Performance Assessment of an Open and Covered Stormwater Wetland System - Aurora, Ontario

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PERFORMANCE ASSESSMENT OF AN OPEN AND COVERED STORMWATER WETLAND SYSTEM – AURORA, ONTARIO

Prepared by:

STORMWATER ASSESSMENT MONITORING
AND PERFORMANCE (SWAMP) PROGRAM

for

Great Lakes Sustainability Fund of the Government of Canada
Ontario Ministry of the Environment
Toronto and Region Conservation Authority
Lake Simcoe Region Conservation Authority
Municipal Engineers Association of Ontario
Town of Aurora

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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 the Environment, 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 the SWAMP studies.

During the mid to late 1980s, the Great Lakes Basin experienced rapid urban growth. Stormwater runoff associated with this growth is a major contributor to the degradation of water quality and the destruction of fish habitats. In response to these environmental 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 designed to address this need.

The SWAMP Program's objectives are:

* to monitor and evaluate the effectiveness of new or innovative 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 sponsoring agencies is included in Appendix A.

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

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EXECUTIVE SUMMARY

Natural wetlands have long been viewed as providing important ecosystem functions in flood control and water quality enhancement. Recognition of these natural ecosystem services has led to the engineering and construction of wetlands to treat wastewater and stormwater. Although the idea of using natural systems for pollutant removal initially met with some resistance, constructed wetlands have now became recognized in Ontario as one of the most effective best management practices for stormwater treatment and runoff control.

Unfortunately, there is still a paucity of local monitoring data to support claims of effectiveness, and published information on the year-round performance of constructed wetlands in temperate climates is also very limited. This three year monitoring study of an extended detention wetland located in Aurora, Ontario was intended to help fill these knowledge gaps and provide a demonstration site against which other stormwater wetland designs could be evaluated. This goal was accomplished through analyses of system hydrology, water quality, temperature, vegetation dynamics, benthic invertebrates and sediment chemistry. Comparative evaluation of a heated greenhouse installed within the wetland offers additional insights into the role temperature plays in wetland treatment and the potential for enhancing performance during the cold season.

Study Site

The study was conducted on a 1.2 hectare stormwater wetland in Aurora, Ontario (Figure 1). The drainage basin for the facility was 82.4 hectares, of which 30% was agricultural and the remainder was medium density residential. This facility was designed and constructed in 1988 as a dry pond, but it evolved into a wetland as moist conditions attracted aquatic plants. Modification to the outlet structure prior to the study created an extended detention capacity of 16 m^3/ha. This modification increased the stormwater residence time within the facility, maintaining a pool of standing water on an intermittent basis, and resulted in a drawdown period of 3 to 5 days after rain events.

The modified facility was unique in its combination of both pond and wetland features. It had no permanent pool, other than in the forebay (40 m^3), and would become dry during periods of infrequent rainfall. The facility did not meet the Ontario Ministry of the Environment (OMOE) stormwater wetland guidelines for permanent pool volume (23 m^3/ha), extended detention volume (40 m^3/ha) and length-to-width ratio (3:1). The length-to-width ratio of the wetland portion of the facility was only 2:1.

The greenhouse was constructed from December, 1995 to July, 1996. The greenhouse was located near the outlet of the facility, had a length-to-width ratio of 4.3:1, and covered 10% (or 210 m^2) of the wetland basin area. The greenhouse consisted of a structural base, upper frame and equipment shed and was heated by a natural gas greenhouse furnace to maintain air temperatures at 10°C. Vegetation within the greenhouse was similar to that of the wetland.

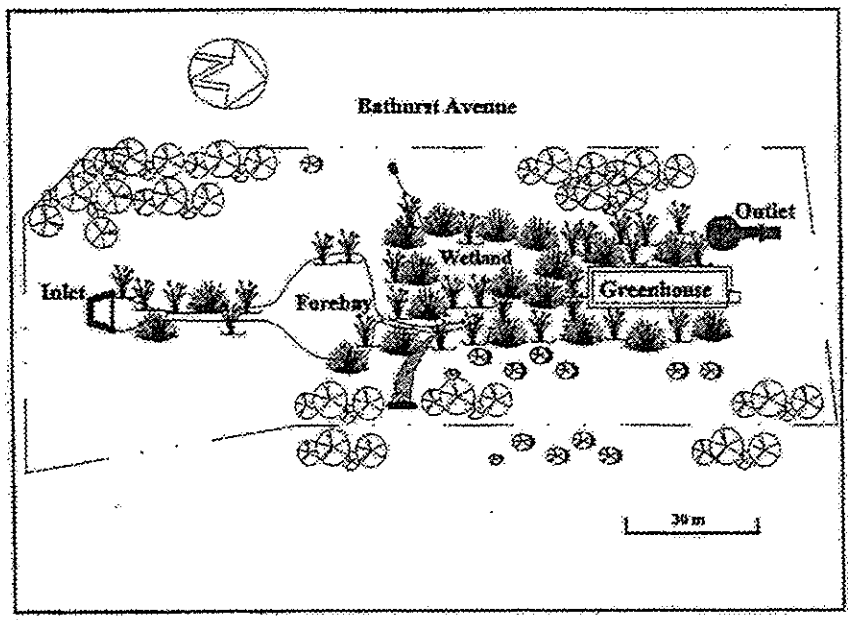

Figure 1: Aurora wetland facility

Study Methods

The water balance of the facility was estimated for 29 runoff events over the study period. Instruments employed for measurement of water balance components included area-velocity flow meters at the inlet and outlet, a water depth sensor in the wetland, groundwater piezometers, a class 'A' evaporation pan, and an automated rain gauge. Temperature of the water, soil substrate and air were recorded at six locations; three in the greenhouse and three in the wetland. Automated samplers at the inlet, outlet, greenhouse and wetland provided water quality data.

Water quality samples were collected at the inlet, outlet, greenhouse and wetland to identify temporal and spatial trends as stormwater passed through the facility, as well as to assess relative differences in water quality among the wetland, greenhouse and outlet monitoring stations. Samples at the inlet were flow-proportioned over the duration of each runoff event, except during the winter, when a single set of 3 grab samples was collected at the inlet near the end of the runoff event. Outlet samples were collected once every hour over a period of 24 hours, then combined into a single composite sample for each day of the drawdown period. Pollutant concentrations of these daily composite samples were later proportioned by daily flow to approximate the effluent event mean concentration. Wetland and greenhouse samples were also collected after each day of the drawdown period, but consisted of a single grab sample, rather than a composite of 24

samples collected at one hour intervals. Since flow was not measured in the greenhouse or wetland, water quality analysis at these locations was based solely on constituent concentrations. All samples were submitted for analysis to the OMOE laboratory in Toronto immediately following collection. Analysis included all the major pollutant groups, including nutrients (N and P), metals, general chemistry, *E.coli*, TSS and phenolics.

Vegetation samples were collected from ten plots within the vegetated portion of the facility; four in the greenhouse and six in the wetland and forebay. Vegetation growth and density were monitored at these locations from April to November, 1997. In June and September, 1997 vegetation samples were collected from locations adjacent to the ten plots. Samples were weighed, dried and submitted as separate above-ground and below-ground samples for lab analysis. Vitality testing was performed on plant rhizomes collected from the greenhouse in January 1997.

Sediment samples were collected from a total of four plots in the greenhouse, wetland (2 plots) and forebay. A control location on the upper southern bank of the facility, above the high water mark, was also sampled. In December, 1996 and June, 1997, sediment samples collected from these plots were submitted for geochemical analysis. In August, 1997, samples from adjacent locations were submitted for bioassessment of toxicity and bioaccumulative potential.

Study Findings

Water Quantity

The Aurora wetland provided a level of runoff control intended to prevent erosion and protect aquatic habitat downstream of the facility. Peak flows were reduced by over 80% during most events. Stormwater drawdown extended over a period of 3 to 5 days and the mean hydraulic detention time was estimated to be 36 hours. Discharge rates in the downstream channel did not exceed 58 L/s, which was sufficient to contain flow within the banks of the channel.

Although the facility was originally designed based on a runoff coefficient of 0.41 (*i.e.* 41% of rainfall within the catchment enters the facility as runoff), flow data for 29 events indicated a mean runoff coefficient of only 0.21. A runoff coefficient of this magnitude is not atypical for a drainage basin with 30% agricultural land use.

Over the study period, water losses to groundwater during wet and dry weather accounted for approximately 13% of the total influent runoff. By comparison, evaporation over the same period accounted for only 0.7% of influent runoff. These quantities are based on the assumption that evaporation and water losses to groundwater are negligible during the winter. For the 29 rainfall events monitored over the study period, inputs (runoff, rain, groundwater discharge) to the wetland system were greater than outputs (outflow, evaporation, groundwater recharge) from the system by an average of 4.7%, which is within the expected error range of the monitoring instruments.

Water Quality

Effluent average event mean concentrations (AEMCs) and load-based removal efficiencies are summarized by season and for the entire study period in Table 1. The major findings regarding water quality were as follows.

Table 1: Summary of wet weather average effluent event mean concentrations (AEMC) and load-based removal efficiencies (R.E.) for selected parameters.

Parameter	Winter[1]		Spring		Summer		Fall		Study Period		PWQOs[2]
	Eff. AEMC	R.E. (%)	Eff. AEMC	R.E. (%)	Eff. AEMC	R.E. (%)	Eff. AEMC	R.E. (%)	Eff. AEMC	R.E. (%)	
General Chemistry											
TSS (mg/L)	28.7	46	16.8	90	25.6	91	26.0	87	23.8	86	
BOD (mg/L)	1.6	7	3.1	55	3.1	34	2.1	22	2.5	32	
COD (mg/L)	29.3	6	30.6	51	31.4	36	22.6	39	28.4	33	
Oil/Grease (mg/L)	1.5	25	1.4	82	1.4	44	0.7	78	1.2	61	
Chloride (mg/L)	429	-15	34	69	105	-51	56	24	45	-1	
Phenolics (µg/L)	1.4	21	1.3	48	1.8	-6	1.1	55	1.5	26	1
E. Coli (c./100ml)	252	42	319	68	475	94	1107	55	464	84	100
Nutrients											
TP (mg/L)	0.16	8	0.16	56	0.10	83	0.18	64	0.14	58	0.03
PO_4 (mg/L)	0.07	26	0.06	14	0.02	86	0.09	31	0.05	44	
TKN (mg/L)	0.88	36	1.19	52	0.81	63	0.79	51	0.90	49	
NH_3+NH_4 (mg/L)	0.05	50	0.17	79	0.03	52	0.05	40	0.06	63	
NO_3 (mg/L)	0.63	27	0.36	26	0.08	82	0.50	11	0.28	41	
Metals											
Copper (µg/L)	5.8	37	5.7	66	4.8	69	5.0	57	5.3	58	5
Zinc (µg/L)	63.8	-87	30.4	59	24.0	53	26.3	58	31.8	17	20
Chromium (µg/L)	2.5	20	1.2	50	1.0	58	1.8	32	1.5	36	8.9[3]
Iron (µg/L)	470	6	353	65	386	52	481	31	415	41	300

1) Mean concentrations and removal efficiencies over the winter period are based on grab samples collected near the end of the runoff event and, therefore, should be interpreted with caution. 2) Underlining indicates concentrations greater than Provincial Water Quality Objectives/Guidelines for receiving waters. 3) 1 µg/L in its less common hexavalent form (Cr VI).

- The load-based TSS removal efficiency was 86% over the study period, ranging seasonally from 46% during the winter to 91% during the summer. The poor removal efficiencies calculated for TSS and other pollutants during the winter may have been a result of low influent concentrations (average = 43 mg/L). Those concentrations may have been partly a consequence of grab sample collection, which would have missed high solids loading associated with the 'first flush' of runoff. Consequently, poor efficiency in winter should not be attributed exclusively to cold weather effects on removal mechanisms.

- Removal efficiencies for most other constituents were considerably less than that of TSS (Table 1). Nutrient removal ranged from 41% for nitrate to 63% for total ammonia, and metal efficiencies were mostly less than 60%. In general, removal efficiencies were greatest during warm weather conditions. However, part of this difference may simply reflect differences in the method of inlet sample collection during the summer and winter (*i.e* composite vs grab).). The effluent concentrations were similar during warm and cold weather.

- Detection frequencies were low in the effluent for several metals, including lead, cadmium, cobalt, nickel and molybdenum. Others such as zinc, copper and iron were often observed at concentrations exceeding receiving water standards (Provincial Water Quality Objectives/Guidelines - PWQOs).

- *E.coli*, TP and phenolics had average effluent concentrations above PWQOs.

- The average particle size distribution (PSD) of suspended particles over the study period indicated a substantial shift to finer particle sizes after the first day of treatment. Based on the samples analyzed, the average particle size of the influent was 3.4 μm (average of PSD medians), compared to the average effluent particle size of 1.7 μm on day 1, and 1.4 μm on day 4.

- Daily constituent concentration data over the treatment period indicated that TSS concentrations decreased substantially after the first day of treatment, but that daily average concentrations of most metals and nutrients displayed a more gradual decline. This gradual decline for the majority of pollutants highlights the importance of residence time in treatment performance. The results are also consistent with the hypothesis that the pollutants are either in dissolved form or associated with the finer suspended particles.

- The maximum water temperature in the wetland (24°C) during the summer was above the 21°C maximum recommended for cold water fisheries habitat. Unlike most wet ponds, the water temperature in the Aurora wetland increased only slightly from the inlet to the outlet, probably due to the small permanent pool and shading of the water column by a dense assemblage of plants.

- Greenhouse air temperatures were higher than the wetland by an average of 5°C, and the growing season in the greenhouse was 12 to 14 weeks longer than the wetland. However, substrate and water temperatures were similar, and the greenhouse failed to provide improved treatment relative to external wetland. This finding is likely explained by the strong influence of soil and water temperatures on root-zone biological removal processes. Also, physical and chemical mechanisms of removal are influenced by temperature and water viscosity.

Vegetation and Sediment Analysis

- The wetland was dominated by *Typha latifolia* (common cattail) and *Scirpus dominus* (bulrush). Vitality testing on both plants indicated that, during the winter, active metabolic functioning was occurring in the

below-ground rhizomes, even though the above-ground tissues were dead. In the greenhouse, these plants started growing and reached their maximum height approximately six weeks earlier than in the external wetland.

- Biomass analysis of above and below-ground tissues for the entire wetland indicated a decrease in the below-ground to above-ground dry weight ratio from 1.3 in June to 0.5 in September. The greenhouse and wetland plants were similar, except that the greenhouse above-ground tissues had slightly higher moisture contents.

- Tissue chemistry analysis showed that, as biomass increased from June to September, the concentrations of most pollutants in the tissues decreased. In terms of mass, however, all nutrients and metals showed significant increases in both above and below-ground tissues over the summer. TKN and TP masses tripled from June to September. The nutrient masses in the plants represented a much greater proportion of influent loads to the wetland than did the metals.

- With the exception of TP, arsenic and chromium, concentrations of pollutants in wetland sediments were greater than concentrations at a nearby control site. However, among pollutants analyzed, only copper had concentrations that exceeded the lowest effect tolerance level for benthic invertebrates.

- Wetland sediments were found to be of good quality based on mayfly, midge and minnow lethal and sublethal endpoints, chemical uptake and bulk sediment chemistry. Forebay sediments showed moderate to severe levels of growth reduction to mayfly nymphs and midge larvae, probably due to high concentrations of PAHs (11 μg/g) in the sediments.

Conclusions and Recommendations

The Aurora facility is unique among stormwater treatment wetlands in that it did not support a substantial permanent pool and would drain dry during periods of infrequent rainfall. As such, the Aurora wetland design did not meet with provincial criteria for this type of facility. Nevertheless, study results indicate that the facility provided good hydraulic control of peak flows and adequate levels of treatment for most pollutants. Extending the time over which stormwater was detained within the facility through modification of the outlet structure was considered to be a key factor in pollutant removal.

The greenhouse did not provide improved cold season performance relative to the wetland, probably because water and substrate temperatures in the greenhouse and wetland were similar. Higher greenhouse air temperatures and a significantly extended growing season were not, by themselves, sufficient to enhance greenhouse treatment capacity. This result is attributed to the key role of root-zone biota in plant removal processes, and the close dependence of root-zone biota on water and soil temperatures, as well as the effect of water temperature on physical and chemical mechanisms of pollutant removal.

Recommendations for facility improvement, maintenance and future research are as follows.

Maintenance and Safety Issues

A considerable amount of trash and debris was carried into the facility during the study period. This trash and debris should be cleaned out at least once a year.

Signs were placed in the facility during the study period warning of human hazards related to fluctuating water levels. Further restrictions on human access to the flooding zone could be achieved by the use of dense perimeter vegetation or fencing with natural materials. Well defined trail grids outside of the flooding zone would help to retain the public amenity function of the facility.

Facility Improvement

Runoff events during the winter are less frequent and intense compared to the summer, but can be large and extend over long periods of time. Treatment of stormwater under this type of runoff regime can be improved by lengthening the cold season detention period through temporary alterations to the outlet control structure. Due to the longer interevent periods and slower release of runoff during the winter, longer extended detention periods may be possible without compromising the water quantity control function of the facility.

If development expands beyond the 58 hectare area documented in this study, some modifications may be warranted such that the facility more closely approximates OMOE design criteria for constructed wetlands. These modifications may include the following: (i) deepening the forebay to the 1 m depth recommended in the SWMP manual; (ii) increasing the length-to-width ratio from 2:1 to 3:1 by moving the forebay closer to the inlet; (iii) reconfiguring the basin such that it retains a 35 cm permanent pool; and (iv) increasing the total depth of extended detention to 1 m, or up to the bottom of the existing outlet weir through installation of a Hickenbottom riser device similar to that installed as part of the study.

Wetlands provide food for several species of mammals and water fowl as well as habitat for amphibians. It was often observed during periods of flooding that waterfowl used the wetland area to forage. Since waterfowl can contribute significantly to the fecal matter in the facility, in addition to stirring up bottom sediments, waterfowl use should be restricted by, for example, providing dense vegetation in open areas.

Future Considerations

Once plant uptake and sediment adsorption pools in the Aurora wetland approach a limit (as defined by site specific environmental conditions and loading regime), removal rates may suffer substantial declines. Deterioration of performance in this regard may indicate the need for dredging, especially in the forebay, and natural or artificial re-establishment of vegetation cover. Short-term monitoring of system performance should be conducted every 3 to 5 years to determine whether maintenance for this purpose is warranted.

The Aurora facility was a dry pond converted to a wetland, but also functioned partly as an infiltration basin, with average water losses to groundwater accounting for 16% of influent runoff during the summer. These

losses helped to further reduce pollutant loading to receiving waters. Enhancing the infiltration component of stormwater wetland facilities should be considered when meeting OMOE permanent pool volume guidelines is not practical, and the potential for contamination of groundwater resources is low.

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1.0 INTRODUCTION

Natural wetlands have long been viewed as providing important ecosystem functions in flood control, water purification, wildlife habitat, carbon cycling and sediment control (Mitsch and Gosselink, 1993).[1] Recognition of these natural ecosystem services has led to the engineering and construction of wetlands designed to enhance and simulate wetland functions of relevance to the treatment of wastewater and stormwater. Over the past decade, the practice of constructing wetlands for water treatment has become well established and is described in the Ontario Stormwater Management and Practices Design Manual (OMOEE, 1994a) as one of the most effective best management practices for stormwater quantity and quality control.

Although constructed wetlands for stormwater treatment are currently in wide use, this technology was not always so well accepted. In 1989, a survey of Ontario municipalities and provincial officials on constructed wetlands (CWs) for stormwater treatment revealed mostly negative responses toward what was perceived to be an unproven or infeasible technology (Carlisle *et al.*, 1991). This perception shifted considerably during the early 1990's. A follow-up survey conducted in 1993 found that, in contrast to the earlier survey, a large majority of respondents favoured the use of CWs for stormwater management (Bennett, 1994). The significant change in attitude was thought to be largely the result of recent findings on the effectiveness of wetland treatment in the United States.

Although attitudes toward the use of CW's for stormwater treatment were changing, monitoring data upon which to assess the performance of these facilities remained very limited. Further, there was no published information on the year-round performance of CWs used for stormwater treatment in Ontario. Most of the monitoring studies of CWs for stormwater treatment have been conducted in the warmer climates of the United States and are, therefore, of limited relevance to Ontario, where winters are colder and seasons more pronounced. This monitoring study of a stormwater wetland in Aurora, Ontario was an attempt to fill some of these knowledge gaps, and provide a demonstration site against which other stormwater wetland designs could be evaluated.

The study site was designed and constructed in 1988 as a water quantity dry pond, but it evolved into a wetland as moist conditions attracted aquatic plants. Modification to the outlet structure prior to the study created an extended detention capacity, which maintained a pool of standing water on an intermittent basis, and resulted in a drawdown period of 3 to 5 days after rain events. The wetland included an experimental greenhouse, with a mechanical furnace and electrical lighting, that was intended to sustain wetland performance during the cold season and provide a standard of comparison with treatment efficiency in the adjacent wetland area.

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[1] Terms used in this study are defined in Appendix B. A brief review of the literature on stormwater wetland treatment systems is provided in Appendix C.

This study was undertaken in two phases. Phase I was conducted by researchers from the University of Waterloo from May 1996 to June 1997. This phase focused on the hydrology and treatment performance of the wetland and greenhouse facility. Phase II was conducted by the SWAMP program from July 1997 to June 1998. The Phase II monitoring program was similar to that of Phase I, but included analyses of sediment, benthic invertebrates and vegetation (tissue chemistry) within the wetland. This report integrates the results of Phases I and II into one comprehensive report. Both phases were funded by the Ontario Ministry of the Environment (formerly the Ontario Ministry of Environment and Energy) and the Government of Canada's Great Lakes 2000 Clean-up Fund (superceded by the Great Lakes Sustainability Fund).

1.1 Study Significance and Objectives

Many of the physical, chemical and biological processes at work in wetlands are inhibited by lower temperatures and shorter photoperiods. However, the effect of these inhibiting factors on the performance of CWs used for stormwater treatment through the cold season is not known. Further, little local information exists on the contribution of wetland vegetation to water quality improvement and the toxicity of stormwater facility sediment.

The overall goals of the study were to address these research needs and provide a comprehensive evaluation of constructed wetlands as a technology for stormwater treatment. Specific aims of the study were to:

- assess and compare the seasonal performance of an open and covered wetland treatment system;
- evaluate the capacity of a heated greenhouse constructed within the wetland to enhance performance during the cold season;
- investigate the contribution of vegetation to water quality improvement;
- assess sediment quality and biological effects through sediment chemistry analysis and sediment toxicity tests;
- document operation and maintenance requirements for the wetland area, and
- provide recommendations for the improvement of wetland treatment technology for stormwater management under Ontario climatic conditions.

2.0 STUDY SITE

2.1 Site Selection and Approvals

The search for and selection of an appropriate site at which to conduct the research was completed during September, 1995. The selection process included site visits to 20 prospective stormwater wetland facilities in south-central and southwest Ontario. Site selection criteria included the following:

- location within commuting distance of the OMOE laboratory in Toronto and the research assistant's home in Aurora, Ontario;
- design with a sediment forebay and wetland and capacity for extended detention of stormwater;
- direct road access, utility services and security fencing;
- acceptable to the local community, municipality and conservation authority.

The site chosen in Aurora, Ontario (Figure 2.1) met all of these criteria.

Based on initial contacts, the municipal site approval process included endorsement by several external agencies, including the Ministry of Natural Resources, Coscan Developments (who maintained the facility) the Lake Simcoe Region Conservation Authority and the local community, particularly that of the adjacent housing co-operative. The municipal approval process, pending endorsement by the agencies, included initial discussions with the staff of the Town of Aurora, project proposal to the Town's Leisure Services Committee, arrangement of legal agreements regarding liability, performance bond issuance and guarantee of facility removal upon completion of the study, Town Council approval and the issuance of a building permit. This approval process occurred between September 27 and November 22, 1995.

2.2 Aurora Stormwater Management Facility

2.2.1 Catchment area

The catchment area draining to the Aurora wetland is 82.4 ha, of which 24.3 ha is in rural agricultural use and 58.1 ha is residential development. The 24.3 ha agricultural block is located to the west of Bathurst Avenue (Figure 2.1), runoff from which drains into the stormwater basin via two culverts beneath Bathurst Ave. The 58.1 ha subdivision area is in mixed residential use and consists of 37.8 ha medium density residential development, 3.6 ha high density multiple family, 0.3 ha commercial, 2.6 ha institutional, 4.4 ha park and open space and 9.4 ha road networks (MMM, 1988). Development of the subdivision area began in 1988 and was completed before this study was initiated. The area-weighted average runoff coefficient used in the stormwater calculations of the agricultural and urbanized areas of the catchment was 0.41

The development is located on the north-draining slope of the Oak Ridges Moraine complex and drains toward Lake Simcoe. The area is at the headwaters of Tannery Creek, which is a subwatershed of the East Holland River, draining to Lake Simcoe. Habitat in this area is classified as Type 2, defined by the Ontario Ministry of Natural Resources (1994) as feeding areas, areas of unspecialized spawning habitat, and areas where pool-riffle-run complexes occur along much of the watercourse. Pre-development terrain was rolling with knolls and basins, descending in a generally northerly direction (Soil-Eng, 1988). Pre-development land use was rural and agricultural.

A soils investigation was conducted for the subdivision development based on 29 boreholes throughout the catchment area (Soil-Eng, 1988). Beneath a topsoil veneer, there exists a strata of sandy silt and clayey silt, intermittently emplaced with a stratum of sandy silt till. At one location a lens of silty gravel was detected. The soils have estimated coefficients of permeability ranging from 10^{-4} to 10^{-6} cm/s (Freeze and Cherry, 1979). Materials in this permeability range are considered relatively pervious to relatively impervious. Signs of soil oxidation at 3.5 to 14.1 m below the surface indicated that the permanent groundwater level lies below the investigated depths, although perched groundwater tables may form temporarily during wet periods. The clayey–silt soil of the basin has a mean natural water content of 15% (Soil-Eng, 1988) and an estimated saturation water content of 35%.

