Development and Demonstration of Approaches to Manage Drinking Water Quality on the Farm

A Rural Conservation Club  
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Final Report

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1.0 Introduction

1.1 Summary of Objectives

The main objective of the Conservation Club was to develop additional understanding of the causes and impacts of groundwater contamination from agricultural activity. The focus of this work was in areas that appeared to be the most susceptible to contamination based on the results of the Ontario Farm Groundwater Quality Survey (OFGQS). These areas included farms relying on shallow wells completed in permeable sediments such as sands and gravels where the water table was shallow and no lower permeability surficial soil was present to provide protection from surface sources of contamination. These conditions are very common throughout Southern Ontario.

Of specific interest during this project was the development of methodologies that could be used to inexpensively yet effectively delineate groundwater contamination in the vicinity of the farmstead and to identify the nature of the sources of the contamination. An additional goal of the project was to develop and implement methods of water well design and placement that would minimize the risk of contaminated groundwater being captured by the farm well.

In order to conduct the field investigations, several farmer collaborators were approached and incorporated with direct involvement in the project. This not only permitted the work to be done on actual farms the were experiencing groundwater contamination problems typical of those encountered during the OFGQS, but also provided for direct input and feedback from the farming community. A total of four farms were selected on the basis of the OFGQS results, field conditions and willingness of the farmer to collaborate. Although investigations were conducted at all four sites, the majority of the work focussed on a farm located in Haldimand-Norfolk County (OFGQS Site 1479). The results of the field work at the other three sites (OFGQS Sites 0806, 1640, and 1644) were presented in detail in the April 1995 Annual Report and are reviewed in the Executive Summary.

As the project developed, approaches to field methodologies and data analysis techniques also evolved. It was possible, for instance to make use of a newly published three-dimensional computer model to evaluate different pumping scenarios based on data from the field sites. This expanded the scope of the Conservation Club will beyond what we were originally intending. In addition, new field techniques were developed for use in mapping groundwater contaminants during the course of the project. These too were incorporated into the field studies and proved to be extremely useful.

1.2 Format of the Report

The report is divided into two main sections. The first part (Section 3.0) outlines the work associated with the field data collection, contaminant source evaluation, field testing of alternative pumping scenarios and computer modelling of the field conditions at Site 1479 along with the
relevant conclusions. The second part (Section 4.0) presents the results of the use of the newly developed field reconnaissance monitoring tool that was used to map groundwater contamination in the vicinity of Site 1479 in substantial detail. An Executive Summary has also been included summarizing the main conclusions from Sections 3.0 and 4.0.

1.3 Unique Aspects of the Project

In addition to the technical results developed through the course of the project, several different and unforeseeable benefits were realized as a result of the project. Specifically, two students from the University of Waterloo were able to pursue research work on different aspects of the project that lead to completion of their thesis requirements. The first was Ms. Parminder Takk, a student in Geological Engineering who worked on the application of the reconnaissance monitoring of the farmsite. Her thesis forms the majority of Section 3.0 of this report. She is now working with an Environmental Consulting firm in Toronto that are involved in the protection of groundwater resources on a regional basis and her experience in the agricultural environment will be a significant benefit to her and her company.

The second student is Ms. Tiffany Svensson who completed her M.Sc. thesis on work she carried out within the scope of the Conservation Club Project. Her work is summarized in Section 2.0. Ms. Svensson was hired by the Ontario Federation of Agriculture (OFA) as Projects Coordinator and has extensive contact with all components of the agricultural community. Her position with OFA and the very significant role she now plays in Ontario agriculture are a direct result of the experience she gained through her participation in the Conservation Club Program.

In addition, the results of the work have been disseminated throughout the scientific community as well as through the agricultural community in the form of poster presentations, technical talks-and journal papers. Currently, two technical papers are in the final stages of completion for submission to international, peer-reviewed journals. The work has continued to enhance the awareness of the farmers involved directly and indirectly in this work, the importance of scientific investigation associated with the development of practical strategies to maintain safe groundwater conditions not only in Ontario, but worldwide.

Many of the salient results of this study were incorporated in the Best Management Plan (BMP) that has recently been developed in the Province for groundwater wells. Ms. Svensson was a member of the committee in charge of developing this BMP.

1.4 Acknowledgements
The primary author acknowledges the support and contributions of Ms. Tiffany Svensson, Ms. Parminder Takk, Dr. Will Robertson, Mr. Paul Johnson, Dr. Tony Endres and Dr. Gary Parkin from the Waterloo Centre for Groundwater Research and Mr. Peter von Bertoldi and Ms. Bibi von Bertoldi from the University of Guelph. The foresight of Agriculture Canada is the main reason for any success achieved by the Conservation Clubs Program and we would specifically like to acknowledge Mr. Mike Hicknell and Ms. Nancy Cherny for their continued support, guidance and patience with our Project.

The main collaborators within this Project are the farmers who graciously contributed their time, land and in many cases their homes and barns to support our project. They have requested to remain anonymous. We could not have completed any of this work without their participation and wisdom. This type of collaboration builds bridges to the future as land-use management becomes a significant issue, one that requires collaboration of all stakeholders especially the Ontario farmer.

2.0 Executive Summary

1). At all farms investigated, nitrate concentration varied both spatially and temporally. At all points, nitrate levels were observed to decrease with depth and were generally below the drinking water limit at depths of 4 m below the water table. This agreed with the results of the OFGQS. Although the general distribution of nitrate did not change significantly during an annual cycle, concentrations were highly variable and may be reflecting fertilizer application schedules and climatic conditions.

2). On several of the farms where the stratigraphic conditions beneath the ground surface were more complex (ie. layers of clay separating sands and gravels), perched water table conditions were encountered. This means that infiltrating water becomes trapped on the lower permeability clay units above the main water table aquifer and very little groundwater circulation occurs. In these cases, if water wells are completed in the perched units, agri-chemicals applied at the ground surface can accumulate in these discontinuous units resulting in elevated levels of contaminants. These perched conditions can be identified by observing hydraulic head in different stratigraphic units on site and by monitoring both vertical and lateral hydraulic gradients. Different surface geophysical techniques were also used to detect perching. Perched conditions should be avoided in the selection of a location to complete a drinking water well.

3). Groundwater contamination at most of the sites appeared to be associated with more than one source. The primary sources included the application of manure and artificial fertilizers at ground surface and point sources of contamination such as septic weeping beds and
feedlots. Geochemical and isotopic tracer species including phosphate, nitrate, chloride, $^{18}$O and $^{15}$N have proven to be very useful in determining the specific source. Another potential source is road salt. This was detected at one of the farms where the drinking water well was located close to the road. New techniques using $^{36}$Cl as a target tracer isotope may be helpful in assessing the impact of road salt on the farm well.

4). In general, the deeper the water wells are constructed, the better chance they will have to encounter and maintain drinking water that is not significantly impacted by agricultural activity. Wells placed at several depths below the water table at a site where stratified nitrate contamination persisted clearly illustrated the benefits to placing the well deeper. In many instances, this will only be a few meters deeper which well not add significantly to the price of the well. Detailed vertical mapping of nitrate concentrations is required and is simple and inexpensive to carry out.

5). Field monitoring techniques that involve portable installation methods proved to be extremely useful in collecting all of the relevant data required to assist in the optimum placement of drinking water wells. These methods include coring and monitoring well installation techniques using hand-held electric or gas hammers and drive-point profilers that permit rapid mapping of the groundwater quality in the vicinity of a farm stead without the need for permanent monitoring well installations. These techniques will be applicable under a wide variety of conditions typical of Southern Ontario.

6). From the data that can now be easily and inexpensively collected at a given farm, newly developed computer simulators that run on PC platforms can be used to evaluate different pumping strategies for an individual farming enterprise to optimize groundwater production and minimize problems related to groundwater contamination.
Section 3.0

Influence of Variable Groundwater Extraction Strategies on Water Quality in Unconfined, Contaminant-Stratified Aquifers
ABSTRACT

An investigation was undertaken to evaluate the effects of variable groundwater extraction strategies on water quality in unconfined, contaminant-stratified aquifers. A suitable field site was chosen and subsequent groundwater monitoring activities revealed the aquifer to have high nitrate concentrations near the water table (approximately 15 mg/L NO$_3$-N) progressively decreasing with depth to less than 1 mg/L NO$_3$-N at depths greater than 5 metres below the water table. The aquifer was therefore considered to be contaminant-stratified with respect to nitrate. Groundwater drawn from the domestic well frequently had nitrate concentrations in excess of the drinking water limit (10 mg/L NO$_3$-N). The selected study site was a tobacco farm located in Haldimand-Norfolk, Ontario, near the town of Delhi. The research project consisted of two main components - field investigations and three dimensional numerical modelling.

In the field investigation the physical and chemical characteristics of the aquifer were determined and the source of the nitrate and its spatial distribution were delineated. In addition, three different well designs were tested under representative pumping conditions to evaluate the effects of different groundwater extraction scenarios on nitrate levels in the extraction water. Groundwater levels and nitrate levels were monitored in each pumping well and adjacent multilevel monitoring well during the pumping test. The field results were then used in numerical simulations and a sensitivity analysis was performed to assess key factors that influence stratified-nitrate capture by a deep short-screened pumping well.

The field results, together with the numerical simulations, demonstrate that appropriate pumping schedules and well design can substantially affect pumped water quality. Lower nitrate levels (< 3 mg/L NO$_3$-N) were noted when using a relatively short well screen placed at greater depths where nitrate levels are lower. Simulations indicate nitrate levels were about the same for intermittent versus continual pumping regimes however, greater vertical transport of nitrate was indicated during continual versus intermittent pumping. Greater differences in steady-state effluent concentrations between well designs were observed if the aquifer had a higher transmissivity. Low nitrate levels in a deep short-screened well were easier to maintain if the aquifer was anisotropic ($K_v < K_x$), or if the magnitude of the regional gradient was increased. Numerical modelling also illustrated that a short screened well increased the down gradient disturbance of the regional nitrate stratification in the aquifer and that steady-state nitrate distributions were reached sooner with a fully screened well versus a short screened well. Notable water quality improvements were made if the total daily volume from any given well was reduced. To capitalize on this observation a three well design was proposed as an approach to improving the drinking water quality. Simulations suggested greatest enhancement of water quality could be obtained by aligning three short screened wells parallel to the flow field. Specifically, two shallow wells bracket one deep well, where the pumping
rate of the deep well is minimized to solely provide water for domestic use with the balance of the water requirements supplied by two shallow wells.

Modelling together with the field evidence suggests that local contributions of nitrate during specific recharge events constitutes only a minor fraction of the total nitrate mass contained in the aquifer therefore a collective effort on a regional scale would be required to substantially reduce excess nitrate levels in the aquifer.
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1.0 INTRODUCTION

Nitrate is perhaps the most wide spread contaminant in groundwater, primarily in shallow unconfined aquifers. The largest contribution of nitrate to groundwater comes from agricultural activities (Follett, 1989; Follett et al., 1991; Hallberg, 1986a, 1986b, 1987a, 1989; Keeney, 1982, 1986; OECD, 1986; Pratt, 1984). This is of particular concern to rural residents since their main source of water is often derived from shallow wells in regional unconfined aquifers. The Ontario Farm Groundwater Quality Survey (Agriculture Canada, 1992) documented that 13% of domestic wells tested in Southern Ontario exceeded the maximum acceptable concentration for nitrate. Several other groundwater quality surveys across the United States have found similar results. In the early 1980's, 2.7% of 603,000 rural domestic wells across the United States had nitrate concentrations above drinking water standards. In agriculturally based states, specifically Iowa, Kansas, and South Dakota, the nitrate contamination in private water supplies rose as high as 25% (Nitrates and Groundwater: A Public Health Concern, 1988).

Although nitrate itself is relatively nontoxic to humans, it can potentially convert to nitrite, which poses a health risk. A landmark article was published in the Journal of the American Medical Association (Cormly, 1945). In the paper Dr. Cormly was the first to clinically recognize high nitrate concentrations in well water as a cause for methemoglobinemia in infants. Since then more than 2000 cases, including fatal poisonings in infants resulting from ingestion of water containing nitrates have been recorded in various countries throughout the world (Shuval et al., 1972; Lukens, 1987; Johnson et al., 1987).
Methemoglobinemia is a condition where hemoglobin, the normal oxygen carrying molecule in the blood, is altered to methemoglobin (chemically, the heme iron is oxidized from the ferrous to the ferric form). Methemoglobin does not have the ability to carry oxygen, therefore as methemoglobin levels rise symptoms of oxygen starvation occur. In extreme cases the victim suffocates. An infant is particularly susceptible to methemoglobinemia due to reduced levels of soluble NADH-cytochrome \( b_5 \) reductase from that of an adult. As a result, the capacity to reduce the unhealthy methemoglobin to healthy hemoglobin at birth is compromised (Lukens, 1987). The mortality rate among affected infants is about 8 to 10 percent (Nitrates and Groundwater: A Public Health Concern, 1988). Furthermore, nitrites derived from ingested nitrates, may react in vivo with secondary nitrogen compounds found naturally in some foods to form N-nitroso compounds, which are potent carcinogens in laboratory animals but their effect on humans is difficult to assess (Fraser et al., 1980).

In many areas, nitrate in groundwater has escalated to become a problem in recent decades (Regional Ground-Water Quality, 1993). Globally, studies show trends which indicate nitrate contamination is increasing. In the Big Spring Basin of Iowa, nitrate contamination of the aquifer has steadily risen from less than 1 mg/L \( \text{NO}_3^-\text{-N} \) in the 1930's to an annual average of 10 mg/L \( \text{NO}_3^-\text{-N} \) (Hallberg et al., 1983, 1984, 1985; Hallberg 1987a, 1987b, 1989; Libra et al., 1986, 1991; Kapp, 1986; Padgitt, 1986; Rowden and Libra, 1990). In Merrick County, Nebraska, the average nitrate concentration in groundwater as assessed through a well monitoring program has increased from 2.5 mg/L \( \text{NO}_3^-\text{-N} \) in the 1940’s to 11- 12 mg/L \( \text{NO}_3^-\text{-N} \) in the mid-1970's (Spalding et al., 1978). The rate of increase in nitrate contamination beneath cultivated land was estimated to be
0.1-1.1 mg/L NO$_3$-N per year (Exner and Spalding, 1974: Spalding and Kitchen, 1988). Aquifers underlying fine textured soils with thicker vadose zones appear to have a lower rate of annual increase in nitrate contamination (Spalding and Kitchen, 1988).

In an agricultural area of the former Soviet Union, average nitrate concentrations in wells rose from 0.8 to 6.5 mg/L NO$_3$-N over a 10-year period (Kudeyarov and Bashkin, 1980). Similarly, in Europe, increasing concentrations have been documented in many agricultural areas (Hallberg, 1986a, 1987b; OECD, 1986; Strebel et al., 1989). In rural England, for example, nitrate concentrations have increased at a rate of 0.2-1.0 mg/L NO$_3$-N per year for the past 10-20 years and are expected to continue to increase over the next 40 years because of slow solute transport rates in some aquifers (Carey and Lloyd, 1985; Howard, 1985). Therefore, nitrate is not only a current but growing problem for farms and is a contaminant that will most likely have heightened attention in the future as more people move to rural settings and become reliant upon groundwater.

Current literature is sparse in the area of minimizing the capture of stratified contaminants (such as nitrate) in a domestic well using screen length, screen placement, and realistic pumping schedules. However, significant attention has been placed on contaminant capture and pump-and-treat methods as remediation techniques (Conant, 1991). High effluent concentrations are desirable during pump-and-treat operations, whereas low effluent concentrations are required for water supplies and domestic wells (Akindunni et al., 1995). The field investigation sought to use previous findings and recommendations in the literature, particularly a study by Gillham et al. (Ontario Ministry of the Environment, 1983), Conant et al. (1995) and Akindunni et al. (1995) to
examine the possibility of minimizing nitrate capture by well design and realistic pumping strategies. In the study conducted by Gillham et al. various well designs were tested near Alliston, Ontario in an attempt to minimize nitrate in the effluent. The investigation used a continual pumping rate while the current study sought to emulate a typical intermittent pumping pattern of a small dairy farm. In the studies by Conant et al. (1995) and Akindunni et al. (1995) the objective was to maximize contaminant capture in a contaminant-stratified unconfined aquifer at C.F.B. Borden, Ontario. A comparison between parameters of the current investigation and the C.F.B. Borden study can be seen in Table 3.

Recognizing the present and growing concern of nitrate contamination in domestic drinking water, the main objective of the current research was to evaluate the influence that common groundwater extraction practices have on the quality of the domestic water supply in shallow, unconfined aquifer systems. On the basis of this evaluation, alternative pumping scenarios were investigated which could potentially minimize the effects of groundwater extraction on the transient water quality in contaminant-stratified aquifers.

Detailed site investigations were carried out at a representative farmstead located in Haldimand-Norfolk County near the town of Delhi, Ontario. The study area was located on a flat-lying sand plain with the regional unconfined aquifer sited as having predominantly horizontal flow (Novakovic and Farvolden, 1974; Ryan, 1994).
The groundwater drawn from the domestic well at the site regularly contained nitrate concentrations in excess of the Provincial Drinking Water Objectives of 10 mg/L NO₃-N. Detailed piezometric, geochemical, and isotopic mapping was conducted to assess the cause of the elevated nitrate levels in the well. This represented one objectives of the field investigation.

The hydraulic parameters of the unconfined aquifer at the study site were determined through a pumping test and falling head permeameter tests conducted on disturbed core samples. This information was then used to design and test three different well configurations in the aquifer sequence. These included a fully screened well, a deep short screened well, and a shallow short screened well. Each of the well designs was subjected to the same pumping pattern and the evolution of the hydraulic head and nitrate distribution within the contaminant-stratified aquifer were monitored.

Incorporating the field observations, numerical modelling was undertaken to provide further insight into the flow and contaminant transport processes occurring under the different pumping scenarios. Simulations were conducted to test various groundwater extraction strategies, in order to determine the parameters which most influence nitrate capture in the domestic well. Further modelling was carried out in an attempt to design alternative methods to minimize the transient effects on the contaminant-stratified aquifer. The hydraulic and geochemical results obtained during the field investigation were used to define the boundary conditions, establish the initial conditions, and to calibrate the three-dimensional variably saturated flow and transport model.
To document the influence of variable groundwater extraction strategies on water quality in unconfined, contaminant-stratified aquifers, three aspects of the study are separately discussed. Firstly, the investigation will establish the aquifer hydraulic properties followed by the spatial and temporal extent of nitrate contamination on site. With the support of chemical indicators this leads to delineating nitrate stratification and source in the domestic water supply. Secondly, three well designs are evaluated in the field for effectiveness in minimizing the capture of stratified nitrate contamination. Thirdly, the design and results from a numerical model were used to provide further insight into the parameters which most influence nitrate capture. A general discussion follows which attempts to correlate the field investigation with the numerical simulations.

In order to more fully understand and interpret the field results some fundamental facts on the persistence and fate of nitrate in groundwater and related information will be presented next.
Nitrate is a highly soluble and mobile compound. The bacterially mediated conversion of nitrate into molecular nitrogen is the only geochemical process that contributes significantly to the depletion of nitrate in contaminated aquifers (Freeze and Cherry, 1979; Starr and Gillham, 1993). Nitrate is usually limited to the upper portion of the aquifer and concentrations tend to diminish with depth (Trudell et al., 1986; Geyer et al., 1992; Starr and Gillham, 1993). This layering of nitrate can be referred to as stratified contamination (Akindunni et al., 1995). A conceptual model to describe a potential mechanism for nitrate-stratification on a regional scale is described by Hallberg and Keeney (Regional Ground-Water Quality, 1993) (Figure 1). On a local scale Ryan (1994) presents a conceptual model (Figure 2) explaining the formation of nitrogen-rich microplumes below the water table. Both conceptual models provide a mechanism leading to contaminant stratification from non-point sources of nitrogen over a large area. Furthermore, there are potential point sources of nitrate, septic weeping beds for example, that generate long pencil like plumes can develop local contaminant stratification (Robertson et al., 1991; Harman, 1992).

In general, possible sources of nitrate contamination include manure, commercial organic and inorganic fertilizers, crop residue, wastes from residential septic systems, and natural organic material in the soils. It is often very difficult to distinguish the major cause of nitrate contamination in a domestic well given the numerous potential sources and transformations of nitrogen that occur in the nitrogen cycle. The nitrogen transformation reactions that occur in shallow aquifers include the following (Freeze and Cherry, 1979: Stumm and Morgan, 1981):
Ammonification

\[ \text{CH}_2\text{O(NH}_3\text{)} + \text{O}_2 \rightarrow \text{NH}_4^+ + \text{HCO}_3^- \]

- **organic material**
- **dissolved oxygen**
- **ammonium**
- **bicarbonate**

Nitrification

\[ \text{NH}_4^+ + 2\text{O}_2 \rightarrow \text{NO}_3^- + 2\text{H}^+ + \text{H}_2\text{O} \]

- **ammonium**
- **dissolved oxygen**
- **nitrate**

Denitrification

\[ 4\text{NO}_3^- + 5\text{CH}_2\text{O} + \text{Bacteria} \rightarrow 2\text{N}_2 + 5\text{HCO}_3^- + \text{H}^+ + 2\text{H}_2\text{O} \]

- **nitrate**
- **organic material**
- **bicarbonate**

Dissimilatory Nitrate Reduction to Ammonium

\[ \text{NO}_3^- + \text{H}_2\text{O} + 2\text{CH}_2\text{O} \rightarrow \text{NH}_4^+ + 2\text{HCO}_3^- \]

- **nitrate**
- **organic material**
- **ammonium**
- **bicarbonate**

Ammonification and nitrification tend to occur in the unsaturated zone, whereas the nitrate reduction reactions of denitrification and dissimilatory nitrate reduction are generally seen below the water table where oxygen deficient conditions are more likely to exist (Freeze and Cherry, 1979).

Denitrification is the only nitrate consuming process which effectively and permanently removes nitrate from the aquifer. Most denitrifying bacteria are facultative anaerobes, and thus denitrification proceeds only in the absence of appreciable dissolved oxygen (Knowles, 1982;
Brock et al., 1984; Trevors, 1985; Starr and Gillham, 1993). Conversely, the presence of dissolved oxygen would indicate that denitrifying conditions are not present at the sample location but does not eliminate the potential for these conditions to exist elsewhere along the flow path. The four conditions required for denitrification are generally taken to be the presence of: (1) nitrate; (2) labile organic carbon; (3) denitrifying bacteria; and (4) reducing conditions (Starr and Gillham, 1993).

