1-1-2003

Performance evaluation of a sediment control pond

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PERFORMANCE EVALUATION OF A SEDIMENT CONTROL POND

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Bachelor of Environmental Studies, University of Waterloo, 2001

A Thesis presented to Ryerson University

In partial fulfillment of the requirement for the degree of

Master of Applied Science in the Program of

Environmental Applied Science and Management

TORONTO, ONTARIO, CANADA, 2003

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ABSTRACT

Performance Evaluation of a Sediment Control Pond

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Current erosion and sediment control practices include the use of sediment control ponds that are designed using the 1994 Ministry of the Environment (MOE) Stormwater Management Practices and Design Manual. These design criteria aim at reducing pollutant loads from developed areas. However, the effectiveness of these design criteria when used for areas undergoing construction has yet to be determined in the field. Thus, this thesis is a performance evaluation of a sediment control pond that was designed using the 1994 MOE stormwater design criteria. The objectives of this thesis include the characterization of the runoff and sediments entering, depositing, and leaving the sediment control pond during the construction phase, and the evaluation of the sediment removal efficiencies of the pond. Generally, the pond was successful in reducing many of the pollutants transported to the pond from the catchment area. Suspended solids were the primary pollutants monitored. Heavy metals and general water quality parameters such as chemical oxygen demand, pH, and alkalinity were also monitored. Suspended solids concentrations were high exiting the pond during several events. The particle size distribution predominantly consisted of fine particles. Most heavy metals including beryllium, cadmium, lead, and nickel were reduced in concentration to levels under their Provincial Water Quality Objectives (PWQO). However, some heavy metals had concentrations above their PWQO when exiting the pond.

ACKNOWLEDGEMENTS

I would like to acknowledge the following parties for their contributions to this thesis.

- Toronto and Region Conservation Authority
- Government of Canada's Great Lakes Sustainability Fund
- Natural Science and Engineering Research Council
- Ryerson University
- Ontario Ministry of the Environment
- Department of Fisheries and Oceans Canada
- Town of Richmond Hill
- National Water Research Institute

I would also like to thank those individuals who participated in the planning and implementation of this thesis. These include my thesis supervisor Dr. James Li, and my research supervisors Dr. Doug Banting, Dr, Jiri Marsalek, Dr. Grace Luk and Dr. Pradeep Behera. I would also like to extend my gratitude to the field work assistants including Harry Manson, Derek Smith, Renata Krasnova, Angelune Des Lauriers, and Kevin Sze.

Dedication

To my family for their love and support.

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CHAPTER 1

1.0 INTRODUCTION

Urban stormwater runoff has generated environmental concern over recent years due to the negative impacts associated with its introduction into natural waterways. It is difficult to characterize stormwater runoff as its sources vary, thus making it difficult to treat (McCuen 1980). The most common contaminants found in urban stormwater runoff include heavy metals, petroleum hydrocarbons, pesticides, sediments, and nutrients (Karouna-Renier & Sparling 2001). Disturbed areas, or areas under development, magnify the problems associated with stormwater runoff considerably. In fact, both the Ministry of the Environment (MOE), and the Metropolitan Toronto and Region Conservation Authority (TRCA) (2002) have stated that soil erosion at construction sites is a major cause of the degradation of rivers and streams in urban areas. According to these government agencies, tons of sediments from construction activities are entering Ontario watercourses each year. This in combination with ineffective erosion control measures is responsible for the degradation of water quality and fish habitats, reduction in navigation in waterways, and increases in flooding (MOE & MTRCA 1992). Rapid development within many jurisdictions is currently underway and will continue to be undertaken. Thus, excessive sediment loads will continue to threaten significant portions of watersheds. According to the Ministry of Municipal Affairs and Housing, Ontario will continue to grow, with an increase in 2.5 million new residents by the year 2015 (MAH 2002). Thus, excessive sediment loads generated from construction sites will continue to threaten significant portions of watersheds in Ontario.

The negative impacts on aquatic ecosystems associated with significant increases in sediment yield due to soil erosion has been well documented (Kerr 1995; Lee & Jones 1984; Nighman & Harbor 1995; Newcombe & MacDonald 1991; Fennessey & Jarret 1994; Bhaduri *et al* 1995; Greb & Bannerman 1997; Wu 1989; MOE 2000; Greenland Inc. & TRCA 2000; Clarifica 2001; Whipple 1979; Waneilista & Yousef 1993; Zarull *et al* 1999). In urban stormwater runoff, suspended solids are considered the dominant pollutant. Evidently, suspended solids have continuously been found to have the greatest concentration in stormwater runoff (Randall *et al* 1982). In addition, suspended solids are associated with myriad pollutants such as nutrients, heavy metals and petroleum-based organics that sorb to their surfaces (Randall *et al* 1982). As an isolated pollutant, suspended solids can inhibit photosynthesis, clog gills, smother eggs, and displace benthic invertebrates. In combination with other pollutants sorbed to their surfaces, suspended solids can inflict serious toxic effects to aquatic ecosystems. The severity of these impacts can alter depending on the concentration of suspended solids and the duration of exposure. Figure 1 helps conceptualize the time scale of water quality problems associated with water pollutants and their detrimental effects on aquatic life.

Water quality concerns related to stormwater discharges can be classified as short-term acute effects and long-term cumulative effects (Behera *et al* 2000). This is illustrated in Figure 1. According to Figure 1, floatables, bacteria, the rapid depletion of dissolved oxygen, and suspended solids can inflict short-term acute effects within hours or days. These effects result from individual events where the pollutants are present in high concentrations. In addition, suspended solids, nutrients and dissolved solids can inflict long term cumulative effects within months or years. These effects result from the gradual build-up of pollutant mass and concentration over a longer period of time. This gradual build-up results in detrimental effects as the pollutant concentrations eventually exceed certain threshold levels (Behera *et al* 2000). According to Figure 1, the time scale demonstrates that suspended solids, if in significantly high concentrations, can hold possible short term acute effects that are toxic within weeks of exposure. In addition, suspended solids can inflict long term cumulative effects within months or a season. This presents a serious dilemma for waterways located downstream from areas undergoing construction and development that are not equipped with effective erosion and sediment control practices.

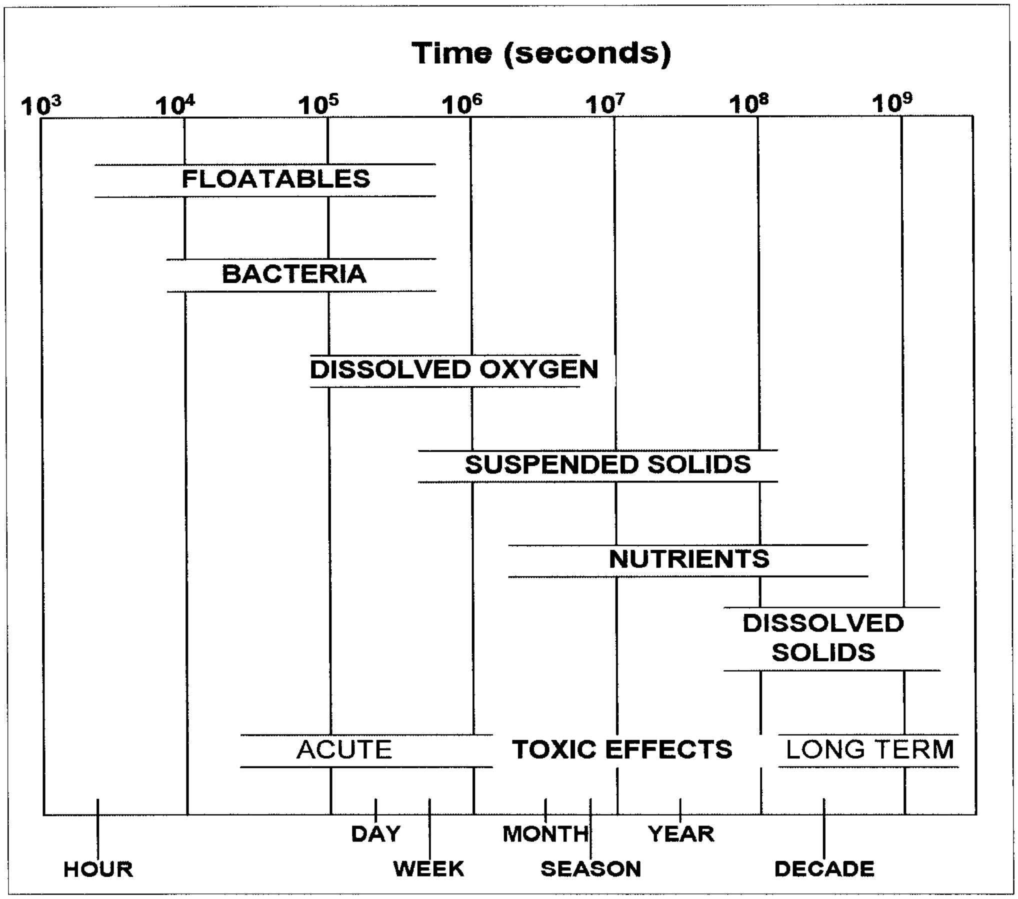


Figure 1: Time Scale of Water Quality Problems (US EPA, 1983)

The volume of sediments accumulating in waterways has increased at alarming rates (Fennessey & Jarrett 1994). Evidently, many reservoirs and basins are experiencing a shorter than expected life due to unforeseen storage losses from sediment loads (Fennessey & Jarrett 1994). Temporary sedimentation basins are commonly used as a means to reduce impacts associated with construction site runoff. Sediment control ponds are constructed according to specific design criteria that aim to trap sediments within the pond thereby reducing impacts downstream. In 1989, the Ministry of Natural Resources produced design criteria for sediment control ponds based on a $125m^3/ha$ in a dry pond setting (Clarifica 2001). However, these design criteria have been targeted as ineffective control measures for erosion at construction sites (Clarifica 2001). In 1994, the Ministry of the Environment (MOE) produced the Stormwater Management Practices Planning and Design Manual (SWMPP) that includes design criteria for stormwater management ponds for developed areas. Since its introduction, these new criteria have also been used in the Toronto area as temporary sediment control ponds before being used as stormwater quality ponds when construction is complete. It is believed that the sizing criteria and permanent pool requirement for the stormwater quality pond will provide increased removal efficiencies over the MNR design criteria for sediment control ponds (Greenland Inc. & TRCA 2001). This new practice has been implemented in the Toronto area, however little data has been collected to evaluate their performance in the field.

Over the years, several studies have been conducted that aimed to improve existing erosion and sediment control practices. The TRCA has led several of these studies due to the development activities that occur within its jurisdiction, and the concern associated with projected future developments. Table 1 outlines the TRCA's multi-phase approach of past studies, current projects in the process of completion, and future studies aimed at improving erosion and sediment control practices.

Table 1: TRCA's multi-phase approach to improving erosion and sediment control practices (Clarifica, 2003)

Urban Construction Sediment Control Study	Phase I 1997
Investigation to Develop an Improve d Sizing Approach for Construction Sediment Control Facilities	Phase II April 2001 – March 2002
Development and implementation of ESC Model by-law	Phase III Aug. 2001-April 2002
Monitoring of Construction Sediment Pond	Phase IV March 2002 – Present
Calibration of hydrologic and water quality models using field monitored data	Phases V – VII Aug 2002 – March 2005

This thesis is a performance evaluation of a construction sediment pond that was designed with the MOE design criteria for stormwater management facilities. Thus, it is contributing research sought out in phase IV outlined in Table 1. Data collected from this thesis will be used for the future calibration of hydrologic and water quality models. In addition, the results from this thesis will help gain a better understanding of the processes of erosion at construction sites, and the performance of sediment control ponds designed with the new criteria that are commonly used at these sites. The outline of this thesis includes the project objectives and scope of the study, concluding Chapter 1. Chapter 2 includes a literature review on the issues with erosion at construction sites and its impacts downstream. This chapter also includes a review on the current practices for erosion and sediment control, and similar studies on erosion and sediment control practices. A description of the project site including the state of the downstream habitat is included in Chapter 3. Chapter 4 describes the methodology undertaken including a detailed account of the development and implementation of the monitoring program. Chapter 5 includes an analysis of the results collected from the site and a discussion of these results. Finally, Chapter 6 concludes the thesis with recommendations.

1.1 Objectives

The pond under study was constructed using the MOE design criteria for stormwater management facilities. The implementation of these new criteria in the field has become a common practice within the Toronto area. However, construction sediment ponds designed using the new criteria for stormwater quality ponds have not been evaluated comprehensively using field monitoring programs (Clarifica 2001). In general, the performance of these facilities during construction stage is not yet known. Therefore, the efficiency of these new design criteria for sediment control purposes is not fully understood. Thus, it is important to understand the accuracy of these sediment pond design criteria through its application in the field (Nighman & Harbor 1995). The ultimate goal of this project is to gain an understanding of the erosion process at construction sites, and to determine the effectiveness of this new criteria used for sediment control ponds during the construction phase. Thus, the objectives of this research project include the following:

- To characterize the runoff and sediments entering, depositing, and leaving the sediment control pond during the construction phase

- To evaluate the sediment removal efficiencies of the pond

1.2 Scope

The objectives include the collection of both water quality and quantity data. Based on these objectives, the scope of the study includes the following:

- To characterize the Event Mean Concentrations (EMC) of sediments and the average concentrations of pollutants entering and exiting the pond

- To characterize the particle size distributions and settling velocities of sediments entering and exiting the pond

- To characterize the bottom sediment accumulation

CHAPTER 2

2.0 LITERATURE REVIEW

The following literature review includes data collected from a plethora of sources that examine the impacts of erosion on the environment. Firstly, the chemical and physical properties of stormwater sediments and their impacts on aquatic communities are examined. In addition, an investigation into the biological, physical, and social impacts associated with soil erosion follows. Secondly, the MNR's *Technical Guidelines on Erosion and Sediment Control,* and the MOE's *Stormwater Management Planning and Design Manual,* are reviewed in order to explore the common practices used to curb the impacts of soil erosion at construction sites. Finally, literature was collected on similar studies that investigated the performance of construction sediment ponds. These similar studies are examined and used as a reference on the monitoring of construction sediment ponds.

2.1 Environmental Impacts of Erosion

2.1.1 Chemical and Physical Properties of Stormwater Sediments

As sediment provides a means for chemical contaminants to be transported to and within natural waterways, aquatic communities receiving stormwater runoff are exposed to a variety of pollutants. Initially, the assumption was that the chemicals bound to sediment particles were unavailable to biota, and therefore posed little threat to aquatic ecosystems (Zarull *et al* 1999). However, laboratory and field experiments began to reveal the risk ecosystems are exposed to with contaminated sediment (Zarull *et al* 1999). Thus, sediment-laden runoff is a major contributor to the increase in chemical transport and deposition within watersheds. Pollutants that are typically associated with sediment, such as heavy metals and nutrients, can vary in their impacts on aquatic ecosystems (Bhaduri *et al* 1995). For example, some contaminants can enter the food chain at the microbe level and in turn bioaccumulate up the food chain to the point where higher life forms such as fish, birds and amphibians may become contaminated (MOE & MTRCA 1992). Furthermore, biota exposed to these contaminants can experience increased mortality, reduced growth and fecundity, or morphological anomalies (Zarull *et al* 1999). Other detrimental impacts include the eutrophication of downstream lakes due to the increased release of contaminants such as phosphates. The chemical constituents sorbed to particles are not the only threat imposed on aquatic ecosystems receiving stormwater runoff. The physical characteristics of the sediment itself are also a function of the severity of impact to the receiving waterbody. For example, Newcombe and Jenson (1996) have claimed that deleterious impacts will increase as a function of increasing particle size. The consequences resulting from these chemical and physical attributes of stormwater pollutants are complex and can be cumulative and synergistic in nature. Thus, the following sections only provide a brief overview of the main issues associated with major stormwater pollutants.

Metals

Stormwater runoff within urban areas, particularly roadways, contain significant quantities of metal elements and solids (Sansalone 1999). These constituents are considered persistent, as they do not degrade in the environment. Depending on the pH of the water, and the nature and quantity of solids, metal elements are partitioned as dissolved or bound to particulates (Sansalone & Buchberger 1997). A major source of trace metals, such as zinc, cadmium, copper, lead, chromium, manganese, aluminum, nickel, and iron is vehicular traffic (Sansalone *et al* 1996).

These metals abundant in urban stormwater are a result of brake and engine wear, fluid leakage, and vehicular component wear and detachment.

It is important to consider the range of impacts metal elements can have on aquatic ecosystems. For example, metal constituents wield both short-term toxicity impacts as a function of concentration or activity as well as long-term toxicity impacts determined through mass accumulation (Sansalone *et al* 1996). A significant concern associated with the release of heavy metals into the aquatic environment is their impacts on organisms. The toxicity of trace metals in aquatic organisms is influenced by various factors. In 1976, Bryan described the various factors including the toxicity of heavy metals in solution (found in Foster & Wittman 1983). These factors are presented in the Table 2.

Table 2: Factors influencing the toxicity of heavy metals in solution

Factors influencing physiology of organisms and possibly form of metal in water	Temperature pH Dissolved oxygen Light Salinity
Condition of Organism	Stage in life history (egg, larva, etc.) Changes in life cycle (e.g. moutling, reproduction) Age and Size Sex Starvation Activity Additional protection (e.g. shell) Adaptation to metals
Behavioral response	Altered behavior

Table 2 demonstrates the variability of circumstances involved with the potential toxicity of trace metals on aquatic organisms. This makes it difficult to characterize the true impacts associated with metal toxicity resulting from urban runoff due to the highly variable loads within stormwater runoff. In 1974, Wood attempted to further classify the toxicity of metals in order to help assess which metals can be toxic and the severity of their toxicity (found in Foster & Wittman 1983). Wood classifies metals according to the following criteria: (1) noncritical, (2) toxic but very insoluble or very rare, and (3) very toxic and relatively accessible. Table 3 groups metals into these classifications. As Table 3 demonstrates, many of the major pollutants found in urban stormwater are found to be very toxic and readily available. These include nickel, zinc, cadmium, and lead.

There are several water quality objectives in place to provide guidance on water quality parameters for surface water in Ontario. These include the Provincial Water Quality Objectives (PWQO), the Canadian Water Quality Guidelines, the Sediment Quality Management Guidelines, and the Ontario Drinking Water Guidelines. The PWQO are the selected governing guidelines for the Richmond Hill sediment pond as they apply to all surface waters within Ontario. They are a set of narrative and numerical ambient surface water quality criteria (MOEE 1994b). The PWQO are continually revised in order to provide the most current and effective criteria for the protection of aquatic life and recreation waters. Because the PWQO are continually revised, pollutants' limits undergo an initial stage before being implemented. During this stage, the PWQO value is referred to as 'interim' before any final approval is given (MOEE 1994b). Table 4 describes both the interim PWQO and the finalized PWQO for the metals tested at the Richmond Hill site.

Table 3: Classification of elements according to toxicity and availability

Non-critical			Toxic but very insoluble or very rare		Very toxic and relatively accessible		
Na	C	F	Ti	Ga	Be	As	Au
K	P	Li	Hf	La	Co	Se	Hg
Mg	Fe	Rb	Zr	Os	Ni	Te	Tl
Ca	S	Sr	W	Rh	Cu	Pd	Pb
H	Cl	Al	Nb	Ir	Zn	Ag	Sb
O	Br	Si	Ta	Ru	Sn	Cd	Bi
N			Re	Ba		Pt	

Table 4: Provincial Water Quality Objectives for metals analyzed at the Richmond Hill site

Metal:	**PWQO**
Aluminum	Interim PWQO At pH 4.5 to 5.5 the *Interim PWQO is 15 µg/L* based on inorganic monomeric aluminum measured in clay-free samples At pH> 5.5 to 6.5, no condition should be permitted which would increase the acid soluble inorganic aluminum concentration in clay-free samples to more than 10% above natural background concentrations for waters representative of that geological area of the Province that are unaffected by man-made inputs At pH>6.5 to 9.0, the *Interim PWQO is 75 µg/L* based on total aluminum measured in clay-free samples If natural background aluminum concentrations in water bodies unaffected by man-made inputs greater than the numerical Interim PWQO (above), no condition is permitted that would increase the aluminum concentration in clay-free samples by more than 10% of the natural background level.
Beryllium	Hardness as CaCO3 PWQO <75 mg/L 11 µg/L >75 mg/L 1100 µg/L
Cadmium	Interim PWQO: Hardness as CaCO3 (mg/L) Interim PWQO (µg/L) 0-100 0.1 >100 0.5
Chlorine	2 µg/L – total residual chlorine, as measured by the amperometric (or equivalent) method.
Chromium	1 µg/L for hexavalent chromium (CrVI) 8.9 µg/L or trivalent chromium (CrIII)
Cobalt	0.9 µg/L
Copper	Hardness as CaCO3 (mg/L) Interim PWQO (µg/L) 0-20 1 >20 5

Table 4 Continued.	
Iron	300 μg/L
Lead	Hardness as CaCO3 (mg/L) Interim PWQO (μg/L) <30 1 30 to 80 3 >80 5
Molybdenum	40 μg/L Interim PWQO
Nickel	25 μg/L
Vanadium	6 μg/L Interim PWQO
Zinc	Interim PWQO 20 μg/L

Source: MOE 1999

Nutrients

Nutrients have been found to increase the productivity of surface-water bodies, including that of streams and estuaries (Novotny & Olem 1994). Although nutrient cycles occur naturally in the aquatic environment, excessive loads that stimulate a surge in productivity can impact the biological balance maintained within a body of water. Excessive nutrient loads are responsible for accelerating the eutrophication of lakes and estuaries. An immediate impact includes the disruption of the dissolved oxygen balance of streams and rivers (Novotny & Olem 1994). The increased algal activity will cause increased respiration by plant and animal life, which can cause a decrease in DO at night (Wanielista & Youseff 1993). According to numerous water quality studies and guidelines, the dissolved oxygen content is considered the most important parameter for protecting fish and aquatic biota (Novotny & Olem 1994).

Nutrients enter the aquatic environment in various forms. The orthophosphorus form of phosphorus and the ammonia and nitrate forms of nitrogen are readily available for plant growth (Wanielista & Youseff 1993). In the absence of treatment facilities such as stormwater ponds, nutrient concentrations in runoff are usually high enough to stimulate the growth of algal and plant species (Wanielista & Youseff 1993). Of the nutrients found in stormwater, the dissolved fraction is readily available for plant and algal populations (Schreiber and Rausch, 1979; Wanielista & Youseff 1993). The particulate fraction will settle, however, it can be resuspended with high flows (Wanielista & Youseff 1993). Of the two major nutrients, phosphorus and nitrogen, the phosphorus loadings are significantly higher in urban areas than from natural areas, whereas nitrogen loads from urban areas are similar to those discharged from natural areas.

In past decades, a tremendous effort has been put forth in controlling nutrient loadings into waterways, specifically phosphorus. This resulted in the control of point sources directly contributing to phosphorus loadings. However, diffuse sources of phosphorus are also a major contributor to phosphorus levels in riverine sediments (Mainstone & Parr 2002). *Mainstone* and *Parr* (2002) state that there are four principal ways in which elevated phosphorus levels can affect aquatic plant communities within rivers:

1. Increase in higher plant growth rates and thereby creating a large standing stock that regrows rapidly following management;
2. Encouragement of higher plants species whose growth rates are geared to higher nutrient levels, thereby altering species composition/balance;
3. Encouragement of epiphytic, epibenthic, filamentous and planktonic algae, thereby reducing the amount of light reaching the leaves and stems of higher plants and interfering with the success of seed germination and seedling growth; and
4. Reduction of rooting depth and thereby making higher plants more susceptible to being ripped out of the substrate under high river flows

Indeed, the impacts of phosphorus loads can be detrimental, and can vary depending on the concentration and duration of the loadings. Eliminating the threat of excessive phosphorus loads from non-point sources is complicated and highly seasonal (Mainstone & Parr 2002). However, since phosphorus characteristically bares a strong affinity for particulates, the majority of the diffuse load is delivered in surface runoff attached to soil particles (Mainstone & Parr 2002). Once in the river, phosphorus becomes highly chemically and biologically active, undergoing numerous transformations and moving between the particulate and dissolved phases, between the sediment and water column, and between the biotic and abiotic environment (Mainstone & Parr 2002). For example, labile phosphorus attached to suspended particulates can rapidly desorb into the water column and become bioavailable. Alternatively, firmly held phosphorus deep within the particle matrix can diffuse slowly into the water column (Mainstone and Parr, 2002). Bhaduri *et al* (1995) found that the bulk of the reactive phosphorous load is associated with larger sediment particles (i.e. sand particles), and that the reactive phosphorus load in the dissolved form is relatively insignificant component of the total reactive phosphorus load. This proved to be a significant finding, as they were able to conclude that by controlling sediment particles that are greater than 2 microns is a good mechanism to control a significant portion of the reactive phosphorus load (Bhaduri *et al* 1995).

For years, anthropogenic activities have significantly increased the supply of nitrogen to freshwaters resulting in the dramatic modification of the global nitrogen cycle (Saunders & Kalff 2001). For example, agricultural land use is recognized as the major non-point source of nitrogen inputs into freshwaters. However, inputs have also been found from the clearing and conversion of land (Saunders & Kalff 2001). The result of increases in nitrogen inputs to freshwater bodies includes the eutrophication of aquatic ecosystems, as well as the acidification of lakes (nitrate) (Saunders & Kalff 2001). Nitrogen in aquatic water bodies may exist in several forms: (a) dissolved nitrogen gas; (b) organic nitrogen incorporated into proteinaceous organic matter; (c) ionized and non-ionized ammonia; (d) nitrite ion; and (e) nitrate ion (Novotny & Olem 1994).

The transformation of nitrogen can affect the dissolved oxygen balance. For example, decomposers break down organic proteinaceous matter releasing ammonia into the aquatic environment. The deionized form of ammonia that is released is toxic to fish. However, the ionized form of ammonia that is released is a nutrient to algae and aquatic plants and exerts dissolved oxygen demand (Novotny & Olem 1994).

The PWQO also includes threshold levels for nutrients due to the problematic consequences of excessive nutrient loads within Ontario's lakes and rivers. Phosphorus and nitrogen in the form of ammonia has been allocated an interim PWQO and PWQO respectively. Table 5 presents these guidelines.

Table 5: Guidelines for Nutrients

Nutrient	Guideline	Source
Total Phosphorus	Interim PWQO 30 µg/L	PWQO
Ammonia	20 µg/L This is dependent on temperature and pH conditions	PWQO

Organics

The production, use, and eventual disposal of organic chemicals have created severe environmental problems (Olem & Novotney 1994). One of the major organic chemicals that are commonly used is pesticides, including herbicides. These chemicals are not exclusive to agricultural use only. Many of these chemicals are also used extensively for urban lawn care (Olem & Novotney 1994). The nature of these chemicals can be toxic and persistant when released into the environment. For example, the herbicides including picloram and 2,4,5-T can often persist in soils for as much as a year after their application. Although generally herbicides are not as persistant as conventional pesticides such as lindane or DDT, these chemicals can accumulate and reach toxic levels (Olem & Novotney 1994).

PAHs can be present in large quantities in urban runoff. These chemicals have a large affinity for adsorption on soils and sediments (Olem & Novoteny 1994). Their source and origin stem back to automobile use, municipal and industrial wastewater effluents, forest fires, and the combustion of coal (Olem & Novotney

1994). Although soil and sediment microorganisms are capable of degrading PAHs, runoff and seepage into groundwater will carry the chemicals before this can occur.

Due to the nature of toxic impacts organic chemicals have had on the environment, PWQOs have been established for many of these chemicals commonly found in urban stormwater. A list of the PAHs, herbicides and pesticides tested in this study, and their subsequent PWQO thresholds are included in Appendix A.

Particle Size

It is recognized that suspended sediments can enter aquatic ecosystems in a number of different shapes and sizes. Newcombe and Jenson (1996) claimed that if other variables are kept constant, ill effects increase as a function on increasing particle size. This coincides with Table 1 presented in Chapter 1. The table demonstrates that suspended solids can impose toxic effects within weeks of exposure, while dissolved solids cause long term effects over lengthy time periods (usually decades). Newcombe and Jenson base their claim on the assumption that during events, fish are often exposed to particle sizes to what they are not normally exposed. For example, Newcombe *et al* (1995) documented that rainbow trout died rapidly (mortality 80 – 100%) when exposed to a silty water discharge whose particles ranged in size from 100 – 170 microns. Newcombe *et* al (1995) compared these results with a similar study where the fish were exposed to particle sizes that were much smaller (mortality 0-10%). Although this study demonstrated that larger particles have a more rapid toxic affect to fish communities, fine particles still impose a threat to the health of aquatic ecosystems. For example, studies have shown that particles in the colloidal size range are capable of entering the fish's cells, and may be accompanied by adsorbed toxicants (Newcombe & Jenson 1996). A process called phagocytosis, which involves the envelopment of fine particles by cells of the fish's gill and gut, can transport the particles into the fish's body. Although these particles may end up in various tissues, the majority ends up in the spleen. As a result, some fishes exposed to fine sediment have spleens that become mineralized to the extent that the tissue damages the cutting edge of the glass microtome blades (an instrument used to cut the organic tissue into thin sections for microscopic examination) (Newcombe & Jenson 1996). Thus, phagocytosis of fine suspended sediments could trigger a sequence of harmful events within the cells of a fish's body leading to ill effects.

Particle size is also considered a key mechanism for pollution transport. For example, fine sediments have been found to represent the majority of particulates in stormwater and carry the bulk of stormwater contaminants (Krishnappan & Marsalek 2002; Greb and Bannerman 1997). Wood and Armitage (1997) claim that it is widely recognized that sediments less than 63 microns in size hold significance in terms of their ability to adsorb and transport contaminants due to their relatively large surface areas and geochemical composition. For example, heavy metals, phosphorus, and petroleum-based organics are believed to have a high affinity for adsorption to these particles (Randall *et al* 1982).

2.1.2 Biological Impacts

One of the main concerns associated with elevated sediment levels in natural waterways is the negative impact on biological communities. The effects of suspended sediments on fish and other aquatic life have been examined extensively in numerous studies (Newcombe & MacDonald 1991). For example, it is recognized that an increase in sediment loads into natural waterways covers streambeds and eliminates habitats of key aquatic species, sometimes suffocating them (Harbor 1999; MOE & MTRCA 1992). In addition, high turbidity levels reduce in-stream photosynthesis, and increase water temperatures while releasing toxic compounds into the ecosystem (Harbor 1999; Whipple 1979; Kerr 1995). Excessive sediment loads can physically alter aquatic ecosystems by dislodging plants, invertebrates, and insects (Whipple 1979). Moreover, some streams are more sensitive to increases in sediment loads than others. For example, cold-water streams, which are continuously threatened, can be dramatically altered with an influx of suspended sediments. The following sections review the negative impacts imposed on biological communities including aquatic vegetation, benthic invertebrates and fish communities.

Aquatic Vegetation

Among the various biological communities are plant life and aquatic vegetation. With an influx in sediments, vegetation suffers from choking as the sediments reduce light penetration, which is necessary for growth (MOE & MTRCA 1992; Kerr 1995; Wood & Armitage 1997). Moreover, high levels of suspended sediment in conjunction with high flow rates can scour algae off streambed substrates and thereby reduce periphyton biomass, a major source of food for several aquatic species (Newcombe & MacDonald 1991). Other documented effects on aquatic vegetation include the physical damage to leaves, slower growth rate, and a reduction in the maximum depth of colonization (Wood & Armitage 1997). These deleterious impacts on aquatic vegetation can be manifested in the invertebrate and fish communities as it disrupts the food chain by reducing or eliminating food sources for other aquatic inhabitants (Wood & Armitage 1997). Although macrophyte communities can benefit aquatic ecosystems by enhancing the settling and deposition of sediments, suspended solid concentrations under extreme conditions can eliminate periphyton and rooted macrophytes from areas where they naturally occur (Wood & Armitage 1997).

Benthic Invertebrates

The deposition of sediments can also have an effect on benthic invertebrates. These organisms are an important part of the food chain, especially for species within higher trophic levels such as fish (US EPA 1997; Maryland DNR 2002). Benthic invertebrates play a critical role in the natural flow of energy and nutrients as they serve as a significant food source for fish, and feed off the algae and other organic matter (Maryland DNR 2002). As a result of senescence, their decay leaves behind nutrients that are reused by aquatic plants and other animals in the food chain (Maryland DNR 2002). In fact, due to their relatively significant position within the aquatic food chain, invertebrates have been used as indicators of water quality as a means to assess the impacts of anthropogenic stress in aquatic systems (US EPA 1997). Unfortunately, influxes of suspended sediment loads from construction sites can create deleterious impacts on benthic communities. For example, excessive sediment loads are responsible for the reduction in food resources and habitat for invertebrates by covering hard substrates and filling interstitial spaces (Nerbonne & Vondracek 2001; Broekhuizen *et al* 2001). Specifically, this can physically alter the riffle/pool ratio, reducing the amount of fish/invertebrate habitat in the stream. In addition, the alteration of the riffle length due to sedimentation ultimately depletes oxygen levels through lessoning the aeration from agitations making survival difficult for many benthic invertebrates (MOE & MTRCA 1992). This is especially a concern with finer sediments, which can scour the substrate by saltation and create a physical disturbance that significantly influences the invertebrate community

(Culp *et al* 1985). This disturbance can result in a significant change in the substrate composition, resulting in changes to the suitability of the substrate for some taxa altering the species composition (Wood & Armitage 1997). Over the years, studies have indicated that an increase in fine sediment load generally causes invertebrates to enter the water column and drift (Shaw & Richardson 2001; Wood & Armitage 1997). As a result, the benthic invertebrate community experiences a reduction in the species density and diversity (Culp *et al* 1985).

Fish

As mentioned previously, benthic invertebrate communities are a source of food for higher trophic levels such as fish (MOE & MTRCA 1992). Thus, negative impacts such as the loss of density and diversity of benthic species, can indirectly impact other aquatic species which are higher in the food chain (Shaw & Richardson 2001). Other impacts to larger fish from excessive deposits of sediment include the filling in of spaces between the gravel and stone of the streambed, thus resulting in a decrease in the spawning success of fish (MOE & MTRCA 1992; Wood & Armitage 1997; Moring 1982; Nerbonne and Vondracek 2001; Broekhuizen *et al* 2001). Furthermore, sedimentation in streams can also physically affect fish gills and eyes, and alter their migration patterns (Wood & Armitage 1997). Higher order species also suffer with the decrease in aquatic invertebrates that live in the substrate, which was covered from sediment deposition.

Impacts on fish can vary depending on the species. Tolerance is dependent on the type of species, various particle sizes and types, and water quality parameters (Kerr 1995). In addition, impacts are based on intensity and the duration of exposure to the organism (Kerr 1995). These are important factors to consider when evaluating a sediment control measure in terms of their potential impacts downstream. The Ministry of Natural Resources has voiced concerns over impacts associated with the continuous development throughout a watershed on biological communities. Their concern stems from the idea that these developments can create greater impacts in terms of the concentration-duration of exposure. It is believed that the increase in duration of exposure results in more harm to fish and other organisms, particularly with higher concentrations (Clarifica 2001). In addition to the risks associated with concentration-duration of exposure, the frequency of pollution episodes, ambient water quality, species and life history stage affected, and the presence of other environmental toxicants may all affect the toxicity of sediments (Newcombe & MacDonald 1991).

Although there are no set PWQO for suspended solids, there are several other guidelines designed to protect aquatic life from increases in suspended sediment levels. Although some of these guidelines are applied outside of Ontario's jurisdiction, they can be used as a reference to maintain water quality objectives. For example, the European Inland Fisheries Advisory Commission developed criteria in 1964 that is designed to protect fisheries from excessive sediment loads (Table 6).

Table 6: Guidelines to Protect Aquatic Resources - European Inland Fisheries Advisory Committee (1964)

Suspended Solids Concentration	**Protection Level**
< 25 mg/L	No evidence of harmful effects on fish and fisheries
25-80 mg/L	Possible to maintain good to moderate fisheries, however the yield would be somewhat diminished relative to waters with < 25 mg/L
80 – 400 mg/L	Unlikely to support good freshwater fisheries
> 400 mg/L	At best, only poor fisheries are likely to be found

Source: Found in Clarifica, pp. 9, 2001

In addition, there are several other guidelines that are useful in determining water quality objectives. Clarifica (2001) outlines in Table 7 recent examples of guidelines to protect aquatic resources.

Table 7: Recent examples of Guidelines to Protect Aquatic Resources

Source	**Criteria**
United States Environmental Protection Agency (1986)	Deposited and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10% from the seasonally established norm for aquatic life
Canadian Council of Resource and Environment Ministers (CCREM) (1987)	Total suspended solids should not be elevated by more than 10 mg/L above background levels when background level is ≤ 100 mg/L. Suspended solids should also not exceed 10% of background concentrations when background concentrations are > 100 mg/L.
British Columbia Ministry of the Environment, Lands, and Parks (1998) and Canadian Council of Ministers of the Environment (CCME, 1999)	Recommendations provided for suspended sediments, turbidity, and streambed substrate. **Suspended Sediments**: *Clear Flow*: Maximum increase of 25 mg/L from background levels for short-term (e.g. < 24 hrs) exposure, and a maximum average increase of 5 mg/L from background from longer-term exposure (e.g. 24 hrs to 30 days) *High Flow*: Maximum increase of 25 mg/L from background levels at any time when background levels are between 25 mg/L and 250 mg/L. Should not increase more than 10% of background levels when background levels are > 250 mg/L

Table 7 Continued.	
	Turbidity *Clear Flow:* Maximum increase of 8 NTUs from background levels for short-term (e.g. 24 hrs) exposures, and a maximum average increase of 2 NTUs from background for longer-terms (e.g. 24 hrs to 30 days). *High Flow or Turbid Waters*: maximum increase of 8 NTUs from background levels at any time when background levels are between 8 NTUs and 80 NTUs. Turbidity should not increase more than 10% of background levels when background levels area >80 NTUs **Stream Substrate:** *Fine Sediments*: The quantity in streambed substrates should not exceed 10% of particles <2mm, 19% of particles < 3mm, and 25% of particles < 6.35 mm *Geometric Mean Diameter* (GMD): The GMD should not exceed 12 mm *Fredle Number* (FN): The Fredle number should not be < 5mm (FN accounts for GMD and sorting coefficient (So = (d75/d25)*0.5). *Inter – Gravel Dissolved Oxygen:* minimum 6.5 mg/L \

Table 7 Continued.			
Department of Fisheries and Oceans – Placer Mining Standards for Sediment Discharges in Yukon Territory (DFO, 1993).	Type	Stream Classification Description	Allowable Sediment Discharge Above Natural Background Levels (mg/L)
	I	Salmonid spawning streams	0
	II	Salmonid rearing streams	<200
	III	Streams with fish having significant use by First Nations, commercial, sport, or domestic fisheries or contributing to biological diversity	<200
	IV	Streams with no fish or streams with fish having no significant use by First Nations, Commercial, sport or domestic fisheries or not contributing to biological diversity	Site specific
	V	Other Streams	0

Source: Clarifica, pp. 11, 2001

Many of the guidelines refer to the natural background levels, making the criteria site specific. In addition, some guidelines are specific to a certain fish species, while others refer to the turbidity of the water as criteria rather than the suspended solid concentrations. It is evident that there are many ways to determine the threshold levels for suspended solids depending on the characteristics of the downstream habitat. For example, cold water fisheries receive a higher order of protection than warm-water fisheries. Another area to consider when implementing water quality criteria for suspended solids is the duration of exposure to aquatic life. For example, Ward (1992) demonstrates in Figure 2 the damage to aquatic organisms by increased suspended sediment concentrations and duration of exposure.

As indicated in Figure 2, if the concentration of suspended solids is less than 25 mg/L, then no ill effects will occur regardless of the duration of exposure during the event (Clarifica 2001). However, Clarifica (2001) examines the graph further and states that for events that extend for 1 to 3 hrs impacts would be considered moderate and would occur at approximately 200mg/L. In addition, Clarifica (2001) also points out that for storms that extend over a 6 to 12 hour period major impacts can occur if the suspended solids concentrations are between 1000 mg/L to 10 000 mg/L.

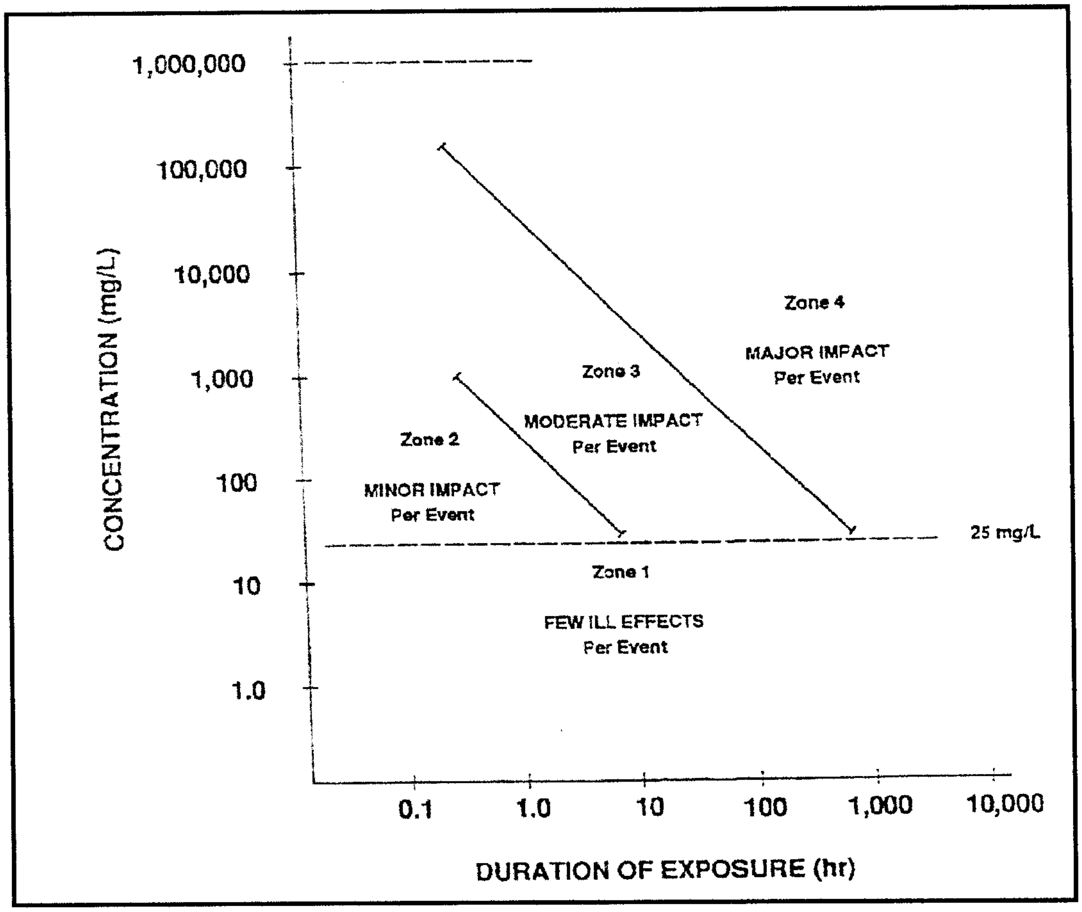


Figure 2: Impacts on aquatic resources according to concentration and duration of exposure (Clarifica, 2001)

2.1.3 Physical and Social Impacts

In addition to the various biological impacts erosion and sedimentation can have on waterways, there are numerous physical impacts worth considering as well. In general, it is understood that a river channel is balanced between the discharge and the amount of suspended sediment transported and bedload stored and moved. Urban construction activity can modify flow quantity, frequency and magnitude of sediment-laden runoff to the stream system causing physical changes to the structure of the channel (MOE & MTRCA 1992). Throughout history, the release of excessive sediment loads causing channels to widen and become shallow has lead to (MOE & MTRCA 1992):

- Increased overbank flooding
- Reduced navigation and
- Lakefront deposition

This physical alteration can impact various streams considerably. In fact, some types of streams and rivers may be able to accommodate the large amount of sediment materials entering its waterway. In some cases, excessive sediment loads carried to a stream from poor construction practices can disrupt or change the natural meander structure of the watercourse (MOE & MTRCA 1992). Negative impacts that result from this include flow restrictions and an increase in the possibility of downstream erosion and flooding. In turn, sedimentation can significantly reduce the navigability of waterways such as channels, canals and harbours. Expensive dredging activities become the only alternative to remedy this situation. Many waterways entering Lake Ontario have already set back governments financially for dredging. For example, the Keating Channel at the mouth of the Don River required over 481,000 m^3 of dredging over a five year period at an overall cost to the taxpayer of over \$5 077 000 (MOE & MTRCA

1992). Unfortunately, the physical impacts resulting in dredging can not only lead to financial stresses, but can also lead to long term ecological impacts. By physically disturbing sediments during dredging practices, the direct destruction of benthic invertebrate habitat can occur as well as the decreased water quality as a result of sediment suspension (Su *et al* 2002). In addition, the resuspension and dispersion of sediments can have ecological impacts that extend beyond the time the physical effects caused by mechanical disturbances occur (Su *et al* 2002). This is of particular concern as these buried sediments have also accumulated pollutants that can be toxic.

Social impacts associated with excessive sediment loads tend to be more evident than physical and biological. Most agree that rivers and streams suffering from high turbidity due to extreme sediment loads are not aesthetically pleasing. In addition, sediment loading into streams also impacts recreation activities. For example, sediment in the stream provides a mechanism for bacteria to be transported to waterfront beaches. Poor erosion and sediment control practices have therefore indirectly caused beach closing (MOE & MTRCA 1992).

2.3 Review of Erosion and Sediment Control Practices

In Ontario, there has been limited research undertaken to address the inadequacies and limitations of erosion and sediment control practices. Although the Toronto and Region Remedial Action Plan recognizes storm runoff as one of the three major pathways for pollutants to enter watercourses, minimal effort has been directed towards the development of remediation measures (Clarifica 2001). In 1999, the Toronto and Region Conservation Authority (TRCA) investigated concerns over the amount of sediment carried through runoff from urban construction sites. The study identified several limiting issues within the erosion and sediment control programs. More specifically, these issues included limiting factors within the adequacy of the planning process, the appropriate selection of sediment control best management practices, and the maintenance and effectiveness of these temporary devices through the servicing and building phases (Clarifica 2001). Following these conclusions, the TRCA, Department of Fisheries and Oceans (DFO), and Great Lakes Sustainability Fund (GLSF) conducted a follow-up study in 2001. The purpose of this study was to review the impacts of construction sediments on fish and fish habitat, develop a generic sediment control by-law, and to investigate the development of a modeling framework to create improved sizing criteria for construction sediment control facilities (Clarifica 2001). The results of this study confirmed that concerns over the effectiveness of sediment control practices on construction sites were valid. It concluded that these control practices provided little protection to fish and fish habitat during wet-weather conditions and can lead to violations of the Ontario Water Resource Act and the Federal Fisheries Act (Clarifica 2001; Greenland Inc. & TRCA 2001).

The use of temporary sediment ponds is a common application for implementing construction sediment control facilities at construction sites. However, according to the previously mentioned studies, these facilities are inadequate in providing protection to downstream fish and fish habitat. The following sections describe the evolution of design criteria in Ontario, more specifically within the TRCA's jurisdiction.

2.3.1 Ministry of Natural Resources Design Criteria

In 1989, the MNR implemented a sizing criterion for sediment control ponds in Ontario based on a storage of 125 m^3/ha and a detention time of 24 hours in a dry pond setting (Clarifica 2001). The TRCA studies identified these design criteria as ineffective in controlling runoff from urban construction sites (Clarifica 2001; Bhaduri *et al* 1995; Greenland Inc. & TRCA 2001). The MNR estimates that during construction, when soils are fully or partially exposed, rates of erosion can increase up to 40,000 times that of undeveloped lands or forests (Clarifica 2001). These erosion rates, coupled with inefficient erosion and sediment control practices, violate regulatory requirements set out by the Federal Fisheries Act that prohibits any harmful alteration, disruption, or destruction of fish habitat.

In the MNR's *Technical Guidelines: Erosion and Sediment Control* (1989), the document outlines various alternatives for dealing with erosion and sedimentation, with an emphasis on problems associated with land development. The guidelines were designed to be applied selectively by the Ministry of Natural Resources to guide development and construction where it is deemed that resources that the Ministry is mandated to protect are threatened (MNR 1989). In addition, the Guidelines were developed to fulfill the need for technical standards and specifications for erosion and sedimentation control measures (MNR 1989).

In addition to providing technical guidelines for sediment control structures such as rip rap outlets, baffles, and vegetated buffers, the Technical Guidelines provided design criteria for sediment basins. The Technical Guidelines report that sediment basins are the most effective means of collecting sediment and suggest constructing the sediment basins through excavation to provide the required storage volume. It also recommends damming low-lying areas or waterways to provide the required volume if a more cost-effective approach is desired (MNR 1989). Other recommendations include the use of sedimentation on large construction sites and on sites adjacent to environmentally sensitive areas. The basins are recommended to be installed prior to grading operations and maintained during the clearing and grubbing phase when the erosion potential of the site is the greatest and then replaced with other measures during the remainder of the construction period (MNR 1989). The guidelines also recognize the practicality of incorporating a sediment basin with a permanent facility which serves some other function such as a stormwater management ponds, groundwater recharge ponds or recreational ponds (MNR 1989). The practice of incorporating sediment basins with permanent facilities was also maintained in the 1994 MOE Stormwater design criteria for stormwater management facilities.

Design specifications within the MNR Technical Guidelines (1989) include the following:

- To design the basin shape, the first step outlined is to establish the length (L) of basin required for a design minimum particle size, with its associated vertical settling velocity (Vs), to reach the bottom of the settling zone
- Below this zone, there should be additional volume for storage of sediment
- The basin settling area (in square meters) should be greater than 570 times the basin outlet capacity (in cms)
- To prevent scouring of the storage zone, the ratio of the basin length to the settling zone depth is to be less than 40 with a minimum settling zone depth of 0.6 meters.
- The storage zone depth should allow for 1 year of estimated sediment yield based on the universal soil loss equation
- Basin length to width ratio should be greater than 2 if less than 10
- A baffle should be used at the entrance to prevent 'short circuiting' and to minimize 'dead zones'

- A minimum basin volume of 125 cubic meters per hectare of contributing drainage area is recommended

The Technical Guidelines based these design criteria on typical stream sediment samples. Based on these samples, it is estimated that a trap efficiency of approximately 90 percent can be achieved if soil particles of 40 microns in diameter or larger, settle (MNR 1989). The MNR (1989) guidelines provide an estimated settling velocity of these particles at approximately 0.0021 m/sec. Basing the design criteria on an estimated settling velocity of a hypothetical particle size has proven to be insufficient. The particle size chosen is indicative of the soils within the TRCA region, found in typical stream sediment. However, erosion at construction sites can generate particles much smaller than this size, making the pond inefficient in trapping these particles. Evidently, these limitations are what triggered the TRCA and other government agencies to begin examining the possibility of improving the design criteria of sediment control basins. In addition, these limitations, especially the minimum storage requirements, were addressed with the introduction and implementation of the new stormwater management design criteria for stormwater management ponds discussed in the next section.

2.3.2 Ministry of the Environment Stormwater Management Planning and Design Manual

In 1994, the Ontario Ministry of the Environment released the Stormwater Management Planning and Design Manual requiring new storage criteria in a wet pond setting for stormwater quality treatment. Several design parameters were altered since the release of the 1989 Technical Guidelines. For example, design criteria are provided for the establishment of a sediment forebay within the stormwater management facility. In addition, a permanent pool is also required for the improvement in the settling of solids by the 1994 Guidelines. The volume required for the permanent pool is considered an improvement from the minimum requirement of 125m3/ha in a dry pond setting from the previous guidelines. Following the release of the MOE Stormwater Management Guidelines, the TRCA recommended that both sediment control ponds during construction phase and stormwater quality control ponds after the construction phase be sized in accordance to Table 3.1 in the guidelines and a permanent pool be established to improve pollutant trapping efficiency (Clarifica 2001).

Although the current use of SWMPP criteria designed for stormwater management facilities as design criteria for construction sediment ponds has improved from the previous guidelines, it still bares some limitations. A major requirement included in the 1994 SWMPP is the water quality storage requirements based on receiving waters. This is included in Table 4.1 in the 1994 guidelines. Protection levels are designated depending on the receiving water bodies. For example, Level 1 protection (implemented for the Richmond Hill Pond) requires 80% TSS removal efficiency through the use of wet ponds (MOE 1994). The table provides storage requirements for the different types of stormwater management facilities according to the level of protection required for downstream habitat. The design criteria were derived using continuous simulation modelling of end-of-pipe stormwater management facilities to determine the variation in pollutant removal with SWMP type and level of imperviousness. This criterion was simulated specifically for urban stormwater runoff, not for construction site runoff. In addition, there are limited field studies to support this data.

In developing the sizing criteria for the MOE Stormwater Guidelines (1994), various sources were used. The MOE includes particle size distributions that are characteristic of urban stormwater entering a pond as a design criterion for sizing the sediment forebay. The sediment forebay is used to trap the bulk of coarse sediment entering the pond. Sizing criteria for this component is derived through the calculation of settling velocities of various particle sizes found in urban stormwater (MOE 1994). Design criteria for sediment forebays, which are included in the Stormwater Guidelines, are based on data collected from the US through their NURP (Nationwide Urban Runoff Program) study (MOE 1994).

