# UTILIZATION OF TRACER EXPERIMENTS FOR THE DEVELOPMENT OF RURAL WATER SUPPLY MANAGEMENT STRATEGIES FOR SECONDARY AQUIFERS

**Report to the**
**WATER RESEARCH COMMISSION**

by

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UTILIZATION OF TRACER EXPERIMENTS FOR THE DEVELOPMENT OF RURAL WATER SUPPLY MANAGEMENT STRATEGIES FOR SECONDARY AQUIFERS

The Steering Committee who directed the project consisted of the following persons:

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# Executive Summary

## 1. Background and Motivation

The semi-arid and arid regions of South Africa form approximately 66% of the country. Most of these regions do not have surface water resources. Groundwater is therefore becoming an important component of the water supply for many South African rural communities in these regions. It also offers a cost-effective solution for rapidly developing settlements, which lack the necessary infrastructure normally associated with water supply. However, rational use of groundwater requires abstraction volumes staying within the assured yield of the aquifer. In addition the catchment area of a borehole must be protected against pollutants such as sewage effluent.

This report presents the findings of research conducted by the Institute for Groundwater Studies on using tracer experiments to develop rural water supply strategies. The study focuses on groundwater velocity estimates, with special reference to the delineation of borehole protection areas. The report provides a summary of a literature survey conducted in the initial phases of the project and detailed discussions on the various types of tracers and tracer tests. These tests are applied to numerous study areas. The results of the tests are used to delineate borehole protection zones. Finally conclusions and recommendations are drawn.

## 2. Statement of Objectives

The objectives of the research project are summarized as:

- Formulating a management strategy for rural water supply in secondary aquifers to determine:
  - i. the catchment area and significant recharge zones which will optimize well-field abstraction rates and
  - ii. the protection zones around boreholes to minimize the influence of potential contaminant sources
- Investigating the use and suitability of artificial tracers in fractured-rock formations and providing the following information:
  - i. travel times of water in the aquifer
  - ii. recharge rates and recharge areas
  - iii. aquifer parameters
  - iv. the geometry of fractures (the importance of the double-porosity concept) and
  - v. the role of fracture apertures in contaminant migration.

# 3. Method and Summary of Results

The project team developed methodologies based on tracer tests to delineate protection zones. Tracer tests can be divided into two main classes, namely natural and forced gradient tracer tests. Natural gradient tracer tests are seldom conducted in fractured formations because of large scale heterogeneities. Because the installation of well points is easier in porous formations (such as unconsolidated sands), natural gradient tracer tests are usually conducted in these formations. Another method called a borehole dilution test was developed to measure the groundwater velocity under natural conditions. This method can be applied in both porous and fractured rock aquifers.

Forced gradient tracer tests involve tracer measurements in radial convergent and radial divergent flow fields.

The tracer tests selected for this study include borehole dilution and the radial convergent tests. The dilution test relates the observed rate of tracer dilution in a borehole to the average groundwater velocity in the aquifer. Since the decline of the tracer concentration in the borehole is exponential, a linear relation is displayed on a semi-log graph. Darcy's velocity can be calculated from the dilution test. A radial convergent flow field is created by pumping a borehole until steady state conditions are reached. A tracer is then introduced in an injection borehole in the vicinity of the abstraction borehole such that a minimum disturbance of the flow field is caused. The tracer breakthrough curve is monitored at the abstraction borehole. Analyses of the resulting breakthrough curves yield estimates of kinematic porosity, aquifer dispersivity and groundwater velocity. The convergent tracer test is attractive because it is theoretically possible to recover the entire tracer from the aquifer. The seepage velocity can be calculated from a radial convergent test. If a dilution test is performed at the same time, the kinematic porosity can also be calculated.

Numerous tracer and dilution tests have been conducted in a number of different geological units such as Karoo sediments, dolerite, dolomite, gabbro/norite, granites, and quartzite/gneiss.

The tracer test results indicate high groundwater velocities along fractures and low velocities in the matrix. Even under natural conditions flow velocities of up to 30 m/d are possible along fractures. The tracer test interpretations also show that the parallel plate model for fractures is not valid for Karoo aquifers, and that the fracture can be viewed as a porous medium with a relatively high kinematic porosity and a small effective thickness.

Under certain circumstances shallow aquifers may be very vulnerable to contamination. The protection of the source for these aquifers is critical in a groundwater quality management strategy. A borehole protection area (BPA) can be defined as the controlled surface area surrounding a production borehole (or well-field), which prevents contaminants from reaching the borehole. It may consist of a capture zone as well as a borehole catchment. In general, the capture zone is delineated to provide a buffer zone for the degradable contaminants to die off, while the borehole catchment is delineated to prevent the persistent contaminants from entering into the borehole.

Delineation of BPAs for groundwater source protection is normally performed by using numerical codes which, possess the ability to handle complex hydrogeological conditions. A delineated BPA is usually irregular in shape. However, it is important and practical to depict a BPA with simple geometric shapes. Protection zones are divided into the following classes:

- **Protection zone I (Fencing):** The fence around a borehole must be at least 5 m from the borehole. For a borehole that is supplying water to less than 20 persons, a well-constructed sanitary seal is sufficient. Quality monitoring is important.
- **Protection zone II (Microbial pollution):** A second protection zone around the borehole is suggested. This zone protects drinking water from microbial contamination.
- **Protection zone III (Hazardous elements):** If persistent hazardous non-degradable elements are present, the whole catchment area of a borehole must be protected.

An EXCEL computer program, BPZONE, was developed to delineate protection zones II and III. The program estimates the maximum expected nitrate value in a borehole by considering the number of people living close to the sanitation area.

## 4. Meeting the Objectives

All the objectives of this project were met except investigating the impact of fracture apertures on contaminant migration. This study was not possible because of logistic problems.

In addition, the Steering Committee decided the project must focus on tracer experiments and delineation of capture zones and not on delineation of recharge areas.

Originally it was foreseen that a Ph.D. student from Switzerland would perform some microsphere tracer tests in South Africa, however, this was not possible as the apparatus was problematic.

## 5. Conclusions

This report presents a systematic approach to conducting tracer tests. If performed together, the borehole dilution and radial convergent tests can yield the following parameters: seepage and Darcy velocities, kinematic porosity and the effective thickness of a fracture zone. The delineation of borehole areas in fractured aquifers are also discussed. Results indicate that borehole protection areas for these types of aquifers are usually much larger than those in typical porous aquifers.

Tracer experiments may assist in revealing additional information concerning fracture characteristics. Furthermore the following can be concluded from the numerous tracers tests conducted:

- Velocity estimates from borehole dilution tests are regarded valid if the value of the effective porosity is known.
- Since effective porosity cannot be derived from hydraulic tests, it must be estimated from a tracer migration test between two boreholes.
- Determining hydraulic conductivity from slug test data in fractured aquifers is scale related, and data are often misinterpreted since the actual aquifer thickness is not known. Tracer tests should be used to calculate this parameter.
- The peak of a tracer breakthrough curve does not represent the mean pore velocity. The data should therefore be analysed with a suitable model as described in Chapter 4.
- Assuming an area is homogeneous is a serious mistake that can be made when assessing the impact of a pollution on groundwater. Vertical fractures should be an important consideration.
- The unsaturated zone is of great importance when dealing with groundwater contamination.
- Tracers are definitely an important aid in gaining a better understanding of groundwater flow mechanisms.

# 6. Recommendations for Future Research

The protection of our groundwater resources should be of major importance and more research should be conducted to ensure a minimal impact of pollution. Therefore the application of the proposed protection zones should be tested extensively.

This research has focused on point pollution sources, however, pollution sources with a larger areal extent, such as mines, waste sites and industrial areas should also be considered.

When conducting an impact study, more emphasis should be placed on tracers to gain more information and data and, it is therefore suggested that tracer tests such as injection withdrawal tests should be investigated.

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# CHAPTER 1

# INTRODUCTION

## 1.1 Project objectives

Groundwater is becoming an important component of the water supply for rural communities. It also offers a cost-effective solution for rapidly developing settlements, which lack the necessary infrastructure normally associated with water supply. Rational use of this resource requires that the planned abstraction volumes have to stay within the assured yield of the aquifer, and that the catchment area of the boreholes has to be protected from polluting activities, such as sewage effluent.

**The objectives of the project were as follows:**

- To investigate the use and suitability of artificial tracers in fractured-rock formations and provide the following information:

  (i) travel times of water in the aquifer;
  (ii) recharge rates and recharge areas;
  (iii) aquifer parameters;
  (iv) the geometry of large and smaller fractures (the importance of the double-porosity concept) and
  (v) the role of fractures of different apertures in contaminant migration.
- To formulate a management strategy for rural water supply in secondary aquifers to determine the following:

  (i) the catchment area and significant recharge zones which will optimize well-field abstraction rates and
  (ii) protection zones (capture zones/residence times of groundwater) around well-heads to minimize the influence of potential pollution sources such as sewage

The study focuses, in particular, on groundwater velocity estimates, with special reference to the delineation of borehole protection areas. In general, it seems that most tracer tests conducted outside the Republic of South Africa are done by government organisations or other research institutions. This is probably due to the expense and complexity of these techniques, as well as time consuming field methods, making tracers tests unattractive to the private groundwater industry, such as consultants. The aim of the project was to facilitate existing techniques for implementation in practise, within the constraints set by meaningful interpretation.

## 1.2 Literature survey

The literature survey included a study on:

(i) The types of existing tracer test techniques,
(ii) tracer tests available in the literature, and,
(iii) the tracers used.

Tracer tests can be divided in two main classes, namely natural and forced gradient tracer tests.

Natural gradient tracer tests in fractured formations are scarce. Due to the large scale of heterogeneity involved with secondary aquifers, these tests are seldom conducted. Since the installation of well points as the plume migrates is much easier in porous rather than in fractured formations, natural gradient tracer tests are usually conducted in porous media such as unconsolidated sands. Another test, the borehole dilution technique, was developed to measure the groundwater velocity under natural conditions and can be successfully applied in both aquifer types (Freeze and Cherry, 1979).

Forced gradient tracer tests involve tracer transport measurements in the following flow fields:

(i) radial convergent,
(ii) radial divergent, and,
(iii) injection withdrawal.

Radial convergent and divergent flow fields are created by pumping water from or into the aquifer respectively. The convergent tracer test is easier to conduct in the field, while the radial divergent test has the advantage that the tracer arrival can be observed at several observation points, using a single tracer. The injection-withdrawal test may be conducted in a recirculating or non-recirculating mode. In the recirculating mode, water that is pumped from the withdrawal well is returned to the aquifer via the injection well. Obviously, data from the non-recirculating method are easier to analyse.

Table 1.1 provides a summary of the tracer tests found in the literature while the type and amount of tracer used during some documented tests, together with the rate of tracer movement in the different aquifer types under the given hydraulic conditions are available in Table 1.2. This was done to provide a guideline on questions like: How much tracer must be used, what should the pumping rate be, how far can observation boreholes be drilled, etc. Table 1.3 summarises the tracer tests conducted in South Africa.

The research was mainly done on the Campus Terrain at the University of the Orange Free State, and is restricted to the groundwater velocity estimations in linear and radial convergent flow fields since both flow fields exist in practice. The tracer tests selected for groundwater velocity determination include the borehole dilution and the radial convergent test.

**Table 1.1: (Sililo *et al.*, 1997) Summary of the tracer tests that has been documented else where in the world.**

| SITE | TRACER | TECHNIQUE | GEOLOGY | AIM/PRAMETERS ESTIMATED | COMMENT | REFERENCE |
|---|---|---|---|---|---|---|
| Sand Ridge State Forest (Illinois) | Dyes<br>- Rhodamine WT<br>- Amino G Acid<br>- Lissamine FF | Natural gradient | windblown sand, glacial outwash and gravel | Linear velocity<br>Longitudinal dispersivity | A solute transport model was used to determine the parameter | Naymik and Sievers (1985) |
| George Village (New York) | Rhodamine WT<br>Tritium | Natural gradient | Delta sand beds | Average velocity | Test done to determine flow of secondary effluent<br>Rhodamine WT better of the 2 tracers :<br>- detected visibly<br>- no potential radioactive hazard | Aulenbach *et al.* (1978) |
| San Joaquin Valley (California) | Fluorescein dye | Forced gradient | Sand, silt and clay | Hydraulic conductivity<br>Longitudinal dispersivity | A flow (MODFLOW) and solute transport model (MT3D) was used to determine the parameters | Hyndman and Gorelick (1996) |
| Neckar River valley (Germany) | Dyes<br>- Rhodamine WT<br>- Flourescein | Forced/natural gradient | Sand and gravel | Hydraulic conductivity<br>Transport velocities<br>Longitudinal dispersivity<br>Validation of reactive transport model | Analytical analysis<br>Rhodamine WT showed sorbing characteristics<br>Fluresceine is non-sorbing | Ptak and Schmid (1996)<br>Ptak and Teutsch (1994) |
| Mobile (Alabama) | Sodium Bromide | Forced gradient | Sand and gravel | Hydraulic conductivity<br>Dispersion coeffiecient<br>Advection | Numerical model used for analysis | Güven *et al.* (1992)<br>Molz *et al.* (1985)<br>Molz *et al.* (1986)<br>Molz *et al.* (1988) |

| | | | | | | |
|---|---|---|---|---|---|---|
| O'ahu<br>(Hawaii) | Helium | Forced gradient | Basaltic extrusive rock forms | Assess helium as a tracer | Advantages of helium :<br>- low background atmospheric concentration<br>- low molecular diffusion constant<br>- moderate solubility in water<br>- easily available and cheap | Gupta *et al.* (1994) |
| Wisconsin | Iodide<br>Bromide | Natural gradient | Sand, silt and clay | Capture zones | Used analytical analysis techniques. Results were not very accurate | Chambers and Bahr (1992) |
| Cape Cod<br>(Massachusetts) | Bromide<br>Fluoride<br>Molybdate<br>Lithium | Natural gradient | Glacial outwash plain | Mean velocity<br>Hydraulic conductivity variabilities<br>Dispersivity<br>Effect of recharge on macrodispersion | Fluoride was abandoned as a tracer because the fluoride concentrations were rapidly attenuated by sorption.<br>Longitudinal mixing dominant dispersion process.<br>Although non reactive solutes concentrations are highly variable and difficult to predict on a small scale, the *average* characteristics can be predicted accurately. | Garabedian *et al.* (1991)<br>LeBlanc *et al.* (1991)<br>Hess *et al.* (1992)<br>Ezzedine and Rubin (1997) |
| Hilly Loess<br>(Washington State) | Enviromental tracers<br>- Tritium<br>- Nitrate<br>- Chloride | Natural gradient | Quaternary loess deposits | Velocity<br>Recharge | Chloride and tritium together provide an understanding of shallow flow and recharge regimes | O'Brien *et al.* (1996) |
| Georgetown<br>(South Carolina) | Potassium Chloride | Forced gradient | Sandy coastal plain | Porosity<br>Dispersivity | To solve for these parameters, a three dimensional characterization using small scale measurements and a three dimensional model was recommended | Mas-Pla *et al.* (1992) |
| Wisconsin | Sodium Bromide | Forced gradient | Sand and gravel | Contaminant migration route | NaBr used because :<br>- not subject to precipitation or | Meiri (1989) |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | | | | sorption<br>- non toxic<br>- very soluble | |
| Forest of Dean Coalfield (Glouchestershire) | Sulpho Rhodamine B | Forced gradient | Coal deposits | Average velocity | Fluorescent dyes cannot be considered as conservative tracers in mined coal measures aquifers because there is substantial adsorption of them onto ferric hydroxide. Sulpho Rhodamine B appears to offer the best resistance to adsorption onto ferric hydroxide | Aldous and Smart (1988) |
| Scarborough (Ontario) | Sodium Chloride | Forced gradient | Fine uniform sands | Hydraulic conductivity<br>Porosity ratios<br>Dispersivity | Linear relationship between dispersivity parameter and time of the arrival of the tracer | Palmer and Nadon (1986) |
| Qttawa River valley (Ottawa) | Iodine 131 | Natural gradient | Silty sand till | Velocity variations<br>Dispersivities Transverse dispersion | Curve fitting techniques used for analysis | Killey and Moltyaner (1988)<br>Moltyaner and Killey (1988) |

**Table 1.2: Summary of the tracers used in case studies.**

Note: $Q_{in}$ = injection flow rate; $Q_{abs}$ = abstraction flow rate; Q = injection-withdrawal / recirculation flow rate

| AQUIFER TYPE | TECHNIQUE | SCALE | TRACER AMOUNT | REFRENCE |
|---|---|---|---|---|
| **Fluorecein** | | | | |
| Sand/gravel deposists | Radial divergent<br>$Q_{in}$ = 4 l/sec | 5-10 m | 15-20g | Ptak and Teutsch (1994) |
| Sands/gravel deposits | Natural gradient | 25-50m | 200 -250g | Ptak and Teutsch (1994) |
| Sand/gravel deposits | Radial convergent<br>$Q_{abs}$ = 3 l/sec | 9-13 m | 10 -20 g | Ptak and Schmid (1996) |
| **Lissamine FF** | | | | |
| Fractured rock,<br>Dolomite | Natural gradient,<br>tracer migration | 30 m | $1{,}7 \times 10^{-1}$g | Novakowski *et al.*(1995) |
| Fractured rock,<br>Dolomite | Natural gradient,<br>borehole dilution | - | $4{,}5 \times 10^{-4}$g | Novakowski *et al.*(1995) |
| Fractured rock | Radial divergent<br>$Q_{in} = 4{,}46\times10^{-3}$-<br>$7{,}35\times10^{-3}$ l/sec | 15m | $4{,}5 \times 10^{-4}$g<br>$9 \times 10^{-4}$g | Novakowski (1992) |
| Fractured rock<br>Dolomite/shales | Radial divergent<br>$Q_{in} = 6{,}25\times10^{-3}$ l/sec | 30 m | $1 \times 10^{-1}$g | Novakowski and Lapcevic<br>(1994) |
| Sand/gravel deposits | Natural gradient | 15m | approximate 400g | Naymik and Sievers (1985) |

| | | | | |
|---|---|---|---|---|
| **Rhodamine WT** | | | | |
| Sand/gravel deposits | Radial convergent $Q_{abs}$ = 3 l/sec | 9-13m | 8-16g | Ptak and Schmid (1996) |
| Sand/gravel deposits | Natural gradient | 15m | approximate $4\times10^{2}$g | Naymik and Sievers (1985) |
| **Sulpho rhodamine B** | | | | |
| Tracer test in a coal mine | Natural gradient, tracer injected in a stream discharging into the mine | 500m | 5g | Aldous and Smart (1988) |
| Tracer tests in coal mine | Tracer injected in flooded mine workings | 3 600m | $5\times10^{3}$ g | Aldous and Smart (1988) |
| **Amino G acid** | | | | |
| Sand/gravel deposits | Natural gradient | 15m | approximate $4\times10^{2}$g | Naymik and Sievers (1985) |
| **Bromide** | | | | |
| Unconsolidated sediments | Natural gradient | 250m | $4,9\times10^{3}$g(LiBr) | Knopman *et al.(*1991) |
| Granular aquifer | Radial divergent $Q_{in}$ = 15,6 l/sec | 5,7m | $2,33\times10^{5}$ g | Molz *et al.(*1988) |
| Granular aquifer | Injection - withdrawal | 40m | - | Molz *et al*.(1988) |

| | | | | |
|---|---|---|---|---|
| | $Q_{in}$ = 16 l/sec | | | |
| Glacial deposits | Natural gradient | - | 7,05×10$^2$g | Chambers and Bahr (1992) |
| **Fluoride** | | | | |
| Unconsolidated sediments | Natural gradient | 250m | 5,95×10$^2$ ($LiF_2$) | Knopman *et al.(*1991) |
| **Molybdate** | | | | |
| Unconsolidated sediments | Natural gradient | 250m | 6,10×10$^2$g | Knopman *et al.(*1991) |
| **Iodide** | | | | |
| Glacial deposits | Natural gradient | - | 8×10$^2$g | Chambers and Bahr (1992) |
| **Chloride** | | | | |
| Coastal sand | Two well, re-circulation, Q = 0,006 l/sec | 5m | 7,16×10$^2$ g KCl | Mas-Pla *et al.*(1992) |
| Uniform sand | Two well, re-circulation, Q = 22 l/sec | 6,5 m | 6,39×10$^4$g | Palmer and Nadon (1986) |
| **Tritium** | | | | |
| Fractured carbonate | Two well, re-circulation 2,7 l/sec | 54m | 18,5 Ci | Grove and Beetem (1971) |
| Fractured carbonate | Two well, re- | 120m | 14,141 Ci | Claassen and Cordes |

| | | | | |
|---|---|---|---|---|
| | circulation, Q = 22,5 l/sec | | | (1975) |
| **Iodine** | | | | |
| Sand | Natural gradient | 40m | $^{131}I$ | Moltyaner and Killey(1988) |
| Fractured rock | Radial divergent | 10 -15 m | $^{131}I$ (70 mCi) | Novakowski *et al*. (1985) |
| **Bromine** | | | | |
| Fractured rock | Injection-withdrawal Q = 0,2 l/sec | 10m | 0,131 GBq of $^{82}Br$ (KBr)<br>0,196 GBq of $^{82}Br$ (KBr) | Novakowski *et al*. (1985) |

**Table 1.3: (Sililo *et al*., 1997) Summary on the tracer tests conducted in South Africa**

| SITE | TRACER | TECHNIQUE | GEOLOGY | AIM/PARAMETERS ESTIMATED | COMMENTS |
|---|---|---|---|---|---|
| Atlantis | Sodium Chloride | Forced gradient | Fine to coarse sands | Porosity<br>Groundwater velocity<br>Permeability<br>Dispersivity | Analytical methods used for data analysis.<br>Dispersivity scale dependent. |
| IGS (1992)<br>(Bloemfontein) | Sodium Chloride | Natural gradient | Fractured sandstones | Flow velocity<br>Hydraulic conductivity<br>Effective porosity<br>Fracture identification | The role of highly conductive fracture zones in flow regimes should be investigated. This would add new information on transport features and thus provide a better understanding of borehole protection measures in secondary aquifers. |
| South Africa | NaI-131 | Forced gradient | Sands | Porosity | Analytical interpretation techniques.<br>Precautionary measures have to be taken against radioactivity |
| Eastern Free State<br>Beaufort West<br>(Bredenkamp and vd Merwe, 1977) | NaI-131<br>Florescent dye | Forced gradient<br>Forced gradient | Dolerite<br>Siltstone and mudstone | Porosity<br>Effective porosity | Analytical interpretation techniques.<br>Precautionary measures have to be taken against radioactivity |

# CHAPTER 2

# OVERVIEW OF TRACERS USED TO STUDY GROUNDWATER MOVEMENT WITH SPECIAL REFERENCE TO ARTIFICIAL TRACERS

## 2.1 Introduction

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The application of tracers in the field of hydrology has made a tremendous contribution to our current understanding of different aspects of the water cycle, and it is foreseen that it will continue to do so in the future. Artificial tracers are applied in a site specific manner and introduced only in the region of interest, while environmental tracers are usually associated with regional studies or the identification of regional water bodies. A summary on the paper by Evans (1983) presents a review on artificial tracers used to study groundwater movement.

## 2.2 Definition

Tracers are identifiable substances which, from the examination of their behaviour in a flowing medium, may be used to infer the general behaviour of the medium. They may be broadly categorised by their origin. Those substances normally present in the medium are generally termed environmental tracers, while those deliberately introduced into the medium for the purpose of the study are termed artificial tracers. Artificial tracers are used in the field of geohydrology to trace fluid movement and dispersion. As such they should be physically compatible with the medium, and interpretation should be a function of the medium and not of the tracer. The scope of the study on the Campus Terrain, as discussed under Chapter 1, is restricted to the application of artificial tracers.