2.2.2 Facility design, vegetation and outlet structure modifications

The stormwater management facility was originally constructed in 1988 as a water quantity dry pond. The 1.1 hectare facility consists of a small sediment forebay (permanent pool = 40 m^3) and a naturally colonized wetland (Figure 2.2) with extended detention provided by an outlet control structure (Figure 2.3), modified for the purposes of this study. Stormwater is discharged from the facility via an intermittent stream through a woodlot to a small pond known as Salamander Pond. This woodlot, like all woodlots in the Town of Aurora, is designated in the Town's Official Plan as an Environmentally Sensitive Area. The facility was designed to store 18,000 m^3 of water at a maximum depth of 3.0 m for a four hour, 1:100 year storm event, with drawdown occurring over a period of less than 6 hours (MMM, 1988).

The facility's two major inlet sewers are 1350 and 900 mm in diameter. The 1350 mm pipe services 49 ha of the urbanized catchment area and the 900 mm pipe collects stormwater from 9 ha of urbanized area and most of the 24 ha agricultural area to the west of the facility. Other minor sources of stormwater runoff to the facility include a 305 mm storm pipe draining a small high density development east of the facility and a 610 mm culvert draining roadside ditch into the facility on Bathurst Ave. The facility is designed to have a 0.8% longitudinal slope and 2.0% bottom crossfall, with side slopes ranging from 5:1 to 3:1.

Note: This photo was taken in 1999. The land outside of the study area was less developed at the time of the study in 1997/98.

Figure 2.1: Study area

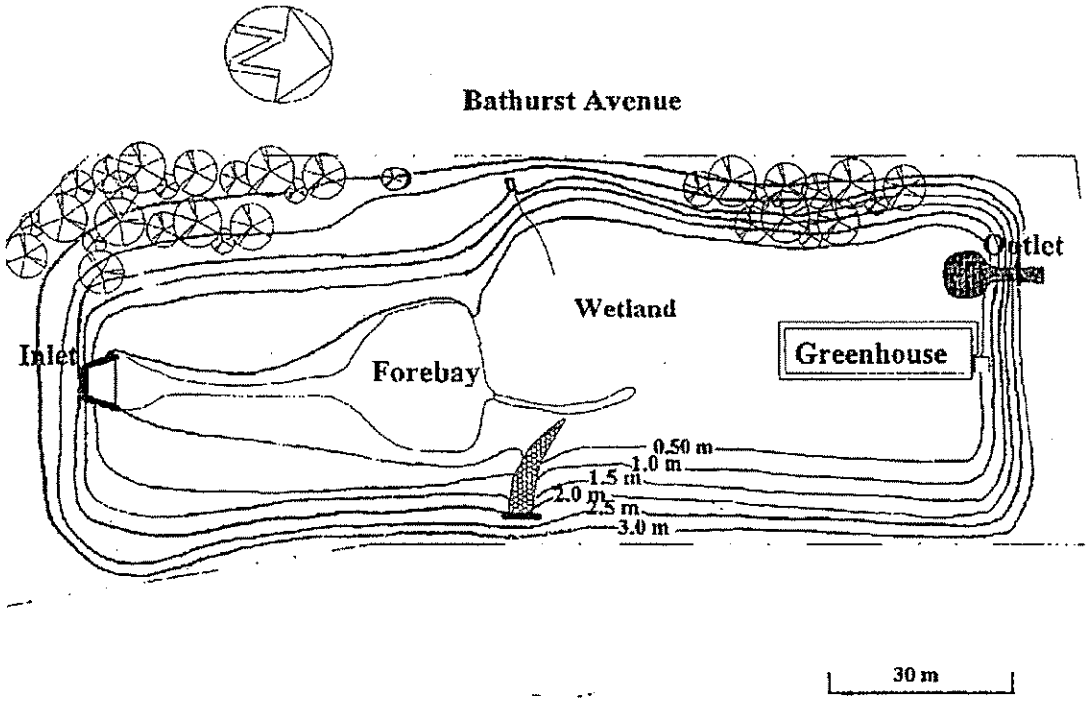


Figure 2.2: The Aurora wetland showing elevations relative to the basin bottom.

Discharge from the facility is controlled through a 457 mm corrugated pipe set approximately 1 m below the basin floor in a cobble-lined depression and a compound rectangular step-weir elevated 2 m above the invert of the discharge pipe (Figure 2.3). These discharge structures provide for rapid basin drainage. In order to extend the detention period of the basin, a single orifice Hickenbottom type riser structure was designed from a section of 457 mm corrugated steel pipe and fitted to the 457 mm discharge pipe. The riser was intended to extend the discharge over four days, which was considered to be adequate for stormwater treatment and fairly representative of the mean interevent period for this area (Gietz, 1983).

The maximum height of the riser was based on the water depth limitations of the wetland plants in the facility. Vegetation in the basin consists of two naturally colonized macrophytic species of wetland plants; *Typha latifolia* (common cattails) and *Scirpus validus* (soft-stem bulrushes). These plants prefer shallow water depths and a fluctuating hydroperiod. *Typha sp.* prefer water depths between 25 and 65 cm, but can withstand depths up to 100 cm for short durations (Thunhorst, 1993; Noller *et al.*, 1994). Neill (1990) found that optimal *Typha* biomass increase occurs when water depths are between 0 and 20 cm, rather than under more deeply flooded 20 to 40 cm depths. Based on these and other similar studies, the riser was designed to

the 53 cm basin depth level, with a 4-day drawdown period, providing adequate flooding over a short hydroperiod. On the shoreline of the basin above the 53 cm level, vegetation consisted of a mixture of grasses, thistles, coniferous and deciduous trees (pine, spruce, maple and oak).

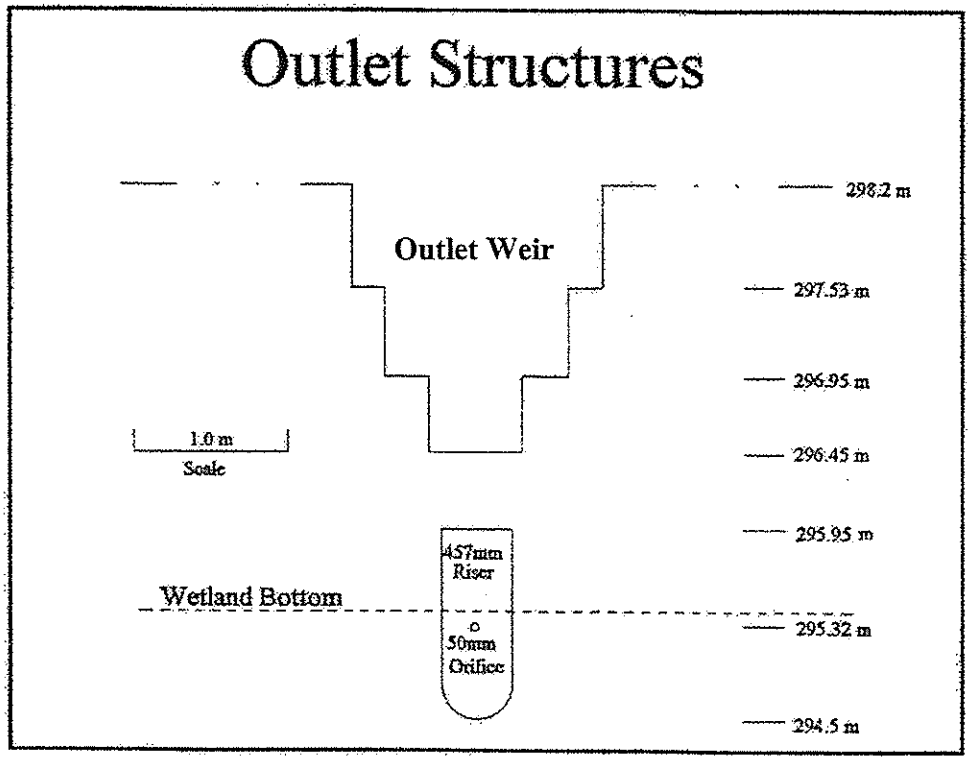


Figure 2.3: Stormwater management facility outlet structures including riser pipe and rectangular step-weir.

The riser structure controls discharge below the 53 cm basin depth level via a single circular orifice 5 cm in diameter (Figure 2.3). The orifice was located 10 cm below the approximate basin floor elevation to ensure complete facility drawdown and greater active storage. Therefore the total effective head from the top of the riser was 63 cm. The diameter of the orifice was sized such that the basin would drawdown from 63 cm to 0.0 cm depth in approximately 96 hours ($Q \approx 3.7$ L/sec). Size calculations were determined from the following freefall discharge relationship derived from Bernoulli's equation (Bedient and Huber, 1988; Daugherty *et al.*, 1985; Wilson, 1990).

$$Q = CA(2gh)^{0.5} \qquad \textit{Equation 1}$$

where:

Q = orifice discharge (0.0037 m^3/s)
C = the discharge coefficient = $C_c C_v$ = 0.647. (Coefficient of contraction (C_c) = 0.66; Coefficient of velocity (C_v) = 0.98)
A = orifice area (m^2)

g = acceleration due to gravity (9.81 m/sec^2)
h = head, distance between free-water surface and orifice (0.43 m = head level at 650 m^3, when the basin is half full)

Discharge through the open mouth of the riser above the 53 cm depth level was calculated in a similar manner as discharge through the orifice.

Stormwater discharged through the step weir at water levels greater than 2 m above the 457 mm outlet pipe (Figure 2.3). Discharge via the step weir was calculated from water level data and the following free-fall rectangular weir equation (Viessman and Lewis, 1996; Wilson, 1990; Bedient and Huber, 1988, Ferguson and Debo, 1990; Daugherty *et al.*, 1985):

$$Q = C\,L\,H^{1.5} \qquad \textit{Equation 2}$$

where:

Q = weir discharge (m^3/s)
C = discharge coefficient (1.84 for free weir discharge) (m/s)
L = length along the weir crest (m)
H = head (m)

The stage:discharge:volume relationship for the modified outlet structure is illustrated in Figure. 2.4.

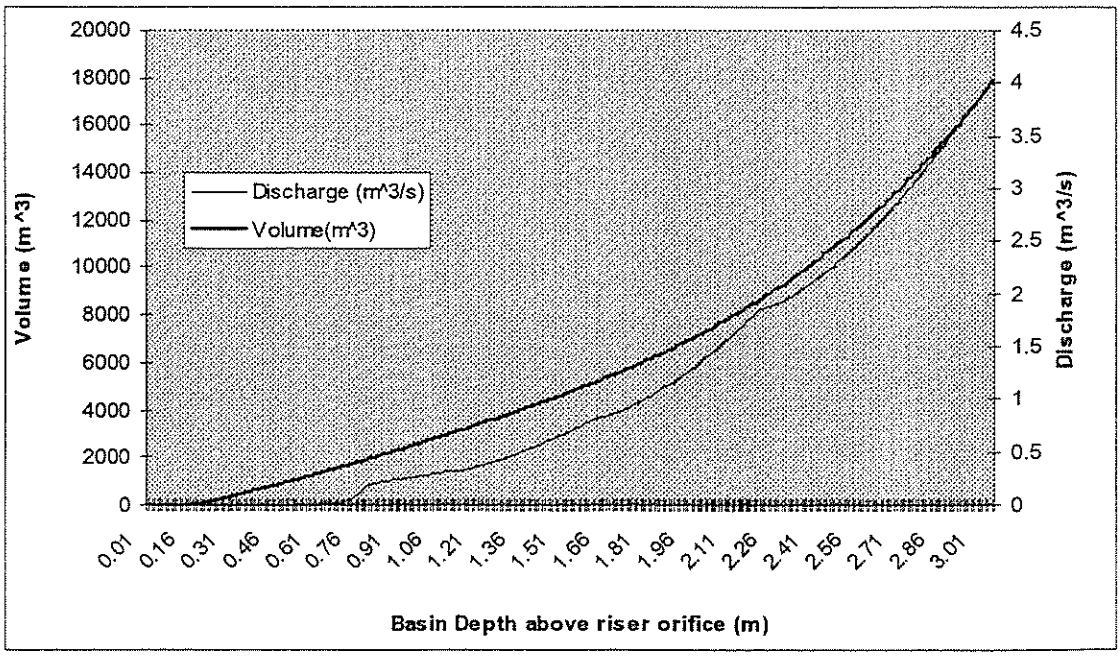


Figure 2.4: Calculated stage:discharge:volume relationship for the modified outlet structure.

Note: due to comparatively low flow rates from the riser, discharges below 0.63 m are not in view. Inflection points in the discharge line represent the elevations at which the riser mouth is fully covered, and the step levels in the weir. The original design discharge follows a similar line to the modified discharge at depths greater than 1 m. At depths below 1 m, discharge declines more gradually to zero.

Table 2.1 compares design features of the modified facility to Ontario constructed wetland design guidelines for the level of aquatic habitat protection (level 2) deemed to be appropriate for this site (OMOEE, 1994a). As recommended, the facility included a sediment forebay and wetland, and provided extended detention sufficient for drawdown over a period of four days. The detention time, defined as the time delay between inlet and outlet hydrograph centroids, was calculated prior to the weir modifications to lie within the range of 1 to 1.5 days. However, the permanent pool in the forebay, which helps to extend the settling period and minimize resuspension of solids, was very small, and the 2:1 wetland length-to-width ratio was less than the 3:1 ratio recommended in the SWMP manual (OMOEE, 1994a) for this type of facility.

Table 2.1: OMOEE (1994a) design guidelines compared to Aurora Wetland design features

Design Feature	**Design Objective**	**OMOEE (1994a) Guidelines**	**Aurora Wetland Facility**
Permanent pool depth (m)	minimize resuspension; avoid anoxic conditions	1-2 (mean); 3 (max.)	less than 1
Permanent pool volume (m^3/ha)	provision of 'Level II' habitat protection	20*	less than 1
Max. extended detention depth (m)	storage & flow control	1	approx. 0.5
Extended detention volume (m^3/ha)	provision of 'Level II' habitat protection	40	16**
Drawdown time (hours)$^{+}$	suspended solids settling	24	96^{++}
Length-to-width ratio	minimize short circuiting	at least 3:1	2:1 (wetland); 4:1 (facility)
Planting strategy	safety, aesthetics, nutrient uptake, filtration of stormwater, reduction of flow velocities to promote sedimentation	five zones – deep water, shallow water, shoreline, flood fringe, upland	natural regeneration of: (i) aquatic plants (esp. cattails and bulrushes); (ii) herbs, grasses and trees on shoreline.

*based on 'Level II' aquatic habitat protection and 35% surface imperviousness (OMOEE, 1994a)
**24.2 m^3 per hectare of developed area (53.7 ha.) within the catchment (82.4 ha.)
$^{+}$ The SWMP manual (OMOEE, 1994a) suggests using 'drawdown time' as an approximate measure of 'detention time'.
$^{++}$ estimated using a drawdown equation for the modified discharge structure. The 'hydraulic detention time' was estimated to be between 1 and 1.5 days for a 25 mm, 4 hour storm.

2.2.3 Greenhouse design

The greenhouse was constructed from December, 1995 to July, 1996. The greenhouse was located near the outlet of the facility, had a length-to-width ratio (4.3:1) similar to that of the facility (4:1), and covered 10%

(or 210 m^2) of the wetland basin area. Vegetation within the greenhouse was similar to that of the wetland. The greenhouse consisted of a structural base, upper frame and equipment shed.

The greenhouse base was 1.5 m high (5 year storm depth) and extended 15 cm below the wetland bottom to avoid cross-mixing of water between the wetland and greenhouse. The base was anchored to the ground with 80 steel 1 to 2 m T-bars. Water entered and exited the greenhouse through 0.3 m wide, 1.4 m high openings fitted with spring-loaded one-way galvanized metal gates. The openings were sized using a rectangular weir discharge equation (equation 2 above) to permit adequate inflow and minimize water level differences between the greenhouse and wetland during storm events.

The greenhouse was manufactured by DeCloet Greenhouse Manufacturing (Simcoe, Ontario) to design specifications outlined in the Ontario Farm Building Code. The greenhouse extended 3.5 m above the wetland bottom and 2 m above the 1.5 m base. During the warm months, ventilation was provided by a 0.9 by 1.8 m aluminum and plexiglass access door and motorized shutters (0.3 m^2) with fans installed at each end of the greenhouse. During the cold season, the greenhouse was heated with a natural gas greenhouse furnace to maintain air temperatures at 10°C. Lighting consisted of four 2.5 m double-tubed, 340 watt fluorescent lights timed to augment natural lighting (*i.e.* at least 13 hours of light) during fall, winter and spring. An equipment shed (2.5 x 2.5 m) was constructed adjacent to the north end of the greenhouse to house the main electrical panel, sampling and maintenance equipment. Construction and operation of the greenhouse from July 1996 to June 1998 cost just over $45,000.

3.0 STUDY METHODS

Water quantity and quality data were collected from July 1, 1996 to June 30, 1998. The selection of physical, chemical and biological parameters for monitoring were based on the original objectives of the study and a careful review of parameters reported by other stormwater wetland performance assessments (see Appendix C). The vegetative and sediment analysis components were added to the second year of monitoring with the aim of better characterizing contaminant pathways within the wetland treatment unit.

3.1 Water Quantity Monitoring

The water quantity monitoring set-up is illustrated in Figure 3.1. The monitoring program consisted of co-ordinated measurements of precipitation, evaporation, groundwater flux, and stormwater depth and

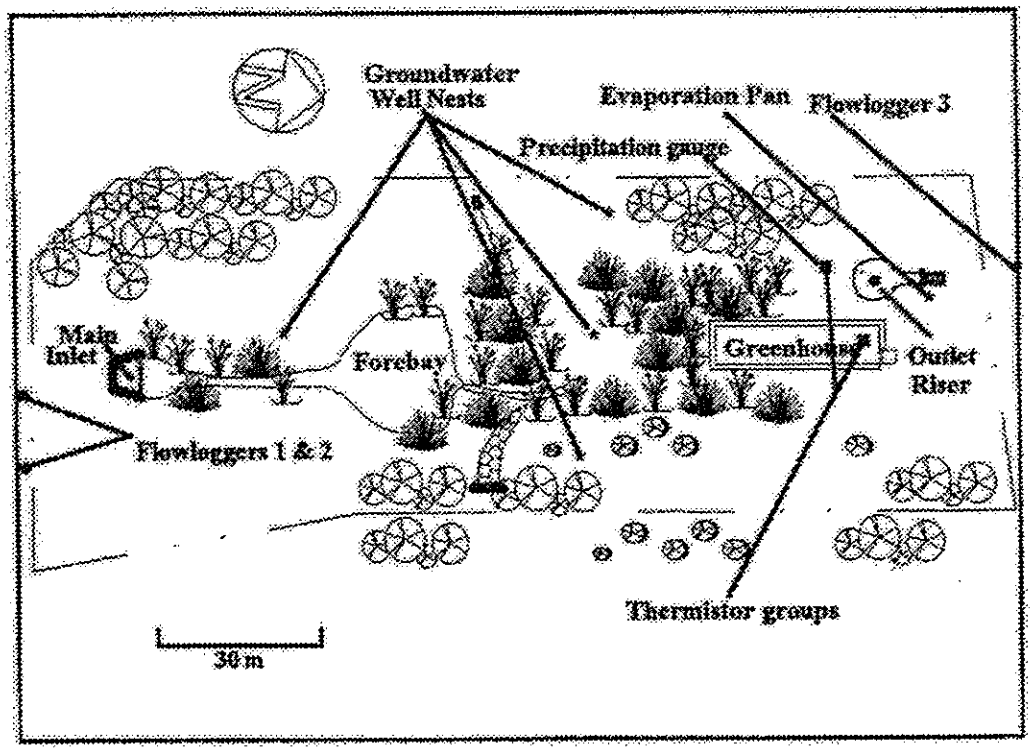


Figure 3.1. Location of hydrological and temperature monitoring equipment.

flow. The water balance for the facility for each storm event was calculated as,

$$P_d + R_i + G_d = R_e + G_r + E_t \qquad \textit{Equation 3}$$

where,

P_d = direct basin precipitation
R_i = influent stormwater runoff
R_e = effluent stormwater runoff
G_d = groundwater discharge to the basin
G_r = groundwater recharge from the basin
E_t = basin evapotranspiration

These water inputs and outputs are illustrated in Figure 3.2. Methods of measurement or estimation for each of these components are described in detail below.

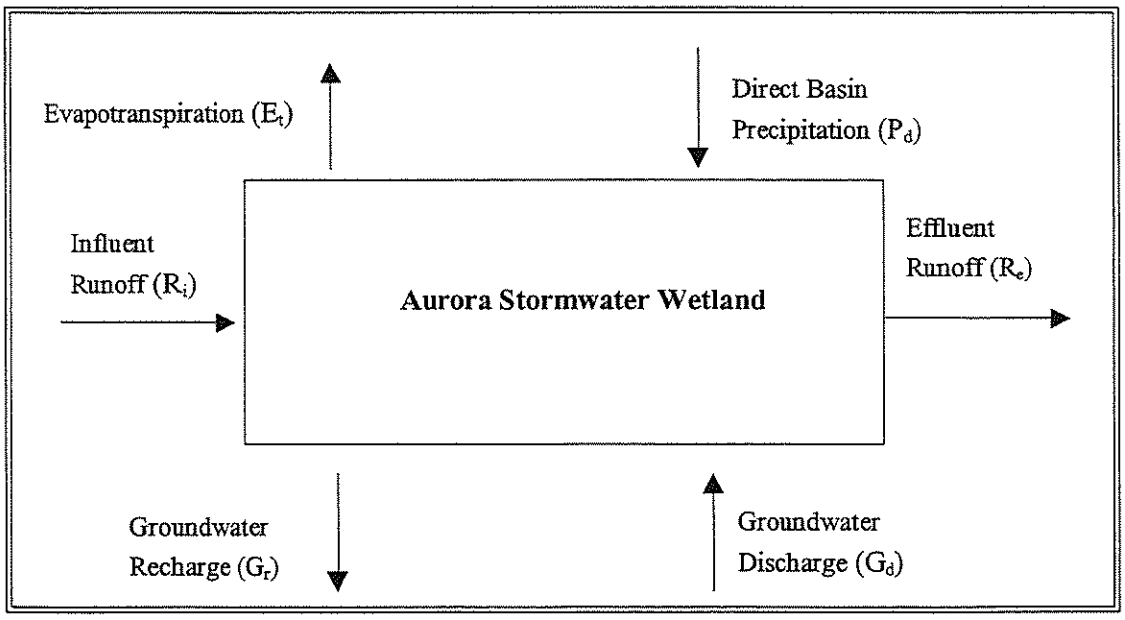


Figure 3.2: Wetland water balance components

3.1.1 Precipitation

Precipitation was determined from two manual rain gauges and an automated tipping bucket rain gauge located within the facility. Snowfall was measured from manual daily depth measurements and snowfall

accumulation in a snow collection cone. Water equivalents of daily accumulations were determined by melting the snow.

Total event precipitation was measured as cumulative precipitation from the beginning of the event to the end of the drawdown period. Seasonal precipitation was the total liquid equivalent of all precipitation events occurring over a given season.

3.1.2 Basin stormwater influent flow volumes

Stormwater runoff in the two inlet sewers was measured continuously at 5 minute recording intervals with two Montedoro-Whitney SonicStar Q-Logger flow recorders. Total inlet runoff volume was calculated as the sum of discharge from both major inlet pipes over the course of the storm event. During winter and early spring when monitoring equipment was removed due to freezing conditions, runoff was estimated from the depth volume and outlet discharge rate of the facility. Discharge volumes into the facility from a townhouse development to the east and from a section of Bathurst Avenue to the west were negligible relative to the total catchment area, and hence runoff from these sources was not monitored.

3.1.3 Groundwater

Groundwater well nests (or piezometers) were used to estimate groundwater discharge to and recharge from the facility between storm events. The location of well nests is shown in Figure 3.1. The wells were constructed from ¾" diameter PVC pipes and set in triangular nests imbedded at 0.75, 1.0 and 1.25 m below the surface. One nest located in the center of the wetland had well intakes set at only two depths (0.5 and 0.75 m) because a consolidated gravelly till prevented deeper augering. The lower 25 cm of all the pipes was perforated and wrapped in 200 μm mesh geotextile. Measurements were taken several times weekly using a water level indicator. The elevation of the piezometers were surveyed and water levels were determined relative to a common benchmark. Groundwater levels were not recorded during runoff events, as the water levels in the wells were that of the ambient basin level when the ground was saturated.

The loss to groundwater (or groundwater recharge) over a runoff event or season (G_r) was determined as the sum of capillary (C_{gw}) and head-induced (H_{gw}) groundwater losses. The capillary groundwater loss represents the volume recharged during the initial part of the runoff event to the soil beneath the basin floor, and was calculated as,

$$C_{gw} = D_s\,(0.20)(2800) \qquad \textit{Equation 4}$$

where:

D_s = water table depth prior to the flooding event or total for a season (m)
0.20 = void space (difference between natural water content and saturation).
2800 = average area of the wetland basin that was flooded (m^2)

The following equation was used to estimate the flood induced groundwater recharge occurring below the flooded zone.

$$H_{gw} = (H_c)(D_r)(0.20)(2800) \qquad \textit{Equation 5}$$

H_c = Head coefficient: a multiplier based on mean basin water depth during the runoff event or season. It was calculated by using the head discharge equation (eqn 1) to determine discharge at the mean basin depth and dividing that value by the discharge at 1 cm depth.
D_r = mean basin water depth for an event or season (m)

3.1.4 Basin depth and volume

The stage:storage relationship provided in the original stormwater management report for the facility (MMM, 1988) was used to estimate the relationship between basin depth and volume. The relationship was judged to be accurate at the beginning of the study based on automated influent flow data and daily depth measurements obtained from a measuring rod attached to the outlet riser.

3.1.5 Evapotranspiration

Evapotranspiration was derived from daily pan evaporation measurements using a 'Class A' evaporation pan set up at the north edge of the basin (Figure 3.1). The pan was filled manually and water losses were measured daily. Class A pan evaporation measurements are known to overestimate open water and wetland evapotranspiration. Pan coefficients ranging from 0.7 for small wetlands to 0.9 for large wetlands have been suggested (Kadlec *et al.*, 1987). Since the Aurora wetland is relatively small, a pan coefficient of 0.7 was employed to convert measured evaporation to actual wetland evapotranspiration. The volume (m^3) of evapotranspiration (E_t) was calculated as:

$$E_t = P_e\,(0.7)(2800) \qquad \textit{Equation 6}$$

where:

P_e = Pan evaporation measured daily (m)
0.7 = small wetland pan coefficient
2800 = The average area of the wetland basin that was flooded (m^2)

3.1.6 Basin stormwater effluent flow volumes

When water levels in the facility were below 1 m, outflow from the facility was measured solely by a flow recorder identical in type and programming to those at the main inlet pipes. The flow logger and area-velocity probe were located about 15 m downstream of the outlet pipe in a section of smooth-walled pipe installed to ensure laminar flow conditions. At water levels greater than 1 m, water flowed over the step weir (Figure

2.3). Weir discharge volumes were estimated from an automated level sensor installed at the outlet and the weir discharge equation discussed in section 2.2.2. During the first phase of the study (until May 1997), depth measurements were performed manually during runoff events by a researcher living immediately adjacent to the study site. The total estimated weir discharge for the precipitation event was then added to the flow calculated from the outlet flow monitor.

