THE IMPACT OF AGRICULTURAL LAND USES ON WATER AND SEDIMENT QUALITY IN THE AGASSIZ / HARRISON HOT SPRINGS WATERSHED, B.C.

by

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Abstract

Non-point source pollution is a significant threat to surface water quality, particularly from agricultural land uses. The Agassiz / Harrison Hot Springs watershed, which contains various pesticide and manure intensive land uses, was studied in order to determine correlations between land use types and water quality.

The technique of correlating water quality with land uses has been well documented in the literature. With the development of computer software applications such as Geographical Information Systems (GIS), spatial analysis of land uses combined with the incorporation of water quality has become more practical. Two techniques were applied to determine the impact of agricultural land uses on water quality. The first, examined land uses within a 100 m buffer around major water courses. This method assumes that land uses adjacent to watercourses are the most relevant to water quality. The second technique divided land into contributing areas and assumes that all land uses within a sub-watershed, regardless of distance from the watercourse, impact water quality. Correlations at each sampling station were determined between various water quality parameters and the total area of different land uses upstream from the sampling point.

A wide variety of water quality parameters were examined, including nitrate, nitrite, ammonia, orthophosphate, pH, specific conductivity, dissolved oxygen and chlorophyll *a* concentrations. Sediment quality was also taken into consideration through the measurement of copper, lead, zinc, manganese, and iron in the <0.063 μm clay/silt fraction. Other sediment characteristics such as loss on ignition and grain size distribution were also examined. Whenever possible, these parameters were compared to provincial water quality guidelines in order to determine the overall state of the watercourses within the watershed.

Water quality parameters were combined with a bioassay experiment using the moss *Fontinalis antipyretica*. Moss stems were placed within mesh bags at select locations within the watershed. Indicators examined were shoot growth, and concentrations of chlorophyll *a* and chlorophyll *b* in each stem. Except for significant differences in shoot growth between stations H4 and H2 (Hogg Slough), the variability of the results was not significant enough to confirm differences in water quality. This is likely because variable environmental conditions between sampling stations made it difficult for the bioassay to

determine subtle differences in water quality. As a result, it was concluded that the use of *Fontinalis antipyretica* is not feasible as a bioassay in small agricultural watercourses.

Another technique utilized to determine water quality was XAD resin adsorption of the pesticide atrazine, which is the primary pesticide used on surrounding corn fields. Bench scale results of the technique showed that recovery of atrazine was affected by pH and the presence of organic matter. Results from the field showed significantly lower recoveries. No significant correlation between corn area and recoverable atrazine could be determined, likely due to variability of site conditions and contamination due to windblown transport of atrazine applied in neighbouring fields.

Use of the buffer technique to determine land use/water quality correlations was found to be a more effective method than examining contributing areas. Overall the buffer technique gave stronger correlations to known agricultural impacts to water quality. It was found that grazing pastures within the Agassiz / Harrison Hot Springs watershed had the greatest impact on water quality, resulting in higher nutrient concentrations in water and increased zinc concentrations in sediments. These impacts are thought to come directly from manure which has been stored or spread onto surrounding fields. Corn and hay/silage fields did not appear to correlate with any particular degradation of water quality, although certain sites appeared to be impacted due to manure application on these surrounding land uses.

The primary areas of degradation within the Agassiz / Harrison Hot Springs watershed are Agassiz Slough and Hogg Slough. Degradation within Agassiz Slough is directly connected with the presence of stormwater outfalls along the watercourse. Along with elevated concentrations of nutrients, there were concentrations of copper above the severe effect level and high levels of zinc and lead present in the sediments outside of the sewer outfall sampling station. Similarly, Hogg Slough had elevated concentrations of nutrients combined with high concentrations of zinc in the sediments. However, the source of degradation in Hogg Slough is primarily manure runoff from surrounding pastures and manure storage. Specific sites of degradation also included station D5 (McCallum Slough), C3 (Miami Creek), and I3 (Clarke Ditch).

Levels of nitrite, dissolved oxygen, pH and iron were outside of accepted water quality guidelines at some stations. Iron and pH, however were assumed to be the result of background concentrations.

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List of Abbreviations

AHHS	Agassiz / Harrison Hot Springs
BCF	Bioconcentration Factor
DFO	Department of Fisheries and Oceans
DO	Dissolved Oxygen
FVRD	Fraser Valley Regional District
GIS	Geographical Information Systems
IQR	Interquartile Range
ISQG	Interim Freshwater Sediment Quality Guidelines
LC_{50}	Lethal Concentration 50
LEL	Lowest Effect Level
MWCO	Molecular Weight Cut-off
PEL	Probable Effect Level
PPB	Parts Per Billion
PPM	Parts Per Million
SEL	Severe Effect Level
USEPA	United States Environmental Protection Agency

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1. Introduction

Pollution from non-point agricultural land use sources has a significant impact on the water quality of rural watersheds. According to the United States Environmental Protection Agency (USEPA), agricultural runoff poses the most significant non-point source pollution risk to surface water quality in the US (Daniel *et al.*, 1994). Within a rural watershed, non-point sources of pollution can come from a variety of different sources. These can include sediment, plant nutrients and agricultural chemicals (Lapp *et al.*, 1998). In order to determine the threat of current agricultural practices to water quality, it is necessary to draw general correlations between land uses and water quality. This not only allows us to predict environmental impacts to water quality, but it is also a useful tool to determine impacts on the future development of the watershed. By using modern computer software combined with comprehensive monitoring of water and sediment quality, it is possible to determine these relationships.

1.1 Study Goals and Objectives

The main objective of this research is to determine the cumulative effects of agricultural land uses on the water quality of watercourses and streams in the Agassiz / Harrison Hot Springs (AHHS) watershed. In addition, the following additional study objectives will be examined:

- To determine if contributing areas or watercourse buffer regions have a greater impact on water quality.
- To establish the overall state of the water quality within the AHHS watershed through the collection and analysis of water and sediment.
- To observe seasonal variations in water and sediment quality within the watershed over one annual hydrological cycle.
- To determine the impact of stormwater outfalls on sediment and water quality in Agassiz Slough.
- To test the ability of XAD resins to adsorb atrazine both in the field and under controlled conditions.
- To evaluate the ability of aquatic mosses to serve as bioindicators of water quality.

1.2 General Methodology

Non-point source pollution is inherently diverse in that it encompasses a variety of both physical and chemical changes to the water and sediment. As a result, it is necessary to use a variety of different techniques to gauge the quality of the water for aquatic organisms. The techniques applied in this study are described below.

1.2.1 Water and Sediment Quality Indicators

The most commonly used indicator of water and sediment quality is an examination of its physical and chemical properties at various points within the watershed. Water quality parameters covered in this study include dissolved oxygen (DO), pH, temperature, specific conductivity, turbidity, chlorophyll *a* concentrations, and nutrient concentrations such as ammonia, nitrate, nitrite, and orthophosphate. All of these parameters must fall within a certain range in order for aquatic ecosystems to function normally.

Quantitative results of parameters can be quickly compared with BC Provincial Water Quality Guidelines and Canadian Water Quality Guidelines for the Protection of Aquatic Life to determine if they are impacting the health of the aquatic ecosystem.

Water quality is also closely associated with the quality of its sediments since there is constant contact between the two, in addition to various chemical interactions. Metal contamination of agricultural watersheds can be quite pronounced, and can indicate both point (i.e. storm sewer outfalls) and non-point pollution sources. Sediment parameters being examined include metal contaminants such as iron, lead, zinc, manganese and copper. As with water quality parameters, these values can be compared with freshwater sediment quality parameters to predict potential adverse impacts to aquatic ecosystems.

Physical parameters such as loss on ignition, which indicate organic matter content within the sediment and grain size distribution, are also important aspects to examine. Organic matter will tend to bind many contaminants such as metals and pesticides and thus incorporate them into sediments. Grain size fractions are an important indicator of the potential for siltation and transport of sediment fines. The greater the percentage of fines, the larger the possible impact during slough maintenance procedures carried out

throughout the watershed. In addition, finer sediments have a greater surface area resulting in a greater potential to adsorb contaminants.

1.2.2 Moss Bioassays

Bioassays are an important tool to determine the actual impact on aquatic life from contaminants and environmental conditions found within watercourses. An examination of physical and chemical water quality parameters tends to give only an indication of water quality at a particular moment in time, resulting in an incomplete representation of the system as a whole (Cenci, 2000). In the bioassay technique, an indicator organism is exposed to the various conditions and contaminants present within the sloughs for a fixed period of time. The overall health and survival of the organism can be used as an indicator of the quality of the aquatic ecosystem. In this particular experiment, the stems of the moss *Fontinalis antipyretica* were used as an indication of possible impacts to the aquatic ecosystem. Previous research has shown that factors such as stem growth, dry weight and chlorophyll *a* and *b* concentrations can change in response to organic pollution and heavy metals (Davies, 2002). This provides possible sub lethal indicators that can be used to show overall environmental quality.

1.2.3 Resin Adsorption of Pesticides

Pesticide use in agricultural watersheds is a serious concern for water quality since only a small percentage of the pesticides applied actually reach their target organisms. Other organisms, particularly in aquatic ecosystems, are invariably affected when these chemicals reach surrounding watercourses. Although chemical analysis of pesticides in water is possible, it only indicates the state of the water at the specific point in time of the sample. Since concentrations of pesticides tend to spike shortly after the time of application and after rainstorm events, grab samples may not adequately represent maximum pesticide concentrations reached in the water. For this reason, resins that can adsorb pesticides and remain in the water for relatively long periods of time are being used in this study. Atrazine is the most common pesticide used in the watershed under study and thus will be used as an indicator of the degree of pesticide contamination within the watercourses.

1.2.4 Geographical Information Systems (GIS)

GIS, developed by ESRI Software, is a computer program which performs spatial analytical functions with geographically referenced data (Berka, 1996). With the advent of new mapping techniques and technologies, GIS has proven to be an excellent tool for the analysis of non-point source pollution in watersheds. The main difficulty when dealing with non-point pollution is determining which land uses significantly affect water quality. In addition, contaminants are subject to complex kinetic reactions and physical processes which are difficult to observe and quantify using traditional methods (McFarland and Hauck, 1999). Thus, the best way to gauge the impacts of the various land uses is to monitor the water quality in various areas of the watershed overtime and derive a correlation between the land uses and changes in water quality. It is almost impossible to determine specific cause and effect relationships applicable to all watersheds. However it is possible to derive certain general consistencies and thus, obtain a general land use/water quality correlation for the AHHS watershed and similar agricultural practices in other watersheds.

1.3 The Agassiz / Harrison Hot Springs Watershed

The Agassiz / Harrison Hot Springs (AHHS) watershed is located in the District of Kent approximately 129 km east of the city of Vancouver. It is situated just north of the Fraser River, and south of Harrison Lake (Figure 1.1). Major municipalities include the village of Harrison Hot Springs, which is adjacent to Harrison Lake, and the town of Agassiz, located north of the Fraser River (Figure 1.2).

The region was first settled in the 1880's and grew quickly with the establishment of the Canadian Pacific Railway in 1885 and the construction of the Lougheed Highway in 1928. Besides agriculture, a variety of industries have had positive impacts on the growth of the region, with logging and road construction providing the most noteworthy spurts of growth. Over the last 50 years, the town of Agassiz has experienced significant development on its southern portion, mostly in the form of single family housing (District of Kent, 2001).

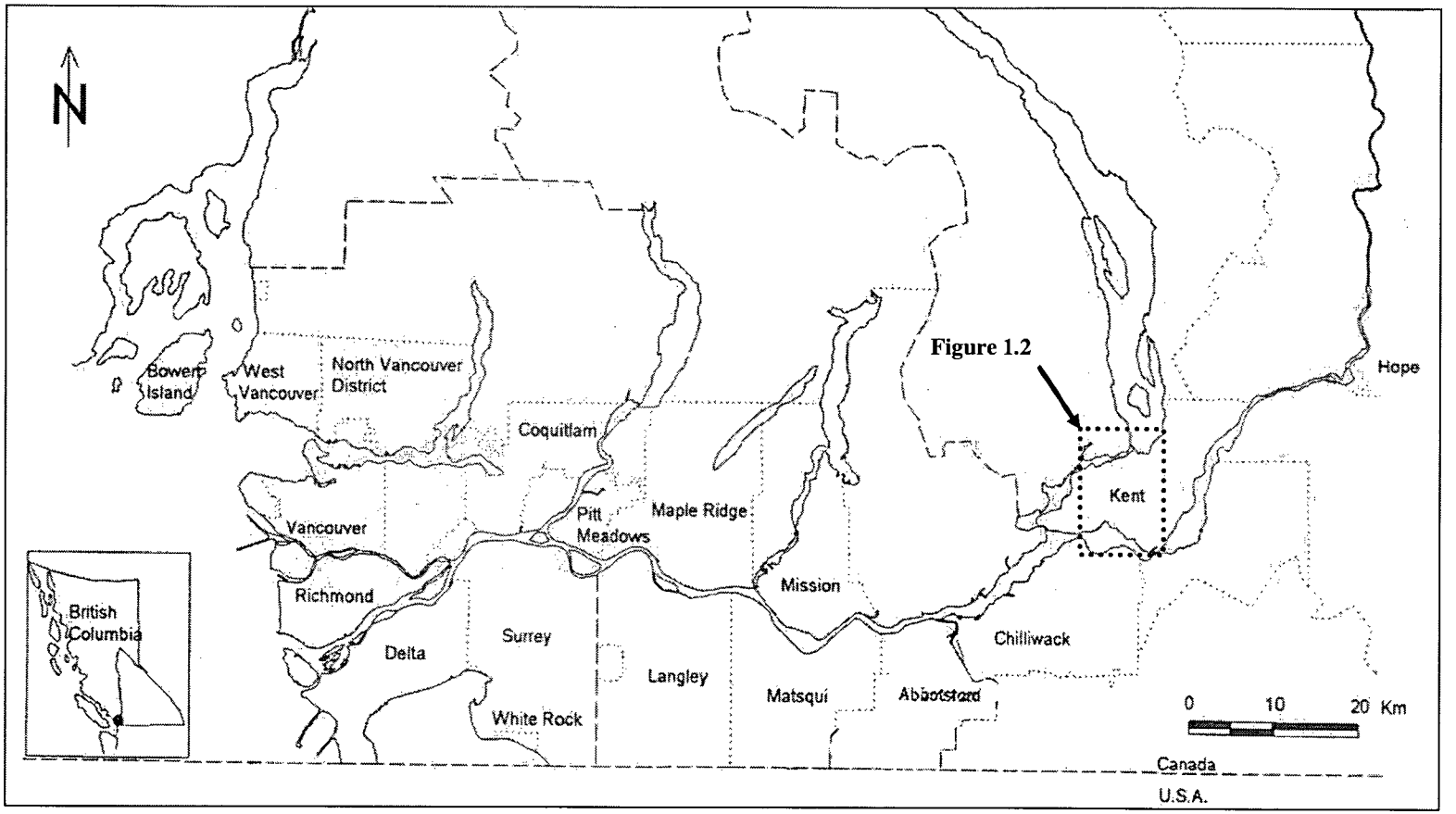

Figure 1.1: Location of Kent District in the Lower Fraser Valley, BC (Berka, 1991).

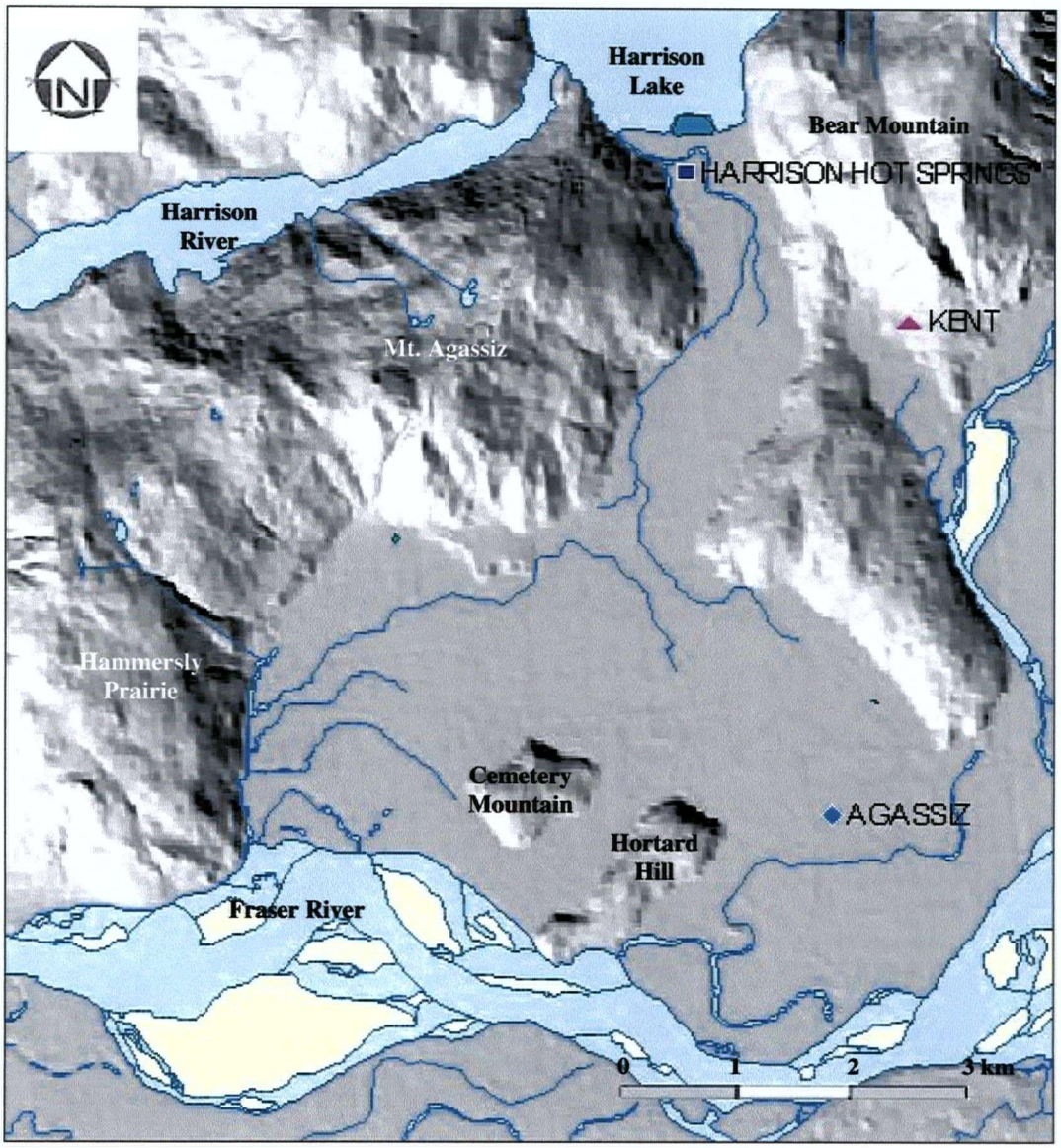

Figure 1.2: Digital Elevation Model (DEM) image hillshade of Agassiz / Harrison Hotsprings Watershed study area (SHIM, 2002).

The Fraser Valley region has one of the fastest growing populations in Canada. The population within the district has grown from 1,281 residents in 1941 to 4,844 residents in 1996. Current growth rates are fairly high, with an estimate of 14% between the year 1996 and 2000. The population is estimated to grow to between 6,500 to 9,400 by the year 2026, with an annual growth rate between 1.3-3.4% (District of Kent, 2001).

1.3.1 Agricultural Land uses

Agriculture is the primary land use within the watershed and is the staple economy of the Kent District. Every year this sector creates $21 million in farm receipts and employs approximately 13% of the labour force (District of Kent, 2001). The protection of agricultural land is an important initiative for the official community plan of the region. Almost all of the current agricultural land is reserved under the Agricultural Land Reserve Act of BC and is under provincial jurisdiction (Figure 1.3).

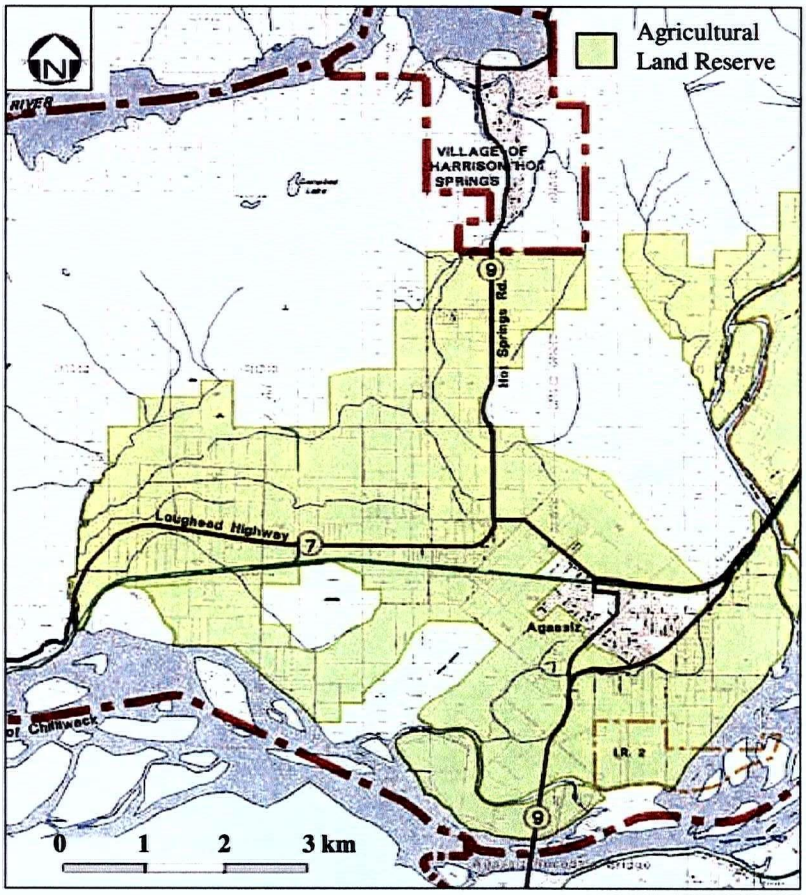

Figure 1.3: Areas of Agricultural Land Reserve within the AHHS Watershed.

The most common agricultural land use is dairy farming along with beef cattle production and processing. Corn crops are also very common within the watershed, giving Agassiz the title of "The Corn Capital of BC". Other common crops grown include hazelnuts, a wide range of vegetable crops, nurseries and greenhouse developments. There is also the presence of hog and poultry production (District of Kent, 2001).

1.3.2 Non-Agricultural Land Uses

Gravel extraction is quite prominent within the watershed, taking place in various locations near Mt. Agassiz on Cemetery Rd. and McCallum Rd. There are also additional plans for extractions to take place within the Fraser River depending on restrictions for the protection of fish.

There is only one heavy industrial site located in the AHHS watershed. The Rimex plant, located at the corner of McCallum and Cameron Rd., produces steel rims for mining industry vehicles. Light industry developments are fairly minor and consist of only two companies located on the north end of Agassiz.

The municipalities of Agassiz and Harrison Hot Springs make up the remainder of the land uses within the watershed. According to the 1996 census, over 80% of the homes within Agassiz are single family homes. Over the next 25 years, it is projected that 580 to 1,560 new homes will be created. Since only a limited amount of space is available within town boundaries, much of the new property will result from converting farmland south of the municipality of Agassiz.

1.3.3 Climate

The AHHS watershed is located in the coastal region of British Columbia. Watersheds within this area are classified as being of the dry maritime sub-variant. Summers tend to be quite cool, while the winters are fairly mild. Mountains in the region tend to receive large amounts of precipitation from moist air which blows eastward from the Pacific Ocean. As a result, the mountain ranges are often characterized by heavy snow packs in the winter (Swain *et al.*, 1997).

Roughly 75% of the precipitation for the Lower Fraser Mainland falls between October and March of each year (Berka, 1996), with precipitation peaking during the months of November, December and January (Figure 1.4) (Bussanich *et al.*, 2000). In the summer, a variety of air movements and the presence of a high pressure system off the coastal area results in less intense systems moving throughout the area (Swain *et al.*, 1997). As a result, the months of July and August typically have the lowest amount of rainfall (Figure 1.5) (Bussanich *et al.*, 2000).

1.3.4 Surficial Geology

The Kent District watershed is surrounded by a number of mountains with the most prominent being Bear Mountain to the northeast. Other mountains include Mount Woodside to the southwest and Agassiz Mountain to the northwest. Cemetery Mountain and Hortard Hill are both located within the watershed (Figure 1.2). The surficial geology of the area can be divided into two regions, the floodplains and the uplands. The floodplain, which dates back to the Quaternary period, was created by post-glacial salish sediments (Bussanich *et al.*, 2000). The floodplain soils are derived from shallow Harrison Lake deposits consisting of organic silt loams ranging in depths from 0.3 m to >10 m. In addition, there is evidence that Fraser River sediments overlie some areas (Bussanich, 2000). The soils of the floodplain are of very high quality for agriculture, ranging from Class 1 to 4 (District of Kent, 2001). The second area, comprising the uplands, originates from the Mesozoic and Upper Paleozoic Era. These areas consist primarily of granite bedrock, but also have layers of glacial, colluvial and eolian sediments no greater than 2 meters in depth (Bussanich *et al.*, 2000).

1.3.5 Flora and Fauna

The plethora of mountainous forests surrounding the AHHS watershed allows for a variety of different animals to inhabit the area. Terrestrial animals include black bears, coyotes, raccoons, black tail deer and cougars while bird species include crows, finches, sparrows, bald eagles, great blue herons, hawks, harlequin ducks, geese and mallard ducks. Besides fish, other aquatic organisms found within the watercourses include toads, snails, salamanders, and fresh water mussels (Bussanich *et al.*, 2000). Flora contained within the upland regions is also quite diverse (Table 1.1).

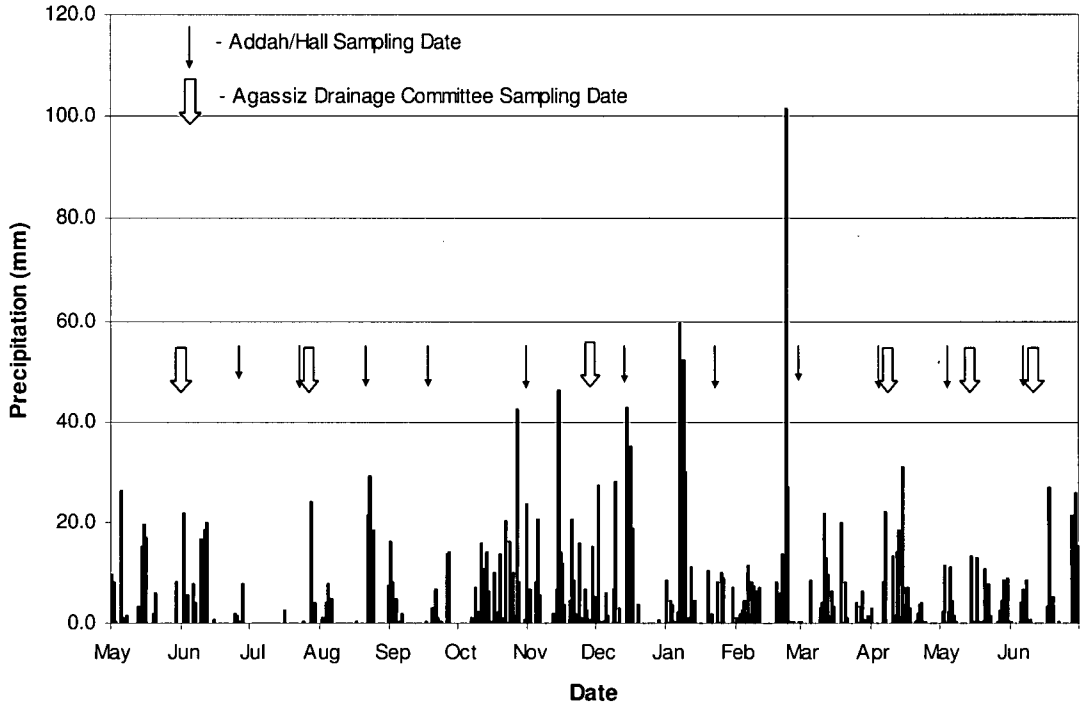

Figure 1.4: Agassiz Research Station recorded average daily precipitation from 2001 – 2002 with sampling dates (Environment Canada, 2002).

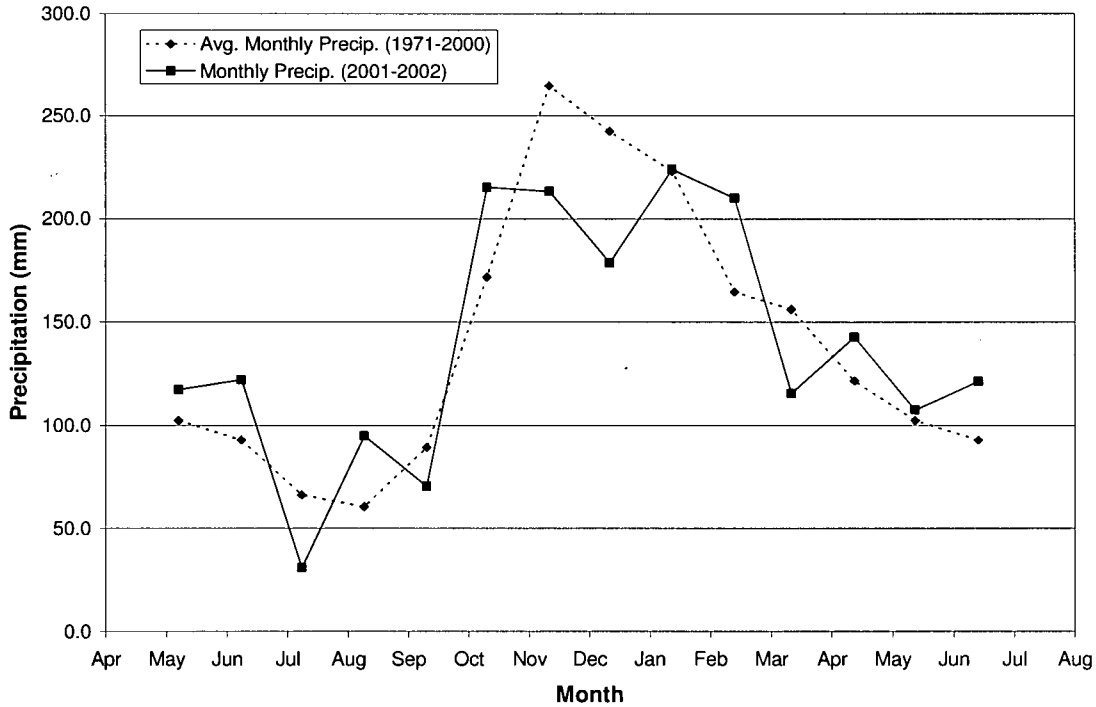

Figure 1.5: Agassiz Research Station Average Monthly Historical Precipitation and Average Annual Monthly Precipitation from 2002-2003 (Environment Canada, 2002).

Table 1.1: Common local flora in the AHHS watershed (Bussanich *et al.*, 2000).

Plant Type	Common Name (*Scientific Name*)
Trees	Western Red Cedar (*Thuja plicata*)
	Red Alder (*Alnus rubra*)
	Black Cottonwood (*Populus balsamifera*)
	Broadleaf Maple (*Acer macrophyllum*)
	Paper Birch (*Betula papyrifera*)
	Douglas Fir (*Pseudotsuga menziesii)*
	Hemlock (*Claytonia sibirica)*
Shrubs	Red Elderberry (*Sambucus racemosa*)
	Black Hawthorn (*Crataegus douglasii*)
	Salmonberry (*Rubus spectabilis*)
	Hardhack (*Spiraea douglasii*)
	Willow (*Salix* sp.)
	Himalayan Blackberry (*Rubus discolour*)
	Evergreen Blackberry (*Rubus laciniatus*)
	Snowberry (*Gaultheria antipoda*)
	Vine Maple (*Acer circinatum*)
	Wild Rose (*Rosa* sp.)
	Red Osier Dogwood (*Cornus stolonifera)*
	Holly
Herbaceous Plants	Lady Fern (*Athyrium filix-femina)*
	Sword Fern (*Nephrolepis exaltata)*
	Licorice Fern (*Polypodium glycyrrhiza*)
	Siberian Miner's Lettuce (*Claytonia sibirica)*
Aquatic Plants	Reed Canary Grass (*Phalaris arundinacea*)
	Duckweed (*Lemna minor*)
	Nightshade (*Solanum* sp.)
	Cattail (*Typha latifolia*)
	Ditch Grass (*Ruppia cirrhosa*)
	Brown and Green Algae
	Periphytes

Riparian zones on the AHHS floodplain surrounding watercourses range from forest covered areas to almost no riparian vegetation. Forest covered riparian zones are common around the northern part of the watershed, where development has been limited around Harrison Hotsprings. In agricultural areas, however, it appears landowners have maintained varying degrees of riparian zones depending on their needs.

1.4 Watercourse Characteristics

The floodplain has an average elevation of 20 m above sea level, and is divided into two sub watersheds (Figure 1.6). The northern sub-watershed is drained almost exclusively by Miami Creek and its tributaries, and occupies a relatively narrow area between Mt. Agassiz and Bear Mountain. This sub-watershed discharges to Harrison Lake. The second, larger sub-watershed is comprised of the southern half of the watershed and is drained by Mountain Slough, McCallum Slough, Westlin Ditch, Hogg Slough, and MS-1 Slough (SHIM, 2002). Water collected in this southern sub-watershed is discharged to the Fraser River.

The largest drainage watercourse is the Mountain Slough system, which drains approximately 30 sq km. The Miami Creek system drains 17 sq km, while the Agassiz – Cheam sloughs drain 14 sq km (Northcote, 2001). The characteristics of each of these watercourses are described in Table 1.2.

1.4.1 Flood Control

As a result of its location on the Fraser River Floodplain, the AHSS watershed is subject to a high frequency of flood events during the spring periods when the usually heavy snow packs from surrounding and upstream mountains melt. Prior to settlement and subsequent clearing for farmland, the area was frequently flooded and was relatively swampy in nature. In 1948, a particularly large flood inundated the vast majority of the watershed causing extensive damage. In order to combat the constant flood threat, an existing dyke was repaired and extended along the north side of the Fraser to prevent flooding during high water levels. Current dyking systems in place are designed to control typical 200 year flood levels (District of Kent, 2001). Along with the dyking system, there is also the Hammersly Pumping Station located at the end of Mountain Slough, which pumps out water to deter any flooding in the Agassiz sub-watershed. Similarly, there is a pumping station located at the mouth of Miami Creek to prevent any flooding within the Harrison Hotsprings sub-watershed.

Table 1.2: Major watercourse characteristics within the AHHS Watershed.

Stream / Slough Channel	Physical Characteristics	Habitat Characteristics (FVRD, 2002)	Current Watercourse Environmental Issues (DFO, 2002)	Possible Pollution Sources
Agassiz Slough	• Natural stream • Wide • Very slow flow • Water level variable depending on Fraser River	• Fish presence • Wetland habitats	• Riparian zone loss • Significant water quality problems	• Agassiz municipal stormwater outfalls • Surrounding farmland
Hogg Slough	• Natural stream 400 meters inland from Mountain Slough, then becomes modified channel • Wide to mid-width • Moderate flow • Water level fairly constant	• Fish presence • Wetland habitats in unmodified areas	• Not surveyed	• Surrounding farmland
Hotsprings Slough	• Natural stream • Mid-width • Slow flow • Water level variable depending on Harrison Lake	• Fish presence • Wetland habitats	• Riparian zone loss • Channelization • Significant water quality problems	• Surrounding farmland
MS-1 Slough	• Natural stream 200 meters inland from Mountain Slough, then becomes modified channel • Narrow • Moderate flow • Water level fairly constant	• Unknown fish presence • Wetland habitats in unmodified areas	• Not surveyed	• Surrounding farmland

Table 1.2 (Cont'd): Major watercourse characteristics within the AHHS Watershed

Stream / Slough Channel	Physical Characteristics	Habitat Characteristics (FVRD, 2002)	Current Watercourse Environmental Issues (DFO, 2002)	Possible Pollution Sources
McCallum Slough	• Natural stream 200 m inland from Mountain Slough, then becomes modified channel • Mid-width • Moderate flow • Water level fairly constant	• Fish presence • Wetland habitats in unmodified areas	• Riparian zone loss • Channelization • Significant water quality problems	• Surrounding farmland • Gravel extraction pits
Miami Creek	• Natural stream with 13 km modified since 1972 • Relatively wide • Slow flow • Water level variable depending on Harrison Lake	• Fish presence • Wetland habitats in unmodified areas	• Riparian zone loss • Channelization • Significant water quality problems • Urbanization	• Municipal outfalls • Surrounding farmland • Harrison Resort Golf Course
Mountain Slough	• Natural stream with 18 km modified since 1972 • Relatively wide • Moderate flow • Water level variable depending on Fraser River	• Fish presence • Wetland habitats along full length of watercourse	• Riparian zone loss • Channelization • Significant water quality problems • Logging	• Surrounding farmland • Village of Harrison Hot Springs
Westlin Ditch	• Modified channel • Narrow • Moderate flow • Water level fairly constant	• Unknown fish presence • Wetland habitats in unmodified areas	• Not surveyed	• Surrounding farmland

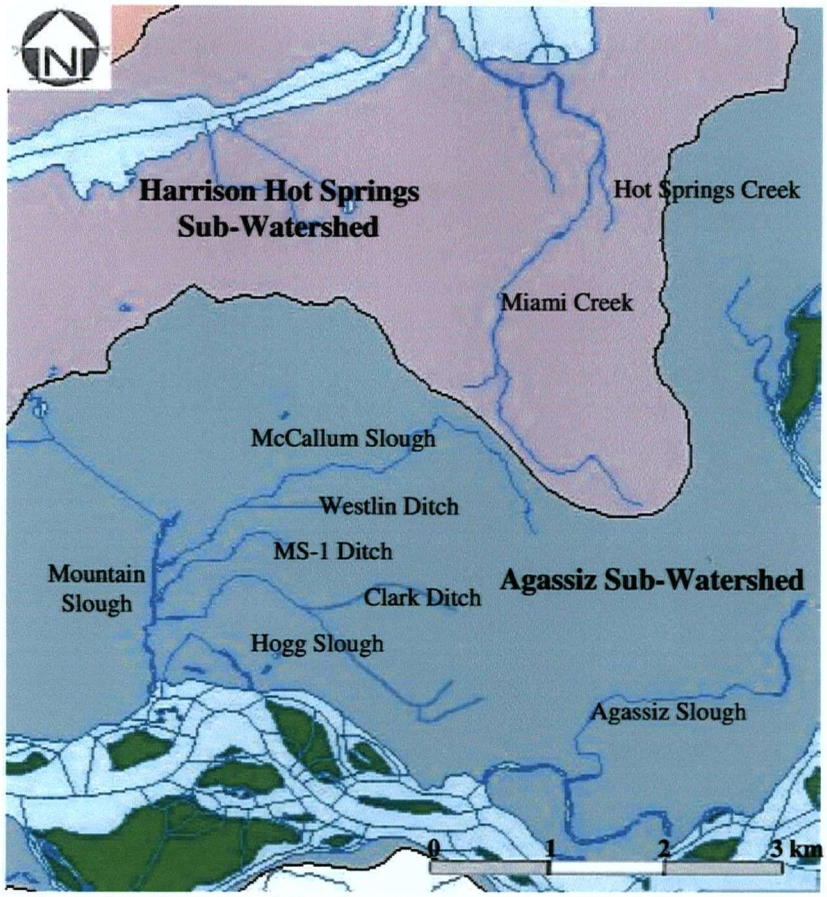

Figure 1.6: Agassiz and Harrison Hot Springs Sub-Watersheds and major watercourses (SHIM, 2002).

1.4.2 Fish Habitats

The presence of fish in watercourses within the watershed has been observed by many of the land owners within the area. Although many of the smaller fish species are present in the more suitable ditches, many land owners have stated that larger fish, such as salmon, were at one time quite common within many of the waterways (Westlin pers. comm., 2002). It is apparent that a significant amount of fish habitat degradation has taken place within the watershed over the past century. In Miami Creek, for example, Coho escapements have been reduced to 25% of the historical capacity (Bussanich *et al.*, 2000).

Miami Creek is one of the most likely candidates to contain fish populations, and the Village of Harrison Hot Springs is interested in protecting any fish species present within it. As a result, there have been a number of initiatives undertaken in order to assess current conditions for a viable fish population and to help in any future management.

Two such surveys include a biophysical over-winter habitat survey by Pacific Initiatives (fall of 2000), and a survey undertaken by Klohn-Leonhoff Consultants (fall of 1989). These surveys showed that the Miami River has significant resources to support a viable fish population (Table 1.3), particularly west of the golf course, which is considered prime spawning habitat for fish.

Table 1.3: Fish species identified in Miami Slough.

Locations Sited	Common Name	Scientific Name
Miami Slough	Coho Salmon [1,2]	*Oncorhynchus kisutch*
	Chum Salmon [1]	*Oncorhynchus mykiss*
	Pink Salmon [1]	*Gasterosteus aculeatus*
	Rainbow Trout [1,2]	*Cyprinella lutrensis*
	Cutthroat Trout[1]	*Oncorhynchus clarki*
	Threespinned Stickleback [2]	*Gasterosteus aculeatus*
	Red Side Shiner [1]	*Richardsonius balteatus*
	Pacific Lamprey [1]	*Lampetra tridentata*

[1]Bassanich *et al.*, 2000
[2]Gregory, 1998

A previous survey of the suitability of the sloughs for fish populations within the District of Kent was conducted in 1998 as part of an assessment of the need for remedial actions which should accompany any watercourse maintenance. The survey classified the ditches into four different groups depending on their ability to support fish populations (Table 1.4).

Table 1.4: Watercourse classifications of fish habitat survey (Gregory, 1998)

Class Designation	Watercourse Characteristics
Class "A"	• Year round or potential for year round habitat for salmonids
Class "A(O)"	• Provide or have potential to provide overwintering habitat for salmonids
Class "B"	• Significant food and nutrient resources, but uninhabited by salmonids
Class "C"	• Insignificant food and nutrient resources, and uninhabited by salmonids

It was found that the majority of the sloughs surveyed within the watershed had Class "A(O)" designations. This designation requires that special precautions be followed for any dredging taking place along those watercourses.

The FVRD has also published a habitat atlas documenting the aquatic habitats of the various sloughs within the AHHS watershed. McCallum Slough, Miami Creek, Mountain Slough and Agassiz Slough all are designated as endangered watercourses with Mountain Slough continuing to degrade in quality with time (FVRD, 2002; DFO, 2002). A variety of conditions such as limited spawning habitat, channel aggregation, and low summer rearing habitat are having a detrimental effect on the ability of these watercourses to sustain fish (Bussanich *et al.*, 2000). The DFO also keeps track of a variety of environmental pressures on various watercourses examined in this study (Table 1.2).

1.4.3 Watercourse Maintenance Issues

In order to ensure that watercourses remain effective drainage structures, seasonal maintenance is commonly completed on an annual basis. This maintenance can take the form of dredging built-up sediments and grasses, widening and bank stabilization, or the trimming of grass within and adjacent to the slough.

Although these actions improve drainage, there is some concern from the DFO that these activities are detrimental to water quality and fish habitats. As a result, the DFO has required a much more stringent and restricted dredging program than current practices. Enhancements to current dredging procedures include:

- "Soft maintenance" cleaning (only cleaning the current channel while leaving surrounding vegetation untouched);
- constructing sedimentation traps along the watercourses;
- enhancing current riparian vegetation zones through the planting of trees and shrubs to encourage shading;
- bioengineering eroded stream banks, rather than using rock and riprap;

- applying specific dredging techniques to each slough present in the watershed, depending on specific riparian zone characteristics and flow patterns; and
- obtaining permits from the DFO before any dredging operations take place.

In addition to these requirements, recent legislation proposed by the BC provincial government states that for fish bearing streams, certain areas of restricted development known as "setbacks" will be required (District of Kent, 2001). This will take a considerable amount of land from farmers for development. Since local farmers see the watercourses as manmade instruments for drainage, there has been some disagreement over the designation of many of the watercourses and tributaries as fish habitat. It is hoped that this report helps to clarify the suitability of the agricultural watercourses within the AHHS watershed to sustain fish populations.

1.5 Non-Agricultural Sources of Pollution

The storm sewer outfalls which drain into Agassiz Slough are the most noteworthy point source of pollution within the watershed. There are a total of five outfalls which drain the Town of Agassiz between Highway 7, the Meadow/McDonald alignments, and the Agassiz Bypass. All of these sewer outfalls discharge into Agassiz Slough. Discharge levels vary depending on the water levels of the slough since pipe grades are relatively flat (District of Kent, 2002).

Another point source of pollution may come from residential nodes outside of the urban boundaries of Harrison Hot Springs and Agassiz which are not serviced by the urban infrastructures. On-site sewage waste disposal such as septic systems are common, and these points may be areas which can contribute nutrients and other pollutants into nearby watercourses. The Harrison Resort Golf Course, located adjacent to the Miami Creek may also be a possible pollution source given that large amounts of herbicides and fertilizers are required to maintain quality golfing facilities. Two decommissioned landfills are also currently present within the study area, albeit not adjacent to any watercourses. These include the Agassiz landfill, located on Cemetery Rd., and a local landfill just north of Cemetery Rd. The Agassiz landfill, decommissioned in 1992, was used to store waste and ash fill from the Kent Incinerator. The other landfill, located in a

wooded area off of Hot Springs Rd., was a general purpose landfill for Harrison Hotsprings, and was closed in the early 1970's.

Various natural processes can contribute to the contamination of the watercourses. Groundwater contamination is a common problem within many agricultural watersheds, particularly within the Lower Fraser Valley (Cook, 1994; Wernick, 1996). Often pollution takes the form of nitrate contamination of nearby waterbodies. Flooding can also result in contamination due to the erosion of adjacent soils which can contain residues of pesticides and fertilizers.

2. LITERATURE REVIEW

2.1 Nutrient Impacts from Agricultural Activities

Levels of nitrogen and phosphorous in watercourses of agricultural regions are commonly found at significantly higher levels than in watersheds untouched by agricultural practices (Hooda *et al.*, 1997; Smolen, 1981). Both of these nutrients are considered "limiting" nutrients and certain minimum concentrations are essential for the survival and maintenance of any aquatic ecosystem. However, if levels rise above the assimilative capacity of the aquatic system, adverse impacts often arise. Eutrophication is one of the most common effects of increased nutrient discharges into watercourses. Higher nutrient concentrations, particularly phosphorous, can result in the overgrowth of macrophytes and undesirable algae (Moss, 1996; Hooda *et al.*, 1997; Stone *et al.*, 1998). Although this may initially lead to higher levels of DO during daylight hours, there is a significant increase in biomass (Heaney *et al.*, 2001), and the subsequent degradation by bacteria will result in severe oxygen depletion creating a stressed aquatic environment. The lack of oxygen often results in fish kills and the loss of species intolerant to low oxygen conditions. In addition, certain algal species such as cyanobacteria often inhabit eutrophic environments when nitrogen becomes limiting, and can produce toxins which pose a health risk to humans and animals (Hooda *et al.*, 1997). Even nutrient levels as low as 0.3 and 0.01 mg/L for inorganic N and P respectively, can result in the growth of noxious aquatic plants and excessive algae blooms (Daniel *et al.*, 1994).

Nutrient concentrations vary depending on the land uses and climate of each watershed. Concentrations generally tend to be at their highest during the early spring season since it is the common time of manure and fertilizer application and there is less crop coverage to stabilize the soil (Lapp *et al.*, 1998). Negative impacts to aquatic ecosystems tend to depend on how well the system can assimilate the additional nutrients. During warmer times of the year, for example, nutrient concentrations will generally decrease as bacterial action and the assimilative capability of riparian zones and aquatic ecosystems increase (Stone *et al.*, 1998). The size of the catchments area can play a role in the assimilative ability of a watershed since larger flood plains allow for an increased capacity for sediment storage (Basnyat *et al.*, 2000). Serious problems often occur when nutrient

concentrations spike during accidental spills of fertilizers, direct defecation into streams by animals or intense rainfall events after fertilizer or manure application (Hooda *et al.*, 1997). Areas with intense animal and manure production are especially susceptible to creating eutrophic water bodies.

2.1.1 Phosphorous and the Phosphorous Cycle

Phosphorous is one of the scarcest elements available in natural systems, but is a key requirement for the growth of algae and higher plants (Moss, 1996). Although present in many different forms, only orthophosphate (PO_4^{3-}) is readily used by the majority of plants and microorganisms (Ginting, *et al.*, 1998). Bacterial transformation of other forms of phosphorous in sediments and water is often required to create orthophosphate in aquatic systems (Figure 2.1). Other inorganic forms include ions such as $H_2PO_4^-$ and HPO_4^{2-}, which usually come from surrounding rocks and sediment. However, in agricultural runoff, many of the inorganic forms come from insoluble inorganic phosphates. These phosphates are dependent on pH. Iron and aluminium phosphates are associated with acidic soils, while calcium phosphates are associated with soils that are more alkaline. Roughly 50 to 90 percent of phosphorous in soils can be in the inorganic form (Daniel *et al.*, 1994).

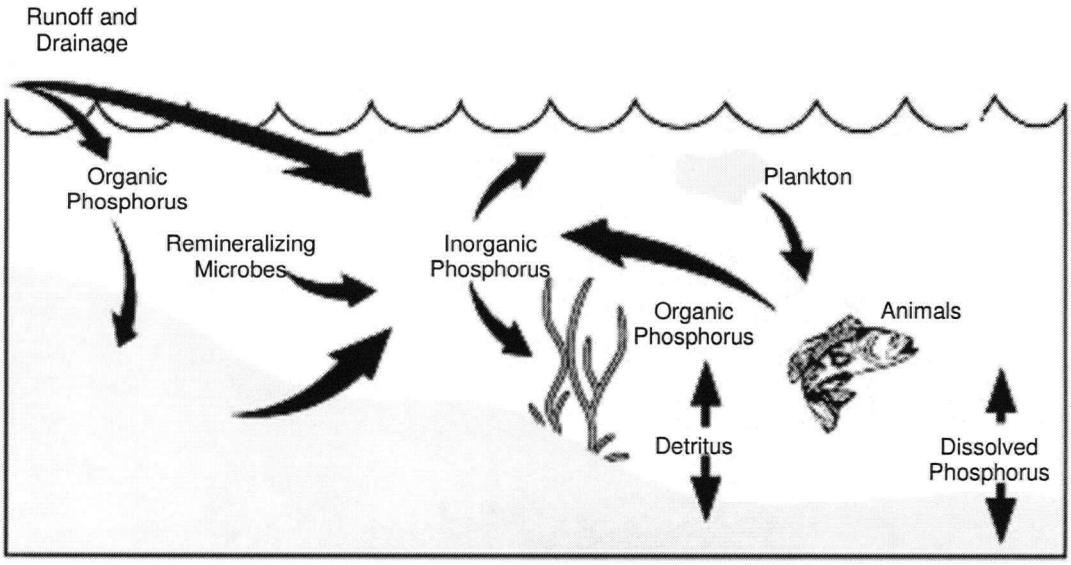

Figure 2.1: The phosphorous biogeochemical cycle in aquatic systems (Adapted from USEPA, 1987).

Organic forms come from animal waste such as manure, or from dead organisms and broken down detritus (Figure 2.1) (Moss, 1996). Common organic forms include glycerophosphates, phosphosugars, phospholipids, and nucleic acids (Daniel *et al.*, 1994). The leeching of phosphorous in soils often takes place when it is in this organic form.

2.1.2 Nitrogen and the Nitrogen Cycle

Nitrogen is an important nutrient for many aquatic organisms, particularly aquatic plants, as it is a crucial component of both proteins and nucleic acids. Although phosphorous is usually a more limiting nutrient, nitrogen can be limiting during certain times of the year to certain forms of algae when the N:P ratio is less than 15:1 (Daniel *et al.*, 1994). Ammonia (NH_3) concentrations as high as 20 mg/L can be toxic to juvenile mammals (Basnyat *et al.*, 2000), and concentrations greater then 2.5 mg/L can be quite toxic to many aquatic organisms (Eghball and Gilley, 1999). Canadian Water Quality Guidelines for the protection of aquatic life give concentrations of 19 μg/L as a safe level for ammonia.

The nitrogen cycle is made up of a variety of stages. Most nitrogen enters into aquatic systems from the atmosphere through a process known as nitrogen fixation. Algae and various nitrogen fixing bacteria primarily perform this function (Figure 2.2). Nitrogen

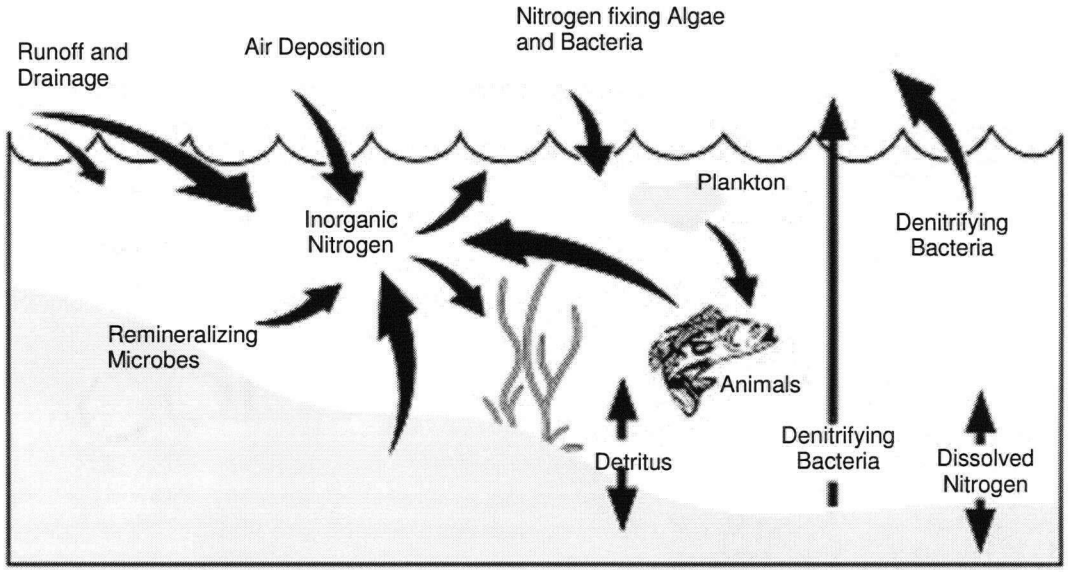

Figure 2.2: The nitrogen biogeochemical cycle in aquatic systems (Adapted from USEPA, 1987).

can also enter aquatic systems due to ammonification, a process where bacteria break down organic matter to create ammonia. Ammonia can appear in the ionic form (NH_4^+) or unionized form (NH_3), with the predominant form depending on the pH of the solution. The unionized form generally tends to be more toxic to fish. Ammonia is the predominant form of nitrogen in anoxic environments, particularly within sediments. If adequate oxygen is available, ammonia can undergo nitrification through the activities of nitrifying bacteria. This is a two step process in which ammonia is converted to nitrite (NO_2^-) and then to nitrate (NO_3^-) (Tchobanoglous *et al.*, 1985). Both nitrate and ammonia are readily taken up by plants as a nitrogen source.

Higher orders of organisms get nitrogen through the consumption of plants and other animals. Under anaerobic conditions, nitrate can be used as an electron acceptor by some microorganisms. This can include the reduction of nitrate to nitrite (anaerobic respiration) or from nitrate directly to nitrogen gas (N_2). This process is known as denitrification (Prescott *et al.*, 1993).

2.1.3 Nitrogen Contamination of Groundwater

The leaching of nitrate into groundwater has become a significant problem in agricultural areas. Groundwater can in turn contaminate stream water, contributing to eutrophication. An increase in the concentration of grazing animals can lead to a marked increase in nitrate levels in groundwater due to the leaching of manure (Stone *et al.*, 1998). The production of corn and concentrated livestock, which are both common in the Kent watershed, are often areas with nitrate contamination in groundwater. The severity of groundwater contamination depends on such factors as quantity, rate, timing, methods of application, management characteristics, hydrogeologic properties of the soil, age of agriculture, and the presence of irrigation (Burkart and Stoner, 2002; Muhammetoglu *et al.*, 2002). Phosphate, with its limited solubility, is often bound to soil particles or immobilized by microorganisms, and thus does not tend to leach deep enough to provide a threat to groundwater (Lapp *et al.*, 1998). The risk of groundwater contamination is greatest during the spring and fall seasons. Manure is often applied during this time, and the assimilative capabilities of the soil are relatively low (Magner and Alexander, 2002).

2.1.4 Nutrient Sources in Agricultural Watersheds

A common source of phosphorous and nitrogen in agricultural watersheds, particularly the AHHS watershed, is manure. Within the AHHS watershed, manure is spread on almost all the corn crops within the region (Zebarth and Paul, 1997). Manure is an indispensable fertilizer for farmers as its incorporation through tillage improves the chemical and physical properties of the soil and increases crop yields (Ginting *et al.*, 1998). For example, soil organic matter in manure will increase ion exchange, improve buffering and chelating capabilities, and improve soil aggregation (Eghball and Gilley, 1999). However, in many cases the spreading of manure can result in excessive nutrient levels in soil. This happens in instances where the nutrient requirements of the soil are underestimated or if the spreading of manure is used as a means for disposal. The resulting surplus of nutrients results in higher nutrient loss through leaching and runoff (Ginting *et al.*, 1998). In many cases, manure is spread over the soil surface rather than incorporated through tilling, and has a greater potential for the runoff loss of phosphorous and NH_4^+ (Daniel *et al.*, 1994; Eghball and Gilley, 1999).

The primary mechanism of entry for manure nutrients into watercourses is through runoff. Nitrate is quite susceptible to this process since it is quite soluble in water. There is also a possibility that manure can fall directly into creeks and sloughs when manure spreaders pass too close to water edges or grazing animals defecate directly into streams. Ammonia from manure is usually converted quite quickly on the aerobic surface of soils and will usually not contaminate surface water under dry conditions. Thus high levels or spikes in concentrations can indicate an overland flow source (Stone *et al.*, 1998). Manure can also contribute a significant portion of phosphorous to the nutrient cycle of a watershed. Animal manure usually has an N:P ratio of approximately 3:1, however most crops require an N:P ratio of 8:1. As a result, excess phosphorous must be added to the soil in order to meet the nitrogen requirements of crops. Impacts can be compounded if additional manure is added to account for nitrogen losses due to volatilization and denitrification (Daniel *et al.*, 1994). This may be of great concern in the south coastal British Columbia since there is almost no carry-over of nitrate in soils between growing seasons (Zebarth and Paul, 1997). However, a study by Wernick (1996), in the nearby

Salmon River Watershed found that there was no linear correlation between high nitrate values in water and spatial inputs of nitrogen from manure, fertilizers and septic systems.

In North America, roughly twice the amount of nitrogen is obtained from fertilizers than from manures. Crops such as corn are especially nitrogen fertilizer dependent (Burkart and Stone, 2002). Nitrogen inputs into water from fertilizers are slightly different than manure since most of the fertilizers are in the anhydrous ammonia form. Although nitrate is easily assimilated by plants, it does not make a good fertilizer due to its leachability into soils. Ammonia, because of its positive charge, readily binds to clay particles in soil and has excellent retention characteristics. The loss of phosphorous from fertilizers is influenced by the rate, time, formulation and technique of fertilizer application (Daniel *et al.*, 1994).

2.2 Erosion

Erosive action within agricultural watersheds is the primary mechanism for the entry of contaminants into the watershed water cycle. Although wind-blown erosion of exposed soils may play a small roll in the process, rainfall induced erosion is the most serious threat to the health of aquatic systems. Agricultural practices invariably create exposed top soils between fall and spring, coinciding with crop harvests and the planting of the coming season's crop. These periods are particularly susceptible to the loss of top soils from rainfall since there is no crop canopy or noteworthy root structure to protect soil integrity (Daniel *et al.*, 1994). Within the AHHS watershed, this is also the time of the greatest amount of precipitation. In many cases, the type of tillage system used in plots can have a remediating effect on the amount of topsoil erosion that takes place (McIsaac, 1991). Generally speaking, however, pasture sites produce less runoff than arable land.