Table 8: Particle Size Distribution in Stormwater (MOEE, Table 3.3, pg. 89, 1994)

Size Fraction	**% of Particle Mass**	**Average vs (m/s) ≤**
$\leq 20\ um$	0 – 20	0.00000254
$20\ um \leq x \leq 40\ um$	20 – 30	0.0000130
$40\ um \leq x \leq 60\ um$	30 – 40	0.00002540
$60\ um \leq x \leq 0.13$ mm	40 – 60	0.00012700
0.13 mm $\leq x \leq$ 0.40 mm	60 – 80	0.00059267
0.40 mm $\leq x \leq$ 4.00 mm	80 – 100	0.00550333

The data presented in Table 8 are not indicative of Ontario stormwater, and do not address construction site runoff. Indeed, the particle size distribution of sediment in urban stormwater runoff varies from the sediment-laden runoff produced from construction sites. For example, in the TRCA study, it was identified that sediment-laden runoff from construction sites in TRCA jurisdiction typically consist of soil particles that are smaller than 40 microns (Clarifica 2001). However, the table indicates that the percentage of particle mass less than 40 microns consists of less than 30% of suspended sediments in urban stormwater. Thus, it is important to investigate the use of these values included in this table as sizing criteria for construction sediment ponds through field monitoring.

Despite the common practice of using sediment control basins to function as water quality and quantity control, there is limited reliable information documented on their ability to trap sediments (Nighman & Harbor 1995). Sites under development have received increasing recognition as causing detrimental impacts on fish and fish habitats if erosion and sediment control practices are not properly implemented. In addition, current erosion and sediment control practices have been criticized for providing little protection during wet-weather conditions (Bhaduri *et al* 1995; Clarifica 2001; Nighman & Harbor 1995).

2.4 Previous Studies on Erosion and Sediment Control Practices

To complete the literature review, an examination of similar studies based on construction sediment ponds is included within this section. The research conducted in each of these studies is compared to the research conducted in this thesis. The development of design criteria for construction sediment ponds has evolved slowly over the years. Initially, sediment basins were intended to be designed and built so as to occupy the least possible area (Oscanyan 1975). However, this approach created an inefficient basin shape of an inverted cone, which was found to provide limited sediment trapping efficiency (Oscanyan 1975). Although improvements have been made on design criteria, little information has been collected through field experience. The following papers discussed in this section have made contributions in gathering research through field experience on construction sediment ponds.

2.4.1. Design of Sediment Basins for Construction Sites – *Oscanyan, 1975*

This paper, written in 1975, examines sediment control basins at construction sites. The author claims that at that time, much more research was needed at actual construction sites before a standard design procedure could be derived (Oscanyan 1975). The paper further recognizes that although research and laboratory studies are beneficial, they could produce extremely different results from studies at construction sites (Oscanyan 1975). To illustrate the ineffectiveness of design criteria, the paper uses a case study based in Maryland. The design criteria used in the case study were formulated through the Maryland

State sediment program at a time when little data was available from construction sites. The following describes the criteria used (Oscanyan 1975):

- The minimum sediment storage volume provided must be not less than one-half acre-inch per tributary acre (approximately 125m^3/ha)
- Sediment basins must be cleared out when the remaining storage is one-fifth acre-inch per tributary acre

The criteria were established to provide an average trapping efficiency of 70% (Oscanyan 1975). The first criterion listed above is approximately the same requirement put forth by the MNR in 1989. The article criticizes this criterion claiming that no reference to variations in soil types or basin configuration was used. The author claims that additional field studies must be implemented in order to determine efficient design criteria for removal of sediments. Today, over twenty-five years later, sediment control ponds using the new MOE design criteria are facing similar criticism. Construction sediment ponds have not been subject to many evaluations in terms of their ability to filter out sediments, including different particle sizes characteristic of construction sites. Indeed the primary pollutant leaving a construction site via runoff is sediment. Therefore, sediment control ponds should be examined in the field for the various particle sizes and types of sediment entering and leaving the pond.

2.4.2 Trap Efficiency of a Stormwater Basin with and without Baffles – *Nighman and Harbor, 1995*

This study compared stormwater sediment basin trap efficiency, with and without baffles during the construction phase. It maintains that despite the importance and cost of basins used to trap sediment, there is very little reliable information on their trap efficiency or on the effectiveness of design modifications intended to enhance the trap efficiency (Nighman & Harbor 1995). The purpose of the study was to evaluate the trap efficiency of a sediment basin, and to evaluate the effect of a baffle on the trap efficiency of the basin.

The study monitored and sampled inflow and outlfow throughout individual storm events. Trap efficiency was defined as the difference between inflow and outflow sediment loads, and is therefore a measure of the effectiveness of a basin in trapping sediment (Nighman & Harbor 1995). The study criticized the use of empirical and model-based estimates of trap efficiency for construction site basins. This is of particular interest considering the MOEE 1994 design criteria are based on model estimates to calculate the permanent pool requirements. The study further supports their claim by arguing that past attempts to predict trap efficiency often involve continuous simulation models which simulate the physical processes operating in stormwater management basins however rigorous field studies of actual construction site basins are rarely attempted (Nighman & Harbor 1995). The authors further argue that using laboratory models to test basin design theory is not sufficient measure of their accuracy because of scaling problems and the inability of lab models to fully simulate field conditions such as turbulence, bed scour, dead storage, and flow patterns within the basin.

The study site includes a sedimentation basin with one inlet and one outlet structure servicing a residential construction area that is approximately 12 acres (48000m^2) in size. The study compared the trap efficiency calculated through their field research with trap efficiencies predicted by theoretical methods. The theoretical methods and their predicted trap efficiencies (TE) are presented in Table 9.

Table 9: Trap Efficiency Predicted by Theoretical Methods (Nighman and Harbour, Table 2, 1995)

Prediction Method:	**Type of Facility:**	**TE (%)**
Brune (1953)	Reservoir	82
Heinemann (1981): A New Sediment Trap Efficiency for Small Reservoirs	Small Reservoirs	76
USDA Soil Conservation Service & Summit SWCD	Basin	75
Fifield (1994)	Basin *	5.8
USEPA (Goldman, 1986)	Basin *	0

*More comprehensive techniques that took into account grain size distribution

The study revealed that their empirical calculations derived using data collected in the field varied in comparison to the theoretical methods. For example, Brune, Heinemann, and the USDA techniques had slightly overestimated trap efficiencies. Whereas the remaining two techniques significantly underestimated the trap efficiencies (Nighman & Harbor 1995). Of the five storms that were collected and analyzed through the duration of the study, their trap efficiencies were 78%, 67%, 56%, -75%, and 79%. The study excluded the negative removal efficiency due to the timing of sampling, which evidently was not representative of the entire event. Overall, the study concluded that theoretical methods used to predict trap efficiency are not accurate (Nighman & Harbor 1995). The study also concluded that more data should be collected before any generalities can be formed regarding the accuracy of the theoretical methods.

2.4.3 Chemical Trap Efficiency of a Construction-Site Stormwater Retention Basin – *Bhaduri et al, 1995*

This study focused on the chemical constituents from construction sites rather than the sediment load (Bhaduri *et al* 1995). It discussed how design specifications for sediment control basins may include sediment trap efficiency requirements. However, the study further suggested that there is little known about how well basins control chemical pollutants such as heavy metals and nutrients. The study acknowledges that there is a dramatic increase in sediment loads from construction sites, sometimes ranging from 2 to 10 times and occasionally 100 times greater in runoff from construction sites compared to undisturbed land. It is also recognized in the paper that most research on the trap efficiencies of stormwater basins has focused on sediments. However, the paper points out how the Nationwide Urban Runoff Program (NURP) demonstrated that heavy metals and nutrients are by far the most prevalent priority pollutant constituents of urban runoff. Interestingly, the study found literature that claimed in street dirt, particles smaller than 43 microns, which represent only 5.98% by weight of the total solids, may contain 50% of the heavy metals and 33-50% of the algal nutrients. This suggests that stormwater basins need to trap very fine sediments that enter the basin in suspension in order to be effective chemical pollutant traps (Bhaduri *et al* 1995). This is particularly interesting when considering that the MOE design criteria target particles that are 40 microns in size. This suggests that the MOE design criteria are not equipped to trap smaller particles.

Bhaduri *et al* (1995) also suggested that based on the limited data available stormwater basins do act as traps for chemical pollutants but that their effectiveness is highly variable. The purpose of the study was to determine the chemical trap efficiency of a construction site stormwater management basin, specifically the behavior and distribution of lead, chromium, and cadmium as well as phosphorus over a hydrograph. The paper highlights the fact that these pollutants exist both in particulate and dissolved forms in stormwater runoff and can change form from dissolved to particulate and vice-versa. The study site was receiving stormwater runoff from a residential construction site. In fact, this study was also conducted on the same site as the study "Trap Efficiency of a Stormwater Basin with and without Baffles – *Nighman and Harbor, 1995*" previously outlined in this paper.

The study concluded that stormwater retention basins are useful in preventing downstream pollution by trapping chemical pollutants associated with sediments. However, since pollutants are found in higher concentrations in finer sediments and also exist in dissolved form, it is extremely important to know the effectiveness of these basins in trapping chemical pollutants. The study also concludes that stormwater retention basins designed to remove traditional pollutants such as sediment may not be equally effective in removing priority pollutants (i.e. heavy metals and nutrients). When calculating the chemical trap efficiencies, the study found negative trap efficiencies for chromium, cadmium, and lead. The study also found low removal efficiencies for the fine particulates and dissolved solids, 26.6% and 18.4% respectively. However, one of the limitations of the study is that it is only based on the collection of two storm events. Given the high degree of variability in pollutant loads carried in stormwater runoff, two events may not be representative or adequate to base solid conclusions on. The study does caution that due to the limited amount of samples collected, to view the results as preliminary.

2.4.4. Performance of Current Sediment Control Measures at Maryland Construction Sites – *Schueller and Lugbill, 1990*

A field and laboratory sampling study was undertaken in 1988 to evaluate the performance of current designs of sediment basins and rip-rap outlet traps in reducing downstream levels of suspended sediment and turbidity. Samples were collected at the inflow and outflow structures during nine storm events at six representative sediment control sites located in the Anacostia River basin. In addition, field and laboratory tests were performed to determine the settling characteristics of suspended sediment in construction site runoff.

The study found that runoff generated from construction sites implementing standard erosion control measures contained suspended sediment and turbidity levels that spanned four orders of magnitude, with median values of 680 mg/L and 450 NTU (Nephelometric Turbidity Units) (Schueller & Lugbill 1990). In addition, particle size distributions of the sediment loads entering the facilities were extremely fine-grained, characterized by fine silts, clays and colloidal materials (Schueller & Lugbill 1990). Interestingly, the study reported that despite the significant sediment removal that occurred within basins, sediment levels within outflows remained elevated, with a median TSS concentration of 283 mg/L and a median turbidity of 200 NTUs.

The outflow experienced significant increases in TSS levels specifically when the study sites were in an advanced stage of construction, the storm events exceeded 1 inch of rainfall, and the sediment basins contained standing water. According to the study, the overall performance of the sediment controls was relatively weak. For example, on average, the instantaneous removal efficiency (IRE) was estimated to be 65% for all storm events. However, for storm events that produced measurable outflow runoff, the IRE was only 46%. The paper claims the 46% removal rate should be considered as the representative estimate of the effectiveness of sediment control designs within the State of Maryland.

The settling velocity analysis of the construction site runoff generated highly variable results due to the finely sized grains found in the runoff. The project analyzed field settling tests, as well as laboratory settling tests. In the laboratory, the tests indicated that initial settling of sediment was quite rapid with as much as 60% removal within 6 hours. It was observed that natural flocculation behavior appeared to accelerate initial settling velocities. However, after the first six hours, any additional increments of sediment removal were more difficult to obtain. For example, it took an average of over 16 more hours to get the next 18% increment of removal, and another 28 hours to get the next 13% increment. After 48 hours of settling, an average sediment removal rate of approximately 90% was achieved. The remaining sediment in suspension was composed of extremely fine clays and colloids that are highly resistant to settling. Field estimates of settling rates after storm events generally support the laboratory settling column data.

The study made some interesting conclusions. Listed below are the major factors influencing the performance of sediment control basins:

- The settling behavior of sediment particles in construction site runoff;
- The size and intensity of the storm event producing runoff at the construction site; and
- The stage of construction activity at a particular construction site.

These factors provided the basis for the experimental design is examining the sediment containment at the Richmond Hill site.

CHAPTER 3

3.0 STUDY SITE

3.1 Site Description

The study site is a wet pond located in the northern part of the Town of Richmond Hill, Ontario. The sediment control pond is releasing stormwater runoff, which will eventually feed into Lake Wilcox downstream. There are two inlets servicing two catchment areas. The catchment, approximately 15.1 hectares in size, is located south of Sunset Beach Drive and west of Bayview Avenue within the Community of Oak Ridges in the Town of Richmond Hill. The catchment area is currently undergoing residential development from agriculture and woodlots.

According to the development engineer, the pond is designed with water quality criteria of 24 hr detention of the runoff from a 25mm short duration storm. Water quantity controls include 2 through 100 year post to pre development to a maximum outflow which equates to the capacity of a downstream roadside ditch (Sabourin Kimble & Associates Ltd. 2000). The developer instituted a permanent pool volume within the detention facility. The permanent pool volume was calculated using the MOE Stormwater Management Practices Planning and Design Manual (SWMPP) assuming a Level 1 fisheries habitat classification (Sabourin Kimble & Associates Ltd. 2000). The pond is equipped with a sediment forebay, where it is anticipated that the bulk of the coarse sediments will initially settle. This is located at the south end of the pond, where the two inlets discharge runoff. To distinguish between the two inlets, the inlet located in the southwest end of the pond is referred to as 'Inlet 1070'. This refers to the diameter of the pipe. The second inlet, located on the opposite side is referred to as 'Inlet 510'. Bulkheads were installed at both inlets to reduce the amount of sediment entering the pond during the construction period. The bulkheads are removed after construction is complete and the sediment control pond is converted to a stormwater management pond. Figure 3 is a photograph of the study site.

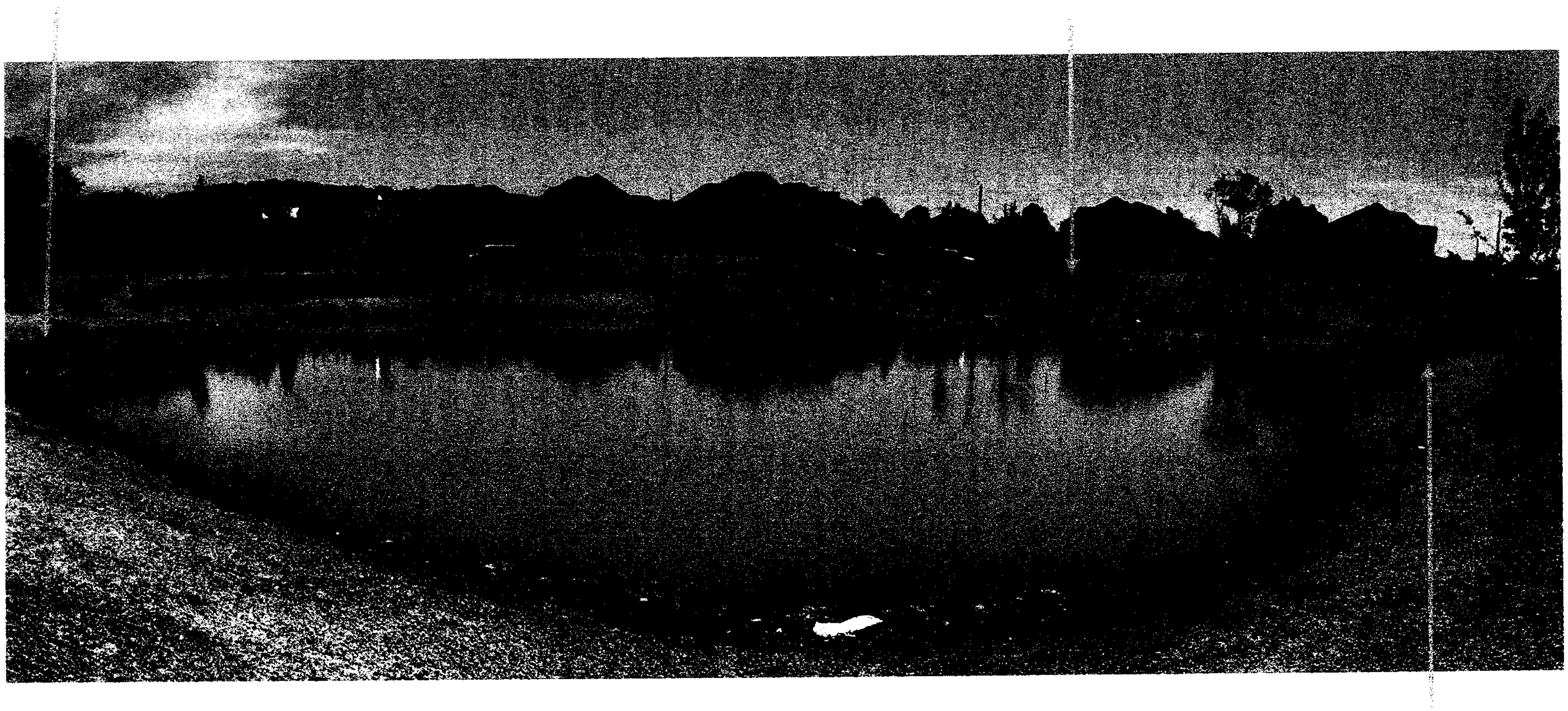


Figure 3: Richmond Hill Study Site

The use of bulkheads is a standard practice for construction sites using sediment control ponds. These are installed in the maintenance chamber of the sewer pipe discharging runoff into the pond. They are designed to decrease the amount of sediments entering the pond.

The total volume provided by the permanent pool is 2361 m^3. In addition, the total active storage volume provided is 5159 m^3. Tables 10 and 11 describe the permanent and active stage storage characteristics of the pond.

Table 10: Permanent pool stage storage characteristics ultimate pond (Sabourin Kimble & Associates, 2000)

Elevation (m)	**Area (m^2)**	**Average Area (m^2)**	**Depth (m)**	**Volume (m^3)**	**Total Volume (m^3)**
297.5	94				0
		397	0.5	198	
298.0	700				198
		891	1.0	891	
299.0	1082				1089
		1413	0.9	1272	
299.9	1744				2361

Table 11: Active storage volume - stage storage characteristics (Sabouring Kimble & Associates, 2000)

Elevation (m)	Area (m^2)	Average Area (m^2)	Depth (m)	Volume (m^3)	Total Volume (m^3)
299.9	2575				0
		2612	0.1	261	
300.0	2650				261
		2958	1.0	2958	
301.0	3265				3219
		3880	1.0	1940	
302.0	4495				5159

The outlet control structure includes a reverse sloped pipe configuration which was designed as the water quality outlet for the pond (Sabourin Kimble & Associates, 2000). According to Sabourin Kimble & Associtès (2000) the structure extends from an elevation of 298.1 within the deep pool at the north end of the facility, to an elevation of 299.9. The elevation at the beginning of the outlet structure is 1.8 m below the permanent pool elevation. In addition, the structure ends at the permanent pool elevation inside a manhole within the pond berm. The outlet control structure is equipped with a 112mm diameter orifice at elevation 299.87m to control the 48 hr drawdown time of the volume equivalent to a 25mm event (Sabourin Kimble & Associates, 2000).

The site conditions were continuously changing throughout the construction process. When the monitoring program began, the catchment area serviced by inlet 510 was sodded, and construction complete. However, the catchment area serviced by inlet 1070 was completely exposed, and construction activities were still in progress. During this time period, dirt piles were present throughout the inlet 1070 catchment area and the roads were covered with soil that had eroded from the site. In addition, the pond was experiencing erosion along its banks. The vegetation growth was not completed for most of the fall sampling period. Figure 4 is a photograph of the erosion that occurred along the banks of the pond.

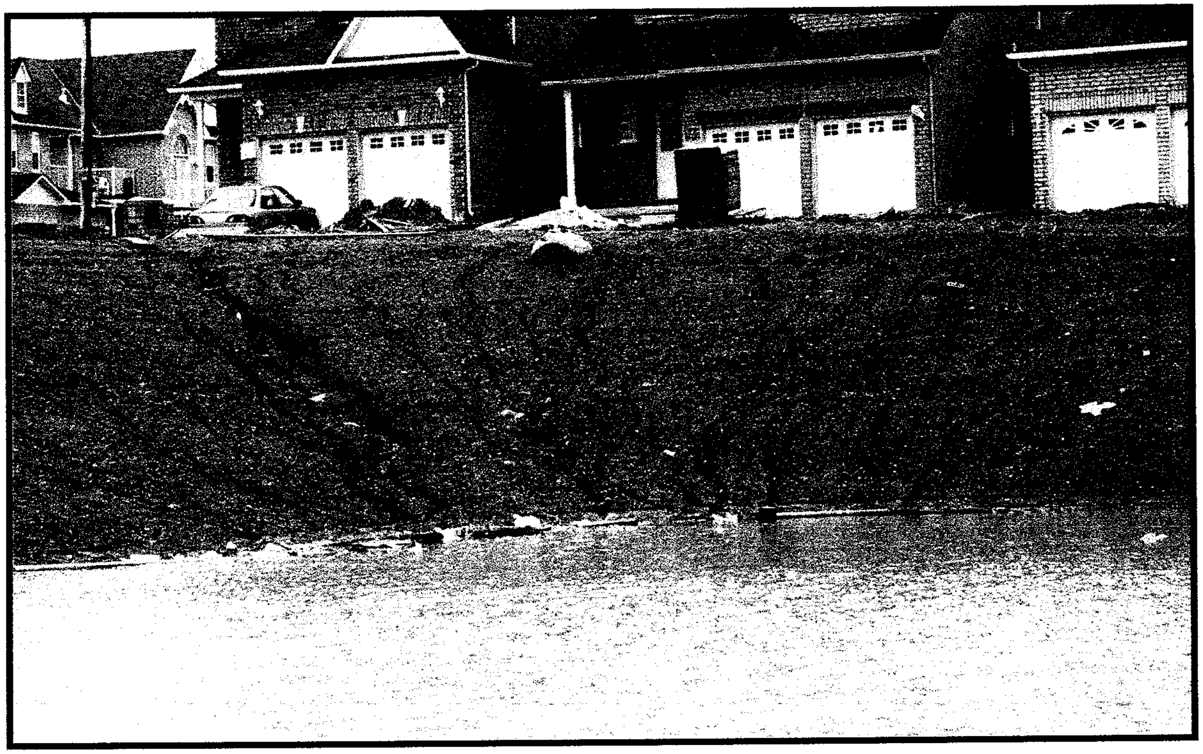
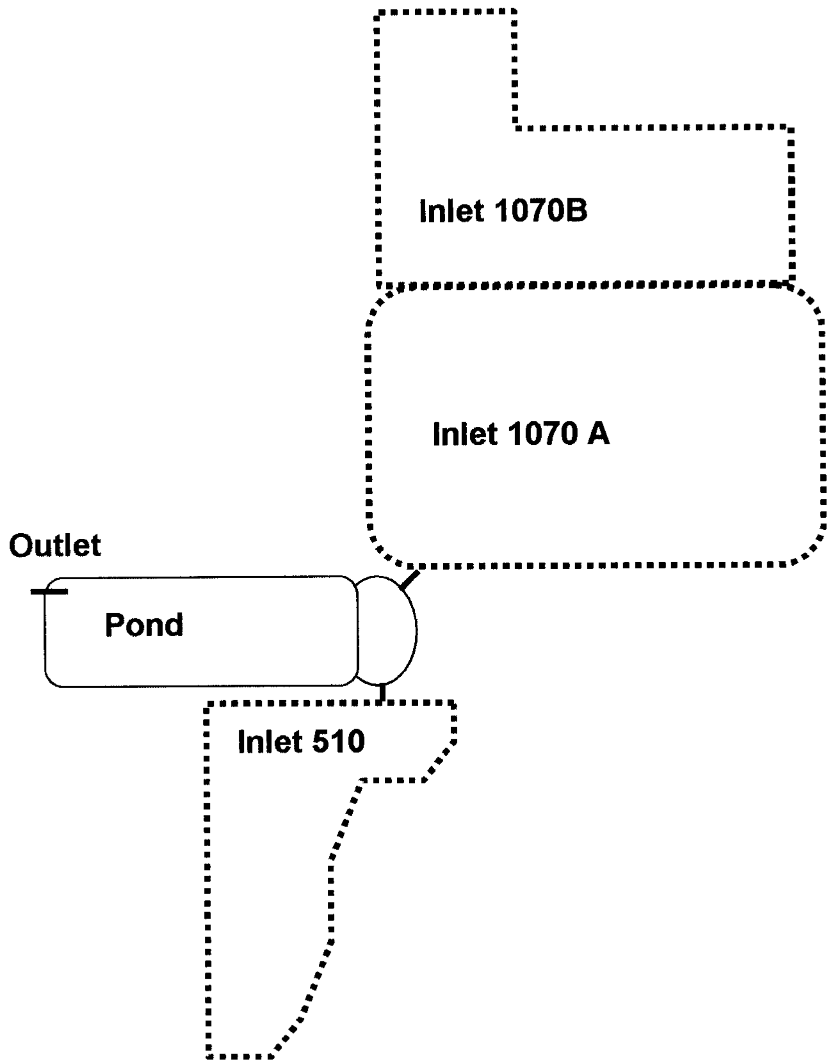


Figure 4: Erosion along banks of pond

3.2 Catchment Area

The catchment area is approximately 15.1 ha in size. The total lot area draining though the inlet 1070 sewer outlet is approximately 9.8 ha. In addition, the total lot area draining into the pond through inlet 510 is 2.39 ha. Figure 5 provides an outline to each of the areas draining into the pond. This figure is further broken down into three more figures that were derived from the site drawings.

Figure 5: Schematic diagram of sediment control pond with catchment area



A soil investigation was also investigated before the site began development. The purpose of the investigation was to determine the subsurface conditions and engineering properties of the soils within the catchment area. The investigation revealed that the site is situated on Markham till plain where glacial tills have been modified by lacustrine sand, silt and clay, or by kames consisting of sand and gravel. The study also revealed that the regional topography is moranic, where occasional thick peat deposits occur in the local depressions. The subsurface conditions included a stratum of silty clay till. In addition, the study encountered layers of fine sand and silt laminated within or below the silty clay till in different areas (Howieson & Chan 1997).

3.3 Downstream Habitat

Lake Wilcox is the primary receiving water body from the outflow of the pond. Located within York Region, and the Oak Ridges Moraine, Lake Wilcox has been recognized as a valuable recreational and environmental resource (FCM 2000; Town of Richmond Hill 2003). Lake Wilcox is a kettle lake, formed by the melting of a remnant block of glacial ice (Town of Richmond Hill 2003). The lake has a surface area of approximately 55.6 ha, with a watershed area extending over 260 ha within the moraine (refer to Figure 5) (Town of Richmond Hill 2003).

Unfortunately, Lake Wilcox has been exposed to various pollutants through the ever increasing anthropogenic activities undertaken within its watershed (Town of Richmond Hill 2003). This has resulted in an unhealthy aquatic ecosystem. According to the Town of Richmond Hill (2003), issues associated with the water quality include the buildup of phosphorus in the water column, a lack of oxygen in the bottom waters, too many nuisance plants in the water and too few fish of desirable species. Nutrients from fertilizers, septic systems, and naturally occurring sources within the watershed have collected within the lake over time (Town of Richmond Hill 2003). Other issues include the lack of clarity in the water, altered shoreline, inadequate water balance creating the need to control the outlet via dam structure, and a shift in aquatic communities (Town of Richmond Hill 2003).

As a response to the degradation of Lake Wilcox, community members have implemented the Lake Wilcox Remediation Strategy whose primary goal is to return the lake to its self-sustaining state (FCM 2000). Past efforts have included controls on external nutrient sources, with some success. However, nutrient cycles and the breakdown of organic material within the lake have resulted in a significant depletion of oxygen in the bottom waters, which in turn are causing the lake to deteriorate rapidly (FCM 2000). The remediation strategy has involved the implementation of a process called 'Lake Lung', where an experimental tool was used to increase bottom water oxygen levels without altering the temperatures or disturbing sediments (FCM 2000). The project was implemented in 1997, and monitoring programs continued through 1998. These efforts improved the health of the Lake, however, efforts are still underway to continue in the community's quest for returning Lake Wilcox to a self-sustaining state. For example, in 2000, the Town of Richmond Hill strengthened its environmental practices with financial support through a municipal funding program called the Green Municipal Funds (GMF) (2000). The funds were allocated to continue monitoring and restoration programs that are continuing today to help restore Lake Wilcox with the goal of maintaining a sustainable community within the area (Town of Richmond Hill 2003).

CHAPTER 4

4.0: MONITORING PROGRAM

In order to meet the objectives, a monitoring program was developed and implemented. This chapter outlines the methodology undertaken during the planning and implementation stages of the project. The chapter begins with a description of the development process for the monitoring program. This is followed by a description of the equipment calibration and installation. In addition, the methodology used to monitor construction activities as well as water quality and flow data are presented. Finally, an overview is provided on the methods used to analyze the data collected.

Various parties participated with the planning and implementation of the monitoring program for this thesis. The basis for the monitoring program was developed by the TRCA, Dr. James Li (Ryerson University), and the Ministry of the Environment. In addition, the equipment calibration and testing was implemented by several graduate students including Harry Manson, Derek Smith, Angelune Des Laurier, Renata Krasnova, and myself. In addition, the analytical procedures implemented on the water quality samples were performed by the MOE Laboratory Services branch.

4.1 Development of the Monitoring Program

Before developing a monitoring program, an overall understanding of the area under study was gained. This helped determine what type of equipment and resources were needed. For example, Wanielista and Youseff (1993) claim that the most reliable sampling is done by manual means, provided that the labour force understands its tasks and its members are punctual. However, due to the unpredictability and highly variable nature of weather events, it is difficult to rely on a team of individuals to be available at all hours. Therefore, instrumentation is typically acquired to collect events.

In order to determine the type of equipment and resources needed, a monitoring protocol was developed. The thesis scope was outlined and the methodologies to attain the objectives were created. From there, the type of equipment and resources acquired in order to perform the methodology was determined. In addition, it was important to approximate a time scale for the different tasks. Thus, Table 10 presents the scope, methodology, equipment and resources needed to pursue the methods used to collected data and samples, and the estimated timescale for performing the methods.

Table 12: Scope and Methodology of Study

To characterize the bottom sediment accumulation	To characterize the particle size distributions and settling velocities of sediments entering and exiting the pond	To characterize the Event Mean Concentrations (EMC) of sediments and the average concentrations of pollutants entering and exiting the pond	**Scope**
The change in elevation of the bottom of the pond was monitored throughout the course of the study. Construction activities were also monitored to relate to the bottom sediment accumulation as well as the performance of the pond.	Particle size distributions were included as a parameter measured in the water quality analysis. Results from particle size distributions were used to estimate settling velocity using Stoke's Law.	Collection of water quality and quantity data. Water quality samples need to represent the majority of the event. Therefore, samples were collected at specified time intervals throughout the duration of the event to calculate EMC values. In addition, each sample was mixed to form a composite or average sample for each event.	**Methodology**
Equipment needed: standard surveying equipment, boat *Resources*: labor	*Equipment needed*: water quality samplers *Resources needed*: laboratory services	*Equipment needed*: water quality samplers and flow measurement devices, and rain gage *Resources needed*: vehicle access, laboratory services and space, labor	**Equipment and Resources Needed for Methodology**
Bottom sediment accumulation was monitored every 1 to 2 months Construction activities were monitored once every two weeks to correspond with water quality data and bottom sediment accumulation	Same time period water quality sampling took place	A few months were required to obtain all equipment. Sampling period extended over fall season, stopped during winter months as equipment cannot withstand subzero temperatures, and then resumed the following spring.	**Timescale**

Based on Table 10, equipment and resources were obtained for the collection of water quality and quantity data, rainfall data, and the measurement of bottom sediment accumulation. The following sections describe the type of equipment and resources used to perform the outlined methodology.

4.1.1. Water Quality and Quantity Data

After gaining an overall understanding of the site, it was determined that three monitoring stations were needed at each inlet and outlet of the pond to characterize the main inputs and outputs of the pond. The scope of this thesis requires that each monitoring station is responsible for collecting water quality and quantity data. Therefore, water quality samplers, and flow measurement devices were acquired for each station to complete this objective. The following sections describe the equipment selection for the study.

Water Quality Samplers

The ISCO 6712 portable samplers were selected for wet weather water quality analysis. These samplers have enough memory to store five sampling programs and sampling data that can be viewed through the sampler display (ISCO 1993). The samplers were chosen based on their ability to collect individual samples over specified time intervals to account for the variability in constituent concentration throughout the course of the runoff event (ISCO 1993). The samplers are equipped with 24 bottles that can collect water quality samples at specified time intervals throughout the course of the event. At the Richmond Hill study site, the portable samplers are installed at the two inlets and one outlet of the pond.

An alternative to using water quality samplers would be to obtain grab samples at the site during an event. However, according to Waneilista and Youseff (1993), grab sampling provides poor estimation of runoff concentration, flow volume, and mass loadings. With grab sampling only instantaneous values are obtained, which lacks even an average representation of the event conditions. In addition, this method fails to characterize the variability and changes that occur during the course of an event. Thus, it was determined that the use of the water quality samplers and the collection of discrete samples throughout the course of the event was the best method for collecting water quality data.

Flow Logger

Flow data is collected through the use of a flow logger, and an area velocity sensor. The *4150 Flow Logger,* manufactured by ISCO Inc., is capable of measuring the average velocity and level, while calculating the flow rate of the stream (ISCO 1993). The 4150 depends on an IBM compatible computer running FLOWLINK3, ISCO's flow-data management software, for programming and calibration (ISCO 1993). The software, FlowLink, is used to program and download data from the loggers. Data downloaded onto this program is converted to an EXCEL spreadsheet for interpretation and analysis.

Area Velocity Sensor

The *Area Velocity Sensors* (a/v probes) are used at each inlet and outlet. Using technology based on the Doppler effect, the velocity sensor directly detects the average velocity of a stream as the flow moves up or downstream (ISCO 1993). In addition, the a/v probes contain a differential pressure transducer that senses the hydrostatic pressure produced by the water above the sensor (ISCO 1993). The velocity sensors can be programmed to trigger the ISCO 6712 samplers where there is an increase in level or flow. At the Richmond Hill study site, the a/v probes are programmed to trigger the samplers based on an increase in level.

Rain Data

A representative study for stormwater quantity and quality monitoring includes the use of a rain gauge on site or within close proximity to the study site's boundary. Measured rainfall at a small scale interval (i.e. 15 to 60 minutes) within the study area will reassure event start times for measured flow and level within the pond. A local rain gauge will also give cleaner hydrographs since the rainfall should reflect the intensity and duration of the event when plotted against level and flow measurements. The Richmond Hill study is fortunate to have access to a rain gauge operated by the Town of Richmond Hill down the street from the site.

4.1.2 Bottom Sediment Analysis

Throughout the life of a construction sediment pond, a significant amount of sediments accumulate at the bottom of the facility. This is due to the high erosion rates experienced at construction sites during a rainfall event. According to Marsalek *et al* (1997), once the pond's bottom sediment levels have exceeded a critical value, the performance of the pond, in terms of its flow control and quality control, can be affected. The cause of this interference can be explained by (i) reduced settling times resulting from reduced storage volume; (ii) potential erosion of bottom deposits, and (iii) release of contaminants from the sediment into the overlying water column and their discharge from the pond into the receiving waters. Thus, the rate at which the sediment accumulates with the pond is an important factor to consider when evaluating the performance of the Richmond Hill pond.

To measure bottom sediment accumulation within the pond, standard surveying techniques were used. A submersible surveying rod was constructed to meet the needs of this task. Figure 6 shows the rod, which includes a flat bottom with spikes driven through to help anchor the rod to the bottom of the pond. By examining the pond elevation drawings, fourteen points were selected, where elevation measurements were taken. Stakes were driven into the pond bottom that extended above the pond's surface to mark where each of these points was selected. At each stake, the rod was submersed and the elevation marked. This was later converted into absolute elevation levels. The difference between the pond's bottom elevation and the measured elevation using the rod was then calculated. This method was repeated twice in the fall of 2002 and once during the spring of 2003. From these measurements, the accumulation of sediments was observed and compared with the different stages of construction at the site.

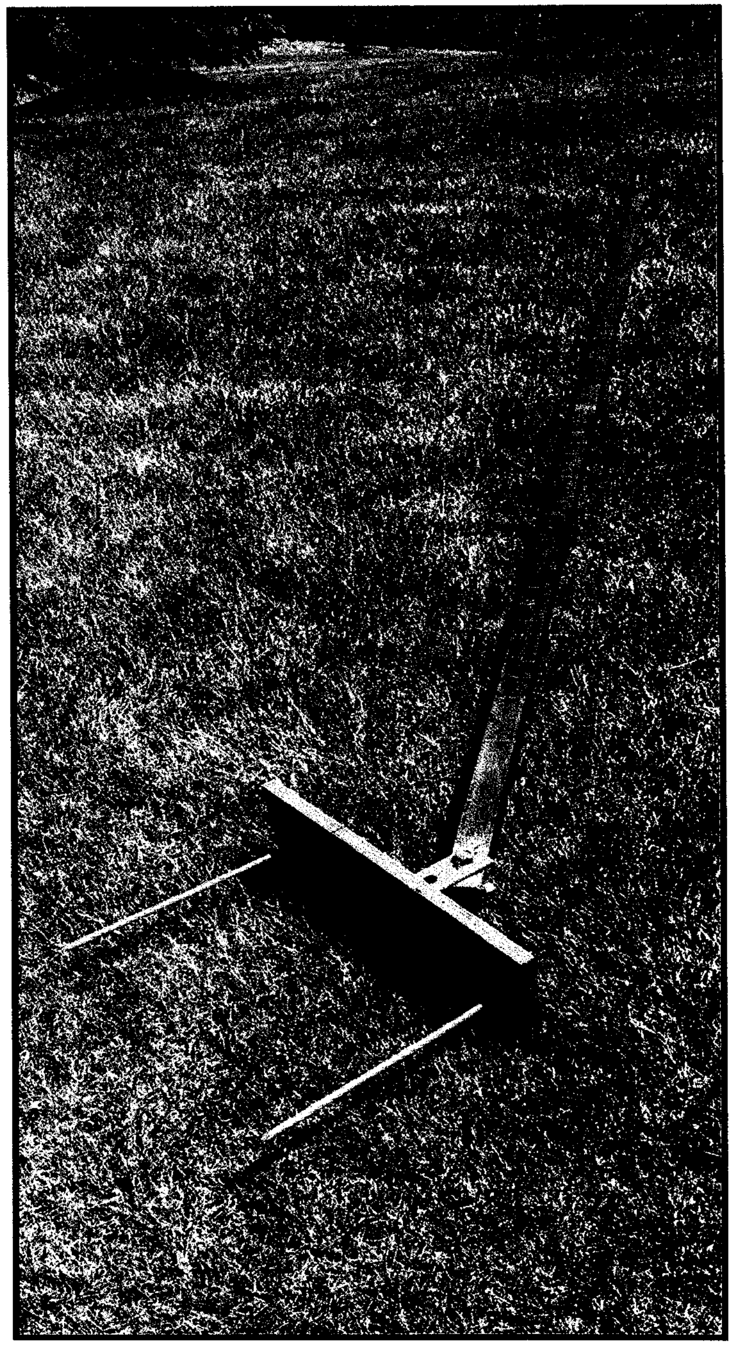


Figure 6: Submersible surveying rod

It is important to note that this method of measurement is regarded only as an approximation of bottom sediment accumulation. It is recognized that in using standard surveying techniques the elevation levels observed may be inaccurate, but can provide a general understanding of the rate of accumulation experienced in the pond.

4.1.3 Other Resources

The monitoring program required several other resources in order to achieve its objectives. These included access to large vehicles (to carry equipment, tools, samples, etc.), laboratory services for analysis, preparation area for collected samples, and tools for maintenance and installation. Fortunately, these resources were provided in kind from several agencies, including the Ministry of Environment, and the Toronto and Region Conservation Authority. The vehicles, tools, and lab facilities were provided by the MOE as well as one monitoring station. The remaining monitoring equipment was provided by the TRCA.

4.2 Equipment Installation and Operation

Before the equipment was installed in the field it was tested and calibrated in the lab. For example, the flow measurement devices were tested and calibrated to ensure accurate readings. Once the calibration of equipment was complete, the installation and set-up of the program began. This process occurred in several stages. First, the location of each monitoring station and all its components was determined. Security measures were then taken in order to avoid vandalism and possible damage incurred through the exposure to weather processes. The samplers and flow loggers were then programmed to collect the required data under the desired conditions. Once in operation, the logging memory and downloading practices were implemented and maintenance procedures were outlined. The following sections review the methodology undertaken to achieve these tasks set out at the Richmond Hill site.

4.2.1 Placement of Monitoring Equipment

The governing concern for determining the location of a monitoring station is the type of data to be monitored. For example, the total flow entering a sedimentation pond should be measured downstream from all connecting sewers at the inlet to the pond. If there is more than one storm sewer inlet, then it must be monitored as well. For example, it was observed that a large bulkhead was positioned upstream of inlet 1070 to the Richmond Hill pond. While the bulkhead did obstruct the inlet flow, it was a characteristic of a typical construction pond and needed to be incorporated into the study design. An AV probe located upstream of the bulkhead could easily be flooded with backflow during a storm event or buried by accumulating sediment. Alternatively, an AV probe located immediately after the bulkhead, would have splashing "waterfall" like conditions that could readily give incorrect measurements. For this study, the AV probe was located downstream of the bulkhead where the flow was smooth and close to the inlet culvert and pond interface. This was determined to be the best site for characterizing the inlet flow under the design characteristics of the pond and the sewershed system (ISCO 1997b). Overland flow along the banks of the pond can also contribute water quantity and quality inputs to the pond, but are difficult to quantify. These inputs are considered in the analysis of water quality results presented in a later section.

Equipment flooding was a concern in the Richmond Hill study. As stated previously, the pond is being monitored during the construction phase of local housing developments where bulkheads are installed upstream of the inlets to limit the discharge of sediment to the pond. As a result, the runoff inflow pools within the sewer maintenance chamber before continuing through the sewer pipe and discharging into the pond. While flooding of the local street does not occur, the bulkheads pool enough water to surpass the engineered level of the maintenance chamber (i.e. over the chamber shelf). This eliminated the possibility of placing the monitoring equipment underground in the chamber. In addition, fluctuating pond levels during an event raised concerns on placing the monitoring equipment along the edge of the pond. As a result, the monitoring equipment was located just above the engineered 100 year-storm waterline. Several metres of intake line and an AV probe extension cable were used to compensate for the distance.

For the Richmond Hill study, all three monitoring stations are positioned above the inlets to the pond, and the outlet in reinforced huts. Each hut is secured to the ground by several 4 foot t-bar posts. Each post was driven into the ground, and attached to the hut (refer to Figure 6). There are several other methods that could have been used to anchor a monitoring station. Usage depends on site characteristics and materials present that may hinder or be used as an advantage in securing the monitoring station. For example, materials underground including the culvert below, large rocks, and packed soils, made driving in the posts a challenging task at the Richmond Hill study site.

When determining a location, factors to consider include the height of the hydraulic head between the sampler and where the intake sieve is located. For example, the maximum height an ISCO 6712 auto sampler can be positioned and still draw samples is 8.5m (28ft). The distance from sample points can affect sampler performance by under/over filling bottles even with programmed adjustments (i.e. hydraulic head, intake line length)(ISCO 1997a; ISCO 2001).

4.2.2 Security Measures

All equipment should be installed in consideration of vandalism. In order to prevent this, each monitoring station requires security and protection to all equipment. At the Richmond Hill site the samplers were placed in reinforced huts with corner brackets and cross braces. The huts for inlet 510 and the outlet were drilled into a concrete block where the openings of the pipes are located. Inlet 1070 was attached to the fence located above the inlet. All monitoring stations were secured with locks for doors and lids. In addition, all tubing and cables were protected with PVC piping. The Richmond Hill study experienced several installation and programming issues that can affect the quality of measured data. For example, in order to protect both the intake lines and the AV probe cable, PVC piping was used to thread the cables through. The PVC was buried at the outlet, and attached to the cement of the culvert at both inlets. Outside of vandalism, the PVC will also protect the lines and cables from damage caused by animals and environmental exposure. By keeping the line exposure to a minimum, risk of a cable being severed or a splitting the line was minimized. Figure 7 demonstrates the security measures taken to protect against vandalism.

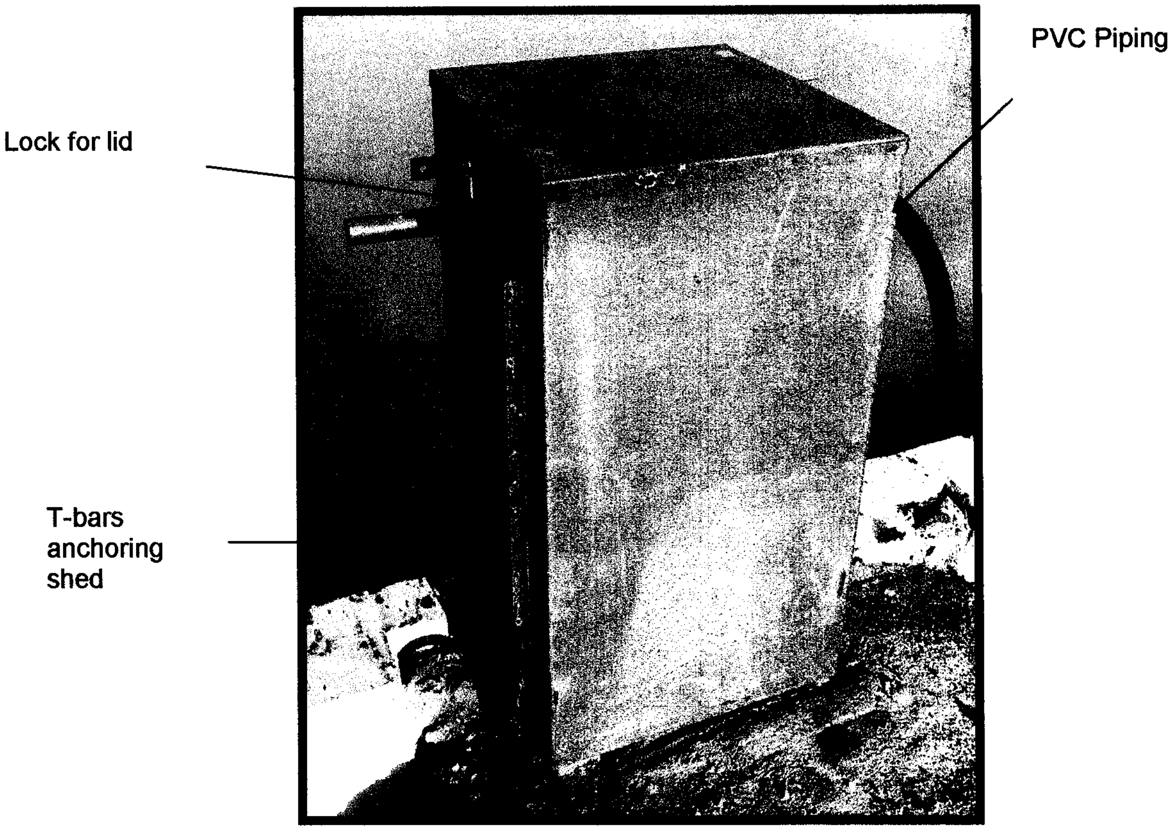


Figure 7: Security measures taken at Inlet 510

4.2.3 Equipment Installation

The position of the AV probe in a culvert should be located in an area that is flat and has no obstructions. For instance, spaces that occur between fitted culverts can cause irregular flow (i.e. riffle water or white capped water). In this case, a probe positioned immediately downstream of this kind of riffled water can cause irregularities in level measurements (ISCO 1997b). A culvert ring can be used to attach the probe and intake. However, in some cases it can introduce error into the measurements. For example, space between the culvert and the bottom of the ring can cause water to flow above and below the ring. If the flow has a high velocity, then the ring can start to vibrate, similar to a hydrofoil. The vibration can cause discrepancies in level measurements up to several millimeters. The Richmond Hill study avoided the use of a culvert ring and simply attached the probe and intake directly to the culvert using metal strapping. The strapping was positioned at the back of the AV probe in order to avoid blocking the Doppler signal (ISCO 1997b).

For representative sampling, the intake lines were composed of Teflon. This prevents any materials from sticking to the inside of the lines, thereby decreasing cross-contamination of samples from past events. Because Teflon lines are rigid, it is impossible for the tubing to be bent around sharp corners. In this case, flexible rubber hosing was used. Furthermore, all the monitoring stations had the intake line closest to the sampler (approximately within the last meter) equipped with rubber tubing. This allowed the sampler to be moved freely without running the risk of kinking or cracking the Teflon line. Any joints made between the Teflon and rubber tubing were reinforced with a pull-tie. This limits the possibility of a separation of the two tubes that will ultimately lead to the splitting of the intake line.

The sampler intake sieve and the AV probe were placed as close together as possible. This ensures that the samples collected are representative of the measurements taken by the AV probe.

4.2.4 Sampling Program and Procedures

Some programming features are difficult to determine until several events have occurred and the data are reviewed. This period is commonly referred as the *trial* and *error* period or 'pilot trials'. For example, in the Richmond Hill study, flow proportionate sampling was requested. However, it was observed that after an event, the pond experiences a slow drawdown time causing back flow to occur up the two inlet pipes. The back flows caused pooling and reverse flow measurements. Thus, the samplers were not collecting water samples under these programming conditions. As a result the samplers were set to trigger on an increase in level and paced to collect every five minutes. This way the logged flow data (at five-minute intervals) and the samples could be calculated to reflect flow proportioning. The samplers were programmed to collect 1 sample per bottle (24 bottles) at five-minute intervals, over a period of 120 minutes. Each bottle can collect up to 1L. The Richmond Hill study collected discrete water samples for each event and analyzed for Total Suspended Solids (TSS). Portions from each discrete sample were extracted and mixed to form composite samples. Composite samples were analyzed as average concentrations for several parameters including metals, nutrients, and organics.

Depending on site characteristics, a sampling program designed to collect time-weighted samples at five-minute intervals (120 minutes total) should be monitored carefully. For example, if the storm duration is less than 120 minutes, baseflow may be collected in some of the 24 bottles, resulting in diluted composite samples. If the storm duration is longer than the 120 minutes, then the sampler will not be able to characterize the entire event. At the Richmond Hill site there is little or no baseflow at the inlets discharging into the pond. Thus, dilution resulting from a short-duration storm event is not a concern. However, the tail of some pollutographs was still neglected due to the length of the sampling period. Since data collected in this thesis will aid in the calibration of future hydrological and water quality models, a greater representation of storm events was desired. Therefore, during the spring (2003) sampling season, it was decided to alter the sampling program and change the sampling intervals from five minutes, to collect every fifteen minutes. This change in time interval extended the sampling period from 120 minutes, to 360 minutes or 6hrs.

4.2.5 Logging Memory and Downloading

In most cases loggers can record data for well over 30 days. However, downloading and review of recorded data should occur more frequently during the monitoring program. Therefore, downloading procedures at the Richmond Hill's site were included in the maintenance schedule and did not exceed two weeks between each download. Malfunctions and unforeseen problems with the equipment in the field can occur.

Unless it is necessary, a larger logging interval should be used. A logger programmed to take measurements every minute verses every five minutes will use up battery power and will vacate memory storage five times faster. A larger sampling interval will allot more time between scheduled site visits. In some cases, the runoff into a facility is rapid and short, requiring a 1 or 2 minute interval. However, at the Richmond Hill site a five-minute interval was deemed appropriate.

4.2.6 Maintenance Procedures

A regular maintenance protocol was developed immediately after the installation. This protocol included equipment cleaning, downloading, equipment tests, and equipment checks for damage. In terms of equipment testing, most automated samplers have maintenance options such as testing the water distributor arm, manual sampling to test the pump, and RAM/ROM testing (ISCO 1997a). These tests were conducted on site during a scheduled maintenance.

Regular cleaning of the a/v probe, both internally and externally was conducted to limit erroneous measurements. For example, fine sediment can accumulate inside the a/v probe. The probes are designed to allow water to enter the probe via small holes; this water then depresses the pressure transducer to measure level (ISCO 1997b). Fine sediments are carried in this water and over time the pressure transducer can become plugged and cause inaccurate measurements. This was especially a concern at the Richmond Hill site where the influent is primarily construction site runoff. Furthermore, sediment can accumulate outside of the probe, and in turn, lower velocity flow. This occurrence can eventually bury the probe and interfere with the Doppler signal. Debris, leaves, and other anomalies can also become entangled around the probe, intake, and connecting lines; therefore, regular cleaning can limit this. In some cases, when the probe is blocked, covered with sediments, or the pressure transducer is clogged, the logger records error, and if not programmed properly will enter a zero value. At the Richmond Hill site, the outlet began recording errors, resulting in frequent zero value entries. Figure 8 demonstrates data that were collected under these conditions.