In unconfined aquifers recharged by precipitation, $\delta^{15}$N$_{NO_3}$ values in shallow groundwater may reflect the sources of nitrate that are present on the overlying landscape (Komor and Anderson, 1993). The ratio of the two stable isotopes of nitrogen, $^{15}$N and $^{14}$N, in nitrate have frequently been used to delineate the source of nitrate contamination (Kreisler et al., 1979; Kreitler and Browning, 1983; Flipse et al., 1984; Flipse and Bonner, 1985; Kaplan and Magaritz, 1986; Wells and Krothe, 1989; Aravena et al., 1993). The ratios of these isotopes are commonly referenced to the international standards atmospheric nitrogen, and reported as $\delta^{15}$N$_{NO_3}$. The reported value is calculated as follows:

$$\delta^{15}$N$_{NO_3}$ = \left[ \frac{(^{15}N/^{14}N)_{sample}}{(^{15}N/^{14}N)_{air}} - 1 \right] \times 1000$$

The $\delta^{15}$N$_{NO_3}$ values found in groundwater have been observed to vary from -1.0% to +40.0%. Chapelle (1993) reports common ranges of isotope composition for different sources of dissolved nitrogen in groundwater. These values are -3.5% to +2.0% for inorganic fertilizer nitrogen, +4.5% to +9.0% for soil organic nitrogen, and +9.0% to +17.8% for human and animal waste nitrogen.
Other references typically state that nitrate derived from commercial fertilizers have $\delta^{15}N_{NO3}$ values from -4% to +4%, nitrate derived from natural organic material in soil have $\delta^{15}N_{NO3}$ values from +3% to +8%, and nitrate derived from animal and human wastes have $\delta^{15}N_{NO3}$ values in the range of +10% to +22% (Freyer and Aly, 1974; Heaton, 1986). The $\delta^{15}N_{NO3}$ values that bound these categories are not rigidly defined, but can be used to classify most $\delta^{15}N_{NO3}$ values according to their respective sources (Komor and Anderson, 1993).

A factor which complicates the use of isotopic signatures in delineating the source of nitrate is that nitrogen isotopes can be fractionated during transport through the unsaturated zone or aquifer. For example, denitrification and dissimilatory nitrate reduction cause an enrichment in $^{15}N$ in the residual nitrate since microorganisms tend to discriminate against the heavier isotopes (Kaplan, 1983; Heaton, 1986; Smith et al., 1991; Chapelle, 1993). Gormly and Spalding (1979) showed an inverse correlation between $\delta^{15}N_{NO3}$ and nitrate concentrations in groundwater. This reflects the fact that groundwater with lower concentrations of nitrate may have undergone more denitrification than the high nitrate groundwater (Chapelle, 1993). Therefore, gradual denitrification and dissimilatory nitrate reduction along groundwater flow paths would cause a decrease in nitrate concentrations and an increase in $\delta^{15}N_{NO3}$ values (Komor and Anderson, 1993).

Volatile loss of ammonium and ammonia from commercial fertilizers and manure is also a cause for $^{15}N$ enrichment since the lighter isotope $^{14}N$ would preferentially volatilize (Kreitler, 1979). It does appear that denitrification, dissimilatory nitrate reduction, and volatilization do not increase $\delta^{15}N_{NO3}$ values above +10%. Therefore, even with the complication of fractionation, this method is used to distinguish animal or human waste contamination from other sources of nitrate (Flipse
and Bonner, 1985). As with all isotope techniques, mixing and exchange of nitrogen among different nitrate sources can occur (Komor and Anderson, 1993). By altering the δ^{15}N_{NO_3} values of the water, these processes can make it difficult to identify the original sources of nitrate in shallow groundwater. It is therefore imperative to correctly assess the physical flow system as well as the geochemistry. Furthermore, supplementary inorganic indicators such as sodium and groundwater electrical conductivity (EC) can be used to identify a human or animal source, such as a septic plume, from background waters. Typical septic system constituents such as sodium and nitrate can occur in plumes at more than 50% of the source concentration for very large distances (Robertson et al., 1989). Measurements of groundwater EC in these plumes have been found to be two to five times higher than background levels (Robertson and Blowes, 1995). This supplementary information can support isotope findings and therefore aid in delineating a nitrate source.
3.0 METHODS AND MATERIALS

3.1 Site Description

The site was selected based on information collected as part of the Ontario Farm Groundwater Quality Survey (OFGQS) data set. This survey was conducted between the fall of 1991 and the summer of 1992. The selected study site has highly permeable soils, high nitrate concentrations in the domestic well, a septic system, and agricultural activities which involved inorganic fertilization and irrigation. The conditions at this site made it highly susceptible to potential nitrate contamination in the domestic water supply and was, in essence, a worst-case scenario (Agriculture Canada, 1992).

The farm study site is located approximately 10 km north of Delhi, Ontario (42º58'N, 80º28'W) (Figure 3) on the Norfolk sand plain. The sand plain is a flat-lying physiographic feature, with a regional topographic gradient typically less than 0.25%, and local slopes with gradients up to 0.7% (Figure 4). The surficial sediments range from very fine to coarse grained sands of glaciolacustrine origin and are on average 9 m thick in the vicinity of the study site. The sands overlie stratified clay, silt and fine sand sediments of lower permeability (Novakovic and Farvolden, 1974; Ontario Institute of Pedology, 1984). Based on the results from the OFGQS in the multilevel monitoring well (ML1), the saturated thickness of the unconfined aquifer was approximately 7 m and groundwater flow was primarily horizontal. Depth to water table from the moderately flat-lying ground surface at the site was approximately 2 m (Novakovic and Farvolden, 1974; Agriculture Canada, 1992).
Figure 5 illustrates the layout of the site. The site accommodates two residential buildings with active and abandoned septic systems, an office, and a set of greenhouses in a garden and lawn setting. Adjacent to the homestead lies an inactive barn and barn yard where historically cattle were maintained until 1975. Surrounding this area are cultivated fields which were cropped to tobacco and rye on a rotating basis. The field received low amounts of inorganic fertilizer (30 kg N per ha) when tobacco was cultivated. Tobacco was grown on the field adjacent to the domestic well in 1990, 1992 and was planned for the 1994 growing season. Irrigation was applied to the field as required to maintain a healthy crop. While livestock existed at the site the solid manure was disposed on remote fields but never applied to the fields adjacent to the house.

Drinking water is produced from a sand point well approximately 4 m deep and is located in the basement of the house. Two additional sand point wells of similar depth and design are located by the office building and near the greenhouses (Figure 5). The total water demand and septic system load of the current farm practice was comparatively low since the residence has only one occupant. Water for irrigation was provided by a pond sustained by groundwater, located approximately 100 m south of the inactive barn but out of range in Figure 5.

The results from the OFGQS showed nitrate concentrations in the domestic well which exceeded provincial drinking water objectives, and vertical stratification of nitrate contamination beneath the cultivated field adjacent to the house. Concentrations ranged vertically from 0-13 mg/L NO₃-N with significantly lower concentrations located at depths greater than 7 m below ground surface.
3.2 Instrumentation and Sediment Coring

In addition to the multilevel monitoring well installed as part of the OFGQS (ML1) three multilevel monitoring wells were installed across the site to more fully assess the seasonal changes in vertical stratification and the lateral distribution of nitrate contamination in the groundwater at the study site. Positioning of the multilevel monitoring wells was based on overlying conditions and proximity to potential nitrate sources. The four multilevel monitoring wells were located in the agriculturally-active tobacco field, on the edge of the active field, and beside the domestic well and active septic system (Figure 5). The monitoring wells were also utilized, together with three uniquely designed pumping wells (described below) to evaluate the influence of extraction strategies and well design on stratified contamination by monitoring vertical gradients and nitrate distributions during pumping.

The multilevel monitoring wells were constructed from a 1.5 cm ID PVC centre stock with external PVC couplings where required. Individual 1.0 cm ID polyethylene tubes were fastened to the centre stock at different levels. Each of the polyethylene tubes and the PVC centre stock were furnished with a 10 cm screen wrapped with NITEX, a fine mesh nylon cloth. The multilevel monitoring wells did not have a sand pack placed around each screen. However the non-cohesive material was allowed to naturally collapse up to the water table where the remaining 0.115 m borehole was backfilled with resident aquifer material to surface.
Multilevel monitoring well 1 (ML1) installed for the OFGQS had 6 sampling points at depths ranging from 2.54 m to 8.94 m whereas multilevel monitoring wells 2, 3 and 4 (ML2, ML3, and ML4) each had 10 sampling points. Construction details are shown on Figures 6, 7, and 8. All well depths were recorded from ground surface to the bottom of the screen. The sample points in the multilevel monitoring wells were numbered consecutively from the shallowest to the deepest point.

Three pumping wells (labelled PW1, PW2, PW3 in Figures 6, 7, and 8) were installed and used to investigate various groundwater extraction scenarios. Two fully screened observation wells (labelled OW1 and OW2), installed beside the fully screened pumping well (PW1), were used in a pumping test to determine bulk aquifer hydraulic properties. Each "pumping well was located adjacent to a multilevel monitoring well (Figure 5).

Pumping well 1 (PW1) was a fully screened well with a screen length of 6.10 m and the pump intake was located at 3.05 m above the bottom of the well. The distance between PW1 and the closest multilevel monitoring well ML3 was 1.10 m (Figure 6).

Pumping well 2 (PW2) was a shallow partially screened well with a screen length of 1.50 m and the pump intake was located at 0.75 m above the bottom of the well. The distance between PW2 and the closest multilevel monitoring well ML2 was 1.70 m (Figure 7).
Pumping well 3 (PW3) was a deep partially screened well with a screen length of 1.50 m and the pump intake was located 0.75 m above the bottom of the well. The distance between PW3 and the closest multilevel monitoring well ML4 was 1.86 m (Figure 8).

Observation wells 1 and 2 (OW 1 and OW2) were fully screened wells with screen lengths of 6.10 m. The distance between PW1, OW1, and OW2 was 1.46 m and 3.98 m respectively (Figure 6).

The pumping and observation wells were constructed from 0.05 m ID Tri Lock flush threaded PVC riser pipe and #10 slot screen with O-ring rubber seals between sections. The screen was divided into 1.50 m sections, of which the centre 1.20 m was screened. The borehole was 0.115 m in diameter and a No. 2 N.J. industrial sand pack was installed around the screen to 0.30 m above the screen. The remaining 0.115 m borehole was backfilled with resident aquifer material to surface. All installations were made using 0.115 m hollow stem augers and no drilling fluids.

To assist in well design and confirm homogeneity noted at the installation of the multilevel monitoring wells, two continuous 9 m geologic cores were taken from ground surface at the study site using the drive point/piston sampler as described by Starr and Ingelton (1992). In this method, a piston sampler allows the collection of high-quality core samples from sand, silt, or clay up to depths of 18 m. The cores are collected in 51 mm OD aluminum tubing with a wall thickness of 1.3 mm and a length of 1.5 m. The samples were obtained without the use of drilling fluids,
rendering pristine geologic samples for physical and geochemical characterization of the subsurface.

Core 1 was obtained within 5 m of PW1 and core 2 was similarly within 5 m of PW2. The two cores were carefully split longitudinally using a circular saw to cut through the aluminum; half the core was visually inspected and logged for moisture content, colour, grain size and grading, and the other half was used for grain size distribution analysis and falling-head permeameter tests.

3.3 Aquifer Hydraulic Properties

The physical and hydraulic properties of the aquifer materials were derived through several different techniques including visual grain size distribution on representative core samples, falling-head permeameter tests, and a pumping test. In addition, the regional piezometric surface was monitored over a period of about 16 months to observe seasonal variations in lateral hydraulic gradients. Each of these methods are outlined in the following paragraphs.
3.3.1 Grain Size Distribution

To perform a grain size distribution analysis, three 100 g samples were removed from three visually representative sections of each of the 9 m cores. The samples were separately sifted through a series of eleven sieves ranging from US standard sieve mesh # <325 to 10 (0.0310-2.0000 mm) and the weight in each sieve determined. The weight in each sieve was then reported as a percentage of the total weight of the original sample. The three samples from each core were then averaged to give one grain size distribution for each of the continuous cores. The percent of the total weight of the original sample which passed through each sieve was then plotted against the grain size (US Standard Sieve Mesh #) and a distribution curve was generated. This data was then used to design the well screen and sand pack for the pumping well installations. The details and plots can be seen in Appendix B.

3.3.2 Falling-Head Permeameter Tests

Hydraulic conductivity profiles along the core were obtained by taking 5 cm long sub-samples from visually different depositional units within the two cores. A total of 15 samples from each core were isolated and tested (Figure 9 and 10). Each sample was prepared and tested using a falling head permeameter described by Freeze and Cherry (1979) according to the method reported by Sudicky (1986). The samples were oven-dried overnight, homogenized through a
sample splitter, then re-packed and sealed into a permeameter cell. The permeameter tests were then performed using de-aired water at 22°C.

3.3.3 **Pumping Test**

A bulk hydraulic conductivity of the aquifer was determined using analytical type-curve matching of time-drawdown measurements from observation wells OW 1 and OW2. The Neuman analytical solution for fully penetrating wells in an anisotropic unconfined aquifer was used (Neuman, 1975).

The pumping test was conducted with the fully penetrating well (PW1) and the pumping and discharge system was designed as follows. A 3.8 cm OD and 2.54 cm ID black polyethylene intake pipe with a plastic foot valve was placed at mid-screen in the well (3 m above the aquitard). An electric, 1 HP Gould Aqua Lawn centrifugal pump was attached to the intake pipe, and the flow rate was regulated by an adjustable valve attached at the outflow of the pump. The pumped water passed through the pump, then through the valve which regulated flow, past a sampling port, and finally through an in-line flow meter with a by-pass option. During initial pumping, this by-pass prevented fine sediment from entering the flow meter. The in-line flow meter was capable of measuring flow rates of up to 80 L/min. The pumped water exited the flow meter and was discharged 50 m down gradient into the tobacco field. The discharge pipe was the same type of tubing as the intake. The flow meter was initially calibrated and continuously checked by
measuring flow out of the discharge pipe using a 10 L bucket, a 1 L graduated cylinder, and a stop watch.

The pumping test was conducted for 30 hours and water levels were taken frequently for the first 2 hours. The pumping rate throughout the test remained constant at 78 L/min ± 0.1 L/min. Time-drawdown data and plot from the two observation wells are contained in Appendix B.

3.3.4 Monitoring the Regional Piezometric Surface

Water levels were measured in three multilevel monitoring wells (ML1, ML2, and ML3) using and electric water-level tape. Water level data were obtained monthly for the fall of 1994 and once in the spring of 1994 and 1995, to determine temporal variation in horizontal and vertical gradients during the study period. Water levels were measured prior to each geochemical sampling. Initially, the three multilevel monitoring wells listed were the only installations available for monitoring. Towards the latter part of the study period, additional wells were installed and used to further determine hydraulic gradient. The water levels were measured from ground surface to the water table, normalized to a datum obtained from a topographic survey discussed below, then contoured to determine the groundwater flow direction.
A topographic survey was performed with a laser theodolight. All notable objects and instrumentation were included. In addition, several ground surface points were included to provide a larger data set for a more accurate general elevation evaluation.

3.4 Geochemical and Isotopic Investigation

The occurrence and distribution of nitrate in the agricultural study area was investigated through several monitoring approaches. The geochemical study included regular monitoring of nitrate contamination in the groundwater at various locations throughout the site, analysis of stable nitrogen isotope $^{15}$N distributions in the nitrate mass, and the collection of supplementary chemical and isotopic data including sodium, dissolved oxygen, and $^3$H. A primary objective of this part of the study was to isolate the source of nitrate that occurs in the domestic well and to monitor the spatial and temporal variability of nitrate. The following describes the sampling protocol, storage, and chemical analysis of these samples.

3.4.1 Nitrate in the Groundwater on Site

Three multilevel monitoring wells were sampled during the study period specifically for regional nitrate mapping. The three multilevel monitoring wells sampled, ML1, ML2, and ML3 are shown in Figure 5. They were chosen for their distinctive location on the farm and proximity to a potential nitrate source. In addition, the domestic well was sampled during the OFGQS in the
winter of 1991 and summer of 1992 and continued to be sampled once in the summer of 1993 and at least once in each season of 1994. ML1 was similarly sampled and ML2 and ML3 were sampled in the summer of 1993 and at least once in each season of 1994.

The domestic well water was sampled from the kitchen tap which avoided any in-line water treatment. The water was allowed to run for 2 minutes before the tap was sampled. Groundwater samples were collected from the multilevel monitoring wells using a peristaltic pump at surface. A 0.5 cm ID Teflon® tube was inserted down the multilevel tube and the intake placed at mid screen (about 5 cm above the bottom of the well). Approximately 1 L of water was purged from each well prior to collecting samples in 20 ml polyethylene vials. With an assumed porosity of 0.33 and a standing volume in the well of 130 ml, then a 1 L purging volume meant samples were representative of groundwater within a 9 cm radius cylinder around the 10 cm long screens (Keely, 1982). The multilevel monitoring wells were sampled from the deepest point up to the shallowest point. The samples were then kept cool during transport and then stored at -2°C prior to analysis, which occurred within 2 weeks. The samples were not chemically preserved or filtered and field blanks and standards where submitted in conjunction with the samples. The field blanks were de-ionized water which passed through the same sampling system after each complete sampling round. The standards were prepared in the laboratory using NaNO₃ and deionized water and ranged in concentration from 0 to 30 mg/L NO₃-N. Following 12 months of storage at -2°C, selected standards and samples were re-submitted for analysis. This confirmed cold storage without any chemical preservatives, was adequate (± 0.5 mg/L NO₃-N) for sample integrity.
Chemical analyses for nitrate were performed in the Department of Land Resource Science at the University of Guelph, Guelph, Ontario. Nitrate analyses were conducted using a Technicon TrAAcs machine and Technicon Industrial Method No. 824-89T. In this method, nitrate was reduced to nitrite and the total amount of nitrate and nitrite in the original sample were measured colourmetrically (Tel and Heseltine, 1990).

3.4.2 Use of the Stable Nitrogen Isotope $^{15}N$

Chapter 2.0, Persistence and Fate of Nitrate in Groundwater, outlines the use of the stable nitrogen isotope $^{15}N$ to delineate a nitrate source. At this site the main objective in using this stable isotope was to isolate the source of nitrate contamination in the domestic well. A successful implementation of this technique would allow for recommendations for approaches to improve the drinking water quality on the farm. Samples were taken from the domestic well to determine the $\delta^{15}N_{NO_3}$ value of the drinking water under two different local flow conditions. In addition, samples were taken from two multilevel monitoring wells (ML1 and ML3), located near two isotopically distinct potential sources of nitrate. The samples obtained from ML1 were expected to reflect an inorganic fertilizer while ML3 might indicate a possible influence from the septic system. Due to its location, ML3 was also representative of the groundwater surrounding the domestic well (Figure 5).
Using the same sampling method outlined above, samples for $^{15}$N analysis were collected in the early spring and late summer of 1994 from the multilevel monitoring wells using a peristaltic pump at the surface. A 0.5 cm ID Teflon® tube was inserted down the multilevel tube and the intake placed at mid screen (about 5 cm above the bottom of the well). Approximately 1 L of water was purged from the well prior to collecting samples in 1 L amber glass bottles. Two litres were extracted from each point through an in-line 45 µm cellulose acetate filter in order to obtain at least 50 mg of NO$_3^-$ or 10 mg of N. The samples were then preserved using 2 ml saturated mercuric chloride (HgCl$_2$) per litre of sample and stored at 4ºC.

Chemical analyses for the stable nitrogen isotope $^{15}$N were performed in the Environmental Isotope Laboratory (EIL), Department of Earth Sciences, at the University of Waterloo, Waterloo, Ontario. In the method used, nitrate was extracted from the groundwater and converted to potassium nitrate using cation exchange columns of Bio Rad AG50W X4 resin (cat# 142-1331). The dry potassium nitrate samples were combusted in a breakseal at 850ºC using copper/copper oxide for the reduction of the nitrogen and calcium oxide for the removal of CO$_2$ and water by absorption (Kendall and Grim, 1990). The resultant nitrogen gas was then analyzed on a mass spectrometer and $^{15}$N / $^{14}$N ratios relative to the international standards atmospheric nitrogen was calculated. The error associated with the reported values was calculated to be ±0.2%.
3.4.3 Supplementary Geochemical Data

Supplementary inorganic indicators such as sodium, dissolved oxygen, and tritium can give insight to biogeochemical and physical processes. Specifically, sodium values can add supportive evidence to isotopic investigations which have determined nitrate contamination as partly or fully due to septic system effluent (Robertson et al., 1989; Robertson et al., 1991; Robertson and Blowes, 1995). Dissolved oxygen and tritium can lend a biogeochemical or physical explanation respectively, to vertical nitrate stratification in an aquifer. The role of dissolved oxygen in the biogeochemical process of denitrification is outlined in Chapter 2.0.

A portion of the sample volumes from the multilevel monitoring well by the house (ML3) collected for nitrate analysis in March and November of 1994, were submitted for sodium analyses at the Water Quality Laboratory, Department of Earth Sciences, University of Waterloo, Waterloo, Ontario. Sodium was selected for analysis because of its potential association with septic field effluent (Robertson et al., 1989). Furthermore, background levels of sodium in the groundwater at the site was obtained from the March 1994 nitrate samples obtained from the multilevel monitoring well beneath the cultivated field (ML1). Sodium was analyzed using atomic absorption spectrophotometry and a Varian Model #1475 Atomic Adsorption Spectrophotometer. The error for sodium analyses was reported as less than 5%.
Samples for measurement of dissolved oxygen were collected from the multilevel monitoring well adjacent to the house (ML3) on two occasions, November 1994 and July 1995, and analyzed on site using the Winkler titration method (American Public Health Association et al., 1981). An additional multilevel monitoring well (ML4) was sampled July, 1995, to confirm the lateral and vertical extent of dissolved oxygen concentrations. The samples were obtained in 50 ml ground glass syringes attached to the effluent tube on a peristaltic pump. Very slow pumping rates were used to minimize the entry of air bubbles into the syringes while sampling, to avoid atmospheric oxygen from coming in contact with the groundwater sample.

As mentioned above tritium can be used to determine waters from different flow paths. Two days after the irrigation pond had been drained following an irrigation event (August 1994) the recharge water was sampled for direct tritium analysis. Similarly, two multilevel monitoring wells (ML1 in the field and ML3 beside the house) were also sampled at this time. If the chemistry of shallower waters is different from deeper waters then the aquifer may be stratified with respect to different aged groundwater. Since direct tritium analysis can give a general indication of pre and post 1960's water it was used to determine if older waters underlay younger waters in the unconfined aquifer (Solomon, 1992). The samples were collected in 20 ml polyethylene vials and kept cool. The samples were submitted to the Environmental Isotope Laboratory (EIL), Department of Earth Sciences, at the University of Waterloo, Waterloo, Ontario. The method used liquid scintillation counting (LSC) for the detection and quantification of tritium. Tritium measurements have a lower
limit of detection of approximately 6 tritium units (TU) and values reported were ±8 TU for water counted directly. EIL used Canberra-Packard Pico-flour LLT (low level tritium) cocktail which had a high carrying capacity for water with high efficiency and low background characteristics. The laboratory standard is NBS-4361 tritium reference material diluted with background water which is then calibrated to NSB-4926C. The background water is from a well with a Radiocarbon activity older than 3500 years and a conductivity of less than 300 µmho (Drimmie et al., 1991).

