through soil. According to Briggs (1984), there is an inverse relationship between solubility and K_{ow} and hence K_d for liquids, whilst for solids an additional factor is the energy required to disrupt the crystal structure. This is related to the melting point (T_m) of the chemical (Briggs, 1981a) by the equation:

$$\log \text{water solubility (mole ml}^{-1}) = -\log K_{ow} - 0.01(T_m - 25)$$

It can be calculated that there can be a 1000-fold variation in water solubility between a liquid and a high melting point solid of the same K_{ow} and a corresponding 36-fold difference in K_d for compounds of the same water solubility (Briggs, 1984). Nicholls, Briggs & Evans (1984) showed that there are few chemicals for which water solubility is an important feature of their behaviour in soil. After an initial equilibration period all soil-applied herbicides are either adsorbed or dissolved at normal field rates, water solubility being a factor only for simazine and lenacil, each with log $K_{ow} < 2$ and a high melting point (Briggs, 1984). The log K_{ow} values/melting points for atrazine and simazine are 2.0-2.5/173°C and 1.5-2.0/225°C, respectively. From equation (4), the water solubility of atrazine (33 mg L^{-1} at 25°C) would be expected to be higher than that for simazine (3.5 mg L^{-1} at 20°C).

Weber (1991a) states that organic matter is more universal in its ability to react with a wide assortment of herbicides with varying degrees of solubility and ionizability. Grover (1965) contends that under high soil water conditions, the availability of atrazine will predominantly be determined by the extent of its adsorption on the hydrophobic adsorptive sites on organic colloids. Under high soil water conditions adsorption on hydrophilic sites could be insignificant because atrazine would be readily desorbed from these sites by the overwhelming number of highly polar water molecules. Under low soil water conditions the availability of hydrophilic adsorptive sites will affect the bioactivity of atrazine. As the water content in the soil is decreased, more atrazine will be adsorbed on the now accessible hydrophilic surfaces (Grover, 1965).

8.3.2.3.3 Electrolyte concentration in soil solution

The differential availability of nutrients and soil-applied pesticides for uptake by plants depends on differences in their solubility in the soil solution and the extent and strength of adsorption. Hurle & Freed (1972) found that atrazine and simazine adsorption increased in the presence of rather high electrolyte (NH_4^+, K^+, Ca^{++}) concentrations and ascribed this effect to depressed herbicide solubility. Since the replacing power of cations on clay and humus in general follows the order $C^+ < C^{++} < C^{+++}$ (Brady, 1974), the divalent Ca^{++} ions would cover more of the negative adsorption sites and were more strongly adsorbed than the monovalent K^+ and NH_4^+ ions (Hurle & Freed, 1972). Herbicide adsorption could, therefore, be limited by competition between atrazine and cations for negatively charged adsorption sites, thereby increasing the concentration of atrazine in the soil solution. These findings were corroborated when Dao & Lavy (1978) showed that atrazine adsorption increased with an increase in the concentration of KCl and NH_4Cl in the soil solution. However, the electrolytic effect of $CaCl_2$ was comparatively low, probably because of competition between Ca^{++} and atrazine for adsorption sites (Dao & Lavy, 1978). Hurle & Freed (1972) are of the opinion that the effect of electrolyte concentration is negated under field conditions by constantly changing uptake by plants, dilution or precipitation.

8.3.2.3.4 Temperature

Temperature affects pesticide activity in various ways, often interrelated with other environmental factors. Temperature affects the bioactivity of pesticides by influencing their adsorption in soil, the rates of absorption by organisms, and their fate in those organisms.

Temperature is believed to exert an indirect influence on the adsorption process through its effect on pesticide solubility. Bailey & White (1970) reported the existence of an inverse relationship between the degree of adsorption and solubility within a number of herbicide groups, in particular *s*-triazines. McGlamery & Slife (1965) suggested that solubility play a role in the temperature effect on desorption as the solubility of atrazine is only 22 mg L^{-1} at 0°C, while it is 70 and 320 mg L^{-1} at 27 and 85°C, respectively.

8.3.2.3.5 Age of pesticides

Another factor affecting the bioavailability and dissipation of pesticides from soil is the length of time they have already been in the soil (Lavy, Mattice, Massey, Skulman, Senseman, Gbur & Barrett, 1996). Several studies have shown that differential dissipation rates exist soon after pesticide application (across days or months) and after the pesticide has aged (across months or years). All of these results are consistent with the compounds becoming less available with time due to diffusion into soil micropores or into or onto soil organic matter. Compounds showing increased adsorption are less available for uptake by plants, as well as for leaching to greater depths.

8.3.2.4 Preferential flow in soil macropores

The discussion in previous sections centred on matrix flow of water and solutes. It would be incorrect to assume that knowledge of the roles of the factors discussed above will make the prediction of water and solute transport easy and accurate. Bypass flow in soil macropores may make prediction of leaching tenuous under certain circumstances. In contrast to matrix flow in micropores, relatively little adsorption of pesticides occurs in macropores and, as a result of intense rainfall or over-irrigation, transport of even normally highly adsorbed compounds may occur beyond the root zone. The presence of preferential pathways in soil may explain fast vertical movement of pesticides. This type of water/solute movement is referred to as preferential flow. The phenomenon of macropore flow is explained in detail by Hall, Murray & Hartwig (1989) and Van Genuchten, Ralston & Germann (1990).

Therefore, although the soil constituents are responsible for retaining and regulating herbicide mobility and bioavailability, the structure and architecture of the soil also play critical roles. Soils containing channels from earthworm activity, dead roots or voids from expansion and contraction normally have high infiltration rates and water may move through these macropores and cracks faster and in greater amounts than through soils with undisturbed matrixes. Pesticides dissolved in void-occurring water should move faster and to greater depths than water in smaller openings (e.g. soil micropores), providing the chemicals were dissolved in the water phase and were not mixed with the soil media (Koskinen & Harper, 1990; Weber, 1991a, b; Aderhold & Nordmeyer, 1995). Fortunately, the latter is normally the case, and therefore, the flow of pesticides is usually impeded, to some extent at least.

8.3.2.5 Volatilization

Pesticides may move from one compartment of the environment to another in vapour form. The propensity of pesticides to convert from the solid or liquid phase to the gaseous state is strongly linked to a compound's vapour pressure. Generally, the higher the vapour pressure of a compound the greater the chance that losses through volatilization will occur. Incorporation or leaching of pesticides into soil normally reduces volatility loss. Compounds that are strongly adsorbed on soil colloids is less likely to vaporize compared to those that are weakly held or occur mainly in the water phase of the soil. In anticipation of volatility loss, volatile compounds are either injected into the soil or incorporated mechanically immediately after application.

8.3.3 Transformation Processes

The above discussion of transfer processes involved in the determination of pesticide behaviour has dealt with some important components of the complex of factors which governs their mobility, bioactivity and persistence in soil. Although degradation of pesticides in soil is only one component of this complex, it is generally considered to be the principal route for loss of most compounds from soil (Wolfe, Mingelgrin & Miller, 1990; Bollag & Liu, 1990; Walker, 1987, 1989). Soils provide an ideal environment for many types of degradative processes. Soil is a dynamic and complex biological and chemical medium in which numerous variables interact to determine the degradation rate of a compound and, hence, its persistence. In this context, factors such as soil water content, temperature, soil texture, nutrient status, organic matter, pH and microbial activity are regarded as important variables. Degradation processes usually involve direct chemical transformation of the compound, and the activities of soil microorganisms.

8.3.3.1 Aspects of soil properties

Organic matter content of soil could be of particular importance in determining the degradation rate and thus the persistence of pesticides. The dominant role of organic matter in the adsorption of pesticides was cited in the above discussion. Organic matter also regulates microbial activity of soil - with generally higher levels of biomass and respiration in more organic soils (Walker, 1989). The clay content of soil may also be important in determining pesticide persistence, because as pointed out earlier, it is an important predictor of a compound's availability in the soil solution. When working with natural soils, it can be difficult to separate the effects of clay and organic matter since they are often correlated (Eagle, 1983a; Reinhardt & Nel, 1989). Clay content is an important component of soil texture, and soil texture can have a marked effect on other soil properties. Coarse-textured soils with low clay content tend to have less organic matter and lower microbial biomass than fine-textured soils (Walker, 1989). Soil texture will also influence pore size distribution in soil and consequently water relations and aeration characteristics.

Soil pH can have a pronounced effect on pesticide degradation rates. As pointed out earlier, pH exerts a direct effect when the stability of the chemical is pH-dependent, or it has an indirect effect through changes in microbial populations, or through colloidal adsorption of the compound.

8.3.3.2 Aspects of temperature and water

In addition to the variability in pesticide degradation rates between soils, rates of loss may vary in a particular soil according to the water and temperature regimes encountered. In general, rates of pesticide loss increase with increases in temperature and soil water (Walker, 1989). Increased atrazine degradation rates with increasing temperature and soil water have been demonstrated in field experiments (Harris, Woolson & Hummer, 1969) and in laboratory incubation studies

(Walker & Zimdahl, 1981). With many pesticides, Briggs (1983) and Walker & Allen (1984) demonstrated a 2 to 2.5 fold increase in half-life with a 10°C decrease in temperature, and a 1.5 to 2.5 fold increase in half-life if soil water is reduced by a factor of two. Weather conditions after herbicide application in the field should have a similar large effect on persistence. The weather, therefore, can have a marked influence on rates of pesticide loss from soil, and it is essential to be aware of this when evaluating field persistence data from different sites.

Duffy (1991) states that in warmer climates (or in years with above normal temperatures) degradation by chemical reactions will typically be faster than that observed in cooler climates (or years having above normal rainfall). This relationship also holds for microbial degradation processes, at least until temperatures are so high that death or dormancy in microbial populations occurs.

In drier climates (or in years having below normal rainfall) the probability that a herbicide will remain in the root zone is usually higher than in wetter climates (or in years having above normal rainfall). This is due to the lower net precipitation of the drier climate or year and/or the greater upward movement of soil water which would move at least a fraction of any pesticide that had been leached deeper into the profile back into the root zone (Duffy, 1991).

8.3.3.3 Chemical degradation

Degradation of herbicides in soils by purely chemical reactions has been discussed by Wolfe *et al.* (1990) and Kearney & Kaufman (1988), and the disposal of pesticide wastes has been discussed by Krueger & Seiber (1984). Chemical hydrolysis proceeds by hydrolysis, oxidation and reduction reactions.

Chemical hydrolysis of atrazine to hydroxyatrazine is regarded as the main avenue of inactivation of the herbicide in soil (Armstrong, Chesters & Harris, 1967; Gamble & Khan, 1985). The atrazine molecule is stable under neutral pH conditions, but rapid chemical hydrolysis occurred under highly acid or alkaline conditions (Armstrong *et al.*, 1967).

In studies with simazine (Walker *et al.*, 1983) and metribuzin (Allen & Walker, 1983), degradation rates decreased as adsorption on soil colloids increased, presumably due to decreased availability for degradation. Adsorption, however, does not always lead to protection from degradation, and examples of adsorption-catalyzed hydrolysis of some chlorotriazines, including atrazine, have been reported (Armstrong & Chesters, 1968; Hance, 1979). Armstrong *et al.* (1967) and Swain (1981) ascribed the major role of organic matter in the inactivation of atrazine in soil to its influence on the rate of atrazine hydrolysis. Organic matter evidently catalyzes hydrolysis through adsorption of compounds on this soil constituent, and therefore, the rate of hydrolysis reactions is rapid in soils with a relatively high organic matter content compared to soil with low organic matter contents.

8.3.3.4 Microbial degradation

Most organic pesticides are prone to microbial breakdown in soil (Bollag & Liu, 1990). Many microorganisms are known to use organic chemicals as energy sources - they feed on them, and in the process degrade (inactivate) them. As atrazine is chemically stable at neutral pH values (Armstrong *et al.*, 1967), it is likely to persist in many soils and chalky groundwater (Wood *et al.*, 1991) unless degraded by microorganisms. Kaufman & Kearney (1970) listed a large number of microorganisms that have the ability to degrade atrazine in pure culture, most of those reported being fungi. There are, however, reports of bacteria including *Arthrobacter* sp., *Bacillus* sp. and *Pseudomonas* sp. (Kaufman & Kearney, 1970; Cain & Head, 1991). Oxidative dealkylation appears to be the major mechanism by which microorganisms degrade atrazine, but degradation has also been observed under anaerobic conditions (Kaufman & Kearney, 1970).

8.3.3.5 Photodecomposition

Excessive activation by sunlight of electrons in pesticide molecules may cause loss of structural integrity which results in biologically inactive residues. This route of degradation is only important for pesticides that remain on the soil surface for a long time, or those that are exposed to sunlight in clear water. Mechanical incorporation of light-sensitive compounds in the soil is advised or else rain is required soon after application in order to leach the pesticide into the upper soil zone.

8.3.4 Significance of Persistence

The above discussion has given an indication of how variations in some soil properties can influence rate of degradation of pesticides through effects on adsorption, chemical transformation, or microbial breakdown. The end result is variability in degradation rate, and therefore, persistence that varies from soil to soil and from site to site. Persistent soil-applied herbicides such as atrazine are an essential component of many crop production systems. The time that herbicides persist in the soil is of particular importance as this has already been shown to have serious implications for the safety of sensitive following crops (Eagle, 1978; Caverley, 1983; Gottesbüren, Pestemer, Wang, Wischnewsky & Zhao, 1991).

From an agronomic viewpoint, the ideal herbicide should persist long enough to provide season-long weed control but not so long that its residues harm sensitive follow-up crops. Carry-over problems are likely to occur when soil and weather factors which favour reduced herbicide degradation prevail periodically or for a significant length of time subsequent to herbicide application; and when the particular follow-up crop is markedly sensitive to the specific herbicide residue. Excessive persistence tends to occur more frequently in some seasons than in others (Eagle, 1983b), reflecting the influence of climatic factors on rates of loss. Some soils are also more prone to carry-over problems than others. These problems are exacerbated by errors in initial application which lead to overdosing.

The persistence of pesticides is the overriding factor in the determination of their potential to reach groundwater sources in detectable amounts. For example, a highly leachable compound that has a very short persistence is unlikely to pose a threat to groundwater, provided the water table is not very high. Even if such a compound does reach the groundwater, its limited persistence will probably render it harmless quickly. On the other hand, a highly persistent compound with low mobility in soil, may eventually accumulate to unacceptable levels in groundwater if given enough time. Thus, both persistence and adsorption/leaching characteristics of pesticides must be considered in assessments of their potential to contaminate groundwater. No pesticide has an inherent persistence (expressed as half-life, i.e. the time required for the original concentration to be halved in a particular situation). In earlier discussion it was pointed out that, due to the influences of many factors and their interactions, the persistence of a compound can vary considerably from soil to soil, from one site to another, from year to year, and from season to season. Leaching behaviour can also not be expected to be constant for a particular compound. For these reasons computer models that predict pesticide leaching and persistence are extremely helpful tools in pesticide behavioural studies. Prerequisites for accurate predictions include the availability of relevant input data and validation of models in the field.

8.3.5 Modelling of Persistence and Leaching

Computer simulation models are used to explain observed dissipation under field conditions as well as to predict the likelihood of herbicide persistence under many soil and weather scenarios. Use of these predictive mathematical models requires information about soil properties and weather patterns at the location for which the simulation is made, as well as data describing the physico-chemical properties of the pesticide and its interaction with the soil constituents which affect pesticide behaviour and fate. Modelling makes it possible to do sensitivity analyses through which the most unfavourable scenarios for a pesticide can be identified. Simulation of worst-case scenarios could guide the establishment of further field tests as necessary.

