Groundwater flow dynamics beneath a wastewater irrigation site, Hautapu, Cambridge.

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Abstract

In this study, a qualitative groundwater hydrological model is developed to assist in evaluating the long-term environmental sustainability of a wastewater irrigation system at Hautapu, Cambridge. The model relies on the integration of site information, numerical groundwater modelling, geochemical data, and surface geophysical data.

The characteristics of the site relating to the groundwater system are examined including climate, topography, natural and artificial recharge, hydrogeology, hydraulic characteristics, seasonal variations, and abstraction activities. Integration of this information indicates groundwater movement is predominantly horizontal and northwestward, with highly restricted vertical movement downwards due to the presence of thick sequences of silt-based materials beginning at an average depth of ~10 m.

A numerical groundwater model was developed incorporating the site information to estimate groundwater flow patterns within the study area. The model suggests that lateral flow northwestward within the unconfined aquifer predominates in the study area and that there is probably minimal vertical movement from the upper aquifer zones to the deep aquifer.

An analysis of recent geochemical data is undertaken in this study to (i) determine the horizontal and vertical extent of modified groundwater movement; and (ii) provide ground truth for the electrical resistivity technique. The results of the analysis reveal elevated levels of sodium in shallow groundwater in a zone centred on the wastewater irrigation farm, with some localised lateral migration to the South, West and Northwest. Movement of modified groundwater vertically has occurred to a maximum depth of ~16 m on-site.

The surface geophysical technique, electrical resistivity was employed at the site to: (i) provide input about flow dynamics beneath the site, through the delineation of the low resistivity modified groundwater plume; and (ii) evaluate the success of the technique in detecting spatial variations in groundwater quality. A comprehensive survey was undertaken involving vertical soundings and horizontal profiling measurements. In general, a modified groundwater plume consistent with that defined by geochemical data was identified thereby validating the application of the

technique and confirming recognised movement patterns. However, the application of the technique is limited to areas where high conductivity modified groundwater controls the resistivity signal. Beyond these areas it is proven that the resistivity data may be an artefact of lithology.

The development of the qualitative groundwater hydrological model was achieved through the integration of the flow information obtained via the outlined investigative methods. The main outputs of the groundwater model are that modified groundwater movement is:

(i) Constrained within the unconfined aquifer; and

(ii) Predominantly horizontal, therefore 2-dimensional.

It is concluded that the current wastewater disposal activity appears to be performing sustainably in terms of nitrogen disposal, however consistent increases in sodium levels over the wastewater disposal site to the boundaries raises the possibility of future increases in sodium in groundwater just beyond the boundaries in some locations.

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I would like to take this opportunity to thank those people who have helped me in countless ways during the course of this research project.

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I would like to thank the residents of the Hautapu area for allowing access to their property, to either obtain groundwater samples or carry out measurements with the electrical resistivity equipment. A special thanks is extended to those that went as far as carrying equipment for me and offering refreshments.

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CHAPTER ONE

Introduction

1.1 Background

In the last 20 years, land application of wastewater has become increasingly popular such that it is now a widely practiced activity in New Zealand. The move towards systems of land treatment has gained momentum in New Zealand in the 1990's largely because direct disposal to surface waterways has become less acceptable. This attitude to protect the water quality of our rivers, lakes and marine environments is exemplified by restrictions imposed by New Zealand's regulatory authorities forcing many industries, like the dairy manufacturing industry, to implement land treatment systems. The main benefits associated with land application of wastewater include (1) equivalent treatment of the wastewater, or removal of nutrients; (2) protection of rivers, lakes and marine environments; and (3) nutrient recycling and water conservation (Balks 1994).

While the benefits associated with the land application of wastewater are clear, effects on groundwater quality are a potential impact associated with this method of disposal (Balks 1994, Parkin and Marshall 1976, Cameron *et al* 1996). This is an important factor to consider, given that the sustainability of land disposal activities is often determined subject to effects on groundwater quality.

Wastewater from Hautapu dairy factory, near Cambridge, has been disposed of onto adjacent farmland (Bardowie farm) for over 20 years. Originally, disposal of wastewater generated at the factory involved large spreadings from tankers or trucks. This method of disposal was then superseded in the 1970's, by a system of large single jet guns. Finally in 1982, a reticulated spray irrigation system, the most efficient option for disposal of dairy manufacturing effluent (Parkin and Marshall 1976), was installed on Bardowie farm.

Traditionally, impacts on shallow groundwater quality at Bardowie farm have been assessed through groundwater quality studies based on geochemical data from monitoring bores installed throughout the wastewater irrigation site. A number of groundwater quality studies have been carried out at the site, mainly considering groundwater quality variation through time (Barnett *et al* 1996, Terra Aqua Consultants 1993). In 1995, the first study considering the spatial variability of groundwater quality was conducted (Ward 1995). The main results of this study was the estimated delineation of a modified groundwater plume centred on the irrigation farm, with the possibility of lateral movement of modified groundwater to localised off-site areas. An attempt was also made to determine the vertical extent of movement however this proved unsuccessful.

Due to the remaining uncertainty with regards to the movement of modified groundwater at the conclusion of Ward's study, it became evident that further research was required at the site to define more accurately, the spatial extent of modified groundwater movement. It was anticipated that this research would assist in evaluating the long-term sustainability of the disposal scheme.

Recognising the significant investment the Hautapu dairy factory has made into directly assessing groundwater quality, the factory sought to trial and test, a non-intrusive method for achieving this. Consequently, the surface geophysical technique, electrical resistivity technique was proposed.

Accordingly, the current study was motivated by the need to:

(i) Define the extent of modified groundwater movement with a greater degree of confidence, to enable the long-term environmental sustainability of the wastewater disposal activity to be assessed; and

(ii) Evaluate the application of the electrical resistivity geophysical technique for assessing groundwater quality at the wastewater irrigation site.

1.2 Objectives of study

The main objective of this study was to develop a qualitative groundwater hydrological model to assist in evaluating the long-term environmental sustainability of the current wastewater disposal activity on Bardowie farm. The model was to rely on a range of information relating to the hydrogeology of the site, geochemical data, modelling work and electrical resistivity data. A subsidiary aim of the study was the implementation and evaluation of the electrical resistivity technique for cost effectively assessing groundwater quality at the site.

1.3 Thesis Outline

Chapter two backgrounds the general issue of groundwater quality impacts, and presents the theory relating to the two methods used in groundwater quality studies, including the advantages and limitations associated with each method. The electrical resistivity technique is reviewed in detail in this chapter.

Chapter three characterises the study area. Particular emphasis is given to the key environmental parameters controlling groundwater movement on and near the wastewater irrigation site, specifically the irrigation activity, local hydrogeology and deep groundwater abstractions.

Chapter four establishes the geochemical setting according to data from the monitoring bore network. The results of previous work and of recent data analysis are included in this chapter.

Chapter five describes the application of the electrical resistivity technique as a measure of groundwater quality in the study area. The extent to which the technique can be used as a water quality index is discussed in this chapter. Some groundwater dynamic possibilities are deduced from the resistivity measurements.

Chapter six describes a simple modelling study carried out to estimate groundwater flow patterns within part of the study area.

Conclusions and recommendations are presented in Chapter seven.

CHAPTER TWO

Groundwater Quality Studies

2.1 Introduction

In recent years, the quality of groundwater resources has come increasingly come under threat, as more and more chemicals originating from human activities have entered the zone of saturation. This increased pressure on groundwater systems has arisen due to population growth and industrial expansion and has necessitated assessments of groundwater quality beneath areas of human activity.

A variety of different chemical sources affecting groundwater quality are described in the literature including landfills, septic tanks, industrial waste containment sites, mining sites, industrial chemical spills, agricultural drainage reservoirs, wastewater irrigation sites and rapid infiltration sites. Non-point sources mostly relating to agricultural practises have also been investigated.

Assessment of groundwater quality has traditionally been achieved through geochemical studies facilitated by monitoring bore networks. However, in recent years there has been a shift towards the use of geophysical techniques such as electrical resistivity, because they provide a cost effective and non-intrusive means of assessing groundwater quality.

2.2 Geochemical Studies

Geochemical studies are generally considered the most accurate means of assessing the extent of chemical migration, as they allow for a direct assessment of groundwater quality. Consequently, this method has been independently employed in many groundwater quality studies. Table 2.1 lists geochemical groundwater quality studies described in the literature.

chemical source	*Study*
landfill	Cherry *et al* (1983)
	MacFarlane *et al* (1983)
	Nicholson *et al* (1983)
wastewater irrigation activity	Baxter (1985)
	Guanghe *et al* (1996)
	Mooers and Alexander (1994)
	Rashed *et al* (1995)
	Ward (1995)
wastewater rapid infiltration activity	Bedient *et al* (1983)
	Tomson *et al* (1985)
chemical spill	Roux and Althoff (1980)
agricultural activities	Bjerg and Christensen (1992)
	Fortina *et al* (1993)
	Rasmussen (1996)
sewage reservoirs (septic tanks)	Hadfield (1995)
	Hameed *et al* (1994)
	van der Kamp *et al* (1994)
	Robertson *et al* (1991)
injected tracers	Freyberg (1986)
	Garabedian *et al* (1991)
	Mackay *et al* (1986)
	Sudicky *et al* (1983)
industrial activities	Liu and Cheng (1997)
	Osiensky *et al* (1984)
	Richerson (1997)
	Rivett *et al* (1994)
	Ryan and Summerfield (1994)
hydrocarbon reservoir	Jengo (1995)
	Madrid *et al* (1998)

Table 2.1 Groundwater quality studies conducted using the geochemical study method.

Geochemical studies are based on direct point measurements of groundwater quality. This is achieved through a network of individual piezometers sampling a common depth, or alternatively a network of multilevel or bundle-piezometers which enable samples to be taken from multiple depths. Rapid groundwater sampling techniques such as the auger-head sampler (Cherry *et al* 1983), the hydropunch tool (Kraemer *et* *al* 1996), and the direct push technique (Jengo 1995, Ryan and Summerfield 1994) can be used in place of or in addition to permanent bore networks.

Although considered the most accurate means of assessing groundwater quality, a number of limitations are associated with geochemical studies:

(i) Possible errors introduced during sample recovery, handling, storage, and sample analysis (Appelo and Postma 1993, Freyberg 1986, Gorelick *et al* 1993, MacKay *et al* 1986, Young and Baxter 1985);

(ii) Chemicals may travel around bores and consequently migrate undetected because groundwater movement is typically along sinuous flowlines due to variations in lithology (Osiensky *et al* 1984, van der Kamp *et al* 1994);

(iii) Chemicals may be either introduced or lost from the groundwater, for example spurious chemicals may be introduced into samples from materials comprising the monitoring bore (Gorelick *et al* 1993);

(iv) Stagnant water in the well may affect the analysis results unless the well water is purged prior to sampling (Appelo and Postma 1993, Gorelick *et al* 1993);

(v) Noise from other known and unknown point and diffuse chemical sources; and

(vi) The intrusive, destructive nature and high costs associated with the method.

Because of the limitations of geochemical studies, in particular those noted in point (vi), many researchers have chosen to seek out alternative methods for investigating groundwater quality, including surface geophysical techniques.

2.3 Surface Geophysical Techniques

Over the years surface geophysical methods have proven useful tools for groundwater quality studies. These methods rely upon the direct relationship between the geophysical properties of aquifer materials and water composition such that changes in water conductivity effect changes in the geophysical properties of the aquifer materials. The techniques most commonly used in groundwater quality studies include ground penetrating radar (GPR), very low frequency electrical technique (VLF), electromagnetics (EM), and electrical resistivity.

Surface geophysical methods offer several major advantages over geochemical studies including the fact that they are inexpensive, non-invasive and are particularly useful where geochemical data is limited. In addition, the equipment is usually portable and easy to operate and for these reasons an expeditious assessment is allowed for.

While the advantages of surface geophysical techniques are noteworthy, their use in groundwater quality studies is limited by a number of factors:

(i) The natural scatter in the data due to variations in lithology, which might be misinterpreted as variations in water quality unless sufficient knowledge about site lithology is known (Greenhouse and Harris 1983);

(ii) The success of geophysical methods in groundwater quality studies largely depends on good chemical contrast occurring between the leached solution and ambient groundwater (Klefstad *et al* 1975, Kelly 1976);

(iii) The presence of electrical conductors such as fences, and underground cables and structures can affect data quality (Mazac *et al* 1986).

(iv) Scatter in the data due to changes in topography, implying fluctuations in the depth of the zone of interest (Nobes *et al* 1994).

(v) Surface geophysical techniques are depth limited, with the quality of the data significantly decreasing where pollution is deep (Klefstad *et al* 1975).

2.3.1 Electrical Resistivity

Of the surface geophysical techniques available, electrical resistivity is the most established methodology used in groundwater quality studies. Where elevated concentrations of chemicals has resulted in a significant increase in the conductivity of the groundwater, electrical resistivity (inverse of conductivity) may be used to delineate the resultant plume. Table 2.2 lists groundwater quality studies in which the electrical resistivity technique has been employed and the chemical sources investigated.

Chemical source	*Study*
sanitary landfill	Bogardi *et al* (1988)
	Broadbent (1992)
	Cartwright and McComas (1969)
	Cartwright and Sherman (1972)
	Frohlich *et al* (1994)
	Greenhous and Harris (1983)
	Kelly (1976)
	Klefstad *et al* (1975)
	Seitz *et al* (1972)
	Senos Matias *et al* (1994)
	Van Duijenbooden and Kooper (1981)
Hydrocarbon reserves	Benson *et al* (1997)
hazardous waste landfill	Zungailia *et al* (1989)
industrial waste site	De Lima *et al* (1995)
	Drozhko *et al* (1997)
	Hackbarth (1971)
	Risk (1980)
	Ulrych *et al* (1994)
injected tracer	Osiensky and Donaldson (1995)
salt water intrusion	Calhoun and Entin (1997)
	Ebraheem *et al* (1997)
mining site	Ebraheem *et al* (1990)
	Karous *et al* (1994)
	Merkel (1973)
wastewater rapid infiltration activity	Fink and Aulenbach (1974)
wastewater irrigation activity	Allen *et al* (1985)

Table 2.2 Groundwater quality studies in which the electrical resistivity technique has been employed.

The theory behind the use of the electrical resistivity technique in groundwater quality studies is simple. A current is passed through the saturated sub-surface between two outer electrodes, while two inner (potential) electrodes measure the potential difference in current flow between the outer electrodes. From the potential difference, given as resistance (Ω), the apparent resistivity[1] is calculated by the investigator. Current flow therefore the resistivity of saturated sub-surface zones will

[1] Apparent resistivity (Ωm) = C x S x R where C = constant 6.28, S = electrode spacing (m), and R = resistance (Ω) (Hempen and Hatheway 1992).

vary according to (1) lithology; and (2) the quality of the water present. Therefore the technique can provide important information about groundwater quality, given uniform background (natural) resistivity conditions.

There are numerous configuration options associated with the electrical resistivity technique, however, the Wenner and Schulemburger arrays are the configurations most commonly used, differing only with regards to the spacing of the electrodes (Robinson and Coruh 1988). The electrode configuration deployed in this study was the Wenner Array, regarded as the best configuration option available (Hempen and Hatheway 1992). The array requires four equally spaced electrodes partially inserted in the ground, in a straight line. The following figures illustrate the Wenner Array electrode configuration and electrical flow in standard 2-layer situations.

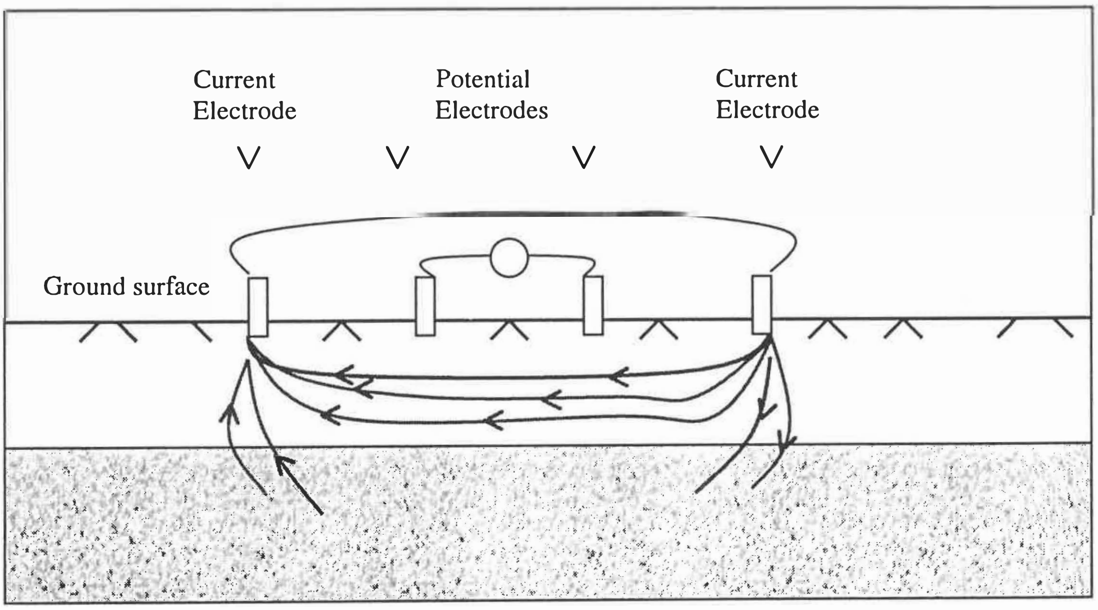


Figure 2.1a Wenner Array electrode configuration including electrical flow lines for a 2-layer case with a higher resistivity second layer.

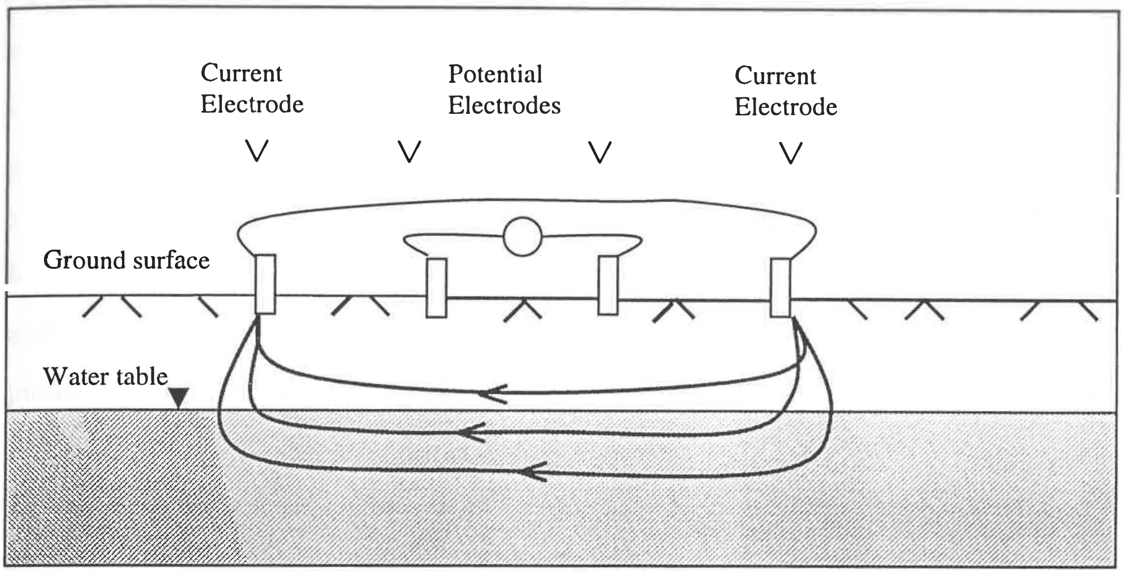


Figure 2.1b Wenner array electrode configuration including electrical flow lines for a water table 2-layer case, with lower resistivity in the saturated zone. It can be seen in Figures 2.1a and 2.1b that electrical flow is retarded in the high resistivity layer and more rapid in the low resistivity layer, as indicated by the directness of the flow lines.

There are two important principles that apply when using the Wenner Array. Firstly, the spacing of the electrodes determines the depth through which the current lines flow, such that the reading obtained is a spatial average over that depth. Secondly, the resistivity of the layer at a depth equal to the electrode spacing will have a proportionally larger effect on the resistivity reading (~73%) (Robinson and Coruh 1988).

The electrical resistivity method can be used to detect both lateral and vertical variations in resistivity. Vertical variations are detected with vertical soundings, which involve progressively increasing the electrode spacing about a fixed mid-point (Robinson and Coruh 1988). Lateral variations are detected in horizontal profiling surveys in which the electrode spacing is maintained and the array is simply moved from site to site (Robinson and Coruh 1988).

2.4 Integrated approaches

Although the advantages of using surface geophysical techniques in groundwater quality studies are obvious, a common perception is that that they will never actually replace drilling (Brown *et al* 1977, Hackbarth 1971, Neislsen 1991). In fact, many groundwater studies have been carried out incorporating both methods of investigation. Benson *et al* (1991) Ebraheem *et al* (1990), Ebraheem *et al* (1997), Kelly (1976), Klefstad *et al* (1975), Rogers & Kean (1980), and Van Duijvenbooden and Kooper (1981) combined geophysical methods together with chemical parameters to delineate zones of contamination.

The reasons for the integrated approach vary. There are those who consider this approach will allow for a more comprehensive survey in which a complete 3-dimensional view of the chemical plume can be obtained. Others have used the two methods as a way to compensate for the disadvantages of each. Most researchers have used chemical parameters in conjunction with resistivity data primarily to validate the resistivity data either quantitatively or qualitatively. For this study, the integrated approach is adopted to allow for:

(i) A comprehensive survey of modified groundwater movement beneath the wastewater irrigation site; and

(ii) An evaluation of the resistivity technique so that the extent to which the technique can be used as an index of water quality modification at the wastewater irrigation site may be evaluated.

2.5 Summary

While the benefits of the traditional method (geochemical studies) for assessing groundwater quality are widely recognised, the limitations associated with this method have led to the development and implementation of alternative methods, namely surface geophysical techniques. These techniques either used independently or in conjunction with geochemical data, are an indirect cost-effective means of assessing groundwater quality. Of the surface geophysical techniques available, electrical resistivity is the most accepted and commonly employed method in groundwater quality studies, reaffirming the application of electrical resistivity in this study.

Before the commencement of any groundwater quality investigation a comprehensive knowledge of the study area is required so that the inherent limitations of the chosen investigative method can be accounted for. The acquisition of site information is also essential for predicting the likely direction/s and rates of groundwater movement, which will ultimately govern the migration of chemicals from a source. Accordingly, the acquisition of site information was undertaken for this study is presented in Chapter Three.

CHAPTER THREE

Characterisation of the study area

3.1 Introduction

The characteristics of the study area presented in this chapter relate primarily to the groundwater system, specifically the irrigation activity, local hydrogeology, and natural and induced groundwater flow regimes. These factors are considered to be the key parameters controlling groundwater movement and ultimately chemical migration (Jaffe and Dinovo 1987, Todd 1980). Several other factors are also described in this chapter including the topography, climate, soils, and deep groundwater abstraction activities occurring near the irrigation farm. Factors in this latter group are important due to the indirect influence they have on the groundwater system.

3.2 Description of the study area

The study area encompasses a ~20 km^2 area 0.5km north of Cambridge (Figure 3.1). The wastewater irrigation farm (Bardowie farm) is located at the southern end of the study area and consists of 140 hectares of relatively flat farmland, of which 110ha is utilised for wastewater irrigation when buffer zones are excluded. Bardowie farm is also used for dairy and dry stock farming, the predominant landuse in the study area.

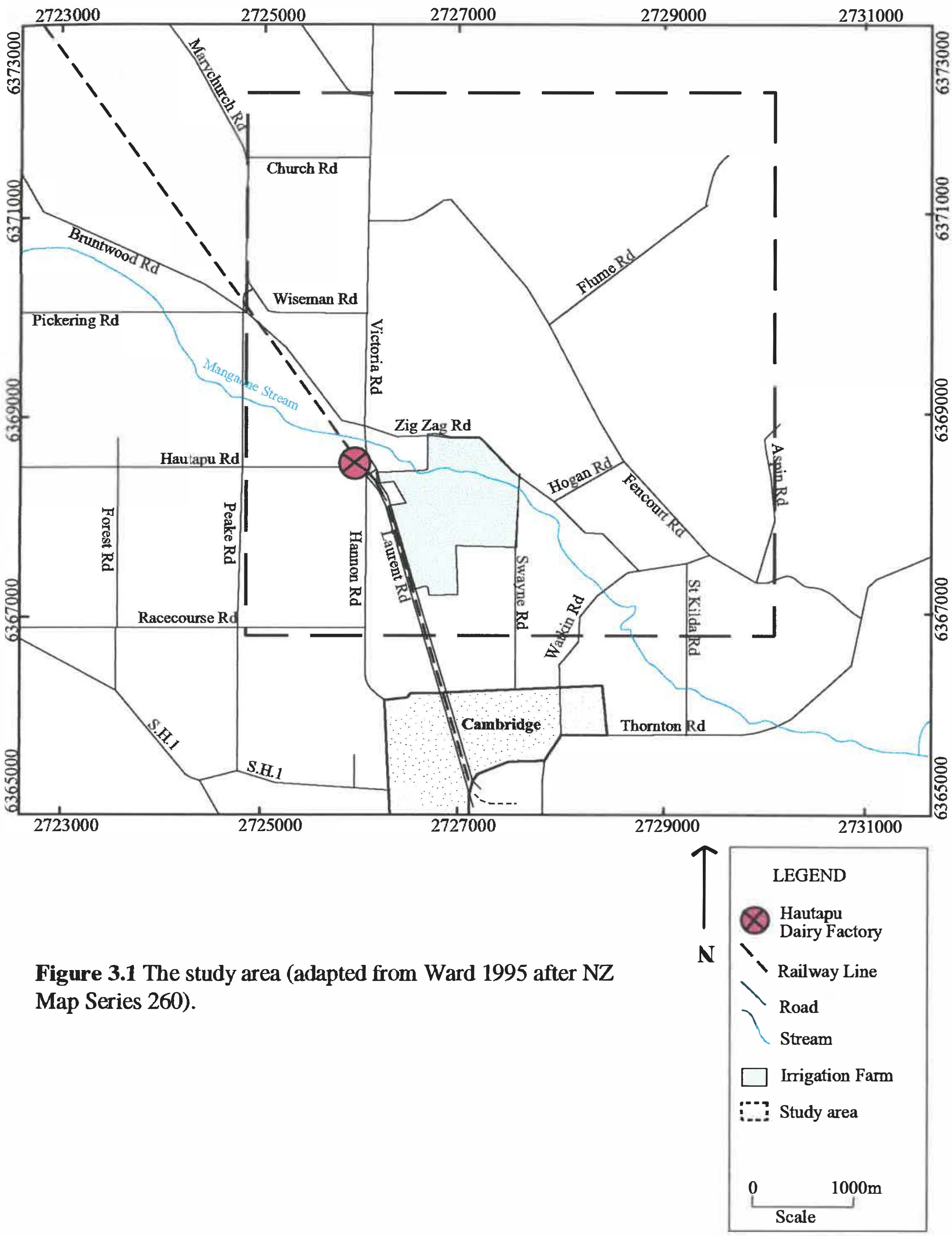


Figure 3.1 The study area (adapted from Ward 1995 after NZ Map Series 260).

3.3 Climate

Warm humid summers and mild winters characterise the Hamilton Basin where the study area is located (de Lisle 1967, Maunder 1973). Monthly means were calculated for a number of climate variables based on long term data collected at Ruakura Climate Research Station, ~13 km Northwest of the study area (Appendix I).

An average annual rainfall value of 1188.7 mm characterises the study area. Maximum rainfall occurs in winter (July ~124.0 mm), while minimum rainfall occurs in summer (February ~78.6 mm) (Figure 3.2). An annual evaporation average of 964.9 mm, based on raised pan measurements at Ruakura represents the study area, with evaporation highest in summer (January ~149.0 mm) and lowest in winter (July ~20.8 mm) (Figure 3.2).

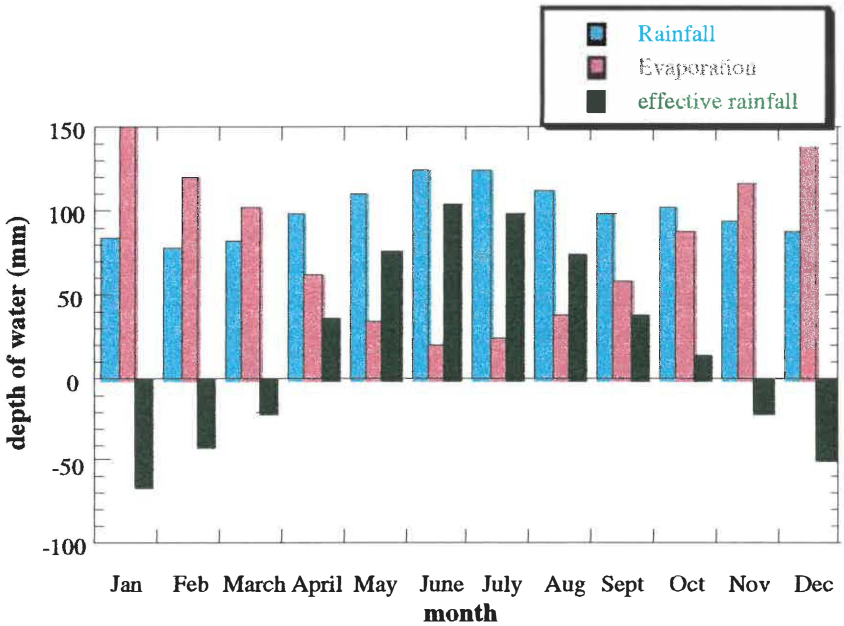


Figure 3.2 Mean monthly rainfall, evaporation, and effective rainfall in the study area.

3.4 Soils

Based on the highly permeable nature of the soils predominating in the area, Bardowie farm is suited to the application of large volumes of wastewater (Figure 3.3). The farm is 70% composed of sandy, friable, free draining soils including the Horotiu and Tamahere soils. These soils were developed from coarse alluvial material deposited by the ancient Waikato River channel system, during times of rapid current flow. Drainage rates to the shallow unconfined aquifer are expected to be high in areas where these soils occur due to the higher saturated hydraulic conductivities associated with coarse textured soils (McLaren and Cameron 1996). In contrast, in the remaining areas of the farm where fine textured soils are found, drainage rates to the shallow unconfined aquifer are predicted to be low (Te Kowhai, Eureka and Bruntwood series).

