SWAMP

Performance Assessment of a Stormwater Retrofit Pond - Harding Park, Richmond Hill, Ontario

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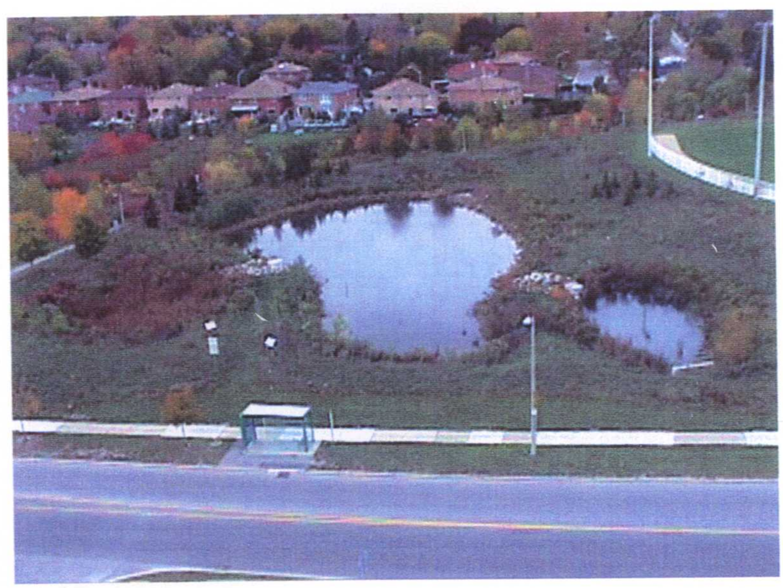
Ontario

Ministry of Environment and Energy



Toronto and Region Conservation

Town of Richmond Hill

PERFORMANCE ASSESSMENT OF A STORMWATER RETROFIT POND - HARDING PARK, RICHMOND HILL, ONTARIO

a report prepared by the

STORMWATER ASSESSMENT MONITORING
AND PERFORMANCE (SWAMP) PROGRAM

for

Great Lakes Sustainability Fund of the Government of Canada
Ontario Ministry of Environment and Energy
Toronto and Region Conservation Authority
Municipal Engineers Association of Ontario
Town of Richmond Hill

June, 2002

NOTICE

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THE SWAMP PROGRAM

The Stormwater Assessment Monitoring and Performance (SWAMP) Program is an initiative of the Government of Canada's Great Lakes Sustainability Fund, the Ontario Ministry of Environment and Energy, the Toronto and Region Conservation Authority, and the Municipal Engineer's Association. A number of individual municipalities and other owner/operator agencies have also participated in SWAMP studies.

During the mid to late 1980s, the Great Lakes Basin experienced rapid urban growth. Stormwater runoff associated with this growth has been identified as a major contributor to the degradation of water quality and the destruction of fish habitats. In response to these concerns, a variety of stormwater management technologies have been developed to mitigate the impacts of urbanization on the natural environment. These technologies have been studied, designed and constructed on the basis of computer models and pilot-scale testing, but have not undergone extensive field-level evaluation in southern Ontario. The SWAMP Program was intended to address this need.

The SWAMP Program's objectives are:

* to monitor and evaluate new and conventional stormwater management technologies; and
* to disseminate study results and recommendations within the stormwater management industry.

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Additional information concerning SWAMP and the supporting agencies is included in Appendix A.

ACKNOWLEDGEMENTS

This report was prepared for the Steering Committee of the Stormwater Assessment Monitoring and Performance (SWAMP) Program. The SWAMP Program Steering Committee is comprised of representatives from:

- the Government of Canada's Great Lakes Sustainability Fund,
- the Ontario Ministry of Environment and Energy,
- the Toronto and Region Conservation Authority,
- the Municipal Engineers Association of Ontario.

This study was jointly funded by the Ontario Ministry of Environment and Energy (OMOEE), the Great Lakes Sustainability Fund (formerly the Great Lakes 2000 Clean-up Fund) and the Town of Richmond Hill. The OMOEE also provided office facilities and logistic support for the SWAMP program. The Laboratory Services Branch of the OMOEE provided laboratory analyses. Staff at the Town of Richmond Hill provided permission to conduct the monitoring study and assisted with the installation and operation of monitoring equipment. The Toronto and Region Conservation Authority provided administrative support for the SWAMP program.

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EXECUTIVE SUMMARY

Background

In 1995, the Town of Richmond Hill converted its Harding Park Stormwater Management Facility from a water quantity dry pond to a multi-celled wetpond/wetland designed to improve stormwater quality and meet current erosion control objectives, while maintaining its original quantity control function. Located within the degraded Don River watershed, the Harding Park project is one of several community action sites identified by the Don Watershed Task Force as demonstrating techniques of regeneration at the local level. The combined effect of many such local projects within the watershed are expected to help restore health to the Don River and, in support of the Toronto and Region Remedial Action Plan (RAP), improve water quality and aquatic habitat along the Toronto waterfront.

Although several quantity-to-quality pond retrofits have been implemented in Ontario over the past 10 years, there is a paucity of data demonstrating the effectiveness of these retrofits, especially for pond-wetland systems. Also, little is known about operation and maintenance issues related to aspects such as the frequency of sediment clean out from stormwater ponds. To generate the necessary data, the Ontario Ministry of Environment and Energy (OMOEE), the Toronto and Region Conservation Authority and the Government of Canada, through the Great Lakes 2000 Clean-up fund (superceded by the Great Lakes Sustainability Fund), jointly agreed to monitor the facility under the Stormwater Assessment Monitoring and Performance (SWAMP) program. This report presents the results of the monitoring program, discusses implications with regard to receiving water impacts, and provides recommendations for improvements to the facility.

Study Objectives

The overall objective of this study was to provide guidance to urban planners, designers and owners of stormwater facilities concerning the design, performance, and maintenance of stormwater retrofit ponds. Within this general context, the specific objectives were to:

- determine hydrologic characteristics of the study catchment and stormwater retrofit pond/wetland (*e.g.* runoff coefficient, peak flows, hydraulic detention times) and evaluate these against the original design objectives;

- assess the stormwater treatment performance of the facility on an average event and seasonal load basis;

- identify aquatic plant species below the high water line of the facility and assess the effectiveness of planting plans;

- evaluate the use of algae as a indicator of spatial variations in stormwater quality within the facility;

- investigate the long-term operation, maintenance and dredging requirements of the facility;

- identify environmental benefits/limitations of the facility and provide recommendations for facility improvement.

Site Description

The retrofit facility incorporates three cells in series: a small sediment forebay, larger wet pond and small wetland (Figure 1). The pond and forebay cover a 0.7 hectare area and have a total storage capacity of 2965 m^3 consisting of a 1015 m^3 permanent pool and 1950 m^3 of active (or extended detention) storage. By contrast, the former dry pond was 0.4 hectares in area and had a total storage capacity of 1650 m^3. Surface drawoff Hickenbottom risers at the forebay and wetland outlets provide for hydraulic control and extended detention. Emplaced sand lenses (or 'French drains') in the berm between the wet pond and wetland help to maintain moist soils in the wetland during dry weather periods. The design of the facility meets the OMOEE's stormwater quality and erosion control guidelines with respect to maximum depth (less than 3 m), drawdown time and storage volume, but the 1:1 length-to-width ratio was less than the 3:1 ratio recommended by the OMOEE.

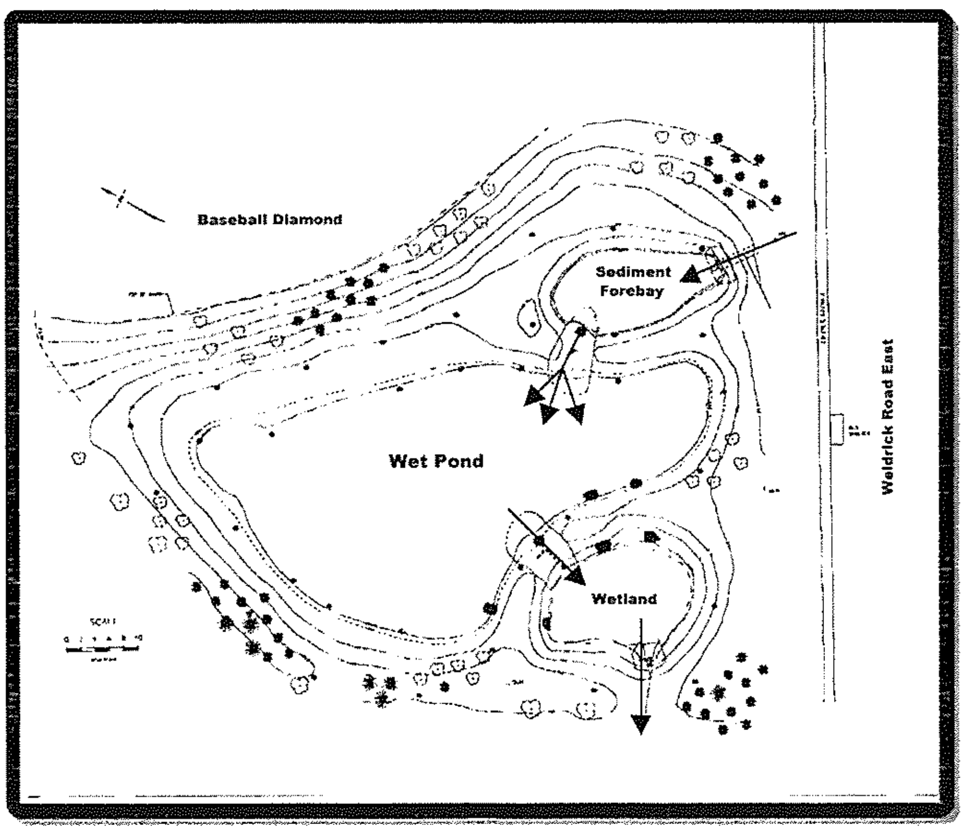

Figure 1: Study site

The facility receives runoff from a 16.8 hectare residential catchment that lies on the southern edge of the Oak Ridges Moraine. The primary local surface water body is German Mills Creek, which drains to the Don River. Land use in the drainage catchment is medium to high density residential, serviced to about 70% by curb, gutter and storm sewers and 30% by roadside ditches and culverts. Prior to construction of the retrofit facility, aquatic habitat in the creek was degraded as a result of urban development and instream erosion control works. Soils in the area were considered to have low infiltration and high runoff potential. Climate is temperate with thermal highs and lows in all seasons moderated by the dominant lake effect from Lake Ontario, 30 km south of the study site.

Study Methods

The monitoring study was conducted from January 1996 to November 1997. Data were analyzed separately for the 'summer/fall' period, from May 1 to November 30, and the 'winter/spring' period, from December 1 to April 30. During the first year of the monitoring period, from January to late August 1996, the berm separating the wet pond and wetland was in a state of disrepair, resulting in unreliable and unrepresentative effluent flow data. Therefore, the focus of the data analysis was on results collected after berm repair (September 1996 to November 1997).

The key components of the monitoring program included rainfall, runoff, water quality, temperature, aquatic vegetation, and in-pond algal communities. Rainfall data in 1996 were obtained from the Buttonville Airport weather station, seven kilometers southeast of the study site, and in 1997, from a tipping bucket rain gauge located immediately adjacent to the Harding Park facility. Temperature impacts of the facility on the downstream watercourse were assessed via continuous temperature monitoring at the inlet, outlet and immediately upstream of the outlet in German Mills Creek. Continuous runoff data were collected at the inlet and outlet using area-velocity flow meters from May to November in 1996 and 1997. During rain events over the same period, flow-proportionate water quality samples were collected using automated instruments. Grab samples were collected during the winter/spring season (December to April). Samples were submitted to the Ministry of Environment and Energy laboratory in Toronto for analysis of total suspended solids (TSS) and all major pollutant groups, including nutrients (phosphorus and nitrogen compounds), metals, organics, bacteria and parameters such as pH, conductivity and particle size distribution.

Subsequent data analysis included calculation of event flow volumes, runoff coefficients, peak flow attenuation, flow durations, lag and hydraulic detention times, hydrological mass balances, average event mean concentrations (AEMCs), 95% confidence intervals, inlet and outlet loading and load-based removal efficiencies. The catchment area and facility were modelled using PCSWMM™ to predict long-term flow rates and suspended solids loading. The model was calibrated using field data collected at the Harding Park site, and a long-term continuous simulation of the model was based on 12 years (1986 to 1997) of rainfall data from the Toronto Buttonville Airport, 7 km southeast of the facility.

The vegetation study inventoried plant species composition and coverage below the high water mark in the facility and assessed the validity of natural colonization as a planting strategy. The aquatic community study focused on the algal communities in the sediment forebay and wet pond with a view to better understand the water quality improvement function of the facility.

Study Findings

Water quantity

The post-berm repair data set includes 6 large storms (greater than 20 mm), 8 medium sized storms (10 to 20 mm) and 1 small storm (less than 10 mm). On average, 34% of catchment rainfall appeared as surface runoff during storm events over the monitoring period. Storms with less than 4.0 mm of rainfall produced negligible runoff, probably due to depression storage and infiltration in roadside ditches. This observation approaches the 5 mm level suggested in the Ontario Stormwater Management Practices and Planning (SWMP) manual for stream baseflow maintenance.

Rain events during the study period generated an average influent storm flow volume of 1451 m^3. The balance between storm flow at the inlet and outlet averaged 11% during monitored rain events, possibly due to exfiltration within the pond (which was not measured), but more likely due to under or over estimation of flow by automated flow instruments. Dry weather baseflow rates were estimated to be 1.5 L/s at the inlet and 1.3 L/s at the outlet, indicating that water losses to pond exfiltration below the permanent pool water line were relatively minor.

During the period after berm repair, mean peak discharge rates were 137 and 27 L/s at the inlet and outlet, respectively. Only two storms had peak outlet flow rates beyond the post-berm repair design threshold of 52 L/s. Peak flow reduction was accompanied by a significant increase in the duration of flow from a 22-hour mean at the inlet to a mean of 46 hours at the outlet.

The hydraulic detention time, defined as the time delay between inlet and outlet hydrograph centroids, provides a measure of the extended detention feature of the facility, by which stormwater influent is temporarily detained, or held back within the facility. The detention time averaged 5.3 hours, with a range between 3.5 and 10.7 hours during individual events. Assuming plug flow displacement conditions (*i.e.* no mixing, no short circuiting), the residence time of an element of fluid within the facility was estimated at roughly 18 hours. This estimate suggests that under actual conditions of short-circuiting and mixing of the influent and permanent pool water, the residence time of a fluid element passing through the facility would be somewhat less than 18 hours. A falling head drawdown equation was used by the designer to meet the OMOEE 'detention time' guideline of 24 hour detention of a 25 mm storm (4 hour Chicago distribution). In general, the observed drawdown time for storms with greater than 20 mm exceeded the 24 hour target.

Water quality

Total suspended solids (TSS) is a critical variable in stormwater quality analysis because several pollutants (*e.g.* phosphorus, metals and some organics) are bound to suspended particles and, hence, the removal of TSS also serves as a measure of the removal of these bound pollutants. For the summer/fall monitoring period after berm repair, removal of TSS averaged 80%, ranging between 26 and 92% during individual events. Influent and effluent AEMCs were 345 and 46 mg/L, respectively. Winter/spring average performance based on grab samples was similar, averaging 78% and ranging between 58 and 97%. Winter/spring influent and effluent average concentrations were 270 and 39 mg/L, respectively. Removal efficiencies during both seasons exceeded the 70% recommended in the SWMP manual for the level of fisheries protection (*i.e.* level 2) deemed appropriate for the reach of German Mills Creek downstream of the Harding Park facility. The geotextile wrapped Hickenbottom risers at the forebay and wet pond outlets, as well as the location of the discharge point at the surface of the permanent pool may have contributed to the reasonably good TSS removal efficiency results.

On average, the median particle size of TSS was 4.5 μm (fine silt) at the inlet and 2.3 μm (clay) at the outlet. Removal efficiencies for sand, silt and clay were estimated at 81, 65 and 48%, respectively. Particles greater than 4 μm (*i.e.* silt and sand size classes) accounted for 55% of the inlet particles compared to only 34% at the outlet, indicating size selective removal of suspended solids.

Removal efficiencies for most parameters were less than observed for TSS (Table 1). During the summer/fall season, load-based removal efficiency was 42% for total phosphorus, 54% for total ammonia, and a range of 11 to 83% for most metals. In contrast, winter removal efficiencies were 56% for total phosphorus, 18% for total ammonia and a range of −13 to 83% for most metals. The surface drawoff configuration of the Hickenbottom risers in the sediment forebay and wet pond did not provide adequate protection from the rapid release of floating oil and grease, which may partly explain the relatively low summer/fall removal efficiency (48%) for this constituent.

During the summer/fall monitoring period, effluent AEMCs of unionized ammonia, zinc, cadmium and copper were less than the Ontario Provincial Water Quality Objectives (PWQOs). PWQO exceedances during warm and cold seasons were noted for average concentrations of *E. coli*, total phosphorus, lead, iron, and during the winter/spring period only, for zinc and copper. Among the 17 herbicides and pesticides and 24 PAHs analyzed in this study, only pentachlorophenol and 2,3,4,6 tetrachlorophenol were found at influent concentrations above laboratory analytical detection limits in greater than 5% of samples analyzed. Effluent concentrations of both pollutants were consistently less than laboratory detection limits, suggesting that the pond was effective in reducing the likelihood that these contaminants will enter the creek.

Table 1: Average seasonal effluent concentrations and overall load based removal efficiencies for selected parameters

Parameter/season	**Summer/fall**		**Winter/spring**	
	Avg. Conc.	**Rem. Eff. (%)**	**Avg. Conc.**	**Rem. Eff. (%)**
TSS (mg/L)	48	80	39	78
Total Phosphorus (mg/L)	0.11	42	0.10	56
Phosphate (mg/L)	0.01	86	0.06	66
Ammonia (mg/L)	0.10	54	0.35	18
TKN (mg/L)	1.0	-24	1.1	31
Copper (μg/L)	4.5	48	10.2	22
Zinc (μg/L)	16.4	70	40.0	38
Lead (μg/L)	7.7	83	6	11
Chromium (μg/L)	2.4	53	2.2	-13
Cadmium (μg/L)	0.5	11	0.1	83
Oil and Grease (mg/L)	0.8	48	1.1	6

Water Temperature

Continuous water temperature measurements indicated that the facility effluent was 6 to 9°C warmer than inlet and upstream creek temperatures during the warm summer months of July and August. The average daily temperatures of the influent, effluent and creek were 14, 23 and 15°C, respectively. Outlet temperatures were frequently above the 21°C limit generally accepted as the threshold for cold water fisheries habitat. However, dilution of facility effluent by the much larger discharge volumes from German Mills Creek would likely result in relatively minor impacts on creek temperatures downstream of the facility.

Aquatic Vegetation and Algae Monitoring

Plants in wet pond treatment systems perform several functions, including bank stabilization, chemical uptake, root zone aeration, surface area attachment for bacteria and aesthetic appeal. Therefore, the type of plants established within the facility, and the success of planting programs was considered to be an important component of the overall performance assessment.

For two years, the aquatic vegetation below the high water level was monitored to determine the success of planted species in colonizing the area and extent of natural colonization by native and non-native species. Results indicated rapid natural colonization with full vegetation cover achieved after only two growing seasons (1996 and 1997). The community structure in all three cells of the facility tended towards a common group of dominant species characterized as aquatic/meadow marsh habitat. Natural colonization of both native and non-native species increased significantly in number, but the ratio of native to non-native species remained generally the same. If rapid natural colonization is found to be a common pattern at stormwater ponds and wetlands, there may be justification for reducing the number and diversity of plant species planted after construction. Although further study is required, the monitoring results suggest that cattail (*Typha*), spikerush (*Eleocharis*), rush (*Juncus*), bulrush (*Scirpus*), water plantain (*Allisma*) and waterweed (*Elodea*) may be worthy candidates for planting plans.

The use of algae as an indicator of biological response to differences in physical and chemical conditions between the forebay and wet pond was investigated. Results showed that the algal community in the forebay was generally poor and dominated by only one genus, whereas the wet pond algal community was significantly more diverse. Low diversity in the forebay was attributed to poor water quality, high and turbulent flow and cool water temperatures relative to the wet pond. Based on the algal community, the conditions in the forebay and wet pond were assessed as hypereutrophic and hypereutrophic-to-eutrophic, respectively. This assessment generally supports the concept of the forebay as a pollutant containment zone and buffer to downstream treatment cells.

Facility Maintenance

The stormwater catchment and facility was modelled to predict total flows, TSS loads, and provide information on the long-term maintenance needs of the facility. The long-term simulation indicated TSS removal efficiency of 75%, which is lower than the short-term removal rate of 80% observed during the study period. Results indicated that, in order to abide by the OMOEE's guidelines, the pond must be dredged every 16 years, with an error range between 13 and 22 years. Actual TSS accumulation within the facility should be field assessed in detail every 5 years. The forebay should also be assessed to ensure sediment deposition in this cell is not clogging the riser. More frequent dredging of the forebay would likely extend the maintenance interval for the wet pond.

Conclusions and Recommendations

Despite constraints inherent in the design of the facility and the relatively short detention time, the facility met design levels of protection with respect to contaminant removal and flow attenuation. This study demonstrates that significant water quality improvement can be achieved through retrofitting existing stormwater quantity control facilities to wet pond and wetland configurations, even in locations where significant site constraints exist.

The following recommendations are provided based on study results and site observations:

(i) Wetland performance could be improved if channelized flow through the wetland were distributed over a larger portion of the wetland via a perforated pipe or similar distribution system installed at the upstream end of the wetland.

(ii) A mid to low level drawoff configuration for the outflow structures would help to improve removal of floating contaminants (e.g. oil and grease, some organics), reduce effluent temperature and minimize adverse effects related to short circuiting across the surface of the pond. Such a structure may, however, result in decreased effluent quality because of reduced sedimentation efficiency over the mean flow path. Data from facilities with different outlet structures should be compared to assess the benefits and weaknesses associated with each design.

(iii) The feasibility of increasing the time period over which stormwater is detained within the facility should be investigated. This objective could be achieved by modifying the outlet structure such that drawdown times more closely match the average interevent period. Before implementing this measure, however, the impact on pond levels and the frequency of overflow should be carefully assessed.

(iv) Further monitoring of vegetation at the site is recommended in order to better characterize the climax community and verify tentative conclusions provided in this study.

(v) Sediment accumulation depths in the forebay and pond should be monitored regularly to determine maintenance requirements.

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1.0 INTRODUCTION

1.1 Background

Wet pond systems are among the most effective and widely applied stormwater Best Management Practices (BMP). Many wet and dry pond facilities were constructed over the past 25 years in the Toronto area and surrounding regions, primarily for flood control purposes. More recently constructed facilities, often in headwater areas, are typically designed to provide water quantity and quality control, as well as downstream erosion control. Recognizing the benefits of these improved designs on receiving waters, several municipalities have already converted, or are in the process of converting, older water quantity dry and wet ponds to multi-function wet pond water quality and erosion control facilities[1].

The Town of Richmond Hill was one of the first municipalities to undertake stormwater pond quantity-to-quality retrofits. The Harding Park retrofit involved converting a stormwater dry pond in Harding Park to a multi-cell facility consisting of a sediment forebay, wet pond and wetland. Storage volumes within the forebay and pond were sufficient to meet current erosion control objectives and enhance the quality of stormwater, while retaining the original water quantity control function. The wetland was to provide additional effluent polishing and improve wildlife corridor linkages in German Mills Creek downstream of the facility.

Located within the degraded Don River watershed, the Harding Park project is one of several community action sites identified by the Don Watershed Task Force as demonstrating techniques of regeneration at the local level (TRCA, 1994). The combined effect of many such local projects within the watershed is expected to restore health to the Don River and, in support of the Toronto and Region Remedial Action Plan (RAP), improve water quality and aquatic habitat along the Toronto waterfront.

There is a paucity of data demonstrating the effectiveness of quantity-to-quality pond retrofits, especially for pond-wetland systems, and little is known about operation and maintenance issues related to aspects such as the frequency of sediment clean out from stormwater ponds. To generate the necessary data, an agreement was made in 1995 among the Government of Canada's Great Lakes 2000 Clean-up fund (now the Great Lakes Sustainability Fund), the Ontario Ministry of Environment and Energy (OMOEE) and the Toronto and Region Conservation Authority (TRCA) to monitor the Harding Park retrofit facility under the Stormwater Assessment Monitoring and Performance (SWAMP) program. This and other SWAMP reports on stormwater treatment technologies are intended to contribute to a local data base on facility performance and provide an empirical basis for confirming and/or updating existing Ontario guidelines for stormwater facilities.

[1] See Appendix B for a glossary of terms and general discussion of concepts related to pond systems

1.2 Study Objectives

The overall objective of this study was to provide guidance to urban planners, designers and owners of stormwater facilities concerning the design, performance, and maintenance of stormwater retrofit ponds. Within this general context, the specific objectives were to:

- determine hydrologic characteristics of the study catchment and stormwater retrofit pond/wetland (*e.g.* runoff coefficient, peak flows, detention times) and evaluate these against the original design objectives;
- assess the stormwater treatment performance of the facility on an average event and seasonal load basis;
- identify aquatic plant species below the high water line of the facility and assess the effectiveness of planting plans;
- evaluate the use of algae as a indicator of spatial variations in stormwater quality within the facility;
- investigate the long-term operation, maintenance and dredging requirements of the facility;
- identify environmental benefits/limitations of the facility and provide recommendations for facility improvement.

This report presents the results of the monitoring program and discusses implications with regard to receiving water impacts. It is hoped that lessons learned from this project will offer insights into the retrofit and monitoring of other similar water quantity pond systems in Ontario.

2.0 STUDY SITE

The Harding Park stormwater quality pond retrofit is only one component of a detailed regeneration plan for German Mills Creek, which is a subwatershed of the Don River (Figure 2.1). The multifaceted nature of the regeneration initiatives is shown in the 'concept plan' for the Harding Park site presented in Figure 2.2 (TRCA, 1994). In addition to two stormwater pond retrofits, the plan includes several integrated actions aimed at improving water quality in the creek and regenerating terrestrial habitat, including pollution prevention, establishment of buffer zones, initiation of native planting programs, improvements to community access of natural areas and enhancement of aquatic and terrestrial habitats. Taken together, these actions were intended to demonstrate regeneration at the local level. An aerial photo of the study site and drainage basin after completion of the pond-wetland retrofit is shown in Figure 2.3.

2.1 Stormwater catchment area

2.1.1 Climate

The climate of the area is temperate with thermal highs and lows in all seasons moderated by a dominant lake effect from Lake Ontario, about 30 km south of the study site. The mean January temperature is -6.5°C, and the mean July temperature is 20°C. The annual average precipitation is 850 mm, and annual average snowfall is about 140 cm. Annual wind speeds average 17 km/h. In March winds are mostly from the W, NW and N at an average velocity of 18.5 km/h, and in August winds are from the SW to N at an average velocity of 13.5 km/h. Annual global solar radiation is 115 Kcal/cm^2 and mean annual net radiation is 40 Kcal/cm^2. Mean annual lake evaporation is about 750 mm, and water balance derived evapotranspiration is about 625 mm (HAC, 1978).

2.1.2 Geology, soils and topography

The geology of the area is dominated by flat-lying sedimentary carbonate rocks, limestones and dolomites. Depth to bedrock regolith in this area is about 70 to 100 m. The area is located in the St. Lawrence Lowlands hydrogeologic region (OMOEE, 1997).

A soils investigation conducted in the catchment indicated that soils consisted of clay till and some sand till, overlain by a veneer of topsoil 20 to 35 cm in depth. At the 5 m borehole depth, the overburden transitions from small to large grain size material in the range of gravel to cobble and boulders. Soils were considered to have low infiltration potential and high surface runoff potential (Soil-Eng, 1982). The catchment area lies on the southern edge of the Oak Ridges Moraine (ORM, 1994).

2.1.3 Hydrology

2.1.3.1 Surface Waters

The primary local surface water body is German Mills Creek, which drains to the Don River (Figures 2.1 and 2.3). North of the study site, the creek has a total watershed area of about 9 km^2. The headwater regions lie within the Oak Ridges moraine complex, which is an area of more permeable soils, and as a result, the Creek does not have a steady baseflow until it reaches Elgin Mills Road (Figure 2.1). Many storm sewers drain directly into the Creek just north of the study site.

2.1.3.2 Groundwater

Groundwater resources in the area are dominated by the presence of the Oak Ridges Moraine. The upper portion of the German Mills Creek watershed, above Elgin Mills Road, lies within the moraine with soils of relatively high permeability. The catchment area of the study site is located at the southern edge of this zone and is dominated by soils of lower permeability and higher runoff potential (Soil-Eng, 1982). Water level elevations at the boreholes drilled during the baseline soils investigation were between 210.2 and 212.4 meters above sea level.

2.1.4 Catchment Land Uses

Land use is medium density residential in the 16.8 ha catchment area draining to the stormwater facility. The upper reaches of the German Mills sub-watershed are dominated by urbanizing areas and existing residential and commercial land uses, north of Elgin Mills Road. South of this area land uses are a mix of residential, commercial, industrial and institutional. The catchment is about 70% serviced by storm sewers and gutters, with the remaining 30% serviced by roadside ditches and culverts. Most roof leaders within the catchment discharge to pervious areas around houses (Gartner Lee, 1995). The facility site is adjacent to a 3.5 ha recreational park. The park area (1.5 ha) was not included in the catchment area calculation, as most inputs of precipitation to the park surface would infiltrate on-site (Figure 2.3).

2.1.5 Terrestrial and Aquatic Resources

Terrestrial habitat in the upper portions of the Creek subwatershed is limited. Urban land use dominates the catchment and, consequently, forest cover is sparse and there are few natural area linkages. Most vegetation in the subwatershed is confined to riparian meadow communities.

In 1987, the creek valley was channelized to provide regional storm discharge capacity. Vegetation planting that reduces the current stream flow capacity is not permitted. However, extensive colonization of willow, dogwood shrub species and various herbaceous plants has occurred. Some planting was undertaken along the edge of the channel banks (*e.g.* pine, spruce, maple and sumac). There are no designated Environmentally Sensitive Areas (ESA) or Areas of Natural Scientific Interest (ANSI) within the study area.

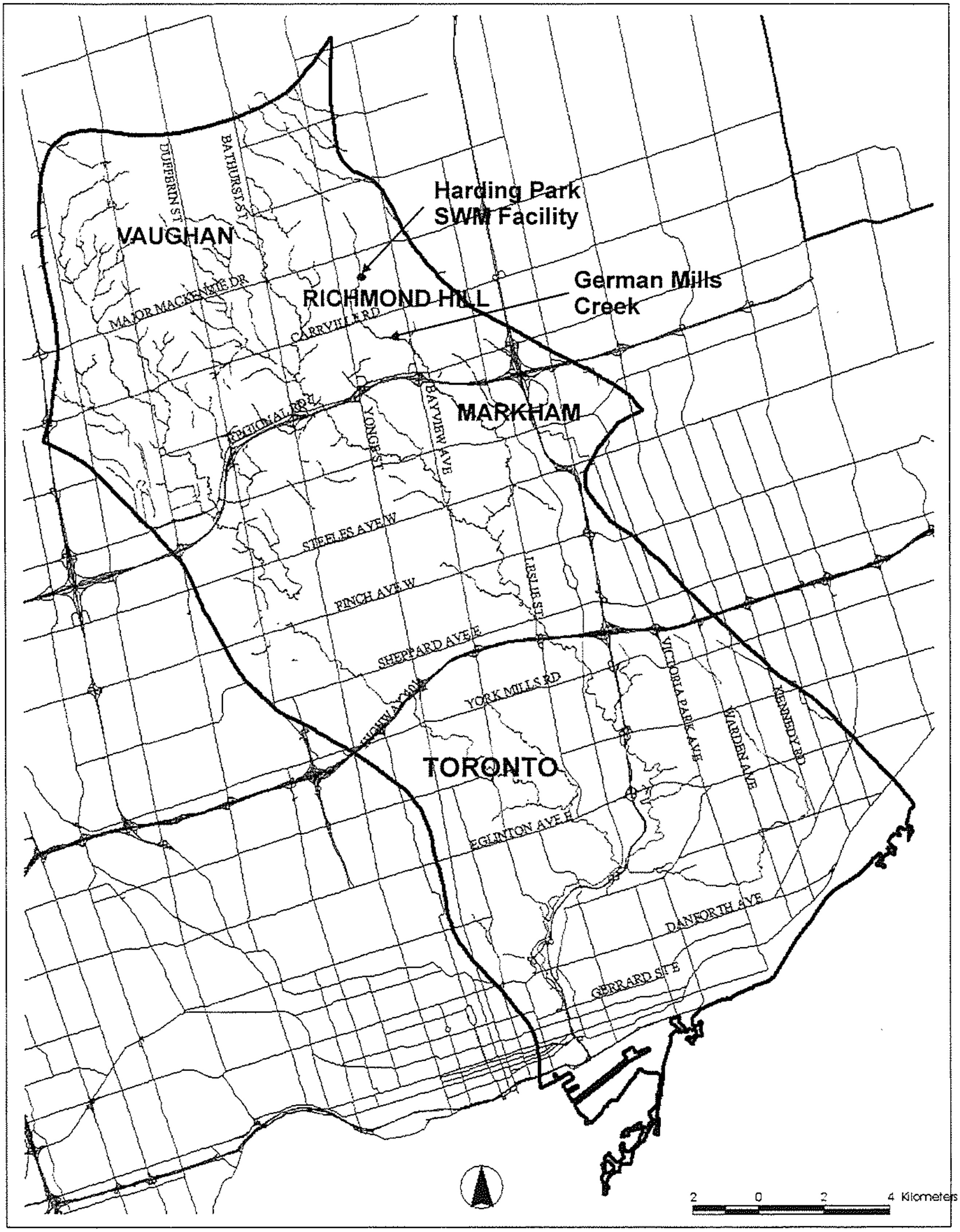


Figure 2.1: Don river watershed

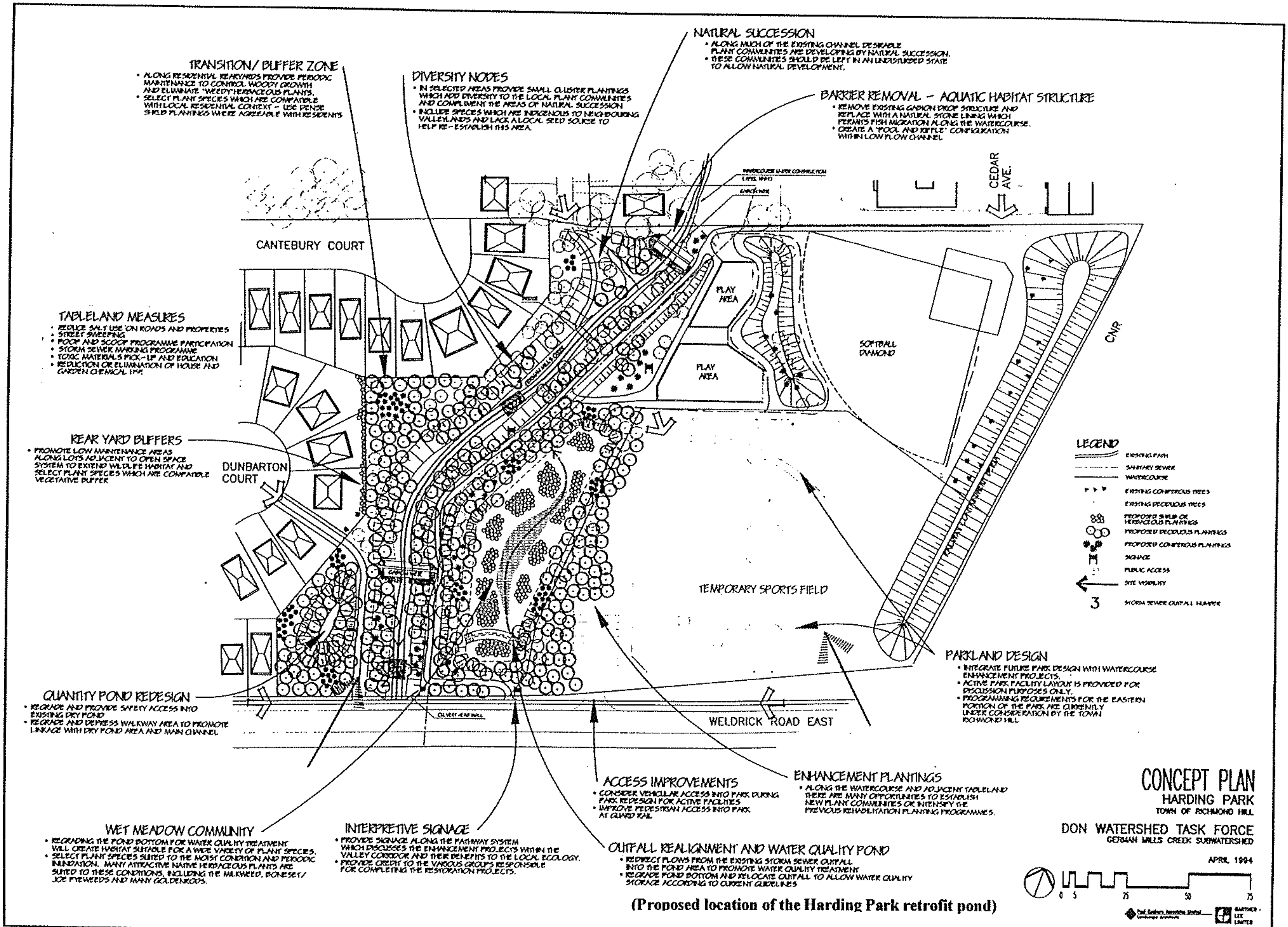


Figure 2.2: 'Concept Plan' of the Harding Park site prior to construction of the retrofit pond.

Figure 2.3: Study area

Aquatic habitat has been degraded as a result of urban development and instream erosion control works. Aquatic habitat is poor in the upper portions of the subwatershed south of Elgin Mills Road where the creek has been channelled underground through a network of sewer pipes. In general, the creek valley above the study site is trapezoidal, with a low flow channel 2 to 3 m wide and 5 to 10 cm deep. A fish inventory conducted in 1991 found six species of fish inhabiting the creek. A subsequent fish inventory conducted in 1994 identified just two species in the immediate area of the Harding Park facility (Gartner Lee, 1995). These species were blacknose dace (*Rhinichthys atratulus)* and creek chub (*Semotilus atromaculatus)*. Both species are common in urban streams and known for their high tolerance to changes in water quality and quantity. Using the Index of Biotic Integrity (IBI) and the fish inventory of 1991, an ecological quality classification of "fair" was assigned to this reach of German Mills Creek.

The creek morphology is unnatural in character as exhibited by the erratic nature of the pool-riffle-run frequency within the stream channel. A considerable amount of sediment accumulation appears to be occurring. As a result of eutrophic conditions and poor shade cover, some algal blooms were also observed. Degraded aquatic habitat has been attributed to poor water quality in the creek, low summer baseflows, narrow water level fluctuations, sediment deposition, poor instream cover, algal growth and the presence of five known instream barriers which limit the movement of fish (Gartner Lee, 1995). Aquatic habitat improves along the downstream portions of the sub-watershed.

2.2 Harding Park Stormwater Management Facility

2.2.1 Soils and Groundwater

Soils at the facility site are moderately permeable silty fine sand with some coarse sand to depths of about 1.5 m. Soils below this depth to the borehole limit of 5.0 m are sandy, silty clay and of low permeability.

An "as constructed" survey conducted in 1997 found that the bottom of the wet pond cell of the facility was between 2 and 3 m below the groundwater level observed in 1982. With increased urbanization since 1982, groundwater levels may have declined. In 1996, groundwater was not discharging into the wetland, even though the wetland cell was elevated approximately 1.5 m above the base of the wet pond. Low flow levels in German Mills Creek reach immediately adjacent to the facility are about 50 cm below the bottom of the wet pond.

2.2.2 Vegetation and Aquatic Biota

The terrestrial vegetation community of the study site was mapped in 1994. Dominant tree species were Crack willow (*Salix fragilis),* Manitoba maple (*Acer negundo*) and Chinese elm (*Ulmus pumila)* (Gartner Lee, 1995). Grasses and herbs were dominated by a larger group including Reed Canary grass (*Phalaris arundinacea)*, Kentucky Bluegrass (*Poa pratensis)*, Smooth Broome (*Bromus inermis)*, Spotted Jewelweed

(*Impatiens capensis*), Asters (*Aster lateriflorus* and *nova - angliae*), Thistle (*Cirsium arvense*) and Ticks (*Bidens frondosa*). The vegetation community was assessed to be an immature, early successional community (RHN, 1994; Gartner Lee, 1995).

2.2.3 Facility design features

The former dry pond covered 0.4 ha and had a total storage capacity of 1650 m^3. In 1995, this dry pond was retrofitted into a three-cell system consisting of a sediment forebay, a wet pond, and a small wetland. A survey drawing of the retrofit facility is presented in Figure 2.4. The sediment forebay and wet pond have permanent pool volumes of 15 and 1000 m^3, respectively. The extended detention volume of the forebay and wet pond is 1950 m^3. The sediment forebay, wet pond and wetland cells are each separated by aggregate berms inlaid with impermeable geotextile. Hickenbottom risers wrapped with geotextile provide hydraulic control from the forebay to the wet pond and again from the wet pond to wetland. French drains (emplaced

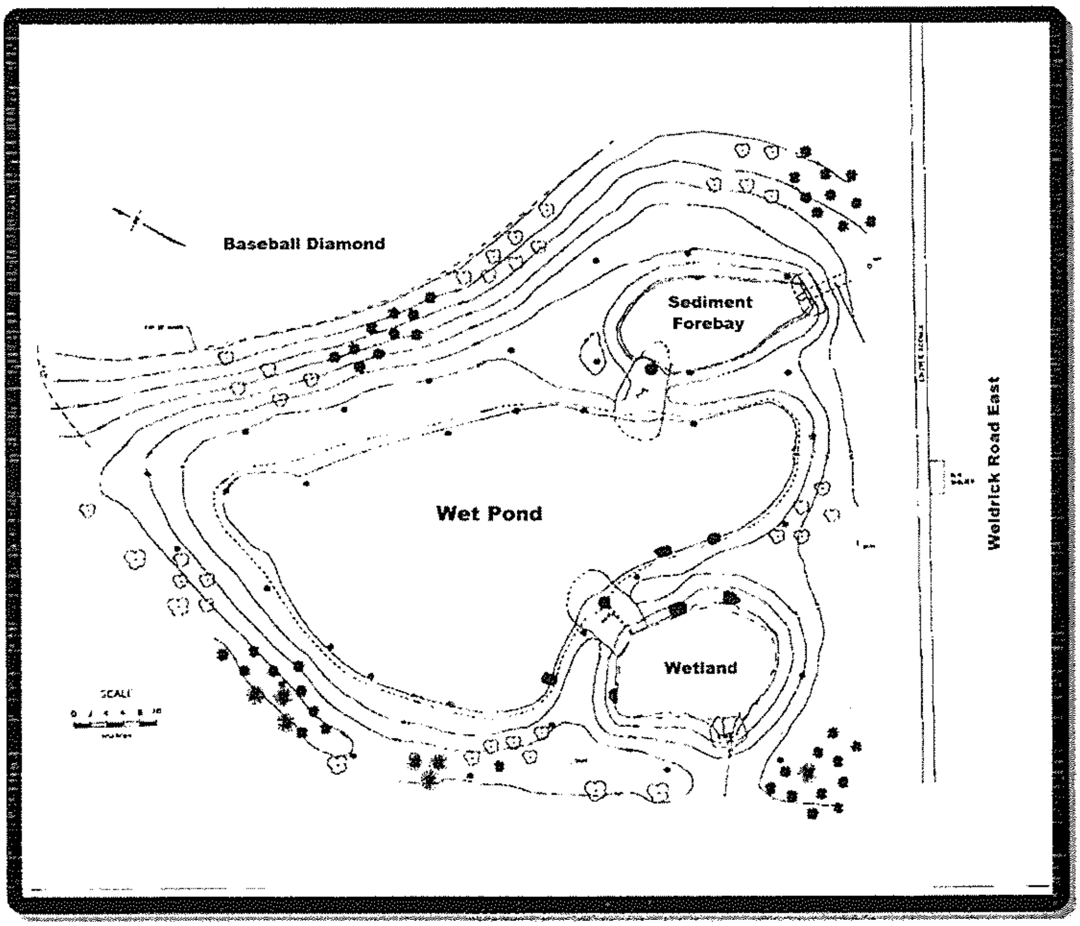


Figure. 2.4: Survey drawing of Harding Park Stormwater Management Facility.

sand lenses) were installed within the berm between the wet pond and wetland to provide moisture input to the wetland during dry weather periods. The wetland does not have a permanent pool and is primarily vegetated with emergent macrophytes. The facility drains west from the wetland to German Mills Creek, about 20 m downstream.