An excellent review of simulation models has been presented by Wagenet & Rao (1990). The models CALF (VARLEACH), LEACHM-P, PESTLA and PRZM (PELMO) (Nicholls, Walker & Baker, 1982; Wagenet & Hutson, 1992; Boesten & Van der Linden, 1991; Carsel, Mulkey, Lorber & Baskin, 1985) are used to simulate leaching and degradation of pesticides in soils. These models do not simulate preferential or bypass flow of trace concentrations of pesticides through soils because they assume that sorption is always at equilibrium during leaching. Although the ability of macropore models to make predictive simulations is still uncertain, because simulated concentrations are extremely sensitive to the parameters that describe macropores, they are probably most useful for simulating lysimeter experiments for which difficult-to-obtain soil physical data are not readily available. The Pesticide Leaching Model (PLM) has been proposed for simulation of pesticide movement through macroporous soils (Nicholls & Hall, 1995).

8.4 POINT SOURCE CONTAMINATION - *(John Weaver)*

8.4.1 Agricultural point source contamination

The previous chapter has described the various processes and transport routes by which pesticides are able to migrate from the soil surface on which they were applied to the water-table. The other manner in which pesticides reach groundwater is termed point-source contamination.

Probably the most (in)famous case is that of the town of Rusic, Wisconsin, USA where spillage at a commercial sprayer and at a dealers' warehouse resulted in 23 out of the towns 27 domestic boreholes being contaminated (Holden, 1986).

Other point-source contamination include the following:

- Filling up a spray-tank directly from a borehole, then when switching off back-siphoning occurs and pesticides are injected directly into the borehole.
- Spills at the mixing area or other points which result in a much higher loading than normal application would.
- Discharging of excess mix either by spraying onto non-crop areas or by draining the spray-tank.
- Washing of the spray-tanks and discharging the rinse-water injudiciously.
- Applying a stronger mix than the label indicates - "just to be sure".
- Poor control at the poison-store, either at the farm or at the distributor's warehouse.

Many, if not all of these point-source errors could be overcome or prevented by awareness campaigns amongst the pesticide users. The question that all pesticide users must continually ask is "If I do this, then is there a possible harmful consequence. And if I wish to do that, what or how can I do it so that no harm can occur". Reputable pesticide distributors have on-going efforts promoting safe-use of pesticides. An example of this is the campaign in South Africa to prevent empty pesticide containers from being used as water containers.

8.4.2 Industrial point-source contamination

Industrial applications is the term used when applying much higher concentrations than would be used for agriculture. This is especially the case for herbicides where the strategy is to ensure that residual concentrations in the soil ensure that follow-up applications need be applied at greater time intervals.

An example of industrial applications is the use of herbicides to control vegetation along railway lines. Vegetation would damage the rail-bed and combined with the vast kilometres of rail-line this results in concentrations of herbicide being applied that greatly exceeds agricultural concentrations.

In Europe the detection of atrazine in groundwater has on a number of occasions been linked to railroad applications (CIBA, 1996). The following examples are quoted from Switzerland: In Belprahon the concentration dropped from 0,8 µg/L in 1988 to 0,2 µg/L in 1991; at Matraon wells above the railroad had residues of atrazine between 0 and 0,1 µg/L. Below the railroad, residue concentrations ranged between 0,82 and 10 µg/L. At Kaltenstein around 20 µg/L were measured in 1990 and after discontinuing atrazine usage the residues had dropped to 2 µg/L in 1993.

This point-source contamination has prompted Ciba-Geigy, the inventors of atrazine, to make the decision that in future the industrial use recommendations for atrazine will no longer be printed on their label (van Biljon, 1996). The purpose of this decision is to prevent bad practices, which result in groundwater contamination, from causing atrazine to be condemned for agricultural use.

A recent South African example of point source pollution has been that of tebuthiuron (herbicide) used for weed control in the sisal line along the Limpopo River north of Messina. (Reinhardt *et al.* 1996). Results of this investigation indicated that the herbicide moved out of the sisal line (either subsurface or overland flow) into the Limpopo River, from where it was pumped and irrigated onto cultivated lands.

8.5 CASE EXAMPLES OF PESTICIDES IN GROUNDWATER

- *(John Weaver)*

8.5.1 Hex River Valley, Western Cape, South Africa

This investigation was carried out in 1989 (Weaver, 1993) by the Groundwater Programme, Watertek, CSIR. The investigation was funded by the Water Research Commission.

The Hex River Valley is a steep-sided valley surrounded by rugged mountains. The valley is 23 kilometres long and varies from 0-5 kilometres in width. The valley sides are quartzites of the Table Mountain Group and the valley floor comprise alluvials overlying shales of the Bokkeveld Group. Where the mountain streams enter the valley extensive alluvial fans are developed.

The area receives winter rainfall and averages between 240-320 mm per annum. During the summer months extensive irrigation takes place. The precipitation due to irrigation in December and January is 125 mm per month which is higher than the average maximum monthly rainfall in June of 55 mm.

Eight shallow wells were chosen for groundwater sampling points. In addition to these eight wells (there were no shallow boreholes available) four tile-drains were identified as sampling points to give a total of twelve sampling sites. The owners of the farms on which the sample sites are located were interviewed and the pesticides used on these farms plus the quantities used were recorded.

The pesticides in use were assessed in terms of:

- potential impact on groundwater;
- toxicity;
- quantity used.

They were grouped in three categories with category 1 being those pesticides which would pose the greater groundwater contamination hazard and thus be those which should have priority when the groundwater is analysed for pesticide content.

A number of laboratories were approached, however, the only laboratory which was equipped and willing to carry out the analysis for pesticides was the South African Bureau of Standards

(SABS). Their estimated cost to carry out the analysis of the full range of pesticides in usage in the Hex River was R5 875 per sample. During the first Steering Committee meeting of 14 July 1989 it was resolved that only certain pesticides should be tested for, namely:

Aldicarb, aldicarb sulphone, dimethoate, fenamiphos, fenamiphos sulphone, fenamiphos sulfoxide, iprodione, methidathion, mevinphos, nuarimol, penconazole, propoxur, prothiofos and simazine.

The cost of analysing for these pesticides was R1 000 per sample.

Due to the irrigation practice and the winter rainfall groundwater samples were collected at the following intervals:

August 1989	-	for the winter rainfall and maximum water-table;
November 1989	-	after maximum pesticide application; and
March 1990	-	when the maximum irrigation water has been applied and water-table levels are reaching a minimum.

The results of the analysis for the identified pesticides show that NONE of the pesticides were present in any of the affected water-samples in quantities above the detection limit. The detection limit was 0,0005 $mg.l^{-1}$ for all the pesticides except aldicarb for which the detection limit was 0,025 $mg.l^{-1}$. A further sampling run was carried out in August and October 1991 (i.e. 2 years later). Three wells and one drain were tested for simazine with a detection limit of 0,00002 $mg.l^{-1}$. All the water-samples at this lower limit showed no simazine.

Water samples were also collected and analysed for nitrate, phosphate and potassium which are extensively used as fertilizers. In all of the samples collected nitrate and potassium were detected. The nitrate varied from 1 $mg.l^{-1}$ to 150 $mg.l^{-1}$ for water with a dissolved salt content of 20 $mg.l^{-1}$ to 1 000 $mg.l^{-1}$. Potassium varied from 9,2 $mg.l^{-1}$ to 25,3 $mg.l^{-1}$. The presence of these agricultural chemicals indicates that leaching of surface applied agricultural chemicals does take place.

The target pesticides were not detected in shallow groundwater in the Hex River Valley. The groundwater and soil conditions of a shallow water-table, unconfined aquifer, coarse soils with a low clay content, low pH and high hydraulic conductivity are all conducive to allowing pesticides to reach the water-table and thus contaminate the groundwater. Furthermore the presence of high levels of nitrate and potassium in the water samples collected shows that leaching of agricultural chemicals does take place. Since this leaching has taken place it is expected that pesticides do have the potential to reach the water-table. However, since no pesticides were detected in the groundwater this indicates that either processes have occurred which have either prevented pesticides and not nitrate and potassium from reaching the water-table or that other processes have degraded pesticides before they have reached the water-table.

This investigation was carried out in 1991.

8.5.2 Vaalharts Irrigation Scheme, Northern Cape, South Africa

Two pesticide in water investigations have been carried out in this irrigation scheme. The first was carried out by the Hydrological Research Institute (HRI), Department of Water Affairs and Forestry in 1989 (White, 1989). This was an intensive investigation of the occurrence of atrazine and other pesticides in surface water (incoming irrigation water) and to a lesser extent in groundwater (outgoing drain water).

The second was carried out by the Groundwater Programme, Watertek, CSIR in 1991 (Weaver, 1993). This investigation, in collaboration with the HRI, investigated atrazine residuals in the soil profile. Some groundwater samples were also analysed.

The Vaalharts Irrigation Scheme is the largest irrigation scheme in the RSA with 32 000 hectares under irrigation, most of which is flood irrigation. The scheme originally comprised approximately 1 200 farms of 25 hectares each. Today many of these smaller farms have been amalgamated into larger units. The main crops are cotton, maize, ground-nuts and lucerne. Crop-rotation is the norm. The area falls within the summer rainfall zone.

For the HRI investigation, seven pesticides were monitored, namely atrazine, parathion, trifluralin, carbofuran, EPTC, bromoxynil and endosulfan. They were selected as being representative of the main pesticide groups, the triazines, the carbamates, the organochlorines and the organophosphates. A total of 11 surface water sampling stations were chosen and 6 groundwater sampling stations.

The surface water sampling programme of the HRI investigation which collected monthly water samples, commenced December 1987 and stopped March 1989. At Spitskop Dam only one sample was collected, at the second site on the main supply canal 5 samples were collected and for the remainder of the stations between 11 and 16 samples were collected. A total of 135 samples were collected. EPTC was detected at low levels once in December 1988. Carbofuran was detected in five samples in December 1988 and in no other sample run. Atrazine was detected regularly throughout the year at varying concentrations at all sample stations. The other four pesticides were not detected.

The groundwater sampling programme commenced in August 1987 for two boreholes. Analysis for pesticides was not possible due to interference by rubber compounds derived from the pumps. On November 1988 the groundwater sampling programme re-started at 3 boreholes which were rubber-free. A total of 12 samples were collected, all of which had no detectable atrazine or other pesticides.

Nitrate (as N) was present in all the borehole water samples collected at values ranging from 2,8 to 20,1 mg.l^{-1}.

The atrazine and nitrate analytical results for the HRI investigation are summarized in Table 8.2.

Table 8.2: Summary of atrazine and nitrate levels detected in incoming and outgoing surface water, Vaalharts Irrigation Scheme.

ATRAZINE	**No of samples**	**No of samples below detection limit**	**Maximum value mg.l^{-1}**	**Median value mg.l^{-1}**
Incoming (irrigation water canal) surface water	18	0	0,00159	0,00042
Outgoing (drain water canal) surface water	76	24	0,00073	0,00006

NITRATE	**No of samples**	**Maximum value mg.l^{-1}**	**Minimum value mg.l^{-1}**	**Median value mg.l^{-1}**
Incoming (irrigation water canal) surface water	19	0,84	0,01	0,18
Outgoing (drain water canal) surface water	90	13,03	0,44	6,28

Thus by applying irrigation water to the fields and allowing the water to percolate through the unsaturated zone atrazine was reduced 7-fold and nitrate was increased 35-fold.

The conclusion that is made is that the incoming irrigation water has low nitrate but has been contaminated with atrazine by up-stream activities, probably due to overland wash-off from dry-land mealie farming areas. After irrigation takes place in the Vaalharts area this atrazine is degraded during the irrigation cycle to produce a drainwater which is virtually atrazine-free. This water has, however, leached nitrate which was applied as fertilizer such that outgoing nitrate concentrations are 35 times that of the incoming water.

For the second investigation by the CSIR of atrazine residues in the soil profile the Vaalharts Agricultural Research Station was selected for the sampling site. This is the only place in the Scheme where atrazine is still being used as a herbicide. Elsewhere atrazine is no longer being used on the recommendations of the pesticide suppliers. This is due to observations that atrazine applied as a pre-emergence herbicide for mealies was being carried over as residuals. This was affecting the following season's crops, especially cotton.

At Vaalharts there were three fields close together where atrazine had been applied 3 months previously, 15 months previously and 27 months previously. A short distance away was another field where atrazine had been applied 63 months previously.

At these sites the soil profile shows that the upper zone is a dark red-brown soil with abundant organic material. The lower zone is mottled grey and red soil and is the zone of water-logging. There are neutron probe tubes in these fields which show that at approximately 1 m the soil is saturated. The grain size distribution curves show that the soil at all the sites is very uniform in nature and is a silt/fine sand. Soil samples were collected from each horizon.

Water samples were collected from two boreholes which are situated down-gradient of the research station. A water sample was also collected from a subsurface drain which drains these fields.

Atrazine was not detected in any of the twelve soil samples or the two borehole water samples submitted for analysis. Atrazine was detected in the drain canal water sample at the very low concentration of 0,00002 $mg.l^{-1}$ which is the method detection limit.

The HRI investigation showed that the incoming water has relatively high atrazine levels and after application as irrigation water the atrazine levels are reduced 7-fold. At the same time the nitrate levels are increased 35-fold. It is also known that atrazine is not used by Vaalharts Irrigation Scheme farmers.

At the Research Station atrazine is applied prior to planting of maize. Four fields where atrazine had previously been applied all showed no residual atrazine in the soil profile. The conclusion made is that:

- either the atrazine has all been flushed out by percolating irrigation water; or
- that the atrazine has been degraded.

CONCLUSIONS FOR HEX RIVER AND VAALHARTS

Several conclusions may be drawn from these two studies:

(i) None of the target pesticides were detected at either study area, even though the presence of elevated nitrate in the groundwater proves that leaching of agricultural chemicals does take place. The most probable cause of the non-detection is that the pesticides have been degraded due to hot weather conditions and low soil pH.

(ii) Low soil pH leads to a lower half-life for atrazine which has been reported as being between 2 weeks (Sophocleous *et al.*, 1990) and up to 106 weeks (Aharanson, 1987). The similarity of the conditions as reported by Sophocleous *et al.* (1990) and the Vaalharts study site indicates that after 90 days atrazine would not be found at Vaalharts.

(iii) Low soil pH conditions, however, lead to longer half-life for other pesticide groups. For instance aldicarb sulfone has a hydrolysis half-life of 3,5 weeks in pH 8,0 water and 347 weeks in pH 6,0 water. Thus soil pH plays a very important role in pesticide breakdown rates. Aldicarb is still in low usage in the Hex River Valley, but is likely to be used in greater quantities. The usage and effects must be closely monitored as the hydrogeological conditions in the Hex River Valley favour groundwater contamination by aldicarb.

8.5.3 California, USA (Holden, 1986)

More than 50 different pesticides have been found in the groundwater of 23 California-counties.

The most extensive monitoring effort was the 1979-1982 dibromochloropropane (DBCP) sampling programme, which revealed that the pesticide was present in approximately one in four wells tested.

Agencies in California have sampled approximately 8000 wells for the nematicide DBCP. The pesticide was found in more than 2000 wells and is known to have contaminated groundwater in an area encompassing 7000 square miles of the San Joaquin Valley. DBCP was used in California from the late 1950's until its registration was cancelled in August 1977.