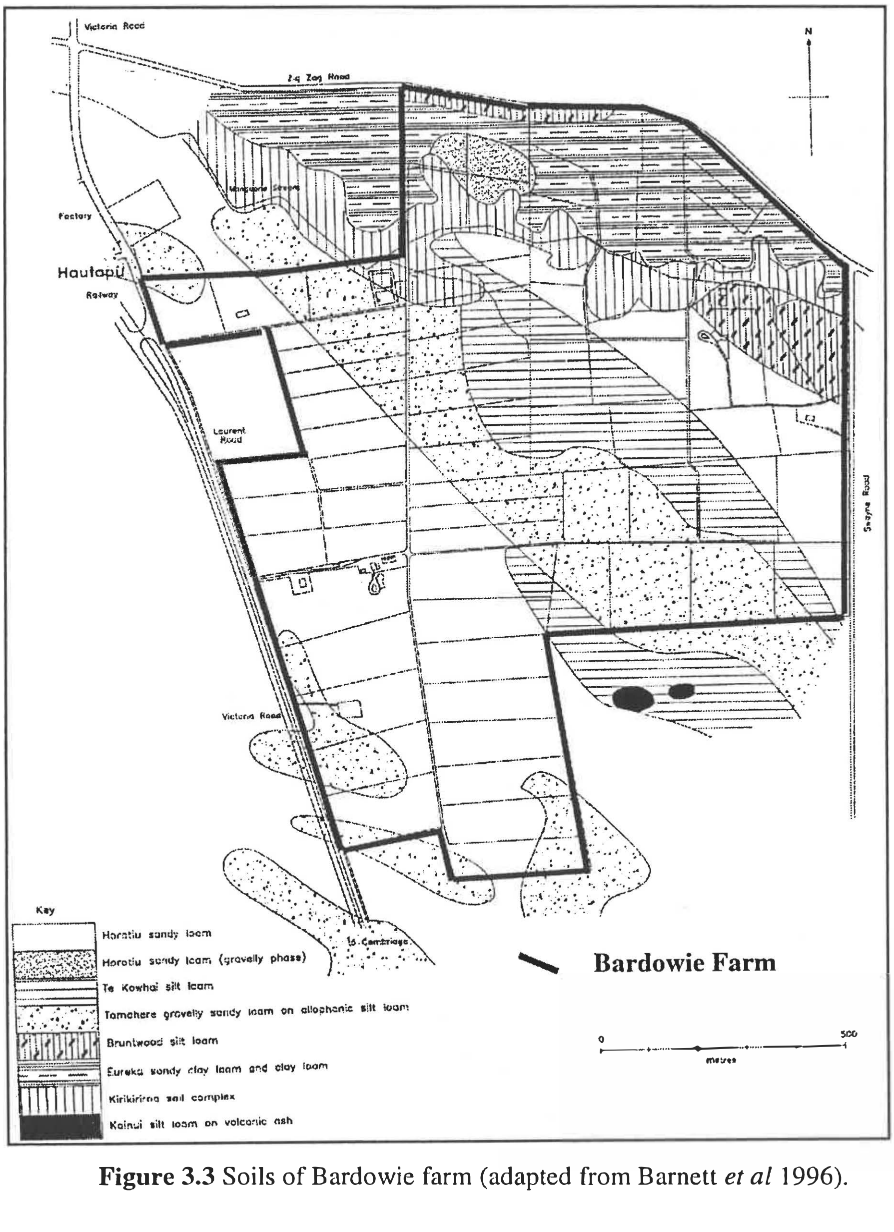


Figure 3.3 Soils of Bardowie farm (adapted from Barnett *et al* 1996).

3.5 Topography

In the lowland areas of Hamilton basin where the study area is located, a large low angle alluvial fan (Hinuera Surface) descending northwestward determines the topography of this region (Schofield 1965). The topography within the study area is relatively flat (average slope between 0-3° according to Crowcroft, 1992) and slopes gently northwestward. Monitoring bore information (Appendix II) was used to generate a contour map of topography on and near the wastewater irrigation farm (Figure 3.4), with interpolation between data points achieved through the kriging[2] function on 'SURFER'. Along the course of the Mangaone stream elevations were inferred based on elevations at nearby sites and the average incision depth of the stream (~2m according to Crowcroft 1992). As illustrated in Figure 3.4, the topography of the irrigation farm and immediate area is relatively flat.

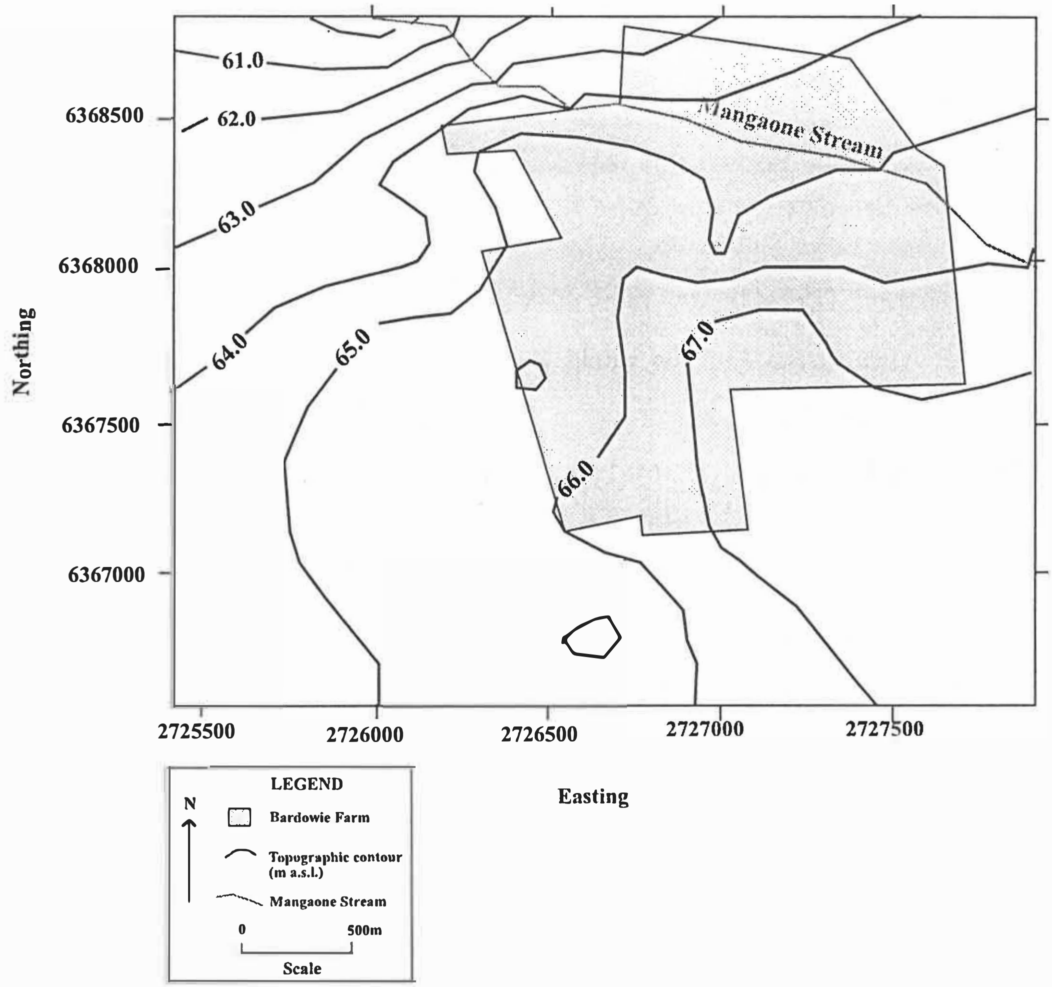


Figure 3.4 Topography within the area defined by the monitoring bore network.

[2] A geo-statistical procedure commonly used to interpolate spatial data.

3.6 Wastewater irrigation at Bardowie farm

Dairy factory wastewater irrigated over Bardowie farm is generated from the manufacture of cheese, casein, whey protein concentrate, milk protein concentrate, milk handling, and plant cleaning operations at the Hautapu factory. Consequently, the main constituents in the wastewater are nutrients such as organic nitrogen, phosphorous, and potassium and ions including sodium and chloride. While the chemical composition of the wastewater varies depending on the dairy products being produced at the factory, generally the composition is consistent with that indicated in table 3.1.

Wastewater Characteristics	
PH*	4.5-6.0
COD*	4108
BOD_5**	8000
ASH%*	0.153
Conductivity (μScm^{-1})*	327.3
K (ppm)*	88.9
P (ppm)**	50
Organic Nitrogen (TKN) (ppm)*	100.6
Na (ppm)*	415

Table 3.1 Concentrations or levels of chemical parameters characterising wastewater irrigated onto Bardowie farm. Note * Data obtained from database at the Hautapu Dairy Factory, ** data obtained from Barnett and Upchurch (1992).

Wastewater is piped from the manufacturing plants within Hautapu dairy factory to a primary treatment area, and finally to three large silos on Bardowie farm where it is stored prior to being pumped throughout the farm through the irrigation system (Figure 3.5). The wastewater irrigation system at Bardowie farm consists of an in-ground sprinkler network involving three main pipelines and sub-main pipelines passing through the center of each paddock. Lateral pipes, with a number of sprinkler points extend from the sub-main pipelines. Each paddock is sprayed for four hours per day, for up to three days, on a 16 day rotation. The irrigation system is computer controlled and sequential spraying of several areas without operator attention is a feature of the system.

On average 765040 m^3 (0.7 m depth based on the 110 ha irrigable area) is spray irrigated onto Bardowie farm each year (Barnett *et al* 1996). During the peak production period (October-December) approximately 2900 m^3 day^{-1} (0.2 cm) is disposed of compared to ~500m^3 day^{-1} (0.04 cm) outside the production period (K.Mischewski pers. comm.).

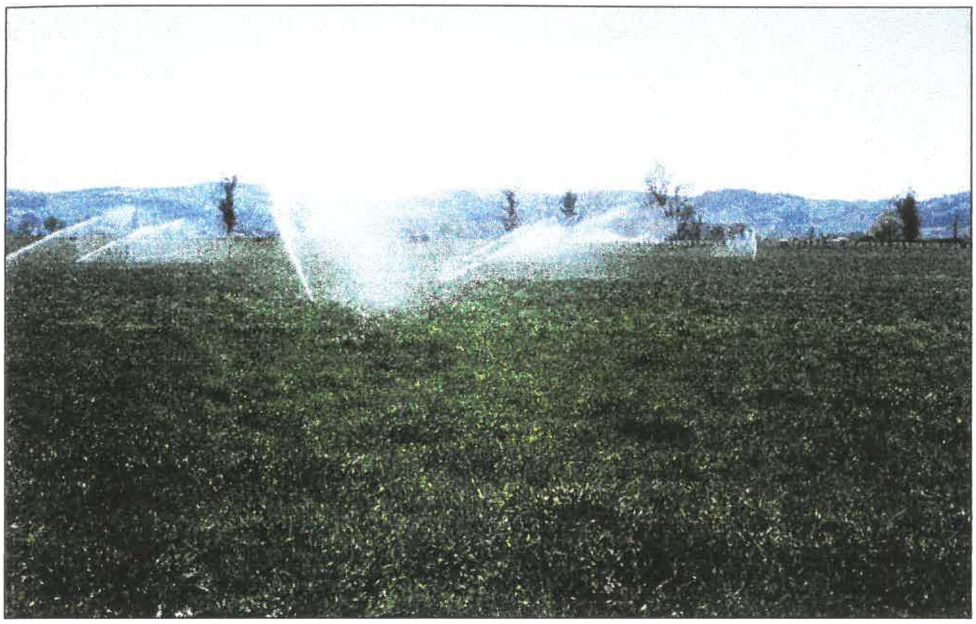


Figure 3.5 The spray irrigation system in operation (Anchor Products 1996).

3.7 The groundwater system

3.7.1 Hydrogeology

A 300m thick sedimentary sequence known as the Tauranga Group dominates the hydrogeology of the Hamilton Basin with the most recent deposits of the group (Piako Subgroup) immediately underlying the study area (Figure 3.6).

Sedimentary Sequence	Group	Formation	Lithologies
Tauranga Group (c. 300m)	Piako	Taupo Pumice Alluvium	Pumice sands and silts
		Hinuera Formation	Sand gravel clay and peat
	Walton	Karapiro Formation	Weathered sand gravel clay and peat
		Puketoka Formation	pumice sand and peat
	Frankton	Whangamarino Formation	sand clay and gravel
		Koromatua Blacksand	sand and clay
		Aberfoyle Siltstone	silts

Figure 3.6 Sedimentary sequence of the Hamilton Basin (adapted from Marshall and Petch 1985).

The Hinuera formation dominates the recent deposits. This formation is characterised by a lack of lithological continuity both vertically and horizontally. This variation is due to fluctuations in sea level, pulses in volumes of sediments, and regular changes in the channel position of the Waikato River, which deposited the sediments of the formation. The thickness of the Hinuera Formation varies spatially from 67m near Huntly (~35 km northeast of Cambridge) to 87m near Karapiro (~ 5 km west of Cambridge) (Healy 1946). The maximum thickness recorded for the formation is 140m (Schofield 1972). Within the Hinuera formation, a shallow unconfined aquifer overlies confined and semi-confined deeper aquifers. In general, the lithologies of the Hinuera formation are 'poorly sorted' consisting of rhyolitic and pumiceous gravelly-sands, low permeability silts and interbedded peats (Marshall and Petch 1985).

The lithological cross sections in Figure 3.8 illustrate the lithologies of the Hinuera formation across Bardowie farm. These cross sections were produced through the correlation of stratigraphic units determined by Petch (1988), based on deep bore logs (Figure 3.7).

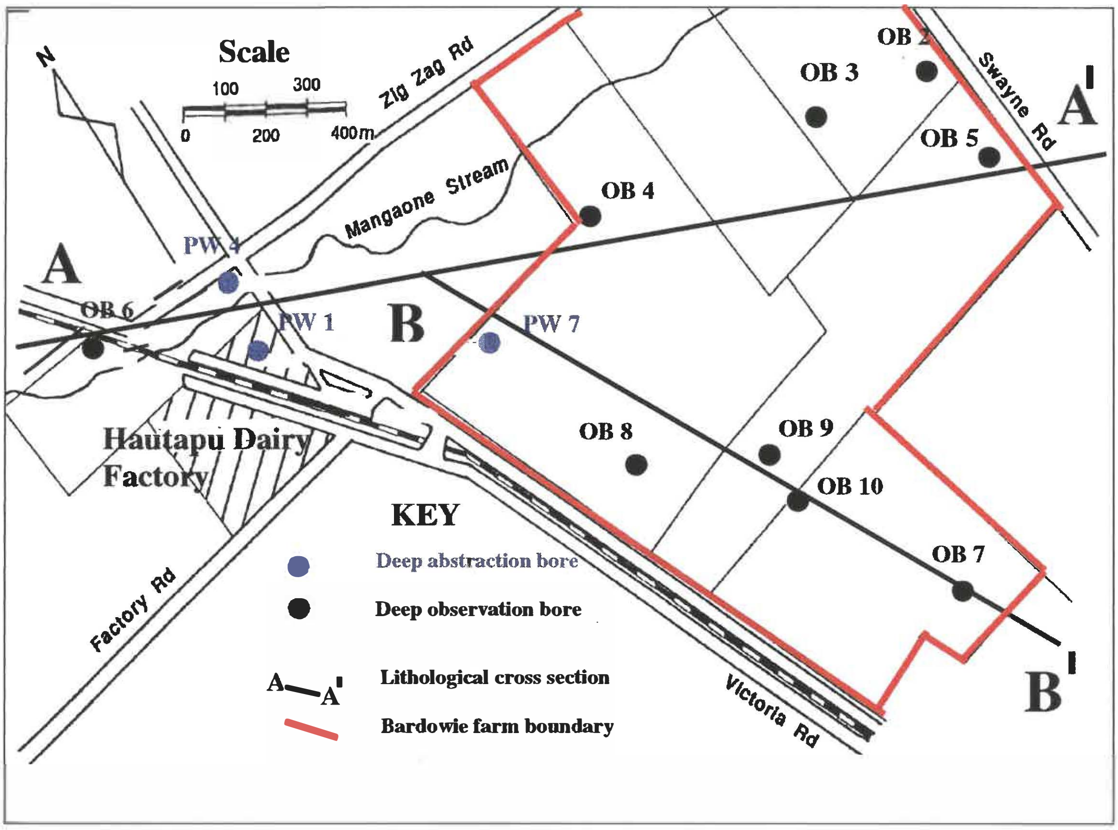


Figure 3.7 Location of deep bores used to construct the lithological cross sections (adapted from Petch 1988).

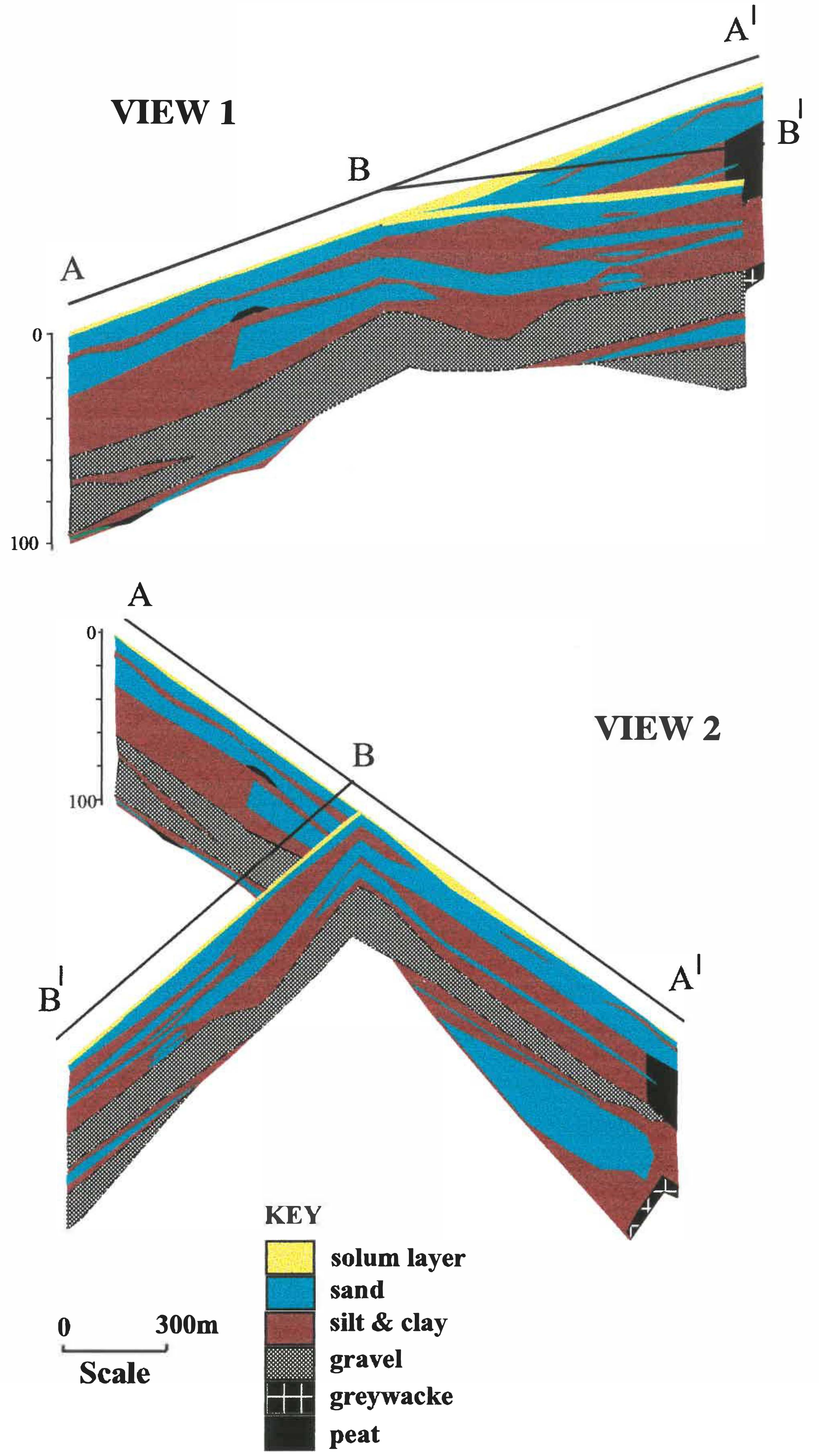


Figure 3.8 Lithological cross sections through Bardowie farm (adapted from Petch 1988).

As expected, a high degree of lithological variation occurs vertically and laterally beneath the irrigation farm. The shallow unconfined aquifer is denoted by the first significant (blue) layer and is ~30m thick, however its thickness varies spatially. The vertical extent of the low permeability confining silt/clay layers (red) also varies spatially. This is important to consider with regards to vertical water fluxes.

3.7.2 Groundwater flow

Hydraulic conductivities

The hydraulic conductivity (K) of geologic materials comprising the groundwater system is an important parameter controlling groundwater flow rates (Todd 1980). The hydraulic conductivity of the lithological material represents its ability to transmit water and has the dimensions of velocity (L/T). It is determined by a variety of physical factors including porosity, and the size, distribution, and arrangement of the particles (Todd 1980).

A hydraulic conductivity value of 2.6 m day^{-1} characterises the shallow unconfined aquifer, using the estimation technique[3] (Ward 1995) and according to other studies (Hadfield pers. comm.). The hydraulic conductivity of the unconfined aquifer is low and is a consequence of the poorly sorted nature of the sediments comprising this layer (Marshall and Petch 1985). A value of 0.5 m day^{-1} has been reported for the silt/clay materials (Marshall and Petch 1985), 5 times smaller than the hydraulic conductivity of the unconfined aquifer. In contrast, a K value of 11.4 m day^{-1}, 4.5 times larger than the K value for the unconfined aquifer, is assumed for the deep aquifer, based on in situ tests on gravel materials of the Hinuera formation (Marshall and Petch 1985). The assumed value for the deep aquifer is reasonable based on the application of the estimation technique, using an average transmissivity value of 542.5 $m^2 day^{-1}$ obtained on the irrigation farm (Petch, 1988), and assuming an average aquifer thickness of ~40m (refer to Figure 3.8).

The average values reported are not assumed to represent all areas within each lithological layer as hydraulic conductivity will vary considerably due to vertical and horizontal changes in sediment texture that occur within the Hinuera formation. However, the average values do give an indication of the likely rates of movement in each layer. It is also noted that anisotropic conditions characterise the Hinuera

[3] *$K = T/B$ where T is transmissivity (m^2 day^{-1}) and B is the thickness of the aquifer (m).*

formation, such that hydraulic conductivities in the vertical direction are generally lower than in the horizontal direction.

Horizontal hydraulic head gradients

The hydraulic head gradient determines the direction of groundwater flow, with flow always occurring in the direction of decreasing head. In this section, horizontal head gradients and inferred directions are defined for the unconfined aquifer.

Horizontal head gradients within the shallow aquifer were estimated by equating horizontal gradient to the water table gradient (Figure 3.10). For most of the monitoring bores, the following data are available: northing and easting co-ordinates, elevation, and long term average water table elevations (Appendix II). For the recently installed bores however, a water table elevation average based on 1998 data was used (Appendix II). The bore information was used to generate a contour map of the water table with data interpolation achieved through the kriging function on 'SURFER'.

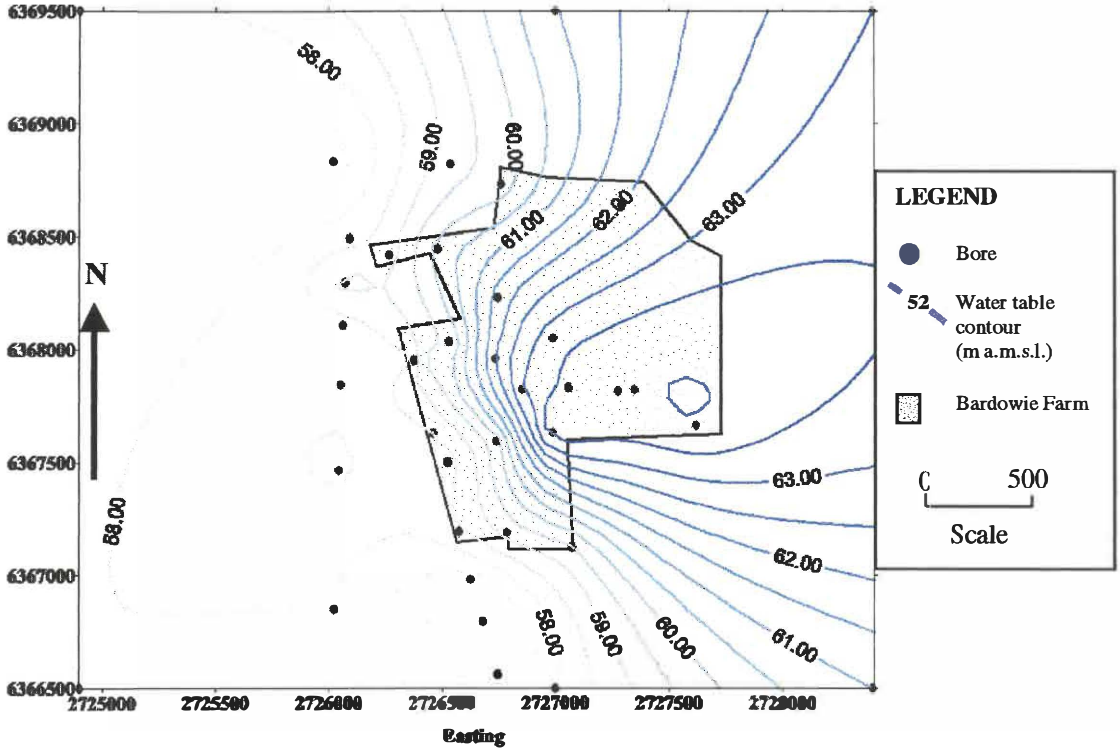


Figure 3.9 Water table configuration within the area defined by the monitoring bore network.

Overall, the water table configuration for the area defined by the monitoring bore network closely matches the pattern of topography (Figure 3.4), with a region of high static head east of the farm. Water table gradients tend to the south, west, and northwest toward the incised Mangaone stream, implying shallow groundwater will flow in these directions. The northwesterly shallow groundwater flow pattern observed is typical of the Mangoanua/Mangaone catchment (Figure 3.10) (Crowcroft 1992). Therefore, based on the locality of the study area with respect to the regional groundwater flow pattern, northwestward flow probably predominates within this area.

Overall the water table gradient within this area is low, averaging 0.003. This value is similar to the average value of 0.002 reported for the Mangaonua/Mangaone catchment (Crowcroft 1992).

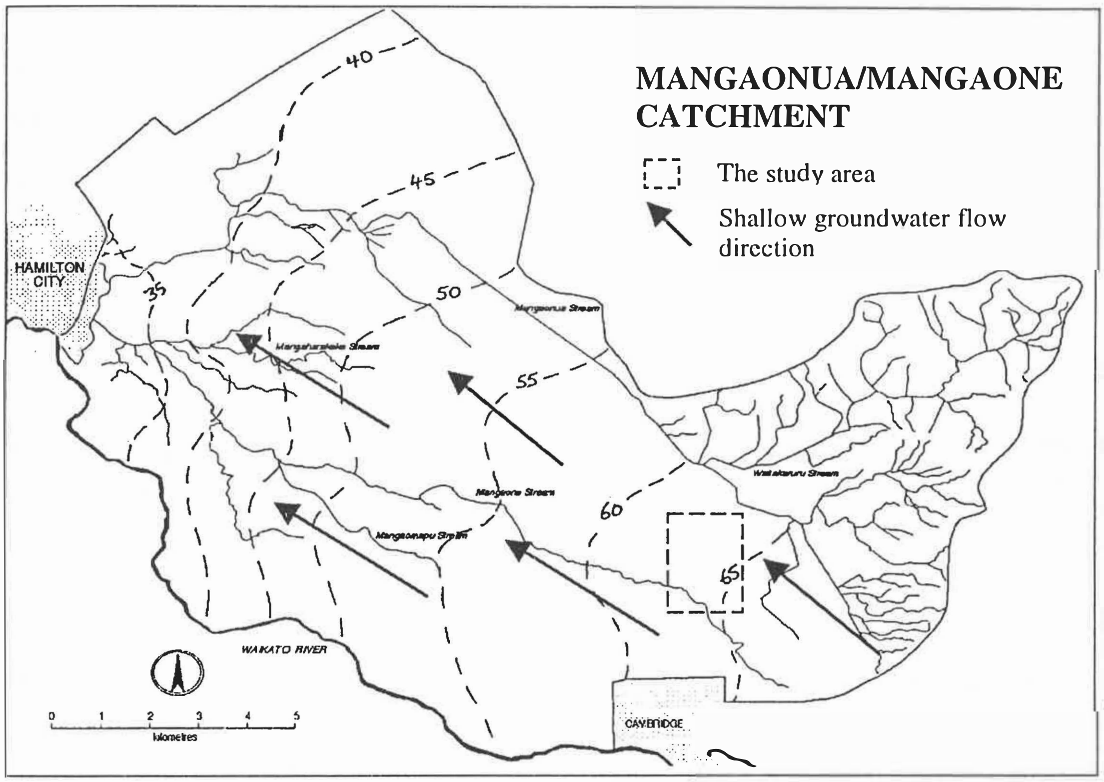


Figure 3.10 Water table contours and direction of shallow groundwater flow within the Mangaonua/Mangaone catchment (adapted from Crowcroft 1992).

Vertical hydraulic head gradients

Within the Hinuera formation there is a general trend towards lower static head values with increasing depth. From this relationship, it is inferred that upper aquifer zones recharge underlying aquifer zones (Marshall and Petch 1985). Vertical head gradients are estimated to be between -0.4 and -0.8 within the Hamilton Basin (Crowcroft 1992), significantly (100-300 times) greater than horizontal head gradients within the study area.

Vertical movement of groundwater is thought to be generally restricted to the upper 30m zone (Marshall and Petch 1985). However, movement of shallow groundwater to deeper aquifers has occurred in some areas within the Hamilton Basin, presumably where the silt/clay layer is thin or discontinuous. At Matangi (~10 km northwest of Cambridge), recent water derived from the shallow unconfined aquifer was detected in deep groundwater samples (>65m), based on environmental isotope work by Marshall and Petch (1985). These workers postulated that downward movement in this area was induced by deep groundwater abstraction by the now closed, Matangi dairy factory. In comparison at Hautapu, where large quantities of groundwater are abstracted from deep aquifer zones, induced recharge from the shallow aquifer to the deeper aquifers does not appear to have occurred. Old water (60-80 yrs old) was determined in the deep aquifer in the region of the Hautapu well field in 1985, and again in 1998 (Terra Aqua Consultants 1998).

While movement of recent water from shallow zones was not observed within the Hautapu well field, induced movement in other areas cannot be discounted especially where the silt/clay confining layer is either thin or discontinuous. The possibility of induced vertical movement from deep groundwater abstractions is addressed in detail in Section 3.8.

Groundwater flow velocities (V)

Groundwater flow velocity is a useful measure of the duration required for groundwater to travel between two points. For this reason, estimated and measured hydraulic parameters (K, head gradients, porosity) were used to calculate horizontal groundwater flow velocities[4] (V). Porosity values of 0.4 and 0.3, based on values reported for various lithologies in Todd (1980), were assumed for the shallow and deep aquifers respectively. According to the calculations, the estimated flow velocities for the unconfined and confined aquifers are 0.013 m day^{-1} and 0.087 m day^{-1} respectively. These are only estimates of flow velocity as lithological variations both vertically and horizontally imply variable rates of flow within each zone.

[4] $V = Q/n$ where $Q = K(dh/dl)$. V is flow velocity (m day^{-1}), Q is discharge rate (m day^{-1}), n is porosity (dimensionless), K = hydraulic conductivity (m day^{-1}), and dh/dl is the hydraulic head gradient (dimensionless).

Lower vertical flow velocities are anticipated for the groundwater system beneath the irrigation site, in comparison to the estimated horizontal flow velocities. This is based on the hydraulic conductivity of the silt/clay materials that comprise the hydrogeology of the groundwater system, and despite the presence of strong vertical head gradients.

3.7.3 Site annual water balance.

Using an annual water balance equation and Darcys Law, an estimate of the boundary head gradient required to export lateral flow from the shallow unconfined aquifer was calculated.

The average volume of groundwater outflow (Q) generated at the site each year, was firstly calculated using the following site annual water balance equation:

$$R + I - E = Q$$

Where:

The total area of the farm is 1.4×10^6 m^2, and the irrigable area is 1.1×10^6 m^2;

R is annual rainfall (1.68×10^6 m^3 yr^{-1});

I is annual irrigation (7.7×10^5 m^3 yr^{-1}); and

E is annual evaporation (1.34×10^6 m^3 yr^{-1}).

Then, using Darcy's Law:

$$Q = K(dh/dl)$$

Where:

Q is groundwater outflow (1.1×10^6 m^3 yr^{-1});

K is hydraulic conductivity (949 myr^{-1}); and

dl is the perimeter length of the site (1183m).

According to the calculation, a head gradient of 0.9 is required to export this volume, 300 times the actual head gradient for the unconfined aquifer (0.003). This estimation relies on the assumption that vertical flow is negligible at the site.