Table 2.1: OMOEE (1994a) design guidelines compared to Harding Park design features

Design Feature	**Design Objective**	**OMOEE (1994a) Guidelines**	**Harding Park Facility**
Permanent pool depth (m)	minimize resuspension; avoid anoxic conditions	1-2 (mean); 3 (max.)	1.8 max
Permanent pool volume (m^3/ha)	provision of level 2 fisheries protection	60*	60
Max. extended detention depth (m)	storage & flow control	1 to 1.5	approx. 2.8
Extended detention volume (m^3/ha)	provision of level II fisheries protection	40	116
Drawdown time (hours)$^{+}$	suspended solids settling	24	26^{++}
Length-to-width ratio	minimize short circuiting	at least 3:1	1:1
Graduated planting strategy	safety, aesthetics, recreational amenity, shading for temperature control	five zones – aquatic to upland	terrestrial and meadow marsh planting; natural regeneration of aquatic plants

* based on level 2 fisheries protection and 45% surface imperviousness (OMOEE, 1994a)
$^{+}$ The SWMP manual (OMOEE, 1994a) suggests using 'drawdown time' as an approximate measure of 'detention time'.
$^{++}$ calculated using equations provided in the SWMP manual (OMOEE, 1994a)

Table 2.1 compares the Harding Park wet pond design parameters to the Ontario Ministry of Environment and Energy's (1994) wet pond guidelines for level 2 fisheries protection, assuming 45% surface imperviousness. In compliance to the regulatory agency design criteria, the area of the cells within the facility was expanded from 0.4 ha to about 0.7 ha. Due to site constraints, the minimal length-to-width requirement of 3:1 could not be accommodated at the site. The pond meets the recommended guidelines (OMOEE, 1994a) for maximum permanent pool wet pond depth, permanent pool volume, extended detention volume above the permanent pool and drawdown time. The OMOEE extended detention depth restriction was intended to protect plant species on the pond banks and prevent the overall depth (permanent pool plus extended detention) from exceeding 5 meters. The Harding Park extended detention depth exceeded the guideline by approximately 1.3 m, but the overall depth was still below the recommended 5 m limit. Extended detention volume in excess of the OMOEE water quality guideline was intended to provide for erosion control of 24 hours detention for a 25 mm storm (4 hour Chicago distribution).

Using equations provided in the SWMP manual (OMOEE, 1994a), the design active volume 'drawdown time' for a 25 mm precipitation event was estimated at 26 hours. The SWMP manual suggests using 'drawdown time' as an approximate measure of 'detention time', but in practice, the two measures often differ substantially. The actual detention time of a fluid element passing through the facility is difficult to estimate prior to monitoring, which may explain why the 'drawdown time' was suggested as a substitute. The term 'drawdown time' is employed in Table 2.1 to highlight the designer's attempt to meet the 24 hour 'detention time' guideline.

3.0 STUDY APPROACH

3.1 Monitoring Program

Assessment of Harding Park facility performance was based on co-ordinated measurements of runoff volumes, water quality and water temperature at the inlet and outlet during the summer/fall period (May to November 1996 and 1997), and grab samples for water quality during the winter/spring period (December to April, 1996 and 1997). During the summer/fall period, separate assessments of wetland vegetation and pond algal community dynamics were undertaken to provide additional insights into the effectiveness of the planting program and ecological status of the pond. Details on instrumentation and statistical methods employed in collecting and analyzing these data are provided below.

3.1.2 Rainfall and Runoff Monitoring

The location and descriptions of monitoring equipment are presented in Table 3.1. Rainfall data were obtained during 1995 and 1996 from Toronto Buttonville Airport, located about 7 km southeast of the facility. During 1997, rainfall data were collected at the Steelworkers Co-op, adjacent to the facility, using a standard tipping bucket rain gauge connected to an Ultralogger™ data logger.

Table 3.1: Location and Description of Monitoring Stations

Station	Description	Quantity
Inlet	Automatic Sampler (Composite Samples)	1
(1050 mm dia.	Flow Logger	1
Storm Sewer)	Temperature Logger	1
	Grab Samples (during the winter/spring)	As Appropriate
Wetland outlet	Automatic Sampler (Composite Samples)	1
(450 mm dia.	Flow Logger	1
Outlet Pipe)	Temperature Logger	1
	Grab Samples (during the winter/spring)	As Appropriate

Hydrologic data were collected from monitoring stations at the inlet and outlet of the facility. The inlet flow monitor was installed approximately 100 m upstream of the facility in a 1050 mm diameter inlet pipe. At the wetland outlet, the flow meter was located at the invert of a 450 mm diameter pipe. Montadoro-Whitney Q-loggers™ were employed at both locations until the spring of 1997 when the inlet Q-logger was replaced with an ISCO™ 4250 flow meter and area-velocity probe.

During the fall of 1997 a depth sensor was installed in the Hickenbottom structure between the wet pond and wetland to determine the active stage-storage relationship of the wet pond. This relationship was employed to determine the active storage and head in the wet pond and, subsequently, to verify the discharge from the wet pond via the orifice in the Hickenbottom riser structure.

As shown in Figure 3.1, a discontinuity in discharge occurred up to the 280 mm active depth level. The discharge discontinuity below this level resulted from clogging by accumulated debris in the rip-rap surrounding the Hickenbottom orifice. Above this point, flow was log-normal as demonstrated by the linearity of the discharge rate when plotted on a logarithmic scale. This discharge discontinuity resulted in observed time lags between the inlet and outlet hydrographs ranging between 2.5 and 18 hours, depending on influent runoff volumes.

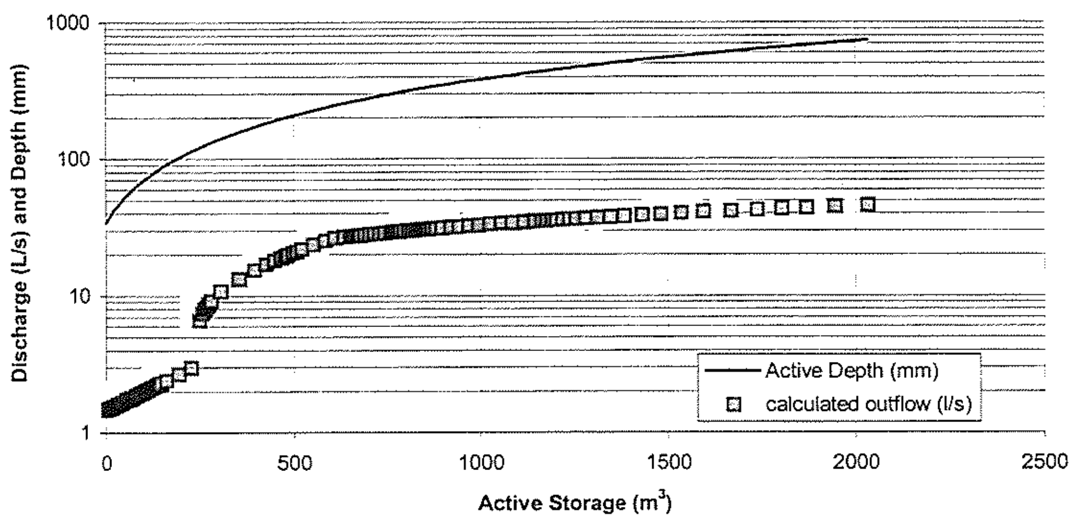


Figure 3.1: Stage-storage (measured) and discharge-storage (calculated from equation 3) relationships for the Harding Park retrofit facility.

Due to the discontinuity in discharge, outlet flow rates were estimated based on two equations. Up to about 280 mm of head (approximately 600 m^3 of storage volume), the head-to-volume relationship was characterized by:

$$\textit{Head (mm)} = 26.46e^{0.0039V} \quad R^2 = 0.97 \qquad (1)$$

where:

V = active storage volume in the wet pond

Above the 280 mm point, no discontinuity in discharge was observed and the head level equation was linear such that,

$$\textbf{\textit{Head (mm)}} = 0.3458V + 74.223 \quad R^2 = 0.99 \qquad (2)$$

Using the above two equations, the following derivation of Bernouli's equation was then used to estimate discharges from the Hickenbottom riser orifice at a given head level (Veissman and Lewis, 1996; Bedient and Huber, 1988, Ferguson and Debo, 1990):

$$Q = CA(2gh)^{0.5} \qquad (3)$$

where:

Q = outlet discharge (m^3/s)
$C = (C_v \times C_c)$ where C_v represents the coefficient of velocity (0.99) and C_c represents the coefficient of compression (0.66)
A = area of the orifice (0.0176 m^2)
g = acceleration due to gravity (9.81 m/s^2)
h = active head depth (m)

The head level provided a measure of dynamic head by discounting from the active volume the amount discharged via the outlet on a five-minute interval. The result was a quantitative method based on empirical data, which described discharge at the outlet reasonably well. The model was verified by comparing outlet flow data from three storms, for which reliable data existed, with the estimated flow data. These three storms were October 18/96, October 26/97 and November 1/97. The correlation coefficients between data sets were 0.83, 0.89 and 0.88, respectively. In light of the paucity of outlet flow data arising from the challenges encountered in collecting reliable flow data from the short and steeply configured outlet pipe, the quantitative model was thought to be a reasonable means of estimating discharge from the outlet.

The influent and effluent volumes were used in a two-element, hydrologic mass balance of the facility, and in the calculation of water quality performance. Groundwater recharge/discharge and evapotranspiration were not monitored because total losses/gains due to these processes were considered negligible relative to event runoff volumes. The estimated annual lake evaporation for this area is about 75 cm and stormwater ponds evaporate approximately the same volume as lakes (HAC, 1978; Kadlec and Knight, 1996). Therefore the estimated annual evaporation would be 1500 m^3, which is small compared to the annual facility runoff input and considering that 1700 m^3 is deposited on the facility in the form of direct precipitation.

3.2 Water Quality Monitoring

3.2.1 Temperature Measurement

Temperature data were continuously logged at 15-minute intervals at the inlet and outlet of the pond, and in German Mills Creek immediately upstream of the pond outlet from May to November 1996. Temperature variations for the months of June, July and August were compared to the suggested limit of 21 C for cold water fisheries.

3.2.2 Water Quality Sampling

During the summer/fall (May to November) monitoring period, water quality samples were collected at the inlet and outlet using ISCO™ 3700 automated wastewater samplers. Samplers were interfaced with flow loggers to collect flow proportioned samples for the duration of each storm. Composite samples collected in this manner were considered to represent the overall mean concentration during storm events, and are hereafter referred to as the Event Mean Concentration (EMC). During the cold season (December to April), monitoring equipment was removed and grab samples were collected at both the inlet and outlet locations. Samples collected in this manner may not represent the actual mean concentration of storm events and, therefore, caution should be exercised in interpreting results.

Samples were submitted to the Ministry of Environment and Energy Lab in Toronto and analyzed following principles outlined in *Standard Methods* (Eaton *et al.*, 1995) for metals, nutrients (P and N), bacteria, organics, and general chemistry. Particle size analysis of suspended solids was undertaken using an optical laser light diffraction method and results were reported by size class in percent by volume. Appendix C summarizes the analytical procedures used in this study.

3.2.3 Statistical methods

Samples were collected from 37 storms at the inlet and 16 storms at the outlet (includes post-berm repair period only) from December 1996 to November 1997. Statistical analysis of water quality parameters was performed using a software package developed by the Ministry of Environment and Energy for use in stormwater quality constituent analysis (Maunder *et al.*, 1995). The package was configured to derive several important statistical parameters, including mean concentrations, upper and lower 95% confidence intervals, standard deviations and estimates of left-censored data. Left-censored data are data below the lower detection limit of the analytical lab equipment. The statistical package accounted for left-censored data by assigning values using Probability Distribution Estimation (PDE) techniques and other statistical methods. This feature was very useful in the analysis of some metals and organic constituents, which were often found at trace concentrations. All statistical calculations were based on log normal probability distributions.

Load-based removal efficiency (*LE*), which requires both the flow volume (V) and event mean concentration (EMC) of constituents, could only be calculated for events during the summer/fall period when the required data were available. Facility removal efficiency during the summer/fall was derived for each sampled event according to the following equation,

$$LE = \left[\frac{[(V_i \times EMC_i) - (V_O \times EMC_O)]}{(V_i \times EMC_i)} \right] \times 100\% \qquad (4)$$

where: o = outlet
i = inlet

The seasonal load based removal efficiency (*SLE)* for the entire summer/fall season is based on cumulative loads, as follows:

$$SLE = \left[\frac{\sum_{j=1}^{m} [(V_{i_j} \times EMC_{i_j}) - (V_{O_j} \times EMC_{O_j})]}{\sum_{i=1}^{n} [V_{i_j} \times EMC_{i_j}]} \right] \times 100\% \qquad (5)$$

where: m = total number of storm events monitored

In the winter/spring season, when flow was not monitored and grab samples were collected, concentration-based removal efficiency (*CE*) was determined by assuming that there was a good hydrologic water balance in the facility (*i.e.* the flow volume entering the pond within an individual event was equal to the volume leaving the pond), such that,

$$CE = \left[\frac{(C_i - C_O)}{C_i} \right] \times 100\% \qquad (6)$$

where: C_i = influent constituent concentration (grab sample)
C_O = effluent constituent concentration (grab sample)

The assumption of a perfect water balance in the winter is probably valid since losses through evaporation and infiltration in the winter are generally very low. During the summer, only minor differences in baseflow entering and exiting the facility were observed.

3.3 Vegetation Monitoring and Aquatic Community Assessment

A vegetation study was conducted from 1995 to 1997 at the Harding Park facility and one other stormwater wet pond in the Greater Toronto Area. The goal of the study was "to develop a list of recommended vascular wetland plant species and recommended planting strategies for stormwater management pond projects in the Greater Toronto Area". The study surveyed the growth and development of planted and naturally colonized plant species within the facility. Results of the study are presented in Appendix D and summarized in Section 6.1.

A study investigating the structure and dynamics of algal communities in the Harding Park facility was also conducted. This study documented the baseline conditions of the summer phytoplankton and periphyton communities, compared in-facility community structure and reports on the relevant physical and chemical monitoring conducted in conjunction with the algal monitoring. Algae were used as an indicator of ecological and water quality conditions of the forebay and wet pond. The results and methods of the study provided in Appendix E are summarized in section 6.2

3.4 Stormwater Facility Modelling

The hydraulic and hydrologic conditions of the stormwater catchment area and facility were modelled using the Stormwater Management Model (SWMM 4.3) developed for the EPA and run as the engine within the PCSWMM™ shell. The model simulates influent and effluent hydrographs and predicts sediment accumulation rates within the facility based on measured flow volumes, TSS loads and twelve years of rainfall recorded near the study site. Estimates of sediment accumulation rates were subsequently used in conjunction with OMOEE guidelines to determine maintenance requirements.

4.0 WATER QUANTITY ANALYSIS

In January 1996, the top portion of the berm separating the wet pond from the wetland cell eroded away, resulting in frequent overflows during storm runoff events. When the berm was repaired in late August 1996, the design outlet peak flow was revised upwards from 40 L/s (Garner Lee, 1995) to 52 L/s. The failed berm and subsequent design change after repair influenced the hydraulic function of the facility. Thus, the water quantity and quality analyses included in this report were based on data collected after the berm was repaired, from September 1996 to November 1997.

The post-berm repair data set includes 6 large storms (greater than 20 mm), 8 medium-sized storms (10 to 20 mm) and one small storm (less than 10 mm). Rainfall and runoff data for the entire monitoring period are presented in Table 4.1. Storm runoff hydrographs and rainfall hyetographs for four representative storms are presented in Figure 4.1.

4.1 Rainfall-runoff

The 1997 monitoring period was drier than 1996, had a lower average rainfall intensity and included several long (greater than 10 day) inter-event periods. During both years, negligible influent was observed for storms with less than 4 mm of rainfall, probably due to depression storage and infiltration of surface runoff in roadside ditches. This observation approaches the 5 mm level suggested in the SWMP manual (OMOEE, 1994a) for stream baseflow maintenance.

As expected, total rainfall during storm events correlated well with total inflow ($r = 0.89$). However, mean rainfall and mean inflow were not well correlated, nor was there a strong relationship between maximum rainfall intensity and peak inflow rates. Both correlations may have been stronger if the 1996 rain data had been collected from Harding Park, rather than Buttonville Airport, 7 km southeast of the study site. Alternatively, factors other than rainfall may be important determinants of mean and peak inflow rates.

4.2 Runoff Coefficient

Runoff coefficients calculated in Table 4.1 represent the fraction of rainfall volume converted to stormwater runoff during an event. The mean runoff coefficient for the 1996/97 study period was 0.34, ranging between 0.21 and 0.57. This mean runoff coefficient is within the expected range for a fully developed catchment with 45% impervious cover (Schueler, 1995).

Table 4.1: Summary of Hydrologic Statistics

Date	Rain					Inlet						Outlet								
	Total Rainfall (mm)	Duration (hr:min)	Mean Intensity (mm/hr)	Maximum Intensity (mm/hr)	Interevent Period (days)	Total Runoff (m^3)	Baseflow (m^3)	Flow Duration (hr:min)	Average Flow (L/s)	Peak Flow (L/s)	Lag Time (hr:min)**	Total Runoff (m^3)	Baseflow (m^3)	Flow Duration (hr:min)	Average Flow (L/s)	Peak Flow (L/s)	Lag Time (hr:min)**	Balance (%) +	Runoff Coefficient	Hydraulic Detention Time (hr:min)*
Before Berm Repair																				
7-Jun-96	14.8	12:18	1.2	3.8		610	93	15:30	10.9	126	4:12								0.21	
22-Jun-96	11.2	5:54	1.9	8.6		679	53	12:00	15.7	552	5:57								0.33	
24-Jun-96	9.2	7:42	1.2	3.0	2.00	448	49	9:00	13.8	103	2:28								0.26	
29-Jun-96	28.6	6:00	4.8	13.4	5.00	1742	100	17:25	27.8	490	4:17								0.34	
7-Jul-96	15.2	4:00	3.8	13.2	8.00	797	73	15:30	14.3	499	2:52								0.28	
15-Jul-96	36	15:00	2.4	5.8	5.90	1461	86	15:40	25.9	218	5:40								0.23	
19-Jul-96	24	10:54	2.2	5.6	3.75	1538	70	17:20	24.6	426	2:44								0.36	
24-Jul-96	7.8	1:00	7.8	7.8	5.25			unreliable data												
29-Jul-96	15.8	17:36	0.9		5.20			unreliable data												
1-Aug-96	9.9	11:00	0.9	6.8	3.00	685	75	13:50	13.8	439	4:18								0.37	
8-Aug-96	36	4:00	9.0	20.0	6.40	1341	33	8:35	43.4	778	3:39								0.22	
After Berm Repair																				
7-Sep-96	64	24:36	2.6	10.2	7.25	5838	402	21:35	75.1	339	15:45	5764	385	111:40	14.3	55	26:28	1.0	0.51	10:43
13-Sep-96	19.6	16:18	1.2	7.4	1.50	1471	150	18:30	22.0	105	5:40	1465	150	30:52	13.2	38	11:38	0.5	0.40	5:58
24-Sep-96	17.8	8:54	2.0	4.8	1.80	1207	85	17:10	19.5	157	3:47	1118	85	29:09	10.7	28	9:38	7.9	0.38	5:50
27-Sep-96	23.8	26:24	0.9	3.8	2.70	1785	247	30:00	16.6	171	21:45	1746	454	50:53	9.8	30	26:20	16.0	0.38	4:35
18-Oct-96	30	30:00	1.0	7.4	8.00	3189	327	13:15	66.9	207	6:32	3013	392	55:00	15.2	69	12:12	8.4	0.57	5:40
29-Oct-96	8.8	10:00	0.9	5.2	6.40	703	102	9:15	21.1	269	6:47	632	135	26:03	6.7	16	13:20	17.3	0.41	6:32
7-Nov-96	12.3	10:18	1.2	4.4	2.50	1239	170	27:30	12.5	248	20:12	1129	170	43:19	7.2	23	24:06	10.3	0.52	3:54
8-Jul-97	15.25	8:00	1.9	6.0	10.20	953	425	16:45	15.8	66	7:53	774	249	23:05	9.3	18	11:58	0.6	0.21	4:05
20-Aug-97	26.25	15:00	1.8	6.6	4.00	1017	150	22:00	12.8	43	11:38	824	156	33:20	6.9	16	15:53	23.0	0.20	4:15
6-Sep-97	13.5	6:00	2.3	5.2	16.50	704	154	25:00	7.8	114	21:42	605	147	54:26	3.1	14	25:15	16.7	0.24	3:33
10-Sep-97	18.75	12:00	1.6	5.0	3.40	993	176	16:10	17.1	50	9:06	791	78	29:00	7.6	16	13:30	12.7	0.26	4:24
29-Sep-97	13.75	13:48	1.0	2.6	3.60	1348	85	39:00	9.6	34	26:06	1143	131	78:25	4.0	13	30:12	19.9	0.55	4:06
26-Oct-97	31	19:24	1.6	6.2	14.00	1735	256	33:50	14.2	55	12:45	1603	239	47:25	9.4	25	17:45	7.8	0.28	5:00
1-Nov-97	19.75	14:06	1.4	5.6	1.00	1344	102	22:55	16.3	58	7:45	1201	131	36:23	9.2	22	13:20	13.8	0.37	5:35
Means																				
All data	20.9	12:24	2.3	7.0	5.54	1427	151	19:14	22.5	263	9:03								0.34	
After Berm Repair	22.5	15:20	1.5	5.7	5.9	1680	202	22:21	23.4	136.9	12:40	1558	207	46:21	9.0	27.4	17:58	11.1	0.38	5:17
1996	21.4	12:19	2.5	7.7	4.67	1573	132	16:49	26.3	346	7:04	1482	253	41:50	11.0	37	14:01	6.5	0.36	6:10
1997	19.8	12:36	1.6	5.3	7.53	1156	193	25:05	13.4	60	13:50	992	162	43:09	7.1	18	18:16	13.5	0.30	4:25

Notes:

+ Balance = (Net Inlet Runoff - Net Outlet Runoff) / Net Inlet Runoff, where net runoff represents total runoff - baseflow

* Hold-back time is calculated as the time delay between the inlet and outlet hydrograph centroids.

** Lag time represents the time delay from the start of rainfall to the influent or effluent hydrograph centroid

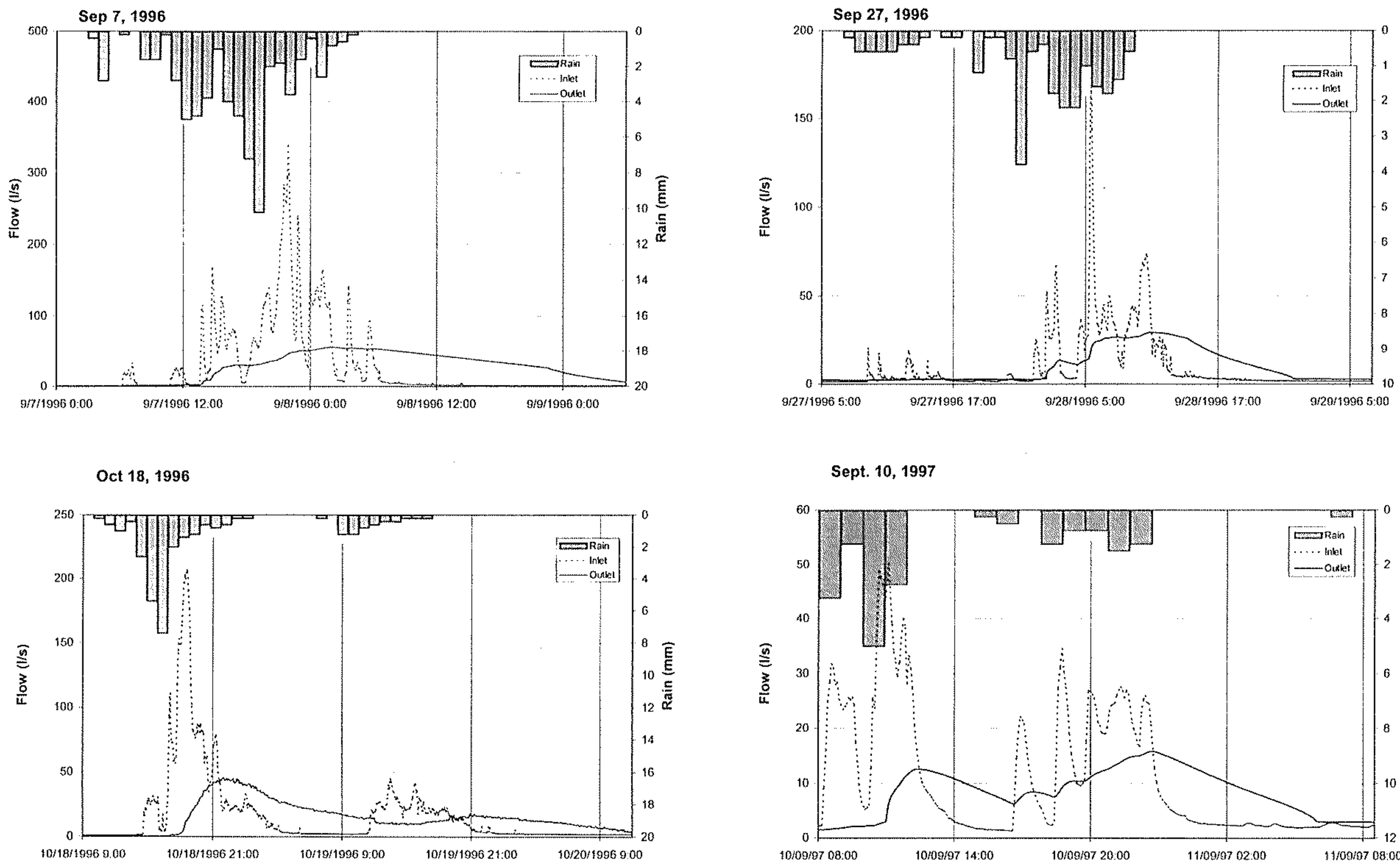


Figure 4.1: Hydrographs and hyetographs of four representative storm events during the study period

The 1996 monitoring period had a higher average runoff coefficient (0.36) than was observed in 1997 (0.30). The discrepancy may be a consequence of differences in rainfall characteristics. Wetter conditions and shorter interevent periods in 1996 would normally be expected to result in higher coefficients. Alternatively, the difference may simply reflect the location at which rainfall data were collected during the two years. As discussed above, rainfall data were collected in 1996 at Buttonville Airport, whereas in 1997 data were collected at the Steelworker's Co-op adjacent to the facility.

4.3 Peak Flow Attenuation

Peak flow rates were substantially attenuated as stormwater passed through the facility (Figure 4.1). From September 1996 to November 1997, the mean peak flow was 137 L/s at the inlet, compared to only 27 L/s at the outlet. After the berm was repaired, the design outlet peak flow was 52 L/s and the facility had a storm runoff storage capacity equivalent to a 5-year, 4-hour storm volume estimated at 2300 m^3. Only two storms (September 7 and October 18/96) exceeded this capacity and both had measured outlet peak flows (55 and 69 L/s, respectively) beyond the 52 L/s design range.

4.4 Volumetric Flow Balance

The mean flow balance during storm events was 11% (ranging from 0.5 to 23%), indicating that more water entered the facility via the inlet than exited by the outlet (excluding differences in baseflow). Due to inherent limitations of the monitoring equipment and frequent low flow conditions, a balance within ±15% cannot be used to conclusively indicate groundwater recharge or discharge within the facility. During dry weather, mean inlet and outlet baseflows were 1.5 and 1.3 L/s respectively, suggesting that water losses or gains were negligible at permanent pool water levels.

4.5 Lag Times

The lag times shown in Table 4.1 were calculated from the start of rainfall to the centroid of inlet and outlet flow hydrographs. The mean lag time for the post berm repair period was approximately 13 and 18 hours at the inlet and outlet, respectively. Variations among individual events are attributed to differences in rainfall intensity and storm flow duration.

The time delay between the start of rainfall and the start of inlet runoff, and between peak rainfall and peak inlet runoff averaged approximately 1.5 hours. The time delay in both instances was likely influenced by depression storage, overland runoff delay from roadside ditches, and the size of the catchment.

4.6 Hydraulic Detention and Plug Flow Residence Times

The hydraulic detention time provides a general measure of the extended detention feature of the facility, by which stormwater flow is temporarily detained or 'held back' within the facility. The hydraulic detention time is calculated as the time delay between inlet and outlet hydrograph flow centroids.

The outlet control structure at the wet pond and, to a lesser extent, the forebay have a strong influence on the detention time, the drawdown time, and the duration of outlet flow. For storms monitored from September 1996 to November 1997, the hydraulic detention time averaged 5 hours and 17 minutes, the drawdown time generally exceeded the 24 hour erosion control target (OMOEE, 1994a) for large storms (*i.e.* greater than 20 mm), and the mean duration of flow at the outlet was more than twice as long (46 hours) as the mean flow duration at the inlet (22 hours) (Table 4.1). The 46-hour outlet flow duration compares to an average inter-event time of five and a half days, indicating that, after most storm events, permanent pool levels were re-established in the facility well before the onset of the next storm. The difference between the mean interevent period and flow duration may provide justification for modifying the outlet structure to prolong the time over which stormwater is detained within the facility.

The hydraulic residence time (or retention time) of runoff within the facility could not be determined from the available data, but if plug flow displacement conditions (*i.e.* no mixing, no short circuiting) are assumed, the facility residence time may be crudely estimated based on permanent and active storage volumes, and the average inflow rate during storm events. At a mean runoff coefficient of 0.34, a rainfall event of greater than 18 mm would be required to totally displace the permanent pool (1015 m^3) volume. During the summer/fall periods of 1996/97, there were 11 such storms. The average flow rate during these 11 storms was 30.7 L/s, which would completely displace the permanent pool and half the extended detention volume (1990 m^3) after 18 hours. During storms with influent volumes of less than 1015 m^3, the plug flow residence time would, of course, be much longer since influent volumes would not be sufficient to displace the permanent pool. Conversely, under non-plug flow conditions (i.e. some short circuiting, some influent-permanent pool mixing), the hydraulic residence time would be less than 18 hours.

Extending this analysis to the entire summer/fall season, it was possible to provide an estimate of the seasonal average residence time. Based on a typical summer/fall year with 464 mm of rain (AES station, Toronto Bloor, 1951-1980), a mean runoff coefficient of 0.34, and an average baseflow rate of 1.5 L/s, a total flow of approximately 54,238 m^3 (26,504 m^3 of storm runoff and 27,734 m^3 of baseflow) would enter the facility. Assuming plug flow conditions, this volume would be sufficient to displace the permanent pool volume an average of 7.6 times per month during the period from May 1 to November 30. Thus, the average seasonal residence time for the facility would be 4.0 days. By comparison, the seasonal average dry weather baseflow residence time is about 7.7 days.

5.0 WATER QUALITY ANALYSIS

The water quality data were partitioned into two "seasons": the winter/spring season, from December 1 to April 30, and the summer/fall season, from May 1 to November 30. As indicated previously, the focus of the analysis is on the post-berm repair period, from September 1996 to November 1997. Prior to this period, erosion of the berm resulted in effluent suspended solids concentrations ranging from 18 to 1340 mg/L during the summer/fall monitoring period, and 29 to 241 mg/L during winter/spring. Since the facility did not operate according to design at the time of berm failure, results from this period were not analyzed in detail.

Inlet and outlet water quality summary tables for the pre and post-berm repair periods are presented in Appendix F. Since failure of the berm did not affect influent data, seasonal summary statistics at the influent monitoring station (*e.g.* seasonal average EMCs, standard deviations, 95% confidence limits, etc.) represent the pre and post-berm repair monitoring period. Removal efficiencies for individual events are calculated from paired inlet/outlet data during the post-berm repair period and are provided in Appendix G.

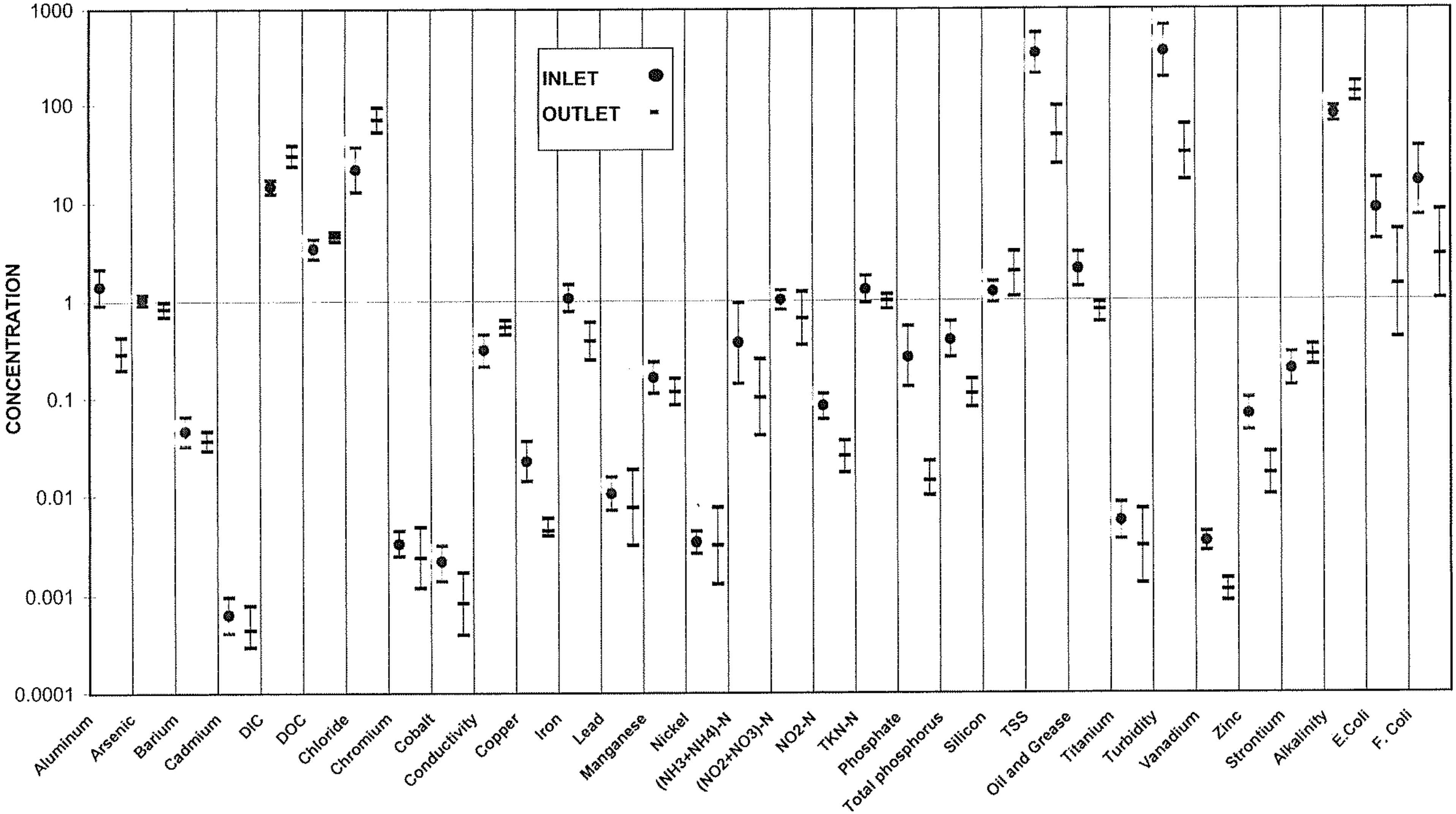
The water quality summary for the summer/fall period represents 22 samples at the inlet, including the period prior to berm repair, and 12 post-berm repair samples at the outlet. The winter grab sample data set includes 15 samples at the inlet and only 4 at the outlet (post-berm repair period only). Due to the small number of outlet samples, winter results should be interpreted with caution. The results and discussion are organized by pollutant group, starting with total suspended solids (TSS) and particle size distribution, and followed by general chemistry (*e.g.* pH and alkalinity), organics, bacteria, nutrients and metals.

Figure 5.1 presents mean summer/fall concentrations and 95% confidence limits for parameters observed at detection frequencies greater than 10%. The summer/fall load-based performance results are presented in Figure 5.2. The mean winter/spring concentrations and concentration-based removal efficiencies are presented in Figures 5.3 and 5.4, respectively.

5.1 Total Suspended Solids

Total suspended solids (TSS) concentration is a critical parameter in stormwater because many stormwater contaminants are strongly associated with suspended solids (Randal *et. al.*, 1982, Neary et. al, 1988). Consequently, TSS concentrations are often used to gauge the general level of water quality improvement in stormwater management facilities. Elevated TSS concentrations can also adversely effect stream benthos, primary photosynthetic productivity, and fish habitat (Waters, 1995).

During the winter/spring period (which includes the pre-berm repair period at the inlet) the mean influent and effluent suspended solids concentrations from grab samples were 270 and 40 mg/L, respectively. The average concentration-based TSS removal efficiency during the post-berm repair period only was 78%, and ranged from a low of 58% to a high of 97%.



NOTE: All concentrations in mg/L, except conductivity (mS/cm), Turbidity (FTU) and bacteria parameters (colonies/100ml x 10^{-3}). This graph is intended to show relative statistical differences between the inlet and outlet means. For numerical values and additional statistical data, see Tables in Appendix E.

***Figure 5.1*:** Average Event Mean Concentrations (AEMCs) and 95% confidence limits at the inlet and outlet for the summer/fall period (September to November 1996; May to November, 1997)

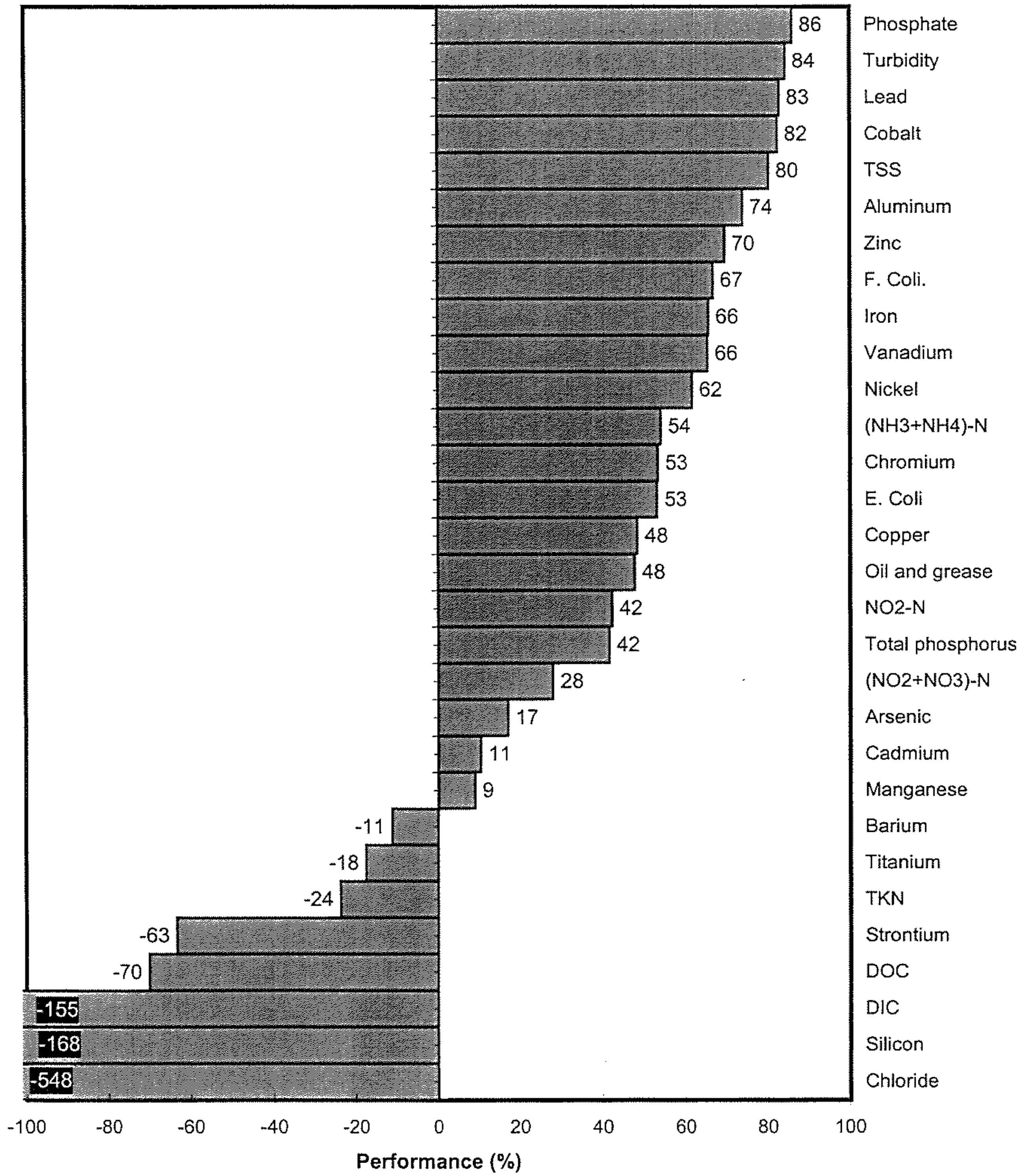
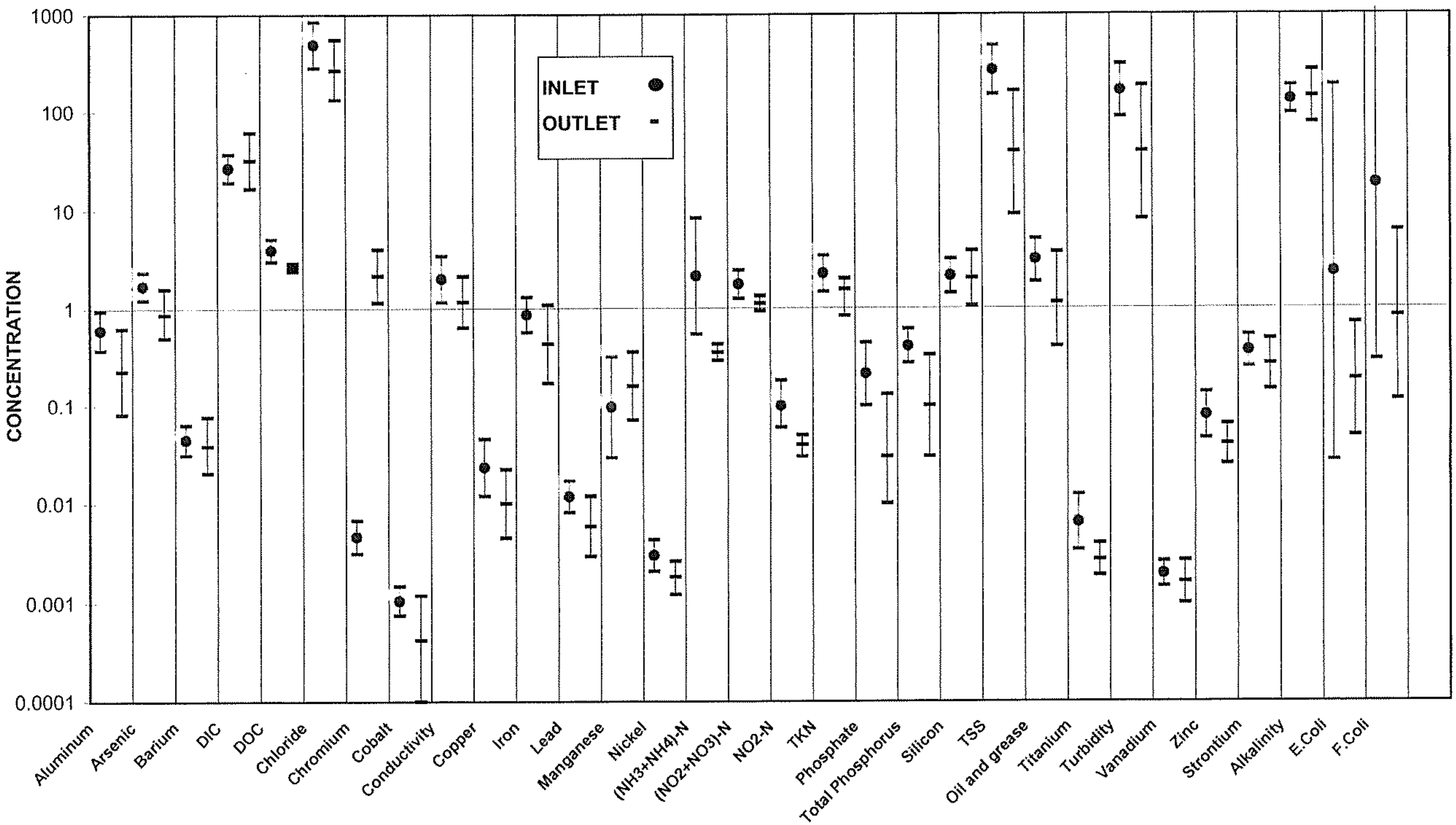


Figure 5.2 : Total load-based removal efficiencies for the summer/fall period from September to November, 1996 and May to November, 1997.