3.5 Evaluation of Variable Screen Length and Depth of Well

The following methods were used to compare nitrate concentrations in the effluent from shorter screened wells placed at different depths versus a fully screened well. The influence of well design on total hydraulic head adjacent to the pumping well and nitrate transport during pumping was also investigated. Water requirements were determined for a household of six people on a small dairy farm with approximately one hundred cows. Three different well designs were installed at the site, a fully screened well (PW1), a shallow short screened well (PW2), and a deep short screened well (PW3). Each well was pumped using a schedule believed to be a typical water consumption pattern. Water levels and nitrate samples were measured and collected from each pumping well (PW1, PW2, and PW3) and adjacent multilevel monitoring wells (ML2, ML3, and ML4 respectively) during the pumping periods.
3.5.1  Installation and Design of Wells

Three wells were installed as described in the preceding paragraphs. All three wells were 0.05 m in diameter with a sand pack placed around the screen as described earlier. The wells were designed to draw groundwater from three different zones within the aquifer sequence in order to evaluate the influence of well depth on transient nitrate distribution.

The first well was a fully screened well that spanned the entire thickness of the aquifer. The second installation was a shallow, short-screened well (1.5 m) that was positioned near the water table in a zone with nitrate concentrations of 5-10 mg/L NO$_3$-N as observed in the multilevel well samples. The final installation was another short-screened well (1.5 m) positioned deeper in the sequence in a zone with nitrate concentrations of 0 to 3 mg/L NO$_3$-N. The location of the wells are shown in Figure 5 and their relative vertical positions are indicated on Figure 11. The nitrate profiles illustrated in Figure 11 were measured prior to any pumping in the adjacent well and were used as a benchmark to monitor vertical changes in nitrate concentrations during extraction.

The wells were developed until fine sediment no longer appeared in the effluent by pumping at a rate of 78 L/min and moving the pump intake up and down along the screen. This procedure was repeated twice and the aquifer was allowed to recover for a period of two days prior to the commencement of the detailed pumping sequences.
Each well design had a multilevel monitoring well installed nearby. The multilevels were used to measure changes in total hydraulic head with depth and nitrate distributions during pumping events. Details on multilevel monitoring well materials and construction were previously discussed. Every polyethylene tube was developed by pumping out 5-6 L of water with a peristaltic pump until no fine sediment was observed in the effluent.

3.5.2 A Typical Water Consumption Pattern

The pumping schedule and rate were designed from information obtained off site. Typical daily water demands for human consumption and use were obtained from the Regional Municipality of Waterloo, Water Services Division (RMOW Long Term Water Strategy Phase I Report, 1994). Typical daily water requirements were approximately 25,000 L per one hundred people (250 L per person) with peak demand periods in the morning and evening. An average daily water requirement for one hundred dairy cows was estimated at a small dairy farm in Waterloo-Wilmont Township, Ontario. The dairy operation consisted of approximately 120 Holstein cows in a moderate northern climate. The estimated daily water demand of one hundred cows on a moderately warm day is approximately 12,000 L (120 L per cow) with peak demands at milking times (morning and evening). From this information the following pumping schedule was designed.
For 100 head of cattle and six people on the farm, the total daily volume extracted was specified at 13680 L. The total daily water requirement was divided into 6 evenly spaced pumping events for a total of six hours, with peak extraction volumes placed at the beginning and end of the day. To extract the specified total daily volume a pumping rate of 38 L/min was required. The first three days had an intermittent pumping schedule that started in the morning with two hours of pumping and the pump was then turned off for one hour. The one hour recovery period was followed by four cycles of one half hour on and one hour off. After the last of the four cycles the pump was turned on for two hour and then turned off over night (13 hours). This pumping schedule was repeated for three days and then slightly altered on the fourth day to observe any differences longer pumping periods at the same pumping rate might have on nitrate transport. The pumping schedule for the fourth day was three cycles of two hours of pumping at a rate of 38 L/min followed by one hour of no pumping. The pumping schedule described above is listed in Table 1.

3.5.3 Measuring and Sampling During Pumping Events

The three different pumping scenarios were monitored in a similar fashion. Water levels were assessed before a pumping event began and at 30 minute intervals during the entire pumping sequence. Both the pumping well and each level of the multilevel monitoring well adjacent to the well being pumped were monitored. Measurements were made with an electric water level tape relative to ground surface. After the water level was determined, a nitrate sample was taken from
each tube in the multilevel monitoring well in accordance with the sampling protocol presented earlier. Nitrate samples of the well water were obtained from the sampling port on the pump. As with the water level measurements, nitrate samples were taken from the entire monitoring network every 30 minutes.
4.0 RESULTS AND DISCUSSION OF THE FIELD INVESTIGATION

On the basis of the field and lab results, the aquifer at the farm study site was determined to be a very permeable, relatively homogeneous, isotropic, sand to gravel aquifer with stratified nitrate contamination. The domestic well had concentrations of nitrate which exceeded or were close to the provincial drinking water objectives (10 mg/L NO$_3$-N) throughout the study period. As Figure 5 shows, the domestic well is located between an active and an inactive septic field and consequently could be vulnerable to human sources of nitrate. Furthermore, the tritium analyses showed post-1960 groundwater throughout the unconfined aquifer suggesting that groundwater in the aquifer had been recharged since the Norfolk sand plain was cultivated (Regional Ground-Water Quality, 1993). The drinking water well has therefore the potential to have been impacted by agricultural activities in the area. The source of nitrate in the domestic well is consequently highly dependent on the direction of the regional hydraulic gradient.

The hydraulic head data from multilevel monitoring wells (ML1 and ML3) indicated that the thickness of the unsaturated zone was 1.5 m in the field and approximately 2.0 m by the house. Fluctuations of ±0.9 m occurred during the two-year study period. The average horizontal hydraulic gradient was 0.003 m/m ranging from 0.0014 to 0.004 m/m. The groundwater flow direction varied from approximately 30 degrees east of north to 10 degrees west of south during the year, the latter
being the predominant flow direction (Figure 5). For example, two plots of the regional hydraulic head are contoured for two unique flow directions in November 1994 and April, 1995 (Figure 12).

The field investigation was designed with a threefold purpose: I) to determine the source and persistence of nitrate in the domestic well on a farm, ii) to evaluate the water quality produced from different well designs, and iii) to obtain physical aquifer properties and geochemical data for use in a numerical model. The first objective was achieved by temporally monitoring concentrations of nitrate, supplementary geochemical data, and groundwater flow directions to delineate the source of nitrate in the domestic well. The second objective was met by monitoring transient hydraulic head and nitrate concentrations in three different well designs and adjacent multilevel monitoring wells under realistic pumping scenarios. The third objective was realized through controlled pumping tests, evaluation of core samples, and through the use of data collected during the pumping scenario with the different well configurations.

4.1 Aquifer Hydraulic Properties

4.1.1 Core Samples

Core 1 and Core 2 were taken approximately 5 m away from the multilevel monitoring well beside the house (ML3) and the multilevel monitoring well beside the field (ML2) respectively
Visual inspection and testing of the geologic cores generally indicated an interbedded very fine to coarse grained sand sequence with the coarsest material observed between 3 and 8 m below ground surface (Figure 13, 14, and 15). Soil recovery ranged from 50-90% for the 1.5 m cored intervals, using the drive point/piston sampler. The poorest recovery occurred in the unsaturated zone (the top 3 m) for both cores.

Each depositional unit, below the water table where recovery was a maximum, ranged in thickness between 0.34-2.29 m for Core 1, and 0.30-1.69 m for Core 2. No direct correlation was noted between the cores however the major depositional units consisted of fine to medium grained brown sand, underlain by a sandy gravel which overlies a medium grained sand. The underlaying, less permeable material appeared to consist of fine grained sand in Core 1 and sandy silt in Core 2. This general stratigraphic structure is consistent with an abandoned beach or lake plain such as the Norfolk Sand Plain. Figures 13 and 14 illustrate the stratigraphy of the two continuous cores.

Hydraulic conductivity measurements were performed on samples taken from each depositional unit of the continuous cores noted in Figure 13 and 14. A total of 15 samples were taken from each continuous core at the locations marked on Figures 9 and 10. From these tests, the most permeable zone was found at 4-8 m below ground surface and had an arithmetic mean hydraulic conductivity of $3.65 \times 10^{-4}$ m/s with minimum hydraulic conductivities observed in the confining layer and unsaturated zone (Figure 15). For Core 1, taken beside the house, the minimum
hydraulic conductivity was measured to be 3.56 x 10^{-6} m/s at ground surface with similar values observed at the water table. For Core 2, taken beside the field, the minimum hydraulic conductivity was determined to be 6.33 x 10^{-5} m/s 0.53 m below ground surface with a similar measurement in the confining layer 8.30 m below ground surface. Maximum hydraulic conductivities were found to be in the most permeable zone as determined by both cores and were 7.51 x 10^{-4} m/s and 6.20 x 10^{-4} m/s for Core 1 and Core 2 respectively. Detailed information and calculations from the falling head permeameter tests can be found in Appendix B.

4.1.2 A Pumping Test

The bulk aquifer hydraulic properties based on the 30-hour pumping test are reported in Table 2. The average horizontal hydraulic conductivity was determined using the Neuman (1975) method to be 4.62 x 10^{-4} m/s. Using the data from both observation wells the average specific storage at early time was found to be 0.003 and at later time the average specific yield was calculated to be 0.22. The pumping test was performed and interpreted as described in the Methods and Materials section. Detailed time-drawdown data can be found in Appendix B.

The horizontal hydraulic conductivity obtained from the pumping test (4.62 x 10^{-4} m/s) and the arithmetic average from the falling head permeameter tests done on the core samples (3.65 x 10^{-4} m/s), agree well. It should be noted that only the samples which coincided with the main
hydraulically conductive unit where used in the arithmetic average from the falling head permeameter tests whereas the pumping test conducted in a fully-screened well incorporates the entire saturated zone. Slug tests were attempted at this site, however, hydraulic head recovery was too fast to accurately obtain water level measurements manually.

The pumping test interpretation rendered an anisotropy ratio ($K_T/K_V$) of 0.10. Furthermore, the falling head permeameter test used the arithmetic and harmonic means of the main conducting unit to calculate an anisotropy ratio. As stated above the arithmetic mean hydraulic conductivity was $3.65 \times 10^{-4}$ m/s and the harmonic mean hydraulic conductivity was calculated to be $1.2 \times 10^{-4}$ m/s. Taking the arithmetic mean and the harmonic mean values to be equivalent to the horizontal ($K_r$) and vertical ($K_v$) hydraulic conductivities, as described by Conant, (1991) and Conant et al., (1995), the anisotropy was calculated to be $3.04 (K_T/K_V)$. This anisotropy is higher than the value obtained by the Neuman (1975) type-curve matching. An anisotropy ratio of 0.10 (ie. $K_V$ 10 times greater than $K_h$ or $K_T$), as obtained from the pumping test, is highly unusual given commonly accepted depositional patterns. Since the Neuman (1975) method is very sensitive and remixing of the permeameter samples destroys some important structure it was difficult to determine an anisotropy ratio.
4.2 Geochemical and Isotopic Investigation

A common observation for nitrate contamination in the rural environment is that higher concentrations are generally restricted to the upper portion of the saturated zone in an unconfined aquifer yielding a contaminant-stratified aquifer (Trudell et al., 1986; Geyer et al., 1992; Starr and Gillham, 1993; Ryan, 1994). At the current study site, concentration profiles at three multilevel monitoring wells (ML1, ML2, and ML3) indicated significant nitrate (5-35 mg/L NO\textsubscript{3}-N) which persisted to approximately 5 m below the water table and much lower nitrate (3-0 mg/L NO\textsubscript{3}-N) at depths greater than 5 m below the water table. Figure 5 shows the locations of the three multilevel monitoring wells ML1 (in the field), ML2 (beside the field), and ML3 (beside the house). As described in previous sections, there are several potential sources of nitrate contamination. The locations for the multilevel monitoring wells were chosen on the basis of proximity to potential nitrate sources ie; the household septic systems and non-point sources such as fertilizers on cultivated fields.

The depth at which the maximum concentration occurred differed between each multilevel monitoring well. With the exception of the shallowest point in August 1993, February 1994, and August 1994, the multilevel monitoring well beside the house (ML3) had steadily declining concentrations of nitrate below the water table (Figure 16). However, in the multilevel monitoring well beneath the field (ML1), groundwater just below the water table was found to have lower nitrate concentrations than groundwater at greater depths (3 to 6 m below ground surface) (Figure
The multilevel monitoring well beside the field showed a blend of these two stratification patterns (Figure 18). The maximum nitrate concentration found at any one point in the multilevel monitoring wells beneath the field (ML1) and beside the field were temporally variable ranging from 12-4 mg/L NO₃-N. The multilevel monitoring well beside the house consistently showed higher nitrate concentrations than observed elsewhere (15-45 mg/L NO₃-N).

The vertical nitrate profiles beneath the field (ML1) showed that during periods of heavy recharge (Spring and Fall) the high nitrate zone mentioned above (3 to 6 m below ground surface) broadened. However, variations in surface infiltration does not appear to significantly affect the profile just below the water table or in deeper portions of the aquifer (Figure 17). This suggests that heavy recharge events, including irrigation and spring snow melt, can affect the thickness of the highly contaminated zone while still maintaining a relatively uncontaminated deeper portion of the aquifer.

Throughout a significant portion of the study period (80%), the nitrate concentration in the domestic well exceeded the provincial drinking water objectives of 10 mg/L NO₃-N (Figure 19). The nitrate concentration ranged from 9 to 22 mg/L NO₃-N and averaged 14 mg/L NO₃-N with a standard deviation of 4 mg/L NO₃-N.

The domestic well is a sandpoint and unlike a fully screened well its intake is only over a very short distance (3.8-4.0 m). In a contaminant-stratified aquifer placement of the sand point well
could potentially influence water quality. The average concentration of nitrate found in the nearby multilevel monitoring well (ML3) at that depth were similar to nitrate concentrations found in the domestic well (±3 mg/L NO₃-N). The multilevel monitoring well was therefore considered representative of a vertical nitrate profile influenced by domestic water production. The nitrate concentrations used in the above comparison did not include measurements made after commencement of the pumping test (December 1994), since the nitrate distribution around ML3 was significantly altered during pumping.

The approaches used to delineate the source of nitrate in the water well included the evaluation of the stable nitrogen isotope $^{15}$N signature in conjunction with the concentrations of nitrate, and sodium, considering the variable groundwater flow direction. Figure 5 illustrates the location of the drinking water well in the house relative to the potential sources of nitrate.

The results from the $^{15}$N analyses are illustrated in Figure 20. The $\delta^{15}$N$_{NO_3}$ value of +10.3 % for the domestic well water in March 1994 was indicative of human and animal waste or denitrification. In August 1994, the $\delta^{15}$N$_{NO_3}$ value in the well of +5.1 % was indicative of inorganic fertilizer which had undergone volatilization during application, or soil organic nitrogen (Aravena et al, 1993).

Beneath the field (ML1) (Figure 20) in March 1994 the $\delta^{15}$N$_{NO_3}$ value of +3.8 % at a depth of 4.37 m was indicative of inorganic fertilizer which may have been enriched in $^{15}$N due to
volatilization during application, soil organic nitrogen, or some denitrification (Aravena et al., 1993; Gormly and Spalding, 1979; Smith et al, 1991; Chappelle, 1993). The same multilevel monitoring well in August 1994 showed a substantial increase in δ\(^{15}\)N\(_{\text{NO}_3}\) between the bottom two levels (+3.1 % at 4.37 m depth versus +12.5 % at 5.89 m depth).

The multilevel monitoring well beside the house (ML3) had a δ\(^{15}\)N\(_{\text{NO}_3}\) value in March 1994 of +6.2 % at 2.8 m below ground surface. This δ\(^{15}\)N\(_{\text{NO}_3}\) value suggests soil organic N but could also represent a value from mixing two different sources such as human or animal waste and inorganic fertilizer. In August 1994 samples from ML3 rendered increasing δ\(^{15}\)N\(_{\text{NO}_3}\) values with depth ranging from +3.7 % at 2.8 m below ground surface to +13.3 % at 5.79 m below ground surface. Although the multilevel monitoring well beside the house (ML3) was considered representative of the domestic well with respect to nitrate distributions, this may not be the case with respect to \(^{15}\)N distribution within the nitrate mass. Slight differences in location between ML3 and the domestic well could introduce discrepancies between the nitrogen isotope values due to changes in the local groundwater flow direction and subsequently the nitrate source. In March 1994 the discrepancy in the isotope values between the two wells at the same depth (+10.3 in the domestic well versus +6.2 in the multilevel monitoring well), could arise due to dissimilar mixing of background waters containing inorganic nitrate and narrow septic plume waters containing anthropogenic nitrate.

In order to accurately assess the isotope results concurrent with nitrate concentration profiles from the wells sampled for \(^{15}\)N in nitrate need to be reviewed. These can be seen in Figure 20.
Nitrate concentrations in the domestic well were 14 and 13 mg/L NO$_3$-N for March and August of 1994 respectively. In March 1994 the multilevel monitoring well beneath the field (ML1) had a vertical profile with a maximum nitrate concentration of 3.79 mg/L NO$_3$-N at a depth of 4.37 m below which the concentrations of nitrate diminished with depth. The same well (MU) in August 1994 showed a maximum nitrate concentration of 7.31 mg/L NO$_3$-N at the same depth of 4.37 m below ground surface which again diminished with depth. The multilevel monitoring well beside the house (ML3) had a maximum nitrate concentration of 12.7 mg/L NO$_3$-N for March of 1994 and 20.35 mg/L NO$_3$-N for August of 1994. In March of 1994 the maximum concentration occurred at 3.78 m below ground surface and in August of the same year the maximum occurred at 2.80 m below ground surface. It should be noted that the water table near the house was higher by 0.24 m in March, 1994 than in August, 1994. For the same time period the water table beneath the field also dropped, however by 0.64 m. The high nitrate concentrations in the multilevel monitoring well beside the house (ML3), as compared to the multilevel monitoring well in the field (ML1), is most likely a septic system influence. The simultaneous decrease of nitrate concentration with depth (to approximately 0 mg/L NO$_3$-N) and increased $\delta^{15}$N$_{NO_3}$ to approximately +12% would suggest that denitrification plays a major role at depth at this site. However, subsequent measurements showed the presence of detectable dissolved oxygen (>2 mg/L) throughout the depth of the aquifer at two different locations.

Figure 21 compares nitrate profiles with dissolved oxygen (DO) profiles in two multilevel monitoring wells, both in close proximity to the house (ML3 and ML4). The DO profiles measured
in November 1994 and July 1995 showed aerobic conditions throughout the depth of the aquifer. The minimum dissolved oxygen concentration was 1.8 mg/L and 1.6 mg/L for ML3 and ML4 respectively. Aerobic conditions (>3 mg/L DO) persisted to approximately 4 m in ML3 and 6 m in ML4. However, greater than 1 mg/L NO$_3$-N persisted to a depth of 8 m below ground surface in both multilevel monitoring wells. As presented earlier, anaerobic conditions is a key requirement for denitrification to occur. Denitrification is therefore not an active process at these sampling locations however it can not be eliminated as a process up gradient.

The low nitrate values combined with elevated $^{15}$N values at depth suggests denitrification. The possibility of two different flow systems being present in the aquifer was assessed with the assistance of tritium ($^3$H) analyses. If tritium values indicate younger waters overlying older waters the nitrate and $^{15}$N isotope stratification could be due to the existence of two distinct flow systems. The tritium values, shown in Figure 22 (7-25 TU), were all indicative of relatively modern post-1960's recharge water throughout the aquifer. Thus the absence of different aged waters suggests that there may be only one flow system present in which nitrate stratification has evolved.

The tritium and dissolved oxygen data would suggest that the nitrate stratification which exists at this location is neither due to differing flow systems nor due to denitrification locally. In light of the conceptual models presented earlier in Figures 1 and 2, a plausible explanation is nitrate originating off-site or at some lateral distance away from the multilevel monitoring well and denitrification occurring along the flow-path prior to the sampling location. As Figure 1 also
suggests, recharge water with low nitrate concentrations could also have mixed up-gradient and produced the measured nitrate stratification.

In order to more accurately define the source of nitrate contamination within the study site, sodium (\(\text{Na}^+\)) concentrations in the groundwater were also evaluated. Sodium at enriched levels (20-100 mg/L) is commonly encountered in septic system plumes (Robertson et al., 1989; Robertson and Cherry, 1992).

Figure 23 compares nitrate and sodium profiles from (a) the multilevel monitoring well beside the house (ML3) with (b) the multilevel monitoring well beneath the field (ML1). In March 1994 the maximum concentration of sodium in the multilevel monitoring well beside the house (ML3) was 32 mg/L as compared to the maximum concentration of 8.6 mg/L beneath the field (ML1). The multilevel monitoring well in the field (ML1) was assumed to represent background levels of sodium concentrations in the aquifer since it is not located near any potential septic system source. The rest of the samples from both multilevel monitoring wells were found to have less than 10 mg/L sodium. The maximum concentrations of sodium were found at a shallower depth than the maximum for nitrate concentrations in both wells. The elevated sodium concentrations in the multilevel monitoring well beside the house (ML3) support the isotopic interpretation that septic system nitrate does impact the domestic well in March 1994.
The position of the domestic well between two septic fields made the task of delineating the source of nitrate particularly sensitive to groundwater flow directions. During two spring melts (March 1994, April 1995) and following several irrigation events in August 1993, local groundwater flow directions shifted away from the regional groundwater flow direction, 10º west of south. This in turn changed the potential source of nitrate in the domestic well from regional agricultural activities to possible capture of septic field effluent. The domestic well and the septic field are separated by approximately 8 m. Using an assumed porosity of 0.33, the average groundwater flow velocity is 0.30 m/day. This suggests that a shift in groundwater flow direction for 27 days could potentially cause the septic plume to influence the domestic well.