3.7.4 Seasonal fluctuations

Shallow and intermediate water levels in the Cambridge-Hamilton region fluctuate seasonally, as illustrated in Figure 3.11. Overall minimal seasonal variation in water levels is observed. Levels typically decline to an annual low over the period May-July followed by a recovery to a peak over the period October-December. This cycle coincides with the seasonal pattern of rainfall infiltration coupled with a delay in time. Recharge occurs in late winter-spring when excess soil moisture conditions lead to infiltration of excess water to saturated zones.

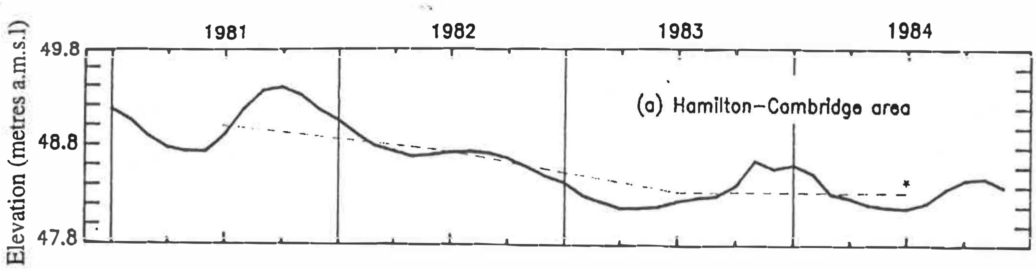


Figure 3.11 Seasonal variation of piezometer levels in the Hamilton-Cambridge area (1981-1984) (from Marshall and Petch 1985). Levels are averages based on data from shallow and intermediate bores.

Seasonal changes in water table levels are not expected to independently effect significant changes in water table gradients, therefore horizontal movement of shallow groundwater. However, when combined with patterns of artificial recharge rates, seasonal fluctuations in the water table can lead to changes in the slope of the water table, hence lateral flow. On-site, artificial recharge causes departure from the natural pattern of recharge, with levels lowest in May and recovering rapidly to August (Figure 3.12a). As a consequence of modified recharge patterns on the farm a head gradient maximum between off-site and on-site areas arises in August (Figure 3.12b), when levels are high on-site but are still recovering off-site. Accordingly, it is inferred that horizontal movement of groundwater from the farm to off-site areas down gradient will be greatest in August. It is noted that in March, head gradients are at a minimum implying reversed flow.

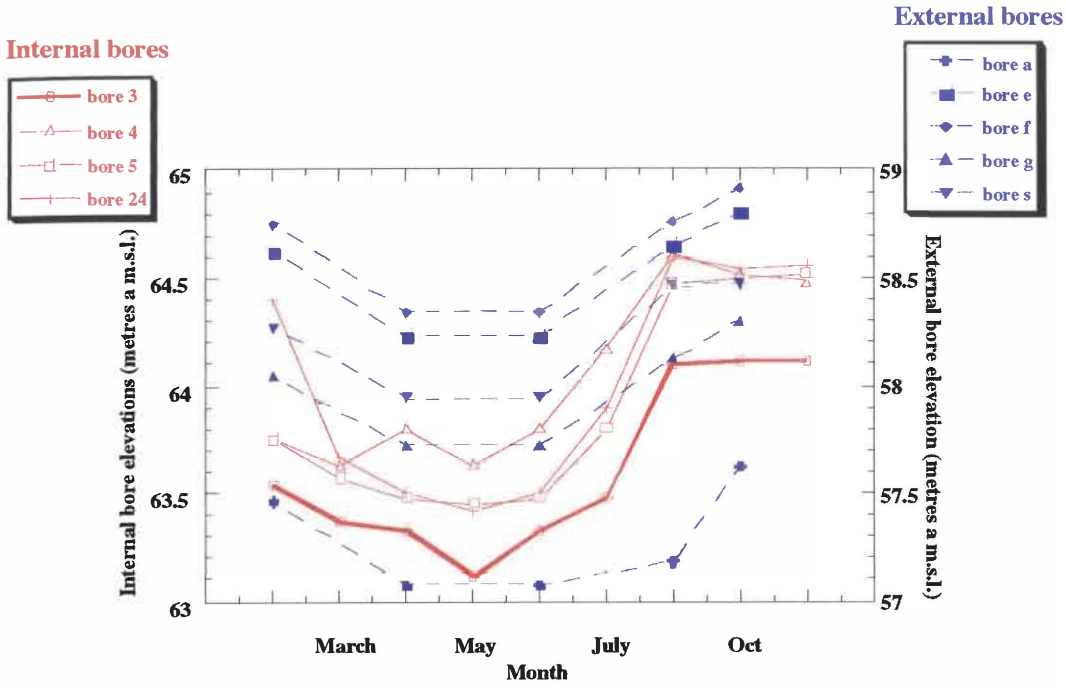


Figure 3.12a Average monthly water levels in monitoring bores for 1998.

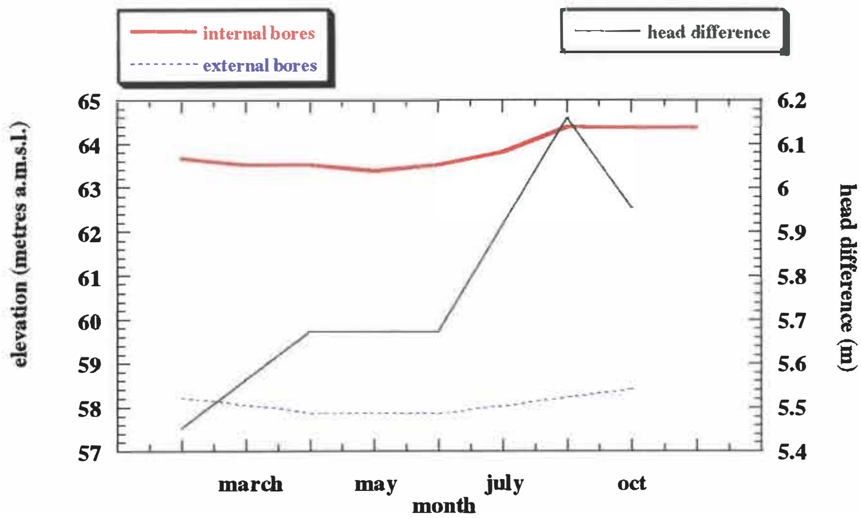


Figure 3.12b Average monthly water levels in off-site and on-site bores for 1998 and average head difference.

3.8 Groundwater abstraction

Groundwater is presently abstracted from five bores (~65-90m) in the deep aquifer to supply clean water for the Hautapu dairy factory. The abstraction bores are located near the factory, ~ 0.5km from the Bardowie farm (Figure 3.13). The bores pump at varying rates during the production season, and rates for the various bores differ between seasons. The average pumping rates for the bores are listed in Table 3.2.

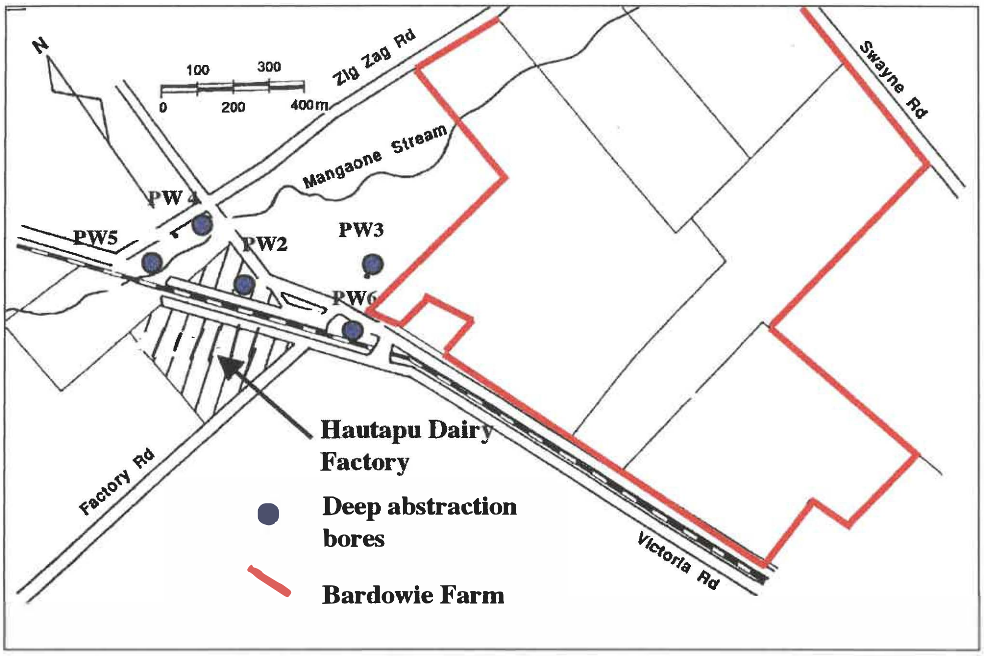


Figure 3.13 Location of deep abstraction bores currently used by the factory (adapted from Anchor Products 1996 after Petch 1988).

Abstraction bore	*Average pumping rate (m^3day^{-1})*
PW 2	369
PW 3	719
PW 4	483
PW 5	902
PW 6	249

Table 3.2 Average pumping rates for the deep abstraction bores (1991-1996) (Anchor Products 1996).

Over the period 1991-1996 total water abstraction averages 2900 m^3day^{-1} with rates typically highest during the dairy season between mid-July and May (Anchor Products 1996). Typically water levels in the abstraction bores decline during the abstraction period (Figure 3.14).

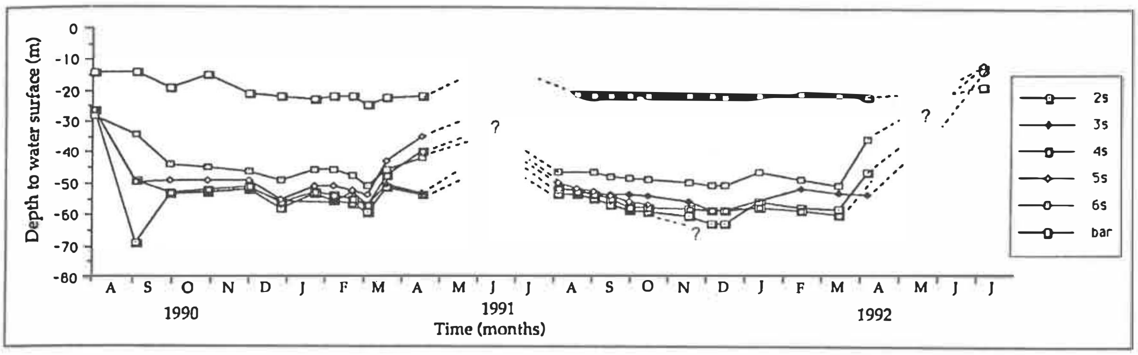


Figure 3.14 Water levels in abstraction bores for the period 1982 – 1984 (after Crowcroft 1992).

According to the Theis theory relating to wells (Todd 1980), pumping will result in an extended drawdown cone. Therefore, induced vertical movement may occur in areas at distance from the well field depending on the areal extent of the drawdown effect.

To determine the areal extent of drawdown, therefore what areas might be affected by induced movement, a drawdown contour plot centred on the production bore field was generated (Figure 3.15). The contour map was created through interpolation between drawdown levels calculated for the deep observation bores (Table 3.3), and the abstraction bores (Figure 3.14).

Observation bore	*Upper water level (Petch 1988)*	*Lower water level (Petch 1988)*	*Jan-99*
OB 4	15	30	19
OB 7	15	20	24
OB 8	10	30	20

Table 3.3 Water levels (m below ground surface datum) in deep observation bores on Bardowie farm as recorded in January 1999 and as reported by Petch (1988).

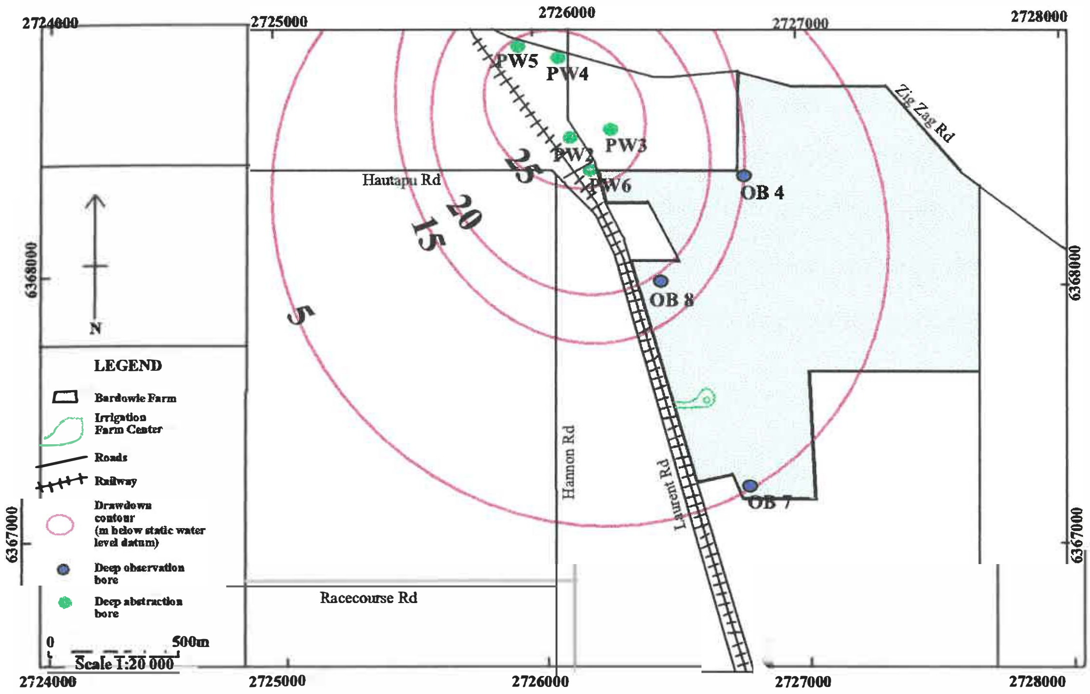


Figure 3.15 Drawdown interpolation contours for the deep aquifer based on abstraction and observation bore data.

The plotted results indicate head levels in the deep aquifer may be lowered between 5-20m on the irrigation farm, indicating the possibility of induced vertical flow within this area. However, the thickness and continuity of the low permeability silt/clay layer will ultimately govern the extent of vertical movement.

3.9 Summary

Having described the important physical aspects or site conditions relating to the groundwater system, several qualitative deductions can be made about the groundwater system and flow dynamics within the study area. Regimes within the shallow aquifer, directly affected by the irrigation activity, are summarised. Horizontal movement of groundwater within the shallow unconfined aquifer is notably slow (0.013 m day^{-1}) and is predominantly northwestward according to:

(i) Local topography;
(ii) The combined effect of natural and artificial recharge rates;
(iii) Seasonal water level fluctuations;
(iv) Water table gradients;
(v) Lateral groundwater velocities estimated for the shallow aquifer; and
(vi) Regional groundwater flow.

It appears that vertical downward movement of groundwater is minimal at the wastewater site based on the hydraulic characteristics of the silt/clay layer, despite the strong vertical head gradients resulting from the effect of deep groundwater abstractions on head levels in the deep aquifer.

Information from this chapter which relates to the groundwater system, specifically head gradients, hydraulic conductivities, hydrogeology, and deep groundwater abstractions, was incorporated into a simple numerical model in order to estimate groundwater flow patterns beneath and near the irrigation site. The simple modelling study is described in Chapter Six.

CHAPTER FOUR

The geochemical setting

4.1 Introduction

Traditionally, assessment of groundwater quality beneath the wastewater irrigation site has been achieved through geochemical studies based on data from the factory's monitoring bore network. A number of such studies have been conducted, considering variations in groundwater quality at the site through both time and space. The main conclusions of these works are incorporated in this chapter, to provide a historic view of the geochemical setting. In addition, recent data has been analysed and interpreted, and the results are presented to re-establish the geochemical setting.

Analysis of recent data in the present study was essential for:

(i) Determining the extent of modified groundwater movement, both laterally and vertically;

(ii) Developing a modified groundwater index as a potential tool for identifying modified groundwater from the wastewater irrigation site; and

(iii) Providing a ground truth for the electrical resistivity technique.

4.2 Groundwater monitoring

As mentioned, shallow groundwater quality near the irrigation site is monitored through a comprehensive bore network installed by the factory (Figure 4.1). The NZMS co-ordinates, depths and screen intervals for each of the monitoring bores are listed in Appendix III. Sampling from the bores is undertaken on either a monthly (on-site bores) or bi-monthly basis (off-site bores). Many of the wastewater constituents are also measured in the shallow groundwater; specifically concentrations of sodium, inorganic nitrogen (nitrate), chloride, phosphorous, and potassium are used by the factory as indicators of water quality.

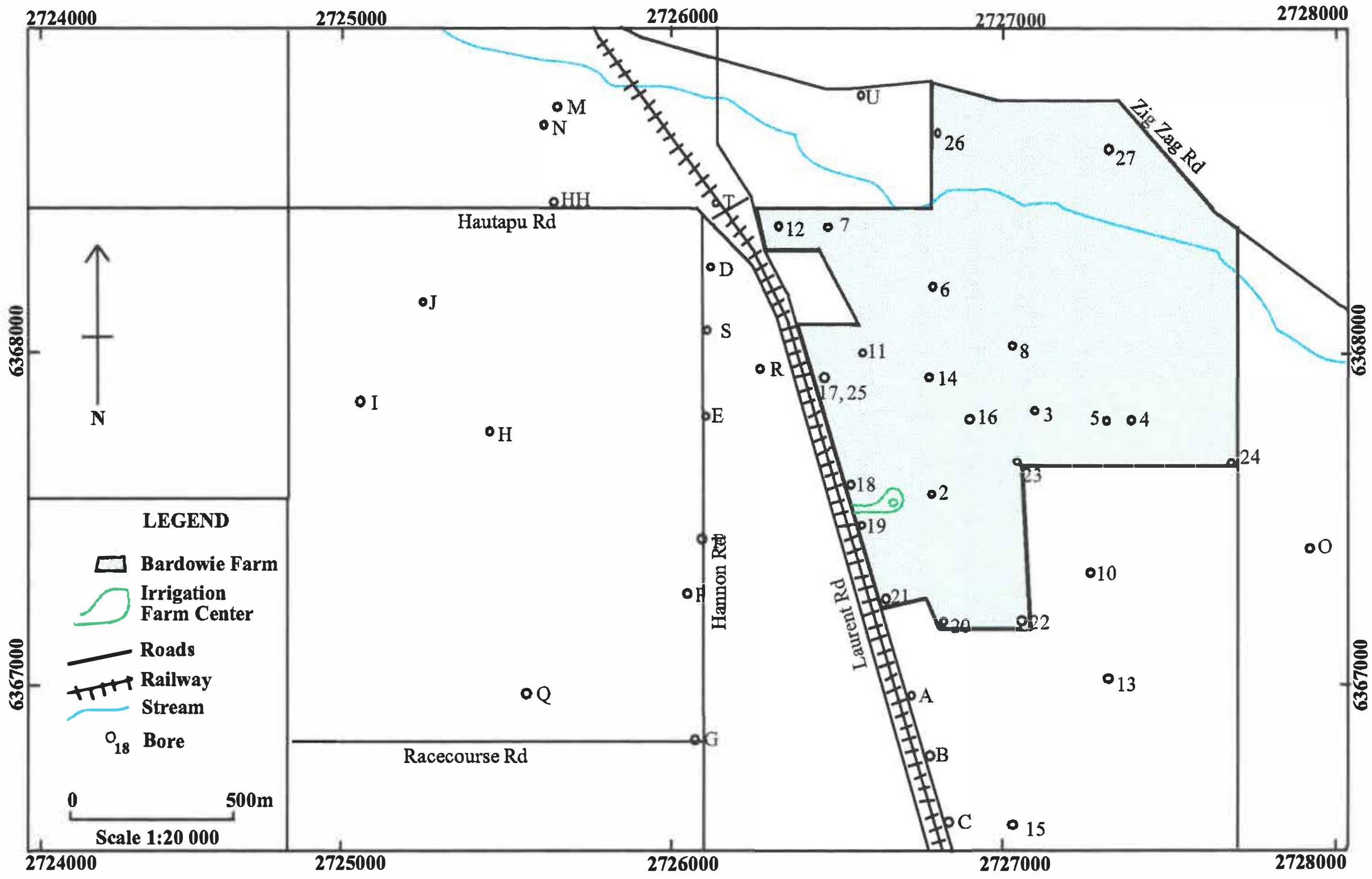


Figure 4.1 The monitoring bore network within the study area (after Ward 1995).

As mentioned, a number of geochemical studies have been conducted based on the data from the monitoring bore network. In both historical studies and the present study, particular emphasis has been placed on nitrate, sodium, and chloride levels, due to their occurrence in the wastewater, their mobility in groundwater, and the potential adverse effects associated with them. For example, nitrate levels are considered important due to the potential adverse affect of eutrophication of natural watercourses.

4.3 Previous work

4.3.1 Terra Aqua Consultants (1993)

In 1993, Terra Aqua Consultants conducted an investigation of shallow groundwater quality at the irrigation site (Bardowie farm). Long-term geochemical data, from 18 original monitoring bores located on Bardowie farm, was used for the study. The results of the time series data analysis indicated nitrate levels in shallow groundwater beneath Bardowie farm had increased significantly as a consequence of the irrigation activity, and that elevated levels of nitrate occurred in most of the monitoring bores on the irrigation farm.

4.3.2 Ward (1995)

Following on from Terra Aqua consultants, Ward (1995) carried out a first comprehensive investigation considering both the temporal and spatial variability of groundwater quality at the wastewater irrigation site. Particular emphasis was placed on nitrate concentrations, however levels of sodium and electrical conductivity were also investigated.

Background levels

In her study, Ward established background levels of the chemical parameters to calibrate with on-site levels. Background nitrate levels in the study area averaged 8.90 ppm, ranging from 1.51 ppm to 17.5 ppm and were often in excess of 10ppm. No trend was observed between levels of nitrate with depth however it was noted that at depths >11m below the ground surface, background concentrations were also often greater than 10ppm. Background levels determined for electrical conductivity and sodium levels averaged 289μScm^{-1} and 20 ppm respectively.

Temporal variation

A time series of nitrate concentration in shallow groundwater on Bardowie farm revealed nitrate levels had increased from 1982 peaking in 1991 at ~70ppm (Figure 4.2). Ward noted that since peaking in 1991 nitrate levels in shallow groundwater on the farm appeared to be decreasing but acknowledged that further long term monitoring was required to confirm this.

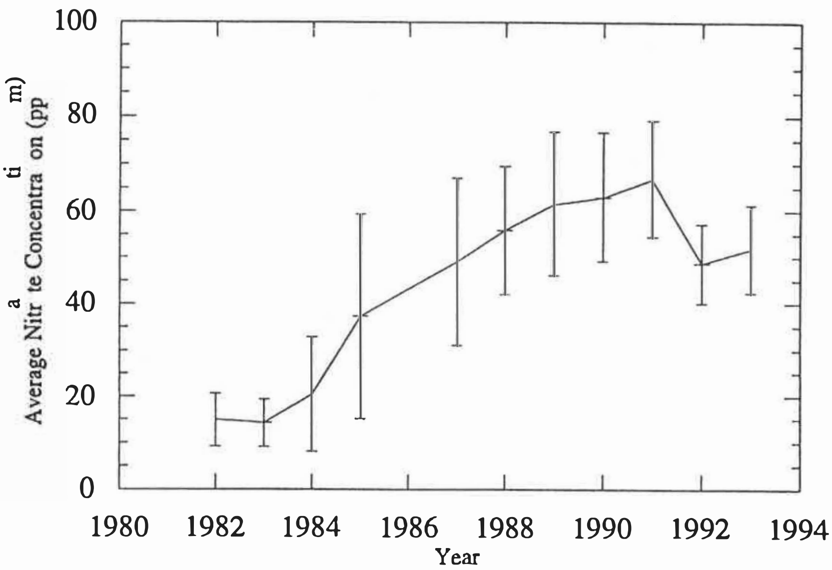


Figure 4.2 Space-averaged nitrate concentration in shallow groundwater on Bardowie farm (after Ward 1995). * Note see Figures 4.7 and 4.11 for nitrate levels since 1993.

Lateral spatial variation

In Ward's study, spatial variation of the chemical parameters was illustrated through estimated iso-chemical contour maps (Figures 4.3, 4.4, 4.5). These contour maps formed the basis for inferences made regarding movement of modified groundwater.

Based on the spatial pattern of nitrate in shallow groundwater (Figure 4.3), movement of modified groundwater appeared to be diverging away from the farm in all directions. Ward suggested that greater movement was occurring in the northwestern direction, although there was no strong evidence for this. Nitrate levels in bores near the southern end of the farm indicated modified groundwater at some stage had moved south, towards Cambridge, around the eastern side of topographic and hydraulic highs. Significant movement westward was also inferred by the fact that the largest area of elevated nitrates occurred on the western side of the farm and the 20 ppm concentration contour, used to estimate the edge of the plume, was present ~1km from the western farm boundary.

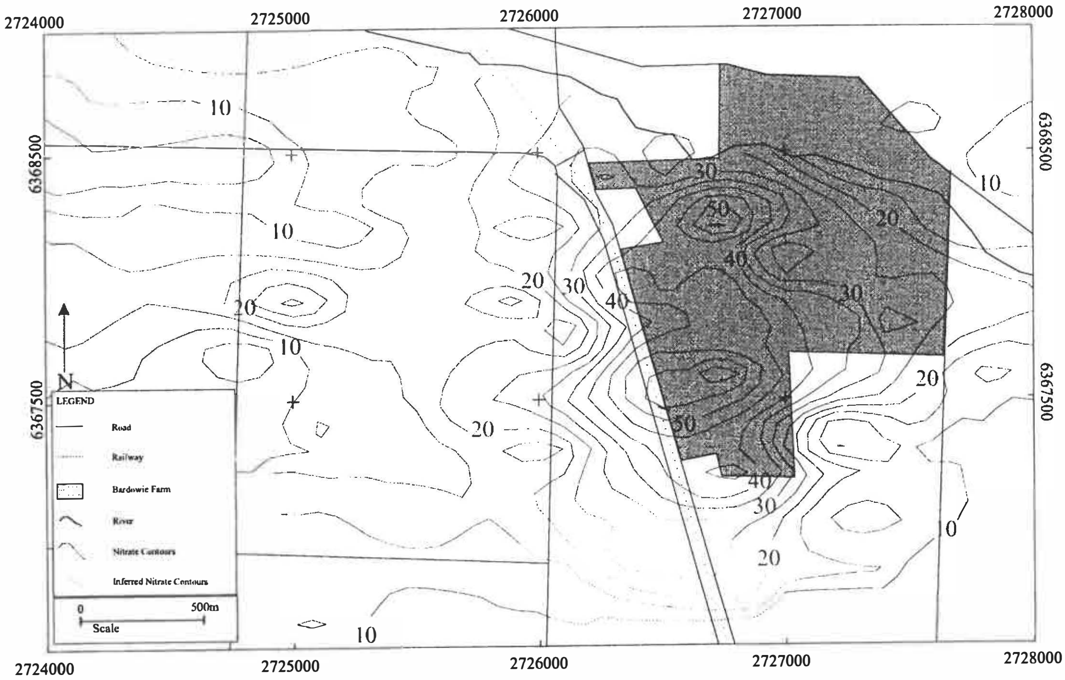


Figure 4.3 Nitrate iso-concentration map (ppm) for August 1994 (after Ward 1995).

The spatial pattern of electrical conductivity (Figure 4.4) also depicted movement of modified groundwater westward, northwestward and southwards. However, the electrical conductivity data indicated movement was greatest westward and that the southern front of the modified groundwater plume had advanced ~300m from Bardowie farm towards Cambridge. The contour map of sodium concentration in shallow groundwater (Figure 4.5) indicated movement patterns consistent with nitrate and electrical conductivity (expansion west, northwest and south) however, a modified groundwater plume of greater areal extent compared to that indicated by nitrate was revealed. This was attributed to the significantly higher concentrations of sodium in irrigated wastewater compared to organic nitrogen (Table 4.1).

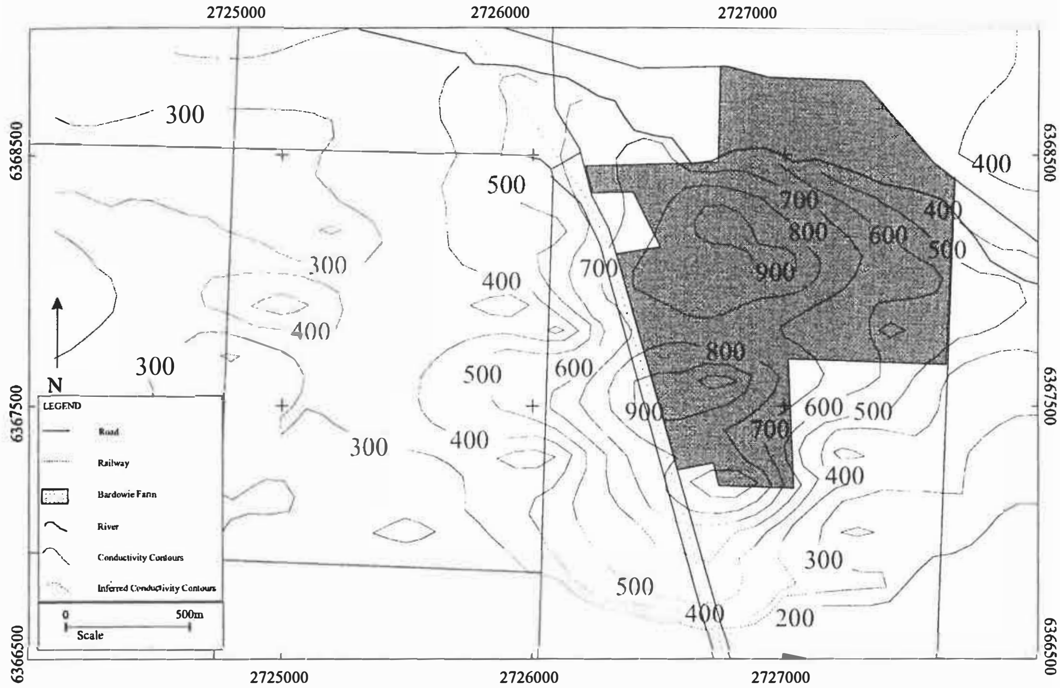


Figure 4.4 Iso-conductivity contour map (μScm^{-1}) for August 1994 (after Ward 1995).

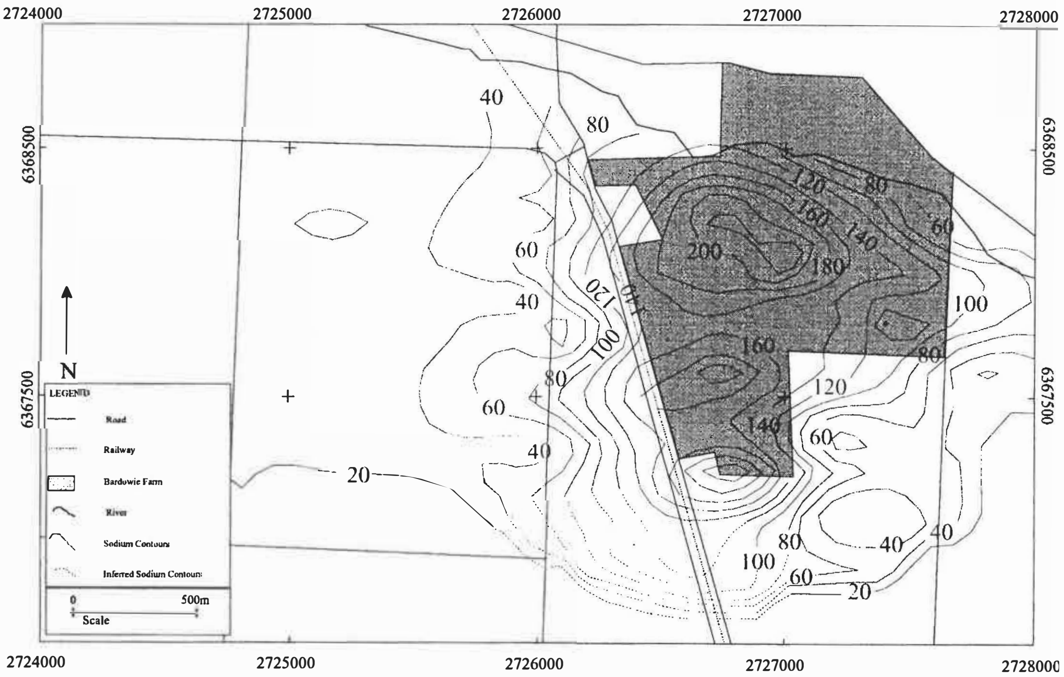


Figure 4.5 Sodium iso-concentration map (ppm) for August 1994 (after Ward 1995).