## 2.3 Selection

An ideal groundwater tracer has the following characteristics:

(i) It is conservative and will follow the movement of the water without loss from the flow due to physical or chemical processes, like adsorption on sediments or equipment.
(ii) It is non-toxic, and can be applied with no administrative or legislative requirements.
(iii) It is detectable with a high sensitivity and can be measured accurately *in-situ* in the field.
(iv) It does not contaminate the terrain of investigation and does not affect the results of further tests.
(v) It is inexpensive and analysis costs are low.

No ideal tracer exists, however, and the selection of the tracer should be based on the most important considerations, which will depend on the nature of the investigation. For instance, if the advective properties of a medium are investigated, care should be taken that the tracer is physically compatible with the fluid, while a study of the diffusive flow through a porous medium will require attention to the diffusion constants and molecular size of the tracer.

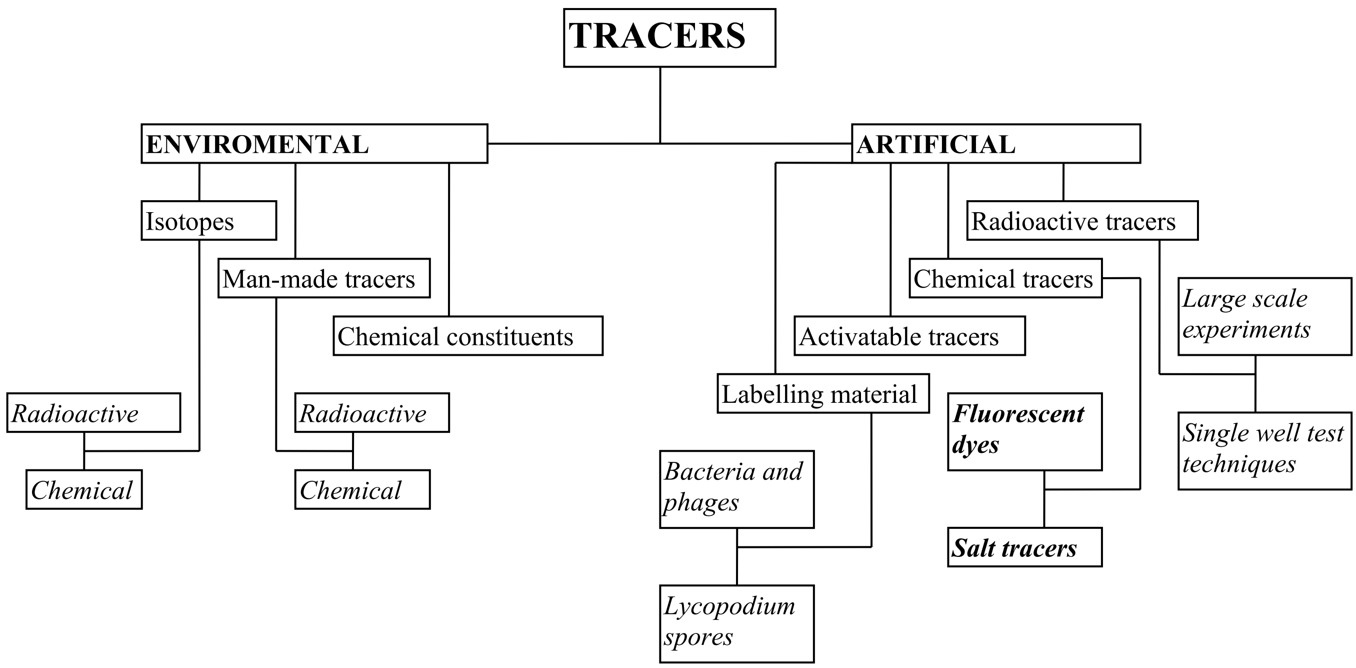
## 2.4 Classification

Artificial tracers can be categorised by their method of analysis, as follows:

(i) Radioactive tracers: These elements are detected by means of their radioactive emissions.
(ii) Activatable tracers: These elements are stable during use, but activated to emit radioactivity for analysis.
(iii) Chemical tracers: Detection is based on mass (mass spectrometry), orbital electron arrangements (chemical reactions) or shell binding energies (energy adsorption or emission properties).

(iv) Particulate tracers: Tracers detected by collection, weighing or counting of individual particles

**Figure 2.1: Classification of groundwater tracers: Those used on the Campus Terrain are printed in italics.**




A brief discussion on the different types of tracers categorised above will follow to illustrate the properties of each type. The classification of groundwater tracers is illustrated in Figure 2.1.

### 2.4.1 Radioactive tracers:

Although some chemical tracers are more toxic than radioactive tracers, and their concentrations do not decrease due to radioactive decay, all radioactive tracers are subjected to legislation in most countries, and the disadvantage of a negative general public reaction to the use of radioactivity limits their applicability. Other associated disadvantages are the need to transport the tracer in special containers and the use of specialised personnel.

In spite of this, radioactive tracers have some special advantages:

(i) A high sensitivity and precision of detection,

(ii) for $\gamma$- emitting tracers, the possibility of accurate *in situ* measurements, and,

(iii) small volumes of tracer are needed for injection in the aquifer which minimises disturbance to the flow system.

Tritium ($^3H$) is commonly used as an environmental and artificial tracer, and since it forms part of the water molecule it will faithfully follow the movement of water. Since the vapour pressure of HTO is lower than $H_2O$, it is extensively used in evaporation and recharge studies. For analyses, samples have to be taken and returned to a laboratory for measurement by liquid scintillation counting or gas counting.

The halides, $^{82}Br$ and $^{131}I$, are the most commonly used radioactive tracers, as anionic bromide or iodide respectively. Results of these tracers are similar to that of tritium.

### 2.4.2 Activatable tracers:

These are stable elements, which are capable of neutron irradiation over a short term during activation to produce a radioactive isotope, and they are generally used as anionic complexes. During irradiation measurements are made with a high resolution Ge(Li) detector. These tracers present no radioactive hazard to natural waters, and no authorisation is required for use. High levels of sensitivity are reached but the costs of analyses are very high. Elements which are applied as activatable tracers are indium, bromide and iodine.

### 2.4.3 Chemical tracers:

They have the advantage that they can be used without authorisation, and that field detection equipment is readily available (especially for the dyes). For some tracers, like many salt tracers for instance, analysis requires sample treatment and has to be carried out in the laboratory.

Inorganic salt tracers, such as the anions $Cl^-$ (NaCl) and $Br^-$ (NaBr, KBr), are commonly used to trace groundwater movement, while the cations are often lost by ion-exchange. Iodide is not as conservative as the bromides or chlorides, but the background concentration is usually less than 10 μg/l compared to the 100 μg/l for bromide and 30 mg/l for chloride. $NaNO_3$ is easy to dissolve and to detect with ion-selective electrodes, but background concentrations of $NO_3$ may be a problem. NaF and KF are detectable in small concentrations, but losses due to precipitation with calcium to form $CaF_2$ could occur. $Na_2SO_4$ too, could precipitate with calcium to form $CaSO_4$ while LiCl is expensive and also takes part in ion exchange reactions with the earth alkalis.

Organic dyes are frequently used as groundwater tracers, since the sensitivity of a fluorometric analysis is very high, and dye tracers are detectable in very low concentrations compared to salt tracers. The main advantage is that field measurements can be performed with filter fluorometers. Some of the dyes, however, are not conservative, and are influenced by the water quality.

Sensitivities as low as 0.001 - 0.1 μg/l could be reached for fluorocarbon compounds ($CCl_3F$, $CCl_2F_2$, $CBr_2F_2$) in groundwater, with detection methods such as gas chromatography. These substances can be used as convenient non-toxic tracers at concentrations as low as a couple of ppm. Atmospheric contact during sampling however, must be avoided to prevent tracer losses due to gaseous exchange, which make them mostly unattractive as artificial groundwater tracers.

### 2.4.4 Particulate tracers:

Particulate tracers have been used in a number of groundwater tracing experiments. The particles may be of biological, botanical, geological or man-made origin, and could be detected by means of counting, fluorimetry, etc. Micro-organisms such as *Serratia marceescens*, bacteriophages, spores of Lycopodium (club moss) have been used in groundwater tracing studies.

Because of the particulate importance of fluorescent dyes, the following chapter will be devoted to it.

# CHAPTER 3

# FLUORESCENT TRACERS

## 3.1 Introduction

Fluorescent dyes are often used as groundwater tracers and have certain advantages over other chemical tracers (Benischke, 1991):

(i) They are visually detectable at high concentrations, which allows them to be used in quantitative experiments.
(ii) They are detectable at much lower concentrations than salt tracers (up to a factor of 1000 times lower).
(iii) The instrumental detection is quick and easy and can be performed in the field without time consuming sample preparation.
(iv) The cost of the analyses of the dye tracers with a filter fluorometer is cheaper than analyses of the salt concentrations with a flame photometer.
(v) Dyes can be detected in extracts of activated carbon so that they can be used in quantitative studies

## 3.2 Principles of fluorescence

A certain amount of energy (excitation energy) is required to excite the electrons of the dye molecules to higher energy level (Benischke, 1991). This is provided by light of a certain wavelength (excitation wavelength) falling onto the solution. If the electrons are excited to this higher energy level they tend to return to their ground state (lowest possible energy level) losing a defined quantum of energy as fluorescent light (emissions) of a certain wavelength. The wavelength of the emission energy is always longer than that of the excitation energy and thus of lower energy.

A summary of a paper by Smart and Laidlaw (1977) on the evaluation of the eight most popular dyes used as groundwater tracers, is presented in this chapter.

The eight dyes are: Amino G acid, photine CU (blue fluorescent dyes), fluorescein, lissamine FF, pyranine (green fluorescent dyes), rhodamine B, rhodamine WT and sulpo rhodamine B (orange fluorescent dyes).

**Table 3-1: Generic and alternative names of the tracer dyes**

| Name in text | Colour Index no. | Generic name | Alternative name |
|---|---|---|---|
| **Blue dyes** | | | |
| Amino G acid | | | 7-amino 1.3 napthalene |
| Photine CU | | CI fluorescent brightner 15 | |
| **Green dyes** | | | |
| Fluorescein | 45 350 | CI acid yellow 73 | Fluorescein LT<br>Uranine<br>Sodium fluorescein |
| Lissamine FF | 56205 | CI acid yellow 7 | Lissamine yellow FF<br>Brilliant sulpho flavine FF<br>Brilliant acid yellow 8G |
| Pyranine | 59040 | CI solvent green 7 | D&C green 8 |
| **Orange dye** | | | |
| Rhodamine B | 45170 | CI basic violet 10 | |
| Rhodamine WT | | | |
| Sulpho Rhodamine B | 45100 | CI acid red 52 | Pontacyl brilliant pink B<br>Lissamine red 4B<br>Kiton rhodamine B<br>Acid rhodamine B |

## 3.3 Analysis

### 3.3.1 Instruments:

Analysis can be performed with a spectrofluorometer or a filter fluorometer. Spectrofluorometers are very specific but their general application is ruled out by their expense, complexity and delicacy.

Filter fluorometers, in contrast, are only moderately expensive, simple to use and sufficiently robust for use in the field, while sensitivity is comparable to that of the spectrofluorometer. Most tracing work has been carried out by filter fluorometers like the Turner111 or the Aminco Bowman fluorometer.

In a filter fluorometer, excitation energy is provided by a light source such as a low - pressure mercury lamp. The light passes through a primary filter before entering the sample compartment, where it is absorbed by the dye sample and re-emitted at a longer wavelength as fluorescence. The light passes through a secondary filter perpendicular to the primary light path. The amount of light passing through the secondary filter is measured by a photomultiplier to produce a read-out of the result.

### 3.3.2 Filters and lamps:

The selection of the best primary and secondary filter combination set will maximise the sensitivity of the analysis. Normally, the filters are selected to have the peak transmission at the excitation and emission wavelengths of the specific dye. In some cases, however, the difference between the excitation and the emission maximum is so small that overlapping of the primary and secondary filters occurs, causing the light that enters the photomultiplier to be scattered and producing a weak read-out. In such cases, the sample has to be excited at a wavelength other than the excitation maximum.

**Table 3-2: Excitation and emission maxima of the tracer dyes and filter combinations for their analyses.**

| Dye | Maximum Excitation nm | Maximum Emission nm | Primary filter | Mercury Line nm | Secondary filter |
|---|---|---|---|---|---|
| **Blue fluorescent dyes** | | | | | |
| Amino G acid | 355(310) | 445 | 7-37* | 365 | 98** |
| Photine CU | 345 | 435(455) | | | |
| **Green fluorescent dyes** | | | | | |
| Fluorescein | 490 | 520 | 98** | 436 | 55** |
| Lissamine FF | 420 | 515 | | | |
| Pyranine | 455(405) | 515 | | | |
| **Orange fluorescent dyes** | | | | | |
| Rhodamine B | 555 | 580 | 2×1-60* + 61** | 546 | 4-97* +3-66* |
| Rhodamine WT | 555 | 580 | | | |
| Sulpho-rhodamine B | 565 | 590 | | | |

Figures in parenthesis refer to secondary maxima. For all spectra, pH = 7.

*Corning filter

**Kodak Wratten filter

Two types of filters are available, the glass type, which is stable under high light intensities with a broad transmission waveband, or the gelatine type, which has a much sharper resolution but which is less stable to high intensity light. Gelatine filters scratch easily and are badly affected by heat, which is a problem in the Turner111 fluorometer where the primary filters are close to the light source. Gelatine filters should, therefore, be glass mounted.

### 3.3.3 Sensitivity and detectability:

The sensitivity of the analysis depends on the efficiency of the dye to convert the excitation energy into fluorescence and the transmission of the filter combination. The detectability depends, however, on the background or blank fluorescence value. The actual dye concentration of the natural water is determined by subtracting the blank fluorescence value from actual fluorescence reading.

**Table 3-3: Sensitivity and minimum detectable concentrations for the tracer dyes.**

| **Dye** | **Sensitivity* $\mu g^{-1}$/scale unit** | **Background Reading** scale units 0-100** | **Minimum Detectability $\mu g^{-1}$** |
|---|---|---|---|
| Amino G acid | 0.27 | 19.0 | 0.51 |
| Photine CU | 0.19 | 19.0 | 0.36 |
| Fluorecein | 0.11 | 26.5 | 0.29 |
| Lissamine FF | 0.11 | 26.5 | 0.29 |
| Pyranine | 0.033 | 26.5 | 0.087 |
| Rhodamine B | 0.010 | 1.5 | 0.010 |
| Rhodamine WT | 0.013 | 1.5 | 0.013 |
| Sulpo rhodamine B | 0.061 | 1.5 | 0.061 |

*At pH =7

** For distilled water

Rhodamine WT and rhodamine B have the lowest minimum detectable concentrations, and sulpho rhodamine B is detectable at lower concentrations than pyranine, despite the latter's higher sensitivity. This illustrates the significance of the absolute value of the background readings.

### 3.3.4 Temperature:

The fluorescence of the rhodamine dyes and photine CU are significantly affected by temperature variations and corrections may be needed in quantitative studies. Temperature variation may also affect the operation of filter fluorometers, as the machine gradually warms up after switching on. If continuous monitoring work is carried out in the field, occasional discrete samples should be taken for later laboratory analysis, so that calibration of the fluorometer can be done.

## 3.4 Effect of water quality

The fluorescence of some dyes is significantly influenced by the water quality. For example structural changes might occur when the pH decreases. Protonation of anionic dyes occurs at low pH values, causing the dye to be adsorbed on the negatively charged sediment surfaces. For pH variation between pH 4 and pH 10 no problems are encountered with lissamine FF and sulpho rhodamine B. The fluorescence of rhodamine B and rhodamine WT is significantly affected at pH values below 5, amino G acid below 6 and fluorescein and photine CU below pH 6.5. Pyranine shows excessive variation in fluorescence with pH changes normally encountered in natural waters.

Apparent or real background fluorescence could be a problem in tracer studies. There are two possible sources for background fluorescence: (i) suspended sediment and (ii) natural fluorescence.

The presence of suspended sediment could contribute to fluorescence because of light scattering by the sediment particles, but this effect is relatively small compared to other sources of background fluorescence. This should not be a problem for sediment concentrations below 1000 mg/l except if the sediment is extremely fine. If the sediment concentration is a problem, the sample could be decanted by allowing the sediment to settle over a period of 10 - 20 hours. If the sample analysis is required immediately, the sample could be centrifuged or diluted with distilled water.

Natural background fluorescence has frequently been wrongly ascribed to the presence of algae. The majority of algae contain chlorophyll which has a fluorescence peaking at 650 nm, causing little interference, and then only with the orange filter combinations. Background fluorescence in natural waters occurs mainly because of the presence of organic and colloidal matter which frequently contain known fluorescent structures. The background values for the blue and green wavelengths are much higher than the orange. Attempts to separate the background from the dye fluorescence by physical and chemical techniques have proved unsuccessful because of the chemical similarity of the compounds and the dye itself.

## 3.5 Non-adsorptive dye loss

Fluorescence is caused when the dye is excited to a higher energy level and reverts to a lower energy state through the emission of light of a longer wavelength. At the high energy state the dye will also take part in chemical reactions with other species, causing it to decompose. The very fast decay rate of photine CU under all light conditions restricts its use as a quantitative tracer. Fluorescein and pyranine are unstable under bright sunlight conditions while the orange dyes, lissamine FF and amino G acid have been reported to have low photochemical decay rates and no correction is required for tests up to 1 week in duration.

Dye loss due to biodegradation should be considered only when tracing work is done in biologically hostile environments, like activated sludge systems where large populations of micro-organisms are present. For the majority of tracing work, this is not considered to be a problem.

## 3.6 Adsorptive dye loss

Adsorption onto sediment surfaces is mainly irreversible; therefore a high resistance to adsorption is of paramount importance for the selection of the dye tracer.

Rhodamine B is, contrary to other the dyes discussed here, cationic of nature and is readily adsorbed on the negatively charged sediment surfaces. This precludes its worth as a quantitative tracer. Pyranine and amino G acid are resistant to adsorption on both organic and inorganic materials, while lissamine FF and sulpho rhodamine B have a higher resistance to organic materials. Rhodamine WT and fluorescein have a moderate resistance to both types of substrate, but photine CU, which is resistant to adsorption on inorganic sediments, shows a marked affinity for organic materials.

Because of its cationic nature, rhodamine B is also adsorbed onto glass surfaces. All the other cationic dyes are repelled by soft (Pyrex) or hard glass surfaces and the dyes can be stored in these containers for periods of up to 10 weeks. No losses were found where lissamine FF or rhodamine WT were stored in polythene bottles, or came into contact with rubber bungs or ‘Parafilm’ laboratory sealing film (Smart and Laidlaw, 1977).

### 3.7 Toxicity

Rhodamine B is toxic to aquatic organisms, probably because it is adsorbed on living tissue, due to its cationic nature. It is recommended that it should not be used as a water tracer. The manufacturers of lissamine FF indicate that it is unlikely to cause unusual toxic hazards. Rhodamine WT, fluorescein and photine CU have relatively low toxicity levels while that of sulpho rhodamine is slightly higher. Fluorecein and pyranine have been certified for use in externally applied drugs, lipsticks and cosmetics in the United States by the Food and Drug Administration.

# CHAPTER 4

# THEORY OF BOREHOLE DILUTION AND RADIAL CONVERGENT TESTS

## 4.1 Borehole dilution test

The dilution test aims to relate the observed rate of a tracer dilution in a borehole to the average groundwater velocity in the aquifer (Freeze and Cherry, 1979). Such a dilution curve is depicted in Figure 4.1. Since the decline of the tracer concentration in the borehole is an exponential process, a linear relation is displayed a semi-log graph.

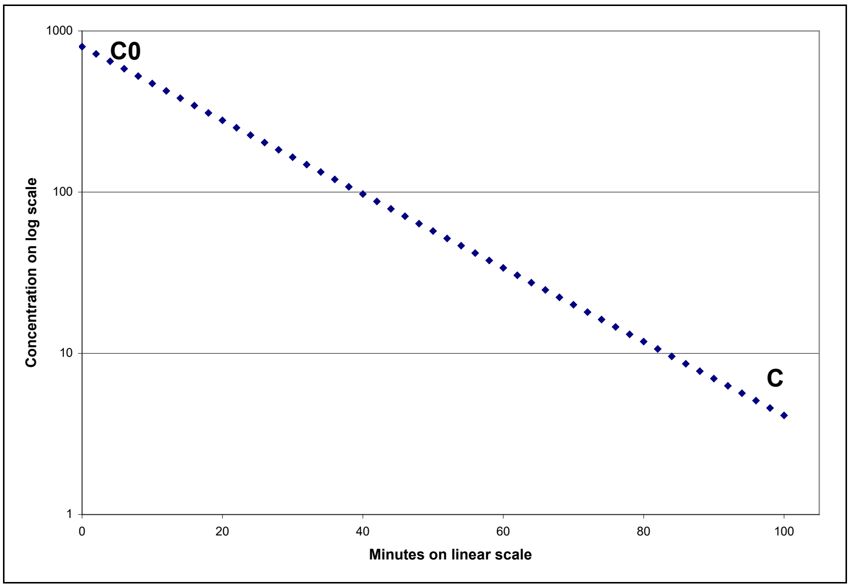



**Fig. 4.1: Illustration of a typical dilution curve**

The volumetric flow rate of groundwater through the borehole, Q, is computed with the aid of the following equation:

**Eq. 4.1:**

$$Q = -\frac{W}{\alpha t} \ln \frac{C}{C_0}$$

With: W: The volume of fluid contained in the test section

$C_0$: Tracer concentration at t=0.

C: Tracer concentration at t=t.

$\alpha$: Borehole distortion factor (between 0.5 and 4, for fractured rocks 2)

t: Time

The Darcy velocity, q, is defined as the volume of water flowing per unit time through a unit cross sectional area normal to the direction of flow. The Darcy velocity is a flux and not a velocity, although it's physical dimensions is that of velocity. It is expressed as:

**Eq. 4.2**

$$q = Q/A$$

Where the A is the cross sectional area normal to the flow direction:

**Eq. 4.3**

$$A=\pi r_w L$$

With: $r_w$: Well radius

L : Length of the test section

Darcy's Law describes the volumetric flow rate through a porous medium as:

**Eq. 4.4**

$$Q=KAi$$

With: K: Hydraulic conductivity

i: Hydraulic gradient ($\Delta H/\Delta L$)

By substituting KAi for Q (equation 4.4) in equation 4.1, the hydraulic conductivity of a porous medium could be computed from a dilution test according to:

**Eq. 4.5**

$$K = -\frac{W}{aAit}\ln\frac{C}{C_0}$$

Since the hydraulic gradient appears in equation 4.5, it is in practice impossible to derive the K-value from a dilution test where only one borehole is available. Problems are also encountered in establishing the gradient in secondary aquifers (for measurements made in open wells) because the piezometeric water levels which would be valid in accounting for the flux through the borehole are concealed, in many cases, by a hierarchy of pressure systems. Due to the sensitivity of the K-value derivations to the hydraulic gradient, it was found appropriate to introduce another entity for velocity/conductivity estimations.

Flow in a porous medium is restricted to that part of the area that is effectively transmitting water, in other words the product of A and the surface porosity of the medium. The seepage or pore velocity of a medium is defined as:

**Eq. 4.6**

$$v = \frac{q}{n_e}$$

With: q: Derived from the dilution test

$n_e$: Kinematic porosity, derived from a radial convergent tracer test

Velocity figures, derived from the dilution tests, require that sound estimates of the kinematic porosity should be ascertained with a radial convergent tracer test. This implies that the dilution test, if valid estimates of the hydraulic gradient or kinematic porosity are lacking, can only supply information regarding the volumetric groundwater flow rate in the aquifer at the position of the well point.

