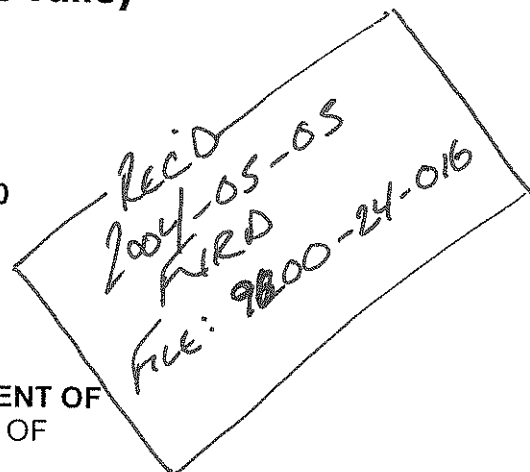


Groundwater and Surface Water Management and Drinking Water Issues in the Hatzic Valley

by
Simone Barbara Magwood
B.Sc., The University of Waterloo, 2000

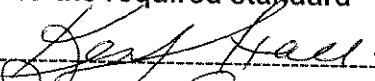


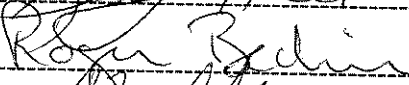
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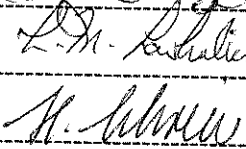
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Abstract

In rural areas, water is often supplied by private groundwater wells. Individual well owners have little control over the quality of their drinking water which can be impacted by adjacent land use.

The objectives of this project were to determine the quality of drinking water in a rural watershed, investigate the links between groundwater, surface water and land use, and compare the results with the residents' perceptions. Field research was conducted in the Hatzic Valley in the Lower Fraser Valley of British Columbia, a rural watershed with mixed land use. With the help of local volunteers, 75 groundwater samples were collected and analysed in July 2002 and March 2003. Stream water samples were collected at 19 stations semi-monthly between May 2002 and May 2003 to determine linkages between 1) land use and surface water and 2) surface water and groundwater quality. A spatial database was created and the percentage of the four predominant land uses were calculated within several radii of the wells sampled. The perceptions and opinions of local residents were collected using a questionnaire and the results compared with the water quality data.

Analysis showed that water quality throughout the valley is good; however, two wells exceeded the health standard for nitrate-N and 10 more wells had nitrate-N levels above 3 mg/L (considered indicative of land use impacts). All the wells with elevated $\text{NO}_3\text{-N}$ were shallow (less than 12 m deep). Percent forest cover surrounding the wells was negatively correlated with nitrate level, while percent urbanised land was positively related to nitrate. The cause of high nitrate in the urban areas is thought to be from septic systems. Surface water quality was also high but showed overall degradation from headwaters to the mouth as agriculture increased in the watershed.

The majority of residents perceived their water quality to be good or excellent. The residents base their perceptions on tangible indicators rather than chemically determined ones as those who have negative perceptions of their water tended to have high levels of iron and manganese rather than nitrate. The latter is detectable without chemical analysis while the former is not. Very few filter, treat or test their water quality on a regular basis.

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Finally, this research was inspired by and dedicated to the town of **Walkerton** and the people who live there. It was great place to grow up – those of us who lived there know it's so much more than just "*E.coli*".

*Those who fail to learn from the past are doomed to repeat it...
let's hope we've all learned a little bit.*

Abbreviations

ANOVA – Analysis of Variance

AQ1 – Aquifer 1. Nicomen Slough Aquifer

AQ2 – Aquifer 2. Hatzic Prairie Aquifer

AQ3 – Aquifer 3. Miracle Valley Aquifer

BC – British Columbia

BCCDC – British Columbia Centre for Disease Control

CAD – Computer Assisted Design

CFU – Colony Forming Unit

EC – Electrical Conductivity

FVRD – Fraser Valley Regional District

GCDWQ – Guidelines for Canadian Drinking Water Quality

GIS – Geographical Information System

LFV – Lower Fraser Valley

MAC – Maximum Allowable Concentration

MDC – Minimum Detection Concentration

NPS – Non Point Source

PHCL – Pacific Hydrology Consultants Ltd.

UK – United Kingdom

1.0 Introduction and Context

“Millions have lived without love, No one has lived without water”
Turkish Businessman (De Villiers, 1999)

“Unlike other important commodities such as oil, copper, or wheat, fresh water has no substitutes for most of its uses” (Postel et al., 1996).

The importance of water for human survival cannot be disputed. A human being can live for 40 days without food but only five days without water. We depend on clean water not only for basic survival but also use it in virtually every human endeavour. It is used for drinking, washing, cooking and cleaning, in industrial processes, to irrigate our farms and to generate electricity. Although water is vital for human survival it is one of our most abused resources (Brandling-Bennett, 1998). Many of the above activities are not consumptive, meaning that some or all of the water returns to the environment after it is used; however, in most cases the water that is discharged bears little resemblance to the water that was initially withdrawn. For example, human habitations discharge water that is contaminated with human waste. Many industries release water laden with chemicals, and when leaving a farm field, water that has been used for irrigation often has a high level of nutrients and pesticides. In almost every case water that is used by humans or passes through areas of human influence is degraded in some way. Since water quality is being degraded all over the planet it is challenging to find water that is safe for drinking. Many of the impacts on water quality are caused by non-point source pollution and are cumulative in nature -- the uncertainty surrounding the causes and sources of potential water quality degradation makes the delivery of safe drinking water a complex problem (Lichtenberg and Penn, 2003). Different areas face different challenges in providing safe drinking water. This project is an examination of the challenges facing a group of rural Canadians in providing themselves with safe drinking water. The focus of the project is on the interactions between rural drinking water quality, land use and individual perceptions of water quality.

Fieldwork was done in a watershed in British Columbia's Lower Fraser Valley. Using this watershed as a focus, the project will attempt to provide a comprehensive discussion of the problems facing, and options available to, rural residents who wish to protect their health and provide themselves with a safe source of water to drink.

1.1 Research Questions:

The purpose of this project is to answer the following questions:

1. What is the surface and groundwater quality in the Hatzic Watershed?
2. Can geological effects on water quality be separated from land use effects?
3. Is there a quantifiable link between groundwater quality and land use?
4. Is there a quantifiable link between surface water quality and land use?
5. Where are there links between groundwater quality and surface water quality?

1.2 Drinking Water Vulnerability in Canada and British Columbia

Drinking water in Canada is relatively safe. In many countries it is routine for people to die from drinking contaminated water. In Canada, however, it is a national tragedy and people are shocked and appalled if people die from drinking water. One such tragedy occurred in Ontario in 2000.

In May of that year, a waterborne outbreak of *Escherichia coli* 0157:H7 and *Campylobacter jejuni* in the town of Walkerton resulted in seven deaths and made 2300 citizens ill (O'Connor, 2002a). The Walkerton tragedy served as a catalyst for jurisdictions across Canada to examine how drinking water is regulated and delivered to the communities who consume it. This outbreak received a lot of attention because people died, however, it was not the first (nor the last) outbreak of waterborne disease to take place in Canada.

British Columbia (BC) alone has experienced 29 waterborne disease outbreaks since 1980 and consistently has the highest rate of enteric disease of any province in Canada (Provincial Health Officer, 2001). A report by BC's auditor general entitled "Protecting Drinking Water Sources" published in 1999 recognized the potential for rural (and urban) drinking water quality problems. The auditor found that there is inadequate protection of drinking water sources in British Columbia and that sources for individual and small water systems are especially vulnerable to the effects of surrounding land uses. Soon after, B.C.'s Provincial Health Officer (2001) issued a report highlighting the public health perspective on drinking water quality in B.C. This report also specified concerns about rural drinking water and highlighted the many knowledge gaps about rural systems. The findings of both reports were then echoed by the Panel Review on British Columbia's Drinking Water Protection Act (2002) which recommended that a new Drinking Water Protection Act "strengthen source protection measures" as well as "provide support to small systems".

The above reports have highlighted the need for more information to be gathered about rural water systems in British Columbia. Among the examples of "what we don't know" listed in the

Provincial Health Officer's report were questions such as "How many water system are there in B.C. today...what is the water source... and the population served by each system... what are the potential sources of contamination... how is the system maintained..."(Provincial Health Officer, 2001). There is currently no centralized data management system to record this information.

1.3 Multiple Barrier Approach

Since the Walkerton *E.coli* contamination in May 2000 much attention has been paid to the concept of using multiple barriers for protection of drinking water supplies (Provincial Health Officer, 2001; Kirmeyer *et al.*, 2001; O'Connor, 2002a, 2002b). Built in redundancy ensures that if one barrier fails there is another in place to prevent contamination. In his report to the Walkerton Inquiry Justice O'Connor outlines the multi-barrier approach to water treatment and delivery a framework that recognizes that delivery of safe drinking water must be an integrated effort. This framework, outlined in Table 1, reminds us that treating water before delivery to consumers might not be enough to ensure their safety. Attention must also be paid to the source of the water, including any watershed processes that might lead to degrading water quality, as well as to what happens to the water in the distribution system after treatment. The final aspect of the multi-barrier approach is the recognition that something could go wrong at all stages and therefore adequate monitoring should be in place to detect problems, as well as a system of response should be in place if all barriers fail (O'Connor, 2002a).

The process of supplying drinking water raises serious public health concerns. A properly structured system for ensuring the safety of drinking water should have multiple barriers: if one protective measure fails, there must be another to back it up. The experts that testified at the inquiry repeatedly emphasized the need for multiple barriers – that is for a robust system with built-in safeguards (O'Connor, 2002a).

Table 1. The Multi-Barrier Approach to Water Treatment and Delivery	
Source of the Water	Protection of source water quality by limiting or prohibiting wastewater discharges and other sources of water pollution
Water Treatment	Adequate treatment such as disinfection and/or filtration
Distribution System	Safeguarding water quality during storage and distribution
Monitoring of Water Quality	Monitoring of the distribution system and enforcement of standards.
Response to Adverse Test Results	If all of the barriers fail, what steps will be taken to protect the public?

Drinking water systems exist in many different forms. In Canada, cities and towns generally have centralized water distribution systems run by the municipality or some other form of public or private utility commission. Rural residents, on the other hand, often have private wells or

surface water intakes serving individual homes and bear the sole responsibility for installation and upkeep of these systems. The importance of using multiple barriers for protection of drinking water supplies has been stressed in the literature since the Walkerton tragedy (O'Connor, 2002a, 2002b), and since May 2000 legislators in Ontario have been working to explicitly include these barriers in legislation (Government of Ontario, 2003) while drinking water protection legislation in other jurisdictions often includes one or more of the barriers. Individual water systems are exempt from most drinking water legislation meaning that there is no legal requirement for rural homeowners to apply any protective barriers to their water supplies. It is dangerous, however, to think that individual well owners do not need to think about these barriers simply because there is no legislation requiring them to do so.

The following sections give an in depth introduction to the drinking water barriers including an overview of how the barriers failed in Walkerton, how they can be applied to municipal systems as well as possibilities for application to individual private wells.

1.3.1 Drinking Water Sources

The fundamental characteristics of water are determined by their source. Parameters such as hardness and in many cases odour and colour are often determined by the parent material through or over which water flows. Land use can also affect water quality to a large degree (Zebarth *et al.*, 1998). The types of contamination a specific water source may be at risk from will be different if the source is in an urban or agricultural area. Walkerton's contaminated groundwater was linked to runoff from a nearby farm, which entered the town's aquifer and subsequently their well after a particularly intense storm event.

1.3.1a Surface Water

Water for human consumption can come from either surface water or groundwater. Surface water can be extracted from one large source such as Lake Ontario, (where Toronto draws their water) or from a combination of smaller sources such as the reservoirs used to supply cities such as St. John's, Newfoundland or Vancouver, BC. In rural areas small streams may be used to directly supply individual homes with the water they need. In some cases all human activity is prohibited within the watersheds used to supply drinking water (as in the case of Vancouver) but in many other cases municipalities have no control over what happens to the rivers and lakes from which they draw their drinking water. Individuals with private water systems have even less control. Surface water has long been recognized as being susceptible to contamination by pathogens and harmful chemicals (Harman *et al.*, 2000).

1.3.1b Groundwater

If a suitable source of surface water cannot be found, municipalities or individuals may use groundwater to meet their water needs. Groundwater is used by 26% of Canadians as their drinking water source. Approximately two thirds, or five million, of these users live in rural areas (Environment Canada, 2003). Groundwater is an attractive option for rural homeowners because it is often of high quality and more broadly distributed than surface water supplies (Summers and FitzGibbon, 2003). In most cases groundwater has traditionally been considered less prone to contamination than surface water (Rudolph, 1998); however the events in Walkerton showed that this resource is not failsafe.

Groundwater is found in the spaces between particles of rock and soil, or in crevices and cracks in rock and sometimes (but very rarely) in caverns and underground rivers. It is stored in water-bearing formations called aquifers of which there are two types, confined and unconfined. Confined aquifers are protected from contamination by an impervious layer of material (such as clay) between the aquifer and surface. This impervious layer is called an aquitard. These aquifers are more secure than unconfined aquifers (also sometimes called shallow aquifers), which lack a protective layer and are therefore more likely to come into contact with water from the surface. Confined aquifers are not always completely protected and may still be at risk depending on the geology of the area. For example, areas of fractured rock (karst topography) can provide rapid conduits for surface contamination of groundwater.

The substrate through which groundwater moves acts as a natural filter for many contaminants and pathogenic microorganisms. The capacity of the ground to act as a filter for water is a function of several factors including characteristics of the soil and underlying bedrock and the nature and concentration of potential pollutants entering the system. Unfortunately, these natural processes have limits which can be stressed to the point that they cease to function leading to groundwater contamination (Summers and FitzGibbon, 2003; Jaffe and DiNovo, 1997).

In British Columbia, the percentage of the population which relies on groundwater is below the national average; only 40% of rural water supplies are provided by wells (Dakin, 1994); however, there are rural areas of the province that rely heavily on groundwater to supply their drinking water. The Lower Fraser Valley is one of these areas and previous studies have shown high levels of nitrate in some aquifers in the valley. In a detailed groundwater study completed on the

Hopington Aquifer (near Langley, B.C.), 30-40% of samples taken were found to be impacted by pollution (3-13% of all samples showed nitrate values above drinking water health standards) (Schreier *et al.*, 1996). In the nearby Abbotsford Aquifer, nearly 60% of samples were found to have nitrate-N concentrations that exceeded health standards (Liebscher *et al.*, 1992).

British Columbia is the only province in Canada lacking groundwater legislation. There are no licensing requirements for groundwater extraction nor do well drillers need to be licensed. This makes gathering of information about groundwater use difficult, as the only source of information is a voluntary registry maintained by the ministry of water, land and air protection. As this registry is voluntary the quality and completeness of the information it contains is variable.

1.3.1c Land Use and Source Protection

Protection of water sources is the first step towards maintaining water that is safe to drink. In most areas, local land use activities can greatly influence water quality (Yanggen and Born, 1990). Surface and groundwater used for drinking is vulnerable to pollution from both point and non-point sources. In the recent past, much attention has been paid to pollution from large identifiable point sources. Many anti-pollution campaigns have successfully focused on operations such as mines, pulp mills and sewage treatment plants to the extent that the pollution from such industries is highly managed and regulated in North America. "Point sources of pollution [in the United States] were regulated through federal legislation... in 1972. As a consequence, 60-80% of the pollution that now occurs in United States waters comes from non-point sources" (Gilliam *et al.*, 1997).

In Canada the majority of contamination in rural areas is from non-point sources of pollution (Berka *et al.*, 2001 and Wernick *et al.*, 1998). This type of pollution is difficult to identify, trace and to mitigate because it is caused by multiple uses within a single watershed. Often individual actions or processes are not of a large enough magnitude to degrade water quality; however, the combination of these actions can lead to water that is seriously contaminated and unfit for human consumption. Consider modern agriculture, which depends on a vast array of chemical inputs to maintain the high productivity needed to feed the growing population of the world; the consequences of these inputs often result in cumulative degradation of water quality, which negatively affects wildlife and humans alike. Most of the inputs to agricultural fields (except pesticides) are not inherently toxic. The nutrients found in fertilizers such as nitrogen (N) and phosphorus (P) are essential for plant growth and development and plants would die without

them. However, when they are applied in excess of plant capacity and drained quickly from fields, they enter water in forms that are toxic to humans (Nitrate-N) or detrimental to aquatic life (sediment bound P). Other land uses that contribute to non-point source (NPS) pollution include forestry activity (including logging, road building and chemical application), urban and subdivision development, septic systems, road transportation, recreation (including swimming and motorized activities), wildlife and mining (Panel Review, 2002). Non-point source pollution is challenging to regulate because conventional legal tools are only designed to address point sources of pollution. Additionally, rural areas often do not have the same resources as their urban counterparts for protecting water sources (DeLoe *et al.*, 2002). Therefore, understanding how the cumulative effects of different land uses can impact water quality is essential for recommending actions to protect rural water sources.

1.3.2 Water Treatment

Most rural homeowners do not treat their drinking water and rely solely on the quality of their drinking water sources to protect them from disease (Jaques and Rhode 2001; Summers and FitzGibbon, 2003; Legault, 2000). Drinking water treatment deals with three types of contaminants, microbial, such as bacteria, viruses and protozoa, chemical contaminants such as metals and pesticides and contaminated sediments and radiological contaminants (Jaffe and DiNovo, 1987). In North America the most common form of water treatment is disinfection using chlorine. Chlorine is toxic to most waterborne pathogens and can inactivate harmful or disease causing organisms (O'Connor, 2002a). In British Columbia disinfection is mandated for all surface water sources and (as of May 2003) "groundwater that, in the opinion of a drinking water officer, is at risk of containing pathogens" (B.C. REG 200/2003). Beyond disinfection there are additional treatment barriers such as coagulation, sedimentation and filtration that are recommended for surface water or groundwater under the influence of surface water when chlorine disinfection alone does not provide for adequate safety. Recently much emphasis has been placed on the importance of filtration as an addition to the treatment barrier.

Failure of treatment can lead to waterborne disease outbreaks. Treatment fails when treatment systems malfunction or a sudden change in water quality overwhelms the ability of the chlorine to disinfect. In Walkerton the operators of the water system (in response to public pressure and based on their own confidence in the quality of the water) routinely did not add adequate levels of chlorine to the water to ensure that pathogens were killed. At the time of the outbreak one well did not have a chlorinator installed at all and it was later revealed that the operators had routinely

lied when recording the amount of chlorine that was added to the system. Past outbreaks in British Columbia have been attributed to problems with treatment such as chlorinator malfunctions, chlorinators being turned off and inadequate chlorination being provided to residents by replacement workers during a strike (Health Canada, 1978-1995, Provincial Health Officer, 2001, Christensen and Parfitt, 2003)

As mentioned previously, most rural well owners do not treat their water. This is true even in cases when the wells have known water quality problems. A study of deterioration of well water environments in Alberta found that “although about 74% of the active wells were reported to have water quality and/or quantity problems, only 31% of these wells were identified as ever having received a treatment and shock chlorination was the only treatment method used in the study area” (Legault, 2000). A similar study in Saskatchewan found that “although 76% [of wells] reported at least one water quality problem only 38% of the wells have been treated at least once in their lifetime” (Jaques and Rohde, 2001). In addition to the fact that treatment is rare in small or individual water systems a study in England found that when small water systems do employ treatment, it is often ineffective because of a lack of maintenance and operational knowledge (Rutter *et al.*, 2000).

1.3.3 Distribution System

The water distribution system is the network of pipes between the source and treatment system and the consumer’s plumbing system, and includes storage of treated water in water towers and reservoirs. Contamination of the distribution system can occur as a result of water main breaks, construction of new mains or infiltration from the surrounding groundwater into the distribution system pipes. The longer water is in the distribution system the greater the risk of a decrease in quality. An American investigation of the incidence of outbreaks in public water systems caused by contaminated water entering the distribution system found that between 1971 and 1998, 18.3% of the outbreaks in the United States were caused by chemical and microbial contaminants entering the distribution system (Craun and Calderon, 2001). The causes of these outbreaks were largely cross-connections, back siphonage and contamination of distribution storage.

While rural water systems do not have extensive distribution systems as described above, this barrier cannot be neglected in discussions of rural water supplies. If we consider the rural distribution system to encompass the well (or intake pipe in the case of surface water) and all subsequent piping between the well and the point of use (home and/or barn(s)) there are many potential avenues for contamination to occur in the distribution system. In addition to the

problems mentioned above, groundwater wells themselves can be a conduit for contamination to enter aquifers. This is because, in a sense, water wells are “a puncture through the protective filter of the earth” (Jaffe and DiNovo, 1987 and Summers and FitzGibbon, 2003). It is important that wells be properly constructed including proper casing and sealing to avoid this “short-cutting” of contaminants to aquifers. An American study found a significant relationship between levels of coliforms in well water and the depth of well casing, further highlighting the need for properly constructed drinking water wells (Tuthill *et al.*, 1998). Other experts feel that most cases of contamination in wells are caused by poor construction of the wells and not from widespread aquifer contamination (Summers and FitzGibbon, 2003). Some non-pathogenic microorganisms can also reduce the value and life of water wells. This process, called bio-fouling, can result in reduced yield quality of water from the well (Legault, 2000 and Jaques and Rohde, 2001).

1.3.4 Monitoring

Monitoring involves taking measurements and collecting samples to ensure the water system is working properly. For municipal systems the frequency of monitoring is usually legislated based on the size of water systems, but as for treatment, there are no legal requirements (in BC) for individual well owners to monitor their drinking water. Many rural residents will test their water at the time a new well is drilled to ensure potability but then do not test the water again. The requirements for frequent monitoring of larger water systems acknowledges the fact that water quality can change either seasonally due to a fluctuating water table, in response to a storm or other watershed event (such as a landslide) or over several years due to changing land uses such as increased intensity of agriculture or density of homes. Rural well owners may be especially vulnerable to these changes as their water sources are often close to home and these land use changes can be happening all around individual wells.

Monitoring budgets are often limited and there is an endless array of pathogens and chemicals that can be detected in water. Time and money are very real constraints that cause monitoring effort to focus on a reduced number of “indicator” organisms and chemicals that are easy to measure and whose presence or absence may indicate contamination with other, harder to measure, but potentially more dangerous chemicals and pathogens. The use of indicators is imperfect (Allan *et al.*, 2000, Craun *et al.*, 1997) but often the only practical method of assessing water quality. For example, most waterborne diseases are caused by microorganisms found in

fecal wastes and therefore if fecal coliforms are found in drinking water this constitutes an unacceptable risk (O'Connor, 2002a).

Microbial parameters should not be the sole focus of monitoring. Other parameters that can be monitored include watershed events (that might be likely to cause contamination such as large storms, landslides or even construction near water sources), rates of disease in the community and other water quality parameters such as turbidity, total dissolved solids, temperature, alkalinity, hardness, pH, hydraulic flow rates, and conductivity (Allen *et al.*, 2000). In order for monitoring to be effective it needs to be carried out in relation to rainfall and runoff events as well as frequently enough to detect changes in parameters that might signify a risk to the health of the people consuming the water.

1.3.5 Response

This final barrier involves the appropriate responses to failing process measures or adverse water quality. This barrier is important because it is the final line of defense if all the other barriers fail and water becomes contaminated. If an outbreak is detected in the community rapid response can limit the scope of the outbreak.

The most common response to higher than acceptable levels of bacteria in water and community gastrointestinal disease outbreaks is for the medical officer of health to issue a "boil water order". This alerts the public to the fact that the water is contaminated with bacteria or other microbial parameters and should not be consumed unless boiled for 5 minutes. In general, issuing a boil water order is a very serious decision as unwarranted boil water advisories can have drastic economic and social consequences as well as the potential to undermine the future credibility of the health unit issuing the advisory on the area for which it is ordered. There are several factors that must be taken into account when responding to adverse monitoring results; do the monitoring results correspond to an increase in illness in the community? Conversely, if people in the community are ill and there haven't been adverse water quality reports, can a link be made from the illness to the community water source? There have been cases where outbreaks have been discovered through clinical monitoring rather than through routine monitoring of water quality and cases where the routine monitoring was poorly carried out, water quality was thought to be compromised and boil water orders were issued with no corresponding evidence of increases in illness in the community (Allen *et al.*, 2000). It must also be kept in mind that a boil water advisory is only issued in cases where the problem with the water is biological. If the water has

been impacted by other chemicals this may be an inappropriate response (in the case of high nitrate values, boiling water will concentrate the chemical rather than dispersing it).