2.2.1 Nutrient Enrichment Due to Soil Erosion

One of the main results of sediment erosion is the flushing of nitrogen and phosphorous into adjacent watercourses. There is a large source of nitrogen and phosphorous associated with agricultural soils, and most nutrient inputs into water are a result of the sediment being mobilized during runoff events. The majority of the nutrient runoff takes place in a short span of time after application, usually after the first one to three runoff events (Daniel *et al.*, 1994). A study of nutrient losses from corn fields have shown that

as much as 160 kg/ha of nitrogen and 70 kg/ha of phosphorous are lost per year (Hargrave, 1995). In addition, when compared with other agricultural crops such as wheat and alfalfa, corn plots consistently showed higher nutrient losses due to erosion. Phosphorous generally tends to associate with particulates making it less available for runoff. It is thus discharged into watercourses at a slower rate than nitrogen. Losses of phosphorous from runoff are generally less than 5 percent of the total applied (Daniel *et al.*, 1994). However, although it is not lost quickly, phosphorous is more subject to accumulation, and high concentrations in soil can provide a long term source of stream contamination (Ginting *et al.*, 1998).

Not every soil type contributes equally to nutrient transport, and each watershed will thus respond differently to runoff events (Basnyat *et al.*, 2000). Fine and very fine sands tend to have lower concentrations of total nitrogen and phosphorous than more coarse gravel fractions (Hubbard, 1982). As a result, different concentrations of nutrients in runoff will vary depending on the intensity of the rainfall event, since the sand fraction will only be mobilized during periods of heavy rainfall. A study in an Ohio agricultural watershed showed that 75% of the runoff was created during only two severe storm events, but accounted for 90% of the total phosphorous lost from soil (Sharpley *et al.*, 1999; Syversen, 2002). Periods of high precipitation may not necessarily result in high pollutant discharges since the presence of a crop canopy, riparian zones and the moisture of the soil affect the assimilative capacity of the terrestrial environment (Lapp *et al.*, 1998). Losses of nitrogen from agricultural fields appear to peak at the beginning of the planting season, and then steadily decrease as the season progresses, most likely due to nitrogen uptake of crops, leaching, and the immobilization of nitrogen in organic matter (Hubbard, 1982).

2.2.2 Sedimentation

Sediment inputs into watercourses from erosive action can have a severe impact on the water quality of agricultural watersheds. A 1996 report by the USEPA showed that the most common agricultural pollutant leading to the impairment of streams was sediment (Nerbonne and Vondracek, 2001). Once in watercourses, sediment can create turbidity in the suspended form or settle onto stream bottoms. In general, community productivity

and respiration are significantly lower in areas dominated by fine sediments since they tend to be more selective for heterotrophic organisms (Rier and King, 1996). One of the most obvious effects of the added turbidity is a decrease in the amount of light which can penetrate to the substrates of the stream (Rier and King, 1996).

An increase in sedimentation affects invertebrates in a number of ways. Benthic invertebrates generally require surfaces free from debris for which to attach. As fine sediments (< 2mm in diameter) cover these surfaces, attachment becomes more difficult. Fine particulates may fill the interstitial spaces in which some invertebrates normally reside (Nerbonne and Vondracek, 2001). In addition, the infilling of these spaces decreases flow through the substrate and reduces substrate heterogeneity (Richards and Host, 1994; Rier and King, 1996). In general, attached algal populations tend to decline when hard surfaces become covered, since these communities require the hard substrates to colonize (Rier and King, 1996; Nerbonne and Vondracek, 2001). Other invertebrates depend on these algae (periphyton) as a food source, and can decrease in number when algal populations decline.

Fish species can be affected by a decrease in invertebrate prey and are also affected by a decrease in visibility since it becomes more difficult to spot prey. Sediment can also cover fish spawning areas and smother eggs, usually located in riffle sections of streams, leading to lower survival rates from egg to smolt (Heaney *et al.*, 2001; Nerbonne and Vondracek, 2001). Observations have shown that a 15% increase in fine sediment results in an egg to fry mortality rate of 97%. At 20% fine sediment loading, the survival rate is subsequently zero (Heaney *et al.*, 2001). In extreme cases, sediment can clog the gills of many fish species making oxygen exchange impossible.

2.2.3 Erosional Impacts of Animals

Pasture animals, cattle in particular, can cause significant mechanical damage to watercourse banks and can also cause increasing turbidity in the watercourse (Trimble, 1994). Mechanical trampling of the riparian vegetation and soft mud adjacent to the bank can result in channel widening and cattle induced slumping of overhanging bands of soil (Williamson *et al.*, 1992). During periods of high flow these impacted sections are more vulnerable to scouring forces. The grazing of the riparian vegetation has a similar effect

in that it exposes the bank to scouring forces (Trimble, 1994). It should be noted, however, that observations of significant impacts to streams from cattle have been limited to mainly smaller streams with widths of 2 metres or less where there is intensive grazing on wet soils adjacent to the watercourse (Williamson *et al.*, 1992). Impacts to larger streams may be more limited.

2.3 Channelization

Channelization is the deepening and widening of drainage channels and water courses in order to improve the drainage of the surrounding land. Such is the case in Agassiz, where sedimentation and the growth of grass and weeds can begin to impede overall flow. Flow alteration has been shown to have a negative impact on the aquatic ecosystems of watercourses (Lapp *et al.*, 1998). In general, it has been observed that chlorophyll levels of channelized streams, in the form of algae and periphyton, are much lower than in natural streams. This is likely because the constantly shifting sands do not provide suitable substrate for colonization (Rier and King, 1996). These organisms are the base of many food chains and are essential for the incorporation of nutrients and energy into ecosystems.

There is also evidence that channel maintenance has an impact on the stability of the drainage channels (Williamson *et al.*, 1992; Rier and King, 1996). Grasses and cobblestones both anchor and protect sediments from erosion during heavy flows after storm events. When this layer of protection is removed, the underlying sediment is more vulnerable to erosive action, resulting in increased turbidity and a greater rate of erosion. In addition, the deepening and straightening effect allows higher water velocities and thus increases sediment transport (Nerbonne and Vondracek, 2001).

The remobilization of these sediments within stream channels can result in sediment deposition in trout spawning areas. Streams which have experienced bank alterations have a significantly higher level of fine sediments downstream than in upstream locations (Heaney *et al.*, 2001). However, it is important to note that, depending on the type of work completed, channel alterations can range from beneficial to detrimental. In beneficial cases, channel work has resulted in increased flow, thereby exposing gravels which salmon can use for spawning (Heaney *et al.*, 2001). One study has shown that

adverse impacts to fish species may not be observed until as much as 10 years after the modification has taken place (Fitzgerald *et al.*, 1998).

One of the main concerns arising from channelization is that contaminants residing in or bound to sediments will be resuspended during the channel work. Metals in particular appear to be quite susceptible to this phenomenon. In general, smaller metal bound particles are more likely to be resuspended for long periods. However, the mobility of metals is also dependent on mechanisms such as sorption-desorption and oxidation from chemical or microbial processes (La Force *et al.*, 1999). Many of these changes come about from the change of an anaerobic to an aerobic environment. For example, zinc has been shown to be released into solution as a result of oxygenation and agitation, while lead tends to stay in its particulate form (La Force *et al.*, 1999).

2.4 Riparian Zone Degradation

Riparian zones are crucial for the proper maintenance of water quality within the watershed. Significant differences have been observed between the physical aquatic habitats of areas with adjacent riparian zones and those without (Richards and Host, 1994; Nerbonne and Vondracek, 2001). It has also been observed that the management of riparian land use can have a greater impact on aquatic habitat quality than controlling land uses upstream (Nerbonne and Vondracek, 2001). In addition to providing shade and temperature control to watercourses, riparian zones act as a buffer from agricultural pollutants in runoff and groundwater. These areas are quite biologically active, and increase the assimilative capacity of the stream, especially for incorporating excessive nutrients in runoff from farm fields (Basnyat *et al.*, 2000).

Much of the impact of sediment fines can be reduced by the presence of buffer zones that stabilize soils on the banks of watercourses and filter soil fines which come from soil runoff (Nerbonne and Vondracek, 2001). The wider the buffer zone, the greater the sediment retention capabilities tend to be.

2.5 Pesticides

Pesticides have become a serious problem endemic to watersheds of intensive agriculture. A variety of pesticides are used to control the growth of various weeds and destructive

organisms, and an estimated 2.5 million tonnes are applied annually to various agricultural crops around the world (Van der Werf, 1996). Unfortunately, no more than 0.3% of pesticides used actually reach, or are consumed by target organisms (Van der Werf, 1996).

Within the AHHS watershed there are a wide variety of pesticides used which have a range of chemical characteristics. The most common pesticides used in the AHHS watershed are listed in Table 2.1. In addition, the toxicological LC_{50} values, which represent the concentration which will kill 50% of the sample population, are also listed. Often, the toxic effects of these pesticides are increased since organisms may be exposed to a combination of different pesticides along with their metabolites (Hunt *et al.*, 1999; Battaglin and Fairchild, 2002). Also, in many instances the pesticides are mixed with additional synergistic compounds which increase their toxicity (McLeay, 1998).

The solubility of each pesticide differs depending on their chemical characteristics, which dictate the mechanism of toxicity. Many of the insoluble pesticides used on corn fields do not necessarily pose a threat under normal rainfall conditions. However, if there are periods of heavy rainfall these pesticides can enter watercourses (Figure 2.3). Pesticide loss can depend on the soil and tillage type, application rates and methods, pesticide properties and the slope of the fields (Ng and Clegg, 1997). The presence of tile drainage structures can also increase the rate of water drainage and subsequently increase the transport of the pesticide. It appears that the critical period for which the majority of pesticides are lost is between 2-6 weeks after the initial application, particularly if there are significant precipitation events within this period (Ng and Clegg, 1997). It is estimated that as much as 2% of the pesticides applied to crops enter into streams during these rainfall events (Van der Werf, 1996; Battaglin and Fairchild, 2002).

Table 2.1: LC_{50} endpoints of common pesticides used within the AHHS Watershed (McLeay, 1998).

Pesticide Trade Name	Herbicide Type	Organism and Test Duration	LC_{50} (μg/L)
2,4-D	Phenoxyacetic herbicide	Rainbow Trout (1.0 g) 96h	24,000
		Rainbow Trout 24h	250,000
		Channel Catfish (1.5 g) 96h	100,000
		Cutthroat Trout 96h	150-1,200
		Daphnia lumholtzi 38h	10,000
Atrazine	Triazine herbicide	Rainbow Trout 96h	2600 – 3200
		Rainbow Trout 96h	4500
		Rainbow Trout 48h	12600
		Daphnia magna 48h	3600
		Bullfrog (*R. catesbeiana*)*	410
Banvel	Dicamba herbicide	Rainbow Trout (0.8 g) 96h	28,000
		Rainbow Trout 96h	135,000
		Rainbow Trout 48h	35,000
		Coho 24h	151,000
		Daphnia magna 48h	>100,000
		Daphnia pulex 48h	11,000
		Amphipod, *G. fasciatus* 96h	>1000,000
		Amphipod, *Gammarus lacustris* 48h	5,800
		Isopod, Asellus 96h	>100,000
Metolachlor	Chloroacetanilide herbicide	Rainbow Trout 96h	2,000
		Carp 96h	4,900
		Daphnia magna 48h	25,100

*(Battaglin and Fairchild, 2002)

Pesticides present in precipitation and air can contribute observable concentrations into surrounding watercourses (Rawn *et al.*, 1999). The effect is most pronounced during the spring since this is often the period of application, and pesticides may volatilize or be subject to wind and dust erosion since little vegetative coverage is present. Observations show that pesticide losses can range from 1-30% due to wind action even during the application process (Van der Werf, 1996).

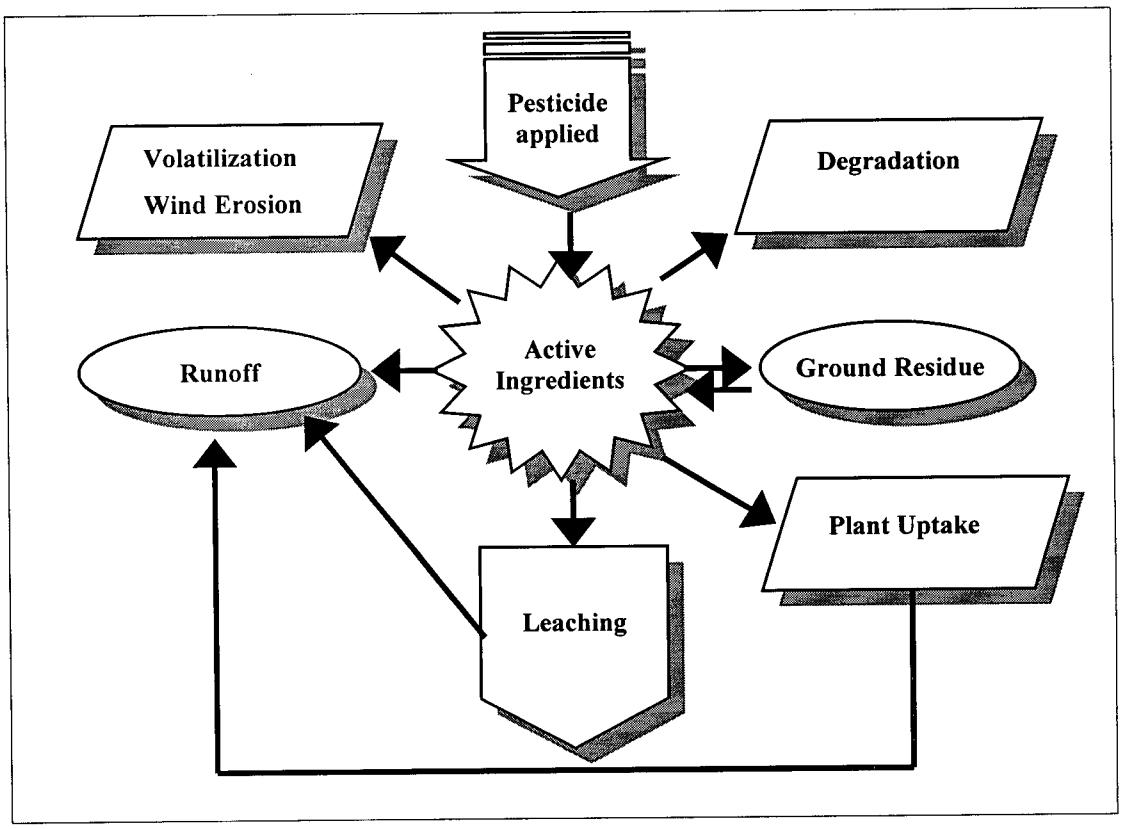

Figure 2.3: Possible routes of pesticide loss from agricultural soils.

2.6 Atrazine

Atrazine is the most widely used herbicide in North America (Pratt *et al.*, 1997), and is the most commonly used herbicide in the watershed under study. It is estimated that atrazine has been used in the AHHS watershed for at least the last 30 years (Boyes pers. comm.., 2002), with application rates within the 2.1-3.1 L/ha recommended by the manufacturer. As a result of its widespread use, it is one of the most commonly found pesticides detected in the water of agricultural watersheds (Kotrikla *et al.*, 1999), and is thus a good indicator of the ability of pesticides to enter the watershed drainage system. Atrazine is both a pre-emergence and post-emergence herbicide, and common application times are between April and late June depending on the climate. It is common for the control of a variety of weeds and it is available in a variety of formulations and mixes. The formulation of atrazine is important since this may dictate its susceptibility to runoff. The most common form within the AHHS watershed is the dispersible liquid, which has

shown losses from soil between 4-8%. Dispersible granule and wettable powder formulations, however, have been shown to have significantly higher rates of runoff (Wauchope, 1987).

During periods of heavy rainfall, atrazine is susceptible to significant movement in runoff, which is the primary mechanism for its entry into watercourses (Lapp *et al.*, 1998). Losses from precipitation make up as much as 75% of the total atrazine lost annually, with the vast majority of the loss taking place within the first 70 days of its application (Ng and Clegg, 1997). The amounts of atrazine available for runoff tend to decrease as much as 50% within the first 24 hours after application (Wauchope, 1987), although a significant rain event can still discharge significant amounts. As a result, levels of atrazine in water tend to peak during times of pesticide application between April and June, depending on the growing season (Ng and Clegg, 1997; Lapp *et al.*, 1998).

2.6.1 Environmental Fate and Chemistry of Atrazine

Atrazine does not tend to associate with suspended particles and has a high to medium mobility potential in soils based on its K_{OC} value (Table 2.2). However it may strongly sorb to organic matter and colloids in water (Hall *et al.*, 1993). Atrazine has a relatively low Henry's Law Constant (Table 2.2) and long volatilization half-life (Table 2.3), indicating that it is normally not very volatile. Nevertheless, significant portions of atrazine and its degradation products have been observed in the air and precipitation. Air samples near Agassiz show an average dry air concentration of 5.529 ng/m^3 while annual air deposition to land is approximately 3.352 μg/m^2 (Belzer *et al.*, 1998). Atrazine concentrations have also been observed during winter periods and at elevated levels during snow melts (Rawn *et al.*, 1999; Hall *et al.*, 1993). It is likely that wind blown atrazine laced soil is present in the air at many locations throughout North America (Hall *et al.*, 1993; Van Der Werf, 1996).

Atrazine ($C_8H_{14}ClN_5$) has a heterocyclic structure (Figure 2.4). Although the eight carbon atoms have a hydrophobic characteristic, there are two amino groups and the

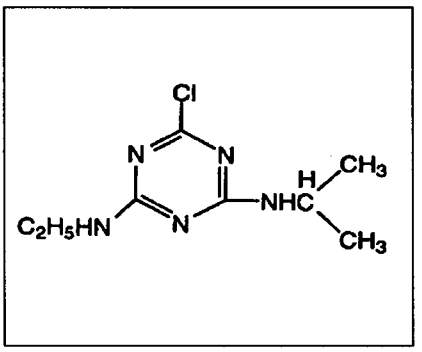

Figure 2.4: Chemical structure of atrazine (2-chloro-4-ethylamino-6-isopropylamino-1,3,5-triazine).

Table 2.2: Chemical properties of atrazine.

Chemical Property	Value	Reference
Log K_{OW}	2.75	(USEPA, 2002)
K_{OC}	122 ml/g	(USEPA, 2002)
Solubility (25^0C)	33 mg/L	(Homes *et al.*, 2001)
Vapour Pressure	3.7×10^{-5} Pa	(Lyman *et al.*, 1990)
Henrys Law Constant	2.9×10^{-4} Pa	(Lyman *et al.*, 1990)
Henrys Law Constant (dimensionless)	1.1×10^{-7}	
Log BCF	0.3 - 2	(USEPA, 2002)

Table 2.3: Half life values of atrazine.

Degradation Conditions	Half Life	Reference
Aqueous Solution	42-180 days	(McEwen, 1979)
Air	60 days	(Homes *et al.*, 2001)
Volatilization	1080 hours	(Lyman *et al.*, 1990)
Microbial Degradation (Aerobic)	146 days	(USEPA, 2002)
(Anaerobic)	159 days	
Photochemical Degradation	2.6 – 25 hours	(Howard, 1991)

unsaturated ring contains three polar nitrogen atoms (Doulia *et al.*, 1997). The heterocyclic ring has a similar structure to that of benzene, but it lacks the same stability due to the presence of nitrogen which is more electronegative than the carbon atoms. As a result, pi-electrons tend to associate more around the nitrogen atoms than the carbon atoms. The higher electron density around the nitrogens creates an area which can act as a weak base within solutions (Doulia *et al.*, 1997). This chemistry plays an important role in the sorption of atrazine to surfaces.

Half-lives for atrazine can vary depending on the pH of the surrounding environment. In general, a lower pH leads to a higher residence time (Doulia *et al.*, 1997). Although the various half lives of atrazine have been determined (Table 2.3), the actual field-measured half lives can be shorter due to the presence of a greater number of degradation pathways in the natural environment (Van der Werf, 1996).

Atrazine degradation can take place due to photolytic reactions and degradation by microorganisms, although field tests show that photolytic reactions play only a minor role (USEPA, 2002). The two major degradation products of atrazine are deethylatrazine and deisopropylatrazine. Both of these products are commonly found with atrazine in stream water (Kotrikla *et al.*, 1999). Deethylatrazine arises from the removal of the propyl group, while deisopropylatrazine arises when the ethyl group is removed. Deethylatrazine generally tends to be the more common form in aquatic systems. It is more soluble than atrazine and is also more resistant to degradation. One study observed that there was no evidence of deethylatrazine degradation during its complete 45-60 day transit time in water (DeLorenzo *et al.*, 2001).

2.6.2 Atrazine Toxicity and Ecological Impacts

Current Canadian Water Quality Guidelines for atrazine give a maximum freshwater concentration of 1.8 μg/L to protect aquatic ecosystems (CCME, 2001). However, based on other research, the no observed effect concentration may be much lower. Atrazine is thought to become ecologically significant at levels between 20 and 50 μg/L (DeLorenzo *et al.*, 2001). Like many other pesticides, atrazine toxicity can be affected by nutrient conditions, with higher levels of nutrients reducing the toxic effect and improving the recovery times of affected organisms.

Atrazine works by blocking the electron transport system in photosystem II during the Hill Reaction (DeLorenzo *et al.*, 2001). Since the primary mode of action is to inhibit photosynthesis within plants, atrazine can be quite toxic to macrophytes, algae, and other aquatic plants present in watercourses. Chlorophyll levels can decrease as much as 41-67% at atrazine concentrations of 1 μg/L (DeLorenzo *et al.*, 2001). A survey conducted by Battaglin (2002) showed that green algae were the most susceptible to triazine herbicides, followed by duckweed. In addition, atrazine concentrations as low as 2.67 μg/L have been observed to inhibit the nutrient uptake of algae by more than 50% (DeLorenzo *et al.*, 2001). Although algae is the most sensitive organism in most ecosystems it is important to note that algal responses can vary depending on concentrations, duration of exposure and the algal species examined. Laboratory tests have shown significant inhibition of overall community productivity, biomass production and species composition when periphyton was exposed to atrazine concentrations as low as 1 mg/kg (Moorhead and Kosinski, 1986). Studies of aquatic microcosms have also shown that high levels of atrazine between 100-300 μg/L can cause a reduction in biovolume and even the collapse of laboratory ecosystems, essentially halting primary production of the community (Moorhead and Kosinski, 1986; DeLorenzo *et al.*, 2001). However, it has been observed that low concentrations between 3-100 μg/L can actually result in an increase in algal biomass and chlorophyll levels in some tests (Pratt *et al.*, 1997). Concentrations of 1.89 μg/L have also been shown to negatively affect planktonic drift populations within creeks (Lakshminarayana *et al.*, 1992). These aquatic organisms are at the base of many food chains, and any long term impacts can also have an impact aquatic habitat as a whole.

Atrazine is generally not very toxic to animals and fish species within ecosystems, and is thus not considered to pose a serious environmental concern, although more recent studies show there may be more significant impacts. Lethal toxicity to fish ranges from 0.8-2.7 mg/L and 6-22 mg/L for invertebrates (Pratt *et al.*, 1997). Atrazine does not have a very high log bio-concentration factor (BCF) (Table 2.2), and will generally not bioaccumulate in fish. However, it has been shown to bioaccumulate slightly in the gall bladder, liver, gut and brain of whitefish (EXTOXNET, 2002).

Atrazine can act as an indirect endocrine disruptor, increasing activity of the aromatase enzyme, which changes testosterone into estrogen (Dodson *et al.*, 1999). A recent study has shown that 10-92% of male leopard frogs (*Rana pipiens*) in the US show gonodal abnormalities from atrazine concentrations as low as 0.2 ppb in water (Hayes *et al.*, 2002). In many cases this has lead to sex reversal and the inhibition of spermatogenesis. The problem is of particular concern since the spring time application of atrazine matches the breeding season of many amphibians. Another study, examining Daphnia, has shown convincing evidence that low levels of atrazine in water can result in a sex determination shift towards the production of more males. This shift is observable at concentrations as low as 0.5 ppb (Dodson *et al.*, 1999). Daphnia and other invertebrates comprise a major portion of the diet of fish, and disturbances to the population can have detrimental impacts on the aquatic food chain.

In areas of high levels of runoff, damaging concentrations of atrazine can persist for several weeks, extending the duration of the impact (Pratt *et al.*, 1997). Generally, recovery from the impacts of atrazine can vary depending on the species and conditions being examined. For example, plankton populations have been shown to recover a few meters down stream from an atrazine runoff source (Lakshminarayana *et al.*, 1992) while a different study showed a plankton recovery time of approximately 7 days after exposure (DeLorenzo *et al.*, 2001).

The two degradation products of atrazine (deethylatrazine and deisopropylatrazine) are both significantly less toxic than the atrazine parent compound, with deethylatrazine being more toxic than deisopropylatrazine (Table 2.4). As a result, a mix of atrazine with its degradation products does not significantly increase its toxicity. However, one experiment has shown that deethylatrazine can significantly reduce chlorophyll *a* content, phototrophic carbon assimilation, phototrophic biovolume, and DO content within a microbial community (DeLorenzo *et al.*, 2001).

Research studying the toxic response of algae exposed to both atrazine and metolachlor show that the combined toxicity does not have as much as an effect as the sum of the individual toxic effects. This would lead to the conclusion that there is actually an

inhibitory effect when atrazine and metolachlor are found together (Kotrikla *et al.*, 1999). Unfortunately, it is difficult to predict the toxic responses of atrazine mixed with a variety of other pesticides in different environments.

Table 2.4: 96-h LC_{50} of atrazine and its degradation products on *Chlorella fusca var-fusca* (Kotrikla *et al.*, 1999).

Chemical		LC_{50} (μg/L)
Atrazine	Replicate 1	68.2
	Replicate 2	76.9
Deethylatrazine	Replicate 1	1,043
	Replicate 2	821
Deisopropylatrazine	Replicate 1	3,824
	Replicate 2	4,504

2.7 XAD Resins

2.7.1 Chemical Properties

The use of macroreticular resins to absorb hydrophobic organic molecules, including atrazine, is gaining in popularity. These solvent impregnated resins have binding and selectivity characteristics combined with good chemical and physical stability. The premise is that they create an environment where the impregnated medium acts as a liquid, but is actually in a solid state which allows for good chemical and physical stability (Juang, 1998). Pesticides are well suited for sorption onto macroreticular resins. One such line of resins that are excellent for this purpose are known as the Amberlite XAD series, created by Rohm and Haas. Ranging in polarity and makeup, the resins XAD-2, XAD-4, and XAD-7 have been shown to give excellent recoveries of common pesticides such as atrazine, 2,4-D, metolachlor, DDT, lindane and aldrin (Doulia *et al.*, 1997; Baun and Nyholm, 1996; Dressler, 1979; Junk *et al.*, 1974).

XAD Amberlite resins are characterized as polystyrene-divinylbenzene copolymers with a mostly hydrophobic nature (Juang, 1998; Junk *et al.*, 1974). These resins have been shown to give good recoveries of atrazine (87±12%) for concentrations as low as 20 parts

per trillion (Junk *et al.*, 1974). XAD-7, a cross linked acrylic ester polymer, contains a more hydrophilic, moderately polar charge (Doulia *et al.*, 1997; Juang, 1998) and appears to give the best recoveries of atrazine in the pH range of the water sampling sites examined in this experiment (Doulia *et al.*, 1997). XAD-7 is in the form of a methacrylate polymer and has a specific surface area of 450 m^2/g with an average pore diameter of 80 Å (Dressler, 1979). Combined with a good pore size distribution and lack of ionically functional groups, the XAD-7 resin is the most chemically compatible with atrazine sorption (Doulia *et al.*, 1997).

2.7.2 Chemistry of Adsorption

XAD resins function by sorbing molecules of similar polarity by mechanisms of dispersion and hydrophobic forces. Sorption efficiency can be affected by a variety of factors, including pH, conductivity and the presence of organic molecules in the sample (Baun and Nyholm, 1996; Doulia *et al.*, 1997). The pH of a solution is thought to alter the adsorptive characteristics of the XAD surface and also cause possible changes to the atrazine molecule due to the ionization of the nitrogen atoms and/or NH groups (Doulia *et al.*, 1997). Lower pHs are generally more favourable for adsorption since the atrazine molecule becomes more cationic allowing for more dipolar and electrostatic interactions. The most favourable pH for the adsorption of atrazine to XAD-7 appears to be at a pH of 5. At this pH it is postulated that there is an increase in polar interactions between the positively charged atrazine molecules in addition to more widespread deprotonation. Below this pH, adsorptive efficiency is reduced as the molecule is less closely packed together, possibly due to the increase in repulsive forces between the positively charged ions (Doulia *et al.*, 1997).

The ionic strength of a solution will also have a significant impact on the adsorptive capacity of the resin. Previous research has shown that an increase in conductivity leads to a decrease in the adsorption of atrazine (Doulia *et al.*, 1997). This is due to the possibility that the atrazine molecules cannot adsorb onto the oppositely charged surface due to competition for sites. Alternatively, the solubility characteristics of the atrazine may be altered, changing the interactions with the resin surface (Doulia *et al.*, 1997).

The presence of organic molecules in a solution would also tend to decrease the adsorptive efficiency of the resin. Although the XAD-7 resin has a large surface area, the presence of other organic molecules with a similar nature to atrazine can take up adsorption sites on the resin. In addition, atrazine will strongly bind to any colloids or organic molecules present in the solution, thus decreasing its availability to bind to the resin.

2.8 Metals

2.8.1 Metal Sources

Metal contamination in agricultural watersheds can come from a variety of different sources. The primary sources are fertilizers, animal feed and the resulting manure. Metals such as copper, zinc, manganese and iron are important nutrients for plants and animals and they are quite common in feeds and fertilizers. Swine manure, in particular, has been shown to contain high concentrations of copper, manganese, and zinc, at levels of 343, 121, and 577 mg/kg respectively (Hsu and Lo, 2000). The most common supplements are zinc, in the form of zinc oxide or zinc sulphate, and copper, in the form of copper sulphate. Animal feed for cattle usually contains at least 25 ppm of zinc and 10 ppm of copper (AFRD, 2002). Like nutrients and pesticides, these metals are also subject to runoff events and can be discharged into surrounding watercourses especially when flows transport sediments to which metals usually bind.

Metal concentrations found in agricultural watercourse sediments should always be compared with background concentrations, since high concentrations, particularly above provincial and federal regulations, can be naturally occurring. Table 2.5 below lists some of the background concentrations of sediments and soil observed throughout North America.

Chronic metal contamination in watercourses close to rural highways and small urban areas is common within agricultural watersheds (Legret and Pagotto, 1999). Areas of impervious land cover, in the form of parking lots, streets and highways are probably the most significant contributors to the observed increases in metals concentrations in freshwater receiving bodies (Smith *et al.*, 2001).

Table 2.5: Typical trace element concentrations in sediments and soils in North America (Cook, 1994).

Material	Cu (mg/kg)	Zn (mg/kg)	Pb (mg/kg)	Mn (mg/kg)	Fe (%)
Western US sediments (<63 μm fraction)	0-110	49-510	9-52	-	-
Upper Illinois R. Basin low order streams median (<63 μm fraction)	23	100	27	-	2.9
NTS 92G Vancouver map sheet (<177 μm fraction)	26	48	7	322	2.02
Streambed Sediment Concentrations in the Brunette Watershed, BC (Lower Fraser Valley urban watershed)	25-279*	60-391*	22-407*	194-3402*	-
US Soils and Surficial Materials	25	60	19	550	2.6
Canadian Soils (uncultivated)	22	74	20	-	-

*(McCallum, 1995)

Compared to the more pervious soils in agricultural areas, only a relatively small amount of rainfall is required to flush metals from the impermeable surfaces into streams. This initial period of runoff is known as the "first flush" event. First flush events refer to the initial period of a rainstorm where runoff flows, particularly from highways, increase significantly (Lawson and Mason, 2001). These periods result in the resuspension of formerly immobile metals deposited on streets, highways and other impervious surfaces. Metals within stormwater outfalls can also be resuspended during these periods resulting in the discharge of high concentrations of metals into receiving water bodies within a very short span of time.

Each metal can have a variety of specific sources. For example, iron is usually the result of the corrosion of raw iron and steel products (Characklis and Wiesner, 1997). Vehicle brakes tend to give off significant amounts of copper, while tire wear tends to deposit high concentrations of zinc on roadways (Table 2.6). The main source of lead is leaded gasoline, however with its recent disuse, concentrations have decreased significantly throughout urbanized watersheds (Legret and Pagotto, 1999).

Table 2.6: Estimated trace metal loadings from automobiles and in stormwater runoff in the Brunette Watershed, BC (McCallum, 1995)

Metal	Automotive Source	Calculated Automotive Trace Metal Loading (kg/day)	Estimated Stormwater Trace Metal Loading (kg/day)
Pb	Fuel	3.1	5.6
Cu	Brake pad wear / fuel	2.1	1.7
Zn	Tire wear / fuel	6.0	2.0
Mn	Fuel additive	3.4	2.7

Common metals present in urban runoff include copper, zinc, iron, lead, manganese, nickel, arsenic and chromium (Sansalone and Buchberger, 1997; Marsalek *et al.*, 1999). Metal concentrations in water and sediment tend to vary significantly depending on the surrounding environmental chemistry, time of season and amount of precipitation.

2.8.2 Metal Chemistry and Environmental Fate

Metals can be found in both the dissolved and particulate bound form. The various proportions of the metals which can be found in each fraction depend on the metal chemistry, rainfall pH, nature and quantity of solids and the solubility of the metal (Sansalone and Buchberger, 1997). The redox potential and dissolved oxygen concentration of the water will also have an impact on overall metal solubility. A significant portion of metals in stormwater sewers are associated with particulates being flushed into receiving water bodies. Metals will normally be associated with the smallest sized particles of sediments in runoff, largely because the smaller size gives an overall larger surface area. Carbonates and iron oxides will bind some metals in sediments (Yu *et al*, 2001), and a study of copper and zinc salt concentrations in agricultural soils showed that there was little partitioning of metals into the soluble phase (Wauchope, 1987). Clay mineralogy will also have an impact on the binding capabilities of the soil and affect the overall mobility of metals. Thus soil loss is highly correlated with metal discharges into adjacent watercourses. Organic matter also has a very high capacity to absorb, and thus associate trace metals with the particulate phase. For example, organic inputs such as fall leaf litter have been observed to have an impact on metal fluxes in sediments throughout a watershed (Lawson and Mason, 2001).

Although not as significant, metals can also partition into the water column in the dissolved ion form. The dissolved form of metals is of particular interest since the dissolved form represents the most available form to organisms, and thus the most likely to result in adverse effects. When the source of dissolved metals is an impervious surface the pavement residence time will play an important role, with longer retention times resulting in higher concentrations of dissolved metals (Sansalone and Buchberger, 1997).

Unlike other forms of contamination, metals do not degrade in the environment, but will instead tend to accumulate in sediments, particularly around stormwater outfalls. Metals generally appear in high concentrations near outfall sources, and then decrease in concentration exponentially with increasing distance (Rhoads and Cahill, 1999). However, there is a significant degree of variability within creeks and river systems depending on the various fluvial-geomorphic conditions and sediment dynamics. This creates various "hot spots" within streams (Rhoads and Cahill, 1999). The highest metal concentrations are usually found in low-energy environments where finer sediments and organic matter are allowed to settle and accumulate. The lowest metal concentrations tend to be in the riffle areas with high rates of erosion and sediment movement.

The residence time of metals within a watershed is dependent on the predominant form of the metal. For example, copper and zinc will tend to be flushed relatively quickly from watershed systems since they are more likely found in the dissolved form. Iron and lead, on the other hand, reside more often in the particulate bound form, and will take much longer to be removed from a system (Sansalone and Buchberger, 1997).

2.8.3 Metal Toxicity

Metal contamination can have significant impacts on affected ecosystems. This can include the loss and degradation of aquatic habitats and the infliction of both short and long term damage to plants and animal life in the receiving water bodies (Rhoads and Cahill, 1999). If toxic concentrations of metals are reached, more tolerant life forms begin to replace and out compete more susceptible organisms, changing the makeup of a community (Lehmann *et al.*, 1999). Quite often the lowest trophic levels, which include bacteria and daphnia, are the first to show the impacts of metal contamination. Because

of this sensitivity, they have often been used as indicator organisms of impacts of urban runoff. This toxic effect is especially prominent in sediment-dwelling bacteria where concentrations of metals in the interstitial space are at their highest (Wei and Characklis, 1994). One study has shown that bacterial communities will begin to shift to a pollutant tolerant population when zinc concentrations as low as 1 μM are exceeded (Lehmann *et al.*, 1999).

Different aquatic plants, including fungi and algae, are able to accumulate different concentrations of metals via specific known and unknown physiological mechanisms (Sawidis *et al.*, 1995). In general, metals will accumulate in the highest concentrations in the roots and rhizomes while the lowest concentrations tend to be in the stem and seeds (Sawidis *et al.*, 1995). Periphyton (algae and fungi) in particular, have the innate ability to build up high concentrations of metals. Periphyton take up metals through direct uptake of the dissolved fraction, bioaccumulation, sorption, or the trapping of metal rich particulates (Besser *et al.*, 2001). Algae, which are also a crucial part of the aquatic community, are quite tolerant to high metal concentrations. In some cases, photosynthesis has actually been observed to increase with higher concentrations of zinc since it may pose as an important micronutrient (Lehmann *et al.*, 1999).

Benthic organisms tend to be exposed to the highest concentrations of metals since they reside within or close to sediments within receiving water bodies (Yu *et al.*, 2001). In addition, consumption of periphyton can become a source of metal exposure to many macrobenthic organisms. Impacts of metals to this trophic level, however, are species specific and dependent on feeding habits. Benthic grazers, for example, have been shown to accumulate high concentrations of zinc and cadmium (Besser *et al.*, 2001).

Fish and daphnia populations are generally quite sensitive to dissolved metals present in the water column. For example, both daphnia and fathead minnow fish cells have shown the same toxicity correlation to increasing concentrations of metals (Dierickx *et al.*, 1996). Fish often receive a significant amount of metal from their prey in addition to exposure from the water column. Invertebrates such as the midge larva, caddis fly nymphs and mayfly nymphs can accumulate metal easily since they reside close to or

within sediments. High concentrations within these invertebrates are invariably transferred to fish. One study has shown that concentrations of zinc, copper and cadmium were at toxic levels in juvenile Chinook salmon due primarily to the consumption of contaminated prey (Sailki *et al.*, 2001). Out of all the metals, copper appears to be the most bioavailable to fish through bioconcentration and biomagnification and may be the limiting factor in many fish populations within contaminated watersheds (Besser *et al.*, 2001).

2.9 Aquatic Moss Bioassays

2.9.1 Advantages of Aquatic Mosses

Recently, the use of aquatic mosses is increasingly useful as an indicator of environmental quality. There are a number of advantages to using aquatic mosses as indicators:

- bryophytes (mosses and other vascular plants) can survive under conditions of high contaminant concentrations;
- contaminant accumulation is rapid as opposed to its slow to medium release rate;
- an absence of root structures greatly simplifies uptake dynamics of contaminants; and
- the mosses are commonly found in the northern hemisphere, and thus can be transported to various locations to monitor different systems (Cenci, 2000).

Moss growth and concentrations of chlorophyll *a* and *b* are connected to a variety of different metabolic processes that can be affected by the presence of herbicides, dissolved metals, or the inadequate presence of the factors necessary for growth. This increased sensitivity to environmental contaminants is due to a lack of developed cuticula and vascular tissue (Davies, 2002).

2.9.2 Aquatic Moss Characteristics

The moss species *Fontinalis antipyretica* was chosen for this study as a bioassay. *Fontinalis antipyretica* is a large, dark green aquatic moss with small leaves present in a

three-ranked formation. The moss can usually be found attached to rocks, logs, or other mobile submerged objects in somewhat shaded sites (Schofield, 1992). *F. antipyretica* is found in many locations throughout BC, including the Agassiz area. There is also a significant population found in the Endowment Lands near the University of British Columbia campus in a tributary known as Cutthroat Creek from where moss samples were collected for this study.

Although the mosses from both locations are of the same species, there are slight physiological differences between the two. It is thought that differences in flow has resulted in greater branching of the stems, in addition to a more wood like attribute to the main stems from the Agassiz area (Davies, 2002). Based on research conducted by Davies (2002), observable physiological changes from contaminants are more prominent in moss samples from the Endowment Lands then those from the Agassiz area.

2.10 Geographical Information Systems (GIS)

2.10.1 Advantages of GIS

Once a land use/water quality correlation has been determined in a specific watershed, it is possible to use GIS to predict the impacts of future developments and changes in land use. This technique can thus replace costly and complicated field surveys, especially over large geographical areas. Richards and Host (1994) successfully used this technique to determine a correlation between nutrient and sediment inputs and the composition of invertebrates and habitat quality, making invertebrate sampling unnecessary.

Another benefit of using GIS is that it can provide a visual database of all pertinent water quality information collected, which makes it easier to observe significant land uses affecting water quality. The technique is especially useful with the increase in the availability of digital spatial information available from both private and public groups. By using this information, land uses can often be ascertained without the use of actual site surveys. A mapping resolution as low as 16 ha/mapping unit is the minimum to predict major water quality trends (Richards and Host, 1994).

2.10.2 GIS Studies of Non-point Source Pollution

GIS has been used successfully in a number of research venues in order to make the connection between land uses and water quality, particularly in agricultural watersheds. Basnyat (2000) developed a "land use/land cover nutrient linkage model" based on data delineated using GIS land use data and water quality parameters of different sites. Many experiments studying land use and water quality only examine a fixed buffer area around watercourses. For example, a study of non-point source phosphorous showed that correlations in phosphorous were more closely associated with near-stream land uses within 60 meters of the stream boundary (Sharpley *et al.*, 1999). However, the exact buffer distance used is dependent on the mobility of the factor causing the impact.

A variety of indices can also be incorporated into GIS databases. These can include sediment delivery ratios such as the Universal Soil Loss equation to determine sediment loadings into streams. Factors such as pesticide sales data in specific areas throughout the watershed have also been used to estimate the amount of pesticide use (Homes *et al.*, 2001). If the specific topography of the watershed can be ascertained from available data, factors such as degree of slope can be incorporated into the database. The database can then be used to determine the relative potential impact of different land uses within the context of different factors affecting runoff. These factors can be subsequently imported into mathematical models (Gardi, 2001). Unfortunately, this information is not available for the District of Kent watersheds. Thus, the most feasible variable is the land uses surrounding the various watercourses. This technique has been successfully applied by McFarland (1999) where agricultural land uses above sampling sites were compared to downstream water quality using correlation and regression analysis. This method proved to be quite effective in finding major sources of non-point source pollution in agricultural watersheds.

A variety of GIS studies have been conducted within the lower mainland area which specifically examine watersheds composed primarily of agricultural land uses. Cook (1994) used GIS to study connections between groundwater quality and land uses within the Salmon River Watershed, with an emphasis on nitrate-N concentrations. Correlations were compared using a density index, which represented land uses immediately upstream

of the sampling point, and a cumulative density index, which incorporated all land uses upstream from the sampling point. Cook determined that stronger correlations were made by using a cumulative density index, thus indicating water quality was representative of all land uses upstream from the point of sampling. A second GIS study within the Salmon River Watershed examined the effect of changes in land use on surface water quality (Wernick, 1996). In this case nitrate-N, ammonia-N, orthophosphate and faecal coliforms were measured in surface water in an attempt to correlate various land uses with a particular water quality impact. GIS was effective in determining if factors such as animal and septic system densities, were having an impact on overall water quality.

The concept of "contributing areas" was used by Berka (1996) in the nearby Sumas Watershed in order to correlate agricultural land uses with water quality. In this technique, areas were divided into sub-watersheds based on topographical data. The contributing area principle is based on the assumption that land use within each sub-watershed upstream of the sampling point is the primary factor affecting water quality. Similar to the research conducted by Cook, it was determined that all land uses upstream from the sampling point, regardless of distance, were shown to give better correlations to water quality then land uses within the sub-watershed of the sampling point.

3. Methodology

3.1 Sampling Methodology

In total, 28 sampling stations were studied throughout the course of this research. Sampling sites were chosen based on the previous sampling program conducted by the Agassiz Drainage Committee to monitor water quality. These site locations ensured an adequate distribution along major watercourses and tributaries, and provided ease of access by road. Initially, 22 sampling locations throughout the watershed were chosen. However, in August 2001, five additional Miami Slough sampling sites (control, CO, C4, C5, and C6) were added to correspond with a similar sampling program being conducted by Lea Elliot. The control station was chosen to represent stream water quality before any form of contamination from the watershed is introduced. Water from this station comes directly from Mount Agassiz and does not travel through any farmland before the point of sampling. In addition, the Mountain Slough sampling station (MTN-1) was added during the month of February 2002 since it may indicate the combined overall water quality of the drainage water before it enters into the Fraser River. Water quality results from the spring station were used as a measure of the general groundwater chemistry.

Samples of water were taken approximately every month for 12 months from June 2001 to June 2002. Sampling was also conducted by the Agassiz Drainage Committee which was incorporated into the analysis of dissolved oxygen, temperature, and pH. Sampling dates for the Agassiz Drainage Committee are shown in Figure 4.1 with data listed in Appendix B. Other water quality variables collected by the Agassiz Drainage Committee, such as NH_3-N and turbidity were not incorporated due to differences in precision and analytical technique. Sediment samples were taken in June 2001 and February 2002 in order to observe any differences in sediment quality between wet and dry seasons. In total, 12 sampling trips were made to the AHHS watershed, with additional trips made in order to install and monitor moss bioassays and resins. Sampling took place under a variety of climatic conditions and during various degrees of precipitation.

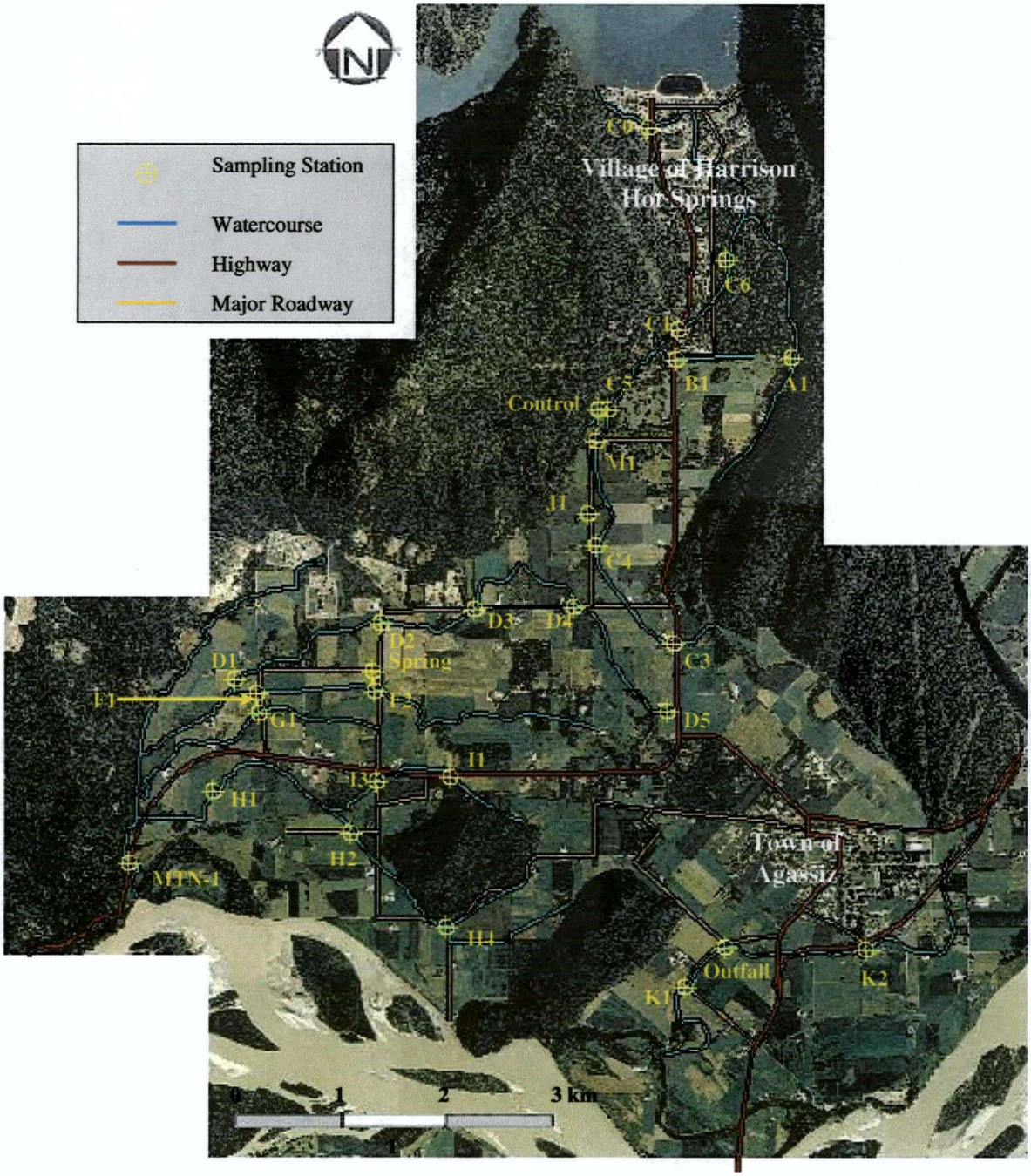

Figure 3.1: Location of water sampling stations throughout the AHHS Watershed.

Table 3.1: Sampling locations and methods of sampling at stations. Stations are arranged from upstream to downstream for each watercourse.

Sampling Station	Watercourse	Sampling Location	Sampling Methods		
			Water Sample	Moss Bioassay	XAD-7 Resin
K2	Agassiz Slough	Northeast side of McDonald Rd., just south of the Haig Hwy.	X	X	
Outfall	Agassiz Slough	Southeast side of the Fir Rd. and Tuyttens Rd. intersection by stormsewer outfall.	X	X	
K1	Agassiz Slough	Northeast side of Whelpton Rd., just southeast of the Whelpton Rd. and Tuyttens Rd. intersection.	X	X	X
I1	Clark Ditch	Southside of the Lougheed Hwy., East of Wells Rd.	X		
I3	Clark Ditch	East side of Cameron Rd., just South of the Lougheed Hwy.	X	X	
H4	Hogg Slough	East side of Hamilton Rd., just south of Pioneer Ave.	X	X	
H2	Hogg Slough	North side of Hunt Rd.	X	X	
H1	Hogg Slough	East side of dirt road in farm field, south of Lougheed Hwy and East of Cameron Sutherland Rd.	X	X	X
D5	McCallum Slough	North side of the driveway, just west of Hwy. 9.	X	X	
D4	McCallum Slough	North side of McCallum Rd, past Golf Rd.	X		
D3	McCallum Slough	North side of McCallum Rd., just east of the gravel pit and Else Rd.	X	X	
D2	McCallum Slough	East side of Cameron Rd.	X		
Spring	McCallum Slough Trib.	North side of Humphrey Rd., just west of Cameron Rd.	X	X	X
D1	McCallum Slough	Behind the house of Ted Westlin off of McCallum Road and Sutherland Rd.	X	X	X

Table 3.1 (Cont'd): Sampling locations and methods of sampling at stations. Stations are arranged from upstream to downstream for each watercourse.

Sampling Station	Watercourse	Sampling Location	Sampling Methods		
			Water Sample	Moss Bioassay	XAD-7 Resin
C3	Miami Creek	East side of Hwy. 9, south of McCallum Rd.	X	X	
C4	Miami Creek	South side of dirt road in farm field, east of Golf Rd.	X	X	
J1	Miami Creek Trib.	East side of Golf Rd.	X		
M1	Miami Creek	North side of Golf Rd., east of the Harrison Resort Golf Course.	X		
Control	Miami Creek Trib.	West side of driveway.	X	X	X
C5	Miami Creek	North side of bank at end of driveway	X	X	
B1	Miami Creek Trib.	South side of McPherson Rd., just east of Hwy. 9.	X		
C1	Miami Creek	West side of Hwy. 9, north of McPherson Rd.	X	X	
C6	Miami Creek	End of walking trail east of McCombs Drive, just north of bridge over Miami Creek.	X	X	
A1	Miami Creek Trib.	North of dirt road on farmland east of McPherson Rd. and McCombs Dr. intersection	X		
C0	Miami Creek	East side of Harrison Hot Springs Rd.,	X	X	X
MTN-1	Mountain Slough	South side of the Lougheed Hwy. bridge, beside Hammersly Pumping Station	X	X	
G1	MS-1 Ditch	West side of Sutherland Rd.	X	X	
F2	Westlin Ditch	West side of Cameron Rd, south of Humphry Rd.	X	X	X
F1	Westlin Ditch	West side of Sutherland Rd.	X	X	

3.1.1 Water Samples

High density polyethylene bottles were used to sample water from each sampling station in order to conduct additional testing for pH and chlorophyll in the laboratory. The same bottles were used each month for the same sampling stations to ensure that there was no cross contamination from other stations. Sample bottles were prepared by machine washing for two cycles, and rinsing with distilled water.

Water samples were obtained by immersing the bottle completely in the water, as close as possible to the middle of the watercourses, but as far away as possible from colonies of algae and other biofloc material present on the surface of the water. In cases where the presence of long grasses and/or steep banks made this impossible, a 4 m pole with an aluminium pot at one end was used to extract the water from the middle of the watercourse. All samples were stored in a cooler with ice packs. Samples were stored in the lab at 4^0C until analysis and analyzed within two days of being sampled.

Samples to be tested for ammonia, orthophosphate, nitrate, and nitrite were collected in 10 ml test tubes. As with the grab samples, the same test tubes were used at each sampling station to ensure there was no cross contamination from the previous sampling of other stations. Test tubes were washed twice in a dishwasher and then rinsed with distilled water and allowed to dry. Before sampling, 5% H_2SO_4 was added to ammonia sampling tubes and phenylmercuric acetone was added to nitrate, nitrite, and orthophosphate sampling tubes as a preservative. As with the grab samples, samples were taken away from algal colonies and under the surface of the water to avoid the incorporation of large groups of algae and other aquatic plants floating on the surface. Samples were filtered using a coarse serum filter before being added to the preservative. Samples were stored in a cooler with ice packs during transport, and then stored at 4^0C until analysis. Samples were analyzed within a week of collection. Results for nitrate, nitrite and ammonia were all given as N, while orthophosphate was given as P.

3.1.2 Sediment Samples

Sediment samples were obtained by using a 4 m pole with an aluminium pan at one end. Before sampling, the pan was rinsed with the water from the sampling station. Samples were taken as close to the middle of the watercourse as possible, and stored in plastic

bags. Each sample was double bagged in order to prevent leakage and cross contamination with other samples. Samples were stored in a cooler with ice packs during transport and refrigerated at 4^0C until analysis for metals, loss on ignition, and grain size was conducted. Samples were analyzed within one month of collection.

3.2 Water Analysis

3.2.1 Field Analysis

Temperature, DO, and specific conductivity were measured using handheld field instruments developed by the Yellow Springs Instrument Company (YSI). Temperature and conductivity were recorded using YSI model 30, while DO was measured using YSI model 95. Both probes were calibrated during each sampling trip. Turbidity samples were analyzed using a Hach P2100 Turbidometer. Sampling vials were rinsed with the sampling site water before analysis.

3.2.2 Laboratory Analysis

Samples for the analysis of chlorophyll *a* were first pre-filtered using cheesecloth to ensure removal of duckweed and larger aquatic plants. The water was then filtered through 0.45 μm nitrocellulose membranes using suction filtration. Filters were transferred into a centrifuge tube filled with 10 ml of 90% acetone/water solution. If analysis was to take place at a later date, filters were frozen until use. Before analysis, tubes were centrifuged at 2500 rpm for five minutes. Measurements were made on a 10-AU Turner Fluorometer and readings were recorded both before and after the addition of a solution of 5% HCl. Final chlorophyll *a* concentrations were determined using the equation below.

Equation 1: Equation for the calculation of chlorophyll *a* in water based on fluorescence

$$Chla = 1.84x1.0326(F_0 - F_a)x\left(\frac{v}{V}\right)$$

Where:
- Chl *a* – Chlorophyll *a* Concentration (μg chl *a*/L)
- F_0 – Fluorescence of sample
- Fa – Fluorescence of sample after addition of 5% HCl
- v – Volume of acetone used for extraction
- V – Volume sample filtered

Measurements for pH were taken using a Beckman Φ 44 pH meter. Calibration of the pH meter was completed by using pH 4 and pH 7 buffers. The instrument was re-calibrated after every 15 samples to ensure that instrument drift was not having a significant effect on results. Samples were stirred continuously during analysis.

A Lachat QuickChem 8000 Flow Injection Analyzer was used for the analysis of nitrate, nitrite, ammonia and orthophosphate samples. Nitrate-N and Nitrite-N samples were analyzed according to Standard Methods No. 4500-NO_3^- - Cadmium Reduction Flow Injection Method. Nitrite was analyzed by removing the cadmium column and repeating the analysis. Ammonia concentrations were determined by Standards Methods No. 4500-NH_3 - Flow Injection Method, while Orthophosphate concentrations were found by applying method 4500-P - Flow Injection Analysis of Orthophosphate (Standard Methods, 1987). The overall detection limits for each technique was 10 μg/L for each water quality variable.

3.3 Sediment Analysis

3.3.1 Physical Properties

Roughly 5 grams of each sediment sample were dried in an oven at 105^0C overnight. In order to determine loss on ignition, exactly 1 gram of each dried sample was put inside a muffle furnace at 550^0C for two hours to combust any organic material. Loss on ignition was then determined by dividing the weight loss between the 105^0C drying and 550^0C combustion by the dry weight.

Grain size distribution was determined by dry sieving samples on a mechanical shaker for 10 minutes with Tyler stainless steel sieves. Sieve numbers used were #18 (1.00 mm), #40 (0.425 mm), #60 (0.250 mm), #120 (0.125 mm) and #230 (0.063 mm).

3.3.2 Metals

The digestion technique was derived from the technique used by Joop (1991). However, due to the high organic content in many of the samples, samples had to first be ashed at 400^0C to reduce the organic content while not volatilizing any of the metals present in the sample. In order to verify that this alteration did not result in any loss of metals, the reference sediment MESS-2 was used to compare the accuracy of both techniques.

The <63 μm fraction, which was taken after grain size separation, was used for metal analysis of the sediment. This fraction represents the clay/silt particles, and is the fraction in which metals tend to accumulate (McCallum, 1995; Berka, 1996). The MDS-81D (Microwave Digestion System) was used to conduct the digestions. Exactly 1 g of ashed sample was put in a Teflon microwave digestion capsule along with 10 ml of an aqua regia solution (3:1 $HCl:HNO_3$). Samples were first digested at 30% power (195 Watts) for 1 minute, then 80% power (520 Watts) for 4 minutes, and then at 100% power (650 Watts) for 30 minutes. Stoppage of the digestion was required approximately every 15 minutes to allow samples to cool and prevent over boiling and loss of sample. Digested samples were then filtered using Whatman 541 filter paper. Volumes were brought to 50 ml in a volumetric flask using deionized distilled water. Analysis took place using a Spectra AA 220 Atomic Adsorption Spectrophotometer. An air/acetylene flame was used for all the metals analyzed. Metals analyzed were copper, lead, zinc, manganese and iron. Table 3.2 below lists the calculated detection limits determined during metal analysis. Final concentrations found in the sediments are presented as mg/kg dry weight for the silt/clay fraction of the sediment.

Table 3.2: Detection limits of analyzed metals

Metal	Detection Limit (mg/kg)
Copper	0.026
Lead	0.015
Zinc	0.010
Manganese	0.016
Iron	0.044

3.4 Atrazine Analysis

3.4.1 XAD-7 Resin Preparation

XAD-7 resins are pre-treated with anti-microbial chemicals and residual monomers which can interfere with uptake and analysis of the substance of interest (Dressler, 1979). As a result, pre-treatment of the membrane was required before use. Resins were initially washed twice in methanol and then decanted to remove any fines present. Then, using a

soxhlet extractor, the resin was washed with sequential 8 hour extractions of methanol, acetonitrile, and diethyl ether. The final, cleaned resins were then stored in methanol to ensure purity. Before use, the resins were rinsed three times with Milli-Q deionized distilled water.

3.4.2 Membrane Preparation

The membrane used to hold the resin was a Spectra/Por 20.4 mm diameter molecular porous membrane with a 6000-8000 molecular weight cut off. The membrane not only holds the resins in place, but also keeps out humic substances and larger molecules which may also absorb onto the resin surfaces. Spectra/Por membranes are packaged in the dry form and thus a pre-treatment procedure was required before use. The membrane was first soaked in deionized distilled water for 30 minutes. The membrane was then placed in a 1 L solution of deionized water and 2% sodium bicarbonate and kept in an oven at 60^0C for 30 minutes. Finally, the membrane was rinsed with deionized water, and stored in a solution of 1% formaldehyde until use.