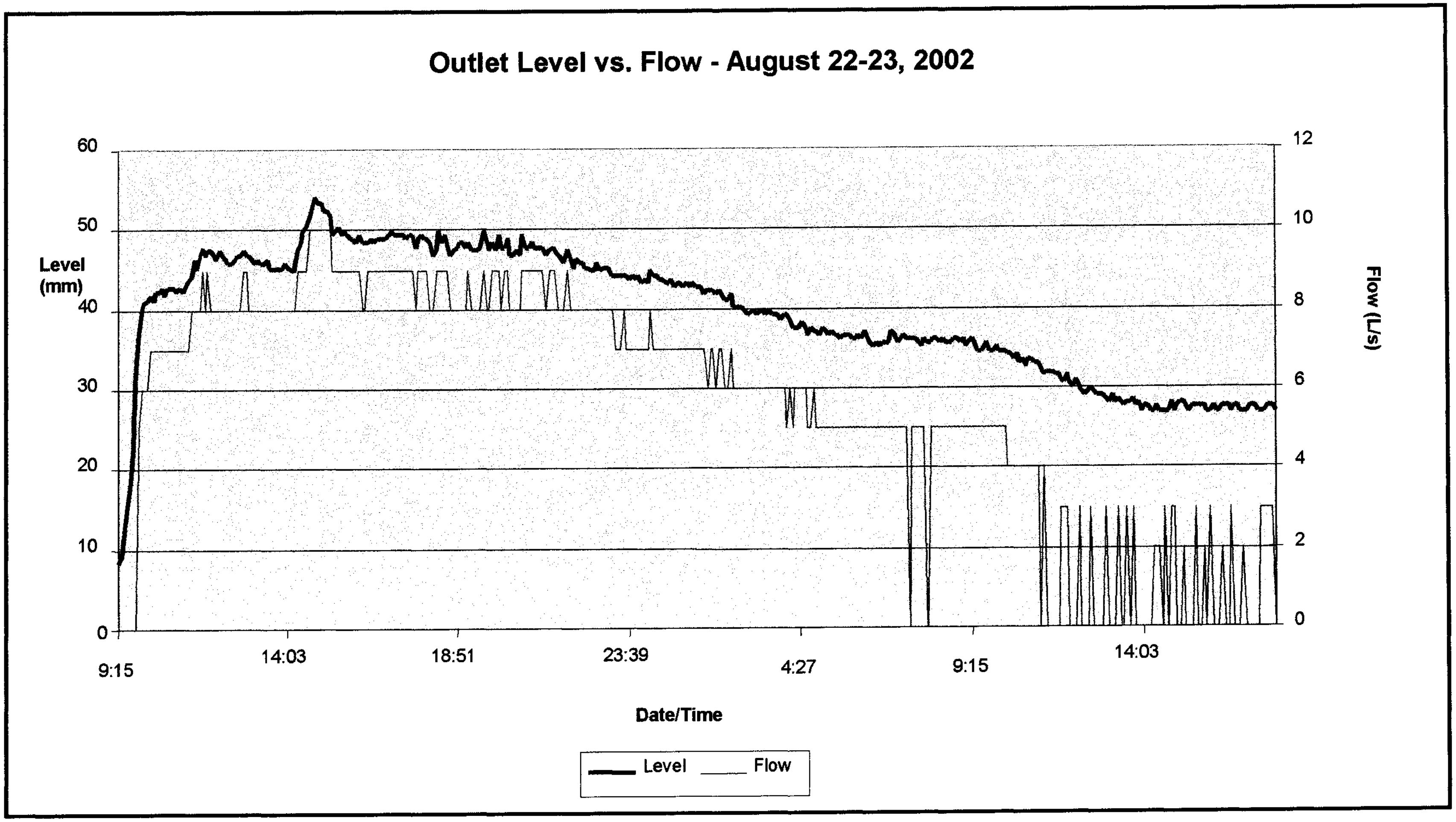


Figure 8: Erroneous outflow measurements

Figure 8 shows the level data was presented as a curve with an initial rising limb and gradual receding limb. However, the flow appears to continually drop to the zero value throughout the event. Frequent maintenance of the probe and intake can prevent this. In addition, a programming feature (Zero Level Offset option) through the logger was used. The software can be programmed to stop recording zero values when there is an error. Instead, the logger was programmed to approximate the values when there was an error by calculating the mean between two recorded measurements (ISCO 1997b).

4.3 Monitoring of water quality and construction activities

The development of the monitoring program and equipment calibration and installation took a considerable amount of time. The monitoring equipment was completely set up at the end of July 2002. However, the month of August served as the trial and error period where the sampling program was adjusting to meet the monitoring objectives. In addition, August experienced very little rain; two events occurred during this month. For these two events, the samplers were set to collect the events, however, during the first event Inlet 510 did not collect, and the second event, the outlet did not collect. The errors that occurred were caused by the back flow up the two inlets. This is shown in Figures 9 and 10.

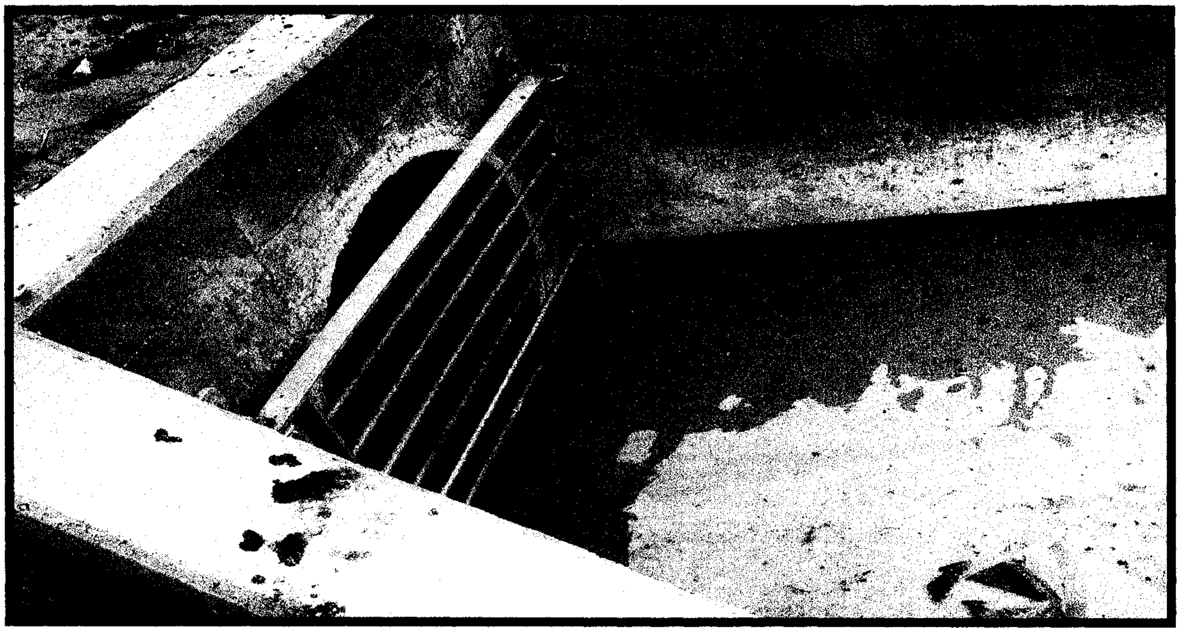


Figure 9: Backflow at Inlet 1070

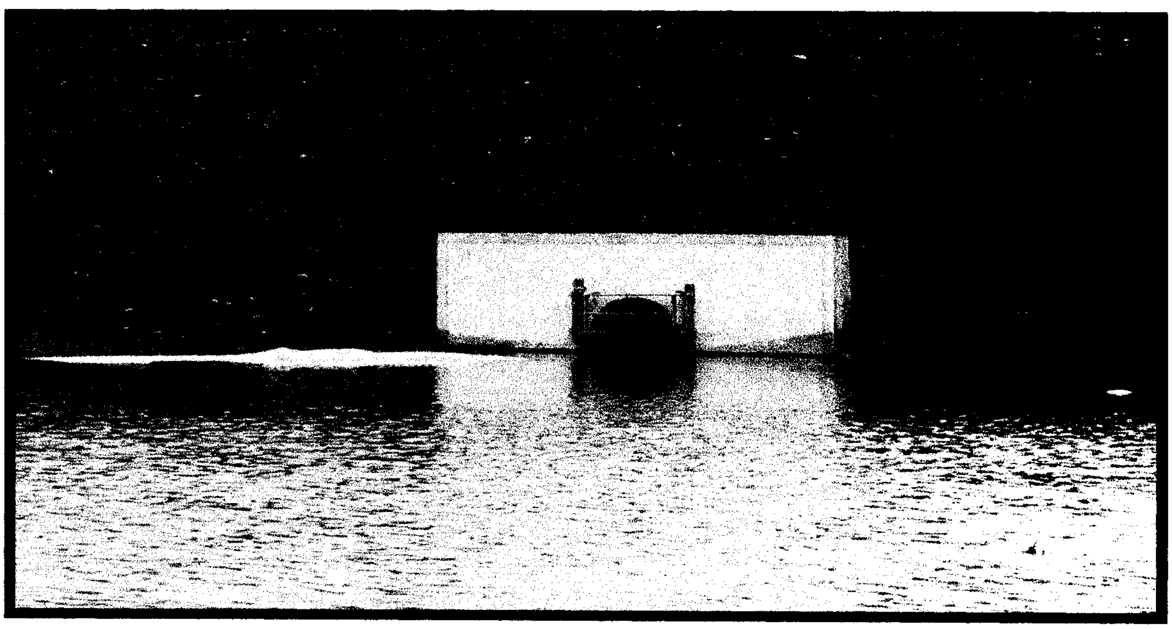


Figure 10: Backflow at Inlet 510

Two sampling periods were collected throughout the course of the thesis. The first sampling period occurred during September – October 2002. A total of 6 events were collected and analyzed. The second sampling period occurred during May – June 2003. A total of five events were collected during the spring 2003 period.

During both sampling periods, the progression of construction was monitored to determine the different stages of construction over the course of the monitoring program. The housing construction was complete at the time of both monitoring periods, thus the state of the lots and driveways were monitored. A checklist was developed that indicated the lot # and whether the lot was sodded or exposed, and whether the driveways were gravel or paved. In addition, notes were taken on the cleanliness of the roads, and if any soil piles were present and for how long. This was an important task as it provided an understanding of the material entering the pond through the two inlets. For example, during the first sampling period it was observed that Inlet 510 was complete in its development. However, the drainage area serviced by Inlet 1070 was completely exposed for the majority of the fall sampling period. Only during the last couple of weeks of October were some of the lots sodded. This still left the majority of the drainage area fully exposed. These conditions were reflected in the samples collected. For example, Figure 11 demonstrates the difference in samples collected from each inlet. The outlet is also included for comparison. In addition, these conditions were also apparent when comparing the conditions of the roads in both areas. The roads within the Inlet 510 drainage area were clean. However, Figure 12 demonstrates the conditions for the drainage area serviced by Inlet 1070.

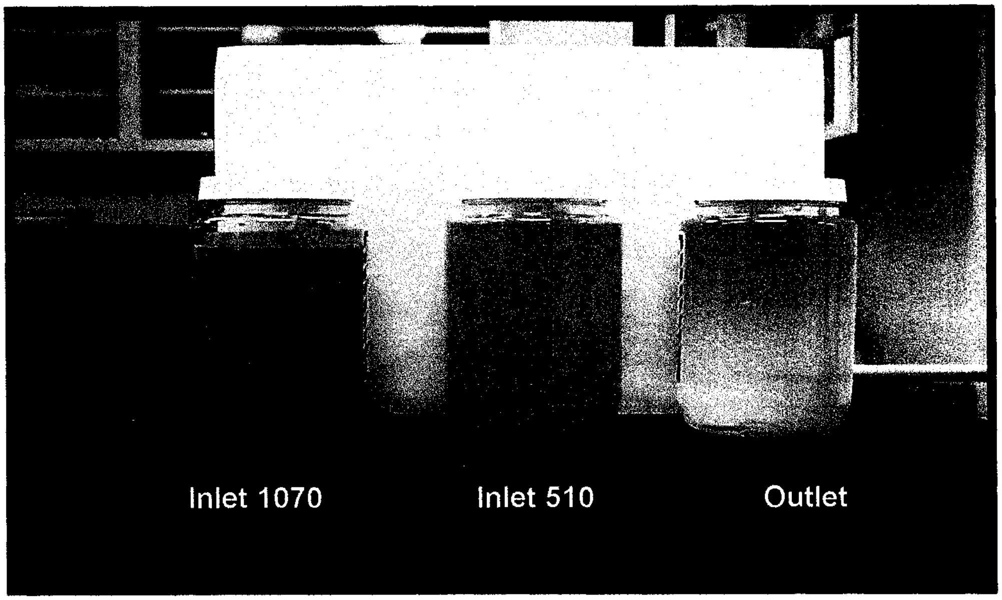


Figure 11: Samples collected from each monitoring station

Figure 12: Road Conditions in Inlet 1070 catchment area

4.4 Analysis of Results

Finally, the last step in the methodology involves the analysis of results. This includes the analysis of water quantity and quality results, settling velocities, and the correlation of the construction activities with the bottom sediment accumulation. These are described in the following sections.

4.4.1 Analysis of Water Quantity Results

Water quantity results include the data collected through the flow and level measurement devices, and the rainfall and pond level data collected through the Town of Richmond Hill's rain gauge. The flow and level measurements were recorded using the ISCO 4150 flow logger and area velocity probe described in the previous section. In addition, rainfall data obtained through the Town of Richmond Hill was also used to aid in the analysis of the runoff data. Table 13 describes what methods of analysis were used to interpret the flow and level measurements, and rainfall and pond level data collected.

Table 13: Analysis used to interpret water quantity data

Analysis	Significance
Rainfall/ runoff	By graphically displaying the runoff flow rate over time compared to the rainfall depth, characteristics of the event can be observed, and the performance of the pond can be examined. For example, if a storm was intense over a long period of time it would be reflected in the inlet hydrograph (i.e. sharp peak and drawn out falling limb). In addition, sediment control ponds are designed to slow runoff volume before being discharged downstream. The hydrograph for the inlets of the pond have sharp inclines and peaks. If the pond is exercising good quantity control, the outlet should show a gradual rising limb and falling limb to the curve.
Runoff Coefficient	This determined the ability of the catchment to retain runoff. For construction sites, this provides an idea of the degree of runoff to be expected. The coefficient represents the relationship between runoff and rainfall depth. Thus the total runoff depth recorded at the inlets was calculated and divided by the total rainfall depth to determine the coefficient.
Lag Times	The lag time of the catchment was calculated (time delay between the centroid of the rainfall hyetograph and the centroid of the inlet hydrograph). This reflected the intensity of the storm and catchment storage. For example, if it is a light storm it may be largely contained in depression storage (SWAMP 2002).

Table 13 Continued.	
Detention Times	Hydraulic detention times were calculated to further assess the performance of the pond in terms of its ability to exercise quantity control. By calculating the centroids of the inlet hydrographs and the outlet hydrograph, the detention time was determined through the difference of the centroids. The centroids were calculated from the first moment. Sediment control ponds are designed with specific quantity control criteria including a minimum of 24 hr detention time. The Richmond Hill pond was designed with a 48 hr detention time. Thus, it is important to determine if the Richmond Hill pond is performing as it was designed.
Drawdown Times	The Town of Richmond Hill has installed a continuous water level meter at the pond. The water level data collected can be plotted versus time. With this data, the drawdown time was computed for each event in two ways. The first method calculates the time difference between the maximum water level and the minimum water level during an event (SWAMP 2002). The second method was determined by calculating the water volume at the average level during an event and dividing it by the average outflow for that same event. Both methods were employed for the Richmond Hill site for comparison.

4.4.2. Analysis of Water Quality Results

The water quality results were determined by the Laboratory Services at the MOE. Table 14 outlines the water quality parameters analyzed in this study. The analytical procedures used by the MOE lab for each parameter measured are included in Appendix B.

Of the water quality parameters listed in Table 14, TSS was measured in the discrete samples collected at the site. The composite samples were analyzed for the remaining parameters. Once the samples are analyzed and results are received from the MOE laboratory services, they were interpreted for analysis of the pond's performance. Several methods were employed to evaluate the pond's performance. The following sections review the methods taken to analyze both discrete and composite data.

Discrete Samples

As previously mentioned, the discrete samples were analyzed for suspended solids. Suspended solids were the only parameter analyzed for discrete samples for two reasons. Firstly, it can be very costly to have discrete samples (72 bottles in total) for each event analyzed for various parameters. Secondly, suspended solids are the primary pollutant found in urban stormwater runoff, particularly from runoff generated at construction sites. In addition, suspended solids are a governing factor that influences pond design criteria.

After the discrete samples were analyzed for suspended solids, pollutographs were generated for each monitoring station for each event. These graphs plot the suspended solids concentration over time. In addition, the Event Mean

Concentrations (EMC) was calculated for each event. The EMC values were used to compare with current guidelines for suspended solids.

Table 14: Water Quality Analysis

PARAMETERS	
General Chemistry:	Conductivity, pH, turbidity, particle size distribution, chemical oxygen demand (COD), alkalinity, total solids, suspended solids, dissolved solids
Nutrients:	Ammonia+ammonium, nitrite, nitrate+nitrite, TKN Phosphate, total phosphorus
Metals:	Aluminum, Arsenic, Barium, Berylium, Calcium, Cadmium, Cobalt, Chromium, Copper, Iron, Magnesium, Manganese, Molybdenum, Nickel, Lead, Selenium, Strontium, Titanium, Vanadium, Zinc
Organics:	**PAHs**: Napthalene, 2-methylnapthalene, 1-methylnapthalene, 2-chloronapthalene, Acenapthylene, Fluroene, Phenanthrene, Anthracene, Fluoranthene, Pyrene, Benzo(a)anthracene, Chrysene, Benzo(b)fluoranthene, Benzo(k)fluoranthene, Benzo(a)pyrene, Dibenzo(a,h)anthracene, Benzo(g,h,i)perylene, 1-chloronapthalene, Perylene, Indole, 5-nitroacenapthene, Biphenyl **Herbicides and Pesticides**: 2,4-dichlorophenol, 2,4,6,-trichlorophenol, 2,4,5-trichlorophenol, 2,3,4-trichlorophenol, 2,3,4,5-tetrachlorophenol, 2,3,4,6-tetrachlorophenol, Pentachlorophenol, Silvex, Bromoxynil, Picloram, Dicambia, 2,4-D-propionic acid, 2,4-D, 2,4,5-T, 2,4-DB, Dinoseb, Diclofop-methyl

The EMC was calculated using Equation 1 (Waneilista *et al*, 1997):

$$EMC = \frac{\sum C_i Q_i}{\sum Q_i} \qquad (1)$$

where

C_i = concentration of sample *i*

Q_i = flow rate of runoff when sample *i* was taken

It should be noted that the EMC values calculated for this thesis are only partial representations of an event. During the fall season, the samplers were set to collect one sample every five minute. This produced suspended solid concentrations that extended over a 120 minute period. The EMC values were only calculated during the sampling time period. Thus, the samples were only representing a specific time period during the event. During the spring sampling period, the samplers were set to collect one sample every fifteen minutes. This extended the sampling period to represent 360 minutes of the event. Although this is still considered a partial representation, it increased the duration of samples collected and in turn extended the characterization of suspended solid concentrations for an event.

Using the EMC values for the inlets and the outlet, the concentration-based efficiency was calculated. This value helps evaluate the performance of the pond as it determines the concentration based pollutant removal efficiency as a percent. This was determined using Equation 2 (SWAMP Appendix B 2002)

$$CE = \frac{EMC_i - EMC_o}{EMC_i} \qquad (2)$$

Where

EMC_i = Event mean concentration entering the pond

EMC_o = Event mean concentration exiting the pond

Composite Samples

The composite samples represented average concentrations of each pollutant from all monitoring stations per event. Pollutants include metals, nutrients, and general chemistry such as chemical oxygen demand, solids, and alkalinity. Several methods were employed to analyze the results gathered through the composite samples. These are described in Table 15.

Table 15: Methods used to analyze water quality results

Method	**Description**	**Significance**
General Statistics	Calculate mean concentrations, upper and lower 95% confidence intervals, and standard deviation. All statistical calculations were based on log normal probability distributions.	To characterize the nature of the data collected.
Constituent Concentrations	The mean, maximum and minimum concentrations are plotted in comparison with the PWQO.	Graphically displays the range of constituents entering and exiting the pond. Compares this range with the PWQO to determine if any are exceeding provincial guidelines.
Performance	Removal efficiency expressed as a percentage. Calculated by determining the total mass of pollutant entering the pond and the total mass of the pollutant exiting the pond. This was calculated using the following equation: $\dfrac{\text{Mass (kg) In} - \text{Mass (kg) Out}}{\text{Mass (kg) In}}$	Demonstrates the performance of the pond based on its removal efficiency for the different pollutants.

Included in the water quality analysis of the composite data are the particle size distributions. The particle size distribution is significant as it describes the nature of particles that are entering and exiting the pond. As discussed in the literature review, it can be assumed that smaller particles tend to be associated with other pollutants such as heavy metals. Therefore, these smaller particles are of concern in evaluating the performance of the pond. The average cumulative particle size distributions were plotted. These graphs plot the percentage by volume of sediment that is greater than a given size against particle size. From these graphs it is possible to obtain several other characteristics of the particle size distributions collected. For example, the D_{50} value is calculated for this study. This value represents the median diameter. This size is assumed to characterize the bulk of the distribution (Simons & Sentrk 1992). The D_{10} values were also determined. This value represents the size of sediments where 10% of the distribution is larger size fractions (Chorley *et al* 1984). Alternatively, everything smaller than the D_{10} values accounts for 90% of the distribution. This value holds significance because the proportion of fine material largely controls many of the dynamic properties of sediments (i.e. porosity, permeability, shearing resistance, etc.) (Chorley *et al* 1984). Furthermore, the finer material is associated with many of the other pollutants found in stormwater.

4.4.3 Settling Velocities

The velocities were calculated using Stoke's law. Stokes law is applicable to discrete particles smaller than 0.062mm, or particles known as silt and clay (Simons & Sentrk 1992). Stoke's law can be determined using Equation 3 (Novotney & Olem 1994):

$$\omega = \frac{gD^2(\rho_s - 1)}{18v} \qquad (3)$$

Where

g = acceleration due to gravity

D^2 = particle diameter

ρ_s = specific gravity

v = kinematic viscosity

The settling velocities of both the D_{50} particle size and the D_{10} particle size will help evaluate the design criteria used to construct the pond. As mentioned in the literature review, the MOE refers to a particle size distribution that is characteristic of urban stormwater entering a pond as a design criterion for sizing the sediment forebay. Sizing criteria for this component is derived through the calculation of settling velocities of various particle sizes found in urban stormwater (MOE 1994). Design criteria for sediment forebays, which are included in the Stormwater guidelines, are based on data collected from the US through their NURP study (MOE 1994). As previously mentioned this is not indicative of the particle size distribution typically found in construction site runoff. Thus, by calculating the settling velocities of both the D_{50} and D_{10} particle sizes, it can be compared to those used to size the sediment forebay (refer to Table 8 in Chapter 2). In addition, Table 8 in Chapter 2 also provides a percentage distribution based on mass typically found in urban stormwater. In a previous study, the TRCA identified that sediment-laden runoff from construction sites in TRCA jurisdiction typically consist of soil particles that are smaller than 40 microns (Clarifica 2001). The table indicates that the percentage of particle mass less than 40 microns consists of less than 30% in urban stormwater. Thus by calculating the D_{50} and D_{10} values and their respective settling velocities, a comparison can be made between the nature of the particle size distribution found in the construction site runoff at the Richmond Hill site, and of that found in the particle size distribution recommended by the MOE Guidelines.

In calculating the settling velocities using Stoke's Law several factors must be considered. For example, the specific gravity can differ depending on the type of particle. The density of mineral particles ranges between 2 and 3 g/cm^3 and is often approximated as 2.6 g/cm^3 (Hemond & Fechner-Levy 2000). However, organic particles usually have a density only slightly greater than water. These particles also tend to contain a high fraction of organic carbon, which is an excellent sorbent for many pollutants (Hemond & Fechner-Levy 2000). Although runoff from construction sites has not been thoroughly characterized in terms of its typical particle sizes, shapes, and densities, urban stormwater runoff has been found to typically hold particles with relatively high specific gravities (Nix *et al* 1988). Table 16 presents different specific gravities according to the type of particle.

Table 16: Specific gravities for different particle types (Nix *et al*, pp.1333, 1998)

Type of Particle	Specific Gravity
Sand, silt, clay	2.65
Wastewater organics	1.0-1.2
Alum and iron flocs	As low as 1.002

Particle size distributions are typically made up of minerals, organic matter, and colloids. To determine these characteristics requires a separate analysis. Although the organic fraction of the particles found at the Richmond Hill site was not determined, the differences in specific gravity will be taken into consideration when calculating the settling velocities. Because urban stormwater runoff typically consists of particles with higher specific gravities, the settling velocities will be determined using the specific gravity outlined in Table 14. However, the specific gravity for wastewater organics was used to determine the settling velocities of lighter particles.

4.4.4 Construction Activities and Bottom Sediment Analysis

Construction activities were monitored on a bi-weekly basis. For each lot, the amount of soil exposed or sodded, and the condition of the driveway (gravel or paved) were recorded. Maps of the residential areas were used where lots were grouped and a size of the area provided (refer to Appendix G). For each of these areas, the percentage of lots exposed (in hectares) was determined based on the monitored activities. This was determined for each day monitored. This analysis demonstrated the level of activity over time and was taken into consideration when analyzing the quality results as well as the bottom sediment accumulation.

The bottom sediment accumulation was recorded every two months during the fall and spring sampling periods. Once the accumulation was determined, the results were compared to the construction activity monitoring results. This determined the relationship between construction activity and bottom sediment accumulation. In addition, the rate of accumulation and ultimate fate of the accumulated sediments will determine if the pond was designed with the capacity to hold the accumulated sediments. As mentioned in the literature study, if the accumulated sediments reach a critical threshold, the performance of the pond can be negatively affected. Thus, it is important to consider this in the analysis of bottom sediment accumulation.

CHAPTER 5

5.0 RESULTS AND DISCUSSION

The following sections examine the results collected throughout the course of this study. As mentioned previously, there were two sampling periods; fall 2002 and spring 2003. Thus, the results are evaluated for the pond's performance during both seasons. The water quantity results are presented first followed by the water quality results, the settling velocity analysis, and the bottom sediment analysis and the results from the construction activity monitoring. Finally, discussion of the issues encountered during the implementation of the monitoring program and analysis of results concludes this chapter.

5.1 Water Quantity Analysis

5.1.1 Rainfall – Runoff

The rainfall intensity varied for each event collected. Total rainfall for the fall sampling season (September – October 2002) was 128.8mm. This is similar to the reported value of 140.9mm, which is the 30 year average calculated for the Toronto Lester B. Pearson International Airport meteorological station for the same months (Environment Canada 2003). This meteorological station is used as it is the closest for the Richmond Hill site where 30 year averages are currently available. Both months experienced slightly less rainfall according to their monthly normal averages reported by Environment Canada (2003a). The month of September 2002 received 71.4mm of rainfall recorded at the Richmond Hill rain gage in comparison to the average 77.5mm reported by Environment Canada. In addition, the month of October experienced a total of 57.4mm while the average rainfall for this month is 63.4mm according to Environment Canada (2003a).

According to Environment Canada, the rainfall statistics included in Table 17 have been determined for the Toronto Lester B. Pearson International Airport meteorological station. These values are based on a thirty year average from 1971 – 2000.

Table 17: Rainfall statistics for the Toronto Lester B. Pearson International Airport meteorological station based on 30 year average (1971 – 2000) (Environment Canada, 2003a)

Days with rain:	September, 02	October, 02	May, 03	June 03
>= 2 mm	10.7	11.5	11.9	11.0
>= 5 mm	4.5	4.0	4.6	5.2
>= 10 mm	2.5	2.1	2.4	2.6
>= 25 mm	0.6	0.3	0.3	0.4

The climatic data provided by Environment Canada demonstrates that the Richmond Hill area typically experiences storms that are less than 10mm throughout the same time periods the samples were taken. This is only recognized as a possibility since the meteorological station is not located adjacent to the site. In addition, it is less likely that a large storm (> 25 mm) will occur during these months. Table 18 compares the rainfall volumes of the events sampled at the Richmond Hill site to the 30 year averages presented in Table 17. In addition, rainfall duration and average intensity are also presented to further characterize the events collected.

Table 18: Rainfall characteristics for events collected at the Richmond Hill site

Event	Total Rainfall (mm)	Rainfall Duration (hr)	Average intensity (mm/hr)
14-Sept-02	29.6	25	1.18
20-Sept-02	20.6	24	0.86
27-Sept-02	18	8.62	2.09
2-Oct-02	9.8	30.08	0.33
19-Oct-02	13	11.38	1.14
25-Oct-02	9.4	7	1.34
2-May-03	6.8	8	0.85
5-May-03	9	5.5	1.67
11-May-03	14.2	6.5	2.18
12-May-03	7	14	0.71
20-May-03	10.8	5.17	2.09

Almost all the events collected in the fall sampling season received greater than 10mm of rain; the events of October 2nd and 25th were 9mm and 9.4mm respectively. The event that occurred on September 14th 2002 was above 25mm at 29.6mm. This is significant since Environment Canada has reported the probability of generating a storm of this size is unlikely during the months of September and October. The events monitored during the spring season were less intense with the largest storm producing 14.2 mm of rain on May 11th.

The size of storms is significant, as they can greatly influence the performance of the construction sediment pond. Schueller and Lugbill (1990) discuss the significance of storm rainfall volume and its influence on sediment levels in the Maryland study presented in the literature review (Chapter 2). Schueler and

Lugbill (1990) reported that storms which generated less than a half inch of rainfall produced a median TSS concentration of 459 mg/L. The TSS concentration almost doubles to 748 mg/L, when storms generating between 0.5 and 0.99 inches were monitored.

The Town of Richmond Hill collected rainfall data through a rain gage located approximately 1km from the site near the intersection of King Rd and Young St. Rainfall intensity is typically graphed as depth per unit of time. This graph is referred to as a hyetograph (Wanielista *et al* 1997). In addition, runoff is typically reported in graphical format and is referred to as a hydrograph (Wanielista *et al* 1997; Waneilista & Youseff 1993). A hydrograph plots the flow per unit of time. Figure 13 includes both the hyetograph and hydrograph for the event of September 14, 2002. The remaining graphs are included in Appendix C.

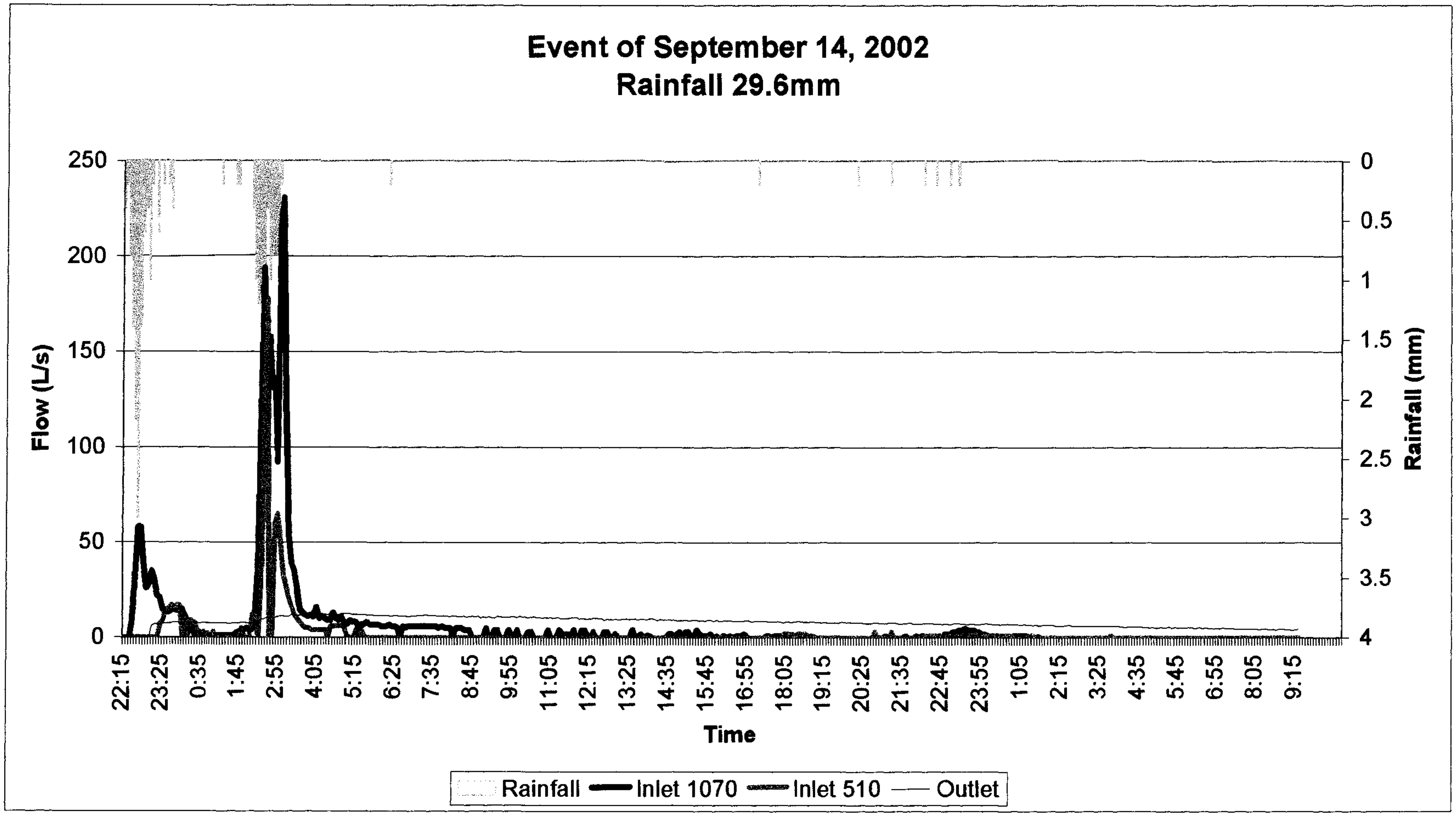


Figure 13: Hydrograph and hyetograph for event of September 14th, 2002

It should be noted that during the month of October the outlet logger experienced some difficulty. All flow and level data collected was lost due to a malfunctioning in the program settings. It is suspected that errors initially occurred when dirt and moisture were trapped in the outlet for the communication cord between the logger and the laptop. Fortunately, the Town of Richmond Hill installed a continuous water level measurement device in the pond. By obtaining the water level measurements of the pond, the outflow was estimated. Equation 4 was used to estimate the outflow.

$$Q = CA\sqrt{2g\Delta h} \tag{4}$$

Where

Q = outflow

C = orifice coefficient

A = area of orifice

g = acceleration due to gravity

Δh = hydraulic head of water level to centre line of orifice

The coefficient is defined by the relationship between the water level in the pond and the outflow. According to Brater *et al* (1996), an orifice coefficient of 0.6 can be applied to the outlet structure in place at the Richmond Hill pond. Thus, this coefficient value was used to approximate the outflow during the month of October.

5.1.2 Runoff Coefficient

The runoff coefficient defines the relationship between rainfall and runoff. This was determined by dividing the total rainfall volume by the total runoff volume for each event monitored. Thus, the runoff coefficient determined for this thesis is referred to as a volumetric runoff coefficient. Table 19 presents the volumetric runoff coefficient values for the catchment area serviced by the Richmond Hill sediment control pond.

Table 19 : Volumetric runoff coefficients for Richmond Hill study site

Storm Date	Runoff Coefficient
14-Sep-02	0.261
20-Sep-02	0.338
27-Sep-02	0.529
2-Oct-02	0.382
19-Oct-02	0.326
25-Oct-02	0.484
2-May-03	0.484
5-May-03	0.260
11-May-03	0.531
12-May-03	0.907
20-May-03	0.528

The volumetric runoff coefficients calculated for the events monitored in this study vary. The broad range extends from 0.26 to 0.9. The average coefficient is 0.457. There are many factors that may explain the variability in coefficients among the events included in Table 19. For example, the continual changes that occur at a construction site should be considered. The monitoring activities implemented for this thesis extended over two seasons. Within this length of time, several changes to the landscape occurred. One of the main factors that can influence the rate of runoff during an event is the amount of depression storage and infiltration activity over the catchment area. Because construction sites incur several changes on the catchment's landscape, the amount of depression storage and rate of infiltration can be altered continuously. Thus, for each event the runoff coefficients were influenced by the state of the site's condition. Secondly, the inter-event times and antecedent moisture conditions can also influence the runoff coefficient values. For example, the event of May 12th produced the highest runoff coefficient (0.907). This may be explained by the moisture conditions at the site during the event. Because there was a short inter-event period between the event of May 11th and May 12th, the site conditions were altered for the May 12th event. For example, depression storage would have been occupied with trapped runoff from the May 11th event. In addition, areas where infiltration would normally occur may be saturated by the runoff from the May 11th event. Thus, during the May 12th event, the runoff coefficient was much higher, as the runoff was transported off the site at a much faster rate.

5.1.3 Lag Times

The lag times were calculated for the catchment and the pond. According to SWAMP (Appendix B 2002), the catchment lag time is the time delay between the start of rainfall, to the start of the inlet runoff or hydrograph. For the Richmond Hill study, there are two inlets servicing two catchment areas. Thus, the catchment lag times are presented in Table 20. In addition, the average catchment lag times are also included. The catchment lag times were calculated by determining the difference between the centroid of the hyetograph and the centroid of the inlet hydrograph. The inlet that received the initial runoff from the event was used to determine the hydrograph centroid.

Table 20, demonstrates some variability of the catchment lag times for each event. One rationalization of this variability includes the condition of the catchment area before each event. For example, the site received a minimal amount of rainfall (<3mm) shortly before the events of September 20th, and October 2nd. The September 20th event received 3mm of rain that was dispersed between 1 am and 5 am, more than 12 hours before the event occurred. It is possible that this rainfall, had infiltrated into the ground making it somewhat saturated. Because of this partial saturation, the rainfall from the following event became runoff at a faster rate than if the ground was completely dry. This may explain why these two events experienced the shortest lag times.

The average catchment lag times at inlet 510 are slightly longer than inlet 1070. One explanation for this can be attributed to the amount of sodded lots. The catchment area serviced by Inlet 510 was fully sodded by September, 2002. If it is assumed that the sodded areas absorb more runoff, allowing it to infiltrate into the soil then Inlet 510 will indeed experience longer lag times. This is also based on the assumption that the catchment area serviced by Inlet 1070 has largely compacted soils from construction equipment. However, these circumstances do not provide an adequate explanation for the events where Inlet 510 experienced shorter lag times than that of Inlet 1070. Unfortunately, no pattern or generalization can be made with the catchment lag times calculated for this study. There are many factors that can account for the different lag times between the two catchments. These include depression storage, impervious cover, and rainfall dispersion.

Table 20: Catchment Lag Times at Richmond Hill study site

Event:	Catchment Lag Time (hrs)
14-Sep-02	1.79
20-Sep-02	3.04
27-Sep-02	3.48
02-Oct-02	0.98
19-Oct-02	0.25
25-Oct-02	1.39
02-May-03	3.02
05-May-03	1.72
11-May-03	3.70
12-May-03	4.09
20-May-03	1.32
Average:	**2.25**

5.1.4 Hydraulic Detention Times and Drawdown Times

Detention time is a function of the water volume in the pond, and the outflow rate. Longer detention times are required for greater pollution removal efficiencies (Waneilista & Youseff 1993). It is suggested that a detention time of 24 hr or more can achieve a 90% or more removal efficiency of suspended solids (Wanielista & Youseff 1993). However, according to Wanielista and Youseff (1993), a study using model detention ponds concluded that a minimum detention time of at least 72 hr is needed to remove more than 95% of the suspended solids and 30 to 70% of nutrients and metals. The latter pollutants are typically found in dissolved form. This finding creates doubt for the current design criteria that requires a 24 hr detention time, or a 48 hr detention time that is commonly used in practice (Sabourin, Kimble & Associates Ltd. 2000). When designing stormwater wet ponds or construction sediment ponds, several inferences are made. These inferences are used to estimate the drawdown time or detention time of the pond. For example, the MOEE SWMPP uses the *Falling Head Drawdown Equation* (Appendix E 1994). This equation assumes a constant pond surface area. Recognizing this may not be an accurate assumption, the Guidelines recommend making the calculation using a linear regression relationship derived between the pond's surface area and wet pond depth (MOE 1994). Moreover, the detention times are calculated based on an estimated peak discharge. The peak discharge is determined through the analysis of a design storm. The Richmond Hill pond used several design storms, 2 through 100 year post to pre-development. However, this technique can be limited, as it cannot account for every type of storm and subsequent peak discharges. This section examines both the hydraulic detention times and drawdown times of the Richmond Hill pond.

The detention times were calculated based on the difference between the centroids of both the inlet hydrographs (combined) and the outlet hydrograph. Table 21 includes the hydraulic detention times. The average hydraulic detention time was 9.79 hrs. This is low in comparison to the expected detention time of 48hrs intended by the design of the pond. This low value may be attributed to the methods used to calculate detention time. For example, when there is back-to-back events it is difficult to determine when the outlet hydrograph ends for one event and begins for another. If the outlet hydrograph is cut off too early the detention time extends for a short period of time. Thus, this may account for the low detention times as the events of September 20th, May 5th, and May 11th were followed immediately by another event. This issue is examined further in the discussion section of this paper.

The drawdown times were calculated using two separate methods. First, the actual drawdown time was calculated by determining the time delay between the maximum elevations of the pond during the event to the minimum elevation at the end of the event. In addition, the average drawdown times were calculated by dividing the pond volume at the average water level by the average outflow from each event. Table 22 presents both the actual and average drawdown times for the fall sampling period. The drawdown times for the spring sampling period are pending.

Table 21: Hydraulic detention times for events monitored at Richmond Hill study site

Event	Detention Time (hrs)
14-Sep-02	11.42
20-Sep-02	10.13
27-Sep-02	13.42
2-Oct-02	8.09
19-Oct-02	9.86
25-Oct-02	15.43
2-May-03	8.99
5-May-03	4.41
11-May-03	5.66
12-May-03	9.15
20-May-03	10.15
Average:	9.7

Table 22: Actual and average drawdown times for events monitored at Richmond Hill study site

Event	Actual Drawdown Time (hrs)	Average Drawdown Time (hrs):
14-Sep-02	41	28.38
20-Sep-02	38	13.77
27-Sep-02	38	14.78
2-Oct-02	27	11.91
19-Oct-02	18	10.99
25-Oct-02	23	11.35

The actual drawdown times are closer to the design criteria than the detention times and the average drawdown times. However, all the estimated actual drawdown times are under the 48 hr detention time design criteria of the pond. Furthermore, Papa *et al* (1999) maintain that the actual drawdown time overestimates a pond's ability to detain runoff from an event.

5.2 Water Quality Analysis

The water quality analysis includes results from both discrete and composite samples. The first section includes the discrete sample results; the remaining sections discuss the results from the composite samples.

5.2.1 Discrete Analysis

Discrete samples were analyzed for suspended solids. These samples were collected using two different time intervals. For the fall season, 24 bottles were collected every 5 minutes, totalling 120 minutes. Subsequent to the analysis of the fall events, it was determined the sampling period could be extending to collect every 15 minutes for a total of 360 minutes. This was to broaden the representation of the pollutant load for an event through water quality samples. From these samples, pollutographs were generated for suspended solids. Figures 14 and 15 depict pollutographs from two events selected from the fall and spring sampling periods; October 2, 2002 and May 20, 2003. These two events were selected as they both share similar rainfall depths but demonstrate the differences in sediment loads from both the fall and spring sampling periods. Although several factors can influence the amount of sediment exported from a catchment area by a rainfall event, rainfall depth is considered a major influence (Schueller and Lugbill, 1990).

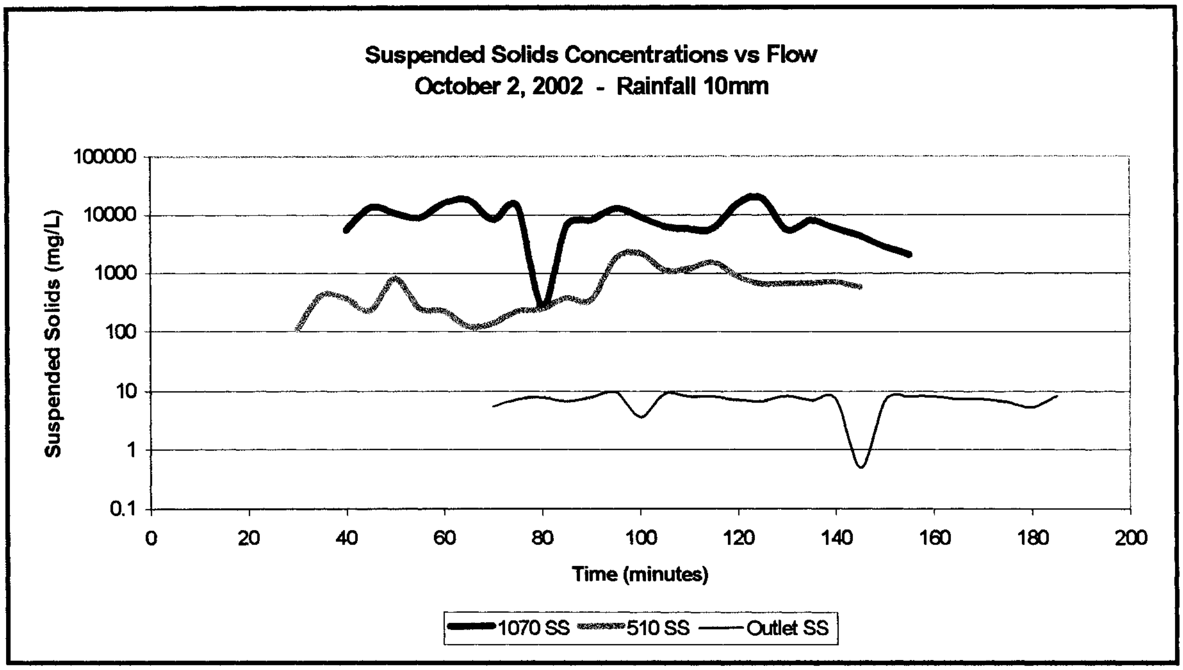


Figure 14: Suspended solids concentrations for event of October 2, 2002

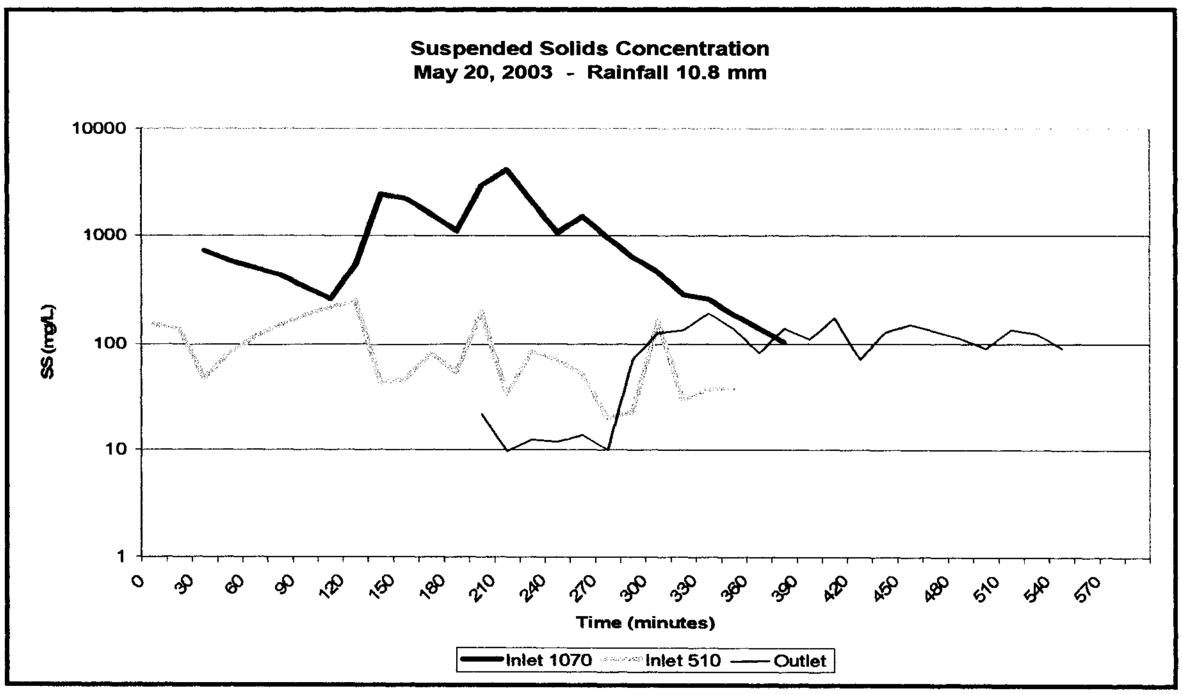


Figure 15: Suspended Solids Concentrations for event of May 20, 2003

Figures 14 and 15 present suspended solid concentrations from two different events sampled in two different seasons, fall and spring. The October 2nd event experienced a higher sediment load than the event on May 20th. Inlet 1070 on October 2nd reached over 10 000 mg/L, while Inlet 510 reached approximately 1000mg/L. It is also evident that the outlet produced a significantly reduced concentration of suspended solids exiting the pond (reaching as high as 10 mg/L). Although both events received similar rainfall volumes (approximately 10mm), the May 20th event did not produce as much sediment, yet produced a significantly higher concentration exiting the pond. On May 20th, Inlet 1070 suspended solid concentrations reached half of that experienced on October 2nd (approximately 5000 mg/L). This may be attributed to the progression in construction activities where additional lots were sodded by the spring sampling period. Markedly, the outlet is experiencing higher suspended solid concentrations exiting the pond, indicating poor performance during the May 20th event. A possible factor to consider is the difference in the sampling start times between the two events. For the October 2nd event, the outlet sampler was triggered approximately 30 minutes after the inlet samplers were triggered. Alternatively, for the May 20th event, the outlet sampler was triggered approximately 160 minutes after the inlet samplers were triggered. The samples are set to trigger on an increase in level. If the trigger level was misestimated, or if there was a surge in the outflow that momentarily increased the level, the outlet samples are collected prematurely. Thus, during the October 2nd event the outlet water quality sampler may have been triggered too early, missing the peak concentrations exiting the pond. Ideally, the outlet should trigger when the fluid from the event has begun to exit the pond through the outlet structure. Initially, the runoff from an event enters the pond and displaces the fluid that was already present from the previous event. Eventually the runoff from the current event begins mixing with the pond's initial volume of water and this mixture containing the now diluted contaminants from the current storm begins to exit the pond. The time required for this to occur can be determined using a tracer study. A tracer study estimates the pond lag time, which is the time required for the runoff from an event to enter and exit the pond. Due to limited time and resources, a tracer study was not conducted at the Richmond Hill pond.

There are several factors that can influence the suspended solid load from an event. One factor to consider is the use of bulkheads at construction sediment ponds. Generally, bulkheads are instituted to limit the discharge of sediment to receiving water bodies. At the Richmond Hill pond, the bulkheads were established within the sewer maintenance chambers at both inlet 1070 and 510. The water quality samples were collected downstream from the bulkhead. The bulkhead was accumulating sediment throughout the course of the study. Thus, during an event, the runoff would enter the sewer drainage system and be carried through inlet 1070 and inlet 510. The runoff would first pass over the bulkhead, with a portion of the sediment accumulating behind the bulkhead. At the end of the fall sampling period visual observations revealed that a significant amount of sediment had accumulated, almost reaching the top of the bulkhead. Therefore, it is possible that as the sediment accumulates behind the bulkhead, the incoming runoff from a new event may stir up the accumulated sediment and carry it into the pond. The level of influence the bulkheads had on the water quality samples was not determined. However, it is important to consider this as a factor that can influence the water quality results.

Other factors that can influence the water quality results include rainfall depth, intensity, duration, and inter-event dry periods. The level of influence each of these factors has on the suspended solid load is difficult to ascertain. However, *Schueller and Lugbill* concluded in the Maryland study that rainfall depth significantly influenced sediment load. For example, storms that generated less than a half inch (12.7mm) of rainfall produced a median TSS concentration of 459 mg/L (1990). In addition, the study found that storms generating between 0.5-0.99 (12.7 – 25 mm) produced a median TSS concentration of 748 mg/L. Finally, the study concluded that storms generating more than 25mm (>1 inch) produced a median TSS concentration of 1372 mg/L (Schueller & Lugbill 1990). This pattern can also be applied to the results found in this study. Table 23 compares rainfall depth with the median concentrations found in this study for each event. Rainfall intensity and duration are also included to provide further insight on the characteristics of each event. The median concentrations are the sum of both inlet 1070 and inlet 510 median concentrations.

Generally, the rainfall depth may influence the suspended solids concentrations. For the larger storms, including September 14th, 20th, and 27th, the median SS concentrations were over 2000mg/L. Alternatively, for events generating rainfall under 10mm the median SS concentration was under 700mg/L. However, there are two events that showed some deviation from this trend. These include September 27th where the rainfall was 18mm, but generated the second highest median SS concentration (over 7000mg/L), and October 2nd when the rainfall depth was only 10mm but generated the highest median SS concentration (over 8000 mg/L).

Several factors may explain the October 2nd and September 27th suspended solids results. These include the timing of sample collection, the intensity and duration of the event, or inter-event dry periods. For example, the October 2nd event was the longest in duration (over 30hrs). In addition, the September 27th event generated one of the highest intensities at just over 2mm/hr. These factors along with the condition of the construction site, and inter-event periods may have influenced the suspended solid loads for these events. The relationship between rainfall depth and suspended sediment median concentrations is presented in Figure 16. Essentially, Figure 16 is demonstrating that there is a general trend in the relationship between the runoff suspended solids load and the rainfall depth of an event. Generally, as the rainfall depth increases the suspended solids concentrations will also increase.

Table 23: Rainfall depth compared to median suspended solids concentrations for events monitored at the Richmond Hill study site

Event	Rainfall Duration (hr)	Average intensity (mm/hr)	Rainfall Depth (mm)	Median SS Concentration (mg/L)	Inter-event Periods (hrs)
14-Sep-02	25	1.18	29.6	3602.05	576
20-Sep-02	24	0.86	20.6	2295.75	99
27-Sep-02	8.62	2.09	18	7748.8	109
2-Oct-02	30.08	0.33	9.8	8499.5	102
19-Oct-02	11.38	1.14	13	1278.75	27
25-Oct-02	7	1.34	9.4	1295.05	70
2-May-03	8	0.85	6.8	406.7	15
5-May-03	5.5	1.67	9	313.4	34
20-May-03	5.17	2.09	10.8	682.5	86

*May 11th event is not included; Inlet 510 did not collect. May 12th inlet 510 results were not received in time for this paper.