Because private well owners rarely treat their water and rarely monitor it, there is not often a chance to respond to adverse results. Additionally, endemic illness in residents on private water supplies are often not linked to water even though many experts feel that water may be the cause of much gastrointestinal illness (Rutter *et al.*, 2000).

1.4 Nitrate and Pathogens – Contaminants of Concern

When examining the impacts of land use on rural groundwater wells and aquifers it quickly becomes apparent that nitrate and pathogens are the most frequently detected contaminants that may impact human health (Coote and Gregorich, 2000; Goss *et al.*, 1998; Rudolph *et al.*, 1998; Reid *et al.*, 2002). Nitrate and pathogens enter groundwater from a variety of sources, the most common being manure and fertilizer applications to agricultural lands and from septic systems, especially in un-sewered rural subdivisions (Coote and Gregorich, 2000; Dakin, 1994; Carmichael *et al.*, 1995; Schreier *et al.*, 1996; Liebscher *et al.*, 1992).

1.4.1 Nitrate

Nitrate (NO_3^-) is a highly soluble nitrogen anion that is not usually adsorbed to soil particles (the terminology Nitrate-N is used to refer to only the nitrogen part of the nitrate molecule when discussing levels of nitrate in groundwater, however, throughout this report nitrate is used interchangeably with nitrate-N when referring to specific measurements of this nutrient.) Supplying crops with adequate nitrogen is vital to ensuring food supplies both nationally and internationally; however, addition of nitrogen fertilizer or manure to farm fields in excess of that required for crops will result in loss of nitrate from the root zone by leaching (Follett, 1989). Nitrate is a human health concern because it has been associated with methemoglobinemia, a condition in which nitrate is converted in the gut to nitrite, which then combines with hemoglobin to form methemoglobin, thus decreasing the ability of the blood to carry oxygen. Infants less than three months of age are most susceptible to this sometimes-fatal condition, which is also called “Blue Baby Syndrome”. Nitrate has also been associated with certain types of cancer although there are no conclusive links between ingesting nitrate and incidence of cancer (Follett, 1989; Coote and Gregorich, 2000; Nolan and Stoner, 2000).

The maximum allowable concentration (MAC) for nitrate in the Guidelines for Canadian Drinking Water Quality (GCDWQ) is 10 mg/L nitrate as nitrogen. However, since background levels of nitrate in groundwater are generally less than 2 mg/L we use 3 mg/L as an indication that the groundwater has been impacted or is under the influence of surface water (Coote and Gregorich, 2000; Schreier *et al.*, 1996; Carmichael *et al.*, 1995). If groundwater has levels of nitrate above 3 mg/L it is an indication that the drinking water may be contaminated other substances of concern.

1.4.2 Coliforms

As mentioned in section 1.3.4 (monitoring), it is impossible to test drinking water for every parameter that might adversely impact human health; this is especially true of disease causing microorganisms. For this reason, indicator organisms are measured to indicate the potential for contamination. The most common indicator is the coliform group of bacteria. Total coliforms include species of bacteria that often inhabit intestines of warm-blooded animals as well as some bacteria that also occur naturally in the environment and are usually not pathogenic. If total coliforms are present in treated drinking water this may indicate a treatment failure or contamination event, however, presence of total coliforms does not necessarily indicate that pathogenic organisms are present. Fecal coliforms are a thermotolerant sub-group of total coliforms. These bacteria can survive at higher temperatures and are only associated with the fecal material of warm-blooded animals, thus the presence of fecal coliforms in a water sample indicates that the water has been contaminated with the fecal matter of humans or animals. The presence of fecal contamination is an indicator of a higher degree of potential risk for individuals exposed to this water. Research has shown (Reynolds, 2003) that >90% of the population of total coliform bacteria in human and animal feces is *Escherichia coli*. Therefore *E. coli* is considered the most accurate indicator of fecal contamination in water. Again it must be kept in mind that the presence of *E. coli* in drinking water does not guarantee that water is contaminated (many sub-species of *E. coli* are harmless) but simply indicates increased risk. Most experts recommend boiling water that has tested positive for fecal coliforms or *E.coli* in order to minimize the risk contracting gastrointestinal illness.

Total and fecal coliforms and *E. coli* are not perfect indicators. They have not proven effective at predicting the presence of viral or protozoan contamination. Large outbreaks of disease caused by protozoan parasites such as *Giradia* and *Cryptosporidium* have occurred in communities where no negative results for coliforms were detected. The most notable of these was an

outbreak of cryptosporidiosis in Milwaukee, Wisconsin in 1993. Despite these limitations, measurements of total and fecal coliforms remain one of the most important and widely used measures of potential microbial contamination and have been effective in preventing outbreaks of cholera, typhoid fever and various gastrointestinal diseases. There are currently no guidelines for levels of the parasites *Giardia sp.* or *Cryptosporidium sp.* nor are there any guidelines for levels of viruses due to the difficulty, expense and unreliability of current detection methods.

The Canadian water quality guidelines for bacteria are currently under review but state that zero fecal coliforms should be detected in 100 ml of sample and no more than 10 total coliforms in 100 ml (of which none should be *E. coli* or thermotolerant bacteria). The vulnerability of surface water to pathogen contamination has long been recognized, leading to the disinfection by chlorine of surface water in Belgium since 1902. Since then, disinfection has become a standard public health practice credited with saving millions of lives around the world (Benidickson, 2002; Hillel, 1998). Groundwater, on the other hand, was long thought to be protected from the bacteria that inhabit surface water with papers published as recently as (1999) claiming that “untreated groundwater is free from fecal contamination as a result of passage through soil over extended periods of time” (VanDerKooij *et al.*, 1999). The Walkerton experience is but one dramatic example that shows that groundwater can be vulnerable to bacterial contamination. Many other studies have linked disease outbreaks, illness and contamination events with groundwater from both public and private wells. (Hass, 1999; Craun and Calderon, 2001; Lippy and Waltrip, 1984; Borchardt *et al.*, 2003). Clearly bacterial contamination is a concern in both surface and groundwater.

1.4.3 Nitrate: A useful indicator for coliform contamination?

Several studies in Ontario and the U.K. have shown that in general, private wells exceed water quality guidelines more often for total and fecal coliforms than for nitrate. Additionally there is often little correlation between the wells that exceed the level for both. For example, a study of private wells in Ontario found that of 1292 wells samples in 1991 and 1992 34% were above the acceptable limit for coliforms and 14% were above 10 mg/L NO₃-N. Only 7% exceeded the MAC for both (Goss *et al.*, 1998). A similar Ontario study using multi-level monitoring wells at 144 farms throughout the province found that 23% of sites had concentrations in 50% or more of the monitored intervals that exceeded the MAC for NO₃-N in both sampling periods. In the same study total coliform levels were exceeded by 66% of wells in winter and 36% in summer. This study also found a clear trend of decreasing NO₃-N with depth but no similar trend with coliform

contamination – which remained relatively similar regardless of depth (Rudolph, 1998). A UK study that examined 1750 wells between 1992 and 1998 found the failure rate for total coliforms to be 41%, 30% for fecal coliforms and 15% for nitrate. Less than 50 of the wells sampled failed for both nitrate and coliforms (Reid *et al.*, 2002).

The studies mentioned above would seem to suggest that nitrate is not a viable indicator of coliform contamination. In every study the number of coliform failures exceeds the number of nitrate failures and there appears to be little correlation between the wells that fail for nitrate and those that fail for coliforms. It must be kept in mind, however that in most places detection of fecal coliform bacteria is considered a failure whereas the 10 mg/L failure level for NO₃-N is much higher than the detection limit and also quite a bit higher than the 3 mg/L level considered by most to be indicative of groundwater that is under the influence of surface water. In terms of establishing a link between nitrate levels and coliform contamination the above studies would be more useful if they reported the percentage of nitrate levels above 3 mg/L as well as nitrate failures when comparing these to coliform levels. Therefore, although the above studies seem to suggest that nitrate is not a useful indicator for coliform contamination, full reporting of nitrate results would be more useful when attempting to establish whether or not a useful link exists between these two parameters. The fact remains that it is easier and cheaper to measure nitrate levels than coliform levels in drinking water supplies.

1.5 Other Water Quality Parameters

1.5.1 Iron and Manganese

Iron (Fe) and manganese (Mn) are non-hazardous elements that can be a nuisance in water supplies. The elements are chemically similar and cause similar problems in drinking water including objectionable taste and odour as well as staining of fixtures, dishes and laundry. Water with high levels of iron and manganese is not considered hazardous to human health and the sources of these elements are not generally anthropogenic in nature. Water (especially with low pH) dissolves Fe and Mn as it percolates through soil and rock and holds the dissolved forms of these metals in solution. Water with high levels of Fe and Mn is typically clear and colourless until exposed to oxygen (O₂). Exposure to oxygen causes the metals to be oxidized and change to coloured solid forms. The oxidized forms of Fe and Mn are visible as particulate matter and cause the water quality problems mentioned above. Additionally, oxidized Fe and/or Mn can build up solid deposits in pipes and other fixtures, eventually reducing water pressure and increasing pumping costs. Finally, there are bacteria associated with high levels of iron which

cause slime buildup in pipes and fixtures. Shock chlorination is required to control iron bacteria. The GCDWQ are 0.3 mg/L for iron and 0.05 mg/L for manganese. These are aesthetic objectives only, water exceeding these parameters are likely to cause the taste, odour and staining problems mentioned above but are not likely to adversely affect the health of those drinking it.

1.5.2 Electrical Conductivity

The electrical conductivity (EC) is a measure of water's ability to conduct current and provides an estimate of the amount of total dissolved salts in the water sample. The greater the content of ions in the water, the more current the water can carry; therefore, conductivity levels are often highly correlated with levels of parameters such as calcium, magnesium and/or silicon. The measurement of conductivity represents an aggregate measure of all dissolved ions in the sample and is therefore influenced by many elements and factors, some natural and some anthropogenic. The geology of the surrounding area will have a large impact on the conductivity of a water sample. For example, groundwater flowing through limestone will have higher conductivity than water flowing over granite because limestone is much more soluble than the former. Anthropogenic activities can increase conductivity by the addition of more dissolved salts into a water system. The activities mentioned above, such as use of commercial fertilizer or spreading of manure, that increase nitrate levels can also increase conductivity levels. There are no water quality guidelines for conductivity.

1.5.3 Orthophosphate (PO_4^{3-})

There is no health standard for phosphorus in drinking water, however, elevated P levels are a concern in surface water, especially in lakes where too much P causes enhanced production of algae which, as it decomposes, consumes dissolved oxygen leading to conditions which are stressful or lethal to certain fish species. This process of nutrient enrichment is called eutrophication.

Orthophosphate (PO_4^{3-}) is a form of phosphorus that is readily available to the biological community and is typically found in very low concentrations in unpolluted water. PO_4 can be produced by natural processes or through manmade influence. Anthropogenic sources include partially treated and untreated sewage, agricultural run-off and some lawn fertilizers (Wetzel, 2001). Although there are no guidelines for P or PO_4 in drinking water there are guidelines for the prevention of eutrophication. Total phosphorus levels above 0.01 mg/L in lakes are considered problematic; for streams, a more conservative guideline of 0.1 mg/L is often used

(Schreier *et al.*, 1996). Orthophosphate is often the form of P measured in water samples because it is the most available form (many forms of P are highly insoluble and tend to bind to sediments rather than dissolve in the water column). PO_4 is a measurement of part of the total phosphorus which means that if PO_4 levels are above the guideline then the total phosphorus is even higher than that. Because of its insoluble nature, P is not generally considered a problem in groundwater. The average level of total P in unpolluted groundwater is about 0.02 mg/L (Wetzel, 2001). This level will vary depending on the amount of phosphorus in the substrate.

1.6 Groundwater in the Lower Fraser Valley

As mentioned in section 1.3.1b, several groundwater aquifers in the Lower Fraser Valley are known to be contaminated with nitrate-N. High nitrate levels (up to 45 mg/L) in the Hoppington and Abbotsford aquifers have been identified and studied in several reports (Schreier *et al.*, 1996, Liebscher *et al.*, 1992; Zebarth *et al.*, 1998). While most research on groundwater quality in the Lower Fraser Valley has focused on the aquifers mentioned above, one comprehensive study looked at 192 community wells and 75 private wells throughout the entire Fraser Lowland and compared the quality throughout. Table 2 shows the number of exceedances of selected parameters measured in this study (Carmichael *et al.*, 1995). This study concluded that elevated levels of nitrate-N occur mostly in highly vulnerable, unconfined aquifers in areas of intense human activity and that elevated nitrate levels are due to human activities. In addition to the 27 wells which had nitrate above health standards 62 wells (26%) had nitrate at levels above 3 mg/L indicating degradation of groundwater from human activities. High levels of Fe and Mn are another well known water quality problem in the Lower Fraser Valley and these parameters have been called “the most troublesome water quality parameters in the Fraser Lowland” (Dakin, 1994).

Table 2. Number of well exceeding the Guidelines for Canadian Drinking Water Quality, Carmichael et al. (1995)			
Parameter	MAC (GCDWQ)	Phase 1 (239 wells sampled)	Phase 2 (240 wells sampled)
Iron (Fe)	0.3 mg/L	38 (16%)	31 (13%)
Copper (Cu)	1 mg/L	5 (2%)	2 (1%)
Zinc (Zn)	5 mg/L	-	1 (0.4%)
Manganese (Mn)	0.05 mg/L	54 (23%)	59 (24%)
Sodium (Na)	200 mg/L	1 (0.4%)	4 (2%)
pH	6.5-8.5 pH units	30 (13%)	31 (13%)
Nitrate-N ($\text{NO}_3\text{-N}$)	10 mg/L	23 (10%)	27 (11%)
GCDWQ = Guidelines for Canadian Drinking Water Quality			
MAC = Maximum Allowable Concentration			

2.0 Research Location

2.1 Introduction to Hatzic Valley

The field research to investigate the issues raised in chapter 1 was conducted in the Hatzic watershed, located in the Fraser Lowland area of British Columbia (BC), Canada. The area was chosen for study for the following reasons: previously, there had been no comprehensive survey of groundwater quality that included all areas of the valley; the valley encompasses several different land uses, a number of which have the potential to negatively impact groundwater quality, and finally, the regional district and local community were interested in having research of this nature undertaken. The watershed is located about 75 km east of the city of Vancouver on the north side of the Fraser River. This predominantly rural watershed is 8370 ha (83.7 km²) in size and includes streams that drain south through Hatzic Lake into the Fraser River. The location of the watershed relative to Vancouver and the Fraser River is shown in figure 1. As the focus of this project was on groundwater resources within the watershed, the field research for this project was confined to the central lowland area of the watershed known as the Hatzic Valley. There are three distinct groundwater aquifers within the Hatzic watershed located in this lowland area. The eastern and western aquifer boundaries correspond to the edge of the valley where the mountains rise steeply from the valley floor. The area of the lowland that encompasses the aquifers considered in the study is approximately 3100 ha (31 km²).

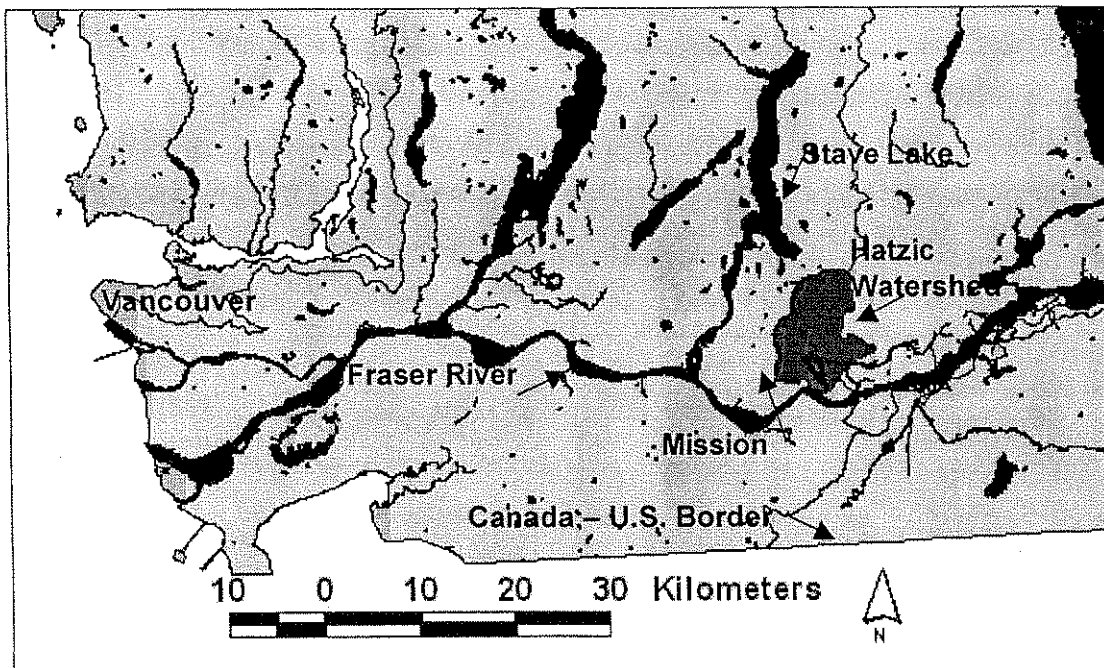


Figure 1. Location of the Hatzic Watershed

The following sections will give an overview of the land use in the valley, as well as detailed descriptions of the geology and geography of the land overlying each aquifer. The aquifer descriptions will also include any previously collected information about the water quality in these areas.

2.2 Land Use in the Watershed

The land in the study area can be split into three predominant categories: agricultural, residential and forested. The land uses are not evenly distributed throughout the valley. Agriculture dominates the central and southeastern portion of the valley, whereas the north is largely covered with forest. Dense residential areas are found in three small subdivisions on Hatzic Prairie (the central valley) and in several subdivisions and trailer parks on Hatzic Island. Hatzic Island is a popular recreational area and there are numerous camps and trailer parks that increase the population seasonally. The uplands to the east and west are largely forest covered and uninhabited due to the steepness of the slopes. Table 3 outlines the area and percentage of each land use for the lowland area (not the entire watershed). Figure 2 shows the complete land use map for the valley.

Table 3. Hatzic Valley Land Use		
Land use	Total (area Ha)	%
Agriculture	1111.003	38.81
Forest	1047.422	36.59
Urban	20.552	0.72
Rural Residential	363.124	12.68
Other^a	286.937	10.02
Total	2862.897	100.00
a = Other includes (water, road, wetland/non-forested riparian, greenhouse)		

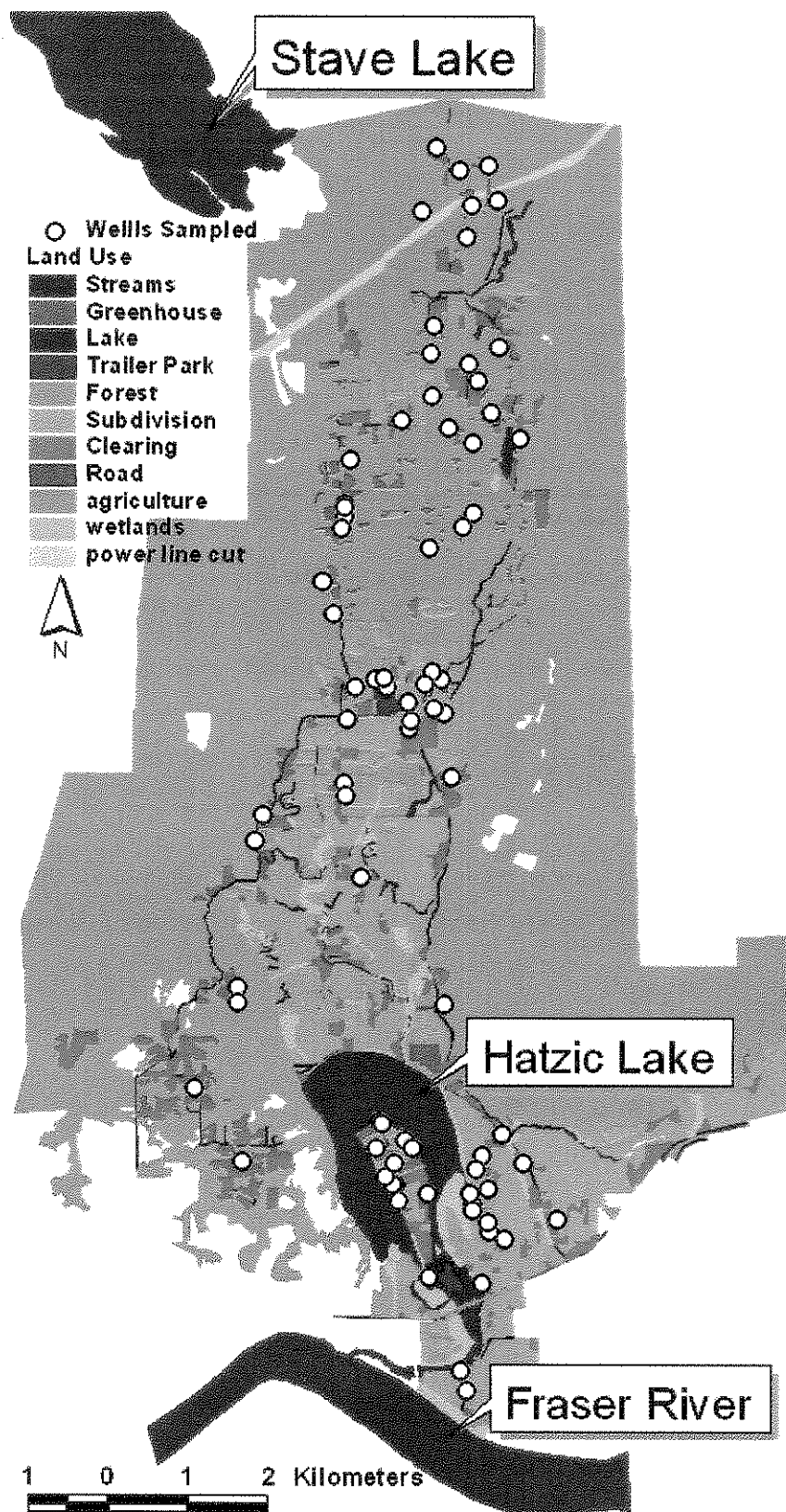


Figure 2. Land Use in the Hatzic Valley

2.3 Drinking Water Sources

Residents of the Hatzic Valley use both groundwater and surface water sources for domestic consumption. As the focus of this project is on the groundwater sources, these will be described in detail in subsequent analysis. The surface water sources used for drinking are mainly small streams that run off the mountains. Although most residents rely on private, individual, water supplies to meet their domestic needs there are a few small community water systems (which serve more than 2 households) within the study area. The three subdivisions in the central valley (Hatzic Prairie) are supplied surface water by community water systems. At least one of these communities (along Sward Road just north of Hatzic Lake) is on a permanent boil water order according to a large sign at the entrance to the area. Information about these water systems was not easy to obtain (Chapman, pers com. 2002) and tap water from these residents was not collected as part of this project. It is not known if this boil water order was issued as a result of a specific contamination event or as a precautionary measure because the water is not treated. The latter is the likely scenario; Hatzic is not included in the list of 28 waterborne disease outbreaks that have occurred in BC since 1980 (Christensen and Parfitt, 2003). There is one community watershed in the valley. The Kenworthy Creek Community watershed is used by the Hatzic Prairie Water User's Community (WUC) for domestic consumption. This WUC holds 10 domestic water licenses that use Kenworthy Creek as a source (Cameron, 1998). It is not known if any of these licences supply the subdivisions mentioned above however the proximity of Kenworthy creek to the subdivision along Sward road suggests this may be likely.