NOTE: All concentrations in mg/L, except conductivity (mS/cm), Turbidity (FTU) and bacteria parameters (colonies/100ml x 10^{-3}). This graph is intended to show relative statistical differences between the inlet and outlet means. For numerical values and additional statistical data, see Tables in Appendix E.

Figure 5.3**:** Average Event Mean Concentrations (AEMCs) and 95% confidence limits at the inlet and outlet for the winter/spring period (December to April, 1997).

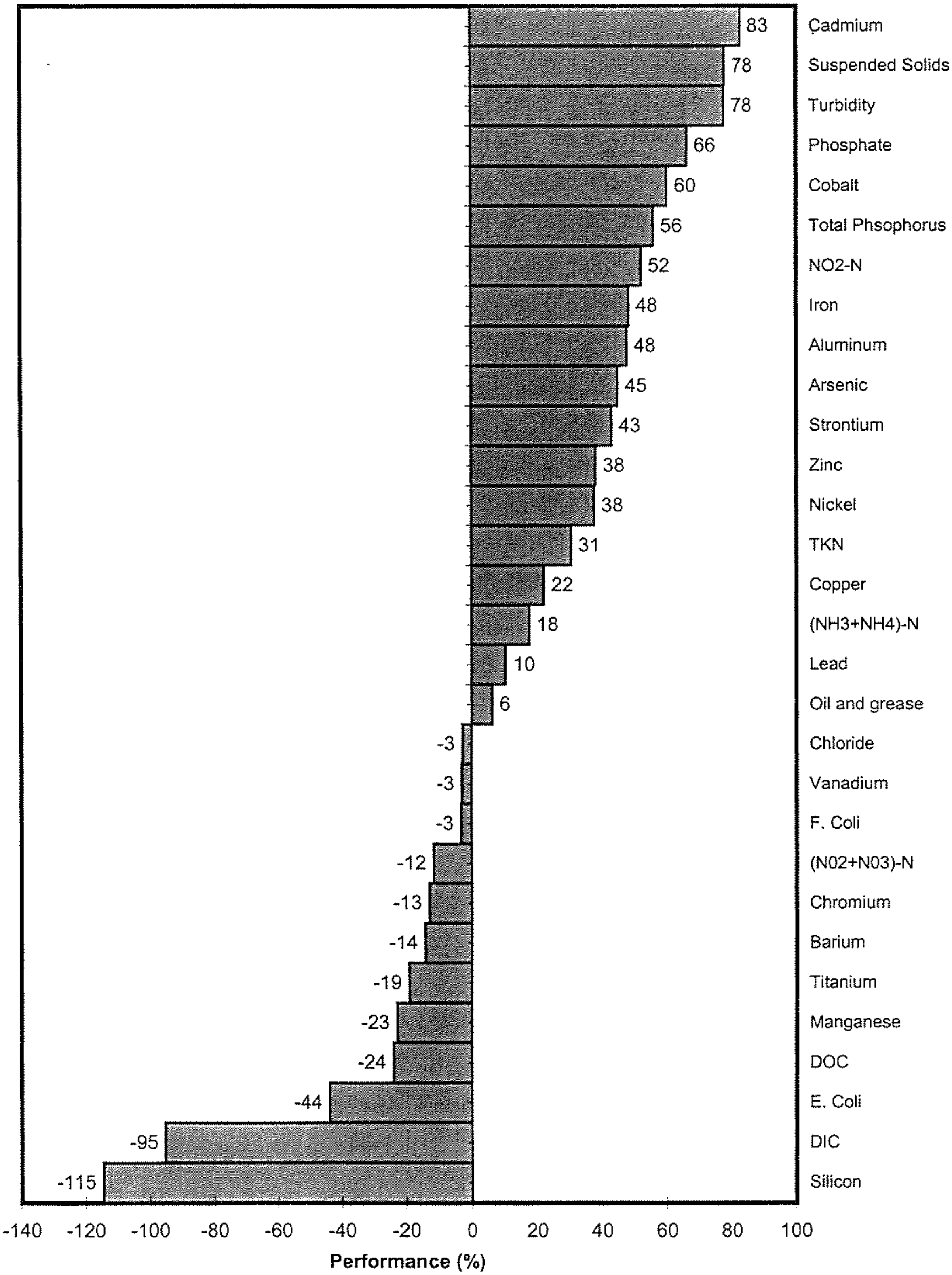


Figure 5.4: Average concentration-based removal efficiencies for the winter/spring period from December to April, 1997.

During the summer/fall period the mean influent and effluent TSS concentrations were 345 and 46 mg/L, respectively. The total load removal efficiency was 80%, ranging from 26 to 94% during the ten individual events for which reliable data were available. The average of the 10 individual event load-based TSS removal efficiencies was 74%. As observed in other studies (*e.g.* Oberts and Osgood, 1991), removal efficiencies were generally better during events with highly concentrated inflows than when the influent was relatively clean.

The removal efficiencies observed in this study are within the expected range for a wet pond facility with an estimated residence time of less than 18 hours (*e.g.* Mathews et al, 1997). In a series of settling column tests, Schueler (1992) found that a batch settling time of only 6 hours was required before removal efficiencies of suspended solids and other constituents ceased to display significant increases. The geotextile wrapped Hickenbottom risers at the forebay and wet pond, as well as the location of the discharge point at the surface of the permanent pool may have contributed to the reasonably good performance results.

TSS removal efficiencies at the Harding Park wet pond-wetland compare well with a TSS removal range of 10 to 85% reported by Scherger and Davis (1982) in a Michigan wet pond-wetland system, and 84% seasonal load-based removal reported by Liang and Thompson (1996) in another stormwater facility located in Richmond Hill, Ontario. The Harding Park facility TSS removal efficiency during both seasons exceeded the Ontario provincial government's (OMOEE, 1994a) target of 70% for level II fisheries protection in the downstream channel.

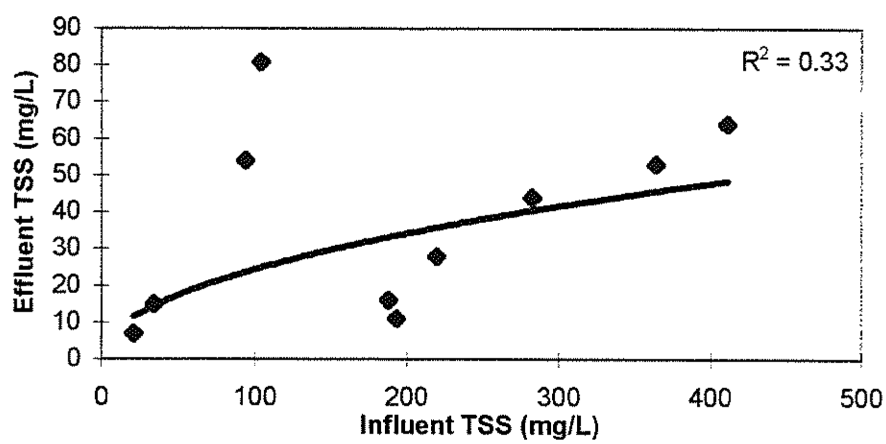


Figure 5.5: Inlet and outlet concentrations of total suspended solids (TSS).

The inlet and outlet concentrations for suspended solids are plotted in Figure 5.5. Although there is some scatter in the 20 to 120 mg/L concentration range, the general pattern is increasing monotonic. The graph also shows that inlet concentrations varied widely from 20 to 410 mg/L, whereas outlet concentrations only varied from 7 to 81 mg/L.

The removal efficiency of TSS followed a recognizable pattern based on influent EMC values. Generally, removal efficiency increased with influent EMC to about 200 mg/L, then levelled off at approximately 90% (Figure 5.6). This pattern suggests that higher TSS concentrations are associated with more settleable particles.

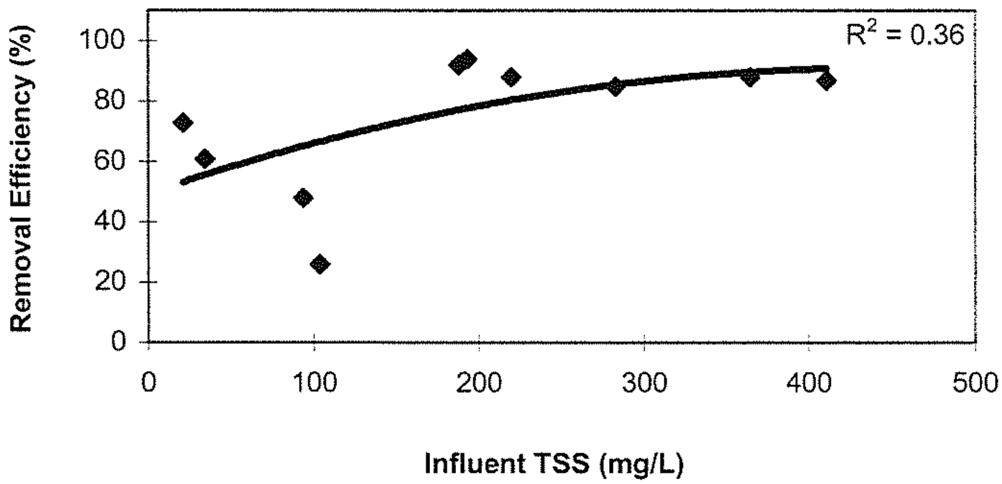


Figure 5.6: Relationship between inlet suspended solids concentrations and removal efficiencies

One of the earliest works on suspended sediment effects on fish was conducted by a group of European scientists who attempted to develop criteria for suspended sediment concentration in relation to fisheries quality (EIFAC, 1965). The criteria for continuous suspended sediment load were as follows: less than 25 mg/L: not harmful, 25 to 80 mg/L: fish yield somewhat reduced, 80 to 400 mg/L: good fisheries unlikely, and greater than 400 mg/L: poor fisheries. Thus the average effluent suspended sediment load at Harding Park would likely reduce fish yield somewhat, but represents a significant improvement over the average influent concentration, which would make improved fisheries unlikely.

5.1.1 Particle size distribution

The particle size distribution (PSD) of stormwater is important for several reasons. First, the relationship between suspended solids removal and the removal of other constituents is greatly influenced by suspended particle size. Clay particles, in particular, have a large capacity to carry nutrients and contaminants due to their high cation exchange capacity (CEC), and large surface area to weight ratio. Second, the change in size distribution observed between the inlet and outlet is an important indicator of size selective particle removal either by settling, flocculation or filtration. Third, the outlet particle size distribution has important implications on effluent impacts to receiving waters both in terms of aquatic habitat and erosion potential.

The generally accepted particle size division between bed load, the larger sized sediment load transported along the bottom of the flow channel, and suspended load, the smaller particles in hydraulic suspension, is about 62 μm. Clay particles are often classified as less than 4 μm in diameter (Waters, 1995).

The mean and individual event PSDs at the inlet and outlet are presented in Figure 5.7. The median particle size of the average PSD was approximately 4.5 and 2.3 μm at the inlet and outlet, respectively. Average inlet and outlet PSDs for particles ranging from 1.7 to 30 μm were not significantly different at the 95% confidence interval. Particles greater than 4 μm accounted for 55% of the inlet PSD compared to only 34% of the PSD at the outlet, indicating size selective removal of TSS. The proportion of particles less than 1 μm in size was only slightly greater at the outlet.

Table 5.1 summarizes the particle class masses and performances as regrouped into very fine sand, silt and clay groups. The analysis assumes negligible density differences among size categories and includes only 5 storms for which PSD data were available at both the inlet and outlet. Particle size masses could only be calculated for the 1.7 to 999 μm size fraction because the laboratory analysis of suspended solids, from which masses were derived, excludes particle sizes less than 1.5 to 2 μm (see summary of analytical procedures in Appendix C). By volume, this 'omitted' fraction accounts for 26 and 36% of all particles at the inlet and outlet, respectively. Among size classes greater than 1.7 μm, removal efficiencies for TSS ranged from 48 to 81, and as expected, higher performance was associated with larger particle size classes.

Table 5.1: Particle size class mass, mass proportion and performance (n=5).

Particle class	Size Range	Inlet mass (kg)	Inlet (%)	Outlet mass (kg)	Outlet (%)	Performance (%)
Fine to Coarse Sand	62 - 999 μm	54	3.8	10	1.1	81
Silt	3.7 – 62 μm	652	51.7	229	33.5	65
Clay	1.69 - 3.7 μm	245	18.8	127	28.9	48
Clay*	0.17 – 1.69 μm	-	25.7	-	36.5	-
Total		951	100	366	100.0	61

* Laboratory measurement of TSS concentrations only includes size fractions greater than 1.5 to 2 μm; therefore, masses could not be calculated for the 0.17 to 1.69 μm size fraction.

a)

b)

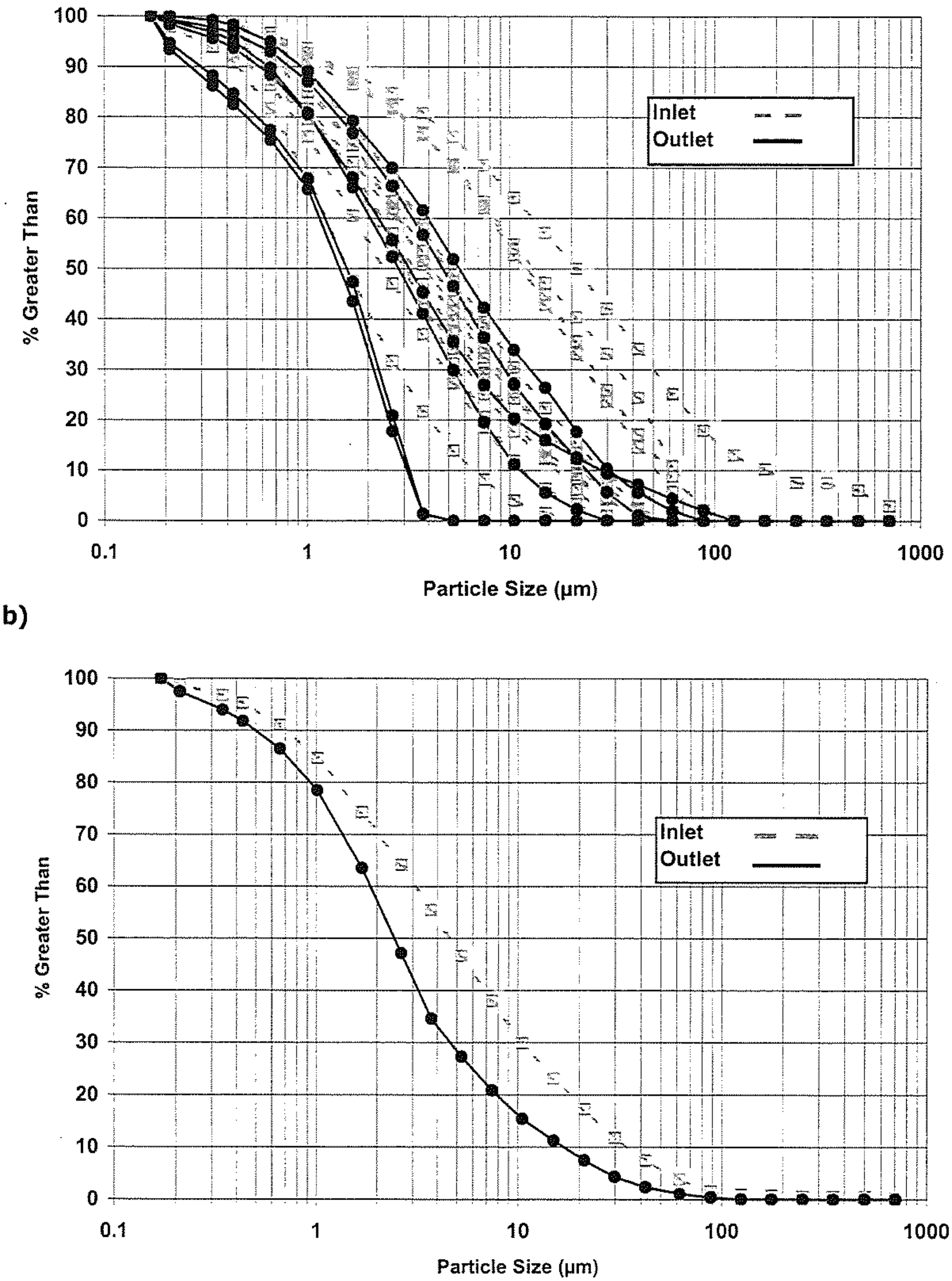


Figure 5.7: a) Individual event and b) average cumulative particle size distributions at the inlet and outlet for the post-berm repair summer/fall monitoring period.

5.2 General Water Chemistry

The average concentrations and removal efficiency results for general physical and chemical parameters analyzed in this study are presented in Table 5.2.

Table 5.2: Average Event Mean Concentrations (AEMCs), grab sample concentrations, removal efficiencies (R.E.) and Provincial Water Quality Objectives (PWQOs) for general chemistry parameters during the summer/fall and winter/spring seasons.

General Chemistry Parameters	Summer/fall			Winter/Spring			PWQO
	AEMC+			Concentration+			
	Inlet**	Outlet	R.E. (%)*	Inlet**	Outlet	R.E. (%)*	
Dissolved Organic Carbon (DOC)	3.4	4.6	-70	3.9	2.6	-24	-
Dissolved Inorganic Carbon (DIC)	14.7	30.7	-155	27.0	32.3	-95	-
pH	8.1	8.1	-	7.9	8.0	-	6.5 to 8.5
Alkalinity	77	133	-	132	143	-	-
Chloride	22	71	-548	497	274	-3	-
Turbidity	360	<u>32</u>	84	164	39	78	-
Conductivity	312	538	-	2003	1165	-	-
Silicon	1.2	2.0	-168	2.1	2.0	-115	-

* Removal efficiencies were calculated for the post berm repair period based on total loads during the summer/fall and based on grab sample concentrations during the winter/spring. Note that removal efficiencies are calculated from paired inlet/outlet samples and, therefore, may be higher or lower than removal efficiencies calculated from the larger concentration data set (*e.g.* DOC).
** Inlet concentrations were averaged over the period before and after berm repair, whereas mean outlet concentrations only include the period after berm repair.
\+ All units in mg/L except for pH (no units), Alkalinity (mg/L $CaCO_3$), Turbidity (FTU) and conductivity (μS/cm).
note: Underlining indicates that outlet concentrations were significantly lower than inlet concentrations at the 95% confidence level.

5.2.1 pH

The mean inlet and outlet pH for the summer/fall period was 8.1. There was little variability in pH levels among storm events. This pattern of slightly alkaline waters with little variability among events and between inlet and outlet sampling points was also observed during the winter/spring period. All pH values observed at the site lay within the 6.5 to 8.5 range recommended by the Ontario Ministry of Environment and Energy's Provincial Water Quality Objectives (OMOEE, 1994b).

5.2.2 Alkalinity

Alkalinity, reported as the concentration of calcium carbonate ($CaCO_3$), displayed considerable variability among events, ranging from 28 to 245 mg/l at the inlet during the summer/fall period. The Average Event Mean Concentration (AEMC) during this period increased from 77 mg/L at the inlet to 133 mg/L at the outlet.

The increase in concentration indicated that the facility was a net source of alkalinity. During the winter/spring, mean alkalinity concentration increased less dramatically from 132 mg/L at the inlet to 143 mg/L at the outlet. Alkalinity levels are an important consideration when discussing the concentrations of some metals since they can significantly influence their mobility and bioavailability (OMOEE, 1994b).

5.2.3 Dissolved organic and inorganic carbon

During the cold and warm seasons, removal efficiencies for DOC and DIC were negative, indicating that the facility is a source of these constituents. Bioactivity within the facility may have contributed both to DOC export in 1997 and the decrease in DOC effluent concentrations from the summer to winter. DIC may have originated from leaching of inorganic bed material in the facility.

Note that average DOC concentrations were greater at the inlet than the outlet, but the removal efficiency for this season was negative. This apparent contradiction stems from the period over which the two sets of statistics were calculated. As stated above, the inlet average encompasses the entire study period, including the period prior to berm repair, when DOC concentrations were relatively high. Removal efficiencies, on the other hand, were determined from paired inlet/outlet concentrations during the post-berm repair period only. Hence, for this and other parameters yet to be discussed, significant differences between concentrations before and after berm repair can result in apparent discrepancies between average concentration data and removal efficiencies.

5.2.4 Chloride and Conductivity

The primary source of chloride to the facility is from de-icing salts applied to sidewalks, driveways and roads during the winter. Studies have shown that chloride accumulates in the pond during the winter, eventually forming a dense chemostratified layer at the pond bottom, and is gradually flushed from the facility during storm events in the spring and summer (SWAMP, 1999). This pattern of winter chloride accumulation followed by gradual flushing during the warm season is shown in Figure 5.8. Depth profiles of chloride concentrations conducted in the forebay and wet pond on August 6, 1998 further indicated that, even after several large rain events, chloride stratification persists well into the summer.

During the winter/spring period, chloride concentrations were generally greater than during other seasons, averaging 497 and 274 mg/L at the inlet and outlet, respectively. Average removal for chloride in 1997 (post-berm repair) was -3% during the winter, with a range from -124% to 88%. During the summer/fall, mean influent and effluent chloride concentrations fell to 22 mg/L and 71 mg/L, respectively. Total load removal efficiency for the summer/fall season was -548%, as winter chloride inputs were flushed out of the facility. As expected, this negative removal rate is also reflected by conductivity concentrations, which increased from 312 μS/cm at the inlet to 538 μS/cm at the outlet.

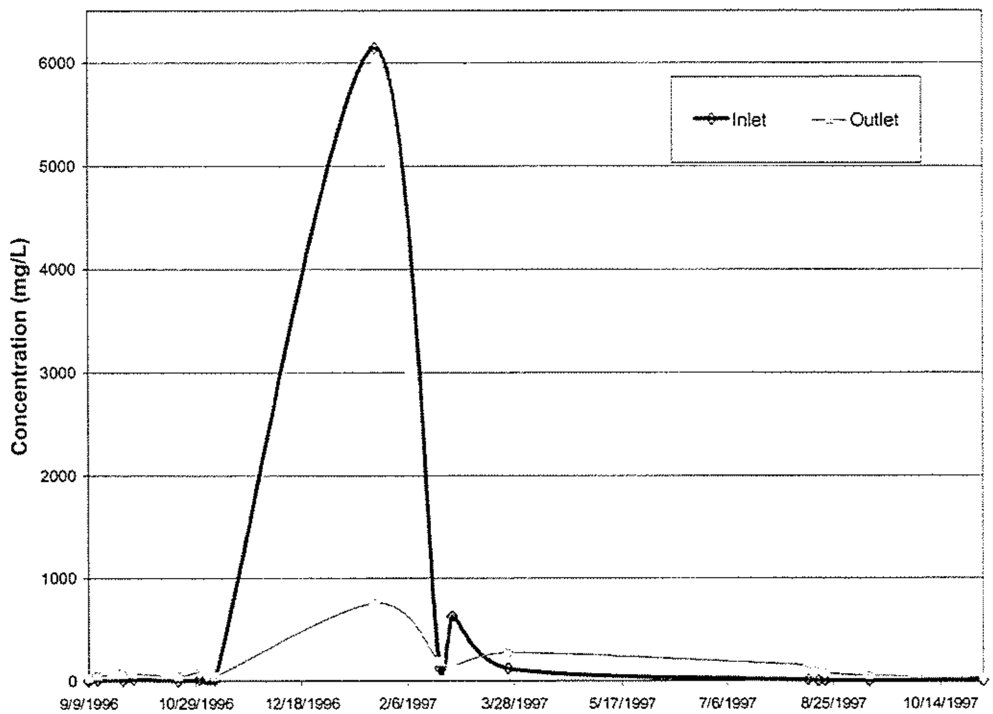


Figure 5.8: Influent and effluent chloride concentrations from September 1996 to November 1997.

5.3 Organics

In the early part of the study, water quality samples were analyzed for an extensive range of organic parameters. A complete list of these parameters, and their respective detection limits and PWQOs are presented in Appendix C. Among those listed, only Pentachlorophenol and 2,3,4,6 Tetrachlorophenol were found at concentrations consistently above the respective detection limits. Note also, however, that several organic parameters listed in Appendix C have PWQOs well below their analytical detection limits, thereby making it impossible to determine whether or not observed concentrations exceeded provincial threshold levels.

Mean pentachlorophenol concentrations at the inlet were 61 ηg/L during the warm season and 70 ηg/L during the cold season, both of which were considerably less than the Provincial Water Quality Objective (PWQO) of 500 ηg/L for this pollutant. All concentrations of pentachlorophenol at the outlet were less than the detection limit of 10 ηg/L. Mean seasonal concentrations of 2346Tetrachorophenol approached the detection limit of 20 ηg/L at the inlet during both the warmer and colder months. Again, no data were available at the outlet due to concentrations below the detection limit. Inlet concentrations were considerably less than the PWQO of 1000 ηg/L for this pollutant.

Solvent extractables (oil and grease) are organic carbon compounds that are less dense than water and therefore tend to float on the surface. The surface-drawoff outlet control structure was relatively ineffective in removing floating contaminants, which may explain the relatively low removal efficiencies of 48% and 6% observed during the warm and cold seasons, respectively. Although there is no PWQO associated with this parameter, average concentrations of 0.8 mg/L in the summer/fall (maximum 1 mg/L) and 1.1 mg/L in the winter/spring were not considered great enough to raise concerns.

5.4 Bacteria

Water quality samples were analyzed for two bacterial parameters: faecal coliforms and *Escherichia coli*. Coliform bacteria are frequently found within the intestinal tract of mammals. Both faecal coliforms and E. coli are used to indicate faecal contaminant levels, and hence, the possible presence of other harmful bacteria in receiving waters. Faecal coliforms often exceed established threshold levels (OMOEE, 1994b) for body-contact recreational activities at downstream beaches in the Toronto area. Die off of faecal coliforms in stormwater treatment facilities occurs naturally and has been shown to be dependent on water temperature and the residence time of stormwater runoff in the facility (Kadlec and Knight, 1996; Reed *et. al.*, 1995)

The mean influent concentration of faecal coliform during the summer/fall period was 16,149 coliforms/100 ml, ranging widely between 1,060 and 51,000 c./100 ml. Effluent concentrations were much lower, averaging only 2,858 c./100 ml, and ranging between 140 to 5,200 c./100 ml. Total load removal of faecal coliforms was 67% during the warm season. Winter/spring influent faecal coliform counts averaged 18,287 c./100 ml (pre and post berm repair period), compared to only 832 c./100 ml at the outlet (post-berm repair period).

During the summer/fall period *E.coli* counts accounted for about half the faecal coliforms counts at both the inlet and outlet with correspondingly similar ranges. The net seasonal *E. coli* removal performance during the summer/fall period was 53%, which compares to 79% *E.coli* removal reported at a nearby wet pond in the same municipality (Liang and Thompson, 1996). In the cold season, mean *E.coli* concentrations were 2,272 c./100 ml at the inlet (pre and post-berm repair period) and 185 c./100 ml at the outlet (post-berm repair period). However, removal was –44% based on paired inlet/outlet concentrations during the post-berm repair period (n=3).

The Provincial Water Quality Objectives (OMOEE, 1994b) for the protection of aquatic and recreational uses indicates that *E.coli* levels are not to exceed 100 c./100 ml. Only one effluent sample during the winter had *E. coli* concentrations less than the PWQO.

5.5 Nutrients

High nutrient (phosphorus and nitrogen) loading can lead to eutrophic conditions in receiving waters, characterized by excessive vegetation and algal production. Algal shading limits photosynthetic oxygen production beneath the water surface, resulting in adverse impacts to aquatic habitat. The nutrient mass ratio between nitrogen and phosphorus in healthy aquatic ecosystems has been estimated to be about 5:1 (Metcalf and Eddy, 1991). Phosphorus uptake is often the limiting factor in nutrient uptake in wastewaters.

Table 5.3 shows the mean seasonal concentrations and removal efficiencies for nutrient species analyzed in this study. Underlined values represent outlet mean concentrations that are lower than inlet mean concentrations at the 95% level of confidence. Values shaded in grey represent concentrations that exceed PWQO threshold values.

Table 5.3: Average Event Mean Concentrations (AEMCs), grab sample concentrations, removal efficiencies (R.E.) and Provincial Water Quality Objectives (PWQOs) for nutrient parameters during the summer/fall and winter/spring seasons.

Nutrient species	Summer/fall			Winter/Spring			PWQO (mg/l)
	AEMC (mg/L)			Concentration (mg/L)			
	Inlet+	Outlet	R.E. (%)*	Inlet+	Outlet	R.E. (%)*	
TKN	1.3	1.0	-24	2.2	1.2	31	-
$NH_3 + NH_4$	0.4	0.1	54	2.1	0.4	18	-
NH_3	0.014	0.003	-	0.014	0.003	-	0.02
$NO_2 + NO_3$	1.0	0.7	28	1.8	1.2	-12	-
NO_2	0.08	0.02	42	0.10	0.04	52	-
TP	0.39	0.11	42	0.40	0.21	56	0.03
PO_4	0.26	0.01	86	0.21	0.06	66	-

+ Inlet concentrations were averaged over the period before and after berm repair, whereas mean outlet concentrations only include the period after berm repair.

* Removal efficiencies were calculated based on loads during the summer/fall and on concentrations during the winter/spring. Mean inlet concentrations include the period before berm repair.

note: underlined values represent outlet concentrations significantly lower than inlet concentrations at the 95% confidence level. Shading indicates values above PWQOs (MOEE, 1994b)

5.5.1 Nitrogen

The nitrogen cycle describes the conversion of nitrogen in its original organic form to ammonia (NH_3) or its ionized form, ammonium (NH_4), then to nitrite (NO_2) and nitrate (NO_3), and finally to nitrogen gas (N_2), nitrous oxide (N_2O), or nitric oxide (NO) (Kadlec and Knight, 1996). Nitrification to nitrite and nitrate and denitrification to the gaseous phase are both biologically mediated processes that typically occur within aerobic and anaerobic environments, respectively.

During the summer/fall, all nitrogen species showed positive removal on a load basis except Total Kjeldahl Nitrogen (TKN) (-24%). Since total ammonia (NH_3 +NH_4) had a removal efficiency of 54% in the summer/fall, and TKN is the sum of organic nitrogen and total ammonia, the poor TKN performance may be attributed to an organic nitrogen source within the facility. Possible sources include waterfowl, mammals or vegetation in the facility. During the cold season, when the facility was covered in ice and biological activity was minimal, the TKN removal efficiency (31%) was greater than that of total ammonia (18%).

Among the nitrogen species analyzed, only nitrite and the ammonium ion had a summer/fall mean outlet concentration lower than its mean inlet concentration at the 95% confidence level. During the cold season, nitrite, total ammonia and ammonia mean outlet concentrations were lower at the 95% confidence level.

Based on a pH of 8.0 and an average observed temperature of 16.2 C in the summer/fall and –5 Cin the winter/spring, the average un-ionized inlet and outlet ammonia (NH_3) concentrations during the two seasons were both less than the 0.02 mg/L PWQO for this parameter (for conversion formula, see OMOEE, 1994b).

5.5.2 Phosphorus

The mean outlet concentrations of total phosphorus and phosphate during the summer/fall were lower than the means at the inlet at the 95% confidence interval. During the same period, both the mean influent (0.39 mg/L) and effluent (0.11 mg/L) total phosphorus (TP) concentrations exceeded the recommended maximum TP concentration of 0.03 mg/L (OMOEE, 1994b) intended to avoid excessive plant growth during the ice-free period. Total load phosphorus removal efficiency during this period was 42%. During the cold season, the mean TP concentration was 0.40 mg/L at the inlet and 0.21 mg/L at the outlet, and TP removal efficiency averaged 56%. A strong linear correlation between TP and TSS ($r = 0.88$) indicates that settling is the primary mechanism for TP removal.

Phosphate (PO_4) represents the dissolved fraction of total phosphorus and hence removal predominantly occurs through mechanisms other than settling, such as plant uptake or fixation by calcium, magnesium or aluminum. The total load performance for phosphate during the summer/fall was 86%. During the winter/spring the inlet and outlet means were not different at the 95% significance level and the removal efficiency was 66%. The better warmer season removal of ortho-phosphate may be attributed to enhanced biological activity in the wet pond and wetland cell.

As mentioned above, aquatic systems utilize TN and TP in approximately a 5:1 ratio, respectively, and in many cases phosphorus is the limiting constituent. The mean TN:TP ratios of the influent and effluent during the summer/fall were about 6:1 and 9:1, respectively, indicating that as expected phosphorus was the limiting factor in biological TN removal via uptake.

5.6 Metals

Several metals are toxic to fish and wildlife even at very low concentrations. The most common toxic metals in stormwater are zinc, lead and copper. Metals in ponds and wetlands can be taken up by plants and bacteria, precipitate as insoluble salts and bind to soluble organics, particulates and sediment (Kadlec and Knight, 1996).

Table 5.4: Average Event Mean Concentrations (AEMCs), grab sample concentrations, removal efficiencies (R.E.), reporting method detection frequencies (%>DL) and Provincial Water Quality Objectives (PWQOs) for metals during the summer/fall and winter/spring seasons.

Metal	RMDL	Summer / Fall					Winter / Spring					PWQO
		Inlet		Outlet			Inlet		Outlet			
		%> DL	AEMC (µg/L)+	%> DL	AEMC (µg/L)	R.E. (%)*	%> DL	Conc. (µg/L)+	%> DL	Conc. (µg/L)	R.E. (%)*	
Aluminum	10	100	1422	100	<u>290</u>	74	100	596	100	226	48	
Arsenic	1.0	23	1.0	8	0.8	17	60	1.7	50	0.9	45	100
Barium	1	100	45.8	100	36.5	-11	100	44.7	100	39	-14	
Beryllium	0.1	45	0.14	8	0.05	-	8	0.05	0	0.05	-	1100
Cadmium	0.1	95	0.64	100	0.45	11	86	1.48	67	0.07	83	0.5
Chromium	0.2	100	3.33	100	2.4	53	100	4.7	100	2.2	-13	100**
Cobalt	0.2	90	2.15	45	0.84	82	79	1.05	75	0.4	60	0.9
Copper	0.2	100	22.2	100	<u>4.5</u>	48	100	23.22	100	10.1	22	5
Iron	20	100	1069	100	<u>386</u>	66	100	861	100	856	48	300
Lead	5	60	10.5	0	2.5	83	79	11.8	100	6.0	10	5
Manganese	0.5	100	162.4	100	115.8	9	100	96.7	100	159.4	-23	
Mercury	0.02	14	0.02	0	0.01	-	7	0.02	0	0.01	-	0.2
Nickel	0.5	95	3.42	92	3.2	62	93	3.04	100	1.8	38	25
Strontium	2	100	195	100	270.5	-63	100	361	100	237	43	
Titanium	1	73	5.5	100	3.1	-18	71	6.5	100	2.7	-19	
Vanadium	0.2	100	3.4	100	<u>1.1</u>	66	100	2.0	100	1.6	-3	6.0
Zinc	0.5	100	66.7	100	<u>16.4</u>	70	100	79	100	40.0	38	20

Note: Underlined values indicate outlet concentrations that are significantly lower than inlet concentrations at the 95% confidence level. Shading indicates values above PWQO.
RMDL stands for Reporting Method Detection Limit for OMOEE laboratories.
* Removal efficiencies were calculated based on loads during the summer/fall and on grab sample concentrations during the winter/spring.
+ Inlet concentrations were averaged over the period before and after berm repair, whereas mean outlet concentrations only include the period after berm repair.
** 1.0 µg/L for Chromium VI; 100 µg/L for Chromium III

Table 5.4 presents detection frequencies, mean inlet and outlet concentrations, seasonal performance results and PWQOs for metals analyzed in this study. As in previous tables, underlined values represent outlet mean concentrations that are lower than inlet mean concentrations at the 95% level of confidence. Shaded values represent concentrations that exceed PWQO threshold values.

The results indicate summer/fall removal efficiencies for metals ranging from –63% for strontium to 83% for lead, with an overall mean of 37%. Stormwater contaminants with summer/fall removal efficiencies above 50% include aluminum, chromium, cobalt, copper, nickel, vanadium, lead and zinc. Removal was generally lower in the winter/spring, averaging only 24% and ranging from –23 to 83%.

Among the metals analyzed, mean outlet concentrations of aluminum, copper, iron, vanadium and zinc were significantly lower than inlet mean concentrations at the 95% level of confidence level (underlined values in Table 5.4). During the winter/spring, inlet and outlet concentrations were not significantly different for any of the parameters, in part because of the very small sample size (n=4) and large variation in pollutant concentrations among events.

During both monitoring seasons, mean effluent concentrations of iron exceeded Ontario's PWQOs. Copper, zinc and lead effluent concentrations exceeded provincial objectives only during the winter/spring period. Trace level amounts of nickel, beryllium, chromium, arsenic and mercury were found in both inlet and outlet samples, all at concentrations well below their respective PWQOs. Mean summer/fall inlet concentrations of mercury were 0.021 µg/L, but were consistently below the Reporting Method Detection Limit (RMDL) of 0.02 µg/L at the outlet. Winter/spring average inlet and outlet mercury concentrations were also below the RMDL.

5.7 Temperature

Temperature data were collected continuously at 15-minute intervals from June 1 to 13 and July 23 to September 30, 1996 at the inlet and outlet to the stormwater facility, and in German Mills Creek immediately upstream of the pond outlet from August 18 to September 30, 1996.

Monthly temperature monitoring results are presented in Figure 5.9. Effluent temperatures were approximately 6 to 9°C higher than inlet and creek water temperatures during July and August. This increase in water temperature as water passes through the facility is greater than a warm season increase of 5.1°C cited in the SWMP manual (OMOEE, 1994a) for extended detention wet ponds.

Influent temperatures rose sharply during storm events, but exhibited relatively constant baseflow temperatures, ranging from 9°C in June to just above 14°C in September. The average daily temperatures of the effluent and creek were 23 and 15°C, respectively, with fluctuations occurring primarily in response to diurnal variations in air temperature. The maximum recorded outlet temperature was 31°C on June 1, 1996.

Outlet temperatures were frequently above the 21°C limit generally accepted as the threshold for cold water fisheries habitat. However, dilution of facility effluent by the much larger discharge volumes from German Mills Creek would likely result in relatively minor impacts on downstream creek temperatures.

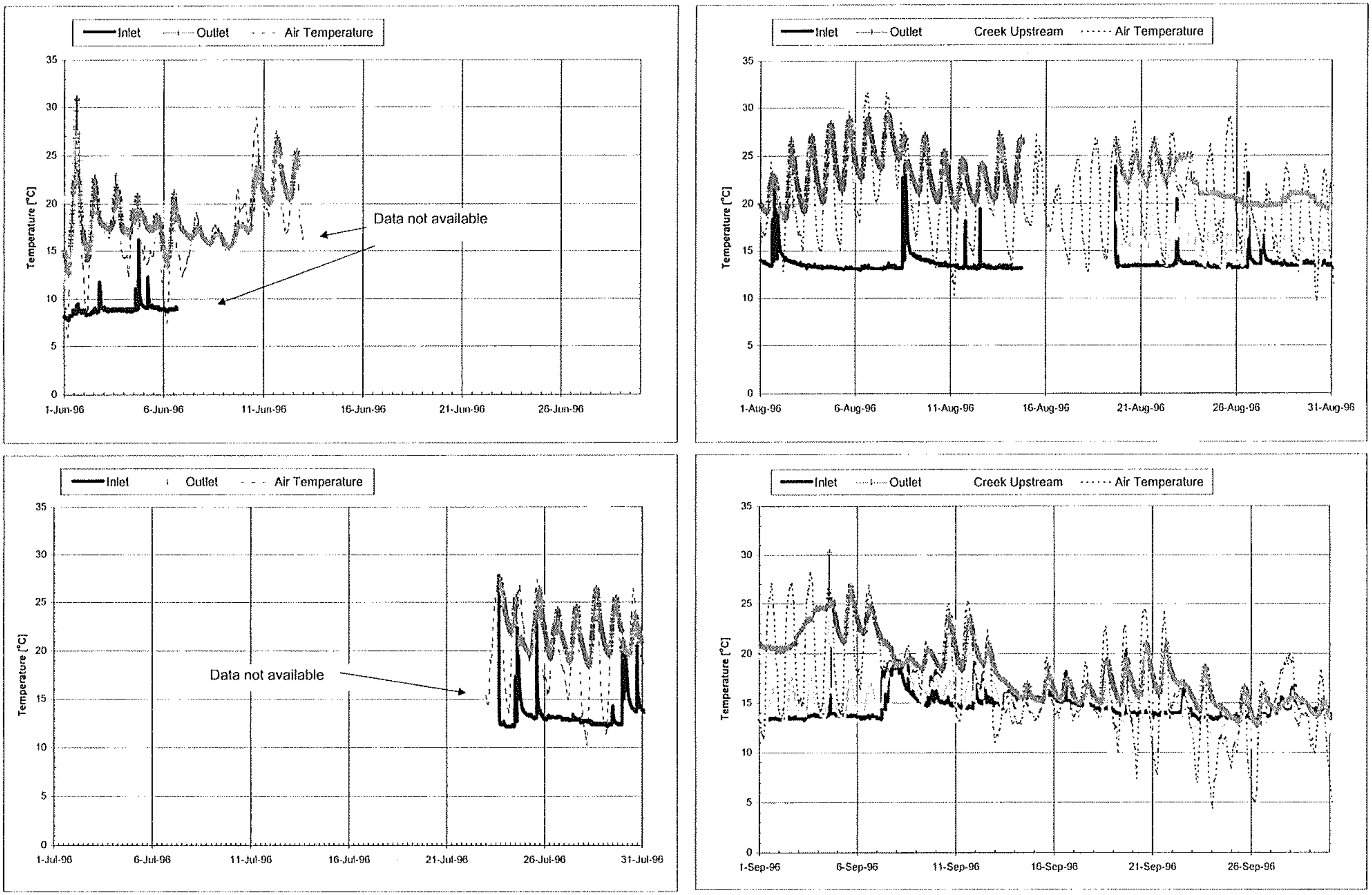


Figure 5.9: Air and water temperature at the inlet, outlet and German Mills Creek from June 1 to September 30, 1996.

6.0 VEGETATION AND AQUATIC COMMUNITY ASSESSMENT

Plants in wet pond treatment systems perform several functions, including bank stabilization, chemical uptake, root zone aeration, surface area attachment for bacteria and aesthetic appeal. Therefore, the type of plants established within the facility, and the success of planting programs was considered to be an important component of the overall performance assessment. Studies of the vegetation and algae communities at the Harding Park stormwater facility are included in Appendices C and D. These studies are briefly summarized in this section.

6.1 Summary of Vegetation Study Results

The vegetation study was conducted on two stormwater management facilities in the Greater Toronto Area: one at the Harding Park Stormwater retrofit pond and the other at the newly constructed Ministry of Transportation (MTO) stormwater wet pond in Scarborough (now the City of Toronto) near a major transportation corridor and adjacent to the Rouge River. The overall goal of the two studies was to develop a list of recommended vascular plant species and corresponding planting strategies for use in facilities treating stormwater. This goal was to be achieved by (i) monitoring the effectiveness of planting plans in developing a balanced vegetation community and, (ii) inventorying plant species in the various cells of the stormwater management facilities.