Other pesticides widely monitored in groundwater include 1,2-dibromoethane (EDB), 1,2-dichloropropane/1,3-dichloropropene (D-D), simazine, atrazine and carbofuran. The confirmed presence of these pesticides in California's groundwater is much less extensive than that of DBCP. A 1983 study by the California Water Resources Control Board found that the water in 67 wells of 266 sampled (25 per cent) contained 1,2-dichloropropane, a manufacturing by-product present in the soil fumigant D-D.

A 1982 monitoring programme by the California Department of Food and Agriculture examined samples from 217 well sites chosen to tap shallow aquifers in four groundwater basins underlying agricultural areas. Four pesticides were tested for:

(i) DBCP-detected in water from 27 wells (12 per cent); concentration: 0,1 - 10,5 ppb;

(ii) the herbicide simazine - detected in water from five wells (2 per cent), concentration: 0,5 - 3,5 ppb. (Originally, the herbicide sampled for was atrazine, but none was detected; instead, simazine was found);

(iii) the nematicide EDB - detected in water from two wells (1 per cent), concentration: 0,1 - 0,2 ppb;

(iv) the insecticide carbofuran - detected in water from one well (0,5 per cent), concentration: 0,5 ppb.

8.5.4 New York State, USA (Holden, 1986)

Aldicarb was the focus of this groundwater pesticide pollution investigation. The results of the aldicarb survey are presented in Table 8.3. In addition to aldicarb approximately 50 other pesticides were tested. The results are presented in Table 8.4.

Table 8.3: Aldicarb results by Suffolk County Township, New York State

Town	Total number of well samples	Wells with concentrations in excess of 7 ppb		Wells with aldicarb concentrations between 1 ppb and 7 ppb		Wells with aldicarb concentrations below detection limits	
		Number	Percent	Number	Percent	Number	Percent
Brookhaven	222	2	0,9	18	8,1	202	91,0
Easthampton	434	43	9,9	46	10,6	345	79,5
Riverhead	2161	351	16,2	345	16,0	1465	67,8
Southampton	1832	270	14,7	256	14,0	1306	71,3
Southold	3160	359	11,4	374	11,8	2427	66,8
Totals	**7809**	**1025**	**13,1**	**1039**	**13,3**	**5745**	**73,6**

Source: Baier, Joseph and Dennis Moran, 1981, p16 in Status Report on Aldicarb Contamination of Groundwater as of September, 1981, Suffolk County Department of Health Services, Long island, New York.

Note: 1 ppb = 1 microgram/L
1 mg.l^{-1} = 1 000 microgram/L

Table 8.4: Suffolk County Department of Health services pesticide testing programme

Compound	Number of samples analysed	Detected?	Total number of detects	Remarks
Aldicarb (Temik)	±18 000	Yes	±4 000	Highest concentration 515 ppb; approximately 11% exceed New York State guideline of 7 ppb.
Aldrin	67	No		
Alachlor (Lasso)	24	No		
Arsenic	700	No		
Atrazine	14	No		
Benomyl (Benlate)	6	No		
Capifol (Difolatan)	24	No		
Carbaryl (Sevin	3945	*		Traces up to 48 ppb found in seven wells; re-samples negative
Carbofuran (Furadan)	5083	Yes*	1535	Highest concentration 65 ppb.
Chlordane	67	No		
Chlorathalonil	24	Yes*	8	Highest concentration 16,3 ppb.
Dacthal	166	Yes*	25	Highest concentration 1039 ppb.
Diazinon	17	No		
Dibrom	17	*		Traces detected; confirmation samples negative
Dicapthon	3	No		
Dichlorvos	6	No		
Dieldrin	67	No		
Dinoseb	66	Yes*	6	Traces up to 4,5 ppb found in wells.
Disulfoton (Disyston)	3	No		
DDT (o,p)	67	No		
DDT (p,p)	67	No		
Endrin	88	No		
Eptam	12	No		
EDB	4000	No*		Trace found at minimum detection level or 3 ppb.
Ethyl parathion		No		
ETU		No		
Guthion		No		
Heptachlor		No		
Heptachlor epoxide		No		
Kelthane (Dicofol)		No		
Lannate (Methomyl)		*		Traces found up to 9 ppb.
Lead		*	29	Traces detected.
Lindane		No		
Linuron (Lorox)		No		
Malathion		No		
Mancozeb		No		
Maneb		No		
Manzate (EBDC)		No		
Methyl Parathion		No		
Monitor (Methamidophos)		No		
Metasystox-R		No		
Methoxychlor	88	No		
Organochlorine screen	15	No		
Organophosphate screen	15	No		
Oxamyl (Vydate)	3945	Yes*	40	Highest concentration 18 ppb.
Paraquat	14	*		Traces found in one well; re-sample negative.

Continued ...

Table 8.4 continued ....

Compound	Number of samples analysed	Detected?	Total number of detects	Remarks
Parathion	3	No		
Permethrin (pounce, Ambush)	29	No		
Penncap-M (encapsulated methyl parathion)	3	No		
Picloram	11	*	1	Trace found at minimum detection level.
Simazine (Princep)	14	No		
Thiodan I and II (Endosulfan)	29	No		
Toxaphene	88			
Telone (D-D)	32	No		
Vapam (Metam-Sodium)	31	*		1,2-Dichloropropane detected.
Vorlex (methyl isothiocynate)	34	No		
Zineb	28	No	±500	Highest concentration ration 300 ppb.
1,2-Dichloropropane	±1 000	Yes*		
2,4-D	88	No		
2,4,5-Tp	88	No		

Source: Dennis Moran Drinking Water Section, Suffolk County Department of Health Services. Values given are on December 1984.

Note: 1 ppb = 1 microgram/L

8.5.5 Wisconsin, USA (Holden, 1986)

Wisconsin agencies identified those pesticides most likely to reach groundwater. A field sampling programme which gave priority to the first 22 chemicals on this list began in July 1983. The results of this ongoing programme for the period July 1983 to December 1984 are presented in Table 8.5.

Table 8.5: Summary of groundwater pesticide monitoring for 7/1/83 through 12/31/84 State of Wisconsin

Chemical name	Trace name	Number of samples	Number of detects	Action level/micro-grams per litre	Number exceeding action level	Highest reported concentration (ppb[a])
Alachlor	(Lasso)	337	47	10	21	88[a]
Metolachlor	(Dual)	126	34	25	17	55[a]
Aldicarb, total[b]	(Temmik)	2293	340	10	70	111
Dinoseb	(Dinitro)	125	3	12,5	0	5,5
Atrazine	(Aatrex)	349	50	215	0	140[a]
Butylate	(Sutan)	38	14	None	0	2,9
Eptam	(Eptam)	100	9	None	0	
Cyanazine	(Bladex)	117	8	60	0	
Carbofuran	(Furadan)	78	2	5	0	7
Chloramben	(Amiben)	29	2	10	0	50
DCPA	(Dacthal)	10	1	300	0	
Metribuzin	(Sencor)	104	1	25	0	1,8
Carbaryl	(Sevin)	121	0	10	0	
Chlorpyrifos	(Lorsban)	25	0	10	0	
Terbufos	(Counter)	78	0	None	0	
2,4-D	(2,4-D)	34	0	100	0	
Diazinon	(Diazinon)	5	0	20	0	
Dicamba	(Banvel)	22	0	12,5	0	
Dimethoate	(Dimethdate)	10	0	10	0	
Disulfoton	(Disyston)	100	0	1	0	
Linuron	(Lorox)	82	0	62,5	0	
Methomyl	(Lannate)	13	0	10	0	
Pentachloro-nitrobenzene	(PCNB)	4	0	10	0	
Pciloram	(Tordon)	2	0	10	0	
Phorate	(Thimet)	84	0	1	0	
Dichlorvos - (DDVP)	(Vapona)	1	0	None	0	
TOTAL		4379	511			

Note: The total number of samples and detections do not directly correspond to the numbers of wells where pesticides were detected in the groundwater. Re-sampling of some wells has occurred, especially those with previous detections (e.g., Aldicarb), consequently a number of the detections may come from samples taken at the same well.

[A] Includes point and non-point sources; important distinction for Alachlor, Metolachlor and Atrazine, which have been detected primarily at trace levels except for the point source in Rusk, Wisconsin.

[B] Aldicarb data from Union Carbide Summary of Wisconsin sampling activity dated January 23, 1985.

8.5.6 Florida, USA (Holden, 1986)

Various monitoring programmes for aldicarb showed that areas with soils having a high pH from calcareous sands produced no contamination problems while up to 600 ppb aldicarb was found in sandy acid soil areas. EDB testing of groundwater from 7609 wells showed that 828 (11%) had EDB present.

The US Geological Survey sampled 96 public water supply systems. Eight of these systems had detectable levels of organics and upon resampling 4 systems were confirmed to be contaminated, of which 2 involved pesticides: Orlando, 0,02-ppb silvex; and Clearwater, 0,01-ppb lindane.

8.5.7 Kansas, USA (Steichen *et al.*, 1988)

The high cost (about $500 per water sample) of laboratory analyses limited the number of wells tested to about 100. Two farm wells were selected at random from each of the 48 counties plus additional wells in those counties with a higher well density.

VOCs, pesticides and inorganic chemicals were analysed for. The pesticides analysed for were:

Alachlor
Aldrin
Altrazine
Chlordane
DCPA
Dieldrin
Metolachlor
Endrin
Heptachlor epoxide
Lindane
Methoxychlor
O,P'DDT
P,P'DDT
Propachlor
Metribuzin
Silvex
Tordon
Toxaphene
2,4,5-T
2,4-D

Eight of the total 103 wells sampled had detectable levels of pesticides. The pesticides identified are detailed in Table 8.6.

Table 8.6: Contaminants Found in Farmstead Wells - Kansas

Chemical	No of wells	Concentration (micrograms/L)		
		Initial	Re-sample	MCL or KAL[b]
Atrazine	4	high = 7,4	High = 40	150 (KAL)
2,4-D[d]	1	1,3	d	100 (MCL)
2,4,5-T[c]	1	1,1	d	700 (KAL)
Tordo	1	5,6	3,3	175 (KAL)
Chlordane[e]	1	0,47	0,58	0,22 (KAL)
Heptachlor epoxide[e]	1	0,26	0,023	0,006 (KAL)
Alachlor[f]	1	0,88	1,8	15 (KAL)

[b] MCL is the maximum contaminant level established by the National primary Water Regulations (US EPA 1987b). Kansas Action Level (KAL) is the level at which KDHE considers the water unacceptable for long-term consumption. The KAL is a guideline for those chemicals without MCLs and is not a regulatory standard.

[c] 2,4-D and 2,4,5-T were found in the same well.

[d] This well could not be re-sampled because the pump has failed and was no longer in use.

[e] Chlordane and heptachlor epoxide were found in the same well.

[f] Alachlor and 1,2-dichloroethane were found in the same well.

8.5.8 Nebraska, USA (Spalding *et al.*, 1989)

Almost all of Nebraska's 330 000 rural households rely on groundwater which provides 90% of the state's drinking water. Each county to be sampled was assigned to one of three strata levels which were based on soil type and land use. For each strata level a computer-generated random sampling list was generated. 2263 well water samples were collected and analysed for pesticides. Thirteen and one-half percent (305 wells) contained detectable levels of atrazine, but only 22 wells exceeded the health advisory of 3,0 ppb. The highest frequency of detections occurred in irrigated corn-growing areas with less than 50 feet to groundwater. Cyanazine, together with the additional triazines -simazine, propazine, prometone and ametryne, also were detected in some well water; however, their frequency of detection was well below that of atrazine. The triazine metribuzin was not detected.

Alachlor, propachlor, and metolachlor also were detected in trace levels in several wells. Five of 2072 samples analysed for alachlor exceeded the health advisory of 0,4 ppb. Almost all fo the contaminated wells were in vulnerable areas. The relatively high frequency of propachlor detections occurred in predominately irrigated corn-growing areas, rather than in areas where propachlor is traditionally applied.

The factors that appear most directly involved in the observed distribution of pesticides in groundwater are the intensity of areal usage, pesticide persistence and mobility, irrigation, soil drainage capacity and depth to groundwater.

Fifteen pesticide residues were detected during this study. If ethylene dibromide and carbon tetrachloride, which were detected in groundwater adjacent to grain elevators are included, a total of 17 pesticide residues were detected in Nebraska's groundwater.

8.5.9 Hawaii, USA (Oki and Giambelluca, 1988)

Thirteen public drinking water wells have been found to be contaminated by EDB, DBCP, and/or trichloropropanes. These wells serve more than 130 000 people.

1,2-dibromo-3-chloropropane (DBCP) and 1,2-dibromoethane or ethylene dibromide (EDB) are soil fumigants previously used by pineapple growers. 1,2,3-trichloropropane (TCP) is an impurity of the soil fumigant D-D and was also detected.

DBCP and TCP contamination of groundwater is derived from non-point and/or point sources associated with nematicide used in pineapple cultivation. EDB may have entered the groundwater as a result of the use and handling of the nematicide by pineapple plantations. Due to the presence of fuel pipeline leaks in the vicinity of the contamination, however, the source of EDB contamination cannot be identified with certainty.

8.5.10 Pennsylvania, USA (Pionke and Glotfelty, 1989)

Waters from 21 wells and two springs located in a typically farmed, mostly agricultural Pennsylvania watershed were analysed for 11 pesticides, NO_3 and Cl and PO_4. Among the pesticides selected according to a farm use survey, only atrazine was commonly found but at extremely low concentrations ranging from 3 nanograms/L to 1,11 micrograms/L.

Atrazine appeared in 74% of cropland wells at least once in a three sampling sequence designed to sample several major groundwater recharge periods. No alachlor, metolachlor, carbofuran, terbufos, chlorpyrifos nor fonofos was found. Cyanazine was found intermittently in two wells. With only one record of application, simazine was found at least once in 32% of the cropland wells, but at extremely low concentrations. Neither 2,4-D nor dicamba were found in the one sampling analysed. The frequency and extent of corn production expressed as Corn Production Intensity (CPI) was strongly related to atrazine concentrations. The higher atrazine concentrations were nearly always associated with continuous corn production. Atrazine concentrations over time varied greatly by well and followed several distinct patterns. Sampling time relative to major groundwater recharge periods affected these concentrations, but a predictive relationship was not apparent.

8.5.11 Arkansas, USA (Cavalier *et al.*, 1989)

During 1985 to 1987, 119 wells, springs and municipal drinking water supplies throughout Arkansas were monitored for the presence of pesticides. Pesticides selected for analysis included acifluorfen, alachlor, atrazine, benomyl, cyanazine, cypermethrin, 2,4-D, dichlorprop, diuron, fenvalerate, fluometuron, hexazinone, linuron, metolachlor, permethrin, picloram, and propanil. Not every sample was analysed for every pesticide. Overall, results indicated that the 18 herbicides, fungicides and insecticides were not present in the groundwater samples studied. (Note: Detectable concentrations of three herbicides - alachlor, atrazine and metolachlor - were found in one irrigation well at 5,5; 5,8 and 6,9 ug/L, respectively.) However, since previous and subsequent sampling failed to detect these compounds, their presence is attributed to a localized spill or handling error rather than agricultural application.