2.4 Aquifer Classification

Many groundwater aquifers in British Columbia have not been extensively studied. One of the most useful sources of information about aquifers in the Fraser Lowland is a report published by the BC ministry of Environment, Lands and Parks (now Water, Land and Air Protection) in 1994 entitled, "A Proposed Classification System for Groundwater Management in British Columbia" (Kreye and Wei, 1994). The aquifer classification system was created to help with prioritizing mapping, assessment, and monitoring of aquifers, to provide more systematic comparison of different aquifers, and as an information tool to increase awareness of groundwater resources. The classification system consists of two parts. The first is the classification component which categorizes aquifers based on their current level of development and vulnerability to contamination, and the second is a ranking component to indicate the relative overall priority for mapping and management of the aquifer based on a number of physical and water use criteria. The classification component is further divided into 2 sub-sections, the first being a development

sub-class and the second a vulnerability sub-class. The development sub-class is based on the level of groundwater use in relation to the aquifer yield or productivity. The classes for development are as follows; heavy (I), moderate (II) or light (III). The vulnerability sub-class assesses the vulnerability of each aquifer to contamination from surface sources, and is based on the type, thickness, and extent of the geological material overlying the aquifer, the depth to water (or top of confining layer), and the porosity of the aquifer materials. The vulnerability rankings are high (A), moderate (B) or low (C). The combination of development ranking and vulnerability ranking results in aquifer class (of which there are nine possible combinations). The ranking for each aquifer is a numerical measurement of the priority for mapping and assessment. The possible ranks range from 5 to 21, where higher-ranking values are an indication of higher relative priority. The rankings are based on the following criteria: productivity, vulnerability, size, demand, type of use, quality concerns, and quantity concerns. See Kreye and Wei (1994) for a more complete description of the method of classifying and ranking the aquifers.

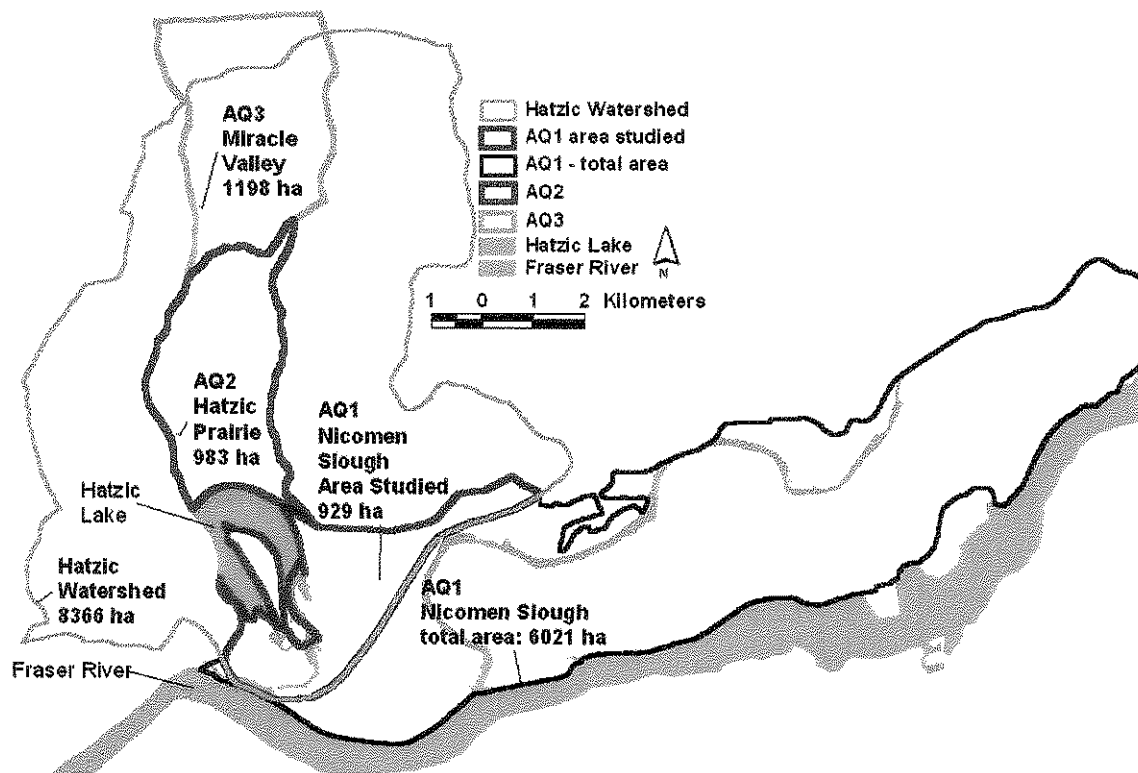


Figure 3. Watershed and Aquifer Boundaries and Areas, Hatzic, British Columbia

The identification of the three distinct aquifers within the Hatzic Watershed comes from the initial report on Aquifer Classification in BC, which used the Fraser Lowland as the pilot study area. The purpose of the aquifer delineation was only to rank, classify, and inventory the aquifers. Detailed analysis of flow systems and variations in water quality were beyond the scope

QUESTION

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Fig. 1. ΔT_{eff} as a function of ΔT_{eff} for various values of ΔT_{eff} .

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of that project. The aquifers were delineated based on previously existing information on surficial geology for shallow aquifers and Halstead's hydro-statigraphic units (Halstead, 1986) for deeper confined aquifers. Information from available consultants reports, other published and unpublished reports, and well records were also used.

As previously mentioned, three separate aquifers within the Hatzic Watershed have been delineated (see figure 3). Two of these aquifers are ranked as IIIA with rankings of 12 and 10, that is, they are not highly developed but are highly vulnerable to contamination. The third aquifer is ranked as IIIC meaning it is not highly developed or vulnerable to contamination. These aquifers will be described in detail in the following three sections.

2.4.1 Nicomen Slough Aquifer

The southern most aquifer within the Hatzic Watershed is the Nicomen Slough aquifer (hereafter known as AQ1). This aquifer has an area of approximately 60 km² and extends to the east several kilometers outside the study area. Only the area within the Hatzic Watershed (less than 10 km²), which includes Hatzic Island and the land directly east of it (which is bounded by Sylvester Road on the east and Loughheed Highway to the south), were monitored as part of this study (see Figure 3 and Figure 4).

AQ1 is a shallow, unconfined, sand aquifer; the depth to the top of the aquifer is estimated to be less than 10 m (in some cases as little as 3 m) (Carmichael *et al.*, 1995; Kreye and Wei, 1994). This aquifer was given a classification of IIIA meaning it is highly vulnerable to contamination but not heavily used. A local exception is on Hatzic Island where groundwater use is high (Carmichael *et al.*, 1995). This means that the aquifer classification for the entire Nicomen Slough aquifer is not accurate on Hatzic Island and that the vulnerability to contamination in this area is high. The ranking of the aquifer is 12 on a scale of 5 to 21, meaning it would have intermediate priority for mapping and assessment from the provincial government. The only water quality concern for this aquifer identified by Kreye and Wei (1994) was locally high iron levels (a non-health related problem). There are several well logs available for wells in this area; however, none of the well logs have stratigraphic information so profiles could not be created.

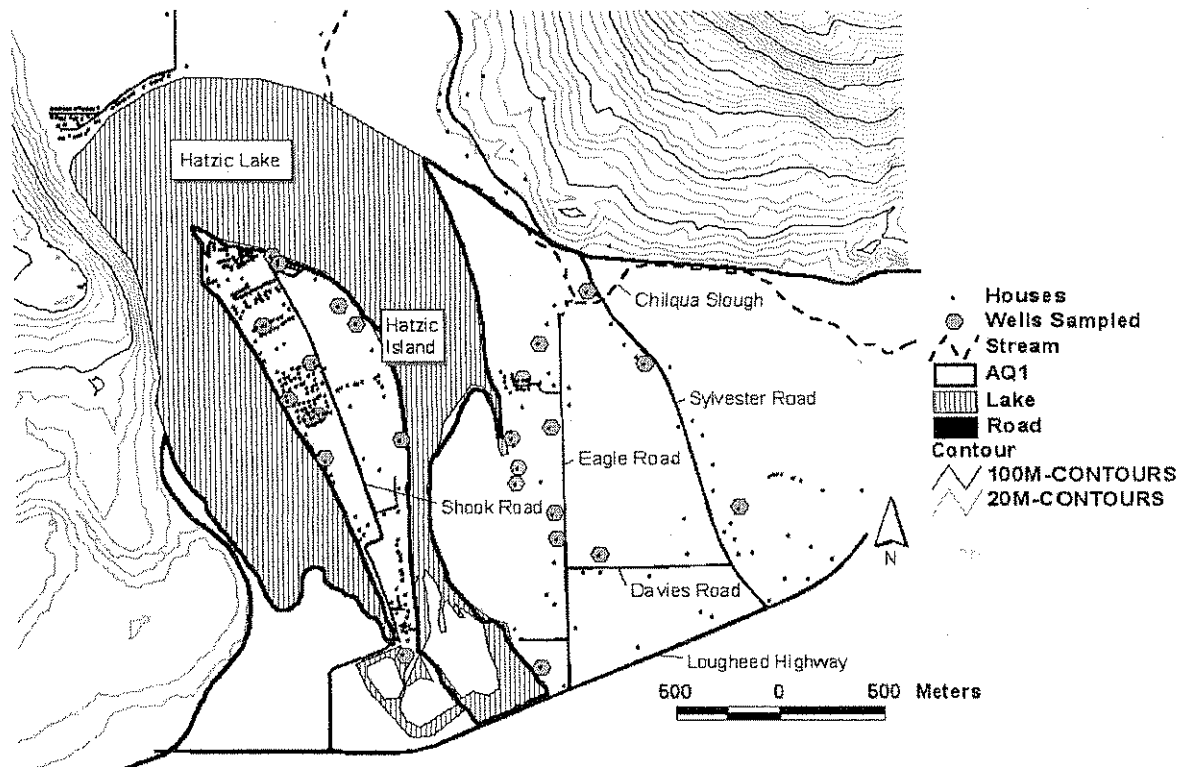


Figure 4. AQ1 – Nicomen Slough Aquifer

Hatzic Island and the region directly east of it have the same surficial geology; the entire area is classified as Fraser River Sediments. In this area these are channelled deposits (expressed at the surface by ridges and swales) up to 10 m thick, consisting of sandy loam and loamy sand that overlie estuarine fine sand to clayey silt (Armstrong, 1980). Soils on the island are moderately well to well drained, and generally medium to moderately coarse in texture. The soils just east of the island are not as uniform as on the island and in this area, in addition to soils similar to those found on the island, there are areas where the drainage is imperfect due to a fluctuating groundwater table. The soils in these areas tend to be medium to moderately fine textured (Luttmerding, 1980a).

AQ1 is one of the only regions in the study area where any previously collected water quality information is available. Eight community wells on Hatzic Island were sampled in 1992-1993 as part of the Fraser Valley Groundwater Monitoring Program (Carmichael *et al.*, 1995). Nitrate values for the wells sampled as part of this report were not above health standards and ranged from 0.06 mg/L to 5.79 mg/L. These wells had depths ranging from 24.38 to 6.1 m. The water table, however, was reported to be as little as 3 m from the surface for two of these wells including the 24.38 m deep well. Water quality information was collected for Hatzic Lake in the

1970's (Drinnan, 1975). At that time, water quality in the lake was reported to be below standards for most parameters, but eutrophication was a concern. At the time, recommendations were made that "investigations should include reviews on the improvement of faulty septic tanks, feasibility of waste collection and treatment... due to poor soil drainage and the possibility of potential health problems, the area may require a sewer system if there is an increase in population," (Drinnan, 1975) (note that Luttmerding's soil maps (1980a) reported the soils on the island as well to moderately well drained whereas this report (written before 1980) says the soil is poorly drained).

Hatzic Lake is the most distinctive geographical feature in the area. This horseshoe shaped lake (likely a former meander of the Fraser River) is quite shallow, with a mean depth of only 2 m and an area of about 300 ha. The water level fluctuates greatly with the season. In fact much of the eastern portion of the lake routinely dries up in the summer exposing vast mudflats through which only a narrow channel continues to run.

Hatzic Island, a peninsula running up the center of the lake (see fig 4), is the most densely populated area of the valley. There is no surface water on the island and, therefore, the residents are entirely dependent on groundwater for their domestic needs. The lake and the island are a popular recreational area with a mixture of seasonal and permanent dwellings in trailer parks, subdivisions, and on larger rural lots. Some agriculture is also present on the island, but it is not a dominant land use, as it is on the area just east of the island. Table 4 outlines the land use for AQ1 in ha and as percentages of the total area considered in the study.

Table 4. Land Use AQ1		
Land use	AQ1 (area ha)	%
Agriculture	410.673	59.39
Forest	42.225	6.11
Urban	30.307	4.38
Rural Residential	75.545	10.92
Other^a	133.774	19.34
Total	691.524	100.00
a = Other includes (water, road, wetland/non-forested riparian, greenhouse)		

2.4.2 Hatzic Prairie Aquifer

The second aquifer within the study area is the Hatzic Prairie Aquifer (hereafter known as AQ2). This aquifer is overlain by the Hatzic Prairie, a flat area that extends north from Hatzic lake and is bounded on the east and west by the hills which rise steeply from the valley floor. The eastern boundary is also defined by Sylvester Road. The northern boundary for this area is just past

Durieu Road, where the topography begins to rise. There are many streams that cross this part of the watershed, including Hatzic Slough, Legace and Pattison creeks, all of which eventually drain into Hatzic Lake. The land is very flat and the dominant land use is agriculture (69%) (see table 5 for complete land use in ha and as percentages of the total). The three rural subdivisions mentioned previously that use surface water for their domestic needs are located on the prairie. Aside from these subdivisions, Hatzic Prairie is not densely populated. The high water table and frequency of flooding may be contributing factors.

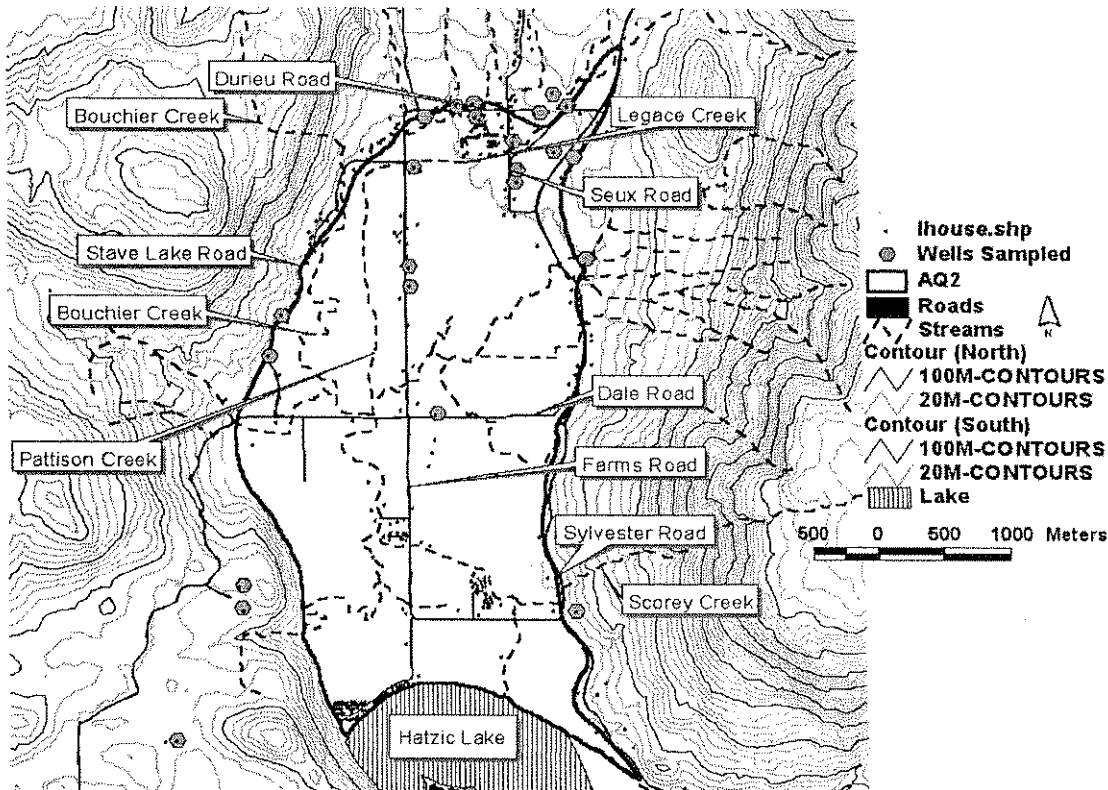


Figure 5. AQ2 – Hatzic Prairie Aquifer

AQ2 was also given a classification IIIA, meaning that it is highly vulnerable to contamination but is not heavily used; again the only water quality concern identified by Kreye and Wei (1994) was non-health related localized iron problems. This aquifer is also shallow and unconfined with the predominant material of the aquifer being sand and gravel. The depth to the top of the aquifer is estimated to be less than 10 m. Because of high levels of iron and the proximity to streams emerging from the headwaters many residents opt to use surface water for drinking in this part of the valley. The wells sampled for this project were concentrated towards the north of the aquifer.

The surficial geology on the Hatzic Prairie is the most varied of the three aquifers within the study area. The entire area is underlain by Fraser River sediments of the type found over all of

AQ1. In some areas these sediments are at the surface, but over a majority of the prairie the dominant material at the surface are Salish Sediments. Some of these include peat, organic silt loam and silty clay loam (0.3 to 10 m thick) but the majority are stream deposits classified as lowland stream channel fill and overbank sandy loam and clay loam, which in places contains disseminated organic material up to 8 m thick. There two areas, one towards the north of the aquifer and the other on the central western side of the valley, where there are slope deposits consisting of fan and landslide gravel, as well as sand and rubble, up to 15+m thick. These deposits also overlie Fraser River Sediments. There are a variety of different soil types on the Hatzic Prairie, which range in texture from fine to medium textured to partially decomposed organic matter. Most of the soils are poorly drained due to a high groundwater table. There are some localized areas of coarse alluvial fan deposits that are well to rapidly drained (Luttmerding, 1980a). This soil is found in two locations and is associated with the slope deposits mentioned in the surficial geology description.

Table 5. Land Use AQ2		
Land use	AQ2 (area ha)	%
Agriculture	617.271	62.79
Forest	126.032	12.82
Urban	21.931	2.23
Rural Residential	103.275	10.50
Other^a	114.641	11.66
Total	983.15	100.00
a = Other includes (water, road, wetland/non-forested riparian, greenhouse)		

2.4.3 Miracle Valley Aquifer

The third aquifer within the study area identified by Kreye and Wei (1994) is the Miracle Valley aquifer (hereafter known as AQ3). This aquifer is a largely sand and gravel aquifer classified as IIIC, meaning that it is neither heavily used nor highly vulnerable to contamination. In contrast to the first two aquifers, AQ3 is a confined aquifer with no identified water quality problems. The top of this deeper aquifer is estimated at 35 to 50 m deep. This aquifer is located at the north end of the Hatzic Valley where the topography rises up to the area known as the Miracle Valley. The human population is denser in this area than on Hatzic prairie, but not as dense as on Hatzic Island. There are no subdivisions in this area. Most of the land is covered in forest and the majority of people live on large treed lots (see table 6 for complete area of each land use on this aquifer). Drinking water is mainly from groundwater sources but some surface water is used for domestic purposes as well. Some of the wells sampled in the Miracle Valley are not located in

the Hatzic watershed but are part of the Cascade Creek system, which drains north into Stave Lake, rather than south into Hatzic Lake.

The surficial geology of the Miracle Valley is mixed and consists of more than one stratigraphic unit. The primary unit is the Fort Langley Formation, a glaciomarine stony silt to loamy clay that can be 8 to 100 m thick. Overlying much of the Fort Langley formation is a layer of sandy till and sub-stratified drift 0.5–2 m thick (Armstrong, 1980). There are several different soils in the Miracle Valley. The predominant soil is a moderately well drained medium to moderately fine textured glaciomarine deposit. There are local areas where 40 to 160 cm of well-decomposed organic material is underlain by moderately fine textured glaciomarine deposits with very poor drainage due to a perched water table. In other areas the soils are well drained gravely outwash deposits. Finally there are also areas along the east of the valley where there are coarse-textured alluvial fan deposits with well to rapid drainage (Luttmerding, 1980a&b).

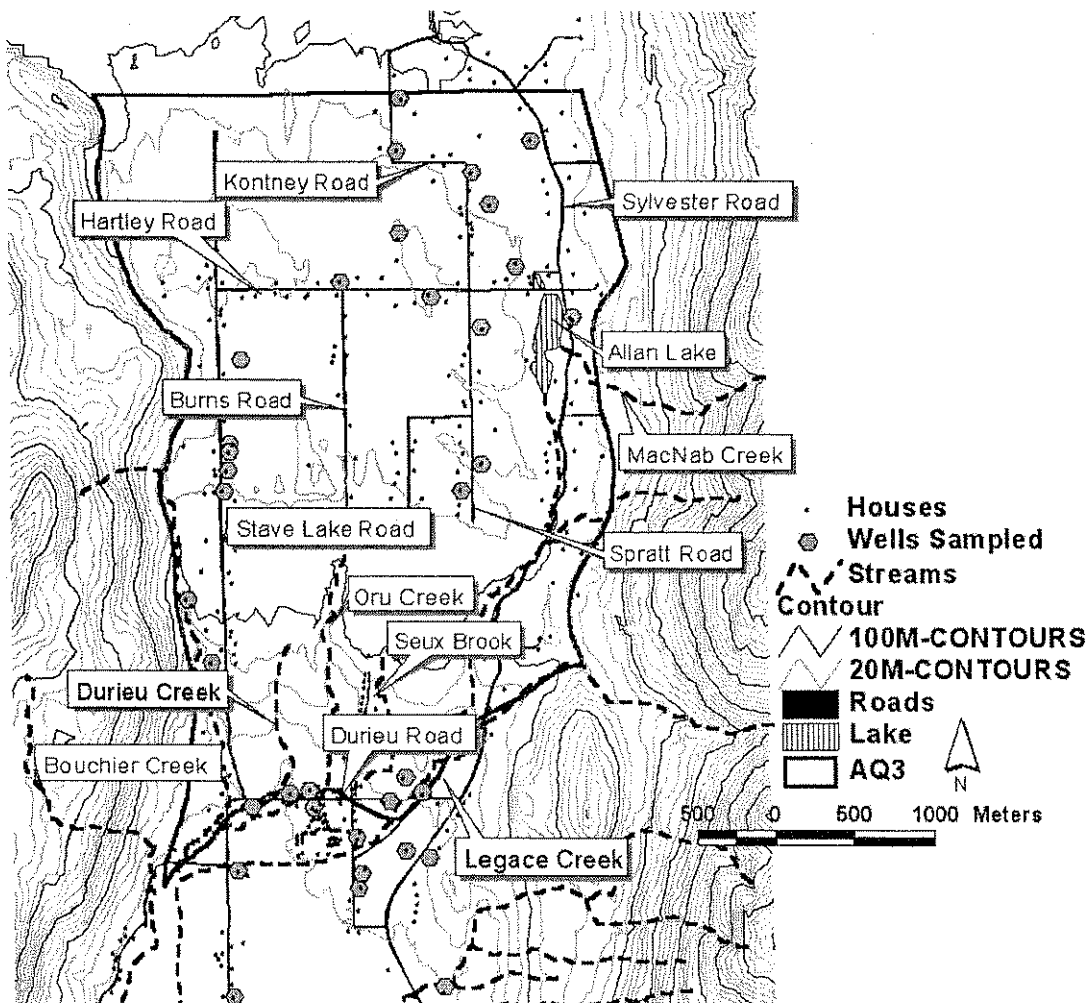


Figure 6. AQ3 - Miracle Valley Aquifer

The most detailed groundwater investigations in the study area have taken place in the Miracle Valley area. Local residents have been concerned about possible impacts on their groundwater quality and quantity from blasting at a rock quarry in the northeast of the area (near the corner of Stave Lake Road and Hartley Road). Reports written by independent consultants have been commissioned, both by the quarry owners, as well as by a group of concerned local residents, to evaluate the possible impact on groundwater resources of this quarry. These reports have contributed to the knowledge of the local groundwater regime by consolidating known information about wells in the area. Residents, coordinated by members of the local ratepayers association, have gone door-to-door collecting information about well depths and locations and, where possible, the well logs for each well within a 1.5 km radius of the quarry. These reports contain information about the groundwater in the Miracle Valley at a finer scale than that used by Kreye and Wei in 1994. The consultants' reports, written in response to the quarry have identified localized "sub" aquifer regimes and made educated guesses about flow direction and recharge areas. The reports mentioned here include one report written by Alan Dakin of Piteau Associates of in 1994 for John Conroy and Neighbours, the concerned citizens group. This report was written at the time of the original quarry application. The other was written by Pacific Hydrology Consultants Ltd. in 2003 as an update to a report by the same company in 1994 of potential impacts by the quarry operation on groundwater quality and quantity. The former report will be referred to as "the Dakin Report" and the latter as "PHCL 2003". These reports confirmed that the predominant source of groundwater in the area is the deep confined regional aquifer (AQ3) but that in addition to this source there are several shallow perched aquifers in the area (this is consistent with Luttmerding, (1980a)). These perched aquifers are evident by the abundant ponds on the western edge of the valley and "consist of sandy lenses in the clay unit that can likely only sustain a few low yielding wells"(Piteau Associates, 1994). Many low yielding bedrock units scattered around the margins of the valley were also identified as the possible source of some groundwater in the region. PHCL 2003 commented on the possible flow directions for groundwater in the area and concluded that, although much of the groundwater flows south towards Hatzic Lake, there is a groundwater divide in the valley and some of the groundwater in the area flows north towards Stave Lake. It is not known if this divide closely mirrors the watershed divide that is present towards the north of the Miracle Valley.