3.2 Temperature Monitoring

Temperature measurements of ambient air, water and upper substrate environments were made within and outside the greenhouse (Figure 3.1). These measurements were made using thermistors connected to a digital thermologger. The thermistor recording ambient air temperature was located outside the greenhouse in a Stevenson screen and within the greenhouse in a white shielded cone. Both devices were set at approximately 2 m above the basin bottom to avoid problems associated with frequent basin flooding. Water temperatures were recorded with the thermistors mounted to pegs at approximately 10 cm above the wetland floor. Upper soil substrate temperatures were measured by thermistor probes imbedded to a depth of 15 cm below the surface. Additional measurements of daily air temperatures inside and outside the greenhouse were recorded manually during morning site visits.

3.3 Water Quality Monitoring

3.3.1 Selection of sampling events

Storm events were selected for sampling based on: (i) the study aim of observing water quality change over a 4-day drawdown period, and (ii) a limit imposed by funding constraints of 16 events per year (or 4 per season). Since the 4-day drawdown period occurs for storms that generate influent runoff of at least 1300 m^3 (basin water levels of at least 53 cm), only storms with runoff greater than this volume were selected. In order to discount differences related to the distribution of rainfall, discrete events with little or no precipitation from the day after the event to the conclusion of sampling four days later were also favoured in the event selection process.

3.3.2 Water quality sampling locations and collection methods

Figure 3.3 shows the location of sampling stations within the facility. During the spring, summer and fall, samples at the inlet were collected by an ISCO 2700 wastewater sampler, which drew samples through 3/8" PVC tubing from an intake strainer strapped to the bottom of the pipe. The sampler was located 5 m inside the outfall of the 1350 mm main inlet pipe. To avoid sampler triggering during small runoff events the sampler was set to sample only after a flow-depth of 13 cm was reached. Once triggered, the sampler was programmed to collect a set of up to 24 flow–weighted samples. Each sample took about 4 minutes to collect including sample line back-purging. The frequency of sampling was proportional to measured flow. When flow rates in the inlet pipe were greater than 20.8 L/s, which occurred frequently during the runoff period, the

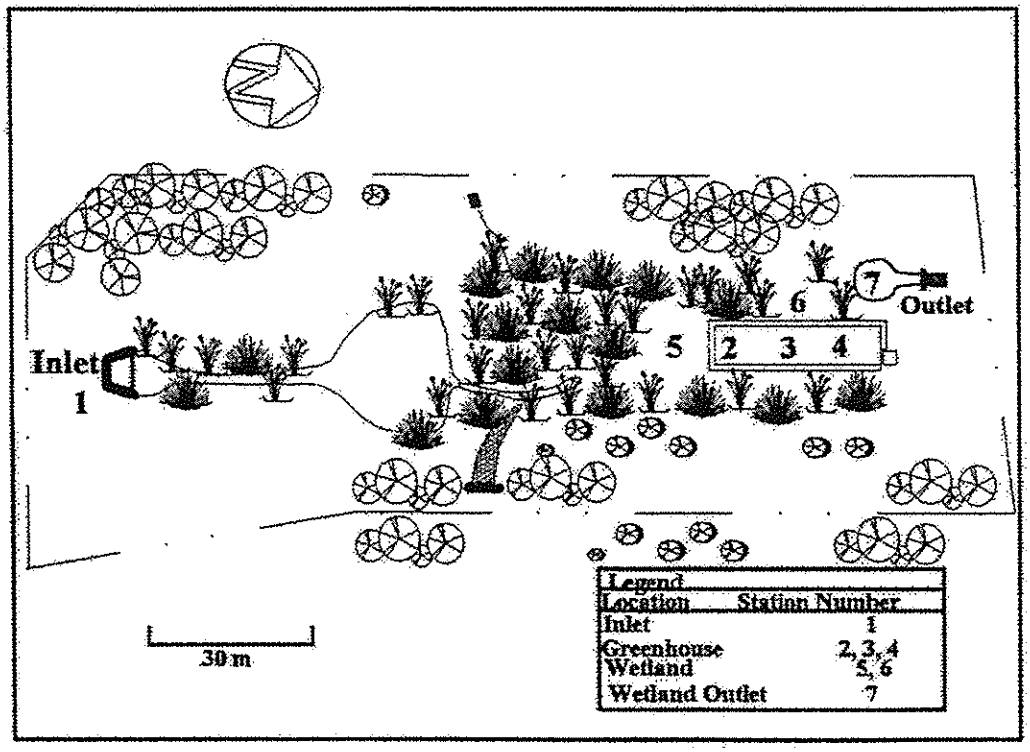


Figure 3.3: Location of water quality sampling points in the stormwater facility.

sampler collected continuously (*i.e.:* the 4-minute pump and purge cycles were contiguous). Since samples collected in this manner gave a flow-weighted sample profile of the storm, the composite of these samples was considered to represent the Event Mean Concentration (EMC). During the winter, when automated sample collection at the inlet was not possible, samples were composited from three 2 L grab samples collected from the inlet outfall (1 grab sample) and sediment forebay (2 grab samples) near the end of the runoff event. Since grab samples may not represent water quality over the entire event, winter performance and influent concentration data should be interpreted with caution.

Sampling stations 2, 3 and 4 were located within the greenhouse at its inlet, midway point and outlet, respectively (Figure 3.3). These stations were used to identify spatial differences in water quality in the greenhouse. Each station included a sample strainer at the intake point to prevent clogging by large debris and was housed inside a 50 cm section of 1.5" ABS pipe, which acted as a protective sleeve to avoid excessive water turbulence when the sample line was back–purged prior to sample collection. The strainer and ABS sleeve pipe were mounted to a wooden block 1.5" thick, which in turn was mounted to a 50 x 50 cm piece of plywood. The plywood was pegged to the bottom of the wetland with the strainer device attached. Samples were collected from each station through clear 3/8" PVC tubing which ran to the equipment shed

where a second ISCO wastewater sampler was located. Single, manually triggered, 5 to 6 L samples were collected from stations 2 to 4 every 24 hours for 4 consecutive days after the storm event. These were not time or flow weighted composite samples, but were single daily collections taken at 24 hour intervals after the storm event. Samples were bottled on site and submitted daily for analysis to the OMOE laboratory.

Stations 5 and 6 were located in the wetland area, outside the greenhouse (Figure 3.3). Station 5 was located about 5 m upstream of the greenhouse inlet and station 6 was located midway along the west side of the greenhouse, adjacent to station 3. These stations were located as such to test for spatial variability in water quality in the wetland as water moved toward the outlet. Also the wetland stations were intended to provide comparison points for the greenhouse stations. Station 5 represented water quality prior to entry into the greenhouse and station 6 represented water in the wetland adjacent to the greenhouse. Station 6 could be used to compare water quality at the greenhouse outlet to water quality near the outlet.

Sampling at stations 5 and 6 were conducted identically to that of stations 2 to 4, using the same ISCO sampler. The sampling stations themselves were also designed identically. The PVC tubing for stations 5 and 6 was manually disconnected and rerouted through the greenhouse. The section of tubing from the greenhouse and the sampling station was housed in 1.5" ABS pipe to protect the sample tubing from damage. The intake strainers for stations 2 to 6 were located 5 to 6 cm off the basin floor.

Station 7 was located at the outlet riser structure. The intake strainer was mounted to the riser pipe approximately 15 cm above the outlet orifice. Set at this level, station 7, labeled the outlet, was at approximately the same elevation from the water surface as stations 2 to 6, but was actually about 1 m above the bottom of the deep outlet depression. Sampling at this station was conducted differently than at stations 2 to 6. Near the end of the rainfall event, once discharge had begun at the outlet, the clear PVC sampling tubing running from the outlet intake strainer to the equipment shed was attached to the ISCO wastewater sampler previously used to collect samples at stations 2 to 6. The sampler was then programmed to collect 750 ml aliquots at 60 minute intervals for 24 hours. These collections were composited into a single sample and at the end of 24 hours of sampling and prepared for lab analysis. Once this procedure was completed, the manually triggered, single samples were collected from stations 2 to 6, after which the wastewater sampler was again connected to the intake line from station 7 and programmed to collect another set of 24 hour composite samples. This process was repeated for four consecutive days.

Due to the time weighted composite nature of the four daily composite samples collected at station 7 over the treatment period, results from this sampling station provided a more representative measure of change in water quality over time in the facility than the results from stations 2 to 6. Also, because each outlet composite was generated from 24 samples, the potential for water quality anomalies was reduced. Flow and sampling results from station 7 were used to calculate the mean event effluent concentration and facility performance over the treatment period.

3.3.3 Water quality parameters and laboratory methods

Water samples were analyzed by the Laboratory Services Branch of the Ministry of the Environment in Toronto, Ontario using principles outlined in *Standard Methods* (Eaton *et al.*, 1995). Constituents analyzed include nutrients (N and P), metals, phenolics, *E.coli*, chloride, oil and grease, TSS and general chemistry. Methods of analysis for these and other parameters are provided in Appendix D. Dissolved oxygen concentrations were measured in the field at the time of sample collection by an Orion DO/Temperature meter.

3.3.4 Surface water quality guidelines

Pollutant concentrations of stormwater effluent from the Aurora facility were evaluated in part using two sets of receiving water quality standards: the federal *Canadian Water Quality Guidelines* for freshwater aquatic life (CCME, 2001), and the *Provincial Water Quality Objectives/Guidelines* (PWQOs) (OMOEE, 1994b). Policy objectives for the protection of aquatic life set out in these documents are based on the quality of receiving waters. Effluent concentrations in excess of guidelines or objectives indicate the potential for water quality problems downstream of the facility. However, dilution of the treatment facility effluent in the receiving stream should be taken into consideration with regard to the environmental impact of the facility. Table 3.1 lists the objectives/guidelines of parameters analyzed in this study.

The discharge of any effluent in Ontario, including stormwater, is governed by the conditions set out through a Certificate of Approval (C of A), issued by the Ontario Ministry of the Environment. Often, but not always, the discharge limits set in the C of A are made with PWQOs in mind. However, the Aurora facility was originally designed solely as a quantity control facility and therefore no site specific quality limits exist. Modifications to the outlet structure prior to this study were intended to improve the pollutant removal capacity of the facility.

Two other guidance documents of relevance to the design of SWM facilities in the Lake Simcoe watershed are the *SWMP Planning and Design Manual* (OMOEE, 1994a) and the *Lake Simcoe Environmental Management Strategy* (LSEMS) (LSRCA, 1995). The SWMP manual proposes a TSS removal target of 70% for constructed wetlands discharging to Type 2 habitat areas, defined as feeding areas, particularly for adult fish, areas of unspecialized spawning habitat and areas where pool-riffle–run complexes occur along much of the watercourse (OMOEE, 1994a). LSEMS is a guidance document addressing water quality concerns in Lake Simcoe and its watershed. This document recommends that, in order to control algae growth and maintain end–of–summer dissolved oxygen levels of 5 mg/L in the lake, total phosphorus concentration of stormwater effluents should not exceed 10 µg/L.

Table 3.1: Provincial Water Quality Objectives/Guidelines (PWQO) and Canadian Water Quality Guidelines (CWQG) for constituents analyzed in this study.

Constituent	PWQO	CWQG
Temperature	Natural thermal regime not altered to the impairment of environmental quality or mixing zones temperatures not to exceed natural ambient temperatures by 10°C.	Thermal additions should not: alter stratification or turnover rates, exceed weekly maximum averages or short term maximums.
Dissolved Oxygen	Cold Water biota (0 - 25°C): 8 – 5 mg/l, 54 – 63% saturation. Warm water biota (0 - 25°C): 7 – 4 mg/l, 47 – 48% saturation	Cold water biota: early life stages: 9.5 mg/l, other stages: 6.5 mg/l Warm water biota: early life stages: 6.0 mg/l, other stages: 5.5 mg/l
PH	6.5 to 8.5	6.5 to 9.0
Escherichia coli	100 counts/100 ml	n/a
Total Suspended solids	TSS should not be added to surface water in concentrations that will change the natural Secchi disc reading by more than 10%.	(i) TSS not to exceed 10 mg/L when background ≤100 mg/l, (ii) TSS not to exceed 10% of background when background TSS > 100 mg/l
Phenols	1 µg/l	1 µg/l
Oil and Grease	Should not be present in concentrations that can: be detected as visible film, sheen, discolouration or odour, cause tainting of edible aquatic organisms or form deposits on shorelines or bottom sediments.	n/a
Total Phosphorus	To avoid excessive : plant growth: <30 µg/l, algal concentrations: < 20 µg/l, aesthetic deterioration: < 10 µg/l	n/a
Ammonia	(un-ionized) 20 µg/l	(total): 1.37 mg/l, pH 8.0, temperature : 10°C
Nitrite	n/a	0.06 mg/l
Nitrate	n/a	Avoid concentrations that stimulate weed growth
Zinc	30 µg/l	20 µg/l
Nickel	25 µg/l	Hardness as $CaCO_3$(mg/l): 0 – 60: 25µg/l, 60 – 120: 65 µg/l, 120 – 180: 110µg/l, > 180: 150 µg/l
Iron	300 µg/l	300 µg/l
Copper	Hardness as $CaCO_3$(mg/l: 0 – 20 : 1 µg/l, > 20 : 5 µg/l	Hardness as $CaCO_3$(mg/l) : 0 – 120: 2 µg/l 120 – 180 : 3 µg/l > 180 : 4 µg/l
Lead	Hardness as $CaCO_3$(mg/l): <30: 1 µg/l, 30 – 80: 3 µg/l, > 80: 5 µg/l	Hardness as $CaCO_3$(mg/l): 0 – 60: 1 µg/l, 60 – 120: 2 µg/l, 120 – 180: 4 µg/l, > 180: 7 µg/l
Cadmium	Hardness as $CaCO_3$(mg/l): ≤100: 0.1 µg/l >100 : 0.5 µg/l	$CaCO_3$(mg/l) 0 – 60: 0.2 µg/l, 60 – 120: 0.8 µg/l, 120 – 180: 1.3 µg/l, >180:1.8 µg/l
Chromium	8.9 µg/l	20 µg/l (to protect fish) 2 µg/l (to protect aquatic microbiota)
Vanadium	6 µg/l	N/A

Note: Objectives/guidelines have not been developed for several of the constituents analyzed in this study.

3.4 Plant Growth, Biomass and Tissue Chemistry

Over the course of the 1996 monitoring season general observations were made on the time and date of sprouting of plant shoots and growth in the greenhouse and the wetland. During the 1997 season vegetation growth was monitored on a monthly basis at 10 plots (1 m^2 each) from shoot break to senescence. Four plots where located in the greenhouse, five were in the wetland, and one was located at the edge of the sediment forebay (Figure 3.4). At the beginning of each month, from early March to November, 1997, vegetation height, shoot density and species composition were recorded at each site.

In early June, plant samples were collected from areas immediately adjacent to each plot. The root, rhizome and aboveground mass were collected, washed of sediment, measured for length, separated into above and below ground mass and wet weighed separately. Samples from each of the 10 plots (above and below ground) were then oven dried at 80°C for 72 hours. Some re-drying was required because of residual moisture in the rhizomes. After desiccation, samples were dry weighed and then submitted for lab tissue analysis. The June collection represented the early season vegetation composition. The collection process was repeated in September just prior to fall senescence. The September collection was dried at 100°C for 72 hours and did not require re-drying. Vegetation samples in the spring and fall were analyzed for micro (TKN and P) and macro (Ca and Mg) nutrients and 16 metals.

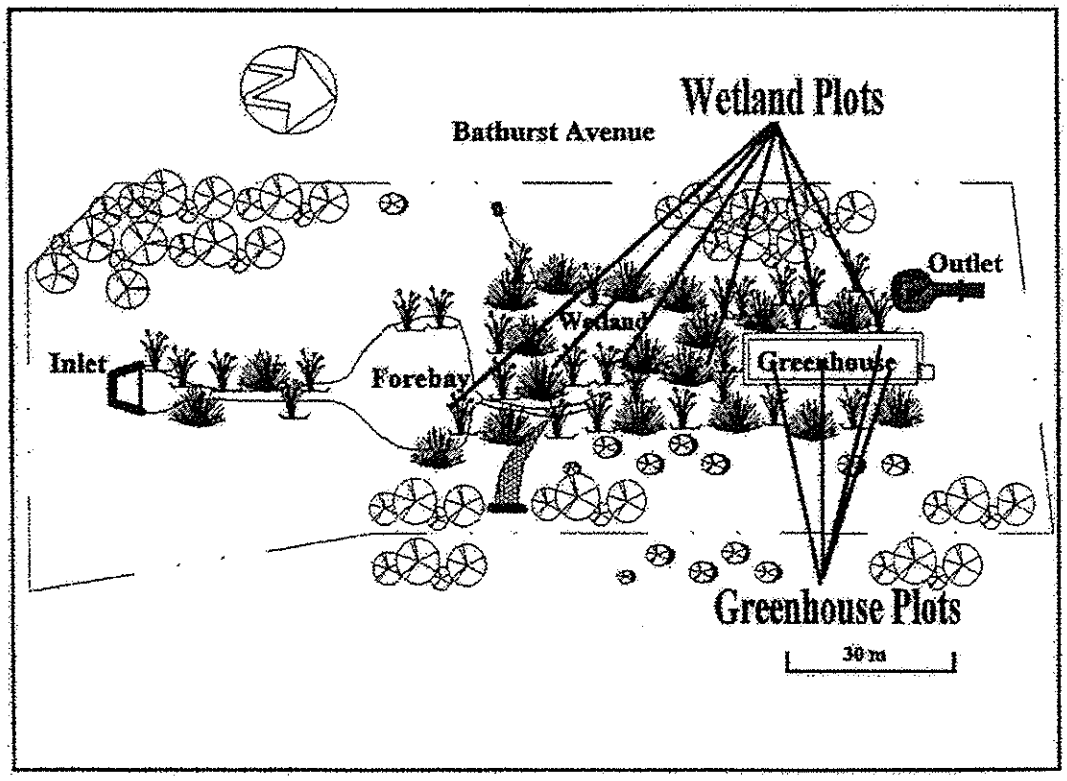


Figure 3.4: Location of vegetation sampling plots within the greenhouse and wetland.

In January 1997, vitality testing was conducted on plant rhizomes collected from the greenhouse to determine whether cells of these rhizomes were metabolizing. The method of vitality testing included cellular staining with uranin and microscopic examination at 10 and 20 times magnification (Stasovski and Peterson, 1991; Peterson and Waite, 1996). In this procedure, living cells can be identified by the presence of staining across the exodermis to the cytoplasm and nuclei, indicating that active osmosis is occurring.

3.5 Sediment Chemistry and Bioassay

Sediment samples were collected in December 1996, June 1997 and August 1997 from four plots in the facility and one control plot located at the southern edge of the SWM facility, above the flooding zone. The locations of sediment sampling plots are shown in Figure 3.5. The four facility plots were located in the greenhouse (1), in the wetland basin (2 and 3), and in the sediment forebay (4). Approximately 5 L of sediment was collected from the upper 5 cm of soil in each plot. The sediment was homogenized in a glass tray, placed in five 250 ml wide mouth amber jars, and subsequently analyzed for the constituents listed in Table 3.2.

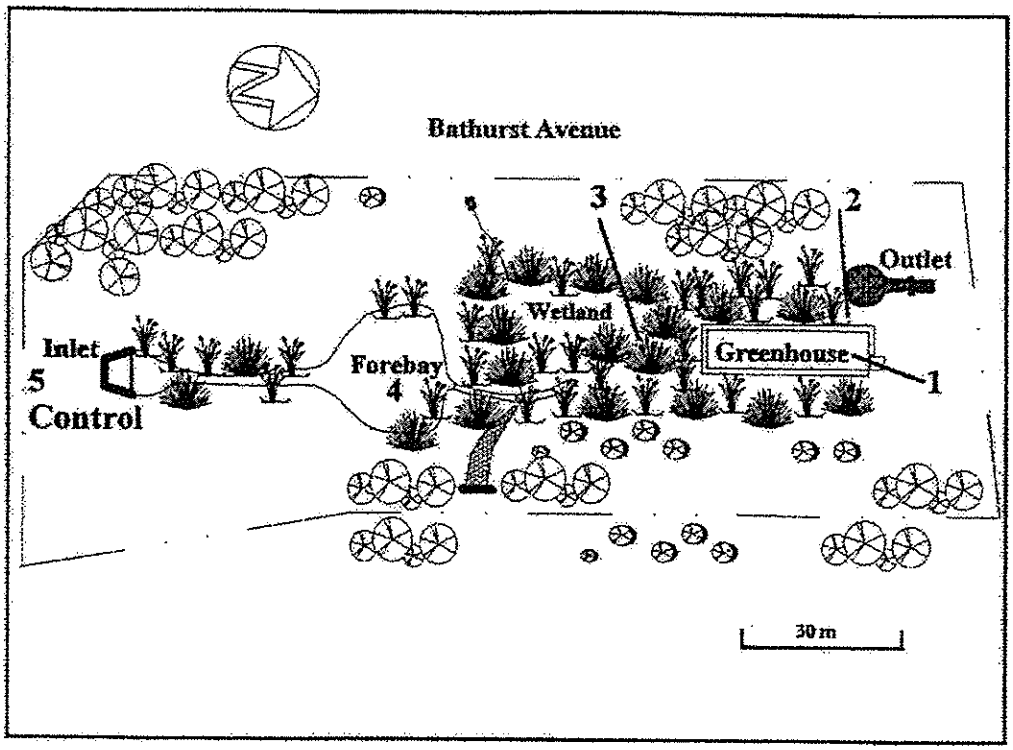


Figure 3.5: Locations of sediment sampling plots.

Table 3.2 Chemical constituents analyzed in sediment.

Constituent group	***Constituents and details***	***Methods***
Total nutrients	TKN and TP	Colourimetry
Chloride		Ion chromatography
Sulphate		Ion chromatography
Oil and Grease	Solvent extractables	Liquid/Liquid extraction
Conductivity		Conductivity meter
pH		pH meter
Metals	18 including Cu, Pb, Zn, Ni, Cd, Cr and Fe	Mass spectrometry/ Inductively coupled plasma
Sediment Particle size	Ranging from <999 - >0.17 μm	Light scattering by Coulter analyzer

In late August 1997, sediment samples for bioassay were collected from the same five plots as those for sediment chemistry. Ten litres of sediment were collected from the upper 5 cm of soil in each plot. Collections were placed into polyethylene-lined buckets; bags were tied to eliminate air pocketing and submitted to OMOE labs for bioassay. OMOE lab protocols for sample collection, preparation and processing were followed.

3.6 Statistical Analysis

The event mean concentration (EMC) was the primary statistic used in the analysis of water quality results during the spring, summer and fall. The EMC is intended to represent the overall average concentration of a pollutant over the duration of a storm event. The influent EMC was based on flow-proportioned composite samples, as described in section 3.3.2. The outlet EMC was determined through daily composites of 24 sample aliquots collected at 60 minute time intervals, and flow weighted by day. Hence, the outlet EMC (EMC^o) is a combination of time and flow-proportioned samples, and as such represents an approximation of the true mean concentration for the event. The outlet EMC was calculated as follows,

$$EMC^o = \frac{\sum_{j=T1}^{T4} V_j^o \times C_j^o}{\sum_{j=T1}^{T4} V_j^o} \qquad \textit{Equation 7}$$

where:

T1 to T4 = day 1 to day 4 of the drawdown period

V^o = daily outlet volume
C^o = daily outlet concentration

Consistent with other stormwater studies (USEPA, 1983; Maunder *et al*, 1995), average event mean concentrations (AEMC) and 95% confidence limits over seasons and the entire study period at each of the monitoring stations were based on the log-normally distributed data.

The load-based removal efficiency (*LE*), which is based on the flow volume (V) and event mean concentration of constituents (EMC), was calculated for single events using the following equation:

$$LE_{event} = \frac{(V^i \times EMC^i) - (V^o \times EMC^o)}{V^i \times EMC^i} \times 100\% \qquad \textit{Equation 8}$$

where V^o represents the outlet flow volume over the 4 day drawdown period.

Performance for each season and the entire study period was calculated based on the sum of loads at the inlet (*i*) and outlet (*o*), as follows:

$$LE_{season} = \frac{\sum_{j=1}^{m}\left[(V_j^i \times EMC_j^i) - (V_j^o \times EMC_j^o)\right]}{\sum_{j=1}^{m}\left[V_j^i \times EMC_j^i\right]} \times 100\% \qquad \textit{Equation 9}$$

where: m = number of events

During the winter, when temperatures fell below the range at which automated samplers can be operated, grab samples were collected at the inlet. Outlet samples were collected in the same manner as during the summer because the sampler was located in a heated shed. Performance during the winter was calculated as:

$$LE_{season} = \frac{\sum_{j=1}^{m}\left[(V_j^i \times C_j^i) - (V_j^o \times EMC_j^o)\right]}{\sum_{j=1}^{m}\left[V_j^i \times C_j^i\right]} \times 100\% \qquad \textit{Equation 10}$$

where C represents a constituent concentration obtained from one or more grab samples.

This study also reports contaminant masses retained within the facility over the event duration. The event mass retention (EMR) for a single event was calculated as the influent mass less the cumulative mass

discharged on each successive day of the drawdown period. Similarly, seasonal retention statistics were calculated from the sum of individual event influent masses less the sum of cumulative effluent masses.

4.0 WATER QUANTITY ANALYSIS

Precipitation and hydrologic data for 29 storms monitored from July 1996 to November 1998 are summarized in Table 4.1. These 29 storms represent 51% of the total storm runoff volume monitored over the study period. Study and season totals and/or averages for selected hydrologic statistics are provided in Table 4.2.