8.5.12 Iowa, USA (EPA, 1987a)

Nine herbicides and two insecticides were detected in monitoring studies conducted in Iowa. For the most part, concentrations were less than one ppb. The major source of these pesticides were attributed to normal agricultural application. Monitoring data indicate that about 27% of the population consumes drinking water which contains low concentrations of pesticides.

8.5.13 Minnesota, USA (Klaseus *et al.*, 1988)

Two parallel but separate surveys were carried out on private and public drinking water wells. 100 Private wells and 400 public wells were sampled. The pesticides analysed for are listed below. The pesticides which were detected are listed in Table 8.7.

Alachlor
Atrazine
Butylate
Chloramben
Metolachlor
Cyanazine
Dicamba
EPTC
Linuron
MCPA

Metribuzin
Picloram
Propachlor
Simazine
Trifluralin
2,4-D
2,4,5-T
2,4,5-TP

Aldicarb
Aldicarb Sulfone
Aldicarb Sulfoxide
Carbaryl
Carbofuran
3-OH Carbofuran
Chlorpyrifos
Disulfoton
Fonofos

Methyl Parthian
Phorate
Phosphamidon
Terbufos
Pentachlorophenol
PCNB

Table 8.7: Occurrence of pesticides in Minnesota public and private wells

Pesticide	Wells with detections*	Median (micrograms/L)	Range (micrograms/L)
Atrazine	154	0,12	0,01 - 42,40
Alachlor	16	0,39	0,07 - 4,03
2,4-D	7	0,22	0,07 - 5,70
Metribuzin	6	0,32	0,10 - 1,05
Dicamba	4	0,17	0,05 - 0,86
Cyanazine	4	0,51	0,18 - 2,90
Picloram	3	0,16	0,08 - 0,63
Pentachlorophenol	3	0,58	0,42 - 0,64
Metolachlor	2	0,42	0,30 - 0,55
Propachlor	2	0,35	0,20 - 0,50
MCPA	2	0,26	0,13 - 2,20
Aldicarb	2	9,0	0,50 - 30,60
Simazine	1	1,40	n.a.
2,4,5-T	1	0,21	n.a.
EPTC	1	0,33	n.a.

* One or more pesticides were detected in 165 (33%) of 500 sample wells.

8.5.14 The National Pesticide Survey of the USEPA, USA (USEPA, 1990)

The EPA completed a national survey for pesticides and nitrates in drinking water wells. Also called the National Pesticide Survey (NPS), this was a stratified random survey that cost approximately $12 million. Planning for the survey began in 1984. The first sample was collected in February 1988 and the final sample was collected in February 1990.

The survey was designed to yield results that are statistically representative of all rural domestic wells and all community water system (CWS) wells in the United States. The survey had two goals. Listed in order of importance they were:

(i) To determine the frequency and concentration of the occurrence of pesticides and nitrates in drinking water wells nationally;

(ii) If sufficient positives are detected, related the results to patterns of agronomic practices and hydrogeology.

Samples were collated from 556 CWS wells and 783 private wells. There are approximately 94600 CWS wells and 10 500 000 rural domestic wells in the United States. Samples from 47 wells were rejected due to quality control problems, leaving usable results from 1292 wells. Analyses were performed for 101 pesticides, 25 pesticides transformation products, and nitrate - a total of 127 analyses. These analyses required the use of eight analytical methods.

Twelve of the 126 pesticides and pesticide metabolites were found in one or more wells at or above the minimum reporting levels of the survey.

The diacid metabolite of the herbicide Dacthal (DCPA; dimethyltetrachlorophthalate) was the most commonly found pesticide analyte, ahead of atrazine. The dacthal metabolite and atrazine hit rates were 6,4% and 1,7% respectively, in private wells, yet the use value of atrazine is probably one order of magnitude greater. Despite being widely detected it is noted that the maximum measured concentration of DCPA metribolite was 7,3 mg.l^{-1} which is well below the health advisory limits which is 4000 mg.l^{-1}. The Health Advisory Level (HAL) is the concentration of a contaminant in water that may be consumed over a person's lifetime without harmful effects. HALs are non-enforceable health-based guidelines that consider only non-cancer toxic effects.

The pesticides with concentrations exceeding HALs were alachlor, atrazine, 1,2-dibromo-3-chloropropane (DBCP), ethylene dibromide (EDB), and gamma-hexachlorocyclohexane (lindane). Other pesticides or metabolites detected were prometon, bentazone, hexachlorobenzene, dinoseb, simazine and ethylene thiourea (ETU). The latter compound is a common metabolite of the widely used EBDC fungicides. The USEPA concluded that 0,8% of the community water supply wells and 0,6% of the rural domestic wells contained at least one pesticide above the HAL. This may not seem many, but if it is your well that is contaminated then it means that your well is 100% contaminated.

8.6 PESTICIDES IN THE ROOTING ZONE OF ANNUAL CROPS, SOUTH AFRICAN CASE STUDIES - *(Charlie Reinhardt)*

The authors are currently (1996) not aware of any case studies done in South Africa other than those presented here. All these investigations involve herbicides. Herbicides are pesticides that are most often detected in groundwater, probably not because they move that much faster down the soil profile than other pesticides, but likely due to the difference in amounts applied. In 1964, in the USA, 32.3 million kilograms of herbicide active ingredients were consumed; this represented 31% of the total amount of pesticides used (Ellis, 1992). Insecticides were the biggest component of pesticides used in 1964, at 52%. Within the next seven years herbicide use had surpassed that of insecticides to reach 82% of total pesticide usage in 1982 (Osteen & Symedra, 1989). Only about 10% of cropland in the USA was treated with herbicides in 1952, but the share jumped to 90-95% by 1980. Today, the trend in the relative usage of pesticides remains, but the total amounts applied had decreased by 1990 (Agricultural chemical usage: Field crops summary, 1990). There are probably three reasons for this trend:

(i) Total hectares planted to major crops is lower.
(ii) Growers are using lower rates.
(iii) New herbicides are more active, and therefore, less active ingredient need be used.

In the light of these developments it may be expected that lower, and even none, residues of herbicides would be detected in groundwater. Unfortunately, this ideal has not yet been reached to the extent hoped for. In Italy, for example, residues of atrazine were detected in well water six years after its use had been banned (Russo, Fava, Achilli & Sassi, 1993). Although atrazine was detected, the trend was progressively lower residual levels over time.

The depth to which herbicides are incorporated in the soil seldom exceeds 15 cm, since the majority of weed seeds germinate in this upper zone. This is also the zone that supports the most rapid dissipation processes (Lavy, Roeth & Fenster, 1973; Moorman & Harper, 1989 and Laird, Yen, Koskinan & Dowdy, 1994). Scrutiny of the mobility characterisitics of several of the commonly used pesticides indicates that under certain soil-moisture regimes some have the potential to move below the upper soil horizons. Evidence of pesticides moving to subsoil depths has been presented by many researchers, including Burnside, Fenster & Wicks (1963), Cohen, Eoden & Lorber (1986), Hall *et al.* (1989), Buhler, Randall, Koskinen & Wyse (1993), Southwick, Willis, Johnson & Selim (1995), Senseman, Lavy, Mattice & Daniel (1996). Factors facilitating downward movement of pesticides were discussed in sections above. Regardless of how a pesticide reaches a lower zone, persistence is likely to increase in those zones, since conditions there are conducive to decreased dissipation. Some of these conditions are reduced microbial activity, decreased uptake by plant roots, and different soil characteristics that affect degradation, adsorption and leaching.

8.6.1 Atrazine and terbuthylazine leaching in 23 soils under controlled conditions (Extracted from Hugo, 1994; Reinhardt & Hugo, 1997)

In contrast to the large knowledge base for atrazine, there is no literature available on the behaviour of terbuthylazine in local soils. In this study selected soil properties were correlated with leaching distances for atrazine and terbuthylazine on soil thick-layer chromatograms, in order to establish the relative importance of the various soil characteristics on the mobility of the herbicides. A further aim was to develop simple models that could be used to roughly rank soils according to their leaching indices for atrazine and terbuthylazine.

Soil chromatograms were made up in the laboratory from 23 soils collected across the main maize producing region of South Africa. Leaching distances for atrazine and terbuthylazine are presented in Table 8.8. The mobility of terbuthylazine was less than atrazine in corresponding soils. In the *Sandfontein*, Viljoenskroon and Wesselsbron soils where atrazine leached 22 cm, 20 cm and 20 cm, respectively, terbuthylazine leached 16, 6 and 6 cm. Atrazine leached at least 12 cm in the Ventersdorp, Eerstegeluk, and Standerton soils, but terbuthylazine only 2, 4 and 6 cm, respectively. Except for the Standerton soil, the % C of all the above-mentioned soils was less than 0.5% and the clay content less than 15%. The Standerton soil had a clay content of 56%, but a relatively low P-reversion value of 32 mg P kg-*l*. P-reversion of the other soils in this group varied from 90-105 mg P kg^{-1}. The low organic matter (0.03-0.28% C), coarse-textured (4-11 % clay) Sandfontein, Viljoenskroon and Wesselsbron soils were the only ones on which significant bioactivity of atrazine was detected at approximately 18 cm from the point of application. In fact, atrazine leached the furthest (22 cm) in the soil (Sandfontein) with the lowest organic matter and clay contents.

Simple correlation coefficients used to describe relationships between soil properties and leaching distance indicated that % C was the single most important soil variable affecting the leaching of atrazine (r=-0.79) and terbuthylazine (r=-0.58). Considering the negative correlation between adsorption and leachability, this finding appears to confirm the well-documented strong relationship between soil organic matter and adsorption of *s*-triazine herbicides.

Appreciable leaching (16 cm) of terbuthylazine occurred only in the Sandfontein soil (Table 8.8). This herbicide leached to beyond 4 cm in only two other soils, namely Viljoenskroon and Standerton. The latter three soils were all low in organic matter content (<0.03% C), although clay content varied from 4-56%.

The combination of % C and pH in multiple regression analysis improved the prediction of the herbicide leaching distances slightly over that provided by % C alone; for atrazine from 63 to 72%, and for terbuthylazine from 34 to 42%. The following models were developed with % C and pH for prediction of atrazine and terbuthylazine leaching:

For atrazine: $y = 27.31 - 12.04x - 1.65z$

For terbuthylazine: $y = 13.35 - 5.86x - 1.02z$

where: y = *potential leaching depth (cm)*
x = *percentage C of the soil*
z = *pH*(H_2O) *of the soil*

Results indicate that atrazine would have a higher propensity to leach in neutral or alkaline soils which are low in organic matter content. Terbuthylazine appears likely to leach appreciably only in soils that are extremely low in organic matter. It is possible that soil pH and clay content would have been more important predictors of the leaching of both herbicides had the following conditions applied: (a) evenly distributed soil pH (H_2O) across the range pH 4-8, and (b) a bigger ratio of soils containing montmorillonite. It follows that the proposed model would be best suited to cases where soil pH ranges between pH 5-7, and where the dominant clay type is kaolinite. It needs to be emphasized that many other factors not considered here influence the leaching of a particular herbicide, e.g., preferential flow, capillary movement, amount of overhead water, and numerous other factors. Preferential flow is particularly difficult to measure or predict, but is generally acknowledged as a major factor in the leaching of pesticides.

The limited number of data pairs that stems from using only 23 soils places further constraints on the utility and predictive ability of the models. The considerable limitations of the models preclude prediction of events of great complexity, such as ground water contamination. It is hoped that the models would be useful in particular cases where rapid, less meticulous identification of soils, which are prone to atrazine and terbuthylazine leaching, would assist in narrowing down soil candidates destined for detailed investigation. The approach has obvious economic advantages in terms of both time and money, but validation in the field is needed.

Table 8.8: **Leaching distances attained in soil thick-layer chromatograms with atrazine and terbuthylazine applied to 23 soils**

	Leaching distance (cm)[1,3]	
Soil	**Atrazine**	**Terbuthylazine**
Sandfontein	22	16
Viljoenskroon	18	6
Wesselsbron	18	4
Eerstegeluk	14	4
Ventersdorp	14	2[2]
Standerton	12	6
Newcastle	10	4
Middelburg A	10	4
Lichtenburg A	10	2
Glencoe A	10	4
Lichtenburg B	10	2
Middelburg B	10	4
Towoomba	10	4
Glencoe B	8	2
Lichtenburg C	8	2
Wolwehoek	8	4
Dundee	6	2
Lichtenburg D	6	2
Glencoe C	6	2
Lichtenburg E	6	2
Lichtenburg F	6	2
Lichtenburg G	6	2
Groblersdal	4	2

[1] The leaching distance of each herbicide was that soil segment (=distance) on the treated plates where significant ($P=0.05$) growth reduction of the test species was noted.

[2] A leaching distance of 2 cm implies that the herbicide did not leach from the zone it was applied to.

[3] The maximum attainable leaching distance was 30 cm (= the total length of soil chromatograms)

Table 8.9: **Terbuthylazine residues in soil samples collected during the 1991/1992 season at different field sites**

Time (days) after herbicide application	Depth	Amount of herbicide (% of total amount on day 1)					
		Bethl	Cedara	Ermelo	PotchR	PotchT	Vaalh
1	0-10 cm	99.03	98.22	99.83	99.69	99.42	98.05
30		60.34	33.30	77.27	39.94	69.31	45.55
60		39.29	13.51	61.20	30.45	52.03	8.887
90		32.08	3.182	44.99	8.052	34.33	1.089
120		4.447	2.223	5.124	3.090	5.134	0.677
150		3.115	1.241	10.26	6.372	7.355	1.280
180		7.457	2.225	5.207	0.861	3.280	0.373
1	10-20 cm	0.732	1.491	0.050	0.194	0.462	1.836
30		0.395	0.516	0.850	1.332	1.019	1.357
60		0.236	0.114	1.062	1.447	2.470	0.916
90		0.091	0.263	0.850	0.067	0.044	0.184
120		0.125	0.108	0.103	0.076	0.086	ND*
150		0.130	ND*	0.187	ND*	0.068	ND*
180		0.130	0.038	0.074	ND*	0.024	ND*
1	20-30 cm	0.236	0.285	0.125	0.116	0.121	0.109
30		0.265	0.230	0.089	0.200	0.114	0.108
60		0.181	0.097	0.124	0.135	0.050	0.268
90		0.105	0.042	0.105	0.078	0.034	0.140
120		0.138	0.237	ND*	ND*	0.024	ND*
150		0.093	0.066	0.140	ND*	0.038	ND*
180		0.042	0.057	0.105	ND*	0.026	ND*

* Residual amount of herbicide in all replicates below minimum level of detection.

8.6.2 Terbuthylazine leaching under field conditions (Extracted from Kamffer, 1997)

Terbuthylazine was applied on the surface of field plots at ARC or Dept. of Agriculture experimental farms at Bethlehem, Cedara, Ermelo, Potchefstroom and Vaalharts during 1991 and again in 1992. The rate used at each site was the same as that recommended for atrazine, since terbuthylazine is not marketed on its own but is available only in mixtures with other herbicides. Soil samples for chemical analysis were collected 24 h after herbicide application, and thereafter at approximately 30-day intervals for 180 days from each site during both seasons.

Only insignificant amounts of terbuthylazine could be detected in soil cores collected, 10-20 cm and 20-30 cm deep, from the different localities at the various sampling dates during the 1991/1992 season (Table 8.9). These amounts were probably due to contamination from the top (0-10 cm) soil layer, as their magnitude corresponded well with the amount detected in these soil layers in samples taken 24 hours after application, when leaching could not yet have contributed to movement of the herbicide to these soil layers. These results are in accordance with those (see previous section) of Hugo (1994) who detected a maximum leaching depth of 16 cm with terbuthylazine in only one of 23 soils. In only two other soils the herbicide leached beyond 4 cm.