The location of groundwater recharge areas has been a concern of local residents worried about land use impacts on their water quality. Several theories about the location of recharge areas

were given in the consultants' reports. It was hypothesized in the Dakin Report that the aquifer was partly recharged from Stave Lake and likely also recharged from side creeks. PHCL 2003 thought it most likely that recharge was largely from the streams entering the valley on the east side. Without more detailed subsurface investigation it is hard to know exactly where the recharge is from. The Dakin report also identified areas of groundwater discharge. These will be discussed in a later section about the interactions between the groundwater and surface water.

Table 6. Land Use AQ3		
Land use	AQ3 (area ha)	%
Agriculture	83.059	6.99
Forest	879.165	73.99
Urban	-	-
Rural Residential	184.304	15.51
Other^a	38.522	3.24
Total	1188.223	100.00
a = Other includes (water, road, wetland/non-forested riparian, greenhouse)		

Miracle Valley was the only area of the study where there were enough well logs available in the BC well database to allow for the creation of profiles. Three cross-sections (profiles) were created using well logs from the BC government "find a well" web site. This labour intensive process involved searching this database for all possible well logs in the study area. Many well logs contained incomplete information. For every well log that was determined to be in the Hatzic Valley the exact location had to be found and mapped. Many logs were not useable because the log did not have adequate street or lot information to allow the location to be mapped. Other logs did not contain any profile information. The logs that did have profile information and could be given a specific location within the aquifer varied greatly in detail. Profiles have also been created in the vicinity of the quarry by PHCL and Dakin and can be found in their respective reports.

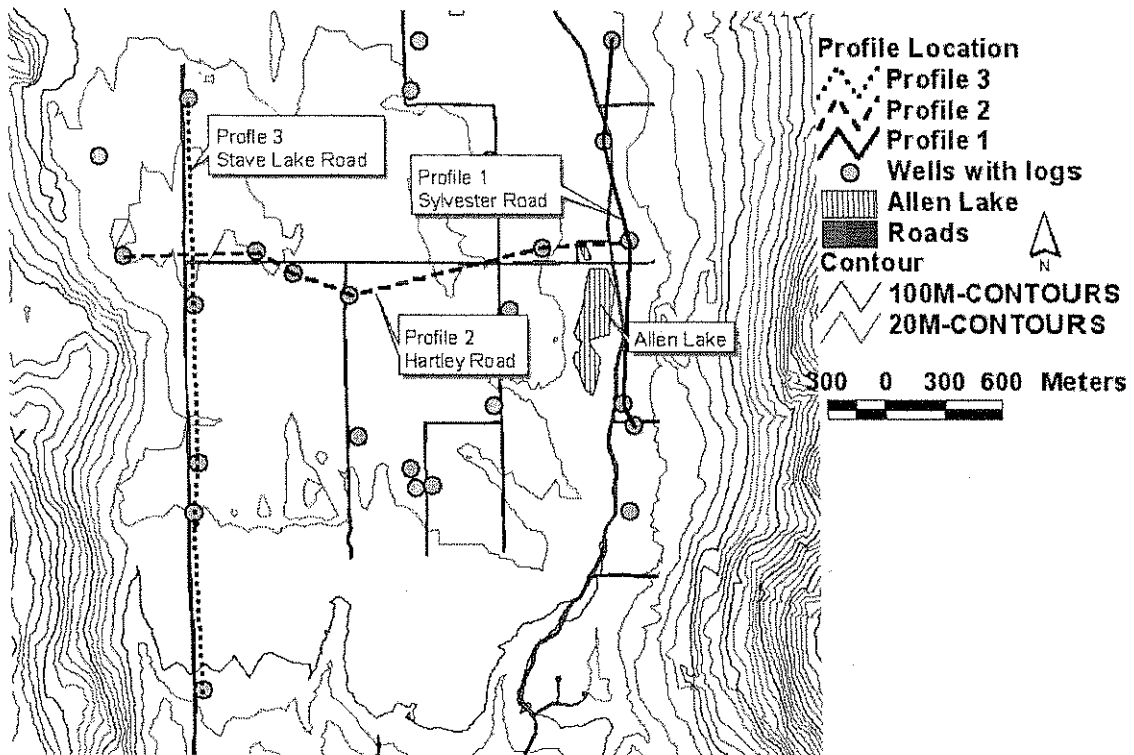


Figure 7. Locations of Well Log Profiles

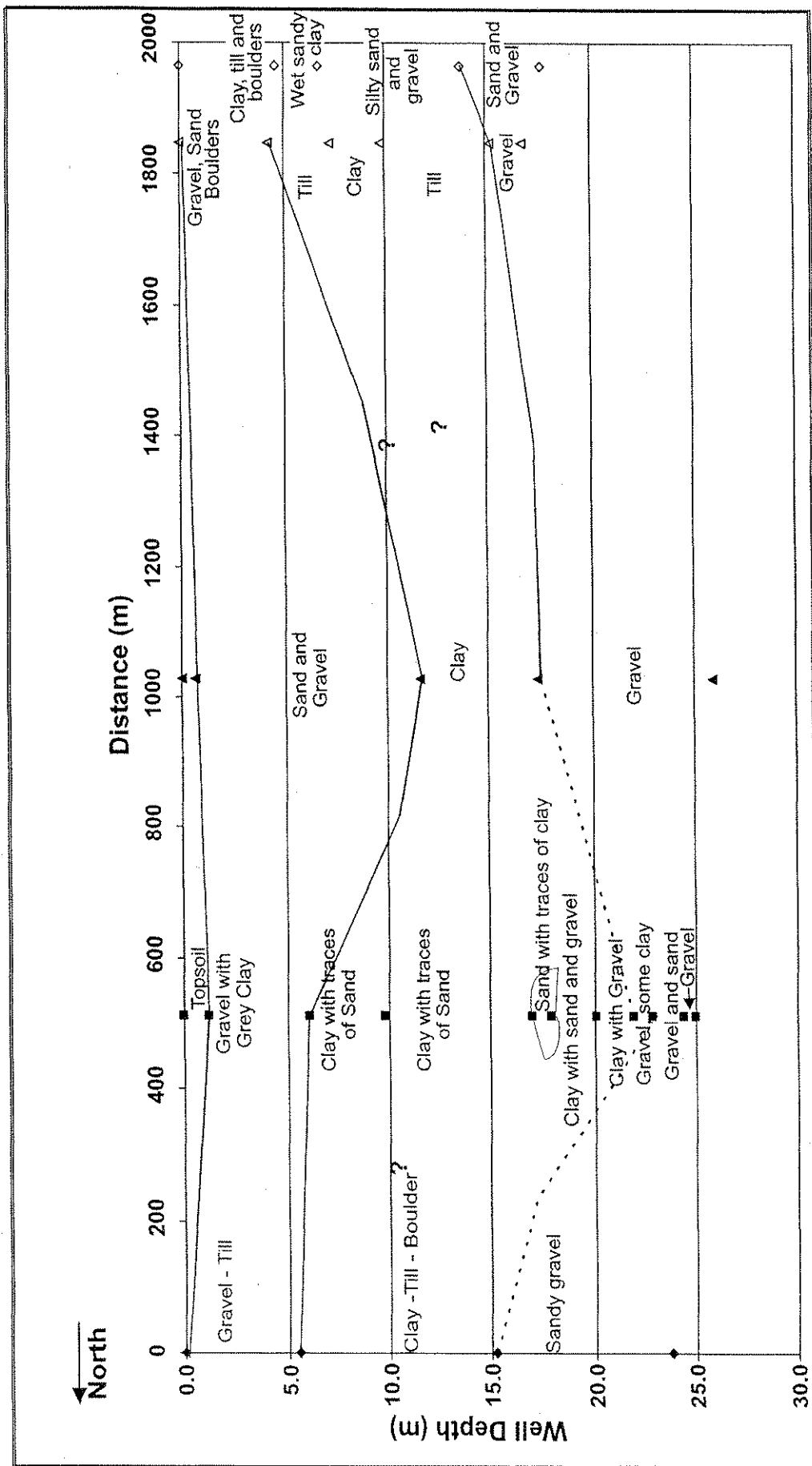


Figure 8. Well Profile 1. Sylvester Road

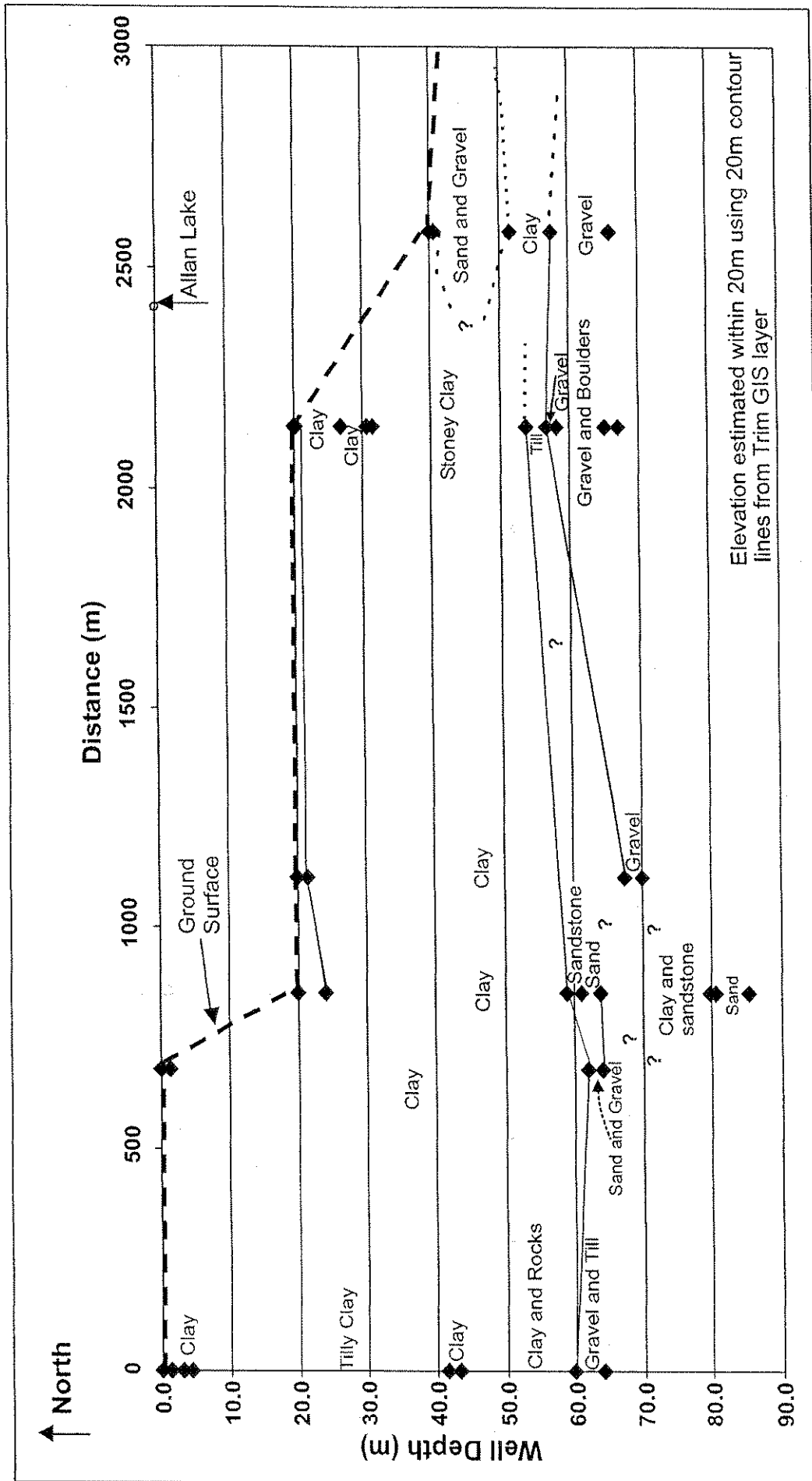


Figure 9. Well Log Profile 2. Hartley Road

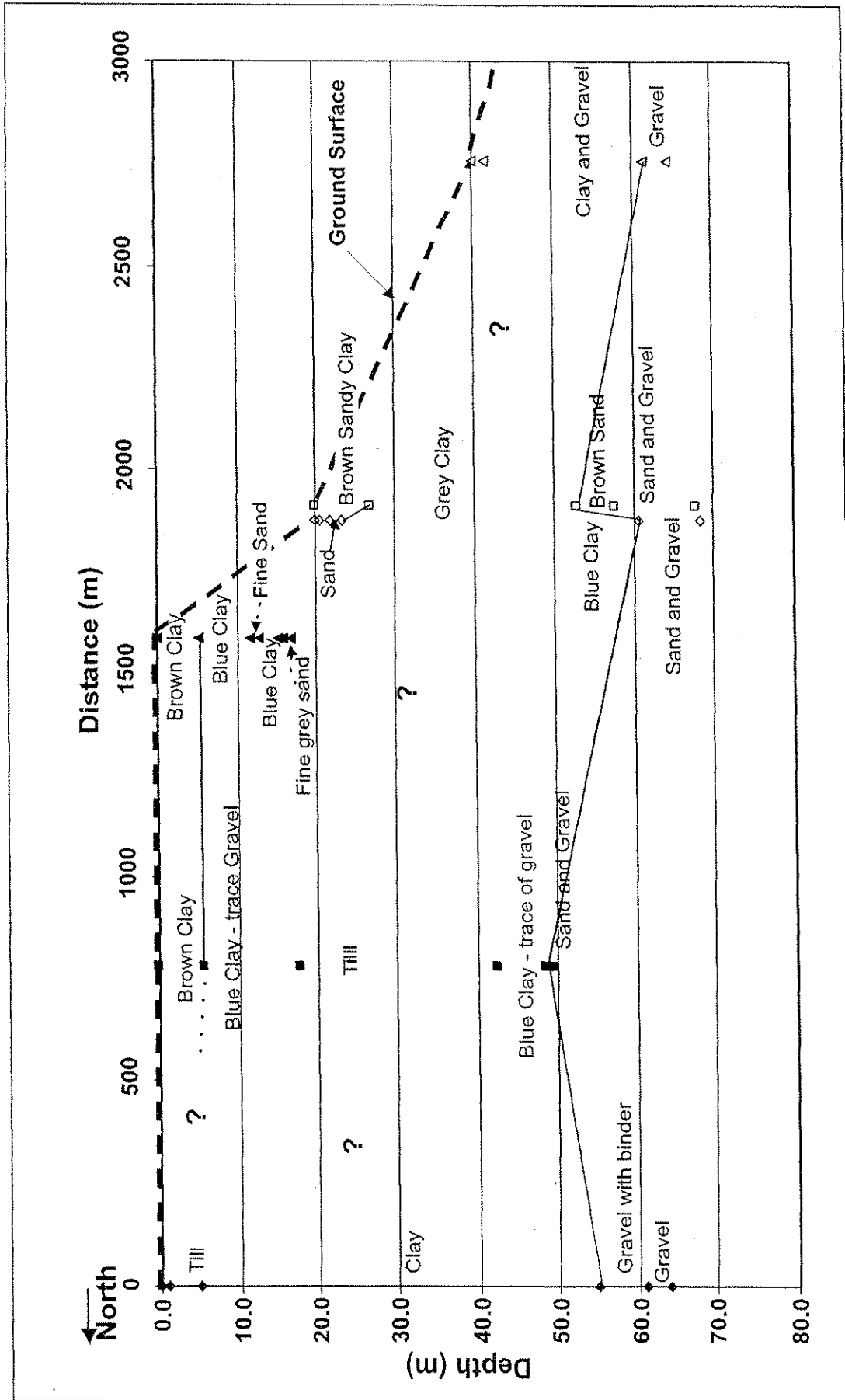


Figure 10. Well Log Profile 3. Stave Lake Road

3.0 Methodology

Three sources of data were used for the majority of the analysis of this project. The first was water quality data obtained from the analysis of surface and groundwater samples, the second was survey results from a survey given to all residents who supplied groundwater samples and the third was land use information created using a Geographical Information System (GIS). Land use throughout the valley was mapped and then detailed descriptions of the land use surrounding each well was calculated using the features of the GIS program. Finally, non-parametric statistical techniques were used to quantify the relationships between the three types of data. The following sections describe in detail the methods used for data collection and analysis.

3.1 Groundwater Survey and Questionnaire

3.1.1 Participant Selection

The method for the collection of groundwater samples was based on a previous groundwater investigation in the Lower Fraser Valley (LFV) (Schreier *et al.*, 1996). Two hundred residents from within the study area were invited by letter to participate in the groundwater survey. In return for providing groundwater samples at two different times of year the participants received free nitrate analysis of their well water. Potential participants were chosen from a database

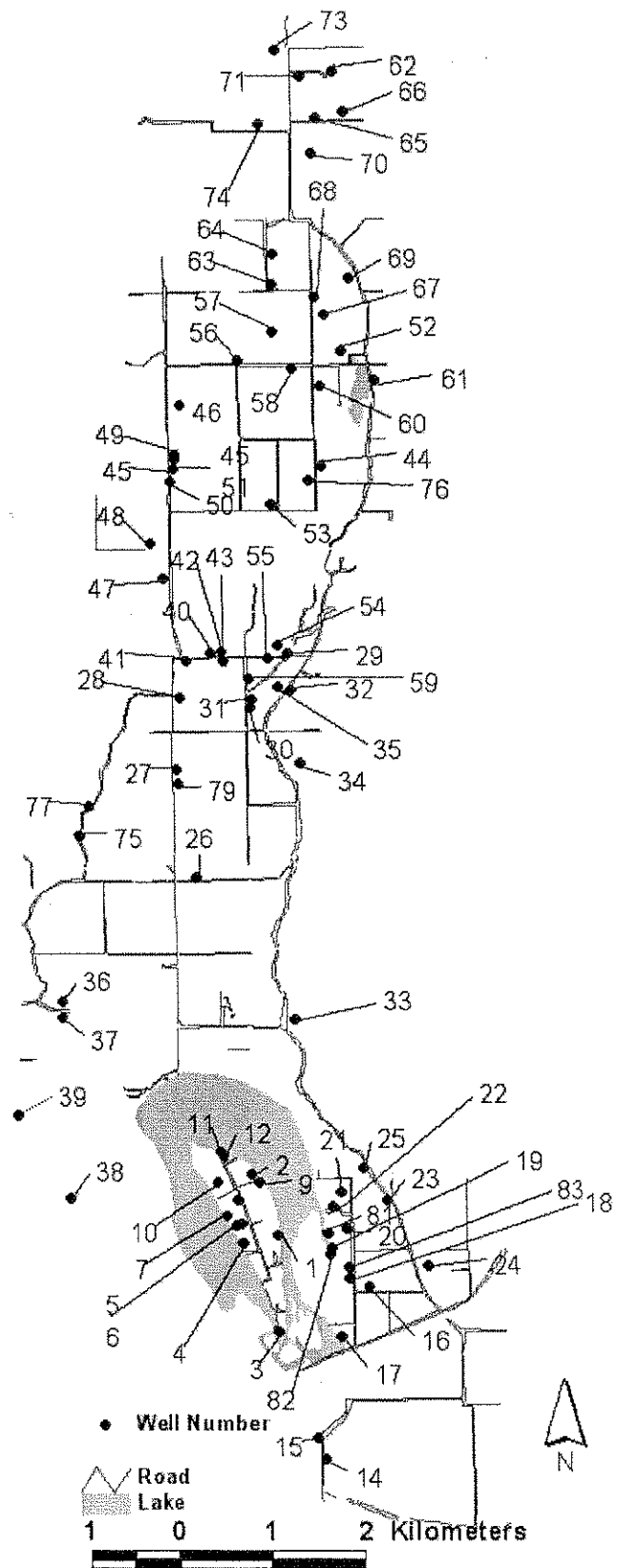


Figure 11. Location of Wells Sampled

provided by the Fraser Valley Regional District (FVRD). The database initially had 960 entries representing each property in the valley. Based on the owner address field, non-permanent residents were removed from the list. Many residents own more than one property in the valley and the list was modified so that each permanent resident only appeared in the list once. From this modified list 200 property owners were selected to receive an invitation to participate in the study. The 200 names were randomly chosen from all areas in the valley in order to maximize the spatial variation of the samples. In addition to the letters sent to these 200 participants a press release about the study was published in a local newspaper inviting the general public of the area to participate as well (see Appendix 1 for copy of the letter of invitation and Appendix 2 for a copy of the Press Release). Bill Walsh, the North Fraser Valley Watershed Stewardship Coordinator volunteered to be the local contact for the project and collected the names of willing participants. Approximately 75 households volunteered to participate in the study (see figure 11 for location of wells sampled.)

Private wells were sampled because this is the most practical way to sample many different sites and get an overview of the water quality in the region without creating new infrastructure (boreholes, monitoring wells, etc.). It is recognized that the testing of private wells may highlight localized water quality problems rather than regional problems thereby giving a false impression of the overall quality of water in the aquifer (Baker, 1990). However, the recognition of local problems is crucial for the individuals using the wells and, as the purpose of this study was to investigate the quality of drinking water, sampling tap water was considered to be the best way to determine the quality of the groundwater at the point of use. Many other studies have found the testing of private wells to be a practical method of gathering regional point of use groundwater quality data (Goss *et al.*, 1998; Legault, 2000; Jaques and Rohde, 2001; Rutter *et al.*, 2000; Murphy, 1992; Borchardt *et al.*, 2003; Tuthill *et al.*, 1998; Reid *et al.*, 2003; Baker 1990; Schreier *et al.*, 1996; Schreier *et al.*, 2003).

3.1.2 Groundwater Survey - Data Collection

Groundwater samples were collected twice, on July 28th, 2002 and on March 10th, 2003. The sampling was done twice in order to observe seasonal variation in the parameters measured and to sample when the water table was at different heights. Three teams of volunteers organized by Leigh Lehmann, secretary of the McConnell Creek/Hatzic Prairie rate-payers association, helped to distribute the bottles and to collect the samples from around the valley. Volunteers distributed sample bottles a week before sampling. Each participant was given a sterile (acid washed) sample bottle along with a short survey and instructions for how to take the water sample (see appendix 3 and 5). On the sampling date, participants filled the sample bottle with tap water (water from their well) between 7 and 9 am. They were asked to run their taps for 5 minutes before filling the bottles in order to minimize the influence of their water pipes on the quality of the water. In the summer, care was taken to ensure that the bottles were not left in direct sunlight so as to

not degrade the nitrate in the samples. Residents were notified of the results of their analysis within two weeks of sample collection (see appendix 4 for sample of the groundwater results letter.)