4.1 Precipitation

The hydrologic data set includes 29 storms, of which 17 were large (rain > 20 mm), 10 were mid-sized (between 10 and 20 mm), and 2 were small (< 10 mm). Mean precipitation for the 29 events was 21.8 mm. Precipitation intensities were calculated for 15, mostly warm season events, when automated tipping bucket rain gauge data were available. The mean rainfall intensity for these 15 events was 1.5 mm/h. During the cold season, and when the tipping bucket rain gauge malfunctioned, daily volumes were determined from a manual rain/snow gauge.

Seasonal rainfall totals in Table 4.2 (including several events not included in Table 4.1) indicate that the summer season had the greatest amount of precipitation (437 mm) followed by the winter (356 mm), spring (302 mm) and fall (297 mm). Among the three years of monitoring, rainfall was greatest in 1996.

4.2 Storm Runoff Coefficients

The storm runoff coefficient is calculated as the ratio of runoff to catchment rainfall. The coefficient represents the ability of the catchment to infiltrate or evaporate rainfall, rather than generate runoff. The runoff coefficient was not calculated during the winter, when retention of precipitation in form of snow, and release of varying quantities of runoff in the form of snowmelt, made the interpretation of winter statistics on an event-by-event basis more difficult. Storm runoff coefficients for the remaining events averaged 0.21, and varied widely from 0.07 to 0.50. This mean runoff coefficient is within the range expected for a catchment with 30% agricultural land use.

During the entire study period, including events monitored but not sampled, the mean runoff coefficient was 0.23, ranging seasonally from 0.17 in the spring to 0.29 in the winter (Table 4.2). The higher winter runoff coefficient reflects snowmelt contributions, and increased runoff from frozen soils in the agricultural area west of Bathurst (Figure 2.1). Note that all of the seasonal runoff coefficients were significantly less than the estimated coefficient (0.41) used in the original design of the facility (MMM, 1988)

Table 4.1: Hydrologic statistics for 29 storm events[1]

Storm	Date	Precipitation (mm)	Mean Intensity (mm/h)	Total Catchment Rain (m^3)	Total Inlet Volume (m^3)[2]	Runoff Coefficient.[3]	Total Outlet runoff (m^3)	Total losses to GW (m^3)	Total Et (m^3)	Total water output (m^3)	water balance volumetric error (%)
1	15/ 07/96	35.8	2.56	29485	5720	0.19	4019	1396	34.5	5450	-4.7
2	19/07/96	17.4	2.9	14331	7479	0.50	5243	1758	43.1	7044	-5.8
3	30/7/96	19.2	1.48	15813	4955	0.31	2985	1475	34.7	4495	-9.3
4	13/09/96	24.8	1.03	20425	7462	0.35	5310	1503	10.8	6824	-8.6
5	18/10/96	31.2	0.87	25696	12275	0.43	9974	930	8.2	10912	-11.1
6	30/10/96	15	0.76	12354	3058	0.25	2140	484	5.5	2630	-14.0
7	1/12/96	19	0.54	15648	3818	0.24	2585	547	0	3132	-18.0
8	17/12/96	22.6	1.41	18613	7565	0.41	7287	559	0	7846	3.7
9	5/01/97	18.8	NA	15484	5619	-	5619	0	0	5619	0.0
10	18/02/97	24.1	NA	19849	18246	-	18246	0	0	18246	0.0
11	25/03/97	21.5	NA	17707	5578	0.32	4269	923	0	5192	-6.9
12	3/05/97	31.6	NA	26026	7443	0.29	6486	925	26.8	7438	-.1
13	15/05/97	19.8	NA	16307	3767	0.23	2557	829	21.4	3407	-9.5
14	9/7/97	13.1	1.64	10789	1189	0.11	1005	231	34.5	1271	-6.9
15	18/7/97	17.4	3.48	14331	1422	0.10	1317	255	38.0	1610	-13.2
16	15/8/97	21	NA	17295	1849	0.11	1594	255	28.6	1877	-1.5
17	20/8/97	30	NA	24708	3084	0.13	2388	255	17.8	2660	13.7
18	17/9/97	19	1.46	15649	2370	0.15	2144	204	24.9	2373	-0.1
19	27/10/97	30.2	0.915	24873	3006	0.12	2412	440	7.1	2859	4.9
20	31/10/97	20.8	0.8	17131	2680	0.13	2312	87	6.7	2405	23.5
21	04/12/97	2.8	0.47	2306	352	0.15	377	0	0	377	-7.3
22	05/1/98	1.6	NA	1318	4710	-	4815	0	0	4815	-2.2
23	17/2/98	22.7	NA	18366	7656	-	7338	0	0	7338	8.1
24	08/3/98	14.4	NA	11860	3655	-	2884	0	0	2884	31.4
25	18/3/98	24.8	NA	20425	2779	-	2943	0	0	2942	5.9
26	16/04/98	38.4	NA	31626	3349	0.11	2753	316	30.4	3099	7.4
27	11/05/98	27.6	NA	22731	2512	0.11	2206	148	40.6	2395	4.6
28	29/05/98	23.3	NA	18943	1228	0.07	571	378	48.2	998	18.8
29	11/06/98	23.3	1.94	18943	1422	0.08	1202	395	49.4	1647	-15.8
	Study Mean	21.8	1.6	17898	4698	0.21	3965	493	17.6	4475	4.7[4]

1. See section 3.1 for a description of water balance components and estimation methods.
2. Inlet volume represents storm and relatively minor quantities of pre-storm storage volumes (see Table 4.5). Direct basin rainfall inputs (P_d in eqn 3) represent a small proportion of total inputs and were not included in the water balance calculations.
3. The runoff coefficient was calculated from storm volumes only. Winter runoff coefficients were not estimated because retention of precipitation in the form of snow, and release of precipitation in the form of snowmelt, rendered runoff coefficients misleading on an event-by-event basis. Evapotranspiration and losses were assumed to be low or negligible during the cold winter months.
4. Calculated from mean influent volume (4698 m^3) and mean water loss (4475 m^3).

Table 4.2. Hydrological summary by season and the entire two year study.

Parameter/Season	*Summer*	*Fall*	*Winter*	*Spring*	*Study Total*
Precipitation (mm)	437	297	356	302	1,393
Total Catchment rainfall (m^3)	360,242	244,774	293,531	248,563	1,147,110
Runoff (m^3)	79,242	60,795	85,020	42,780	267,840
Runoff coefficient	0.22	0.25	0.29	0.17	0.23
Total Evapotranspiration (m^3)	837	251	-	730	1,817
Total Groundwater losses (m^3)	20,829	7,003	-	7717	35,639
Total Outlet runoff (m^3)	61,412	48,807	85,020	30,548	225,787
Volumetric error (%)	-4.9	7.8	0.0	8.8	1.7

4.3 Stormflow Volume, Drawdown and Hydraulic Detention Time

The mean influent volume during the study period was 4,698 m^3. Several events generated runoff at rates sufficient to exceed the facility extended detention volume of 1,300 m^3. During these large and more intense storms, excess flow exited via the step weir. Summer and winter were the seasons of greatest total runoff, followed by fall, then spring. The large winter runoff volumes are attributed to snow melt, which often occurred before the onset of spring.

The drawdown period ranged from 3 to 6 days during storm events. The hydraulic detention time is typically calculated as the lag-time between the inlet and outlet hydrograph centroids. Due to the manner by which outflow was determined, precise calculations could not be provided. However, the approximate average was 36 hours, with a range between 15 and 48 hours during individual events.

4.4 Groundwater

Among the four groundwater monitoring well nests installed in the stormwater facility only one, located in the middle of the wetland (Figure 3.1), exhibited regular subsurface water level fluctuations. Other wells surrounding the area of active water level fluctuations were always empty, despite intake elevations below the basin floor. This observation suggests that recharge occurs mostly in a vertical direction beneath the flooded area, with little horizontal transport within the upper 1.25 m of soil. Further, in well nests at the periphery of the basin, there was no evidence of perched water tables, which sometimes form when a low-permeability clay layer exists within a more permeable sand or silt formation.

Groundwater recharge losses from the basin for the 29 storms sampled are presented in Table 4.3. Seasonal and study total groundwater recharge is presented in Table 4.4. After a rain event, groundwater

recharge from the facility (or drawdown), as measured in the wetland well nest (50 and 75 cm below the surface), occurred at an average rate of approximately 0.06 m/d. This average rate is considered to be a conservative estimate of the actual rate of recharge occurring in the basin since it does not include subsurface drawdown during the period of stormwater detention.

Table 4.3. Groundwater drawdown and losses for the 29 events sampled

Storm Date	Drawdown (days)	Mean GW Drawdown rate (m/d)	Capillary losses (m^3)	Mean Basin Depth (m)	Head induced Drawdown (m^3)	Total GW loss (m^3)
15/ 07/96	4	0.091	0	0.47	1396	1396
19/07/96	5	0.091	0	0.48	1758	11758
30/7/96	4	0.091	28	0.50	1447	1475
13/09/96	5	0.091	0	0.35	1503	1503
18/10/96	6	0.032	224	0.47	706	930
30/10/96	5	0.032	0	0.30	484	484
1/12/96	5	0.032		0.37	547	547
17/12/96	5	0.032		0.39	559	559
5/01/97	4	-	-	0.32	-	-
18/02/97	5	-	-	0.64	-	-
25/03/97	5	0.048		0.47	923	923
3/05/97	5	0.048	56	0.41	869	925
15/05/97	5	0.048	0	0.37	829	829
9/7/97	3	0.091	78	0.27	153	231
18/7/97	5	0.091	0	0.27	255	255
15/8/97	5	0.091	0	0.33	255	255
20/8/97	5	0.091	0	0.38	255	255
17/9/97	4	0.091	196	0.31	204	400
27/10/97	5	0.031	353	0.38	87	440
31/10/97	5	0.031	0	0.31	87	87
04/12/97	3	-	-	0.09	-	-
05/1/98	5	-	-	0.53	-	-
17/2/98	5	-	-	0.53	-	-
08/3/98	5	-	-	0.37	-	-
18/3/98	5	-	-	0.32	-	-
16/04/98	5	0.053	168	0.38	148	316
11/05/98	5	0.053	0	0.34	148	148
29/05/98	5	0.053	230	0.27	148	378
11/06/98	5	0.053	346	0.27	148	395

Note: see section 3.1.3 (equations 4 and 5) for a description of methods.

Table 4.4. Groundwater parameters presented seasonally and for the study period.

Parameter/ Season	Summer	Fall	Winter	Spring	Study period
Mean Drawdown rate (m/d)	0.091	0.031	-	0.052	0.059
Days basin Flooded	89	82	67	66	304
Mean Basin Depth (m)	0.36	0.27	0.38	0.28	0.32
Capillary losses (m^3)	3220	1405	-	2576	7301
Head induced GW losses (m^3)	17609	5597	-	5141	28347
Total GW losses (m^3)	20829	7002	-	7717	35639

The average drawdown rate was considerably higher during summer, at 0.09 m/d, followed by spring and fall, at 0.05 and 0.03 m/d, respectively. Increased evapotranspiration during the warm summer months may partly account for the higher summer drawdown rates. Observed drawdown rates range from 3.5 x 10^{-7} to 1.05 x 10^{-6} m/s and are consistent with hydraulic conductivities in the silty clay to clayey classes (Freeze and Cherry, 1979). Water losses via groundwater recharge were highest during summer and assumed to be negligible during the winter. During summer, fall and spring, groundwater losses as a percent of inlet runoff were estimated at 26, 12 and 18%, respectively. On an annual basis, groundwater losses accounted for approximately 13% of total inlet runoff volumes (Table 4.2).

4.5 Evapotranspiration

Event and seasonal evapotranspiration estimates are presented in Tables 4.1 and 4.2, respectively. Over the summer, fall, spring and entire study period, estimated evapotranspiration accounted for about 1.0, 0.4, 1.7 and 0.7% of inlet runoff. Evaporation pan measurements began in late August 1996, therefore the rate used to calculate evapotranspiration for storms 1 to 3 was based on the July 1997 pan evaporation rate. Also the seasonal total for the first summer of monitoring in 1996 is based on the second summer of monitoring in 1997 (until late August).

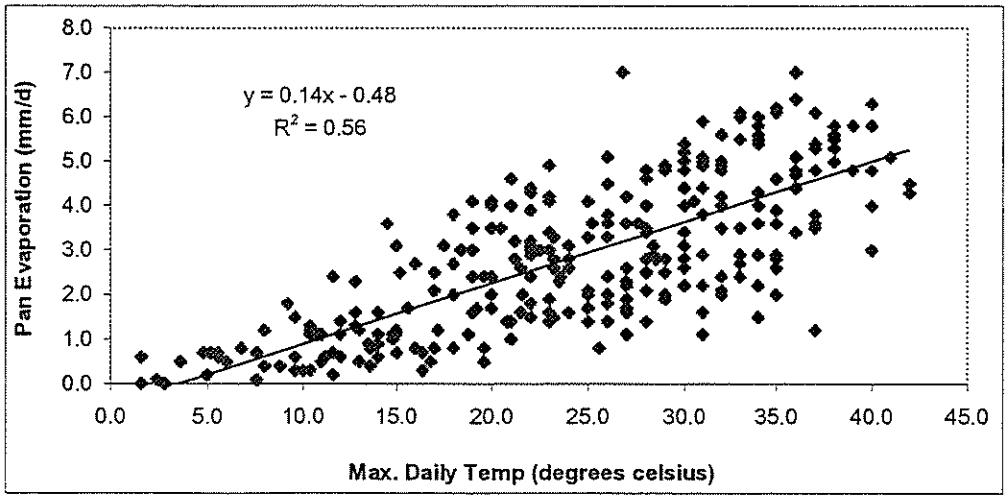


Figure 4.1: Linear relationship between daily pan evaporation and daily maximum temperatures.

The relationship between maximum daily temperatures and pan evaporation records is shown in Figure 4.1. The R^2 statistic indicates that 56% of the variation in daily evaporation rates is explained by maximum daily temperatures. Other influencing factors may include wind, atmospheric pressure, average daily temperature and the number of hours of sunlight.

4.6 Basin stormwater outlet flow volumes

Basin outlet flow volumes were calculated from water level data and the orifice and weir equations described in section 2.2.2. The flowlogger installed at the outlet was replaced seven times in 1996 due to equipment malfunction. Therefore, outflow data during 1996 were derived primarily from discharge equations based on water level measurements. This method was thought to be reasonably representative of actual discharge especially below the upper lip of the riser. For water levels above the riser lip to the first step of the weir, terminal discharge was estimated at 58 l/s. This maximum discharge estimate was later confirmed to be accurate using measured outflow data in 1997.

Figure 4.2 shows inlet and outlet hydrographs for a storm event on August 20th. A total of 30 mm of rainfall fell over a period of 5 days, 17 mm of which fell during the first day. Comparison of the inlet and outlet hydrographs shows a reduction in peak flow from 310 L/s at the inlet to 25 L/s at the outlet. The sudden increase in runoff at the outlet on August 21st represents a transition of flow from the 50 mm orifice at the bottom of Hickenbottom riser to flow through the open mouth of the riser (457 mm), which occurs when water levels in the facility rise to 53 cm above the bottom of the basin (see section 2.2.2).

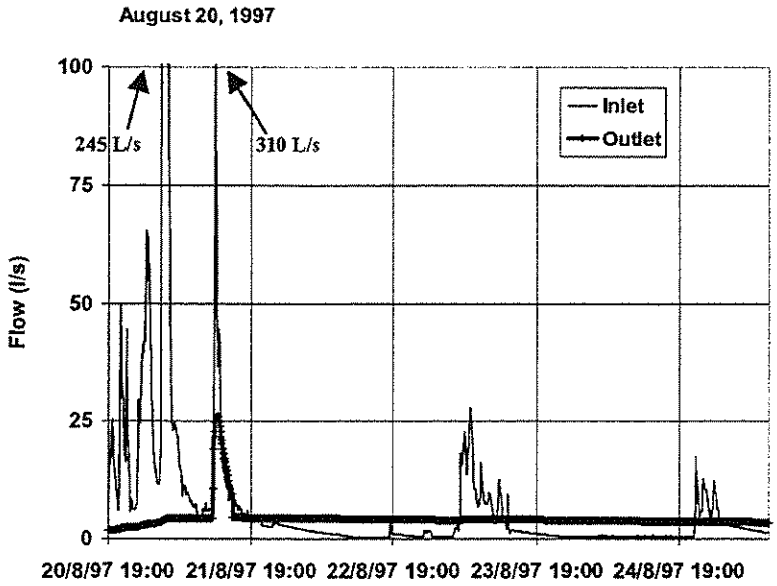


Figure 4.2: Inlet and outlet hydrographs for a 30 mm rain event on August 20th, 1997.

Table 4.5: Storm event flow summaries

Storm	Date	Pre-storm vol. (m^3)	Inlet Runoff (m^3)	Total Event Volume (m^3)	Day 1 Outlet Runoff (m^3)	Day 2 Outlet Runoff (m^3)	Day 3 Outlet Runoff (m^3)	Day 4 Outlet Runoff (m^3)	Volume after Day 4 (m^3)	Total Outlet Volume (m^3)
1	15/ 07/96	0	5720	5720	1577	1159	962	0	321*	4019
2	19/07/96	321	7158	7479	2946	1002	633	344	670	5243
3	30/7/96	0	4955	4955	1195	400	360	360	670	2985
4	13/09/96	377	7085	7462	4348	357	330	275	81	5310
5	18/10/96	1200	11075	12275	5000	3575	367	354	913	9974
6	30/10/96	0	3058	3058	1195	344	288	257	153	2140
7	1/12/96	0	3818	3818	1195	337	326	315	521	2585
8	17/12/96	0	7565	7565	5000	1684	351	283	81	7287
9	5/01/97	0	5619	5619	5000	344	275	0	0	5619
10	18/02/97	0	18246	18246	5000	2020	5000	5000	1226	18246
11	25/03/97	0	5578	5578	2435	360	307	377	975	4269
12	3/05/97	0	7443	7443	243	5000	377	370	670	6486
13	15/05/97	0	3767	3767	1195	360	326	233	610	2557
14	9/7/97	0	1190	1190	543	270	193	0	0	1005
15	18/7/97	0	1422	1422	302	339	290	287	100	1317
16	15/8/97	0	1849	1849	286	357	321	280	350	1594
17	21/8/97	0	3084	3084	730	352	342	314	650	2388
18	17/9/97	0	2370	2370	482	274	337	322	730	2144
19	27/10/97	0	3007	3007	742	413	395	399	463	2412
20	31/10/97	463	2218	2681	1294	306	275	229	208	2312
21	04/12/97	0	351	351	220	157				377
22	05/1/98	0	4710	4710	573	1493	760	635	1355	4815
23	17/2/98	330	7326	7656	4305	1153	536	181	1163	7338
24	08/3/98	551	3104	3655	1849	111	101	94	730	2884
25	18/3/98	0	2779	2779	1945	274	246	214	264	2943
26	16/04/98	0	3349	3349	1203	824	258	228	240	2753
27	11/05/98	0	2512	2512	1586	263	201	139	17	2206
28	29/05/98	0	1228	1228	263	138	110	61	0	571
29	11/06/98	0	1423	1422	88	364	337	287	127	1202
Total		3242	133009	136250	52737	24030	14603	11837	13287	114982
Mean		112	4586	4698	1818	829	522	423	475	3965

* Volume in facility after day – 3 at the start of storm 2.

Table 4.5 shows the daily distribution of wetland outlet flow over the duration of each event monitored. As expected, runoff was highest on the first and second days of the drawdown period. During some smaller rain events, there was no measurable flow after day 3 of the event.

In the downstream channel, flow during long interevent periods declined considerably near Salamander Pond, approximately 0.5 km downstream of the facility. Runoff from the facility increased stream flow to an average width of 1m and depth of about 10 cm. At maximum estimated discharges of 58 L/s, the highest mean stream velocity was approximately 0.6 m/s.

4.7 Water balance volumetric error

Expressed as the difference between inputs to and outputs from the facility (equation 4, chapter 3), the average volumetric error in the event water balances was 4.7% (Table 4.1). This event-based error compares to a mean volumetric error for the entire study period (including small storms) of 1.7% (Table 4.2), and seasonal volumetric errors of less than ±10%. Errors of this magnitude are within the range expected for data collection instruments used in this study.

5.0 WATER QUALITY ANALYSIS

Water quality results are presented in this report based on four pollutant groups: general chemistry (including suspended solids and particle size distribution), bacteria, nutrients and metals. Mean seasonal concentrations, removal efficiencies and cumulative mass retention[2] over the treatment period are calculated for selected constituents. The mean concentrations represent 29 events sampled from July 1996 to June 1998. Among these events, 9 occurred during the summer, 7 each during the fall and spring, and 6 during the winter. In general, runoff volume was not correlated with influent or effluent quality.

A summary of seasonal water quality statistics (*e.g.* 95% confidence intervals, standard deviations, etc.) and removal efficiencies are provided in Appendices E and F, respectively. The significance of pollutant concentration data is gauged in part by comparison to receiving water objectives for the protection of aquatic habitat, as set forth in the Ontario Provincial Water Quality Objectives/Guidelines (PWQOs) (OMOEE, 1994b), .

5.1 Total Suspended Solids

Total suspended solids (TSS) concentrations are often used as a general indicator of stormwater quality because of their ability to transport adsorbed pollutants, such as phosphorus and heavy metals. Large concentrations of suspended particulate matter can also reduce light penetration into the water column, cause abrasion of fish gills, disrupt aquatic food webs, and destroy feeding and spawning grounds through sediment blanketing.

TSS results are summarized in Table 5.1. Influent TSS concentrations and removal efficiencies varied considerably over the study period. Influent and effluent mean concentrations were highest in the summer, followed by the fall, spring and winter. The grab method of sampling during the winter likely explains the low influent concentrations observed during this season. Since winter grab samples were collected near the end of the influent runoff, the 'first flush', often associated with higher solids loading, was not sampled.

Seasonal load based TSS removal efficiencies were above 85% during the spring, summer and fall. In contrast, load-based removal over the winter season was only 44%. The poor winter removal results can be in part attributed to relatively low influent concentrations, which, as mentioned above, is likely an artifact of the grab method of sampling during this season. As shown in Figure 5.1, influent concentrations are associated with a wide range of efficiencies up to a threshold value of between 70 and 110 mg/L, presumably because of changes in size distribution and particle density. At higher influent concentrations, most particles are large and dense, and consequently the efficiency is typically above 85%. The correlation between influent and effluent suspended solids

2. Cumulative mass retention represents the influent load less the cumulative effluent load from day 1 to day 3.

concentrations was weaker (R^2 = 0.44) than between influent concentrations and efficiencies, particularly at influent concentrations below 213 mg/L.

Table 5.1: Suspended solids mean influent/effluent concentrations, facility performance and mass input and retention

Total Suspended Solids	Infl. Conc. (mg/L)		Effl. Conc. (mg/L)		Performance (%)		Mass Input and Cumulative Mass Retention (Tonnes)			
	Min	Max	Min	Max	Min	Max	Input	Day 1	Day 2	Day 3
Summer	194		26		91					
	15.5	856	5.4	155.4	49	96	13	12	12	12
Fall	121		26		87					
	30	945	9.4	83.1	40	94	7	7	6	6
Winter	43		29		46					
	24	108	16.4	39.9	-60	72	2	2	1	1
Spring	110		17		90					
	14	335	4.8	36.4	71	95	5	5	5	5
Study Period	111		24		86					
	14	945	4.81	155.4	-60	96	28	26	25	24

Note: All seasonal concentrations represent an approximation of the average event mean concentration, except during the winter, when grab samples were manually collected at the inlet near the end of the runoff event.

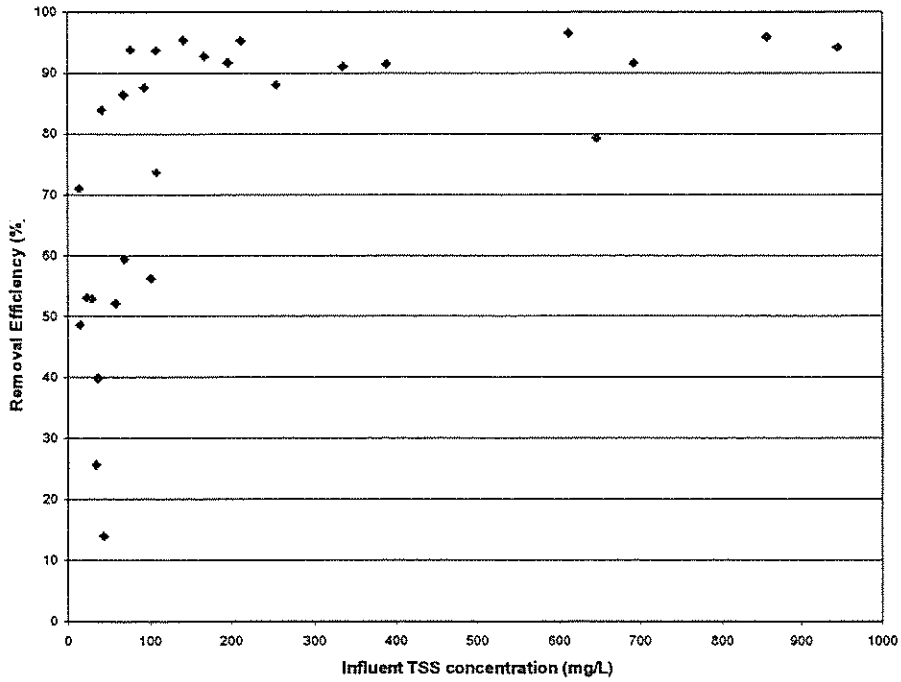
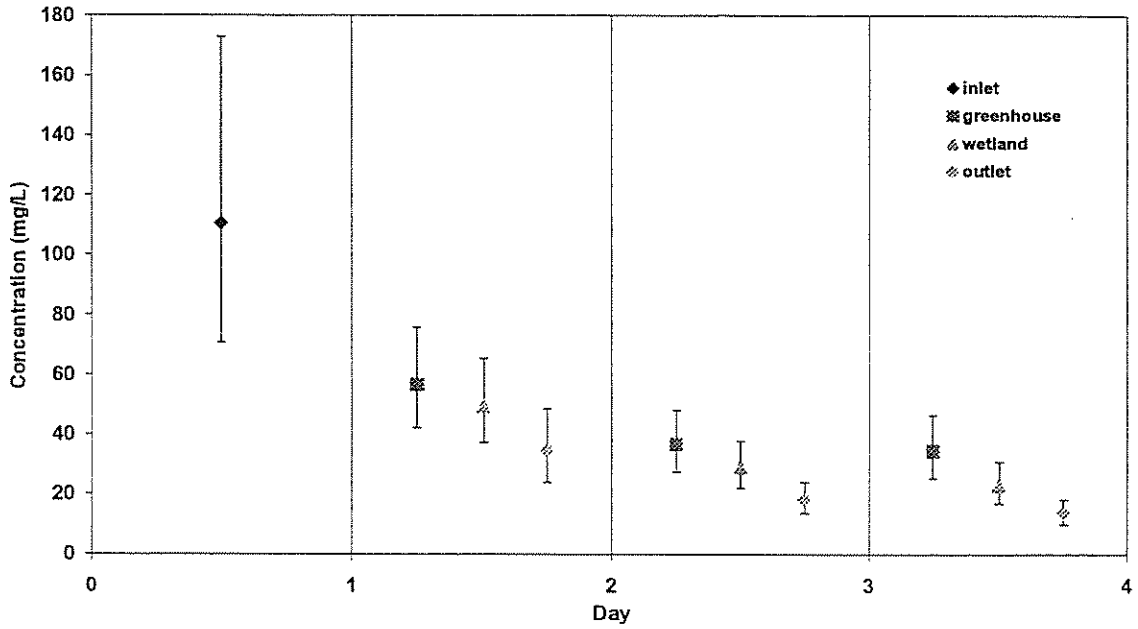


Figure 5.1: Relationship between influent TSS concentrations and removal efficiencies.