At Harding Park, vegetation inventories were conducted in June and September 1996 and in June, August and September 1997. All plants below the high water or active storage elevation were identified and classified as being native or non-native to the region. Habitats were identified as disturbed, meadow, meadow marsh, upland, shallow marsh, emergent, aquatic, riparian or mesic (see Appendix D for definitions).

Planting was carried out after the Harding Park facility was retrofitted. Four native and two non-native species were planted in the sediment forebay and wet meadow, and four native species were planted in the wet pond. By the end of 1996, the total number of species in the sediment forebay increased from four native and two non-native original plantings to 14 native and 5 non-native species. By the end of 1997, native species in the sediment forebay increased to 30 and non-native increased to 12. Similar increases in species diversity were also observed in the wet pond and wet meadow cells.

Results from the two years of vegetation monitoring indicated several trends. At the end of 1996, vegetation coverage was sparse and much of the shoreline was bare. By the end of 1997, these bare patches were well vegetated and plant diversity had increased significantly. Although species dominance varied among cells, there did appear to be a trend toward aquatic to meadow marsh species dominance in all three cells of the facility. Diversity was greatest during the month of August. Although, the total numbers of native and non-native species naturally colonizing the facility increased significantly over the two years of study, the proportion of native to non-native species remained generally the same.

Based on monitoring results, the following recommendations were made regarding vegetation establishment in stormwater facilities.

1. If substantial natural colonization is observed at other sites, it may justify a reduction in the original planting plan.
2. The vegetation community tended to evolve toward a common group of dominant species, which may warrant a reduction in the diversity of the original planting plan.
3. Original planting species were often substituted by plant suppliers with alternative, non-native species, which highlights the need for increased quality control at the planting site.
4. Further study is needed to:
   - evaluate the effect of alien invasive species such as purple loosestrife,
   - determine the seasonality of species dominance,
   - assess the success of natural seed establishment, and
   - compare the results of this study with the findings of other comparable studies.

6.2 Summary of the Phytoplankton and Periphyton Assessment

Algae are a broad group of unicellular to multicellular photosynthetic plants and bacteria that lack the tissue specialization of higher plants and organisms. Algae may be grouped as filamentous algae growing at or near the water surface, free-floating plankton within the water column, periphyton attached to plant or other surfaces or benthic algae growing on the bottom substrate. Algae require relatively high sunlight levels to thrive and, if provided in a nutrient enriched environment, can grow exponentially forming extensive algal blooms. These algal blooms inhibit the transmission of light to aquatic plants and may lead to anoxic conditions in the lower water column. In an open aquatic or wetland system, they provide the food source for a higher trophic level of heterotrophs and microbia. As the basis of an autochthonous food chain of organisms both intrinsic and essential to wetland treatment, algae are an important component of biological treatment.

The periphyton and phytoplankton communities of the sediment forebay and wet pond of Harding Park were assessed during the summer of 1997. The study examined phytoplankton and periphyton species abundance as well as bio-volume. It also examined several chemical and physical characteristics of these different areas within the stormwater facility.

The sediment forebay of Harding Park was colonized by few species and dominated by *Euglena sp.* Phytoplankton. Periphyton within the forebay showed a distinct floristic assemblage, which were nearly two magnitudes higher biovolume than the areal periphyton of the wet pond. The periphyton community in the forebay differed from the assemblage in the wet pond. The species poorness and dominance by one genus in the forebay was considered to be indicative of considerable environmental stress at this location. Relative to the wet pond, the environmental stressors were poor water quality, high and turbulent flow, and cool water temperatures. Samples collected from the forebay had higher concentrations of total phosphorus (TP), total nitrogen (TN), and conductivity and lower pH and surface water temperatures compared to samples collected in the wet pond. Based on chemical and physical characteristics of the algae in the forebay, conditions in the forebay were classified as hypereutrophic.

The wet pond area was not dominated by a single genus, but showed a more balanced and species rich algal community, probably due to improved water quality, less flow turbulence and higher surface water temperatures. In the densely vegetated periphery of the wet pond, periphyton volumes were less than expected, as the dense macrophytic plants appeared to outcompete and shade out periphyton. The algal community in the wet pond cell was classified as eutrophic to hypereutrophic.

Although, surface water quality improved from the forebay to the wet pond, a dense, chemostratified layer of cooler saline water was detected at the bottom of the wet pond. Although this layer weakened somewhat over the summer, it did persist into September. At mid-summer, water below 2 m depth in the facility was anoxic.

There were no blue-green algae in the facility. This finding was unexpected as blue-green algae are common in nutrient-rich aquatic environments, such as the Harding Park wet pond. While an explanation for this result can not be given, other factors such as low N:P ratios, turbulence, temperature, light, pH, carbon dioxide and zooplankton populations have been known to influence blue-green algae establishment.

The observations of improved water quality from the forebay to the wet pond and chemostratified layer support the findings of the main body of the study. As indicated in Chapter 5, water quality generally improves from the inlet to outlet, in the case of some constituents quite markedly, so it would be expected that an improvement in water quality would be observed between the forebay and wet pond. The study results provide evidence that the sediment forebay acts as an initial pollutant containment zone and buffer to downstream treatment cells. The latter observation supports the recommendation that maintenance and dredging of the forebay is required more frequently than downstream cells. Finally, the results of the algal study demonstrate the validity and effectiveness of algae as a surrogate measure of water quality and treatment performance in stormwater management systems.

7.0 STORMWATER FACILITY MODELLING

The purpose of the modelling exercise was to assess the long–term performance of the Harding Park stormwater facility and estimate suspended solids accumulation rates. The model used in this study (SWMM 4.3) is a parametric, spatially distributed model that utilizes rainfall data to simulate runoff from a user–defined catchment area. To accurately represent the observed runoff quantity and quality to and from the site, several runoff events were used to conduct comparisons, both for model calibration and verification. While it is impossible to develop a stormwater facility model that will simulate observed conditions with 100% accuracy, the final model does so with reasonable accuracy and its degree of error is known. As a result, the model output is viewed as a good estimate of real conditions within a pre-defined range of possible conditions.

7.1 Model Set-up and Calibration

The structure and calibration of the model is described in detail in Appendix H. In general, observed inlet flow volumes and peak flows were matched within 25% by the simulated results. Observed and simulated outlet flow volumes were also strongly correlated. Observed and simulated effluent peak flows were less consistent, but when averaged over the season, were matched within 10%.

Total suspended solids loading estimates were based on the assumption that solids build-up during interevent periods, and are subsequently washed off during runoff events. The mean observed and simulated loads during the summer/fall period were within 16% of the observed values, although there was significant variation among individual events. For the seven storms used in the calibration, the average load-based removal efficiency of 72% was similar to the modelled removal efficiency of 75%.

7.2 Continuous Simulation

Long–term continuous simulation of the calibrated model was used to estimate the cumulative loads. The continuous model was based on 12 years of rainfall data obtained from Toronto Buttonville Airport, 8-km southeast of the facility. The rainfall data set started on May 23, 1986 and ended on November 1, 1997 and covered the period from May 1 to November 1 of each year. The data set did not include the period from November to April. Therefore, simulation results for the period from May 1 to November 1 were extrapolated to include the November to April period by using a multiplier factor of 1.7, representing the 12 year rainfall average from May to November (499 mm) divided by the annual average precipitation (850 mm) for the area (HAC, 1978).

Table 7.1 shows precipitation, infiltration, evaporation and runoff amounts for the May 1 to November 1 simulation and annual extrapolation. Over the 12 year simulation period, 5993 mm of precipitation fell on the catchment, of which 1995 mm ran off as event flow. This yielded a continuous, long-term runoff coefficient of 33%, which was considered to be reasonably accurate when all precipitation events were considered. Over the entire catchment surface, infiltration accounted for 52% of all precipitation. Surface evaporation accounted for 15% of all precipitation.

Table 7.1: Simulated and extrapolated totals for model results over the entire catchment area.

Parameter	**Simulated Periods (May – Nov.1)**		**Extrapolation (Jan/86-Dec/97)**	
	mm	m^3	*mm*	m^3
Total Precipitation	5,993	1,006,785	10,200	1,713,548
Total infiltration	3,121	524,262	5,311	892,294
Total evaporation	890	149,427	1,514	254,325
Surface runoff	1,890	317,421	3,216	540,250
Channel/Pipe/Inlet flow	1,995	335,080	3,395	570,306
Error (%)	5.3			

Baseflow as a percent of the total simulated flow during event mode was small compared to continuous mode. When total baseflows during both dry and wet weather periods were accounted for in conjunction with surface event runoff, the total amount of runoff entering the facility increased to 878,240 m^3 during the simulation periods. Total estimated baseflows account for 49% of all facility inflow. Evaporation from the facility accounted for 1% of inlet flow and basin infiltration accounted for 10% of total inflow.

Table 7.2 summarizes the water quality results for the continuous simulation and extrapolation time periods. The long-term TSS removal efficiency of the facility was estimated at 75%, which compares to 80% net removal efficiency estimated from the summer/fall season.

The SWMP Manual (OMOEE, 1994a) recommends that wet ponds be dredged after the annual average TSS removal efficiency declines by 5%. Therefore, based on continuous simulation results, maintenance of the Harding Park facility would be required when removal efficiency falls to approximately 70%. A reduction in removal efficiency is a function of a reduction in total storage capacity from sediment accumulation, which is in turn influenced by the level of imperviousness in the pond catchment. Estimates from the SWMP Manual (OMOEE, 1994a) indicate that for the Harding Park pond storage capacity (62 m^3/ha) and catchment impervious level (approx. 45%), a decrease in removal efficiency from 75 to 70% would equate to a storage reduction of about 10 m^3/ha. The loss of 10 m^3/ha in permanent pool capacity indicates that the pond will require dredging when the original storage capacity of 1040 m^3 has been reduced by 168 m^3 to 872 m^3.

Table 7.2: Simulated and extrapolated results for water quality parameters over a 12-year period.

Parameter	Simulated Periods (May – Nov.1)		Extrapolation (Jan/86-Dec/97)	
	SS conc. (mg/L)	SS load (Tonnes)	Total Annual SS load (Tonnes)	Annual SS load (Tonnes)
Inlet mean / total	135	118	201	17
Loss by infiltration	8	0.3	0.5	0.04
Loss by decay	106	88	151	12
Outlet mean / total	36	30	50	4
Removal Efficiency	74	75		
Error (%)		0.4		

Assuming a bulk density for TSS of 1.23 tonnes/m^3 (OMOEE, 1994a), the Harding Park facility accumulates 10.2 m^3/yr of sediment. This suspended solids accretion rate will exhaust the storage buffer of 168 m^3 in 16.5 years. Recalling that the TSS removal efficiency (or the concomitant accumulation rate) had an error range of ±25%, the accumulation rate may range from 7.7 to 12.8 m^3/yr. This accretion range converts to a storage buffer depletion range between 13 and 22 years, respectively.

Since the estimates provided here are subject to considerable uncertainty, direct field assessments of TSS accumulation are recommended every 5 years.

8.0 CONCLUSIONS AND RECOMMENDATIONS

8.1 Conclusions

The primary objectives of this study were to assess the performance of the retrofitted facility in terms of water quantity and quality, evaluate the status of wetland vegetation and algal communities, and investigate long term maintenance requirements of the facility. The major study findings relating to each of these objectives are summarized below.

8.1.1 Water Quantity

From January to August 1996, the berm between the wet pond and wetland was damaged by erosion. When the berm was repaired in late August, the design outlet peak flow rate was revised upwards from 40 to 52 L/s. Among the 14 storms monitored after berm repair, only two had peak effluent flows greater than 52 L/s.

Event monitoring indicated that the runoff coefficient for the Harding Park catchment averaged 0.34, and ranged between 0.20 and 0.57. Storms with precipitation depths less than 4.0 mm produced negligible runoff. Mean inlet storm flow over the study period was 1427 m^3, which compares to a permanent pool volume of 1015 m^3. On average, peak flows were reduced by 90%. Dry weather baseflow was estimated at 1.5 and 1.3 L/s at the inlet and outlet, respectively, suggesting that, on average, approximately 17 m^3 of water per day is lost to evaporation and exfiltration of pond water.

The hydraulic detention time, as calculated from the time delay between inlet and outlet volumetric centroids, averaged 5.3 hours, and ranged from a low of 3.5 to a high of 11 hours. The residence time could not be directly determined from the available data, but a crude estimate assuming plug flow displacement conditions suggested a conservative value of 18 hours. Since there is probably some short circuiting and mixing of influent, the actual residence time for an element of fluid passing through the facility would likely be less than 18 hours. A falling head drawdown equation was used by the designer to meet the OMOEE 24 hour 'detention time' guideline (for 25 mm storm) suggested in the SWMP manual (OMOEE, 1994a). The drawdown time observed during large storms (greater than 20 mm) generally exceeded this 24 hour guideline.

8.1.2 Water Quality

Total load TSS removal was 80% during the summer/fall season, and ranged between 26 and 92% during individual events. The mean outlet TSS concentration was 46 mg/L. By contrast, during the period before berm repair, when the outlet control structure was not functional, the mean outlet concentration was 308 mg/L. Removal efficiencies during the winter/spring period after berm repair averaged 78%; this estimate was based on only four outlet grab samples and therefore should be interpreted with caution. All mean TSS removal efficiencies exceeded the 70% target recommended by the Ontario Ministry of Environment and

Energy for level 2 fisheries protection (OMOEE, 1994a). Possible explanations for the good TSS removal efficiencies may include the geotextile wrapped Hickenbottom risers at the forebay and wet pond, the location of the discharge point at the surface of the permanent pool, and extended detention volume exceeding OMOEE guidelines for water quality control.

Removal efficiencies for other constituents were generally less than observed for TSS. During the summer/fall season, total load-based removal was 42% for total phosphorus, 54% for total ammonia, 48% for oil and grease and an average of 37% for metals. In contrast, winter removal efficiencies were 56% for total phosphorus, 18% for total ammonia, 6% for oil and grease and an average of 30% for metals.

During both monitoring seasons, average outlet concentrations of iron, lead, *E. Coli* and phosphorus exceeded their respective PWQOs. In the winter/spring, mean concentrations of copper and zinc were also greater than PWQO threshold levels.

Among the 41 organic parameters (herbicides, pesticides and PAHs) analyzed in this study, only pentachlorophenol and 2,3,4,6 tetrachlorophenol were found at concentrations consistently above laboratory detection limits, and these were well below their respective PWQOs.

On average, particle sizes in the sand, silt and clay size classes had removal efficiencies of 81, 65 and 48%, respectively. Removal efficiencies could not be determined for particles within the 0.17 to 1.7 µm size range. This size range represented 26% of the influent particles and 36% of the effluent particles (by volume). The median particle size of the average distribution was 4.5 µm at the inlet, compared to 2.3 µm at the outlet.

As noted in other studies, the pond had a significant warming effect on the stormwater entering the facility. During July and August, the two warmest months, average influent and effluent temperatures were 13 and 23°C, respectively. Upstream creek temperatures were similar to those of the influent.

8.1.3 Hydrologic Modelling

The stormwater catchment and facility was modelled using SWMM 4.3 to predict, as accurately as possible, peak and total flows, TSS concentrations and loadings as well as the long-term, major maintenance frequencies for the facility. Twelve years (1986 to 97) of May-November precipitation data from Buttonville Airport, seven km away, were used to run a long-term simulation of facility operation. In general, surface infiltration, evaporation and surface runoff account for 52, 15 and 33%, of all the precipitation that falls on the catchment, respectively. Baseflow accounted for 49% of all facility inflow. Using a multiplier to calculate the annual rates of various model parameters, the annual average TSS input to and output from the facility are 16.8 and 4.2 tonnes, respectively, leaving a net retained mass of 12.5 tonnes. The predicted long term TSS removal efficiency was 75%, which is lower than the short-term removal efficiency rate of 80% observed during the study period.

Based on guidelines provided in the SWMP Manual (OMOEE, 1994a), the average dredging interval is estimated at 16 years, with an error range between 13 and 22 years. In order to provide a more precise estimate of the interval required for major maintenance activities, the depth of sediment accumulation should be assessed every five years.

8.1.4 Vegetation and Algae Monitoring

Vegetation surveys of the wetland indicated that the diversity of plant species increased from six initially planted at the site to 25 and 52 by the end of the first and second growing seasons respectively. This rapid establishment suggests that natural colonization may be an effective planting strategy for constructed stormwater wetlands.

The algae community study showed dominance by a single species in the forebay with significantly greater diversity in the pond. This difference reflects more turbulent flow, higher nutrient loading and cooler temperatures in the forebay. Based on the algal community assessment, the forebay and wet pond were identified as hypereutrophic and eutrophic-to-hypereutrophic, respectively.

8.2 Recommendations

The following recommendations are provided based on study results and site observations:

(i) Wetland performance could be improved if channelized flow through the wetland were distributed over a larger portion of the wetland via a perforated pipe or similar distribution system installed at the upstream end of the wetland.

(ii) A mid to low level drawoff configuration for the outflow structures would help to improve removal of floating contaminants (e.g. oil and grease, some organics), reduce effluent temperature and minimize adverse effects related to short circuiting across the surface of the pond. Such a structure may, however, result in decreased effluent quality because of reduced sedimentation efficiency over the mean flow path. Data from facilities with different outlet structures should be compared to assess the benefits and weaknesses associated with each design.

(iii) The feasibility of increasing the time period over which stormwater is detained within the facility should be investigated. This objective could be achieved by modifying the outlet structure such that drawdown times more closely match the average interevent period. Before implementing this measure, however, the impact on pond levels and the frequency of overflow should be carefully assessed.

(iv) Further monitoring of vegetation at the site is recommended in order to better characterize the climax community and verify tentative conclusions provided in this study.

(v) Sediment accumulation depths in the forebay and pond should be monitored regularly to determine maintenance requirements.

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APPENDIX A

Historical Context of the SWAMP Program

HISTORICAL CONTEXT OF THE SWAMP PROGRAM

Over the past 15 years, the Great Lakes Basin experienced rapid urban growth. Stormwater runoff associated with this growth has been identified as a major contributor to the degradation of water quality and the destruction of fish habitats. In response to these concerns, a variety of stormwater management programs have been developed in the Great Lakes basin.

A number of complementary programs have been established at the international, national, provincial and municipal levels to protect the Great Lakes ecosystem. The SWAMP program and the study that is the subject of this report are parts of the overall effort.

International Joint Commission

The International Joint Commission (IJC) prevents and resolves disputes between the United States of America and Canada under the Boundary Waters Treaty of 1909. The IJC pursues the common good of both countries as an independent and objective advisor of the two governments.

In particular, the IJC rules upon applications for approval of projects affecting boundary or transboundary waters and may regulate the operation of these projects; it assists the two countries in the protection of the transboundary environment. Among the responsibilities of the IJC is the implementation of the Great Lakes Water Quality Agreement.

Great Lakes Water Quality Agreement

The first Great Lakes Water Quality Agreement (GLWQA) between Canada and the United States was signed in 1972 in recognition of the urgent need to improve environmental conditions in the Great Lakes. The focus of the agreement was to improve water quality through pollution control programs. Objectives included the reduction of nuisance conditions and control of toxic substances. Specific numerical targets were included for the reduction of phosphorus loadings.

The Great Lakes Water Quality Agreement was amended in 1978 to include the objective of controlling persistent toxic substances. The new agreement also incorporated the ecosystem approach to environmental management.

In 1987, the Canadian and U.S. governments signed a protocol that identified local Areas of Concern (AOC's) where beneficial uses of the ecosystem had been significantly degraded. Remedial Action Plans (RAP's) were to be prepared by various levels of government for the AOC's. The plans would contain strategies to clean up problem areas in the Great Lakes region. In addition, the 1987 protocol included annexes addressing specific subjects such as non-point contaminant sources and contaminated sediments.

In total, 43 Areas of Concern were identified throughout the Great Lakes basin. Of the total, 17 AOC's were in Canada.

Great Lakes Sustainability Fund

The Canadian federal government's commitment to the Great Lakes ecosystem was initially managed through the Great Lakes Action Plan (GLAP). In 1990, the Great Lakes Cleanup Fund (GLCuF) was created to provide support for environmental projects designed to benefit the Great Lakes basin ecosystem.

In 1994, GLAP was replaced by the Great Lakes 2000 Program. GLCuF was extended and renamed the Great Lakes 2000 Cleanup Fund. In 2000, the Great Lakes Basin 2020 Action Plan was introduced in addition to the successor to the GLCuF, the Great Lakes Sustainability Fund (GLSF). The new plan and fund place priority on the restoration of environmental quality in Canada's remaining 16 Areas of Concern.

The GLSF supports the implementation of remedial actions falling within federal responsibilities that will lead to the restoration of beneficial uses in the Canadian Great Lakes Areas of Concern. The five-year, $30 million GLSF builds on past successes and is administered by Environment Canada on behalf of eight Government of Canada departments.

To restore these beneficial uses in the Great Lakes Areas of Concern, joint Canada-Ontario teams work in consultation with local Public Advisory Committees to develop Remedial Action Plans (RAPs) aimed at eliminating or reducing the major sources of contamination in these areas. When all beneficial uses in an AOC have been restored, the area is delisted. The RAPs have had some important successes. Collingwood Harbour was delisted in 1994, and Spanish Harbour was designated an Area of Recovery in 1999.

Canada – Ontario Agreement

Canada and Ontario have had Great Lakes environmental agreements in effect since 1971. The latest version of the Canada-Ontario Agreement Respecting the Great Lakes Basin Ecosystem (COA) was signed in June, 2002. The agreement provides the framework for systematic and strategic coordination of shared federal and provincial responsibilities for environmental management in the Great Lakes basin. The main objectives are to restore degraded areas, to prevent and control pollution, and to conserve and protect human and ecosystem health.

Ontario Ministry of Environment and Energy

The Ontario Ministry of Environment and Energy (OMOEE) manages a number of programs that contribute to the protection and clean-up of the Great Lakes basin. The Provincial Water Protection Fund assists municipalities to address water and sewage treatment problems and to undertake related studies. The Ontario

Great Lakes Renewal Foundation, established in 1998, provides seed money to support local projects that include habitat restoration and stormwater management. The OMOEE works in partnership with federal and state agencies and municipal governments to achieve numerous environmental goals; the Great Lakes Remedial Action Plans have been a prominent example of such work.

Toronto and Region Conservation Authority

The Toronto and Region Conservation Authority (TRCA) is one of 38 conservation authorities in Ontario that develop and implement programs for the management of water and natural resources on a watershed basis. Conservation authorities are created and given their mandate under the Conservation Authorities Act and involve a partnership of the municipalilties within a watershed and the Province of Ontario. The TRCA jurisdiction includes nine watersheds in the Toronto Region.

The TRCA and the Waterfront Regeneration Trust are the local coordinating agencies for the Toronto and Region Remedial Action Plan. The two agencies help the provincial and federal governments fulfill their obligations under the Great Lakes Water Quality Agreement and Canada-Ontario Agreement. The TRCA's general RAP role is to focus implementation activities on an individual watershed basis and provide technical expertise to its implementation partners. Stormwater management and the remediation of combined sewer overflows are integral to the restoration of the Toronto and Region Area of Concern.

SWAMP

In 1995, the Storm Water Assessment Monitoring and Performance Program (SWAMP) was created as a cooperative initiative of agencies interested in monitoring and evaluating the performance of various stormwater management technologies. The SWAMP program acts as a vehicle whereby federal, provincial, municipal and other interested agencies can pool their resources in support of shared research interests.

The objective of SWAMP is to collect data and report on the performance of stormwater treatment facilities. SWAMP is supported by the Great Lakes Sustainability Fund, the Ontario Ministry of Environment and Energy, the Toronto and Region Conservation Authority, the Municipal Engineers Association, a number of individual municipalities in Great Lakes Areas of Concern, and other owner/operator agencies.

A variety of stormwater management technologies have been developed to mitigate the impacts of urbanization on the natural environment. Prior to the creation of SWAMP, these technologies had been studied using computer models and pilot-scale testing, but had not undergone extensive field-level evaluation in southern Ontario.

The objectives of the SWAMP Program are:

- to monitor and evaluate the effectiveness of new or innovative stormwater management technologies,
- to disseminate study results and recommendations within the stormwater management community.

Technologies that have been addressed by the SWAMP program include:

- wet ponds and constructed wetlands,
- underground storage tanks,
- flow balancing systems,
- oil and grit separators,
- conveyance exfiltration systems.

A number of people have been part of the SWAMP team since the inception of the program. In alphabetical order, the staff members have been:

David Averill	Program Coordinator [July 2001 to present]
David Fellowes	
Rene Gagnon	
Dajana Grgic	
Weng Liang	Program Coordinator [1995 to 2000]
Serge Ristic	
Derek Smith	
Sheldon Smith	
William Snodgrass	Program Coordinator [December to June 2001]
Michael Thompson	
Tim Van Seters	

In addition, several student employees contributed to the success of the projects. Staff of the Ontario Ministry of Environment and Energy, Standards Development Branch, provided administrative and facility support. In addition, Standards Development Branch staff have contributed their technical expertise through informal advice and review of draft reports.

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APPENDIX B

Glossary and Fundamental Concepts of Pond Systems

1.0 GLOSSARY

active storage: see 'extended detention storage'.

adsorption: The adhesion of a liquid, gaseous or dissolved substance to a solid, resulting in a higher concentration of the substance (Raven *et al.*, 1992).

alga; *pl.* algae: Traditional term for a series of unrelated groups of photosynthetic eukaryotic organisms lacking multicellular sex organs (except for the charophytes); the 'blue-green algae,' or cyanobacteria, are one of the groups of photosynthetic bacteria (Raven *et al.*, 1992).

autochtonous: Pertaining to organisms or organic sediments that are indigenous to a given ecosystem (Parker, 1989)

autotroph: An organism that is able to synthesize the nutritive substances it requires from inorganic substances in its environment (Raven *et al.*, 1992)

average event mean concentration (AEMC): The arithmetic mean of two or more individual storm runoff Event Mean Concentrations.

bankfull stage: Typically defined as the elevation of the active floodplain surface. The bankfull stage corresponds go the bankfull discharge, often considered to be the dominant channel forming discharge and has been shown to occur with a frequency of about 1.5 years (Badelt, 1999).

benthic: Pertaining to occurrence on or in the bottom sediments of wetland and aquatic ecosystems (IWA, 2000).

best management practice (BMP): A device, practice, or method for removing, reducing, retarding, or preventing targeted stormwater runoff constituents, pollutants, and contaminants from reaching receiving waters (ASCE, 1999).

catchment: That area determined by topographic features within which falling rain will contribute to runoff to a particular point under consideration. The area tributary to a lake, stream, sewer or drain. See also drainage area, drainage basin, river basin, catchment area, watershed (James and James, 2000).

climax community: The final stage in a successional series; its nature is determined largely by the climate and soil of the region (Raven et al., 1992)

diatom: The common name for algae composing the class *Bacillariophyceae*; noted for the symmetry and sculpturing of the silicieous cell walls (Parker, 1989).

drawdown time: During a storm runoff event, the time required for water levels in a pond, retention basin or tank to return to the water level existing prior to the storm event, beginning at the peak level..

emergent macrophytes: A rooted, vascular aquatic plant that grows in periodically or permanently flooded areas and has portions of the plant (stems and leaves) extending through and above the water column (adapted from IWA, 2000).

eutrophic: pertaining to a water body containing a high concentration of dissolved nutrients; often shallow, with periods of oxygen deficiency (Parker, 1989).

evapotranspiration: The combined processes of evaporation from the water or soil surface and transpiration of water by plants (IWA, 2000).

event mean concentration (EMC): The arithmetic mean concentration of an urban pollutant measured during a storm runoff event. The EMC is calculated by flow-weighting either grab samples or consecutive composite concentrations collected over the course of an entire storm event. (James and James, 2000).

extended detention storage: The storage provided by temporarily retaining water within a basin, tank or reservoir. Also called active storage.

flora: Plants (Parker, 1989).

geotextile: A woven or nonwoven fabric manufactured from synthetic fibers or yarns that is designed to serve as a continuous membrane between soil and aggregate in a variety of earth structures.

glacial till: Unsorted and unstratified drift consisting of a heterogeneous mixture of clay, sand, gravel and boulders which is deposited by and underneath a glacier (Parker, 1989).

groundwater recharge: Replenishment of groundwater naturally by precipitation or runoff or artificially by spreading or injection (James and James, 2000).

groundwater table: The upper surface of groundwater, or the surface below which the pores of rock or soil are saturated (James and James, 2000).

heterotroph: an organism that cannon manufacture organic compounds and so must feed on organic materials that have originated in other plants and animals (Raven *et al.*, 1992)

hydraulic detention time: The time delay in a pond or reservoir between the inlet and outlet hydrograph centroids.

hydraulic residence time (or hydraulic retention time): A measure of the average duration over which an element of fluid occupies a given volume or vessel, as estimated from tracer studies with conservative tracers such as lithium or dyes (adapted from IWA, 2000).

hydraulic conductivity: The rate of water flow through a cross section under a unit hydraulic gradient (Parker, 1989).

hydrograph: A graph showing, for a given point on a stream or conduit, the discharge, stage, velocity, available power, or other property of water with respect to time (James and James, 2000)

hyetograph: A graphical representation of the variation in rate of rainfall over time (James and James, 2000).

hyper- : prefix meaning 'above' or 'over'.

infiltration rate: The rate at which water enters the soil or other porous material under a given condition (James and James, 2000) (also see hydraulic conductivity and permeability)

lag time: In this study, the time delay between the start of rainfall and the influent hydrograph centroid. In other studies, lag time is often calculated as the time delay between the centroids of the rainfall hyetograph and influent hydrograph, or as the time delay between peak rainfall and peak runoff.

left-censored data: Data sets including pollutant concentrations at or below the laboratory analytical detection limit.

mass balance: An accounting for all identified materials entering, leaving, or accumulating within a defined region.

matric forces: Forces acting on soil water that are independent of gravity but exist due to the attraction of solid surfaces for water, the attraction of water molecules for each other, and a force in the air-water interface due to the polar nature of water (Parker, 1989).

olfactory: Of or relating to the sense of smell (Oxford Dictionary, 1995).

peak discharge: The maximum instantaneous flow at a specific location resulting from a given storm condition (James and James, 2000).

peak-shaving: Reduction of peak discharge rates by providing temporary detention in a BMP. Also called peak flow attenuation (adapted from James and James, 2000).

perched water table: The water table or upper surface of groundwater that is unconfined and separated from an underlying main body of groundwater by an unsaturated zone (Parker, 1989)

performance: A measure of how well a BMP meets its goals for stormwater that the BMP is designed to treat. (ASCE, 1999)

periphyton: The community of microscopic plants and animals that grows on the surface of submergent subjects in water bodies (IWA, 2000).

permanent pool volume: A volume of water that is stored permanently in a pond, reservoir or tank, as compared to extended detention volume, which exists only temporarily during storm runoff events.

permeability (of soil): property of soil which governs the rate at which water moves through it (James and James, 2000) (also see infiltration rate and hydraulic conductivity)

phytoplankton: Microscopic algae that are suspended in the water column and are not attached to surfaces (IWA, 2000).

plug flow: Flow in which fluid particles are discharged from a tank or pipe in the same order in which they entered it. The particles retain their discrete identities and remain in the tank for a time equal to the theoretical detention time. A flow value used to describe a constant hydrologic condition. Also a sequence of parcels of water. (James and James, 2000)

porosity: The fraction of a solid, as a percent of its total volume, occupied by minute channels or open spaces (Parker, 1989).

recharge basin: A basin excavated in the earth to receive the discharge from streams or storm drains for the purpose of replenishing groundwater supply (James and James, 2000).

regolith: The layer of rock or blanket of unconsolidated rocky debris of any thickness that overlies bedrock and forms the surface of the land (Parker, 1989).

removal efficiency: A percentage reduction in a specific contaminant or constituent of the wastewater or runoff, as measured across a treatment system or an individual treatment unit.

runoff: That part of the precipitation which runs off the surface of a drainage area and reaches a stream or other body of water or a drain or sewer (James and James, 2000).

runoff coefficient: The ratio of the depth of runoff from the drainage basin to the depth of rainfall (James and James, 2000)

taxon; *pl.* taxa: general term for any one of the taxonomic categories, such as species, class, order or division (Parker, 1989).

transpiration: The transport of water vapour from the soil to the atmosphere through actively growing plants (IWA, 2000).

unsaturated zone: A subsurface zone containing water below atmospheric pressure and air or gases at amospheric pressure (Parker, 1989).

vascular: pertains to any plant tissue or region consisting of or giving rise to conducting tissue *e.g.* xylem, phloem, vascular cambium (Raven et al, 1992).

watercourse: A natural or artificial channel for passage of water (James and James, 2000).

watershed: A topographically defined area drained by a river or a stream or a system of connecting rivers and streams such that all outflow is discharged through a single outlet (James and James, 2000).

zooplankton: microscopic animals that move passively in aquatic ecosystems (Parker, 1989).

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2.0 FUNDAMENTAL CONCEPTS FOR POND SYSTEMS

The purpose of this section of the appendix is to explain the basic principles of stormwater storage pond systems. Material balance principles will be used to derive important relationships and to explain relevant definitions.

2.1 System Definition

Figure B1 illustrates the basic system diagram for a stormwater pond. A fundamental feature of this system is that its operation is not steady-state; the hydraulic and pollutant loadings vary appreciably with time. Storage within the vessel makes the effluent hydrograph differ from that of the influent. Separation of the pollutants, in both suspended and dissolved forms, within the pond can result in both positive and negative removal efficiencies as a function of time and the many mechanisms that control the process. If there is a continuous dry-weather flow through the pond, the effect of storm events is modified by that flow, and vice-versa. In cases without a continuous dry-weather flow (baseflow), operation of the system is completely intermittent and both the storm event and the inter-event quiescent period must be considered.

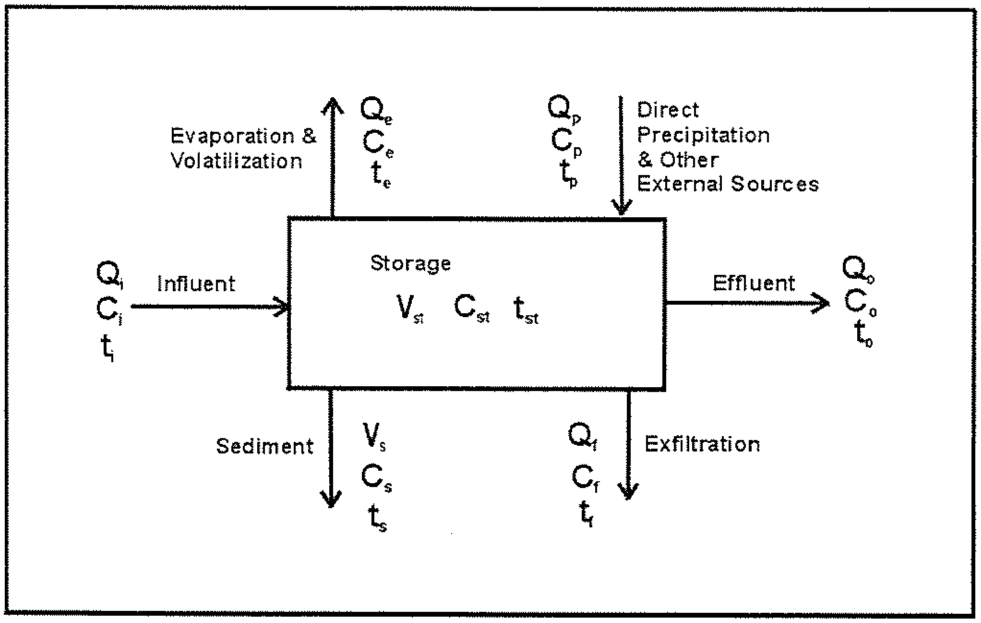

Figure B1: Stormwater Pond Material Balance Diagram

In Figure B1, "Q" represents a flow rate, "C" represents a pollutant concentration and "V" represents a volume. The symbol "t" represents a time period over which the respective flows, concentrations or volumes are being considered, or are of significance. As will be discussed, this time frame is of particular importance in the determination of system performance, particularly in situations that include long inter-event periods (quiescent or low-flow conditions) or emptying of the vessel between events.

Inlet flow (Q_i) and outlet flow (Q_o) are typically represented by time-series graphs called *hydrographs*. Monitoring of the inlet and outlet concentrations may not always continue for the full duration of the respective flows, or for sufficient time to establish complete mass balances. Methods of sampling also vary and can affect the reliability of the resulting performance data.

The volume of water in the pond is typically variable, resulting from the flow-throttling effect of the effluent structure. Concentrations in the pond may be measured only in the more intensive studies. Storage time in the pond has various meanings, as will be discussed.

Exfiltration, through the pond sides or a semi-pervious dam, may be a significant factor in some installations. Conversely, a high water table in the vicinity of the pond may result in infiltration of groundwater into the treatment facility. The quality of infiltration/exfiltration is generally estimated by summing the other flows.

In most stormwater pond studies, losses and gains to and from the atmosphere are seldom considered. These factors are more relevant to lake studies and lake modeling. However, other non-point contributions to the pond can result from waterfowl and other wildlife, including overland drainage from the surrounding area.

The volume and quality of the sediment are important considerations in stormwater ponds. The residence time is governed by decomposition rates and clean-out frequency.

The material balance diagram provides the basis for computing material (mass and volume) balances for the system. An understanding of the dynamics of the system is also necessary to design monitoring programs, and to define parameters representing system performance.

2.2 Quantity Considerations

Stormwater ponds are often designed in accordance with runoff quantity, quality and erosion control objectives. The characteristics relevant to runoff quantity and erosion control will be discussed with reference to actual data from a stormwater storage pond (Figure B2). This example will help to illustrate not only the basic principles but also some of the constraints associated with the analysis of real-world data.

Figure B2a contains the rainfall *hyetograph* and the runoff *hydrograph*. The hyetograph is a plot of rainfall depth versus time; unlike the example in Figure B2a, this data set is often plotted as a bar graph using an inverted y-scale. The hydrograph is a plot of runoff flow rate versus time; in this case, the hydrograph contains the inflow to the stormwater pond. Given the surface area of the catchment, both data sets can be converted to volumes of water, or to a uniform depth of water over the catchment area. The runoff coefficient for the catchment is the ratio of the runoff volume (or depth) to the rainfall volume (or depth); in this case, the value of the runoff coefficient was 0.28. The runoff coefficient is a measure of the ability of the catchment to retain rainfall, such that it percolates into the ground or returns to the atmosphere through evaporation and transpiration, rather than generating runoff. A high value of the runoff coefficient is indicative of a large percentage of impervious surfaces in the catchment. In this example, a little more than one-quarter of the rainfall was measured as runoff.

Various event characteristics related to time and intensity can be extracted from Figure B2a:

- The lag time of the catchment may be expressed as the time delay between the start of the rainfall and the start of runoff at the point of measurement. This quantity may be influenced by the frequency of observation; in the example data set, the rainfall was reported hourly and the runoff was reported every 5 minutes. Lag times also reflect the intensity of the storm, since a light rainfall may be largely contained in depression storage.
- The centroids of the hyetograph and hydrograph may be computed (from the first moment) and used to represent the variables as existing in points of time. This approach is useful in computing inter-event times. The time difference between the centroids also provides an alternative means of characterizing the catchment lag time, one that takes the total volume into consideration and is not biased by the initial rainfall intensity. Baseflow is not included in the calculation of the runoff hydrograph centroid, such that the centroid represents the average runoff conditions independent of the dry-weather flow.
- The durations of both the rainfall event and the runoff are also of interest. Because of the distance over which the runoff must flow, and the resistance to flow created by different surfaces and different paths of flow, the duration of runoff must exceed the duration of rainfall. The duration of the runoff event is measured from the appearance of a flow greater than the baseflow (or dry-weather flow) and ending with the return to baseflow. However, the end of the runoff event may be defined somewhat subjectively because surface and subsurface storage can cause the tails of the runoff curves to persist for long time periods.

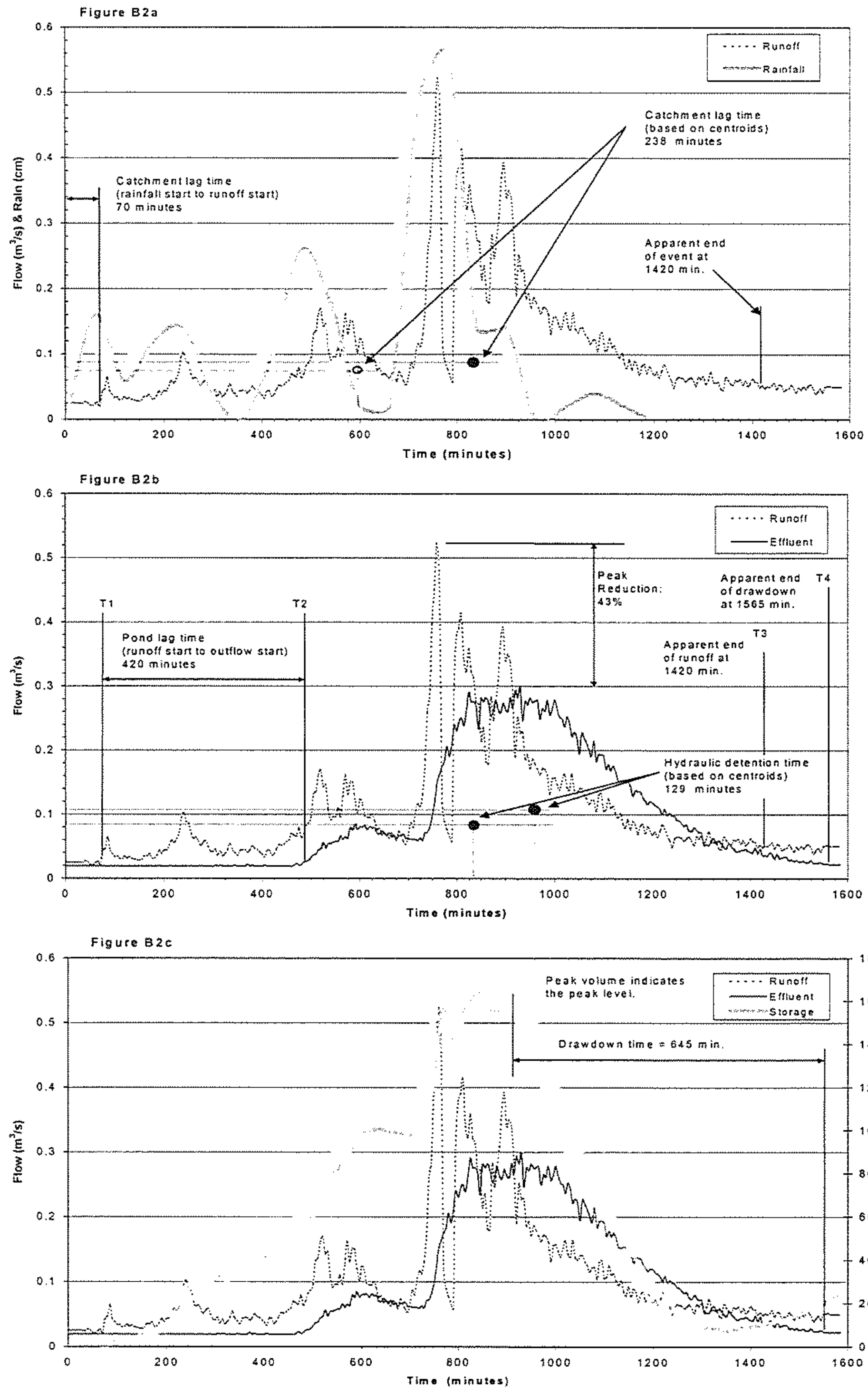

Figure B2: Hydrologic time series graphs for a sample event

- Each curve may be represented by its peak factor: the ratio of the maximum value to the mean. Because of flow attenuation in the catchment, the peak factor for the runoff is expected to be less than that of the rainfall. In some cases, the temporal relationships of the rainfall and runoff peaks may be documented (e.g., a peak-to-peak lag time); however, in events with multiple peaks, the significance of such relationships is not clear. In this case, the peak rainfall and the peak runoff flow were essentially simultaneous, a situation which would not be expected under most (simpler) conditions.

- The base flow, or dry-weather flow, may be different before and after the event. A prolonged dry period before the event would cause a small base flow. The rainfall event would be expected to increase the elevation of the groundwater table, promoting infiltration into the sewer system, and residual surface and subsurface water would enter subgrade drains and other parts of the system slowly. Consequently, the baseflow after the event would be elevated for a considerable time, making estimation of the duration of runoff difficult. The base flow may not return to the initial conditions before the next rainfall event. In the example, the initial and final base flows were smoothed and extended for illustrative purposes; the initial value was 0.025 m^3/s and the final value was 0.050 m^3/s.