3.4.3 Bench Scale Experiment

In order to test if atrazine recovery was possible by XAD resins, flasks filled with atrazine spiked water were used to replicate environmental conditions. Distilled water was also used to observe any impacts of the site water on uptake. Roughly 2 grams of XAD-7 resin (dry weight) was put into an 8 cm long section of membrane and filled with distilled water to create the resin packets. All air bubbles were removed before the packets were sealed. Each packet of resin was sealed on both ends with Spectra/Por membrane clips.

Water samples were spiked with Atrazine 480 commercial herbicide, which is the actual atrazine mixture used on farm fields within the AHHS watershed. Concentrations of 1, 10 and 100 ppb were examined to determine if differences in concentration affected uptake efficiency. Since pH is the main factor dictating uptake, samples were kept at a pH of 5 and a pH of 7 to replicate both the lower and upper limit of pH expected in the watercourses. Hydrochloric acid and sodium hydroxide were used to control the pH on a daily basis. Conductivity was also measured daily to ensure that solutions of the same pH had similar conductivity potentials. In order to better simulate environmental

conditions of flow and temperature, samples were put on a shake table rotating at 80 rpm while under a constant temperature of 10^0C for 7 days.

3.4.4 Field Experiment

Downstream sampling sites of each of the five major tributaries in the watershed and the control were chosen for placement of the resin packets. Besides the control site, resin locations were at H1 (Hogg Slough), F2 (Westlin Ditch), C1 (Miami Creek), K2 (Agassiz Slough), and D1 (McCallum Slough) (Figure 3.1). Two resin packets were put at site D1 to determine the instream variability of the technique. Each packet consisted of three 2 g packets of XAD-7 resin separated by knots in the membrane, with both ends sealed by membrane clips to obtain results in triplicate. Resin packets were then suspended within a 7 cm diameter by 30 cm long perforated plastic pipe for protection. The pipe was tied down inside a concrete block to ensure that the device would stay stationary during periods of high flow. Photos of the resin apparatus setup are shown in Appendix A. Within each watercourse, pipes were placed parallel to the flow of water in areas of higher flow.

Resins were left at each station for a total of 28 days between May 10, 2002 and June 7, 2002. Resin apparatuses were inspected on a weekly basis to ensure that periphyte growth on the membrane did not interfere with water flow through the pipe and across the membranes.

3.4.5 Resin Analysis

Both field and lab scale experiment resins were analyzed using the same technique. After collection, membrane pouches were cut open and the contents were put into a 50 ml test tube with a Teflon lined screw cap. An elution solution of 10% diethyl ether in n-hexane has been shown to give the best recoveries for atrazine (Dressler, 1979). Ten ml of the elution solution was added to the test tube, and the contents were shaken by hand for approximately 30 seconds. The test tube was then centrifuged at 3000 rpm for 5 minutes to settle out the XAD resin. The top layer (elution solution) was poured into a 25 ml test tube and the extraction procedure was then repeated. Two grams of anhydrous sodium sulphate was then added to the elution solution and shaken to ensure the absorption of

water from the solvent. An additional 0.5 ml of iso-octane was also added to discourage the volatilization of the sample.

The elution solution was put into a 50 ml round bottom flask and concentrated using a Buchi Rotavapor R-110 rotary evaporator connected to a vacuum. The volume was reduced to 2-5 ml. Samples were transferred to a 10 ml test tube, and nitrogen gas was used to reduce the volume to less than 1 ml. The final solution was put into a GC analysis tube, with the final volume brought up to 1 ml. Analysis was completed on a Hewlett Packard 6890 Series GCMS using a helium carrier gas at a flow rate of 38 cm/sec. The column (#RTX-5M5) measured 30 x 0.25 metres and had a film thickness of 0.25 μm. A splitless injection was used with an injection port temperature of 250^{0}C. An EPA 505/525 Pesticide Mix solution in acetone supplied by Supelco was used for the atrazine standards to create a calibration curve and to determine recoveries. Positive identification of atrazine required the presence of peaks at a molecular weight of 200 and 215. The technique gave an overall detection limit of 0.01 μg/L, with recoveries of 73%.

3.5 Moss Bioassay

3.5.1 Moss Collection and Field Conditions

Moss samples were collected from Cutthroat Creek in the Endowment Lands. Care was taken to ensure that collected specimens were submerged and collected from the deeper parts of the stream with a stronger current as recommended by Cenci (2000). Moss samples were stored in 20 L pails filled with the stream water during transport and stored in a refrigerator at 4^{0}C under fluorescent light.

Stems were cut to a length of 2 cm before being put into a heat sealed mesh bag. Mesh bags were made from mesh screen material (charcoal grey), and measured approximately 20 cm x 20 cm. Each bag contained 10 stems. Twenty one sampling stations were chosen for placement of the moss samples. These locations each maintained an adequate flow required to conduct the bioassay study. Bags were tied to a wooden stake by fishing line (monofilament) which was then hammered into place. Photos of the apparatus setup are shown in Appendix A. Sites at each location were chosen based on maximum exposure to light, and a sufficient distance from grasses or algae which may interfere

with light penetration. Replicates were placed at sites D1, H2, and the outfall station. Mesh screens were checked on a weekly basis in order to ensure that the bags had maintained proper positioning in the water and to clean away any algae and periphyte growth accumulated on the mesh.

Bags were left at sampling stations for 28 days between May 10, 2002 and June 7, 2002. After collection, bags were stored in site water until analysis could be conducted. The following day all stem lengths were measured, cut to 1 cm for chlorophyll analysis, stored in 10 ml vials and frozen.

3.5.2 Shoot Lengths and Chlorophyll Measurements

After one week, stems were thawed and 5 ml of dimethyl sulphoxide (DMSO) was added to each vial. After sealing, each vial was put in a 60^0C oven for 12 hours. Absorbance was measured using a UV300 Spectronic/Unicam Spectrophotometer at wavelengths 648.2 and 664.9. In order to determine dry weight, shoot tips were allowed to dry in an oven at 80^0C for 24 hours, and then weighed. Equation 1 and 2 (below) were used to calculate the chlorophyll *a* and chlorophyll *b* levels, respectively.

Equation 2: Equation for the calculation of chlorophyll a in *Fontinalis antipyretica*

$$Chla = (14.85A^{664.9} - 5.14A^{648.2})x\frac{V}{DW}$$

Equation 3: Equation for the calculation of chlorophyll b in *Fontinalis antipyretica*

$$Chlb = (25.48A^{648.2} - 7.36A^{664.9})x\frac{V}{DW}$$

Where:
Chla – Chlorophyll *a* Concentration (mg chl/mg moss)
Chlb – Chlorophyll *b* Concentration (mg chl/mg moss)
$A^{648.2}$ – Absorptive measurement at 648.2 nm
$A^{664.9}$ – Absorptive measurement at 664.9 nm
V – Volume of dimethyl sulphoxide (DMSO) solvent (ml)
DW – Dry weight of 1 cm moss tips (mg)

3.6 Quality Analysis and Quality Control

During each sampling period, two sets of replicates were taken on each sampling day during the course of the experiment in order to determine the sampling variability. The results of this replicate analysis are presented in Section 4.1.1. Raw data is given in Appendix I. Replicates were taken at different sampling sites during each day to ensure an examination of variability in all the watercourses. The coefficient of standard deviation was calculated for each sampling day, and the average was taken to represent the variability. The complete average of the variability was then calculated.

The reference sediment MESS-2 was used to determine the accuracy and precision of the metal digestion technique. Results are given in Section 4.1.2.

3.7 GIS Methodology

Land uses within the AHHS watershed were obtained from a variety of different sources. The primary source of information was from a "windshield" survey in which land use types were visually inspected from a vehicle and drawn onto a mylar overlay of a colour orthophoto. These orthophoto maps were obtained from the Town of Agassiz Municipal Office. Watercourse routes were determined from the examination of colour orthophotos and from 1:20,000 TRIM maps (1992) which show the route of watercourses.

Land use and watercourse routes were then traced over the colour orthophotos in GIS Arcview, along with sampling station locations and significant features such as highway routes and major streets. Through the use of the spatial analysis tool in Arcview, it was possible to calculate the areas of the various land uses within the watershed.

3.7.1 Buffer Area Delineation

The relatively flat topography of the AHHS lowlands is generally not conducive to runoff traveling long distances. As a result, it was assumed that a 100 m buffer was adequate to observe any water quality/land use correlations. Buffers were restricted to watercourses sampled during the course of the study. Smaller watercourses were not included due to their seasonal nature and relatively lower flows. Land uses within the buffer were calculated for individual watercourses and between sampling points. All lands within a

100 m buffer upstream from the sampling point, including tributaries, were including in the total land use type calculations.

Sampling stations along Agassiz Slough were not included in the land use correlations due to the presence of stormwater outfalls. These outfalls have a significant impact on water quality that is not related to surrounding land uses.

3.7.2 Contributing Area Delineation

As with the buffers, sampling station results were correlated with all land uses upstream from the sampling point. Contributing areas were estimated taking into account topographical features such as major roads and highways where ditches can impede and redirect runoff from adjacent land. Due to the relatively flat topography of the watershed, and the lack of detailed elevation maps, midpoints between watercourses were the main factor used in determining contributing areas. As with the buffer calculations, Agassiz Slough was not included in the calculations since the water quality is not representative of surrounding land uses.

3.8 Statistical Analysis

Box and Whisker plots were created using *SPSS for windows (Release 11.0).* The Shapiro-Wilks test was applied to confirm the non-parametric distribution of data for each season before median concentrations were calculated. A description of the box and whisker plots is described in Figure 3.2. The "box" area, known as the interquartile range (IQR) represents the range of 50% of the data values obtained for the variable being examined. The location of the median within the box can be used as an indication of the data skewness. The "whisker" area of the plots represent data points between the IQR, and 1.5 times the IQR. Values which are between 1.5 and 3 times the IQR range are labelled with a small circle, and are deemed as outlier values. Extreme values, marked with a star, are those values which are outside the outlier range.

Correlations were used to determine the relationships between water quality and land uses. Statistically significant positive correlations between a specific land use and a variable indicated that as the area of a certain land use increased, the water or sediment

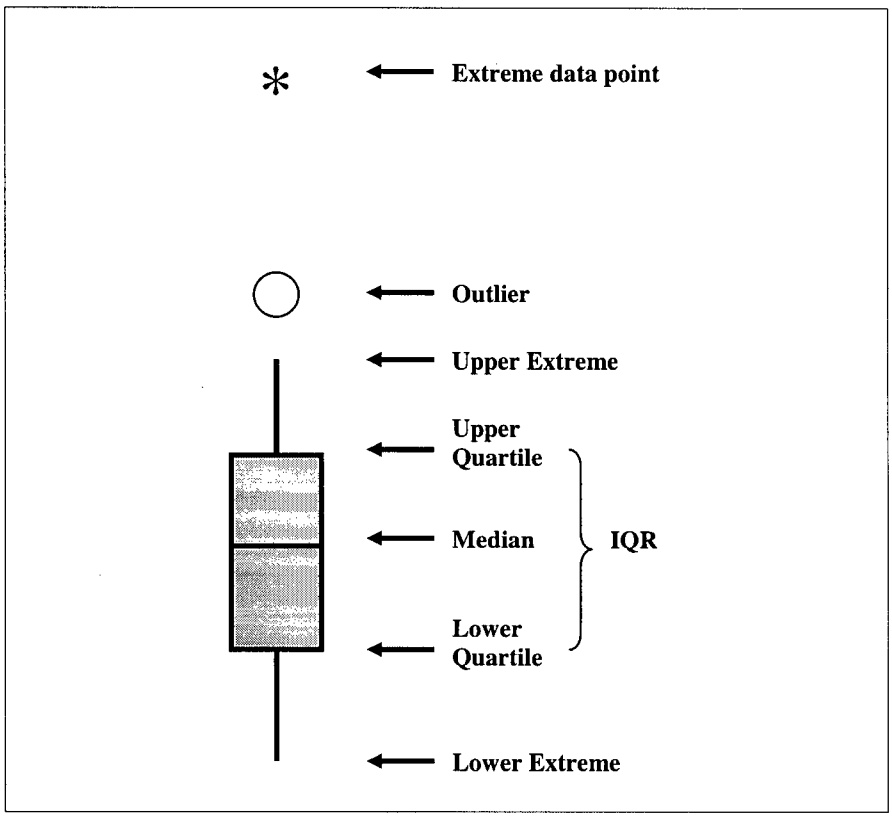

Figure 3.2: Box and whisker plot characteristics

quality variable also increased. Similarly, a negative correlation indicated that an increase in a specific land use lead to a decrease in the water or sediment quality variable.

Spearman Rank Test correlations were conducted in order to determine correlations between land uses and the various water and sediment quality medians for both seasons. The null hypothesis was that there was no relationship between the water quality indicators. A one-tailed test was used since it was assumed that there was a relationship between water quality and land uses. Correlations were deemed significant at $\alpha = 0.05$ for a one-tailed test.

A one-way analysis of variance test was applied to the bioassay results to determine if significant differences were present between the various sampling stations.

4. Results

Results are broken down into the wet season and dry season since wet seasons typically produce more runoff. Based on the monthly precipitations recorded during the study (Figure 1.5), it is evident that the wet season lies between the months of October and March, while the dry season is between the months of April and September.

Box and whisker plot data are grouped by individual watercourses. Sampling stations within each grouping are also arranged from upstream (left) to downstream (right) in order to easily identify spatial trends within the data. Raw data for the results are located in Appendix B and C.

4.1 Accuracy and Precision

4.1.1 Method Precision for Nutrient Analysis

Method precision for orthophosphate-P, nitrate-N, nitrite-N and ammonia-N was determined by calculating the percent difference between random duplicates. Duplicates were analyzed during each sampling run, and the average percent difference of all duplicates was used as a measure of method precision. Table 4.1 below summarizes the results.

Table 4.1: Method precision for analyzed nutrients.

Parameter	Average % Difference	% Difference Standard Deviation
Orthophosphate-P	6.3	9.8
Nitrate-N	0.2	0.3
Nitrtie-N	4.3	5.6
Ammonia-N	16.3	23.0

From these results, it is apparent that ammonia-N has the largest percent difference for method precision, however results may have been skewed due to the presence of an unusually high duplicate on May 3, 2002 which gave a 100% difference (Appendix I).

4.1.2 Method Precision and Accuracy for Metal Analysis

Duplicate samples were also used in the metal analysis of sediments to determine method precision. Duplicates were analyzed in both the wet season and dry season for each metal in order to calculate the percentage difference. Results are presented below in Table 4.2.

Table 4.2: Precision of analyzed metals for wet and dry season.

Metal	Dry Season (June, 2001)		Wet Season (February, 2002)	
	Average % Difference	Average % Difference Standard Dev.	Average % Difference	Average % Difference Standard Dev.
Copper	3.1	2.9	6.2	8.7
Lead	5.7	2.4	4.9	2.4
Zinc	3.0	1.3	2.3	2.1
Manganese	5.1	1.2	5.2	6.6
Iron	9.1	1.9	5.6	4.3

Accuracy of the metal analytical technique involved the analysis of MESS-2 reference sediment. MESS-2 sediment analysis was conducted in triplicate and average concentrations were calculated. Results, given in Table 4.3, show that very poor recoveries were obtained for lead. In addition, recoveries for manganese and iron were outside of the 5% error range. However, recoveries match results obtained in previous studies analyzing metals in sediments (Cook, 1994 and McCallum, 1995).

Table 4.3: Accuracy of analyzed metals using MESS-2 reference sediment.

	MESS-2 Certified Concentration (mg/kg)	Average Analytical Concentration (mg/kg)	% Error Outside Lowest Certified Concentration
Copper	39.3 ± 2.0	35.5	4.8
Lead	21.9 ± 1.1	7.6	63.3
Zinc	172.0 ± 8.6	165.7	0.0
Manganese	365.0 ± 19.2	263.7	23.7
Iron	62200.0 ± 3110.0	41368.8	30.0

4.1.3 Site Variability for Water Quality Parameters

Variability of sites was measured by taking triplicate samples at random sites during the course of sampling. Site variability was measured by calculating the coefficient of variance for each replicate set and then taking the overall average for each water quality variable. Variability was examined for orthophosphate-P, nitrate-N, nitrite-N, ammonia-N, pH, and chlorophyll concentrations. Results are listed below in Table 4.4.

Table 4.4: Average site variability for water quality parameters.

Water Quality Variable	Average Coefficient of Variance (%)	Average Coefficient of Variance Standard Deviation (%)
Orthophosphate-P	15.5	15.6
Nitrate-N	14.9	25.2
Nitrtie-N	15.5	12.3
Ammonia-N	27.7	43.0
pH	0.8	1.1
Chlorophyll	27.1	22.7

4.1.4 Site Variability for Sediment Quality Parameters

Metal concentrations in the sediments, in addition to % LOI and grain size distributions were examined for variability at specific sites. Results from both the wet and dry season are given in Table 4.5 and 4.6.

Table 4.5: Average site variability for analyzed metals.

Sediment Quality Variable	Dry Season Average Coefficient of Variance (%)	Wet Season Average Coefficient of Variance (%)
Copper	2.4	4.7
Lead	6.7	9.6
Zinc	6.3	4.3
Manganese	8.6	7.8
Iron	2.2	8.4
% LOI	0.7	11.1

Table 4.6: Average site variability for grain size fractions

Grain Size Fraction	Dry Season Average Coefficient of Variance (%)	Wet Season Average Coefficient of Variance (%)
> 1 mm	15.8	3.7
> 0.425 mm	46.0	7.0
> 0.250 mm	28.3	3.8
> 0.125 mm	7.7	2.7
> 0.063 mm	100.3	9.0
< 0.063 mm	101.4	14.4

It is apparent that there was very high variability in grain sizes less then 0.125 mm during the dry season. A review of the raw data (Appendix J) shows that this is largely due to the influence of one replicate which had significantly higher percentages of grain sizes < 0.125 mm.

4.2 Variations in DO

4.2.1 Seasonal Variations

DO levels were quite variable throughout the sampling period ranging from concentrations of almost 0 mg/L at the spring station to supersaturated levels of 17.4 mg/L at the control station. In general, slightly higher levels of DO were observed during the wet/winter season (Figure 4.1 and 4.2). This was expected since the colder temperatures allow for greater oxygen saturation. The winter season is also characterized by lower productivity, a lower rate of decomposition and increased flushing of organic matter from the system. As a result, there is less oxygen demand within the aquatic system. The expected decrease in photosynthetic activity did not appear to have an impact on overall DO levels.

4.2.2 Spatial Variability

The highest median DO concentration during the dry season was present at site H2, with a level of 10.44 mg/L. The high variability of this value is due in part to two extremely low values of 0.2 mg/L and 0.7 mg/L recorded on August 22, 2001 and September 18,

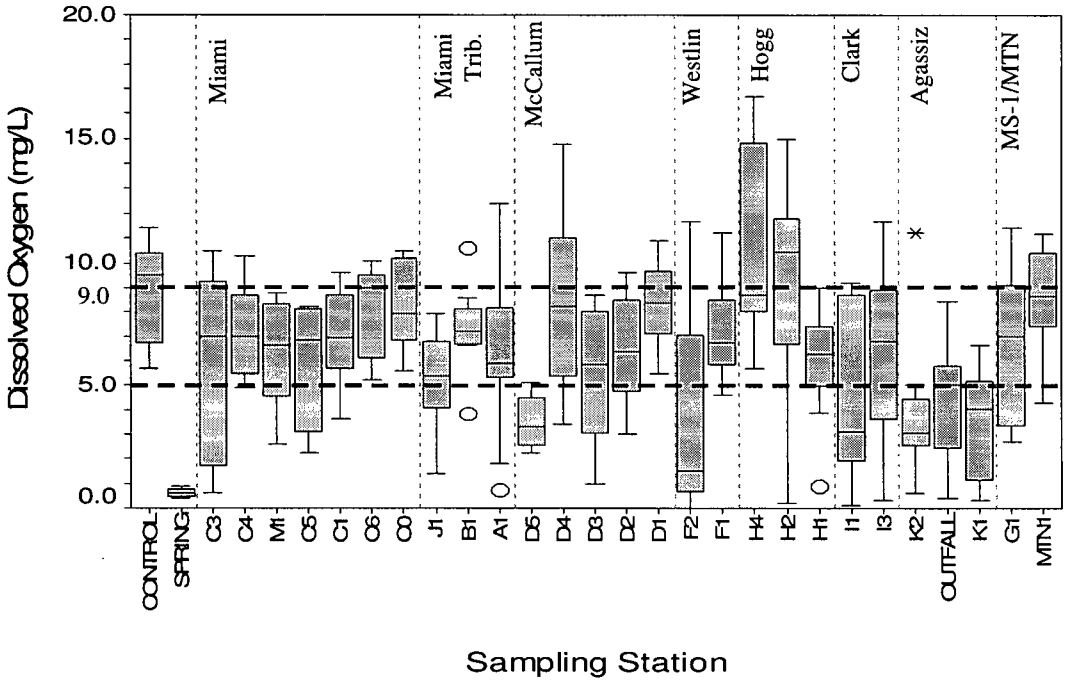

Figure 4.1: Box and whisker plot of dry season median dissolved oxygen concentrations at sampling stations throughout the AHHS Watershed. The BC Water Quality Guideline for DO is shown at 5.0 mg/L and 9.0 mg/L.

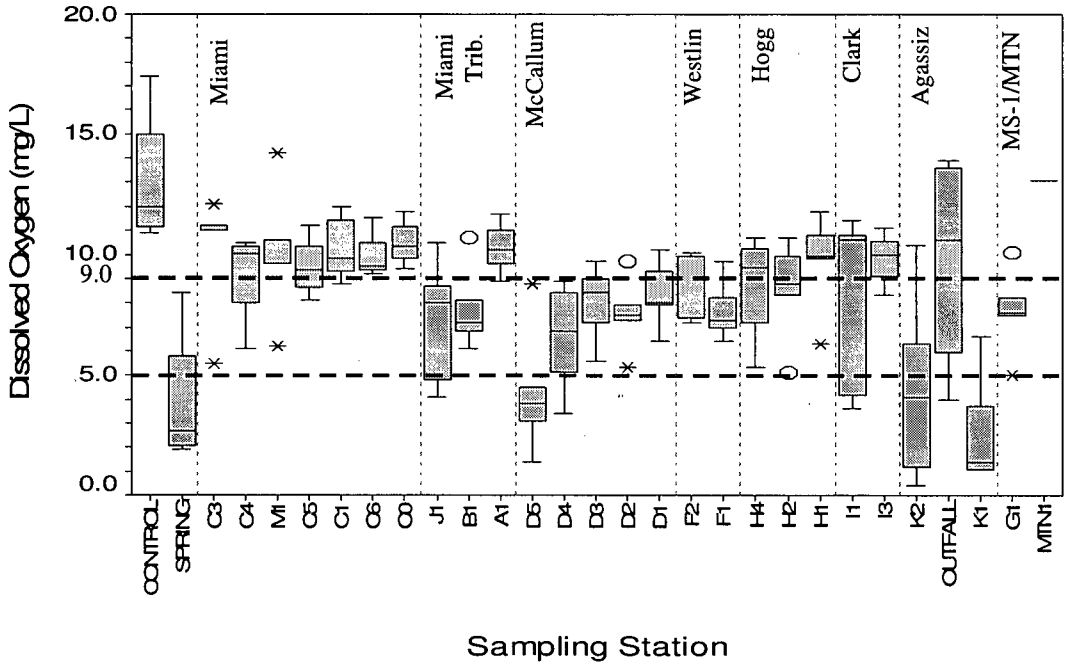

Figure 4.2: Box and whisker plot of wet season median dissolved oxygen concentrations at sampling stations throughout the AHHS Watershed. The BC Water Quality Guideline for DO is shown at 5.0 mg/L and 9.0 mg/L.

2001 (Appendix B). Sites C6 (Miami Creek) and H4 (Hogg Slough) also showed considerably high median concentrations of 9.00 mg/L and 8.70 mg/L respectively. In contrast, sites F2 and K2 had very low DO levels of 3.05 mg/L and 1.51 mg/L. Agassiz Slough showed overall low levels of DO compared to the other watercourses throughout the watershed. In addition, it was observed that DO levels in McCallum Slough were at their lowest at the uppermost station (D5). As expected, the spring station, which was used as a measure of groundwater characteristics, had the lowest median concentration of the watershed at 0.60 mg/L. The control had the highest median level of 9.48 mg/L (Figure 4.1).

During the wet season station C3 had the highest DO concentration at 11.20 mg/L. Similarly, a high median concentration of 10.60 mg/L was observed at M1, I1 and the outfall station. Surprisingly, station K1 had a median DO concentration of 1.35 mg/L, which was even lower then the spring sample station. Generally speaking, DO levels outside the outfall were significantly greater then concentrations at the other sampling points within Agassiz Slough, likely due to the entrainment of oxygen during turbulence as the water travels through the outfall system. As in the dry season, relatively low DO levels were recorded at station D5 in McCallum Slough, with concentrations slowly increasing downstream towards an equilibrium concentration. During both seasons, DO concentrations appear to increase significantly between stations D4 and D5 (Figure 4.2).

Compared to BC Water Quality Guidelines, it is obvious that the majority of the dry season results are above the 5.0 mg/L of DO recommended for adult and juvenile salmonids. However a large number of stations are below the 9.0 mg/L guideline for salomonid embryos. Only the control and station H2 had median DO concentrations above the 9.0 mg/L guideline. In the wet season, although more stations are above the DO guidelines, particularly in Miami Creek, there still appear to be a number of stations which were below minimum levels. In particular D5, K2, K1 and the spring station gave median values well below the 5.0 mg/L criteria.

4.3 Variations in Percent Saturation of DO

4.3.1 Seasonal Variations

As opposed to direct measurements of DO at sampling stations, the percent saturation of DO is a good indicator of the actual oxygen demand in the water. Percent saturation takes into account temperature effects to give a clearer picture of overall DO levels compared with saturation potential. Except for Hogg Slough, saturation levels are generally higher in the wet/winter season than the dry/summer season. Concentrations during the winter, particularly at the control station, often reached supersaturated levels (Figure 4.3 and 4.4). As mention earlier, increases in percent DO saturation are expected during the wet/winter season since increased flushing of organic matter from watercourses combined with lower rates of decomposition result in a lower overall oxygen demand.

4.3.2 Spatial Variability

Median values of DO saturation percentages were generally quite variable during the dry season, particularly in Hogg Slough. The highest median was observed at station H4 (Hogg Slough) which had a high median concentration of 147.4%. This unexpectedly high value is likely the result of an abundance of large aquatic plants observed at the site which cause an increase in dissolved oxygen during photosynthesis, emphasised during the dry/summer season when productivity is at its peak. The large drop in median concentrations downstream from H4 is likely due to a combination of oxygen demand and a return of the system to equilibrium concentrations. It is interesting to note the difference between the variability of station F1 and F2, which is likely due to the influence of groundwater on dissolved oxygen concentrations. Decreased DO levels at station F1 would not be reflective of ambient water temperatures, thus making results more variable. The increased outfall station variability indicates that there is an impact on percent DO saturation values. As mentioned earlier, high values are likely the result of oxygen entrainment due to turbulence. As expected, the spring station gave the lowest value at 2.0% saturation (Figure 4.3).

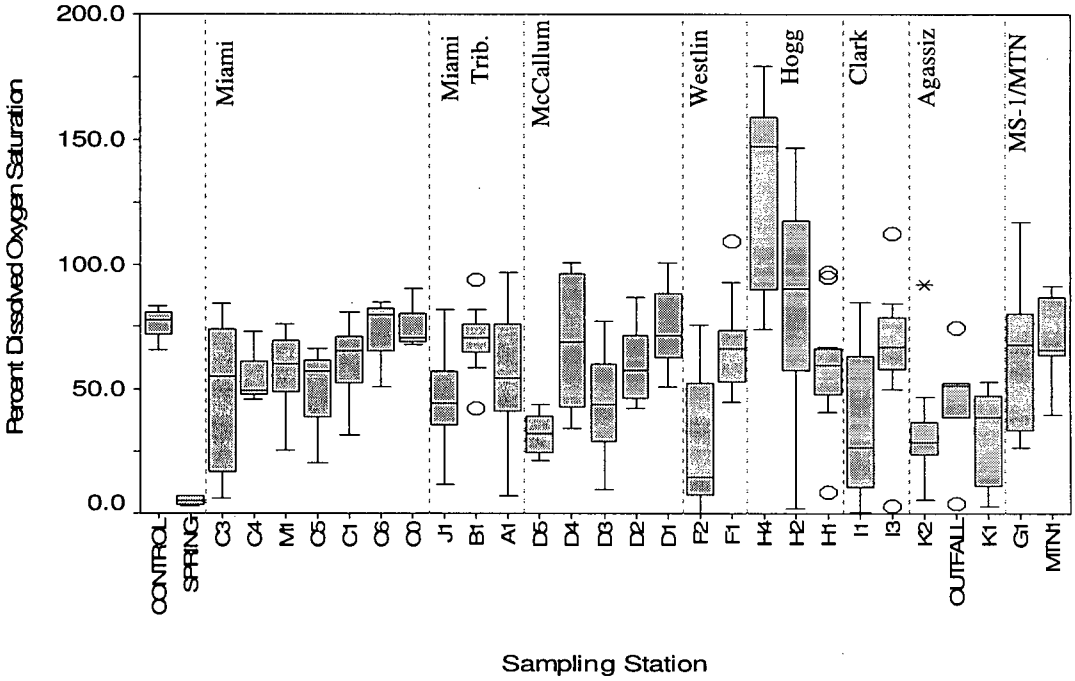

Figure 4.3: Box and whisker plot of dry season median percent dissolved oxygen saturation at sampling stations throughout the AHHS Watershed.

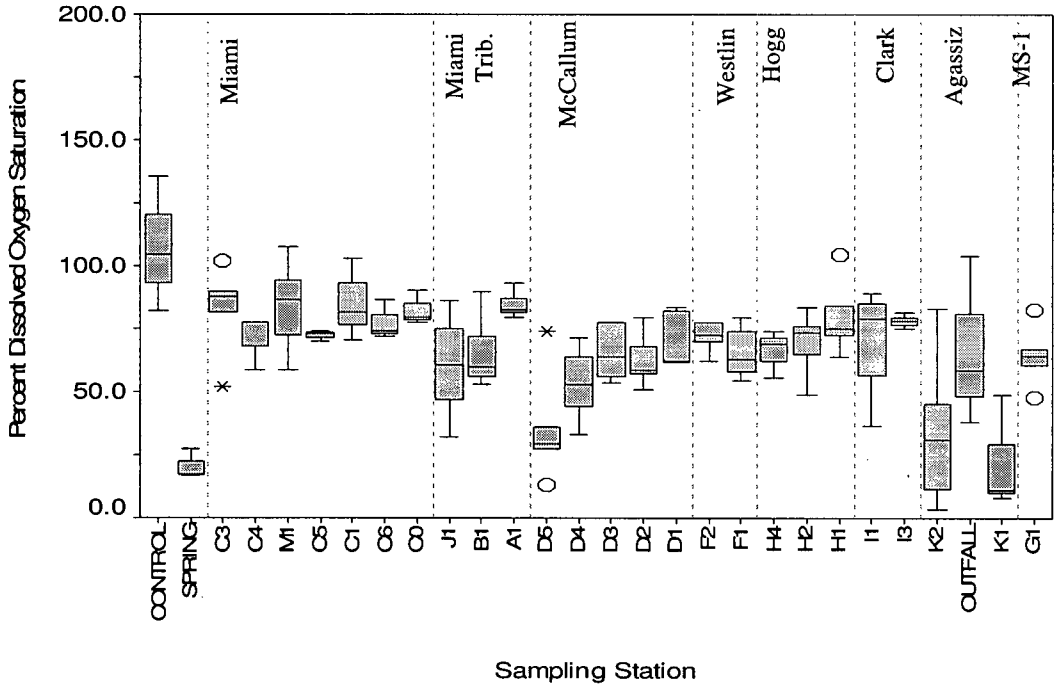

Figure 4.4: Box and whisker plot of wet season median percent dissolved oxygen saturation at sampling stations throughout the AHHS Watershed.

The wet season, in addition to giving higher dissolved oxygen saturation percentages, also gave more variable median values. It is interesting to note that the control station showed supersaturated levels with a median percentage of 104.8%. High percentages were also observed at station A1, which also gets a significant amount of runoff from the adjacent mountain slopes. McCallum Slough generally gave higher percentage DO saturation levels then other watercourses within the AHHS watershed. Compared to other stations within McCallum Slough, site D5 gave a much lower percent DO saturation value, indicating a possible area of oxygen deficiency. This matches higher nutrient concentrations observed at this site discussed further on in this section. Lower variability between the sampling stations in Hogg Slough during the wet season show that dry season variability was likely the result of photosynthetic activity. As in the dry season, the outfall station continued to show increased variability as a result of the stormwater outfall (Figure 4.4).

4.4 Temperature

Temperature results should be read with caution since readings were affected depending on the time of day which the sample was taken. For example, based on the order of the sampling program, those stations which were sampled in the morning were generally expected to have lower temperatures than those stations sampled later on in the day. However, temperature results do give an indication of possible influxes of groundwater. Generally speaking, temperatures from the spring station were higher than average temperatures in the winter and lower than average temperatures in the summer. By making this observation, it was obvious that there was groundwater infiltration between certain sampling points, particularly between sites F1 and F2 in Westlin Ditch. During the wet/winter season, the temperature increases on average 2.5^0C, from 5.9^0C to 8.4^0C between the two stations.

4.5 Variations in Specific Conductivity

4.5.1 Seasonal Variations

Specific conductivity, an estimate of dissolved solids within the watercourses, was generally higher in the dry/summer season then in the wet/winter season. In addition, the dry/summer season had much more variable specific conductivity values (Figure 4.5 and 4.6).

4.5.2 Spatial Variations

High variability in specific conductivity values were observed in Agassiz Slough in both the wet and dry seasons. Station K2 showed the highest dry season mean conductivity at 328 μs/cm. The outfall station had the highest wet season mean conductivity at 257.5 μs/cm. In McCallum Slough, station D5 appeared to have the highest median specific conductivity in both the wet and dry seasons. It was interesting to note the relative low specific conductivity of station D3, obvious in both the wet and dry seasons. This may indicate that there is some input from Mt. Agassiz tributaries before this sampling point. Other sites of interest include Westlin Ditch, where specific conductivity increased with progression downstream. Site F2 gave a specific conductivity of 99 μs/cm while F1 had a level of 206 μs/cm (Figure 4.5). The high conductivity of the spring shows that the changes in Westlin Ditch are likely due to groundwater input, discussed in more detail in Chapter 5.4.2. The same reasoning would also explain the high conductivity at station G1 during the dry and wet season (Figure 4.6).

Areas in which there was a significant proportion of water coming from the surrounding uplands, showed relatively low specific conductivities. For example, the control site had a specific conductivity ranging from 18.3 to 30.4 μs/cm. In addition, site A1 maintained a specific conductivity between 63.0 to 72.5 μs/cm. Since the granite rock which dominates much of the upland areas is not very soluble, water coming from these areas is not expected to have very high conductivity values (Figure 4.6).

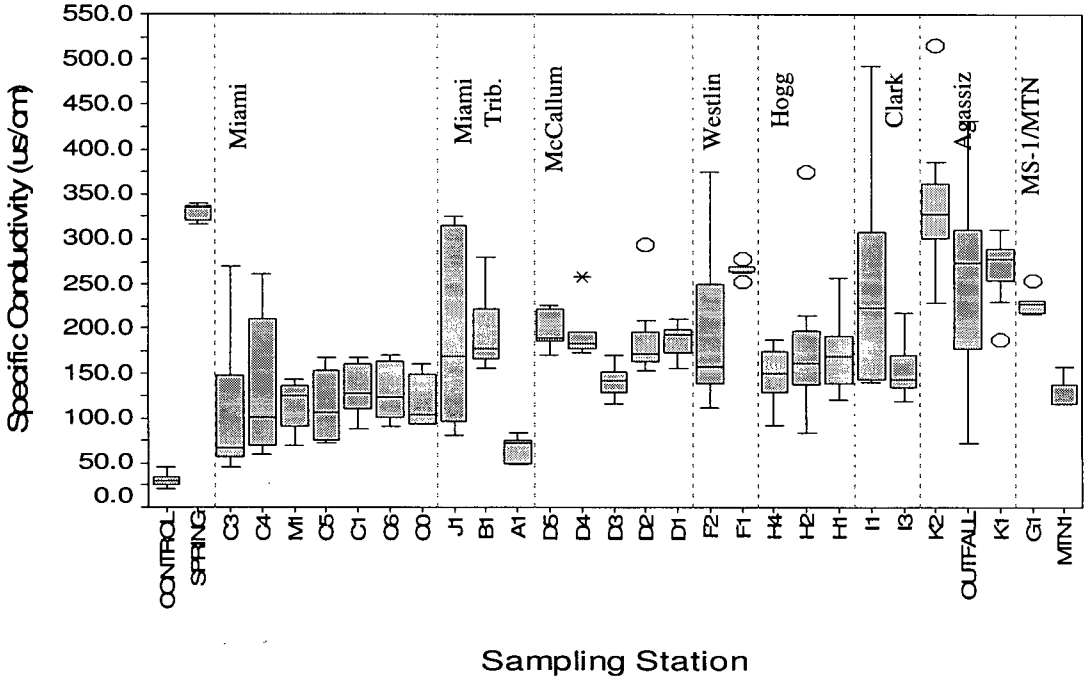

Figure 4.5: Box and whisker plot of dry season median specific conductivity levels at sampling stations throughout the AHHS watershed.

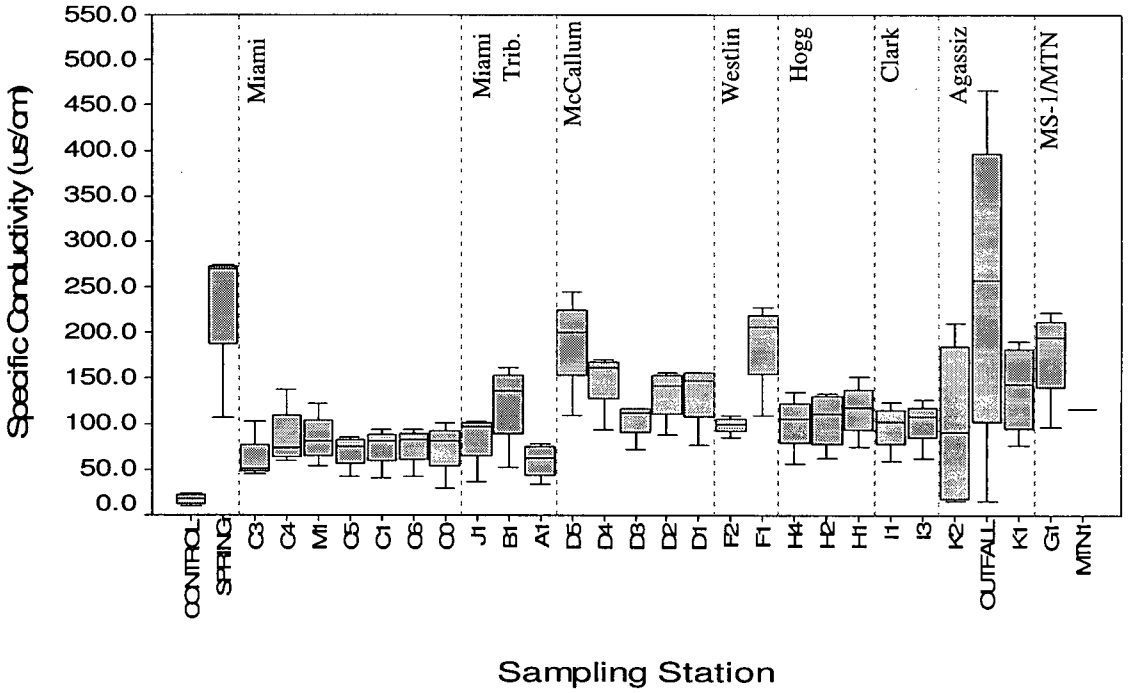

Figure 4.6: Box and whisker plot of wet season median specific conductivity levels at sampling stations throughout the AHHS watershed.

4.6 Variations in Turbidity

4.6.1 Seasonal Variations

Turbidity appeared to increase only slightly during the wet season, but was much more variable than during the dry season (Figure 4.7 and 4.8). Turbidity values were expected to be higher during the wet season, since roots and canopy cover during the dry/summer season provide better protection from erosion. Significant turbidity has been observed coming from cornfields during the wet/winter season when there was significant soil exposure. It should be noted that the results were dependent on the weather of the sampling days chosen.

4.6.2 Spatial Variations

The most significant increase in turbidity appears around the outfall sampling location. The highest turbidity reading recorded throughout the study was measured outside the outfall at 626 NTU's during the wet season. Sediment discharges, in the form of a large plume, have been observed outside the stormwater outfall pipe even during minor precipitation events (Appendix A). Site C3 has more variable turbidity readings during the dry season then any of the other sampling points within Miami Creek. This may indicate a localized turbidity source upstream from this location. Hogg Slough appears to show a general increase in turbidity as one moves further downstream during the wet season, indicating a constant influx of turbidity along the length of the watercourse due to runoff. High variability and turbidity values at station F2 during the dry season (24 NTU's) is likely due to presence of iron oxides and periphytic growth at the site. Similarly, high turbidity levels at station I1 during the dry season is more a reflection of duckweed and algae present due to the stagnated conditions at the sampling point.

Low turbidity values at station D5 are likely a reflection of the very slow flow rate in this area. The lowest turbidity values were measured at the control sampling station, where a lack of fine particulates in the mountain tributaries provide little opportunity for turbidity to enter into the water.

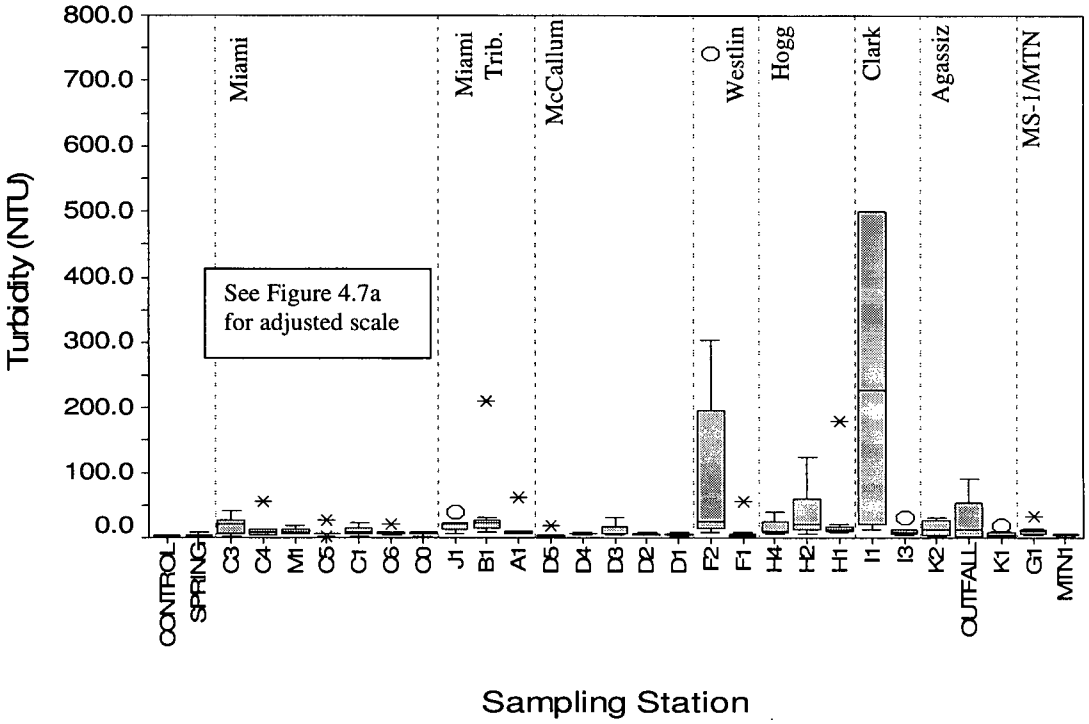

Figure 4.7: Box and whisker plot of dry season median turbidity levels at sampling stations throughout the AHHS watershed.

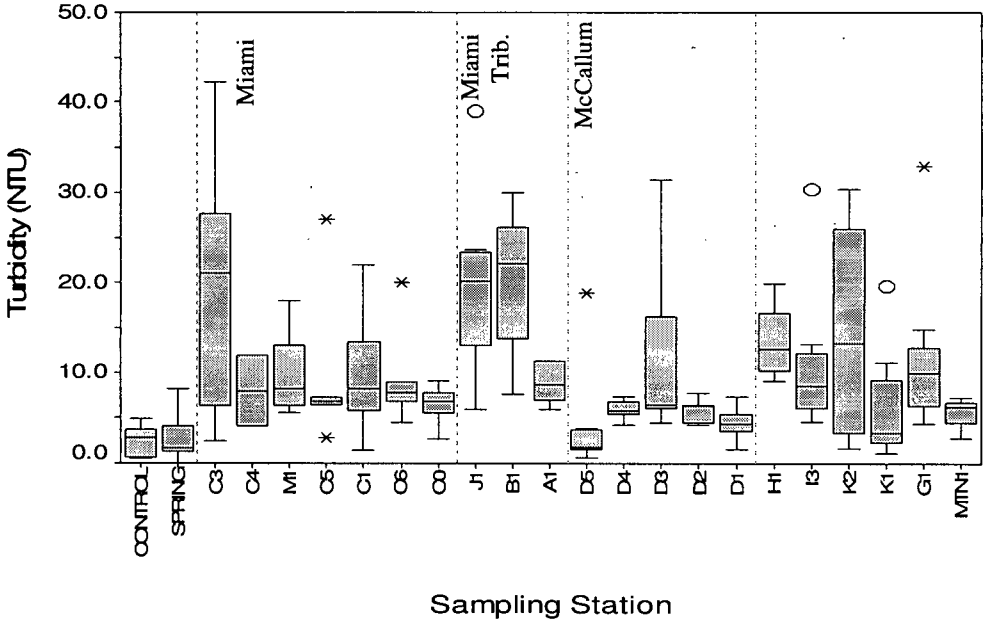

Figure 4.7a: Box and whisker plot of dry season median turbidity with adjust scale.

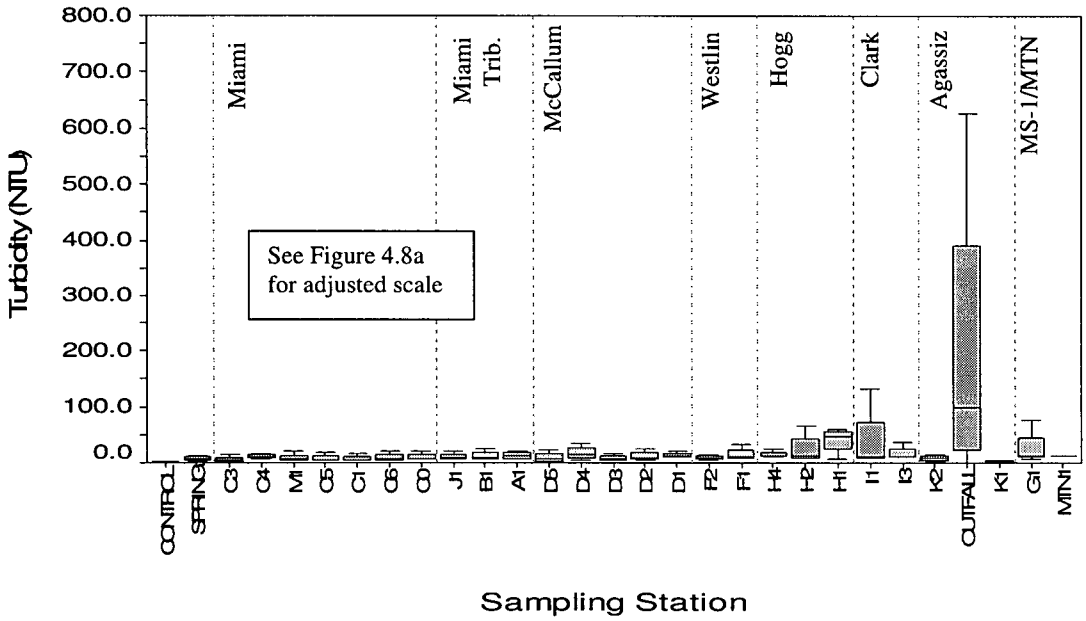

Figure 4.8: Box and whisker plot of wet season median turbidity levels at sampling stations throughout the AHHS watershed.

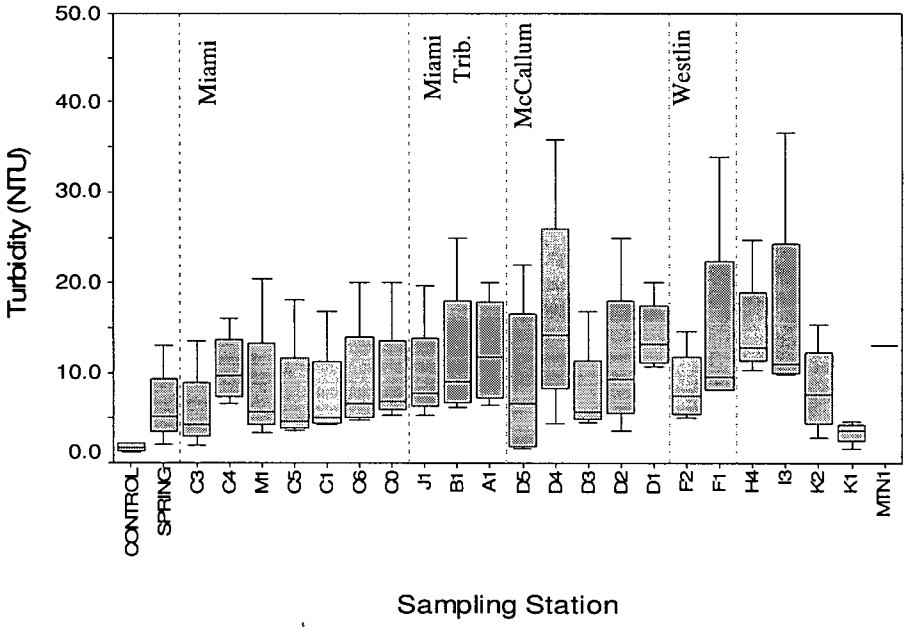

Figure 4.8a: Box and whisker plot of wet season median turbidity with adjusted scale.

4.7 Variations in pH

4.7.1 Seasonal Variations

The median pH values in the wet/winter season were noticeably lower then pH values in the dry/summer season. This was as expected since higher rates of photosynthesis by aquatic plants in the dry/summer period use CO_2, causing an increase in pH. Levels did not vary significantly between the various sampling sites and variability between the two seasons was generally similar (Figure 4.9 and 4.10).

4.7.2 Spatial Variations

Agassiz Slough had the highest pH levels during the dry season, with K1 and K2 having a pH of 6.8 and 6.7 respectively. More acidic dry season sampling sites included station J1 and the control with pH levels of 5.9 and 5.7 (Figure 4.9).

The highest pH levels were observed during the wet season at stations F1 and G1, with a median pH of 6.5 and 6.4 respectively, reflecting inputs of groundwater upstream from these stations. McCallum Slough had relatively high pH's ranging from 6.1 to 6.3. Within Hogg Slough, pH appeared to increase along the length of the watercourse from 5.7 at station H4 to 6.1 at H1. The most significant increase took place between stations H4 and H2. Station J1 also had a fairly low median pH of 5.5. The spring sampling station maintained a relatively high pH during the course of the experiment, indicating that the groundwater generally had a higher pH then the surface water. In contrast, the control station median pH was relatively lower compared to the other sampling stations. Flow from the control station into Miami Creek obviously results in a reduction in pH as can be observed at Station C5 during the dry and wet season (Figure 4.9 and 4.10).

4.8 Variations in Orthophosphate

4.8.1 Seasonal Variations

As can be observed in Figure 4.11 and 4.12, phosphorous levels between the seasons remained relatively constant throughout the watershed. Levels appeared to increase and decrease independent of sampling sites and watercourses. Except for the Agassiz stormwater outfall, sampling stations generally appeared to be subject to greater variability during the dry season.

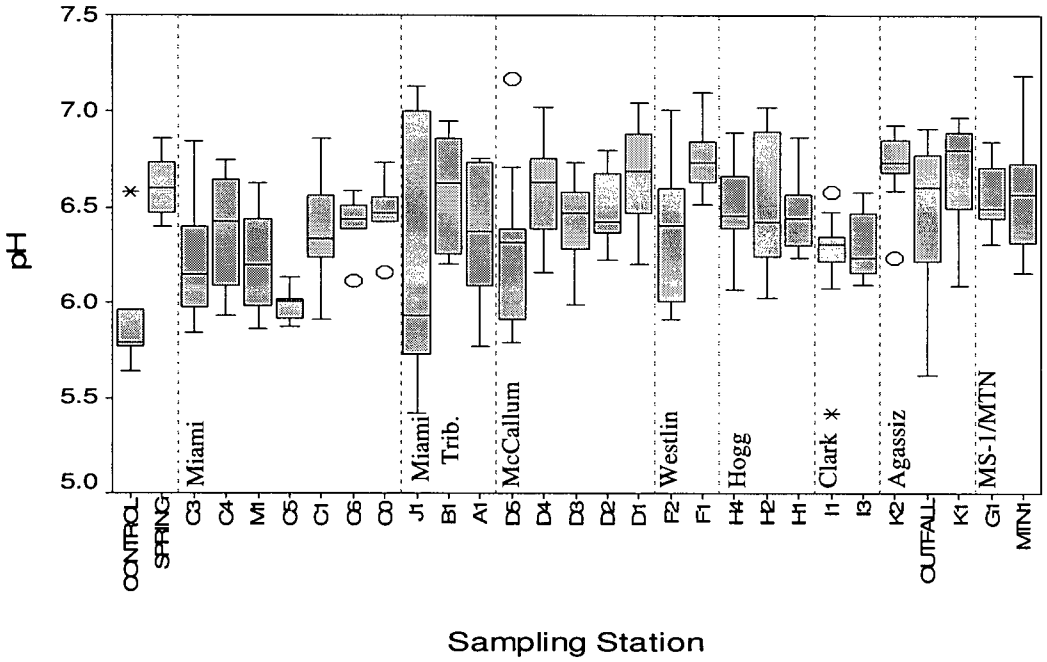

Figure 4.9: Box and whisker plot of dry season median pH at sampling stations throughout the AHHS watershed.

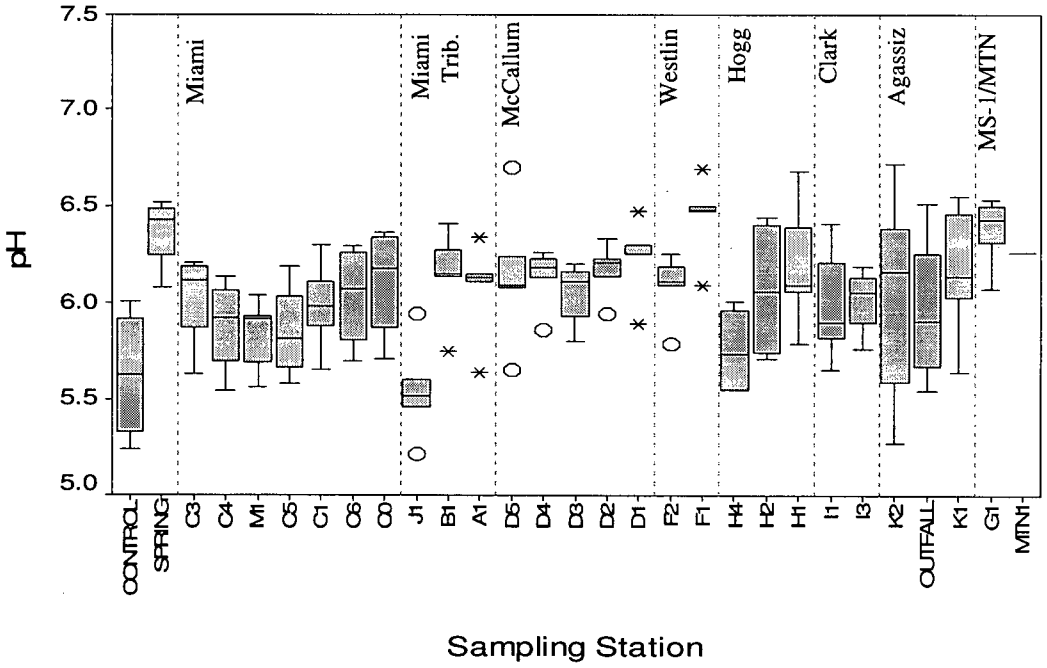

Figure 4.10: Box and whisker plot of wet season median pH at sampling stations throughout the AHHS watershed.

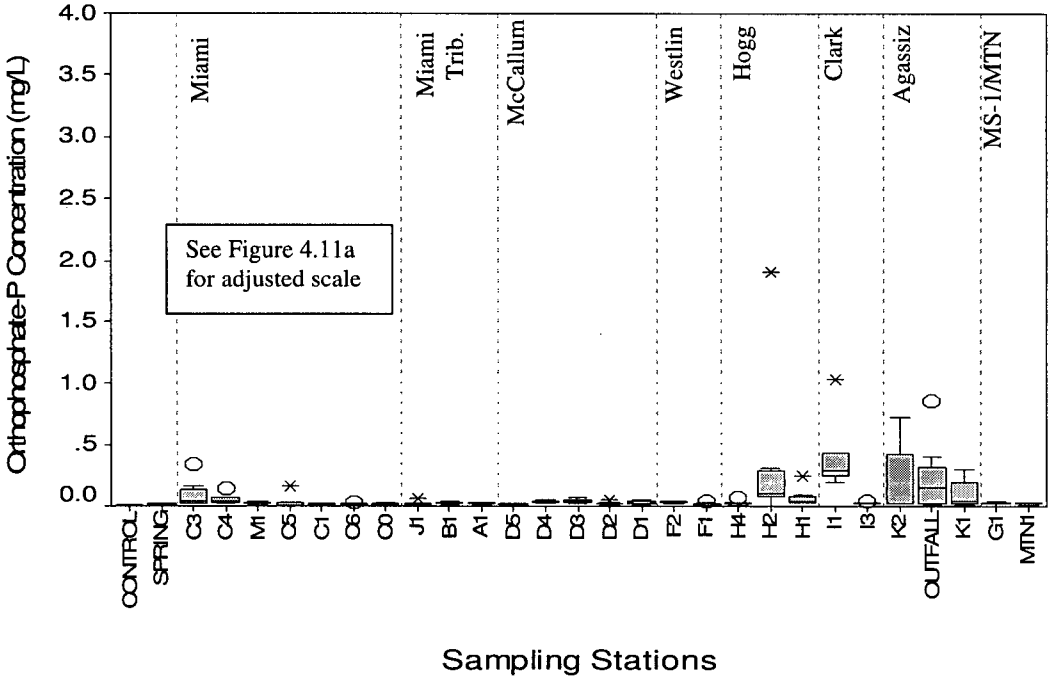

Figure 4.11: Box and whisker plot of dry season median orthophosphate concentrations (PO_4-P) at sampling stations throughout the AHHS watershed.

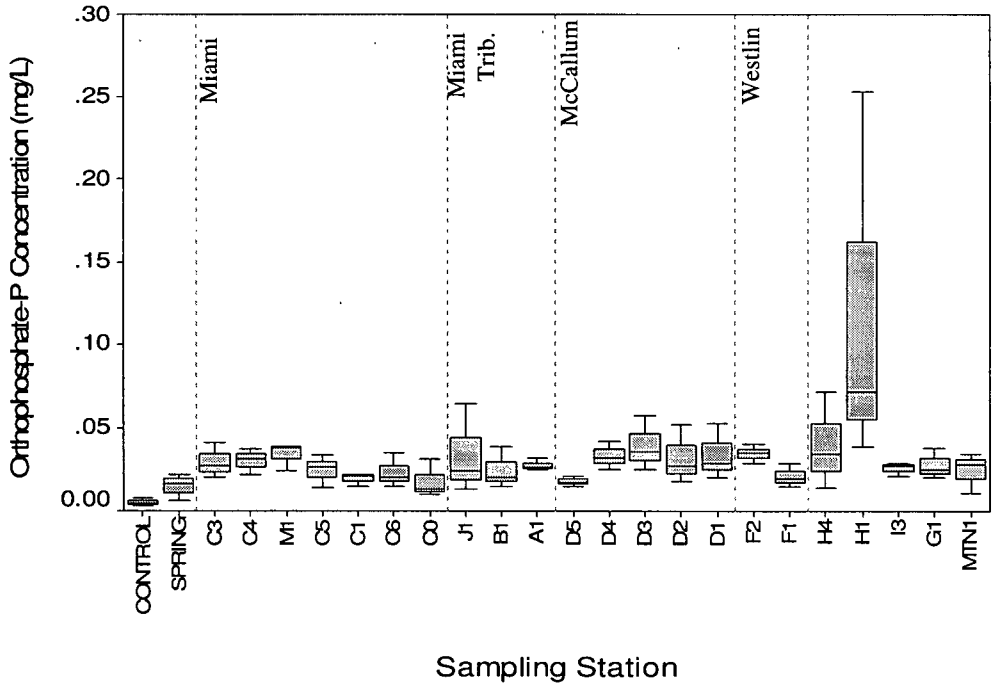

Figure 4.11a: Box and whisker plot of dry season median orthophosphate concentrations (PO_4-P) with adjusted scale.