Figure 5.2 shows the change in mean TSS concentrations from the inlet (station 1) to the greenhouse (station 4), external wetland (station 6) and outlet (station 7) over each of the first three days following the start of the event (see Figure 3.2 for location of stations). Over 90% of the total concentration reduction from the inlet to the greenhouse, wetland and outlet occurred after the first day of treatment. Daily load reduction, as represented in Table 5.1 as mass retention (input mass less cumulative daily output mass), show a similar pattern in all seasons (i.e. most of the load reduction occurred during the first day of the event). By the second day of treatment, all three monitoring stations had TSS concentrations below influent concentrations at the 95% level of significance. Mean concentrations were highest at the greenhouse, followed by the wetland and outlet. Only on the second and third day of treatment were mean greenhouse and outlet concentrations significantly different.



Note: Spring, summer and fall influent samples (day 0 to 1) were flow-proportioned over the duration of the inlet hydrograph. During the winter, grab samples were collected at the inlet near the end of the runoff event.

Figure 5.2: Mean TSS concentrations and 95% confidence intervals at the inlet, greenhouse, wetland and outlet monitoring stations.

The overall TSS removal efficiency (86%) observed in this study compares with a median efficiency of 81% for 14 stormwater wetlands in the United States. (Strecker *et al.*, 1992). Removal efficiencies above 85% have been reported in pond-wetland systems (*e.g.*: Oberts and Osgood, 1991).

5.1.1 *Suspended solids accumulation and facility maintenance*

The study period mass input and retention in Table 5.1 includes data gathered from the 29 storm events sampled over the two year study period. Total estimated inlet mass was 27.9 tonnes and total mass retained was 24.2 tonnes. The 29 storms sampled represented about 51.4% of total estimated runoff entering the facility. Most unmonitored runoff was in the form of smaller storm events and baseflow with generally less concentrated TSS relative to larger events. Thus, it can be conservatively estimated that the total inlet TSS mass was roughly 50 tonnes (30 tonnes during wet weather and 20 tonnes during dry weather and small rain events) over the two year monitoring period. Since both the 1996-97 and 1997-98 years were unusually dry, a reasonable estimate of TSS loading during a climatically normal year would be approximately 35 tonnes (or 70 tonnes over two years), of which 30.4 tonnes of TSS would be retained within the facility. Based on an estimated TSS bulk density of 1.23 tonnes/m^3 (OMOEE, 1994a), the annual solids volume retained would be 24.7 m^3. If 20% of the suspended solids mass decomposes over time to 10% of its original mass, then the long term solids accretion within the facility is 20.2 m^3/yr.

The SWMP manual recommends facility maintenance based on a reduction in permanent storage associated with a 5% decline in performance (see OMOEE, 1994a for estimation methods). Using graphical plots of storage vs wetland performance provided by the OMOEE (1994a), a 5% annual performance reduction in the Aurora facility would result from a storage loss of approximately 20 m^3/ha, or 1647 m^3. Assuming an even distribution of solids over the entire basin, it would take 81 years of sediment accretion to reduce the storage capacity of the facility by this volume. Of course, since accumulation is much greater in the forebay than the wetland, cleanout of the forebay would be required sooner.

5.2 Particle Size Distribution

Comparison of influent and effluent particle size distributions (PSDs) offers insights into the size selective nature of sedimentation in the Aurora wetland. These insights contribute to a better understanding of removal processes in wetlands, but are also important because small grain sizes are generally associated with greater and more bio-available pollutant concentrations than large grain sizes (Sharpley et al., 1992; Bavor et al., 2001).

The PSD of suspended solids was determined on composite samples collected from August 1997 to June 1998. Average PSDs for events occurring over this period are shown in Figure 5.3. There were more influent samples analyzed than effluent samples because, during several events, the effluent sample submitted to the laboratory was too clear for particle size analysis by laser light diffraction. Thus, average PSDs shown in Figure 5.3 represent 13 samples at the inlet, and 12, 7, 6 and 5 samples at the outlet on day 1 to 4, respectively[3]. A comparison of PSDs for single events is provided in Appendix G.

[3] Average PSDs of the five events for which data were available at both the inlet and outlet over all 4 days of treatment showed a more gradual shift in distributions from the inlet to day 3 (see Appendix G). The median influent particle size for these events was 2.6 μm and the median effluent particle size was 2.1 μm on day 1, 1.7 μm on day 2, and 1.4 μm on days 3 and 4.

The average PSDs show a substantial shift towards smaller particle sizes from the inlet to the outlet. Most of the reduction in median particle size occurred by the end of the first day. However, if it had been possible to include the samples designated as too clear for analysis in the average effluent PSD plots, the difference among effluent PSDs on day 1 to 4 would likely have been more pronounced. Particles greater than 3.7 μm (mostly silt and fine sand size classes) accounted for 47% of the influent PSD, compared to between 11 and 20% of the effluent PSDs. The median particle size of the average PSDs was approximately 3.4 μm at the inlet, and between 1.4 and 1.7 μm at the outlet. Bavor et al. (2001) suggest that, when compared to wet ponds, extensive vegetation in wetlands enhances removal of fine particle sizes (less than 2 μm). However, effluent PSDs at the Aurora wetland were similar to those of other stormwater ponds monitored in Ontario (SWAMP, 2002a; 2002b)

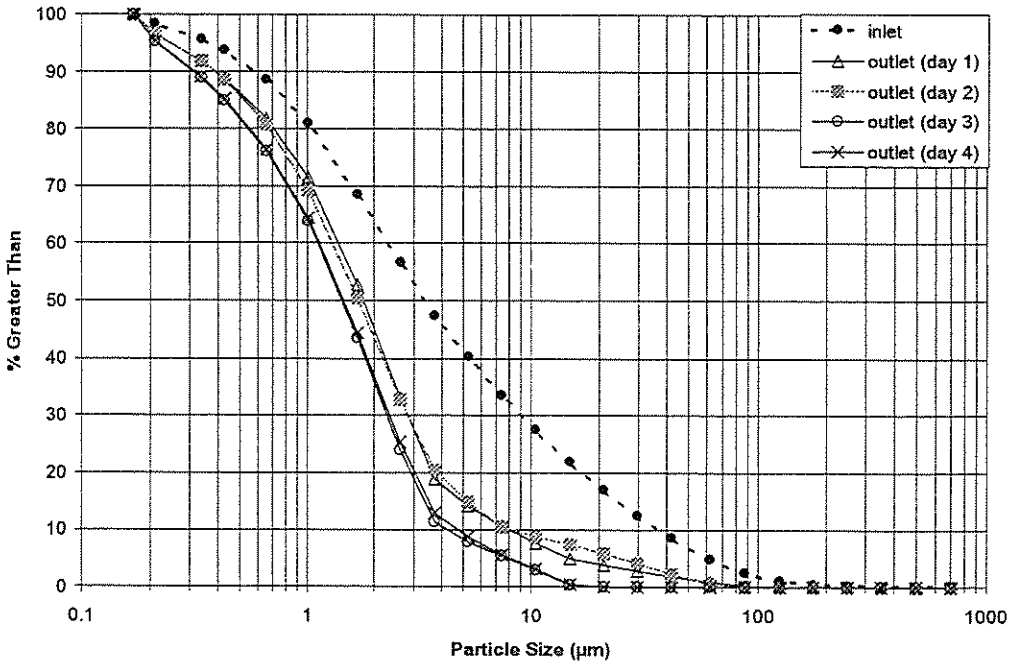


Figure 5.3: Inlet and outlet average particle size distributions

5.3 General Chemistry

5.3.1 Dissolved Oxygen

Dissolved oxygen (DO) measurements were taken manually at the time of water sample collection during the first year of the study. DO concentrations increased during cold periods and fell during the warm summer months, when micro-organisms are most active. The lowest DO concentrations were observed in the wetland and greenhouse (minimum 2.4 and 2.6 mg/L), in part due to low flow rates and poor water mixing at these two stations. By comparison, the summer DO concentration at the outlet, where flow throttling generates

some turbulence, averaged 5 mg/L. Over the observed temperatures, the lower DO threshold for warm water aquatic species is considered to be between 4 and 5 mg/L, or at a 47% saturation level (OMOEE, 1979, LSRCA, 1995). The lower threshold for cold water biota is slightly higher at 5 mg/L, or 54 to 62% saturation (OMOEE, 1994b).

Load based DO removal ranged from 18% in the spring to 41% in the summer. Decomposition of dissolved and particulate matter by microbes such as bacteria, fungi and actinomycetes are partly responsible for reduction of DO concentrations within the facility.

5.3.2 pH

Effluent pH levels ranged from 7.5 to 8.2 over the study period, which is within the 6.5 to 8.5 range stipulated in the Provincial Water Quality Objectives (OMOEE, 1994b). Influent pH levels exhibited a wider range (7.3 to 9.4), and averaged 8.0 over the study period.

5.3.3 Biochemical and chemical oxygen demand

Biochemical Oxygen Demand (BOD) seasonal influent concentrations varied considerably over the study period, with the largest mean values observed during the spring (5.3 mg/L) and summer (3.9 mg/L). The greenhouse and wetland values were similar to influent concentrations, except on day 1 when mean wetland concentrations rose well above those at the inlet. This increase may have been caused by resuspension of organic and inorganic matter within the facility during event flooding. Mean outlet concentrations over the study period were significantly less than at all other monitoring stations. Removal was greatest in the spring (55%), followed by summer (34%), fall (22%) and winter (7%), averaging 32% over the study period.

In general, the COD influent and effluent concentrations were approximately an order of magnitude higher than BOD, but followed similar patterns over the treatment period. Maximum influent values were noted in spring (102 mg/L) and summer (92 mg/L). Removal efficiency over the study period was 33%.

5.3.4 Chloride and related parameters

Since chloride is an important component of dissolved solids and a significan contribution to conductivity, concentrations of the three parameters displayed similar fluctuations seasonally and over the treatment period. Hence, the following discussion of chloride is also relevant to conductivity and dissolved solids. Statistical summaries of the latter two constituents may be found in Appendicies E and F.

Seasonal differences in chloride concentrations and removal are presented in Table 5.2. Variations in concentrations among events are shown in Figure 5.4. Chloride exhibited strong seasonal fluctuations due to winter application of road salts. In stormwater ponds (SWAMP, 2002a, 2002b), these salts typically accumulate during the winter and are flushed out during large runoff events in late winter and spring, resulting in negative removal efficiencies during this period. However, event concentrations shown in Figure

5.4 show higher effluent concentrations during the winter of 1997. This result may be explained in part by the grab sample method of winter influent sample collection, which likely missed the first flush when peak chloride concentrations are typically observed. Winter effluent samples were collected with an automated sampler (inside a heated hut) and are, therefore, more representative of the true mean concentration for the entire event (see section 3.6). Note also that baseflow, small rain events and some snowmelt events were not monitored, making it impossible to show storage curves or calculate mass balances. Storage of chloride within the facility during these periods may also help explain why winter and spring effluent concentrations were higher than influent concentrations in 1997.

Table 5.2: Chloride influent/effluent concentrations, facility performance and mass input and retention

Chloride	**Infl. Conc. (mg/L)**		**Effl. Conc. (mg/L)**		**Performance (%)**		**Mass Input and Cumulative Mass Retention (Tonnes)**			
	Min	**Max**	**Min**	**Max**	**Min**	**Max**	**Input**	**Day - 1**	**Day – 2**	**Day - 3**
Summer	4.2		10.5		-51					
	1.8	11.2	4.9	23.2	-305	33	0.18	0.09	0.05	-0.02
Fall	37.6		55.8		24					
	5.2	402	8.0	428.6	-451	86.9	2.05	1.44	1.15	0.93
Winter	277.4		429.4		-15					
	86	949	104.9	1567	-526	69	19.55	9.70	5.57	2.82
Spring	43.3		34.5		69					
	11.8	507.3	13.9	97.4	-185	87	3.43	3.00	2.73	2.65
Study Period	29.8		45.1		-1					
	1.8	949	4.9	1567	-526	87	25.20	14.23	9.46	6.39

Note: All seasonal concentrations represent an approximation of the average event mean concentration, except during the winter, when inlet samples were manually collected at a single point in time near the end of the runoff event.

Concentrations of chloride above 250 mg/L have been shown to adversely effect aquatic life (Environment Canada, 2000). This level was exceeded at the outlet during most runoff events in the winter, but by spring, concentrations had declined to levels below 97 mg/L.

Over the treatment period, mean chloride concentrations within the facility were similar (Figure 5.5). The low concentration at the inlet may be a function of the grab sampling method and storm selection process discussed above. In general, mean concentrations at the outlet were less than at the greenhouse and wetland stations. However, none of the observed concentrations, either among monitoring stations or sampling days, were significantly different at the 95% level of confidence.

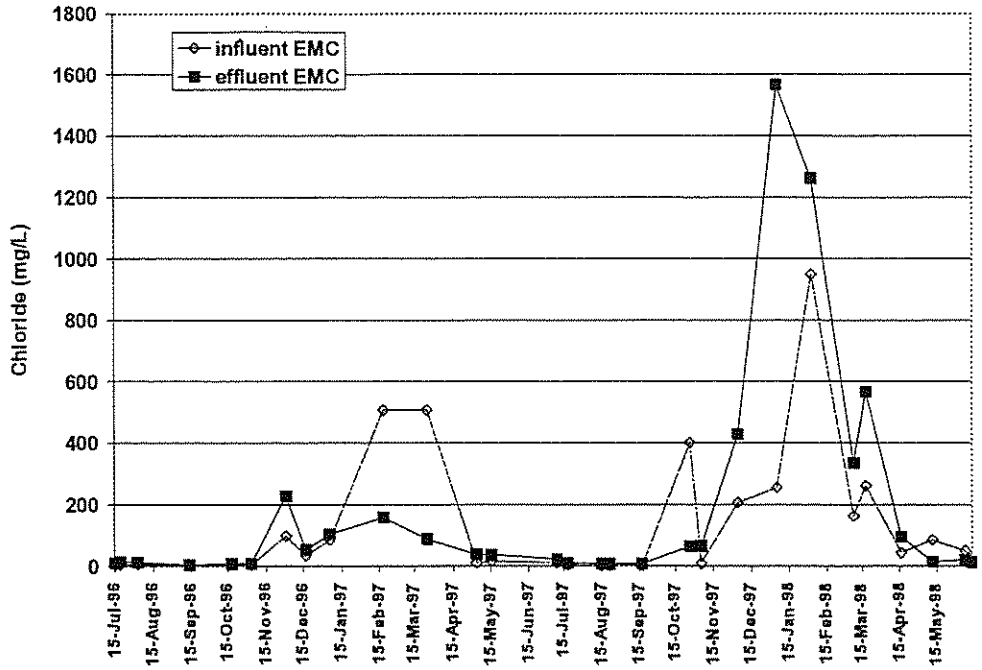
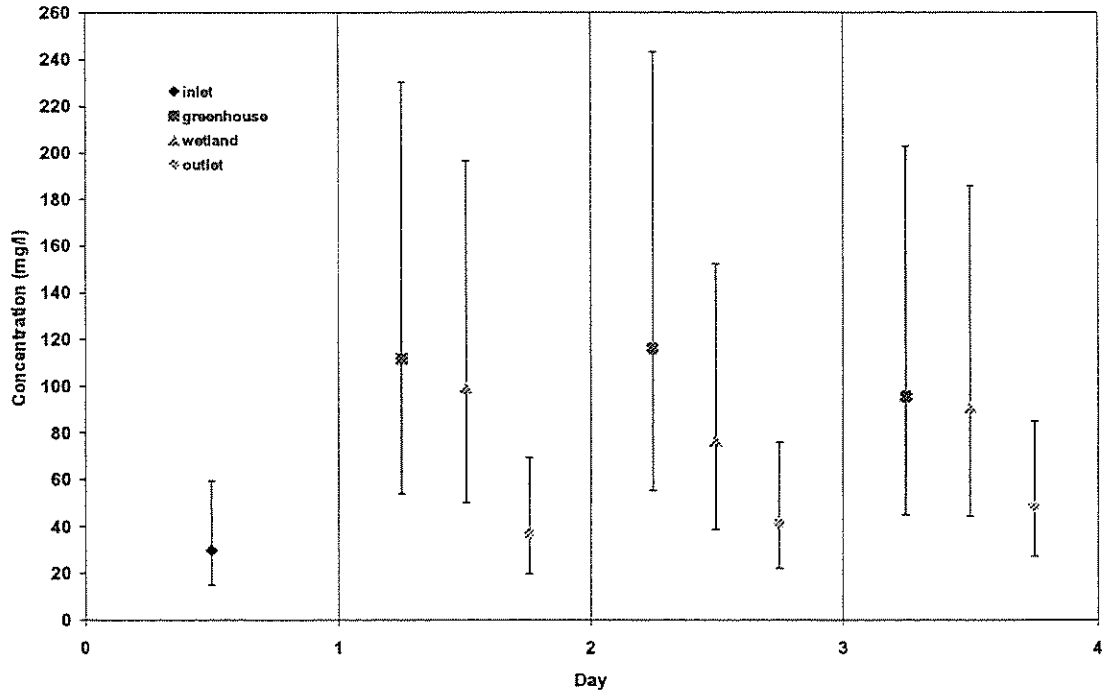


Figure 5.4: Chloride influent and effluent concentrations over the study period



Note: The duration of influent sampling (day 0 to 1) varied among events.

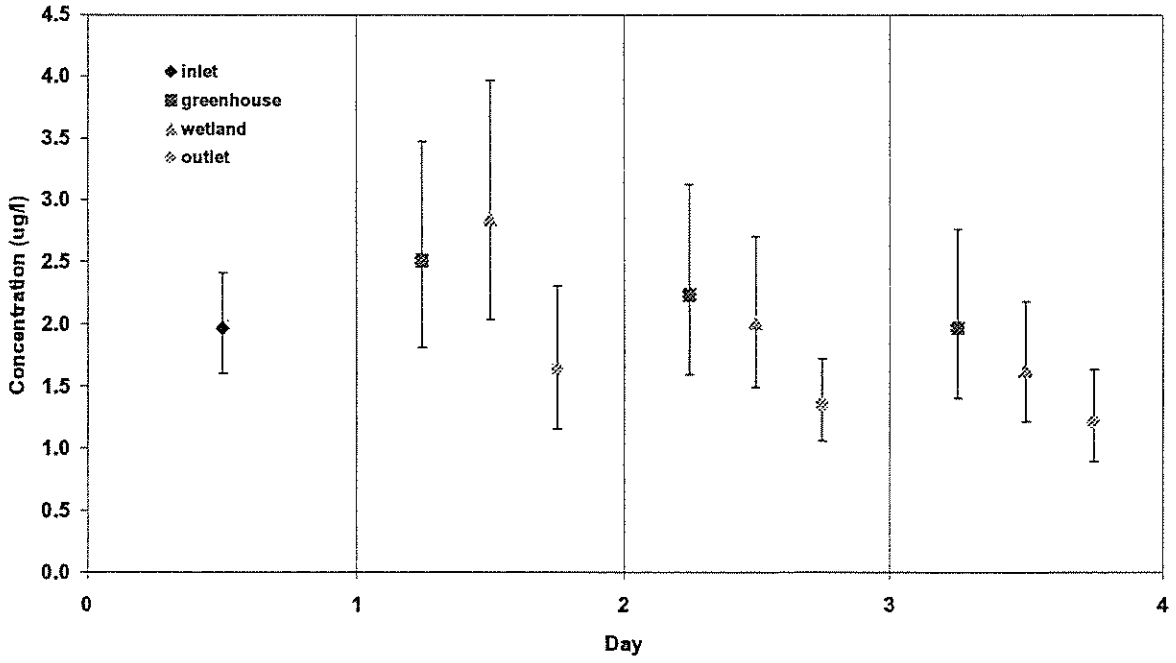
Figure 5.5: Mean chloride concentrations and 95% confidence intervals at the inlet, greenhouse, wetland and outlet monitoring stations.

5.3.5 Phenols

Phenols are organic byproducts of petroleum refining, tanning, textile dyeing and other manufacturing processes, but may also be released from aquatic plants and decaying vegetation. As germicides they act as disinfectants and, at high concentrations, are toxic to humans and aquatic organisms.

Table 5.3: Phenols influent/effluent concentrations, facility performance and mass input and retention

Phenolics	**Infl. Conc. (µg/L)**		**Eff. Conc. (µg/L)**		**Performance (%)**		**Mass Input and Cumulative Mass Retention (kg)**			
	Min	**Max**	**Min**	**Max**	**Min**	**Max**	**Event**	**Day - 1**	**Day - 2**	**Day - 3**
Summer	1.8		1.8		-6.4					
	1.2	4.2	1.1	11.6	-95	66	70.20	25.95	13.46	-1.161
Fall	2.2		1.1		55					
	1.4	4.8	0.7	1.9	41	73	59.26	42.96	36.57	35.43
Winter	2.1		1.4		21					
	2.0	2.2	1.2	1.7	14	44	48.85	35.85	32.07	16.68
Spring	2.1		1.3		48					
	1.6	2.7	1.2	1.5	26	64	35.07	27.94	20.08	19.12
Study Period	2.0		1.5		26					
	1.2	4.8	0.7	11.6	-95	73	213.4	132.7	102.1	70.1



Note: The duration of influent sampling (day 0 to 1) varied among events.

Figure 5.6: Mean phenol concentrations and 95% confidence intervals at the inlet, greenhouse, wetland and outlet monitoring stations.

Mean effluent concentrations were always above the Provincial Water Quality Objective/Guideline (OMOEE, 1994b) of 1.0 µg/L (Table 5.3). Unlike other constituents, summer removal of phenols was less than in other seasons. Performance over the entire study period was 26%.

Mean influent and effluent phenol concentrations did not vary greatly over the study period. Inspection of confidence intervals in Figure 5.6 shows that mean concentrations among monitoring stations were not significantly different.

5.3.6 Oil and grease

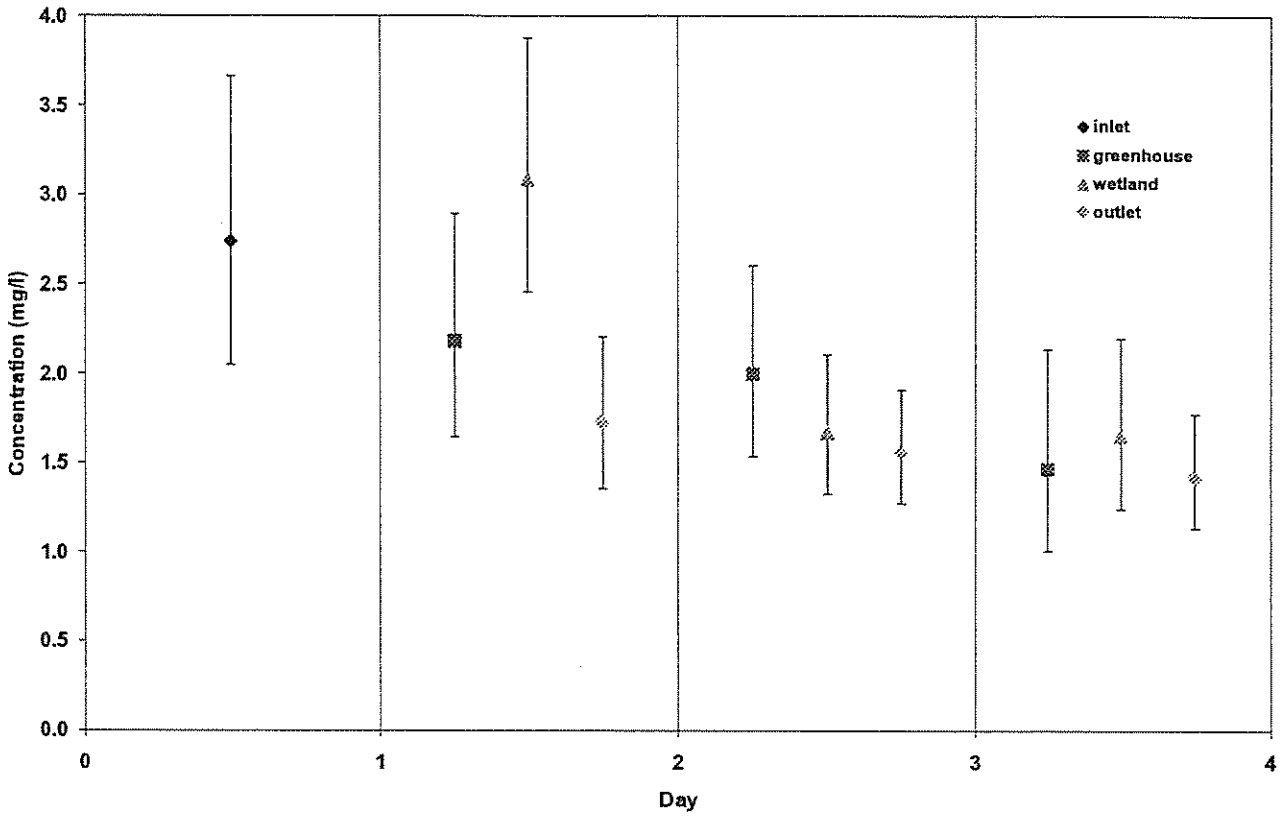
Oil and grease (solvent extractables) contains hydrocarbon compounds toxic to aquatic life at relatively low concentrations. These compounds are lighter than water but have a strong affinity for surfaces (particularly hydrophobic or poorly-wetted surfaces) and are therefore removed partly by attachment to sediments, but also by attachment to plant and other surfaces in the wetland. Removal efficiencies ranged from 25% in the winter to 82% in the spring (Table 5.4). Seasonal average effluent concentrations ranged from 0.7 to 1.5 mg/L. Concentrations were below analytical detection limits in 10% of samples collected.