Figure B2b contains the runoff hydrograph and the pond effluent hydrograph. Several system characteristics can be determined.

- The lag time of the pond may be expressed as the time delay between the start of the runoff flow (pond influent) and the start of the pond effluent flow. Several factors can influence this variable. In the example, the base effluent flow was often too small to be measured with the installed equipment and some manual extrapolation was employed to adjust the curve[1]. In some cases, a combination of evaporation and exfiltration from the pond can lower the surface of the water below the effluent control structure, producing a storage volume that would otherwise be unavailable and delaying the start of the effluent flow.

- The centroids of the hydrographs may be computed (from the first moment) and used to represent the variables as existing in points of time. The time difference between the centroids is defined as the *hydraulic detention time*, or the average time by which the bulk of fluid is held back or detained by the pond. The hydraulic detention time is determined primarily by the throttling effect of the effluent control structure. It is a measure of the ability of the facility to smooth and extend the runoff hydrograph to reduce its impact on the receiving stream.

- Differences in the durations of the influent and effluent hydrographs are another measure of the flow throttling effect of the facility. Normally, the effluent duration would be expected to exceed the influent duration. However, in this case, the effluent duration was less than that of the influent because of the shapes of the curves and the possible (extra) storage volume. In addition, the effluent was seen to exceed the influent at times, as a result of the irregularity of the rainfall and runoff curves; hence, the pond provided a flow smoothing function as well as attenuation. Also in this case, the average effluent flow was observed to be greater than the average influent flow, as a consequence of uncertainty in the initial conditions.

- Because of flow attenuation in the pond, the peak factor for the effluent is expected to be less than that of the runoff (influent). In some cases, the temporal relationships of the influent and effluent peaks may be documented (e.g., a peak-to-peak lag time); however, in events with multiple peaks, the significance of such relationships is not clear.

[1] Further examination of the effluent level and flow signals may lead to re-interpretation of the initial flow data. Instrument data will be the subject of discussion in a future report.

- The effluent base flow may be less than the influent base flow because of evaporation and exfiltration losses from the pond. At other sites, groundwater may flow into the pond causing the effluent base flow to exceed that of the influent. Also, the initial and final effluent base flows may be different because of changes in these gain or loss rates and in the influent base flow. In this example, the initial effluent base flow was 0.019 m^3/s and the final value was 0.022 m^3/s. The initial and final evaporation/exfiltration losses were therefore approximately 0.006 m^3/s and 0.028 m^3/s respectively. These estimates were affected by the poor quality of the initial data; if the initial effluent base flow had actually been closer to zero, the losses would have been similar.

Figure B2c contains the active (or dynamic) storage volume of the pond together with the influent and effluent hydrographs. The storage volume is calculated from the two sets of flow data. This graph is particularly useful as a means of testing the volumetric balance of the data set. Any deviation from zero storage at the end of the event indicates inaccuracy in the flow measurements and/or the estimation of other gains or losses. In this case, the evaporation/exfiltration losses were estimated from the initial data alone. Failure to include the final baseflow conditions in the calculation procedure is evident in the upward slope of the storage curve after the event. The overall volumetric error was 9%; if measurement of the small initial outflow had been feasible, the computed error may have been smaller.

The water level in the pond is another variable of interest. Water level measurements provide an independent check on volumetric data, providing that a reasonable stage-storage relationship can be derived for the pond based on its geometry. In the example, the pond level was not measured but survey data resulted in a linear stage-storage relationship over the range of active storage volumes. Hence, the pond level is proportional to the stored volume. Knowledge of the water level also permits the computation of another typical pond parameter:

- The *drawdown time* is defined as the period between the maximum water level and the minimum level (dry-weather or antecedent level) in the pond. A theoretical drawdown curve for a pond may be taken as the stage-discharge relationship of a specific effluent control structure. The theoretical value would be approached in practice only if there was no influent flow at the time that the pond was draining. Because there is typically some inflow during this time, the value of the actual drawdown time is expected to exceed that of the theoretical curve.

2.2.1 Summary – stormwater quantity

Table B1 summarizes the hydraulic characteristics of the pond stormwater event used as an example in Figure B2. The underlying principle for runoff quantity analysis is that the displacement of water is acknowledged. In other words, the emphasis is on bulk water quantities. The actual molecules of water entering the system are not necessarily those exiting the system within the timeframe considered. Hence, these quantity relationships should not be confused with the water quality relationships discussed in the next section.

Table B1: Hydraulic Characteristics – Example Pond Event

Parameter	Rainfall	Runoff - Influent	Pond Effluent
Volume (cubic metres)	32,380	8,950	8,130
Duration (minutes)	1,200	1,350	1,075[1]
Runoff Coefficient		0.28	
Pond Volumetric Error[2] (%)			9
Peak Factor	7.6	5.2	3.3
Peak Reduction			43%
Lag Time (minutes)			
- start-to-start		70	420[1]
- centroid-to-centroid		238	129
- peak-to-peak[3]		n/a	n/a
Pond Drawdown Time (minutes)			645

Notes: [1] Difficulty measuring initial effluent flow reduced the duration and increased the lag time.
[2] Volumes and volumetric error are determined after accounting for baseflow.
[3] Peak-to-peak time intervals can not be adequately defined in a multi-peak event.

2.3 Quality Considerations

Stormwater quality refers to the pollutants in the water. Runoff pollutant concentrations typically vary with time as a result of erosive forces (flow rate) and the duration of runoff events. Consequently, water quality data are often represented by *pollutographs*. Pollutographs are measured by collecting discrete samples at uniform time intervals, and are graphed as time-series data sets.

The fate of pollutants in a pond or other treatment system is determined by the physical, chemical and biological forces to which the pollutants are exposed, and the duration of exposure. Each element of fluid that enters the treatment system has a specific residence time (or retention time) within that system. The *hydraulic residence time* is determined by the pond volume, the flow rate and the flow patterns within the pond. The flow rate and the volume of water within the pond vary as described under the heading of "quantity considerations". The flow patterns are determined by several factors including the geometry of the pond, hydraulic conditions at the inlet and outlet, thermal stratification, density stratification and wind effects.

Because different elements of fluid can take different paths through the pond, a range of residence times exists for each facility. This range is quantified as a residence time distribution, which is measured through the use of an inert tracer material. The tracer is added to the inlet flow at a point in time and concentrations in the outflow are measured as a function of time. The average residence time is measured as the centroid of the residence time distribution curve.

The fate of pollutants in a treatment system may be predicted knowing the hydraulic residence time and a "decay rate" specific to each pollutant. The decay rate is the rate of reaction for substances that are destroyed or transformed within the treatment system, or the settling rate for suspended material that is retained within the system. Reaction rates for specific pollutants depend on many physical, chemical and biological factors. Some pollutants may be both settled and reacted. Some substances may be produced within the pond, for example by photosynthesis. Hence, the residence time of a pollutant is specific to each situation. For inert suspended materials, residence time is determined in part by the frequency of clean-out operations. Inert soluble materials such as chloride may follow the flow paths and leave the ponds in the effluent or the exfiltration flow, but may also be stored for extended periods of time in density layers within the ponds.

No tracer tests were undertaken for the pond used as an example above. Hence, the hydraulic residence times were not determined. A general impression of the hydraulic residence time may be obtained by assuming steady-state flow, an average pond volume and plug-flow conditions (no mixing of influent and pond contents and no short-circuiting of flow). If the average flow were 0.1 m^3/s and the average volume were 7000 m^3 (both consistent with the above example), the hydraulic residence time would be 1,170 minutes (19.4 hr.) under plug-flow conditions. Short-circuiting of flow, internal mixing and other factors would tend to reduce that value, on average. However, since many rainfall/runoff events are shorter than 19 hours, some of the runoff may be expected to reside in the pond for several days (inter-event periods). Also, eddy currents and dead spaces within the ponds can hold elements of water and associated pollutants for extended periods of time and produce long tails on the residence time distribution curves.

2.4 Discussion

Hydraulic detention time and hydraulic residence time (a.k.a. hydraulic retention time) are two distinctly different concepts and are used for different purposes. Detaining, delaying or holding back runoff is an important aspect of hydraulic control – the flattening of runoff hydrographs. Retaining, storing or holding volumes of stormwater is an important aspect of pollution control – the destruction or separation of pollutants. Detention times and residence times can be vastly different within any given system. Figure B3 illustrates extreme conditions that emphasize the choice of appropriate system characteristics.

A long, narrow pond with inlet and outlet structures at either end (Figure B3a) forces the flow to proceed under essentially "plug-flow" conditions, such that each element of flow entering the pond has essentially the same residence time as well as the maximum time permitted by the pond volume and flow rate. The average residence time under such conditions could be measured in days. The water level in the pond, however, responds quickly to inflow. If there is minimal effluent flow throttling, the effluent hydrograph could follow very quickly after the influent hydrograph, resulting in a hydraulic detention time of minutes.

The other extreme case is a long, thin pond with the inlet and outlet structures located very close together (Figure B3b). The pond may be large with good effluent flow throttling, resulting in a long hydraulic detention time. However, elements of the influent flow can proceed quickly from the inlet structure to the outlet structure or, if stored for longer periods of time, would not migrate far from the two structures such that they are discharged before significant treatment can occur. The hydraulic residence time in this case is very short, and much of the volume of the pond is essentially inactive from the perspective of quality control.

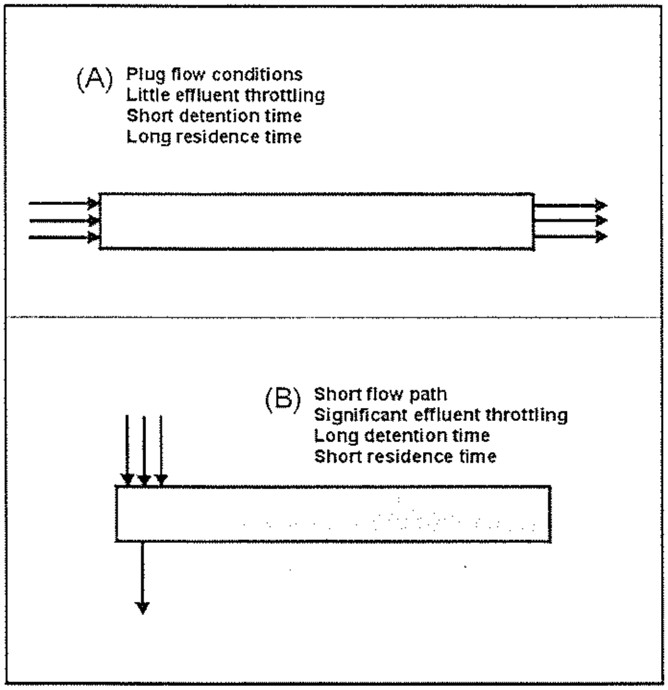

Figure B3: Detention and Residence Time Scenarios

There is a tendency in the stormwater literature to interchange – or at least confuse - hydraulic detention time and hydraulic residence time. Hydraulic detention time may be discussed (incorrectly) in the context of settling rates or treatment efficiency. Assuming that pond geometry guidelines are followed, a reasonable correlation between detention and retention times would likely exist and, by extension, a correlation between detention time and treatment efficiency. However, such correlations do not imply a cause-and-effect relationship, nor can they be used to examine removal mechanisms. Only an extensive review of performance data would indicate whether any such correlations may be reliable, and within what range of system geometry.

2.5 Performance

2.5.1 Volume, mass and concentration

The total volume and total pollutant mass found in any water or wastewater stream may be determined by summation over the appropriate time intervals. For example, with reference to Figures B1 and B2, the influent volume (V_i) and influent pollutant mass (M_i) are calculated as:

$$V_i = \sum_{k=T1}^{T3} Q_{i_k} \, \Delta t_k \qquad \text{(B-1)}$$

$$M_i = \sum_{k=T1}^{T3} C_{i_k} \, Q_{i_k} \, \Delta t_k \qquad \text{(B-2)}$$

where: Q = flow measured over finite time interval, Δt
C = concentration of a specified pollutant measured over finite time interval, Δt
T1 represents the start of the runoff (influent) flow
T3 represents the end of the runoff (influent) flow

The flow-weighted average influent pollutant concentration ($\overline{C}_i$) may be determined from the total influent mass and the total influent volume:

$$\overline{C}_i = \frac{M_i}{V_i} \tag{B-3}$$

Similarly, the volume, mass and a flow-proportioned mean concentration may be calculated for the effluent or any other significant flow.

Ideally, the average pollutant concentration measured at a specific location for one event is determined by integration of continuous data or the summation of multiple flow-weighted discrete observations. However, sampling programs seldom generate sufficient data for a rigorous analysis. The average concentration is often determined from composite samples. Important considerations include whether or not the composite sample was flow-proportioned (flow-weighted) and whether the sampling period included all of the runoff event[2].

Given appreciable temporal variation in most storm events (i.e., in hydrograph and pollutograph shapes), the lack of flow-proportioned samples can result in appreciable error. The worst case scenario consists of simultaneous peaking of the hydrograph and pollutograph, such that high concentrations occur at high flow and a large mass of pollutant is transported during that part of the event. Hence, the type of sampling should be indicated when an average concentration is reported.

An average concentration, measured at a specified location over the duration of one event, is typically called the *event mean concentration* (EMC). Ideally, the type of sampling used to determine the EMC should be indicated:

EMC^p = flow-proportioned event mean concentration

EMC^t = time-averaged or non-flow-proportioned event mean concentration

B-2.5.2 Event efficiency -- load-based

Load-based efficiency (LE) is defined as the ratio of the mass of a specific pollutant removed to the corresponding influent concentration[3]. This parameter may also be referred to as *mass efficiency.* The *LE* is determined by considering the entire event cycle: the time from the start of the stormwater flow to the end of the effluent drawdown curve. Equation 4 is written using the summation of incremental mass quantities (the product of flow and pollutant

[2] The selection and programming of sampling equipment, as well as other sampling logistics considerations will be the subject of a subsequent report.

[3] In this document, removal efficiency is expressed as a fraction rather than a percentage, primarily to simplify the equations.

concentration over finite observation intervals). Ideally, all sources and destinations of flow and pollutants would be considered; practically, only the influent and effluent are included in the definition of efficiency.

$$LE = \frac{\sum_{k=T1}^{T3} Q_{i_k} C_{i_k} \Delta t_k - \sum_{k=T2}^{T4} Q_{o_k} C_{o_k} \Delta t_k}{\sum_{k=T1}^{T3} Q_{i_k} C_{i_k} \Delta t_k} \qquad \text{(B-4)}$$

Equation B-4 may also be written using the sums of all mass loads entering *(SOL_{in})* and leaving *(SOL_{out})* the facility.

$$LE = \frac{SOL_{in} - SOL_{out}}{SOL_{in}} \qquad \text{(B-5)}$$

In Equation B-5, the summations are assumed to be over the time periods relevant to the influent and effluent.

B-2.5.3 Event efficiency - concentration-based

In stormwater studies, flow and volume data may not always be available. In such cases, the Event Mean Concentration *(EMC)* is an average concentration that has been obtained without flow-proportioned sampling. Using the *EMC* values, a concentration-based pollutant removal efficiency for a single event may be defined as follows:

$$CE = \frac{EMC_i - EMC_o}{EMC_i} \qquad \text{(B-6)}$$

$$CE = 1 - \frac{EMC_o}{EMC_i} \qquad \text{(B-7)}$$

This expression for concentration-based efficiency is the definition of efficiency commonly used for continuous-flow clarifiers with negligible underflow.

B-2.5.4 Residence time and intermittent operation

There are further complications to be considered when examining effluent samples and calculating removal efficiencies. These considerations are consequences of the long residence times and intermittent operation common to stormwater treatment systems.

Ideally, removal efficiency should be associated with each element (or incremental volume) of suspension that enters the treatment system. Each element of fluid entering the system contains a specific matrix of pollutants that will be removed in accordance with their characteristics, the hydraulic and other conditions in the system, and the time during which the element of fluid resides in the system. Comparison of the characteristics of that element of fluid, as it leaves the system, with its initial characteristics would provide a true measure of treatment efficiency.

Consider a large wet pond treatment system:

- Effluent flow at the start of an event consists primarily of displaced fluid that had been in the pond since the previous event or had accumulated during the intervening dry-weather period. The long residence times for these elements of fluid would probably result in pollutant concentrations equivalent to the non-settleable (non-treatable) residual concentrations.
- As the event progresses, the component of the effluent flow generated by the current event begins to increase. Some influent flow will mix with the pond contents and some elements of the influent may short-circuit to reach the effluent structure before the majority of the flow. The result is measurement in the effluent stream of partly diluted and partly settled current-event influent.
- In moderate-size events, the remainder of the influent fluid elements would reside in the pond until the next event or until they are gradually displaced by dry-weather flow. These elements would be expected to receive the maximum treatment efficiency possible for the specific installation.
- In large events, the total contents of the pond may eventually be exchanged. The effluent would then reflect only the current influent conditions and the treatment efficiency of the pond in continuous (flow-through) operation mode.

Effluent samples are typically collected during each runoff event and only for the duration of the event hydrographs. Effluent quality from that sampling period may be compared directly to the influent quality from the same event to estimate treatment efficiency. The result is a measure of the change in water quality across the pond, and the reduction in pollutant loading during that specific event. However, that procedure ignores the residence time in the system and may introduce significant errors in examining the removal mechanisms and determining the overall environmental loadings from the facility.

Ideally, the least error would result from continuous measurement of influent and effluent during both wet-weather and dry-weather. Short-term efficiency would be best represented by comparison of influent samples to effluent samples with the latter offset by the residence time in the system. However, the residence time could not be measured on a continuous basis because it is a distribution that is influenced by many physical factors, and it is measured by a pulse addition of a tracer. The concept of following elements of fluid through the treatment system may be appropriate to numerical simulation techniques[4].

Inter-event (or dry-weather) flow and pollutant loading are often not considered. Low flows and small concentrations may be difficult to measure, and differential concentrations (removals) may not be significant numbers. However, the long dry-weather time periods can conceptually result in large volumes and pollutant masses.

Practically, composite samples are collected for each event and few - if any - samples are collected between events. Hence, the data analysis options are: (1) compare the effluent data to the influent data of the same event, (2) compare the effluent data to the influent data of the previous event, or (3) calculate efficiency based only on long time periods

[4] Numerical simulation of stormwater ponds will be the subject of a subsequent report.

considering the total influent and effluent masses (long-term mass efficiency). The latter option will provide the best estimate of system efficiency.

2.5.5 Long-term efficiency - load-based

Load-based efficiency calculations provide the most accurate method of determining long-term efficiency. In this procedure, the summations are made over the full time frame of interest (several events, a season, a year or several years).

The sum-of-loads concept may be expressed in terms of *EMC* values and event volumes *(V)*. Hence, the efficiency ratio based on mass load for a single event is:

$$LE_{emc} = 1 - \frac{EMC_o \times V_o}{EMC_i \times V_i} \tag{B-8}$$

An average efficiency ratio could be calculated for several events:

$$ALE_{emc} = \frac{\sum_{j=1}^{m} LE_j}{m} \tag{B-9}$$

where: m represents the number of events.

However, a simple average of efficiencies gives equal importance (weight) to each event, regardless of event size. A better estimate of long-term efficiency is obtained by totaling the mass quantities over the time period of interest:

$$SLE_{emc} = 1 - \frac{\sum_{j=1}^{m} EMC_{o_j} \times V_{o_j}}{\sum_{j=1}^{m} EMC_{i_j} \times V_{i_j}} \tag{B-10}$$

Table B2 contains an example of the extent to which averaging of event performance can distort the estimate of long-term efficiency. In this hypothetical example, one large event, one small event and two moderate-sized events each have reasonable TSS removal efficiencies. A simple average of the four efficiencies, however, does not adequately represent actual system performance.

These definitions of efficiency are not as rigorous as those derived from material balance principles. The difference is that the composite samples that are used to determine the *EMC* values were not necessarily flow-proportioned. However, from a practical perspective (given current sampling practice), mass loading based on *EMC* values and averaged over as large a variety of events as possible is the best feasible method of representing stormwater pond performance.

Table B2: Hypothetical Data Set - Effect of Averaging Performance Data

Event No.	Volume	*EMC* in	*EMC* out	% Rem.	Mass in	Mass out
1	2,000	125	50	60	250,000	100,000
2	500	110	15	86	55,000	7,500
3	10,000	165	120	27	1,650,000	1,200,000
4	1,500	115	30	74	172,500	45,000
ALE				62		
Total					2,127,500	1,352,500
SLE				36		

2.5.6 Long-term efficiency – concentration-based

Flow and volume data are not always available; consequently, pollutant mass can not be determined. In such cases, an average event mean concentration *(AEMC)* may be calculated for several events, for example over one year or a runoff season.

$$AEMC = \frac{\sum_{j=1}^{m} EMC_j}{m} \qquad \text{(B-11)}$$

where: m represents the number of events.

Similarly, long-term *average efficiency (ACE)* can be calculated from *AEMC* values:

$$ACE^{*} = \frac{AEMC_i - AEMC_o}{AEMC_i} \qquad \text{(B-12)}$$

$$ACE^{*} = 1 - \frac{AEMC_o}{AEMC_i} \qquad \text{(B-13)}$$

Alternatively, individual efficiencies can be averaged. Numerically, averaging the concentrations over a season and calculating a seasonal efficiency based on averages is not the same as calculating individual *EMC*-based efficiencies and averaging them $(ACE^{*} \quad ACE^{\#})$.

$$ACE^{\#} = \frac{\sum_{j=1}^{m} CE_j}{m} \qquad \text{(B-14)}$$

APPENDIX C

Analytical Procedures and List of Organic Parameters Analyzed

***Table C1:* OMOEE Analytical Procedures Employed in the Harding Park Stormwater Pond Retrofit Study**

Method Number	Product Number	Constituent	Procedure	Comments
E3016A	CL3016	Chloride	Colourimetry following two-stage reaction with mercuric thiocyanate and ferric iron	interferences from bromide, iodide, sulphide, cyanide, thiosulphate
E3060B	HG3060	Mercury	Cold vapour flameless atomic adsorption spectrophotometry (CV-FAAS) at 253.7 nm following acid digestion and reduction with stannus chloride solution	method is suitable for "clean" waters
E3080A*	MET3080	Metals	Inductively coupled plasma-optical emission spectroscopy (ICP-OES) analysis following preconcentration and digestion with nitric acid and aqua regia -- only the supernatant is analyzed	A preconcentration step (evaporation) used in a previous method was found to result in lower levels for some elements.
E3089A	ASSE3089	Arsenic, Selenium & Antimony	Flameless atomic adsorption spectroscopy (FAAS) following acid digestion and hydride generation	
E3119A	CPA3119	Chlorophenols and phenoxyacid herbicides	Solid phase extraction (SPE) using pre-conditioned C_{18} cartridges followed by elutriation with solvent, treatment with diazomethane and analysis gas chromatography with electron capture detectors (GC-ECD)	16 compounds
E3120B	OCS3120	Organochlorine pesticides (OC's), polychlorinated biphenols (PCB's) and other chlorinated organic compounds	GC-ECD following solvent extraction and clean-up with Florisil™	38 compounds
E3265A	PAH3265	Acid/base and neutral compounds	In-situ acetylation and liquid/liquid extraction with dichloromethane, followed by drying with sodium sulphate, concentration and analysis by gas chromatography with a mass selective detector (GC-MSD) with single ion monitoring (SIM) data capture	a wide range of organic compounds including phenolics and PAH's
E3289A	PHALCO 3289	Conductivity, pH, Alkalinity	Automated system using electrodes in a constant temperature bath for conductivity, a calibrated potentiometric system for pH and titration for TFE alkalinity (to an end-point of pH 4.5)	Supernatant or filtrate is analyzed. Gran alkalinity by special request only
E3311A	TURB3311	Turbidity	Measurement of light scattering at 90° ±30° by nephelometry calibrated to Formazin turbidity standards	

E3328A	PART3328	Particle size	Optical – laser light diffraction (Coulter LS130 Particle Size Analyzer)	0.1to 900 µm in 27 size channels. Reported as % by volume (no count data)
E3334A	ID3334 SXT3334	Organic Solvent Extractable Matter (liquid-liquid or liquid-solid extraction)	Diffuse reflectance infrared Fourier transform spectroscopy (DR-IR) after extraction with dichloromethane	3 major groups: non-volatile petroleum hydrocarbons, cooking oils, soaps and detergents (Presence of humic acid detected in several samples).
E3364A	DISNUT 3364	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 - phosphorus, as orthophosphate, by colourimetric method folowing reaction with ascorbic acid	
E3365A	SS3365	Suspended Solids	Suspended solids are determined as the material removed from suspension by a 1.5 to 2.0 µm glass fibre filter, after drying at 103° ±2°C	
E3367A	TOTNUT 3367	Total nutrients: total P TKN	Total P: digestion in sulphuric acid, mercuric oxice, 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	
E3370A	DCSI3370	Silicon: reactive silicate Dissolved organic carbon 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 measured 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.	

E3371A	ECFSPS 3371	Escherichia coli Fecal streptococcus	Membrane filtration procedures are used to recover and enumerate several bacteria or bacterial groups. The culture media, incubation temperatures and incubation periods are specific to each bacterial analyte.	
E3386A*	MET3386	Metals	Inductively coupled plasma (ICP) following ultrasonic nebulizer	Digestion is not used.

* Note: The analytical method for metals changed from MET3080 up to and including Aug. 9, 1996 to MET3386 after this time.

Table C2: Analytical Detection Limits and Provincial Water Quality Objectives (PWQOs) for Herbicides, Phenols and PAHs Analyzed in this Study

Polynuclear Aromatic Hydrocarbons	**Reporting Method Detection Limit (µg/L)**	**PWQO Limit (µg/L)**
Napthalene	1.6	7
2-methylnaphthalene	2.2	2
1-methylnaphthalene	3.2	2
2-chloronaphthalene	1.8	0.2
Acenaphthene	1.3	
Acenaphthylene	1.4	
Fluorene	1.7	0.2
Phenanthrene	0.4	0.03
Anthracene	1.2	0.0008
Fluoranthene	0.4	0.0008
Pyrene	0.4	
Benzo(a)anthracene	0.5	0.0004
Chrysene	0.3	0.0001
Benzo(b)fluoranthene	0.7	
Benzo(k)fluoranthene	0.7	0.00002
Benzo(a)pyrene	0.6	
Indeno (1,2,3-c,d) pyrene	1.3	
Dibenz(a,h)anthracene	1.3	0.002
Benzo(g,h,i)perylene	0.7	0.00002
1-chloronaphthalene	2.5	0.1
Perylene	1.5	0.00007
Indole	1.9	
5-nitroacenaphthene	4.3	
Biphenyl	0.6	0.2
Herbicides and Pesticides		
2,4-dichlorophenol	2.0	0.2
2,4,6-trichlorophenol	0.02	18
2,4,5-trichlorophenol	0.1	18
2,3,4-trichlorophenol	0.1	18
2,3,4,5-tetrachlorophenol	0.02	1
2,3,4,6-tetrachlorophenol	0.02	1
Pentachlorophenol	0.1	0.5
Dicamba	0.05	200
Bromoxynil	0.05	
2,4 - D-propionic acid	0.1	
2,4 -D	0.1	4
Silvex	0.02	
2,4,5 -T	0.05	
2,4 -DB	0.2	
Dinoseb	0.02	
Picloram	0.1	
Diclofop-methyl	0.1	

Note: Only pentachlorophenol and 2-3-4-6 Tetrachlorophenol were observed at concentrations consistently above laboratory analytical detection limits.

APPENDIX D

Vegetation Monitoring

The following report was produced by the Toronto and Region Conservation Authority (TRCA). It includes studies of a highway stormwater retention pond adjacent to the Rouge River in Toronto and the Harding Park retrofit pond in Richmond Hill. This document has been reformatted but is otherwise reproduced as submitted by TRCA.

STORM WATER ASSESSMENT MONITORING & PERFORMANCE (SWAMP) PROGRAM

VEGETATION MONITORING COMPONENT

FINAL REPORT for YEARS 1 & 2

June, 1998

Prepared by: Jennifer Vincent (TRCA) and Gavin Miller (TRCA)

STORM WATER ASSESSMENT MONITORING & PERFORMANCE (SWAMP) PROGRAM

AQUATIC VEGETATION MONITORING COMPONENT

Final Report for Years 1 & 2 – June, 1998

1.0 BACKGROUND

The Storm Water Assessment Monitoring and Performance (SWAMP) Program is an initiative of the Government of Canada's Great Lakes Sustainability Fund, the Ontario Ministry of Environment and Energy (OMOEE), The Toronto and Region Conservation Authority (TRCA), and the Municipal Engineer's Association. A number of individual municipalities and other owner/operator agencies have also participated in SWAMP studies.

As urban areas within the Great Lakes Basin expanded during the mid to late 1980s, stormwater runoff associated with urban growth increased. The increase has had a pronounced environmental effect on water quality and fish habitat raising concerns over stormwater management. In response to these concerns, a variety of stormwater management technologies have been developed to mitigate the impacts of urbanization on the natural environment. These technologies have been studied using computer models and pilot-scale testing, but have not undergone extensive field level evaluation in southern Ontario. The SWAMP Program evaluates these technologies at the field level. The purpose of the SWAMP Program is: to monitor and evaluate new and conventional stormwater management technologies; to disseminate study results; and to make recommendations to the stormwater management (SWM) industry. Monitoring components include: rainfall, flow, water quality and temperature, sediment particle size distribution, sediment quality, toxicity, and vegetation.

The research addresses questions raised by SWM practitioners concerning the performance of SWM facilities in improving stormwater quality. Studies will also respond to questions regarding appropriate plant species and effective planting strategies in facilities with a constructed wetland component. Based on the Toronto area experience, the aquatic plant component of a SWM pond facility can represent up to 7% of the total facility construction cost to the developer. Aquatic plants can represent up to 30% of the total planting plan cost. Therefore, the developer and the municipality (which often becomes the owner of the facility) both have an interest in ensuring that the plant species selected and the planting strategy employed will be the most suitable for conditions found in the stormwater management facility. Conservation agencies and municipalities are interested in ensuring that the plants fulfil short term objectives of soil stabilization and provide optimal pollutant removal over the life of the facility. Vegetation community monitoring and assessment have, therefore, been included as part of the monitoring program applied to these facilities.

It is important for project managers to design and implement planting plans that are compatible with site conditions, will provide a basis for other plants to colonize, meet sediment and erosion control objectives while plants are establishing, and are cost effective. In order to provide insight into which plant species would best accomplish these objectives, this study examined the establishment of a plant community, documenting which plants were present in, or dominated, the aquatic vegetation community and at what times of the year. Dominance is a function of the plant's ability to compete within the community structure and a function of season, as plants mature at different times of the year.

In 1996 and 1997, the TRCA undertook aquatic vegetation community monitoring on behalf of the SWAMP Program. This monitoring program looks at the ***aquatic*** vegetation community only. Monitoring of the algal community was undertaken by Dan Olding, and results are documented in a separate report. Monitoring of the terrestrial component of planting plans was deemed to be beyond the scope of this program.

This report summarizes the results of the first two years of aquatic vegetation community monitoring at two newly constructed SWM ponds within the Greater Toronto Area: the Ontario Ministry of Transportation's (MTO) Rouge/401 SWM pond and the Town of Richmond Hill's Harding Park SWM retrofit pond.

1.1 Literature Review

Loiederman Associates, Inc. (1996) note findings from a literature review on the subject of vegetation in stormwater wetlands:

- Vegetation contributes to the water quality function of stormwater ponds. Nutrients are assimilated into plant biomass, providing temporary storage. Dead and decaying biomass can fuel reduction/oxidation processes such as nitrification/denitrification, providing both substrate and carbon sources. Plants transport oxygen deeper into the soil than it would travel by diffusion alone.

- Many of the biogeochemical processes involved with water quality treatment, including nitrification/denitrification, phosphorus retention, and pollutant immobilization, can be linked with oxygen availability.

- The average root depth penetration of wetland plants varies with species. (E.g., cattails root down to 30 cm, reeds root down to 60 cm, and bulrushes root down to 75 cm.) Wetlands with a *variety* of plant species can therefore expand the aerobic zone of the soil, enhancing removal of biological oxygen demand (BOD).

- The nutrient assimilative capacity of wetland plants varies with species, even in the same habitat.

- Constructed wetlands were found to exhibit a high percent cover of non-native species, which peaked one to four years after construction and declined to natural levels in seven years.

- The diversity in new wetlands many diminish after a few years, due to competitive exclusion and species dominance.

- Planted wetlands tend to maintain or have increased species richness and diversity when compared to wetlands that are not planted but instead rely on natural colonization. Unplanted wetlands may be dominated by a few species. Planted wetlands resist domination by invading colonizers.

- Stormwater ponds planted with a greater diversity of plant species may perform water quality functions, such as nutrient removal, better than those having few species.

Data from Loiderman Associates (1997) indicate little difference in species richness and diversity in stormwater wetlands that were five to seven years old compared to stormwater wetlands that were ten to twelve years old, in Maryland and Virginia. This may indicate that the wetland plant dynamics have stabilized in five years.

1.2 Goal

The goal of the Aquatic Vegetation Monitoring Program is to develop a list of recommended vascular wetland plant species and recommended planting strategies for stormwater management pond projects in the Greater Toronto Area.

1.3 Objectives

This goal will be achieved through the following objectives:

- To monitor the effectiveness of planting plans in developing a balanced desirable aquatic vegetation community.

- To identify the presence of plant species below the "top of active storage" line for each cell of the stormwater management pond.

2.0 STUDY SITES

2.1 Harding Park Regeneration Project, Richmond Hill, Ontario

This constructed wet pond/wetland facility is a retrofit of an existing dry flood control pond. The retrofit project was constructed in 1995 by the Town of Richmond Hill, in response to recommendations in *Forty Steps to a New Don*, a regeneration strategy for the Don River watershed. The facility consists of a sedimentation forebay, a wet pond, and a wet meadow area. The total storage volume meets current guidelines for stormwater quality and erosion control and maintains the original flood storage capacity.

The planting plan for this facility concentrated on terrestrial and meadow marsh type plants. No true aquatic plants were introduced. As such this site presented the opportunity to monitor which aquatic plants would colonize naturally.

2.2 Highway 401/Rouge River Stormwater Management Facility, Scarborough, Ontario

This extended wet pond was constructed in 1994 by the Ministry of Transportation as part of a Highway 401 widening project. It was constructed to address water quality and fisheries concerns originating from highway water runoff. The facility is designed with a submerged impermeable weir which partitions the pond into a forebay and a quiescent treatment zone. The outflow structure consists of a reversed slope pipe to draw water from below the permanent pool level. This minimizes the impact of the elevated runoff water temperature on the cooler waters of the Rouge River.

The planting plan for this facility comprised both a terrestrial and aquatic component. This study is looking at the aquatic component only. There were 5 aquatic/meadow marsh species planted at the Rouge/401 SWM pond: 156 common arrowhead, 350 softstem bulrush, 60 fragrant waterlily, 88 curled pondweed, and 496 reed canary grass. Of these five species, curled pondweed is a nonnative submergent, fragrant waterlily is a floating leaf, and arrowhead, bulrush, and reed canary grass are emergent to meadow marsh. While reed canary grass is native, it is considered an invasive.

3.0 METHODS

Many aquatic plants (e.g., sedges) are difficult to identify without having the fruiting bodies. By visiting the sites multiple times throughout the growing season we were able to confirm the identification of some of these more difficult plants. All plants were identified at minimum to the genus level. The vast majority of the plants found were identified on-site to the species level. For those species that were not identified in the field, a sample was taken and identification of the plant was verified by a botanist using the appropriate keys referenced at the end of this report. The dominant plant species within each area was determined visually.

The TRCA began vegetation monitoring at the Town of Richmond Hill's Harding Park Stormwater Management Pond in 1996 after completion of the reconstructed pond facility. The newly developing vegetation community was inventoried twice in 1996 (June 22 and September 17) and three times in 1997 (June 26, August 5, and September 24).

On August 21, September 15 and October 12, 1995 the MTO visited the Rouge/401 pond to inventory the newly establishing vegetation community. It is believed that the MTO followed a similar methodology and therefore the results from 1995 are comparable to 1996 and 1997. In 1996 and 1997 the TRCA continued with this role at the Rouge/401 pond, inventorying the vegetation community on three occasions each year (June 22, August 1, and September 17, 1996; June 26, August 5, and September 24, 1997).

The intent of this monitoring program is to identify **aquatic** plant community establishment. To do this it was decided that all plants found below the "top of active storage" line would be identified. This recognized that due to the frequent water fluctuations of a stormwater management pond, the transition zone between aquatic and terrestrial is blurred.

"Top of active storage" is the maximum height to which stormwater will rise within the facility. The difference between the top of active storage and the permanent storage can be as high as one metre. This water fluctuation zone develops into a diverse vegetation community consisting of both terrestrial and aquatic vegetation species. The "top of active storage" was determined using the design drawings, an "as-built" bathymetric (contour) map, and confirmed visually in the field. At the Rouge SWM pond this was verified as the absence of woodchips used in the planting beds (i.e., wood chips float to shore at the highest water level). At the Harding Park SWM pond this was verified using the locations of the concrete pillars, incorporated into the pond design for the purposes of future monitoring.

Vegetation establishment is not an instantaneous event. It takes five years or more for a wetland community to mature. A couple of years of very dry or very wet weather can dramatically affect this process in a wetland. For these reasons, aquatic vegetation monitoring should be continued over several years until the vegetation community stabilizes.

4.0 RESULTS AND DISCUSSION

Tables in Annex D1 list all plant species found at the Rouge/401 MTO SWM Pond and the Harding Park SWM Pond. The tables also provide information about each plant's native status and habitat requirement. The status of plants observed at the two study sites was determined using *Distribution and Status of the Vascular Plants of Central Region, Ontario Ministry of Natural Resources* (Riley, 1989). Symbols used in these and other tables in this report are defined by Riley as follows:

n	the species is considered native to Ontario's Central Region.
+	the species is introduced or escaped from cultivation in Central Region
(+)	the species may be considered native in some regions but is introduced to Central Region.
?	the status of the species was unknown.

The following symbols and definitions are used in this report for the habitat in which these plants may be found.

d	*disturbed*	a recently altered natural state (e.g.: due to construction)

m	*meadow*	closed graminoid and herb vegetation behind areas of shoreline emergent vegetation and on wet floodplains adjacent to open water systems. Usually seasonally flooded or subject to storm floods.
mm	*meadow marsh*	having a canopy of 75% to 100% with standing water and/or muck/mud flats beneath canopy or between clumps; characterized by more or less continuous stands of dominant graminoids of medium to low stature with surface water; water depth up to 1 m (flooded), but usually shallower, or exposed mud, during much of the summer.
upland	*upland*	well-drained hilltops, steep to moderate slopes, sand flats, etc. Stands normally dominated by dryland species of trees, shrubs, and/or herbaceous ground vegetation.
sm	*shallow marsh*	having a canopy of 75% to 100% with standing water and/or muck/mud flats beneath canopy or between clumps. Characterized by more or less continuous stands of tall emergent aquatics with surface water up to 1 m (flooded), but usually less during much of the summer months.
e	*emergent*	emergent aquatic vegetation in or adjacent to open shallow water, pools or channels; commonly interspersed or dominated by clumps of vegetation (rooted, unconsolidated, or floating) with open water channels between or with open water beneath the canopy of sedges, grasses, reeds, cattails; cover by emergents or shrubs greater than 25%.

(The above-noted definitions are from *Ontario Wetland Evaluation System for Southern Ontario* - Ontario Ministry of Natural Resources, 1993)

a	*aquatic*	adapted to living partially or wholly submerged in water or in waterlogged soils.
r	*riparian*	growing adjacent to a river or stream including shores and floodplains.

(The above-noted definitions are from *Wetland Plants of Ontario* - Newmaster *et al*, 1997)

m	*mesic*	characterized by moderately moist conditions; neither too moist nor too dry (*Dictionary of Biology* - Steen, 1971)
sub	*submergent*	growing below the water surface
f	*floating*	the majority of the plant grows on the water's surface

4.1 Harding Park Stormwater Management Pond

Two years of monitoring the aquatic vegetation community at the Harding Park facility has resulted in some interesting observations. Due to the fact that wetland vegetation takes, on average, five years or more to become well established, it is too early, after two years, to make any clear conclusions. Nevertheless, several trends are beginning to emerge.

Table D1 illustrates which meadow marsh plants were introduced and which ones are still found on-site after two years of monitoring. It is interesting to note that of the 11 species originally planted, seven can still be found within the pond. Four plant species did not survive. The reasons for this could be improper placement for their habitat requirements or the possibility that the stock received was not in good health. One species of concern that was planted is Common Reed. It is considered an invasive plant that, while native to Central Region, is not normally suggested in plantings, as it will almost always colonize on its own and has a strong tendency to "take over" an area. This reduces the vegetation community's plant diversity that reduces its ability to provide good quality habitat for fauna.

Table D1: The fate of marsh meadow plants planted at the Harding Park SWM pond.

COMMON NAME	SCIENTIFIC NAME	Status	Originally Planted	Present in 1996	Present in 1997
New England Aster	*Aster novae-anglia*	n	x	x	x
Turtlehead	*Chelone glabra*	n	x	x	x
Spotted Joe-pye-weed	*Eupatorium maculatum*	n	x	x	x
Boneset	*Eupatorium perfoliatum*	n	x		x
Sweet Joe-pye-weed	*Eupatorium purpure*	n	x		
Helen's flower	*Helenium autumnale*	+	x	x	x
Stella d'or daylily	*Hemerocallis "stall d'oro"*	+	x		
Bergamot	*Monarda didyma*	n	x		
Common Reed	*Phragmites australis*	n	x		x
False Dragonhead	*Physostegia virginiana*	?	x	x	x
Black eyed susan	*Rudbeckia hirta*	n	x		

Status: n = native species + = introduced species ? = unknown status

Table D2 summarizes the total number of plant species that were found in the Harding Park SWM pond. Table D3 summarizes the number of meadow marsh (mm) and aquatic (a) plants that were identified. Of these plants identified, Table D4 summarizes the total number of native and non-native plant species found.

There were significant changes in the plant community from 1996 to 1997. In 1996 there was often no dominant plant species and the shoreline still had large patches of bare, unvegetated ground. By the end of the 1997 growing season, these barren areas were well-vegetated and the diversity of plants had increased significantly. The total number of plants found below the "top of active storage" line since the 1996 meadow

Table D2: Harding Park Pond - Total Number of Plant Species Found

Location	1996			1997			
	June 22	Sept. 17	Total # of species found	June 26	Aug. 5	Sept. 24	Total # of species found
Sediment Forebay	11	12	19	28	28	24	43
Main Pond	7	19	23	25	28	23	47
Wet Meadow	15	18	25	24	29	31	52

Table D3: Harding Park Pond - Total Number of Aquatic (a) and Meadow Marsh (mm) Plant Species Found at the End of Two Growing Seasons (* if the plant is considered both "mm" and "a" it will be counted as "a" for this Table)

Location	Total # of mm & a Wetland Habitat Species		Total # of Planted mm & a Wetland Habitat Species		Total # of Colonized mm & a Wetland Habitat Species	
	mm	a	mm	a	mm	a
Sediment Forebay	30	6	7	0	23	6
Main Pond	32	9	5	0	27	9
Wet Meadow	36	9	6	0	30	9

mm = meadow marsh species a = aquatic species

Table D4: Harding Park Pond - Native vs. Non-native Plant Species Found

Location	1996			1997			Original Planting	
	native	non-native	unknown	native	non-native	unknown	native	non-native
Sediment Forebay	14	5	0	30	12	1	4	2
Main Pond	19	4	0	27	16	4	4	0
Wet Meadow	14	9	2	33	19	0	4	2

marsh planting has increased from 11 plants species to 43 species in the sediment forebay. In the main pond, 47 plant species became established, and in the wet meadow 52 plant species became established. A significant increase in diversity has been observed. To the best of our knowledge, all of these new plant species have naturally colonized the site.

Dominance is a function of season and competition. For example in the Harding Park SWM pond, the rush species (*Juncus spp.*) tended to dominate in the early part of the season (June) and were succeeded by water plantain *(Allisma plantago-aquatica)* in August. This change in dominance as the season progresses is something that needs to be considered in SWM pond designs. If good vegetative cover is required throughout the growing season, plants that mature at different times of the season may be required to meet this objective.

