AN EVALUATION OF SEPTIC TANK EFFLUENT MOVEMENT IN SOIL AND GROUNDWATER SYSTEMS

By

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Dedication

To my Parents
and
Marie

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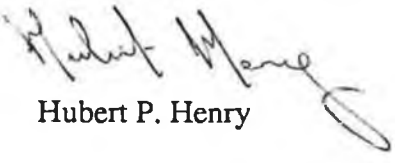
Declaration

This thesis has not previously been submitted to this or any other college and, with acknowledged exception, is entirely my own work.



Hubert P. Henry

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Abstract

Recent studies have shown that septic tank systems are a major source of groundwater pollution. Many public health workers feel that the most critical aspect of the use of septic tanks as a means of sewage disposal is the contamination of private water wells with attendant human health hazards.

In this study the movement and attenuation of septic tank effluents in a range of soil/overburden types and hydrogeological situations was investigated. The suitability of a number of chemical and biological tracer materials to monitor the movement of septic tank effluent constituents to groundwater sources was also examined. The investigation was divided into three separate but interrelated sections.

In the first section of the study the movement of septic tank effluent from two soil treatment systems was investigated by direct measurements of soil nutrient concentrations and enteric bacterial numbers in the soil beneath and downgradient of the test systems. Two sites with different soil types and hydrogeological characteristics were used. The results indicated that the attenuation of the effluent in both of the treatment systems was incomplete. Migration of nitrate, ammonium, phosphate and fecal bacteria to a depth of 50 cm beneath the inverts of the distribution tiles was demonstrated on all sampling occasions. The lateral migration of the pollutants was less pronounced, although on occasions high nutrients levels and fecal bacterial numbers were detected at a lateral distance of 4.0 m downgradient of the test systems. There was evidence that the degree and extent of effluent migration was increased after periods of heavy or prolonged rainfall when the attenuating properties of the treatment systems were reduced as a result of saturation of the soil.

The second part of the study examined the contamination of groundwaters downgradient of septic tank soil treatment systems. Three test sites were used in the investigation. The sites were chosen because of differences in the thicknesses and nature of the unsaturated zone available for effluent attenuation at each of the locations. A series of groundwater monitoring boreholes were installed downgradient of the test systems at each of the sites and these were sampled regularly to assess the efficiency of the overburden material in reducing the polluting potential of the wastewater. Effluent attenuation in the septic tank treatment systems was shown to be incomplete, resulting in chemical and microbiological contamination of the groundwaters downgradient of the systems. The nature and severity of groundwater contamination was dependent on the composition and thickness of the unsaturated zone and the extent of weathering in the underlying saturated bedrock.

The movement of septic tank effluent through soil/overburdens to groundwater sources was investigated by adding a range of chemical and biological tracer materials to the three septic tank systems used in section two of the study. The results demonstrated that a single tracer type cannot be used to accurately monitor the movement of all effluent constituents through soils to groundwater. The combined use of lithium bromide and endospores of *Bacillus globigii* was found to give an accurate indication of the movement of both the chemical and biological effluent constituents.

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CHAPTER 1
INTRODUCTION

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1:1 Introduction

Septic tank systems have been widely used in both developed and developing countries for the treatment of domestic wastewater in rural areas for over 100 years. In recent decades they have become increasingly popular in suburban areas not serviced by public sewer systems. The widespread use of the septic tank system has continued in the face of a consistent history of failure, with severe localised groundwater pollution, and almost unanimous disapproval by researchers in the field. The feasibility of using septic tank systems as a method of treating domestic wastewater was being questioned as early as 1956 when Kiker suggested that ' at best a septic tank is a poor substitute for centralised sewage collection and should be avoided whenever possible '.

As a primary treatment system septic tanks do not significantly reduce the polluting potential of the wastewater. The bulk of the treatment takes place in the soil through various physical, chemical and biological interactions between the effluent and soil colloids. In the United States (U. S.) approximately three billion m^3 of septic tank effluent is discharged into the soils for treatment annually (Bitton and Gerba, 1984). However, less than 50% of these soils are thought to be capable of achieving an adequate reduction in the pollution potential of the waste (Patterson et al, 1971). In Ireland there are an estimated 300,000 septic tank systems serving a population in the region of 1.2 million people and discharging approximately 78 million m^3 of wastewater to soil annually (Henry, 1988). Again only half of these soils are considered capable of providing sufficient treatment to prevent groundwater pollution (O' Hegarthy, 1976 and Daly, 1987).

Recent reports in the United States (Canter and Knox, 1985 and Kaplan, 1987), Ireland (Henry et al, 1987 and Aldwell et al, 1988) and New Zealand (Sinton, 1986) have highlighted septic tanks as one of the major polluters of groundwater supplies. Despite this overwhelming evidence and the serious implications for groundwater quality and human health, little research has focused on the mechanisms and situations by which the wastewater migrates through the unsaturated zone to groundwater. Futhermore, much of the research to date has been site specific concentrating on a single set of site conditions and in many cases focusing on a limited number of effluent constituents e. g. nitrate, phosphate or fecal bacteria. Few studies have investigated effluent migration and groundwater contamination in a number of sites with different characteristics with a view to using the information in the development of guidelines or codes of practice. This lack of research interest was aptly

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summarised by Winneberger (1984). He concluded that:

> *Knowledge from other disciplines had not found their way into septic tank technology. But most surprisingly, a device as important and deceptively complex as the septic tank system was, had received appallingly little scientific attention.*

There is an urgent need for strict regulations and guidelines on the use of septic tank systems in areas dependent on groundwater supplies. These regulations can only be formulated and implemented as a result of accurate scientific research. The research must adopt a multidisciplinary approach as the efficiency of effluent treatment by a system depends not only on its design, construction and maintenance but also on the chemical/biological nature of the effluent, the soil type and hydrogeological setting of the sites. This study attempts to achieve this multidisciplinary dimension by assessing the migration of septic tank effluent and the resulting contamination of groundwaters in a number of sites with different soil types and hydrogeological conditions and by investigating tracer materials suitable for monitoring effluent movement from treatment systems to groundwater. The scope of the study is summarised in Figure 1.1, (p4).

1:2 Aims and Objectives

The aim of this study is to assess the movement and attenuation of septic tank effluent in a range of soils, its subsequent pollution of groundwater and to identify tracer materials suitable for monitoring the effluent movement in a range of soil types and hydrogeological settings. The aim will be achieved by meeting the following objectives:

(i) Assessing the quality and variation in septic tank effluent

(ii) Assessing the movement and attenuation of septic tank effluent in two soil treatment systems

(iii) Assessing the survival of enteric indicator bacteria in a range of soil types

(iv) Investigating the influence of the nature and depth of the unsaturated zone on the groundwater quality downgradient of a number of septic tank treatment systems

(v) Investigating seasonal variations in groundwater quality downgradient of a septic tank system

(vi) Investigating the relative usefulness of a range of chemical and biological tracers for monitoring the movement of septic tank effluent to groundwater in a number of different soil types and hydrogeological settings.

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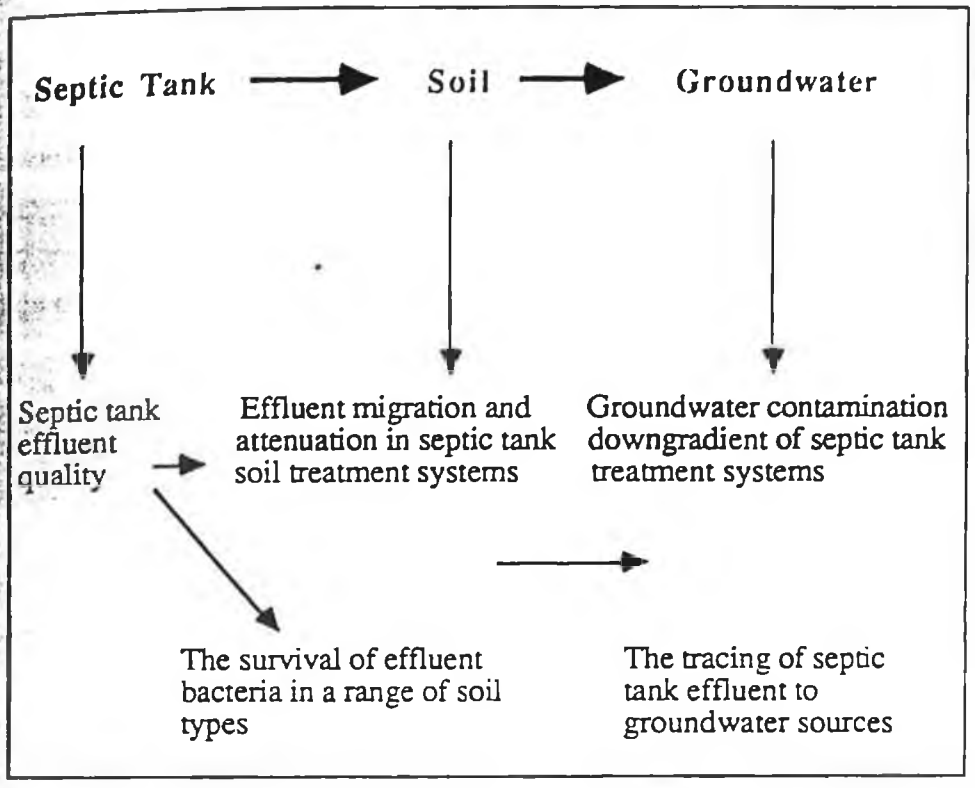
Figure 1.1

The Scope of the Study



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1:3 Thesis Layout

The thesis is divided into three separate but interrelated sections:

1. **The Movement and Attenuation of Septic Tank Effluent in Soils (Chapter 2)**

This section presents the results of a study of the movement and attenuation of septic tank effluent in two soil distribution systems with different soil types and thicknesses of unsaturated zone. The chemical and biological nature of the effluents are also examined. In a separate but related laboratory study bacterial survival in a range of soil types was also assessed.

2. **The Contamination of Groundwater Downgradient of Septic Tank Treatment Systems (Chapter 3)**

Samples were taken from a number of monitoring boreholes installed downgradient of the septic tank systems at three test sites. This section presents the results of a study on the extent of groundwater contamination at these sites.

3. **The Tracing of Septic Tank Effluent to Groundwater Sources (Chapter 4)**

This section presents the results of tracing experiments in which a range of chemical and biological tracer materials was introduced into the septic tanks at the three test sites noted above (Chapter 3). The monitoring boreholes were used to detect the presence of the tracers in groundwater downgradient of the sites.

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1:4 Septic Tank Systems - An Overview

1:4.1 General

A septic tank is a buried watertight container, designed and constructed to receive wastewater from a household, to separate solids from liquids and to provide limited anaerobic digestion of organic matter. The solids are stored in the tank and the liquid supernatant is allowed to overflow into the surrounding soil for further treatment. A septic tank functions primarily as a settlement chamber with only limited reduction of the Biochemical Oxygen Demand (B. O. D.) and Suspended Solids (S. S.) content of the wastewater. The settled solids (sludge) on the floor of the tank are partially digested by anaerobic microorganisms with the liberation of gases, principally carbon dioxide (CO_2) and methane (CH_4). Oils, greases, fats and soaps in the wastewater are floated to the surface by this gas and form a thick scum over the liquid mass (an indication that the tank is functioning properly).

The wastewater from the household includes toilet flushings (black water or sewage), washbasin and bathtub washings and kitchen waste (grey water or sullage). In some systems the kitchen and bathtub washings bypass the septic tank and are piped directly to a separate soil treatment system. In the past it was felt that the high concentrations of detergents and salt in such wastes might upset the treatment process within the tank (Patterson et al, 1971). Current opinions do not uphold this theory and suggest that average household concentrations of detergents and salts should not adversely affect the proper functioning of a septic tank (Dewhurst, 1970).

1:4.2 Septic Tank - Design, Construction and Maintenance

The septic tank was first developed by Mouras in 1860. It was introduced to the United Kingdom by Cameron some 20 years later and was patented there in 1881. It's first appearance in the United States was in Boston in 1883. Since then it has undergone a number of changes in design and operation but there has been no major modification to the system in the last 50 years. The tank can be prefabricated or built in - situ and can be manufactured from a range of materials, the most common of which are concrete and fibre - glass. The shape of the tank is important with regard to velocity flow, circulation, depth of sludge and the amount of dead space available. Current practices favour rectangular tanks although studies have shown little difference in performance between rectangular and cylindrical designs when sludge storage capacities were similar (Canter and Knox, 1985).

Ideally a tank should have at least two chambers connected in series, the aim being to minimise disturbance of the liquid solid mass by the incoming effluent by providing a baffle to stem any strong flows. In general the length of the tank should be three times its width with a minimum internal width of 0.75 metres. The depth of

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liquid in the tank should be between 1.2 to 1.7 metres with a free air space of 0.3 metres (Paz Maroto, 1960). This is to prevent streaming of the effluent from inlet to outlet pipe which would dangerously reduce the retention time. T - pipes are used at both ends of the tank to minimise disturbance of the liquid mass. The tank should be of sufficient size to allow a retention time of at least 24 hours. The total capacity of the tank should be calculated to allow for sludge storage between desludgings. Most European countries have codes of practice for the sizing of septic tanks. There is considerable variation in the design recommendations due to the different climatic, social and cultural conditions in each country. It is accepted that the size of tank required depends on the following factors:

(i) the influent wastewater flow
(ii) the retention time required for effective solids removal
(iii) the sludge accumulation rate
(iv) the frequency of desludging.

(Anon., 1982)

The influent wastewater flow depends on water consumption within the house while the retention time for effective solids removal depends on the number of users. The sludge accumulation rate varies from country to country, depending on climatic conditions, and ranges from 30 litres/person/year in Southern Europe to 70 litres/person/year in Northern Europe. The frequency of desludging depends on the sludge accumulation rate and the cost of emptying the tank. The recommended time interval between desludging also varies considerably between countries from every six months to once every four years, although the former is most commonly recommended. Where local conditions do not permit regular desludging additional sludge storage capacity should be allowed for (Anon., 1982).

In very small septic tanks there is a danger that peak flows would disturb the settled sludge and scum causing them to be washed out into the soil treatment system. Consequently a minimum design size is usually stipulated. The United States public health service have recommended that the minimum tank capacity should be 1.5 times the daily volume of wastewater, where the volume is between 1900 and 5700 litres. A capacity of less than 1900 litres is not encouraged. In Ireland the Institute of Industrial Research and Standards (I. I. R. S.) recommend a minimum size of 2720 litres.

Regular maintenance of the tank is of paramount importance to the efficient working of the system. Lack of maintenance leads to poor effluent quality and may result in solids being washed into the soil treatment system with ultimate clogging and failure. Most countries produce guidelines for the construction and maintenance of septic tank systems. In Ireland, reference should be made to I. I. R. S. S.R. 6 (Anon., 1975) and in New Zealand to CP - 44 (Anon., 1961).

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1:4.3 Septic Tanks - Efficiency of Treatment

A septic tank functions primarily as a settlement chamber and as such only affords limited digestion of the wastewater. The efficiency of treatment within the tank depends on many factors, primarily the design, construction and maintenance of the system. The volume and nature of the waste is also important.

In general, approximately 50% of the solids will be removed but this can increase to 70% in a well - constructed two chamber tank. B. O. D. removal within the tank is considerably less, ranging from 15 to 30 %, although this can also be extremely variable (Patterson et al, 1971 and Goldstein and Wenk, 1972).

The effluent from a septic tank is of poor quality and highly polluting if it reaches surface or groundwaters. The effluent contains high numbers of fecal bacteria and viruses and large amounts of phosphorous and nitrogen (mainly as ammonia), as well as having a high B. O. D. and S. S. content. It is a common misconception that the tank will effectively remove the bacteria and other microorganisms contained in the waste. Studies have shown that the removal of these organisms within the tank is negligible (Patterson et al, 1971). Even the most efficient tank can only offer partial treatment, hence the physical, chemical and biological quality of the effluent is such that it cannot be discharged directly to surface or groundwaters without further treatment. This treatment takes place in the soil treatment system into which the effluent is channelled on leaving the tank.

1:4.4 The Soil Treatment System

The soil is an integral part of the process by which the effluent strength is reduced before reaching the saturated zone. It has been suggested by a number of authors that the term *septic tank system* or *treatment works* be used to indicate that the septic tank and the soil treatment system should be regarded as a single treatment unit. Once the effluent leaves the septic tank it enters the soil treatment system where it interacts with the soil colloids. There are two types of soil treatment system commonly in use :

(i) Soakage pits

(ii) Distribution fields (also called percolation, tile or absorption fields).

The first system simply allows the effluent to flow into an excavated hole filled with stone or rubble. The main disadvantage of this is that the effluent is concentrated into a small area which may become clogged and quickly fail (Patterson et al, 1971). The use of soakage pits as a disposal option is not recommended (Patterson et al, 1971 and Anon., 1975).

Absorption fields are designed to evenly distribute the effluent through a large volume of soil via perforations in a pipe distribution network. The soil's ability to

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effectively treat the waste depends on the design, configuration and loading of the pipe distribution network, maintenance of the tank, and the constituents of the waste (in addition to the soil characteristics).

The extent to which attenuation of the effluent takes place in the regolith (soil and overburden) depends on the ion exchange capacity, the porosity, permeability and texture of the regolith, the thickness of the regolith beneath the site, the depth of the water table and the slope of the ground surface (Huddleston and Olson, 1967 and Patterson et al, 1971).

1:4.5 Failure of the Soil Treatment System

Not all soils are capable of effectively treating septic tank effluent. More than half the soils in the United States are unsuitable for septic tank systems with respect to percolation rate. Failure of the systems has been reported to be between 25 and 50 % (O'Hegarty, 1976 and Patterson et al, 1971). It is estimated that half of these failures can be attributed to the location of absorption fields in soils of low permeability, a characteristic of over 50% of Irish soils (Daly, 1987). Another major reason for failure is location in an area with a high water table. This can cause ponding of the effluent on the surface with resulting health hazards. In addition failure can occur if the density of septic tank systems in the area is too high, causing the soil to be overloaded.

Failure can also occur in a septic tank system situated in a soil with high permeability. Although it is unlikely to become clogged, severe groundwater pollution can occur by the rapid passage of wastewater through the unsaturated zone without sufficient contact time with the soil for treatment (Caldwell, 1938 and Patterson et al, 1971).

1:4.6 Groundwater Pollution by Septic Tank Effluent

Septic tank systems are the most frequently reported source of groundwater contamination (Geraghty and Miller, 1978). Recent studies have conclusively proven that septic tank failure is a major cause of groundwater pollution (Patterson et al, 1971; McCoy and Hagedorn, 1979; Stewart and Reneau, 1981; Sinton, 1986; Daly, 1987 and Henry et al, 1987). Many public health workers feel that the most critical effect of septic tank systems is the contamination of private water wells. The human health implications of such contamination are considerable. Outbreaks of typhoid fever, infectious hepatitis, gastrointestinal infections and infantile methaemoglobinemia have all been linked to malfunctioning septic tanks.

Almost half the reported water disease outbreaks in the United States every year are due to the consumption of contaminated groundwater (Keswick et al, 1982). Overflow from septic tanks was responsible for 42% of the reported outbreaks of disease (Craun, 1979).

Pollution of groundwater by septic tank effluent can be chemical or biological in nature, or both. The poor microbiological quality of domestic well water supplies has been well documented. A recent study of rural groundwater sources in the U. S. showed 92% to be contaminated with coliform bacteria (Bitton and Gerba, 1984), while a similar study in western Ireland found that 68% of all rural groundwater supplies contained fecal Coliforms, fecal Streptococci or both (Aldwell et al, 1988). Septic tank effluent was believed to be the main source of contamination in both cases.

CHAPTER 2
THE MOVEMENT AND ATTENUATION OF SEPTIC TANK EFFLUENT IN SOIL TREATMENT SYSTEMS

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2:1 Introduction

A septic tank functions primarily as a settlement chamber and as such affords limited digestion of the wastewater entering it. Consequently the effluent from the system is of very poor quality containing large concentrations of chemical and biological pollutants, and cannot be discharged directly to ground or surface waters without further treatment. This treatment takes place in the soil into which the effluent is channelled on leaving the tank.

The soil plays an integral part in the process by which the effluent strength is reduced before reaching the saturated zone. Soils have the ability through physical, chemical and microbiological processes to effectively restrict and attenuate the various effluent constituents and reduce their polluting potential. Different soils have different attenuation or renovation capacities depending on many factors including permeability, ion exchange capacity, texture, thickness of overburden beneath the site, depth to the water table and slope of the land (Huddleston and Olson, 1967).

Not all soils are capable of effectively treating the waste and the failure of soil treatment systems resulting in ground and surface water pollution is well documented in the literature (Patterson et al, 1971 and Bitton and Gerba, 1984). In the United States (U. S.) the number of system failures appears to be very large. O'Hegarty (1976) reports that the percentage of failures is in the region of 24 to 50, while Scalf et al (1977) quote a figure of 50%. The primary reason for failure is the unsatisfactory permeability of the soil beneath the treatment system. Excessive permeability can result in rapid migration of the effluent through the soil profile. Because of insufficient contact with the soil particles the effluent may not receive adequate treatment and this can lead to severe localised pollution of groundwater. Conversely, insufficient permeability causes ' ponding ' of effluent in the treatment system which often results in the pollution of surface and groundwaters, as well as the generation of a considerable human health risk. There are no comparable figures available for the number of system failures in the Republic of Ireland. However, it has been reported that approximately half of the soils have low permeability and that over 20% of them are waterlogged for at least part of the year, which indicates that the number of system failures may be quite high (O'Hegarty, 1976 and Daly, 1988).

It is difficult to define the type of soil which is ideally suited to the disposal of septic tank effluents. A well aerated soil with a medium permeability provides excellent conditions for the breakdown of effluent organic constituents through the action of heterotropic bacteria. However, these conditions also favour the nitrification of ammonia and organic nitrogen in the wastewater which can give rise to severe nitrate pollution of groundwater. The problem is further complicated by the fact that rainfall

can affect the degree of treatment within the system by physically washing pollutants through the soil profile or reducing the soils sorptive capacity by decreasing the ionic strength of the soil solution. Another factor that can make the assessment of soil suitability difficult is that pollutants can travel rapidly through cracks and joints in the subsurface and can bypass the purifying soil layers.

Much research has been carried out in the U. S. on the suitability of various soils for effluent disposal. To date this research has been focused on either the chemical (Ellis, 1973; Crites and Pound, 1976 and Rea and Upchurch, 1980) or microbiological (Hagedorn et al, 1978; Reneau et al, 1978; Viraraghavan, 1978; McCoy and Hagedorn, 1979 and Sinton, 1986) aspects of wastewater migration, despite the fact that a septic tank effluent is a complex mixture of both chemical and biological constituents. Furthermore, guidelines and regulations on the design, construction and maintenance of septic tank systems are strictly enforced in the U. S. and few studies have been carried out in countries where regulations are less stringent.

This section of the study attempts to address these problems by assessing the movement and attenuation of both the chemical and microbiological constituents of the septic tank effluent in two soil treatment systems, neither of which have been designed, constructed or maintained in compliance with any recognised standard. The survival of *Escherichia coli* bacteria in a range of soil types is also examined as is the quality of and variation in septic tank effluents. The study is presented in the following sections:

2:1 Introduction

2:2 Literature review: This section reviews the relevant literature on the processes and mechanisms of septic tank effluent attenuation in soils. The treatment efficiency within the tank and the resulting effluent quality is also described

2:3 Site characteristics: A report on the two sampling sites used is given in this section. Included are descriptions of geology, soil type, topography and land use, as well as details on the design, construction and maintenance of the two septic tank treatment systems

2:4 Materials and Methods: The materials and methods used in the sampling, analysis and description of the test sites are presented here. The statistical methods used in the analysis of the results obtained are also described

2:5 Results: The main results of the study are presented in this section

2:6 Discussion: The results are discussed in detail with reference to previous investigations

2:7 Conclusions.

2:2 Literature Review

2:2.1 Septic Tank Treatment Efficiency and Effluent Quality

A septic tank is a buried watertight container designed and constructed to receive wastewater from a household, to separate solids from liquids and to provide limited digestion of organic matter. The solids are stored in the tank and the liquid supernatent is allowed to overflow into the surrounding soil for further treatment. The settled solids (sludge) on the floor of the tank are partially digested by anaerobic microorganisms, with the liberation of gases, principally carbon dioxide and methane. Oils, greases, fats and soaps in the wastewater are floated to the surface by this gas and form a thick scum over the liquid mass (an indication that the tank is functioning properly).

The wastewater from the house includes toilet flushings (blackwaters), washbasin and bathtub washings and kitchen waste (greywaters or sullage). In some systems the kitchen and bathtub washings bypass the septic tank and are piped directly to a separate soil treatment system. In the past it was felt that the high concentrations of detergents and salts in such waters might upset the treatment process within the tank (Patterson at al, 1971). Current opinions, however, do not uphold this theory and suggest that the average household concentrations of detergents and salts should not effect the proper functioning of a septic tank (Dewhurst, 1970 and Canter and Knox, 1985).

The primary function of a septic tank is to condition the sewage so that it can percolate more readily through the subsoil (Cain and Beatty, 1965). It acts primarily as a settlement chamber and as such affords only limited digestion of the wastewaters. The efficiency of treatment within the tank depends on many factors, primarily the design, construction and maintenance of the system. The volume and nature of the waste is also important (Patterson et al, 1971). Patterson et al (1971) and Goldstein and Wenk (1972) investigated the efficiency of treatment within a range of septic tanks. They both concluded that in general approximately 50% of the solids in the sewage are removed within the tank, which can be increased to 70% in a well constructed two chamber tank. B. O. D. removal was considerably lower, ranging from 15 to 30%, although very variable reductions were reported by both authors.
Canter and Knox (1985) cite research carried out by Viraraghavan (1976) who found that B. O. D. and C. O. D. removal efficiencies were in the order of 50% while total suspended solids removal was less than 25%. Canter and Knox (1985) also refer to a study by Laurence (1973) who observed that 34 to 35% of the suspended solids concentration was removed with B. O. D. removals of 15% or less.

Most authors agree that the effluent from a septic tank is of poor quality and highly polluting if it reaches ground or surface waters directly. It contains high numbers of fecal bacteria and viruses and large amounts of phosphorous and nitrogen (mainly as ammonia), as well as having a high B. O. D. and S. S. content (Patterson et al, 1971). Patterson et al (1971) carried out a review of the literature on septic tank effluent quality and summary results of their findings are presented in Table 2.2.1. More recently Canter and Knox (1985) reviewed research by the University of Wisconsin (Anon., 1978) and the U. S. Environmental Protection Agency (Anon., 1980). Based on this research they compiled a summary of the typical physical and chemical quality of septic tank effluent (Table 2.2.2, p17).

It is a common misconception that the septic tank will effectively remove the bacteria and other microorganisms in the waste (Cain and Beatty, 1965). Studies have shown that removal of these organisms within the tank is negligible (Patterson et al, 1971). McCoy and Ziebell (1975) demonstrated that there are large numbers of fecal bacterial in septic tank effluents, indicating that removal during the treatment process is low (Table 2.2.3, p18).

Similarily Canter and Knox (1985) report that the anaerobic process within the septic tank is largely ineffective in reducing the concentration of phosphorous and nitrogen in the influent wastewater and converts most of the influent phosphorous, in both the organic and condensed forms, to soluble orthophosphate. They report that the average influent phosphorous concentration was 25 mg/l, of which 8.8 mg/l (35%) was in the inorganic or orthophosphate form, with the remaining 16.2 mg/l (65%) in the organic form. A study by Doyle and Thom (1986) found orthophosphate concentrations in the effluent as high as 52 mg/l. Bourma (1979) reports on studies by others who found that more than 85% of the total phosphorous in septic tank effluents was in the soluble orthophosphate form.

Canter and Knox (1985) found that the average concentration of total Kjeldahl nitrogen (organic and ammonia nitrogen) in septic tank influent wastewaters was 38 mg/l, with 12 mg/l (32%) in the ammonium form. Anaerobic conditions within the tank result in much of the organic nitrogen being converted to ammonium. The nitrate (NO_3) concentration of septic tank effluent is usually low due to the lack of oxygen present in the treatment process.

2.2.2 Soil Treatment Systems

Once the effluent leaves the septic tank it is channelled into a soil treatment system where it is receives further treatment by reactions with soil colloids. There are

two soil treatment systems commonly in use:

(i) Soakage pits

(ii) Distribution fields (also known as percolation, absorption or tile fields).

The former simply consists of an excavated hole filled with stone or rubble into which the effluent is allowed to flow. The main disadvantage associated with this practice is that it concentrates the effluent into a small area which may quickly clog and fail (Patterson et al, 1971). Consequently, the use of soakage pits as a disposal option is not recommended (Bender, 1961 and Patterson et al, 1971). Distribution fields are designed to distribute the effluent through a large volume of soil via a pipe distribution network. Many countries produce guidelines for the construction and maintenance of these soil distribution systems e. g. in Ireland reference should be made to the I. I. R. S. S. R. 6 (Anon., 1975).

The primary requirements for the location of a septic tank waste disposal system were listed by Bourma (1979). These included:

(i) A satisfactory soil hydraulic conductivity as measured by the percolation test

(ii) Adequate soil volume and/or depth for effluent renovation

(iii) The lack of other site constraints such as slope, flooding, water tables, bedrock etc..

The ability of the system to effectively treat the waste depends, not only on location and the soil characteristics, but also on the design, configuration and loading of the pipe distribution network, the maintenance of the tank and the nature of the waste (Patterson et al, 1971). Soil characteristics of importance include ion exchange capacity, porosity, permeability, texture, thickness of overburden beneath the site, depth of the water table and slope of the ground surface (Bender, 1961; Huddleston and Olson, 1967; Patterson et al, 1971 and Miller and Wolf, 1975). Arguably the most important of these in determining the success or failure of a treatment system is permeability or hydraulic conductivity. Not all soils are hydraulically suitable for septic tank systems. Patterson et al (1971) reports that more than half of the soil in the United States is unsuitable for septic tank systems with respect to its permeability. Similarily O'Hegarty (1976) reports that 50% of Irish soils are unsuitable due to low permeability. Two types of system failure are noted. The first, due to insufficient permeability, occurs when the capacity of the soil to absorb effluent from the septic tank is exceeded and the waste backs up to the soil surface.

Table 2.2.1

Average Composition of Septic Tank Effluent.

Parameter	Study		
	Preul (1965)	Robeck et al (1964)	Polkowski and Boyle (1970)
pH (pH units)	7.0 - 7.5	7.5 - 8.1	-
B. O. D. (mg/l)	130	90	102
D. O. (mg/l)	0	-	-
S. S. (mg/l)	40	46	70
Alkalinity (mg/l)	300	400	-
Organic - N (mg/l)	10	5.4	16.2
NO_3 - N (mg/l)	0.15	0.11	0.13
NH_3- N (mg/l)	25	22	14
PO_4 - P (mg/l)	20	-	-
Potassium (mg/l)	10 - 15	-	-
Sodium (mg/l)	100	-	-
Sulphates (mg/l)	50	-	-
Chlorides (mg/l)	70	75	43
Coliform Bacteria (c. f. u. 's/100ml)	-	5.0×10^6	9.7×10^7

After Patterson et al (1971)

Table 2.2.2

The Physical and Chemical Quality of Septic Tank Effluent

Parameter	Concentration (mg/l)
S. S. (mg/l)	75
B. O. D. (mg/l)	140
C. O. D. (mg/l)	300
Total N (mg/l)	40
Total P (mg/l)	15

After Canter and Knox (1985)

Table 2.2.3

Bacteriological Quality of a Septic Tank Effluent

Organism	Bacteria/100 ml		
	No. of Samples	Mean	95% Confidence
Fecal Streptococci	97	3.8×10^3	$2.0\text{-}7.2 \times 10^3$
Fecal Coliforms	94	4.2×10^5	$2.9\text{-}6.2 \times 10^5$
Total Coliforms	91	3.4×10^6	$2.6\text{-}4.4 \times 10^6$
Pseudomonas aeroginosa	33	8.6×10^3	3.8×10^3 - 1.9×10^4
Total Bacteria	88	3.4×10^7	$2.5\text{-}4.8 \times 10^7$

After McCoy And Ziebell (1975)

This results in soil clogging and subsequent loss of infiltration capacity and is caused by a combination of physical, chemical and biological factors. When system failure occurs in this way, effluent constituents may travel by overland flow to contaminate lakes, rivers and inadequately sealed wells (Canter and Knox, 1985). The second type of failure is when pollutants move too rapidly through soils. Soils with a high permeability can quickly become overloaded with organic and inorganic chemicals and microorganisms, thus permitting rapid movement of contaminants to groundwater (Scalf et al, 1977).

The permeability of the soil intended to be used for the disposal of sewage is usually assessed using a percolation test. The first percolation test was developed by Henry Ryan in 1928 (Cain and Beatty, 1965 and Winneberger, 1984). Today there are many varieties in use. The most common test in the U. S. is the U. S. Public Health Service (U. S. P. H. S.) percolation test, while in Ireland the test described in I. I. R. S. S. R. 6 (Anon., 1975) is used. All percolation tests involve the excavation of a test hole on site and filling it with water to a specified depth. The rate at which the water empties is then measured and is expressed as minutes per inch (or millimeters per second). On the basis of this result the site is deemed suitable or unsuitable for effluent disposal. Most guidelines also use the results of percolation tests to specify the dimensions of the soil distribution field needed to adequately treat the waste and prevent hydraulic overloading of the system (Anon., 1961 and Anon., 1975). Over dependence on the test to the point where other important site characteristics are overlooked has been criticised by a number of authors. Stryker and Steele (1977) report that the ' perc. test ' procedure does not recognise many of the soil and site conditions that affect the correct functioning of septic tank treatment systems. They add that the variability in testing procedures and seasonal fluctuations make the percolation test an unsatisfactory technical base for the assessment of site suitability. These misgivings are also reported by Winneberger (1984) who stated that the percolation test is a far from a reproducible standard, irrespective of which local variety is used. However, Salvato (1975), while recognising that the validity and reliability of the test was questionable, felt that a properly constructed test provided a sound basis for determining the absorptive capacity of the soil.

2:2.2.1 The Failure of a Soil Treatment System due to Clogging

One of the main difficulties experienced with the accurate measurement of soil permeability is that the percolation rate of a soil changes with time (Cain and Beatty, 1965). It has been noted that during a long testing period the permeability of a soil varies in three distinct phases (Figure 2.2.1, p21). Phase 1 is a period of initial decrease in percolation rate, phase 2 is a period of increase and phase 3

is a period of gradual but steady decrease (Cain and Beatty, 1965; Winneberger, 1984 and Vigneswaran and Suazo, 1987). This typical curve has been demonstrated when soils have been inundated with fresh water and/or sewage effluents. It is agreed that the initial decrease in permeability noted in phase 1 is due to swelling of soil particles and dispersion of soil aggregates. The subsequent increase noted in phase 2 is attributed to an increase in the amount of pore space available for water movement, which occurs as a result of the dissolution of trapped air. The gradual reduction of permeability in phase 3, which is the most important aspect of the process in relation to septic tank effluent disposal, is due to the mechanical disintegration of soil aggregates, the clogging of soil pores by biological materials and the dispersion of soil aggregates by microorganisms (Cain and Beatty, 1965). In soil distribution systems, where wastewaters high in solids are added, all three phases are greatly compressed and the clogging phase proceeds rapidly (Laak, 1974; Miller and Wolf, 1975 and Winneberger, 1984). Whereas inundation of soil with freshwater results in a decrease in permeability in the upper 7.5 to 15 centimetres of soil, wastewater disposal results in the formation of a clogged zone at the wastewater/soil interface (Winneberger, 1984). This zone is formed by three distinct processes:

(i) Physical, where solids in the effluent physically clog the soil pores
(ii) Chemical, where soil colloids swell as a result of chemical processes
(iii) Biological, where bacteria or bacterial breakdown products reduce pore size

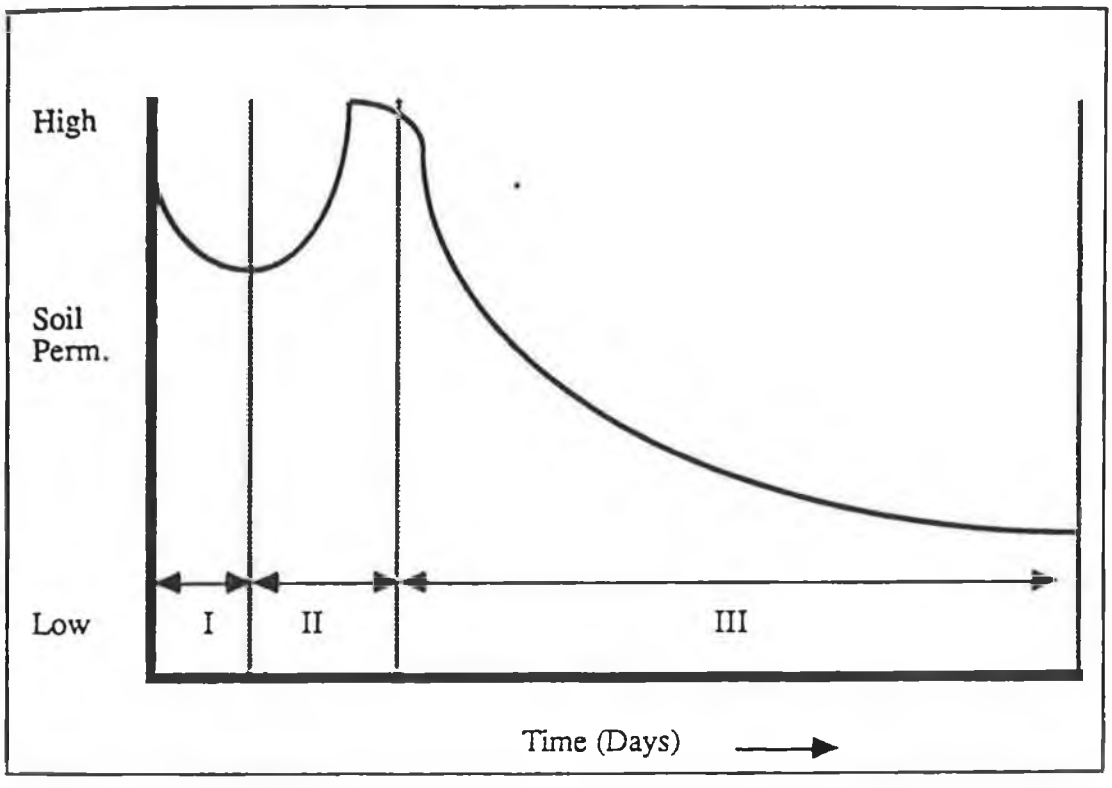
(Patterson et al, 1971)

The clogged zone (also referred to as biomat or biocrust) has many implications for the degree and effectiveness of effluent attentuation. It plays an important role in reducing the numbers of fecal bacteria in the percolating effluent (McCoy and Ziebell, 1975) and has also been reported to effectively remove other effluent constituents by various sorption reactions (Laak, 1974; McCoy and Ziebell, 1975 and Miller and Wolf, 1975). However, if the clogged mat reaches a stage where the infiltration rate of the soil is seriously reduced and the effluent can no longer percolate into the ground, the system will quickly fail. Often the immediate result of this failure is the saturation of surface soil with sewage which may move by overland flow to pollute surface waters (Patterson et al, 1971).

Once clogging has occurred and caused failure of the system there are some remedial measures which can be taken to reduce the clogged zone and restore soil permeability. Harkin et al (1975) recommend that the crust be partially dissolved using hydrogen peroxide. They report that H_2O_2 can operate under neutral or slightly alkaline conditions, rapidly destroying the crust by converting sulphide to soluble

Figure 2.2.1

The Effect of Prolonged Inundation on the Permeability of Soil



After Cain and Beatty (1965)

sulphate without the creation of noxious byproducts. However, Laak (1974) reported that the clogging rate is largely dependent on the B. O. D. and suspended solids in the effluent and conclude that reduction of the concentration of these elements in effluents can reduce soil clogging and allow the use of smaller soil treatment systems. Laak (1974) emphasises that the clogged zone is a living mat and that treatment processes designed to destroy the mat do not appear to be beneficial to the effluent treatment process. There is some evidence that the ' resting ' of percolation fields which have become clogged can result in a breakdown in the biomat with a subsequent return to the original permeability (Laak, 1974 and Miller and Wolf, 1975). Most guidelines recommend the installation of two percolation fields which can be operated alternately.

2:2.2.2 Attenuation of Septic Tank Effluent in Soil Treatment Systems

When assessing the polluting potential of a septic tank soil distribution system the movement and fate of effluent constituents in the underlying soils and overburden materials must be considered. The characteristics of septic tank effluents are such that several major components must be removed by soil renovation or attenuation if groundwater pollution is to be avoided. These include:

(i) Biological contaminants

(ii) Chemical contaminants

(a) Organic compounds yielding B. O. D., C. O. D. and total Coliform loads

(b) Compounds consisting of nitrogen, phosphorous and salts of varying solubilities.

(Miller and Wolf, 1975).

The depth within the soil profile at which removal is complete varies with the size of the particles, the soil texture and the rate of water movement. The effluent constituents will travel greater distances in coarse soils where the application rate is high (Canter and Knox, 1985). A study by Legrand (1972), cited by Canter and Knox (1985), examined the hydrogeological factors affecting pollutant movement from a treatment system. He concluded that the most important factors were the presence of clays which retard pollutant movement and facilitate sorption, and the depth to the water table to provide sufficient time for attenuation to occur. A good soil treatment system for receiving septic tank effluent should be permeable enough to absorb all effluent generated, provide a high level of treatment before the effluent reaches groundwater and have a long useful lifespan.

The soil acts as a chemical and biological filter through the many organic and inorganic reactions which occur when wastewaters pass through the soil profile. It may also chemically alter the waste contaminants and reduce or, in some cases, increase their polluting potential (Ellis, 1973). Therefore it is important to understand the chemical reactions that occur in the soil in order to accurately assess the suitability of a soil for the disposal of sewage. The properties of the soil medium are such that several mechanisms are available to act on the waste components thereby effecting at least some degree of renovation. These mechanisms include filtration, sorption, precipitation, chemical alteration, oxidation and biological transformation (Ellis, 1973; Miller and Wolf, 1975; Bourma, 1979 and Canter and Knox, 1985).

(i) Filtration

The mineral skeleton of the soil and the biocrust can serve as an effective effluent filter. Soil filtration processes not only remove suspended solids but also serve to retain microorganisms and facilitate biological treatment of both dissolved and suspended organic matter (Miller and Wolf, 1975 and McCoy and Ziebell, 1975). The greatest removal by filtration occurs at the wastewater/soil interface where the clogged zone exists. As progressively more of these particles accumulate on the filtering surface the size and quantity of the particles being filtered decreases (Miller and Wolf, 1975). This may eventually lead to severe clogging and system failure. The clogging is, however, beneficial in very permeable soils composed of coarse sands and gravels where the effluent movement is rapid and filtration would otherwise be minimal. A build - up of solids in the clogged zone would improve the treatment process by slowing down effluent transport and increasing contact time between effluent constituents and soil particles. This also enhances the probability of sorption, B. O. D. reduction and bacterial dieoff (Miller and Wolf, 1975). It has been shown that bacteria behave like other effluent particulate matter in soil treatment systems and as such filtration plays a vital role in restricting their movement through the soil profile to groundwater (McCoy and Ziebell, 1975; Miller and Wolf, 1975; Lewis et al, 1982 and Canter and Knox, 1985). This is discussed in more detail in 2:2.3.3.

(ii) Sorption

Miller and Wolf (1975) state that sorption is the binding of one substance by another through the mechanisms of absorption (taking in or reception by molecular or chemical action), adsorption (binding of a substance onto a surface) and persorption (the adsorption of materials in pores only slightly wider than the dimensions of the absorbed molecule). Sorption mechanisms have been shown to play a major role in the

renovation of sewage effluent constituents in soil profiles (Ellis, 1973 and Bourma, 1979). The iron, aluminium and hydrous oxides coating the subsoil clay minerals and magnesium - hydroxy clusters or coatings on the weathered surfaces of ferromagnesium minerals provide excellent sorption sites (Miller and Wolf, 1975). Elements such as phosphates, boron, ammonium and many others are removed from the percolating effluent in this way.

The electrostatic properties of clay minerals and organic matter provide an ion exchange capacity which is capable of sorbing ionic and biological constituents from the septic tank effluent. Cation exchange is the more dominant process in soils. Generally ion exchange reactions are electrostatic and can be influenced by the valence and hydration of the ion involved and the location and density of the charge on the solid component of the soil (Ellis, 1973). The cation exchange capacity (C. E. C.) of soils can range from 2 to 60 meq/100 grams of soil, with most soils having a C. E. C. of between 10 and 30. The difference arises because of the large variation in the quantity of clay and humus present in different soil types (Canter and Knox, 1985). Heavier textured soils have a higher C. E. C. (Miller and Wolf, 1975) and consequently are more efficient in restricting the movement of effluent constituents by this process.

(iii) Precipitation

Precipitation denotes a rapid crystallisation of a product of chemical reaction which is relatively insoluble in the medium in which it is found (Ellis, 1973). Many ions introduced to the soil by the addition of wastewater may combine with other ions in the soil solution to produce an insoluble product. This process is an important mechanism by which phosphate is removed from percolating septic tank effluent. In alkaline soils monocalcium phosphate is generally the product formed whereas aluminium and iron phosphates are more common in acid soils. This process is described in more detail in 2:2.3.2.

(iv) Oxidation and Biological Transformations

Septic tank effluents have been shown to contain a variety of natural and synthetic organic compounds collectively expressed in terms of B. O. D., C. O. D. or total organic carbon (T. O. C.) content. The effective treatment of this effluent requires oxidation of both the carbonaceous material and nitrogenous material consisting of organic or ammonia nitrogen (Miller and Wolf, 1975). Aerobic bacteria which use the carbon in the organic material as an energy source need oxygen for the decomposition process. This is supplied through the unsaturated soil pores by diffusion and/or oxygen dissolved in the percolating waters. The aerobic decomposition process results in the production of CO_2, H_2O, NO_3^-, SO_4^{2-} and microbial cells. Aerobic conditions

should prevail in the soil treatment system (Miller and Wolf, 1975 and Bourma, 1979). A well drained soil with an adequately designed and installed disposal system is capable of supplying sufficient oxygen for B. O. D. and C. O. D. breakdown. However, when oxygen is deficient anaerobic metabolism occurs, resulting in the production of CH_4, H_2, NH_4^+, H_2S, CO_2 and H_2O. Anaerobic breakdown occurs at a slower rate and is less complete than the aerobic process, with the production of intermediates such as acids, alcohols, amines and mercaptans. The biological transformations that occur in the soil treatment system include organic matter decomposition and nutrient assimilation by plants. Greater biological activity will therefore occur in the upper layers of soil beneath the soil distribution system (Canter and Knox, 1985).

2:2.3 The Movement and Attenuation of Specific Constituents in Soil Treatment Systems

2:2.3.1 Nitrogen

As described in 2:2.1 the treatment process within a septic tank is largely ineffective in reducing the nitrogen content of the influent wastewater. Consequently the effluent discharged to the soil treatment system contains high concentrations of nitrogen, mainly in the form of ammonia. The transport and fate of the nitrogen discharged to the subsurface is largely dependent on the form in which it entered the system and the subsequent biological conversions which may take place. Nitrogen transformations which occur in the subsurface are summarised by Freeze and Cherry (1979) and are presented in Figure 2.2.2 (p27).

The two forms of nitrogen of major concern to the pollution of groundwater systems are ammonium ions (NH_4^+) and nitrates (NO_3^-) (Canter and Knox, 1985). Ammonium ions can be discharged directly from the septic tank to the soil treatment system or they can be formed by ammonification of organic nitrogen in the upper layers of the soil system. Movement of ammonium ions in soil can be retarded by adsorption, cation exchange, incorporation into microbial biomass or release into the atmosphere in gaseous form. Canter and Knox (1985) report that the adsorption process is probably most effective in removing the ions from the percolating effluent. Under anaerobic conditions the NH_4^+ ions are readily adsorbed onto negatively charged particles (Bourma, 1979). It is documented that adsorption of the ammonium ions is essentially complete in the first few centimetres of soil. However, once the adsorptive capacity of the soil is exceeded the ammonia must travel through the saturated soil to find unoccupied sites. Since anaerobic conditions are usually associated with saturated soils the movement of ammonia from soil treatment systems to groundwater can occur if the

effluent is continuously added to a saturated soil profile (Miller and Wolf, 1975 and Canter and Knox, 1985).

A distinct attenuation of ammonia with increasing distance from a soil treatment system located in fine soils was observed in a study by Viraraghavan and Warnock (1976), (cited by Canter and Knox, 1985). The concentration was reduced from 40 mg/l beneath the distribution tiles to 5 mg/l less than 3.0 metres from the system. They also found that ammonia was the dominant form of nitrogen detected beneath the soil treatment system during the periods of operation. However, when loading of the system ceased for a number of weeks the concentration of ammonia decreased and a corresponding increase in the concentration of nitrate was noted.

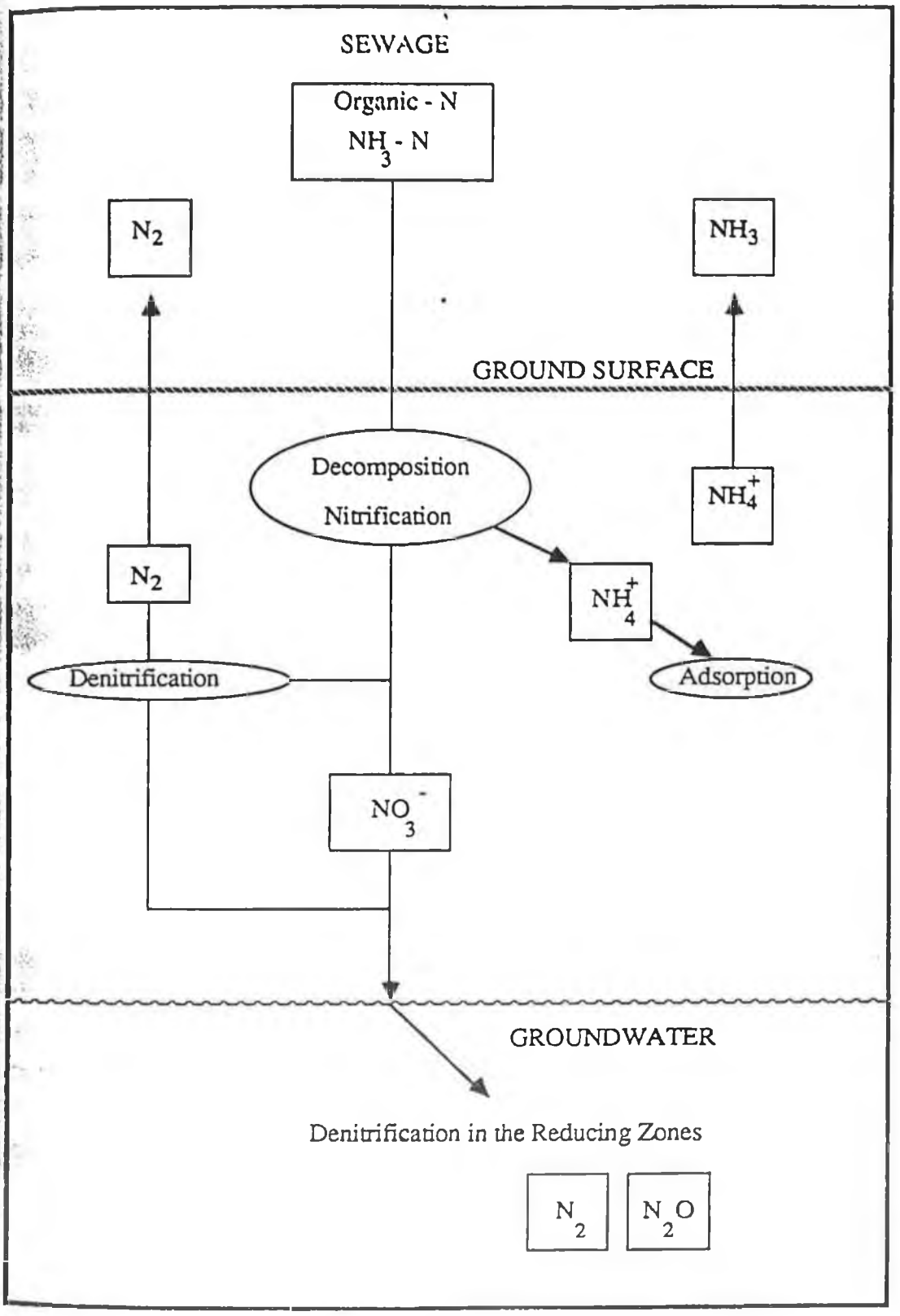
The cation exchange process may also contribute to the retention of ammonia by soil treatment systems (Miller and Wolf, 1975; Bourma, 1979 and Canter and Knox, 1985). Bourma (1979) reports that, as with the adsorption process, the exchange capacity of the soils can be exceeded. In such conditions an equilibrium would be reached between the cations in the soil and the percolating effluent and eventually the effluent moving to groundwater would remain largely unchanged in terms of its cation content. Ammonium nitrogen can be lost by incorporation into microbial or plant tissue but such losses are minimal in septic tank treatment systems. Finally, ammonium can be removed by conversion to NH_3 gas which can be released from the soil. This has been shown to occur at higher pH values (Bitton and Gerba, 1984 and Canter and Knox, 1985).

Nitrate can be present in septic tank effluent. However, as described in 2:2.1, concentrations are very low, with most of the nitrate generated in the upper layers of the soil beneath the distribution tiles by the nitrification process. Nitrification is an aerobic process performed mainly by obligate autotrophic microorganisms, with NO_3 as the main end product (Bourma, 1979). Peavy (1978) reported that the rate of nitrification is dependent on the aeration of the soil which is in turn dependent on soil characteristics, percolation rate, leaching rate, distance to impervious strata and distance to groundwater. Bourma (1979) demonstrated that effluents discharged to sand and gravel soils undergo predominately aerobic reactions with the production of nitrate. However, incomplete nitrification may occur in more clayey soils.

Because nitrate is a negatively charged ion it is not attracted to negatively charged soil colloids and as such is more mobile than the ammonium ion in both saturated and unsaturated soils (Miller and Wolf, 1975; Bourma, 1979; Canter and Knox, 1985 and Lewis et al, 1982). The leaching of nitrate from soil treatment systems and subsequent pollution of groundwater is well documented in the literature (Polkowski and Boyle, 1970; Patterson et al, 1971; Jones and Lee, 1979 and Lewis et al, 1982) and is discussed in more detail in Chapter 3.

Figure 2.2.2

The Form and Fate of Nitrogen in the Subsurface Environment



(After Freeze and Cherry, 1979)

Nitrate can be removed from the system by denitrification. This is the process by which certain bacteria convert nitrates back to nitrites and ultimately to N_2 gas which can be released through the soil. Denitrification occurs in soils containing an abundance of denitrifying bacteria that use NO_3^- as a substitute electron acceptor in the absence of free oxygen. In addition there must be a ready source of carbon present to supply the bacteria with energy (Bitton and Gerba, 1984). An increase in soil moisture content above field capacity can significantly increase the rate of denitrification. Bitton and Gerba (1984) cite research which demonstrates that a 2 to 6% increase in the soil moisture content above field capacity can double the rate of denitrification. Nitrate can also be immobilised by plants and microbial action in a soil treatment system but reduction in the overall concentration is minimal as the nitrogen concentration in effluents greatly exceeds that which can be utilised by these mechanisms (Bourma, 1979).

2:2.3.2 Phosphate

The treatment process within a septic tank is largely ineffective in reducing the phosphorous content of the influent wastewater although much of the organic phosphorous is converted to the soluble orthophosphate form (2:2.1). Phosphate is restricted from moving in the soil by a combination of adsorption and precipitation reactions and can be removed from percolating septic tank effluent at practically all pH values. Phosphate ions can become sorbed onto iron and aluminium minerals in acid soils and onto calcium minerals in neutral or alkaline soils. As the concentration of phosphate within the soil solution increases phosphate precipitates may form. Hydroxyapatite is the stable calcium phosphate precipitate ($Ca_{10}(PO_4)_6(OH)_2$) formed in the pH range found in most septic tank systems (Bourma, 1979). However, under the high phosphate concentrations found in most effluents, dicalcium phosphate and octacalcium phosphate are formed initially with a gradual conversion to hydroxyapatite (Bourma, 1979). Sawhrey and Starr (1977) (cited by Canter and Knox, 1985) carried out a series of laboratory column sorption experiments by passing septic tank effluent (containing 18 mg/l phosphorous) through columns containing 60 centimetres of sandy fill underlain by 30 centimetres of silt loam. The concentrations of phosphate observed in the leachate were initially very low but increased after 20 days (at a daily application rate of 8 centimetres/day). It was concluded that the leaching of phosphorous through soils was minimal until all sorption sites had been occupied, after which the rate increased depending on the percolation rate, application rate and pH of the soil.

A four year study by Jones and Lee (1979) concluded that, although the effluent from the septic tank systems under investigation did migrate to groundwater, there was no evidence of phosphate contamination. They concluded that phosphate

removal in the soil was complete. It has, however, been demonstrated that under some circumstances phosphate transport from a septic tank soil treatment system can occur with subsequent contamination of groundwater (Chapter,3). Ellis (1973) and Miller and Wolf (1975) reported that the sorptive capacity of a soil can be seriously reduced over a period of time as the exchange sites are filled. However, this can be restored if the discharge of effluent is discontinued for a few months. This is possibly due to the crystallisation of adsorbed phosphate into less soluble compounds and to the production of more aluminium and iron oxides by weathering.

2:2.3.3 Fecal Bacteria

The persistence and movement of enteric bacteria in soils is of major importance in determining the suitability of a soil for the disposal of septic tank effluent. Bacteria and most other microorganisms which are present in excreta pass easily through septic tanks and into the soil treatment system (Patterson et al, 1971). There is, however, some dispute about survival of the organisms within the tank but most authors agree that the numbers of fecal bacteria in septic tank effluent are very high (2:2.1). The microorganisms are not capable of self propulsion and are carried along by the liquid in which they are suspended. Several mechanisms combine to remove bacteria from the wastewater as it percolates through the soil. The physical process of straining and the chemical process of adsorption are the most significant of these (Schaub and Sorber, 1977; Canter and Knox, 1985 and Lewis et al, 1982). Additional factors include competition for nutrients and the production of antibiotics by large populations of Actinomycetes, *Pseudomonas* and *Bacillus* in the aerobic layer beneath the clogged zone. Bourma (1979) reports that these antibiotics play an important role in the reduction of the fecal Coliform and fecal Streptococci numbers beneath septic tank distribution lines.

The filtration or straining of fecal bacteria at the wastewater/soil interface is a major limitation to their transport through soils. The extent of filtration depends on the soil type and the nature of the liquid in which the bacteria are suspended and is generally more pronounced in fine - grained soils than in coarser sands and gravels. It has been demonstrated that the removal of bacteria from liquid percolating through a given soil profile is inversely proportional to the particle size of the soil (Gerba et al, 1975).

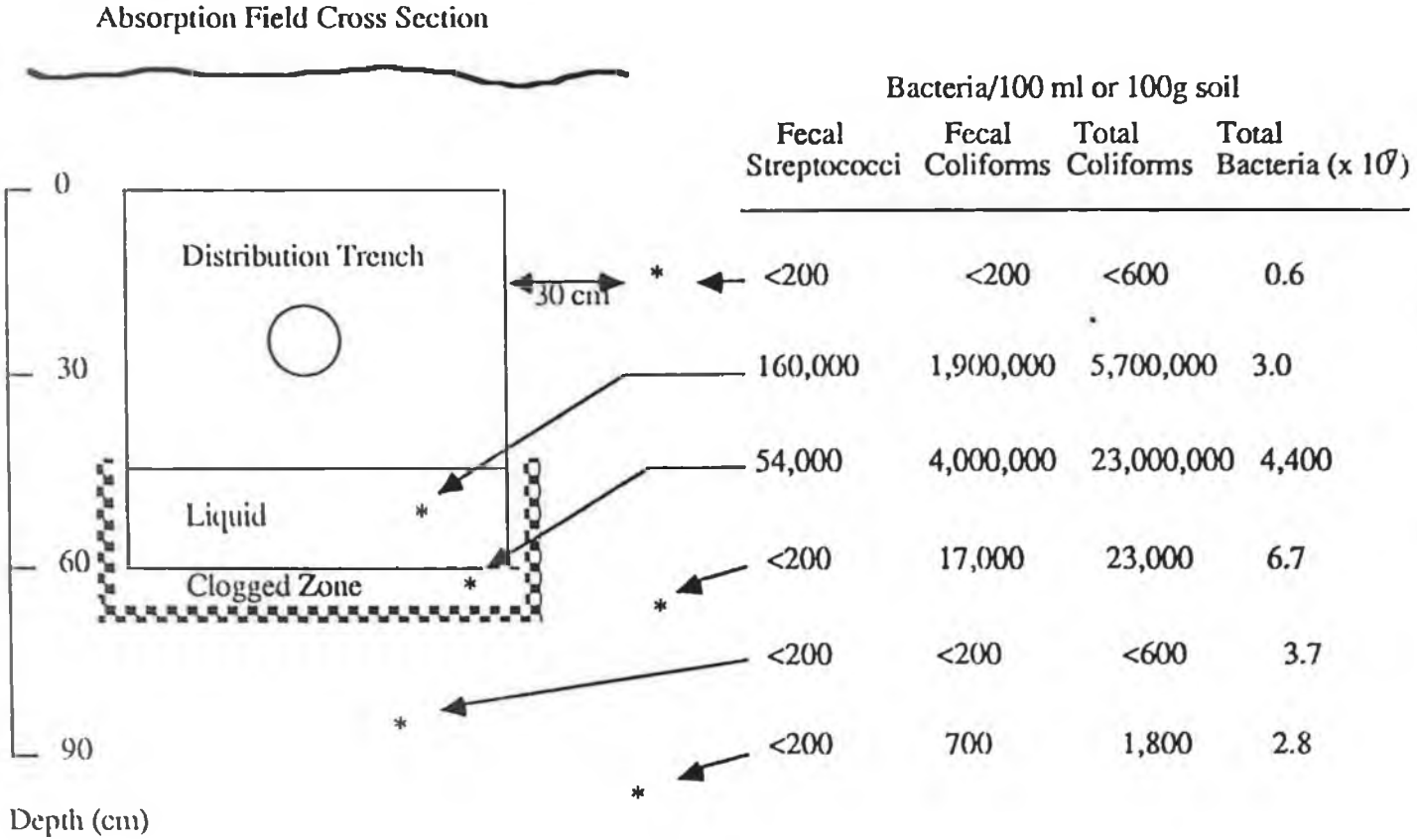
Physical straining occurs when the bacteria are larger than the pore openings in the soil and partial clogging of the soil pores by the suspended solids in the septic tank effluent can increase the efficiency of this process (Peavy, 1978). Straining of bacteria from septic tank effluents is greatest in the clogged zone or biological mat beneath the distribution lines, where the bacteria are removed by a combination of

mechanical straining and sedimentation of bacterial clusters. Ziebell et al (1974) and McCoy and Ziebell (1975) observed that up to 99.9% of the original coliform population were retained over a distance of less than 30 centimetres (Figure 2.2.3, p31).

Adsorption can also restrict the movement of bacteria in soils. Factors which reduce the repulsive forces between two surfaces, such as the presence of cations, allow closer interaction between them enabling adsorption to proceed (Bitton and Gerba, 1984). Thus the C. E. C. of a soil is of great importance. Clay soils with a high C. E. C. will have a higher degree of adsorption of bacteria onto soil surfaces than a sandy soil with a low C. E. C. (Bitton and Gerba, 1984). Peavy (1978) reports that adsorption occurs in soils with a high ionic strength and neutral or slightly acidic pH. Cations such as Ca^{++}, Mg^{++}, Na^{+} and H^{+} in the soil water neutralise and sometimes supersaturate the surface of the bacteria and in doing so enable them to be adsorbed to negatively charged soil particles. However, Schaub and Sorber (1977) conclude that filtration is the dominant process by which bacteria are removed in the soil and that adsorption plays a minor role in bacterial retention in soils. Both filtration and adsorption can be influenced by the rate of effluent disposal from the septic tank system (Bourma, 1979). Rapid movement decreases the contact time between the bacteria and the soil particles and reduces the degree of purification. Viraraghavan (1978) demonstrated that Coliform bacteria, fecal Coliform bacteria and fecal Streptococci migrated through a clay/sand soil to groundwater sources. This movement was attributed to the fluctuating water table which limited travel distance in the unsaturated zone to 0 to 0.15 metres, thereby preventing bacterial removal by filtration and adsorption. Similarily, Hagedorn et al (1978) and Stewart and Reneau (1981) report that bacteria can migrate through a significant thickness (up to ten metres) of saturated soil with subsequent threat to groundwater quality. In a study reported by Patterson et al (1971), no fecal bacteria were isolated below 0.6 metres depth in a soil treatment system consisting of a fine textured clayey soil. However, in more permeable soils bacteria were observed at greater depths. The retention of fecal bacteria in soil treatment systems is complicated by the fact that previously retained bacteria can be mobilised during periods of heavy or prolonged rainfall. This is attributed to the flushing of the bacteria through pores by the physical force of the percolating rainwater and the desorption of previously adsorbed organisms due to a decrease in the ionic strength of the soil solution. The increased bacterial pollution of groundwater after rainfall events is well documented in the literature (Patterson et al, 1971; Hagedorn et al, 1978; Bitton and Gerba, 1984 and Sinton, 1986). Furthermore, bacteria have been demonstrated to migrate at specific depths in zones of higher permeability (i. e. cracks or joints) in the soil profile. This

Figure 2.2.3

Cross Section of an Absorption Field Showing Typical Bacterial Counts at Various Locations



After McCoy and Ziebell (1975)

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can result in bacteria being transported over long distances in a relatively short period of time (Rahe et al, 1978; McCoy and Hagedorn, 1979; Bitton and Gerba, 1984 and Sinton, 1986).

The survival of fecal bacteria in soil systems is an important factor in determining the risk to groundwater of pollution by effluents from septic tank soil treatment systems. Factors known to influence bacterial survival in soils are summarised in Table 2.2.4 (p33). However, the three main factors governing survival are:

(i) Climate

(ii) The nature of the soil

(iii) The nature of the microorganism.

(Bitton and Gerba, 1984)

Climate in turn controls two additional factors, temperature and rainfall. Bacterial survival can be greatly prolonged in a cool environment and at temperatures below 4 ^{o}C they can survive for months or even years (Bitton and Gerba, 1984). At higher temperatures dieoff is rapid and is roughly doubled with every ten degree rise in temperature between 5 and 35 ^{o}C (Reddy et al, 1981). The nature of the soil is critical to the survival of the organisms. Soil properties influence moisture holding capacity, pH and organic matter content, all of which directly control the survival times. Reports of bacterial survival times in soils vary and results are often complicated by the possibility of regrowth. In most situations it appears that two to three months is sufficient for the numbers of enteric bacteria to be reduced to negligible levels once they have entered the soil (Bitton and Gerba, 1984). However, survival times of up to two years have been reported (Patterson et al, 1971).

Table 2.2.4

Factors Affecting the Survival of Enteric Bacteria in Soils

Factor	Remarks
Moisture Content	Longer survival time in moist soils and in times of high rainfall
Moisture Holding Capacity (M. H. C.)	Shorter survival time in soils with low M. H. C. i. e. Sandy soils
Temperature	Longer survival time at low temperatures
pH	Shorter survival time in acid soils (pH 3 - 5)
Sunlight	Shorter survival time at the soil surface
Organic Matter	Increased survival time and possible regrowth if sufficient organic matter is present
Antagonism from Soil Microflora	Increased survival time in sterile soils.

After Bitton and Gerba (1984)

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2:3 Site Characteristics

Two test sites were used in the study, site 1 at Dromahaire, County Leitrim (G 805 315) and site 2 at Cregg, County Sligo (G 653 395). Their geographical location is given in Figure 2.3.1 (p37). Each site consists of a single dwelling with an on - site waste disposal system (i. e. a septic tank and soil treatment system) receiving both household and sewage waste from a population of at least four people. The sites were chosen primarily because of differences in the thickness and nature of the unsaturated zone available for effluent attenuation. Site 1 had a consistently high water table which resulted in an unsaturated zone of less than 0.5 metres beneath the soil treatment system for most of the sampling period. On occasions, particularly in the winter months, the soil treatment system was flooded. Site 2 was underlain by a considerably thicker unsaturated zone which remained well drained throughout the sampling period. Other variables which could affect the movement and attenuation of the effluent constituents in the soil such as construction, design and maintenance of the system, nature of the waste and the proximity of other polluting sources were similar at the two sites.

2.3.1 Site 1 (Dromahaire, County Leitrim)

Site 1 is located in County Leitrim on the outskirts of Dromahaire. This small village is built on the river Bonet, which drains into the eastern end of Lough Gill, and is situated approximately 16 kilometres south east of Sligo town. Most of the village is served by centralised sewage collection but on - site waste disposal systems are used in the outlying areas.

The geology of the area is complex. The Ox mountain fault runs along the southern shores of Lough Gill. To the south of the fault the geology is dominated by a metamorphic inlier composed of schists and gneiss from the Moinian period. This band of metamorphic rock extends over four kilometres to the south where it forms a faulted contact with carboniferous limestone. Dromahaire village is underlain by this limestone, described as Dartry limestone by Oswald (1955).

No information is available on the nature and distribution of soil/overburden material in the area. A survey of the area suggests that the region is overlain by a varying thickness of poorly drained podzolics (1.0 to 3.0 metres). Soil augering at the site revealed greater than 1.2 metres of heavy, saturated soil which was found to contain 10% clay, 25% silt and 65% sand and gravel sized particles. The soil is classified as a sandy loam under the United States Department of Agriculture (U. S. D. A.) classification system.

The test site consists of a single dwelling served by a septic tank and soil treatment system in operation for one year. The septic tank is a prefabricated concrete, one chambered structure. It is not constructed to recognised standards or specifications such as I. I. R. S. S. R. 6 (Anon., 1975) and was not inspected, maintained or desludged during the study period. The treatment system serves a household of two adults and two children and receives both sewage (toilet flushings) and sullage (wash basin and bathtub washings, laundry and kitchen wastes) household wastes. The household does not use a garbage grinder.

The layout of the septic tank treatment system is presented in Figure 2.3.2 (p38). The effluent is channelled into a distribution box and passes into a soil treatment system which is not constructed to recognised standards or specifications. The soil system consists of three distribution trenches (4.0 x 0.75 x 0.5 metres), a distance of 1.5 metres apart. The bottom of each distribution trench was lined to a depth of approximately 20 centimetres with coarse gravels. Three lengths of five centimetre diameter plastic drainage pipes were placed in the trenches, covered by a small amount of gravel and backfilled to ground level with indigenous soil material. The inverts of the distribution pipes are 0.3 metres below ground surface. No venting or maintenance/ observation ports were installed in the distribution field. A levelling survey showed that the ground surface at the site slopes gently away from the soil treatment system. The density of septic tanks within a 450 metres radius of the site is eight and the nearest polluting source (domestic or agricultural) is greater than 100 metres away.

2.3.2 Site 2 (Cregg, County Sligo)

Site 2 is located in the townland of Cregg on the northern side of Sligo bay, 4.5 kilometres north west of Sligo town. The Cregg/Ballincar area has witnessed a marked increase in residential development in recent years, particularly along the L16 link road. The area is not served by a municipal sewerage system and consequently there has been a significant increase in the number of septic tank systems.

The geology is dominated by a metamorphic inlier. The rocks, composed mainly of metasedimentary schists and are described as Moinian in age (Max, 1984). No information is available on the nature and distribution of soil/overburden material in the region. An area survey suggests that the region is overlain by a varying thickness of glacial till (1.0 to 5.0 metres), although there are extensive rock outcrops to the north east of the test site. Soil augering at the site revealed a significant thickness (greater than 1.2 metres) of unsaturated soil which was found to contain 8% clay, 30% silt and 62% sand and gravel sized particles (the thickness of the material was demonstrated to be over 2.5 metres in subsequent investigations at the site, Chapter 3).

The soil is described as a sandy loam under the U. S. D. A. classification system.

The test site consists of a four year old dwelling served by a septic tank and soil treatment system which has been in operation for 1.5 years. The septic tank is constructed in - situ using concrete block (3.0 x 1.6 x 1.5 metres) and contains no baffle wall. It does not comply with standard specifications such as I. I. R. S. S. R. 6 (Anon., 1975) and was not inspected, maintained or desludged during the study period. It is also worth noting that there was no scum present on the liquid surface in the tank. The treatment system serves a household of four adults and receives both sewage (toilet flushings) and sullage (wash basin and bathtub washings, laundry and kitchen wastes) household wastes. The household does not use a garbage grinder.

A schematic layout of the septic tank treatment system is presented in Figure 2.3.3 (p39). The effluent is channelled into a distribution field via a distribution box and passes into three trenches (5.0 x 1.0 x 0.6 metres), a distance of 2.0 metres apart. The field is not constructed to recognised specifications. The bottom of each trench was lined to a depth of 20 centimetres with coarse gravels and a perforated 5.5 centimetre PVC plastic pipe was placed in the trench with the invert 0.40 metres below ground surface. Each pipe was then covered by a small amount of gravel and the trench was backfilled with indigenous soil material. No vent or maintenance ports were installed. A levelling survey showed that the ground surface at the site slopes gently away from the soil treatment system. The density of septic tanks within a 450 metres radius of the site is 15 and the nearest polluting source (domestic or agricultural) is greater than 100 metres away.

The site was also used in a study on the contamination of groundwater sources by septic tank systems. The results of this investigation are presented in Chapter 3 where a more detailed account of the site geology and hydrogeology is also given.

Figure 2.3.1

Location Map Showing the Two Sampling Sites

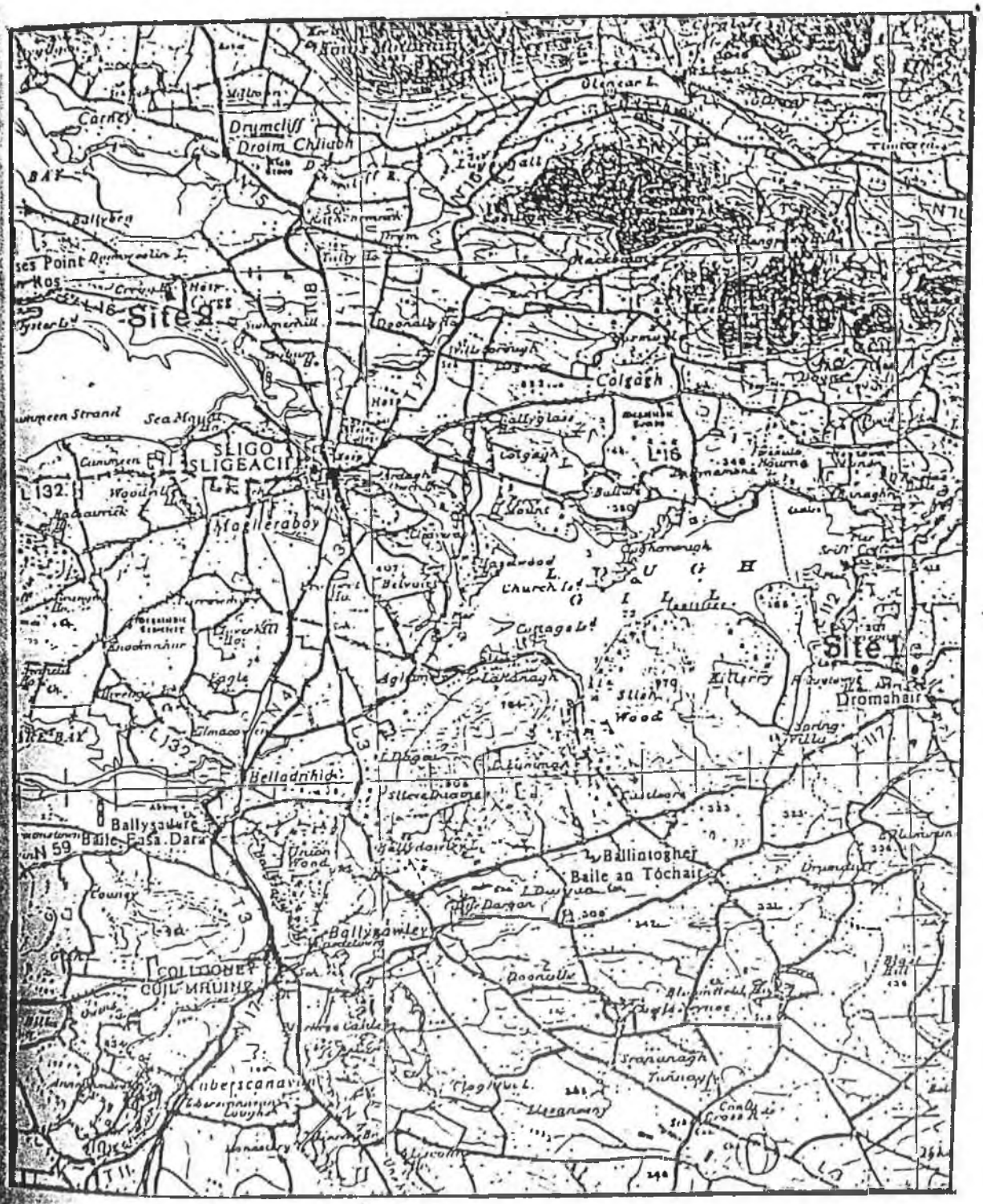

Figure 2.3.2

Schematic Representation of the Soil Treatment System at Site 1(Dromahaire)

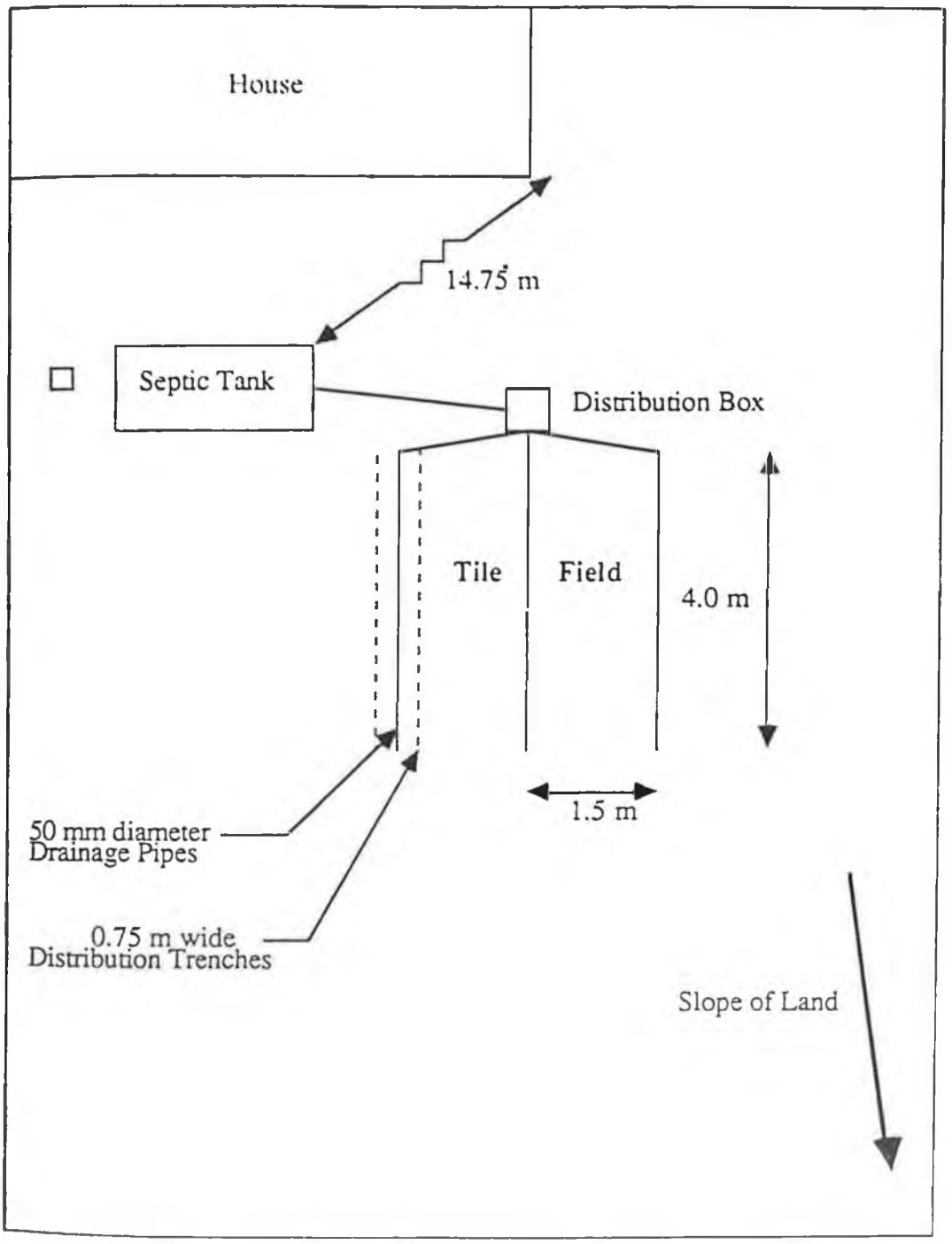

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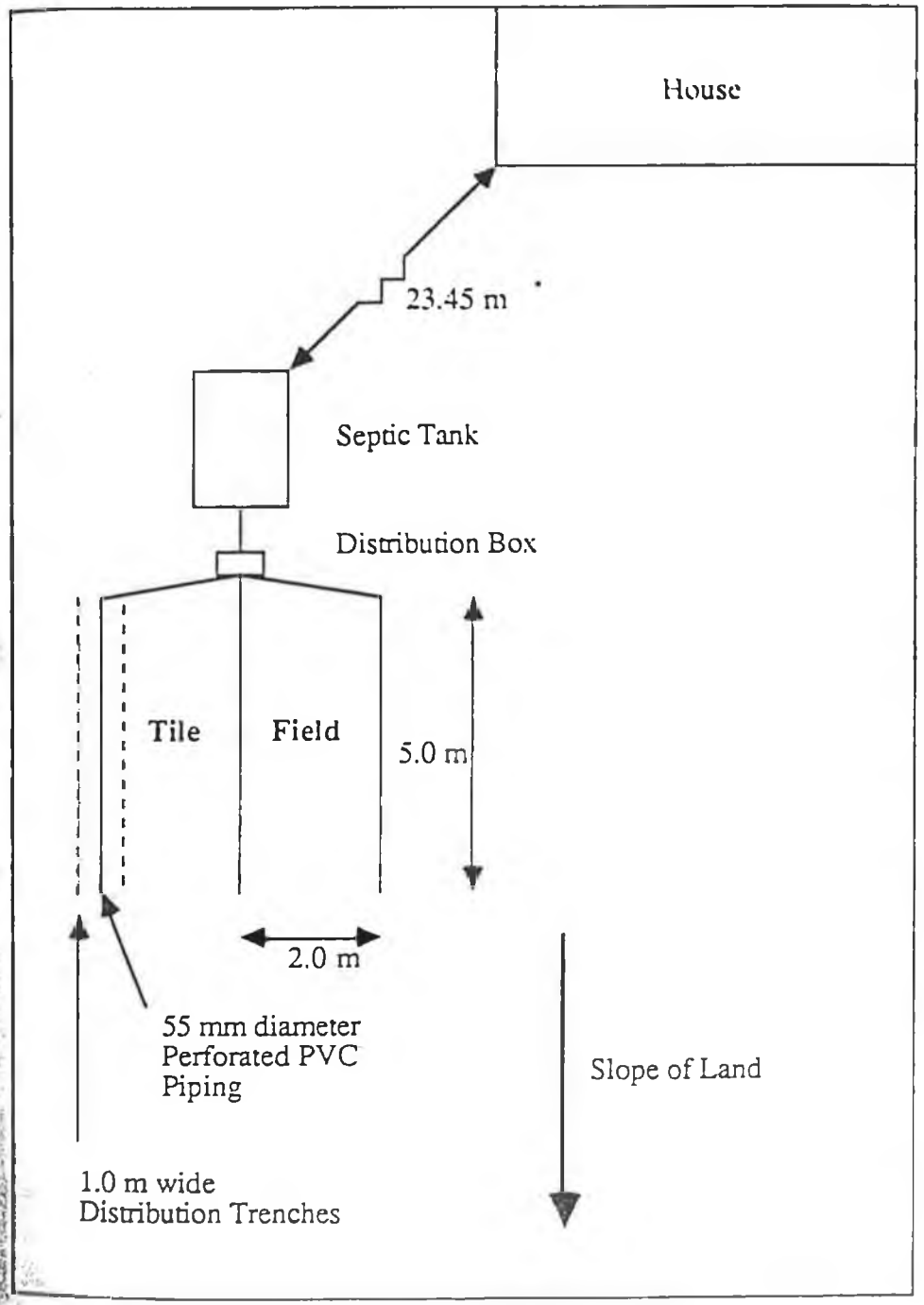
Figure 2.3.3

Schematic Representation of the Soil Treatment System at Site 2 (Cregg)



2:4 Materials and Methods

2.4.1 Sampling

(i) Effluent

The septic tank effluent at both sites was sampled at two monthly intervals from April 1987 to June 1988 (a total of eight sets of samples). The samples were taken directly from the tanks at the outlet T - pipes. Samples for hydrochemical analysis were collected in clean acid washed one litre polypropylene sampling bottles. Microbiological samples were taken in sterile 250 millilitre (wide neck) glass sampling bottles. On all sampling occasions the tank was inspected for the development of surface scums and for blockages in the tank inlet and outlet pipes or the soil distribution box.

(ii) Soil

All soil samples were taken using Eiljkelkamp heterogeneous soil hand augering equipment. A preliminary set of samples was taken at both sites to determine the physical and chemical properties of the material. The soil was sampled at 20 centimetres intervals to a depth of 1.2 metres. To ensure that samples were representative of soil conditions over the entire site, three sets were taken at different locations around the soil treatment system. All samples were transferred to polythene bags and transported to the laboratory where they were refrigerated until analysis.

Regular monitoring of the soil was carried out at bimonthly intervals from April 1987 to June 1988 to assess the movement and attenuation of effluent pollutants in the soil treatment systems. A total of ten sampling stations were monitored at each of the sites, the locations of which are shown in Figures 2.4.1 (p43) and 2.4.2 (p44). Two samples were taken at each station. Within the soil treatment system samples were taken a lateral distance of 20 centimetres from the distribution trench and at depths of 20 (sample A) and 50 centimetres (sample B) below the invert of the distribution pipes (Figure 2.4.3, p45). Samples downgradient of the system were taken at depths of 50 (sample A) and 80 centimetres (sample B) at site 1 and 60 (sample A) and 90 centimetres (sample B) at site 2.

Two samples were taken at each depth, one for microbiological analysis and one for nutrient analysis. Both samples were removed in a similar fashion, using a soil auger. A sample of approximately 100 g was taken for nutrient analysis. Aseptic precautions were taken with the sample for microbiological analysis. Once the required depth was reached the auger head was removed and sterilised by flaming with 100% ethanol. The auger was then carefully replaced and a soil sample of approximately

50 g was taken. Both samples were transferred to plastic bags (U. V. sterilised for microbiological samples) for transport to the laboratory.

(iii) Soil Water

(a) Piezometer Installation

A total of six shallow piezometers were installed at site 1 in order to assess the migration of effluent constituents from the soil distribution system. Two piezometers were installed within the soil distribution field, three more were placed between 2.5 and 5.0 metres downgradient of the system and a control piezometer was installed upgradient of the field (Figure 2.4.4, p48).

Eiljkelkamp augering equipment was used to excavate the piezometer bores to a depth of 1.0 metre beneath the soil water table. Standard six centimetre perforated drainage piping was then used as casing to facilitate free movement of soil water. The ground surface around each piezometer was sealed with a concrete apron to prevent contamination of the soil water by surface pollutants. The opening was then covered using removable polypropylene caps.

(b) Sampling

Water samples were taken on three occasions from all six piezometers using a modification of the method described by Caldwell (1938). Approximately one litre of sample was removed from each piezometer using a hand suction pump and was filled directly into sterile two litre glass bottles. The samples were then transferred to a polystyrene 'cool box' for transport to the laboratory where they were refrigerated until analysed.

2.4.2 Analysis

(i) Effluent

Table 2.4.1 (p46) summarises the chemical parameters analysed for and the methods used. Electrical conductivity and pH were measured on site. On return to the laboratory, half of the one litre sample was filtered through a 0.47 µm membrane filter and placed in a second acid washed one litre polypropylene bottle. Analysis for ammonia, nitrate, B. O. D. and C. O. D. was carried out on the unfiltered samples. The filtrate was used for all other analysis.

Microbiological analysis was carried out within six hours of sampling. Total, fecal Coliform and fecal Streptococci bacteria were analysed for using a pour plate technique with serial dilution in 1/4 strength Ringer's solution. The methods used are summarised in Table 2.4.2 (p46).

(ii) Soil

Tables 2.4.3 and 2.4.4 (p47) present the soil parameters analysed for and the methods used. All analyses were carried out in accordance with British Standards B. S. 1877 (Anon., 1975), ' Chemical Analysis of Agricultural Materials ' (Byrne, 1979) and ' Methods of Soil Analysis ' (Black et al, 1965). Soil permeability was determined using the I. I. R. S. S. R. 6 (Anon., 1975) percolation test. All extractions and analyses were completed within 24 hours of sampling. Moisture content was calculated for all samples analysed so that results could be expressed as mg/kg of dry soil.

The fecal bacteria were extracted from the soil samples using a modification of the method described by Black et al (1965). A five gram sample was aseptically transferred to 45 mls of sterile Ringer's solution in a bottle containing a number of glass beads. The bacterial cells were then extracted from the soil colloids by shaking on a mechanical shaker for two minutes. The extracted organisms were enumerated and identified using recognised standard methods (Figure 2.4.4, p48). All results are reported in colony forming units per 100 grams of soil (c.f.u.' s/100g)

(iii) Soil Water

Table 2.4.5 (p49) summarises the methods used for the chemical and biological analysis of the soil water samples. On return to the laboratory, microbiological analysis was carried out immediately. Total and fecal Coliform bacteria were enumerated using a pour plate technique with serial dilution in 1/4 strength Ringer's solution. Half of the remaining sample was filtered through a 0.47 μm membrane filter and placed in a clean, acid washed one litre polypropylene bottle. Analysis for ammonia and nitrate was carried out on the unfiltered samples and the filtrate was used for phosphate analysis.

Figure 2.4.1

Schematic Representation of the Soil Treatment System at Site 1 (Dromahaire) Showing the Location of the Soil Sampling Stations and the Water Sampling Piezometers

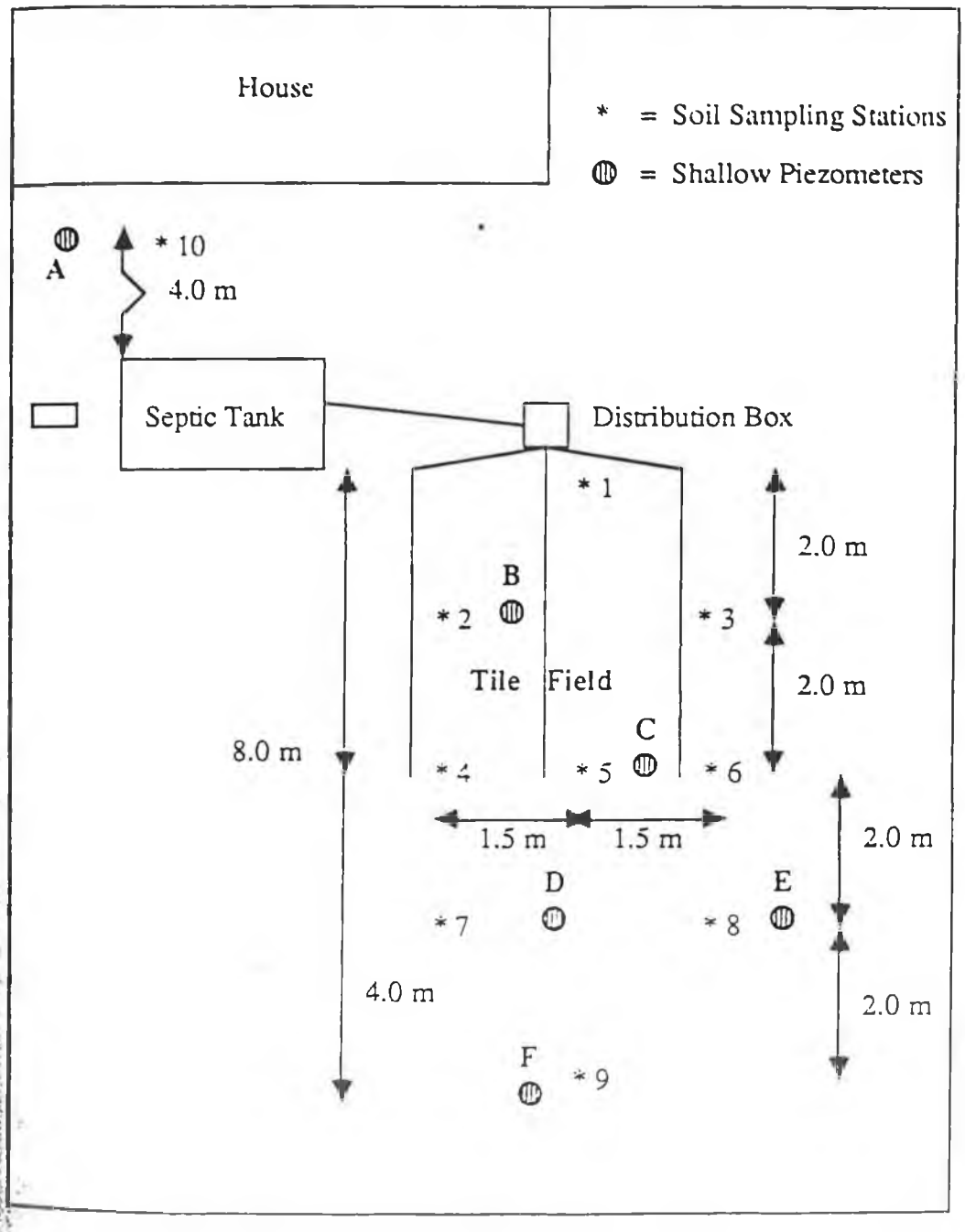


Figure 2.4.2

Schematic Representation of the Soil Treatment System at Site 2 (Cregg) Showing The Location of the Soil Sampling Stations

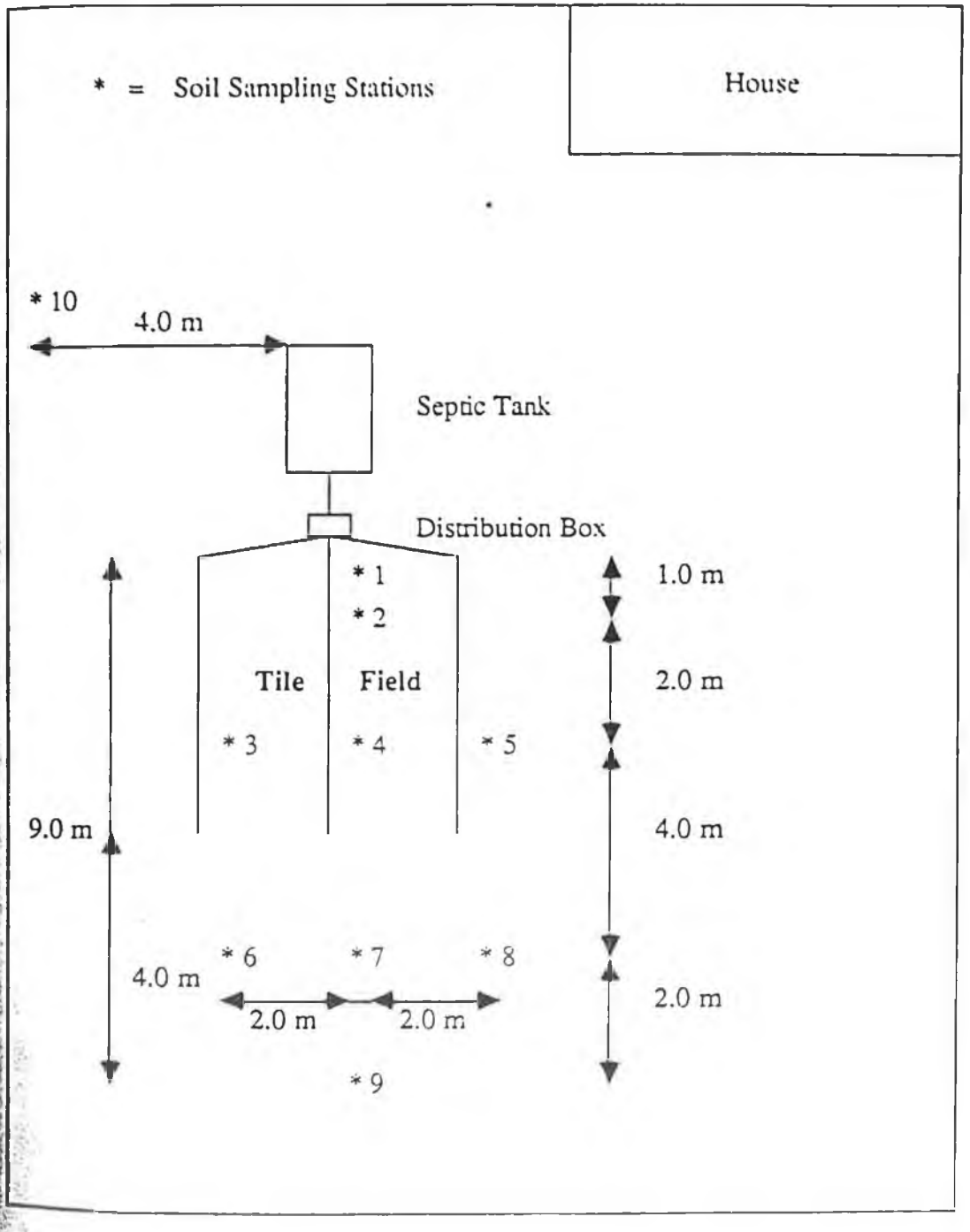


Figure 2.4.3

A Cross Section of a Distribution Trench Showing The Location of the Soil Samples in Relation to the Distribution Pipe Invert

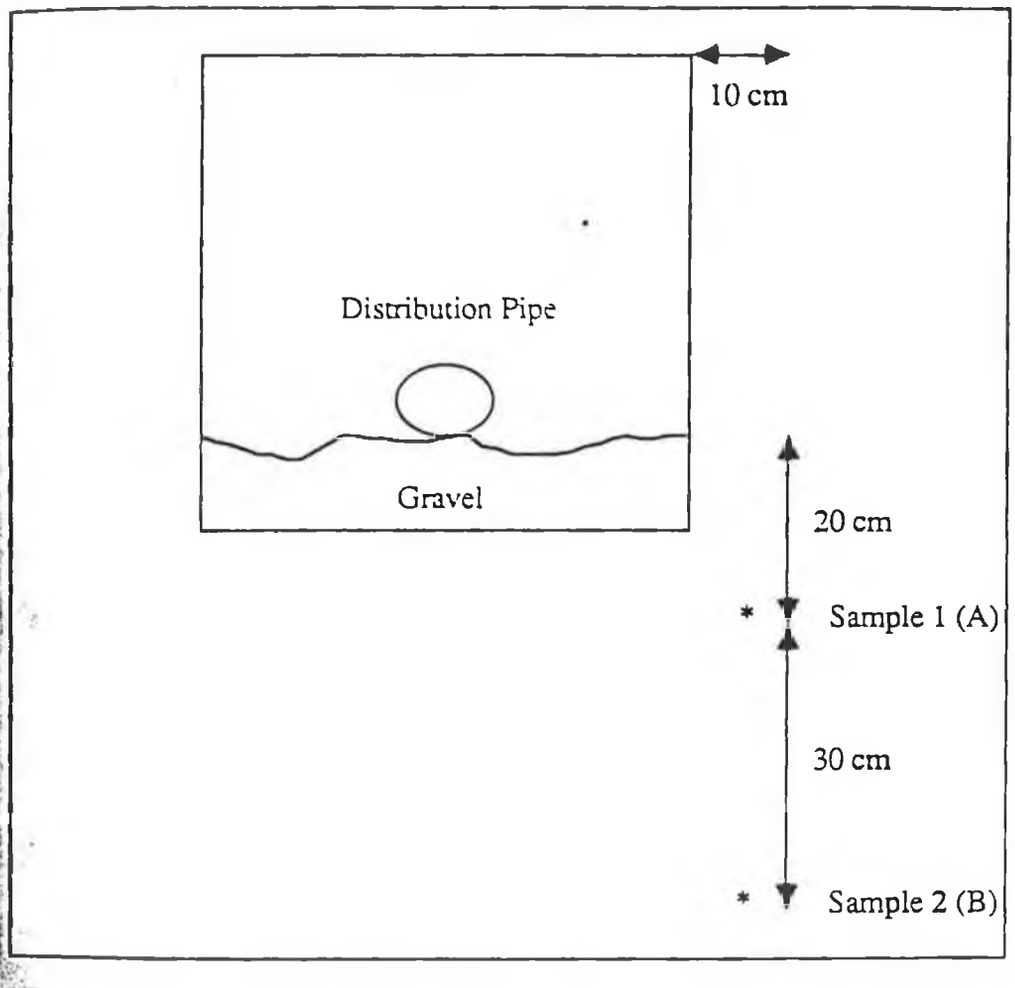


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Table 2.4.1

Methods Used in the Hydrochemical Analysis of the Septic Tank Effluent Samples

Parameter	Method
pH	Electrometric
Conductivity	Electrometric
B. O. D.	Electrometric
C. O. D.	C. O. D. Reactor Method
Suspended Solids	Gravimetric
Nitrate (NO_3 - N)	Markham Still Distillation
Ammonia (NH_3 - N)	Markham Still Distillation
Phosphate (PO_4 - P)	Visible Spectrophotometry
Sulphate (SO_4)	Gravimetric (ignition of residue)
Chloride (Cl^-)	Argentometric Titration
Sodium (Na^+)	Flame Photometry
Potassium (K^+)	Flame Photometry

Table 2.4.2

Methods Used in the Microbiological Analysis of the Septic Tank Effluent Samples

Parameter	Method
Total Coliform Bacteria	Pour plate method with serial dilution in 1/4 strength Ringer's solution and isolation on Violet Red Bile Agar (V. R. B. A. - Oxoid) growth media
Fecal Coliform Bacteria	Pour plate method with serial dilution in 1/4 strength Ringer's solution and isolation on Violet Red Bile Agar (V. R. B. A. - Oxoid) growth media.
Fecal Streptococci Bacteria	Pour plate method with serial dilution in 1/4 strength Ringer's solution and isolation on KF Streptococcus (Oxoid) growth media

Table 2.4.3

Methods Used in the Assessment of the Physical and Chemical Properties of the Soil/Overburden Material

Parameter	Method
pH	Electrometric
Moisture Content	Loss on Ignition
Soil Texture	Sieving and Sedimentation
Porosity	Bulk Density/Particle Density
Organic Matter Content	Loss on Ignition
Organic Carbon Content	Walkley - Black Titration
Cation Exchange Capacity	Sodium Saturation Method
Permeability	I. I. R. S. Percolation Test

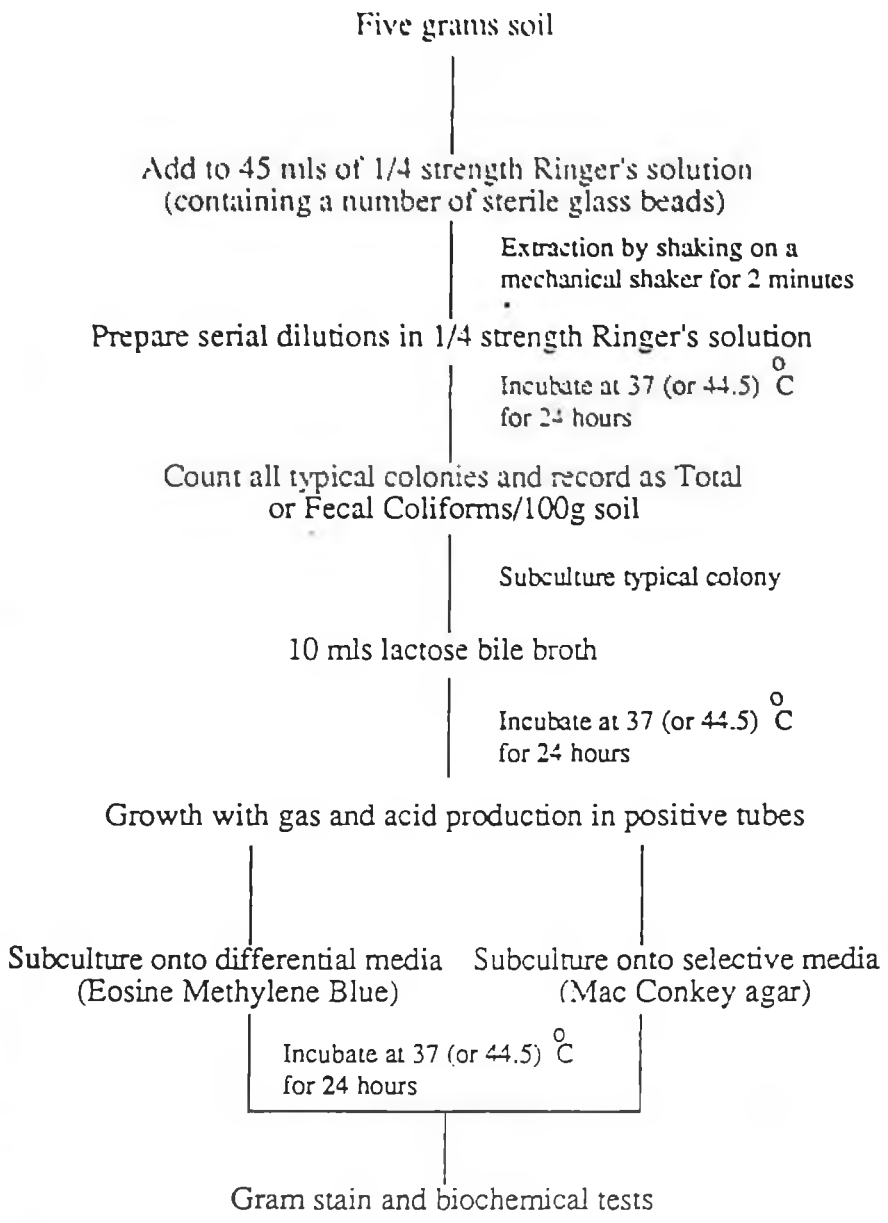
Table 2.4.4

Methods Used in the Chemical Analysis of Soil Samples

Parameter	Method
Nitrate nitrogen	Markham Still Distillation (extraction in 2 N KCl)
Ammonia nitrogen	Markham Still Distillation (extraction in 2 N KCl)
Ortho phosphate	' Ag ' Reagent Method (extraction in Morgan's solution)

Figure 2.4.4

Isolation and Identification of Coliform Bacteria from the Soil Samples



Note:
Incubate at 37 °C for Total Coliform Bacteria
Incubate at 44.5 °C for Fecal Coliform Bacteria

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Table 2.4.5

Methods Used in the Chemical and Biological Analysis of the Piezometer Soil Water Samples

Parameter	Method
Nitrate (NO_3 - N)	Markham Still Distillation
Ammonia (NH_3 - N)	Markham Still Distillation
Phosphate ($P0_4$ - P)	Visible Spectrophotometry
Total and fecal Coliform bacteria (c. f. u. ' s/100ml)	Pour plate method with serial dilution in 1/4 strength Ringer's solution and isolation on Violet Red Bile Agar (V. R. B. A. - Oxoid) growth media

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2:4.3 Bacterial Survival Experiments

This section details the methods used in assessing the survival of *Escherichia coli* bacteria in a range of soil types. It was described in 2:2 that the movement of fecal bacteria from septic tank treatment systems and the resulting risk of groundwater pollution is largely dependent on the fate of the organisms once released into the soil i. e. their retention and survival. The inconsistency of literature reports on the survival of enteric bacteria in the soil/overburden necessitated the following soil survival studies.