Table 5.4: Oil and grease influent/effluent concentrations, facility performance and mass input and retention

Oil & Grease	**Infl. Conc. (mg/L)**		**Effl. Conc. (mg/L)**		**Performance (%)**		**Mass Input and Cumulative Mass Retention (kg)**			
	Min	**Max**	**Min**	**Max**	**Min**	**Max**	**Input**	**Day - 1**	**Day - 2**	**Day - 3**
Summer	1.9		1.4		44					
	0.2	7.5	0.6	2.5	-165	83	69.07	50.11	44.64	34.72
Fall	2.2		0.7		78					
	0.2	9	0.2	2.8	-70	94	82.74	67.34	61.54	60.71
Winter	1.6		1.5		25					
	0.2	5	0.6	2.8	-42	48	88.07	56.93	42.48	31.12
Spring	4.3		1.4		82					
	1.5	11	0.6	3.6	-12	91	142.31	128.79	121.14	118.98
Study Period	2.3		1.2		61					
	0.2	11	0.2	3.6	-166	94	382.2	303.2	269.8	245.5

Note: All seasonal concentrations represent an approximation of the average event mean concentration, except during the winter, when inlet samples were manually collected at a single point in time near the end of the runoff event.

Mean concentrations at the greenhouse, wetland and outlet decreased continuously over the treatment period (Figure 5.7). On the second and third days of treatment, mean outlet concentrations were similar to those at the greenhouse and wetland. Oily sheens on the water surface were frequently observed during the treatment period. Since sampler intake strainers were located below the water surface, the automated sampling equipment may not have captured these surface residues.



Note: The duration of influent sampling (day 0 to 1) varied among events.

Figure 5.7: Mean oil and grease concentrations and 95% confidence intervals at the inlet, greenhouse, wetland and outlet monitoring stations.

5.4 Nutrients

Nutrients such as nitrogen (N) and phosphorus (P) originate mainly from organic and inorganic fertilizer application to lawns, gardens and agricultural fields, as well as from animal waste and atmospheric sources. Some of these nutrients are cycled internally within the wetland and accumulate in the sediments for extended periods of time as organic material and fallen litter (Tanner, 2001). Excess nutrients cause eutrophication of surface waters by stimulating algal and aquatic plant growth, which deplete oxygen when they die resulting in fish kills and stress to bio-communities. As the limiting nutrient for plant growth in most inland waters, phosphorus removal plays an important role in rehabilitating water bodies altered by excessive nutrient inputs. The total phosphorus threshold (OMOEE, 1994b), beyond which excessive plant growth in rivers and streams occurs, is currently set at 0.03 mg/L in Ontario.

5.4.1 Phosphorus

Samples were analyzed for total phosphorus (TP) and reactive ortho-phosphate (OP). A large fraction of TP is in the particulate phase, of which only a portion is considered to be bioavailable (Sharpley et al., 1992). In contrast, OP is mostly soluble and considered immediately available for algal growth. Phosphorus removal from water in wetlands occurs through plant and microbial uptake; precipitation of aluminum, iron and calcium phosphates, and adsorption by sediment, organic matter and aluminum and iron oxides and hydroxides (Richardson, 1985).

Average seasonal phosphorus concentrations and load-based removal efficiencies are presented in Table 5.5. Average influent concentrations of TP were lowest during the winter (0.21 mg/L) and highest during the spring (0.36 mg/L) and fall (0.41 mg/L). Average effluent concentrations were lowest during the summer (0.10 mg/L) and highest during the fall (0.18). All seasonal averages were above the 0.03 mg/L provincial guideline for TP (OMOEE, 1994b).

Table 5.5: Total phosphorus (TP) and orthophosphorus mean influent/effluent concentrations, removal efficiencies and mass input and retention

TP	**Inf. Conc. (mg/L)**		**Eff. Conc. (mg/L)**		**Performance (%)**		**Mass Input and Cumulative Mass Retention (kg)**			
	Min	**Max**	**Min**	**Max**	**Min**	**Max**	**Event**	**Day - 1**	**Day - 2**	**Day - 3**
Summer	0.30		0.10		83					
	0.08	1.08	0.04	0.19	-90	91	19.1	16.8	16.3	16.0
Fall	0.41		0.18		64					
	0.12	2.15	0.07	0.32	13	92	18.0	13.7	12.2	11.9
Winter	0.21		0.16		8					
	0.13	0.33	0.09	0.38	-16	45	11.2	4.78	3.86	2.23
Spring	0.36		0.16		56					
	0.18	1.03	0.09	0.36	18	91	10.44	9.06	6.73	6.38
Study Period	0.31		0.14		58					
	0.08	2.15	0.04	0.38	-90	92	58.79	44.33	39.15	36.47
Orthophosphate										
Summer	0.11		0.02		86					
	0.04	0.44	0.01	0.08	38	98	7.43	6.73	6.63	6.40
Fall	0.13		0.09		31					
	0.03	0.34	0.02	0.25	-111	72	6.34	3.41	2.38	2.18
Winter	0.11		0.07		26					
	0.07	0.2	0.03	0.21	14	68	7.14	3.38	2.81	2.56
Spring	0.10		0.06		14					
	0.08	0.24	0.02	0.23	-173	88	2.99	2.30	0.87	0.72
Study Period	0.11		0.05		44					
	0.02	0.44	0.01	0.25	-173	98	23.90	15.82	12.69	11.86

Note: All seasonal concentrations represent an approximation of the average event mean concentration, except during the winter, when grab samples were collected at the inlet near the end of influent runoff.

TP was removed most effectively during the summer, probably due in part to phosphorus sequestering by plants. The reduction in phosphorus was also more rapid in the summer, with over 90% of phosphorus removed within the first day of treatment. Winter removal of phosphorus was only 8%, but effluent concentrations were not significantly higher than other seasons, suggesting that, like TSS (Figure 5.1), relatively low influent concentrations during the winter influenced removal rates.

Orthophosphate mean effluent concentrations ranged from 0.02 mg/L in the summer to 0.09 mg/L in the fall. The large initial reduction on the first day of treatment during the summer was similar to that observed for TP. An increase in effluent concentrations was observed during the fall and winter in 1997, and during the fall in 1998 (Figure 5.8). This trend of increased OP effluent concentrations during the fall has been noted in other wetland treatment systems and is usually attributed to the release of phosphorus as plants and microbes die (Tanner, 2001; Richardson, 1985). The trend of phosphorus release in the fall is also displayed by inlet and outlet OP:TP ratios. The fall effluent OP:TP ratio (0.5) was not only greater than that of other seasons, but also increased more significantly from the inlet to the outlet. Although release of nutrients during the cold season results in poorer performance, the effect is relatively minor since storage of nutrients in live plant tissues typically accounts for only a small fraction of total nutrient removal (Tanner, 2001).

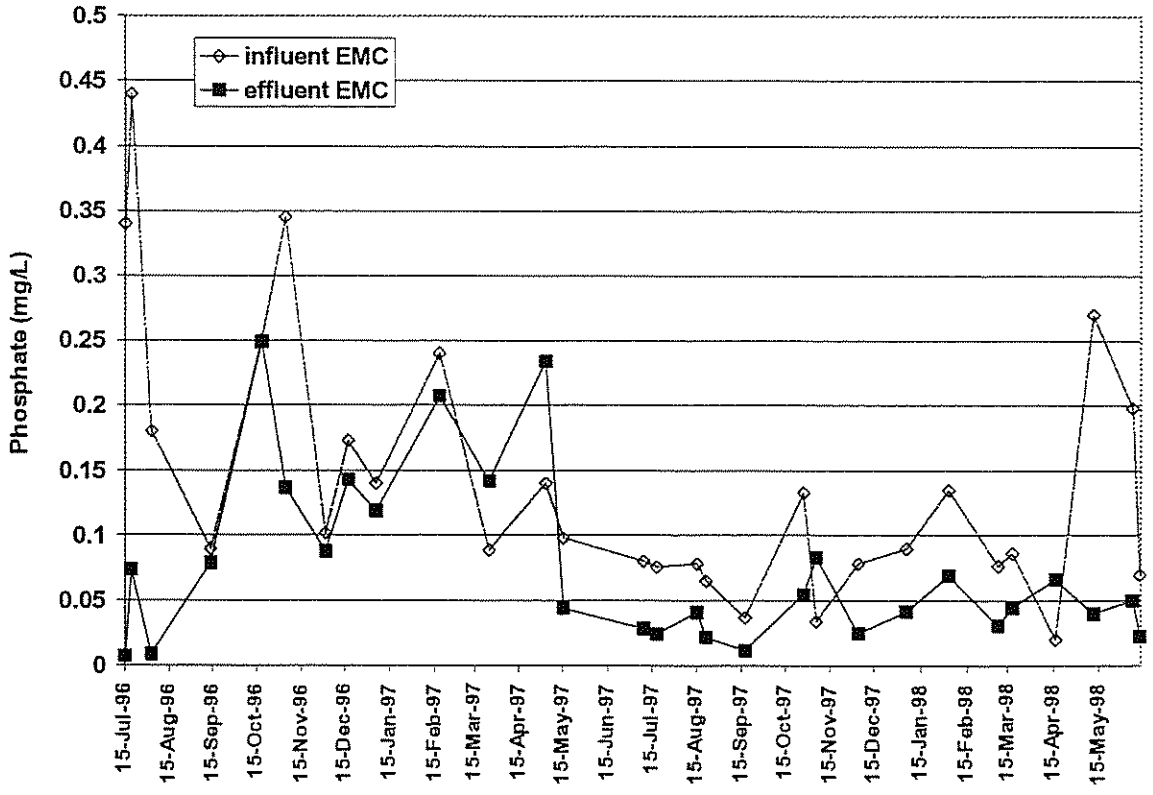
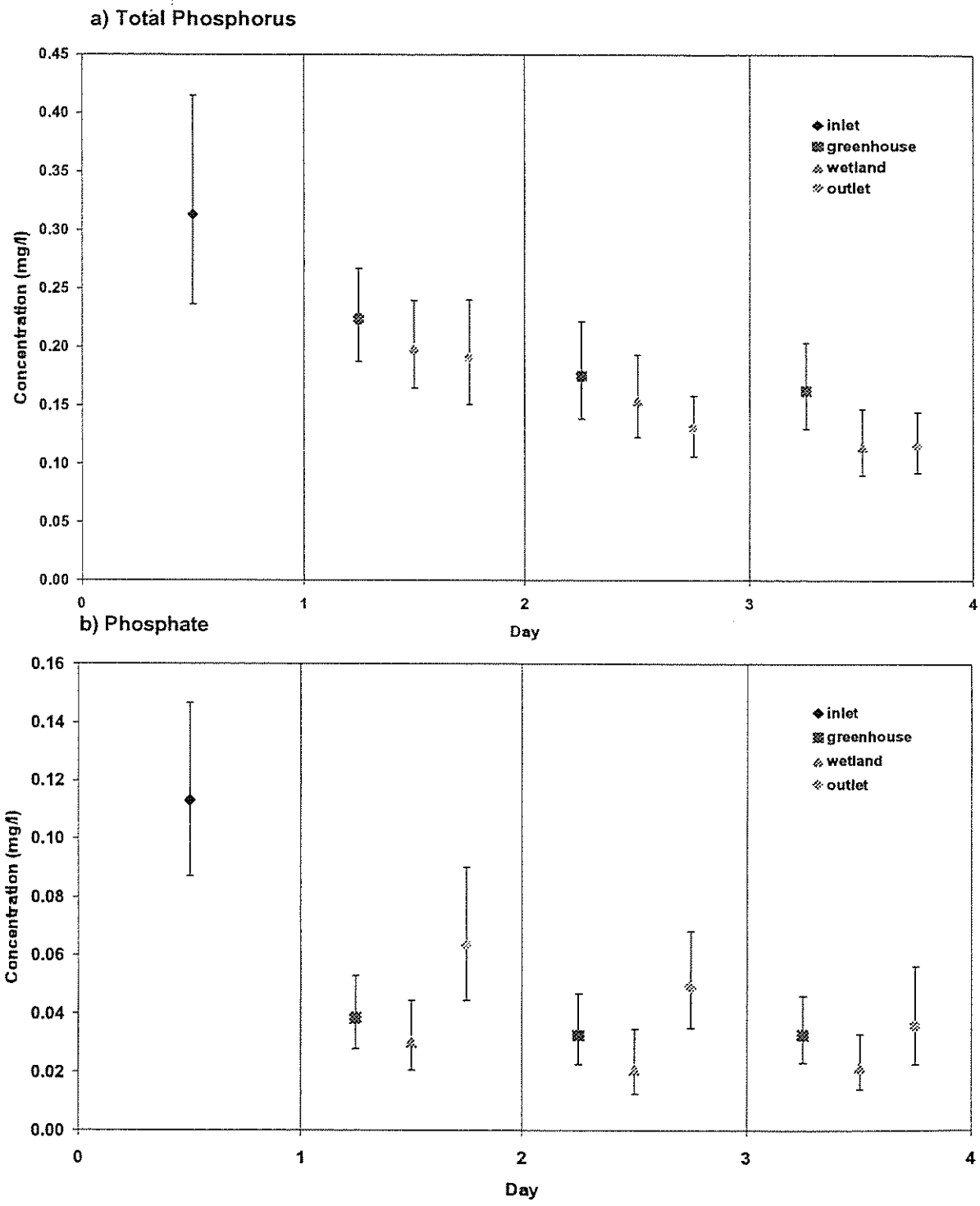


Figure 5.8: Mean influent and effluent orthophosphate concentrates during the study period



Note: Variable influent runoff durations are represented on the graph as day 0 to 1.

Figure 5.9: Mean TP (a) and ortho-phosphate (b) concentrations and 95% confidence intervals at the inlet, greenhouse, wetland and outlet monitoring stations.

Average TP and OP concentrations over the treatment period are illustrated in Figure 5.9 (parts a and b). In general, the reduction in TP concentrations over the treatment period at the outlet, wetland and greenhouse were similar. Although most of the reduction in average TP concentrations occurred after the first day of treatment, the decline in concentrations was more gradual than observed for TSS (Figure 5.2). Only by the second day of treatment were concentrations at the greenhouse, wetland and outlet significantly lower than influent concentrations at the 95% level of confidence.

5.4.2 Nitrogen

Cycling of nitrogen in wetland systems is a complex process involving mineralization of organic nitrogen to ammonium (NH_4^+), followed by nitrification of ammonium to nitrite (NO_2^-) and nitrate (NO_3^-). Nitrate may be subsequently assimilated by plants or microbes, and when nitrate diffuses into anoxic water, anaerobic bacteria reduce nitrate (denitrification) to nitrogen gas (N_2). A small amount of ammonia may also be converted to ammonia gas (NH_3), which dissolves in the soil pore water and combines with protons to form the ammonium ion. Most nitrogen absorbed by plants is in the form of nitrate, although plants are also able to synthesize plant proteins directly from the ammonium ion. Upon the death of plants, nitrogen compounds are returned to the soil where they are reprocessed by soil microbes, translocated to, or taken up by roots, washed away in drainage water, or denitrified to nitrogen gas or nitrous oxide (Raven *et al.*, 1992). The biological and chemical nitrification/denitrification processes account for the majority of nitrogen removal observed in constructed wetlands (*e.g.* Jansson *et al.*, 1994, Gilliam, 1994). Unlike phosphorus, only approximately 10 to 20% of nitrogen is transported in particulate form (Vought *et al.*, 1994)

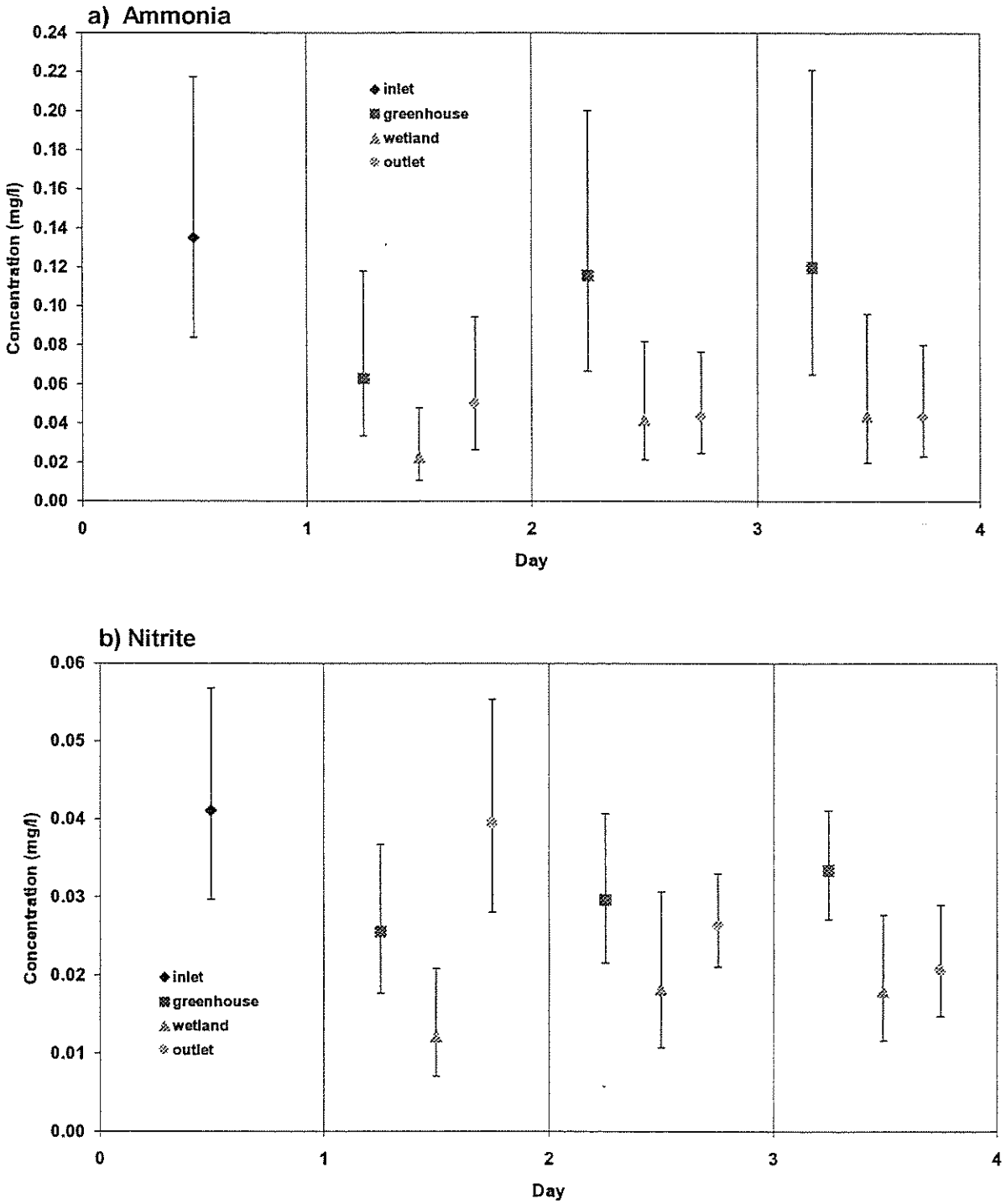
Samples at all stations were analyzed for Total Kjeldahl Nitrogen (TKN), total ammonia (NH_3-N + NH_4-N), nitrite and nitrate. TKN is the sum of total ammonia and organic nitrogen. Concentrations, removal efficiencies and mass input and retention over a three day period for nitrogen compounds are presented in Table 5.6. Mean influent and effluent concentrations and 95% confidence intervals over three days of treatment are shown in Figure 5.10 (a to d). In general, mean concentrations for nitrogen compounds were low relative to other studies. Nitrogen influent concentrations were greatest in the spring, when the majority of organic and inorganic fertilizers are applied to lawns and agricultural fields. Although average effluent concentrations of TKN, nitrite and total ammonia were also greater during the spring, the unionized ammonia effluent concentration, based on the temperature and pH of the water, was consistently less than the 0.02 mg/L PWQO for this constituent (OMOEE,1994b). Total ammonia was also less than the CCME (2001) guideline of 1.37 mg/L during all seasons and at all monitoring stations.

Removal rates over the entire study period ranged from 22% for nitrite to 63% for total ammonia. Nitrate removal efficiency was highest, and nitrate effluent concentrations were lowest during the summer (Figure 5.11). Low dissolved oxygen associated with high rates of microbial respiration during the warmer summer temperatures (see section 5.3.1) may have contributed to nitrate removal through the process of denitrification, which occurs preferentially in oxygen limiting environments of bottom sediments. Unlike TP, total nitrogen (TKN + NO_2+NO_3) effluent concentrations did not increase significantly during the fall or

Table 5.6: Influent/effluent concentrations, removal efficiencies and mass input and 3-day cumulative retention for nitrogen compounds

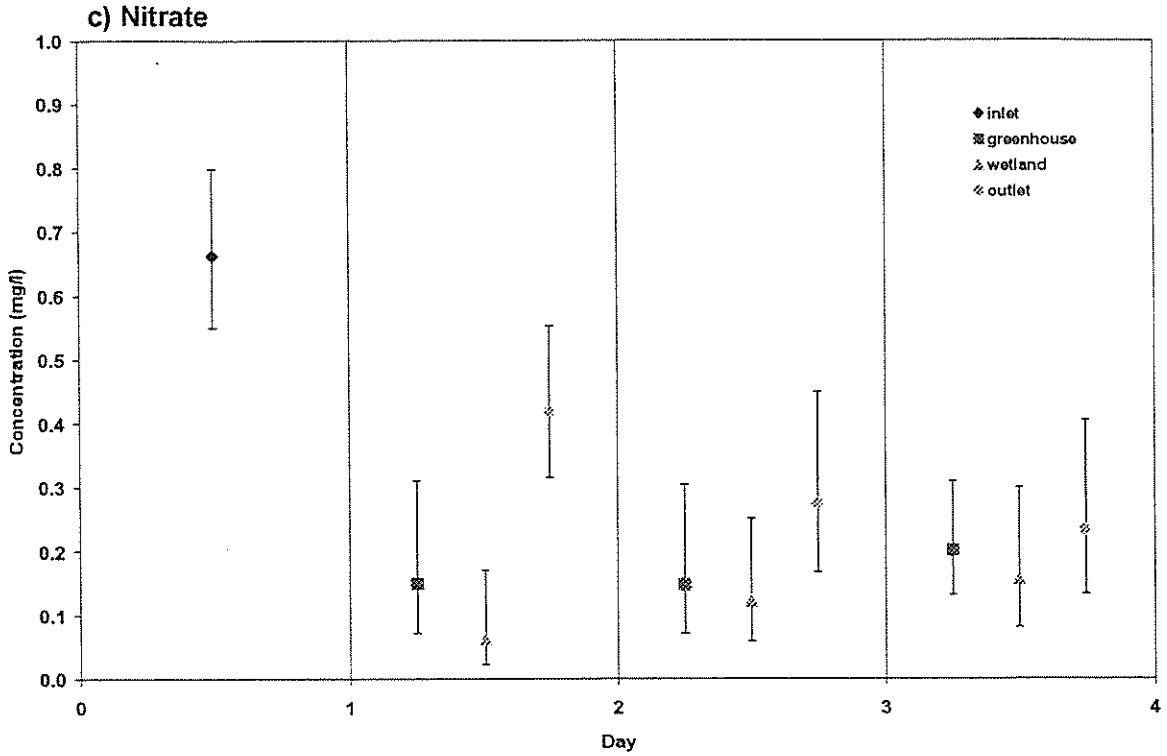
TKN	**Inf. Conc. (mg/L)**		**Effl. Conc. (mg/L)**		**Performance (%)**		**Mass Input and Cumulative Mass Retention (kg)**			
	Min	**Max**	**Min**	**Max**	**Min**	**Max**	**Event**	**Day - 1**	**Day - 2**	**Day – 3**
Summer	1.34		0.80		63					
	0.6	3.3	0.50	1.09	12	77	59.84	48.04	44.45	41.11
Fall	1.34		0.79		51					
	0.64	4.10	0.51	1.09	14	87	47.65	33.14	27.68	26.10
Winter	0.99		0.88		36					
	0.6	2.96	0.52	1.64	-52	45	76.06	46.75	41.05	33.77
Spring	2.06		1.19		52					
	1.3	9.5	0.66	2.03	-13	94	59.78	50.39	37.52	34.82
Study Period	1.40		0.90		49					
	0.6	9.5	0.50	2.03	-52	94	243.3	178.3	150.7	135.8
Ammonia + ammonium										
Summer	0.08		0.08		52					
	0.01	0.48	0.002	0.44	-52	100	6.028	5.660	5.564	3.115
Fall	0.09		0.05		40					
	0.05	0.13	0.03	0.12	-2	79	3.236	2.013	1.591	1.482
Winter	0.10		0.05		50					
	0.01	0.7	0.003	0.32	-48	95	15.18	9.746	8.756	8.526
Spring	0.50		0.17		79					
	0.18	5.65	0.04	0.42	-10	99	21.27	19.39	17.70	17.09
Study Period	0.14		0.06		63					
	0.01	5.65	0.002	0.44	-52	100	45.71	36.81	33.28	30.22
Nitrite										
Summer	0.03		0.02		31					
	0.002	0.10	0.008	0.20	-1159	92	2.07	0.78	0.69	0.63
Fall	0.03		0.03		11					
	0.02	0.05	0.01	0.06	-200	63	0.98	0.38	0.24	0.19
Winter	0.03		0.03		-25					
	0.02	0.04	0.02	0.06	-58	39	1.520	0.903	0.695	-0.005
Spring	0.09		0.05		52					
	0.03	0.22	0.02	0.11	-53	86	2.25	1.72	1.34	1.27
Study Period	0.04		0.03		22					
	0.002	0.22	0.008	0.20	-1159	92	6.82	3.80	2.97	2.09
Nitrate										
Summer	0.72		0.08		82					
	0.29	2.06	0.003	0.57	60	100	32.98	32.38	32.08	31.74
Fall	0.46		0.50		11					
	0.24	0.84	0.25	0.71	-76	60	15.53	8.362	4.650	3.589
Winter	0.70		0.63		27					
	0.43	1.09	0.43	0.89	-23	52	35.52	23.03	19.14	16.75
Spring	0.82		0.36		26					
	0.37	1.45	0.04	0.94	-120	96	19.86	14.13	8.032	7.094
Study Period	0.66		0.28		41					
	0.24	2.06	0.003	0.94	-120	100	103.9	77.92	63.91	59.18

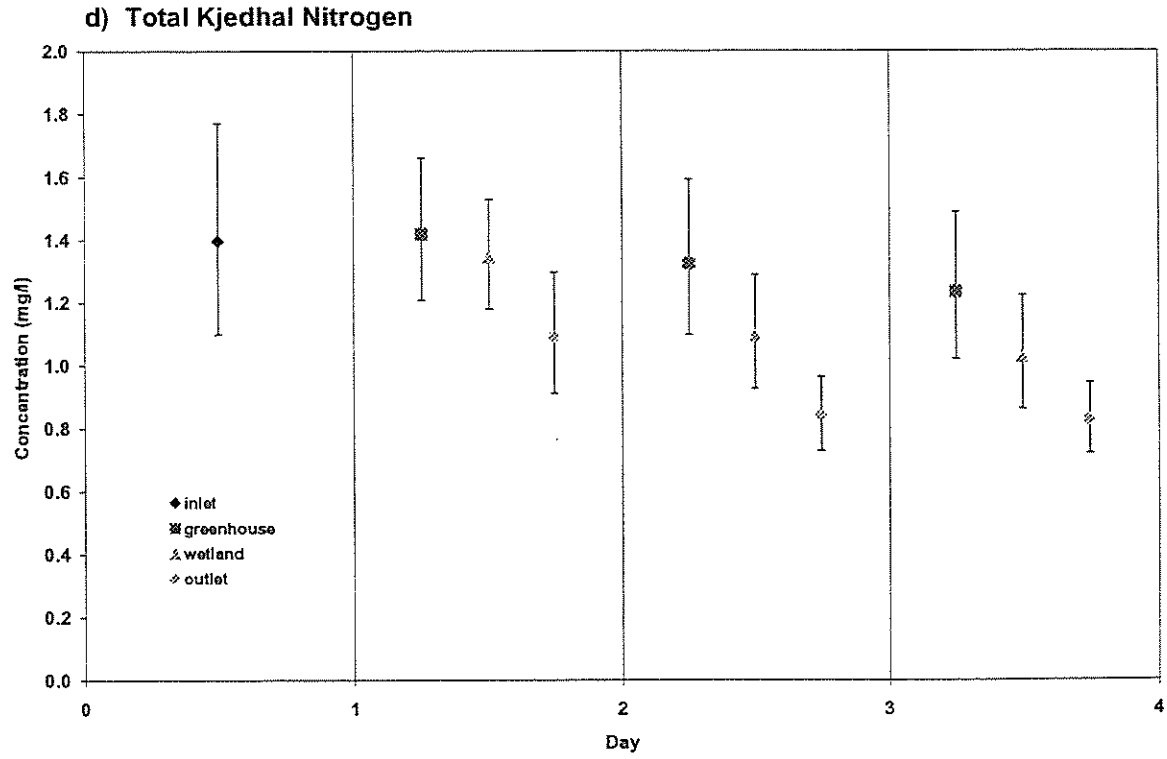
Note: All seasonal concentrations represent an approximation of the average event mean concentration, except during the winter, when inlet samples were manually collected at a single point in time near the end of the runoff event.