It is also important to examine the dominant plant species within the community composition. Often the plant species introduced are not the dominant species found after one or two growing seasons. By examining systems, such as the Harding Park SWM pond, that are naturally colonizing, we can get a better idea of which plants will dominate the community structure. In September 1996, the first growing season, the dominant species in the sediment forebay was pale smartweed (*Polygonum lapathifolium*) a plant often found in disturbed meadow marsh type habitats. This is consistent with the disturbance the area received due to construction. As the area began to recover from this disturbance, the community structure changed toward a more stable aquatic/ meadow marsh habitat. By August 1997, the dominant species found were purple loosestrife *(Lythrum salicaria)*, broad-leaf cattail (*Typha latifolia*), water plantain (*Allisma plantago-aquatica*), and softstem bulrush (*Scirpus validus*). These plants are all aquatic to meadow marsh in habitat. This trend toward an aquatic to meadow marsh dominated system was prevalent in all three areas of the Harding Park SWM pond.

There are plant species present that are not recommended for planting, as they are considered invasive, difficult to remove once established, and provide poor habitat. Within the Harding Park facility these species include common reed, reed canary grass, and purple loosestrife. These plants will likely colonize a site naturally and have a strong tendency to result in a monoculture of one or two species.

The number of both native and non-native plants increased between 1996 and 1997. However, there was no significant change in the proportion of native to non-native plants in each area. The ideal would be to have a facility that has only native plant species. The reality is that non-native plants are common in urban areas and without intensive management are impossible to remove entirely from the facility. Permitting and wildlife agencies recommend that planting plans include only native plant material, in an effort to reduce the number of non-natives introduced to the facility. The problem with non-natives is that they can out-compete and displace native species. Their seeds may be transported to other, more natural areas of the watershed.

4.2 Rouge/401 MTO Stormwater Management Pond

After three growing seasons, all the plant species introduced are still present in the facility (see Table D5). Based on the planting plan, all the areas planted, except one, have thrived and expanded. A grouping of 44 softstem bulrush was planted adjacent to the submerged weir in the main pond. These plants have survived along the pond edges but not out into the pond. This is probably due to the currents that flow through this area during a storm event.

In the two growing seasons since the facility was planted, 76 aquatic and meadow marsh plant species have naturally colonized the main pond of the facility (Table D6). In the same time period, 50 aquatic and meadow marsh plant species have naturally colonized the sediment forebay. This is not unexpected for this pond as it is located within the Rouge River Valley, adjacent to high quality habitat. The high quality of this adjacent habitat is also evident in the high number of native plant species that have colonized in comparison to non-native species (Table D7). These natural colonizations are probably a result of wind, water and animal transportation. When we visited this site, we often observed deer tracks, and saw leopard frogs, dragonflies, and several species of birds using the site.

Table D5: Fate of aquatic and meadow marsh plants planted at the Rouge/401 SWM pond

COMMON NAME	SCIENTIFIC NAME	Status	Originally Planted	Present in 1995	Present in 1996	Present in 1997
Common arrowhead	Sagittaria latifolia	n	x	x	x	x
Softstem bulrush	Scirpus validus	n	x	x	x	x
Fragrant waterlily *(horticultural variety)*	Nymphaea odorata	+	x	x	x	x
Curled pondweed	Potamogeton crispus	+	x	x	x	x
Ribbon reed canary grass	Phalaris arundinacea var. picta	+	x		x	x

Status: n = native species + = introduced species

Table D6: Rouge/401 MTO SWM Pond - Total Number of Plant Species Found

Date:	Sediment Forebay	Main Pond
Original Planting	1	5
Aug. 21/95	0	8
Sept. 15/95	0	13
Oct. 12/95	0	11
TOTAL 1995	0	13
June 22/96	9	9
Aug. 1/96	8	25
Sept. 17/96	12	37
TOTAL 1996	16	45
June 26/97	28	45
Aug. 5/97	33	48
Sept. 24/97	34	60
TOTAL 1997	51	81

Table D7: Rouge/401 MTO SWM Pond - Native vs. Non-native Plant Species Found

Location	Originally Planted		1995		1996			1997		
	native	non-native	native	non-native	native	non-native	unknown	native	non-native	unknown
Sediment Forebay	0	1	0	0	12	3	1	32	17	2
Main Pond	2	2	10	3	35	8	2	53	28	0

The long term results of this initial planting will not be known for several years. Within the emergent aquatic and meadow marsh area the dominant plants in the sediment forebay were spikerush (*Eleocharis)* throughout the growing season with water plantain (*Allisma plantago-aquatica*) becoming dominant in late summer to early fall. In the main pond the dominant emergent/meadow marsh plants were cattail (*Typha*), water plantain (*Allisma plantago-aquatica*), spikerush (*Eleocharis*), and jointed rush (*Juncus articulatus*). Of these dominant plants, none of them were introduced through the planting plan. Within the submergent plant community, the dominant species in both the sediment forebay and the main pond was Canada waterweed (*Elodea*). This native submergent is a well-known food source for ducks. Curled pondweed (*Potomogeton crispus*) is still significant within the submergent community, however, it is no longer the dominant plant species.

It was observed that a non-native horticultural variety of the Fragrant Water Lily (*Nymphaea odorata [hort.]*) was substituted for the native Fragrant Water Lily (*Nymphaea odorata*) specified in the planting plan. Similarly, a non-native horticultural variety of Ribbon Reed Canary Grass (*Phalaris arundinacea var. picta*) was substituted for the native Reed Canary Grass (*Phalaris arundinacea*) specified in the planting plan. This was probably because the supplying plant nursery did not have the correct plants available in stock. In order to prevent these problems, project managers should specify in their Planting Specifications and their Plant Order that "NO SUBSTITUTIONS" are to be allowed. A botanist or horticulturalist should be on-site to receive and confirm the plant material.

The evolution of the non-native plant species introduced through the planting plan is interesting. While they are all still present within the system, only the waterlily has remained a dominant species, with some spreading. This observation is not unexpected, as there are no other competing species. The water lily situation requires careful monitoring, as there is some risk that it may escape into the Rouge Valley system. Ribbon Reed Canary Grass *(Phalaris arundinacea variety picta)*, the native variety of which is known for its aggressiveness, has remained with only limited spreading along the north shore of the SWM pond. Its relative dominance has decreased significantly as natives naturally colonized the site. Curly pondweed (*Potomogeton crispus*) is a common non-native submergent aquatic plant. Its dominance within the submerged community appears to be lessening with the natural establishment of other pondweeds.

The community developing at the Rouge/401 SWM pond ranges from meadow marsh to aquatic. As the development of the vegetation community at this site proceeds it is expected that the diversity of plant species will be reduced as the early successional colonizing plant species are out-competed and the meadow marsh to aquatic species continue to spread and dominate.

The main pond consistently shows more plant diversity and more plant colonization than the sediment forebay.

5.0 OBSERVATIONS AND RECOMMENDATIONS

In summary, several trends have been observed in the evolution of vegetation communities at the two SWM facilities. These trends may have implications for the design of planting plans in future facilities.

1. Substantial natural colonization appears to occur at the sites, even after only one growing season. If this trend is common to other SWM facilities, there may be justification for a reduction in the number of plant species identified in the initial planting plan.

2. Vegetation communities at these sites have tended to evolve toward a common group of dominant species. While considerably more work will need to be completed to confirm this, the results from these two ponds would suggest that cattail (*Typha*), spikerush (*Eleocharis*), rush (*Juncus*), bulrush (*Scirpus*), water plantain (*Allisma*) and waterweed (*Elodea*) may be effective species for inclusion in planting plans.

3. At both sites there was evidence that alternative, often non-native species were substituted by the plant supplier for the species prescribed in the planting plan. This finding underscores the need for instructions stating that no substitutions will be accepted and closer inspection of plant material delivered should be made by the landscape supervisor.

As it is premature to draw any conclusions after monitoring only two ponds for two growing seasons, the following are recommendations made by the authors for further work:

1. The monitoring of both the Rouge/401 SWM pond and the Harding Park SWM pond can be discontinued in years 3 and 4, except for a single inspection of the status of invasive (e.g., purple loosestrife) and potentially invasive (e.g., horticultural variety of water lilies) plants. If these species appear to be expanding, recommendations should be made to the pond operators for implementing control measures. In year 5, a complete assessment of the vegetation community should be undertaken, following the methodology described in this report.

2. The results of this study should be compared to the results of inventories at other SWM ponds within the GTA that have an established vegetation community before any general conclusions can be made about the appropriateness of planting species in SWM pond systems. Each SWM pond is unique (i.e., differing catchments, differing chemical issues, etc.). Recommendations about the types of wetland plants and planting techniques suitable in general for all SWM pond facilities should not be based on the results of two SWM ponds.

3. Dominance will change as the growing season progresses. Dominance is often a function of "time of year" rather than "number of plants present". To determine the dominant species of a system, the site should be allowed to evolve for at least five growing seasons, and the plants should be identified at least three times over a growing season.

4. A similar study should be undertaken to address the terrestrial planting portion of these facilities. This type of study would probably best be undertaken by a municipality in partnership with SWAMP.

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ANNEX D1: List of Plant Species

Definitions of Symbols used in Tables 1 and 2

n	the species is considered native to Ontario's Central Region.
+	the species is introduced or escaped from cultivation in Central Region
(+)	the species may be considered native in some regions but is introduced to Central Region.
?	the status of the species was unknown.

d	*disturbed*	a recently altered natural state (e.g.: due to construction)
m	*meadow*	closed graminoid and herb vegetation behind areas of shoreline emergent vegetation and on wet floodplains adjacent to open water systems. Usually seasonally flooded or subject to storm floods.
mm	*meadow marsh*	having a canopy of 75% to 100% with standing water and/or muck/mud flats beneath canopy or between clumps; characterized by more or less continuous stands of dominant graminoids of medium to low stature with surface water; water depth up to 1 m (flooded), but usually shallower, or exposed mud, during much of the summer.
upland	*upland*	well-drained hilltops, steep to moderate slopes, sand flats, etc. Stands normally dominated by dryland species of trees, shrubs, and/or herbaceous ground vegetation.
sm	*shallow marsh*	having a canopy of 75% to 100% with standing water and/or muck/mud flats beneath canopy or between clumps. Characterized by more or less continuous stands of tall emergent aquatics with surface water up to 1 m (flooded), but usually less during much of the summer months.
e	*emergent*	emergent aquatic vegetation in or adjacent to open shallow water, pools or channels; commonly interspersed or dominated by clumps of vegetation (rooted, unconsolidated, or floating) with open water channels between or with open water beneath the canopy of sedges, grasses, reeds, cattails; cover by emergents or shrubs greater than 25%.
a	*aquatic*	adapted to living partially or wholly submerged in water or in waterlogged soils.
r	*riparian*	growing adjacent to a river or stream including shores and floodplains.
m	*mesic*	characterized by moderately moist conditions; neither too moist nor too dry
sub	*submergent*	growing below the water surface
f	*floating*	the majority of the plant grows on the water's surface

Sources: Newmaster et al., 1997; OMNR, 1993; Steen, 1971 (see references on Page D-14)

Table 1 Plant species found at Rouge/401 Stormwater Management Pond

LOCATION	STATUS	HABITAT	COMMON NAME	SCIENTIFIC NAME	Original Planting	Date Aug. 21/95	Sept. 15/95	Oct. 12/95	June 22/96	Aug. 1/96	Sept. 17/96	June 26/97	Aug. 5/97	Sept. 24/97
Sediment Forebay	n	a	Narrow Leaf Cattail	Typha angustifolia							x	x	x	x
	n	a	Broad Leaf Cattail	Typha latifolia										x
	n	mm	Cattail	Typha x Glauca								x	x	x
	n	a - mm	Black Bulrush	Scirpus atrovirens					x	x	x	x	x	x
	n	mm	Soft Stem Bulrush	Scirpus validus					x	x	x	x	x	x
	?	mm	Barberpole Sedge	Scirpus marcocarpa (rubrotinctus)									x	
	r	a	Floating Arrowhead	Sagittaria cuneata									x	
	+	am-mm	Purple Loosestrife	Lythrum salicaria								x	x	x
	n	mm - mesic	Common reed	Phragmites australis										x
	+	mm - mesic	Reed canary grass	Phalaris arundinacea var. picta	x				x	x	x	x	x	x
	n	a - mm	Water plantain	Allisma plantago-aquatica							x*	x	x*	x (eaten)
	n	mm	Bebb's sedge	Carex bebbii/cristatella								x	x	x
	n	mm	Fox sedge	Carex vulpinoidea								x	x	x
	n	mm	Sedge	Carex hystricina								x		
	n	mm	Rush	Juncus sp.					x		x			
	n	mm	Jointed Rush	Juncus articulatus								x	x	x
	n	mm	Rush	Juncus compressus									x	
	n	a - mm	Dudley's rush	Juncus dudeyi								x		x
	n	mm	Tory's rush	Juncus toreyi										x
	n	a - mm	Rush	Juncus bufonius					x	x	x			
	n	mm	Spikerush	Eleocharis sp.						x	x*	x*	x*	x
	n	mm	Spikerush	Eleocharis erythropoda										x
	n	a	Canada waterweed	Elodea canandensis						x	x	x	x*	x
	+	a	Curly leaved pondweed	Potomogeton crispus						x	x		x	x
	n	a	Sago pondweed	Potomogeton pectinatus						x	x	x	x	x
	n	a	Chara	Chara sp.							x		x*	x
	+	d - m	Curled dock	Rumex crispus								x	x	
	+	mm	Hairy willowherb	Epilobium hirsutum										x
	n	mm	Devil's Beggarticks	Bidens frondosa					x					x
	n	m	Canada goldenrod	Solidago canadensis									x	
	n	mm	Rough bed-straw	Galium asprellum					x					
	+	mm - m	Rayless Aster	Aster brachyactis										x
	n	mm - m	Tall white Aster	Aster lanceolatis								x		
	n	upland	Bird's foot trefoil	Hosackia americana								x	x	x
	n	mm	Pale smartweed	Polygonum lapathifolium										x
	+	m	Chickory	Cichorium intybus										x
	n	mm	Cursed crowfoot	Ranunculus sceleratus								x	x	
	n	mm	Blue vervain	Verbena hastata									x	
	+ salty	m - mm	Alkali grass	Puccinellia distens								x	x	
	+	r - mm	Crack willow	Salix fragilis					x					
	n	r mm	Heart-leaved willow	Salix eriocephala								x		
	n	r - mm	Peach-leaved willow	Salix amygdaloides									x	x
	n	not a	Manitoba maple	Acer negundo										x
	n	mm	Canada bluejoint	Calamagrostis canadensis									x	x
	?	mm		Agrostis sp.					x			x		
	+	mm	Redtop	Agrostis gigantea									x	x
	(+)	mm	Creeping bentgrass	Agrostis stolonifera									x	x
	n	m	Fescue	Fescue sp.								x		
	+	mesic	Perennial rye-grass	Lolium perenne								x	x	x
	+ salty	m	Foxtail barley	Hordeum jubatum								x	x	
	+	mm - mesic	Barnyard grass	Echinochloa crusgalli									x	x
	+	m	Meadow fescue	Festuca pratensis								x		
	?	m	Tall fescue	Festuca gigantea									x	
	(+)	upland	Canada blue grass	Poa compressa								x	x	
	+	mesic	Timothy grass	Phleum pratense								x		
	+		Quack grass	Elymus (Agropogon) repens										x
Main Pond	n	a	Broad Leaf cattail	Typha latifolia		x	x	x	x	x	x	x	x*	x*
	n	a	Narrow Leaf Cattail	Typha angustifolia						x	x	x		x
	n	a	Cattail	Typha x Glauca					x			x	x	x*
	n	mm	Black Bulrush	Scirpus atrovirens						x	x	x	x	
	n	a - mm	Soft Stem Bulrush	Scirpus validus	x	x	x	x		x	x	x	x	x
	n	a - sm	Pickerelweed	Pontederia cordata		x	x			x			x	
	n	a - sm	Arrowhead	Sagittaria latifolia	x	x	x	x		x	x	x	x	x

Table 1. Plant species found at Rouge/401 Stormwater Management Pond

LOCATION	STATUS	HABITAT	COMMON NAME	SCIENTIFIC NAME	Original Planting	Date Aug. 21/95	Sept. 15/95	Oct. 12/95	June 22/96	Aug. 1/96	Sept. 17/96	June 26/97	Aug. 5/97	Sept. 24/97
	+	sm - mm	Purple Loosestrife	Lythrum salicaria		x	x	x	x	x	x	x	x	x
	n	mm - mesic	Common reed	Phragmites australis										x
	+	mm - mesic	Reed canary grass	Phalaris arundinacea var. picta	x					x	x	x	x	x
	n	a - mm	Water plantain	Allisma plantago-aquatica								x*	x*	x*
	n	mm	Bebb's sedge	Carex bebbii/cristatella								x	x	x
	+	m - mm	sedge	Carex spicata								x		
	n	mm	Fox sedge	Carex vulpinoidea								x	x	x
	n	mm	Sedge	Carex hystricina							x		x	
	?	mm	Rush	Juncus sp.										
	n	mm	Jointed Rush	Juncus articulatus						x		x*	x	x
	+	mm	Rush	Juncus compressus						x				
	n	a - mm	Dudley's rush	Juncus dudleyi						x		x	x	x
	n	mm	Rush	Juncus nodosus						x				
	n	mm	Rush	Juncus toreyi									x	x
	n	a - mm	Rush	Juncus tenuis						x				
	n	a - mm	Rush	Juncus bufonius						x	x		x	
	n	a - mm	Rush	Juncus effusus								x		
	n	mm	Spikerush	Eleocharis sp.			x			x	x	x*		
	n	mm	Spikerush	Eleocharis erythropoda									x	x
	+	a	Fragrant water lily	Nymphaea odorata (hort.)	x	x	x	x	x	x	x	x	x	x
	n	r - mm	American water-horehoun	Lycopus americanus										x
	+	r - mm	European water horehoun	Lycopus europaeus						x	x	x	x	x
	n	a	Canada waterweed	Elodea canandensis		x	x	x		x	x	x	x*	x
	n	a	Eel grass	Valisneria americana		x	x	x			x			
	R	a	Horned pondweed	Zannichellia palustris						x	x			
	+	a	Curly leaved pondweed	Potomogeton crispus	x		x	x		x	x	x		
	R	a	Pondweed	Potomogeton nodosus			x	x						
	n	a	Sago pondweed	Potomogeton pectinatus			x	x			x	x	x	
	n	a	Chara	Chara sp.			x	x		x	x	x	x	x
	+	d - m	Curled dock	Rumex crispus								x	x	x
	n	mm	Field horsetail	Equisetum arvense								x		
	n	sm - mm	Blue Flag	Iris versicolor										x
	n	mm	Boneset	Eupatorium perfoliatum									x	x
	n	mm	Swamp milkweed	Asclepias incarnata									x	x
	n	mm - mesic	Lady's thumb	Persicaria persicaria								x	x	x
	+	mm	Hairy willowherb	Epilobium hirsutum						x	x			x
	n	mm	Sticky willowherb	Epilobium ciliatum							x		x	x
	n	mm	Mad-dog Skullcap	Scuterllaria lateriflora										x
	+	m	Chickory	Cichorium intybus										x
	n	mm	Devil's beggarticks	Bidens frondosa							x	x	x	x
	n	m - mm	Tall goldenrod	Solidago altissima									x	
	n	m	Canada goldenrod	Solidago canadensis							x			
	n	m	Late goldenrod	Solidago gigantea										x
	n	m - mm	Narrow-leaved goldenrod	Solidago graminifolia					x	x		x		
	?	mm	Bedstraw	Galium sp.									x	
	n	mm	Bedstraw	Galium palustre										x
	n	mm	Rough bed-straw	Galium asprellum							x	x	x	
	+	m - mm	Sweet Coltsfoot	Tussilago farfara										x
	n	upland	Bird's foot trefoil	Hosackia americana							x	x	x	x
	n	m	Rugel's Plantain	Plantago rugelii										
	+	m	English Plantain	Plantago lanceolata								x	x	
	n	d - m	Common Ragweed	Ambrosia artemisifolia									x	x
	n	m	New England Aster	Aster novae-angliae							x		x	x
	n salty	mm salty	Rayless Aster	Aster sp.							x		x	x
	n	m - mesic	Tall white Aster	Aster lanceolatis							x	x	x	x
	n	mm	Nodding bur-marigold	Bidens cernua										
	n	m	Lily of the Valley	Convallaria magalis					x		x			
	n	m	Oxeye daisy	Helopsis helianthoides					x	x	x	x		
	+	mesic	White clover	Trifolium repens					x		x	x	x	
	+	m	Red clover	Trifolium pratense									x	
	n	mm	Cursed crowfoot	Ranunculus sceleratus							x	x	x	x
	+	m	Common sunflower	Helianthus annus							x			
	n	mm	Blue vervain	Verbena hastata							x		x	x

Table 1. Plant species found at Rouge/401 Stormwater Management Pond

LOCATION	STATUS	HABITAT	COMMON NAME	SCIENTIFIC NAME	Original Planting	Date									
						Aug. 21/95	Sept. 15/95	Oct. 12/95	June 22/96	Aug. 1/96	Sept. 17/96	June 26/97	Aug. 5/97	Sept. 24/9
	n	d- mm	Pale smartweed	Polygonum lapathifolim										x
	n	d - m	Witch grass	Panicum capullare										x
	n	mm	Rice cut-grass	Homalocenchrus oryzoides							x			x
	+ salty	m - mm	Alkali grass	Puccinellia distens								x		x
	n	d - m	Bull thistle	Circium vulgare							x			
	n	d - m	Canada thistle	Circium arvense										
	+	r mm	White willow	Salix alba										x
	+	r - mm	Crack willow	Salix fragilis							x			
	n	mm	Narrow Heart-leaved willo	Salix eriocephala							x	x		x
	n	r - mm	Peach-leaved willow	Salix amygdaloides										x
	+	m	European Pussy willow	Salix caprea								x		x
	n	r - mm	Sandbar willow	Salix exigua										x
	+	m	Bay leaved willow	Salix pentandra								x		x
	+	m	Quack grass	Agropyron repens									x	x
	?	mm		Agrostis sp.					x	x				
	+	mm	Redtop	Agrostis gigantea									x	x
	+	mm	Creeping bentgrass	Agrostis stolonifera								x	x	x
	(+)	mesic	Perennial rye-grass	Lolium perenne								x	x	
	+ salty	m	Foxtail barley	Hordeum jubatum									x	x
	+	mm - mesic	Barnyard grass	Echinochloa crusgalli							x		x	x
	+	mm - mesic	Barnyard grass	Echinochloa microstachya										x
	+	mesic	Meadow fescue	Festuca pratensis								x		x
	+	mesic	Fescue	Festuca trachyphylla									x	
	(+)	upland	Canada blue grass	Poa compressa								x		
	+	mesic	Timothy grass	Phleum pratense										
	n	m	Red Ash	Fraxinus pennsylvanica var. pennsylvanica	x									
	n	m	Green ash (seedling)	Fraxinus pennsylvanica var. subintegerrim	x							x		x
	n	not a	Manitoba maple (seedling)	Acer negundo								x	x	x
	(+)	m	Eastern Cottonwood	Populus deltoides								x		x

Table 2. Plant species identified at Harding Park Stormwater Management Pond

LOCATION	STATUS	HABITAT	COMMON NAME	SCIENTIFIC NAME	Original Planting	Dates June 22/96	Sept. 17/96	June 26/97	Aug. 5/97	Sept. 24/97
Wet Meadow										
	n	a	Soft Stem Bulrush	Scirpus validus		x	x	x	x*	x
	n	mm	Barberpole Sedge	S. rubrotinctus (macrocarpus)				x	x	x
	n	mm	Black Bulrush	S. atrovirens			x		x	x
	n	a	Broad leaf cattail	Typha latiflia			x	x*	x*	x*
	n	a	Narrow-leaved Cattail	Typha angustifolia						x
	+	mm	Purple Loosestrife	Lythrum salicaria		x	x	x	x*	x
	n	mm	Turtlehead	Chelone glabra		x	x			x
	n	mm	Spotted Joe Pye-weed	Eupatorium maculatum		x	x	x	x	x
	n	mm - mesic	Reed Canary Grass	Phylaris arundinacea		x		x		
	+	mm - mesic	Barnyard grass	Echinochloa crusgalli			x			
	n	mm	Spotted Jewelweed	Impatiens campensis			x		x	x
	n	d - mm	Pale smartweed	Polygonum lapathiifoium		x	x			
	+	mm	Charlock mustard	Brassica kabor		x		x		
	(+)	upland	Canada bluegrass	Poa compressa		x				
	n	mm	Fowl Meadow grass	Poa palustris				x		
	+	mesic	Timothy grass	Phleum pratense				x		
	+	mesic	Perennial Rye-grass	Lolium parenne				x		
	n	mm	Field Horsetail	Equiseturn arvense				x		x
	n		Red Baneberry	Actaea rubra		x				
	+	mesic	White clover	Trifolium repens			x			
	?	mesic	Fescue	Festuca sp.		x				
	?	mm	False dragonhead	Physostegia virginiana			x		x	x
	+	mm	Helen's Flower	Helenium autumnale			x			x
	n	mm - mesic	Tall white Aster	Aster simplex			x	x		x
	n	m	New England aster	Aster novae-angliae			x			x
	+	m	Ox-eye daisy	Chrysanthemum leucanthemum				x		
	n	m - mm	Tall goldenrod	Solidago altissima						x
	n	m - mm	Narrow-leaf goldenrod	Solidago graminifolia		x	x			
	n	m	Canada goldenrod	Solidago canadensis		x			x	
	+	m	White campion	Lychnis alba				x		
	+	m	Indian mustard	Brssica juncea		x				
	+	d m	Canada Thistle	Circium arvense		x				x
	+	d m	Bull Thistle	Circium vulgare						x
	n	mm	Cursed crowfoot	Ranunculus scelera tus				x	x	
	+	upland	Queen Anne's Lace	Daucus carota					x	
	+	upland	Bird's Foot trefoil	Hosackia americana				x	x	
	+	m - mm	Sweet Coltsfoot	Tussilago farfara					x	
	n	mm	Boneset	Eupatorium perfoliatum					x	
	n	mm	Sedge	Carex sp.						x
	n	mm	Fox Sedge	Carex vulpinoidea					x	x
	n	mm	Bebb's sedge	Carex bebbeyi/crystatella				x		
	?	mm	Rush	Juncus sp.		x	x			
	n	a - mm	Rush	Juncus tenuis			x			
	n	a - mm	Rush	Juncus bufonius			x			
	n	a - mm	Didley's sedge	Juncus dudlyii					x	
	n	mm	Jointed sedge	Juncus articulatus				x	x	

Table 2. Plant species identified at Harding Park Stormwater Management Pond

LOCATION	STATUS	HABITAT	COMMON NAME	SCIENTIFIC NAME	Original Planting	Dates June 22/96	Sept. 17/96	June 26/97	Aug. 5/97	Sept. 24/97
Wet Meadow										
	n	mm	Torrey's sedge	Juncus torreyi					x	x
	+	mm	Rush	Juncus compressus				x		
	n	a - mm	Common Rush	Juncus effusis				x		x
	n	sm - mm	Tall manna grass	Glyceria grandis				x*	x*	x
	+	mm	Redtop	Agrostis gigantia					x	x
	(+)	mm	Creeping bent grass	Agrostis stolonifera					x	
	n	mm	Spikerush	Eleocharis erythropoda					x	x
	n	mm	Spikerush	Eleocharis sp.				x		
	+	mm	Rice cut grass	Homalocenchus oryzoides					x	
	n	a - mm	Water-plantain	Alisma plantago-aquatica				x	x	
	n	mm	Nodding bur-marigold	Bidens cernua					x	x
	n	mm	Devil's beggartricks	Bidens frondosa					x	x
	+	mm	Great hairy willowherb	Epilobum hirsutum					x	x
	n	mm - r	Red osier dogwood	Cornus stolonifera						x
	n	r - mm	Peach-leaved willow	Salix amygdaloides				x	x	x
	+	r - mm	Crack willow	Salix fragilis					x	x
	n	not a	Manitoba maple	Acer negundo						x
Main and wet Pond	n	mm	Spotted Joe pye-weed	Eupatorium maculatum		x				
	n	mm	Jewelweed	Impatiens campensis		x	x			
	+	mm	Purple Loosestrife	Lythrum salicaria			x	x	x	x
	n	mm - mesic	Common reed	Pragmites autralis					x	x
	n	mm - mesic	Reed Canary Grass	Phylaris arundinacea		x	x			x
	+	r - mm	European water horehound	Lycopus europaeus		x	x			
	n	mm	Common mint	Mentha arvensis						x
	n	mm	Common horsetail	Equisetum arvense		x				
	n	mm	Boneset	Eupatorium perfoliatum		x				
	+	m	chickory	Chicorium intybus					x	
	n	mm	Cursed crowfoot	Ranunculus sceleratus			x	x		
	+	upland	Bird's foot trefoil	Hosackia americana				x	x	
	n	m	New England Aster	Aster novae-angliae				x		
	n	mm - m	Tall white Aster	Aster lanceolatus (simplex)						x
	+	m	Prickly lettuce	Lactuca virosa						x
	(+)	mm	False dragonhead	Physostegia virginiana			x			
	+	mm	Broad-leaved Plantain	Plantago major				x	x	x
	n	mm	Nodding bur-marigold	Bidens cernua			x		x	x
	n	mm	Devil's beggarticks	Bidens frondosa					x	
	n	mm	Bur-marigold	bidens sp.				x		
	n	mm	Pale smartweed	Polygonum lapathifolium			x		x	x
	?	mm	Smartweed	Polygonum sp.				x		
	n	m	Tall goldenrod	Solidago altissima						x
	n	m	Grass-leaved goldenrod	Solidago graminifolia		x				
	n	m	Canada goldenrod	Solidago canadensis				x		
	+	m	Creeping thistle	Circium arvense				x	x	
	+	m	Oakleafed goosefoot	Chenopodium glaucum				x		
	+	m - mm	Sweet Coltsfoot	Tussilago farfara				x		x
	+	m - mm	Common charlock	Simapis sp.				x		

Table 2. Plant species identified at Harding Park Stormwater Management Pond

LOCATION	STATUS	HABITAT	COMMON NAME	SCIENTIFIC NAME	Original Planting	Dates June 22/96	Sept. 17/96	June 26/97	Aug. 5/97	Sept. 24/97
Wet Meadow										
	n	d - m	Common ragweed	Ambrosia elatior					x	x
	?	d - m	Cross-straight knotweed	Polygonum aviculare				x		
	+	mesic	Perennial Rye-grass	Lolium perenne				x	x	
	+	m	Common flax	Linum usitatissimum			x			
	+	d - m	Curly dock	Rumex crispus					x	x
	n	d - m	American water horehound	Lycopus americanus			x			
	?	mm - mesic	Barnyard grass	Echinochloa sp.					x	x
	n	mm - mesic	Barnyard grass	Echinochloa crusgalli			x			
	n	d -mm	Witch grass	Panicum capillare						x
	+	m	Red clover	Trifolium pratense						x
	n	mm	Blue vervain	Verbena hastata					x	x
	+	d - m	Sunflower	Helianthus annuus					x	
	+	upland	Queen Anne's Lace	Daucus carota					x	
	n	mm	Spikerush	Eleocharis sp.			x			
	n	a - mm	Broad leaf cattail	Typha latifolia			x*	x		x
	n	a - mm	Narrow leaf cattail	Typha angustifolia				x	x	
	n	a - mm	Water plantain	Allisma plantago-aquatica			x	x	x*	x*
	n	mm	Fox sedge	Carex volupinoides				x	x	x
	n	mm	Crested sedge	Carex cristatella					x	
	n	a	Soft Stem Bulrush	Scripus validus				x	x	x
	n	mm	Black Bulrush	Scripus atrovirens				x	x	
	n	a - mm	Rush	Juncus tenuous			x			
	n	mm	Toad Rush	Juncus bufonius			x			
	n	a - mm	Dudley's rush	Juncus dudleyi				x*	x	
	n	a	Sago pondweed	Potamogeton pectinatus			x	x		
	n	a	Stonewort	Chara sp.			x	x*	x	
	(+)	r	Cottonwood seedling	Populus deltoides				x	x	
	n	r - mm	Peached-leaved willow	Salix amygdaloides			x		x	
	n	r -mm	Heart-leaved willow	Salix eriosephella					x	x
	+	r - mm	Crack willow	Salix fragilis					x	x
	?	r - mm	Willow seedling	Salix sp.				x		
Sediment Forebay	n	mm	Turtlehead	Chelone glabra					x	
	n	mm	Cursed crowfoot	Ranunculus sceleratus		x				
	n	mm	Water hemlock	Circuta maculata				x		x
	n	r - mm	Water horehound	Lycopus unifloris				x	x	
	n	mm	Spotted Joe pye-weed	Eupatorium maculatum		x	x		x	x
	n	mm	Blue vervain	Verbena Hastata				x	x	
	n	mm - mesic	Common reed	Phragmites autralis		x	x	x	x	x
	n	mm - mesic	Reed canary grass	Phylaris arundinacea				x	x	
	+	mm	Purple Loosestrife	Lythrum Salicaria					x*	x
	n	m	Tall goldenrod	Solidago altissima			x			x
	n	m	Canada goldenrod	Solidago graminifolia		x				
	n	m - mm	Narrow-leaf goldenrod	Euthiamia graminifolia				x	x	x
	n	mm	Spotted Jewelweed	Impatiens campensis		x	x		x	x
	n	d - m	Common Ragweed	Ambrosia artemisiifolia		x		x		x
	n	d - m	Daisy Fleabane	Erigeron annus				x		

Table 2. Plant species identified at Harding Park Stormwater Management Pond

LOCATION	STATUS	HABITAT	COMMON NAME	SCIENTIFIC NAME	Original Planting	Dates June 22/96	Sept. 17/96	June 26/97	Aug. 5/97	Sept. 24/97
Wet Meadow										
	+		Common wintercress	Barbarea vulgaris						
	+	m	Prickly lettus	Lactuca virsa				x	x	
	n	mm	Common mint	Mentha rotundifolia				x	x	
	+	upland	Bird's foot trefoil	Hosackia americana				x	x	
	?	m	Forget-me-not	Myosotis sp.				x		
	n	mm - mesic	Tall white aster	Aster lanceolatus			x	x		x
	+	mm	Harry willow herb	Epilobium hirsutum			x		x	
	+	mm	Indian mustard	Brassica Juncea		x				
	+	m	Pennycress	thlapsi arvense		x				
	+	m	Chickory	Chicorium intybus			x*		x	
	+	mm	Sneezeweed	Helinium autumnale					x	x
	(+)	mm	False dragonhead	Physostegia virginiana					x	x
	n	mm	Pale smartweed	Polygonum lapathifolium			x*		x	x
	+	d - m	Canada thistle	circium arvense						x
	n	mm	Boneset	Eupatorium perfoliatum		x	x			
	+	m - mm	Barnyard grass	Echinochloa crusgalli					x	x
	n	m - mm	Barnyard grass	Echinochloa microstachya						x
	+	mm	Rice cut grass	Homalocenchus oryzoides				x		
	n	sm - mm	Tall Manna Grass	Glyceris grandis				x	x	x
	+	d - m	Curly dock	Rumex crispus			x		x	x
	n	mm	Spikerush	eleocharis erythropoda				x	x	
	n	a	Broad leaved cattail	Typha latifolia					x*	x
	n	a	Cattail	Typha x glauca				x		
	n	a	Soft Stem Bulrush	Scripus validus		x		x	x*	x
	n	mm	Black Bulrush	Scripus atrovirens				x (?)	x	x
	n	mm	Woolgrass	Scripus cyperinus			x	x		
	n	mm	Barberpole sedge	S. microcarpus (rubrotinctus)						
	n	a - mm	Rush	Juncus tenuous				x*		
	n	a - mm	Dudley's rush	Juncus dudleyi				x	x	x
	n	mm	Fox sedge	Carex vulpinoidea				x	x	x
	n	mm	sedge	Carex stipata				x		
	n	mm	sedge	Carex granularis				x		
	n	a - mm	water plantain	Allisma plantago-aquatica			x	x	x*	x*
	+	d - m	common plantain	Plantago major				x		x
	n	not a	manitoba maple seedling	Acer negundo		x				
	n	r -mm	Peach leaf willow	Salix amygdaloides				x	x	

APPENDIX E

Assessment of Phytoplankton and Periphyton Communities

The following report was produced by Daniel D. Olding, Consulting Biologist, for the Toronto and Region Conservation Authority (TRCA). It includes studies of a highway stormwater retention pond adjacent to the Rouge River in Toronto and the Harding Park retrofit pond in Richmond Hill. This document has been reformatted but is otherwise reproduced as submitted by TRCA.

Stormwater Management Ponds:

Assessment of Phytoplankton and Periphyton Communities (1997)

Final Report

Harding Pond
Rouge Pond

January 13, 1998 **Report # 9801.1**

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1.0 INTRODUCTION

This report describes the investigation into algal dynamics in two stormwater management ponds as part of the Stormwater Assessment Monitoring & Performance (SWAMP) Program.[1] The report describes baseline conditions of the summer phytoplankton and periphyton, compares within and between pond differences, and reports on chemical and physical field monitoring relevant to the algal dynamics. The two ponds included the following:

a) Harding Pond - Harding Park Pond in Richmond Hill, Ontario.
b) Rouge Pond - MTO Pond at Highway 401 and the Rouge River, Scarborough, Ontario.

2.0 METHODS

2.1 Phytoplankton and Chemical Analysis

Sites were sampled every two and one half weeks from late June to early September. Water samples for phytoplankton analysis were taken from the deepest point of each basin in the study locations. Samples were taken by Kemmerer bottle at one metre intervals to twice the secchi depth, and were field composited and preserved with Lugol's Iodine Solution. Field parameters were measured including temperature-depth profiles, conductivity, pH, and water transparency (secchi depth). In mid August, dissolved oxygen profiles were taken and samples were collected in the same manner as for phytoplankton and field preserved for total nitrogen and total phosphorus.

Phytoplankton samples for each site were composited from the five samples over the summer period. Samples were prepared by Utermohl sedimentation and identified to species level, where possible, at 625X magnification under phase contrast on an inverted microscope. Diatom species identifications were confirmed from peroxide cleaned mounts using DIC microscopy at 1250X magnification. Identifications were based on Prescott (1962), Taft and Taft (1990), Kramer and Lange-Bertalot (1986, 1988, 1991a, 1991b), Anton and Duthie (1981), Komarkova-Legnerova (1969) and Starmach (1985). Algal biovolumes were determined through measurements of individual algal species and calculations based upon geometric shapes (MOEE 1992). Estimated chlorophyll a was calculated by the following conversions:

1. Wet weight (mg/l) = Biovolume (mm^3/l)* 1.1
2. Dry weight (mg/l) = Wet weight * 0.2
3. Chlorophyll a (mg/l) = Dry weight (mg/l) * 0.01

Final Calculation:Chlorophyll a (mg/l) = Biovolume (mm^3/l)*0.0022

[1] See Appendix B for a glossary of terms used in this report.

The phytoplankton counting procedure was developed to ensure consistent enumeration of rare species. Each sample was appropriately diluted or concentrated so that complete transects were viewed until a minimum of 500 individuals of all taxa were counted. The number of fields of view was recorded. A second count on the same sample was performed covering the same number of fields of view, but only recording those taxa which occurred at less than four in 500 (0.8%) individuals. The results of the two counts were combined, and only those phytoplankton taxa which were recorded at a percentage greater than 0.4% (four in 1000) in the combined count were identified and recorded in the taxa lists.

2.2 Periphyton

Periphyton were sampled through the use of artificial substrate tiles (10 x 10 cm unglazed ceramic), acid washed in dilute HCl prior to installation. Tiles were placed at the time of first sampling in late June. Two tiles were placed in the littoral zone of each sampling site at depths varying from 20 to 50 cm, and fixed in a vertical position with aluminum pegs so as to prevent accumulation of sedimented particles. Tile locations (i.e. rock or sand) were chosen to reflect the different types of bottom substrate available in each sampling site.

Harvesting of the tiles was performed in early September at the time of the final sampling. Attached algae were harvested by removing the tiles from the sampling location, with care not to disturb any loosely attached algae, and placing them in a sampling bin. The macroalgae was first removed into the bin with a wide blade scraper, and the scraper was rinsed into the bin after use with distilled water. The tile was then scrubbed vigorously with a fine plastic brush and rinsed into the bin a minimum of three times, or until no additional algae could be seen to be removed. The samples were made up to 250 ml with distilled water, transferred to sampling bottles and field preserved with Lugol's Iodine Solution.

Periphyton samples were prepared and enumerated in the same manner as phytoplankton samples, except that diatoms were identified to species level (where possible) and other groups were identified to genus level. Higher taxonomic resolution for the diatoms was required to take advantage of extensive monitoring data based on species level identifications (i.e. Hofmann 1996, Lowe 1974).

3.0 RESULTS

3.1 Harding Pond

3.1.1 Site Description

Harding Pond consisted of four distinct areas:

- Sediment Forebay
- Wet Pond
- Wetland Pool

- Wet Meadow

Only two of these areas, Sediment Forebay and Wet Pond, contained sufficient water for a phytoplankton survey. The other two sites were shallow and heavily vegetated. Phytoplankton and chemical sampling locations were established at the deepest points of the Sediment Forebay (HP1), and the Wet Pond (HP2), and a third location was added in the southern bay of the Wet Pond (HP3). Periphyton sampling tiles were established in the littoral zone of each of the three locations. In HP1, Tile 1 was placed at 25 cm depth in a sandy substrate, and Tile 2 was placed at 50 cm depth in proximity to rocks. In HP2, Tile 1 was placed at 40 cm depth among rocks, and Tile 2 was placed at 50cm depth in a sandy substrate. In HP3, Tile 1 was placed at 25 cm depth in proximity to rocks, and Tile 2 was placed at 25 cm depth in a sandy substrate. The purpose of the third location (HP3) was to evaluate whether consistent differences in phytoplankton/ periphyton and chemical composition were observed between the deepest part of the main basin and its associated bays.

3.1.2 Phytoplankton Survey

HP1 contained 13 taxa (Table E1) and was dominated (i.e. groups comprising 10% or more of the total) in numerical abundance by green algae and euglenoids (Table E2). Key taxa (>10% by numerical abundance) were *Spermatozoopsis* sp. and *Euglena* sp. When biovolume corrections were added to compensate for the relative size of different phytoplankton, the only dominant group (greater than 10% by biovolume) was the euglenoids, made up entirely of *Euglena* sp., with an abundance of 88.8% (Table E3). The total biovolume was 41.1 mm^3/l approximately equivalent to 90.4 μg/l chlorophyll a. 87.5% of the phytoplankton biovolume had a GALD (greatest axial linear dimension) of less than 35 microns. This class of phytoplankton is generally considered to represent those phytoplankton easily susceptible to grazing (Watson and McCauley 1988).

HP2 contained 19 taxa (Table E1). Four groups were numerically dominant; green algae, followed by cryptophytes, diatoms and euglenoids (Table E2). *Pyramimonas* sp. and *Euglena* sp., were the primary taxa. The majority of the biovolume was split between the euglenoids (59.3%) and the cryptophytes (19.3%), represented primarily by *Euglena* sp. and *Cryptomonas erosa* (Table E3). The total biovolume was 9.5 mm^3/l (20.9 ug/l chlorophyll a). The biovolume was approximately equal between the size classes with 55% being grazable (GALD < 35 μm), and 45% ungrazable (GALD >35 μm).

Nineteen phytoplankton taxa were recorded in HP3 (Table E1), with the same dominant groups as HP2 (Table E2). The dominant taxa numerically were *Euglena* sp., *Fragilaria nanana*, and *Stephanodiscus* sp. Biovolume dominants were the same as HP2 with *Euglena* sp. being the dominant taxa, comprising 71.7% of the biovolume (Table E3). The total phytoplankton biovolume was 8.0 mm^3/l (17.6 μg/l chlorophyll a) with 59.5% of the biovolume having a GALD of less than 35 μm.

The three Harding Pond sites showed a similarity in being dominated by one taxa of phytoplankton, *Euglena* sp. However, there were distinct differences in phytoplankton assemblages between the Sediment Forebay

(HP1) and the Wet Pond (HP2 and HP3). HP1 while having a much higher total phytoplankton biovolume, was comprised of fewer taxa than HP2 and HP3. The rank-abundance distribution of HP1 (Figure E1) showed a phytoplankton community which was unbalanced in favour of one taxa, whereas HP2 and HP3 showed a more even distribution. Seven of thirteen taxa present in HP1 were not found in either HP2 or HP3. In contrast, the two locations in the Wet Pond were similar with respect to total biovolume, number of taxa recorded, and size distribution. However, there were some differences in phytoplankton assemblages between HP2 and HP3, with only thirteen of 26 taxa being common between the two sites, and slight differences in the evenness of rank-abundance distributions.