The reported survival times of enteric bacteria in soils vary widely and are often complicated by the possibility of regrowth in the test systems. However, in most situations it appears that two to three months is sufficient for the reduction of enteric bacteria to negligible levels once they have been introduced into soil (Bitton and Gerba, 1984). The large variation in bacterial survival times reported may be due to different experimental procedures and conditions used by the various researchers. Much of the research to date has been laboratory based (Kibbey et al, 1978 and Rhodes and Kator, 1988) using small leaching containers or shaking flasks under predetermined laboratory environmental conditions. No previous investigation has attempted to assess the survival of enteric bacteria in a simulated septic tank soil treatment system using a range of soil types.

In this study a set of bacterial leaching containers was set up to assess the survival of *Escherichia coli* bacteria in a range of soil types and to determine realistic survival times (T 90 and T 99) for fecal bacteria discharged to septic tank soil treatment systems. A number of special features were incorporated into the experimental setup in order to overcome some of the inherent errors associated with earlier laboratory based studies:

(i) Large (1.5 m^3) wide diameter leaching containers were used to minimise the boundary effects and soil disturbance interferences associated with smaller diameter leaching tubes

(ii) The leaching containers were located outdoors and were thus exposed to normal environmental conditions

(iii) The bacteria were introduced to the containers in subsurface innoculation tubes similar to the distribution lines used in a soil treatment system. The bacteria were also added to sterilised septic tank effluent before addition to the soils in order to simulate effluent disposal conditions

(iv) Three markedly different soil types were used in the study in order to make the results more widely applicable.

A process which may be significant in removing effluent bacteria in a soil percolation system is the anti - microbial nature of antibiotics produced by actinomycetes and some bacteria in the clogged zone. It was not possible to account for this in the experimental setup. The following section describes the experimental setup, the sampling methods and the analytical and statistical procedures used in the assessment of the survival of the organisms in a range of soil types.

2:4.3.1 Experimental Procedure

Four plastic barrels of 1.5 m^3 volume (Plate 2.4., p54) were filled with three different soils types (peat, loam and sand). The fourth barrel was also filled with the loam soil and was used as a control for the experiment. The barrels were filled to a fixed depth and a sod cover representative of the vegetation indigenous to each soil type was placed on top. Figure 2.4.5 (p53) gives a diagrammatic representation of one of the leaching containers. An innoculation tube was placed in the barrel through which the effluent containing the bacterial population was passed into the soils. The innoculation tubes were designed to closely resemble the perforated pipes in a septic tank distribution field. The bottom of the barrels was also perforated and lined with a thin layer of coarse gravel to allow rainwater to pass through and prevent saturation and ponding in the soils. Soil thermometers and tensiometers were installed in two of the barrels to monitor the soil temperature and moisture status throughout the test period.

Samples of the three soil types were retained for laboratory analysis of their physical and chemical properties. The methods used in the analysis of the physical and chemical properties of the soils are similar to those described in 2:4.2. The soil in the barrels was then allowed to settle and compact for a period of four months. When the soil had compacted sufficiently the barrels were irrigated to saturation with sterile water and then allowed to drain for 24 hours, bringing the soils to field capacity. The barrels were then immediately covered and no further moisture was allowed in during the test period.

A concentrated culture of *Escherichia coli* was obtained using a Multigen fermenter. A working solution (10^6 *E. coli* /ml) was then prepared in sterile septic tank effluent. A 1.5 litre aliquot of this solution was added to each of the three soils on the 19th of November 1988 at 1400 hours and the first samples were taken three hours later. Sampling continued for a period of 44 days. The soil samples were removed from the barrels through the sampling portholes (Figure 2.4.5, p53) using a small hand auger. They were immediately transferred to U. V. sterilised plastic sampling bags. The auger was sterilised between samples by swabbing and flaming with ethanol. The amount of soil removed was minimised in order to reduce the effects of soil disturbance. On each sampling occasion the temperature and moisture status of the

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soil was also noted. The *Escherichia coli* bacteria in the soil samples were extracted and enumerated using the procedure described in 2:4.2.

2:4.3.2 Statistical Analysis

The relationship between the die - off of the *Escherichia coli* bacteria with time was statistically analysed using Pearson's correlation. It was assumed that the number of fecal bacteria isolated from the soil samples is a normally distributed variable. Scatter plots of the log of bacterial numbers (c. f. u.' s/g) isolated from each of the soil containers against time (days) were prepared. These plots appeared to follow an exponential decay pattern. When an exponential curve was fitted to the plots a straight line was obtained from which Pearson's coefficients of determination (r^2) and correlation (r) were estimated. In order to test the normality assumption, a plot of the residuals and the residuals against expected ' y ' values was prepared for each of the above straight line graphs. The model is satisfied if the residuals are randomly distributed about zero with no obvious dependencies. The model would be rejected in the event of a spurious residual distribution.

The significance of the correlation coefficient (r) was tested using the following test statistic:

$$t = r\sqrt{\frac{n-2}{1-r^2}} \sim t_{n-2} \qquad \text{(at P = 0.01 and 0.001)}$$

Ho : $\partial = 0$ non linear relationship

H_1 : $\partial \neq 0$ linear relationship

(where r estimates ∂)

Figure 2.4.5

Cross Section Through a Leaching Barrel

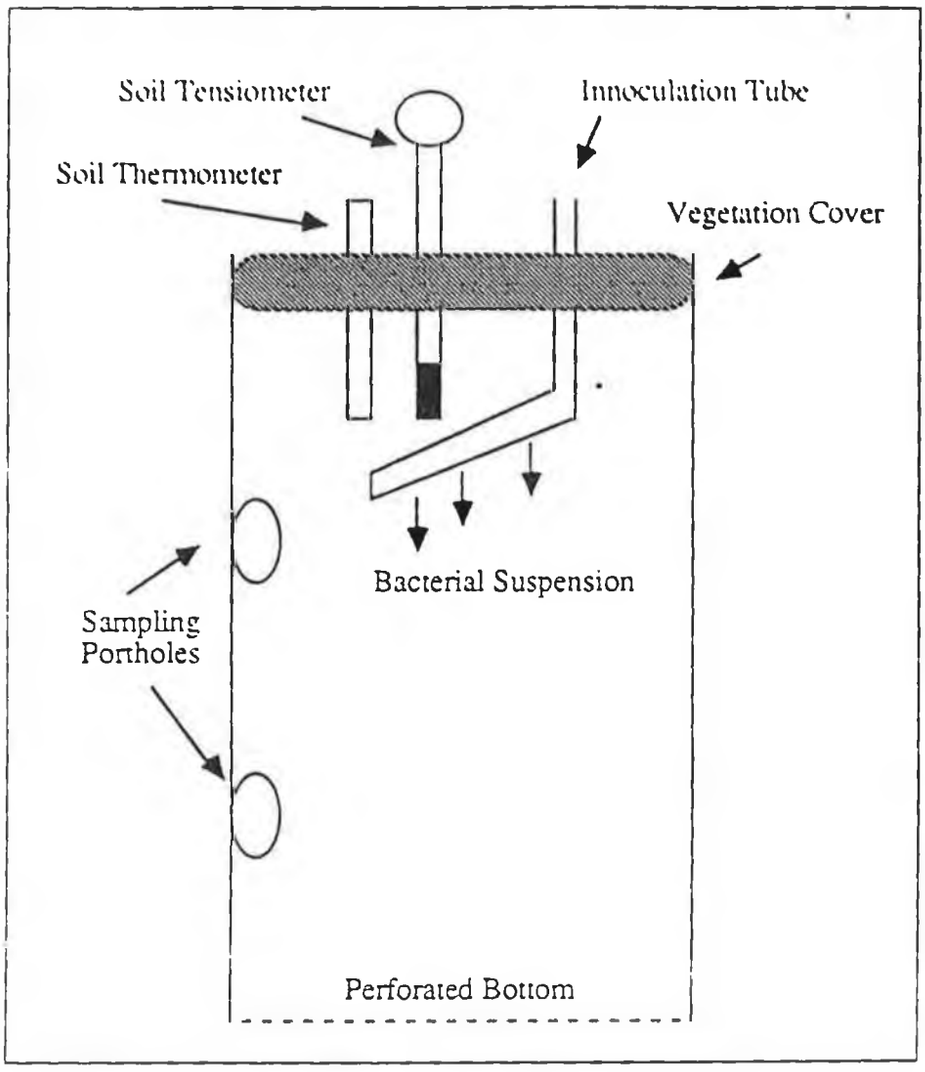

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Plate 2.4.1

The Four Leaching Containers Used in the Bacterial Survival Experiments

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2:5 Results

The results of the investigation are presented in five sections as follows:

(i) Effluent Analysis

Summary results of the chemical and microbiological analyses of septic tank effluent at the two sites are presented in Tables 2.5.1 (p57) and 2.5.2 (p58)

(ii) Soil Analysis

The physical and chemical properties of the soil/overburden material at the two test sites are presented in this section. The results are average values taken from the analysis of three sets of samples from various locations within the test sites. The results are presented in Tables 2.5.3 to 2.5.6 (p59 to 60), as follows:

Tables 2.5.3 and 2.5.4 (Site 1, Dromahaire)

Tables 2.5.5 to 2.5.6 (Site 2, Cregg)

(iii) The Movement and Attenuation of Septic Tank Effluent Constituents from the Soil Treatment Systems

Summary results of the chemical and microbiological analysis of the soil samples on the eight sampling occasions (April 1987 to June 1988) are presented in Tables 2.5.7 to 2.5.11 (p61 to 77). The results are also graphically presented in Figures 2.5.1 to 2.5.40 (p62 to 85), as follows:

Figures 2.5.1 to 2.5.8	Ammonium
Figures 2.5.9 to 2.5.16	Nitrate
Figures 2.5.17 to 2.5.24	Phosphate
Figures 2.5.25 to 2.5.32	Total Coliform bacteria
Figures 2.5.33 to 2.5.40	Fecal Coliform bacteria

The monthly rainfall (mm) during the sampling period is plotted in Figure 2.5.41 (p81) while the total rainfall (mm) for the five day period preceeding sampling is presented in Figure 2.5.42 (p81).

(iv) Analysis of the Sampling Piezometers at Site 1

The results of the analysis of the piezometers at Site 1 (Dromahaire) are presented in Table 2.5.12 (p82).

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(v) The Survival of *Escherichia coli* in a Range of Soil Types

The results of the physical and chemical analysis of the three soils used in the study are presented in Tables 2.5.13 and 2.5.14 (p83). *Escherichia coli* bacteria numbers isolated from the soil samples on the various sampling occasions after the addition of the organisms are given in Table 2.5.15 (p84). The die - off of the organisms in the soils is presented graphically in Figures 2.5.43 to 2.5.45 (p85). The results of statistical calculations are presented in Table 2.5.16 (p86).

Table 2.5.1

Summary Results of the Chemical and Microbiological Analysis of the Septic Tank Effluent at Site 1 (Dromahaire)

Parameter	No. of Samples	Max.	Min.	Mean	Std. Deviat.	Co - eff. of Variation
B. O. D. (mg/l)	8	562.0	140.0	346.0	143.0	41.3
C. O. D. (mg/l)	8	1050.0	316.0	596.0	258.0	43.4
S. S. (mg/l)	8	280.0	68.0	160.0	82.1	51.3
pH (pH units)	8	7.60	7.19	7.42	0.15	2.10
Cond. (µS/cm)	8	1110.0	881.0	953.0	79.4	8.3
NO_3- N (mg/l)	8	1.4	0.0	0.4	0.5	117.0
NH_3 - N (mg/l)	8	36.1	13.5	27.8	7.50	27.1
PO_4- P (mg/l)	8	43.6	13.9	28.7	9.8	34.2
Potassium (mg/l)	8	87.0	17.0	42.2	21.4	50.6
Sodium (mg/l)	8	180.0	101.0	140.0	28.7	20.5
K/Na Ratio	8	0.57	0.15	0.31	0.16	51.30
Chloride (mg/l)	8	97.0	39.0	64.3	18.2	28.4
Sulphate (mg/l)	8	70.0	31.1	48.4	12.5	25.7
Total Coliforms (c. f. u. 's/100ml)	8	6.5×10^6	1.0×10^6	3.0×10^6	2.0×10^6	65.7
Fecal Coliforms (c. f. u. 's/100ml)	8	3.2×10^6	1.0×10^5	1.3×10^6	1.2×10^6	91.7
Fecal Streptococci (c. f. u. 's/100ml)	8	5.3×10^5	1.2×10^3	2.0×10^5	2.2×10^5	109

Table 2.5.2

Summary Results of the Chemical and Microbiological Analysis of the Septic Tank Effluent at Site 2 (Cregg)

Parameter	No. of Samples	Max.	Min.	Mean	Std. Deviat.	Co - eff. of Variation
B. O. D. (mg/l)	8	822.0	342.0	564.0	170.0	30.1
C. O. D. (mg/l)	8	1800.0	509.0	1050.0	488.0	46.4
S. S. (mg/l)	8	708.0	91.0	265.0	198.0	74.6
pH (pH units)	8	8.50	6.27	7.55	0.69	9.19
Cond. (µS/cm)	8	1670.0	968.0	1100.0	235.0	21.4
NO_3- N (mg/l)	8	1.1	0.0	0.4	0.5	113.0
NH_3- N (mg/l)	8	71.8	22.9	43.7	14.5	33.1
PO_4 - P (mg/l)	8	63.2	31.9	49.8	11.2	22.5
Potassium (mg/l)	8	89.0	22.0	35.1	22.4	63.7
Sodium (mg/l)	8	112.0	89.0	98.8	8.4	8.5
K/Na Ratio	8	0.93	0.22	0.36	0.24	66.50
Chloride (mg/l)	8	117.0	32.0	64.4	25.2	39.1
Sulphate (mg/l)	8	68.9	27.0	46.1	12.9	27.9
Total Coliforms (c. f. u. 's/100ml)	8	4.0×10^7	8.9×10^5	1.6×10^7	1.5×10^7	93.0
Fecal Coliforms (c. f. u. 's/100ml)	8	1.6×10^7	3.6×10^5	5.1×10^6	6.0×10^6	117
Fecal Streptococci (c. f. u. 's/100ml)	8	1.8×10^5	1.8×10^3	8.8×10^4	6.9×10^4	78.0

Table 2.5.3

Particle Size Distribution of the Soil/Overburden Material at Site 1(Dromahaire)

PARTICLE SIZE	Sample Depth (cm)						
	20	40	60	80	100	120	Mean
	Percentage of Total						
Clay (< 0.002 mm)	10	11	12	10	9	8	10
Silt (0.002 - 0.06 mm)	25	21	26	27	28	24	25
Sand (0.06 - 2.0 mm)	57	58	54	51	55	57	55
Gravel (> 2.0 mm)	8	10	8	12	8	11	10

Table 2.5.4

Physical and Chemical Properties of the Soil/Overburden Material at Site 1 (Dromahaire)

PARAMETERS	Sample Depth (cm)						
	20	40	60	80	100	120	Mean
pH (pH units)	7.21	7.30	7.17	7.26	6.88	7.11	7.15
Porosity (%)	42.0	39.0	38.0	39.0	37.0	31.0	38.0
Organic Matter (%)	20.3	12.6	11.0	10.3	10.8	10.2	12.5
Organic Carbon (%)	7.1	6.3	5.9	5.6	5.7	5.1	6.0
C. E. C. (meq/100g)	36.0	35.0	38.0	32.0	34.0	36.0	35.0
Permeability (mm./sec.)	← 0.023 →						

Table 2.5.5

Particle Size Distribution of the Soil/Overburden Material at Site 2 (Cregg)

PARTICLE SIZE	Sample Depth (cm)						
	20	40	60	80	100	120	Mean
	Percentage of Total						
Clay (< 0.002 mm)	7	8	6	10	10	7	8
Silt (0.002 - 0.06 mm)	30	29	33	30	25	32	30
Sand (0.06 - 2.0 mm)	41	37	31	38	40	32	36
Gravel (> 2.0 mm)	22	26	30	22	25	29	26

Table 2.5.6

Physical and Chemical Properties of the Soil/Overburden Material at Site 2 (Cregg)

PARAMETERS	Sample Depth (cm)						
	20	40	60	80	100	120	Mean
pH (pH units)	7.35	7.40	7.42	7.48	7.55	7.51	7.45
Porosity (%)	37.0	35.0	38.0	33.0	32.0	32.0	35.0
Organic Matter (%)	12.2	9.6	8.2	10.6	9.5	9.3	9.9
Organic Carbon (%)	5.8	5.2	5.1	5.5	5.3	5.0	5.3
C. E. C. (meq/100g)	29.0	28.0	22.0	21.0	24.0	22.0	24.0
Permeability (mm./sec.)	← 0.04 →						

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Table 2.5.7

Summary Results of the Ammonium - N Concentrations (mg/kg) Detected in the Soil Samples From the Two Test Sites

Sample Number	Sampling Site									
	Site 1 - Dromahaire					Site 2 - Cregg				
	Max.	Min.	Mean	S. D.	Coeff. of Var	Max.	Min.	Mean	S. D.	Coeff. of Var.
1a	88.0	9.0	47.0	24.0	51.1	98.0	19.0	47.6	34.6	51.7
1b	65.0	16.0	36.5	16.3	44.6	89.0	2.0	38.2	25.0	65.5
2a	41.0	10.0	27.9	12.6	45.1	86.8	17.2	48.8	30.7	62.9
2b	90.0	2.0	32.8	27.0	82.4	78.4	9.0	41.6	26.0	62.6
3a	60.0	13.0	23.1	16.2	70.0	14.0	1.0	5.8	4.5	77.1
3b	29.4	10.0	18.0	6.9	38.3	11.2	2.0	5.1	3.3	64.5
4a	95.0	17.6	56.3	27.2	48.3	78.0	2.0	18.7	26.2	140.0
4b	105.0	10.2	47.2	30.4	64.3	25.2	3.0	11.6	6.9	59.3
5a	75.0	12.0	39.0	24.0	61.5	22.4	0.0	5.9	7.1	120.0
5b	61.8	10.0	33.7	20.6	61.2	16.2	0.0	5.0	5.2	105.0
6a	140.0	31.0	74.8	41.3	55.2	8.4	1.0	5.0	2.5	49.7
6b	78.0	29.0	51.0	18.4	36.2	11.2	1.0	5.8	3.6	62.5
7a	12.6	1.80	7.2	3.5	49.5	25.0	6.6	17.6	6.4	36.4
7b	21.0	2.6	7.0	6.1	87.5	23.0	9.0	14.4	4.0	28.0
8a	74.2	3.0	14.0	24.6	175.0	8.6	2.4	4.8	2.1	43.1
8b	63.0	3.0	1.4	2.0	146.0	7.8	2.0	3.9	1.8	45.0
9a	6.6	1.8	3.7	1.6	43.1	7.6	2.1	4.6	2.0	43.0
9b	5.0	0.0	3.0	1.6	52.1	5.9	1.9	3.4	1.3	38.0
10a	4.8	1.0	2.3	1.4	59.8	5.4	0.0	2.3	1.7	74.5
10b	3.2	0.0	1.6	1.0	64.3	5.6	0.0	2.4	1.6	66.8

Max. = Maximum Concentration Recorded
Min. = Minimum Concentration Recorded
Mean = Geometric Mean (x)
S. D. = Standard Deviation
Coeff. of Var. = Coefficient of Variation [(S. D. / x) %]

Figure 2.5.1
Plot of the Ammonium - N concentration (mg/kg) detected in the soil samples at the two test sites on the April 1987 sampling date

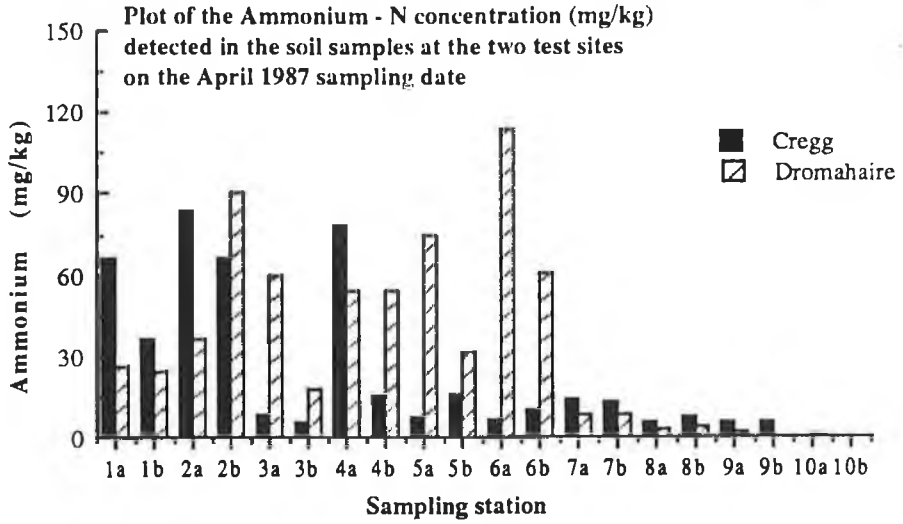

Figure 2.5.2
Plot of the Ammonium - N concentration (mg/kg) detected in the soil samples from the two test sites on the June 1987 sampling date

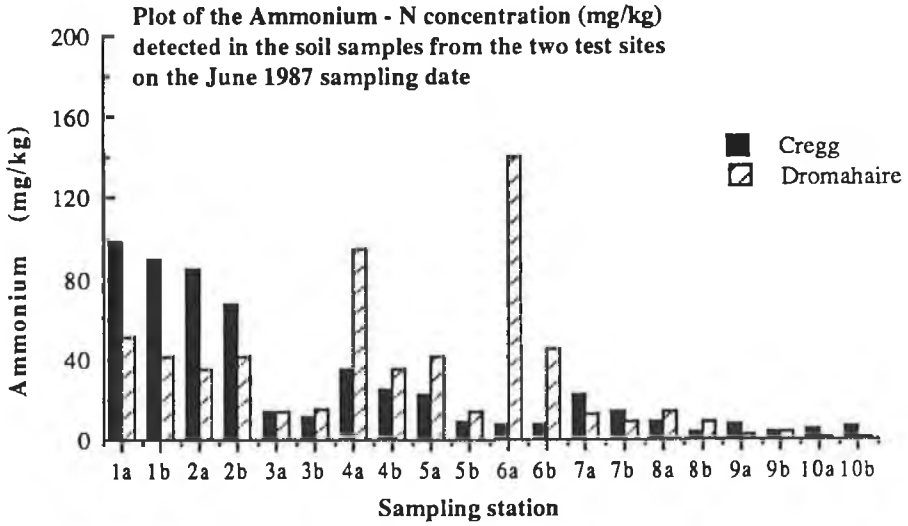

Figure 2.5.3
Plot of the Ammonium - N concentration (mg/kg) detected in the soil samples from the two test sites on the August 1987 sampling date

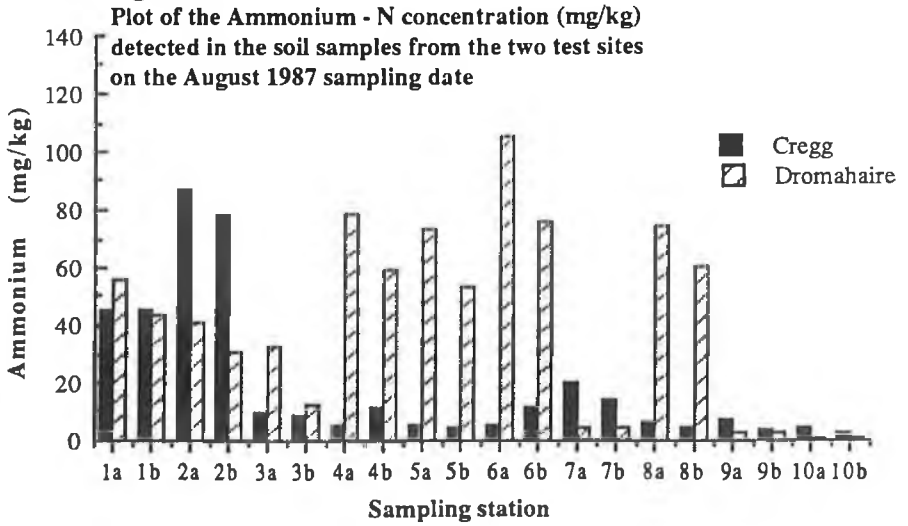

Figure 2.5.4
Plot of the Ammonium - N concentration (mg/kg) detected in the soil samples from the two test sites on the October 1987 sampling date

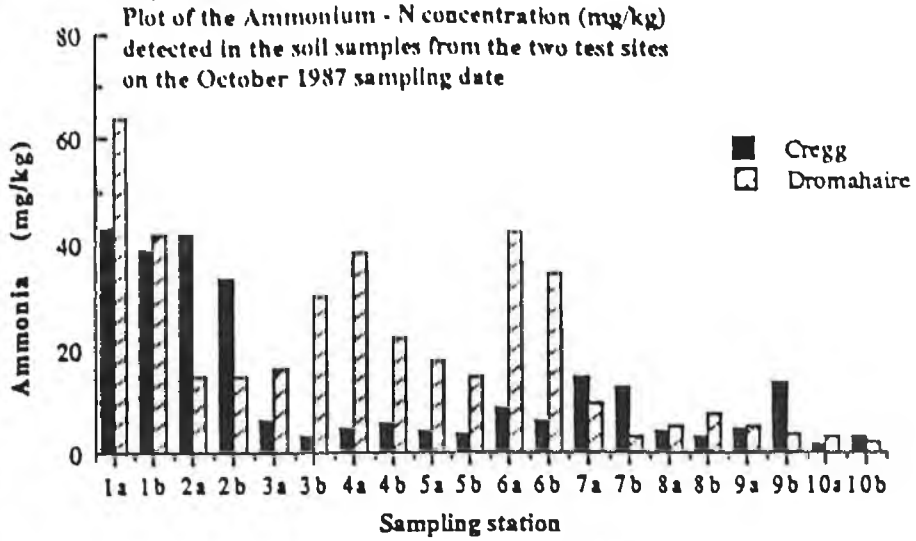

Figure 2.5.5
Plot of the Ammonium - N concentration (mg/kg) detected in the soil samples from the two test sites on the December 1987 sampling date

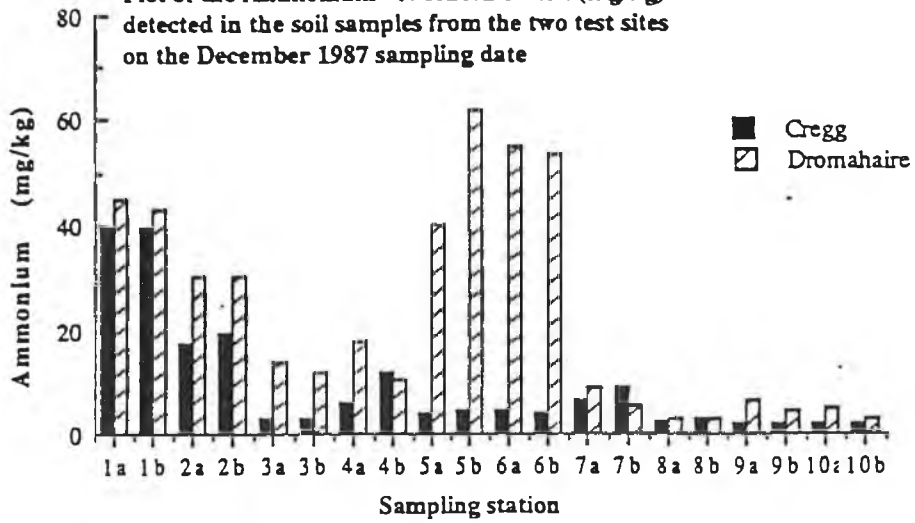

Figure 2.5.6
Plot of the Ammonium - N concentration (mg/kg) detected in the soil samples from the two test sites on the February 1988 sampling date

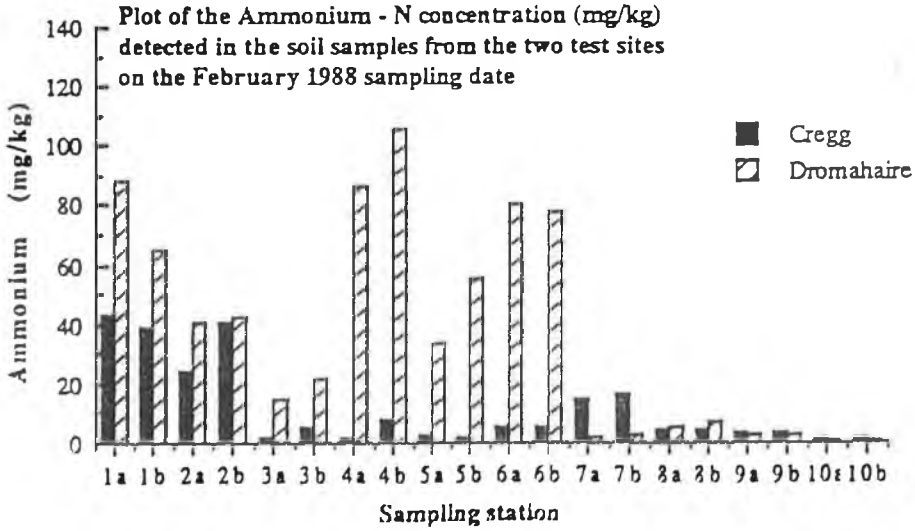

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Figure 2.5.7
Plot of the Ammonium - N concentration (mg/kg) detected in the soil samples from the two test sites on the April 1988 sampling date

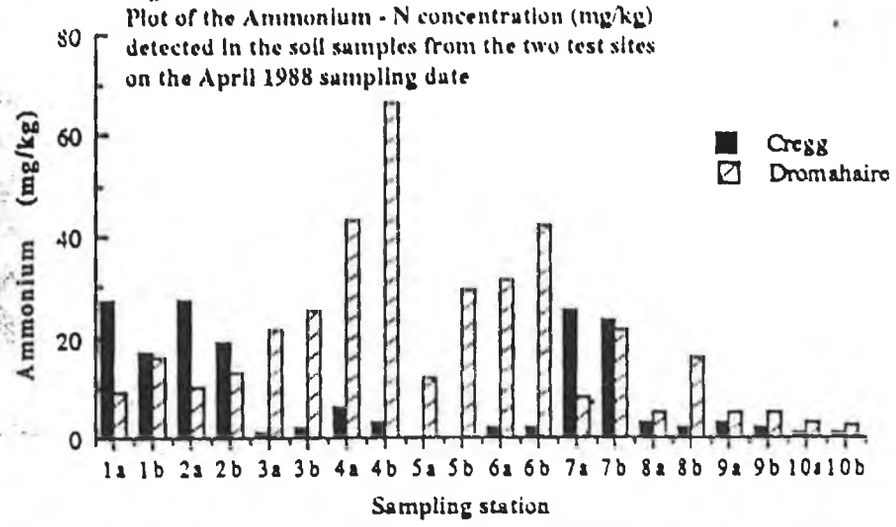

Figure 2.5.8
Plot of the Ammonium - N concentration (mg/kg) detected in the soil samples from the two test sites on the June 1988 sampling date

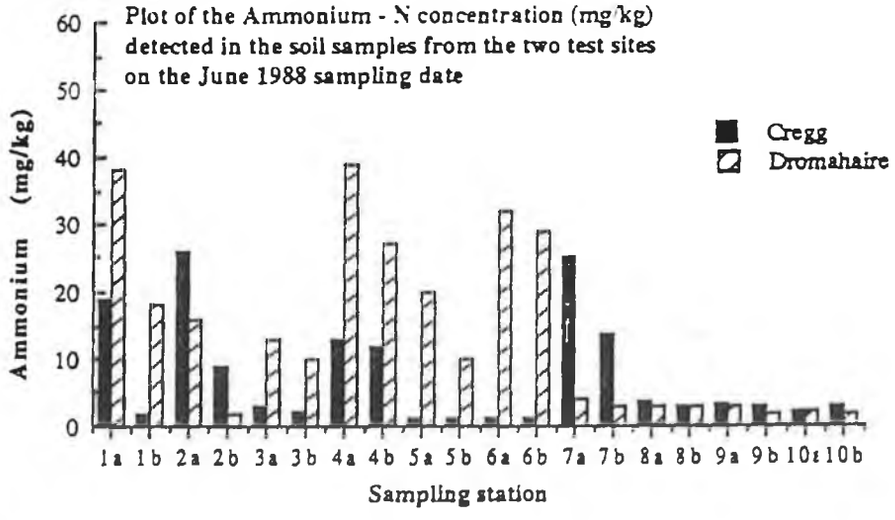

Table 2.5.8

Summary Results of the Nitrate - N Concentrations (mg/kg) Detected in the Soil Samples From the Two Test Sites

Sample Number	Sampling Site									
	Site 1 - Dromahaire					Site 2 - Cregg				
	Max.	Min.	Mean	S. D.	Coeff. of Var	Max.	Min.	Mean	S. D.	Coeff. of Var
1a	14.0	0.0	6.3	5.4	84.6	8.4	1.0	4.2	3.3	78.7
1b	20.4	0.0	7.9	8.0	102.0	18.2	2.0	7.3	5.1	70.7
2a	11.2	0.0	4.1	4.2	105.0	14.0	1.0	6.0	4.6	76.4
2b	42.5	0.0	13.1	17.9	137.0	35.0	1.0	10.1	10.6	105.0
3a	10.5	0.0	5.3	3.6	68.9	9.8	1.2	4.6	3.2	69.8
3b	14.0	1.0	7.8	5.8	73.9	14.0	1.8	6.0	4.0	66.2
4a	7.0	1.0	2.5	1.9	76.1	19.6	1.8	7.6	5.7	74.9
4b	7.0	0.9	2.9	2.1	71.5	25.2	1.8	10.1	8.0	79.5
5a	14.0	0.0	3.4	4.5	134.0	14.0	1.8	5.4	4.3	79.4
5b	14.0	0.0	3.7	4.6	124.0	26.6	1.2	8.6	8.9	103.0
6a	11.2	0.0	5.2	4.1	80.0	16.8	1.9	5.5	4.9	88.5
6b	13.0	0.0	7.4	5.4	72.6	43.4	1.2	9.2	14.0	152.0
7a	8.6	1.8	4.0	2.6	64.4	8.4	2.0	4.6	2.3	50.7
7b	14.0	2.0	6.0	4.0	66.2	13.1	1.2	7.8	4.7	56.9
8a	7.0	1.0	2.9	1.9	67.0	10.7	1.8	4.8	3.2	67.5
8b	8.4	1.4	4.5	2.5	55.0	15.5	1.4	6.4	5.9	85.1
9a	2.8	1.0	1.3	0.6	47.0	6.6	2.1	3.8	1.6	42.6
9b	2.8	1.0	1.6	0.8	49.1	9.5	1.8	5.2	3.1	59.7
10a	3.5	1.1	2.6	0.8	30.8	3.0	1.0	1.8	0.8	42.3
10b	3.5	1.0	2.4	0.8	33.1	7.1	1.0	2.6	2.0	75.3

Max. = Maximum Concentration Recorded
Min. = Minimum Concentration Recorded
Mean = Geometric Mean (x)
S. D. = Standard Deviation
Coeff. of Var. = Coefficient of Variation [(S. D. / x) %]

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Figure 2.5.9
Plot of the Nitrate - N concentration (mg/kg) detected in the soil samples from the two test sites on the April 1987 sampling date

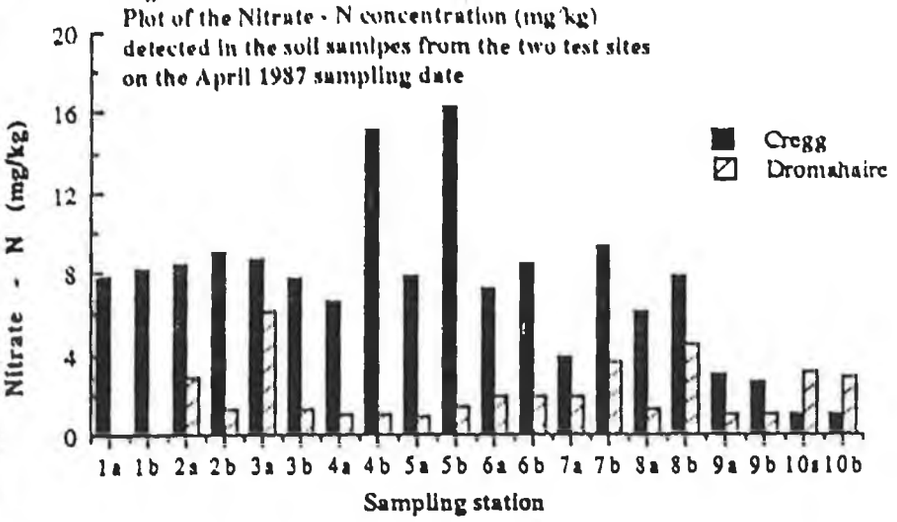

Figure 2.5.10
Plot of the Nitrate - N concentration (mg/kg) detected in the soil samples from the two test sites on the June 1987 sampling date

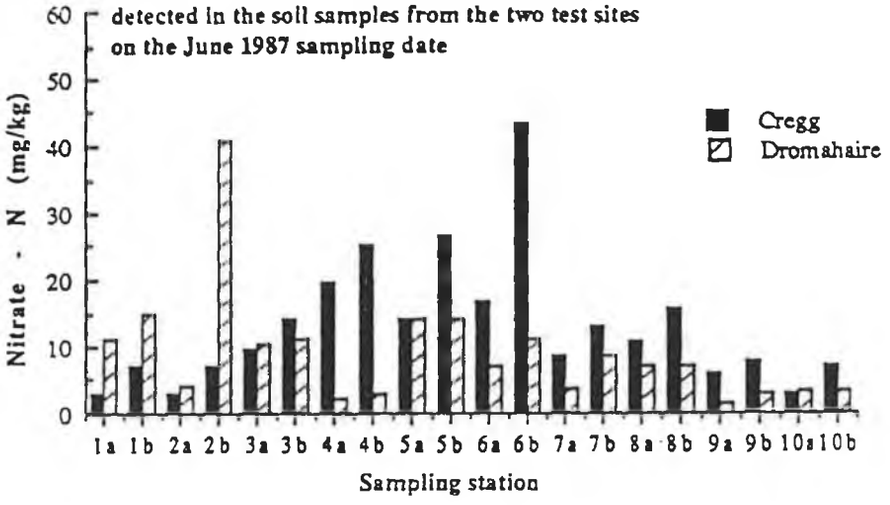

Figure 2.5.11
Plot of the Nitrate - N concentration (mg/kg) detected in the soil samples from the two test sites on the August 1987 sampling date

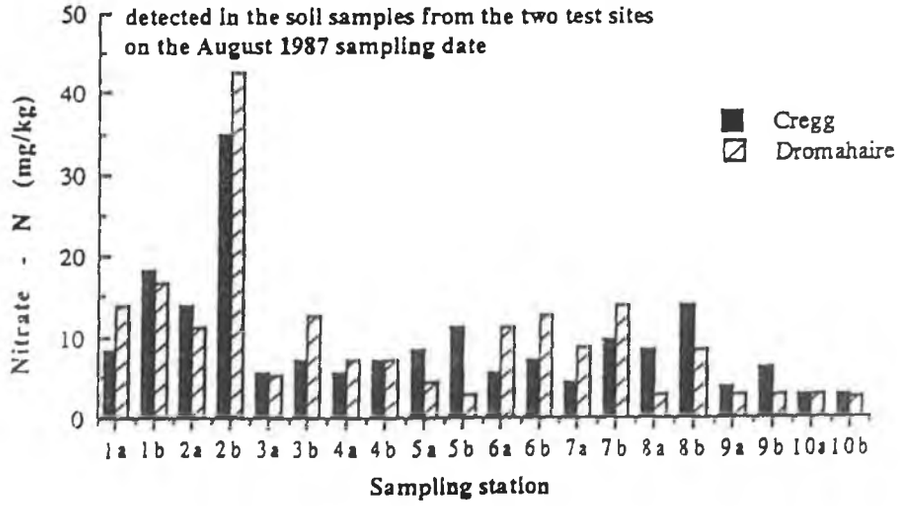

Figure 2.5.12
Plot of the Nitrate - N concentration (mg/kg) detected in the soil samples from the two test sites on the October 1987 sampling date

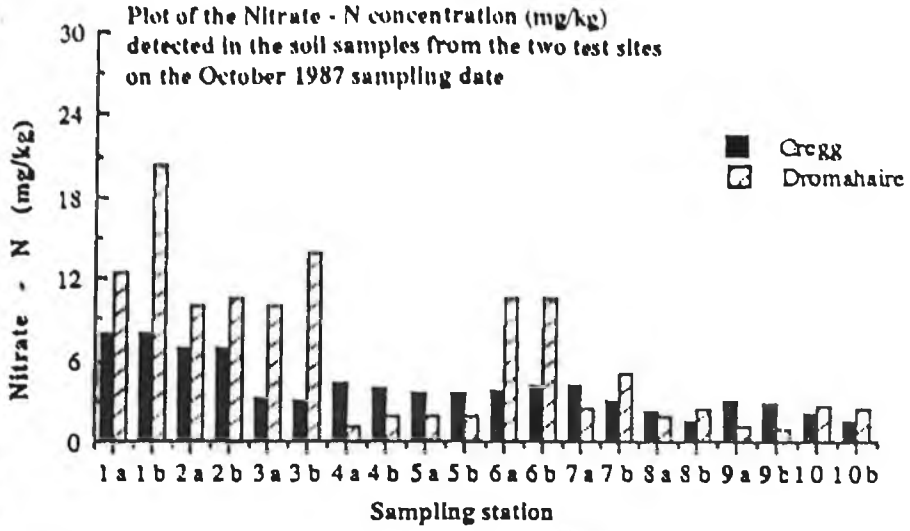

Figure 2.5.13
Plot of the Nitrate - N concentration (mg/kg) detected in the soil samples from the two test sites on the December 1987 sampling date

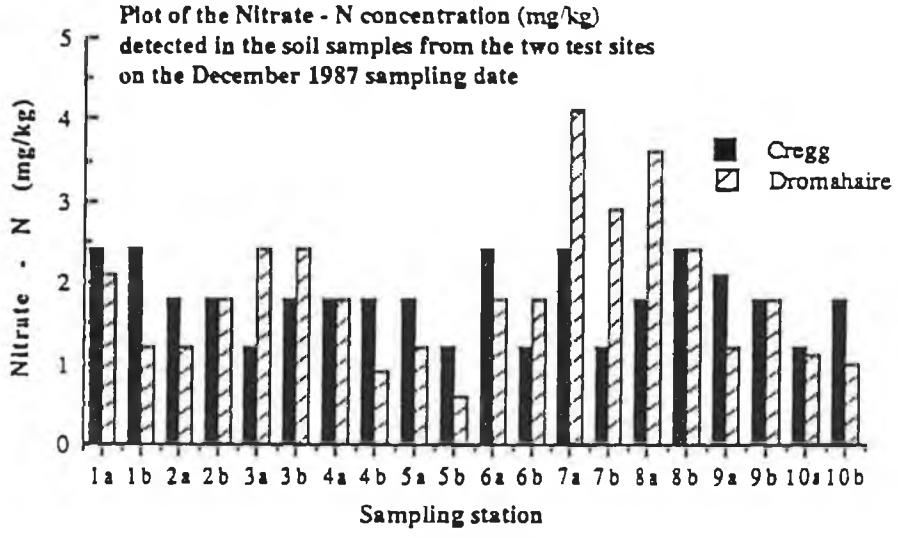

Figure 2.5.14
Plot of the Nitrate - N concentration (mg/kg) detected in the soil samples from the two test sites on the February 1988 sampling date

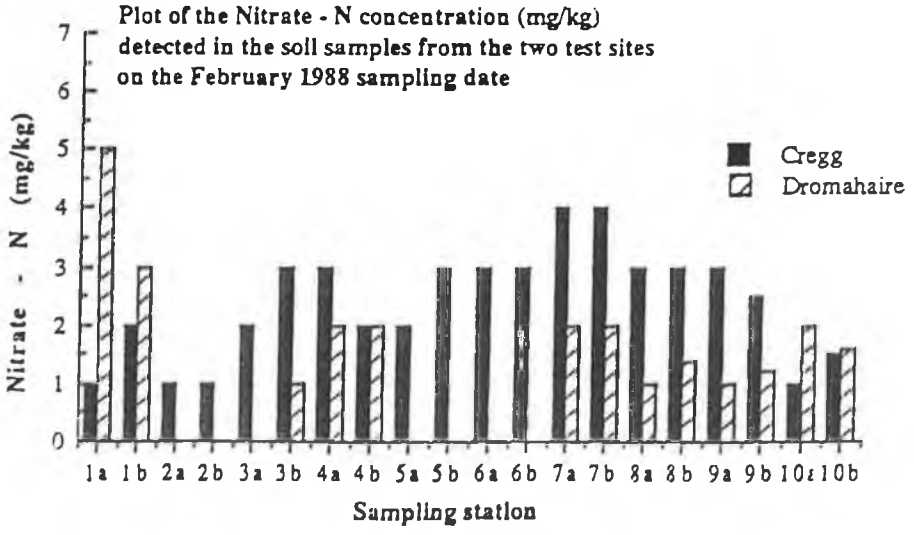

Figure 2.5.15
Plot of the Nitrate - N concentration (mg kg) detected in the soil samples from the two test sites on the April 1988 sampling date

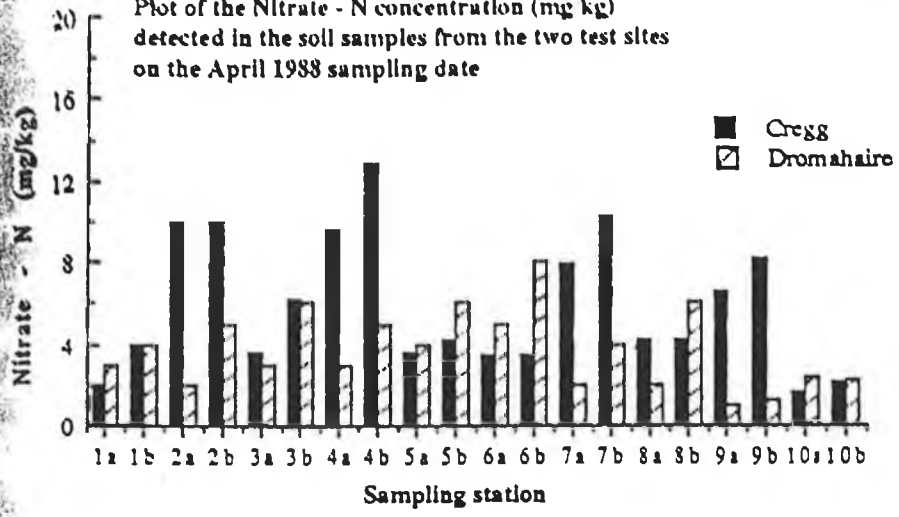

Figure 2.5.16
Plot of the Nitrate - N concentration (mg/kg) detected in the soil samples from the two test sites on the June 1988 sampling date

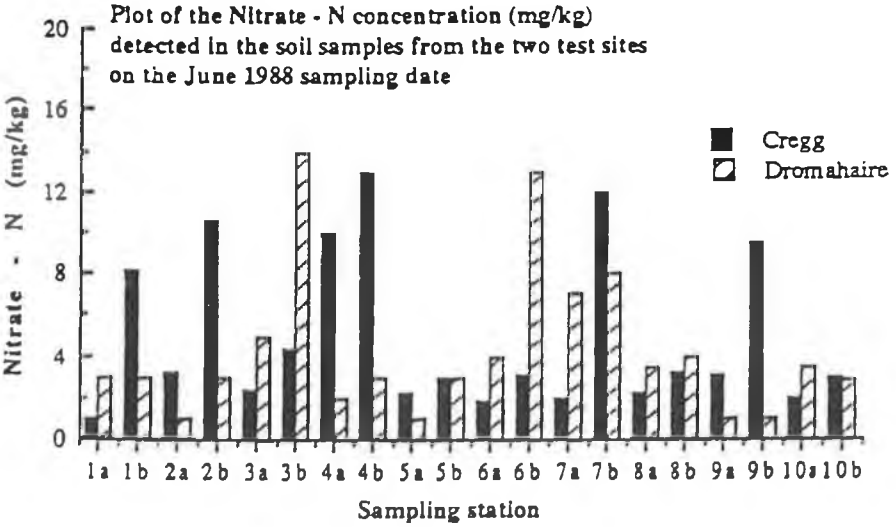

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Table 2.5.9

Summary Results of the Phosphate - P Concentrations (mg/kg) Detected in the Soil Samples From the Two Test Sites

Sample Number	Sampling Site									
	Site 1 - Dromahaire					Site 2 - Cregg				
	Max.	Min.	Mean	S. D.	Coeff. of Var	Max.	Min.	Mean	S. D.	Coeff. of Var
1a	23.1	3.4	11.3	7.3	64.6	40.6	9.5	25.1	11.2	46.6
1b	33.0	1.7	12.4	10.5	84.7	41.9	6.9	22.3	14.7	64.2
2a	31.5	2.5	11.9	9.3	77.9	35.5	9.8	24.5	9.4	38.2
2b	14.6	1.1	9.5	5.7	60.1	36.1	5.5	23.7	11.5	48.3
3a	16.3	5.5	10.3	3.4	33.1	5.0	1.9	2.7	1.1	39.8
3b	34.6	6.5	14.6	9.2	63.1	7.9	1.1	3.0	2.3	77.4
4a	14.7	1.8	8.2	5.2	63.5	22.7	8.5	16.1	4.7	29.4
4b	17.4	1.2	9.1	5.0	54.8	24.3	9.5	15.1	4.9	32.4
5a	13.3	2.6	7.7	4.4	57.0	3.0	0.7	1.7	0.9	53.4
5b	31.7	1.8	12.5	9.8	78.3	2.2	0.5	1.3	0.2	39.9
6a	36.3	5.4	19.4	10.6	54.5	5.6	1.0	2.4	1.6	64.1
6b	30.1	15.7	20.8	5.5	26.3	2.2	0.8	1.5	0.5	32.5
7a	10.9	2.0	5.6	3.0	53.4	23.0	1.9	10.1	7.8	77.6
7b	8.7	1.8	5.0	2.8	55.6	30.6	1.5	10.7	10.1	94.0
8a	9.6	2.3	5.2	2.6	50.1	5.5	0.9	3.0	1.6	55.6
8b	8.5	2.1	5.6	2.2	39.6	8.0	0.9	3.0	2.3	79.0
9a	8.6	1.0	3.8	3.0	78.7	10.5	0.5	3.8	3.1	82.8
9b	8.2	0.7	3.9	3.0	76.9	6.9	0.3	2.1	2.1	99.0
10a	2.2	1.0	1.7	0.6	33.7	3.0	0.0	1.3	1.1	81.8
10b	2.7	0.5	1.5	0.8	51.1	3.0	0.1	1.5	1.1	78.8

Max. = Maximum Concentration Recorded
Min. = Minimum Concentration Recorded
Mean = Geometric Mean (x)
S. D. = Standard Deviation
Coeff. of Var. = Coefficient of Variation [(S. D. / x) %]

Figure 2.5.17
Plot of the Phosphate - P concentrations (mg/kg) detected in the soil samples from the two test sites on the April 1987 sampling date

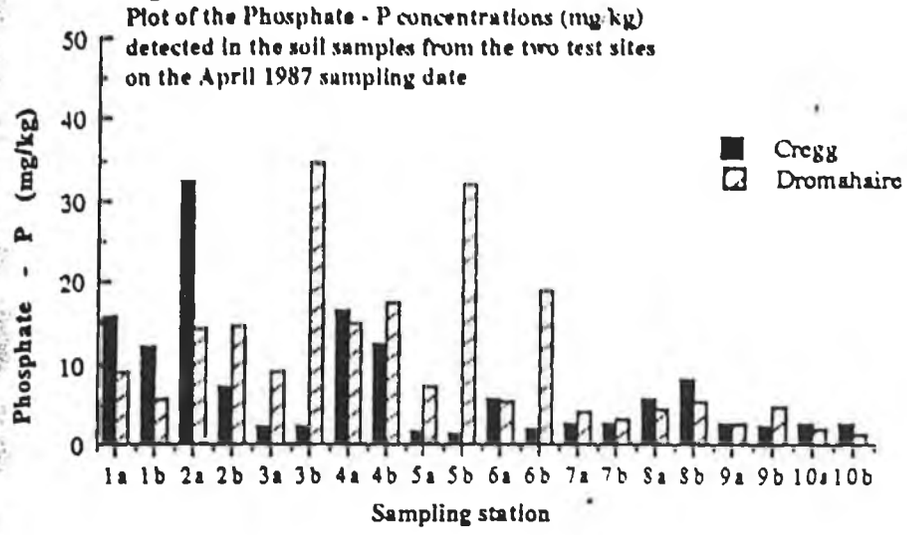

Figure 2.5.18
Plot of the Phosphate - P concentration (mg/kg) detected in the soil samples from the two test sites on the June 1987 sampling date

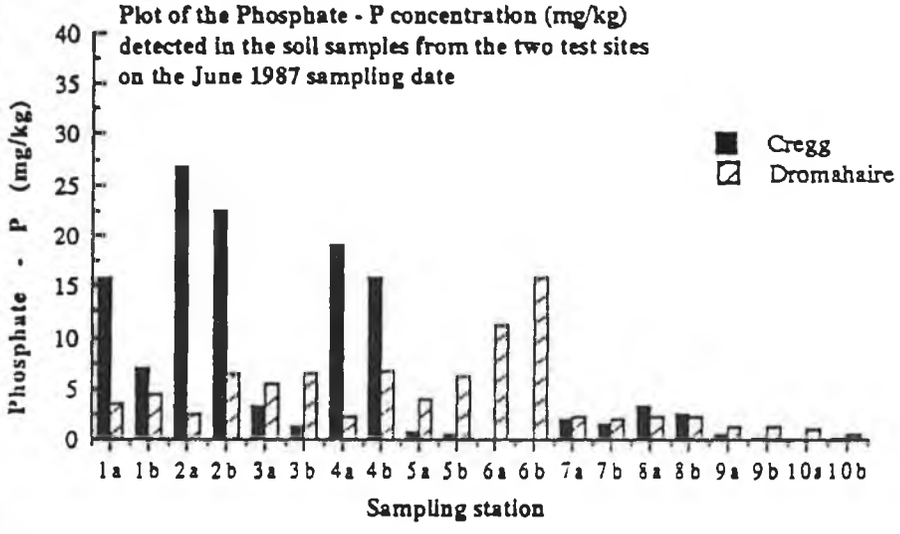

Figure 2.5.19
Plot of the Phosphate - P concnetration (mg/kg) detected in the soil samples from the two test sites on the August 1987 sampling date

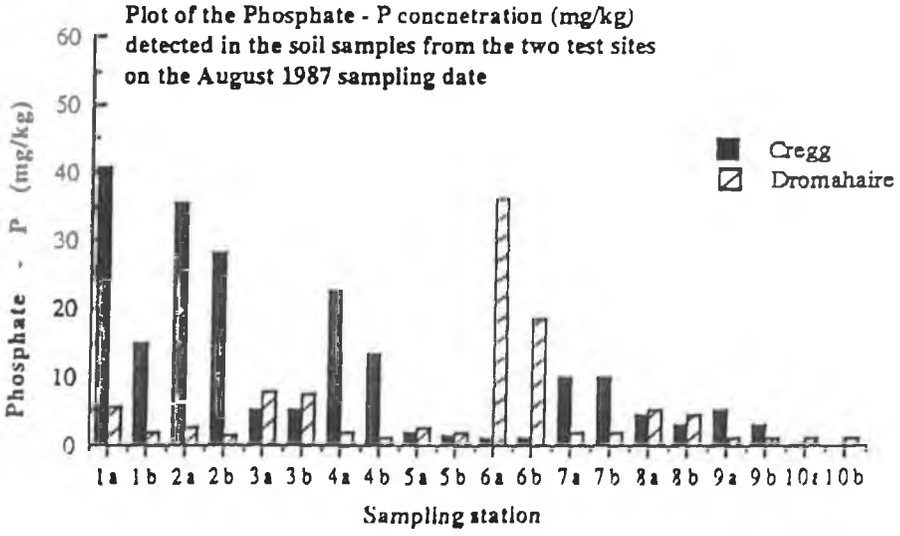

Figure 2.5.20
Plot of the Phosphate - P concentration (mg kg) detected in the soil samples from the two test sites on the October 1987 sampling date

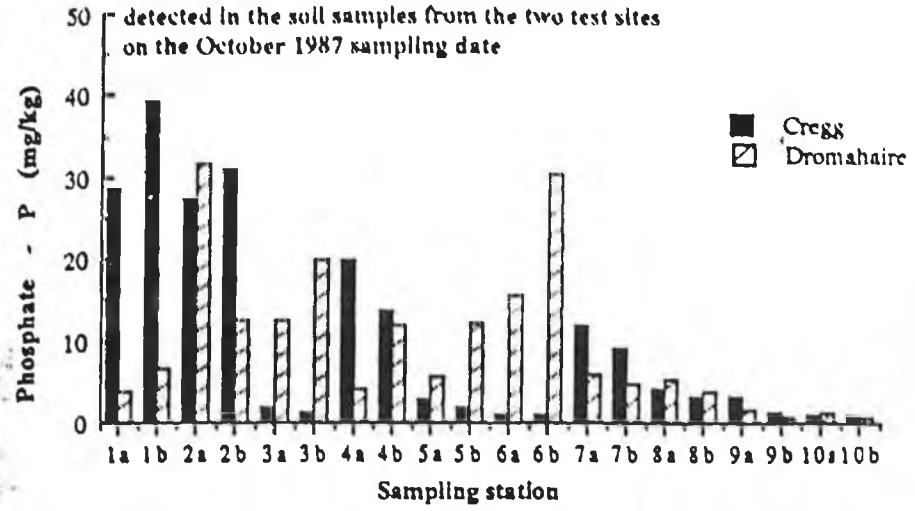

Figure 2.5.21
Plot of the Phosphate - P concentration (mg/kg) detected in the soil samples from the two test sites on the December 1987 sampling date

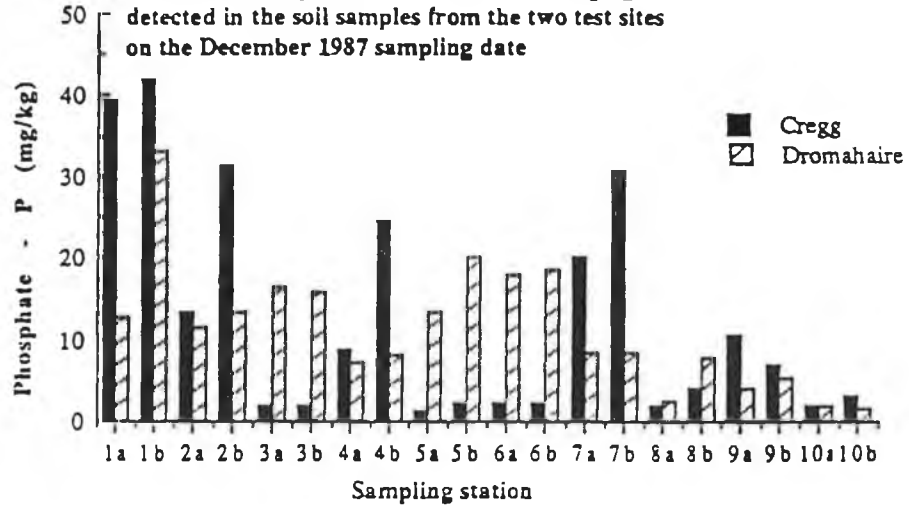

Figure 2.5.22
Plot of the Phosphate - P concnetration (mg/kg) detected in the soil samples from the two test sites on the February 1988 sampling date

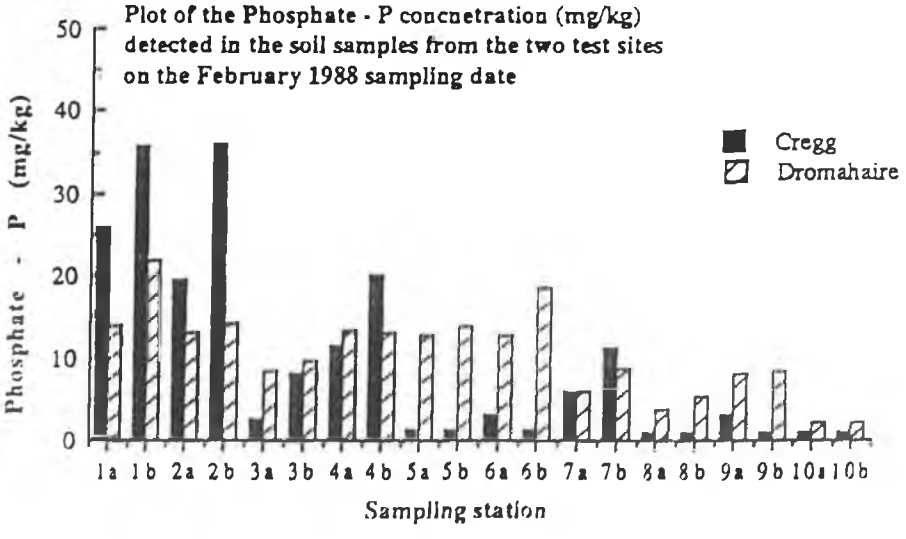

Figure 2.5.23
Plot of the Phosphate - P concentration (mg/kg) detected in the soil samples from the two test sites on the April 1988 sampling date

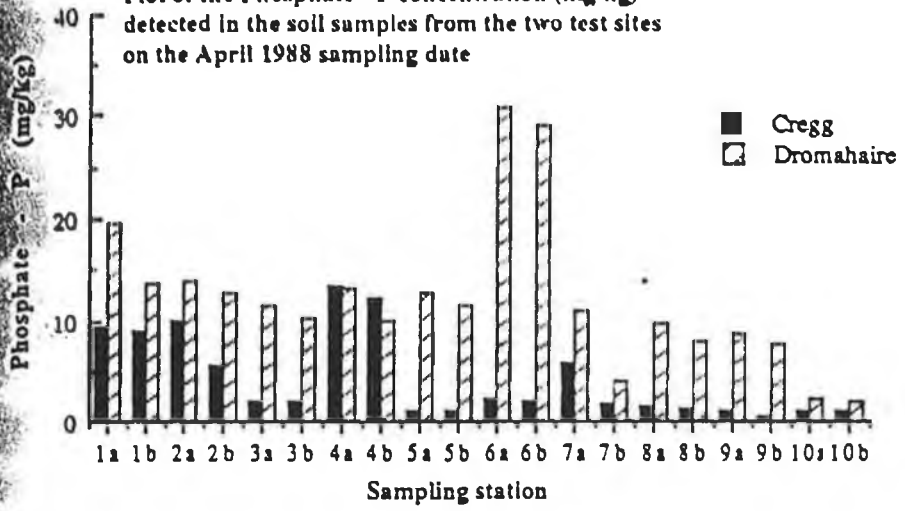

Figure 2.5.24
Plot of the Phosphate - P concentration (mg/kg) detected in the soil samples from the two test sites on the June 1988 sampling date

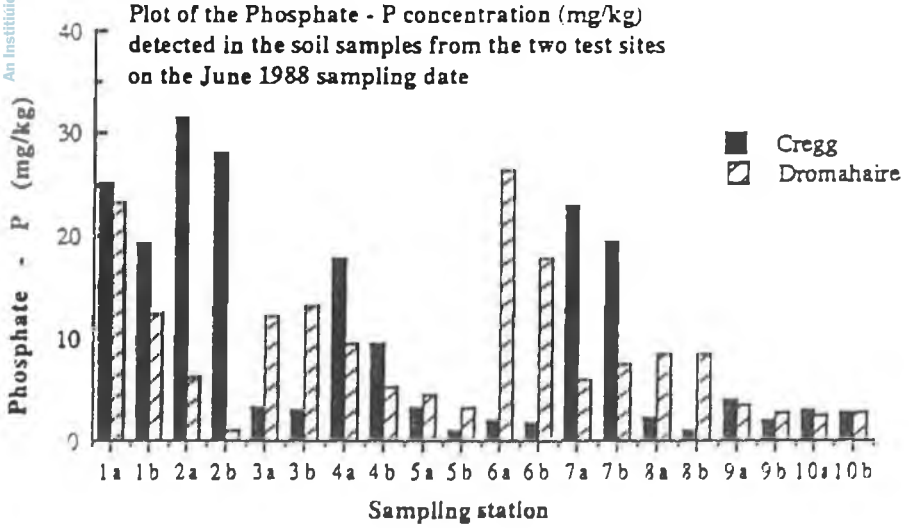

Table 2.5.10

Summary Results of the Numbers of Total Coliform Bacteria (c. f. u. 's/100g) Isolated from the Soil Samples from the Two Test Sites

Sample Number	Sampling Site							
	Site 1 - Dromahaire				Site 2 - Crégg			
	Max.	Min.	S. D.	Coeff. of Var.	Max.	Min.	S. D.	Coeff. of Var.
1a	250.00	3.50	83.00	155.00	1500.00	3.00	560.00	148.00
1b	110.00	0.80	39.00	116.00	2500.00	2.00	890.00	193.00
2a	140.00	1.60	47.00	124.00	3900.00	80.00	1300.00	157.00
2b	70.00	0.80	22.00	129.00	1500.00	2.90	640.00	129.00
3a	1110.00	2.90	370.00	178.00	60.00	1.00	21.00	126.00
3b	630.00	1.30	270.00	141.00	21.00	0.00	8.80	128.00
4a	200.00	1.90	67.00	187.00	64.00	5.00	250.00	90.00
4b	22.00	2.10	6.70	69.00	580.00	2.10	240.00	122.00
5a	67.00	2.30	19.00	101.00	49.00	0.29	18.00	63.00
5b	16.00	1.30	5.80	66.00	140.00	0.20	47.00	160.00
6a	900.00	3.00	310.00	223.00	25.00	0.00	8.80	151.00
6b	100.00	3.00	39.00	84.00	11.00	0.00	4.90	170.00
7a	31.00	2.20	13.00	98.00	260.00	16.00	88.00	123.00
7b	48.00	0.10	17.00	159.00	400.00	2.50	160.00	140.00
8a	86.00	1.60	29.00	128.00	2.60	0.10	0.94	92.00
8b	70.00	0.13	24.00	148.00	2.10	0.00	0.83	133.00
9a	24.00	0.11	8.10	155.00	3.20	0.60	0.94	65.00
9b	60.00	0.10	21.00	240.00	3.50	0.30	1.10	88.00
10a	0.36	0.08	0.10	52.00	0.60	0.10	0.16	62.00
10b	0.50	0.00	0.16	105.00	0.44	0.09	0.12	69.00

Max. = Maximum Concentration Recorded
Min. = Minimum Concentration Recorded
S. D. = Standard Deviation
Coeff. of Var. = Coefficient of Variation [(S. D. / x) %]

Figure 2.5.25
Plot of the numbers of total Coliform bacteria (c. f. u.'s/100g) isolated from the soil samples at the two test sites on the April 1987 sampling date

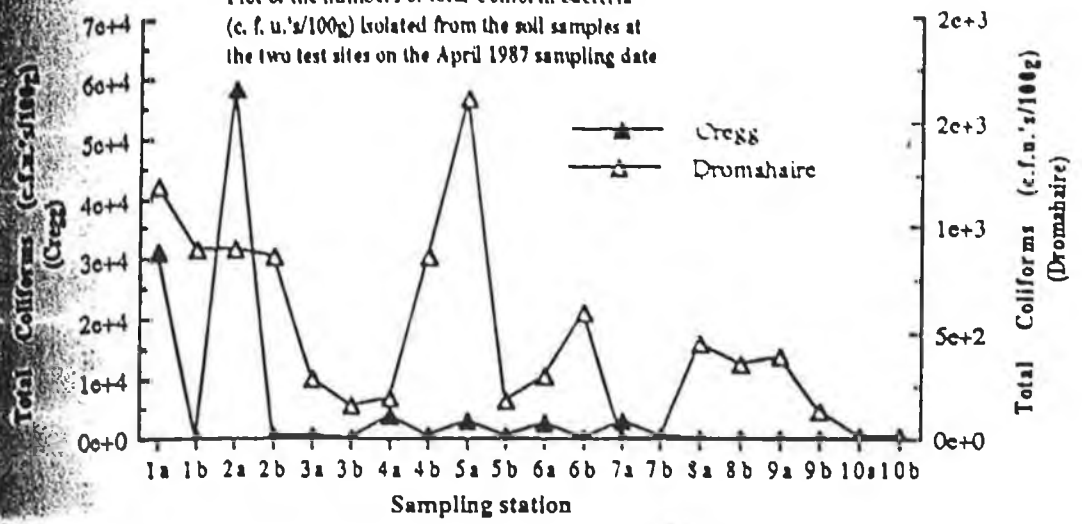

Figure 2.5.26
Plot of the numbers of total Coliform bacteria (c. f. u.'s/100g) isolated from the soil samples at the two test sites on the June 1987 sampling date

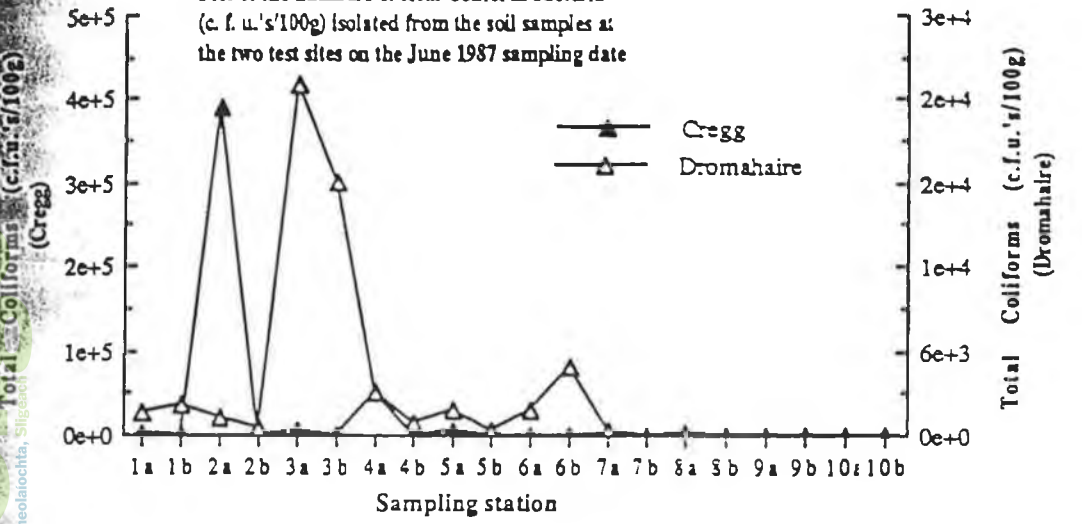

Figure 2.5.27
Plot of the numbers of total Coliform bacteria (c. f. u.'s/100g) isolated from the soil samples at the two test sites on the August 1987 sampling date

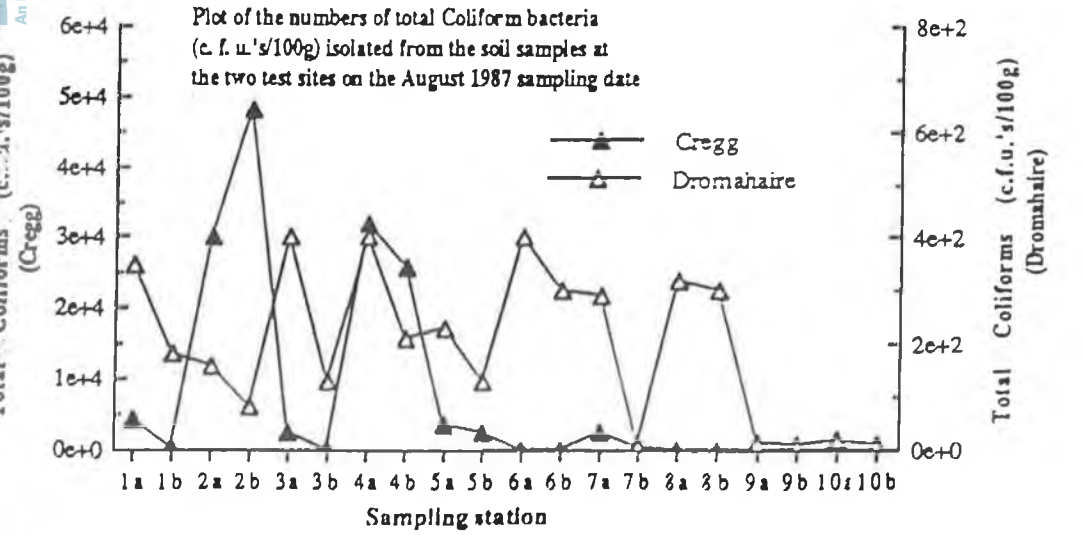

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Figure 2.5.28
Plot of the numbers of total Coliform bacteria (c.f.u.'s/100g) isolated from the soil samples at the two test sites on the October 1987 sampling date

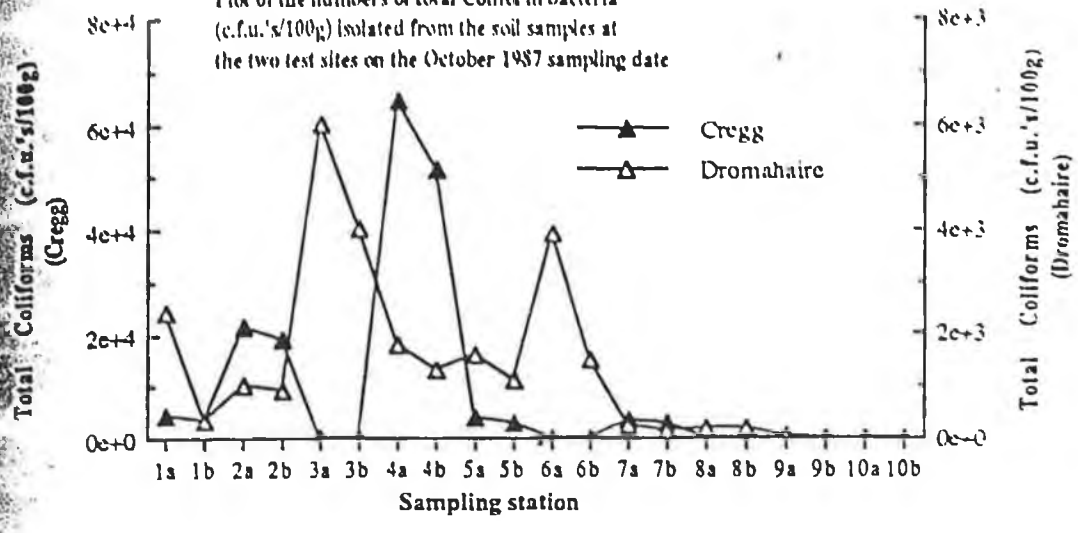

Figure 2.5.29
Plot of the numbers of total Coliform bacteria (c.f.u.'s/100g) isolated from the soil samples at the two test sites on the December 1987 sampling date

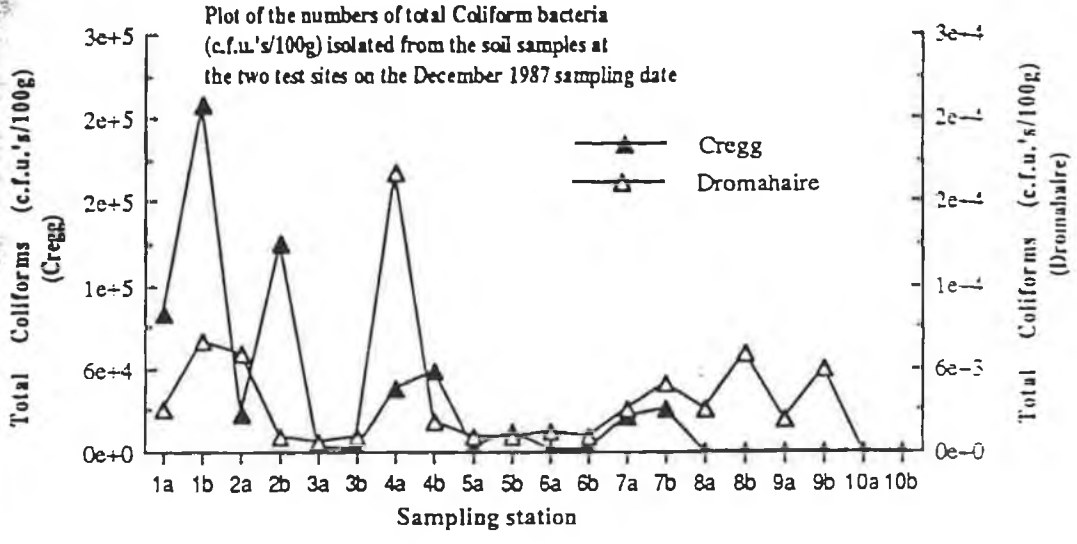

Figure 2.5.30
Plot of the numbers of total Coliform bacteria (c.f.u.'s/100g) isolated from the soil samples at the two test sites on the February sampling date

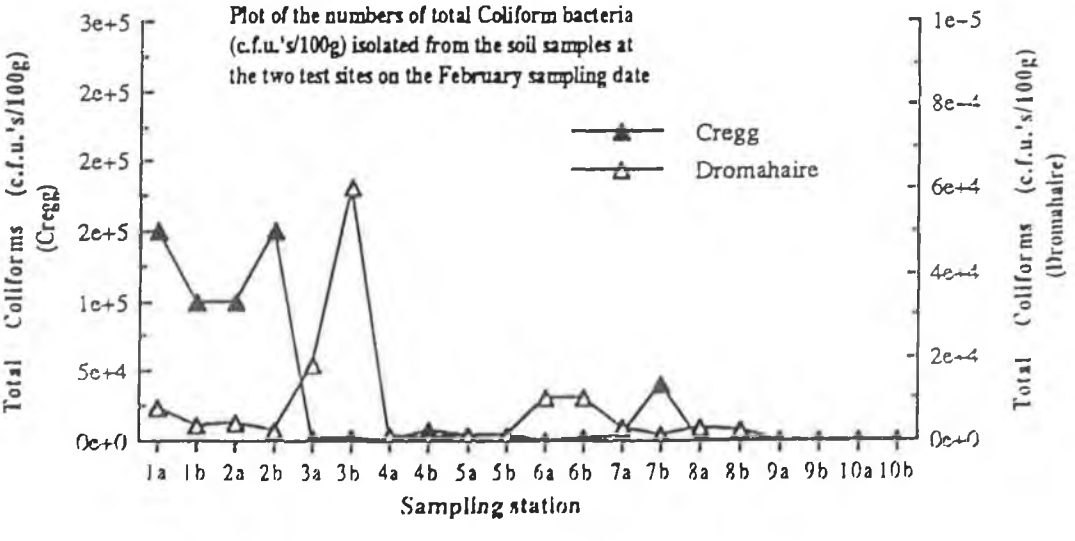

Figure 2.5.31
Plot of the numbers of total Coliform bacteria (c. f. u.'s/100g) isolated from the soil samples at the two test sites on the April 1988 sampling date

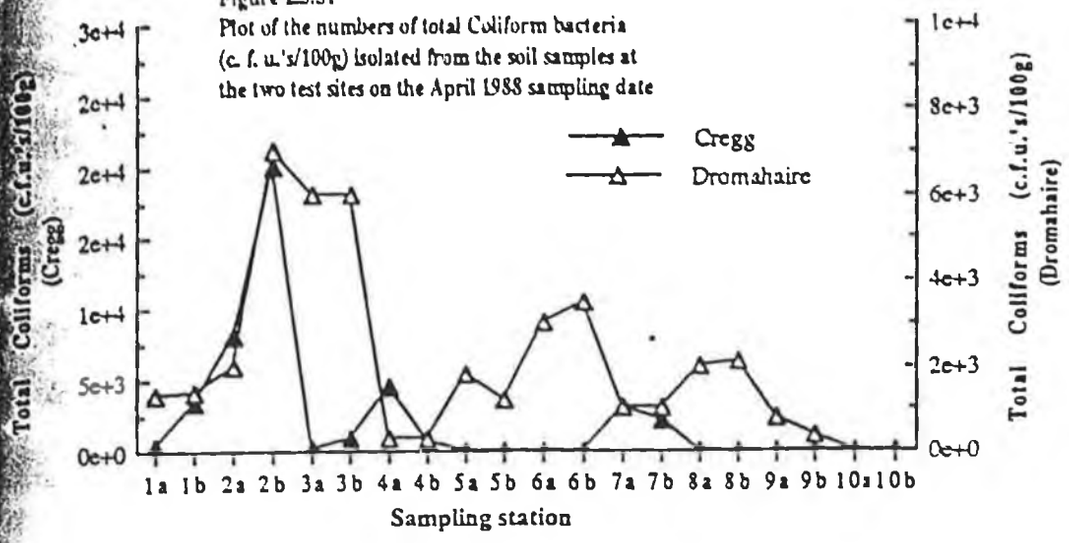

Figure 2.5.32
Plot of the numbers of total Coliform bacteria (c. f. u.'s/100g) isolated from the soil samples at the two test sites on the June 1988 sampling date

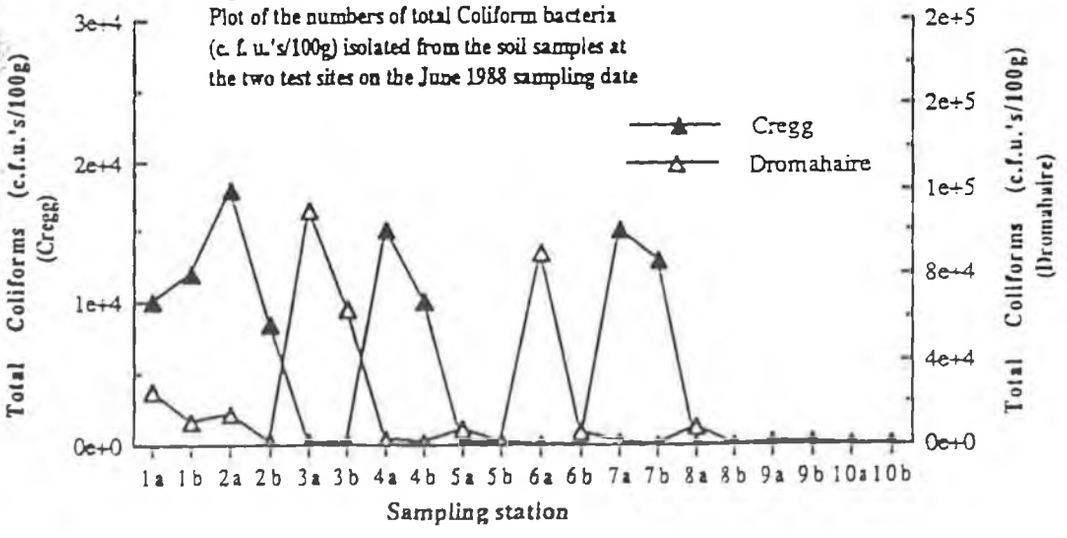

Table 2.5.11

Summary Results of the Numbers of Fecal Coliform Bacteria (c. f. u. 's/100g) Isolated from the Soil Samples from the Two Test Sites

Sample Number	Sampling Site							
	Site 1 - Dromahaire				Site 2 - Cregg			
	Max.	Min.	S. D.	Coeff. of Var.	Max.	Min.	S. D.	Coeff. of Var.
1a	36.00	1.50	14.00	107.00	320.00	2.00	110.00	162.00
1b	12.00	0.71	4.10	67.00	400.00	0.20	140.00	190.00
2a	26.00	0.60	8.80	151.00	71.00	8.00	23.00	64.00
2b	91.00	0.00	31.00	234.00	600.00	2.90	210.00	227.00
3a	150.00	1.20	57.00	152.00	3.00	0.00	1.10	106.00
3b	120.00	0.80	44.00	159.00	2.00	0.00	0.88	163.00
4a	21.00	0.10	7.40	149.00	82.00	10.00	22.00	55.00
4b	9.00	0.10	3.00	145.00	100.00	0.29	37.00	117.00
5a	12.00	0.89	4.40	120.00	34.00	0.00	12.00	207.00
5b	4.20	0.90	1.10	67.00	2.70	0.00	0.96	182.00
6a	50.00	2.00	16.00	167.00	17.00	0.00	6.00	275.00
6b	9.60	0.40	2.90	74.00	0.23	0.00	0.10	149.00
7a	6.00	0.00	2.00	108.00	80.00	1.20	25.00	109.00
7b	4.60	0.00	1.70	125.00	100.00	0.09	34.00	190.00
8a	5.90	0.10	2.20	81.00	1.00	0.00	0.35	217.00
8b	7.00	0.00	2.50	143.00	0.11	0.00	0.04	283.00
9a	1.80	0.00	0.69	161.00	0.36	0.00	0.13	214.00
9b	4.10	0.00	1.40	252.00	0.19	0.00	0.07	205.00
10a	0.25	0.00	0.09	161.00	0.10	0.00	0.05	180.00
10b	0.14	0.00	0.05	283.00	0.10	0.00	0.04	283.00

Max. = Maximum Concentration Recorded
Min. = Minimum Concentration Recorded
S. D. = Standard Deviation
Coeff. of Var. = Coefficient of Variation [(S. D. / x) %]

Figure 2.5.33
Plot of the number of fecal Coliform bacteria (c. f. u.'s/100g) isolated from the soil samples at the two test sites on the April 1987 sampling date

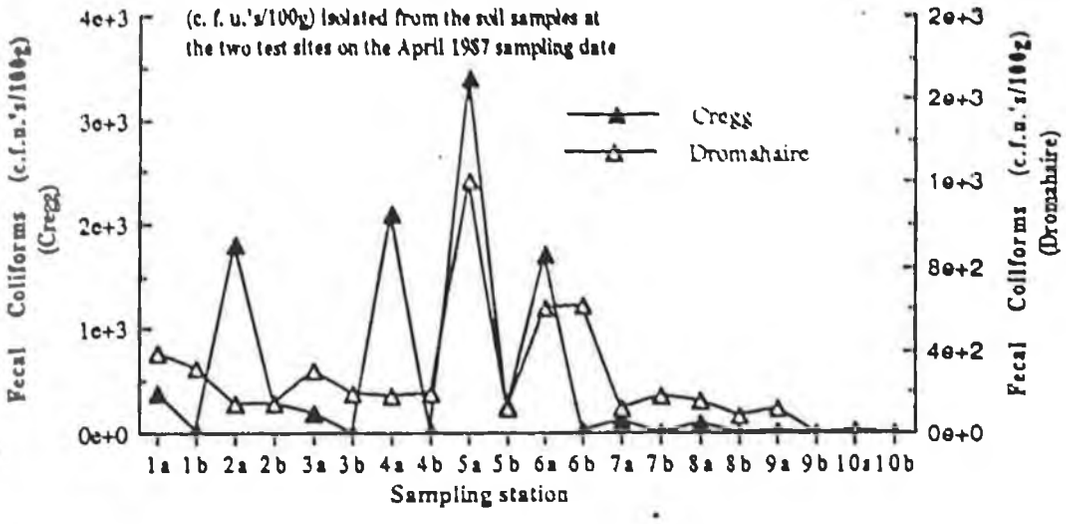

Figure 2.5.34
Plot of the number of fecal Coliform bacteria (c. f. u.'s/100g) isolated from the soil samples at the two test sites on the June 1987 sampling date

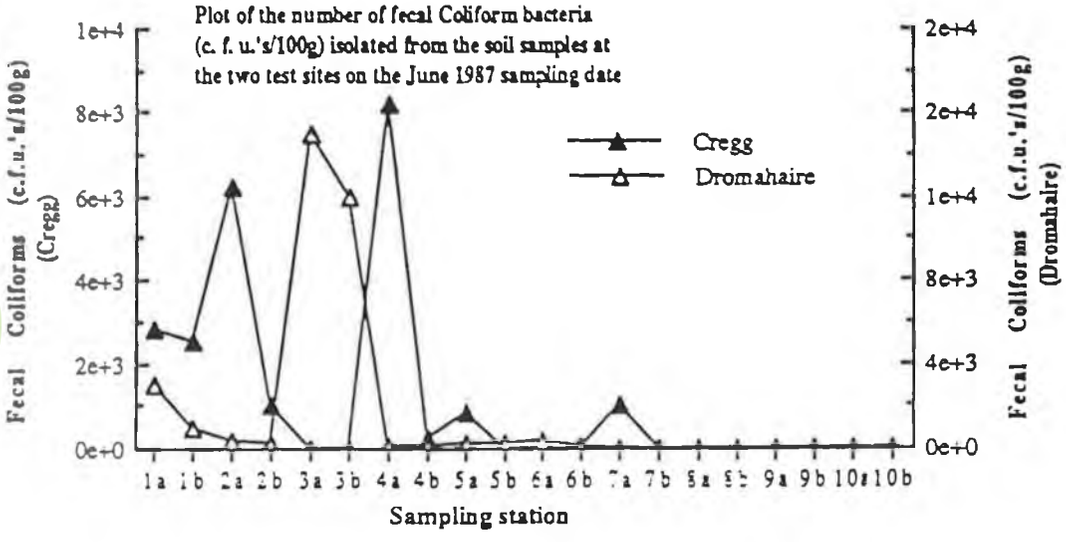

Figure 2.5.35
Plot of the numbers of fecal Coliform bacteria (c. f. u.'s/100g) isolated from the soil samples at the two test sites on the August 1987 sampling date

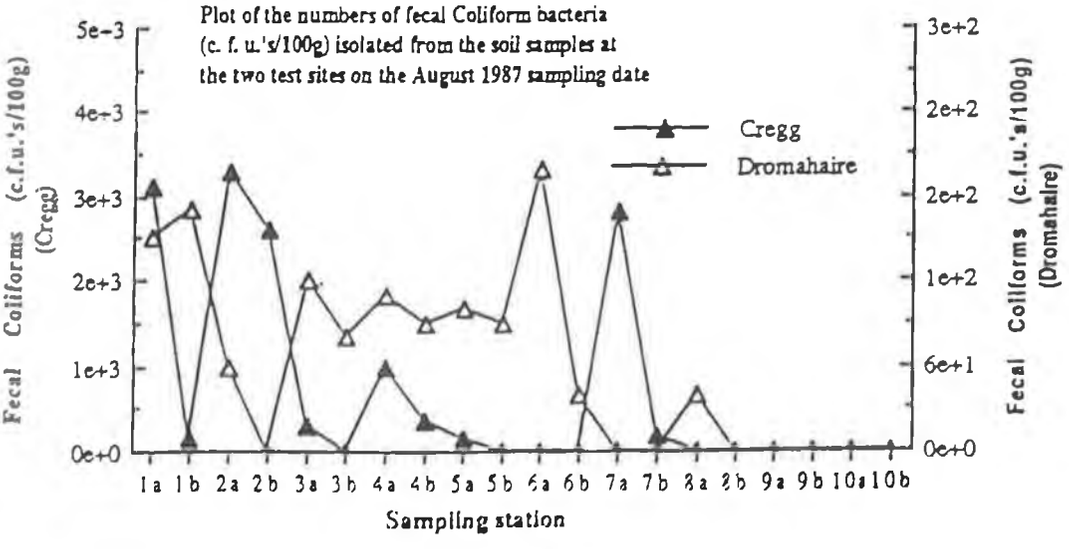

Figure 2.5.36
Plot of the numbers of fecal Coliform bacteria (c. f. u.'s/100g) isolated from the soil samples at the two test sites on the October 1987 sampling date

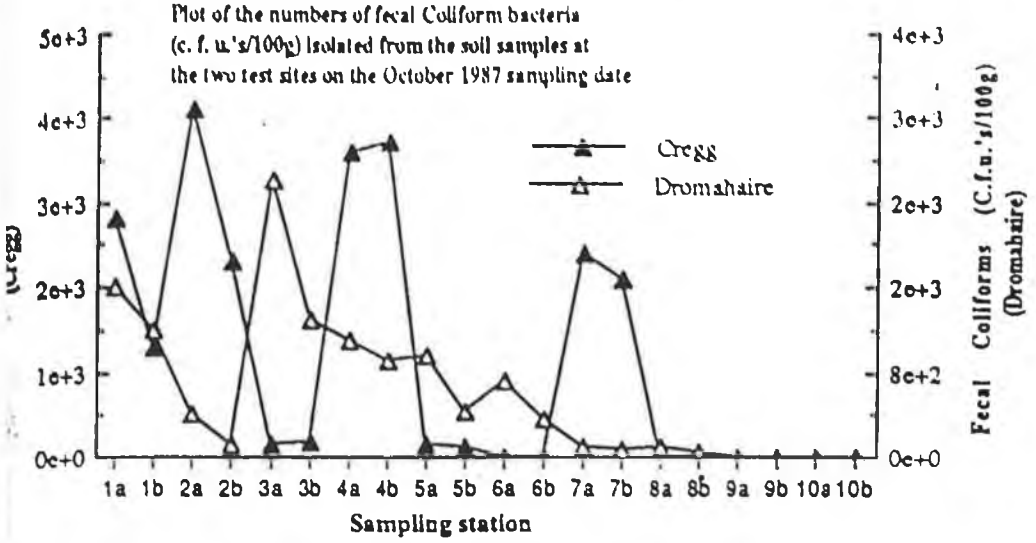

Figure 2.5.37
Plot of the numbers of fecal Coliform bacteria (c. f. u.'s/100g) ioslated from the soil samples at the two test sites on the December 1987 sampling date

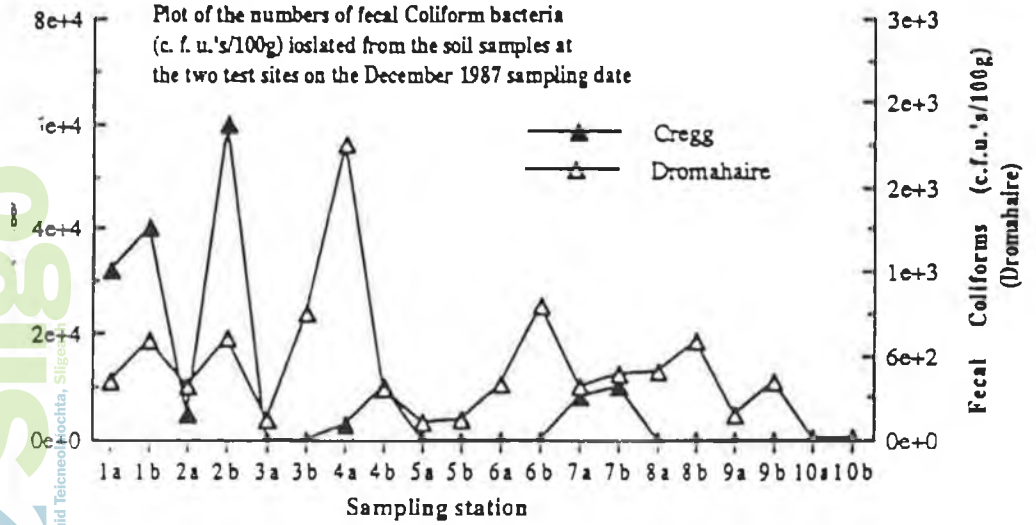

Figure 2.5.38
Plot of the numbers of fecal Coliform bacteria (c. f. u.'s/100g) isolated from the soil samples at the two test sites on the February 1988 sampling date

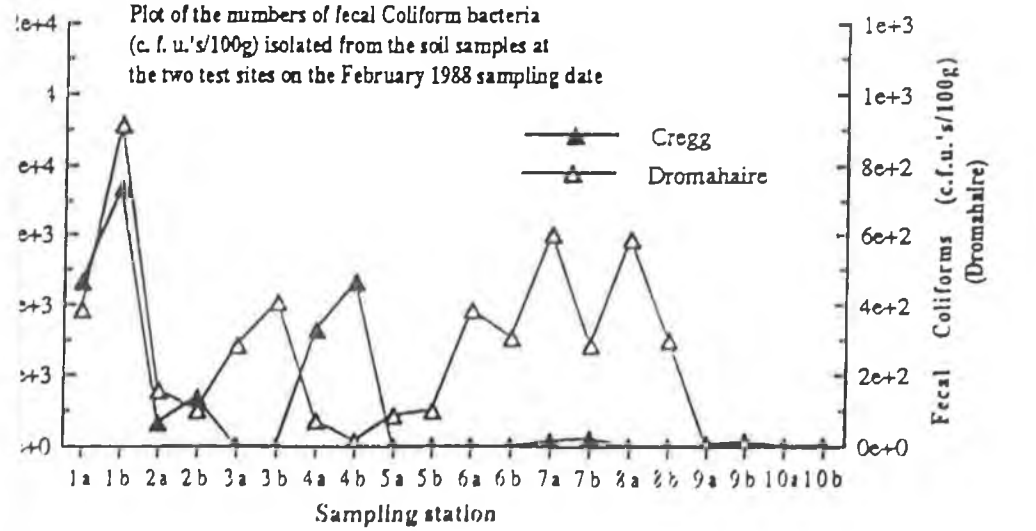

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Figure 2.5.39
Plot of the numbers of fecal Coliform bacteria
(c. f. u.'s/100g) isolated from the soil samples
at the two test sites on the April 1988 sampling date

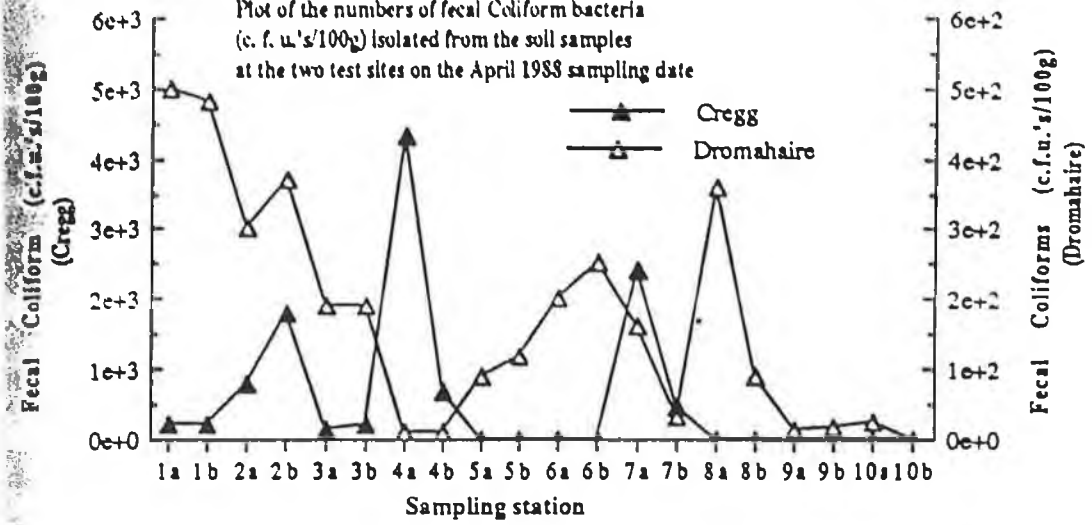

Figure 2.5.40
Plot of the numbers of fecal Coliform bacteria
(c. f. u.'s/100g) isolated from the soil samples
at the two test sites on the June 1988 sampling date

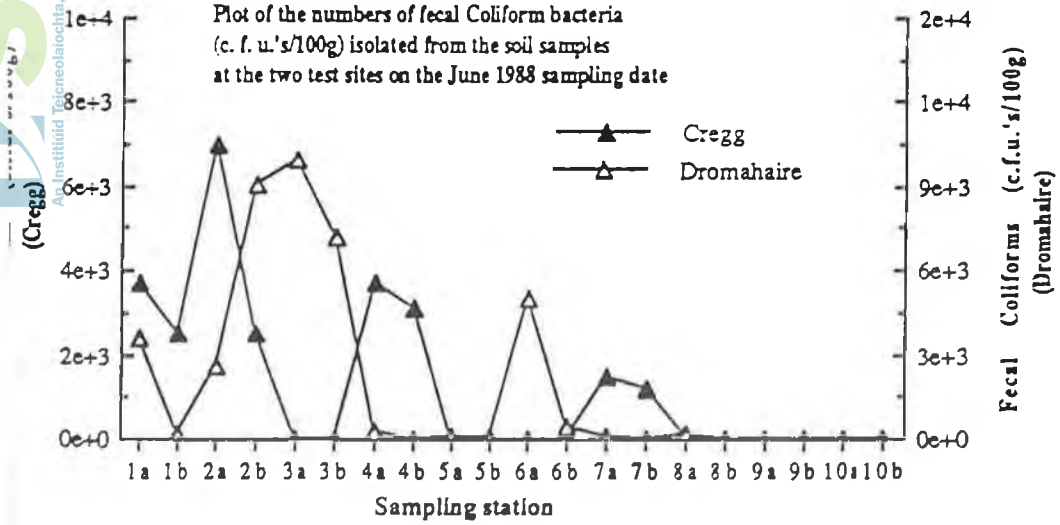

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Figure 2.5.41
Total monthly rainfall (mm) during the sampling period April 1987 to June 1988

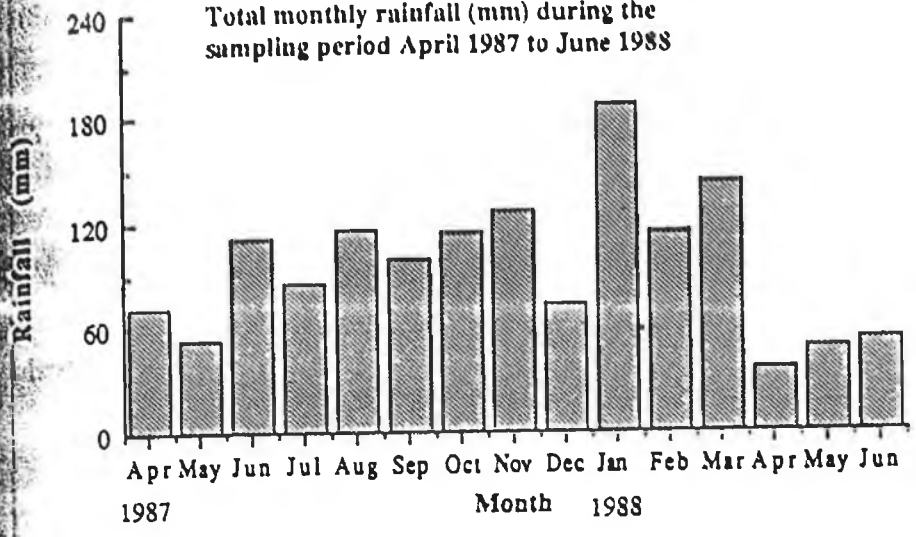

Figure 2.5.42
Total rainfall (mm) for the five day period preceeding sampling

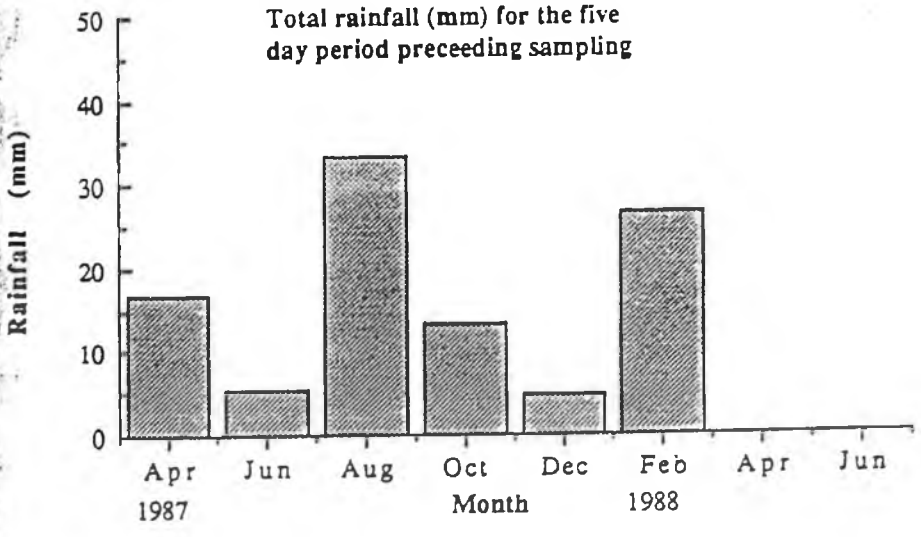

Table 2.5.12

Results of the Analysis of the Sampling Piezometers Installed at Site 1 (Dromahaire)

Piezometer (Sampling Date)		Ammonia - N (mg/l)	Nitrate - N (mg/l)	Phosphate - P (mg/l)	Total Coliforms (c. f. u. 's/100ml)	Fecal Coliforms (c. f. u. 's/100ml)
A	June '87	0.00	3.10	0.00	110.00	22.00
	Feb. '88	0.05	0.90	0.05	280.00	100.00
	June '88	0.00	2.60	0.00	100.00	18.00
B	June '87	3.55	6.10	3.01	3500.00	290.00
	Feb. '88	6.82	1.80	4.60	7200.00	3000.00
	June '88	1.92	4.50	2.00	4000.00	600.00
C	June '87	4.35	8.20	2.92	3900.00	480.00
	Feb. '88	9.65	1.90	2.99	6500.00	2200.00
	June '88	6.10	4.90	1.16	1000.00	290.00
D	June '87	0.90	4.80	1.11	100.00	38.00
	Feb. '88	2.90	1.10	1.67	1100.00	340.00
	June '88	1.56	3.60	0.95	190.00	111.00
E	June '87	1.00	4.30	0.50	39.00	28.00
	Feb. '88	2.10	0.90	0.90	860.00	200.00
	June '88	0.90	3.20	0.00	16.00	14.00
F	June '87	1.11	5.10	0.18	42.00	10.00
	Feb. '88	2.10	1.00	0.58	4000.00	1800.00
	June '88	1.00	3.30	0.00	39.00	12.00

Table 2.5.13

Particle Size Distribution of the Three Soil Types Used in the Bacterial Survival Study

PARTICLE SIZE	Soil Type		
	Loam	Sand	Peat
	Percentage of Total		
ay (< 0.002 mm)	17	2	--
lt (0.002 - 0.06 mm)	38	12	--
and (0.06 - 2.0 mm)	33	69	--
ravel (> 2.0 mm)	12	17	--

Table 2.5.14

Physical and Chemical Properties of the Three Soil Types Used in the Bacterial Survival Study

PARAMETERS	Soil Type		
	Loam	Sand	Peat
pH (pH units)	7.40	7.55	4.79
Porosity (%)	38.0	19.0	-
Organic Matter (%)	19.9	7.4	75.0
C. E. C. (meq/100g)	31.0	10.1	68.0

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Table 2.5.15

The Number of Escherichia coli Bacteria (e. f. u. 's/g) Isolated From the Three Soil Samples at Various Time Intervals After Bacterial Culture Addition

Time (days)	Escherichia coli (e. f. u. 's/g) (x 1000)		
	Loam soil	Sand soil	Peat soil
0.125	380.00	270.00	900.00
0.250	450.00	200.00	250.00
1.0	2400.00	280.00	370.00
1.5	380.00	350.00	1500.00
2.0	180.00	140.00	790.00
3.0	170.00	170.00	500.00
4.0	250.00	250.00	270.00
5.0	220.00	220.00	240.00
6.0	200.00	230.00	650.00
7.0	140.00	130.00	520.00
8.0	310.00	60.00	530.00
9.0	150.00	70.00	110.00
10.0	190.00	120.00	250.00
11.0	120.00	120.00	250.00
12.0	70.00	50.00	290.00
13.0	70.00	30.00	80.00
15.0	60.00	20.00	20.00
17.0	50.00	10.00	50.00
19.0	10.00	60.00	80.00
21.0	20.00	20.00	30.00
23.0	14.00	20.00	20.00
25.0	28.00	15.00	14.00
28.0	4.00	11.00	18.00
31.0	2.00	11.00	6.00
35.0	1.00	3.40	3.80
39.0	0.45	0.70	2.50
44.0	0.69	0.66	2.60

Figure 2.5.43
Plot of the number of Escherichia coli bacteria isolated from the soil samples against time (days) after the addition of the culture - Loam soil

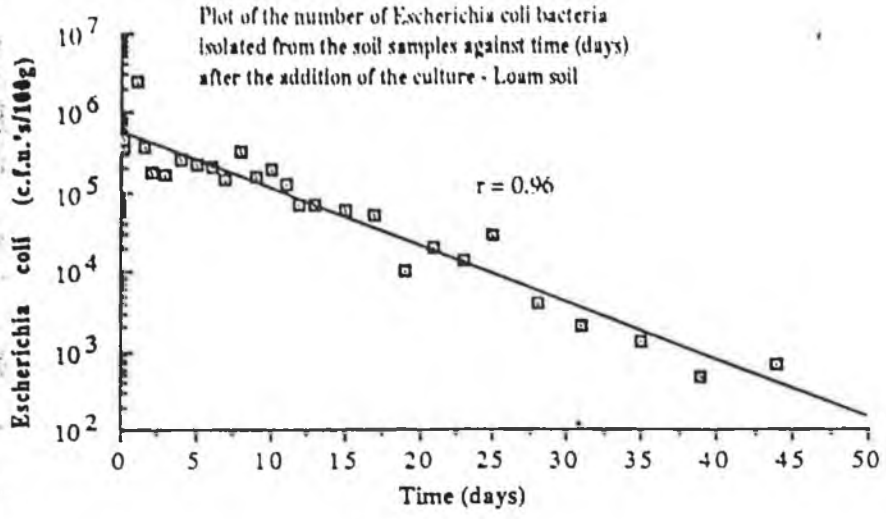

Figure 2.5.44
Plot of the numbers of Escherichia coli bacteria isolated from the soil samples against time (days) after the addition of the culture - Sand soil

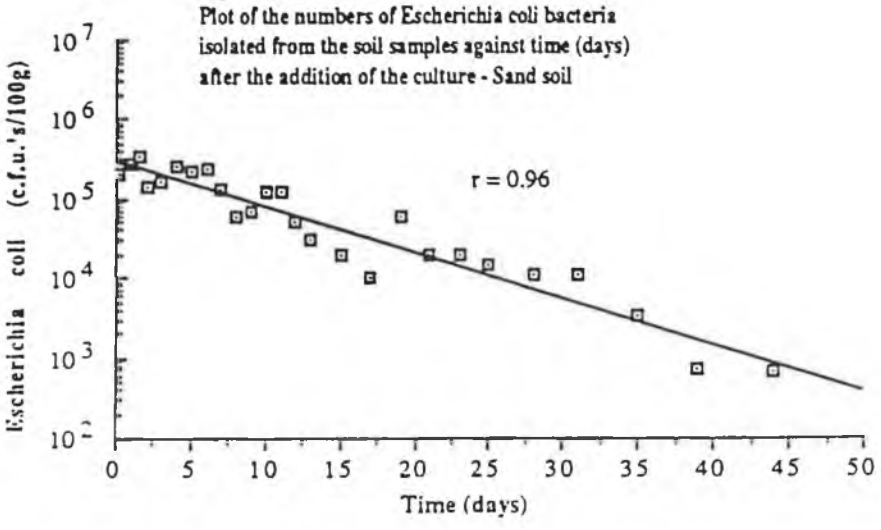

Figure 2.5.45
Plot of the numbers of Escherichia coli bacteria isolated from the soil samples against time (days) after the addition of the culture - Peat soil

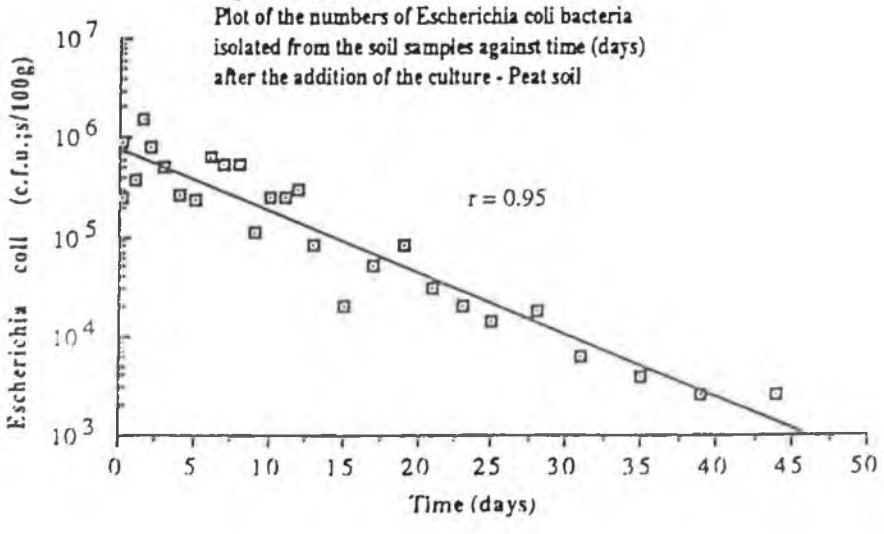

Table 2.5.16

Table Showing the Pearson's Correlation Coefficient (r) and Coefficient of Determination (r^2) Calculated from the Regression Lines of the Numbers of *E. coli* Bacteria (c. f. u. 's/g) Isolated from the Soil Samples Against the Sampling Time (days). The Significance of the Correlation was Calculated Using the t - Statistic at P = 0.01 and 0.005 Confidence Levels

Regression Lines *E. Coli* (no./g) Vs Time (days)	r	r^2	t $\left(t = r\sqrt{n-2} / \sqrt{1-r^2}\right)$	Significance Level (P)	
				0.01	0.005
Loam	0.96	0.92	17.14	+	+
Sand	0.96	0.92	17.14	+	+
Peat	0.95	0.90	15.21	+	+

Note:

+ = The correlation is significant at the given significance level (P)
- = The correlation is not significant at the given significance level (P)

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The discussion of the results is divided into four main sections as follows:

2:6.1 Septic Tank Effluent Quality

2:6.2 Soil Overburden Analysis

2:6.3 The Movement and Restriction of Septic Tank Effluent Constituents in the Two Septic Tank Soil Treatment Systems:

- 2:6.3.1 Nitrogen
- 2:6.3.2 Phosphate
- 2:6.3.3 Coliform bacteria

2:6.4 The Survival of *Escherichia coli* Bacteria in a Range of Soil Types

2:6.1 Septic Tank Effluent Quality

Summary results of the septic tank effluent analysis at the two test sites are presented in Tables 2.5.1 (p57) and 2.5.2 (p58). The poor quality of the effluent is clearly demonstrated in the elevated concentrations of Biochemical Oxygen Demand (B. O. D.), Chemical Oxygen Demand (C. O. D.) and Suspended Solids (S. S.) recorded. At site 1 mean values observed were 346, 596 and 160 mg/l for B. O. D., C. O. D. and S. S. respectively. The corresponding values at site 2 were 564, 1050 and 265 mg/l for B. O. D., C. O. D. and S. S. respectively. These levels are significantly high, given that the average concentrations in raw municipal sewage are 200, 350 and 240 mg/l (Hammer, 1977). The maximum concentrations recorded were considerably higher than have previously been reported in the literature. B. O. D. levels of up to 562 and 822 mg/1 were recorded at sites 1 and 2 respectively with corresponding C. O. D. maxima of 1050 and 1800. On one occasion the S. S. levels recorded at site 2 were as high as 700 mg/l. These results contrast sharply with reports by many authors including Robeck et al (1964), Preul (1968), Polkowski and Boyle (1970) and Canter and Knox (1985) who quote effluent concentrations of to 140 mg/l B. O. D., 30 mg/l C. O. D. and 40 to 75 mg/l S. S. The large difference in the results is attributed to the poor design, installation and maintenance of the septic tank systems, in particular the absence of a baffle wall and failure to desludge the systems. At site 2 the very high maximum concentrations observed could have been caused by a process known as sludge overloading where gas production in the anaerobic sludge on the floor of the tank floats solids to the surface. In the absence of a baffle wall these solids can be washed from the tank in the effluent with serious implications for the longevity of the soil treatment system.