3.1.3 Chemical Analysis

The collected water samples were put on ice and brought to the UBC pedology laboratory where they were analyzed for several chemical parameters. pH was measured using an Orion 420A pH meter with auto-calibration using two buffers. Conductivity was measured using a Bach-Simpson limited Conductivity Meter type CDM 2e, No. 121323 with an electrode from radiometer analytical; part 945-000. The pH meter and the conductivity meter were calibrated to known standards prior to each use. A suite of 21 elements including aluminium, arsenic, boron, barium, calcium, cadmium, cobalt, chromium, copper, iron, potassium, magnesium, manganese, molybdenum, sodium, nickel, phosphorus, selenium, silicon, strontium and zinc were measured using a Varian Vista Pro Radial CCD Simultaneous ICP-OES. Nitrate-N, Ammonia-N and Orthophosphate-P were analyzed using a QuikChem FIA+ 8000 Series model from Lachat Instruments. The nitrate was analysed by QuikChem method 12-107-04-1-B. In this method nitrate is quantitatively reduced to nitrite by passage of the sample through a copperized cadmium column. The nitrite (reduced nitrate plus original nitrite) is then determined by diazotising with sulfanilamide followed by coupling with N- (1-naphthyl) ethylenediamine dihydrochloride. The resulting water-soluble dye has a magenta colour which is read at 520 nm (QuikChem method 12-107-04-1-B). The orthophosphate was analyzed by QuikChem method 10-115-01-1-A. In this method the orthophosphate ion reacts with ammonium molybdate and antimony potassium tartrate under acidic conditions to form a complex. This complex is reduced with ascorbic acid to form a blue complex that absorbs light at 880 nm. The absorbance is proportional to the concentration of orthophosphate in the sample (Diamond, 1995). Ammonia is analysed by QuikChem method 10-107-06-2-A the principle of which is as follows, when ammonia is heated with salicylate and hypochlorite in an alkaline phosphate buffer an emerald green colour is produced which is proportional to the ammonia concentration. The colour is intensified by the addition of sodium nitroprusside (Prokopy, 1992).

Quality was assured by using bottles that were acid washed prior to use and by the requirement to run the tap for 5 minutes prior to filling the sampling bottle. The laboratory technicians use standards every 10 samples to ensure proper calibration of the equipment. Two blind duplicates were used in the winter sampling. The results of the duplicates are found in appendix 6.

3.1.4 Coliform Analysis

Selected samples that had nitrate-N levels above 4 mg/L were analyzed for total and fecal coliforms by the British Columbia Centre for Disease Control (BCCDC). The following method was used for coliform enumeration. All samples are filtered as follows: put a 0.45µm filter on the base of the funnel then put

the funnel on. Pour in about 20ml of buffer and pipette an aliquot of water (for the watershed and surface water, 1 and 0.1ml was chosen for most of the samples) into the buffer. Turn on the funnel tap and drain the sample and buffer, then rinse the sample 2 xs with about 30ml of buffer. For total coliform, take the filter out and put onto Les Endo Agar plate. Incubate the plate for 22-24 hours at $35 \pm 0.5^{\circ}\text{C}$. For fecal coliform, take the filter out and put onto m-FC Agar plate and incubate the plate for 24 ± 2 hours at $44.5 \pm 0.2^{\circ}\text{C}$.

After incubation, all plates are read using stereomicroscope: 1) for total coliform, typical colonies (are those form green metallic sheen one) and atypical colonies (are dark red with or without sheen) will be counted and verified. Translucent or pale pink colonies are non-coliforms and won't be verified. Typical and atypical colonies will be transferred to lauryl tryptose broth and Brilliant Green bile salt broth and incubated at $35 \pm 0.5^{\circ}\text{C}$ for confirmation. Gas formed in brilliant green bile salt broth will be considered positive total coliform. 2) For fecal coliform, typical colonies (blue colonies) and atypical colonies (greyish blue, greenish blue) will be verified by transferring to lauryl tryptose broth and incubated at $35 \pm 0.5^{\circ}\text{C}$ for 48 hours. If lauryl tryptose broth is positive (gas formation), a loopful will be transferred to EC broth for confirmation and incubated at 44.5°C for 24 hours. Positive (gas formation) EC will be considered as Fecal positive. After all confirmation is done, count will be calculated according to the confirmation results (Wong, 2004).

3.1.5 Opinion Survey

A short survey, based on a previous survey distributed by Schreier *et al.*, (1996), was included with the sampling bottles to get more information about the wells sampled such as the age and depth of the wells as well as information about the habits and perceptions of the owners of the sampled wells. Questions were asked about water treatment, land use near the wells, septic systems and perceptions of the importance of various activities in degrading water quality. A copy of the survey is included in appendix 5.

3.2 Stream Sampling

Stream samples were taken from 19 locations throughout the study area between May 2002 and May 2003. pH, Conductivity and temperature were measured in stream using and Oakton Waterproof pH/conductivity deluxe meter kit: model no. WD-35631-60. Since temperature and distance affect the conductivity levels the probe is constructed with electrodes a standard distance apart (1cm) and the ability to correct the temperature to 25°C . Conductivity measurements that are corrected for temperature are called specific conductivity measurements. Nitrate-N, Ammonia-N and Orthophosphate-P were analyzed in the lab using the same equipment and methods as for the groundwater samples (see section 3.1.3).

Figure 12 shows the location of the stream sampling stations. Samples were taken in sterile (acid washed) bottles that were rinsed twice with water from the sampling location prior to collection.

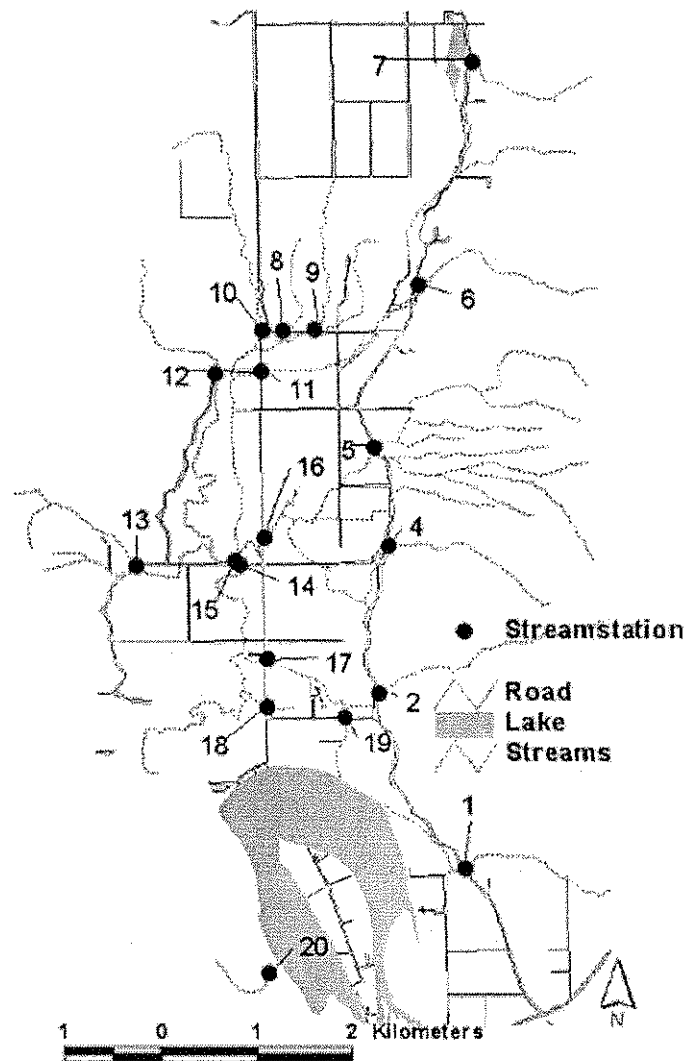


Figure 12. Location of Stream Sampling Stations

3.3 GIS Database.

A spatial database was created in the Arc View 3.2 Geographical Information System. Locations of the wells sampled were plotted using a combination of orthophotos and CAD (Computer Assisted Design) drawings supplied by the FVRD. The exact locations of the wells were not plotted but rather the location of the house on the property where the wells were located. It was assumed that the wells were relatively close to the homes they were serving.

A map representing the land use in the valley was created using the orthophotos provided by the FVRD. See table 7 for a summary of the land use categories created and figure 2 for the entire land use map. From these 11 categories, 4 were used in the analysis; forest (5), agriculture (9), rural residential (clearing) (7) and urban (trailer park and subdivision combined) (4,6). The rural residential category

encompassed all of the non-agricultural clearings and was intended to capture land that was non-forested but was not being used to produce crops. It was assumed that this land would have different inputs than the agricultural fields. Most polygons categorized as rural residential included houses and some included barns and barnyards as well. Forest cover, agriculture, rural residential and urban were chosen as the final four land use categories because they covered the largest amount of the area around the wells and therefore they could be considered to be the dominant land uses in the valley. These four categories also represent different potential stresses to the groundwater system. For example it would be expected that forested land would not be receiving large inputs of nutrients, the agricultural land would have large nutrient inputs due to applications of manure and commercial fertilizer and the urban areas would receive nutrient inputs from septic systems. Each well was also given a location code that corresponded to the aquifer it was in. The locations were determined from the GIS maps of the aquifers from the British Columbia Ministry of Water Land and Air Protection (WLAP) and only take into account difference in location and not in depth when grouping the samples.

Table 7. Land use Categories Coded into GIS	
1	Big Streams
2	Greenhouses
3	Lake
4	Trailer Park
5	Forest
6	Subdivision (Dense Residential)
7	Clearing (Extensive rural Residential, including hobby farms)
8	Road
9	Agricultural
10	Wetland/ non forest/non agricultural riparian
11	Hydro Cut

Using the buffer function of the GIS software, circles (called buffers) with radii of 50, 100 and 200 m were created surrounding each well. These buffers were then used to clip a corresponding circle from the land use map around each well (see figure 13.). The area of each land use type within each buffer was calculated and then used to calculate percentages of land use type within each buffer. A similar process was used to calculate the percentage of land use associated with the streams. In this case 50 and 100 m buffers were created around line data (streams) rather than point data (wells). In the case of the stream stations, the percentage of land use was calculated for the entire stream network upstream of that point meaning that the land use percentages are cumulative. Figure 13 shows what the 100 m buffers around the wells on AQ1 look like.

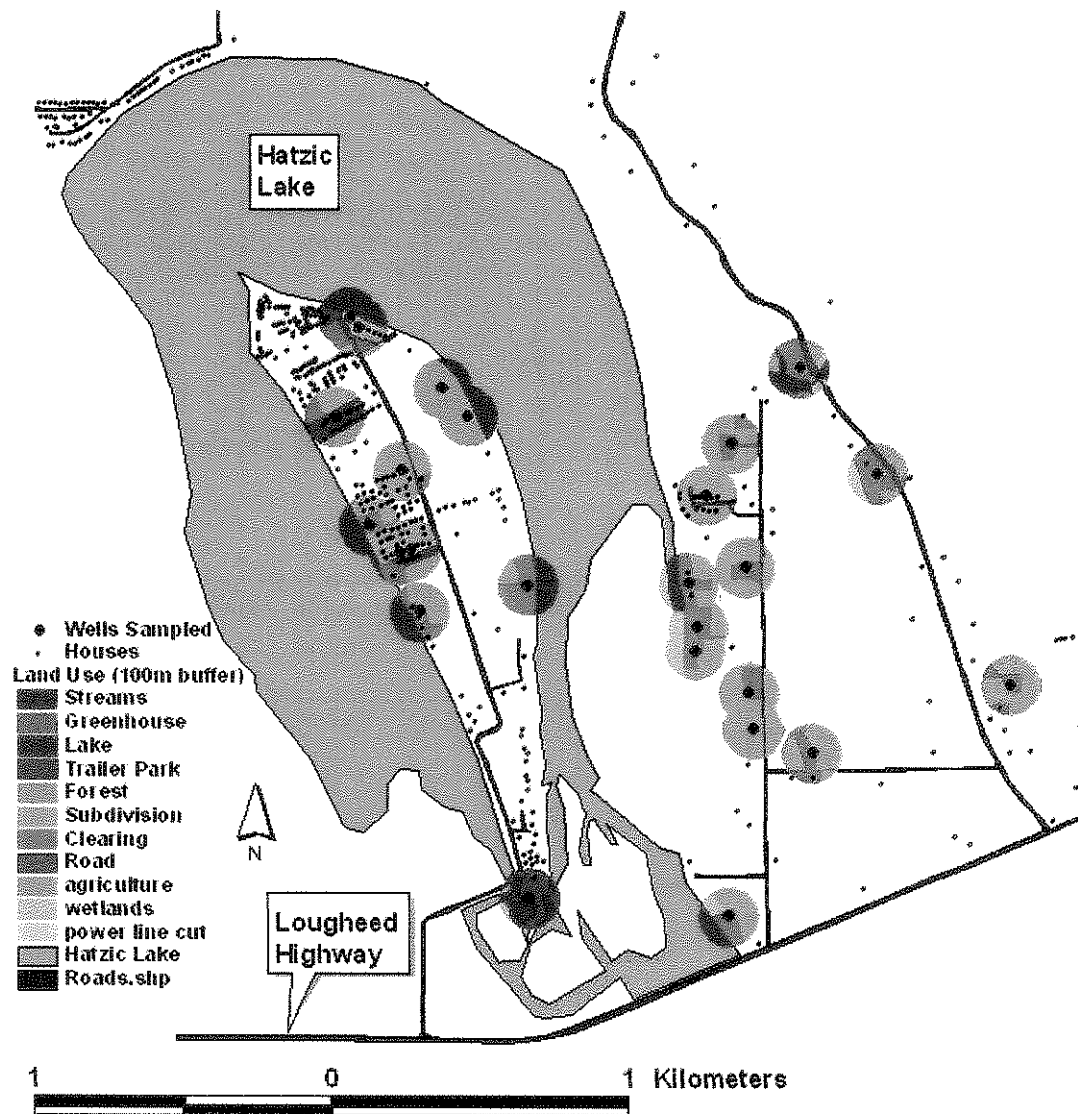


Figure 13. Land use buffers surrounding wells in AQ1

Information on the soil in the area was taken from *Soils of the Langley-Vancouver Map Area* Volumes 1 and 2 (Luttermiding 1980a and 1980b). The soil type for each well was determined visually and then coded into the data base. The information taken from the soils map is listed in appendix 7. Surficial Geology was added to the GIS in the same manner as soils information. The surficial geology was taken from Armstrong (1980) *Surficial Geology Mission, British Columbia*. Appendix 7 also shows the texture classification created from the surficial geology map. The categories were coded so that lower numbers represented the least permeable option and the higher numbers represented more permeable conditions.

3.4 Statistical Analysis

Data analysis was performed using non-parametric statistical techniques. These methods, although less powerful than traditional techniques, are useful when the assumptions of parametric procedures are not met. Non-parametric techniques are used when data is not normally distributed and when some of the

3.4.3 Kruskal-Wallis Test

The Kruskal-Wallis (K-W) test is the non-parametric equivalent to the analysis of variance (ANOVA). This test is used to test for significant differences between three or more groups using the following hypothesis.

H_0 : The populations all have the same median.

H_1 : At least one of the populations has a significantly different median.

The results of the Kruskal-Wallis test do not show which groups are different from the others; a rejection of the null hypothesis only tells us that at least one of the groups is significantly different than one of the others. When using parametric methods, procedures such as Bonferroni, Tukey and Sheffe are available to test which groups are different from each other after running a one-way ANOVA. These techniques have been developed to determine which groups are different because the number of comparisons made will influence the probability of making a type I error (saying that there is a difference when the two groups are really from the same population). For example, if one made six comparisons in this manner the overall probability of making a type I error would be 0.26 rather than 0.05 (Zar, 1984). Many descriptions of the Kruskal-Wallis test do not give any instructions on how to determine which groups are different following this test (Siegel, 1956; Statsoft, 2002). The most commonly quoted source of methods for similar non-parametric procedures is Zar (1984) but these are not included in the analytical capabilities of the SPSS software. In order to continue to use SPSS for statistical analysis but to attempt to lessen the increase in type I error that results from making multiple comparisons using the Mann-Whitney test, the following method will be used to determine which groups are different from each other after running a Kruskal-Wallis test. If three or more groups are to be compared a K-W test will be used. If the null hypothesis is rejected, meaning that at least one of the groups is different, boxplots will be created to give a visual representation of what is being tested. In this case the boxplots will be used to visually determine which medians are closest together. The Mann-Whitney U test will be used to determine if these medians are significantly different from each other. If they are, no further Mann-Whitney tests need to be used as it follows that if the closest medians are significantly different from each other the other medians will be too and it will not be necessary to test the other variables. If the first medians tested are not significantly different from each other subsequent Mann-Whitney tests will be used.

3.4.4 Boxplots

Boxplots are a method of graphically displaying data that shows the dispersion of the data. In SPSS the box consists of the median and the interquartile range. The interquartile range includes 50% of the distribution. The whiskers on the plot cannot extend more than 1.5 times the length of the box (the interquartile range). If there are values greater than 1.5 times the interquartile range they are considered outliers. Values greater than 3 times the interquartile range are considered extreme values. Outliers are represented by a circle "o" on the boxplots and extreme values are represented by an asterisk "*".

3.4.5 Cluster Analysis

The purpose of cluster analysis “is to join together objects into successively larger clusters, using some measure of similarity or distance” (Statsoft, 2002). Cluster analysis is not a statistical technique which has significance associated with it. It is a descriptive technique that is used to look for patterns in data as an exploratory step in data analysis. Usually this procedure is used in order to find unknown patterns in data. In hierarchical cluster analysis (the type used for this analysis) each object begins in a separate cluster and they are gradually linked together into one large cluster. There are two measures that must be chosen when performing a cluster analysis. The first is the distance measure. The distance measure determines how far apart each individual case is from all other cases. The Euclidian distance was used in this analysis because it is a good measure to use on data that has not been standardized. The second measurement is called the amalgamation or linkage rule. These rules determine how the distance between groups is measured. The method chosen for this analysis is called ‘complete linkage’ or ‘furthest neighbour’. This method measures the furthest distance between members of different clusters to determine which cluster to link next. This method was chosen through a combination of trial and error and because it was recommended for groups that form naturally distinct clumps. The output from the analysis is typically a dendrogram and it is left to the discretion of the researcher to determine how many clusters are appropriate.

4.0 Results and Discussion

The results of the analysis of the three types of data mentioned in section 3 (water quality data, GIS data and survey data) are presented in this section. First, the overall groundwater quality is discussed with a focus on those parameters that exceeded the Guidelines for Canadian Drinking Water Quality (GCDWQ). Then an examination of the relationships between parameters is made including seasonal variation, variation with depth and correlations between the water quality parameters. Because groundwater samples were taken from three separate aquifers, analysis was done to determine if the water from the different aquifers had distinct chemical signatures. Using the GIS database and the water quality database an examination of the interactions between groundwater quality and land use was made. The next section of analysis focuses on the surface water quality and the interactions between these parameters and land use. The following section is a discussion about interactions between groundwater and surface water. The final section of analysis looks at the results of the opinion survey and the relationships between perceptions of water quality and the physical parameters that were measured as well as regional difference in the survey responses.

4.1 Groundwater Quality

Twenty-six water quality parameters were measured for each sample of groundwater taken. Of these, 21 were elements measured using the ICP (see section 3.1.3 for the complete list of elements measured). Only 13 of these elements were found above the method detection concentration (MDC) in any of the samples. Due to changes in analytical procedure the detection limits varied between winter and summer sampling for a number of elements. In some cases this drastically changed the number of samples above the detection limit. Table 8 outlines the summary statistics for all parameters that had at least one sample in winter or summer above the MDC. The mean, median, standard deviation, maximum and minimum recorded value as well as the detection limit and the number and percentage of samples that were above the detection limit are listed. Due to an equipment problem with the Lachat the winter samples for orthophosphate were invalid, therefore only the July PO_4 data will be used. Aluminium, barium and boron concentrations were predominantly below the detection limit and are therefore excluded from the analysis.

Table 8. Groundwater Sampling Results: Summary Statistics for Each Variable with at Least One Sample above the Detection Limit (MDC)^b

Element	MDC ^b (mg/L)	# Above detection	% Above detection	Mean	Median	Max	Min	Standard Deviation
EC_S ^{a,c}	N/A	76	100%	217	115	3200	30	503.0
EC_W	N/A	75	100%	219	123	3290	17	515.2
pH_S ^a	N/A	76	100%	6.68	6.40	8.35	5.82	0.693
pH_W	N/A	75	100%	6.69	6.37	9.03	5.73	0.79
PO4_S	0.050	56	74%	0.167	0.084	0.542	0.050	0.142
PO4_W	<i>Orthophosphate results in winter were inaccurate due to analytical problems.</i>							
NO3_S	0.050	61	80%	0.938	0.362	10.129	0.050	1.60
NO3_W	0.050	60	80%	1.242	0.498	11.962	0.050	1.973
NH3_S	0.050	8	10%	0.076	0.050	0.988	0.050	0.132
NH3_W	0.050	26	35%	0.068	0.050	0.826	0.050	0.091
Al_S	0.1000	0	0%	-	-	-	-	-
Al_W	0.050	2	2.7%	0.050	0.050	0.076	0.050	0.003
B_S	0.100	5	6.6%	0.183	0.100	3.704		0.467
B_W	0.010	7	9.3%	0.106	0.014	3.366	0.010	0.439
Ba_S	0.5000	0	0.0%	-	-	-	-	-
Ba_W	0.010	20	27%	0.013	0.010	0.052	0.010	0.0086
Ca_S	0.50	75	99%	15.40	12.28	183.76	0.50	21.923
Ca_W	0.10	75	100%	14.04	11.65	200.20	0.13	23.378
Cu_S	0.025	36	47%	0.076	0.025	0.708	0.025	0.117
Cu_W	0.050	20	27%	0.096	0.050	1.388	0.050	0.176
Fe_S	0.200	10	13%	0.251	0.200	1.155	0.200	0.170
Fe_W	0.050	20	27%	0.150	0.050	2.175	0.050	0.3229
K_S	0.10	73	96%	1.33	0.49	18.72	0.10	2.801
K_W	0.50	34	45%	1.36	0.50	16.89	0.50	2.55
Mg_S	0.025	75	99%	4.845	2.761	31.377	0.025	5.960
Mg_W	0.010	75	100%	4.445	2.740	28.581	0.091	5.248
Mn_S	0.050	10	13%	0.070	0.050	0.718	0.050	0.086
Mn_W	0.005	31	41%	0.029	0.005	0.352	0.005	0.06
Na_S	1.000	75	99%	23.778	4.018	588.915	1.000	88.633
Na_W	0.250	68	91%	21.003	2.665	573.931	0.250	82.48
Si_S	0.500	76	100%	8.650	7.714	19.125	2.517	3.900
Si_W	0.150	75	100%	8.034	7.072	19.676	1.813	3.978
Sr_S	0.002	58	76%	0.074	0.047	0.850	0.020	0.104
Sr_W	0.002	73	97%	0.060	0.038	0.802	0.002	0.099
Zn_S	0.200	5	6.6%	0.336	0.200	6.677	0.200	0.803
Zn_W	0.010	33	44%	0.153	0.010	3.338	0.010	0.551

a = all parameters are measured in mg/L with the exception of Conductivity which was measured in $\mu\text{S}/\text{cm}$ and pH which was measured in pH units

b = Method Detection Concentration (detection limit)

c = _S = results for samples taken in summer (July 28, 2002) (n= 76)

_W = results for samples taken in winter (March 10, 2003) (n=75)

4.1.1 Parameters exceeding the Guidelines for Canadian Drinking Water Quality

Some wells exceeded the Guidelines for Canadian Drinking Water Quality (GCDWQ) for one or more parameter. Table 9 outlines the maximum allowable concentration (MAC) for these parameters and the number of wells in each sampling period that exceeded the guidelines. For the purpose of comparison, table 2 in chapter 1 outlines the number of exceedances of the same elements from a previous study that looked at water quality of community wells throughout the Lower Fraser Valley (Carmichael *et al.*, 1995).