3.1.3 Periphyton

Both artificial substrate tiles were recovered from HP1, and microscopic analysis recorded nine taxa (Table E4). The diatoms dominated numerically (Table E5), with two small pennate diatoms, *Achnanthes minutissima /Cymbella microencephala,* comprising over 80% of the individuals counted. Despite the presence of relatively few green algae numerically (<5%), the one taxa recorded (*Oedegonium* sp.) was a large filamentous green macroalgae which dominated the biovolume, accounting for 58.5% of the total (Table E6). Most of the rest of the biovolume was made up of diatoms (32.7%). The total biovolume expressed by surface area was 179 $mm^3/100\ cm^2$.

HP2 and HP3 were similar in composition, with eight and ten taxa recorded respectively (Table E4). Only one substrate tile was recovered from each location (Tile 2 from HP2, Tile 1 from HP1). Both sites were dominated numerically by diatoms, blue-green algae and green algae (Table E5). Taxa comprising greater than 10% numerically included *Achnanthes minutissima /Cymbella microencephala* (HP2, HP3), *Leptolyngbya* sp. (HP2, HP3), *Protococcus viride* (HP2) and *Cocconeis placentula* (HP3). The total biovolume was primarily made up of green algae and diatoms in each of the sites (Table E6), with green algae being more abundant in HP2 and diatoms more abundant in HP3. Dominant species included *Achnanthes minutissima /Cymbella microencephala* in both sites, *Protococcus viride* in HP2, and *Cocconeis placentula* and *Oedegonium* sp. in HP3. Total areal biovolumes were similar at 1.8 $mm^3/100\ cm^2$ (HP2) and 0.8 $mm^3/100\ cm^2$ (HP3).

The Sediment Forebay of Harding Pond (HP1) showed a distinct floristic assemblage compared to the Wet Pond (HP2 and HP3) with only two taxa from the Sediment Forebay being found in the Wet Pond. Additionally, the areal periphyton biovolume of HP1 was approximately two magnitudes higher than HP2 or HP3. Rank-abundance distributions of all three sites were similar in length and evenness (Figure E2).

3.1.4 Chemical and Physical Characteristics

The three Harding Pond locations differed in key chemical and physical characteristics over the summer sampling period (Tables E7, E12, E13 and E14). HP1 was characterized by higher total phosphorus, total nitrogen and conductivity, lower pH, and cooler surface water temperatures. Mean summer transparencies were similar between the three sites, although HP1 showed greater variation throughout the summer.

Nitrogen:phosphorus levels were similar between HP1 and HP2 at 11.9:1 and lower at HP3 with a ratio of 7.8:1. Littoral zone macrophyte vegetation was sparse in HP1.

HP2 was the deepest site and appeared to be chemically stratified (meromictic) with a dense layer of cooler saline water underlying lighter warmer water. The chemical stratification weakened gradually throughout the summer, but was still present on September 5. At midsummer, the bottom waters (2 metres) were anoxic. Littoral zone macrophyte vegetation was dense throughout HP2 except near the rocky flow-through structure from HP1.

HP3, being a shallower site in the same basin as HP2, was outside the area of meromixis. Heavy growths of macrophytes were present at the phytoplankton sampling site and in the littoral zone.

Based on trophic state indicators such as secchi depth, total phosphorus, total nitrogen, and estimated chlorophyll a, HP1 can be classed as hypereutrophic, and HP2 and HP3 as eutrophic bordering on hypereutrophic.

3.1.5 Comparison with Previous Studies

Two previous studies were performed on the algal communities of Harding Pond. The first survey by Melkic (1996), undertaken in mid-September, identified only the dominant macroalgae, i.e. those large filamentous algae which were visually seen to colonize substrates around the ponds perimeter. Comparison of results from the sediment forebay and the wet pond with the current study reveal one similarity. In both cases, *Oscillatoria* (= *Phormidium*) was recorded in the sediment forebay. Additional similarities to the present study might have been realized, had the tile placed near the flowthrough from the sediment forebay to the wet pond been able to be recovered, as this was a site specifically sampled in Melkic's study. However, the tile could not be located due to extensive growths of vegetation/algae. A more formal comparison between the two studies, identifying possible trends, would be difficult since sampling techniques were very different between the two studies.

The second study (Olding 1997) covered the early summer phytoplankton using an identical sampling methodology as the current study. The phytoplankton analysis was less stringent, with only 200 individuals per sample being counted. The phytoplankton species composition and richness of the early summer phytoplankton was not seen to differ much from the summer composite conditions (i.e. not many new taxa were found, and not many lost). Both studies recorded 32 taxa in all three locations of the pond. However, the relative proportion of phytoplankton groups did change, with green algae and euglenoids increasing in the summer composite, and diatoms decreasing, a typical summer transition. Additionally, total biovolume showed some location specific changes, with HP1 increasing greatly (1.7 to 41.1 mm^3/l), HP2 decreasing (18.2 to 9.5 mm^3/l) and HP3 staying constant.

Errata: The species identified in Olding (1997) as *Spondylomorum quaternium* (HP1) has since been confirmed at *Spermatozoopsis* sp., and the species *Achnanthes minutissima* also includes individuals of *Cymbella microencephala.*

3.2 Rouge Pond

3.2.1 Site Description

Rouge Pond consisted of two distinct areas:

- Sediment Forebay
- Quiescent Treatment Zone

Both of these areas were suitable for phytoplankton study, and sampling sites were established in the deepest points of each of the basins. The sampling sites were identified as RP1 (Sediment Forebay) and RP2 (Quiescent Treatment Zone). Periphyton sampling tiles were established in the littoral zone of each of the two locations. In RP1, both tiles were placed in sandy substrates with Tile 1 at 25 cm depth and Tile 2 at 50 cm depth. In RP2, Tile 1 was placed at 20cm depth in a sandy substrate and Tile 2 was placed at 25 cm depth among rocks.

3.2.2 Phytoplankton

RP1 contained 21 taxa (Table E8) which were numerically dominated by diatoms, green algae and cryptophytes (Table E2). Key taxa were *Achnanthes minutissima/Cymbella microencephala, Cryptomonas phaseolus,* and *Carteria* sp. Four groups were dominant by biovolume, euglenoids, green algae, diatoms and cryptophytes, with key taxa *Euglena* sp., *Carteria* sp. and *Cryptomonas erosa* (Table E3). The total phytoplankton biovolume was 0.7 mm^3/l (1.5 ug/l chlorophyll a), and 57.2% of the biovolume had a GALD less than 35 μm.

Twenty-four taxa were recorded at RP2 (Table E8), numerically dominated by green algae, chrysophytes and cryptophytes (Table E2). Key taxa were *Dinobryon divergens* and chlorophyte cells. The dominant groups by biovolume were euglenoids, dinoflagellates, cryptophytes and green algae, with *Euglena* sp., *Peridinium* sp. and *Cryptomonas erosa* being the dominant taxa (Table E3). The total phytoplankton biovolume was 3.4 mm^3/l (7.5 ug/l chlorophyll a) with 54.3% having a GALD less than 35 μm.

RP1 and RP2 showed similarities in species richness, size distribution, and rank-abundance distribution. However, RP2 had a higher total phytoplankton biovolume and only seventeen of thirty-two taxa were common between the two sites. Additionally, diatoms were much more prevalent both numerically and by biovolume in RP1, whereas chrysophytes numerically and dinoflagellate biovolume were more abundant in RP2.

3.2.3 Attached Algae

Two tiles were recovered from RP1, and only four taxa were identified at levels greater than 0.4% (Table E9). The diatoms dominated numerically (Table E5) with *Achnanthes minutissima /Cymbella microencephala* comprising over 90% of the individuals found. The biovolume (Table E6) was dominated primarily by the diatoms *Achnanthes minutissima /Cymbella microencephala,* and *Gomphonema parvulum/angustatum,* and secondarily by a large green filamentous macroalgae (*Spirogyra* sp.), which despite being found at numerical abundance less than 0.4%, accounted for 23.8% of the biovolume. The total areal biovolume was 54.0 $mm^3/100\ cm^2$.

Two tiles were recovered from RP2, but only Tile 1 was analyzed. Tile 2 was located in rock and had extensive periphyton growth so far in excess of the other tiles that comparison would have be difficult. This tile may be analyzed separately for comparison of periphyton growths between rock and sand substrates. Eighteen taxa were recorded from Tile 1 of RP2 (Table E9), dominated numerically by the diatoms *Achnanthes minutissima /Cymbella microencephala* (Table E5). Total biovolume (Table E6) was dominated by large filamentous green (*Mougeotia* sp.) and blue-green (*Oscillatoria* sp.) algae. The total areal biovolume was 470 $mm^3/100\ cm^2$.

The sediment forebay (RP1) and quiescent treatment zone (RP2) showed dramatic differences in periphyton assemblages, especially with regards to species richness (4 vs. 18) and areal biovolume (54.0 vs. 470 $mm^3/100\ cm^2$). Rank-abundance distributions reflect the differences in community structure, with low length and evenness in RP1 compared to RP2 (Figure E2).

3.2.4 Chemical and Physical Characteristics

The two Rouge Pond sites differed in many chemical and physical properties (Tables E7, E10 and E11). RP1 had a higher surface conductivity and nitrogen:phosphorus ratio, slightly lower pH and lower total nitrogen and total phosphorus levels. In addition, the surface waters were several degrees cooler than RP2, averaging 16.5°C throughout the summer. The transparency in RP1 extended right to the bottom sediments (1.7 m) all summer, in contrast with the mean transparency of RP2 (1.3 m). RP2 had a maximum depth of 4.0 m and was strongly chemically stratified with the chemocline between two and three metres. The chemical stratification persisted strongly throughout the summer, although some erosion of the saline layer was evident, especially at 3 meters (Table E11). At mid summer the bottom waters of RP2 below 3 meters were anoxic. Littoral zone macrophyte vegetation was sparse in RP1 and moderate to dense in RP2.

Based on trophic state indicators such as secchi depth, total phosphorus, total nitrogen, and estimated chlorophyll a, RP1 can be classed as oligotrophic, and RP2 as eutrophic.

3.2.5 Comparison with Previous Studies

Two previous studies were performed on the algal communities of Rouge Pond. The first survey by Melkic (1996), undertaken in mid-September, identified only the dominant macroalgae, i.e. those large filamentous algae which were visually seen to colonize substrates around the ponds perimeter. Comparison of results from the sediment forebay and the wet pond with the current study show two similarities, i.e. in both studies, *Spirogyra* was recorded as the only filamentous macroalgae in the sediment forebay, and *Oedegonium* was present in the quiescent treatment zone. The analysis of Tile 2 from the quiescent treatment zone might reveal additional similarities as this was an area extensively sampled in Melkic's study. A more formal comparison between the two studies, identifying possible trends, would be difficult since sampling techniques were very different between the two studies.

The second study (Olding 1997) covered the early summer phytoplankton using an identical sampling methodology as the current study. The phytoplankton analysis was less stringent, with only 200 individuals per sample being counted. The phytoplankton species composition and richness of the summer composite phytoplankton community was seen to differ considerably from that in the early summer. The same taxa were recorded in both studies, but the summer composite contained many new species (i.e. 30 compared to 16 in the early summer). In general, from early summer to summer composite conditions, the proportion of green algae and dinoflagellates decreased in both locations, and proportion of diatoms (RP1), chrysophytes (RP2) and cryptophytes (RP1 and RP2) increased. The biovolume of RP1 remained constant between the two studies, and the summer composite biovolume of RP2 was approximately double that of the early summer.

Errata. The species identified in Olding (1997) as *Achnanthes minutissima* also includes individuals of *Cymbella microencephala.*

4.0 DISCUSSION

Harding Pond and Rouge Pond are both shallow stratifying stormwater management ponds with high overall productivity. However, considerable differences in physical and chemical parameters between the two ponds are observed. The majority of these differences can be explained with reference to the water quality of the incoming stormwater quality, i.e. Rouge Pond treats stormwater received primarily from a major highway, with high levels of heavy metals, petroleum hydrocarbons and road salt (Marsalek et. al. 1997), while Harding Pond treats stormwater received from an urban subdivision, with elevated levels of nutrients. The algae of the two sites, both phytoplankton and periphyton, reflect the quality of the incoming water. The algae of Rouge Pond, while having some ubiquitous "tolerant" taxa , i.e. *Cryptomonas erosa*, *Achnanthes minutissima* (Lowe 1974), *Cymbella microencephala* (Lowe 1974) and some representatives indicative of nutrient rich conditions, i.e. *Gonium sociale* (Prescott 1962), *Nitzschia acicularis* (Lowe 1974), and *Euglena* sp., showed an exceptional number of salt tolerant marine or brackish water diatoms, i.e. *Caloneis amphibaena, Entomoneis alata, Navicula pygmae, Fragilaria fasciculata, Diatoma tenuis* (Germain 1981, Lowe 1974). Harding Pond was similar to the Rouge Pond in having "tolerant" taxa, but had relatively more nutrient rich taxa, and no marine or brackish water diatoms. The presence of taxa such as *Cocconeis placentula* (Germain 1981), and *Protococcus viride* (Prescott 1962) in the Wet Pond (HP2 and HP3) reflected the extensive littoral zone macrophyte vegetation which characterized these locations.

The algal communities also provide insight into the performance of the ponds from a biological perspective. It appears as though the incoming stormwater has a strong impact on the algal composition of the Sediment Forebay of both Harding and Rouge Ponds, resulting in disturbed algal communities. The effects of the disturbance is not identical between the two ponds, but instead reflects the qualities of the incoming stormwater. For example, in the Sediment Forebay of Rouge Pond (RP1), the periphyton is characterized by being extremely species poor, with only four taxa recorded, and this is strong evidence that some environmental factor(s) related to the incoming stormwater is/are having a strong negative impact on the periphyton community. The impact is likely related to the sediments since the phytoplankton community does not seem to be affected in the same way. While further investigation would be necessary to conclusively isolate the causative agents, evidence from another study links exclusive domination of the two main periphyton taxa in RP1, (*Achnanthes minutissima* /*Cymbella microencephala* and *Gomphonema parvulum/angustatum*) with high levels of heavy metals (Whitton 1984). As heavy metals are often present in roadway runoff, a similar explanation could be hypothesized in this case. The effects of disturbance are also seen in the Sediment Forebay of Harding Pond. In this case, the phytoplankton community is impacted, being relatively species poor, and dominated by a large bloom of *Euglena* sp. (greater than 88% by biovolume). The cause of the disturbance is almost certainly related to the input of excessive nutrient concentrations (i.e. total nitrogen and total phosphorus in hypereutrophic range) which is typical in urban stormwater runoff.

As the inputs of stormwater move through the various compartments of the stormwater treatment ponds, evidence is seen of change towards a healthier and more diverse algal community. In the Rouge Pond, this effect is quite dramatic, with the species richness of the periphyton increasing from four in the sediment forebay (RP1) to eighteen in the quiescent treatment zone (HP2). Similarly, in Harding Pond, the species richness of the phytoplankton increased from thirteen in the sediment forebay (HP1) to nineteen in the Wet Pond (HP2 and HP3), and the bloom of *Euglena* sp. was significantly reduced in magnitude. These biological changes appear to be well correlated with pollutant reductions documented in the main body of this report. Further, the role of the ponds in protecting the biological communities of the receiving waters of Rouge River (Rouge Pond), and German Mills Creek (Harding Pond) can begin to be clearly seen. The use of biological community monitoring such as phytoplankton and periphyton, provides an essential link between designated substance reductions (i.e. heavy metals, nutrients) and the objectives for which these guidelines were established, the protection of biological habitat and communities.

Several other factors relevant to the biological functioning of the two ponds should be mentioned. First, the pathway through the Rouge Pond and Harding Pond systems can be seen to be fundamentally different. In the Sediment Forebay of the Rouge Pond, nutrients are low, the phytoplankton community biovolume is sparse, and periphyton communities are impaired. As we move into the Quiescent Treatment Zone, nutrients increase, the phytoplankton and periphyton community biovolume increases, and the disturbing effects on the periphyton are reversed. In contrast, in the Sediment Forebay of Harding Pond, nutrients are extremely high, and phytoplankton and periphyton community biovolume is high. As we move through the system into the Wet Pond, nutrients decrease, as do phytoplankton and periphyton biovolumes.

Second, the effects of aquatic vegetation can have a modifying effect on the periphyton communities. This can be seen most strongly in the Wet Pond sites of Harding Pond (HP2 and HP3), where the periphyton biovolumes were much lower than expected. At both sites the littoral macrophyte community was able to almost completely outcompete the periphyton community throughout the summer. The comparison of periphyton communities across the sites in Harding and Rouge Ponds needs to take into consideration the extent of macrophyte vegetation at the sampling sites. The present study fairly accurately reflects the actual differences in littoral zone periphyton, related to both biotic and abiotic factors. In order to separate the biotic differences from the abiotic, the sampling tiles would need to be placed out of the zone of vegetation influence, perhaps by suspending them just below the surface in the middle of the pond. In this way, the periphyton community would more accurately represent the water quality of the system, free from the confounding effects of vegetation. However, this change in sampling location also has drawbacks in that the sampling tiles would be removed from the influence of the sediments, an important factor affecting some periphyton communities. Ultimately, periphyton sampling locations need to be chosen to reflect the design objectives of the study, thereby providing information appropriate to the questions being posed.

Thirdly, there is a conspicuous absence of blue-green algae in all locations of the two stormwater ponds. This absence is somewhat unexpected, given the usual association of blue-green algae with nutrient rich conditions. Studies have suggested that the dominance of blue-green algae may be related to additional factors such as low nitrogen:phosphorus ratios, low (or moderate) turbulence levels, high water temperatures, low light levels, high pH and/or low carbon dioxide levels, or high zooplankton grazing levels (Shapiro 1990). Interestingly, most of these factors are consistent with conditions in Harding Pond, and to a lesser extent, Rouge Pond. The absence of blue-green algal dominance represents a beneficial state for the two stormwater ponds, and further study may reveal what critical factor(s) are behind this state.

Finally, while there are many attributes of the algal community composition which are documented in this study (i.e. absence of blue-green dominance), most will only be able to be interpreted in comparison with other sites. The comparison of the two ponds in this study could only be performed with reference to established ecological interpretations of a few indicator taxa. Similarly, environmental variables could only be related to algal composition in an ad hoc manner due to the small sample size. Future studies will incorporate Harding Pond and Rouge Pond into a larger set of sites and allow for a more formal establishment of relationships between environmental parameters and the entire algal community species composition. Additionally, the utility and effectiveness of other phytoplankton community measures (GALD, algal taxonomic or functional groups, etc.) will be able to be evaluated, perhaps leading to a greater understanding of the dynamics within these ponds.

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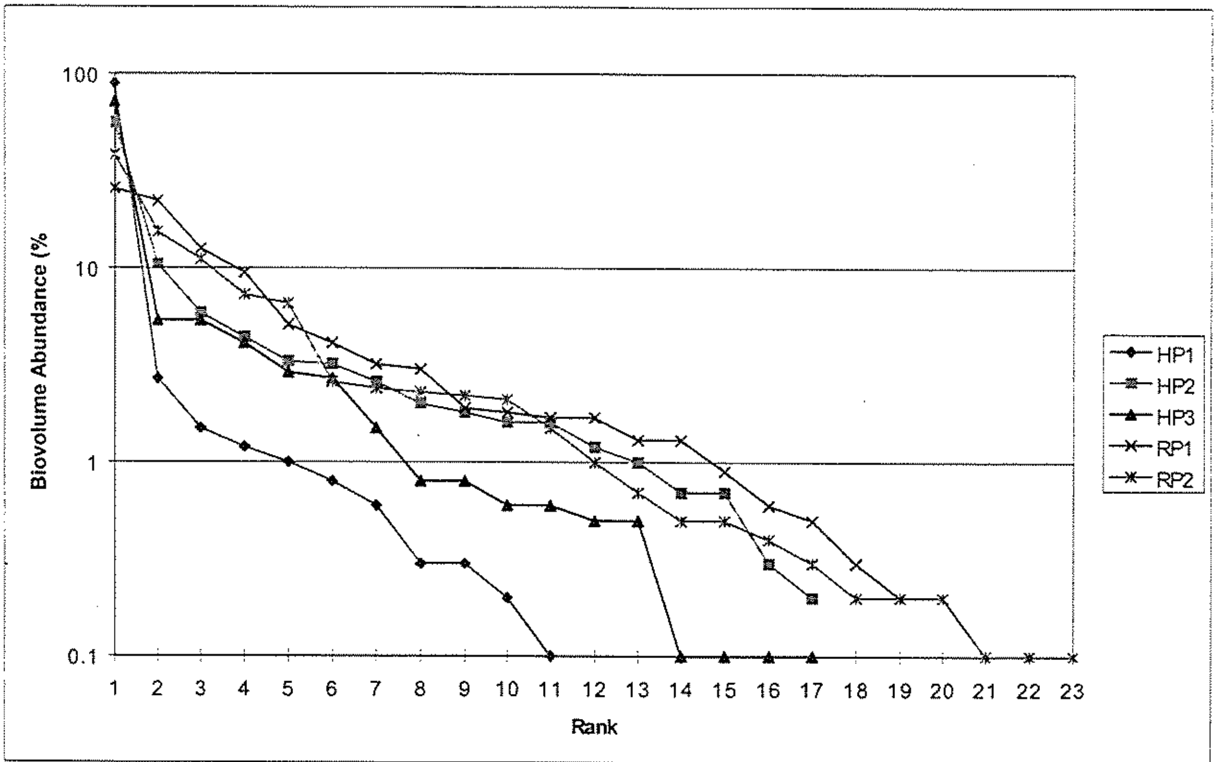

Figure E1: Rank-Abundance for Phytoplankton of Harding Pond and Rouge Pond.

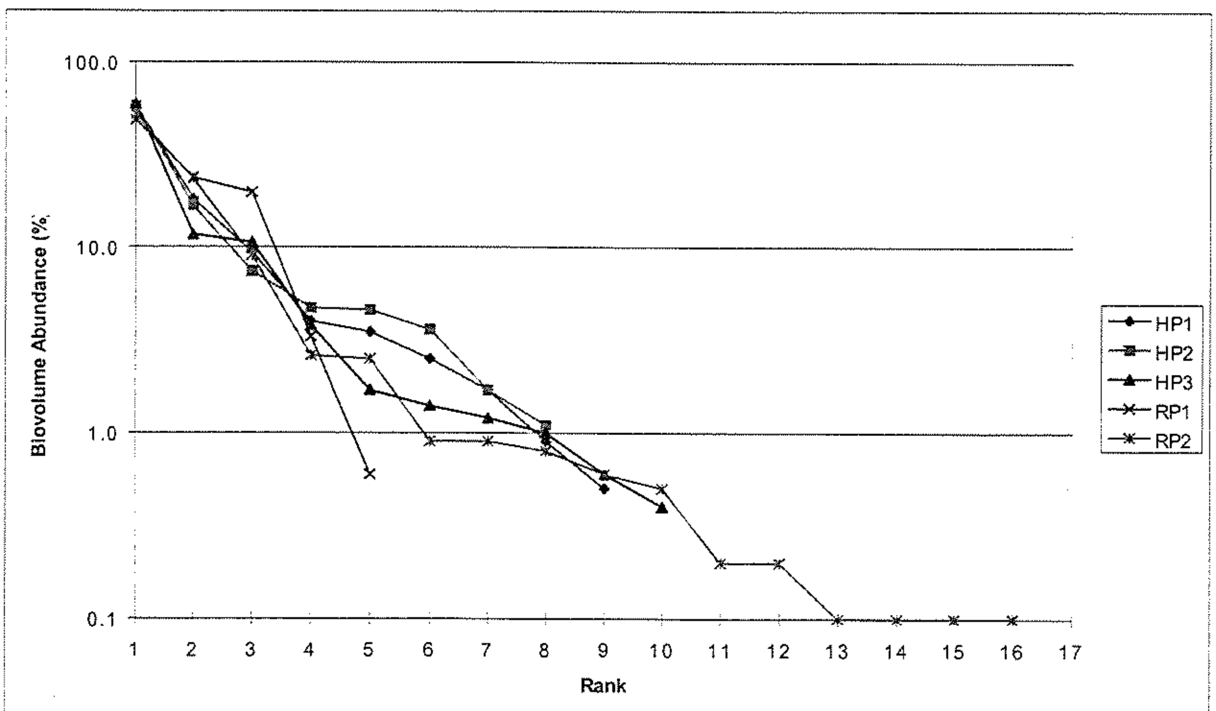

Figure E2: Rank-Abundance for Periphyton of Harding Pond and Rouge Pond.

Table E1: Phytoplankton taxa recorded from Harding Pond in numerical abundance and biovolume. Group symbols are: bg=blue-green algae, ch=chrysophyte, cr=cryptophyte, d=diatom, df=dinoflagellate, e=euglenoids, g=green algae, x=xanthophyceae.

			% by Number			% by Biovolume		
	Taxa	**Group**	**HP1**	**HP2**	**HP3**	**HP1**	**HP2**	**HP3**
1.	Achnanthes minutissima	d		0.9	3.0		0.2	0.5
	Cymbella microencephala							
2.	Carteria sp.	g		0.9	0.4		1.0	0.6
3.	Chlamydomonas sp.	cr		1.9	3.8		0.7	2.7
4.	Chlorophyte cells	g	0.5	1.8	6.8	0.3	0.3	1.5
5.	Chroococcus dispersus	bg			1.1			<0.1
6.	Cryptomonas erosa	cr	0.9	5.3	2.3	2.7	10.5	2.9
7.	Cryptomonas erosa v. reflexa	cr	0.8	2.5	7.9	1.2	2.6	5.4
	Cryptomonas pyrenoidifera							
8.	Cryptomonas ovata	cr		1.9	2.8		4.4	4.1
9.	Cryptomonas phaseolus	cr		4.9			1.8	
10.	Dinobryon divergens	ch	0.7			0.3		
11.	Euglena sp.	e	13.3	10.5	21.8	88.8	55.7	71.7
12.	Fragilaria nanana	d		7.7	19.8		3.2	5.4
13.	Golenkinia radiata	g	5.4			0.8		
14.	Mallomonas tonsurata	ch		0.6			0.7	
15.	Monoraphidium braunii	g	0.4			<0.1		
16.	Monoraphidium contortum	g		0.9			<0.1	
17.	Monoraphidium setiforme	g	1.2			0.1		
18.	Nitzschia acicularis	d			0.6			0.1
19.	Nitzschia sp.	d	1.7			1.0		
20.	Peridinium sp.	df		0.7			3.3	
21.	Phormidium sp.	bg	0.9			0.6		
22.	Pyramimonas sp.	g		35.5	4.9		5.9	0.5
23.	Rhodomonas minuta	cr	1.0	9.1	8.9	0.2	1.2	0.8
24.	Scenedesmus longus	g			0.7			0.1
25.	Selenastrum minutum	g	0.5			<0.1		
26.	Spermatozoopsis sp	g	70.3	0.4		1.5	<0.1	
27.	Sphaerellopsis sp.	g			0.5			0.1
28.	Stephanodiscus sp.	d		9.5	10.7		1.6	0.8
29.	Trachelomonas sp.	e		0.4			1.6	
30.	Trachelomonas volvocina	e		0.9	0.5		2.0	0.6
31.	Unidentified volvocale sp.1	g			0.5			<0.1
32.	Unidentified volvocale sp.2	g			0.4			0.1

Table E2: Numerical abundance (%) of phytoplankton groups and species richness across stormwater pond locations.

Algal Group	**HP1**	**HP2**	**HP3**	**RP1**	**RP2**
Green Algae	78.3	41.4	18.0	26.9	39.4
Euglenoids	13.3	11.8	22.3	4.6	3.2
Dinoflagellates		0.7		0.6	5.4
Diatoms	1.7	18.1	34.1	33.9	6.2
Cryptophytes	2.7	23.7	21.9	26.3	16.2
Chrysophytes	0.7	0.6		4.4	24.3
Blue-green Algae	0.9		1.1	0.6	
Xanthophyceae				0.5	3.3
Species Richness	13	19	19	21	24

Table E3: Biovolume abundance (%) of phytoplankton groups and total biovolume (mm^3/l), chlorophyll a (ug/l) and % edible algae across stormwater pond locations.

Algal Group	**HP1**	**HP2**	**HP3**	**RP1**	**RP2**
Green Algae	2.7	7.9	5.6	25.7	13.6
Euglenoids	88.8	59.3	72.3	27.0	38.4
Dinoflagellates		3.3		1.9	17.6
Diatoms	1.0	5.0	6.7	23.0	3.7
Cryptophytes	4.1	19.3	13.2	18.7	16.0
Chrysophytes	0.3	0.7		0.9	6.6
Blue-green Algae	0.6		<0.1	0.3	
Xanthophyceae				<0.1	0.3
Total Biovolume	41.1	9.5	8.0	0.7	3.4
Chlorophyll a*	90.4	20.9	17.6	1.5	7.5
Edible Algae**	87.4	54.8	59.3	57.2	54.2

* calculated from biovolumes

** proportion of phytoplankton biovolume with GALD (greatest axial linear dimension) less than 35 microns.

Table E4: Periphyton taxa recorded from Harding Pond in numerical abundance and biovolume. Group symbols are: bg=blue-green algae, cr=cryptophyte, d=diatom, e=euglenoids, g=green algae.

			% by Number			% by Biovolume		
	Taxa	Group	HP1	HP2	HP3	HP1	HP2	HP3
1.	Achnanthes minutissima Cymbella microencephala	d	83.3	27.4	17.3	18.2	16.7	10.6
2.	Characium sp.	g		1.2			3.6	
3.	Chlamydomonas sp.	g			1.7			0.6
4.	Chlorophyte cells	g		2.1	3.7		1.7	1.4
5.	Chlorophyte colony	g		1.0	0.8		1.1	0.4
6.	Cocconeis placentula	d		3.9	31.2		7.4	59.5
7.	Cryptomonas sp.	cr	0.6			0.5		
8.	Euglena sp.	e	0.8			4.0		
9.	Filamentous blue-green sp.	bg			2.7			1.0
10.	Gomphonema parvulum Gomphonema angustatum	d	1.1	1.2		2.5	4.6	
11.	Gomphonema truncatum	d	2.9			9.4		
12.	Leptolyngbya sp.	bg		44.1	35.5		4.7	3.8
13.	Navicula sp.	d	1.1			1.7		
14.	Nitzschia sp.	d	1.4			0.9		
15.	Oedegonium sp.	g	4.5		0.5	58.5		11.7
16.	Phormidium sp.	bg	3.3			3.5		
17.	Protococcus viride	g		17.9	2.3		57.9	1.7
18.	Scenedesmus sp.	g			1.9			1.2

Table E5: Numerical abundance (%) of periphyton groups and species richness across stormwater pond locations.

Algal Group	**HP1**	**HP2**	**HP3**	**RP1**	**RP2**
Green Algae	4.5	22.2	10.9		7.5
Diatoms	89.8	32.5	48.5	98.3	83.1
Blue-green Algae	3.3	44.1	38.2		7.9
Euglenoids	0.8				
Cryptophytes	0.6				
Species Richness	9	8	10	4	18

Table E6: Biovolume abundance (%) of periphyton groups and total biovolume ($mm^3/100cm^2$) across stormwater pond locations.

Algal Group	**HP1**	**HP2**	**HP3**	**RP1**	**RP2**
Green Algae	58.5	64.3	17.0	23.8*	61.2
Diatoms	32.7	28.7	70.1	72.4	9.2
Blue-green Algae	3.5	4.7	4.8		23.5
Euglenoids	4.0				
Cryptophytes	0.5				
Total Biovolume	179	1.8	0.8	54.0	470

* found at less than 0.4% numerical abundance

Table E7: Summer average chemical and physical characteristics across stormwater pond locations.

Parameter	**HP1**	**HP2**	**HP3**	**RP1**	**RP2**
Transparency (m)	0.8	1.0	0.8b	1.7b	1.3
pH*	7.8	8.5	8.8	7.8	7.9
Conductivity* (us/cm)	1150	520	530	2950	1900
Temperature* (°C)	20.5	24.5	24.0	16.5	23.5
Maximum Depth (m)	1.2	2.0	0.8	1.7	4.0
Total Nitrogen (mg/l)	3.7	0.75	0.69	0.33	0.61
Total Phosphorus (mg/l)	0.31	0.063	0.089	0.012	0.035
Nitrogen:Phosphorus	11.9	11.9	7.8	27.5	17.4
Dissolved Oxygen	oxic	anox	oxic	oxic	anox

b: transparent to bottom *: at surface
oxic: oxygenated at bottom anox: anoxic at bottom

Table E8: Phytoplankton taxa recorded from Rouge Pond in numerical abundance and biovolume. Group symbols are: bg= blue-green algae, ch=chrysophyte, cr=cryptophyte, d= diatom, df=dinoflagellate, e=euglenoids, g=green algae, x=xanthophyceae.

			% by Number		% by Biovolume	
	Taxa	**Group**	**RP1**	**RP2**	**RP1**	**RP2**
1.	Achnanthes minutissima Cymbella microencephala	d	20.9	2.4	3.0	0.4
2.	Caloneis amphisbaena	d	1.0		9.5	
3.	Carteria sp.	g	12.9	3.3	22.1	7.3
4.	Chlamydomonas sp.	g	3.8	1.7	0.6	0.1
5.	Chlorophyte cells	g	7.0	15.4	1.7	2.4
6.	Chrysophyte flagellate	ch	4.4		0.9	
7.	Cryptomonas erosa	cr	8.2	6.5	12.6	11.1
8.	Cryptomonas erosa v. reflexa Cryptomonas pyrenoidifera	cr	2.2	1.4	1.8	1.5
9.	Cryptomonas marsonii	cr		1.1		1.0
10.	Cryptomonas phaseolus	cr	14.3	6.4	4.1	2.3
11.	Dinobryon divergens	ch		24.3		6.6
12.	Entomoneis alata	d	0.8		5.1	
13.	Euglena sp.	e	4.2	3.2	25.7	38.4
14.	Fragilaria nanana	d		1.3		0.5
15.	Fragilaria sp.	d		0.6		2.1
16.	Gonium sociale	g	1.6	2.2	1.3	2.6
17.	Gymnodinium sp.	df		1.9		2.2
18.	Microactinium pusillum	g		5.8		0.7
19.	Monoraphidium braunii	g	1.6	8.5	<0.1	0.2
20.	Navicula pygmaea	d	1.8		1.7	
21.	Nitzschia acicularis	d	1.4	1.5	0.5	0.5
22.	Nitzschia sp.	d	8.0	0.4	3.2	0.2
23.	Oocystis sp.	g		0.4		0.2
24.	Ophiocytium capitatum	x	0.5	3.3	<0.1	0.3
25.	Peridinium sp.	df	0.6	3.5	1.9	15.4
26.	Phormidium sp.	bg	0.6		0.3	
27.	Rhodomonas minuta	cr	1.6	0.9	0.2	0.1
28.	Scenedesmus longus	g		0.4		0.1
29.	Selenastrum minutum	g		1.7		<0.1
30.	Trachelomonas sp.	e	0.4		1.3	

Table E9: Periphyton taxa recorded from Rouge Pond in numerical abundance and biovolume. Group symbols are: bg=blue-green algae, d=diatom, g=green algae.

	Taxa	Group	% by Number		% by Biovolume	
			RP1	RP2	RP1	RP2
1.	Achnanthes minutissima Cymbella microencephala	d	92.1	67.9	48.7	2.6
2.	Chlorophyte cells	g		1.7		0.1
3.	Chroococcus sp.	bg		0.4		0.1
4.	Cyclotella sp.	d		0.9		0.6
5.	Denticula elegans	d	0.6	2.8	3.3	0.9
6.	Diatoma tenuis	d		2.5		0.8
7.	Fragilaria fasciculata	d		1.7		0.9
8.	Fragilaria ulna	d		2.2		2.5
9.	Gomphonema parvulum Gomphonema angustatum	d	4.8	2.0	19.8	0.5
10.	Gomphonema truncatum	d		0.4		0.2
11.	Merismopedia sp.	bg		5.6		<0.1
12.	Mougeotia sp.	g		1.2		51.8
13.	Navicula sp.	d		1.2		0.1
14.	Nitzschia sp.	d	0.8	1.5	0.6	0.1
15.	Oedegonium sp.	g		2.6		9.0
16.	Oscillatoria sp.	bg		1.9		23.4
17.	Protococcus viride	g		0.9		0.3
18.	Scenedesmus sp.	g		1.1		<0.1
19.	Spirogyra sp.	g	**		23.8	

** less than 0.4%

Table E10: Physical and Chemical Characteristics at Rouge Pond Sampling Site 1 (RP1) through the summer of 1997.

	Sampling Date				
Parameter	**June 27**	**July 16**	**Aug 1**	**Aug 20**	**Sept 5**
Secchi Depth (m)	1.7b	1.7b	1.7b	1.7b	1.7b
pH	7.9	7.8	7.8	7.8	8.0
Temperature (°C)					
0 m	17	18	17	17	13
1 m	13.5	14.5	15	14.5	12
Conductivity (uS/cm)					
0 m	3100	2900	3000	2500	3200
1 m	3200				
Dissolved Oxygen (mg/l)					
0 m				13.0	
1 m				13.0	
Total Phosphorus (mg/l)				0.012	
Total Nitrogen (mg/l)				0.33	

b: to bottom

Table E11: Physical and Chemical Characteristics at Rouge Pond Sampling Site 2 (RP2) through the summer of 1997.

	Sampling Date				
Parameter	**June 27**	**July 16**	**Aug 1**	**Aug 20**	**Sept 5**
Secchi Depth (m)	0.8	1.7	1.9	1.0	1.1
pH	8.2	7.8	7.9	7.6	7.9
Temperature (°C)					
0 m	28	27	23	21	18
1 m	21	21.5	22	19.5	18
2 m	18.5	19	19	18	17
3 m	14.5	15.5	16	16	16
3.8 m	14	14	13	13	12
Conductivity (uS/cm)					
0 m	1200	2500	2200	1300	2400
1 m	1900	3050	2450	1600	2400
2 m	2250	3150	2650	1700	2500
3 m	12500	13000	10000	10000	9500
3.8 m	14500	15500	15000	14000	14000
Dissolved Oxygen (mg/l)					
0 m				8.0	
1 m				7.8	
2 m				6.7	
3 m				0	
3.8 m				0	
Total Phosphorus (mg/l)				0.035	
Total Nitrogen (mg/l)				0.61	

Table E12: Physical and Chemical Characteristics at Harding Pond Sampling Site 1 (HP1) through the summer of 1997.

	Sampling Date				
Parameter	**June 27**	**July 16**	**Aug 1**	**Aug 20**	**Sept 5**
Secchi Depth (m)	0.8	0.7	1.0	0.3	1.1b
pH	7.7	7.4	7.7	8.2	7.8
Temperature (°C)					
0 m	24	24	20	17	16.5
1 m	18	16.5	18	15	14.5
Conductivity (uS/cm)					
0 m	1200	1200	1200	900	1200
Dissolved Oxygen (mg/l)					
0 m				>20	
1 m				18.2	
Total Phosphorus (mg/l)				0.31	
Total Nitrogen (mg/l)				3.7	

b: to bottom

Table E13: Physical and Chemical Characteristics at Harding Pond Sampling Site 2 (HP2) through the summer of 1997.

	Sampling Date				
Parameter	**June 27**	**July 16**	**Aug 1**	**Aug 20**	**Sept 5**
Secchi Depth (m)	1.0	0.9	1.0	0.8	1.1
pH	8.7	8.8	8.4	9.0	7.8
Temperature (°C)					
0 m	29	28.5	23	20	21
1 m	22.5	24	23	19	18
2 m	15	16	18	17.5	17
Conductivity (uS/cm)					
0 m	480	600	600	450	520
1 m	650	750	700	490	550
2 m	4850	4450	4400	3000	1150
Dissolved Oxygen (mg/l)					
0 m				17.4	
1 m				8.4	
2 m				0	
Total Phosphorus (mg/l)				0.063	
Total Nitrogen (mg/l)				0.75	

Table E14: Physical and Chemical Characteristics at Harding Pond Sampling Site 3 (HP3) through the summer of 1997.

	Sampling Date				
Parameter	**June 27**	**July 16**	**Aug 1**	**Aug 20**	**Sept 5**
Secchi Depth (m)	0.8b	0.8b	0.8b	0.6v	0.6v
pH	8.7	8.9	8.6	9.2	8.7
Temperature (°C)					
0 m	27	29	23	20	21
Conductivity (uS/cm)					
0 m	480	600	600	450	520
Dissolved Oxygen (mg/l)					
0 m				18.6	
Total Phosphorus (mg/l)				0.089	
Total Nitrogen (mg/l)				0.69	

b: to bottom
v: to vegetation

APPENDIX F

Summary of Water Quality Results

Table F1: Summary of water quality statistics at the inlet from May to November 1996 and 1997.