Year	*Na (ppm)*	*TKN (ppm)*
1989/1990	415	206
1990/1991	327	131
1991/1992	307	127
1992/1993	367	130
1993/1994	356	155
1994/1995	449	140
1995/1996	457	96

Table 4.1 Sodium and organic nitrogen levels in wastewater for the period 1989-1996 (after Barnett *et al* 1996).

Vertical spatial variation

To determine the vertical variation in groundwater quality, or the vertical extent of modified groundwater movement, Ward considered the vertical variation in nitrate levels. The results of the analysis revealed no trend in nitrate concentrations with depth on the irrigation farm (Figure 4.6). It was concluded in the study that the vertical extent of nitrate movement associated with irrigation at Bardowie farm could not be determined from the limited data and Ward acknowledged further work was required to ascertain the vertical thickness of the plume (an issue addressed in Chapter Four).

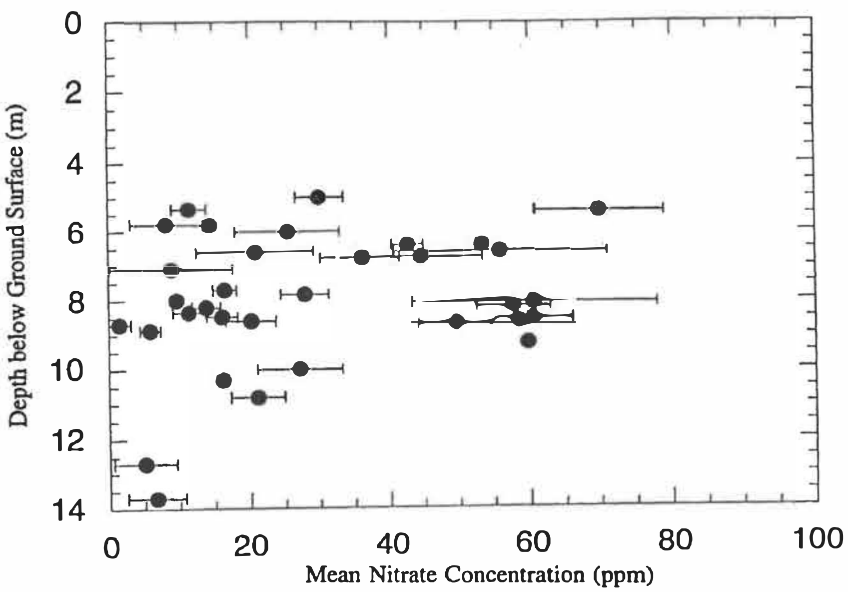


Figure 4.6 Concentration of nitrate in groundwater with depth below ground surface, on Bardowie farm (after Ward 1995).

4.3.3 New Zealand Dairy Research Institute (1996)

In 1996 Barnett *et al* (1996), re-established the geochemical setting with a study involving analysis of long-term shallow groundwater quality data from the monitoring bore network. The study re-confirmed the occurrence of elevated levels of chemical parameters characterising the irrigated wastewater, specifically nitrate, sodium, conductivity, and potassium. In addition, the study re-emphasised that, with the exception of nitrate, levels of all chemical parameters had increased steadily since 1982.

An important finding of the study was that nitrate levels in shallow groundwater had decreased steadily since 1991. The decrease in nitrate levels in shallow groundwater was attributed to a decrease in organic and ammonium nitrogen (TKN) applied to the irrigation site (Figure 4.7).

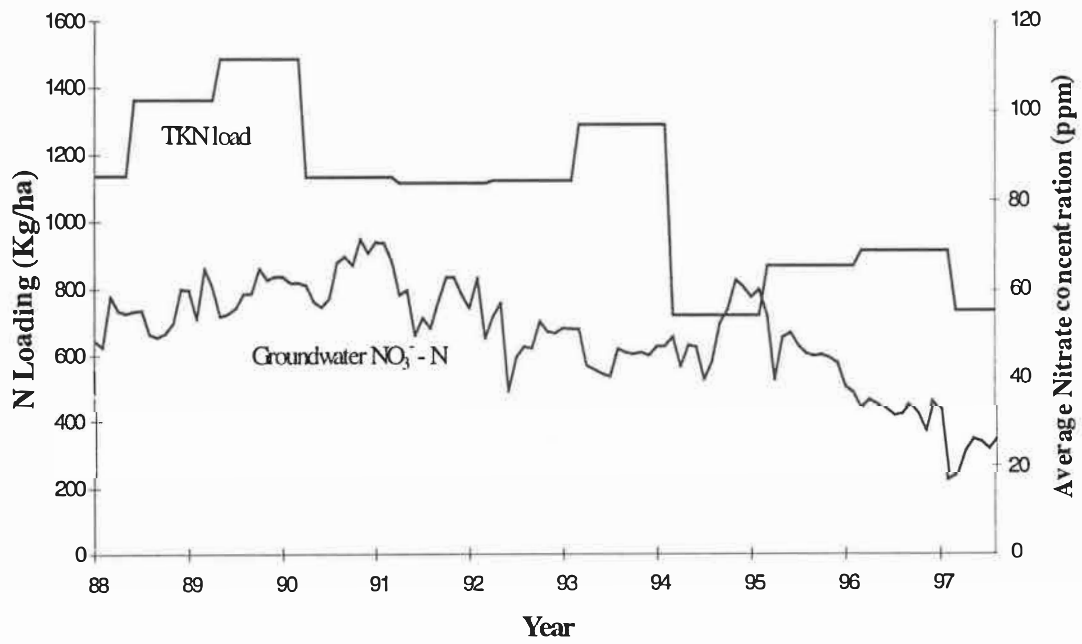


Figure 4.7 Time series of N loadings and groundwater nitrate levels on Bardowie Farm (after Barnett *et al* 1996)

4.4 Present study

Temporal changes in nitrate levels revealed in previous studies indicated it was necessary to re-establish the geochemical setting. The first stage of this investigation involved a time series analysis to detect temporal variations in levels of important chemical parameters characterising the wastewater, namely sodium, chloride, nitrate and conductivity. A current assessment of the spatial variation of the important chemical parameters was then carried out, followed by an investigation of the vertical extent of modified groundwater movement, which prior to this study had not been established.

4.4.1 Temporal variation

A time series analysis of the levels of chemical parameters characterising the wastewater, namely sodium, chloride, nitrate and electrical conductivity, was carried out for the period 1990–1998. Data from thirteen internal (on-site) shallow bores (bores 2, 3, 4, 5, 6, 7, 8, 11,12, 14, 16, 17 and 19) were used for the time series analysis for the period 1990–1998. Information concerning these bores is included in Appendix III. The thirteen bores were selected according to the availability of long-term data and because they are all located in areas subject to moderate-high irrigation rates (Figure 4.8).

Yearly data for each bore was standardised about the respective mean and standard deviation, using the following equation:

$$\text{Std value} = \frac{x - \bar{x}}{S}$$

Where $\bar{x}$ is an average of concentration through time for the respective bore, S is standard deviation for the respective bore, and x is chemical parameter level. The yearly data was then averaged to produce a space averaged concentration for each year in the period considered (1990-1998).

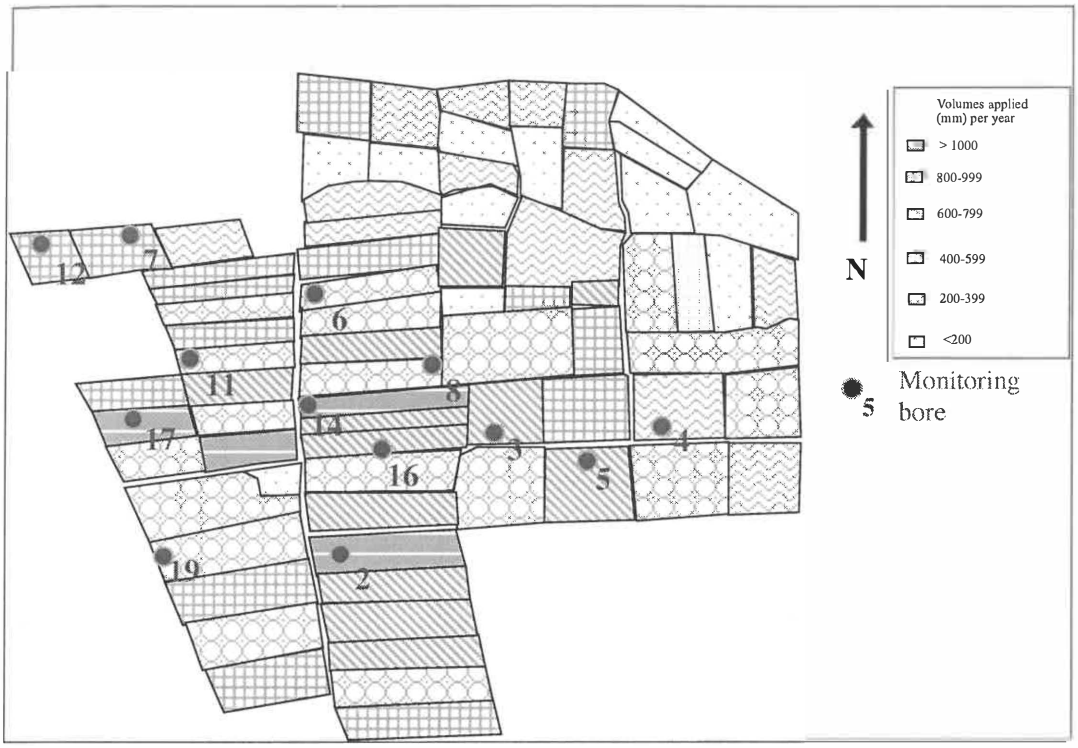


Figure 4.8 Locations of selected bores and wastewater irrigation rates per paddock (1 October 1993 to 1 September 1994). Not to scale. (adapted from Ward 1995).

The results of the analysis of groundwater conductivity revealed that conductivity has increased steadily since 1990 (Figure 4.9). This increase is strongly correlated with an increase in groundwater sodium levels during the same period (Figures 4.10a and 4.10b).

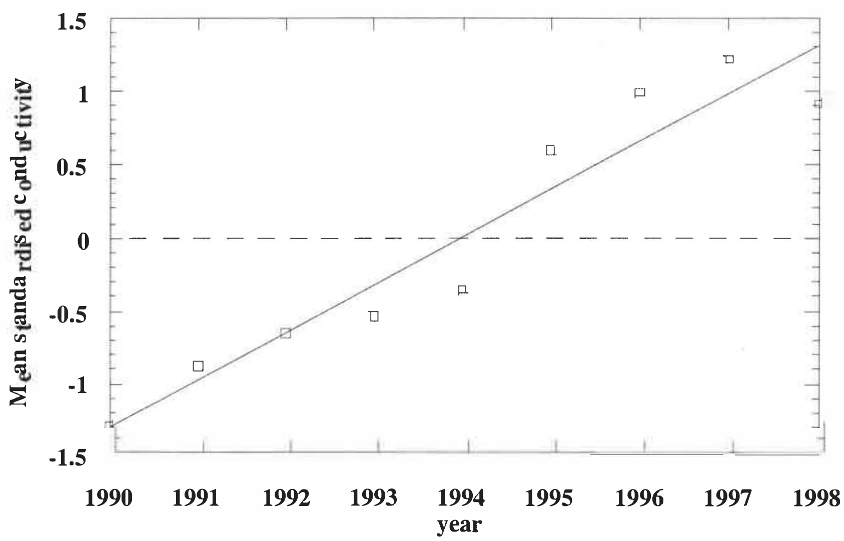


Figure 4.9 Mean standardised conductivity of shallow groundwater on Bardowie farm (1990-1998).

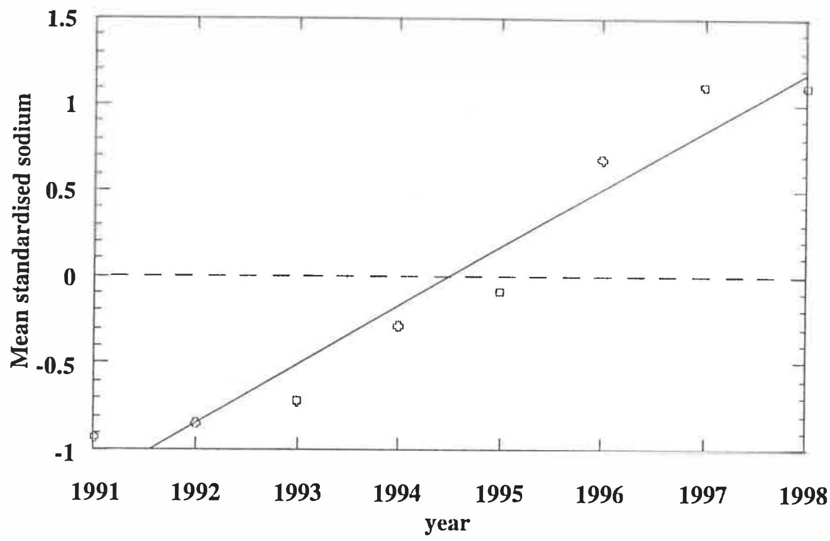


Figure 4.10a Mean standardised sodium in shallow groundwater on Bardowie farm (1991-1998).

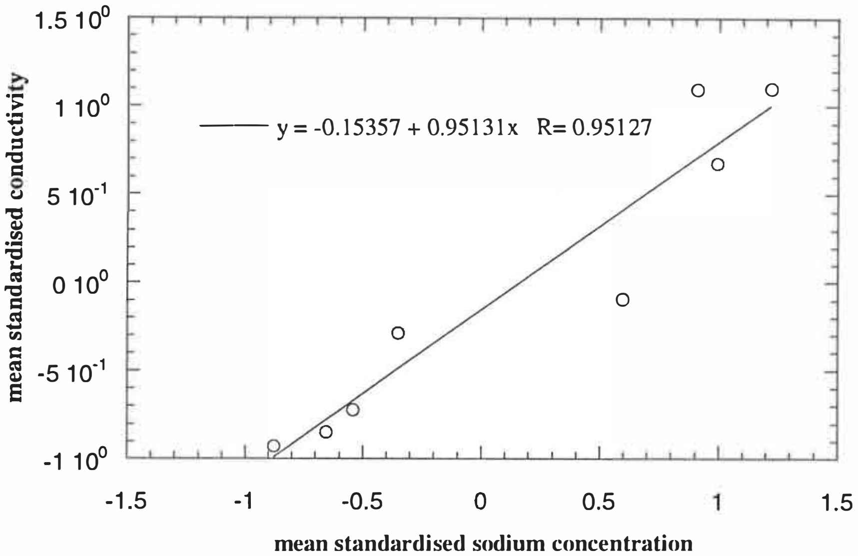


Figure 4.10b Conductivity versus sodium in shallow groundwater on Bardowie farm (1991-1998).

The increase in sodium levels in shallow groundwater may be largely attributed to decreased adsorption of this monovalent cation in the solum layer, as determined by the New Zealand Dairy Research Institute (Barnett *et al* 1996). It is important to note that volumes of wastewater and sodium concentrations applied to Bardowie farm have remained relatively constant over this time (Anchor Products 1996).

With regards to nitrate levels in shallow groundwater on Bardowie farm, levels of this constituent have remained low to 1998 because of on-going initiatives taken by the factory to reduce nitrogen loadings to the irrigation farm (Figure 4.11). Chloride levels have tended to fluctuate over the analysis period (Figure 4.12). Considering chloride is a conservative tracer, these fluctuations can only be attributed to variations in concentrations of this constituent in the wastewater resulting from a change in factory processes.

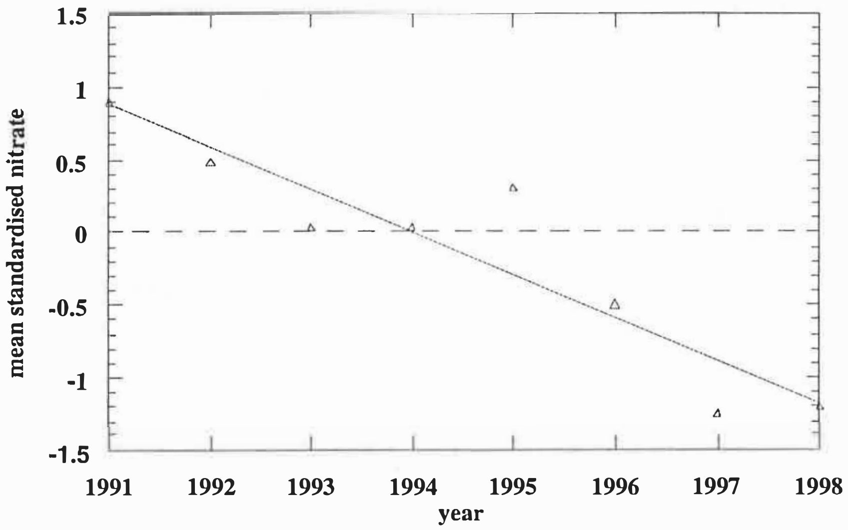


Figure 4.11 Mean standardised nitrate in shallow groundwater on Bardowie farm.

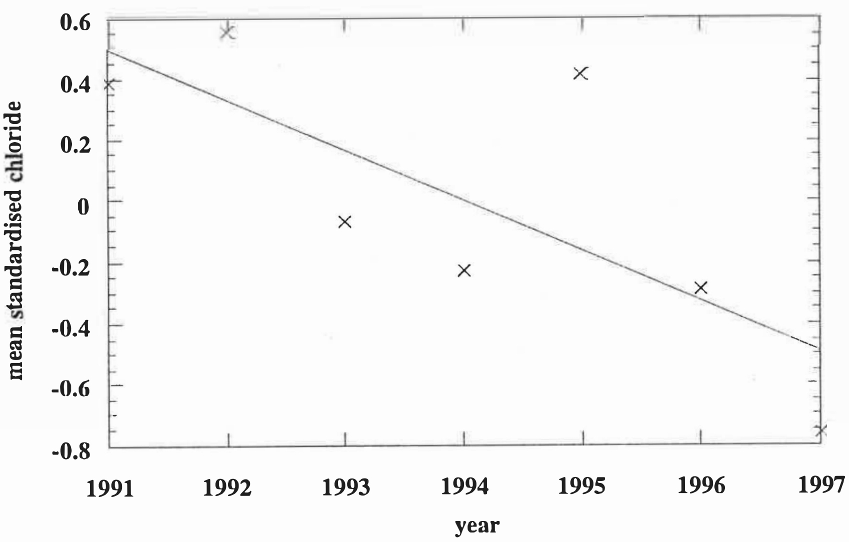


Figure 4.12 Mean standardised chloride in shallow groundwater on Bardowie farm.

4.4.2 Spatial Variation

Following on from the work carried out by Ward (1995), maps illustrating the spatial variation of the chemical parameters predominating in the wastewater were generated in the present study to further define the movement of modified groundwater. Figures 4.13, 4.14 and 4.16 are the contour maps of levels of electrical conductivity, sodium and nitrate respectively, in shallow groundwater. These contour maps were created using bore averages based on 1998 data and with data interpolation via the kriging technique within 'SURFER'.

The spatial patterns exhibited by sodium and conductivity (Figure 4.13 and 4.14) are similar to those observed in Ward (1995). The recent data indicates a modified groundwater plume centred on the irrigation farm with movement suggested mainly to the Northwest, but also to the South and Westward. It is acknowledged that the spatial distribution of these chemical parameters is a reflection of the distribution of the data points. In some areas, for example immediately off-site to the South and West, there is an apparent lack of data control for movement in these directions, particularly in the western direction. A degree of uncertainty therefore surrounds the suggestion of modified groundwater movement southwards and westwards.

As expected, current levels of sodium and conductivity in shallow groundwater on-site are overall higher compared to in 1994. A comparison between the 1994 contour maps generated by Ward and the 1998 maps generated for this study indicates the areal extent of modified groundwater movement has not increased in any of the main movement directions (Northwest, South and West) during this time.

Long-term conductivity data from bores forming a transect extending north south through the farm, was analysed to investigate more closely modified groundwater movement southwards (Figure 4.15). The absence of long term off-site data to the West and Northwest prevented investigations of movement in these directions.

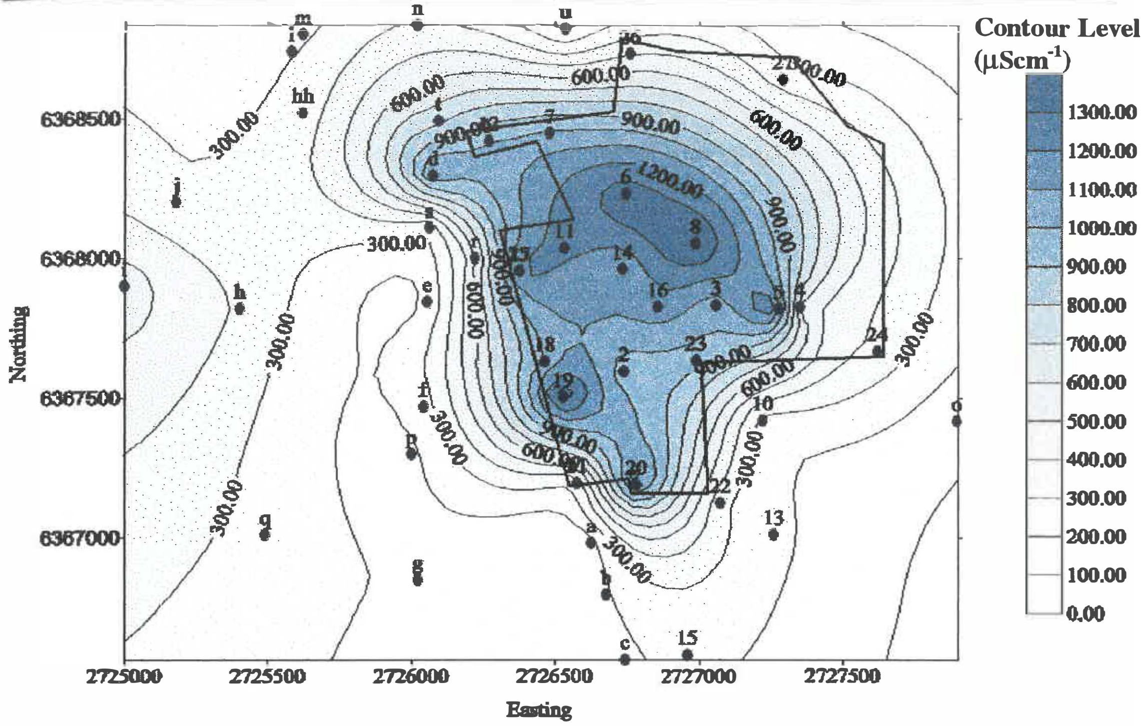


Figure 4.13 Iso-conductivity contour map (μScm^{-1}) for Bardowie farm and surrounding areas based on 1998 data. * Note conductivity levels in Bore I, off-site to the east, creating inaccurate movement patterns in this direction.

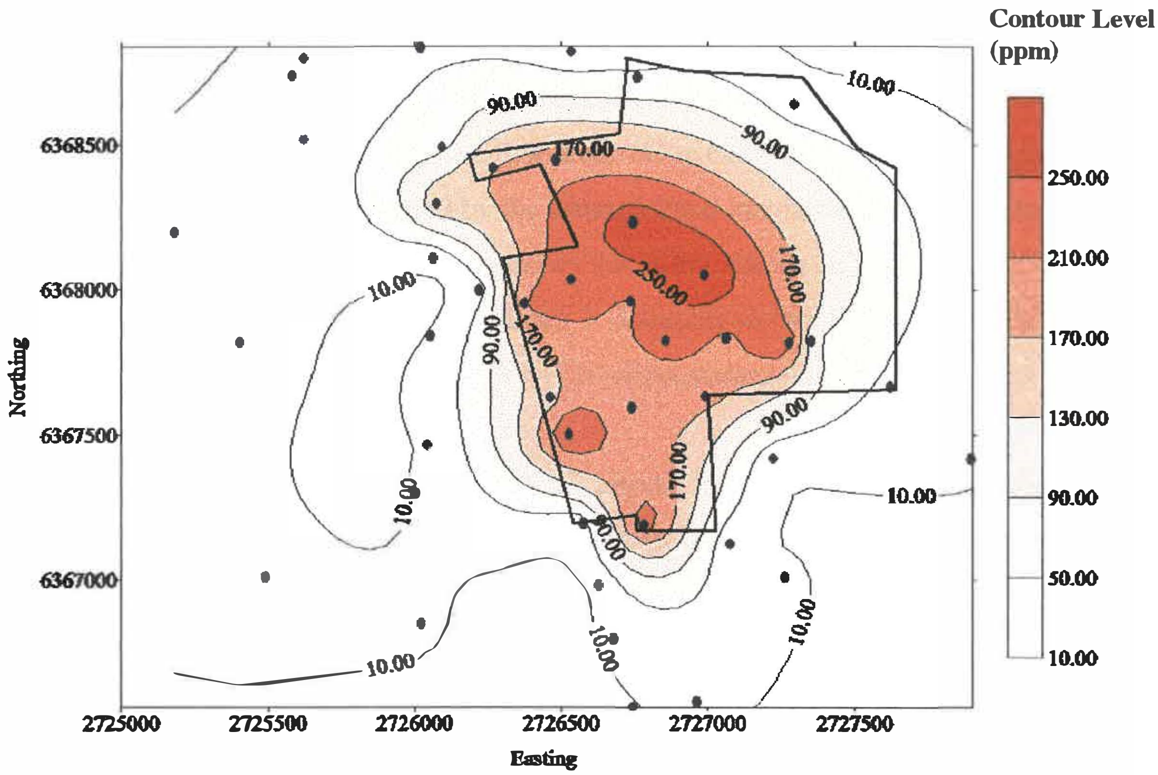
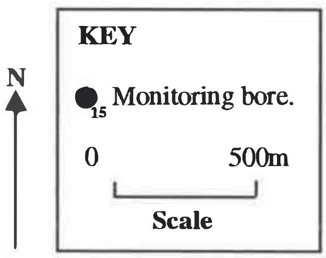


Figure 4.14 Sodium iso-concentration map (ppm) for Bardowie farm and surrounding areas based on 1998 data.



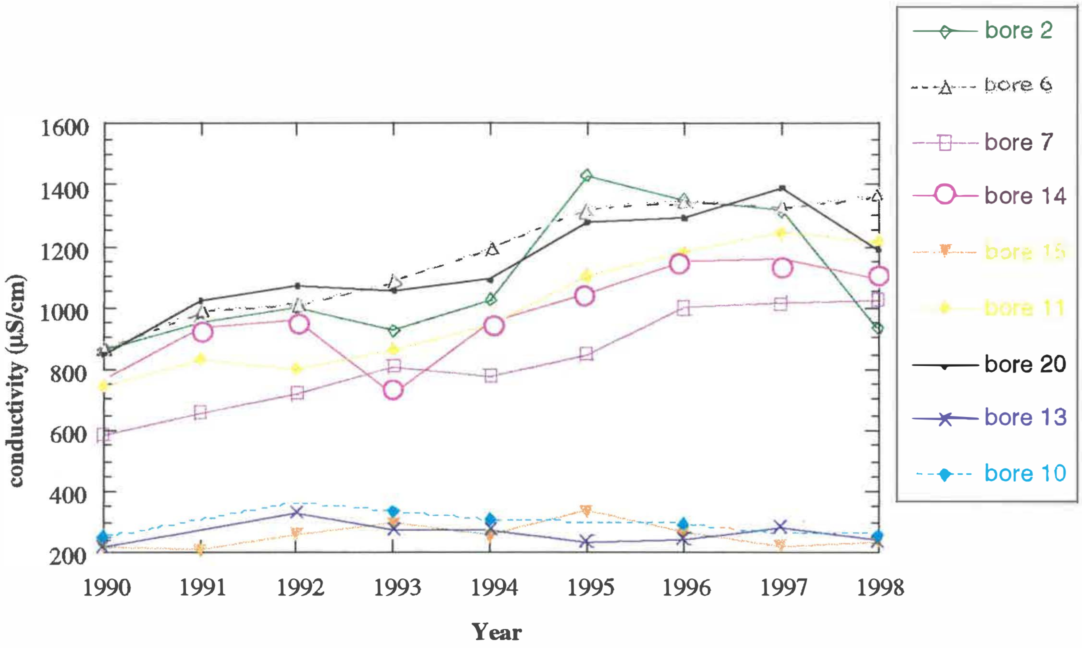


Figure 4.15 Conductivity in bores forming a north south transect through Bardowie farm.

The results (Figure 4.15) show conductivity of shallow groundwater in off-site bores located to the south (bores 10, 13, 15) has remained constant compared to on-site. This suggests significant movement has not occurred up to 500m south of the irrigation farm (location of bore 15 furthermost point) and that an equilibrium dilution situation may be present in this area. However, slow southern migration may still be occurring, as indicated by the upward trend in conductivity levels in bore 20, located at the farm's southern boundary (location shown in Figure 4.13). It is also noted that conductivity levels in Bore 7 (location shown in Figure 4.13) have also increased, confirming the possibility of Northwestward movement.

With regards to the 1998 contour map of nitrate levels in shallow groundwater (Figure 4.16), the pattern is notably dissimilar to the spatial pattern exhibited by conductivity and sodium with a less pronounced modified groundwater plume apparent. In contrast to the sodium and conductivity contour maps, strong northwestward movement is indicated by the spatial distribution of nitrate. However, this movement pattern appears be an artefact of the elevated nitrate readings at the western boundary, and off-site to the northwest. The spatial pattern of nitrate also appears to be related to a consequence of the lower levels of this constituent in shallow groundwater compared to sodium. Another possibility is that spatial variability in denitrification at the site is governing the distribution of this constituent. The disparity between the spatial patterns exhibited by nitrate with those shown by sodium and conductivity, indicates that the latter chemical parameters should be used to define modified groundwater movement. Of these, conductivity will provide the best ground truth for the resistivity technique.