The results presented also demonstrate the large concentrations of phosphate and nitrogen contained in the effluents. Mean phosphate concentrations of 28.7 and 49.8 mg/l were recorded at site 1 and 2 respectively. The higher value at site 2 (maximum concentration of 63.2 mg/l PO_4 - P) is attributed to the consistently heavy use of phosphate based detergents in the household. Nitrogen in the effluent was mainly in the form of ammonia with low and variable concentrations of nitrate. These results are in agreement with previous investigations by Canter and Knox, 1985 and Patterson et al, 1971 who report that the anaerobic process within the tank converts most of the influent organic nitrogen to the ammonium form. The phosphate and ammonium concentrations recorded were significantly greater than those previously reported in the literature, which range from 15 to 20 mg/l PO_4 - P and 14 to 25 mg/l NH_4 - N (Robeck et al, 1964; Preul, 1968; Polkowski and Boyle, 1970 and Canter and Knox, 1985).

High concentrations of sodium and potassium were also recorded. The mean concentrations were 98.8 and 35.1 mg/l for sodium and potassium at site 2 and 140.0 and 42.2 at site 1. The sodium levels were greater than the potassium levels in all samples analysed. This is reflected in the calculated K/Na ratios which ranged from 15 to 0.57 at site 1 and from 0.22 to 0.93 at site 2. The mean K/Na ratios recorded were 0.31 and 0.36 at site 1 and 2 respectively. The results conflict with those of Daly and Daly (1982) who suggest that groundwaters polluted by point sources such as septic tank effluent may have high K/Na ratios. This is discussed in more detail in Chapter 3.

The poor microbiological quality of the effluents is also clearly demonstrated in the results obtained. High numbers of total and fecal Coliform and fecal Streptococci bacteria were isolated at both sites on all sampling occasions. At site 1 mean bacterial numbers of 3.0×10^6 total Coliforms, 1.3×10^6 fecal Coliforms and 2.0×10^5 fecal Streptococci (c. f. u. ' s/100ml) were isolated. Coliform bacterial numbers recorded at site 2 were significantly greater i. e. 1.6×10^7 total Coliforms and 5.1×10^6 fecal Coliforms/100 mls. However, the numbers of Streptococci bacteria isolated were markedly lower i. e. 8.8×10^4 per 100 mls. In general the numbers obtained were greater than those reported by McCoy and Ziebell (1975) who recorded 3.4×10^6 total Coliform, 4.2×10^5 fecal Coliform and 3.8×10^3 fecal Streptococci bacteria. The larger numbers can again be related to the improper design and maintenance of the system i. e. the elevated bacterial numbers may be associated with the high solids level in the effluent. The average ratio of fecal coliforms to fecal streptococci (FC : FS) calculated from the results was 6.5 at site 1 and 57.0 at site 2. The reason for this large difference is unclear but the results are in general agreement with reports by Wheater et al (1976) who observed that the FC : FS ratio for human

sewage was very variable, ranging from 2 to 34, but was consistently greater than 1. The high coefficient of variation calculated for the bacterial results at the two sites demonstrates the large fluctuation in effluent bacteriological quality compared to chemical constituent fluctuations. In general the coefficients of variation were similar at the two sites indicating that, although the effluent at site 2 was consistently poorer in microbiological quality, the fluctuation in both tanks was similar throughout the sampling period.

2:6.2 Soil Analysis

Summary results of the analysis are presented in Tables 2.5.3 to 2.5.6 (p59 to 60). Particle size analysis of the soil at site 1 (Table 2.5.3) showed it to contain 10% clay, 25% silt, 55% sand and 10% gravel sized particles which is classified as a ' sandy loam ' under the U. S. D. A. classification system. The results show little variation in the particle size distribution with increasing depth from 20 to 120 centimetres. The particle size distribution of the soil has an important influence on the physical and chemical properties of the soil and consequently on its effluent attenuation capacity, which is reflected in the results of the physical and chemical analysis of the soil (Table 2.5.4). The moderately high percentage of fine clay and silt sized particles in the soil give it a large total porosity of 31 to 42% (mean 38%). However, the individual pore size is small (capillary pores) which restricts water transmission and lowers the overall permeability of the soil (0.023 mm/second). The high organic matter content (10.2 to 20.3%), together with the high clay fraction, gives the soil a moderately high cation exchange capacity of 32 to 38 meq/100g (mean 35). This again has an important influence on its attenuation capacity.

It would appear from the results that the soil cover at the site has good attenuating properties. The low permeability would ensure adequate contact time between the effluent and the soil particles for ion exchange and adsorption reactions to proceed, with the high C. E. C. of the material providing exchange sites for the immobilisation of the effluent constituents. The medium to small pore size of the material would facilitate bacterial restriction by the process of filtration. However, because the water table at the site frequently rises into the soil treatment system, the soils attenuating ability could be greatly reduced. The high water table would affect the efficiency of the adsorption and ion exchange processes preventing oxidation and biological transformations in the treatment system. In addition bacterial immobilisation could also be greatly reduced.

The results obtained from the analysis of the soil material at site 2 are presented in Tables 2.5.5 and 2.5.6 (p60). The results show that the material is composed of 8% clay, 30% silt, 36% sand and 26% gravel sized particles, also

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classified as a ' sandy loam ' under the U. S. D. A. classification system. As was observed at site 1 there was little variation in the particle size distribution with increasing depth between 20 and 120 centimetres. The lower percentages of clay and higher gravel sized particles present at this site have a significant influence on the nature and efficiency of treatment which can take place. The lower clay content results in a lower cation exchange capacity (21 to 29 meq/100 g) while the larger percentage of coarse gravel sized particles could facilitate greater oxygen diffusion, encouraging oxidation and biological transformations in the percolating septic tank effluent. The high percentage of gravel also results in a lower total porosity of 32 to 38% and higher permeability of 0.04 mm/second.

The results indicate that the effluent from the distribution field would have adequate contact time with the soil particles to facilitate ion exchange and sorption reactions. In addition the relatively large percentage of coarse sized particles would allow oxygen diffusion between pores and permit oxidation of effluent constituents. It should also be noted that the soil in the treatment system was freely drained throughout the study period. At its highest level the water table was over 2.0 metres below the bottom of the distribution trenches (Chapter 3).

In summary, the results presented suggest that the particle size distribution and the physical and chemical composition of the soils at both sites are similar. Consequently it is the thickness of the unsaturated zone and the degree of saturation within the distribution fields which would have the dominant effect on the relative efficiency of effluent treatment within the test systems.

2:6.3 The Movement and Attenuation of Effluent Constituents in the Two Soil Treatment Systems

The following section describes and discusses the results of the movement and restriction of various effluent constituents from the two soil treatment systems investigated. Where possible the results obtained are compared to previous investigations. The section is further subdivided as follows:

- 2:6.3.1 Nitrogen
- 2:6.3.2 Phosphate
- 2:6.3.3 Coliform bacteria.

Reference should be made to Figures 2.4.1 (p43), 2.4.2 (p44) and 2.4.3 (p45) which present the location of the sampling stations at each of the test sites.

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(i) Ammonium

The results of the soil analysis for ammonium are presented in Table 2.5.7 (p61) and Figures 2.5.1 to 2.5.8 (p62 to 64). Table 2.5.7 presents summary results of the ammonium concentrations recorded at the two test sites. Very high ammonium levels were observed in the soil samples at site 1 where levels of up to 140 mg/kg were recorded within the distribution field at sampling stations 1 to 6. There is evidence of extensive movement of the ammonium with increasing depth in the soil. This may be attributed to the fact that the adsorptive capacity of the soil has been exceeded and the effluent ammonium must travel through the saturated soil to find unoccupied sites. Migration to depth is clearly demonstrated at sampling station 2 where the mean ammonium concentration recorded throughout the study was 12.6 mg/kg at 20 centimetres depth beneath the distribution pipe invert and 27.0 mg/kg at 50 centimetres depth. These results are similar to previous reports by Miller and Wolf (1975) who found that migration of ammonium through soils can occur if the effluent is continuously added to a saturated soil profile. Lateral movement of the ammonium from the soil distribution system is less marked although on occasions concentrations of up to 74.2 mg/kg were recorded at station 8, a distance of 2.0 metres downgradient of the distribution lines (compared to a background concentration of 4.8 mg/kg).

At site 2 very high levels of ammonium were recorded at stations 1, 2 and 4 within the distribution system and occasionally at station 7, a lateral distance of 2.0 metres from the end of the distribution lines. Significantly lower levels were recorded at stations 3 and 5 within the treatment system and at stations 6, 8 and 9 downgradient of the field. This trend continued for most of the research period suggesting uneven distribution of the sewage in the distribution lines, a factor which could seriously affect the longevity of the treatment system. Similar problems of non - uniform distribution of sewage in soil treatment systems have been reported by Otis et al (1974). Ammonium levels of up to 98 mg/kg at a depth of 20 centimetres below the pipe invert and 89 mg/kg at 50 centimetres depth were recorded at station 1 showing that, as at site 1, movement to depth through the soil profile was significant. This is attributed to the exhaustion of adsorption sites by excessive addition of effluent resulting in migration of the ammonium ion to unoccupied sites at greater depths. It is noted that the highest concentrations at the 50 centimetres depth were recorded at the sampling stations 1, 2 and 4. It is possible that the excessive addition of sewage along this leach line created temporary saturation of the soil beneath the trench permitting more rapid movement of the ammonium into the soil profile, excluding oxygen diffusion and preventing nitrification. With the exception of station 7, lateral movement of the

ammonium from the system was minimal. This is consistent with reports by Viraraghavan and Warnock (1976) who observed a distinct attenuation of ammonium ions in a septic tank effluent with increasing distance from the soil treatment system in fine textured unsaturated soils.

Figures 2.5.1 to 2.5.8 (p62 to 64) present the concentration of ammonium recorded at the two test sites on various sampling dates between April 1987 and June 1988. In April 1987 (Figure 2.5.1) the movement of ammonium at site 1 was more pronounced than at site 2 due to saturation of the soil profile. This could facilitate migration of the ammonium ion by reducing the efficiency of the adsorption process and prevent nitrification by reducing oxygen diffusion. Figure 2.5.2 (June 1987) shows that the levels of ammonium recorded at both sites are similar to those recorded in April. Migration of the ion to 50 centimetres depth beneath the distribution trenches was again significant. However, with the exception of station 7 at site 2, lateral movement downgradient of the treatment systems was minimal. In addition it was noted that the widespread movement of ammonium within the distribution system at site 1 was much less pronounced than had previously been observed. This may be attributed to a drying out of the soil beneath the system as the water level dropped, with a subsequent increase in the nitrification of ammonium to nitrate which was then leached from the system. It should also be noted that the total rainfall for the five day period preceeding sampling was significantly lower than that recorded on the April 1987 sampling date (Figure 2.5.42, p81). Figure 2.5.3 (August 1987) shows that the ammonium concentration recorded at site 1 was significantly higher than on the previous sampling occasion. In addition movement through the distribution field and downgradient of the system was more extensive. Very high levels of ammonium were recorded at station 8, 2.0 metres downgradient of the leach lines and migration to depth at all sampling stations was observed. It is noted in Figure 2.5.42 that there was very high rainfall (33 mm) for the five day period preceeding sampling. It is possible that this rainfall resulted in ponding and saturation of the distribution field by exceeding the infiltration capacity of the soil. Under the resulting anaerobic conditions nitrification was inhibited and the migration of ammonium from the soil system was facilitated by the lower ionic strength of the soil. It may also be possible that the percolating rainwater mobilised previously retained ammonium in the bio - zone beneath the distribution lines permitting its transport from the treatment system. There are, however, no literature reports to support this claim.

Figures 2.5.4 and 2.5.5 (p63) show similar trends in the movement of ammonium in October and December of 1987. At site 1 high levels of up to 60 mg/kg were detected throughout the distribution system. However, lateral movement in the direction of stations 7, 8 and 9 was minimal. At site 2 high levels were again noted at

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stations 1 and 2 within the treatment system. Levels well above background were also recorded at station 7 downgradient of the system. Figure 2.5.6 (February 1988, p63) shows increased concentrations in all sampling stations at site 1 over those recorded on the previous sampling occasions. This increase may again be related to increased rainfall prior to sampling (Figure 2.5.42, p81). In June 1988 (Figure 2.5.8, p64) migration of the ammonium ion was again very significant at site 2. Levels of greater than 25 mg/kg were recorded at station 7 compared to background levels of 3 mg/kg or less.

In summary, the migration of ammonium beneath the soil treatment systems was found to be very significant on all sampling occasions. Very high concentrations of ammonium were recorded at a depth of 50 centimetres below the bottom of the distribution trenches. The movement of the ion at site 2 was concentrated along the centre trench in the system suggesting uneven distribution of the sewage from the distribution box. At site 1 high levels were evenly distributed throughout the system. Lateral migration of the ion downgradient of the test systems was demonstrated on a number of occasions. On two of the sampling dates high levels of ammonium were recorded at site 1, 2.0 metres downgradient of the distribution system (sampling station 8). On both of these occasions this may have been related to increased rainfall events prior to sampling, reducing the efficiency of the adsorption process in the soil. Similarily at site 2 high levels of the ion were recorded at sampling station 7, a lateral distance of 2.0 metres downgradient of the system. It would appear that the overloading of the centre trench resulted in locally high moisture contents in the soil profile with corresponding high flow rates. This result is in agreement with reports by Otis et al, (1974) who suggested that uneven distribution in a septic tank treatment system can have a serious effect on treatment efficiency by reducing the filtration, adsorption, oxidation and ion exchange mechanisms which restrict of the movement of effluent constituents.

(ii) Nitrate

The results of the soil analysis for nitrate are presented in Table 2.5.8 (p65) and Figures 2.5.9 to 2.5.16 (p66 to 68). Summary results of the nitrate concentrations recorded in the soil samples are presented in Table 2.5.8. At site 1 high nitrate concentrations above background levels were recorded in all the sampling stations 1 to 9 at both sampling depths. The highest level recorded was 42.5 mg/kg at station 2 at a depth of 50 centimetres. The mean concentrations of nitrate detected at the various sampling stations were consistently greater at 50 centimetres depth, demonstrating the mobility of the nitrate ion. Similarly, at site 2, concentrations above background were recorded at all of the sampling stations 1 to 9. As at site 1 highest levels were recorded

at 50 centimetres depth. These results are consistent with observations by Miller and Wolf (1975), Bourma (1979), Canter and Knox (1985) and Lewis et al (1982) all of whom report mobility of nitrate in soil profiles.

In general the concentration of nitrate recorded at site 2 was greater than at site 1, as was the increase in concentration with increasing depth from 20 to 50 centimetres. This difference is attributed to the different moisture status of the soil at the two sites i. e. at site 2 the soil beneath the distribution field remained freely drained throughout the study facilitating the nitrification of effluent ammonium.

Lateral movement of nitrate from the two treatment systems was also demonstrated. At site 1 the mean concentration of nitrate recorded at sampling station 8 over the entire sampling period was only marginally above background levels. However, during drier periods when the moisture content of the soil was at a minimum levels of up to 5.0 mg/kg above background levels were recorded downgradient of the treatment system at a depth of 50 centimetres. This result indicates that migration of nitrate is more prevalent during drier periods. Lateral movement of nitrate downgradient of the treatment was greater at site 2. The results presented clearly demonstrate that the mean and maximum values recorded at stations 6, 7, 8 and 9 downgradient of the system were significantly higher than background levels. Mean concentrations of up to 3 mg/kg above background were recorded at station 9, a distance of 4.0 metres downgradient of the distribution field.

The levels of nitrate detected at the two test sites on the various sampling occasions are presented graphically in Figures 2.5.9 to 2.5.16 (p66 to 68). Figure 2.5.9 shows that significantly higher concentrations of nitrate were recorded at site 2 in April 1987. This is attributed to the high water table and saturated conditions in the distribution field at site 1 which inhibited nitrification of effluent ammonium. The result is confirmed by the high levels of ammonium recorded on the same sampling date (Figure 2.5.1). Significant lateral movement of the ion from the soil treatment system occurred at site 2. Levels well above background were recorded in sampling stations 6, 7, 8 and 9 at distances of between 2.0 and 4.0 metres downgradient of the distribution trench.

Similar trends were observed on the following sampling date (June 1987). However, nitrate concentrations were significantly greater than those recorded on the previous sampling occasion. This is attributed to a continued drying out of the treatment system permitting increased oxidation of ammonium with subsequent leaching of nitrate. A notable increase in the nitrate concentrations recorded at site 1 is also demonstrated and was accompanied by a corresponding decrease in the ammonium levels (Figure 2.5.2, p62). This result is consistent with reports by Raraghavan and Warnock (1976) who found that the concentration of nitrate increased

and that of ammonium decreased downgradient of a septic tank soil treatment system during drier periods. Significantly lower concentrations were recorded during the period October 1987 to February 1988 (Figures 2.5.12 to 2.5.14, p67). It is likely that nitrification of the ammonium in the treatment systems was prevented due to an increase in the soil moisture content thereby limiting the availability of free oxygen.

Figures 2.5.15 and 2.5.16 (p68) show an increase in the levels recorded at both sites. On both sampling occasions (April 1988 and June 1988) vertical and lateral migration of the ion was very significant. At site 2 levels markedly above background were recorded at stations 1, 2, 4, 7 and 9. This increase may be due to nitrification of previously adsorbed ammonia beneath the centre distribution leach line which, as described earlier, appeared to receive most of the effluent from the system. The high concentrations of nitrate recorded at station 9, 4.0 metres downgradient of the system, is significant (up to 9.5 mg/kg compared to a background concentration of 3 mg/kg) and again demonstrates the mobility of the nitrate ion in the subsurface particularly during the summer months.

In summary, the results show that the nitrate ion is migrating from the soil treatment systems at both sites. Vertical movement of the ion was very considerable demonstrating the ability of the ion to migrate through both saturated and unsaturated soil profiles. Although no samples were obtained from depths greater than 50 centimetres below the invert of the distribution lines it is possible that nitrate is moving to greater depths and reaching groundwater. This was confirmed when the groundwater beneath the distribution field at site 2 was analysed (Chapter 3). The highest levels of nitrate were recorded in the soils during the drier periods (i. e. April to June) when nitrification of effluent ammonium was maximised. Concentrations were lower during the wetter winter months. Lateral movement of the ion downgradient of the systems was also demonstrated especially at site 2 where levels of up to 9.5 mg/kg were recorded at a depth of 90 centimetres and a lateral distance of 4.0 metres downgradient of the soil system.

(iii) Soil Piezometers

The results of the movement of nitrogen from the soil distribution system was further investigated by analysis of the six shallow sampling piezometers installed at site 1. Figure 2.4.1(p43) shows the location of the sampling piezometers in relation to the soil sampling stations. The result of the analysis is presented in Table 2.5.12 (p82). The piezometers were sampled on three of the eight sampling occasions (June 1987, February 1988 and June 1988). High levels of ammonium were recorded in piezometers B and C within the distribution system with maximum concentrations of 6.8 and 9.7 mg/l NH_3 - N recorded in February 1988. This

represents a 75.4% and 65.8% reduction in the effluent ammonium concentration and demonstrates a significant removal within the treatment system. Maximum ammonium concentrations in piezometers D and E, 2.0 metres downgradient of the system, were 2.90 and 2.10 mg/l respectively compared to a background level of 0.05 mg/l in piezometer A. The concentration recorded in piezometer F, a lateral distance of 4.0 metres downgradient of the system, was also found to be 2.10 mg/l. This represents a 92.4% reduction in the concentration recorded in the effluent introduced into the system. These results are similar to reports by Viraraghavan and Warnock (1976) who found that the ammonium concentration beneath a soil treatment system was reduced from 40 mg/l to 5 mg/l less than 3.0 metres from the system i. e. an 87.5 % reduction.

The nitrate concentrations recorded in the piezometers show that lowest levels were recorded during the February 1988 sampling date when the moisture content in the treatment system was high, preventing the nitrification of the effluent ammonium. Higher levels of up to 6.10 mg/l were recorded in piezometer B during the summer sampling dates (compared to a background concentration of 3.10 mg/l) when water levels were reduced allowing oxygen diffusion and nitrification to proceed. Although there was some evidence of lateral migration of nitrate from the treatment system during the drier periods, the extent of this movement appeared to be minimal. Maximum concentrations of 5.1 mg/l were recorded in piezometer F in June 1987 compared to a background level of 3.1 mg/l. These results also indicate that the migration of nitrate from the distribution system is limited to the summer months and that the observed increase in concentration during these periods is accompanied by a corresponding decrease in ammonia. This is consistent with reports by Viraraghavan and Warnock (1976) described earlier.

2:6.3.2 Phosphate

The results of the analysis of the soil samples for phosphate are presented in Table 2.5.9 (p69) and Figures 2.5.17 to 2.5.24 (p70 to 72). Migration of the phosphate ion to depth beneath the soil treatment systems was very significant. At site 1 the mean concentration of phosphate recorded at station 1 over the entire sampling period was 7.3 mg/kg at a depth of 20 centimetres below the invert of the distribution pipe, and 10.5 mg/kg at 50 centimetres depth. Similarily at site 2 the mean concentration recorded at a depth of 20 centimetres (Station 1) was 25.1 mg/kg and 22.3 mg/kg at 50 centimetres depth. These results are contrary to reports by Patterson et al (1971), Jones and Lee (1979) and Canter and Knox (1985) who found that phosphate movement in the soil profile is minimal due to adsorption and precipitation reactions. However, the results are consistent with observations by Miller and Wolf (1975) and Sawhrey and Starr (1977) who report that continuous

ddition of phosphate rich effluent to soils can seriously reduce the sorptive capacity of
ie material and result in the migration of the ion to greater depths. Lateral movement
f the phosphate from the distribution fields was also very significant. At site 1 high
oncentrations of 8.6 mg/kg were recorded at sampling station 9, 4.0 metres
owngradient of the system, compared to a corresponding background level of 2.2
ig/kg. Similar high levels were recorded at site 2. A concentration of 10.5 mg/kg
hosphate was detected at station 9 with maximum concentrations recorded along the
entre line in the distribution field. It is likely that the overloading of this leach line
sulted in an increase in the moisture content beneath the system, reducing the
fficiency of the adsorption and precipitation processes and promoting the transport of
ie phosphate ion downgradient of the treatment system in the direction of sampling
ation 9. The migration of phosphate downgradient of the system was greatest during
e wetter periods.

Figures 2.5.17 to 2.5.19 (p70) present the concentrations of phosphate
corded on the first three sampling dates (April 1987, June 1987 and August 1987).
he graphs show significant migration of phosphate to 50 centimetres depth (sample B)
all sampling stations at both test sites. The preferential migration of the ions along
entre trench in the distribution field at site 2 is also demonstrated and there is
idence of lateral movement of the ion to sampling stations 2 and 4. These results are
agreement with observations by Dole (1986) that phosphate can migrate laterally
om a septic tank soil treatment system to contaminate groundwater downgradient of
e system. On the following three sampling occasions (October 1987 to
ebruary 1988) the concentration of phosphate recorded at both sites was significantly
gher. This is attributed to a decrease in the efficiency of the soil colloids in restricting
e movement of the ion during wetter periods when the moisture content within the
stribution system was higher (Figure 2.5.42, p81). Migration of the ion from the
stribution field at site 2 was very significant on the December sampling date when the
ncentration of phosphate recorded at station 7 was over 30 mg/kg (at 50 centimetres
pth) compared to a corresponding background level of 3 mg/kg.

Figure 2.5.23 (p72) presents the phosphate concentrations recorded at the two
sites on the April 1988 sampling date. Concentrations recorded at site 1 were high
th both vertical and lateral movement of the ion from the treatment system taking
ace. In contrast the levels recorded at site 2 were markedly lower than those observed
the previous sampling occasions. The reason for this reduction is unclear. It may
due to crystallisation of adsorbed phosphate into less soluble compounds, thereby
nerating more exchange sites and restoring the sorptive capacity. This process has
eviously been described by Miller and Wolf (1975) and may occur if the discharge of
luent to the system is discontinued for a number of months. Throughout the study

period discharge of the effluent continued as usual. However the drying out of the distribution system during the months February to April may have been sufficient to stimulate this process. Figure 2.5.24 (p72) shows that increased concentrations were again recorded on the following sampling date with extensive migration of the ion along the centre line of the distribution field in the direction of sampling station 7, 2.0 metres downgradient of the system. The concentration recorded at 50 centimetres depth was over 22 mg/kg compared to a corresponding background level of 2.5 mg/kg.

Table 2.5.12 (p82) presents the results of the analysis of the sampling piezometers at site 1. Again high phosphate concentrations were recorded on the February 1988 sampling date when concentrations of 4.6 and 2.9 mg/l were obtained in the piezometers B and C respectively compared to a background concentration (piezometer A) of 0.05 mg/l. These values represent an 83.9 and 89.5% reduction in the concentration of phosphate which was introduced to the soil in the effluent. The concentrations recorded in piezometers C and D a lateral distance of 2.0 metres downgradient of the system were 1.7 and 0.9 indicating a distinct attenuation of phosphate with increasing distance from the system. This decrease was further emphasised on analysis of piezometer F where a concentration of 0.6 mg/1 was recorded, representing a 97.9% decrease in effluent phosphate concentration over a distance of 4.0 metres from the distribution field. On the other sampling dates the concentrations were considerably less, ranging from 3.0 mg/l in piezometer B within the distribution field to 0.00 at piezometer F. The results obtained confirm previous observations above that the phosphate from the treatment system is migrating downgradient of the field in the direction of sampling station 9 (piezometer F) and that maximum migration of phosphate is occurring during wetter periods. The results, however, also demonstrate that the percentage attenuation of the ion over a distance of 4.0 metres from the system is high, up to 97.9%.

In summary, the results presented demonstrate that the phosphate ion migrated from the soil treatment systems at both sites and that attenuation of the ion within the system was incomplete. High concentrations of phosphate were recorded at depths of 20 and 50 centimetres beneath the invert of the distribution lines. This is attributed to the inability of the treatment systems to restrict the migration of the ion by precipitation or sorption reactions. Although no samples were obtained at depths greater than 50 centimetres it is possible that the phosphate migrated to greater depths to unoccupied sorption sites. Lateral movement of the phosphate from the treatment systems was also recorded. At both sites high concentrations were recorded at distances of 4.0 metres downgradient of the test systems. The results also suggest that the migration of phosphate in both systems was greatest during the wetter periods when the attenuation capacity of the soils may have been reduced.

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2:6.3.3 Coliform Bacteria

The results of the analysis of the soil samples for coliform bacteria are presented in Tables 2.5.10 (p73) and 2.5.11 (p77) and in Figures 2.5.25 to 2.5.40 (p74 to 80). Similar patterns and trends were observed for both the total and fecal bacteria. Consequently this discussion is restricted to the movement and attenuation of the fecal Coliform bacteria within the two treatment systems.

Table 2.5.11 presents summary results of the numbers of fecal Coliform bacteria (c. f. u. ' s/100 g soil) isolated from the various sampling stations at the two test sites. High numbers of the organisms were recorded within the distribution system at site 1 (sampling stations 1 to 6). Maximum numbers were recorded at station 3 where 1.5×10^4 bacteria (c. f. u. ' s/100 g) were isolated at 20 centimetres depth beneath the distribution pipe invert and 1.2×10^4 c. f. u. ' s/100 g at a depth of 50 centimetres. This demonstrates a marked migration of the organisms vertically through the soil profile and is attributed to the loss of efficiency in soil retention properties (i. e. filtration and adsorption) due to saturation by percolating effluent. High bacterial counts were also recorded at the other sampling stations throughout the field where numbers recorded were typically in the region of 1.0×10^3 c. f. u. ' s/100 g. At all sampling stations there is evidence of extensive migration of fecal coliform bacteria to 50 centimetres depth.

The numbers isolated at sampling stations 7, 8 and 9 were considerably lower. However, on occasions bacterial numbers at all three stations were greater than those recorded at the control station 10. There is evidence of a decrease in the numbers isolated with increasing distance from the treatment system. For example the maximum number of bacteria isolated from the sampling stations 7 and 8 at both depths was between 4.6 and 7.0×10^2 c. f. u. ' s/100 g compared to numbers of 1.8 to 4.1×10^2 at station 9. This result, however, demonstrates the ability of fecal coliform bacteria to migrate laterally from a soil distribution system. Previous reports by Hagedorn et al (1978), Viraraghavan (1978) and Stewart and Reneau (1981) have all demonstrated the ability of coliform bacteria to migrate through significant thicknesses of saturated soil with subsequent threat to groundwater quality. Although the numbers of bacteria isolated in the soil samples appear to be high they represent a very considerable reduction in the numbers which were introduced to the system in the septic tank effluent. The maximum numbers recorded at sampling station 3 (described above) represent a 98.8 and 99.0% reduction in the concentrations of bacteria added to the system at depths of 20 and 50 centimetres respectively. This large reduction is attributed to the removal of large numbers of effluent bacteria in the biological matt (biocrust) beneath the distribution tiles. These figures are slightly lower than those reported by McCoy and Ziebell (1975) who observed reductions of up to

99.9% in the original coliform population over a distance of less than 30 centimetres beneath the distribution trench. Percentage reductions at sampling station 9, 4.0 metres downgradient of the system, were calculated as 99.98 and 99.96 at 20 and 50 centimetres respectively indicating the greater attenuation of the bacteria with increasing distance from the system.

Results of analysis at site 2 show that the highest bacterial numbers were again isolated along the centre line of the distribution trench, giving further evidence of uneven distribution of effluent through the treatment system. Maximum numbers were recorded at station 1 where 3.2×10^4 and 4.0×10^4 bacteria/10 ml were isolated at 20 and 50 centimetres depths respectively. These results again demonstrate significant migration of the fecal bacteria vertically through the soil profile. Similar high numbers were isolated at stations 2, 4, 5 and 7 where migration to depth was considerable. Lateral movement of the organisms to station 7, 2.0 metres downgradient of the system was also demonstrated. Numbers of up to 8×10^3 and 1×10^4 c. f. u. ' s/100 g were isolated at 20 and 50 centimetres depth compared to a corresponding background level of 10 c. f. u. ' s/100 g at both sampling depths. The movement of bacteria vertically and laterally from the treatment system is attributed to the overloading of the centre leach line, with the subsequent temporary saturation of the system. This would reduce the efficiency of the filtration and adsorption mechanisms which are crucial to the retention of the effluent bacteria. Although migration of the organisms to sampling station 7 was very significant the numbers isolated at sampling station 9, 4.0 metres downgradient of the system, were reduced to near background levels, indicating that bacterial removal effiency improved with increasing distance from the overloaded area. As was described at site 1 the percentage reduction in the original number of fecal coliform bacteria introduced to the system appears to be very high. Maximum values were recorded at sampling station 1, representing a 99.3 and 99.2% reduction in the numbers of organisms introduced to the system. Furthermore the percentage reduction in numbers at station 9, 4.0 metres downgradient of the system, was calculated to be greater than 99.99% indicating that lateral migration from the treatment system was minimal.

It should be noted that although the percentage reduction in the numbers of fecal bacteria discharged to the system in the effluent was large, the reduction with depth beneath the soil treatment system was much less significant. The migration of such large numbers of bacteria to depth beneath a soil treatment system is of some concern. These organisms can survive for a considerable period of time (2:6.4) and can subsequently be mobilised by heavy rainfall to migrate through cracks or joints in the subsurface and contaminate groundwater supplies. This occurrence has previously been reported by a number of authors, including Patterson et al (1971),

Hagedorn et al (1978), Bitton and Gerba (1984) and Sinton (1986).

The numbers of fecal coliform bacteria isolated at the two sites on the various sampling dates are given in Figures 2.5.33 to 2.5.40 (p78 to 80). The graphs show that the migration of fecal coliform bacteria from the treatment system at site 1 was more pronounced in April 1987 (Figure 2.5.33), August 1987 (Figure 2.5.35) and February 1988 (Figure 2.5.38). On all of these sampling occasions large numbers of organisms were isolated at one or more of the sampling stations 7, 8 and 9 downgradient of the test system. In addition the numbers isolated were significantly lower than on the other sampling dates. It is noted from Figure 2.5.42 (p81) that the total rainfall for the five day period before the three sampling dates was significantly greater than that recorded prior to the other sampling occasions. It is possible that this increased rainfall resulted in the development of saturated conditions within the soil treatment system thereby promoting the movement of the bacteria from the system. The increased mobilisation of fecal bacteria from soil treatment systems after periods of heavy or prolonged rainfall is well documented in the literature and is described in more detail in Chapter 3. The high bacterial counts recorded at station 8 in April 1988 is also attributed to the mobilisation of previously fixed bacteria by percolating rainwater i. e. Figure 2.5.41 (p81) shows that the total monthly rainfall for the month of March was high (over 130 mm).

The graphs again illustrate the preferential movement of effluent along the centre tile in the distribution system at site 2. Although significant numbers of bacteria were isolated at sampling stations downgradient of the system on a number of occasions (Figures 2.5.35, p78; 2.5.36, p79 and 2.5.39, p80) movement to station 9, 4.0 metres downgradient of the system, was minimal.

The results of the bacteriological analysis of the sampling piezometers at site 1 are presented in Table 2.5.12 (p82). High numbers of total and fecal Coliform bacteria were isolated from piezometers B and C within the distribution system. As was noted with the other test parameters maximum numbers were isolated from all piezometers in February 1988. This result is consistent with literature reports which have documented that bacterial contamination of groundwater sources is maximised during wetter periods or after heavy or prolonged rainfall (Patterson et al, 1971; Hagedorn et al, 1978; Bitton and Gerba, 1984 and Sinton, 1986).

The highest numbers isolated were in piezometer B in February 1988 when 7.2×10^3 total Coliform and 3.0×10^3 fecal Coliform bacteria/100 ml were recorded. These figures represent a 99.7% reduction in the numbers of bacteria introduced to the system in the septic tank effluent and confirm observations above that the majority of the bacteria in the percolating effluent are being removed in the biological zone beneath the distribution trenches. The results also demonstrate that lateral migration of the

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bacteria from the treatment system in February 1988 was very significant. Elevated bacterial numbers of 4.0×10^3 and 1.8×10^3 were recorded in piezometer F, a lateral distance of 4.0 metres downgradient from the end of the distribution trenches. This result provides further evidence of the mobility of the organisms in saturated soil treatment systems.

The reason for the consistently high numbers of Coliform bacteria isolated in the control piezometer A is unclear. It may be due to the development of a wastewater mound beneath the distribution trench which resulted in the effluent bacteria migrating upgradient of the treatment system. This has previously been documented by Sinton (1986).

2:6.4 The Survival of *Escherichia coli* in a Range of Soil Types

This section describes and discusses the results of the bacterial survival experiments. The results of the physical and chemical analysis of the soil used in the study are presented in Tables 2.5.13 and 2.5.14 (p83). Particle size analysis of the loam soil showed it to be composed of 17% clay, 38% silt, 33% sand and 12% gravel (classified as a loam under the U. S. D. A. classification system). Analysis of the sand soil showed it to contain 2% clay, 12% silt, 69% sand and 17% gravel sized particles (classified as a sand under the U. S. D. A. system). No textural analysis was carried out on the peaty soil. As was described in 2:2.3.3 the dominant factors affecting the survival of enteric bacteria in soils are pH, exposure to sunlight, moisture content, temperature and availability of organic matter. Since moisture content and sunlight were controlled throughout the study and the temperature in all three leaching barrels was similar, the other factors were more significant in determining the relative survival rates of the bacteria. Table 2.5.14 shows the results of the chemical analysis of the three soils. In the peaty soil for example the high organic matter content would be expected to favour bacterial survival. However, this would be offset by the adverse affect of the low pH value recorded (4.79). Thus die - off in the peat soil could be rapid and significant. Survival in the sandy soil could be limited by the poor moisture holding capacity and by the low organic matter content recorded (7.4%). Similarily bacteria might be expected to live longest in the loam soil with its near neutral pH, high organic matter content and moderate water holding capacity combining to optimise survival.

The results of the analysis of the soil samples for *Escherichia coli* bacteria at various intervals after the addition of the culture to the soils are presented in Table 2.5.15 (p84). The results show that significant numbers of bacteria were isolated after 44 days sampling in all three soil types. The die - off in all three soils followed a similar pattern. The largest decrease in bacterial numbers occurred between 10 and 13

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days after innoculation. This increased mortality may have been due to a reduction in the availability of easily obtainable nutrients within the soils. Alternatively it may have been due to a decrease in the moisture status in the leaching containers although this was not registered on the soil tensiometers. Although die - off in all three soils followed a similar pattern the largest numbers were isolated in the peat soil, with the lowest numbers recorded in the sandy soil. This is attributed to the efficiency of the peaty soil in restricting the movement of bacteria by adsorption (i. e. its high cation exchange capacity) and filtration. Bacterial restriction in the sandy soil was significantly reduced due to the low cation exchange capacity and large pore size of the material which combined to minimise adsorption and filtration machanisms.

In order to demonstrate trends in the die - off patterns of the organisms and to calculate survival times, a plot of the numbers of bacteria (c. f. u. ' s/g soil) against time was prepared. The resulting plots for all three soil types appeared to follow an exponential decay pattern. A second set of plots of the log of bacterial numbers isolated versus time was prepared. This yielded curves which appeared to demonstrate a linear relationship. Exponential curves were fitted to the three graphs yielding straight lines. The goodness of fit of the resulting plots was tested by calculating Pearson's correlation coefficient (r) and coefficient of determination (r^2). High linear correlation coefficients of 0.96, 0.96 and 0.95 were calculated for the loam, sand and peat soil plots respectively (Figures 2.5.43 to 2.5.45, p85). Clearly in all three soils die - off followed an exponential decay pattern. The significance of the correlation for each of the regression lines was tested using a t - statistic (Table 2.5.16, p86). The results show that the correlation coefficients calculated from the regression lines for each of the soil types were highly significant at both levels (P = 0.01, P = 0.05).

From the regression line curves T90 and T99 values (the time taken for 90 and 99% die - off to occur) were calculated. These results are presented in Table 2.6.1 (p105). The results obtained conflict with previous literature reports and general expectations given the nature of the soils. The *Escherichia coli* bacteria were shown to survive longer in the peat (T99 of 31 days) than in the loam (T99 of 28 days) despite the fact that the the acidic nature of peat has supposed anti - microbial properties. Furthermore maximum survival was recorded in the sandy soil (T99 of 36 days), which again is a reversal of the expected trend.

The results obtained indicate that the effluent in which the bacteria were suspended provided a buffering capacity for the organisms within the soils, permitting greater survival times by minimising the adverse effects of the soil environment. The large concentration of available nutrients and the neutral to slightly alkaline pH of the effluent may have provided a favourable micro - environment for the survival of the bacteria and consequently the effect of soil type on the survival of the bacteria was not

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significant. A previous investigation, cited by Lewis et al (1982), reported similar results. They observed that the survival of *Escherichia coli* was doubled in soils amended with 50 mm of cow manure per week. In this study, however, T_{90} values of 8.5 days were calculated which are considerably lower than those reported above. Miller and Wolf (1975) report survival times of enteric bacteria in peat soils as low as 24 hours. The results are, however, comparable to reports by Kowal (1983) who estimated that a common maximum survival time of enteric bacteria in soils was in the region of two months. Similarily Bitton and Gerba (1984) concluded that two to three months is sufficient for the reduction of enteric bacterial to negligible numbers once released to soils.

The implications of the results obtained are significant. Bacteria released to the soil from septic tank systems may survive for a considerable period of time. This may result in a build up of viable organisms in the soil beneath the soil distribution system which may be flushed into groundwater after periods of heavy or prolonged rainfall with attendant human health hazards.

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Table 2.6.1

Estimated T 90 and T 99 Survival Times for

Escherichia coli Bacteria in the Three Soil Types

SOIL TYPE	Survival Time (days)	
	T 90	T 99
Loam	14	28
Sand	18	36
Peat	15	31

2:7 Conclusions

The main conclusions of the study are as follows:

(i) The effluent from the two septic tanks investigated was of very poor chemical and microbiological quality containing high concentrations of B. O. D., C. O. D. and S. S. in addition to elevated levels of nitrogen and phosphorous. Large numbers of fecal bacteria were also isolated from the wastewater. The high concentrations of pollutants recorded in the effluents were significantly greater than have previously been reported in the literature. This is attributed to the improper design, construction and maintenance of the test systems. The quality of the effluent was such that it could not be discharged directly to waterbodies without further treatment.

(ii) The attenuation of septic tank effluent in the soil treatment systems at the two test sites was shown to be incomplete. Migration of effluent nutrients and microbial constituents from the treatment system was clearly demonstrated. Movement of nitrate, ammonium, phosphate and fecal bacteria to a depth of 50 centimetres beneath the invert of the distribution trench tiles occurred on all sampling dates. The lateral migration of pollutants from the treatment systems was less pronounced, although on a number of occasions the concentrations recorded at distances of 2.0 and 4.0 metres downgradient of the test systems were well in excess of background levels. There is evidence that the degree and extent of effluent migration from the treatment systems increases during or after periods of increased rainfall when the attenuating properties of the soil have been reduced as a result of saturation of the soil profiles.

The failure of the treatment system at site 1 to effectively immobilise the septic tank effluent is attributed to the consistently high water table, giving rise to saturated conditions in the treatment system. This had the effect of reducing the efficiency of the filtration, adsorption, oxidation and ion exchange mechanisms which restrict the movement of effluent constituents. Conversely, the treatment system at site 2 remained freely drained throughout the study. It is considered that the non - uniform distribution of the effluent in this distribution field resulted in the overloading of the centre leach line. This in turn led to the exhaustion of the soil attenuating properties in this area and the development of temporary saturated conditions beneath the trench with resulting adverse effect on treatment efficiency.

(iii) The migration of ammonium beneath the soil treatment systems at the two test sites was very significant. This is attributed to the saturation and overloading of the soil distribution system with a resulting decrease in adsorption efficiency and restriction of the nitrification process. There is some evidence that this migration to depth increased during wetter periods. Lateral movement of ammonium was noted on a number of occasions, especially during wetter periods when concentrations above background levels were recorded at distances of between 2.0 and 4.0 metres downgradient of the treatment systems.

The analysis of samples from the piezometers at site 1 showed a distinct attenuation of ammonium with increasing distance from the distribution trenches. The concentration of ammonium introduced to the system in the effluent was reduced by 92.4 % over a lateral distance of 4.0 metres downgradient of the system.

(iv) The results obtained show a significant migration of nitrate from the treatment systems at both of the test sites. The highest concentration and most extensive lateral movement of the ion was recorded during the drier periods (April, June and August sampling dates) when drying out of the soil in the treatment systems permitted oxygen diffusion and nitrification of ammonium to occur. Conversely the lowest concentrations were recorded during the wetter winter months (October to February sampling dates) indicating that nitrification was inhibited due to an increase in soil moisture status. The increase in nitrate concentration during the summer months was accompanied by a corresponding decrease in the ammonium levels, particularly at site 1.

It is noted that the mean nitrate concentrations recorded were consistently greater at a depth of 50 centimetres, a result which demonstrates the mobility of the nitrate ion in both saturated and unsaturated soil profiles. Lateral movement of the ion from the treatment systems was less pronounced although at site 2 concentrations of up to 9.5 mg/kg were recorded at a depth of 90 centimetres, 4.0 metres downgradient of the distribution field

(v) Attenuation of the phosphate ion within the treatment systems at both sites was incomplete. High concentrations of phosphate were detected at a depth of 50 centimetres beneath the invert of the distribution lines at both of the test sites. This is attributed to a decrease in efficiency of the soils sorption properties under saturated or overloaded site conditions. Although no samples were taken at depths greater than 50 centimetres it is likely that the ion migrated to unoccupied sorption sites at greater depths. There was also evidence of lateral migration of the ion from the treatment systems. On a number of occasions the phosphate concentrations detected at a distance

of 4.0 metres downgradient of both treatment systems were significantly greater than background levels. Maximum migration of the ion from both systems occurred during or after wetter periods.

The results of the piezometer analysis at site 1 confirm the lateral migration of the ion to a distance of 4.0 metres downgradient of the test system and that maximum movement occurred during wetter periods. A distinct attenuation of phosphate with increasing distance from the treatment system was also recorded i. e. a 97.9 % reduction in the effluent phosphate concentraton was observed over a distance of 4.0 metres from the treatment system.

(vi) The bacterial counts recorded in the soil samples at both test sites represent a large reduction in the numbers of organisms introduced to the treatment systems in the septic tank effluents. The maximum numbers recorded at a depth of 50 centimetres beneath the treatment systems represent a reduction of between 99.0 and 99.2 % in the numbers originally introduced to the system. This large reduction is attributed to the removal of large numbers of effluent bacteria in the bio - zone (biocrust) beneath the distribution tiles.

Although the percentage reduction in the effluent fecal bacteria was very large the reduction in numbers with depth was much less significant. The results obtained at both of the test sites demonstrate a marked migration of Coliform bacteria vertically through the soil profiles to a depth of 50 centimetres beneath the distribution trenches. Increased mobilisation of the organisms after periods of heavy or prolonged rainfall was also noted. This is attributed to a loss in efficiency of soil retention properties (i. e. filtration and adsorption) due to the saturation (site 1) and overloading (site 2) of the treatment systems.

Lateral migration of the organisms from the soil treatment system at site 1 was significant especially during wetter periods. At site 2 lateral movement was less pronounced indicating a marked improvement in bacterial removal efficiency with increasing distance from the overloaded area.

(vii) The survival of *Escherichia coli* bacteria in a range of soil types was demonstrated to be very significant. Large numbers of up to 2.6×10^3 viable bacteria (c. f. u. ' s/g soil) were isolated from the soil samples after a 44 day sampling period. The principle findings of the experiments were as follows:

(a) The die - off of the bacteria in all three soil types followed an exponential decay pattern

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(b) The calculated survival times for the organisms were markedly greater than those previously reported. The T99 values calculated for the organisms in each of the three soil types displayed a reversal of the expected trend i. e. longest survival time was observed in the sandy soil and lowest in the loamy soil. The calculated T99 survival times were:

Sand	36 days
Peat	31 days
Loam	28 days

(c) Soil type did not have a significant effect on the survival of the bacteria. It would appear that the septic tank effluent in which the bacteria were suspended provided a buffering capacity for the organisms within the soils permitting greater survival times and minimising the adverse effects of the soil environment

(d) The anti - microbial properties of peaty soils were not demonstrated in the results obtained.

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CHAPTER 3
GROUNDWATER CONTAMINATION DOWNGRADIENT OF SEPTIC TANK TREATMENT SYSTEMS

3:1 *Introduction*

Groundwater is one of the earth's most widely distributed natural resources. It exists wherever the rate of water infiltration beneath the surface is sufficient to saturate a significant thickness of bedrock. It becomes a useful resource when the saturated bedrock is permeable enough to transmit appreciable quantities of water to wells or springs. Groundwater represents a major proportion of the earth's useable water resources. Excluding the water in icecaps and glaciers it accounts for 97% of all non-marine water and in some areas it is the only source of water available.

About 25% of the total water usage in Ireland is supplied by groundwater sources (Daly, 1985). A similar figure is quoted for the United States where groundwater accounts for greater than 50% of all drinking water (Bitton and Gerba, 1984). In other countries such as Germany groundwater sources are used to a much greater extent, providing up to 80% of the total supply. Dependence on groundwater can be more marked at a regional level. In parts of south - east England and the Irish midlands groundwater provides greater than 90% of the total water requirements. It is envisaged that the growing demand for cleaner water supplies will result in a significant increase in the exploitation of groundwater resources.

Groundwater has traditionally been regarded as a pure source of water because of the filtering and cleansing processes which take place as it percolates through the soil and subsurface material. However, pollution of groundwater sources does occur. In 1977 a report by the United States Environmental Protection Agency (E. P. A.) suggested that zones of polluted groundwater are extremely difficult to detect because of the complexity of and large variation in subsurface water systems. Aquifer contamination has often only been discovered after a water supply has been affected and in many cases it is then too late for remedial action. Consequently in protecting groundwater resources emphasis must be placed on the prevention of pollution rather than subsequent rehabilitation. The problems encountered in protecting groundwater are highlighted by Freeze and Cherry (1979) who report that personnel involved in the protection of groundwater must ' identify the areas and mechanisms by which pollutants can enter groundwater flow systems and develop reliable predictions of the transport of contaminants within the flow system '.

Subsurface disposal of domestic sewage by on - site disposal facilities, in particular septic tank systems, has been identified as one of the main sources of groundwater pollution (Geraghty and Miller, 1978). Many public health workers feel that the most critical effect of septic tank systems is the contamination of private water wells. Outbreaks of typhoid fever, infectious hepatitis, gastrointestinal infections and infantile methaemoglobinemia have all been linked to malfunctioning septic tank

systems (Patterson et al, 1971 and Hutton and Gerba, 1984). There is an urgent need for strict regulations and guidelines on the use of septic tank systems in areas dependent on groundwater for domestic supply. These regulations must be based on the results of accurate scientific investigations into the movement and attenuation of the effluent through the various overburden materials. Research to date has mainly been focused on single site investigations often involving just one effluent constituent, despite the fact that septic tank effluent is a complex heterogeneous mixture of chemical and biological constituents with a variety of migration patterns in different site, overburden and hydrogeological situations.

This section of the study examines the migration of a large number of the chemical and biological constituents of the effluent through a range of soil/overburden materials and in a number of hydrogeological settings. The aim of the study is to assess the degree and mechanisms of effluent movement in different site situations thus providing the necessary information to develop reliable predictions of effluent transport to groundwater systems. The Chapter is divided into subsections as follows:

3:1 Introduction

3:2 Literature Review: This section reviews the relevant literature on the pollution of groundwater sources by both the chemical and microbiological constituents of septic tank effluent

3:3 Site Characteristics: A detailed report on the sampling sites used in the study is presented here. Included are descriptions of the geology, hydrogeology, topography and land use at the three test sites as well as details on the construction, design and maintenance of the septic tank treatment systems

3:4 Materials and Methods: This section describes the materials and methods used in the installation and development of the monitoring boreholes and in the sampling/analysis of the overburden, septic tank effluent and groundwater. The statistical methods used in the analysis of the results are also detailed

3:5 Results: The main results of the study are presented in this section. The results of analysis of the monitoring boreholes are presented graphically, while results of the overburden analysis, percentage reduction in effluent constituents and statistical summaries are given in tabular form

3:6 Discussion: The results are discussed in detail with reference to previous investigations

3:7 Conclusions.

3:2 Literature Review

3:2.1 The Effects of Septic Tank Systems on Groundwater Quality

The septic tank was first introduced into the United States (U. S.) in 1883. Since then it has become the most frequently used method for on - site disposal of domestic wastewater. It is estimated that there are between 17 and 20 million households in the U. S. (30% of the population) served by septic tank systems discharging in the region of three billion m^3 of wastewater to soils annually (Bitton and Gerba, 1984; Chen, 1988 and Scalf et al, 1977). At present 25% of all new houses built in the U. S. are installing septic tank systems (Canter and Knox, 1985). In Ireland there are an estimated 300,000 septic tanks serving a population in the region of 1.2 million people and discharging approximately 78 million m^3 of wastewater to soils annually (Henry, 1988). In Canada approximately one million homes are served by septic tanks (Viraraghaven, 1978) while 20% of New Zealand households dispose of their wastewater in this manner (Sinton, 1986).

The efficiency of treatment within a septic tank depends on many factors primarily the design, construction and maintenance of the system. The volume and nature of the waste is also important. The effluent contains high numbers of fecal bacteria and viruses and large amounts of phosphorous and nitrogen as well as having a high Biochemical Oxygen Demand (B. O. D.) and suspended solid (S. S.) content (Patterson et al, 1971). It is a common misconception that the tank will effectively remove bacteria and other microorganisms from the wastewater. Studies have shown that the removal of these organisms within the tank is negligible (Patterson et al. 1971 and Canter and Knox, 1985). Even the most efficient tank can only offer partial treatment and the chemical, physical and biological quality of the effluent is such that it cannot be discharged to surface or groundwaters directly without further treatment. This secondary treatment takes place in the soil which forms an integral part of the process by which the effluent strength is reduced before entering the saturated zone. The efficiency of treatment within the septic tank and the role of the soil in the treatment process is discussed in more detail in Chapter 2.

Recent studies have conclusively proven that septic tank failure is a major cause of groundwater pollution (Patterson et al. 1971; Viraraghaven. 1978; McCoy and Hagedorn, 1979; Stewart and Reneau, 1981; Sinton, 1986 and Aldwell et al, 1988). In many countries septic tank soil distribution systems rank highest in the total volume of wastewater discharged to groundwater systems. In addition they are the most frequently reported source of groundwater pollution (Geraghty and Miller, 1978).

The poor microbiological quality of domestic wellwater in the vicinity of septic tank systems is well documented. A recent study of rural groundwater sources in the United States showed that 92% of all wells monitored were contaminated with Coliform bacteria (Bitton and Gerba, 1984). A similar study in western Ireland found that 68% of all rural groundwater supplies tested in the study area contained fecal Coliforms, fecal Streptococci or both (Aldwell et al, 1988). Septic tank effluent was believed to be the main cause of contamination in both cases. The health implications of such contamination are considerable. Outbreaks of typhoid fever, infectious hepatitis, gastrointestinal infections and infantile methaemoglobinemia have all been linked to malfunctioning septic tanks (Patterson et al, 1971 and Bitton and Gerba, 1984). Almost half the reported disease outbreaks in the United States each year are caused by the consumption of contaminated groundwater (Keswick et al, 1982 and Yates, 1985) with the overflow from septic tanks being responsible for 42% of these outbreaks (Craun, 1979). In 1971 and 1974, 98 cases of viral hepatitis (Arkansas) and 1200 cases of gastrointestinal infections (Florida) were directly traced to malfunctioning septic tank systems (Bitton and Gerba, 1984).

The chemical pollution of groundwater supplies by septic tank effluent is also well documented. As early as 1938 Caldwell demonstrated the ability of the chemical pollutants in the effluent from on - site waste disposal systems to pass through a sandy soil into groundwater, migrating over ten metres in the direction of groundwater flow within a ten day period. Several potentially harmful chemicals present in septic tank wastewater have consistently been shown to contaminate groundwater. These include nitrates (Chen, 1988 and Lewis et al, 1982), detergents (Alhajjar et al, 1989) and toxic organic contaminants (Viraraghaven and Hashem, 1985).

3:2.2 Pollution of Groundwater by Chemical Effluent Contituents

3:2.2.1 Phosphorous

Phosphate has been recently identified as the key element in controlling the growth of algae and other aquatic plants (Jones and Lee, 1979). It is considered to be the major cause of eutrophication in freshwater and seawater bodies (Patterson et al, 1971). The phosphorous concentration in septic tank effluent is generally high and is mainly in the ' available ' state (i. e. soluble orthophosphate) as the anaerobic digestion process within the tank converts most of the organic phosphorous in the influent to soluble phosphorous (Canter and Knox, 1985). Bourma (1979) reported that 85% of the phosphorous in septic tank effluent was in the orthophosphate form. Canter and Knox (1985) found that the average total phosphorous concentration in domestic influent is 25 mg/l, with only 35% of this in

soluble form. Doyle and Thorn (1987) detected soluble orthophosphate effluent concentrations as high as 52 mg/l.

Pollution of groundwater by phosphate from septic tank effluent rarely occurs because of the soil's ability to fix and restrict the movement of the phosphate ion (Chapter 2). Jones and Lee (1979) demonstrated that monitoring boreholes downgradient of a septic tank system in glacial deposits (medium to fine grained sand, mean grain diameter 0.125 to 0.5 mm) were not contaminated by total or soluble phosphorous at any stage during a four year test period, despite the fact that the wells were shown to be contaminated by other septic tank effluent constituents. They concluded that there was no transport of phosphate from the septic tank tile field to groundwater. Polkowski and Boyle (1970) reported that orthophosphate concentrations in septic tank effluent were reduced from 5.15 mg/l to 0.04 mg/l in a shallow well located 4.5 metres downgradient of the adsorption field.

Contamination of groundwater sources by phosphorous in septic tank effluent can, however, occur particularly in sand/gravel overburdens where the adsorptive capacity can be quickly exceeded. Dole (1986) reports that a borehole 5.0 metres downgradient of a septic tank soakage pit contained 4.13 mg/l of orthophosphate compared to a background level of 0.01 mg/l. Similarly, research cited by Canter and Knox (1985) states that phosphate concentrations ranging from less than 1 to 20 mg/l were detected beneath septic tank tile fields, with the higher concentrations detected when groundwater levels were high. Jones and Lee (1979) and Canter and Knox (1985) suggest that if phosphate pollution of groundwater by septic tank effluent becomes a problem it is possible to reduce the phosphorous concentrations in the effluent using certain chemicals. Addition of aluminium sulphate, lime or ferric chloride to the tank can significantly reduce phosphate concentrations in effluent. Phosphorous is completely removed from solution when aluminium is present in large excess. It reacts with the aluminium to form insoluble aluminium phosphates which precipitate to the bottom of the tank, as follows:

$$Al_2(SO_4)_3 + 2PO_4 \rightarrow 2AlPO_4 + 3SO_4$$

3:2.2.2 Chlorides

Chloride salts are natural constituents of surface and groundwaters and are also commonly found in domestic wastewater. Septic tanks are ineffective in removing chloride from the waste and consequently the effluent discharged to the soil adsorption system can contain high amounts of the ion (Patterson et al, 1971 and Canter and Knox, 1985). Polkowski and Boyle (1970) have reported that chloride contamination was easily detectable in groundwater 76.2 metres downstream from a septic tank adsorption field. However, it does not pose a threat to public health or livestock

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except when present in extremely high concentration.

The concentration of chloride in effluents can range from 37 to 101 mg/l (Canter and Knox, 1985). Many authors suggest that because of its anionic form (Cl^-) the chloride ion is highly mobile and can migrate long distances in the unsaturated zone, with minimal interaction with soil/overburden material. Jones and Lee (1979) reported that monitoring of wells installed downstream from a septic tank tile field in glacial drift deposits showed that the conservative elements in the effluent (nitrates, conductivity and chlorides) migrated through the overburden while phosphate was restricted. Saffigna and Keeney (1977) concluded that chloride is as mobile as nitrate in soils and that spreading potassium chloride fertiliser on sand and gravel outwash soils can result in the leaching of chloride to groundwater.

3:2.2.3 Nitrogen

Excessive nitrate concentrations in drinking waters have been shown to result in a deterioration in human health. A maximum concentration of 11.3 mg/l NO_3 - N in drinking water has been recommended by the World Health Organisation (W. H. O.) and the European Community (E. C.). Consumption of nitrate rich water by young children can give rise to methaemoglobinemia or infantile cyanosis (Stewart and Stoleman, 1961; Patterson et al, 1971; Bitton and Gerba, 1984 and Keeney, 1986). Bitton and Gerba (1984) report that most of the cases of infantile cyanosis occurred when the nitrate concentration in the water consumed was greater than 22 mg/l NO_3 - N, although cases have been reported where concentrations were less than 10 mg/l. A link between high nitrate levels in water and gastric cancer has also been suggested (Anon., 1985 and Keeney, 1986).

Nitrogen occurs principally as ammonium and organic nitrogen in septic tank effluent. Canter and Knox (1985) report that 75% of the nitrogen in septic tank effluent occurs as ammonium, with only 25% in the organically bound form. The anaerobic digestion processes which take place within the tank are largely ineffective in removing nitrogen although some conversion of organic nitrogen to the ammonium form does occur. The nitrate concentration of the effluent is generally low due to the anaerobic conditions within the tank. Both ammonium and organic nitrogen are readily converted to nitrate (NO_3^-) under aerobic conditions. This is dealt with in more detail in Chapter 2.

Nitrate is a conservative and highly mobile ion and the pollution of groundwater by nitrate from septic tank adsorption fields is well documented (Polkowski and Boyle, 1970; Patterson et al, 1971; Jones and Lee, 1979; Lewis et al, 1982 and Canter and Knox, 1985). Polkowski and Boyle (1970) report

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that the concentration of nitrate detected downstream of a septic tank adsorption field was significantly higher than that measured upstream. Jones and Lee (1979) report that both nitrate and chloride migrated through outwash sands and gravels to contaminate monitoring boreholes downstream of the soil treatment system. Quan et al (1974) investigated the impact of rapid residential development (using septic tank disposal systems) on the quality of the groundwater in East Portland, Oregon. He reported that the area was transformed over a thirty year period from ' a rural suburban to a basically urban community '. The area described was located on level terraces consisting of fluviolacustrine sediments (pleistocene period) underlain by partially cemented gravels. He estimated that a daily volume of 34,000 to 38,000 m^3 of sewage was disposed of to the soil in this area. It was concluded that this had resulted in serious degradation of the groundwater quality in the region. The nitrate concentration in the groundwater had increased significantly in comparison to adjacent sewered areas. Values of 4.7 to 11.9 mg/l NO_3 - N (mean 7.7 mg/l) were recorded in shallow springs and wells in comparison to 1.0 mg/l NO_3 - N in the sewered area. The contamination was most evident in wells in the upper portion of the saturated zone. A similar study by De Walle and Scaff (1980) noted a decrease in groundwater quality over a thirty year period in Pierce County, Seattle. Most of the contamination was attributed to an increase in the number of septic tank systems in the area. The nitrate values detected in the monitoring wells were highest near the surface and decreased with depth with the lowest values recorded below 43 metres. Increased nitrate levels were noted in the winter months and after periods of heavy rainfall, indicating leaching of nitrate which had previously been ' fixed ' in the soil.

Saffigna and Keeney (1977) report that losses of nitrate through soil profiles can be very significant leading to severe groundwater pollution. Daly and Daly (1984) noted that, although nitrate levels in Irish groundwaters were not a significant problem, contamination by point sources should be prevented by sensible location, design and management of these sources and by proper location and construction of wells. Doyle and Thorn (1987) reported that a borehole downgradient of a septic tank soakage pit contained 45.0 mg/l NO_3 - N, compared to a background concentration of 1.0 mg/l. It was concluded that the ammonium nitrogen from the septic tank was rapidly converted to nitrate in the sand/silt overburden and was subsequently leached to groundwater.

3:2.2.4 Detergents and Trace Organics

Synthetic detergents such as Alkyl Benzene Sulphonates contained in septic tank effluents have been shown to have serious effects on the quality of groundwater

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beneath septic tank soil treatment system (Patterson et al, 1971). They report that approximately 90% of the alkyl benzene sulphonates (A. B. S.) entering the tank passed directly into the soil treatment system and that removal in soils was minimal. The introduction of biodegradable linear alkylated sulphonates (L. A. S.) in 1965 greatly reduced this problem. L. A. S. substances are not degradable under the anaerobic conditions in a septic tank. However, virtually complete degradation of the substances can occur in the soil treatment system, provided that conditions remain aerobic (Patterson et al, 1971). Polkowski and Boyle (1970) also report low concentrations of methylene blue active substances (M. B. A. S.) in a septic tank adsorption field groundwater plume.

Few data are available on the potential of septic tank effluent to contaminate groundwater with trace organic chemicals. Some of these chemicals are known carcinogens and could pose a serious threat to human health. Canter and Knox (1985) report that there is evidence of organic chemical contamination of aquifers by septic tank effluents. The organic chemical most frequently detected in groundwater is trichloroethylene, a constituent of the most commonly used septic tank cleaners. Other volatile organics found include tetrachloroethylene, 1, 1, 1 - trichloroethane, 1, 1 - dichloroethane and dichloroethane (United States E. P. A., 1980). Viraraghaven and Hashem (1986) investigated the levels of trace organics present in septic tank effluent. They point out that in most areas of the U. S., householders can purchase septic tank cleaning fluids containing trichloroethylene (TCE), benzene or methylene chloride and that the greatest contribution of toxic compounds to household wastewater originates from cleaning products and cosmetics (Table 3.2.1, p120). They concluded, however, that the concentration of trace organics in septic tank effluent was comparatively low and may not pose a threat to aquatic life or human health when the attenuating capacity of the soil and dilution by groundwater is taken into account. A survey by Burmaster (1982) indicated that groundwater in the U. S. is contaminated by toxic organic compounds. The survey suggests that septic tank systems may be contributing to the increase detected in the concentration of organic compounds in drinking water. De Walle and Scaff (1980) measured several volatile compounds in septic tank influents and effluents and identified five priority pollutants in the raw water i. e. toluene, dichloromethane, chloroform, tetrachloroethane and ethylbenzene. None of these compounds were removed to any great extent in the tank (Table 3.2.2, p120).

3:2.3 Pollution of Groundwater by Microbiological Effluent Constituents.

One of the most commonly reported effects of septic tank systems has been the pollution of water wells with fecal bacteria contained in the effluent. Most of the

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microorganisms present in excreta pass directly through the septic tank and into the soil around the tile field. They are not capable of self - propulsion but are carried along with the effluent flowing through the soil. In a new soil adsorption bed the travel of microorganisms through the unclogged soil may be rapid and far reaching (Caldwell, 1938 and Patterson et al, 1971). However, the ability of the soil to restrict microbial movement has been shown to increase with time as a clogged zone or biological mat forms beneath the soil treatment system. McCoy and Ziebell (1975) observed that these mat formations were capable of straining up to 99.9% of the original effluent Coliform bacterial numbers.

The soil itself also has the ability to restrict the movement of microorganisms mainly by filtration and adsorption. However, the degree to which these mechanisms are effective is largely dependent on the nature of the soil (Cain and Beatty, 1965; McCoy and Ziebell, 1975; Lewis et al, 1982 and Bitton and Gerba, 1984). Under certain soil and climatic conditions effluent organisms can travel considerable distances through soil/overburden materials and contaminate groundwater (McCoy and Ziebell, 1975 and Bitton and Gerba, 1984).

Caldwell (1938) was one of the first to demonstrate the potential of subsurface sewage disposal to cause microbiological contamination of groundwater. He found that *Escherichia coli* contained in pit latrine effluent migrated a maximum distance of 3.05 metres downgradient of the pit. He also reported a decrease in numbers and travel distances of the bacteria with time. This was attributed to the development of a biological mat. In the same study Caldwell showed that the chemical pollutants in the waste travelled much greater distances. He concluded that ' the *B. coli* stream formed an inner core of less width and markedly less extent than the chemical '.

McCoy and Ziebell (1975) reported that a high degree of purification can be achieved by passing effluent through relatively small amounts of soil. However, in certain situations contamination can occur with resulting disease outbreaks. They cite an example of a family of four members in Yakima County, Washington who contacted typhoid (caused by *Salmonella typhi*) from well water which had been contaminated by a septic tank disposal system 61 metres away.

Viraraghaven (1978) demonstrated that Coliform, fecal Coliform and fecal Streptococci bacteria migrated through a clay/sand soil to groundwater. The organisms were isolated from monitoring boreholes in numbers which declined with increasing distance from the tile field (in the direction of groundwater flow). The numbers isolated from boreholes at 15.25 meters from the field were still high (10^2 to 10^4 bacteria/100 ml). This was attributed to a fluctuating watertable which limited the travel distance in the unsaturated zone to 0 to 0.15 metres thereby preventing organism removal by filtration and adsorption. Stewart and Reneau (1981)

Table 3.2.1

Typical Toxic Compounds in Household Products
(after Viraraghavan and Hashem, 1986)

Products	Toxic Compounds
Toilet bowl cleaners Drain pipe cleaners Septic tank cleaners Stove and oven cleaners Tile and tub cleaners	Solvents such as benzene, toluene, dichlorobenzene, trichloroethane phthalates, dichloropropanes, dichloroprophylene, isophorone, trichloroethylene, carbon tetrachloride, phenol, chlorophenol
Rug and upholstery cleaners Paint brush cleaners Paint thinners	Solvents such as benzene, dichloroethane, chloroethylether, dichlorobenzene, dichloroethylene, isophorone
Cosmetics	Pigments (heavy metals), perfumes (aromatics), antibacterial agents (phenol and chlorophenols)

Table 3.2.2

Volatile Organics in Septic Tank Influent and Effluent
(averaged over 7 day 24 hour composites)
(after De Walle and Scaff, 1980)

Organics	Concentration (micrograms/litre)			
	Influent	Effluent	Scum	Solids
Toluene	34.60	38.80	0.7	0.02
Dichloromethane	3.60	3.40	0.9	0.25
Chloroform	1.70	0.76	0.1	0.06
Tetrachloroethene	0.76	0.28	5.8	7.60
Ethylbenzene	0.10	0.10	6.9	6.00

demonstrated that fecal Coliforms from a septic tank soil treatment system migrated through 10.0 metres of saturated soil in the direction of groundwater flow. They concluded that the use of septic tank systems in areas where the water table is permanently high can result in bacterial contamination of shallow groundwater. Hagedorn et al (1978) used an antibiotic resistant strain of fecal bacteria to monitor the movement of microorganisms from a septic tank soil treatment system. They demonstrated that bacteria migrated long distances through saturated soils in a relatively short length of time. They also noted that peak numbers were recorded in the monitoring wells after periods of heavy rainfall and that survival of organisms in the soil treatment system during the 32 day sampling period was considerable. Similarly Reneau (1978) examined the migration of fecal bacteria from a soil treatment system into a drained tile field. He reports, that although artificial drainage may render an area hydraulically suitable for subsurface effluent disposal, it is more difficult to assess the adequacy of the system with respect to the penetration of the biological constituents present in the wastewater. He concluded that these systems have greatest potential for failure when the water table is high (i. e. in winter and spring). Pitt (1974) found that bacterial contamination of groundwater was occurring to a depth of 18.29 metres directly beneath a soil treatment system. However, no fecal Coliform bacteria were found below 0.91 metres. He also showed that bacterial pollution of groundwater was greater where the density of septic tank systems per unit area was higher. The results of a detailed study of 17 septic tank systems located near the lakeshores of eight lakes in New York state showed that significant contamination of the lakeshore groundwaters was occurring with nitrate and fecal Coliform bacteria being the main pollutants detected (Chen, 1988). De Walle and Scaff (1980) investigated the groundwater quality in Pierce County, U. S.. They detected a gradual increase in groundwater contamination over a 30 year period. Most of the pollution was reflected in increased Coliform counts due to contamination from septic tank systems.

Two methods of septic tank effluent disposal were investigated by Sinton (1986) in the Canterbury plains, New Zealand. The movement of fecal Coliform bacteria nine meters from a 5.5 metre deep soakage pit into an unconfined aquifer and 42 metres from a 18.1 metres deep injection borehole into a confined aquifer was recorded. There was evidence of a groundwater mound beneath the soakage pit and injection bore which caused radial spread of the leachate to occur. Marked diurnal variations were found in the bacterial numbers isolated from the monitoring boreholes. These variations coincided with periods of effluent discharge, indicating that frequent sampling may be necessary to establish peak contamination levels. The report also noted the migration of bacteria in specific zones in the subsurface material. Sinton concluded that the migration in specific zones of higher

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permeability and the radial spread of effluent in response to recharge mound formation indicates that regional piezometric surveys may not accurately reflect leachate flow paths over short distances. The ability of effluent bacteria to migrate rapidly through zones of high permeability at specific depths in the subsurface has also been demonstrated by Rahe et al (1978), McCoy and Hagedorn (1979), Lewis et al (1982), Sinton (1986) and Henry (1987).

There is a distinct lack of information available regarding the movement of enteric viruses from septic tank soil treatment systems to groundwater. In recent years their potential as a possible pollutant has been recognised. Because of their small size and longer survival time they are less susceptible to losses in the overburden material and are more likely to reach and contaminate groundwater than most bacterial species (Yates and Yates, 1989). The persistance of polio virus 1, echovirus 1 and M S-2 coliphage viruses in groundwater was investigated by Yates (1985). They concluded that temperature was the only variable that correlated with decay rates for all three viruses. No significant differences in survival rates were detected between the three species. Significant numbers of the viruses were found to be viable in the groundwater after a 30 day sampling period. The setback distances of septic tank systems from groundwater sources was also investigated by Yates et al (1986) and Yates and Yates (1989). Lewis et al (1982) report an outbreak of gastroenteritis which was directly linked to a well contaminated with poliovirus. The pollution was traced to a septic tank drain field located 43 metres from the well. Coliform levels in the well ranged from 0 to 16 colony forming units /100 mls but Salmonella or Shigella organisms were not found. Vaughn et al (1985) carried out a study of the movement of naturally occurring human enteroviruses from subsurface wastewater disposal systems. The movement of viruses 67.05 metres downgradient of the disposal system at a depth of 18 metres in the aquifer was demonstrated. The study also showed the limitations of current microbial water quality indicators for predicting the virological quality of groundwater.

3:2.4 Assessment of the Groundwater Polluting Potential of a Septic Tank System

3:2.4.1 Numerical Methods

Several techniques may be used to evaluate the potential of septic tank systems to pollute groundwater. These methods range from the calculation of empirical indices to the use of sophisticated mathematical models. Various mathematical models exist, some of which address groundwater flow using analytical methods, and others which adopt a numerical analytical approach amalgamating both flow and solute

transport considerations (Canter and Knox, 1985). Most authors are agreed on the necessity of carrying out site suitability studies in areas selected for the disposal of septic tank effluent. However, very little modelling has been done on the effect of septic tanks on groundwater systems, probably because the large scale modelling operations needed are complex and time consuming and to date have only been applied to large centralised sewage disposal systems.

Cartwright and Sherman (1974) proposed a simple empirical index for assessing the risk of pollution from individual septic tank systems. They consider that permeability of soil, volume of effluent and thickness of overburden above groundwater sources are the most significant factors in determining pollution potential. They used a standard percolation test to determine soil permeability and housing density as an approximation of the volume of effluent. Using these parameters, they developed the following formula to estimate the groundwater pollution potential of a septic tank system:

$$Pi = \frac{200\sqrt{D}}{\sqrt{I}\,(T - 5)}$$

where Pi = pollution index
D = density of housing served by the septic tank within a 450 metres radius
I = time taken in minutes for water to fall six inches in the percolation test
T = thickness in feet of soil between discharge level and groundwater.

A Pi value of less than ten indicates that the potential for significant pollution is fairly low while a Pi value of greater that ten suggests that some potential for pollution exists and that a more detailed assessment of the site is necessary.

3:2.4.2 Groundwater Monitoring Programmes

Accurate and reliable monitoring of groundwater quality at sites where systems are already in use is equally important in asssessing the pollution potential of septic tanks. Regular monitoring is essential in areas where the density of septic tank systems is high. According to Canter and Knox (1985) the first requirement for a groundwater monitoring programme is a clear delineation of monitoring objectives.

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Nelson and Ward (1982) suggest two basic objectives based on the detection of system failure:

(i) The detection of temporary overloads of high pollutant concentrations in groundwater

(ii) The detection of permanent overloads of high pollutant concentrations.

The three primary components of a septic system monitoring programme are given as:

(i) Determination of sample location

(ii) Selection of parameters to be monitored

(iii) Selection of the required number of samples.

(Nelson and Ward, 1982)

It is proposed that, since the treatment system consists of the septic tank, the soil treatment system and the unsaturated soil zone beneath the tile field, the most logical sampling point is the upper layer of the saturated zone directly beneath the treatment system. The parameters analysed for should include biochemical oxygen demand (B. O. D.), chemical oxygen demand (C. O. D.), total solids, suspended solids, total organic carbon (T. O. C.), total nitrogen, ammonia, nitrate, total phosphorous, orthophosphate, total bacteria, total Coliform bacteria, fecal Coliform bacteria, fecal Streptococci bacteria and enteric viruses. The pollutants of greatest concern are those which could constitute a health hazard i. e. bacteria, nitrates and viruses (Canter and Knox, 1985).

Another important consideration is the representative sampling of aquifer water. Special precautions should be taken to ensure that the sample is neither altered or contaminated during sampling and handling and that it is representative of the water in the aquifer. Marsh and Lloyd (1980) found that chemical reaction between groundwaters and well steel casing resulted in alterations in the pH, Eh, T. D. S., major ion and trace ion concentrations in extracted samples compared to levels generally detected in the aquifer. They also point out that stagnation in a monitoring borehole modifies the groundwater chemistry to the extent that samples from such sections may be totally unrepresentative of the natural water in the aquifer. They conclude that all stored water must be flushed from the well or that a sampling tube be used to ensure that water is taken from within the aquifer. The pumping time required to completely flush out the well can be calculated if the casing dimensions and the flow rates are known. The flushing progress can be monitored by observing changes in pH, electrical conductivity or temperature readings. In addition they recommend that well casing be constructed of an inert corrosion resistant material such as PVC. Keeley and

Boateng (1987) also suggest that purging of a monitoring well is important prior to sampling. Purging can help to remove sediment accumulated between sampling events. They recommend removing at least two borehole volumes of water in addition to monitoring the pH, Eh, temperature and specific conductance of the purged water. Scalf et al (1977) recommended removal of between four and ten well volumes before sampling.

3:3 Materials and Methods

3:3.1 *Borehole Installation and Development*

A total of eight monitoring boreholes were installed at three test sites by the Geological Survey of Ireland (G. S. I.) B 53 ' Explorer ' mobile drilling rig (Plate 3.3.1, p142). A variety of techniques was used to drill through the different overburden materials encountered. At site 1 an auguring attachment was used to bore through the thin loamy clay overburden to bedrock while a Tricomb rotary method with water flush was required to drill through the stony gravel overburden at site 2. A combination of the above methods was used at site 3. Drilling through the bedrock at all sites was carried out using a rotary wireline coring method. Soil/overburden material and bedrock cores removed during the drilling process were retained for detailed laboratory analysis and assessment. The solid rock cores were also measured and described.

Standard 5.5 centimetre Wavin PVC ducting was used as casing material in all the monitoring boreholes. Horizontal slits were cut at the required depth in the casing to provide a well screen. The area surrounding the screen was packed to rock head with an envelope of coarse sand and gravel. The ground surface around the well head was then covered with a concrete apron to prevent water running down the inside of the casing. Protective caps and locks were fitted to the top of each of the monitoring bores (Plate 3.3.2, p142).

Natural gamma logs were taken of all the monitoring wells using the Geological Survey of Ireland ' Widco ' porta logger. The geophysical (gamma) and the geological logs for each of the monitoring wells are presented in Appendix B 1 and B 2. Table 3.4.2 (p154) shows summary statistics for each of the monitoring boreholes and a schematic representation of the general borehole construction is given in Figure 3.3.1 (p136).

Development of the boreholes involved removing muds and fine material left in the area outside the well screen after the drilling process. A purging technique similar to that described by Scalf at al (1977) and Keeley and Boateng (1987) was used. A heavy bailer was dropped to the bottom of the boreholes and successively raised and lowered opposite the well screen. This was carried out daily over a seven day period.

3:3.2 Hydrogeological Investigations

A series of pumping tests using recognised standard methods were carried out on the boreholes in June 1988 to obtain information on the hydrological characteristics at the three test sites. The Geological Survey of Ireland ' Seba '

pneumatic - submersible pump was used to pump water from each bore and water level probes were used to measure the resulting drawdown on the other wells at the site. The pump was operated at a fixed discharge rate of 273 litres per hour. The results of these tests are described in 3:4.

Problems were encountered at all three sites. Dirt and grit in the polluted wells at site 1 resulted in clogging of the pump intake valve and the test had to be abandoned before sufficient data could be collected. At site 2 investigations were complicated by the fact that the groundwater occurs in two water bearing bodies, one in sands and gravel, and the other in the underlying limestone. Pumping was successful and drawdown was measured when the water was in the gravel overburdens, but once it entered the limestone the borehole was pumped dry very quickly. Consequently, because of the short duration of the test, insufficient data were obtained to ensure an accurate hydrogeological assessment. The high permeability at site 3 resulted in little or no measurable drawdown in either the pumped or monitoring wells even after two hours pumping at maximum pump discharge of 318 litres per hour and no reliable results were obtained. The pumping test analysis therefore resulted in only a rough estimation of the hydrological characteristics at the sites. All results obtained were plotted on semi - log paper and the resulting drawdown and residual curves were analysed using the Jacob method (Kruseman and De Ridder, 1970).

Because of the problems encountered it was decided to carry out another series of tests using a non discharge displacement method. This involved lowering a displacement object into the monitoring well and measuring the resulting instantaneous rise in water level and the subsequent fall to it's original position (falling head test). The displacer was then removed and the rise in water level to it's original position was also monitored (rising head test). Problems were again encountered at all three sites. Because the monitoring wells were primarily constructed as water sampling wells, water may have passed out through the well screen during the displacement test into the surrounding gravel envelope and the resulting rise and fall in water levels was not sufficient to yield enough results for accurate hydrogeological assessment. The data obtained at sites 1 and 2 were analysed using the method described by Bouwer and Rice (1976) and a series of transmissivity values was then calculated. The test was again unsuccessful at site 3 because of the high permeability of the overburden material.

3:3.3 Sampling

3:3.3.1 Soil

Samples of soil and overburden were taken at the three sites during drilling in

order to assess their ability to attenuate septic tank effluent constituents. The samples were taken at a number of different depths, as follows:

Site 1	1.0 and 2.0 metres
Site 2	1.0, 2.0 and 3.0 metres
Site 3	1.0, 2.0 and 2.5 metres

At sites 1 and 3 the samples were taken directly from the drilling auger flights. A different method of drilling was, however, used at site 2 (tricombing with water flush). Here samples were taken from an excavated hole at the site. It was not possible to obtain a sample of the permeable coarse overburden material underlying the boulder clay at site 3. In this case it was necessary to rely on information given by the borehole drillers.

To ensure that an accurate, representative sample was obtained a large quantity of overburden material (minimum of 1 kg weight) was taken at each sampling point. All samples were brought to the laboratory and were refrigerated until analysis.

3:3.3.2 Water

The series of pumping tests described in 3:3.2 above were used to determine the volume of water which should be removed from the monitoring boreholes to obtain a representative groundwater sample. Samples were taken for chemical analysis at various intervals during the pumping process. The results of the analysis were plotted (Appendix B 3) and the minimum volume of water required to be removed before stabilisation of the groundwater chemistry occurred was determined. From these graphs it can be seen that stabilisation occurred between 15 and 25 minutes (pumping rate of 273 litres per hour). This represents the equivalent of approximately four well volumes. The pumping tests were unsuccessful at site 1 and no data were obtained. However, on the basis of the results obtained at the other two sites and reports by Scalf et al (1977) and Keeley and Boateng (1987), it was decided to remove four well volumes from the monitoring wells prior to sampling.

Between July 1988 and March 1989 samples were taken at monthly intervals from the eight monitoring boreholes (when possible) and the two control sites. A final set of samples was taken in June 1989. As described above four well volumes were removed from each of the boreholes prior to sampling (a well volume was considered to be the total volume of water within the borehole at the time of sampling). The water was removed from the monitoring wells using a hand operated Wild ' Gusher ' diaphragm pump. Samples were taken at a depth of 1.0 metre below the static water level (S. W. L.).

Pumping was achieved at the control borehole C2 by allowing a tap to run for five minutes (the exact dimensions of the well were not known). No pumping was necessary at C1 since the sample was taken directly from a free flowing spring. Samples for hydrochemical analysis were filled directly from the diaphragm pump discharge hose to clean acid washed one litre polypropylene sampling bottles. The bottles were rinsed with the sample water and were filled to capacity to exclude air, as recommended by Coxon and Thorn (1989).

Samples for microbiological analysis were taken in 250 millilitre sterile bottles. The sampling bottles were secured to a length of string with lead weights attached (Figure 3.3.2, p137). Before lowering the sampler into the borehole the cap was replaced with a sterile cork stopper attached to another length of string. When the bottle was lowered to 1.0 metre beneath S. W. L. the cork was removed by pulling the attached string, allowing the bottle to fill with water. The bottle was then slowly withdrawn from the borehole taking care to avoid contact with the sides of the casing. A separate sampler was used at each borehole to avoid cross contamination. When larger samples were needed, as in analysis for sulphite reducing Clostridia and Salmonella, the water was extracted using a sterile hand bailer, as described in Chapter 4. Samples from the control sites C1 and C2 were taken directly from the flowing spring and tap to one litre polypropylene and to sterile 250 millilitre (wide neck) glass sampling bottles. A water softener on the borehole C2 was turned off prior to sampling and the tap was allowed to run for five minutes. The tap was swabbed with ethanol and flamed before taking the microbiological sample. All samples were transferred immediately to a polystyrene ' cool box ' for transport to the laboratory.

3:3.3.3 Effluent

Effluent samples were taken from the septic tanks at the test sites on three occasions, in July and November 1988 and in March 1989, to assess the percentage reduction in the effluent constituents from the septic tank to the monitoring boreholes. The samples were taken directly from the tanks at the outlet T - pipes. Samples for chemical analysis were taken in one litre polypropylene bottles as described in 3:3.4.2, while those for microbiological analysis were taken in sterile 250 millilitre (wide neck) glass sampling bottles. They were transported to the laboratory as described above.

3:3.4 Analysis

3:3.4.1 Soil

Table 3.4.1 (p144) summarises the parameters analysed for and the methods

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used in the analysis of the soil samples. All analysis was carried out in accordance with British Standards B.S. 1877 (Anon., 1975), ' Chemical Analysis of Agricultural Materials ' (Byrne, 1979) and ' Methods of Soil Analysis ' (Black et al, 1965). Soil permeability (infiltration rate) was determined using the I. I. R. S. percolation test (Anon., 1975).

3:3.4.2 Water

All analysis was completed within 24 hours of sampling with the exception of the determination of calcium and magnesium. These samples were acidified and refrigerated. Table 3.3.2 (p134) summarises the chemical parameters analysed for and the methods used. Electrical conductivity, pH and temperature were measured on site. On return to the laboratory, half of the one litre sample was filtered through a 0.47 µm membrane filter and placed in a second acid washed one litre polypropylene bottle. Analysis for ammonia, alkalinity, biochemical oxygen demand and chemical oxygen demand was carried out on the unfiltered samples. All other analyses were carried out on filtered samples.

A 50 ml portion of the filtered water was added to two mls of 2% lanthanum chloride (Anon., 1985) and the samples were stored in the refrigerator for analysis of metals. Analysis for detergents, as methylene blue active substances (M. B. A. S.), iron and manganese was only carried out on two occasions.

Ion balance errors were calculated from the results of all samples analysed. Because water is electrically neutral the concentration of anions (meq/litre) should equal the sum of the cations (meq/litre):

$$\text{Ion balance error (\%)} = \frac{\sum \text{sum of cations} - \sum \text{sum of anions}}{\sum \text{sum of cations} + \sum \text{sum of anions}} \times 100$$

(Lloyd and Heathcote, 1985)

This calculation therefore provides an indication of the overall accuracy of the analysis. The error should be less than 5% with modern analytical methods.

All microbiological analyses were carried out within six hours of sampling. Total and fecal Coliforms and fecal Streptococci were routinely analysed for. All analysis was by membrane filtration in accordance with Standard Methods (Anon., 1985). A 50ml aliquot of the water sample was added to a sterile filtration apparatus (autoclaved at 121 °C and 15 p. s. i. for 15 minutes) using sterile glass cotton wool stoppered pipettes (heat sterilised in an oven at 160 °C for 1.5 hours). The

filters were then transferred to a selective media and incubated at the appropriate temperature. After the required incubation period counts of typical colonies were taken and recorded as colony forming units (c. f. u. 's) per 100 mls water. On a number of occasions further confirmatory tests were carried out and on two occasions the samples were also analysed for the presence of sulphite reducing Clostridia and Salmonella species. Details of the analytical procedures used, including those for the Clostridia and Salmonella analysis, are presented in Figures 3.3.3 to 3.3.6 (p138 to 141).

3:3.4.3 Effluent

Table 3.3.3 (p135) summarises the chemical parameters analysed for and the methods used. The electrical conductivity and pH were measured on site. On return to the laboratory, half of the one litre sample was filtered through a 0.47 µm membrane filter and placed in a second acid washed one litre polypropylene bottle. Analyses for ammonia, nitrate, biochemical oxygen demand and chemical oxygen demand were carried out on the unfiltered samples. All other analyses were carried out on the filtered samples. Many of the methods of analyses differed from those used for the groundwater samples because of the significantly higher concentrations present and the high sample turbidity.

Microbiological analysis was carried out within six hours of sampling. Total and fecal Coliforms and fecal Streptococci were analysed for using a pour plate technique with serial dilution in 1/4 strength Ringer's solution. The methods used are summarised in Table 3.3.4 (p135).

3:3.5 Statistical Analysis

All statistical analyses were carried out using Apple Macintosh ' Statworks ' and IBM ' Systat ' software packages.

3:3.5.1 Wilcoxon Signed Rank Test

Statistical analysis of the data was carried out using the Wilcoxon Signed Rank test, a non - parametric alternative to the t - test for paired samples (Snedecor and Corchran, 1963). The test is a useful method for assessing the difference between measurements when no assumptions can be made about the distribution of a population. The absolute values of the differences between the two measurements are ranked (ignoring signs), with the smallest difference being assigned the rank 1. Zero differences contribute no information to the comparison and are ignored (Campbell, 1984). Signs are then restored to the rankings and the sum of the positive (R^+) and negative (R^-) ranks are calculated. The test statistic T is the smaller of R^+ and

R^-. The following can be used as a check on the calculations:

$$R^+ + R^- = 1/2\, n\, (n + 1),$$

The rank with the less frequent sign will usually, though not always, give the smaller rank total. The test statistic is then referred to the statistical tables developed by Wilcoxon. The null hypothesis in this test is that the frequency distribution of the original measurements are the same for the treated and untreated members of a pair (i. e. there is no significant difference between the measurements). A test statistic of less than the critical value indicates rejection of the null hypothesis H_0 (i. e. there is a significant difference between the two measurements, in this case assumed to be greater - a one tailed test). Statistics were calculated at P levels of 0.025 (2.5% level, significant) and 0.005 (0.5% level, highly significant).

Null Hypothesis:	$H_0 : \mu_1 = \mu_2$ (no difference between measurements)
Alternative Hypothesis:	$H_1 : \mu_1 \neq \mu_2$ (difference between measurements)
	$H_1 : \mu_1 > \mu_2$ i. e. a one tailed test

The above test was applied to the data to assess:

(i) If concentrations of the various parameters recorded in the monitoring boreholes were significantly greater than those at the control sites

(ii) If there was a significant decrease in the contamination of groundwater with increasing distance from the septic tank soil treatment system.

3:3.5.2 Correlation Coefficients

The relationship between increased rainfall and elevated fecal Coliform bacterial numbers in the groundwater samples was statistically analysed using Pearson's correlation. To apply this method it was nesessary to assume that the number of fecal bacteria in the groundwater samples is a normally distributed variable. Scatter plots of the bacterial numbers (c. f. u. 's/100ml) isolated in the monitoring boreholes and control sites against the total rainfall (mm) for the five days preceding sampling were prepared (Appendix B 4). A simple regression curve was fitted to the plots and Pearson's coefficients of determination (r^2) and correlation (r) were then calculated. In order to test the normality assumption a plot of the residuals and the residuals against expected ' y ' values was prepared for each of the fitted lines. The model is satisfied if the residuals are randomly distributed about zero with no obvious dependencies. The model is rejected in the event of spurious residual distributions.

The significance of the correlation coefficient (r) was tested using the following test statistic:

$$t = r\sqrt{\frac{n-2}{1-r^2}} \sim t_{n-2} \qquad \text{(at P = 0.01 and 0.001)}$$

Ho : $\partial = 0$ non linear relationship

Ho : $\partial \neq 0$ linear relationship

(where r estimates ∂)

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Table 3.3.1

Analytical Methods Used in the Assessment of the Physical and Chemical Properties of the Soil/Overburden Material

Parameter	Method
pH	Electrometric
Moisture Content	Loss - on - Drying
Soil Texture	Sieving and Sedimentation
Porosity	Bulk Density/Particle Density
Organic Matter Content	Loss - on - Ignition
Organic Carbon Content	Walkley - Black Titration
Permeability	I. I. R. S.. (1975) Percolation Test

Table 3.3.2

Analytical Methods Used in the Hydrochemical Analysis of the Groundwater Samples

Parameter	Method
Water Level	Water Level Probe
Temperature	Thermometric
pH	Electrometric
Conductivity	Electrometric
B. O. D.	Electrometric
C. O. D.	Modified C. O. D. Reactor Method
Nitrate (NO_3 - N)	Ultra Violet Spectrophotometry
Ammonia (NH_3 - N)	Visible Spectrophotometry
Phosphate (PO_4 - P)	Visible Spectrophotometry
Sulphate (SO_4)	Visible Spectrophotometry
Chloride (Cl^-)	Argentometric Titration
Sodium (Na^+)	Flame Photometry
Potassium (K^+)	Flame Photometry
Alkalinity ($CaCO_3$)	Titrimetric
Calcium (Ca^+)	Atomic Absorption Spectrophotometry
Magnesium (Mg^+)	Atomic Absorption Spectrophotometry
Iron (Fe^+)	Atomic Absorption Spectrophotometry
Manganese (Mn^+)	Atomic Absorption Spectrophotometry
M. B. A. S.	Methylene Blue Method (with Chloroform extraction)

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Table 3.3.3

Analytical Methods Used in the Hydrochemical Analysis of the Septic Tank Effluent Samples

Parameter	Method
pH	Electrometric
Conductivity	Electrometric
B. O. D.	Electrometric
C. O. D.	Modified C. O. D. Reactor Method
Suspended Solids	Gravimetric
Nitrate (NO_3 - N)	Markham Still Distillation
Ammonia (NH_3 - N)	Markham Still Distillation
Phosphate (PO_4 - P)	Visible Spectrophotometry
Sulphate (SO_4)	Gravimetric (with ignition of residue)
Chloride (Cl^-)	Argentometric Titration
Sodium (Na^+)	Flame Photometry
Potassium (K^+)	Flame Photometry

Table 3.3.4

Analytical Methods Used in the Microbiological Analysis of the Septic Tank Effluent Samples

Parameter	Method
Total Coliform Bacteria	Pour plate method with serial dilution in 1/4 strength Ringer's solution and isolation on Violet Red Bile Agar (V. R. B. A. - Oxoid) growth media
Fecal Coliform Bacteria	Pour plate method with serial dilution in 1/4 strength Ringer's solution and isolation on Violet Red Bile Agar (V. R. B. A. - Oxoid) growth media
Fecal Streptococci Bacteria	Pour plate method with serial dilution in 1/4 strength Ringer's solution and isolation on KF Streptococcus (Oxoid) growth media.