Note: Variable influent runoff durations are represented on the graph as day 0 to 1.

Figure 5.10: Mean ammonia (a), nitrite (b), nitrate (c) and TKN (d) concentrations and 95% confidence intervals at the inlet, greenhouse, wetland and outlet monitoring stations.





Note: Variable influent runoff durations are represented on the graph as day 0 to 1.

Figure 5.10 (continued): Mean ammonia (a), nitrite (b), nitrate (c) and TKN (d) concentrations and 95% confidence intervals at the inlet, greenhouse, wetland and outlet monitoring stations.

winter when wetland plants were dormant. The absence of nutrient release during senscence may be explained by internal recycling of nutrients assimilated during the active growth phase through translocation to below-ground tissues (rhizomes, bulbs, corms) (Tanner, 2001).

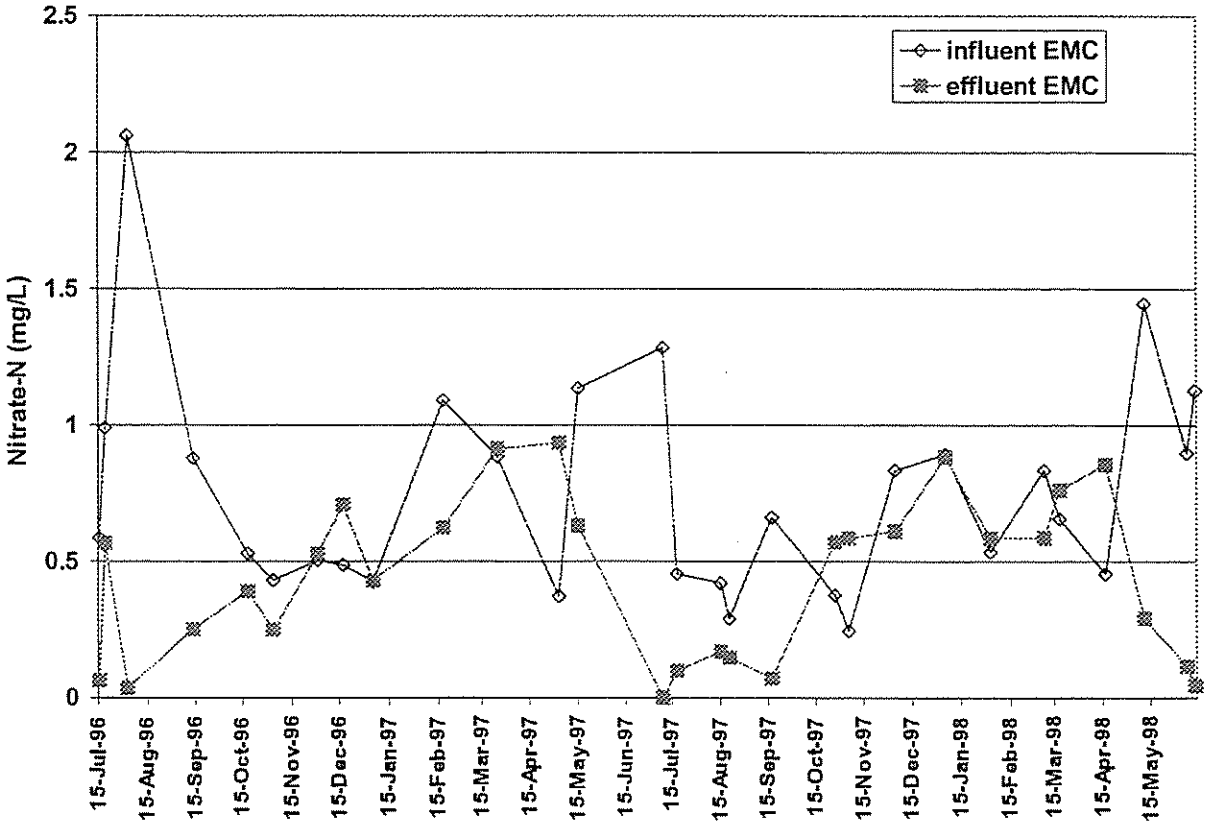


Figure 5.11: Relationship between influent and effluent nitrate concentrations over the study period.

5.5 Metals

Most metals originate from roadways and roofing materials, and are usually present in particulate and highly complexed forms. Although aquatic organisms require trace amounts of some heavy metals, such as copper, iron, cobalt, vanadium and zinc, these and other metals can be toxic to living organisms, even at very low concentrations. The dominant pathway for most metals is accumulation in sediments. Relatively small quantities of metal loads accumulate in standing vegetation (*eg.* Nolte and Assoc., 1999).

Sixteen metals were analyzed in this study, including copper, zinc, lead, cadmium, chromium and nickel. A list of all metals and their respective detection frequencies is provided in Table 5.7. Statistical summaries for these metals are provided in Appendices E and F.

Constituents with effluent concentration detection frequencies of less than 50% include nickel, cadmium, cobalt, lead, molybdenum and beryllium. Influent detection frequencies for the same metals were higher, but a significant number of samples still had concentrations below laboratory detection limits. Low concentrations of these metals made the reliability of removal efficiencies difficult to authenticate. Copper, chromium, zinc and iron are common stormwater pollutants that had detection frequencies above 60%. Results for these four metals are presented in Table 5.8 and Figure 5.12.

Table5.7: Detection frequencies (%) for metals analyzed in this study

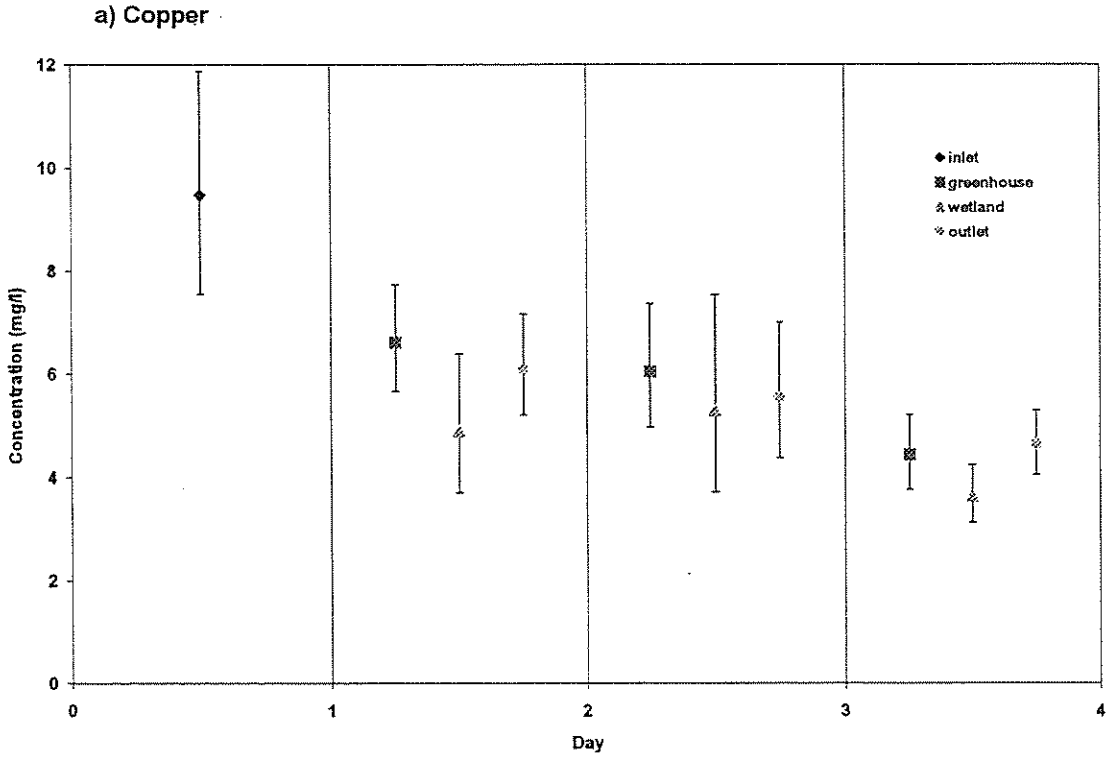
	RMDL (µg/L)	Influent Conc.	Greenhouse	Wetland	Outlet	Effluent Conc.
Copper	1.6	100	100	98	100	100
Nickel	1.3	69	73	60	43	38
Zinc	0.6	100	100	100	100	100
Cadmium	0.6	62	13	6	13	14
Cobalt	1.3	38	20	11	6	3
Chromium	1.4	76	64	43	54	65
Lead	10	10	5	1	2	3
Iron	0.8	100	100	100	100	100
Manganese	0.2	100	99	99	99	100
Aluminum	11	100	100	100	100	100
Vanadium	1.5	76	64	50	56	59
Molybdenum	1.6	0	0	0	0	0
Barium	0.2	100	100	100	100	100
Beryllium	0.02	76	70	62	49	41
Strontium	0.1	100	100	100	100	100
Titanium	0.5	100	100	100	100	100

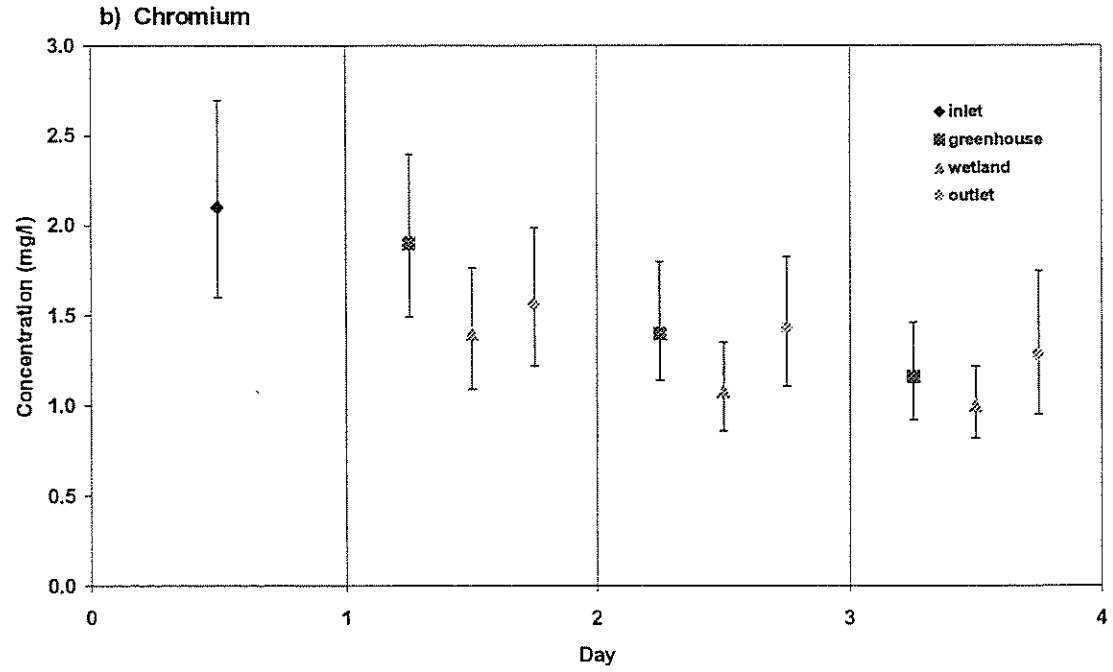
Load based removal efficiencies for the entire study period were 58, 36, 17 and 41% for copper, chromium, zinc and iron, respectively. In general, removal efficiencies were greatest in the summer and spring, and least during the winter. The presence of metals in decomposing plant tissues may have contributed to lower removal efficiency during the winter. Elevated winter salinity levels, which promote desorption of some metals from sediments, may have had a similar effect. In general, winter data are less reliable because influent samples were collected by the grab method, and therefore do not represent event mean concentrations.

Table 5.8: Mean influent/effluent concentrations, removal efficiencies and mass input and 3-day cumulative retention for copper, chromium, zinc and nickel

Copper	**Inf. Conc. (µg/L)**		**Effl. Conc. (µg/L)**		**Performance (%)**		**Mass Input and Cumulative Mass Retention (g)**			
	Min	**Max**	**Min**	**Max**	**Min**	**Max**	**Event**	**Day - 1**	**Day - 2**	**Day - 3**
Summer	11.0		4.8		69					
	3.4	37.2	3.0	6.6	-23	92	448.3	364.4	341.7	323.7
Fall	9.4		5.0		57					
	5.2	21.1	4.1	8.4	8	79	295.52	221.42	193.02	182.18
Winter	7.5		5.8		37					
	4.8	14.1	4.0	9.4	-28	49	377.0	259.4	217.4	181.3
Spring	9.6		5.7		66					
	4.1	20.1	2.6	28.3	-66	78	331.5	294.5	246.2	237.8
Study Period	9.5		5.3		58					
	3.4	37.2	2.6	28.3	-66	92	1452.4	1139.7	998.3	924.6
Chromium										
Summer	1.8		1.0		58					
	0.7	4.4	0.7	2.5	10	86	96.0	67.0	61.0	58.4
Fall	2.4		1.8		32					
	1.6	4.5	0.7	2.9	-24	67	84.7	49.7	36.1	31.6
Winter	2.6		2.5		20					
	0.7	7.8	1.7	7.2	-152	40	149.8	93.1	72.1	47.9
Spring	1.6		1.2		50					
	0.7	4.2	0.7	2.3	-18	71	68.4	57.1	43.6	41.1
Study Period	2.0		1.5		36					
	0.7	7.8	0.7	7.2	-152	86	398.9	266.9	212.7	179
Zinc										
Summer	22.5		23.9		53					
	1.6	99.4	15.2	63.5	-1033	86	1395.9	993.0	857.9	796.7
Fall	41.3		26.3		58					
	19.0	102.0	15.7	42.2	-138	83	1339.5	990.5	875.7	823.5
Winter	33.9		63.8		-87					
	20.5	74.3	11.3	192.7	-404	45	1540.9	394.5	-99.9	-527.6
Spring	49.3		30.4		59					
	21.6	162.0	16.6	127.6	-23	86	1412.5	1133.3	936.2	902.8
Study Period	34.3		31.8		17					
	1.6	162.0	11.3	192.7	-1033	86	5688.9	3511.4	2570.0	1995.6
Iron										
Summer	508		386		52					
	12	1920	130	1081	-886	80.0	36.6	23.8	21.3	19.7
Fall	673		481		31					
	359	1480	200	1046	-11	74	23.3	12.3	9.0	8.0
Winter	487		470		6					
	312	707	388	575	-39	34	22.4	12.9	99.7	42.3
Spring	757		353		65					
	357	1560	188	649	51	80	25.3	22.2	18.4	17.8
Study Period	593		415		41					
	12	1920	130	1081	-886	80	109.6	71.21	58.7	49.8

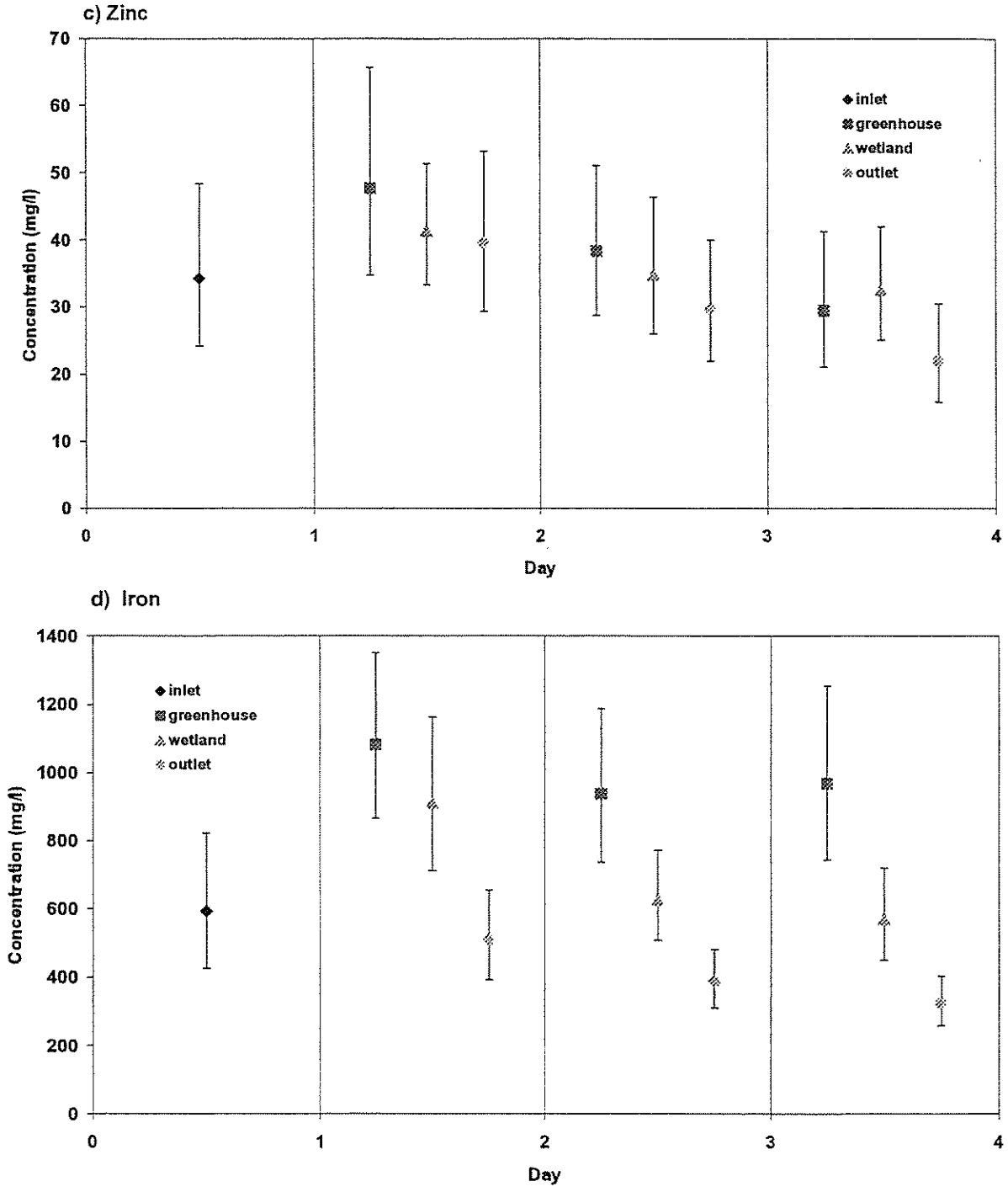
Note: All seasonal concentrations represent an approximation of the average event mean concentration, except during the winter, when inlet samples were manually collected at a single point in time near the end of the runoff event.





Note: Variable influent runoff durations are represented on the graph as day 0 to 1.

Figure 5.12: Mean copper (a), chromium (b), zinc (c) and iron (d) concentrations and 95% confidence intervals at the inlet, greenhouse, wetland and outlet monitoring stations.



Note: Variable influent runoff durations are represented on the graph as day 0 to 1.

Figure 5.12 (continued): Mean copper (a), chromium (b), zinc (c) and iron (d) concentrations and 95% confidence intervals at the inlet, greenhouse, wetland and outlet monitoring stations.

Seasonal effluent concentrations for copper, zinc, and iron were mostly above their respective PWQOs (OMOEE, 1994b) of 5, 20 and 300 µg/L, respectively. An exception was the summer average event mean concentration for copper, which was 4.8 µg/L. Seasonal average effluent concentrations for chromium (all species) were greater than the 1.0 µg/L PWQO for hexavalent chromium, but well below the 100 µg/L PWQO for the more common trivalent form of this constituent. Chromium concentrations were also below the 20 µg/L Canadian Water Quality Guideline (CCME, 2001) for the protection of fish.

Effluent concentrations over the study period for other metals (e.g. vanadium, nickel, beryllium, cadmium) were also below their respective PWQOs. Since lead concentrations were frequently below the detection limit, and the limit for lead (10 µg/L) is above its PWQO (5 µg/L), guideline exceedance levels for this constituent could not be determined. Metals such as copper and zinc often have a strong affinity for solid surfaces, but in this study effluent concentrations of total suspended solids and individual metals were not well correlated.

Average copper concentrations at the greenhouse, wetland and outlet were significantly less than the average influent concentration by day 2 of the treatment period (Figure 5.12 a to d). . Chromium concentrations at the greenhouse and wetland were also significantly lower, but outlet and inlet concentrations were similar. The reverse was true for iron: outlet concentrations by day 3 were significantly less than the influent concentration, but daily concentrations at other stations were similar. Zinc concentrations at all stations were similar.

5.6 Bacteria

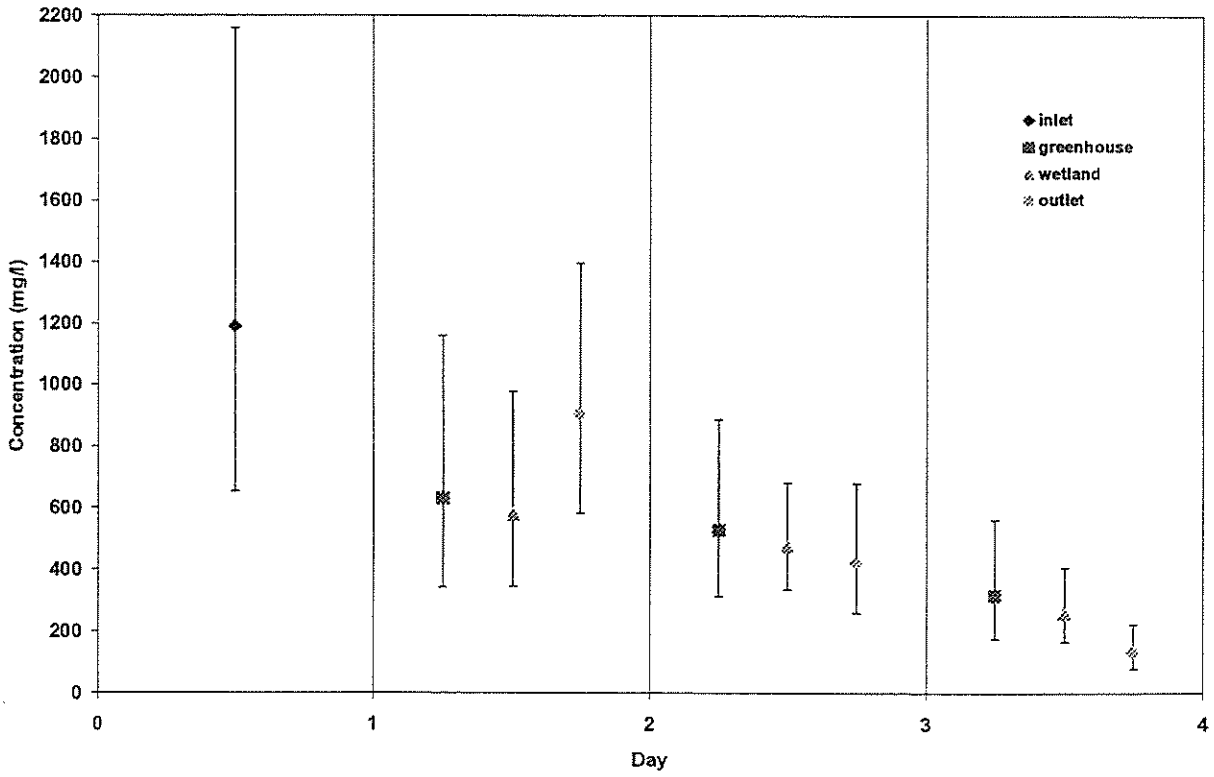
The *Escherichia Coli* group of bacteria is often used as a general indicator of the presence of fecal wastes and other bacteria harmful to humans. At high concentrations, these bacteria present a threat to the recreational use of receiving waters. *E.coli* originates primarily from organic fertilizers applied to agricultural fields, and fecal matter from dogs, racoons and other wildlife living in the facility or within the catchment or storm sewer system. Removal of *E.coli* occurs through sedimentation, predation and natural die off due to prolonged exposure to sunlight and other physical, chemical and biological conditions (Kachka and Turner, 1996). A residence time of 2 to 3 days is typically required for effective removal of these bacteria.