## 4.2 Radial convergent tracer test

A radial convergent flow field is created by pumping a well until steady state conditions are reached (Moench, 1989). A tracer is then quickly introduced in an injection well in the vicinity of the pumping well in such a way that the minimum disturbance of the flow field is caused, while the tracer breakthrough curve is monitored at the pumping well. An example of such a breakthrough curve is displayed in Figure 4.2. Analyses of the resulting breakthrough curves yield estimates of kinematic porosity, aquifer dispersivity and groundwater velocity. The convergent tracer test is attractive because it is theoretically possible to recover all the tracer from the aquifer. For 1-D uniform flow, the concentration distribution at any time is given by (Sauty *et al*., 1992) equation 4.7.

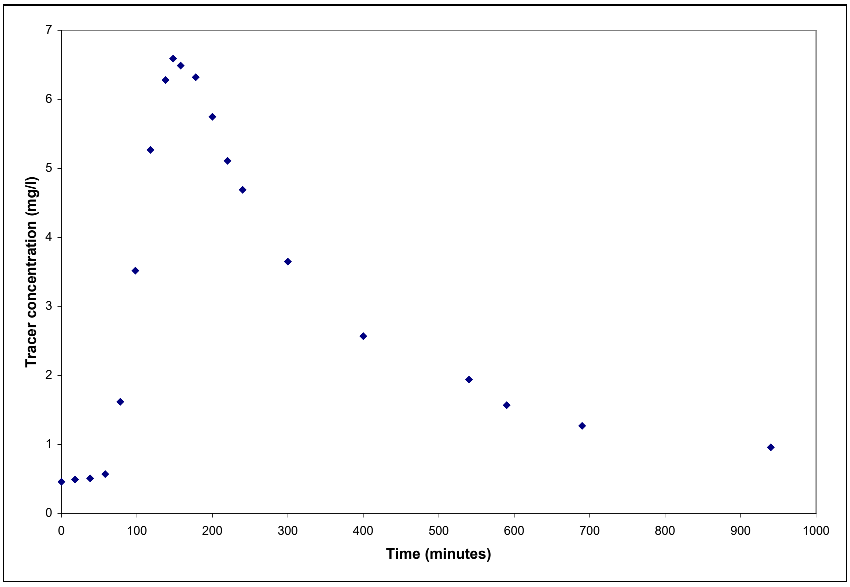



**Fig. 4.2 Ilustration of a breakthrough curve detected at the pumping well.**

**Eq. 4.7**

$$c(r,t) = \frac{\Delta M}{2Q\sqrt{\pi * \alpha_L * vt^3}} \exp\left[-\frac{(r-vt)^2}{4D_L t}\right]$$

With: $\Delta M$: Injected mass of tracer per unit section

$\Delta M$=Mass(kg)/Thickness(m)

A: Cross sectional area of groundwater flow (m$^2$)

$r$: Radial distance (m)

$\alpha_L$: Longitudinal dispersivity (m)

$D_L$: Longitudinal dispersion coeff. ($m^2/s$)

$v$: Pore velocity (m/s)

$Q$: Pumping rate of well ($m^3/s$)

The field method was improved in such a way that estimations of the kinematic porosity between the tracer injection well and the abstraction well are independent of measurements of hydraulic head. This is achieved by using the tracer injection well for a dilution test, which allows the darcy velocity q to be computed under the given hydraulic conditions. Since analyzing the tracer breakthrough curve enables the derivation of the velocity with the help of equation 4.7, the kinematic porosity can then be estimated with the aid of equation 4.6. Since the porosity is a fixed property of the medium, it could then be employed in estimating velocity from the results of dilution tests conducted under hydraulic conditions different from those of the tracer test. To yield the natural groundwater velocity, equation 4.6 is again applied. For this application the Darcy velocity (q) is derived from a dilution test performed under the natural flow regime. From the above it is evident that velocity and effective porosity estimations can be accomplished independently from hydraulic gradient measurements.

**Eq. 4.8**

$$v = \frac{Ki}{n_e}$$

With the aid of equation 4.8, the hydraulic conductivity of the rock fracture can be ascertained from the radial convergent tracer test q having been established in the tracer injection well under the given flow regime. The kinematic porosity and pore velocity are then ascertained from equations 4.6 and 4.7 respectively, and substituted in the conductivity formula (equation 4.8). However, since this expression of the hydraulic conductivity (or Darcy's Law) requires knowledge of the hydraulic gradient, the hydraulic conductivity of a rock fracture/fracture system could only be established within a range of certainty having upper and lower boundaries, according to the degree of certainty in hydraulic head.

Poor well efficiencies in the abstraction borehole are likely to contribute to this uncertainty of the actual ΔH value of the aquifer, as depicted in Figure 4.3.

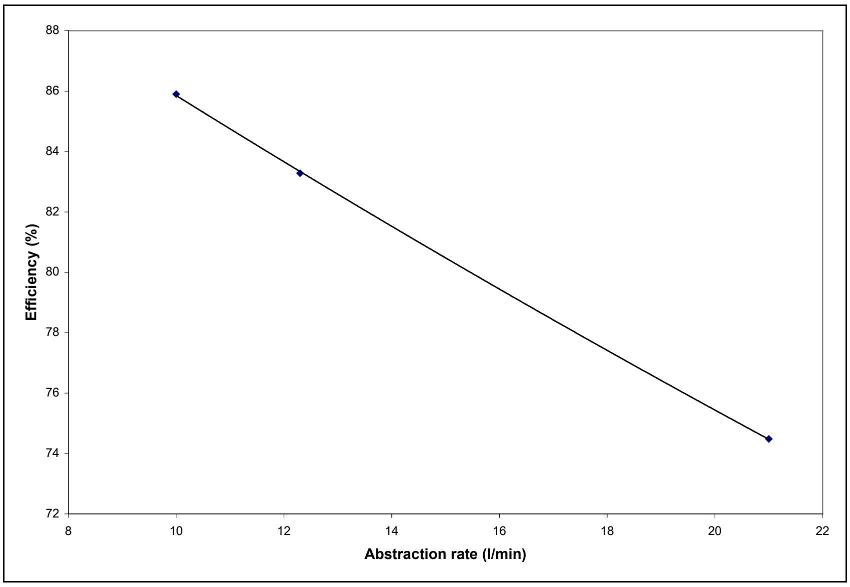



**Fig. 4.3: Well efficiency of UO5 on the Campus Terrain**

## 4.3 Borehole dilution test: Field method, instrumentation and interpretation:

As a rule, a single fracture is usually isolated with straddle packers. A tracer is then quickly introduced to the packer chamber and the tracer recession is monitored against time as the groundwater flow removes the tracer from the well bore. Due to the high costs of purchasing commercial packers, an alternative, cheaper technique was developed and utilised. By means of the open-well method, several fracture intersects could be tested at once. This technique does away with packer systems, yet the pump inlet position and the position of the pumping line outlet delimits the test section, illustrated in Figure 4.4. The test section, similar to packer systems, can then be then shifted upwards and downwards in the borehole to establish a vertical profile of groundwater through-flow.

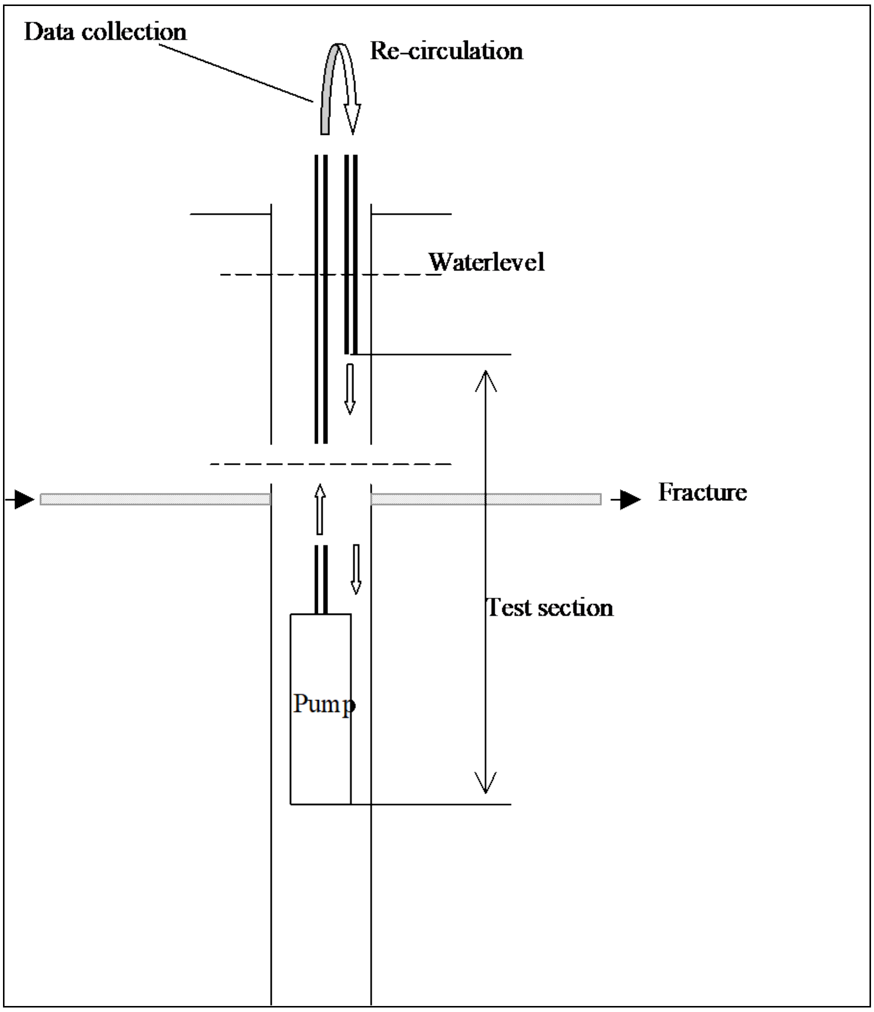



**Fig. 4.4: Illustration of the design of the open well dilution test.**

Circulation of the tracer plume through the test section is achieved by pumping the water to the surface, where the tracer concentration is ascertained before it is placed back into the borehole via the pumping injection line (Figure 4.4). This way, the necessity of expensive in-situ tracer detection systems, required for the packer systems, is eliminated. Since the decline in the NaCl concentration can be measured as electrical conductivity, table salt is found to be a very practical tracer for use in dilution tests.

Several tests were conducted to ascertain the best tracer injection procedure. Introducing the tracer in such a way as to establish a uniform distribution of tracer through the entire test section, was found to be a difficult task in practice. Instead, the best injection procedure established was to "spike" the segment to create a tracer plume that circulates through the system. Each peak of the measured field curve

(Figure 4.5) then depicts the moment the plume midpoint passes the data collection point, which is controlled by the flow-rate of the pump. By computing the amount of tracer that is removed from the borehole by groundwater throughflow each time the plume passes the data collection point, a tracer decline curve could be established (Figure 4.6). Interpretation of this curve is then done according to the procedures outlined.

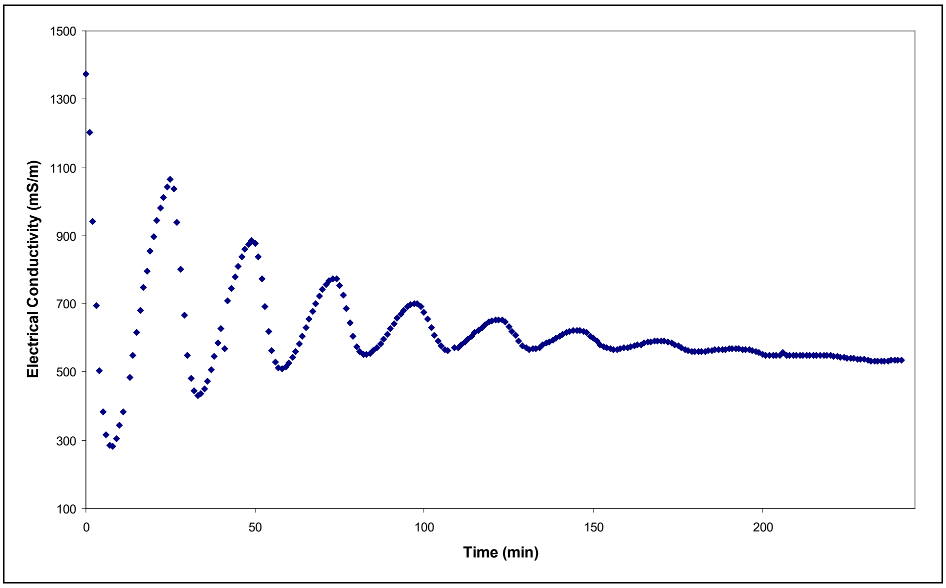



**Fig. 4.5: Measured field curve with spiked tracer injection.**

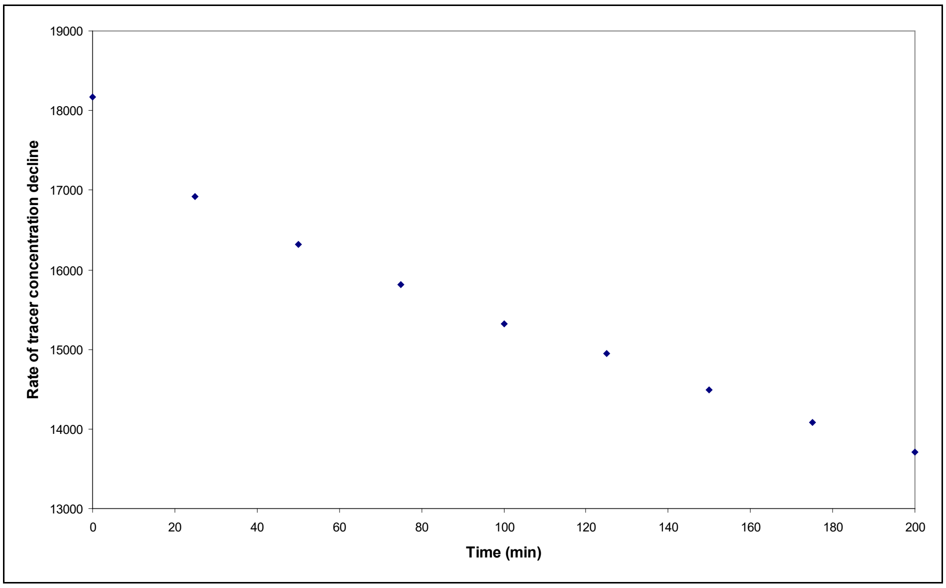



**Fig. 4.6: Integration below each of the measured tracer plumes allows the calculation of the rate of tracer concentration decline due to groundwater advection.**

The borehole dilution experiment is able to shed light on hydraulic conductivity while a tracer experiment under pumping conditions is able to reveal effective porosity and dispersivity of the aquifer involved.

# CHAPTER 6

# CASE STUDIES: TRACER TESTS

## 6.1 Introduction

This chapter focuses on numerous tracer and dilution tests that have been conducted in a number of different geological units, such as Karoo sediments, dolerite, dolomite, gabbro/norite, granites, and quartzite/gneiss. Experience has shown that due to uncertainties when measuring the hydraulic gradient in open wells, establishing the K-value of fractures is difficult. Due to this, the research discussed in this chapter has been directed towards establishing methods to ascertain groundwater velocities without the aid of hydraulic head measurements.

## 6.2 Campus Test Site

### 6.2.1 Saturated zone

A detailed discussion of the field method and the theory applied to the Campus Tracer Tests is provided in Chapter 4. The methodology is illustrated in Figure 6.2. In particular, the study focused on velocity and kinematic porosity assessment, the calculation of the hydraulic conductivity of rock fractures, validation of the parallel plate model and slug tests. Most of the experiments were conducted on the main fracture encountered in the sandstone unit (Botha *et al.*, 1998). A number of radial convergent tracer tests were conducted in the surroundings of UO5, which was used in each case as the abstraction well.

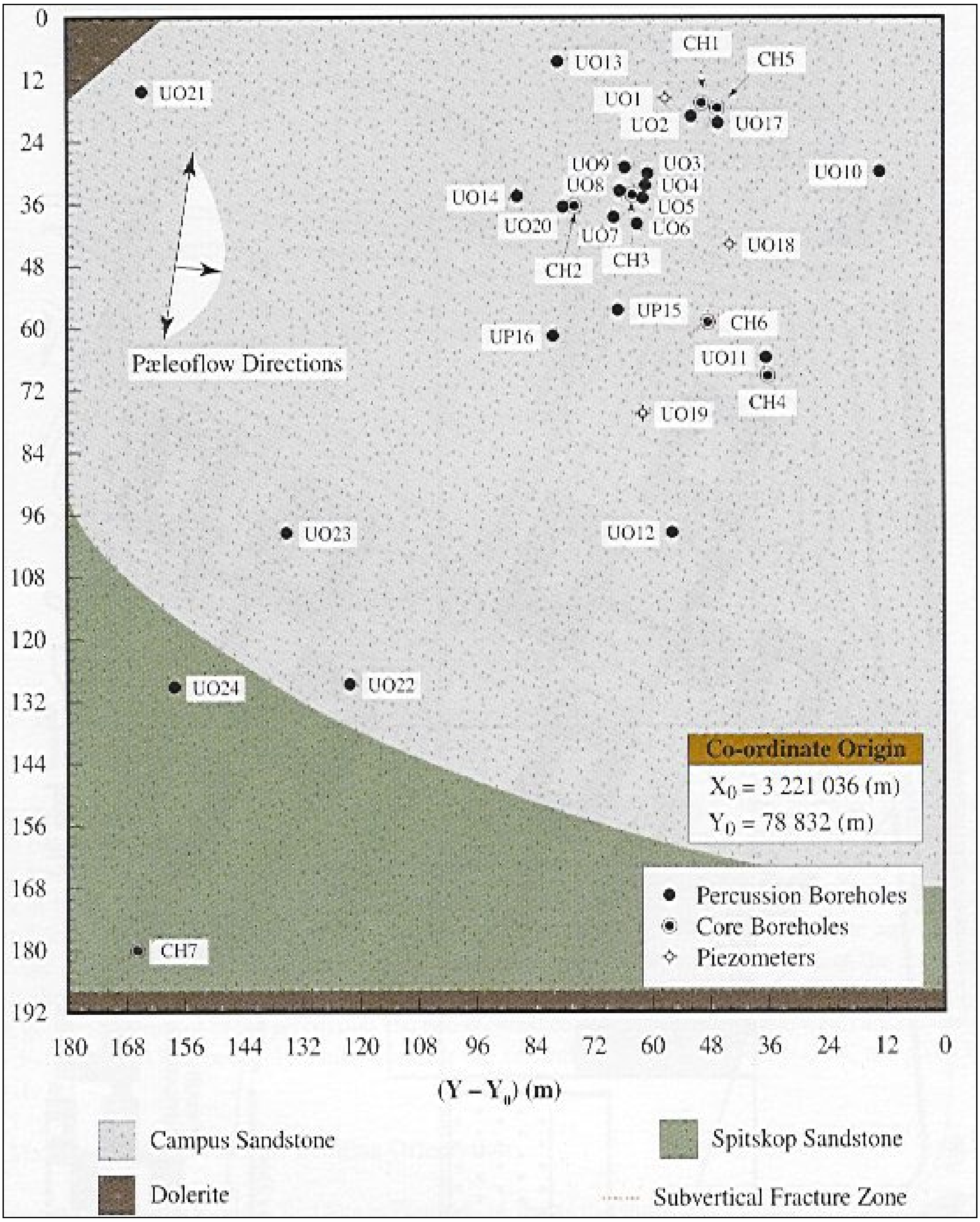



**Fig. 6.1: Borehole positions of the Campus Test Site**

The dilution tests focused specifically on the UO20 – UO5 borehole pair, the purpose being to study the behaviour of the fracture under different hydraulic circumstances as far as groundwater flow rates and velocities are concerned. The specifications of both the radial convergent, as well as the dilution tests, are summarized in Tables 6.1 and

6.2 respectively. It should be noted that for effective porosity estimation, these two tests are employed simultaneously, as depicted in Figure 6.2.

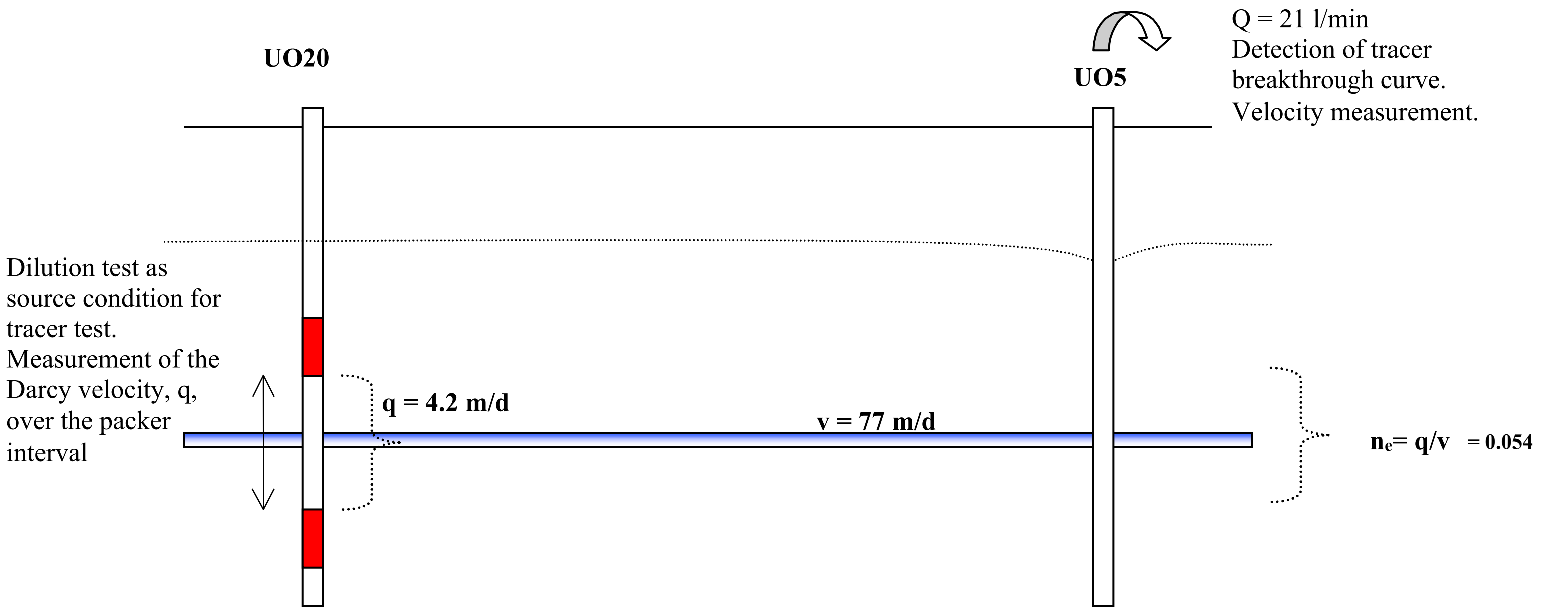
**Table 6.1: Summary of the field methods and results of the Campus tracer tests**

| **TEST ID** | **CV1 UO20-UO5** | **CV2 UO20-UO5** | **CV3 UO6-UO5** | **CV4 UO8-UO5** | **CV5 UO7-UO5** |
|---|---|---|---|---|---|
| **FIELD PROCEDURE** | | | | | |
| Mass (g) | 35.8 | 0.50 | 1.5 | 1.25 | 0.9 |
| Distance between wells (m) | 15 | 15 | 4.5 | 4.5 | 6.3 |
| Abstraction rate (l/min) | 21 | 12.3 | 13 | 13 | 13.5 |
| **6.1 RESULTS** | | | | | |
| Mean pore velocity (m/d) | 76.65 | 67.55 | 22.57 | 29.03 | 30.195 |
| Longitudinal dispersivity | 2.65 | 2.24 | 0.84 | 1.255 | 1.0868 |
| Aquifer thickness (m) | 0.173 | 0.178 | 0.055 | 0.103 | 0.0695 |

**Table 6.2: Field method and results of the Campus dilution tests**

| **TEST ID** | **DT1** | **DT2** | **DT3** | **DT4** | **DT5** |
|---|---|---|---|---|---|
| **FIELD PROCEDURE** | | | | | |
| Flow regime | Natural | Natural | Forced | Forced | Forced |
| Dilution well | UO20 | UO20 | UO20 | UO20 | UO20 |
| Abstraction well | None | None | UO5 | UO5 | UO5 |
| Well conditions | Packer | Packer | Packer | Packer | Open |
| Distance from abstraction well (m) | | | 15 | 15 | 15 |
| Abstraction rate at pumping well (l/min) | | | 10 | 21 | 13.5 |
| Position in aquifer | Fracture | Matrix | Fracture | Fracture | Fracture |
| **RESULTS** | | | | | |
| Darcy velocity (m/d) | 1.459 | 0.02 | 2.92 | 4.2 | 0.406 |
| Kinematic porosity | 0.054 | | 0.054 | 0.054 | 0.0065 |
| Groundwater flow rate through well (l/d) | 240 | 3 | 480 | 693 | 556 |
| Pore velocity (m/d) | 27 | | 54 | 77 | 62 |

**Figure 6.2: Field methology of the Campus tracer tests to establish the groundwater velocity and kinematic porosity.**



It was found that the peak of the tracer breakthrough does not represent the mean adjective velocity of the fracture zone, since for the same velocity, the peak of simulated breakthrough curves is displaced in time for different values of longitudinal dispersivity.  Therefore, a model was developed to analyse tracer breakthrough curve for pore velocity, longitudinal dispersivity and the aquifer thickness according to equation 4.7 in chapter 4.