Figure 3.3.1

Schematic Diagram of the Monitoring,Borehole Construction

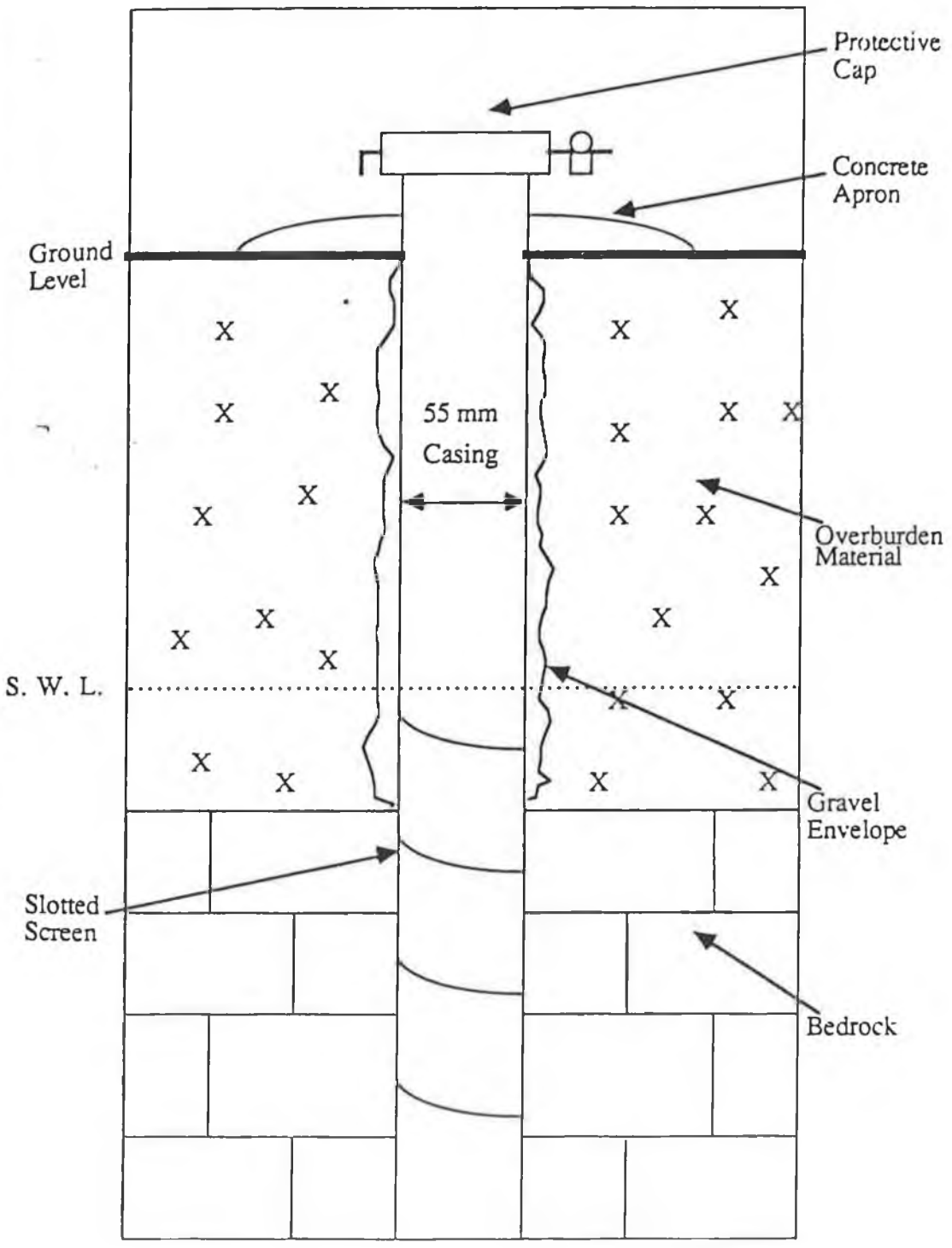

Figure 3.3.2

Sampling Device Used to Obtain Microbiological Samples from the Monitoring Boreholes

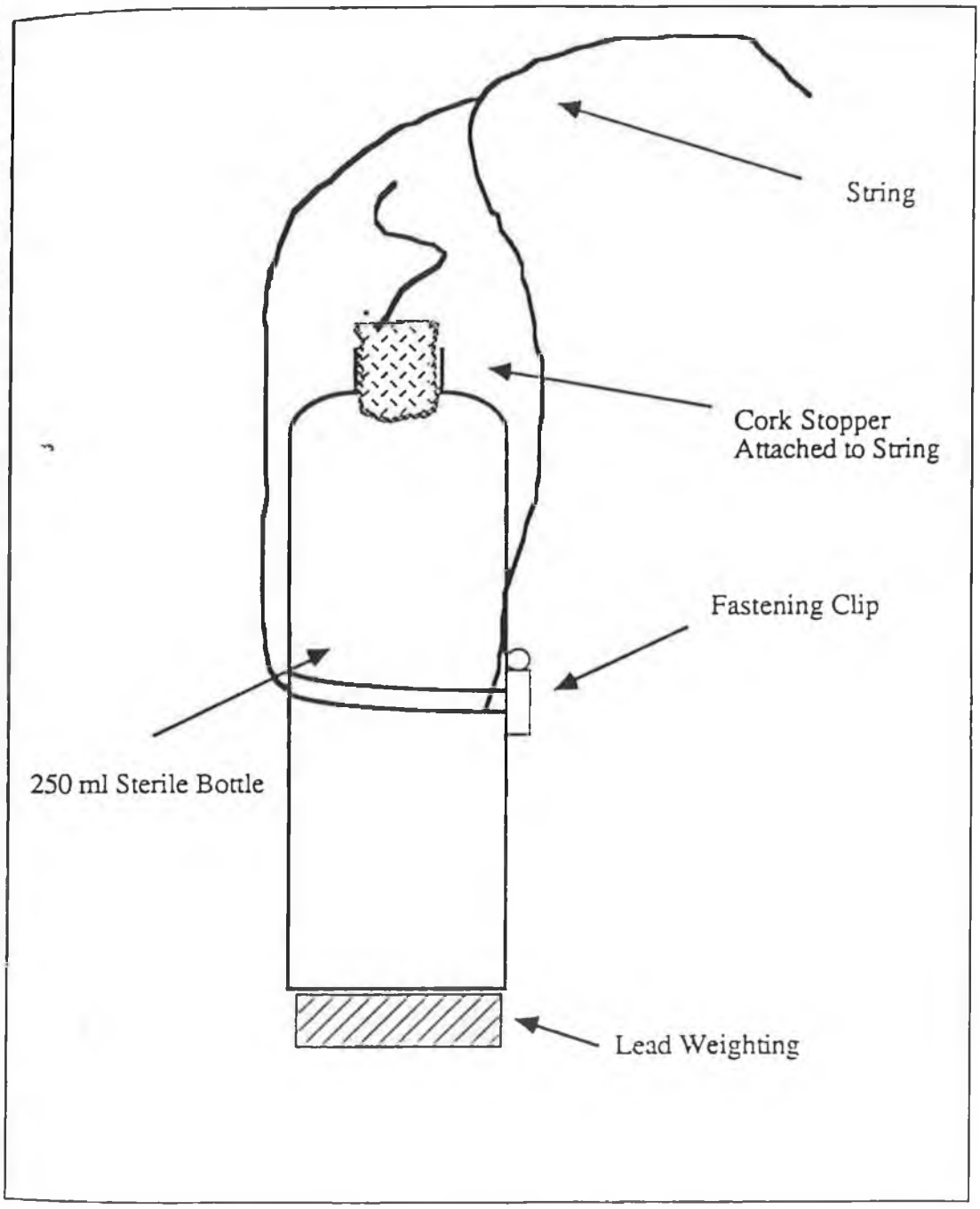

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Figure 3.3.3

Isolation and Identification of Total and Fecal Coliform Bacteria from the Groundwater Samples

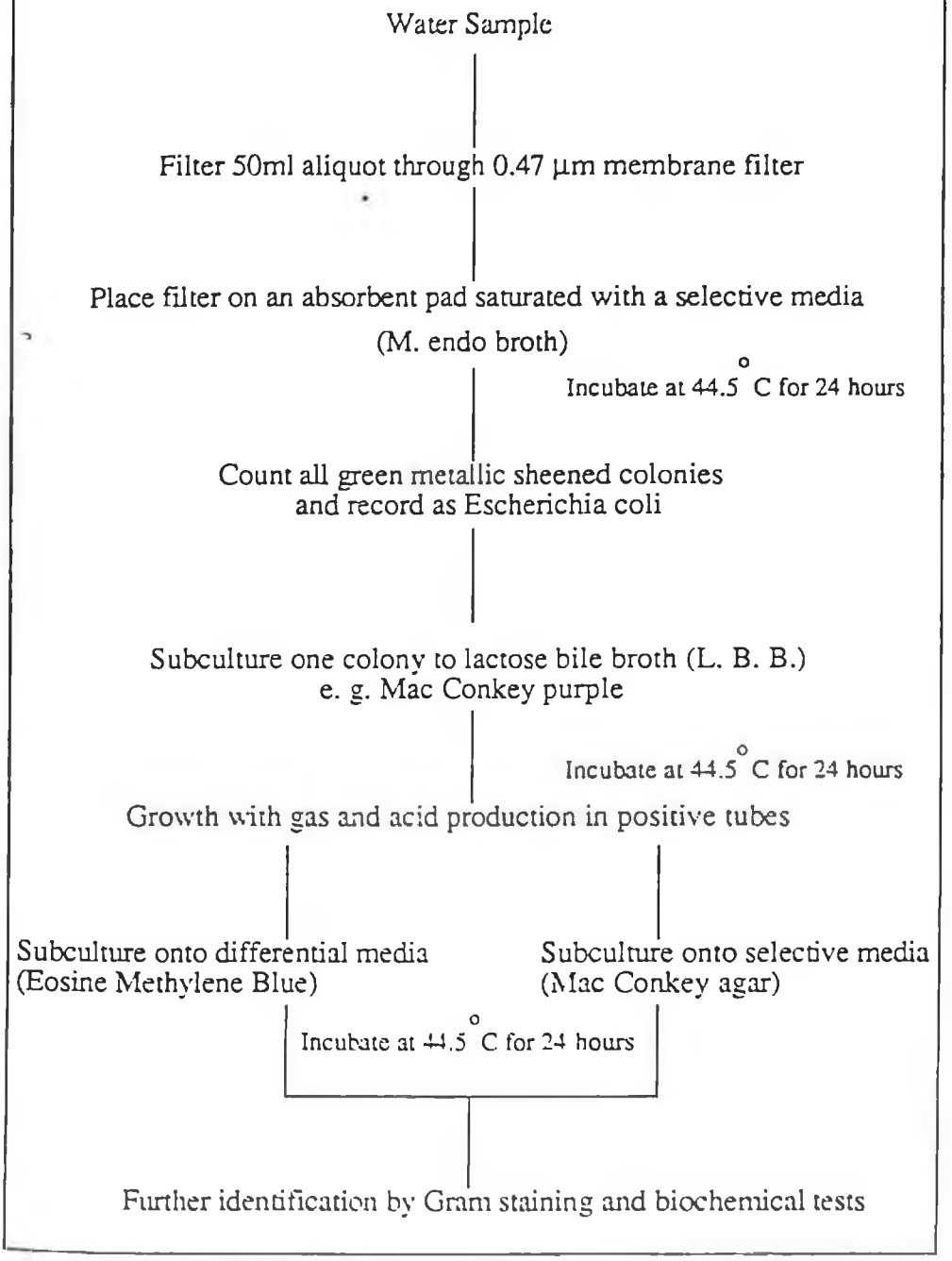

Figure 3.3.4

Isolation and Identification of Fecal Streptococci Bacteria from the Groundwater Samples

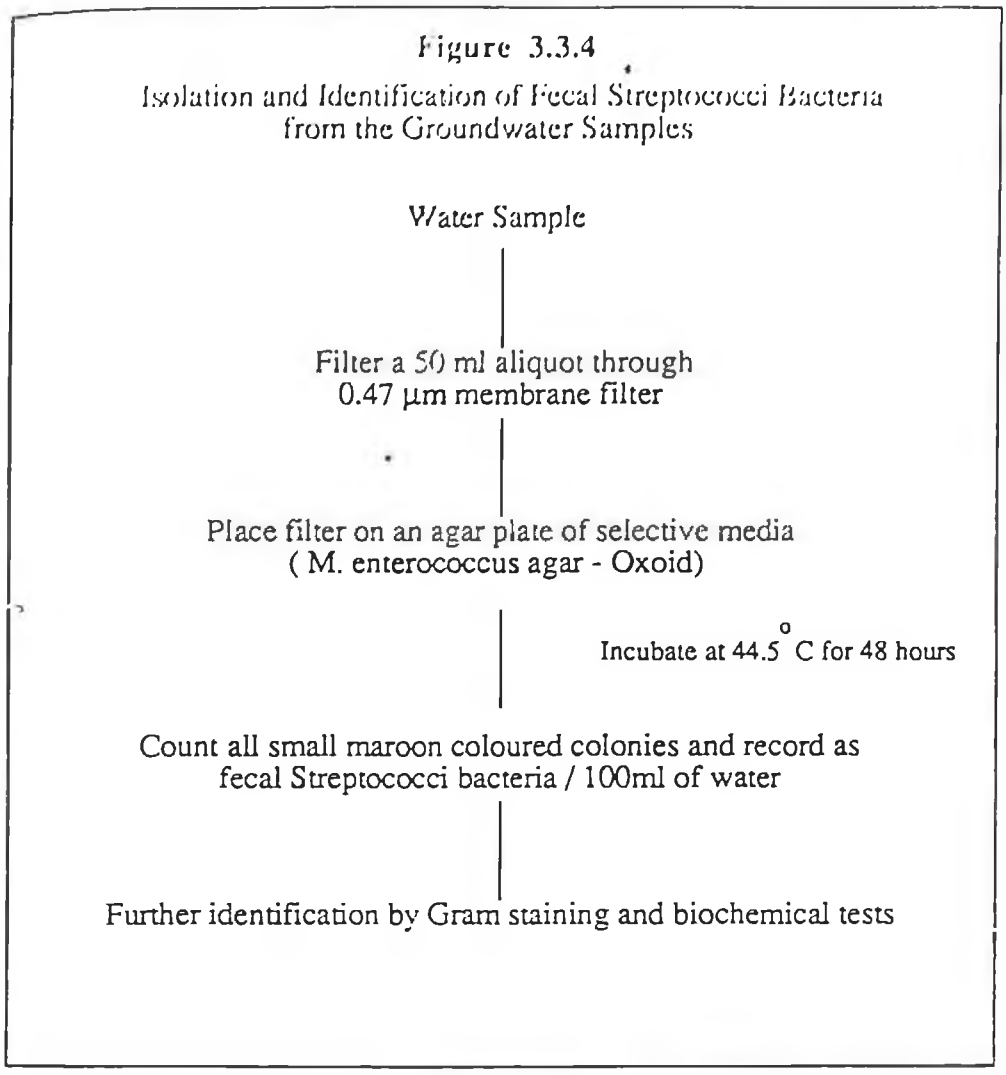

Figure 3.3.5

Isolation and Identification of Sulphite Reducing Clostridia from the Groundwater Samples

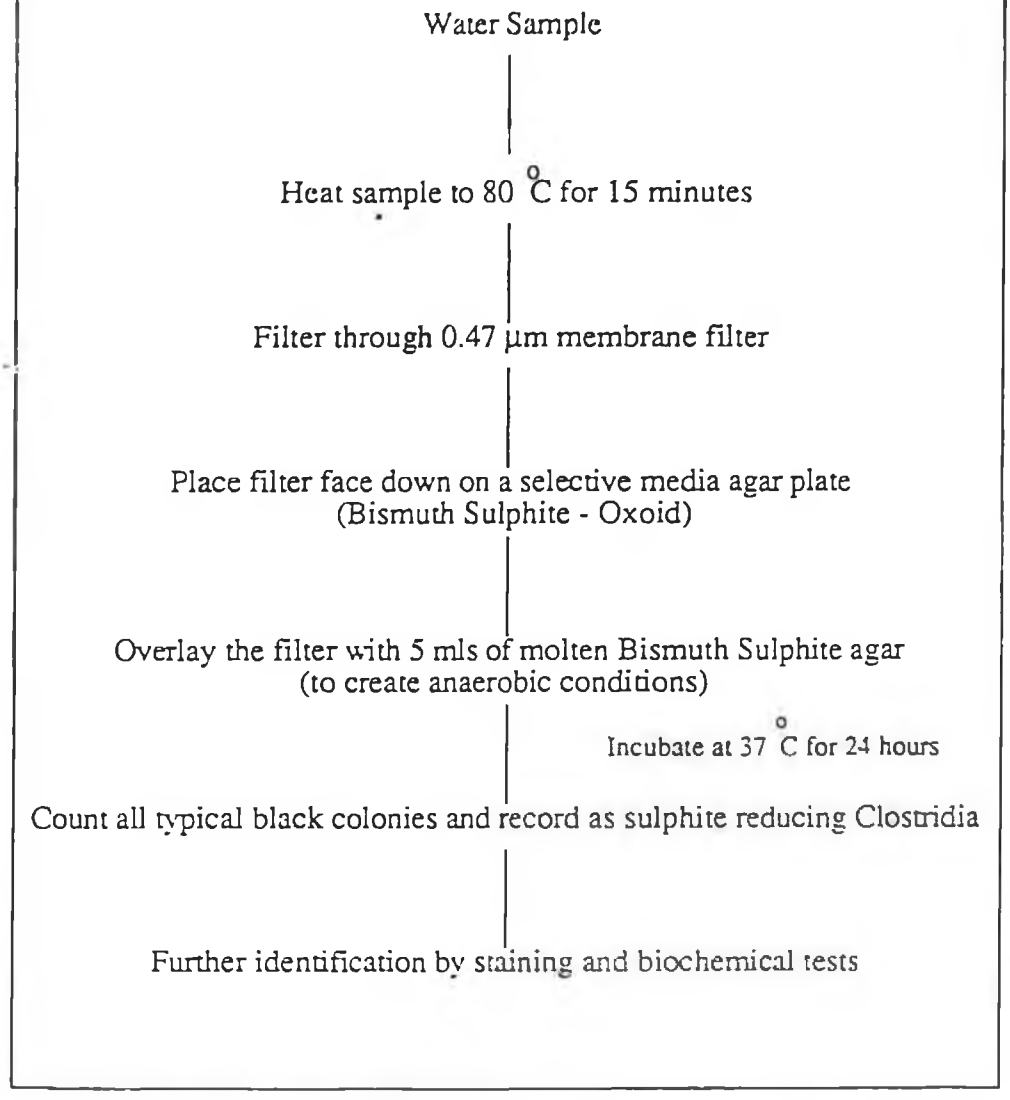

Figure 3.3.6

Isolation and Identification of Salmonella Species from the Groundwater Samples

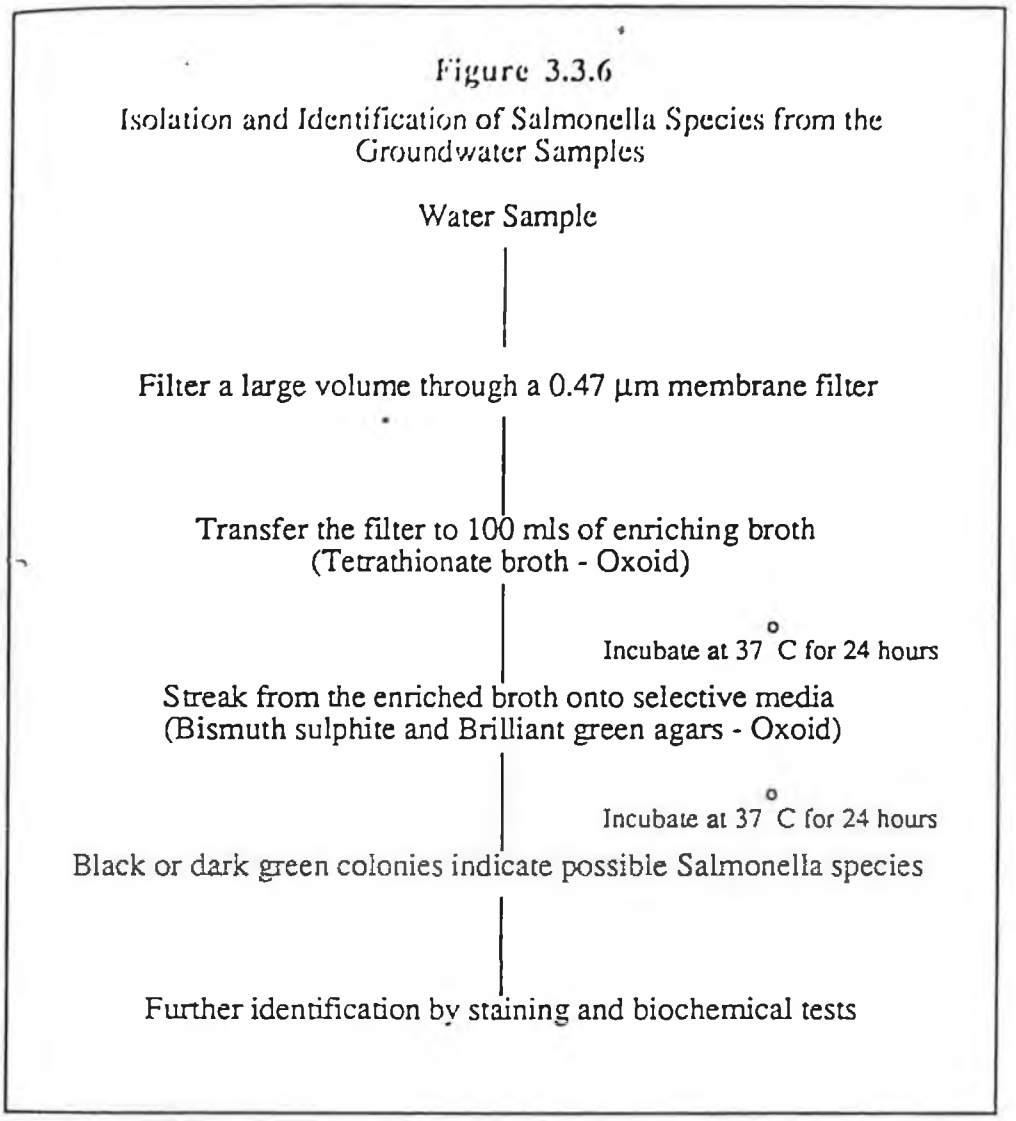

Plate 3.3.1
Geological Survey of Ireland Mobile Drilling Rig

Plate 3.3.2
Monitoring Well - Head Protection

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3:4 Site Characteristics

A total of five sampling locations in the County Sligo area were used in the study. These included three test sites (Sites 1, 2 and 3) and two control stations (C1 and C2), the grid references of which are given in Table 3.4.1 (p144).

Each of the test sites consisted of a two to four year old dwelling with a septic tank and soil treatment system serving a population of at least four people. The sites were chosen because of differences in the thickness and nature of the unsaturated zone available for effluent attenuation. Other variables which could affect groundwater quality such as the construction, design and maintenance of the system, the nature of the waste and the proximity of other polluting sources were similar at all three sites.

3:4.1 Sites 1 and 2

Both sites are located on the Knocknarea peninsula to the south - west of Sligo town. The area is bounded to the north and south by Sligo and Ballysadare bays respectively and occupies an area of approximately 20 square kilometres (Figure 3.4.1, p150).

The centre of the peninsula is dominated by undulating ' kame and kettle ' topography with varying thicknesses (0.0 to 15.0 metres) of outwash deposits from the Midlandian Glaciation which ended circa 13,000 years ago. These deposits consist mainly of sands and gravels with some clays and were laid down by the ice sheets as they retreated. The land slopes gently southwards to Ballysadare bay. Here shallow soils, mainly degraded podzolics, dominate varying in thickness from 0.0 to 3.0 metres. Some low lying parts of this area are subject to periodic flooding especially in the winter months.

The geology of the area is predominantly limestone from the Lower Carboniferous period. The rock, described as Dartry limestone by Oswald (1955), is a massive well jointed grey crinoidal rock devoid of shale and containing nodules of black or grey chert. It is often locally dolomitised and it's relative purity makes it particularly susceptible to karstification.

Annual rainfall in the area is in the region of 1200 mm of which 75% (900 mm) infiltrates as groundwater recharge. Few data are available on the hydrogeology of the area. A study on the groundwater potential in the area around Sligo town was carried out in 1974. A total of ten boreholes were drilled, two of which were in the vicinity of the Knocknarea peninsula. Only small amounts of water were encountered in the two boreholes. The results, presented in an unpublished Geological Survey of Ireland report (Daly, 1975), note that the groundwater potential of the limestones was ' very disappointing '. However, until recently, most of the

Table 3.4.1

Table Showing the Townlands and Grid References of the Sampling Sites

Site	Townland	Grid Reference
Site 1	Kilmacowen	G 662 307
Site 2	Knocknahur	G 646 333
Site 3	Cregg	G 653 395
C1	Tobernaveen	G 662 342
C2	Cregg	G 653 340

Knocknarea area was not served by mains water supply and groundwater sources were used for both domestic and agricultural purposes. Local people report many high yielding and reliable wells (both dug and bored). Much of the area has since been linked to the Sligo County Council mains water although a number of householders have chosen to retain their groundwater supplies.

Grassland farming dominates in the area although some dairy and to a lesser extent sheep farming is also practised. In recent years there has been a marked increase in residential development. This is reflected by a sharp rise in population. For example in the Kilmacowen electoral area there has been a 73% increase in the period from 1971 to 1986 (Central Statistics Office, 1981 and 1986). The area is not served by a municipal sewerage system and consequently this growth in population has resulted in a marked increase in the number of septic tank systems in use.

A large rising in the townland of Tobernaveen towards the centre of the Knocknarea peninsula (C1 in Figure 3.4.1, p150) was used as the control station for both sites 1 and 2. The spring drains the eastern end of the peninsula and is located in an area of undulating topography with outwash sands, gravels and glacial till overburden overlying fissured Carboniferous limestone bedrock. It is partially fed by concentrated recharge into the underlying fissured limestone aquifer (with a velocity varying from 36 metres/hour at low stage to 99 metres/hour at high flow) and partially by diffuse recharge through the overlying glacial deposits (Thorn and Coxon, 1989). The spring has a discharge of 60 to 70 litres per second in high flow and 10 to 15 litres per second in low flow conditions. Previous groundwater investigations in the area have shown that the spring water is of good quality although subject to periodical Coliform bacterial contamination. The groundwater from the spring is representative of the general water quality in the Carrowmore peninsula area and there are no septic tank systems in use within a 100 metre radius.

3:4.1.1 Site 1 (Kilmacowen)

Site 1 in the townland of Kilmacowen is located two kilometres north of Ballysadare village on the L 132 link road. The site consists of a two year old dwelling served by a septic tank treatment system.

The septic tank is a prefabricated concrete structure (3.0 x 1.5 x 2.0 metres in dimension). The tank is not constructed to recognised standard specifications such as I. I. R. S. S. R. 6 (Anon., 1975) and was not inspected, maintained or desludged during the study period. It serves a household of three adults and one infant and receives both sewage and sullage household waste. The effluent from the tank is channelled into a soakage pit (3.0 x 2.0 x 2.0 metres) filled with coarse gravel and building rubble. Because of the high water table (see overleaf) the soakage pit is

partially submerged in water for most of the year and consequently the effluent is discharged directly to groundwater with no effective attenuation. A survey of the ground levels at the site shows that the surface slopes gently away from the septic tank towards the house. Drilling revealed a 2.0 to 2.5 metre thickness of brown loamy soil overlying bedrock. On analysis the soil was found to contain 11% clay, 34% silt and 55% sand and gravel sized particles and is classified as a loamy soil under the U. S. D. A. classification system. Limestone bedrock was encountered at 2.0 to 2.5 metres and drilling continued to 9.0 metres (B2) and 12.0 metres (B1). The rock was extensively weathered with a number of highly dolomotised zones.

Two monitoring boreholes were installed at the site downgradient of the septic tank soil treatment system. The boreholes, named B1 and B2, were located 2.0 metres and 8.1 metres downgradient of the soakage pit (Figure 3.4.2, p151). The number and location of the boreholes installed was restricted by the soft and wet site conditions, which limited the mobility of the drilling rig. Summary statistics on the boreholes are shown in Table 3.4.2 and the detailed borehole logs are presented in Appendix B 1.

The water table in the monitoring boreholes was observed to fluctuate between 0.8 and 2.0 metres during the study period. Transmissivity values of between 4 and 12 m^2/day were estimated from a series of hydrogeological investigations (pumping tests and displacement methods) carried out on the boreholes. However, as described in 3:3, problems were encountered which resulted in insufficient data being collected to ensure accurate analysis. Consequently the results should be treated with some degree of caution.

A site survey showed that the density of septic tank systems within a 450 metre radius of the boreholes is less than ten and that the distance to the nearest polluting source (domestic or agricultural) is 100 metres.

3:4.1.2 Site 2 (Knocknahur)

Site 2 in the townland of Knocknahur North situated two kilometres north west of site 1 and 4.5 kilometres south west of Sligo town (Figure 3.4.1). The site also consists of a two year old dwelling served by a septic tank and soil treatment system.

The septic tank is manufactured from concrete blocks and was constructed in - situ (3.0 x 1.8 x 1.5 metres in dimension). The design of the tank does not comply with any recognised standard specifications such as I. I. R. S. S. R. 6 (Anon., 1975) and it was not inspected, maintained or desludged during the study period. The tank serves a household of two adults and four children and receives both sewage and sullage wastes. The effluent is channelled to a 2.0 metre deep soakage pit backfilled

with local sands and gravels.

Drilling revealed that the area is overlain by 6.0 to 6.5 metres of permeable outwash sands and gravels with large boulders up to 0.5 metres in diameter (Plate 3.4.3, p156). On analysis the overburden was found to consist of 1% clay, 6% silt and 93% sand and gravel sized particles and is classified as a sandy soil under the U. S. D. A. classification system. Limestone bedrock was encountered at 6.0 to 6.5 metres and was found to be significantly less weathered and dolomitised than that described at site 1 (Appendix B 1).

Three boreholes, B3, B4, and B5, were installed 3.85, 4.90 and 10.22 metres downgradient of the treatment system (Figure 3.4.3, p152). The exact location of the monitoring boreholes was limited by the mobility of the drilling rig. B3 was bored to rock head only (a depth of 6.5 metres) therefore only receiving water from the surrounding overburden. It was intended to determine the degree of effluent attenuation taking place in the overburden by monitoring the water quality in this borehole. B4 and B5 were installed into the limestone to a depth of 14.0 and 12.2 metres respectively. Monitoring of these boreholes should give a good indication of the combined effect of effluent attenuation in the overburden and subsequent dilution in bedrock. Summary statistics on the boreholes are shown in Table 3.4.2 (p154) and the detailed borehole logs are presented in Appendix B 1.

The groundwater levels in the monitoring boreholes were observed to fluctuate between 4.7 to 5.6 metres. Calculated transmissivity values ranged from two to 8.0 m^2/day, the higher values being for the sands and gravels and the lower for the underlying limestones. As before, because the tests were of insufficient duration, the data should be treated with caution.

A site survey showed that the density of septic tank systems within a 450 metres radius of the boreholes is 15 and that the distance to the nearest polluting source (domestic or agricultural) is greater than 100 metres.

3:4.2 Site 3

Site 3 is located in the townland of Cregg on the northern side of Sligo bay, 4.5 kilometres north west of Sligo town. The geology of the area is dominated by a metamorphic inlier described in detail by Max (1984). The rocks, composed mainly of metasedimentary schists, are described as Moinian in age and are classified as the Liscarragh formation. These rocks are structurally the lowest unit in the Rosses Point group. No information is available on the nature and distribution of soil/overburden material in the region. An inspection of the area suggests that it is overlain by varying thickness of glacial till (1.0 to 5.0 metres) although there are extensive rock outcrops to the north east of the test site.

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Annual rainfall in the area is 1200 mm of which approximately 800 mm infiltrates as groundwater recharge. No hydrogeological studies have been carried out in this area but, as in the Knocknarea peninsula, locals report some very good yielding bored wells. A large well was bored at Cregg House (a psychiatric hospital) in 1935. The hospital, with over 200 permanent residents, is located 0.5 kilometres north of the test site. This borehole has been reported to yield significant quantities of water and acts as an auxiliary supply to the mains water.

The Cregg /Ballincar area has also witnessed a marked increase in residential development in recent years, particularly concentrated along the L16 link road. The area is not served by a municipal sewerage system and there has been a marked increase in the number of septic tank systems. Farming is dominated by cattle husbandry, although some sheep rearing is practised on the poorer land. Many of the farms have been replaced by residential development and most of the agricultural activity is now located to the east and north of Cregg House away from the L16 link road.

A private bored well in the townland of Cregg was used as the control site. The well (C2 in Figure 3.4.1, p150) is bored into permeable overburden material and metamorphic bedrock to a total depth of 37.5 metres, although the exact thickness of the overburden and metamorphic rock is not known. The metamorphic schist bedrock in the area is referred to as the Cregg House formation by Max (1984). The well is located 0.5 kilometres north west of the test site and serves a private dwelling which uses a septic tank system for sewage disposal. The septic tank is located a distance of 50 metres downgradient from the borehole. There are no other septic tank systems within a 100 metre radius.

3:4.2.1 Site 3 (Cregg)

The site consists of a four year old dwelling served by a septic tank and soil treatment system which has been operation for three years. The septic tank is manufactured from concrete block and was constructed in - situ (3.0 x 1.6 x 1.5 metres). The tank does not comply with standard specifications such as I. I. R. S. S. R. 6 (Anon., 1975) and was not inspected, maintained or desludged during the study period. It serves a household of four adults and receives both sewage and sullage household wastes. The effluent is channelled into a distribution box and passes into three 1.0 metre deep distribution trenches 5.0 metres in length and 1.5 metres apart (the distribution field is not constructed to recognised specifications).

Drilling revealed 2.5 metres of glacial till which on analysis was found to consist of 7% clay, 30% silt and 63% sand and gravel sized particles. The soil is described as a sandy loam under the U. S. D. A. classification system. Bedrock was

encountered at 6.0 to 6.5 metres during the drilling process and was extremely weathered to a considerable depth. In B6, for example, this weathered zone was found to penetrate to a depth of 3.0 metres. However, the rock in B8 was found to be much tighter with significantly less fissuring (Appendix B 1). It was not possible to obtain a sample of the overburden material between 2.5 and 6.5 metres. The material was described by the drillers as permeable sands and gravels in the first 2.5 metres overlying a further 1.0 to 1.5 metres of very coarse gravels and cobbles (probably highly weathered gruss of parent rock origin).

Three monitoring boreholes B6, B7 and B8 were installed at the site downgradient of the soil treatment system. Boreholes B6 and B7 were installed 1.5 metres downgradient of the percolation field. B7 was installed to within 1.0 metre of bedrock at a depth of 5.5 metres, while B6 and B8 were installed into the bedrock to a total depth of 13.0 and 11.3 metres respectively (Figure 3.4.4, p153). As at site 2 it was intended to examine separately the effluent attenuation in the overburden (B7) and the combined effect of both overburden attenuation and dilution in bedrock (B6 and B8). B8 was installed a lateral distance of 9.77 metres downgradient of the soakage pit. Summary statistics on the boreholes are shown in Table 3.4.2 (p154) while the more detailed borehole logs are presented in Appendix B 1.

Hydrogeological investigations at the site were largely unsuccessful as pumping and displacement tests yielded insufficient data for accurate analysis. A two hour pumping test at monitoring borehole B8, pumping at a rate of 318 litres per hour, resulted in less than two centimetres drawdown in the water level, substantiating local reports of high yielding wells. The water table in the monitoring wells was found to fluctuate between 3.0 metres and 4.9 metres beneath ground level during the period of the study.

A site survey showed that the density of septic tank systems within a 450 metre radius of the boreholes is 15 and that the distance to the nearest polluting source (domestic or agricultural) is greater than 100 metres.

Figure 3.4.1

Map of Part of County Sligo Showing the Location of the Sampling Sites

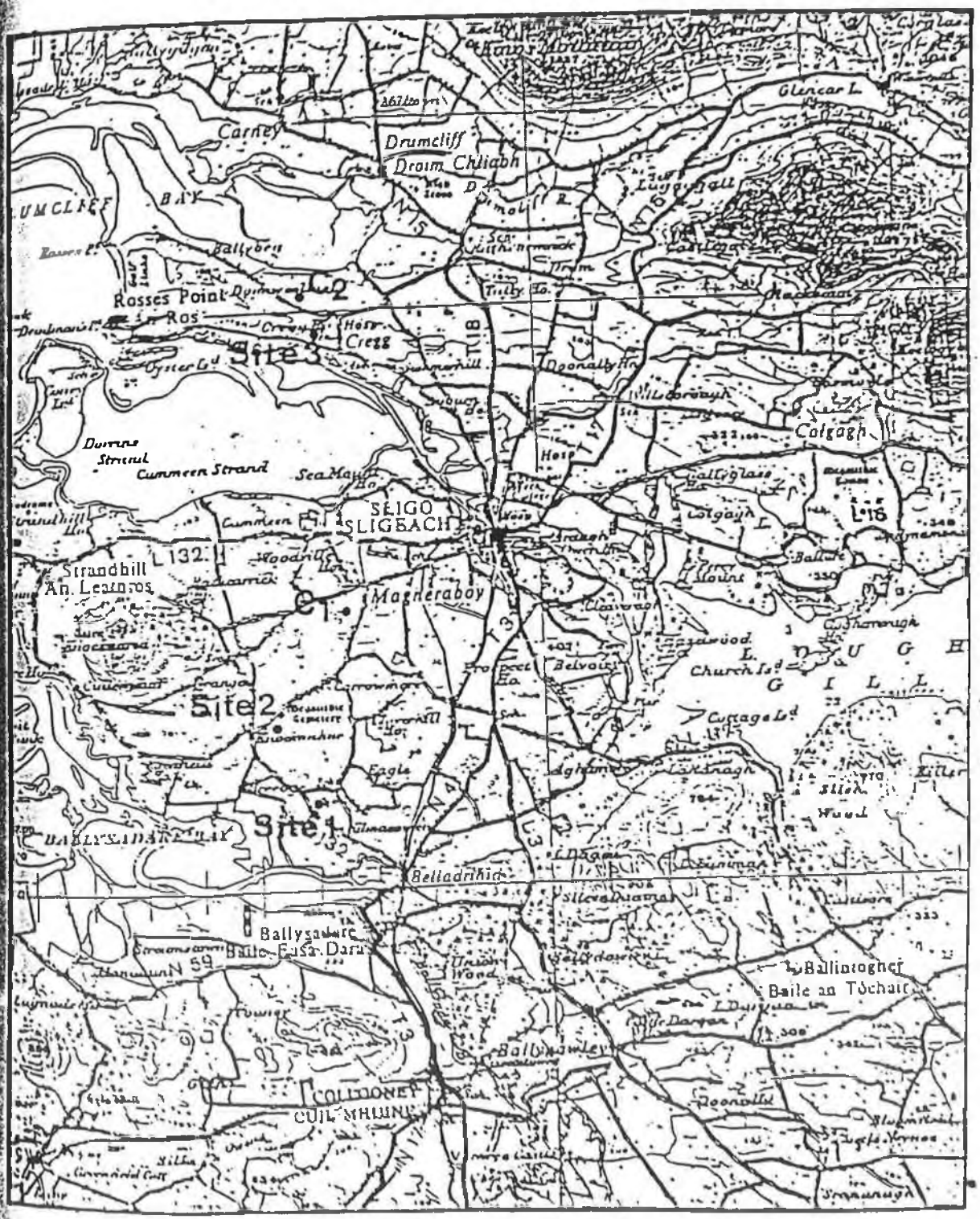

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Figure 3.4.2

A Schematic Representation of the Septic Tank Treatment System at Site 1(Kilmacowen) Showing the Location of the Monitoring Boreholes B1 and B2

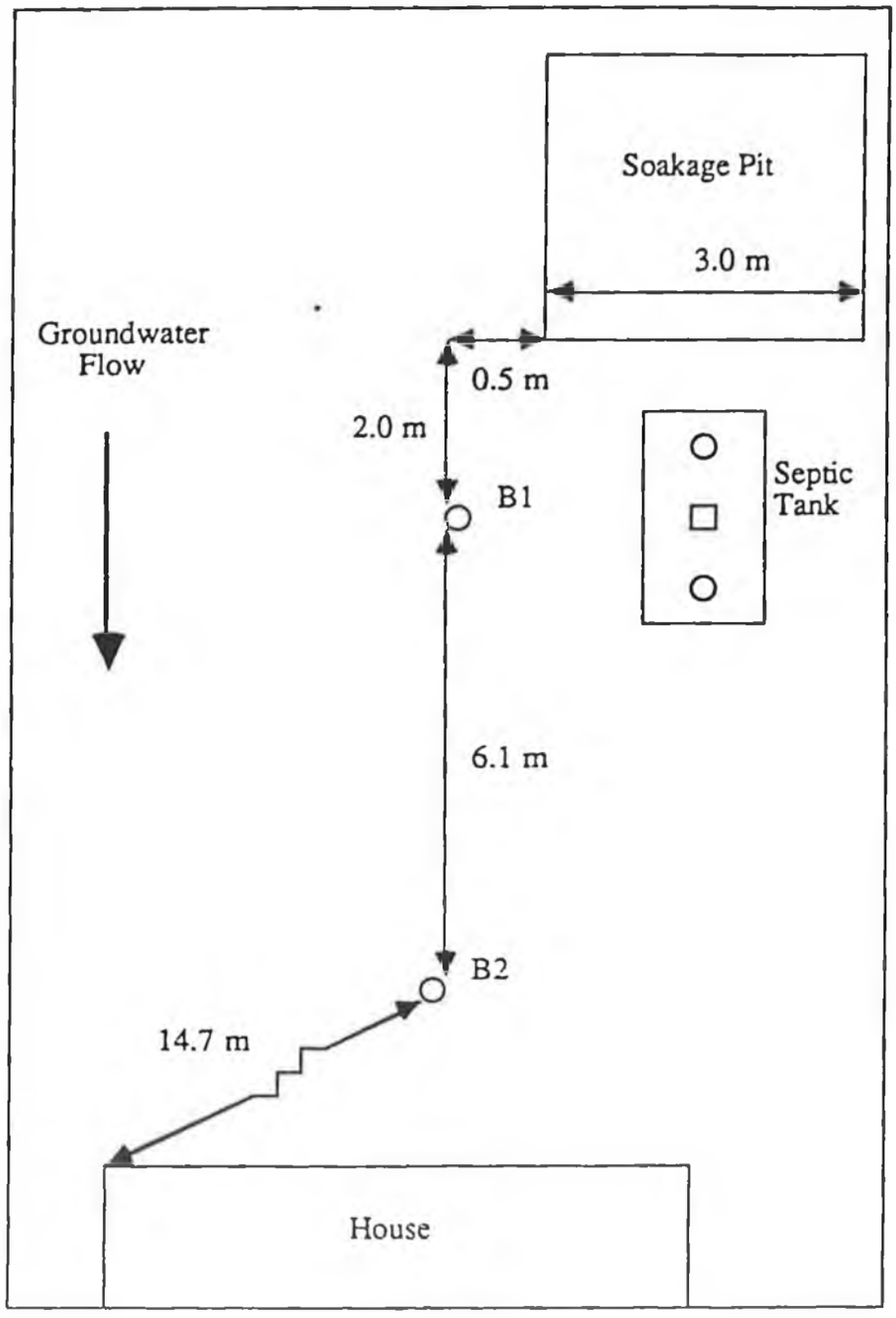

Figure 3.4.3

A Schematic Representation of the Septic Tank Treatment System at Site 2 (Knocknahur) Showing the Location of the Monitoring Boreholes B3, B4 and B5

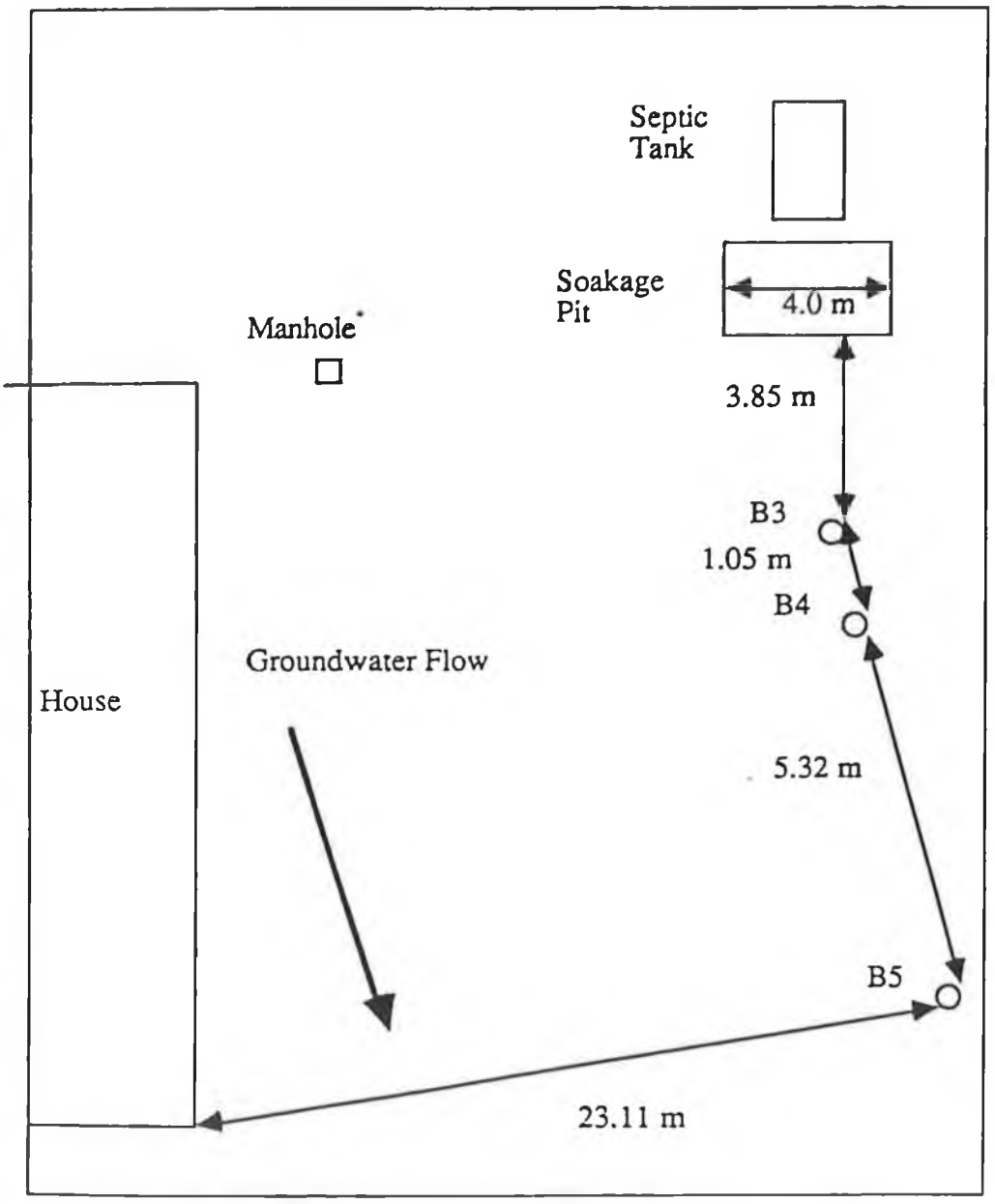

Figure 3.4.4

A Schematic Representation of the Septic Tank Treatment System at Site 3 (Cregg) Showing the Location of the Monitoring Boreholes B6, B7 and B8

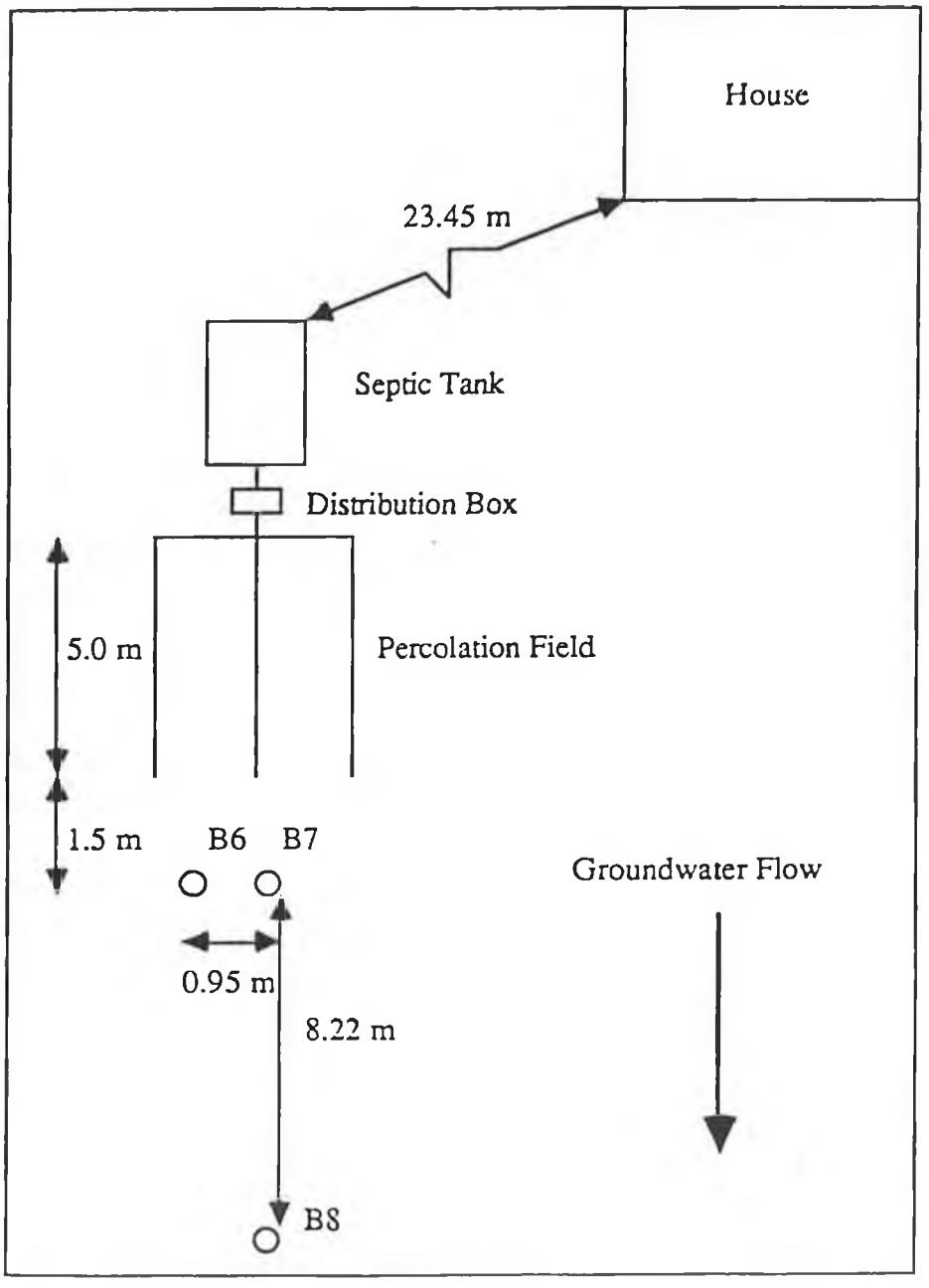

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Table 3.4.2

Summary Details on the Groundwater Monitoring Boreholes

	Type of Overburden	Depth to Rock (m)	Rock Type	Static Water Level S. W. L. (March 1988)	Total Depth of Well (m)	Casing (Slotted from)	Drilling Method
B1	brown clay loam	2.0	limestone	1.86	12.0	55 mm wavin ducting (2.0m)	rotary
B2	brown clay loam	2.0	limestone	1.36	9.0	55 mm wavin ducting (2.0m)	rotary
B3	coarse sands & gravels with large boulders	6.5	limestone	5.25	6.5	55 mm wavin ducting (3.0m)	rotary
B4	coarse sands & gravels with large boulders	6.5	limestone	5.25	14.0	55 mm wavin ducting (3.0m)	rotary
B5	coarse sands & gravels with large boulders	6.0	limestone	5.38	12.2	55 mm wavin ducting (3.0m)	rotary
B6	brown loamy soil overlying coarse gravels	6.5	schists	3.00	13.0	55 mm wavin ducting (3.0m)	rotary
B7	brown loamy soil overlying coarse gravels	6.5	schists	3.00	5.5	55 mm wavin ducting (3.0m)	rotary
B8	brown loamy soil overlying coarse gravels	6.0	schists	3.30	11.3	55 mm wavin ducting (3.0m)	rotary

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Plate 3.4.1

Site 1 at Kilmacowen, County Sligo

Plate 3.4.2

Site 2 at Knocknahur, County Sligo.

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Plate 3.4.3

The Outwash Sands and Gravels with Large Boulders Found in the Knocknahur Area

Plate 3.4.4

Site 3 at Cregg, County Sligo

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3:5 Results

3:5.1 Presentation of Results

The results of the investigation are presented in four sections as follows:

3:5.1.1 Soil/Overburden Analysis

The results of analyses of the soil/overburden material beneath the septic tank disposal systems are presented in this section

3:5.1.2 The Contamination of Groundwater Downgradient of Septic Tank Systems

In this section the chemical and microbiological quality of the groundwater in the monitoring boreholes at the test sites is presented in graphical form. The rainfall events and the water level fluctuations in the monitoring boreholes during the sampling period are also presented

3:5.1.3 Effluent Attenuation at the Three Test Sites

The reduction in the concentration of the effluent constituents from the septic tank to the monitoring boreholes downgradient of the soil treatment system was calculated on three of the sampling occasions. The results are presented as percentage reductions in Tables 3.5.5 to 3.5.7 (p184 to 186)

3:5.1.4 Statistical Analyses

The results presented were statistically analysed using the Wilcoxon Signed Rank test. The relationship between increased bacterial numbers in groundwater samples and rainfall events was investigated using Pearson's correlation method. The significance of the correlation was tested using the t - statistic. Summary results are presented in Tables 3.5.8 to 3.5.12 (p188 to 192).

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Table 3.5.1

Particle Size Distribution of the Overburden Material

SITE	PARTICLE SIZE											
	CLAY			SILT			SAND			GRAVEL		
	< 0.02mm			0.002 - 0.06mm			0.06 - 2.0mm			> 2.0mm		
	Sample Depth (m)											
	1m	2m	3m	1m	2m	3m	1m	2m	3m	1m	2m	3m
Site 1	11	9	-	34	40	-	44	41	-	11	10	-
Site 2	1	1	1	6	6	7	47	50	41	46	43	51
Site 3**	7	9	6	30	36	33	40	35	42	23	20	19

** At this site samples were taken at 1.0m, 2.0m and 2.5m depths

Table 3.5.2

Physical and Chemical Properties of the Overburden Material

PARAMETERS	SITE 1			SITE 2			SITE 3		
	Sample Depth (m)								
	1m	2m	3m	1m	2m	3m	1m	2m	2.5m
pH (pH units)	7.31	7.36	-	7.51	7.49	7.48	7.35	7.46	7.41
Porosity (%)	41.0	39.0	-	28.0	25.0	23.0	38.0	36.0	33.0
Organic Matter (%)	18.3	13.6	-	5.2	4.8	4.3	12.9	10.2	9.0
Organic Carbon (%)	6.9	5.3	-	1.9	1.6	1.5	5.2	4.4	3.9
Permeability (mm/s)	0.029			0.125			0.040		
C. E. C. (meq/100g)	35.0	32.0	-	4.8	3.6	3.1	28.0	23.0	26.0
Moisture Content (%)	23.5	26.3	-	3.0	4.1	3.8	19.2	19.8	19.8

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3.5.1.2 The Contamination of Groundwater Downgradient of Septic Tank Treatment Systems

The results of analysis of the groundwater samples at the five sampling sites are presented in Appendix B 5. In this section the chemical and microbiological quality of the groundwater in the monitoring boreholes compared to background levels is presented in graphical form. The rainfall events during the sampling period and the water level fluctuations in the monitoring boreholes are also presented graphically. Detergent concentrations, sulphite reducing Clostridia and Salmonella bacterial numbers isolated from the groundwater samples are shown in Tables 3.5.3 and 3.5.4 (p182). The results are presented as follows:

(i) The chemical and microbiological quality of the groundwater in the monitoring boreholes at the three test sites compared to background concentrations is detailed in Figures 3.5.1 to 3.5.41 (p161 to 174), as follows:

Figures 3.5.1 to 3.5.3	pH
Figures 3.5.4 to 3.5.5	C. O. D.
Figures 3.5.6 to 3.5.8	B. O. D.
Figures 3.5.8 to 3.5.11	Conductivity
Figures 3.5.12 to 3.5.14	Chloride
Figures 3.5.15 to 3.5.17	Nitrate
Figures 3.5.18 to 3.5.20	Ammonia
Figures 3.5.21 to 3.5.23	Phosphate
Figures 3.5.24 to 3.5.26	Sodium
Figures 3.5.27 to 3.5.29	Potassium
Figures 3.5.30 to 3.5.32	K/Na
Figures 3.5.33 to 3.5.35	Total Coliform bacteria
Figures 3.5.36 to 3.5.38	Fecal Coliform bacteria
Figures 3.5.39 to 3.5.41	Fecal Streptococci bacteria

(ii) The relationship between the nitrate concentrations recorded and the ammonia, chloride and fecal Coliform bacteria levels is shown in Figures 3.5.42 to 3.5.50 (p175 to 177), as follows:

Figures 3.5.42 to 3.5.44	nitrate - N vs ammonia - N
Figures 3.5.45 to 3.5.47	nitrate - N vs chloride
Figures 3.5.48 to 3.5.50	nitrate - N vs fecal Coliform bacteria

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(iii) The numbers of fecal bacteria isolated from the monitoring boreholes are plotted against the total rainfall (mm) for the five days preceding sampling in Figures 3.5.51 to 3.5.56 (p178 to 179):

Figures 3.5.51 to 3.5.53 Fecal Coliform bacteria

Figures 3.5.54 to 3.5.56 Fecal Streptococci bacteria.

(iv) The monthly rainfall (mm) during the sampling period is plotted in Figure 3.5.57 (p180) while the total rainfall (mm) for the five day period preceding sampling is presented in Figure 3.5.58 (p180).

(v) The fluctuation in water levels in the monitoring boreholes during the sampling period is given in Figure 3.5.59 (p189).

(vi) Table 3.5.3 (p182) gives details of the concentrations of L. A. S. detergent (as M. B. A. S.) recorded in the groundwater samples on two of the sampling dates. The numbers of sulphite reducing Clostridia and Salmonella bacteria isolated from the groundwater samples are given in Table 3.5.4 (p182).

Figure 3.5.1
pH concentration detected in the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

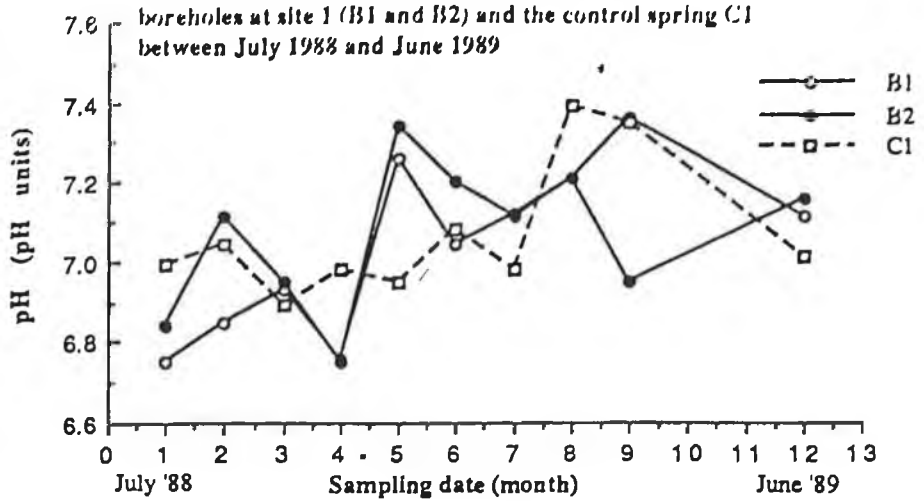

Figure 3.5.2
pH concentration detected in the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

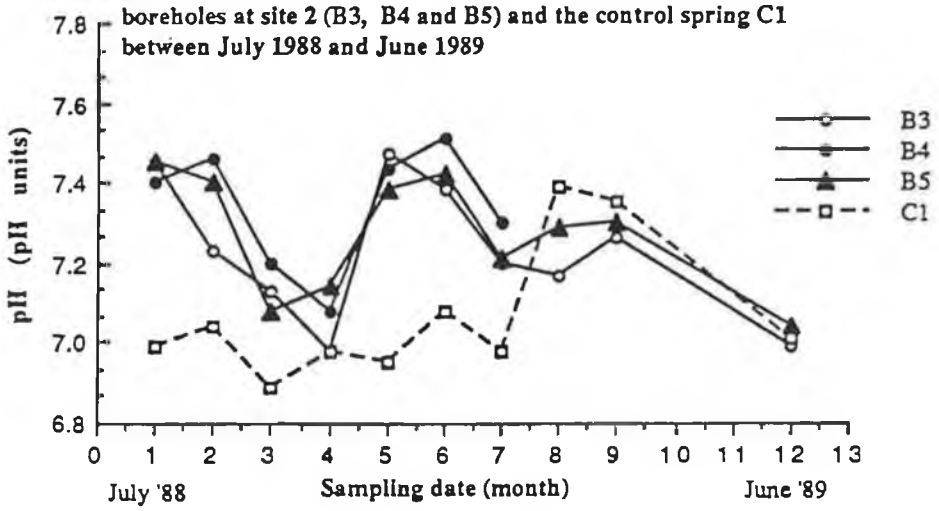

Figure 3.5.3
pH concentration detected in the monitoring boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

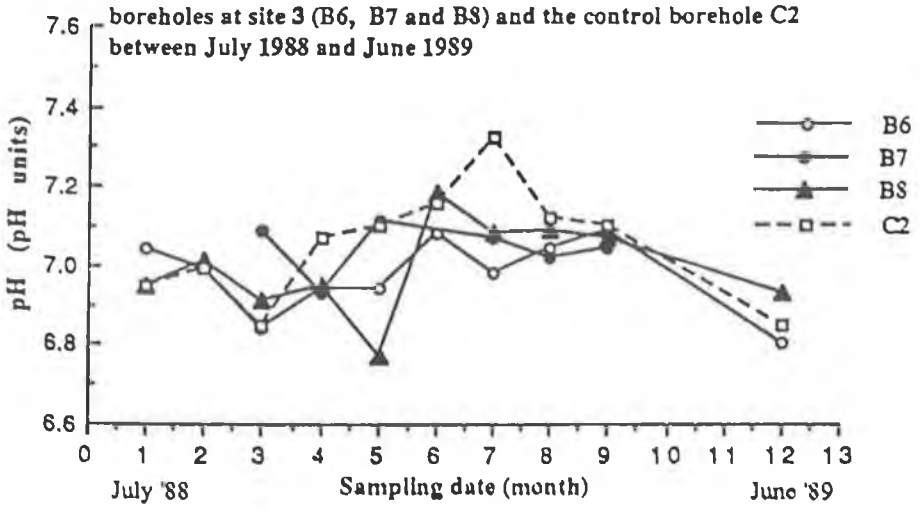

Figure 3.5.4
B.O.D. concentration (mg/l) detected in the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

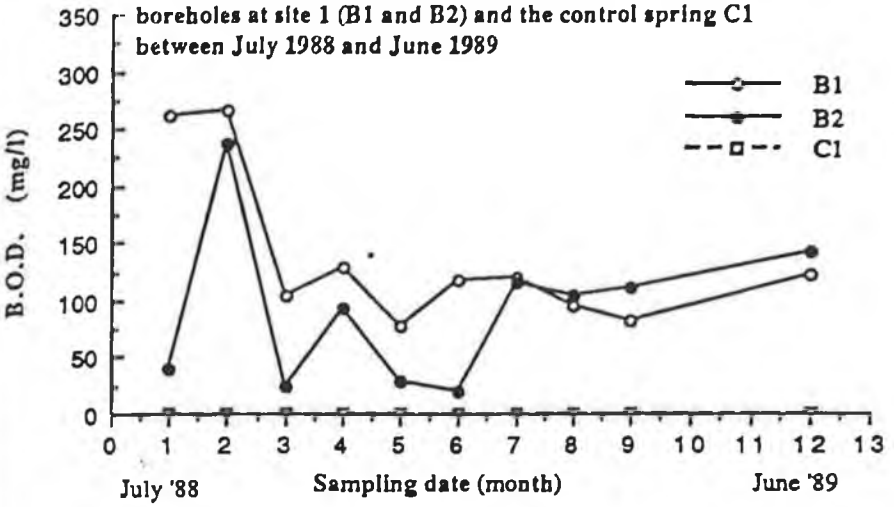

Figure 3.5.5
B.O.D. concentration (mg/l) detected in the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

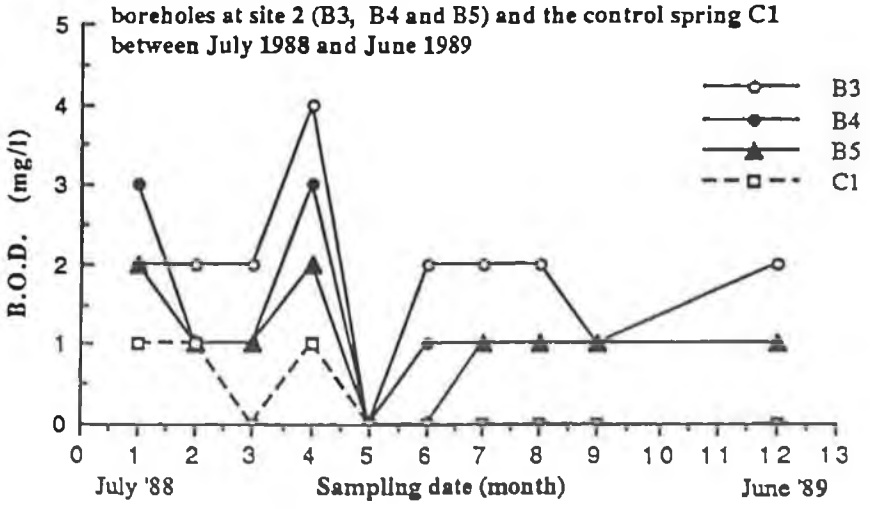

Figure 3.5.6
C.O.D. (mg/l) concentration detected in the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

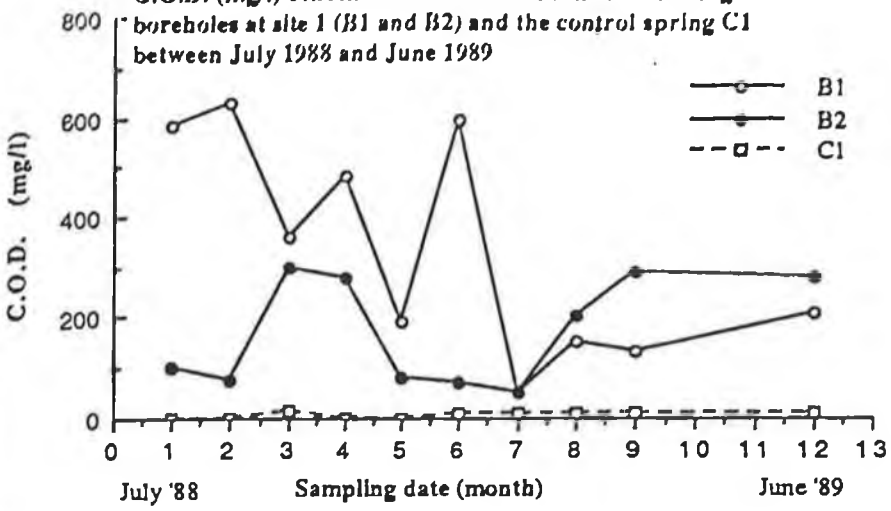

Figure 3.5.7
C.O.D. (mg/l) concentration detected in the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

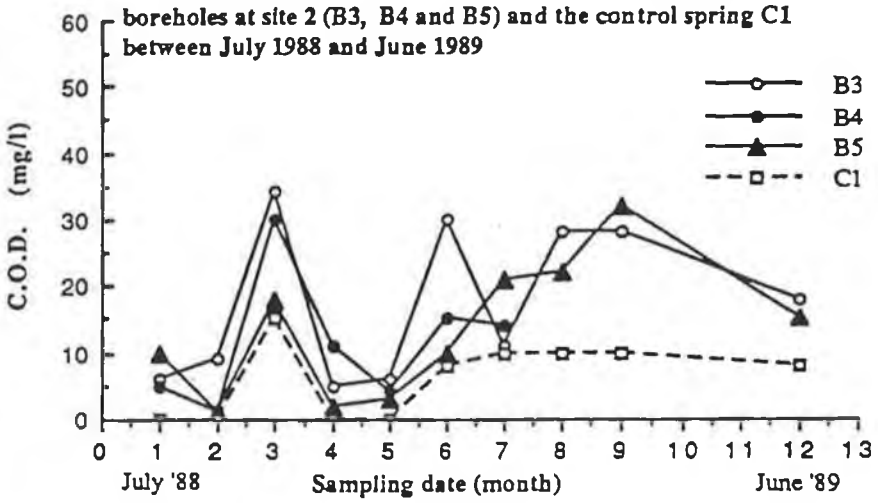

Figure 3.5.8
C.O.D. (mg/l) concentration detected in the monitoring boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

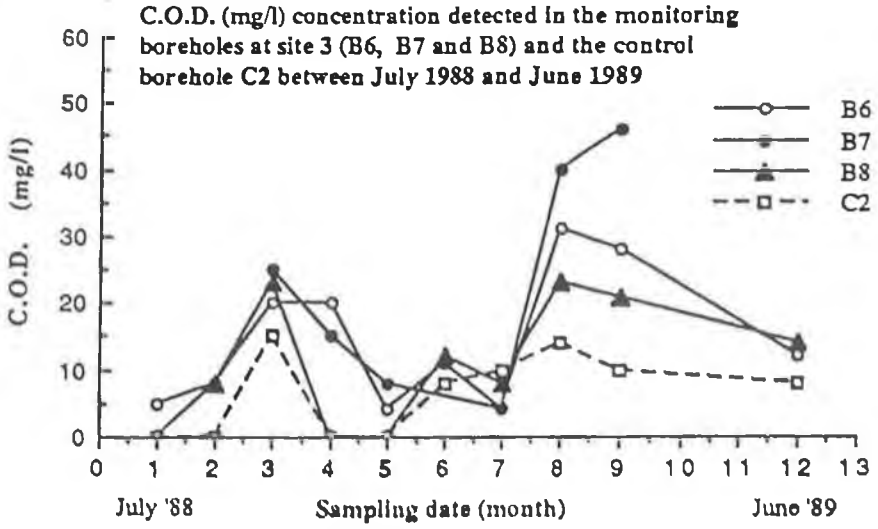

Figure 3.5.9
Conductivity (micro S/cm) detected in the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

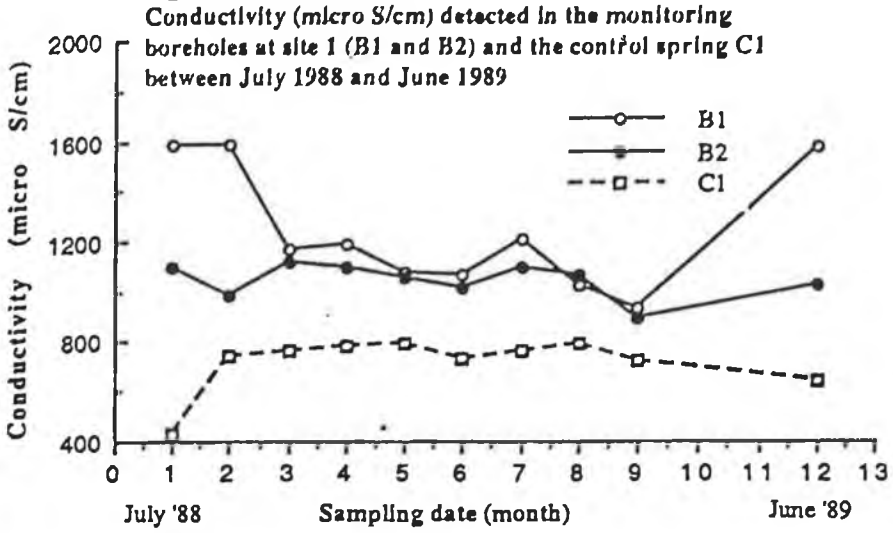

Figure 3.5.10
Conductivity (micro S/cm) detected in the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

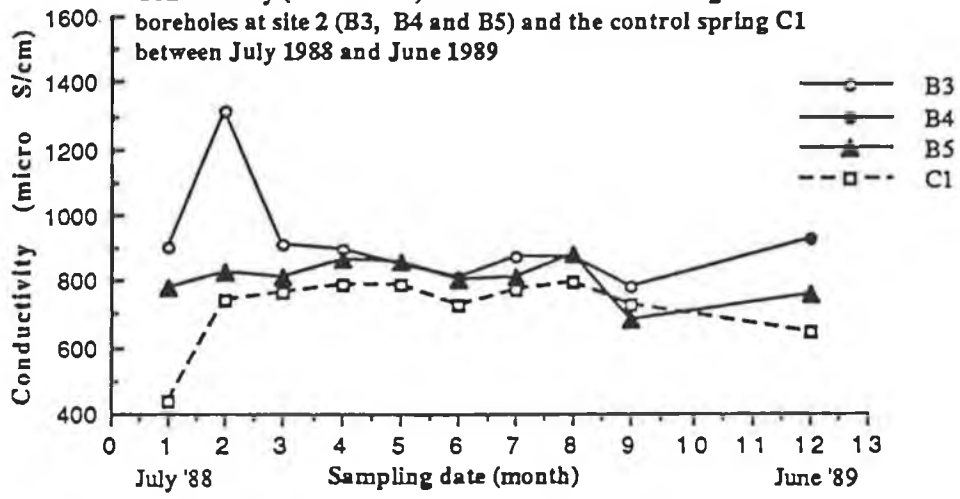

Figure 3.5.11
Conductivity (micro S/cm) detected in the monitoring boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

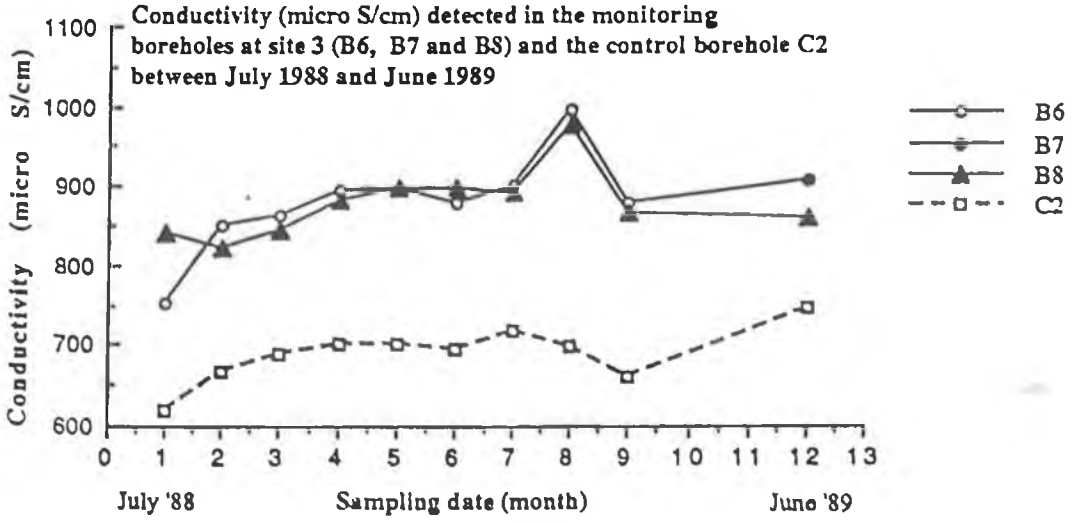

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Figure 3.5.12
Chloride concentration (mg/l) detected in the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

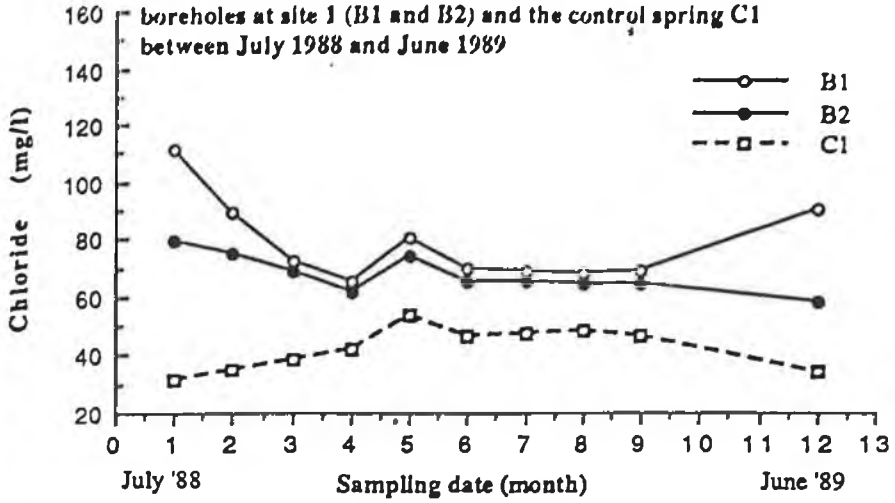

Figure 3.5.13
Chloride concentration (mg/l) detected in the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

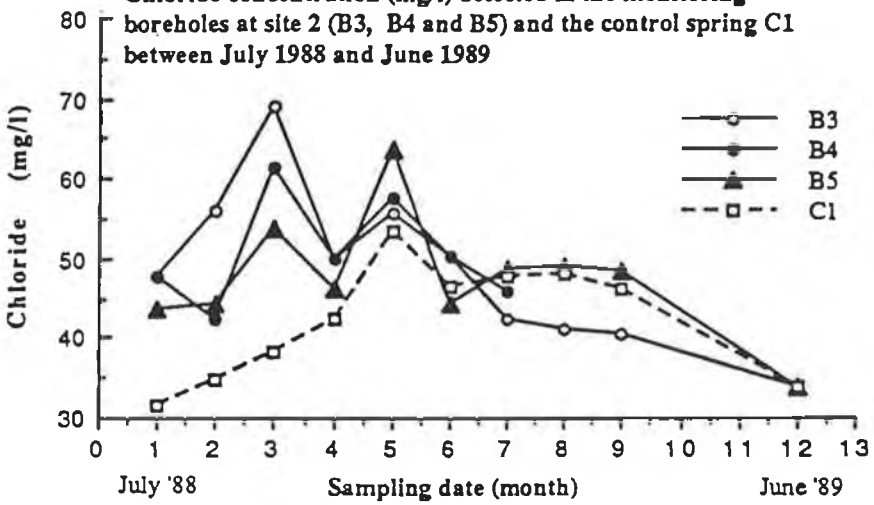

Figure 3.5.14
Chloride concentration (mg/l) detected in the monitoring boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

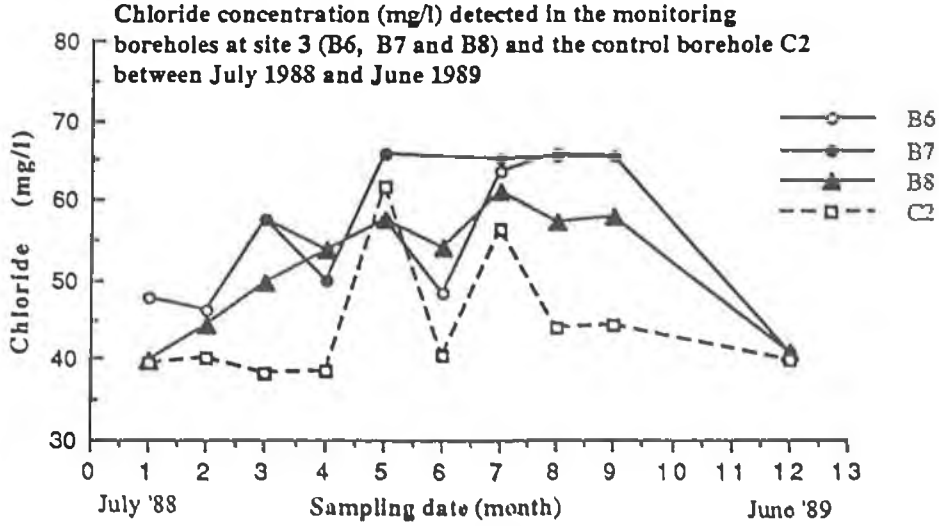

Figure 3.5.15
Nitrate - N concentration (mg/l) detected in the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

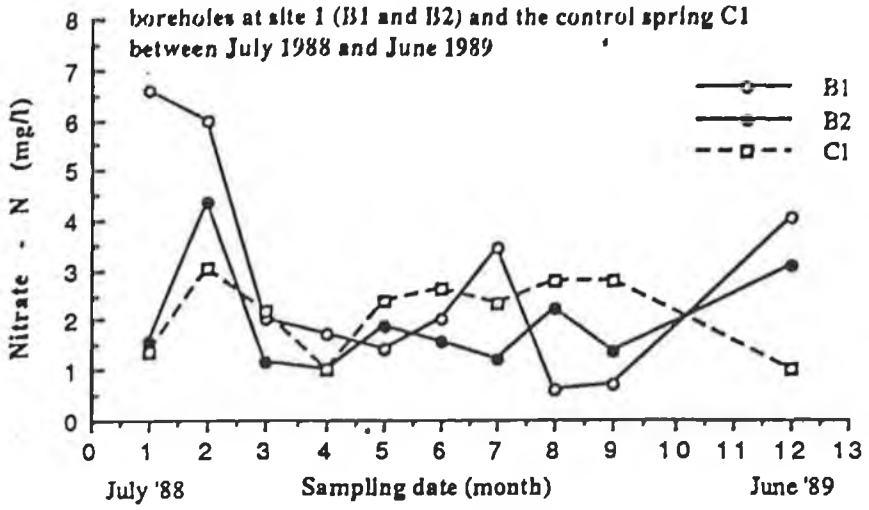

Figure 3.5.16
Nitrate - N concentration (mg/l) detected in the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

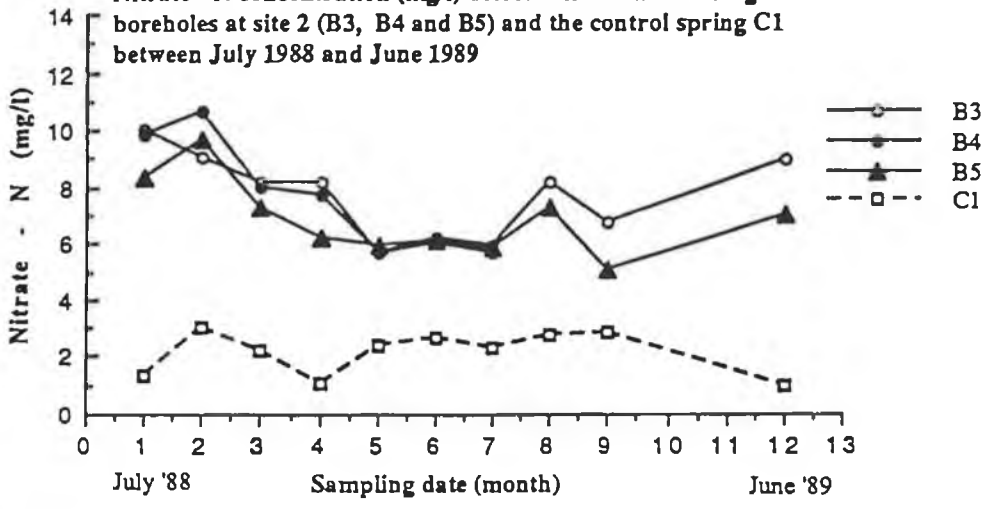

Figure 3.5.17
Nitrate - N concentration (mg/l) detected in the monitoring boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

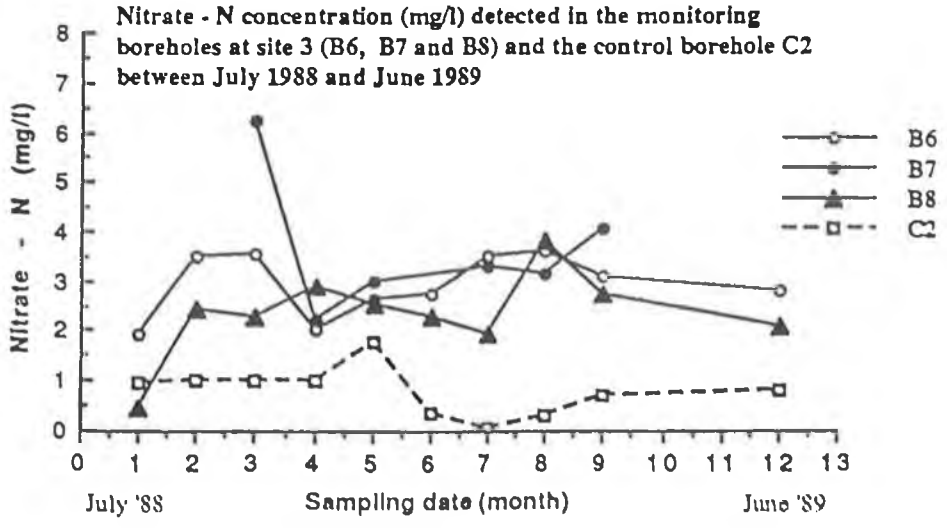

Figure 3.5.18
Ammonia - N concentration (mg/l) detected in the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

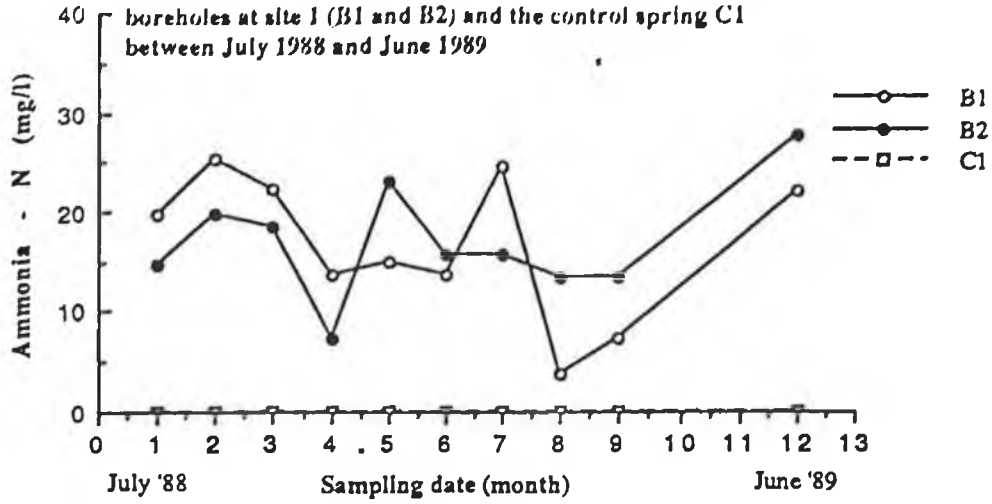

Figure 3.5.19
Ammonia - N concentration (mg/l) detected in the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

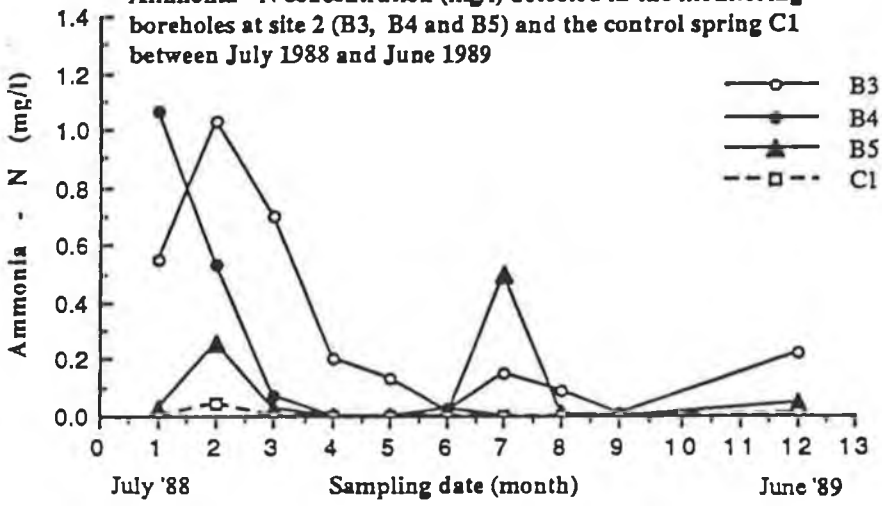

Figure 3.5.20
Ammonia - N concentration (mg/l) detected in the monitoring boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

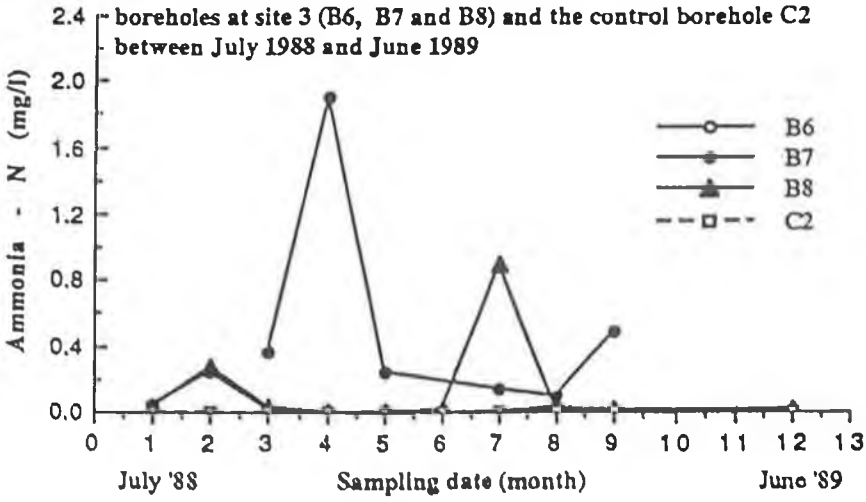

Figure 3.5.21
Phosphate - P concentration (mg/l) detected in the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

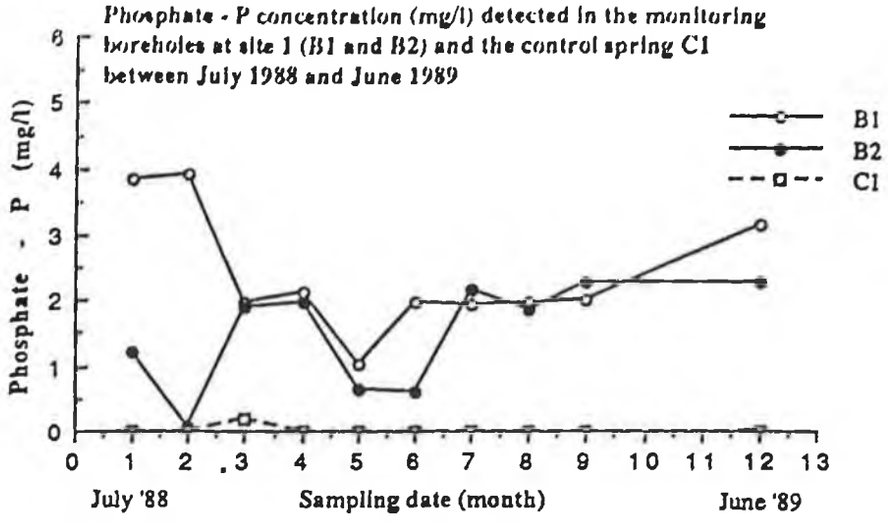

Figure 3.5.22
Phosphate - P concentration (mg/l) detected in the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

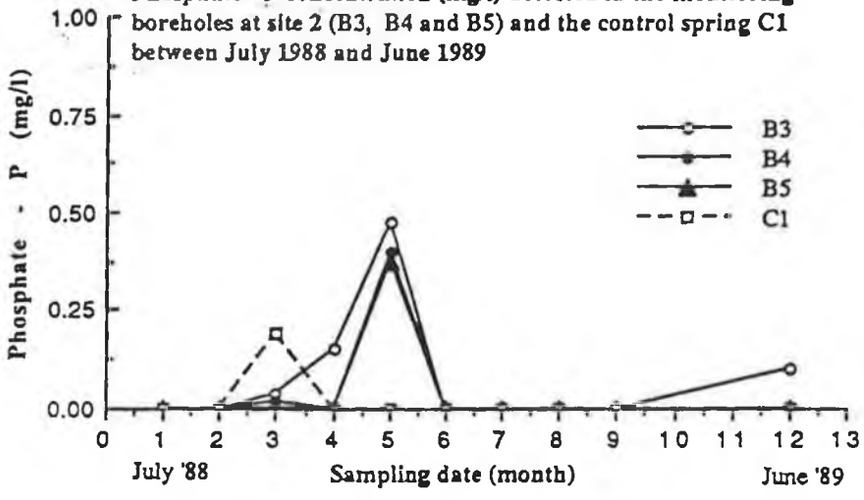

Figure 3.5.23
Phosphate - P (mg/l) concentration detected in the monitoring boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

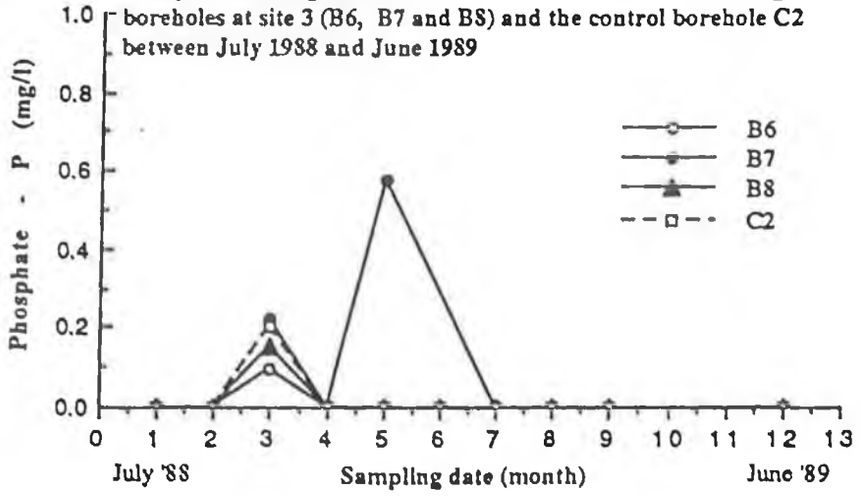

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Figure 3.5.24
Sodium concentration (mg/l) detected in the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

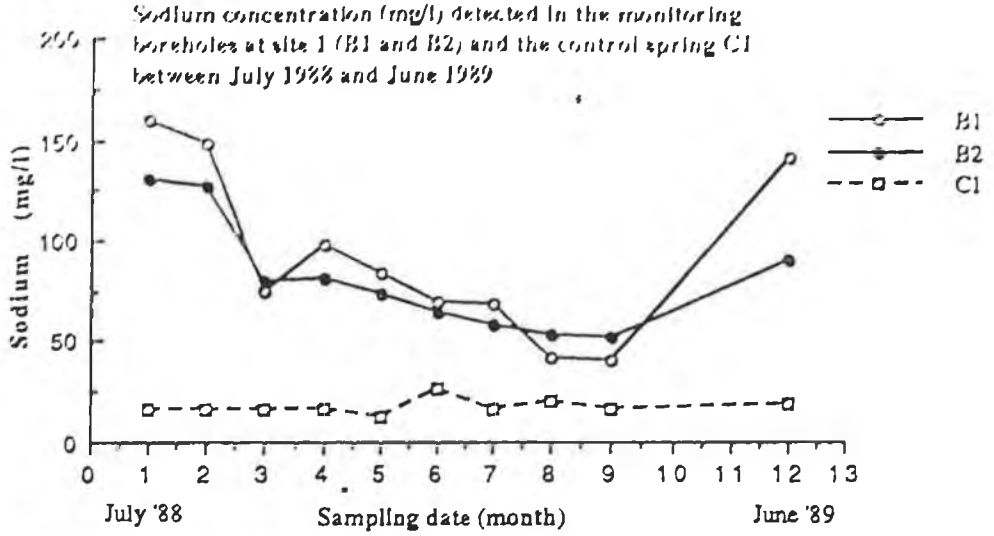

Figure 3.5.25
Sodium concentration (mg/l) detected in the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

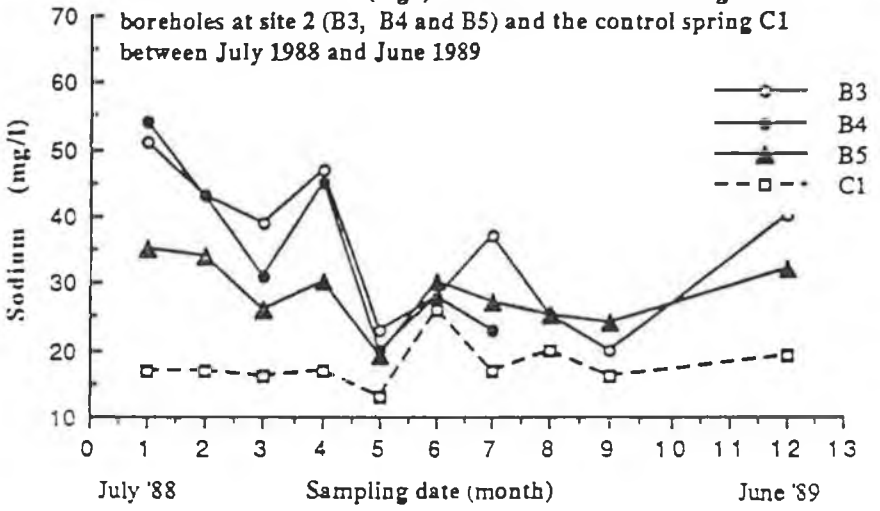

Figure 3.5.26
Sodium concentration (mg/l) detected in the monitoring boreholes at site 3 (B6, B7 and B8) and the control spring C2 between July 1988 and June 1989

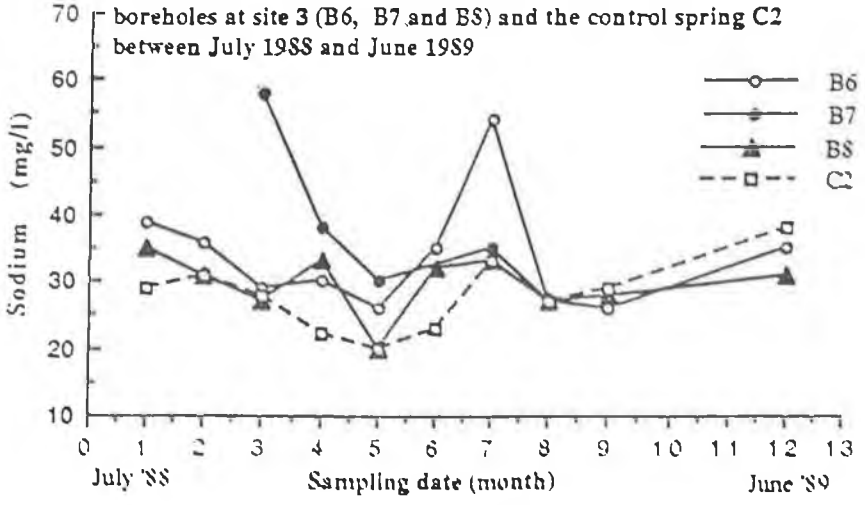

Figure 3.5.27
Potassium concentration (mg/l) detected in the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

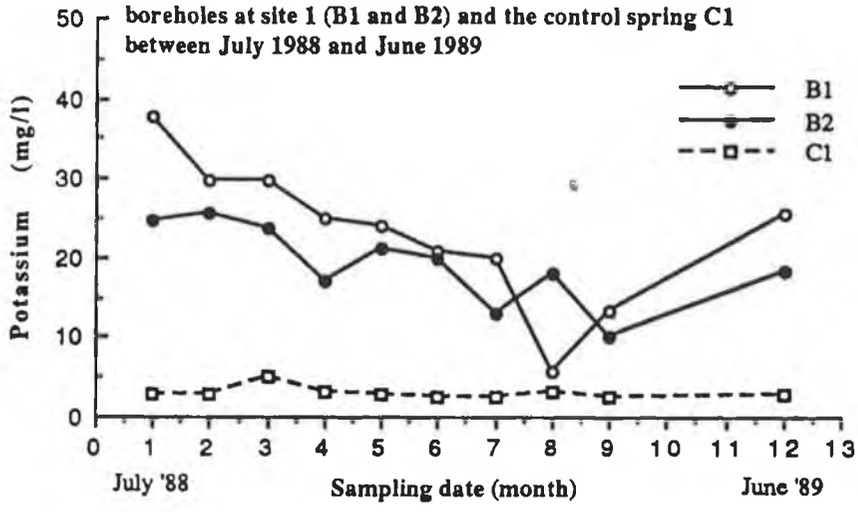

Figure 3.5.28
Potassium concentration (mg/l) detected in the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

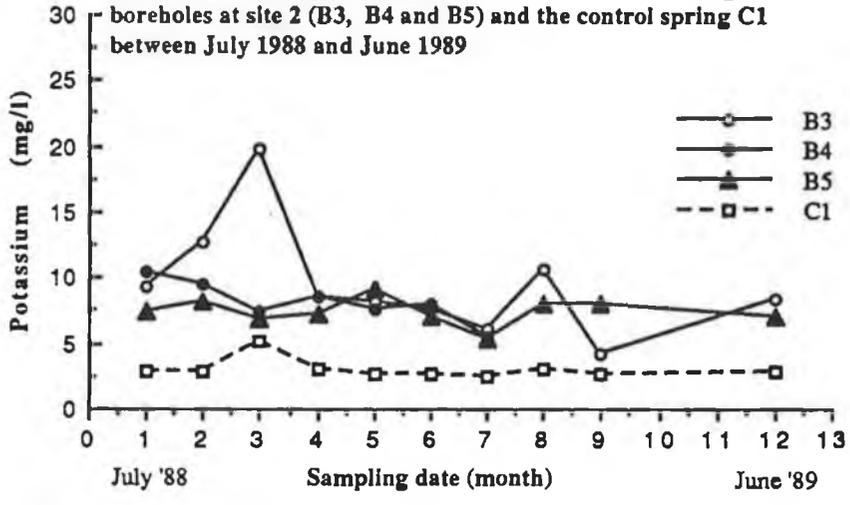

Figure 3.5.29
Potassium concentration (mg/l) detected in the monitoring boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

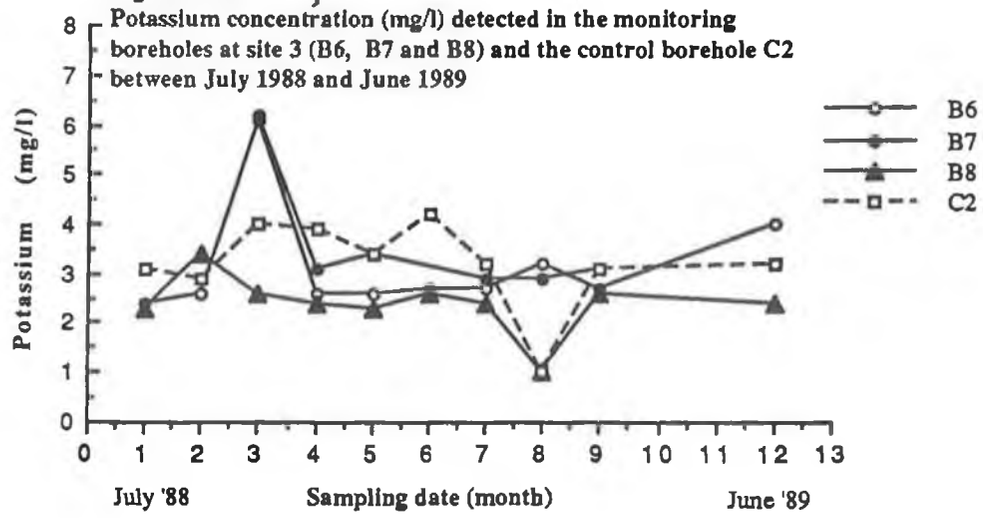

Figure 3.5.30
Calculated K/Na ratio for samples from the monitoring boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

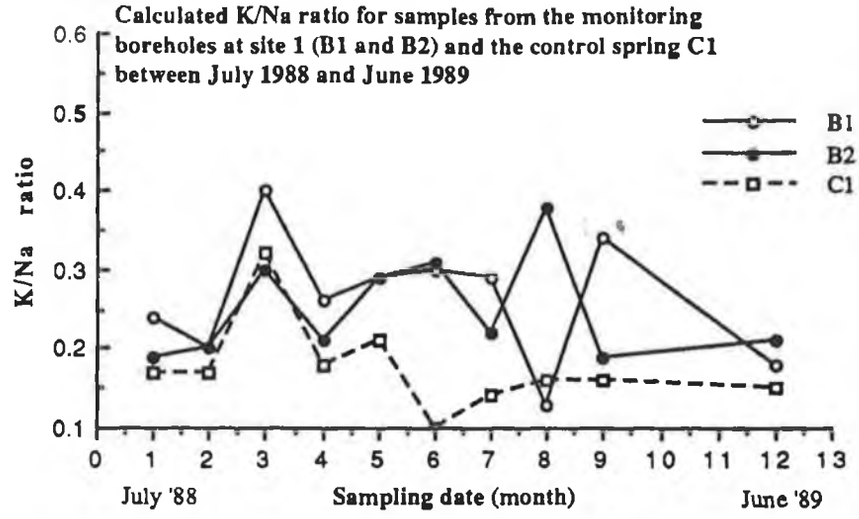

Figure 3.5.31
Calculated K/Na ratio for samples from the monitoring boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

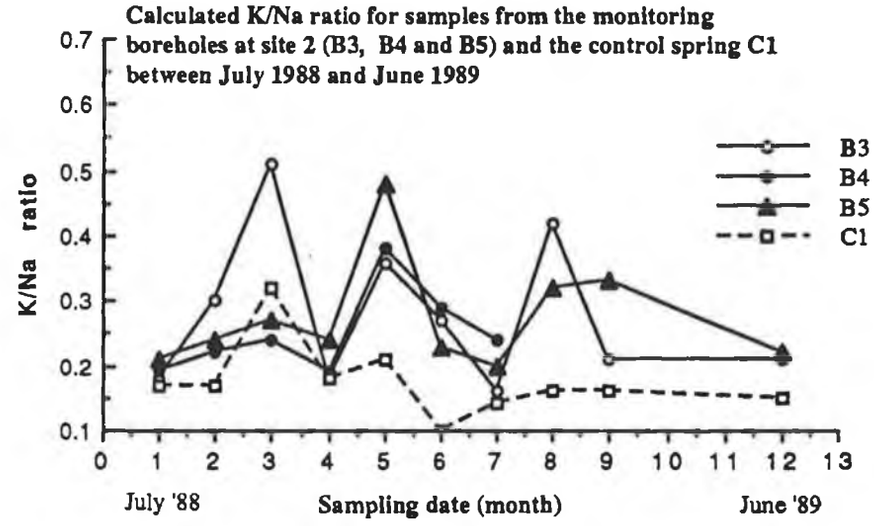

Figure 3.5.32
Calculated K/Na ratio for samples from the monitoring boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

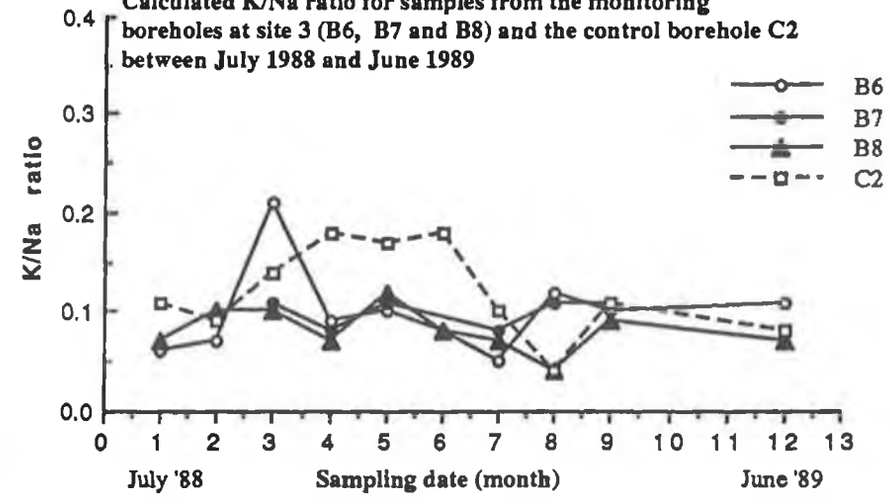

Figure 3.5.33
Numbers of total Coliform bacteria (c.f.u.'s/100ml) detected in the boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

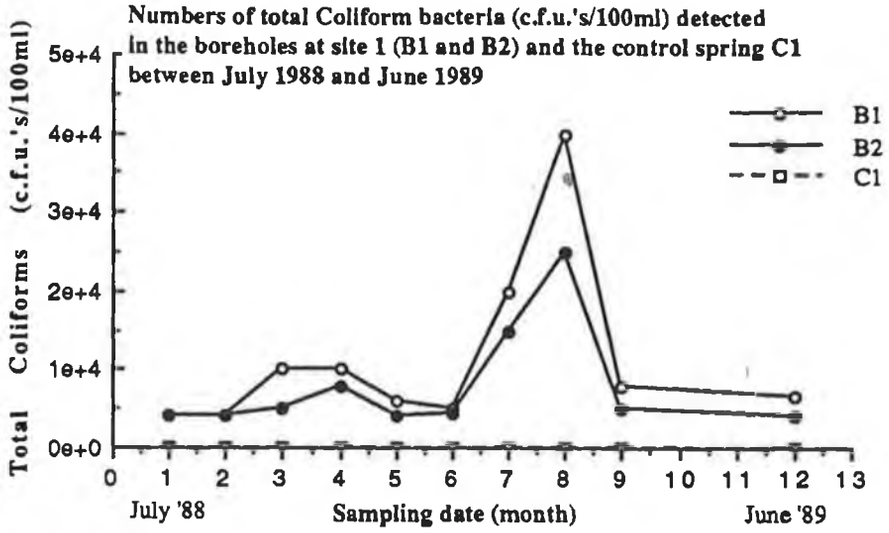

Figure 3.5.34
Numbers of total Coliform bacteria (c. f. u.'s/100ml) detected in the boreholes at site 2 (B3, B4 and B5) and the control spring C1 between July 1988 and June 1989

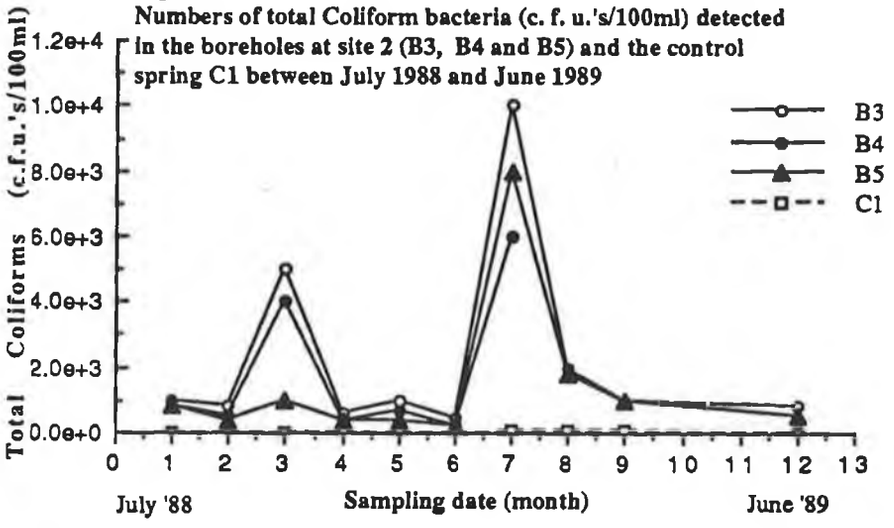

Figure 3.5.35
Numbers of total Coliform bacteria (c. f. u.'s/100ml) detected in the boreholes at site 3 (B6, B7, B8) and the control borehole C2 betewwn July 1988 and June 1989

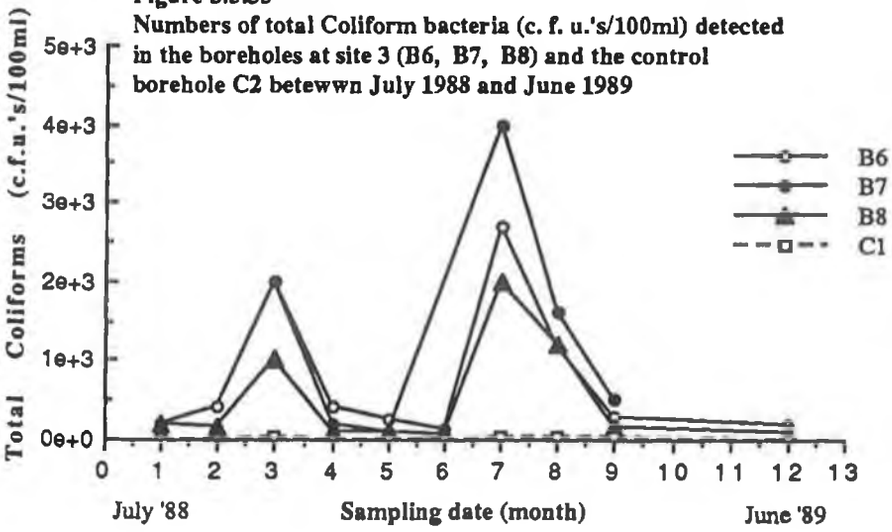

Figure 3.5.36
Numbers of fecal Coliform bacteria (c. f. u.'s/100ml) detected in the boreholes at site 1 (B1and B2) and the control spring C1 between July 1988 and June 1989