In summary, the interpretation of the nitrogen isotope and sodium results, together with consideration of the potential transient nature of the groundwater flow field can assist in delineating the origin of nitrate in the domestic well. In March 1994 the domestic well captured waters containing nitrate with $^{15}$N values typical of human waste (Chapelle, 1993). The most likely up gradient source for human waste under the given groundwater flow conditions of March 1994 was the active septic system. Furthermore, comparatively high sodium values (32 mg/L) are indicative of septic effluent. The maximum sodium concentration beside the house (32 mg/L) was significantly higher than the maximum concentration of sodium in the field (7 mg/L). The results of the sodium analyses therefore supports the findings from the nitrogen isotope analyses for March 1994.
Similarly, considering nitrogen isotope values and the groundwater flow direction in August 1994, inorganic fertilizer from agricultural activities appears to be the source of nitrate in the domestic well. Aravena et al. (1993) showed that when the ammonium (NH$_4^+$) in inorganic fertilizers volatilizes, the $^{15}$N value can be significantly enriched, by up to 6%. Soil organic nitrogen could produce a similar isotopic signature (approximately +6%). However, the nitrate in the groundwater would not likely reach such high concentrations solely from soil organic nitrogen.

The depletion of nitrate with depth in concert with the increased isotopic values with depth in both multilevel monitoring wells is consistent with $^{15}$N enrichment due to denitrification (Smith et al, 1991; Chapelle, 1993). Gormly and Spalding (1979) showed that there is an inverse correlation between $\delta^{15}$N$_{NO3}$ values and nitrate concentrations in groundwater as also occurs here (Figure 20). However, the persistence of oxygen with depth would suggest that denitrification was not a significant process beneath this farm (Starr and Gillham, 1993; Gillham and Cherry, 1978). Although, denitrifying conditions do not appear to be present at this site, favourable conditions for denitrification may have existed up gradient.

4.3 Evaluation of Variable Screen Length and Depth of Well

Field evaluation of the well construction geometries presented in Figure 11 will be presented next including transient drawdown behaviour and nitrate concentrations. Firstly, the results are
presented for the pumping well followed by the results for the surrounding aquifer as measured in the adjacent multilevel monitoring wells are presented.

A review of the pumping schedule presented in Table 1 is required to more easily interpret the presentation of the results. The pumping schedule was designed to closely represent the water usage pattern on a dairy farm with 100 cows and six residents. Peak demand periods were mimicked in the morning and evening with shorter intermittent pumping throughout the day. A total of six pumping events were performed on days 1, 2, and 3 but only three pumping events on day 4. All four days had a total pumping time of six hours at a pumping rate of 38 L/min which extracted the same total daily volume. The hydraulic and nitrate distribution measurements are presented separately for each well design.

The temporal variations in drawdown in the pumping well and nitrate concentration in the effluent for each pumping well design are illustrated in Figure 24. The upper plots on each graph show the maximum drawdown measured during each pumping event over four days. The lower plot on the same graph shows the nitrate concentration as monitored in the pumping well effluent during pumping events. The intermittent pumping schedule is visible in the graphs since drawdown oscillates between maximum and zero. Simultaneous transient nitrate concentrations are plotted for each well design.
The well design which produced the largest drawdown (0.64 m) in the pumping well was the shallow short screened well. Conversely the smallest drawdown (0.16 m) was measured in the fully screened pumping well. The maximum drawdown in the deep short screened well was 0.44 m.

The nitrate concentration in the effluent samples from the pumping wells oscillated during the pumping schedule for each case (Figure 24). The greatest fluctuation in nitrate concentration occurred in the fully screened well and the least variation was found in the deep short screened well. In the shallow short screened well, the magnitude of this oscillation decreased over time. A discussion on possible reasons for the variability differences will follow the presentation of the results from the adjacent multilevel monitoring wells.

The nitrate concentration in the deep short screened well asymptotically increased for 35 hours then appeared to approach a reasonably steady state concentration whereas scatter in the data makes this observation difficult for the other two well designs. The nitrate concentrations in the effluent from the deep short screened well were lower than the fully screened well or shallow short screened well. After four days of pumping the nitrate concentrations appeared to be approximately 9 mg/L, 6 mg/L, and 4 mg/L NO₃-N for the fully screened well, shallow short screened well, and the deep short screened well, respectively.

The observations related to the hydraulic response of the surrounding aquifer and vertical distributions of nitrate concentrations in the adjacent multilevel monitoring wells during each
pumping scenario are illustrated separately. Figure 25, 26, and 27 show the vertical gradient profiles for each pumping event and Figure 28, 29, and 30 show the vertical evolution of nitrate concentration in the multilevel monitoring wells during the pumping schedule for each well design.

Hydraulically the aquifer responded very consistently during each pumping event for all well designs. This resulted in several drawdown measurements co-plotting. In Figures 25, 26, and 27 the initial water level conditions are marked as a set of vertically connected points. The four plots in each figure present the data from each day of the four day investigation.

The resultant vertical gradient profile was unique to each well design (Figures 25, 26 and 27). The largest drawdown measured in each of the multilevel monitoring well corresponded to the depth of the pump intake. The largest drawdown for all well designs was measured after the final pumping event of each day. The measured drawdown was very reproducible between pumping events for all four days and all three well designs. The water level throughout the aquifer recovered within 0.5 centimetres, 10 minutes after the pump was turned off. The fully screened well design (PW1) had small vertical gradients develop across the entire thickness of the aquifer, 1.1 m away from the pumping well. Pumping the shallow short screened well (PW2) produced no measurable vertical gradient in the upper 1 metre of the aquifer, 1.70 m away from the pumping well. Similarly, pumping the deep short screened well (PW3) produced no measurable vertical gradient in the upper 2 m of the saturated thickness of the aquifer 1.86 m away from the pumping well. The maximum
difference in total hydraulic head between multilevel monitoring points was approximately 2.5 cm for the fully screened well (PW1), and 3.5 cm for both short screened wells (PW2 and PW3).

The evolution of vertical nitrate distribution at the multilevel monitoring wells during the pumping schedule are illustrated in Figures 28, 29, and 30. Essentially no change in nitrate concentration was seen at the second deepest sampling point for all three well designs. However, the deepest point at all three multilevel monitoring wells consistently showed increasing nitrate concentrations with time. These elevated concentrations at the deepest point declined overnight to near zero concentrations. Re-analysis of these samples confirmed the original concentration (±1 mg/L NO₃-N) and field blanks eliminated the possibility of cross contamination through sampling. Furthermore, sodium analysis on the same samples for ML3 mimicked the same trend suggesting the possibility of interception or drawing up of a deeper septic plume.

The multilevel monitoring well located adjacent to the fully screened well demonstrated the highest nitrate variability between sampling times with a maximum range at any one point of 21 mg/L NO₃-N (day 1 at 2.80 m depth below ground surface). The same observation for the shallow (PW2) and deep (PW3) short screened wells were 9 and 7 mg/L NO₃-N, respectively. For PW2 the maximum occurred on day 1 at 2 m below ground surface (0.05 m below the water table) and for PW3 the maximum occurred day 3 at 3 m below ground surface (1.1 m below the water table). Excluding the deepest sampling point, the majority of fluctuation in the nitrate concentration was confined between the water table and just above the well screen of the shallow short screened
pumping well. Similar observations were made beside the deep short screened well where the majority of fluctuation in the nitrate concentration occurred between the water table and just above the well screen.

Although the initial stratification of nitrate was similar for each well design, the vertical nitrate profile by the fully screened well showed a large change after the initial commencement of pumping. The other two well designs demonstrated some vertical transport of nitrate, however the change in maximum nitrate concentration was measurably less after pumping had been initiated. Two plausible reasons for this difference between well designs are as follows: (i) The vertical gradients near the water table induced by a fully screened pumping well could more readily capture nitrate mass contained in the capillary fringe or just below the water table. And (ii) The lateral location of the fully screened well risks interception of a septic plume.

As described previously the fourth day pumping schedule for each well design was slightly adjusted to three prolonged pumping events while still only producing the same total daily volume at the same pumping rate as days 1, 2, and 3 (Table 1). No measurable difference could be determined for days 1, 2, and 3 versus day 4 pumping schedules in hydraulic and nitrate measurements. The absence of a measurable difference suggests that frequency and duration of pumping while extracting the same total daily volume has no measurable effect on the capture of nitrate by the pumping well given the specific aquifer properties at this site.
The scatter in effluent nitrate concentrations of the fully screened and the shallow short screened wells versus the more consistent effluent concentrations measured in the deep short screened well is consistent with the findings reported by Ryan (1994). Ryan (1994) described 'microplumes' with high concentrations of nitrate which moved horizontally with the regional hydraulic gradient. In their study, the mid-portion of the unconfined aquifer, approximately 1 to 2 metres below the water table, appeared to exhibit greater temporal variability in nitrate concentrations than the bottom 2 metres of the aquifer. If there is a significant amount of temporal variability at the mid-aquifer depth in a horizontal flow system one would also expect a significant amount of spatial variability in nitrate concentrations at the mid-aquifer depth. The conceptual model presented by Hallberg and Keeney (Regional GroundWater Quality, 1993) and the conceptual model presented by Ryan (1994) are illustrated in Figures 1 and 2.

Assuming strictly radial flow, a porosity of 0.33, and a total daily extraction volume of 13,680 L the radius of capture from a 6 m screen would be approximately 0.5 m/day. The same calculation for a 1.5 m screen yields a 1.7 m/day radius of capture. In the same aquifer, over four days of equivalent extraction volumes the capture radius of a 6 m screen and a 1.5 m screen would be 1.9 and 6.8 m respectively. Initially when pumping a well it is reasonable to assume the major portion of the effluent is derived from horizontal flow from the aquifer surrounding the well screen. It would therefore be reasonable to expect the variability in nitrate distribution in the aquifer adjacent to the well screen to be reflected in the concentration of nitrate in the effluent. Furthermore, studies by Kaleris (1989) and Barczeweslcı and Marschall (1989) indicated that increased flux could
occur near the pump intake. In the fully screened well and the shallow short screened well the pump intake was located at mid aquifer (4.5 m and 6 m below ground surface) where the majority of nitrate mass and variability was measured (Figures 16, 17, and 18). Superimposed on the variability of nitrate distribution within the aquifer is the effect of intermittent pumping which might induce vertical mixing. The deep short screened well initially drew water from the less contaminated portion of the aquifer. In the less contaminated portion of the aquifer the nitrate concentration was not only lower but the nitrate distribution was more homogeneous. This in turn produced less variability in the nitrate concentration of the effluent.

The length and vertical placement of the well screen produced distinct hydraulic head profiles in the adjacent multilevel monitoring well. This can become an important consideration in an aquifer where solutes are vertically stratified since solute transport is dictated by aquifer hydraulics. Furthermore, the multilevel monitoring wells adjacent to the two short screened wells measured essentially horizontal flow for half a metre below the water table for the shallow short screened well and for one metre below the water table for the deep short screened well. Vertical gradients between points were measured in the multilevel below these depths for both well designs. The nitrate concentrations measured in the multilevel monitoring wells during the pumping events showed a large amount of nitrate variability in the upper 4 m of the aquifer. This suggests that vertical transport of shallow waters with high nitrate concentrations occurred where vertical gradients were measured between monitoring points. The second deepest point in each multilevel monitoring well showed little to no evidence of vertical nitrate transport. This observation is
consistent with the lack of variability in nitrate concentration in the effluent from the deep short screened well. Again, this was probably due to the low nitrate concentration and relatively homogeneous nitrate distribution at this depth. However, the deepest point showed notable fluctuations in nitrate concentration during the day's pumping schedule. Cross contamination and erroneous lab analysis were eliminated as plausible explanations. With the support of sodium analysis of the samples from the multilevel monitoring well beside the house (ML3) (Figure 31), one possible interpretation could be the possible capture of a deep septic plume. However, this evidence alone is inadequate to make this conclusion.

Pumping from different depths clearly influences mixing in a chemically stratified aquifer as shown in the field results. Furthermore, the evaluation of the three well designs demonstrated that pumping with a short-screened well from deep in a contaminant-stratified unconfined aquifer can improve the drinking water quality. These results agree with field results from a similar investigation by Gillham et al. (Ontario Ministry of the Environment, 1983) but differ from the findings of Conant et al. (1995). Conant et al. (1995) reported that pumping-well screen placement in an unconfined aquifer with vertically stratified contamination initially effected the water quality in the effluent but after three days of continual pumping both well designs had asymptotically approached a vertically averaged concentration of the aquifer. Several differences between the investigation by Conant et al. (1995) and the current study may have lead to dissimilar observations. The dissimilar results are likely due to different aquifer materials, to the nature of the contaminant stratification, and the
pulsing nature of the pumping schedule in the current study. Table 3 offers a comparison between the current study and the study conducted by Conant et al. (1995). These differences may all contribute to less mixing of vertically stratified contaminants. If the duration of the current test was extended, creating a greater total extraction volume, the vertically averaged concentration of the aquifer may have been reached by all three well designs. However, after 15 hours of pumping all three well designs appeared to have reached a reasonable steady-state pattern and the deep short screened well showed superior water quality as compared to the fully screened well and the shallow short screened well.
5.0 NUMERICAL MODELLING

5.1 Introduction

The purposes of the numerical simulations were threefold: i) to provide insight into the flow and transport processes associated with the various well geometries and pumping scenarios investigated during the field program, ii) to evaluate the sensitivity of the different parameters which most influence nitrate capture in the domestic well, and iii) to investigate alternative methods of groundwater extraction designed to improve drinking water quality on the farm. A three-dimensional numerical model was selected for the simulations in order to incorporate the radial response of the pumping wells. The model was constructed to closely represent the study site based on the field observations. A series of comparisons were then performed numerically to investigate the influence of various physical parameters on the steady state nitrate concentration in the effluent of the domestic well under different pumping scenarios. The numerical simulations included: the sensitivity to nitrate concentration in the recharge waters originating at surface around the well, the effect that total amount of surface recharge has on nitrate distribution in the aquifer, intermittent versus continual pumping for extracting the same total daily volume of groundwater, isotropy versus anisotropy, high versus low regional gradients, and the total daily volume extracted by a single well. Finally, a multiple well design was simulated as one way to minimize nitrate in the domestic water supply. The numerical investigation generally sought to be conservative and as such the aquifer was assumed to have no potential for denitrification. Furthermore, the well design chosen for all
comparisons represented the best design, as determined from the field investigation, to minimize nitrate in the effluent of the domestic well.

5.2 Theory

The numerical simulations were made with FRAC3DVS - a three-dimensional, finite-element, variably-saturated flow and transport model with one-dimensional line elements superimposed on the three-dimensional grid to represent pumping wells. Details describing this model were taken from Therrien and Sudicky (1994). The variably-saturated flow equation was solved using the control volume finite element method (Forsyth, 1991; Kropinski, 1990) and a standard time-marching finite element approach was used to solve the solute transport equation. The model assumed the fluid was essentially incompressible, the porous media was non-deformable, the system was under isothermal conditions, and the air phase was infinitely mobile. The effect of hysteresis was not included in the model and this was determined to be a reasonable exclusion. The control volume finite element approach makes use of the Newton-Raphson linearization technique (Forsyth and Simpson, 1991) and allows the saturation term to be exactly represented therefore eliminating mass balance concerns.

5.2.1 Variably-Saturated Flow

FRAC3DVS uses a modified form of the Richards' equation to describe three-dimensional transient groundwater flow in a variably-saturated porous medium. The equation has the following
general form in three dimensions, using an indicial notation and the summation convention (Cooley, 1983; Huyakorn et al., 1984):

$$ \frac{\partial}{\partial x_i} \left( K_{ij, \text{sr}} \frac{\partial (\psi + z)}{\partial x_j} \right) = Q - \frac{\partial}{\partial t} \left( \theta s \right), \quad i, j = 1, 2, 3 \quad (1) $$

where $K_{ij, \text{sr}}$ is the saturated hydraulic conductivity tensor, $k_{rw} = k_{rw} (S_w)$ represents the relative permeability of the medium with respect to the degree of water saturation $S_w$, $\psi = \psi(x_i, t)$ is the pressure head, $z$ is the elevation head and $\theta_s$ is the porosity, which is equal to the saturated water content. The water saturation is related to the water content $\theta$ according to:

$$ S_w = \frac{\theta}{\theta_s} \quad (2) $$

The effect of any source or sink on the flow in the porous media such as pumping can be represented in (1) by $Q$. Since equation (1) is non-linear a function needs to be defined for the primary variables $\psi$ and $S_w$. FRAC3DVS solves equation (1) in terms of $\psi$ where $S_w = S_w(\psi)$. The relative permeability is also assumed to be a function of $\psi$ or $S_w$. In FRAC3DVS the functional relationship by van Genuchten (1980) was used. This function is empirically determined based on earlier work by Mualem (1976). The van Genuchten (1980) saturation-pressure relation is expressed by:
and the relative permeability is determined using:

\[ k_{rw} = S_e^{1/2} \left[ 1 - (1 - S_e^{1/m})^m \right]^2 \]  

(4)

where \( \alpha, m, \) and \( n \) are parameters obtained from a fit of (3) and (4) to experimental results, \( P_c \) is the capillary pressure \( (P_c = -\psi) \) and \( S_e = (S_w - S_{wr})/(1 - S_{wr}) \), \( S_{wr} \) being the residual saturation. The storage term in equation (1) is expanded in a similar way to Cooley (1971) and Neuman (1973) and expressed as:

\[ \frac{\partial}{\partial t} \theta S_w - S_w S_e \frac{\partial \psi}{\partial t} + \theta \frac{\partial S_w}{\partial t} \]  

(5)

5.2.2 Solute Transport

FRAC3DVS solves for three-dimensional transport in a variably-saturated porous media as described by Bear (1972) and expressed in the following equation:

\[ \theta S_w R \frac{\partial c}{\partial t} + q_i \frac{\partial c}{\partial x_i} - \frac{\partial}{\partial x_i} \left( \theta S_w D_{ij} \frac{\partial c}{\partial x_j} \right) + \theta S_w R \lambda c = 0 \quad i, j = 1, 2, 3 \]  

(6)

where \( c = c(x, t) \) is the solute concentration, \( D_{ij} \) is the hydrodynamic dispersion coefficient and \( \lambda \) is a first-order decay constant. Equation (10) also assumes fluid incompressibility.

The retardation factor, \( R \), as described by Freeze and Cherry (1979) is:
\[ R = 1 + \frac{\rho_p - K_d}{\theta_s S_w} \]  \hspace{1cm} (7)

where \( \rho_p \) is the bulk density of the porous medium and \( K_d \) is the equilibrium distribution coefficient describing a linear Freundlich adsorption isotherm. It should be noted that for variably-saturated conditions, the water saturation appears in the definition of \( R \).

Darcy’s equation yields the fluid flux for equation (6):

\[ q_i = -K_y k_{rw} \frac{\partial(\psi + z)}{\partial \xi_i} \]  \hspace{1cm} (8)

The hydrodynamic dispersion coefficient \( D_{ij} \) is given by (Bear, 1972):

\[ \theta_s S_w D_{ij} = (\alpha_l - \alpha_t) \frac{|q| q_i}{|q|} + \alpha_l q_i \delta_{ij} + \theta_s S_w \tau D_d \delta_{ij} \]  \hspace{1cm} (9)

where \( \alpha_l \) and \( \alpha_t \) are the longitudinal and transverse dispersivities, respectively, \( |q| \) is the magnitude of the Darcy flux, \( \tau \) is the porous medium tortuosity, \( D_d \) is the free solution diffusion coefficient and \( \delta_{ij} \) is the Kronecker delta. The effective diffusion coefficient in the porous medium is simply the product of \( \tau \) and \( D_d \).

The boundary conditions needed to define equation (1) can be first or second type as defined by Huyakorn and Pinder (1983). Similarly the boundary conditions required to solve equation (6) can be first, second, or third type (Huyakorn and Pinder, 1983).
5.3 Description of the Model Domain

The domain used for all simulations was 40 m long, 30 m wide, and 9 m deep and oriented with the long axis parallel to the predominant regional flow direction. The pumping well was positioned 15 m from the upstream boundary along the centre line of the domain. The depth and screen length of the pumping well was variable depending on the scenario under investigation.

The assigned physical properties and boundary conditions of the domain are illustrated in Figure 32. The regional gradient of 0.003 equivalent to an average annual value observed on site was imposed across the domain with first type boundary conditions of constant hydraulic head \( H = 7.00 \) m and \( H = 6.88 \) m at the inflow boundary and outflow boundary, respectively. The bottom surface and lateral boundaries were defined as a no-flow boundary and the upper surface, or ground surface, had a constant recharge of 0.39 m per year with a concentration of 15 mg/L NO\(_3\)-N. This recharge rate and concentration was determined from estimated recharge rates in the Region of Waterloo and reasonable nitrate concentrations from shallow groundwater at the farm study site. Bulk recharge rates for the Region of Waterloo have been estimated to be between 0.15 m and 0.25 m per year (Martin, 1994). Glacial tills, such as those found in the Region of Waterloo, are less permeable than fine to medium sand found at the farm site (Freeze and Cherry, 1979; Martin, 1994). It is assumed that increased permeability allows for an increased recharge rate. Furthermore, rural areas generally allow more recharge to the aquifer since more land surface is available for infiltration. For these
reasons the recharge rate was in excess of the estimated bulk recharge rate given for the Region of Waterloo.

The domain was divided into four distinct depositional units. The physical properties of the four units within the domain were assigned average hydraulic conductivities as determined from the falling head permeameter tests and pumping test conducted during the field investigation (Figure 33). The best hydraulic head calibration was observed with increasing each of the average falling head permeameter measurements in each depositional unit by a factor of 1.25. This was within an acceptable margin of error for determining hydraulic conductivities within most aquifer systems. Furthermore, the bulk hydraulic conductivity derived from the pumping test was a factor of 1.26 greater than the average hydraulic conductivity determined from the falling head permeameter tests. From the top of the domain to the bottom these hydraulic conductivities were 5.14 x 10⁻⁵ m/s, 2.61 x 10⁻⁴ m/s, 5.76 x 10⁻⁴ m/s, and 5.14 x 10⁻⁵ m/s and the porosities were 0.33, 0.33, 0.326, and 0.33, respectively. The four units were defined as isotropic and the van Genuchten parameters for variably saturated flow were defined from coupling Mualem's catalogue of soils (Mualem, 1976) with the van Genuchten method of predicting the hydraulic conductivity of unsaturated soils (van Genuchten, 1980; Bennett and Rumpf, 1995). The two catalogued soil types used within the model domain were soil# 4135 and soil# 4136, gravelly sand and fine sand G.E.#3, respectively. The van Genuchten parameters for these soils were; \( S_{wt} = 0.243, \alpha = 1.505 \text{ m}^{-1}, n = 2.8388, m = 0.6477 \) for soil# 4135; and \( S_{wt} = 0.1844, \alpha = 0.719 \text{ m}^{-1}, n = 3.894, m = 0.7432 \) for soil# 4136.
The initial nitrate distribution and boundary conditions used to solve the solute transport equation (6) are illustrated in Figure 34. As stated above, the bottom of the domain was a no-flow boundary and the top had a recharge of 0.39 metres per year with a nitrate concentration of 15 mg/L NO₃-N. The inflow boundary had five different nitrate concentrations as first type boundary conditions. These were fixed concentrations representing the regional nitrate concentration profile. This regional nitrate profile was taken from a representative nitrate profile obtained from the three multilevel monitoring wells beside the pumping wells at the study site (Figure 35). These concentrations also defined the initial conditions throughout the domain. The outflow boundary for transport was defined as a second type boundary with zero dispersive flux, this forces the concentration gradient to zero while still allowing advective mass flux to cross the boundary. The dispersivities were the same throughout the domain and were modified from Sudicky (1986) to conservatively represent the current study site. These dispersivities were 0.5 m longitudinally, 0.05 m transverse horizontally, and 0.005 m transverse vertically. The free solution diffusion coefficient used for nitrate was $3.2 \times 10^{-3} m^2/\text{year}$.