Analyses of the tracer breakthrough curves produced fracture velocities that varied under the different pumping rates between 22 and 76 m/d (Figure 6.3).

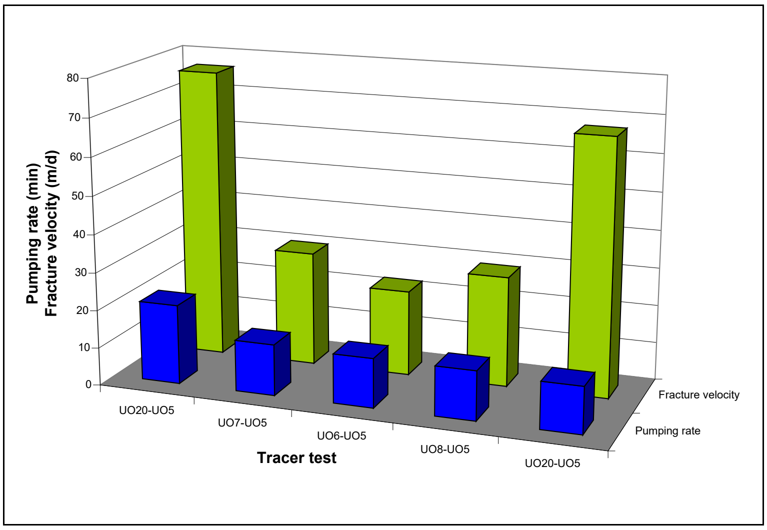



**Figure 6.3: Comparison of the fracture velocities and the pumping rates.**

Analysis of the dilution tests reveals a remarkable linear relationship between the groundwater flow rate through UO20, the dilution well, and the abstraction rate in UO5, the pumping well, which are located 15 m apart from each other.  The measured baseflow in the aquifer, established at the position of UO20, was estimated as 240 l/d with a natural pore velocity of 27 m/d (Figure 6.4).

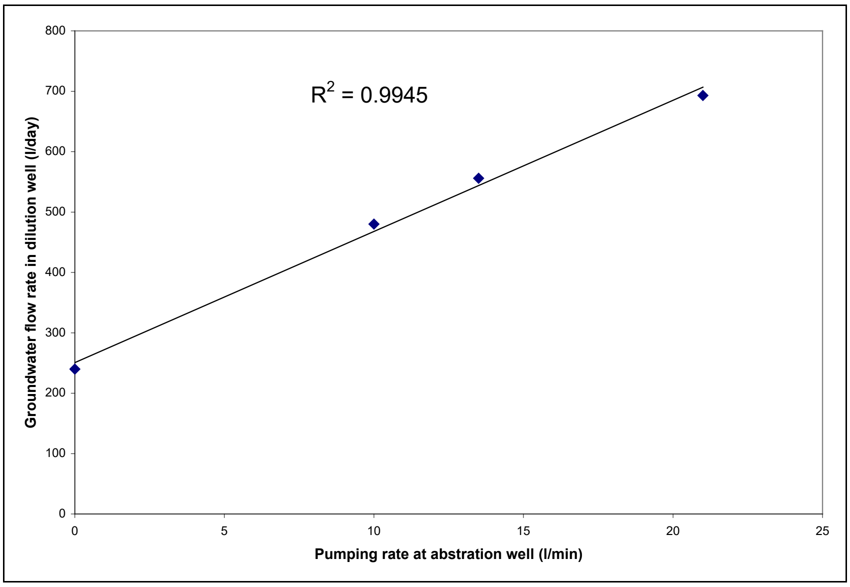



**Fig. 6.4: Groundwater flow rate at UO20 versus the pumping rate in UO5.**

The estimation of the fracture velocities from the dilution tests (Figure 6.5) was enabled by kinematic porosity measurements obtained from the radial convergent tracer tests (Figure 6.2). The methodology implies that the Darcy velocity, q, and the actual pore velocity can be measured with a tracer test to produce the effective porosity. This porosity value is then applied to dilution tests under different hydraulic conditions to yield the pore velocity under a certain hydraulic gradient. Note that this method is independent of hydraulic head measurements. It seems that the magnitude of these dilution test velocities are highly plausible, since the resulting velocity from the DT5 (dilution test) at an abstraction rate of 10 l/min in UO5, is 54 m/d, which is in agreement with a value of 62 m/d acquired from a tracer test (CV2) at 12.3 l/min. These tests were conducted independently of each other.

Figure 6.6 illustrates that the estimated porosity value is dependent on the thickness of the fracture zone. However, the computation of the hydraulic conductivity according to equation 4.8 in Chapter 4 requires the $n_E$-value at the scale of the fracture thickness (Figure 6.6), which in this case is 0.173m. The corresponding values for q and $n_E$ of the fracture then turn out to be 10.4 m/d and 13.5% respectively.

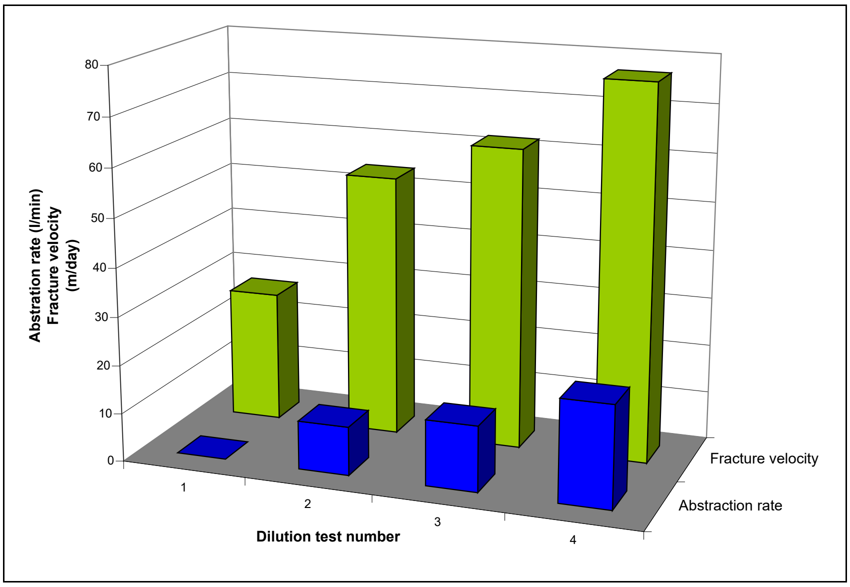



**Fig. 6.5: Groundwater velocities, established at UO20 at different pumping rates in UO5.**

### Validation of equivalent apertures

Velocity estimates from the dilution tests are valid for porosity measurements obtained from the radial convergent test. This implies that in practice, a convergent test should be performed to establish the kinematic porosity, which is then applied to a dilution test under natural conditions to estimate the baseflow velocity of the aquifer. Since the field method would be simplified significantly if the convergent tracer test could be substituted with a hydraulic test, the validity of the hydraulic aperture to replace kinematic porosity in the velocity equation of the dilution test was investigated.

The concept of the hydraulic aperture, idealizes the rock fracture as two parallel plates separated with a constant thickness 2b, called the parallel plate model. It is related to the transmissivity of the fracture and is derived from measurements of volumetric flow and pressure drop, and does not involve any tracer transport. For radial flow, $2b_H$ is given by (Tsang, 1992):

**Eq. 6.1**

$$2b_H=\{(6\mu Q/(\pi\gamma\Delta H))\ln(r_1/r_0)\}^{1/3}$$

with: $r_0$ = pumping well radius

$r_1$ = radius to the observation well

$\Delta H$ = hydraulic head

$\gamma$ = weight density of the fluid = 9804 n/m$^3$ at 8°C

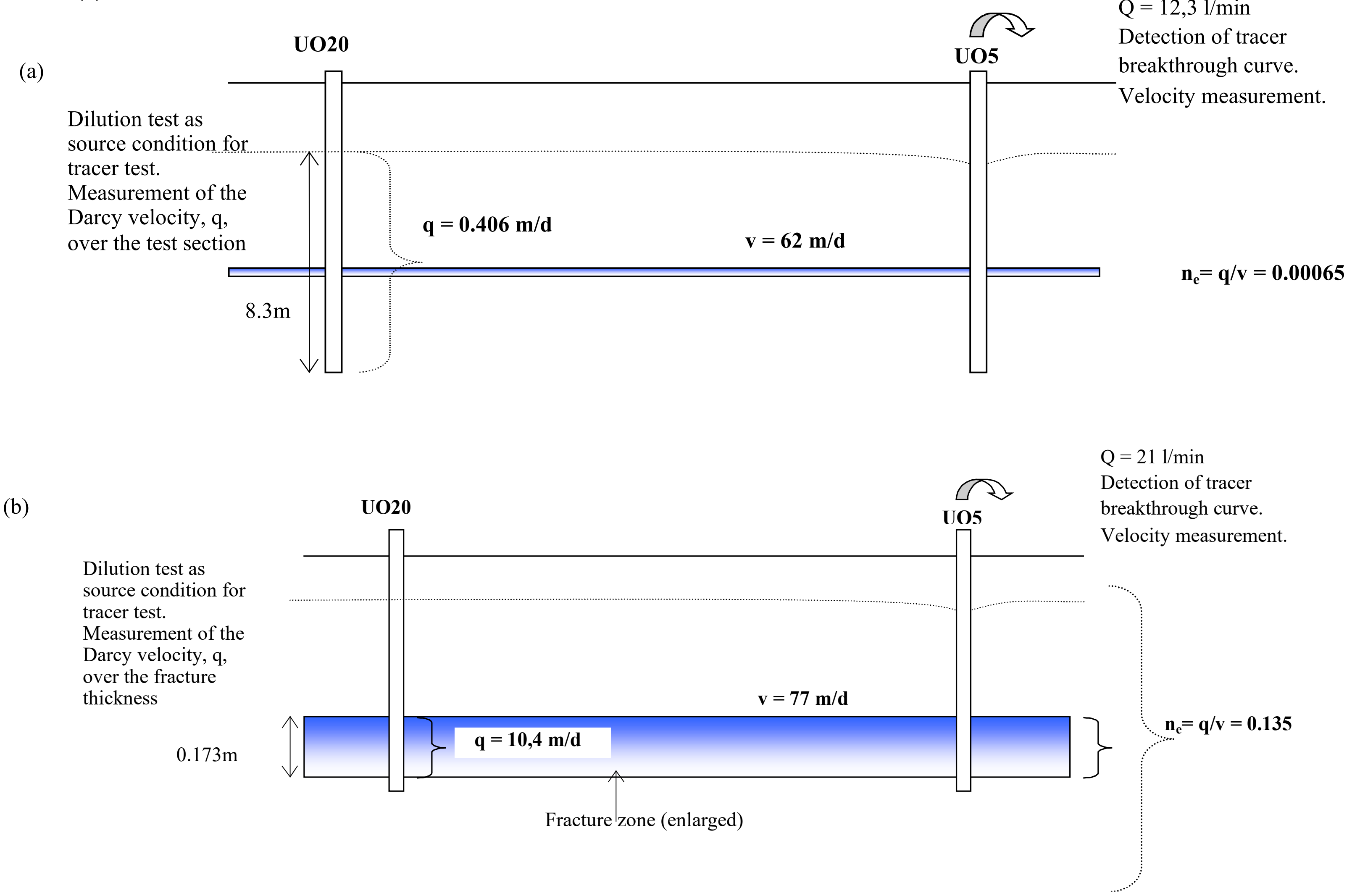
$\mu$ = viscosity of the fluid = $1{,}386 \times 10^{-3}$ N.s/m$^2$ at 8°C

Q = volumetric flow rate.

The hydraulic aperture between UO20 and UO5 was established by means of a pumping test, and hydraulic measurements were made in both wells. Under the same hydraulic circumstances the real pore velocity was obtained from a tracer test, and a dilution test was conducted in UO20. However, from the consequent $2b_H$ of 1.15 mm a velocity of 270 m/day was calculated from the dilution test, which does not correspond with the real velocity of 77 m/d. The same was done for another aperture model, the frictional loss aperture (Tsang, 1992), which requires the measurement of the hydraulic head and the mean residence time of the tracer transport. This time, the resulting values for $2b_F$ and fracture velocity were 0.5 mm and 500 m/d respectively.

Since both models incorporate measurements of $\Delta H$, these inconsistencies with measured velocities are due to incorrect hydraulic head measurements made in open wells, as well as non-linear well losses in the pumping well. Nevertheless, increasing experimental evidence has demonstrated that the parallel plate fracture approach is not consistent with field observations since it fails to recognize the spatial heterogeneity of real fractures and is therefore too idealistic and obviously unrealistic (Johns and Roberts, 1991).

**Figure 6.6: Campus site: Estimation of the kinematic porosity at the scale of (a) the saturated thickness in the borehole and (b) the fracture**



**Hydraulic conductivity of the fracture**

Packers were employed during a tracer test (CV2 and DT4) to establish the hydraulic gradient of the fracture. The following values for each parameter were accomplished:

| | |
|---|---|
| Pore velocity: | 77 m/d |
| Fracture thickness: | 0.173m |
| Darcy velocity of the fracture: | 10.4 m/d |
| Kinematic porosity of the fracture: | 13.5% |
| Hydraulic gradient: | 0.0235 |

Substituting the velocity and effective porosity into equation 4.8 in Chapter 4 yielded a K-value of 440 m/d for the fracture. The resulting T-value, for a fracture thickness of 0.17m is then 75 $m^2$/d.

Although T and K-values are fixed properties of the medium, it can be illustrated that the interpretation of the data yielding from slug tests, K-value of rock fractures is scale related, and often incorrect. The hydraulic conductivity is expressed by the relationship K=T/b where b is the fracture thickness. The T-value, which is the averaged hydraulic conductivity over the saturated thickness of the aquifer D is a known unit, since D is known. The K-value, however, is dependent on what is regarded as the fracture thickness, b, which cannot be derived from hydraulic measurements. For instance, for T=10 $m^2$/d, the resulting K-value is 10m/d if b is taken as 1m. However, if the actual fracture thickness is 0.01m, then the real K-value is 1000m/d. This shows that for fractured rock aquifers, the K-values are usually higher than the T-values. Since a tracer test is the only way to establish the effective thickness of a rock fracture, a T-value can only be established from slug tests if data are analysed with a method such as the Cooper method.

### 6.2.2 Unsaturated (vadose zone)

Tracer tests in the unsaturated zone were conducted to simulate flow from a septic tank or pitlatrine. Fluorecein was used as a tracer to determine vertical travel times of a pollution source through the unsaturated zone and the further migration thereof towards a production borehole.

Two shallow boreholes were drilled to a depth of 5 m on the upstream side of boreholes UO7 and UO23. Fluorecein was introduced into these "pitlatrine" boreholes and UO7 and UO23 were pumped for a duration of 1 hour per day at 0.2 L/s. The reason for the low abstraction rate was to keep conditions as natural as possible and not to create a too large cone of depression. While the borehole was pumped, the outflow was channelled through a fluorometer to measure the concentration of Fluorecein in the water.

Figure 6.7 is a graphic representation of the setup that was used on the Campus Test Site.

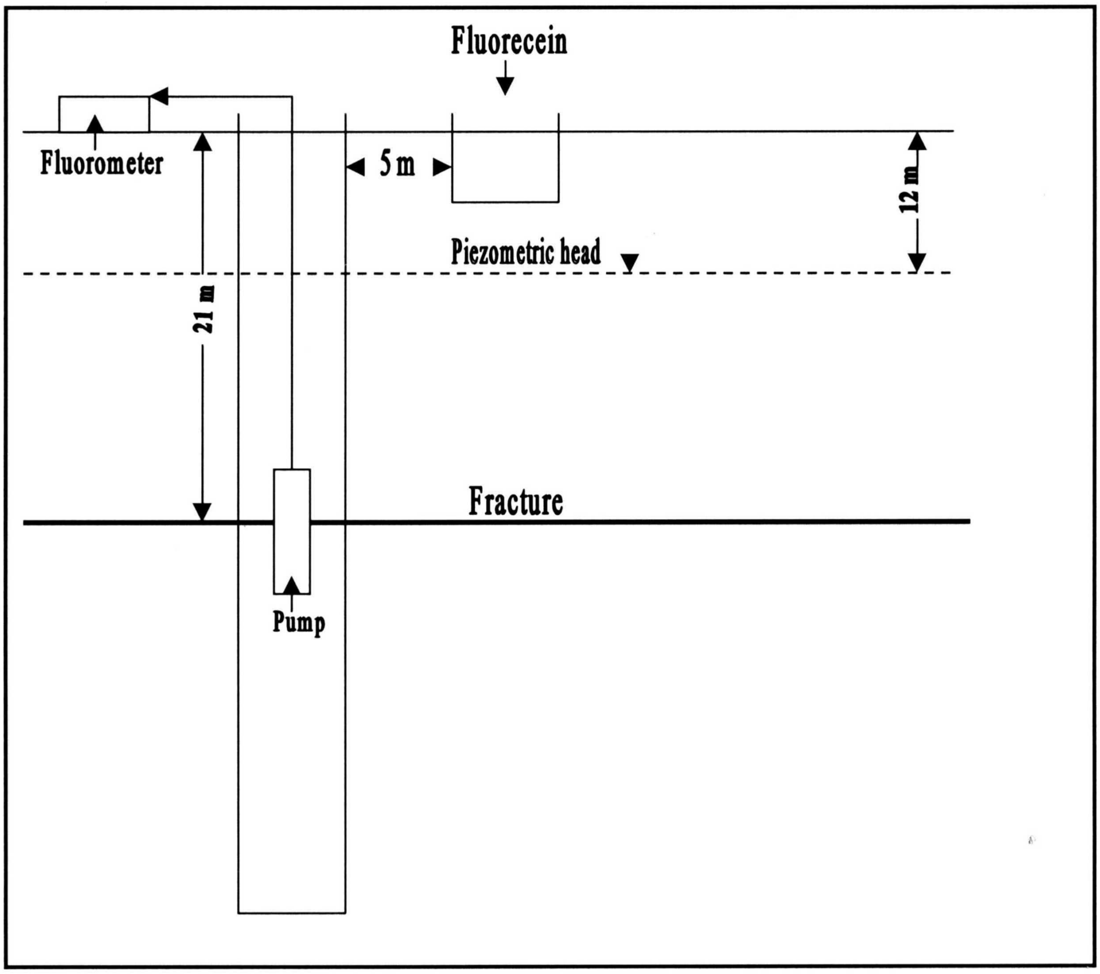



**Figure 6.7: Graphic representation of tracer test setup for the unsaturated zone at the Campus Test Site.**

When the migration of the tracer is considered, the importance of piezometers cannot be stressed enough. The "waterlevels" measured in boreholes UO7 and UO23 must be the piezometric head of the bottom and main aquifer. This can be seen if we

compare the waterlevel of borehole UO 1 at ±8 m with the piezometric heads measured in boreholes UO7 and UO23 of ±12 m. Any leakage from the first aquifer down to the second aquifer by means of preferred pathways (boreholes) will be compensated for by the high K-value of the fracture.
The importance of piezometers is realised when the migration process of the tracer is considered. The reasons can be described by the following two scenarios:

1. If the level measured in boreholes UO7 and UO23 are waterlevels, then the gradient from the "pitlatrine" borehole towards boreholes UO7 and UO23 will have a major influence on the tracer migration as soon as it reaches the watertable.
2. If the level measured is the piezometric head of the bottom aquifer, then the tracer will have to travel through the unsaturated zone and matrix of the aquifer to reach the fracture. Only then will it be possible to detect the tracer when water is abstracted from the borehole.

On March 10, 1999 the tracer tests were started. 30 g of Fluorecein was mixed with 10 L of water and introduced into the "pitlatrine" boreholes. For each day afterwards the waterlevels in the "pitlatrines" were monitored and UO7 and UO23 were pumped for one hour at 0.2 L/s while the Fluorecein concentration was measured by means of a fluorometer.
Table 6.3 summarizes the tracer test that was started on March 10, 1999.

**Table 6.3: Summary of tracer tests conducted on March 10, 1999 at the Campus Test Site.**

| Date | Fluorecein concentration (ppb) | | Waterlevels (m) | |
|---|---|---|---|---|
| | **UO7** | **UO23** | **Pitlatrine UO7** | **Pitlatrine UO23** |
| March 10, 1999 | 1.030 | 3.199 | 1.270 | 3.530 |
| May 24, 1999 | 0.720 | 4.537 | 1.380 | 4.070 |

As was anticipated, Table 6.3 confirms that the tracer has not reached the fracture and no Fluorecein was detected. The difference in the concentrations between UO7 and UO23 is because UO23 has already been contaminated by earlier tests conducted on it.

By making use of the derived equation (Equation 6.2) of Darcy's law for falling-head experiments (De Marsily, 1981), the K-values for the formations under the two "pitlatrine" boreholes were calculated.

$$K = \frac{aL}{A(t-t_0)} \quad \ln\frac{h}{h_0} \qquad \textbf{(Equation 6.2)}$$

Where:

K = Saturated vertical hydraulic conductivity.
A = Larger cross-sectional area.
a = Smaller cross-sectional area.
L = Length over which flow takes place.
$t_0$ = Start time.
T = Time at end of test.
$h_0$ = Head at time $t_0$.
h = Head at time t.

The diameter of the "pitlatrines" are constant and this implies that $a = A$ in Equation 6.2 and thus it becomes:

$$K = \frac{L}{(t-t_0)} \quad \ln\frac{h}{h_0} \qquad \textbf{(Equation 6.3)}$$

By using Equation 6.3 the values in Table 6.4 were obtained for the saturated vertical K-value at the two "pitlatrine" boreholes.

**Table 6.4: Saturated vertical K-values calculated for "pitlatrine" boreholes UO7 and UO23.**

| Borehole Nr. | Saturated vertical K (m/day) |
|---|---|
| "Pitlatrine" UO7 | $1.17 \times 10^{-4}$ |
| "Pitlatrine" UO23 | $1.7 \times 10^{-2}$ |

The values calculated show a greater difference (than expected) on two locations approximately 80 m apart in the same area. This justifies the decision made in chapter 6 where it is proposed that protection zone II should cover the whole extent of the fracture, if one is present.