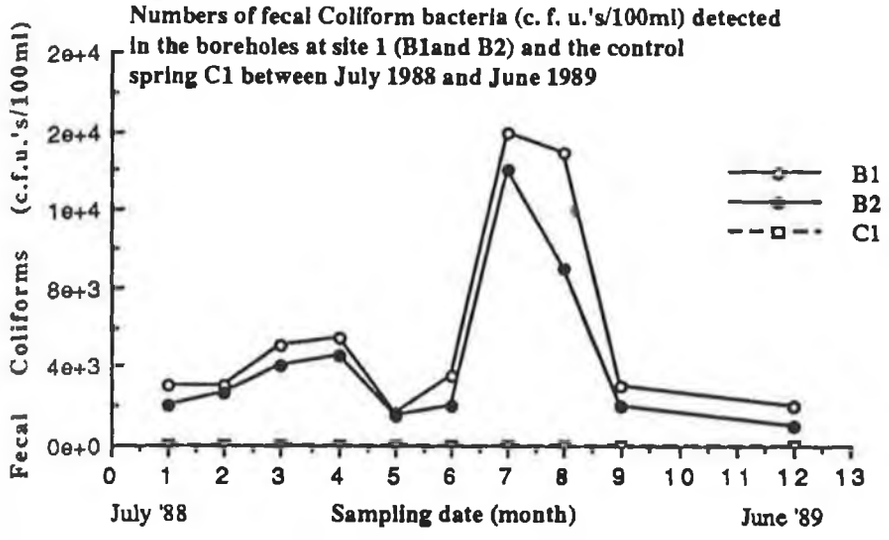

Figure 3.5.37
Numbers of fecal Coliform bacteria (c. f. u.'s/100ml) detected in the boreholes at site 2 (B3, B4, B5) and the control spring C1 between July 1988 and June 1989

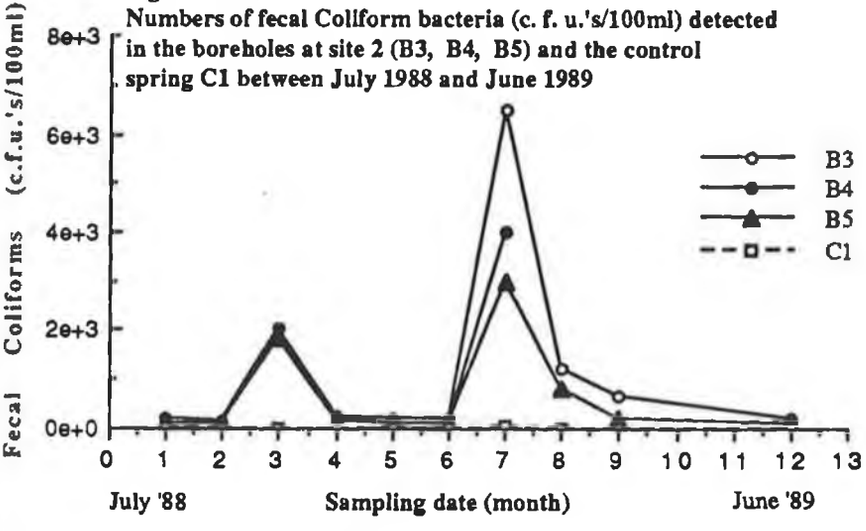

Figure 3.5.38
Numbers of fecal Coliform bacteria (c. f. u.'s/100ml detected in the boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

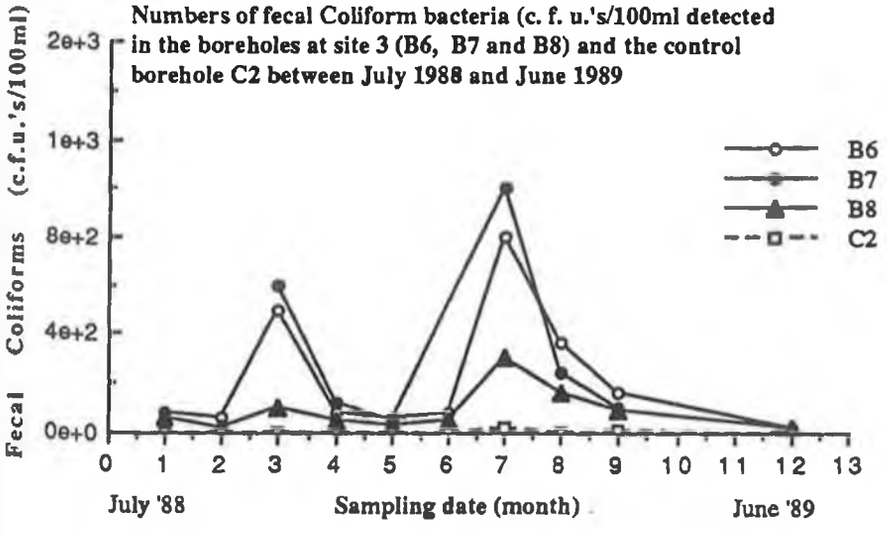

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Figure 3.5.39
Numbers of fecal Streptococci bacteria (c. f. u.'s/100ml) detected in the boreholes at site 1 (B1 and B2) and the control spring C1 between July 1988 and June 1989

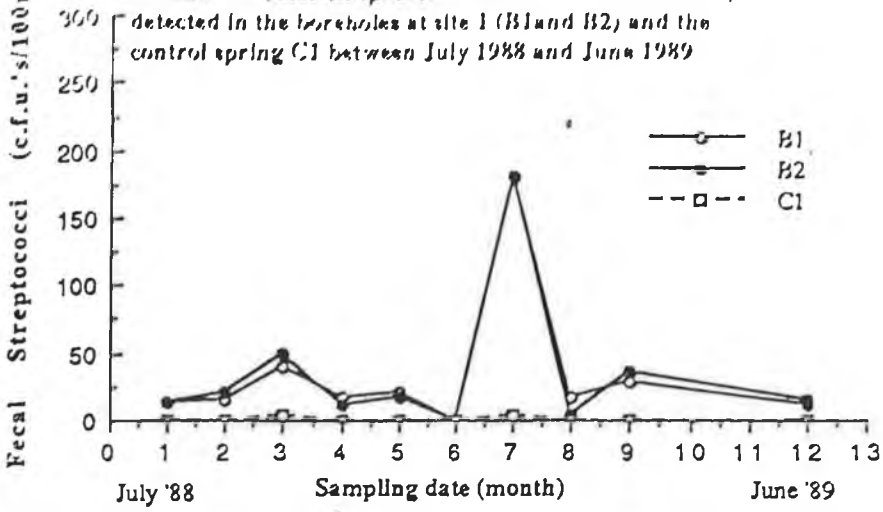


Figure 3.5.40
Numbers of fecal Streptococci bacteria (c. f. u.'s/100ml) detected in the boreholes at site 2 (B3, B4 and B5) and the control Spring C1 between July 1988 and June 1989

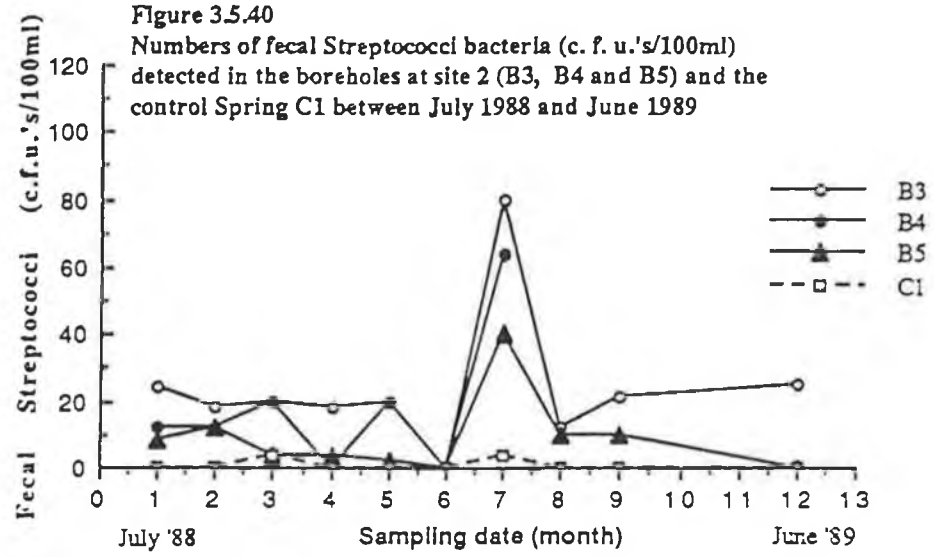


Figure 3.5.41
Numbers of fecal Streptococci bacteria (c. f. u.'s/100ml) detected in the boreholes at site 3 (B6, B7 and B8) and the control borehole C2 between July 1988 and June 1989

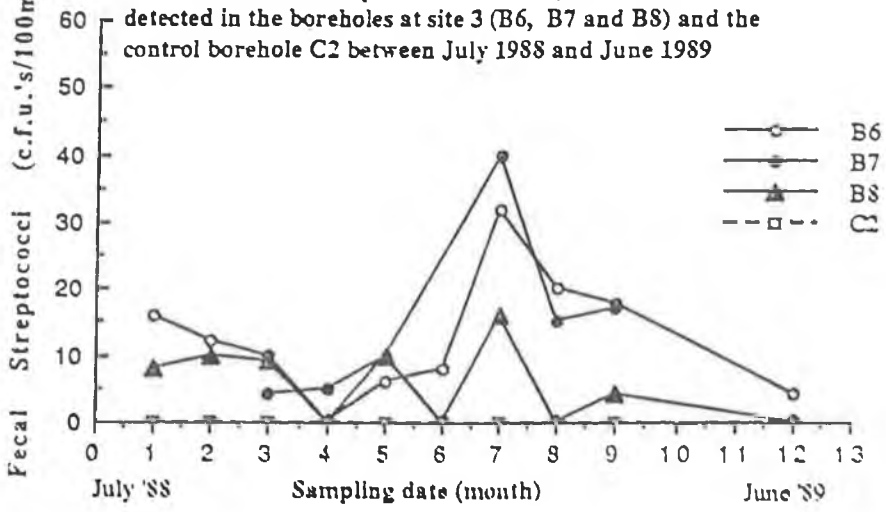


Figure 3.5.42
Comparison of the nitrate and ammonia concentrations detected in at site 1 (B1) between July 1988 and June 1989

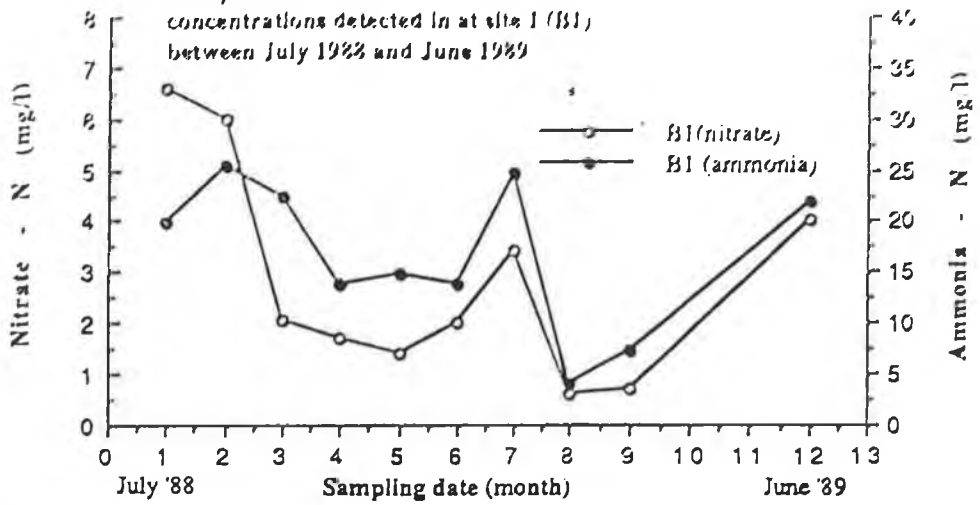


Figure 3.5.43
Comparison of the nitrate and ammonia concentrations detected at site 2 (B3) between July 1988 and June 1989

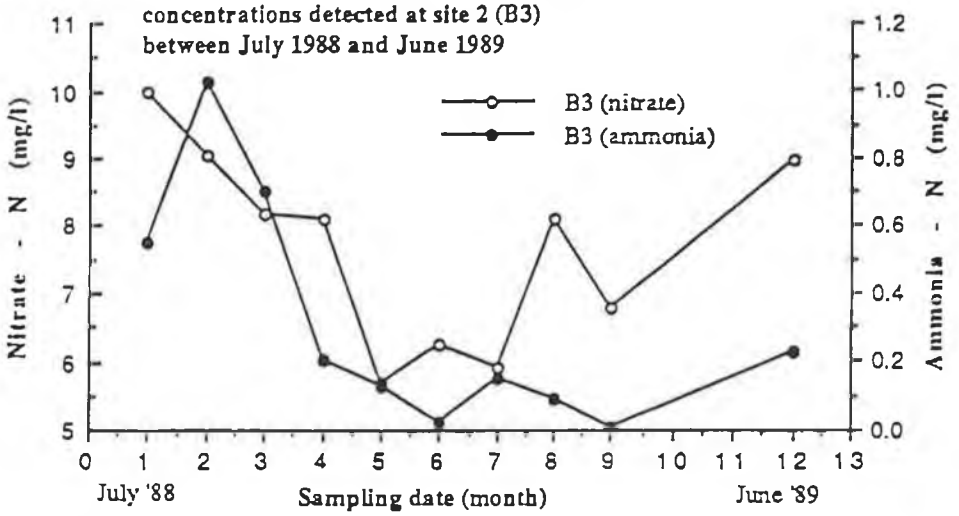
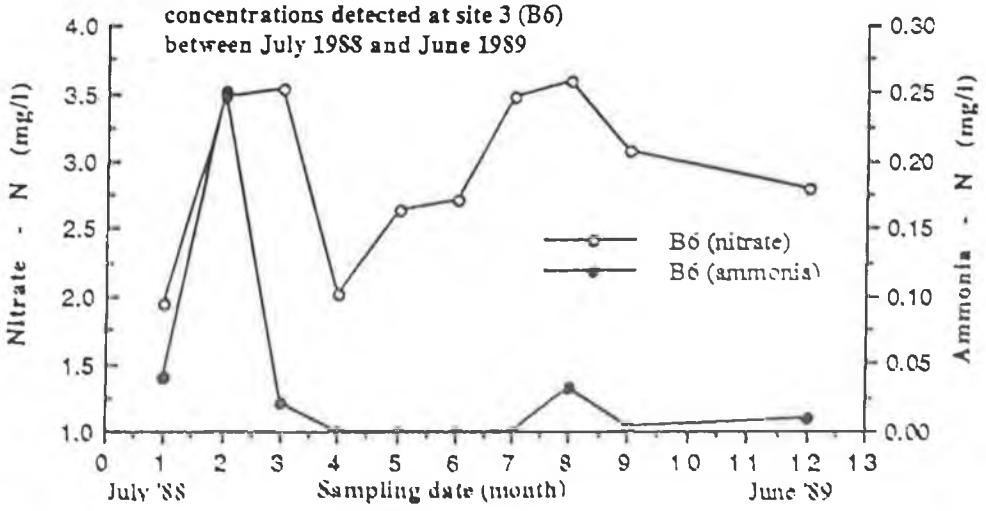


Figure 3.5.44
Comparison of the nitrate and ammonia concentrations detected at site 3 (B6) between July 1988 and June 1989



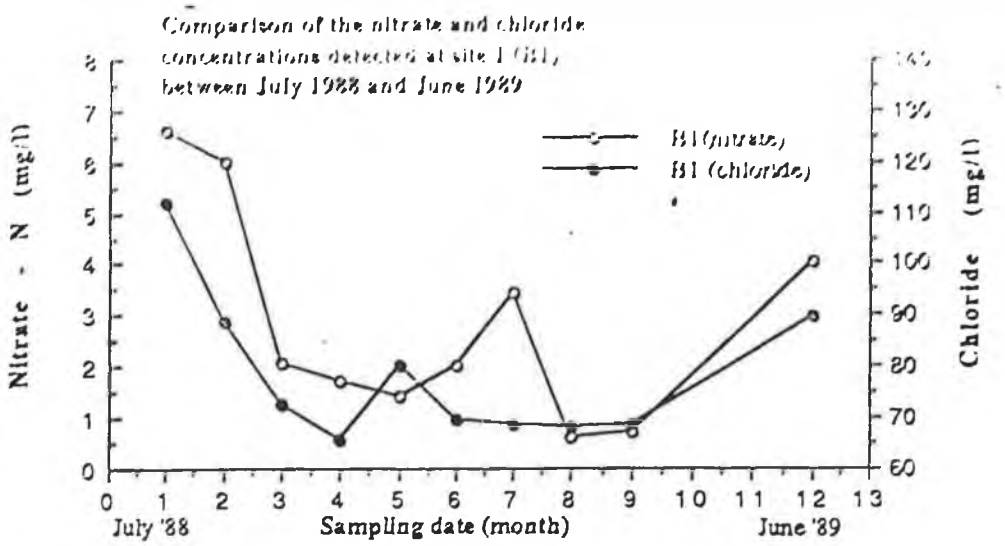


Figure 3.5.46
Comparison of the nitrate and chloride concentrations detected at site 2 (B3) between July 1988 and June 1989

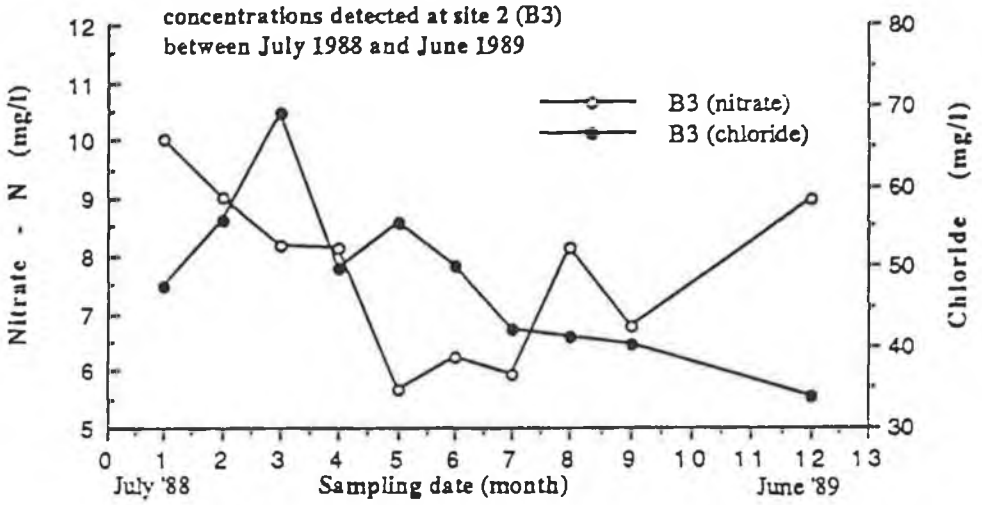


Figure 3.5.47
Comparison of the nitrate and chloride concentrations detected at site 3 (B6) between July 1988 and June 1989

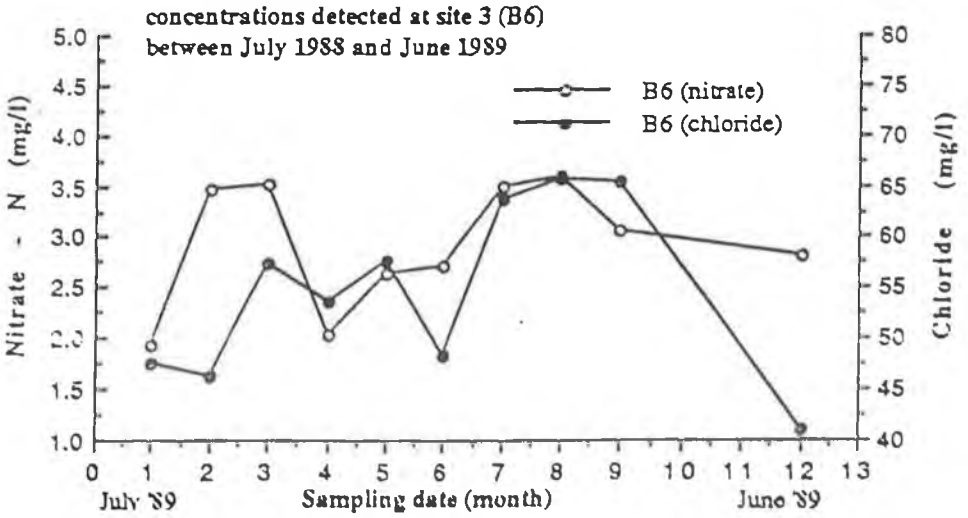


Figure 3.5.48
Comparison of the nitrate concentration and fecal Coliform bacteria numbers detected at site 1 (B1) between July 1988 and June 1989

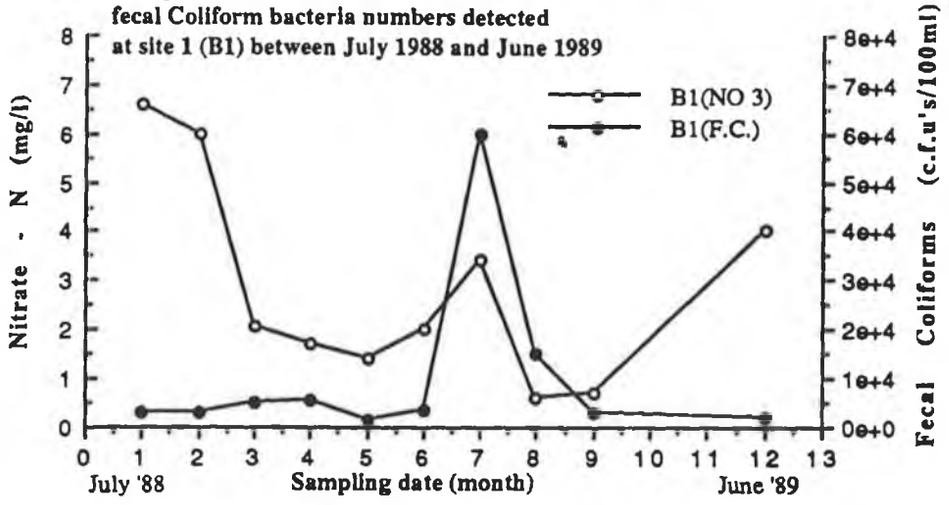


Figure 3.5.49
Comparison of the nitrate concentration and fecal Coliform bacteria numbers detected at site 2 (B3) between July 1988 and June 1989

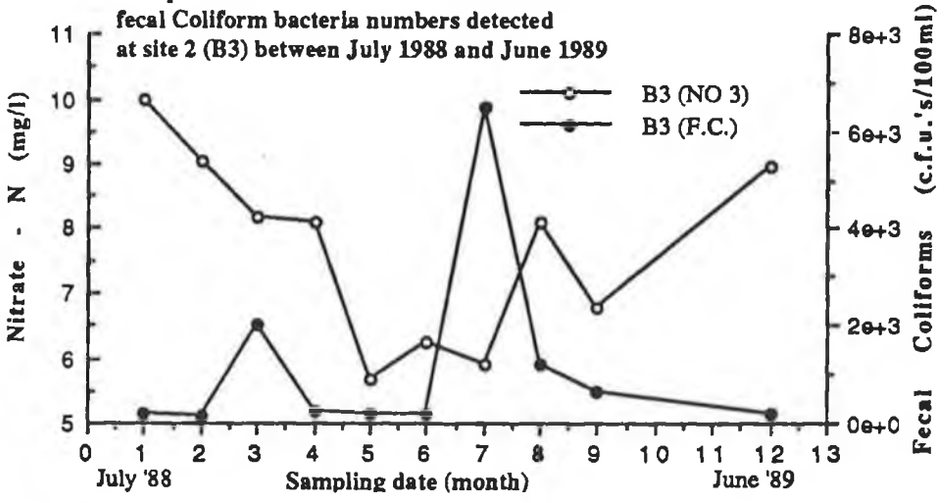


Figure 3.5.50
Comparison of nitrate concentration and fecal Coliform bacteria numbers detected at site 3 (B6) between July 1988 and June 1989

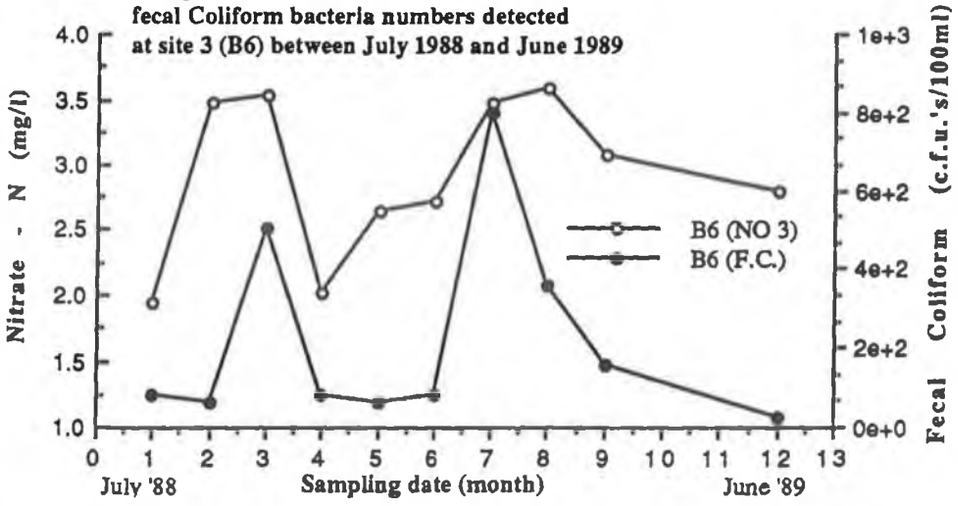


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Figure 3.5.51
Plot of the numbers of fecal Coliform bacteria detected at site 1 (B1) and the total rainfall (mm) for the 5 day period preceeding sampling

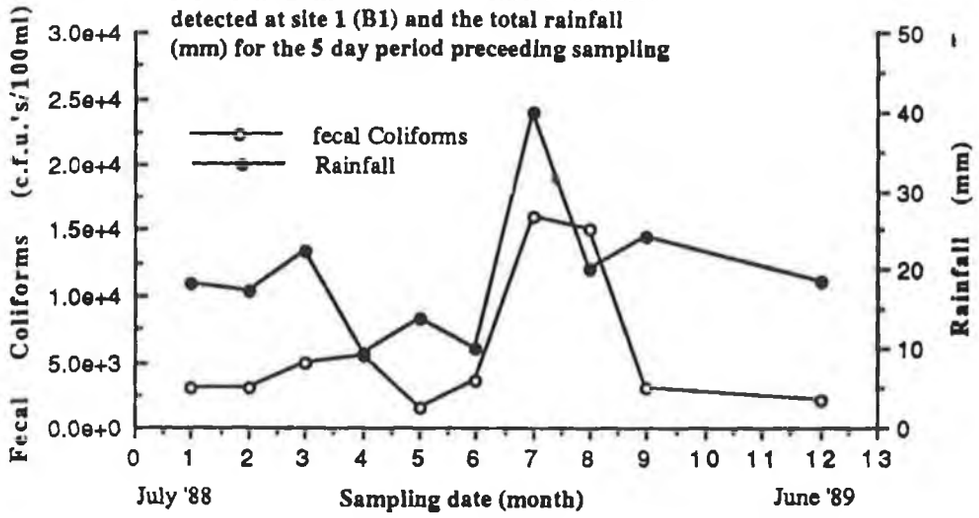

Figure 3.5.52
Plot of the number of fecal Coliform bacteria detected at site 2 (B3) and the total rainfall (mm) for the 5 day period preceeding sampling

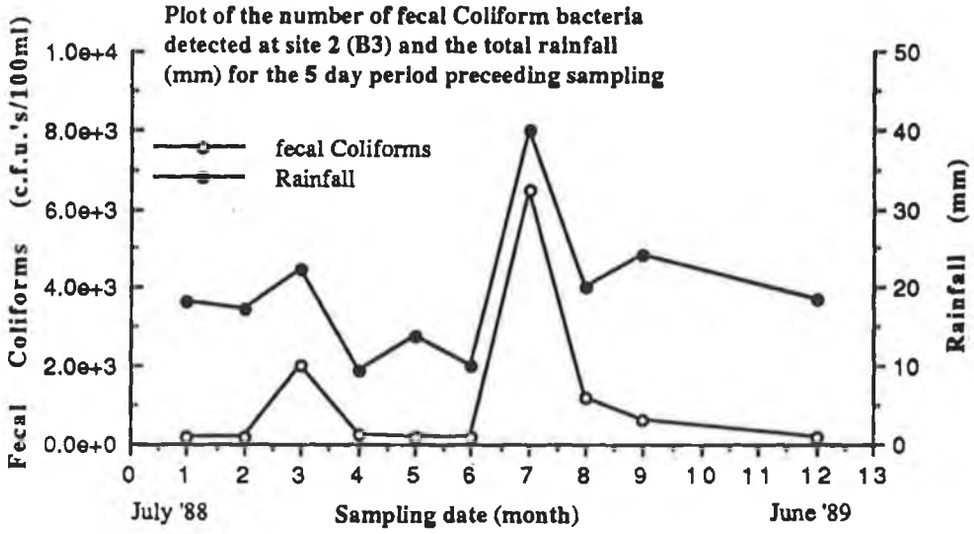

Figure 3.5.53
Plot of the number of fecal Coliform bacteria detected at site 3 (B6) and the total rainfall (mm) for the 5 days preceeding sampling

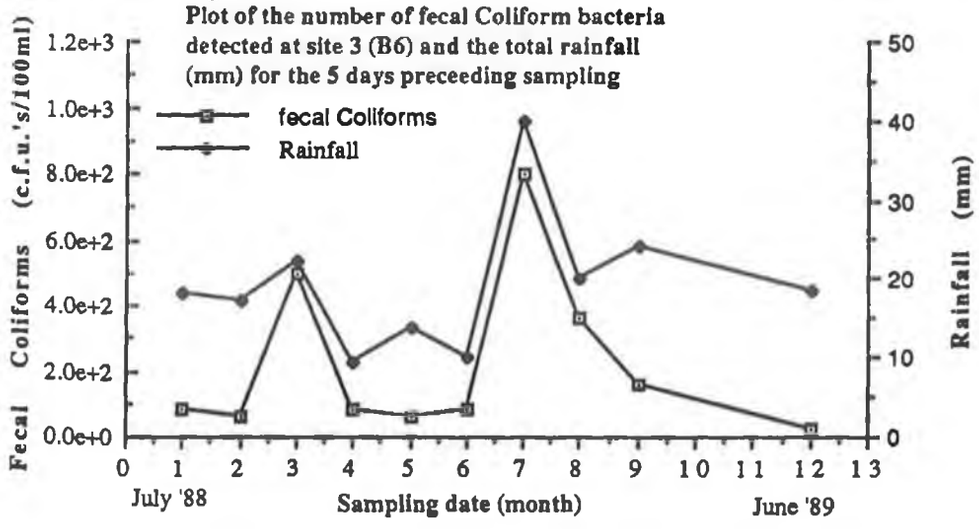

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Figure 3.5.54
Plot of the numbers of fecal Streptococci bacteria detected at site 1 (B1) and the total rainfall (mm) for the 5 day period preceeding sampling

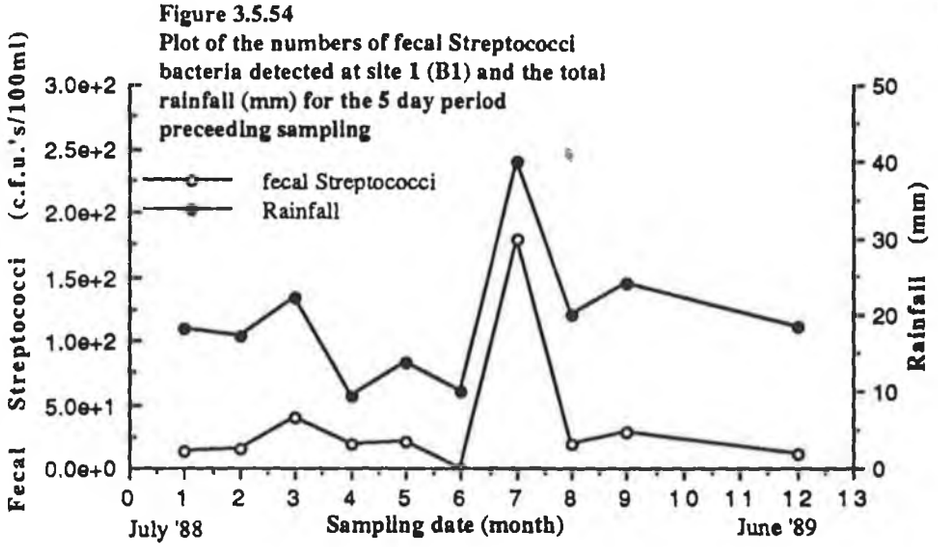

Figure 3.5.55
Plot of the number of fecal Streptococci bacteria detected at site 2 (B3) and the total rainfall (mm) for the 5 day period preceeding sampling

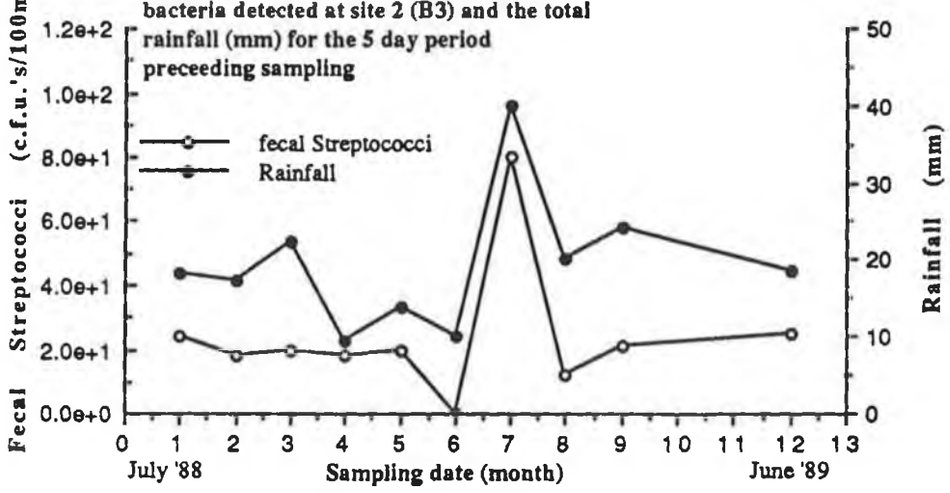

Figure 3.5.56
Plot of the numbers of fecal streptococci bacteria detected at site 3 (B6) and the total rainfall (mm) for the 5 day period preceeding sampling.

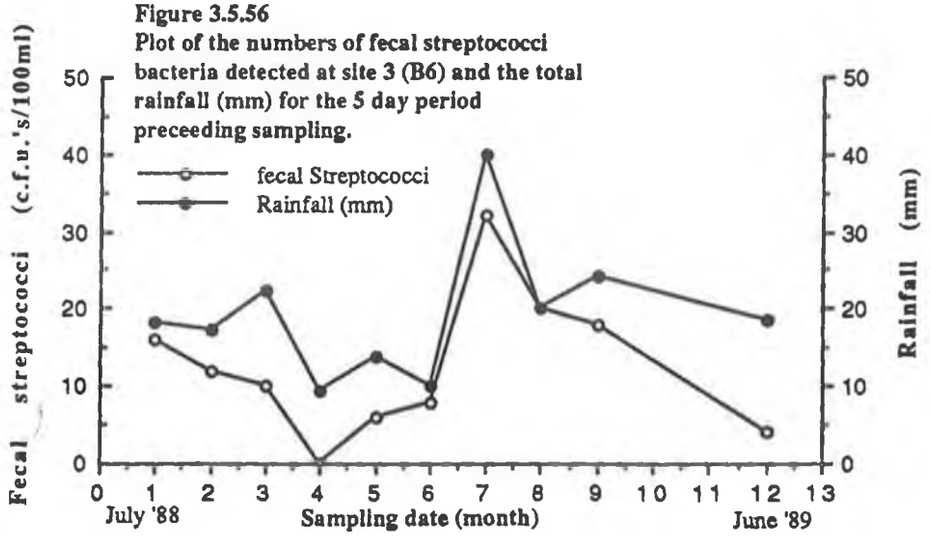

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Figure 3.5.57
Monthly rainfall (mm) for the the groundwater sampling period (June 1988 to June 1989)

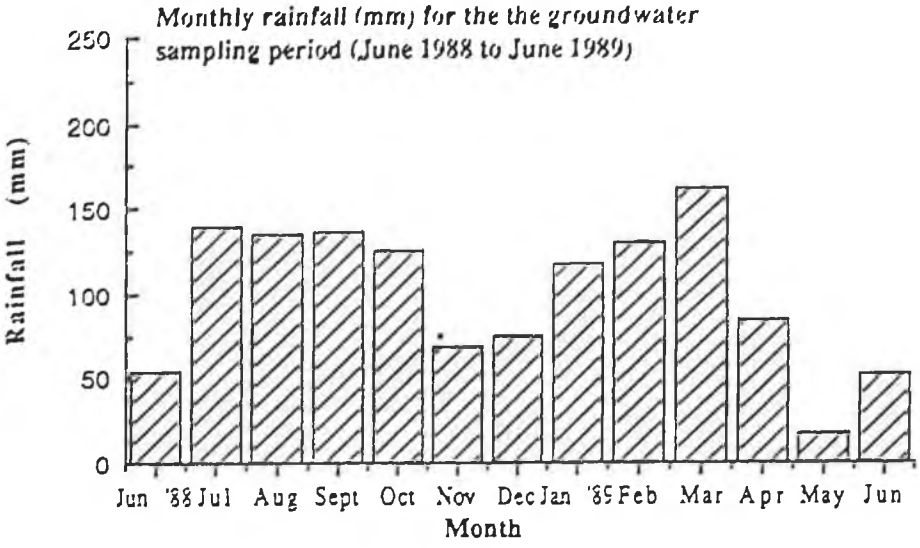

Figure 3.5.58
Total rainfall (mm) for the 5 day period preceeding sampling (July 1988 to June 1989)

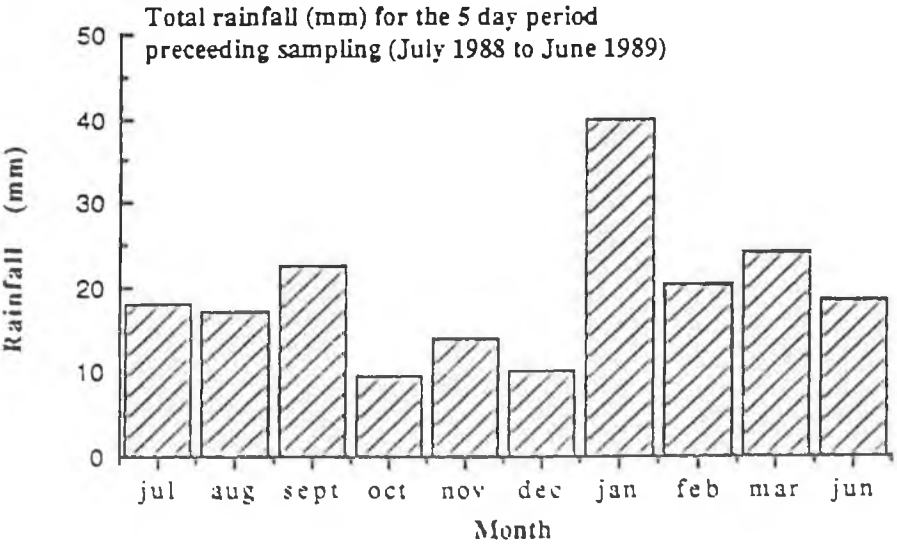

Figure 3.5.59
Water level fluctuations in the monitoring boreholes at the three sampling sites between July 1988 and June 1989

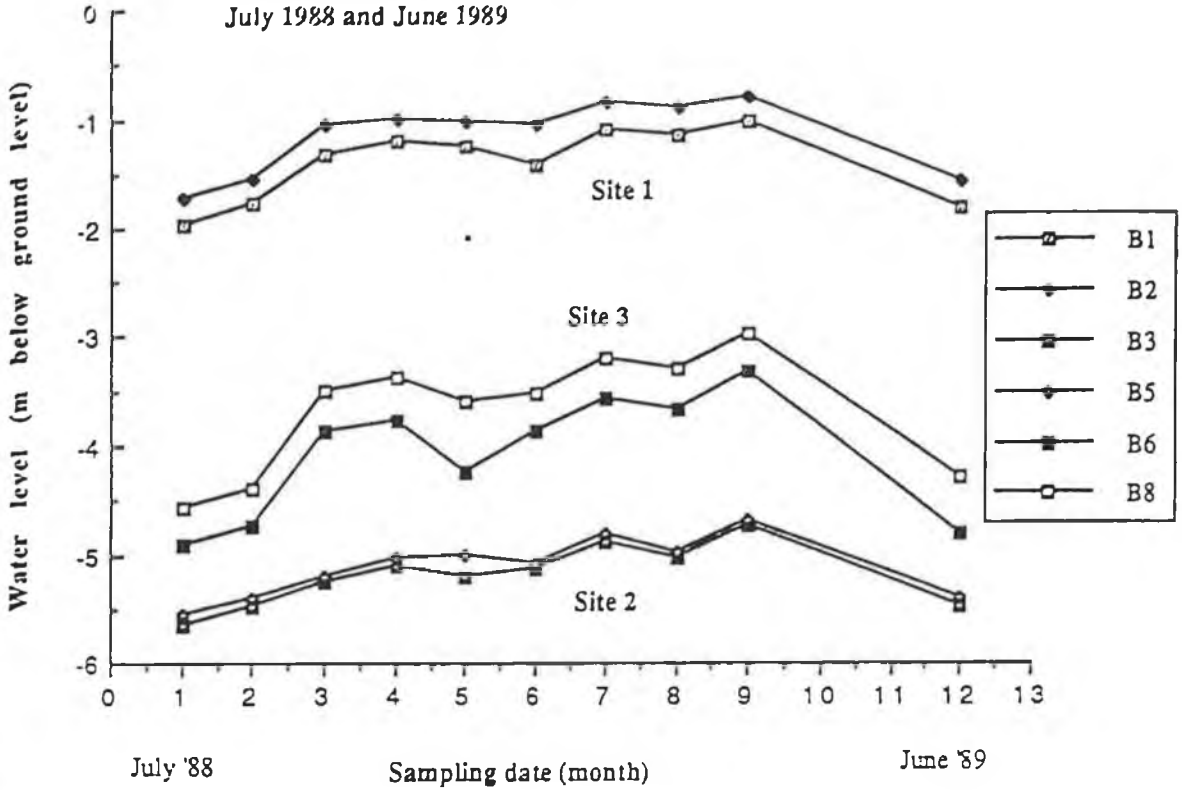

Table 3.5.3

Numbers of Sulphate Reducing Clostridia and Salmonella Bacteria (c. f. u. 's/100ml) Isolated From the Groundwater Samples

ORGANISM	SAMPLING DATE																	
	November 1988										June 1989							
	Monitoring Borehole																	
	B1	B2	B3	B4	B5	B6	B7	B8	C1	C2	B1	B2	B3	B4	B5	B6	B7	B8	C1	C2
Sulphite - Reducing Clostridia (c. f. u. 's/100ml)	20	12	-	-	-	-	-	-	-	-	12	12	-	-	-	-	-	-	-	-
Salmonella Spp. (Presence or absence)	← Not detected in any of the samples →																	

Table 3.5.4

Concentration (mg/l) of L. A. S. Detergents (as M. B. A. S.) Detected in the the Groundwater Samples

SAMPLING DATE	M. B. A. S. (mg/l)									
	Monitoring Borehole									
	B1	B2	B3	B4	B5	B6	B7	B8	C1	C2
Nov. 1988	5.63	4.98	3.23	1.00	1.02	0.20	0.85	0.30	0.10	0.10
June 1989	8.96	5.39	4.10	-	1.98	0.62	-	0.35	0.20	0.20

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3:5.1.3 Effluent Attenuation at the Three Test Sites

The effluent from the septic tanks at the test sites was sampled and analysed on three occasions. The results of the analysis are presented in Appendix B 5. The percentage reduction in the concentration of the effluent constituents from the septic tank to the groundwater monitoring boreholes was calculated using the results obtained. These results are presented in Tables 3.5.5 to 3.5.7 (p184 to 186) as follows:

(i) Table 3.5.5
Percentage reduction in the effluent constituents from the septic tank to the groundwater monitoring boreholes at site 1 (B1 and B2)

(ii) Table 3.5.6
Percentage reduction in the effluent constituents from the septic tank to the groundwater monitoring boreholes at site 2 (B3, B4 and B5)

(iii) Table 3.5.7
Percentage reduction in the effluent constituents from the septic tank to the groundwater monitoring boreholes at site 3 (B6, B7 and B8).

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Table 3.5.5

The Percentage Reduction in the Concentration of Effluent Constituents from the Septic Tank to the Groundwater Monitoring Boreholes at Site 1 (B1 and B2)

PARAMETER	Monitoring Borehole					
	B1			B2		
	Sampling Date					
	July	Nov.	Mar.	July	Nov.	Mar.
B. O. D.	11.22	72.40	60.77	86.73	89.96	47.37
C. O. D.	+	54.02	60.12	61.68	80.72	14.11
Nitrate (NO_3 - N)	+	100	100	+	100	100
Ammonia (NH_3 - N)	32.4	38.4	61.5	49.5	4.8	29.4
Phosphate (PO_4 - P)	79.5	96.5	88.8	93.5	97.8	87.3
Chloride (Cl)	+	62.3	59.2	24.3	71.0	67.8
Potassium (K)	+	23.2	50.9	34.2	33.9	66.4
Sodium (Na)	+	12.3	69.6	+	25.9	54.4
Total Coliforms	>99.9	>99.9	>99.9	>99.9	>99.9	>99.9
Fecal Coliforms	>99.9	>99.9	>99.9	>99.9	>99.9	>99.9
Fecal Streptococci	99.9	99.3	99.3	99.9	99.4	99.1

Note:

+ = The concentration recorded in the groundwater sample exceeds that in the septic tank effluent

Table 3.5.6

The Percentage Reduction in the Concentration of Effluent Constituents from the Septic Tank to the Groundwater Monitoring Boreholes at Site 2 (B3, B4 and B5)

PARAMETER	Monitoring Borehole								
	B3			B4			B5		
	Sampling Date								
	July	Nov.	Mar.	July	Nov.	Mar.	July	Nov.	Mar.
B. O. D.	99.6	100	99.7	99.2	100	-	99.6	100	99.7
C. O. D.	98.8	98.4	97.1	98.9	98.9	-	97.9	99.2	96.4
Nitrate(NO_3 - N)	+	+	+	+	+	-	+	+	+
Ammonia(NH_3 - N)	98.1	99.6	99.9	96.3	100	-	99.9	100	100
Phosphate (PO_4 - P)	100	98.3	100	100	98.7	-	100	98.7	100
Chloride (Cl)	47.0	96.6	100	47.0	93.4	-	60.3	83.4	100
Potassium (K)	66.1	81.4	92.3	60.3	83.5	-	76.2	78.0	71.9
Sodium (Na)	56.9	89.6	94.6	53.2	92.7	-	77.2	93.8	89.1
Total Coliforms	>99.9	>99.9	>99.9	>99.9	>99.9	-	>99.9	>99.9	>99.9
Fecal Coliforms	>99.9	>99.9	>99.9	>99.9	>99.9	-	>99.9	>99.9	>99.9
Fecal Streptococci	99.7	99.5	99.9	99.7	99.9	-	99.8	99.9	99.9

Note:

+ = The concentration recorded in the groundwater sample exceeds that in the septic tank effluent

The Percentage Reduction in the Concentration of Effluent Constituents from the Septic Tank to the Groundwater Monitoring Boreholes at Site 3 (B6, B7 and B8)

PARAMETER	Monitoring Borehole								
	B6			B7			B8		
	Sampling Date								
	July	Nov.	Mar.	July	Nov.	Mar.	July	Nov.	Mar.
B. O. D.	100	100	99.8	-	100	99.6	100	100	100
C. O. D.	99.0	99.2	97.4	-	98.4	94.5	100	100	98.4
Nitrate(NO_3 - N)	90.4	+	+	-	+	+	100	+	+
Ammonia(NH_3 - N)	99.8	100	99.9	-	98.9	98.7	99.9	100	99.9
Phosphate (PO_4 - P)	100	100	100	-	98.2	100	100	100	100
Chloride (Cl)	87.9	100	57.3	-	91.6	57.3	99.7	100	72.8
Potassium (K)	100	100	100	-	100	100	100	100	100
Sodium (Na)	90.0	93.0	100	-	83.4	100	94.0	100	100
Total Coliforms	>99.9	>99.9	>99.9	-	>99.9	>99.9	>99.9	>99.9	>99.9
Fecal Coliforms	>99.9	>99.9	>99.9	-	>99.9	>99.9	>99.9	>99.9	>99.9
Fecal Streptococci	>99.9	>99.9	>99.9	-	>99.9	>99.9	>99.9	>99.9	>99.9

Note:

+ = The concentration recorded in the groundwater sample exceeds that in the septic tank effluent

3:5.1.4 Statistical Analyses

Statistical analysis of results obtained in the study are presented in Tables 3.5.8 to 3.5.12 (p188 to 192). The statistical evaluations are divided into three groups:

(i) The Wilcoxon Signed Rank test was applied to determine whether the concentrations of the various parameters recorded in the monitoring boreholes were significantly greater than those in the control samples. The results of this analysis are presented in Table 3.5.8

(ii) The Wilcoxon Signed Rank test was applied to assess if there was a significant decrease in the contamination of groundwater with increasing distance from the septic tank soil treatment system. These results are given in Tables 3.5.9 to 3.5.11

(iii) Scatter plots were prepared of the numbers of fecal Coliform bacteria in the groundwater samples against the total rainfall for the five days preceding sampling (Appendix B 4). A simple regression line was fitted to the plots and Pearson's correlation coefficients were calculated for the linear fits. The results are presented in Table 3.5.12.

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Table 3.5.8

Statistical Evaluation of the Increase in the Measured Parameters in the Monitoring Boreholes Compared to the Control Samples - Using the Wilcoxon Signed Rank Test (at P = 0.025 and 0.005)

Parameter	Monitoring Borehole															
	B1		B2		B3		B4		B5		B6		B7		B8	
	Significance Level (P)															
	.025	.005	.025	.005	.025	.005	.025	.005	.025	.005	.025	.005	.025	.005	.025	.005
B. O. D.	+	+	+	+	+	+	+	+	+	+	+	-			+	-
C. O. D.	+	+	+	+	+	+	*	*	+	+	*	*			*	*
pH	-	-	-	-	+	-	+	-	+	-	+	-			+	+
Cond.	+	+	+	+	+	+	+	+	+	+	+	+			+	+
NO_3 - N	-	-	-	-	+	+	+	+	+	+	+	+			+	+
NH_3 - N	+	+	+	+	+	+	*	*	*	*	*	*			*	*
Cl	+	+	+	+	-	-	+	-	+	-	+	+	*		+	+
Na	+	+	+	+	+	+	+	+	+	+	+	-			-	-
K	+	+	+	+	+	+	+	+	+	+	-	-			-	-
K/Na	+	+	+	+	+	+	*	*	+	+	-	-			-	-
T. C.	+	+	+	+	+	+	+	+	+	+	+	+			+	+
F. C.	+	+	+	+	+	+	+	+	+	+	+	+			+	+
F. S.	+	+	+	+	+	+	*	*	+	+	+	+			+	+
PO_4 - P	+	+	+	+	+	-	-	-	-	-	-	-			-	-

Note:

+ = There is a significant increase in the concentration recorded in the monitoring
- = There is no significant increase in concentration recorded in the monitoring borehole
* = The paired observations are not sufficient to yield an accurate assessment

Table 3.5.9

Statistical Evaluation of the Decrease in Concentrati
Effluent Parameters with Increasing Distance f
Septic Tank System at Site 1 (from B1 to
Using the Wilcoxon Signed Rank Test (at P = 0.02

PARAMETER	Significance Level	
	0.025 (2.5%)	((
B. O. D.	-	-
C. O. D.	-	-
Nitrate (NO_3 - N)	-	-
Ammonia (NH_3 - N)	-	-
Phosphate (PO_4 - P)	-	-
Chloride (Cl)	+	+
Potassium (K)	-	-
Sodium (Na)	-	-
Potassium/Sodium	-	-
Conductivity	+	+
Total Coliforms	+	+
Fecal Coliforms	+	+
Fecal Streptococci	+	+

Note:

+ = There is a significant reduction in concentration between the monitoring boreholes

- = There is no significant reduction in concentration between the monitoring boreholes

Table 3.5.10

Statistical Evaluation of the Decrease in Concentration of Various Effluent Parameters with Increasing Distance from the Septic Tank System at Site 2 (from B3 to B5) Using the Wilcoxon Signed Rank Test (at P = 0.025 and 0.005)

PARAMETER	Significance Level (P)	
	0.025 (2.5%)	0.005 (0.5%)
B. O. D.	+	+
C. O. D.	-	-
Nitrate (NO_3 - N)	+	-
Ammonia (NH_3 - N)	+	-
Phosphate (PO_4 - P)	*	*
Chloride (Cl)	-	-
Potassium (K)	-	-
Sodium (Na)	+	-
Potassium/Sodium	-	-
Conductivity	+	-
Total Coliforms	+	+
Fecal Coliforms	+	+
Fecal Streptococci	+	+

Note:

+ = There is a significant reduction in concentration between the monitoring boreholes

- = There is no significant reduction in concentration between the monitoring boreholes

* = Insufficient data to make an accurate prediction (paired observations too low)

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Table 3.5.11

Statistical Evaluation of the Decrease in Concentration of Various Effluent Parameters with Increasing Distance from the Septic Tank System at Site 3 (from B6 to B8) Using the Wilcoxon Signed Rank Test (at P = 0.025 and 0.005)

PARAMETER	Significance Level (P)	
	0.025 (2.5%)	0.005 (0.5%)
B. O. D.	*	*
C. O. D.	-	-
Nitrate (NO_3 - N)	+	-
Ammonia (NH_3 - N)	-	-
Phosphate (PO_4 - P)	*	*
Chloride (Cl)	-	-
Potassium (K)	+	-
Sodium (Na)	+	-
Potassium/Sodium	-	-
Conductivity	-	-
Total Coliforms	+	+
Fecal Coliforms	+	+
Fecal Streptococci	+	+

Note:

+ = There is a significant reduction in concentration between the monitoring boreholes

- = There is no significant reduction in concentration between the monitoring boreholes

* = Insufficient data to make an accurate prediction (paired observations too low)

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Table 3.5.12

Table Showing the Pearson's Correlation Coefficient (r) and the Coefficient of Determination (r^2) Calculated from the Regression Lines of the Fecal Coliform Bacteria Numbers in the Groundwater Samples Against the Total Rainfall for the Five Day Period Preceeding Sampling. The Significance of the Correlation was Calculated Using the t - Statistic at P = 0.01 and 0.005 Significance Levels

Regression Lines Fecal Coliform Bacteria Vs Rainfall	r	r^2	t $\left(t = r\frac{\sqrt{n-2}}{\sqrt{1-r^2}}\right)$	Significance Level (P)	
				0.01	0.005
B1	0.66	0.40	2.31	-	-
B2	0.73	0.53	2.97	+	-
B3	0.89	0.79	5.53	+	+
B5	0.84	0.71	4.41	+	+
B6	0.85	0.72	4.46	+	+
B8	0.86	0.73	4.68	+	+
C1	0.94	0.87	7.42	+	+
C2	0.85	0.72	4.49	+	+

Note:

\+ = The correlation is significant at the given significance level (P)
\- = The correlation is not significant at the given significance level (P)

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3:5.2 Description of Results

3:5.2.1 Soil/Overburden Analysis

This section describes the results of the analysis of the soil/overburden material at the test sites. Particle size analysis (Table 3.5.1, p158) of the material at site 1 showed it to be composed of 9 to 11% clay, 34 to 40% silt, 41 to 44% sand and 10 to 11% gravel. The material is classified as a ' loam ' under the U. S. D. A. classification system. This has a major influence on its physical and chemical properties which is clearly demonstrated in the results presented in Table 3.5.2 (p158). The large percentage of clay and silt sized particles gives the soil a high porosity (33.0 to 41.0%), but the nature of these particles is such that the individual pore size is small (capilliary pores) and hence transmission of water is low. This is reflected in the high moisture content (23.5 to 26.3%) of the material and in the low permeability recorded (0.029 mm/second). The material also has a moderately high cation exchange capacity (C. E. C.) of 32 to 35 meq/100 g due to the high clay content and significant quantities of organic matter present. The results obtained show little variation in particle size distribution or physical and chemical properties with increasing depth.

It would appear that the soil cover at the site has good attenuating properties. The moderately low permeability would ensure adequate contact time between the effluent and soil particles for ion exchange and adsorption reactions to proceed, with the high C. E. C. of the material providing exchange sites for the immobilisation of effluent constituents. However, because the effluent is discharged from the septic tank into a 2.0 metre deep soakage pit (described in 3:3), it is unlikely that any of the attenuation capacity of the overburden material would be utilised.

Conversely, results obtained from the analysis of the overburden at site 2 (Table 3.5.1, p158) show that the material is largely composed of coarse - sized sand and gravel particles (41 to 50% sand and 43 to 52% gravel), with only small quantities of the finer clays and silts present (clay 1% and silt 6 to 7%). The material is classified as a ' sand ' under the U. S. D. A. classification system. The high percentage of coarse - sized particles results in a relatively low total porosity (23 to 28%) in comparison to site 1. However, because of the lack of fine particles, the individual pore size is large permitting rapid transmission of water, as indicated by the percolation rate of 0.125 mm/second and the low moisture holding capacity (moisture content 3.0 to 4.1%). In addition the large pore spaces can facilitate oxygen diffusion through the overburden material and permit oxidation reactions with the percolating effluent constituents. The results also show that the C. E. C. of the material is low (3.1 to 4.8 meq/100 g).

The results obtained indicate that the effluent from the septic tank soakage pit may only receive limited attenuation in the overburden material. The high permeability and low C. E. C. would curtail most of the attenuation, facilitating rapid migration of the effluent with few exchange sites available for adsorption and ion exchange. However, because the material is well aerated, some oxidation reactions with effluent constituents may occur.

Site 3 is shown to be overlain by 2.5 metres of ' sandy loam ' material (U. S. D. A. classification) containing 6 to 9% clay, 30 to 36% silt, 35 to 42% sand and 19 to 23% gravel sized particles (Table 3.5.1, p158). The physical and chemical properties of the material are similar to those recorded at site 1 (Table 3.5.2, p158). However, the higher percentage of coarse sized particles relative to that at site 1 result in a lower total porosity (33 to 38%) and cation exchange capacity (23 to 28 meq/100 g). In addition the larger individual pore size gives the material a moderate permeability of 0.039 mm/second, with a reduced moisture holding capacity (moisture content of 19.2 to 19.8%). Although no samples were obtained at depths of greater than 2.5 metres, it was noted during drilling to be largely composed of permeable sands and gravels overlying coarser gravel/cobble sized material.

3:5.2.2 Contamination of Groundwater Downgradient of Septic Tank Treatment Systems

This section describes the results obtained in the analysis of the boreholes at the three test sites and compares the concentrations detected to the background levels in the control sites. In addition, in order to illustrate the level of contamination detected. the results obtained are compared to the E. C. control levels for drinking water (Anon., 1980). The results of the percentage reduction in the concentration of the effluent constituents from the septic tank to the monitoring boreholes are also described, as is the statistical evaluation of the data.

(i) Site 1

The monitoring boreholes B1 and B2 (2.0 and 8.1 metres downgradient of the soakage pit) were grossly polluted by septic tank effluent on all sampling dates. With few exceptions the concentrations of measured parameters were markedly above the background concentration in the control spring C1 (Table 3.5.8, p188) and in many instances exceeded E. C. guide levels and maximum admissable concentrations (M. A. C.). Figures 3.5.5 (p162) and 3.5.6 (p163) show the B. O. D. and C. O. D. concentrations detected in the boreholes and the control spring C1. The highest concentrations detected were 630 mg/l (C. O. D.) and 265 mg/l (B. O. D.) in August 1988. This degree of contamination is significant when it is considered that the average

B. O. D. of domestic sewage is 250 mg/l (Hammer, 1977). No statistically significant reduction in the B. O. D. or C. O. D. levels was noted with increasing distance from the septic tank system (Table 3.5.9, p189). The reduction between B1 and B2 was shown to be very variable i. e. there was a noticeable decrease in concentration between the boreholes until January 1989 after which concentrations in B2 exceeded those recorded in B1.

Conductivity values in B1 and B2 (Figure 3.5.9, p164) were on average 400 µS/cm above the background levels recorded in C1. Highest concentrations were recorded in the summer months when the water levels in the monitoring boreholes were lowest (Figure 3.5.59, p181). During these periods values of up to 930 µS/cm above background were observed. Reduction in concentration from B1 to B2 was minimal during high water conditions (October to March) but significant reductions were noted in July/August 1988 and again in June 1989. On these sampling dates up to a 40% reduction was recorded between the boreholes (1583 µS/cm to 992 µS/cm). Table 3.5.9 (p189) shows that there was a statistically significant reduction (at P = 0.005) in the conductivity concentrations from B1 to B2 over the entire sampling period.

Chloride concentrations in the monitoring boreholes (Figure 3.5.12, p165) were very high and markedly above those recorded in the control spring (Table 3.5.8, p188). The maximum concentration recorded of 111.7 mg/l at the monitoring borehole B1 greatly exceeded the corresponding control value in C1 of 37.7 mg/l. The greatest decrease in chloride concentration from B1 to B2 occurred during the period of lowest water levels in the monitoring wells (maximum difference of 39.6 mg/l). During this period maximum concentrations were also recorded. The reduction in concentration from B1 to B2 from October 1988 to June 1989 was minimal (i. e. 5 mg/l or less) although a statistically significant reduction(at P = 0.005) throughout this sampling period was demonstrated (Table 3.5.9, p189).

Nitrate concentrations in the boreholes (Figure 3.5.15, p166) were in general low and similar to those recorded at the control site C1. Table 3.5.8 (p188) shows that there was no significant difference (at P = 0.025) between the nitrate concentrations in the control samples and the monitoring boreholes B1 and B2. This is attributed to the fact that the anaerobic septic tank effluent contained nitrogen mainly in the form of ammonia (demonstrated by the results of the effluent analysis presented in Appendix B 5) and was discharged directly to groundwater, with little if any contact with oxygen in the overburden material. This is confirmed by the results presented in Figure 3.5.18 (p167) which show that very high ammonia concentrations were detected in the boreholes. The maximum value detected was 25.3 mg/l ammonia (August 1988 in B1) which is 60 times greater than the E. C. maximum admissable concentration for

drinking water. The concentration of NH_3 - N does not follow the pattern observed for other parameters. The concentrations detected were very variable and no decrease was noted with increasing water levels in the monitoring boreholes. No significant reduction in concentration between the two boreholes was noted (at P = 0.025) throughout the sampling period (Table 3.5.9, p189).

The phosphate concentrations recorded in the boreholes were high, with maximum concentrations of 3.84 mg/l in B1 and 2.29 mg/l in B2. No phosphate was detected in the control spring C1. Maximum levels were recorded during the periods of lowest water levels with the largest difference in concentrations between B1 and B2 also noted under these conditions. During the period October 1988 to March 1989 when water levels were higher the concentrations recorded were mainly below 2.0 mg/l. There was no significant reduction (at P = 0.025) in the phosphate concentration between the boreholes B1 and B2 for the entire sampling period (Table 3.5.9).

The sodium (Figure 3.5.24, p169) and potassium (Figure 3.5.27, p170) levels in the boreholes greatly exceeded levels detected in the control spring (significant at P = 0.005). The E. C. guide level for sodium of 25 mg/l was exceeded by all samples on all sampling dates while the E. C. maximum admissable concentration of 150 mg/l was exceeded on only one occasion. Potassium levels in both boreholes were also extremely high and the E. C. maximum admissable concentration of 12 mg/l was exceeded in 90% of the samples analysed. Again maximum concentrations were recorded in the summer months, as was the greatest reduction in concentration from B1 to B2. The lowest concentrations occurred in the winter months when water levels were highest.

High levels of bacterial contamination were recorded on all sampling dates. The total Coliform bacterial numbers detected on the various sampling occasions are presented in Figure 3.5.33 (p172). Numbers as high as 2.0×10^4 c. f. u. 's/100 ml were recorded (B1, January 1989). Although a general reduction in numbers from B1 to B2 over the whole sampling period was noted (Table 3.5.9, P = 0.005) the fluctuation in numbers does not follow the pattern described for some of the other parameters. Highest numbers were recorded in both boreholes in September 1988 and in January 1989. This increase coincided with increased rainfall for the five day period preceding sampling (Figure 3.5.58, p180). Similar patterns were recorded for fecal Coliform (Figure 3.5.36, p173) and fecal Streptococci (Figure 3.5.39, p174) bacteria. Microbiological analysis for Clostridia and Salmonella (Table 3.5.3, p182) showed that the numbers of sulphite reducing Clostridia in boreholes B1 and B2 on both sampling dates were low (12 to 20 c. f. u. 's/100 ml), while no Salmonella species were isolated from any of the samples analysed.

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Detergent levels, as methylene blue active substances (M. B. A. S.), were analysed for on two occasions (Table 3.5.4, p182). Concentrations of up to 8.96 mg/l were recorded in B1 and 5.39 mg/l in B2 compared to a background concentration of 0.1 mg/l and an E. C. maximum admissable concentration of 0.2 mg/l.

On three sampling occasions (July 1988, November and March 1989) effluent samples were taken from the septic tanks at all sites in addition to the routine borehole groundwater samples. The percentage reduction in effluent concentration from the septic tank to borehole groundwater was then calculated (Table 3.5.5, p184). The results indicate that the reduction in concentration of effluent parameters between the septic tank and the monitoring boreholes was generally low and variable. Reductions in B. O. D. and C. O. D. showed no distinct pattern, ranging from 11.22 to 89.9% for B. O. D. and from 0 to 80.7% for C. O. D.. Percentage reductions of ammonia, potassium, chloride and sodium were also very low. These results confirm observations in 3:3 and 3:5.2.1 that the effluent is entering groundwater directly with minimum attenuation in the overburden material. In contrast a significant reduction in the phosphate concentrations detected was noted (79.5 to 97.8%). Similar high reductions in the fecal bacterial numbers were recorded (Table 3.5.5). Total and fecal Coliform bacterial numbers were reduced by greater than 99.9% on all three sampling dates, while reduction in fecal Streptococci numbers was slightly lower ranging from 99.3 to 99.9%.

(ii) Site 2

The three monitoring boreholes B3, B4 and B5 (3.85, 4.90 and 10.22 metres downgradient of the septic tank soakpit) were shown to be significantly contaminated with septic tank effluent, although to a lesser degree than at site 1. Elevated levels of NO_3 - N, NH_3 - N, K and conductivity were detected, as well as high numbers of fecal bacteria. The levels recorded for many of the parameters were consistently above background concentrations recorded in the control spring C1 (Table 3.5.8, p188) and on a number of occasions exceeded E. C. guide levels and maximum admissable concentrations.

The B. O. D. (Figure 3.5.5, p184) and C. O. D. (Figure 3.5.7, p186) concentrations detected were above the background levels recorded in C1 (significant at $P = 0.005$, Table 3.5.8). However, the B. O. D. concentrations were low with a maximum value of 4.0 mg/l compared to a background of 1.0 mg/l. Significantly higher C. O. D. levels were recorded. The maximum concentration was 34 mg/l at B3 compared to a background level of 15 mg/l. Although the highest B. O. D. and C. O. D. concentrations were recorded in B3, the reduction in concentration from B3 to B4

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and B5 was minimal. Table 3.5.10 (p190) shows that the reduction in B. O. D. concentration from B3 to B5 was statistically significant (at P = 0.005) but no such reduction was noted in the C. O. D. concentrations recorded (at either P = 0.005 and 0.025).

Conductivity values (Figure 3.5.10) detected in the boreholes were significantly above background values (significant at P = 0.005, Table 3.5.8, p188). As in site 1, the highest concentration (1315 µS/cm at B3) and the largest reduction between monitoring boreholes was recorded in the summer months when water levels in the monitoring wells were low (Figure 3.5.59, p181). For most of the sampling period the reduction in concentration between B3 and B4 and the other borehole B5 was less than 100 µS/cm. Table 3.5.10 shows that there was a statistically significant reduction in the conductivity values recorded between B3 and B5 throughout the sampling period (significant at P = 0.025). However, this reduction could not be demonstrated at a high degree of confidence (not significant at P = 0.005).

Chloride (Figure 3.5.13, p165) concentrations also exceeded the recorded background concentrations (Table 3.5.8). The maximum concentration detected was 68.9 mg/l compared to a corresponding background concentration of 38.2 mg/l (B3, September 1988). However, the levels detected were extremely variable, as was the reduction in concentration in the wells with increasing distance from the soakage pit. No statistically significant reduction in the chloride concentration between B3 and B5 was noted (Table 3.5.10).

Figure 3.5.16 (p166) presents the nitrate concentration detected in the boreholes and the control site C1. The levels recorded show a large increase over background values (significant at P = 0.005). Of the 27 samples analysed 26 (96%) had NO_3 - N concentrations in excess of the E. C. guide level of 5.6 mg/l NO_3 - N, although the E. C. maximum admissable concentration of 11.3 mg/l was not exceeded. The maximum concentration recorded was 10.62 mg/l NO_3 - N (B4, August 1988) with highest levels detected in the summer months when water levels were low. Little reduction in concentration was noted between B3 and B4 but levels recorded in these boreholes were generally greater than in B5 (significant at P = 0.025 but not at 0.005). Conversely the ammonia concentrations detected were low in comparison to those at site 1 (Figure 3.5.19, p167). The levels recorded were highest in B3 although the maximum concentration of 1.06 mg/l was detected in B4. This value is almost three times the E. C. maximum admissable concentration. Of the 27 samples analysed 13 (48%) had NH_3 - N levels in excess of the E. C. guide level of 0.038 mg/l NH_3 - N.

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Figure 3.5.22 (p168) presents the concentration of phosphate detected in the boreholes. The results show a marked reduction in the PO_4 - P recorded in comparison to site 1. Highest levels were detected in B3 where concentrations above background level were recorded on four occasions, the maximum recorded value being 0.47 mg/l PO_4 - P. In the other boreholes B4 and B5, levels above background were recorded on only one occasion (November 1988) when the concentrations detected were 0.39 and 0.37 mg/l respectively.

Sodium (Figure 3.5.25, p169) and potassium (Figure 3.5.28, p170) were present in the boreholes in concentrations significantly above background (Table 3.5.8, p188). Sodium levels of up to 54 mg/l were recorded compared to an average background concentration of 20 mg/l. There was a large variation in the sodium concentration on the different sampling dates. Levels were again highest in the summer months and on most sampling occasions the level of sodium in B3 and B4 was greater than in B5. Potassium levels in all three boreholes were well in excess of the background levels recorded in C1 (Table 3.5.8). Highest concentrations were detected in B3 where the maximum value of 19.7 mg/l was 14 mg/l above the background concentration recorded. The E. C. guide level of 10.0 mg/l K was exceeded by four of the 27 (14.8%) samples analysed, two of which (7.4%) also exceeded the E. C. maximum admissable concentration of 12.0 mg/l (both from the monitoring borehole B3).

Large numbers of fecal bacteria were recorded in the groundwater samples. Figure 3.5.34 (p172) shows the numbers of total Coliform bacteria detected in the monitoring boreholes and the control spring C1. Numbers up to 1.0×10^4 c. f. u. 's/100 ml were isolated from B3 and a general decrease in numbers isolated with increasing distance from the septic tank system was noted (significant at $P = 0.005$). The fluctuation noted in the numbers detected was closely related to increased rainfall events prior to sampling (Figure 3.5.58, p180). Similar increases in response to rainfall events were noted in the numbers of fecal Coliforms (Figure 3.5.3, p173) and fecal Streptococci (Figure 3.5.41, p174). Table 3.5.3 (p182) shows that no sulphite reducing Clostridia or Salmonella species were isolated from the boreholes on either of the two sampling dates.

Table 3.5.4 (p182) shows that high concentrations of detergents (M. B. A. S. mg/l) were detected in the boreholes on both sampling dates (November, 1988 and June, 1989). The highest concentration detected of 4.1 mg/l in B3 is over 20 times the E. C. maximum admissable concentration of 0.2 mg/l. Values of 1.00 and 1.98 mg/l were recorded for B4 and B5 respectively which is still considerably greater than the E. C. maximum admissable concentration.

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The results presented in Table 3.5.6 (p185) demonstrate that the reduction in concentration of various parameters from the effluent to the monitoring boreholes is significantly greater than those recorded at Site 1. B. O. D. and C. O. D. reductions were significant, ranging from 99.2 to 100% for B. O. D. and 96.4 to 99.2% for C. O. D. Similar high reductions were noted for ammonia (96.3 to 100%) and phosphate (98.3 to 100%). However, the poor adsorption capacity of the overburden material (noted in 3:5.2.1) is demonstrated in the low and variable percentage reductions recorded for the potassium (60.3 to 92.3%) and sodium (56.9 to 94.6%) ions . As at site 1 very significant reductions in fecal bacterial numbers were recorded for both the Coliform (greater than 99.9%) and the fecal Streptococci bacteria (99.5 to 99.9%).

(iii) Site 3

Analysis of the monitoring boreholes B6, B7 and B8 (1.50, 1.50 and 9.77 metres downgradient of the septic tank percolation field) showed that the groundwater was contaminated to some extent by the septic tank effluent. However, the degree of contamination was considerably less than at the other two sites as the majority of the parameters measured were similar to background concentrations recorded in the control borehole C2 (Table 3.5.8, p188).

Organic contamination as indicated by B. O. D. and C. O. D. concentrations (Figure 3.5.8) was minimal. B. O. D. concentrations recorded were similar to background levels in all three boreholes. However, C. O. D. values above background were frequently recorded. The maximum concentration detected of 47 mg/l was markedly greater than the background concentration of 10 mg/l. Other parameters such as pH (Figure 3.5.3), sodium (Figure 3.5.26) and potassium (Figure 3.5.29) were all reduced to background levels, confirming observations in 3:5.2.1 that the soil/overburden cover had a high attenuation capacity.

Nitrate levels were however significantly greater than background levels (Table 3.5.11, p191, significant at $P = 0.005$) in all three monitoring boreholes (Figure 3.5.1, p1667), although concentrations were in general well below the E. C. guide level of 5.6 mg/l NO_3 - N. The highest value recorded of 6.21 mg/l at B7 is much higher than the general concentration observed. A mean NO_3 - N concentration of 2.76 mg/l was recorded compared to a mean background concentration of 0.80 mg/l. A general decrease in concentration from B6 to B8 was noted although this decrease could not be demonstrated with a high level of confidence (significant at $P = 0.025$ but not at 0.005). Figure 3.5.17 shows that, unlike the trend in nitrate concentrations reported at sites 1 and 2 (where a decrease in concentration was recorded

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in the winter/spring months), there appears to be a general decrease in NO_3 - N from October 1988 to March 1989 with the lowest concentration being recorded in the summer months.

Ammonia (Figure 3.5.20, p167) concentrations were noticeably lower than levels recorded at the other two sites. However, elevated concentrations above background levels were detected, especially in B7 where the highest level was 1.9 mg/l (almost five times the E. C. maximum admissable concentration). All six samples from the monitoring boreholes B7 exceeded the E. C. drinking water guide level, ranging from 0.085 to 1.9 mg/l NH_3 - N. The levels recorded in B6 and B8 were considerably lower, exceeding the E. C. guide level on only three occasions (0.25 mg/l at B6, 0.28 mg/l and 0.9 mg/l at B8). Table 3.5.11 (p191) shows that there was no statistically significant reduction in concentration between the boreholes B6 and B8 (at $P = 0.025$).

Phosphate contamination of the groundwater was minimal (Figure 3.5.23, p168). On two occasions the concentration detected in B7 exceeded background levels (0.22 and 0.58 mg/l) while none of the samples from the other two boreholes exceeded background concentrations at any time.

High numbers of fecal bacteria were recorded in all three monitoring wells. The results are presented in Figures 3.5.35 (total Coliform bacteria, p172), 3.5.38 (fecal Coliform bacteria, p173) and 3.5.41 (fecal Streptococci bacteria, p174). The fluctuation in numbers detected followed the same pattern in response to rainfall as was noted at the other sites. Maximum numbers recorded were again in the monitoring borehole B7, with a significant decrease (significant at $P = 0.005$) in numbers in the other two boreholes. This reduction is clearly demonstrated in Table 3.5.13 (p202). Further bacterial analysis showed that no sulphite reducing Clostridia or Salmonella species were present in the monitoring wells on either of the two sampling dates (Table 3.5.3, p182).

Detergent (M. B. A. S.) concentrations (Table 3.5.4, p182) were considerably lower than those reported for the other two sites. However, the levels detected were still notably above the background concentrations recorded at the control borehole C2 and the E. C. maximum admissable concentration was exceeded.

Table 3.5.7 (p186) shows the reduction in concentration in effluent parameters from the septic tank to the monitoring boreholes. The results confirm observations in 3:5.2.1 that the overburden material has a high attenuation capacity. Greater percentage reductions than those recorded at the other two sites were evident. B. O. D. and C. O. D. concentrations were markedly reduced. The reduction in B. O. D. (99.6 to 100%) was, however, more significant than that of C. O. D.

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Table 3.5.13

The Reduction in the Numbers of Fecal Bacteria Isolated from the Monitoring Boreholes at Site 3 (January 1989) with Increasing Distance from the Soil Treatment System

Monitoring Borehole	Total Coliforms (c. f. u. 's/100 ml)	Fecal Coliforms (c. f. u. 's/100 ml)	Fecal Streptococci (c. f. u. 's/100 ml)
B7	4.0×10^3	1.0×10^3	40
B6	2.7×10^3	8.0×10^2	32
B8	2.0×10^3	3.0×10^2	16

(94.5 to 100%). Other parameters were also shown to be more significantly reduced than at the other two sites e. g. NH_3 - N (98.7 to 100%), PO_4 - P (98.2 to 100%), K (100%) and Na (83.4 to 100%). The reduction in the chloride concentration was again extremely variable, ranging from 57.3 to 100%.

As at the other sites, very high reductions in fecal bacterial numbers were recorded. Less than 0.1% of total and fecal Coliform and fecal Streptococci bacteria discharged in the effluent were isolated in the groundwater samples (i. e. greater than 99% reduction). A notable exception to the results observed is the percentage reduction recorded for the nitrate ion. On two of the three sampling occasions the concentration of NO_3 - N in the boreholes exceeded that in the effluent.

Another anomaly is the clear increase in the conductivity values detected in the monitoring boreholes compared to those in the control samples (Figure 3.5.11, p164). This increase is not, however, attributed to contamination by septic tank effluent. It is more likely due to the use of a water softener on the control borehole C2. Although the softener was turned off prior to sampling (as described in 3:3) the exact dimensions of the well were not known and it is possible that the sample was taken before all the softened water had been removed from the borehole. Similar increases in Ca, Mg and alkalinity above those observed in the control borehole are also attributed to the inadvertent sampling of partially softened water (Figures 3.5.60 and 3.5.61, p204).

It is also noted that the concentrations of Ca and Mg recorded in B6, B7, and B8 were markedly greater than would be expected given the geology of the area. However, as described in 3:3, the overburden material in the vicinity of the boreholes is composed of 2.5 metres of loamy soil overlying 4.0 metres of permeable sands/gravels and coarser material. It is possible that much of this material (up to 3.0 metres) is composed of outwash sands and gravels of glacial origin. The material would have a high storage capacity and undoubtedly contributes much of the water to the monitoring wells. This in turn may have resulted in the ' carbonate ' nature of the groundwaters analysed. Furthermore it is likely that the thickness of these glacial deposits is greater in the area around the control borehole C2, which would explain the need for a water softner.

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Figure 3.5.60
Concentration of calcium (mg/l) detected in the boreholes at site 3 (B6 and B8) and the control borehole C2 between July 1988 and June 1989

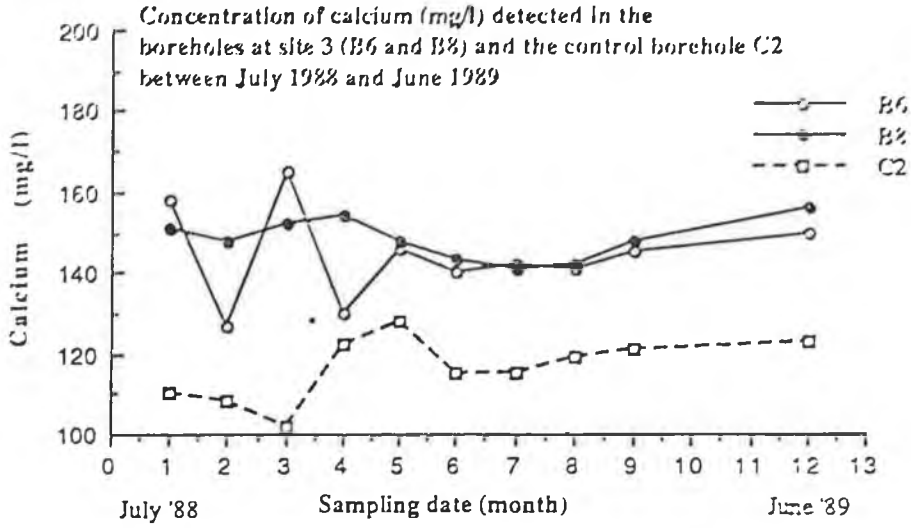
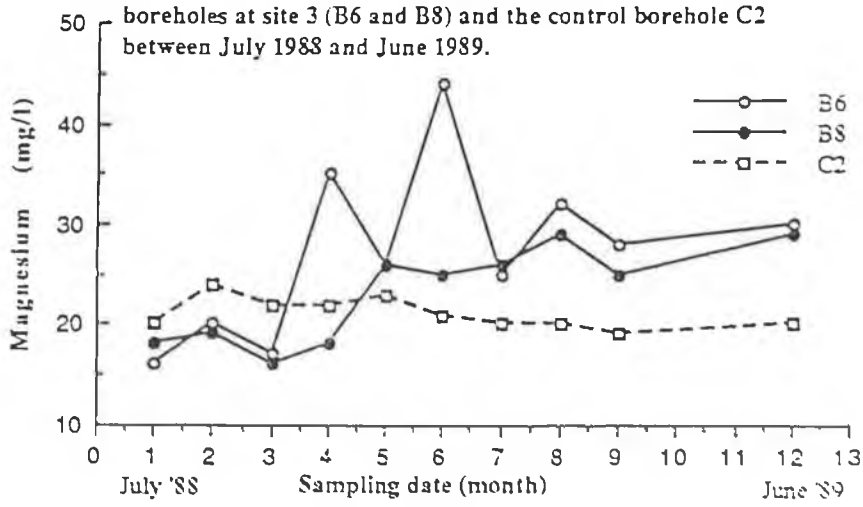


Figure 3.5.61
Concentration of magnesium (mg/l) detected in the boreholes at site 3 (B6 and B8) and the control borehole C2 between July 1988 and June 1989.



The discussion is divided into two sections:

3:6.1 The main findings of the study are summarised.

3:6.2 This section discusses the pollution of the groundwaters by specific effluent constituents and is further subdivided as follows:

3:6.2.1 Nitrogen

3:6.2.2 Phosphate

3:6.2.3 Sodium/Potassium

3:6.2.4 Fecal bacteria

3:6.2.5 Chloride, Conductivity and Detergents.

3:6.1 Summary

The results of the 11 month monitoring programme show that the groundwaters at the three test sites were contaminated by septic tank effluent. The nature and severity of groundwater contamination at each site was dependent on the composition and thickness of the soil/overburden material in the unsaturated zone, and the extent of weathering in the underlying saturated zone.

(i) Site 1

At site 1 the septic tank effluent was discharged to a 2.0 metres deep soakage pit which was partially submerged beneath the water table for the entire sampling period. Although some of the effluent may have percolated laterally through the sides of the soakage pit into the surrounding loamy soil (a process referred to as side wall attenuation by Winneberger, 1984) it is likely that most of the effluent entered groundwater directly and was largely unaltered by reactions with the overburden material. This was confirmed on analysis of the groundwater in the monitoring boreholes (B1 and B2) downgradient of the system. In general the parameters analysed for were markedly above background concentrations and were on many occasions in excess of E. C. directive maximum permissible levels for waters intended for human consumption (Anon., 1980). High concentrations of ammonia, potassium, sodium, chloride, phosphate and detergents, in addition to elevated conductivity values and fecal bacterial numbers, were recorded on all sampling dates. The highest level of contamination was noted during the summer months when water levels in the monitoring wells were lowest. This was attributed to increased dilution during the winter months rather than additional attenuation in the overburden material during the drier periods. An exception to this observation was the numbers of fecal bacteria isolated from the groundwater samples. The fluctuation in the numbers isolated did not

show any seasonal trends. However, a clear relationship between fecal bacterial numbers in the groundwater samples and the rainfall intensity for the five day period preceding sampling was noted (Figures 3.5.51, p178 and 3.5.54, p179).

Reductions in the concentrations of pollutants with increasing distance from the septic tank system was minimal. With the exception of chloride, conductivity and fecal bacteria, no significant reduction was noted from B1 to B2. The small reduction in pollutant concentration with increasing distance from the treatment system was attributed to the rapid transport of contaminants in the highly weathered, permeable material in the saturated zone. In contrast, the numbers of bacteria isolated in borehole B2 were significantly lower than in B1. This reduction was possibly due to adsorption and filtration losses in the subsurface materials. Alternatively it may also be attributed to the migration of the bacteria in specific high permeability fracture/fissure zones, thus bypassing B2.

(ii) Site 2

At site 2 the septic tank effluent was discharged to a 2.0 metres deep soakage pit underlain by 2.8 to 3.6 metres of unsaturated sand and gravel overburden (depending on the water table level). Analysis showed that the percolating effluent may receive limited treatment in the overburden. However, the high infiltration rate and low cation exchange capacity recorded would curtail most of the attenuation, facilitating rapid migration of the effluent with few exchange sites available for adsorption and ion exchange. It was, however, noted that the large pore spaces would enable oxygen to diffuse through the overburden material and permit oxidation reactions with the percolating effluent constituents.

These observations were also confirmed on analysis of the groundwater samples. Elevated concentrations of nitrate, ammonia, potassium, sodium and detergents as well as high conductivity values and fecal bacteria numbers were recorded in the three monitoring boreholes. As at site 1 the highest levels of contamination were recorded during the summer months when water levels in the monitoring boreholes were lowest. Again this was attributed to increased dilution during the winter months rather than increased attenuation in the overburden material during the drier periods. The shallow borehole B3 was consistently more contaminated than the others (B4 and B5). The high levels of nitrate recorded in the monitoring boreholes were attributed to rapid nitrification of ammonia in the permeable overburden material, with subsequent leaching of the nitrate ion to groundwater. The highest levels were recorded in the monitoring borehole B3 during the summer months, when levels greater than those known to be toxic to infants were detected (Stewart and Stoleman, 1961; Patterson et al, 1971; Bitton and Gerba, 1984 and Keeney, 1986).

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High levels of sodium and potassium were detected in the monitoring boreholes in comparison to the control sample, indicating the poor ion exchange and adsorptive properties of the overburden material. The ratio of potassium to sodium in the samples was very variable but generally less than 0.4. Phosphate contamination of the groundwater was minimal, with the soil/overburden material effectively immobilising the ion, probably by a combination of adsorption and precipitation mechanisms. This is consistent with results reported by Jones and Lee (1979).

High numbers of fecal bacteria were isolated from the boreholes. This concurs with literature reports which have demonstrated that sand and gravel overburden is largely ineffective in restricting the movement of bacteria (Caldwell, 1938; Patterson et al, 1971; Bitton and Gerba, 1984 and Sinton, 1986). The relationship between elevated bacterial numbers in the groundwater samples and increased rainfall prior to sampling was again demonstrated. A high linear correlation was shown to exist between the numbers of fecal Coliform bacteria and rainfall for the five day period preceding sampling. This result is attributed to a combination of the physical ' flushing ' of bacteria through the soil overburden material with the percolating rainfall, and the desorption of previously fixed bacteria due to a decrease in the ionic strength of the soil solution.

The reduction in concentration of the contaminants with increasing distance from the soakage pit was again shown to be minimal. Small reductions in the concentration of B. O. D., nitrate, ammonia, sodium and conductivity were noted from B3 to B5 but these were not statistically significant at a high confidence level (Table 3.5.10, p190). This result demonstrates the mobility of septic tank effluent pollutants in permeable fissured subsurface materials. However there was a significant reduction in the numbers of fecal bacteria isolated with increasing distance from the system (B3 to B5). This may be due to the migration of the organisms in fissures and cracks in the saturated bedrock, thus bypassing the monitoring borehole.

(iii) Site 3

The septic tank effluent at site 3 was discharged into a series of one metre deep distribution trenches underlain by an unsaturated zone consisting of 1.5 metres of a sandy loam material and 0.5 to 2.0 metres of permeable sands, gravels and cobble sized material (depending on the water table level). Analysis of the material indicated that, because of the moderate infiltration rate and high cation exchange capacity, the effluent from the soil percolation field would have adequate contact time with the soil particles for ion exchange and adsorption reactions to proceed. Most attenuation would take place in the 1.5 metres of overburden material described above, with little or no reduction occurring in the underlying permeable layers.

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The high attenuation capacity of the overburden material was confirmed on analysis of the groundwater samples. Groundwater contamination by the septic tank effluent was considerably lower than at the other two sites. The concentration of many of the parameters including potassium, sodium, B. O. D., C. O. D., phosphate and chloride was reduced to the background levels recorded in the control borehole C2. However, the groundwater samples were shown to be consistently contaminated by high levels of fecal bacteria. This result is significant as the nature and thickness of the unsaturated zone available for effluent attenuation has previously been reported to be effective in restricting the movement of fecal bacteria by a combination of filtration and adsorption mechanisms (Bourma et al, 1972; Ziebell et al, 1974; McCoy and Ziebell, 1975; Hagedorn et al, 1981; Bitton and Gerba, 1984; and Kaplan, 1987). It is possible that the organisms migrated through cracks and jointings in the sandy loam soil material into the underlying permeable layers where they would meet little resistance to their transport to groundwater. Migration of fecal bacteria in specific zones of higher permeability through cracks and joints in overburden material is well documented in the literature (Lewis et al, 1982; Sinton, 1986 and Chen, 1988). The numbers of fecal bacteria isolated from the monitoring boreholes were again shown to have a direct relationship with increased rainfall prior to sampling.

The levels of nitrate recorded in the boreholes B6, B7 and B8 were significantly lower than those detected at site 2. It is likely that much of the ammonia in the percolating effluent was immobilised by adsorption to clay surfaces in the loamy overburden material. However, the levels detected were markedly greater than those in the control borehole C2. This was attributed to partial nitrification of ammonia with subsequent leaching of the nitrate ion to groundwater. The fluctuations in the nitrate levels recorded did not follow the pattern observed at sites 1 and 2. The lowest values were recorded during the summer months, with variable but generally higher concentrations detected during wetter periods when water levels in the monitoring wells were higher. This was attributed to an increase in the amount of nitrate leached due to the mobilisation of previously ' fixed ' nitrogen in the loamy soil as water levels in the boreholes and the soil moisture content increased. This is consistent with reports by De Walle and Scaff (1980) and Lewis et al (1982).

As was observed at the other sites, the reduction in concentration of the contaminants with increasing distance from the soakage pit was minimal. Reduction in the concentration of nitrate, ammonia, sodium and potassium from B6 to B8 was low, demonstrating the mobility of the pollutants in permeable fissured subsurface material. However, a significant reduction ($P = 0.005$) in the concentration of fecal bacteria isolated with increasing distance from the system (B6 to B8) was again observed (Table 3.5.11, p191).

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3:6.2 Contamination of the Groundwaters by Specific Effluent Constituents

3:6.2.1 Nitrogen

Nitrogen from the septic tank effluents was shown to be one of the major pollutants of the groundwaters at the three test sites. Concentrations recorded in the monitoring boreholes were well above background levels on all sampling dates (Table 3.5.8, p188). On a number of occasions (especially at site 1 and 2) E. C. guide levels and maximum admissable concentrations for drinking water were exceeded. High levels of ammonia were detected in the boreholes at site 1. This result confirms observations in 3:4 and 3:5.1 that the effluent from the septic tank was being discharged directly to groundwater with little effective attenuation or chemical alteration in the overburden material. It also indicates that the anaerobic process within the septic tank releases nitrogen mainly in the form of ammonia (Appendix B 5). This has previously been reported by a number of authors including Patterson et al (1971) and Canter and Knox (1985). Conversely high nitrate concentrations were recorded in the boreholes at sites 2 and 3, where it is likely that nitrification of ammonia in the unsaturated zone occurred with subsequent leaching of nitrate to groundwater.

The nitrate concentrations detected in the groundwater at the three sites are presented in Figures 3.5.15 to 3.5.17 (p166). The highest levels were detected in the boreholes at site 2 (B3, B4 and B5) where concentrations were well in excess of the E. C. drinking water guide level of 5.6 mg/l. The maximum concentrations of 10.0 (B3) and 10.62 mg/l (B4) were only slightly lower than the E. C. drinking water M. A. C. of 11.3 mg/l and exceeded concentrations known to have caused methaemoglobinemia (infantile cyanosis) in infants (Stewart and Stoleman, 1961; Bitton and Gerba, 1984 and Lewis et al, 1982). At site 3 (Figure 3.5.17) the concentration of nitrate in the boreholes was also well above background levels. However, the E. C. guide level was only exceeded on one occasion (6.21 mg/l in B7). The mean nitrate concentration of 2.76 mg/l detected in the groundwater at the site was markedly above the mean background concentration of 0.8 mg/l recorded in the control borehole C2. In contrast the nitrate concentrations detected at site 1 were generally low and below those recorded in the control spring C1.

Seasonal variations in the concentration of nitrate detected were similar at sites 1 and 2. Maximum concentrations were recorded in the summer months when water levels were low (water levels in the monitoring wells at both sites fluctuated by approximately 1.0 metre, Figure 3.5.59, p181). The lowest water level recorded in the monitoring wells at site 1 was 1.98 metres, still leaving 0.02 metres of the soakage pit below the water table. Thus the effluent leaving the pit was discharged directly into groundwater throughout the year and the increased concentrations of nitrate were

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unlikely to be due to an increase in the nitrification of ammonia. Conditions at site 2 were similar during the summer months. The high permeability and low moisture holding capacity of the overburden material, together with the large pore size (3:5.1), would permit rapid nitrification and leaching of the nitrate ion and as such a 1.0 metre increase in the thickness of the unsaturated zone during the summer months would have little effect on this process. Continuous leaching of nitrate from permeable overburdens has previously been reported by Canter and Knox (1985) who recorded that effluent from septic tanks located in sandy soils can be expected to undergo predominantly aerobic reactions resulting in nitrification of ammonia and subsequent leaching of nitrate to groundwater. It would thus appear that the increase in nitrate noted during the summer months at sites 1 and 2 was due to reduced dilution in the monitoring wells rather than an increase in nitrification of ammonia.

In contrast, the levels detected at site 3 (Figure 3.5.17, p166) were lowest during the summer months. Concentrations recorded during periods of higher water levels in winter were generally higher and more variable. Figure 3.5.59 (p181) shows that the fluctuation in water levels in the boreholes was more significant than at the other sites. A maximum difference of 1.58 metres was recorded between the spring and summer sampling dates. The increased thickness of the unsaturated zone (due to the drop in water levels) during the summer months could have resulted in an increase in the immobilisation of the nitrate by interaction with overburden material. However, this is unlikely as the overburden material in the zone of water table fluctuation is composed of permeable sands and gravel material (3:4). The fluctuation in nitrate levels was thus more likely to be due to an increase in the quantity of nitrate leached in periods of higher water levels. It is possible that the higher nitrate levels in the groundwater samples were due to increased leaching of nitrate, previously immobilised in the 1.5 metres thick loamy overburden material as water levels in the boreholes rose and the soil moisture content increased. This leaching of nitrate may have been sufficient to offset any possible reduction in the levels by dilution as the water levels in the wells increased. In addition it is demonstrated in Figure 3.5.17 that the peaks in nitrate concentrations recorded in the boreholes occurred in September 1988 and January/February 1989 after periods of heavy or increased rainfall (Figure 3.5.58, p180). An increase in the nitrate level of groundwaters during periods of higher water levels and after heavy rainfall has been reported by a number of authors (De Walle and Scaff, 1980 and Lewis et al, 1982).

The reduction in nitrate concentration with increasing distance from the soil disposal systems was minimal at all three test sites. No statistically significant reduction was noted between the boreholes B1 and B2 at site 1 (Table 3.5.9, p189). The reduction in concentration at site 2 between B3 and B5 was generally less than 1.0

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mg/l (a mean difference of 0.83 mg/l). Table 3.5.10 (p190) shows that this reduction was statistically significant at P = 0.025 but not at a higher level of confidence (at P = 0.005). Similarly, at site 3, reductions in concentrations from B6 to B8 were very low (a mean difference of 0.58 mg/l NO_3 - N). Again the reduction was significant at P = 0.025 but not at a higher level of confidence (at P = 0.005). These results are similar to literature reports by Patterson et al (1971), De Walle and Scaff (1980) and Canter and Knox (1985), all of whom document the mobility of the nitrate ion in permeable subsurface materials beneath septic tank treatment systems.

The levels of ammonia detected in the monitoring boreholes at the three test sites are presented in Figures 3.5.18 to 3.5.20 (p167). Figure 3.5.18 clearly demonstrates the high concentrations recorded at site 1. A mean ammonia concentration of 14.8 mg/l NH_3 - N was detected in the monitoring boreholes with maximum levels of 25.3 and 23.05 mg/l NH_3 - N in B1 and B2 respectively (up to 60 times greater than the E. C. drinking water M. A. C.). No seasonal fluctuation in levels was observed in the boreholes B1 and B2. In addition there was no reduction in concentration with increasing distance from the soakage pit i. e. from B1 to B2 (at P = 0.025). This is reflected in the ammonia levels detected in the monitoring borehole B2 (a lateral distance of 8.1 metres downgradient of the soakage pit) which were similar to those in B1, 1.5 metres downgradient of the pit. On 50% of the sampling dates the levels in B2 exceeded those recorded in B1. Ammonia concentrations detected in the groundwaters at the other two sites were significantly lower although on occasions, especially in the shallow boreholes B3 (site 2) and B7 (site3), concentrations above background were recorded. The E. C. drinking water guide level of 0.039 mg/l NH_3 - N was also frequently exceeded (Figures 3.5.19 and 3.5.20). Again there was no evidence of a seasonal variation in the levels recorded at both sites.

3:6.2.2 Phosphate

The phosphate concentrations detected in the monitoring boreholes at the three test sites are presented in Figures 3.5.21 to 3.5.23 (p168). High phosphate concentrations of up to 3.84 mg/l were recorded at site 1 (B1 and B2) with maximum levels detected in the summer months during dry periods. The concentrations recorded from October 1988 to March 1989 were generally less than 2.0 mg/l PO_4 - P. No phosphate was detected in the control spring C1. These findings are similar to those observed by Dole (1986) who reported that a borehole downgradient of a septic tank system contained 4.13 mg/l PO_4 - P compared to a background concentration of 0.01 mg/l. Although the boreholes B1 and B2 contained high levels of phosphate, the

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reduction in concentration from the septic tank effluent to groundwater was high given the nature of effluent disposal at the site (3:3). Table 3.5.5 (p184) shows that the reduction in phosphate concentration ranged from 79.5 to 97.8%. This may be due to immobilisation of the ion by reactions within the soakage pit. Phosphate can be immobilised by direct anion exchange or adsorption by calcium minerals within the soakage pit and the underlying clogged zone (3:6.2.4) forming insoluble calcium phosphate precipitates such as hydroxyapatite (Chapter 2). The results are similar to reports by Doyle and Thorn (1986) who recorded an 88 % reduction in the septic tank effluent phosphate within a 2.0 metres deep soakage pit.

The phosphate levels recorded in the monitoring wells at sites 2 and 3 were reduced to background levels (Figures 3.5.22 and 3.5.23, p168). It would appear that the overburden material at both sites has the ability to ' fix ' and prevent movement of the ion, probably by a combination of adsorption and precipitation reactions (Chapter 2). The percentage reduction in phosphate levels (Tables 3.5.6 to 3.5.7, p185 and 186) from the septic tank effluent to the monitoring boreholes was very high at sites 2 and 3 where reductions of 98 to 100 % were recorded. These results are consistent with reports by Polkowski and Boyle (1970), Patterson et al (1971), Ellis (1973) and Jones and Lee (1979) who found that the risk of phosphate pollution of groundwaters from septic tank effluent is minimal due to immobilisation of the ion in the overburden material.

3:6.2.3 Sodium/Potassium

The concentration of sodium (Figures 3.5.24 to 3.5.25, p169) and potassium (Figures 3.5.27 to 3.5.28, p170) detected in the monitoring boreholes at sites 1 and 2 was consistently above the background levels recorded in the control spring C1. At site 1 very high sodium concentrations of up to 160 mg/l were recorded (Figure 3.5.24). High potassium concentrations (up to 37.0 mg/l) were also detected. The E. C. maximum admissable concentration of 12 mg/l K was exceeded in 90% of the samples analysed. At site 2 high concentrations of sodium (Figure 3.5.25) and potassium (Figure 3.5.28) were recorded in the wells B3, B4, and B5. The high levels of both ions in the groundwater samples is an indication of the poor adsorptive capacity (3:5.1) of the sand and gravel overburden material, as these ions are immobilised in soil mainly by cation exchange reactions (see Chapter 2).

Higher levels of sodium were recorded during the summer months at both sites. This result is contrary to reports by De Walle and Scaff (1980) who observed that waters polluted by septic tank effluent had lower sodium concentrations in summer. He attributed this to the greater electrolytic strength of the soil solution during the summer, which increased the rate of sodium - calcium ion exchange mechanisms.

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Because of the permeable nature of the overburden material at site 2 it is unlikely that the electrolytic concentration in the soil would significantly change throughout the year and consequently no increase in ion exchange reactions would be expected. Similarly, at site 1, there was no decrease in sodium concentrations during the summer months as the effluent was discharged directly to groundwater throughout the sampling period, with minimal interaction with the overburden material. The decrease in sodium concentrations during the winter months at both sites was possibly due to higher water levels, giving increased dilution in the monitoring wells.

The concentrations detected at site 3 (B6, B7, and B8) were significantly lower than those recorded at the other two sites. On average the concentrations of both ions detected in the monitoring boreholes (B6 and B8) did not exceed those in the control borehole C2 (no significant difference at $P = 0.025$). However, on a number of sampling occasions the sodium concentrations in the boreholes B6 and B7 (Figure 3.5.26, p169) were substantially greater than those recorded in the control borehole. This increase may have been due to a decrease in the electrolytic strength of the soil solution (described by De Walle and Scaff, 1980) and an increase in soil moisture content during the wetter periods, resulting in increased leaching of sodium ions to groundwater.

The potassium concentrations in the boreholes (B6, B7 and B8) are presented in Figure 3.5.29 (p170). Elevated potassium levels were recorded on only one sampling occasion (6.1 and 6.2 mg/l at B6 and B7 respectively) in September 1988. In general the levels were lower than those recorded in the control borehole C2, reflecting the ability of the overburden material to effectively immobilise the potassium in the percolating septic tank effluent.

The percentage removal of the sodium and potassium ions in the overburden material at the three sites is shown in Tables 3.5.5 to 3.5.7 (p184 to 186). The reduction in concentration of both ions was very low and variable at site 1 (Table 3.5.5) where the concentrations in the groundwater samples occasionally exceeded that in the effluent. Similarly, at site 2 (Table 3.5.6), the percentage reductions were low, ranging from 56.9 to 94.6% for sodium and 60.3 to 92.3% for potassium. The greater attenuation capacity of the overburden material at site 3 is reflected in the high percentage reductions noted in boreholes B6, B7 and B8 (Table 3.5.7). Reductions in sodium concentrations of 84 to 100% were recorded and the potassium ion was completely immobilised (100% reduction). These results are in agreement with observations by Ellis, B. (1973) and Ellis, J. (1980) who report that the potassium ion is more readily adsorbed onto overburden material than the sodium ion.

Figures 3.5.30 to 3.5.32 (p171) present the K/Na ratios calculated for the groundwater samples from the three test sites and the control sites C1 and C2.

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Summary statistics of these results are presented in Table 3.6.1 (p215). The ratio of potassium to sodium in the samples analysed was generally low. However, Figures 3.5.30 and 3.5.31 show that the ratios recorded at sites 1 and 2 were very variable (0.13 to 0.51) and were significantly greater than the background values recorded in the control spring C1 (significant at P = 0.005). At site 3 (Figure 3.5.32), where contamination of groundwater by both ions was minimal, the calculated ratios were consistently less than 0.2 and similar to the background values recorded in the control borehole C2. It was noted that 95.2% of the samples analysed from the test sites (B1 to B8) had a K/Na ratio less than 0.4, while none of the ratios calculated for the control sites (C1 and C2) exceeded this value. The reduction in the K/Na ratio in the groundwater samples from sites 1 and 2 (mean 0.26 and 0.27 respectively) to site 3 (mean 0.10) demonstrates the high adsorptive capacity of the overburden material at site 3 and confirms observations made above that the potassium ion is more strongly adsorbed than the sodium ion.

In summary, the results indicate that although the groundwaters at sites 1 and 2 contained elevated concentrations of potassium and sodium, the K/Na ratio was generally low (less than 0.4). At site 3 the pollution of groundwater by both ions was negligible and the K/Na ratio was reduced to background levels of 0.2 or less.

3:6.2.4 Fecal Bacteria

Large numbers of fecal bacteria were isolated from the monitoring boreholes at the three test sites on all sampling dates. The total Coliform (Figure 3.5.33 to 3.5.35, p172), fecal Coliform (Figure 3.5.36 to 3.5.38, p173) and fecal Streptococci (Figure 3.5.39 to 3.5.41, p174) bacteria numbers recorded were markedly above the background levels in the control samples C1 and C2. None of the groundwater samples from the three test sites were microbiologically fit for human consumption under E. C. drinking water standards.

The highest numbers were isolated from the boreholes at site 1 (B1 and B2) where levels of 2.0×10^4 c. f. u. 's/100 ml total Coliform (Figure 3.5.33, p173), 1.6×10^4 c. f. u. 's/100 ml fecal Coliform (Figure 3.5.36, p173) and 1.8×10^2 c. f. u. 's/100 ml fecal Streptococci (Figure 3.5.39, p174) bacteria were recorded in Janurary 1989. However, when compared with the very large numbers of fecal bacteria in the effluent (up to 1.0×10^6 c. f. u. 's/100 ml fecal Coliform and 8.0×10^4 c. f. u. 's/100 ml fecal Streptococci bacteria) and the nature of effluent disposal at the site, the numbers isolated from the monitoring wells were relatively low. This is highlighted in the results presented in Table 3.5.5 (p184) where it is shown that less than 0.1% of the Coliform bacteria discharged to the soakage pit

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Table 3.6.1

Summary Results of the Calculated K/Na Ratio in Groundwater Samples from the Monitoring Boreholes (B1 to B8) and the Control Samples (C1 and C2)

Site	No. Samples.	Max.	Min.	Median	Mean	Std. Dev.
1 (B1, B2)	20	0.40	0.13	0.25	0.26	0.07
2 (B3, B4, B5)	27	0.51	0.16	0.24	0.27	0.09
3 (B6, B7, B8)	26	0.21	0.04	0.10	0.10	0.04
C1	10	0.33	0.10	0.17	0.18	0.06
C2	10	0.18	0.04	0.10	0.12	0.05

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were isolated from the monitoring boreholes B1 and B2 (i. e. greater than 99.9% reduction). Similar high reductions of 99.1 to 99.9% were recorded for fecal Streptococci. It is possible that many of the organisms in the percolating effluent were filtered out in the ' clogged zone ' or biological mat at the base of the soakage pit. The formation of such clogged zones beneath septic tank soakage pits has been reported by Winneberger (1984) and Kaplan (1987). The large reduction in the numbers of fecal bacteria is consistent with reports by Ziebell et al (1974) and McCoy and Ziebell (1975) who observed that fecal bacterial reductions of greater than 99.9% could be achieved in the top few centimeters of the biological mat formed beneath a septic tank distribution field. The results are also consistent with those obtained in Chapter 2. The high levels of ammonia in the groundwater boreholes (up to 25 mg/l NH_3 - N) may also have been directly toxic to the fecal bacteria, contributing to the reduced numbers in the samples analysed (Krieg and Holt, 1984). The marginally lower reduction in the fecal Streptococci bacteria numbers may indicate superior survival of the Streptococci species over the Coliforms in soil and groundwater systems. This has been reported by a number of authors (Wheater et al, 1976; Bitton and Gerba, 1984 and Lewis at al, 1982). Figures 3.5.33 (p172) and 3.5.36 (p173) show a significant reduction in the numbers of total and fecal Coliform bacteria isolated from the boreholes B1 and B2 (significant at P = 0.005). This reduction may be attributed to filtration and adsorption losses in subsurface weathered material or to the migration of bacteria in fissures or zones of higher permeability, thus bypassing the monitoring borehole B2. The movement of bacteria in zones of higher permeability is well documented in the literature (Patterson et al, 1971; Lewis et al, 1982; Sinton, 1986 and Chen, 1988). Similar statistically significant reductions in the numbers of fecal Streptococci isolated from B1 and B2 were noted (at P = 0.005).