It seems, therefore, that leaching can be disregarded as a factor affecting the dissipation of terbuthylazine. The Locality X Time interaction was significant for both seasons, indicating dissimilar dissipation patterns at the various localities. Dissipation was the slowest at the Ermelo and Potchefstroom Turf sites, followed by the Bethlehem and the Potchefstroom Red sites, with the highest dissipation rates at the Cedara and Vaalharts sites during the 1991/1992 season (Table 8.9). During the 1992/1993 season, the relative dissipation rates in plots adjacent to trials of the previous season remained approximately the same, except for the Potchefstroom Red plot which now exhibited the highest dissipation rate.

8.6.3 Metolachlor mobility under controlled and field conditions (Extracted from Van Biljon, 1991)

8.6.3.1 Metolachlor mobility in an aeolian soil

The field experiment was done on a typical drift (aeolian) sand soil in the Viljoenskroon district. Metolachlor was applied and rain subsequently simulated with the same small-plot sprayer. Soil samples in 25 mm increments were collected from the treated site up to a depth of 175 mm, four days after herbicide application. The same soil used in the field trial was also used to measure the leaching and capillary movement of metolachlor under controlled conditions in a greenhouse. The leaching experiment done in the field gave results very similar to those obtained in the greenhouse.

The results of the leaching studies in the drift sand soils showed that metolachlor readily leached in these soils. The findings suggested that only 12.5 mm rain on wet or dry soil may leach metolachlor 75 mm deep, and in the event of 25 mm rain the herbicide may leach 100-125 mm deep. It was also shown that after herbicide application followed by water application, metolachlor can, through capillary forces, remain in the surface soil layers. Prerequisites for such an event are sufficient water to facilitate capillary movement and sufficient evaporation at the soil surface. The unrestricted downward and upward movement of metolachlor, which is readily available for plant uptake in low adsorptive soils, may be a very important factor in phytotoxicity. This could be expected during periods of abundant rain when the water table in the north-western Free State may be very shallow and would promote capillary movement. Capillary movement of 500 mm is likely in these soils. A field experiment is needed to monitor metolachlor movement under such conditions.

8.6.3.2 Metolachlor leaching in 19 soils

In order to obtain a more complete picture of the leaching behaviour in other South African soils, the leaching behaviour of metolachlor in the Viljoenskroon soil was compared with that in some other soils. Soil columns 300 mm in length were made up by attaching 25 (height) x 100 (diameter) mm PVC pipe segments to each other with silicone sealer. The columns were hand-packed with soil collected in the field. Care was taken to attain the same bulk densities as those that existed at soil sampling sites in the field. Metolachlor and water were applied on the soil surface at the top of columns.

Table 8.10: **Leaching depths attained by metolachlor in 19 soils exposed to differential amounts of simulated rainfall**

Soil	Metolachlor leaching depth* (mm)		
	Simulated rain (mm)		
	12.5	25.0	50.0
Cullinan	25	50	125
Viljoenskroon	75	100	225
Vredefort	75	100	175
Towoomba	50	75	125
Lichtenburg	75	100	150
Pretoria (K1)	75	125	200
Delmas	75	75	150
Sasolburg	25	50	75
Kempton Park	50	75	125
Ventersdorp	67	125	225
Wolwehoek	25	25	50
Vereeninging	25	25	50
Geluksburg	25	50	75
Pretoria (K4)	50	75	100
Koedoeskop	50	50	75
Pretoria (Turf)	50	75	100
Roodeplaat	50	75	125
Cedara	25	25	25

* Maximum attainable depth was 300 mm (the soil column length).

The results for the nineteen soils (Table 8.10) showed that little leaching of metolachlor occurs in soils with relatively high organic matter contents due to greater adsorption. As expected, leaching depth (=herbicide mobility) was positively correlated with the amount of overhead water used to effect leaching.

Data in Table 8.11 show that percentage organic carbon (C) was the soil property which gave the best prediction of metolachlor leaching depth (r=-0,87). Percentage clay (r = -0,56) and CEC (r = -0,47) dit not correlate well with leaching depth. A good correlation (r = 0,70) was also obtained with percentage sand. This latter result, however, needs confirmation.

Table 8.11: **Linear correlation coefficients (r) indicating the relationship between soil properties and leaching depth of metolachlor.**

Soil variable	**Correlations with leaching depth**	
	r	**r^2**
% C	-0.87**	0.76
% Silt	-0.78**	0.61
Bulk density	0.75**	0.57
% Medium sand	0.71**	0.50
% Sand	0.70**	0.50
% Clay	-0.56	0.32
CEC	-0.47	0.32
% Coarse sand	0.36 ns	-
% Fine sand	0.08 ns	-

** Significant at P=0.05.
ns = Not significant

The leaching behaviour of metolachlor in this limited number of soils could be predicted by using percentage C in a simple model. The model is described by the following equation:

$$y = 77{,}93 - 67{,}99x + 1{,}89z$$

where: y = *potential leaching depth (mm) of metolachlor*
x = *percentage C content of the soil, and*
z = *amount of rainfall/irrigation (mm)*

These results confirmed the work of several researchers who found that organic matter was the most important soil property related to the leaching behaviour of metolachlor. Mobility was greater and phytotoxicity more likely on soils low in organic matter. Results from overseas are mostly based on soils with higher organic matter contents than South African soils.

The 19 soils were grouped in three percent C categories; <0.5 percent; 0.5 - 1.0 percent, and >1.0 percent. By using the leaching depth at each water level it was possible to establish a regression line for each percentage C category. It was shown that <1.0 percent C can influence leaching substantially. Since there is a good relationship between leaching and adsorption and between adsorption and bioactivity, these results confirm those of Reinhardt & Nel (1989) with respect to the bioactivity of alachlor and metolachlor, which was also strongly affected by soil organic matter content, despite the inherently low C contents in the soils used (predominantly <1 %C).

8.7 CASE STUDIES OF PESTICIDES IN SOUTH AFRICAN SURFACE WATER - *(Charlie Reinhardt)*

Although distinction is made in the following case studies between investigations on the presence of pesticides in dam water, groundwater and in the root zone of annual crops, the compounds have a common origin in that they are applied either aboveground, on the soil surface, or directly in the uppermost soil horizon. In a general sense, the risk of groundwater pollution increases if a pesticide leaches readily in the upper soil horizons. This is particularly true for those compounds that persist for a long time.

8.7.1 Atrazine in pan and dam water (Extracted from Coetzee & Vahrmeijer - undated)

Although this topic does not strictly fit the framework of this chapter, it is considered important in the broader context of understanding the full implications of pesticide usage. In fact, the consequences of the presence of pesticides in all types of water sources are similar, since potable waters and water used for irrigation in South Africa include both surface water and groundwater.

8.7.1.1 Pan and soil water

Water from a pan in the Viljoenskroon district was monitored for the presence of atrazine. Soil water from the same area was also analyzed. The groundwater level was relatively close to the soil surface due to the presence of a horizon with high clay content that restricts the movement of water. Covering this layer is wind-transported (aeolian) sand with a high water permeability. Due to the shallow water table, a hole was made with an auger and water removed with a specially designed container.

The assumptions were that adsorption of all the atrazine in the soil solution will not take place under field conditions. Thys, atrazine will be present in the water fraction of the soil. The relatively weak binding energy of atrazine is a further indication that atrazine might well be transported in soil by water. The detection in this investigation of atrazine in both drainage and run-off water confirmed these assumptions. A portion of the atrazine administered to the surface of the soil is thus removed from the soil profile. This may lead to contamination and pollution of water sources.

During the study of the mobility of atrazine where a rain simulator was used, it was determined that the majority of the administered atrazine was not removed from the soil profiles. The portion that was indeed removed from the profile of the soil of the vertisol form (60% clay), was mainly removed by run-off. The conclusion is in accordance with findings in literature. In the case of sandy soil (12% clay - avalon form), atrazine was present throughout the whole profile with the most atrazine accumulating at a depth of 100 to 200 mm. In contrast, atrazine was restricted to the upper 50 mm of the vertisol, although more water moved through the profile. If the specific surfaces with which atrazine in the soil of the vertisol and avalon forms came into contact is taken into consideration, it would appear that clay mineralogy plays an important role in the adsorption of atrazine. However, the following findings from the laboratory study where the adsorption of atrazine by the different clay minerals was characterised, indicate the opposite.

- In nature vertisol soils are saturated mainly with calcium or magnesium. In the laboratory there was a drastic decrease in the adsorption of atrazine by calcium and magnesium saturated montmorillonite.
- In the laboratory study no adsorption of atrazine occurred in potassium saturated montmorillonite if the pH of the suspension was not reduced to 7.5 at least. The pH of the natural vertisol was 8.1.

The fact that atrazine in the vertisol soil did not move as readily as in the case of the soil of the avalon form, is probably due to the fact that the latter soil contained 0.42% and the soil of the vertisol 2.96% organic material. In foregoing sections it was pointed out that organic matter content is the soil property with the greatest influence on the activity and mobility of atrazine -

the higher the organic matter content, the lower the bioactivity and mobility of atrazine.

The role of clay content on the physical restriction of water movement through a profile must however not be underestimated. The fact that more atrazine was removed from the soil of the vertisol by run-off water, while run-off and drainage water removed almost the same amount of atrazine in the case of soil of the avalon form, confirms that physical factors do play a role in the movement of water and solutes in it.

Various authors found that run-off is the most important transport mechanism of atrazine. In literature it is reported that more than two percent of the administered quantity of a herbicide can be removed by runoff within a fortnight after application. Thunderstorms occur regularly on the Highveld, thus it can be expected that runoff will play an important role in the transport of atrazine. Run-off is influenced by the physical properties of soil and the intensity of rainfall. If the infiltration rate of a soil is low as a result of the high clay content, the forming of crusts or as the result of compaction, run-off will be the most important mechanism for transport of atrazine. The role of the transport mechanism may increase in clay soil if the soil is wet and plate erosion takes place, because most of the applied atrazine will probably be restricted to the upper zones of the vertisol. A characteristic of a vertisol is the forming of cracks when the soil is dry. If water is added to the soil, atrazine will move down into the cracks. According to literature the presence of atrazine in the profile of a vertisol is associated with cracks in the profile. Cracks promote preferential flow of water. The role of preferential flow in pesticide transport was discussed in foregoing sections. If enough water is added to a cracked soil the cracks will close due to swelling, and run-off will again increase in importance. In better drained soils run-off will make an important contribution to the transportation of atrazine if the rate at which the water is administered or the intensity of rainfall exceeds the infiltration rate. The time that elapses from the application of atrazine to subsequent water administrations or rain will also play a role. The shorter the time-span between atrazine application and an intense rainfall event, the greater will be the propensity for atrazine to be leached or removed in run-off water.

The transporting of atrazine by drainage water is also an important mechanism, and confirmation of this was found in this study where atrazine was detected deep in the profile of the soil of the avalon form and in the soil water of that soil at a site near Viljoenskroon. Although electrolyte concentration did not have a direct influence on the adsorption of atrazine, the pH of the soil may rise or fall in response to fertilization, depending on the type and amount of fertilizer. Should the pH rise it will result in better protection of atrazine from chemical degradation, resulting in atrazine being available longer for transportation to water, hence increasing the risk to water reservoirs.

The presence of atrazine in the soil water and water of a pan which were examined during the study, may be explained as follows:

- The concentration of atrazine in the soil water and water in the pan correlated with the monthly rainfall - the higher the rainfall, the greater the mobility of atrazine, and therefore, also the amount reaching the pan water .

- Since the pH of the soil water and that of the water in the pan was higher than seven, and also because water drained freely in the sandy soil, atrazine moved deep into the profile where depth-protection against chemical and micro-organism degradation was possible. This type of protection of atrazine against degradation probably explains the carryover of atrazine that was detected from one season to the next.

- The drastic increase in the concentration of atrazine that was measured in the soil water at the beginning of the season was probably due to the movement of atrazine in a front.

- The decrease in the concentration of atrazine in the soil water and water of the pan was logarithmic with time. This indicates that degradation and/or dilution of atrazine occurred. Thus, most of the atrazine, as already discussed, is apparently transferred swiftly to the soil water. The atrazine is then probably spread further in the ground water by diffusion, which is a slower process than mass flow. Atrazine is mainly broken down at the surface of the soil where aeration is better and the temperature is higher. Successive addition of water with lower concentrations of atrazine causes further dilution of atrazine in the groundwater.

8.7.1.2 Dam water

The following dams in the main maize producing area of South Africa were monitored for the presence of atrazine:

- Wentzel Dam in the Harts River (Schweizer-Reneke);
- Bloemhof dam in the Vaal River (Bloemhof);
- Strydom Dam in the Vals River (Kroonstad);
- Koppies Dam in the Renoster River (Koppies);
- Loskop Dam in the Olifants River (Middelburg).

From the Vaal Dam in the Vaal River samples were taken at Deneysville and Oranjeville. Catch samples were collected monthly from fixed sites in each dam and the concentrations of atrazine in them were determined.

Atrazine was found in all the dams that were examined (Table 8.12). The presence of atrazine in the dams follows a seasonal tendency, which was to be expected, because atrazine is mainly used as a pre-germination herbicide and is sprayed at the beginning of the planting season. The highest concentration of atrazine was present in the Strydom Dam, followed by the Bloemhof Dam and the Vaal dam at Oranjeville (Table 8.12), where the concentration was the second highest. The maximum concentration of atrazine in the Wentzel Dam occurred a month later than the maximum concentration of atrazine in the other dams. This is probably due to the fact that the Wentzel Dam is situated in the northern Cape and planting time here is later than that of regions situated further east. The maximum concentration of atrazine in the different dams was mainly in January and February (Table 8.12). The exception, however, is the Vaal Dam at Deneysville and Oranjeville, where the concentration of atrazine in the water reached a maximum in November 1991 and April 1992, respectively. It was also noticeable that a drastic increase in the concentration of atrazine in the different dams was accompanied by drastic increases in the inflow of water. The transference of atrazine from one planting season to the next did not take place in any of the dams, although the pH values of the water in all the dams were alkaline. The decrease in atrazine concentration during the growth season of maize shows a logarithmic relationship with time, which indicates that degradation of atrazine took place probably through micro-organisms and algae, or by means of other chemical processes not dependent on high pH values.

The concentrations of atrazine in the water of the different dams show the same tendencies as those for soil water and pan water discussed in the previous section, namely:

- A seasonal tendency existed. The maximum concentration of atrazine occurred at the beginning of the crop (maize) growing season.
- Drastic increases in the concentration of atrazine in surface waters was directly related to increases in the inflow of water into the different dams.
- Water pH levels above seven were conducive to atrazine persistence.
- Decreases in atrazine concentration in water showed a logarithmic association with time.

These relationships indicate that the mechanisms of transport and dissipation of atrazine in the soil water, pan and dam water were probably the same.

The presence of atrazine in the water of the Bloemhof and Koppies Dams correlated with certain variables. Dilution was probably not the only reason for the decrease in the concentration of atrazine in the two dams, since atrazine is assimilated by algae and micro-organisms, which result in degradation. Chemical degradation probably did not dominate in these instances, since the water of the two dams was alkaline.