To further demonstrate the importance of protecting the whole fracture extent, the program ***BPZONE*** (Chapter 7) was used to determine the travel time of a pollutant when a vertical fracture is present. The hydraulic conductivity of a fracture can be over 1000 m/day (Van Wyk, 1998). If a hydraulic conductivity value of 1000 m/day is used, the travel time for a pollutant is 0.002 days to reach the horizontal fracture. Since it is impossible to determine the positions of vertical fractures, the importance of protecting the whole horizontal fracture extent is again fully justified.

### 6.3 Tracer and dilution tests conducted outside the Campus Terrain

Although the Campus tracer tests have illustrated that reliable velocity estimates can only be achieved with tracer tests, establishing borehole pairs in the field between which the tracer tests can be conducted is both difficult and expensive. Future research will endeavor to measure the effective porosity with the aid of single well tracer tests, such as the drift and pump-back test.

The kinematic porosity of the fracture on the Campus Terrain has been established at 13.5 %, on the scale of the fracture thickness of 0.17m. Yet, for tracer experiments accomplished at Hartebeesfontein, the porosity of dolomites are 94% and 2% respectively, taken over the actual aquifer thickness of 0.7m (Figure 6.8) and the saturated thickness in the borehole (Figure 6.9). A groundwater velocity of 4 m/d, and a fracture thickness of 0.022m were obtained with a tracer test conducted in a dolerite sill with intersecting layers of Karoo sediments at Jacobsdal, at an abstraction rate of 10 l/min (Figure 6.10). This is low compared to the Campus measurements of 54 m/d, in the sandstone fracture at the same pumping rate and distance. Under induced hydraulic conditions, velocities as high as 5000 m/d were observed at Windhoek in a quartzite fault for a radial divergent tracer test that was conducted over a distance of 780m. At the Meadhurst Test Site, Bainsvlei, Bloemfontein, a groundwater velocity of 10 m/d was established at an abstraction rate of 1 l/s. It is believed that most of the groundwater flow occurs within sand that overlays a sandstone unit. The kinematic porosity of this sand is estimated at 30%, computed over an aquifer thickness of 280mm.

**Figure 6.8: Hartebeesfontein: Kinematic porosity calculation at the scale of the aquifer thickness**

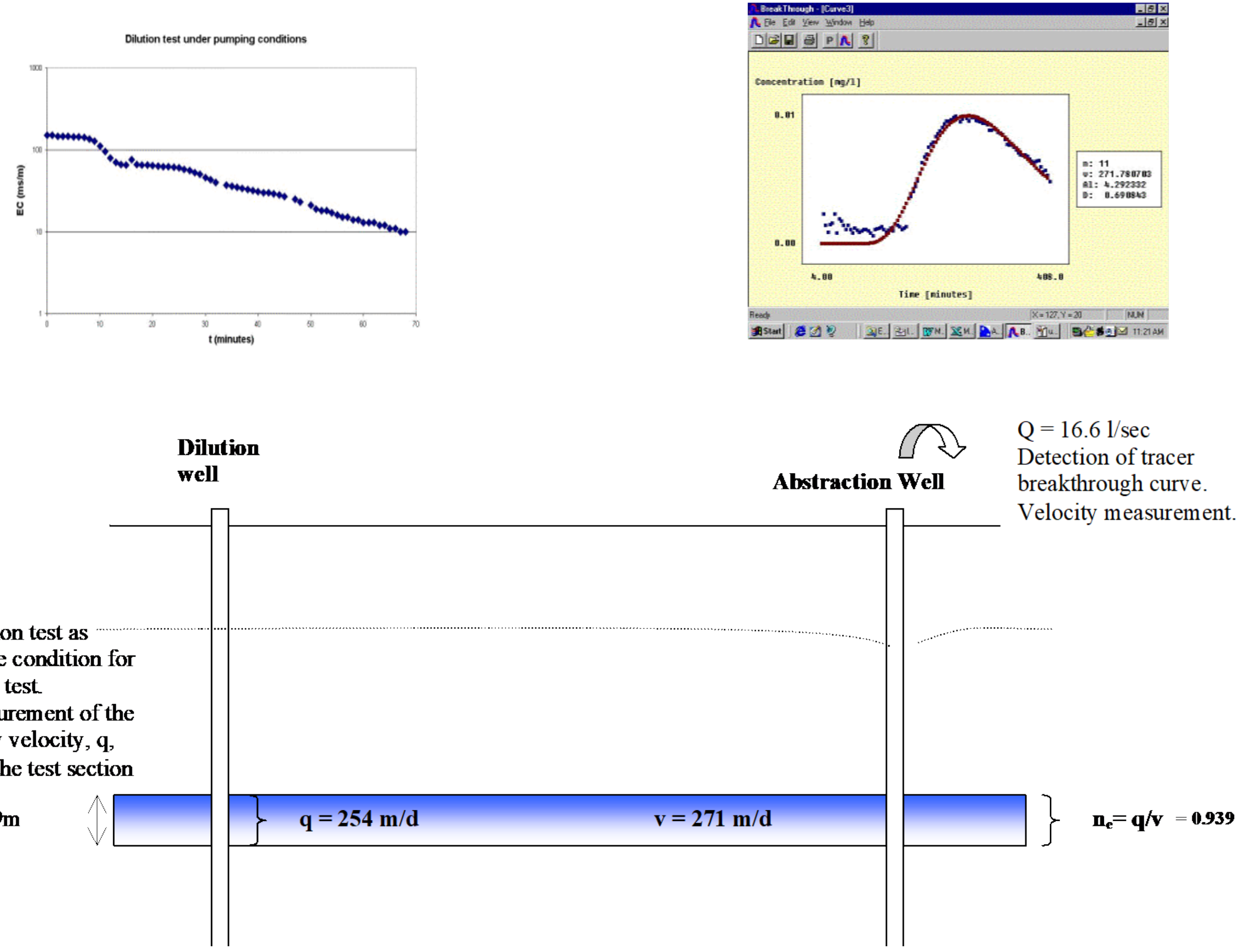
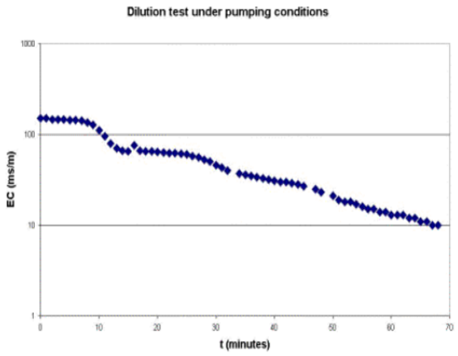
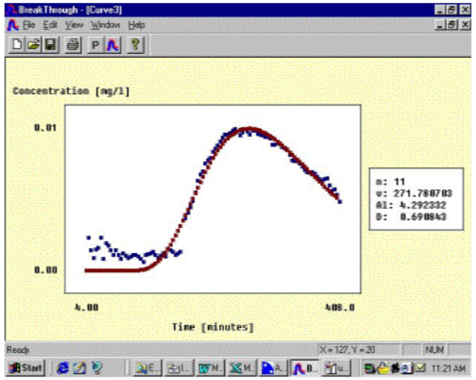


**Figure 6.9: Hartebeesfontein: Kinematic porosity calculation at the scale of the saturated thickness in the borehole.**



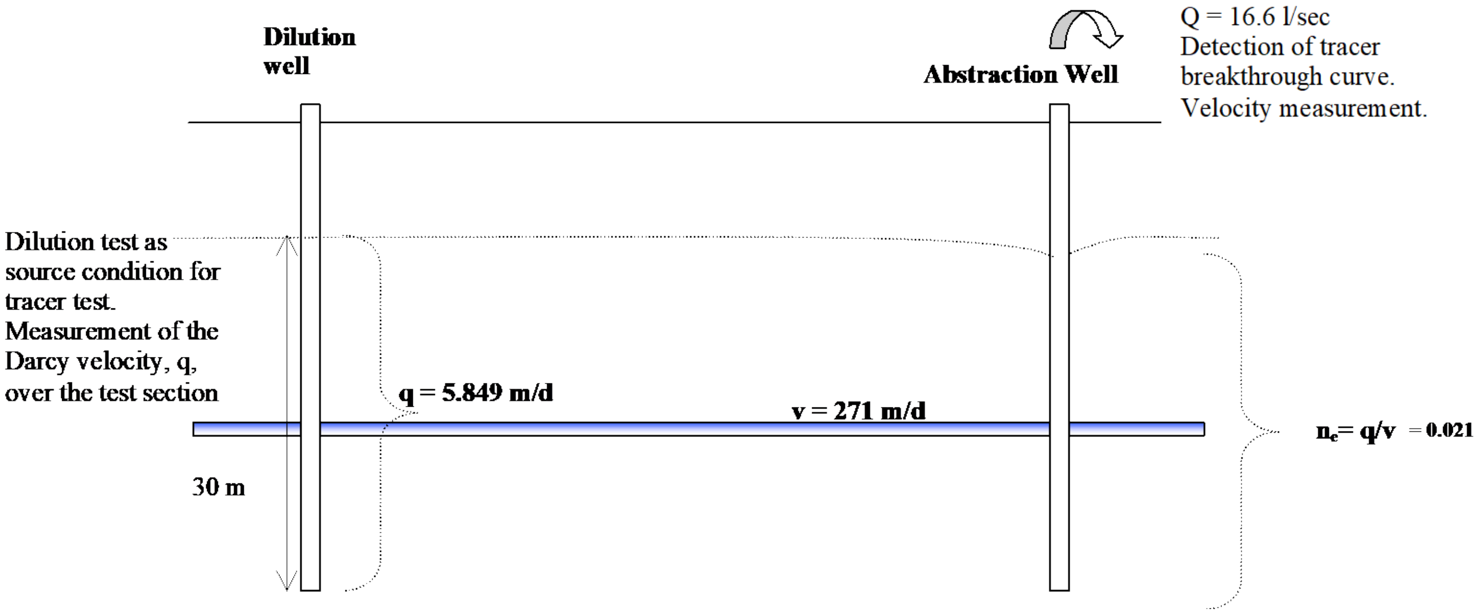
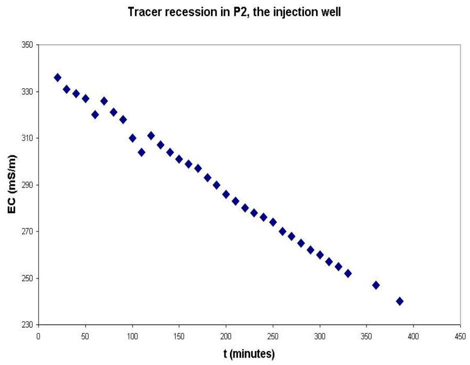
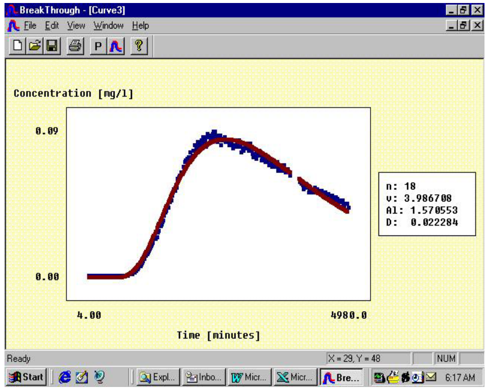
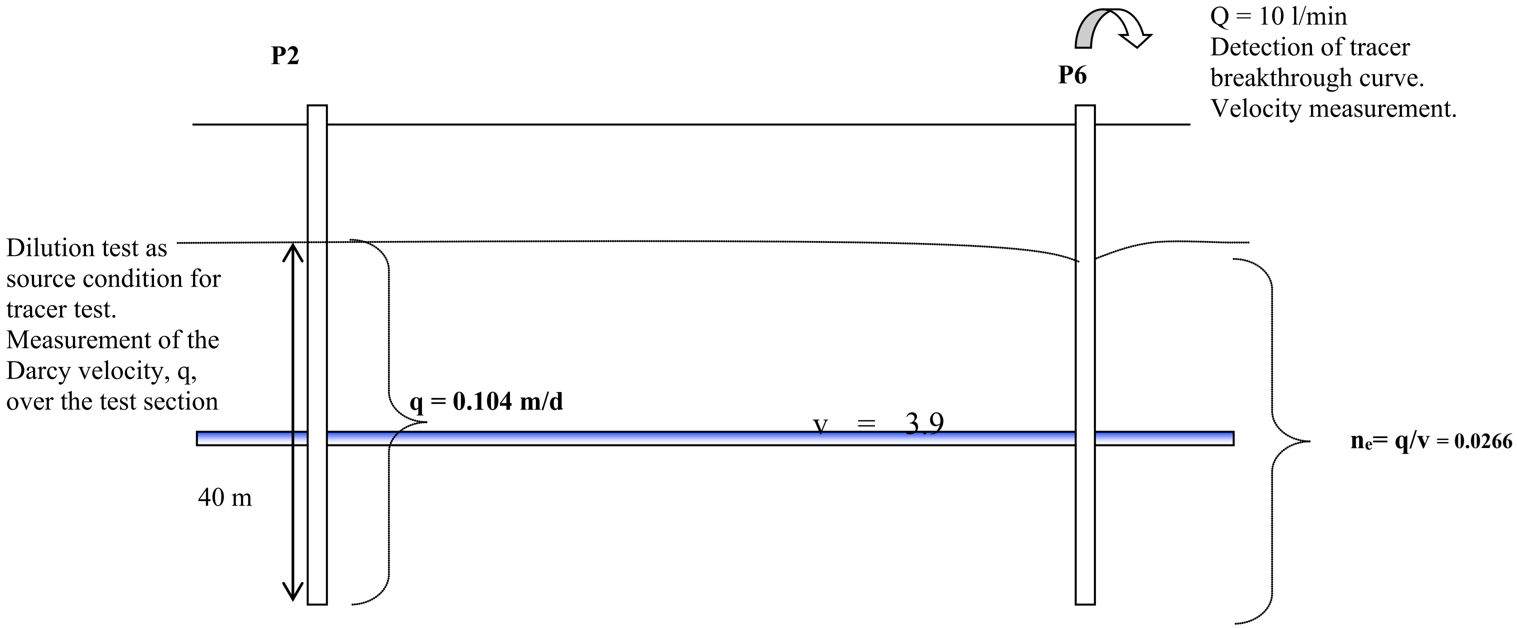



**Figure 6.10: Jacobsdal: Velocity measurements in dolerite sill, with interlacing units of Karoo sediments**.



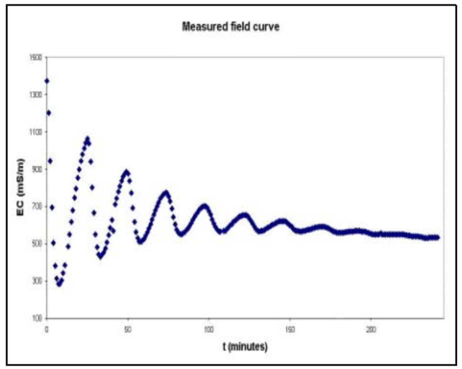



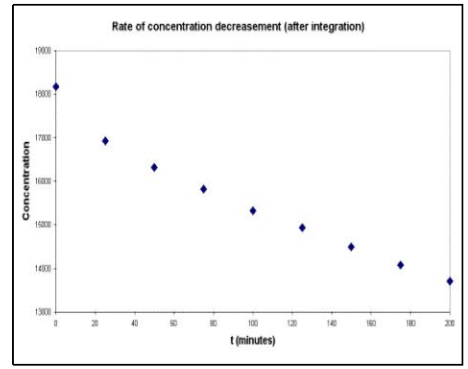



In spite of the fact that only borehole dilution tests could yield the groundwater flow rate when the kinematic porosity-value is not available, these tests provide actual flow measurements and not an interpretation of hydraulic head observations. In studies where boreholes are absent dilution tests provide important information for establishing the average hydraulic gradient, such as downstream of a single borehole at a waste site. Establishing the flow rate through the well provides insight into the risk of contamination, since groundwater chemistry could be coupled to the waterflux, from which the rate of mass migration at the position of the borehole could be established.

Of particular interest for pollution studies, dilution tests have been found useful for establishing the degree of groundwater flow through dolerite/diabase/syenite dykes. At a dolerite dyke in the Witbank area (Figure 6.11), the hydraulic head difference between opposite sides of the dyke suggested that the dyke is impervious to groundwater flow. However, a dilution test conducted in a borehole drilled into the dyke places a question mark on its integrity, since a flux of 300 l/d was established at this position. In the Rustenburg area on the other hand, low flux rates were obtained for a number of tests conducted on the side of a diabase dyke (Figure 6.12), which underlies the base of process water storage dams. From this, there is no evidence that the dyke provides preferred pathways for groundwater flow and contamination.

**Figure 6.11: Witbank: Establishing the integrity of a dolerite dyke towards groundwater through-flow.**




$$Q = W/\alpha \times \ln(C/C_0)$$
$$W = 0.315\ m^3$$
$$C_0 = 18\ 166$$
$$C = 13\ 711$$
$$\alpha = 2$$
$$t = 200\ min$$

*Thus*

$$Q = 317\ l/d$$

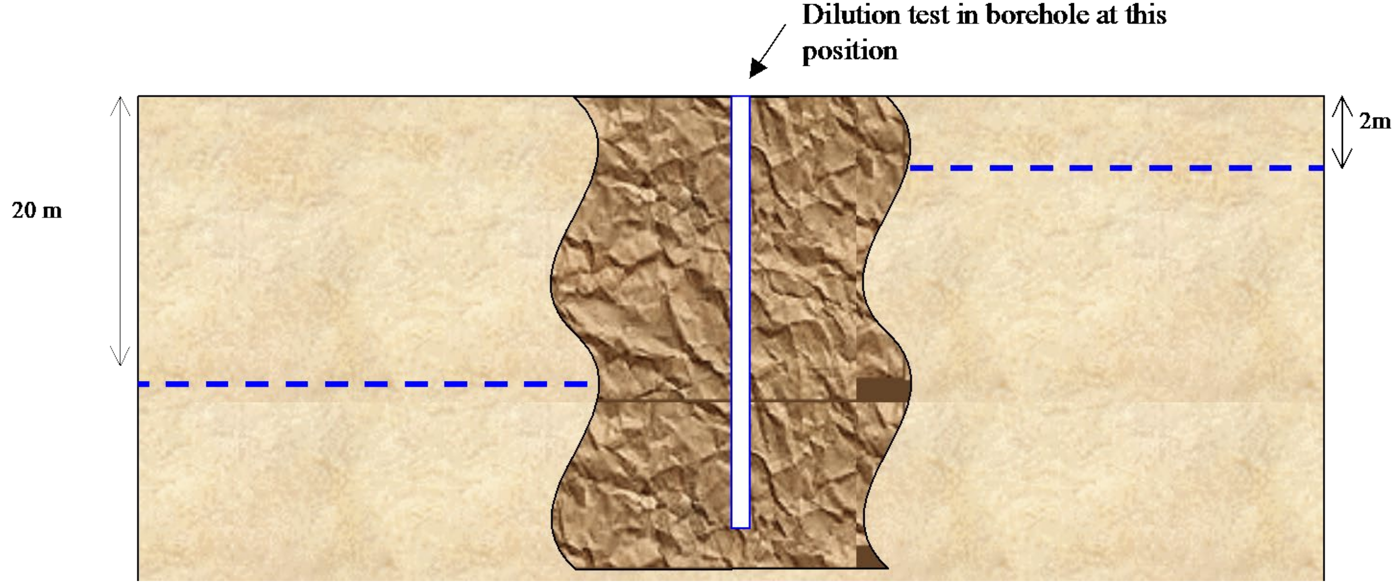
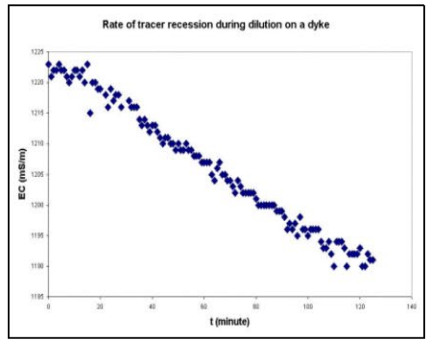


**Figure 6.12: Rustenburg: Establishing the likeliness of groundwater flow along preferred pathways along the side of the dyke.**




$Q = W/\alpha t \times \ln(C/C_0)$

$W = 0.165 m^3$

$C_0 = 1\ 223$ mS/m

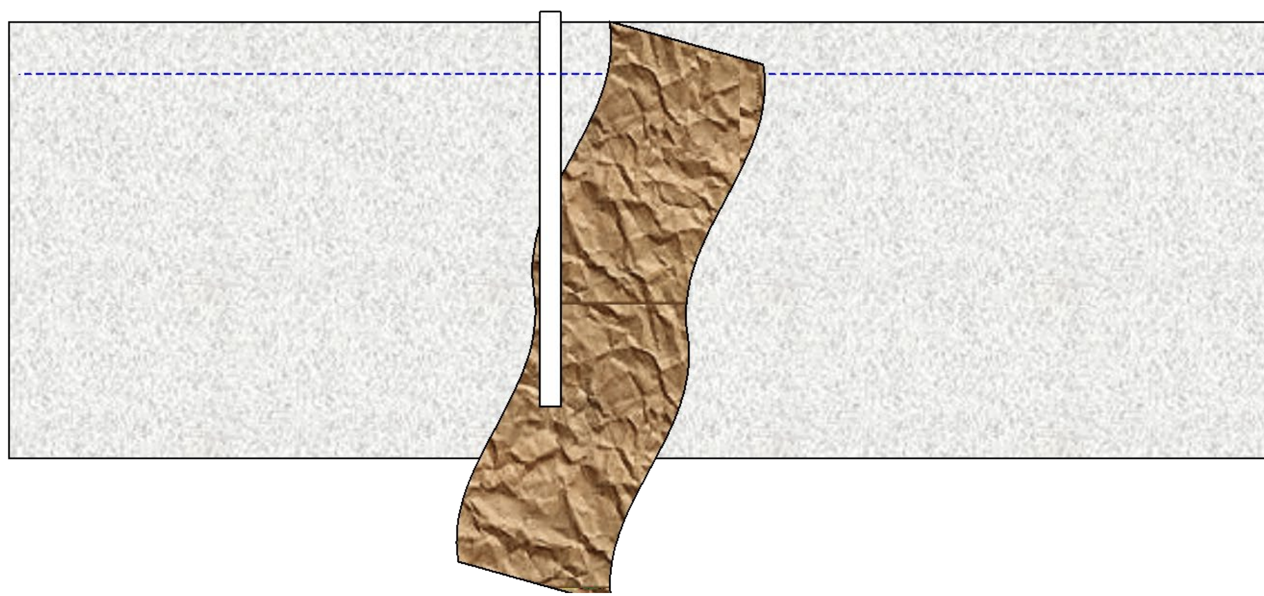
$C = 1\ 191$ mS/m

$\alpha = 2$

$t = 210$ min

$Q = 15$ l/d

Dilution tests were conducted in boreholes positioned in the contact zone of the dyke



In many cases it is difficult to present proof that seepage takes place from slimes dams, water containment dams, etc. into underlying aquifers. Considering that dilution tests are inexpensive and easy to conduct in practice, a method is now available to obtain a first estimate of the likeliness of such facilities impacting the groundwater system. For instance, downstream of an unlined slimes dam at a gold mine (Figure 6.13), fluxes between 400 and 1300l/d were established, which were remarkably higher than fluxes ascertained at other locations within the aquifer. At a coal mine near Witbank, the difference in fluxes between the mined areas and undisturbed geology could readily be established (Figure 6.14).