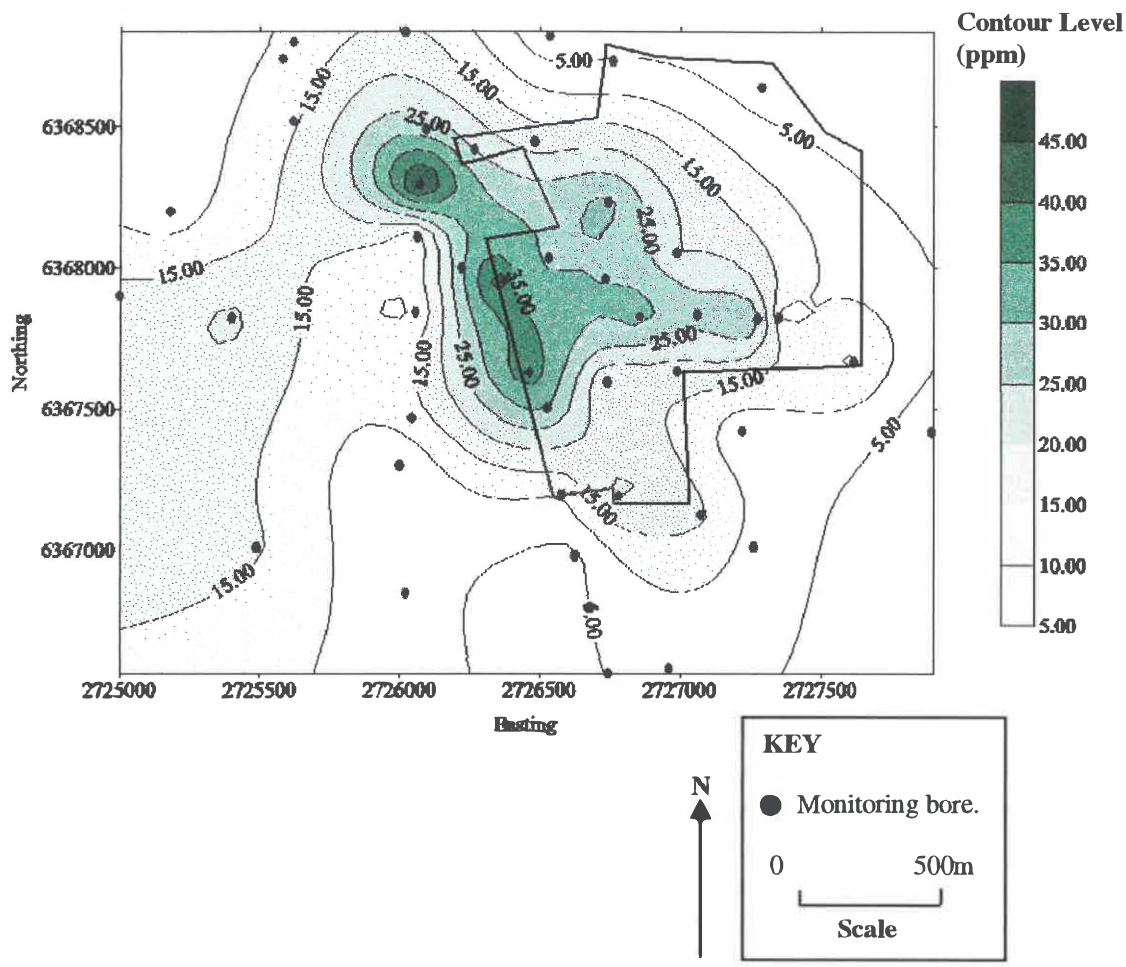


Figure 4.16 Nitrate iso-concentration map (ppm) for Bardowie farm and surrounding areas based on 1998 data. * Note nitrate levels in bores off-site to the east creating inaccurate movement patterns in this direction.

Modified groundwater index

On the basis that sodium and conductivity appear to be the best indicators of modified groundwater movement a chemical index was derived for these parameters in order to examine more closely, the size and geometry of the modified groundwater plume. Because monitoring programs differ between on-site and off-site bores and between chemical parameters, both 1997 and 1998 data was used to increase the accuracy of the chemical index. The chemical index was derived by:

(i) Generating monthly spatial standardised values for each chemical parameter based on their respective maximum levels (1510 μScm^{-1} for conductivity and 296 ppm for sodium);

(ii) Averaging the monthly standardised values, to obtain spatial annual averages (1997/1998) for each chemical parameter; and

(iii) Calculating a single spatial annual average thereby combining levels of sodium and conductivity and generating the modified groundwater index.

A contour map was then generated based on the single annual average with data interpolation achieved through the kriging function in 'SURFER'.

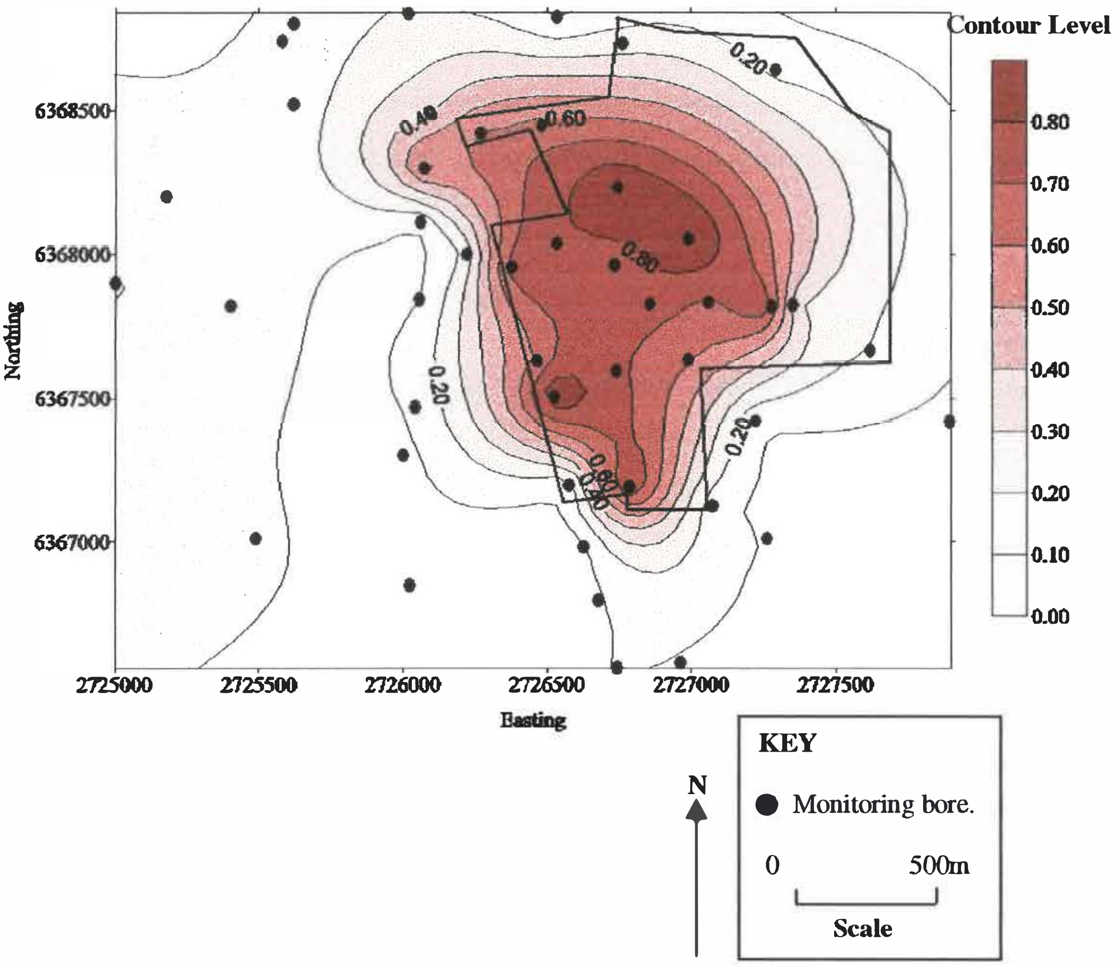


Figure 4.17 Contour map based on the modified groundwater index (1997-1998).

The map created based on the spatial modified groundwater index values (Figure 4.17) shows similar spatial patterns to those observed in the original sodium and conductivity contour maps (Figures 4.13 and 4.14). The modified groundwater index contour map indicates a plume centred on the farm with some lateral movement indicated to the West and South, but mainly to the Northwest. Highest levels of sodium and conductivity are found near the centre of the farm, with levels in this area being in the order of 0.8 of the maximum values for sodium and conductivity.

The high levels in this area of the farm are attributed to:

(i) The close proximity of the water table with respect to the ground surface, indicating minimal chemical attenuation in the solum; and

(ii) The long history of disposal and the higher irrigation rates associated with this area.

Levels of sodium and conductivity decrease radially outwards from the centre of the farm. In a large area (>70%) on the wastewater irrigation site, sodium and electrical conductivity levels exceed 0.5 of the maximum values. Immediately off-site, levels are typically between 0.1–0.4 of the maximum levels recorded on the wastewater irrigation farm.

4.4.3 Vertical variation

As mentioned, prior to the present study, the vertical extent of modified groundwater movement had not been established. Consequently, an investigation of the vertical spatial variation in levels of sodium, nitrate and electrical conductivity was undertaken in the present study, to elucidate this unknown parameter. The investigation was also motivated by:

(i) The inference that lateral movement is notably slow, therefore minimal at the site; and

(ii) Identification of vertical head gradients (Chapter Three), therefore the potential for vertical movement downwards.

Assessment of the vertical extent of modified groundwater was achieved using geochemical data from two existing bores (16m and ~65m deep) and from nested piezometers installed specifically for the present study. The locations of these bores are shown in Figure 4.18.

The installation of the nested piezometers, in particular, was considered fundamental to this investigation. This part of the investigation permitted groundwater samples to obtained from 13.6, 20.4 and 36.5 metres beneath the site in an area characterised by (i) a long history of disposal; (ii) medium to high irrigation rates; and (iii) elevated levels of the sodium and electrical conductivity in the shallow groundwater. Therefore, it was anticipated that the results would prove useful for determining the vertical extent of modified groundwater movement at the site. Table 4.2 lists the levels of the various chemical parameters found at various depths beneath the site, including geochemical data from existing bores and the nested piezometers.

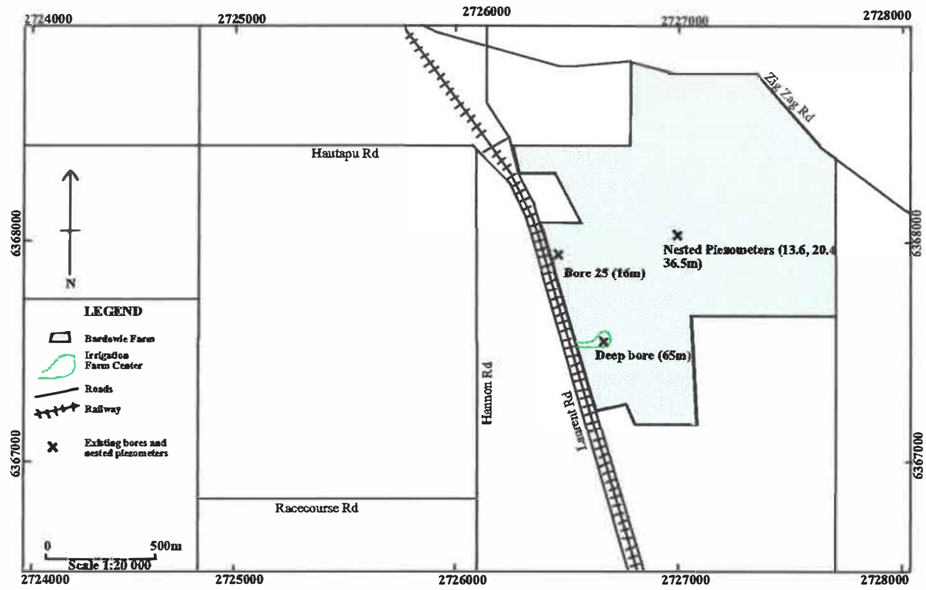


Figure 4.18 Locations of existing bores and nested piezometers on Bardowie farm used to determine the vertical extent of modified groundwater movement.

Bore		*Sodium (ppm)*	*Nitrate (ppm)*	*Conductivity (μScm^{-1})*
Deep Bore (65m)		19.8	0.02	130.0
Bore 25 (16m)		206.0	44.4	1245.0
Nested piezometers -	**13.6m**	29.6	0.141	463.0
	20.4m	57.5	<0.02	302.0
	36.5m	23.0	0.103	187.0

Table 4.2 Concentrations of chemical parameters in existing bores and in nested piezometers on Bardowie farm.

Given the background levels for sodium, nitrate and electrical conductivity (20 ppm, 8.9 ppm, and 289 μScm^{-1} respectively) the results from the existing deep bore (65m) suggest extensive downward movement of modified groundwater has not occurred on the western side of the farm. However, geochemical data from Bore 25 indicates vertical movement to at least 16m in this area, based on an assumed screen length of 1-2m (average screen length for all other on-site shallow bores). Here, levels of the chemical parameters are close to or near the current (1998) maximum levels recorded on site (285 ppm for sodium, 44 ppm for nitrate, and 1426 μScm^{-1} for electrical conductivity). Levels of the chemical parameters in the nested piezometers suggest near the centre of the farm, vertical movement is highly restricted by the thick sequences of low-permeability silt based materials beginning at ~8m (Figure 4.19). Nitrate, sodium and conductivity levels in the nested piezometers are either at or slightly above background levels of these constituents.

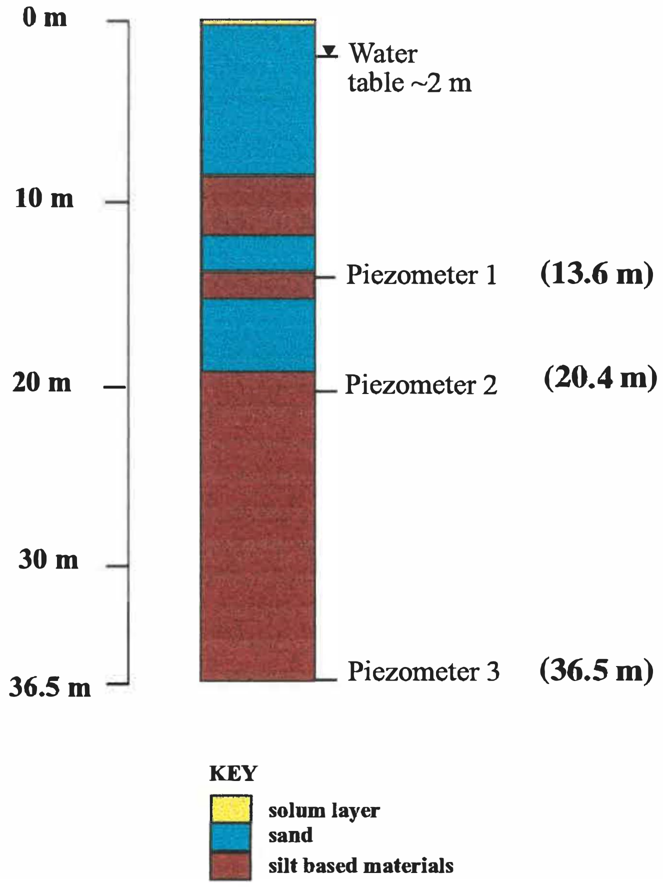


Figure 4.19 Lithological log recorded by Terra Aqua Consultants, during the installation of the nested piezometers on Bardowie farm, February 1999.

The increase in water conductivity upward from depth in this area of the farm, may reflect slight vertical movement downwards. It is more likely that this upward trend represents some natural variation in the aquitard groundwater.

4.5 Significance of sodium levels

Analysis of sodium levels in shallow groundwater beneath the wastewater irrigation site has indicated that elevated levels of sodium occur on site (130–250 ppm) and are steadily increasing. Due to non-utilisation of shallow groundwater on-site the only expected impact associated with these elevated sodium levels is with respect to soil structure degradation (Chapman, 1996, Dojlido and Best 1993). Immediately off-site levels are between 50–130 ppm in some areas, and it is anticipated that levels in such areas are also increasing, however long term data is required to confirm this.

4.6 Conclusions

In all three earlier studies (Barnett *et al* 1996, Terra Aqua 1993, Ward 1995) it was determined that the irrigation activity on Bardowie farm has led to elevated levels of chemical parameters in shallow groundwater and that, with the exception of nitrate, levels have steadily increased through time. With regards to the spatial variability of groundwater quality, a localised modified groundwater plume centred on the farm, and emanating in all directions was delineated in an earlier study (Ward 1995).

In the present study, variations in groundwater quality were considered through both time and space leading to the following main conclusions:

(i) Electrical conductivity levels in shallow groundwater on the wastewater irrigation site have increased steadily since 1990;

(ii) This increase is reflective of a steady rise in levels of sodium in shallow groundwater;

(iii) A less pronounced modified groundwater plume was revealed by nitrate, highly dissimilar in form to that indicated by sodium and conductivity. This is largely due to lower levels of nitrate relative to the other chemical parameters as a result of:

   (a) On going initiatives taken by the factory to reduce nitrate loadings to the farm; and

   (b) Nitrate removal through the denitrification process, likely to be favoured in areas where low permeability soils occur.

(iv) Sodium and electrical conductivity are thus considered the best groundwater quality indicators at the wastewater irrigation site, with electrical conductivity being the best ground truth for the electrical resistivity technique;

(v) Levels of sodium and electrical conductivity in groundwater indicate a localised modified groundwater plume centred on the irrigation site, with the possibility of slow lateral movement to the South, West and predominantly to the Northwest.

(vi) Geochemical data obtained for various depths below the site indicates vertical movement of modified groundwater is generally restricted to within the shallow unconfined aquifer, and in some places may extend to a depth of at least ~16m depending on the depth to the silt/clay layer. There is some possibility of minor deeper vertical movement.

CHAPTER FIVE

The application of electrical resistivity

5.1 Introduction

As indicated, a subsidiary aim of the present study was the assessment of the effectiveness of the electrical resistivity technique at the wastewater irrigation site. The main purpose of this work was to:

(i) Provide input about flow dynamics beneath the wastewater irrigation site through the delineation of the low resistivity modified groundwater plume; and

(ii) Evaluate the success of the technique in detecting spatial variations in groundwater quality.

Employment of the electrical resistivity technique to assess groundwater quality within the study area required preliminary investigations to test the feasibility of using the technique in this environment. Following on from preliminary work, a detailed assessment was carried out to determine the vertical and lateral extent of the modified groundwater plume previously delineated based on geochemical data. The Nilsson model 400 4-pin soil resistance meter was deployed for the resistivity survey.

5.2 Preliminary investigations

5.2.1 Laboratory experiment

Initial tests of the application of the electrical resistivity technique began with a laboratory experiment to estimate ambient (or background) and on-site groundwater resistivities. A 0.059m^3 volume plastic container was partially filled (0.036m^3) with sand/gravel material, similar in texture to the sediments of the Hinuera formation. The sand/gravel material was saturated with groundwater from monitoring Bore G, located approximately 0.5km South of Bardowie farm (refer to Figure 4.1), in an area considered to represent background groundwater conditions according to geochemical data (Chapter Four).

An electrode spacing of 0.15m was used to measure the resistivity of the ambient groundwater. The procedure was then repeated using pre-irrigation wastewater. The results of the laboratory experiment on these two samples are shown in table 5.1.

Media	***resistivity (Ωm)***
Wastewater / gravel	113.02
Ambient groundwater / gravel	207.24

Table 5.1 Inferred background and on-site resistivities of groundwater.

The resistivity of the wastewater/gravel combination was almost 50% lower than that of the ambient groundwater/gravel mix, indicating a good water quality contrast. These results suggested the possibility of mapping the high conductivity plume beneath the wastewater irrigation site.

5.2.2 Field investigations

Following on from the laboratory experiment, two initial field investigations were undertaken to test the applicability of the electrical resistivity technique in this environment.

High and Low Conductivity Areas

In the first of the field investigations, vertical sounding profiles at an on-site location (paddock 20) and an off-site location (Racecourse Rd) were conducted (Figure 5.1). According to geochemical data these areas are characterised by high and low conductivity waters respectively. The main objective of this work was to detect lateral variations in groundwater quality indicated by geochemical data. A second objective was to determine the most appropriate electrode spacing for horizontal profiling work so that the zone of interest, the shallow unconfined aquifer, would be adequately assessed.

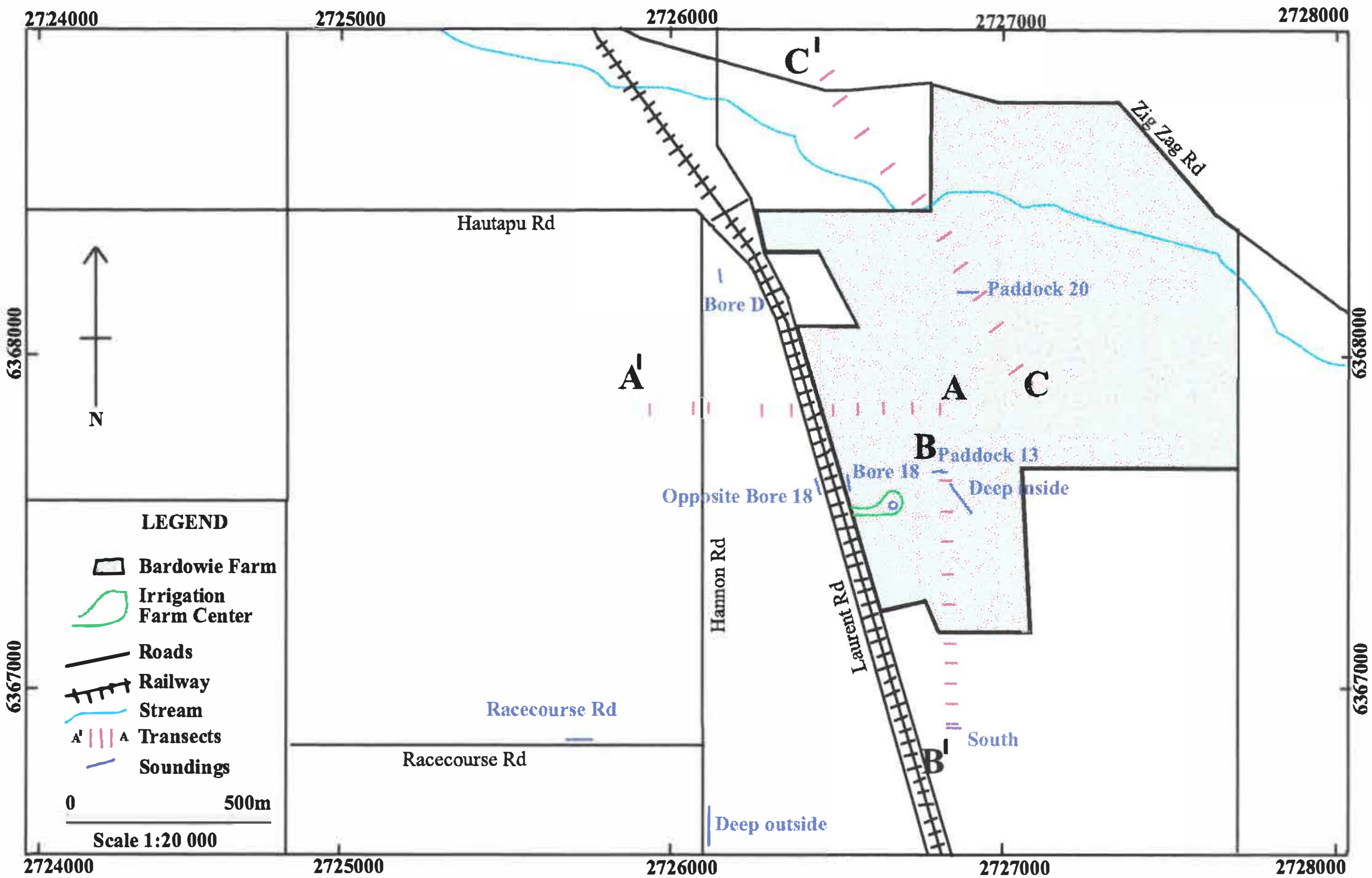


Figure 5.1 Locations of vertical soundings and transects on and near the irrigation farm.

To maximize the sensitivity to water quality changes, the vertical sounding profiles were aligned parallel with equipotential contours determined by the monitoring bore water levels (Chapter Three) and with water conductivity contours (Chapter Five). Alignment in this manner was observed, where possible, for all resistivity measurements carried out. This alignment approach has been used in other geophysical studies including those by Cartwright and McComas (1969), Seitz *et al* (1972), and Rogers and Kean (1980).

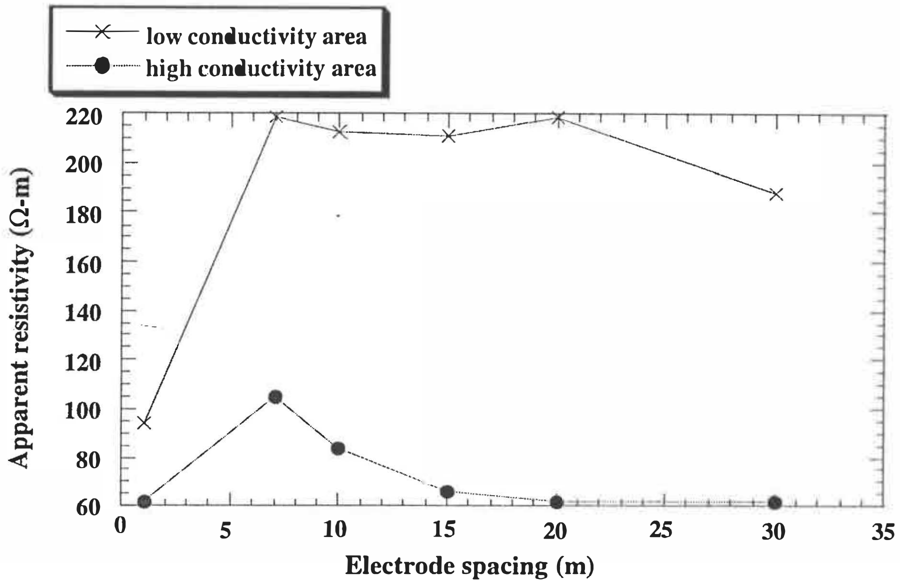


Figure 5.2 Resistivity sounding curves in areas characterised by high conductivity and low conductivity groundwater.

The horizontal spatial changes in resistivity are in agreement with the water quality pattern indicated by conductivity data, reaffirming the application of the electrical resistivity to detect changes in water conductivity (Figure 5.2). Low resistivity values are associated with the site characterised by high-conductivity groundwater (paddock 20) and high resistivity values are indicated for the low-conductivity area (Racecourse Rd). The low on-site resistivities are comparable to resistivities reported by Allen *et al* (1985) at a wastewater irrigation site in the United States. It is likely that the near surface (1-7m) vertical variations in resistivity are due to variations in water content and lithology, however this was not confirmed.

Transects

The next stage of the preliminary investigation involved taking horizontal profiling measurements along two transects aligned east west (A-A') and north south (B-B') respectively. The primary aim of this work was to track lateral changes in water quality indicated by geochemical data. Each transect extended ~1km outwards from the center of the farm to off-site areas and consisted of ten horizontal profile measurements located ~50-100m apart (Figure 5.1). An electrode spacing of 15m was adopted based on the results of the preliminary sounding work. Measurements along these transects were made in May, and again in July of 1998 to assess data repeatability. The resistivities obtained for each transect, including the two repetitions, are shown in Figures 5.3a and 5.3b.

Overall the patterns were similar between repetitions, however off-site the magnitude of the readings differed notably between repetitions. The July readings were on average 64% lower than the reading obtained in May. This variation in resistivity between repetitions could not be explained by changes in depth to water table, therefore any differences may be attributed to greater soil moisture in July compared with May.

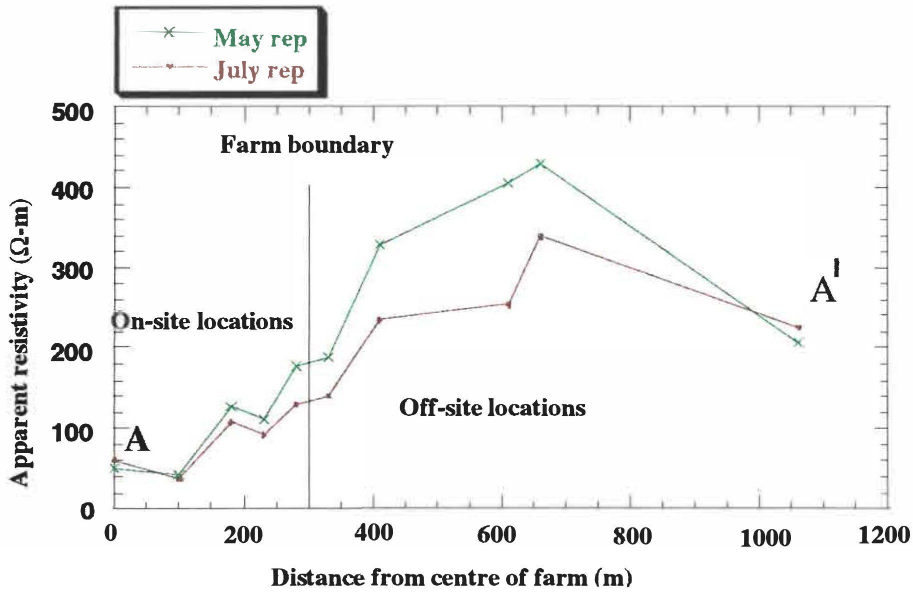


Figure 5.3a Resistivities along the Western transect (A-A').

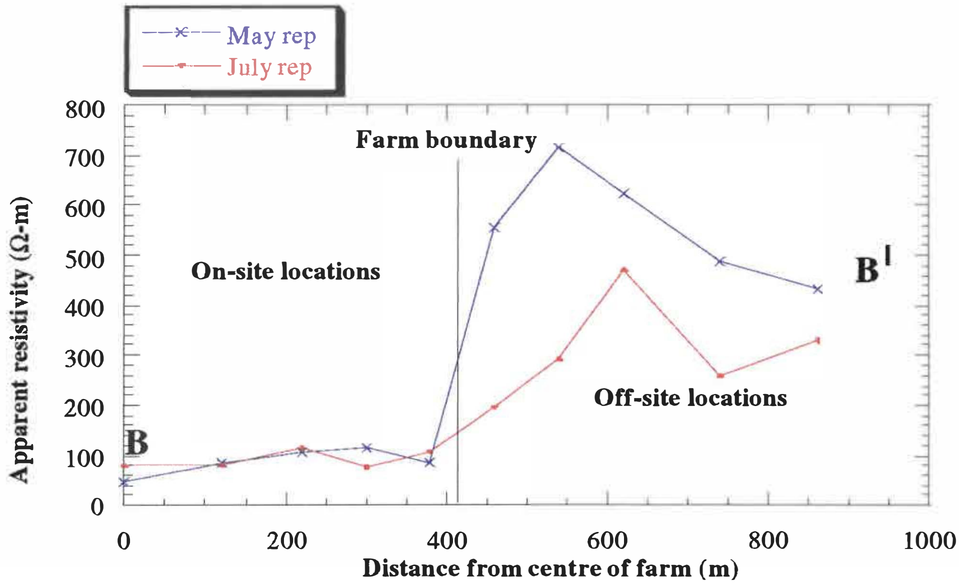


Figure 5.3b Resistivities along the southern transect (B-B').

The results of the southern and western transects are consistent and reflect the patterns indicated by geochemical data. Low on-site resistivities characterise both transects, with notably higher resistivities recorded off-site. Low resistivities on-site may be attributed to high conductivity modified groundwater derived from the irrigation activity, while dilution with ambient groundwater and rainwater as the modified groundwater moves off-site may explain the higher resistivities recorded in off-site areas. It is noted that the furthest off-site location along the western transect produced a notably lower reading, departing from the increasing trend that had developed. This low reading may be attributed to other point and non-point chemical sources, including fertiliser applications on adjacent land (Hannon pers. comm).

Although the basic form of both transects is similar, (low on-site resistivities compared with off-site) the rate of increase in resistivity and the magnitudes of the resistivity readings obtained off-site areas differs considerably between the two transects. A more gradual increase in resistivity occurs along the western transect compared to the southern transect, characterized by a large increase in resistivity immediately outside the farm boundary. It is important to acknowledge that some of the large increase in resistivity off-site to the south may be in fact attributed to variations in lithology, however most of the change is likely to be related to changes in groundwater conductivities. The off-site readings for the western transect, are at least 50% lower than the off-site readings south of the farm. Therefore, greater lateral movement of modified groundwater in the westward direction appears to have occurred compared to southwards. This is consistent with the patterns indicated by geochemical data (Chapter Five).