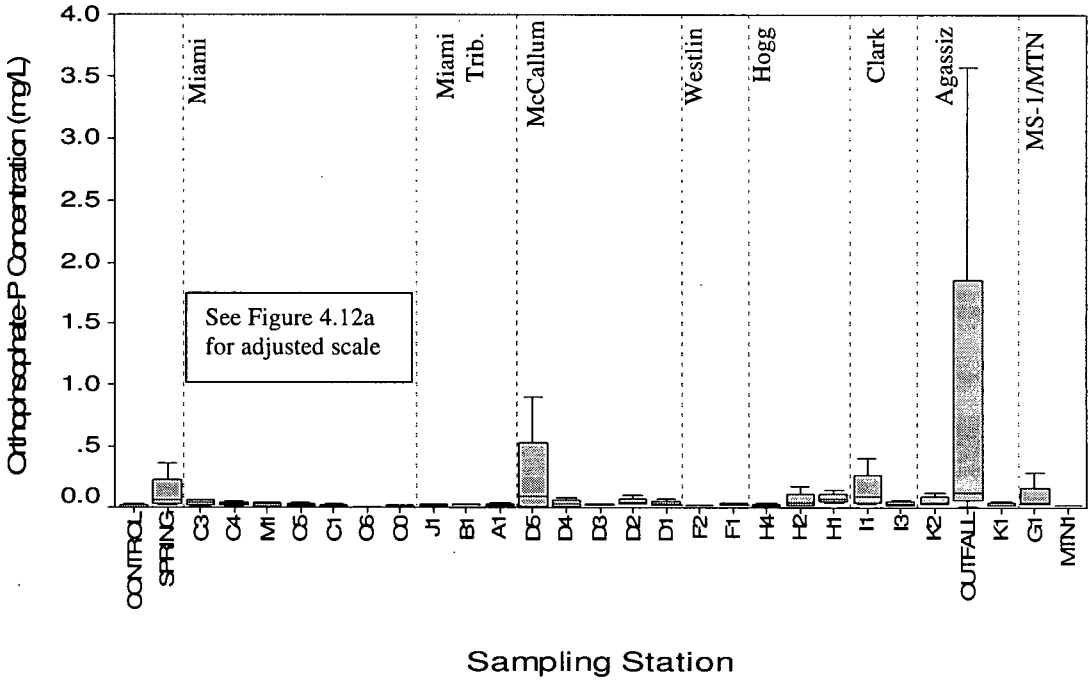

Figure 4.12: Box and whisker plot of wet season median orthophosphate concentrations (PO_4 - P) at sampling stations throughout the AHHS watershed.

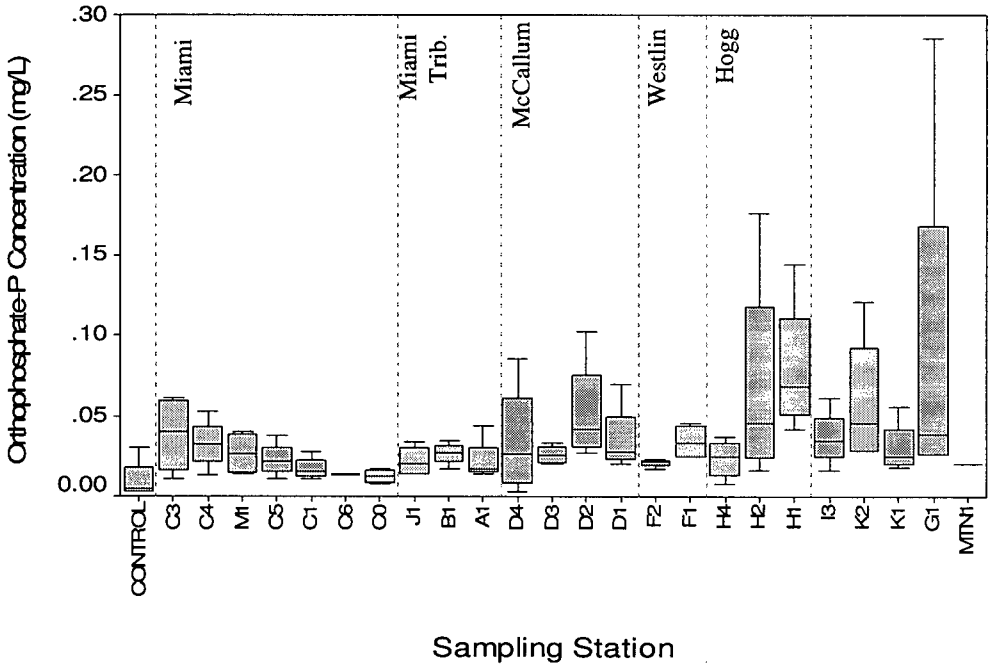

Figure 4.12a: Box and whisker plot of wet season median orthophosphate concentrations (PO_4 - P) with adjusted scale.

4.8.2 Temporal Variations

The highest orthophosphate concentrations appeared within Agassiz and Hogg Slough during the dry season. The stormwater outfall station gave a median concentration of 0.153 mg/L while downstream, station K1 had a concentration of 0.044 mg/L. Within Hogg Slough, site H2 showed the highest orthophosphate concentration at 0.103 mg/L. Spikes in orthophosphate concentrations also appeared at stations I1 and C3, with levels of 0.295 mg/L and 0.041 mg/L respectively. However, the stagnated environment at station I1 during the dry season means that orthophosphate concentrations were likely not the result of surrounding land uses. The spring and control station both gave the lowest dry season concentrations at 0.010 mg/L and 0.008 mg/L respectively (Figure 4.11).

High wet season concentrations were observed at sites D5 (McCallum Slough), G1 (MS-1 Ditch), the spring, and the outfall station. Of these, the highest median concentrations were observed at the outfall and D5 station with concentrations of 0.123 mg/L and 0.089 mg/L respectively. Station I1 showed relatively high concentrations at 0.088 mg/L. Hogg Slough also showed increased concentrations of orthophosphate at sites H2 and H1. It is obvious that there is an influx of phosphorous before station H2, which contributed to the high levels at site H1 during the wet season. It is also interesting to note that variability in orthophosphate decreases as one moves downstream from site C3 in Miami Creek. The control site showed the lowest wet weather orthophosphate concentration at 0.005 mg/L (Figure 4.12).

4.9 Variations in Nitrate

4.9.1 Seasonal Variations

Nitrate levels were generally higher in the wet/winter season than in the dry/summer season (Figure 4.13 and 4.14). Except for stations in Agassiz Slough (K1, K2 and outfall), variability among the sampling sites also appeared to increase during the wet season. It is assumed that the stormwater outfalls along Agassiz Slough contributed significant amounts of nitrate year round to Agassiz Slough, making it difficult to observe seasonal variances.

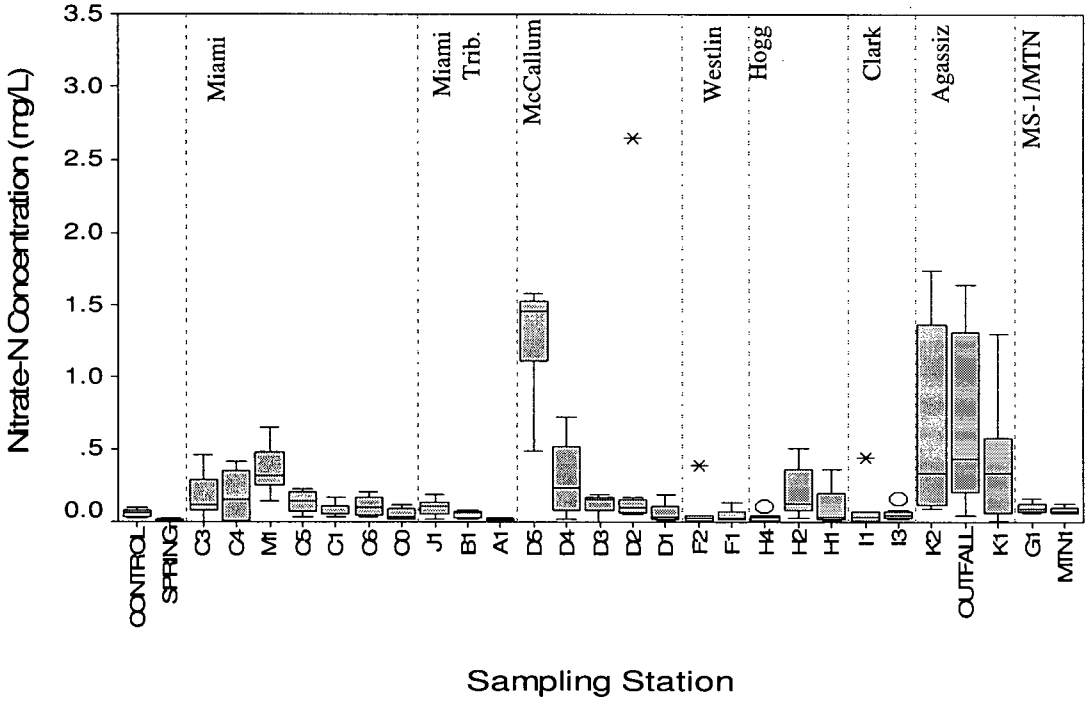

Figure 4.13: Box and whisker plot of dry season median nitrate concentrations (NO_3^-- N) at sampling stations throughout the AHHS watershed.

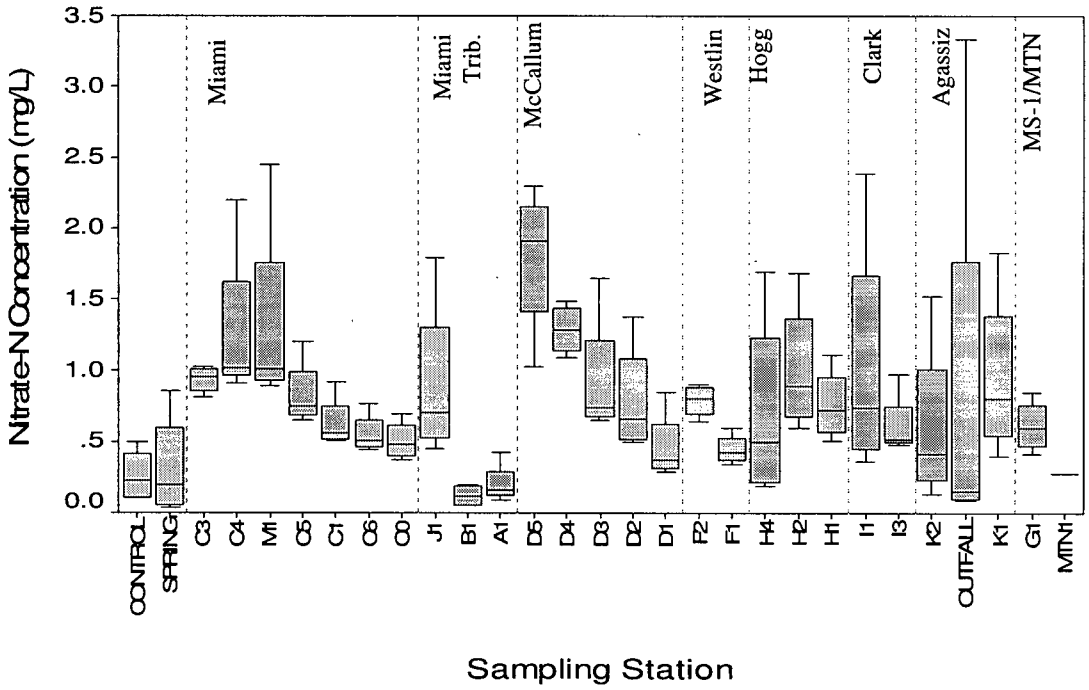

Figure 4.14: Box and whisker plot of wet season median nitrate concentrations (NO_3^-- N) at sampling stations throughout the AHHS watershed

4.9.2 Spatial Variations

Station D5 (McCallum Slough) appeared to maintain relatively high levels of nitrate during both the wet and dry season at 1.90 mg/L and 1.46 mg/L respectively. Concentrations then decreased moving downstream along McCallum Slough. During the dry season, Agassiz slough showed relatively high median concentrations with significant variability. The lowest nitrate concentration of the dry season was 0.017 mg/L, observed at the spring station (Figure 4.13)

In addition to station D5, stations C3, C4, and M1 all appeared to maintain high concentrations of nitrate during the wet season at 0.958 mg/L, 1.025 mg/L and 1.0143 mg/L respectively. In Hogg Slough, station H2 gave elevated concentrations of nitrate in both the dry and wet seasons compared to the adjacent stations H4 and H1. As expected, both the control and the spring showed relatively low levels of nitrate (Figure 4.14).

4.10 Variations in Nitrite

4.10.1 Seasonal Variations

Concentrations of nitrite appeared to be fairly constant between the wet and dry seasons, although levels were much more variable during the wet seasons, particularly along McCallum Slough (Figure 4.15 and 4.16).

4.10.2 Spatial Variations

During the dry season, site H2 gave high nitrite concentrations in both the dry and wet seasons at 19 μg/L and 26.5 μg/L respectively. The dry season median, however, was skewed due to the August 22, 2001 result where a concentration of 263 μg/L was recorded (Appendix B). The outfall station gave a median concentration of 20.0 μg/L during the dry season. Site F2 and I1 also gave notably high dry season median nitrite concentrations at 17 μg/L and 30 μg/L respectively (Figure 4.15).

The highest concentrations during the wet season were observed in Agassiz Slough, where the outfall station gave a median concentration of 38.5 μg/L. The control and spring station both had the lowest nitrite concentrations during the wet season of 1 μg/L and 7 μg/L respectively (Figure 4.16).

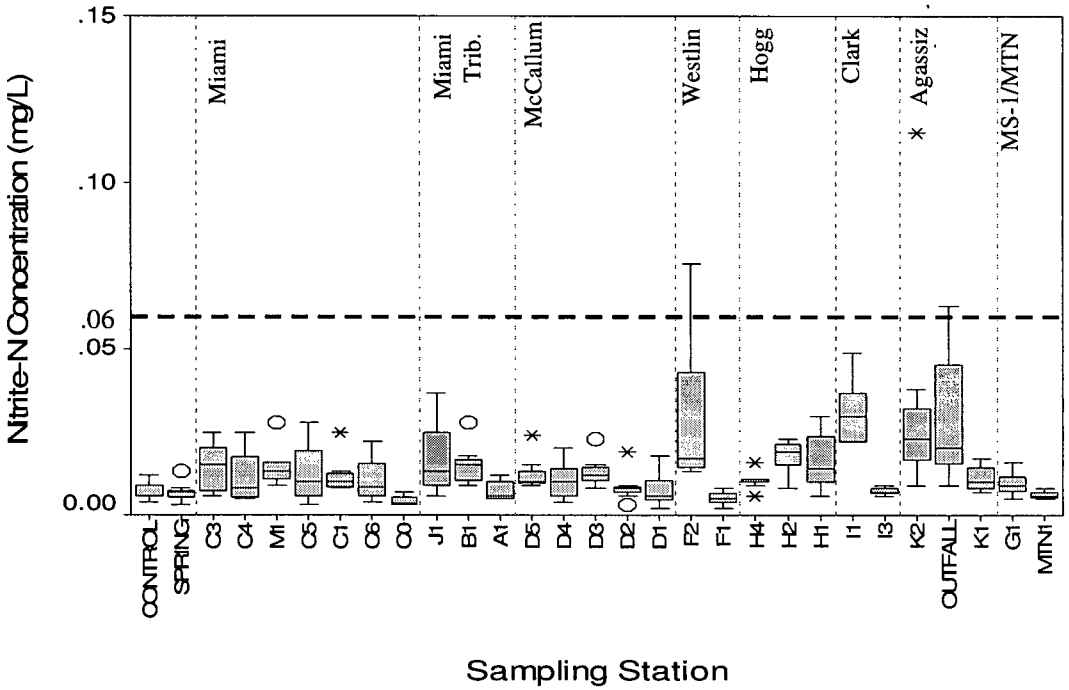

Figure 4.15: Box and whisker plot of dry season median nitrite concentrations (NO_2^-- N) at sampling stations throughout the AHHS watershed. The BC Water Quality Guideline for nitrite is shown at 0.06 mg/L.

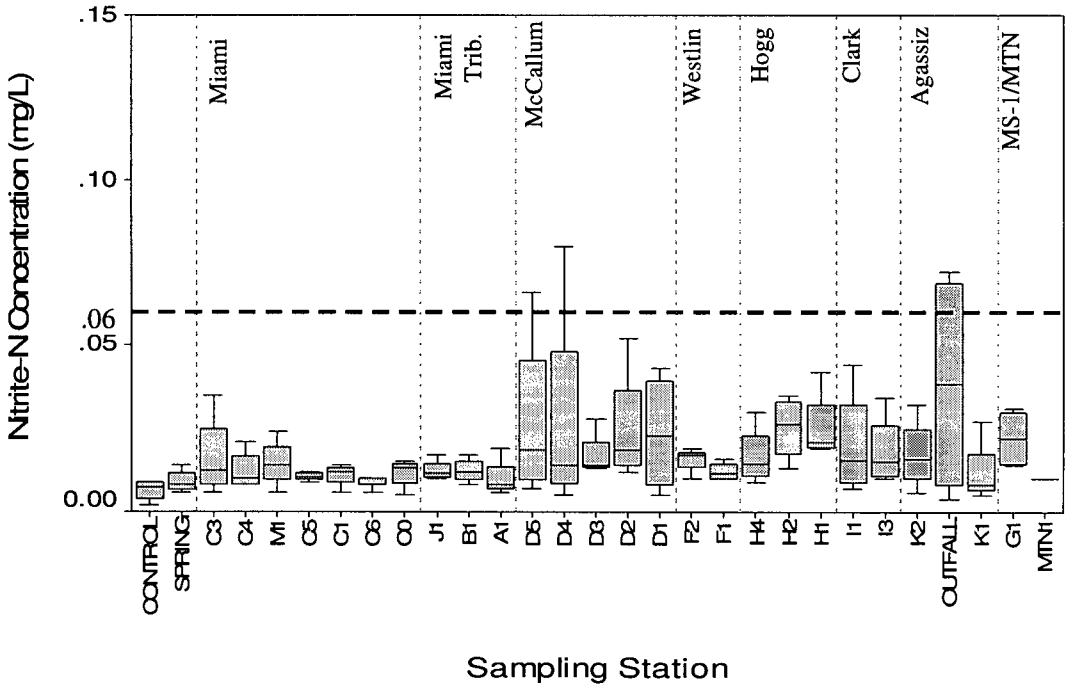

Figure 4.16: Box and whisker plot of wet season median nitrite concentrations (NO_2^-- N) at sampling stations throughout the AHHS watershed. The BC Water Quality Guideline for nitrite is shown at 0.06 mg/L.

When compared with BC Water Quality Criteria, it is clear that there are a few stations which had values above the 0.06 mg/L guideline. During the dry season, these stations were F2, the outfall, and an outlier value at station K2. During the wet season, stations D5, D4, and the outfall had values above the guideline, with the outfall showing the most sampling days above this value.

4.11 Variations in Ammonia

4.11.1 Seasonal Variations

Concentrations of ammonia appeared only slightly more variable in the wet season than in the dry season with more obvious changes in variability observed at sites D5 and D4 on McCallum Slough (Figure 4.17 and 4.18). On average, ammonia concentrations throughout the watershed were higher in the wet season then the dry.

4.11.2 Spatial Variations

Within the dry season, site H2 showed a high median ammonia concentration at 0.619 mg/L. It is interesting to note that the high concentration at H2 did not impact the downstream station of H1, which had a median ammonia concentration of only 0.045 mg/L. In Agassiz Slough, the K2 and outfall stations showed relatively elevated levels during the dry season with concentrations of 0.111 mg/L and 0.090 mg/L respectively. Both of these sites were quite variable in ammonia levels. Station I1 also had high levels of ammonia, however concentrations most likely reflected the stagnated nature of the site (Figure 4.17).

Similar to the dry season, the highest ammonia concentrations during the wet season appeared in Agassiz and Hogg Slough. The outfall site had the highest median ammonia concentration at 0.822 mg/L, although K2 also had a relatively high ammonia concentration of 0.436 mg/L. Within Hogg Slough, site H1 had the highest median concentration of 0.512 mg/L while H2 gave a median concentration of 0.251 mg/L. In McCallum Slough, site D5 also had a relatively high median concentration of 0.244 mg/L. Ammonia concentrations at the D5, H2 and outfall stations were all quite variable throughout the wet season. The control station gave the lowest median ammonia concentrations of 0.006 mg/L (Figure 4.18).

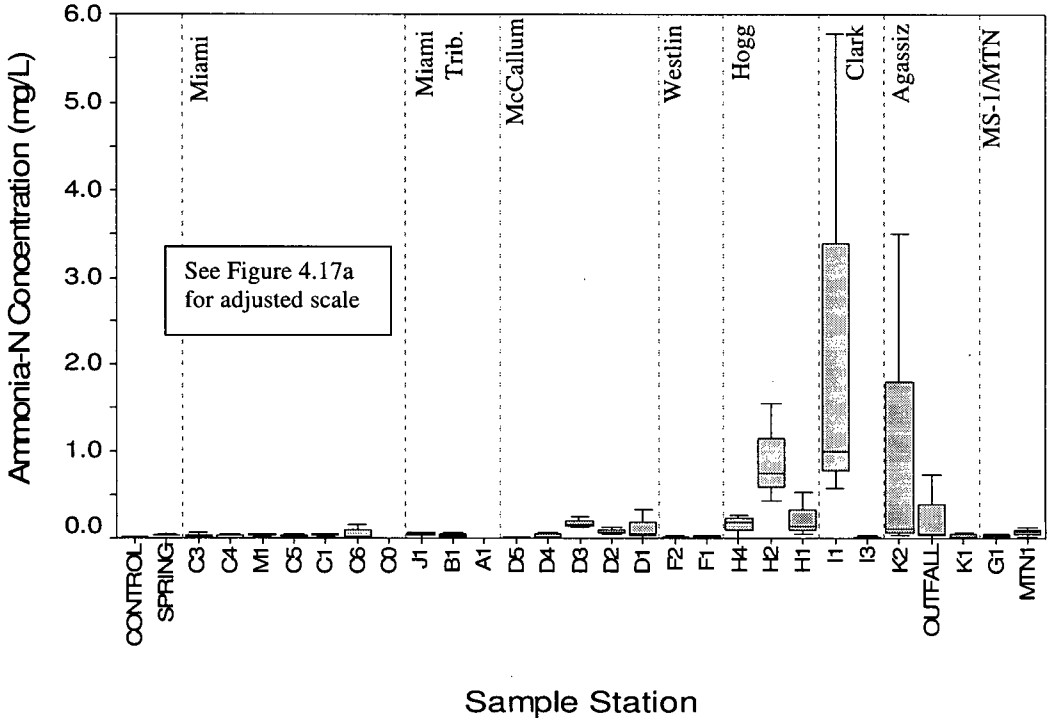

Figure 4.17: Box and whisker plot of dry season median combined ammonia concentrations (NH_3-N and NH_4^+-N) at sampling stations throughout the AHHS watershed.

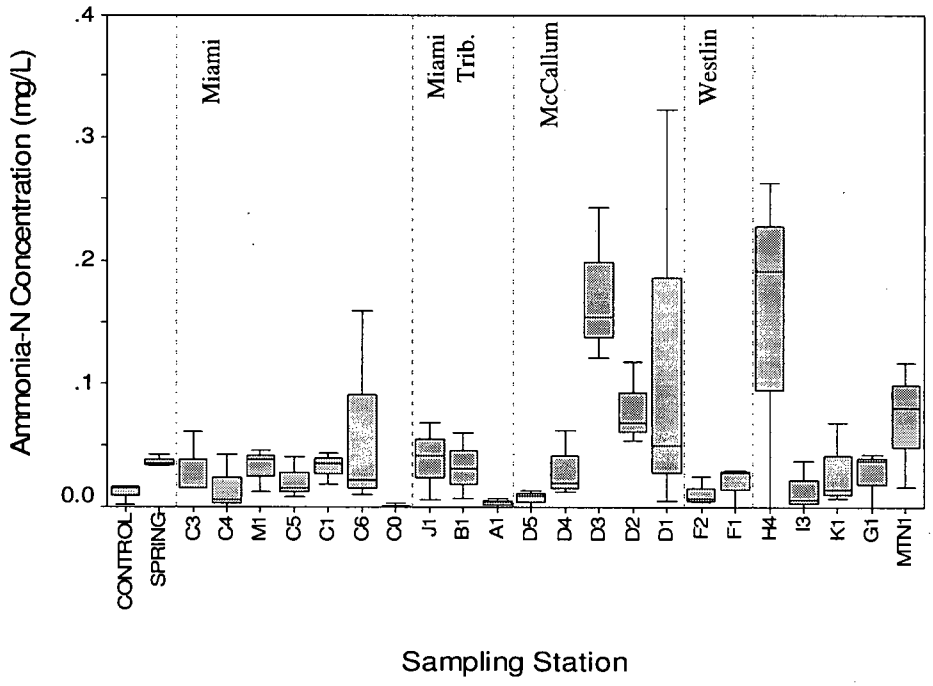

Figure 4.17a: Box and whisker plot of dry season median combined ammonia concentrations (NH_3-N and NH_4^+-N) with adjusted scale.

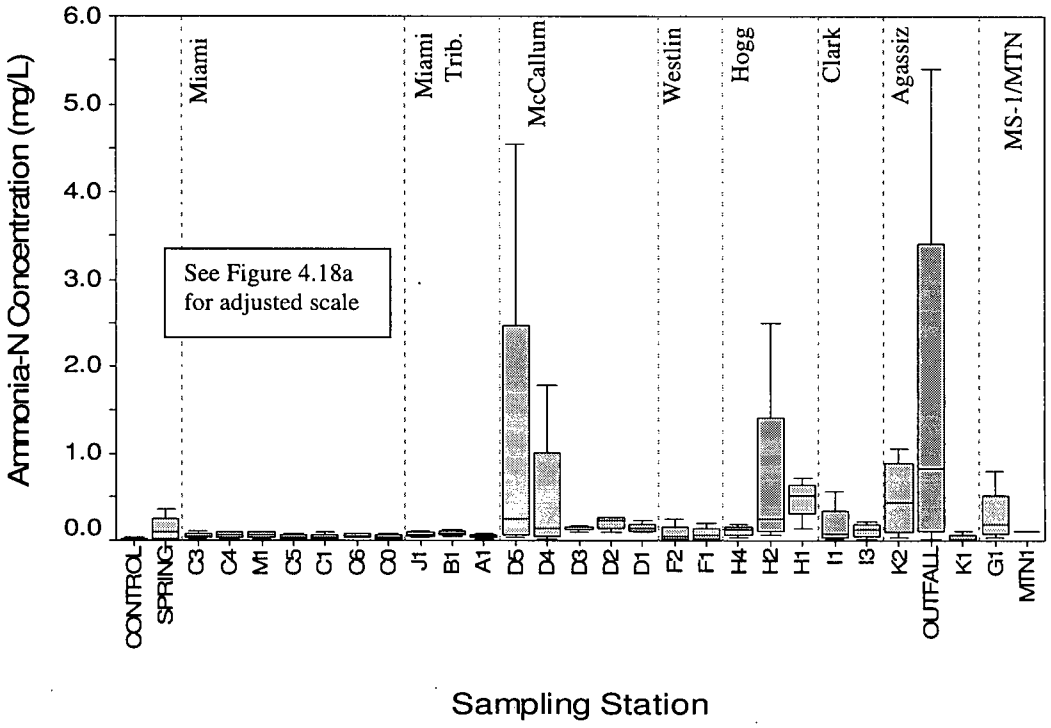

Figure 4.18: Box and whisker plot of wet season median combined ammonia concentrations (NH_3-N and NH_4^+-N) at sampling stations throughout the AHHS watershed.

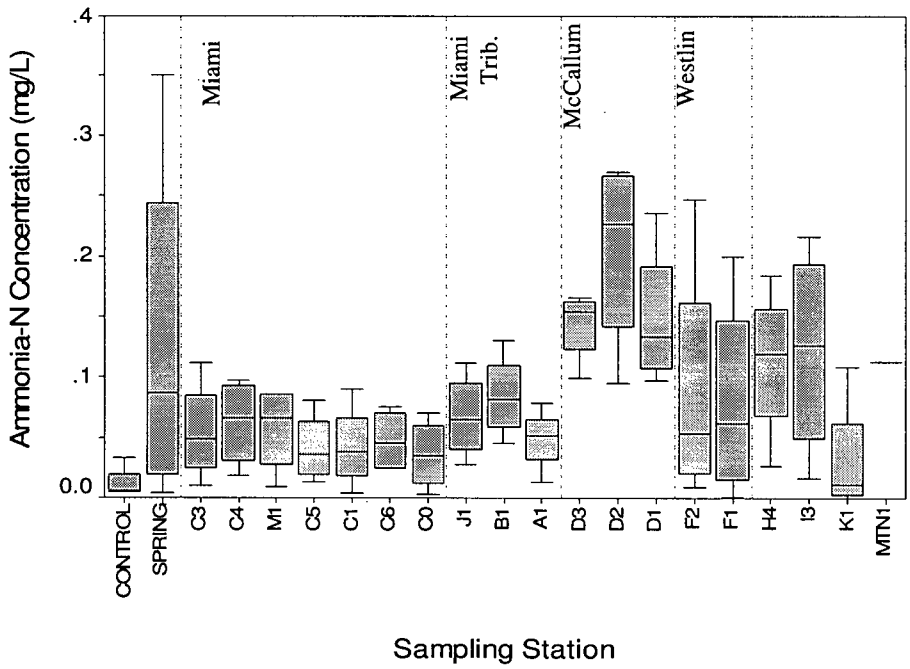

Figure 4.18a: Box and whisker plot of wet season median combined ammonia concentrations (NH_3-N and NH_4^+-N) with adjusted scale

4.12 Variations in Chlorophyll

4.12.1 Seasonal Variations

As can be expected, there were significant differences in chlorophyll *a* levels between the wet (winter) and dry (summer) seasons. This is obviously due to the more prolific algae growth during the warmer temperatures of summer (Figure 4.19 and 4.20). The highest chlorophyll levels were reached during the months of July and August, while the lowest levels were observed in January and February (Appendix B).

4.12.2 Spatial Variations

No particular slough appeared to have significantly higher concentrations of chlorophyll in the dry season. The highest median chlorophyll concentration was 41.23 μg/L at site I3, which similarly had a high degree of variability. Site F2 and C3 had high median chlorophyll concentrations of 26.76 μg/L and 15.86 μg/L respectively. Elevated levels were also present at sites K2 and H2 (Figure 4.19).

Although low compared to dry season values, relatively high chlorophyll concentrations were observed throughout Agassiz Slough during the wet season period. In particular, the outfall had the highest median for chlorophyll at 3.66 μg/L while stations K1 and K2 had levels of 1.64 μg/L and 1.96 µg/L respectively. High levels were also observed in McCallum Slough at stations D5 and D4 with concentrations of 1.64 μg/L and 2.41 μg/L respectively. Site H1 had chlorophyll levels higher then site H2, with a median concentration of 3.12 μg/L. Both the control and spring showed the lowest levels of chlorophyll during the wet season (4.20).

4.13 Sediment Properties

4.13.1 Loss on Ignition (LOI)

Results for LOI, a measure of the organic matter content of sediments, were quite variable within the watershed without any clear pattern between sites or watercourses. In addition, there does not appear to be any significant difference between the results from the wet and dry season (Figure 4.21).

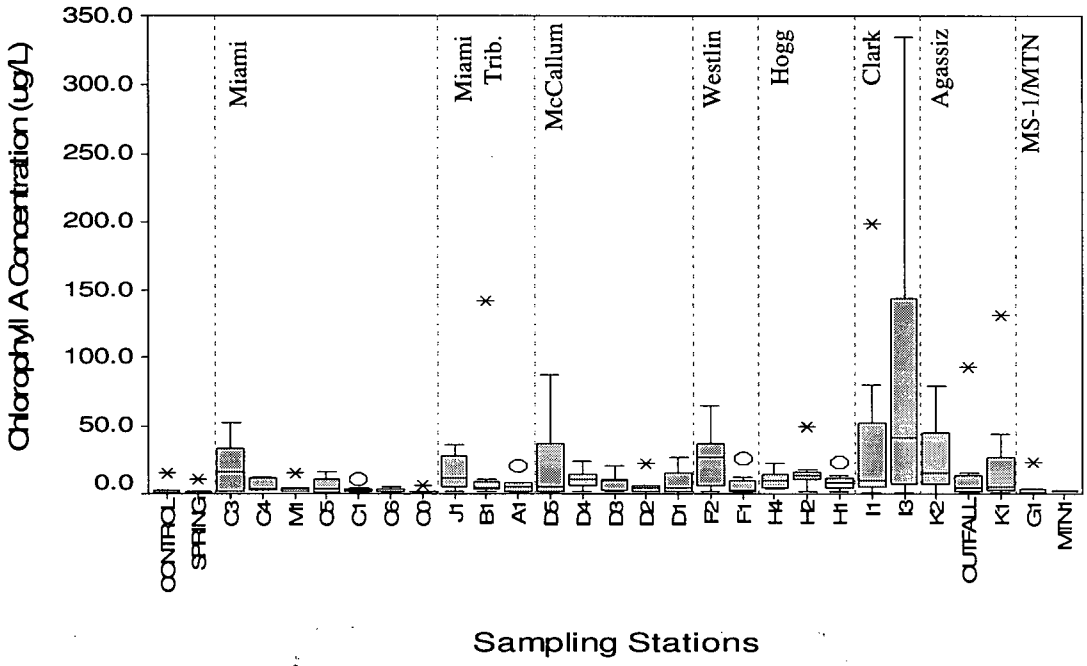

Figure 4.19: Box and whisker plot of dry season median chlorophyll concentrations at sampling stations throughout the AHHS watershed.

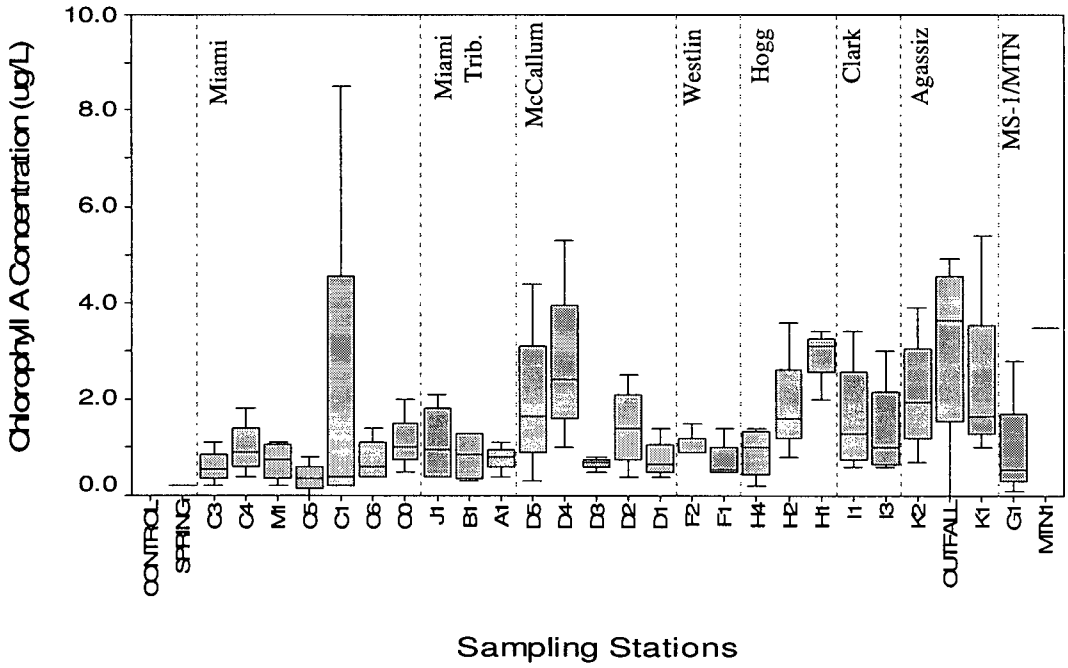

Figure 4.20: Box and whisker plot of wet season median chlorophyll concentrations at sampling stations throughout the AHHS watershed. Scale is adjusted to match lower seasonal concentrations.

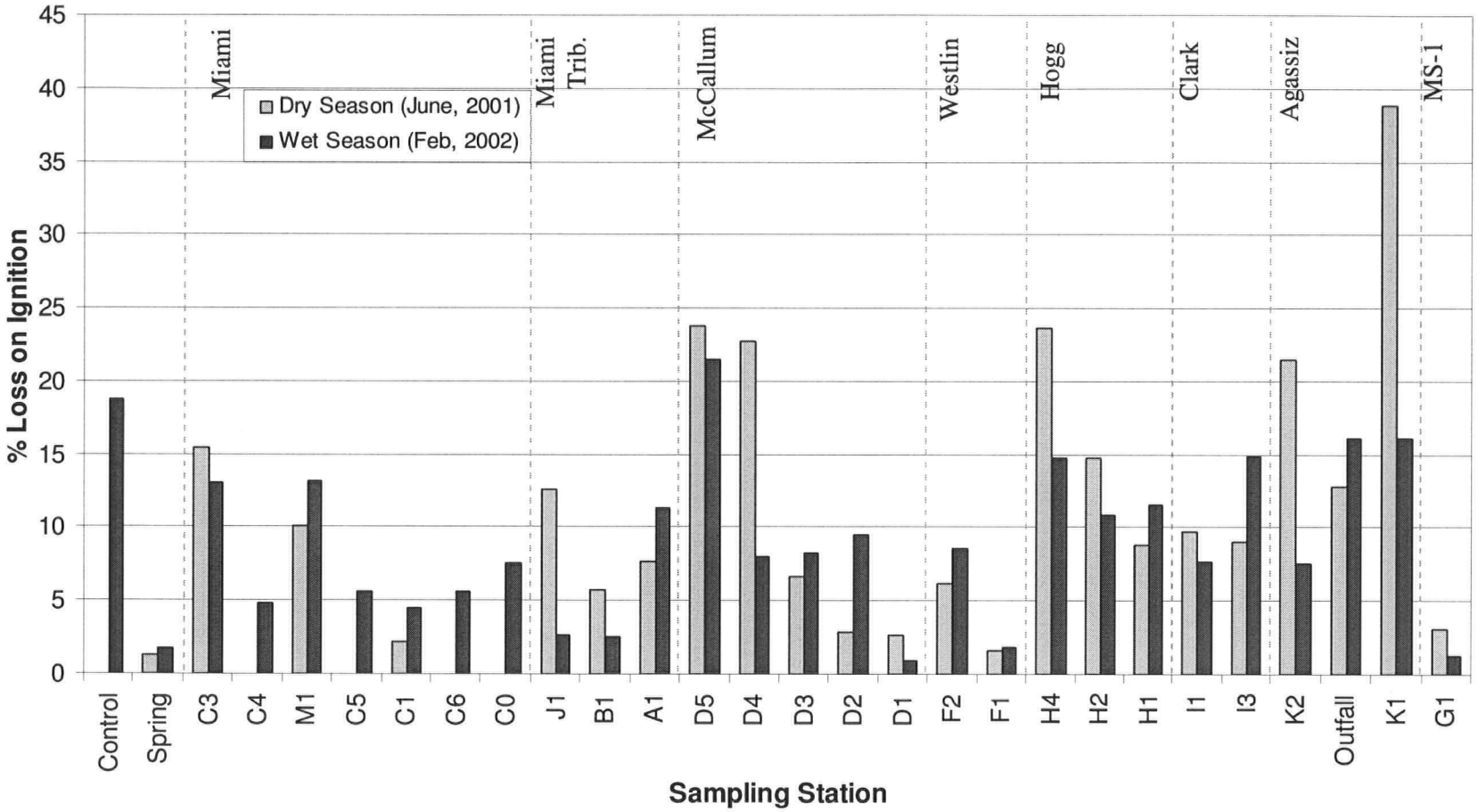

Figure 4.21: Percentage loss on ignition for dry and wet seasons at sampling stations throughout the AHHS Watershed.

During the June 2001 dry season the highest LOI percentage was 38.9% at station K1. In McCallum Slough, sites D5 and D4 gave relatively high LOI percentages of 23.7% and 22.8%. The H4 station also had a high LOI percentage of 23.6%. Relatively low values were observed at the Spring, F1 and C1 at 1.3%, 1.6% and 2.2% respectively (Figure 4.20). An examination of wet seasonal data showed that D5 maintained a relatively high LOI percentage of 21.5%. However, D4 showed a significant decrease in organic content with only an 8% LOI. In fact, many sites which showed high organic contents in the dry season (K1, K2, and H4) proved to have considerably lower LOI percentages in the wet season. The only other sampling site which showed a relatively high LOI percentage was the control station at 18.8%. Low LOI percentages were observed at D1 (1.0%), G1 (1.2%) and the spring station (1.7%).

4.13.2 Grain Size Fractions

Similar to LOI, grain size fractions were quite variable throughout the watershed in addition to being variable between the dry and wet seasons. Grain sizes <0.063 mm have the greatest potential to become resuspended and thus only the results of that grain size will be discussed. During the dry season, the station with the largest percentage of grain sizes <0.063 mm was station A1 at 60.6%. Significant percentages of the silt-clay component were also observed at site M1 (50.7%) and C3 (45.9%) (Figure 4.22).

During the wet season, sampling sites with fine grained compositions appeared at different sites then the dry season. The highest percentages appeared at sites F2 and C3 with 48.6% and 52.8% respectively. In addition, sampling stations A1, C4, H1, I1 and I3 all had a <0.063 mm fractions above 40%.

4.14 Metals in Sediments

Metal concentrations in sediments within the AHHS watershed appear to be quite variable between the wet and dry seasons. In general, manganese and iron had the highest metal concentrations throughout the watershed, however a significant proportion is assumed to come naturally from the surrounding soils. Although no data is available

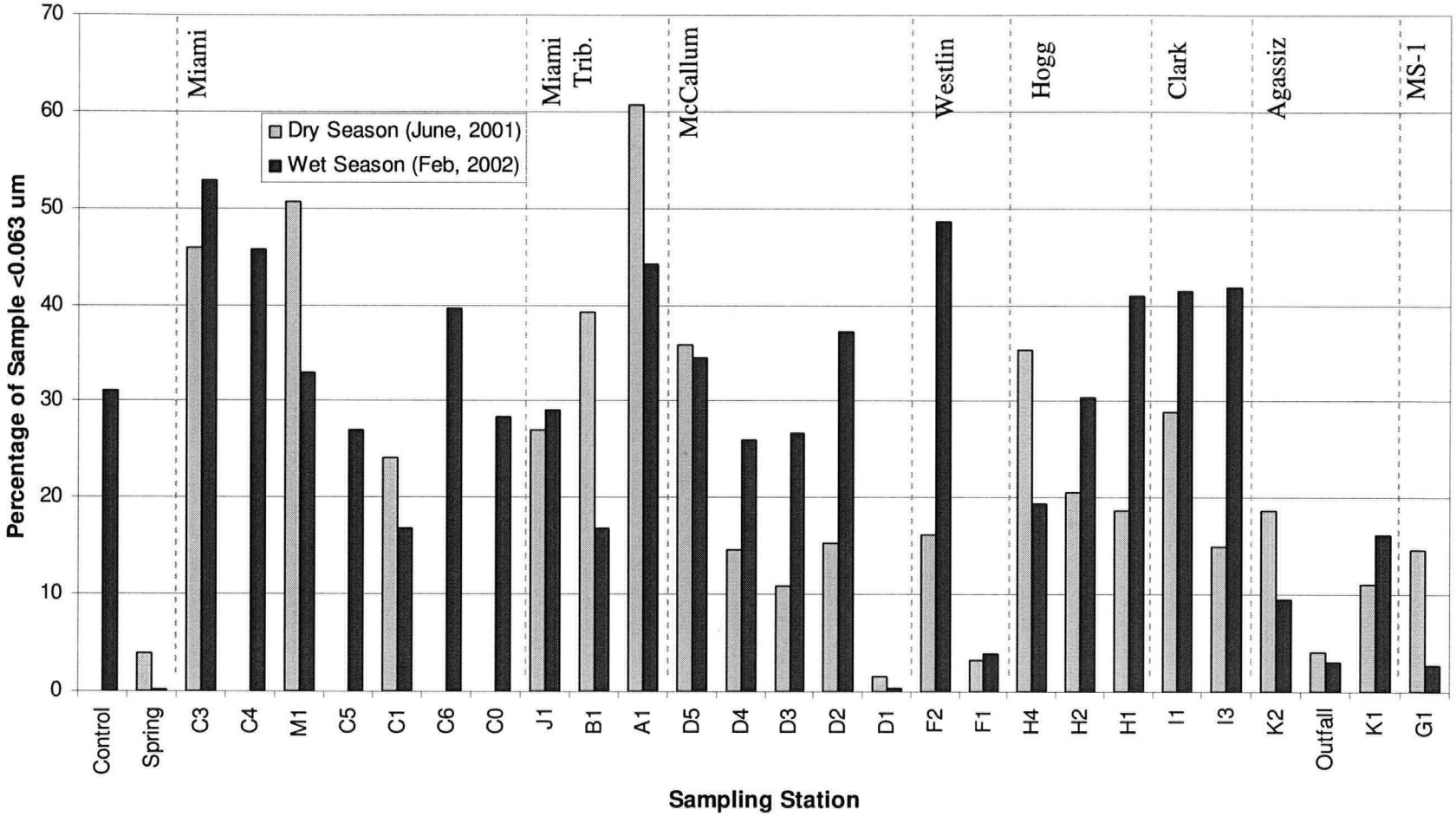

Figure 4.22: Percentage of sample in the <0.063µm grain size fraction at sampling stations throughout the AHHS Watershed.

for the AHHS watershed, both iron and manganese matched other ranges measured in the Lower Fraser Valley (McCallum, 1995 and Cook, 1996).

4.14.1 Lead

Dry season sampling showed that the highest lead concentrations were present in Agassiz Slough. The outfall station and K2 gave concentrations of 148.8 mg/kg and 104.0 mg/kg respectively. Site K1 also had high levels of lead with a sediment concentration of 72.2 mg/kg. Other notably high concentrations were observed at site D5 (70.9 mg/kg) and I3 (75.9 mg/kg). The spring station gave the lowest lead concentrations at 18.7 mg/kg (Figure 4.23).

Similar to the dry season, the highest lead concentrations during the wet season were recorded in Agassiz Slough. The outfall station had the highest lead concentrations at 144 mg/kg, while site K1 and K2 had concentrations of 110.3 mg/kg and 85.8 mg/kg respectively. Apart from Agassiz slough, other notably high lead concentrations were observed at sites C5, D5, F1 and I3 with concentrations ranging from 65.5 mg/kg to 91.8 mg/kg (Figure 4.23). Wet season samples for lead were generally higher during the wet season than the dry season.

It is apparent from the data that most of the samples from Agassiz Slough are above 31 mg/kg, the lowest effect level (LEL) of BC Sediment Quality Guidelines. None of the samples within the AHHS watershed were near the severe effect level (SEL) of 250 mg/kg.

4.14.2 Copper

Agassiz slough presented the highest copper concentrations during the dry season. The outfall station had the highest concentration of 148.8 mg/kg, while station K1and K2 had concentrations of 72.2 mg/kg, and 104.0 mg/kg respectively. Besides Agassiz Slough, high concentrations were observed at site I3 (75.9 mg/kg), H2 (65.0 mg/kg) and D5 (70.9 mg/kg) (Figure 4.24).

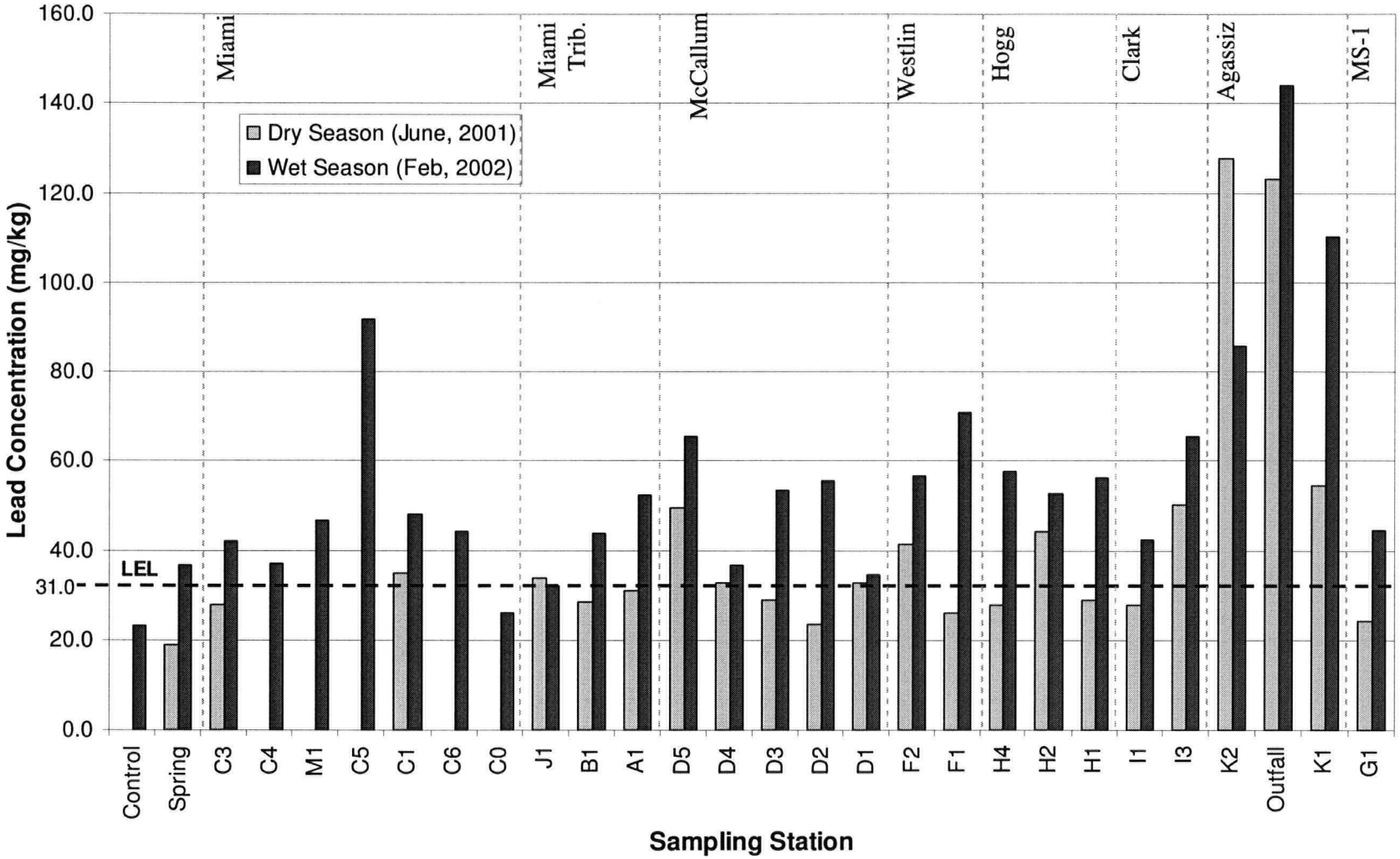

Figure 4.23: Lead Concentrations for Dry and Wet Seasons at Sampling Stations throughout the AHHS Watershed. BC Sediment Quality Guideline LEL is shown at 31 mg/kg.

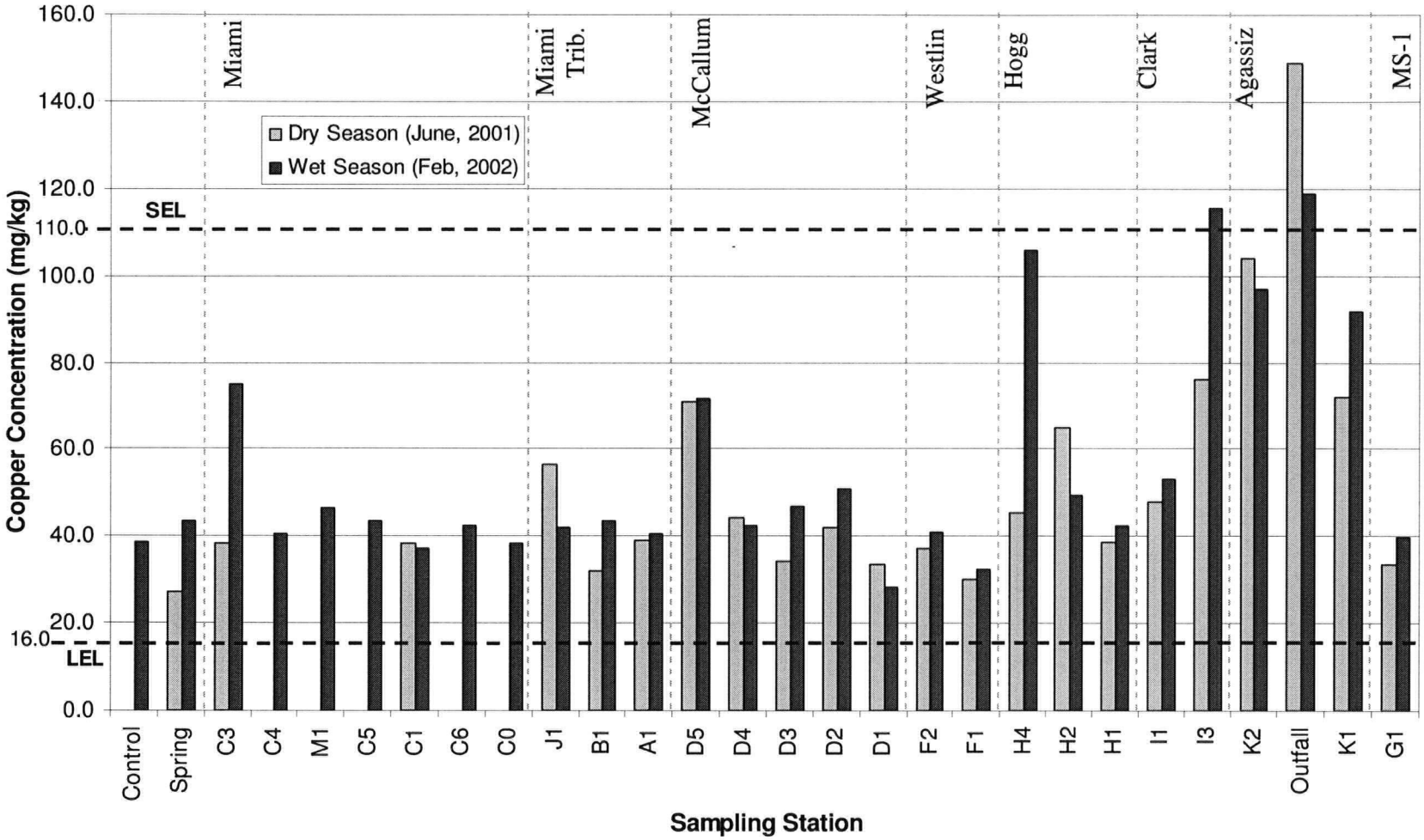

Figure 4.24: Copper Concentrations for Dry and Wet Seasons at Sampling Stations throughout the AHHS Watershed. BC Sediment Quality Guidelines are shown at 16 mg/kg (LEL) and 110 mg/kg (SEL).

The wet season results showed the highest concentrations of copper around the outfall at 118.9 mg/kg. In addition to sites K1 and K2, which continued to show high concentrations through the wet season, sites C3 and D5 showed relatively high copper levels at 74.9 mg/kg and 71.6 mg/kg respectively. High copper concentrations were also apparent at station H4 with a concentration of 105.6 mg/kg. It did not appear that the high levels at site H4 were experienced further downstream in Hogg Slough at stations H2 and H1.

It is interesting to note that all of the sampling stations recorded copper levels well above BC LEL's concentration of 16 mg/kg. Both the outfall station and station I3 had levels above the provincial SEL's of 110 mg/kg, indicating copper concentrations at these sites are at toxic levels.

4.14.3 Zinc

Similar to other metals, zinc concentrations were highest in Agassiz Slough. In particular, high levels were observed at the outfall station during the dry season with a concentration of 737.7 mg/kg. High zinc concentrations were also observed at stations I3 (231.0 mg/kg), F2 (191.9 mg/kg) and H2 (238.2 mg/kg).

During the wet season, the outfall showed a lower concentration of 508.2 mg/kg. Zinc concentrations at the other Agassiz Slough stations remained high with K2 and K1 giving concentrations of 253.3 mg/kg and 289.0 mg/kg respectively. The wet season also showed relatively high concentrations of zinc in Westlin Ditch with the concentrations of sites F1 and F2 at 184.3 mg/kg and 152.9 mg/kg. Other notable locations with relatively high zinc concentrations included I3, H2, and C5. The concentration at C5 (143.3 mg/kg) appears to be significantly higher than zinc concentrations at other sampling stations within Miami Creek (Figure 4.25).

The high zinc levels observed in Agassiz Slough are well above the freshwater sediment LEL of 120 mg/kg. In addition to the stations in Agassiz Slough and Westlin Ditch, other stations such as C5 (McCallum Slough), I3 (Clark Ditch), and H2 (Hogg Slough) are above freshwater sediment LEL's, and there may be adverse impacts to the aquatic ecosystem at these stations.

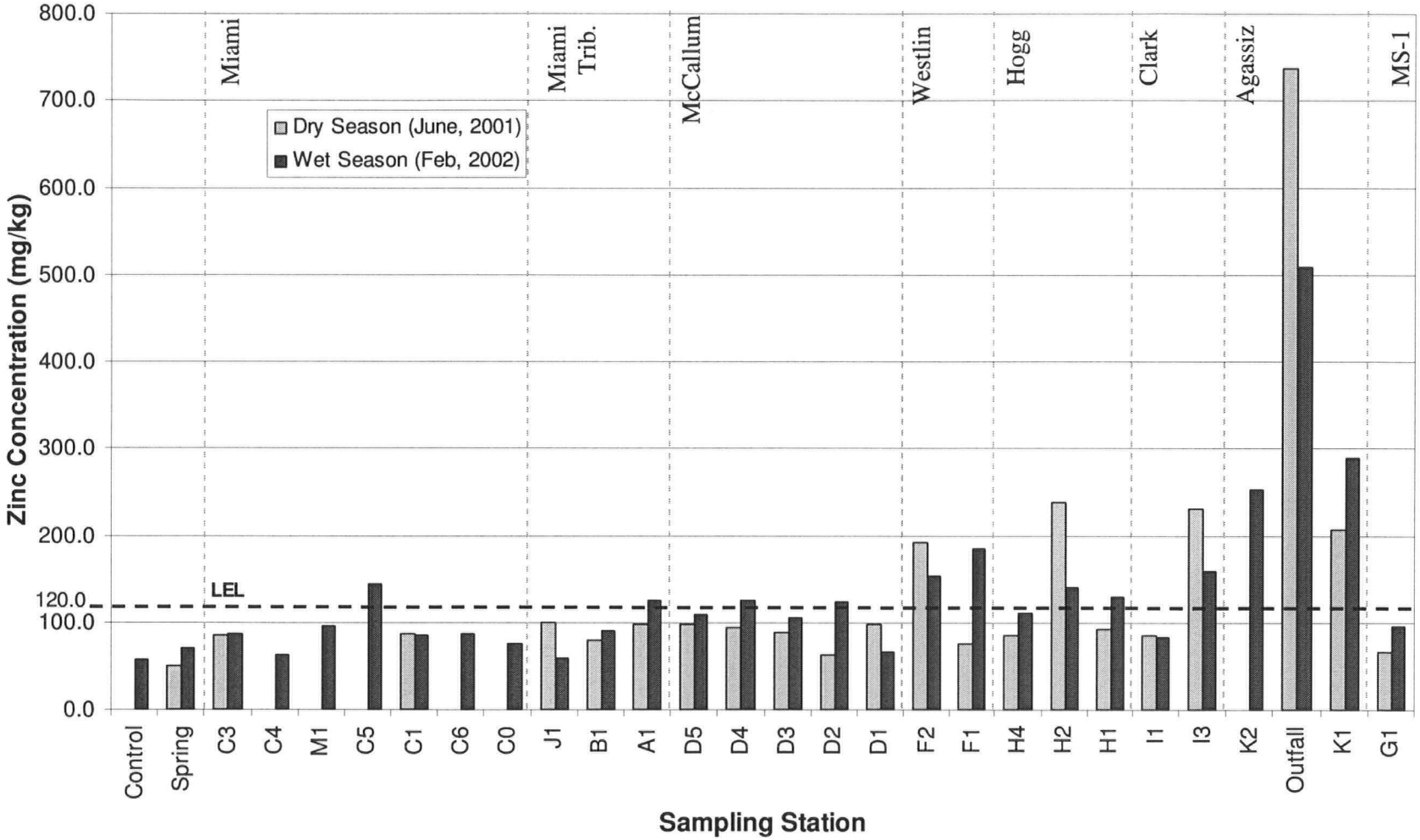

Figure 4.25: Zinc Concentrations for dry and wet seasons at sampling stations throughout the AHHS Watershed. BC Sediment Quality Guidelines are shown at 120 mg/kg (LEL).