### 5.4 Model Calibration

The model was calibrated for flow and transport using the field data obtained from the variable well pumping tests. Figure 36 illustrates the calibration results for flow. The calculated drawdown range defined the drawdown that would be observed in a multilevel monitoring well 2 m up gradient and down gradient from the pumping well. The multilevel monitoring well in the
field was located cross gradient 1.86 m away from the pumping well. The calculated results were closest to field values near the water table and a greater discrepancy between measured and calculated drawdown was observed with depth. All calculated drawdown values were within 1.25 cm of the average field measurements in the multilevel monitoring well beside the deep short screened well measured over 4 days. The standard deviation of the average drawdown measurement in the multilevel monitoring well was less than 0.5 cm. The small discrepancy can likely be attributed to greater heterogeneities in hydraulic conductivities with depth (Figure 33). A realistic range of hydraulic conductivities were tested within each defined layer of the model domain in an attempt to improve calibration results however the most characteristic curve match was found with the chosen hydraulic conductivities. Furthermore, observed discrepancies may be due to a difference of 14 centimetres between the field multilevel and the horizontal distance between the pumping well and the designated observation point's location in the model since limitations were made by exact nodal positions in the domain grid. The model was also calibrated using the drawdown in the deep short screened pumping well (Figure 36). The calculated pumping well drawdown was 16 centimetres more than measured values. This is most likely due to slight differences in measured hydraulic conductivities and average hydraulic conductivities defined for the zones within the model. The pumping rate and duration used for the model calibration was 38 L/min for 2 hours, this was equivalent to the longest pumping event in the field investigation. In the field study drawdown values did not change measurably after 15 minutes. Two hours was therefore assumed to represent steady-state conditions. Since most of the comparative numerical simulations were done with the deep short screened well design a calibration for flow has only
been illustrated for the deep short screened well. However, the calibration curves for the shallow short screened well and the fully screened well designs tested were also acceptable. The two modes of flow calibration discussed above showed good agreement between field and model results.

The model was calibrated for transport using the effluent concentrations in the three pumping wells. On the basis of a study by Harmsen (1989) an initial assumption was made that continual pumping and intermittent pumping while still extracting the same total daily volume would produce equivalent results. Further numerical simulations were performed to confirm this assumption of equivalency between pumping patterns and will be discussed below. The numerical simulation used a continual pumping rate of 9.5 L/min which yielded the equivalent total daily volume extracted in the field investigation at 38 L/min intermittently for a total of 6 hours. Figure 37 shows the nitrate concentrations measured at the farm compared to the calculated values from the numerical simulations. The variability in nitrate concentrations in (a) the fully screened and (b) the shallow short screened well span a large range making it difficult to assess the accuracy of the modelled values. Generally however, the field data bridges the calculated concentrations for these two well designs. The deep short screened well (c) has less oscillation in the data and as such can be compared more accurately to the calculated concentrations. The calculated values slightly over estimate the nitrate in the effluent (less than 1 mg/L NO₃-N). Since most of the comparative numerical simulations were done with the deep short screened well design and the model calibration appears to slightly over estimates the effluent concentration the numerical
simulations were considered conservative. The differences between the measured and calculated concentrations were determined to be acceptable within field heterogeneities in nitrate concentrations.

5.5 Results and Discussion of the Numerical Simulations

The results will be presented and discussed in the following order. First, a sensitivity analysis will be presented to illustrate the effect quantity and quality of the recharge has on the nitrate distribution within the domain. Secondly, a comparison will be made between intermittent versus continual pumping. Thirdly, the simulated results for the transient and steady-state nitrate distributions in the aquifer for each well design tested at the farm study site will be presented. Fourthly, a series of physical properties of an aquifer will be compared to determine the principal factors that influence nitrate capture in a domestic well and lastly, one possible solution for minimizing nitrate capture in the domestic well will be illustrated and discussed.

The numerical simulations investigated the effect of well geometry in a contaminant-stratified unconfined aquifer. The three well designs closely simulated the field test conditions. A long screen spanned from 0 m to 6 m and two short well screens spanned from 1 m to 2.5 m and from 4 m to 5.5 m above the aquitard in the model domain.
Careful consideration was given to the influence of the no-flow lateral boundary conditions. Examination of the steady-state drawdown at the lateral boundaries of the model domain indicated that interference imposed by boundary conditions was negligible.

The nitrate concentrations ranged from 0 to 15 mg/L NO$_3$-N throughout the model domain for all simulations. Excluding the simulations with variable recharge quantity and quality, only the 5 mg/L and 10 mg/L contours are illustrated on the figures as they represent concentrations of regulatory interest.

5.5.1 Sensitivity Analysis of Recharge Quality and Quantity

As mentioned in the section describing the model domain, the surface recharge rate and nitrate concentration in the recharge waters were estimated from recharge values used for the Region of Waterloo and nitrate values observed in the shallow groundwater at the study site. It is therefore useful to assess the magnitude of the error possibly created from these estimations.

Figure 38 illustrates the sensitivity of the simulated steady-state nitrate distribution to variable surface recharge conditions when pumping a deep short screened well. Comparing (a) estimated recharge with high nitrate concentration and (b) estimated recharge with zero nitrate concentration, the results showed no observable difference after 2 years. This implies that the nitrate in the recharge water over this surface area (1200 m$^3$), such as may be derived from excess fertilizer,
does not significantly contribute to the nitrate distribution under the field. Instead, the nitrate distribution is dominated by the regional nitrate concentrations and stratification for an aquifer with predominantly horizontal flow. Comparing (a) estimated recharge with high nitrate concentration and (c) no recharge, a greater effect was observed. Less recharge appeared to slightly diminish the vertical movement of high nitrate waters down gradient from the pumping well. This suggests that large recharge events such as irrigation in the summer or spring snow melt, enhance vertical mixing and increased depths of penetration of nitrate in the aquifer.

5.5.2 Intermittent versus Continual Pumping

The simulated nitrate concentrations in the effluent from the deep short screen well after two weeks of pumping intermittently versus continually are illustrated in Figure 39. The same total daily volume is extracted in both pumping schedules however the two pumping schemes have different pumping rates and consequently different pumping durations. In the first simulation the pump was extracting 38 L/min for six hours and then turned off for eighteen hours and in the second simulation the pump was continually extracting 9.5 L/min. Initially there appeared to be more nitrate captured by the intermittent pumping. However, after forty eight hours the calculated nitrate concentration in the effluent of both pumping strategies appeared to be equivalent. This supports the work of Harmsen (1989) who has shown that the particle trajectories are essentially identical under intermittent, daily cyclic pumping and steady state pumping, when the total volume of water pumped per day is the same in both cases (Harmsen (1989); Harmsen et al., 1991a; Harmsen et al., 1991 b).
This observation suggests that pumping intermittently versus continually to yield equivalent total daily volumes makes no measurable difference in the final effluent concentration for the case of a relatively deep short screened well in an aquifer with homogeneous geochemical stratification given the currently defined physical parameters.

The numerical simulations also allowed for an examination of the nitrate distribution throughout the model domain after different pumping schedules with the deep short screened well. Again, for the same total daily volume of 13,680 L, a comparison of four different pumping schedules are illustrated in Figures 40 and 41. Figure 40 compares (a) an intermittent pumping schedule equivalent to that used during the field tests (see Table 1) and (b) a continual 6-hour extraction at the same rate with both scenarios resulting in the same total extraction volume. There were no observable differences in the nitrate distribution both longitudinally and laterally after one day of pumping. In contrast, Figure 41 shows some differences in nitrate distribution between intermittent (a 6 hour pumping period with an 18 hour recovery period as in scenario (b) above) versus continual extraction after one month of pumping. Here the higher nitrate concentrations near the water table penetrate deeper, down gradient from the pumping well with continual low flux pumping. Figure 42 shows that the location of the increased vertical nitrate penetration corresponds to the location of essentially vertical velocity vectors. An increased magnitude of the velocity vectors, and hence an increased potential for transporting nitrate deeper, is dependant on the magnitude of the vertical gradient induced from pumping assuming the same hydraulic conductivities and porosities. It should be noted that after one month, the nitrate distribution around
the well screen and consequently the nitrate concentrations in the pumping effluent is representative of steady-state conditions.

The unique numerical results for each pumping well designs will be discussed next. Starting with the differences in flow fields induced from equivalent pumping rates under different well geometries, a discussion will follow to present differences in nitrate distributions parallel to the flow field and perpendicular to the flow field under each scenario.

5.5.3 Hydraulic Head Distributions Under Different Pumping Scenarios

The longitudinal, transverse, and plan view (at mid screen depth) flow fields generated from continually pumping each well design at 9.5 L/min are illustrated in Figures 43, 44 and 45 respectively. The head distributions would suggest larger vertical gradients are created with short screened well geometries as compared to fully screened wells. The larger vertical gradients would create greater potential for vertical transport of nitrate under short screened well pumping conditions.

The simulated total hydraulic head distributions in Figures 43, 44, and 45 reflect the characteristic hydraulic head distributions across the depth of the aquifer as measured in the multilevel monitoring wells beside each well design. When pumping at a rate of 38 L/min the maximum differences in total hydraulic head across the depth of the aquifer at a radial distance of
2 m from a shallow and deep short screened well were calculated to be approximately 2.0 cm and 4.0 cm, respectively. These values differed from the maximum measured drawdown difference across the aquifer of 3.5 cm for both short screened well designs as measured in the multilevel monitoring wells in the field investigation at 1.70 m away from the shallow short screened pumping well and 1.86 m away from the deep short screened well. The divergence is most probably due to the difference in radial distances from the pumping well between the model and the field investigation although, differences in anisotropy within the geologic units could also offer an explanation.

As Figures 46 and 47 show, the differences in total hydraulic head distributions across the depth of the aquifer are influenced by anisotropy ratios and gradient differences. Anisotropy appears to increase the observed range in total hydraulic head across the depth of the aquifer. Furthermore, a higher regional gradient appears to decrease the range in total hydraulic head across the depth of the aquifer longitudinally but only small differences in the flow field were seen transversely.

In light of the flow fields illustrated above, the vertical transport of stratified nitrate contamination can more fully be appreciated and understood. In the discussion that follows, nitrate distributions are illustrated and discussed for the flow field scenarios presented above. In addition, simulated nitrate distributions for lower pumping rates resulting in lower total daily extraction volumes will be illustrated and discussed.
5.5.4 The Evolution of Nitrate Distribution

Figures 48 and 49 compares the steady-state nitrate distributions for each well design both longitudinally and transversely. The model results suggest that a fully screened well causes the least disturbance to the regional vertical distribution of nitrate compared to the short screened wells. The fully screened well design consequently reaches steady state before (three months) the short screened wells (six months). The effect of pumping each of the well designs described above appears to influence longitudinal nitrate distributions more than transverse distributions. In Figures 50, 51, and 52 illustrating the evolution of nitrate distributions, only the longitudinal distributions are illustrated since transverse effects are measurable but limited.

As stated Figures 50, 51, and 52 illustrate the longitudinal evolution of the nitrate distribution within the model domain from initial concentrations to steady-state concentrations for each well design. The nitrate concentrations in the vicinity of the well do not measurably change after one month for all three well designs. This was reflected in steady-state nitrate concentrations being reached in the pumping well effluent after approximately 1 month (720 hours) for all three well designs (Figure 53). The majority of change in nitrate distribution after one month of pumping occurs down gradient from the pumping well. The deep well scenario tends to cause the deepest spreading of higher nitrate concentrations. This is due to the nature of the flow field induced by the deep pumping well (Figures 43, 44, and 45) which results in the most significant migration downward of high nitrate waters. Conversely, the shallow short screened well design tends to
improve the deeper groundwater quality by reducing the thickness of the contaminated zone down gradient from the pumping well. Each well was continually pumped to extract 13,680 L per day from the aquifer. The maximum change in the vertical distribution of nitrate occurred approximately 2.5 m down gradient from the pumping well for all three well screen designs with the current aquifer properties.

Although the effect of extracting water from an unconfined contaminant-stratified aquifer is more salient parallel to the flow field there is measurable influence perpendicular to the flow field. Figure 49 illustrating the transverse steady-state nitrate distribution for each well design also incorporates the transport effects of radial flow towards the well screen. This factor would not be included in a two-dimensional simulation. This consideration is most significant in the short screened wells. The above observation confirms the importance for the use of a three dimensional numerical simulation.

The numerical simulations offered insight into the high nitrate variability in the upper sampling points of the multilevel monitoring wells during the intermittent pumping tests in the field. Both field measurements and model simulations would suggest measurable vertical gradients and consequently vertical nitrate transport up to a radial distance of 2 m under the present aquifer system.
As the simulated flow fields would suggest anisotropy also influence vertical transport of nitrate. Figure 54 shows the longitudinal and transverse steady-state nitrate distributions in an isotropic and anisotropic aquifer system. Both illustrated cases are under a regional gradient of 0.003 with a continual pumping schedule. The simulations suggest that an anisotropic ratio of 10 limits downward movement of high nitrate groundwater near the water table. Another aquifer properties also able to restrict vertical transport of nitrate are higher regional gradients.

Figure 55 is a comparison of nitrate distributions when continually pumping a deep short screened well under a regional gradient of 0.003 (observed in the field) and a high regional gradient of 0.01. The steady-state longitudinal and transverse nitrate contours would suggest that a higher regional gradient also restricts vertical transport of nitrate.

Given the same physical aquifer properties and reducing the total daily volume extracted by continually pumping at a lower extraction rate can also limit vertical movement of nitrate. Figure 56 (a) and (b) illustrates this by comparing a higher continual pumping rate of 0.57 m³/hour (9.5 L/min) to a lower continual pumping rate of 0.057 m³/hour (0.95 L/min) in an isotropic porous media. Furthermore, the results shown in Figure 56 (b) and (c) suggest that extracting a smaller total daily volume combined with anisotropy are additive in limiting vertical nitrate transport. This lower pumping rate combined with an anisotropy ratio of 10 resulted in minimal changes to the initial nitrate distribution within in the model domain.
Although the steady-state nitrate distributions for each pumping scenario discussed above can offer insight to the potential effluent concentrations from the pumping well, a closer examination of the transient effluent concentrations in the pumping well is required.

5.55 Physical Properties of an Aquifer and Steady State Effluent Concentrations

Previous investigations by Conant et al. (1995) and Akindunni et al. (1995) involved a contaminant-stratified unconfined aquifer with variable screen lengths and continual pumping. The former describes a field evaluation with supporting evidence through numerical simulations and the later is a more general numerical evaluation of variable well geometries in a contaminant-stratified unconfined aquifer. Conant et al. (1995) and Akindunni et al. (1995) show that at early times of pumping the effluent concentrations were similar to the concentrations adjacent to the well screen, but at late times, the effluent concentrations of the pumping well approached the vertically averaged concentration in the shallow unconfined aquifer. It was also found that the time to reach the vertically averaged concentration was defined by the well geometry, initial location of the contaminant plume in relation to the well screen, and hydraulic properties of the aquifer (See Table 3 for a site comparison). The model used in the above study was a transient axisymmetric variably saturated flow model coupled with a particle tracking model. The present study builds on these findings by further investigating the effects of regional gradient, anisotropy ratios within depositional units, transmissivity, and surface recharge quantity and quality.
The results from the current numerical simulations differ from the findings of Conant et al. (1995) and Akindunni et al. (1995). Figure 57 illustrates that the vertically averaged concentration of the shallow unconfined aquifer (5.6 mg/L NO$_3$-N) is approached by all three well designs but differences do exist between them. It is important to note that a reduction of 1 mg/L NO$_3$-N represents 6.5% of the maximum concentration, therefore apparently small analytical improvements actually represent significant relative improvements. The discrepancy between the results of the current study and the above mentioned is most likely due to an additive effect of surface recharge, a regional hydraulic gradient, and transmissivity differences.

When the transmissivity was increased by increasing the hydraulic conductivity within each depositional unit by an order of magnitude a substantial difference was observed between the well geometries tested. These simulation results of effluent concentrations in three well geometries over time can be observed in Figure 58. These findings imply that screen length and placement should not be weighed independently, but that transmissivity should also be taken into consideration for enhancement of water quality.

In Figure 57 the steady state concentration in the effluent was reached after approximately one month for the short screened wells and within four days for the fully screened well. This supports the finding by Conant et al. (1995) that well geometry controls the time to reach the steady state concentrations in the effluent.
Figure 59 illustrates three other factors which improve the steady-state concentration in the pumping well effluent. These are regional hydraulic gradient, anisotropy of the porous medium, and reduced total daily volume extracted. Figure 60 shows that within a range of regional hydraulic gradients the difference in steady-state nitrate concentrations from a deep short screened well was approximately 2 mg/L NO₃-N (13% of the maximum concentration within the domain). Furthermore, for the same well design Figure 59 (a) shows that an anisotropy ratio of 10 improves the final water quality significantly, as does a relatively high regional hydraulic gradient. However, the anisotropy of the aquifer does increase the time to reach steady state concentrations. As stated above the downward migration of nitrate was significantly reduced as the lateral regional gradient increased and as the anisotropy ratio increased. When a smaller total daily volume was extracted from the well (Figure 59 (b)), the final concentration improved from 5 mg/L NO₃-N to 2 mg/L NO₃-N, a 20% relative improvement. Figure 59 (b) also shows a combined improvement of an anisotropic aquifer coupled with a smaller total daily volume. Although continual pumping at a lower pumping rate improved water quality in the effluent leaving the regional nitrate distribution undisturbed, only 10% of the daily water requirements are met. This would in turn require 10 wells of similar design to continually pump at the same low rate to meet the total daily water demand of the farm. This option is economically not feasible although something between these two options might be viable.
5.5.6 Groundwater Extraction Using a Multiple Well Design

The results and discussion above has illustrated that improvements were made with a shorter screen length and deeper placement of a well, higher regional gradients, and an anisotropy ratio of 10 for the porous media. The lower pumping rate did improve the quality of the effluent however failed to meet the water supply needs of the farm. Combining these observations a three well design was simulated to further enhance water quality. The strategy was to align three wells along the gradient 5 m apart with two shallow wells pumping at higher rates than the deeper domestic well (Figure 61). The two 'skimmer' wells on either side of the domestic water supply well maintained the higher nitrate concentrations near the water table and supplied water for the general dairy farming needs. The three well design improved the domestic drinking water quality to approximately 1 mg/L NO₃-N and the two skimmer wells had a nitrate concentration of approximately 7 mg/L NO₃-N. This was a significant improvement in drinking water quality as compared to a single deep short screened well (Figure 62) (5-6 mg/L NO₃-N).

The location of the stagnation zone is a function of the regional gradient and pumping rate. The efficiency of the three well design to minimize nitrate in the drinking water is dependent upon both of these factors. The sensitivity to these parameters may be a limitation for this design in environments where regional gradients significantly change. Economically three wells would still involve additional cost to the farm owner however many farms commonly have two to three wells on site (Agriculture Canada, 1992). Ideal separation distances and pumping rates for minimizing
nitrate in the drinking water could be determined on the basis of water needs and physical properties of the aquifer.
6.0 GENERAL DISCUSSION

The following is a discussion of the field investigation in light of the numerical simulations. The effectiveness of well design in conjunction with pumping strategies for improving water quality will be reviewed. Furthermore, the origin of the nitrate in the groundwater and need for reduced delivery of nitrate mass to the regional aquifer will be addressed.

The field investigation saw improvement in water quality with a deep short screened well (Figure 24). The effluent concentration seemed to have reached a relatively steady-state concentration although the numerical model would suggest that a one month test would have better evaluated the effectiveness of well design on improved water quality. Furthermore, the higher water quality of the field investigation as compared to modelled results in a deep short screened well in a contaminant-stratified unconfined aquifer could be attributed to differences in three physical properties of the aquifer. The main physical properties contributing to the model's conservative results are possibly higher hydraulic conductivities, anisotropy, and regional gradients in the field.

The field investigation supported by the modelling showed that pumping schedules, while still extracting the same total daily volume, made no measurable difference to the nitrate concentration in the effluent. Furthermore, the model demonstrated that for the same total daily volume intermittent versus continual pumping at a lower pumping rate made no difference to the
effluent concentration but did show differences in nitrate distribution down gradient of the pumping well.

The observed variability in nitrate concentrations at shallower sampling points on the multilevel monitoring well can be explained by downward transport of waters with high nitrate concentrations, down or cross gradient of the pumping well.

Significant spatial and temporal variability of nitrate throughout the study site were documented. However, detectable nitrate concentrations were consistently limited to the upper 5 metres of the unconfined aquifer. The majority of spatial and temporal variability occurred at mid-aquifer depths which was consistent with the conceptual model (Figure 2) and findings of Ryan (1994) where 'microplumes' of contamination moved horizontally, 1 to 2 metres below the water table. These results together with the numerical sensitivity analysis of recharge quality and quantity suggest that the majority of nitrate mass originates up gradient and is transported horizontally in the regional groundwater flow. Nitrate entering as recharge along the top boundary penetrated the aquifer primarily during large recharge events at highly permeable areas of the cultivated field, from septic systems, or to a lesser extent from the drawdown effects of a pumping well. Reviewing the field results with the numerical simulations it is likely the nitrate reached the water table as a 'multi-point' source up gradient through zones of preferential infiltration and water table mounding, or as a 'point' source such as under a septic field. The numerical modelling showed that the local contribution of nitrate represents only a fraction of the total nitrate mass flowing through the
system. This suggests that the current nitrate mass in the aquifer originated from many cultivated fields and septic systems and has accumulated over at least 30 years.