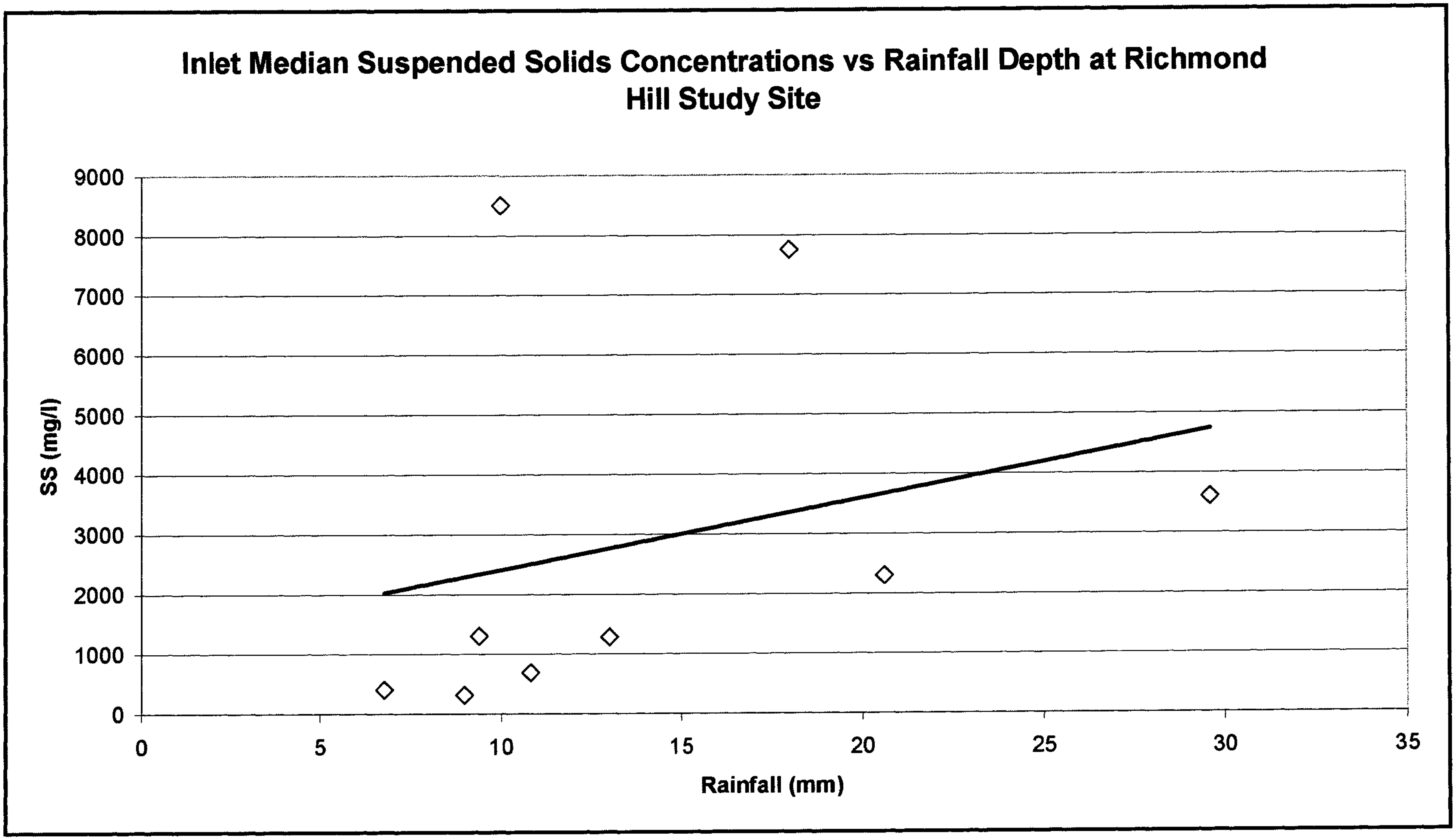


Figure 16: Relationship between inlet median suspended solids concentrations and rainfall depth

The EMC values were calculated using the results collected from the discrete samples. These are presented in Table 24.

It should be noted that Inlet 510 did not trigger for the May 11th event. Thus, the Inlet 510 EMC value from May 20th substituted the missing data as this event also produced similar rainfall characteristics. This event was chosen as it produced similar rainfall characteristics (i.e. intensity, duration, and total volume). In addition, the October 25th event experienced flow measurement error at inlet 1070. During the event, the area velocity sensor recorded zero flow at the time the samples were collected. Thus, the EMC could not be calculated without flow data. Therefore, data from the event of October 2nd was used as it produced similar suspended solids concentrations and generated similar rainfall characteristics. It is recognized that the use of substituted data can affect the outcome of the results. This is particularly true when calculating the concentration-based removal efficiencies. For example, if the substituted data is underestimating the true pollutant load, the concentration-based removal efficiency could be high.

According to the concentration-based removal efficiencies (CE) presented in Table 24, the pond is performing very well. In fact, these concentration-based removal efficiencies are much higher than the trap efficiencies predicted using theoretical methods included in the *Nighman and Harbor* (1995) study outlined in Chapter 2. The theoretical trap efficiencies were ranging from 0 – 82% (refer to Chapter 2, section 2.4.2, Table 9). However, percentage removal efficiencies are not necessarily the most reliable method for determining the pond's performance. Ultimately, the determining factor is the outflow concentrations. Of the Guidelines outlined in Chapter 2 (Table 7), protection requirements are based on increases from background levels. Unfortunately, there are no background levels to compare for this study. However, Table 6, Chapter 2 presents the European Inland Fisheries Advisory Commission's *Guidelines to Protect Aquatic Resources.* These protection guidelines offer concentration thresholds that if exceeded could cause harm to downstream fisheries. A comparison is made between the results found in this study, and the protection guidelines in Figure 17.

Table 24: Event mean concentrations and the concentration based removal efficiencies for events monitored

Date:	EMC In (mg/L)	EMC Out (mg/L)	CE(%)
14-Sep-02	2761.85	276.61	90%
20-Sep-02	8484.90	26.51	≈100%
27-Sep-02	2591.17	74.88	97%
2-Oct-02	2797.28	7.24	≈100%
19-Oct-02	657.89	28.51	96%
02-May-03	279.81	37.52	87%
05-May-03	753.68	35.18	95%
11-May-03	1613.98	223.54	86%
20-May-03	863.84	100.41	88%
02-May-03	279.81	37.52	87%

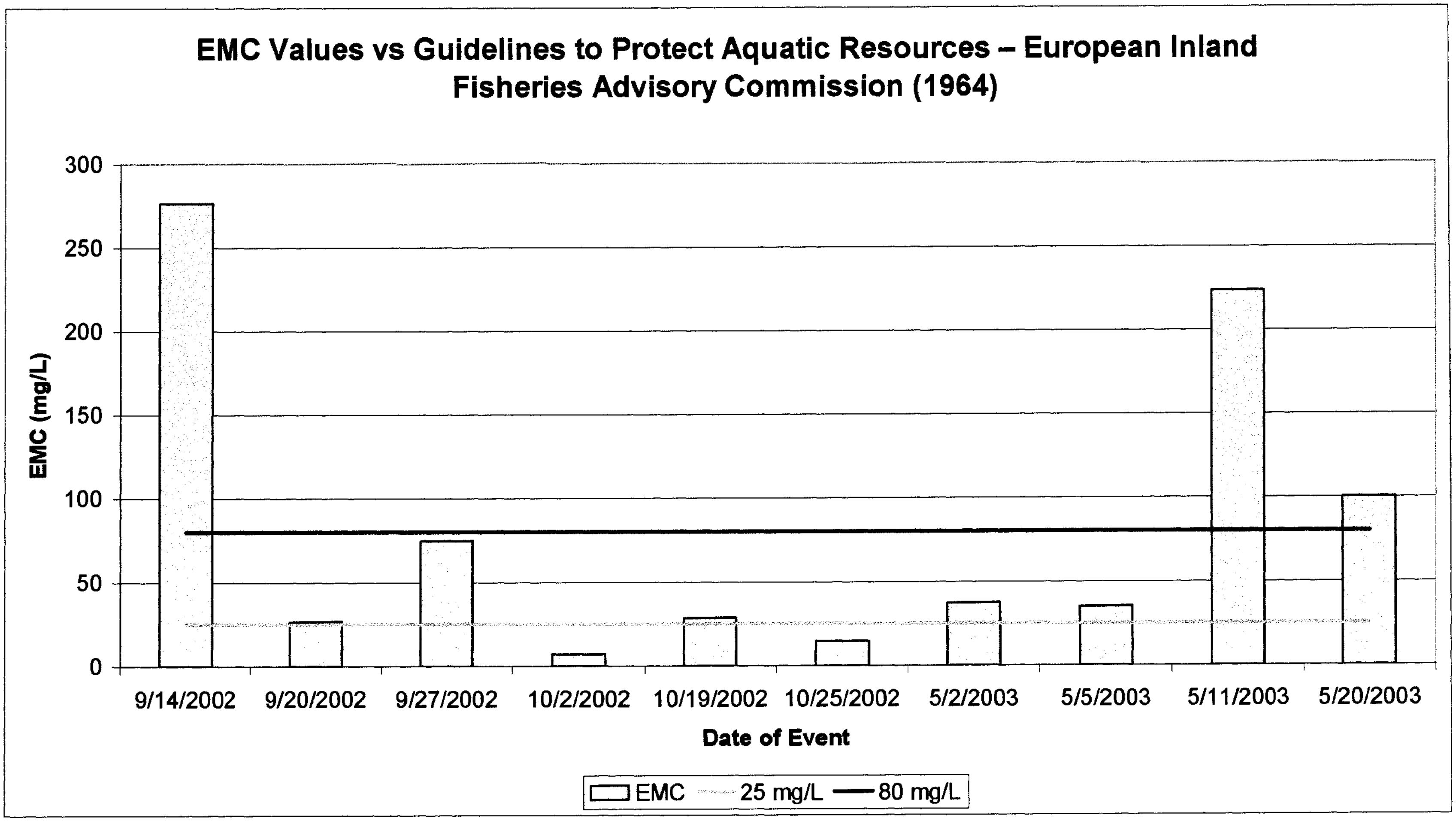


Figure 17: EMC concentrations compared to recommended guidelines in Europe

Figure 17 shows the EMC values of each event compared to two different threshold levels recommended by the European Inland Fisheries Advisory Commission. The two threshold levels used are the 25mg/L and 80 mg/L. The September 14th event shows an EMC value exiting the pond at over 270 mg/L. This is significantly high. When compared to the European Inland Fisheries Protection criteria, this value is well above the 80mg/L threshold shown in Figure 16, which falls within the "Unable to support freshwater fisheries" range (80 – 400 mg/L). This event was the most intense event in the fall season, and received the largest amount of rainfall (29.6mm). Moreover, only two of the ten events produced EMC values that were under the 25 mg/L threshold, as shown in Figure 16. According to the European Inland Fisheries Advisory Commission (1964), if concentrations remain above this threshold it is "possible to maintain good to moderate fisheries, however the yield would be somewhat diminished relative to waters with < 25 mg/L "(Clarifica 2001). Only the events of October 2nd, and 25th produced suspended solid concentrations below 25mg/L. The Advisory Commission considers this level of concentration to have "no evidence of harmful effects on fish and fisheries" (Clarifica 2001).

The majority of events collected had EMC values under 80 mg/L, and all had high removal efficiencies. It is important to consider why the pond was unable to reduce the concentration of suspended solids to adequate levels for protecting downstream habitat. The events of September 14th, and May 11th are of particular interest, as the EMC values were both above 270 mg/L. Indeed, this high level of concentration would be detrimental if the facility was located upstream from coldwater fisheries. Several factors can influence the pond's ability to reduce concentrations including rainfall volume and intensity, or inter-event dry periods. The September 14th event extended over a long period of time (approximately 25 hrs). In addition, the event generated a significant amount of rainfall: 29.6mm. This event also had an average intensity of 1.18mm/hr as it extended over 25 hr. The May 11th storm did not generate as much rainfall volume as the September 14th event; a total of 14.2 mm was generated. However, the storm only lasted for approximately 6.5 hrs, with an intensity of 2.18 mm/hr. Although the total rainfall volume was considerably less than the September 14th event, the high intensity may account for the significantly high concentrations of suspended solids exiting the pond.

The EMC values presented in Table 24 are suggesting that the pond is unable reduce suspended solids concentrations to safe levels under certain conditions. However, it is important to note the limitations of these results due to the methods applied during the monitoring program. For example, the samples collected only represent partial events. For the fall season, samples were collected at five minute intervals. This enables the samples to represent 120 minutes of each event during the fall season. During the spring events, the sample collection increased to intervals of fifteen minutes. In addition, the samplers were not always triggered at an ideal time (i.e. at the beginning of the event). Thus, on some occasions the samplers were only able to collect at the end of an event, or sometimes event triggered prematurely. To illustrate how these issues may affect the results Table 25 describes event times and when the samplers were triggered.

Table 25: Event start times, duration, and sampler start times

Event	Event Start Time	Event Duration (hrs) *Based on length of hydrograph*	Sampler Start Time		
			Inlet 1070	**Inlet 510**	**Outlet**
14-Sep-02	22:25	25	22:35	23:35	22:50
20-Sep-02	20:47	31	21:42	23:35	1:37
27-Sep-02	8:50	13	11:10	12:25	11:30
02-Oct-02	18:30	14	18:45	18:35	19:15
19-Oct-02	0:35	19	0:35	4:40	5:15
25-Oct-02	21:35	17	17:42	0:36	0:27
02-May-03	0:15	19	1:45	1:45	4:10
05-May-03	15:20	7	16:20	15:35	17:50
11-May-03	7:20	8	9:30	-	11:30
12-May-03	5:35	24	11:35	-	16:35
20-May-03	12:50	15	13:35	13:05	16:20

Table 25 demonstrates the circumstances in which the samples were collected. For example, during the event of September 14th, the outlet sampler triggered 45 minutes before the inlet 510 sampler was triggered. Because the samples collected are only partial representations (extending 120 minutes over the hydrograph for the fall collection period), the portion of samples collected at the outlet may be missing some of the pollutants entering the pond through inlet 510. Ideally, the outlet should be triggered after both inlet samplers are triggered. This also occurs on September 27th. These issues are further examined in the discussion section of this thesis.

5.2.2 Composite Analysis

The composite samples are a mixture of each discrete sample. For example, of the 24 bottles collected, a portion was extracted and mixed into one bottle. The composite bottle was then divided into several bottles for the analysis of water quality parameters outlined in Table 12 (refer to Section 4.4.2, Chapter 4). It should be noted that all samples analyzed for PAHs, herbicides and pesticides produced 'no measurable response' according to the MOE laboratory services. Because these parameters were not detected in large enough quantities in the samples, these results are not included.

The total performance and removal efficiencies were calculated for the fall sampling period. The spring sampling results from the composite samples are still pending. During the fall sampling period, the pond was able to significantly reduce the mass load of pollutants entering the facility. Yet, some pollutants were able to escape in larger quantities, producing lower removal efficiencies. In fact, some pollutants seemed to have experienced negative removal efficiencies.

Figure 16 shows how the removal efficiencies for suspended solids are quite high, almost reaching 100%. Yet, dissolved solids are much lower, with a removal efficiency of less than 60%. In addition, nutrients appear to be successfully trapped within the basin. However, many of the heavy metals are experiencing lower removal efficiencies. In particular, iron, titanium, and cadmium experienced negative removal efficiencies. This suggests there is an internal source within the pond (i.e. bottom material), or from the eroded banks along the pond's edge. Another possible explanation for the negative removal efficiencies includes the gradual accumulation of pollutants when there is no outflow. For example, residents that received new sod on their lots were encouraged to water their lawns frequently. This may have produced enough runoff to enter the pond, while carrying some pollutants from the site. These pollutants can gradually build up so that during the next event a higher concentration may exit the pond than what initially entered the pond for that event.

As with the discrete analysis, the removal efficiencies for the composite results are not the determining factor of the pond's performance. The actual concentrations of the pollutants exiting the pond are considered to be the ultimate indication of the pond's ability to protect downstream habitat. For example, the results of this study are indicating that high removal efficiencies do not determine safe levels of pollutant concentrations. Appendix D is a list of the outlet concentrations for every event monitored compared with the removal efficiencies and the PWQOs. Figure 18 demonstrates that some of the pollutants experienced high removal efficiencies during the fall season. This includes Total Phosphorus, which demonstrated a removal efficiency of over 90%. However, phosphorus concentrations exiting the pond were above the recommended 0.03 mg/L for most of the fall events. This also occurred for other pollutants including cobalt, copper, and iron. In addition, Figure 19 demonstrates that many of the pollutants are still experiencing high removal efficiencies. However, the removal efficiencies calculated for the spring monitoring season were not as high as the fall season. For example, suspended solids did not reach the 80% removal efficiency. In fact, based on the spring results the removal efficiencies did not meet the 1994 MOE guidelines requirement. This requirement is 80% removal efficiency for TSS.

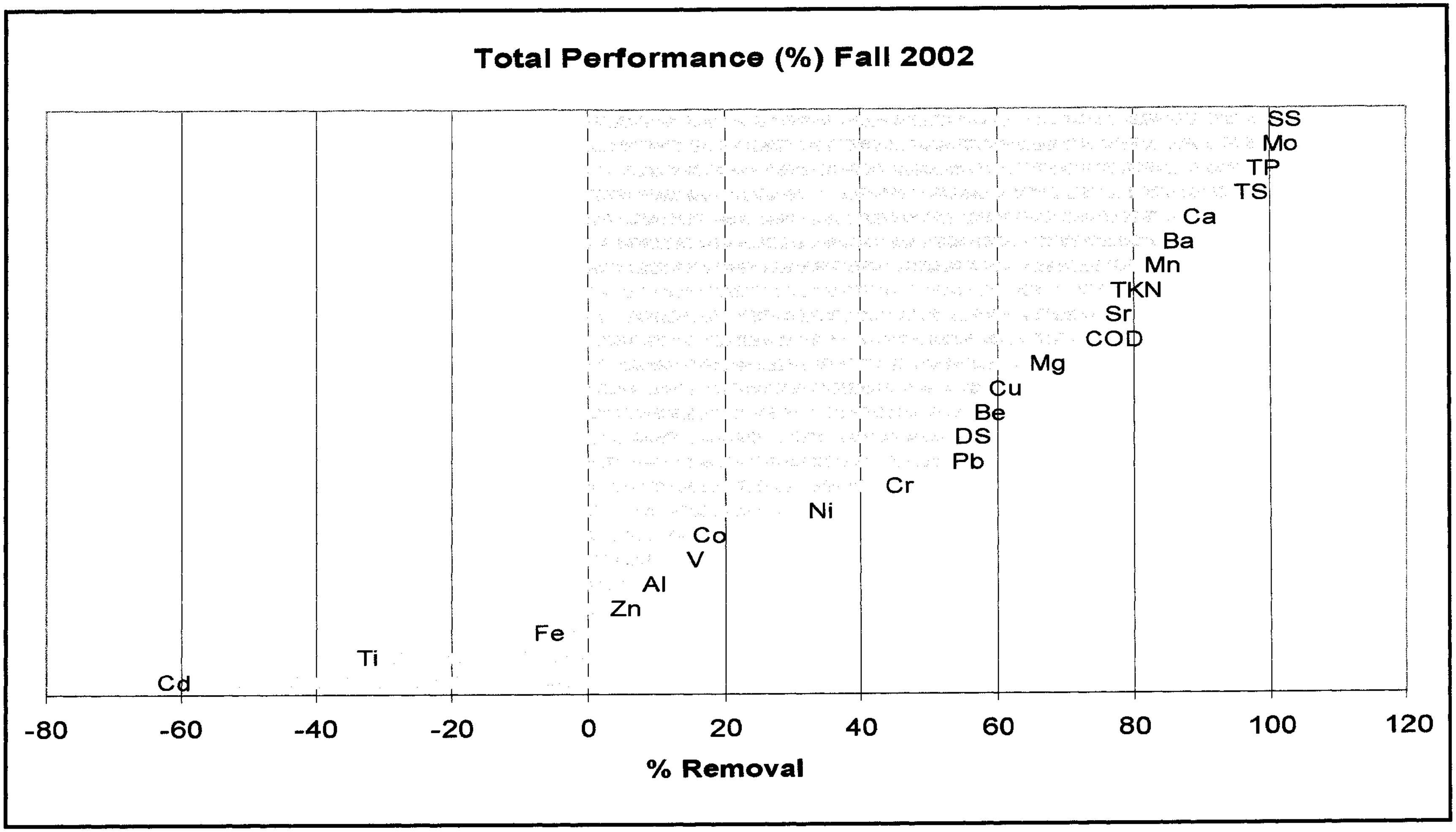


Figure 18: Total Performance (%) – Fall 2002

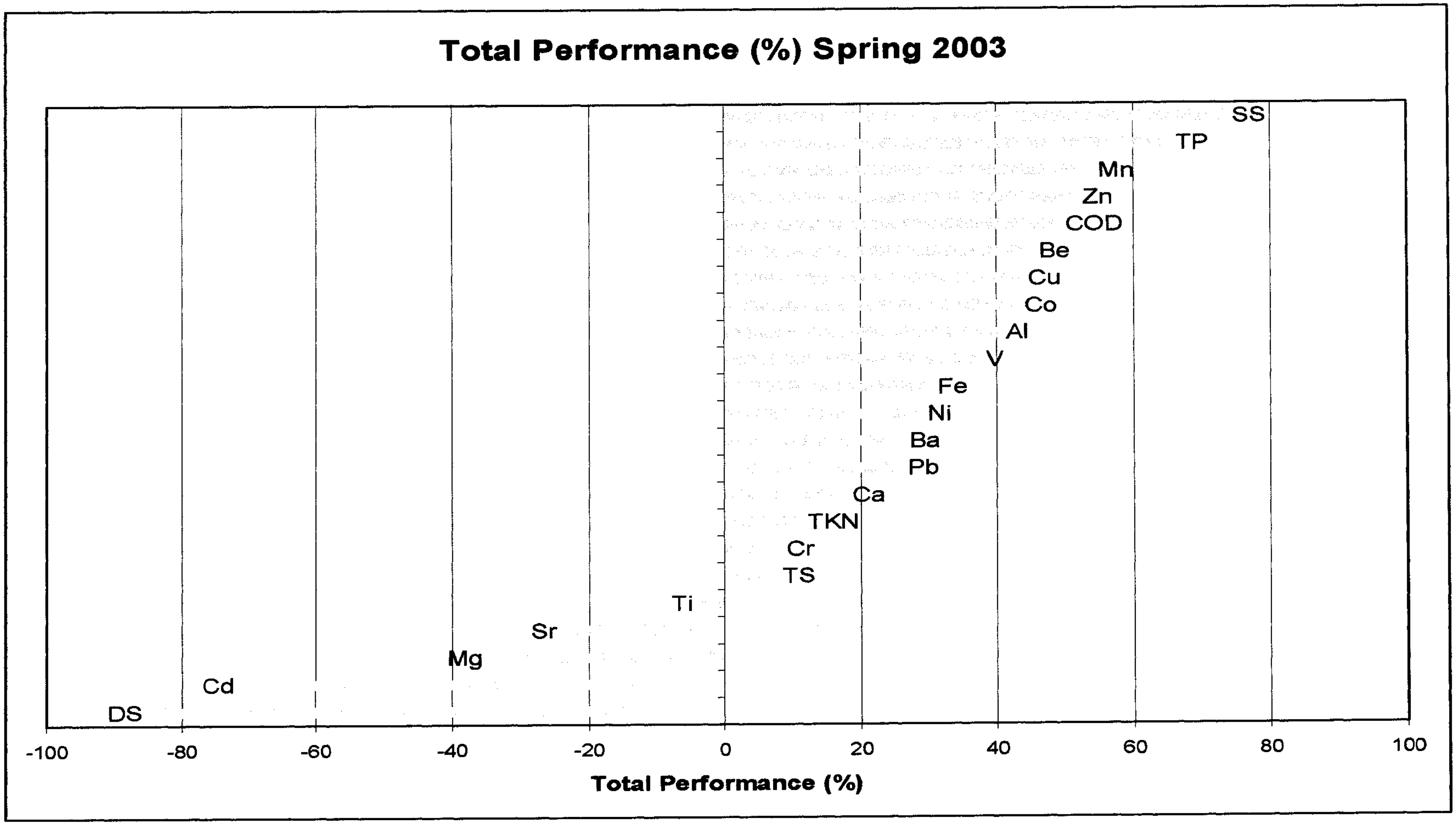


Figure 19: Total Performance (%)

The following sections examine each of the water quality parameters monitored as composite samples during the study. The results of the water quality parameters are grouped into the following: general chemistry, nutrients, and metals.

5.2.2.1 General Water Chemistry

The general water chemistry was analyzed and includes the following.

- Chemical Oxygen Demand (COD)
- Suspended Solids
- Total Solids
- Dissolved Solids
- Conductivity
- pH
- Alkalinity
- Turbidity
- Chromium
- Dissolved Organic Carbon
- Dissolve Inorganic Carbon
- Silicon

Figures 20 and 21 present the mean values, maximum and minimum concentrations found at the outlet during the fall 2002 and spring 2003 sampling periods. In addition, the subsequent PWQOs for each of the parameters analyzed are also included in Figures 20 and 21. The mean, maximum and minimum values for the inlets are included in Appendix E.

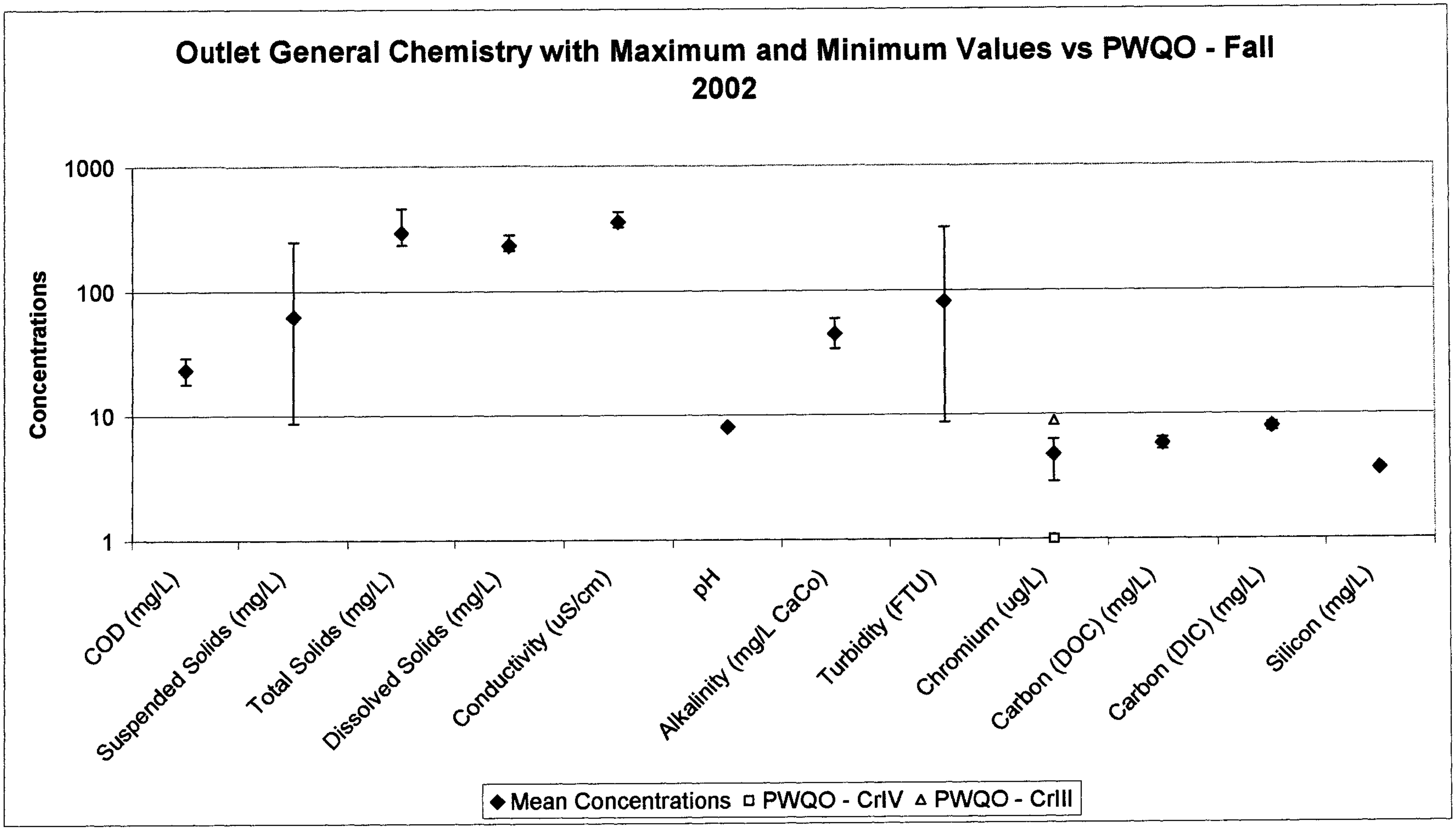
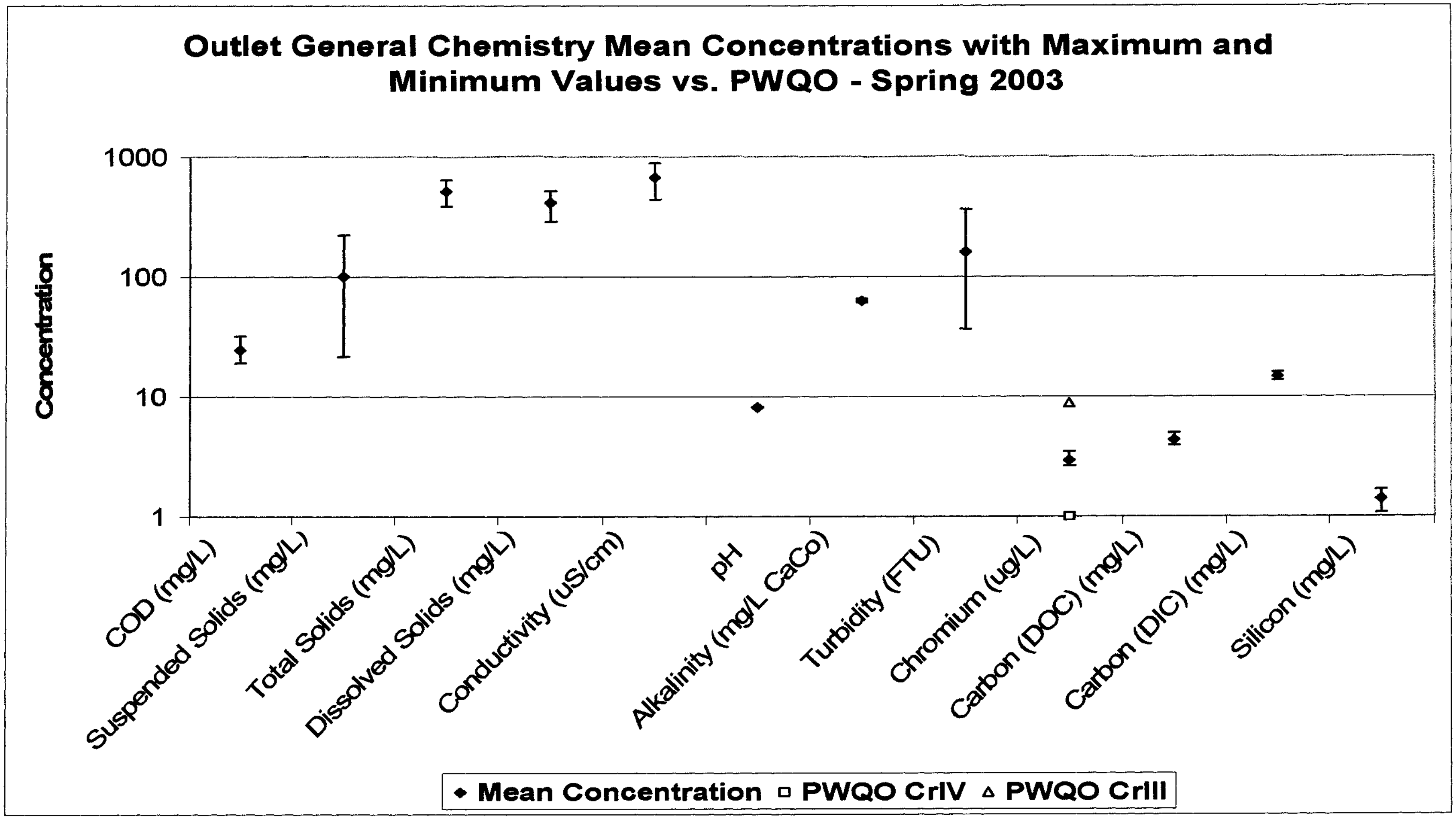


Figure 20: Outlet general chemistry with mean, maximum and minimum values vs. PWQO – 2002



Figures 21: Outlet general chemistry with mean, maximum and minimum concentrations vs. PWQO – Spring 2003

With the exception of chromium and pH, there are no PWQOs for the parameters included in the general chemistry analysis. For hexavalent chromium (CrVI), the PWQO is 1 μg/L, for trivalent chromium (CrIII) the PWQO is 8.9 μg/L (MOE 1999). This is included in the above figures. The pH range recommended by the PWQO is between 6.5 and 8.5. This is not indicated in the above figures, but is included in the following analysis.

Appendix E includes the inlet mean, maximum and minimum values compared to their assigned PWQO values. At inlet 1070, most of the parameters varied little in their concentrations among the six events sampled during the fall season. However, turbidity, chromium, suspended solids and total solids all experience some variability with a significantly low minimum concentration. Generally, inlet 510 also shows little variability with suspended solids and turbidity experiencing a wider range in concentration. The outlet concentrations, including suspended and total solids and turbidity, show a high maximum concentration when compared to the mean. This suggests some extreme events when the concentrations were quite high exiting the pond. In fact, for the event of September 14th, the average outlet concentration of suspended solids was over 270 mg/L. This concentration is well over the European Commission Advisory's recommendation of below 25 mg/L to promote healthy fisheries. Evidently, this concentration affected the results presented for total solids and turbidity. Figure 22 compares all of the suspended solid average composite concentrations collected during the fall sampling period with the European Guidelines.

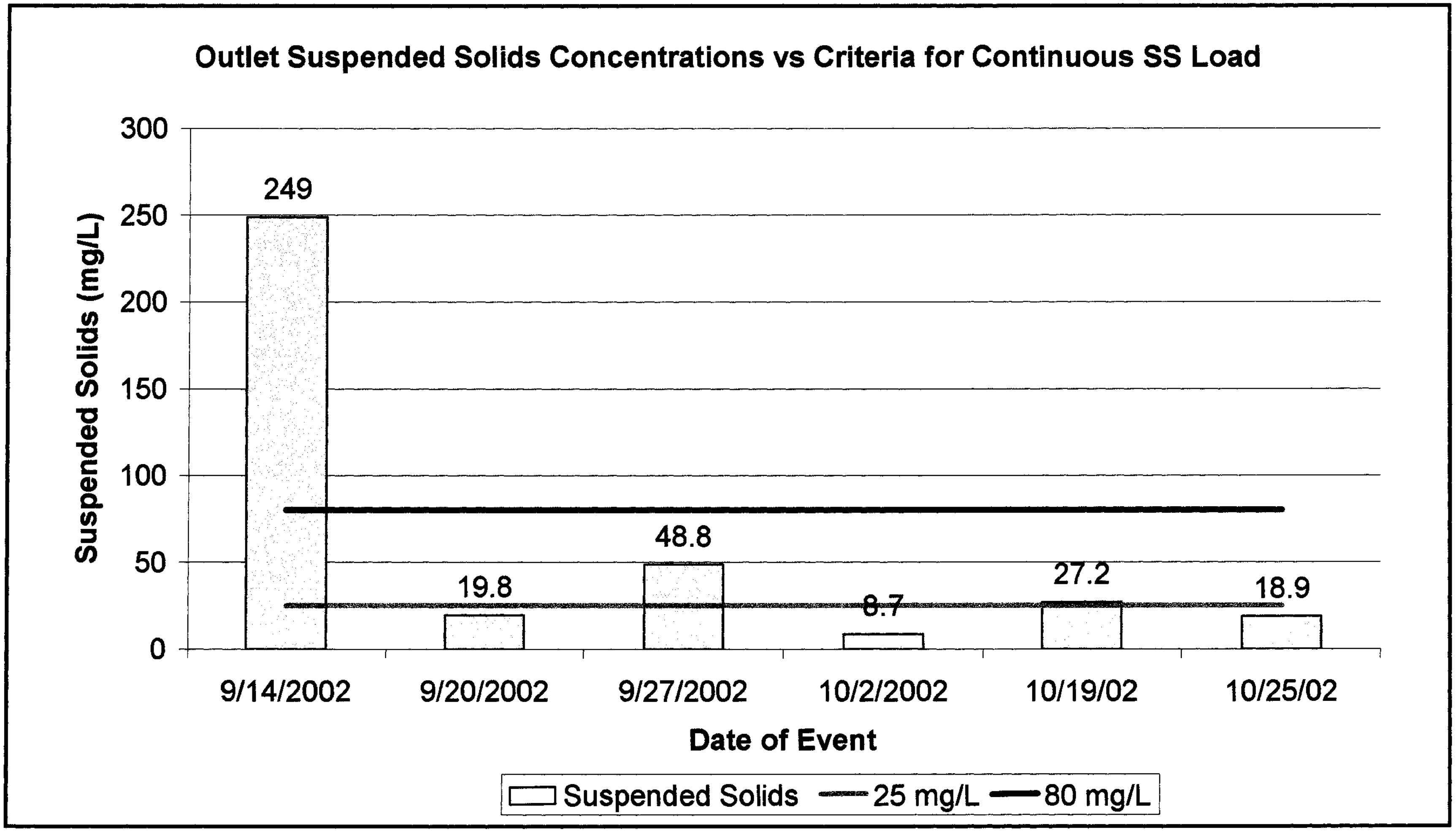


Figure 22: Outlet suspended solids concentrations compared to recommended guidelines

Figure 22 shows that three of the six events were above the 25 mg/L threshold, and two of the six events were above the 80 mg/L threshold recommended by the European Advisory Commission. This shows that the outflow coming from the pond could have some deleterious effects if the downstream habitat was coldwater fisheries. For Lake Wilcox, the introduction of additional sediments to its already altered and oxygen-depleted bottom may slow down the community's remediation efforts. However, for three of the six events the pond performed well, and ultimately reduced the concentrations below the 25mg/L threshold.

Parameters that experienced little variability in concentrations for all monitoring stations during the fall sampling period include COD, conductivity, pH, alkalinity, DIC, DOC, and silicon. The chemical oxygen demand (COD) is a measure of the total quantity of oxygen required to oxidize all organic material into carbon dioxide and water (Brown and Caldwell, 2001). The COD measured at the two inlet demonstrated some variability, yet the concentrations at the outlet showed less variation ranging from 18 to 29 mg/L. In addition, the outlet concentration of chromium varies little in comparison to the other parameters measured for all of the fall events, ranging between 2.88 to 6.3 μg/L. Conductivity ranged from 325 to 433 μS/cm at the outlet. This parameter is an indicator of other water quality problems (i.e. indication of concentration of dissolved ions) (Brown and Caldwell, 2001). Furthermore, the pH ranged from 7.89 to 8.84 entering the pond, and 7.7 to 8.3 exiting the pond. As mentioned previously, the PWQO recommends a range from 6.5 to 8.5. The results show that the pond was able to successfully maintain this range at its outflow.

5.2.2.2 Nutrients

Nutrients were analyzed during the fall and spring seasons, and their mean, maximum and minimum concentrations at the outlet are presented in Figures 23 and 24. The inlet concentrations are included in Appendix E.

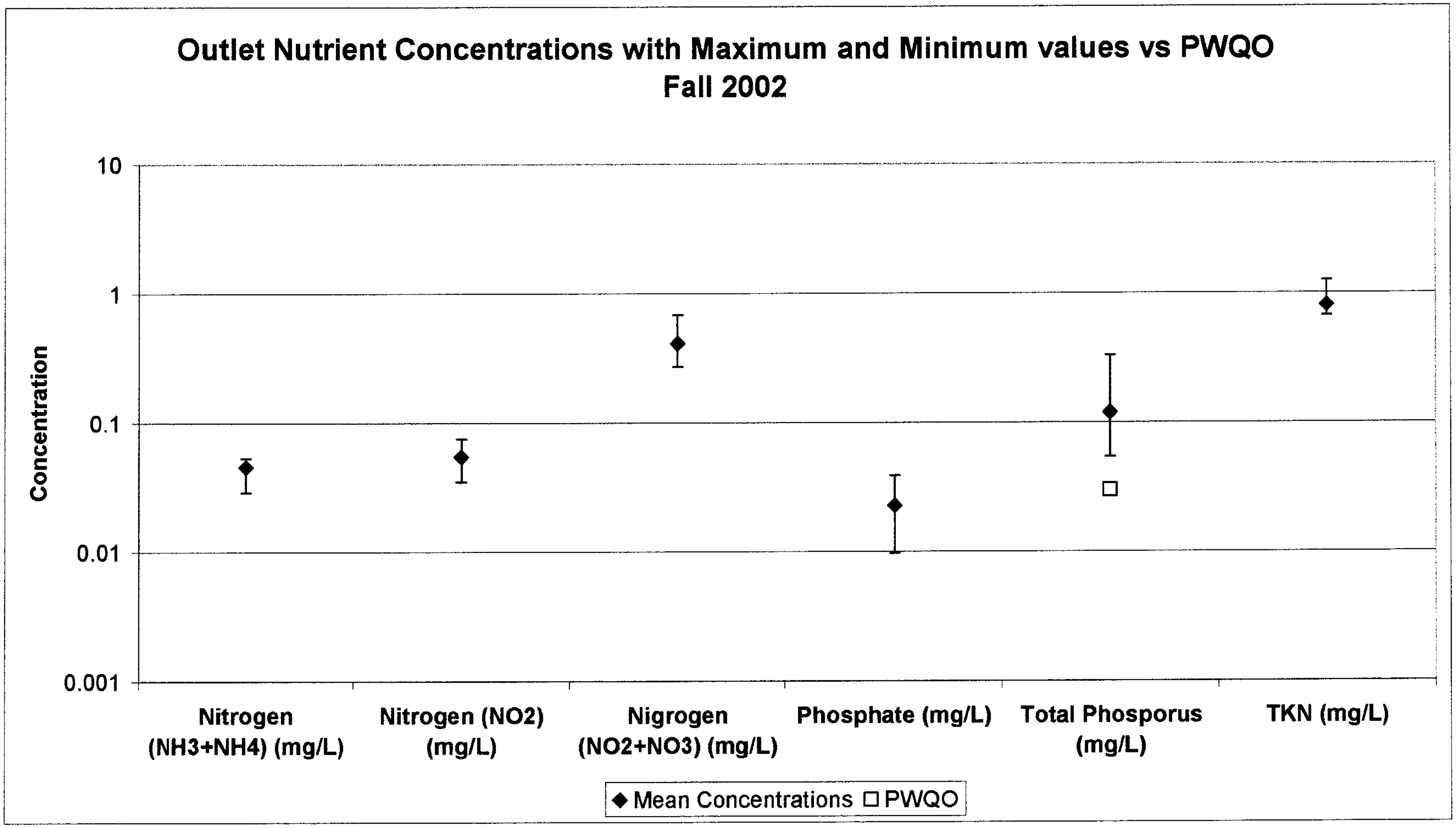


Figure 23: Outlet nut concentrations with mean, maximum, and minimum values vs PWQO – Fall 2002

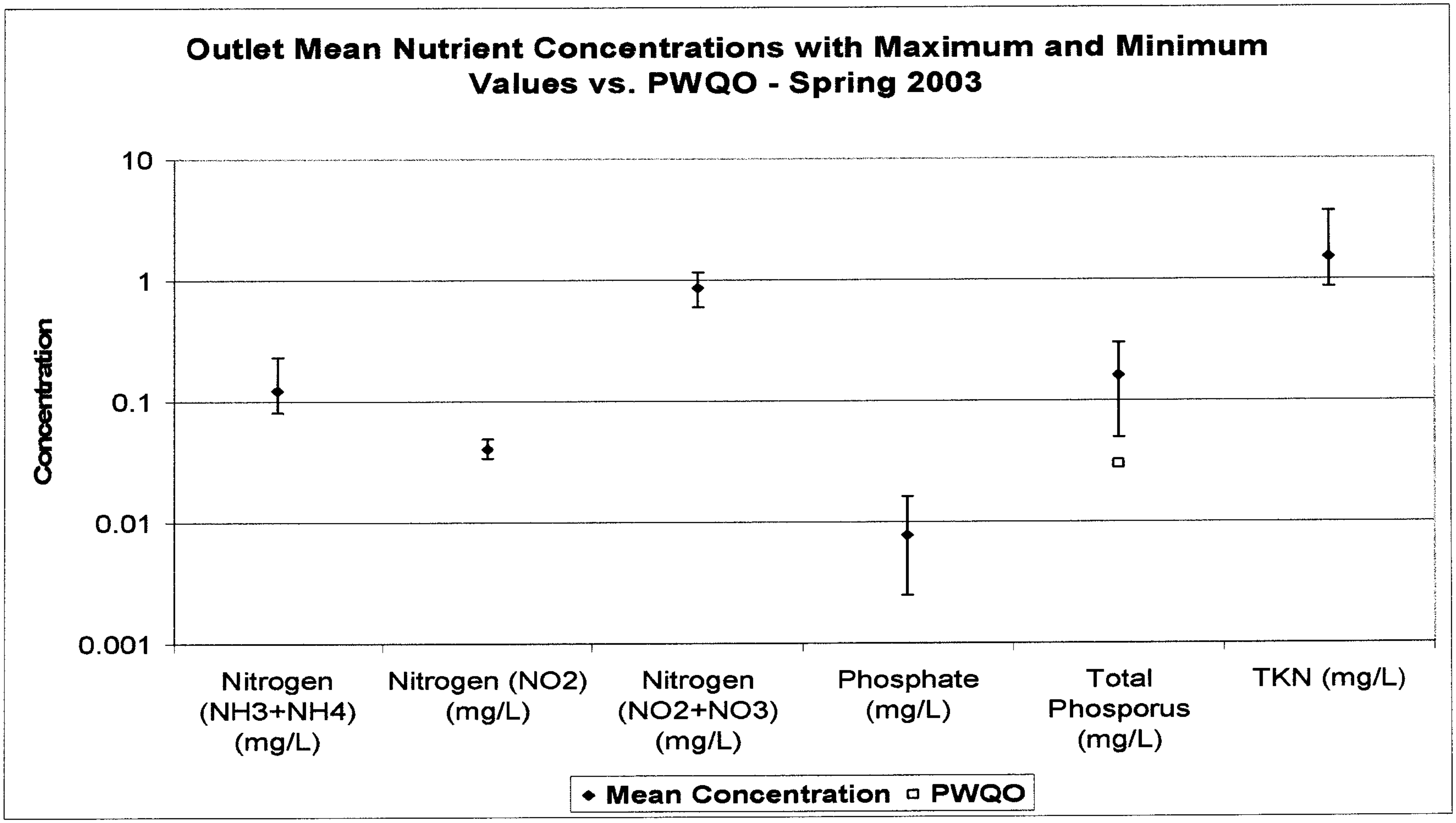


Figure 24: Outlet nutrient concentrations with mean, maximum, and minimum values vs. PWQO – Spring 2003

For the results presented in Figures 23 and 24, it is important to note that phosphate and all nitrogen parameters (excluding TKN) were not analyzed for in the first two events of the fall sampling period. Thus, Total Phosphorus and TKN consisted of a larger sample size. As mentioned previously, the total phosphorus levels were above the PWQO during most events for the fall sampling season. Although it was reduced significantly as seen in the outlet mean concentration the pond still failed to reduce the concentration to a safe threshold. This is important, as Lake Wilcox located downstream is burdened with excessive nutrient loadings. In addition, the dissolved fraction of total phosphorus, analyzed as phosphate, experienced little variation during the fall sampling period. Although this may be attributed to the smaller sampling size, it is indicating consistency in the performance of the pond.

For the fall and spring sampling seasons, linear correlations were derived for the total phosphorus (TP) and TKN data with the suspended solids (SS) and dissolved solids (DS) entering and exiting the pond. The composite sampling results from the spring season are still pending. Table 26 presents the relationship between nutrients and solids.

Table 26: Relationship between nutrient concentration vs. suspended solids and dissolved solids

Correlation Coefficients for Nutrient Concentrations						
	Inlet 1070		**Inlet 510**		**Outlet**	
Nutrient	**SS**	**DS**	**SS**	**DS**	**SS**	**DS**
TP	0.97	-0.27	0.98	0.05	0.97	-0.21
TKN	0.99	-0.66	0.75	0.18	0.29	0.12

Table 26 demonstrates that there is a strong relationship between TKN and suspended solids at the inlets. However, this relationship becomes less significant at the outlet. Alternatively, total phosphorus demonstrates a strong relationship with suspended solids at both inlets and the outlet of the pond. This suggests that the settling of solids will serve as a significant influence on the removal of nutrients from the pond.

5.2.2.3 Metals

A total of 18 different metals were analyzed in this study. Figures 25 and 26 presents the mean, maximum, and minimum concentrations of all metals analyzed for during the fall season.

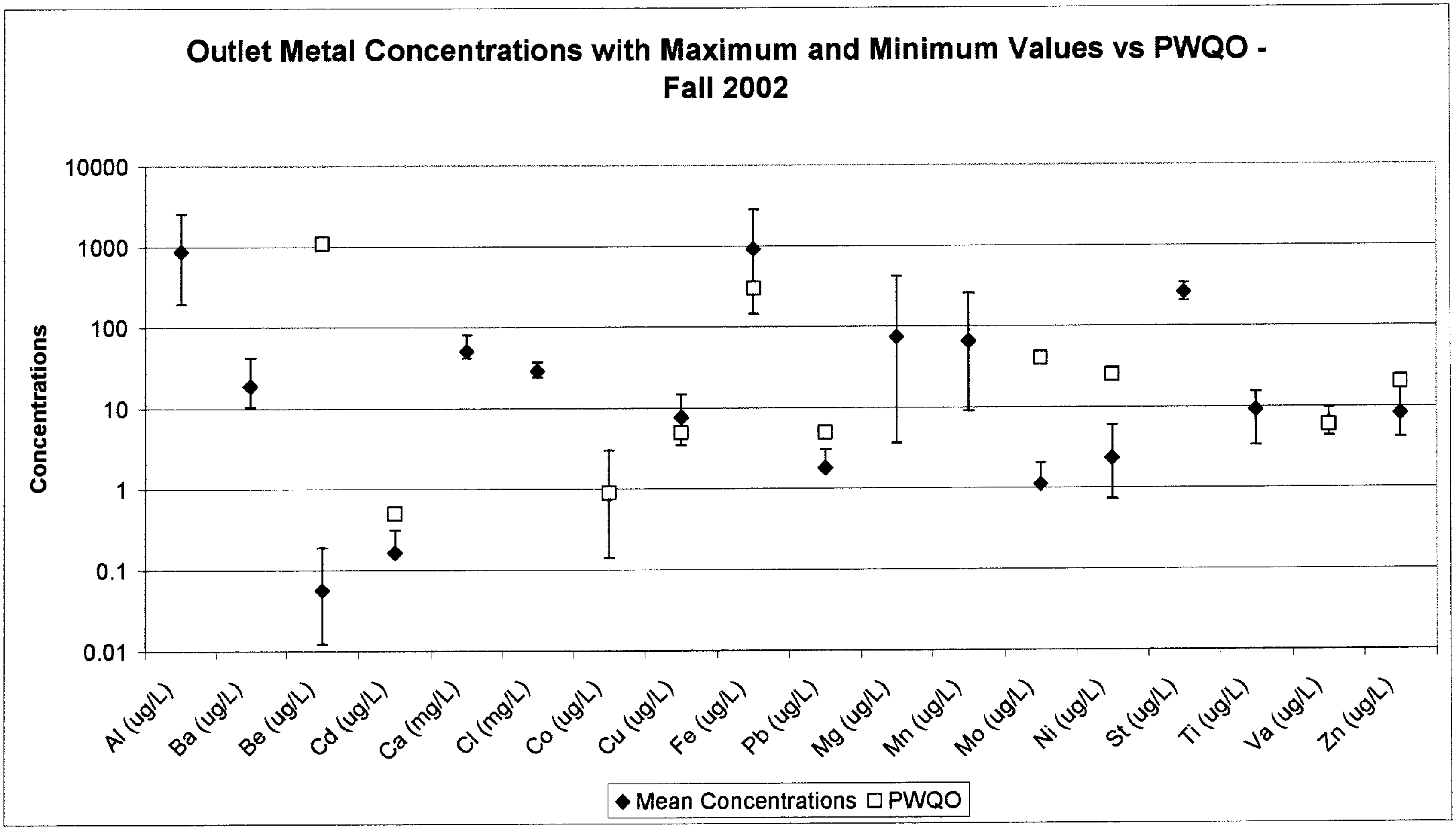


Figure 25: Outlet metal ncentrations with mean, maximum and minimum values vs. PWQO – Fall 2002

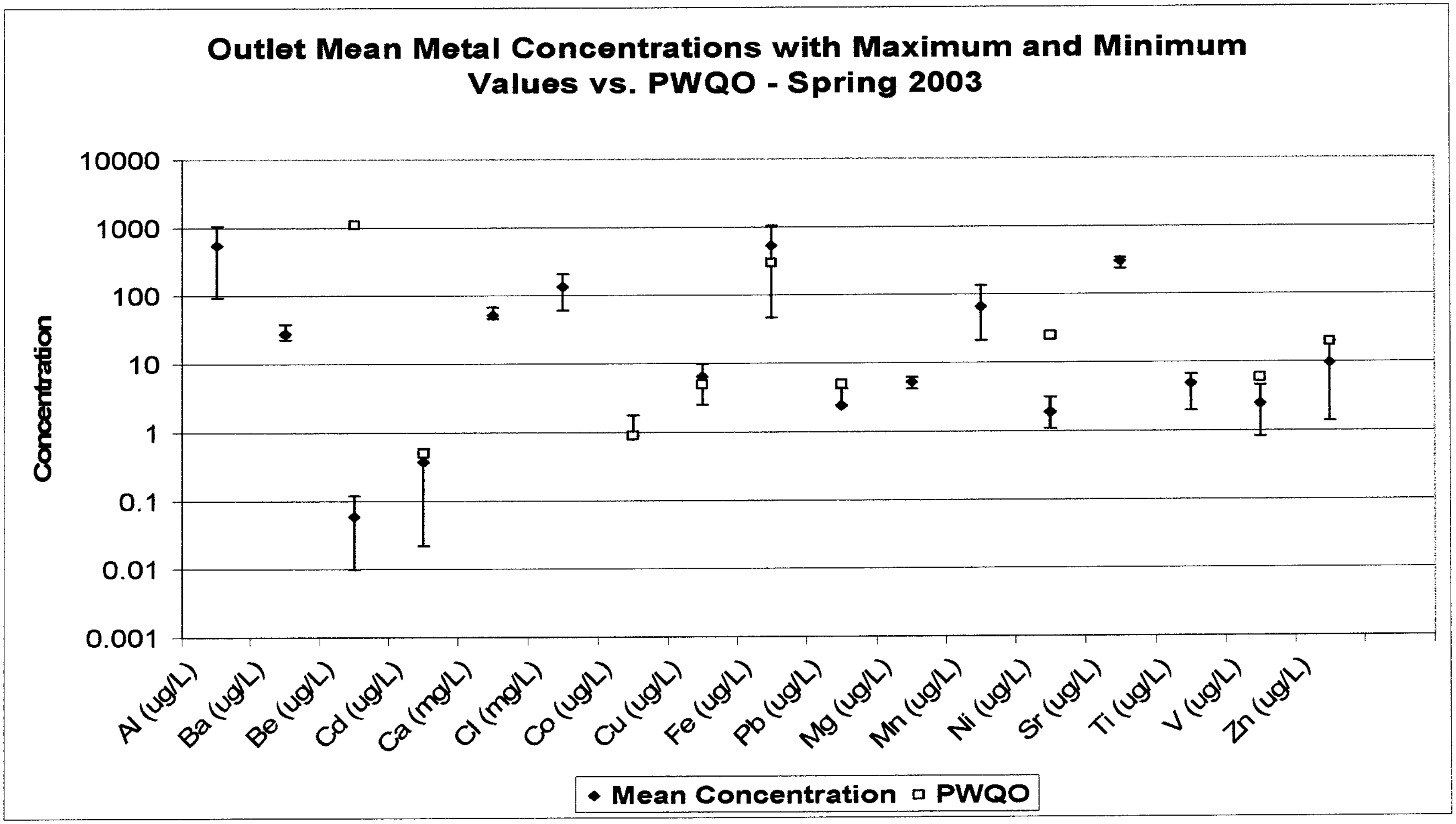


Figure 26: Outlet metal ncentrations with mean, maximum and minimum values vs. PWQO – Spring 2003

Generally, the metals entering and exiting the pond had some variability in concentrations for` different events. For example, iron, lead, and aluminum showed a significant range between maximum and minimum concentrations at inlet 1070. Inlet 510 showed less variability. Furthermore, the outlet showed even less variability, demonstrating some consistency in its removal capabilities.