The amount of atrazine detected in the Loskop Dam represented only a small percentage of the amount of atrazine estimated to have been used in the catchment area. This finding apparently confirms that the largest portion of the applied atrazine accumulates in the soil.

The presence of atrazine in the water of dams means that atrazine is present in water which is used for domestic purposes. It is obvious that the methods used for the purification of water is not adequate to remove atrazine. Concentrations of atrazine higher than the guidelines which will be valid from 1994 were found in potable waters.

Table 8.12: **Atrazine concentrations ($\mu g.L^{-1}$) detected at regular intervals in different dams from August 1990 until February 1993.**

Date	Bloemhof Dam	Vaaldam at Deneysville	Vaaldam at Oranjeville	Koppies Dam	Loskop Dam	Wentzel Dam	Strydom Dam
Aug '90	< 0.05	< 0.05	< 0.05		0.08	< 0.05	< 0.05
Sep '90	< 0.05	< 0.05	< 0.05		< 0.05		
Oct '90	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05
Nov '90	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05
Dec ' 90	< 0.05	< 0.05	1.20	< 0.05	< 0.05	< 0.05	< 0.05
Jan '91	1.35	1.64	2.22	3.33	2.82	< 0.05	11.48
Feb '91	4.55	2.71	0.39	1.22	3.89	0.55	1.69
Mar '91	2.67	0.84	0.39	0.46	1.24	1.46	1.09
Apr '91	1.86	0.43	0.39	0.30	0.46	0.56	0.20
May '91	1.09	0.48	0.33	0.26	0.38	0.44	0.22
Jun '91	0.98	0.46	0.35	0.19	0.34	0.43	0.31
Jul '91	1.95	0.89	0.52	1.10	0.41	1.93	2.35
Aug '91	1.81		0.64	2.25	0.51	1.81	1.99
Sep '91	1.70	< 0.05	< 0.05	0.92	< 0.05	1.57	2.12
Oct '91	< 0.05	2.22	< 0.05	< 0.05	2.12	< 0.05	
Nov '91	< 0.05	0.89	6.6	2.26	2.20	< 0.05	

Continued....

Table 8.12 continued.....

Date	Bloemhof Dam	Vaaldam at Deneysville	Vaaldam at Oranjeville	Koppies Dam	Loskop Dam	Wentzel Dam	Strydom Dam
Dec '91	2.13	3.57	3.22	1.71	3.86	1.57	
Jan '92	2.90	4.73	< 0.05	2.56		3.81	
Feb '92	6.18	< 0.05	< 0.05	4.82		3.61	
Mar '92	3.17	1.49	< 0.05	2.72	< 0.05	3.75	
Apr '92		4.77		0.86	< 0.05		
May '92		1.15		< 0.05	< 0.05		
Jun '92		< 0.05		< 0.05	< 0.05		
Jul '92		0.07		< 0.05	< 0.05		
Aug '92		< 0.05		< 0.05	< 0.05		
Sep '92		< 0.05		< 0.05	< 0.05		
Oct '92		< 0.05		< 0.05	< 0.05		
Nov '92		0.41		0.98	0.53		
Des '92		0.55			0.56		
Jan '93		1.87			0.47		
Feb '93				0.30	0.43		

8.8 MONITORING FOR CHLORO-TRIAZINE RESIDUES IN SOUTH AFRICA - *(J J van Biljon)*

8.8.1 Introduction

During the 1980s the presence of pesticide residues in groundwater became an issue in Europe and the USA. The presence of triazine herbicides and particularly atrazine became a matter of concern for people in Europe especially. As a pro-active measure, Ciba-Geigy (now Novartis), as the largest manufacturer of triazine herbicides, implemented use-reduction programmes and started water monitoring studies at various localities.

As large quantities of atrazine and terbuthylazine are used for weed control in maize in South Africa, it was decided to evaluate the situation in this country too. Surface water, groundwater and tap water were monitored for residues of atrazine and terbuthylazine from 1989 onwards.

8.8.2 Water Monitoring Sites

Surface water
Samples were taken from the Bronkhorstspruit dam, Vaal River (near Orkney), Bloemhof dam and Hartbeespoort dam.

The catchment area of the Bronkhorstspruit river which flows into the Bronkhorstspruit dam covers the intensive maize growing region of Delmas. The Vaal River drains the largest part of the maize growing area of the Highveld and eventually flows into the Bloemhof dam. The Hartbeespoort dam drains the industrial area of Gauteng where hardly any maize is grown.

Groundwater
Boreholes were randomly selected on farms in the Bethlehem, Lichtenburg, and Kroonstad districts while the Bethal borehole was on a farm where it is known that the farmer fills his herbicide sprayers close to the borehole.

Tap water
Samples of household water were collected in the towns of Bethlehem, Lichtenburg, Kroonstad and Bethal. These towns are all situated in the maize growing area of the country and use surface water for household purposes.

8.8.3 Results

Surface water

The results of the surface water monitoring programme are given in Table 8.13 for atrazine and Table 8.14 for terbuthylazine :

Table 8.13: Residues of atrazine in surface water in South Africa ($\mu g.L^{-1}$)

Sampling dates	Bronkhorst-spruit Dam (Maize area)	Vaal River (Maize area)	Bloemhof Dam (Maize area)	Hartbeespoort Dam (Industrial area)
January 1989	1,0	1,3	0,7	0,8
June 1989	1,0	1,5	0,7	0,3
March 1990	0,9	0,1	0,44	0,2
August 1990	0,4	1,6	0,3	0,1
September 1994	0,45	2,09	1,07	0,86
September 1995	0,1	-	0,05	<0,05

Table 8.14: Residues of terbuthylazine in surface water in South Africa ($\mu g.L^{-1}$)

Sampling dates	Bronkhorst-spruit Dam (Maize area)	Vaal River (Maize area)	Bloemhof Dam (Maize area)	Hartbeespoort Dam (Industrial area)
January 1989	2,0	0,8	0,4	0,6
June 1989	0,3	0,9	0,4	0,1
March 1990	0,3	<0,05	0,05	<0,1
August 1990	0,2	<0,1	<0,1	<0,1
September 1994	0,19	0,06	0,14	0,26
September 1995	0,01	-	0,05	<0,05

Groundwater

The results of the ground water monitoring programme are given in Table 8.15 for atrazine and Table 8.16 for terbuthylazine:

Table 8.15: Residues of atrazine in ground water in South Africa from various boreholes (μg.L^{-1})

Sampling dates	Bethlehem	Kroonstad	Bethal	Lichtenburg
August 1992	<0,05	0,33	1.15	<0,05
November 1992	<0,05	0,25	0,92	<0,05
January 1993	<0,05	0,21	0,07	<0,05
March 1993	0,46	0,30	0,15	<0,05
June 1993	<0,05	0,33	<0,05	<0,05
September 1993	<0,05	0,08	<0,05	<0,05
December 1993	<0,05	0,30	0,56	<0,05
July 1994	<0,05	0,30	1,00	<0,05
October 1994	<0,05	0,16	0,95	<0,05
September 1995	<0,05	<0,05	0,06	<0,05

Table 8.16: Residues of terbuthylazine in ground water in South Africa from various boreholes (μg.L^{-1})

Sampling dates	Bethlehem	Kroonstad	Bethal	Lichtenburg
August 1992	<0,05	0,13	0,61	<0,05
November 1992	<0,05	0,10	0,41	<0,05
January 1993	<0,05	0,07	0,05	<0,05
March 1993	0,15	<0,05	<0,05	<0,05
June 1993	<0,05	0,06	<0,05	<0,05
September 1993	<0,05	0,26	<0,05	<0,05
December 1993	<0,05	0,06	0,48	<0,05
July 1994	<0,05	0,10	0,85	<0,05
October 1994	0,05	0,09	0,60	0,05
September 1995	<0,05	<0,05	0,06	<0,05

Tap water

The results of the tap water (household water) monitoring programme are given in Table 8.17 for atrazine and Table 8.18 for terbuthylazine :

Table 8.17: Residues of atrazine in tap water of various South African towns ($\mu g.L^{-1}$)

Sampling dates	Bethlehem	Lichtenburg	Kroonstad	Bethal
August 1992	0,23	<0,05	0,34	-
November 1992	0,36	<0,05	0,66	0,44
January 1993	0,52	<0,05	-	0,22
March 1993	0,47	0,05	0,33	0,53
June 1993	0,63	<0,05	0,39	0,55
September1993	0,10	<0,05	0,10	<0,05
December 1993	0,06	<0,05	0,34	0,53
July 1994	0,30	<0,05	0,40	0,50
October 1994	0,52	<0,05	0,40	0,50
October 1995	0,19	<0,05	0,59	-

Table 8.18: Residues of terbuthylazine in tap water in various South African towns ($\mu g.L^{-1}$)

Sampling dates	Bethlehem	Lichtenburg	Kroonstad	Bethal
August 1992	0,05	<0,05	0,05	-
November 1992	0,09	<0,05	0,12	0,12
January 1993	0,18	<0,05	-	0,06
March 1993	0,15	0,05	0,09	0,13
June 1993	<0,05	<0,05	0,11	0,13
September 1993	0,42	<0,05	0,51	0,13
December 1993	0,28	<0,05	0,06	0,32
July 1994	0,15	<0,05	0,21	0,13
October 1994	0,14	<0,05	<0,05	0,09
October 1995	<0,05	<0,05	0,05	-

8.8.4 Interpretation of the Water Monitoring Results

In general the water monitoring programme clearly shows that much higher concentrations of triazines occur in surface water than in ground water. The tap water contains similar concentrations to the surface water which is not surprising since this water is obtained from surface water sources.

Reasons for higher concentrations in rivers and dams could be: Direct discharge from production and formulation sites, surface run-off and erosion from treated areas, drift at application or careless disposal of spray rests and containers.

Due to the short period of observation no increasing or decreasing trends can be determined.

8.8.5 Ecotoxicological Risk Assessment

All the available ecotoxicological data on atrazine produced by Ciba-Geigy as well as by independent scientists, has been reviewed recently by Prof. W. Huber (Technical Univeristy of Munich, Germany). He concludes that atrazine causes no irreversible damage to aquatic life as long as the concentration does not exceed the limit of 20 $\mu g.L^{-1}$ for a prolonged period of time. The ecotoxicity of terbuthylazine is similar to that of atrazine.

Based on this data it can be assumed that the concentrations found in the monitored surface waters do not present a danger to the aquatic life.

8.8.6 Toxicological Risk Assessment

Based on the product-specific toxicological data the World Health Organisation (WHO) proposes limit values for pesticide residues in drinking water. For atrazine, the WHO proposed a limit value of 2 micrograms per litre of drinking water. This means that a person could drink 2 litres of water per day containing 2 micrograms of atrazine per litre for the entire life and without health risk.

The Australian Health Authorities who have recently re-evaluated the toxicological data of atrazine even set the limit for residues of atrazine in drinking water at 15 $\mu g.L^{-1}$.

For terbuthylazine no WHO evaluation exists, but based on Ciba-Geigy's data base it can be concluded that the toxicity of terbuthylazine is similar to that of atrazine.

Based on this data, it can be assumed that the concentrations found do not present any danger to human health.

8.8.7 Conclusion

In water monitoring programmes, residues of atrazine and terbuthylazine were detected in ground, surface, and tap water. However, the concentrations found are within safe limits, and do not present a danger to human or aquatic life. Specific use-reduction programmes have been designed by former Ciba-Geigy (now Novartis) with the goal to reduce the level of residues in water. A strict application of the measures proposed (see : J J van Biljon; Good Farming Practice for Chloro-triazine Herbicides) will contribute to an improvement of the situation. Any progress in this field requires the commitment of all partners involved, industry, authorities and farmers.

8.8.8 Good Farming Practice for Chloro-Triazine Herbicides - J J van Biljon

➢ Introduction

The presence of traces of mainly atrazine in ground, surface and drinking water, and the public concern associated herewith has led Ciba-Geigy (now Novartis) to implement several measures which are expected to contribute to a substantial reduction of residues in water. The measures taken are based on the assessment of the potential causes for the presence of residues. In Europe these measures are called the 'Good Farming Practice Programme' (GFPP) and in the USA 'Best Management Practices' (BMP). The GFPP for atrazine was first implemented in Switzerland. Based on the success of this programme, Ciba-Geigy decided to apply this philosophy to all chloro-triazine herbicides and to all countries. The goal is to contribute to a substantial reduction of residues in water and to retain these herbicides for uses with major benefit to the farmer and minimal risk to the environment.

The changes in use patterns after implementing the GFPP in Europe, resulted in a clear reduction of residue concentrations in ground, surface and drinking water. (A. Seiler, F. Mülebach : Good farming practices to reduce residues of atrazine in ground and surface water;

1995, BCPC Monograph No 62 : Pesticide movement to water; 389 - 396. These positive developments are essential for the continued support of the registration of chloro-triazines in Europe.

Similar use-adaptations were also implemented in South Africa in 1995 through the Agricultural and Veterinary Chemical Association of South Africa (AVCASA) in collaboration with the Registrar of Agricultural Remedies and Fertilisers (Act 36 of 1947). Certain uses have been eliminated and specific recommendations have been included on the chloro-triazines product labels. Although these new recommendations are based on an initiative of the former Ciba-Geigy, they are now binding for all suppliers and users of atrazine-containing herbicides.

- **Uses no longer permitted**

The following uses and rates were eliminated, because they were recognised to contribute the most to residues in ground and surface water :

- *Industrial uses :* Registration of atrazine for industrial uses has been withdrawn. The use of atrazine-containing herbicides in these areas is therefore no longer permitted.

- *High use rates :* The maximum use rate of atrazine or total chloro-triazines in maize, sorghum and sugarcane is limited to 2,5 kg a.i./ha and year.

- **Label recommendations**

In rural areas, empty crop protection product containers are still often washed in dams, rivers and streams and reused for other purposes. This is why the following recommendations are given to support the careful handling of the spray solution and in order to protect ground and surface water:

- *Reduction of ground water contamination:*
  No mixing of filling within 15 m of any borehole.
  Prevent back-siphoning to the water source when herbicides are applied through the irrigation system.

- *Reduction of surface water contamination :*
  No mixing or filling within 15 m of rivers, dams or other water bodies.
  No application within 20 m of the points where field run-off enters streams or rivers.
  No application within 60 m of dams and lakes.
  Proper rinsing of equipment and disposal of rinsate.

It would obviously be beneficial for the protection of the environment if similar recommendations were developed for other products as well.

- **Additional recommendations**

The following measures contribute further to the protection of ground and surface water. These measures are not specific for atrazine-containing herbicides, but they may apply to all pesticides used.

- Promote cropping practices to improve erosion control such as reduced tillage, contour farming, buffer strips and waterways.
- Minimise wastes by calculating the correct amount of spray solution needed.
- Match application rates to soil characteristics
- Reduce spray drift to a minimum.
- Proper disposal of containers.

In addition, the chemical industry is actively pursuing new innovative formulation and packaging concepts in order to minimise point source contamination.

Water monitoring programmes will reveal if these use-reductions contribute to a clear reduction of residues in water (see J J van Biljon : Water monitoring for chloro-triazine residues).

8.9 CONCLUSIONS AND GUIDELINES FOR FUTURE STUDIES

Investigations of pesticide impact on groundwater in South Africa have been random shots in the dark, rather than following a well developed plan of investigation. This is similar to the early investigations in the USA prior to the establishment of the National Pesticide Survey. A contributing factor which reduces collaboration is the great sensitivity of the pesticide industry towards publicity and the consequent desire to keep investigations as quiet as possible. An example of this non-encouragement of collaboration of investigation occurred when Weaver (1993) and Reinhardt (Hugo (1994) with C R Reinhardt as supervisor) were both investigating the occurrence of atrazine in the soil horizon at the same time yet never knew about each others work until well after completion. For any other scientific investigation, in the small community of South African researchers, two researchers in the same field would have had knowledge of each others work, if not collaboration.