Denitrification is believed to be the primary mechanism below the water table where by the nitrate mass can be depleted. The $\delta^{15}N_{\text{NO}_3}$ values suggest that denitrification had occurred up gradient from the site. This would imply that the aquifer has the potential to remove the current nitrate mass contained therein.

Excess fertilizer applications on a regional scale combined with numerous conventional septic systems will likely continue to cause further degradation of water quality within the regional aquifer system. It is therefore important to emphasise that collective efforts by the rural community and agricultural industry are required to prevent nitrate from reaching the water table. The economic issues of production loss are recognized. However, the method of variable fertilization application rates have proven effective in minimizing excess leachable nitrogen while maintaining productivity (Kachanoski and Fairchild, 1993). Furthermore, alternative septic system technologies have been shown effective in reducing the nitrate mass in the septic effluent (Carmichael, 1994; Robertson and Cherry, 1995).
7.0 SUMMARY AND CONCLUSIONS

The problem of groundwater contamination from nitrate has long been recognized in the rural community and some evidence indicates the water quality is steadily declining. The current investigation has evaluated the influence of different extraction strategies in an unconfined aquifer on regional nitrate stratification and nitrate concentrations in the pumping well effluent.

Monitoring water quality in the domestic well at the study site confirmed the findings of the OFGQS (Agriculture Canada, 1992) that the drinking water at a farm located on highly permeable soil, can be susceptible to nitrate contamination. The shift in groundwater flow directions during large recharge events increased the number of potential sources of nitrate. The well became vulnerable to septic system effluent during spring thaw and substantial irrigation events. The use of nitrogen isotopes together with supporting physical and chemical data proved to be a useful tool in delineating the source of nitrate contamination in the drinking water. The field investigation successfully determined the source of nitrate in the domestic well. The results showed that for part of the year the domestic well was influenced by septic system effluent.

The field results together with numerical simulations demonstrated that pumping schedules and well design can improve the drinking water quality. Under the defined physical parameters the effluent concentration was essentially the same for intermittent versus continual pumping for the
equivalent total daily volume, however, greater vertical transport of nitrate was documented numerically with continual versus intermittent pumping. Improvements were noted with a shorter well screen placed at a depth with minimal to no nitrate contamination. Greater differences in steady-state effluent concentrations between well designs were observed if the aquifer had a higher transmissivity. Furthermore, nitrate contamination in the effluent from a deep short screened well was improved if the aquifer had an anisotropy ratio of 10, or if the magnitude of the regional gradient was increased. A cumulative improvement was demonstrated numerically when combining the effect of anisotropy with an increased gradient. Numerical modelling illustrated that a short screened well increased the down gradient disturbance of the regional nitrate stratification in the aquifer and that steady state nitrate distributions were reached with a fully screened well before a short screened well. Notable water quality improvements were made if the total daily volume from any given well was reduced. To capitalize on this observation a three well design was proposed as an approach to improving the drinking water quality. Significant enhancement of the water quality was numerically simulated by aligning three short screened wells at different depths parallel to the flow field. The two shallow 'skimmer' wells on either side of a deep domestic well all pumped at low rates but collectively supplied the total daily water requirements of the farm. The success of this design is dependent upon the correct separation distance between the three wells given specific regional gradients and pumping rates.
Furthermore, numerical modelling together with field evidence concluded that the local contribution of nitrate during recharge events was only a fraction of the total nitrate mass carried by the regional flow system. This suggested that the current nitrate mass in the aquifer originated from many cultivated fields and septic systems and has accumulated over at least 30 years. Implications are that a collective effort on a regional scale is required to reduce excess nitrate from reaching the water table.

The conclusions drawn from the results of this study allow the following recommendations to be made. Primarily, a collective effort needs to be made by the rural community to reduce total nitrogen loading to a regional unconfined aquifer from which many rural residents draw their drinking water. Recent studies for variable fertilizer application rates have demonstrated that excess leachable nitrogen can be reduced while still maintaining agricultural productivity (Kachanoski and Fairchild, 1993). Similarly, alternative septic system technologies can reduce the nitrate mass released beneath the septic field (Carmichael, 1994; Robertson and Cherry, 1995).

In lieu of collective reduction of nitrate to the flow system some interim steps can be taken to minimize nitrate in the drinking water. Prior to installing a domestic well or septic field a seasonal assessment of local groundwater flow directions needs to be made. This would prevent possible nitrate contamination from the septic field during times of shifted hydraulic gradients. Water quality improvements can also be obtained through shorter well screens placed deeper in a
contaminant-stratified unconfined aquifer. Furthermore, a series of three staggered wells, such as the proposed three well design, can substantially reduce the nitrate uptake in the domestic well.
REFERENCES


Table 1: Designed pumping schedule for variable screen length and well depth investigation.

<table>
<thead>
<tr>
<th>Number of Days</th>
<th>Time (Hours)</th>
<th>Pumping Rate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Day 1</td>
<td>0.0-2.0</td>
<td>38 L/min</td>
</tr>
<tr>
<td>Day 2 and Day 3</td>
<td>2.0-3.0</td>
<td>OFF</td>
</tr>
<tr>
<td></td>
<td>3.0-3.5</td>
<td>38 L/min</td>
</tr>
<tr>
<td>Day 3</td>
<td>3.5-4.5</td>
<td>OFF</td>
</tr>
<tr>
<td></td>
<td>4.5-5.0</td>
<td>38 L/min</td>
</tr>
<tr>
<td></td>
<td>5.0-6.0</td>
<td>OFF</td>
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<td>6.0-6.5</td>
<td>38 L/min</td>
</tr>
<tr>
<td></td>
<td>6.5-7.5</td>
<td>OFF</td>
</tr>
<tr>
<td></td>
<td>7.5-8.0</td>
<td>38 L/min</td>
</tr>
<tr>
<td></td>
<td>8.0-9.0</td>
<td>OFF</td>
</tr>
<tr>
<td></td>
<td>9.0-11.0</td>
<td>38 L/min</td>
</tr>
<tr>
<td></td>
<td>11.0-24.0</td>
<td>OFF</td>
</tr>
<tr>
<td>Day 4 only</td>
<td>0.0-2.0</td>
<td>38 L/min</td>
</tr>
<tr>
<td></td>
<td>2.0-3.0</td>
<td>OFF</td>
</tr>
<tr>
<td></td>
<td>3.0-5.0</td>
<td>38 L/min</td>
</tr>
<tr>
<td></td>
<td>5.0-6.0</td>
<td>OFF</td>
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<tr>
<td></td>
<td>6.0-8.0</td>
<td>38 L/min</td>
</tr>
<tr>
<td></td>
<td>8.0-24.0</td>
<td>OFF</td>
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Total Daily Water Volume is 13680 L
Table 2: Bulk aquifer hydraulic properties calculated using Neuman's fully penetrating well solution for evaluating time-drawdown curves from a pumping test with a fully screened pumping well and two fully screened observation wells.

<table>
<thead>
<tr>
<th>Time</th>
<th>$K_r$ (m/s)</th>
<th>$K_V$ (m/s)</th>
<th>$K_T / K_V$</th>
<th>$SS$ (1/m)</th>
<th>$S_y$</th>
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</thead>
<tbody>
<tr>
<td>OW1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>early</td>
<td>5.4 x 10^{-5}</td>
<td>1.1 x 10^{-3}</td>
<td>0.05</td>
<td>0.005</td>
<td></td>
</tr>
<tr>
<td>late</td>
<td>6.7 x 10^{-4}</td>
<td>1.4 x 10^{-2}</td>
<td>0.05</td>
<td>0.34</td>
<td></td>
</tr>
<tr>
<td>OW2</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>early</td>
<td>1.6 x 10^{-4}</td>
<td>4.3 x 10^{-4}</td>
<td>0.37</td>
<td>0.001</td>
<td></td>
</tr>
<tr>
<td>late</td>
<td>9.7 x 10^{-4}</td>
<td>2.6 x 10^{-3}</td>
<td>0.37</td>
<td>0.1</td>
<td></td>
</tr>
<tr>
<td>Avg</td>
<td>4.6 x 10^{-4}</td>
<td>4.4 x 10^{-3}</td>
<td>0.10</td>
<td>0.003</td>
<td>0.22</td>
</tr>
</tbody>
</table>

Additional Information

PW1 $Q = 1.3 \times 10^{-3} \text{ m}^3/\text{sec}$
OW1 $b = 6.56 \text{ m}, \ r = 1.46 \text{ m}$
OW2 $b = 6.53 \text{ m}, \ r = 3.98 \text{ m}$

Where $K_r = \text{Radial Hydraulic Conductivity}$
$K_V = \text{Vertical Hydraulic Conductivity}$
$S_S = \text{Specific Storage}$
$S_y = \text{Specific Yield}$
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<table>
<thead>
<tr>
<th></th>
<th>Current Study Site</th>
<th>Base Borden Site</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pumping Pattern</td>
<td>Intermittent</td>
<td>Continual</td>
</tr>
<tr>
<td>Total Volume Extracted</td>
<td>54,720 L over 4 days</td>
<td>207,360 L and 108,000 L over 3 days</td>
</tr>
<tr>
<td>Pumping Rate (Q)</td>
<td>38 L/min</td>
<td>48 L/min, 25 L/min</td>
</tr>
<tr>
<td>Hydraulic Conductivity (K_{sat})</td>
<td>10^{-4} m/s</td>
<td>10^{-3} m/s</td>
</tr>
<tr>
<td>Anisotropy Ratio</td>
<td>&lt; 10</td>
<td>&lt; 10</td>
</tr>
<tr>
<td>Regional Gradient</td>
<td>0.003 m/m</td>
<td>0.005 m/m</td>
</tr>
<tr>
<td>Average Saturated Thickness (b)</td>
<td>6.92 m</td>
<td>5.86 m</td>
</tr>
<tr>
<td>Average Thickness of Contaminant Zone</td>
<td>5.0 m</td>
<td>1.5 m</td>
</tr>
<tr>
<td>Location of Contaminant Zone</td>
<td>Top of aquifer</td>
<td>Bottom of aquifer</td>
</tr>
</tbody>
</table>
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Section 4.0

Using the Waterloo Profiler to Delineate Rural Contamination
ABSTRACT

The level of groundwater quality in rural communities is decreasing with time due to the increased density of residents and fertilizer use. In a rural environment there is a potential for many sources of groundwater contamination. Farmers need a quick and inexpensive method of determining their water quality. The main objective of this project was to demonstrate profiling as a useful tool to map inorganic plumes in a regional field system, in a manner that is both economical and simple. It was found that in one day profile holes can be driven around the septic field, nearby roads, and the agricultural field, allowing for the general impact of the many sources to be determined. During the two day field investigation a total of 64 samples were collected.

In almost all cases the nitrate concentrations at the site were above the drinking water standard of 10 mg/L. These high concentrations persisted with depth to the location of the drinking water well. High nitrate concentrations were found near the septic field, beside the office and the road. Two major potential sources of nitrate were present at the site (agricultural activity and septic systems). The more distal sources (agricultural activity, and septic systems of neighboring owners) formed a plume of nitrate that underlayed the nitrate plume at the field site producing nitrate stratification. These deeper plumes are of concern since they have a greater potential of contaminating wells that have been placed deeper, in hopes of avoiding local contaminants.

The chloride concentrations at the site were found to be slightly below the drinking water limit of 250 mg/L. The major sources of chloride at the site were: water softeners, agricultural activity, and road salt application. Once again the more distal sources formed a plume which underlayed the local sources.

The Waterloo Profiler has the potential to reveal many of the groundwater problems that may be present in a rural environment, in a short period of time, and at a low cost.
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1.0 INTRODUCTION

In Canada, over six million people (26% of the population) rely on groundwater for domestic use. Approximately two-thirds of these people live in rural areas (Environment Canada, 1990). Contamination of these drinking water supplies by malfunctioning septic systems, increasing agricultural activity, and application of road salt is on the rise. One or a combination of the aforementioned sources can lead to increases in chloride and nitrate levels in groundwater.

Nitrate ($\text{NO}_3^-$) is the most widespread contaminant in groundwater, mainly in shallow unconfined aquifers. The two major contributors to nitrate in groundwater are septic systems and agricultural activities. This is of concern because the main source of water for rural residents is generally a shallow well in an unconfined aquifer. Nitrate itself is not toxic, but becomes a health concern when it is reduced to nitrite ($\text{NO}_2^-$) - a toxic substance. Methemoglobinemia is an acute toxic response to nitrite exposure that results in oxygen starvation. Infants of less than six months of age are most at risk, since their higher stomach pH allows the survival of a reducing type bacterial population (Bouchard et al., 1992). The drinking water standard of nitrate-N is 10 mg/L (Health and Welfare Canada, 1993).

Chloride ($\text{Cl}^-$) is a very common contaminant in groundwater. It may be due to the application of fertilizers such as potassium chloride (KCl), water softened sewage from septic systems, and road salt. The drinking water standard for chloride is 250 mg/L (Health and Welfare Canada, 1993).

Approximately thirty percent of wastewater in the United States and Canada is disposed of by on-site subsurface disposal. This leads to 1.4 billion liters of wastewater being "poured into the ground" daily (Robertson, 1993). Consequently groundwater degradation due to septic systems has become a public issue, particularly in rural areas where the possibility of well water contamination exists. Whereas a septic system is a point source contamination, agricultural activity is non-point source contamination. Non-point source contamination occurs when the source of contaminants are...
spread over a large area (e.g. cultivation field). This is also a growing problem in North America due to the increasing use of fertilizers. Research by Weigmann and Kroehler (1988) showed that on average approximately half of the nitrogen applied as fertilizer is harvested with the crops, and the excess is available to enter the groundwater system.

With many sources of groundwater contamination present in rural areas; the use of conventional monitoring techniques can become very expensive and time consuming to map the thickness and extent of a plume. The Waterloo Centre for Groundwater Research at the University of Waterloo has developed a drive-point sampling device for such purposes (The Waterloo Profiler), (Pitken, 1994). Since this device has only been used previously to delineate organic contaminant plumes, the main objective of the current research was to demonstrate profiling as a useful tool to map inorganic plumes in a regional field system in a manner that is both economical and simple. The secondary objectives were to outline the geochemistry of the area, to delineate the sources of contamination, and to predict the impact of the contaminants on the well drinking water. The water samples collected with the profiler were used to measure the geochemistry of the plumes. In addition chlorine isotope analyses and the geochemistry of the plumes were combined to help delineate the sources of contamination present at the farm site.

The site investigation was carried out at a typical farmstead located in Haldimand-Norfolk County near the town of Delhi, Ontario. The study area is situated on a highly permeable, relatively homogenous, isotropic, unconfined sand-gravel aquifer where the groundwater flow is predominantly horizontal (Svensson, 1996).
2.0 GROUNDWATER CONTAMINATION IN THE RURAL ENVIRONMENT

2.1 Nitrate Contamination in Groundwater

2.1.1 Nitrogen Transformation Reactions

Other than nitrate and nitrite, other examples of nitrogen compounds found in naturally occurring waters include ammonia (NH₃) and ammonium ion (NH₄⁺). The action of microorganisms transform gaseous atmospheric nitrogen into fixed forms such as ammonium and nitrate (Behnke, 1975). Other microorganisms denitrify fixed nitrate back into gaseous forms such as nitrous oxide and free nitrogen (N₂ gas).

Ammonia is highly soluble in water; it reacts with water and forms ammonium and hydroxyl ions (Behnke, 1975).

\[
NH_3 + H_2O \leftrightarrow NH_4^+ + OH^-
\]

Ammonia ammonium

Ammonia is present at high pH whereas while ionic ammonium is present at lower pH. This process tends to occur in the unsaturated zone.

Nitrification is the oxidation of the ammonium ion to nitrate (Delwiche, 1981). Ammonium is oxidized by nitrifying bacteria (Faure, 1991; Robertson et al., 1991).

\[
NH_4^+ + 2O_2 \leftrightarrow NO_3^- + 2H^+ + H_2O
\]

ammonium dissolved nitrate oxygen

This process depends upon the presence of oxygen, temperature, pH, and concentration of nitrifying organisms (Behnke, 1975). It decreases the pH of the water due to the production of hydrogen ions and tends to occur in the unsaturated zone.
Denitrification is a process where the nitrate serves as an electron acceptor for an organism oxidizing an organic compound to obtain energy (Delwiche, 1981).

\[
4NO_3^- + 5CH_2O + Bacteria \rightarrow 2N_2(g) + 5HCO_3^- + H^+ + 2H_2O
\]

When organic carbon is unavailable, autotrophic bacteria (derive energy from light and inorganic sources) accomplish nitrate reduction by utilizing energy derived from the oxidation of reduced sulphur or ferrous ion (Robertson et al., 1996; Robertson and Cherry, 1992). However, aquifer solids often have sufficient organic material to enable heterotrophic (bacteria derive cell energy from organic carbon) denitrification. This process tends to occur in the saturated zone in the absence of dissolved oxygen (Pederson et al., 1991; Robertson et al., 1991; Stuart et al., 1995). Four conditions are required for denitrification to occur (Starr and Gillham, 1993): nitrate, labile organic carbon, denitrifying bacteria, and reducing conditions (i.e. absence of dissolved oxygen). The presence of these four factors are important since denitrification is the only known process that can permanently remove nitrate from an aquifer. If there is no natural reduction of nitrate there is the risk of reaching nitrate concentrations above the allowable limit of 10 mg/L.

2.1.2 Properties of Nitrate

Nitrate is a highly mobile and soluble compound. Nitrate concentrations are higher near the water table and tend to decrease with depth (Walker et al., 1973; Starr and Gillham, 1993; Robertson et al., 1991). This produces a thin layering of nitrates in the aquifer from a point source, and has been referred to as "long pencil like" plumes (Robertson et al., 1991). Dispersion in sand and gravel often causes little attenuation of nitrate, therefore in some cases nitrate undergoes little dilution with ambient groundwater (Robertson and Cherry, 1992). Because there may be little attenuation of nitrate (under oxidizing conditions) in a flow system it may be used as a conservative tracer for septic system contamination.
2.2 Contamination Due to Septic Systems

Septic systems were designed so that some sewage degrades in the tank and in the surrounding sand and subsoil. They prevent wastewater from ponding on the surface, but do not necessarily prevent groundwater contamination. Three main factors affect the potential for groundwater pollution (Weigmann and Kroehler, 1988): the amount of time wastewater stays in the soil, types of waste in effluent, and the type of soil through which the wastewater moves. The longer the wastewater stays in the unsaturated zone the greater the potential for it to dilute and degrade before reaching the groundwater. The level of toxicity and concentration of the wastes in the effluent will affect the amount of groundwater pollution. Bacterial, viruses, detergents, and household cleaners are common contaminants that enter groundwater systems from septic systems. The soil surrounding the leach fields should adsorb all the effluent generated, provide a high level of treatment before the effluent reaches the groundwater, and have a long, useful life (Weigmann and Kroehler, 1988). Coarse sand can allow wastewater to move through it too quickly; heavy clays impede wastewater flow, therefore it is sometimes difficult to find a type of soil that will meet the above criteria. The best soil type would allow the contaminants to move through it at such a rate that they would either be adsorbed or oxidized before they reach the water table.

2.2.1 Septic System Plume Geochemistry

There are a variety of chemicals in septic effluent, but the major ones commonly studied are: sodium (Na$^+$), potassium (K$^+$), calcium (Ca$^{2+}$), magnesium (Mg$^{2+}$), ammonium (NH$_4^+$), nitrate (NO$_3^-$), chloride, (Cl$^-$), sulfate (SO$_4^{2-}$), phosphate (PO$_4^{3-}$). Other parameters that are measured include: dissolved oxygen (DO), pH, alkalinity, and dissolved organic carbon (DOC). A thorough analysis of these parameters assists in clearly define the plume and the extent of contamination.

Robertson et al., (1991) studied all of the above parameters and how they effect plume character for two small septic systems on sand aquifers in Ontario. They found sodium to be the best indicator of determining the impact from a septic system, since the sodium levels were ten to twenty times
greater than background levels. Chloride and nitrate could not be used as tracers because of the high background levels as a result of agricultural practices at one of the sites. Approximately 130 m down gradient from the tile field sodium concentrations were more than 50 percent of the effluent concentrations. This showed that there was little attenuation of the plume as a result of little dispersion laterally and vertically. There were slightly lower values of pH (mainly a result of nitrification) in the plume core in comparison with the background levels. Phosphate was present only in the area below the tile field.

Three major differences between the tile effluent and the plume concentrations were that much less DOC and NH\text{4}^{+}-N, and much higher NO\text{3}\text{--N} were present in the plume than the effluent. This showed that the organic carbon in the effluent was converted to carbon dioxide and water by aerobic biodegradation; microbial nitrification converted the ammonium in the effluent to nitrate. The low DOC levels and high NO\text{3} noted near the water-table indicate that these processes occurred mostly in the unsaturated zone. In the saturated zone, denitrification was occurring at one site where reducing conditions developed. As chloride values rose the nitrate level remained constant.

Viraraghavan and Warnock (1975) studied groundwater pollution in Ontario due to a septic tile field on sandy-silt soil. They found that pollutant concentrations in the groundwater were much lower than the septic tank effluent due to movement through soil and dilution. However, the concentrations were quite high compared to background levels, mainly chemical oxygen demand (COD), soluble organic carbon (DOC), total phosphate, ammonia, and chloride. Their results showed that although the soil removed a large amount of the pollutants, the concentrations in the groundwater varied significantly because of differences in the depth to the groundwater table and non-uniform loadings.

Robertson and Cherry (1992) studied a large-flux septic system situated in an unconfined sand aquifer. They found that the plume waters were easily differentiated in the field by elevated electrical conductivity values as a result of the high number of ions in the groundwater (primarily
Na, Cl, Ca, and NO₃⁻). The plume was also distinguished by higher DOC values and slightly lower pH values. Analysis of the chloride distribution showed that beneath the area of the tile bed the plume is within the entire thickness of the aquifer, however down gradient it is overlain by non-impacted water as it approaches its discharge point. The boundary between the non-impacted water and the plume was distinct in both the longitudinal and transverse directions. They also found that chloride (non-reactive solute) concentrations down gradient were similar to those in the effluent. The distinct boundary between the plume and the non-impacted water indicated that the sand aquifer did not cause much dispersion in any direction. Down gradient from the tile bed the chloride concentrations increased with depth.

2.3 Groundwater Contamination Due to Agriculture

The septic system under study for the proposed project is set near an agricultural field. Therefore, the background levels of many solute species may be quite high due to agricultural activities. To understand the extent of septic contamination, the extent of agricultural contamination must also be accounted for and understood. Extensive research has been done in this area of non-point source contamination, since it is a growing problem in North America due to the increasing use of fertilizers. On average, approximately only half of the nitrogen applied as fertilizer is harvested with the crops, and the excess may enter the groundwater (Weigmann and Kroehler, 1988).