The mean concentrations of copper and iron exceeded the PWQO recommended value to protect aquatic ecosystems. In addition, cobalt, vanadium, and zinc all had maximum concentrations that exceeded the PWQO value.

The interaction between metals and solids entering and exiting the pond can indicate the ability of the pond to remove several of these pollutants. For example, a fraction of the total metals entering the pond may have a high affinity to the suspended solids. Since the settling of solids is the primary removal mechanism of the pond, many of the metals will settle out with the suspended solids. However, a fraction of the metals will be dissolved, increasing the chances of not settling and ultimately exiting the pond. Table 27 presents linear correlations between selected metals and the suspended solids (SS) and dissolved solids (DS) entering and exiting the pond during the fall and spring sampling periods.

Table 27: Relationship between heavy metals vs. suspended solids and dissolved solids

Correlation Coefficients for Metal Concentrations						
	Inlet 1070		Inlet 510		Outlet	
Metal	**SS**	**DS**	**SS**	**DS**	**SS**	**DS**
Al	-0.63	0.44	0.99	0.22	0.85	-0.35
Ba	-0.54	0.31	0.99	0.3	0.93	-0.22
Be	-0.51	0.26	0.99	0.22	0.96	-0.23
Ca	-0.66	0.63	0.92	0.47	0.91	0.001
Co	-0.51	0.26	-0.31	-0.06	-0.17	0.39
Cu	-0.69	0.31	0.95	0.02	0.85	0.02
Fe	-0.61	0.48	0.38	0.03	-0.31	-0.42
Pb	-0.79	0.32	0.99	0.19	0.82	-0.37
Mg	0.53	0.33	0.55	0.02	0.02	-0.34
Mn	0.61	-0.35	0.91	0.51	0.43	0.55
Ni	0.2	-0.35	0.99	0.15	0.94	-0.2
St	-0.7	0.67	0.96	0.23	0.84	-0.27
Ti	0.51	0.06	0.63	0.79	0.27	0.63
Va	-0.73	0.76	0.7	0.32	0.41	-0.41
Zn	-0.75	0.37	0.93	0.4	0.48	-0.7

Table 27 shows a strong correlation of metals and suspended solids entering the pond through inlet 510. However, inlet 1070 showed several negative correlations between metals and suspended solids with stronger correlations between dissolved solids and metals. The outlet experienced a similar trend as inlet 510. Although inlet 1070 shows somewhat contradictory results when compared to inlet 510, the correlation coefficients at inlet 510 and the outlet suggest that settling does play a significant role in removing the metal pollutants. The difference in the catchment site conditions may explain why the inlet 1070 catchment area experienced lower concentrations in metals when suspended solid concentrations were high. Inlet 510 was completed in its development at the time of sample collection. Naturally, residents began to move in and occupy the area as the construction was completed. In turn, this increased the vehicular traffic within the catchment area of inlet 510. As discussed in the literature review, vehicular traffic is a significant source of heavy metals in stormwater. Thus, inlet 1070 experienced lower levels of heavy metals because construction was still in progress at the time of sampling and residents had not moved in.

5.2.2.4 Particle Size Distribution

The average cumulative particle size distributions were determined with the composite water quality results obtained throughout the monitoring period. Figure 27 shows the average cumulative particle size distributions for the fall sampling season (September – October, 2002). The individual event particle size distributions are included in Appendix F.

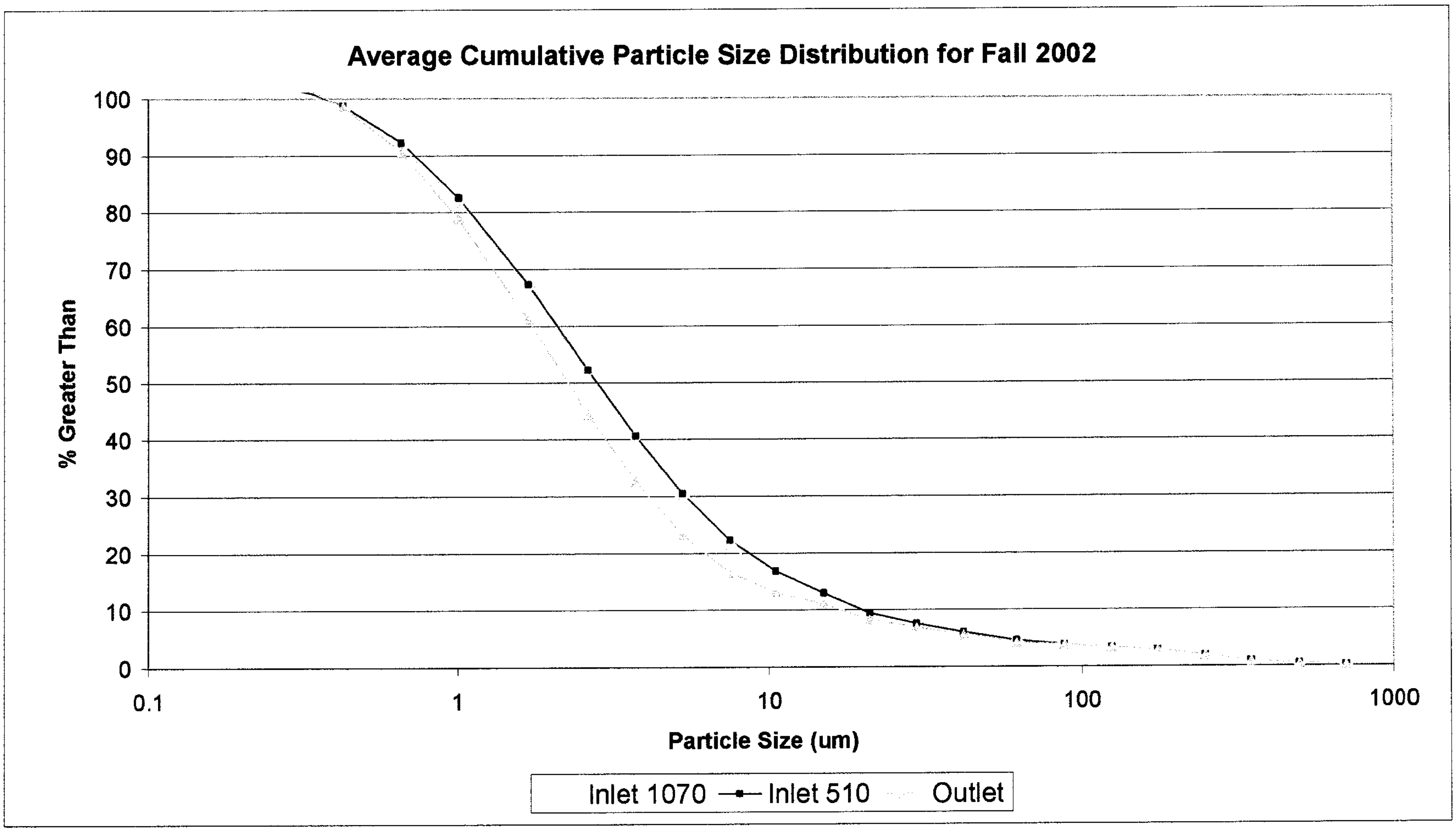


Figure 27: Average Cumulative Particle Size Distribution – Fall 2002

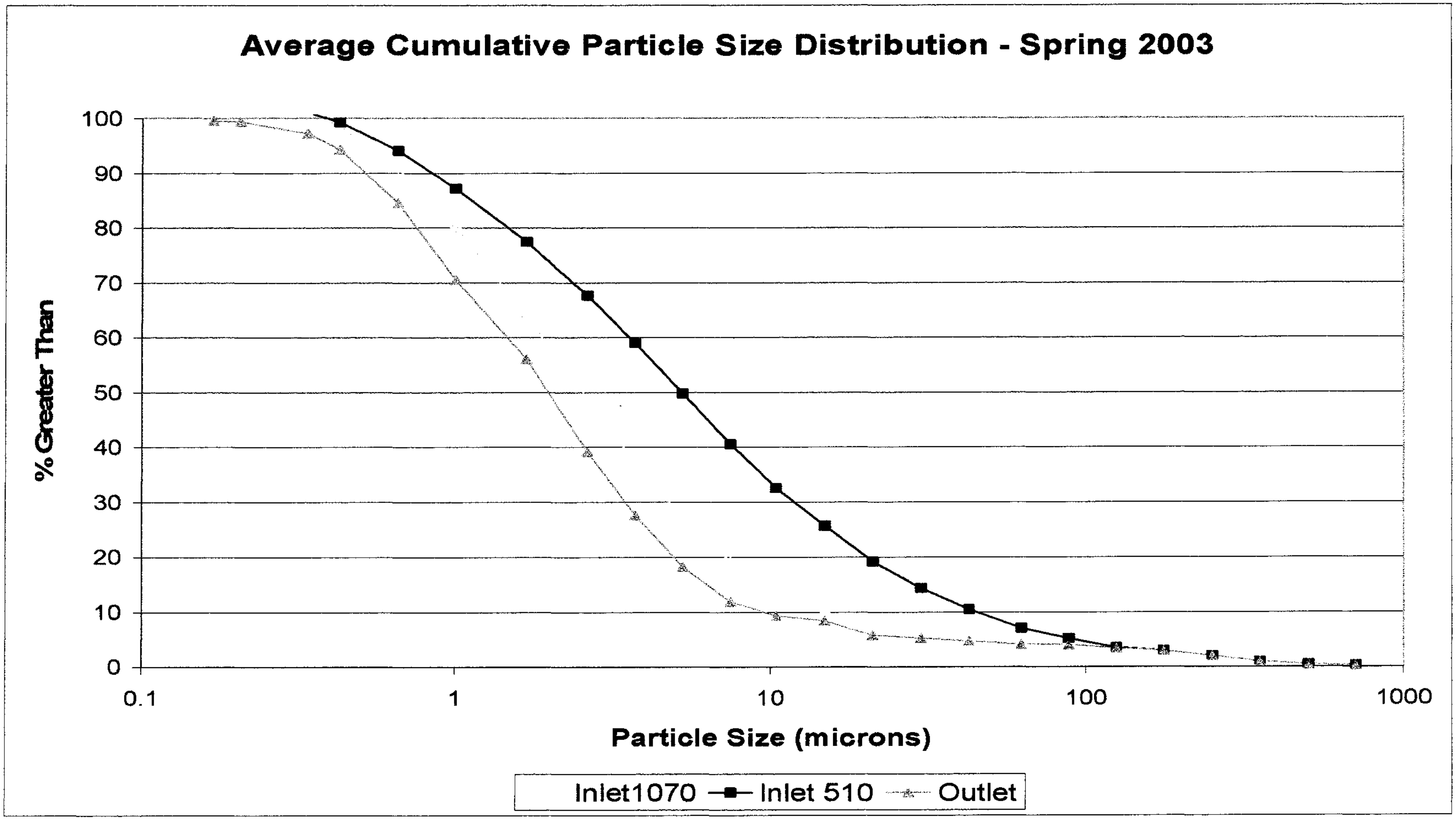


Figure 28: Average Cumulative Particle Size Distribution - Spring 2003

Figures 27 and 28 show that the majority of the particles found at each inlet and the outlet during the fall and spring sampling seasons are fine sediments. During the fall season, both inlets received runoff with approximately 98.5% of the particle size distribution less than 62 μm. Alternatively, the outlet shows a shift towards finer sediments with 99% of the particle size distribution less than 62 μm. In addition, the two inlets have just over 50% of the particle size distribution less than 2.63 μm, whereas the outlet shows an increase in finer particles with over 60% of its distribution under 2.63 μm. During the spring season, approximately 99% of the particle size distribution was less than 62 microns at inlet 1070, and 96% was less than 62 microns at inlet 510. In addition, approximately 55.4% was less than 2.63 microns at inlet 1070 and 35.8% was less than 2.63 microns at inlet 510. Conversely, approximately 99% of the particle size distribution was less than 62 microns at the outlet. In addition, 65% of the particle size distribution was less than 2.63 microns at the outlet.

Table 28 includes the D_{10} and D_{50} values of each of the distributions during the fall and spring sampling periods. In addition, the average D_{10} and D_{50} values are included.

Table 28: D_{10} and D_{50} Values for Fall 2002 and Spring 2003

	Inlet 1070	Inlet 510	Outlet	Inlet 1070	Inlet 510	Outlet
Event	**d10**			**d50**		
14-Sep	20.985	41.5	14.42	7.067	3.15	2.23
20-Sep	-	19.28	-	-	3.45	-
27-Sep	13.47	15.68	19.3	2.57	2.86	2.41
02-Oct	20.1	-	-	3.6	-	-
19-Oct	39.9	18.5	-	5.6	2.32	-
25-Oct	-	-	-	-	-	-
02-May	10.72	24.8	6.086	2.17	4.26	1
05-May	14.208	47.95	16.94	2.59	6.34	2.17
11-May	14.79	-	7.25	2.98	-	2.16
12-May	16.84	31.8	14.14	2.39	2.98	1.99
20-May	19.7	76.44	7.41	3.35	8.86	2.43
Average:	18.97	27.95	16.86	3.59	3.73	2.32

The event of October 25th received no analysis for the particle size distribution. This occurred on several other occasions for the different monitoring stations and can explain the absence of the D_{10} and D_{50} values for some of the events. Table 28 shows that the average D_{10} value for Inlet 1070, 510, and outlet is 18.97, 27.95, and 16.86 microns respectively. In addition, the average D_{50} value for Inlets 1070 and 510, and the outlet are 3.59, 3.73, and 2.32 microns respectively. The mean D10 values differ at the two inlets; inlet 1070 is 18.97 microns and inlet 510 is 27.95 microns. This difference may be attributed to the difference in the site conditions of each catchment area. For example, the inlet 1070 catchment area is undergoing construction and may be generating finer sediments due to the exposed soils.

Although Table 28 is demonstrating that there is a shift in particle sizes at the outlet to finer particles, this trend is difficult to ascertain. Indeed, 90% of the particle size distribution at the outlet is finer than 16.86 microns; at the inlets 90%

is less than 27 microns. However, it should be noted that the particle size analysis does not account for flocculation, or the agglomerated particles that is typical for stormwater sediment. The MOE laboratory uses a particle analyzer that requires the breakdown of grouped particles before passing through the particle analyzer. Thus, the results are reflective of the primary particles; the individual particles before they grouped together to form larger particles that settle out of the water column. Alternatively, if the particles that are agglomerating and flocculating out of the water column are the result of the incoming sediments, then the particles that did not settle out because they did not flocculate is what exited the pond at the outlet. Therefore the outlet particle size distribution may not have been as influenced by the preprocessing of the samples before entering the particle analyzer. Overall, it can be said that the particles entering the pond are fine sediments, and the particles exiting the pond are fine sediments. This is demonstrated in Table 28. This trend in particle size distribution should be noted for construction site runoff as it is especially important in terms of design specifications for sediment control ponds. This will be examined further in the next section.

5.3 Settling Velocities

The settling velocities were calculated for the mean D_{10} and D_{50} values presented in the previous section. In the MOEE SWMPP manual (1994), settling calculations are used to determine the forebay volume and length. As mentioned in the literature review, the Guidelines recommend the use of settling velocities included in a table that were derived through stormwater particle size distribution monitoring data which was collected in the NURP study (MOE 1994). Although these settling velocities are derived from stormwater samples not indicative of construction site runoff, the Guidelines support their method by claiming it is more conservative than using Stoke's Law or Newton's Law (MOE 1994). This is because a continuous model of settling rates was derived for the particles during both quiescent and dynamic conditions; Stoke's Law assumes ideal settling conditions (MOEE, 1994). According to the Guidelines, the forebay settling length is one of the design criteria determined and is accomplished with the following equation (MOEE Equation 3.3., pg. 89, 1994):

$$Dist = \sqrt{\frac{rQ_p}{V_s}} \tag{5}$$

Where

Dist = distance

r = length to width ratio of forebay

Q_p = Peak flowrate from the pond during design quality storm

V_s = Settling velocity (dependent on desired particle size)

The Guidelines deter any 'oversizing' of the forebay, by recommending that the forebay be designed to settle out 150 μm particles (MOE 1994). Based on their table (presented in Chapter 2), the settling velocity for 150 μm is estimated to be 0.0003 m/s. This particle size may seem rather large, especially when considering runoff from construction sites. However, the Guidelines further support this recommendation by claiming that the settling velocity estimated for the 150 μm particle is approximately one order of magnitude less than the settling velocity for a 40 μm particle given by Stoke's Law (MOE 1994). In addition to the forebay settling length, the length needed to slow a jet discharge is also required as the design criterion, and is referred to as the dispersion length. The dispersion length is recommended as a check on the forebay settling length to ensure adequate dispersion (MOE 1994).

The dispersion length is determined using the following equation (MOEE Equation 3.4 1994).

$$Dist = \frac{8Q}{dV_f} \qquad (6)$$

where

Dist = distance

Q = inlet flowrate (m^3/s)

d = depth of the permanent pool in the forebay (m)

V_f = desired velocity in the forebay

The Richmond Hill pond uses the settling velocity recommended for the 150 μm particle (0.0003m/s). Using the Guidelines' forebay settling length equation and the recommended settling velocity, the Richmond Hill construction sediment pond's forebay was calculated and required 11.25m (Sabourin Kimble & Associates Ltd 2000). This was then checked with the dispersion length equations. The dispersion equations determined for inlet 1070 and inlet 510 required 30m and 3.2m respectively. The Guidelines recommend that the forebay length should be greater than, or equal to the larger of the two forebay length equations (MOE 1994). For the purposes of this study, the settling velocities of the D_{10} and D_{50} particles were calculated and then used in the forebay settling length equation provided by the Guidelines. This allows comparison of the forebay settling length calculations using the recommended settling velocity with the settling velocities calculated using actual particle sizes collected at the site. Tables 29 and 30 present the settling velocities calculated for D_{10} and D_{50} mean particle sizes respectively, and the resulting forebay settling length using the MOEE equation. It should be noted that during the sampling period, equipment limitations prevented the collection of temperature data. Thus, the ambient air temperature was used to estimate the water temperature at the time of sampling. Air temperature data was obtained through the Environment Canada website from the Toronto Buttonville Airport weather station (Environment Canada, 2003b). This temperature was then used to determine the kinematic viscosity needed to complete Stoke's Law. Because the kinematic viscosity (ν) was to be estimated, a range was used based on the ambient air temperature. The range for ν was obtained from *Water Resources Engineering* (Mays, Table 2.1.2, pg 16, 2001). This range is also presented in Tables 27 and 28.

Table 29: Settling velocities and forebay settling length for mean D_{10} values from the fall and spring sampling periods

		D10 (microns)		Settling Velocity (m/s)			Forebay Settling Length (m)	
Event	Ambient Air Temp (°C)	Inlet 1070	Inlet 510	Temp for Kinematic Viscosity	Inlet 1070	Inlet 510	Inlet 1070	Inlet 510
14-Sep	21	20.985	41.5	15	0.0004	0.0014	10.42	5.21
				20	0.0004	0.0015	9.87	5.03
				25	0.0004	0.0017	9.29	4.73
20-Sep	24	-	19.28	20		0.0003		10.73
				25		0.0004		10.13
				30		0.0005		9.09
27-Sep	15.7	13.47	15.68	10	0.0001	0.0002	17.44	15.00
				15	0.0001	0.0002	16.30	14.00
				20	0.0002	0.0002	15.32	13.39
2-Oct	18	20.1	-	15	0.0003		10.93	
				20	0.0004		10.27	
19-Oct	7.5	39.9	18.5	5	0.0009	0.0002	6.35	13.68
				10	0.0007	0.0002	7.57	12.72
25-Oct		-	-					
2-May	10	10.72	24.8	5	0.0001	0.0004	23.64	10.22
				10	0.0001	0.0004	21.93	9.48
				15	0.0001	0.0005	20.43	8.85
5-May	8.3	14.21	47.95	5	0.0001	0.0014	17.87	5.28
				10	0.0001	0.0016	16.53	4.90
11-May	10.1	14.79		5	0.0001		17.13	
				10	0.0002		15.90	
				15	0.0002		14.85	
12-May	9.3	16.84	31.8	5	0.0002	0.0006	15.04	7.97
				10	0.0002	0.0007	13.96	7.39
20-May	13.9	19.7	76.44	10	0.0003	0.0040	11.93	3.08
				15	0.0003	0.0046	11.15	2.87

Table 30: Settling velocities and forebay settling length for mean D_{50} values from fall and spring sampling periods

		D_{50} (microns)		Settling Velocity (m/s)			Forebay Settling Length (m)	
Event	Ambient Air Temp (°C)	Inlet 1070	Inlet 510	Temp for Kinematic Viscosity	Inlet 1070	Inlet 510	Inlet 1070	Inlet 510
14-Sep	21	7.067	3.15	15	3.94E-05	0.000008	31.06	68.92
				20	4.46E-05	0.000009	29.19	64.98
				25	0.00005	0.00001	27.57	61.64
20-Sep	24	-	3.45	20		1.06E-05		59.87
				25		1.19E-05		56.51
				30		1.47E-05		50.84
27-Sep	15.7	2.57	2.86	10	4.5E-06	5.6E-06	91.89	82.38
				15	5.2E-06	6.4E-06	85.49	77.06
				20	5.9E-06	7.3E-06	80.25	72.15
02-Oct	18	3.6	-	15	0.00001		61.64	
				20	0.000012		56.27	
19-Oct	7.5	5.6	2.32	5	0.000018	3.2E-06	45.95	108.97
				10	0.000021	3.7E-06	42.54	101.34
25-Oct		-	-					
2-May	10	2.17	4.26	5	2.79E-06	1.07E-05	116.71	59.59
				10	3.24E-06	1.25E-05	108.30	55.14
				15	3.7E-06	1.43E-05	101.34	51.55
5-May	8.3	2.59	6.34	5	3.97E-06	2.38E-05	97.84	39.96
				10	4.61E-06	2.76E-05	90.79	37.11
11-May	10.1	2.98		5	5.26E-06		85.00	
				10	6.11E-06		78.86	
				15	6.99E-06		73.73	
12-May	9.3	2.39	2.98	5	3.38E-06	5.26E-06	106.03	85.00
				10	3.93E-06	6.11E-06	98.33	78.86
20-May	13.9	3.35	8.86	10	7.72E-06	5.39E-05	70.16	26.55
				15	8.84E-06	6.18E-05	65.56	24.80

The D_{10} values represent 10% of the distribution that is greater than this value. Thus, if the forebay settling length is calculated based on this particle size, it may only be capable of capturing 10% of the particles greater than or equal to this size. The forebay settling length calculated for the Richmond Hill sediment control pond using the MOE Guidelines and recommended settling velocity (0.0003m/s) was computed as 11.25m. According to Table 29, the computed forebay settling length using the settling velocity of D_{10} may be adequate for settling out this particle size. Unfortunately, the D_{10} particle size only accounts for 10% of the distribution.

The D_{50} particle size was used in Table 30 to calculate the settling velocities using Stoke's Law, and then used in the forebay settling length equation. This table shows much larger forebay settling lengths; all over 27m. These lengths are attributed to the very fine particles entering the pond. Moreover, the D_{50} value is representative of 50% of the particle size distribution, a more representative particle size to use based on the runoff samples collected in this study.

Indeed, the settling forebay lengths are significantly altered to account for the fine particles. In Chapter 4, specific gravities of different particle types were discussed. It is evident that not all particles will have a specific gravity of 2.65 that was used to calculate the settling velocities presented in the previous two tables. Thus, Table 31 takes this into account, and uses a smaller specific gravity value to account for the organic particles, which weight much less than other solids. These settling velocities were calculated using a specific gravity of 1.2. This is the higher end of the specific gravity found in wastewater organics outlined in Table 13 in Chapter 4.

Table 31: Settling velocities of D_{50} particles with specific gravity 1.2

		D50		Settling Velocity			Forebay Settling Length	
Event	**Ambient Air Temp**	**Inlet 1070**	**Inlet 510**	**Temp for Kinematic Viscosity**	**Inlet 1070**	**Inlet 510**	**Inlet 1070**	**Inlet 510**
14-Sep	21	7.067	3.15	15	5E-06	9E-07	87.18	201.06
				20	5E-06	1E-06	87.18	193.02
				25	6E-06	1E-06	79.58	177.95
20-Sep	24	-	3.45	20		1E-06		172.30
				25		1E-06		162.11
				30		2E-06		146.11
27-Sep	15.7	2.57	2.86	10	5.5E-07	7E-07	262.85	236.39
				15	6.3E-07	8E-07	245.60	220.72
				20	7.1E-07	9E-08	231.35	657.13
02-Oct	18	3.6	-	15	1.2E-06		175.77	
				20	1.4E-06		164.75	
19-Oct	7.5	5.6	2.32	5	2.3E-06	4E-07	129.96	316.23
				10	2.6E-06	4E-07	120.89	291.24
25-Oct		-	-					
02-May	10	2.17	4.26	5	7.7E-07	0.0011	222.68	5.95
				10	8.9E-07	0.0009	206.61	6.65
				15	1E-06	0.001	192.99	6.21
05-May	8.3	2.59	6.34	5	8.4E-07	0.0009	213.04	6.49
				10	9.7E-07	0.001	197.67	6.02
11-May	10.1	2.98		5	9E-07		205.70	
				10	1E-06		190.86	
				15	1.2E-06		178.28	
12-May	9.3	2.39	2.98	5	8E-07	0.0006	217.36	7.83
				10	9.3E-07	0.0007	201.69	7.27
20-May	13.9	3.35	8.86	10	1.1E-06	0.0012	185.36	5.53
				15	1.3E-06	0.0014	173.14	5.17

Tables 29 to 31 demonstrate that there is a general trend on how the particle size can influence the sediment forebay length calculations; the smaller and lighter the particle size the larger the sediment forebay. Moreover, if the smaller particle sizes collected in this study were used to derive the settling characteristics within a wet pond stormwater management facility, the permanent pool storage may also be altered. This is because the storage requirements were derived through the settling model based on the distribution of particle sizes presented in Table 3.3 in the Guidelines and presented in Chapter 2 in this study.

Ideally, the estimation of particle settling velocities would involve settleability testing in columns with samples collected from the Richmond Hill sediment control pond. Unfortunately, due to time and resource constraints, this was not a possibility. Thus, it is recognized that there are limitations in using Stoke's Law for the settling velocity analysis of this thesis. The analysis did not account for flocculated particles. This would influence the distribution found at the two inlets. Thus, the inlet distributions are only representative of the primary particles rather than the distribution entering the pond under natural conditions. This may account for the high removal efficiencies discussed in Section 5.5.2. For example, if it is assumed that pollutants are adsorbing to the fine particles found at the inlets and that these particles are flocculating, then the agglomerated particles will settle at faster rates than what is presented in Table 29 and 30. In turn, the pollutants and sediments will settle out of the water column resulting in high removal efficiencies. Other limiting factors in this analysis include the assumption in Stoke's Law that particles were spherical and were exposed to ideal settling conditions. Furthermore, the composition of the particle size distribution was not accounted for. For example, particles with a lower specific gravity were applied to Stoke's Law to account for organic particles. However, particles entering the pond may consist of mineral, organic matter, and colloids. The compositions of these types of particles were not analyzed. Thus, the analysis is limited in determining what types of particles are expected to be exported off a construction site during an event. This is also related to the soil composition of the catchment area.

Despite these limitations, a general trend is noted in the particle size distribution and settling velocity analysis. Thus, these analyses have proven to be useful as they help to characterize the relationship between particle size and pond design. For example, the MOE Guidelines (1994) recommend the use of the 150 micron particle size with a settling velocity of 0.0003m/s. This recommended particle size was derived from the U.S. NURP program, which is indicative of urban stormwater runoff. This settling velocity is the equivalent of the settling velocity for a 20 micron particle as determined by Stoke's Law. In addition, this settling velocity is commonly used in practice, just as it was used for the design specifications of the Richmond Hill sediment control pond. However, both the particle size distribution analysis and the settling velocity analysis outlined in the last two sections demonstrate two useful factors. Firstly, there are fine particles entering the pond (90% <23.65 microns). This may differ from the particle size commonly found in urban stormwater runoff, particularly those reported and used in the MOE Guidelines. Secondly, the smaller particle sizes found at the site have an affect on the sediment forebay length equation. When using the settling forebay length equation provided by the MOE Guidelines, it becomes evident that the smaller particles sizes result in larger sediment forebay lengths. These factors need to be taken into consideration when implementing design criteria for sediment control ponds used at construction sites.

5.4 Bottom Sediment Accumulation and Construction Activities

The bottom sediment accumulation was recorded during the fall and spring sampling season. Tables 32 and 34 present the increase in elevation recorded during each monitoring period. In addition, the total accumulation recorded for each sampling season is included. From these tables, the total sediment load from the catchment area during the monitoring period was computed. This was determined by multiplying the average sediment depth (m) recorded during the monitoring period by the area of the pond (m^2) at the bottom elevation. This resulted in the total sediment load during the monitoring period (m^3). This value was then divided by the catchment area (15.1 ha) to obtain the seasonal sediment accumulation rate based on a three month time period. The three month period is used to represent both the fall (September – November) and spring (April – June) months.

According to the data presented in Table 32, the total sediment load from the catchment area during the three month period is 6.67 m^3/ha. The annual accumulation is determined through the addition of seasonal rates. This is because during the winter months, the rate of accumulation is expected to slow down. This is a result of the halting of construction activities and the changes in precipitation. The summer months will also slow down, as precipitation is typically lower than the spring and fall seasons. Thus, a three month season is assumed for the fall and spring months. If the rate of accumulation according to Table 32 was typical for the fall season, the estimated seasonal sediment load would total 10.01 m^3/ha/fall season.

The construction activities were monitored approximately every two weeks during the same monitoring period as the bottom sediment accumulation. Each drainage area serviced by the two inlets was divided into subgroups. The subgroups are made up of residential lots, and roads. The subgroups were determined according to the site plan, where each subgroup is outlined and their respective areas are given in hectares. The maps used to determine the subgroups are included in Appendix G.

Table 32: Bottom sediment accumulation for fall 2002

Station	Elevation of Bottom of Pond	Elevation in August (m)	Elevation in October (m)	Total Accumulation (August - October) (m)
1	298	299.235	300.555	0.085
2	297.5	298.29	299.4	0.32
3	297.5	298.405	299.44	0.13
4	298	299.25	300.715	0.215
5	298	299.63	301.235	-0.025
6	298	299.135	300.375	0.105
7	298	299.195	300.435	0.045
8	298	299.47	301.02	0.08
9	298	298.995	300.255	0.265
10	298	298.5	298.79	-0.21
11	298	298.355	298.695	-0.015
12	298	298.525	298.86	-0.19
13	298	298.375	298.695	-0.055
Accumulation rate (m^3/ha)				6.67
Fall season accumulation (m^3/ha/fall season)				10.01

Table 33: % of total area exposed for fall, 2002

Drainage Area	Total Area (ha)	% Total Area Exposed for Fall, 2002						
		14-Aug	22-Aug	4-Sep	13-Sep	26-Sep	4-Oct	22-Oct
Inlet 1070A	1.8525	90%	90%	90%	90%	90%	90%	57%
Inlet 1070B	2.1375	100%	100%	100%	100%	100%	100%	100%
Inlet 510	0.99	43%	10%	0%	0%	0%	0%	0%

Monitoring observations of each lot included whether the land was sodded or the soil exposed, and whether the driveway was paved or left as gravel. In addition, any other observations were recorded such as dirt piles, and the condition of the road. The information gathered for each lot was analyzed. According to each subgroup located within the catchment area (Appendix F), the amount of area where the soil was exposed was determined and included in Table 33.

For the fall sampling period, the area located within Inlet 1070 B drainage area was completely exposed. In addition, Inlet 1070 A was not receiving sod until near the end of October. Inlet 510 was completed by the beginning of September. Thus, inlet 510 was totally developed during the fall and spring sampling period. This proved to be beneficial as it allowed for comparisons between developed areas versus areas undergoing construction.

During the spring sampling period, the bottom sediment accumulation was monitored once in June. Table 34 shows that the total sediment load from the catchment area during this monitoring period (October 02 – June 03) is 3.64 m^3/ha. Because the pond was completely frozen over during the months of January through March, it can be assumed that there was little to no accumulation during this time. No flow was observed entering or exiting the pond. In addition, the flow that was observed during the months of November and December were very minimal as temperatures dropped and there was little precipitation. Thus, the pond was receiving little (very fine sediments) to no sediments during this five-month period. Based on these observations, it can be assumed that the majority of the sediments accumulated before the spring survey was completed (end of June), accumulated over a three month period (April - June). Thus, the estimated sediment load for the spring season can be calculated using the October 2002 surveying results as the initial elevation points for the spring accumulation. Therefore, the accumulation for the spring season is 5.46m^3/ha/spring season. This rate of accumulation is lower than the fall sampling results. This can be related to the progression of construction activities during this time.

Table 34: Bottom sediment accumulation for spring 2003

Station	**Elevation of Bottom of Pond**	**Elevation in October (m)**	**Elevation in June (m)**	**Total Accumulation (October 02 - May 03**
1	298	299.32	299.56	0.24
2	297.5	298.61	298.64	0.03
3	297.5	298.535	298.49	-0.045
4	298	299.465	299.47	0.005
5	298	299.605	299.67	0.065
6	298	299.24	299.27	0.03
7	298	299.24	299.35	0.11
8	298	299.55	299.55	0
9	298	299.26	299.14	-0.12
10	298	298.29	298.37	0.08
11	298	298.34	298.31	-0.03
12	298	298.335	298.33	-0.005
13	298	298.32	298.37	0.05
Accumulation rate (m^3/ha)				3.64
Spring season accumulation (m^3/ha/spring season)				5.46

Table 35: % Total area exposed for spring 2003

Drainage Area	Total Area (ha)	% Total Area Exposed, June 27, 2003
Inlet 1070A	1.8525	0
Inlet 1070B	2.1375	100
Inlet 510	0.99	0

The construction activities during the spring sampling period gradually progressed towards completion. At the beginning of the spring sampling period the majority of the catchment area draining into inlet 1070 was still exposed. Piles of soil were a constant presence in addition to dirty roads. The construction activities were monitored on two occasions during this time period. Unfortunately, the first monitoring results recorded were misplaced and never recovered. The second set of results were recorded and presented in Table 35.

Table 35 demonstrates that the lots in Inlet 1070A were almost completed in their development, with only the backyards left exposed and front yards sodded. Unfortunately, these recorded observations were misplaced. Yet, by the end of June, these lots were completely sodded and gravel comprised driveways. Thus, this progression in construction activities is reflected in the rate of bottom sediment accumulation observed in the pond. The rate of accumulation decreased in the spring sampling season, as fewer lots were exposed to erosion processes.

5.5 Discussion

During the implementation of the monitoring program and the analysis of results, several issues emerged. It became evident that these issues were pivotal in reaching the study's outcome, which is ultimately the evaluation of the performance of the pond. These issues concerned the methodology used to monitor stormwater events, and the methodology used to analyze both water quality and quantity data. The following section presents the importance of these issues as they can greatly influence the outcome of any study with similar goals and objectives.

The methodology employed to monitor stormwater events is a critical factor that can significantly influence the results of a study. Indeed, monitoring programs are initiated with the best intentions of collecting samples that bear the highest degree of representation of an event. However, limited resources often restrict the techniques used to collect these samples. Ideally, the best monitoring program will include numerous samples collected at minimum time intervals extending over the entire length of a hydrograph. Yet, water quality samplers are not equipped with an infinite source of bottles. This ideal situation is also impractical if the program involved the collection of grab samples. Thus, stormwater monitoring programs typically involve water quality samples taken at specified time intervals. This method provides a 'snapshot' of a specific time period during an event. The timing of these 'snapshots' can significantly influence the outcome of any monitoring program. For example, stormwater runoff is highly variable and difficult to characterize. For some events, the pollutant load peaks before the hydrograph, thus encouraging sampling programs to begin collecting the bulk of the samples at the start of an event

(Novotny & Olem 1994). However, this is not characteristic of all runoff events. For example, Figures 28 – 29 describe the suspended solids concentrations entering the Richmond Hill pond.

The May 5th event shows a distinct peak in suspended solids concentration before the flow hydrograph reaches its peak. However, for the September 27th event, the suspended solids pollutograph shows a similar temporal pattern as the hydrograph. Indeed, the start time of sampling is important. If the pollutants carried by runoff are peaking in concentration before the hydrograph then it is critical to collect samples before the hydrograph reaches its peak. If this portion of the pollutograph is missing, it can significantly alter the results and ultimately misrepresent the pollutant load of an event. Thus, collecting numerous samples over the entire event is ideal. However, determining the ideal time interval is difficult. Initially, samples were collected every five minutes, as seen in the September 27th pollutograph (extends over 120 minutes). However, it is evident in the September 27th pollutograph that the tail of the hydrograph was missing water quality samples. Taking this into consideration and the highly variable characteristics of stormwater pollutants, the sampling time interval was increased to fifteen minutes to better represent the storm event. Figures 29-32 demonstrate the benefits of increasing the sampling interval.

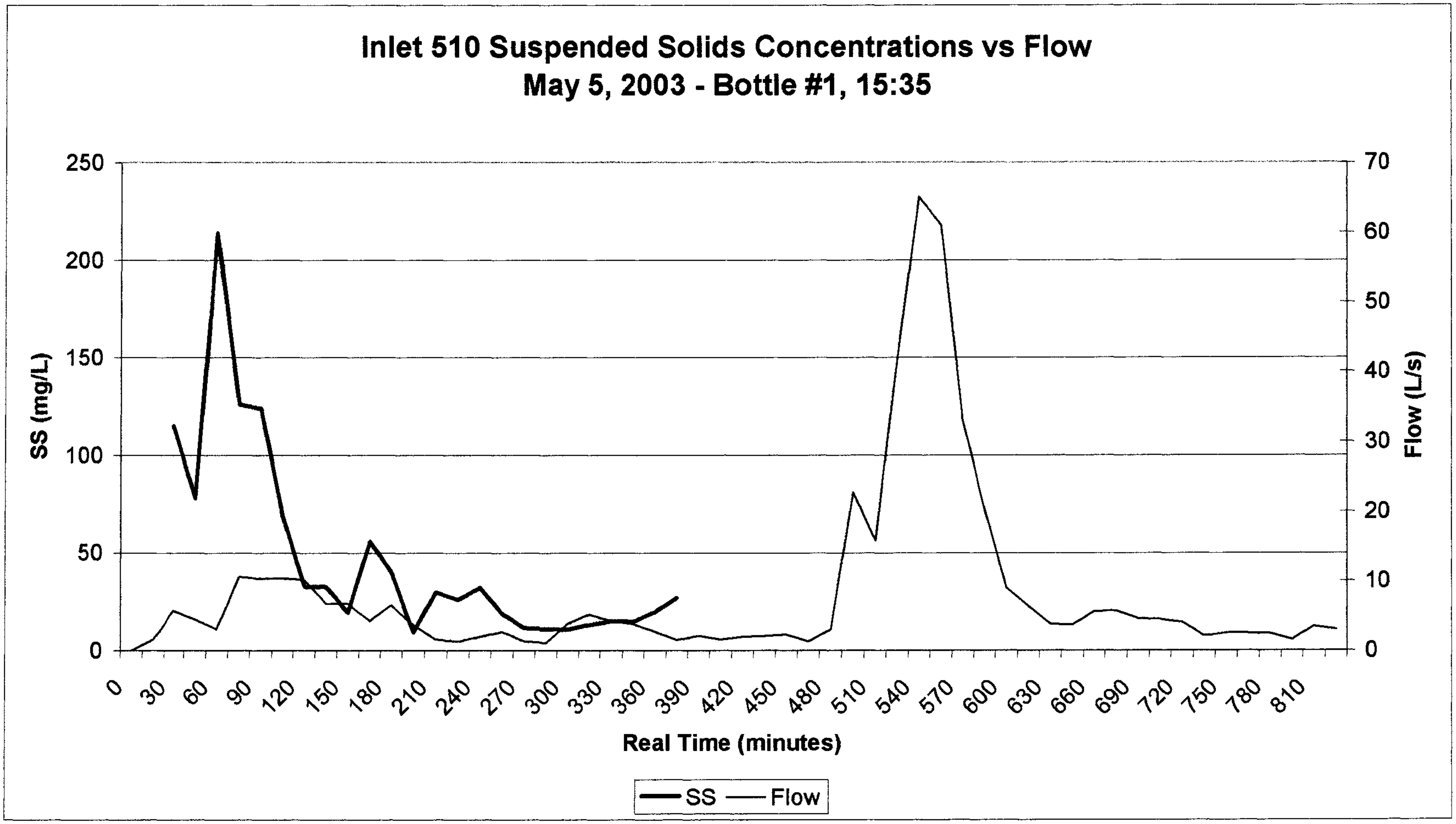


Figure 29: Inlet 510 snded solids concentrations May 5, 2003

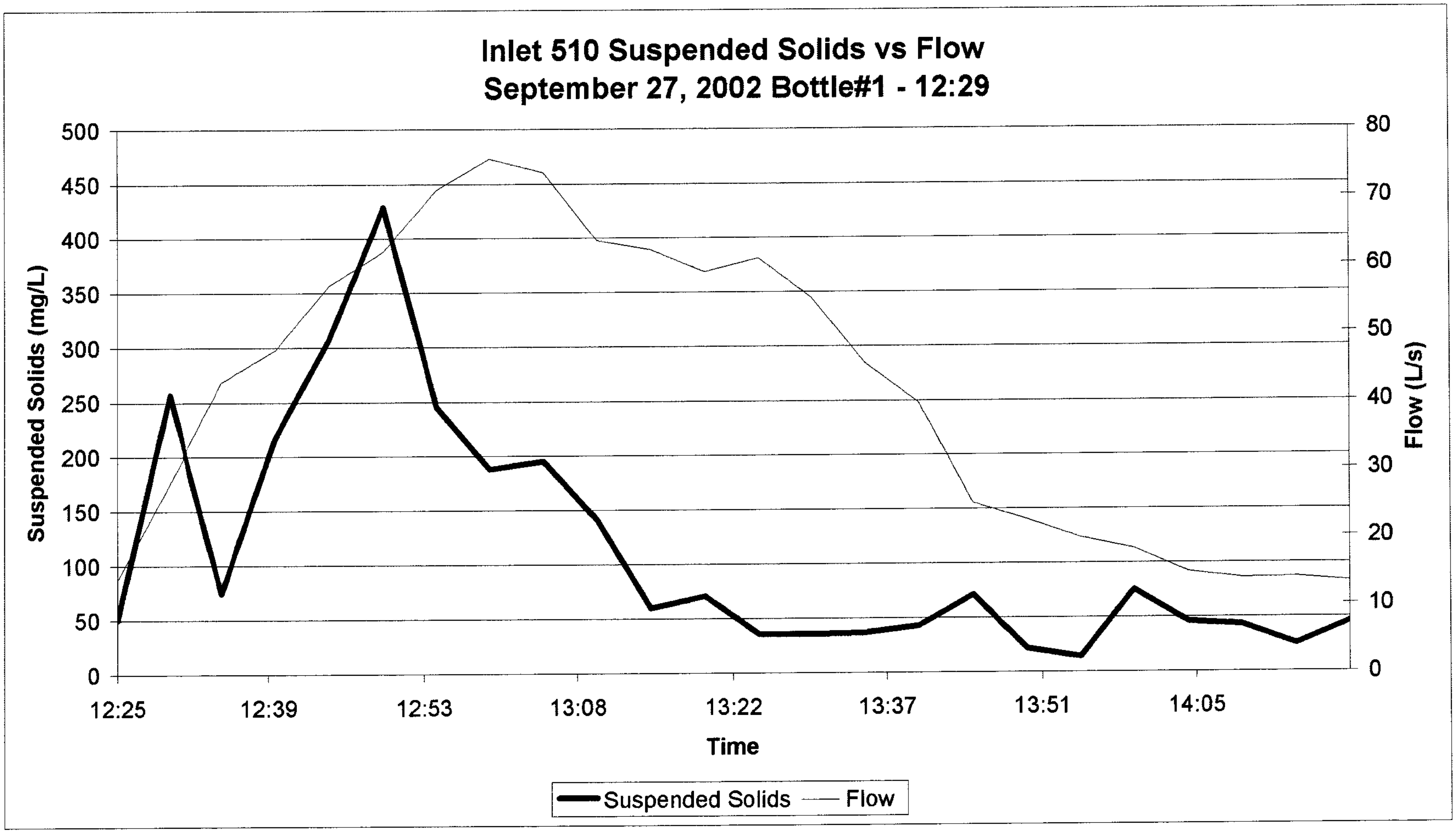


Figure 30: Inlet 510 suspended solids concentrations September 27th, 2002

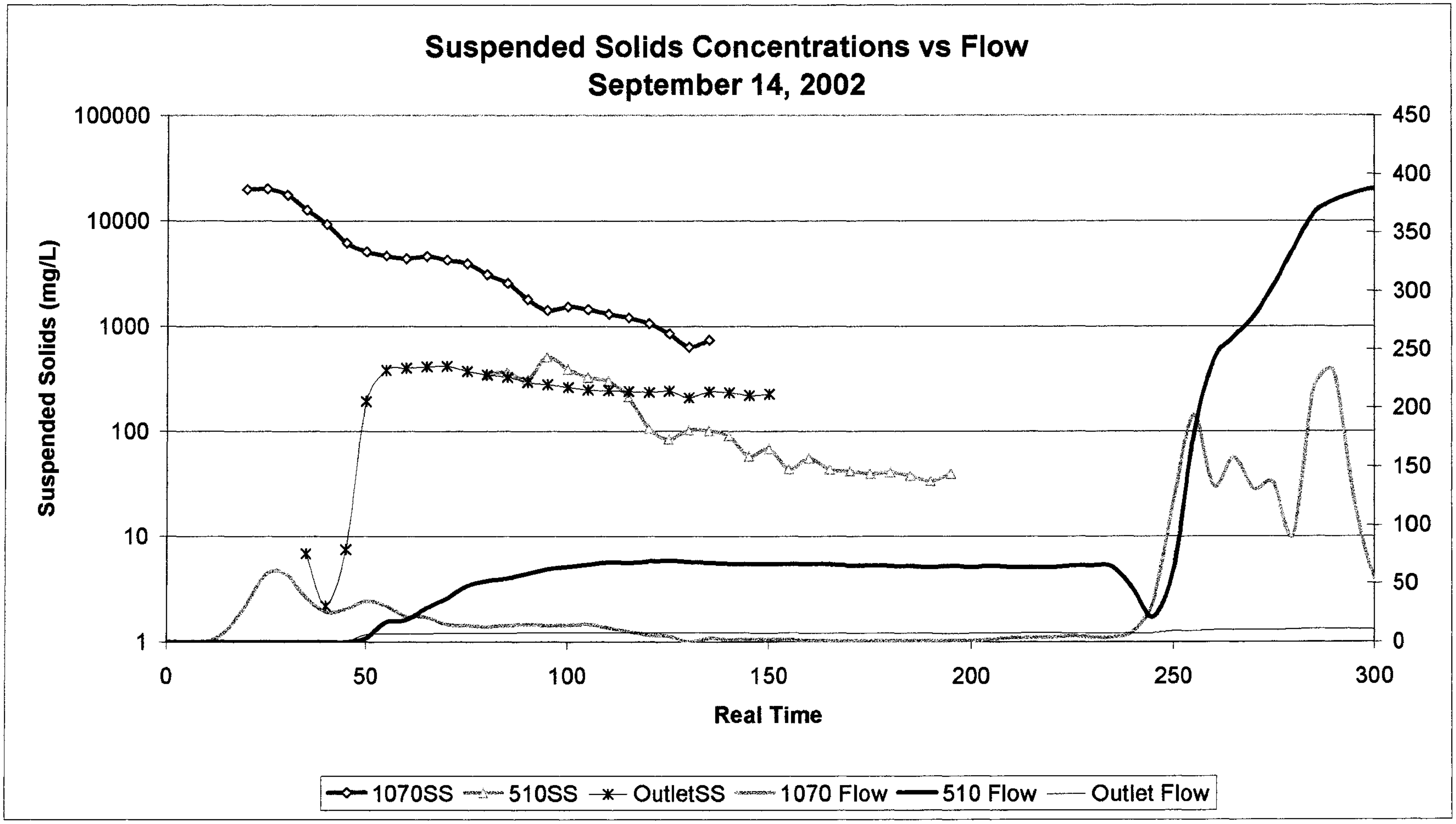


Figure 31: Suspended solids concentrations September 14, 2002

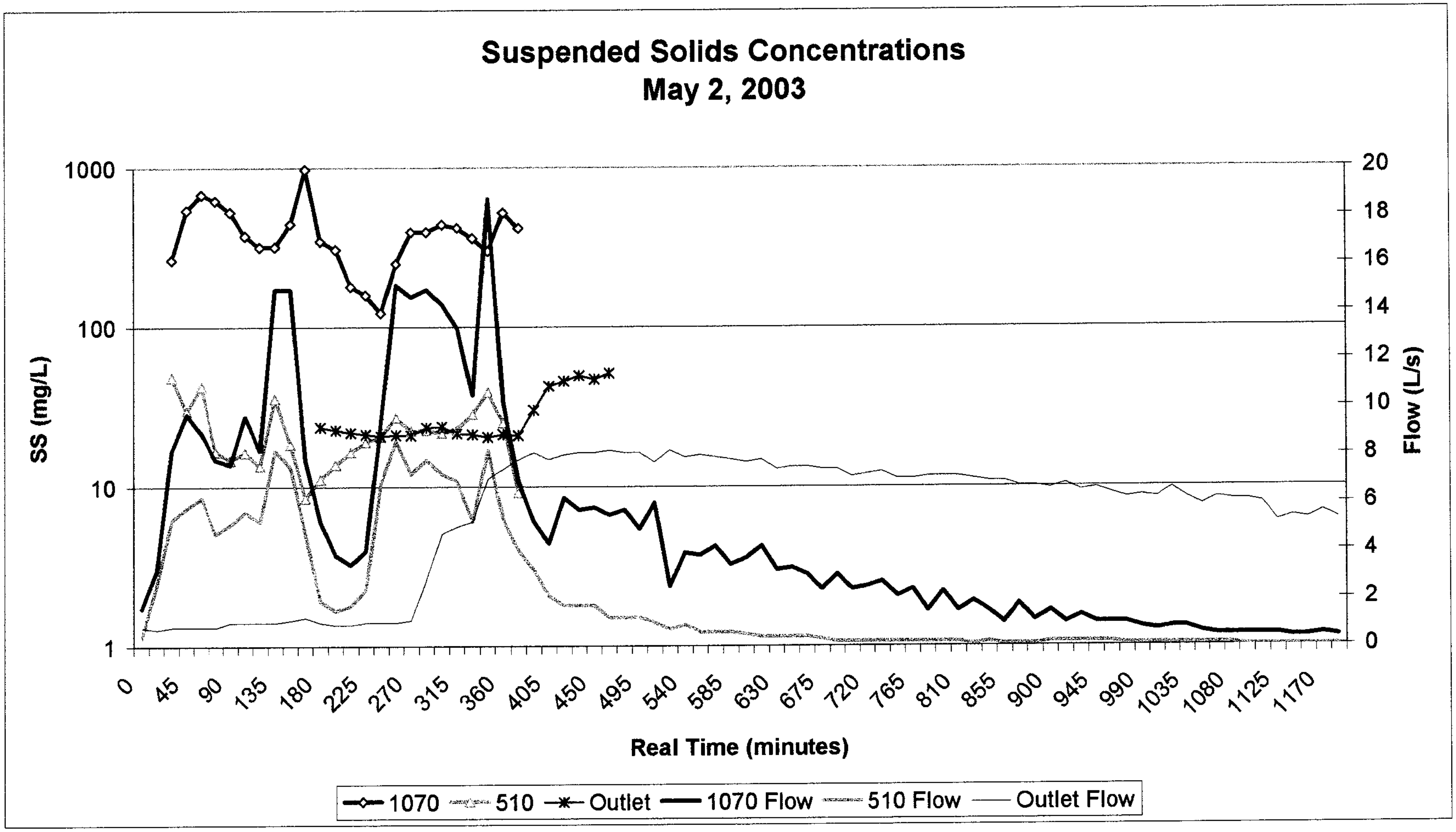


Figure 32: Suspended solids concentrations May 2, 2003

The samples collected during the fall sampling event on September 14 were collected at the beginning of the event before the flow reached its peak in the hydrograph. These samples were collected over a shorter duration; one bottle every five minutes. The pollutographs for the May 2nd event clearly demonstrate a better representation as it began at the beginning of the event, and extended almost to the end of the hydrographs. It is also evident that the samplers may have been triggered prematurely for the September 14th event. This occurred on a few occasions when there was an initial bump in the hydrograph that raised the incoming flow levels high enough to trigger the sampler before the event occurred. A possible explanation for this initial bump includes the watering of lawns by residents who have received new sod. When new sod is established, residents are encouraged to water their lawns frequently. Depending on the number of residents and the timeframe when the lawns are watered, this may produce flow at the inlets. This can also affect the water quality results as the runoff produced by the watering of lawns can also carry pollutants to the pond. In order to avoid the premature triggering of the samplers, two practices were amended in the monitoring program. First, the trigger levels were raised. Second, the sampling time intervals were extended. By extending the sampling time intervals, the water quality samplers collected over a longer period of time, increasing the chances of collecting an event if the sampler were triggered prematurely.