Large numbers of fecal bacteria were also isolated from the monitoring boreholes at site 2. Maximum numbers were isolated in January 1989 when 1.0×10^4 c. f. u. 's/100 ml total Coliform (Figure 3.5.34, p172), 6.5×10^3 c. f. u. 's/100 ml fecal Coliform (Figure 3.5.3, p173) and 8.0×10^1 c. f. u. 's/100 ml fecal Streptococci (Figure 3.5.40, p174) bacteria were recorded. The large numbers isolated are consistent with literature reports which indicate that the type of overburden material present at the site is largely ineffective in restricting the migration of bacteria. The high permeability and the large percentage of coarse sized particles in the overburden material at the site (3:5.1) offer little resistance to the migration of bacteria as the two main processes by which bacterial migration is prevented (i. e. filtration and adsorption) are largely ineffective under such conditions (Caldwell, 1938; Patterson et al, 1971; Bitton and Gerba, 1984 and Sinton, 1986). However, as was observed at site 1, the reduction in the numbers of fecal bacteria

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from the effluent to groundwater samples was very significant (Table 3.5.6, p185). Reductions of greater than 99.9% were recorded for the Coliform bacteria while reductions of between 99.5 and 99.9% were observed for the Streptococci bacteria. It is again possible that many of the organisms were removed in the biological mat at the base of the soakage pit, with minimal removal occurring in the underlying permeable and porous overburden material.

There was a significant reduction in the numbers of bacteria isolated with increasing distance from the soakage pit i. e. from B3 to B5. Highest numbers were recorded in the shallow borehole B3, with significantly lower numbers in B4 and B5 (at P = 0.005). This reduction may be due to filtration and adsorption losses in the subsurface material or to the migration of the organisms in specific depth zones as described above. Boreholes B6, B7 and B8 contained significantly fewer fecal bacteria than the other two sites. However, numbers of 4.0×10^3 c. f. u. 's/100 ml total Coliforms (Figure 3.5.35, p172), 1.0×10^3 c. f. u. 's/100 ml fecal Coliforms (Figure 3.5.38, p173) and 40 c. f. u. 's/100 ml fecal Streptococci bacteria (Figure 3.5.41, p174) were recorded in January 1989. These relatively large numbers are surprising given the nature of the overburden material at the site and are contrary to literature reports which observe that similar type soil material underlying distribution trenches was very effective in removing fecal bacteria from percolating effluents by a combination of filtration and adsorption mechanisms (Bourma et al, 1972; Ziebell et al, 1974; McCoy and Ziebell, 1975; Hagedorn et al, 1981; Bitton and Gerba, 1984 and Kaplan, 1987). Table 3.5.7 (p186) shows that the percentage reduction in the numbers of fecal bacteria from the effluent to groundwater was greater than 99.9% for both the Coliform and Streptococci bacteria. Much of this reduction may be due to filtration losses in the biological mat at the base of the distribution trenches. It is possible that some of the organisms which passed through the biological mat were not restricted in the soil but migrated through cracks or jointings in the loamy soil/overburden. Once the organisms passed through this layer to the underlying sands and gravels there would be little resistance to their migration to groundwater. The migration of fecal bacteria in specific zones of higher permeability through cracks and joints in overburden material has been documented in the literature (Lewis et al, 1982; Sinton, 1986 and Chen, 1988). The largest numbers of fecal bacteria were isolated from the shallow borehole (B7) with a significant reduction in numbers with increasing distance from the distribution system i. e. from B6 to B8 (significant at P = 0.005). Large numbers up to 2.0×10^3 c. f. u. 's/100 ml total Coliform, 3.0×10^2 c. f. u. 's/100 ml fecal Coliform and 10 c. f. u. 's/100 ml fecal Streptococci bacteria were isolated from monitoring borehole B8, a lateral distance of 9.77 metres downgradient of the distribution system. These reports are consistent with those of

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Viraraghavan (1978) who observed that fecal bacteria numbers in a series of monitoring boreholes decreased with increasing distance from a soil distribution system underlain by sandy loam soil, and that large numbers were isolated from a borehole at a distance of 15.25 metres from the end of the distribution trenches.

The ratio of fecal Coliform to fecal Streptococci bacteria (FC : FS) for all samples analysed was extremely variable both between sites and on the different sampling dates. Highest ratios were recorded at site 1 where the FC : FS ratio was consistently greater than 100. Lower ratios were recorded at site 2 (mean 67) and at site 3 (mean 26). All samples had a ratio greater than four, which is similar to reports by Wheater et al (1976) and Mara (1974) that waters contaminated by human sewage have high FC : FS ratios. The decrease in the ratio with increasing attenuation capacity of the overburden material (i. e. from site 1 to site 3) may be due to the superior survival characteristics of the Streptococci bacteria in the soil/overburden systems. Alternatively it may indicate that the fecal Coliform bacteria are more susceptible to losses by filtration and adsorption in the overburden. There are no literature reports to support this observation. Recent reports, however, do not recommend the use of the ratio to distinguish between animal and human contamination because of the difference in the survival and migration rates of the two species in both soil and groundwater systems (Lewis et al, 1982).

No seasonal fluctuation in the numbers of bacteria isolated from the boreholes was observed. As described above maximum numbers were recorded at the three test sites in January 1989. Another peak in the numbers isolated was noted in September 1988. Figure 3.5.57 (p180) presents the monthly rainfall (mm) during the sampling period. No relationship between the levels of bacteria in the groundwater and the monthly rainfall was noted. However, when the total rainfall (mm) for the five day period preceeding sampling was plotted and compared to bacterial numbers (Figure 3.5.58, p180), the correlation between rainfall events and bacterial contamination of the groundwater was clearly demonstrated. Maximum numbers were isolated in January 1989 when a total of 39.8 mm of rainfall was recorded for the five days preceeding sampling (Figure 3.5.5, p184). Conversely the lowest numbers of bacteria were recorded in the period October to December 1988 when the rainfall preceeding sampling was low, ranging from 10 to 13.9 mm. This relationship between heavy or prolonged rainfall and increased bacterial contamination of groundwater sources is well documented in the literature (Patterson at al, 1971; Bitton and Gerba, 1984; Lewis et al, 1982; Sinton, 1986; Kaplan, 1987; Chen, 1988 and Drew, 1987) and is attributed to a combination of the physical ' flushing ' of bacteria through the soil overburden material and desorption of previously fixed bacteria due to a decrease in the ionic strength of the soil solution. The

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relationship is more clearly demonstrated in Figures 3.5.51 to 3.5.53 (fecal Coliforms, p178) and Figures 3.5.54 to 3.5.56 (fecal Streptococci, p179). The graphs show a marked increase in the numbers of bacteria isolated in the boreholes in response to increased rainfall preceeding sampling. However, the relationship is complex. The graphs 3.5.51 to 3.5.53 show that large numbers of fecal Coliform bacteria were isolated from the boreholes at all sites in January 1989. Low numbers were then recorded on the following three sampling occasions despite the fact that the rainfall before sampling ranged from 18.4 to 24.2 mm, a level which resulted in large numbers of organisms appearing in the monitoring boreholes in September 1988. It would appear that the heavy rainfall preceeding sampling in January caused the ' flushing ' of large numbers of previously ' fixed ' organisms to the groundwater boreholes. As a result subsequent rainfall events did not give rise to a corresponding high level of bacteria contamination.

The relationship was further investigated by preparing scatter plots of the number of fecal Coliform bacteria isolated from the monitoring wells and control sites (c. f. u. 's/100ml) against the total rainfall (mm) for the five days preceeding sampling (Appendix B 4). Examination of the distribution suggested that a linear relationship existed. Simple regression curves were fitted to the plots and Pearson's coefficients of correlation (r) and determination (r^2) were then calculated. The significance of the correlation was tested using the t statistic (3:4.5.2) and the results are presented in Table 3.5.12 (p192). Correlation coefficients of 0.66 and 0.73 were obtained for the regression lines at site 1. The correlations were, however, not significant at B1. Although the correlation at B2 was significant at P = 0.01, this was not demonstrated at the higher confidence level (at P = 0.005). Higher correlation coefficients were obtained at site 2 (0.89 and 0.84) and site 3 (0.85 and 0.86), all of which were shown to be significant at the upper confidence level (P = 0.005). It would thus appear that there is a stronger linear relationship between increased rainfall and elevated bacterial numbers at sites 2 and 3 than at site 1. A strong linear relationship (significant at 0.005) between increased rainfall and elevated bacterial numbers was also noted in the control sites C1 and C2 (r = 0.94 and 0.85 respectively).

Low numbers of sulphite reducing Clostridia were isolated from the boreholes at site 1 (12 to 20 c. f. u. 's/100ml) but no Salmonella species were recorded on either of the two sampling dates (Table 3.5.3, p182). Neither of the two organisms were isolated from the groundwaters at sites 2 and 3.

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3:6.2.5 Chloride, Conductivity and Detergents

(i) Chloride

The results of the chloride analyses are presented in Figures 3.5.12 to 3.5.14 (p165). Chloride concentrations in the monitoring boreholes at all three sites were significantly greater than those detected in the control sites C1 and C2. This is consistent with reports by Canter and Knox (1985) who refer to chloride as a common groundwater contaminant and a useful indicator of pollution from septic tank systems.

At site 1 the chloride levels in the boreholes B1 and B2 (Figure 3.5.12) were high and markedly greater than those recorded in the control spring C1. The maximum concentration detected of 111.7 mg/l at monitoring borehole B1 greatly exceeded the corresponding value of 37.7 mg/l in the control site C1. The fluctuations in chloride concentrations recorded in the boreholes on the various sampling dates were similar to those noted for the nitrate ion. Highest concentrations were detected in the boreholes during the summer months when water levels were lowest, while levels in the control spring C1 were shown to increase in the wetter periods. This is clearly demonstrated in Figure 3.5.45. The percentage reduction in the concentration of chloride from the septic tank to the monitoring boreholes (Table 3.5.8, p188) was generally very low and variable ranging from 24.3 to 71.0%. However, the reduction was statistically significant over the entire sampling period at P = 0.005 (Table 3.5.9). Similar results have been reported by Saffigna and Keeney (1977) and Jones and Lee (1979). They found that chloride was highly mobile in overburden materials and groundwater systems and that its movement was similar to that of the nitrate ion.

Chloride concentrations detected in the boreholes at site 2 (Figure 3.5.13, p165) were, on occasions, far in excess of those recorded in the background spring C1. The maximum concentration detected was 68.9 mg/l (B3, September 1988) compared to a corresponding background concentration of 38.2 mg/l. However, the fluctuation in the levels detected was extremely variable as were the percentage reductions in concentration from the septic tank to the monitoring boreholes (Table 3.5.6, p185). In general the highest concentrations were recorded in the first half of the sampling period with a general decrease in the levels detected thereafter. The chloride concentrations detected in borehole B3 were notably lower than those in the control sample on all sampling dates after December 1989. No statistically significant difference between the levels recorded in B3 and the control spring was noted over the entire sampling period (Table 3.5.8, p188). In addition, although the chloride concentrations in the other two boreholes B4 and B5 were significantly greater than those recorded in the control spring (at P = 0.025), this could not be demonstrated with a high degree of confidence (at P = 0.005). Unlike the

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results obtained at site 1 and previous reports in the literature, no relationship was found between the concentrations of chloride and nitrate ions detected in the groundwater samples (Figure 3.5.46, p176). The reason for the sharp reduction in the concentrations recorded in the monitoring boreholes for the period December 1988 to June 1989 is not known.

The chloride levels recorded in the boreholes at site 3 were significantly greater than those recorded in the control borehole C2. Maximum concentrations were recorded in the winter months when water levels in the monitoring boreholes were highest. This may indicate increased leaching of the chloride ion from the loamy overburden material during wetter periods when the electrolytic strength of the soil was reduced. Similar results were obtained for the sodium and nitrate ions. Reduction in concentration from the septic tank to the monitoring boreholes was higher than that observed at site 1, but lower and more variable than that observed for some of the other parameters (Table 3.5.7, p186). Figure 3.5.47 (p176) shows a distinct relationship between the concentrations of chloride and nitrate detected in the boreholes, with similar seasonal fluctuations recorded for both ions. This is consistent with literature reports as discussed above.

(ii) Conductivity

The electrical conductivity values detected in the groundwater samples are presented in Figures 3.5.9 to 3.5.11 (p164). High conductivity values were recorded at site 1 (Figure 3.5.9) where values were on average 400 µS/cm above background levels in C1. As was observed with most of the effluent pollutants at this site the highest concentrations were recorded in the summer months when the water levels in the monitoring boreholes were lowest (Figure 3.5.59, p181). The maximum value recorded of 1583 µS/cm (B1, August 1988) was 930 µS/cm above the corresponding background level. These results are consistent with reports by Jones and Lee (1979) who observed that monitoring boreholes in the vicinity of septic tank systems contained high concentrations of chloride and nitrate, and elevated electrical conductivity values. There was a statistically significant reduction (at $P = 0.005$) in the conductivity concentrations from B1 to B2 over the entire sampling period (Table 3.5.9, p189).

Conductivity values detected in the boreholes at site 2 (Figure 3.5.10, p164) were significantly above the background values recorded in the control spring C1. As in site 1, the highest concentration (1315 µS/cm in B3) and the largest reduction between monitoring boreholes was recorded in the summer months when water levels in the monitoring wells were low (Figure 3.5.59, p181). Table 3.5.10 (p190) shows that there was a statistically significant reduction in the conductivity values between B3 and B5 throughout the sampling period (at $P = 0.025$). However, this reduction could

not be demonstrated at a higher confidence level (not significant at P = 0.005). Again these results are similar to literature reports by Jones and Lee (1979).

The results obtained in the analysis of the boreholes at site 3 show a clear increase in conductivity values over those detected in the control sample C2 (Figure 3.5.11, p164). However, as described in 3:5.2.2, this large increase was not attributed to contamination by septic tank effluent. It is more likely to be due to the inadvertent sampling of partially softened water from the control borehole C2. The fluctuation in the levels of conductivity values in the boreholes were, however, similar to those observed for the majority of the parameters measured at the site. Maximum levels were recorded in the winter months when water levels were highest in the monitoring wells.

(iii) Detergents

The concentrations of detergents detected in the groundwater samples as M. B. A. S. mg/l) are presented in Table 3.5.4 (p182). High levels (4.98 to 6.96 mg/l) were detected in the boreholes at site 1, compared to a background concentration of 0.1 to 0.2 mg/l and an E. C. maximum admissable concentration of 0.2 mg/l. These results are contrary to reports by Polkowski and Boyle (1970) who found that the concentration of M. B. A. S. detected in a borehole downgradient of a septic tank system was very low. This was attributed to the complete degradation of the detergent in the aerobic soil treatment system. Patterson et al (1971) reported that M. B. A. S. concentrations in septic tank effluent are almost totally degraded in a soil treatment system provided that conditions remain aerobic. The nature of effluent disposal at this site resulted in the discharge of the anaerobic effluent directly to groundwater (3:3), with little oxygenation occurring in the overburden. This would explain the results obtained.

High levels were also detected at site 2 where concentrations of 1.0 to 4.0 mg/l were recorded. Maximum concentrations were recorded in the shallow borehole B3 and there was a notable reduction with increasing distance from the soakage pit (from B3 to B5). Although conditions at this site would allow some oxidation of the effluent to occur, the overburden material is highly permeable and the contact time between the detergents in the percolating effluent and the soil/overburden material may not have been sufficient to ensure complete degradation of the substances.

The greater attenuation capacity of the overburden material at site 3 was demonstrated as the detergent levels recorded were considerably lower than those detected at the other two sites. The concentrations detected ranged from 0.2 to 0.5 mg/l and were consistently greater than background concentrations of 0.1 to 0.2 mg/l.

6:7 Conclusions

6:7.1 Effluent Attenuation at the Three Test Sites

The main conclusions of the study are as follows:

(i) The attenuation of effluent by the septic tank treatment systems at the three test sites was incomplete resulting in chemical and microbiological contamination of groundwaters downgradient of the systems. The nature and severity of contamination was dependent on the composition and thickness of the soil/overburden material in the unsaturated zone and the extent of weathering in the underlying saturated bedrock. At all three sites the groundwater was unsuitable for human consumption under the E. C. drinking water directive (Anon., 1980).

(ii) The groundwater downgradient of the treatment system at site 1 was grossly polluted by septic tank effluent. On all sampling occasions high concentrations of ammonia, potassium, sodium, chloride, phosphate and detergents, in addition to elevated conductivity values and fecal bacterial numbers, were recorded in the groundwater at a distance of 8.1 metres downgradient of the treatment system. The highest levels of contamination were recorded during the summer months when the water levels in the monitoring wells were lowest. This is attributed to increased dilution during the winter months rather than additional attenuation in the overburden material during the drier periods.

The results suggest that the use of septic tank/soakage pit treatment systems in areas where fissured bedrock is close to the surface should be discontinued and replaced with suitable alternatives.

(iii) At site 2 the results demonstrate that 2.7 to 3.6 metres of unsaturated sandy overburden material was insufficient to effectively restrict the movement of effluent pollutants and prevent the contamination of the underlying groundwaters. Elevated levels of nitrate, ammonia , potassium, sodium and detergents, as well as high conductivity values and fecal bacterial numbers, were recorded in the groundwater at a distance of 10.22 metres downgradient of the treatment system. As was noted at site 1, the highest levels of contamination were recorded during the summer months when water levels in the monitoring boreholes were lowest. Again this is attributed to increased dilution during the wetter periods.

(iv) The higher attenuation capacity of the overburden material at site 3 was demonstrated in the results obtained. The contamination of the groundwater downgradient of the treatment was considerably less than was observed at the other two sites. Many of the test parameters including B. O. D., C. O. D., potassium, sodium, phosphate and chloride were reduced to background levels. However, high numbers of fecal bacteria and elevated nitrate levels were consistently recorded at a distance of up to 9.77 metres downgradient of the treatment system. This result indicates that the unsaturated zone, consisting of 1.5 metres of a sandy loam soil overlying 0.5 to 2.4 metres (depending on the water table) of unsaturated sands and gravels, was insufficient to effectively attenuate the septic tank effluent constituents. It is considered that the effluent pollutants migrated through high permeability cracks and joints in the sandy loam overburden limiting the contact time between the effluent and the soil colloids and reducing the efficiency of the soil filtration, adsorption and ion exchange mechanisms.

(v) The reduction in the concentration of pollutants recorded in the groundwaters with increasing distance from the treatment systems was minimal, demonstrating the high mobility of effluent contaminants in the permeable weathered material of the saturated zone. It would appear that most of the effluent treatment and attenuation occurred in the septic tank soil treatment systems and unsaturated soil/overburden layers, with little reduction in the underlying saturated material.

3:7.2 The Contamination of Groundwater by Specific Effluent Constituents

The main conclusions on the contamination of groundwater by specific effluent constituents are:

(i) Nitrogen

Nitrogen from the septic tank effluent was shown to be one of the major groundwater pollutants at the three test sites. High concentrations of ammonia were recorded in the monitoring boreholes at site 1 (B1 and B2) indicating that the effluent was discharged directly to groundwater with little effective attenuation or chemical alteration in the overburden material. Ammonia concentrations recorded at the other two sites were significantly lower although on a number of occasions, especially in the shallow boreholes B3 (site 2) and B7 (site 3), concentrations above background levels were recorded. No seasonal fluctuations were noted in the levels of ammonia recorded.

High concentrations of nitrate nitrogen were detected at site 2 and 3 where it is likely that nitrification of ammonia in the unsaturated zone occurred with subsequent leaching of the nitrate to groundwater. The highest levels were recorded at site 2

indicating rapid nitrification of effluent ammonia in the permeable sandy overburden material. The lower levels recorded at site 3 are attributed to the partial immobilisation of effluent ammonia by adsorption to clay surfaces in the loamy overburden material.

The seasonal fluctuations in the levels of nitrate recorded were similar at site 1 and 2. Maximum concentrations were recorded in the summer months when water levels were lowest. It is considered that this increase is due to reduced dilution in the monitoring wells rather than an increase in the nitrification of effluent ammonia during drier periods. In contrast the nitrate levels detected at site 3 were lowest during the summer months and generally greater during periods of higher water levels. There was also evidence of elevated nitrate levels after heavy or increased rainfall events. It is possible that this increase was due to the mobilisation and leaching of previously fixed nitrate in the overburden material as the water levels and soil moisture content increased.

The reduction in nitrate concentrations with increasing distance from the three treatment systems was minimal demonstrating the mobility of the ion in permeable saturated materials.

(ii) Sodium/Potassium

High concentrations of sodium and potassium were recorded in the groundwaters at sites 1 and 2 illustrating the poor treatment efficiency of both systems. At site 1 the removal of the ions within the treatment system was minimal. On many occasions the concentrations recorded in the groundwaters exceeded that in the septic tank effluent. Similarly, at site 2, the high levels of both ions recorded in the groundwater samples demonstrated the poor adsorptive capacity of the sandy overburden material. Again the percentage removal of the effluent sodium and potassium in the overburden material was low i. e. 56.4 to 94.6% for sodium and 60.3 to 92.3% for potassium. At both sites the highest levels were recorded during the summer months. This was again attributed to increased dilution in the monitoring wells during the wetter periods.

The greater attenuation capacity of the overburden material at site 3 is reflected in the low levels of both ions recorded in the groundwaters (B6, B7 and B8). The results demonstrate that 1.5 metres of unsaturated sandy loam overburden was sufficient to effectively immobolise both ions and prevent their transport to groundwater. High percentage removals of the effluent sodium and potassium in the overburden material were noted. Sodium concentrations recorded in the groundwater samples were reduced by between 84 and 100% of that discharged in the septic tank effluent. There was, however, evidence of increased leaching of sodium during wetter periods possibly due to a reduction in the electrolytic strength of the

soil/overburden with increased soil moisture content. Complete immobilisation of the potassium ion was recorded (i. e. 100% reduction) indicating that the potassium ion is adsorbed onto the overburden material in preference to the sodium ion.

The results obtained also demonstrate that the contamination of groundwater by septic tank effluent can result in an increase in the K/Na ratio above background levels although the ratios recorded rarely exceeded 0.4. It was also noted that 95.2% of the groundwater samples analysed at the test sites (B1 to B8) had a K/Na ratio less than 0.4 while none of the control samples exceeded this value. It is thus considered that the K/Na ratio may be a useful test parameter in distinguishing between groundwater contamination from septic tank systems and other point sources of contamination such as landfill leachates, animal slurries and farmyard runoff which have considerably higher K/Na ratios.

(iii) Phosphate

The phosphate levels recorded in the groundwaters at sites 2 and 3 were not significantly greater than background levels. It would appear that the overburden material at both sites has the ability to fix and prevent the movement of the phosphate ion, probably by a combination of sorption and precipitation mechanisms.

Higher concentrations were recorded at site 1 (B1 and B2). However, the reduction in the concentration of phosphate from the septic tank to the groundwater was very high (ranging from 79.5 to 97.8%) given the nature of effluent disposal at the site. This is attributed to immobilisation of the ion by direct anion exchange or adsorption/precipitation reactions within the soakage pit.

The results indicate that the risk of groundwater pollution by phosphate contained in septic tank effluents is minimal due to the immobilisation of the ion in the treatment system and overburden materials.

(iv) Fecal Bacteria

Large numbers of total Coliform, fecal Coliform and fecal Streptococci bacteria were isolated from the groundwaters at the three test sites on all sampling dates. At sites 1 and 2 the large numbers isolated are attributed to the absence of an effective unsaturated zone (site 1) in addition to the high permeability and large percentage of coarse sized particles in the overburden material (site 2) whereas at site 3 the contamination is possibly due to the migration of the organisms through cracks and joints in the loamy soil overburden.

Unlike the trends noted for many of the other parameters, there was a distinct reduction in the numbers of fecal bacteria isolated with increasing distance from the treatment systems at all three test sites. This reduction may be attributed to filtration

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and adsorption losses in the subsurface weathered material or the preferential migration of the bacteria in fissured zones of higher permeability thereby bypassing the monitoring boreholes.

Less than 0.1% of the effluent fecal Coliform bacteria discharged to the soil treatment systems at the three test sites were isolated from the groundwater samples. This represents a percentage reduction of greater than 99.9%, considerably greater than recorded for the other measured parameters. It is possible that the majority of the bacteria in the percolating effluent were filtered out in the clogged zone or biological mat at the base of the soakage pits and percolation trenches. The percentage reduction in the numbers of fecal Streptococci bacteria samples was slightly lower than recorded for the Coliform organisms (i. e. 99.1 to 99.9%). This result may indicate superior survival characteristics of the Streptococci bacteria in soil and groundwater systems.

No seasonal fluctuations in the numbers of bacteria isolated from the groundwater samples was observed. However, the results show that the maximum numbers of fecal bacteria were isolated from the groundwaters at the three sites after periods of heavy or prolonged rainfall and that, conversely, the lowest numbers were isolated following drier periods. Statistical analysis of the results at sites 2 and 3 demonstrated a strong linear relationship between increased rainfall for the 5 day period prior to sampling and elevated bacterial numbers in the groundwater samples. This increase is attributed to a combination of the physical flushing of bacteria through the overburden material by the percolating rainwater and the desorption of previously fixed bacteria as a result of a decrease in the ionic strength of the soil solution. No statistically significant relationship was demonstrated in the results obtained at site 1.

(v) Detergents

Degradation of the effluent detergents (as M. B. A. S.) within the septic tank treatment systems was incomplete. High concentrations (in excess of E. C. drinking water maximum permissible levels) were recorded in the groundwaters downgradient of all three treatment systems. This is attributed to:

(a) The discharge of the anaerobic effluent directly to groundwater thereby preventing the oxidation of effluent detergents (site 1)

(b) Insufficient contact time between the effluent and soil colloids due to the high permeability of the overburden material (site 2) or migration of the effluent through cracks and joints in the overburden material (site 3).

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CHAPTER 4

AN ASSESSMENT OF THE SUITABILITY OF A RANGE OF CHEMICAL AND BIOLOGICAL TRACERS TO MONITOR THE MOVEMENT OF SEPTIC TANK EFFLUENTS TO GROUNDWATER

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4:1 Introduction

A tracer is matter or energy carried to or by groundwater which will give information regarding the direction of movement and/or velocity of the groundwater and the potential contaminants which might be transported therein. A tracer can be natural, such as heat carried by warm spring waters or accidental, such as fuel leakage from a ruptured storage tank, or it may be intentionally introduced by man such as dye tracing in cave water systems. The purpose and importance of tracing was eloquently described by Dole (1906). His sentiments are perhaps even more relevant today in light of the increased incidence of groundwater pollution:

> It is often a matter of much importance to know whether the flow is from a cesspool toward a neighbouring well or in the opposite direction; it may be necessary to determine whether or not water seeps from a contaminated brook into boreholes of a neighbouring region; whether collecting galleries for public water supplies receive seepage from well established sources of contamination; whether, in general, known foci of pollution are in immediate, though obscured, connection with sources of drinking water. Knowledge of this nature is especially important in the study of waters passing through formations full of seams or crevices, where there is opportunity for rapid circulation without much purification.
>
> (Dole, 1906)

In recent decades there has been a rapid increase in the development and refinement of water tracers and the analytical procedures used in their detection. The use of naturally occurring radioisotopes and ' labelled ' microbiological tracers has allowed very selective and accurate detection of tracer movement. Fluorinated organic acids and halocarbons have also been used and found to be highly sensitive. Despite this, the relatively uncomplicated dye and ion tracers have remained very popular with improvements in analytical detection techniques giving greater sensitivity.

The tracing of septic tank effluent from soil treatment systems to groundwater supplies involves monitoring the percolation of the tracer through varying thicknesses of soil/overburden cover. This limits the number of materials that can be used successfully as some are subject to high adsorption losses in certain soil types while others are lost by physical filtration in soil and rock pores or by various chemical and biological alterations in the unsaturated zone. Tracing is further complicated by the nature of the septic tank effluent which is a complex heterogeneous mixture of chemical and biological waste. As such, a suitable tracer must be capable of tracing the movement of both the chemical and the biological constituents. It has been suggested that tracer dyes do not always accurately represent the movement of microbes (Rahe et al, 1979). Conversely microbiological tracers, which can be subject to high

sorption and filtration losses, may not accurately reflect the movement of highly obile effluent constituents such as nitrate.

To date, few data have been published on the tracing of septic tank effluent to oundwater despite the fact that it has been identified as one of the major pollutants of oundwater supplies. In 1984 septic tank effluent was the most frequently reported rce of groundwater contamination in the United States and was responsible for 58% illnesses caused by the use of contaminated, untreated well water ates et al, 1989). Similar high statistics are reported for Ireland, Canada and New aland. It is likely that the problem is increasing with the rapid expansion of sewered suburban development occurring in these and many other countries.

Research into the tracing of septic tank effluent movement through soils has inly focused on the migration of the microbial constituents of the effluent to oundwater (Bitton and Gerba, 1984) although there has been a limited number of ccessful dye traces. The studies to date have, however, been site specific and no empt has been made to assess the usefulness of a range of tracer types in a number of l/overburden regimes and hydrogeological settings.

In this study the effluents from three septic tanks at sites with different l/overburdens and depths of unsaturated zones were traced to a series of monitoring reholes. A variety of tracer materials were used. The primary aim of the vestigation was to evaluate the relative success of a number of tracer types and to ntify those best suited to the specific site conditions. The investigation is presented follows :

1 Introduction

2 Literature review: This section reviews the relevant literature on the chemical and microbiological tracing of organic effluents in soil and groundwater systems

3 Site characteristics: A report on the three sampling sites used is given in this section

4 Materials and Methods: The materials and methods used in the preparation, sampling and analysis of the chemical and microbiological traces are presented here

5 Results: The main results of the study are presented and described in this section

6 Discussion: The results are discussed in detail with reference to previous investigations

7 Conclusions.

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4:2 Literature Review

4:2.1 Historical

The first tracing experiment on record was carried out about 2000 years ago when Philip the Tetrarch of Trachonitus threw chaff into a crater lake. He reported that the chaff reappeared downgradient of the lake in one of the springs at the headwaters of the river Jordan. This link was later shown to be highly unlikely by Mazor in 1976. The first serious attempts at tracing were carried out in the karst areas of Europe at the turn of the century, using mainly dyes and salt as tracers. Dole (1906) refers to work carried out by a French doctor tracing the origin of a typhoid fever outbreak in Paris in 1882. The invention of the fluoroscope in 1901 by M. Tillant (refined by M. Marbirtin) greatly increased the precision of fluorescent dye measurements. Tracer work in the early 1900's on soils and karst systems using sodium fluorescein is described by Dole, who is himself given credit for pioneering the use of fluorescein in English speaking countries. Around the same time sodium chloride was being used by Adolf Theim to determine the flow velocity of water. In his work samples were taken which were then analysed for chloride in the laboratory. In 1902, Slichter modified Theim's work by obtaining a continuous field read out of electrical conductivity. Ammonium chloride was also used by Slichter in his experiments.

In the 1950's radioactive tracers were developed allowing very precise and selective tracer measurements. They proved very popular but in recent years their use has been curtailed because of public health concerns. The development of tracers such as naturally occurring radio - isotopes in the 1960's revolutionised the science of water tracing but these too have been overtaken by the development of extremely sensitive organic acid and halocarbon tracers. Recent advances have also resulted in an increase in popularity of microbiological tracers (mainly bacteria and viruses). They have been particularly useful in tracing microbial - rich wastes to groundwater sources (Sinton, 1980; Keswick et al, 1982 and Bitton and Gerba, 1984).

4:2.2 Criteria for a Good Tracer

A good tracer should possess the following properties:

(i) The tracer should not be removed by interaction with solid material i. e. adsorption or ion exchange. In addition, it should not be precipitated by changes in pH or reaction with other ions.

(ii) It should be relatively inexpensive and easily detected using available technology with minimum effort and labour.

(iii) It should be non - toxic and should not require any special precautions in handling or in the storage of samples

(iv) It should not modify the hydraulic conductivity or other properties of the medium being studied.

In addition the following factors should be taken into consideration when assessing the relative usefulness of a tracer material:

(i) The chemical and physical behaviour of the tracer in soil or groundwater should be understood

(ii) It should be present in concentrations well above the background level in the natural system

(iii) Careful consideration must also be given to the possible health implications before artificially introducing foreign substances into a system.

(Elrick and Lawson, 1969; Glover, 1972; Ellis, 1980 and Davis et al, 1984)

4:2.3 Types of Tracers

The following section presents a brief description of the tracer materials which have previously been used in soil and groundwater studies. They are broadly grouped into the following categories:

4:2.3.1 Physical

4:2.3.2 Chemical

4:2.3.3 Microbiological.

4:2.3.1 Physical

(i) Temperature Variation

The use of water temperature as a groundwater tracer is one option which, to date, has found little application in the field of groundwater tracing. Water has a much greater heat capacity than most of the subsurface material it comes in contact with and consequently it's temperature either remains the same or changes very slowly as it passes through the subsurface. Any change in temperature due to outside influences would therefore quickly become noticeable. Using water temperature as a tracer is particularly useful in areas overlain by coarse - grained overburden, fractured weathered subsurfaces or karst regimes (Davis et al, 1984). The main advantages to

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using variations in water temperature as a tracer are:

(i) A direct field measurement is easily obtainable eliminating the need for laborious and time - consuming laboratory analysis and reducing the error incurred in transporting the sample from the field site to the laboratory

(ii) The equipment involved is relatively inexpensive and easy to use

(iii) Minimal environmental disruption is caused as only clean water is injected.

The main disadvantage is that changes in water temperature are accompanied by changes in viscosity and density which in turn alter the velocity and direction of flow. The warmer water has a slightly lower density and tends to float on top of the cooler water. Water injected into groundwater at a temperature of 40 ^{o}C will travel twice as fast as water in the same aquifer, under the same hydraulic gradient, at a temperature of 5 ^{o}C (Davis et al, 1984).

The use of cool water as a tracer was examined by Simpson et al (1985). Icicles of water containing a radioactive isotope of Iodine, I_{131}, were injected into a borehole in an alluvial aquifer. No temperature or radiation changes were detected in the monitoring boreholes. The trace was, however, successful when a standard liquid trace was used. It was thought that the higher density of the cooler water caused the trace to sink and miss the monitoring boreholes. In order to minimise errors due to changes in density and viscosity, Keys and Brown (1978) used smaller temperature differences between the tracer and groundwater and successfully detected the readings using a probe with a sensitivity of 0.02 ^{o}C.

It would appear that the most important application for water temperature as a tracer is to serve as an aid in groundwater tracing experiments. It can be used to estimate the breakthrough time of other tracers, and as a simple inexpensive means for determining the best location for monitoring boreholes.

(ii) Solid Particles

Solid materials in suspension can be used successfully as groundwater tracers in areas of large conjugate flow or with highly fissured permeable overburden. It has been reported that bales of hay, wheat chaff and even live geese were at one time used as tracers in the karst regions of Missouri. In recent times particulate tracers have been much smaller in size. Their use, however, is of limited value as they are only suitable for conjugate flows and cannot be used successfully in studies of soil and groundwater systems. Although some authors include bacteria, yeasts and viruses as solid particle tracers, they are dealt with separately in 4:2.3.3.

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(a) Paper and Sample Floats

These tracers range in sophistication from uniform pieces of paper to polypropylene floats. The particles will usually float on the surface of the water and consequently their velocity will be greater than the mean velocity of the water body. Their main application is in approximating flow velocity and establishing flow paths (Davis et al, 1984). One advantage with using floats is that little equipment is required. However, the particles must be recovered and counted manually which is laborious and time consuming.

(b) Spores

The most common non - bacterial spore used as a tracer is Lycopodium, a club moss whose spores have a mean diameter of 33 µm. They are slightly denser than water and require a certain degree of turbulence to keep them in suspension (Davis et al, 1984). A mean velocity of a few miles per hour is generally sufficient. The spores survive in polluted water but do not perform well where the water velocity is slow or where suspended sediment concentrations are high. Their use has largely been confined to karst regions. Buchtela et al (1968) compared the use of Lycopodium stained with uranin (a fluorescent dye) to sodium chloride. The Lycopodium was found to travel more rapidly. This was attributed to the fact that the spores tend to remain in suspension in waters with a higher velocity.

The advantages of using Lycopodium as a tracer are that the spores are relatively small and the injection concentration can therefore be very high. In addition, they are not adversely affected by water chemistry, or adsorbed onto clay or silt surfaces. The movement of the spores also tends to approximate the flow of the surrounding water. Greater than five traces can be run simultaneously because the spores are easy to dye and there are no health risks associated with their use. The main disadvantages are that the preparation of the spores for the trace, injection of the spores into the groundwater system and subsequent sampling and analysis is time consuming. Sampling and analysis can also be difficult if the silt content of the water is very high.

4.2.3.2 Chemical

The use of chemical water tracers is discussed under the following headings:

1. Dyes
2. Ions

1. Dyes

Inorganic dyes (both fluorescent and non fluorescent) have been used in water tracing experiments since the late nineteenth century (Dole, 1906). The use of fluorescent dyes became widespread during the 1960's, possibly because they are easily detected. However, dyes travel more slowly than water due to adsorption onto overburden materials and other surfaces and are generally not as conservative as ionic or radioactive tracers (Smart and Laidlaw, 1977). Non - fluorescent dyes are seldom used nowadays.

Unlike physical and particle tracers dyes can be used to trace the movement of waters through soils to groundwater. McLaughlin (1982), who reviewed this application, concluded that fluorescent dyes have high detectability at low concentrations. No single dye is suited to all applications. The dyes can vary greatly in their susceptibility to environmental conditions especially during long - term field studies. In addition, because they consist of large molecules, they are adsorbed in varying degrees onto soil surfaces which can give problems in the quantitative interpretation of results (McLaughlin, 1982 and Smart and Laidlaw, 1977).

As yet, no dye has been developed which can accurately mimic the movement of water in soils. However, in comparison to other tracers, dyes appear to offer great potential for simultaneously relating solute distribution in soils to soil physical and morphological characteristics (McLaughlin, 1982). The advantages of using fluorescent dyes as tracers are:

(i) Ease of detection
(ii) The relatively low cost
(iii) They are generally not environmentally disruptive or toxic.

The main disadvantages are:

(i) They are unstable and must be stored in dark glass bottles to prevent photochemical decay. In addition analysis must be carried out as soon as possible after sampling
(ii) Analysis usually involves the procurement of very costly equipment
(iii) The dyes themselves are visually objectionable

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(iv) Adsorption onto surfaces can be significant making quantitative estimation difficult. Detection levels can also be influenced by pH, salinity and calcium carbonate concentrations in groundwater thereby affecting the validity of comparisons made between different sites.

(Smart and Laidlaw, 1977; McLaughlin, 1982 and Davis et al, 1984)

The most commonly used fluorescent dyes are sodium fluorescein, pyranine, lissamine FF, rhodamine B, rhodamine WT, sulpho rhodamine B and the optical brighteners e. g. leucophor PBS (Davis et al, 1984).

(i) Sodium Fluorescein

Sodium fluorescein (also known as uranin, fluorescein and pthalein) is a green fluorescent dye which has found widespread use in groundwater tracing. It's use is well documented and was first reported by Dole in 1906. It has been widely used for tracing studies in karst regions (Atkinson et al, 1973) and to trace leachate movement from a sanitary landfill (Smart, 1985). Few data are, however, available on the use of sodium fluorescein in tracing experiments from soil to groundwater. Omoti and Wild (1979) reported that sodium fluorescein was one of the best tracers for soil studies. However, Rahe et al (1979) did not recover any injected dye from the soil 2.5 metres downslope of the injection point, despite the fact that bacterial tracers were used successfully in the same study.

One of the main problems with the use of sodium fluorescein has been interference from high background concentrations, making interpretation of results difficult (Davis et al, 1984). It also has a high photochemical decay rate (Smart and Laidlaw, 1977). It has been recommended in the literature that sodium fluorescein traces be restricted to slightly alkaline or neutral waters as the structure of the molecule can change to a colourless form under acidic conditions (Corey, 1968; Knuttson, 1968 and Smart and Laidlaw, 1977). There is conflicting evidence on the effect of salinity on the fluorescent nature of the dye. Davis et al (1984) reports that an appreciable decrease in fluorescence is displayed with increasing salinity while no decrease in fluorescence was reported by Smart and Laidlaw (1977).

There are a number of advantages to using sodium fluorescein in tracing experiments. It can be visually detected in a water sample at one part in 40 million and at one part in 10 billion with a fluorimeter (Corey, 1968). The dye is also less costly than many other tracer dyes but due to its high photochemical decay rate and high adsorption losses it's relative cost is increased (Davis et al, 1984).

(ii) Pyranine

Pyranine is the second most commonly used green fluorescent dye. It has the advantage of having stronger fluorescence than sodium fluorescein although it is considerably more expensive (Davis et al, 1984). It has been successfully used as a soil water tracer and was considered to be the most promising of the dyes tested for tracing in an acidic sandy soil (Reynolds, 1966). Reynolds also concluded that pyranine was superior to sodium fluorescein as a tracer of the percolation of rain water through soil profiles. Omoti and Wild (1979) found that pyranine as well as sodium fluorescein were the most useful dyes for soil water tracing. They did, however, express some concern about the stability of pyranine with increasing soil organic matter content.

The main disadvantages in using pyranine as a tracer are that it has a high photochemical decay rate and is adversely affected by the pH of most natural waters. Pyranine has also been found to be unstable in experiments lasting longer than 24 hours. This instability appears to be due to the fact that it can solubilize organic matter in the soil resulting in colour quenching (McLaughlin, 1982).

(iii) Lissamine FF

There is little information available on the performance of lissamine FF but Smart and Laidlaw (1977) suggest that it is the best of the green dyes. They report that it is less susceptible to photochemical decay and is absorbed onto humus to a lesser extent than other dyes.

(iv) Rhodamine B

Rhodamine B is an orange fluorescent dye which has frequently been used as a water tracer. Both this dye and rhodamine WT have the lowest minimum detectability of all the tracer dyes (Smart and Laidlaw, 1977). The degree of fluorescence is not affected by the fluctuations in pH experienced in most natural waters and it does not suffer significantly from photochemical decay (Smart and Laidlaw, 1977). However, Smart and Laidlaw noted that rhodamine B was more susceptible to chemical decay than rhodamine WT. The main disadvantage associated with the use of the dye is that it is readily adsorbed onto materials (Reynolds, 1966; Knuttson, 1968; Smart and Laidlaw, 1977 and Davis et al, 1984) (Figure 4.2.1).

Knuttson (1968) reports that rhodamine B is more useful than fluorescein as a tracer because of it's lower photochemical decay rate and greater resistance to decay by bacterial action. The dye is, however, sensitive to temperature fluctuations and displays optical quenching in the presence of high concentrations of suspended solids.

The use of rhodamine B as a water tracer was also recommended by Marston and Schofield (1962).

Rhodamine B is generally regarded as the most toxic of the tracer dyes as it is readily adsorbed onto body tissue (Smart and Laidlaw, 1977 and Davis et al, 1984). In toxicity experiments it was found to be significantly more toxic than fluorescein (Smart and Laidlaw, 1977). Two year old oysters died after two days exposure to 100 mg/ l of Rhodamine B. No ill effects were demonstrated on exposure to 1 mg/l. Both rhodamine B and sodium fluorescein have been classified as having a toxicity level of C111 (toxicity rating) by the Food and Agriculture Organisation and the World Health Organisation (Davis et al, 1984).

(v) Rhodamine WT

This dye is one of the most useful tracers for quantitative studies, based on its low detectability levels, photochemical/chemical decay rates and adsorption losses (Smart and Laidlaw, 1977). No significant problems were noted with pH fluctuations within the normal range encountered in natural waters. Mc Laughlin (1982) reported that the dye was useful in tracing the movement of waste applied to land. In his experiments the dye was detected up to 321 metres from the point of application. Rhodamine WT is less toxic than rhodamine B, having a toxicity level comparable to that of sodium fluorescein (Smart and Laidlaw, 1977).

(vi) Sulpho Rhodamine B

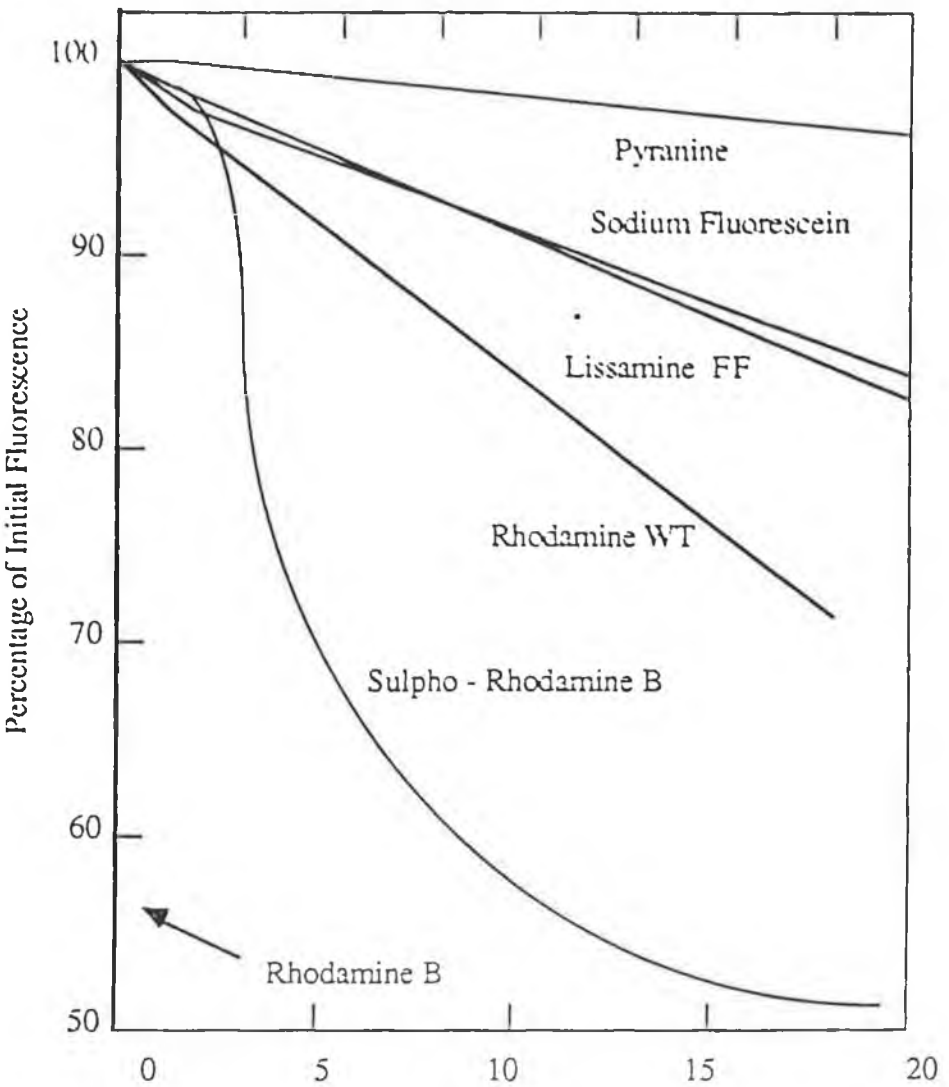
This dye, also known as brilliant pink, is the least used of the orange dyes mainly because of its high cost. It's adsorption to clay surfaces has been shown to be intermediate between that of rhodamine B and rhodamine WT. It is not significantly affected by photochemical, chemical or biological decay nor is it susceptible to changes in pH (Smart and Laidlaw, 1977 and Davis et al, 1984). Sulpho rhodamine B has been shown to be affected by salinity changes (Smart and Laidlaw, 1977). It is also more toxic than rhodamine WT.

(vii) Optical Brighteners

Optical brighteners are blue fluorescent dyes which have been used in recent times by the textile and paper industry to enhance the whiteness of the material. They appear colourless in normal daylight but fluoresce blue under U. V. light. They are responsible for the ' whiter than white ' appearance of white clothes in discotheques where U. V. lights are used for lighting effects. Optical brighteners are perhaps the simplest to use of all the dye tracers. However, they have a high affinity for organic matter and adsorption losses are normally high. Their use in areas contaminated with

Figure 4.2.1

The Relative Adsorption Rates of Various Dyes onto Kaolinite



(After Smart and Laidlaw, 1977)

domestic waste is questionable because of high background interference (Glover, 1972).

The toxicity of optical brighteners has been shown to be very low and they do not constitute a health hazard, even at the excessive dosing levels used in tracing experiments.

2. Ions

Ion compounds have been used extensively as groundwater tracers. The charge on the ion is of paramount importance in determining it's movement through soils and groundwater (Davis et al, 1984). The ions most commonly used as tracers are:

(i) Chloride (Cl^-)
(ii) Bromide (Br^-)
(iii) Lithium (Li^+)
(iv) Sodium (Na^+)
(v) Potassium (K^+)
(vi) Nitrate (NO_3^-)

The adsorption of different ions onto surfaces can vary considerably. Cations are generally less useful as tracers as they are more readily adsorbed by clay minerals, normally through cation exchange processes. The ions replace those already present on the clay surface (usually sodium and calcium). In most cases negatively charged ions (anions) are not affected to any great extent by the medium through which they pass. The capacity of clay minerals to hold anions has, however, been shown to increase with decreasing pH. Precipitation reactions within soils are another means by which anion movement is restricted (Ellis, 1973). The process of ion exchange and the relative restriction of ion movement in soils is discussed in more detail in Chapter 2.

One advantage of ion tracers is that they do not decompose and are therefore not lost to the system. Background concentrations can, however, be high and large concentrated injections may be required. This can result in density separation and gravity segregation during a tracing experiment, altering flow patterns and the degree of ion exchange and secondary precipitation which takes place (Davis et al, 1984). One way of overcoming this is to raise the temperature of the injection mixture to a level above that of the receiving water (Grisak et al, 1979). Alternatively, the tracer may be introduced in powder form and allowed to dissolve in the injection borehole/stream.

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(i) Chloride

Chloride is a conservative tracer and is only weakly adsorbed onto soil surfaces. It's use as a tracer is complicated because of the high background concentrations often encountered. Large injection levels may be required to obtain a detectable increase in ion concentration. This can lead to significant density effects and can alter the permeability of soils by ion exchange (Davis et al, 1984). Injection concentrations of greater than 3000 mg/l are not recommended.

(iii) Lithium and Bromide

Lithium Bromide was successfully used to trace the connection between a landfill site and a spring by Murray et al (1980). It was chosen over conventional tracers for the following reasons:

(a) Two measurable parameters were included both of which are relatively insensitive to sub - surface removal
(b) Low background levels were found in the sampling spring making detection possible even under high levels of dilution
(c) Sensitive analytical procedures are well established
(d) Lithium bromide is non - toxic even under conditions of moderate dilution with potable water.

(Murray et al, 1981)

Tracing with rhodamine WT proved unsuccessful in the same study. There are some conflicting reports on the relative adsorption loss of lithium. Murray et al (1981) and Smith and Davis (1974) reported little loss while Davis et al (1984) reported that lithium losses were ' high due to ion exchange '.

The use of bromide as an ionic tracer has many advantages. It has low background levels in groundwater (generally less than 1 mg/l) and it is not significantly affected by precipitation or absorption. Smith and Davis (1974) report that the rate of movement of bromide through soils to groundwater is similiar to that of nitrate. The slight difference in the movement of the two anions recorded in the study was attributed to microbiological activity involving nitrate. This was confirmed by repeating the experiment using sterile soils. The report concluded that bromide can be used to follow the movement of nitrate through soils.

(iii) Potassium

Potassium, as with most other cations, has a relatively high adsorption rate in soils and is also lost by ion exchange thereby reducing it's value as a tracer. It's

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advantages include simplicity of analysis and generally low background levels (Davis et al, 1984).

Ellis (1980) evaluated the usefulness of potassium as a tracer of the migration of municipal landfill leachate to surface and groundwaters. He concluded that potassium successfully indicated leachate pollution of groundwater and that it could serve as a simple parameter for assessing the movement of leachate in groundwater. It is particularly useful in preliminary investigations for determining the direction of movement of the leachate and the extent of dilution by receiving waters.

4:2.3.3 Microbiological

The contamination of groundwater supplies by microbial rich wastes with attendant health hazards is well documented in the literature. The ability to trace the movement of this waste to groundwater is essential in assessing the potential transmission of disease causing organisms (Keswick et al, 1982). It has been established that chemical tracers do not always reflect the movement of microorganisms in soils or groundwater systems (Rahe et al, 1978 and Bitton and Gerba, 1984). Techniques are therefore needed to predict the movement of pollutants from suspect sources using highly distinctive tracer organisms which can be easily identified in the groundwater system. As tracers microorganisms have the advantage of being non - mutagenic, non - toxic and having a finite lifetime (Keswick et al, 1982 and Bitton and Gerba, 1984). In addition, they can be easily obtained in large numbers in a relatively small volume yet they can be detected in high dilutions. Three types of microbial tracers are commonly used:

(i) Yeasts
(ii) Bacteria
(iii) Viruses

(i) Yeasts

A yeast is a single celled organism, ovoid in shape and with a diameter of 4 to 10 μm. Bakers' yeast (*Saccharomyces cerevisiae*) is a non - pathogenic species which has been most commonly used (Bitton and Gerba, 1984 and Davis et al, 1984). Wood and Ehrlich (1978) reported that yeast cells penetrated 7.0 metres into a sand and gravel overburden less than 48 hours after injection. They found that lateral movement of the cells occurred up to 7.6 metres downgradient of the injection point. In later experiments they showed that, although initial breakthrough times for all three tracers were similar, the peak in breakthrough concentrations for bromide and iodide lagged significantly behind that of yeast. This anomaly was attributed to the fact that the largest soil pores which carry the particles are also the paths of highest velocities. The

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yeast cells travelled through solution openings but were filtered out in intergranular pores while the movement of the conservative tracers occurred simultaneously through both solution openings and intergranular pores.

The survival of yeast cells in the environment is largely dependent on the availability of nutrients (Keswick et al, 1982). Thus the use of yeast in long term, long distance tracing experiments is not advisable. Another limitation is that it's relatively large size renders it susceptible to filtration losses in soils and fine grained aquifers. The advantages include the ease and low cost of sampling and analysis.

(ii) Bacteria

Bacteria are the most commonly used microbial tracers (Keswick, 1982 and Bitton and Gerba, 1984). Species which have been used successfully include *Escherichia coli, Streptococcus faecalis, Bacillus stearothermophilus, Serratia marcescens and Serratia indica*. The organisms range in size from 1 to 10 µm (Davis et al, 1984), although the majority of species are less than 2 µm.

Fecal bacteria have been successfully used in tracing wastewater flow from pit latrines (Caldwell, 1938) and septic tank disposal systems (McCoy and Hagedorn, 1979). However, some difficulty can arise in differentiating between those derived from the suspected source and those from other sources (Bitton and Gerba, 1984 and Keswick et al, 1982). In addition many of these organisms are pathogenic to man. Although the enumeration of Coliform and fecal Coliform bacteria, fecal Streptococci and *Clostridium perfringens* may give some indication of the proximity and size of the contamination source, a positive identification of a particular source of fecal material cannot normally be made. Techniques are needed to permit the ' labelling ' of suspected sources of fecal contamination with highly distinctive ' tracer ' organisms which can be easily identified and counted if they enter the groundwater system (Sinton, 1980). This labelling process may be in the form of antibiotic resistance or the ability to produce a distinctive end product. Alternatively, a tracer organism not usually found in sewage may be used (Rahe et al, 1978; Sinton, 1980; Bitton and Gerba, 1984 and Davis et al, 1984). Sinton (1980) suggests *Serratia marcescens*, although the species is commonly isolated from animal intestines (Krieg and Holt, 1984). The identification of *Serratia marcescens* is simplified by the fact that the organism produces highly coloured colonies which are easily distinguished on agar plates (Rippon, 1963 and Bitton and Gerba, 1984). It's use, however, tends to be hampered by the growth of other aerobic species especially in sewage polluted waters (Sinton, 1980). This problem can be partially solved by the use of an antibiotic resistant mutant strain (Rippon, 1963 and Wimpenny et al, 1972). The use of *Serratia marcescens* as a tracer of estuarine water movement was investigated by

Rippon (1963). It was concluded that the organism compared favourably with dyes and radioisotopes as far as cost and sensitivity is concerned. There was little difference in the time taken for analysis. Wimpenny (1972) reported that *Serratia marcescens* was well suited to river water tracing and the use of an antibiotic resistant strain greatly improved identification by significantly reducing the growth of indigenous organisms. However, a significant drop in recovery was also noted on the antibiotic - containing media which made quantitative measurements difficult. *Serratia indica* was also successfully used to trace sewage pollution at sea in Norway (Ormerod, 1964). However, as it is now recognised that Serratia species are potential pathogens, their use as water tracers is not recommended (Davis et al, 1970; Keswick et al, 1982; Bitton and Gerba, 1984 and Davis et al, 1984).

The use of antibiotic resistant strains of *E. coli* and *Streptococcus faecalis* to trace subsurface soil water flow in western Oregon soils was successful for Hagedorn et al (1978). The organisms travelled 500 centimetres in a septic tank drainfield under saturated conditions over a 32 day sampling period. It has been suggested that these organisms provide an excellent microbiological method for assessing the suitability of different soil types for septic tank drainfields. Similar results were reported by Rahe et al (1978). He found that an antibiotic resistant strain of *E. coli* was a suitable soil - water tracer having survived for periods of at least 96 hours. However, when markers like antibiotic resistance are used it is sometimes difficult to distinguish between the natural or introduced organisms because of interaction with natural populations. Another concern is that antibiotic resistance can be transferred to human pathogens, especially if the water is to be consumed. The likelihood of this can be greatly reduced by using bacteria which do not carry the genetic information for antibiotic sensitivity on plasmids (Keswick et al, 1982; Bitton and Gerba, 1984 and Davis et al, 1984). Rahe et al (1979) raised serious doubts about the use of dyes as groundwater tracers for microorganisms. They reported the results of a tracing experiment using sodium fluorescein and *E. coli* in western Oregon soils where only *E. coli* was detected in the soil downgradient of the point of introduction.

A series of tracer experiments using different bacteria was conducted by Sinton (1980) at Canterbury, New Zealand. Two species were used, *Bacillus stearothermophilus* and a H_2S producing strain of *Escherichia coli*. Both species travelled over 920 metres in the underlying groundwater system. However, *B. stearothermophilus* exhibited a better recovery rate which was attributed to it's superior survival characteristics. The bacteria was, however, found to be naturally occurring in Canterbury soil/groundwater systems and the numbers present tended to increase following rainfall events thereby limiting it's use to periods of low rainfall.

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Neither *B. stearothermophilus* or *E. coli* (H_2S^+) were considered suitable for use as tracers in sewage polluted waters as both occurred naturally in sewage (Sinton, 1980). The lateral movement of bacteria through shallow groundwater in Alaska was investigated using *Serratia marcescens, Bacillus globigii and Chromobacterium violaceum*. None of the organisms were satisfactory due to an inability to distinguish them from those naturally present in the soil (Bitton and Gerba, 1984).

Some disadvantages of using bacteria as tracers are:

(a) The organisms are capable of regrowth in the environment thus producing erroneous results

(b) Bacteria are susceptible to loss by filtration and adsorption in certain soil types

(c) The movement of bacteria does not necessarily reflect the movement of other microorganisms e. g. viruses.

Advantages of using bacteria include:

(a) They are easy to grow in large numbers and easy to assay

(b) If markers such as anti - biotic resistance are used they are usually distinguishable from other flora likely to be present in the groundwater or soil.

(Keswick et al, 1982; Bitton and Gerba, 1984 and Davis et al, 1984)

(iii) Viruses

Animal, plant and bacterial viruses have all been used as groundwater tracers (Davis et al, 1984). Viruses are generally much smaller than bacteria, ranging in size from 0.2 to 1.0 µm. The use of human enteric viruses is not recommended because of the danger of disease outbreaks (Bitton and Gerba, 1984). To overcome this problem vaccine strains of polio virus 1 have been used as a water tracer but even these are capable of causing disease (Keswick et al, 1982). Most animal strains are known not to infect man and bovine enterovirus has been successfully used to trace the movement of viruses from septic tank drainfields (Bitton and Gerba, 1984 and Keswick et al, 1982).

The use of bacterial viruses (or bacteriophage) as tracers was first evaluated by Wimpenny et al (1972). A lamda - like phage of *E . coli*, K 12, was found to be an excellent tracer of water movement, particularly in polluted rivers. The following reasons were given for the organism's success as a tracer:

(a) Detection is easy, sensitive and extremely rapid

(b) The phage is highly selective for it's host organism

(c) No problems with, or interference from, high levels of naturally occurring river bacteria were encountered

(d) The phage used did not decrease in viability over a one week laboratory survival experiment

(e) Phages are oblivious to all organisms other than their hosts thus there are no public health problems associated with their use.

(Wimpenny et al, 1972)

Similarly, in a recent study on the use of bacteriophages as groundwater tracers, the principle advantages were reported as:

(a) Greater safety when compared to bacterial tracers

(b) Potential for simulating the possible movement of enteric viruses in the aquatic environments

(c) The possibility of freezing the samples for assay at a later date.

(Sinton and Ching, 1987)

Hilton and Stotzky (1973) investigated the feasibility of using coliphages (phages of coliform bacteria) as indicators of sewage pollution. They found no relationship between coliform and coliphage levels in polluted waters and concluded that coliform bacteria were not suitable for tracing virus movement in polluted waters. Martin and Thomas (1974) employed type 2 phage of *Aeromonas aerogenes* 243 to follow the movement of shallow groundwater in South Wales over a distance of 680 metres. Ten litres of phage suspension were introduced through shallow piezometers into the sandstone strata. Phage were detected in the monitoring boreholes at a distance of at least 680 metres for nine days after tracer addition. They concluded that the phage had good potential as a groundwater tracer in small scale investigations especially where non polluting tracers are required. The use of phage 80 of *Staphylococcus aureus* appears to have considerable potential as a sewage tracer but coliphage MWD 1 of *E. coli* was unsuccessful in sewage polluted waters and may be best suited to tracing secondary treated effluents (Sinton and Ching, 1987).

The methods for carrying out a phage trace are well established. In recent years research on procedures for the concentration (Hilton and Stotzky, 1973; Primrose and Day, 1977; Seeley and Primrose, 1979; Goyal et al, 1980 and Farrah and Preston, 1985) and enumeration (Kennedy et al, 1985) of bacteriophage from water samples has been undertaken, resulting in the development of highly refined and accurate techniques.

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4:3 Site Characteristics

Three test sites in the county Sligo area were used for the tracing experiments. Sites 1 and 2 are located on the Knocknarea peninsula south - west of Sligo town and Site 3 in the townland of Cregg, to the north of Sligo bay. Two control stations, C1 and C2, were also included in this study. C1, positioned near the centre of the Knocknarea peninsula, was used as a background reference for sites 1 and 2. C2, used as a control for Site 3, is situated approximately 1 kilometre north of the site in Cregg. The grid references for these sites are given in Table 4.3.1 (p249). These sites were used in an earlier investigation on the impact of septic tank systems on groundwater quality (Chapter 3) and further information on site characteristics including location maps, soil type, geology, hydrogeology and the construction/ maintenance of the septic tank treatment systems and monitoring boreholes is presented there.

Each site consisted of a two to five year old dwelling with a septic tank and soil treatment system serving a population of at least four people. The three sites were chosen because of differences in the thickness and nature of the unsaturated zone available for effluent attenuation.

(i) Site 1

Site 1 is located in an area with a high water table (0.8 to 2.0 metres below ground level) and a thin cover of sandy loam soil overlying highly fissured and weathered limestone bedrock. The effluent receives no effective attenuation as it is discharged to a 2.0 metres deep soakage pit and therefore enters groundwater directly.

(ii) Site 2

Site 2 is located in an area with 6.0 to 6.5 metres of permeable sands and gravels overlying moderately fissured limestone. The water table normally fluctuates between 4.7 and 5.6 metres below ground level. The effluent is discharged into a soakage pit 2.0 metres deep and therefore percolates through 2.7 to 3.6 metres (depending on the watertable level) of unsaturated permeable overburden before entering groundwater.

(iii) Site 3

Site 3 is located in an area with 2.5 metres of sandy loam soil overlying 3.5 to 4.0 metres of permeable gravels and cobbles. The underlying bedrock consists of highly fissured and weathered gneiss. Water table levels vary between 3.0 and 4.9 metres below ground level. The effluent is discharged

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into a series of 1.0 metre deep distribution trenches and as such percolates through a 1.5 metres thickness of heavy boulder clay and 0.5 to 2.4 metres of permeable gravels (depending on the watertable level) before entering groundwater.

A total of eight groundwater monitoring boreholes were installed at the sites by the Geological Survey of Ireland (G. S. I.). Two monitoring boreholes were installed at site 1 (B1 and B2), 2.0 and 8.1 metres downgradient of the septic tank soakage pit. At site 2 three boreholes (B3, B4 and B5) were installed 3.85, 4.90 and 10.22 metres downgradient of the soakage pit. A further three boreholes (B6, B7, and B8) were sunk at site 3, 1.5, 1.5, and 9.77 metres downgradient of the percolation field. The exact location of the monitoring boreholes was dictated by the mobility of the drilling rig under the soft site conditions at the time of drilling.

The location of the monitoring boreholes in relation to the septic tank treatment systems is presented diagramatically in Chapter 3 (Figures 3.4.2 to 3.4.4. p151 to 152). In addition detailed information on the soil type and geology in the vicinity the bores is given in section 3:4 (Chapter 3).

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Table 4.3.1

Table Showing the Townlands and Grid References of the Sampling Sites and Control Stations

Site/Station	Townland	Grid Reference
Site 1	Kilmacowen	G 662 307
Site 2	Knocknahur	G 646 333
Site 3	Cregg	G 653 395
C1	Tobernaveen	G 662 342
C2	Cregg	G 653 340

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4:4 Materials and Methods

The tracer materials used in this study can be broadly classified into two groups, chemical and microbiological (Figure 4:4.1, p251). Information on the experimental procedures, including sampling and analytical methods used for each tracer material, is presented in this section.

4:4.1 Optical Brightener

The three sites were traced using leucophor PBS optical brightener between 16/1/89 and 21/1/89. Absorbent cotton wool was used as a detector. Before the detectors were assembled the materials used were checked for background fluorescence under an Ultra Violet hood. The wool was then tied in nylon stockings, weighted and attached to a length of string. The detectors were lowered down into each of the monitoring boreholes and the control spring C1, and fixed at a depth of 1.0 metre beneath the static water level (S. W. L.) by securing the string to the well head. An initial set was placed in the boreholes (16/1/89) to monitor the groundwaters for background fluorescence. After 72 hours they were removed and placed in separate plastic bags for transport to the laboratory.

Fresh cotton wool detectors were then placed in the boreholes and secured as described above. This was carried out before the optical brightener was handled to reduce the risk of contamination and false positive results. A 3.0 litre aliquot of leucophor PBS was then added to the septic tank inlet manhole at each site. The dye was washed into the tank by flushing clean tap water through the manhole. After a further 72 hrs (21/1/89) the second set of detectors were removed and transported to the laboratory for inspection. All detectors were examined for fluorescence under an Ultra Violet hood.

4:4.2 Fluorescent Dyes (Sodium Fluorescein and Rhodamine B)

Sites 1, 2, and 3 were traced using sodium fluorescein and rhodamine B between 7/2/89 and 14/2/89. These two dyes were selected as being generally representative of all the fluorescent tracer dyes. It had been intended to also use rhodamine WT and lissamine FF but these dyes were not available at the time of tracing.

Boreholes 1, 2, 3, 5, 6, 7 and 8 were used as monitoring boreholes, with spring C1 again being used as the test control. Because of problems encountered with the well casing in B4 it was not possible to obtain a sample and this borehole was not used in any further tracing experiments.

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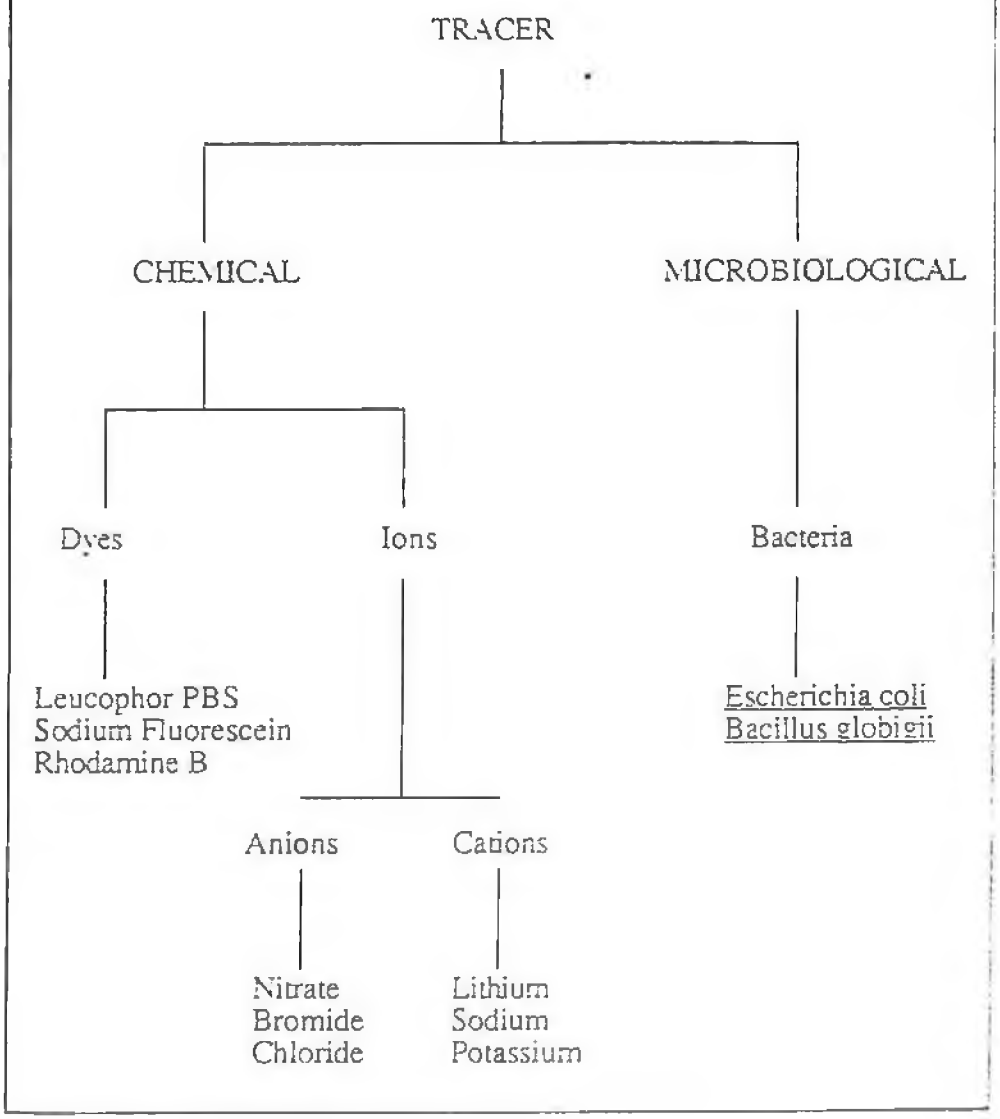
Figure 4.4.1

Tracer Materials Used in the Study



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The volumes of the three septic tanks were determined and the weight of dye powder required to give a known effluent dye concentration was then calculated (Table 4.4.1, p253). This was determined by taking into account the potential dye loss in the unsaturated zone, the sensitivity of the analytical equipment used and the reported toxicity of the material. Smart and Laidlaw (1977) was used as a reference for the expected photochemical and chemical decay of the dyes. The loss of dye due to adsorption onto organic matter and interaction with other chemicals in the septic tank was not taken into account as this would be extremely difficult to quantify because of the range and variability of chemicals and organic materials present in septic tanks.

Groundwater samples were taken from all the monitoring boreholes and the control spring prior to the addition of the tracer to determine the background dye concentrations. The tracer dye powder was then dissolved in one litre of clean tap water, mixed well, added to the septic tank influent manhole and flushed into the tank with tap water. Samples were taken directly from the boreholes at eight hourly intervals after addition of the tracer using a 0.5 litre hand bailer. Cross contamination between the three sites was avoided by using three separate bailers. In order to obtain a representative groundwater sample, two well volumes were removed from each monitoring well prior to sampling. After each sample was removed the bailer was carefully washed with 80% ethanol and rinsed thoroughly with clean water before proceeding to the next borehole. All samples were transferred directly from the bailer to clean acid washed glass universal bottles and were immediately placed in a dark container for transport to the laboratory. The sampling interval was increased after the tracer dye had visibly appeared in the groundwater samples.

The excitation and emission maxima were calculated for each dye by running a scan of a standard dye solution on a spectrofluorimeter. Absorption values for prepared standards and the groundwater samples were then measured on the instrument.

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Table 4.4.1

The Quantities of Dye Tracer Added to the Septic Tanks and the Resulting Effluent Concentrations

SITE	*Expected concentration in effluent (mg/l)			Quantity (g) added
1	Sodium fluorescein	=	50	100
	Rhodamine B	=	10	20
2	Sodium fluorescein	=	75	150
	Rhodamine B	=	20	40
3	Sodium fluorescein	=	75	150
	Rhodamine B	=	20	40

Table 4.4.2

The Quantities of Ion Tracers Added to the Septic Tanks and the Resulting Effluent Concentrations

SITE	*Expected concentration in the effluent (mg/l)			Quantity (g) added
1	Sodium	=	100	500 (as NaCl)
	Potassium	=	58	300 (as KNO_3)
	Nitrate	=	92	100 (as LiBr)
	Chloride	=	150	
	Bromide	=	41	
2	Sodium	=	100	500 (as NaCl)
	Potassium	=	77	400 (as KNO_3)
	Nitrate	=	122	200 (as LiBr)
	Chloride	=	150	
	Bromide	=	82	
3	Sodium	=	100	500 (as NaCl)
	Potassium	=	77	400 (as KNO_3)
	Nitrate	=	122	200 (as LiBr)
	Chloride	=	150	
	Bromide	=	82	

* Assuming no loss by adsorption or chemical alteration within the septic tank

4:4.3 Ions

Sites 1, 2 and 3 were traced with the cations sodium, potassium and lithium and the anions chloride, nitrate and bromide between 29/3/89 and 5/4/89. The electrical conductivity of all samples was also measured as an indication of general tracer movement. Boreholes 1, 2, 3, 5, 6, 7 and 8 were used as monitoring boreholes. The quantity of tracer added was then determined as described in 4:4.2. Table 4.4.2 (p253) shows the quantities of tracer added to the systems and the expected effluent concentration. Effluent concentrations were estimated by taking into account the possible losses within the unsaturated zone, the sensitivity of the analytical detection procedures, the toxicity of the tracer material and the expected background concentration of the ion.

The required quantities of reagent grade sodium chloride (NaCl), potassium nitrate (KNO_3) and lithium bromide (LiBr) were partially dissolved in distilled water before addition to the septic tanks. The tracer mixtures were transported to each site in acid - washed polypropylene containers and were flushed into the tanks with clean tap water.

An initial set of groundwater samples was taken from the monitoring boreholes and the control spring in order to establish the background concentrations of the ions. All samples were taken directly from the monitoring boreholes using a 0.5 litre volume bailer. A representative groundwater sample was obtained by removing two well volumes from the boreholes before taking a sample. One litre samples were then removed from each monitoring borehole and transferred directly to clean acid washed polypropylene bottles. Contamination between sites and boreholes was avoided using the same procedure adopted in the dye tracing experiments (4:4.2). Samples were taken at eight hourly intervals after tracer addition. This interval was increased when the tracer breakthrough peak had been observed.

On return to the laboratory electrical conductivity readings were taken of all samples. They were then frozen for future analyses. When the tracing experiment was complete the samples were removed from the freezer, allowed to thaw and brought to room temperature. A 200 ml portion of the sample was filtered through a 0.47 µm filter and stored in clean polypropylene bottles. Analysis for Na^+, K^+, NO_3^-, Cl^- and Br^- was carried out on the filtrate. Because of problems encountered with analytical equipment (the photometric filter for lithium could not be obtained) the lithium ion was not analysed for. Table 4.4.3 (p255) summarises the analytical methods used in the detection of the ion tracers .

Table 4.4.3

Summary of the Analytical Procedures Used in the Detection of the Ion Tracers

Parameter	Method
Conductivity (µS/cm)	Electrometric
Sodium/Potassium (mg/l)	Flame Photometry
Chloride (mg/l)	Titrimetric
Nitrate (mg/l)	Ultraviolet Spectrophotometry
Bromide (mg/l)	Visible Spectrophotometry

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4:4.4 Microbiological

The three test sites were traced using two bacterial species, an antibiotic resistant strain of *Escherichia coli* (resistant to 200 µg/ml streptomycin) and endospores of *Bacillus globigii*. The traces were carried out between 7/5/89 and 23/5/89. Boreholes 1, 2, 3, 5, 6, 7 and 8 were sampled regularly after tracer addition. The spring C1 was sampled on one occasion as a test control.

In order to assess the growth characteristics of the *Escherichia coli* bacterial tracer a growth curve and plot of bacterial numbers against optical density was carried out on each stock culture prior to the tracing tests. This curve was then used to estimate the fermentation time required. Two New Brunswick Multigen bacterial fermenters (Plate 4.4.1, p260) were used to culture the bacteria. Brain Heart Infusion broth (B. H. I. Oxoid) was the culture media used. Single colonies of the *E. coli* stock culture were innoculated into 100 mls of the broth and incubated overnight at 37 °C. The two fermenters were then set at 37 °C and allowed to reach the required temperature before innoculation. A 20 ml portion of the overnight culture was then added aseptically to the fermenters, giving a 2% startup culture. The liquid cultures were constantly aerated and agitated and were fermented to a concentration of approximately 1 x 10^8 cells/ ml. This concentration was estimated by measuring the optical density of the growing culture. On completion of the fermentation, which lasted for four hours, an accurate count was carried out using a standard serial dilution/plate count procedure with Tryptone Soya Agar (T. S. A. Oxoid) as the growth media.

The *Bacillus globigii* culture was fermented at the laboratory of Biocon Ltd.. The fermentation process is an industrial secret but the technique resulted in a culture concentration of approximately 1 x 10^{10} cells/ml. A high percentage of the organisms (> 95%) were in spore form. Accurate counts of the numbers of organisms present in the culture were taken using a standard serial dilution/plate count procedure with Tryptone Soya Agar (T. S. A. Oxoid) as the growth media.

It was not considered necessary to separate the bacterial cells from the nutrient rich growth culture media prior to addition to the tracer system (although this has been recommended by Sinton, 1980) because the cultures were being added to a septic tank already rich in nutrients. The liquid cultures were transferred to sterile one litre glass bottles for transport to the test sites. Background bacterial counts were determined in all monitoring boreholes before the tracer was added to the septic tank systems. The tracers were then introduced into the system through the inlet manhole and washed into the tank with clean tap water.

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Samples were taken from the monitoring boreholes using the 0.5 litre bailers described above. A separate bailer was used at each of the sites traced. Sterilisation of the bailers between the monitoring boreholes was achieved by thorough washing with 80% ethanol and rinsing with sterile water. The sample was transferred directly from the bailer to sterilised 250 millilitre glass sampling bottles (sterilised by autoclaving at 121 °C / 15 p. s. i. for 15 minutes). A representative water sample was ensured by removing two well volumes from each monitoring well prior to sampling. Samples were taken at 12 hourly intervals although this interval was increased once the tracer breakthrough peak had been observed. All the samples were placed in a cool box and transported immediately to the laboratory for analysis.

The samples were analysed within six hours of removal from the monitoring boreholes. The water samples were analysed using the membrane filtration system given in A. P. H. A. Standard Methods (Anon., 1985). This method is recommended by Sinton (1980) because ' it greatly increases the sensitivity of detection over plate count methods '. Millipore membrane filtration apparati and 0.47 µm membrane filters were used. The filtration apparati were sterilised between samples by autoclaving at 121 °C / 15 p. s. i. for 15 minutes. A 50 ml aliquot of the sample (diluted if necessary) was transferred to the filtration apparatus using sterile glass, cotton wool - stoppered pipettes (heat sterilised in an oven at 160 °C for 2 hours). After filtration the membrane filter was transferred to selective media and incubated at the required temperature for an appropiate period. Bacterial counts were taken after incubation and recorded as colony forming units (c. f. u. 's) per 100 mls of sample (Plates 4.4.2 and 4.4.3, p261). On a number of occasions further confirmatory tests were carried out on the organisms isolated. The analytical procedures used in the isolation and identification of the bacterial tracers are detailed in Figures 4.4.2 (p258) and 4.4.3 (p259).

Figure 4.4.2

Isolation and Identification of The Escherichia coli Bacterial Tracer

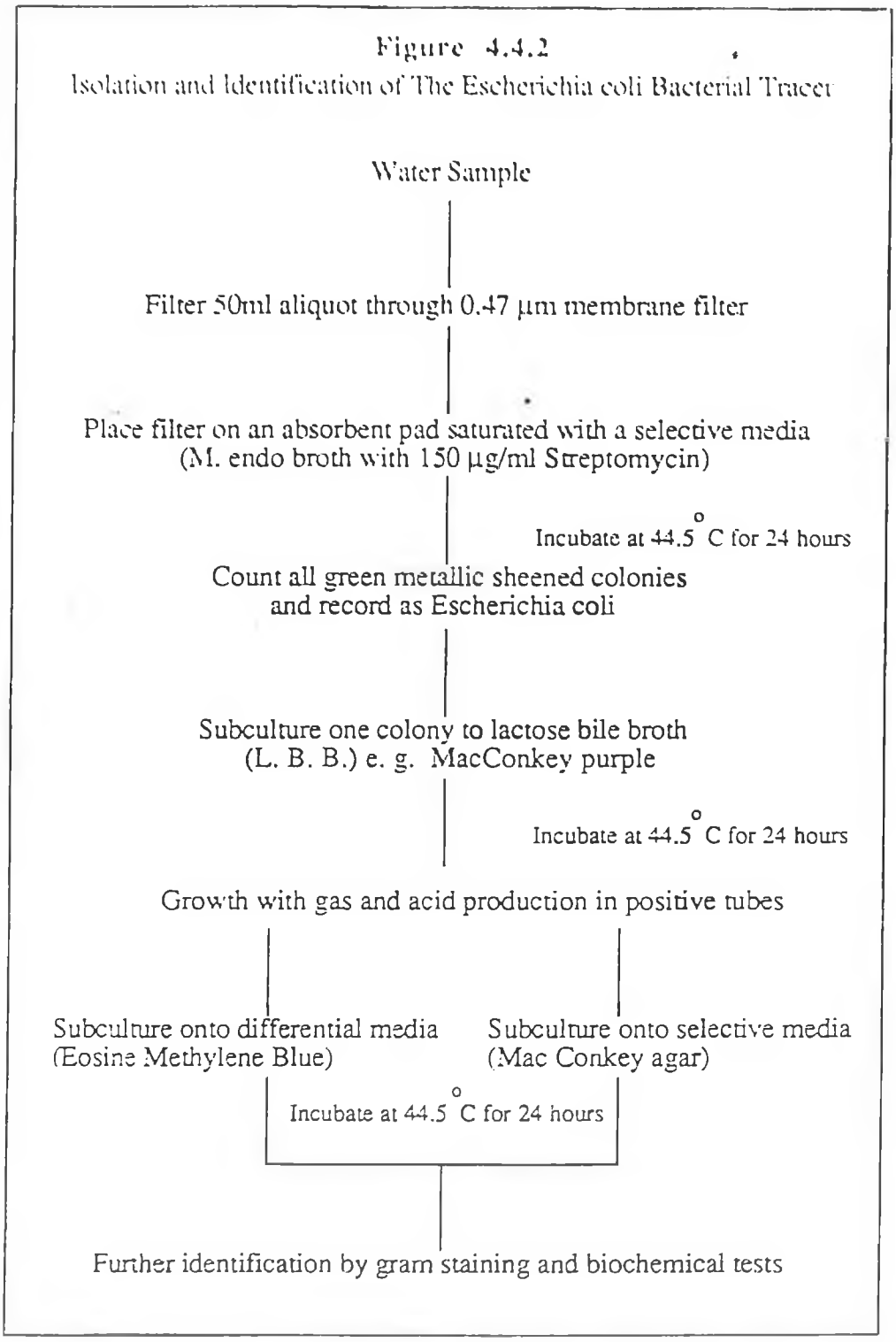

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Figure 4.4.3

Isolation and Identification of the Bacillus globigii Bacterial Tracer

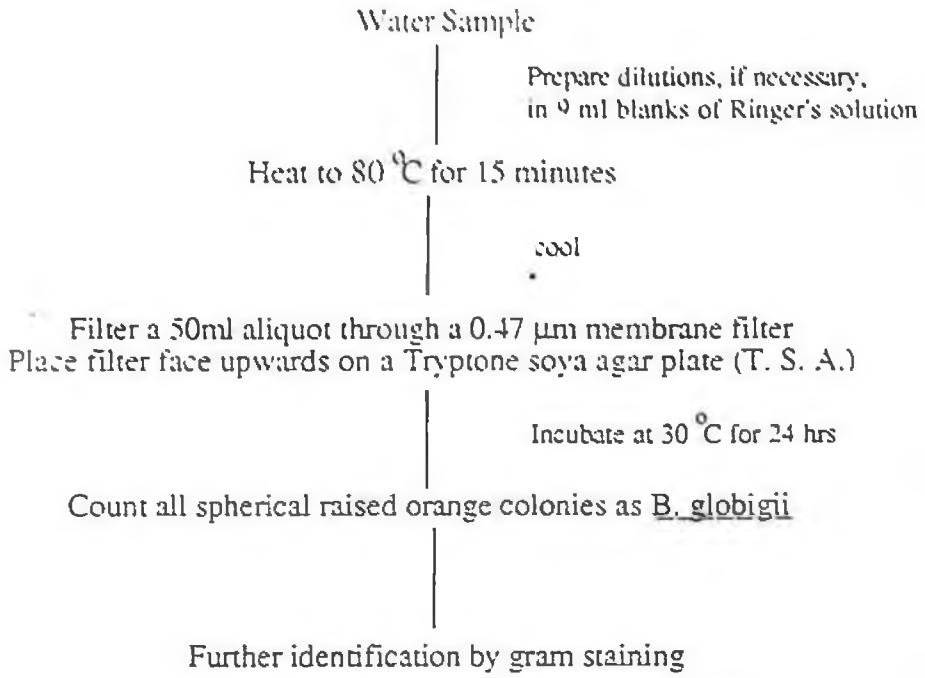

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Plate 4.4.1

The Fermentation of the *Escherichia coli* Bacterial tracer

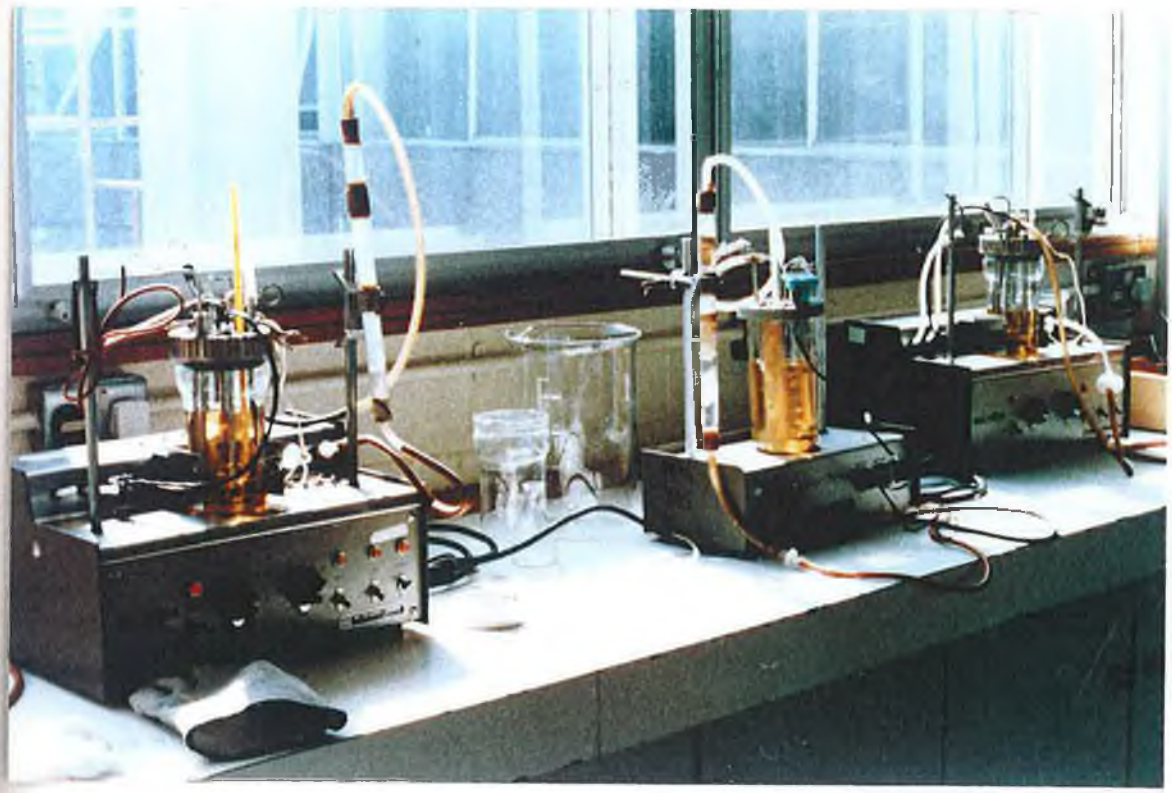

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Plate 4.4.2

Escherichia coli Tracer Bacterial Colonies Isolated on M. endo Media

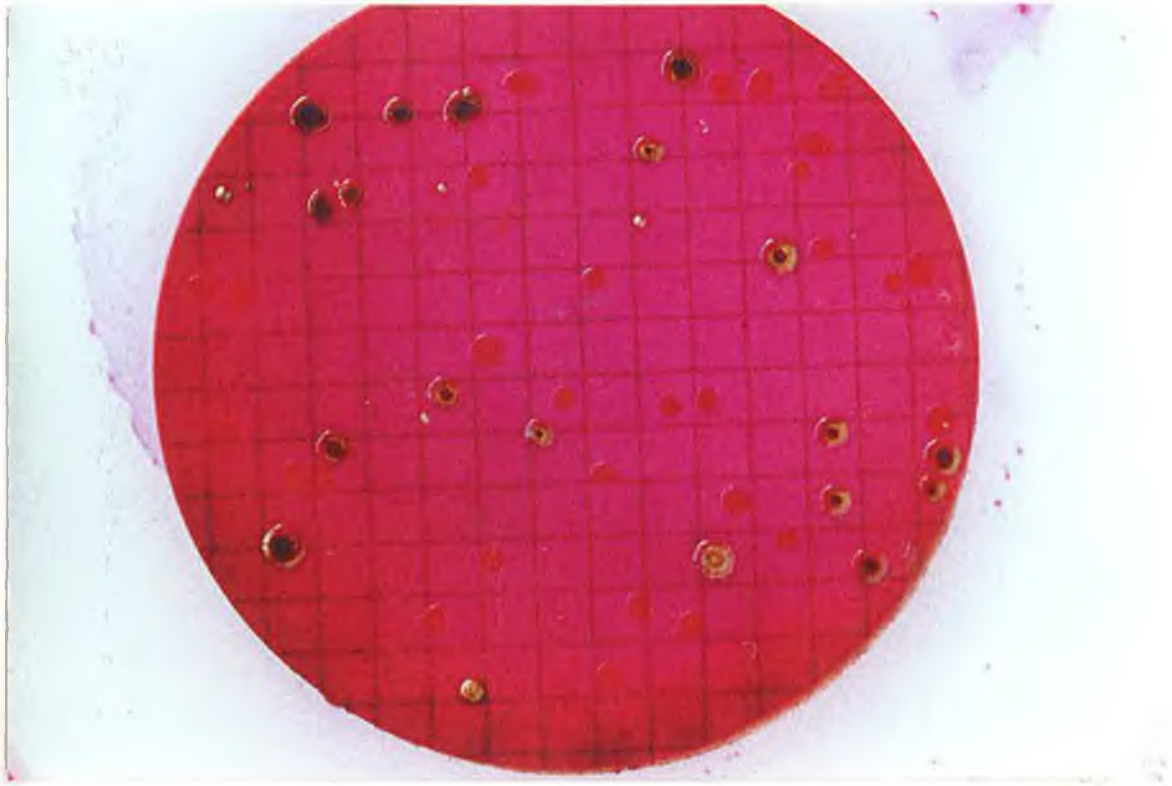
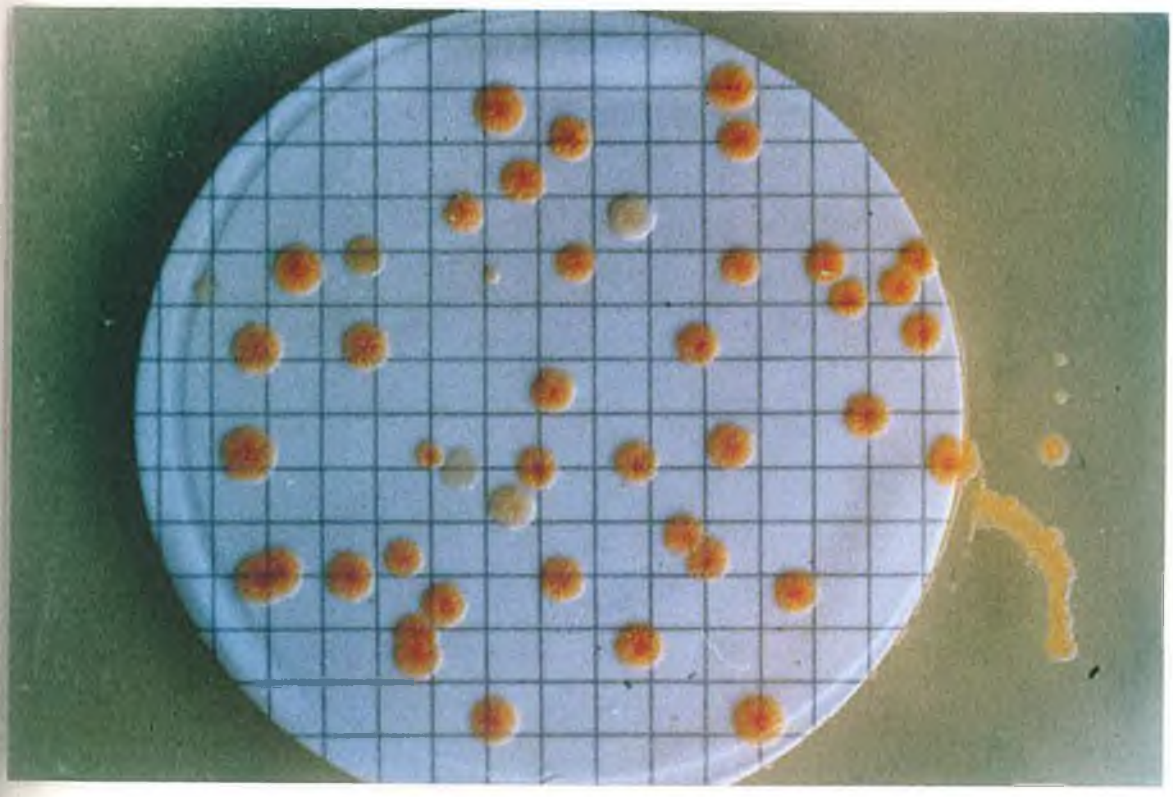


Plate 4.4.3

Bacillus globigii Tracer Bacterial Colonies Isolated on T. S. A. Media



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4:5 Results

4.5.1 Presentation of Results

The main results of the investigation are presented in this section as follows:

(i) Optical brightener trace

The results of the Lecuophor PBS optical brightener trace are shown in Table 4.5.1 (p264)

(ii) Fluorescent dye, ion and bacterial traces

The results of the fluorescent dye, ionic and bacterial traces are presented graphically in Figures 4.5.1 to 4.5.28 (p265 to 274). The figures, referred to as breakthrough curves, are presented as follows:

Figures 4.5.1 to 4.5.3	Sodium fluorescein
Figures 4.5.4 to 4.5.5	Rhodamine B
Figures 4.5.6 to 4.5.8	Electrical Conductivity
Figures 4.5.9 to 4.5.11	Nitrate
Figures 4.5.12 to 4.5.14	Bromide
Figures 4.5.15 to 4.5.17	Chloride
Figures 4.5.18 to 4.5.20	Sodium
Figures 4.5.21 to 4.5.23	Potassium
Figures 4.5.24 to 4.5.25	*Escherichia coli*
Figures 4.5.26 to 4.5.28	*Bacillus globigii*

Comparisons of the dye, ion and bacterial tracers are illustrated in Figures 4.5.29 to 4.5.50 (p275 to 282) as follows:

Figures 4.5.29 to 4.5.30 Sodium fluorescein v Rhodamine B
Figures 4.5.31 to 4.5.33 Sodium fluorescein v Nitrate
Figures 4.5.34 to 4.5.36 Sodium fluorescein v Bromide
Figures 4.5.37 to 4.5.39 Sodium fluorescein v *Bacillus globigii*
Figures 4.5.40 to 4.5.42 Nitrate v Bromide
Figures 4.5.43 to 4.5.45 Nitrate v *Bacillus globigii*
Figures 4.5.46 to 4.5.48 Bromide v *Bacillus globigii*
Figures 4.5.49 to 4.5.50 *Bacillus globigii* v *Escherichia coli*

(iii) Information on the rainfall recorded in the Sligo during each of the tracer is presented in Figures 4.5.51 to 4.5.54 (p283 and 284)

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(iv) The maximum recoverability ratio i. e. the highest ratio of the concentration of tracer material recovered to tracer material added (C/Co max) was calculated for each of the tracers at all of the monitoring boreholes. The results are presented in Table 4.5.2 (p285).