Irrigation effects water quality in different ways depending on the original water quality, type of soil, underlying geology, type of irrigation, crop grown, and farming method utilized (Environment Canada, 1991). If the original water quality is low, any use of fertilizer will only decrease the quality of the water. A soil with a high permeability and hydraulic conductivity will spread the contamination quickly. Certain crops will absorb more nitrates than others, while certain fertilizers contain a higher percentage of nitrates.
Bohlke and Denver (1995) studied the history and fate of nitrate contamination in two agricultural watersheds in Maryland. They noted that NO$_3^-$ gradients can have a variety of different configurations that depend locally on both transient and steady state variables: changes in nitrate contents of recharging groundwaters over time are partially related to changes in land use practices; the direction, distance, and time between discharge and recharge of contaminated groundwaters (scale, geometry, and velocity of local flow systems); and the distribution and competence of natural remediation (e.g., denitrification). To understand the fate of nitrate contamination the above factors affecting nitrate distribution need to be analyzed. The uncontaminated groundwater near the area of study contained mainly sodium or calcium type chemicals. Contaminated groundwater under the cultivated fields contained magnesium, calcium, nitrate, and chloride solutions; they were largely a result of the agricultural use of crushed dolomite, potash, and various nitrogen fertilizers or manures. All of these contaminants are also present in septic system plumes making it difficult to distinguish between the level of agricultural contamination compared to septic contamination. By comparing the nitrate discharge rates with recharge rates they found that the NO$_3^-$ discharge rates were less than the recharge rates. They determined this difference to be due to the discharge of relatively old waters with low initial NO$_3^-$ concentrations, and local denitrification.

A similar study was done by Jackson et al. in 1973. They studied nitrate in surface and subsurface flow from a small agricultural watershed. Nitrate-N concentrations were found to be higher in subsurface runoff than surface runoff for the period of study. Recharge in the form of rainfall will infiltrate and move laterally as shallow subsurface flow above a low permeability zone. However, NO$_3^-$ tends to move vertically toward the water-table and become a part of sub-surface flow. Therefore, measurements of surface runoff do not represent the total runoff and chemical loss from an agricultural watershed.

Svensson (1996) studied the water-quality of the agricultural field that is under study for this project. Nitrate profiles compared with dissolved oxygen profiles near the house (i.e. close to the septic
weeping beds) showed aerobic conditions throughout the depth of the aquifer. Eight meters below ground surface, concentrations greater than 1 mg/L NO₃-N remained. Denitrification was probably not occurring near the house, since anaerobic conditions are required for this process to proceed.

At depth, she found there were low nitrate values and elevated $^{15}$N (stable isotope of nitrogen) values possibly suggesting denitrification. Tritium was used to determine if there were two different flow systems present in the aquifer. The tritium values revealed that the groundwater was of relatively the same age. Therefore the idea of more than one flow system present was eliminated; nitrate stratification had evolved within one flow system.

Svensson also compared sodium and nitrate profiles from near the house with those at a distance away from the house in the farmer's field. Sodium was tested since it is commonly enriched in septic system plumes. The maximum concentration of sodium near the house was 32 mg/L, whereas the maximum concentration in the field was 8.6 mg/L. The sodium concentration values in the field were assumed to represent background values for the aquifer. Therefore, sodium may be used to differentiate between septic and agricultural contamination. For both areas of study (near the septic field and in the farmer's field) the maximum concentrations of nitrate were at a greater depth than the maximum sodium concentrations.

3.0 ASPECTS OF GROUNDWATER CONTAMINANT TRANSPORT

3.1 Contaminant Transport Processes

For the proposed study, several chemicals in the groundwater will be analyzed. To effectively analyze the results, a good understanding of contaminant transport processes is required. A thorough knowledge of this aspect would help explain why some contaminants have moved further than others in the plume, and why their concentrations may be different than source concentrations.
Three aspects that effect the movement of contaminants in groundwater are dispersion, adsorption, and biodegradation. To predict the movement of contaminants in groundwater, the aquifer is thought of as a two phase system in which contaminants partition between immobile solid constituents and the mobile aqueous phase. As stated earlier, sodium chloride (NaCl) can be used as a conservative tracer, since both of these ions (Na⁺, Cl⁻) are only weakly sorbed if at all, and therefore move at the same rate as the groundwater. Sodium and chloride move under the processes of advection (Freeze and Cherry, 1979). Contaminants that are sparingly soluble in water have a high tendency to sorb or precipitate on to aquifer media and are therefore retarded (to move slower than the rate at which the groundwater flows). Colloids are particles with a diameter less than 10 μm and are able to suspend freely in water. They are believed to facilitate contaminant transport (Gshwend and Ryan, 1990). A variety of organic and inorganic materials exist as colloids in groundwater including macromolecular components of dissolved organic carbon.

Diffusion is a process where ionic or molecular constituents move under the influence of their own kinetic activity in the direction of their concentration gradient. If the solution is flowing, diffusion and mechanical dispersion lead to the mixing of the constituents in the groundwater (Freeze and Cherry, 1979). Mechanical dispersion results from changes in the groundwater velocity due to different layers within the aquifer and variations in pore geometry (Fetter, 1994). For non-reactive solutes in a homogeneous, isotropic medium under steady-state uniform flow the coefficient of hydrodynamic dispersion ($D_l$) can be described by the following equation (Freeze and Cherry, 1979):

$$D_l = d_i \times v_x + D^*$$

where

- $d_i = \text{dynamic dispersivity}$
- $v_x = \text{average linear groundwater velocity}$
- $D^* = \text{coefficient of molecular diffusion}$

Dispersion is normally stronger in the direction of flow (longitudinal dispersion) than in the direction perpendicular to flow.
Adsorption is the accumulation of an ion at the surface of a solid and causes a retardation. Generally, adsorption related forces are weak, making the process reversible. Nitrate ion is too large to be effectively adsorbed by sand particles (Fetter, 1994). The adsorption of solute contaminants causes a decrease in the liquid-phase concentration without a loss of the total mass of the contaminants. This results in the retardation of the contaminants relative to the groundwater flow.

Biodegradation is the biologically mediated conversion of a compound to more simple products. Many environmental conditions influence the type and rate of the reaction: pH, temperature, state of oxidation or reduction, microorganisms present, and chemicals present (Fetter, 1994). The degradation processes of nitrogen compounds were discussed in Section 3.1 of this paper.

3.2 Delineating The Source Of Contamination

In this project delineation of the extent of contamination from each source was attempted. This would be to differentiate between contamination due to agricultural activities, sewage disposal and road salting. The parameters used are: sodium, chloride, and phosphate concentrations, electrical conductivity (EC), and nitrogen isotopes. For all three techniques, it is important to measure background levels as a means of comparison.

3.2.1 Sodium, Chloride, and Phosphate Concentrations

Robertson and Cherry (1992) noted that a septic system may be delineated by monitoring relatively non-reactive effluent constituents such as Na⁺ and CF. These chemicals are commonly encountered in septic systems at enriched values (20 - 100 mg/L) (Svensson, 1996). Using sodium to differentiate between agricultural and septic contamination was discussed earlier (see Section 5.0). The main problem with using these parameters is that salt applied to the roads during winter may also increase their concentrations. Phosphate is generally limited to septic system contamination and is not often found as a result of agricultural contamination in this area (Robertson, personal communication,
Therefore to delineate between these two plumes phosphate would be the most appropriate contaminant species.

3.2.2 Electrical Conductivity

Electrical Conductivity (EC) is a measure of the ability of a solution to conduct an electrical current. Since, an electrical current is transported by the ions in a solution, the conductivity increases as the concentration of ions in the solution increases (Tchobanoglous and Schroeder, 1987). Electrical conductivity is commonly measured in microsiemens per centimeter (µS/cm). High levels of groundwater EC can be used to identify a human or animal source of contamination (e.g., septic plume) from background waters. Since, the farmer in the area under study does not use manure as fertilizer for his fields, the high electrical conductivity values will be a result of the septic plume. Robertson and Blowes (1995) found the EC values to be two to five times higher than the background levels in the area of septic contamination. Stuart et al. (1995) also found that wells with the highest nitrate contents also had relatively high specific conductivity values. Electrical conductivity is a useful tool; it can be measured directly in the field with a portable metre. This helps to make decisions of where to drill next and allows for the plume to be outlined prior to the installation of permanent monitoring devices.

3.2.3 Nitrogen Isotopes

Isotopes are atoms whose nuclei contain the same number of protons but a different number of neutrons. Differences in nitrogen isotope values in shallow groundwater may be used to determine the source of nitrate for unconfined aquifers recharged by precipitation (Svensson, 1996). As nitrogen moves through the unsaturated zone, it can be isotopically fractionated by biological, chemical and physical processes. The processes of nitrification, denitrification, ammonia volatilization, and mineralization are the main processes that lead to fractionation (Lindau and Spalding, 1984).
Kreisler (1979) used the ratio of two stable isotopes, $^{15}\text{N}$ (denser) and $^{14}\text{N}$ (lighter isotope), of nitrogen in nitrate to distinguish between groundwater nitrate from cultivation and nitrate resulting from animal waste material. The ratio of these two stable isotopes is presented as $^{8}\delta^{15}\text{N}$ in accepted isotopic terminology, and is calculated as follows:

$$
\delta^{15}\text{N} \, (\%) = \left( \frac{^{15}\text{N} / ^{14}\text{N(sample)} - ^{15}\text{N} / ^{14}\text{N(standard)}}{^{15}\text{N} / ^{14}\text{N(standard)}} \right) \times 1000
$$

Mass spectrometry is used to determine the $\delta^{15}\text{N}$ values for nitrate and ammonium. Nitrate derived from different sources will have a corresponding range of $\delta^{15}\text{N}$ values. Nitrate from the decomposition of animal waste sources (organic) have a characteristic isotopic composition ($\delta^{15}\text{N}$) of approximately +9 to +22%. Commercial fertilizers (inorganic) have a $\delta^{15}\text{N}$ range of -7.4 to 6.8% (Lindau and Spalding, 1984). Similar values were determined by Kreider (1979). He also determined that nitrate originating from non-fertilized cultivated fields had a $\delta^{15}\text{N}$ range of +2 to +8% with an average of 4.9%. Commercial fertilizers lead to higher $\delta^{15}\text{N}$ values than non-fertilized fields because the volatilization of ammonia depletes $^{15}\text{N}$ during and after fertilizer application, and this leaves nitrate enriched in $^{15}\text{N}$ (Kreider, 1979).

This technique of using nitrogen isotopes, although quite accurate is expensive and large sample volumes are required. For a project with a large budget this would be an ideal technique to use to determine the source of nitrate.

### 3.2.4 Chlorine Isotopes

There are two stable isotopes ($^{37}\text{Cl}$ and $^{35}\text{Cl}$) of chlorine. The dominant ion is $^{35}\text{Cl}$ with an abundance of 75.77% (Eggenkamp, 1994). Chlorine interacts little with its surrounding environment leading to its highly conservative behavior. Therefore, significant chlorine variation in natural environments does not occur (Kaufmann et al., 1984). The small variation that does occur is because the heavier isotope, $^{37}\text{Cl}$ has a slightly smaller diffusion coefficient (1.0012) than the lighter isotope, $^{35}\text{Cl}$ (1.0022) (Eggenkamp, 1994). Due to this small variation in the isotopic composition, the actual
isotope ratio is not measured, rather the isotope ratio of the sample and a standard is measured. This difference is expressed as per mil as follows (Eggenkamp, 1994):

\[
\delta^{37}\text{Cl} = \frac{(R_{\text{sample}} - R_{\text{standard}})}{R_{\text{standard}}} \times 1000
\]

In the above equation R is the isotope ratio between \(^{37}\text{Cl}\) and \(^{35}\text{Cl}\). Since the isotopic composition of seawater is constant it is the accepted standard and referred to as SMOC (Standard Mean Ocean Chloride). The value of \(\delta^{37}\text{Cl}\) is the deviation from sea water; a sample that is depleted in \(^{37}\text{Cl}\) will have a negative value of \(\delta^{37}\text{Cl}\), and a sample with a positive value will be enriched in \(^{37}\text{Cl}\) (Eggenkamp, 1994). The University of Waterloo's Inorganic Geochemistry Lab uses SMOC as its standard (Flatt, 1997).

Kaufmann et al. (1984) showed that the ratio of the two chlorine stable isotopes can be used to indicate sources of groundwater chloride. This is possible since the chemistry of chlorine limits chlorine isotope variation in most low temperature environments to diffusion. Kaufmann et al. (1984) found variations in chlorine isotope compositions in hydrothermal springs, deep and shallow groundwater, and deep brines.

Different sources of chloride at a site may have different isotopic signatures. Therefore, the isotopic signature of road salt may be different than that from water softener salt. These differences may be due to the different sources and/or the different manufacturing processes used.

### 3.3 Groundwater Contamination Monitoring Devices

The main objectives of this study is to demonstrate that the Waterloo Profiler is an appropriate tool to efficiently map and characterize an inorganic plume. The Waterloo Profiler will be compared with two conventional monitoring techniques (piezometer and multilevel point samplers) to demonstrate its usefulness.
3.3.1 Waterloo Drive-Point Profiler

The Waterloo Profiler allows driving, purging, and sampling to be accomplished quickly. The main purpose of drive-point profiling is to obtain water samples from discrete depth positions so that a vertical profile of water chemistry (concentration versus depth profiles) at each monitoring location can be obtained.

The following information regarding the profiler has been obtained from Pitkin et al (1994). The profiler includes a stainless steel drive point with circular ports covered with stainless steel screen (Figure 1). The ports empty into a reservoir in the profiler tip. Stainless steel tubing attached to the reservoir is a conduit for water from the ports to the ground surface.

The lightweight version is advanced using a Bosch electric jackhammer, and the drill-rod version is advanced using a pneumatic piercing tool (air hammer) (Figure 2). The work area consists of three tiers of scaffolding sitting on two sets of wheels.

During sampling groundwater is extracted from the profiler using a peristaltic pump (Figure 3). Therefore, sampling needs to be done where the water table or potentiometric surface is within suction limit (8-9 m). This will not be a problem in the case of this project because of the shallow water table and aquifer at the farm site. By sampling from a column of water that is not in contact with gases, loss of volatiles due to application of suction can be reduced.

If the hydraulic conditions in which drive point profiling is feasible, it provides many advantages over conventional monitoring techniques. These advantages are as follows: minimal drill rig costs, no drill cuttings at surface and therefore no handling and disposal costs, little formation disturbance of geochemical conditions during installation, minimal purging for sample collection and therefore little cost of disposing of purge water, avoidance of using heavy equipment, and detailed chemical profiles at minimal cost per data point (Cherry et al., 1993). The major disadvantages of the Waterloo Profiler are samples can not generally be collected beyond nine meters, if transient
conditions of a system are desired permanent monitoring devices need to be installed (e.g., piezometer and multi-level wells), and it is limited to fairly permeable sediments.

Broholm et al. (1994) used a drive-point profiler for the detailed profiling of volatile organic carbon (VOC) plumes in groundwater aquifers. They found profiling with the single stainless steel tube profiler for suction sampling to be very effective in heterogeneous aquifers with permeabilities varying between 0.002 and 0.031 cm/s. The screen became blocked when the profiler was driven through the silt layer. Forty eight sampling points were obtained in approximately 12 hours to a depth of 14 m. This corresponds to the time required for the installation and sampling of a single well of the same depth.

Pitken et al. (1994) used a drive-point sampling device for the detailed characterization of PCE plume in a sand aquifer. They installed conventional monitoring wells after obtaining the solute concentration distributions provided by the profiler. However, with this prior knowledge in hand and the placement of the well screens in the areas of highest concentration, the wells were not able to provide the quantity or quality of information that was obtained by the multi-level point sampling device. A bailer was used to collect the water sample from a monitoring well, and in every case the highest concentration was obtained with the use of a bailer than with a peristaltic pump (used with drive-point profiler). A peristaltic pump when compared to bailing results in a negative bias with a mean of 28.3%. Even with these conditions the profiler provided more information on the distribution of contaminants than would be possible with conventional techniques. The cross-contamination from one depth to another was negligible using the profiler. Cross-contamination at low concentrations (lower portions of the profile hole) is unclear.

With the use of a drive-point profiling technique one is able to spend a relatively short amount of time in the field and have an adequate three dimensional outline of the plume. Therefore, money is not lost trying to find the plume with the more expensive conventional monitoring techniques. They
can be installed after the initial profile with the knowledge of where it would be most beneficial to install them.

3.3.2 Standpipe Piezometer

A piezometer is a non-pumping well, commonly of a small diameter, and is used to measure the elevation of the water table or potentiometric surface (Fetter, 1994). Most standpipe piezometers have a short screen tip through which water can enter (Figure 4). A screen for a piezometer tip generally constitutes slots along the side of the pipe (e.g. PVC) and then it may be wrapped with a fiberglass or nylon filter cloth to ensure that fine silt or sand will not accumulate in the piezometer (Cherry et al., 1983). Coarse-grained sand or gravel serves as a filter around the piezometer tip and is usually topped with a fine grained sand layer. Bentonite clay lies above the sand layer to hydraulically isolate the filter. The remainder of the hole may be backfilled with cement or cement-bentonite grout. It is important to isolate the tip since water pressures can vary from one horizon to the next (Franklin and Dusseault, 1989).

Piezometers are most useful in determining the depth to the water table in order to define the flow field. A study completed by Cherry et al, 1983 in Borden, Ontario found the piezometer served three main purposes: (1) hydraulic head measurements at discrete depths in the aquifer; (2) to complete rising or falling head tests to obtain hydraulic conductivity measurements; and (3) collection of samples for chemical analysis. However, the piezometer had quite a high cost per measurement point. It also does not allow for the collection of samples from numerous depths below the water table. For this to be possible, standpipe piezometers would require many boreholes at each site and a lot of PVC piping creating higher costs.

The main advantage of the standpipe piezometer is that it can be used for water-level response tests to determine the hydraulic conductivity of the aquifer. One standpipe in a borehole has a geometry of the intake zone that is suitable for hydraulic conductivity tests. The ability of a piezometer to
accurately measure the hydraulic conductivity of an aquifer depends on the amount of disturbance of the borehole surrounding the piezometer tip and the efficiency of the piezometer screen (Cherry et al., 1983).

As stated earlier, the main disadvantage of standpipe piezometers is that samples can not be collected at numerous points. Concentrations of contaminant in groundwater determined from piezometer samples are often averaged over the collection depth. This averaged result shows the peak concentrations of the contaminants being closer to the source than it really is in the field.

Robertson et al. (1991) installed a network of water table piezometers in the initial field investigation of two small septic systems on sand aquifers to determine the regional groundwater flow. Once the regional groundwater flow direction was known, detailed groundwater monitoring networks were installed consisting of multilevel samplers.

3.3.3 Multilevel Point Sampler
Multilevel sampling devices are installed in a single borehole to sample groundwater from different depths (Figure 5). There are two main types of multilevel samplers: suction-type and positive displacement. The suction-type will be discussed in this paper since it is more common. The device consists of a rigid PVC pipe inside of which are multiple tubes, each of these end at a sampling port at a different depth (Fetter, 1994). Samples can be collected as close as 30 cm apart. Since, only a small sample of water is withdrawn, the sample represents water from a very small portion of the aquifer. They provide a detailed picture of the vertical distribution of the groundwater contamination. It is installed by augering a boring to the desired depth using hollow stem. PVC pipe is lowered through the hollow stem and as the auger is removed sand collapses in the space around the pipe and seals it off (Fetter, 1994). One major disadvantage of the multilevel device is that water levels can not be measured. Therefore, it is difficult to determine the hydraulic gradient and hydraulic head distribution. It is generally restricted to sand and gravel aquifers and has a suction
limit of approximately 9 m. If contaminant concentrations are required at a greater depth, the positive-displacement multilevel sampler may be employed.

Cherry et al. (1983) installed a very detailed monitoring network in Borden to investigate the hydrogeochemical and dispersion characteristics of a plume. The construction and installation of multilevel samplers required many hours. Since, a drill rig was also required the costs of this technique were very high. Samples were obtained by applying suction to each tube, draining water from the tube, and then collecting the sample.

4.0 METHODS AND MATERIALS

4.1 Site Description
The farm site located near Delhi, Ontario (Figure 6) was selected based on a research project conducted by Svensson (1966). She found the site to have highly permeable soils, high nitrate concentrations, a septic system, and agricultural activities (inorganic fertilization and irrigation). A combination of all these conditions made it susceptible to nitrate contamination in the drinking water supply.

Figure 7 illustrates the layout of the site. The site includes a residential home with an active septic system. Drinking water is derived from a well approximately 4 m deep located in the basement of the house. The office building also contains a well within it. There are also two greenhouses. Beside the office is an inactive barn and barn yard where cattle were maintained until 1974. Adjacent to the greenhouses are cultivated fields which are cropped to tobacco and rye on a rotating basis (Svensson, 1996).

The loading to the septic system is quite low since the residence has only one occupant. Therefore, the total demand for water is quite low. Water for irrigation is provided by a creek that is sustained by groundwater, located approximately 100 m south of the inactive barn. This creek is also located
105 m west of the residence (running north-south), but changes direction in the south-west corner of the site to run west-east. Considering that the creek is a discharge point, it is believed that the direction of the hydraulic gradient is south-west to south.

4.1.1 Aquifer Properties

Svensson (1996) performed sieve analyses, falling head permeameter tests, and a pumping test to determine the physical and hydraulic properties of the aquifer material at the study site. The sieve analyses showed that the majority of the aquifer material consisted of sand with a diameter between 0.2 and 0.6 mm. Hydraulic conductivity measurements were performed on two cores: core 1, 5 m from ML3 (Figure 7), and core 2, beside ML2 (Figure 7). She found the most permeable zone to be 4-8 m below ground surface with an average hydraulic conductivity of 3.7 x10^{-4} m/s. Beside the house, the minimum hydraulic conductivity was observed at the water table with a value of 3.6 x 10^{-6} m/s. The core taken beside the field showed that the minimum hydraulic conductivity was 6.3 x10^{-5} m/s, 0.5 m below ground surface. The average hydraulic conductivity determined using the Neuman method was determined to be 4.6 x10^{-4} m/s. The hydraulic head data from the multilevel monitoring wells indicated that the average horizontal hydraulic gradient was 0.003 m/m, ranging from 0.0014 to 0.004 m/m. Due to the low hydraulic gradient however, it was difficult to determine an exact groundwater flow direction, due to inherent water level measurement error.

4.2 Instrumentation and Field Investigation

The farm site already had four multilevel monitoring wells present. These wells were located in the tobacco field, on the edge of the field, and beside the domestic well (Figure 7). All multilevel domestic wells were monitored except the one in the tobacco field.