The National Pesticide Survey in the USA (EPA 1990) has been an example of a well-planned and statistically valid national exercise which has placed the impact of pesticides in groundwater into its proper context. Following on from this project a strategy for further research is in place.

However, to apply such an exercise in South Africa will probably not be possible due to lack of manpower and also the high monetary cost. The alternative is to learn and adopt from the work that is being done and to apply this in South Africa. Prediction of the fate of pesticides in the subsurface by means of computer-models is today well advanced from the rudimentary models of 10 years ago. To use these models for prediction purposes in South Africa should be possible. Thus if a problem is suspected in a specific area then the recommended first step would be to carry out modelling of the site. If modelling shows that there is a probable real impact then site investigations should be carried out.

The main differences between South African conditions and those of Europe and North America where most of the investigations have occurred are:

- higher ambient temperatures in SA which speeds up pesticide degradation
- lower rainfall in SA decreasing recharge
- higher clay content of soil increasing retardation
- often deeper water-tables which increases travel time
- lower soil carbon content which decreases retention

8.9.1 Legislation

Implementation of South Africa's pollution control and waste management legislation is imminent. The aim of the National Policy on Integrated Pollution Control (IPC) for South Africa was to develop a holistic approach to pollution control, i.e. to integrate regulations on water, air, soil (land) poisoning and waste into a cohesive and workable form. The stated objectives of IPC are to:

- "ensure the attainment or maintenance of ambient standards which are conducive to good human health and safety and which allow for sustainable ecosystems maintenance;
- consider discharges to all environmental media (air, water and land) in the context of their effect on the environment as a whole;
- prevent or minimise pollutants and render harmless any pollutants which may be released;
- ensure that the costs of pollution are carried as directly as possible by the polluter".

It is against this background that the information provided in this chapter should be interpreted.

8.9.2 Regulation

Pesticides are highly regulated. To quote Riley (1995): "Regulations give guidance on data required, methods for assessing risk and how to decide if a risk is acceptable. They include defining how a pesticide can be used, not just an "approval" of the active ingredient. Regulations also give guidance on how information should be communicated to the user via the product label. They also provide a legal framework for enforcing the proper use of pesticides".

Earlier Tooby & Marsden (1991) remarked that the lack of understanding of the mechanisms involved in environmental fate and behaviour of pesticides in the past has prevented good interpretation of data for regulatory purposes. They stated that the development of risk assessment procedures and fundamental research that lends support have often not been co-ordinated in terms of generation of data that is directly relevant to that required for registration purposes. Another apparent constraint was that the principles involved in environmental fate and behaviour were not universally accepted by regulatory authorities, environmental groups or the agrochemical industry. Although total concensus is still elusive there has been progress in recent years in terms of risk assessment and decision-making processes. Progress has been made in terms of field monitoring techniques (Carter & Fogg, 1995), simulation modelling (Jarvis, 1995; Hollis, Keay, Hallett & Gibbons, 1995; Pestemer & Günther, 1995) and regulatory measures (Riley, 1995).

Tooby & Marsden (1991) proposed that in order to obtain the most relevant data for registration purposes certain basic questions should be asked by the regulatory scientist or those involved in research and development. The following points are essentially those which Tooby & Marsden (1991) proposed:

- Is the active ingredient or degradation product(s) so persistent that it presents a risk to the consumers of the present and following crops?
- Is the active ingredient or degradation product(s) so persistent that it is likely to have an adverse effect on following crops?
- Are the concentrations in soil likely to present a risk to beneficial soil organisms?
- Are the predicted concentrations in water resulting from leaching or run-off likely to present a risk to aquatic organisms?
- Are the predicted concentrations in ground- and surface water likely to present a toxicological risk?
- Are the predicted concentrations in ground- and surface water likely to exceed maximum levels fixed by legislation?

8.9.3 Research, modelling and risk assessment/management

The smooth transition between the questions above and the objectives of studies for generating relevant data is obvious. The scientific approach that is necessary for obtaining such data usually has a tiered or stepwise approach. Data generated at each step need evaluating which requires an expert judgement. Clearly, computer models can be used at each stage of evaluation but their application in decision-making is currently not universally accepted. Nevertheless, almost all registration authorities carry out some sort of assessment which could be approximated by use of a single model or more likely a number of models.

Tooby & Marsden (1991) proposed a decision-tree for the assessment of the contamination of groundwater. Important data components used in the decision-tree are:

- halflife (time required for 50% degradation of the applied amount) of the parent or degradation products;
- mobility (leaching) as denoted by solid/solution soil partition coefficients (K_d- and/or K_{oc}-values);
- hydrolysis rate (time required for 50% hydrolysis loss) ;
- application rate (dosage) and timing; and
- the toxicity of metabolites or degradation products.

The regulatory authority in South Africa relies on the local agrochemical industry to furnish the types of data presented above. These data sets usually originate with parent companies which do the basic research abroad. Therefore, it is to be expected that the data are seldom appropriate for all edaphic and climatic combinations, and this would explain why pesticides often show aberrant behaviour in countries far removed from the compounds' origin. As foregoing discussions indicate, the complex interactions involving pesticides in the dynamic soil system demand data generation with local soils under conditions which simulate relevant climatic situations.

Pesticides are essential crop production inputs that will have to be depended upon until alternative measures for pest management can be found. The role of pesticide use in food production has been referred to as "dependence on technological crutches". Because there are no effective alternatives to chemical options in the foreseeable future, a form of tacit acceptance of this situation involves the introduction of agricultural management opportunities aimed at minimizing pesticide movement from target areas. These initiatives are also referred to as "Good Farming Practices" (Seiler & Mühlebach, 1995), which aim to avoid or at least minimize environmental impacts of pesticides. It is not surprising that these type of initiatives are usually launched by the agricultural chemical companies, since the industry is essentially self-regulatory with regard to the maintenance of stature and avoidance of "damage" liability claims. It remains the responsibility of regulatory authorities in individual countries to monitor the local situation, to appraise measures aimed at protecting the environment, and to institute strategies and legislation to this effect.

8.9.4 Knowledge requirements

Knowledge of the leaching behaviour of a pesticide in soil can be used for assessing the potential to contaminate groundwater, and for identifying soils where excessive leaching may occur. Leaching of pesticides is controlled by variations in numerous edaphic and weather factors. The most important soil factors that affect the leaching of most pesticides are reportedly soil texture and architecture, organic matter content and pH. If preferential flow can be excluded the vertical distribution of most pesticides in soil is determined primarily by the strength of sorption at adsorptive sites on colloidal particles, and also by the amount of water moving through the system. In addition, factors that preserve the chemical stability of pesticides, and hence, prolong their persistence, would increase the pollution risk for groundwater.

In previous sections numerous mechanisms of pesticide movement to groundwater were considered. Knowledge of the factors governing those processes should be used for prediction of environmental impacts of pesticides. The best approach in assessing potential negative impacts is to focus on worst-case scenarios. This implies that those characteristics of the environment which promote pesticide leaching should be identified and measured in suspected sensitive areas, and the data be used to select the most sensitive sites for long-term monitoring of pesticide presence. Simulation modelling of pesticide fate and behaviour could be an extremely useful tool in the identification of sites for monitoring.

8.10 RECOMMENDATIONS

At the moment knowledge and expertise of pesticide impact on the environment and groundwater in particular is widely scattered in South Africa. Contact between the various persons is informal and occasional.

It is strongly recommended that a vehicle be established that will encourage focussing of this expertise. This will enable increased public and/or government interaction with more effective results.

The ideal vehicle would be electronic interlinking of the expertise and opening the group to interested parties. The first step would be to identify the role players and establish a working group. This group will be formally connected electronically and there would also be an open electronic discussion forum. Ultimately the modelling expertise would be available via Internet.

8.11 GOOD FARMING PRACTICES

The two broad categories of sources of pesticide pollution are:

- point sources and
- non-point or diffuse sources

Each of these requires individual attention and care in managing losses of pesticide to the environment.

8.11.1 Point sources

Point sources are almost all related to the storage and mixing of pesticides and the disposal of excess or unwanted pesticides. A number of examples are discussed but it is important that the user of pesticides at all times is aware and thinks about the possible implications of his actions.

To illustrate the impact that a minor spillage has consider the washing of a spray tank. A standard pesticide spray application would be 500 g of active ingredient (a.i.) per hectare applied in 200 litres of mixture. At the end of the spraying there is 10 litres of mixture left. Poor practice would then allow the work water to drain into a sump or pit. This 10 litre mixture has 2.5 grams of active ingredient. Now the European Union Standards state that the maximum allowable concentration in drinking water is 0.1 μg/L. So if we take the above 2,5 g of a.i. it has to be diluted with 25,000,000 litres of water to reach this level.

Typical point sources that require care are:

- Disposing of excess pesticides down a sump, the sewer-system or down disused or dry boreholes are very bad practices. Excess pesticide should be sprayed onto fields where the degradation will be more rapid.

- Washing of spray-tanks, the wash water must also be treated as pesticide and sprayed onto fields.

- Back siphoning can occur when filling the spray-tank directly out of a borehole, stream or irrigation canal. Good practice is to have an overhead water-tank which is filled with water which then under gravity is used to fill the spray tanks.

- Always read the label and follow the recommendations. But always remember that the label instructions relate to the safety of the product with respect to impact on the target crop and other crops and seldom if ever make positive recommendations for disposal of excess or unwanted pesticides.

- A recommended measure for users who want to reduce the danger of spillage in the mixing area is to construct a pesticide spillage safety trap.
  "Have you constructed your PSST! Yet?"

8.11.2 Pesticide Spillage Safety Traps (PSST)

It is possible to permanently trap and to inactivate pesticides in a controlled way in order to prevent water pollution. The concept involved is the containment of pesticides in a relatively small compartment of the environment that can be managed in such a way that point-source contamination, which may result from spillages during filling and rinsing of spray equipment, is avoided. As mentioned in previous sections the inactivation of pesticides may be temporary through reversible retention (sorption) on adsorptive surfaces, or permanent through chemical or microbial degradation. The suggestion here is for a practical solution to the problem of the hazards involved in spillages of pesticide which often occur during filling or washing operations.

A site large enough to accommodate all sizes of pesticide application apparatus available on a farm needs to be selected. A pit 1.5 m deep and of adequate surface are must be prepared. Organic matter that is either already well-composted, or have the ability to compost readily is used to fill the pit. The organic matter is compacted well enough that the weight of a tractor and/or sprayer can be supported. The organic matter is covered with a layer of gravel that forms the working surface on which the tractor/sprayer is parked during filling/cleaning operations.

The environment inside the trap system should ideally be acidic so as to promote chemical hydrolysis of pesticides, and at the same time be conducive to high microbial activity. In other words do not add lime. The internal environment should be constantly moist in order that microbial activity and chemical degradation processes proceed at optimal levels.

A roof, either a lean-to or a carport, is strongly recommended to prevent excessive flooding. The water content of the organic matter should be checked regularly and adjusted if need be by addition of water. The organic matter must not be allowed to dry out completely.

Over time the surface will subside. The system can be rejuvenated from time to time by addition of organic matter to the system.

8.11.3 Diffuse pesticide pollution

It is recognised that pesticides have an important role to play in modern farming. The guiding credo should be "careful use of pesticides will increase crop production and have a minimum negative impact on the environment".

At all times the pesticide user must consider how, where and why he is applying the pesticide and in what manner can that pesticide have an impact on the environment and on water resources.

The major mode by which pesticides leave the area of application and reach rivers or groundwater is by rain followed by runoff or infiltration. So when planning a spraying listen to the weather report. Complicating the planning is that some pesticides need rain to be effective. Thus pre-emergent herbicides and soil-applied pesticides benefit from a light rain which washes the pesticide into the upper soil layers. Too much rain, however, washes the pesticide away or too deep into the soil profile. Post-emergent herbicides and plant applied fungicides and insecticides however do not benefit from rain as the pesticide will wash-off, reducing efficacy. Thus listening to the weather-report could save money.

Calibration of spray equipment is important to ensure that one does not over-apply, thus increasing the pesticide load that may impact and also costing more. Before spraying check the proper size of the nozzle tip, spray width per nozzle, and flow rate from nozzle.

Overland transport by flooding of both pesticide and soil to which the pesticide adheres to is the main cause of pesticide reaching dams and rivers. Thus all soil erosion prevention practices are good pesticide pollution prevention measures. These include minimum tillage, no tillage next to streams, tree or bush buffers along stream banks and contour terraces are good practice.

Integrated pest control is a very effective method for reducing the amount of pesticide used and thus the total negative impact of pesticide on the environment.

CHAPTER 9

CONCLUSIONS AND RECOMMENDATIONS

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9.1 SUMMARY OF CONCLUSIONS

The revised objectives of the project are listed below with a summary of the main findings.

1. *To confirm the contamination impact on groundwater quality resulting from those agricultural activities identified as posing the most serious threat.*

The literature survey identified intensive animal husbandry (IAH), sewage sludge application, inorganic fertilizer application, irrigation and pesticide use as the activities posing the most serious threat to groundwater quality.

Sufficient literature was available on the impact of irrigation and pesticide use in South Africa and this has been summarised in chapters 7 and 8. The other activities were studied in the field, each at three potentially vulnerable sites. The impact on groundwater quality was interpreted in terms of the agricultural practices at each site.

An impact on groundwater quality was seen at all the field sites with vulnerable, shallow, unconfined or semi-confined conditions. Nitrate was the most common agricultural contaminant evident in groundwater sampled. Nitrate distribution and isotopic analyses of NO_3-N indicated the most important sources to be sludge, manure and soil biota. Elevated DOC levels were associated with sludge application and IAH. Potassium, ortho-phosphate and microbiological indicators of faecal pollution, contaminated groundwater as a result of IAH practices.

The greatest impact was seen at a site where sludge was applied to agricultural land. Nitrate levels were elevated to 268 mg.l^{-1} in the dolomitic aquifer directly beneath the sludge applied area and were persistent to a level of 30 mg.l^{-1} to a distance of greater than 1 km. However it should be noted that sludge was applied at this site primarily as a means of disposal and application rates were not tied to crop requirements. There was therefore a long history (15 years) of over application. At the other sludge sites, where sludge was used to condition sandy soils, limited increases in nitrate levels were seen. At one site, shallow groundwater (< 3mbgl) was contaminated to a maximum of 38 mg.l^{-1} but this was not persistent with depth. This site also had a long history of application. At the other site, only a 2 mg.l^{-1} increase in nitrate levels was noted during the first season of application. Cumulative impacts of sludge application therefore appear to be significant.

Increased DOC levels were seen at 2 of the sludge field sites in shallow groundwater samples: to 30 $mg.l^{-1}$ at site C and to 10 $mg.l^{-1}$ at site A.

Significant impacts on groundwater quality were seen at IAH site B, overlying a shallow alluvial aquifer. Pollution was associated with livestock concentration in pasture and effluent irrigation. The highest nitrate levels seen at this site were 156 $mg.l^{-1}$. Increased levels of DOC, potassium, ortho-phosphate and faecal coliforms were also seen. IAH site A showed some contamination by nitrate and faecal streptococci and site C showed no contamination as a result of natural aquifer protection and good practices.