Indeed, the start time and time intervals of sample collection can greatly influence the results of a study. Another area where this issue became significant is the timing of sample collection at the outlet. It became evident that an ideal sample collection at the outlet would occur within a close approximation to the pond lag times. Section 5. 2.1, *Discrete Analysis*, discussed the effects of triggering the outlet sampler prematurely, essentially missing an important part of the pollutant load during an event. Programming the outlet sampler to trigger after the two inlets have triggered proved to be difficult. The samplers are not connected to a central programming device. Thus, the outlet triggering level had to be adjusted to sense an increase in level after the inlets received runoff. Evidently, the triggering level was set too low for the September 14th event as it triggered only 30 minutes after the two inlets were triggered. Unfortunately, the appropriate triggering level can only be attained through the trial and error process. This became increasingly difficult, especially if there was back-to-back events. Under these circumstances, the outlet is still experiencing outflow from the previous event. This outflow is continuously decreasing in level as it drains the pond. Therefore, several factors must be considered when setting the trigger level. For example, the start time of the next event needs to be considered. In addition, the extent of decrease in water level should also be considered.

Ultimately, the timing of sampling became a sensitive issue. This issue was governed by many factors. Some of these factors are preventable and some are unavoidable. For example, the duration of a storm and its intensity is unpredictable. An event can generate minimal rainfall, causing the initial bump in the hydrograph as seen in September 14th event, and trigger the sample prematurely. Although these issues may never be fully resolved, they can be minimized by extended the sampling time interval, and becoming familiar with the sensitivities in the triggering level.

In analyzing water quantity data, several issues became apparent that could ultimately alter the outcome of the study. One of these issues involved the interpretation of an event. This involves determining the start and end of an event. At first glance, this seems simple, especially at the Richmond Hill site where there is no baseflow. However, this can prove to be difficult, particularly when there are back-to-back events. Any choice regarding the start and end times of an event can alter both water quality and quantity results. This is because the start and end times will influence the calculation of the total volume of a hydrograph. In turn, total volumes are used to calculate the centroids of the hydrograph, which are used to determine the hydraulic detention time. In addition, total volumes are used to calculate the mass loads and performance data. Indeed, it is important to maintain a standard method of determining the start and end times of an event throughout the analysis. Yet, some events are difficult to determine these two elements of a storm. An example of this is the event of May 5^{th}, 2003. Figure 33 presents the hydrographs for all runoff volumes, and hyetograph of the rainfall for the event.

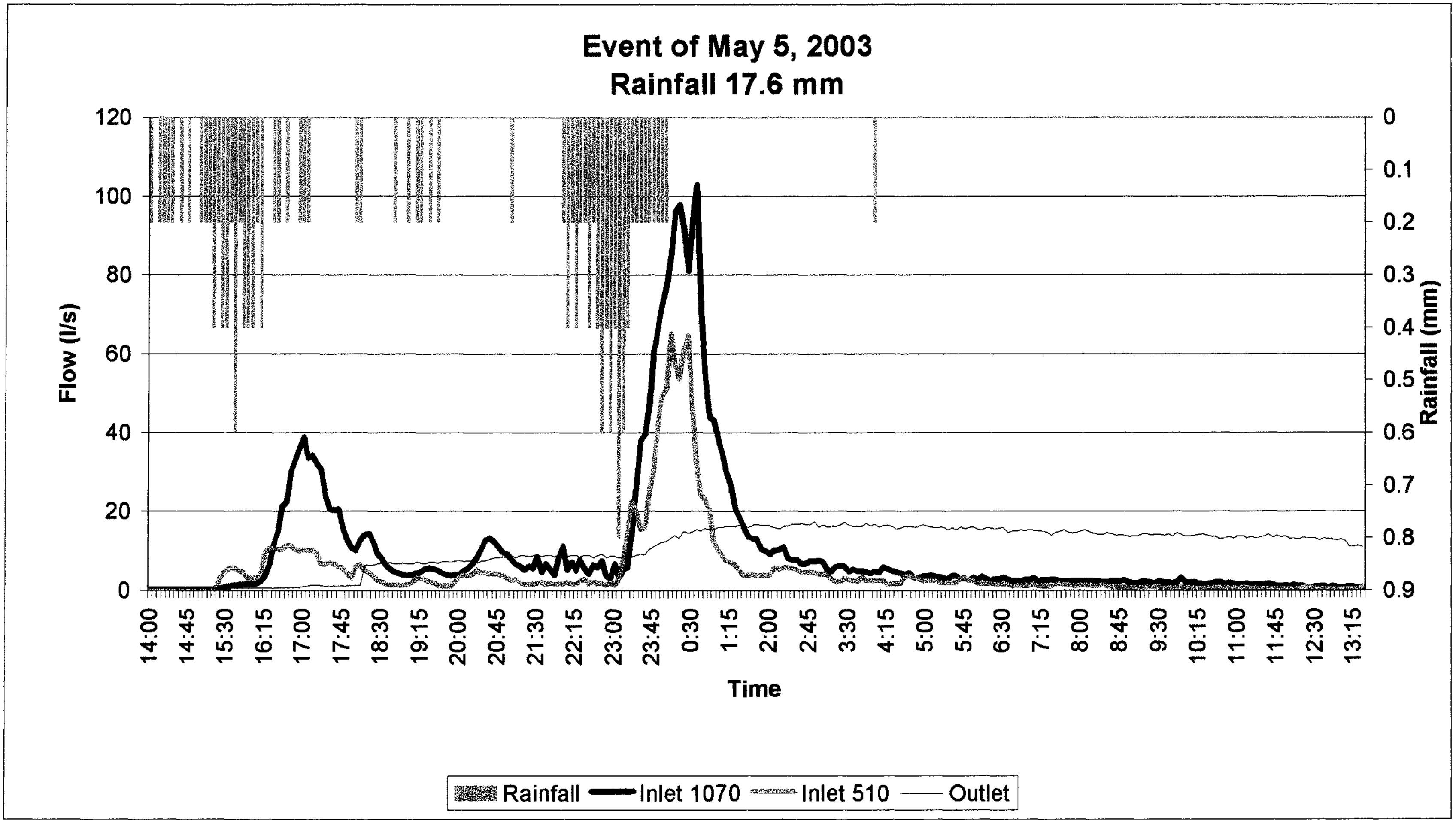


Figure 33: Event of May 5 – 6, 2003

Evidently, the event started on May 5th and gradually receded before continuing throughout the 6th with a much stronger peak in the hydrograph. The water quality samples were taken during the first peak in the hydrograph. Thus, the first peak in the hydrograph is the desired event for analysis. However, Figure 33 raises an interesting question. When does the first peak or event end? More specifically, this event raises the issues regarding inter-event times. Inter-event times can be determined in various ways. For example, when reporting rainfall events with duration of rainfall in days the minimum inter-event dry period is one day (Wanielista & Youseff 1993). However, the choice of inter-event dry period is dependent on the analysis, as long as the rainfall events are independent of each other. Typical inter-event dry periods extend over 4 to 6 hrs (Wanielista & Youseff 1993). The May 5th event shows two distinct rainfall periods in the hyetograph followed by two peaks in the hydrographs. The time difference between the two rainfall periods is 2.5 hrs. Indeed this is less than the typical inter-event dry period. However, the samplers collected water quality samples over the duration of the first peak in the hydrograph. No water quality samples were collected for the second peak. If this were to be treated as one event, the results may be misrepresentative of the event. This would be particularly evident when calculating the mass load and performance results, as the total runoff volume of the entire event would significantly affect the mass load calculations. Thus, this event was considered as two events; May 5th, and May 6th. When separating these two events, the inlet hydrographs were simple as they tailed off to relatively low flows before the second peak. However, the cutoff for the outlet hydrograph was not as obvious. Because the outlet structure is designed to slow the outflow, it significantly reduces the peak in the outlet hydrograph creating a much longer tail until its flow is finally reduced, signaling the end of the event. Thus, when there is a back-to-back event it is difficult to determine when the outlet hydrograph ends for one event and begins for the other. There are many factors that should be considered to determine the start and end times of an event, particularly at the outlet. For example, the displacement of water stored in the pond by the following event may influence the decision to delay the start time in the analysis until the hydrograph has begun its rising limb. In the case of May 5th, a factor that influenced this decision is the pollutograph. Figure 34 presents the outflow hydrograph and pollutograph.

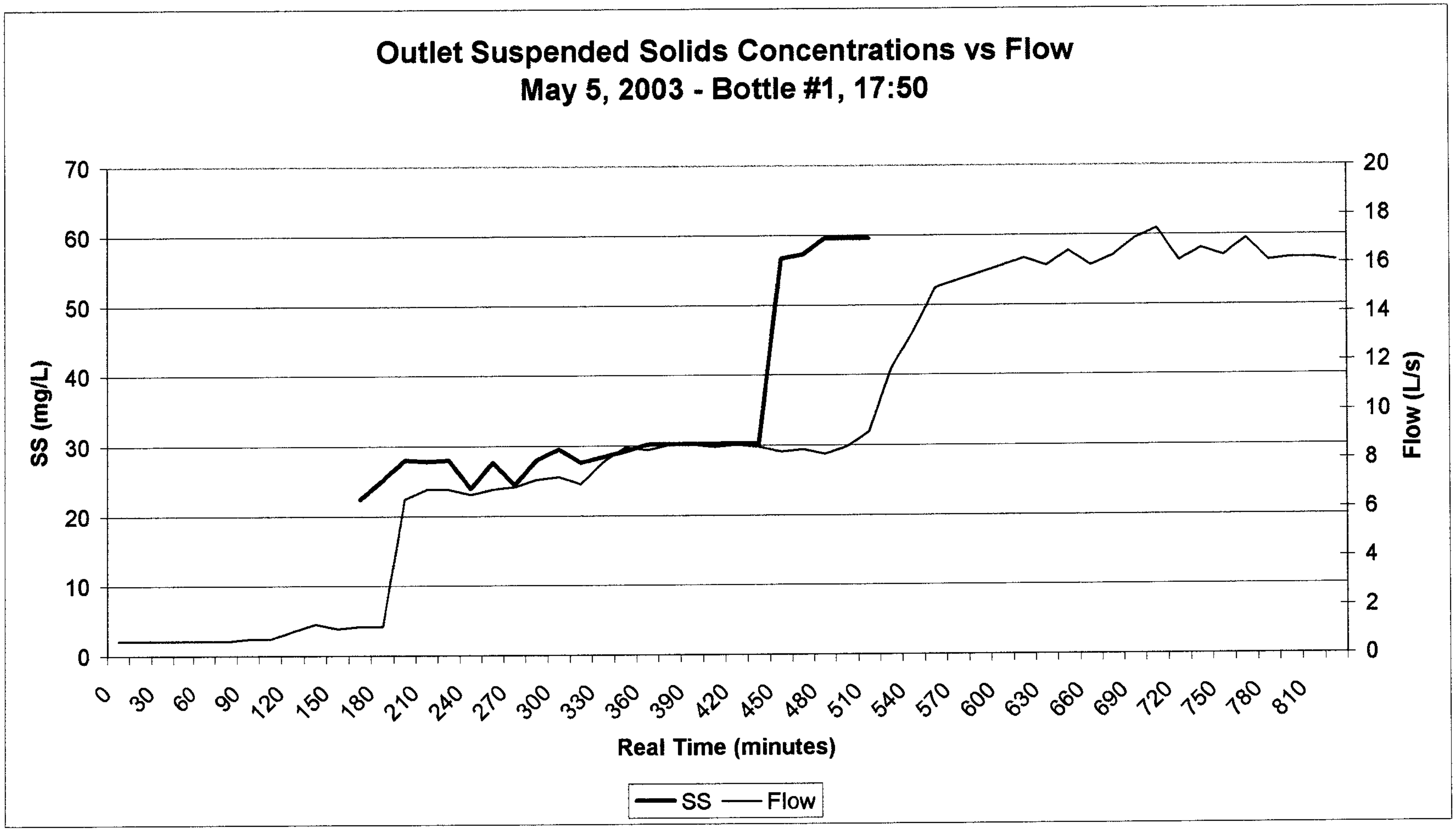


Figure 34: Outlet suspended solids concentrations May 5, 2003

Figure 34 shows the two increases in the hydrograph where the two peaks from the inlets affected the outflow. There is a sharp increase in suspended solids concentration just before the second increase of the hydrograph. Due to the temporal pattern of the outlet hydrograph in relation to the inlet hydrographs, it can be assumed that this sharp increase is a result of the second peak in the inflow hydrograph. Therefore, before the increase in suspended solids occurs, the outflow hydrograph is cut-off, signalling the end of the May 5th event. In addition, suspended solids contributing to the sharp increase seen in Figure 34 are also cut-off from the analysis.

Indeed, interpreting water quality and quantity data is relative to an individual's discretion. The choices made for interpreting hydrographs, inter-event times, and sampling time intervals are examples of how results can vary when analyzing stormwater data.

CHAPTER 6

6.0 CONCLUSIONS AND RECOMMENDATIONS

The scope of the study outlined specific undertakings that were implemented to meet the study's objectives. The scope of this study involved the determination of the event mean concentrations and average concentrations of suspended solids and stormwater pollutants respectively. It also involved the determination of the particle size distribution and settling velocities of particles entering and exiting the pond. Finally, the scope included the characterization of bottom sediment accumulation in relation to the construction activities on site. These tasks were successfully performed and the following conclusions are made.

Event Mean Concentrations and Average Concentrations

- Suspended solids entering and exiting the pond are significantly high, reaching as much as 34,000 mg/L during one event (September 20th, 2002).
- Rainfall intensity, duration and total volume influence the sediment load entering the pond. In addition, the stage in construction activities also affected the sediment load entering the pond.
- Removal efficiencies are significantly high for all events. All events had over 80% concentration based removal efficiencies for the event mean concentrations. Despite these high removal efficiencies, the EMC exiting the pond on some occasions were significantly high, exceeding water quality protection guidelines. In light of this finding, the concentrations exiting the pond should serve as the determining factor of the pond's performance rather than the removal efficiencies.

- Metals and nutrients varied in their removal efficiencies, some at 100% and others with negative removal efficiencies. Certain metals showed strong correlation with suspended solids. This indicates settling as the main process for removing pollutants. However, other metals showed strong correlation with dissolved solids. This may account for the low removal efficiencies for some of the metals. Nutrients shared similar results, indicating a strong relationship with suspended solids. However, the actual concentration exiting the pond exceeded PWQO's for total phosphorus.
- It should be noted that there is several alternative reasons why the pond experienced such high removal efficiencies. For example, suspended sediment entering the pond is predominantly very fine particles. These particles are subject to flocculation processes that encourage the settling of the agglomerated particles. This may account for such high removal efficiencies. However, this assumption may not be accurate depending on the timing of sampling and length of sampling. For example, if the outlet is triggered prematurely, it will collect water quality samples that consist of the displaced water that initially exits the pond at the beginning of an event. The displaced water has been present in the pond since the last event, and has undergone settling. This results in pollutant concentrations that are low in comparison to what the event actually produced. The inflow of water from an event will eventually mix and scour some of the bottom sediments, then exit the pond. This outflow is the portion of the event that is representative. In addition, partial sampling, which was employed during this study, is limited in its representation of the event. For example, the samples collected in the fall season extended over a 2 hr

period. For most events, particularly at the outlet, the hydrograph extended beyond the 2hrs. Thus, a portion of the event is not sampled.

- Hydraulic detention times and drawdown times are significantly low. This indicates the pond was unable to fulfill 48 hr detention time requirement. This may also have an effect on the pond's performance. Low detention time minimizes the pond's settling time required to remove most of the solids and pollutants entering the pond.

Particle Size Distributions and Settling Velocities

- Particle size distributions are indicating that fine sediments dominate the sediment loads entering the pond. These fine sediments increased in the outflow of the pond. The D_{10} and D_{50} particle sizes were significantly lower than the targeted particle size recommended by the MOEE Guidelines to use when sizing the sediment forebay.
- The settling velocities calculated for the D_{10} and D_{50} average particle size using Stoke's Law are very low. In addition, the settling velocities calculated for particles with lower specific gravities are even lower. The settling velocities calculated for the mean D_{50} particle size significantly altered the sizing of the forebay when using the MOEE forebay settling length. This indicates the design criteria needs to target a smaller particle size for construction sediment ponds. The Guidelines are currently targeting the 40 μm particle size. The forebay is designed to trap coarse sediments while the deeper part of the pond is designed for settling of finer particles. If the runoff sediments are of large sizes, the forebay will allow easy maintenance. If the sediments are of fine sizes, the function of the forebay will be insignificant. It may well be better to eliminate the forebay.

- Although eliminating the sediment forebay may improve the settling conditions in a sediment control pond, stormwater management ponds benefit from sediment forebays as they receive runoff with larger particles. Thus, the sediment forebay should be constructed within the pond after the construction activities are complete in the catchment area.

Bottom Sediment Accumulation and Monitoring of Construction Activities

- Bottom sediment accumulation corresponded with the progression in construction activities. In order to maintain the required storage for trapping runoff sediments, pond maintenance should be schedule in accordance to construction stages.
- The bottom sediment accumulation was negligible during the winter months as the pond was completely frozen and received zero runoff.
- The rapid filling of the pond as construction activities progress can affect the performance of the pond.

Overall, the pond achieved high removal efficiency and significantly reduced the pollutant loads entering the facility. However, the concentrations exiting the pond are the main concern. Despite the high removal efficiencies, many of these concentrations exceeded the Province's water quality objectives and have the potential to affect aquatic organisms. Based on the study's findings and the conclusions outlined above, the design criteria recommended by the MOE for stormwater water quality ponds may not be adequate for sediment control ponds. The hydraulic detention time requirement of 48 hrs was not met, and the sediment forebay was undersized according to the particle size distribution found in the construction site runoff. Moreover, the results indicate a significant difference in the runoff characteristics from construction sites when compared to surface runoff from developed areas. Areas undergoing active construction are continuously changing in landscape and differ considerably from developed areas where stormwater management ponds are implemented. For example, the disturbance caused by construction activities is particularly evident in the water quality results at inlet 1070. This inlet produced the significant levels of suspended solid concentrations reaching as high as 34, 000 mg/L. Suspended solid concentrations at inlet 510 reached as high as 509 mg/L. The catchment area serviced by inlet 510 had completed construction at the time of sampling. Indeed, construction activities can alter the landscape and increase erosion processes significantly. Moreover, the water quality results analyzed at the outlet indicate that the construction sediment pond is inconsistent in its ability to reduce suspended solid concentrations to acceptable levels. The pond was designed using the 1994 MOE stormwater design criteria. These criteria require that stormwater management facilities reduce total suspended solid concentrations by 80%. This research has proven that an 80% reduction does not guarantee an acceptable range of total suspended solid concentrations exiting the pond. Thus, the 1994 MOE design criteria concerning detention times, sediment forebay sizing, and total suspended solids removal (permanent pool volume), may not have provided an adequate pond design needed to protect downstream habitat from the impacts of erosion at construction sites.

Indeed, construction site runoff differs considerably from stormwater runoff over developed areas. If the runoff characteristics of the Richmond Hill site are typical of most construction sites then design criteria for sediment control ponds need to take into consideration the smaller particle size that dominates the distribution found in runoff. In addition, the criteria should also consider extending the hydraulic detention times as the pond will be receiving smaller particle sizes and therefore will require longer settling times. Moreover, physical settling of construction sediments has its own limitations. In order to protect aquatic ecosystem from construction sediments, non-conventional techniques such as chemical assisted precipitation may be necessary.

Although the construction site implemented erosion control practices (i.e. mud mats, sediment shield and snow fence, and filter fabric sediment traps) the site experienced considerable erosion rates which in turn increased the dependence on the sediment control pond for the protection of downstream habitat. Because the pond was inconsistent in reducing suspended solid concentrations, a greater importance should be stressed on the control of erosion processes at the site. The sediment control pond is an end-of-pipe facility. If this type of facility fails, or is unable to perform optimally under all conditions, then the level of protection is diminished. Thus, by improving erosion control practices at the site rather than depending on the sediment control facility, the potential to decrease the detrimental impacts from construction activities will increase.

The results produced from this study demonstrate the benefits and limitations of sediment control ponds. It is important to note that these results are relative to the methods used to collect and analyze the data. When implementing the monitoring program for the Richmond Hill study several issues became apparent in collecting the samples and analyzing the data. These issues were examined in the discussion section of this study. Based on the discussion section and experiences gained through the implementation of the monitoring program, the following conclusions and recommendations can be made.

- Setting the samplers to collect at the appropriate time is a crucial element in gaining representative results for any study. This can be influenced by the type of equipment used, positioning of the equipment, storm characteristics, depression storage, etc. Thus, it is important to gain an overall understanding of the site characteristics, and to become familiarized with the equipment used. By including a sufficient trial and error period in the study timeline, many of the issues associated with the methodology can be managed.
- Under certain conditions, analyzing the event hydrographs and hyetographs can be difficult. It became apparent that when a back-to-back event occurred, it was unclear where the outflow from one event ended, and where the next event began. Thus, it is important to develop a standard method when analyzing storm events. The method should consider factors such as the pollutograph, detention times, and pond lag times when determining the start and end times of an event.
- It is recommended to initially develop a greater understanding of the hydraulic processes of the pond under study to minimize the errors that can occur in the timing of sampling and analysis of the hydrographs. For example, if the hydraulic residence time was determined, the monitoring program may set the outlet sampler to trigger at a more appropriate time. Thus, a greater understanding of the performance of the pond will ensue.

Overall, the monitoring program was implemented with considerable success. Events were collected and analyzed; goals and objectives were met. However, several issues were raised during the implementation of monitoring program. These issues revealed how sensitive the results are to the methodology used to collect and analyze the data. Ideally, an ultimate monitoring program would include the continuous collection of water quality samples and flow measurements. However, for obvious reasons this is unrealistic. Thus, when implementing a monitoring program it is important to be aware of the limitations and issues that can affect the results and how to develop the program to minimize these concerns.

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KEN=42781982&RequestTimeout=600&LocateBy=&StationName=Toronto&SearchType=BeginsWith&Province=

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Appendix A

PAH's, Herbicides and Pesticides included in Spring Composite Samples

Polynuclear Aromatic Hydrocarbons	PWQO (μg/L)
Napthalene	7.0
2-methylnapthalene	2.0
1-methylnapthalene	2.0
2-chloronapthalene	0.2
Acenapthylene	--
Fluroene	0.2
Phenanthrene	0.03
Anthracene	0.0008
Fluoranthene	0.0008
Pyrene	--
Benzo(a)anthracene	0.0004
Chrysene	0.0001
Benzo(b)fluoranthene	--
Benzo(k)fluoranthene	0.0002
Benzo(a)pyrene	--
Dibenzo(a,h)anthracene	0.002
Benzo(g,h,i)perylene	0.00002
1-chloronapthalene	0.1
Perylene	0.00007
Indole	--
5-nitroacenapthene	--
Biphenyl	0.2
Herbicides and Pesticides	
2,4-dichlorophenol	0.2

2,4,6,-trichlorophenol	18.0
2,4,5-trichlorophenol	18.0
2,3,4-trichlorophenol	18.0
2,3,4,5-tetrachlorophenol	1.0
2,3,4,6-tetrachlorophenol	1.0
Pentachlorophenol	0.5
Silvex	--
Bromoxynil	--
Picloram	--
Dicambia	200.0
2,4-D-propionic acid	--
2,4-D	4.0
2,4,5-T	--
2,4-DB	--
Dinoseb	--
Diclofop-methyl	--

Appendix B

Analytical Procedures, Ontario Ministry of the Environment, Laboratory Services

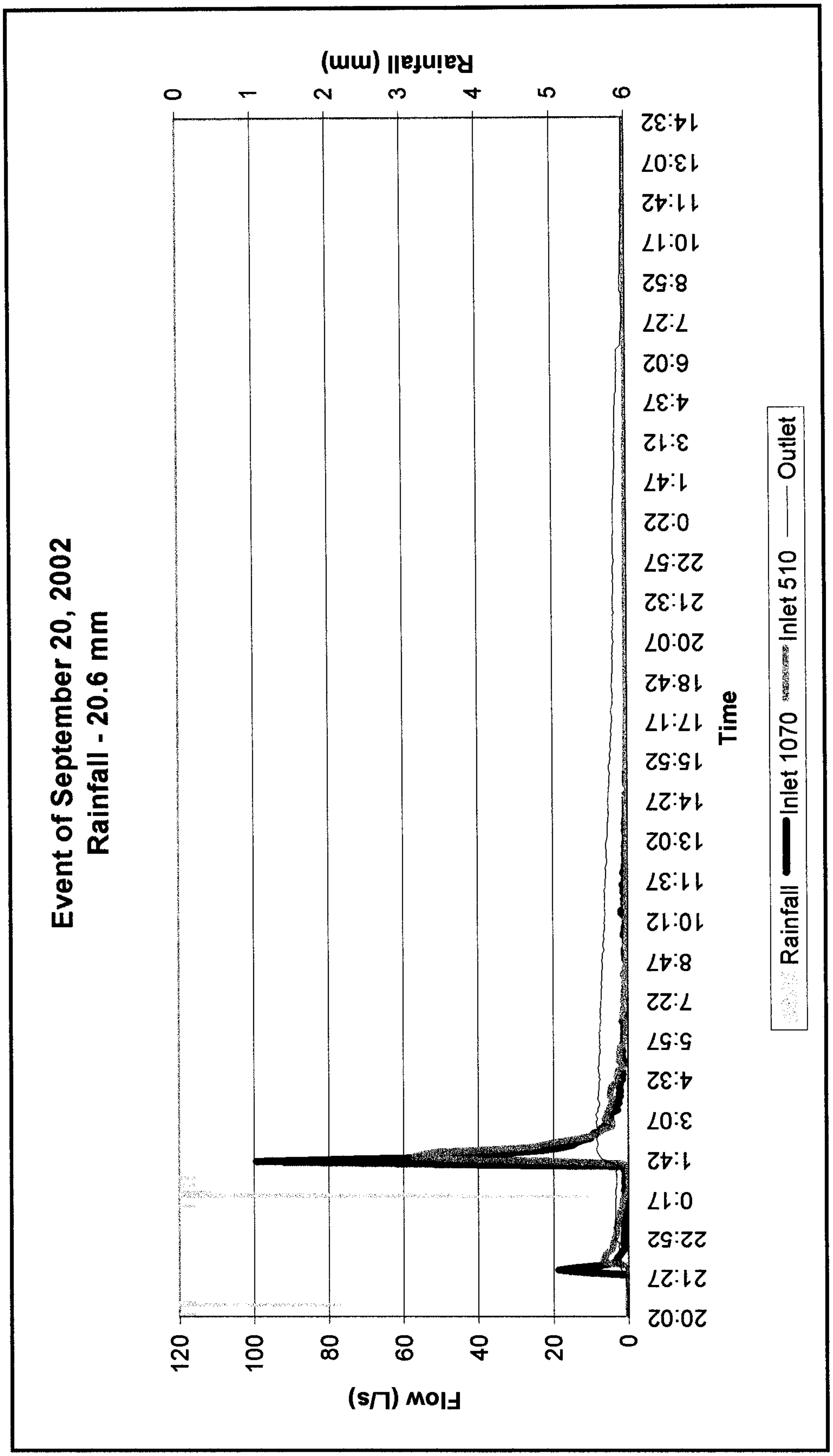
	Parameter	Method
General Chemistry	Chloride	Colourimetry following two-stage reaction with mercuric thiocyanite and ferric iron
	Conductivity, pH, Alkalinity	Automated system using electrodes in a constrant temperature bath for conductivity, a calibrated potentiometric system for pH and titration for TFE alkalinity (to an end-point of pH 4.5).
	Silicon: reactive silicate; Dissolved organic carbon; and Dissolved inorganic carbon	Molybdate reactive silicates: dissolved reactive silicate ions are measured through the formation of molybdenum heteropoly blue complex Dissolved inorganic carbon (+ carbon dioxide) are measured by acidifying the sample supernatant, extracting the CO_2 through a dialysis membrane and reacting it with phenolphthalein and colourimetric measurement Organic carbon is measrued in the sample supernatant by acidification followed by nitrogen flushing to remove inorganic carbon and UV digestion in an acid-persulphate medium. The resulting CO_2 is analyzed as above

	Solids: Suspended Dissolved Total	Total Solids: refers to the material (residue) which remains after an aliquot of well-mixed sample is transferred into a tared dish and evaporated to dryness (20 hrs minimum) at 102 ± 2° C. Suspended Solids refers to all material (residue, particulate) which is removed from a sample when a well-mixed sample aliquot is filtered through a 1.5 to 2.0 µm glass fibre filter. The material on the filter is dried at 103 ± 2°C. Dissolved solids refers to the material (residue, filtrate) which remains in solution when a well-mixed sample is filtered through a 1.5 – 2.0 µm glass fibre filter. An aliquot of the filtrate (50 or 100 mL) is transferred, using a transfer pipette, into a tared dish and evaporated to dryness (20 hrs minimum), at 103 ± 2°C.
	Chemical Oxygen Demand	Samples are mixed with an acidified Potassium Dichromate Solution which contains mercuric sulphate to suppress chloride interference. Concentrated sulphuric acid containing silver sulphate as a catalyst is added and the mixture is digested in a mechanical-convection oven for 3 hrs at 149 ± 1°C. Analysis is completed by automated colourimetric measurement of trivalent chromium.
	Particle Size	Optical – laser light diffraction (Coulter LS130 Particle Size Analyzer) This method is capable of analyzing from 0.1 to 900 µm in 27 size channels. However, it is reported as a percent distribution by volume (no count data).
	Turbidity	Measurement of light scattering at 90° ± 30° by nephelometry calibrated to Formazin turbidity standards

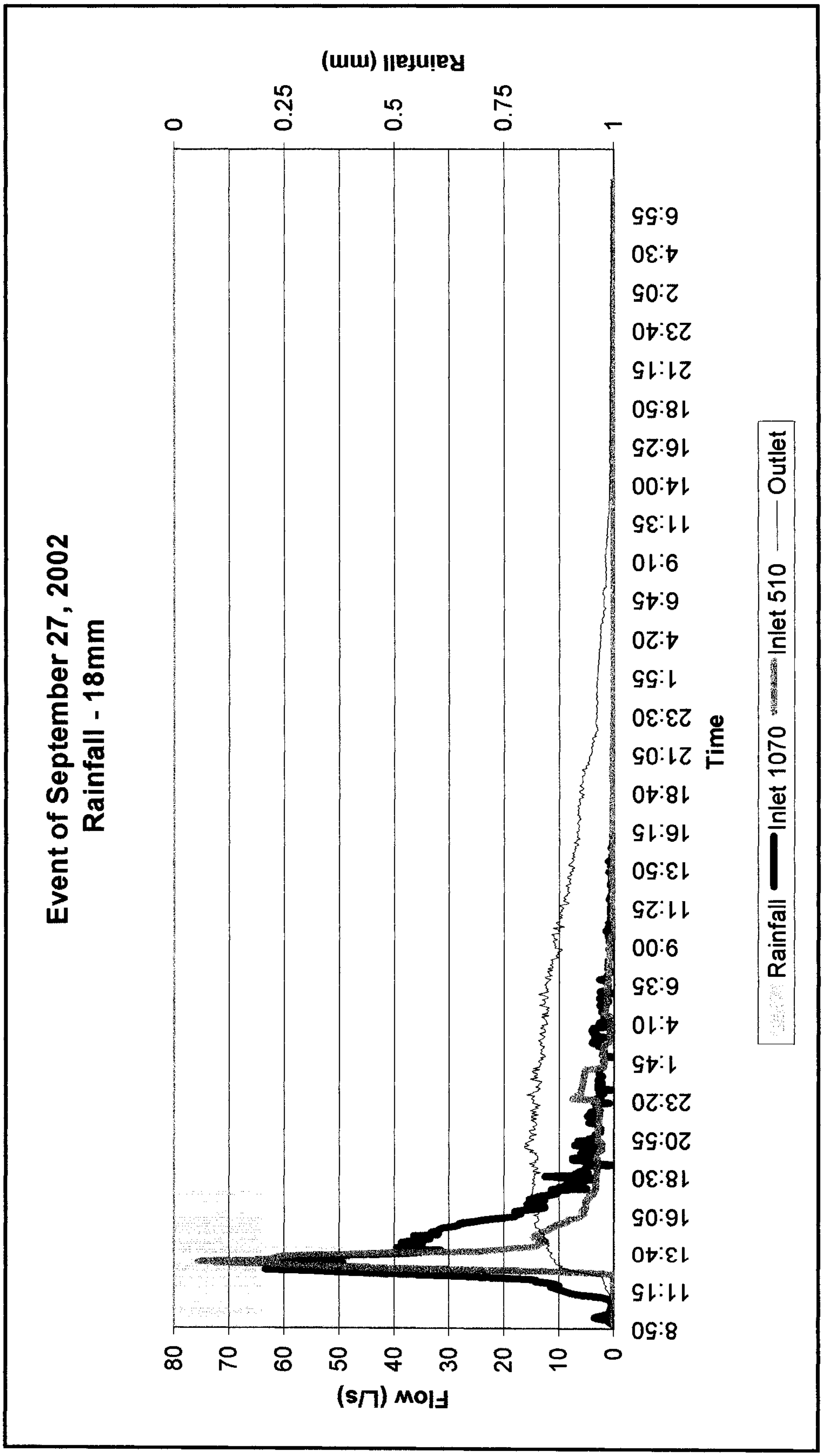
Metals	Total Metals: Aluminum, Barium, Beryllium, Calcium, Cadmium, Cobalt, Chromium, Copper, Iron, Magnesium, Manganese, Molybdenum, Nickel, Lead, Strontium, Titanium, Vanadium, and Zinc	Inductively coupled plasma (ICP) following ultrasonic nebulizer
Nutrients	Total nutrients: Total P TKN	Total P: digestion in sulphuric acid, mercuric sxice, potassium sulphate media followed by reduction with ascorbic acid – measured a orthophosphate Total Kjeldahl Nitrogen: digestion with Kjeldahl's reagent, neutralization and analysis for ammonia species by colourimetry
	Dissolved Nutrients: Ammonia+ Ammonium Nitrite Nitrate+nitrite Phosphate	Simultaneous, automated analysis of one aliquot of sample: • Ammonia by conversion to indophenol blue with sodium nitroprusside as a catalyst • Nitrite by colourimetric method after reaction with sulphanilamide and N (1-napthyl) ethylenediamine dihydrochloride • Nitrate + nitrite by colourimetric method following conversion of nitrate to nitrite • Phosphours, as orthophosphate, by colourimetric method following reaction with ascorbic acid

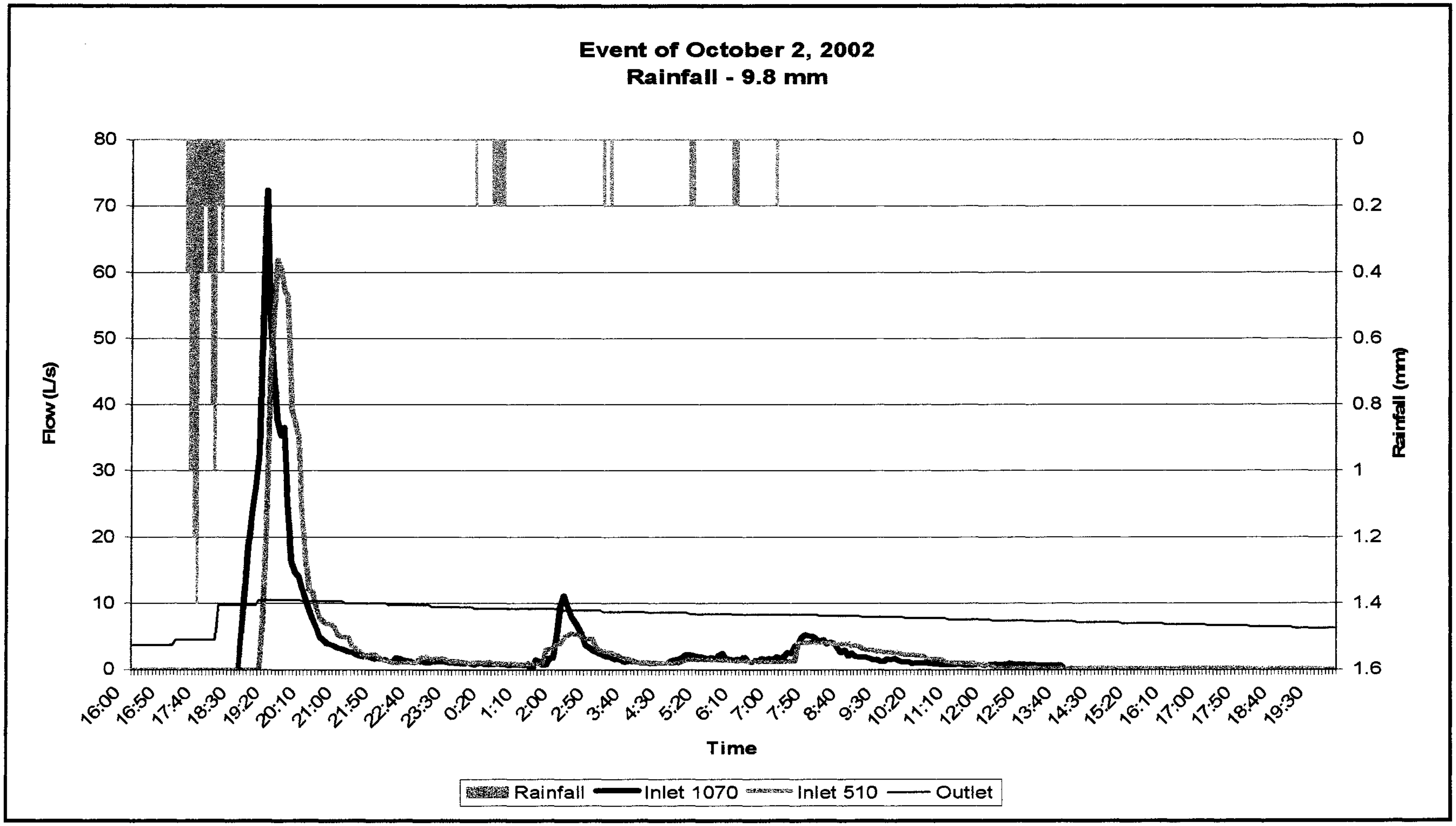
Appendix C

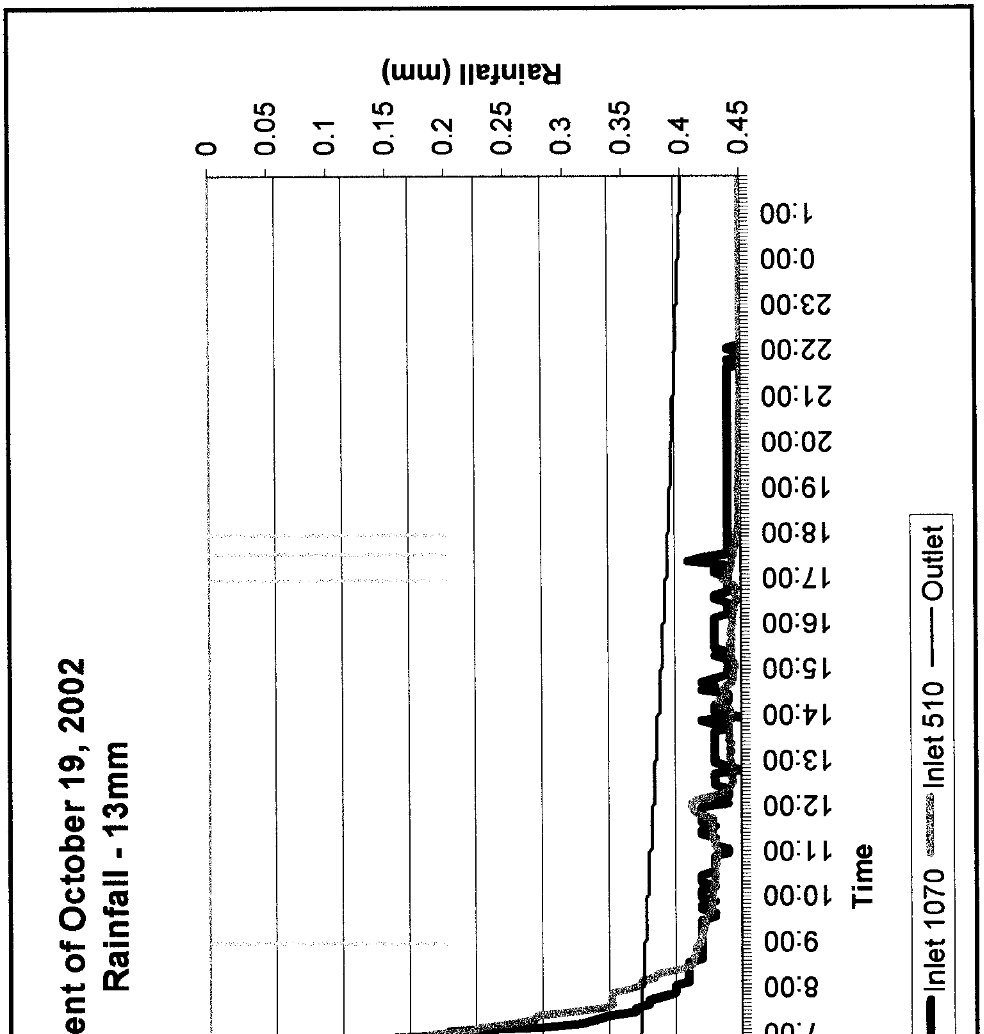
Hydrological Data for Events Monitored

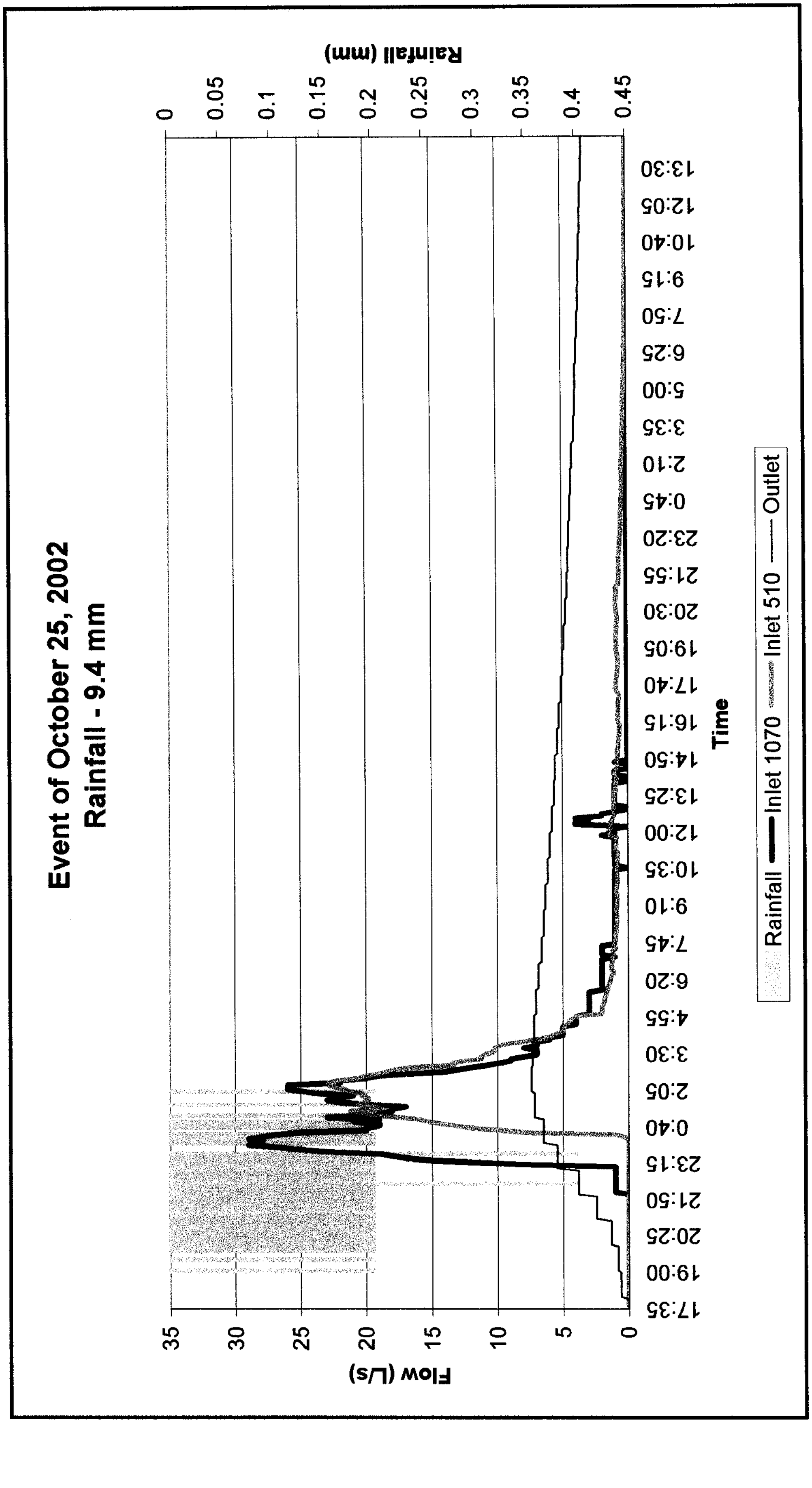


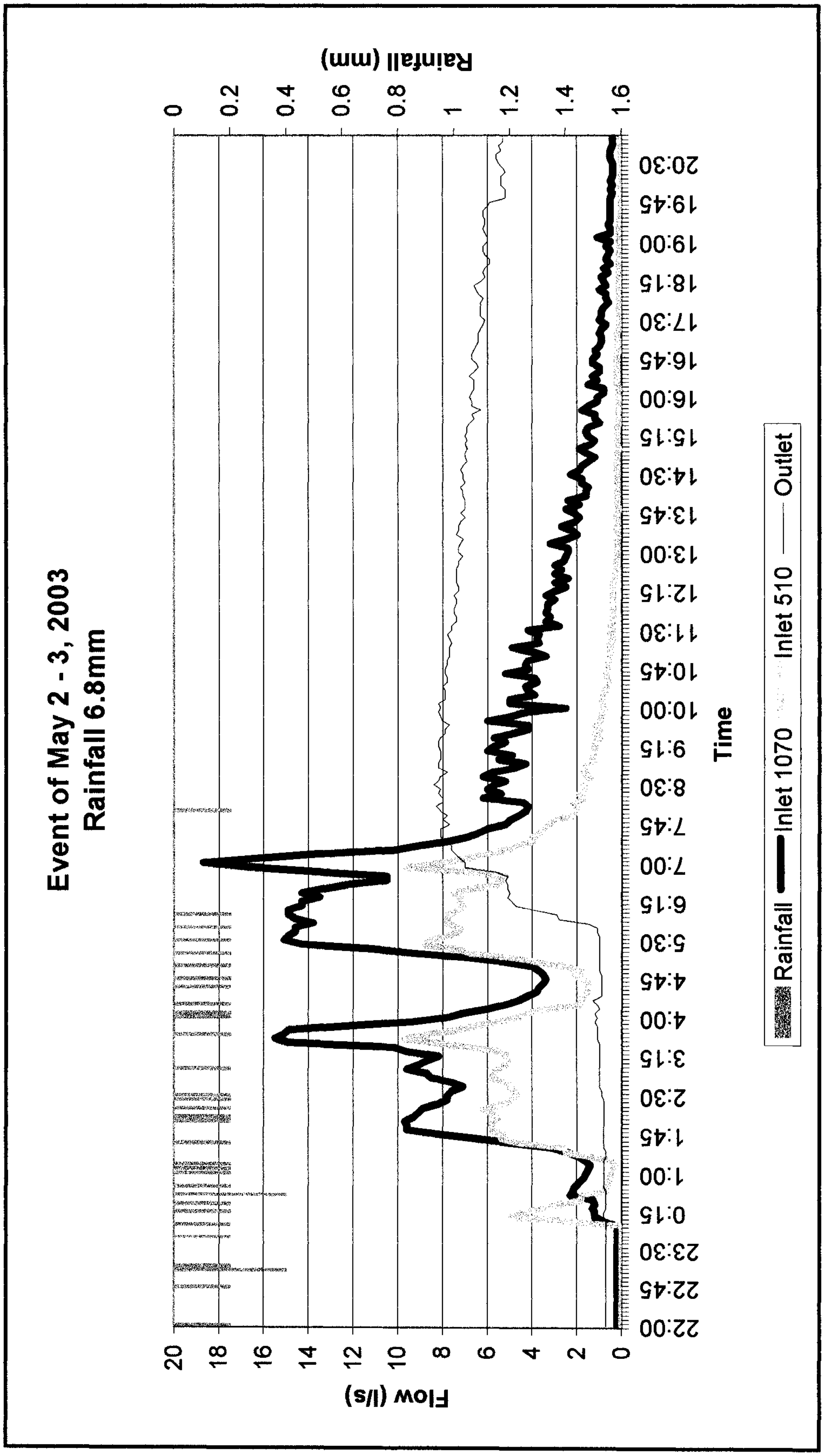
Event of September 20, 2002
Rainfall - 20.6 mm
Flow (L/s)
Rainfall (mm)
Time
120
100
80
60
40
20
0
1
2
3
4
5
6
20:02
21:27
22:52
0:17
1:42
3:07
4:32
5:57
7:22
8:47
10:12
11:37
13:02
14:27
15:52
17:17
18:42
20:07
21:32
22:57
0:22
1:47
3:12
4:37
6:02
7:27
8:52
10:17
11:42
13:07
14:32
Rainfall
Inlet 1070
Inlet 510
Outlet