A hand auger was used to auger down to the water table and retrieve soil samples; thus allowing the shallow stratigraphy of the area to be outlined. The lightweight version of the Drive-Point Profiler was used to determine the distribution of the inorganic plume. A total of 12 profile holes were driven
Water samples were collected at 0.4 metre intervals except at H1 (0.2 and 0.3 metre intervals) and H4 (0.3 and 0.4 metre intervals). Profiling was completed to a depth of 2.8 m below ground surface except for H1 (3.8 m), H6 (3.8 m), H7 (3.2 m), H10 (3.6 m), and H13 (3.6 m). A total of 64 samples were collected during the two day field investigation period (including nine samples collected from multilevels).

The lightweight profiler was advanced using a Bosch electric jackhammer. The profiler was driven with ease due to the unconsolidated material of the sand material. After the collection of a sample, water was pumped through the ports to ensure that the ports did not clog and to prevent cross contamination between samples. Distilled water was purged through the ports prior to sample collection. The purged water was continuously tested for electrical conductivity (EC) and pH using field-portable meters (Table 1); when these values stabilized the sample was collected. Allowing these values to stabilize ensured that only formation groundwater was collected, not distilled water. The samples were filtered through a 45 µm filter and preserved with sulfuric acid immediately after collection. The samples that had the peak EC values were sent to MDS Laboratories, Halifax and analyzed for nitrate, ammonia, sodium, chloride, and phosphate. The remaining samples were analyzed at the Waterloo Water Quality Lab, for nitrate and chloride (Table 2).

It was found that the time required to collect a sample was approximately 10 minutes, and this included driving, purging, and sampling. Fifty-five samples were collected using the profiler in two days and this included driving out to the site (1 hour), setting up, sampling, moving between profile holes, and packing up the equipment.

Three samples were collected from each multilevel monitoring well. They were sampled between 1 and 4 metres below ground surface (bgs). Electrical conductivity measurements were also taken from the nearby creek. These measurements reflect the background levels outside the area of the septic system.
A preliminary analysis of the geochemical data showed that several sources of contamination may exist at the site. Therefore, it was decided to return to the field site and collect samples for chlorine isotopes to further help delineate the source of contamination. Three samples were collected from two areas due to their proximity to a potential chloride source, and one from an area where more than one type of source was present. One litre samples were collected. Samples were obtained from ML3 at 2.3 m depth, ML2 at 3.0 m depth, and H13 at 2.8 m depth. The sample from H13 was collected by using the profiler and driving it down manually with a monkey bar. All three of these samples were analyzed for chloride prior to chlorine isotope analysis. It was decided not to analyze the sample from ML3 for chlorine isotopes, but rather a sample near the septic field. Sample H4 at 2.6 m was selected due to its high chloride concentration (185 ppm). This sample was collected during the initial investigation, a 1 L sample was not required in this case due to the high concentration of chloride (185 ppm).

5.0 RESULTS AND DISCUSSION OF FIELD INVESTIGATION

5.1 Geological Information

The study site is located near Delhi, Ontario on the Norfolk Sand Plain. The sand plain is relatively flat and has a topographic gradient of less than 0.25% (Svensson, 1996). The surficial sediments consist mainly of top soil (approximately 30 cm), followed by a fine sand layer, then a coarse sand and gravel layer, and coarse sand to the water table and to greater depths (Figure 9). Hydrochloric acid (HCl) showed that carbonate minerals were present in the coarse sand layer. This coarse grained sand layer is of glaciolacustrine origin and averages 9 m thick in the general area of the study site (Svensson, 1996). The coarse grained sand and gravel layer was only found in the vicinity of the septic field; it is presumed to be a backfill material for the septic system. The depth to the water table was between 1.3 and 1.4 m in the area of the septic field.
5.2 Geochemical Investigation

The geochemical data serves two main purposes: to determine the source and persistence of nitrate and chloride at the farm site and to evaluate the water quality. Vertical profiles of nitrate and chloride vs. depth were produced to aid in the analysis as were plan views and x-sections of the nitrate and chloride concentrations.

The peak concentrations for both nitrate and chloride were generally between 2.0 and 2.4 m below ground surface (bgs) in the area of the septic field (Figure 10). Near the agricultural field the nitrate concentrations were quite consistent with depth, being between 5 and 10 mg/L (Figure 11). This was not the case with the chloride concentrations near the agricultural field. The peak occurred at a depth of 3 min ML2 (42.7 mg/L) with the concentrations increasing with depth from 1 m; in the case of ML4 the high chloride levels were at 2.0 bgs (49.4 mg/L) with the concentrations decreasing with depth. The profile hole closest to the creek (H6) revealed high nitrate (9.80 mg/L) and chloride (105 mg/L) levels 3.6 m bgs and the concentrations generally increased with depth (Figure 12). The profile hole (H13) closest to the road showed very high nitrate (35 mg/L) and chloride (167 mg/L) concentrations 2.8 m bgs (Figure 12). Nitrate (25.5 mg/L) and chloride (194 mg/L) concentrations near the office (H7) peaked between 2.5 and 3.0 m below ground surface and generally decreased with depth (Figure 13).

A large variation of these peak concentrations with depth may be due to the several processes acting on the nitrate and the potential different sources of nitrate present in the area. The many potential sources of chloride may also cause the varying chloride levels with depth. These ideas will be discussed in more detail in the following sections.

5.2.1 Nitrate Distribution and Behavior

Generally, the highest nitrate concentrations were in close proximity to the septic field, except H13. The NO$_3$-N values ranged from 13 to 29 mg/L between a depth of 2.0 and 2.8 m. The deepest samples were collected up to a depth of 3.8 m and the NO$_3$-N values were 12.2 and 22.2 mg/L, for
H1 and H4 respectively. Although these average concentrations have decreased with depth; they are still above the drinking water standard (10 mg/L), and consequently pose a problem to the well supply. This is of concern since the drinking water is derived from a well approximately 4 m deep.

Nitrate concentrations versus depth in H4 (Figure 10) shows that the nitrate values are approximately 22 mg/L between 2 and 2.5 m, quickly fall to 10 mg/L between 2.5 and 3.5 m, and then increase back to 22 mg/L between 3.5 and 4.0 m. This abrupt decline in NO₃-N may be due to localized denitrification within isolated anaerobic zones. In H1 it is noticed that the NO₃-N concentrations steadily decrease to about 11 mg/L. Since nitrate does not readily adsorb to soil particles, the main processes acting on the nitrate should be advection, dispersion, and diffusion. These are generally the only processes that apply to chloride due to its conservative nature, thus by comparing the chloride concentrations versus depth to that of nitrate it may indicate if other processes are acting on the nitrate. Since in H1 both the nitrate and chloride concentrations generally decrease with depth, the same processes are leading to their decreasing concentrations. namely advection, dispersion, and diffusion. However in H4 the chloride concentrations eventually do decrease with depth, the nitrate levels do not. This may be due to the persistent nature of nitrate.

To better understand the effects of the different sources on the nitrate levels, plan views were produced at depths of 2.0 m (Figure 14), 2.4 m (Figure 15), and 2.8 m (Figure 16). Two cross sections were also produced (Figures 17 and 18). See Figure 8 for the location of the x-sections. The background levels near the agricultural field were between 6 and 9 mg/L NO₃-N. These high levels are mainly due to the application of fertilizer and irrigation. Generally, background levels are between 2 and 3 mg/L NO₃-N. A combination of highly permeable soil and the absence of denitrification may lead to the higher background levels present.

The plan view at a depth of 2.8 m (Figure 16) suggest that possibly four sources of nitrate exist at the site: in the area of the septic field, near the road, close to the office, and high background levels (i.e. the agricultural field). Figure 17 also clearly portrays the first three sources of nitrate listed.
This figure shows contamination coming from the side of the road, but it shows up at a greater depth (approximately 2.6 m bgs). Nitrate contamination from a source a fair distance from the study site would be at greater depth, since the plume of contamination would move deeper as it proceeded along its flowpath (Robertson, personal communication, 1997). Therefore it is presumed that this source of nitrate is either from the septic system or the agricultural field of the farmer across the road, or is a combination of both. The persistent high nitrate concentrations at a greater depth in the area of the study site may be due to the agricultural practices of the neighboring farmers and study site along with the absence of denitrification with depth. The high nitrate levels near the office may be due to a second small septic system placed near the office building. Although farm animals were kept in the inactive barn beside the office until 1974; they should no longer affect the nitrate levels in the region due to the relatively high velocities at the site (Svensson, 1996).

These many sources of contamination from distal and proximal sources lead to local nitrate stratification. Non-point sources of nitrate distributed over a large area produce a plume that underlies or overlies a plume from a point source.

5.2.2 Chloride Distribution and Behavior

The peak chloride levels were in the vicinity of the septic field. The peak values were between 1.9 and 2.6 m bgs and ranged between 124 to 195 mg/L. The deepest samples were collected up to a depth of 3.8 m and the chloride values were 28.4 and 52.1 mg/L, for H1 and H4 respectively. These average concentrations have decreased rapidly with depth; they are below the drinking water standard (200 mg/L) and consequently of little concern since the drinking water is derived from a well approximately 4 m deep.

Chloride concentrations versus depth in H4 (Figure 10) show that the chloride values are less than 6 mg/L above 1.7 m, approximately 170-180 mg/L between 2.0 and 3.2 m, and then quickly fall to 50 mg/L beyond 3.5 m. This abrupt increase in chloride may be due to the impact of a water softener...
at the farmhouse. The owner uses a water. Therefore these chloride levels are further elevated due to the presence of a water softener, which becomes another potential source of contamination of groundwater in the area surrounding the well. Essentially the well is recirculating this softened water that has high chloride levels back through the water softener. Since salt is used to flush the resin in the softener, the possibility exists that the tap water is high in sodium chloride. This chloride profile in H4 clearly outlines the thickness of the chloride plume to be approximately 1.2 m. The distribution of the chloride plume is much greater in the longitudinal direction than the transverse direction, therefore little mixing with non-impacted waters is occurring in the transverse direction.

In H1 it is noted that the chloride concentrations steadily decrease with depth. Chloride is a conservative tracer; the main processes acting on the chloride should be advection, dispersion, and diffusion. These processes will decrease the concentration of chloride laterally and longitudinally. Although both H1 and H4 are in close proximity the behavior of chloride in these two profile holes is quite different. It is believed that H4 may be closer to the main area of sewage loading leading to a thicker chloride plume in this profile hole.

To better understand the effects of the different sources on the chloride levels, plan views were produced at depths of 2.0 m (Figure 19), 2.4 m (Figure 20), and 2.8 m (Figure 21). Two cross sections were also produced (Figures 22 and 23). See Figure 8 for the location of the x-sections. The background levels near the agricultural field were between 4 and 49.4 mg/L Cl, and are largely due to the application of fertilizer (namely potassium chloride) and irrigation. Chloride levels at the shallowest depth tested, 2.0 m (Figure 19) displayed high levels around the septic bed with concentrations decreasing outwards in all directions. At a deeper depth of 2.4 m (Figure 20), the chloride concentrations in the vicinity of the septic bed decrease, but increase near the road and the office. At a depth of 2.8 m (Figure 21), the chloride concentrations further decrease in the vicinity of the field while there is a drastic increase in the chloride levels near the road. These high chloride levels near the road may be due to fertilizer being applied at the farm across the road and a septic
system across the road, thereby coinciding with the high nitrate concentrations at this approximate depth. However heavy fertilizer application from a distant source would not solely lead to the heightened chloride levels, therefore it is believed that a combination of both road salt and fertilizer application have led to the high chloride concentrations. The deeper plume mainly due to fertilizer application and a septic system across the road can be easily differentiated from the septic chloride plume in Figure 22.

A combination of the many sources of chloride: septic system, fertilizer application, water softener, and road salt in a farm setting have the potential of leading to chloride levels that are above the drinking water standard of 250 mg/L. Also, over time the plume near the road will move southward in the direction of the well, and since this plume is deeper it provides the possibility of further increasing the level of chloride in the well drinking water.

5.2.3 Phosphate
Phosphate (PO$_4^{3-}$) levels are expected to be higher in water impacted by human waste and washwater than in background levels (Harman et al., 1996: Robertson, 1995). At the study site no detectable phosphate (<0.02 mg/L P) was found at depth (Table 2). At shallow depths however (1.4 - 1.7 m), the amount of phosphate (as P) was between 0.01 and 0.58 mg/L in the area of the septic field. These higher levels of phosphate are indicative of sewage wastewater. At depths greater than 2.0 m the greatest amount of phosphate was 0.02 mg/L at the study site. The behavior of phosphate is mainly controlled by sorption, therefore much of the phosphate has been attenuated in the unsaturated zone and the shallow portions of the saturated zone. Robertson (1995) found that the initial depletion of phosphate was due to adsorption, and the steady-state concentrations were due to additional mineral precipitation reactions.
5.2.4 Ammonia and pH

The highest concentration of ammonia - as N (NH₄⁺-N) at the field site was 0.05 mg/L. Therefore either oxidation of NH₄⁺ to NO₃⁻ is occurring, or sorption of NH₄⁺ (Harman et al., 1996). The high levels of nitrate and the corresponding low levels of ammonia at the site would suggest that the oxidation of ammonia (as N) is the dominant process. This reaction is as follows:

\[ \text{NH}_4^+ + 2\text{O}_2 = \text{NO}_3^- + 2\text{H}^+ + \text{H}_2\text{O} \]

The requirement of dissolved oxygen ensures that the majority of the reaction is carried out in the unsaturated zone. The above reaction increases the acidity of the water through the production of hydrogen ions. However the pH values in the area of the septic bed remain near neutral (Table 1), ranging from 6.56 to 7.13. The buffering capacity of carbonate minerals in the soil can maintain a pH close to 7 by the following reaction:

\[ \text{CaCO}_3 + \text{H}^+ = \text{HCO}_3^- + \text{Ca}^{2+} \]

Also, a pH decrease can be buffered by the alkalinity in the effluent (Robertson, 1997). A combination of these two processes can stop the development acidic conditions in the groundwater plume.

5.3 Delineating The Source of Contamination

5.3.1 Chlorine Isotopes

Three samples were analyzed for chlorine isotopes: in the area of the septic field, beside the agricultural field, and near the road. Chlorine isotope analysis was conducted to help delineate between the three major sources of chloride: water softener salt, road salt, and fertilizer. The sample taken from the septic field (H4-2.6m) had $\delta^{37}\text{Cl} = -0.03 \%$, the sample taken from beside the agricultural field (ML2-3.0m) had the same value for $\delta^{37}\text{Cl} (-0.03 \%)$. H13-2.8m (near the road) had $\delta^{37}\text{Cl} = -0.09 \%$. Therefore all samples are slightly depleted in $^{37}\text{Cl}$ with respect to sea water, although this deviation was very small.
A summary of chlorine isotope data (Table 3, Frape, 1997) is a means of comparison. The water softener salt values in this table are similar to the sample taken from the septic field (close to zero). The values of $\delta^{37}\text{Cl}$ for road salt are also near 0.0, and consequently compare well with the sample obtained near the road. It is difficult to positively identify the source of chloride contamination using chlorine isotopes in this case because the isotopic signatures for the different sources are quite similar. Because mixing of the sources of chloride may also be occurring to some degree it is difficult to identify a source with certainty. Also if different types of road salt, fertilizer, or water softener salt were used there will be a mixing of different isotopic signatures.

5.3.2 Field and Geochemical Data

A summary of the geochemical data in the postulated major source areas are shown in Table 4. A comparison of the data shows distinct geochemical characteristics for the different sources of contamination. Electrical Conductivity, sodium, and chloride levels were the highest in the area of the septic system. Although, the peak nitrate (as N) level in the septic field was high (24 mg/L), it was much higher near the road (35 mg/L). In all instances except for phosphate and ammonium, the background (agricultural field) concentrations were much lower.

The alkalinity beside the agricultural field (193 mg/L as CaCO$_3$) was lower than the one near the road (281 mg/L as CaCO$_3$). The alkalinity near the septic bed, beside the house was 335 mg/L as CaCO$_3$. Generally the alkalinity of sewage impacted groundwater in calcareous terrain is around 350 (Robertson, 1997). In all natural waters the presence of carbonate ($\text{CO}_3^{2-}$), bicarbonate ($\text{HCO}_3^-$), and hydroxyl (OH-) ions account for essentially all of the alkalinity (Tchobanoglous and Schroeder, 1987). The groundwater was near neutral in all cases, therefore the high alkalinity in the wastewater sewage would be largely due to the concentration of the carbonate and bicarbonate ions.

Ratios of sodium and chloride (Na’/Cl’) were determined from the source data presented by Frape (1997) and compared with data at the study site (Table 3). Excluding the one extreme value for
fertilizers the following averages of the ratios were obtained: water softener salt (0.59), fertilizer (1.85), road salt-CaCl (0.03), and road salts-MgCl (0.43). The results collected from the data at the farm site (Table 4) at the peak EC values are: septic field (0.65), agricultural field (1.34), and near road (0.24). Peak EC conductivity values close to each source were used to help ensure that the data was not a result of a combined number of sources. The two sets of data correlate well, however the there is a lot of variability in the data by Frape (1997). The ratios for fertilizers and the agricultural field are above 1, and the highest of the three sources. In the case of the water softeners and the septic field, the ratios are very similar, 0.59 and 0.65, respectively. It was confirmed that the farm house has water softened sewage. Also these ratios are intermediate between the fertilizers and the road salts. Finally, the road salts have the lowest sodium to chloride ratios as seen in both data sets. Although it can not be concluded which type of road salt it is, it can be differentiated from the other two sources (agricultural and septic).

5.4 Profiling Technique

The main objective of this study was to demonstrate the Waterloo Profiler as a useful tool in determining the extent of inorganic constituents in rural settings. This tool is shown to be very useful in characterizing groundwaters around farm houses where there is a shallow water table, and permeable aquifer material. Densely packed or clay material would make it difficult to drive down the profiler with either an electric jackhammer or a pneumatic piercing tool. Also, clay particles have the potential of clogging the sample ports. Since many farm sites are on sandy material however, the device should find widespread application.

Using the profiler in the field was a relatively simple process. Heavy machinery was not required, and a team of two people could easily handle the operation of the profiler and the collection of water samples. Throughout the two day field investigation, no problems were encountered with the use of the profiler.
A total of 64 samples were collected during the two day investigation period, including 9 from the multilevels. During the field study electrical conductivity and pH measurements were taken, as well as hydraulic head measurements, and the general stratigraphy of the study site was determined. From the collected samples the major sources and extent of groundwater contamination at the farm site were determined. Interestingly it was not known that owner had a water softener, but it was with the use of the profiler that this was deduced. The Waterloo Profiler has the potential to reveal many of the groundwater problems that may exist in a rural area, in a short period of time, and at a low cost.

6.0 SUMMARY AND CONCLUSIONS
Groundwater quality in the rural community is declining with time as a result of the increased density of residents and fertilizer use. With the many potential sources of contamination present in a farm setting, farmers need a quick and inexpensive method of determining their water quality. The Waterloo Profiler provides this. In one day, profile holes can be driven around the septic field, nearby roads, and the agricultural field in order to obtain a general idea of the impact of these sources. The collected samples should be analyzed for nitrate, chloride, and sodium in particular, and also for other source indicators such as phosphate and chlorine isotopes.

Nitrate at the site was in almost all cases above the drinking water standard of 10 mg/L, and in some areas it was three times this value. These high nitrate concentrations persisted to depth, where the drinking water well was placed (4 m). The profiler revealed that high nitrate concentrations were present near the road, around the septic field, and beside the office. The background levels in the agricultural field remained around 10 mg/L. Various sources appeared to lead to nitrate stratification. The more distal sources (agricultural activity, and septic systems of neighboring owners) formed a plume of nitrate that underlayed the nitrate plume from the septic system around the area of the septic field. These deeper nitrate plumes are of more concern since they have a
greater potential of affecting the water supply wells, that were emplaced deeper in hopes of avoiding local contaminants.

The chloride concentrations at the site were found to be below the legal drinking limit of 250 mg/L. However, in the vicinity of the septic field some of the chloride concentrations approached this value. It is important for farmers to analyze for chloride, since it is a conservative tracer and provides an indication of the physical processes that may be occurring at the farm site. Also, many potential sources of chloride were found: agricultural activities, septic systems including water softeners, and road salt. Once again the more distal sources formed a plume which underlayed the local. Since there was very little mixing of water in the transverse direction, these plumes remain separated by less impacted water.

Many farmers are reluctant to analyze their water quality due to the associated costs and inconvenience. However this project has shown that the Waterloo Profiler can help rural residents recognize if the quality of water they are drinking meets regulatory requirements at a minimal cost. This project has also shown that further research needs to be done to find safer ways of on-site sewage disposal and agricultural practices.

7.0 RECOMMENDATIONS

Dissolved oxygen tests should be conducted to determine if an anaerobic environment exists. This would help determine if denitrification is possible. The Winkler Titration method can be done directly in the field at a low cost.

The effluent concentration from the septic system should be measured. This would provide an initial concentration of the constituents, and thus be a basis of comparison of how the concentrations have changed down gradient of the discharge point.
Chlorine isotope analysis should be conducted on the water softener salt, road salt, and fertilizer. By knowing their initial isotopic signatures, they can be better used as tracers.

Profiling should be completed once during the winter season and once during the summer season. If the profile holes are properly marked, they can be re-driven manually without the aid of electrical equipment. It is important to complete a profile in the summer months to determine the impact of the increased agricultural activity during this season. During the winter months, the impact of the increased road salt application can be deduced.

Farmers should be informed on how their agricultural practices and those around them are effecting the water quality of their region. This would allow the possibility to find measures that would increase the level of drinking water quality on a community basis.
REFERENCES


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Figure 23: X-Section across Septic Field of Nitrate Concentrations (mg/L)
### Table 1: Field Data

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Table 3: Chlorine Isotope Data (Frape, 1997)

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<tr>
<th>Water Softeners</th>
<th>Sodium (ppm)</th>
<th>Chloride (ppm)</th>
<th>Nitrate (ppm)</th>
<th>$^{37}$Cl</th>
<th>Na/Cl</th>
<th>NO$_3$/Cl</th>
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<tr>
<td>Sifto Crystal</td>
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<td>0.02</td>
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Table 3: Chlorine Isotope Data (Frape, 1997) (cont’d)

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<th>Road Salts (MgCl)</th>
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<th>Chloride (ppm)</th>
<th>Nitrate (ppm)</th>
<th>$^{37}$Cl</th>
<th>Na/Cl</th>
<th>NO3/Cl</th>
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Table 4: Summary of Field and Geochemical Data

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<td>EC (µs/cm)</td>
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