5.3 Detailed investigation

Having completed the preliminary investigation, which indicated the resistivity data was adequately detecting spatial changes in water quality, a detailed assessment was undertaken to delineate the modified groundwater plume. The detailed assessment comprised two parts, an investigation of the lateral spatial extent of modified groundwater movement and an investigation of the vertical extent of movement.

5.3.1 Vertical variation

For the examination of the vertical extent of modified groundwater movement (the vertical thickness of the modified groundwater plume) vertical soundings were conducted at a number of on-site and off-site locations (Figure 5.1). The results of the on-site and off-site soundings are presented in Figure 5.4.

Of particular importance are the results from a deep sounding (65m) carried out near the center of the farm. The results of the deep sounding appear consistent with the existence of a low-resistivity zone between 15-65m. Resistivity averaged 68.70 Ωm for the deep vertical sounding on-site compared with 237.70 Ωm for the deep sounding carried out approximately 1km southwest of the irrigation farm. One possibility was the vertical downward movement of some of the high conductivity modified groundwater. Another possibility was the presence of a highly conductive silt/clay zone, however the absence of such a layer off-site would be required to explain the increasing resistivities off-site with depth.

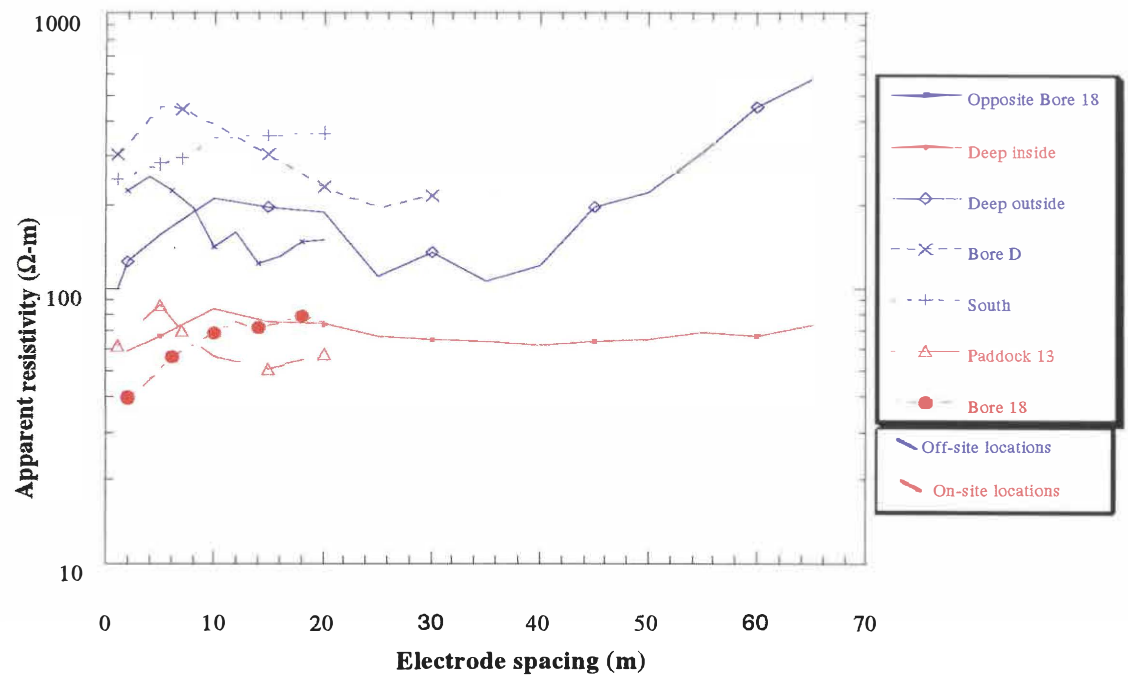


Figure 5.4 Resistivity soundings for on-site and off-site locations (refer to Figure5.1 for locations).

Geochemical data from nested piezometers near the location of the deep sounding (Chapter Four), showed low levels of the chemical parameters characterising the wastewater at 13.6, 20.4 and 36.5 m. The geochemical data obtained at various depths below the site, indicated vertical movement of modified groundwater movement is highly restricted by the presence of thick sequences of low permeability silt-based materials beginning at a depth of ~8 m (Chapter Four). Consequently, it appears that the low-resistivity readings obtained between 15–65 m reflect the presence of these highly conductive low permeability materials and do not signify the presence of high conductivity modified groundwater at depth.

With regards to the shallow soundings undertaken, which was influenced mainly by conductivities within the shallow aquifer, significant differences in the magnitude and form of the sounding curves are apparent. The low and consistent results of the vertical soundings conducted on-site suggest the presence of high conductivity modified groundwater within the shallow aquifer to an average depth of ~15m (Figure 5.4). This could only be confirmed at one location on the wastewater irrigation farm. Recent water quality data from bore 25, extending to 16m depth with an assumed screen length of ~1-2m, indicates elevated levels of the chemical parameters in shallow groundwater (Table 5.2) thereby suggesting the validity of the resistivity data.

Nitrate (ppm)	44.4
Sodium (ppm)	206.0
Conductivity (μScm^{-1})	1245

Table 5.2 Average levels of the chemical parameters in Bore 25 (based on 1998 data).

At off-site locations, resistivity is notably higher at depths within the shallow aquifer and fluctuates significantly presumably due to variations in lithology or ambient groundwater quality changes. Overall, the differences in resistivity within the saturated zone (≥15m), between on-site and off-site locations, are thought to be largely due to differences in water conductivity.

5.3.2 Lateral variation

The next stage of the detailed assessment involved an extensive horizontal profiling survey using a 15m-electrode spacing to delineate lateral variations in groundwater conductivity. Originally, measurements were made adjacent to the factory's on-site and off-site monitoring bores (Chapter Four) to obtain a relationship between resistivity and groundwater conductivity. However, assessment of the results as they were collected indicated uncertainty with regards to the boundaries of the conductivity plume, therefore the survey was expanded. A total of 56 horizontal profile measurements were made within the study area (Figure 5.5). Measurements at eight sites were repeated in November to determine whether the technique is affected by temporal changes in sub-surface conditions such as watertable elevations and soil moisture content.

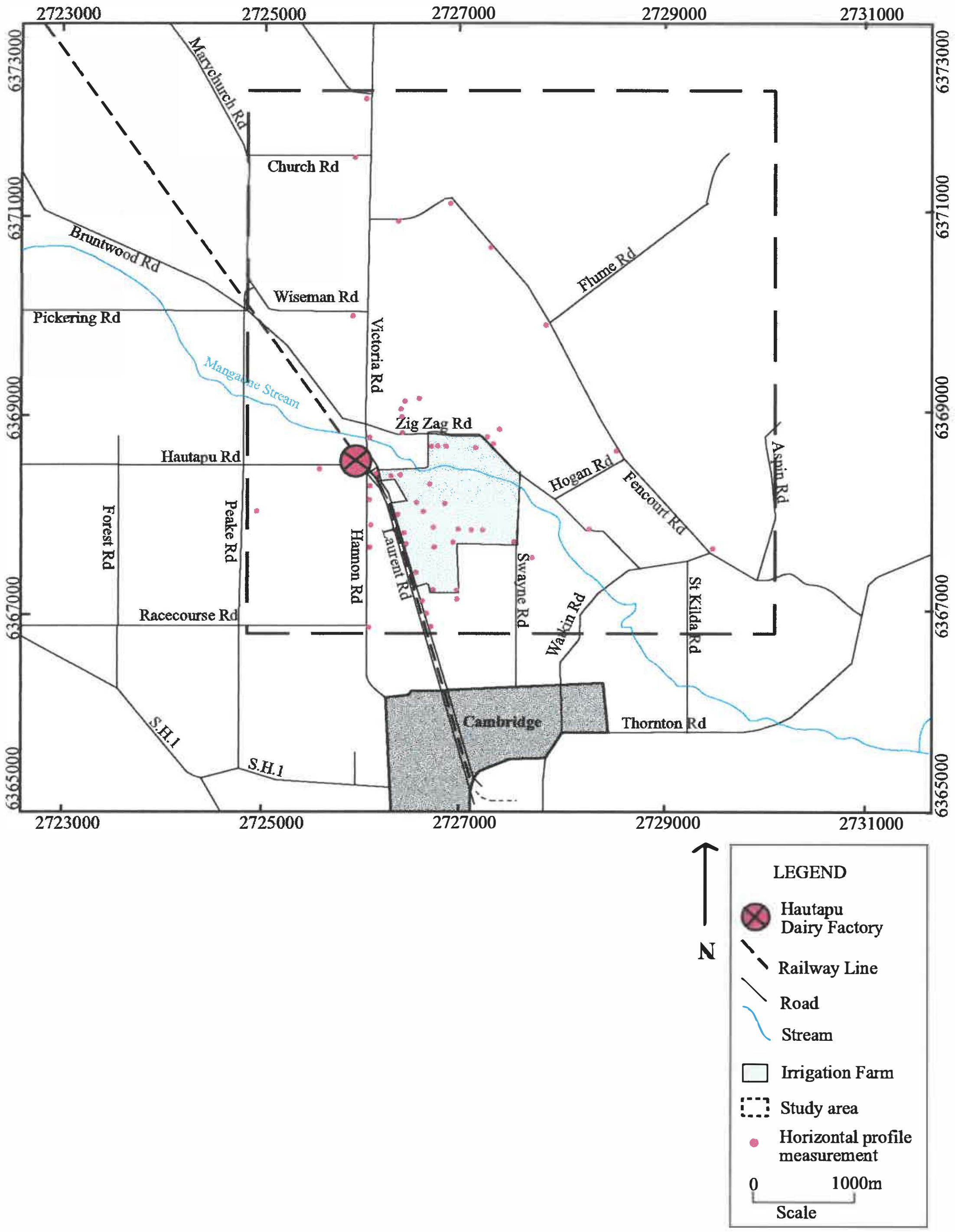


Figure 5.5 Horizontal profiling measurement sites (adapted from Ward 1995 after NZ Map Series 260).

Correlation between resistivity and geochemical data

As mentioned, the original strategy of the electrical resistivity survey was to obtain resistivity data at the sites of the monitoring bores, to derive an empirical relationship between the electrical resistivity data and water conductivity. It was anticipated that this would enable a quantitative evaluation of the success of the technique, as an index of water quality modification in this environment.

In a study by Ebraheem *et al* (1990), good correlation was observed between (i) earth resistivity and total dissolved salts and; (ii) total dissolved salts and water conductivity. Therefore in the present study, water conductivity values were converted to water resistivity using the following equation:

$$R = 1/C$$

Where R is water resistivity (Ωm), and C is water conductivity (Sm^{-1}). This enabled water resistivity to be plotted against earth resistivity. Although a reasonable correlation was observed between the two variables (Figure 5.6a), it was postulated that some of the scatter might be a result of the difference in depth considered by the two techniques. Therefore, measurements were repeated at several sites, with the spacing of the electrodes at each site determined by the maximum depth sampled by the corresponding bore. The relationship between earth resistivity and water resisitivity was thus further improved (Figure 5.6b).

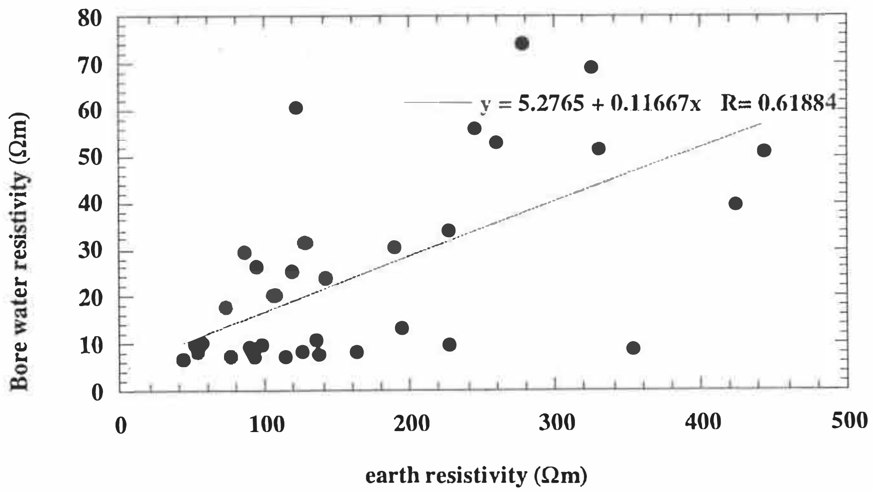


Figure 5.6a The relationship between bore water resistivity and earth resistivity at bore locations (15m electrode spacing).

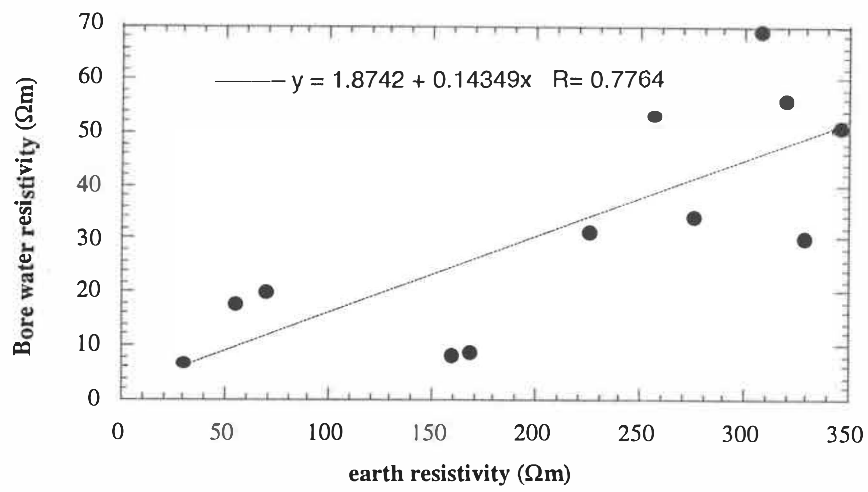


Figure 5.6b The relationship between bore water resistivity and earth resistivity at bore locations (electrode spacing defined by maximum sampling depth of each bore).

The relationship based on the second dataset indicates earth resistivity is largely affected by groundwater conductivity, however the remaining scatter (0.3) suggests the technique is subject to errors due to:

(i) Lithological variations;

(ii) Variations in the depth to the water table; and

(iii) The fact that the resistivity data is a vertical average whereas the water resistivity values are a point measure.

Resistivity mapping

The next stage of the investigation, considering the lateral spatial variation in resistivity involved mapping the horizontal profiling data. The computer program 'SURFER' was employed for this task. Data interpolation was achieved using the kriging function, a geostatistical procedure commonly used for interpolating spatial water quality data (Cooper *et al* 1988, Cooper *et al* 1988, Dou and Woldt 1994, Parks *et al* 1994, Zirschky *et al* 1986, and Zirschky *et al* 1986).

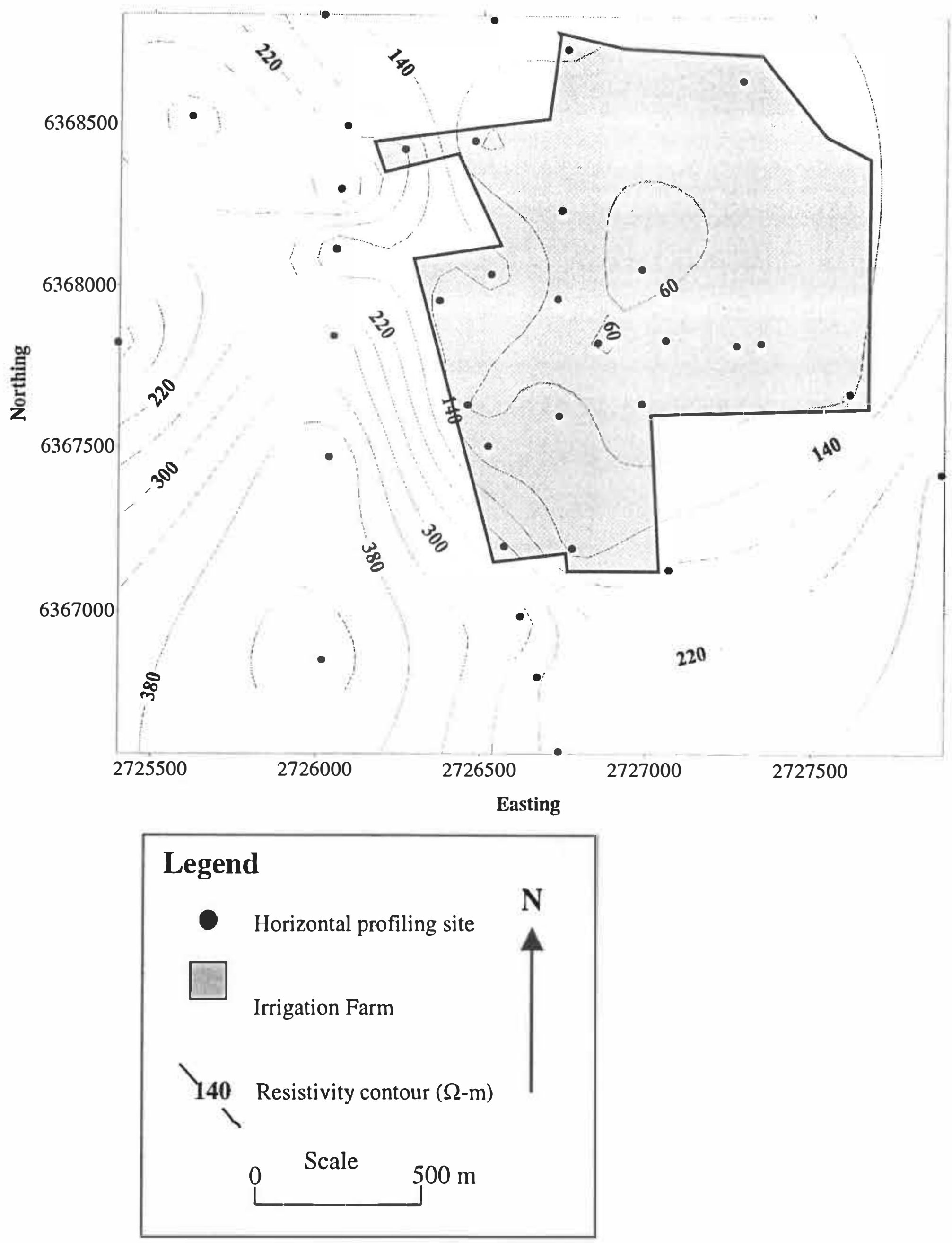


Figure 5.7 Iso-resistivity contour map of the area defined by the monitoring bore network.

The iso-resistivity contour map generated based on measurements at monitoring bore sites (Figure 5.7) indicates the presence of a localised low resistivity (60-100 Ω-m) plume centered on the wastewater irrigation farm. This is consistent with the irrigation and groundwater recharge of high-conductivity wastewater within this area. Resistivity increases outwards from the farm boundary as modified groundwater spreads, and is diluted by ambient groundwater and rainwater close to the boundary. In general, the size and geometry of the plume indicated by the resistivity data is consistent with that indicated by the water conductivity data (Chapter Four) with possible movement of low-resistivity modified groundwater northwestward, southward and westward indicated. However, disparity between the water conductivity data and the resistivity data is evident at some locations. For example at Bore D (refer to Figure 4.1), where high-conductivity water is found, a high resistivity reading was obtained when it was anticipated that a low resistivity reading would result. This inconsistency between the geochemical data and the resistivity data is attributed to variation between the depths considered by the bore (~10m) and the resistivity technique (~15 m). Another possibility is that the high conductivity water is highly localised and the resistivity measurements were not taken in the region where this water occurs.

Further inconsistency between the geochemical data and the resistivity data was evident where notable movement of modified groundwater indicated north-northeastwards and to the east by the low resistivity readings obtained off-site in these directions, relative to on-site. Accordingly, further research was undertaken in these areas, with particular emphasis on defining the northern edge of the plume based on the assumption that northwest flow probably predominates in the area.

To check the resistivity readings, ten horizontal profiling measurements, forming a third transect (C-C'), were undertaken. The horizontal profiles were aligned parallel to each other and positioned so the transect extended northwest from the center of the farm to off-site areas (Figure 5.1).

The results show that the pattern for the northwestern transect differs greatly from those that developed along the western and southern transects (Figures 5.8b and 5.8c). Along this third transect (Figure 5.8a), resistivity remains low off-site relative to on-site areas, suggesting the possibility of significant movement of modified groundwater northwestward.

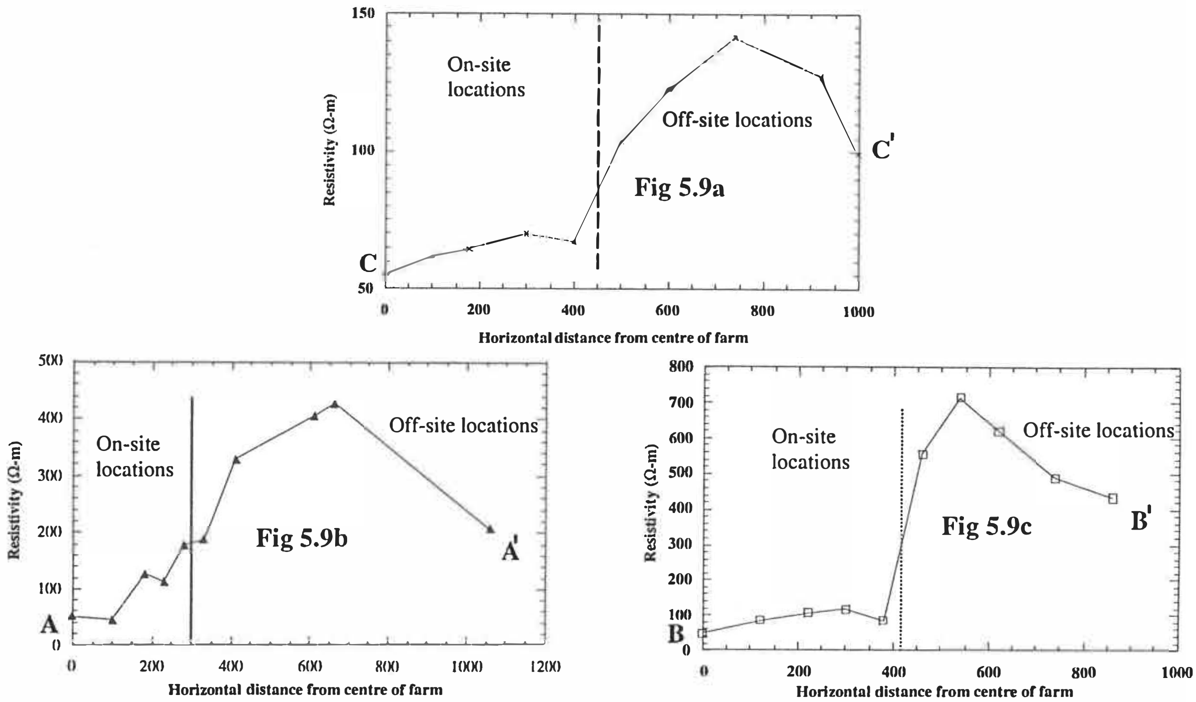


Figure 5.8a Resistivities along the Northwestern transect **Figure 5.8b** Resistivities along Western transect **Figure 5.8c** Resistivities along Southern transect. Dashed lines denote the farm boundary.

The results of the northwestern transect necessitated the expansion of the resistivity survey. Accordingly, additional horizontal profiling measurements were taken northwards to define the northern edge of the plume. Due to the apparent uncertainty existing east of the farm additional measurements were also taken in this area.

A revised iso-resistivity map (Figure 5.9) was generated incorporating the additional off-site measurements taken northwards and eastward, as well as the resistivities of the Northwestern, Southern and Western transects. The results show abrupt boundaries to the West and South suggesting limited lateral movement of modified groundwater. However to the north-northwest and eastwards, inferred lateral movement of modified groundwater is indicated given that resistivity remained low, relative to on-site resistivities, and over considerable distances in these directions (~3-4 km north-northwestward and 1-2 km eastward).

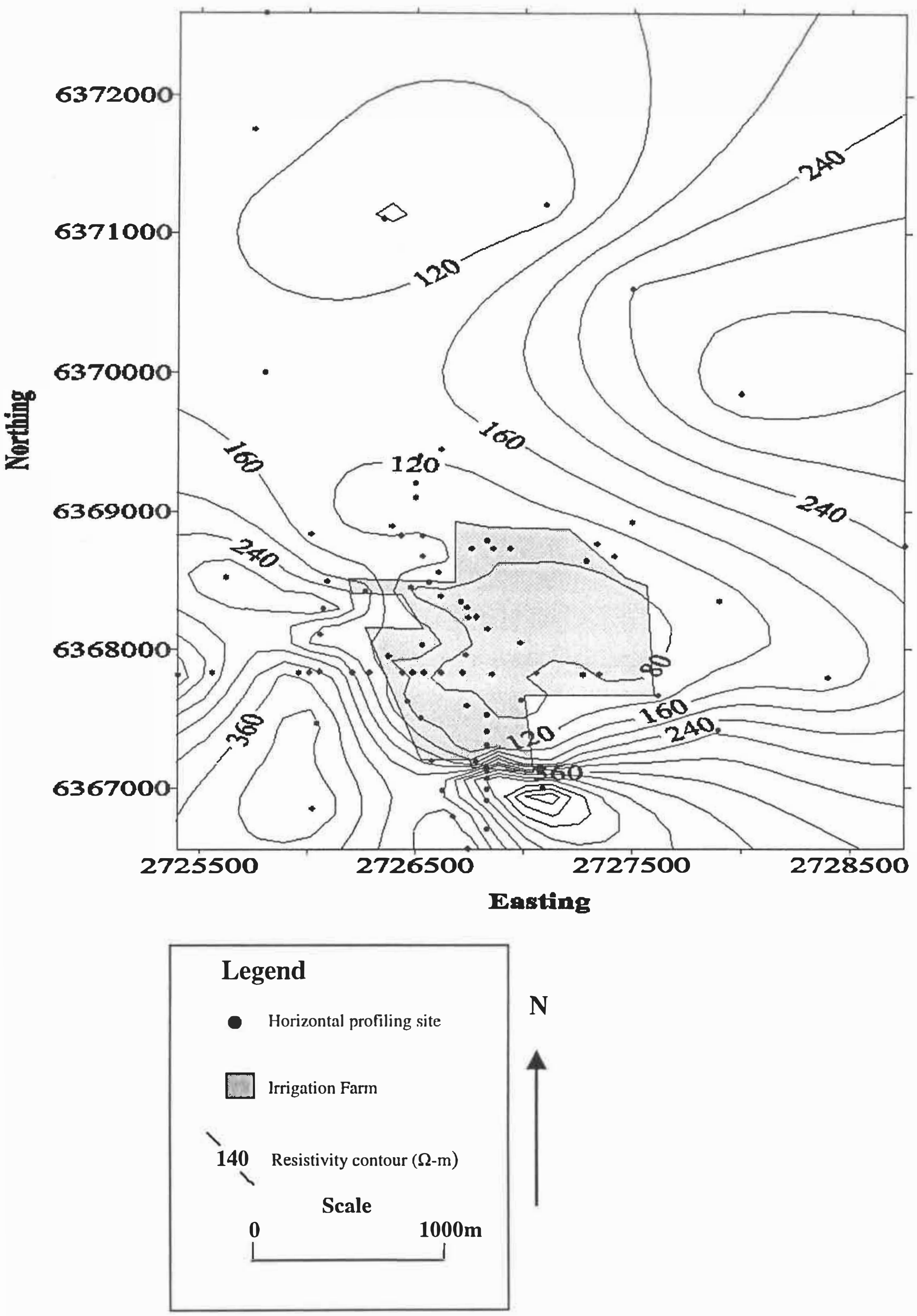


Figure 5.9 Revised iso-resistivity contour map of the study area.

The northwestern movement pattern seemed plausible based on the regional groundwater flow pattern characterising the study area (Chapter Three). However, contrary to this theory, Bore U located north of the farm (refer to Figure 4.1) indicates low levels of conductivity, sodium and nitrate (Table 5.3). Consequently, further geochemical data was required to confirm the possibility of northwestward movement. Geochemical data was also required to validate the pattern indicated east of the farm.

Sodium (ppm)	17.5
Nitrate (ppm)	0.8
Electrical conductivity (μScm^{-1})	165

Table 5.3 Average levels of chemical parameters in Bore U, located off-site to the northwest (based on 1998 data).

Groundwater samples were obtained from private bores in the northern and eastern areas (Figure 5.10) and analysed for sodium, conductivity, chloride and nitrate, by the New Zealand Dairy Research Industry. Ward (1995) had also obtained groundwater samples from these bores and at that time background levels of these chemical parameters were found. Past and present levels of the chemical parameters in the northern and eastern private bores are presented in Tables 5.4 and 5.5.

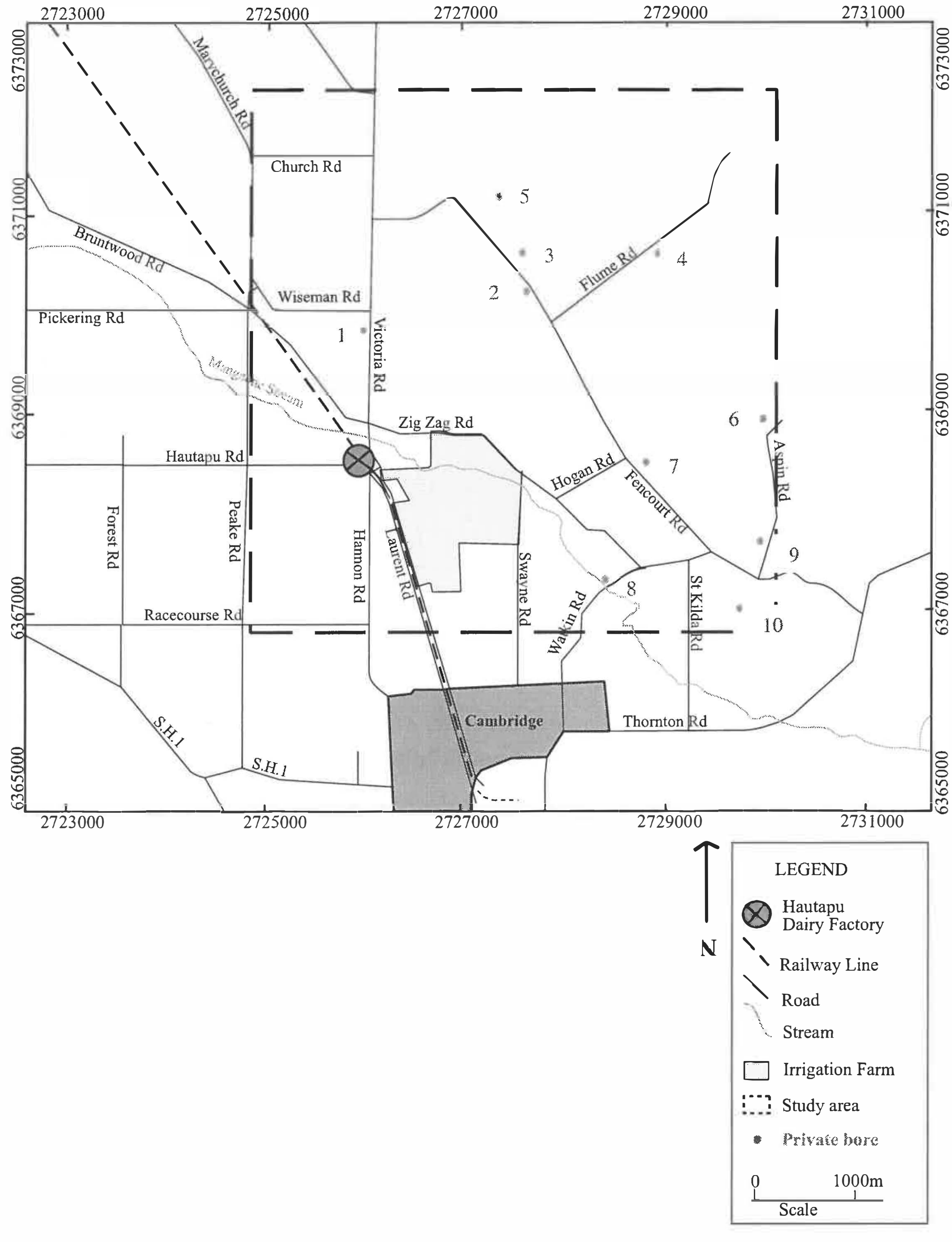


Figure 5.10 Location of private bores in the northern and eastern areas.