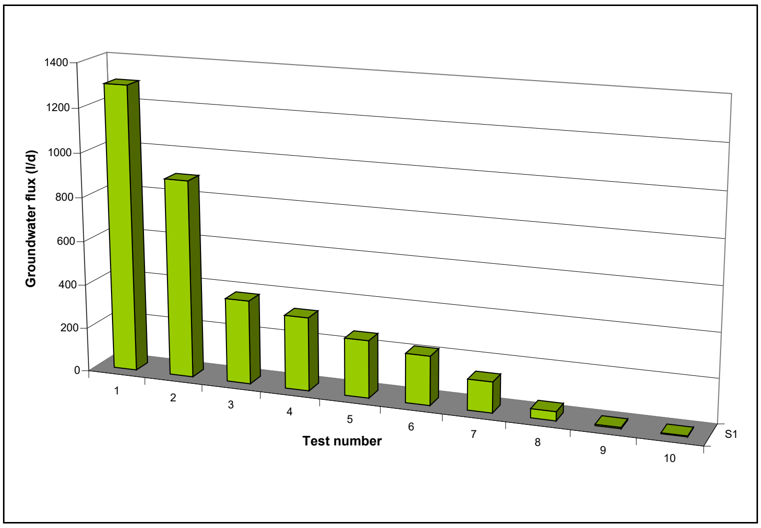



**Fig. 6.13: Dilution tests conducted below a slimes dam indicate much higher fluxes than the rest of the aquifer.**

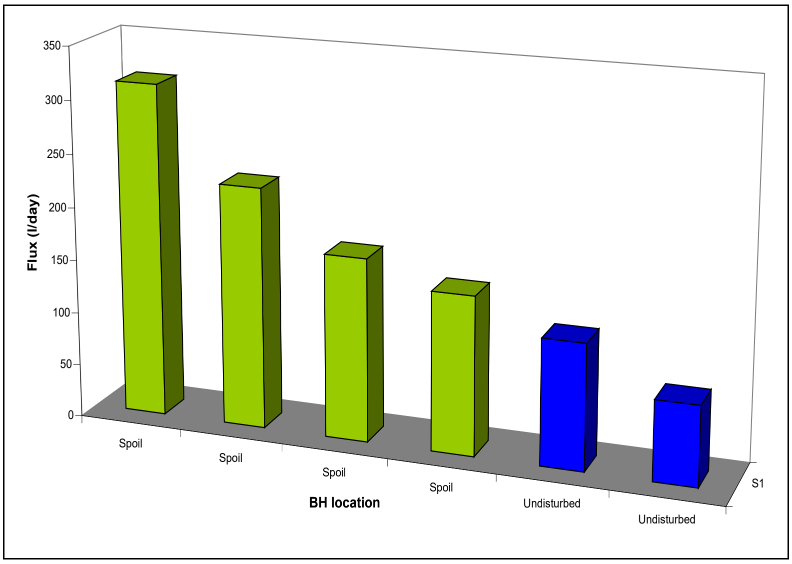



**Fig. 6.14: Groundwater fluxes in rehabilitated mine pits and undisturbed geology at a coalmine.**

From tracer dilution tests conducted on the Windhoek aquifer, boreholes that were reacting to artificial injection could be identified. It was found that boreholes with the larger fluxes were not necessarily among those that exhibited the greatest hydraulic response. Here, the usefulness of dilution tests was further exploited by conducting several tests at different intervals in the borehole, thereby establishing the exact position of the fractures reacting to the induced injection. This information was later applied to select a sampling position for a consequent tracer test. After introducing fluorecein in the injection borehole (injection rate: 16 l/s), the fluorecein was detected in an observation borehole situated 800 m away from the injection borehole after 5.2 hours. This yields a flow velocity of about 3700 m/d.

## 6.4 Meadhurst Test Site

### 6.4.1 Introduction

The Meadhurst Test Site is situated approximately 16 km outside of Bloemfontein in the Bainsvlei smallholdings area. In the 1950's, the farm Meadhurst was subdivided into smaller plots. A number of pitlatrines and septic tanks, as well as active cultivation of the land, have been in operation since then. Boreholes were drilled by the owners to supply water for domestic purposes, irrigation and stock.

During June 1997, IGS started developing the new Meadhurst Test Site. The reasons for picking the Meadhurst terrain as a site were:

1. It is an opportunity to conduct tracer tests along a dyke and to move away from the type of formation that was prevalent on the Campus Test Site at the University of the Free State.
2. The nature of the setup is such that expensive equipment can be situated inside one of the residences, to minimize the possibility of theft.
3. The whole of the test can be monitored 24 hours a day by one of the live-in students.
4. The presence of existing pitlatrines and septic tanks to estimate their impact on the quality of the groundwater.
5. Twenty boreholes already exist on the site of which twelve are equiped with pumps.

**The program for developing the test site consisted of the following:**

1. Geophysics. Siting of new boreholes was done by using a magnetometer.
2. Drilling of six new boreholes as well as cleaning up and casing of existing boreholes.
3. Slug tests on all the open boreholes.
4. Pumping tests on some of the boreholes.
5. Tracer tests.

Figure 6.15 shows the layout of the area and the position of the boreholes, pitlatrines and septic tanks.

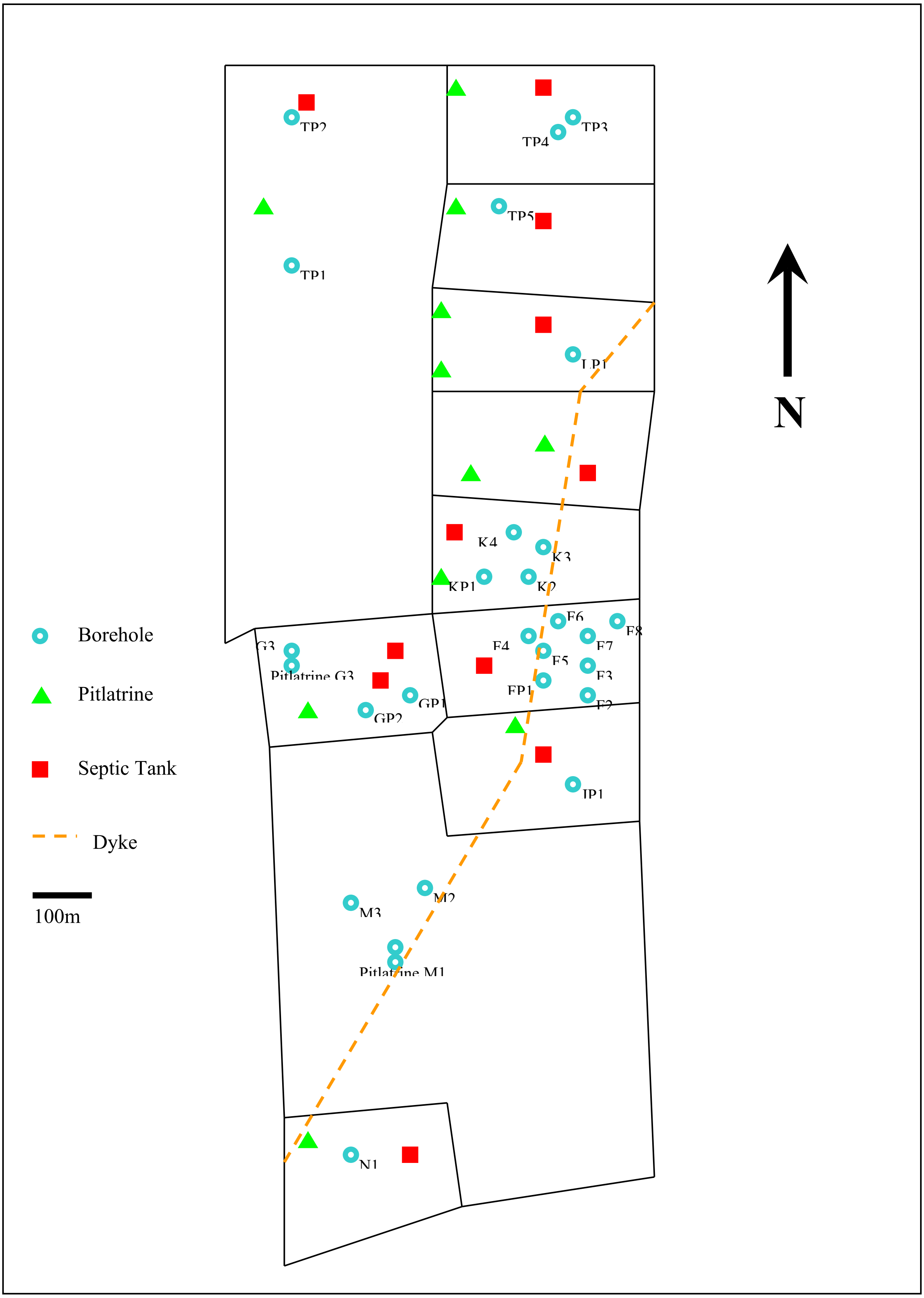



**Figure 6.15: Meadhurst Test Site with the borehole, pitlatrine and septic tank positions.**

On February 13, 1998, additional six boreholes were sited and drilling commenced. These boreholes were used in an attempt to simulate pitlatrines and their effect on the groundwater.

### 6.4.2 Geology

The general geology of the area is mainly made up of Karoo sandstone, dolerite sills and dykes and shale. The top layer consists of soil and the sand is very deep. The depth varies between 18 – 21 m with an average water level of about 19 m below surface.
During the drilling phase at the site several of the boreholes were logged. A typical log of one of the boreholes is given below (Table 6.5).

**Table 6.5: Lithology of borehole FP1.**

| Depth | Formation |
|---|---|
| 0-10m | Reddish soil |
| 10-18m | Orange-red sand |
| 18-21m | Light brown sand |
| 21-28m | Sandstone |
| 28-31m | Sandstone (coarser) |
| 31-36m | Shale |

5 boreholes were drilled along the dyke with yields varying between 0.7 and 3.0 l/s.

### 6.4.3 Water Quality

Samples were taken from all the boreholes for chemical analysis. The results from the laboratory can be seen in Table 6.6.

**Table 6.6: Results of chemical analysis at Meadhurst test site.**

| Nr. | pH | EC | Ca | Mg | Na | K | PAlk | MAlk | Cl | SO4 | NO3 as N | F | Br | COD | TDS |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **FP1** | 7.94 | 67 | 64 | 34 | 39 | 2.9 | ♦ | 250 | 39 | 9 | 12 | 0.21 | 0.36 | ♦ | 491 |
| **F2** | 8 | 61 | 60 | 31 | 36 | 2.9 | ♦ | 248 | 40 | 9 | 4.2 | 0.2 | 0.28 | ♦ | 447 |
| **F3** | 8.01 | 61 | 60 | 32 | 36 | 2.7 | ♦ | 270 | 35 | 9 | 0.5 | 0.2 | 0.24 | ♦ | 449 |
| **F4** | 7.91 | 66 | 62 | 34 | 39 | 2.7 | ♦ | 255 | 38 | 10 | 11.4 | 0.21 | 0.34 | <10 | 489 |
| **F5** | 7.95 | 406 | 109 | 54 | 726 | 12.8 | ♦ | 254 | 999 | 11 | 8.3 | 0.27 | 751 | ♦ | 2965 |
| **F7** | 7.71 | 64 | 61 | 33 | 43 | 3.2 | ♦ | 269 | 28 | 14 | 6.99 | 0.22 | 0.27 | ♦ | 482 |
| **F8** | 8.23 | 65 | 67 | 32 | 38 | 2.7 | ♦ | 265 | 23 | 8 | 12.5 | 0.18 | 0.26 | ♦ | 491 |
| **G3** | 8.04 | 67 | 64 | 38 | 34 | 2.7 | ♦ | 222 | 34 | 9.5 | 22.5 | 0.2 | 0.56 | ♦ | 505 |
| **KP1** | 8.34 | 67 | 67 | 38 | 36 | 3 | 5 | 244 | 34 | 11 | 18.3 | 0.2 | 0.46 | ♦ | 487 |
| **K2** | 7.95 | 70 | 64 | 37 | 37 | 2.7 | ♦ | 245 | 34 | 11 | 17.2 | 0.21 | 0.44 | ♦ | 507 |
| **K3** | 7.91 | 64 | 63 | 36 | 36 | 2.9 | ♦ | 242 | 35 | 7.5 | 14.3 | 0.21 | 0.35 | ♦ | 486 |
| **K4** | 8.03 | 62 | 58 | 31 | 43 | 3 | ♦ | 270 | 27 | 8 | 6.5 | 0.28 | 0.19 | ♦ | 469 |
| **GP1** | 7.84 | 67 | 62 | 37 | 38 | 2.8 | ♦ | 233 | 35 | 9 | 19.1 | 0.22 | 0.45 | ♦ | 509 |
| **GP2** | 7.89 | 70 | 69 | 40 | 38 | 2.8 | ♦ | 238 | 33 | 13 | 25 | 0.23 | 0.55 | ♦ | 545 |
| **J1** | 8.08 | 63 | 57 | 30 | 35 | 2.1 | ♦ | 256 | 36 | 7 | 8.1 | 0.21 | 0.32 | ♦ | 471 |
| **LP1** | 7.92 | 72 | 70 | 40 | 40 | 3.3 | ♦ | 263 | 43 | 13 | 16.6 | 0.21 | 0.5 | ♦ | 547 |
| **N1** | 8.4 | 63 | 47 | 31 | 37 | 2.3 | 10 | 190 | 32 | 9 | 25.5 | 0.25 | 0.72 | ♦ | 473 |
| **TP1** | 7.45 | 65 | 66 | 39 | 31 | 2.4 | ♦ | 249 | 29 | 10 | 17.4 | 0.19 | 0.43 | ♦ | 503 |
| **TP2** | 7.91 | 72 | 69 | 41 | 35 | 2.5 | ♦ | 257 | 34 | 12 | 23.1 | 0.21 | 0.45 | ♦ | 552 |
| **T3** | 7.79 | 72 | 67 | 38 | 36 | 2.3 | ♦ | 265 | 37 | 7 | 17.8 | 0.22 | 0.399 | ♦ | 536 |
| **T4** | 7.8 | 72 | 60 | 35 | 35 | 2.2 | ♦ | 249 | 35 | 6 | 24.4 | 0.21 | 0.43 | ♦ | 535 |
| **T5** | 7.57 | 73 | 68 | 40 | 35 | 2.2 | ♦ | 282 | 39 | 10 | 14.3 | 0.21 | 0.45 | ♦ | 544 |
| **M1** | 7.67 | 64 | 59 | 35 | 34 | 2.5 | ♦ | 202 | 27 | 8 | 24 | 0.26 | 0.62 | 3 | 486 |

According to the South African standards for drinking water the maximum level for nitrate as N is 9 mg/l while the recommended maximum is 5 mg/l. Only boreholes F2 and F3 have values less than the recommended value. The high values for the rest of the boreholes could be a result of contamination from the pitlatrines and septic tanks in the area. Another origin of the nitrates could be the fertilizer that is used during the planting season.

Samples of boreholes GP2, G3, FP1, LP1 and TP2 were sent to the Council for Scientific and Industrial Research (**C.S.I.R.**) for isotopic analysis to determine the ratio of contribution from the above mentioned sources of contamination.

The results from the **C.S.I.R.** for the $^{15}N$ values were as follow (Table 6.7)

**Table 6.7: Results of $^{15}N$ isotopic analysis from samples at the Meadhurst Test Site.**

| Borehole Nr. | $^{15}N$ value (‰) |
|---|---|
| FP1 | +4.8 |
| GP2 | +5.5 |
| G3 | +5.0 |
| LP1 | +5.2 |
| TP2 | Analytical error in laboratory. |

If the values of the water samples are compared to Figure 5.1 (chapter 5), it seems as if the major contributor to the nitrate abundance is the soil organic matter (Figure 6.16). It will also seem as if the risk of nitrate contamination by means of pitlatrines and septic tanks is very low at the Meadhurst Test Site. The only other source of nitrate, according to the isotopic analysis, could then be the fertilizer used by the farmers in the vicinity.

The high values in borehole F5 for most of the parameters is because NaCl, NaBr and Fluorecein were used to conduct a tracer test between borehole F5 and borehole F4.

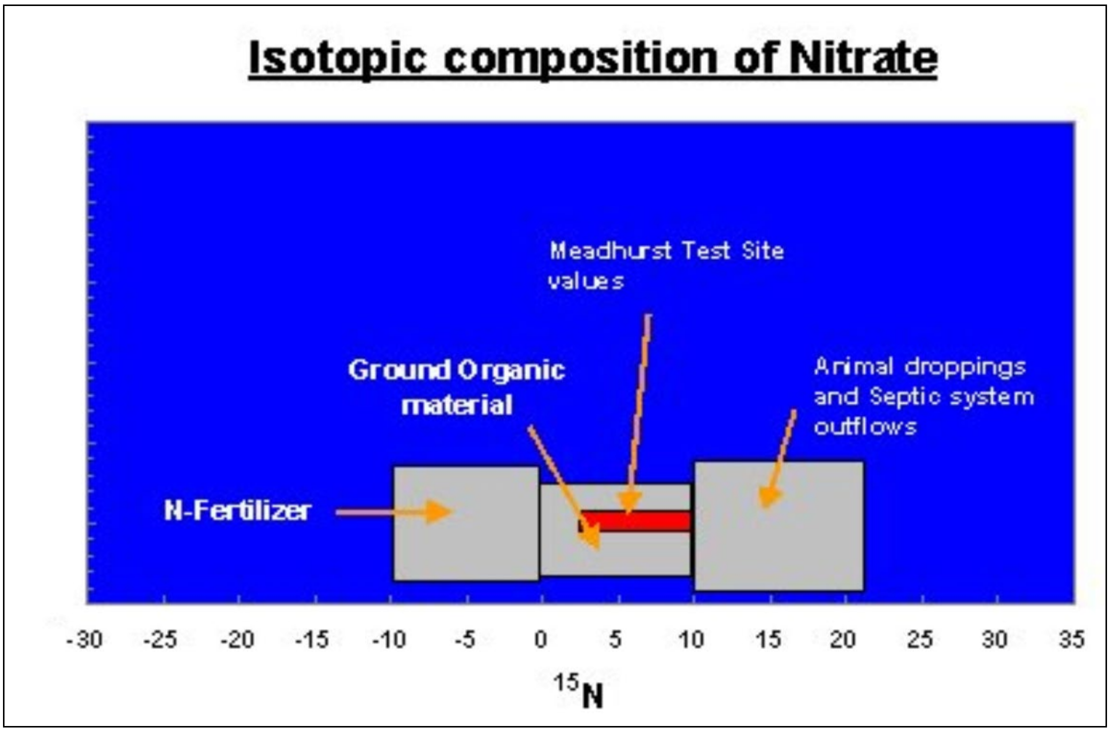



**Figure 6.16: $^{15}N$ analysis at Meadhurst Test Site.**

### 6.4.4 Borehole Tests

#### 6.4.4.1 Slug Tests

Several slug tests were performed on boreholes at the Meadhurst Test Site. The results of the slug tests are given in Table 6.8 below. The borehole yields were obtained from a graph proposed by Viviers (1993).

**Table 6.8: Slug test results at Meadhurst Test Site.**

| Borehole Nr. | Recovery time (s) | Yield (l/s) |
|---|---|---|
| FP1 | 56 | 0.49 |
| F4 | 18 | 1.9 |
| FP1 | 9 | 4.6 |
| K3 | 26 | 1.3 |
| G3 | 16 | 2.3 |

#### 6.4.4.2 Pumping Tests

Pumping tests were conducted on boreholes FP1, F4, GP1 and G3. Boreholes F4, FP1, F6 and F7 were used as observation boreholes during the duration of the tests. No observation holes were available for the tests conducted on GP1 and G3

The tests were analysed with the programs ***Aquitest*** and ***RPTSOLV*** to calculate values for the storativity (**S**) and the transmisivity (**T**) of the aquifer. The calculated values can be seen in Table 6.9.

**Table 6.9: Calculated T- and S-values.**

| Borehole nr. | Cooper - Jacob | | Theis - Recovery | | RPTSOLV | |
|---|---|---|---|---|---|---|
| | T-Value($m^2$/d) | S-Value | T-Value($m^2$/d) | | T-Value($m^2$/d) | S-Value |
| F4 | 34.2 | 4.63E-03 | 34.6 | ♦ | 30 | 2.90E-03 |
| F4 (Obs) | 123 | ♦ | 133 | ♦ | 260 | 3.50E-02 |
| F4 (Obs) | 205 | 1.13E-06 | ♦ | ♦ | ♦ | ♦ |
| FP1 | 41.7 | 2.29E-05 | ♦ | ♦ | ♦ | ♦ |
| FP1 (Obs) | 116 | 9.14E-04 | 119 | ♦ | 260 | 4.90E-03 |
| FP1 (Obs) | 129 | 1.11E-03 | ♦ | ♦ | ♦ | ♦ |
| F6 (Obs) | 120 | 1.52E-03 | 127 | ♦ | ♦ | ♦ |
| F6 (Obs) | 329 | 1.92E-04 | ♦ | ♦ | ♦ | ♦ |
| F7 (Obs) | 305 | 1.05E-03 | ♦ | ♦ | ♦ | ♦ |
| GP1 | 157 | 4.21E-12 | 223 | ♦ | ♦ | ♦ |
| G3 | 133 | 1.14E-01 | 194 | ♦ | ♦ | ♦ |

**Harmonic mean T-value = 97.5 $m^2$/d.**

**Harmonic mean S-value = 2.4 E-004.**

### 6.4.5 Tracer Tests

#### 6.4.5.1 Saturated Zone

On October 27, 1997 a tracer test was conducted between boreholes FP1 and F4 over a distance of 30.4 m. Borehole F4 was used as the injection borehole and borehole FP1 as the abstraction borehole. The test was conducted under forced gradient conditions. This means that borehole FP1 was pumped until the water level stabilised and a pseudo steady state was reached. The abstraction rate was 1 L/s (Table 6.10).

Two types of tracer tests were combined during this exercise. A point dilution test was performed on borehole F4, while a forced gradient test was conducted between borehole FP1 and borehole F4. The graphical representation of the point dilution test can be seen in Figure 6.17.

**Table 6.10: Summary of tracer test between boreholes FP1 and F4.**

| **TEST ID** | **CV 1** |
|---|---|
| Date | 27/10/97 |
| Injection well | F4 |
| Pumped well | FP 1 |
| Well radius(m) | 0.0825 |
| Radial distance (m) | 30.4 |
| **INJECTION WELL** | |
| Well conditions | Open |
| Position of mixing pump (m) | 22 |
| Injection well volume (l) | 20 |
| Injection method | In-situ |
| Tracer | Fluorecein |
| Volume of tracer solution (ml) | 5000 |
| Concentration of tracer solution (g/l) | 20 |
| Time of tracer injection | 13:55 |
| Tracer concentration after injection ($C_0$) (mg/l) | 78 |
| Tracer mass injected (g) | 100 |
| Rate of tracer decay in the injection well: | See figure 6.17. |
| **PUMPING WELL** | |
| Well conditions | Open |
| Pump position (m below collar level) | 22 |
| Abstraction rate (l/sec) | 1 |
| Std. Deviation of flow rate (l/sec) | |
| Head difference after steady state (m) | 0.195 |
| Tracer detection technique | Sample collection |
| Time of first tracer detection (min) | 1028 |
| Time of peak arrival (min) | 3412 |
| Tracer concentration of peak, $C_P$ (mg/l) | 0.07 |
| Tracer mass recovery (g) | 64 |

The shape of the dilution graph in Figure 6.17 represents an almost straight line, which indicates that the mixing of the tracer in borehole F4 was successful. From the point dilution test the value of the Darcy velocity (q) can be determined.

The data of the tracer test was fitted with an analytical model and the groundwater velocity was estimated as 10 m/d (Figure 6.17).