4.14.4 Manganese

Dry season results showed a high level of manganese present at station K2 of 1296.4 mg/kg. Similarly, in Agassiz Slough, a high concentration was observed outside of the outfall at 835.2 mg/kg. This is likely because of impermeable surface runoff from manganese based fuel additives which have been flushed into the sewer system during storm events. Besides Agassiz Slough, a high manganese concentration was observed at station B1 of 734.0 mg/kg (Figure 4.26).

Results during the dry season gave very different results than the wet season, particularly in Agassiz Slough with dry season concentrations higher then wet season concentrations. The highest concentration was recorded at site D2 (1955.4 mg/kg). It is thought that this spike in concentration may be due to gravel mining activities near this station. Sites F1 and B1 also gave relatively high concentrations at 917.0 mg/kg and 779.2 mg/kg respectively. In contrast to other metals, site D5 had relatively low concentrations of manganese in both the wet and dry seasons. Although there are no criteria for manganese concentrations, the majority of stations did not appear to deviate far from expected background concentrations found in sediments throughout the region (Table 2.5).

4.14.5 Iron

Due to the high concentrations of iron present throughout the watershed, results were given as percentages of the total sample. Overall concentrations did not appear to vary between watercourses, however there appeared to be a general pattern of increasing concentrations as one moved downstream during the dry season. The highest concentrations were observed at stations I3, K2, and C1 (8.69%, 8.60% and 8.44% of the sample respectively). Site D5 gave the lowest iron percentages during the dry season at 1.94% of the sample (Figure 4.27).

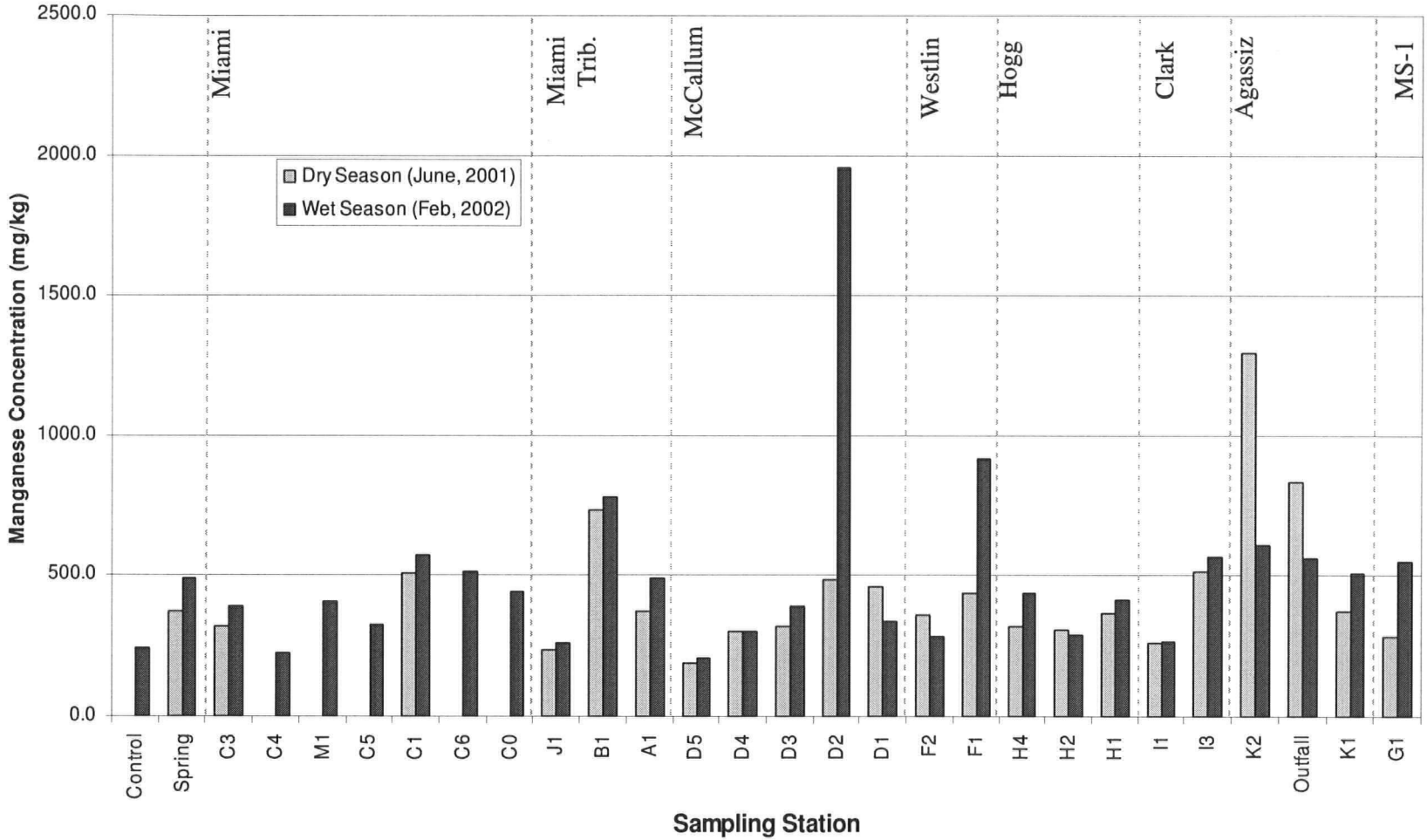

Figure 4.26: Manganese Concentrations for Dry and Wet Seasons at Sampling Stations throughout the AHHS Watershed.

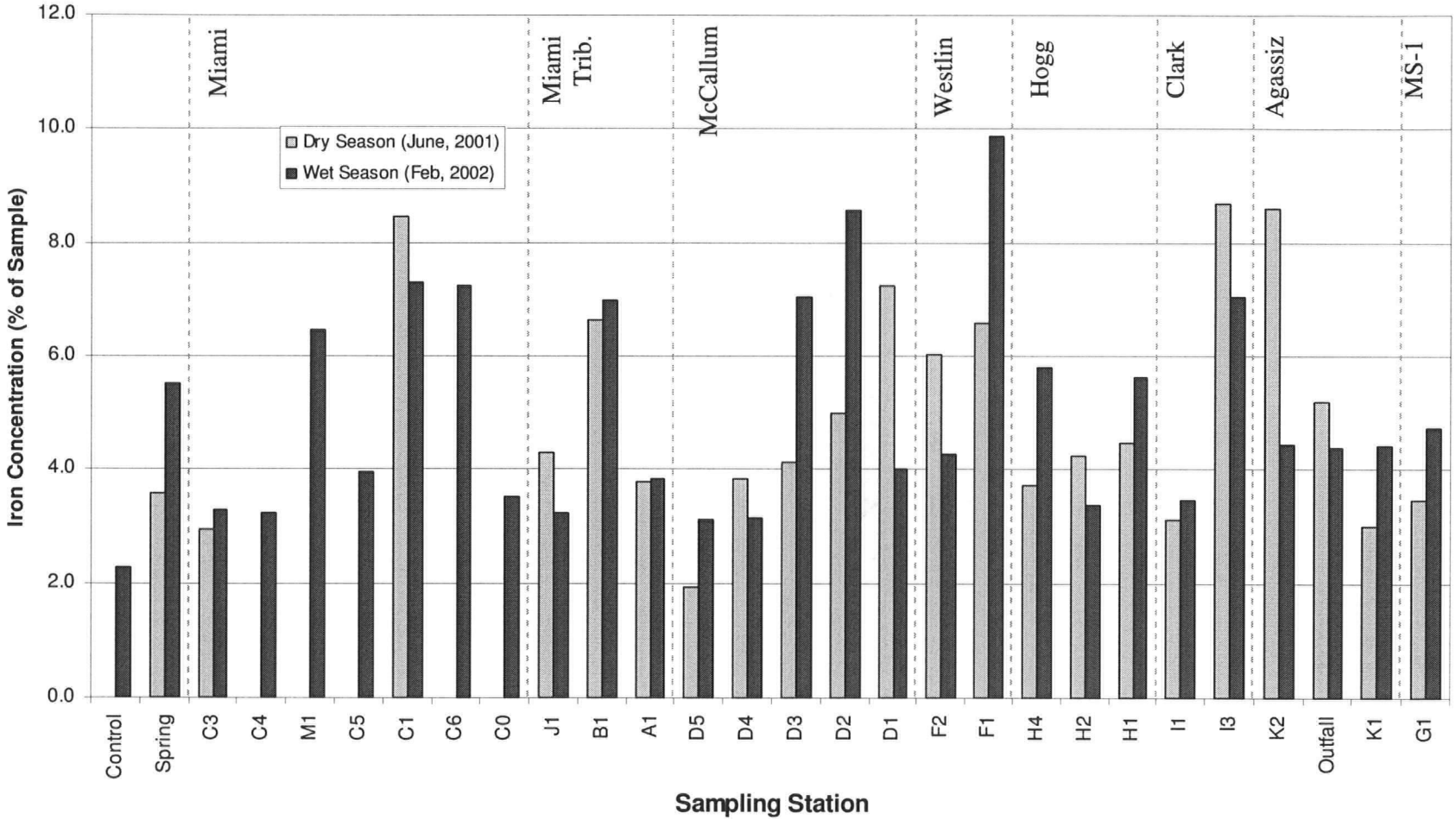

Figure 4.27: Iron Concentrations for dry and wet seasons at sampling stations throughout the AHHS Watershed.

The wet season showed high iron concentrations at different locations. Most interesting was that the relatively high levels of iron in Agassiz Slough were not observed. The highest iron concentrations were observed at stations F1 and D2 at 9.87% and 8.57% of the respective samples. The lowest iron level was observed at the control station at 2.29% of the sample (Figure 4.27).

4.15 Bioassay Results

4.15.1 Bioassay Field Conditions

Field conditions of the various locations differed considerably depending on the water chemistry and the conditions of the riparian zones surrounding the stations. Stations such as D5 (McCallum Slough) and C3 (Miami Slough) had an abundance of trees and vegetation occupying the riparian zones, and thus sunlight exposure was not as intense as in sites such as F2 (Westlin Slough) where there was little to no riparian vegetation. Many of the sampling sites ranged from areas of almost no flow, to streams with moderate flow (Appendix F).

The coverage of algae and iron oxide deposits appeared to be a significant problem in the Miami and Westlin Sloughs. Coverage often ranged from moderate to severe (Appendix F) and cleaning of the mesh was often required. The one exception to this was the control sampling station where only a slight amount of algae was found to cover the mesh during the weekly inspections. The MTN-1 station also appeared to be moderately free of any periphyte and algae growth. Although periphyton coverage was quite severe between the first two weekly checks (May 7 – May 24), the coverage rate appeared to slow near the end of the month.

Water levels were fairly constant during the course of the study, however there was a significant increase in the water levels of Agassiz Slough, Miami Creek and Hot Springs Slough during the last two weeks of the experiment. This resulted in the loss of samples at sites K2, Outfall, C6, and C1 since the mesh bags were too deep for retrieval. Sample bags from sites H1, D2, D3 and D4 went missing during the course of the experiment, most likely due to movement by local farmers.

4.15.2 Mesh Conditions

No damage was apparent to the mesh bags after retrieval. Invertebrates such as snails, small leeches, and worms were sometimes present inside the meshes. The presence of algae, periphyton, and duckweed were also noted between the mesh and the pores of the mesh.

Moss stems ranged in colour from a light green to brown colour (Appendix G). In many of the stations the base of the stems were brown compared with the green colour of the shoots. Although there was considerable variability of stem appearances between stations, stems were quite similar within each individual station, including those where duplicates were located. Some stems were observed to be missing from meshes. It is thought that the disappearance was due to grazing from invertebrates since no holes were viewed on the meshes. Dead stems were observed at sites H2, D1, D5, G1, C3 and I3. Dead stems with the growth buds missing were not included in the results since it was assumed growth inhibition did not result from the environmental quality of the water.

4.15.3 Moss Growth

Shoot length growth and the average dry weight/mm for each site is presented in Figure 4.28. Overall, differences in growth were only statistically significant within Hogg Slough. Mosses at site H4 showed a significant amount of growth at 21.3 mm, and gave the greatest amount of growth out of all the sampling sites within the watershed. However, downstream from this location site H2 gave the shortest average shoot growth of 2.4 mm. The replicate sample at this location confirmed that station H2 was not conducive to significant growth.

A high amount of growth was noted at sites I3, F1 and MTN-1 with length increases of 18.3, 19.9 and 18.3 mm respectively. Site F2 also showed fairly robust growth at 15.8 mm. Both McCallum Slough and Miami Creek did not show any significant differences in growth, and overall only showed average increases in length. Both replicates, at D1 and H2, showed a relative low variability at their respective stations.

The highest values for average dry weight/mm were 0.043 g/mm and 0.033 g/mm observed at station H2 and the H2 replicate respectively. As with the shoot lengths, there

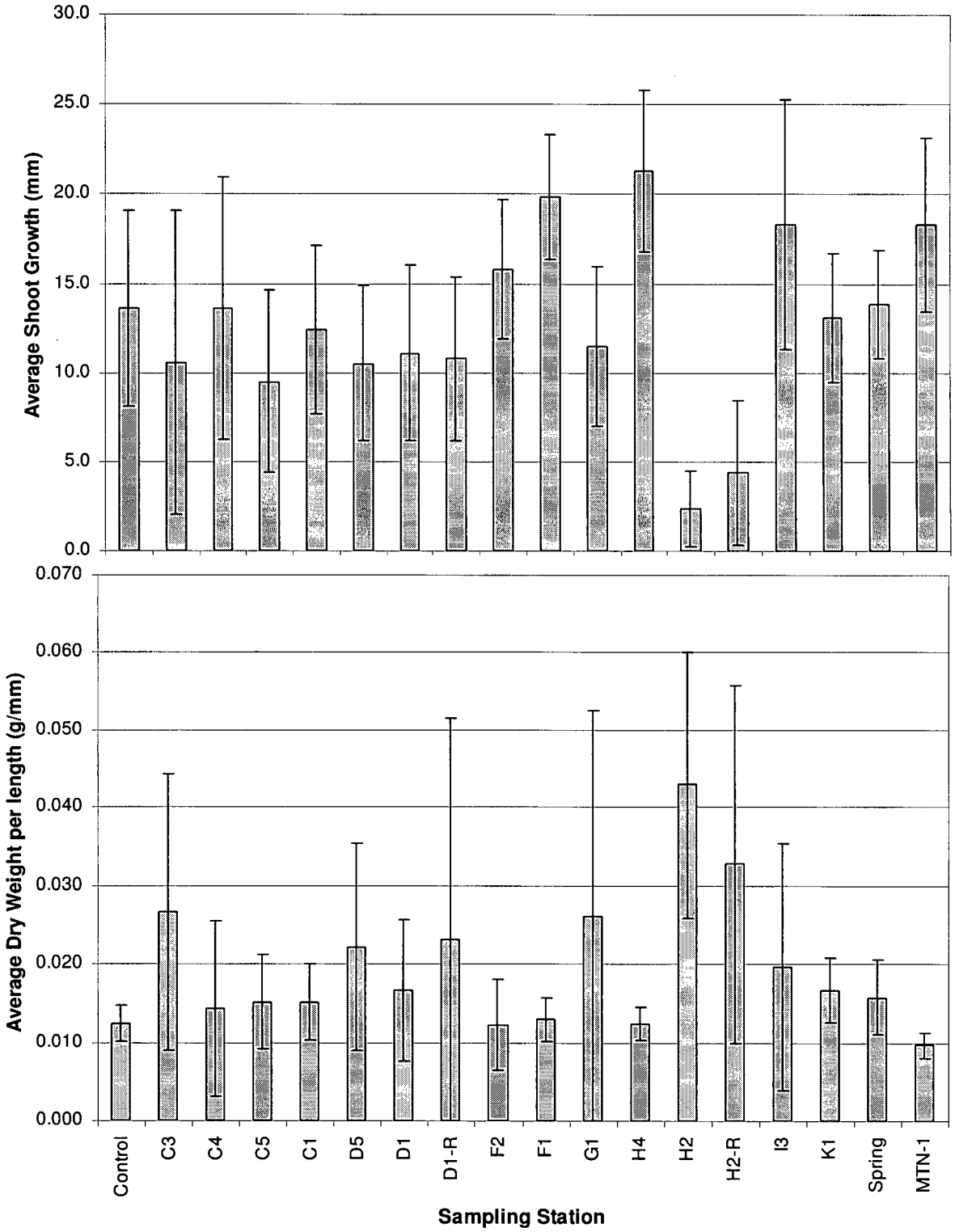

Figure 4.28: Average shoot growth and average dry weight/mm of the moss *Fontinalis antipyretica* stems and with standard deviations at selected sampling stations throughout the AHHS Watershed (between May 10, 2002 and June 7, 2002).

was a significant difference between stations H4 and H2. The H4 value was significantly lower with an average dry weight of 0.012 g/mm. The lowest average dry weight was observed at the MTN-1 station at 0.010 g/mm. Although not statistically significant, it is interesting to note that high average weights were also observed at stations C3, D5, and I3 with values of 0.027 g/mm, 0.025 g/mm and 0.020 g/mm respectively. All three of these stations show relatively high nutrient concentrations compared to other sites within the watershed.

4.15.4 Concentrations of Chlorophyll A and Chlorophyll B

Chlorophyll *a* concentrations in stems within the various sloughs were quite variable and did not appear to show any correlations with measured water quality parameters. The highest mean chlorophyll *a* level was recorded at site C5, with a value of 12.94 mg chl/mg moss. Station C4 and MTN-1 also gave relatively high readings at 12.70 mg chl/mg moss and 12.03 mg chl/mg moss respectively. The spring station had the lowest recorded chlorophyll *a* concentrations at 5.89 mg chl/mg moss. The only statistically significant difference was recorded between the spring station and sites C4 and C5 (Figure 4.29).

Results measuring chlorophyll *b* showed significantly lower concentrations then that of chlorophyll *a*. However, chlorophyll *b* results were much more variable. Analysis of variance tests did not show a statistically significant difference between any of the sampling stations based on chlorophyll *b* concentrations. The highest notable value was obtained at the MTN-1 site where concentrations of chlorophyll *b* reached 2.39 mg chl/mg moss. The H4 site also had relatively high concentrations of chlorophyll *b* at 2.19 mg chl/mg moss. The lowest results were observed at the spring, and site D5 at 0.424 and 0.413 mg chl/mg moss respectively (Figure 4.29).

4.16 XAD Resin Results

4.16.1 Labscale Results

As can be seen by Figure 4.30, the efficiency of atrazine adsorption to XAD-7 resins is dependent primarily on the atrazine concentration. The use of distilled water compared to actual site water also appeared to have an impact on adsorption capacity. The best

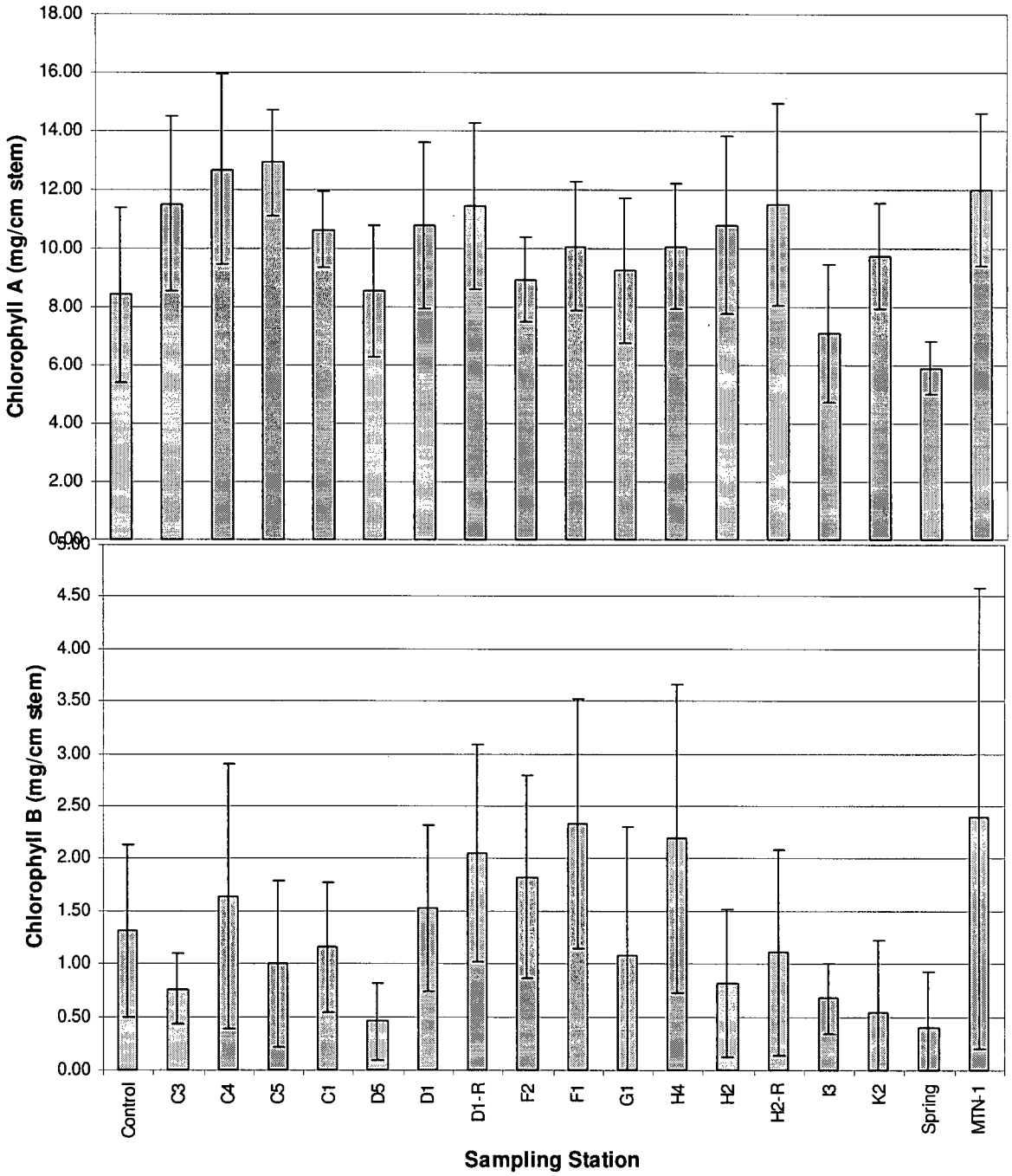

Figure 4.29: Average chlorophyll *a* and chlorophyll *b* concentrations of *Fontinalis antipyretica* stems with standard deviations at selected sampling stations throughout the AHHS Watershed (between May 10, 2002 and June 7, 2002).

adsorption efficiency was found to be 76.0% in distilled water at pH 7, with an atrazine concentration of 100 ppb. An atrazine concentration of 1 ppb in pH 7 sample water gave the lowest recovery of 3.1%. Based on previous research, it was expected that a pH of 5 would provide the most suitable environment for adsorption, however it gave slightly lower efficiencies of 58.5%. In general it was observed that a pH of 7 gave slightly better recovery efficiencies then a pH of 5. The one exception were samples run with an atrazine concentration of 10 ppb. In this case, the conductivity of the samples did not appear to affect the efficiency of the adsorption process.

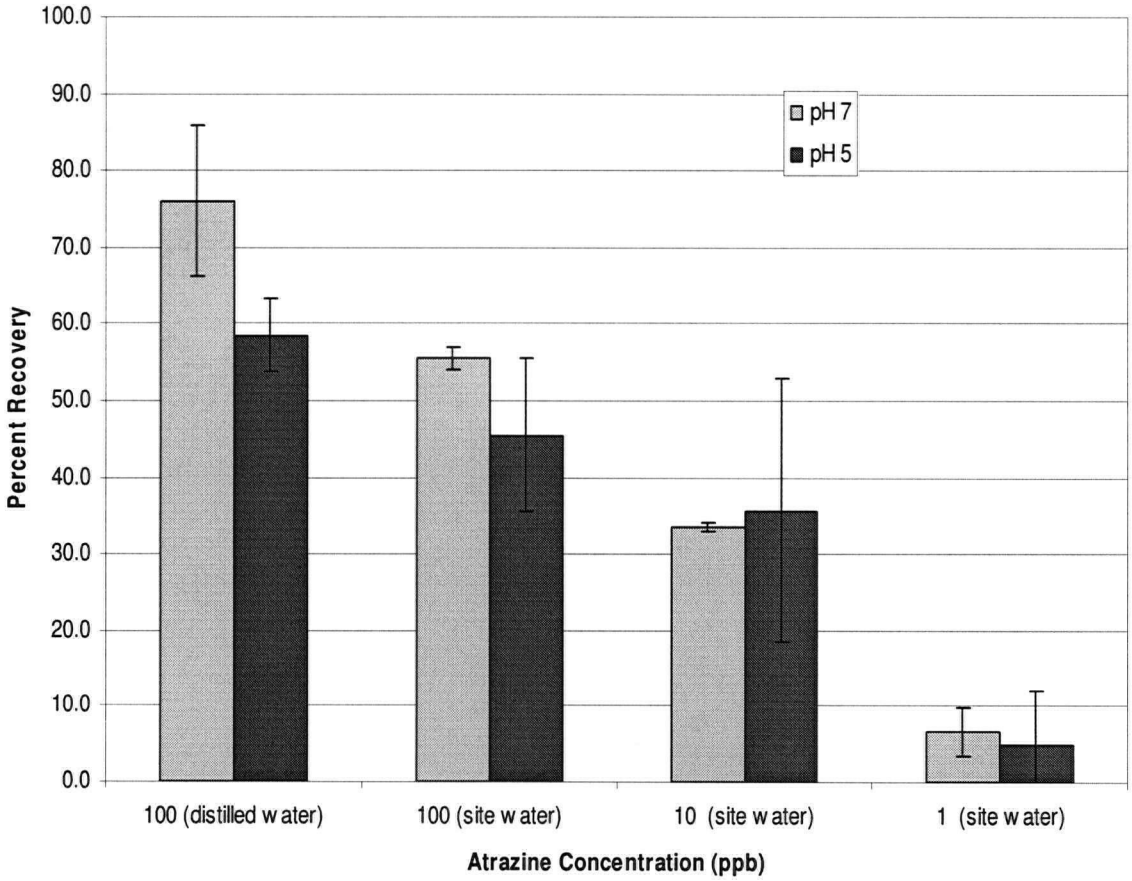

Figure 4.30: Average laboratory scale atrazine recoveries using XAD-7 resin under different pH and atrazine concentrations. Results and standard deviations are based on duplicate samples.

4.16.2 Resin Apparatus Field Conditions

The combination of iron oxides and periphyton growth was quite prevalent at sites F2 (Westlin Ditch) and D1 (McCallum Slough). The cleaning of the equipment was often required on the weekly visits to the sites. Due to the setup of the apparatus, direct cleaning of the membranes was not possible. However, it may not have been desirable given their delicate nature. Photos of the resins after retrieval (Appendix A) show that there was considerable fouling around the membranes from site D1, compared to growth around the membranes of the control.

As with the moss bioassays, flooding of Agassiz and Miami Slough became a problem for the retrieval of the resins. Although the resins at station K1 (Agassiz Slough) were retrieved, resins at C0 (Miami Slough) could unfortunately not be found.

4.16.3 Field Performance

The results of the field experiment show a gradient effect between resins at the same sampling station (Figure 4.31). The first resin pouch of each group, located at the front of the sampling apparatus, retained the highest amount of atrazine while resin pouches immediately behind the first showed progressively lower concentrations. As a result of the obvious concentration gradient, only the first pouch was considered for comparison between sampling stations. The one exception to the concentration gradient observation was at the control site where the final bag adsorbed a significantly higher amount of atrazine. However, this is most likely due to contamination during the analytical process.

After 28 days of exposure, the highest atrazine recovery was found in the first resin packet at site K2 (Agassiz Slough), with a recovery concentration of 0.090 μg atrazine/g resin. Sites F2 (Westlin Ditch) and H1 (Hogg Slough) showed similar recovery concentrations of 0.085 μg atrazine/g resin and 0.075 μg atrazine/g resin respectively. Surprisingly, the control station showed some concentrations of atrazine, with the first resin packet giving a concentration of 0.045 μg atrazine/g resin. The resins at station D1 and its accompanying replicate had the lowest recovery concentrations of 0.023 μg atrazine/g resin and 0.025 μg atrazine/g resin respectively. It is interesting to note that the 2nd and 3rd resin packets for both D1 and D1-R did not show any atrazine recovery.

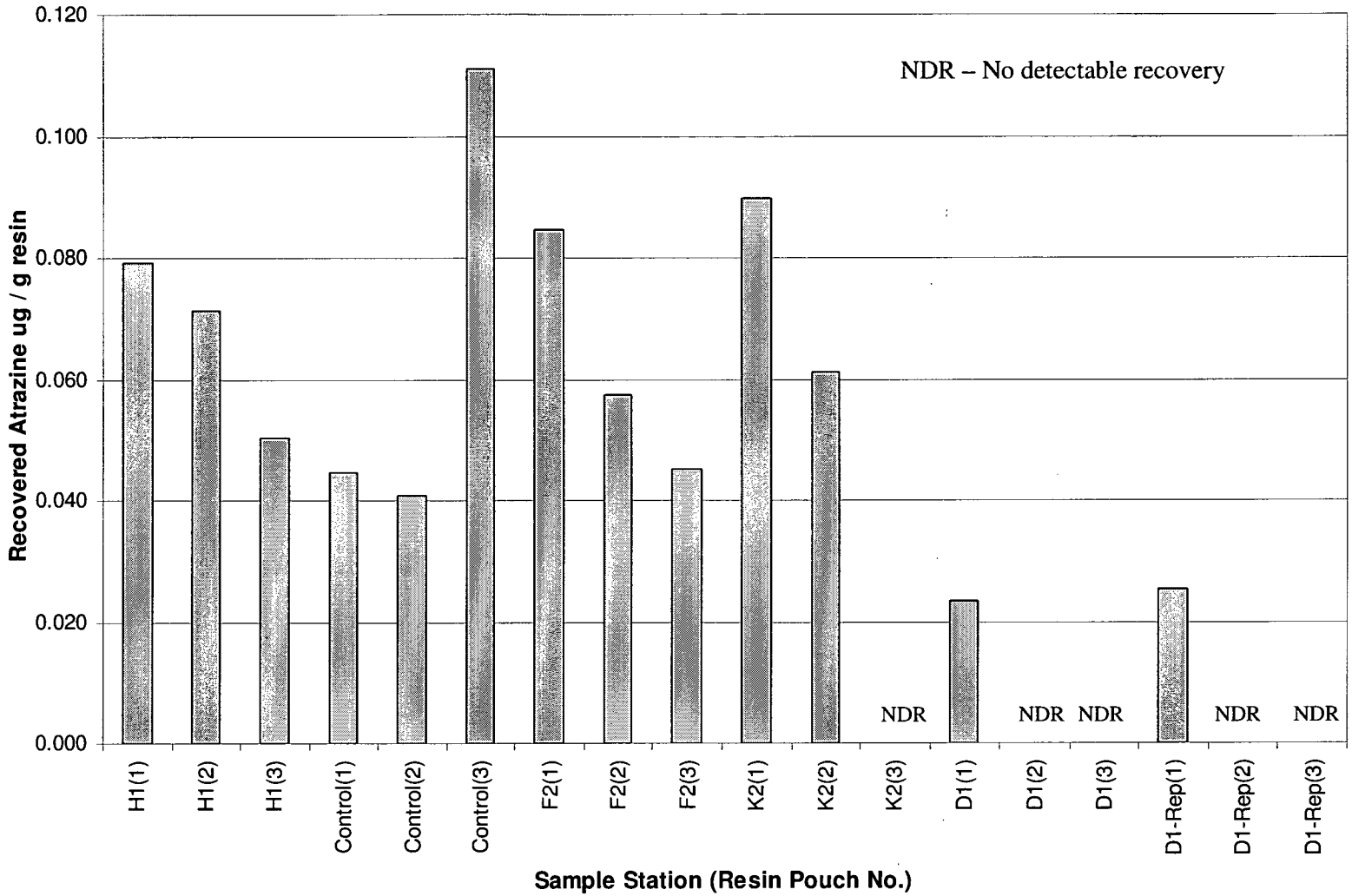

Figure 4.31: Atrazine recoveries using XAD-7 resins at selected sampling stations throughout the AHHS Watershed (Resins placed in field between May 10, 2002 and June 7, 2002 for a total of 28 days exposure)

No significant positive correlation was observed comparing atrazine recovery with corn fields within a 100 m buffer around watercourses.

When compared to labscale recoveries, it was obvious that the field recoveries gave much lower concentrations of recovered atrazine. At its closest level, recovery concentrations measured from the field experiment were at least one order of magnitude lower then recovery concentrations observed in the lab. This indicates either that concentrations were significantly lower then 1 μg/L in the watercourse, or that there are additional factors affecting the recovery of atrazine which are not represented in the laboratory.

4.17 GIS Results

Figure 4.32 shows the various land uses determined by "windshield surveys" and orthophoto interpretation. The total area of the AHHS watershed was calculated to be 6362.8hectares (15723.3 acres). It is apparent that hay/silage fields comprise the principal agricultural land use within the watershed. Corn crops appear to be another significant land use, particularly adjacent to Agassiz Slough. Forest cover takes up the largest area within the watershed due to the surrounding mountains. A summary of the AHHS watershed land uses and there respective areas are located in Appendix E.

Contributing area delineation shows that the MTN-1 (Mountain Slough) encompasses the largest overall area (Figure 4.33). When upstream land uses are taken into account, almost half the land area within the watershed drains to this station. A summary of contributing area land use types and areas is summarized in Appendix E. Delineation of each contributing area proved to be quite difficult due to the overall flat topography, and estimations of contributing area boundaries may have adversely affected correlation results. Many of the contributing area boundaries follow major roads and highways which pose as barriers to surface flow.

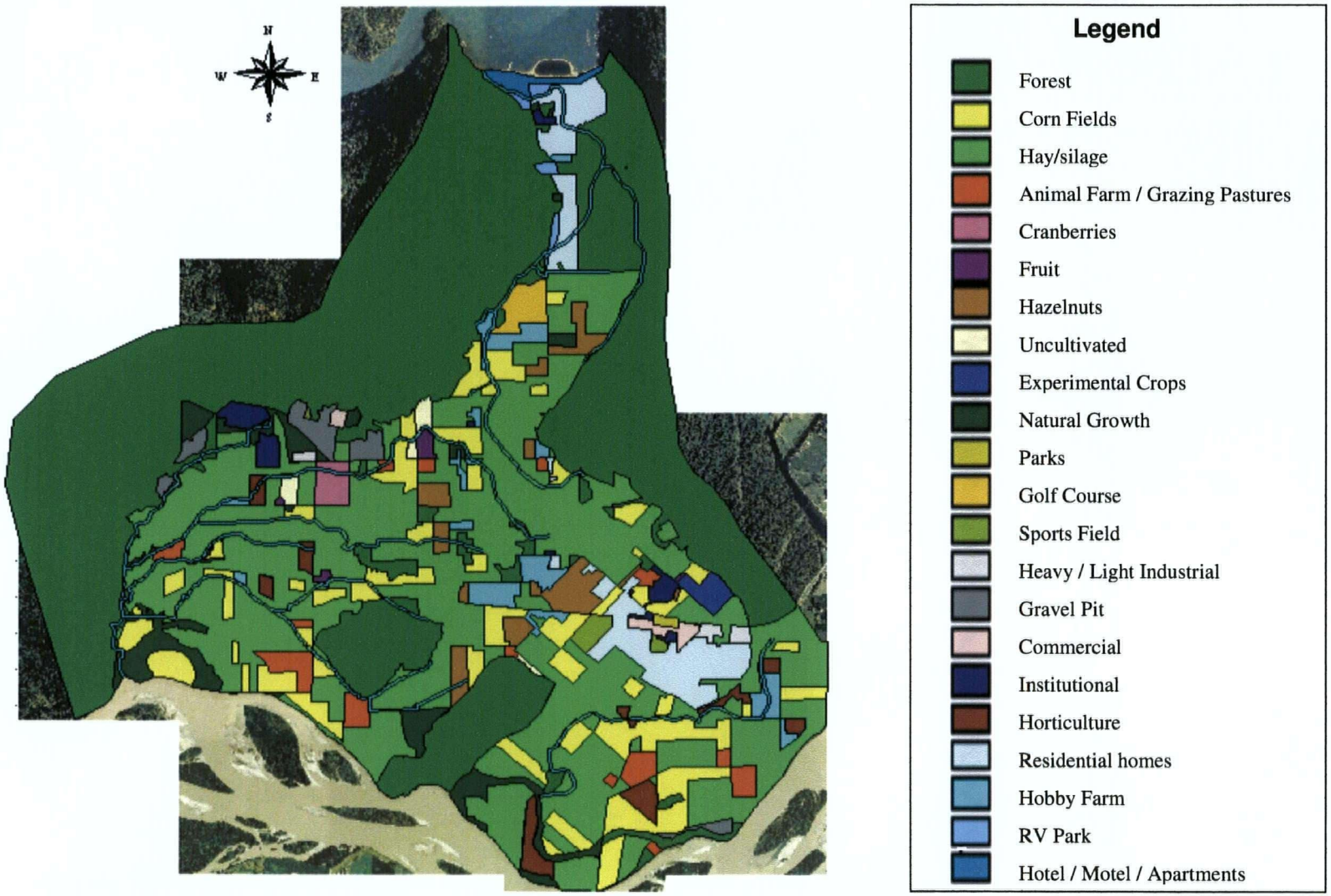

Figure 4.32: Land Uses within the AHHS Watershed

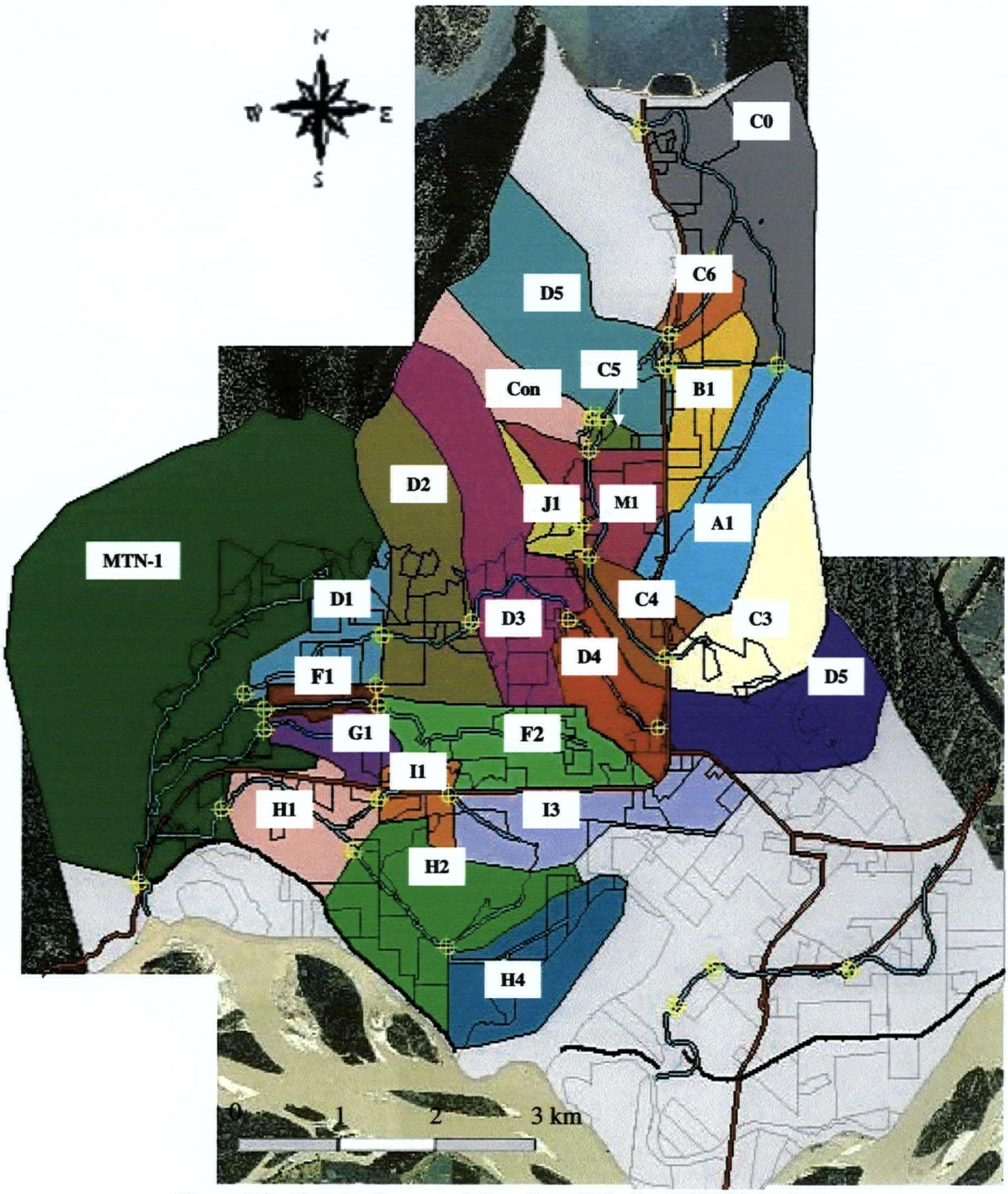

Figure 4.33: Contributing area delineation within the AHHS Watershed

4.18 Water and Sediment Parameter Correlations

4.18.1 Buffer Land use Correlations

Table 4.7 shows the results of Spearman Rank correlations between water quality parameters and land uses within a 100 m buffer around watercourses. Both corn and hay/silage fields showed significant negative correlations for chlorophyll and turbidity. In addition, hay/silage fields showed a positive correlation to pH. Pastures, showed a significant correlation with nitrite, and although not statistically significant, there were also positive correlations with orthophosphate, ammonia, chlorophyll and turbidity during the wet season.

Obvious correlations were also observed for non-agricultural land uses. Residential areas showed strong negative correlations for the nitrate, chlorophyll and turbidity. In particular, nitrate gave strong negative correlations in both the wet and dry seasons. The only positive water quality correlation from residential areas was with median pH levels during the dry season. Positive correlations between chlorophyll and turbidity for natural growth were also observed during the dry/summer season while forest cover showed negative correlations with nitrate levels.

4.18.2 Buffer Land use and Sediment Quality Correlations

The Spearman Rank correlation values for metals and loss on ignition are shown in Table 4.8. Hay/silage fields appear to have the most obvious correlations with sediment quality. Significant positive correlations were observed for manganese and iron, while negative correlations were observed for copper, lead, LOI, and percentage sediment fines <0.063 μm. Corn fields also showed positive correlations for manganese and iron, and a negative correlation for LOI. Grazing pastures showed only a positive correlation for zinc during the wet season, while iron had strong positive correlations with the presence of hobby farms during both seasons.

Some unexpected correlations were a positive correlation between zinc and natural shrub areas in the dry season, and a negative correlation between copper and residential areas during the wet season.

Table 4.7: Spearman rank correlation coefficients between water quality indicators and land uses using the buffer zone method.

Water Quality Indicator	Season	Land Use Type						
		Forest Cover	Corn	Hay/silage	Pasture	Natural Vegetation	Residential	Hobby Farm
Orthophosphate	Dry	-0.19	-0.23	-0.14	-0.15	0.25	0.07	-0.30
	Wet	-0.24	-0.22	-0.10	0.77	0.32	-0.44	-0.40
Nitrate	Dry	***-0.49****	0.04	-0.18	-0.03	-0.10	***-0.76****	-0.21
	Wet	***-0.39****	-0.24	-0.21	0.27	0.07	***-0.73****	-0.28
Nitrite	Dry	-0.19	-0.18	-0.33	0.44	0.39	-0.58	-0.09
	Wet	0.03	-0.12	0.11	***0.85****	0.54	-0.06	-0.03
Ammonia	Dry	-0.19	-0.23	-0.14	-0.15	0.25	0.07	-0.30
	Wet	-0.24	-0.22	-0.10	0.77	0.32	-0.44	-0.40
Chlorophyll	Dry	-0.26	***-0.40****	***-0.49****	0.03	***0.71****	***-0.83****	-0.41
	Wet	0.10	-0.11	-0.14	0.74	0.49	0.34	-0.34
DO	Dry	0.01	-0.04	0.34	0.35	0.36	0.59	-0.01
	Wet	0.06	0.15	0.05	0.29	0.34	-0.34	-0.02
Specific Conductivity	Dry	-0.01	-0.33	-0.07	0.50	-0.19	0.39	-0.03
	Wet	-0.08	-0.22	0.05	0.18	-0.29	0.59	-0.05
pH	Dry	0.00	-0.09	***0.42****	-0.57	-0.39	***0.71****	-0.06
	Wet	-0.08	-0.04	0.26	-0.09	-0.56	0.49	0.00
Turbidity	Dry	-0.26	***-0.40****	***-0.49****	0.03	***0.71****	***-0.83****	-0.41
	Wet	0.10	-0.11	-0.14	0.74	0.49	0.34	-0.34

* Significant correlation at α=0.05, for one tailed test

Table 4.8: Spearman rank correlation coefficients between sediment quality indicators and land uses using the buffer zone method.

Water Quality Indicator	Season	Land Use Type						
		Forest Cover	Corn	Hay / Silage	Pasture	Natural Vegetation	Residential	Hobby Farm
Copper	Dry	0.30	-0.19	***-0.45****	0.32	0.37	-	-0.70
	Wet	-0.24	-0.22	***-0.51****	-0.27	0.20	***-0.74****	-0.03
Lead	Dry	0.30	-0.19	***-0.45****	0.32	0.37	-	-0.70
	Wet	0.26	0.17	0.08	-0.15	0.00	-0.15	0.23
Zinc	Dry	0.28	0.17	-0.07	0.56	***0.71****	-	-0.26
	Wet	0.17	0.07	0.16	***0.74****	-0.05	-0.27	0.18
Manganese	Dry	0.17	***0.65****	***0.45****	0.03	0.34	-	0.81
	Wet	0.32	0.32	0.32	-0.29	-0.66	0.63	0.61
Iron	Dry	0.29	***0.66****	***0.55****	0.44	0.39	-	***0.90****
	Wet	0.27	***0.43****	***0.41****	-0.62	-0.31	0.51	***0.50****
LOI	Dry	-0.27	***-0.46****	***-0.53****	0.09	0.32	-	***-0.63****
	Wet	-0.10	0.01	-0.25	0.24	0.37	-0.30	-0.37
% of Sediment <0.063 μm	Dry	0.15	-0.03	***-0.52****	0.24	0.22	-0.80	0.40
	Wet	-0.05	-0.01	-0.27	0.56	0.17	-0.45	-0.31

*Significant correlation at α=0.05, for one tailed test

4.18.3 Contributing Area Land use and Water Quality Correlations

Water quality correlations using contributing areas were generally not observed to match correlations using buffer areas. Agricultural land uses did not appear to show any significant positive correlations for nutrients (Table 4.9). However, pastures and hazelnut farms showed a negative correlation with nitrate during the wet season. The most significant impact from agricultural land uses appears to be in pH levels. Corn, hay/silage, and fruit all had significant negative correlations with pH, while hazelnut fields had a positive correlation. The connection between land uses and pH is not precisely clear, however since pH is affected by surrounding soils, it may be connected more to the surrounding geology then agricultural land uses. As in the case of buffer land use correlations, turbidity did not appear to correlate with increased areas of corn and pasture. However, there was a significant positive correlation between turbidity and hay/silage fields. The fact that this observation occurs during the dry season when ground cover is at its peak indicates that increased algae growth is the likely result of the turbidity rather than the runoff transport of sediment and other particulates.

Overall, contributing area correlations showed that non-agricultural land uses had a significant impact on water quality. Forest cover was one of the few land uses which showed expected correlations using the contributing area method. In addition to nitrite, there were decreases in pH, specific conductivity and chlorophyll during the dry season. There was also a significant positive correlation with dissolved oxygen. All of these factors match expected correlations since most of the forest cover is present in the mountainous areas of the watershed. Runoff would be expected to have a lower pH and specific conductivity since much of the water travels over granite rock rather then through low land soils. In addition, a higher DO reflects the more turbulent nature of the mountain tributaries in addition to the lower nutrient concentrations.

Surprisingly, natural vegetation gave a significant positive correlation with ammonia and orthophosphate during the wet season. This could explain the resulting positive correlation with chlorophyll and pH due to increases in photosynthetic activity.

Table 4.9: Spearman rank correlation coefficients between water quality indicators and land uses using the contributing area method.

Water Quality Indicator	Season	Land Use									
		Forest Cover	Corn	Hay/silage	Pasture	Fruit	Hazelnuts	Uncultivated	Natural Vegetation	Residential Homes	Hobby Farm
Orthophosphate	Dry	0.06	0.12	0.18	0.31	0.32	0.33	-0.45	0.39	-0.06	-0.07
	Wet	-0.09	0.17	0.25	0.31	-0.43	0.20	-0.89	***0.48****	***-0.43****	-0.05
Nitrate	Dry	0.18	0.01	0.08	-0.37	-0.09	-0.28	-0.89	-0.10	***-0.47****	-0.06
	Wet	-0.10	-0.11	-0.07	***-0.45****	-0.40	***-0.38****	-0.89	0.05	***-0.46****	0.00
Nitrite	Dry	***-0.43****	-0.15	-0.25	-0.01	-0.56	0.18	***-0.94****	0.29	-0.17	-0.30
	Wet	-0.10	0.20	0.21	0.20	-0.32	0.26	0.00	0.31	-0.34	0.03
Ammonia	Dry	0.06	0.12	0.18	0.31	0.32	0.33	-0.45	0.39	-0.06	-0.07
	Wet	-0.09	0.17	0.25	0.31	-0.43	0.20	-0.89	***0.48****	***-0.43****	-0.05
Chlorophyll	Dry	***-0.55****	-0.18	-0.31	-0.05	-0.13	0.06	-0.45	0.55	***-0.45****	-0.36
	Wet	-0.03	0.22	0.27	0.22	0.43	0.29	0.00	***0.73****	0.21	0.15
DO	Dry	***0.35****	0.24	0.24	0.09	0.36	0.23	0.89	0.14	0.19	-0.09
	Wet	0.26	0.09	-0.03	0.29	0.23	***0.40****	0.45	0.37	0.52	**0.42***
Specific Conductivity	Dry	***-0.42****	-0.07	0.01	-0.01	-0.45	0.02	0.00	-0.13	-0.26	-0.23
	Wet	-0.19	0.14	0.28	0.10	-0.09	0.01	0.45	-0.05	-0.28	-0.07
pH	Dry	***-0.36****	***-0.38****	***-0.56****	0.17	***-0.85****	***0.39****	-0.89	0.07	0.37	-0.15
	Wet	-0.05	0.13	0.18	0.35	0.13	0.30	0.89	***0.66****	0.07	0.01
Turbidity	Dry	0.18	0.21	***0.43****	0.23	0.54	0.02	0.89	-0.17	-0.08	0.05
	Wet	0.18	0.08	0.28	0.41	0.20	-0.15	0.89	-0.26	-0.21	0.22

* Significant correlation at α=0.05, for one tailed test

Similar to correlations using the buffer, contributing area correlations show that residential areas result in a decrease in nitrate concentrations. There is also a negative correlation with factors such as orthophosphate and ammonia during the wet season. The matching decrease in chlorophyll during the dry season may be the result of overall nutrient reductions in these residential areas, primarily along Miami Creek.

4.18.4 Contributing Area Land Use and Sediment Quality Correlations

Contributing area correlations did not show that agricultural land uses were impacting sediment quality (Table 4.10). Negative correlations between corn and hay/silage fields with the percentage of sediment fines <0.063 μm indicates that runoff during the dry season does not add sediment to the watercourses. This is likely due to plant growth and root structures anchoring soil in addition to reducing runoff flow filtering sediment fines before they are allowed to enter watercourses. It was unexpected, however that no positive correlation was evident between corn fields and the clay/silt fraction during the wet season since it was common to see turbidity entering streams from corn fields during storm events. Another unexpected observation was the significant positive correlation between corn and iron concentrations in both the wet and dry season. It is difficult to determine the reason behind the correlation since iron is not a common component of fertilizers, manure, or pesticides. Due to the high concentrations of iron naturally present in the surrounding soils, the correlation may be due to top soil runoff or simply an anomaly based on natural changes in background concentrations of iron.

It is obvious that natural vegetation has the most significant impact on sediment quality within the watershed based on contributing area correlations. A positive correlation between LOI and natural vegetation are expected since plants may be contributing leaves and other woody debris to the watercourses. However, unexpected positive correlations were observed for copper, lead, zinc. This may be due to the higher concentration of organic matter in the watercourses during the dry season and its ability to bind metals. The correlation is not significant during the wet/winter season likely because much of the organic debris has been decomposed or flushed from the system.

Table 4.10: Spearman rank correlation coefficients between sediment quality indicators and land uses using the contributing area method.

Water Quality Indicator	Season	Land Use								
		Forest Cover	Corn	Hay / Silage	Pasture	Fruit	Hazelnuts	Natural Vegetation	Residential Homes	Hobby Farm
Copper	Dry	-0.06	-0.01	-0.18	-0.34	-0.32	0.22	***0.70****	0.40	0.05
	Wet	-0.21	-0.12	-0.21	-0.18	-0.23	0.30	0.31	-0.08	-0.05
Lead	Dry	-0.06	-0.01	-0.18	-0.34	-0.32	0.22	***0.70****	0.40	0.05
	Wet	-0.14	0.19	0.11	0.00	-0.41	0.03	0.31	0.06	0.20
Zinc	Dry	-0.03	0.23	0.01	0.05	0.03	-0.06	***0.62****	0.06	0.02
	Wet	-0.23	0.18	0.16	0.13	-0.41	-0.04	0.29	-0.09	0.14
Manganese	Dry	0.17	0.29	0.27	0.44	0.23	0.22	0.02	0.17	0.22
	Wet	0.24	0.13	0.15	0.21	-0.49	0.27	-0.46	0.26	0.32
Iron	Dry	0.05	***0.53****	0.40	0.58	0.32	0.23	0.04	0.16	0.22
	Wet	0.21	***0.35****	0.31	0.35	-0.01	0.28	-0.33	0.12	0.30
Loss on Ignition	Dry	-0.18	-0.25	-0.30	-0.52	-0.61	0.00	***0.55****	0.06	-0.24
	Wet	-0.11	0.04	-0.02	-0.14	-0.41	0.13	***0.57****	-0.01	0.01
% of Sediment <0.063 μm	Dry	-0.11	***-0.43****	***-0.52****	-0.37	-0.41	0.00	0.32	0.44	-0.16
	Wet	-0.10	-0.18	-0.28	0.19	-0.41	-0.08	0.24	0.30	0.15

*Significant correlation at α=0.05, for one tailed test

4.19 Water Quality Compared to Water Quality Criteria

In this study, emphasis is placed on provincial water quality guidelines since they are more reflective of the natural conditions and environmental goals of British Columbia. However, it should be noted, that enforcement of guidelines usually takes place under the *Fisheries Act*, which adheres to Canadian guidelines. It is also important to note that the long term and cumulative impacts of contaminated water and sediments are not necessarily taken into account for the determination of provincial and federal guidelines. Most toxicity tests are often no longer then 48-96 hours and are conducted under controlled conditions. As a result, elevated concentrations which are below acceptable levels may still be having a significant impact on the aquatic system, particularly if, as is often the case, other contaminants and environmental stressors are present.

Overall water quality within the AHHS watershed appears to be within most provincial water quality guidelines (Table 4.11). Median concentrations of nitrate and ammonia were well below provincial guidelines at all sampling points throughout the watershed. This includes compliance of all field measurements taken throughout the sampling program. However, nitrite levels were in exceedence of BC Water Quality Guidelines at a number of locations (Table 4.11). The most prominent of these is the stormwater outfall station which exceeded maximum allowable concentrations during both seasons.

In McCallum Slough, the exceedence in nitrite at stations D5 and D4 was associated with a period of particularly intense runoff during a week of manure application. Figure 4.34, shows the percentage of samples at each of these stations which exceeded BC Water Quality Guidelines for nitrite. The outfall station shows the highest percentage of samples above provincial guidelines, particularly in the wet season where 50% of all samples are out of compliance.

Table 4.11: Sampling stations within the AHHS Watershed exceeding provincial water quality guidelines and Canadian Water Quality Guidelines for the Protection of Aquatic Life during the dry and wet seasons (BCMELP, 2002).

Water Quality Parameter	BC Water Quality Guidelines	Sites Above or Below BC Water Quality Guidelines	Canadian Water Quality Guidelines For the Protection of Aquatic Life	Sites Above or Below Canadian Water Quality Guidelines
Orthophosphate	No criteria	-	No criteria	-
Nitrate	≤40 mg/L (avg) 200 mg/L (max)	None	No criteria$^{\psi}$	None
Nitrite	0.02 mg/L (avg)$^{\gamma}$ 0.06 mg/L (max)$^{\gamma}$	Dry Season – I1*, K2*, F2^{+}, Outfall^{+} Wet Season – D1*, D4*, D5^{+}, G1*, H2*, H1*, Outfall^{+}	0.06 mg/L	Dry Season – F2, Outfall Wet Season – D5, D4, Outfall
Ammonia (Total)	16.5-27.5 mg/L$^{\beta}$	None	3.37-153 mg/L$^{\beta}$	None
DO	5.0 mg/L (adult / juvenille life stages)	Dry Season – All stations below criteria except C6, C0, Control, D1 and H4 Wet Season - D5, D4, I1, J1, and Agassiz Slough (all sites)	6.5 mg/L (other life stages)	Dry Season – All stations below criteria Wet Season - All stations below criteria except F2, C1, C5, C6, C0, Control and A1
	9.0 mg/L (buried embryo / alevin life stage)	Dry Season – All stations Wet Season – All stations except C6, C0 and control	9.5 mg/L (early life stages)	Dry Season – All stations below criteria Wet Season – All stations except control

β – Values determined using ranges of temperature and pH observed within the AHHS watershed
γ – When chloride is less then 2 mg/L
* - Stations above average criteria value
+ - Stations above maximum instantaneous criteria value
ψ – Concentrations which stimulate weed growth should be avoided

Table 4.11 (Cont'd): Sampling stations within the AHHS Watershed exceeding provincial water quality guidelines and Canadian Water Quality Guidelines for the Protection of Aquatic Life during the dry and wet seasons (BCMELP, 2002).