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Table 4.5.1

The Results of the Leucophor PBS Optical Brightener Trace

Sampling Station / Detector Fluorescence	Sample								
	B1	B2	B3	B4	B5	B6	B7	B8	C1
Degree of detector fluorescence before tracer addition	++	++	++	+	+	-	-	-	-
Degree of detector fluorescence after tracer addition	+++	+++	++	+	+	-	-	-	-

Note:

+++ = Very positive fluorescence
++ = Positive fluorescence
+ = Traces of fluorescence
- = No fluorescence

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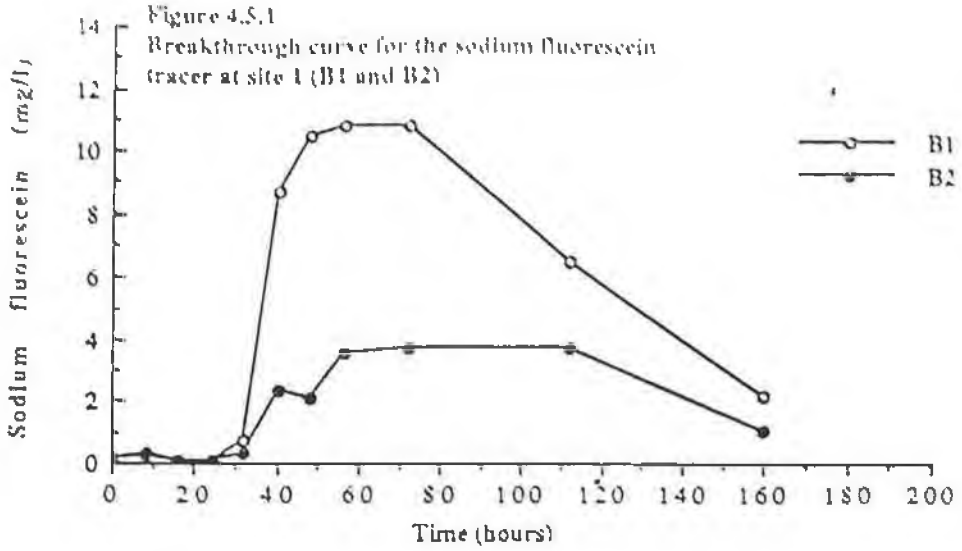

Figure 4.5.1
Breakthrough curve for the sodium fluorescein tracer at site 1 (B1 and B2)

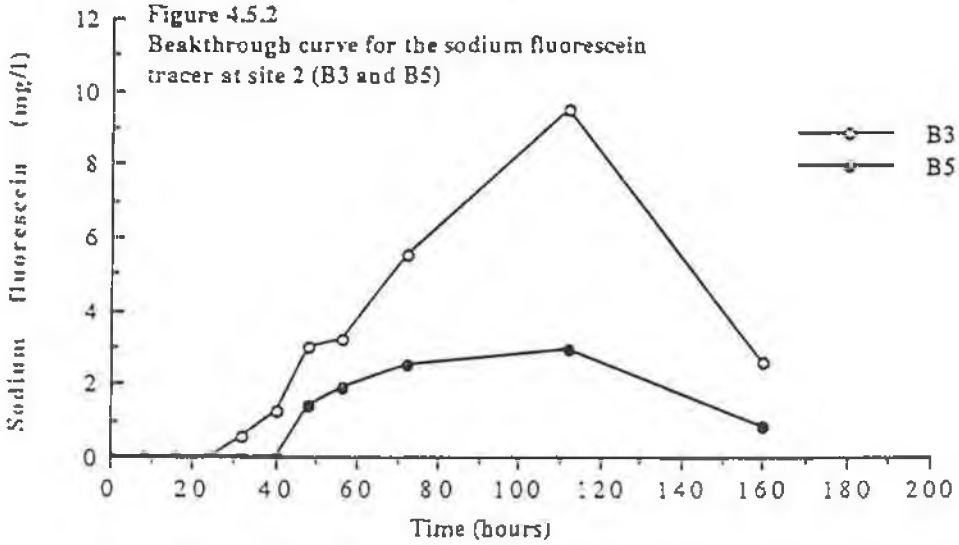

Figure 4.5.2
Beakthrough curve for the sodium fluorescein tracer at site 2 (B3 and B5)

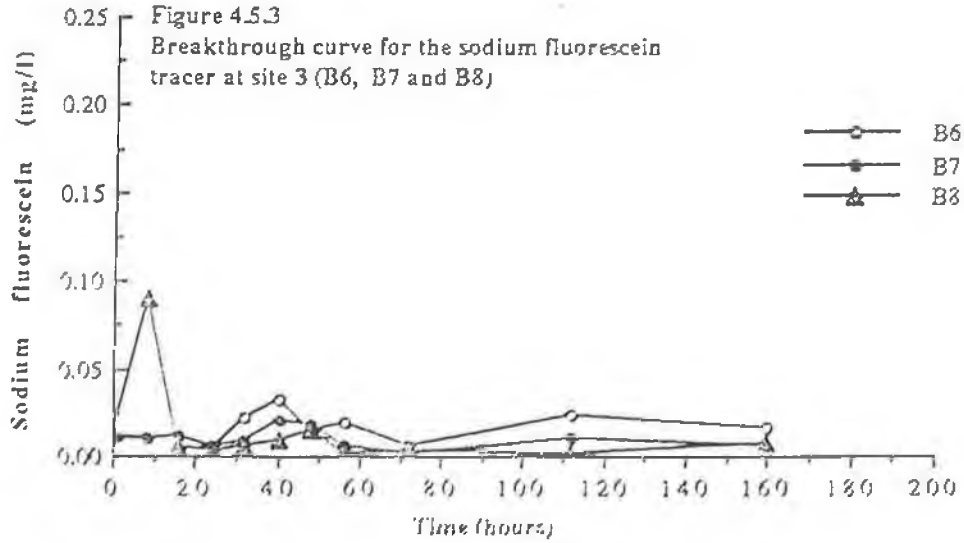

Figure 4.5.3
Breakthrough curve for the sodium fluorescein tracer at site 3 (B6, B7 and B8)

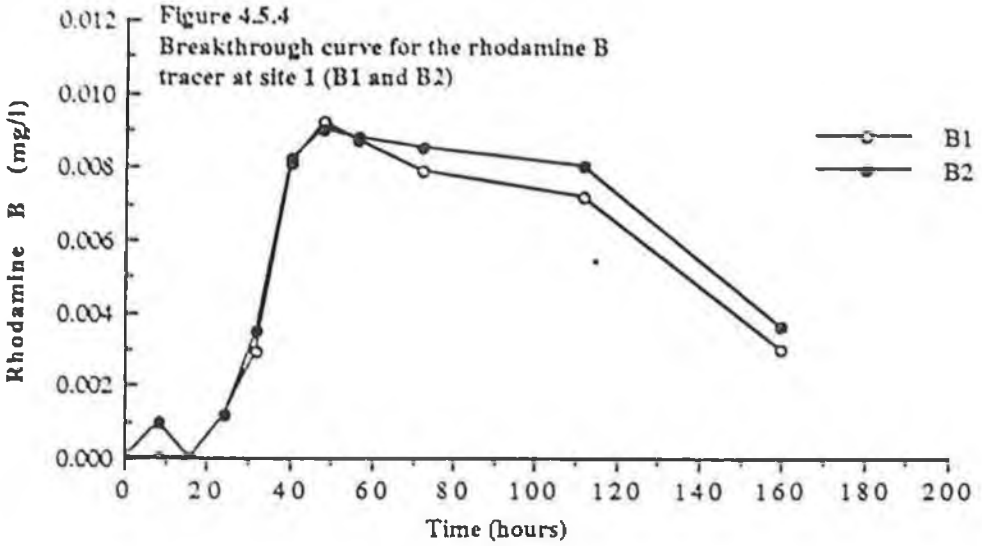

Figure 4.5.4
Breakthrough curve for the rhodamine B tracer at site 1 (B1 and B2)

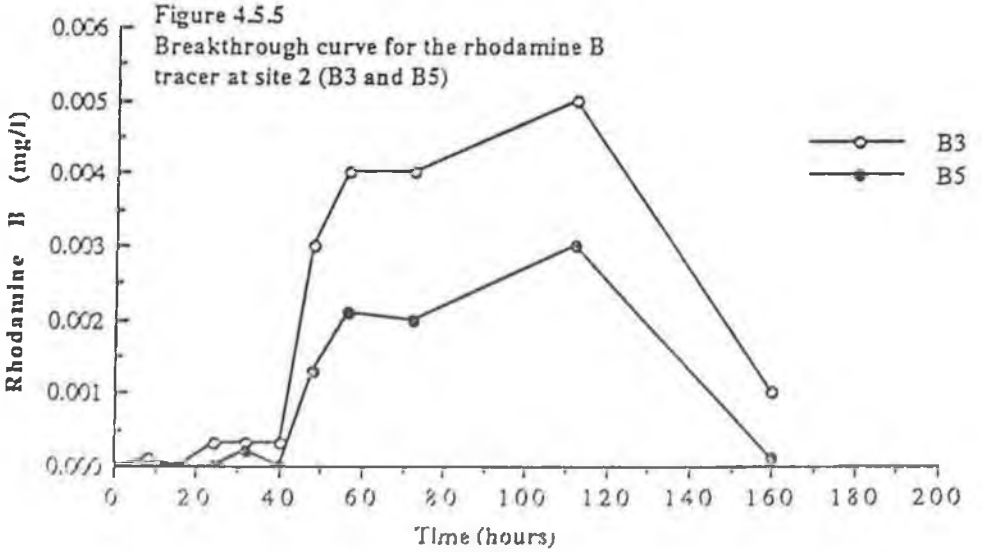

Figure 4.5.5
Breakthrough curve for the rhodamine B tracer at site 2 (B3 and B5)

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Figure 4.5.6
Breakthrough curve for the ion tracers
(measured by electrical conductivity)
at site 1 (B1 and B2)

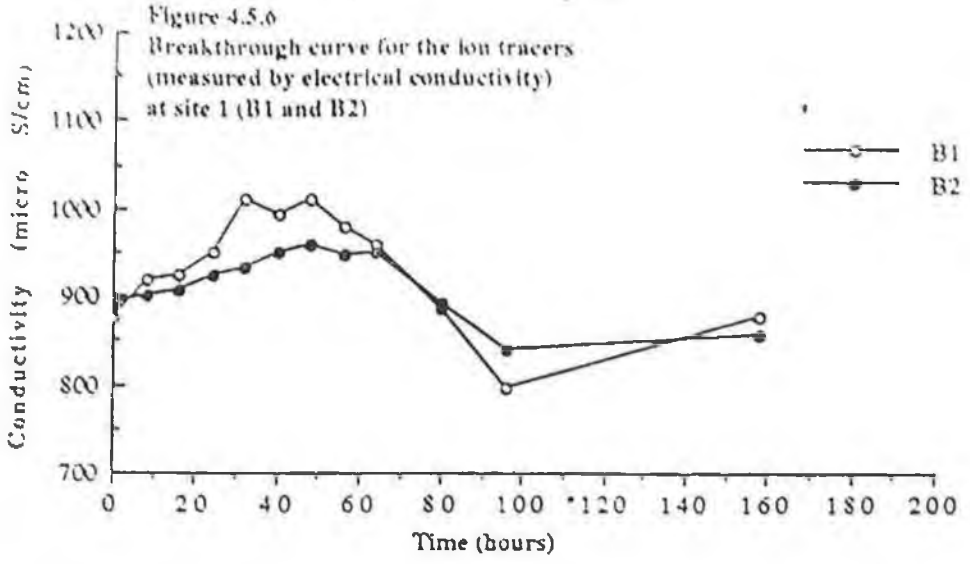


Figure 4.5.7
Breakthrough curve for the ion tracers
(measured by electrical conductivity)
at site 2 (B3 and B5)

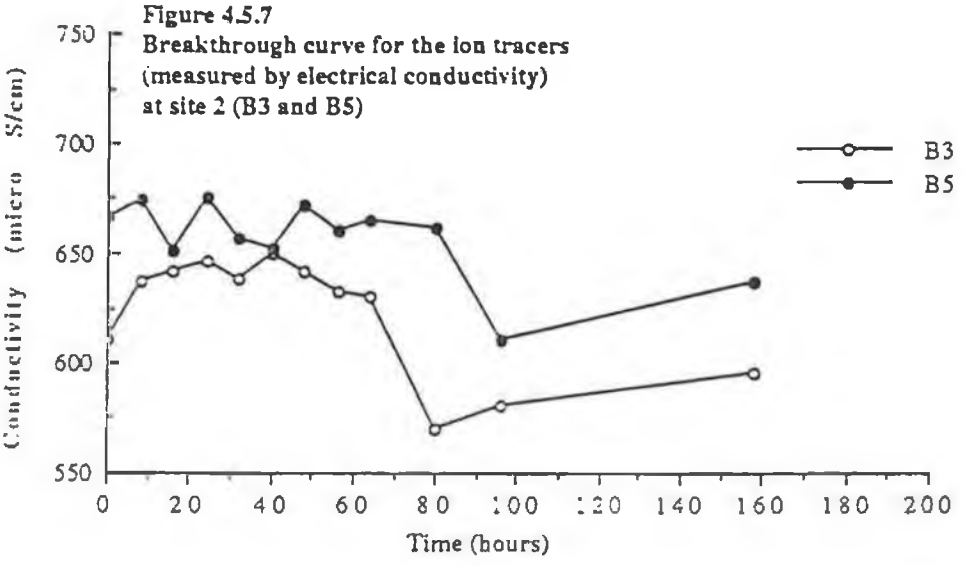
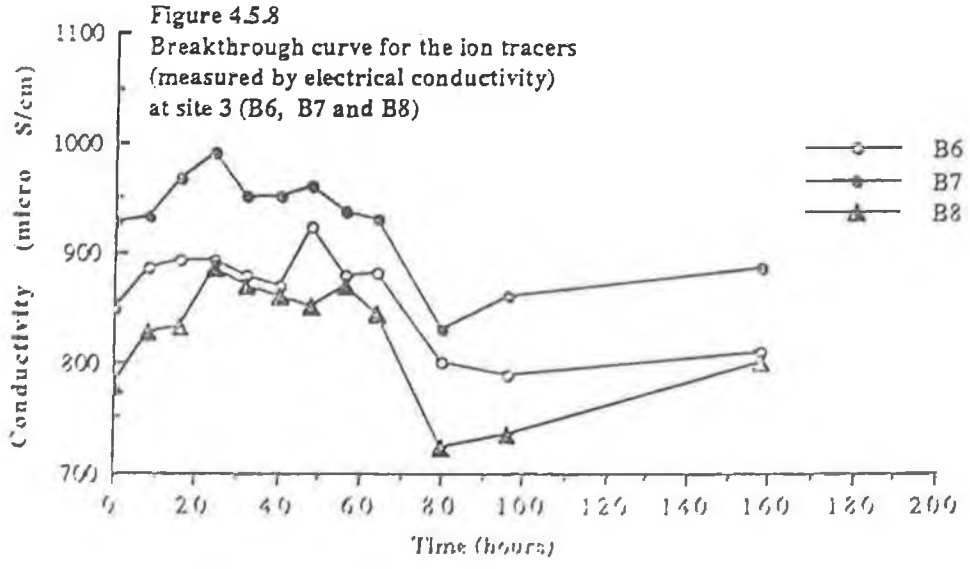


Figure 4.5.8
Breakthrough curve for the ion tracers
(measured by electrical conductivity)
at site 3 (B6, B7 and B8)



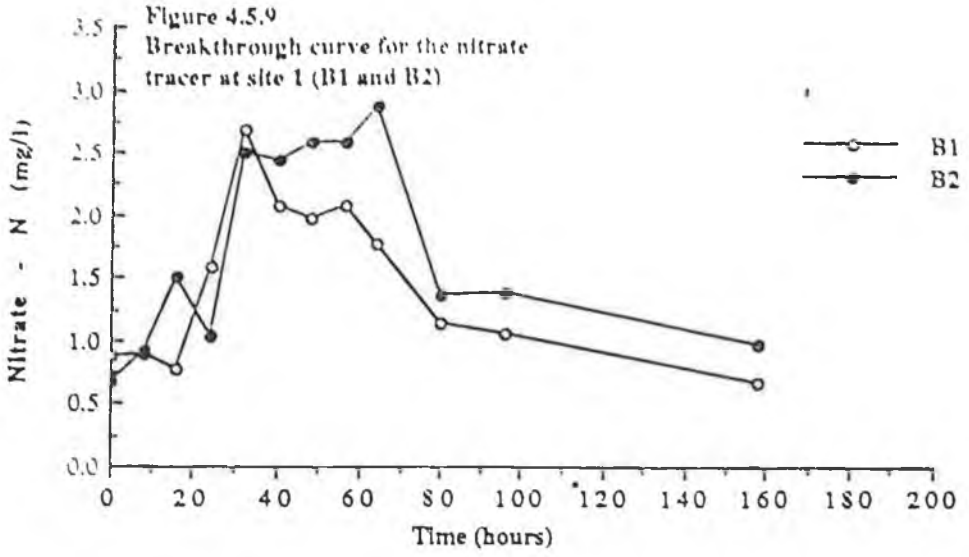


Figure 4.5.9
Breakthrough curve for the nitrate tracer at site 1 (B1 and B2)

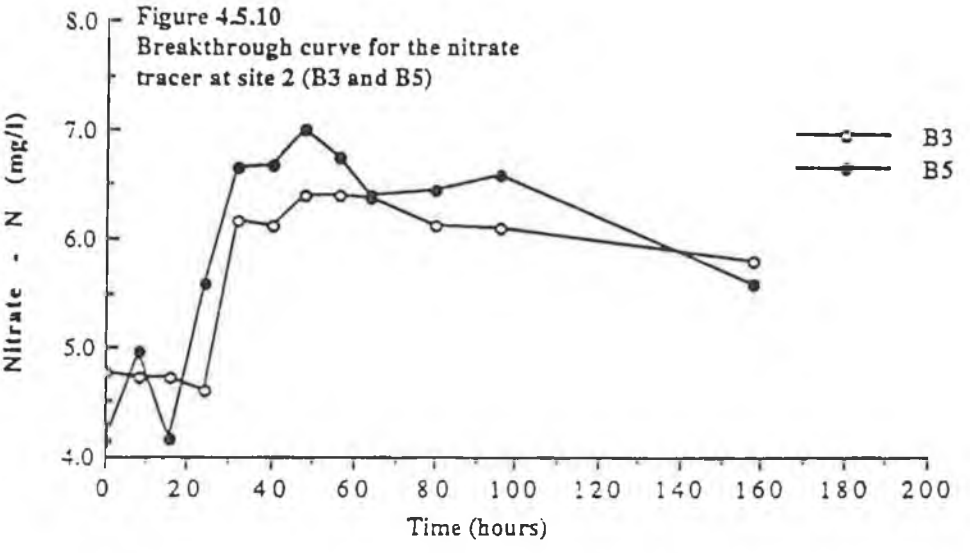


Figure 4.5.10
Breakthrough curve for the nitrate tracer at site 2 (B3 and B5)

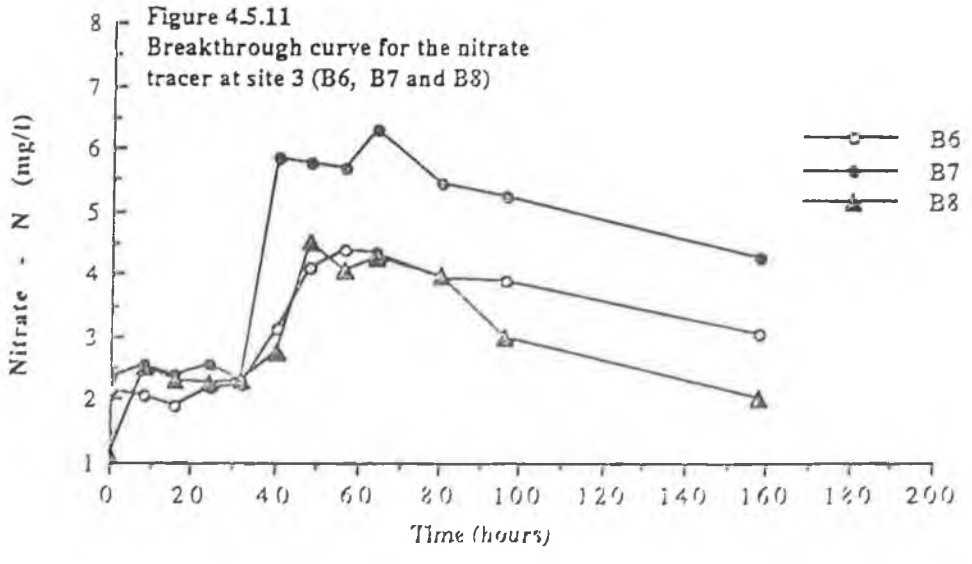


Figure 4.5.11
Breakthrough curve for the nitrate tracer at site 3 (B6, B7 and B8)

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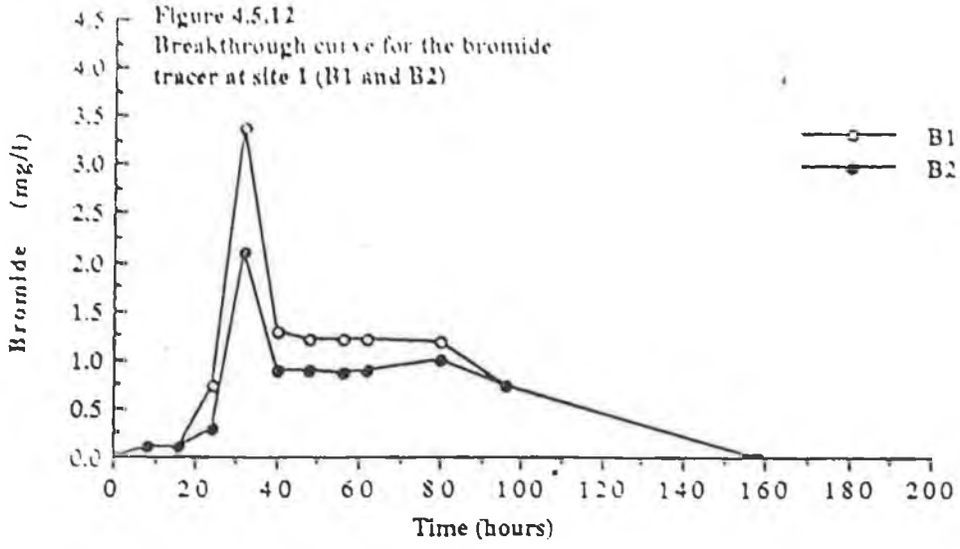

Figure 4.5.12
Breakthrough curve for the bromide tracer at site 1 (B1 and B2)

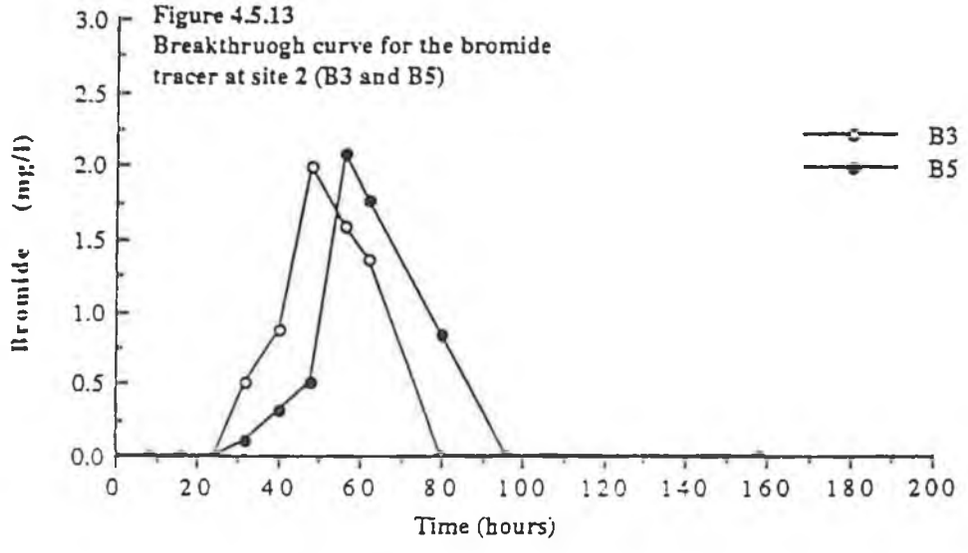

Figure 4.5.13
Breakthruogh curve for the bromide tracer at site 2 (B3 and B5)

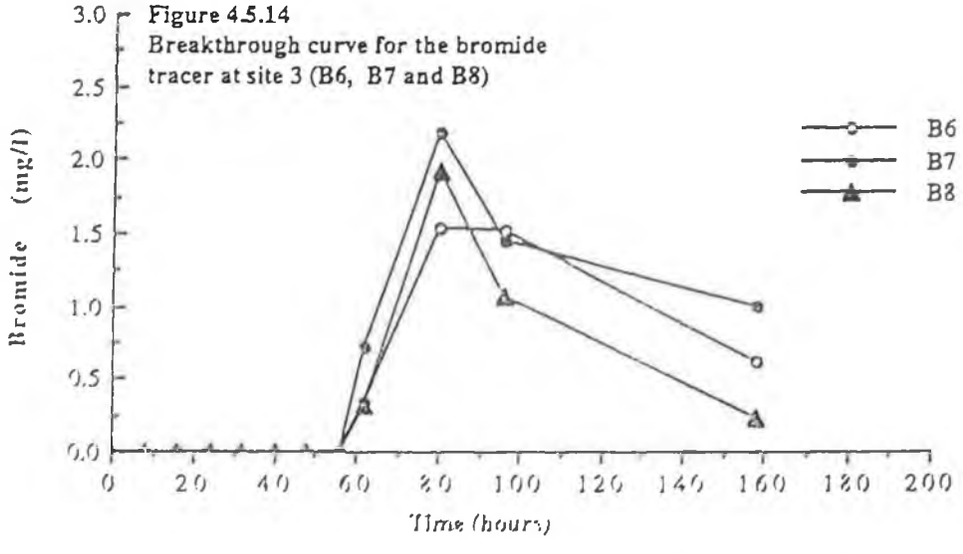

Figure 4.5.14
Breakthrough curve for the bromide tracer at site 3 (B6, B7 and B8)

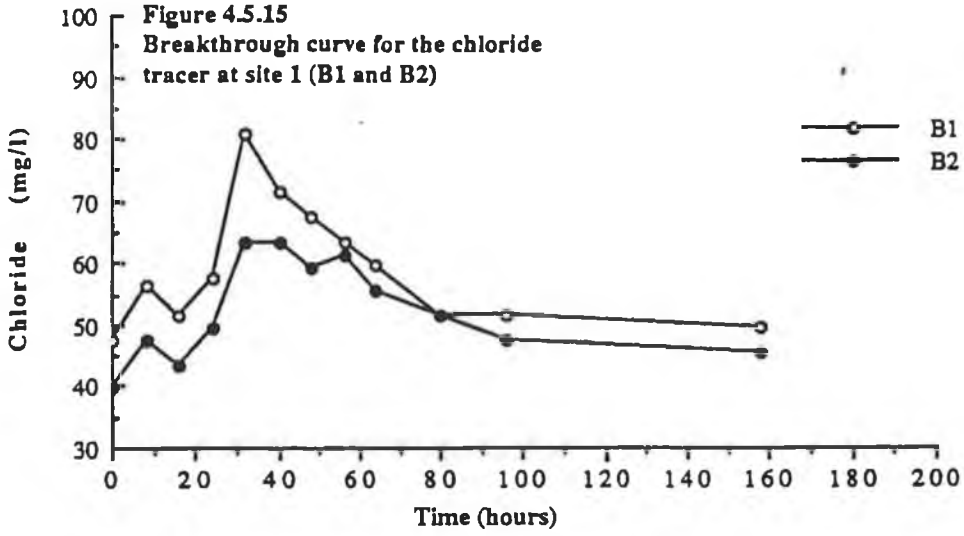

Figure 4.5.15
Breakthrough curve for the chloride tracer at site 1 (B1 and B2)

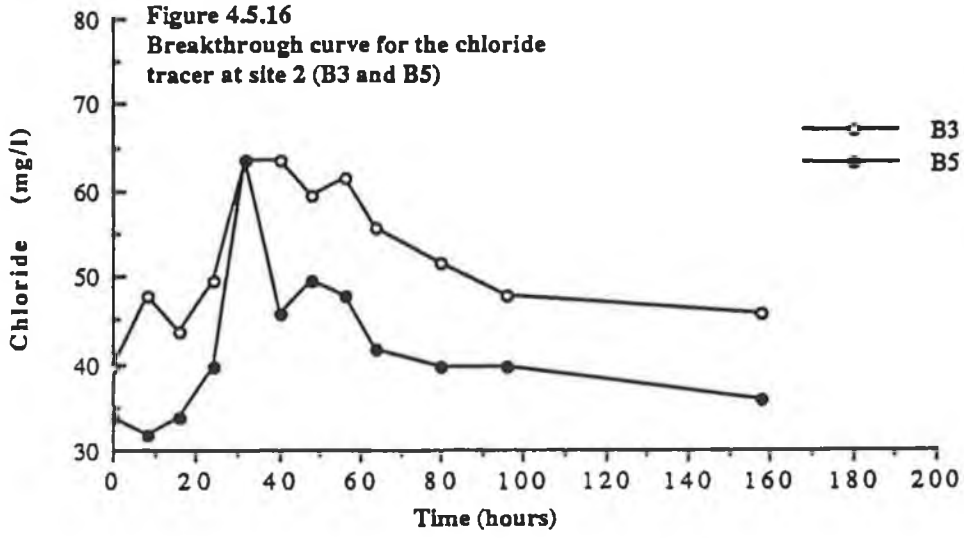

Figure 4.5.16
Breakthrough curve for the chloride tracer at site 2 (B3 and B5)

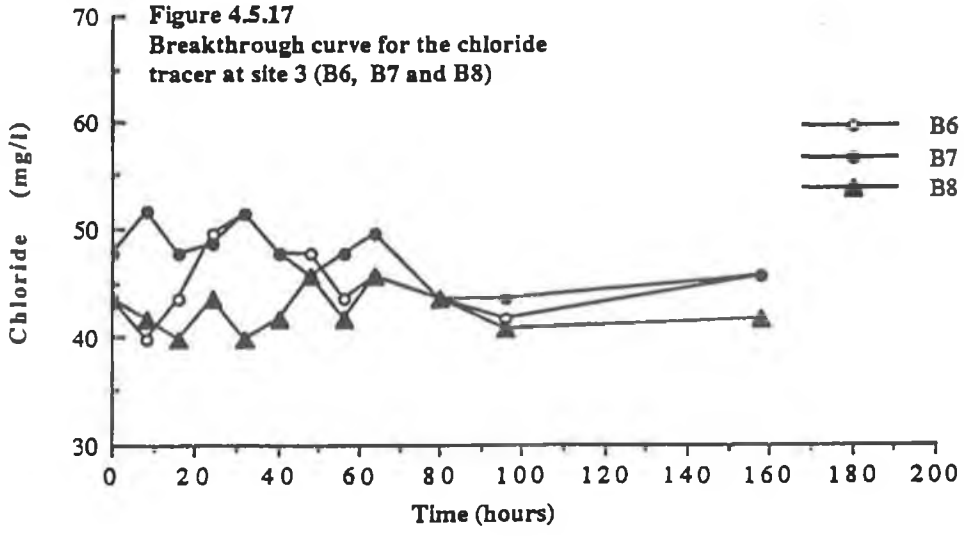

Figure 4.5.17
Breakthrough curve for the chloride tracer at site 3 (B6, B7 and B8)

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Figure 4.5.18
Breakthrough curve for the sodium tracer at site 1 (B1 and B2)

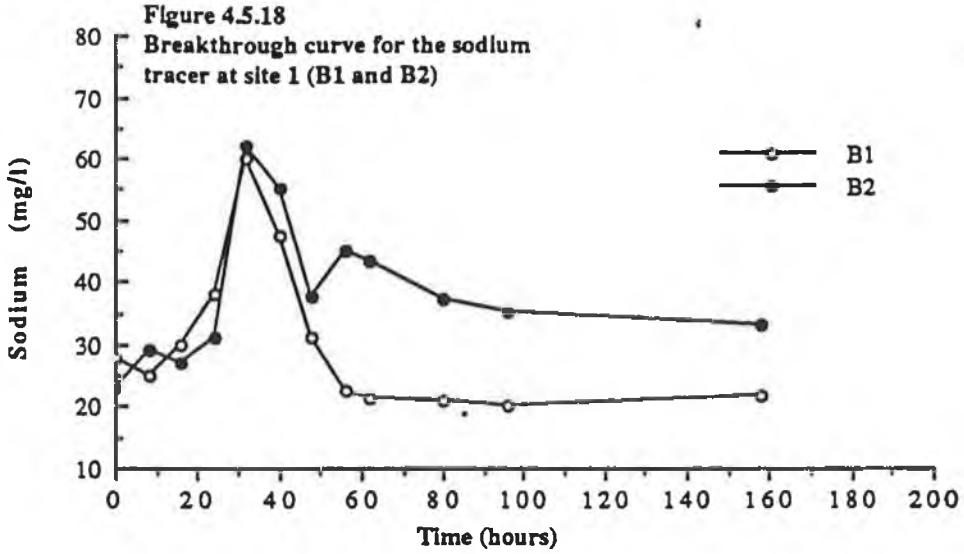


Figure 4.5.19
Breakthrough curve for the sodium tracer at site 2 (B3 and B5)

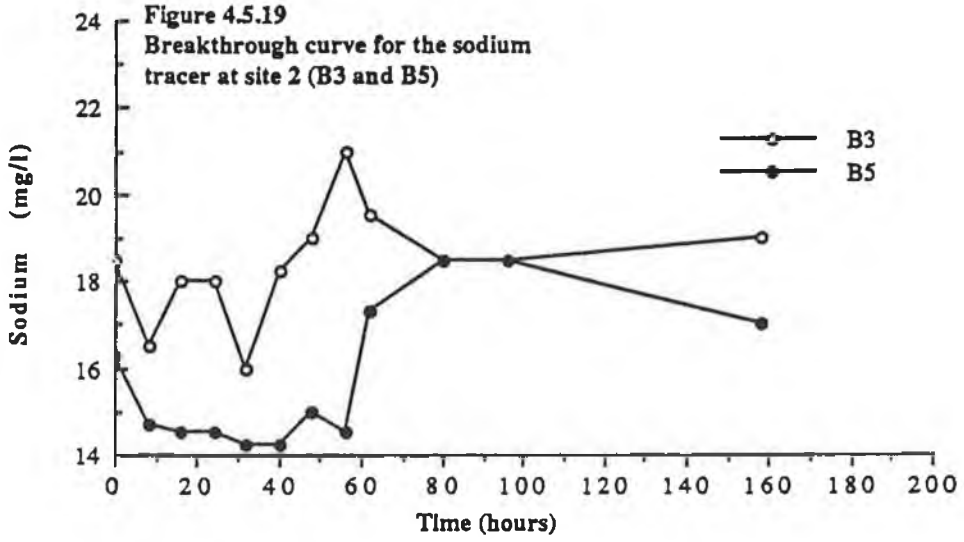
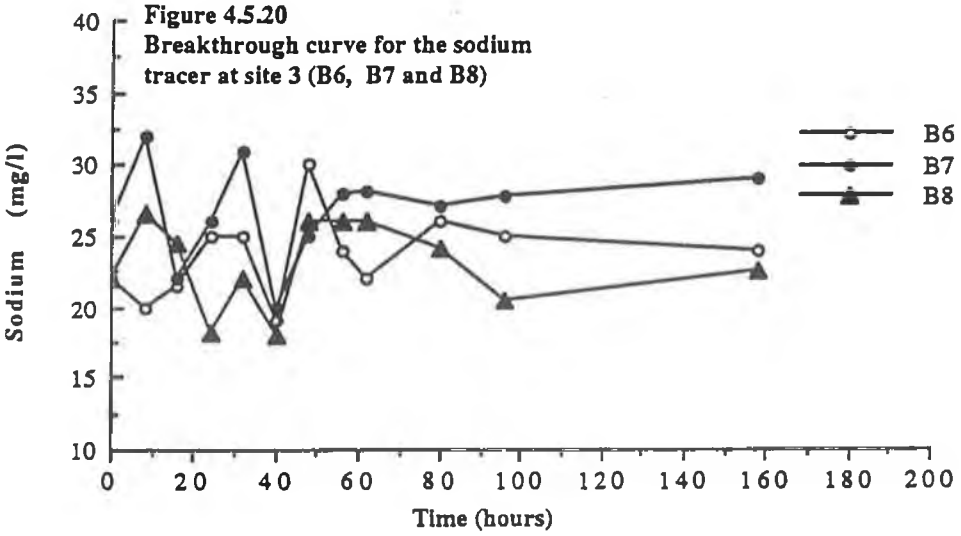


Figure 4.5.20
Breakthrough curve for the sodium tracer at site 3 (B6, B7 and B8)



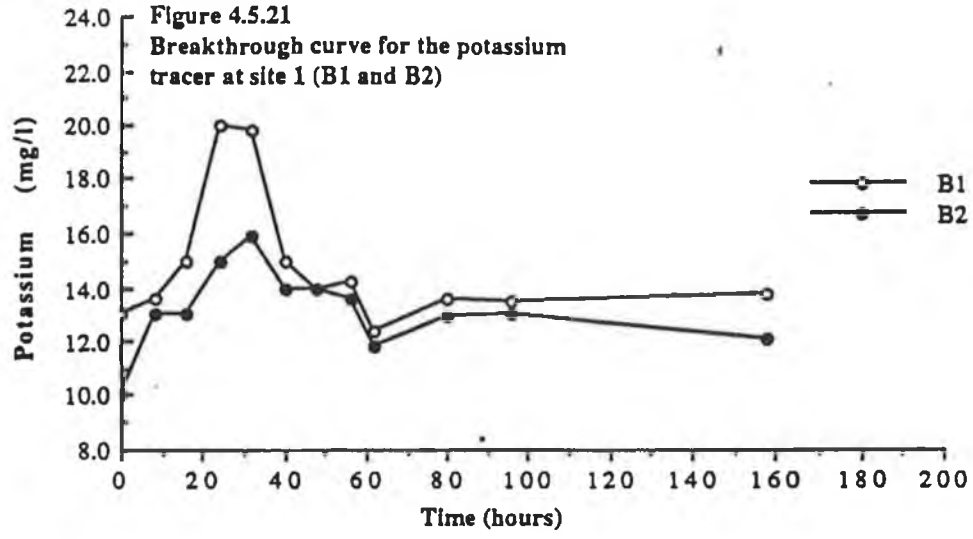


Figure 4.5.21
Breakthrough curve for the potassium tracer at site 1 (B1 and B2)

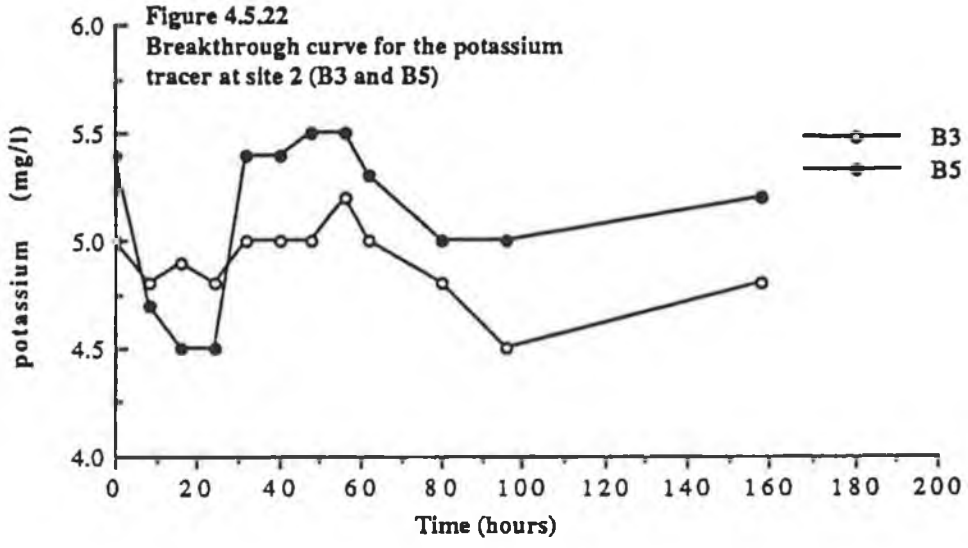


Figure 4.5.22
Breakthrough curve for the potassium tracer at site 2 (B3 and B5)

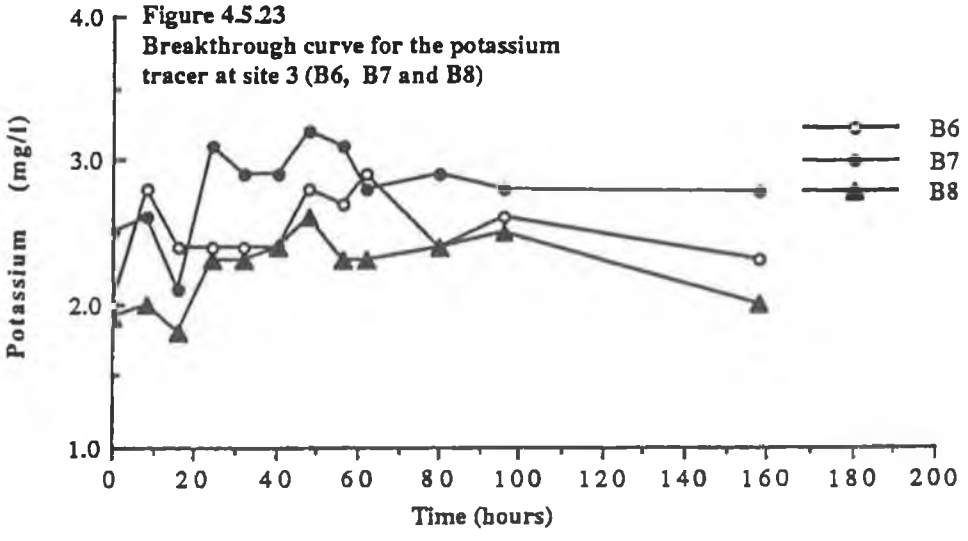


Figure 4.5.23
Breakthrough curve for the potassium tracer at site 3 (B6, B7 and B8)

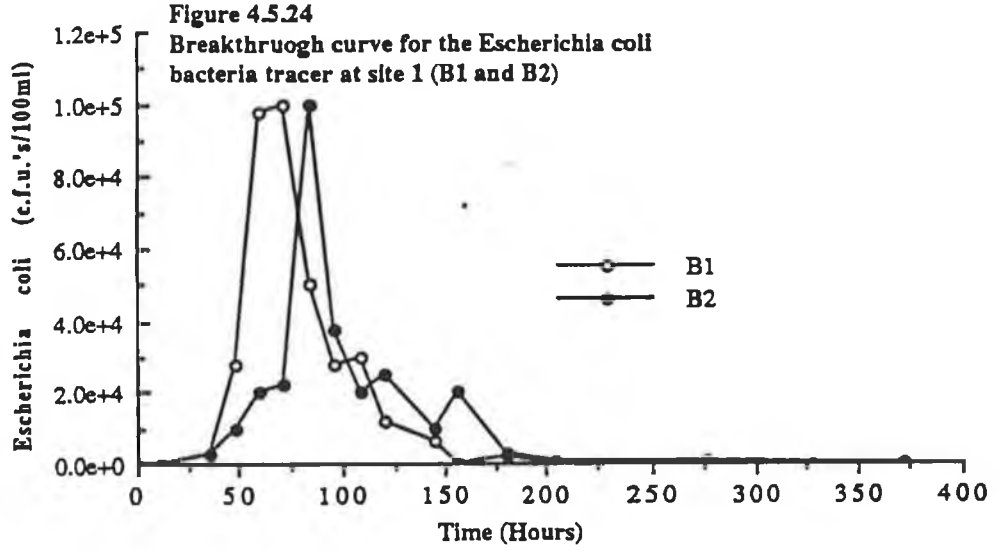


Figure 4.5.24
Breakthruogh curve for the Escherichia coli bacteria tracer at site 1 (B1 and B2)

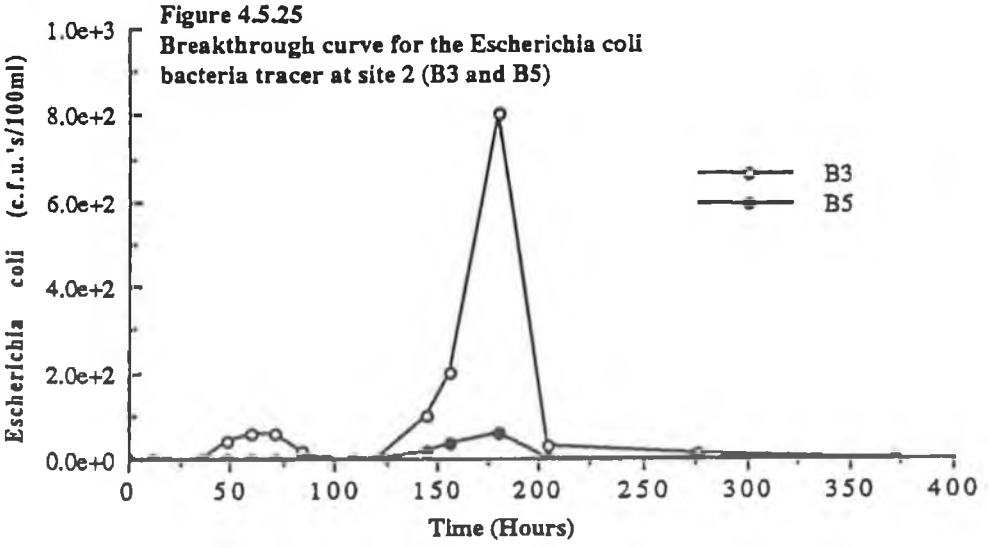


Figure 4.5.25
Breakthrough curve for the Escherichia coli bacteria tracer at site 2 (B3 and B5)

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Figure 4.5.26
Breakthrough curve for the Bacillus globigii bacteria tracer at site 1 (B1 and B2)

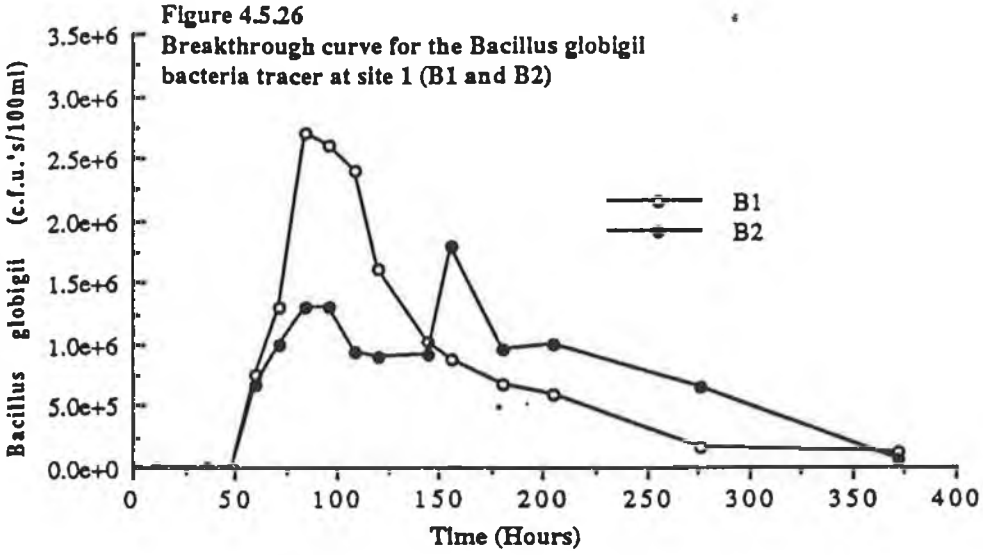

Figure 4.5.27
Breakthrough curve for the Bacillus globigii bacteria tracer at site 2 (B3 and B5)

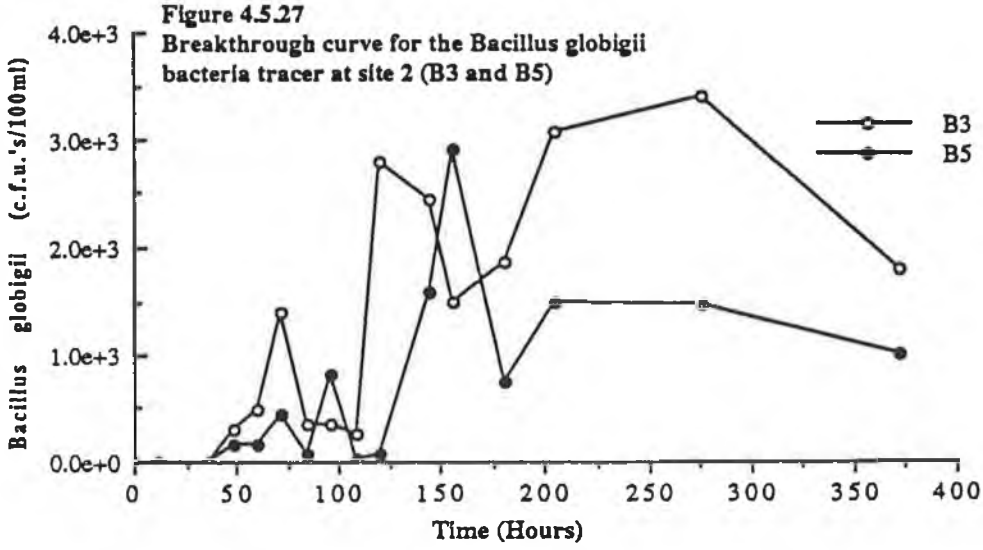

Figure 4.5.28
Breakthrough curve for the Bacillus globigii bacteria tracer at site 3 (B6 and B8)

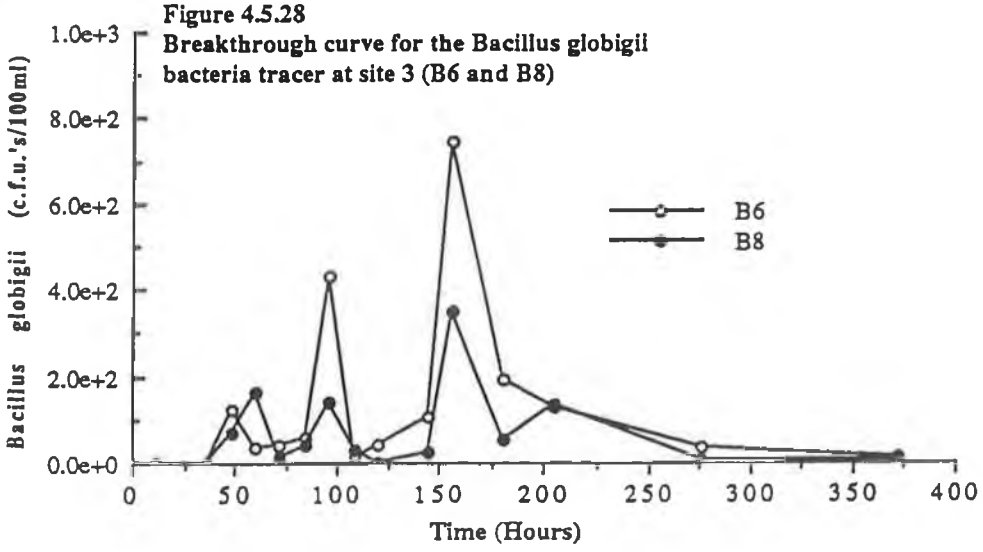

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Figure 4.5.29
Comparison of the breakthrough curves for the sodium fluorescein and rhodamine B tracers at site 1 (B1)

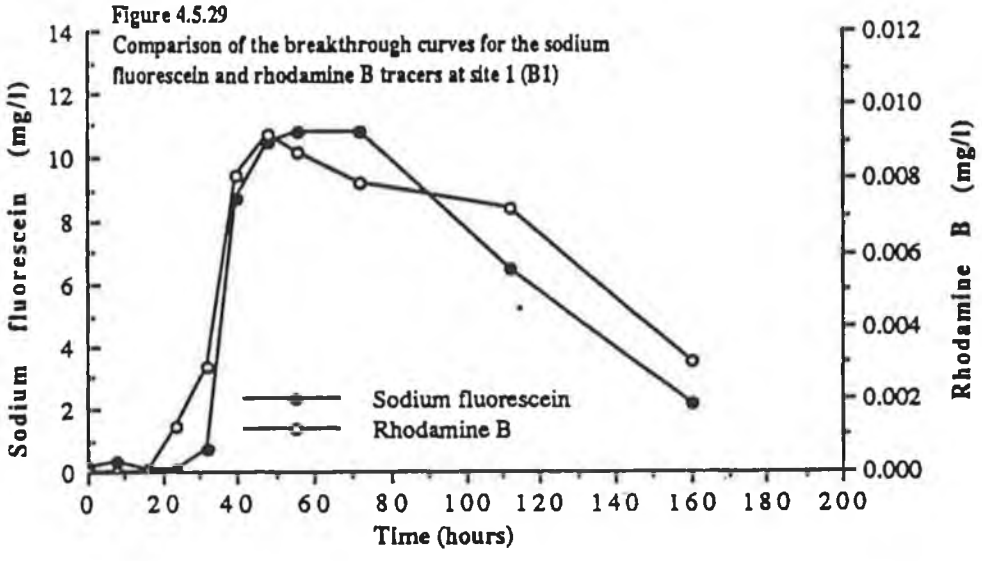


Figure 4.5.30
Comparison of the breakthrough curves for the sodium fluorescein and rhodamine B tracers at site 2 (B3)

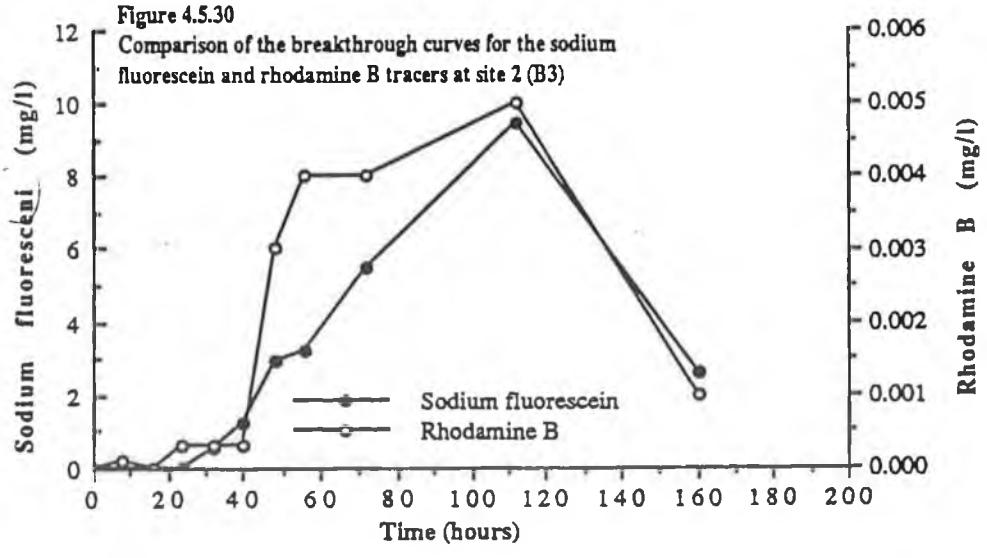


Figure 4.5.31
Comparison of the breakthrough curves for the sodium fluorescein and nitrate tracers at site 1 (B1)

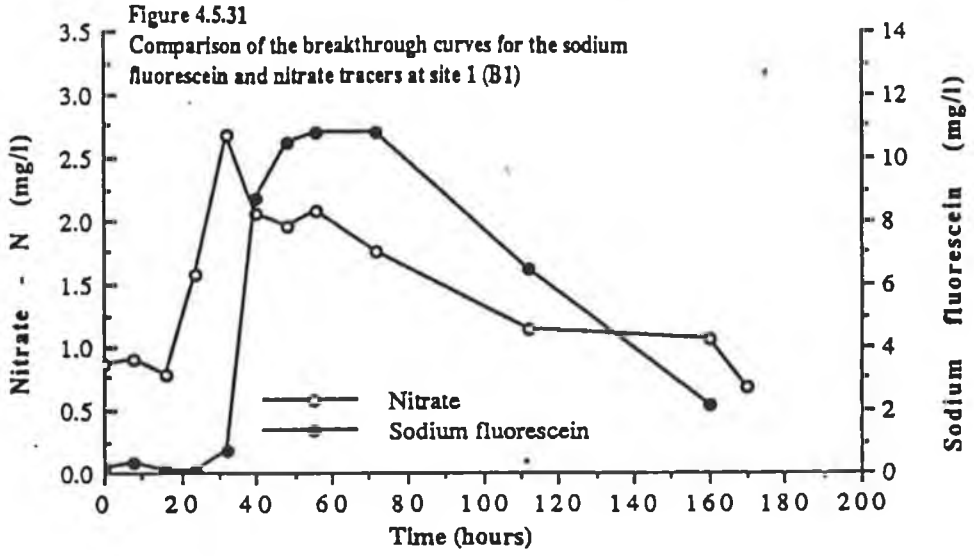


Figure 4.5.32
Comparison of the breakthrough curves for the sodium fluorescein and nitrate tracers at site 2 (B3)

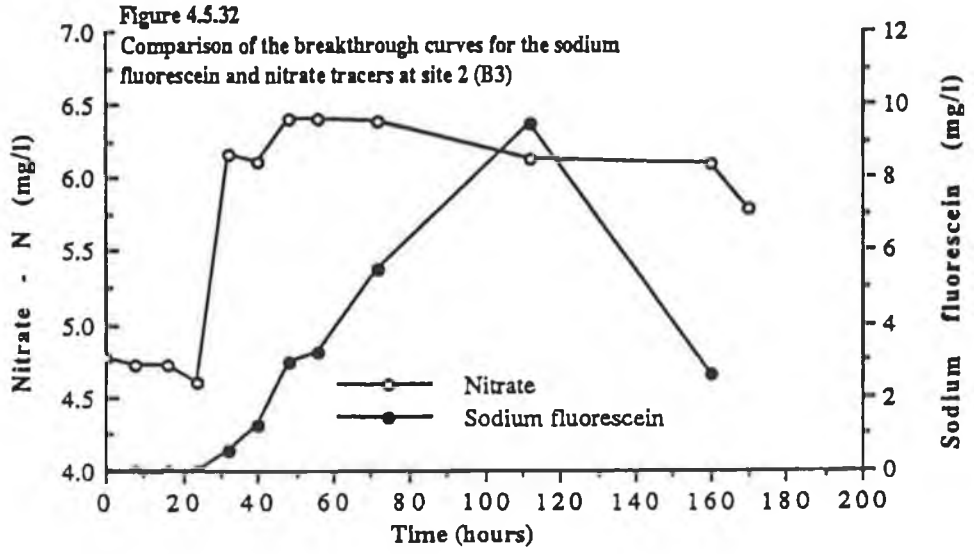


Figure 4.5.33
Comparison of the breakthrough curves for the sodium fluorescein and nitrate tracers at site 3 (B6)

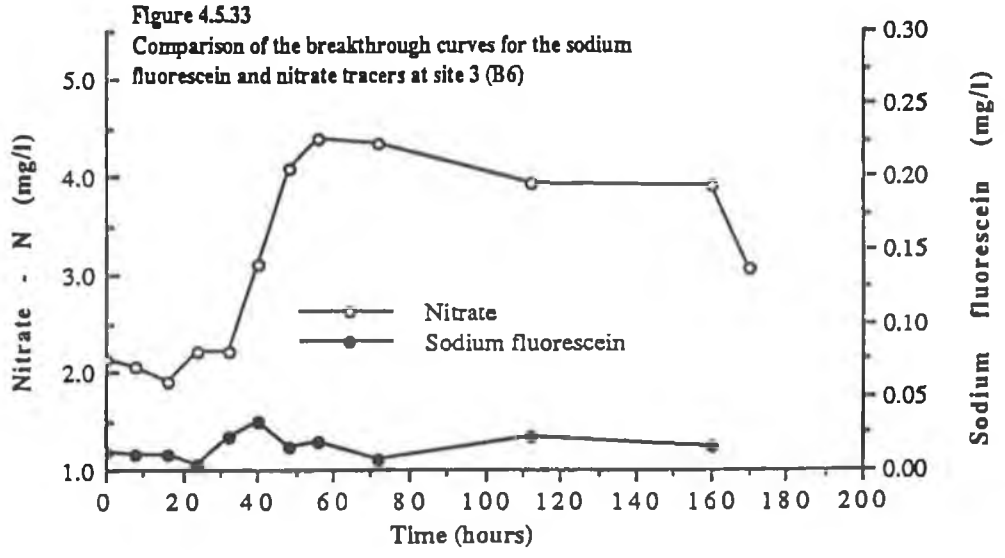


Figure 4.5.34
Comparison of the breakthrough curves for the sodium fluorescein and bromide tracers at site 1 (B1)

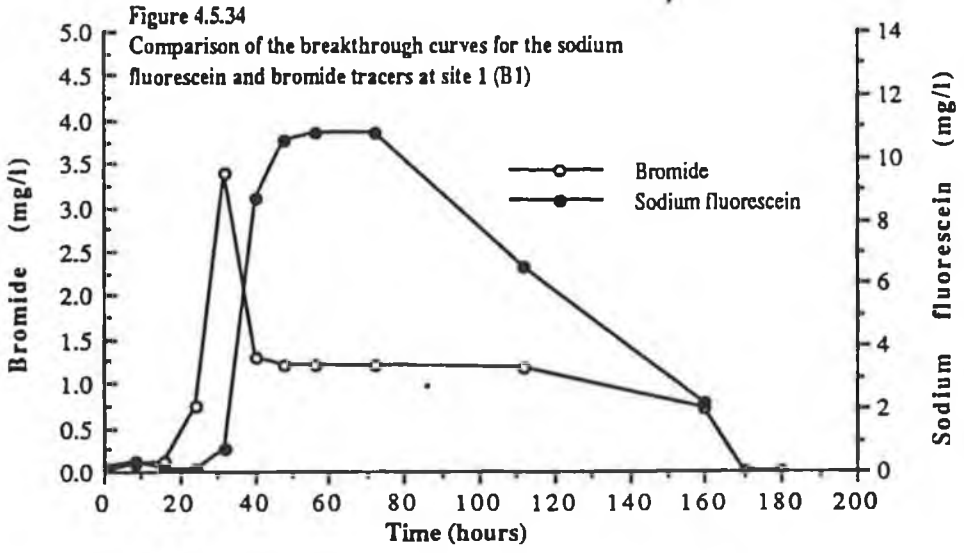


Figure 4.5.35
Comparison of the breakthrough curves for the sodium fluorescein and bromide tracers at site 2 (B3)

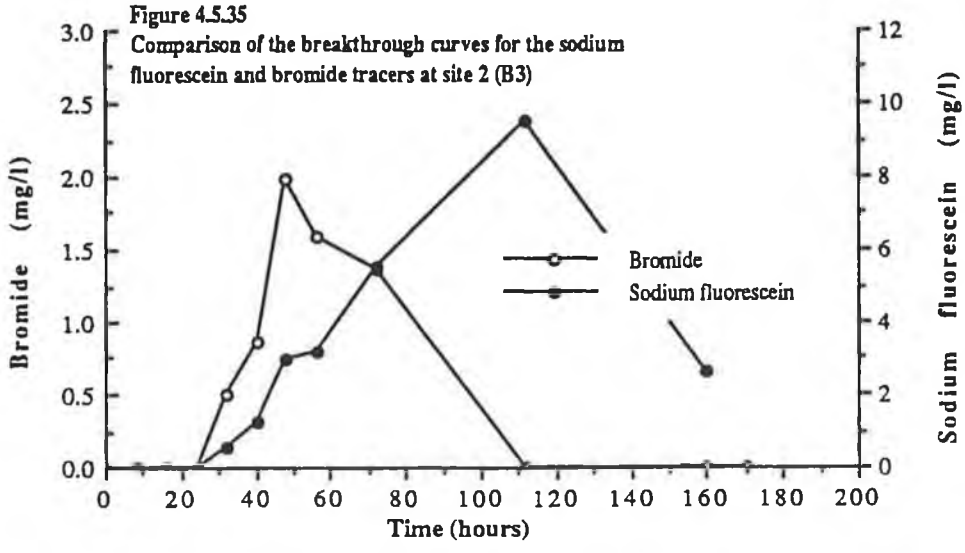


Figure 4.5.36
Comparison of the breakthrough curves for the sodium fluorescein and bromide tracers at site 3 (B6)

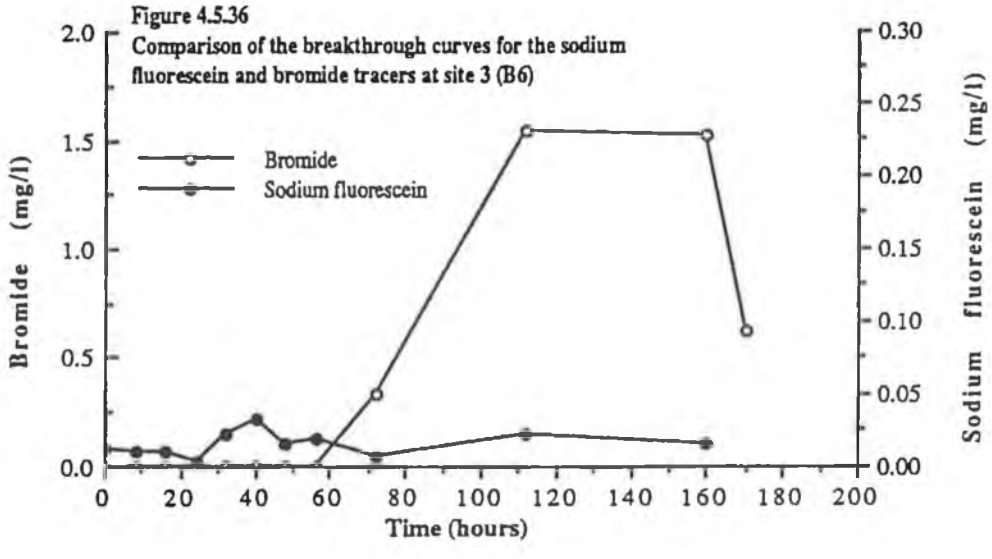


Figure 4.5.37
Comparison of the breakthrough curves for the sodium fluorescein and Bacillus globigii bacteria tracers at site 1 (B1)

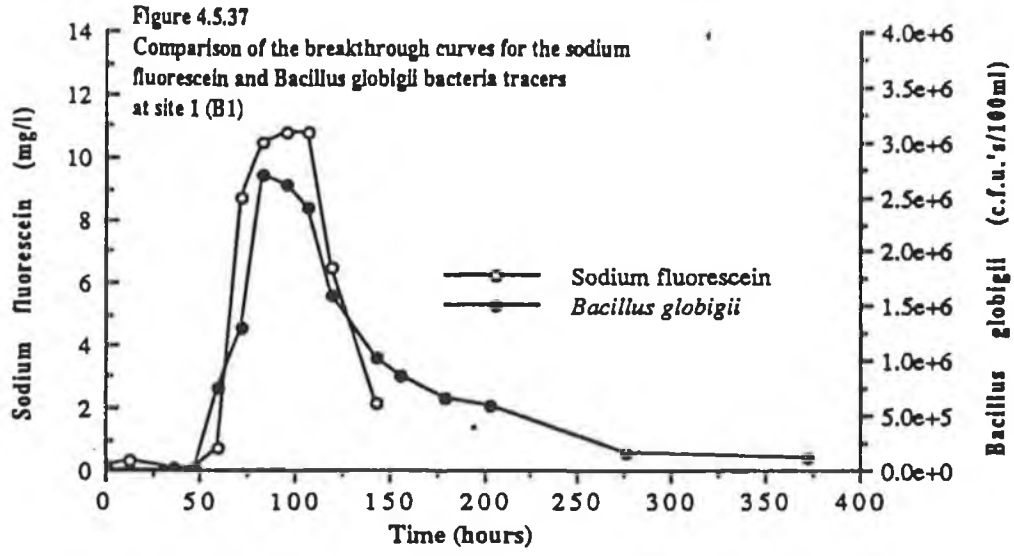


Figure 4.5.38
Comparison of the breakthrough curves for the sodium fluorescein and Bacillus globigii bacteria tracers at site 2 (B3)

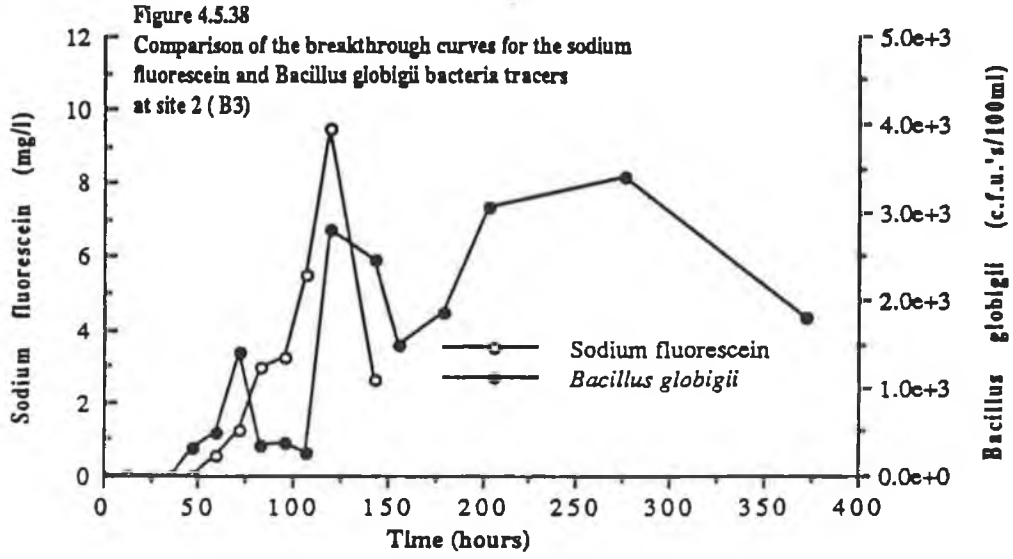
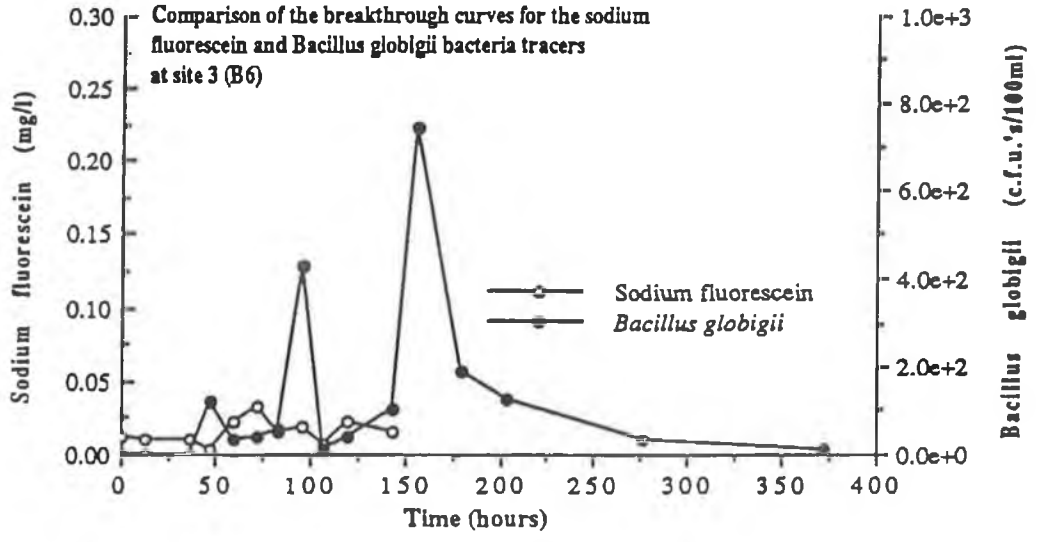


Figure 4.5.39
Comparison of the breakthrough curves for the sodium fluorescein and Bacillus globigii bacteria tracers at site 3 (B6)



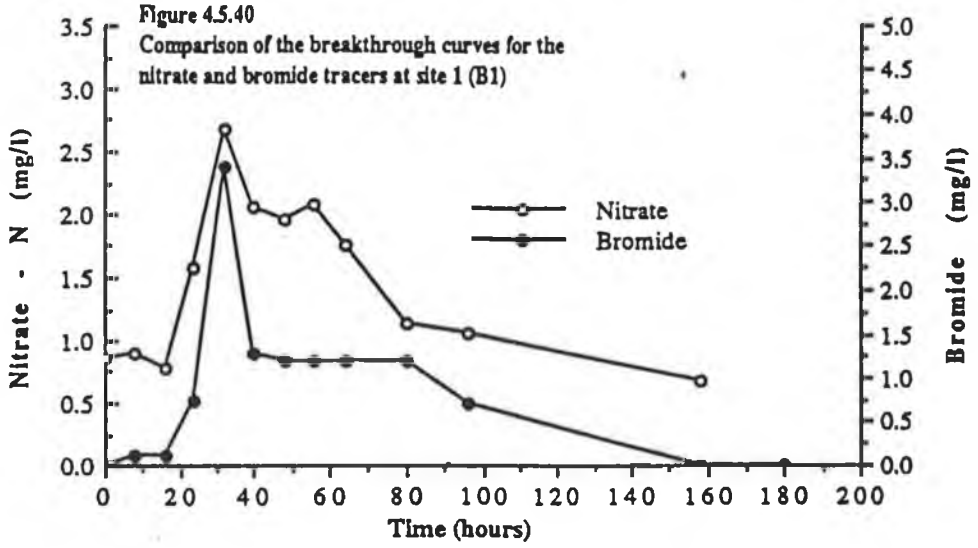


Figure 4.5.40
Comparison of the breakthrough curves for the nitrate and bromide tracers at site 1 (B1)

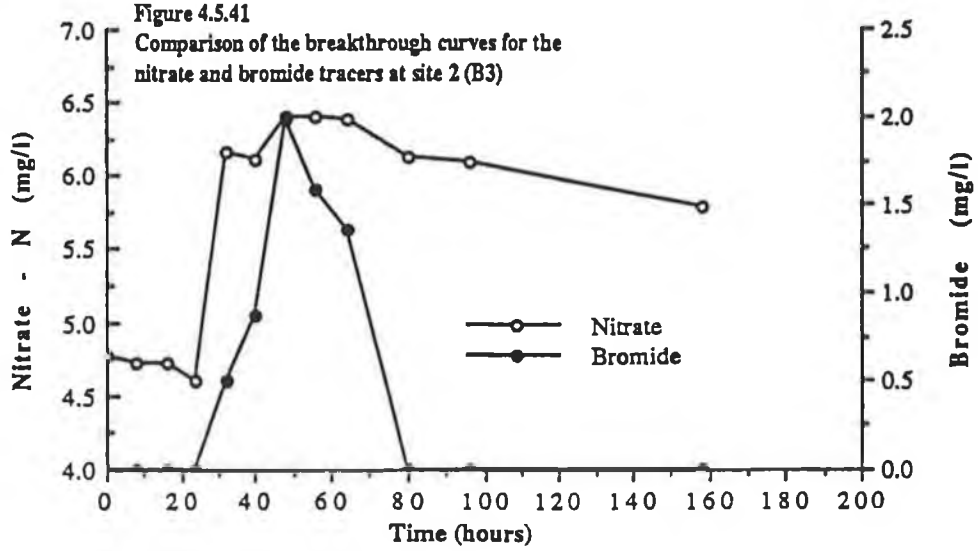


Figure 4.5.41
Comparison of the breakthrough curves for the nitrate and bromide tracers at site 2 (B3)

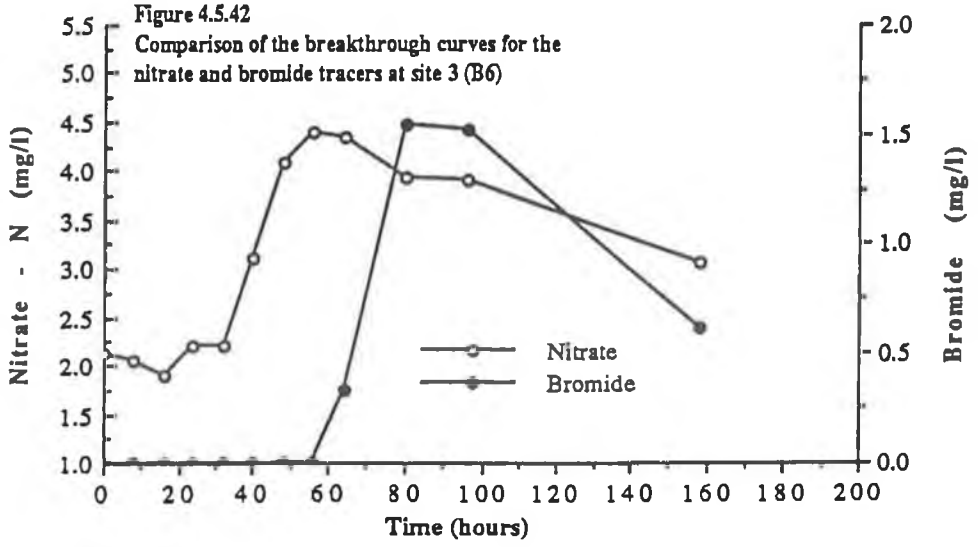


Figure 4.5.42
Comparison of the breakthrough curves for the nitrate and bromide tracers at site 3 (B6)

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Figure 4.5.43
Comparison of the breakthrough curves for the nitrate and Bacillus globigii bacteria tracers at site 1 (B1)

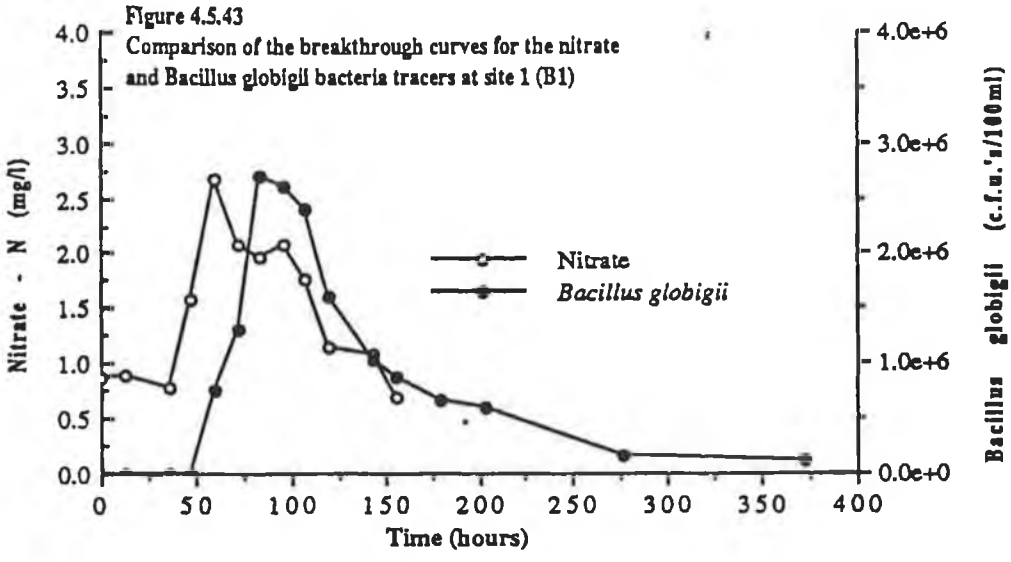


Figure 4.5.44
Comparison of the breakthrough curves for the nitrate and Bacillus globigii bacteria tracers at site 2 (B3)

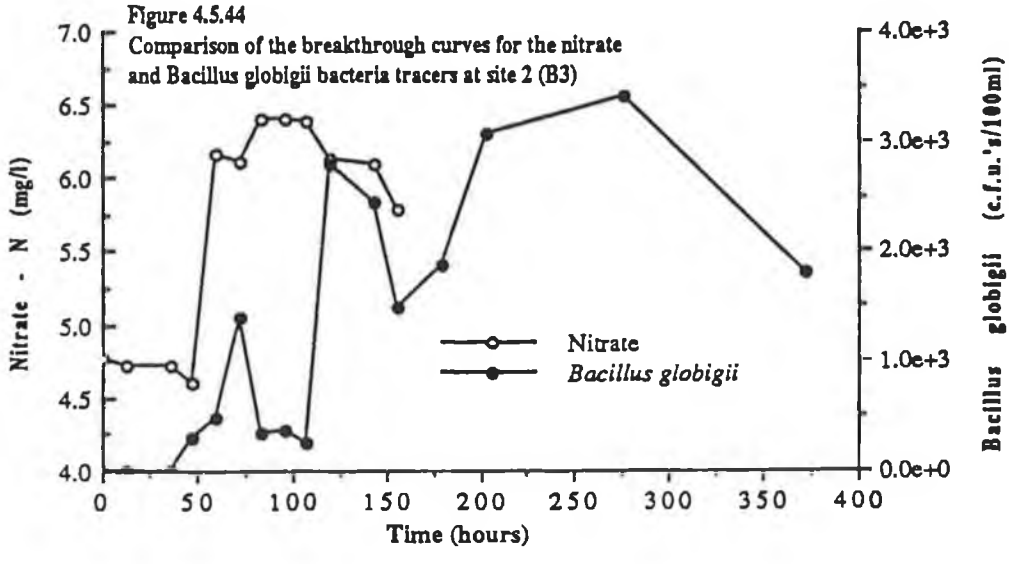


Figure 4.5.45
Comparison of the breakthruogh curves for the nitrate and Bacillus globigii bacteria tracers at site 3 (B6)

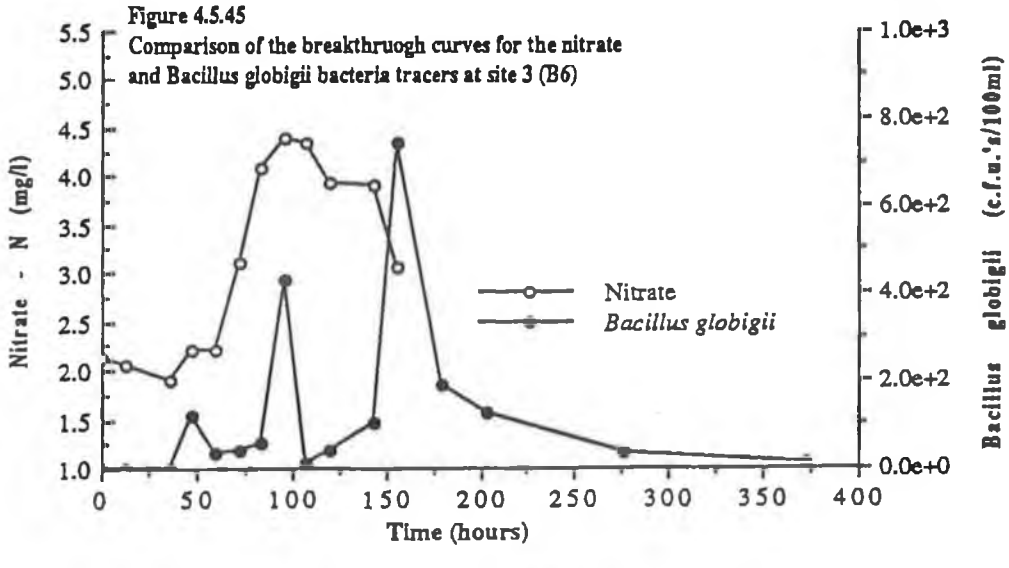


Figure 4.5.46
Comparison of the breakthrough curves for the bromide and Bacillus globigii bacteria tracers at site 1 (B1)

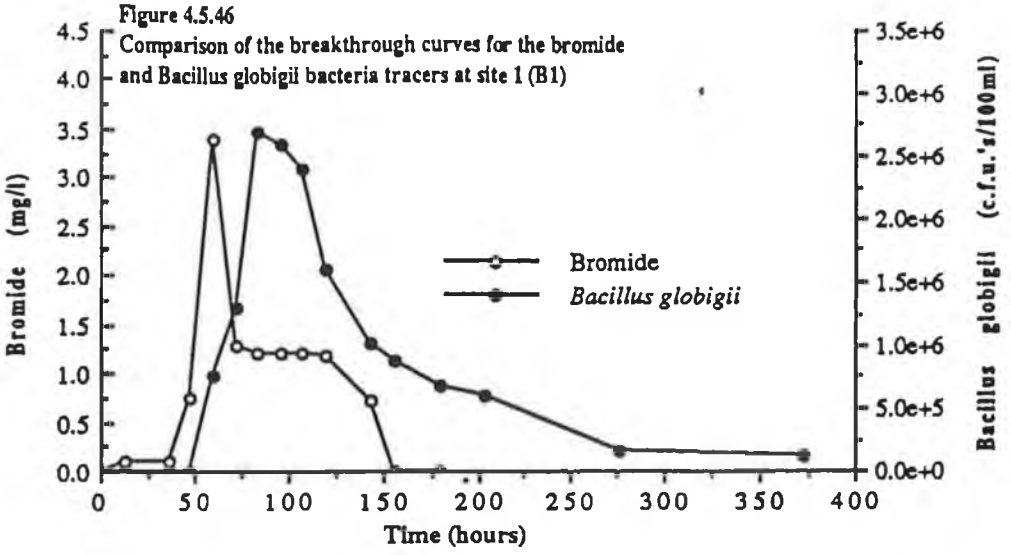


Figure 4.5.47
Comparison of the breakthrough curves for the bromide and Bacillus globigii bacteria tracers at site 2 (B3)

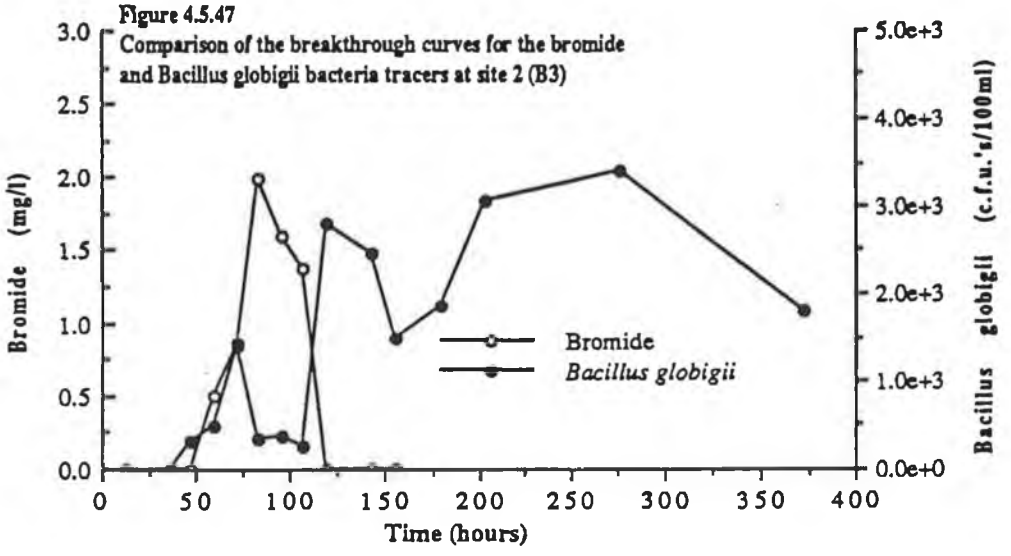
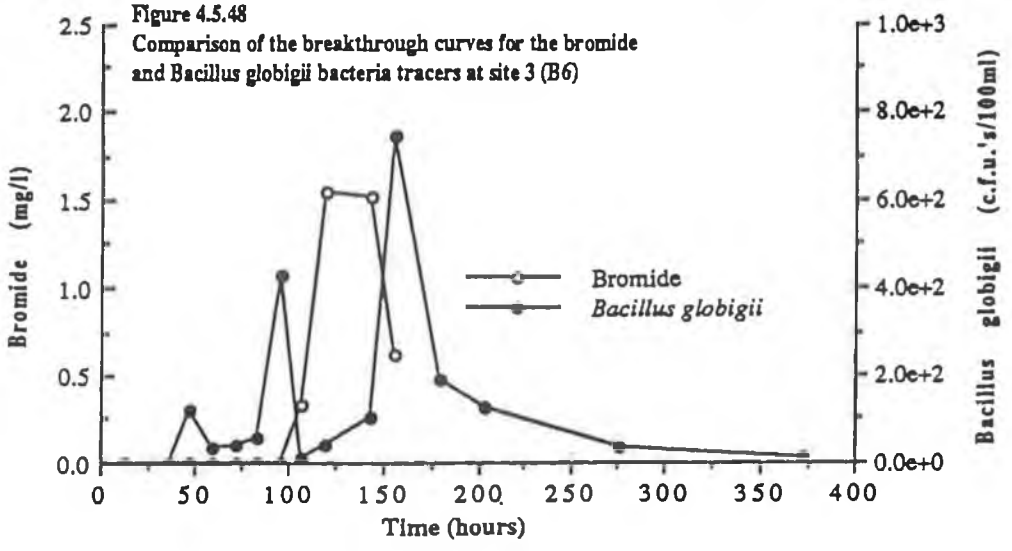


Figure 4.5.48
Comparison of the breakthrough curves for the bromide and Bacillus globigii bacteria tracers at site 3 (B6)



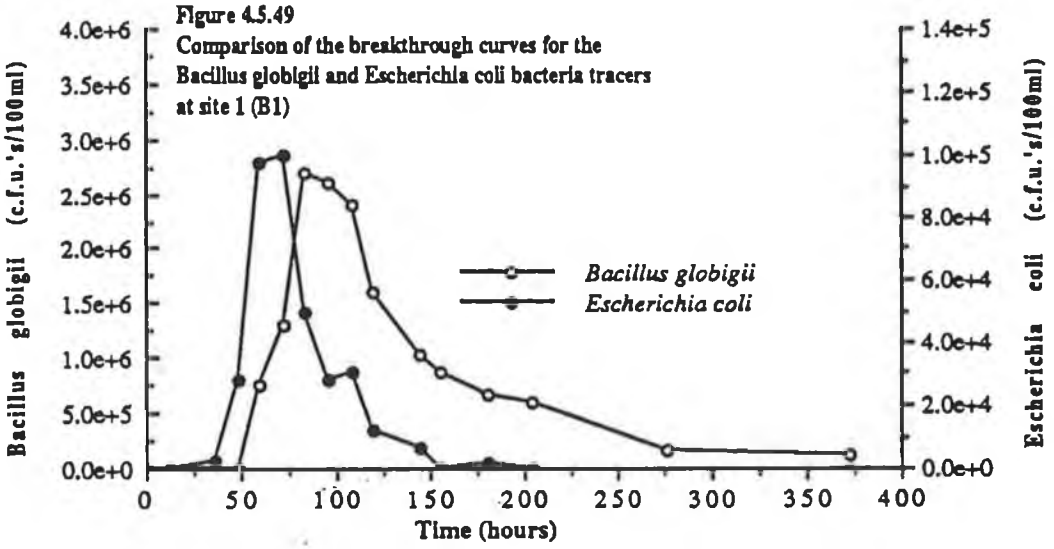


Figure 4.5.49
Comparison of the breakthrough curves for the Bacillus globigii and Escherichia coli bacteria tracers at site 1 (B1)

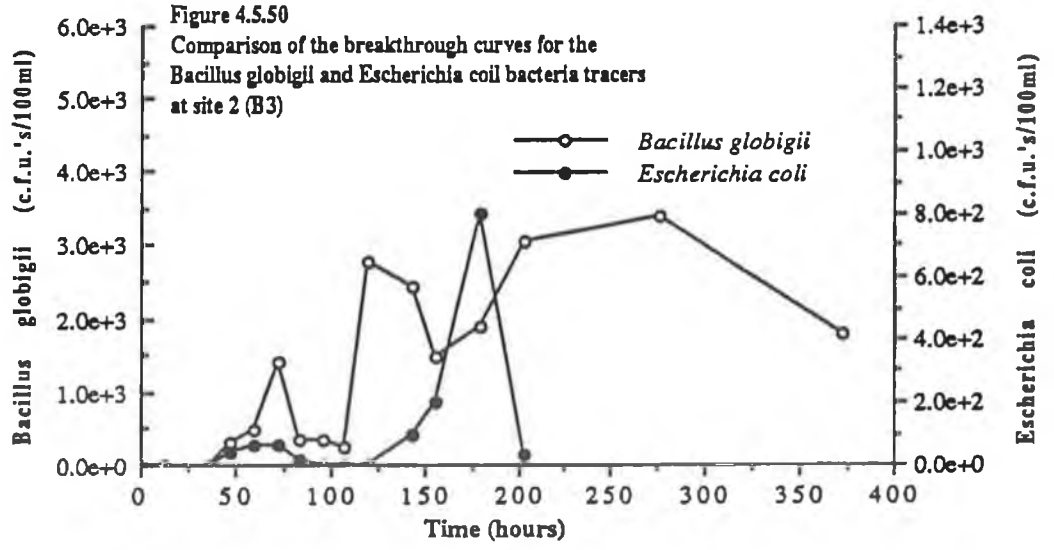


Figure 4.5.50
Comparison of the breakthrough curves for the Bacillus globigii and Escherichia coli bacteria tracers at site 2 (B3)

Figure 4.5.51
Daily rainfall (mm) in the Sligo area during the optical brightner tracing experiments

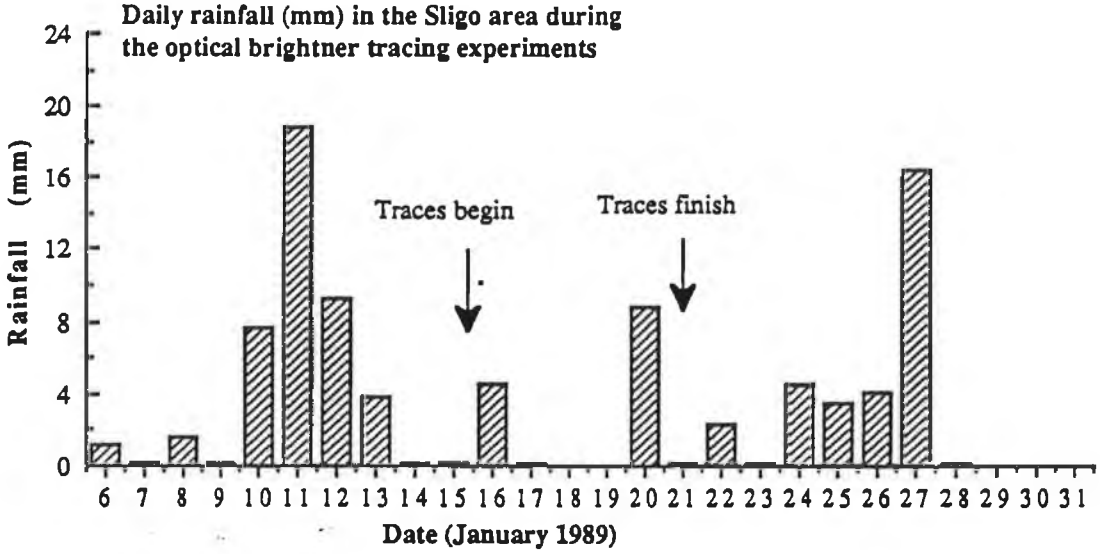


Figure 4.5.52
Daily rainfall (mm) in the Sligo area during the fluorescent dye tracing experiments

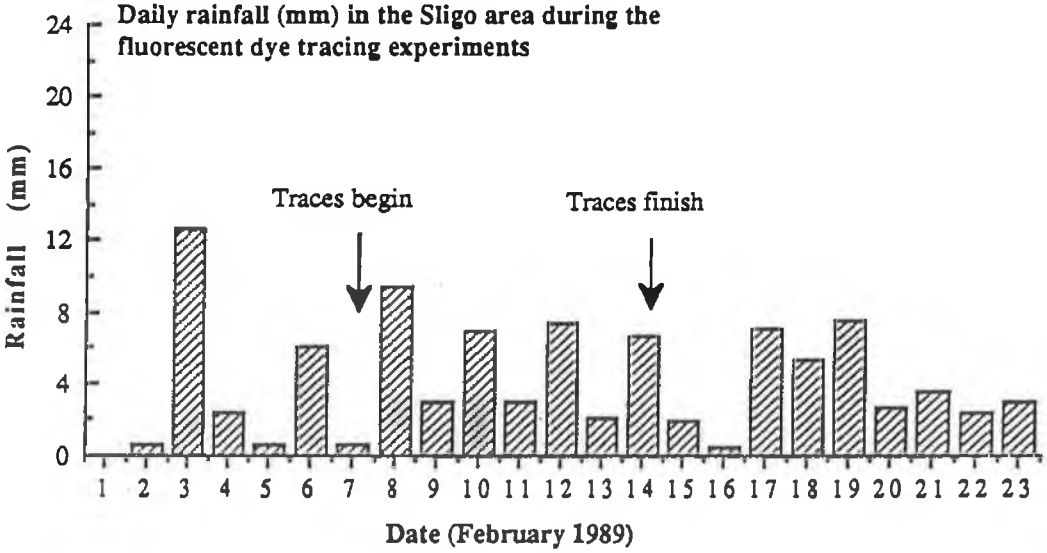


Figure 4.5.53
Daily rainfall (mm) in the Sligo area during the ion tracing experiments

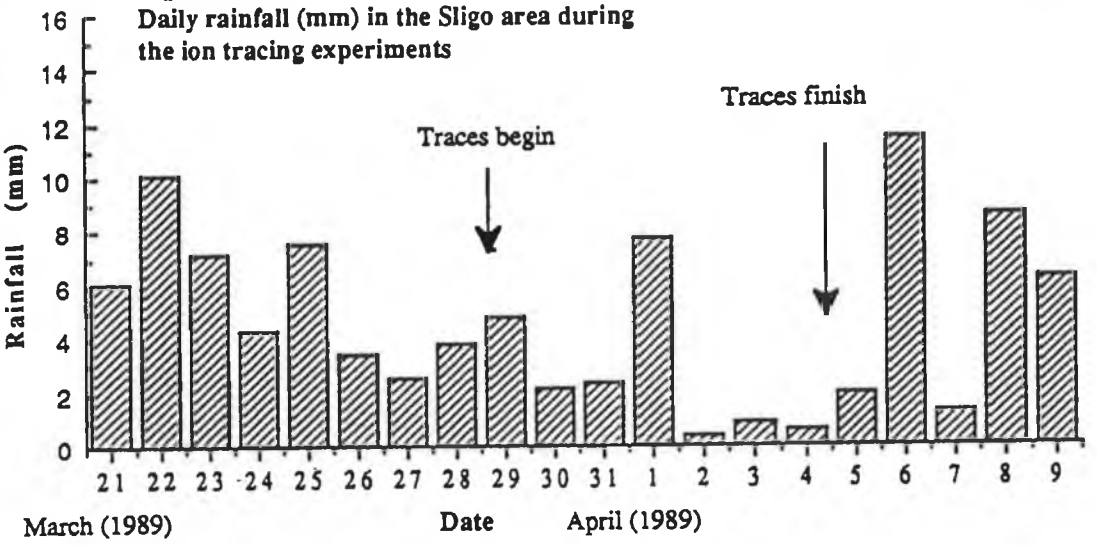


Figure 4.5.54
Daily rainfall (mm) in the Sligo area during the bacterial tracing experiments

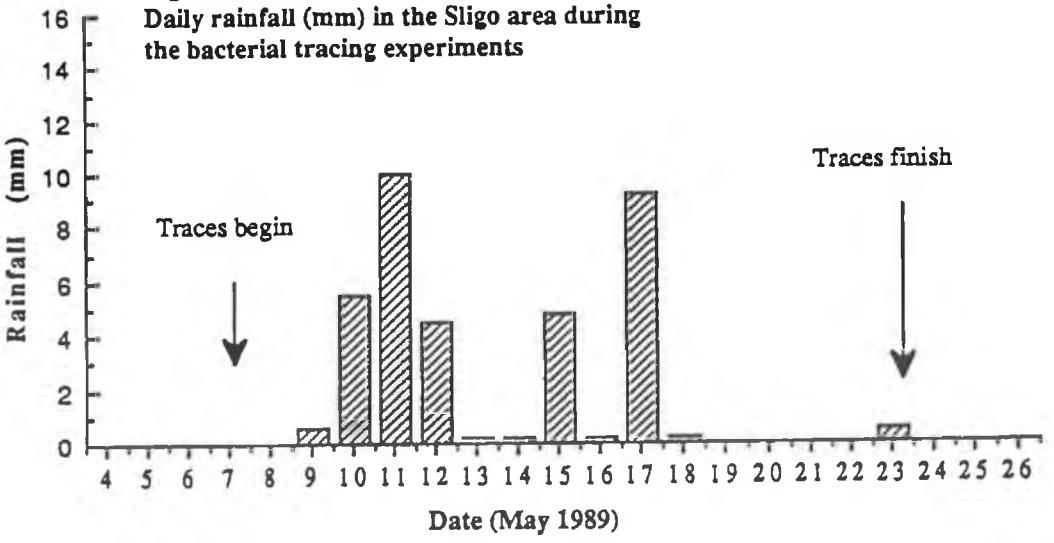


Table 4.5.2

Tracer Materials and Calculated C/Co (max) Ratios for Each of the Monitoring Boreholes

TRACER MATERIAL	C/Co max						
	B1	B2	B3	B5	B6	B7	B8
Fluorescein	0.213	0.072	0.127	0.039	-	-	-
Rhodamine	0.00092	0.00092	0.00045	0.00025	0.00011	-	-
Bromide	0.083	0.051	0.024	0.025	0.019	0.027	0.023
Potassium	0.121	0.102	0.003	0.001	0.013	0.001	0.001
Sodium	0.32	0.39	0.03	0.02	0.08	0.05	0.04
Nitrate	0.02	0.024	0.013	0.024	0.019	0.032	0.027
Chloride	0.225	0.158	0.198	0.132	0.052	0.026	0.013
B.globigiii	0.0054	0.0036	6.8 E-06	5.8 E-06	1.5 E-06	-	
E.coli	0.020	0.020	0.00016	1.2E-05	-	-	-

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4:6 Discussion of Results

The following section discusses the findings of the tracer experiments and compares the results obtained to previous investigations. The results of the fluorescent dye, ion and bacterial traces are presented graphically. Plots of the concentration of tracer material detected in the groundwater samples against time after tracer addition are referred to as breakthrough curves. The following measurements are important in the interpretation of the curves and are continually referred to in the discussion:

(i) Breakthrough Time
The time taken for the first appearence of the tracer material in the groundwater samples to occur after tracer addition is referred to as the breakthrough time.

(ii) Breakthrough Peak
The time at which the maximum concentration of tracer material was recorded in the groundwater samples is the breakthrough peak or breakthrough maxima.

(iii) Maximum Recoverability Ratio (C/Co max)
This is the maximum ratio of the concentration of tracer material recovered in the groundwater samples to the tracer concentration initally added to the system and gives an important indication of the loss/restriction of the tracer material by interactions in the septic tank and saturated zone.

(iv) Background Tracer Concentrations
The concentration of the tracer detected in the groundwaters before tracer addition is referred to as the background tracer concentration. This is an important measurement as it dictates the clarity of the breakthrough curve, the quantity of tracer material required and the sensitivity of analytical equipment needed for tracer detection.

The discussion is formated as follows:

4:6.1 The Suitability of the Various Tracer Materials to Monitor the Movement of Septic Tank Effluents to Groundwaters:
- 4:6.1.1 Optical Brightener
- 4:6.1.2 Fluorescent Dyes
- 4:6.1.3 Ions
- 4:6.1.4 Bacteria

4:6.2 Summary.

4:6.1 The Suitability of the Various Tracer Materials to Monitor the Movement of Septic Tank Effluents to Groundwaters

4:6.1.1 Optical Brightener

The results of the leucophor PBS optical brightener trace are presented in Table 4.5.1 (p264). A high degree of background fluorescence was recorded in the detectors from the monitoring boreholes at sites 1 (B1 and B2). Although an increase in fluorescence was noted after the tracer addition it was not sufficient to enable an accurate interpretation of the test results. Similarly, at site 2, positive background fluorescence was recorded in the monitoring boreholes (B3, B4 and B5). However, there was no increase in fluorescence after the addition of the leucophor. No background fluorescence or breakthrough of the tracer dye was noted at site 3 (B6, B7 and B8) where it is likely that the dye was rapidly adsorbed onto organic material in the loamy overburden material.

The results indicate that the optical brightener trace was unsuccessful due to high background fluorescence in the polluted groundwaters (site 1) and the complete immobilisation of the dye by interactions within the septic tank and the unsaturated zone (sites 2 and 3). These findings are similar to reports by previous investigators who found that the use of optical brighteners in soil - groundwater studies is complicated by the high affinity of leucophor for organic matter (Davis et al, 1984) and the high and variable background concentrations present, especially in polluted groundwater systems (Glover, 1972).

4:6.1.2 Fluorescent Dyes

(i) Sodium Fluorescein

Figures 4.5.1 to 4.5.3 (p265) present the breakthrough curves for the sodium fluorescein trace. Successful traces were recorded at sites 1 and 2. The recorded breakthrough time was 32 hours at both B1 and B2 with breakthrough peaks occurring at 72 hours (Figure 4.5.1). The breakthrough times at site 2 were 32 hours and 40 hours for B3 and B5 respectively, with the peaks occurring at 112 hours in both of the monitoring boreholes (Figure 4.5.2). Table 4.5.2 (p285) shows the calculated C/Co max ratio for the trace at both sites. No breakthrough of the tracer dye was recorded at site 3 (Figure 4.5.3).

The recoverability of the dye in the groundwaters at sites 1 and 2 was high. Ratios of up to 0.213 were recorded at site 1 (B1) with a corresponding ratio of 0.127 at site 2 (B3). This result suggests that loss of the dye by chemical/biological interactions in the septic tank and overburden material was low. However, a marked

reduction in the dye recovered in the groundwater samples with increasing distance from the septic tank systems may indicate a significant restriction or decay of the dye in the saturated subsurface materials. This is clearly demonstrated in the results presented in Table 4.5.2 (p285):

	C/Co max		C/Co max
Site 1 (B1 to B2)	0.213	->	0.072
Site 2 (B3 to B5)	0.127	->	0.039

The reduction in concentration in transit between monitoring boreholes may be due to a combination of loss by dilution, chemical decay and microbial transformation. Microbial transformations and breakdown of the dye in soil and groundwaters have previously been reported as a significant factor in the loss of sodium fluorescein from a system (Knuttson, 1968).

The high and variable levels of sodium fluorescein recorded in the groundwater samples at site 1 (up to 289 µg/l) resulted in some difficulty in the interpretation of the test results and raise some concerns about the use of the dye as a tracer in heavily polluted soil groundwater systems. These results are in agreement with findings by Reynolds (1966), Knuttson (1968), Smart and Laidlaw (1977) and Davis et al (1984), all of whom report difficulty in interpreting tracer results due to high background concentrations.

The failure of the trace at site 3 (Figure 4.5.3, p265) was probably due to the high adsorption losses in the loamy overburden material, in addition to chemical decay and microbiological - related losses.

In summary, the use of sodium fluorescein as a septic tank effluent tracer was sucessful at sites 1 and 2. However, because of adsorption losses in more complex overburden material (site 3) it's use as tracer of the migration of septic tank effluents to groundwater is limited.

(ii) Rhodamine B

The breakthrough curves for the rhodamine B trace are presented in Figures 4.5.4 to 4.5.5 (p266). Successful traces were again recorded at sites 1 (Figure 4.5.4) and 2 (Figure 4.5.5). Breakthrough times at both sites were similar to those observed for the sodium fluorescein trace. In all boreholes (B1 to B5) the first appearance of the dye was recorded at 32 hours with breakthrough peaks occurring after 48 hours in B1, B2 and B3, and 112 hours in B5. Again, no breakthrough of the rhodamine B tracer dye was recorded at site 3.

The concentration of tracer detected in the monitoring boreholes at sites 1 and 2 was very low ranging from 5 to 9 µg/l. This result suggests that losses of the dye in

the septic tank and overburden materials were high and may have been due to adsorption and to chemical decay of the dye in the septic tanks and the overburden material. High adsorption losses of rhodamine B are well documented in the literature (Reynolds, 1966; Knuttson, 1968; Smart and Laidlaw, 1977 and Davis et al, 1984). The large losses are reflected in the C/Co max ratio calculated from the breakthrough curves at sites 1 and 2 (Table 4.5.2, p285):

		C/Co max
Site 1	B1:	0.00092
	B2:	0.00092
Site 2	B3:	0.00045
	B5:	0.00025

Despite these large losses it was possible to obtain clear, interpretable breakthrough curves because of the low background concentration of the dye. The results are in agreement with reports by Smart and Laidlaw (1977) who found that rhodamine B, together with rhodamine WT, has the lowest minimum detectability of all the tracer dyes.

Figures 4.5.29 and 4.5.30 (p275) present a comparison of the breakthrough curves for sodium fluorescein and rhodamine B. The curves demonstrate that the migration patterns of both dyes were similar in the permeable subsurface materials at sites 1 and 2. However, both dyes were restricted from migrating through the loamy overburden at site 3. In contrast to the results observed for the sodium fluorescein dye, the rhodamine concentrations recorded in the monitoring boreholes did not show a significant decrease with increasing distance from the septic tank treatment systems. This is again demonstrated in the recoverability ratios calculated from the breakthrough curves (Table 4.5.2):

	C/Co max		C/Co max
Site 1 (B1 to B2)	0.00092	->	0.00092
Site 2 (B3 to B5)	0.00045	->	0.00025

These results confirm observations by Knuttson (1968) who reported that rhodamine B was more useful as a groundwater tracer than sodium fluorescein because of its lower chemical decay rate and greater resistance to microbiological breakdown.

In summary, the low background concentrations of rhodamine B recorded in groundwaters, together with its high detectability, enabled very sensitive analysis at sites 1 and 2. However, the large adsorption losses in the septic tank and overburden materials restrict the use of the dye in certain soil/overburden and hydrogeological

conditions (site 3) and thus its application as a tracer of the potential movement of septic tank effluents to groundwater systems is limited.

4:6.1.3 Ions

(i) Electrical Conductivity

The breakthrough curves for the conductivity measurements following the addition of the mixture of ion tracers are presented in Figures 4.5.6 to 4.5.8 (p267). A clear increase in electrical conductivity readings was noted in the monitoring boreholes at site 1 (Figure 4.5.6), with breakthrough times of 32 hours at B1 and 40 hours at B2. These breakthrough times are similar to those observed for both of the fluorescent dye tracers. No clear breakthrough was observed at either sites 2 or 3. A sharp decrease in the measured conductivity was recorded in the groundwater samples at all three sites between 64 and 96 hours after tracer addition. This reduction is, however, attributed to instrumental error rather than a sudden decrease in the electrical conductivity of the groundwater samples as a result of tracer material movement.

In summary, the use of conductivity measurements to monitor the movement of ion tracers through the unsaturated zone to groundwater was largely unsuccessful. This can be attributed to differences in the rates of movements of the relative ions through the overburden material. As expected, the most notable breakthrough curve occurred at site 1 (Figure 4.5.6) where, because of the absence of an unsaturated zone, the ion constituents migrated at approximately equal rates. At sites 2 (Figure 4.5.7) and 3 (Figure 4.5.8), breakthrough curves were difficult to interpret as no clear breakthrough times or peaks were recorded.

(ii) Nitrate

The breakthrough curves for the nitrate ion tracer are presented in Figures 4.5.9 to 4.5.11 (p268). Successful traces were recorded at all three sites. Breakthrough at site 1 (Figure 4.5.9) was recorded after 24 hours in B1 and 32 hours in B2 with maximum concentrations occurring after 32 and 64 hours respectively. Similarly, at site 2, the first appearance of the nitrate was recorded at 32 hours in boreholes B3 and B5, with maximum concentrations detected in both at 48 hours (Figure 4.5.10). The breakthrough of the nitrate ion at site 3 was observed at 40 (B6 and B7) and 48 hours (B8) with corresponding breakthrough maxima at 48, 56 and 64 hours respectively (Figure 4.5.11). Background nitrate concentrations were variable ranging from 0.8 mg/l NO_3 - N at site 1 to 4.7mg/l NO_3 - N at site 2. The tracer quantity used at all three sites gave a sufficient rise above background levels for a clear

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breakthrough peak to be observed and resulted in an increase in nitrate concentration of 2 to 4 mg/l.

Table 4.5.2 (p285) presents the calculated recoverability ratios (C/Co max) for the nitrate tracer at the three test sites. The calculated values demonstrate that the recoverability of the ion at sites 1 and 2 was significantly lower than recorded for many of the other ions or the sodium fluorescein dye. However, unlike all the other tracer materials used, no decrease in the ratio was observed with increasing complexity of overburden material from site 1 to site 3:

	C/Co max
Site 1 (B1)	0.019
Site 2 (B5)	0.024
Site 3 (B8)	0.028

The result demonstrates the 'conservative ' nature of the ion in soil and groundwater systems and suggests that the loss of the nitrate ion occurred primarily within the septic tanks.

Figures 4.5.31 to 4.5.33 (p276) present a comparison of the breakthrough curves for nitrate and sodium fluorescein. Breakthrough times for both tracers at sites 1 and 2 were similar but in both cases the conservative ion tracer reached a maximum concentration (breakthrough peak) in the boreholes significantly before the dye. This result highlights the different migration rates of the two tracer materials and raises serious doubts about the use of dyes to monitor the movement of the more conservative constituents of septic tank effluent through soils.

In summary, the nitrate ion was shown to migrate from the septic tanks to the groundwater monitoring boreholes downgradient of the treatment systems at all three sites. The results indicate that the nitrate ion could be successfully used to monitor movement of the ion constituents of septic tank effluent in a range of soil/overburden types and hydrogeological conditions.

(iii) Bromide

The breakthrough curves for the bromide trace are presented in Figures 4.5.12 to 4.5.14 (p269). Breakthrough was again observed in all three sites. The bromide ion was recorded in all the monitoring boreholes at sites 1 and 2 after 32 hours reaching a peak concentration at 24, 32, 48 and 56 hours in B1, B2 (Figure 4.5.12), B3 and B5 (Figure 4.5.13) respectively. At site 3 (Figure 4.5.14) breakthrough was recorded at 64 hours with peak concentrations recorded in all three boreholes after 80 hours. Background concentrations of bromide detected in all the monitoring boreholes were either very low or absent. This permitted accurate detection

at low concentrations and a distinct breakthrough curve was obtained at each site.

The recoverability ratios were similar at all three sites, illustrating that the ion remains relatively unchanged in its passage through the soil/overburden material:

	C/Co max
Site 1 (B1)	0.083
Site 3 (B8)	0.023

A comparison of the bromide and sodium fluorescein breakthrough curves are presented in Figures 4.5.34 to 4.5.36 (p277). Breakthrough times of the two tracers were similar but the bromide tracer reached a peak in the monitoring boreholes significantly before the dye. These results are similar to comparisons between nitrate and sodium fluorescein traces and again illustrate the different migration rates of the dyes and ions in soil/overburden materials.

Figures 4.5.40 to 4.5.42 (p279) present a comparison of the breakthrough curves obtained for bromide and nitrate. At site 1 (Figure 4.5.40) the breakthrough curves were very similar with peaks occurring at 32 hours. Similarly, at site 2, the breakthrough time for both ions was 32 hours. These results are in agreement with reports by Saffigna and Keeney (1977) who found that the migration rates of the bromide and nitrate ions in soils were similar. However, the migration patterns of the two ions were significantly different at sites 2 (Figure 4.5.41) and 3 (Figure 4.5.42). For example at site 3 (Figure 4.5.42) the nitrate ion initially migrated more rapidly than bromide but its breakthrough curve was more elongated indicating possible interactions with soil/overburden microorganisms. In contrast, the bromide ion produced a much tighter breakthrough curve suggesting little interaction with overburden material. This result is contrary to reports by Smith and Davis (1974) who found that the migration of bromide through soils was marginally faster than that of nitrate.

In summary, the bromide ion was shown to migrate through the soil/overburden materials at all three sites demonstrating its usefulness as a tracer in a range of soil type and hydrogeological situations. The low or absent background concentrations permitted clear delineation and interpretation of breakthrough curves. The ion therefore has considerable potential as a tracer in monitoring the migration of the ion constituents of septic tank effluents to groundwater sources.

(iv) Chloride

The breakthrough curves for the chloride ion tracer are presented in Figures 4.5.15 to 4.5.17 (p270). Clear breakthrough of the tracer was observed at sites 1 (Figure 4.5.15) and 2 (Figure 4.5.16) demonstrating the high mobility of chloride in permeable overburden material. Breakthrough peaks were observed after 32

hours in the monitoring boreholes B1, B2 and B3 and at 48 hours in B5. The recoverability of the chloride tracer in the groundwaters at sites 1 and 2 was significantly greater than either nitrate or bromide. This is illustrated in the recoverability ratios calculated from the breakthrough curves (Table 4.5.2, p285) and suggests that the chloride ion is much less susceptible to losses/restrictions at these two sites than either of the other conservative ions (nitrate and bromide):

	C/Co max
Site 1 (B1)	0.225
Site 2 (B5)	0.132

No clear breakthrough of the chloride tracer was observed at site 3 (Figure 4.5.17, p270) due to the high and variable background concentrations of chloride encountered in the groundwaters (fluctuations of up to 10 mg/l). This observation is in agreement with previous reports by Davis et al (1984) and Saffinga and Keeney (1977) who found that the high background concentration of the chloride ion is a major drawback to its effective use as a soil - groundwater tracer. It is noted that the recoverability ratios calculated for the tracer at this site (Table 4.5.2) were similar to those recorded for the nitrate and bromide ions i. e. 0.052, 0.026, and 0.013 for B6, B7 and B8 respectively. However, because the background concentration of chloride was significantly higher than that detected for nitrate or bromide, a clear, recognisable breakthrough curve could not be produced. From the results it was estimated that a recoverability ratio in the region of 0.1 would be necessary to produce a clear breakthrough. The large volume of tracer material needed to achieve this would, however, be far in excess of the concentrations encountered in the septic tank effluent and as such the use of the chloride ion as a tracer of septic tank effluent movement is restricted to sites with low background concentrations.