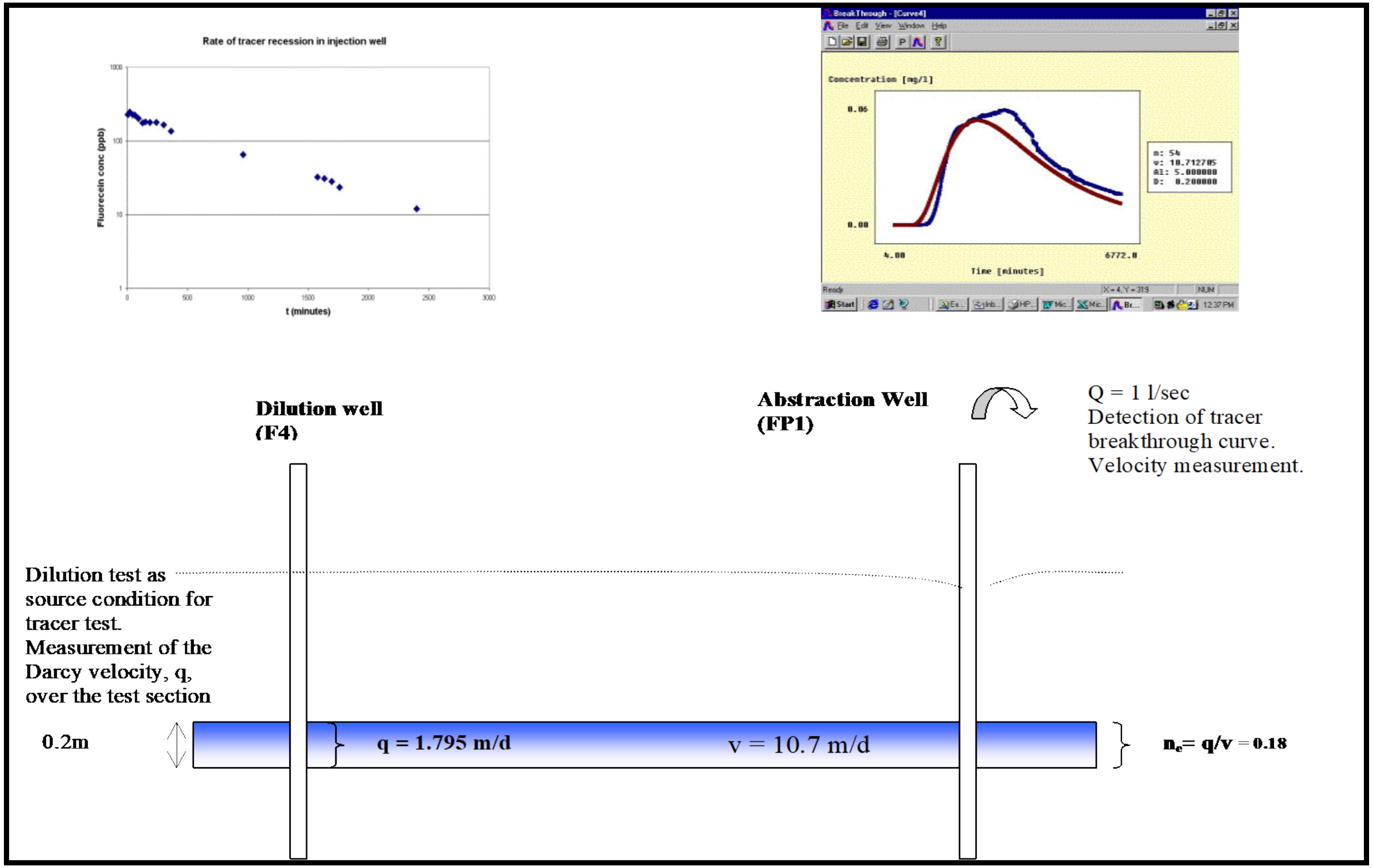



**Figure 6.17: Output of analytical model used to determine the groundwater velocity.**

#### 6.4.5.2 Unsaturated zone

The new boreholes drilled at the Meadhurst Test Site on February 13, 1998 were used to try and simulate pitlatrines and the effect of the unsaturated zone on the effluent from pitlatrines. Three boreholes were drilled to a depth of 38 m. These boreholes were positioned to form three corners of a triangle. This was done to ensure that the gradient of the watertable could be determined. Another borehole was drilled to a depth of 15m. The water levels measured in the other three boreholes are approximately 18 – 19m. An unsaturated zone of ± 3m was left to conduct the tests on.

Two other shallow boreholes were drilled (boreholes not reaching water table). One was drilled upstream from borehole FP1 to a depth of 12m and another to a depth of 9m upstream from borehole G3.

Samples were taken at depths between 14 and 24 m in borehole M1. Lab measurements on these samples, yielded the following parameters:

1. K = 5 m/d
2. Porosity (n) = 0,41
3. Kinematic porosity = 0,33

Tracer tests were also conducted to determine the effect of the vadose zone on bacteriological contaminants. In these tests, bacteriophages were used as tracers together with Fluorecein. A bacteriophage is a virus, which infects bacteria. The bacteriophages are believed to be more representative of the behaviour of viral and bacteriological contaminants in the groundwater environment. The reason for this line of thinking lies in the fact that their small size puts them on the same scale as pathogenic viruses of eucaryotic organisms, and they are non-pathogenic as well as non-toxic (Rossi, 1994). Similar tests on surface water, karst type aquifers as well as porous medium aquifers in Switzerland seem to support this theory (Rossi, 1994).

The bacteriophages that were used as tracers, held no risk for the groundwater environment, as it is non-reactive as long as it is not in contact with its host bacteria. Their host bacteria only occur naturally in the marine environment and not in groundwater, therefore there is no risk of the bacteriophages being able to reproduce and contaminate the groundwater. The bacteriophages are united with the host bacteria only after the samples have been taken from the extraction wells. This happens in the laboratory. The use of bacteriophages is not expensive and the analysis can be done within two days after the sample has been taken.

Tracer tests where a fluorescent dye as well as bacteriophages were used, were conducted at Willerwald, Switzerland (Rossi, 1994). Although the tests were not done in a fractured type of groundwater environment, the results were very informative in terms of migration differences between a conservative type tracer and a bacteriological tracer.

From the breakthrough curves it was observed that bacteriophage migration was faster than both chemical and fluorescent tracers in a porous medium (Figure 6.18). Another observation was that the bacteriophages moved over greater distances and in larger quantities through an aquifer with permeable interstitial porosity, while in aquifers that are less permeable and more silty, the adsorption of bacteriophages on the matrix substrate is believed to be the main cause for the dramatic reduction in the number of particles in suspension (Rossi, 1994).

Bacteriophages will logically have a higher migration velocity in the most granular part of a system. This characteristic might be used to try and predict certain types of contaminant flow and behaviour where conventional tracers fail to give any conclusive results (Rossi, 1994).

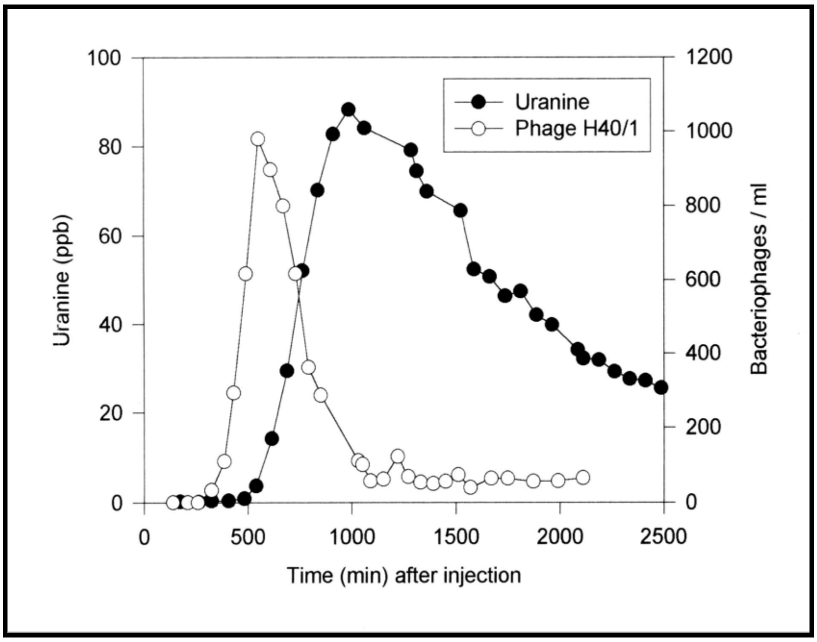



**Figure 6.18: Graph showing the difference in travel time between Fluorecein (Uranine) and bacteriophage H40/1.**

From the data gathered during the tracer tests on the unsaturated zone, it was thought the Fluorecein were detected after a time of about three days. No traces of the bacteriophages were detected even after sampling continued for a duration of one month. After the second series of tracer tests it became apparent that the differences in concentration were only oscillations when performing the analysis for Fluorecein.

The apparent breakthrough curves over a period of 1 day for the Fluorecein in the different boreholes are given in Figure 6.19.

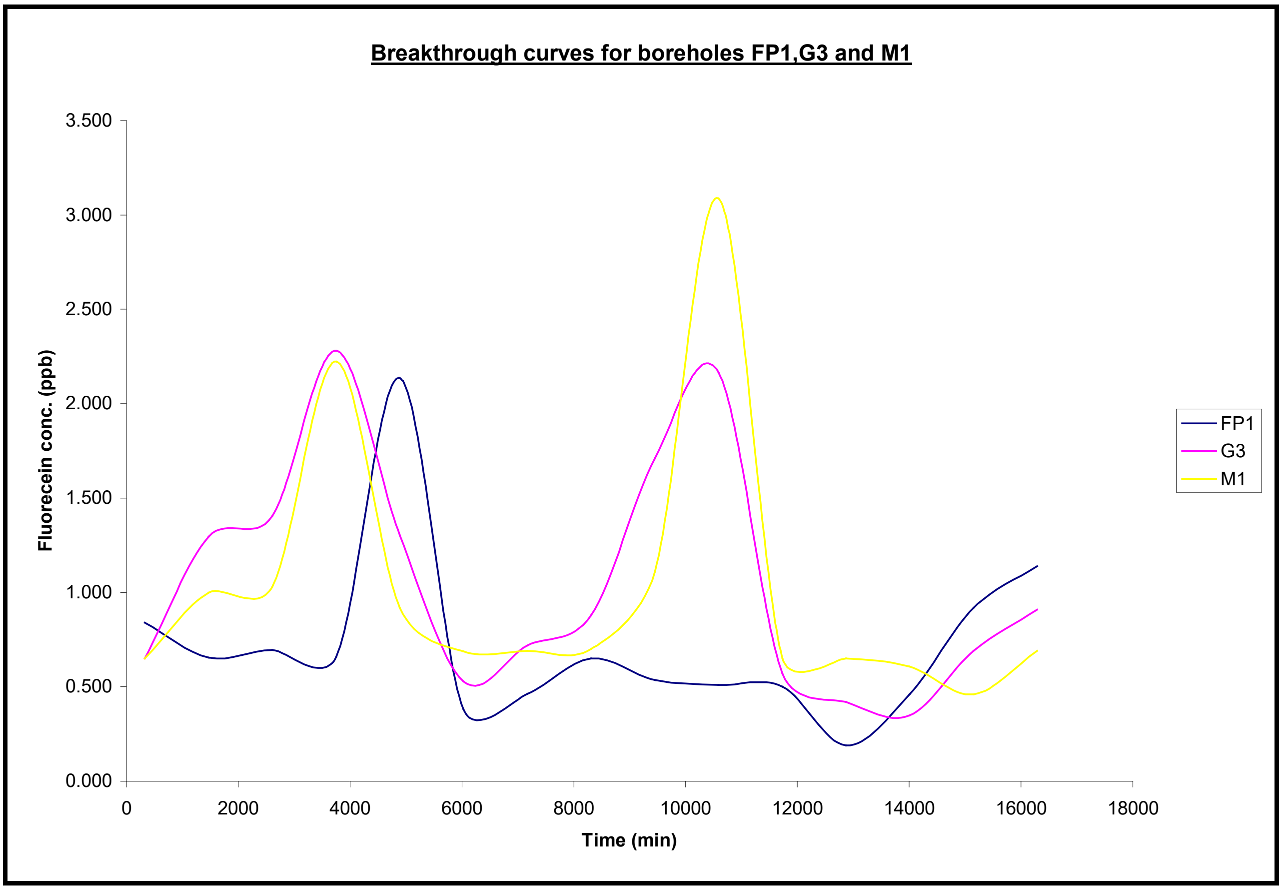



**Figure 6.19: Apparent Fluorecein breakthrough curves in boreholes FP1, G3 and M1 for a period of 1600 minutes.**

The tracer tests were again carried out after a drying out period of about three months. For a duration of one week, 10 liters of water were added to the “pitlatrine” boreholes every second day. Thereafter 5 liters of water was added every third day for a peiod of three weeks. For the further duration of the test, 5 liters of water were added only when no water level could be measured in the “pitlatrine” boreholes. No bacteriophages were used for these tests and only Fluorecein was used.

The results from the tracer tests are given in Table 6.11.

**Table 6.11: Fluorecein concentration at the start and end of the tracer tests conducted at the Meadhurst Test Site.**

| Date | Fuorecein concentration (ppb) | | |
|---|---|---|---|
| | FP1 | M1 | G3 |
| March 10, 1999 | 0.531 | 4.080 | 0.990 |
| May 24, 1999 | 736.29 | 4.690 | 1.530 |

**Table 6.12: Differences in waterlevels in the "pitlatrine" boreholes over a one-day period.**

| Date | Difference in waterlevel (m) | | |
|---|---|---|---|
| | Pitlatrine FP1 | Pitlatrine M1 | Pitlatrine G3 |
| 03/23/99-03/24/99 | 0.9 | 0.5 | 0.2 |

After applying equation 6.3 the values in Table 6.13 were calculated for the saturated K-value of the unconsolidated zone.

**Table 6.13: Saturated vertical K calculated for "pitlatrine" boreholes FP1, M1 and G3.**

| Borehole Nr. | Saturated vertical K (m/day) |
|---|---|
| "Pitlatrine" FP1 | 2.2 |
| "Pitlatrine" M1 | 0.7 |
| "Pitlatrine" G3 | 0.009 |

Again, three totally different saturated vertical K-values are encountered in an area no larger than 1 $km^2$ which emphasises the problems when homogeneity is assumed.

The K-values in Table 6.13 explains why Fluorecein was detected in only borehole FP1 and not in the other boreholes. The small K-value at borehole G3 correlates very well with the geology encountered when it was drilled. Much more clay was encountered during the drilling phase and after heavy rains the water usually accumulates in that part of the test site.

Borehole FP1 also has the advantage in that the pump is permanently installed whereas in boreholes M1 and G3 a smaller pump was only inserted for the duration of the sampling period. More emphasis was therefore placed on the tracer test at borehole FP1 because of the better conditions for continuous sampling that exists.

After the saturated vertical K-value for the unconsolidated zone was calculated for "pitlatrine" FP1, this value together with the travel time was used in the programme BPZONE (Chapter 7) to determine the saturated vertical K-value for the consolidated zone (Figure 6.20).

**(c) Estimation of travel time from surface to water strike: for bacterial die-off time**

**K of unconsolidated and consolidated material**

| **Unconsolidated material** | | **Consolidated material Vertikal K-value (if unknown)** | | |
|---|---|---|---|---|
| clay content (%) = | 8 | Use K (m/d) = | 0.01 | for first estimate |
| **Cambell equation: K (m/d) =** | 2.722 | (usually between 0,001 and 0,1 m/d) | | |

| | Thickness | K (m/d) | Porosity | t. time (d) |
|---|---|---|---|---|
| (a) unconsolidated material | 12 | 2.7 | 0.2 | 0.889 |
| (b) consolidated material to water strike | 9 | 0.02 | 0.1 | 45.000 |
| (c) Depth to water level (m) = | 18.68 | | | |
| Travel time to water strike (d) = | 45.89 | | | |
| Travel time to water level (d) = | 34.29 | | | |
| Risk of microbial pollution in BH = | Low | | | |

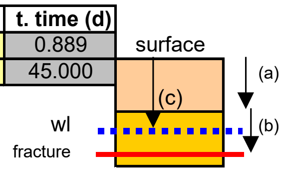



**Figure 6.20: Estimation of travel time from surface to water strike with emphasis on the K-value of 0.02 m/d suggested by the programme BPZONE.**

The breakthrough curve measured in borehole FP1 is depicted in Figure 6.21.

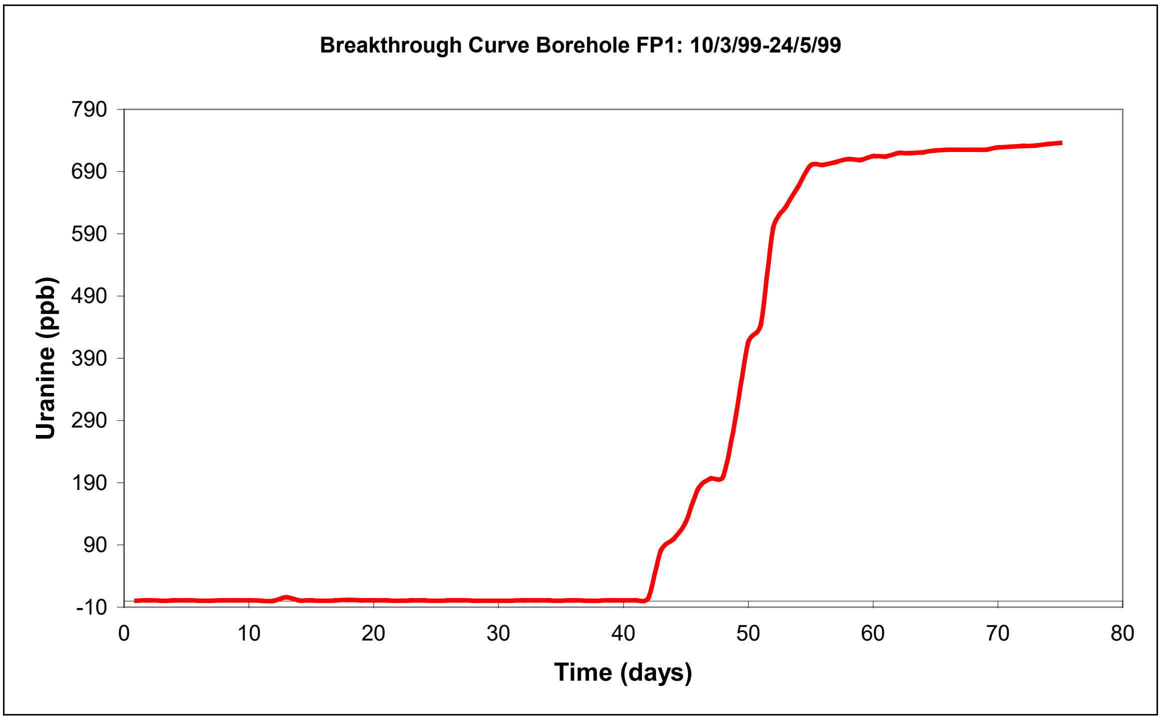



**Figure 6.21: Breakthrough curve measured in borehole FP1 for a period of 75 days.**

## 6.5 Conclusions

The tracer test results indicate high groundwater velocities along fractures and low velocities in the matrix. Table 6.14 is a summary of all the tracer tests discussed in this chapter. Even under natural conditions flow velocities of up to 30 m/d are possible along fractures.

The tracer test interpretations show that the parallel plate model for fractures is not valid for Karoo aquifers, and that the fracture can be viewed as a porous medium with a relatively high kinematic porosity and a small effective thickness. The results discussed in this chapter highlight the complexity of the unsaturated zone and therefore the vulnerability of shallow aquifers. The protection of the source for these aquifers is critical in a groundwater quality management strategy.
Much more effort will have to go into trying to determine the effect of the unsaturated zone where the effect of known pollutants on the environment is concerned.

Table 6.13: Aquifer parameters obtained from tracer tests

| Location | Rock type | Distance of transport (m) | Abstraction rate (l/s) | Groundwater flow rate through injection/dilution well (l/d) | Pore Velocity (m/d) | Effective thickness of fracture (m) | Kinematic porosity of fracture |
|---|---|---|---|---|---|---|---|
| Campus Terrain, Bloemfontein | Karoo Sandstone fracture | 15 | 0.35 | 700 | 77 | 0.17 | 0.135 |
| Meadhurst, Bainsvlei | Sand, sandstone | 30 | 1 | 100 | 10 | 0.2 | 0.18 |
| Jacobsdal | Dolerite sill with interlacing units of Karoo sediments | 10 | 0.166 | 500 | 5 | .022 | |
| Windhoek | Quartzite/gneiss fault | 780 | Injection rate 17 l/s | | 3500 | | |
| Hartebeesfontein | Dolomite | 60 | 16 | | 270 | 0.7 | 0.94 |
| Witbank | Dolerite Dyke | | 0 | 317 | | | |
| Rustenburg | Diabase Dyke | | 0 | 27 | | | |

# CHAPTER 8

# CONCLUSIONS AND RECOMMENDATIONS

## 8.1 Conclusions

This report presents a systematic approach to the conducting of tracer tests in the field. If performed together, the borehole dilution and radial convergent tests can yield values of the following parameters:

- Seepage and Darcy velocities
- Kinematic porosity
- Effective thickness of the fracture zone

The report also focuses on the delineation of borehole areas in fractured aquifers, which are dominant in the Republic of South Africa. It has been found that the borehole protection area is usually much larger than in typical porous aquifers. The protection zones proposed in this document should be considered wherever the possibility of groundwater contamination is present. The program ***BPZONE*** is a more practical approach when delineating boreholes.
It is emphasised that the tracer experiments may assist in revealing additional information on fracture characteristics.

From the research it can be concluded that:

(i) Velocity estimates from the borehole dilution test are in agreement with the results of the radial convergent tracer tests, and the method is regarded valid if the value of the effective porosity is known.
(ii) Equivalent apertures are not regarded as valid for velocity estimations with the dilution test method.
(iii) Since the effective porosity cannot be derived from hydraulic tests, porosity estimation involves a tracer migration test in the aquifer between two boreholes (porous model).

(iv) The derivation of the K-value from slug test data in fractured aquifers is scale related, and data are often misinterpreted since the actual aquifer thickness, b, is not known. In the case of pollution studies, it is recommended that a tracer test should be conducted to derive the K-value.

(v) The peak of a tracer breakthrough curve does not represent the mean pore velocity of the fluid in the medium. Thus the data should be analysed with a suitable model such as described in Chapter 4.

(vi) To assume areal homogeneity is one of the worst mistakes that can be made when assessing the impact of a pollution source on groundwater. Vertical fractures should be an important consideration.

(vii) The unsaturated zone is of great importance when dealing with groundwater contamination.

(viii) Tracers are definitely an important aid in gaining a better understanding of groundwater flow mechanisms as well as the effect of the groundwater environment on groundwater movement.

## 8.2 Recommendations

1. The protection of our groundwater resources should be of major importance and more research should be done on how to ensure a minimal impact of pollution sources on groundwater quality.

2. The application of the proposed protection zones should be tested over a wide variety of regions.

3. Now that the effect of point pollution sources on groundwater has been established, this type of study should be expanded to pollution sources with a larger areal extent, like mines, waste sites and industrial areas.

4. When conducting an impact study, more emphasis should be placed on the use of tracers to gain more information and data.

5. Future research projects should be carried out for practical problems. For instance, research on injection withdrawal tests should be taken into consideration.

# REFERENCES


Aldous, P. J. and Smart, P.L. (1988). Tracing groundwater movement in abandoned coal mine aquifers using fluorescent dyes, *Ground Water*, 26(2), 125-178.

Ardakani, M.S., Rehbock, J.T. and McLaren, A.D. (1974)(a). Oxidation of ammonium to nitrate in a soil column. *Soil Sci. Soc. Am. Proc.* 38: 96-99.

Ardakani, M.S., Schulz, R.K. and McLaren, A.D. (1974)(b). A kinetic study of ammonium and nitrate oxidation in a soil field plot. *Soil Sci. Soc. Am. Proc.* 38: 273-277.