Table 9. Number of wells exceeding the Guidelines for Canadian Drinking Water Quality, Hatzic Valley Sampling, 2002, 2003			
Parameter	MAC (GCDWQ)	# Winter n=75	# Summer n=76
Iron (Fe)	0.3 mg/L	8 (10%)	8 (10%)
Copper (Cu)	1 mg/L	1 (1.3%)	-
Zinc (Zn)	5 mg/L	-	1 (1.3%)
Manganese (Mn)	0.05 mg/L	10 (13.5%)	10 (13.5%)
Sodium (Na)	200 mg/L	2 (2.7%)	2 (2.7%)
pH	6.5-8.5 pH units	47 (63%)	45 (60%)
Nitrate-N (NO₃-N)	10 mg/L	1 (1.3%)	1 (1.3%)
GCDWQ = Guidelines for Canadian Drinking Water Quality MAC = Maximum Allowable Concentration			

4.1.1a pH

The most common exceedance in the study area was pH; approximately 60% (47 wells in winter and 45 wells in summer) of the wells sampled had pH below 6.5. Low groundwater pH in the LFV had been noted in previous reports (Dakin, 1994 and Carmichael *et al.*, 1995). The low pH is likely due to the granitic bedrock that makes up a large part of the Hatzic watershed. Granite substrates are a common cause of acidic waters. Low pH in itself is not a health concern but water with low pH can cause enhanced corrosion of metal in pipes (esp. copper) or in the substrate (iron and manganese), which can then lead to elevated levels of metal in water (Leitch, 2000).

4.1.1b Metals

Eight exceedances of iron and 10 of manganese were recorded for wells spread throughout the watershed (see fig. 14). Conversations with well owners suggested that high iron is a known problem for people who have drilled wells into AQ2 accounting for the high use of surface water for domestic consumption in this area. Based on the sampling results, however, all areas seemed equally prone to high iron and manganese levels with no immediately evident spatial trends (except that there were no exceedances on Hatzic Island). The aquifer classification report (Kreye and Wei, 1994) did not mention high iron as being a problem in AQ3; however, I found that 4 of 8 iron exceedances and 3 of 10 manganese exceedances were in wells on this aquifer, suggesting that there is the same potential for high iron to occur here as in the other two aquifers. As mentioned in the introduction, iron and manganese levels are

generally related to the substrate through which water flows before reaching the well and not from anthropogenic influence.

In addition to the exceedances of iron and manganese, one well had copper above the CGDWQ and one

had zinc above the limit. Corroding pipes have been identified as the most probable source of elevated levels of Cu and Zn in groundwater in the Lower Fraser Valley (Carmichael *et al.*, 1995). It is not surprising that a few homes in the valley have high copper and zinc based on the low pH of the water in the area.

A 1979 study entitled "a National Survey for Cd, Cr, Cu, Pb, Zn, Ca and Mg in Canadian Drinking Water Supplies" sampled the previously mentioned elements in surface and well water across Canada (Méranger *et al.*, 1979). In this study Cu and Zn levels were consistently higher in distributed water than in raw water (due to corrosion of pipes). We can compare the maximum values for untreated water in the 1979 study for Cu, Zn, Ca and Mg to the maximum concentrations of the same elements measured in the Hatzic watershed study (of the metals measured in the 1979 study these were the only ones in the present study above the detection limit). Some samples from the Hatzic Valley had values for Cu, Zn and Ca that greatly exceeded the highest level recorded in the

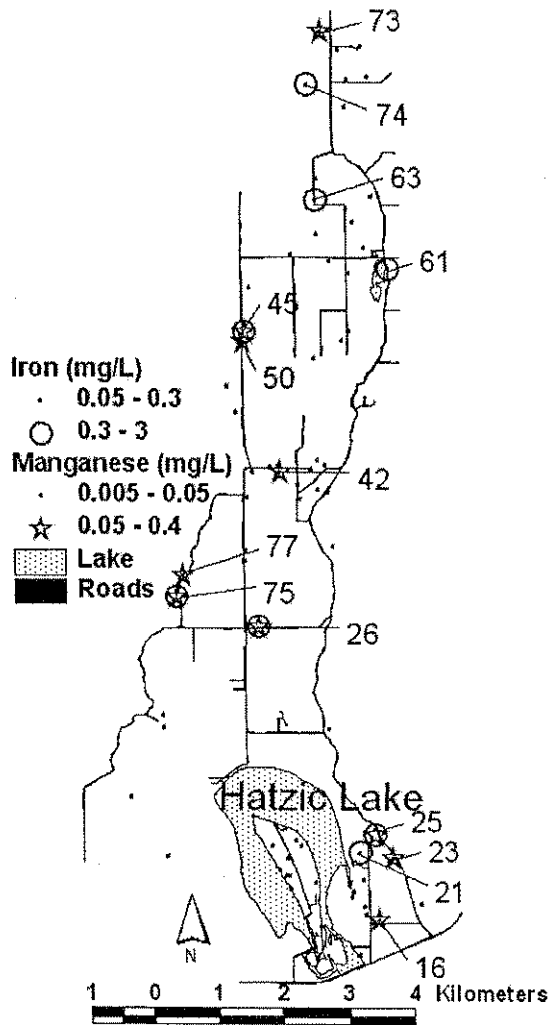


Figure 14. Wells exceeding the CGDWQ for Fe and Mn

1979 study (see table 10). This is likely because the study sampled community drinking water supplies rather than private wells. Communities often have more choice about which water supplies to develop than private well owners; a municipality may be able to reject water supplies with high metals in favour of another site, or drill a deeper well to access an aquifer with better water quality whereas a private well owner is limited to the water that is available on their own property and may not be able to afford to drill a deeper well.

Table 10. Comparison of maximum Cu, Zn, Ca and Mg levels in Hatzic Valley with results of Méranger et al, 1979

Element ^a	Max DW (1979 study)	Max RW (1979 study)	Max Hatzic Valley Summer	Max Hatzic Valley Winter	Number of Hatzic Samples above 1979 maximum
Cu	0.260	0.11	1.388	0.7079	3
Zn	0.130	0.02	3.33	6.67	9
Ca	70.1	67.6	200	183.1	1
Mg	54.7	81.7	28.5	31.3	0

a = all values are given in mg/L
DW= Distributed Well Water
RW = Raw Well Water

4.1.2 Conductivity

Conductivity had the greatest range of all parameters measured. Values covered three orders of magnitude from 17 $\mu\text{S}/\text{cm}$ to 3290 $\mu\text{S}/\text{cm}$. The values were not evenly distributed in this range and only six samples (8%) had values above 300 $\mu\text{S}/\text{cm}$ (see table 11). Figure 15 clearly shows how much higher these values are than the majority of conductivity value and figure 16 shows the conductivity plots with the six outliers removed.

Table 11. High conductivity wells

Well Number	Aquifer	Conductivity $\mu\text{S}/\text{cm}$		Sodium mg/L		Depth of Well
		Winter	Summer	Winter	Summer	
26	AQ2	3290	3200	422.13	497.25	91.44 (artesian)
77	AQ2	3100	3100	573.93	588.91	Unknown (artesian)
42	AQ2	1030	1000	121.008	137.61	60.96
40	AQ2	610	550	80.75	99.55	60.96 (artesian)
25	AQ1	390	410	27.42	31.80	7.62
45	AQ3	340	330	6.52	8.55	19.20

In the Lower Fraser Valley there are deep regional saline aquifers underlying many of the identified aquifers (Carmichael *et al.*, 1995). The water from wells 26, 77, 40 and 42 (which are all located on AQ2) are likely from this deep regional aquifer (see the sodium levels and depths presented in table 9). Three out of four of these wells are known to be flowing artesian wells and are among the deepest wells sampled. There are two wells sampled which are deeper than well 26 that do not have abnormally high conductivity or sodium levels, however these wells are located at several hundred metres higher elevation than well 26 and if the depths were corrected for elevation it is likely that wells 26, 77, 40 and 42 would be deeper on an absolute scale. Wells 25 and 45 do not fit the same pattern as the other four high conductivity wells, they are not extra deep nor do they have extraordinarily high sodium levels. The high conductivity levels of these wells are therefore due to different factors.

Figure 16 also shows that in general, once these six outliers are removed, the wells on aquifer one have higher conductivity values than those in AQ2 and AQ3 (the conductivity values of AQ1 are significantly ($\alpha = 0.05$) different than those of AQ3, the difference between AQ1 and AQ2 and AQ2 and AQ3 were not significantly different from each other).

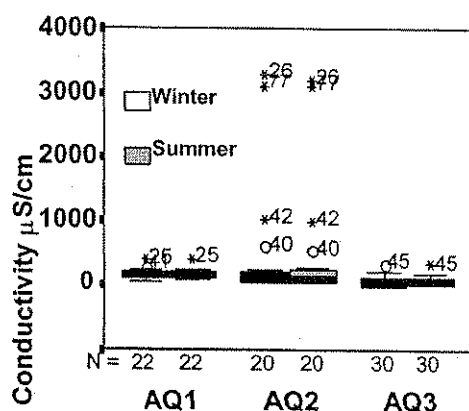


Figure 15. Boxplots of conductivity showing outliers

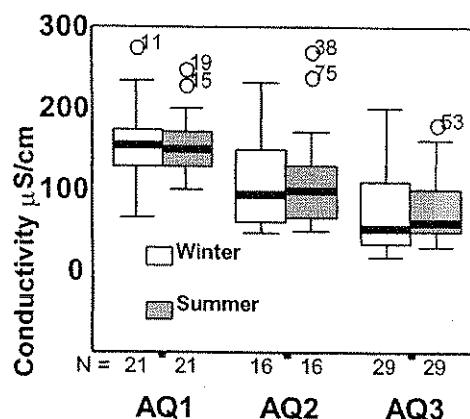


Figure 16. Boxplots of Conductivity, outliers (wells 26, 77, 42, 40, 25, 45) removed

4.1.5 Orthophosphate: PO_4

The orthophosphate results were surprising because they showed very strong regional variation. The highest levels of PO_4 were found in wells on Hatzic Island. These were significantly higher ($\alpha = 0.05$) than the wells from the area just east of the island and higher again than those found on AQ2 and AQ3 (the PO_4 levels in AQ2 and AQ3 were not significantly different from each other).

The results are worth mention because PO_4 is generally considered to be a parameter, like nitrate, for which the levels are determined by land use. The pattern seen in Hatzic suggests that the levels of PO_4 are influenced by different process than the nitrate levels. Several theories can be suggested to explain the observed pattern. Hatzic Island has the densest concentration of homes in the study area and each of these homes is on a septic system. Some studies have suggested that septic systems can release significant quantities of PO_4 into the subsurface. While the source of PO_4 may be from septic systems it must be kept in mind that all of the wells sampled on Hatzic Island had higher levels of PO_4 than the other areas of the valley but not all the wells sampled (even on Hatzic Island) were located in densely populated areas. The elevated PO_4 levels are seen consistently across the region.

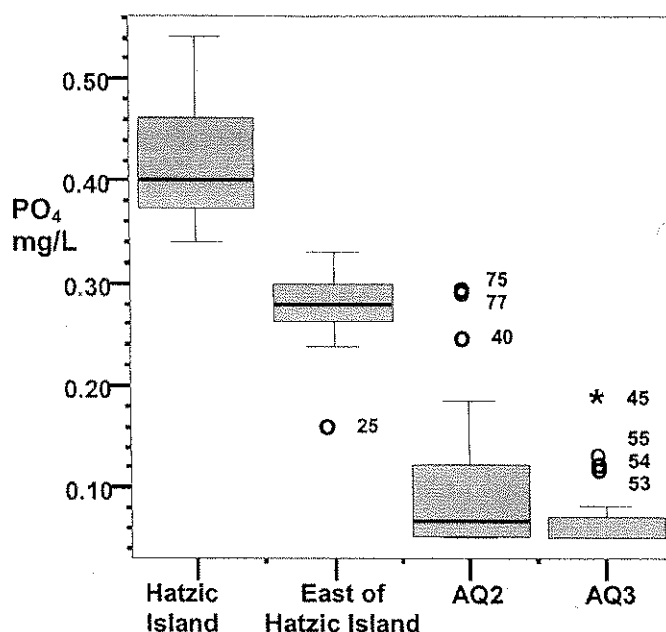


Figure 17. Boxplot of PO_4 grouped by aquifer (aquifer 1 split into two groups, Hatzic Island and area east of Hatzic island)

The water table on Hatzic Island is quite high and interaction between groundwater and water from Hatzic Lake would seem to be inevitable. No water quality data for Hatzic Lake was collected during this study but previous research from as far back as 1973 suggests that eutrophication has been a concern in this body of water for some time (Drinnan, 1975). All of the values of PO_4 measured in wells on Hatzic Island are above the levels considered to be of concern for eutrophication of streams or lakes (0.01 mg/L P level is often used as a standard for the prevention of lake eutrophication (Schreier *et al.*, 1996)). Since the flow and interaction between Hatzic Lake water and groundwater is not known it is difficult to determine if eutrophic lake water may be influencing the groundwater PO_4 or if the interaction is in reverse with the groundwater impacting the lake. Likely the relationship is both ways, when the water table is high, the septic systems on the island may not be able to function correctly, releasing PO_4 to the subsurface and to the lake; however, if the lake water is already mesotrophic or eutrophic, phosphorus rich water could be impacting the groundwater. The relationship with phosphorus is further complicated but the fact that large portions of the lake bottom are exposed during parts of the year. Decomposition of organic matter during periods of low water could cause releases of phosphorus to the lake when the lake floods again. It is known that large powerboats are continually stirring up sediments in the lake bottom, which could lead to continual re-suspension of phosphorus rich sediments.

The highest levels of PO_4 on AQ2 are from the three deepest wells, 75, 77 and 40, (the same wells with very high conductivity) suggesting that in this case geological processes are determining levels of this nutrient (Eyre and Pepperell, 1999). Further research would have to be done on Hatzic Island to

determine the interactions between Hatzic Lake water and the groundwater on the island and how these two systems influence the levels of PO_4 , which, if the groundwater from the island is discharging into the lake are high enough to cause serious concern about eutrophication.

4.1.2 Nitrate-N ($\text{NO}_3\text{-N}$)

The Guideline for Canadian Drinking Water Quality for nitrate-N is 10 mg/l (the level is 45 mg/L if reported as nitrate (NO_3^-)), however, levels above 3 mg/L are considered to be indicative of groundwater that is under the influence of surface water (Carmichael *et al.*, 1995; Schreier *et al.*, 1996). Table 12 shows the wells with nitrate-N values above 3 mg/L during at least one of the sampling periods. Wells with nitrate-N above 4 mg/L were re-sampled (within 2 weeks in the summer and within a month in the winter) and these values are presented in brackets beside the original. Six wells had higher nitrate values in winter than in summer and five wells showed the reverse trend. The well with the greatest change in nitrate-N value was well 11, which showed a 7.4 mg/L increase between summer and winter.

Table 12. Samples with Nitrate-N values above 3 mg/L in at least one sampling.

Well Number	$\text{NO}_3\text{-N-Winter}^a$	$\text{NO}_3\text{-N-Summer}$	Change	Depth (m)
1	3.008	1.003	2.005	9.45
5	3.075	4.828 (4.624)	-1.753	7.01
6	4.933 (3.793)	5.286 (5.325)	-0.353	6.10
11	11.962* (12.131)	4.538 (2.093)	7.424	N.A.
12	3.369	4.705 (7.06)	-1.336	7.62
15	2.168	3.284	-1.116	N.A.
16	5.307 (6.813)	0.579	4.728	9.14
19	7.499 (6.251)	10.129* (9.240)	-2.63	4.27
43	3.447	3.141	0.306	5.49
58	4.061 (4.296)	2.784	1.277	7.62
79	6.111 (2.973)	0.901	5.209	0.91
81	3.141	N/A	N/A	N/A

a – all nitrate-N values, including “change” are in mg/L

* Bold values are above the GCDWQ.

() re-sampled nitrate level

N/A = data not available

Both the wells with nitrate levels above 10 mg/l were located on the Nicomen slough aquifer as were the vast majority of the samples that measured above three. Previous research on groundwater quality in the Lower Fraser Valley found no nitrate values above 10 mg/L in wells located on the Nicomen slough aquifer. This study only examined community wells and acknowledged that private wells are more likely to be contaminated than community wells (Carmichael *et al.*, 1995). The three wells that were not on the Nicomen slough that had nitrate above three were also very shallow (5.49 m, 7.62 m, and 0.91 m).

In the summer, well 19 (located on the east of Hatzic Island) was the only well that exceeded the guideline for nitrate (10.13 mg/L). In order to try and determine the extent of the nitrate problem around

well 19, three additional wells were sampled as part of the March sampling. In March the nitrate level in well 19 had dropped to 7.5 mg/L, below the health standards but still well above background levels. Of the additional wells sampled well 81, had a nitrate level of 3.14 mg/L while the other two wells, which were slightly farther away had levels below 3 mg/L (1.8 mg/L and 0.9 mg/L). This suggests that the nitrate contamination seen in well 19 may be also impacting well 81 but that the problem is not widespread as the levels of nitrate dropped quickly as one moved away from this well.

The highest nitrate value was seen in the winter in well 11 (11.96 mg/L) This well (which is located in a small rural subdivision on Hatzic Island) did not exceed drinking water quality guidelines in the summer, however both well 11 and well 12 (located next door) had nitrate levels above 3 mg/L in both winter and summer suggesting there may be a more widespread nitrate pollution problem in this area.

Seasonal variations in nitrate level do not appear to follow a consistent pattern. There were more wells in the winter that had nitrate values above 3 mg/L, however the well with the highest nitrate in the summer decreased substantially in the winter. In the summer there were more wells on Hatzic Island with nitrate levels above 3. The nitrate values in some of these wells decreased in the winter while others remained fairly constant. In the winter high nitrate values were found throughout a wider area in the valley. This is consistent with previous research that has found that nitrate levels tend to vary with the season but that there is no discernable trend in that variation that can be related to a specific season (Carmichael *et al.*, 1995; Liebscher *et al.*, 1992).

4.1.4 Coliforms

Groundwater samples that measured above 4 mg/L nitrate were re-sampled to confirm the NO₃-N value as well as being sent to the British Columbia Centre for Disease Control (BCCDC) for bacterial testing (total and fecal coliforms). The results of the coliform analysis can be found in table 13. None of the samples from Hatzic Island, which had the highest nitrate values, had any detectable coliforms in the water. Total coliforms were detected in the winter in one well on AQ2 and one on AQ3. Both of these wells were very shallow and were located in areas with livestock close by. Well 58 appears, based on the airphotos, to be located in a forest and have no agricultural or urban land use nearby which makes the high nitrate and coliform values seem to be anomalous, however upon visiting the property it was observed that there was livestock such as sheep and chickens freely wandering the property which could be a source for nitrate and coliforms. The highest levels of total coliforms were found in the summer in well 76, which did not have a nitrate problem (the nitrate levels were below the detection limit in this well). However the owner informed us that this well had significant structural problems that likely led to the coliform contamination. This well was also quite shallow (2.74 m) suggesting that that depth may also be a significant factor in determining coliform contamination.

processes. Nitrate-N levels were not significantly correlated with any of the variables in table 15 suggesting that different processes determine what the levels of NO₃-N will be.

Table 15. Correlations between selected parameters (Spearman's Rank correlation coefficients (r)).							
	Ca	EC	PO ₄	Sr	Si	Na	Mg
Ca	1						
EC	0.816	1					
PO ₄	0.671	0.768	1				
Sr	0.851	0.775	0.722	1			
Si	0.701	0.823	0.927	0.703	1		
Na	0.624	0.869	0.618	0.597	0.683	1	
Mg	0.895	0.814	0.782	0.878	0.802	0.615	1
All correlations are significant at 0.01							

4.2.3 Depth

Shallow wells in unconfined aquifers have long been known to be more susceptible to contamination than deeper wells. Therefore, the depth of well is an important determinant of the vulnerability of a particular well to contamination. In the Hatzic Valley there are significant differences between the depths of wells in AQ1 and AQ3 ($\alpha = 0.05$) (Figure 18 shows the relationship between depths in the three aquifers). The wells in AQ1 are significantly shallower than those in AQ3. The differences in depth between AQ1 & AQ2 and AQ2 & AQ3 are not significant. AQ2 has the greatest range of depths because of the deep wells that tap into a deep regional aquifer (see section 4.1.2 for further discussion of these wells).

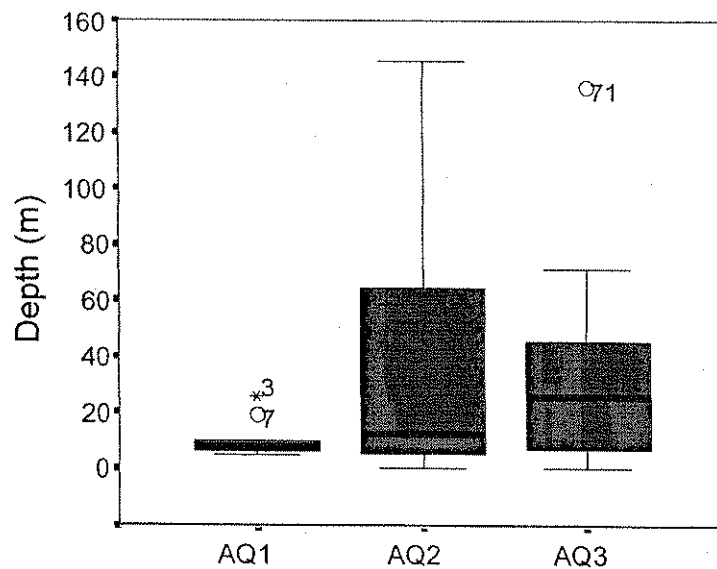


Figure 18. Depth Boxplot (Division by Aquifer)

Some studies mention a strong relationship between nitrate and depth (Schreier *et al.*, 1996; Hudak, 1999; Burkart and Stoner, 2002; Muhammetoğlu *et al.*, 2002; Reid *et al.*, 2002). The results of this study show that although there are many shallow wells with low nitrate values there are no deep wells with nitrate above 3 mg/l (see figure 19). A previous study in the LFV found that, "nitrate-N where present in wells

generally less than 60 m deep. This trend of nitrate-N with depth indicated that the source of nitrate-N is related to activities at the land surface" (Carmichael *et al.*, 1995). In the present study an even more

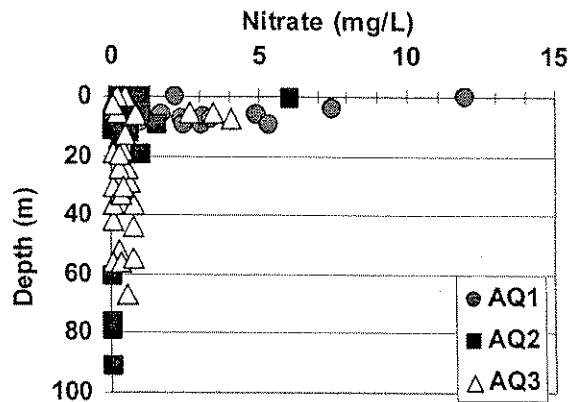


Figure 19. Nitrate vs. Depth (winter data)

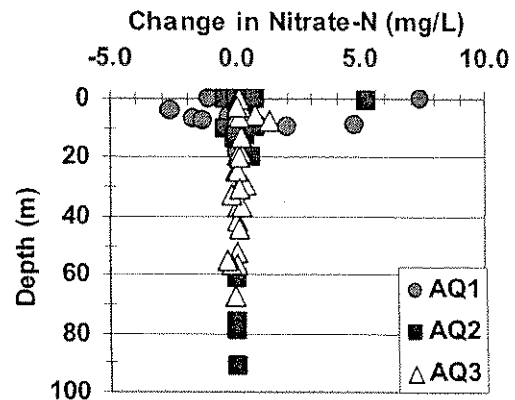


Figure 20. Seasonal Change in Nitrate vs. Depth

conservative depth was found. There are no wells deeper than 12 m with nitrate levels above 3 mg/l.