Limited impacts on groundwater quality were found to be associated with the application of inorganic fertilizers. High nitrate levels (>10 $mg.l^{-1}$) were seen at all the fertilized sites however, the source of this nitrate was not exclusively inorganic fertilizers. Fertilizers are believed to contribute to high nitrate levels but other activities such as the application of organic fertilizers and deep rip ploughing are also thought to play a significant role.

The literature reviewed in the essay on the impact of irrigation showed that percolates from the root zone and irrigation return flow can cause the salinisation of groundwater in irrigated aquifer systems. Significant salinisation has been reported in groundwater underlying the irrigated lands of the Great Fish-Sundays River basin (TDS increased from 2 000 to 3 400 $mg.l^{-1}$). At the Vaalharts irrigation scheme it is estimated that between 17 and 63 million cubic metres percolate to the water table annually carrying nearly 30 000 tons of dissolved salts. This has resulted in increased groundwater salinity, a rise in the water table and some water logging of soils.

The review of pesticide contamination of groundwater showed that limited, scattered information is available. In the early 1990s, a study showed that the herbicide atrazine was present in most rivers and dams in the maize producing areas of South Africa. These included the Olifants, Vals, Vaal, and Renoster rivers. This river water is used in many areas for irrigation and therefore where aquifers are recharged, acts as a source of pesticide contamination. Tests of the herbicides methochlor and terbuthylazine have shown that they leach beneath the root zone in a wide variety of South African soils.

2. *To identify practical and easy-to-implement strategies and practices which can be used to prevent or reduce contamination resulting from agricultural activities.*

   Detailed and illustrated recommendations for Good Farming Practices are given in the farmers' handbook. A summary of these recommendations is given below in section 9.3.

3. *To prepare a non-technical booklet, which transfers the information to farmers and members of the public associated with the agricultural sector.*

   This handbook has been prepared and was approved in the final steering committee meeting for this project. It is aimed at commercial farmers, as it is this group whose activities have the greatest potential to impact groundwater quality. Although originally termed a 'non-technical' booklet, it covers many good farming practices in a necessarily technical manner. It is assumed that most commercial farmers are used to working with technical information and are educated to at least matric level.

   The handbook gives an introduction to groundwater, its occurrence, vulnerability and the implications of deteriorations in groundwater quality. General guidelines for groundwater management are given, such as a methodology for obtaining reliable hydrochemical data. As nitrate is the main contaminant of concern, a chapter is devoted to an explanation of nitrate in groundwater and ways of limiting nitrate contamination.

   A chapter is given on each of the 5 agricultural activities examined by this project, quoting examples of groundwater contamination in South Africa and overseas. Details of good farming practices to minimise contamination are given. In addition, there is a chapter on potentially polluting features at the farmstead, such as pit latrines and storage tanks.

   Useful addresses are provided for many relevant organisations. An example groundwater sampling sheet is given and guidelines for drinking water quality.

   The booklet will be available in English and Afrikaans, as a complete publication, as separate information sheets covering the different activities and on the Internet. It is hoped that it will be endorsed by DWAF and NDA as well as the Agricultural Unions. It should be freely available from the WRC, DWAF, NDA and various agricultural departments.

9.2 RECOMMENDATIONS FOR FURTHER RESEARCH

Details of further scientific work related to the specialist studies are given in those chapters. It is recommended that longer term monitoring of groundwater quality and agricultural activities at the study sites listed below would improve the understanding of the processes at work.

- Sludge site B.
- IAH site B
- Fertilizer sites B and C.

As nitrate is the main contaminant of concern, it is recommended that further investigations into the impact of elevated nitrate levels on the environment and human health should be conducted. More detailed studies are required on the processes of nitrate accumulation in groundwater from anthropogenic and natural sources.

New areas of study warranting further research are:

- The impact on groundwater quality of deep 'rip' ploughing new areas for cultivation. This is thought to release nitrates derived from soil biomass decaying as a result of exposure, but the level or persistence of contamination is not well documented.

- Methods to determine the rate of bioavailable nitrogen release from organic sources such as sludge, different soil types and manures.

The mobility and persistence of hormones and steroids used in Intensive Animal Husbandry is a new area of research that has not received attention in South Africa. If effective, analytical techniques are available, this area should be investigated.

At the moment knowledge and expertise of pesticide impact on the environment and groundwater in particular is widely scattered in South Africa. Contact between the various persons is informal and occasional. It is strongly recommended that a vehicle be established that will encourage focussing of this expertise. This will enable increased public and/or government interaction with more effective results. The ideal vehicle would be electronic interlinking of the expertise and opening the group to interested parties. The first step would be to identify the role players and establish a working group.

9.3 RECOMMENDATIONS FOR GOOD FARMING PRACTICES

Recommendations for GFPs are given in the farmers' handbook (Handbook of Groundwater Quality Protection (*or Farmers*). A summary of these recommendations is given below.

9.3.1 Intensive animal husbandry

Natural processes can be harnessed to help with waste management at intensive animal farms, if they are understood. For instance, in the case of active feedlot pads and effluent lagoons, a layer with very low permeability will naturally form and serve to protect groundwater if it is maintained.

The potential for seepage of nutrients to the groundwater below **effluent lagoons** is reduced due to the accumulation of solids and clogging by bacterial cells and fine organic matter. Infiltration may occur from new unlined ponds but, with several months accumulation, many become self-sealing. *A settling basin or solid separator is usually required to maintain an efficient life-span for an effluent lagoon.*

Self-sealing may not be established in areas with coarse sands, fractures or fissures. *In these areas artificial pond lining is required.* In anaerobic effluent lagoons any nitrate that forms is usually denitrified. This chemical reduction combined with low permeability at the base of the lagoon tend to mitigate nitrate leaching.

Cases of groundwater contamination from effluent lagoons are associated with the rupture of the lagoon seal by seasonal drying out. *Moist conditions at the base of a lagoon should be maintained at all times and scraping out the base lagoon avoided as this will destroy the self sealing layer.*

Feedlot pads receive a high loading of bovine waste but generally low levels of infiltration and contaminant transport are recorded in well managed pads. The concentration of cattle in the pad area, whilst resulting in high levels of manure deposition, also acts to compact manure forming a low permeability, anaerobic layer at the manure - soil interface. Water cannot easily permeate this layer therefore *care should be taken when removing surface manure from pads not to remove the interface layer.*

Correct siting of feedlot pads and good drainage goes a long way to prevent contamination. Feedlots should not be sited in areas with greater than 750 mm rainfall per annum. Pads should be sited on a 2 to 5% slope with feeding and drinking troughs at the upper end. The slope will minimise standing water and run-off should be collected in channels and directed to an effluent lagoon. The risk of ground and surface water contamination is greatest during the first rains of a wet season. This is known as the first flush effect. *Careful management of this run-off is required to prevent it reaching surface water or areas where it may infiltrate to groundwater.* When a pad is abandoned the manure dries and shrinkage cracks form. The permeability of the pad base or interface layer is then increased and the risk of infiltration increases. *At this point all manure and the interface layer should be removed.*

The literature shows that nitrate leaching to groundwater from areas of **land disposal** of waste and effluent is significant and widespread. Application to land is widely practised as it provides an economic means of disposing of effluent and manure. Other forms of disposal, such as discharge to a surface water body, would require expensive treatment. Disposal to land is a beneficial use option where fodder crops are grown.

Problems of contamination occur due to the high levels of contaminants in the waste, high application rates and aerobic conditions in the disposal area which enhance nitrification and subsequent leaching. The risk of leaching is greatest where high volumes of effluent are applied or irrigation is carried out in addition to waste application. A limit of 170 kg of nitrogen per hectare per annum in manure applied to land is recommended in Europe. *Additional irrigation should be avoided and the nitrogen loading in the effluent or manure should be matched to the requirements of the crop or pasture. Minimise effluent irrigation to land overlying shallow, vulnerable aquifers.*

For raw piggery waste, the following calculation can be carried out:

Application rate (pigs per hectare) = nitrogen requirement of the crop or pasture x 0.1

If the waste has undergone some treatment (anaerobic or aerobic) then the nitrogen requirement may be multiplied by 0.15.

General good practice should also include the maintenance of grassed buffer zones several tens of metres wide next to surface water features. No effluent of manure should be applied to these buffers.

Significant leaching of nitrate to groundwater occurs beneath pastures. The leaching is mainly associated with urine patches which release nitrate more quickly than manure. The very irregular distribution of livestock wastes across a paddock results in many small patches where extremely high nitrogen concentrations result in leaching. Irregular patches occur due to excretion during grazing in pasture and heavier manure loads occur due to periodic livestock concentration such as at feeding or water troughs and gates. *The impact of this can be minimised by not exceeding the carrying capacity of the pasture and using mobile feeding and drinking troughs.*

9.3.2 Fertilizer application

Nitrate is more likely to infiltrate to groundwater under the following conditions:

- the water table is shallow and the unsaturated zone is permeable
- fertilizer is applied at a time when it is not being taken up by the crop
- fertilizer is applied at a rate which is greater than the crop can use it
- rain or irrigation water is present to carry the nitrate beneath the root zone and out of reach of crops.

If fertilizers are applied during a rainy period when the ground is bare and there is no uptake by plants, it is likely that a significant portion of the nitrogen supplied will be washed beneath the root zone. This means that the crop will not receive the correct amount of nitrogen for optimal growth and there is a high risk of contaminating groundwater with nitrate.

In planning an effective schedule for fertilizer application the farmer needs to know:

- **When** does the crop require the most nutrients, particularly nitrate? This is usually around early periods of growth following planting and prior to fruiting.
  And
- **How much** fertilizer should be applied at the different times?

Working out the nitrogen balance for a crop can be a complicated procedure. It is usually carried out over a few years and is based on the practical experience of the farmer. Field observations, such as signs in the crop of excessive growth rates, and measurements of the crop tissue, soil nutrient levels and groundwater chemistry all contribute to getting the balance right. Measurement of nitrate in the root zone soil (as opposed to organic nitrogen) gives an indication of some of the residual nitrogen that will be available. (See Chapter 2)

Farmers need to be particularly aware of over fertilization risks in high rainfall areas or during rainy periods. Over irrigation may also lead to greater leaching losses where there is excess nitrogen in the soil. Irrigation should be scheduled to the crop requirements.

Fertilization

1. If groundwater is used to apply fertilizer, the concentrated fertilizer and water should not be mixed within 15 m of the borehole.
2. Fertilizer should not be applied within 50 m of a borehole.

Contamination from fertilizers can also occur during storage or handling. Chapter 8 of the handbook outlines good practices to minimise contamination risk around the farm yard (werf).

9.3.3 Sludge application

Particular care should be taken with sludge application where any of the following conditions apply:

- shallow aquifers underlie the fields where sludge will be applied
- groundwater is used as drinking water for people or livestock on your farm or in the surrounding area
- the quality of the groundwater is marginal

Guidelines recommending safe sludge application rates exist in many countries. In South Africa guidelines are given by the Department of National Health and Population Development (DNH&PD). The maximum rate of application recommended is 8 t (dry mass)/ha/year. The concentrations of potentially contaminating substances such as heavy metals may restrict the application rate to less than this and limit the cumulative loading over a period of several years.

From the point of view of metal mobility, sludge should not be applied to soils with a pH of less than 6. The guidelines also recognise that sludge application to agricultural land should not exceed the crop nitrogen requirements to prevent leaching of nitrogen to groundwater. Given the difficulties in accounting for available nitrogen from sludge and any inorganic fertilizers that may be used, they provide only a general guide for crop nitrogen demand.

Sludge is classified into four types by the DNH&PD depending on its source, how it is treated and the level of potentially harmful substances it contains, as shown in the table below.

Guidelines for sludge use

Type	*Source and treatment*	*Stability and uses*
A	Sludge from primary settling tanks, possibly mixed with other sludge types but which has received no treatment other than dewatering	Unstable with a potentially high fly and odour nuisance and high levels of pathogenic organisms. May be used for composting, landfill and tree plantations with certain restrictions.
B	Waste activated sludge, humus sludge or sludge which has been digested. Generally some dewatering has taken place	Stable with low fly and odour nuisance and reduced levels of pathogens. May be used as above plus on crops (excluding vegetables), grazing, public gardens and parks, etc, and nurseries with certain restrictions.
C	Sludge to which some specific sterilization treatment has been applied, eg. composting, pasteurisation, lime stabilization	Stable with insignificant odour and fly nuisance and insignificant levels of pathogens. May be used as above with certain restrictions.
D	Tertiary sludge with low heavy metals content. Similar hygienic quality to Type C and metal contents are below stated limits.	Stable with insignificant odour and fly nuisance and insignificant levels of pathogens. May be used as above with certain restrictions plus on household vegetables and private gardens with certain recommendations.

Type A sludge should not be applied to a slope of greater than 6% or to land underlain by an aquifer at less than 5 m. The sludge applied area should be greater than 500 m from a dwelling and greater than 200 m from a river, dam or borehole. Type B sludge should not be applied to a slope of greater than 4% or to land underlain by an aquifer at less than 2 m. The sludge applied area should be greater than 200 m from a dwelling, river, dam or borehole.

9.3.4 Salinisation as a result of irrigation

The best solution to the mineralization problem of the groundwater is to minimize saline irrigation return flow. This can be achieved by

- reduction in the leaching fraction and over-irrigation.
- reduction in the leakage from canals and dams.
- application of a better quality irrigation water in order to reduce the input of water borne soluble salts.
- artificial drainage to remove percolates rapidly from the soil body and to prevent build up of perched water tables.
- choice of area and timing of new land development.

9.3.5 Pesticides

The recommended pesticide **application limits** on the label should always be adhered to. The limit recommended for atrazine is 2.5 kg of active ingredient per hectare per year. Be aware of the effectiveness of the pesticide in your environment and lower the application rate if possible. During cooler dry periods it may be able to attain the same level of control with a lower dose (depending on the pesticide).

If **pesticides** are mixed with groundwater this should be done at least 15 m from the borehole and the hose removed from the mixing tank before switching off the water supply. This avoids the risk of back-siphoning which may inject pesticides directly into the borehole. A back-flow prevention device should be installed if possible. Remember that spills at the mixing area result in a much higher loading than normal application would.

Some pesticides such as pre-emergent herbicides and soil-applied pesticides benefit from a light rain which washes the pesticide into the upper soil layers. Too much rain, however, washes the pesticide too deep into the soil profile or carries it away with surface run-off. Post-emergent herbicides and plant applied fungicides and insecticides do not benefit from rain as the pesticide will wash-off. The farmer should be aware of the local weather predictions and avoid application before heavy rainfall.

Pesticide containers should be disposed of responsibly to a registered landfill site or returned to the supplies. These containers should never be used for water. The bottom of the container should be punctured to ensure that it will not be used. Similarly, **excess mixture** and rinse-water should be disposed of responsibly.

Spray equipment should be calibrated to ensure that pesticide is not over-applied. Before spraying, the size of the nozzle tip, spray width per nozzle, and flow rate from the nozzle should be checked.

At farms where pesticides are regularly used, the best practice to inactivate pesticides in a controlled way is to construct a Pesticide Spillage Safety Trap (PSST). This makes use of the natural processes of breakdown and pesticide adsorption to clays and organic matter. Once it is constructed all pesticide handling should be carried out over the special area. Construction details are given in the handbook.

CHAPTER 10

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