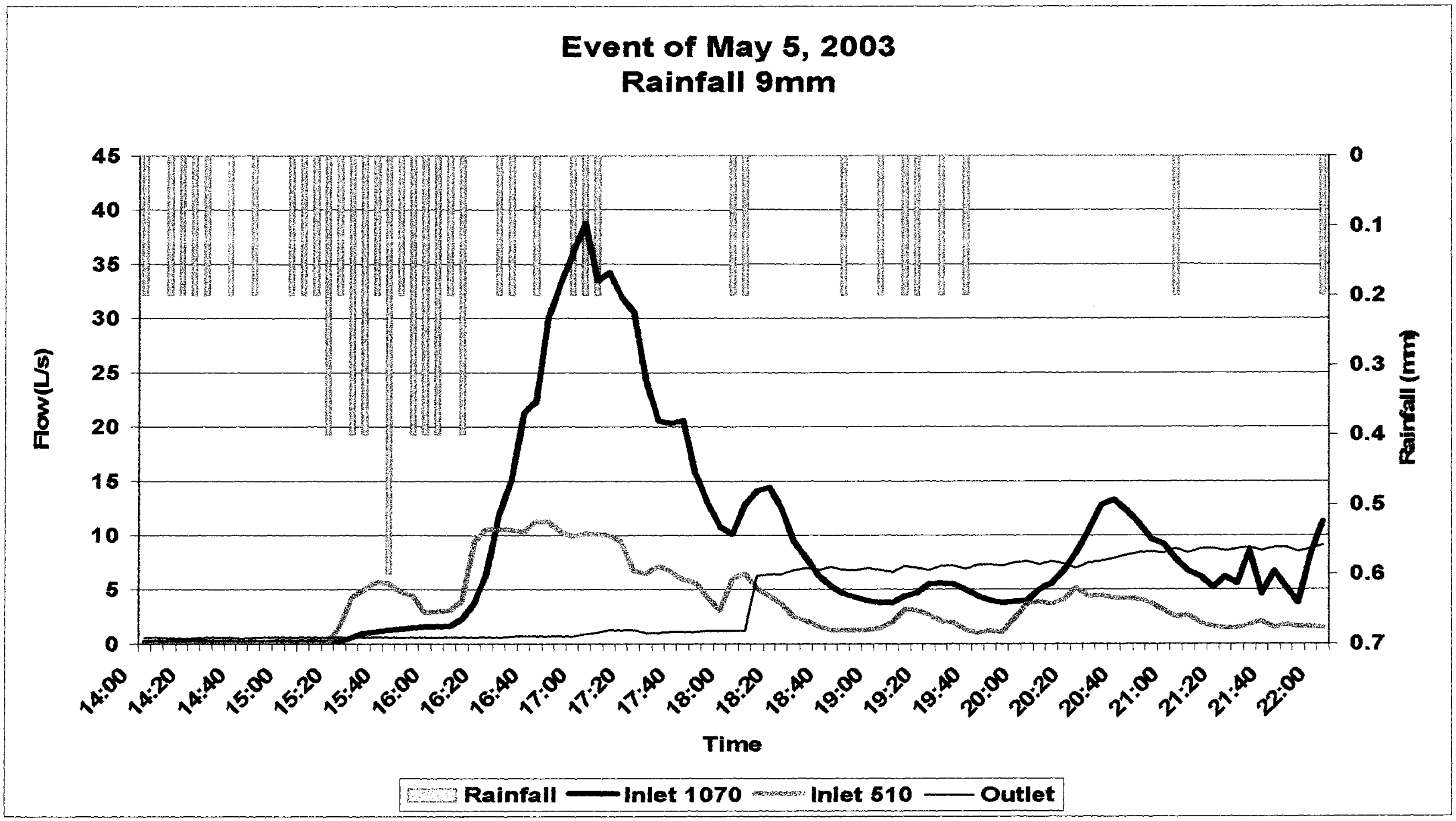

Event of September 27, 2002
Rainfall - 18mm
Flow (L/s)
Rainfall (mm)
Time
80
70
60
50
40
30
20
10
0
0
0.25
0.5
0.75
1
8:50
11:15
13:40
16:05
18:30
20:55
23:20
1:45
4:10
6:35
9:00
11:25
13:50
16:15
18:40
21:05
23:30
1:55
4:20
6:45
9:10
11:35
14:00
16:25
18:50
21:15
23:40
2:05
4:30
6:55
Rainfall
Inlet 1070
Inlet 510
Outlet

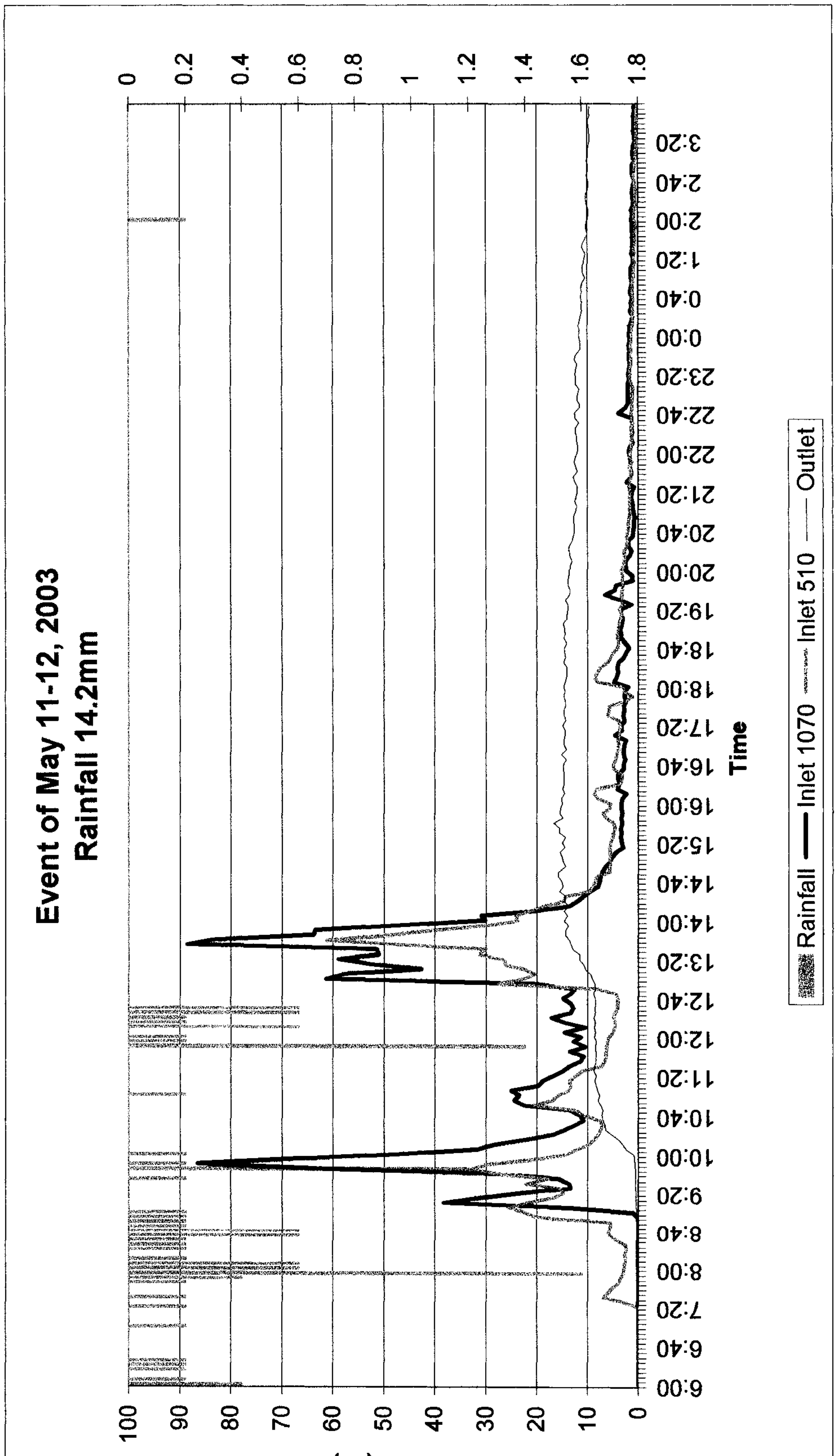

Event of October 2, 2002
Rainfall - 9.8 mm
Flow (L/s)
Rainfall (mm)
Time
Rainfall
Inlet 1070
Inlet 510
Outlet

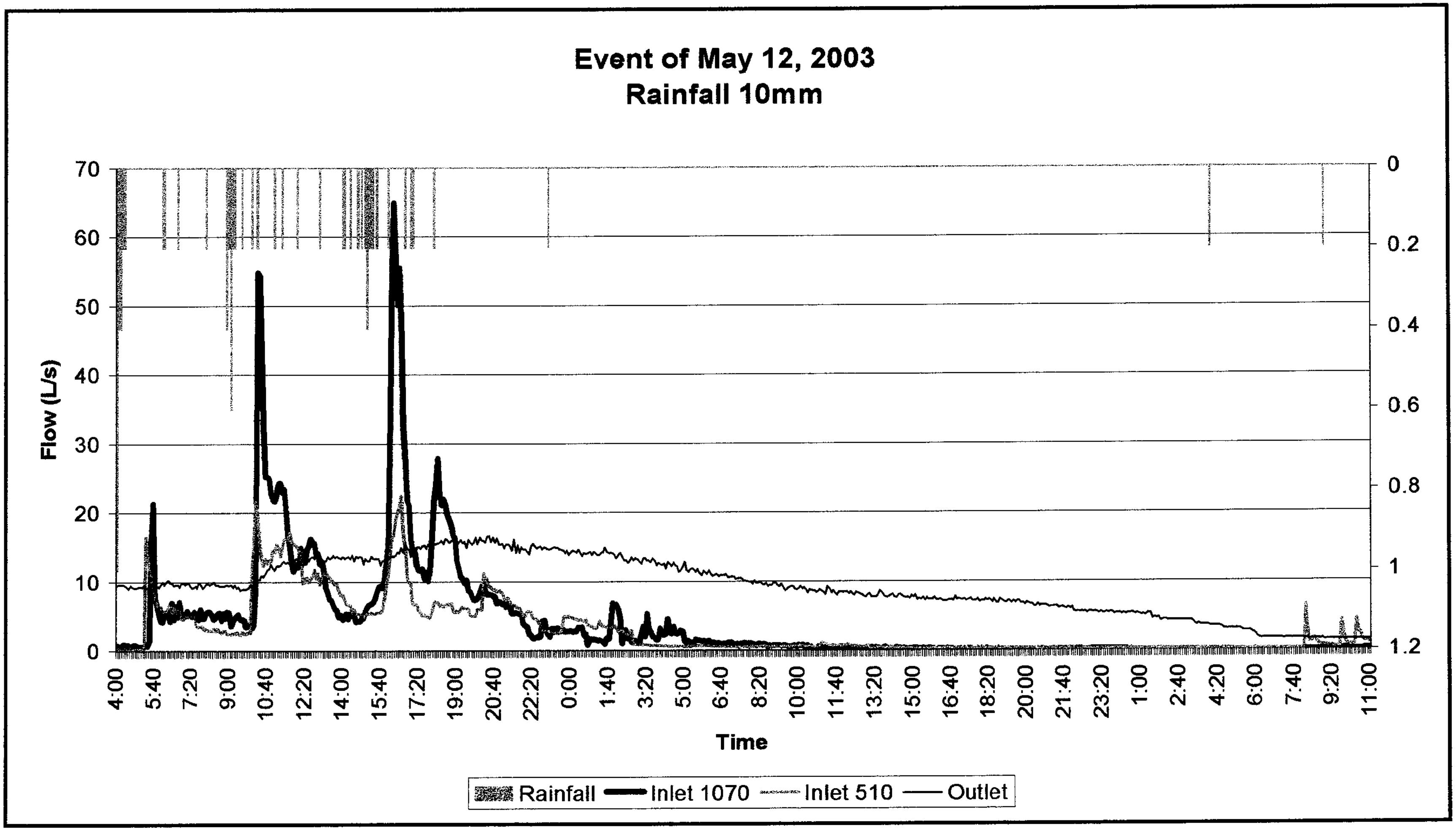

Event of October 19, 2002
Rainfall - 13mm
Flow (L/s)
0
5
10
15
20
25
30
35
40
Rainfall (mm)
0
0.05
0.1
0.15
0.2
0.25
0.3
0.35
0.4
0.45
19:00
20:00
21:00
22:00
23:00
0:00
1:00
2:00
3:00
4:00
5:00
6:00
7:00
8:00
9:00
10:00
11:00
12:00
13:00
14:00
15:00
16:00
17:00
18:00
19:00
20:00
21:00
22:00
23:00
0:00
1:00
Time
Rainfall
Inlet 1070
Inlet 510
Outlet

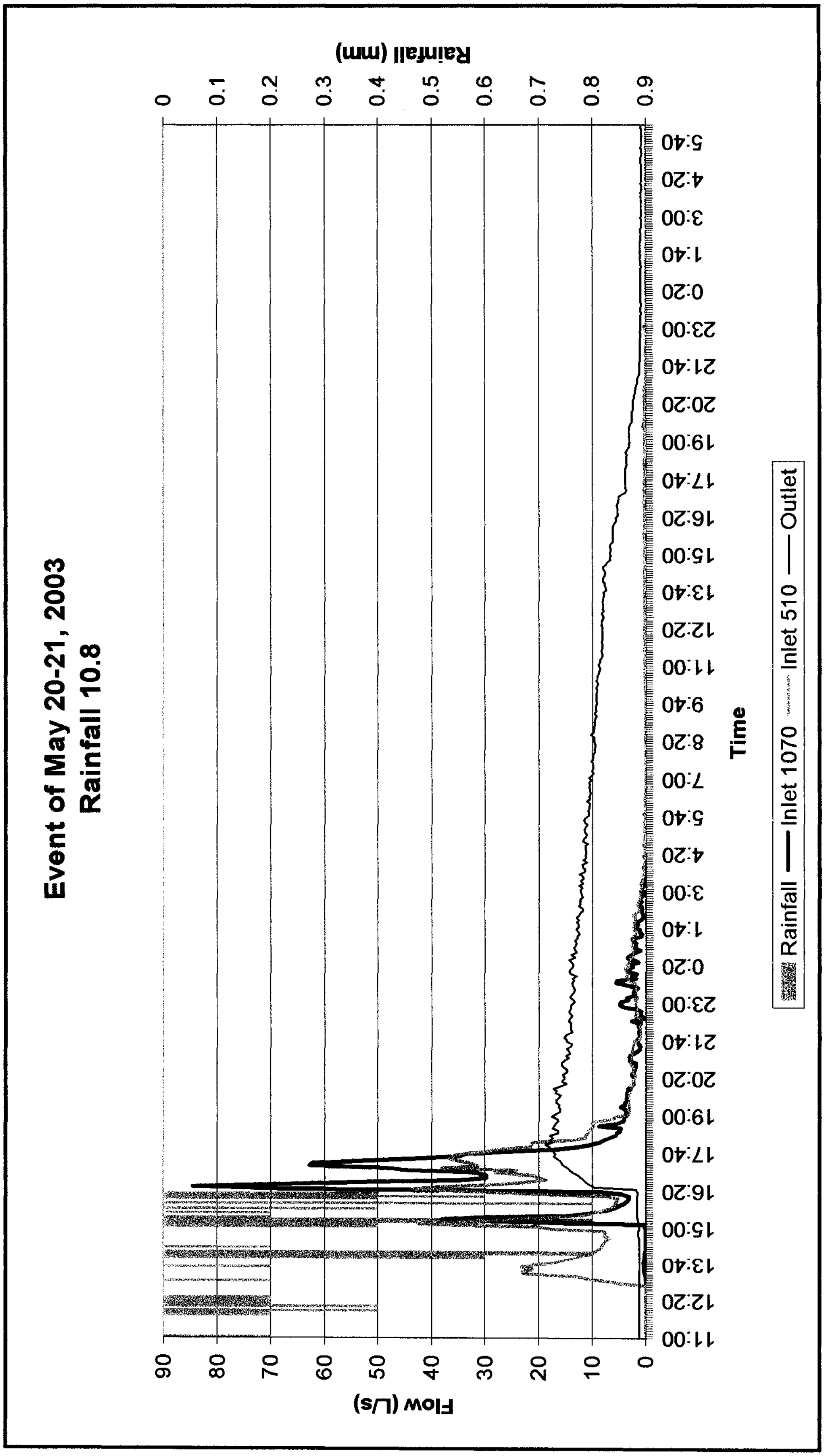

Event of October 25, 2002
Rainfall - 9.4 mm
Flow (L/s)
Rainfall (mm)
Time
Rainfall
Inlet 1070
Inlet 510
Outlet
35
30
25
20
15
10
5
0
0.05
0.1
0.15
0.2
0.25
0.3
0.35
0.4
0.45
17:35
19:00
20:25
21:50
23:15
0:40
2:05
3:30
4:55
6:20
7:45
9:10
10:35
12:00
13:25
14:50
16:15
17:40
19:05
20:30
21:55
23:20
0:45
2:10
3:35
5:00
6:25
7:50
9:15
10:40
12:05
13:30


Event of May 2 - 3, 2003
Rainfall 6.8mm
Flow (l/s)
20
18
16
14
12
10
8
6
4
2
0
Rainfall (mm)
0
0.2
0.4
0.6
0.8
1
1.2
1.4
1.6
22:00
22:45
23:30
0:15
1:00
1:45
2:30
3:15
4:00
4:45
5:30
6:15
7:00
7:45
8:30
9:15
10:00
10:45
11:30
12:15
13:00
13:45
14:30
15:15
16:00
16:45
17:30
18:15
19:00
19:45
20:30
Time
Rainfall
Inlet 1070
Inlet 510
Outlet


Event of May 5, 2003
Rainfall 9mm
Flow(L/s)
45
40
35
30
25
20
15
10
5
0
Rainfall (mm)
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
14:00
14:20
14:40
15:00
15:20
15:40
16:00
16:20
16:40
17:00
17:20
17:40
18:00
18:20
18:40
19:00
19:20
19:40
20:00
20:20
20:40
21:00
21:20
21:40
22:00
Time
Rainfall
Inlet 1070
Inlet 510
Outlet


Event of May 11-12, 2003
Rainfall 14.2mm
Flow (l/s)
100
90
80
70
60
50
40
30
20
10
0
0
0.2
0.4
0.6
0.8
1
1.2
1.4
1.6
1.8
6:00
6:40
7:20
8:00
8:40
9:20
10:00
10:40
11:20
12:00
12:40
13:20
14:00
14:40
15:20
16:00
16:40
17:20
18:00
18:40
19:20
20:00
20:40
21:20
22:00
22:40
23:20
0:00
0:40
1:20
2:00
2:40
3:20
Time
Rainfall
Inlet 1070
Inlet 510
Outlet


Event of May 12, 2003
Rainfall 10mm
Flow (L/s)
70
60
50
40
30
20
10
0
0
0.2
0.4
0.6
0.8
1
1.2
4:00
5:40
7:20
9:00
10:40
12:20
14:00
15:40
17:20
19:00
20:40
22:20
0:00
1:40
3:20
5:00
6:40
8:20
10:00
11:40
13:20
15:00
16:40
18:20
20:00
21:40
23:20
1:00
2:40
4:20
6:00
7:40
9:20
11:00
Time
Rainfall
Inlet 1070
Inlet 510
Outlet


Event of May 20-21, 2003
Rainfall 10.8
Flow (L/s)
Rainfall (mm)
Time
Rainfall
Inlet 1070
Inlet 510
Outlet

Appendix D

Outlet Water Quality Analysis vs. PWQO

% Removal efficiencies, outlet concentrations, and PWQO's for September 2002

Parameter	PWQO	14-Sep-02		20-Sep-02		27-Sep-02	
		Outlet	% Removal	Outlet	% Removal	Outlet	% Removal
COD (mg/L)		29.0	78.8	25.0	83.9		
Total Phosporus (mg/L)	0.03	0.3	93.6	0.1	98.7	0.1	100.0
TKN (mg/L)		1.3	75.2	0.8	89.0	0.7	88.4
Al (ug/L)		2560.0	-38.8	388.0	-56.4	799.0	34.8
Ba (ug/L)		43.2	73.3	15.3	94.4	15.8	89.2
Be (ug/L)	1100	0.2	2.8	0.0	71.1	0.0	45.8
Ca (mg/L)		81.4	83.9	42.2	90.3	44.8	89.4
Cd (ug/L)	0.5	-0.1	100.0	0.2	20.5	0.2	-94.8
Co (ug/L)	0.9	3.0	-181.9	0.5	67.4	0.1	84.8
Cr (ug/L)	1.0 / 100*	4.6	-6.3	2.9	64.6	4.8	40.2
Cu (ug/L)	5	8.4	29.5	10.9	86.8	14.8	-28.6
Fe (ug/L)	300	2840.0	-41.2	429.0	-105.9	891.0	34.6
Mg (mg/L)		6.0	55.6	4.0	68.1	3.6	70.8
Mn (ug/L)		256.0	68.3	27.7	89.1	44.1	94.8
Mo (ug/L)	40	-0.1	100.0	1.6	80.1	0.7	100.0
Ni (ug/L)	25	6.1	10.8	1.9	37.3	2.0	46.5
Pb (ug/L)	5	-1.5	100.0	1.2	54.3	3.1	43.1
Sr (ug/L)		273.0	73.5	205.0	75.8	228.0	76.3
Ti (ug/L)		7.6	5.6	7.7	-134.7	10.7	-9.4
V (ug/L)	6	9.6	-5.2	4.4	42.8	6.9	-35.2
Zn (ug/L)	20	21.3	-34.7	5.1	44.7	6.8	49.3

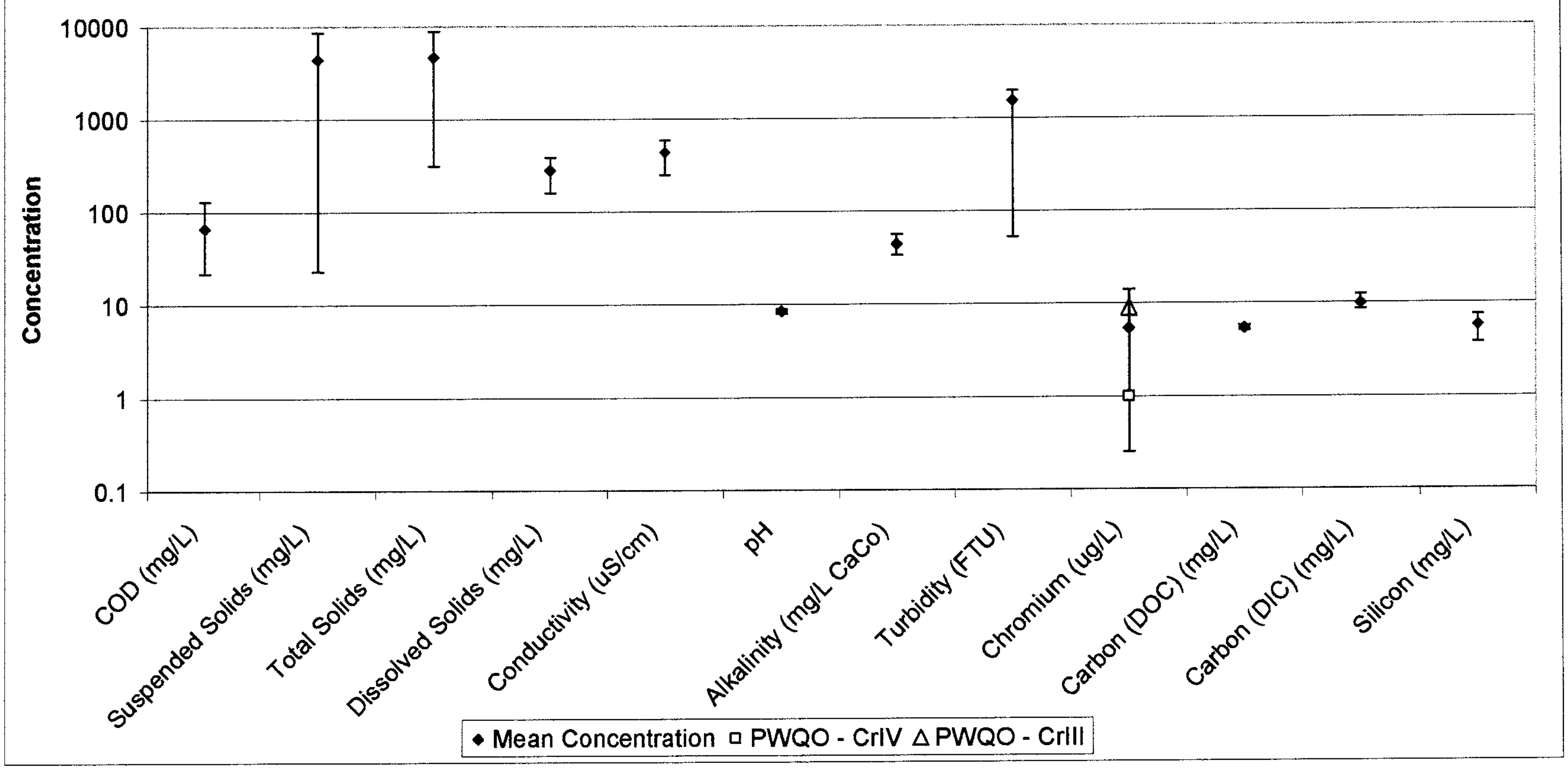
% Removal efficiencies, outlet concentrations, and PWQO's for October, 2002.

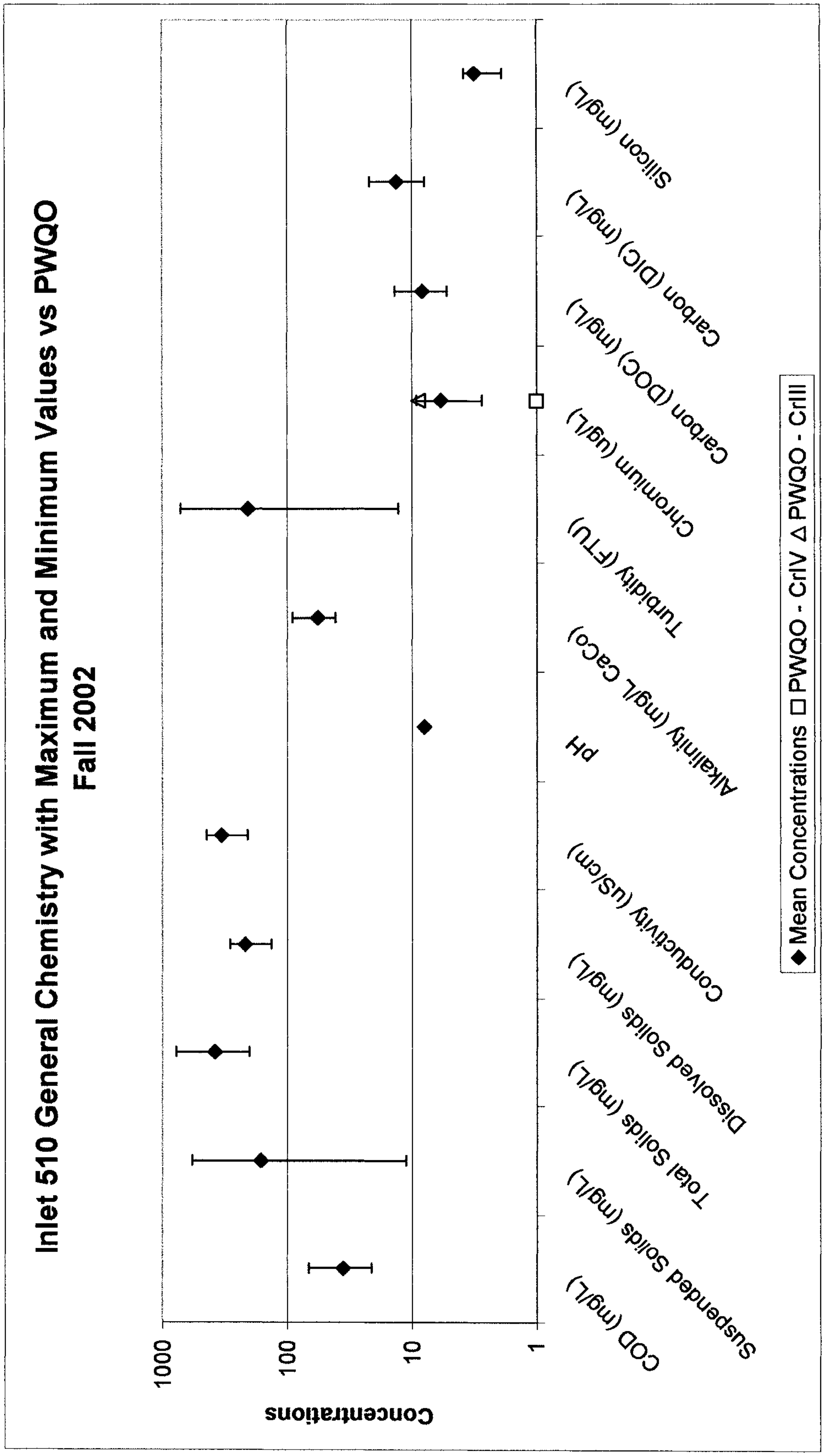
Parameter	PWQO	02-Oct-02		19-Oct-02		25-Oct-02	
		Outlet	% Removal	Outlet	% Removal	Outlet	% Removal
COD (mg/L)		18.0	-4.1	26.0	61.6	20.0	50.0
Suspended Solids (mg/L)		8.7	99.2	134.0	97.5	18.9	44.0
Dissolved Solids (mg/L)		230.0	58.1	287.0	62.5	281.0	47.4
Total Solids (mg/L)		239.0	91.0	421.0	84.8	300.0	47.2
Total Phosporus (mg/L)	0.03	0.1	84.8	0.2	91.7	0.1	100.0
TKN (mg/L)		0.7	23.1	1.1	75.4	0.7	-32.6
Al (ug/L)		196.0	-2032.4	1550.0	88.1	582.0	43.7
Ba (ug/L)		10.5	48.5	34.0	89.1	14.8	44.3
Be (ug/L)	1100	0.0	-318.7	0.1	56.7	0.0	30.0
Ca (mg/L)		42.0	79.4	67.4	81.9	50.4	43.8
Cd (ug/L)	0.5	-0.1	0.0	0.0	0.0	0.1	11.1
Co (ug/L)	0.9	-2.0	-542.7	1.7	96.0	0.4	73.4
Cr (ug/L)	1.0/100*	3.9	-87.9	9.2	72.5	6.3	47.5
Cu (ug/L)	5	4.3	-231.9	6.5	75.9	3.4	39.7
Fe (ug/L)	300	145.0	-3675.0	1720.0	88.2	556.0	30.9
Mg (mg/L)		3.9	35.0	6.8	79.0	5.5	45.1
Mn (ug/L)		9.0	50.3	106.0	95.1	29.2	12.1
Mo (ug/L)	40	2.0	100.0	0.1	-812.5	1.4	57.8
Ni (ug/L)	25	0.7	-193.7	3.5	87.0	1.5	31.1
Pb (ug/L)	5	3.1	-217.9	4.0	77.4	2.8	-87.3
Sr (ug/L)		254.0	71.2	405.0	74.0	345.0	46.2
Ti (ug/L)		3.4	-359.0	12.9	6.1	10.3	52.3
Va (ug/L)	6	4.8	-114.4	8.8	69.9	6.0	42.6
Zn (ug/L)	20	4.8	-1069.1	15.8	87.3	4.2	28.0

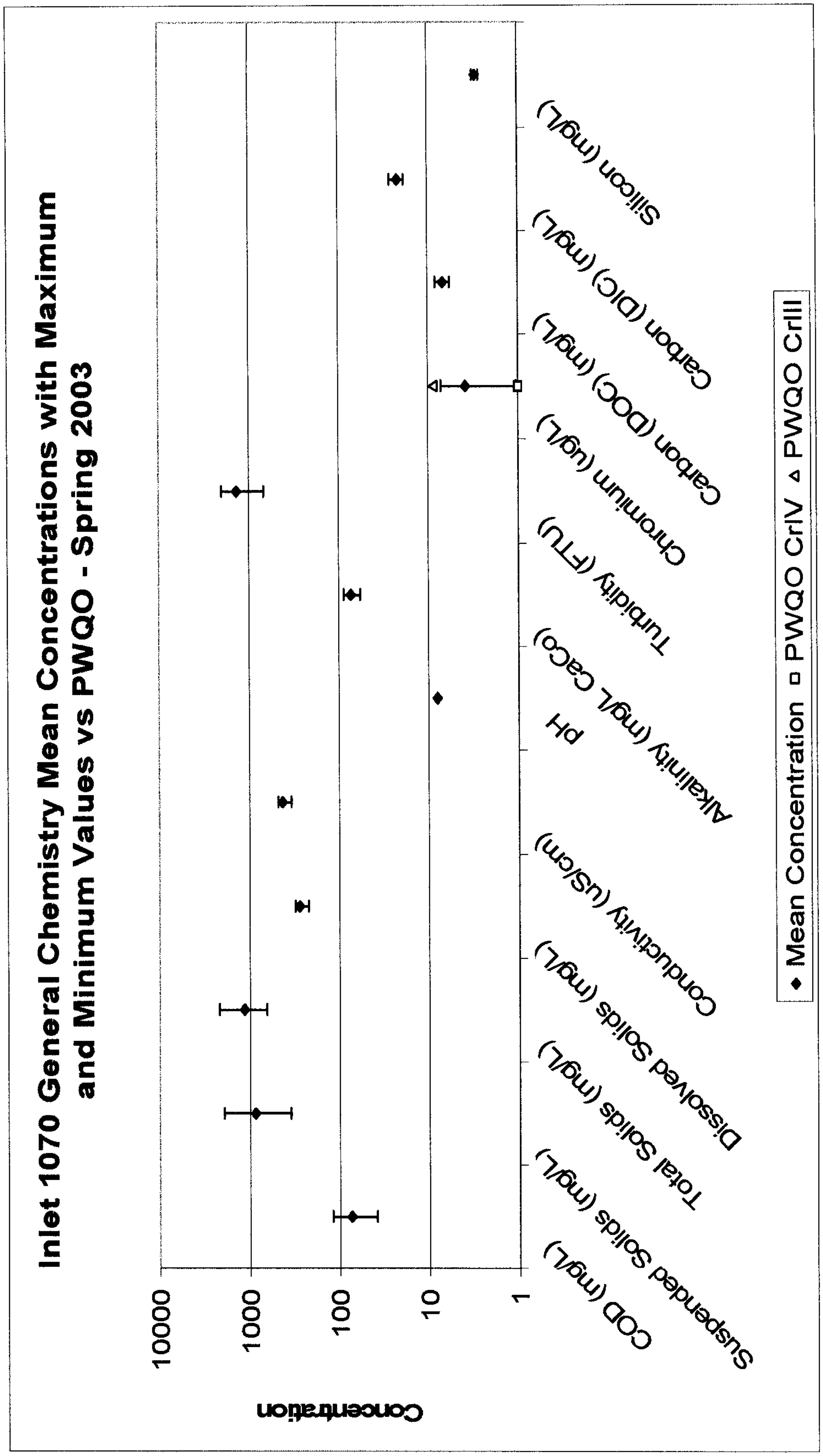
Appendix E

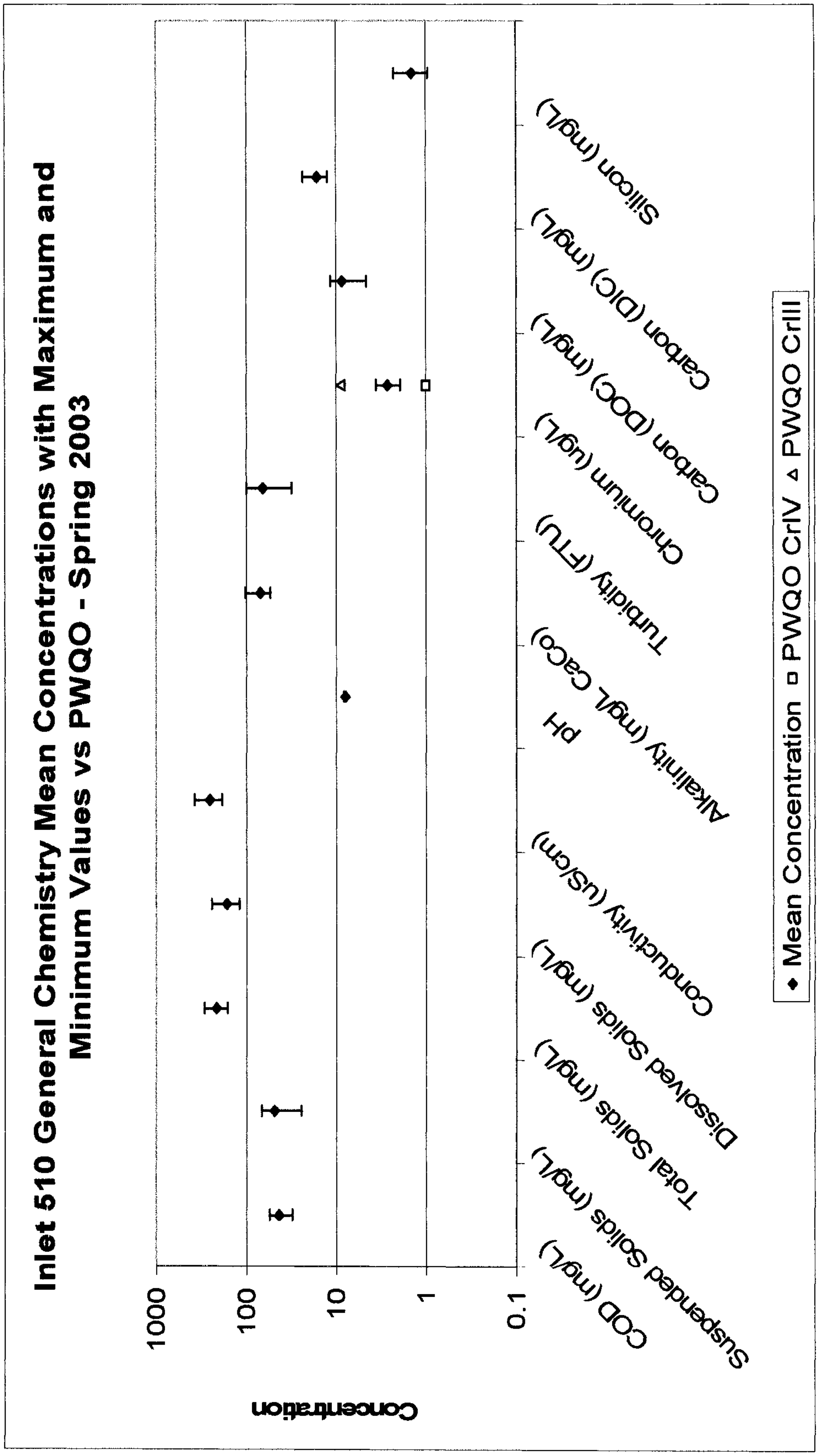
Inlet Mean, Maximum and Minimum Concentrations for fall 2002, and spring 2003 sampling periods

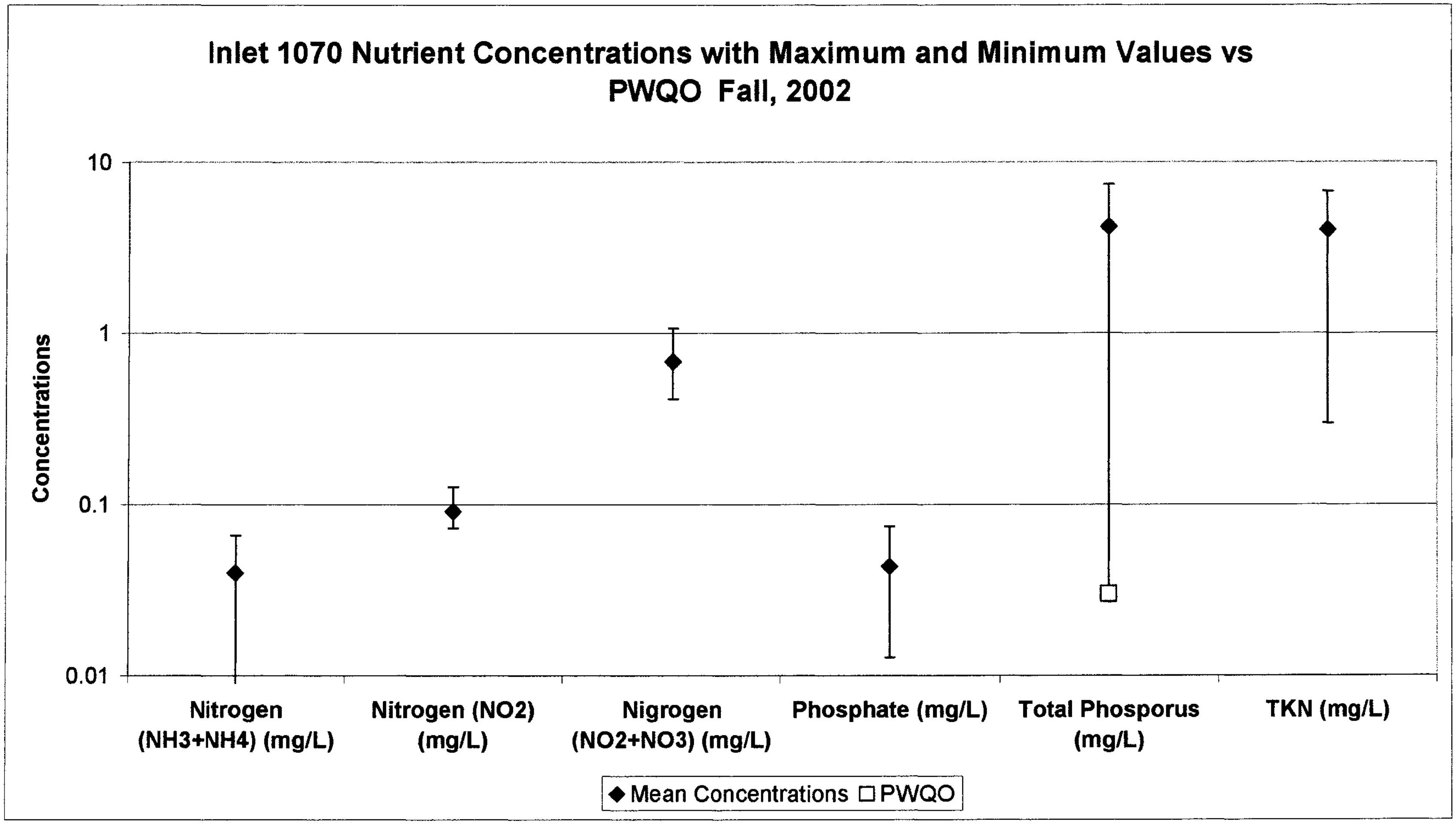
Inlet 1070 General Chemistry with Maximum and Minimum Values vs PWQO Fall, 2002

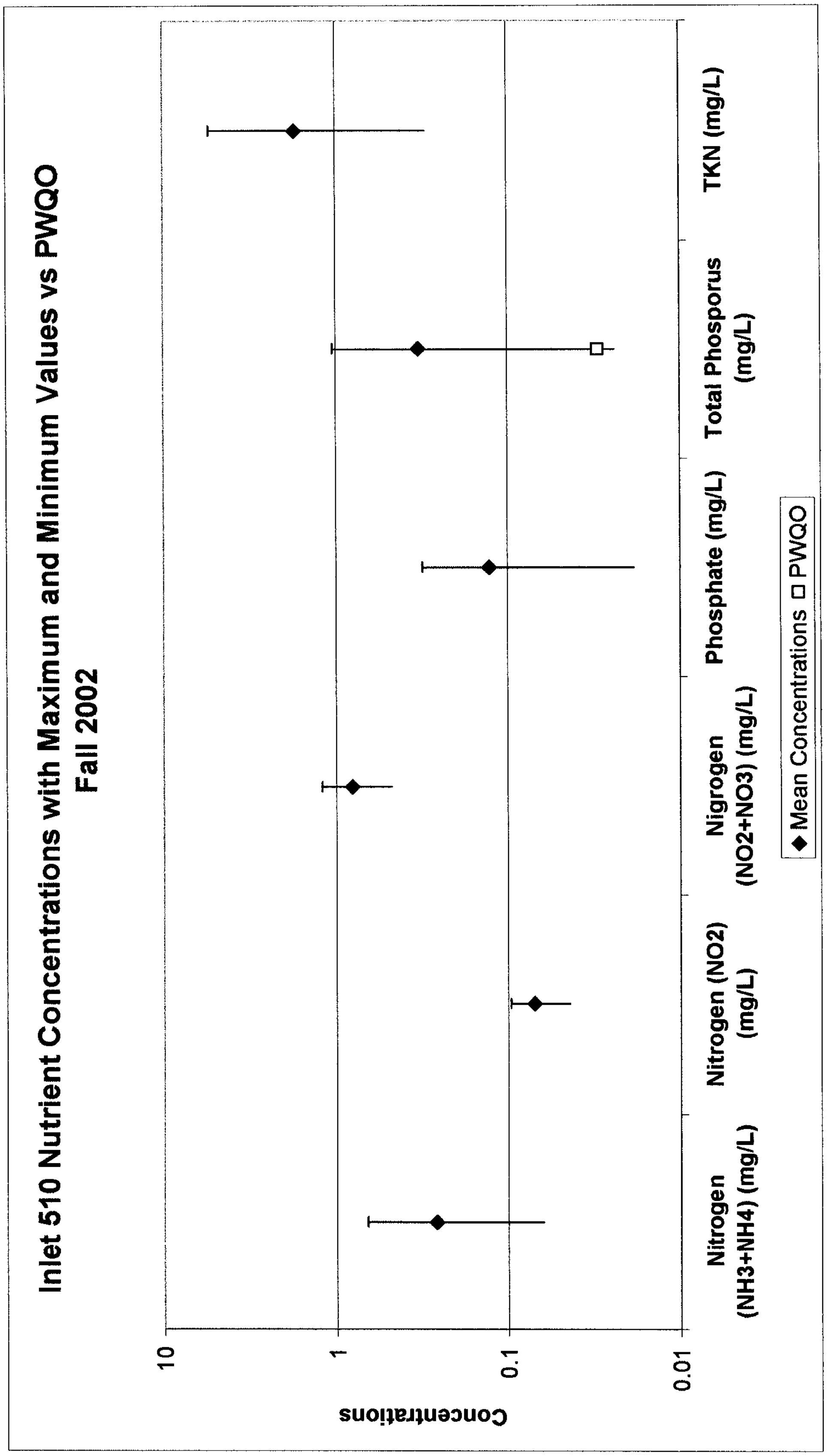


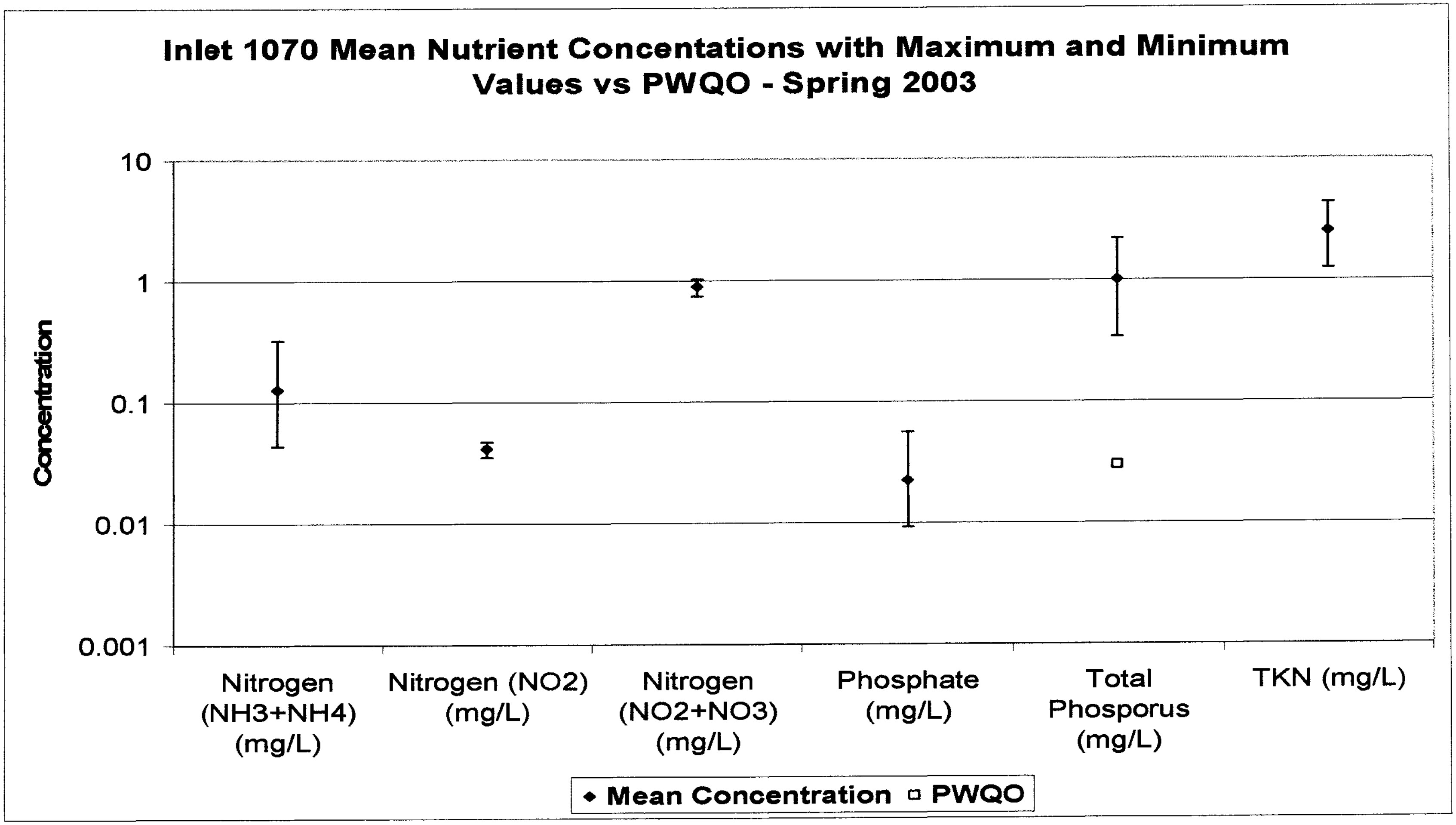

Inlet 510 General Chemistry with Maximum and Minimum Values vs PWQO
Fall 2002
Concentrations
1000
100
10
1
COD (mg/L)
Suspended Solids (mg/L)
Total Solids (mg/L)
Dissolved Solids (mg/L)
Conductivity (uS/cm)
pH
Alkalinity (mg/L CaCo)
Turbidity (FTU)
Chromium (ug/L)
Carbon (DOC) (mg/L)
Carbon (DIC) (mg/L)
Silicon (mg/L)
Mean Concentrations
PWQO - CrIV
PWQO - CrIII

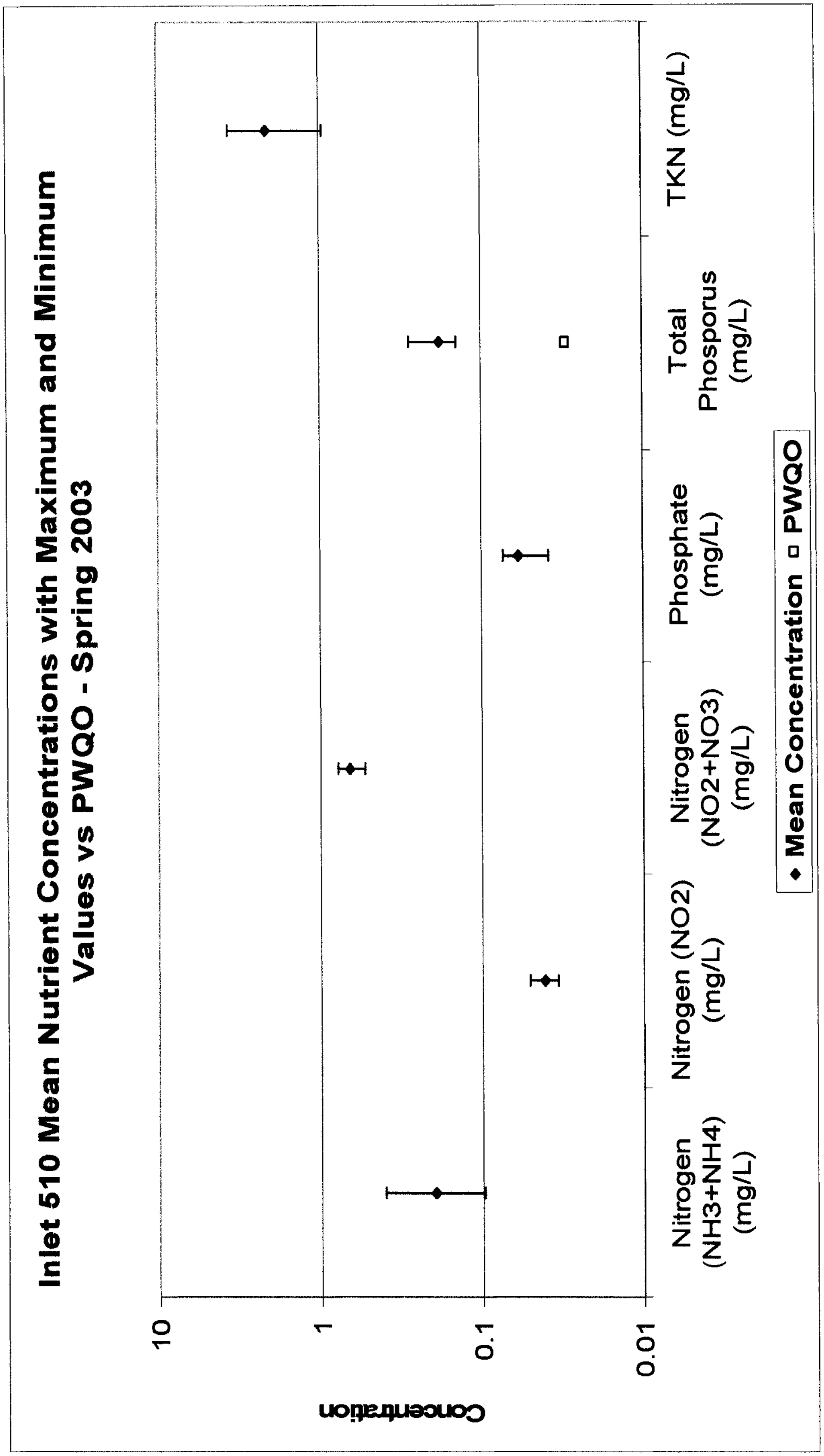

Inlet 1070 General Chemistry Mean Concentrations with Maximum and Minimum Values vs PWQO - Spring 2003
Concentration
10000
1000
100
10
1
COD (mg/L)
Suspended Solids (mg/L)
Total Solids (mg/L)
Dissolved Solids (mg/L)
Conductivity (uS/cm)
pH
Alkalinity (mg/L CaCo)
Turbidity (FTU)
Chromium (ug/L)
Carbon (DOC) (mg/L)
Carbon (DIC) (mg/L)
Silicon (mg/L)
Mean Concentration
PWQO CrIV
PWQO CrIII