Although there were high correlations between the winter and summer sampling for most parameters (including Nitrate) (section 4.2.1) figure 20 shows the importance of depth in influencing the nitrate

values in the valley. No wells deeper than 12 m had significant fluctuations (greater than 1 mg/L) in

nitrate values between winter and summer however, several wells that were shallower than 12 m did.

This suggests that these shallower wells are under the influence of surface water and therefore land use,

whereas the deeper wells are not. The parameters in table 15 that were highly correlated with each other

did not show the same relationship with depth as nitrate did. Rather than being influenced by depth it is

likely safe to assume that the levels of these parameter are dependent on the geology of the substrate

through which they flow rather than by anthropogenic influences.

4.3 Parameters Influenced by Geology

In this study groundwater samples were taken from three distinct aquifers; it was the first study to look at these three aquifers as a system. In order to further understand the processes that affect water quality in the valley we need to characterize the water quality in each aquifer by determining if each aquifer has a distinct chemical signature. To examine the differences between the aquifers, the wells sampled were grouped according to location. Based on the findings in subsections 4.2.1 - 4.2.3, the following parameters were determined to be influenced largely by geology: calcium, silicon, orthophosphate, conductivity, magnesium, sodium and strontium. These parameters were chosen because they are highly correlated with each other but not with nitrate and because none of these parameters shows a relationship with depth.

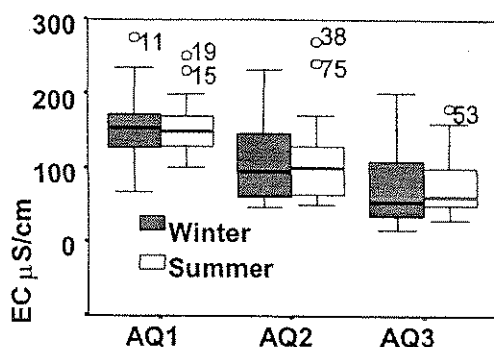


Figure 21. Conductivity Boxplot

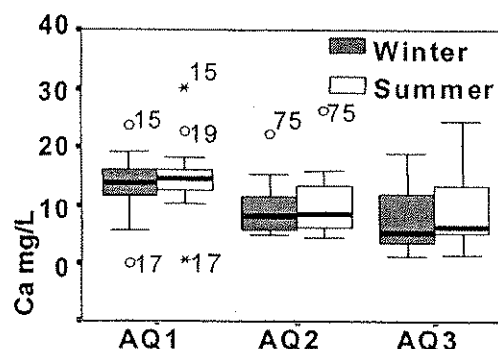


Figure 22. Calcium Boxplot

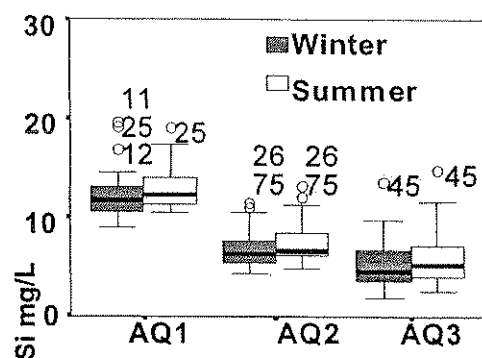


Figure 23. Silicon Boxplot

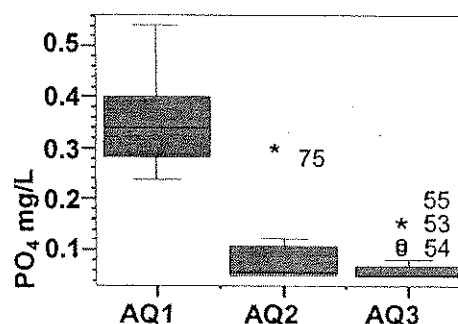


Figure 24. Orthophosphate Boxplot

For this aquifer specific analysis, wells 36-39 were removed from the database, as they are located west of AQ2 on the uplands just east of Mission. Additionally the four wells with the highest conductivity were removed because, as previously discussed (section 4.1.2) they have unique chemistry and likely draw from a different, deeper, aquifer than the shallower wells on AQ2. Figures 21-24 show boxplots for some of the geologically influenced parameters grouped by aquifer. These plots show that AQ1 has higher values for these parameters than the other two aquifers. Kruskal-Wallis tests followed by Mann-Whitney tests were run to determine which parameter were significantly different from each other when grouped according to aquifer. The K-W test showed that there were significant differences between at least 2 of the three aquifers for all parameters tested. The results of the Mann-Whitney tests showed that AQ1 and AQ3 have very distinct chemical signatures, the median was significantly different for each parameter. A surprising result was that despite the physical similarities between AQ1 and AQ2 (both are shallow and unconfined) the water in AQ2 is much more chemically similar to that of AQ3 (the deeper confined aquifer) than to that of AQ1. There were significant differences between the water quality of AQ1 and AQ2 for all parameters except sodium but between AQ2 and AQ3 only sodium and magnesium (in summer) had significantly different levels. Several explanations can be given for these results. The substrate of AQ2 may be geologically similar to that of AQ3 resulting in water that is chemically similar. It is also possible that there is continuous flow between AQ2 and AQ3 and that the water in AQ2

originated in AQ3 and has simple flowed south. More detailed technical analysis would need to be conducted to confirm either of these theories.

Table 16. Significance testing by aquifer, Kruskal-Wallis and Mann-Whitney tests							
	Kruskal-Wallis	Mann-Whitney U					
		AQ1-AQ2		AQ1-AQ3		AQ2-AQ3	
		S	W	S	W	S	W
EC	**	**	**	**	**		
Mg	**	**	**	**	**	**	
Ca	**	**	**	**	**		
Na	**			**	**	**	**
Si	**	**	**	**	**		
Sr	**	**	**	**	**		
PO ₄	**	**	N.A.	**	N.A.		N.A.
(**) = significant difference							

4.4.1 Cluster Analysis

In addition to testing for significant differences between aquifers, cluster analysis was used to determine if the chemical properties of the water led to natural groupings. As for the previous significance testing, wells 36-39 and the high conductivity wells were removed before starting this analysis so that only wells from the 3 aquifers were being compared.

Usually this procedure is used in order to find unknown patterns in data. In this case it was used to see if a predetermined pattern could be justified. The purpose was to discover how the wells would cluster based on chemistry alone and not on our predetermined aquifer separation.

Because cluster analysis is a subjective operation many different combinations of calcium, conductivity, orthophosphate and silicon were run. No matter which combinations were used, a fairly consistent pattern emerged (see fig 25 and appendix 10 for the dendrogram on which the diagram was based for a typical example). The clustering shown in figure 25 is based only on the winter results for Calcium and Silicon which were chosen because they both had 100% of samples above the detection limit and because these parameters are known to be influenced mostly by geology. The results show a pattern that seems to confirm that there are at least two separate groundwater regimes in the valley. One cluster was found in the northeast of the valley and covers much of AQ3 (cluster 1, □). Another cluster was found which covered mainly the south of the valley including Hatzic Island (cluster 3 ○). There was very little overlap between clusters two and three. Finally, cluster 2(*) encompassed the west side of AQ3 and the north of AQ2. This cluster was less consistent than the other clusters as it covered parts of AQ2 and AQ3.

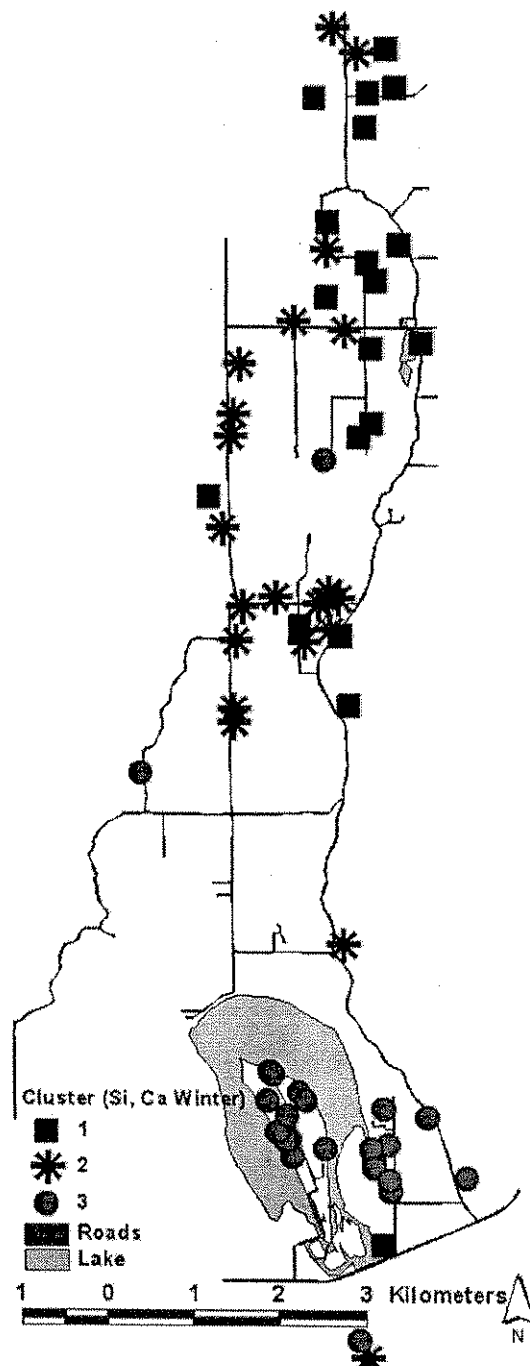


Figure 25. Clusters based on Silicon and Calcium

When a Kruskal-Wallis test was run using the clusters as grouping factors it was found that there was a significant difference between the groups for both calcium and silicon suggesting that one parameter was not dominating the other in determining the clusters.

Figure 26 shows Ca vs. Si grouped by aquifer. This graph clearly shows the distinct chemistry of AQ1 while at the same time highlighting the similarities between AQ2 and AQ3. This figure supports the argument that the water in AQ1 is chemically distinct and that AQ2 and AQ3 are difficult to separate into

distinct chemical groups. Figure 27 shows the same thing using different parameters; again AQ1 is distinct from AQ2 and AQ3.

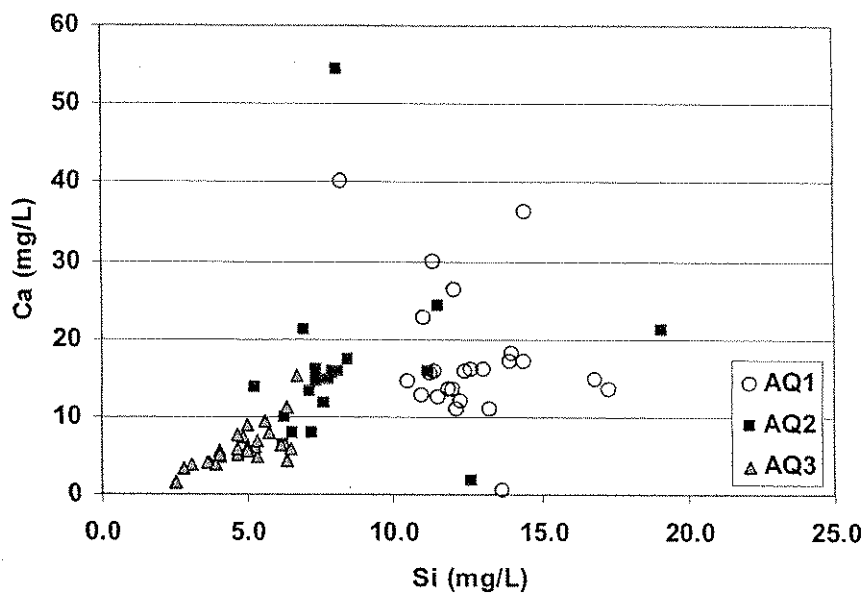


Figure 26. Silicon vs. Calcium (by aquifer)

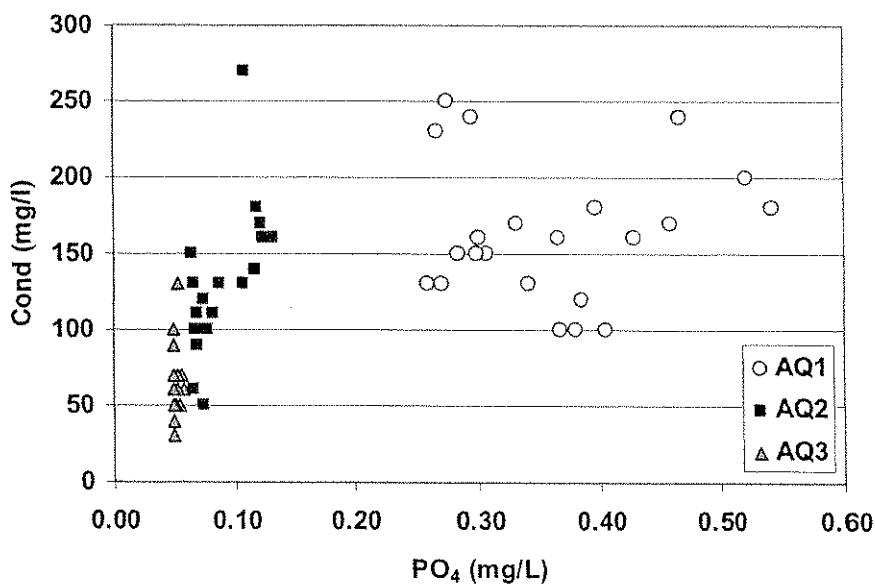


Figure 27. EC vs. PO₄ (by Aquifer)

Finally, figure 28 again shows AQ1 as having different water quality than AQ2 and AQ3, in this case the relationship is more linear; however, the graph shows the same trend, that AQ1 has water quality that is quite distinct from AQ2 and AQ3.

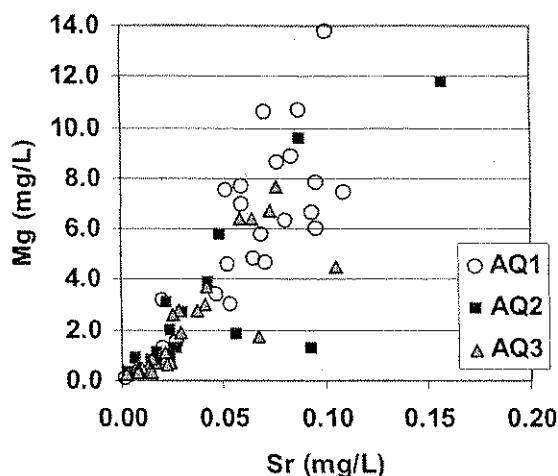


Figure 28. Mg vs. Sr (by aquifer)

4.4 Groundwater and Land use Interactions

In order to better protect the quality of drinking water sources we need to understand how land use influences water quality. The previous sections have shown that, because it is not correlated with any other parameter and because of its relationship with depth, nitrate is the parameter most influenced by land use in the Hatzic Valley. This section will explore how land use influences nitrate levels.

As mentioned in the methodology, GIS was used to calculate the percentage of various land use types around each well within three fixed radii. The percentage of forested land, agricultural land, intensively urbanised land as well as rural residential land (non-forested land around houses not in subdivisions but not used for agriculture) was calculated for 50, 100 and 200 m radii around each well.

The first part of the analysis involved looking at correlations between nitrate-N levels and the percentage of the four major land use categories mentioned above. All the wells sampled were used for the initial analysis regardless of depth or location. The correlation coefficients are shown in table 17. There were no significant correlations between nitrate and the percentage of agriculture or the percentage of rural residential land surrounding the wells. There was, however, a significant negative correlation between the percentage of forested land and nitrate levels, meaning that as nitrate levels in the wells increased, the percentage of land surrounding the wells that was forested tended to decrease. A positive correlation between the percentage of urban land surrounding the wells and nitrate levels was also seen, meaning that nitrate levels tended to be higher in wells that were surrounded by more heavily urbanised land.

In previous sections it was shown that there are no wells with nitrate levels above 3 mg/L that are deeper than 12 m. This confirms that land use is likely to impact wells that are shallow and dug into unconfined aquifers. Based on the depths as well as the geological profiles presented in section 2.4.3 it is evident that many of the wells in AQ3 are protected by an aquitard. Additionally, as mentioned in section 2.4 the land use mirrors the aquifer boundaries to a large extent. This means that the deeper wells in AQ3 are also more likely to be surrounded by forest than many of the shallower wells in AQ1 and AQ2. This make it difficult to know if the reason that wells with a greater percentage of forest surrounding them have lower nitrate is due to the forest or the depth and protected nature of the wells.

Table 17. Spearman's Rank Correlation Coefficients (r): Land Use vs. Nitrate							
Wells Correlated		Ag 50	Ag 100	Ag 200	For 50	For 100	For 200
All Wells	Winter	0.192	0.201	0.207	-0.349**	-0.344**	-0.346**
	Summer	0.153	0.154	0.153	-0.318**	-0.282*	-0.277**
<15 m Deep	Winter	0.064	0.072	0.049	-0.425**	-0.449**	-0.440**
	Summer	0.05	0.063	0.042	-0.469**	-0.448**	-0.464**
		Urb 50	Urb 100	Urb 200	Rur 50	Rur 100	Rur 200
All Wells	Winter	0.220	0.204	0.327**	0.062	0.028	-0.125
	Summer	0.255*	0.240*	0.298**	0.062	0.041	-0.047
<15 m Deep	Winter	0.315*	0.364*	0.449**	0.185	0.356*	0.216
	Summer	0.471*	0.514*	0.521**	0.120	0.255	0.207
** Correlation is significant at the 0.01 level (2-tailed)							
* Correlation is significant at the 0.05 level (2-tailed)							

In order to attempt to remove the bias where deeper wells, which are largely surrounded by forest may appear to have low levels of nitrate solely because of the amount of forest surrounding them the correlations were re-calculated using only those wells that were 15 m deep or less. This left 57% of the wells sampled or about 44 wells. The same relationships with forest and urban land were found (see table 17). When looking only at the shallow wells the correlations between the percentage of urban land and nitrate were even stronger. The correlation coefficient for the 200 m buffer in summer was 0.521 (the strongest correlation in table 17).

In order to further understand these relationships the wells were separated into three groups based on the land use percentages for each well. Using agriculture as an example, the groups were created as follows. Group one contained those wells with between 0 and 5% agricultural land within the buffer (the 100 m buffers were used for this analysis). Group two contained those wells with 5-50% agriculture and finally group three encompasses the remaining wells, those with 50-100% agricultural land within the 100 m buffer. The wells were grouped in this manner for each of the four major land uses. The nitrate levels were tested based on these three groups to determine if they were significantly different. As with the correlation coefficients the results of the Kruskal-Wallis test showed that the only significant differences in nitrate levels when grouped by land use ranks were seen when comparing the forested ranks and the urban ranks. There were no significant differences in nitrate levels if they were based on agriculture or

rural residential land percentages. Figures 29 and 30 show the boxplot of the differences in nitrate when divided by forest and urban land use ranks.

Figure 29 shows that the nitrate levels are higher where there is the least forested land. There were significant differences between group 1 & 2 and groups 1 & 3 but not between groups 2 & 3. The same results were obtained regardless of whether or not all the wells or only the shallowest wells were tested. These ranks were applied to and tested on the 100 m and 200 m buffers. Figures 29 and 30 are boxplots of the 100 m buffer.

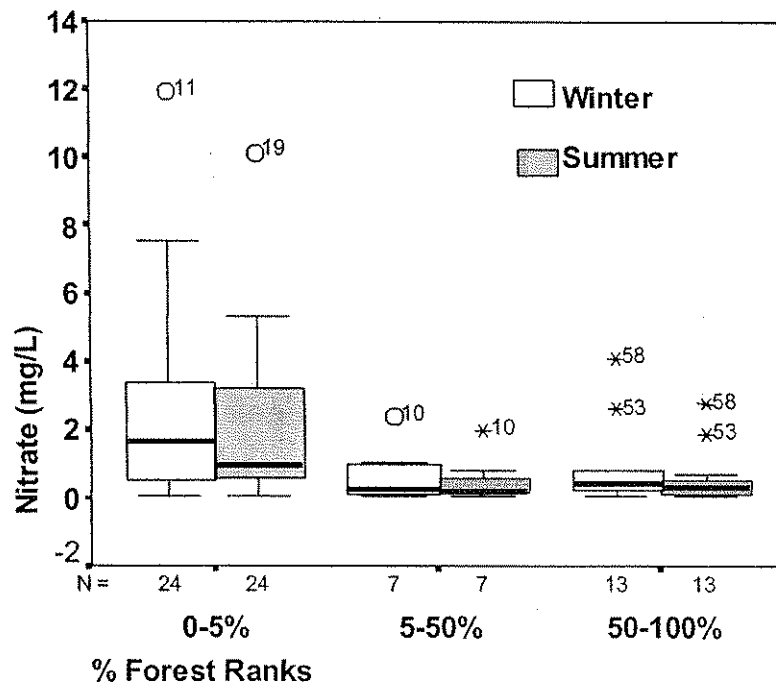


Figure 29. Boxplot - Nitrate vs. Forest Ranks (winter and summer) for a 100 m land use buffer

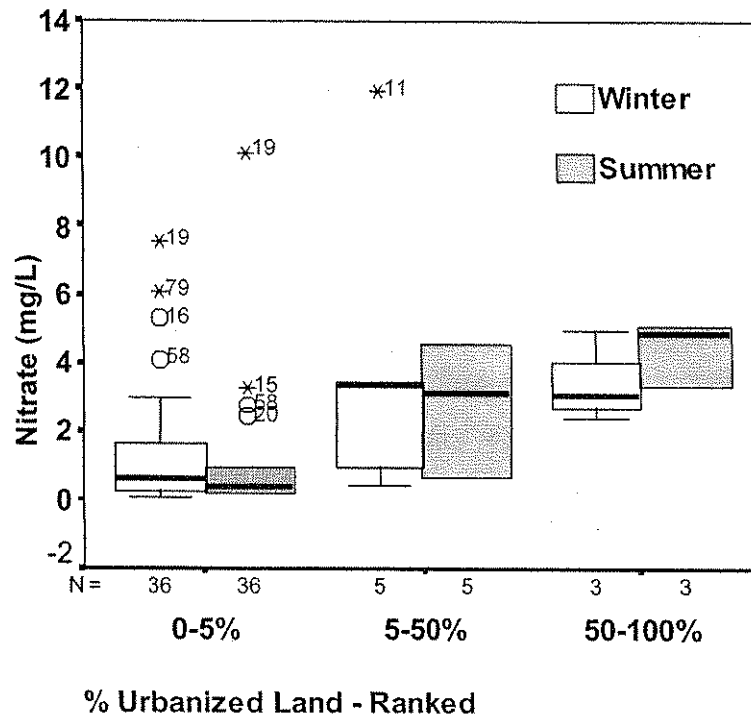


Figure 30. Boxplot - Nitrate vs. Urban Ranks (winter and summer) for a 100 m land use buffer

Figure 30 displays the relationship between nitrate level and the percentage of urban land surrounding the wells. The level of nitrate increases as the amount of urban land increases. In this case there are slight seasonal differences evident with the most urbanised wells being most highly impacted in the summer. This trend is also evident in the correlation coefficient where the coefficient is higher for the summer for the urbanised land.

The results can be explained as follows. Land that is forested protects groundwater because these areas do not have as many inputs as the land that is not forested. As discussed in section 4.1.5, in one case (well 58) high levels of nitrate were evident in a well, which, based on air photos appears to be completely surrounded by forest. A visit to this property revealed numerous livestock roaming freely throughout the yard, which is a very probable source for nitrate. This well was also quite shallow (7.62 m) increasing its vulnerability. Forested land does not cause low nitrate but generally activities that are likely to contribute nitrogen to groundwater are not found in forests (the above example is one of many possible exceptions).