Bore	NO_3^{2-} (ppm)	Ward (1995)	Cl (ppm)	Ward (1995)	Conductivity (μScm^{-1})	Ward (1995)	Na (ppm)	Ward (1995)
1	14.5	15.2	18.3	*	267.0	314.0	15.2	17.8
2	3.4	3.5	18.8	*	175.0	232.0	17.2	18.4
3	9.1	14.1	18.3	*	253.0	337.0	18.4	23.4
4	8.1	4.2	57.7	*	425.0	530.0	26.2	34.3
5	0.1	10.5	24.5	*	284.0	341.0	18.9	27.0
* No data								

Table 5.4 Results of chemical analysis of groundwater samples taken from private bores north of the irrigation farm.

Bore	NO_3^{2-} (ppm)	Ward (1995)	Cl (ppm)	Ward (1995)	Conductivity (μScm^{-1})	Ward (1995)	Na (ppm)	Ward (1995)
6	6.1	11.3	23.6	*	249.0	347.0	12.8	14.7
7	8.7	13.3	14.7	*	213.0	212.0	12.6	13.6
8	2.7	*	4.4	*	149.0	*	9.0	*
9	0.4	10.2	1.2	*	255.0	223.0	12.4	*
10	3.1	12.1	7.9	*	186.0	375.0	13.1	28.4
* No data								

Table 5.5 Results of chemical analysis of groundwater samples taken from private bores east of the irrigation farm.

The results of the chemical analysis of groundwater samples from private bores north and east of the irrigation farm indicate background levels of the chemical parameters, similar to those reported by Ward (1995). The levels in the eastern private bores are expected considering the position of this area with respect to regional groundwater flow (Chapter Three) and the irrigation farm. Therefore, it is likely that subsurface resistivities in this area are determined by some other factor other than water conductivity, for example lithology.

The geochemical results from the northern private bores seem to suggest extensive modified groundwater movement has not occurred in this direction. A possible explanation for this apparent lack of movement to the north-northwest is that the Mangaone Stream partially intercepts shallow modified groundwater as it migrates in this direction. This theory is supported by the fact that flow rates increase as the stream moves through the farm (Karl Mischewski pers. comm.).

In terms of the low resistivity readings obtained in this area, there are a number of possibilities including:

(i) The presence of natural stagnant high conductivity groundwater in this area;

(ii) The existence of continuous thick sequences of low resistivity materials such as silt or clay, a definite possibility given the results of the investigation considering vertical variations in resistivity; and

(iii) Movement of modified groundwater northwestward at depth, beyond the Mangaone stream and the bores in this area

Temporal variation

Based on the variation between the June and November resistivities, it appears that the technique is affected to a minor extent, by temporal variations in subsurface conditions (Figure 5.11). For six of the eight sites a lower resistivity reading was recorded in November compared to the June reading. This decrease in resistivity to November may be attributed to increased moisture in the solum layer and a decrease in the depth to the water table, noted to have occurred at the sites (Table 5.6). Experimental error, specifically inconsistent alignment of the electrode array, may account for the variation at the remaining two sites where a higher resistivity reading was recorded despite a rise in the water table. Overall, temporal variations in resistivity are considered minor on the basis that the same spatial pattern is exhibited between the bores in June as in November.

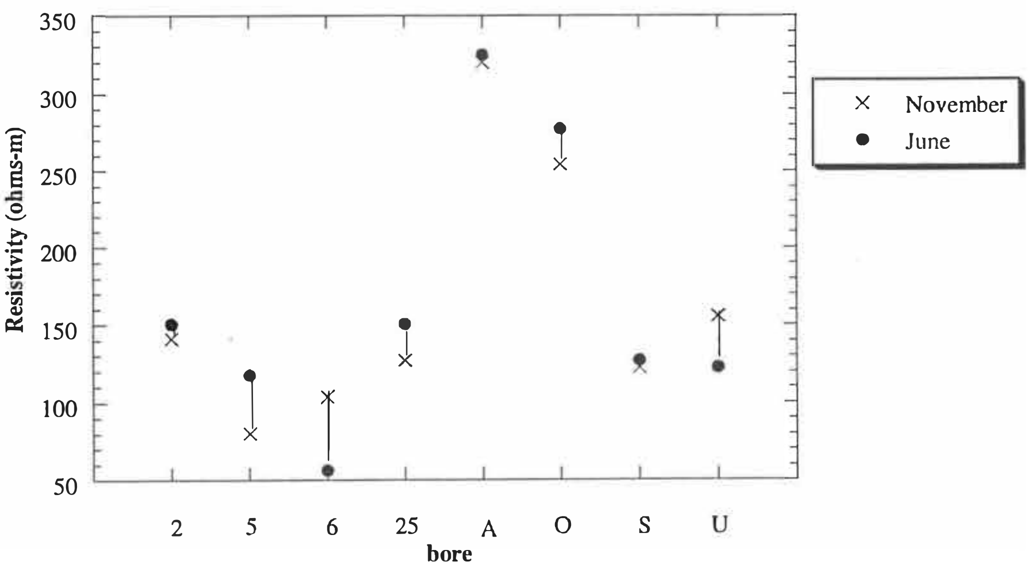


Figure 5.11 Resistivity at on-site and off-site bores in June and November 1998. Black line indicates difference between the readings.

Bore	***Depth to water table (m)***	
	June-98	November-98
2	6.56	5.83
5	3.85	3.11
6	4.26	3.72
25	7.31	6.65
A	8.48	7.93
O	3.95	2.93
S	5.13	4.60
U	3.41	2.72

Table 5.6 Water levels in selected bores for June and November 1998.

5.4 Conclusions

In general, the application of the electrical resistivity technique at the wastewater irrigation site has proven to be relatively successful. The geometry and size of the modified groundwater plume delineated using the electrical resistivity technique is highly similar to the conductivity plume described in Chapter Four.

The following main conclusions with regards to the inferred size and geometry of the plume are drawn from the results of the resistivity survey:

(i) The modified groundwater plume is localised and centered on the farm with definite boundaries occurring to the South, West and East;

(ii) While uncertainty remains in the northwestern direction, the direction of regional groundwater flow, it is likely that the northern edge of the plume occurs at the Mangaone Stream based on geochemical data beyond the stream and to the north-northwest; and

(iii) The modified groundwater plume occurs within the shallow unconfined aquifer, the thickness of which varies, however appears to extend to an average depth of ~15m.

The overall consistency between geochemical data and the resistivity data suggests the geophysical technique is effective for determining variations in shallow groundwater quality beneath the wastewater irrigation site. However, the disparity between the two datasets observed off site and at depth on-site, indicates the resistivity technique is limited to areas where high conductivity modified groundwater dominates the resistivity signal. Beyond these areas, lithology may become the determining factor for resistivity. Consequently misinterpretations may be made with regards to water quality, and ultimately modified groundwater movement, unless monitoring bore information (geochemical data and lithological log) is available as a ground truth for the resistivity technique. Therefore, it is concluded that the electrical resistivity technique should not be solely relied upon for assessing groundwater quality near the site, but should be used in conjunction with drilling for lithological and groundwater sampling purposes.

CHAPTER SIX

A simple modelling study

6.1 Introduction

A simple modelling study was undertaken to estimate groundwater flow patterns within part of the study area, including the area of the irrigation farm (Figure 6.1). The main aim of this study was to find the most probable pathways and indicative rates of groundwater movement in this area, including possible vertical downward movement into the deep aquifer.

The various stages of the modelling study are described in this chapter, from the development of the conceptual model through to calibration and generation of output information. The model was generated with the following structure:

(i) Steady state conditions (seasonal variations are therefore not accounted for in the model and the bores pump at a constant mean rate);

(ii) Three homogenous layers of constant thickness, including the silt/clay layer causing this layer to act as an aquitard;

(iii) The wells fully penetrate the aquifer.

The generalisations made were necessary to develop the numerical model however they are seldom met in reality. Therefore, it is anticipated that variation will occur between values predicted by the model and those occurring in reality at any given time. The development of a more detailed numerical model to represent transient groundwater effects and detailed lithology with a higher degree of accuracy was beyond the scope of this study, therefore the results of the study should be considered in view of the above conditions.

6.2 The conceptual model

The conceptual model, a schematic diagram of the physical framework of the groundwater system including the main components and the various interactive processes occurring within the system, is shown in Figure 6.1. A 10.5 km^2 region within the study area was included in the model (the 'zone of interest'), encompassing the irrigation farm, the abstraction well field and a buffer zone (0.5-1 km) surrounding these entities.

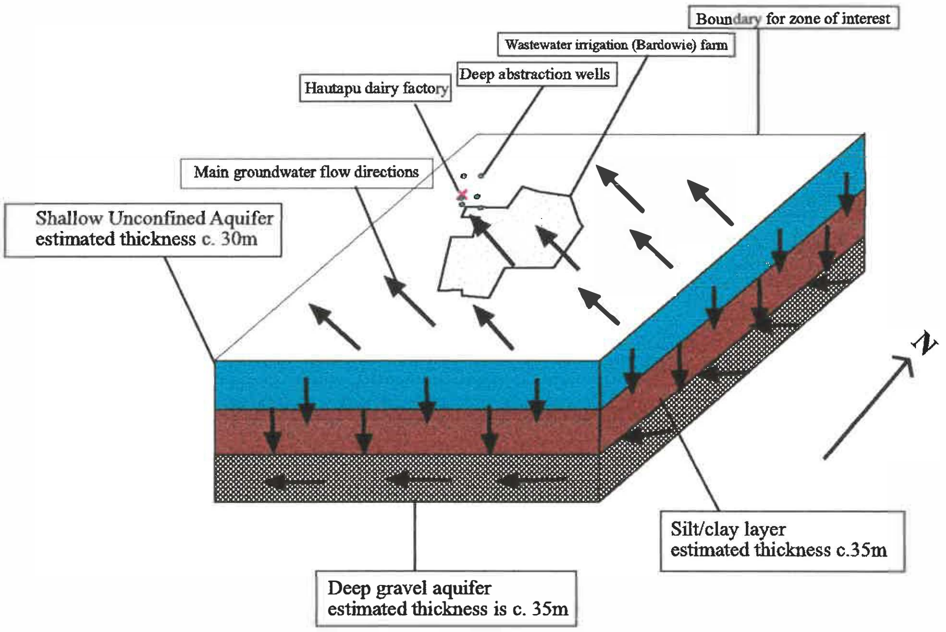


Figure 6.1 Conceptual model of the study area (not to scale).

As outlined in Chapter Three, the unconfined sand aquifer (blue) and the confined deep gravel aquifer (hatched) of the Hinuera formation are the main water bearing zones in the area and are separated by a low-permeability silt/clay layer (red). In reality, the thicknesses of these three layers vary however for the purpose of this simple model the thickness of the unconfined aquifer was set at an estimated average value of 30m and the thicknesses of the silt/clay layer and gravel aquifer were each set at 35m. These thicknesses were determined based on lithological information presented in Chapter Three. It is expected that the predicted results of the model results may underestimate vertical groundwater fluxes, particularly in locations where the silt/clay layer is thin or discontinuous.

6.3 The Hautapu groundwater flow model

The next stage of the modelling study involved the transfer of information from the conceptual model into a numerical model. A 3-dimensional finite difference modelling computer package (visual MODLFOW) based on the 3-dimensional finite difference code (MODFLOW) developed by McDonald and Harbaugh (1988) was employed for this. The basic theory behind this type of numerical model is covered in Appendix IV. The 3-D finite difference numerical code developed by McDonald and Harbaugh is constructed on a system of finite points (nodes), located in the centre of regularly spaced grid cells. In this model, linear head-dependent functions (flow equations) simulate flow between cells (in 3-dimensions) and are based on the difference in head (h) values between nodal points. Solution of the flow equations requires specification of all known input parameters and appropriate boundary conditions.

6.3.1 System geometry

The transfer of information from the conceptual model into visual MODLFOW began with generating the grid for the zone of interest defined in the conceptual model. A grid was set up consisting of ~7600 grid cells covering the 10.5 km^2 region within the study area, including the irrigation farm (Figure 6.2). The grid spacing was decreased around the pumping well field to simulate the influence of the wells more realistically. The final stage of defining the geometry of the system involved importing surface topography grid data, and defining the thicknesses of the hydrogeological layers (shallow unconfined aquifer, silt/clay aquitard, and deep gravel aquifer) (Figure 6.3). Layer 1 – the shallow aquifer was defined as unconfined, layer 2 – the silt/slay layer as confined, and layer 3 - the deep gravel aquifer as confined.

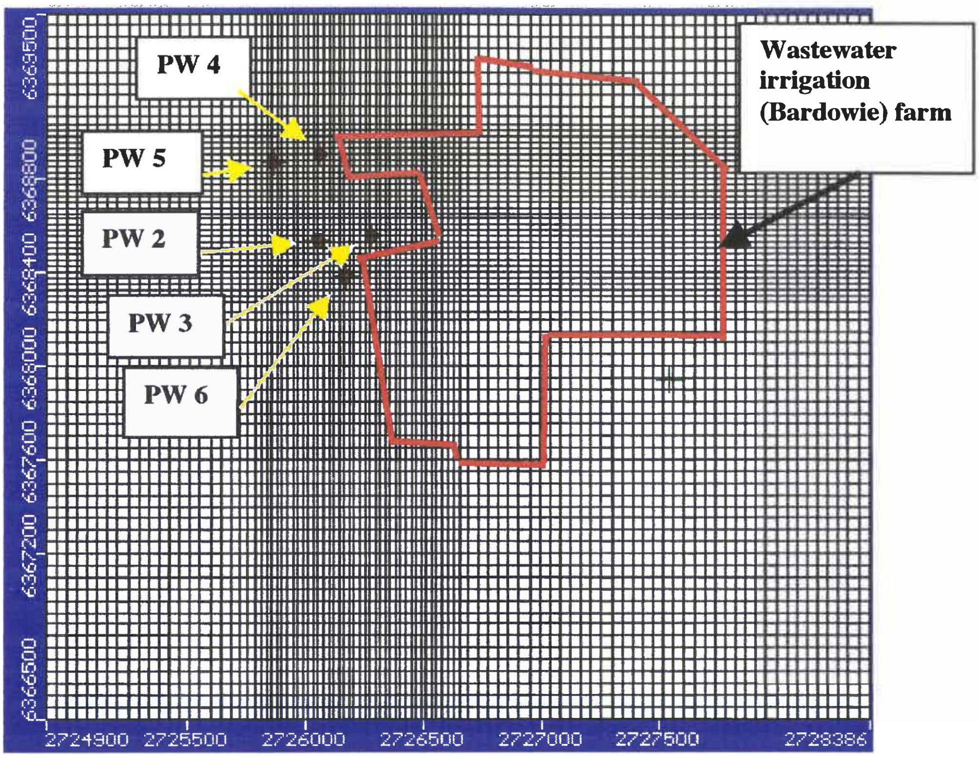


Figure 6.2 The grid structure and boundary of the Hautapu groundwater model, including the locations of the production wells. Grid based on NZMS 260 series.

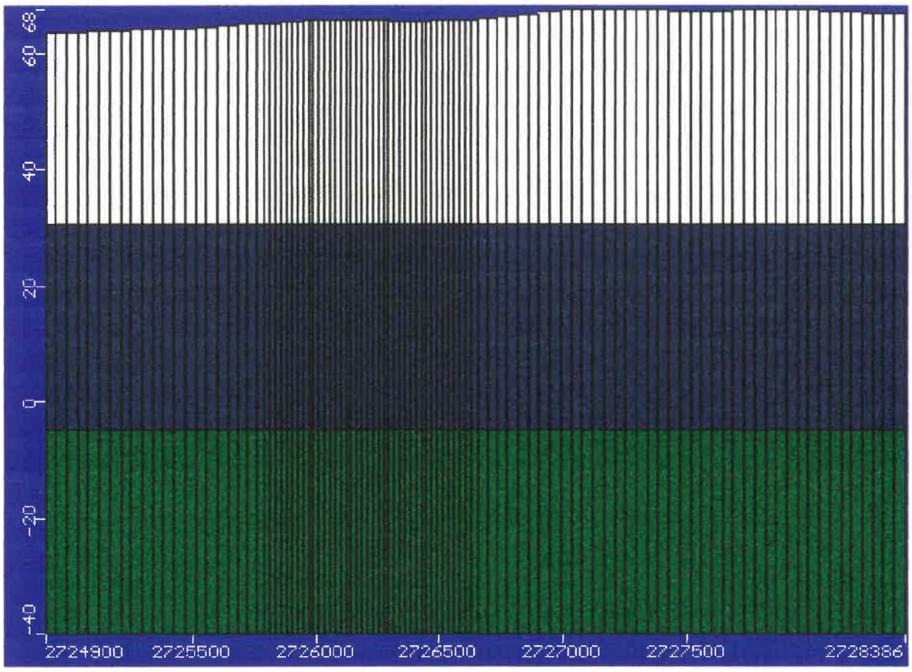


Figure 6.3 The vertical structure of the Hautapu groundwater model. The top layer (white) denotes the shallow aquifer, the second layer (blue) the silt/clay aquitard, and the third layer (green) is the deep gravel aquifer.

6.3.2 Input data

Standard numerical groundwater models, such as this, use all known head values (boundary conditions) to solve the flow equations for head at each node. The model is then calibrated which essentially involves refinement of hydraulic parameters (for example hydraulic conductivity) until outputs predicted by the model are realistic, or close to those observed. Calibration often requires intensive refinement of the model to develop an accurate representation of the groundwater system, therefore can be time consuming and difficult.

For this study, the calibration stage was minimised by specifying fixed head values for every cell within the shallow aquifer layer of the model, with these fixed head values being set at mean water table elevation (Chapter Three). Natural and artificial recharge and evaporative losses were hence taken into account by the model, in the sense that these parameters determine the water table position. Specification of head within the cells of the shallow aquifer layer involved:

(i) Extending and dividing the long-term water table contour map for the area defined by the monitoring bore network (Chapter Three) into sub-zones;

(ii) Assigning each sub-zone a long-term head value; and

(iii) Inputting the long-term head values, into the appropriate grid cells within the shallow aquifer layer of the visual MODFLOW model.

For each layer, both available and estimated values for hydraulic conductivity were then entered into the model. The best estimates for this parameter, and other K values assigned to the layers during the model refinement stage are included in Table 6.1.

hydraulic conductivity	*unconfined aquifer*	*silt/clay layer*	*deep gravel aquifer*
Kx (m day^{-1})	2.6*	0.018*	11.4*
	4.5	0.5	9.26
Ky (m day^{-1})	2.6*	0.018*	11.4*
	4.5	0.5	9.26
Kz (m day^{-1})	0.26 *	0.0018*	0.114*
	0.45	0.05	0.926

Table 6.1 Hydraulic conductivity values entered into the Hautapu groundwater model. * Best estimates determined through refinement of the model.

Pumping from the deep gravel aquifer was modelled by entering relevant information into the model, specifically the locations and average pumping rates of the abstraction bores (Table 6.2). The positions of the well in the model grid are shown in Figure 6.2. It is acknowledged that the predicted effects from pumping (hydraulic head, drawdown) will differ from those observed because an annual average pumping rate was used. The MODFLOW assumption that the wells are screened through the entire thickness of the aquifer may also allow for variation between the observed and predicted values.

Abstraction Bore	*Average pumping rate $m^3 day^{-1}$*	*Northing*	*Easting*
PW 2	369	6368532	2726055
PW 3	719	6368554	2726277
PW 4	483	6368901	2726058
PW 5	902	6368863	2725863
PW 6	249	6368386	2726172

Table 6.2 Information for the deep abstraction bores (location information from Dewhurst 1987, and pumping rate information from Anchor Products 1996).

While visual MODLFOW can account for both transient and steady state conditions, the latter option in which conditions remain constant were simulated in this model.

6.3.3 Boundary conditions

Specification of boundary conditions is required for any numerical groundwater model to allow the flow equations to solve for h. Constant head boundary conditions were set up for the deep aquifer, for which some limited head information was available. The boundary conditions for the deep aquifer, were established through the determination of a head gradient transect extending through the modelled area.

The head gradient was determined by plotting the elevation gradient from the recharge zone (Te Miro hills) to the discharge zone (Waikato River). Using the average static head level for the deep aquifer (on average ~20m below ground surface datum based on deep observation bore data on the irrigation farm) the head gradient for the deep aquifer was plotted beneath the elevation gradient line, from the recharge zone to the discharge zone. The positions of the western and eastern boundaries of the zone of interest were defined on the transect enabling the appropriate constant head boundaries to be determined. The western and eastern constant head boundaries were set as 48 and 52.2 metres above m.s.l. respectively.

6.3.4 Calibration

Calibration of the model required adjustment of the hydraulic parameter values for the layers of the model so that the flow regimes and zone budgets seemed realistic. This involved re-modelling using two estimates of hydraulic conductivity values for the each of the three lithological layers (Table 6.1).

6.4 Model outputs

The models predictions of flow within the Hautapu groundwater system are presented here as a series of equipotential contour plots. The contour plots incorporate scaled vectors indicating the direction and magnitude of flow. Groundwater flow within the system is viewed both vertically and laterally through plan view and cross sectional equipotential contour plots (north south, east west) respectively. Plan view and cross sectional drawdown contour plots are also presented, to further examine the effects of pumping from the deep aquifer.

6.4.1 Horizontal flow

Horizontal flow in the unconfined aquifer (Figure 6.4) reflects the specified fixed water table elevations entered into the model, with flow tending in all directions from the high head region on the eastern side of the modelled area. Flow is predominantly to the West, Southeast, and Northwest however the magnitudes of the vectors indicate flow velocities are greatest in the northwestern direction.

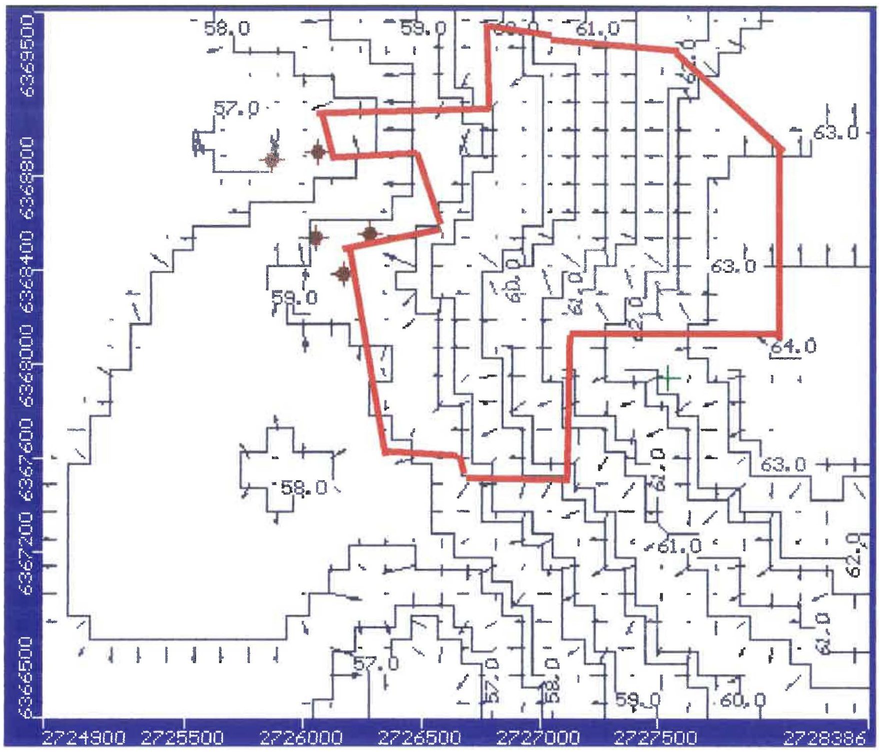


Figure 6.4 Horizontal groundwater flow within the shallow unconfined aquifer (scale x2). The approximate location of the irrigation farm (red) is shown.

Within the deep aquifer (Figure 6.5), westward flow predominates, as determined by the designated constant head boundaries. Apparent outward flow from the well field is a consequence of the differential head values in areas adjacent to the wells relative to inside the wells. Convergence to the pumping wells occurs from all directions as indicated by the concentric nature of the equipotential contours about the well field. Deviation from the normal pattern of flow (east – west) is observed up to 2km south of the well field and is more pronounced as the well field is approached. For example immediately west of the well field reversed flow (west – east) is predicted by the model.

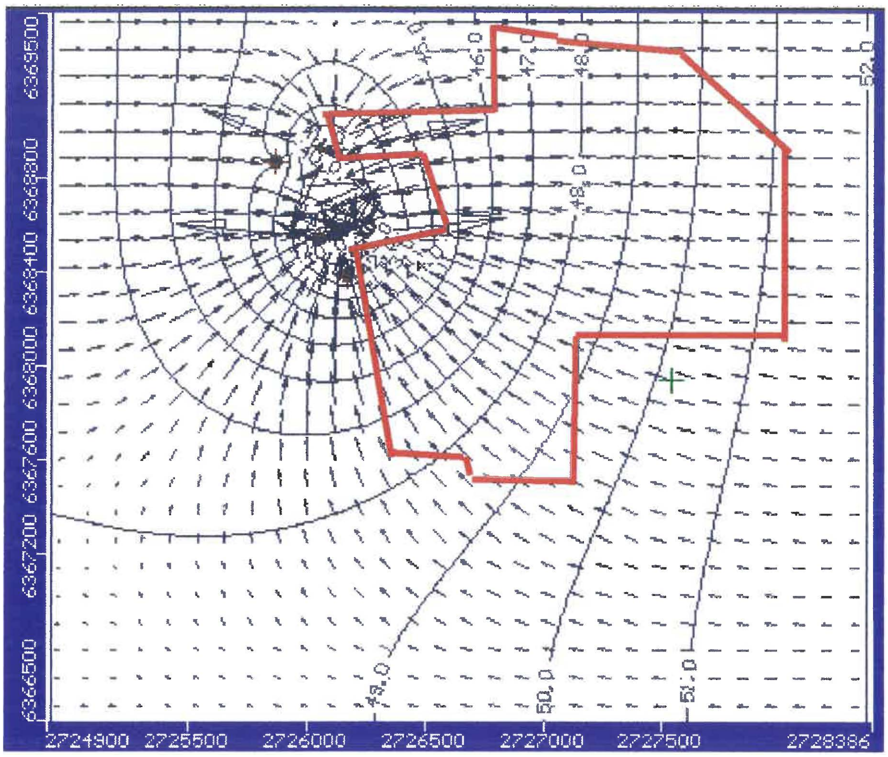


Figure 6.5 Horizontal groundwater flow within the deep gravel aquifer (scale x1). Apparent outward flow from some of the wells is due to the different predicted head levels inside wells relative to adjacent areas. The approximate location of the irrigation farm (red) is shown.

6.4.2 Vertical flow

Vertical flow within the groundwater system is represented by a series of cross sections across the modelled area. Cross sections in areas both within and near the well field are presented to illustrate the predicted effects on vertical flow from pumping. The locations of these cross sections are shown in Figure 6.6.

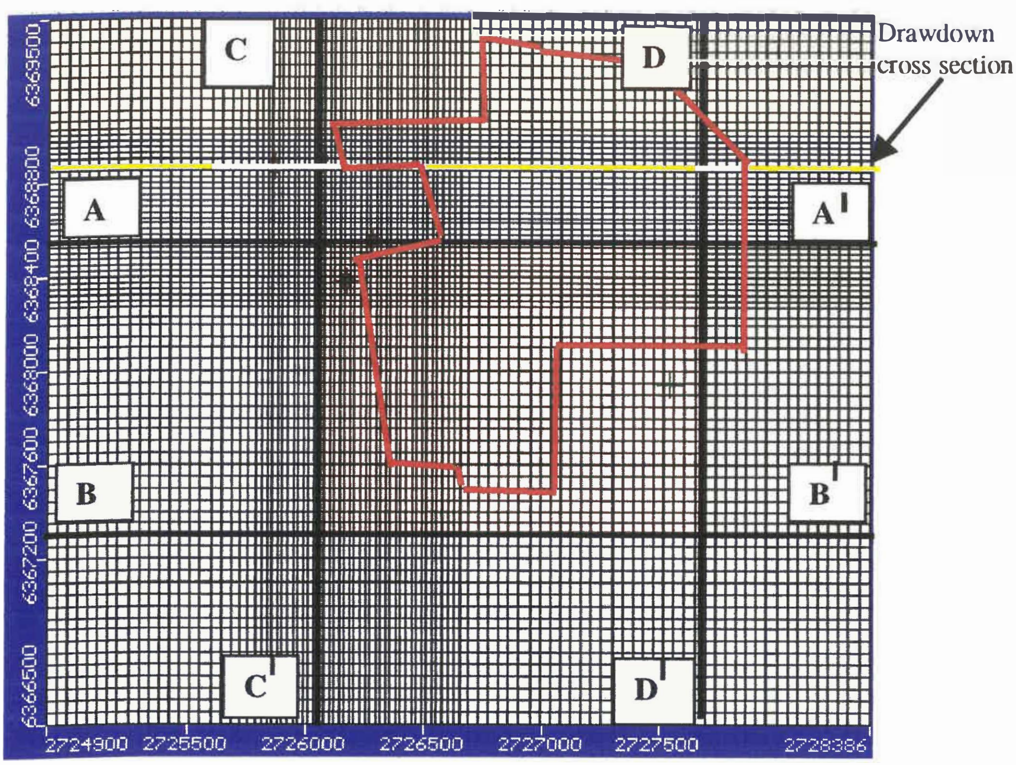


Figure 6.6 Locations of cross sections.

Within the well field, the model predicts (Figures 6.7a and 6.7b) steep vertical gradients therefore strong downward vertical flow from upper zones into the deeper aquifer. At distance from the well (Figure 6.8a and 6.8b), vertical flow downwards from the upper zone is still a feature of the groundwater system however the magnitudes of the vectors indicate slower vertical flow velocities in these areas. Away from the well field, vertical flow appears to be governed by natural vertical head gradients, with less effect from pumping apparent. It is noted that the vertical flow regimes in these areas are largely determined by the fixed head values assigned to the shallow unconfined aquifer and the constant head boundaries set up for the deep aquifer.

The total vertical flux from the silt clay layer to the deep aquifer, over the entire area of the model, is predicted to be 5917 m^3 day^{-1} equating to a vertical flow velocity of 0.00056 m day^{-1} or 0.5 mm day^{-1}). This vertical flow velocity estimate does not take into account porosity therefore is likely to be a conservative value.