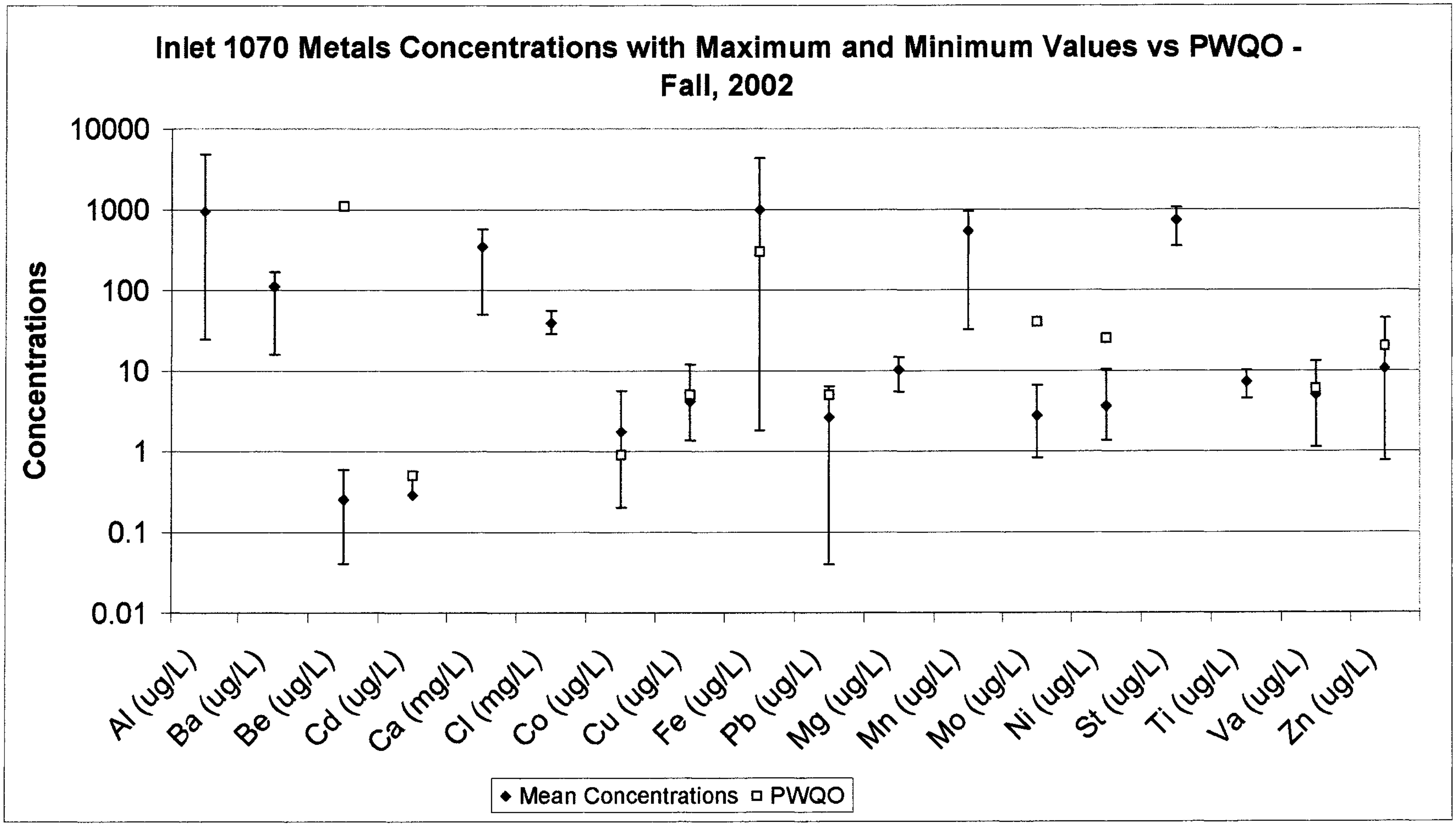

Inlet 510 General Chemistry Mean Concentrations with Maximum and Minimum Values vs PWQO - Spring 2003
Concentration
1000
100
10
1
0.1
COD (mg/L)
Suspended Solids (mg/L)
Total Solids (mg/L)
Dissolved Solids (mg/L)
Conductivity (uS/cm)
pH
Alkalinity (mg/L CaCo)
Turbidity (FTU)
Chromium (ug/L)
Carbon (DOC) (mg/L)
Carbon (DIC) (mg/L)
Silicon (mg/L)
Mean Concentration
PWQO CrIV
PWQO CrIII


Inlet 1070 Nutrient Concentrations with Maximum and Minimum Values vs PWQO Fall, 2002
Concentrations
10
1
0.1
0.01
Nitrogen (NH3+NH4) (mg/L)
Nitrogen (NO2) (mg/L)
Nigrogen (NO2+NO3) (mg/L)
Phosphate (mg/L)
Total Phosporus (mg/L)
TKN (mg/L)
Mean Concentrations
PWQO

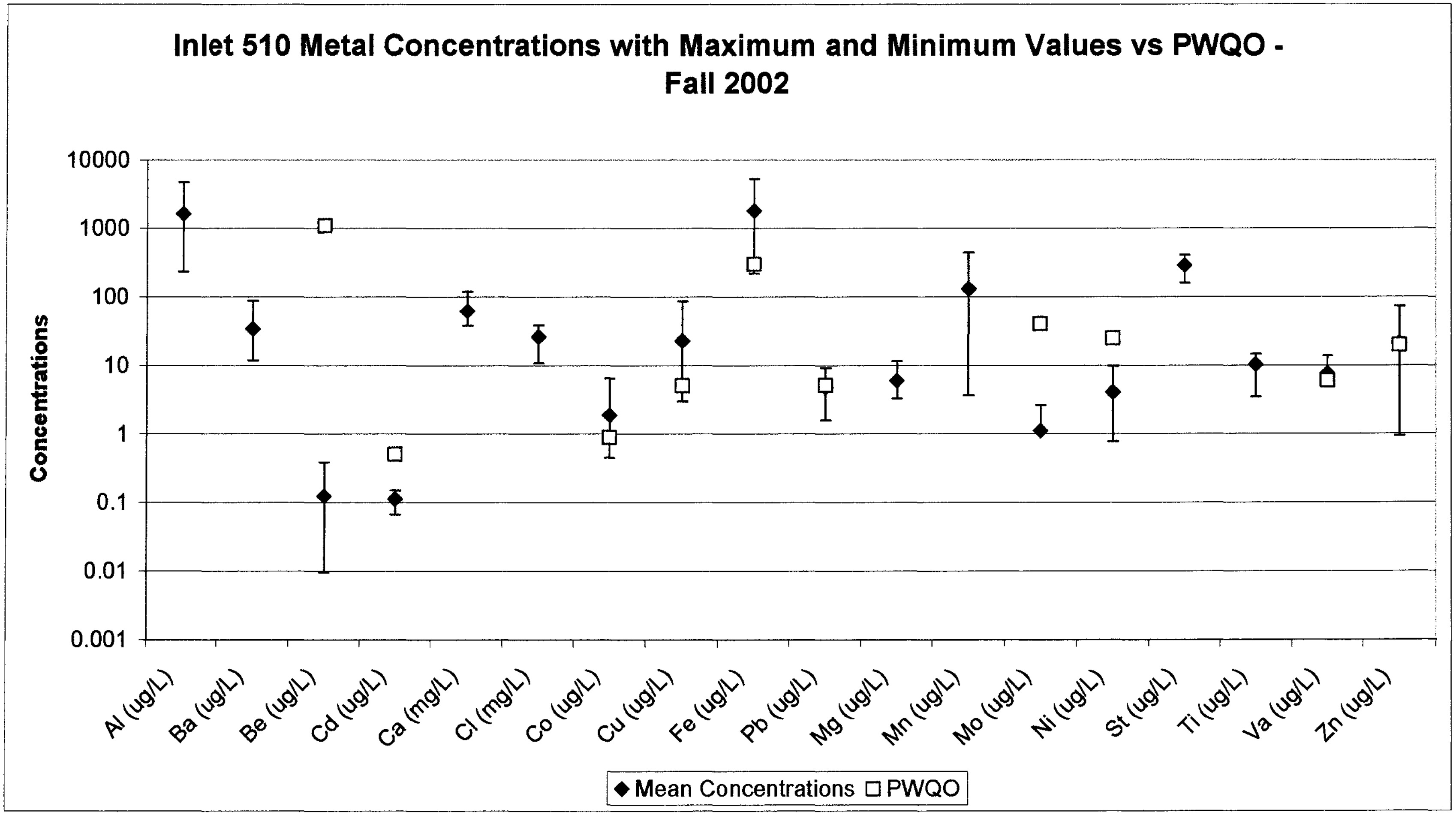

Inlet 510 Nutrient Concentrations with Maximum and Minimum Values vs PWQO
Fall 2002
Concentrations
10
1
0.1
0.01
Nitrogen (NH3+NH4) (mg/L)
Nitrogen (NO2) (mg/L)
Nigrogen (NO2+NO3) (mg/L)
Phosphate (mg/L)
Total Phosporus (mg/L)
TKN (mg/L)
Mean Concentrations
PWQO


Inlet 1070 Mean Nutrient Concentations with Maximum and Minimum Values vs PWQO - Spring 2003
Concentration
10
1
0.1
0.01
0.001
Nitrogen (NH3+NH4) (mg/L)
Nitrogen (NO2) (mg/L)
Nitrogen (NO2+NO3) (mg/L)
Phosphate (mg/L)
Total Phosporus (mg/L)
TKN (mg/L)
Mean Concentration
PWQO

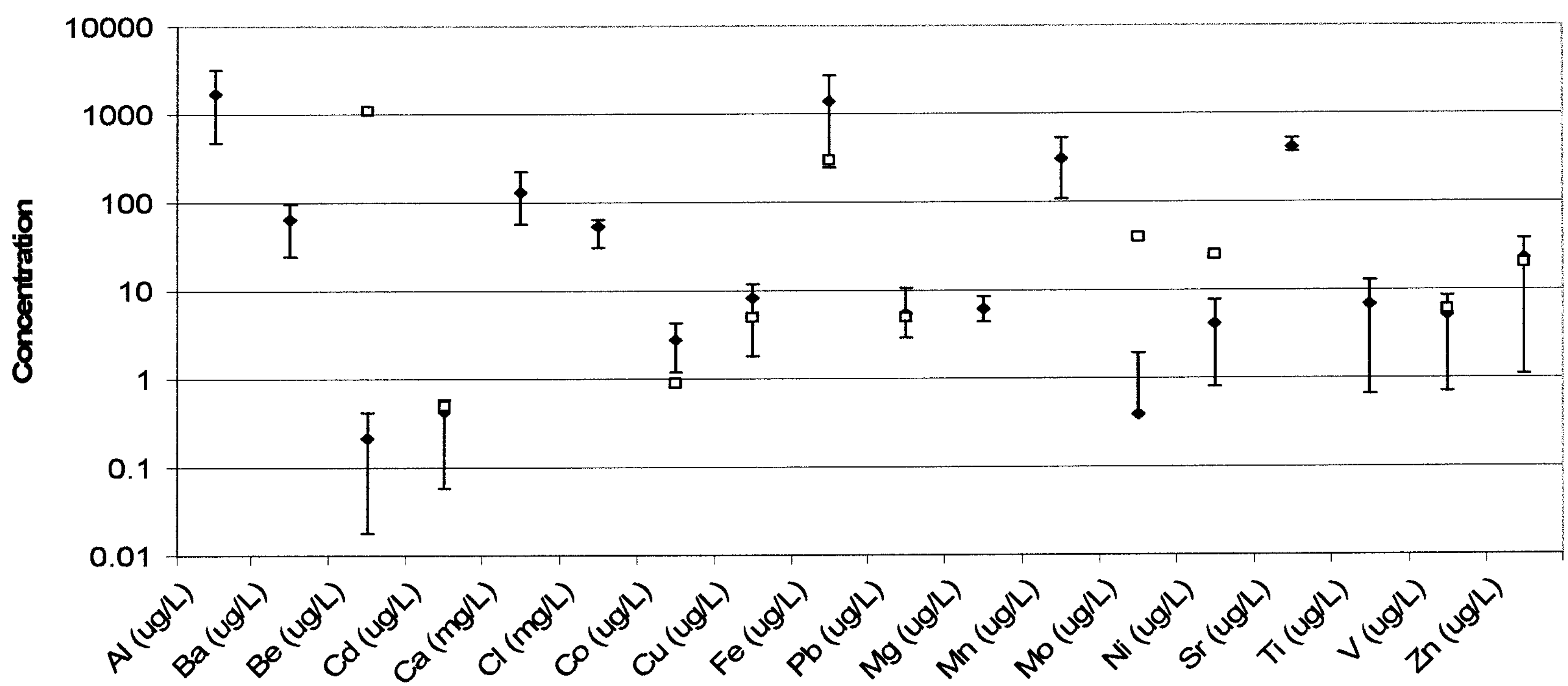

Inlet 510 Mean Nutrient Concentrations with Maximum and Minimum Values vs PWQO - Spring 2003
Concentration
10
1
0.1
0.01
Nitrogen (NH3+NH4) (mg/L)
Nitrogen (NO2) (mg/L)
Nitrogen (NO2+NO3) (mg/L)
Phosphate (mg/L)
Total Phosporus (mg/L)
TKN (mg/L)
Mean Concentration
PWQO



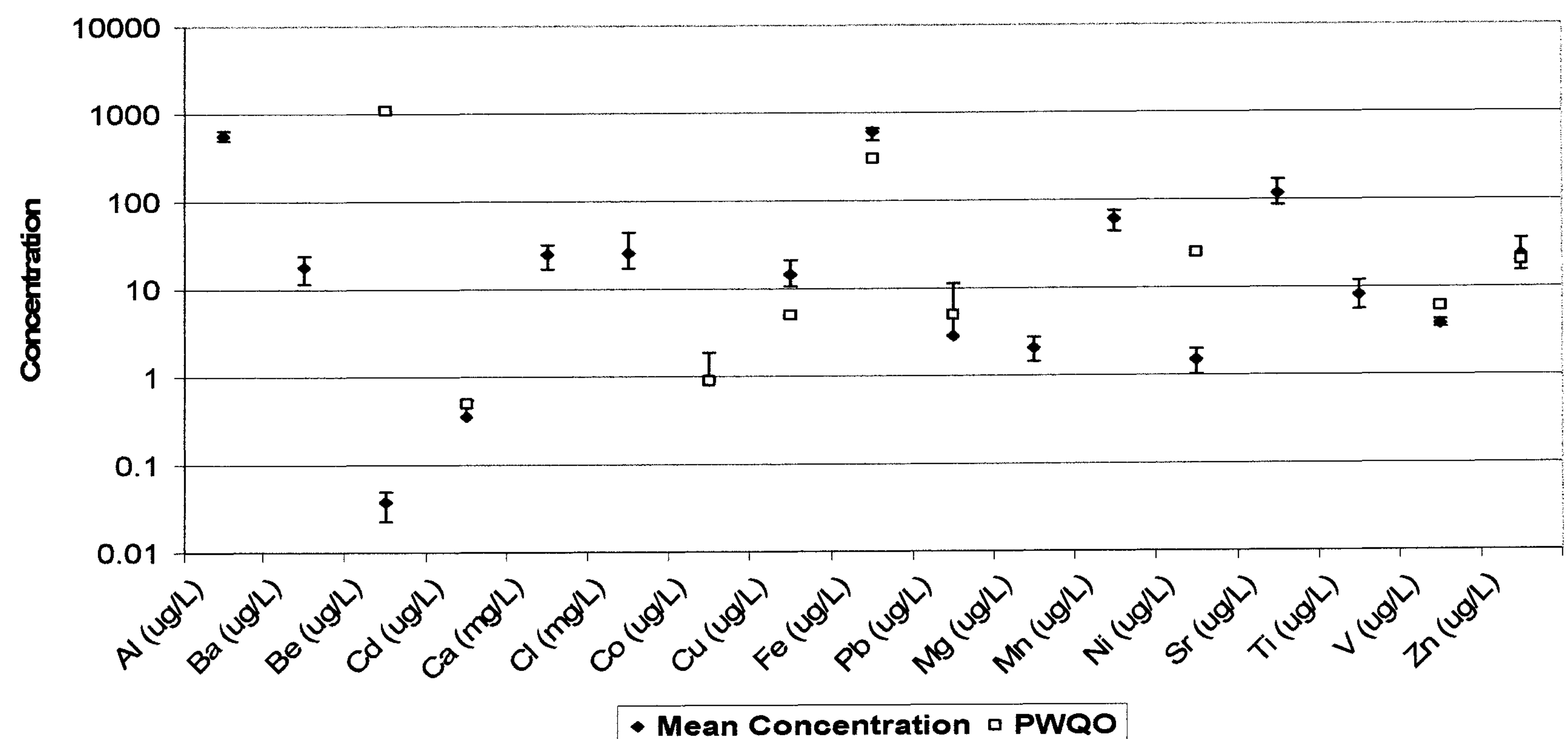
Inlet 1070 Metals Concentrations with Maximum and Minimum Values vs PWQO - Fall, 2002
Concentrations
10000
1000
100
10
1
0.1
0.01
Al (ug/L)
Ba (ug/L)
Be (ug/L)
Cd (ug/L)
Ca (mg/L)
Cl (mg/L)
Co (ug/L)
Cu (ug/L)
Fe (ug/L)
Pb (ug/L)
Mg (ug/L)
Mn (ug/L)
Mo (ug/L)
Ni (ug/L)
St (ug/L)
Ti (ug/L)
Va (ug/L)
Zn (ug/L)
Mean Concentrations
PWQO


Inlet 510 Metal Concentrations with Maximum and Minimum Values vs PWQO -
Fall 2002
Concentrations
10000
1000
100
10
1
0.1
0.01
0.001
Al (ug/L)
Ba (ug/L)
Be (ug/L)
Cd (ug/L)
Ca (mg/L)
Cl (mg/L)
Co (ug/L)
Cu (ug/L)
Fe (ug/L)
Pb (ug/L)
Mg (ug/L)
Mn (ug/L)
Mo (ug/L)
Ni (ug/L)
St (ug/L)
Ti (ug/L)
Va (ug/L)
Zn (ug/L)
◆ Mean Concentrations □ PWQO

Inlet 1070 Mean Metal Concentrations with Maximum and Minimum Values vs PWQO - Spring 2003

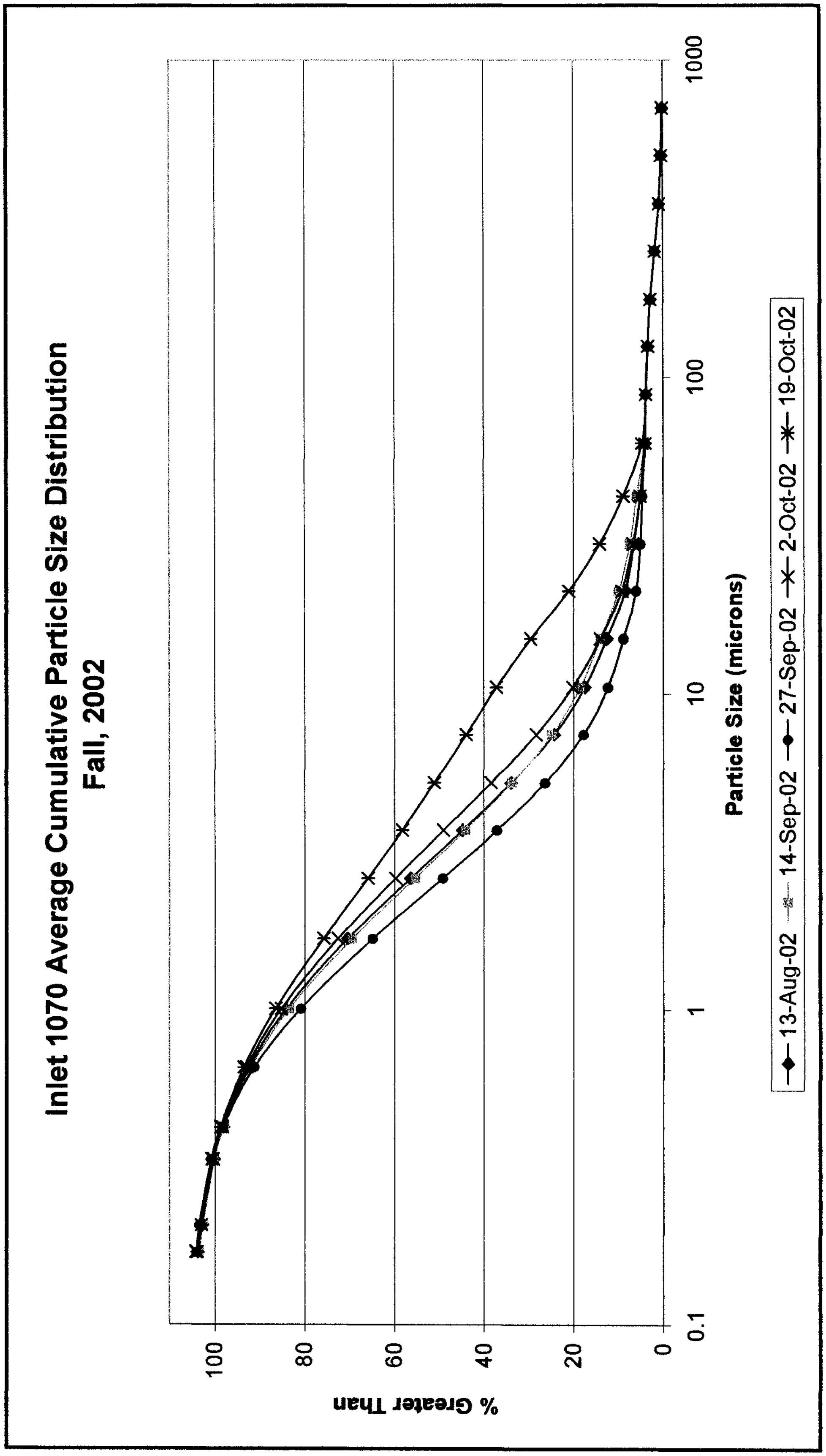


Inlet 510 Mean Metal Concentrations with Maximum and Minimum Values vs PWQO - Spring 2003

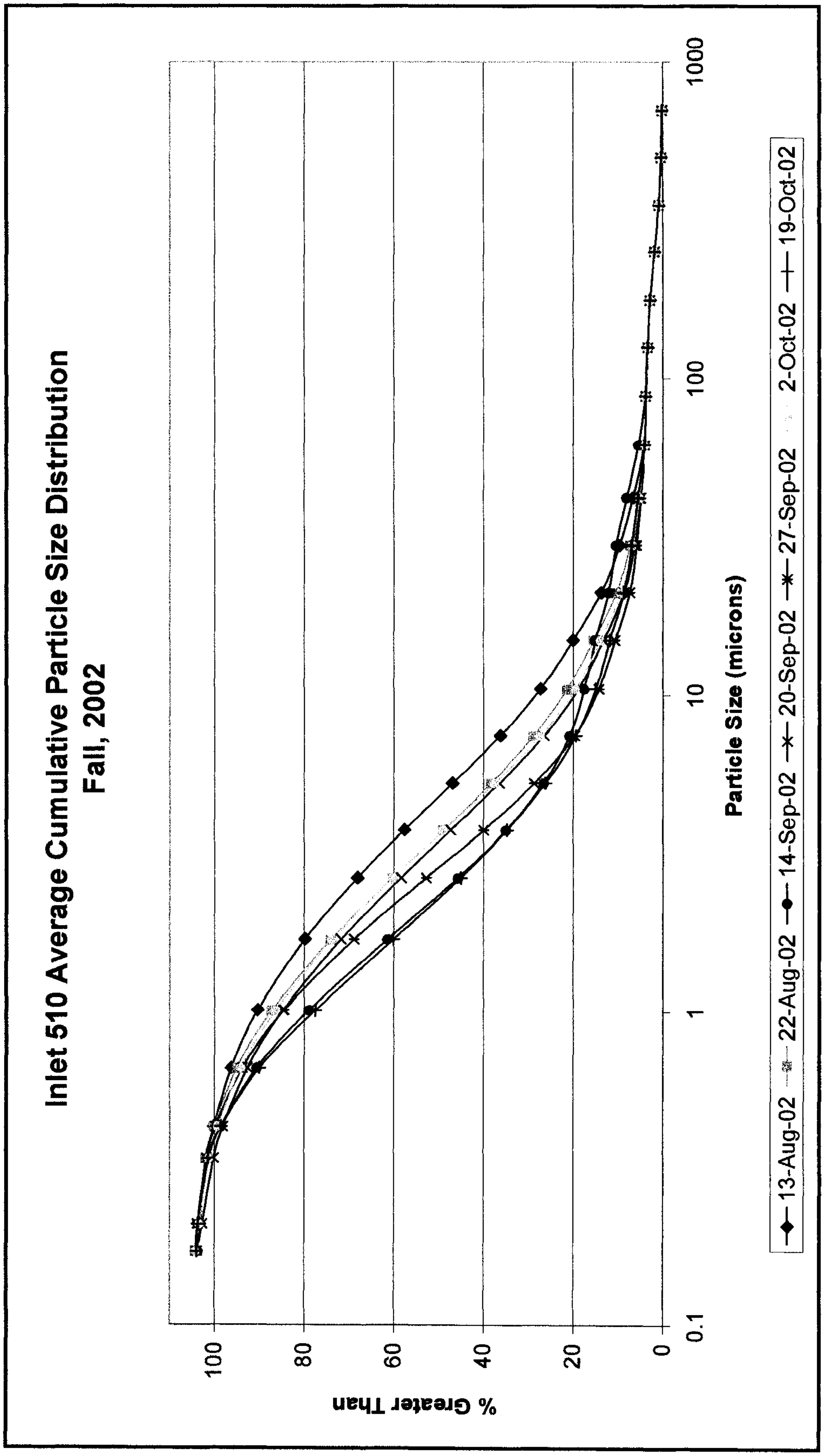


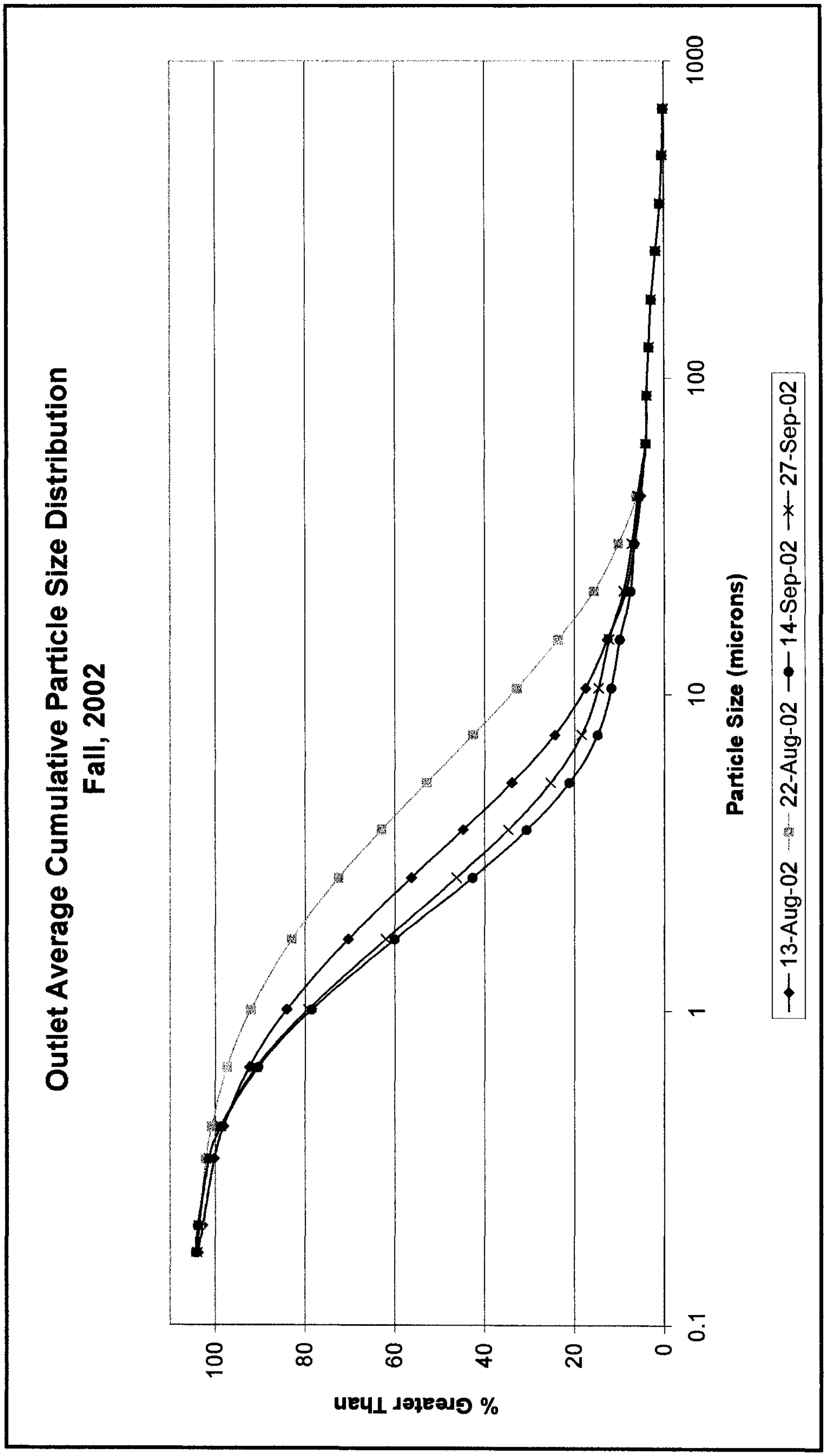
Appendix F

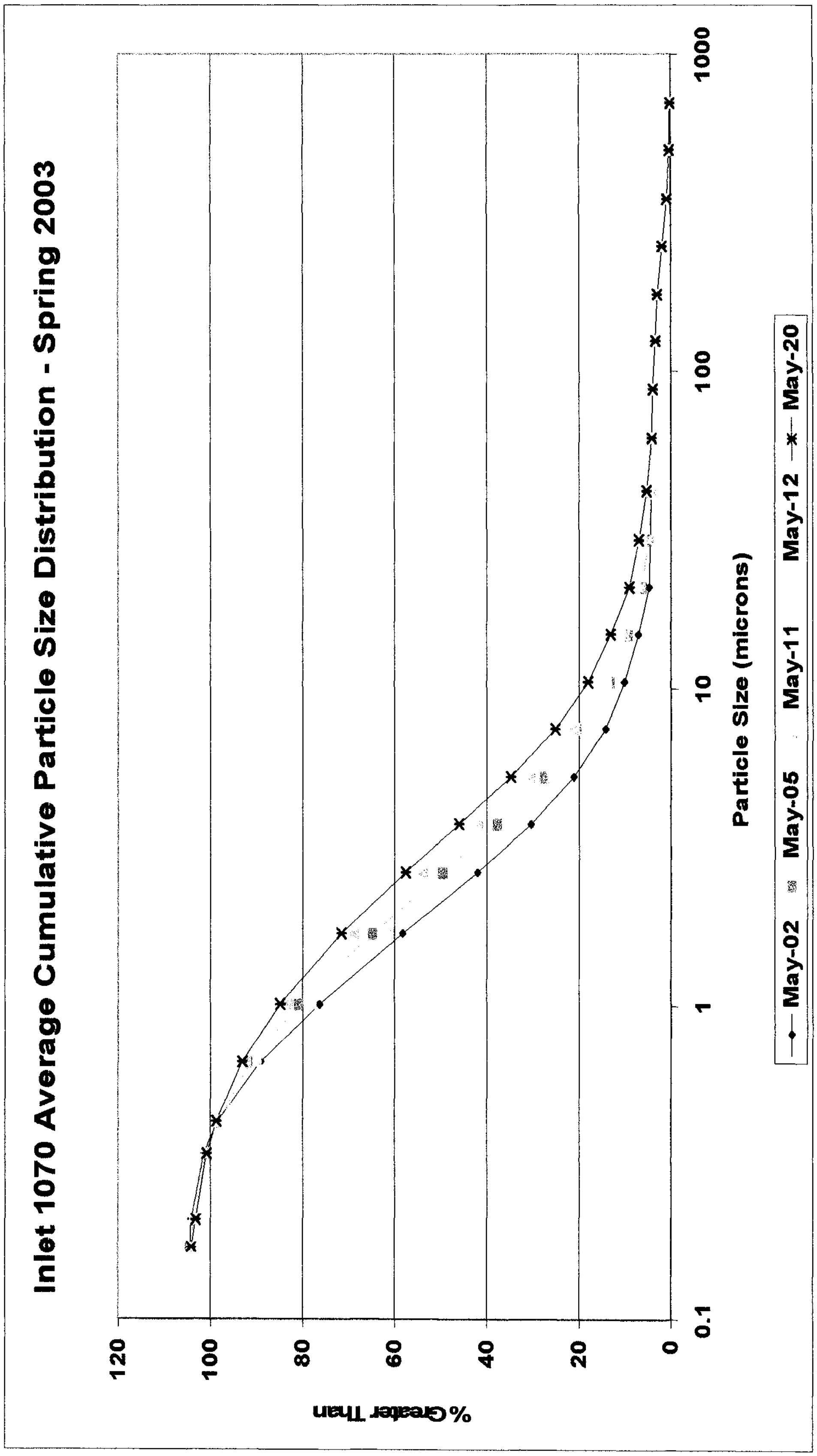
Particle Size Distributions

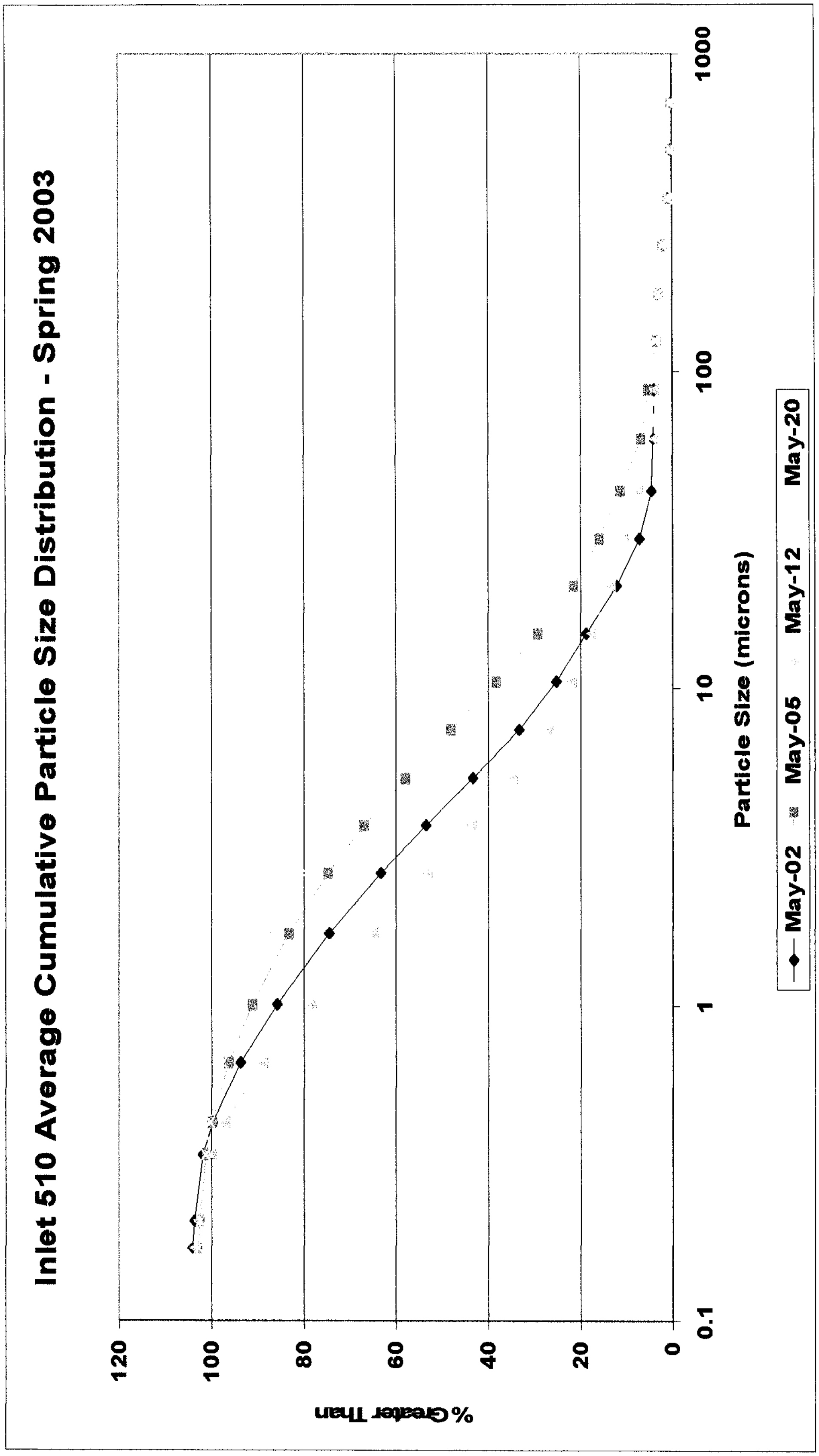


Inlet 1070 Average Cumulative Particle Size Distribution
Fall, 2002
% Greater Than
100
80
60
40
20
0
0.1
1
10
100
1000
Particle Size (microns)
13-Aug-02
14-Sep-02
27-Sep-02
2-Oct-02
19-Oct-02

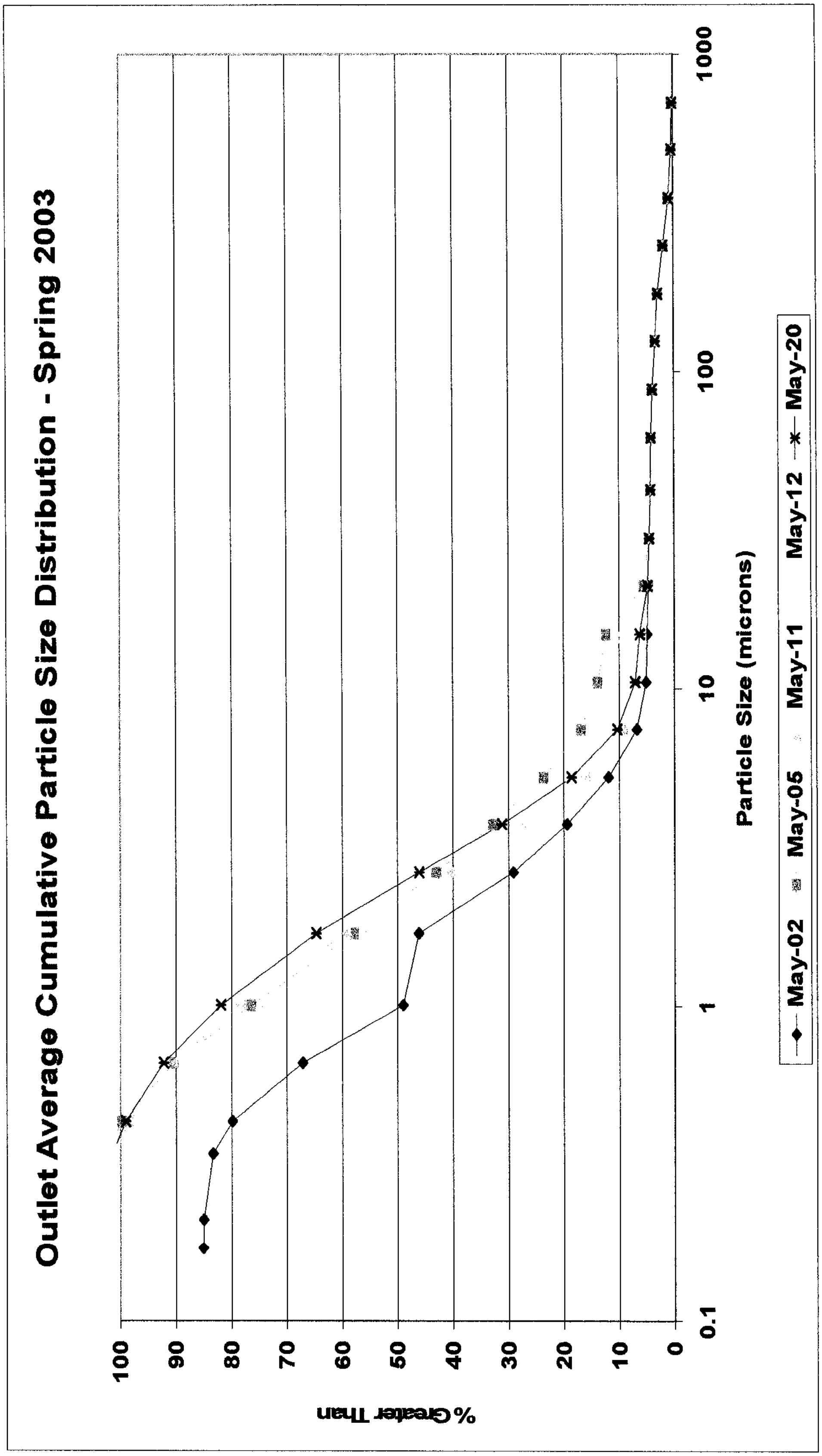

Inlet 510 Average Cumulative Particle Size Distribution
Fall, 2002
% Greater Than
100
80
60
40
20
0
0.1
1
10
100
1000
Particle Size (microns)
13-Aug-02
22-Aug-02
14-Sep-02
20-Sep-02
27-Sep-02
2-Oct-02
19-Oct-02


Outlet Average Cumulative Particle Size Distribution
Fall, 2002
% Greater Than
100
80
60
40
20
0
0.1
1
10
100
1000
Particle Size (microns)
13-Aug-02
22-Aug-02
14-Sep-02
27-Sep-02


Inlet 1070 Average Cumulative Particle Size Distribution - Spring 2003
% Greater Than
120
100
80
60
40
20
0
0.1
1
10
100
1000
Particle Size (microns)
May-02
May-05
May-11
May-12
May-20



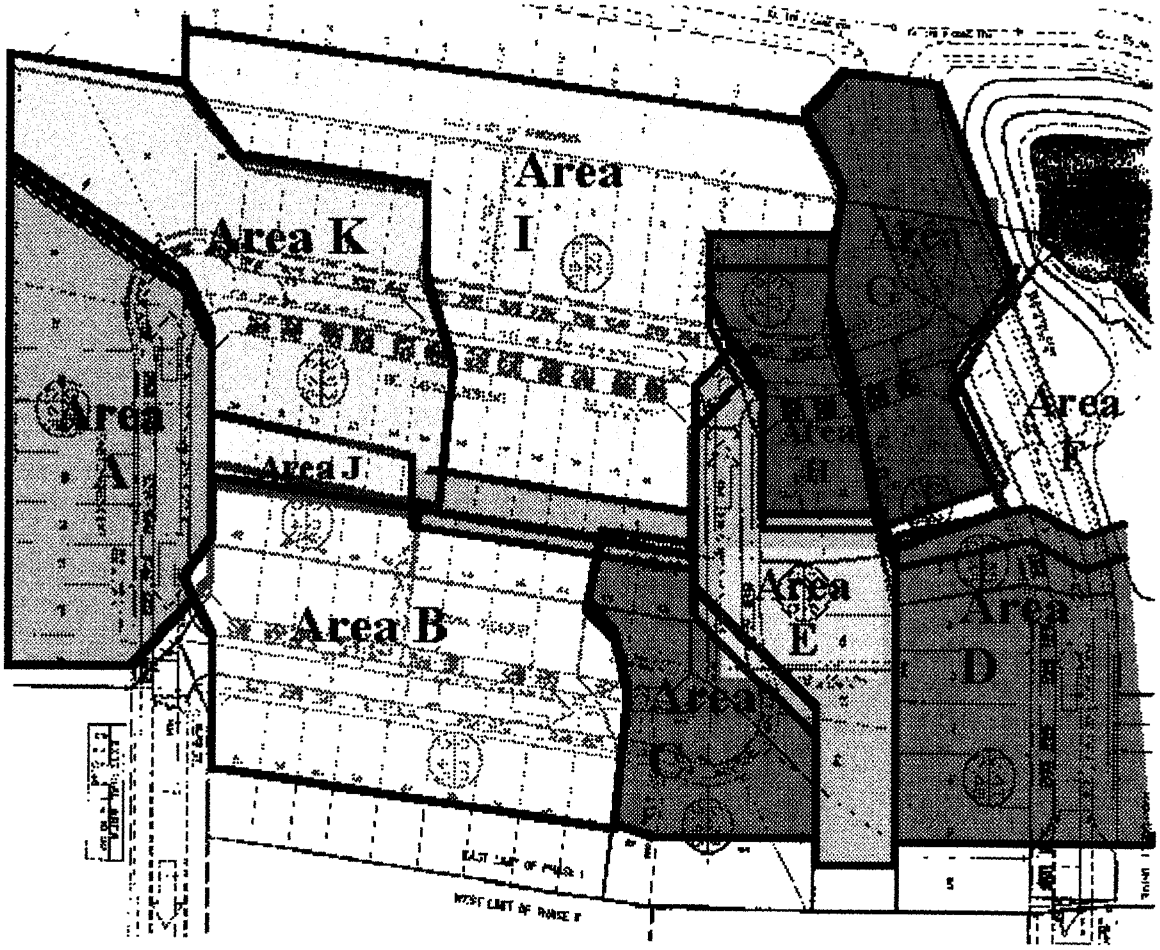
Inlet 510 Average Cumulative Particle Size Distribution - Spring 2003
% Greater Than
120
100
80
60
40
20
0
0.1
1
10
100
1000
Particle Size (microns)
May-02
May-05
May-12
May-20



Outlet Average Cumulative Particle Size Distribution - Spring 2003
% Greater Than
100
90
80
70
60
50
40
30
20
10
0
0.1
1
10
100
1000
Particle Size (microns)
May-02
May-05
May-11
May-12
May-20

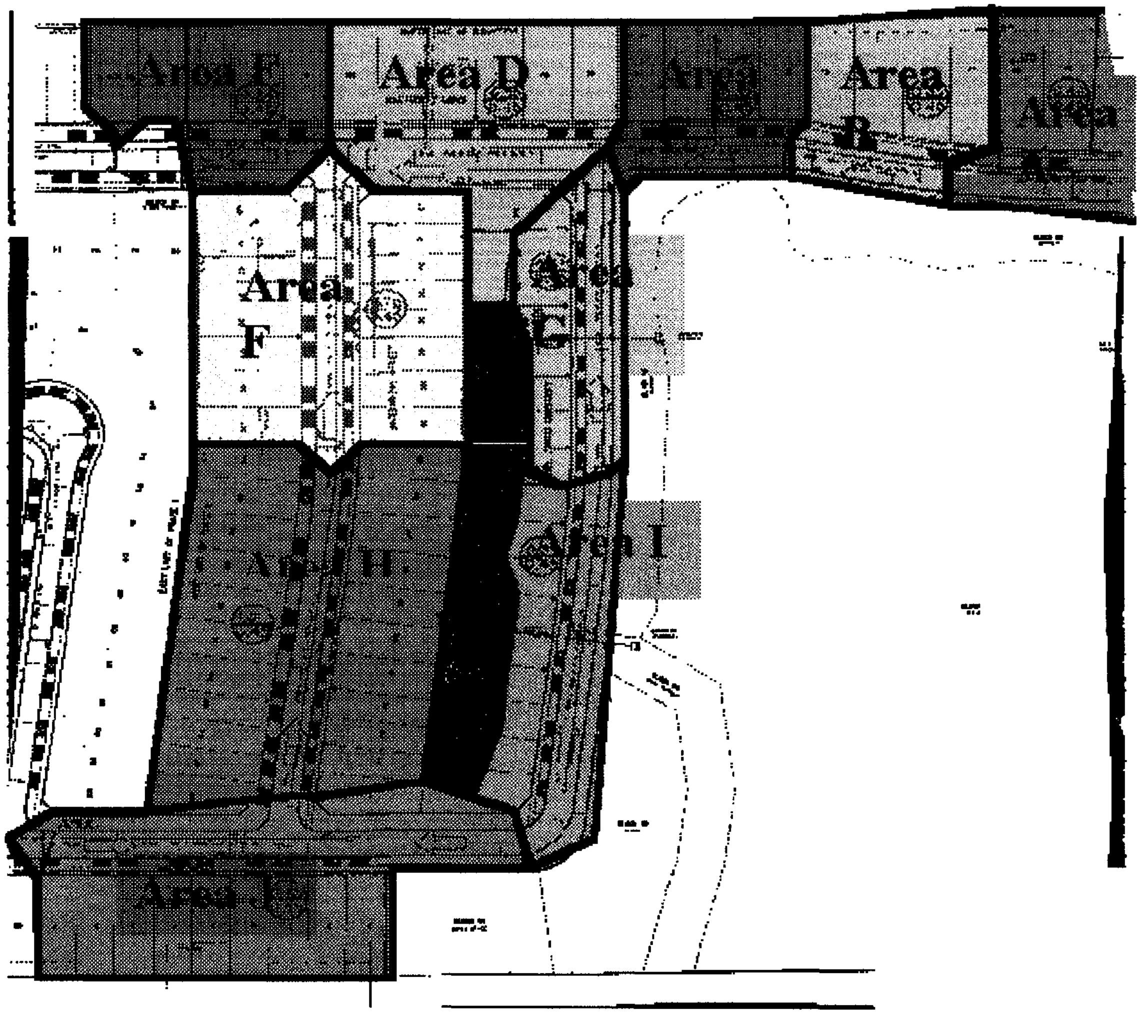
Appendix G

Residential Site Maps for Catchment Areas

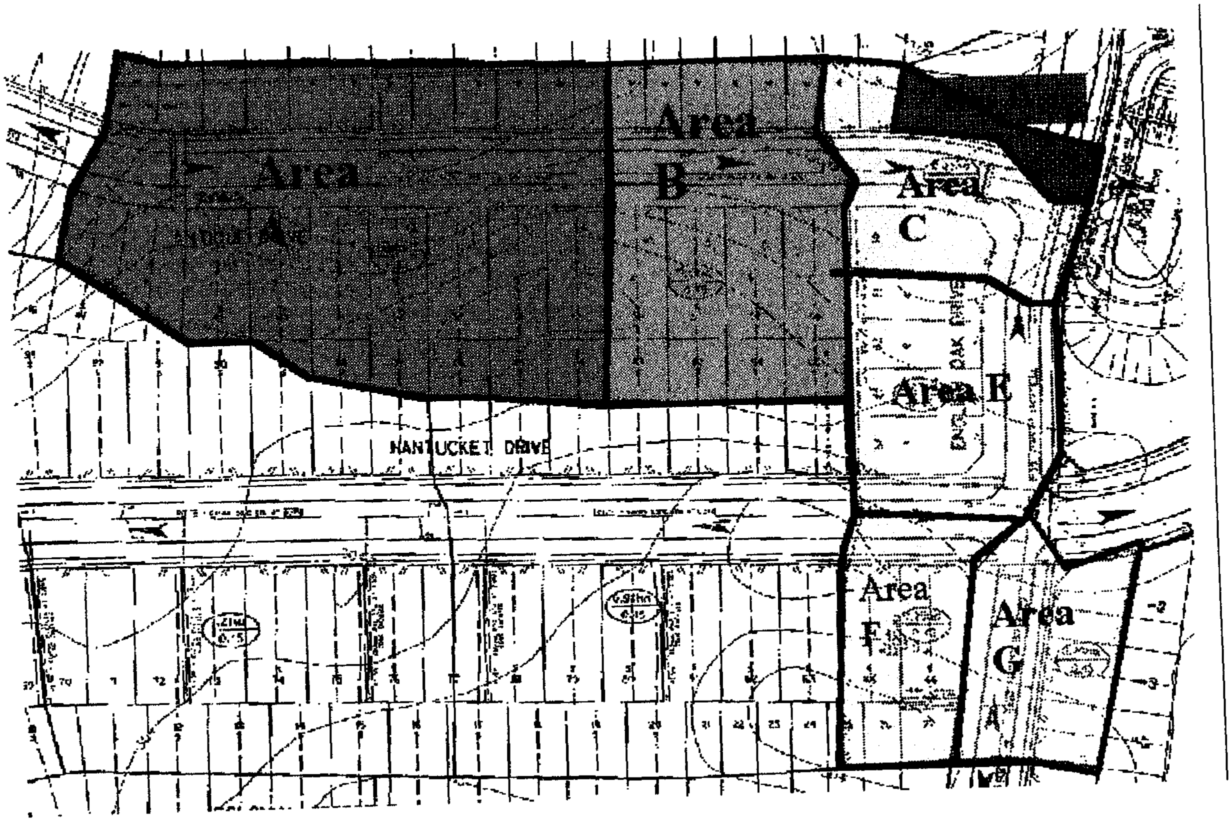


Area K
Area I
Area A
Area J
Area B
Area C
Area E
Area G
Area H
Area F
Area D

Inlet 1070 A



Inlet 1070 B


Area A
Area B
Area C
Area E
Area F
Area G
NANTUCKET DRIVE
OAK DRIVE

Inlet 510