Aulenbach, D.B., Bull, J. H. and Middlesworth, B.C. (1978), Use of tracers to confirm groundwater flow, *Ground Water.*, 16(3), 149-157.

Bauman, B.J. and Schafer, W.M. (1985). Estimating ground-water quality impacts from on-site sewage treatment systems. In Proc. 4th National Symposium on Individual and Small Community Sewage Systems., ASAE, St. Joseph, MI. pp. 285-294.

Benischke, R.(1991), Fluorescent tracers in hydrology, Unpublished manuscript, Inst. For Geothermics, Graz.

Bitton, G., Farrah, S.R., Ruskin, R.H., Butner, J. and Chou, Y.J. (1983). Survival of pathogenic and indicator organisms in groundwater. *Groundwater*. 21: 405-410.

Bitton, G., Lahav, N. and Henis, Y. (1974). Movement and retention of klebsiella aerogenes in soil columns. *Plant and Soil*. 40: 373-380.

Bosch, H.M., Rosefield, A.B., Huston, R., Shipman, H.R. and Woodward, F.L. (1950). Methemoglobinemia and Minnesota well supplies. *J. Am. Water Works Assoc*. 42: 161-170.

Botha, J.F., Verwey, J.P., Van der Voort, I., Vivier, J.J.P., Buys, J., Colliston, W.P. and Loock, J.C. (1998). Karoo Aquifers. Their geology, geometry and physical properties. WRC report no. 487/1/98.

Bouma, J., Ziebell, W.A., Walther, W.O., Olcott, P.G., McCoy, E. and Hole, F. (1972). Soil adsorption of septic tank effluent. University of Wisconsin, Madison, WI.

Bouwer, H., Lance, J.C. and Riggs, M.S. (1976). High-rate land treatment II. Water quality and economic aspects of the Flushing Meadows Project. *J. Water Poll. Con. Fed*. 46: 844-859.

Bredenkamp, D.B., Janse van Rensburg, H., van Tonder, G.J., and Botha, L.J. (1994), Manuel on quantitative estimation of groundwater recharge and aquifer storativity. Progress report to the Water Research Commission of SA.

Brown, K.W., Donnelly, K.C., Thomas, J.C. and Slowey, J.F. (1984). The movement of Nitrogen species through three soils below septic fields. *J. Environ. Qual*. 13: 460-465.

Brown, K.W., Slowey, J.F. and Wolf, H.W. (1983). The movement of salts, nutrients, fecal coliforms and virus below septic leach fields in three soils. In Proc. 2nd National Home Sewage Treatment Symposium., ASAE, St. Joseph, MI. pp. 208-217.

Bryson, D. D. (Chairman) (1988). Nitrate in drinking water. ECETOC Technical Report No. 27, Brussels.

Chambers, L. W. and Bahr, J. M. (1992), Tracer Test evaluation of a drainage ditch capture zone, *Ground Water*, 30(5), 667-674.

Claassen, H.C. and Cordes, E.H. (1975), Two recirculating tracer test in fractured carbonate rock, Nevada, *Hydrological Science Bulletin*, XX.3, 367-379.

Cogger, C.G. and Carlile, B.L. (1984). Field performance of conventional and alternative septic systems in wet soils. *J. Environ. Qual*. 13: 137-142.

Craun, C. (1992). Waterborne Disease Outbreaks in the United States of America: Causes and Prevention. WHO Statistics Quarterly Vol. 45, 192-199.

Dagan, G., Zaltztein & Gorodischer (1988). *Eur. J. Pediatr. Vol* 147, 87-89.

De Lange, F. (1999), Environmental impact of point pollution sources. MSc thesis in the Department of Geohydroloy at the University of the Free State, Bloemfontein.

De Marsily, G. (1981). Quantitative hydrogeology. Groundwater hydrology for engineers. Paris school of mines, Fontainebleau, France.

DeWalle, F.B. and Schaff, R.M. (1980). Ground-water pollution by septic tank drainfields. *J. Environ. Eng. Div. ASCE*. 106: 631-646.

DWAF, (1997). A Protocol to manage the potential of groundwater contamination from on-site sanitation (National Sanitation Co-ordination Office, Directorate of Geohydrology).

EPA, (1991). Delineation of wellhead protection areas in fractured rocks. Wisconsin geological and natural history survey. Ground-water protection division, Office of groundwater and drinking water, U.S. environmental protection agency, Washington, DC 20460.

Evans, G.V. (1983), Tracer techniques in hydrology, *Int. J. Appl. Isot.*, 34(1), 451-475.

Ezzedine, S. and Rubin, Y. (1997), Analysis of Cape Cod tracer data, *Water Resour. Res.*, 33(2), 1-11.

Ford, K.L., Schoff, J.H.S. and Keefe, T.J. (1980). Mountain residential development minimum well protective distances - well water quality. *J. Environ. Health*. 43: 130-133.

Foster, F. (1985). In: DWAF, (1997). A Protocol to manage the potential of groundwater contamination from on-site sanitation (National Sanitation Co-ordination Office, Directorate of Geohydrology).

Foster, F. and Hirata, H.O.T. (1991). In: DWAF, (1997). A Protocol to manage the potential of groundwater contamination from on-site sanitation (National Sanitation Co-ordination Office, Directorate of Geohydrology).

Fourie, A.B. and Van Ryneveld, M.B. (1994). Environmental impact of on-site sanitation - a literature review with particular application to South Africa; WRC Report No KV 57/94, Water Research Commission, Pretoria.

Franceys, R., Pickford, J. and Reed, R. (1992). A guide to the development of on-site sanitation, World Health Organization, Geneva.

Freeze, R.A., and Cherry, J.A. (1979), Groundwater. Prentice-Hall, Englewood Cliffs, New Jersey.

Garabedian, S.P., LeBlanc, D.R., Gelhar, L.W. and Celia, M.A. (1991), Large-scale naturalgradient tracer test in sand and gravel, Cape Cod, Massachusetts. Analysis of spatial moments for a nonreactive tracer, *Water Resour. Res.*, 27(5), 911-924.

Gerba, C.P. and Lance, J.C. (1978). Poliovirus removal from primary and secondary sewage effluent by soil filtration. *Appli. Environ. Microbiol*. 36: 247-251.

Goyal, S.M. and Gerba, C.P. (1979). Comparative adsorption of human enteroviruses, simian rotavirus, and selected bacteriophages to soils. *Appli. Environ. Microbiol*. 38: 241-247.

Green, K.M. and Cliver, D.O. (1975). Removal of virus from septic tank effluent by sand columns. In Proc. National Home Sewage Disposal Symposium., ASAE, St. Joseph, MI. pp. 137-143.

Gringarten, A. C. and Ramey, H. J. (1974). Unsteady-state pressure distributions created by a well with a single horizontal fracture, partial penetration, or restricted entry. *Society of Petroleum Engineers Journal*. **14** (5), 413-426.

Grove, D.B. and Beetem, W.A. (1971), Porosity and dispersion constant calculations for fractured carbonate rock using the two well method, *Water Resour. Res.*, 7(1), 128-134.

Gupta, S. K., Lau, L. S. and Moravick, P. S. (1994), Groundwater tracing with injected helium, *Ground Water*, 32(1), 96-101.

Hagedorn, C., Hansen, D.T. and Simonson, G.H. (1978). Survival and movement of fecal indicator bacteria under conditions of saturated flow. *J. Environ. Qual.* 7: 55-59.

Heaton, T.H.E. (1986). Isotopic studies of nitrogen pollution in the hydrosphere and atmosphere: A review.

Hess, K.M., Wolf, S. H. and Celia, M. A. (1992), Large scale natural gradient tracer test in sand and gravel, Cape Cod, Massachusetts, hydraulic conductivity variability and calculated macrodispersivities. *Water Resour. Res.*, 28(8), 2011-2027.

Hurst, C.J., Gerba, C.P. and Cech, I. (1980)(a). Effects of environmental variables and soil characteristics on virus survival in soil. *Appli. Environ. Microbiol*. 40: 1067-1079.

Hurst, C.J., Gerba, C.P., Lance, J.C. and Rice, R.C. (1980)(b). Survival of enteroviruses in rapid-infiltration basins during the land application of wastewater. *Appli. Environ. Microbiol*. 40: 192-200.

Hyndman, D.W., and Gorelick, S. M. (1996), Estimating lithologic and transport properties in three dimensions using seismic and tracer data: The Kesterson Aquifer , *Water Resour. Res.*, 32(9), 2659-2670.

Jackson, A. (1994). In: DWAF, (1997). A Protocol to manage the potential of groundwater contamination from on-site sanitation (National Sanitation Co-ordination Office, Directorate of Geohydrology).

Jackson, A. (1997). In: DWAF, (1997). A Protocol to manage the potential of groundwater contamination from on-site sanitation (National Sanitation Co-ordination Office, Directorate of Geohydrology).

Johns, R.A. and Roberts, P.V. (1991). A solute transport model for channelized flow on a fracture, Water Resour. Res., 27(8), 1797- 1808.

Jorgensen, P.H. and Lund, E. (1985). Detection and stability of enteric viruses in sludge, soil and ground water. *Water Sci. Tech*. 17: 185-195.

Kibbey, H.J., Hagedorn, C. and McCoy, E.L. (1978). Use of fecal streptococci as indicators of pollution in soil. *Appli. Environ. Microbiol*. 35: 711-717.

Killey, R.W.D. and Moltyaner, G.L. (1988), Twin lake tracer tests: Setting, methology and hydraulic conductivity distribution, *Water Resour. Res.*, 24(10), 1585-1612.

Kinzelbach, W., Marburger, M., Chiang, W. (1991). Determination of groundwater catchment areas in two and three spatial dimensions. *Journal of Hydrology*, 134 (1992): 221-246.

Knopman, D.S., Voss, C.I. and Garabedian, S.P. (1991), Sampling design for groundwater solute transport: Test methods and analysis of the Cape Cod tracer test data, *Water Resour. Res.*, 27(5), 925-949, 1991.

Koya, K.V. and Chaudhuri, M. (1977). Virus retention by soil. *Prog. Wat. Tech*. 9: 43-52.

Kruseman, G.P. and De Ridder, N.A. (1994). Analysis and evaluation of pumping test data. ILRI publication 47, Wageningen, The Netherlands.

Lance, J.C. (1972). Nitrogen removal by soil mechanisms. *J. Water Poll. Con. Fed*. 44: 1,352-1,361.

Le Blanc, D.R., Garabedian, S.P., Hess, K.M., Gelhar, L.W., Quadri, R.D., Stollenwerk, K.G. and Wood, W.W. (1991), Large scale natural gradient tracer test in sand and gravel, Cape Cod, Massachusetts :1. Experimental design and observed tracer movement. *Water Resour. Res.*, 27(5), 895-910.

LeChevallier, M.W. and Seidler, R.J. (1980). Staphylococcus aureus in rural drinking water. *Appli. Environ. Microbiol*. 30: 739-742.

Lenda, A. and Zuber, A. (1970). Tracer dispersion in Groundwater experiments. Isotope Hydrology 1970. International Atomic Energy Agengy, Vienna.

Lewis, J.W., Foster, S.S.D., Drasar, B.S. (1980). The risk of groundwater pollution by on-site sanitation in Developing countries: a literature review, International

Conference for Waste Disposal (IRCWD), Report # 01/82, Duebendorf, Switzerland.

Maloszewki, P. and Zuber, A. (1990). Mathematical modeling of tracer behaviour in short term experiments in fissured rocks. WRR, 26(7) July.

Mas-Pla, J., Yeh, T.C.J., McCarthy, J.F. and William, T.M. (1992), A forced gradient tracer experiment in a coastal sandy aquifer, Georgetown Site, South Carolina, *Ground Water*, 30(6), 964-959.

Masters, G. (1990). (Personal communication) from: Introduction to Environmental Engineering and Science. Professor, Stanton University.

McFetters, G.A., Bissonnette, G.K., Jezeski, J.J., Thompson, C.A. and Stuart, D.G. (1974). Comparative survival of indicator bacteria and enteric pathogens in well water. *Appl. Microbiol*. 27: 823-829.

McGinnis, J.A. and DeWalle, F. (1983). The movement of typhoid organisms in saturated, permeable soil. *J. Am. Water Works Assoc*. 75: 266-271.

Meiri, D.(1989), A tracer test for detecting cross contamination along a monitoring well column, *GMWR.*, 79-81.

Moench, A.F. (1989), Radial convergent dispersion: A Laplace transform solution for aquifer tracer testing, *Water Resour. Res.*, 25(3), 439-447.

Moltyaner, G.L. and Killey, R.W.D. (1988), Twin Lake tracer tests: Transverse dispersion, *Water Resour. Res.*, 24(10), 1628-1637.

Molz, F.J., Melville, J.G., Guven, O., Crocker, R.D. and Matteson, K.T. (1985), Design and performance of single well tracer tests at the Mobile Site, *Water Resour. Res.*, 21(10), 1497-1502.

Molz, F.J., Guven, O., Melville, J.G., Nohrstedt, J.S. and Overholtzer, J.K. (1988), Forced gradient tracer tests and inferred hydraulic conductivity distributions at the Mobile Site, *Ground Water*, 26(5), 570-579.

Molz, F.J., Guven, O., Melville, J.G., Crocker, R.D. and Matteson, K.T. (1986), Performance, analysis and simulation of a two well tracer test at the Mobile Site, *Water Resour. Res.*, 22(7), 1031-1037.

Muller, M. (1989). In: DWAF, (1997). A Protocol to manage the potential of groundwater contamination from on-site sanitation (National Sanitation Co-ordination Office, Directorate of Geohydrology).

Murray, J.P. and Laband, S.J. (1979). Degradation of poliovirus by adsorption on inorganic surfaces. *Appli. Environ. Microbiol*. 37: 480-486.

Naymik, T.G., and Sievers, M.E. (1985), Characterisation of dye tracer plumes , *Ground Water*, 23(6), 746-752.

Ncube E. (1998). The influence of pitlatrines on the quality of borehole water in the Thaba Nchu area. M.Sc.-short dissertation (in progress), UFS, Bloemfontein.

NHMRC (1990). Rural water supplies-nitrate health effects. In Report of the 10th Session, National Health and Medical Research Council, Canberra, Australia, p 22.

Novakowski, K.S. and Lapcevic, P.A. (1994), Field measurements of radial solute transport in fractured rock, *Water Resour. Res.*, 30(1), 37-44.

Novakowski, K.S., Evans, G.V., Lever, D.A. and Raven, K.G. (1985), A field example of measuring hydrodynamic dispersion in a single fracture, *Water Resour. Res.*, 21(8), 1165-1174.

Novakowski, K.S., Flavelle, P.A., Raven, K.G. and Cooper, E.C. (1985), Determination of groundwater flow path ways in fractured plutonic rock using a radioactive tracer, *Int. J. Appl. Isot.*, 36(5), 339-404.

Novakowski, K.S., Lapcevic, P.A., Voralek, J. and Bickerton, G. (1995), Preliminary interpretation of tracer experiments in a discrete rock fracture, *Geophysical Research Letters*, 22(11), 1417-1420.

Novakowski, K.S, (1992), The analysis of tracer experiments conducted in divergent radial flow fields, *Water Resour. Res.*, 28(12), 3215-3225.

O'Brien, R.O., Keller, C.K. and Smith, J.L. (1996), Multiple tracers of shallow groundwater flow and recharge in hilly loess, *Ground Water*, 34(4), 675-682.

Palmer, A.D. and Nadon, R.L. (1986), A radial injection tracer experiment in a confined aquifer, Scarborough, Ontario, Canada, *Ground Water*, 24(3), 322-321.

Palmer, I. (1997). In: DWAF, (1997). A Protocol to manage the potential of groundwater contamination from on-site sanitation (National Sanitation Co-ordination Office, Directorate of Geohydrology).

Parker, W.F. and Mee, B.J. (1982). Survival of Salmonella Adelaide and fecal coliforms in coarse sands of the Swan coastal plain, western Australia. *Appli. Environ. Microbiol*. 43: 981-986.

Parsons, R. and Jolly, J. (1994). In: DWAF, (1997). A Protocol to manage the potential of groundwater contamination from on-site sanitation (National Sanitation Co-ordination Office, Directorate of Geohydrology).

Patrick, W.H. and Wyatt, R. (1964). Soil nitrogen loss as a result of alternate submergence and drying. *Soil Sci. Soc. Am. Proc*. 28: 647-653.

Peterson, T.C. and Ward, R.C. (1988). Impact of adverse hydrological events on bacterial translocation in coarse soils near on-site wastewater treatment systems. In On-site wastewater treatment, Proc. 5th National Symposium on Individual and Small Community Sewage Systems., ASAE, St. Joseph, MI. pp. 87-93.

Peterson, T.C. and Ward, R.C. (1989). Bacterial retention in soils: New perspectives, new recommendations. *J. Environ. Health*. 51: 196-200.

Powelson, D.K., Simpson, J.R. and Gebra, C.P. (1990). Virus transport and survival in saturated and unsaturated flow through soil columns. *J. Environ. Qual*. 19: 396-401.

Ptak, T. and Teutsch, G. (1994), Forced and natural gradient tracer tests in a highly heterogeneous aquifer: instrumentation and measurements, *Journal of Hydrology*, 159, 79-104.

Ptak, T., and Schmid, G. (1996), Duel tracer transport experiments in a physically and chemically heterogeneous porous aquifer: effective transport parameters and spatially variability, *Journal of Hydrology*, 183, 117-138.

Reneau, R.B. and Pettry, D.E. (1975). Movement of methylene blue active substances from septic tank effluent through two coastal plain soils. *J. Environ. Qual*. 4: 370-375.

Reneau, R.B., (Jr.) (1979). Changes in concentrations of selected chemical pollutants in wet, tile-drained soil systems as influenced by disposal of septic tank effluents. *J. Environ. Qual*. 8: 189-196.

Rossi, P. (1994). Advances in biological tracer techniques for hydrology and hydrogeology using bacteriophages; Otimisation of the methods and investigation of the behaviour of the bacterial viruses in surface waters and in porous and fractures aquifers. Faculty of sciences, University of Neuchâtel.

Sandhu, S.S., Warren, W.J. and Nelson, P. (1979). Magnitude of pollution indicator organisms in rural potable water. *Appli. Environ. Microbiol*. 37: 744-749.

Sauty, J.P. (1978). Identification des parameters du transport hydrodispersif dans les aquifers par interpretation de tracages en ecoulement cylindrique convergent ou divergent. J.Hydrol. 39, 69-103.

Sauty, J.P., Kinzelbach, W. and Voss, A. (1992), CATTI: Computer Aided Tracer Test Interpretation program documentation.

Schwab, M. and Genthe, B. (1998). Environmental health risk assessment: A primer for South Africa. CSIR report section 1C: ENV/S-I 98029.

Short, S. (1999) Groundwater Digest mailing list No 1200. New South Wales, Australia.

Sikora, L.J. and Keeney, D.R. (1976). Denitrification of nitrified septic tank effluent. *J. Water Poll. Con. Fed*. 48: 2 108-2 025.

Sililo, O., van der Voort, I. and Cave, L. (1997), Technologies for predicting water and solute movement on the subsurface: Aquifer parameters and tracer techniques, Internal report ENV/S 197006, CSIR, Stellenbosch.

Smart, P.L., and Laidlaw, I.M.S. (1977), An evaluation of some fluorescent dyes for water tracing, *Water Resour. Res.*, 13(1), 15-33.

Sobsey, M.D. and Hickey, A.R. (1985). Effects of humic and fulvic acids on poliovirus concentration from water by microporous filtration. *Appli. Environ. Microbiol*. 49: 259-264.

Sobsey, M.D., Dean, C.H., Knuckles, M.E. and Wagner, R.A. (1980). Interactions and survival of enteric viruses in soil materials. *Appli. Environ. Microbiol*. 40: 92-101.

Stenstrom, T.A. and Hoffner, S. (1982). Reduction of enteric microorganisms in soil infiltration systems. In Eikum, A.S., and R.W. Seabloom (eds.) Alternative Wastewater Treatment. D. Reidel Publ. pp. 169-181.

Stewart, L.W. and Reneau, R.B., (Jr.) (1988). Shallowly placed, low pressure distribution system to treat domestic wastewater in soils with fluctuating high water tables. *J. Environ. Qual*. 17: 499-504.

Tare, V. and Bokil, S.D. (1982). Wastewater treatment by soils: Role of particle-size distribution. *J. Environ. Qual*. 11: 596-602.

Tate, R.L., (III) (1978). Cultural and environmental factors affecting the longevity of Escherichia coli in Histo soils. *Appli. Environ. Microbiol*. 35: 925-929.

Tredoux, G. (1993). A preliminary investigation of the nitrate content of groundwater and limitation of the nitrate input; WRC Report No. 368/1/93, Water Research Commission, Pretoria.

Tsang, Y.W. (1992). Usage of "equivalent apertures" for rock fractures as derived from hydraulik and tracer tests, WRR, Vol. 28, No.5, 1451 - 1455.

Van Tonder, G. J. (1999). Personal communication. Professor, IGS, UFS, Bloemfontein, South Africa.

Van Wyk, A.E. (1998). Tracer experiments in fractured rock aquifers. M.Sc. dissertation. Institute for Groundwater Studies, University of the Free State, P.O. Box 339, Bloemfontein 9300, South Africa.

Ver Hey, M.E. and Woessner, W.W. (1988). Documentation of the degree of waste treatment provided by septic systems, vadose zone and aquifer in intermontane soils underlain by sand and gravel. In On-site waste-water treatment, Proc. 5th National Symposium on Individual and Small Community Sewage Systems., ASAE, St. Joseph, MI. pp. 77-86.

Vivier, J.J.P. (1993). Korrelasie tussen die hersteltye van giet toetse en die lewering van pomptoetse. (Honneurs projek, IGS, UFS).

Volz, M.G. and Starr, J.L. (1977). Nitrate dissimilation and population dynamics of denitrifying bacteria during short term continuous flow. *Soil Sci. Soc. Am. J*. 41: 891-896.

Walker, W.G., Bouma, J., Keeney, D.R. and Magdoff, F.R. (1973). Nitrogen transformations during subsurface disposal of septic tank effluent in sands: I. Soil transformations. *J. Environ. Qual*. 2: 475-480.

Walton, G. (1951). Survey of literature relating to infant methemoglobinemia due to nitrate-contaminated water. *Am. J. Public Health*. 41: 986-996.

Wang, H.Q. and Crampon, N. (1995). Method for interpreting tracer experiments in radial flow using modified analytical solutions. J.Hydrol. 165 11-31.

Wellings, F.M., Lewis, A.L. and Mountain, C.W. (1974). The fate of virus in Florida soils following secondary effluent spray irrigation. In Individual on-site wastewater systems, Proc. 1st Nat. Conf. Ann Arbor, MI. pp. 121-126.

Wellings, F.M., Lewis, A.L. and Mountain, C.W. (1976). Demonstration of solids-associated virus in wastewater and sludge. *Appli. Environ. Microbiol*. 31: 354-358.

Wellings, F.M., Lewis, A.L., Mountain, C.W. and Pierce, P.W. (1975). Demonstration of virus in ground-water after effluent discharge onto soil. *Appl. Microbiol*. 29: 751-757.

WHO (1985). Health hazards from nitrates in drinking water. Report on a WHO meeting, Copenhagen.

Xu, Y. and Braune, E (1995). A Guideline for Groundwater Protection for the Community Water Supply and Sanitation Programme.

Xu, Y. (1998). Delineation of borehole protection areas in fractured aquifers. Unpublished Ph.D. thesis, UOFS, Bloemfontein.

Yates, M.V., Gerba, C.P. and Kelley, L.M. (1985). Virus persistence in groundwater. *Appli. Environ. Microbiol*. 49: 778-781.

Yeager, J.G. and Obrien, R.T. (1979). Structural changes associated with poliovirus inactivation in soil. *Appli. Environ. Microbiol*. 38: 702-709.