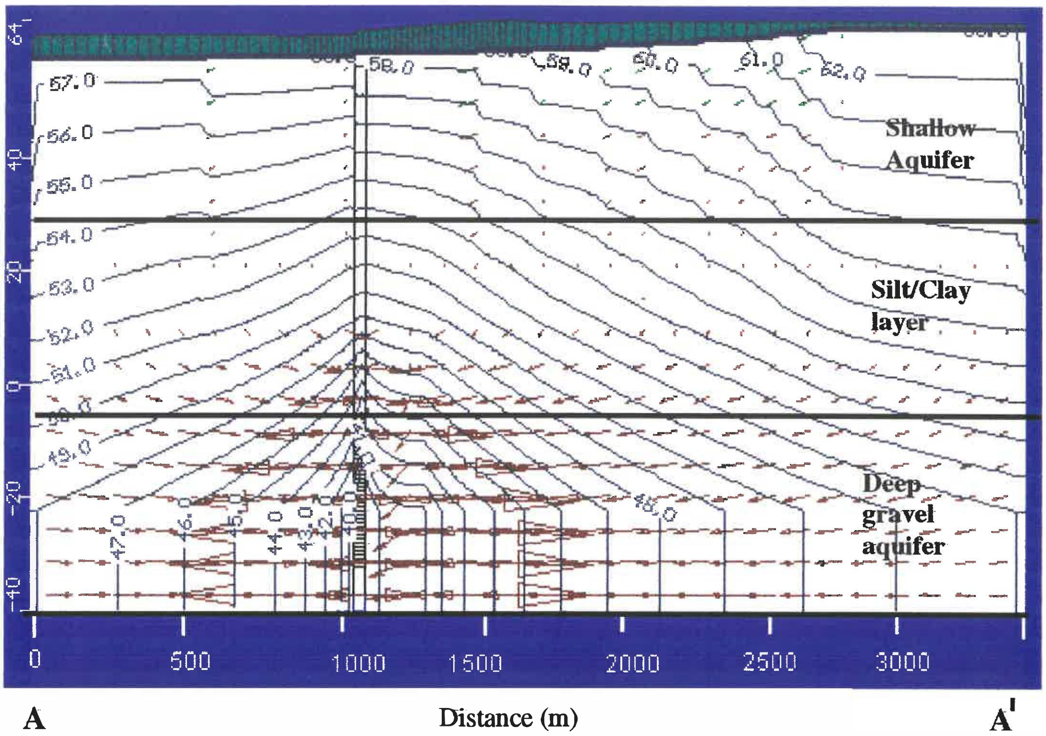


Figure 6.7a Cross section A-A' of the groundwater system within the well field (scale x1). Well shown is production well 3. Note cross section vertically exaggerated (x20).

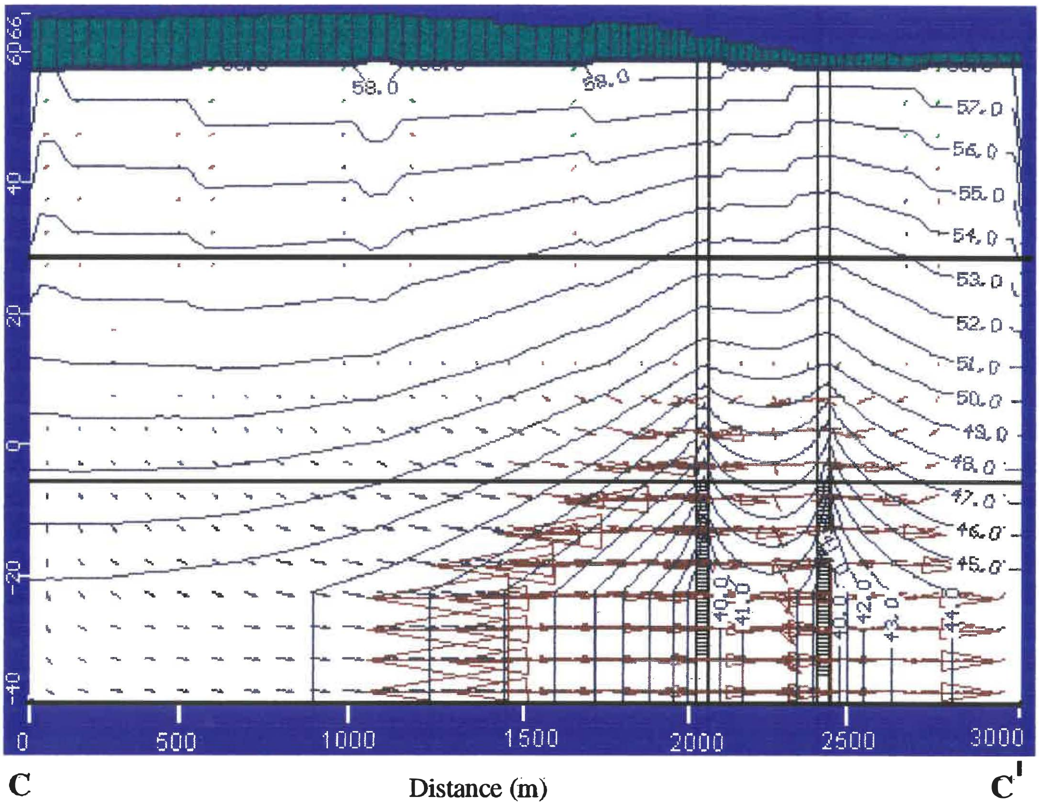


Figure 6.7b Cross section C – C' of the groundwater system within the well field (scale x1). Wells shown are production wells 2 and 4. Note cross section vertically exaggerated (x20).

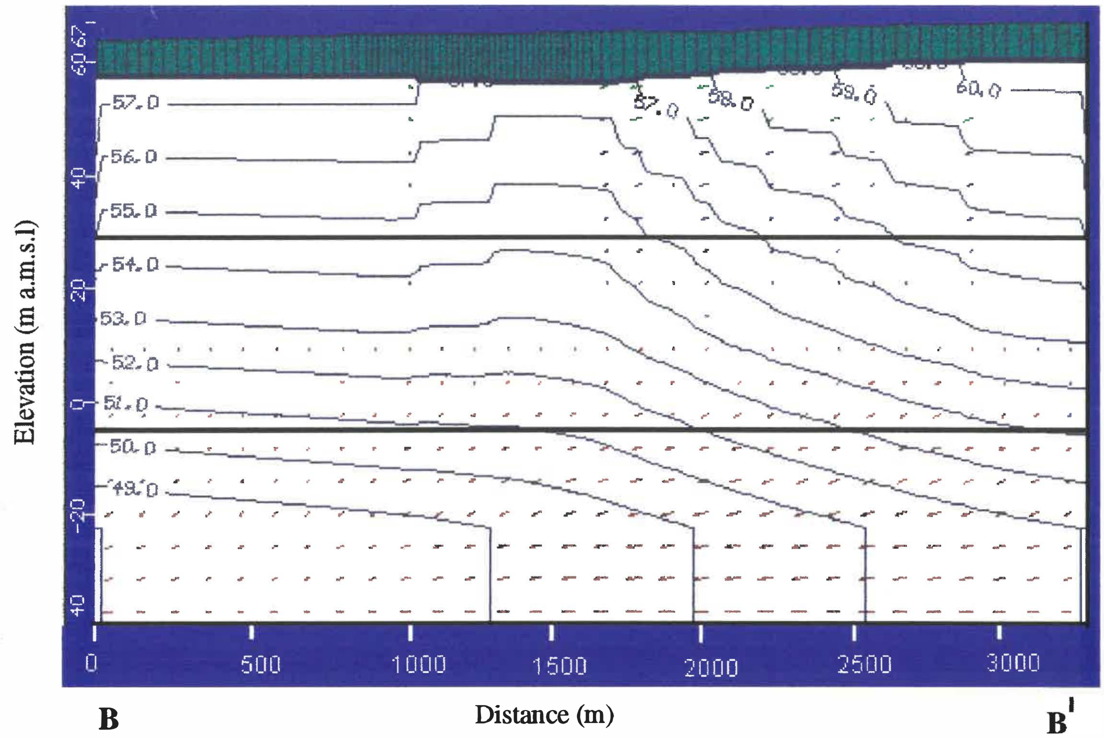


Figure 6.8a Cross section B – B' of the groundwater system at distance from the well field (Scale x2). Note cross section vertically exaggerated (x20).

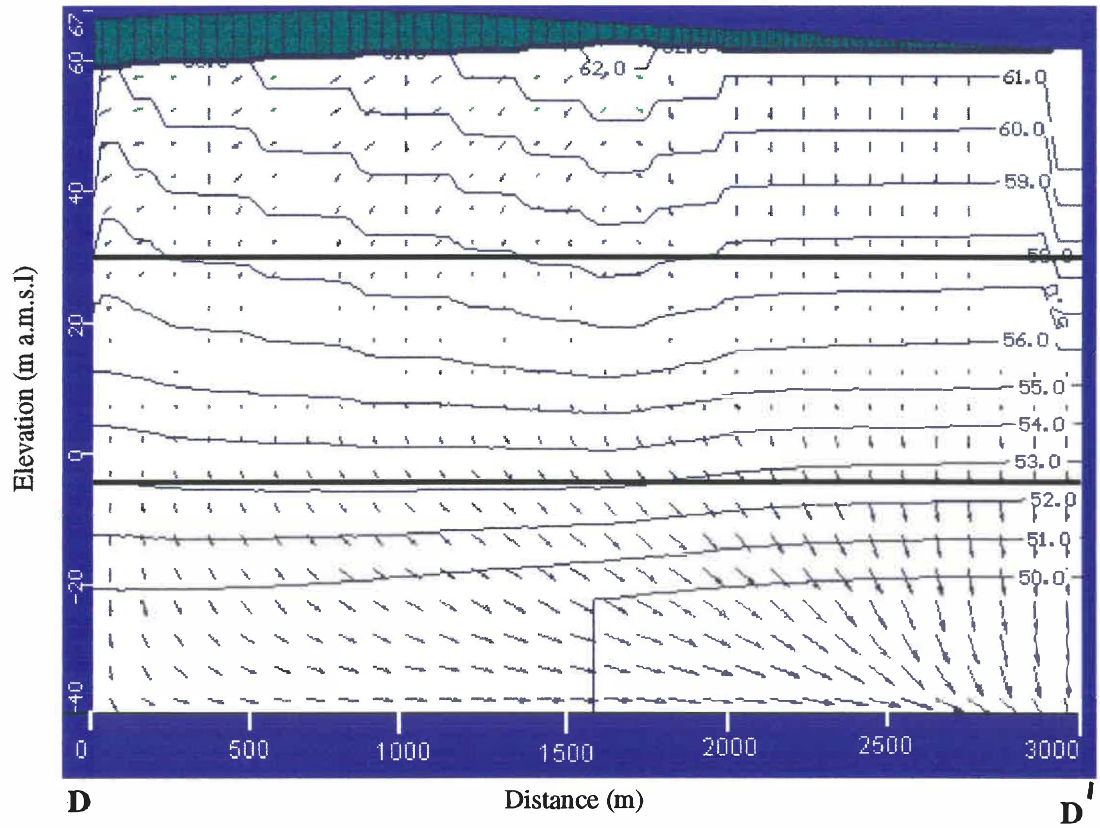


Figure 6.8b Cross section D – D' of the groundwater system at distance from the well field (Scale x1). Note cross section vertically exaggerated (x20).

6.4.3 Drawdown effects

As observed in the equipotential contour plots, the effects of pumping on groundwater flow are notable within the groundwater system, especially in areas nearest the well field. The model predicts convergence of flow within the deep aquifer toward the well field even at notable distances, implying some lowering of head levels (drawdown) over a radial distance of ~2 km. To define the areal extent of the drawdown effect associated with the annual average pumping rates, drawdown contour plots are presented.

According to the model, the drawdown effect from pumping is greatest in the deeper aquifer, and decreases notably to the surface such that, a drawdown range of 0-5m is predicted for the shallow unconfined aquifer (Figure 6.8a). Significant attenuation of the drawdown effect is predicted through the silt/clay aquitard layer because of the thickness and low hydraulic conductivity (0.5 m day^{-1}) of this layer.

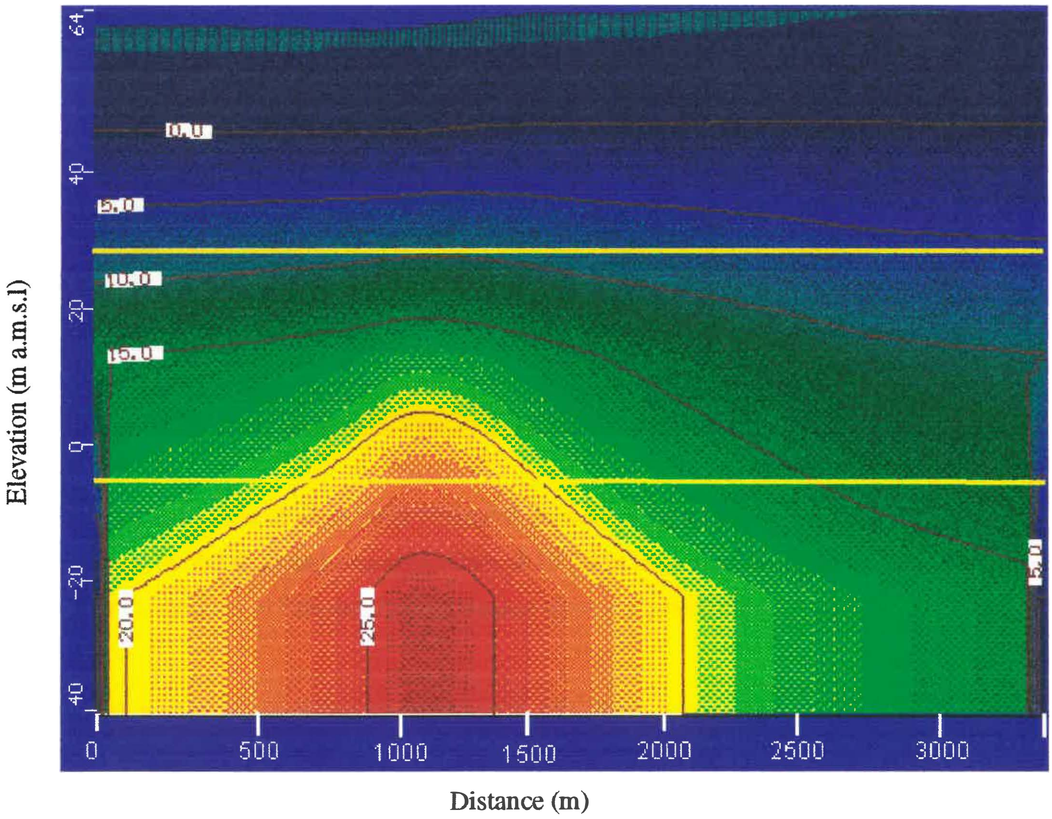


Figure 6.9a A cross section of drawdown within the groundwater system (refer to Figure 6.6 for location). Note water levels in the wells are higher than outside the wells resulting in an inverse drawdown cone centred over the well, thereby departing from reality. Well water levels are impossible to predict accurately, given that average annual pumping rates are used in the model.

On the basis that drawdown is mainly observed in the deep aquifer, a drawdown plot across the deep aquifer is shown (Figure 6.8b). The lateral extent of the drawdown effect in the deep aquifer, as predicted by the model, is clearly shown in this visualization. The model predicts a drawdown cone within the deep aquifer of considerable lateral extent. The radius of the drawdown cone is ~2km, with drawdown levels ranging from between 0m at the eastern boundary of the modelled area to 25m within the well field (northwest corner of the modelled area). These results are consistent with those obtained in a similar modelling study (Terra Aqua 1998).

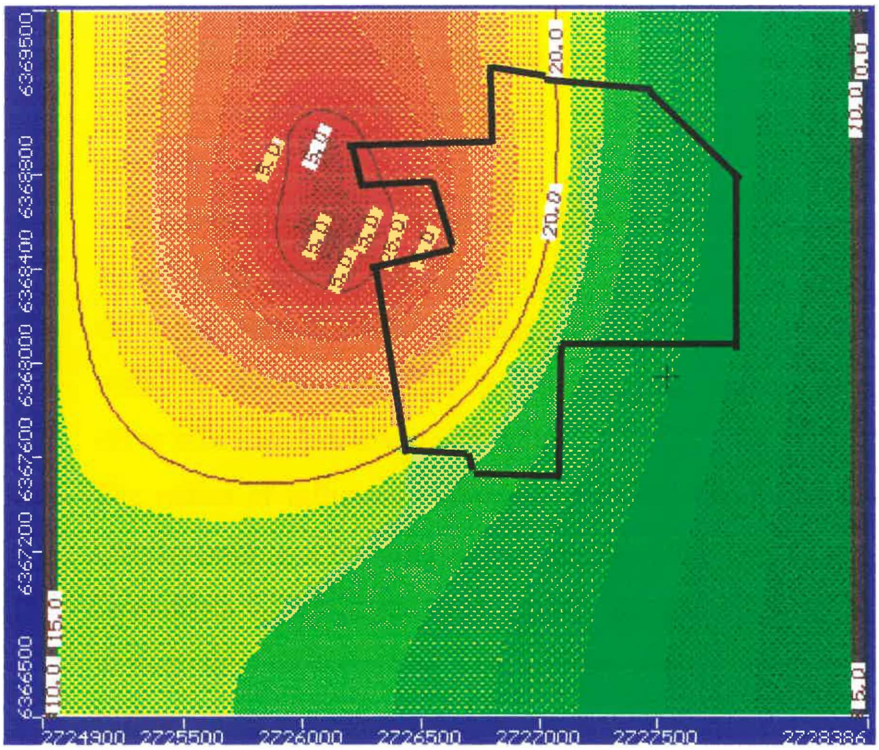


Figure 6.9b Areal extent of the annual averaged drawdown effect in the deep aquifer. Note that the effect observed on the farm and expected effect of 15-20 m drop in static head levels – realistic value based on levels reported in Chapter Three (changes observed averaging ~15m).

The model predicts drawdown values of 15–20m in the deep aquifer, beneath the irrigation farm. These values are consistent with the estimates for the deep observation wells based on limited data (Chapter Three). The results of the model confirm that induced vertical movement of modified groundwater from the shallow aquifer to the deep aquifer is possible, however the volumes involved are likely to be insignificant.

6.5 Conclusions

The finite difference numerical model, visual MODLFOW, used to simulate groundwater flow at Hautapu, has proven a useful tool for obtaining a first approximation to three dimensional flow dynamics within the study area. It has also permitted the effect of the pumping activity to be estimated to the extent that a steady state model has approximated a seasonal pumping regime.

According to the visual MODFLOW model:

(i) Slow vertical movement downward (0.0005 m day^{-1}) from the upper zones to the deep aquifer occurs within the study area;

(ii) Lateral flow in the shallow unconfined aquifer occurs in all directions however mainly to the northwest, a reflection of the water table gradient;

(iii) Groundwater flow in the deep aquifer is from east to west and at significantly greater velocities than predicted for the other layers of the model largely due to the higher hydraulic conductivity associated with this layer (11.4 m day^{-1}). Highest flow velocities in this layer occur near the abstraction well field.

(iv) The radius of influence within the deep aquifer due to the pumping activity (radius of the drawdown cone) is ~2km;

(v) Drawdown levels in the deep aquifer are estimated to be ~15-20m beneath the irrigation farm area, similar to drawdown levels observed within the deep aquifer on the farm.

The predicted drawdown levels for the deep aquifer in the area beneath the irrigation farm reaffirm that induced movement of modified groundwater is possible at the site. However, the thickness of the silt/clay layer will ultimately govern rates of vertical flow beneath the irrigation farm, such that the volumes involved are likely to be extremely small. Given that only small volumes of modified groundwater are likely to recharge the deep aquifer and that dilution of any modified groundwater in the deep aquifer will render concentrations of the wastewater constituents insignificant, it is anticipated that the deep aquifer will not be affected by the irrigation activity.

CHAPTER SEVEN

Conclusions and recommendations

7.1 Conclusions

7.1.1 Introduction

At Bardowie farm, the disposal of wastewater from the Hautapu dairy factory has impacted on the quality of shallow groundwater beneath the site. Traditionally, groundwater quality has been assessed directly through a comprehensive monitoring bore network installed by the factory, thereby involving a considerable financial investment. The present study aimed to develop a qualitative groundwater hydrological model to enable the long-term environmental sustainability of the current wastewater disposal system to be evaluated, and to test the application of electrical resistivity for indirectly and cost effectively assessing groundwater quality beneath the site.

7.1.2 The qualitative groundwater hydrological model

Integration of the results from the following investigations undertaken in the present study, has enabled the development of the qualitative model of groundwater movement beneath the site:

(i) The acquisition of site characteristic information;

(ii) A three-dimensional numerical groundwater modelling study using visual MODFLOW;

(iii) Analysis of recent geochemical data from the monitoring bore network installed by the Hautapu dairy factory; and

(iv) A groundwater quality survey, in which the electrical resistivity technique was employed.

Based on these investigations the following main conclusions (model outputs) are drawn with regards to the groundwater flow dynamics beneath the wastewater irrigation site:

- Slow horizontal movement of unconfined groundwater northwestward predominates at the site, the direction of regional groundwater flow, with some possibility of movement southward and westward;

- Vertical movement of shallow groundwater is highly limited due to the presence of thick sequences of low permeability silt-based materials at the base of the shallow unconfined aquifer;

- A high conductivity/low resistivity modified groundwater plume exists within the shallow unconfined aquifer beneath the wastewater disposal site;

- The high conductivity of the plume is largely attributed to the presence of elevated levels of sodium in shallow groundwater;

- Levels of sodium in the shallow groundwater have increased steadily since 1991 to the present. In contrast, nitrate levels have decreased steadily due to initiatives taken by the factory to reduce loadings of this nutrient to the wastewater disposal site;

- Horizontal movement of sodium rich modified groundwater is currently minimal, with the outer edges of the zone of increasing sodium coinciding with site boundaries. However, given the rise in levels at the boundaries of the farm, it may be that sodium levels in shallow groundwater in some areas immediately off-site are also increasing.

7.1.3 The application of the electrical resistivity technique

In general, the electrical resistivity technique has proven a relatively effective tool for assessing groundwater quality near the wastewater disposal site. The accuracy of the technique in detecting spatial variations in groundwater quality on-site is apparent, however beyond zones where a large contrast between ambient groundwater and modified groundwater exists, this indirect method may be subject to influences from other factors such as lithology. Consequently, drilling is advised in these areas to supplement the results of the surface investigations.

7.3 Recommendations

Although the study has greatly increased the knowledge concerning the geometry and size of the modified groundwater plume and flow regimes at the site, some further research is recommended to improve the understanding of modified groundwater movement. It is recommended that a temporal (transient) numerical groundwater model be developed to yield quantitative predictions of the long-term movement of modified groundwater and subsequent plume development. In addition, regular checks for increasing trends in sodium levels in off-site bores to the northwest should be undertaken to enable early detection of lateral movement in this direction.

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APPENDIX I

Climate Data

Table A1.1 Average monthly rainfall and evaporation. Rainfall data (1905-1997) and evaporation data (1970-1997) collected at Ruakura Climate Research Station, Hamilton.

Month	*Rainfall (mm)*	*Pan Evaporation (mm)*	*Effective Rainfall (mm)*
Jan	83.3	149.0	-65.7
Feb	78.6	120.9	-42.3
March	81.0	101.0	-20.0
April	97.2	61.7	35.5
May	110.1	34.4	75.7
June	124.0	20.8	103.2
July	123.1	24.7	98.4
Aug	111.6	37.7	73.9
Sept	97.4	58.7	38.7
Oct	101.3	87.3	14.0
Nov	93.8	115.3	-21.5
Dec	88.7	137.0	-48.3

APPENDIX II

Water level and topography data

Table AII.1 Monitoring bore information - including NZMS co-ordinates, elevations, and long term water level. X = data not available or not used.

Bore	Elevation (m)	Easting	Northing	Bore depth (m)	Depth to water table (m) 1983	1986	1990	1991	1992	1993	1998	Average (m)	Height (m a.m.s.l.)
2	65.94	2726740	6367595	8.1	X	X	X	6.19	6.26	6.30	6.25	6.25	59.7
3	67.22	2727060	6367833	6.6	4.14	3.42	3.45	X	X	X	3.61	3.66	63.6
4	66.3	2727349	6367824	5.3	2.87	2.09	2.26	X	X	X	2.26	2.37	63.9
5	67.33	2727275	6367817	6.6	4.11	3.36	3.27	X	X	X	3.44	3.55	63.8
6	65.8	2726744	6368232	5.5	4.42	3.94	4.00	X	X	X	4.05	4.10	61.7
7	65.21	2726480	6368446	7.0	4.88	4.70	4.91	X	X	X	5.30	4.95	60.3
8	64.88	2726988	6368052	7.0	2.12	1.38	1.33	X	X	X	1.54	1.59	63.3
11	64.97	2726532	6368036	6.8	6.22	5.32	5.16	X	X	X	5.23	5.48	59.5
12	64.91	2726268	6368420	7.9	6.79	6.05	5.89	X	X	X	6.12	6.21	58.7
14	66.14	2726734	6367959	6.4	4.43	4.13	3.94	X	X	X	4.00	4.13	62.0
16	66.64	2726856	6367825	8.2	X	2.64	3.73	3.98	X	X	4.19	3.64	63.0
17	65.17	2726376	6367954	8.5	X	5.60	5.36	5.52	X	X	5.82	5.58	59.6
18	64.86	2726463	6367630	8.6	X	6.61	6.25	6.27	X	X	6.47	6.40	58.5
19	64.97	2726526	6367503	9.3	X	6.96	6.52	6.53	X	X	6.69	6.67	58.3
20	66.46	2726783	6367189	8.7	X	7.59	7.08	7.12	X	X	7.24	7.26	59.2
21	66.2	2726576	6367196	8.9	X	X	X	X	X	X	7.74	7.74	58.5
22	67.25	2727074	6367125	11.4	X	X	X	X	X	X	7.83	7.83	59.4
23	67.45	2726990	6367634	5.5	X	X	X	X	X	X	3.48	3.48	64.0
24	66.7	2727618	6367667	5.2	X	X	X	X	X	X	2.69	2.69	64.0
25	65.2	2726374	6367954	16.0	X	X	X	X	X	X	6.84	6.84	58.4
26	62.81	2726760	6368732	9.9	X	X	X	X	X	X	3.15	3.15	59.7
27	64.23	2727290	6368640	8.7	X	X	X	X	X	X	2.26	2.26	62.0
A	65.55	2726625	6366982	10.0	X	X	X	X	X	X	8.27	8.27	57.3
B	64.8	2726678	6366796	10.0	X	X	X	X	X	X	8.10	8.10	56.7
C	65.65	2726744	6366562	13.0	X	X	X	X	X	X	9.04	9.04	56.6
D	64.45	2726074	6368297	10.0	X	X	X	X	X	X	5.15	5.15	59.3
E	65.1	2726054	6367843	10.5	X	X	X	X	X	X	6.59	6.59	58.5
F	65.75	2726042	6367468	10.0	X	X	X	X	X	X	7.13	7.13	58.6
G	65.05	2726022	6366849	11.5	X	X	X	X	X	X	7.07	7.07	58.0
N	59.25	2726018	6368834	4.0	X	X	X	X	X	X	2.06	2.06	57.2
S	63.07	2726061	6368110	10.0	X	X	X	X	X	X	4.85	4.85	58.2
T	62.97	2726091	6368492	9.6	X	X	X	X	X	X	4.68	4.68	58.3
U	62.37	2726534	6368822	10.3	X	X	X	X	X	X	3.05	3.05	59.3

APPENDIX III

Monitoring bore information

Table AIII.1 The depths and screen intervals of the Hautapu dairy factory monitoring bores. X = data not available.

Bore	*Northing*	*Easting*	*depth*	*screen interval*
2	6367595	2726740	8.1	6.085-8.085
3	6367833	2727060	6.6	4.9-6.4
4	6367824	2727349	5.3	x
5	6367817	2727275	6.6	5.07-6.59
6	6368232	2726744	5.5	3.9-5.41
7	6368446	2726480	7	4.48-6
8	6368052	2726988	7	5.12-6.6
10	6367420	2727220	6.09	4.29-5.79
11	6368036	2726532	6.8	x
12	6368420	2726268	7.9	x
13	6367010	2727260	8.3	6.842-8.342
14	6367959	2726734	6.4	x
15	6366580	2726960	x	x
16	6367825	2726856	8.2	x
17	6367954	2726376	8.5	x
18	6367630	2726463	8.6	x
19	6367503	2726526	9.3	x
20	6367189	2726783	8.7	x
21	6367196	2726576	8.9	x
22	6367125	2727074	11.4	x
23	6367634	2726990	5.5	x
24	6367667	2727618	5.2	x
25	6367954	2726374	16	x
26	6368732	2726760	9.9	x
27	6368640	2727290	8.7	4.7-8.7
a	6366982	2726625	10	8.9-10.0
b	6366796	2726678	10	8.9-10.0
c	6366562	2726744	13	11.9-13.0
d	6368297	2726074	10	8.9-10.0
e	6367843	2726054	10.5	9.4-10.5
f	6367468	2726042	10	8.9-10.0
g	6366849	2726022	11.5	10.4-11.5
h	6367820	2725400	8.47	4.47-8.47
hh	6368520	2725620	5.8	3.8-5.8
i	6367900	2725000	7.84	3.84-7.84
j	6368200	2725180	8.87	4.87-8.87
l	6368740	2725580		
m	6368800	2725620	6.9	5.8-6.9
n	6368834	2726018	4	2.9-4.0
o	6367417	2727895	5.223	x

p	6367300	2726000	x	x
q	6367010	2725490	x	x
r	6368000	2726220	10	x
s	6368110	2726061	10	x
t	6368492	2726091	9.6	x
u	6368822	2726534	10.3	x

APPENDIX IV

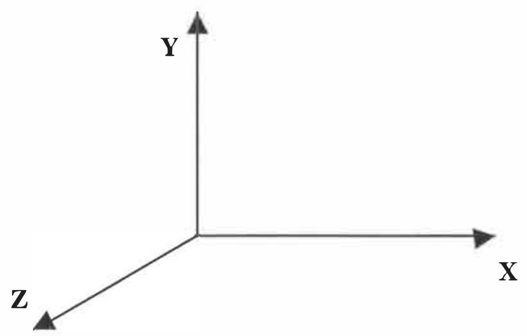
3D finite difference numerical models – basic theory

In numerical mathematical models, groundwater systems are approximated through a grid of points (nodes). A large number of simple linear equations (one for each node point) are solved for (by computer) to determine unknowns at each node (becomes the average for the cell), for example hydraulic head. Solution of these flow equations permits groundwater flow within the system to be simulated.

Finite difference numerical models, one of two types of numerical models commonly used for groundwater problems, are characterised by:

- (i) Regular rectangular grids;
- (ii) A finite set of nodal points inside or at the edges of grid cells;
- (iii) The computation of unknown variables such as hydraulic head (h), at the nodal points based on differences between cells.

Three-dimensional finite models, as the name suggests, considers flow in three dimensions (x, y, z).



As with all numerical models, 3D finite difference numerical models, aim to approximate, by way of numerical functions, the governing equations for groundwater flow to predict unknown variables such as hydraulic head levels throughout the groundwater system. The governing equations for three-dimensional flow in terms of

head are differential equations derived by combining the continuity equation and Darcy's law for discharge:

The continuity equation for groundwater (Namjou 1996):

$$\nabla.q + \frac{S\partial h}{\partial t} = q_s \quad \text{(for transient conditions)}$$

$$\nabla.q = qs \quad \text{(for steady state conditions)}$$

where ∇ is the divergence operator
q is the specific discharge (ms^{-1})
S is the specific storage (m^{-1})
H is the hydraulic head (m)
q_s is a volumetric flux per head unit volume and represents sources and/or sinks of water

Discharge according to Darcys Law (Namjou 1996):

$$Q = KA\,\frac{h_2 - h_1}{l}$$

or

$$q = \frac{Q}{A} = \frac{K\,h_2 - h_1}{l_2 - l_1}$$

where Q is the rates of flow (m3s-1)
k is the hydraulic conductivity (ms-1)
l is flow length.
A is the cross sectional area

Therefore for a volume in an aquifer the volume discharge rate (Q) is directly proportional to the head drop and to the cross sectional area.

The differential equation for three-dimensional flow in terms of head, combining Darcy's Law and the continuity equation (Namjou 1996) (For groundwater flow through an anisotropic, heterogenous aquifer under steady-state conditions):

$$\frac{\partial}{\partial x}\left(K_{xx}\frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_{yy}\frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial z}\left(K_{zz}\frac{\partial h}{\partial x}\right) + q_s = 0$$