Water Quality Parameter	BC Water Quality Guidelines	Sites Above or Below BC Water Quality Guidelines	Canadian Water Quality Guidelines For the Protection of Aquatic Life	Sites Above or Below Canadian Water Quality Guidelines
pH	6.5-9.0	Dry Season – All stations below criteria except K2 Wet Season – All stations below criteria	6.5-9.0	Dry Season – All stations below criteria except K2 Wet Season – All stations below criteria
Temperature	12^0C (incubation maximum for fall and spring)	Fall Season – All stations above criteria Spring Season – B1, G1, D1, K2, K1, Westlin Ditch (all sites), Hogg Slough (all sites), Clark Ditch (all sites)	No applicable criteria	-
	19^0C (maximum daily temperature)	Dry Season – B1, K1, Hogg Slough (all sites) Wet Season – All stations below criteria	No applicable criteria	-

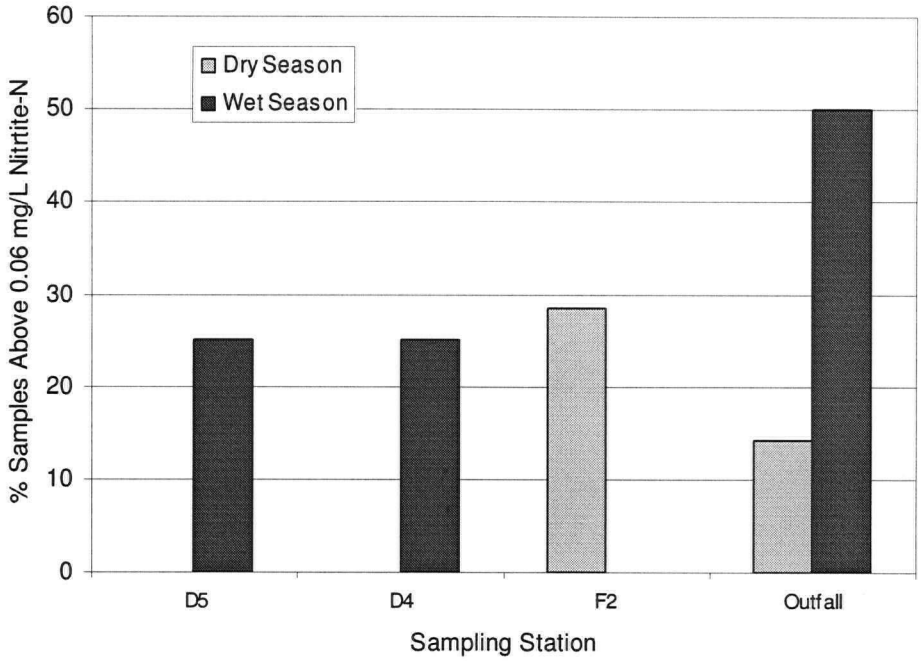

Figure 4.34: Percentage of samples above BC Water Quality Criteria for nitrite-N at stations throughout the AHHS Watershed.

There were also problems in the compliance of provincial guidelines for DO. The majority of sampling stations within the watershed had at least one instantaneous sampling DO concentration below the provincial criteria of 5.0 mg/L, which represents the minimum DO level required for adult and juvenile fish. The problem is especially apparent for Agassiz Slough, which dropped below 5.0 mg/L in both the dry and wet seasons. Although this may be a reflection of the slow flowing, stagnated nature of the watercourse, the input of nutrients from the storm sewer outfalls is likely contributing to the low DO levels. All of the stations fell below the minimum requirement of 9.0 mg/L of DO for buried embryo development at least once during the course of sampling. Based on the data obtained, the only stations which could support embryonic fish development are C6, C0 and the control. However, this is only during the wet/winter season between October and March.

A clearer picture of DO levels can be observed in Figure 4.35, which shows the frequency at which sampling stations were out of compliance with BC guidelines for DO.

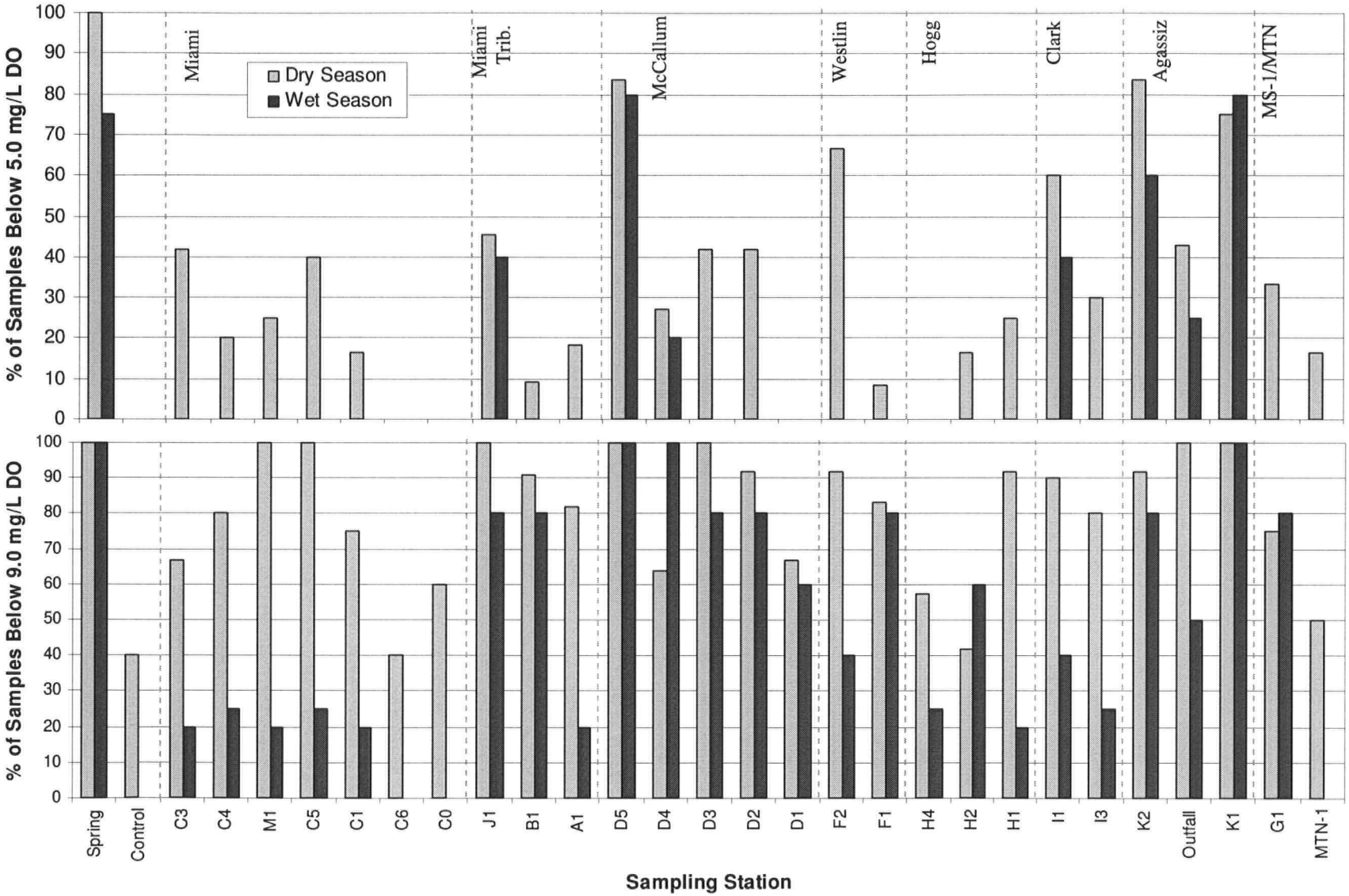

Figure 4.35: Percentage of samples below BC Water Quality Criteria for dissolved oxygen at stations throughout the AHHS Watershed.

From this figure it is obvious that Agassiz Slough is the most impacted watercourse. McCallum Slough is also impacted with the most oxygen deficient station being D5, which also shows high nutrient concentrations. This appears to be having a detrimental impact along the length of the watercourse, although the oxygen deficiency is less pronounced further downstream. Overall, it appears that DO levels are closer to acceptable levels during the wet season then the dry season, particularly in Miami Creek.

Temperature results (Table 4.11) show that there are a number of stations which had higher temperatures then current provincial guidelines. None of the sampling stations in the AHHS watershed were below the maximum temperature for the incubation of fish embryos during the fall season. Overall, it appears that Miami and Hot Springs Creek are the only watercourses which are able to maintain temperatures below the 12^0C incubation guideline, and only during the spring season. All stations within Hogg Slough (H4, H2, and H1) in addition to stations K1 and B1 appear to be areas of relatively high temperatures, with levels above the daily maximum during the dry/summer season. As expected, all stations were in compliance of the 19^0C maximum during the wet/winter season. The relative number of exceedences of the 19^0C and 12^0C guidelines is shown in Figure 4.36 and 4.37. As stated earlier, data should be used with caution since temperatures vary slightly depending on the time of day of sampling. However, results show watercourses within the AHHS watershed are limited in their ability to support the incubation of salmonid embryos. In addition, it is evident that Hogg Slough, represented by stations H4, H2 and H1, is likely not able to support a viable fish population based on its high temperatures.

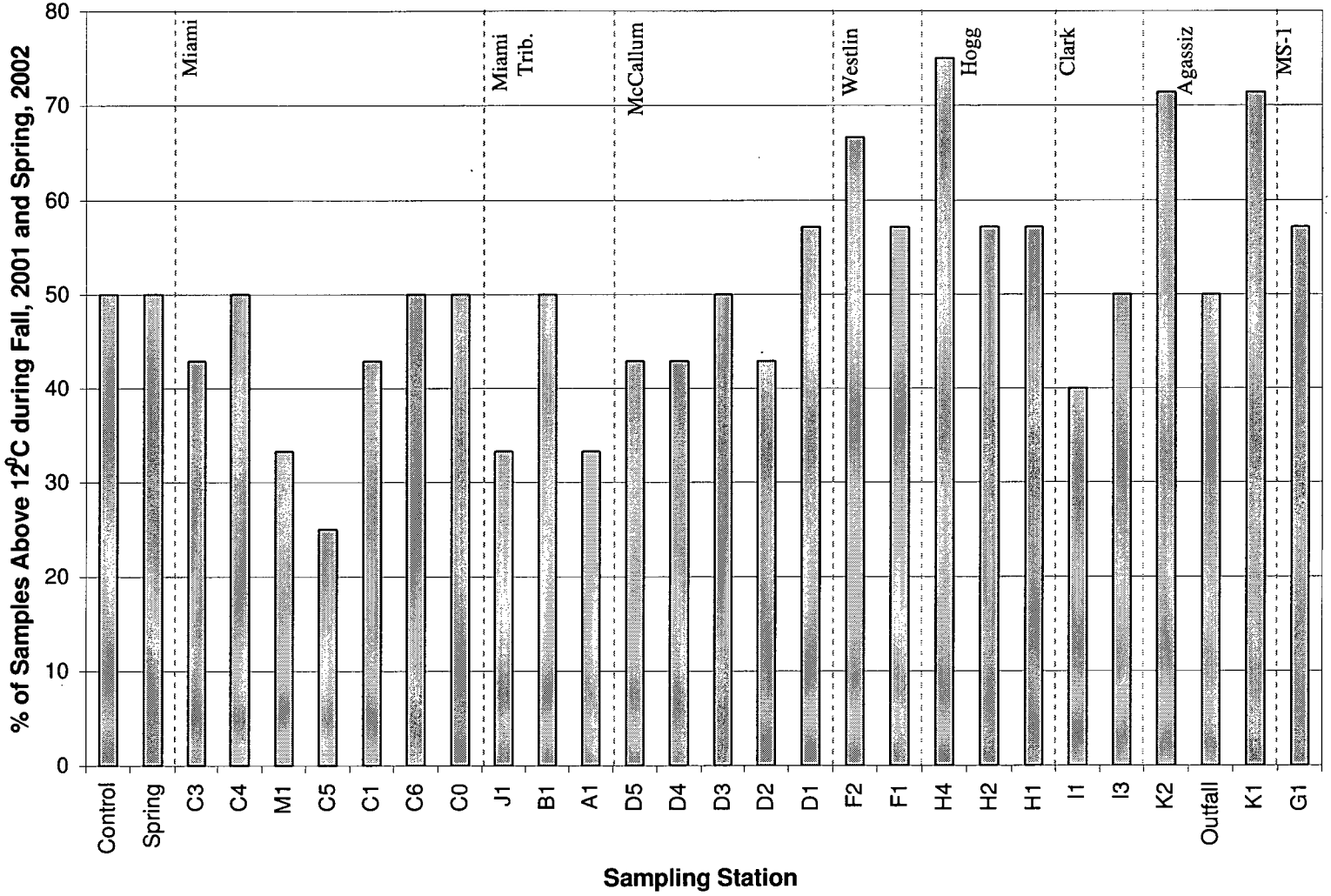

Figure 4.36: Percentage of water samples above BC Water Quality Guideline for temperature for the incubation of fish during the spring and fall.

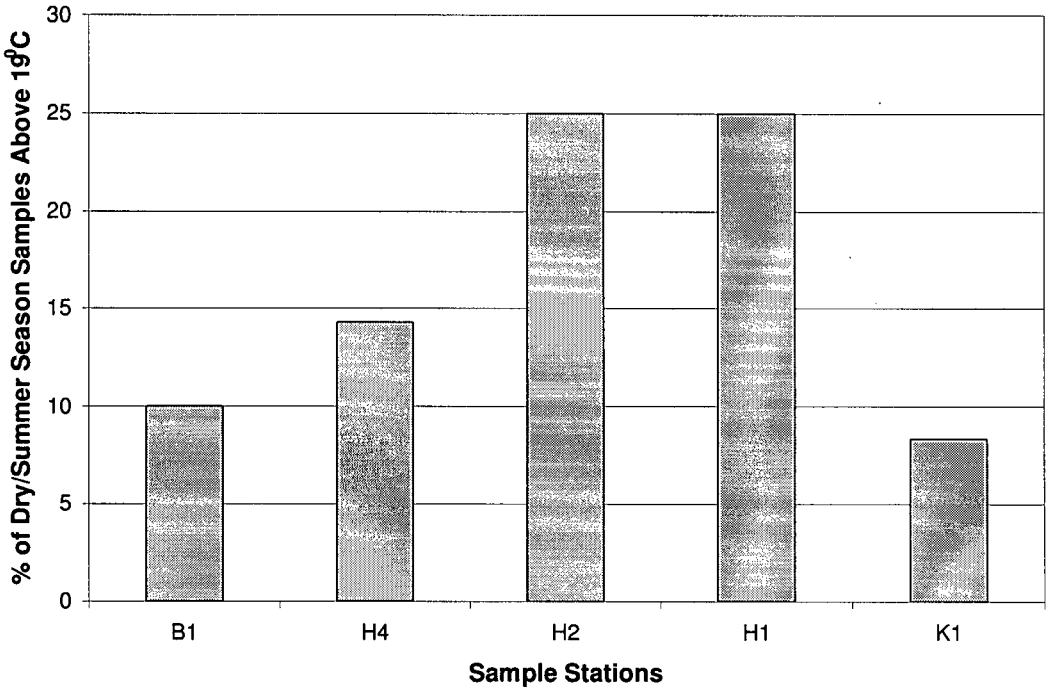

Figure 4.37: Percentage of water samples above BC Water Quality Guidelines for daily maximum temperature for the protection of freshwater fish at stations throughout the AHHS Watershed.

Almost all the sampling stations were out of compliance with provincial standards for pH (Table 4.3). Although some stations were able to maintain median pH levels within provincial guidelines, no station was consistently between the required guidelines during the course of the sampling program. It is assumed that fluctuations in pH were more likely the result of the surrounding geology and water chemistry rather then a result of adjacent land uses.

In terms of overall compliance with the water quality variables examined, Miami and Hot Springs Creek are the best watercourses for aquatic life, while Agassiz and Hogg Slough are the most degraded.

4.20 Sediment Quality Compared to Sediment Quality Criteria

There are a few areas within the AHHS watershed which are above BC sediment quality guidelines for metals. The most obvious is at the outfall station outside a stormwater sewer servicing Agassiz. High concentrations of copper above the severe effect level (SEL) likely points to some toxicity in these sediments, particularly for fish. This may also be the case for station H4 which also has high copper levels, although below SEL concentrations. Copper concentrations were above LEL levels for all other stations, including the control, and are assumed to be due to natural background levels. Concentrations of lead and zinc were also above LEL levels at many stations, however it is not known if this is having an impact on the aquatic system. If metal concentrations were impacting the aquatic ecosystem, it would most likely be observed in Agassiz Slough which maintained the highest overall concentrations. Natural background concentrations in sediments and the local geology likely played a role for high concentrations of manganese and iron. Iron, in particular had concentrations considerably higher then the severe effect level (SEL) of provincial guidelines.

Overall, it appears that the wet season results in higher metal concentrations, and is thus a time of most concern in terms of environmental compliance (Table 4.12).

Table 4.12: Sampling stations within the AHHS Watershed exceeding provincial sediment quality guidelines (BCMELP, 2002) and Canadian Water Quality Guidelines for the Protection of Aquatic Life (CCME, 2001) during the dry and wet seasons.

Sediment Quality Parameter	Provincial Guidelines (mg/kg)	Stations Above BC Sediment Quality Guidelines	Canadian Sediment Quality Guidelines (mg/kg)	Stations Above Canadian Sediment Quality Guidelines
Copper	16 (LEL)	Dry Season - All stations above guideline Wet Season - All stations above guideline	35.7 (ISQG)	Dry Season - All stations above guideline except D3, D1, F1, G1 and Spring Wet Season - All stations above guideline except D1 and F1
	110 (SEL)	Dry Season – Outfall Wet Season – I3 and Outfall	197 (PEL)	Dry Season - All stations below guideline Wet Season - All stations below guideline
Lead	31 (LEL)	Dry Season – C1, J1, D5, D4, D1, F2, H2, I3 and Agassiz Slough Wet Season – All stations above guideline except C0 and Control	35 (ISQG)	Dry Season - D5, F2, H2, I3, K1, K2 and Outfall Wet Season - All stations above guideline except Control, CO, D1 and J1
	250 (SEL)	Dry Season - All stations below guideline Wet Season - All stations below guideline	91.3 (PEL)	Dry Season - K2 and Outfall Wet Season - K1 and Outfall
Zinc	120 (LEL)	Dry Season – F2, H2, I3, Outfall and K1 Wet Season – C5, A1, D4, D2, F2, F1, H2, H1, I3 and Agassiz Slough (all stations)	123 (ISQG)	Dry Season - F2, H2, I3, Outfall and K1 Wet Season - A1, C5, D4, D2, F2, F1, H2, H1, I3, K2, K1 and Outfall
	820 (SEL)	Dry Season - All stations below guideline Wet Season - All stations below guideline	315 (PEL)	Dry Season - Outfall Wet Season - Outfall
Iron	2100 (LEL)	Dry Season – All stations above guideline Wet Season – All stations above guideline	no criteria	-
	4380 (SEL)	Dry Season – All stations above guideline Wet Season – All stations above guideline	no criteria	-
Manganese	no criteria	-	no criteria	-

5. Discussion

5.1 Relevant Land Use Determination

Within the AHHS watershed, the use of watercourse buffers to correlate land uses with water quality impacts is a more effective technique then the use of contributing areas. Although both techniques were able to make significant positive and negative correlations between land uses and water quality, contributing area correlations did not correspond well with known impacts of agricultural land use. The use of the buffer technique, however, resulted in a greater number of expected correlations.

There are a number of reasons why the buffer technique resulted in superior correlations. The most likely is due to the lack of topographic relief in the surrounding lowlands. The flat surface generally does not encourage extensive runoff, and thus only the land adjacent to the watercourse likely makes a significant impact on water quality. Although smaller tributaries may collect runoff from land outside the designated buffer region, it is apparent that the overall impact is not significant.

A limitation inherent in both techniques, which may have been enhanced in the contributing area method, is the inability to quantitatively distinguish land use distances from sampling points and their relative importance. This shortcoming was also observed within the Salmon River Watershed (Cook, 1994). Each measurement in correlation analysis is treated an independent entity and spatial distances are not taken into account. For example, there is no distinction between a land use located adjacent to a sample station and a land use located at the headwaters of the watercourse, even though the adjacent land use likely has a more significant impact on the water quality at the station. The effect is exacerbated at stations near the mouth of the watercourse since they are located at the most downstream point. Unfortunately it is difficult to determine a numerical value to distinguish relative importance based on distance. Conditions are constantly changing within the watershed, and each contaminant acts differently depending on its chemical and physical properties. In this sense, the buffer technique is useful in that it minimizes the incorporation of irrelevant land uses included in the correlation technique.

5.2 Impacts of Agricultural Land Uses on Water Quality

5.2.1 Nutrients

Through the use of the buffer correlation technique, it is clear that grazing pastures significantly increase nutrient concentrations in watercourses throughout the AHHS watershed. This was expected since similar associations have been made in other studies of agricultural watersheds (Stone *et al.*, 1998). The most obvious example of this is in Hogg Slough. The H4 sampling station, which is the most upstream location, shows relatively low or average concentrations of nutrients. However, the water quality just downstream from this location, at site H2, shows high concentrations of nitrate, nitrite, ammonia and orthophosphate, corresponding to an increased area of grazing pastures. The high variability of H2, particularly during the course of the wet season, supports the assumption that there is a nutrient influx from runoff sources. This influx has resulted in wet season nitrite concentrations above acceptable levels at stations H2 and H1. However, it is important to note that these high nutrient levels may come from runoff where manure is stored rather than from the defecation of animals in adjacent fields. The observation that nutrients were more pronounced during the wet/winter season, when animals were more likely to be kept inside, confirms this possibility. High nutrient concentrations are obviously carried on to station H1, which also shows high concentrations of nutrients and increased variability.

Other studies within the Lower Fraser Valley did not appear to make strong correlations between combined nitrate and nitrite using the contributing area method. For example, Cook (1996) only found a significant correlation with poultry and fur bearing animals, while Wernick (1996) did not find any significant correlations.

Station D5 (McCallum Slough), and to a lesser extent, C3 (Miami Creek), also seem to be areas of fairly significant nutrient accumulation. Both these sampling sites are within the same general area and surrounded by hay/silage fields and corn. It is assumed that the spreading of manure on adjacent land is having an effect on nutrient concentrations at these sampling sites, particularly at station D5 where low DO levels likely contributed to nitrite concentrations above provincial guidelines during the wet season. Although nutrient concentrations may, in fact, come from fertilizers, the presence of high sediment

copper levels above background concentrations (Table 2.5) indicates that the source is most likely swine manure. As mentioned earlier, copper is an important supplement in pig feed (Hsu and Lo, 2000) and will often appear within manure and the resulting contaminated runoff.

Corn crops did not appear to have any impacts on nutrient concentrations within the watershed. Although there has been a significant body of research connecting corn fields and manure use to water quality degradation (Hargrave, 1995; Zebarth and Paul, 1997; Eghball and Gilley, 1999; Burkart and Stone, 2002; Muhammetoglu et al., 2002), it does not appear to have made a clear impact during the course of sampling within the AHHS watershed.

In general, nutrient concentrations, aside from orthophosphate, were found to be both elevated and more variable during the wet season. The likely reason for this is the increase in manure application during the fall season. The greater amount of rainfall combined with the exposed soil gives a greater potential for nutrient rich runoff to enter adjacent watercourses. There was also a significant amount of manure application during the spring, however it is thought that the greater amount of vegetative cover combined with the increased assimilative capabilities within the biotic community keep nutrient concentrations in soil at lower levels (Lapp *et al.*, 1998; Magner and Alexander, 2002).

The control and spring sampling stations consistently showed low levels of nutrients, verifying that nutrient sources throughout the floodplain were not from groundwater or surrounding highlands where there has been no agricultural development. The low nutrient levels observed at the spring station during the course of the monitoring confirm that groundwater contamination was not affecting surface water quality in this particular area.

5.2.2 Chlorophyll

High chlorophyll concentrations were well correlated with orthophosphate, nitrite and ammonia concentrations in watercourses, particularly in the summer (Appendix D). Agricultural land uses which produce significant nutrient runoff, such as grazing pastures, can thus result in increases in algae growth. In Hogg Slough for example,

increasing chlorophyll concentrations are observed with increasing concentrations of orthophosphate. In addition, station D5 (McCallum Slough) showed high levels of chlorophyll corresponding to its high median nutrient concentrations assumed to come from manure runoff.

In addition to nutrient concentrations, it is apparent that there are a number of different factors affecting overall chlorophyll levels. High chlorophyll concentrations at D4 were somewhat unexpected as nutrient levels were found to be quite low in this area except for nitrate. This may be due to residual chlorophyll concentrations from the upstream location, station D5, travelling downstream. The same process can explain station I3 (Clarke Ditch), where residual concentrations from station I1 are most likely having an impact on median chlorophyll levels. However, it is difficult to explain high chlorophyll concentrations at station F2 (Westlin Ditch), since nutrient levels were generally low at this site. Turbidity and chlorophyll showed positive correlations with one another (Appendix D), indicating that much of the turbidity within the watercourses may be the result of algae.

Both the spring and the control maintained low concentrations of chlorophyll year round, which would be expected due to the oligotrophic conditions present at each station.

5.2.3 Turbidity

Turbidity did not show any significant positive correlation with agricultural land uses. This includes a lack of significant correlation between turbidity and pasture areas, even though previous research has shown that pastures areas generally contribute considerable turbidity due to the erosion of land and stream banks from animals (Chandler and Walter, 1997). However, the correlation of 0.74 during the wet season, although not statistically significant, signifies that there still may be a connection between the turbidity and pastures. Turbidity values were generally quite variable, especially since samples were taken on days with conditions ranging from substantial run off to no runoff at all. A positive correlation may have been present if samples were restricted only to times of significant runoff, verified by the higher values obtained during the wet season in Hogg Slough.

Turbidity correlations for the wet season were generally more positive then the dry season, however no statistically significant positive correlations could be made for any particular agricultural land use. Signficant negative correlations of turbidity with hay/silage and corn fields during the dry season may be explainable by the presence of vegetation, not present during the wet season, which stabilizes soil and acts to slow overland flow of runoff (Nerbonne and Vondracek, 2001). Visual observations of turbidity coming from cornfields during rainfall events in the wet/winter season points to the strong possibility the vegetation reduces the amount of turbidity in runoff.

The presence of forested areas and natural vegetation did not appear to have any significant negative correlations with turbidity within the watercourses. Forest cover and natural riparian areas should show a negative correlation with turbidity by their ability to increase bank stability and act as a filter for runoff entering into watercourses (Rier and King, 1996; Chandler and Walter, 1997; Nerbonne and Vondracek, 2001). However, it is important to note that correlations may have been affected by a number of factors which could not be integrated in the correlation technique. For example, riparian and forested areas would not have a significant impact on turbidity if the source of the turbidity is at an upstream site with little riparian protection. In areas such as Miami Creek and Hogg Slough, most of the natural growth and forested areas are located near the end of the watercourse, and thus an appreciable effect on turbidity would not be expected. In addition, it was observed that there was a positive correlation between turbidity and chlorophyll *a*. It is possible that the majority of the turbidity was the result of algae rather then sediments that could be screened out by riparian growth.

It is obvious that the storm sewer outfall has the largest response to runoff events and creates the greatest variability in turbidity. This has also been confirmed both by the data obtained and visually (Appendix A). Compared to similar studies of stormsewer outfalls in the Brunette River Watershed (Macdonald *et al.*, 1997), the range of turbidity values outside of the Agassiz outfall appear to be considerably higher, particularly during the wet season and storm events.

5.2.4 Dissolved Oxygen

Although DO levels were often below provincial water quality guidelines it did not appear to have any strong correlations with any particular agricultural land use. Better correlations may have been possible if the time and location of manure and fertilizer application was known during the course of the monitoring. However, areas of high nutrient accumulation such as Agassiz Slough and station D5 (McCallum Slough) showed expected decreases in DO, verified by results of percent DO saturation for each station. The low DO likely contributed to increases in nitrite and ammonia concentrations at these sites. Hogg Slough is an exception to this observation with high nutrient concentrations at H2 and H1 corresponding to high DO levels. However, it is though that the high DO levels upstream at site H4, as a result of photosynthetic activity, may be ameliorating impacts further downstream. It is interesting to note that DO did not have a significant correlation with nitrite or ammonia concentrations within the watershed (Appendix D). However, results may have been skewed due to Hogg Slough and the outfall station, which had relatively high nutrient concentrations combined with high dissolved oxygen concentrations.

As expected, the control station had high DO levels, likely due to the low nutrient concentrations combined with the turbulent nature of the mountain tributaries feeding into the site. Also, as expected, the spring station gave expectedly low DO concentrations due to groundwater infiltration.

It is important to note that at the majority of stations within the watershed, DO levels consistently fluctuated above and below minimum requirements for aquatic life. This means that almost all the stations surveyed were unable to support a stable aquatic ecosystem year round. This is especially important for fish, where even one day below minimum requirements could kill a significant portion of the population. Based on DO results, the only watercourse which could adequately support a stable fish population would be Miami Creek during the wet/winter season.

5.2.5 pH

Although there is a positive correlation with hay/silage fields, the most likely determinant of pH in the watershed is the surrounding geology and soil. If this is the case, land use

inputs would be buffered by the natural background pH levels and would have a significant effect on the water quality.

The impact of natural background concentrations can be seen in the differences between the control and spring stations. Water at the control site comes from water which flows over granite bedrock and bedrock derived soils. As a result, the lack of buffering capacity results in a much lower pH. On the other hand, the spring sampling station represents water which has percolated through the more basic lowland soils, resulting in a higher overall pH. Photosynthetic activity, reflected by chlorophyll concentrations, is likely the reason for higher overall pH values during the dry/summer period. The high chlorophyll values during the summer indicate increased photosynthetic activity which results in the removal of CO_2 from water, affecting alkalinity.

5.2.6 Specific Conductivity

Specific conductivity did not correlate with any specific agricultural land use. Surprisingly, corn fields, which are known to be fertilizer and pesticide intensive, did not show any expected correlations to conductivity. Overall changes in specific conductivity within the AHHS watershed are likely due to natural sources as dissolved solids enter the water from surrounding soils and sediment. A similar situation was noted in the Salmon River watershed, particularly during low flow conditions in the dry season when groundwater has a greater impact (Cook, 1994; Wernick, 1996).

5.3 Impacts of Agricultural Land Uses on Sediment Quality

5.3.1 Metals

Correlations using the buffer and contributing area techniques showed that pastures had a significant correlation with concentrations of zinc. As previously mentioned, small concentrations of zinc are present in animal feed as a mineral supplement in the form of zinc oxide or zinc sulphate. The resulting manure can become a possible non-point pollution source for zinc. The presence of this correlation during the wet season indicates that it is likely due to manure contaminated runoff from grazing pastures and farms. Zinc was also found to correlate well with agricultural land uses in the agricultural

watercourses of the nearby Sumas River Watershed, although correlations with specific agricultural land use types and metals was not conducted (Berka, 1996).

The majority of the correlations between agricultural land and metals in sediment did not show expected results. It is thought that manure and fertilizers used on hay/silage fields and corn would produce a positive correlation for copper and zinc from fertilizers and pesticides, however this correlation was not evident. A negative correlation with lead along with a positive correlation with manganese and iron is equally unexpected for corn and hay/silage fields. Correlations may have been affected by limitations in the correlation technique previously discussed. Data may also be skewed by natural anomalies in metal concentrations of soils and sediments as has been observed in the Sumas River Watershed within the Lower Fraser Valley (Berka, 1996).

Based on previous data taken within the Lower Fraser Valley, it is obvious that high levels of manganese and iron are present naturally in soils and sediments within the AHHS watershed (Cook 1994; McCallum 1995). However, manganese concentrations may be supplemented by inputs from automobile fuel near major roadways (McCallum, 1995). This would explain slightly elevated concentrations at station B1 and I3. Runoff from impermeable surfaces would also explain elevated levels observed near the storm sewer outfall in Agassiz Slough. It is not likely that high iron and manganese levels are having an impact on the aquatic community, since it does not appear to deviate significantly from background concentrations.

5.3.2 Sediment Properties

Corn and hay/silage fields did not appear to contribute significant amounts of organic matter to adjacent watercourses. This may indicate that there is not a significant influx of manure from these agricultural land uses. However, positive correlations for LOI were obtained for stations which had forest and natural vegetation upstream from the sampling sites. Although not statistically significant, it is possible that natural riparian zones contribute more organic matter to the watercourses then corn and hay/silage field. This would show an overall decrease of organic matter in sediments as watershed development moves toward agriculture.

There was no significant correlation between corn fields and the clay/silt fraction of sediments. The observation is confirmed by the lack of correlation between the clay/silt fraction and turbidity during the wet season (Appendix D). The absence of a significant correlation was somewhat unexpected since, in many cases, turbidity from sediment was observed flowing from corn fields into adjacent watercourses during storm events. Differences of flow in the various watercourses may explain why no significant correlation could be made since areas of more robust flow would result in a smaller clay/silt fraction. In addition, there is a high amount of variability between sampling locations, and even between the same sampling sites during different times of the year. This may make any correlations with the clay/silt fraction difficult to observe.

5.4 Natural Variables

5.4.1 Storm Events

Storm events generally resulted in more variable data, as can be observed from the increased variability between the wet and dry seasons. However, the most obvious impacts can be observed at the sampling stations along Miami Creek, Hogg Slough, and Agassiz Slough. Turbidity, nitrate, nitrite, ammonia and orthophosphate levels were also generally elevated in samples taken during runoff periods, particularly at the outfall station. The most obvious was on November 1, 2002, when there was considerable runoff after a period of extensive manure application. Station D5, in particular, showed spikes in nitrite, nitrate, specific conductivity and turbidity, while Hogg Slough showed spikes in nitrate concentrations (Appendix B).

However, not all sampling stations and watercourses responded in the same manner to storm events. It is assumed that this was the result of different land uses adjacent to each watercourse, in addition to specific agricultural activities taking place upstream from each station. Runoff events in agricultural watersheds are often the primary mechanisms of turbidity and nutrient enrichment in surface water (Daniel *et al.*, 1994), and it is likely that additional sampling during runoff events would provide a clearer picture of storm event impacts.

5.4.2 Groundwater Infiltration

Groundwater effects are quite prominent throughout the AHHS watershed. Sites of groundwater emergence range from permanent locations, such as the spring station, to temporary groundwater upwellings in Hot Springs Creek and McCallum Slough. As can be seen from the water chemistry at the spring station, groundwater has the ability to significantly alter specific conductivity, DO, and temperature within watercourses.

Hammersly Prairie is an area of considerable groundwater emergence, particularly in Westlin Ditch between stations F2 and F1. Station F1, which is downstream from station F2, has a higher specific conductivity, lower DO, and less variable temperature range. Temperature variations are especially apparent from December, 2002 to March, 2002 where station F2 has a considerably higher temperature then F1. It is assumed that there is also groundwater influence of the MS-1 Ditch since the water chemistry between the spring station and G1 tend to be quite similar.

5.5 Moss Bioassay Performance

Results from the moss bioassay experiment show a strong possibility that there are detrimental impacts to aquatic plants at sampling site H2. Unfortunately bioassays do not allow determination of the specific causes of the growth inhibition, however water quality data points to the possible toxic response from high concentrations of zinc and nitrite, in addition to decreased light penetration from turbidity. The presence of pesticides other than atrazine may also be the cause of differences in the average concentration of chlorophyll *a* and *b* in addition to growth inhibition. The similar results obtained for site replicates confirm that this technique did not produce significant variability.

Chlorophyll *a* and *b* concentrations did not show any significant differences between sites. Given the variability of recorded chlorophyll levels, it does not appear to be an appropriate measure of possible environmental impacts to aquatic organisms. However, moss dry weights may provide a possible indicator to water quality. Although it was quite variable, the moss dry weight appeared to increase at those sights with the highest nutrient concentrations. Station H2, D5, C3 and I3, which generally showed high nutrient concentrations, all showed relatively higher average dry weights. This may

indicate that dry weights are sensitive enough to show nutrient increases within watercourses.

Although the moss bioassay did work for Hogg Slough, there are many indications that the technique may not be suitable for detecting more subtle changes in agricultural watercourses. The variable nature of the watercourses makes it extremely difficult to ensure similar environmental conditions between sites. For example, flows at the various stations tended to differ significantly, resulting in different rates of exposure to contaminants and causing variations in growth. Another important variable was the presence of extensive periphytic growth on the moss shoots and mesh bags holding the moss stems. Periphyte growth is quite prominent in many of the watercourses, varying from moderate to severe. In some instances, the moss stems were almost totally inundated with periphytic growth (Appendix A). This growth reduces the amount of light reaching the moss stems, which would explain the high variability of chlorophyll *a* and *b* observed. Periphytic growth may also cause changes in nutrient and DO concentrations around the moss stems. The inhibition of growth and chlorophyll production is a reflection of complicated metabolic processes within the moss stems which are sensitive to changes in environmental quality (Davies, 2002). However, with environmental conditions between sampling sites showing considerable variability, it is likely that the moss *Fontinalis antipyretica* is not adequate for detecting low concentrations of contaminants and overall water quality in the agricultural watercourses of the AHHS watershed.

A more appropriate measure of environmental quality may be the use of native periphyton within the watercourse. Small plates can be put at sampling stations to encourage periphytic growth, which can then be measured for chlorophyll. This technique would minimize some of the disadvantage of using the *Fontinalis antipyretica* moss stems in the field. Another option could be to bring water samples into the laboratory and conduct controlled experiments with *Fontinalis antipyretica* to limit site variability.

5.6 XAD-7 Adsorption Efficiency

Field observations showed that atrazine concentrations, determined by adsorption to XAD-7 resins, had no significant correlation with corn fields where atrazine is commonly used. Results may have been slightly skewed due to the high concentrations of atrazine present in Westlin Ditch, which has a relatively small area of corn coverage within its 100 m buffer. Due to the long use of atrazine in the watershed, it is possible that background concentrations may be affecting correlations with corn fields. Background concentrations within agricultural watershed streams have been found to range from 0.1 – 30.3 μg/L (Lakshminarayana *et al.*, 1992; Dodson *et al.*, 1998). In addition, atmospheric deposition of atrazine in AHHS watercourses on the order of 3.35 $\mu m/m^2$ (Belzer *et al.*, 1998) likely played a significant role in the concentrations recovered by the XAD resins. Unfortunately, a survey of air borne concentrations of atrazine was beyond the scope of this experiment. However, there is a large area of corn fields near the control site which were sprayed with atrazine during the course of monitoring (Boyes, pers. comm., 2002). The observation of relatively high atrazine levels at the control station verifies that wind blown transport of atrazine likely contributed significant amounts to other watercourses within the watershed.

There is some question as to the applicability of the XAD adsorption technique. Previous research has shown that XAD-7 resin should give better recoveries than actually observed in the benchscale experiment (Junk *et al.*, 1974). However, previous experiments involved the direct application of water samples onto XAD-7 resin surfaces. In this case, the apparatus requires that atrazine first diffuse through a membrane before coming into contact with the XAD, providing a possible rate limiting step to the adsorption process.

Recoveries of atrazine using XAD-7 appear to be inhibited by organic material present in the water. As stated previously, atrazine can bind tightly to colloids and other particulates such as algae and other organic matter (Hall *et al.*, 1993), inhibiting adsorption to the XAD resin. This observation is confirmed by the benchscale experiment where adsorption differences between distilled and sample water were quite significant. Organic compounds less then the 6000-8000 MWCO of the membrane may have also sorbed onto the XAD resin, taking up possible adsorption sites and reducing overall adsorption efficiency.

In this study, the use of XAD-7 resins does not appear to be a useful technique to determine relative atrazine concentrations in agricultural watercourses in the AHHS watershed. As with the moss bioassays, there is significant variability between sites. Differences in channel flow, for example, make it difficult to compare results. Extensive periphytic growth on the membranes at sampling stations is also a problem (Appendix A). Not only does this decrease the surface area in contact with water, but it also results in the fouling of the membrane. A more useful method for atrazine recovery may be to obtain samples in the field and directly apply the sample to an XAD-7 resin column as used in previous studies (Junk *et al.*, 1974; Baun and Nyholm, 1996). However, to minimize the limitations of point samples, sampling should be conducted during the first runoff events after pesticide application.

5.7 Impacts of Non-Agricultural Land Uses

5.7.1 Storm Sewer Outfalls

The most prominent non-agricultural impacts within the AHHS watershed are from the storm sewer outfalls located along the banks of Agassiz Slough. Results obtained from the outfall sampling station show significant increases in the median concentrations of ammonia, nitrate, nitrite, orthophosphate, specific conductivity and turbidity. Nitrite concentrations are of particular concern since levels are above both provincial and federal guidelines. It is likely that toxic concentrations are present in these areas. Contaminant sources are assumed to be impermeable surfaces, which include residential areas, municipal roads and highways. It is assumed that higher concentrations of chlorophyll and low DO levels result from the outfall discharges, although these variables may also be attributable to the slow flowing nature of the watercourse. Changes in pH do not appear to persist very far downstream from the outfall location, most likely due to the buffering effect of the water and surrounding soils.

The extremely slow flows within Agassiz Slough allow more time for the attenuation of adverse impacts, however results from sampling stations K2 and K1 indicate that there is still significant water degradation and variability downstream from outfall locations. Nutrients, for example, appear to maintain high concentrations over relatively long distances.

The most significant impacts of the storm sewer outfalls appears to be from the elevated concentrations of copper, lead and zinc in the sediments immediately surrounding discharge areas. Concentrations of copper metals above SELs are particularly worrisome as it points to the strong possibility that there are toxic impacts to aquatic organisms. There may also be long term effects from the lower concentrations of lead and zinc. It is likely that there has been significant degradation of the aquatic ecosystem in these areas. The slow moving nature of Agassiz slough seems to prevent significant impact to other sites downstream from the outfalls, however metal concentrations still appear to be elevated throughout the watercourse. Metal concentrations of Pb, Cu, and Mn found outside the outfall station are within the range found by McCallum (1995) in Brunette Watershed stream bed sediments (Table 2.5). However, results for zinc are significantly higher then those found by McCallum, and thus, may be higher then what is typically expected for an urban centre in BC.

5.7.2 Urban Land Uses

Without taking into account the impact of the Agassiz storm sewer outfalls, land use correlations show that residential areas within the AHHS are associated with reduced levels of nitrate, chlorophyll and turbidity, in addition to increasing the overall pH. Levels of copper in sediment also appear to decrease with increasing residential area. Results were unexpected since urban centres are usually associated with a significant increase in nonpoint source pollution from impermeable surfaces, particularly for metals (McCallum, 1995; Legret and Pagotto, 1999; Smith *et al.*, 2001). However, it is important to note that the majority of residential areas included in the correlation are from the Village of Harrison Hot Springs. Compared to other residential areas, Harrison Hot Springs is relatively small, and made up primarily of residential ares. These residences are connected to the municipal sewage infrastructure, and thus nutrient inputs from sources such as septic systems should not have been observed to any large degree. In addition, sewage and storm sewer collection redirects impermeable surface pollutants such as metals, away from the watercourses. As a result, the impacts of residential areas are less pronounced compared to agricultural land uses, and the negative correlation likely reflects natural attenuation from upstream agricultural impacts.

5.7.3 Forested Areas

Forested areas were one of the few land uses which showed predictable water quality correlations using the contributing area method. In general, an increase in forested areas resulted in water quality more reflective of the control station. This includes a lower pH, specific conductivity and chlorophyll concentration combined with a higher level of DO. As stated earlier, most of the forested areas within the watershed cover the mountainous areas. The resulting correlations reflect the considerable influx of mountain runoff into the watercourses. A negative correlation with nitrite may be the result of higher concentrations of DO.

The contributing area method works well for forested areas likely because of the relatively steep, rocky slopes of the mountains. This environment is conducive to significant runoff, which likely travels the length of the contributing area. Soils in the agricultural lowlands, however, are more conducive to the adsorption and percolation of water rather than significant runoff. As a result, the amount of land in which runoff directly drains into watercourses is much smaller. Delineation of the contributing areas in the lowland areas is also more difficult due to lack of topographical relief which is much more obvious in the mountainous regions.

Previous studies of agricultural watersheds have noted that an increase in forest cover results in an overall decrease in nitrate levels (Hooda *et al.*, 1997; Smolen, 1981). Although this relationship was not evident using the contributing area method, a significant negative relationship was observed using the buffer technique.

5.8 Areas of Degradation

Based on the water and sediment quality parameters measured, it is apparent that Agassiz Slough is in the most degraded watercourse within the AHHS watershed. Degradation comprises of elevated levels of nitrate, orthophosphate, ammonia, turbidity and decreased DO concentrations. However, the main concerns are toxic concentrations of nitrite in water and copper in sediments outside the outfall station. It is assumed that areas adjacent to storm sewer outfalls along the watercourse are in a similarly degraded state. Although most water quality variables are within the range of similar stormwater studies conducted in the Lower Fraser Valley (McCallum, 1995; Macdonald *et al.*, 1997),

concentrations of zinc in sediments and turbidity in water are considerably higher than other municipal outfalls. Various initiatives such as Best Management Practices may have to be implemented in order to control these contaminants.

Hogg Slough also has high nutrient concentrations combined with elevated concentrations of zinc and copper in its sediment. However, unlike Agassiz Slough, the reason for degradation can be linked to the animal farms and grazing pastures present along the banks of the watercourse. The results of the bioassay also confirm that there may be significant impacts to the aquatic ecosystem at station H2. Toxicity may be the result of high levels of zinc in the sediment and nitrite and turbidity in water.

Individual sampling points within some watercourses also show degraded water quality due to surrounding agricultural land uses. The most significant is station D5 (McCallum Slough) where nutrient levels are consistently high relative to other sampling points along the watercourse. In addition, nitrite levels exceeded maximum allowable concentrations on one of the sampling days. Elevated concentrations of zinc in the sediments of D5 indicate that the source is likely manure being spread on the surrounding fields. Seasonal tributaries draining into the slough may be transporting some of these nutrients from outside the 100 m buffer used in the study. Station C3 (Miami Creek) is influenced by this same process except to a lesser extent. Another station of concern is site I3 (Westlin Ditch), which shows high concentrations of zinc and copper in its sediments. Sources are assumed to be related to the close proximity of the watercourse to the Lougheed highway, in addition to the use of manure on adjacent agricultural land uses.

It is important to note that water quality guidelines do not take into account the long term impacts or the cumulative effects of contaminants on the aquatic ecosystem. As stated earlier, guidelines are often determined by exposing organisms to contaminants for limited periods of time under controlled conditions. The combined cumulative effects of a variety of environmental stressors, such as temperature, may result in detrimental effects on organisms within the aquatic ecosystem even if water quality variables are below provincial and federal water quality guidelines.

6. Summary

6.1 Land Uses and Water Quality Impacts

Based on GIS calculations, the total area within the AHHS watershed is 6362.8 hectares. Forested areas comprise the largest area in the watershed. The majority of the lowland areas are used for agricultural purposes, with the most significant uses being hay/silage fields and corn crops.

Calculating land uses within a designated buffer around watercourses was found to be more effective then delineating contributing areas. It is thought that this is due to the ability of the buffers to incorporate more relevant land uses that affect water quality, particularly for areas where the overall topography is flat such as in the AHHS watershed. In addition, the flat topography made it difficult to delineate actual contributing areas, which may have adversely affected the resulting correlations.

By using the buffer technique, it was found that agricultural land uses can have a significant impact on water quality within the AHHS watershed. Pastures, in particular, appear to result in higher concentrations of nutrients in water, and increased levels of zinc in sediments due to runoff from manure. Hay/silage and corn fields did not appear to have any significant impacts on water quality, even during the winter period when soils were most subject to erosion and runoff.

Forest covered areas were not observed to impact water quality within the AHHS watershed, except for an observed reduction in nitrate concentrations. Residential areas were observed to not have any appreciable effect on overall water quality.

6.2 State of Water and Sediment Quality in the AHHS Watershed

Watercourses throughout the watershed are well within range of provincial guidelines for the nutrients orthophosphate, nitrate and ammonia. However, levels of nitrite are not in compliance in a number of areas in the watershed, exacerbated by low levels of DO found at many sites. Levels of DO and temperature consistently remained below required provincial and federal guidelines, particularly for developing salmonid embryos. Variables such as pH in water and iron in sediments both have natural background

concentrations outside of provincial regulations thus they are not assumed to have an impact on the natural aquatic system.

When compared with provincial guidelines, Agassiz Slough is in the most degraded state. Based on the results of the outfall station, it is likely that outfalls along the watercourse contribute toxic levels of nitrite to the water column, in addition to detrimental concentrations of lead, zinc, and especially copper that accumulates in the sediments. Station I3 (Clarke Ditch), similarly appears to have toxic levels of copper combined with elevated zinc concentrations in its sediment likely due to a combination of nearby impermeable surfaces and the spreading of manure on adjacent agricultural lands.

The presence of animal farms and grazing pastures adjacent to Hogg Slough has resulted in significant degradation, likely due to runoff contaminated by manure. Although concentrations of ammonia and nitrate are well within provincial guidelines, nitrite and zinc exceeded current provincial standards for water quality and sediment quality respectively. Results of the moss bioassay indicate that there may be adverse impacts to the aquatic community, particularly at site H2. Station D5 (McCallum Slough) and C3 (Miami Creek) also showed elevated nutrient and metal concentrations compared to other sampling sites due to nearby manure application, and may be areas of future concern.

6.3 Applicability of Bioassays

The aquatic moss *Fontinalis antipyretica* did not prove to be an effective bioassay for agricultural watercourses within the AHHS watershed. Although differences were observed between stations H4 and H2, no other statistically significant differences were observed between other sampling stations. The main reason for the high variability between sites is thought to reflect the variable site conditions. Factors such as light, flow and periphytic growth, can affect moss stem growth and in addition to levels of chlorophyll *a* and *b* within the stems. Under these variable conditions, it is not likely that field experiments using *Fontinalis antipyretica* moss stems can show the impacts of minor changes in the environmental quality of small agricultural watercourses.

6.4 Atrazine Adsorption by XAD-7 Resin

Atrazine adsorption to XAD-7 resins was found to be primarily dependent on the concentration of atrazine and the presence of organic matter in the water. Benchscale lab results showed that the best recovery of 76% was obtained using distilled water with a concentration of 100 ppb atrazine, which was the highest sample water concentration studied. When actual site water was used, much lower recoveries were observed. It is thought that this was most likely due to the presence of organic matter in the water binding atrazine and making in unavailable for sorption to the XAD resin. In addition, organic matter may be binding to the resin and blocking membrane pores.

The field component gave significantly lower recovered concentrations per gram of resin then to benchscale results. This could be due to significantly lower concentrations of atrazine at the field stations or the presence of additional environmental factors which are inhibiting adsorption. Overall, the highest atrazine concentration was recovered from Agassiz Slough and Westlin Ditch, while the lowest recovery came from McCallum Slough. No correlations were observed between corn field area and recoverable atrazine.

7. Recommendations

7.1 Areas of Further Research

Improvements to Atrazine Adsorption Technique

XAD-7 resins show some promise for being used as an indicator to atrazine contamination and research has shown that better recoveries should be possible. However, there are a number of areas that should be investigated in order to improve on the results obtained. Improvements to atrazine extraction from the resin should be examined. Variations in analytical technique and changes to the extraction solution may significantly increase recoveries. An internal standard should also be incorporated into the analytical technique in order to ensure that there is no degradation during adsorption or recovery. Further improvements to the resin apparatus should also be made in order to eliminate the gradient effect observed in the results.

For future studies, a method for measuring atrazine concentrations in air should be developed to confirm possible atmospheric inputs due to windblown transport from nearby fields.

Comprehensive Study of Agassiz Slough

Agassiz Slough is obviously impacted as a result of the storm sewer outfalls from the Town of Agassiz. A toxicological examination of the water and sediments should be conducted to determine long term and cumulative impacts on the aquatic ecosystem. In addition, the sediments outside all the outfall locations should be sampled to determine the full extent of metal contamination downstream from each discharge point.

Determination of Watercourse Maintenance Impacts

It was not possible during the course of this experiment to adequately gauge the impact of watercourse maintenance on water quality. However, high concentrations of copper and zinc in the sediments of some stations may pose a risk to the aquatic system if maintenance activities are allowed to take place. A dedicated study should be conducted in collaboration with the Agassiz Drainage Committee and relevant contractors in order to determine any adverse impacts during the annual maintenance of sloughs. A comprehensive sampling program examining both water and sediment quality should be

considered, particularly if any work is done in Agassiz Slough, Hogg Slough, or Clarke Ditch

Alternative Bioassays for Agricultural Watercourses

The use of *Fontinalis antipyretica* as a bioassay does not appear to be feasible within agricultural watercourses. As a result, other bioassays should be investigated as better replacements to this particular species. An examination of chlorophyll levels in native periphyton may be an alternative method. Small plates can be used as a Periphyte growing surface within watercourses and kept at a constant depth in order to minimize any variability. Another option would be to use the *Fontinalis antipyretica* moss bioassay in the laboratory rather then the field in order to minimize variability in the environmental conditions between stations.

Database Creation

A database covering areas such as agricultural land uses, pesticide use, animal density, crop types should be generated for future studies concerning water quality in the AHHS watershed. A determination of stream flow and channel depth should also be made to facilitate comparisons between stations.

Impacts of Rural Highways

There is a possibility that the source of metal contaminants in watercourses is a result of runoff coming from of the local highways which pass through the area. A study examining the contribution of highway runoff to watercourse water quality should be undertaken to confirm this possibility.

7.2 Management Recommendations

Best Management Practices

It is obvious that discharges from storm sewer outfalls in Agassiz Slough must be controlled in order to limit the impact to the watercourse and aquatic ecosystem. It is recommended that best management policies be implemented throughout the storm sewer infrastructure in order to bring discharges to within current provincial guidelines. More

information can be obtained by referring to Best Management Practices Guide for Stormwater (GVSDD, 1999).

Implementation of Nutrient Management Practices

Fertilizer and manure application should be controlled and monitored to ensure that nutrients do not build up to excessive levels within soils. A program for relocating excess manure is another option, to ensure that farmers are not inclined to spread manure as a form of waste disposal. The November deadline for the application of manure should also be revised take into account climatic conditions in order to minimize nutrient runoff into watercourses. Other factors such as protecting manure piles from rain and restricting the spreading manure to established grasslands are recommended by Manure Management Guidelines for the Lower Fraser Valley.

Development of a Best Agricultural Waste Management Plan can be implemented within the AHHS watershed to meet the specific needs of the farmers and the community. Care should be taken to ensure that agricultural activities concerning manure do not contravene the *Fisheries Act*, or the *Agricultural Waste Control Regulation* under the *Waste Management Act*. More information can be obtained by consulting the *Manure Management Guidelines* (BCMAF, 2002)

Separation of Drainage Slough and Aquatic Habitat Designations

A clear designation between agricultural drainage sloughs and aquatic habitats which may hold sensitive aquatic ecosystems should be sought. It is essential that local farmers are able to properly drain their land without fear of imposed habitat maintenance responsibilities. Areas which need to be protected, such as areas of fish spawning, should have controlled development in order to minimize any further degradation. This study has shown that there are a number of manmade drainage sloughs which are incapable of supporting healthy aquatic life. Unless these sites show potential to support a viable fish population, they should not be subject to the stringent criteria required in more sensitive areas.

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Personal Communication

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Appendix A: Photo Appendix

Photo: Watercourse maintenance activities along Hogg Slogh (Aug 22, 2001)

Photo A.2: Manure application of hay/silage fields (Nov. 1, 2001)

Photo A.3: Sheep farm adjacent to McCallum Slough (June 7, 2002)

Photo A.4: Sediment plume from sewer outfall in Agassiz Slough (Feb. 8, 2002)

Photo A.5: Moss bioassay mesh bag with float at the Control station (May 24, 2002). No periphytic growth present.