Parameter	Unit	DL	n	%>DL	Min	Max	Mean	SD	95%CI-UL	95%CI-LL
Aluminum	µg/L	10	22	100	113	3300	1422	1.0	2200	919
Arsenic	µg/L	1	22	23	0.5	2.0	1.05	0.3	1.2	0.9
Barium	µg/L	1	22	100	7.0	100.0	45.8	0.8	65.3	32.1
Beryllium	µg/L	0.1	22	45	0.1	0.2	0.1	0.3	0.2	0.1
Cadmium	µg/L	0.1	21	95	0.1	3.0	0.6	0.9	1.0	0.4
Chromium	µg/L	0.2	21	100	0.7	9.0	3.3	0.7	4.5	2.5
Cobalt	µg/L	0.2	21	90	0.0	5.0	2.2	0.9	3.2	1.4
Copper	µg/L	0.2	22	100	1.3	130.0	22.2	1.1	35.7	13.8
Iron	µg/L	20	22	100	156	2000	1069	0.7	1459	784
Lead	µg/L	5	20	60	1	33	11	0.8	15	7
Manganese	µg/L	0.5	22	100	15.0	380.0	162.4	0.9	236.7	111.5
Mercury	µg/L	0.02	22	14	0.01	0.03	0.02	0.1	0.02	0.02
Molybdenum	µg/L	0.2	12	55	0.1	1.8	0.6	0.8	1.0	0.4
Nickel	µg/L	0.5	22	95	0.2	6.5	3.4	0.6	4.4	2.6
Strontium	µg/L	2	22	100	7	450	195	0.9	289	131
Titanium	µg/L	1	22	73	1	30	6	1.0	8	4
Vanadium	µg/L	0.2	22	100	1.0	7.0	3.4	0.5	4.3	2.8
Zinc	µg/L	0.5	22	100	4.0	330.0	66.7	0.9	98.7	45.0
Total Ammonia	mg/L	0.002	21	77	0.001	1.20	0.37	2.1	0.96	0.14
Nitrite	mg/L	0.001	21	100	0.03	0.30	0.083	0.6	0.11	0.06
Nitrite + Nitrate	mg/L	0.005	21	100	0.33	1.95	1.02	0.5	1.29	0.81
TKN	mg/L	0.02	21	100	0.14	2.90	1.30	0.7	1.80	0.95
Total Phosphorus	mg/L	0.002	21	100	0.024	0.78	0.39	1.0	0.61	0.26
Phosphate	mg/L	0.0005	21	100	0.0015	1.2	0.26	1.6	0.54	0.13
TSS	mg/L	1	22	100	21	807	345	1.1	562	212
DOC	mg/L	0.1	22	100	1.5	8.6	3.5	1.7	4.3	2.8
DIC	mg/L	0.2	22	100	11.2	67.4	14.7	1.5	17.4	12.5
Conductivity	µS/cm	1	22	100	88	2550	312	0.9	457	213
pH	none		22	100	7.3	8.3	8.1	0.0	8.2	8.0
Alkalinity	mg/L CaCO	0.2	22	100	27.6	245.0	77.3	0.4	92.7	64.5
Chloride	mg/L	0.02	22	100	2	638	22	1.2	38	13
Turbidity	FTU	0.01	22	100	8.5	889.0	360.2	1.4	672.8	192.9
Silicon	mg/L	0.2	21	100	0.4	3.2	1.2	0.5	1.6	1.0
Oil and Grease	mg/L	0.5	21	82	0.3	8.0	2.1	0.9	3.1	1.4
E. Coli	c./100 mL	4	12	100	990	39000	8362	1.1	17180	4070
Fecal Coliforms	c./100 mL	4	12	100	1060	51000	16149	10292.0	36697	7107
Pentachlorophenol	ng/L	10	5	100	24	110	61	0.7	138	27
2,3,4,6-Tetrachlorophenol	ng/L	20	5	0	10	10	10			

Table F2. Summary of oultet water quality statistics from September to November, 1996 and May to November, 1997 (post-berm repair period)

Parameter	Unit	DL	n	%>DL	Min	Max	Mean	SD	95%CI-UL	95%CI-LL
Aluminum	µg/L	10	12	100	83	510	290	0.60	428	196
Arsenic	µg/L	1	12	8	0.5	1.0	0.8	0.34	1.0	0.7
Barium	µg/L	1	12	100	16.0	63.0	36.5	0.37	46.2	28.9
Beryllium	µg/L	0.1	12	8	0.05	0.26	0.06			
Cadmium	µg/L	0.1	9	100	0.2	1.9	0.5	0.70	0.8	0.3
Chromium	µg/L	0.2	10	100	0.7	15.4	2.4	0.99	4.9	1.2
Cobalt	µg/L	0.2	10	45	0.1	2.0	0.8	1.00	1.7	0.4
Copper	µg/L	0.2	12	100	2.0	8.0	4.5	0.40	5.8	3.5
Iron	µg/L	20	12	100	50	650	386	0.70	607	245
Lead	µg/L	5	4	0	2.5	15.0	7.7	0.55	18.3	3.2
Manganese	µg/L	0.5	12	100	38.2	174.0	115.8	0.50	157.6	85.1
Mercury	µg/L	0.02	12	0	0.01	0.01	0.01	0.00		
Molybdenum	µg/L	0.2	4	100	0.23	1.00	0.53			
Nickel	µg/L	0.5	10	92	0.25	43.0	3.2	1.24	7.7	1.3
Strontium	µg/L	2	9	100	176	452	271	0.30	342	214
Titanium	µg/L	1	3	100	2.0	4.0	3.1	0.35	7.3	1.3
Vanadium	µg/L	0.2	10	100	0.50	2.00	1.12	0.35	1.44	0.87
Zinc	µg/L	0.5	10	100	6.1	41.7	16.4	0.69	26.8	10.0
Total Ammonia	mg/L	0.002	12	85	0.001	0.112	0.102	1.43	0.252	0.041
Nitrite + Nitrate	mg/L	0.001	12	100	0.025	1.00	0.66	0.99	1.24	0.35
Nitrite	mg/L	0.005	12	100	0.004	0.034	0.024	0.62	0.036	0.017
TKN	mg/L	0.02	12	100	0.70	1.62	1.00	0.30	1.16	0.82
Total Phosphorus	mg/L	0.002	12	100	0.04	0.20	0.11	0.55	0.16	0.08
Phosphate	mg/L	0.0005	12	100	0.0045	0.040	0.014	0.70	0.022	0.010
TSS	mg/L	1	12	100	5	145	48	1.10	95	25
DOC	mg/L	0.1	12	100	3.3	6.3	4.6	0.16	5.1	4.1
DIC	mg/L	0.2	12	100	18.8	69.2	30.7	0.40	39.2	24.1
Conductivity	uS/cm	1	12	100	322	813	538	0.30	638	454
pH			12	100	7.7	8.4	8.1	0.03	8.3	8.0
Alkalinity	mg/L CaCO	0.2	12	100	77.2	256.0	133.0	0.40	171.4	104.5
Chloride	mg/L	0.02	12	100	27	152	71	0.50	94	53
Turbidity	FTU	0.01	12	100	3.2	131.0	32.1	1.00	61.2	16.8
Silicon	mg/L	0.2	12	100	0.36	3.66	2.00	0.82	3.15	1.10
Oil and Grease	mg/L	0.5	12	62	0.25	1	0.8	0.40	1.0	0.6
E. Coli	c/100mL	4	8	100	40	2700	1440	1.50	5081	408
Fecal Coliforms	c/100mL	4	8	100	140	5200	2858	1.20	8011	1019

Table F3: Summary of water quality statistics at the inlet from January to April, 1996 and December to April, 1997.

Parameter	Unit	DL	n	%>DL	Min	Max	Mean	SD	95%CI-UL	95%CI-LL
Aluminum	µg/L	10	14	100	10.0	1200	596.3	0.8	954.5	372.5
Arsenic	µg/L	1	15	60	0.5	5	1.7	0.6	2.4	1.2
Barium	µg/L	1	15	100	14.0	110	44.7	0.6	63.9	31.3
Beryllium	µg/L	0.1	15	8	0.05	0.10	0.05			
Cadmium	µg/L	0.1	15	86	0.05	1.00	0.48	0.8	0.74	0.30
Chromium	µg/L	0.2	14	100	1.0	13.0	4.7	0.7	6.9	3.2
Cobalt	µg/L	0.2	15	79	0.1	2.0	1.1	0.6	1.5	0.8
Copper	µg/L	0.2	14	100	1.3	39.0	23.2	1.2	46.3	11.7
Iron	µg/L	20	14	100	63	1400	861	0.7	1309	566
Lead	µg/L	5	14	79	2.5	21.0	11.8	0.6	16.9	8.2
Manganese	µg/L	0.5	14	100	0.1	480	96.7	2.1	318	29.4
Mercury	µg/L	0.02	15	7	0.01	0.01	0.01			
Molybdenum	µg/L	0.2	13	77	0.1	3.0	0.5	0.8	0.8	0.3
Nickel	µg/L	0.5	14	93	0.3	7.0	3.0	0.6	4.4	2.1
Strontium	µg/L	2	14	100	98	900	361	0.7	527	247
Titanium	µg/L	1	12	71	0.5	20	6.5	1.0	12.3	3.4
Vanadium	µg/L	0.2	14	100	1.0	4.0	2.0	0.5	2.6	1.5
Zinc	µg/L	0.5	14	100	4.0	130.0	79.0	0.9	136.0	45.9
Total Ammonia	mg/L	0.002	15	100	0.001	2.52	2.12	2.5	8.33	0.54
Nitrite	mg/L	0.001	15	100	0.004	0.25	0.10	1.1	0.18	0.06
Nitrite + Nitrate	mg/L	0.005	15	100	0.59	4.56	1.75	0.6	2.44	1.26
TKN	mg/L	0.02	13	100	0.68	6.90	2.25	0.7	3.42	1.48
Total Phosphorus	mg/L	0.002	12	100	0.104	1.15	0.403	0.7	0.61	0.27
Phosphate	mg/L	0.0005	15	100	0.005	0.33	0.21	1.3	0.44	0.10
TSS	mg/L	1	13	100	28	859	270	1.0	483	151
DOC	mg/L	0.1	15	100	1.5	8.6	3.9	0.5	5.1	3.0
DIC	mg/L	0.2	15	100	11.2	67.4	27.0	0.6	37.4	19.5
Conductivity	µS/cm	1	13	100	585	18300	2003	0.9	3425	1171
pH	none		13	100	7.70	8.20	7.94	0.0	8.05	7.83
Alkalinity	mg/L CaCO	0.2	13	100	51	286	132	0.6	186	95
Chloride	mg/L	0.02	15	100	121	6140	497	1.0	852	290
Turbidity	FTU	0.01	15	100	17	820	164	1.2	310	87
Silicon	mg/L	0.2	15	100	0.5	5.1	2.1	0.7	3.2	1.4
Oil and Grease	mg/L	0.5	15	87	0.3	9.5	3.1	0.9	5.1	1.9
E. Coli	c/100mL	4	3	100	140	3600	2272	1.8	189486	27
Fecal Coliforms	c/100mL	4	3	100	940	26000	18287	1.7	1146151	292
Pentachlorophenol	ng/L	10	11	91	5	140	71	1.0	140	36
2,3,4,6-Tetrachlorophenol	ng/L	20	11	0	10	10	10			

Table F4: Summary of oultet water quality statistics from December to April, 1997 (post-berm repair period)

Parameter	Unit	DL	n	%>DL	Min	Max	Mean	SD	95%CI-UL	95%CI-LL
Aluminum	µg/L	10	4	100	50	456	226	2.8	625	82
Arsenic	µg/L	1	4	50	0.5	1.5	0.87	1.9	1.6	0.5
Barium	µg/L	1	4	100	23	100	39	2.0	77	20
Beryllium	µg/L	0.1	4	0	0.1	0.1	0.1	1.0		
Cadmium	µg/L	0.1	3	67	0.1	0.1	0.1	1.9	0.1	0.0
Chromium	µg/L	0.2	4	100	0.9	4.1	2.2	1.9	4.0	1.2
Cobalt	µg/L	0.2	4	75	0.1	1.37	0.4	3.0	1.2	0.1
Copper	µg/L	0.2	4	100	3.25	20.4	10.2	2.2	22.3	4.6
Iron	µg/L	20	4	100	112	856	431	2.6	1093	170
Lead	µg/L	5	4	100	3	15	6	2.0	12	3
Manganese	µg/L	0.5	4	100	70.3	494	159.4	2.3	355.1	71.5
Mercury	µg/L	0.02	4	0	0.01	0.01	0.01	0.0		
Nickel	µg/L	0.5	4	100	1.09	2.43	1.85	1.5	2.67	1.29
Strontium	µg/L	2	4	100	157	522	264	1.8	480	145
Titanium	µg/L	1	3	100	2	4	3	1.4	4	2
Vanadium	µg/L	0.2	3	100	1.1	2.6	1.6	1.6	2.7	1.0
Zinc	µg/L	0.5	4	100	22.1	70.5	40.03	1.6	64	25
Total Ammonia	mg/L	0.002	4	100	0.298	0.464	0.35	1.2	0.4	0.3
Nitrite	mg/L	0.001	4	100	0.032	0.059	0.04	1.3	0.05	0.03
Nitrite + Nitrate	mg/L	0.005	4	100	0.87	1.3	1.1	1.2	1.3	0.9
TKN	mg/L	0.02	4	100	0.74	1.6	1.1	1.4	1.56	0.84
Total Phosphorus	mg/L	0.002	4	100	0.018	0.22	0.1	3.3	0.33	0.03
Phosphate	mg/L	0.0005	4	100	0.0035	0.084	0.03	4.4	0.13	0.01
TSS	mg/L	1	4	100	5	117	39	4.3	164	9
DOC	mg/L	0.1	4	100	2.4	3	2.6	1.1	2.9	2.4
DIC	mg/L	0.2	4	100	18.8	82.2	32.3	1.9	61.9	16.9
Conductivity	µS/cm	1	4	100	707	2760	1165.5	1.9	2,136	636
pH	none		4	100	7.76	8.14	8.0	1.0	8.15	7.83
Alkalinity	mg/L CaCO	0.2	4	100	87.6	355	143.1	1.9	268.7	76.2
Chloride	mg/L	0.02	4	100	144	758	274.4	2.1	560	135
Turbidity	FTU	0.01	4	100	3.93	136	39.0	4.9	186	8
Silicon	mg/L	0.2	4	100	1.06	4.92	2.0	1.9	3.82	1.06
Oil and Grease	mg/L	0.5	4	75	0.25	4.5	1.1	3.3	3.7	0.4
E. Coli	c/100mL	4	3	100	50	490	185	3.2	703	49
Fecal Coliforms	c/100mL	4	3	100	110	2380	832	5.8	6046	114

Table F5. Summary of oultet water quality statistics from May to August, 1996 (pre-berm repair period)

Parameter	Unit	DL	n	%>DL	Min	Max	Mean	SD	95%CI-UL	95%CI-LL
Aluminum	µg/L	10	6	100	5	680	640	1.3	2411	170
Arsenic	µg/L	1	6	0	0.5	0.5	0.5	0.0		
Barium	µg/L	1	6	100	41.0	78.0	56.3	0.3	73.8	43.0
Beryllium	µg/L	0.1	6	25	0.05	0.05	0.05	0.0		
Cadmium	µg/L	0.1	6	75	0.05	1.70	0.51	1.0	1.42	0.19
Chromium	µg/L	0.2	6	8	0.60	3.40	1.44	0.7	2.92	0.70
Cobalt	µg/L	0.2	6	88	0.10	1.20	0.88	0.3	1.22	0.63
Copper	µg/L	0.2	6	100	5.0	38.0	11.7	0.8	25.8	5.3
Iron	µg/L	20	6	88	10	640	551	1.4	2415	126
Lead	µg/L	5	6	25	2.5	10.0	5.8	0.3	7.9	4.3
Manganese	µg/L	0.5	6	100	70	180	129	0.4	202	83
Mercury	µg/L	0.02	6	13	0.01	0.04	0.02	0.3	0.03	0.02
Molybdenum	µg/L	0.2	6	50	0.1	0.6	0.3	0.5	0.5	0.2
Nickel	µg/L	0.5	6	100	1.5	2.5	2.1	0.2	2.6	1.7
Strontium	µg/L	2	6	100	170	390	259		369	182
Titanium	µg/L	1	6	88	0.5	4.0	2.8	0.5	4.8	1.6
Vanadium	µg/L	0.2	6	88	0.1	3.00	2.03	1.0	5.98	0.69
Zinc	µg/L	0.5	6	100	9.0	21.0	14.3	0.3	19.2	10.6
Total Ammonia	mg/L	0.002	8	75	0.001	0.36	0.26	2.1	1.45	0.05
Nitrite	mg/L	0.001	8	100	0.014	0.063	0.039	0.6	0.065	0.024
Nitrite + Nitrate	mg/L	0.005	8	100	0.56	1.38	0.96	0.3	1.22	0.75
TKN	mg/L	0.02	8	100	0.38	1.72	0.99	0.5	1.53	0.65
Total Phosphorus	mg/L	0.002	8	100	0.06	0.61	0.23	0.9	0.47	0.12
Phosphate	mg/L	0.0005	8	100	0.004	0.180	0.036	1.2	0.102	0.013
TSS	mg/L	1	8	100	18	1340	308	1.3	939	101
DOC	mg/L	0.1	8	100	2.8	4.8	3.8	0.2	4.4	3.3
DIC	mg/L	0.2	8	100	23.4	57.2	35.8	0.3	47.4	27.1
Conductivity	uS/cm	1	8	100	352	2290	819	0.6	1326	506
pH			8	100	8.07	8.42	8.22	0.0	8.32	8.13
Alkalinity	mg/L CaCO	0.2	8	100	109	252	160	0.3	207	123
Chloride	mg/L	0.02	8	100	22	1130	176	1.2	473	65
Turbidity	FTU	0.01	8	100	10.4	1570.0	300.3	1.6	1131.9	79.7
Silicon	mg/L	0.2	8	100	1.66	3.78	2.32	0.2	2.85	1.88
Oil and Grease	mg/L	0.5	8	38	0.25	1.00	0.63	0.3	0.82	0.48
E. Coli	c/100mL	4	5	100	1200	4500	2233	0.6	4524	1103
Fecal Coliforms	c/100mL	4	5	100	940	10100	4709	0.9	14806	1498
Pentachlorophenol	ng/L	10	4	75	5	160	53	1.3	450	6
2,3,4,6-Tetrachlorophenol	ng/L	20	4	0	10	10	10	0.0		

Table F6: Summary of oultet water quality statistics from January to April, 1996 (pre-berm repair period)

Parameter	Unit	DL	n	%>DL	Min	Max	Mean	SD	95%CI-UL	95%CI-LL
Aluminum	µg/L	10	11	100	180	710	419	0.42	556	315
Arsenic	µg/L	1	11	18	0.5	3.0	1.2	0.33	1.5	0.9
Barium	µg/L	1	11	100	24.0	64.0	42.0	0.35	53.0	33.3
Beryllium	µg/L	0.1	11	18	0.05	0.20	0.11	0.21	0.13	0.10
Cadmium	µg/L	0.1	11	73	0.05	1.60	0.45	0.91	0.83	0.25
Chromium	µg/L	0.2	11	100	1.60	8.20	3.65	0.54	5.23	2.54
Cobalt	µg/L	0.2	11	45	0.1	0.8	0.6	0.53	0.8	0.4
Copper	µg/L	0.2	11	100	5.0	30.0	15.9	0.62	24.5	10.5
Iron	µg/L	20	11	100	220	700	500	0.35	633	396
Lead	µg/L	5	11	45	2.5	20.0	8.6	0.57	12.6	5.9
Manganese	µg/L	0.5	11	100	68.0	270.0	123.1	0.35	155.9	97.2
Mercury	µg/L	0.02	10	0	0.01	0.01	0.01	0.00		
Molybdenum	µg/L	0.2	11	64	0.1	1.0	0.4	0.6	0.7	0.3
Nickel	µg/L	0.5	11	91	1.5	2.5	2.1	0.19	2.6	1.7
Strontium	µg/L	2	11	100	130	380	231	0.38	298	179
Titanium	µg/L	1	11	91	0.5	12.0	5.7	0.73	9.2	3.5
Vanadium	µg/L	0.2	11	82	0.1	4.00	2.12	1.00	4.15	1.10
Zinc	µg/L	0.5	11	100	20.0	86.0	44.8	0.53	63.9	31.4
Total Ammonia	mg/L	0.002	11	100	0.036	2.07	0.75	1.25	1.73	0.32
Nitrite	mg/L	0.001	11	100	0.019	0.120	0.073	0.60	0.110	0.050
Nitrite + Nitrate	mg/L	0.005	11	100	0.8	2.6	1.4	0.38	1.8	1.1
TKN	mg/L	0.02	9	100	0.6	5.5	2.0	0.64	3.1	1.2
Total Phosphorus	mg/L	0.002	9	100	0.07	0.43	0.26	0.68	0.44	0.15
Phosphate	mg/L	0.0005	11	91	0.0	0.2	0.2	1.8	0.6	0.1
Suspended Solids	mg/L	1	9	100	29	241	124	0.71	213	72
DIC	mg/L	0.1	10	100	2.3	8.9	4.2	0.37	5.5	3.3
DIC	mg/L	0.2	10	100	16.0	54.4	34.4	0.50	49.0	24.2
Conductivity	µS/cm	1	9	100	839	1780	1332	0.26	1620	1095
pH	none		9	100	7.73	8.29	7.98	0.02	8.12	7.90
Alkalinity	mg/L CaCO	0.2	9	100	72	232	162	0.40	220	120
Chloride	mg/L	0.02	11	100	147	1680	413	0.63	629	271
Turbidity	FTU	0.01	11	100	13.7	151.0	73.5	0.87	131.5	41.1
Silicon	mg/L	0.2	10	100	0.84	3.04	2.31	0.40	3.10	1.73
Oil and Grease	mg/L	0.5	11	82	0.25	4.5	1.8	0.68	2.8	1.1
E. Coli	c/100mL	4								
Fecal Coliforms	c/100mL	4								
Pentachlorophenol	ng/L	10	10	100	10	79	42	0.86	78	23
2,3,4,6-Tetrachlorophenol	ng/L	20	10	0	10	10	10	0.00		

APPENDIX G

Removal Efficiencies

Table G1: Load-based performance results (%) for the summer/fall period from September to November, 1996 and May to November, 1997.

Parameter	7/9/96	13/9/96	24/9/96	27/9/96	18/10/96	29/10/96	7/11/1996	20/08/97	10/9/1997	1/11/1997	Average*	Overall Removal Efficiency+
Aluminum	81	88	84	78	7	81	30	39	41	50	58	74
Arsenic	1	14	7	51	6	10	9	59	20	11	19	17
Barium	27	42	37	-95	-107	-148	-88	-88	-232	-95	-75	-11
Cadmium	-15		68	-286	0		22	-579	-1	38	-94	11
Chromium	62	87	69	76	3		17		16	53	48	53
Cobalt	94	78	99	89	3	73	89	11		-201	37	82
Copper	14	72	73	49	45	85	56	35	60	75	56	48
Iron	74	80	75	69	8	76	33	74	15	46	55	66
Lead		91	98	63	95	84			1	31	66	83
Manganese	11	57	62	-112	-31	-43	-12	-9	-124	-49	-25	9
Nickel	75	78	87	43	4	55	-10	65	-365	40	7	62
Strontium	-59	-54	-52	-20	-140	-128	-117	11	-339	-221	-112	-63
Titanium	12	-224	-748	19	-103	82	-65	-812	27	66	-175	-18
Vanadium	68	74	79	70	48	58	61	16	53	61	59	66
Zinc	78	94	89	50	-7	84	8	-129	57	72	40	70
Total Ammonia	1	-845	-219	-76	74	14	97	-1764	75	-347	-299	54
Nitrite	36	43	63	43	22	58	53	79	69	-18	45	42
Nitrite + Nitrate	39	12	-27	26	31	3	-1	34	65	37	22	28
Nitrogen, TKN	-122	17	2	-23	-7	35	29	-9	38	36	0	-24
Phosphorus, Total	2	46	79	75	20	85	51	80	46	51	53	42
Phosphate	90	96	96	87	73	94	64	96	67	-10	75	86
TSS	85	87	88	94	26	92	48	88	73	61	74	80
Diss. Carbon, Org.	-85	-76	-104	-142	-72	12	-51	-130	-69	-4	-72	-70
Diss. Carbon, Inorg.	-120	-169	-181	-249	-131	-275	-147	-65	-168	-135	-164	-155
Chloride	-515	-460	-675	-291	-909	-403	-364	-1165	-1883	-985	-765	-548
Turbidity	90	86	91	90	28	82	61	91	48	47	71	84
Silicon	-135	-143	-148	-257	-248	-322	-181	-48	-1	-95	-158	-168
Oil and Grease	1	43	54	51	53	85	9	19	60	70	44	48
E. Coli	60		89	95	-132			83			39	53
Faecal Coliforms	61		80	97	-150			83			34	67

* Simple average of 10 individual event removal efficiencies

+ Removal efficiency based on the sum of influent and effluent loads (see section 3.2.3)

Table G2: Concentration-based performance results (%) for the winter/spring period from December 1 to April 30, 1997.

Parameter	1/22/97	2/21/97	2/27/97	3/25/97	AVERAGE
Aluminum	89	19	-	35	48
Arsenic	50	40	-	-	45
Barium	11	-32	38	-74	-14
Cadmium	-	58	91	99	83
Chromium	86	21	-191	32	-13
Cobalt	39	33	94	75	60
Copper	91	-10	-22	30	22
Iron	89	19	-	37	48
Lead	86	-27	-46	29	10
Manganese	-145	-30	76	7	-23
Nickel	5	30	75	40	38
Strontium	41	46	67	19	43
Titanium	-	-	-	-19	-19
Vanadium	-	-18	-2	10	-3
Zinc	74	-13	72	19	38
Total Ammonia	58	-5	-1	19	18
Nitrite	85	35	57	32	52
Nitrite +Nitrate	-17	20	-47	-2	-12
Nitrogen, Total Kjeldahl	81	-7	48	0	31
Phosphorus, Total	95	19	76	34	56
Phosphate	92	63	70	41	66
TSS	**97**	**58**	**96**	**60**	**78**
DOC	-20	7	-100	17	-24
DIC	-209	-4	-64	-104	-95
Chloride	88	-52	77	-124	-3
Turbidity	97	68	94	51	78
Silicon	-257	16	-73	-145	-115
Oil and Grease	95	0	-100	31	6
E. Coli	99	-	-145	-86	-44
Fecal Coliforms	100	-	44	-153	-3
Pentachlorophenol	-	-	-	-	-

Note: In the absence of flow measurements during the winter/spring period, the removal efficiencies are calculated based soley on influent and effluent concentrations. See section 3.2.3 for method of calculation.

APPENDIX H

Numerical Simulation

1.0 MODEL STRUCTURE AND CALIBRATION

1.1 Model Structure

SWMM 4.3, originally developed by the USEPA, was designed to predict stormwater runoff quality and quantity values based on meteorological and land-form characteristic data (Huber and Dickinson, 1988). It is a parametric, spatially distributed model, which can operate in single event or continuous mode. The program consists of several modules or blocks, each one simulating an aspect of the hydrologic cycle and incorporating the hydraulic conditions of the collection and or storage system, and the pollutant removal efficiency of the treatment facility (Huber and Dickinson, 1988). For this study, the modules applied were:

1.1.1 Rain

This block processes lengthy precipitation records from the National Weather Service (U.S.) or Atmospheric Environment Service (Canada) for use by the Runoff block.

1.1.2 Runoff

This module generates surface runoff and pollutant loads in response to precipitation and surface pollutant accumulation. It also has a limited ability to route flows through pipes and gutters. However, the more sophisticated routines in the Transport and Extran blocks are employed for this purpose. Rainfall data can also be executed within the Runoff module, which was the case in this exercise where the precipitation events used to develop, calibrate and verify the model were directly read into the Runoff module. The catchment was not subdivided, but was modelled as one lumped subcatchment.

1.1.3 Transport

This block routes flows and pollutant loads through a sewer system. The flows and loads are generated by the Runoff block, and input to points throughout the system. Transport also has the ability to generate and route the continuous dry weather flow component (baseflows) which occurs at the facility inlet. Though surcharge conditions did not occur during this study, it is important to note that Transport cannot simulate backwater effects and surcharge conditions. In cases involving surcharge conditions, the Extran block should be used.

1.1.4 Storage/Treatment

This module is designed to route flow and pollutant loads through a storage/treatment facility. In our case, these flows and loads come from the Transport block. The user is given a great deal of flexibility by the block's ability to connect as many as five storage/treatment units, which may be given storage characteristics or be modelled as a simple flow-through device.

1.2 Simulation of Storage/Treatment

The Harding Park model was set up with three storage / treatment components:

1. *Wet pond.* The wet pond was the primary storage and treatment unit of the facility. The sediment forebay was not modelled due to its very small capacity (permanent pool of 15 m^3) and the wetland cell was not modelled because it did not maintain any detention function. Both these cells were flow-through units and while of limited importance to hydraulic storage, were considered to be important to water quality improvement. Therefore, the removal efficiency of the facility, though based on a single wet pond unit, was developed to represent the performance of the entire facility from inlet to outlet.

2. *Baseflows.* Based on data collected during the monitoring period, the average inlet baseflow was estimated at 1.5 l/s and the average outlet baseflow was estimated at about 1.25 l/s. To model baseflow a second storage/treatment unit was set–up which accounted for baseflow volume both in event and continuous (wet and dry weather) mode.

3. *Infiltration.* The water balance from inlet:outlet baseflows indicated a 0.25 l/s water loss, which was accounted for in the model as infiltration to the surrounding soils. Flows in excess of the baseflow limit were routed to the baseflow storage treatment unit where 0.25 l/s infiltration was discounted and then the excess flow was routed to the outflow from the facility.

1.3 Sensitivity Testing, Calibration and Verification

For each module or block, the entire validation process includes successive sensitivity analysis, calibration and verification. Sensitivity analysis is intended to prioritize the model parameters. It proceeds by holding all parameters but one constant at their expected value, and perturbing that parameter within reasonable expected limits such that the variation of an objective function can be examined. If apparently small perturbations of a parameter produce large changes in the objective function, the system is said to be sensitive to that parameter. In this case, the influence of parameters was investigated on flow volumes and rates for water quantity, and suspended solids (TSS) loads for water quality. Sensitivity analysis determines which parameters are going to be set during calibration as well as the accuracy required for these parameters.

Calibration is performed on all storms available but one, which is used for verification. During calibration, the input parameters are set sequentially until the results of the model match the observed data within a reasonable range. The objective range for this study was 10% at the inlet and 20% at outlet, for flow volumes, peak flows, and TSS loads.

Verification is intended to establish the credibility of the model for data different from the calibration data set. During verification, the model was run with the parameters obtained during calibration for the remaining events. If the results match the observed data within a reasonable range, the calibration can be considered to be independent from the data set on which it was performed.

2.0 MODELLING CONSTRAINTS

There were some constraints within the monitoring design, which limited the functionality of the program. Rainfall data for 1996 was obtained from Toronto Buttonville Airport located 8 km to the southeast of the site. Analysis of rainfall-runoff results in Chapter 4 indicated that the distance between the facility and weather station might explain observed inconsistencies between rainfall and runoff in 1996. As several storms from 1996 were used to develop the model, this constraint added some uncertainty to the model. Rainfall data for the long-term simulation was also obtained from Buttonville airport.

Failure and delayed repair of the berm separating the wet pond and wetland cells limited the number of events available for analysis. Since berm failure did not affect inlet runoff, a total of 14 storms from this site were used to compare simulated results. The model was set up using the stage-discharge relationship developed after berm repair, and calibrated using 7 storms available during 1997 and the fall of 1996.

Only seven storms with complete inlet and outlet sample collection were available for use in calibrating and verifying the storage/treatment block. Therefore a larger factor of error was expected for the storage/treatment component of the model, as opposed to the runoff and transport blocks, which had ample observed data.

Since the flow monitoring equipment was removed during the cold season, the modelling exercise was based on flow rates recorded during the warmer season (May – November). Snowfall or snowmelt events were not modelled.

Finally, the facility was not monitored at cell junctions. As a result, the model was not set up to simulate the cells of the facility. Therefore, the treatment performance and solids accumulation amounts within each cell were not simulated. It is intuitive to assume that larger particles would settle and accumulate within the forebay, but due to its limited storage capacity, the accumulation threshold would be reached rapidly, after which scouring and resuspension might be expected. Site reconnaissance indicated that a well–defined and somewhat incised channel routed through the wetland cell, which led to the assumption that little solids accumulation occurred within the wetland cell.

The model was set up, sensitized, calibrated and verified in each block for runoff quantity and peak flows. After the parameters affecting flow were set, the water quality parameters were established. It should be noted that flow parameters were not adjusted again in the quality calibration exercise. As SWMM calculated the suspended solids concentration based on build–up and wash off from the facility, errors in flow are reflected in SS loading calculations. Consequently, the widest margin of error is expected in quality results. The estimate of error can be used as the best and worst case scenarios, and the actual results can be expected to fall within these extremes.

For complete documentation and discussion of SWMM modules, parameters and functions see Huber and Dickenson (1988) and James and James (1995).

3.0 WATER QUANTITY MODELLING

3.1 Runoff block

Sensitivity analysis of parameters in the runoff block was performed using three simulated rainfall scenarios: (i) events of short duration (20 minutes) and high intensity (76 mm/h), (ii) medium duration (60 minutes) and high intensity (25 mm/h) and, (iii) long duration (600 minutes) and low intensity (5 mm/h).

PCSWMM computes mean-normalized sensitivity gradients. They can range from -1 to 1 and indicate the relative variation of the objective function when one parameter is modified and the others kept constant. A gradient close to ±1 indicates that the parameter is very sensitive. Results for the five most sensitive parameters are shown in Table H1. For all other parameters, gradients are smaller than 0.01.

Table H1: Sensitivity analysis of Runoff block parameters

Parameter	Total Flow Volume				Peak Flow			
	SDHI	**MDHI**	**LDHI**	**Rank**	**SDHI**	**MDHI**	**LDHI**	**Rank**
% Imperviousness	0.98	0.99	1.00	1	0.73	0.98	1.00	1
Impervious Depression Storage	-0.05	-0.05	-0.03	2	-0.03	-0.01	-0.01	5
Width	0.02	0.02	0.01	3	0.29	0.02	0.01	2
Imperv. Manning's *n*	-0.02	-0.01	-0.01	4	-0.27	-0.02	-0.01	3
Slope	0.02	0.01	0.01	5	0.13	0.01	0.01	4

note: SDHI: short duration, high intensity; MDHI: medium duration, high intensity; LDHI: long duration, high intensity

Percent imperviousness was the most sensitive parameter affecting both total flow volume and peak flow. The percent imperviousness was calibrated up from an initial value of 35% to 44%. As the simulation accuracy of peak flow is secondary to total flow, percent imperviousness was not recalibrated again to match peaks. Instead, for the second most sensitive peak flow parameter, catchment width was adjusted from an initial value of 300 m to 200 m, to match peak flows. Impervious Manning's *n* was adjusted from an initial value of 0.015 to 0.013; impervious depression storage was increased from 1.6 to 2.0 mm and slope was increased from 1 to 1.5%.

Table H2: Observed and simulated inlet runoff and peak flow rate summary for 10 storms

Date	**Total Flow Volume (m^3)**			**Peak Flow(l/s)**		
	Observed	**Simulated**	**% difference**	**Observed**	**Simulated**	**% difference**
June 7/96	610	948	-55.4	126	125	0.8
June 29/96	1742	2188	-25.6	552	450	18.5
July 7/96	797	1120	-40.5	499	500	-0.2
Sept.7/96	5838	4517	22.6	339	255	24.8
Sept.13/96	1465	1521	-3.8	105	92	12.4
Sept. 24/96	1207	1213	-0.5	157	116	26.1
Oct. 18/96	3189	2377	25.5	207	169	18.4
Aug. 20/97	1017	1250	-22.9	43	116	-169.8
Sept.10/97	993	996	-0.3	50	88	-76.0
Nov. 1/97	1344	1176	12.5	58	108	-86.2
Total flow and Mean Peaks	18202	17306	4.9	213.6	201.9	5.5
Standard. Dev.	0.79	0.57	4.98	0.79	0.68	10.9
Correlation (r)		0.98			0.97	

Table H2 presents a comparison of observed and simulated inlet runoff volumes and peak flow rates for 10 storms for the 1996 and 1997 summer/fall periods. Inlet total flow was matched within approximately 25% by all storms except June 7 and July 7, 1996. Both these storms were relatively small, but intense. Storms in the medium duration-high intensity to long duration-high intensity range, which account for the majority of real storm events, were well matched by the simulated flow volumes. Note that the observed and simulated mean storm volumes are quite similar, with a difference of only about 5%. A strong correlation ($r = 0.98$) was found between the observed and simulated flow volumes.

Similarly, peak flow rates were also matched within about 25%, with the exception of the Aug. 20, Sept. 10 and Nov.1, 1997 storms, which were also small, but of low intensity. Storms in the more common MDHI to LDHI ranges were well matched in peak flow. The simulated mean peak flow rate was only 5.5% less than the observed. Again, a strong correlation ($r = 0.97$) was noted between the observed and simulated peak flow results.

3.2 Transport Block

The Transport block routes flows through a sewer system, using the kinematic wave approximation of the St. Venant Equations. It cannot model backwater effects and surcharge conditions. When surcharge conditions are encountered, flows exceeding the open capacity of a conduit are simply stored at the upstream end of the conduit and released when the flow is less than this capacity. In this situation, the flow is limited and downstream conditions are distorted.

In this study, routing effects were small and did not justify the use of the Transport block. A constant inlet baseflow rate of 1.5 l/s, estimated as the average baseflow, was input in this module.

3.3 Storage/Treatment block

As indicated above, the storage/treatment module was subdivided into three parallel units: the wet pond, baseflow and infiltration components. Both baseflow and infiltration rates were constant and vary in volume only as a function of run time. Output volume and flow rate from unit one, the wet pond, were determined from the field calibrated stage-storage-discharge relationship presented in section 2.3.1.3.

The most sensitive parameter was the initial volume within the facility at the start of the storm, followed closely by the discharge rate from the facility. As the mean interevent period was over 5 days during 1996-97 and the shortest was one day, the initial volume at the start of the storm was set at the permanent pool equilibrium level of 1040 m^3. The relationship between stage, storage and discharge was previously verified from field data and therefore considered quite accurate. As a result, discounting the temporally variable infiltration volume and evaporation, and as long as the simulation ran long enough for the facility to re-attain hydraulic equilibrium, there was little difference between inlet and outlet flow volumes.

As mentioned, the berm separating the wet pond and wetland was not repaired until August, 1996. Therefore, Table H3 presents the observed and simulated flow volume and peak flow rates for 7 storms studied during 1996/97 after berm repair. Simulated outlet flow volume was matched within the same 25% range as that of inlet flow volume, with the exception of the Aug.20/97 storm. Mean simulated outlet volume was approximately 10% less than the observed outlet volume. The correlation coefficient between observed and simulated outlet flow volume was 1.0.

Table H3: Observed and simulated outlet runoff and peak flow rate summary for 7 storms after berm repair.

Date	Total Outlet Flow Volume (m^3)			Outlet Peak Flow (l/s)		
	Observed	Simulated	% difference	Observed	Simulated	% difference
Sept.7/96	5764	4463	22.6	55	81	-47.3
Sept.13/96	1465	1486	-1.4	38	30	21.1
Sept. 24/96	1118	1183	-5.8	28	32	-14.3
Oct. 18/96	3013	2326	22.8	69	40	42.0
Aug. 20/97	824	1211	-47.0	16	29	-81.3
Sept.10/97	791	961	-21.5	16	25	-56.3
Nov. 1/97	1201	1143	4.8	22	29	-31.8
Total mean peak flow	14176	12773	9.9	34.9	38.0	-9.0
St. Deviation	0.8	0.6	2.2	0.5	0.4	-4.2
Correlation (*r*)		1.0			0.6	

Outlet peak flow rates were less consistent than outlet flow volumes as indicated by the percent difference between observed and simulated results. The mean peak flow rates were more similar, with the mean simulated peak 9% greater than the observed. It should also be noted that the similarity of the observed and simulated means and the respective mean–normalized standard deviations, indicate that the simulated peak flow rates should fall within the deviation range of the observed peak flow rates.

In summary, inlet observed and simulated flow volume and peak rates were matched to within 25%. Outlet observed flow volumes, based on a smaller storm set, were also matched within 25% by the simulated results. The outlet peak flow rates were more inconsistent with error ranges of about 50%, but the simulated and observed means were matched within 10%. On a mean basis, inlet flow volumes and peak flow rates were 4.9 and 5.5% less than the observed results, respectively. Simulated outlet flow volumes were 9.9% less than the observed results and simulated outlet peak flow rates were 9% greater than the observed mean. Note when comparing mean inlet and outlet values that the outlet sample included only seven storm events.

4.0 WATER QUALTIY MODELLING

Stormwater quality modelling is an imprecise exercise because of the many variable and interdependent factors that can influence water quality (Huber and Dickenson, 1988). The primary objectives of modelling at the Harding Park facility were to assess long-term suspended solids performance and predict dredging and maintenance frequencies.

4.1 Runoff Block

SWMM operates under the assumption that pollutants build-up during interevent periods, with the option of setting an upper limit, and is then washed off during runoff events. The runoff rate can be exponential.

Sensitivity analysis was performed on the washoff coefficient, the maximum potential amount of pollutant per hectare in the subcatchment, the pollutant build-up rate and the exponent on the runoff rate. The sensitivity analysis was conducted using one and three dry days prior to the event and was consistent with other sensitivity analyses runs in SDHI, MDHI and LDHI modes. The sensitivity results are presented in Table H4.

Table H4: Sensitivity analysis of Inlet runoff water quality parameters.

Parameter	One dry day prior				Three dry days prior			
	SDHI	MDHI	LDHI	Rank	SDHI	MDHI	LDHI	Rank
Washoff Coefficient	1.0	0.99	1.0	1	1.0	0.99	1.0	1
Max. pot. Poll. Mass	0.01	0.01	0.01	4	0.33	0.34	0.33	2
Poll. Build-up rate	0.33	0.34	0.33	2	0.01	0.01	0.01	4
Runoff rate exponent	-0.09	-0.03	-0.01	3	-0.09	-0.03	-0.01	3

The washoff coefficient is by far the most sensitive parameter during all events with either a short or moderate interevent period. Pollutant build-up rate was somewhat sensitive during the short interevent period and maximum potential pollutant mass within the facility was somewhat sensitive during the three-day interevent period simulations. Huber and Dickenson (1988) indicate that the sensitivity of the pollutant build-up rate and maximum potential pollutant mass depends on the interevent period. When the maximum pollutant mass is not yet reached, build-up rate is sensitive, but after the maximum is attained, build-up rate is redundant. The runoff rate exponent was least sensitive as it affects the shape of the pollutograph more so than the total pollutant loading.

The maximum potential pollutant mass was calculated at an average of 800 kg for the catchment (47 kg/ha). During the July 15/96 storm, the maximum pollutant load was washed off after 1.2 dry days and therefore the build-up rate was calibrated as 41 kg/ha/day. The washoff coefficient was calibrated from an initial value of 5 to 2.4 and the runoff exponent from an initial value of 1 to 1.2.

As previously noted, errors in flow volume can be reflected in pollutant load estimates because pollutant concentrations are multiplied by runoff volume to derive the loads. Table H5 presents the observed and simulated Inlet TSS loads.

The storm events of June 7/96, Sept.10/97 and Nov.1/97 were not included as the observed SS concentrations were considered to be very low (less than 66 mg/L). In the above cases the modelled concentrations ranged from 288 to 339 mg/L and were thought to better approximate storm concentrations. The largest storm monitored, Sept. 7/96, was also excluded because the composite sample represents only half the total storm volume of 5838 m^3.

Table H5: Observed and simulated Inlet suspended solids loads.

Date	**Inlet SS load (kg)**		
	Observed	**Simulated**	**% difference**
June 29/96	674	608	9.8
July 7/96	420	423	-0.7
Sept.13/96	602	433	28.1
Sept. 24/96	266	389	-46.2
Oct. 18/96	332	580	-74.7
Aug. 20/97	309	372	-20.4
Total load	2603	2805	-7.8
St. Deviation	0.35	0.20	-4.47
Correlation (*r*)		0.50	

The storm of Oct.18/96 also had a large volume, yet relatively low observed TSS concentration (104 mg/L). The simulated concentration of 182 mg/L was thought to provide a reasonable estimate of this large storm's loading concentration. With the exception of the Oct 18/96 storm and the storm on Sept. 24/96, observed TSS loads are matched by simulated loads within 30%. The total observed and total simulated TSS loads for the six storm events differed by only 7.8%.

4.2 Transport Block

The transport block was viewed as a completely mixed reactor. No quality data on dry weather flows were available and as a result, baseflows routed through the block were assigned a TSS concentration of 3 mg/L.

4.3 Storage/Treatment Block

In SWMM a detention facility may either function as a plug flow system or as a completely mixed reactor. As the wet pond permanent pool volume was large compared to the average event runoff volume, and mixing was facilitated by the short length:width ratio of the pond (1:1), the completely mixed reactor scenario was used. In this scenario incoming material was instantly distributed uniformly throughout the facility yielding uniform pollutant concentrations throughout the facility. This method is incapable of using particle size selective settling velocities to calculate removal efficiency. The removal efficiency was determined as a function of the pollutant decay coefficient, which in conjunction with time determines the decay or removal rate. The initial value of the decay rate was assessed as 2×10^{-5}/s. Simulating to match the average performance and detention times, the decay rate was calibrated to 4.9×10^{-5}/s. Table H6. presents the observed and simulated TSS loads and performance levels for storms after berm repair.

Table H6: Observed and simulated outlet loads and performance summary for seven storms after berm repair.

Date	Outlet load (kg)			Performance (%)		
	Observed	Simulated	% difference	Observed	Simulated	% difference
Sept.7/96	254	218.5	14.0	85	71	16.5
Sept.13/96	94	106.5	-13.3	84	75	10.7
Sept. 24/96	31	99	-219.4	88	75	14.8
Oct. 18/96	244	154	36.9	27	73	-170.4
Aug. 20/97	44	81	-84.1	86	78	9.3
Sept.10/97	5.5	58	-954.5	74	80	-8.1
Nov. 1/97	18	81	-350.0	61	76	-24.6
Total load Mean Perf.	690.5	798	-15.6	72.1	75.4	-4.6
St. Deviation	0.9	0.4	-20.0	0.3	0.0	-12.8
Correlation (*r*)		0.9			0.2	

As mentioned above, the storms of Sept.10/97 and Nov.1/97 had extremely low inlet TSS concentrations. The storm of Sept. 24/96 had a low outlet concentration (28 mg/L), resulting in a large difference between the observed and simulated results. Observed and simulated outlet loads for the Sept 7/96, Sept. 13/96 and Oct 18/96 storms are matched within 40%. It should be noted that due to smaller loads at the outlet, relatively small mass discrepancies can amount to large proportional differences. The performance analysis indicates that within the estimated range of error, the observed and simulated performance were quite similar.

Table H7 presents error ranges for the water quantity and quality parameters of total flow volume, peak flow rate, loading and performance.

Table H7: Summary of inlet and outlet simulation error ranges for comparison parameters

Parameter	*Inlet Error Range (±%)*	*Outlet Error Range (±%)*
Event flow volume	25	25
Total Flow volume	10	10
Event peak flow rate	25	50
Mean peak flow rate	5.5	9
Event SS load	30	40
Total SS load	10	15
Event SS Performance		25
Mean SS Performance		5

References

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