It was surprising to find no correlations between nitrate levels and agriculture. Other studies in the Lower Fraser Valley (Schreier *et al.*, 2003; Schreier *et al.*, 1996; Leibsher *et al.*, 1992; Zebarth *et al.*, 1998) and elsewhere (Nolan and Stoner, 2000; Burkart and Stoner, 2002) have implicated agricultural activity as the major cause of nitrate pollution in various aquifers throughout North America and the world. Many agricultural activities, especially use of commercial fertilizer and spreading of manure often lead to

situations where the amount of nitrogen applied to the land is in excess of the requirements of crops or land cover. Since nitrate is soluble, the surplus ends up in groundwater. The lack of statistical correlation between groundwater nitrate and the percentage of agriculture surrounding wells in the Hatzic Valley is likely due to one or more of the following factors. The type of agriculture practiced in the Hatzic Valley is not as intensive as that practiced elsewhere in the Lower Fraser Valley. There are no large chicken or pig farms nor are there large areas of intensive horticulture within 200 m of any of the wells sampled. Additionally, Hatzic Prairie is the area of the valley with the most agriculture but it also the area where the least groundwater samples were taken. The samples that were taken were largely clustered towards the north of the area where the land use is more mixed. This un-even sampling can largely be attributed to chance, invitations to participate were sent to household throughout the watershed but there was limited control over the spatial distribution of those who agreed to participate. Additionally, residents reported that surface water was a popular alternative to groundwater for domestic use in the AQ2 area due to high iron levels found in the groundwater. There were several wells with high nitrate levels that were located in primarily agricultural areas. These wells, were, however the exception rather than the rule. All wells in primarily agricultural areas with high nitrate levels were less than 12 m deep.

The majority of wells with high nitrate were found in the small subdivisions on Hatzic Island. Significant correlations were found between the percentage of urbanised land and the level of nitrate. These areas are quite densely populated but lack the services found in most urban areas, that is, each household has an individual well and individual septic system. The source of nitrate in these areas is likely from the septic systems, which are too dense to function properly. The septic density is very high and the water table is near the surface at different times of the year. This creates challenging conditions for septic systems to function properly all year round.

4.6 Surface Water Quality

Surface water samples from 19 stations throughout the valley were taken between May 2002 and March 2003 (see figure 12 for location of stations). Seventeen of these stations will be analyzed here. Station 7 is omitted because it was dry for most of the summer and therefore cannot be included in seasonal comparisons and station 20 is omitted because it was not located in the study area.

The following parameters were investigated: nitrate, temperature and conductivity. The results were divided into two groups representing the “dry season” (May to Oct.) and the “wet season” (Nov. to March). The average of each parameter was calculated for these seasons so that each station has a dry (summer) and a wet (winter) value. In contrast with the groundwater samples there were significant differences between seasons for all parameters measured when the Mann-Whitney U tests were run on the stream data. In order to understand how the water quality changes as the streams run their course, each stream has been divided up into a mainstem and tributaries and changes in parameter levels examined from upstream to downstream. Figure 31 shows the stations on the mainstem and the location of stations on three of the tributaries. Each stream was buffered by 100 m using the GIS software. The area and percentage of each land use type upstream of the stations was calculated. The land use was calculated cumulatively and therefore downstream stations include all the land area upstream of that station.

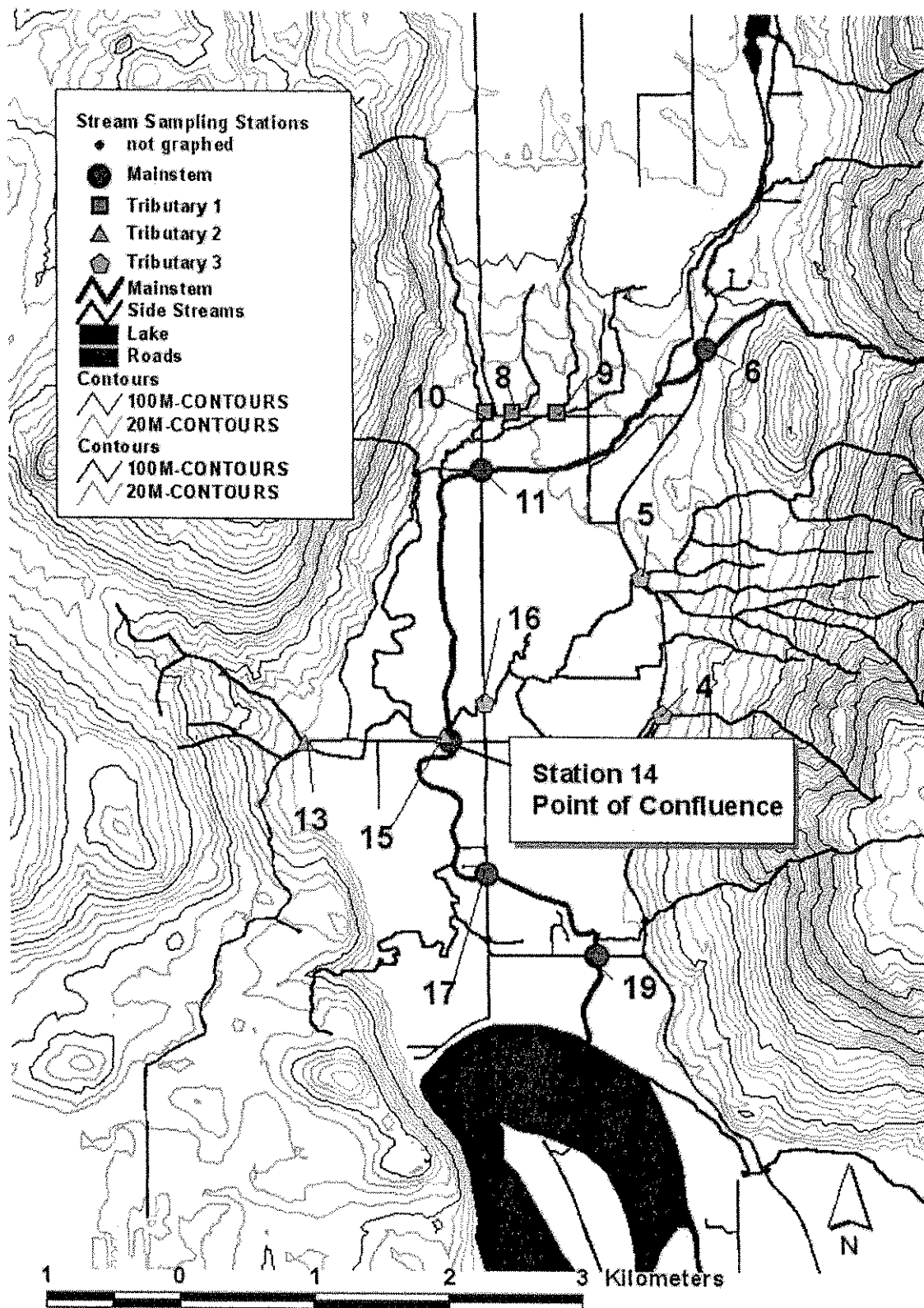


Figure 31. Hatzic Valley Mainstem and Selected Tributaries

4.6.1 Temperature

Figures 32-35 show the average wet and dry season temperature at each station on the mainstem and for three tributaries (Note that station 14 represents the confluence of the mainstem and 3 tributaries; therefore, each graph is the same beyond station 14. This is denoted by a dashed line running through station 14 on figures 32-43). As one would expect the temperature is higher in the dry season (summer) than in the wet season (winter). The streams in the Hatzic Watershed flow through two main land uses, forested and agricultural. In general the amount of agricultural land upstream from any station increases as one moved downstream. Significant correlations were found between the temperature and the percentage of forested land and the percentage of agriculture. Temperature tended to increase as the percentage of agriculture upstream increased and decrease at those stations where the percentage of forested land upstream was greater (See table 18 for the correlation coefficients). In winter the temperature is quite constant for all the tributaries. In the summer the temperature increases from upstream to downstream. This fits with the correlations mentioned earlier. Stations 8 and 10 have lower temperatures in the summer than most of the other stations. This is likely due to the influence of groundwater on the flow of the streams in these locations. Groundwater and stream water interactions will be discussed in more detail in a subsequent section.

Table 18. Spearman's Rank Correlation Coefficients (r) Surface Water vs. Land Use		
	%Forested	% Agriculture
Temperature (Dry)	-0.696 (**)	0.732 (**)
Temperature (Wet)	-0.662 (**)	0.656 (**)
Conductivity (Dry)	-0.581 (*)	0.631 (**)
Conductivity (Wet)	-0.569 (*)	0.592 (*)
(**) significant at 0.01		
(*) significant at 0.05		

4.6.2 Nitrate

Figures 36-39 show the dry and wet season values for nitrate from upstream to downstream. The graph of the mainstem shows a spike at station 14. By examining the tributaries it is evident that the elevated nitrate is coming from tributary 2 in the vicinity of station 13. The source of this nitrate is likely agricultural as there are farms nearby. It should be noted that these nitrate values are all quite low, below the 3 mg/L that we considered indicative of polluted water when looking at the groundwater samples. However, the maximum value measure at station 13 (2.51 mg/L) is higher than the background and therefore, indicates anthropogenic influence. The nitrate levels in the streams throughout the valley are higher in winter than in summer for several reasons. The main source of nitrate in streams is from run-off from agricultural fields. In the summer there is less run-off because it rains less; therefore, any nitrate that is on the fields or runs off is being taken up by the plants and aquatic biota that are growing during this time. In winter, the opposite occurs, run-off increases as the weather gets wetter and uptake

decreases because the plants are not growing. If the summer nitrate levels were high it would likely be due to groundwater contamination. The low summer nitrate values indicate that this is not the case.

4.6.3 Conductivity

The final parameter measured is conductivity (figures 40-43). The conductivity values are higher in the summer than in the winter due to the fact that there is less water in the streams in summer than in winter which therefore leads to a concentrating effect. There is a big jump in conductivity between station 6 and 11 on the mainstem. This is not due to agricultural activity but likely due to some in stream work and other urban activity that occurs near this station. A high conductivity (over 2000 μ S/cm) rivulet was found draining into the mainstem at station 11. It was not possible to follow this to the source as it came from under a fence on private property. The conductivity at station 8 is also quite high, likely from the influence of groundwater recharge at this station. In the winter when there is more water in the streams the conductivity is lower due to dilution. Stations 3 and 4 sample streams that come off the mountains after flowing over a fairly inert substrate. Since these streams are not influenced by land use, they have similar conductivity throughout the year. There were significant correlations between the land use and the conductivity seen in the streams. Streams tend to have lower conductivity at stations that run through forests than those that run through agricultural land (see table 18). None of the streams run past the urban areas on Hatzic Island (there is no surface water on the island) so no comparisons can be made.

These results show that land use in the Hatzic Valley influences the water quality in the streams that run through the valley. Locally, agricultural activity can raise nitrate levels although this trend is not seen consistently throughout the valley. There is a consistent significant ($\alpha = 0.05$) correlation between temperature and conductivity and amount of agricultural or forested land throughout the watershed. The next section will discuss how surface water and groundwater interact in the valley.

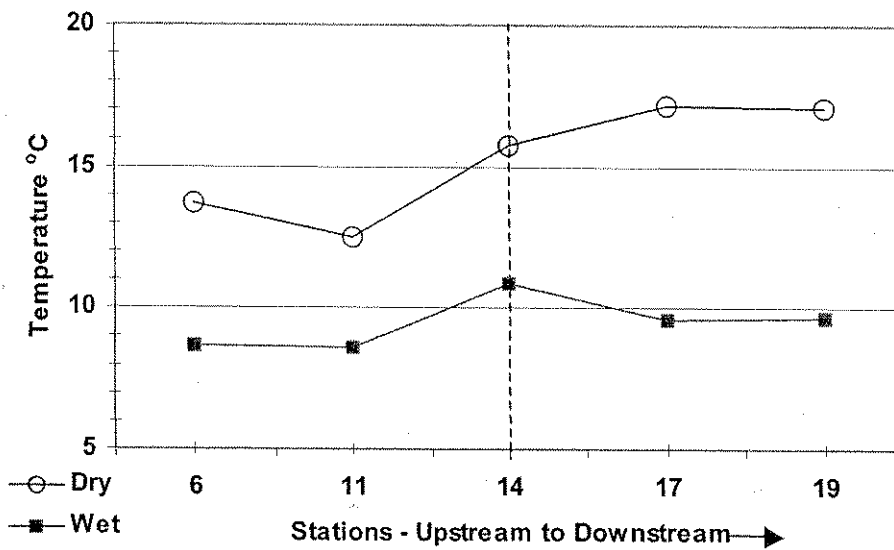


Figure 32. Average Dry and Wet Season Temperatures – Mainstem

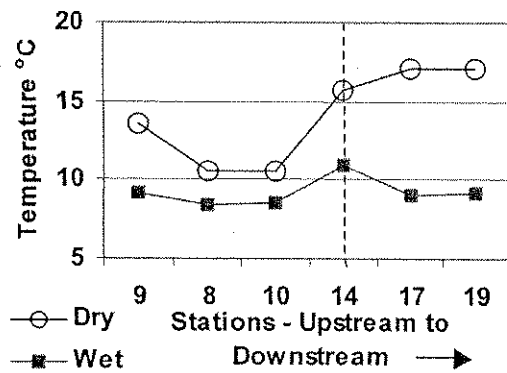


Figure 33. Average Dry and Wet Season Temperatures - Tributary 1

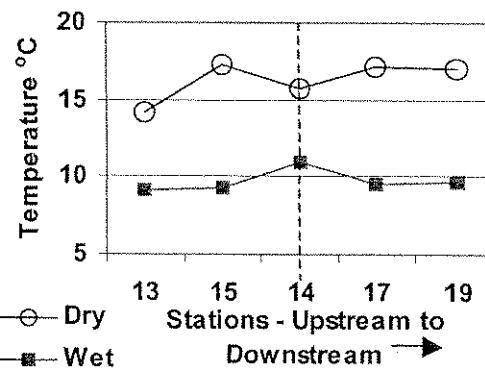


Figure 34. Average Dry and Wet Season Temperatures - Tributary 2

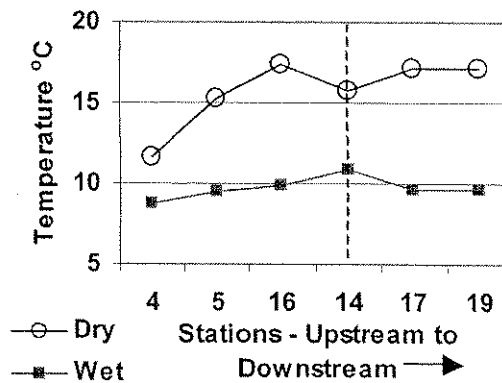


Figure 35. Average Dry and Wet Season Temperatures - Tributary 3

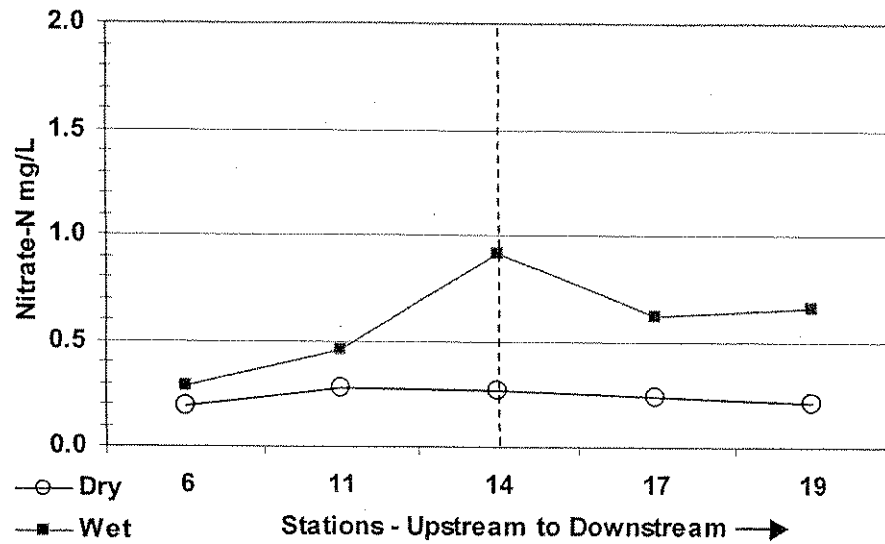


Figure 36. Average Wet and Dry Season Nitrate Levels - Mainstem

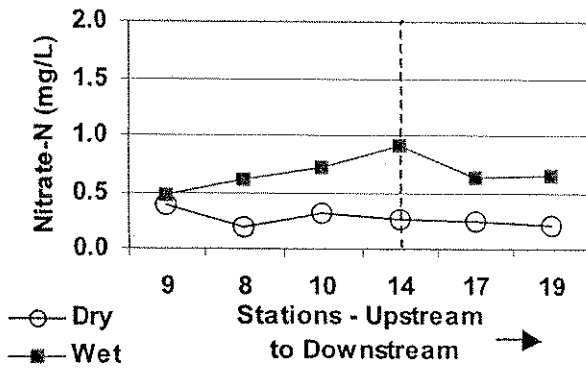


Figure 37. Average Wet and Dry Season Nitrate Levels - Tributary 1.

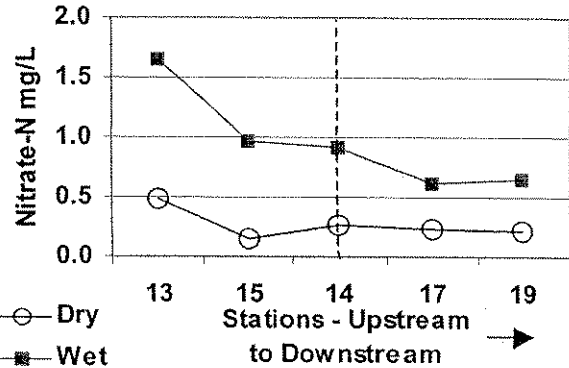


Figure 38. Average Wet and Dry Season Nitrate Levels - Tributary 2.

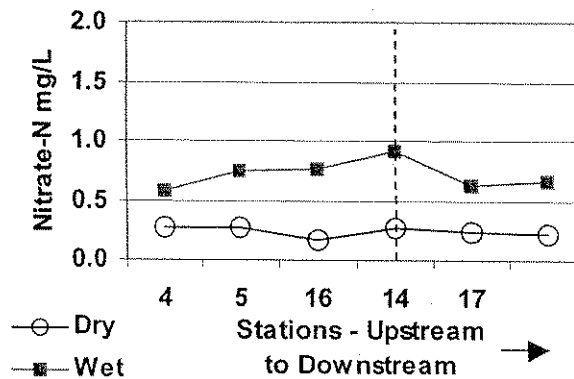


Figure 39. Average Wet and Dry Season Nitrate Levels - Tributary 3.

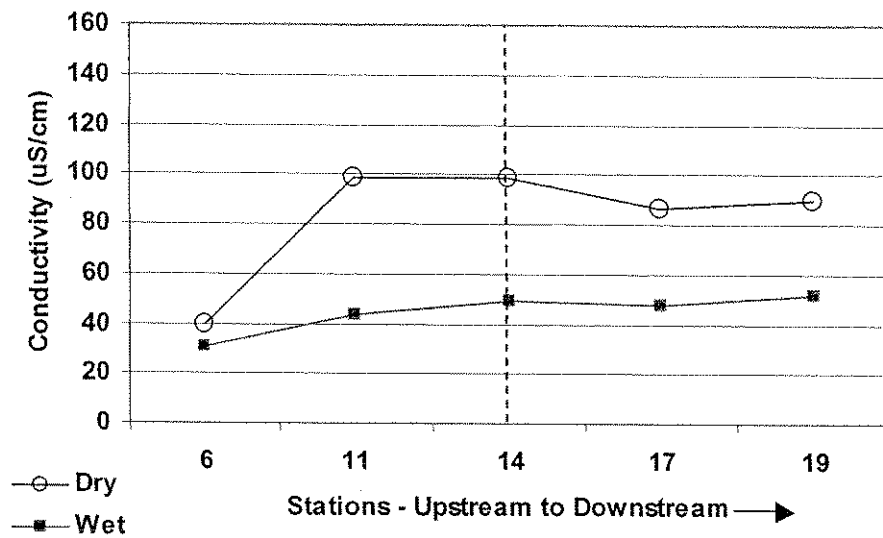


Figure 40. Average Dry and Wet Season Conductivity - Mainstem

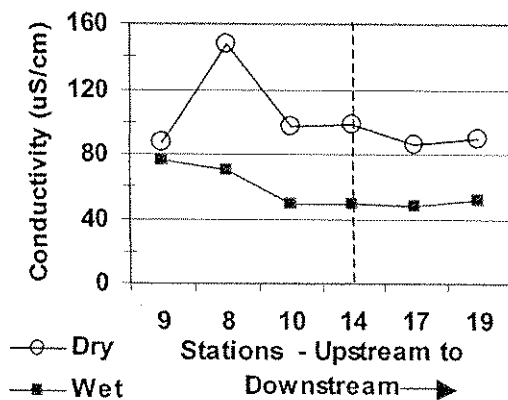


Figure 41. Average Dry and Wet Season Conductivity - Tributary 1.

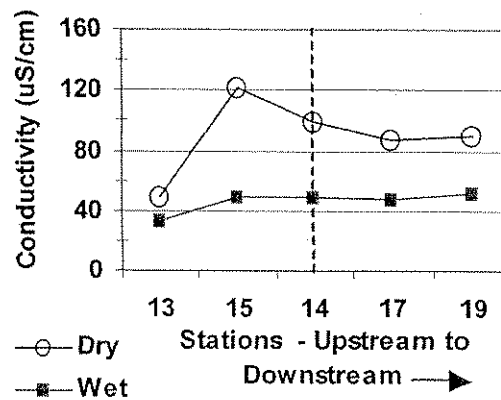


Figure 42. Average Dry and Wet Season Conductivity - Tributary 2.

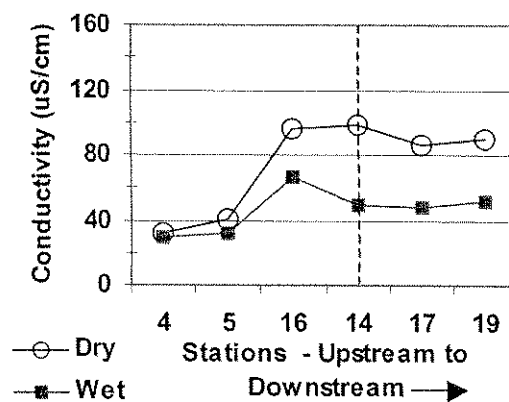


Figure 43. Average Dry and Wet Season Conductivity - Tributary 3.

4.7 Groundwater and Surface Water Interactions

Interactions between groundwater and surface water can have a controlling influence on stream chemistry as well as influence the biological conditions in and near streams (Oxtobee and Novakowski, 2002).

Previous research has shown that deterioration of surface water quality in areas of intensive agriculture can result from contaminated shallow groundwater discharging into streams (Cey *et al.*, 1998; Berka *et al.*, 2001). It is, therefore, important to understand the interaction between groundwater and surface water in order to determine the migration pathways for contaminants. These interactions become even more important in watersheds where groundwater makes up most or all of the flow during dry periods.

This section will look at the results of the groundwater and stream water sampling in the Hatzic Valley and attempt to determine if there are sections of the stream sampling network where groundwater influence can be seen or areas where surface water influence on the groundwater quality can be seen.

As the majority of the stream sampling stations are located on Hatzic Prairie (AQ2), which corresponds to the area where the least groundwater samples were taken, there is only a small part of the sampling network where there is sufficient overlap between the stream sampling network and groundwater sampling network to attempt to observe groundwater and stream water interactions based on direct comparisons. This area is illustrated in figure 44 and is located in the middle of the valley where AQ3 and AQ2 meet. The area includes the sampling stations on Oru, Durieu and Belcharton creek (stations 8-10) along Durieu Road and the nearby groundwater sampling stations. The discussion of groundwater influence on stream water quality will draw from several sources and include a general comparison of surface water quality and groundwater quality and how these may be related, a specific look at the water quality in the area mentioned above and reference to a previous study that speculated on the same problem. Two parameters that were measured both in the streams and groundwater (nitrate and conductivity) as well as temperature (which was only measured in the surface water) will be discussed and compared.