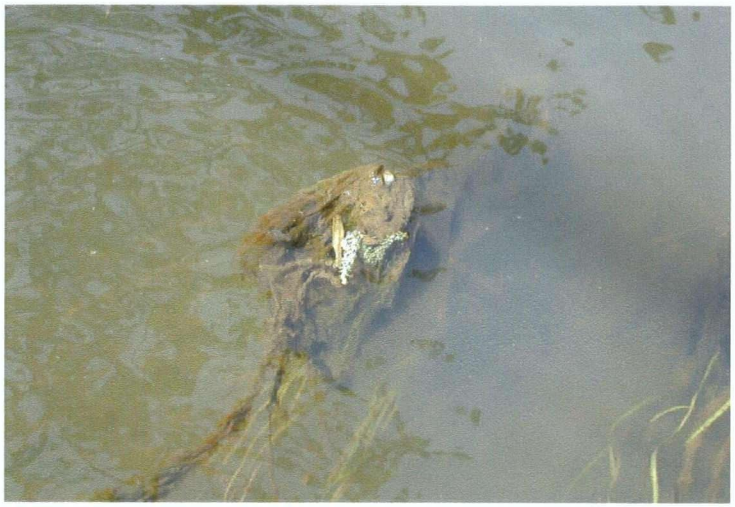

Photo A.6: Severe periphytic growth around moss bioassay mesh bag at station C1 in Miami Creek (May 24, 2002)

Photo A.7: XAD-7 resin apparatus at the Control station (May 24, 2002)

Photo A.8: Severe periphytic growth around XAD-7 resin apparatus at station F2 in Westlin Ditch (May 24, 2002)

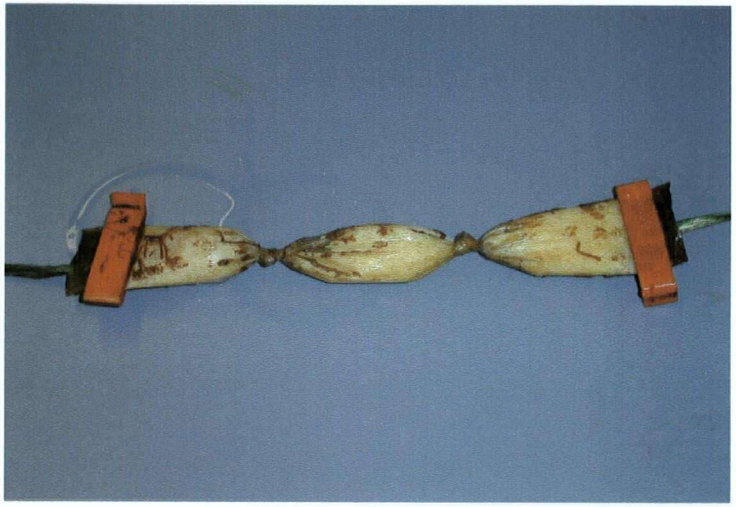

Photo A.9: Control station XAD-7 resin pouch after removal from apparatus (June 7, 2002)

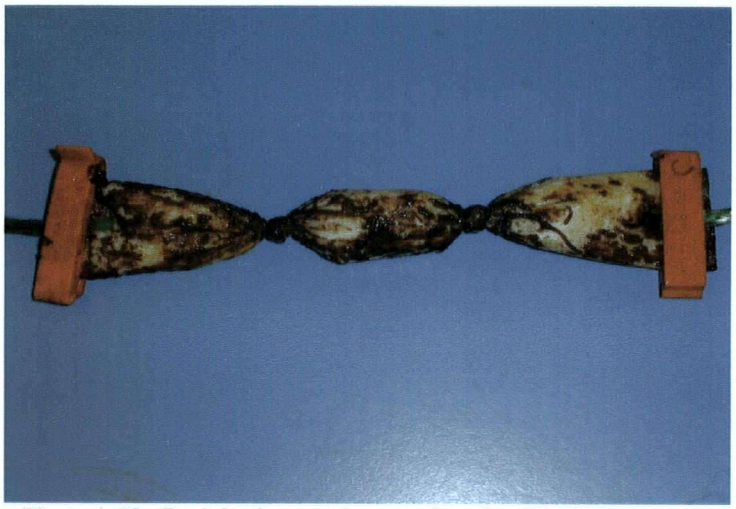

Photo A.10: Periphytic growth around station F2 XAD-7 resin pouch after removal from apparatus (June 7, 2002)

Photo A.11: XAD-7 resin pouch suspended through pipe in resin apparatus (June 7, 2002)

Photo A.12: Concrete block and pipe configuration for XAD-7 Resin apparatus (June 7, 2002)

Appendix B: Water Quality Data

Table B.1: Water Quality Parameter Median Values for the Wet and Dry Seasons

	Ammonia (mg/L)		Nitrate (mg/L)		Nitrite (mg/L)		Orthophosphate (mg/L)		Dissolved Oxygen (mg/L)		Chlorophyll (µg/L)		Specific Conductivity (µg/cm)		pH		Turbidity (NTU)	
	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
A1	0.008	0.052	0.023	0.159	0.006	0.008	0.026	0.017	5.9	10.2	5.2	0.8	73	63	6.4	6.1	8.7	11.8
B1	0.059	0.082	0.060	0.114	0.015	0.012	0.021	0.028	7.2	7.2	4.8	0.8	178	136	6.6	6.2	22.1	9.1
C3	0.037	0.049	0.114	0.958	0.015	0.013	0.041	0.040	7.0	11.2	15.9	0.6	67	51	6.2	6.1	21.0	4.2
Control	0.017	0.007	0.062	0.225	0.009	0.008	0.008	0.005	9.5	12.0	0.8	0.0	30	18	5.8	5.6	2.9	1.7
C0	0.003	0.035	0.043	0.476	0.004	0.013	0.013	0.013	7.9	10.4	1.6	1.0	104	81	6.5	6.2	6.8	6.8
C4	0.042	0.067	0.151	1.025	0.008	0.010	0.037	0.033	7.0	10.1	3.9	0.9	101	74	6.4	5.9	7.9	9.7
C5	0.040	0.037	0.149	0.751	0.010	0.011	0.026	0.022	6.8	9.4	3.3	0.3	106	76	6.0	5.8	6.8	4.7
C6	0.046	0.045	0.095	0.509	0.009	0.010	0.020	0.014	9.0	9.5	1.7	0.6	123	82	6.4	6.1	7.7	6.6
M1	0.038	0.066	0.316	1.014	0.013	0.014	0.024	0.027	6.6	10.6	3.2	0.8	125	81	6.2	5.9	8.2	5.7
C1	0.039	0.039	0.056	0.560	0.010	0.012	0.016	0.016	6.9	9.8	2.5	0.4	128	81	6.3	6.0	8.1	5.1
D5	0.013	0.244	1.457	1.902	0.010	0.019	0.015	0.089	3.3	3.8	5.8	1.6	189	201	6.3	6.1	1.8	6.6
D4	0.055	0.145	0.233	1.282	0.010	0.014	0.036	0.026	8.2	6.8	11.2	2.4	183	162	6.6	6.2	5.9	14.2
D3	0.154	0.154	0.153	0.738	0.012	0.014	0.043	0.026	5.9	8.4	9.6	0.7	142	113	6.5	6.1	6.5	5.7
D2	0.054	0.226	0.096	0.656	0.008	0.019	0.018	0.042	6.4	7.5	4.2	1.4	172	143	6.4	6.2	6.3	9.3
D1	0.005	0.133	0.032	0.370	0.006	0.023	0.020	0.028	8.4	8.0	4.6	0.7	193	148	6.7	6.3	4.4	13.2
F2	0.011	0.054	0.028	0.803	0.017	0.017	0.035	0.022	1.5	9.9	26.8	0.9	158	99	6.4	6.1	24.4	7.4
F1	0.011	0.063	0.027	0.424	0.005	0.012	0.015	0.034	6.7	7.3	2.8	0.6	265	206	6.7	6.5	5.0	9.6
G1	0.042	0.185	0.090	0.595	0.009	0.022	0.020	0.039	7.0	7.6	1.2	0.6	227	194	6.5	6.4	10.0	12.4
H4	0.055	0.119	0.032	0.498	0.011	0.015	0.024	0.025	8.7	9.5	10.2	1.0	151	106	6.5	5.7	9.4	12.8
H2	0.441	0.251	0.130	0.897	0.019	0.027	0.103	0.046	10.4	8.8	13.7	1.6	162	111	6.4	6.1	20.3	13.4
H1	0.045	0.512	0.040	0.724	0.014	0.021	0.039	0.069	6.3	9.9	8.1	3.1	169	118	6.5	6.1	12.7	47.7
I1	0.619	0.079	0.038	0.744	0.030	0.016	0.295	0.088	3.1	10.6	9.8	1.3	223	102	6.3	5.9	227.0	10.8
I3	0.006	0.127	0.041	0.518	0.007	0.015	0.026	0.035	6.7	10.0	41.2	1.0	144	108	6.2	6.1	8.5	11.0
J1	0.030	0.066	0.106	0.705	0.013	0.012	0.016	0.020	5.4	8.0	11.6	1.0	169	97	5.9	5.5	20.2	7.7
K2	0.111	0.437	0.332	0.421	0.023	0.016	0.024	0.046	3.1	4.1	15.6	2.0	328	91	6.7	6.2	13.3	7.6
Outfall	0.090	0.822	0.436	0.152	0.020	0.039	0.153	0.123	5.0	10.6	4.3	3.7	275	258	6.6	5.9	12.2	98.8
K1	0.030	0.011	0.332	0.810	0.010	0.008	0.044	0.025	4.0	1.4	5.3	1.6	278	144	6.8	6.1	3.3	3.7
Spring	0.038	0.087	0.017	0.199	0.007	0.008	0.010	0.063	0.6	2.7	0.4	0.2	336	270	6.6	6.4	1.7	5.2
MTN-1	0.081	0.113	0.067	0.284	0.006	0.010	0.028	0.020	8.6	13.1	2.2	3.5	117	116	6.6	6.3	6.2	13.1

Table B.2: Monthly Ammonia Concentrations (mg/L)

Sampling Station	Date										
	26-Jun-01	23-Jul-01	22-Aug-01	18-Sep-01	01-Nov-01	13-Dec-01	23-Jan-02	28-Feb-02	04-Apr-02	03-May-02	07-Jun-02
A1	0.015	0.008		0.178	0.079		0.052	0.013	0.004	0.007	0.000
B1	0.004	0.160	0.059	0.080	0.090	0.045	0.130	0.073	0.031	0.060	0.007
C3	0.004	0.037	0.158	0.606	0.112	0.058	0.040	0.010	0.016	0.015	0.061
Control			0.238	0.241	0.007	0.006	0.033	0.005	0.002	0.017	0.016
C0			0.025	0.038	0.021	0.049	0.070	0.003	0.003	0.000	0.000
C4			0.068	0.093	0.090	0.097	0.043	0.019	0.042	0.006	0.000
C5			0.080	0.121	0.027	0.013	0.046	0.081	0.016	0.008	0.040
C6			0.151	0.046	0.025	0.065	0.075	0.025	0.010	0.159	0.022
M1	0.009	0.012	0.076	0.073	0.086	0.046	0.086	0.009	0.038	0.012	0.045
C1	0.039	0.018	0.073	0.059	0.034	0.043	0.090	0.004	0.019	0.043	0.035
D5	0.013	0.008	0.047	0.086	4.546	0.391	0.024	0.097	0.013	0.009	0.000
D4		0.048	0.127	0.116	1.790	0.219	0.071	0.014	0.012	0.020	0.062
D3	0.018	0.199	0.046	0.197	0.159	0.148	0.165	0.099	0.121	0.243	0.154
D2	0.009	0.007	0.031	0.057	0.263	0.270	0.189	0.095	0.118	0.068	0.054
D1	0.000	0.000	0.065	0.004	0.119	0.236	0.147	0.097	0.050	0.005	0.322
F2	0.000	0.011	0.045	0.249	0.032	0.247	0.075	0.009	0.025	0.004	0.007
F1	0.005	0.000	0.080	0.011	0.001	0.199	0.095	0.030	0.028	0.000	0.030
G1	0.004	0.591	0.132	0.047	0.241	0.796	0.129	0.026	0.042	0.000	0.037
H4	0.103	0.011	0.055	0.052	0.184	0.128	0.110	0.027	0.191	0.000	0.263
H2	0.027	0.142	8.888	0.380	0.163	0.339	2.498	0.069	0.441	0.741	1.544
H1	0.060	0.017	0.044	0.039	0.552	0.471	0.712	0.144	0.523	0.137	0.045
I1	0.617	0.057	0.621		0.551	0.120	0.037	0.010	0.995	5.790	0.568
I3	0.000	0.000	0.056	0.236	0.171	0.216	0.082	0.017	0.037	0.000	0.006
J1	0.012	0.012	0.030	0.069	0.078	0.053	0.112	0.028	0.041	0.068	0.006
K2	0.062	0.026	5.593	0.974	0.024	0.148	1.054	0.725	0.025	3.497	0.111
Outfall	0.019	0.090	0.808	8.793	5.407	0.218	1.425	0.008	0.049	0.731	0.000
K1	0.011	0.030	0.075	0.543	0.000	0.109	0.006	0.016	0.068	0.014	0.007
Spring	0.038	0.027	0.080	0.094	0.035	0.350	0.138	0.004	0.035	0.042	0.033
MTN-1								0.113	0.117	0.017	0.081

Table B.3: Monthly Nitrate Concentrations (mg/L)

Sampling Station	Date										
	26-Jun-01	23-Jul-01	22-Aug-01	18-Sep-01	01-Nov-01	13-Dec-01	23-Jan-02	28-Feb-02	04-Apr-02	03-May-02	07-Jun-02
A1	0.018	0.027		0.010	0.422		0.16	0.09	0.03	0.01	0.03
B1	0.022	0.019	0.075	0.037	0.201	0.171	0.05	0.06	0.06	0.07	0.08
C3	0.072	0.114	0.095	0.002	0.914	1.031	0.81	1.00	0.46	0.39	0.18
Control			0.023	0.032	0.498	0.339	0.11	0.11	0.10	0.08	0.06
C0				0.014	0.699	0.536	0.37	0.42	0.12	0.06	0.02
C4				0.000	2.201	1.017	0.91	1.03	0.42	0.29	0.01
C5				0.037	1.204	0.647	0.72	0.79	0.23	0.18	0.11
C6				0.032	0.766	0.474	0.44	0.54	0.21	0.13	0.06
M1	0.143	0.463	0.275	0.647	2.448	0.895	0.97	1.06	0.49	0.32	0.24
C1	0.052	0.056	0.052	0.032	0.918	0.528	0.51	0.59	0.17	0.14	0.08
D5	0.488	1.580	1.457	1.532	1.030	1.799	2.01	2.30	1.41	1.53	0.81
D4		0.082	0.168	0.014	1.388	1.092	1.18	1.48	0.72	0.52	0.30
D3	0.041	0.171	0.125	0.003	1.642	0.778	0.65	0.70	0.19	0.15	0.18
D2	0.057	0.096	2.648	0.055	1.378	0.784	0.50	0.53	0.17	0.13	0.07
D1	0.009	0.027	0.032	0.014	0.853	0.390	0.29	0.35	0.13	0.09	0.19
F2		0.016	0.049	0.039	0.746	0.906	0.64	0.86	0.39	0.00	0.01
F1	0.027	0.023	0.109	0.014	0.449	0.595	0.34	0.40	0.14	0.02	0.04
G1	0.059	0.166	0.132	0.080	0.675	0.848	0.42	0.52	0.12	0.07	0.09
H4	0.032	0.049	0.017	0.007	1.695	0.759	0.19	0.24	0.11	0.01	0.04
H2	0.130	0.029	0.100	0.071	1.680	1.046	0.60	0.75	0.51	0.23	0.50
H1	0.040	0.014	0.020	0.012	1.108	0.818	0.51	0.63	0.36	0.15	0.25
I1	0.036	0.069	0.012		2.388	0.934	0.55	0.36	0.44	0.01	0.04
I3	0.058	0.041	0.085	0.015	0.980	0.515	0.48	0.52	0.16	0.03	0.02
J1	0.188	0.032	0.068	0.019	1.790	0.455	0.81	0.60	0.11	0.16	0.11
K2	1.048	1.686	0.332	0.099	0.135	0.339	0.50	1.52	0.14	0.09	1.75
Outfall	1.646	0.436	0.351	0.048	0.186	0.087	0.12	3.34	0.06	1.33	1.30
K1	0.750	0.110	0.332	0.009	0.394	0.685	0.93	1.83	0.01	0.41	1.30
Spring	0.025	0.019	0.015	0.010	0.855	0.338	0.06	0.04	0.02	0.01	0.02
MTN-1								0.28	0.13	0.07	0.06

Table B.4: Monthly Nitrite Concentrations (mg/L)

Sampling Station	Date										
	26-Jun-01	23-Jul-01	22-Aug-01	18-Sep-01	01-Nov-01	13-Dec-01	23-Jan-02	28-Feb-02	04-Apr-02	03-May-02	07-Jun-02
A1	0.005	0.006		0.005	0.019		0.008	0.006	0.012	0.010	0.006
B1	0.011	0.028	0.015	0.015	0.017	0.013	0.011	0.008	0.018	0.010	0.009
C3	0.007	0.015	0.020	0.025	0.035	0.015	0.010	0.006	0.020	0.006	0.008
Control			0.012	0.009	0.006	0.009	0.009	0.002	0.009	0.006	0.004
C0				0.004	0.014	0.015	0.012	0.005	0.003	0.007	0.003
C4				0.025	0.021	0.012	0.008	0.008	0.005	0.010	0.006
C5				0.003	0.012	0.011	0.009	0.010	0.028	0.011	0.009
C6				0.004	0.010	0.010	0.010	0.006	0.022	0.009	0.008
M1	0.016	0.012	0.013	0.016	0.024	0.015	0.013	0.006	0.028	0.009	0.010
C1	0.025	0.013	0.012	0.010	0.012	0.014	0.012	0.006	0.009	0.008	0.008
D5	0.010	0.024	0.011	0.009	0.066	0.025	0.012	0.007	0.015	0.010	0.009
D4		0.007	0.006	0.004	0.080	0.005	0.016	0.012	0.020	0.014	0.013
D3	0.009	0.014	0.008	0.015	0.028	0.014	0.013	0.014	0.023	0.012	0.012
D2	0.006	0.008	0.019	0.003	0.052	0.021	0.016	0.012	0.008	0.008	0.009
D1	0.006	0.004	0.005	0.002	0.043	0.005	0.011	0.035	0.010	0.011	0.018
F2	0.067	0.076	0.019	0.017	0.017	0.019	0.017	0.010	0.016	0.013	0.013
F1	0.008	0.003	0.006	0.002	0.013	0.016	0.010	0.010	0.007	0.005	0.005
G1	0.009	0.016	0.014	0.005	0.031	0.029	0.015	0.014	0.009	0.009	0.006
H4	0.011	0.011	0.006	0.009	0.030	0.016	0.013	0.009	0.016	0.011	0.011
H2	0.015	0.008	0.263	0.015	0.035	0.022	0.031	0.013	0.019	0.020	0.023
H1	0.030	0.011	0.006	0.009	0.042	0.022	0.020	0.019	0.027	0.020	0.014
I1	0.049	0.022	0.023		0.044	0.020	0.011	0.007	0.037	0.037	0.022
I3	0.007	0.006	0.008	0.006	0.034	0.018	0.012	0.010	0.007	0.009	0.008
J1	0.025	0.009	0.009	0.006	0.012	0.011	0.010	0.017	0.025	0.037	0.013
K2	0.023	0.014	0.115	0.026	0.006	0.014	0.018	0.032	0.009	0.038	0.019
Outfall	0.012	0.019	0.063	0.032	0.065	0.012	0.072	0.004	0.009	0.059	0.020
K1	0.010	0.014	0.008	0.017	0.008	0.027	0.008	0.005	0.008	0.007	0.015
Spring	0.013	0.003	0.006	0.005	0.009	0.014	0.007	0.006	0.008	0.007	0.007
MTN-1								0.010	0.006	0.005	0.008

Table B.5: Monthly Orthophosphate Concentrations (mg/L)

Sampling Station	Date										
	26-Jun-01	23-Jul-01	22-Aug-01	18-Sep-01	01-Nov-01	13-Dec-01	23-Jan-02	28-Feb-02	04-Apr-02	03-May-02	07-Jun-02
A1	0.018	0.014		0.030	0.044		0.014	0.017	0.032	0.025	0.026
B1	0.023	0.021	0.038	0.019	0.029	0.017	0.035	0.026	0.039	0.020	0.015
C3	0.023	0.093	0.167	0.340	0.061	0.058	0.022	0.011	0.027	0.020	0.041
Control			0.014	0.011	0.003	0.030	0.003	0.006	0.008	0.003	0.005
C0			0.021	0.012	0.008	0.016	0.017	0.009	0.031	0.013	0.010
C4			0.073	0.144	0.031	0.053	0.034	0.013	0.037	0.022	0.031
C5			0.162	0.007	0.038	0.023	0.020	0.011	0.033	0.014	0.026
C6			0.023	0.011	0.014	0.014	0.014	0.013	0.035	0.020	0.015
M1	0.020	0.023	0.031	0.012	0.038	0.040	0.014	0.015	0.039	0.024	0.038
C1	0.016	0.008	0.016	0.009	0.014	0.017	0.028	0.011	0.022	0.015	0.021
D5	0.023	0.011	0.013	0.008	0.895	0.160	0.018	0.008	0.021	0.017	0.015
D4		0.055	0.040	0.028	0.085	0.038	0.003	0.014	0.042	0.025	0.032
D3	0.034	0.073	0.043	0.053	0.029	0.033	0.022	0.020	0.057	0.025	0.036
D2	0.019	0.016	0.017	0.014	0.049	0.102	0.035	0.027	0.052	0.027	0.018
D1	0.017	0.015	0.052	0.008	0.029	0.070	0.027	0.020	0.029	0.020	0.053
F2	0.045	0.033	0.043	0.016	0.017	0.022	0.023	0.021	0.035	0.029	0.040
F1	0.045	0.005	0.015	0.012	0.042	0.046	0.025	0.025	0.029	0.015	0.019
G1	0.020	0.015	0.028	0.005	0.051	0.285	0.027	0.026	0.038	0.025	0.020
H4	0.030	0.016	0.017	0.024	0.037	0.030	0.019	0.008	0.071	0.014	0.034
H2	0.002	0.103	1.908	0.269	0.032	0.060	0.176	0.016	0.082	0.091	0.305
H1	0.088	0.035	0.029	0.028	0.077	0.144	0.060	0.042	0.253	0.071	0.039
I1	0.251	0.194	0.262		0.406	0.129	0.047	0.029	0.327	1.034	0.439
I3	0.022	0.026	0.045	0.020	0.037	0.061	0.033	0.016	0.027	0.021	0.029
J1	0.021	0.016	0.012	0.013	0.026	0.034	0.014	0.014	0.024	0.064	0.013
K2	0.023	0.024	0.727	0.265	0.029	0.029	0.121	0.063	0.023	0.577	0.022
Outfall	0.011	0.035	0.405	0.859	3.577	0.116	0.130	0.007	0.227	0.153	0.013
K1	0.012	0.019	0.044	0.236	0.056	0.027	0.018	0.023	0.296	0.160	0.013
Spring	0.010	0.003	0.019	0.010	0.081	0.365	0.044	0.024	0.006	0.022	0.016
MTN-1								0.020	0.028	0.011	0.034

Table B.6: Monthly Chlorophyll Concentrations (μg/L)

Sampling Station	Date										
	26-Jun-01	23-Jul-01	22-Aug-01	18-Sep-01	01-Nov-01	13-Dec-01	23-Jan-02	28-Feb-02	04-Apr-02	03-May-02	07-Jun-02
A1	21.0	7.3		1.2	0.4		1.1	0.8	3.0	7.9	1.5
B1	11.2	141.7	1.9	7.4	1.3	1.3	0.3	0.4	3.8	4.8	3.3
C3	32.9	34.7	15.9	52.2	0.6	1.1	0.2	0.5	1.4	1.8	1.8
Control			15.6	2.6	0.0		0.0	0.0	0.3	0.8	0.7
C0			1.6	1.5	2.0	1.0	0.5	1.0	1.1	2.2	6.3
C4			11.7	3.9	1.0	1.8	0.8	0.4	2.5	12.8	2.4
C5			16.2		0.3	0.8	0.0	0.4	1.3	5.4	1.1
C6				1.8	1.4	0.8	0.4	0.4	1.6	5.3	1.3
M1	2.3	4.3	3.9	1.5	1.1	1.0	0.2	0.5	3.2	15.2	1.5
C1	10.7	1.2	2.5	3.6	8.5	0.6	0.2	0.2	1.1	4.5	1.8
D5	1.5	88.0	53.5	20.8	4.4	1.5	0.3	1.8	1.2	5.8	2.0
D4	24.1	12.7	11.2	2.1	5.3	2.2	2.6	1.0	9.7	16.0	2.4
D3	2.5	21.0	9.6	12.2	0.7		0.5	0.8	3.6	10.1	1.4
D2	22.3	6.2	1.8	0.3	0.4	2.5	1.7	1.1	4.2	6.6	2.0
D1	24.5	26.8	4.6	0.4	0.6	1.4	0.4	0.7	2.5	6.7	1.8
F2	43.9	26.8	65.5	29.6	0.9		0.9	1.5	5.8	6.3	2.1
F1	25.8	1.5	12.9	0.5	0.5	1.4	0.6	0.5	2.8	6.6	1.3
G1	0.5	23.1	2.7	1.2	0.6	2.8	0.1	0.5	0.7	3.7	1.0
H4	10.2	17.2	22.7	12.6	0.7	1.4	0.2	1.3	3.3	4.3	5.5
H2	18.1	13.7	49.7	14.9	1.6		3.6	0.8	1.9	10.8	10.3
H1	13.3	3.5	1.5	10.8	2.0		3.4	3.1	8.1	23.6	6.1
I1	80.7	23.7	0.9	4.5	0.6	3.4	0.9	1.7	9.8	199.0	5.7
I3	67.6	334.5	143.4		1.3	3.0	0.7	0.6	14.9	7.5	1.2
J1	6.5	36.6	29.5	26.0	2.1	1.5	0.4	0.4	1.6	11.6	4.1
K2	0.1	18.2	10.7	72.4	3.9	0.7	2.2	1.7	4.6	79.9	15.6
Outfall	12.3	2.1	1.3	15.2	3.1	4.2	4.9	0.0	93.1	0.8	4.3
K1	8.8	2.3	1.3	130.9	1.0	5.4	1.6	1.7	44.6	3.6	5.3
Spring	2.0	11.0	1.3	0.4	0.2		0.2	0.2	0.2	0.4	0.1
MTN-1								3.5	1.8	2.2	2.7

Table B.7: Monthly Specific Conductivity Levels (μs/cm)

Sampling Station	Date										
	26-Jun-01	23-Jul-01	22-Aug-01	18-Sep-01	01-Nov-01	13-Dec-01	23-Jan-02	28-Feb-02	04-Apr-02	03-May-02	07-Jun-02
A1	75	75		84	78	34	73	53	50	70	48
B1	198	280	156	246	126	52	146	162	178	168	164
C3	67	106	190	270	102	51	52	46	45	46	67
Control			34	45	10	15	24	22	22	25	30
C0			149	161	77	30	101	84	94	94	104
C4			262	210	138	60	80	67	70	60	101
C5			154	168	79	42	85	73	76	72	106
C6			163	171	80	42	94	84	91	101	123
M1	125	141	143	131	122	54	86	76	81	69	102
C1	128	160	161	167	78	41	94	83	88	97	126
D5	189	187	170	185	245	109	198	203	226	226	218
D4		184	196	259	171	94	161	163	182	173	177
D3	142	157	146	170	109	72	117	116	117	118	140
D2	172	183	294	209	135	88	150	157	154	154	172
D1	193	205	193	210	140	77	157	156	168	156	178
F2	263	237	158	375	99	85	99	110	112	130	149
F1	269	278	270	265	201	110	211	227	265	253	263
G1	232	254	219	227	185	97	203	221	231	216	217
H4	174	188	151	176	135	57	101	110	92	110	150
H2	161	181	375	215	133	63	129	93	84	114	162
H1	169	190	194	257	152	76	124	111	121	134	145
I1	188	493	308		123	60	96	108	144	258	141
I3	144	161	180	217	127	62	108	108	132	119	139
J1	169	311	325	320	102	37	94	100	89	81	105
K2	328	312	386	338	22	16	160	210	516	229	290
Outfall	223	313	72	432	326	15	466	189	310	131	275
K1	290	311	290	278	77	113	174	190	230	188	278
Spring	338	339	337	336	275	107	272	267	324	319	317
MTN-1								116	117	116	158

Table B.8: Monthly Turbidity Levels (NTU)

Sampling Station	Date										
	26-Jun-01	23-Jul-01	22-Aug-01	18-Sep-01	01-Nov-01	13-Dec-01	23-Jan-02	28-Feb-02	04-Apr-02	03-May-02	07-Jun-02
A1	63.0	6.0		7.6	15.7	20.0	7.9	6.5	11.2	9.8	7.0
B1	30.0	210.0	22.1	22.3	10.9	25.0	6.2	7.2	7.6	12.8	14.8
C3	30.2	25.0	21.0	42.2	4.1	13.6	1.9	4.3	7.0	2.5	5.6
Control			4.9	3.8	2.1	2.2	1.3	1.2	0.5	0.7	2.9
C0			7.7	9.0	6.5	20.0	7.2	5.3	6.8	5.6	2.7
C4			11.9	55.3	6.6	16.0	8.0	11.5	7.9	4.1	4.1
C5			27.0	7.2	4.1	18.1	5.2	3.6	6.8	6.4	2.9
C6			8.9	4.5	5.2	20.0	8.0	4.8	20.0	6.9	7.7
M1	5.7	12.4	18.0	7.0	6.1	20.4	5.2	3.4	13.8	5.6	8.2
C1	8.1	15.9	22.0	11.0	4.4	16.8	5.8	4.3	6.2	5.3	1.4
D5	1.8	3.5	3.9	18.9	21.9	11.1	1.5	2.1	0.6	1.4	1.8
D4		5.4	6.8	6.2	12.1	16.2	35.8	4.4	7.3	5.5	4.2
D3	5.9	21.1	6.5	31.4	4.5	16.8	6.0	5.4	11.5	4.5	6.2
D2	4.2	6.7	4.7	6.3	3.6	24.9	11.1	7.4	6.3	4.4	7.7
D1	4.0	3.2	4.4	1.6	10.7	20.0	14.8	11.6	7.4	5.3	5.5
F2	90.0	303.0	14.6	743.0	5.0	14.6	9.0	5.9	7.7	13.7	24.4
F1	56.0	1.9	3.3	1.7	8.1	33.9	10.8	8.3	7.6	5.1	5.0
G1	10.0	33.0	10.7	5.3	6.7	76.7	12.5	12.3	14.9	7.4	4.4
H4	16.0	5.5	9.1	40.3	10.4	24.7	12.4	13.1	34.5	9.4	5.6
H2	12.4	6.2	61.4	123.0	7.4	66.7	18.2	8.5	58.6	20.3	12.3
H1	12.7	9.0	11.1	19.9	6.7	60.0	53.3	42.0	179.0	13.4	9.4
I1	11.5	500.0	500.0		13.6	132.0	8.1	7.3	29.9	424.0	21.1
I3	11.2	4.6	8.5	30.3	9.8	36.5	12.0	10.0	5.8	6.4	13.2
J1	23.7	39.0	23.0	20.2	5.3	19.7	8.0	7.4	6.9	6.0	19.3
K2	1.7	3.2	24.0	28.0	2.8	15.4	9.2	6.0	13.3	30.3	3.4
Outfall	1.8	3.8	82.0	91.0	42.5	155.0	626.0	1.0	12.2	27.3	1.9
K1	1.2	2.8	7.3	19.6	3.5	3.8	4.7	1.5	11.1	3.3	1.8
Spring	8.1	1.3	0.0	1.3	5.6	13.1	4.8	2.1	2.6	5.7	1.7
MTN-1								13.1	7.2	2.8	6.2

Table B.9: Monthly Dissolved Oxygen Levels (mg/L)

Date	A1	B1	C3	CON	C0	C4	C5	C6	M1	C1	D5	D4	D3	D2	D1
30-May-01	***5.4***	***6.6***	***8.6***						***7.2***	***5.7***	***2.9***	***14.8***	***6.5***	***3.0***	***8.8***
26-Jun-01	8.4	6.6	7.3						6.5	7.4	2.6		5.9	8.9	9.7
23-Jul-01	5.2	3.8	1.2						5.9	6.9	2.9	8.2	4.0	5.9	9.6
25-Jul-01	***0.7***		***1.6***						***5.0***	***4.9***	***2.2***	***5.8***	***3.2***	***4.9***	***8.2***
22-Aug-01		6.7	1.8	5.7	5.6	7.0	3.1	5.2	3.5	5.7	2.5	3.8	2.8	4.2	5.5
18-Sep-01	6.7	7.9	0.6	6.7	6.8	4.9	2.2	9.0	2.6	6.9	4.0	3.4	1.0	4.6	7.3
01-Nov-01	8.9	6.1	5.5	10.9	9.4	6.1	8.1	9.2	6.2	8.8	1.4	3.4	5.6	5.3	6.4
28-Nov-01	***9.6***	***8.1***	***12.1***						***10.6***	***12.0***	***3.1***	***5.1***	***9.0***	***7.9***	***9.3***
13-Dec-01	11.7	10.7	11.2	12.6	11.8	10.5	11.2	11.5	10.6	11.4	8.8	8.9	9.7	9.7	10.2
23-Jan-02	10.2	7.2	11.2	17.4	10.5	9.9	9.5	9.5	14.2	9.8	3.8	6.8	7.2	7.3	8.0
28-Feb-02	11.0	6.8	11.0	11.4	10.2	10.2	9.2	9.5	9.6	9.3	4.5	8.4	8.4	7.5	7.9
04-Apr-02	12.4	8.6	10.5	11.4	10.2	10.3	8.2	10.1	8.7	9.6	5.0	12.6	8.6	8.2	8.4
09-Apr-02	***5.6***	***7.2***	***9.2***						***8.0***	***8.4***	***4.1***	***8.0***	***7.5***	***6.8***	***6.9***
03-May-02	9.5	7.9	10.1	10.4	10.5	8.7	8.1	9.5	8.6	9.0	4.9	10.7	8.7	9.6	10.9
15-May-02	***5.9***	***10.6***	***9.3***						***8.8***	***9.6***	***3.7***	***11.3***	***8.5***	***8.8***	***9.7***
07-Jun-02	7.9	8.3	6.7	9.5	7.9	5.5	6.8	6.1	6.7	6.3	5.1	9.0	5.8	7.0	8.3
12-Jun-02	***1.8***	***7.0***	***3.8***						***4.1***	***3.6***	***2.5***	***5.0***	***2.9***	***4.9***	***5.7***

Bold – Agassiz Drainage Committee sampling date and data

Table B.9 (Cont'd): Monthly Dissolved Oxygen Levels (mg/L)

Date	F2	F1	G1	H4	H2	H1	I1	I3	J1	K2	Outfall	K1	Spring	MTN-1
30-May-01	***0.5***	***6.5***	***5.8***		***11.9***	***5.0***	***8.7***	***7.1***	***3.7***	***4.4***		***1.7***		
26-Jun-01	0.2	6.9	7.6	8.7	11.7	6.2	3.2	5.2	4.6	2.6	5.7	4.1	0.4	
23-Jul-01	0.8	7.4	3.6	16.7	13.1	8.9	0.4	6.4	7.9	3.0	5.0	0.3	0.4	
25-Jul-01	***0.8***	***6.3***	***2.7***		***10.4***	***3.9***				***2.6***		***1.4***		
22-Aug-01	2.2	4.6	3.0	5.7	0.2	5.0	3.0	2.4	6.7	0.6	5.9	0.9	0.9	
18-Sep-01	0.8	5.0	3.1	15.4	0.7	0.9		0.3	7.3	0.6	0.4	0.4	0.5	
01-Nov-01	7.4	6.4	5.0	5.3	5.1	6.3	3.6	8.3	4.8	1.2	4.0	1.1	1.9	
28-Nov-01	***7.2***	***8.2***	***7.5***		***8.8***	***11.8***	***4.2***		***8.7***	***0.4***		***1.4***		
13-Dec-01	9.9	9.7	10.1	10.7	10.7	10.8	11.4	11.1	10.5	10.4	13.9	3.7	8.4	
23-Jan-02	9.9	7.3	8.2	9.1	8.3	9.9	10.8	10.0	8.0	4.1	13.3	1.1	2.2	
28-Feb-02	10.1	7.0	7.6	9.8	9.9	9.8	10.6	9.9	4.1	6.3	7.9	6.6	3.2	13.1
04-Apr-02	11.7	8.8	8.1	7.6	7.6	6.3	2.5	10.5	5.4	11.2	0.4	5.5	0.6	10.4
09-Apr-02	***8.6***	***6.3***	***8.9***		***7.2***	***7.3***	***9.2***	***8.9***	***4.5***	***4.5***		***6.6***		***11.16***
03-May-02	8.0	11.2	11.4	14.3	15.0	9.0	8.8	11.7	6.8	3.1	8.4	4.8	0.8	9.6
15-May-02	***3.3***	***10.5***	***10.3***		***10.5***	***7.4***	***1.9***		***3.7***	***5.0***		***4.0***		***7.4***
07-Jun-02	6.1	8.2	9.3	8.4	10.8	7.4	5.5	7.7	5.4	4.2	4.5	5.6	0.8	7.7
12-Jun-02	***0.0***	***5.4***	***6.4***		***6.1***	***4.9***	***0.1***	***3.6***	***1.4***	***2.5***		***4.4***		***4.3***

Bold – Agassiz Drainage Committee sampling date and data

Table B.10: Monthly Temperatures (^{0}C)

Date	A1	B1	C3	CON	C0	C4	C5	C6	M1	C1	D5	D4	D3	D2	D1
26-Jun-01	15.6	18.7	13.0						14.5	12.8	12.3		14.0	14.5	14.0
23-Jul-01	18.1	21.3	15.6						16.9	13.2	13.0	18.4	15.3	15.8	15.2
25-Jul-01	***16.9***		***14.3***						***15.4***	***12.9***	***14.1***	***17.4***	***14.6***	***15.2***	***15.2***
15-Aug-01	***18.4***		***17.4***							***12.7***	***15.3***	***18.9***	***17.8***	***16.5***	***16.6***
22-Aug-01		17.1	15.0	15.9	17.1	16.3	12.1	13.9	15.0	12.4	13.6	16.2	15.4	15.0	14.7
18-Sep-01	15.0	15.6	13.9	14.8	14.2	14.9	11.3	13.4	13.5	12.4	13.9	15.7	14.8	14.1	14.0
01-Nov-01	9.6	10.9	8.9	8.6	9.5	10.3	9.5	9.6	10.1	9.4	10.8	9.9	9.5	9.8	10.2
28-Nov-01	***6.2***	***8.0***	***6.0***						***7.2***	***7.3***	***8.9***	***6.9***	***6.6***	***7.3***	***7.8***
13-Dec-01	5.0	5.0	5.3	5.7	4.8	5.0	5.1	5.2	4.1	5.2	5.7	5.3	5.2	5.1	5.4
23-Jan-02	3.0	5.3	3.7	2.7	4.0	4.3	4.6	4.2	4.4	4.7	6.6	4.4	4.2	5.2	5.3
20-Feb-02	***5.6***	***7.8***	***6.0***						***6.4***	***6.2***	***8.1***	***6.8***		***7.2***	***7.8***
28-Feb-02	3.4	7.9	4.0	2.9	5.3	4.6	4.7	4.8	4.7	5.0	6.8	5.9	6.0	7.0	7.8
04-Apr-02	6.5	14.3	6.3	6.4	9.6	8.1	7.5	8.3	7.9	7.7	8.3	11.1	10.2	11.1	12.1
09-Apr-02	***9.0***	***10.3***	***8.5***						***9.1***	***8.2***	***9.1***	***9.4***	***8.8***	***9.2***	***9.6***
03-May-02	8.4	9.6	7.9	7.3	9.2	8.5	8.5	8.6	8.9	8.6	9.0	8.5	8.5	9.4	9.8
15-May-02	***9.8***	***12.4***	***8.2***						***10.4***	***9.5***	***10.0***	***9.9***	***10.3***	***10.9***	***11.1***
07-Jun-02	12.2	13.2	11.0	9.9	15.2	12.5	10.4	11.9	11.9	11.1	10.5	11.1	11.4	11.5	11.3
12-Jun-02	***15.9***	***17.3***	***15.5***						***15.9***	***14.6***	***12.8***	***15.5***	***15.9***	***15.1***	***14***

Bold – Agassiz Drainage Committee sampling date and data

Table B.10 (Cont'd): Monthly Temperatures (^{0}C)

Date	F2	F1	G1	H4	H2	H1	I1	I3	J1	K2	Outfall	K1	Spring	MTN-1
26-Jun-01	16.4	14.2	14.1	17.2	18.0	19.2	17.6	14.3	14.0	13.3	11.1	13.6	11.4	
23-Jul-01	15.8	14.4	16.5	19.1	21.5	20.2	17.0	17.1	17.5	14.4	18.0	18.4	12.3	
25-Jul-01	***15.3***	***14.2***	***15.9***		***20.6***	***17.6***				***13.2***		***16.8***		
15-Aug-01	***15.4***	***15.9***	***17.9***		***19.4***	***20.7***				***14.9***		***16.6***		
22-Aug-01	14.4	15.0	15.3	17.4	16.8	17.5	16.0	15.7	16.8	14.8	18.8	15.3	12.4	
18-Sep-01	13.6	15.0	14.5	18.0	14.7	16.4		15.5	15.7	14.9	14.6	12.8	12.2	
01-Nov-01	9.5	11.1	10.4	9.6	9.6	10.2	9.8	9.8	9.8	9.0	11.5	8.5	11.6	
28-Nov-01	***5.9***	***7.9***	***7.6***		***5.9***	***5.8***	***5.6***		***7.1***	***6.3***		***6.2***		
13-Dec-01	5.1	6.1	5.4	4.9	5.1	4.9	4.4	5.1	4.5	4.6	5.0	5.2	4.9	
23-Jan-02	2.9	5.9	6.8	4.2	3.9	3.7	2.2	4.0	5.7	2.1	3.0	3.3	9.1	
20-Feb-02		***8.9***	***8.4***		***5.9***	***6.3***	***6.5***	***6.6***	***6.9***	***7.8***		***6.4***		
28-Feb-02	6.1	10.2	11.3	7.8	7.3	7.5	7.3	7.1	6.9	5.9	8.6	5.4	9.4	7.4
04-Apr-02	13.6	15.9	17.3	17.6	15.9	18.2	14.4	14.4	8.9	18.5	10.2	21.0	10.6	11.1
09-Apr-02	***9.8***	***10.7***	***10.6***		***9.5***	***10.5***	***11.1***	***10.7***	***9.3***	***12.5***		***12.4***		***9.5***
03-May-02	9.6	10.4	9.9	10.7	10.8	10.8	12	9.8	8.5	12.8	9.3	11.2	10.8	9.3
15-May-02	***11.5***	***12.1***	***11.3***		***12.9***	***13.7***	***10.8***		***8.9***	***14.4***		***12.5***		***12.1***
07-Jun-02	12.4	11.4	10.9	11.4	11.9	12.7	12.1	11.7	11.6	10.8	12.1	12.6	10.8	12.5
12-Jun-02	***15.1***	***14.3***	***13.4***		***15.7***	***17.2***	***15.8***	***16.1***	***13.9***	***11.9***		***14.6***		***16.7***

Bold – Agassiz Drainage Committee sampling date and data

Table B.11: Monthly pH Readings

Date	A1	B1	C3	CON	C0	C4	C5	C6	M1	C1	D5	D4	D3	D2	D1
26-Jun-01	6.7	6.8	6.5						6.6	6.9	6.4		6.6	6.8	7.1
23-Jul-01	6.2	6.6	6.1						6.5	6.7	6.2	6.8	6.4	6.7	6.8
25-Jul-01	***6.8***		***6.2***						***6.5***	***6.8***	***6.3***	***6.8***	***6.7***	***6.7***	***7.0***
22-Aug-01		6.2	5.9	5.8	6.4	6.7	5.9	6.4	6.2	6.3	5.9	6.4	6.2	6.4	6.2
18-Sep-01	6.1	7.0	6.1	5.8	6.5	6.8	6.1	6.6	6.2	6.4	5.9	6.6	6.0	6.4	6.7
01-Nov-01	6.1	6.2	5.9	5.2	6.0	5.9	5.8	5.9	5.7	5.9	6.2	6.2	5.9	6.2	6.3
28-Nov-01	***6.3***	***6.1***	***6.2***						***5.9***	***6.1***	***6.7***	***6.1***	***6.2***	***6.2***	***6.3***
13-Dec-01	5.6	5.8	5.6	5.4	5.7	5.5	5.6	5.7	5.6	5.7	5.7	5.9	5.8	5.9	5.9
23-Jan-02	6.1	6.3	6.2	6.0	6.4	6.1	6.2	6.3	6.0	6.3	6.1	6.3	6.1	6.3	6.5
28-Feb-02	6.2	6.4	6.1	5.8	6.3	6.0	5.9	6.2	5.9	6.0	6.1	6.2	6.2	6.1	6.3
04-Apr-02	6.4	6.9	6.3	6.6	6.7	6.4	6.0	6.4	6.3	6.1	6.4	7.0	6.5	6.7	6.9
09-Apr-02	***6.3***	***6.2***	***6.6***						***5.9***	***6.2***	***6.7***	***6.5***	***6.5***	***6.4***	***6.5***
03-May-02	6.0	6.9	5.9	6.0	6.6	6.1	6.0	6.5	6.0	6.3	6.3	6.3	6.4	6.3	6.4
15-May-02	***6.8***	***6.4***	***6.2***						***5.9***	***6.3***	***5.9***	***6.7***	***6.7***	***6.7***	***6.9***
07-Jun-02	5.8	6.3	5.8	5.6	6.2	5.9	5.9	6.1	6.0	5.9	5.8	6.2	6.0	6.2	6.5
12-Jun-02	***6.6***	***6.7***	***6.9***						***6.4***	***6.4***	***7.2***	***6.7***	***6.5***	***6.4***	***6.7***

Bold – Agassiz Drainage Committee sampling date and data

Table B.11 (Cont'd): Monthly pH Readings

	F2	F1	G1	H4	H2	H1	I1	I3	J1	K2	Outfall	K1	Spring	MTN-1
26-Jun-01	6.6	6.8	6.4	6.7	6.9	6.5	6.5	6.1	5.4	6.9	6.4	6.9	6.9	
23-Jul-01	6.0	6.7	6.5	6.6	7.0	6.6	6.2	6.5	7.0	6.8	6.9	6.9	6.4	
25-Jul-01	***6.7***	***7.1***	***6.6***		***7.0***	***6.5***				***6.9***		***6.9***		
22-Aug-01	6.0	6.5	6.3	6.4	6.5	6.6	5.4	6.2	7.1	6.7	5.6	6.6	6.4	
18-Sep-01	5.9	6.7	6.3	6.9	6.3	6.4		6.1	7.1	6.6	6.6	6.3	6.6	
01-Nov-01	6.1	6.5	6.3	5.6	5.7	6.1	5.8	6.1	5.5	5.3	5.8	5.6	6.5	
28-Nov-01	***6.3***	***6.5***	***6.5***		***6.4***	***6.7***	***6.4***		***5.9***	***6.7***		***6.6***		
13-Dec-01	5.8	6.1	6.1	5.6	5.7	5.8	5.7	5.8	5.2	5.6	5.5	6.0	6.1	
23-Jan-02	6.1	6.7	6.5	5.9	6.4	6.4	6.2	6.2	5.6	6.2	6.5	6.5	6.4	
28-Feb-02	6.2	6.5	6.4	6.0	6.1	6.1	5.9	6.0	5.5	6.4	6.0	6.1	6.5	6.3
04-Apr-02	6.6	7.0	6.8	6.1	6.2	6.3	6.3	6.6	5.8	6.6	6.8	7.0	6.7	6.5
09-Apr-02	***6.4***	***6.5***	***6.5***		***6.0***	***6.2***	***6.4***	***6.2***	***6.0***	***6.9***		***6.7***		***6.2***
03-May-02	6.4	6.6	6.5	6.4	6.4	6.3	6.1	6.2	5.7	6.2	6.1	6.1	6.5	6.3
15-May-02	***6.5***	***6.9***	***6.9***		***6.9***	***6.9***	***6.6***		***5.9***	***6.9***		***6.9***		***7.2***
07-Jun-02	6.0	6.8	6.8	6.5	6.2	6.3	6.3	6.6	5.7	6.7	6.8	6.8	6.8	6.6
12-Jun-02	***7.0***	***6.7***	***6.7***		***6.4***	***6.5***	***6.4***	***6.5***	***6.2***	***6.7***		***6.5***		***6.7***

Bold – Agassiz Drainage Committee sampling date and data

Appendix C: Sediment Quality Data

Table C.1: Metal Concentrations in Sediments during the Dry and Wet Seasons

	Copper (mg/kg)		Lead (mg/kg)		Zinc (mg/kg)		Managanese (mg/kg)		Iron (% of sample)	
	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
A1	39.2	40.3	39.2	52.4	96.9	125.8	373.1	492.5	3.8	3.8
B1	31.8	43.4	31.8	43.6	79.3	89.4	734.0	779.2	6.6	7.0
C3	38.4	74.9	38.4	41.8	85.5	87.3	322.0	387.6	2.9	3.3
Control		38.5		23.2		57.2		240.0		2.3
C0		38.4		25.9		75.1		445.2		3.5
C4		40.5		37.0		63.0		221.9		3.2
C5		43.4		91.8		143.3		322.6		4.0
C6		42.5		44.2		87.5		513.7		7.2
M1		46.3		46.7		95.7		410.4		6.5
C1	38.3	37.1	38.3	48.0	86.9	84.0	507.2	575.4	8.4	7.3
D5	70.9	71.6	70.9	65.3	98.0	108.0	188.5	204.1	1.9	3.1
D4	44.0	42.3	44.0	36.7	94.2	126.0	302.7	300.8	3.8	3.1
D3	34.3	46.8	34.3	53.3	89.2	104.7	316.5	390.7	4.1	7.0
D2	42.0	50.8	42.0	55.6	62.8	123.6	487.0	1955.4	5.0	8.6
D1	33.4	28.4	33.4	34.6	97.5	67.2	460.8	336.8	7.2	4.0
F2	37.1	40.9	37.1	56.4	191.9	152.9	358.2	284.0	6.0	4.3
F1	30.1	32.4	30.1	70.7	74.8	184.3	439.9	917.0	6.6	9.9
G1	33.5	39.6	33.5	44.3	66.3	95.5	284.8	550.7	3.5	4.7
H4	45.4	105.6	45.4	57.5	84.1	110.3	320.7	436.0	3.7	5.8
H2	65.0	49.4	65.0	52.8	238.2	140.3	306.8	287.3	4.2	3.4
H1	38.8	42.2	38.8	56.3	92.8	128.2	367.4	416.6	4.5	5.6
I1	47.9	53.2	47.9	42.4	84.9	83.0	260.8	264.7	3.1	3.5
I3	75.9	115.6	75.9	65.5	231.0	157.7	514.9	565.2	8.7	7.0
J1	56.4	41.9	56.4	31.9	100.4	59.3	238.3	262.9	4.3	3.2
K2	104.0	97.0	104.0	85.8	0.0	253.3	1296.4	608.5	8.6	4.4
Outfall	148.8	118.9	148.8	144.0	737.7	508.2	835.2	558.7	5.2	4.4
K1	72.2	91.6	72.2	110.3	207.1	289.0	373.0	506.8	3.0	4.4
Spring	27.0	43.4	27.0	36.7	50.3	70.9	370.4	493.1	3.6	5.5

Table C.2: Grain Size Composition Weights and Percent Composition for the Dry Season

Sample Station	Total Weight (g)	Grain Size Mass (g)						Percent Composition					
		> 1 mm	> 0.425 mm	> 0.250 mm	> 0.125 mm	> 0.063 mm	< 0.063 mm	> 1 mm	> 0.425 mm	> 0.250 mm	> 0.125 mm	> 0.063 mm	< 0.063 mm
A1	110.88	6.07	11.23	5.28	6.75	14.31	67.24	5.47	10.13	4.76	6.09	12.91	60.64
B1	116.98	3.38	7.38	4.57	17.65	38.00	46.00	2.89	6.31	3.91	15.09	32.48	39.32
C1	147.84	17.69	22.52	15.31	25.15	31.39	35.78	11.97	15.23	10.36	17.01	21.23	24.20
C3	73.56	5.24	8.49	4.16	9.21	12.66	33.80	7.12	11.54	5.66	12.52	17.21	45.95
D1	420.82	6.85	116.32	219.57	63.12	8.76	6.20	1.63	27.64	52.18	15.00	2.08	1.47
D2	267.17	15.14	17.65	19.76	99.60	74.25	40.77	5.67	6.61	7.40	37.28	27.79	15.26
D3	334.73	53.04	44.38	21.82	103.51	75.48	36.50	15.85	13.26	6.52	30.92	22.55	10.90
D4	166.10	9.56	23.34	18.15	49.13	41.56	24.36	5.76	14.05	10.93	29.58	25.02	14.67
D5	26.42	1.94	3.59	2.17	3.97	5.27	9.48	7.34	13.59	8.21	15.03	19.95	35.88
F1	389.13	0.53	16.88	123.11	196.05	40.28	12.28	0.14	4.34	31.64	50.38	10.35	3.16
F2	176.00	17.91	18.72	14.53	42.15	54.24	28.45	10.18	10.64	8.26	23.95	30.82	16.16
G1	203.70	3.08	4.72	23.12	102.45	40.59	29.74	1.51	2.32	11.35	50.29	19.93	14.60
H1	242.31	7.94	13.45	30.88	97.18	47.62	45.24	3.28	5.55	12.74	40.11	19.65	18.67
H2	92.28	19.29	23.86	11.48	10.02	8.61	19.02	20.90	25.86	12.44	10.86	9.33	20.61
I1	168.28	9.64	11.95	7.33	34.39	56.34	48.63	5.73	7.10	4.36	20.44	33.48	28.90
I3	58.19	19.80	15.00	5.21	5.05	4.41	8.72	34.03	25.78	8.95	8.68	7.58	14.99
J1	130.22	16.37	21.23	13.58	19.05	24.81	35.18	12.57	16.30	10.43	14.63	19.05	27.02
K1	60.45	11.00	18.21	9.67	9.48	5.43	6.66	18.20	30.12	16.00	15.68	8.98	11.02
K2	33.69	7.88	7.22	3.33	4.58	4.37	6.31	23.39	21.43	9.88	13.59	12.97	18.73
M1	95.97	6.72	11.07	5.73	7.83	15.96	48.66	7.00	11.53	5.97	8.16	16.63	50.70
H4	101.90	18.06	17.49	9.33	12.07	8.95	36.00	17.72	17.16	9.16	11.84	8.78	35.33
Outfall	99.51	17.72	31.87	26.31	15.37	4.13	4.11	17.81	32.03	26.44	15.45	4.15	4.13
Spring	416.27	0.96	2.03	15.03	306.19	75.69	16.37	0.23	0.49	3.61	73.56	18.18	3.93

Table C.3: Grain Size Composition Weights and Percent Composition for the Wet Season

Sampling Station	Total Weight (g)	Grain Size Mass (g)						Percent Composition					
		> 1 mm	> 0.425 mm	> 0.250 mm	> 0.125 mm	> 0.063 mm	< 0.063 mm	> 1 mm	> 0.425 mm	> 0.250 mm	> 0.125 mm	> 0.063 mm	< 0.063 mm
A1	168.40	9.08	19.02	12.50	19.88	33.40	74.52	5.39	11.29	7.42	11.81	19.83	44.25
B1	252.21	1.03	2.81	6.15	124.33	75.62	42.27	0.41	1.11	2.44	49.30	29.98	16.76
C3	113.11	3.00	8.22	5.43	13.86	22.88	59.72	2.65	7.27	4.80	12.25	20.23	52.80
Control	146.95	11.76	18.59	12.06	22.52	36.28	45.74	8.00	12.65	8.21	15.32	24.69	31.13
C0	168.35	1.60	7.66	9.48	39.20	62.68	47.73	0.95	4.55	5.63	23.28	37.23	28.35
C4	300.04	20.57	32.59	17.80	38.31	53.86	136.91	6.86	10.86	5.93	12.77	17.95	45.63
C5	254.27	7.16	13.80	11.86	64.44	88.11	68.90	2.82	5.43	4.66	25.34	34.65	27.10
C6	203.60	0.37	2.36	5.74	46.89	67.67	80.57	0.18	1.16	2.82	23.03	33.24	39.57
M1	119.73	2.43	6.75	8.72	27.93	34.40	39.50	2.03	5.64	7.28	23.33	28.73	32.99
C1	267.26	5.51	8.77	16.39	113.29	78.52	44.78	2.06	3.28	6.13	42.39	29.38	16.76
D5	49.06	2.80	5.51	4.48	8.57	10.78	16.92	5.71	11.23	9.13	17.47	21.97	34.49
D4	245.38	7.98	15.69	19.28	61.81	76.76	63.86	3.25	6.39	7.86	25.19	31.28	26.02
D3	155.69	6.42	7.30	7.89	50.55	41.90	41.63	4.12	4.69	5.07	32.47	26.91	26.74
D2	166.61	1.26	3.03	6.83	39.08	54.43	61.98	0.76	1.82	4.10	23.46	32.67	37.20
D1	393.01	5.82	129.38	208.70	43.17	4.37	1.57	1.48	32.92	53.10	10.98	1.11	0.40
F2	123.38	1.60	5.30	5.30	4.89	46.39	59.90	1.30	4.30	4.30	3.96	37.60	48.55
F1	363.49	8.83	20.26	125.33	165.51	29.53	14.03	2.43	5.57	34.48	45.53	8.12	3.86
G1	298.86	4.52	50.43	134.03	81.45	20.15	8.28	1.51	16.87	44.85	27.25	6.74	2.77
H4	124.21	25.58	27.68	16.75	17.61	12.48	24.11	20.59	22.28	13.49	14.18	10.05	19.41
H2	117.97	8.79	12.64	18.38	21.81	20.52	35.83	7.45	10.71	15.58	18.49	17.39	30.37
H1	117.17	5.87	11.69	9.34	19.80	22.47	48.00	5.01	9.98	7.97	16.90	19.18	40.97
I1	208.72	6.65	15.91	9.61	25.31	64.70	86.54	3.19	7.62	4.60	12.13	31.00	41.46
I3	142.87	9.94	16.86	11.28	19.77	25.29	59.73	6.96	11.80	7.90	13.84	17.70	41.81
J1	206.38	4.56	22.06	19.50	43.80	56.36	60.10	2.21	10.69	9.45	21.22	27.31	29.12
K2	148.40	30.11	51.56	23.87	17.21	11.63	14.02	20.29	34.74	16.08	11.60	7.84	9.45
Outfall	116.39	14.15	27.02	23.32	21.23	27.13	3.54	12.16	23.22	20.04	18.24	23.31	3.04
K1	83.95	20.73	14.97	8.36	14.18	12.2	13.51	24.69	17.83	9.96	16.89	14.53	16.09
Spring	257.24	0.84	1.95	16.52	226.66	10.62	0.65	0.33	0.76	6.42	88.11	4.13	0.25

Table C.4: Loss on Ignition Values for the Dry and Wet Season

Station	Dry Season (June, 2001)	Wet Season (Feb, 2002)
A1	7.7	11.3
B1	5.7	2.5
C3	15.4	13.0
Control		18.8
C0		7.5
C4		4.8
C5		5.6
C6		5.6
M1	10.0	13.2
C1	2.2	4.5
D5	23.7	21.4
D4	22.8	8.0
D3	6.6	8.3
D2	2.9	9.4
D1	2.6	1.0
F2	6.2	8.5
F1	1.6	1.8
G1	3.1	1.2
H4	23.6	14.8
H2	14.8	10.9
H1	8.8	11.6
I1	9.8	7.7
I3	9.0	14.8
J1	12.6	2.7
K2	21.4	7.5
Outfall	12.8	16.1
K1	38.9	16.1
Spring	1.3	1.7

Appendix D: Water and Sediment Quality Correlations

Table D.1: Dry season correlations between water and sediment quality variables

	PO_4^{3-}-P	NO_3^--N	NO_2^--N	NH_3^+-N	Chl a	DO	Sp. Cond.	pH	Turbidity	Cu	Pb	Zn	Mn	Fe	LOI	% Silt/Clay
PO_4^{3-}-P	1.00															
NO_3^--N	0.30	1.00														
NO_2^--N	**0.54**	**0.38**	1.00													
NH_3^+-N	**0.51**	**0.36**	**0.57**	1.00												
Chl a	**0.55**	0.15	**0.57**	0.13	1.00											
DO	-0.12	-0.06	-0.30	0.05	-0.24	1.00										
Sp. Cond.	0.01	0.06	0.22	0.21	0.11	**-0.51**	1.00									
pH	0.10	-0.06	-0.14	0.13	-0.10	-0.07	**0.61**	1.00								
Turbidity	**0.46**	-0.07	**0.70**	**0.34**	**0.54**	-0.10	-0.05	-0.22	1.00							
Cu	**0.43**	**0.62**	**0.41**	0.21	**0.49**	-0.18	0.06	-0.20	0.14	1.00						
Pb	**0.43**	**0.62**	**0.41**	0.21	**0.49**	-0.18	0.06	-0.20	0.14	**1.00**	1.00					
Zn	**0.46**	0.41	0.29	-0.19	**0.53**	-0.09	-0.13	-0.15	0.12	**0.69**	**0.69**	1.00				
Mn	-0.07	-0.10	-0.11	-0.10	-0.22	0.00	0.13	**0.40**	-0.07	0.01	0.01	0.01	1.00			
Fe	-0.18	-0.17	-0.04	-0.10	0.04	0.14	-0.05	0.16	0.12	-0.04	-0.04	0.16	**0.72**	1.00		
LOI	**0.47**	**0.66**	**0.45**	0.22	**0.58**	-0.01	-0.03	-0.10	0.14	**0.78**	**0.78**	**0.54**	-0.32	**-0.42**	1.00	
% Silt/Clay	0.06	0.09	**0.35**	0.08	0.27	0.03	**-0.60**	**-0.63**	**0.47**	0.22	0.22	-0.01	-0.23	-0.26	**0.36**	1.00

*Significant correlation at α=0.05, for one tailed test

Table D.2: Wet season correlations between water and sediment quality variables

	PO_4^{3-}-P	NO_3^--N	NO_2^--N	NH_3^+-N	Chl a	DO	Sp. Cond.	pH	Turbidity	Cu	Pb	Zn	Mn	Fe	LOI	% Silt/Clay
PO_4^{3-}-P	1.00															
NO_3^--N	0.16	1.00														
NO_2^--N	**0.61**	0.19	1.00													
NH_3^+-N	**0.74**	0.11	**0.75**	1.00												
Chl a	**0.35**	0.14	**0.44**	**0.57**	1.00											
DO	-0.25	-0.08	-0.06	-0.27	-0.38	1.00										
Sp. Cond.	**0.55**	-0.18	**0.39**	**0.60**	**0.34**	**-0.59**	1.00									
pH	0.21	-0.27	-0.10	-0.27	0.05	**-0.44**	**0.56**	1.00								
Turbidity	**0.35**	-0.16	**0.55**	**0.67**	**0.58**	0.07	**0.41**	0.21	1.00							
Cu	**0.50**	0.19	0.27	**0.38**	**0.49**	-0.06	0.17	-0.24	0.02	1.00						
Pb	**0.32**	0.05	0.26	0.28	**0.35**	-0.22	0.28	-0.04	0.09	**0.54**	1.00					
Zn	0.29	0.06	0.30	**0.32**	**0.49**	-0.23	**0.33**	0.15	0.26	**0.48**	**0.88**	1.00				
Mn	0.09	**-0.56**	0.00	0.11	-0.02	-0.26	0.29	**0.50**	0.07	0.10	**0.38**	**0.39**	1.00			
Fe	0.02	**-0.34**	0.03	0.01	-0.16	0.20	0.26	0.31	-0.01	0.05	**0.40**	0.29	**0.81**	1.00		
LOI	0.11	0.26	0.14	0.06	**0.43**	**-0.36**	-0.13	**0.46**	-0.06	**0.58**	**0.38**	**0.37**	-0.20	-0.22	1.00	
% Silt/Clay	-0.09	**0.45**	-0.06	-0.20	0.09	**0.59**	**-0.58**	**-0.39**	-0.07	0.10	-0.09	-0.09	**-0.41**	-0.26	**0.42**	1.00

*Significant correlation at α=0.05, for one tailed test

Appendix E: Land Uses

Table E.1: Total area of various land uses within the AHHS Watershed.

Land Use	Acres	Hectares	Percent of Watershed Area
Forest	6941.1	2809.0	44.15
Corn	1260.5	510.1	8.02
Hay/silage	4789.5	1938.2	30.46
Pastures	226.7	91.7	1.44
Cranberries	60.3	24.4	0.38
Fruit	31.1	12.6	0.20
Hazelnuts	281.9	114.1	1.79
Uncultivated	55.7	22.5	0.35
UBC Experimental Farm	50.4	20.4	0.32
Natural Shrubs	446.1	180.5	2.84
General Recreation	9.8	4.0	0.06
Golf Course	76.0	30.7	0.48
Sports Field	45.7	18.5	0.29
Heavy / Light Industry	37.6	15.2	0.24
Gravel	145.6	58.9	0.93
Commercial	42.9	17.4	0.27
Institutional	124.1	50.2	0.79
Horticulture	189.9	76.8	1.21
Residential Homes	537.5	217.5	3.42
Hobby Farms	278.8	112.8	1.77
RV Parks	39.9	16.1	0.25
Hotel/Motel/Apartments	52.3	21.2	0.33
Total Area	***15723.4***	***6362.8***	***100.0***

Table E.2: Total area of various land uses within a 100 m buffer around major watercourses throughout the AHHS Watershed.