(v) Sodium

Figures 4.5.18 to 4.5.20 (p271) present the breakthrough curves for the sodium ion trace. Successful traces were recorded at sites 1 and 2. Breakthrough of the ion was observed in the boreholes at site 1 at 32 hours when maximum concentrations were also recorded (Figure 4.5.18). At site 2 breakthrough of the ion was recorded significantly later in the monitoring boreholes than at site 1, suggesting some interaction with the sand and gravel overburden. The ion was first observed in B3 and B5 at 48 and 62 hours respectively, with corresponding maximum concentrations at 56 and 80 hours (Figure 4.5.19).

Table 4.5.2 shows that the recoverability of the sodium tracer at site 1 was significantly higher than that recorded for the other tracer materials, suggesting that

losses within the septic tank were minimal. However, the recoverability ratio (C/Co max) decreased sharply at site 2 demonstrating a marked loss of the tracer material in the overburden material probably by adsorption and ion exchange reactions (Table 4.5.2):

Site 1	C/Co max		Site 2	C/Co max
B1	0.320	->	B3	0.03
B2	0.390	->	B5:	0.02

No clear breakthrough of the ion was observed at site 3 (Figure 4.5.20) although a large fluctuation (up to 11.0 mg/l) in the sodium concentrations was noted in B6. This was probably due to the complete immobilisation of the ion by adsorption and ion exchange mechanisms in the loamy overburden material.

(vi) Potassium

The breakthrough curves for the potassium ion tracer are presented in Figures 4.5.21 to 4.5.23 (p272). A successful trace was again recorded at site 1. Breakthrough of the tracer at the monitoring boreholes B1 and B2 was recorded at 24 hours with maximum concentrations occurring at 24 and 32 hours respectively (Figure 4.5.21). The recoverability of the ion in the groundwaters was high ranging from 0.120 to 0.121 (Table 4.5.2, p285). These ratios are, however, significantly lower than those calculated for sodium ion tracer at the same site. This may be attributed to the preferential sorption of potassium to organic matter within the septic tank.

As was noted for the sodium ion the recoverability of the potassium was high at site 1 but decreased sharply at site 2 due to immobilisation of the anion in the overburden material (Table 4.5.2):

Site 1	C/Co max		Site 2	C/Co max
B1	0.121	->	B3	0.003

No clear breakthrough curves were observed at sites 2 and 3 (Figures 4.5.22 and 4.5.23). Concentrations detected in the monitoring boreholes at site 2 fluctuated between 4.5 and 5.5 mg/l potassium. It could not, however, be conclusively stated that the increase in concentration noted in the boreholes at 40 to 48 hours represented a breakthrough, since fluctuations in excess of 1 mg/l in the potassium concentration had been noted during a previous investigation involving regular monitoring of the boreholes (Chapter 3). Large fluctuations in potassium ion concentrations were detected at site 3 (1.9 to 3.1 mg/l). Again, no clear breakthrough of the tracer could be identified (Figure 4.5.23). It is likely that the ion became tightly fixed to soil matrices

in the overburden materials at sites 2 and 3 thereby preventing its migration to groundwater.

It is, however, worth noting that the results presented in Chapter 3 clearly demonstrate that leaching of potassium did occur at site 2, resulting in potassium concentrations in the groundwater monitoring boreholes B3 and B5 which were significantly greater than background levels detected in the area (C1). It is possible that this is a long - term cumulative process where the potassium in the percolating effluent is adsorbed until no other exchange sites are available and leaching then occurs. The single ' slug ' of potassium tracer used may not have been sufficient to exhaust the available exchange sites and consequently no breakthrough of the tracer was observed.

In summary, excessive adsorption losses of the potassium ion in the septic tank and the unsaturated zone mitigate against it's use as a tracer in monitoring the movement of septic tank effluent to groundwater systems.

4:6.1.4 Bacteria

The results of the two bacterial traces are presented in Figures 4.5.24 to 4.5.28 (p273 and 274). The tracer bacteria displayed a more complex migration pattern than the conservative ion or dye tracers. Although breakthrough times for both of the bacterial species used were similar to those observed for many of the other tracer materials, the breakthrough peaks and migration patterns recorded were significantly different.

(i) *Escherichia coli*

The breakthrough curves for the *Escherichia coli* bacterial tracer are presented in Figures 4.5.24 and 4.5.25 (p273). Successful traces were recorded at sites 1and 2. The bacteria were detected in the monitoring boreholes B1 and B2 at 36 hours with maximum numbers recorded at 72 and 84 hours (Figure 4.5.24). At site 2 breakthrough was observed at 48 hours and 144 hours in B3 and B5 respectively with peaks in the numbers isolated occurring in both boreholes at 180 hours (Figure 4.5.25). No breakthrough of the bacteria was recorded in site 3 probably due to the complete immobilisation of the organisms by filtration/adsorption mechanisms in the sandy loam overburden material.

The recoverability of the bacteria at site 1 was low indicating a significant reduction in the organisms possibly within the septic tank. This is reflected in the recoverability ratio (C/Co max) values calculated for the monitoring boreholes B1 and

B2 (Table 4.5.2, p285):

Site 1	C/Co max
B1	0.02
B2	0.02

A sharp reduction in the numbers recovered was noted from site 1 to site 2. This reduction in the numbers of bacteria isolated was probably due to high adsorption and filtration removal of the organisms as they percolated through the overburden material. The C/Co max values calculated from the breakthrough curves at boreholes B3 and B5 (Table 4.5.2) clearly demonstrate this reduction:

Site 1	C/Co max		Site 2	C/Co max
B1	0.02	->	B3	0.00016
B2	0.02	->	B5	0.000012

The results also illustrate the high level of accuracy possible in a bacterial trace i. e. a recoverability ratio as low as 0.000012 can still produce a recognisable breakthrough peak. This contrasts sharply with the results obtained for chloride ion trace (4:6.3) when it was noted that a recoverability ratio in the region of 0.1 was necessary to produce a clear, interpretable breakthrough curve.

The initial breakthrough of the bacteria at site 2 (Figure 4.5.25, p273) was recorded in the monitoring well B3 at 48 hours. However, a second and significantly larger breakthrough was noted at 144 hours following a period of heavy rainfall (Figure 5.4.54, p284). Mobilisation of bacteria during or after periods of heavy or increased rainfall is well documented in the literature (Patterson et al, 1971; Bitton and Gerba, 1984; Drew, 1987 and Henry, 1988) and is attributed to a decrease in the soil's ionic strength with increased soil moisture status, which in turn reduces the soil's ability to retain the adsorbed organisms. They are subsequently released and subjected to physical ' flushing ' through the soil profile by the percolating rainwater. The results validate those presented in Chapter 3 which indicated that bacterial pollution of groundwaters by septic tank effluents can reach a peak after periods of heavy or prolonged rainfall.

The reasons for the failure of the trace at site 3 are unclear. The type of unsaturated overburden material present at this site has previously been shown to be an effective barrier to the movement of bacteria contained in domestic wastewater (McCoy and Ziebell, 1975 and Bitton and Gerba, 1984). It is possible that the initial injection concentration of the bacterial tracer was insufficent to overcome the filtration and adsorption losses in the biological mat and overburden material (the injection concentration of the *Bacillus globigii* was 100 times greater than that used for the

Escherichia coli tracer). This is contrary to reports by Hagedorn et al (1978) who used an antibiotic resistant *Escherichia coli* to successfully trace septic tank effluent through soils. However, that particular study was carried out under saturated soil conditions which would explain the success of the trace.

Alternatively, the tolerance of the bacteria to the streptomycin antibiotic may have been altered or reduced by interactions with other bacteria within the septic tank itself or with the indigenous soil microorganisms. This has been previously reported by a number of authors (Keswick et al, 1982; Bitton and Gerba, 1984 and Lewis et al, 1982). Finally, it should be noted that the *Escherichia coli* bacterial strain used in the study was a much cultured laboratory strain the viability of which may have been significantly reduced in the hostile soil/overburden environment.

(ii) Bacillus globigii Endospores

The breakthrough curves for the *Bacillus globigii* bacterial tracer are presented in Figures 4.5.26 to 4.5.28 (p274). The spores migrated from the septic tanks through the soil/overburden to the groundwater at all three sites. Breakthrough of the bacteria was observed in all monitoring boreholes at 48 hours. Large numbers of the bacteria (up to 2.7×10^6 c. f. u.'s/100ml) were isolated in the boreholes at site 1 (Figure 4.5.26) where peak concentrations were noted at 84 hours. Similarly, at site 2, large numbers of the organisms were isolated from the monitoring boreholes B3 and B5 (Figure 4.5.27) where peaks in the numbers isolated were recorded at 156 hours and 276 hours respectively. As stated earlier in Chapter 2 and 3, this can be attributed to the absence of an effective unsaturated zone (site 1) and the large pore size and low organic matter content of the overburden material which minimise filtration and adsorption (site 2). Breakthrough at site 3 was also recorded at 48 hours with peaks in the numbers isolated occurring at 156 hours.

Removal of the bacteria in the septic tank and overburden material is again demonstrated in the low recoverability ratios recorded and the decrease in the C/Co max ratio with increasing complexity of overburden material from site 1 to site 3 (Table 4.5.2, p285):

Site 1	C/Co max	->	Site 2	C/Co max	->	Site 3	C/Co max
B1	0.0054		B3	0.0000068		B6	0.0000015
B2	0.00036		B5	0.0000058		B8	0.0000007

Recoverability ratio values for the two monitoring boreholes site 1 were 0.0054 and 0.0036 (B1 and B2 respectively) indicating significant bacterial removal, possibly within the septic tank itself. The ratios were significantly lower than those recorded for the *E.coli* bacterial tracer. This result was somewhat surprising given the fact that a

very large percentage of the bacillus organisms were in spore form and were thus theoretically more tolerant to adverse environmental conditions. It is possible that on entering the organic rich septic tank some of the organisms reverted to vegetative forms and were thus not detected in the heat treated water samples. Alternatively, it may be possible that the spores of the bacteria, which are considerably smaller than the vegetative cells of the bacillus or the *E. coli* tracer, migrated through smaller cracks and crevices in the fissured bedrock thereby bypassing the monitoring borehole. Migration of bacteria in specific zones within the unsaturated and saturated zone is well documented in the literature (McCoy and Ziebell, 1975; Bitton and Gerba, 1984 and Henry, 1987) and was noted in earlier investigations at the test sites (Chapters 2 and 3).

The large numbers of the organism which were recorded at sites 2 and 3 may be attributed to the smaller size of the organism which reduced the number of organisms removed by filtration in the soil/overburden material. The high numbers may also be due to the larger injection concentration of the bacillus organism i. e. the injection concentration was over 100 times greater than that of the *E.coli* tracer. The breakthrough curves at both site 2 (Figure 4.5.27, p274) and site 3 (Figure 4.5.28, p274) display distinct peaks and troughs and differ considerably from the more regular curves obtained for the dye and ion tracers. The peaks and troughs suggest more complex interactions between the bacteria and the overburden material. This is possibly due to the immobilisation and subsequent re - release of bacteria in response to changes in soil moisture status e. g. percolating water following rainfall events. This was confirmed on examination of the rainfall events which occurred during the trace (Figure 4.5.54, p284).

The reason for the significant difference in the breakthrough peaks in boreholes B3 and B5 (Figure 4.5.27) is unclear but it may be related to the relatively tight nature of the limestone bedrock at this site (Appendix B 1, p330). In contrast, it is noted that breakthrough peaks at the monitoring boreholes B6 and B8 occurred simultaneously. This result reflects the high mobility of the bacteria in the fissured/weathered subsurface material and reaffirms observations in Chapter 3 that immobilisation of the effluent bacteria occurs mainly in the unsaturated overburden materials, with minimal restriction in the underlying saturated permeable layers.

A comparison between the breakthrough curves for sodium fluorescein and *Bacillus glogigii* at the three test sites (B1, B3 and B6) is presented in Figures 4.5.37 to 4.5.39 (p278). The graphs demonstrate that, with the exception of site 1, the breakthrough patterns of the tracer dye do not accurately represent that of the tracer bacteria. Thus it can be concluded that fluorescent dyes cannot be used to accurately assess the migration of the microbial effluent constituents through soil/overburden materials to groundwater sources. These results are in agreement with previous reports

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by a number of authors (Rahe et al, 1979 and Bitton and Gerba, 1984).

Figures 4.5.43 to 4.5.48 (p280 and 281) present a comparison between the breakthrough curves for nitrate, bromide and *Bacillus globigii* tracers. The results again show that the breakthrough curves of the conservative ion tracers and the bacteria are markedly different. Breakthrough for the conservative ions (nitrate and bromide) occurred at a much faster rate and breakthrough maxima were obtained significantly earlier than for the tracer bacteria. The results suggest that the use of a non - conservative tracer bacteria such as *Bacillus globigii* does not accurately represent the potential migration of the ion constituents of a septic tank effluent to groundwater systems and as such cannot be used in isolation as a tracer of the potential movement of the chemical and biological constituents to groundwaters.

In summary, the results of the trace demonstrate that *Bacillus globigii* can be used effectively to monitor the migration of septic tank effluent in a range of hydrogeological and soil/overburden situations. The small size and superior survival characteristics of the bacillus endospores compared to conventional indicator bacteria may make the tracer more representative of the potential movement of pathogenic bacteria and enteric viruses to groundwaters.

4:6.2 Summary

The results of the study illustrate the heterogeneity of the pollution plume generated by the septic tank effluent. The breakthrough curves observed in the monitoring boreholes for the tracers NO_3, Br, K, Na, fluorescent dyes and bacteria were markedly different in terms of breakthrough times and patterns. It was demonstrated that the tracers, most of which were themselves effluent constituents, behaved differently in the unsaturated zone. This result eliminates the idea of a single homogeneous pollution plume migrating uniformly in the direction of groundwater flow and raises serious doubts about the use of a single tracer material to represent the migration of a complex heterogeneous waste such as septic tank effluent. The results of this investigation suggest that the use of both the bromide ion (or nitrate if conditions are suitable) and *Bacillus globigii* endospores would give a good indication of the potential for chemical and microbiological contamination of groundwater by septic tank effluents in a range of soil types and hydrogeological conditions.

4.7 Conclusions

The main conclusions of this study are as follows:

(i) The optical brightener leucophor PBS was not a suitable tracer of the migration of septic tank effluent through soil to groundwaters. No clear breakthrough of the dye was recorded at any of the test sites. The tracer was unsuccessful due to high background fluorescence in the polluted groundwaters (site 1) and the complete immobilisation of the dye possibly by adsorption processes and chemical/microbiological interactions within the septic tank and in the soil/overburden materials (sites 2 and 3).

(ii) At sites 1 and 2 the fluorescent dyes sodium fluorescein and rhodamine B were successfully traced from the septic tanks to the groundwater monitoring boreholes downgradient of the test systems. Both dyes were, however, found to be unsuitable as tracers of the migration of the septic tank effluent through the sandy loam overburden at site 3. It is likely that both of the dyes were restricted from moving in the loamy overburden by a combination of adsorption, chemical decay and microbial transformations. Thus the use of the fluorescent dyes as effluent tracers is limited to sites where the overburden layer is thin and/or low in organic matter content.

The rate at which the fluorescent dyes migrated through the overburden to groundwaters at sites 1 and 2 differed significantly from that observed for the ion and bacterial tracers. Consequently it is concluded that the movement of the tracer dyes does not accurately represent the migration of the septic tank effluent constituents to groundwater systems.

The use of the fluorescent dye sodium fluorescein is complicated by its high and variable background concentrations especially in polluted groundwaters. In contrast the lower background concentrations of rhodamine B, together with its high detectability, permitted very accurate detection in the groundwater samples. Rhodamine B was, however, shown to be subject to high adsorption losses in both the septic tank and soil/overburden material.

(iii) The potassium and sodium ions traces were unsuccessful. It would appear that the ions are subjected to complex ion - exchange and sorption interactions with the overburden material resulting in breakthrough curves which are unclear and difficult to interpret. Consequently, the cations cannot be used to trace the movement of septic tank effluent to groundwater systems.

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(iv) The chloride ion was successfully traced to the groundwaters downgradient of the septic tank systems at sites 1 and 2 where breakthrough curves were similar to those recorded for the bromide and nitrate ion tracers. However, the use of the ion was complicated by the high and variable background concentrations encountered in the groundwaters at site 3 (Cregg). The quantity of tracer material needed to overcome this would make the trace unrepresentative of concentrations encountered in septic tank effluents and hence the use of the ion in such site conditions is not recommended.

(v) The nitrate and bromide ions migrated through the overburden material to groundwater at all three sites. The results indicate that the nitrate ion could be successfully used to monitor the percolation of the ion constituents of septic tank effluent in a range of soil/overburden types and hydrogeological conditions. In situations where background nitrate levels are high bromide was shown to be a suitable alternative.

(vi) The tracer bacteria displayed a more complex migration pattern than the conservative ion or dye tracers. An increase in the numbers of the organisms recorded in the groundwater samples was noted after periods of increased rainfall. This was probably due to the immobilisation of the organisms by filtration and adsorption mechanisms and their subsequent re - release following changes in the soil moisture status after increased rainfall events.

The *Escherichia coli* bacterial tracer was successfully traced to the groundwater monitoring boreholes at sites 1 and 2. However, no breakthrough of the organisms was observed at site 3 despite the fact the groundwaters at this site had previously been shown to be contaminated by septic tank effluent bacteria (Chapter 3). It is concluded that the tracer bacteria did not accurately monitor movement of effluent bacteria through the sandy loam soil overburden. The bacteria may have been restricted from moving through the overburden by a combination of filtration and adsorption mechanisms. It may also be possible that the antibiotic resistance or viability of the laboratory culture was lost or reduced through interactions with other organisms within the septic tank and the overburden material.

The *Bacillus globigii* spores showed good potential as tracers. The results of the study suggest that the organisms could be used effectively to monitor the migration of septic tank effluent in a range of hydrogeological and soil/overburden conditions. The small size and superior survival characteristics of the organism compared to conventional indicator bacteria would suggest that the spores give a better indication of the potential migration of pathogenic bacteria and enteric viruses to groundwaters.

(vii) The effluent plume migrating from the soil treatment systems formed a complex heterogeneous mixture of chemical and biological constituents which moved through the overburden material to groundwater at different rates. Therefore a single tracer type cannot be used to accurately trace the movement of all the effluent constituents. A combination of chemical and biological tracer materials must instead be used to gain a more accurate representation of the movement of specific pollutant types in the effluent.

The combined use of the *Bacillus globigii* endospores and bromide ion tracers (or nitrate if background concentrations are low) would give an accurate indication of the risk of groundwater pollution from septic tank effluent in a range of soil/overburden types and hydrogeological conditions. The tracers could also be used to assess the suitability of new sites for the disposal of septic tank effluent and in investigations into the sources of pollution incidences involving septic tank systems.

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CHAPTER 5
FUTURE RESEARCH NEEDS

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5:1 Treatment Efficiency in Septic Tanks and Soil Treatment Systems

The results of this study have established that the efficiency of wastewater treatment within the septic tanks is very low and that the attenuation of the effluent constituents in the soil treatment system is incomplete. It was clearly demonstrated that the migration of effluent nutrients and microbial constituents from the treatment system can occur, resulting in the contamination of groundwater with attendant human health hazards. In view of these findings future research is necessary to improve the design performance of both the septic tank and soil treatment systems. Studies on septic tank treatment efficiency should examine the performance of the septic tanks, giving particular attention to size, shape, incorporation of baffles, degree of maintenance and nature and volume of the waste. It should endeavour to maximise treatment efficiency and minimise unit costs. The performance of the soil treatment system could be enhanced by facilitating even distribution of the effluent through the treatment system thereby preventing overloading. Any research in this area should refer to a publication by Otis et al (1974).

In certain situations, such as that described at site 1 (Dromahaire) in this study, the presence of a high water table precludes the effective use of a septic tank treatment system. Research is urgently needed to develop alternative on - site treatment facilities for such areas. The suitability of alternative systems presently on trial in the United States should be assessed in the Irish situation. These would include raised (mound) beds, evapotranspiration, spray irrigation and peat bed systems, or simply the replacement of existing soil with material having more favourable attenuation properties. In the assessment of alternative disposal systems consideration should be given to the treatment efficiency, the longevity and unit cost compared to conventional septic tank soil distribution systems.

It was stated earlier that a large percentage of Irish soils are unsuitable for septic tank disposal systems because of excessive or insufficient permeability. Furthermore much of the Irish midlands and western regions are characterised by a thin soil cover underlain by highly weathered and fractured limestone bedrock. In such situations the use of septic tank disposal systems is questionable due to the inability of the regolith to effectively treat the percolating wastewater. In view of this it would appear that a primary research need is the development of reliable soil suitability indices which could identify the areas where septic tank disposal systems should be replaced by an alternative method of treatment.

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A number of other areas have also been identified where there is considerable scope for future research:

(i) Most of the research to date has focused on assessing the survival and transport of Coliform bacteria (e. g. *Escherichia coli*) with a view to predicting the suitability of various soil types for domestic wastewater disposal. Efforts are now needed to assess the fate of other potentially pathogenic organisms in a range of soil types and their migration patterns from soil treatment systems. This research should include bacterial species such as Salmonella, Shigella and Clostridium. In addition there is a need to investigate the survival and movement of enteric viruses in a range of soil types and septic tank treatment systems. This research is urgently required as it is now accepted that many of the waterbourne disease outbreaks have a viral etiology and their behaviour in subsurface systems may bear no relationship to that observed for enteric bacteria.

(ii) Recent studies have established that the effluent from a septic tank can contain significant quantities of toxic organic chemicals such as trichloroethylene, dichloromethane, benzene, toluene and methylene chloride, some of which are known carcinogens and pose a serious threat to human health (Viraraghavan and Hashem, 1986 and De Walle and Scaff, 1980). It is therefore important that an assessment is made of the degradation and attenuation of such chemicals in a range of soil types with a view to predicting the risk of groundwater contamination resulting from increased usage of septic tank systems.

5:2 The Contamination of Groundwater by Septic Tank Effluent

The results of this study have clearly demonstrated that the groundwaters downgradient of septic tank treatment systems are contaminated by effluent chemical and microbiological constituents. At all three test sites the groundwater was unsuitable for human consumption under the E. C. drinking water standards. In view of these findings it is apparent that a national survey on the nature and severity of groundwater contamination should be conducted in areas using septic tank systems for on - site wastewater disposal. Where groundwater contamination is identified, consideration should be given to the treatment of the water before use in the household. To this end the development of new water treatment and disinfection methods suitable for installation in single homes is essential. The results of this survey should also be used in conjunction with soil suitability assessments (5:1) to identify the areas where septic

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tank systems should be prohibited and replaced by suitable alternatives.

There are a number of additional concerns which require immediate scientific evaluation:

(i) In response to recent findings in the U. S. which have established that the effluent from a septic tank contains high concentrations of toxic organic chemicals (Viraraghaven and Hashem, 1986) and that septic tank systems are contributing to increased levels of these chemicals in drinking water (Burmaster, 1982), efforts should be made to:

(a) Improve and simplify the sampling and analytical methods for the detection of these chemicals in groundwaters

(b) Assess the degree of contamination in Irish groundwaters and the resulting human health risk.

(ii) The results of this study have demonstrated the ability of indicator fecal bacteria to migrate through various soil/overburden types and contaminate the underlying groundwater. Further research is necessary to assess the fate of potentially pathogenic organisms in groundwater systems. This should include an investigation of the persistence and migration patterns of bacterial species such as Salmonella, Shigella and Clostridium and enteric viruses such as Polio and Hepatitis.

(iii) Research should be undertaken to determine the extent of degradation of L. A. S. detergents in septic tank treatment systems and in various soil overburden materials. The persistence of the substances in groundwater systems should also be examined.

5:3 The Use of Tracer Materials to Monitor the Movement of Septic Tank Effluents to Groundwater

The tracer experiments have identified a number of materials suitable for use in a range of soil/overburden conditions and hydrogeological settings. These techniques must now to be applied under field conditions. This would involve tracing from excavated test holes to groundwater sources at the test site e. g. dug or drilled boreholes, springs etc.. It is only through such field based research that the methodologies described can be developed and refined. A number of other areas have also been identified where there is considerable scope for future research:

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(i) The results of this study indicate that the fluorescent dyes, fluorescein and rhodamine B, are not suitable as tracers in some situations, probably because of excessive adsorption losses in the overburden material. Future research should focus on other fluorescent dyes such as pyranine which is reported to be much less suceptible to adsorption losses in the overburden material. The use of rhodamine WT which has the same high detectability as rhodamine B but is much less susceptible to adsorption should also be investigated.

(ii) The bromide ion and *Bacillus globigii* spores were identified as suitable tracers for monitoring the movement of conservative ions and enteric bacteria contained in septic tank effluent. However, in recent times there has been increasing concern regarding the pollution of groundwater by trace organics and enteric viruses. Further research is necessary to determine the potential of the tracers described in this study to accurately monitor the migration of these contaminants. The use of host specific bacteriophages as tracers of virus migration to groundwater should also be investigated.

(iv) The *Bacillus globigii* spores were shown to have good potential as tracers of the bacterial constituents in septic tank effluents. However, further research is necessary in the following areas:

(a) The survival characteristics of the organism in a range of soil/overburden materials must be established

(b) The ability of the spores to revert to vegetative cells in the organic rich septic tank or soil/overburden material must be assessed. It is possible that the incorporation of some specialised growth requirements into the strain may prevent the spores from vegetating, ensuring more accurate results

(c) The suitability of the spores for assessing the potential movement of enteric viruses should be investigated.

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APPENDIX A
THE MOVEMENT AND ATTENUATION OF SEPTIC TANK EFFLUENT IN SOIL TREATMENT SYSTEMS

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A 1

Results of the Chemical and Microbiological Analysis of the Soil Samples at the Two Test Sites on the Various Sampling Dates Between April 1987 and June 1988

Table 1

The Ammonium - N Concentrations (mg/kg) Recorded in the Soil Samples at Site 1 (Dromahaire) on the Various Sampling Occasions (April 1987 to June 1988)

Sampling Station	Sampling Date							
	Apr '87	June	Aug	Oct	Dec	Feb '88	Apr	June
1a	25.8	50.4	56.0	63.6	45.0	88.0	9.0	38.0
1b	24.4	40.6	43.4	41.4	43.2	65.0	16.0	18.0
2a	36.1	35.0	40.6	14.2	30.0	41.0	10.0	16.0
2b	90.0	40.6	30.8	14.2	30.0	42.0	13.0	2.0
3a	60.0	14.0	32.2	16.0	13.8	15.0	21.0	13.0
3b	18.0	15.0	12.6	29.4	12.0	22.0	25.0	10.0
4a	54.0	95.0	78.4	37.8	17.6	86.0	43.0	39.0
4b	54.0	35.0	58.8	21.6	10.2	105.0	66.0	27.0
5a	75.0	40.6	72.8	17.4	40.2	34.0	12.0	20.0
5b	31.8	14.0	53.2	14.4	61.8	55.0	29.9	10.0
6a	114.0	140.0	105.0	42.0	54.6	80.0	31.0	32.0
6b	61.0	44.8	75.6	34.2	53.2	78.0	42.0	29.0
7a	8.1	12.6	4.2	9.6	8.9	1.8	8.0	4.0
7b	8.0	8.2	4.6	3.0	5.2	2.6	21.0	3.0
8a	3.0	14.0	74.2	4.8	3.0	5.3	5.0	3.0
8b	3.4	8.6	60.3	7.2	3.0	7.2	16.0	3.0
9a	1.8	2.8	2.8	4.8	6.6	2.8	5.0	3.0
9b	0.0	4.2	2.8	3.6	4.2	2.5	5.0	2.0
10a	1.0	1.5	1.2	3.2	4.8	1.0	3.1	2.4
10b	0.0	1.0	1.0	2.1	3.2	1.0	2.5	2.0

Table 2

The Ammonium - N Concentrations (mg/kg) Recorded in the Soil Samples at Site 2 (Cregg) on the Various Sampling Occasions (April 1987 to June 1988)

Sampling Station	Sampling Date							
	Apr '87	June	Aug	Oct	Dec	Feb '88	Apr	June
1a	66.0	98.0	44.8	43.2	39.6	43.0	27.0	19.0
1b	36.0	89.0	44.8	38.4	39.6	39.0	17.0	2.0
2a	84.0	84.0	86.8	41.4	17.2	24.0	27.0	26.0
2b	66.0	67.0	79.4	33.0	19.2	41.0	19.0	9.0
3a	8.0	14.0	9.8	6.0	3.0	2.0	1.0	2.9
3b	6.0	11.2	8.4	3.0	3.0	5.0	2.0	2.0
4a	78.0	35.0	5.6	4.2	6.0	2.0	6.0	13.0
4b	16.2	25.2	11.2	5.4	11.9	8.0	3.0	12.0
5a	7.2	22.4	5.6	4.1	3.9	3.0	0.0	1.0
5b	16.2	8.4	4.2	3.6	4.2	2.0	0.0	1.0
6a	6.7	7.0	5.6	8.4	4.6	5.0	2.0	1.0
6b	10.2	7.0	11.2	6.0	3.9	5.0	2.0	1.0
7a	13.6	21.9	19.8	14.2	6.6	15.0	25.0	25.0
7b	12.9	14.2	14.2	12.2	9.0	16.0	23.0	13.6
8a	6.0	8.6	6.3	3.9	2.4	4.0	3.2	3.6
8b	7.8	4.1	4.2	3.2	3.0	4.0	3.2	3.6
9a	5.8	7.6	6.9	4.4	2.1	3.5	3.0	3.2
9b	5.3	4.0	3.9	13.2	2.2	3.1	1.9	2.9
10a	0.0	5.4	4.2	1.7	2.2	2.0	1.0	2.0
10b	0.0	5.6	2.8	2.8	2.2	2.0	1.1	3.0

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Table 3

The Nitrate - N Concentrations (mg/kg) Recorded in the Soil Samples at Site 1 (Dromahaire) on the Various Sampling Occasions (April 1987 to June 1988)

Sampling Station	Sampling Date							
	Apr '87	June	Aug	Oct	Dec	Feb '88	Apr	June
1a	0.0	11.2	14.0	12.4	2.1	5.0	3.0	3.0
1b	0.0	14.7	16.8	20.4	1.2	3.0	4.0	3.0
2a	2.8	4.2	11.2	10.0	1.2	0.0	2.0	1.0
2b	1.2	40.6	42.5	10.6	1.8	0.0	5.0	3.0
3a	6.0	10.5	5.1	10.0	2.4	0.0	3.0	5.0
3b	1.2	11.2	12.6	13.8	2.4	1.0	6.0	14.0
4a	1.0	2.1	7.0	1.2	1.8	2.0	3.0	2.0
4b	1.0	2.8	7.0	1.8	0.9	2.0	5.0	3.0
5a	0.9	14.0	4.2	1.8	1.2	0.0	4.0	1.0
5b	1.3	14.0	2.8	1.8	0.6	0.0	6.0	3.0
6a	1.8	7.0	11.2	10.6	1.8	0.0	5.0	4.0
6b	1.8	11.2	12.6	10.6	1.8	0.0	8.0	13.0
7a	1.8	3.8	8.6	2.4	4.1	2.0	2.0	7.0
7b	3.6	8.4	14.0	5.0	2.9	2.0	4.0	8.0
8a	1.2	7.0	2.8	1.9	3.6	1.0	2.0	3.5
8b	4.4	7.0	8.4	2.4	2.4	1.4	6.0	4.0
9a	1.0	1.4	2.8	1.1	1.2	1.0	1.0	1.0
9b	1.0	2.8	2.8	1.0	1.8	1.2	1.2	1.0
10a	3.1	3.4	2.9	2.5	1.1	2.0	2.3	3.5
10b	2.8	3.5	2.4	2.4	1.0	1.6	2.2	2.9

Table 4

The Nitrate - N Concentrations (mg/kg) Recorded in the Soil Samples at Site 2 (Cregg) on the Various Sampling Occasions (April 1987 to June 1988)

Sampling Station	Sampling Date							
	Apr '87	June	Aug	Oct	Dec	Feb '88	Apr	June
1a	7.8	2.8	8.4	8.0	2.4	1.0	2.0	1.0
1b	8.2	7.0	18.2	8.0	2.4	2.0	4.0	8.2
2a	8.4	2.8	14.0	6.8	1.8	1.0	10.0	3.2
2b	9.0	7.0	35.0	6.8	1.8	1.0	10.0	10.6
3a	8.7	9.8	5.6	3.2	1.2	2.0	3.6	2.4
3b	7.7	14.0	7.0	2.9	1.8	3.0	6.2	4.3
4a	6.6	19.6	5.6	4.3	1.8	3.0	9.6	10.0
4b	15.0	25.2	7.0	3.9	1.8	2.0	12.9	13.0
5a	7.8	14.0	8.4	3.6	1.8	2.0	3.6	2.2
5b	16.2	26.6	11.2	3.5	1.2	3.0	4.2	3.0
6a	7.2	16.8	5.6	3.7	2.4	3.0	3.5	1.9
6b	8.4	43.4	7.0	4.0	1.2	3.0	3.5	3.1
7a	3.8	8.4	4.2	4.1	2.4	4.0	7.9	2.0
7b	9.2	13.1	9.6	3.0	1.2	4.0	10.3	12.0
8a	6.0	10.7	8.2	2.3	1.8	3.0	4.2	2.2
8b	7.8	15.5	14.0	1.4	2.4	3.0	4.2	3.2
9a	3.0	6.0	3.6	2.9	2.1	3.0	6.6	3.1
9b	2.6	7.8	6.1	2.8	1.8	2.5	8.2	9.5
10a	1.0	3.0	2.8	2.1	1.2	1.0	1.6	2.0
10b	1.0	7.1	2.8	1.4	1.8	1.5	2.1	3.0

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Table 5

The Phosphate - P Concentrations (mg/kg) Recorded in the Soil Samples at Site 1 (Dromahaire) on the Various Sampling Occasions (April 1987 to June 1988)

Sampling Station	Sampling Date							
	Apr '87	June	Aug	Oct	Dec	Feb '88	Apr	June
1a	8.8	3.4	5.6	3.7	12.6	13.9	19.6	23.1
1b	5.7	4.5	1.7	6.5	33.0	22.0	13.6	12.3
2a	14.3	2.5	2.6	31.5	11.4	12.9	13.9	6.2
2b	14.6	6.4	1.3	12.5	13.2	14.2	12.6	1.1
3a	8.8	5.5	7.7	12.3	16.3	8.2	11.4	12.2
3b	34.6	6.5	7.3	19.9	15.6	9.7	10.1	13.1
4a	14.7	2.2	1.8	4.0	7.1	13.4	13.2	9.5
4b	17.4	6.7	1.2	11.8	7.9	12.9	9.8	5.3
5a	7.0	3.9	2.6	5.5	13.3	12.6	12.6	4.4
5b	31.7	6.2	1.8	11.9	20.0	13.8	11.3	3.2
6a	5.4	11.2	36.3	15.3	17.8	12.6	30.5	26.4
6b	18.7	15.7	18.4	30.1	18.4	18.4	29.0	17.9
7a	3.9	2.2	2.0	15.8	8.3	5.9	10.9	6.0
7b	3.2	2.0	1.8	4.7	8.3	8.7	3.9	7.5
8a	4.3	2.3	5.2	5.2	2.6	3.8	9.6	8.3
8b	5.4	2.1	4.4	3.8	7.6	5.2	7.8	8.5
9a	2.6	1.3	1.0	1.4	3.9	7.9	8.6	3.4
9b	4.6	1.2	1.0	0.7	5.1	8.2	7.7	2.6
10a	2.0	1.0	1.0	1.2	1.9	2.1	2.2	2.5
10b	1.3	0.5	1.0	0.7	1.6	2.2	2.0	2.7

Table 6

The Phosphate - P Concentrations (mg/kg) Recorded in the Soil Samples at Site 2 (Cregg) on the Various Sampling Occasions (April 1987 to June 1988)

Sampling Station	Sampling Date							
	Apr '87	June	Aug	Oct	Dec	Feb '88	Apr	June
1a	15.8	15.7	40.6	28.7	39.4	25.9	9.5	25.1
1b	11.9	6.9	15.0	39.2	41.9	35.8	8.8	19.3
2a	32.5	26.7	35.5	27.2	13.4	19.4	9.8	31.6
2b	7.0	22.4	28.3	31.0	31.3	36.1	5.5	28.2
3a	2.1	3.3	5.0	1.9	1.9	2.5	1.9	3.3
3b	2.1	1.2	5.0	1.1	1.9	7.9	1.9	3.0
4a	16.4	18.9	22.7	19.9	8.5	11.3	13.4	17.7
4b	12.2	15.7	13.2	13.7	24.3	20.2	12.2	9.3
5a	1.4	0.7	2.0	2.8	1.3	1.2	1.0	3.3
5b	1.3	0.5	1.4	1.9	2.2	1.2	1.0	1.1
6a	5.6	0.0	1.2	1.0	2.1	3.1	2.1	1.9
6b	1.7	0.0	1.1	0.8	2.2	1.2	1.9	1.8
7a	2.5	1.9	10.0	11.7	20.0	6.0	5.6	23.0
7b	2.4	1.5	10.0	8.9	30.6	11.2	1.7	19.5
8a	5.5	3.2	4.6	3.9	1.8	0.9	1.4	2.2
8b	8.0	2.5	3.0	3.1	3.9	0.9	1.2	0.9
9a	2.6	0.5	5.2	3.2	10.5	3.1	1.0	4.0
9b	2.2	0.3	2.9	1.3	6.9	1.0	0.5	2.0
10a	2.6	0.0	0.2	1.0	1.9	0.9	1.0	3.0
10b	2.6	0.3	0.1	1.0	3.0	0.9	1.0	2.7

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Table 7

The Numbers of Total Coliform Bacteria (c.f.u.'s/100 g) Recorded in the Soil Samples at Site 1 (Dromahaire) on the Various Sampling Occasions (April 1987 to June 1988)

Sampling Station	Sampling Date							
	Apr '87	June	Aug	Oct	Dec	Feb '88	Apr	June
1a	1200	1600	350	2400	3000	8000	1300	25000
1b	900	2100	180	360	8000	3600	1400	1000
2a	900	1200	160	1000	7000	4000	2000	14000
2b	860	460	80	900	1100	2500	7000	1000
3a	290	25000	400	6000	800	18000	6000	110000
3b	160	18000	130	400	1000	60000	6000	63000
4a	190	3000	400	1800	20000	1000	300	2000
4b	860	900	210	1300	2200	500	290	1500
5a	1600	1800	230	1600	1000	1000	1800	6700
5b	180	360	130	1100	1000	1500	1200	1600
6a	300	1800	400	3900	1500	10000	3000	90000
6b	600	4800	300	1500	1000	10000	3500	6000
7a	0	300	290	220	3100	3000	1000	1000
7b	0	25	10	130	4800	1400	1000	200
8a	460	160	320	200	3000	3200	2000	8600
8b	360	13	300	190	7000	2600	2100	300
9a	390	11	13	29	2400	300	800	200
9b	130	14	10	16	6000	300	350	100
10a	15	12	18	10	36	8	22	29
10b	13	17	10	0	50	0	12	18

Table 8

The Numbers of Total Coliform Bacteria (c.f.u.'s/100 g) Recorded in the Soil Samples at Site 2 (Cregg) on the Various Sampling Occasions (April 1987 to June 1988)

Sampling Station	Sampling Date							
	Apr '87	June	Aug	Oct	Dec	Feb '88	Apr	June
1a	31000	3200	4300	4400	100000	150000	300	10000
1b	200	610	240	3600	250000	100000	3400	12000
2a	58000	390000	30000	21000	27000	100000	8000	18000
2b	300	290	48000	19000	150000	150000	20000	8500
3a	300	6000	2600	160	3000	1000	100	180
3b	200	0	14	190	2000	2100	900	130
4a	3600	52000	32000	64000	45000	500	4500	15000
4b	210	280	26000	51000	58000	7000	800	10000
5a	3100	4900	3500	3900	4000	3300	29	110
5b	210	280	2600	2900	14000	3300	20	100
6a	2500	180	0	10	1000	900	30	39
6b	100	0	0	21	1100	1100	10	12
7a	2800	16000	2400	3200	26000	3000	3000	15000
7b	300	250	320	2900	30000	40000	2100	1300
8a	200	81	10	15	260	150	10	89
8b	0	32	16	19	210	180	10	32
9a	100	60	90	200	315	80	65	225
9b	85	75	56	140	350	45	30	180
10a	21	24	10	19	33	30	10	60
10b	10	14	10	19	28	9	10	44

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Table 9
The Numbers of Fecal Coliform Bacteria (c.f.u.'s/100 g) Recorded in the Soil Samples at Site 1 (Dromahaire) on the Various Sampling Occasions (April 1987 to June 1988)

Sampling Station	Sampling Date							
	Apr '87	June	Aug	Oct	Dec	Feb '88	Apr	June
1a	390	3000	150	1600	410	390	500	3600
1b	310	1000	170	1200	700	910	480	1900
2a	150	360	60	410	380	160	300	2600
2b	140	210	0	110	720	100	370	9100
3a	300	15000	120	2600	150	290	190	10000
3b	190	12000	80	1300	890	410	190	7200
4a	180	110	110	1100	2100	69	10	270
4b	190	90	90	900	360	11	10	19
5a	1200	280	100	950	120	89	90	110
5b	120	180	90	420	150	100	120	110
6a	600	340	200	720	390	390	200	5000
6b	610	140	40	350	960	310	250	460
7a	120	16	0	100	380	600	160	130
7b	80	0	0	64	460	290	32	36
8a	160	10	40	90	480	590	360	150
8b	90	0	0	51	700	300	91	0
9a	120	0	0	0	180	10	15	16
9b	0	0	0	0	410	15	18	11
10a	10	0	0	0	10	0	25	0
10b	0	0	0	0	14	0	0	0

Table 10
The Numbers of Fecal Coliform Bacteria (c.f.u.'s/100 g) Recorded in the Soil Samples at Site 2 (Cregg) on the Various Sampling Occasions (April 1987 to June 1988)

Sampling Station	Sampling Date							
	Apr '87	June	Aug	Oct	Dec	Feb '88	Apr	June
1a	390	2800	3100	2800	32000	7000	200	3700
1b	20	2500	160	1300	40000	11000	210	2500
2a	1800	6200	3300	4100	5000	1000	800	7000
2b	290	1000	2600	2300	60000	2000	1800	2500
3a	200	0	300	150	13	16	150	14
3b	0	0	10	190	19	11	200	0
4a	2100	8200	1000	3600	3000	5000	4300	3700
4b	29	210	360	3700	10000	7000	700	3100
5a	3400	820	150	150	11	10	10	0
5b	270	13	0	120	21	0	1	0
6a	1700	0	0	0	21	0	10	10
6b	23	0	0	0	22	0	10	0
7a	120	1000	2800	2400	8000	210	2400	1500
7b	9	25	190	210	10000	310	480	1200
8a	100	0	0	0	0	0	0	31
8b	0	0	0	0	0	0	0	11
9a	36	0	0	0	0	0	0	12
9b	0	0	0	0	0	0	0	19
10a	0	0	10	0	0	0	0	10
10b	0	0	0	0	0	0	0	10

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APPENDIX B
GROUNDWATER CONTAMINATION DOWNGRADIENT OF SEPTIC TANK TREATMENT SYSTEMS

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B 1

Borehole Logs

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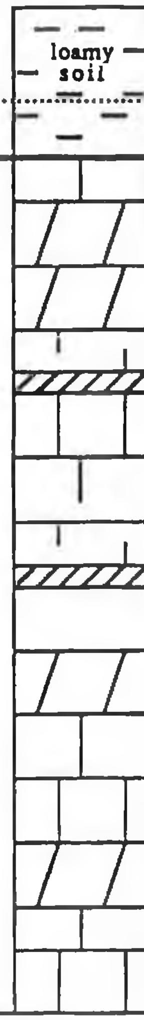
Figure 1
Borehole log - Kilmacowen B1

Drilling method: Rotary	Site : 1 Kilmacowen Location : Co Sligo G 662 307	G.S.I. Ref. No : 88/2

S.W.L	Well construction	Description of strata	Depth (m)	Symbolic log
water table 1.3m	bedrock		0, 1	loamy soil
	55 mm wavin pipeing slotted from 2m	highly weathered grey dolomotised limestone with calcite deposited along the fissures	2, 3	
		dark grey fossiliferous limestone with large fissures and some shales	4	
		highly weathered zone	5	
		moderately fissured fossiliferous limestone with some shales	6, 7	
		highly dolomotised limestone	8	
		weathered limestone with calcite deposition	9	
		dark grey dolomotised limestone with some fossilisation and calcite deposition	10, 11	
		borehole completed	12, 13	

Logged by: H.Henry	Driller: K.Crilly	BOREHOLE No. : B1
Remarks: highly weathered dolomitic lst		

Figure 2
Borehole log - B2 Kilmacowen

Drilling method: Rotary	Site : 1 Kilmacowen Location : Co Sligo G 662 307	G.S.I. Ref. No : 88/3

S.W.L	Well construction	Description of strata	Depth (m)	Symbolic log
water table 1.8m			0 1	loamy soil
	bedrock		2	
	55 mm wavin pipeing slotted from 2m	highly weathered grey limestone dolomotised limestone highly fissured limestone with calcite and iron deposition fissured dolomotised limestone highly fissured lst. with calcite and iron deposits - some shales less fissured limestone with a high degree of fossilisation dark grey weathered limestone with large fissures	3 4 5 6 7 8	
		borehole completed	9 10 11 12 13	

Logged by: H.Henry	Driller: K. Crilly	BOREHOLE No. : B2
Remarks: highly weathered dolomitic lst.		

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Figure 3
Borehole log - B4 Knocknahur

Drilling method: Rotary	Site : 2 Knocknahur Location : Co Sligo G 646 333	G.S.I. Ref. No : 88/4

S.W.L	Well construction	Description of strata	Depth (m)	Symbolic log
water table 5.25m			0, 1, 2, 3, 4, 5	permeable outwash sands and gravels
	bedrock		6	
	55 mm wavin pipeing slotted from 3m	dark grey/black lst. moderately pure with some degree of fissuring&fossilisation some shales present	7, 8	
		tight black limestone with some dolomitisation	9, 10	
		tight black limestone with small number of fissures	11, 12	
		less pure fossiliferous limestone borehole completed	13	

Logged by: H.Henry	Driller: K. Crilly	BOREHOLE No. : B4
Remarks: Pure lst. with some fissuring		

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Figure 4
Borehole log - B5 Knocknahur

Drilling method: Rotary	Site : 2 Knocknahur Location : Co Sligo G 646 333	G.S.I. Ref. No : 88/5

S.W.L	Well construction	Description of strata	Depth (m)	Symbolic log
water table 5.38m			0, 1, 2, 3, 4, 5	permeable outwash sands and gravels
	bedrock			
	55 mm wavin pipeing slotted from 3m	tight dark grey/black limestone moderately pure with some degree of fossilisation and dolomotisation	6, 7, 8	
		tight black limestone some fossils present some shales	9	
		tight black pure limestone with some fissuring	10, 11	
		dolomotised limestone	12	
		borehole completed	13	

Logged by : H.Henry	Driller : K Crilly	BOREHOLE No. : B5
Remarks: pure lst. with some fissuring		

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Figure 5
Borehole log - B6 Cregg

Drilling method: Rotary	Site : 3 Cregg Location : Co Sligo G 653 395	G.S.I. Ref. No : 88/7

S.W.L	Well construction	Description of strata	Depth (m)	Symbolic log
water table 3.05 m			0 1 2	loamy soil
			3 4 5 6	coarse gravel with boulders
	bedrock 55 mm wavin pipeing slotted from 3m	highly weathered and fissured schists tight less weathered schist with some fissures highly weathered and fissured, brittle schist with iron deposition in the larger fissures less weathered zone	7 8 9 10 11 12	
		borehole completed	13	

Logged by: H.Henry	Driller: K. Crilly	BOREHOLE No. : B6
Remarks: highly weathered metamorphic schist		

Figure 6
Borehole log - B8 Cregg

Drilling method: Rotary	Site : 3 Cregg Location : Co Sligo G 653 395	G.S.I. Ref. No : 88/9

S.W.L	Well construction	Description of strata	Depth (m)	Symbolic log
water table 2.70m			0, 1, 2, 3, 4, 5	loamy soil; coarse gravel with boulders
	bedrock 55 mm wavin pipeing slotted from 3m	weathered schist tight-pure schist with some iron and pyrite deposition weathered schist tight pure schist with some small fissures	6, 7, 8, 9, 10	
		borehole completed	11, 12, 13	

Logged by: H.Henry	Driller: K. Crilly	BOREHOLE No. : B8
Remarks: tight metamorphic schist (some fissures)		

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B 2

Gamma Logs

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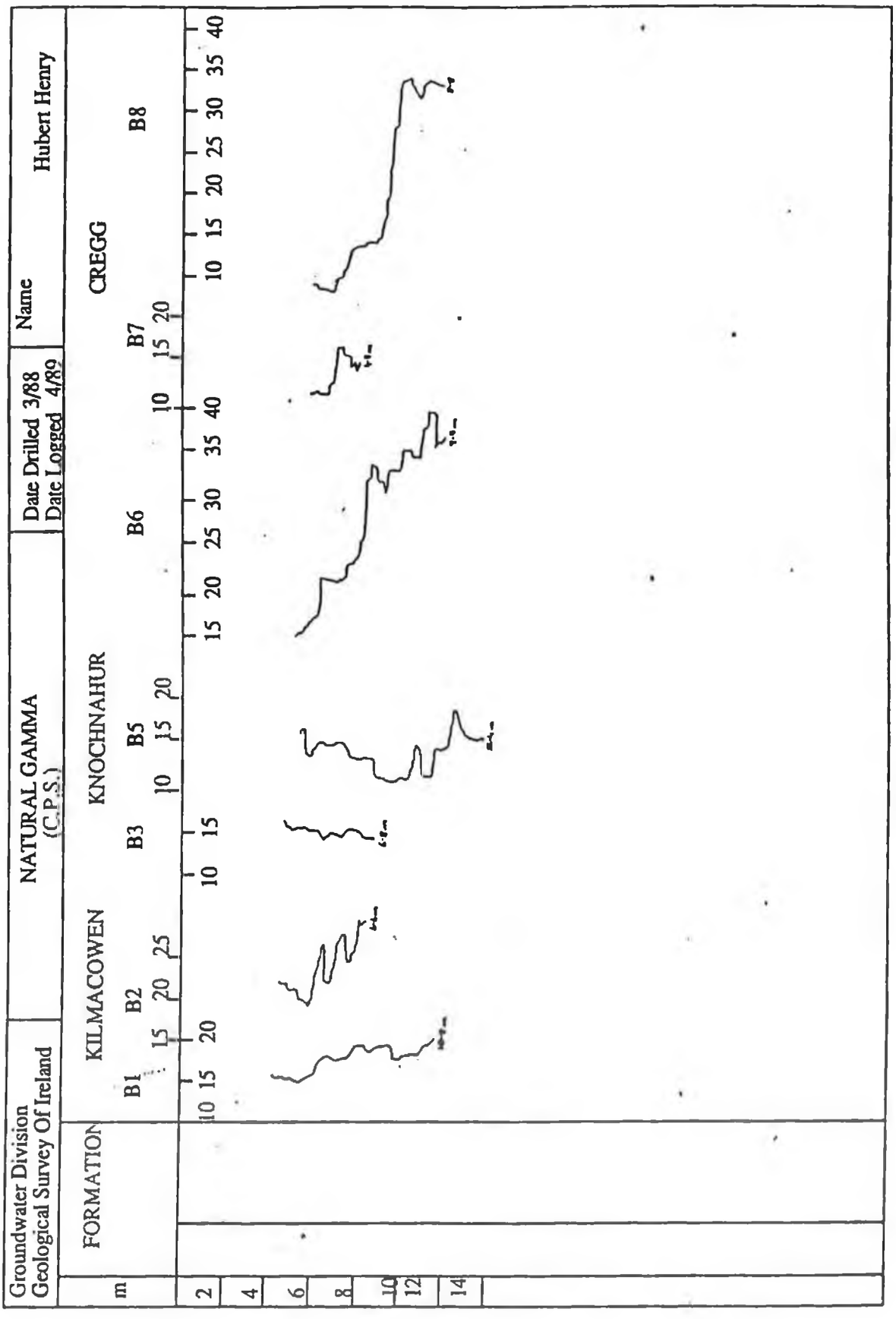
Figure 7

Gamma Logs For The Monitoring Boreholes At The Three Test Sites



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B 3

Changes in the Concentrations of Groundwater Constituents in The Monitoring Boreholes at Sites 2 and 3 During a Pumping Test

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Figure 8
Changes in the pH, nitrate and potassium concentrations with time during a pumping test at site 2 (B4) (pumping rate of 273 litres/hr)

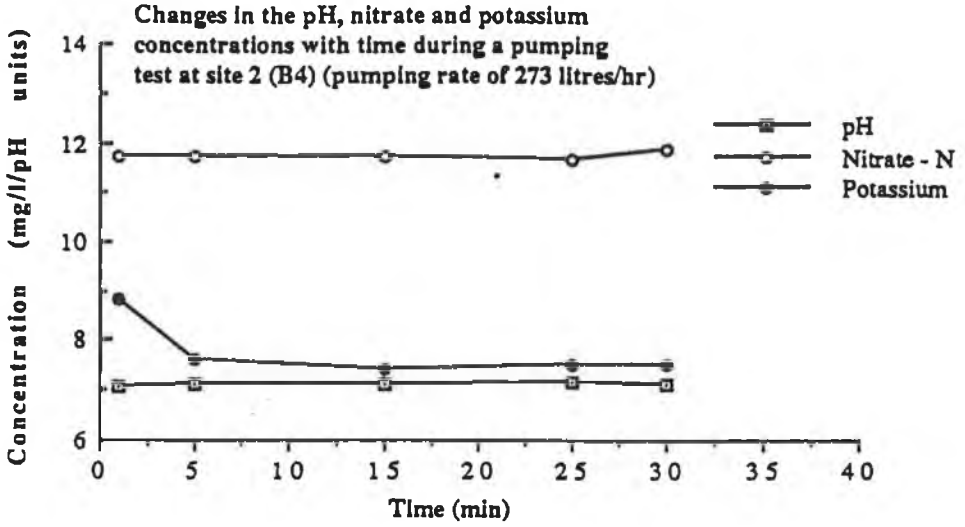


Figure 9
Changes in the pH, nitrate and Potassium concentrations with time during a pumping test at site 3 (B8) (pumping rate of 273 litres/hr)

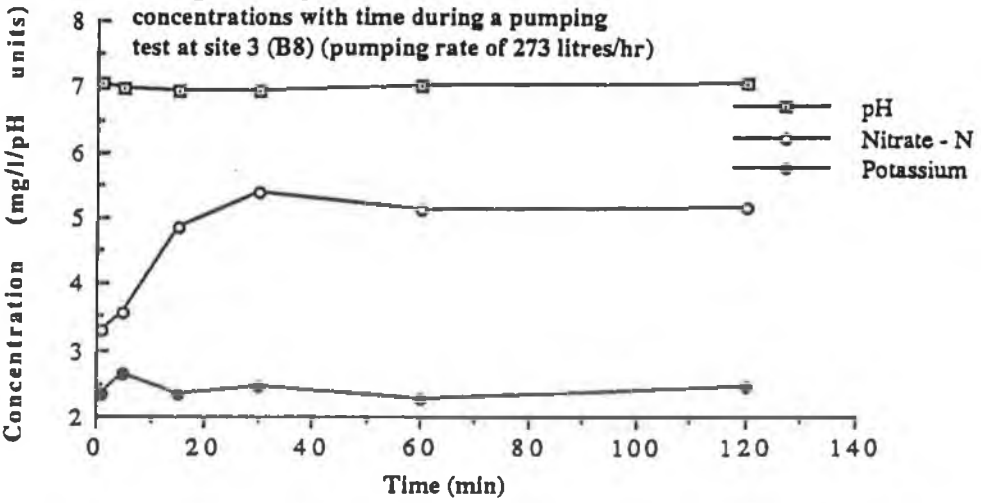


Figure 10
Changes in the conductivity and alkalinity concentrations with time during a pumping test at site 2 (B4) (pumping rate of 273 litres/hr)

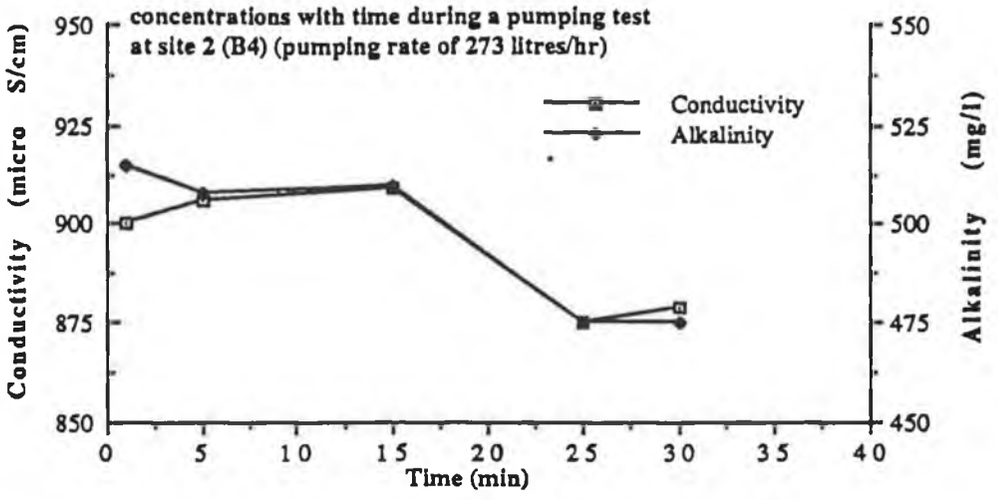


Figure 11
Changes in the conductivity and alkalinity concentrations with time during a pumping test at site 3 (B8) (Pumping rate of 273 litres/hr)

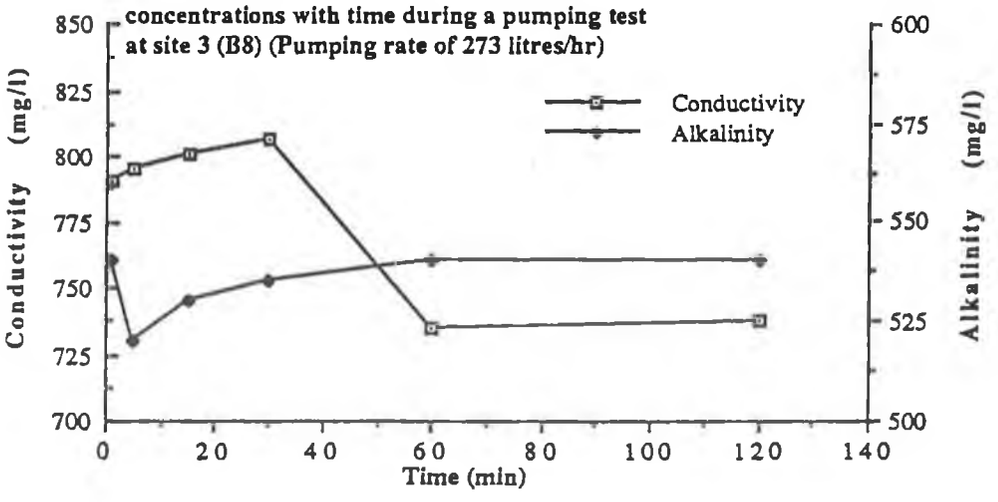


Figure 12
Changes in the sulphate, chloride, and sodium concentrations with time during a pumping test at site 2 (B4) (pumping rate of 273 litres/hr)

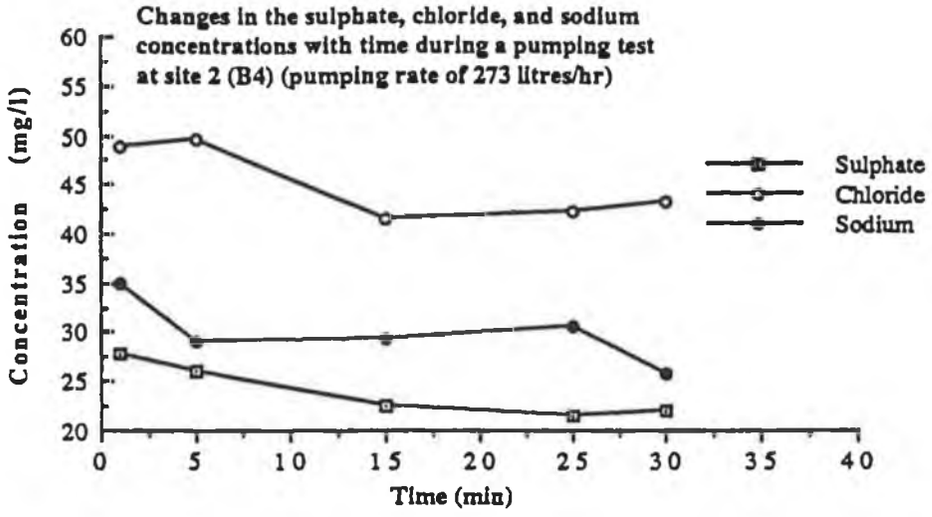
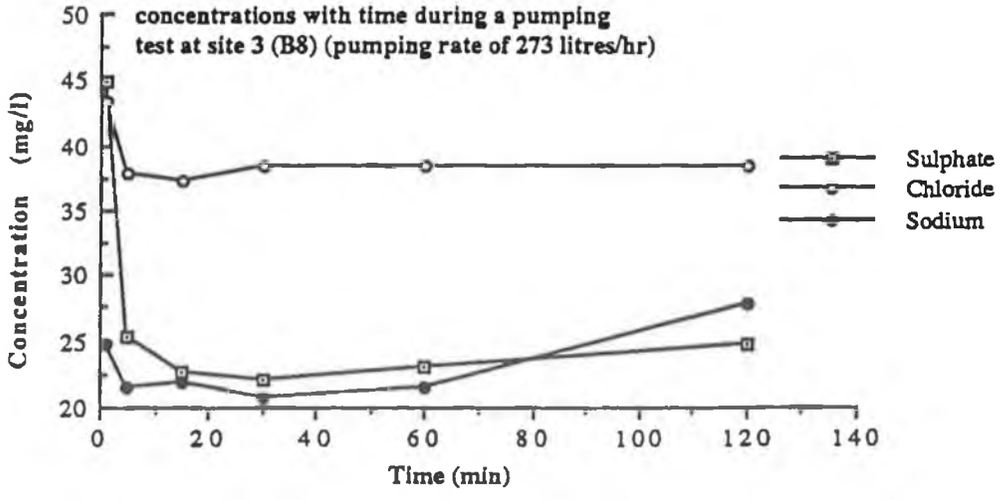


Figure 13
Change in the sulphate, chloride, and sodium concentrations with time during a pumping test at site 3 (B8) (pumping rate of 273 litres/hr)



B 4

Scatter Plots and Simple Regression Lines of the Total Rainfall Preceding Sampling Against Fecal Coliform Numbers Isolated From the Monitoring Boreholes at the Three Test Sites

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Figure 14
Linear regression line fitted to a plot of the total rainfall for the 5 days preceding sampling against the numbers of fecal Coliform bacteria (c.f.u.'s/100ml) isolated from the monitoring borehole B1 (r = Pearson's correlation coefficient)

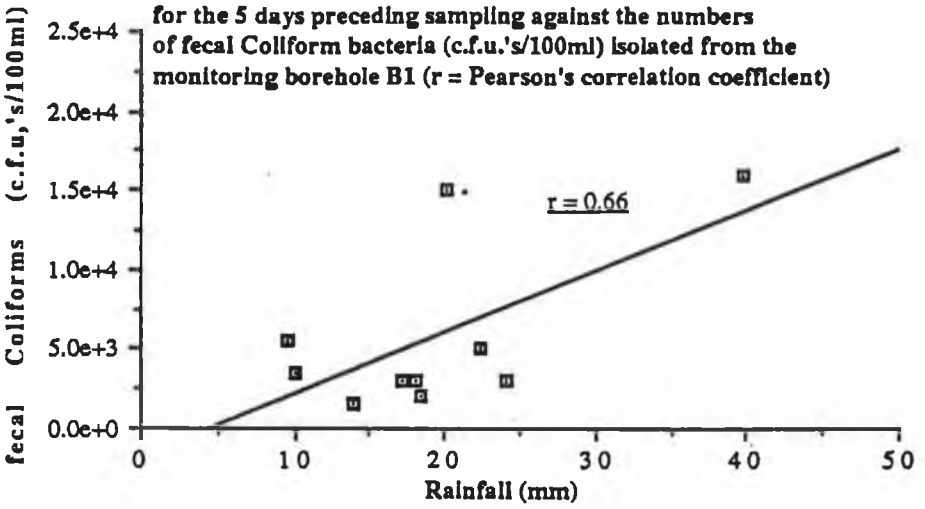
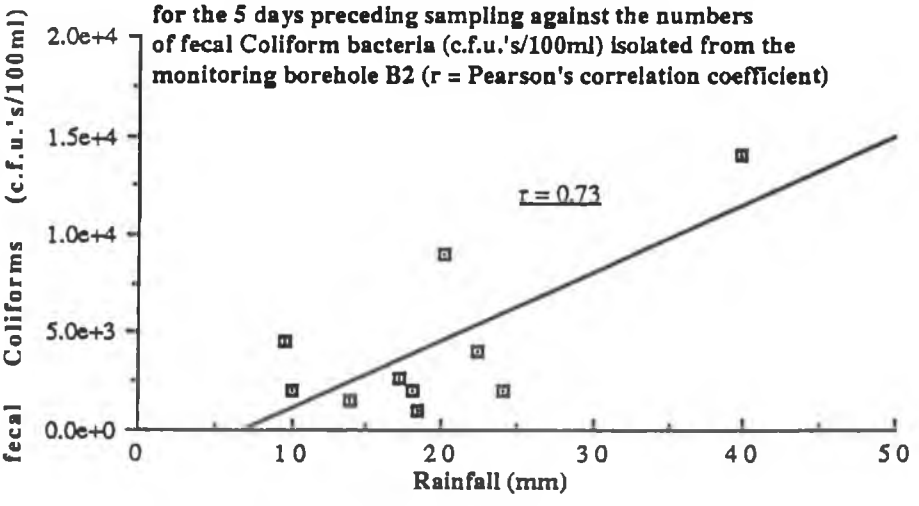


Figure 15
Linear regression line fitted to a plot of the total rainfall for the 5 days preceding sampling against the numbers of fecal Coliform bacteria (c.f.u.'s/100ml) isolated from the monitoring borehole B2 (r = Pearson's correlation coefficient)



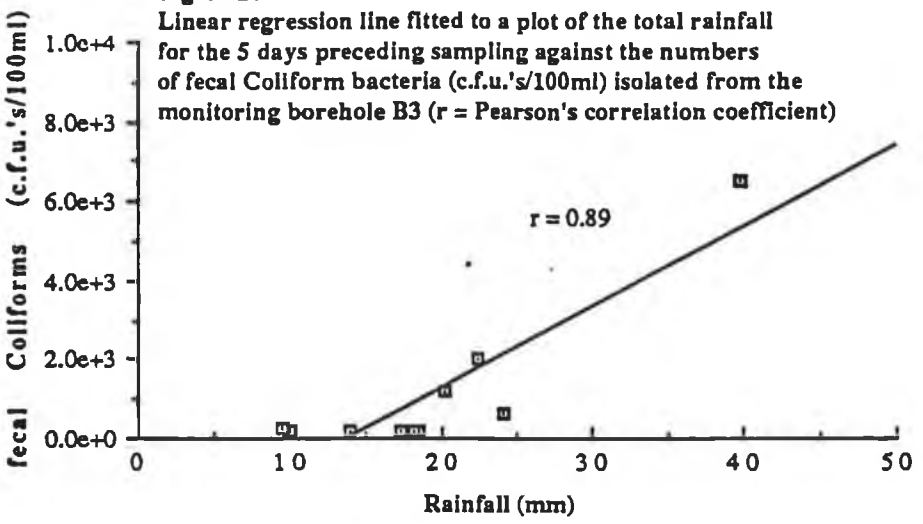


Figure 16
Linear regression line fitted to a plot of the total rainfall for the 5 days preceding sampling against the numbers of fecal Coliform bacteria (c.f.u.'s/100ml) isolated from the monitoring borehole B3 (r = Pearson's correlation coefficient)

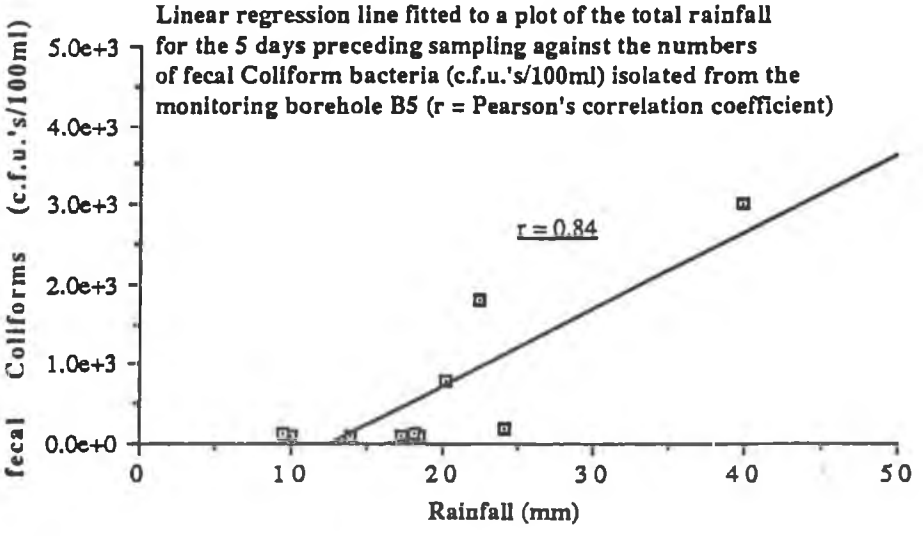


Figure 17
Linear regression line fitted to a plot of the total rainfall for the 5 days preceding sampling against the numbers of fecal Coliform bacteria (c.f.u.'s/100ml) isolated from the monitoring borehole B5 (r = Pearson's correlation coefficient)

Figure 18
Linear regression line fitted to a plot of the total rainfall for the 5 days preceding sampling against the numbers of fecal Coliform bacteria (c.f.u.'s/100ml) isolated from the monitoring borehole B6 (r = Pearson's correlation coefficient)

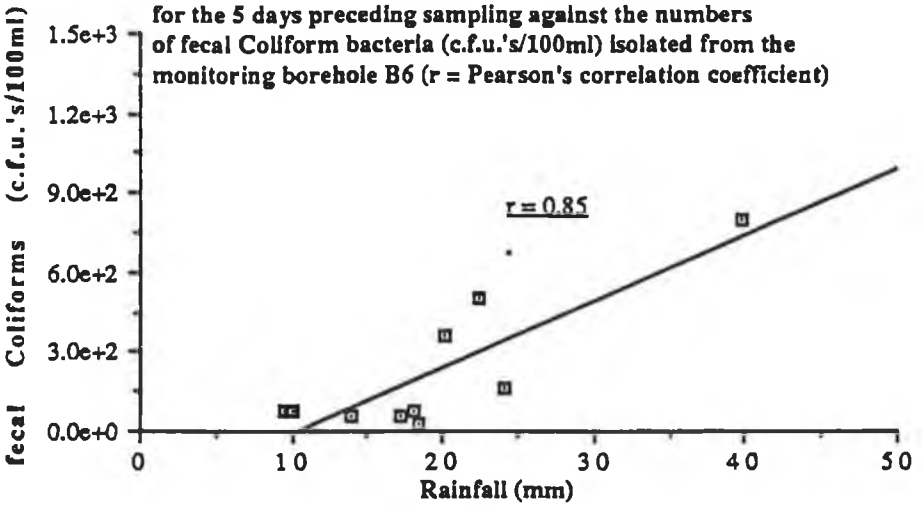


Figure 19
Linear regression line fitted to a plot of the total rainfall for the 5 days preceding sampling against the numbers of fecal Coliform bacteria (c.f.u.'s/100ml) isolated from the monitoring borehole B8 (r = Pearson's correlation coefficient)

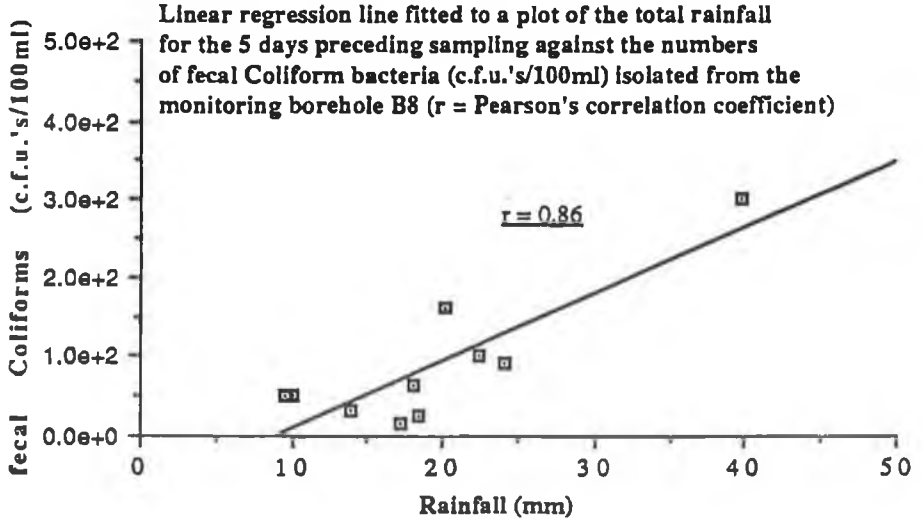


Figure 20
Linear regression line fitted to a plot of the total rainfall for the 5 days preceding sampling against the numbers of fecal Coliform bacteria (c.f.u.'s/100ml) isolated from the control spring C1 (r = Pearson's correlation coefficient)

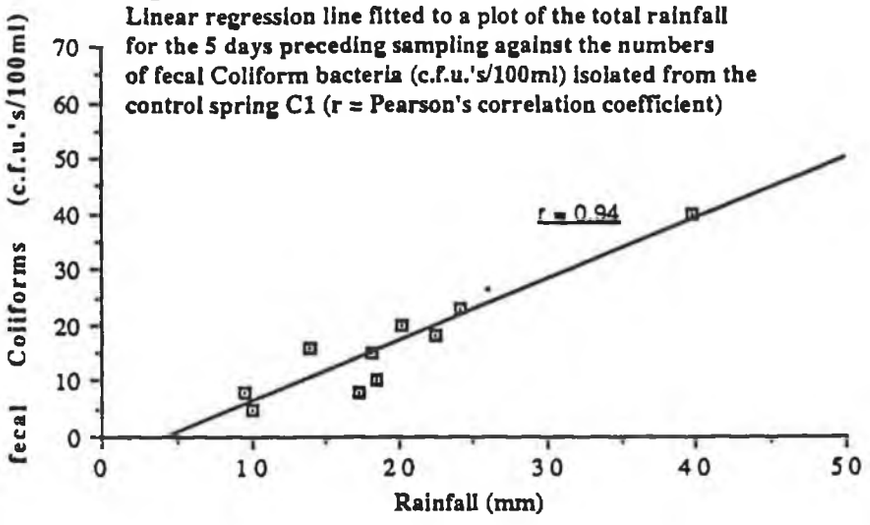
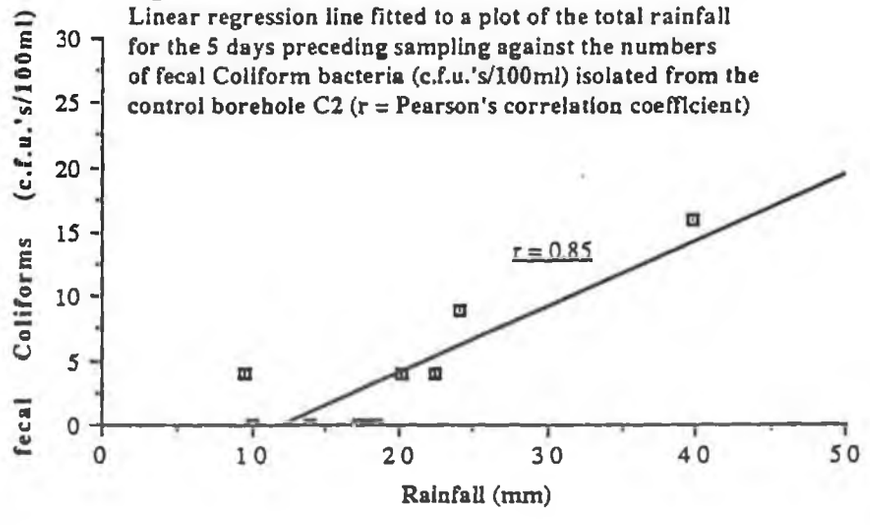


Figure 21
Linear regression line fitted to a plot of the total rainfall for the 5 days preceding sampling against the numbers of fecal Coliform bacteria (c.f.u.'s/100ml) isolated from the control borehole C2 (r = Pearson's correlation coefficient)



B 5

The Physical, Chemical and Biological Data From the Analysis of the Groundwater Samples From the Test Sites and Control Stations During the Eleven Month Monitoring Programme

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Table 1

Temperature (° C) Of The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	13.0	13.0	11.5	11.0	12.0	12.5	-	12.5	11.0	10.9
August	13.1	13.3	11.7	12.2	11.2	12.9	-	12.1	11.0	10.9
September	12.3	12.8	11.3	11.0	10.7	12.2	13.5	12.8	10.8	11.1
October	11.5	11.8	11.2	11.1	10.8	13.0	13.0	12.0	11.0	11.5
November	10.5	10.8	11.0	10.7	10.8	10.9	10.7	10.7	10.8	10.9
December	10.2	9.9	10.0	10.3	10.3	10.1	-	10.5	10.3	11.0
January 1989	10.4	9.4	9.8	10.0	10.0	10.4	10.2	10.1	10.0	10.0
February	9.8	9.0	9.3	-	9.8	10.0	9.9	10.0	9.5	10.3
March	9.6	8.8	9.1	-	9.2	10.2	9.8	10.3	9.0	8.8
June	11.0	10.9	10.0	-	9.9	10.4	-	10.5	9.8	9.5

Table 2

Water Level (m) In The Monitoring Boreholes On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	1.98	1.73	5.65	5.65	5.55	4.92	-	4.57	-	-
August	1.79	1.55	5.48	5.47	5.41	4.74	-	4.44	-	-
September	1.33	1.05	5.25	5.24	5.20	3.87	3.87	3.51	-	-
October	1.21	1.01	5.11	5.10	5.04	3.79	3.79	3.39	-	-
November	1.25	1.02	5.20	5.19	5.12	4.24	3.92	3.61	-	-
December	1.42	1.16	5.14	5.13	5.08	3.87	-	3.53	-	-
January 1989	1.11	0.87	4.88	4.87	4.82	3.57	3.60	3.22	-	-
February	1.15	0.91	5.04	5.02	4.98	3.68	3.70	3.32	-	-
March	1.04	0.82	4.75	4.74	4.68	3.34	3.29	3.00	-	-
June	1.82	1.59	5.49	5.48	5.40	4.81	-	4.29	-	-

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Table 3

pH Values (pH units) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	6.75	6.84	7.45	7.42	7.45	7.04	-	6.95	6.99	9.95
August	6.85	7.11	7.23	7.46	7.40	6.99	-	7.01	7.04	6.99
September	6.93	6.95	7.13	7.20	7.08	6.84	7.09	6.91	6.89	6.85
October	6.76	6.75	6.98	7.08	7.14	6.94	6.93	6.95	6.98	7.07
November	7.26	7.34	7.47	7.43	7.38	6.94	7.11	6.77	6.95	7.10
December	7.04	7.20	7.38	7.51	7.42	7.08	-	7.18	7.08	7.16
January 1989	7.12	7.11	7.20	7.30	7.21	6.98	7.07	7.08	6.98	7.3
February	7.21	7.21	7.17	-	7.29	7.04	7.02	7.09	7.39	7.10
March	7.36	7.65	7.26	-	7.30	7.09	7.04	7.07	7.35	7.10
June	7.11	7.16	6.99	-	7.04	6.80	-	6.93	7.01	6.85

Table 4

Electrical Conductivity Values (µS/cm) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	1580	1103	902	839	781	752	-	841	435	619
August	1583	992	1315	948	825	850	-	822	742	666
September	1174	1124	907	880	809	864	1057	845	762	690
October	1188	1094	896	875	861	893	947	883	789	702
November	1079	1054	845	850	858	899	941	897	790	702
December	1070	1015	812	805	805	880	-	896	730	695
January 1989	1210	1099	871	828	807	901	1010	892	769	718
February	1026	1064	874	-	878	995	1017	892	769	718
March	938	899	776	-	683	879	888	865	728	659
June	1576	1029	923	-	757	906	-	859	644	745

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Table 5

B.O.D. Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	262	40	2	2	3	1	-	1	1	1
August	265	237	2	1	1	1	-	1	1	1
September	104	24	2	1	1	2	1	0	0	0
October	128	93	4	3	2	1	1	1	1	1
November	77	28	0	0	0	0	0	0	0	0
December	118	21	2	1	0	0	-	0	0	0
January 1989	120	115	2	1	1	1	2	0	0	0
February	96	104	2	-	1	1	3	0	0	0
March	82	110	2	-	1	1	2	0	0	0
June	121	142	2	-	1	1	-	0	0	0

Table 6

C.O.D. Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	580	100	6	5	10	5	-	0	0	0
August	630	75	9	1	1	8	-	8	0	0
September	360	300	34	30	18	20	25	23	15	15
October	480	280	5	11	2	20	15	0	0	0
November	190	80	6	4	3	4	8	0	0	0
December	590	70	30	15	10	11	-	12	8	8
January 1989	52	52	11	14	21	4	4	8	10	10
February	150	200	28	-	22	31	40	23	10	14
March	130	290	28	-	32	28	46	21	10	10
June	210	280	18	-	15	12	-	14	8	8

Table 7

Nitrate - N Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	6.57	1.56	10.00	9.86	8.33	1.94	-	0.48	1.35	0.95
August	5.95	4.35	9.02	10.62	9.63	3.47	-	2.41	3.04	1.00
September	2.05	1.14	8.17	7.97	7.27	2.52	6.21	2.28	2.18	1.03
October	1.73	1.03	8.11	7.70	6.23	2.02	2.23	2.89	1.03	1.03
November	1.41	1.88	5.68	5.68	5.91	2.64	2.97	2.54	2.36	1.79
December	2.03	1.59	6.23	6.03	6.11	2.72	-	2.29	2.63	0.36
January 1989	3.43	1.24	5.92	5.67	5.86	3.48	3.28	1.94	2.34	0.46
February	0.61	2.23	8.11	-	7.31	3.59	3.15	3.79	2.79	0.31
March	0.71	1.38	6.76	-	5.05	3.07	4.07	2.71	2.80	0.72
June	4.04	3.10	8.95	-	7.00	2.80	-	2.10	1.00	0.80

Table 8

Ammonia - N Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	19.80	14.80	0.55	1.06	0.03	0.04	-	0.03	0.00	0.00
August	25.30	19.80	1.03	0.53	0.26	0.25	-	0.28	0.04	0.00
September	22.40	18.60	0.70	0.07	0.03	0.02	0.37	0.03	0.00	0.00
October	13.79	7.30	0.20	0.00	0.00	0.00	1.90	0.00	0.00	0.00
November	14.94	23.05	0.13	0.00	0.00	0.00	0.24	0.00	0.00	0.00
December	13.75	15.75	0.02	0.03	0.00	0.00	-	0.01	0.00	0.00
January 1989	24.55	15.75	0.15	0.00	0.50	0.00	0.14	0.90	0.00	0.00
February	3.90	13.36	0.09	-	0.01	0.03	0.09	0.01	0.00	0.00
March	7.36	13.53	0.01	-	0.00	0.01	0.48	0.01	0.00	0.00
June	22.00	27.50	0.23	-	0.04	0.01	-	0.01	0.01	0.00

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Table 9

Phosphate - P Concentrations (mg/l) Recorded Of The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	3.84	1.21	0.00	0.00	0.00	0.00	-	0.00	0.00	0.00
August	3.92	0.06	0.00	0.00	0.00	0.00	-	0.00	0.00	0.00
September	1.99	1.88	0.04	0.02	0.00	0.01	0.22	0.16	0.19	0.20
October	2.11	1.96	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00
November	1.03	0.66	0.48	0.40	0.38	0.00	0.58	0.00	0.00	0.00
December	1.97	0.60	0.00	0.00	0.00	0.00	-	0.00	0.00	0.00
January 1989	1.92	2.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
February	1.97	1.86	0.00	-	0.00	0.00	0.00	0.00	0.00	0.00
March	2.00	2.26	0.00	-	0.00	0.00	0.00	0.00	0.00	0.00
June	3.15	2.29	0.10	-	0.00	0.00	-	0.00	0.00	0.00

Table 10

Chloride Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	111.7	72.4	47.6	47.6	43.6	47.6	-	39.7	31.7	39.5
August	88.7	75.2	55.9	42.4	44.3	46.2	-	44.3	34.7	40.0
September	72.7	68.9	68.9	61.2	53.6	57.4	57.4	49.7	38.2	38.2
October	65.3	61.4	49.9	49.9	46.1	53.7	49.9	53.7	42.2	38.4
November	80.1	74.0	55.5	57.5	63.7	57.5	65.8	57.5	53.4	61.7
December	69.4	65.6	50.1	50.1	44.3	48.2	-	53.9	49.6	40.4
January 1989	68.7	65.1	42.3	45.7	48.7	63.7	65.2	61.0	47.6	56.2
February	67.9	64.2	41.2	-	49.1	65.7	65.6	57.1	48.1	43.9
March	68.6	63.9	40.4	-	48.2	65.5	65.5	57.8	46.2	44.3
June	89.9	57.9	33.9	-	33.9	40.9	-	40.9	33.9	39.9

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Table 11

Sodium Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	B2	B3	B4	B5	B6	B7	B8	C1	C2
July 1988	160	131	51	54	35	39	-	35	17	29
August	148	126	43	43	34	36	-	31	17	31
September	75	80	39	31	26	29	58	27	16	28
October	98	81	47	45	30	30	38	33	17	22
November	84	73	23	20	19	26	30	20	13	20
December	69	65	28	28	30	35	-	32	26	23
January 1989	68	58	37	23	27	54	35	33	17	33
February	42	53	25	-	25	27	27	27	20	27
March	40	52	20	-	24	26	28	28	16	29
June	140	90	40	-	32	35	-	31	19	38

Table 12

Potassium Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	B2	B3	B4	B5	B6	B7	B8	C1	C2
July 1988	37.7	24.6	9.3	10.4	7.4	2.4	-	2.3	2.9	3.1
August	29.6	25.7	12.8	9.4	8.2	2.6	-	3.4	2.9	2.9
September	29.8	23.7	19.7	7.4	6.9	6.1	6.2	2.6	5.2	4.0
October	25.1	17.0	8.5	8.5	7.3	2.6	3.1	2.4	3.1	3.9
November	24.2	21.2	8.2	7.6	9.2	2.6	3.4	2.3	2.7	3.4
December	20.9	19.9	7.5	8.0	7.0	2.7	-	2.6	2.6	4.2
January 1989	19.8	12.9	6.1	5.5	5.3	2.7	2.9	2.4	2.4	3.2
February	5.6	17.9	10.6	-	7.9	3.2	2.9	1.0	3.1	1.0
March	13.4	10.0	4.1	-	8.0	2.7	3.1	2.6	2.6	3.1
June	25.5	18.5	8.4	-	7.0	4.0	-	2.4	2.8	3.2

Table 13

Potassium/Sodium Ratios (K/Na) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	0.236	0.188	0.182	0.193	0.211	0.062	-	0.066	0.171	0.107
August	0.200	0.204	0.298	0.219	0.241	0.072	-	0.110	0.171	0.09
September	0.397	0.296	0.505	0.239	0.265	0.210	0.106	0.096	0.325	0.143
October	0.256	0.210	0.181	0.189	0.243	0.067	0.081	0.073	0.182	0.177
November	0.288	0.290	0.357	0.380	0.484	0.100	0.113	0.115	0.208	0.170
December	0.303	0.306	0.268	0.286	0.233	0.077	-	0.081	0.100	0.183
January 1989	0.291	0.222	0.165	0.240	0.196	0.050	0.083	0.073	0.141	0.097
February	0.133	0.338	0.424	-	0.316	0.119	0.107	0.037	0.155	0.037
March	0.335	0.192	0.205	-	0.333	0.104	0.111	0.093	0.163	0.107
June	0.183	0.206	0.210	-	0.219	0.114	-	0.077	0.147	0.084

Table 14

Alkalinity Values (mg/l $CaCO_3$) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	567	513	432	415	388	410	-	405	383	307
August	568	520	368	380	368	380	-	376	356	325
September	592	529	436	390	378	407	462	399	365	294
October	560	493	410	410	395	400	480	450	400	318
November	546	491	382	361	378	399	404	399	361	306
December	500	440	380	370	395	380	-	388	356	296
January 1989	421	450	410	405	392	405	370	379	306	298
February	433	449	373	-	382	399	405	411	348	310
March	435	392	370	-	368	378	407	395	306	334
June	564	484	400	-	384	420	-	416	308	360

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Table 15

Calcium Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	87	81	123	113	114	158	-	151	129	110
August	83	82	102	98	119	127	-	148	98	108
September	96	78	125	127	128	165	158	152	105	102
October	83	84	127	126	125	130	162	154	132	122
November	91	88	118	119	123	146	148	148	126	128
December	84	75	120	122	122	140	-	143	102	115
January 1989	83	77	119	122	120	142	140	141	115	115
February	92	78	120	-	119	141	142	142	121	119
March	96	93	115	-	118	145	148	148	108	121
June	90	89	120	-	122	150	-	156	112	123

Table 16

Magnesium Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	38	22	28	26	28	16	-	18	22	20
August	40	38	29	27	30	20	-	19	23	24
September	48	41	26	27	26	17	18	16	23	22
October	40	38	27	27	28	35	37	18	22	22
November	51	42	28	28	28	26	26	26	21	23
December	42	34	26	25	27	44	-	25	24	21
January 1989	32	45	27	27	26	25	26	26	15	20
February	49	47	28	-	28	32	30	29	20	20
March	54	49	26	-	27	28	23	25	15	19
June	57	53	28	-	28	30	-	29	22	20

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Table 17

Total Coliform Bacteria (c.f.u.'s/100ml) Isolated From The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	4000	4000	1000	800	800	200	-	200	24	8
August	4000	4000	800	550	400	400	-	160	19	6
September	10000	5000	5000	4000	1000	2000	2000	1000	21	18
October	10000	8000	600	400	400	400	200	104	12	9
November	6000	4000	1000	650	400	250	100	80	26	4
December	5000	4500	450	220	200	120	-	50	10	0
January 1989	20000	15000	10000	6000	8000	2700	4000	2000	100	38
February	40000	25000	2000	-	1800	1160	1600	1200	56	18
March	8000	5000	1000	-	1000	300	500	160	51	26
June	6500	4000	800	-	500	190	-	100	18	6

Table 18

Fecal Coliform Bacteria (c.f.u.'s/100ml) Isolated From The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	3000	2000	190	184	120	80	-	64	15	0
August	3000	2600	166	176	97	60	-	16	8	0
September	5000	4000	2000	2000	1800	500	600	100	18	4
October	5500	4600	250	200	140	80	120	52	8	4
November	1600	1500	200	120	100	60	52	32	16	0
December	3500	2000	200	100	100	80	-	50	5	0
January 1989	16000	14000	6500	4000	3000	800	1000	300	40	16
February	15000	9000	1200	-	800	360	240	160	20	4
March	3000	2000	650	-	200	160	100	92	23	9
June	2000	1000	180	-	80	24	-	24	10	0

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Table 19

Fecal Streptococci Bacteria (c.f.u.'s/100ml) Isolated From The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	14	13	24	12	8	16	-	8	0	0
August	15	20	18	12	12	12	-	10	0	0
September	40	50	20	20	4	10	4	9	4	0
October	18	12	18	0	4	0	5	0	0	0
November	21	18	20	20	2	6	10	10	0	0
December	0	0	0	0	0	8	-	0	0	0
January 1989	180	180	80	64	40	32	40	16	4	0
February	18	4	12	-	10	20	15	0	0	0
March	28	36	21	-	10	18	17	14	0	0
June	12	16	25	-	0	4	-	0	0	0

Table 20

Sulphate Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	83.7	72.1	16.1	16.8	17.2	30.7	-	27.9	11.9	36.0
August	115.0	102.0	22.5	14.3	12.2	21.9	-	30.5	5.6	33.0
September	25.3	19.3	20.1	17.3	14.2	19.1	29.1	25.4	9.5	27.9
October	35.2	19.5	34.8	38.9	28.9	51.8	42.5	44.5	11.4	31.7
November	84.0	13.0	24.5	23.3	19.4	24.6	24.6	22.6	7.9	34.8
December	11.4	9.2	10.9	17.9	13.0	24.1	-	23.0	9.0	22.0
January 1989	52.9	26.6	17.9	26.6	17.9	33.1	45.1	45.0	15.9	34.5
February	38.4	21.7	32.9	-	19.0	22.7	30.7	23.6	11.7	27.5
March	65.4	113.9	18.4	-	17.2	21.5	31.8	34.8	10.2	28.9
June	52.0	58.0	23.1	-	21.0	32.4	-	38.2	12.3	34.0

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Table 21

Iron Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2		< 0.2	< 0.2	< 0.2
February 1989	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2

Table 22

Manganese Concentrations (mg/l) Recorded In The Groundwater Samples On The Various Sampling Dates

Sampling Date	Monitoring borehole/spring									
	B1	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B8**	**C1**	**C2**
July 1988	0.75	0.57	0.18	0.09	< 0.05	< 0.05		< 0.05	< 0.05	< 0.05
February 1989	0.62	0.73	0.14	0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05

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Table 23

Chemical And Microbiological Quality Of The Septic Tank Effluent At Site 1

Parameter	Sampling Date		
	Jul. 1988	Nov. 1988	Mar. 1989
B.O.D. (mg/l)	194	279	209
C.O.D. (mg/l)	261	415	326
S.S. (mg/l)	78	103	95
pH (pH units)	7.7	7.5	7.4
Conductivity (µS/cm)	1110	1210	998
NO_3-N (mg/l)	1.15	1.35	1.00
NH_3-N (mg/l)	29.3	24.2	19.2
PO_4-P (mg/l)	18.7	29.0	17.8
Chloride (mg/l)	63	71	55
Sodium (mg/l)	112	81	79
Potassium (mg/l)	33	28	22
K/Na ratio	0.29	0.35	0.28
Sulphate (mg/l)	31	46	50
Total Coliforms (c.f.u.'s/100 mls)	2000000	1500000	1300000
Fecal Coliforms (c.f.u.'s/100 mls)	1000000	900000	500000
Fecal Streptococci (c.f.u.'s/100 mls)	80000	3000	4000

Table 24

Chemical And Microbiological Quality Of The Septic Tank Effluent At Site 2

Parameter	Sampling Date		
	Jul. 1988	Nov. 1988	Mar. 1989
B.O.D. (mg/l)	262	187	316
C.O.D. (mg/l)	498	382	615
S.S. (mg/l)	98	80	151
pH (pH units)	7.1	7.5	7.8
Conductivity (µS/cm)	995	1060	1112
NO3-N (mg/l)	0.06	0.20	0.80
NH3-N (mg/l)	28.8	31.5	30.2
PO4-P (mg/l)	31.5	28.2	36.0
Chloride (mg/l)	30.0	62.1	35.5
Sodium (mg/l)	79	96	74
Potassium (mg/l)	18.9	29.6	19.2
K/Na ratio	0.24	0.31	0.26
Sulphate (mg/l)	18.1	22.4	37.0
Total Coliforms (c.f.u.'s/100 mls)	1300000	38000000	1800000
Fecal Coliforms (c.f.u.'s/100 mls)	260000	1100000	50000
Fecal Streptococci (c.f.u.'s/100 mls)	4000	180000	14000

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Table 25

Chemical And Microbiological Quality Of The Septic Tank Effluent At Site 3

Parameter	Sampling Date		
	Jul. 1988	Nov. 1988	Mar. 1989
B.O.D. (mg/l)	385	298	426
C.O.D. (mg/l)	511	513	690
S.S. (mg/l)	215	126	202
pH (pH units)	7.4	7.3	7.4
Conductivity (µS/cm)	1015	1000	1210
NO3-N (mg/l)	0.52	0.00	0.14
NH3-N (mg/l)	24.6	21.5	35.6
PO4-P (mg/l)	57.3	31.5	42.8
Chloride (mg/l)	67.1	48.5	49.7
Sodium (mg/l)	100	86	93
Potassium (mg/l)	28.0	30.1	38.0
K/Na ratio	0.24	0.35	0.41
Sulphate (mg/l)	32.8	38.6	61.0
Total Coliforms (c.f.u.'s/100 mls)	13000000	6200000	7800000
Fecal Coliforms (c.f.u.'s/100 mls)	1100000	800000	100000
Fecal Streptococci (c.f.u.'s/100 mls)	130000	9000	13000

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APPENDIX C
AN ASSESSMENT OF THE SUITABILITY OF A RANGE OF CHEMICAL AND BIOLOGICAL TRACER MATERIALS TO MONITOR THE MOVEMENT OF SEPTIC TANK EFFLUENTS TO GROUNDWATER

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C 1

Results of the Analysis of the Groundwaters at the Three Test Sites For the Chemical and Biological Tracer Materials at Various Time Intervals After Tracer Addition To the Septic Tank Systems

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Table 1

Results of the Sodium Fluorescein Concentrations (mg/l) Detected in the Groundwater Samples at Various Time Intervals After Tracer Addition

Time after Tracer addition	Monitoring borehole						
(hours)	B1	B2	B3	B5	B6	B7	B8
0	0.157	0.189	0.021	0.021	0.012	0.010	0.013
8	0.289	0.206	0.016	0.020	0.010	0.011	0.090
16	0.062	0.051	0.013	0.029	0.011	0.012	0.006
24	0.101	0.089	0.020	0.020	0.004	0.005	0.003
32	0.721	0.308	0.562	0.023	0.023	0.009	0.006
40	8.721	2.381	1.257	0.032	0.032	0.021	0.082
48	10.50	2.112	2.981	1.036	0.015	0.018	0.015
56	10.50	3.618	3.204	1.886	0.019	0.005	0.001
72	10.82	3.776	5.509	2.536	0.006	0.002	0.004
112	6.51	3.729	9.489	2.931	0.023	0.010	0.001
160	2.165	1.051	2.611	0.841	0.016	0.006	0.008

Table 2

Results of the Rhodamine B Concentrations (mg/l) Detected in the Groundwater Samples at Various Time Intervals After Tracer Addition

Time after Tracer addition	Monitoring borehole						
(hours)	B1	B2	B3	B5	B6	B7	B8
0	0	0	0	0	0	0	0
8	0	0	0.0010	0.0001	0	0	0
16	0	0	0	0	0	0.0001	0
24	0.0012	0.0012	0.0001	0.0003	0	0	0
32	0.0029	0.0029	0.0035	0.0003	0.0002	0	0
40	0.0081	0.0081	0.0082	0.0003	0	0	0
48	0.0092	0.0092	0.0090	0.0030	0.0013	0	0
56	0.0087	0.0087	0.0088	0.0040	0.0021	0	0
72	0.0079	0.0079	0.0085	0.0040	0.0020	0	0
112	0.0072	0.0072	0.0080	0.0050	0.0030	0	0
160	0.0030	0.0036	0.0010	0.0001	0	0	0

Table 3

Results of the Electrical Conductivity Concentrations (µS/cm) Detected in the Groundwater Samples at Various Time Intervals After Tracer Addition

Tracer addition	Monitoring borehole						
(hours)	B1	B2	B3	B5	B6	B7	B8
0	873	892	610	666	850	927	778
8	917	900	637	674	885	933	828
12	925	906	642	651	892	967	832
24	951	924	646	675	893	990	887
32	1011	932	638	657	880	950	870
40	995	950	650	652	870	952	860
48	1010	959	642	672	923	960	852
56	980	946	633	660	880	937	870
64	958	950	630	665	881	930	845
80	885	891	570	662	800	830	724
96	795	840	580	610	789	860	736
158	877	856	595	637	810	887	801

Table 4

Results of the Bromide Concentrations (mg/l) Detected in the Groundwater Samples at Various Time Intervals After Tracer Addition

Time after Tracer addition	Monitoring borehole						
(hours)	B1	B2	B3	B5	B6	B7	B8
0	0	0	0	0	0	0	0
8	0.10	0.11	0	0	0	0	0
12	0.12	0.10	0	0	0	0	0
24	0.74	0.29	0	0	0	0	0
32	3.39	2.11	0.50	0.10	0	0	0
40	1.29	0.90	0.87	0.32	0	0	0
48	1.02	0.89	1.09	0.51	0	0	0
56	1.02	0.87	1.59	2.07	0	0	0
64	1.20	0.90	1.36	1.76	0.33	0.71	0.31
80	1.19	1.00	0	0.84	1.54	2.19	1.91
96	0.72	0.74	0	0	1.52	1.44	1.06
158	0	0	0	0	0.62	0.99	0.22

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Table 5

Results of the Nitrate - N Concentrations (mg/l) Detected in the Groundwater Samples at Various Time Intervals After Tracer Addition

Time after Tracer addition	Monitoring borehole						
(hours)	B1	B2	B3	B5	B6	B7	B8
0	0.87	0.68	4.77	4.14	2.13	2.40	1.13
8	0.89	0.91	4.72	4.95	2.05	2.54	2.49
12	0.78	1.51	4.72	4.17	1.89	2.38	2.30
24	1.58	1.05	4.61	5.57	2.19	2.55	2.22
32	2.67	2.50	6.15	6.66	2.21	2.32	2.29
40	2.07	2.43	6.12	6.67	3.11	5.85	2.76
48	1.97	2.58	6.40	7.00	4.09	5.77	4.49
56	2.07	2.58	6.40	6.73	4.40	5.69	4.07
64	6.77	2.86	6.38	6.39	4.35	6.28	4.26
80	1.15	1.36	6.13	6.44	3.93	5.45	3.55
96	1.07	1.38	6.01	6.58	3.90	5.24	3.01
158	0.68	0.99	5.78	5.59	3.05	4.27	2.02

Table 6

Results of the Chloride Concentrations (mg/l) Detected in the Groundwater Samples at Various Time Tntervals After Tracer Addition

Time after Tracer addition	Monitoring borehole						
(hours)	B1	B2	B3	B5	B6	B7	B8
0	47.5	39.6	33.7	29.8	43.6	47.6	43.6
8	56.3	47.6	31.7	25.8	39.7	51.6	39.7
12	51.6	43.6	33.7	25.8	43.6	47.6	41.6
24	57.5	49.6	39.7	27.7	49.6	48.5	43.6
32	81.3	66.4	63.4	43.6	51.5	51.5	39.7
40	71.4	63.4	45.6	47.6	47.6	47.6	41.6
48	67.4	59.5	49.6	49.6	47.6	45.6	45.6
56	63.4	61.3	47.6	47.6	43.6	47.6	41.6
64	59.5	55.5	41.6	41.6	45.6	49.6	45.6
80	51.6	51.6	39.7	37.9	43.6	43.6	43.6
96	51.6	47.6	39.7	31.7	41.6	43.5	40.6
158	49.6	45.6	35.7	31.7	45.6	45.6	41.6

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Table 7

Results of the Sodium Concentrations (mg/l) Detected in the Groundwater Samples at Various Time Intervals After Tracer Addition

Time after Tracer addition	Monitoring borehole						
(hours)	B1	B2	B3	B5	B6	B7	B8
0	28.0	23.1	18.5	16.2	22.0	26.0	22.1
8	25.0	29.2	16.5	14.7	19.9	32.0	26.5
12	30.0	27.4	18.0	14.5	21.5	22.1	24.5
24	38.0	31.0	18.6	14.5	25.0	26.3	18.3
32	60.8	62.1	16.0	14.3	25.0	31.7	22.5
40	47.5	55.0	19.3	14.3	19.0	20.0	18.0
48	31.2	37.5	19.0	15.0	30.0	25.6	26.1
56	22.5	45.0	21.3	14.5	24.0	28.0	26.4
64	21.5	43.5	19.5	17.3	22.0	28.1	26.1
80	20.8	37.2	18.5	18.5	26.0	27.1	24.2
96	20.0	35.1	18.5	18.5	25.0	27.8	20.4
158	22.0	33.0	19.1	17.5	24.0	29.0	22.5

Table 8

Results of the Potassium Concentrations (mg/l) Detected in the Groundwater Samples at Various Time Intervals After Tracer Addition

Time after Tracer addition	Monitoring borehole						
(hours)	B1	B2	B3	B5	B6	B7	B8
0	13.4	10.2	5.6	5.4	2.0	2.5	1.9
8	13.6	13.0	4.8	4.7	2.8	2.6	2.0
12	15.0	13.0	4.9	4.5	2.4	2.1	1.8
24	20.0	15.0	4.8	4.5	2.4	3.1	2.3
32	19.8	15.9	5.0	5.4	2.4	2.9	2.3
40	15.0	14.0	5.0	5.4	2.4	2.9	2.4
48	14.0	14.0	5.0	5.5	2.8	3.2	2.6
56	14.2	13.6	5.2	5.5	2.7	3.1	2.3
64	12.4	11.8	5.0	5.3	2.9	2.8	2.3
80	13.6	12.9	4.8	5.0	2.4	2.9	2.4
96	13.5	13.0	4.5	5.0	2.6	2.8	2.5
158	13.8	12.1	4.8	5.2	2.3	2.7	2.0

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Table 9

Results of the Numbers of *Escherichia coli* Tracer Bacteria (c.f.u.'s/100ml) Isolated From the Groundwater Samples at Various Time Intervals After Tracer Addition

Time after Tracer addition (hours)	Monitoring borehole						
	B1	B2	B3	B5	B6	B7	B8
0	15	18	0	0	0		0
12	10	20	0	0	0		0
36	2500	2500	0	0	0		0
48	28000	10000	40	0	0		0
60	98000	20000	60	0	0		0
72	100000	22000	60	0	0		0
84	50000	100000	20	0	0		0
96	28000	38000	0	0	0		0
108	30000	20000	0	0	0		0
120	12000	25000	0	0	0		0
144	6000	10000	100	20	0		0
156	2200	20000	200	35	0		0
180	1800	2500	800	60	1		0
204	200	1000	30	1	0		0
276	100	500	12	0	0		0
372	80	100	1	0	0		0

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Table 10

Results of the Numbers of *Bacillus globigii* Tracer Bacteria (c.f.u.'s/100ml) Isolated From the Groundwater Samples at Various Time Intervals After Tracer Addition

Time after Tracer addition (hours)	Monitoring borehole						
	B1	**B2**	**B3**	**B5**	**B6**	**B7**	**B8**
0	0	0	0	0	0		0
12	15	10	3	5	0		0
36	10	10	5	4	0		0
48	1360	940	306	160	120		70
60	750000	680000	480	160	36		161
72	1300000	990000	1390	452	40		10
84	2700000	1300000	340	60	60		40
96	2600000	1300000	360	820	430		140
108	2400000	930000	250	18	12		28
120	1600000	902000	2790	60	41		0
144	1020000	910000	2450	1590	105		21
156	870000	1800000	1488	2900	744		351
180	670000	960000	1870	748	190		50
204	590000	1000000	3060	1500	125		136
276	167000	650000	3400	1460	36		3
372	121000	56000	1800	1000	10		3

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