Land Use	Acres	Hectares	Percent of Watershed Area
Forest	460.0	186.2	2.93
Corn	210.6	85.2	1.34
Hay/silage	1026.8	415.5	6.53
Pastures	35.2	14.2	0.22
Cranberries	20.9	8.5	0.13
Fruit	12.1	4.9	0.08
Hazelnuts	32.9	13.3	0.21
Uncultivated	16.6	6.7	0.11
Natural Shrubs	88.7	35.9	0.56
Golf Course	12.8	5.2	0.08
Sports Field	0.3	0.1	0.00
Heavy / Light Industry	8.7	3.5	0.06
Gravel	8.1	3.3	0.05
Institutional	43.7	17.7	0.28
Horticulture	73.8	29.9	0.47
Residential Homes	70.9	28.7	0.45
Hobby Farms	13.2	5.4	0.08
RV Parks	3.7	1.5	0.02
Hotel/Motel/Apartments	460.0	186.2	2.93
Total Area	***2139.1***	***865.7***	***13.61***

Table E.3: Area within 100 m buffer upstream from sampling stations within the AHHS Watershed.

Sampling Station	Tributary	Buffer Area (hectares)	Percentage of Watershed	Total Upstream Buffer Area (hectares)	Percentage of Watershed
A1	Tributary to Miami	30.3	0.48	30.3	0.48
B1	McPherson Road	16.1	0.25	16.1	0.25
C3	Miami Creek	12.0	0.19	12.0	0.19
Control	Miami Creek	6.0	0.09	6.0	0.09
C0	Miami Creek	75.0	1.18	244.6	3.84
C4	Miami Creek	27.2	0.43	39.1	0.62
C5	Miami Creek	6.6	0.10	80.1	1.26
C6	Miami Creek	17.5	0.27	139.3	2.19
M1	Miami Creek	23.1	0.36	67.5	1.06
C1	Miami Creek	25.6	0.40	121.9	1.92
D5	McCallum Slough	3.0	0.05	3.0	0.05
D4	McCallum Slough	29.7	0.47	32.7	0.51
D3	McCallum Slough	29.3	0.46	61.9	0.97
D2	McCallum Slough	20.3	0.32	82.3	1.29
D1	McCallum Slough	32.8	0.52	115.1	1.81
F2	Westlin Ditch	50.3	0.79	50.3	0.79
F1	Westlin Ditch	22.9	0.36	73.2	1.15
G1	Miami Slough Ditch	24.8	0.39	24.8	0.39
H4	Hogg Slough	30.4	0.48	30.4	0.48
H2	Hogg Slough	46.2	0.73	76.7	1.21
H1	Hogg Slough	43.2	0.68	154.0	2.42
I1	Clark Ditch	18.1	0.28	18.1	0.28
I3	Clark Ditch	16.0	0.25	34.1	0.54
J1	Miami Slough Tributary	5.2	0.08	5.2	0.08
MTN-1	Mountain Slough	173.1	2.72	574.3	9.03

Table E.4: Composition of land uses within a 100 m buffer upstream of sampling stations

	A1	B1	C3	Control	C0	C4	C5	C6
Forest	160737.2	43904.9	38054.6	50445.1	1107036.1	38054.6	113588.0	424634.5
Corn Fields			22338.6		162203.3	24491.0	162203.3	162203.3
Hay / Silage	96973.0	88776.4	55627.6		608868.8	315710.6	405655.7	511804.6
Pastures								
Cranberries								
Fruit								
Hazelnuts	44866.6				44866.6			
Uncultivated								
Natural Growth					2789.3			
General Recreation								
Golf Course					51776.0		12752.0	51776.0
Sports Field								
Industrial								
Gravel Pit								
Institutional					3155.7			
Horticulture								
Residential Homes		28756.1	3755.0		293340.3	4196.3	4196.3	119839.7
Hobby Farms				9783.7	103026.3	8893.7	103026.3	103026.3
RV Park					53529.5			19880.1
Hotel/Apartments					15082.5			

Table E.4 (cont'd): Composition of land uses within a 100 m buffer upstream of sampling stations

	M1	C1	D5	D4	D3	D2	D1	F2
Forest	41667.3	331909.0		38269.5	38269.5	52629.3	52629.3	53570.2
Corn Fields	162203.3	162203.3		12917.4	132491.5	148611.2	148611.2	41224.8
Hay / Silage	405655.7	511804.6	29618.3	261817.0	335598.8	395856.9	571060.5	367238.8
Pastures					95.2	28359.7	28359.7	
Cranberries						81966.0	84657.1	
Fruit					30824.9	30824.9	30824.9	
Hazelnuts								41106.6
Uncultivated					39849.0	39849.0	67296.9	
Natural Shurbs					24932.4	25055.8	52276.3	
General Recreation								
Golf Course	142.7	51776.0						
Sports Field								
Industrial							1213.6	
Gravel Pit						2040.2	2040.2	
Institutional								
Horticulture							62250.0	
Residential Homes	4196.3	45203.9						
Hobby Farms	60891.7	103026.3		13775.4	17386.5	17386.5	49827.5	
RV Park		12592.3						
Hotel/Apartments								

Table E.4 (cont'd): Composition of land uses within a 100 m buffer upstream of sampling stations

	F1	G1	H4	H2	H1	I1	I3	J1
Forest	53570.2			130637.7	242958.8	112321.1	112321.1	
Corn Fields	77107.0	623.5	14899.7	83372.2	208106.6	22307.7	68598.4	27923.7
Hay / Silage	559547.9	210864.9	191505.9	422500.8	924652.5	46242.3	142174.6	24321.7
Pastures			17571.9	49824.1	49824.1			
Cranberries								
Fruit					17970.3		17970.3	
Hazelnuts	41106.6		47133.6	47133.6	47133.6			
Uncultivated								
Natural Shurbs	658.8	475.3	33315.0	33315.0	33315.0			
General Recreation								
Golf Course								
Sports Field								
Industrial								
Gravel Pit								
Institutional								
Horticulture		35873.2			299.7			
Residential Homes								
Hobby Farms					15668.2			
RV Park								
Hotel/Apartments								

Table E.4 (cont'd): Composition of land uses within a 100 m buffer upstream of sampling stations

	K2	Outfall	K1	Spring	MTN-1
Forest					866977.6
Corn Fields	46912.9	165701.7	185200.8		573675.6
Hay / Silage	64623.4	214553.5	421870.4		3266683.0
Pastures					142397.9
Cranberries					84657.1
Fruit					66765.6
Hazelnuts					88240.2
Uncultivated					67296.9
Natural Shurbs					356011.4
General Recreation					
Golf Course					
Sports Field					
Industrial					1213.6
Gravel Pit					35285.4
Institutional					29787.8
Horticulture		26454.1	78413.7		98422.9
Residential Homes		385.2	5415.2		
Hobby Farms			118417.6		65495.6
RV Park					
Hotel/Apartments					

Table E.5: Area of contributing areas from sampling stations within the AHHS Watershed.

Sampling Station	Tributary	Contributing Area (hectares)	Percentage of Watershed	Total Upstream Contributing Area (hectares)	Percentage of Watershed
A1	Tributary to Miami	199.9	3.1	199.9	3.1
B1	McPherson Road	103.1	1.6	103.1	1.6
C3	Miami Creek	190.1	3.0	190.1	3.0
Control	Miami Creek	88.7	1.4	88.7	1.4
C0	Miami Creek	323.6	5.1	1354.2	21.3
C4	Miami Creek	82.8	1.3	272.9	4.3
C5	Miami Creek	12.4	0.2	532.8	8.4
C6	Miami Creek	46.2	0.7	830.7	13.1
M1	Miami Creek	113.5	1.8	431.7	6.8
C1	Miami Creek	268.2	4.2	904.1	14.2
D5	McCallum Slough	192.9	3.0	192.9	3.0
D4	McCallum Slough	97.8	1.5	290.7	4.6
D3	McCallum Slough	264.8	4.2	555.5	8.7
D2	McCallum Slough	250.2	3.9	805.6	12.7
D1	McCallum Slough	101.7	1.6	907.4	14.3
F2	Westlin Ditch	170.9	2.7	170.9	2.7
F1	Westlin Ditch	33.6	0.5	204.5	3.2
G1	Miami Slough Ditch	59.5	0.9	59.5	0.9
H4	Hogg Slough	268.9	4.2	268.9	4.2
H2	Hogg Slough	163.9	2.6	432.8	6.8
H1	Hogg Slough	133.4	2.1	797.6	12.5
I1	Clark Ditch	180.8	2.8	180.8	2.8
I3	Clark Ditch	50.6	0.8	231.4	3.6
J1	Miami Slough Tributary	45.3	0.7	45.3	0.7
MTN-1	Mountain Slough	1084.0	17.0	3052.9	48.0

Table E.6: Composition of land uses upstream of sampling stations in contributing areas

	A1	B1	C3	Control	C0	C4	C5
Forest	1506779.4	213298.6	1479246.9	874903.7	9748549.7	1575266.2	2972605.3
Corn Fields	11697.1	25352.0	40823.7		458805.0	77458.6	447107.9
Hay / Silage	417655.3	543096.6	380797.4		1867028.5	945691.0	1448576.6
Pastures					22510.6	22510.6	22510.6
Cranberries							
Fruit							
Hazelnuts	62881.8	136304.9			146464.1	34460.6	83582.3
Uncultivated							
UBC Experimental Farm							
Natural Growth		61369.9			16857.3		
Golf Course					62288.9		62288.9
Sports Field							
Industrial							
Gravel Pit							
Commercial							
Institutional					38523.6	2024.0	2024.0
Horticulture							
Residential Homes		51824.4	105.4		804882.2	18272.8	18272.8
Hobby Farms		118.5		12220.4	294751.8	53631.5	271190.7
RV Park					79694.6		
Hotel/Apartments					2110.3		

Table E.6 (cont'd): Composition of land uses upstream of sampling stations in contributing areas

	C6	M1	C1	D5	D4	D3	D2
Forest	5727940.4	2074664.5	5489959.5	842662.7	909126.2	2320914.9	3624134.3
Corn Fields	447107.9	447107.9	490138.1	77492.9	142839.4	480094.6	524945.3
Hay / Silage	1448576.6	1448576.6	2039878.7	992869.8	1733514.1	2213690.3	2768082.4
Pastures	22510.6	22510.6	22510.6	1202.5	1202.5	40556.6	68360.4
Cranberries							240726.8
Fruit						80134.5	80134.5
Hazelnuts	83582.3	83582.3	219887.2	4953.5	4953.5	125801.4	125801.4
Uncultivated						112398.6	112398.6
UBC Experimental Farm							
Natural Growth	0.1		61404.3			34163.5	82992.5
Golf Course	62288.9	974.0	310194.5				
Sports Field							
Industrial							
Gravel Pit							241755.4
Commercial							40038.5
Institutional	2024.0	2024.0	2024.0	10044.9	14076.6	20488.9	20488.9
Horticulture							
Residential Homes	204972.5	18272.8	85869.2		31.6	31.6	31.6
Hobby Farms	271190.7	219627.8	308075.5		101102.6	126449.1	126449.1
RV Park	37134.7		11289.6				
Hotel/Apartments							

Table E.6 (cont'd): Composition of land uses upstream of sampling stations in contributing areas

	D1	F2	F1	G1	H4	H2	H1
Forest	3653085.1	246622.9	246622.9		1032358.6	1490081.6	1953245.7
Corn Fields	526875.8	224549.8	277118.1	655.1	210588.9	536553.4	889754.2
Hay / Silage	3233547.7	1084594.6	1360473.2	518611.4	959569.2	1482921.8	3454224.7
Pastures	68360.4				18656.7	350450.7	382608.3
Cranberries	245141.1						
Fruit	92209.2			705.0			24625.9
Hazelnuts	125801.4	81582.9	81582.9		248617.1	248617.1	628956.5
Uncultivated	198261.5						
UBC Experimental Farm							
Natural Growth	172773.8		7175.9	427.4	197010.5	197014.5	197178.2
Golf Course							
Sports Field	3704.6						
Industrial	38466.6						
Gravel Pit	316811.4						
Commercial	40038.5						
Institutional	118604.7						423.0
Horticulture	80918.4			74704.1			47463.9
Residential Homes	31.6						88959.2
Hobby Farms	158886.2	72089.9	72089.9		22213.7	22213.7	308209.5
RV Park							
Hotel/Apartments							

Table E.6 (cont'd): Composition of land uses upstream of sampling stations in contributing areas

	I1	I3	J1	MTN-1
Forest	368208.3	463164.1	251939.6	13847824.5
Corn Fields	66064.5	222753.8	81674.9	1911761.1
Hay / Silage	638721.7	868518.3	111120.5	10322008.8
Pastures				515182.8
Cranberries				245141.1
Fruit		24565.0		117540.1
Hazelnuts	380339.3	380339.3		836340.7
Uncultivated				198261.5
UBC Experimental Farm				
Natural Growth				797058.7
Golf Course				
Sports Field				8841.0
Industrial				38466.6
Gravel Pit				509741.0
Commercial				40038.5
Institutional	423.0	423.0		309476.1
Horticulture				203086.3
Residential Homes	88959.2	88959.2		88990.8
Hobby Farms	265456.2	265456.2	8488.3	539185.6
RV Park				
Hotel/Apartments				

Appendix F: Bioassay and Resin Site Conditions

Table F.1: Moss bioassay site conditions.

Station	Tributary	Flow	Shading	Periphyte Coverage of Apparatus			
				17-May-02	24-May-02	31-May-02	07-Jun-02
Control	Miami Creek	v. slow	negligible	slight	slight	negligible	negligible
C3	Miami Creek	slow	negligible	slight	slight	slight	slight
C4	Miami Creek	moderate	slight	moderate	moderate	slight	negligible
C5	Miami Creek	moderate	significant	severe	severe	severe	slight
C1	Miami Creek	moderate	moderate	moderate	severe	slight	negligible
D5	McCallum Slough	v. slow	significant	moderate	moderate	moderate	moderate
D1	McCallum Slough	moderate	negligible	severe	severe	severe	severe
D1-R	McCallum Slough	moderate	negligible	severe	severe	severe	severe
F2	Westlin Ditch	slow	negligible	severe	severe	severe	severe
F1	Westlin Ditch	slow	negligible	severe	severe	severe	moderate
G1	Miami Slough Ditch	slow	moderate	moderate	severe	severe	severe
H4	Hogg Slough	moderate	negligible	negligible	negligible	moderate	slight
H2	Hogg Slough	slow	negligible	moderate	slight	severe	moderate
H2-R	Hogg Slough	slow	negligible	moderate	slight	severe	moderate
I3	Clark Ditch	slow	moderate	slight	negligible	slight	slight
K2	Agassiz Slough	v. slow	moderate	severe	severe	severe	slight
Spring	Humphrey Road	slow	negligible	moderate	moderate	moderate	moderate
MTN-1	Mountain Slough	moderate	moderate	moderate	moderate	slight	slight

Table F.2: XAD-7 Resin Apparatus Site Conditions.

Station	Tributary	Tributary Flow	Periphyte Coverage of Apparatus			
			17-May-02	24-May-02	31-May-02	07-Jun-02
Control	Miami Creek	v. slow	slight	slight	slight	slight
CO	Miami Creek	v. slow	negligible	moderate	moderate	moderate
D1	McCallum Slough	moderate	moderate	moderate	moderate	moderate
D1-R	McCallum Slough	moderate	moderate	moderate	moderate	moderate
F1	Westlin Ditch	slow	moderate	severe	severe	severe
H1	Hogg Slough	v. slow	slight	moderate	moderate	moderate
K1	Agassiz Slough	v. slow	moderate	-	-	-

Appendix G: Bioassay Data

Appendix G.1: Moss Bioassay Raw Data

Station	Stem #	Length (mm)	Average Length	Standard Deviation	Absorbance 648.2	Absorbance 664.9	Dry Weight (mg)	Average Dry Weight	Dry Weight S.D.	Chl A	Average Chl A	Chl A S.D.	Chl B	Average Chl. B	Chl B S.D.
I3	1	37	38.3	6.9	0.020	0.047	0.52	0.68	0.33	5.72	7.08	2.37	0.53	0.67	0.34
	2	44			0.033	0.078	0.50			9.89			0.89		
	3	34			0.014	0.035	0.43			5.21			0.33		
	4	43			0.025	0.061	0.51			7.62			0.57		
	5	21			0.080	0.164	1.29			7.85			1.34		
	6	42			0.047	0.091	1.05			5.28			1.10		
	7	38			0.018	0.042	0.37			7.18			0.69		
	8	42			0.028	0.061	1.05			3.63			0.49		
	9	44			0.031	0.083	0.46			11.67			0.37		
	10	38			0.024	0.060	0.57			6.73			0.42		
MTN-1	1	32	38.3	4.8	0.030	0.063	0.36	0.37	0.06	10.85	12.03	2.59	1.69	2.39	2.19
	2	32			0.022	0.032	0.30			6.04			2.70		
	3	37			0.061	0.076	0.36			11.32			7.16		
	4	36			0.029	0.065	0.33			12.37			1.46		
	5	40			0.037	0.089	0.50			11.31			0.92		
	6	42			0.026	0.059	0.28			13.26			1.48		
	7	46			0.030	0.074	0.36			13.12			0.90		
	8	41			0.036	0.088	0.41			13.68			1.00		
	9	43			0.032	0.077	0.40			12.24			0.99		
	10	34			0.063	0.102	0.37			16.09			5.56		
G1	1	32	31.5	4.5	0.022	0.057	0.49	0.73	0.46	7.48	9.25	2.50	0.34	1.08	1.23
	2	36			0.052	0.103	0.57			11.07			2.13		
	3	32			0.065	0.123	0.69			10.81			2.41		
	4	31			0.032	0.076	0.55			8.76			0.77		
	5	32			0.034	0.099	0.61			10.62			####		
	6	35			0.039	0.109	0.71			9.99			0.13		
	7	32			0.030	0.070	0.44			10.06			0.97		
	8	20			0.061	0.180	2.02			5.84			####		
	9	35			0.020	0.046	0.59			4.92			0.51		
	10	30			0.074	0.130	0.60			12.92			3.59		

Appendix G.1: Moss Bioassay Raw Data (Cont'd)

Station	Stem #	Length (mm)	Average Length	Standard Deviation	Absorbance 648.2	Absorbance 664.9	Dry Weight (mg)	Average Dry Weight	Dry Weight S.D.	Chl A	Average Chl A	Chl A S.D.	Chl B	Average Chl. B	Chl B S.D.
Spring	1	34	33.9	3.0	0.016	0.039	0.51	0.53	0.12	4.87	5.89	0.93	0.36	0.42	0.50
	2	36			0.035	0.075	0.64			7.30			1.05		
	3	35			0.011	0.032	0.43			4.87			0.00		
	4	37			0.017	0.046	0.49			6.08			0.16		
	5	34			0.009	0.028	0.34			5.43			0.00		
	6	27			0.027	0.075	0.69			7.07			0.11		
	7	35			0.033	0.063	0.62			6.18			1.34		
	8	33			0.017	0.042	0.50			5.36			0.37		
	9														
	10														
C1	1	41	32.4	4.7	0.026	0.068	0.34	0.47	0.09	12.88	10.66	1.31	0.54	1.15	0.62
	2	34			0.027	0.056	0.33			10.50			1.71		
	3	25			0.042	0.118	0.60			12.80			0.14		
	4	37			0.040	0.077	0.47			9.98			2.11		
	5	34			0.030	0.061	0.40			9.40			1.65		
	6	32			0.034	0.072	0.44			10.16			1.53		
	7	34			0.041	0.092	0.53			10.90			1.28		
	8	31			0.033	0.078	0.56			8.83			0.80		
	9	29			0.036	0.082	0.50			10.33			1.13		
	10	27			0.036	0.091	0.54			10.80			0.62		
D5	1	40	30.5	4.4	0.029	0.079	0.47	0.72	0.28	10.89	8.53	2.25	0.26	0.46	0.36
	2	25			0.021	0.058	0.96			3.92			0.07		
	3	33			0.027	0.074	0.56			8.57			0.18		
	4	29			0.032	0.069	0.54			7.96			1.12		
	5	32													
	6	27			0.031	0.074	0.45			10.44			0.89		
	7	33			0.031	0.086	0.63			8.87			0.15		
	8	30			0.041	0.102	0.63			10.35			0.67		
	9	30			0.050	0.124	1.27			6.24			0.41		
	10	26			0.054	0.142	0.96			9.54			0.38		

Appendix G.1: Moss Bioassay Raw Data (Cont'd)

Station	Stem #	Length (mm)	Average Length	Standard Deviation	Absorbance 648.2	Absorbance 664.9	Dry Weight (mg)	Average Dry Weight	Dry Weight S.D.	Chl A	Average Chl A	Chl A S.D.	Chl B	Average Chl. B	Chl B S.D.
D1-R	1	20	30.8	4.6	0.171	0.478	1.95	0.61	0.53	15.95	11.44	2.88	0.21	2.05	1.04
	2	34			0.034	0.065	0.30			13.17			2.85		
	3	35			0.044	0.087	0.36			14.80			2.86		
	4	32			0.040	0.087	0.52			10.45			1.41		
	5	35			0.032	0.064	0.53			7.41			1.38		
	6	31			0.049	0.090	0.44			12.33			3.01		
	7	31			0.035	0.061	0.34			10.68			3.03		
	8	29			0.046	0.100	0.77			8.11			1.10		
	9	30			0.026	0.047	0.28			10.08			2.58		
	10														
D1	1	34	31.1	5.0	0.038	0.075	0.44	0.49	0.19	10.44	10.78	2.86	2.03	1.53	0.79
	2	33			0.032	0.067	0.34			12.21			1.92		
	3	39			0.022	0.047	0.57			5.13			0.75		
	4	32			0.025	0.053	0.35			9.41			1.41		
	5	32			0.033	0.061	0.28			13.15			3.15		
	6	30			0.068	0.146	0.89			10.22			1.46		
	7	31			0.026	0.062	0.39			10.09			0.86		
	8	33			0.023	0.061	0.40			9.85			0.35		
	9	20			0.081	0.186	0.72			16.29			1.71		
	10	27			0.042	0.089	0.50			11.06			1.66		
C3	1	34	30.6	8.5	0.054	0.133	0.46	0.69	0.26	18.45	11.54	3.00	1.29	0.76	0.33
	2	45			0.025	0.057	0.40			8.97			0.98		
	3	38			0.042	0.104	0.51			13.02			0.87		
	4	20			0.072	0.166	1.12			9.35			0.96		
	5	36			0.038	0.093	0.56			10.59			0.77		
	6	26			0.053	0.136	0.68			12.85			0.65		
	7	25			0.047	0.123	0.84			9.43			0.39		
	8	20			0.058	0.161	1.08			9.69			0.16		
	9	31			0.043	0.106	0.59			11.47			0.79		
	10														

Appendix G.1: Moss Bioassay Raw Data (Cont'd)

Station	Stem #	Length (mm)	Average Length	Standard Deviation	Absorbance 648.2	Absorbance 664.9	Dry Weight (mg)	Average Dry Weight	Dry Weight S.D.	Chl A	Average Chl A	Chl A S.D.	Chl B	Average Chl. B	Chl B S.D.
F2	1	39	35.8	3.9	0.025	0.043	0.30	0.44	0.23	8.50	8.93	1.48	2.50	1.83	0.96
	2	40			0.032	0.059	0.50			7.12			1.72		
	3	36			0.025	0.050	0.37			8.30			1.54		
	4	37			0.050	0.122	1.03			7.55			0.56		
	5	33			0.022	0.053	0.33			10.21			0.82		
	6	36			0.043	0.079	0.45			10.58			2.58		
	7	33			0.023	0.039	0.20			11.52			3.53		
	8	42			0.030	0.060	0.50			7.37			1.37		
	9	33			0.035	0.059	0.40			8.70			2.71		
	10	29			0.022	0.051	0.34			9.47			0.95		
K2	1	29	33.1	3.7	0.019	0.057	0.51	0.55	0.10	7.34	9.73	1.83	-0.11	0.54	0.67
	2	35			0.026	0.077	0.41			12.31			-0.12		
	3	28			0.035	0.082	0.67			7.74			0.73		
	4	34			0.030	0.078	0.43			11.68			0.52		
	5	37			0.034	0.079	0.49			10.19			1.01		
	6	30			0.030	0.087	0.50			11.38			-0.02		
	7	39			0.026	0.069	0.57			7.82			0.28		
	8	33			0.037	0.093	0.64			9.30			0.56		
	9	33			0.057	0.111	0.69			9.82			2.00		
	10														
F1	1	40	39.9	3.4	0.050	0.086	0.36	0.52	0.12	14.17	10.06	2.21	4.17	2.33	1.18
	2	43			0.057	0.115	0.64			11.05			1.99		
	3	41			0.046	0.077	0.49			9.26			2.93		
	4	37			0.037	0.070	0.42			10.11			2.26		
	5														
	6														
	7	40			0.051	0.115	0.70			10.33			1.19		
	8	44			0.046	0.071	0.52			7.86			3.06		
	9	34			0.025	0.059	0.49			7.63			0.69		
	10														

Appendix G.1: Moss Bioassay Raw Data (Cont'd)

Station	Stem #	Length (mm)	Average Length	Standard Deviation	Absorbance 648.2	Absorbance 664.9	Dry Weight (mg)	Average Dry Weight	Dry Weight S.D.	Chl A	Average Chl A	Chl A S.D.	Chl B	Average Chl. B	Chl B S.D.
H4	1	44	41.3	4.4	0.050	0.088	0.39	0.49	0.06	13.46	10.07	2.18	3.72	2.19	1.46
	2	40			0.038	0.080	0.53			9.36			1.44		
	3	39			0.050	0.088	0.49			10.71			2.96		
	4	38			0.021	0.046	0.52			5.53			0.73		
	5	36			0.051	0.076	0.50			8.66			3.67		
	6	39			0.029	0.071	0.53			8.54			0.62		
	7	47			0.054	0.086	0.43			11.62			4.18		
	8	42			0.044	0.106	0.60			11.23			0.90		
	9	50			0.047	0.080	0.46			10.29			3.12		
	10	38			0.029	0.074	0.42			11.31			0.60		
H2	1	23	22.4	2.1	0.066	0.175	1.37	0.95	0.38	8.25	10.81	3.05	0.29	0.82	0.70
	2	22			0.079	0.187	0.72			16.46			1.47		
	3	22			0.074	0.160	0.81			12.32			1.71		
	4	28			0.030	0.079	0.47			10.84			0.42		
	5	22			0.045	0.130	1.50			5.66			0.00		
	6	22			0.061	0.159	1.02			10.04			0.43		
	7	20			0.042	0.109	0.53			13.23			0.60		
	8	22			0.078	0.222	1.46			9.92			0.06		
	9	22			0.060	0.123	0.89			8.53			1.45		
	10	21			0.070	0.152	0.74			12.82			1.74		
H2-R	1	25	24.4	4.1	0.040	0.099	0.58	0.78	0.54	10.90	11.50	3.44	0.73	1.11	0.97
	2	20			0.087	0.185	0.92			12.50			1.85		
	3	25			0.064	0.141	0.67			13.17			1.68		
	4	23			0.031	0.082	0.52			10.18			0.37		
	5	23			0.070	0.129	0.55			14.14			3.42		
	6	22			0.046	0.121	0.46			16.96			0.66		
	7	22			0.047	0.124	0.61			13.11			0.50		
	8	25			0.037	0.099	0.58			11.03			0.35		
	9	24			0.068	0.151	2.27			4.17			0.51		
	10	35			0.042	0.095	0.68			8.79			1.00		

Appendix G.1: Moss Bioassay Raw Data (Cont'd)

Station	Stem #	Length (mm)	Average Length	Standard Deviation	Absorbance 648.2	Absorbance 664.9	Dry Weight (mg)	Average Dry Weight	Dry Weight S.D.	Chl A	Average Chl A	Chl A S.D.	Chl B	Average Chl. B	Chl B S.D.
C5	1	32	29.5	5.1	0.045	0.099	0.39	0.42	0.10	15.88	12.94	1.79	2.04	1.01	0.79
	2	27			0.042	0.077	0.42			11.04			2.71		
	3	33			0.029	0.070	0.40			11.13			0.89		
	4	22			0.037	0.095	0.51			11.97			0.60		
	5	38			0.025	0.064	0.35			11.74			0.60		
	6	24			0.033	0.087	0.47			11.94			0.46		
	7	30			0.025	0.066	0.36			11.83			0.44		
	8	34			0.029	0.081	0.36			14.64			0.20		
	9	31			0.030	0.073	0.32			14.53			1.10		
	10	24			0.061	0.150	0.65			14.72			1.04		
Control	1	45	33.6	5.5	0.037	0.094	0.55	0.42	0.09	10.96	8.39	3.00	0.60	1.32	0.82
	2	38			0.027	0.065	0.39			10.60			0.86		
	3	32			0.037	0.070	0.39			10.89			2.43		
	4	32			0.021	0.045	0.31			9.04			1.30		
	5	33			0.020	0.046	0.51			5.69			0.59		
	6	28			0.029	0.053	0.29			11.00			2.73		
	7	29			0.017	0.032	0.40			4.85			1.10		
	8	32			0.018	0.034	0.50			4.12			0.93		
	9														
	10														
C4	1	38	33.6	7.4	0.024	0.058	0.34	0.41	0.19	10.85	12.70	3.23	0.86	1.64	1.26
	2	39			0.029	0.069	0.25			17.51			1.52		
	3	38			0.032	0.060	0.31			11.72			2.69		
	4	41			0.028	0.070	0.46			9.73			0.61		
	5	34			0.033	0.082	0.51			10.28			0.67		
	6	40			0.026	0.059	0.26			14.28			1.59		
	7	31			0.036	0.065	0.25			15.60			4.01		
	8	33			0.045	0.090	0.32			17.27			3.21		
	9	22			0.048	0.114	0.64			11.30			0.99		
	10	20			0.039	0.105	0.80			8.49			0.25		

Appendix H: Resin Data

Table H.1: Recoveries of lab scale atrazine apparatus

Sample pH / Atrazine Conc.	Percent Recovery (Rep 1)	Percent Recovery (Rep 2)	Average	Standard Deviation
pH 7 / 100 (dist)	69.2	82.9	76.0	9.7
pH 5 / 100 (dist)	61.9	55.2	58.5	4.8
pH 7 / 100	54.5	56.4	55.4	1.4
pH 5 / 100	38.5	52.5	45.5	9.9
pH 7 / 10	33.9	33.2	33.6	0.5
pH 5 / 10	23.5	47.8	35.6	17.2
pH 7 / 1	4.4	8.8	6.6	3.1
pH 5 / 1	0.0	10.0	5.0	7.1

Table H.2: Field recoveries of atrazine from XAD-7 resin apparatus.

Station (Resin Pouch #)	Recovered Conc./g of resin
H1(1)	0.079
H1(2)	0.071
H1(3)	0.050
Control(1)	0.045
Control(2)	0.041
Control(3)	0.111
F2(1)	0.085
F2(2)	0.057
F2(3)	0.045
K2(1)	0.090
K2(2)	0.061
K2(3)	0.000
D1(1)	0.023
D1(2)	0.000
D1(3)	0.000
D1-Rep(1)	0.025
D1-Rep(2)	0.000
D1-Rep(3)	0.000

Appendix I: Accuracy and Precision

Table I.1: Percentage difference values (precision) for water quality variables. All values in units of mg/L.

Orthophosphate-P				Nitrate-N			Nitrite-N			Ammonia-N	
Sampling Station	Date	Duplicate Conc. (mg/L)	Sampling Station	Date	Duplicate Conc. (mg/L)	Sampling Station	Date	Duplicate Conc. (mg/L)	Sampling Station	Date	Duplicate Conc. (mg/L)
H2	26-Jun-01	0.082	D1	23-Jul-01	0.024	D1	23-Jul-01	0.004	D1	23-Jul-01	0.004
		0.086			0.024			0.005			0.005
	% diff	4.7		% diff	0.0		% diff	20.0		% diff	20.0
F2	26-Jun-01	0.044	K2	23-Jul-01	1.232	K2	23-Jul-01	0.014	C3	18-Sep-02	0.598
		0.045			1.245			0.015			0.606
	% diff	2.2		% diff	1.0		% diff	6.7		% diff	1.3
I3	23-Jul-01	0.025	J1	23-Jul-01	0.032	J1	23-Jul-01	0.009	Outfall	18-Sep-02	8.647
		0.026			0.032			0.009			8.793
	% diff	3.8		% diff	0.0		% diff	0.0		% diff	1.7
D5	23-Jul-01	0.01	J1	5-Nov-01	1.802	J1	5-Nov-01	0.012	D4	18-Sep-02	0.112
		0.01			1.803			0.013			0.116
	% diff	0.0		% diff	0.1		% diff	7.7		% diff	3.4
I3	22-Aug-01	0.045	CO	5-Nov-01	0.713	CO	5-Nov-01	0.014	C1	18-Sep-02	0.06
		0.048			0.714			0.014			0.062
	% diff	6.3		% diff	0.1		% diff	0.0		% diff	3.2
J1	22-Aug-01	0.012	H1	5-Nov-01	1.15	H1	5-Nov-01	0.041	C1	2-Nov-02	0.031
		0.013			1.152			0.042			0.034
	% diff	7.7		% diff	0.2		% diff	2.4		% diff	8.8
M1	22-Aug-01	0.028	K1	5-Nov-01	1.233	K1	5-Nov-01	0.087	C4	2-Nov-02	0.09
		0.029			1.233			0.088			0.09
	% diff	3.4		% diff	0.0		% diff	1.1		% diff	0.0
C4	5-Nov-01	0.029	C3	13-Dec-01	1.046	C3	13-Dec-01	0.015	K1	2-Nov-02	0.17
		0.031			1.047			0.016			0.172
	% diff	6.5		% diff	0.1		% diff	6.3		% diff	1.2
C1	5-Nov-01	0.008	H2	13-Dec-01	1.068	H2	13-Dec-01	0.022	C4	13-Dec-02	0.097
		0.015			1.068			0.023			0.111
	% diff	46.7		% diff	0.0		% diff	4.3		% diff	12.6
D2	13-Dec-01	0.102	G1	13-Dec-01	0.87	G1	13-Dec-01	0.028	C1	13-Dec-02	0.043
		0.104			0.877			0.029			0.07
	% diff	1.9		% diff	0.8		% diff	3.4		% diff	38.6
A1	13-Dec-01	0.146	C4	29-Jan-01	1.076	C4	29-Jan-01	0.008	C4	23-Jan-02	0.097
		0.152			1.083			0.009			0.111
	% diff	3.9		% diff	0.6		% diff	11.1		% diff	12.6
D2	23-Jan-01	0.031	C1	29-Jan-01	0.614	C1	29-Jan-01	0.012	C1	23-Jan-02	0.043
		0.033			0.616			0.012			0.07
	% diff	6.1		% diff	0.3		% diff	0.0		% diff	38.6
C5	28-Feb-02	0.01	D3	29-Jan-01	0.78	D3	29-Jan-01	0.013	C4	23-Jan-02	0.042
		0.011			0.782			0.013			0.043
	% diff	9.1		% diff	0.3		% diff	0.0		% diff	2.3
C4	4-Apr-02	0.037	J1	1-Mar-02	0.671	J1	1-Mar-02	0.017	C1	23-Jan-02	0.085
		0.037			0.676			0.018			0.09
	% diff	0.0		% diff	0.7		% diff	5.6		% diff	5.6
H4	4-Apr-02	0.071	CO	1-Mar-02	0.46	CO	1-Mar-02	0.005	C4	4-Apr-02	0.042
		0.071			0.46			0.005			0.044
	% diff	0.0		% diff	0.0		% diff	0.0		% diff	4.5

Table I.1 (Cont'd): Percentage difference values (precision) for water quality variables. All values in units of mg/L.

Orthophosphate-P			Nitrate-N			Nitrite-N			Ammonia-N		
Sampling Station	Date	Duplicate Conc. (mg/L)	Sampling Station	Date	Duplicate Conc. (mg/L)	Sampling Station	Date	Duplicate Conc. (mg/L)	Sampling Station	Date	Duplicate Conc. (mg/L)
C1	3-May-02	0.015	C5	4-Apr-02	0.332	C5	4-Apr-02	0.008	H4	4-Apr-02	0.189
		0.017			0.334			0.008			0.191
	% diff	11.8		% diff	0.6		% diff	0.0		% diff	1.0
G1	3-May-02	0.025	C3	4-Apr-02	0.466	C3	4-Apr-02	0.006	CO	3-May-02	0
		0.025			0.468			0.007			0.009
	% diff	0.0		% diff	0.4		% diff	14.3		% diff	100.0
C1	7-Jun-02	0.021	C5	3-May-02	0.226	C5	3-May-02	0.011	F2	3-May-02	0.004
		0.021			0.226			0.011			0.006
	% diff	0.0		% diff	0.0		% diff	0.0		% diff	33.3
C3	7-Jun-02	0.04	D4	3-May-02	0.618	D4	3-May-02	0.014	D1	3-May-02	0.007
		0.041			0.619			0.014			0.011
	% diff	2.4		% diff	0.2		% diff	0.0		% diff	36.4
H2	7-Jun-02	0.305	C5	7-Jun-02	0.031	C5	7-Jun-02	0.008	C3	7-Jun-02	0.06
		0.309			0.031			0.009			0.061
	% diff	1.3		% diff	0.0		% diff	11.1		% diff	1.6
K2	7-Jun-02	0.019	C5	7-Jun-02	0.145	C5	7-Jun-02	0.009	C5	7-Jun-02	0.04
		0.022			0.145			0.009			0.043
	% diff	13.6		% diff	0.0		% diff	0.0		% diff	7.0
A1	7-Jun-02	0.023	D4	7-Jun-02	0.365	D4	7-Jun-02	0.013	D4	7-Jun-02	0.047
		0.025			0.365			0.013			0.062
	% diff	8.0		% diff	0.0		% diff	0.0		% diff	24.2
% Diff. Std. Dev.		*9.8*	*% Diff. Std. Dev.*		*0.3*	*% Diff. Std. Dev.*		*5.6*	*% Diff. Std. Dev.*		*23.0*
Average % Difference		*6.3*	*Average % Difference*		*0.2*	*Average % Difference*		*4.3*	*Average % Difference*		*16.3*

Table I.2: Dry Season Method Precision for Metals in Sediment

Sampling Station	Copper (mg/kg)	Lead (mg/kg)	Zinc (mg/kg)	Manganese (mg/kg)	Iron (mg/kg)
H1	30.0	28.9	110.2	320.3	39580.1
	31.6	30.1	114.7	340.6	42873.0
% diff	5.1	4.0	3.9	6.0	7.7
C3	27.0	27.6	103.3	299.7	39837.2
	27.2	29.8	105.5	313.0	44479.7
% diff	0.7	7.4	2.1	4.2	10.4
Avg % diff. std. dev.	***3.1***	***2.4***	***1.3***	***1.2***	***1.9***
Avg % diff.	***2.9***	***5.7***	***3.0***	***5.1***	***9.1***

Table I.3: Wet Season Method Precision for Metals in Sediment

Sampling Station	Copper (mg/kg)	Lead (mg/kg)	Zinc (mg/kg)	Manganese (mg/kg)	Iron (mg/kg)
A1	33.4	30.3	150.9	451.5	28512.7
	33.4	31.3	152.1	453.7	31211.5
% diff	0.0	3.2	0.8	0.5	8.6
D3	34.8	31.1	127.1	371.6	70193.3
	39.7	33.3	132.1	412.2	71995.1
% diff	12.3	6.6	3.8	9.8	2.5
Avg % diff. std. dev.	***8.7***	***2.4***	***2.1***	***6.6***	***4.3***
Avg % diff.	***6.2***	***4.9***	***2.3***	***5.2***	***5.6***

Table I.4: Measurement of Method Accuracy Using MESS-2 Marine Reference Sediment

	Copper (mg/kg)	Lead (mg/kg)	Zinc (mg/kg)	Manganese (mg/kg)	Iron (mg/kg)
Replicate Concentrations	32.6	8.1	168.3	256.5	44337.0
	31.8	7.9	163.4	266.2	41940.9
	42.1	6.9	165.4	268.4	37828.7
Average Conc.	35.5	7.6	165.7	263.7	41368.8
Certified Conc.	39.3	21.9	172.0	365.0	62200.0
Lowest Cert. Conc.	37.3	20.8	163.4	345.8	59090.0
% Error outside range	***4.8***	***63.3***	***0.0***	***23.7***	***30.0***

Appendix J: Site Variability

Table J.1: Water quality parameter replicates and coefficient of variance values. All data in units of mg/L

	Orthophosphate-P		Nitrate-N		Nitrite-N		Ammonia		pH		Chlorophyll	
	Rep. 1	Rep. 2	Rep. 1	Rep. 2	Rep. 1	Rep. 2	Rep. 1	Rep. 2	Rep. 1	Rep. 2	Rep. 1	Rep. 2
23-Jul-01	0.010	-	1.428	-	0.024	-	0.006	-	6.8	6.2	9.1	70.1
	0.009	-	1.511	-	0.024	-	0.007	-	6.8	6.2	4.2	22.8
	0.006	-	1.532	-	0.023	-	0.006	-		6.2	21.3	64.9
Average	0.008	-	1.490	-	0.024	-	0.006	-	6.8	6.2	11.5	52.6
Std. Dev.	0.002	-	0.055	-	0.001	-	0.001	-	0.0	0.0	8.8	25.9
Coef. of Var. (%)	**25.0**	**-**	**3.7**	**-**	**2.4**	**-**	**9.1**	**-**	**0.0**	**0.3**	**76.2**	**49.3**
22-Aug-01	0.082	0.047	-	-	0.014	0.006	0.070	0.046	6.2	6.2	4.8	3.1
	0.057	0.046	-	-	0.014	0.005	0.063	0.047	6.1	6.1	3.9	1.2
	0.063	0.070	-	-	0.014	0.009	0.082	0.057	6.1	6.1	4.4	2.9
Average	0.067	0.054	-	-	0.014	0.007	0.072	0.050	6.1	6.1	4.4	2.4
Std. Dev.	0.013	0.014	-	-	0.000	0.002	0.010	0.006	0.1	0.1	0.5	1.0
Coef. of Var. (%)	**19.4**	**25.0**	**-**	**-**	**0.0**	**31.2**	**13.4**	**12.2**	**0.9**	**0.9**	**10.6**	**42.7**
18-Sep-01	0.009	0.287	0.043	0.128	0.006	0.021	0.057	0.947	6.4	6.2	3.4	85.7
	0.011	0.292	0.052	0.155	0.006	0.028	0.062	0.898	6.5	6.7	5.4	89.0
	0.011	0.257	0.046	0.080	0.007	0.018	0.064	0.825	6.5	6.7	4.4	102.8
Average	0.010	0.279	0.047	0.121	0.006	0.022	0.061	0.890	6.4	6.5	4.4	92.5
Std. Dev.	0.001	0.019	0.005	0.038	0.001	0.005	0.004	0.061	0.1	0.3	1.0	9.1
Coef. of Var. (%)	**11.2**	**6.8**	**10.0**	**31.5**	**9.1**	**23.0**	**5.9**	**6.9**	**0.8**	**4.3**	**22.9**	**9.8**
1-Nov-01	0.033	0.061	1.589	0.614	0.090	0.005	0.193	0.000	6.3	6.1	0.2	3.2
	0.039	0.061	1.751	0.635	0.089	0.006	0.168	0.000	6.3	5.9	0.4	7.1
	0.034	0.057	1.762	0.672	0.088	0.010	0.172	0.000	6.3	5.9	0.6	2.0
Average	0.035	0.060	1.701	0.641	0.089	0.007	0.178	0.000	6.3	6.0	0.4	4.1
Std. Dev.	0.003	0.002	0.097	0.030	0.001	0.003	0.013	0.000	0.0	0.1	0.2	2.6
Coef. of Var. (%)	**9.1**	**3.9**	**5.7**	**4.6**	**1.1**	**37.8**	**7.6**	**0.0**	**0.2**	**1.6**	**50.2**	**64.3**
13-Dec-01	0.024	0.146	0.270	0.815	0.015	0.026	0.061	0.505	5.7	5.8	-	1.7
	0.022	0.206	0.253	0.821	0.011	0.028	0.066	0.476	5.7	5.9	-	1.9
	0.022	0.170	0.274	0.801	0.013	0.022	0.061	0.450	5.7	5.9	-	-
Average	0.023	0.174	0.265	0.812	0.013	0.025	0.063	0.477	5.7	5.8	-	1.8
Std. Dev.	0.001	0.030	0.011	0.010	0.002	0.003	0.003	0.028	0.0	0.0	-	0.2
Coef. of Var. (%)	**5.1**	**17.4**	**4.2**	**1.3**	**15.4**	**12.1**	**4.6**	**5.8**	**0.2**	**0.2**	**-**	**10.3**

Table J.1: Water quality parameter replicates and coefficient of variance values. All data in units of mg/L.

	Orthophosphate-P		Nitrate-N		Nitrite-N		Ammonia		pH		Chlorophyll	
	Rep. 1	Rep. 2	Rep. 1	Rep. 2	Rep. 1	Rep. 2	Rep. 1	Rep. 2	Rep. 1	Rep. 2	Rep. 1	Rep. 2
28-Feb-02	0.009	0.017	0.123	0.317	0.009	0.013	0.010	0.055	6.1	6.3	0.6	0.7
	0.025	0.016	0.111	0.313	0.005	0.012	0.017	0.044	6.1	6.3	0.8	0.8
	0.011	0.016	0.114	0.321	0.005	0.016	0.007	-	6.1	6.3	0.9	0.9
Average	0.015	0.016	0.116	0.317	0.006	0.014	0.011	0.050	6.1	6.3	0.8	0.8
Std. Dev.	0.009	0.001	0.006	0.004	0.002	0.002	0.005	0.008	0.0	0.0	0.2	0.1
Coef. of Var. (%)	**58.1**	**3.5**	**5.5**	**1.4**	**36.5**	**15.2**	**45.3**	**15.7**	**0.3**	**0.2**	**20.0**	**11.0**
4-Apr-02	0.018	0.023	0.322	0.129	0.006	0.005	0.018	0.088	6.0	6.6	1.5	2.0
	0.020	0.027	0.288	0.137	0.006	0.008	0.024	0.120	6.0	6.6	1.4	1.3
	0.022	0.052	0.300	0.128	0.008	0.005	0.028	0.108	6.1	6.7	1.3	2.0
Average	0.020	0.034	0.303	0.131	0.007	0.006	0.023	0.105	6.0	6.6	1.4	1.8
Std. Dev.	0.002	0.016	0.017	0.005	0.001	0.002	0.005	0.016	0.0	0.0	0.1	0.4
Coef. of Var. (%)	**10.0**	**46.2**	**5.7**	**4.0**	**17.3**	**28.9**	**21.6**	**15.3**	**0.2**	**0.3**	**6.7**	**21.1**
3-May-02	0.026	0.015	0.016	0.085	0.008	0.009	0.007	0.007	6.0	6.5	7.1	7.5
	0.022	0.016	0.009	0.080	0.006	0.008	0.002	0.010	6.0	6.6	5.3	7.4
	0.025	0.016	0.019	0.087	0.008	0.008	0.002	0.003	-	-	-	-
Average	0.024	0.016	0.014	0.084	0.007	0.008	0.004	0.007	6.0	6.6	6.2	7.4
Std. Dev.	0.002	0.001	0.005	0.004	0.001	0.001	0.003	0.004	0.0	0.0	1.3	0.1
Coef. of Var. (%)	**8.6**	**3.7**	**36.1**	**4.7**	**15.7**	**6.9**	**78.7**	**52.7**	**0.4**	**0.3**	**20.7**	**0.9**
7-Jun-02	0.026	0.030	0.003	0.155	0.006	0.012	0.000	0.131	6.4	6.8	1.5	2.2
	0.023	0.033	0.009	0.137	0.005	0.012	0.000	0.138	6.0	6.8	0.8	2.1
	0.025	0.032	0.028	0.138	0.005	0.012	0.051	0.131	6.1	6.8	0.7	2.1
Average	0.025	0.032	0.013	0.143	0.005	0.012	0.017	0.133	6.2	6.8	1.0	2.1
Std. Dev.	0.002	0.002	0.013	0.010	0.001	0.000	0.029	0.004	0.2	0.0	0.4	0.0
Coef. of Var. (%)	**6.2**	**4.8**	**97.7**	**7.0**	**10.8**	**0.0**	**173.2**	**3.0**	**2.6**	**0.4**	**41.5**	**2.2**

Average CV(%)	***15.5***	***14.9***	***15.5***	***27.7***	***0.8***	***27.1***
Maximum CV(%)	***58.1***	***97.7***	***36.5***	***173.2***	***4.3***	***76.2***

Table J.2: Coefficient of variance values for metals and LOI in sediments.

	Copper	Lead	Zinc	Manganese	Iron	% LOI
Dry Season, June 2001 (Station F1)	21.4	49.6	126.4	786.2	89944.7	11.3
	22.0	43.4	111.6	665.6	92740.7	11.3
	20.9	45.8	121.1	702.2	-	11.4
Average	21.45	46.30	119.70	717.98	91342.70	11.3
Std. Dev.	0.52	3.12	7.53	61.82	1977.05	0.08
Coef. of Var. (%)	**2.4**	**6.7**	**6.3**	**8.6**	**2.2**	**0.7**
Wet Season, Feb. 2002 (Station D5)	69.1	52.3	90.3	212.7	36001.9	1.7
	63.9	43.8	93.6	187.4	31772.3	1.5
	69.7	45.6	98.4	185.3	30840.5	1.8
Average	67.57	47.22	94.08	195.14	32871.54	1.7
Std. Dev.	3.20	4.52	4.09	15.27	2750.70	0.18
Coef. of Var. (%)	**4.7**	**9.6**	**4.3**	**7.8**	**8.4**	**11.1**

Table J.3: Coefficient of variance values for grain size distributions.

	> 1 mm	> 0.425 mm	> 0.250 mm	> 0.125 mm	> 0.063 mm	< 0.063 mm
Dry Season, June 2001 (Sring Station)	0.03	1.26	31.35	66.70	0.64	0.01
	0.03	1.05	18.08	76.80	3.95	0.09
	0.03	0.45	22.40	76.18	0.91	0.02
Average	0.03	0.92	23.95	73.23	1.83	0.04
Std. Dev.	0.00	0.42	6.77	5.66	1.84	0.04
Coef. of Var. (%)	**15.8**	**46.0**	**28.3**	**7.7**	**100.3**	**101.4**
Wet Season, Feb. 2002 (Station F1)	0.18	5.00	28.89	52.56	9.44	3.93
	0.19	5.76	31.19	51.11	8.65	3.09
	0.18	5.44	30.18	49.77	10.35	4.08
Average	0.18	5.40	30.09	51.15	9.48	3.70
Std. Dev.	0.01	0.38	1.15	1.39	0.85	0.53
Coef. of Var. (%)	**3.7**	**7.0**	**3.8**	**2.7**	**9.0**	**14.4**

Appendix K: Seasonal Trends

Figure K.1: Seasonal Trends of Dissolved Oxygen in McCallum Slough.

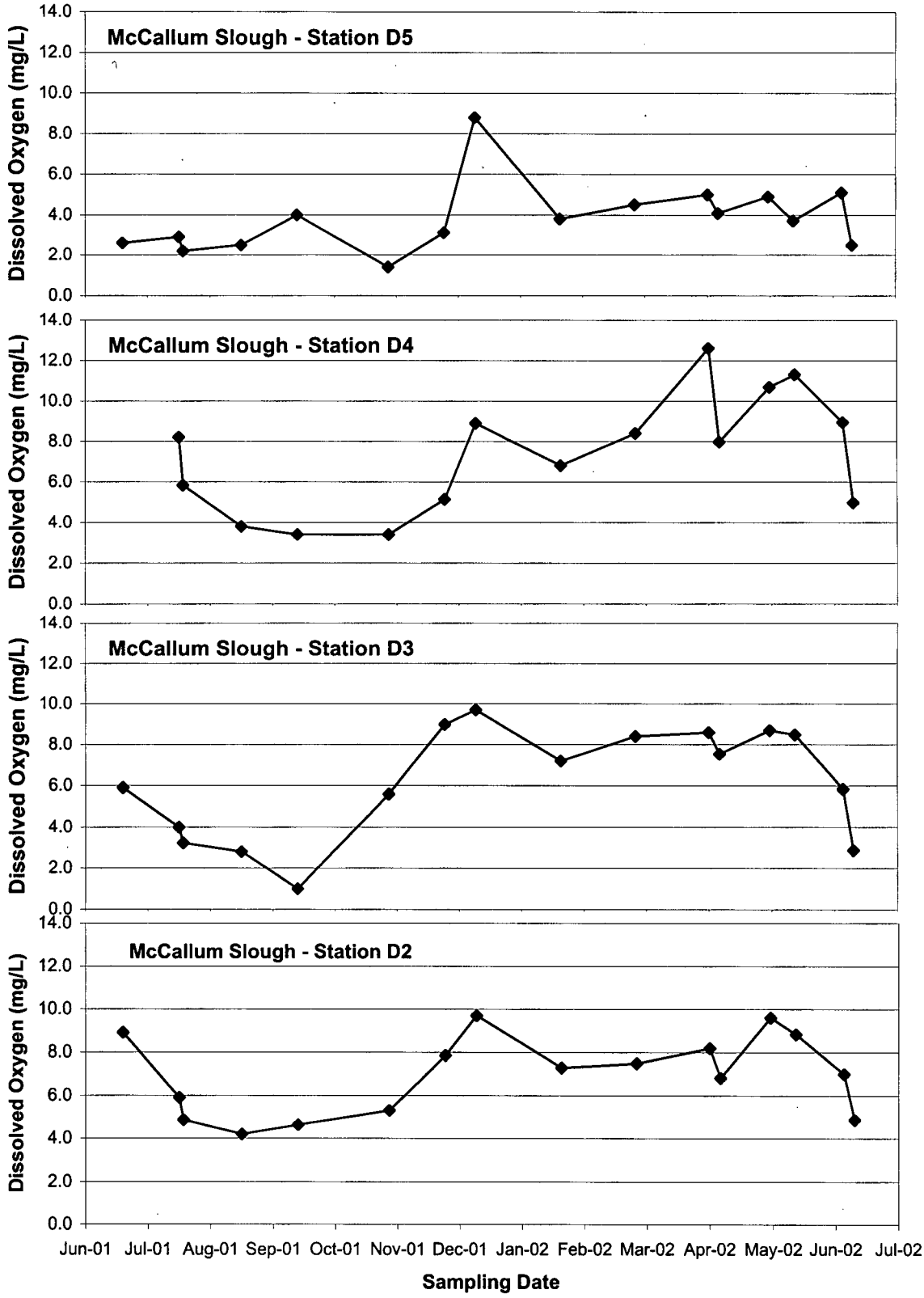

Figure K.1 (Cont'd): Seasonal Trends of Dissolved Oxygen in McCallum Slough.

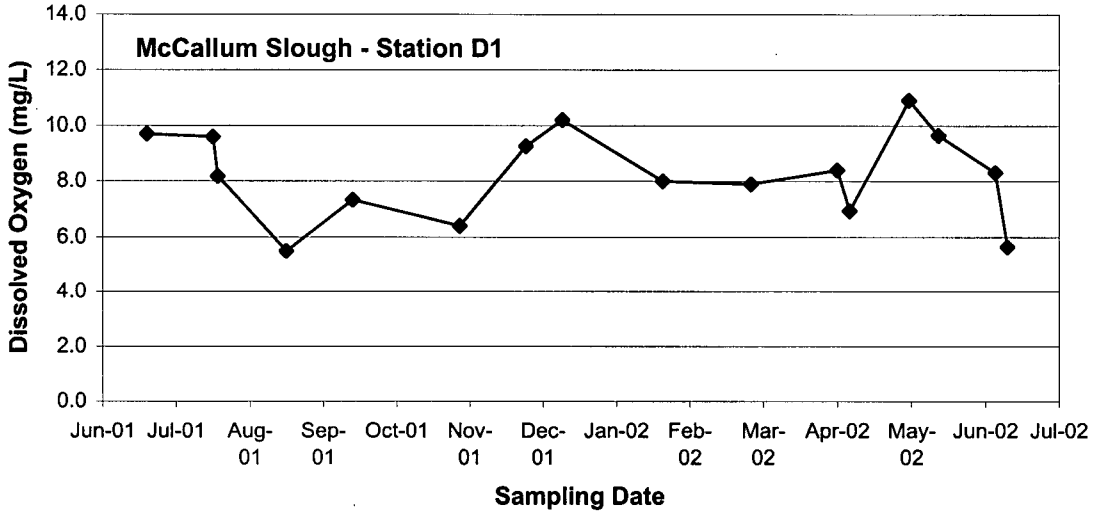

Figure K.2: Seasonal Trends of Turbidity in McCallum Slough.

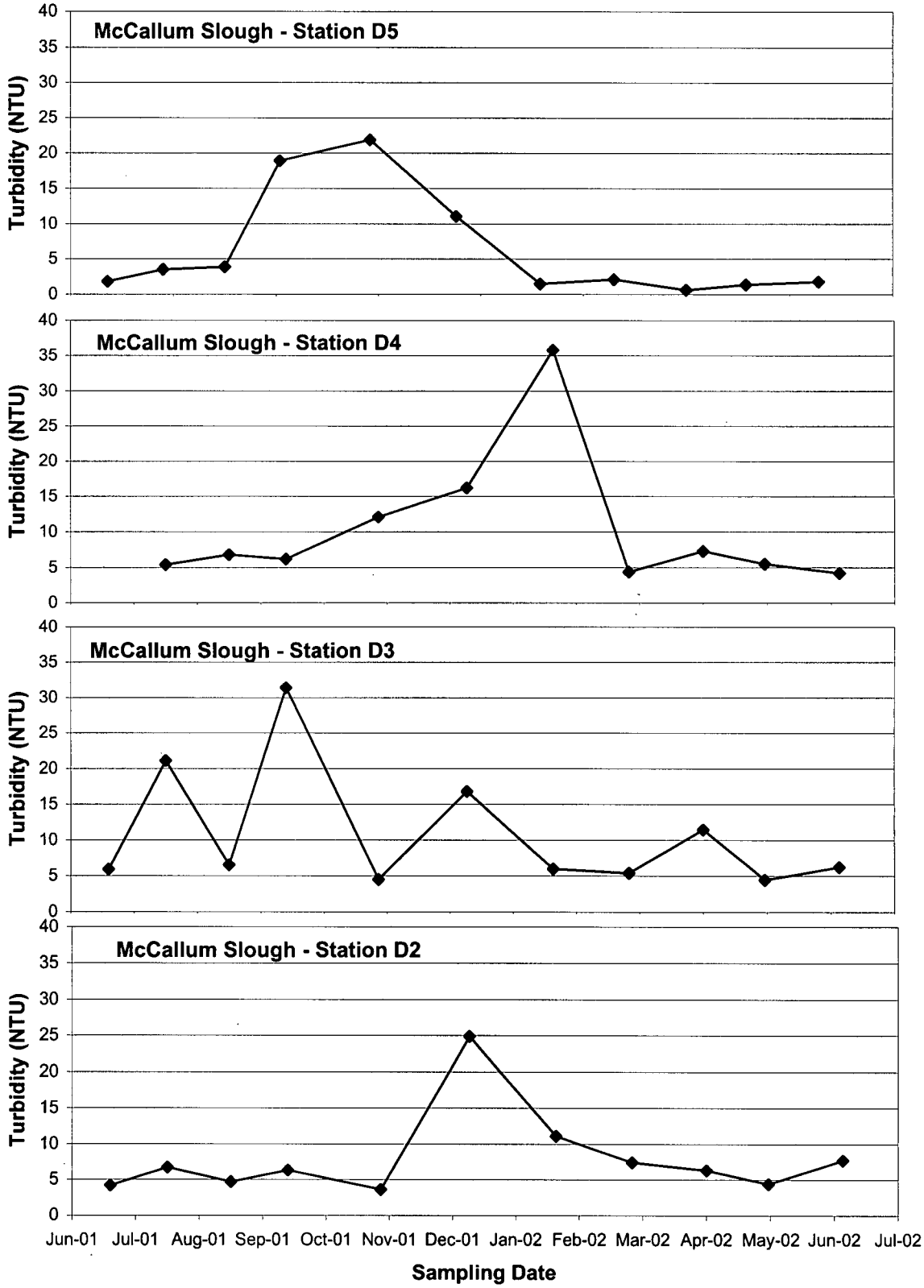

Figure K.2 (Cont'd): Seasonal Trends of Turbidity in McCallum Slough.