The sample size for *E.coli* (n = 24) was smaller than for other parameters because the laboratory requirement for rapid analysis following sample collection could not always be met. Since *E. coli* survival is temperature dependent; influent loading during the summer was much greater than during the winter (Table 5.9). Effluent concentrations ranged from 252 c/100 ml during the winter to 1107 c./100 ml during the fall. The removal efficiency for the study period was 84%, but effluent concentrations in all seasons were still above the 100 c./100 ml provincial threshold (OMOEE, 994b) for recreational use of waters.

Table 5.9: *E. coli* mean influent/effluent concentrations and removal efficiencies

E.coli	Infl. Conc. (c./100ml)		Eff. Conc. (c./100ml)		Performance (%)	
	Min	Max	Min	Max	Min	Max
Summer	4659		475		94	
	1780	24000	135	1351	74	97
Fall	941		1107		55	
	100	8600	563	3208	-656	80
Winter	278		252		42	
	40	780	57	616	-641	76
Spring	1518		319		68	
	410	5400	44	1115	20	98
Study period	1190		464		84	
	40	24000	44	3208	-656	98

Note: All seasonal concentrations represent an approximation of the average event mean concentration, except during the winter, when inlet samples were manually collected at a single point in time near the end of the runoff event.



Note: Variable influent runoff duration is represented on the graph as day 0 to 1

Figure 5.13: Mean *E. coli* concentrations and 95% confidence intervals at the inlet, greenhouse, wetland and outlet monitoring stations.

Daily effluent concentrations decreased gradually from day 1 to day 3 (Figure 5.13). Although substantial variation in influent concentrations over the study period resulted in wide confidence intervals, by day 3, average concentrations at all monitoring stations were significantly lower than the average influent concentration at the 95% confidence level.

5.7 Temperature

Mean monthly air, surface water and substrate temperatures in the greenhouse and wetland are presented in Figures 5.14 to 5.16, respectively. Surface water temperatures, taken about 10 cm above the ground surface, were recorded during wet and dry periods. Data were not available from early January to mid–February, 1996.

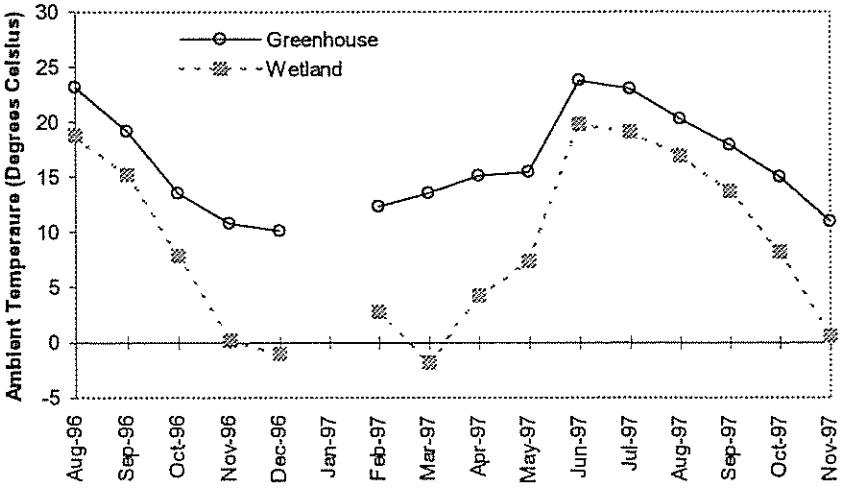


Figure 5.14: Mean monthly ambient temperatures for the greenhouse and external wetland.

Mean monthly ambient air temperatures were about 5°C warmer in the greenhouse than in the wetland during the summer. During winter, the temperature difference between the greenhouse and external wetland was greater than 10°C (Figure 5.14). By comparison, mean monthly surface water temperatures in the greenhouse were about 2°C warmer than the external wetland in the summer, and 5°C warmer in the winter (Figure 5.15). Substrate temperatures at 15 cm depth varied the least over the study period. Greenhouse and wetland substrate temperatures were similar during the winter, spring and summer. In the fall, greenhouse substrate temperatures were 2 to 3°C higher those in the wetland. Substrate temperatures never dipped below freezing in either the greenhouse or external wetland (Figure 5.16).

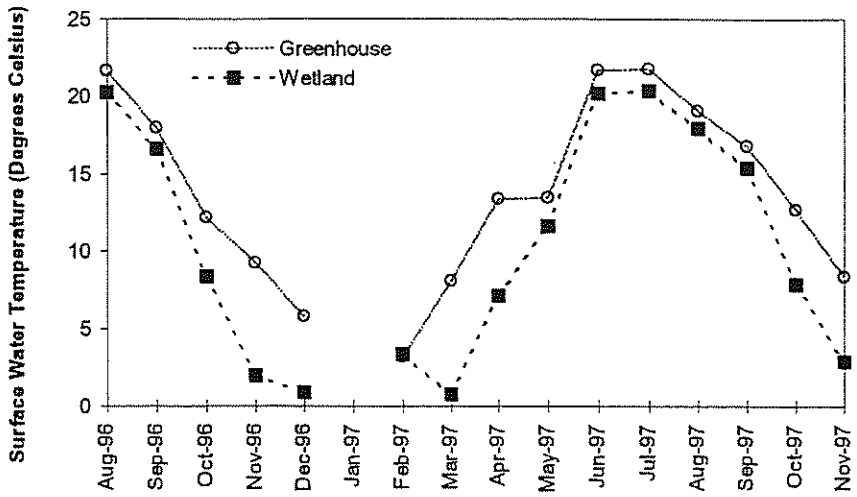


Figure 5.15: Mean monthly water temperatures recorded at 10 cm above the ground surface at the greenhouse and external wetland.

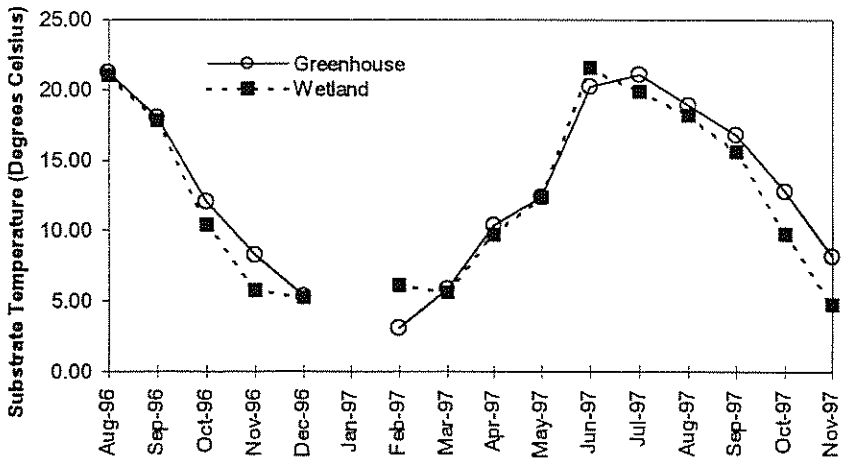


Figure 5.16: Mean monthly substrate temperatures recorded at 15 cm depth for the greenhouse and external wetland.

Table 5.10 shows mean daily air and water temperatures during storm events from August 1996 to November 1997. Greenhouse water temperatures were 1°C higher than the wetland during the summer, and almost 3°C higher during the fall. In the winter, despite significantly higher greenhouse air temperatures, the reverse was true: mean daily wetland water temperatures were higher than the greenhouse by 0.6°C. This trend may be largely attributed to a very large event on the 18th of February, during which daily external ambient temperatures reached a high of 15°C, and water temperatures in the external wetland were almost 4°C higher than those within the greenhouse. During spring, greenhouse water temperatures were an average of 4.6°C greater than the wetland. On an annual basis, the average temperature of water in the greenhouse was 1.8°C greater than outside water temperatures. In general, ambient air temperatures were reasonably well correlated to water temperatures (r between 0.78 and 0.84) at both monitoring stations.

Table 5.10: Mean daily seasonal and annual ambient and water temperatures (°C) in the greenhouse and wetland during sampled storms.

Station	**Season**	**Entire Event**		**Day 1**		**Day 2**		**Day 3**		**Day 4**	
		Air	H_2O	Air	H_2O	Air	H_2O	Air	H_2O	Air	H_2O
Greenhouse	Summer	18.8	19.0	20.3	19.1	19.9	19.6	19.3	19.6	18.8	18.4
Wetland		15.9	18.6	16.6	18.3	16.4	18.6	15.2	18.5	16.6	17.3
Greenhouse	Fall	12.7	8.6	11.7	8.0	11.4	7.6	11.2	8.0	12.0	8.7
Wetland		6.0	5.8	6.1	6.5	4.0	6.3	1.6	5.8	2.3	5.9
Greenhouse	Winter	12.9	1.6	12.1	1.6	10.0	1.8	13.6	1.7	13.5	1.7
Wetland		5.8	2.2	7.7	3.2	2.3	3.2	8.7	2.4	5.6	4.2
Greenhouse	Spring	14.2	9.7	14.0	9.0	13.8	9.2	12.6	8.5	17.9	11.9
Wetland		2.5	5.1	5.7	7.1	5.2	6.7	5.8	6.0	7.8	8.0
Greenhouse	Annual	14.6	9.7	14.5	9.4	13.8	9.5	14.2	9.5	15.5	10.2
Wetland		7.5	7.9	9.0	8.8	7.0	8.7	7.8	8.2	8.1	8.9

The maximum water temperature of 24°C during the summer was above the maximum acceptable to cold water fish species post-spawning and egg development growth stages (OMOEE, 1979). However, the maximum was well below the 30°C maximum discharge temperature set out in the Ontario Provincial Water Quality Objectives (OMOEE, 1994b). Unlike the situation in ponds, where effluent temperatures are strongly influenced by warming of the permanent pool during interevent periods, influent and effluent water temperatures in the Aurora wetland were not significantly different during storm events.

5.8 General Discussion of Water Quality Results

In general, for most constituents, observed annual removal efficiencies were within the range of efficiencies reported by other studies of wetland stormwater management systems (see Appendix C). Removal of TSS was particularly impressive (84%), but unlike other studies, pollutants with a strong affinity for suspended solids, such as phosphorus, zinc and copper, had significantly lower removal rates (17 to 58%). The weak correlation between TSS and pollutants typically associated with sediments (e.g. heavy metals) suggests that processes other than sedimentation may be acting on these constituents.

Another difference between TSS and other pollutants was in the rate at which average concentrations were reduced through the facility. Ninety percent of the total TSS load reduction occurred after the first day, whereas most metals and nutrients displayed a more gradual 3-day decline. This gradual decline for most pollutants highlights the importance of residence time in treatment performance.

This study differs from other wetland and wetpond studies in that the Aurora facility did not have a permanent pool, but instead drained 'dry' after each event. Water that entered the facility during a storm event either exfiltrated or was discharged from the facility over the 3 to 5 day drawdown period. Therefore, composite effluent concentrations for storm events are composed of water that entered the facility during the same events. By contrast, in wetlands or wet ponds with permanent pools, composite effluent samples collected over the duration of the outlet hydrograph are typically composed of displaced water and/or effluent flow generated by the current event and, as such, consist of a combination of current and past event water. Average residence times in these systems tend to be longer than in 'dry' ponds or wetlands because all or some of the water that enters the facility during an event (depending on the size of the permanent pool relative to the inflow volume) resides within the facility until the next event flushes it out, or it is gradually displaced by dry weather flow. Thus for similar sized 'wet' and 'dry' stormwater facilities, the 'wet' facility would typically provide for improved treatment.

As expected, performance over the winter period for most constituents was less than during other seasons. Winter load-based seasonal removal efficiencies did not exceed 50% for any of the parameters analyzed. A portion of this seasonal difference in efficiencies may be attributed to lower influent concentrations during the winter. As noted previously, since low influent concentrations are often associated with finer and less dense particle sizes, removal tends to be lower. Additional factors contributing to poor winter performance may include leaching of nutrients accumulated in plant tissues, slowed rates of microbial activity, less exfiltration of storm water, and lower filtration and sedimentation rates caused by ice formation on the wetland. If the facility had a permanent pool, redirection of water beneath the ice layer may have significantly enhanced performance. The grab method of sampling influent during the winter may be another important factor influencing winter performance results.

An important goal of this study was to assess whether treatment could be enhanced during the cold season by extending the metabolic function of wetland plants and associated microbia into the winter. Contrary to expectations, enhanced treatment in the greenhouse was not observed during the cold months, and overall water quality in the wetland and greenhouse were similar. The failure to provide enhanced treatment is attributed to the similarity of water and substrate temperatures in the greenhouse and wetland. Ice and snow cover above flowing waters in the wetland may have insulated wetland stormwater and substrates, providing a natural form of greenhouse over the wetland. Under these conditions, a large difference in biological activity between the greenhouse and wetland would not be expected. Another contributing factor may relate to the pattern of plant senescence within the greenhouse. By maintaining thermal and lighting conditions similar to those during the growing period, greenhouse plants were expected to remain active much longer into the winter and possibly even to spring. Although the growing season within the greenhouse was extended by more than three months, in late November all the wetland plants were in senescence, and new growth did not commence until the following March. Hence, during the winter (late December to late March), root metabolic activity was sustained, but the plant community likely contributed little to pollutant removal.

Pollutant concentrations at the outlet were generally less than at the greenhouse and wetland monitoring stations. Location of the monitoring stations along the flow path help explain part of this difference, but two other factors may also have been important. The first relates to the location of sampler intakes. The outlet sampler intake was elevated 1 m above the bottom of the outlet depression near the riser orifice, whereas sampler intakes in the greenhouse and wetland were located only 8 to 10 cm above the bottom. If a briny layer formed near the bottom, as indicated by an increase in dissolved constituent concentrations over the treatment period, this layer would have been sampled at the wetland and greenhouse stations, but not at the outlet. A second factor influencing differences in concentrations among monitoring stations may relate to the filtering effect of the geotextiled berm surrounding the outlet depression. While the berm may have had only a minor influence on dissolved constituents, it would likely have enhanced removal of suspended solids.

6.0 VEGETATION AND SEDIMENT ANALYSIS

6.1 Vegetation

In surface flow wetlands, plants and litter help directly and indirectly to improve performance by physically filtering suspended matter, moderating flow rates, stabilizing sediments, shading and sheltering the water column, altering soil/water redox potentials, and providing habitat and refuge for a wide range of microbia (Tanner, 2001). Direct pollutant uptake by vegetation can also play a role in seasonal removal, but the benefit of plants in treatment stems primarily from their complex interactions with system hydrology, sediments, periphyton (biofilm) and other biota. In comparative studies of planted and unplanted subsurface flow treatment wetlands, Tanner (2001) found that plant-derived organic matter and root zone oxygen release were key factors influencing nutrient retention, immobilization and transformation processes.

6.1.1 Plant growth, dominance and vitality

Contrary to expectations in the design stages of the study, there appeared to be no adverse impacts on plant communities due to high water levels. Water levels during events often fluctuated within a range of 0.5 to 1.5 m, but the wetland plant community remained healthy.

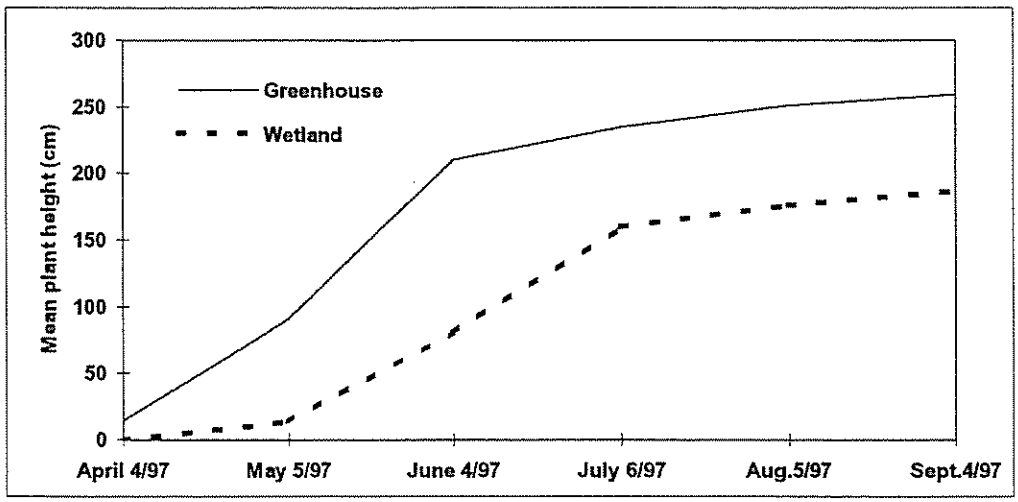
Figure 6.1a compares mean plant growth within the greenhouse and wetland in 1997. A similar growth pattern was also observed in 1996. The greenhouse extended the growing season by six weeks in spring and by six to eight weeks in the fall, or by a total of 12 to 14 weeks annually. The greenhouse also increased the rate of growth to plant maturity and altered the pattern of plant senscence. The first frost resulted in senescence of the whole wetland plant population, but in the greenhouse, plants entered dormancy more slowly and at different times, until by the end of November all plants were in senescence.

The vegetated wetland area covered approximately 2080 m^2, of which 210 m^2 were within the greenhouse. The diversity of plants was generally quite low. Low plant diversity is often associated with homogeneous substrata, with less colonization by microorganisms and poor pollutant retention relative to more diverse systems (Wetzel, 2001). In the wetland area, *Typha latifolia* (common cattail) was found to be the dominant plant species in 9 of the 10 plots, and *Scirpus dominus* (bulrush) was dominant in the remaining plot. *Scirpus* was distributed in patches and occupied approximately 10% of the wetland basin. As shown in Figure 6.1b, *Typha* and *Scirpus* experienced shoot eruption at nearly the same time (late March to early April), but the two plant species exhibited markedly different growth rates. Initially, *Typha* grew faster and higher than the *Scirpus*, until early June, when growth rates were similar.

Plants were considerably denser in the wetland compared to the greenhouse. June plant density averaged 34 and 61 plants/m^2 in the greenhouse and wetland areas, respectively. These average densities increased in September to 41.2 and 69.0 plants/m^2. The average density of *Typha* and *Scirpus* in September was 47 and 152 plants/m^2, respectively.

Vitality testing was performed in January, 1996 on *Typha* and *Scirpus* sp. rhizomes collected from the greenhouse. Although the aboveground plant mass was dead, belowground rhizome cross-sections of the two plant species indicated that active metabolic functioning was occurring. This observation suggests that uptake by plants may have continued in the winter, albeit at a much slower rate.

a.



b.

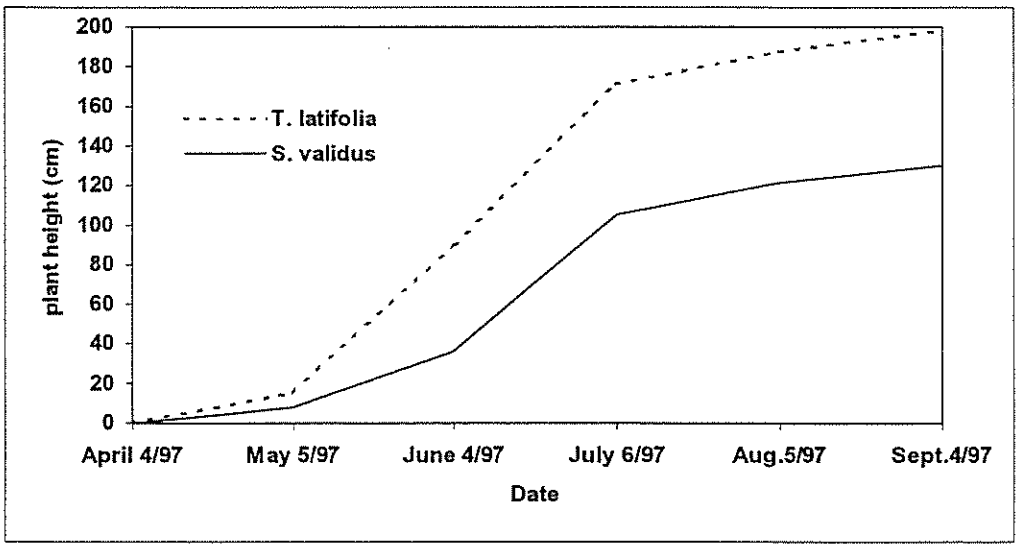


Figure 6.1: (a) Mean 1997 plant growth in the greenhouse and wetland during the growing season. (b) Mean 1997 growth of the wetland area's two dominant vegetative species : *Typha latifolia* and *Scirpus validus.*

6.1.2 Plant biomass

Plant density, and sample wet and dry weights were determined from the ten vegetation plots in June and September. Table 6.1 summarizes plant biomass data on a per plot, total greenhouse, total external wetland and total wetland (greenhouse + wetland) basis. In June, the greenhouse wet and dry plant weights were signficantly greater than those of the wetland; a consequence of the earlier start to the growing season in the greenhouse (Figure 6.1a). After June, growth rates of greenhouse plants slowed, as shown by the more modest increase from June to September of greenhouse above and below-ground masses relative to the wetland. The difference between June wetland and greenhouse dry weight proportions is probably linked to the stage of plant maturity in the two areas. The larger and more rapidly growing plants in the greenhouse had smaller dry weight proportions (i.e. higher moisture content) in June, but by September, when growth rates in the two areas paralleled one another (Figure 6.1a), the difference between greenhouse and wetland dry weights and dry weight proportions narrowed considerably.

The September collection was thought to be a good representation of total biomass within the facility, having been sampled just prior to fall senescence. The total biomass dry weight was 6 T, or 28 T/ha. This biomass weight compares well to values reported by Kadlec and Knight (1996) of 30 and 20 T/ha for *Typha* and *Scirpus*, respectively. Over the entire wetland, the above-ground mass, with a wet weight of 19 T (or 92 T/ha.), was almost twice the below-ground mass. Harvesting of plants in the fall to improve performance is often recommended for constructed wetlands, but with such a significant mass of plants, the effort would not likely be cost efficient.

Table 6.1: Plant Biomass results from June and September collections

1997	Field Collection (average kg/plot)				Extrapolated Totals (kg)				Wetland system total (kg)		
	Greenhouse		Wetland		Greenhouse		Wetland				
	Above	Below	Above	Below	Above	Below	Above	Below	Above	Below	Total
JUNE											
Wet	4.1	3.3	0.9	1.6	860	689	1729	2907	2590	3696	6186
Dry	0.5	0.6	0.2	0.3	99	136	473	597	573	733	1306
Dry (%)	12	20	27	21	15		23		22	20	21
SEPTEMBER											
Wet	8.4	3.8	9.3	4.2	1756	804	17382	7933	19138	8736.8	27875
Dry	1.2	0.9	1.9	0.9	255	182	3608	1723	3864	1905	5769
Dry (%)	15	22	20	22	17		21		20	22	21

6.1.3 Tissue chemistry

Nutrients and metals found in vegetation tissues originate primarily from the soils, groundwater and surface water. Nutrients from surface inflows are assimilated mostly by microbiota such as bacteria, algae, fungi and

protists attached to wetted surfaces, although some nutrients are also assimilated by the mycorrhizae and roots of higher plants (Wetzel, 2001). Below-ground tissues survive over the winter and can, therefore, store nutrients over extended periods of time. As plants enter maturity and senescence, a large store of nutrients in above-ground tissues are translocated to rooting tissues for storage and use in new shoot development (Wetzel, 2001).

Results of the tissue chemistry analysis are presented in Table 6.2. From June to September, concentrations of most nutrients and metals decreased in the above and below-ground tissues of greenhouse and wetland vegetation. Exceptions were TP in belowground wetland tissues, and some greenhouse concentrations of zinc, iron and nickel. A decrease in concentration as plants grow is generally expected since living material is mostly composed of carbohydrates, lipids, proteins and nucleic acids, of which nitrogen and phosphorus form only a small part.

Table 6.2: (a) June and (b) September, 1997, plant tissue chemistry results and extrapolation values for the entire greenhouse, wetland and combined wetland system

a.

June 1997	Tissue concentrations (ug/g for metals mg/g for TKN & TP)				Extrapolated Totals (g for metals & kg for TKN &TP)				Wetland system total (g for metals and kg for TKN & TP)		
	Greenhouse		Wetland		Greenhouse		Wetland		Above	Below	Total
	Above	Below	Above	Below	Above	Below	Above	Below			
TKN	17.2	6.4	23.2	10.2	1.7	0.9	11.0	6.1	12.7	7.0	19.6
TP	5.2	3.2	3.4	2.9	0.5	0.4	1.6	1.7	2.1	2.2	4.3
Cu	7.5	16	12.3	18.1	0.7	2.2	5.8	10.8	6.6	13.0	19.6
Ni	0.8	4.0	2.5	2.4	0.08	0.6	1.2	1.4	1.2	2.0	3.2
Zn	28	52.8	31.8	64.2	2.8	7.2	15.1	38.3	17.8	45.5	63.3
Cd	BD	0.5	0.2	0.4	NA	0.07	0.1	0.2	0.1	0.3	0.4
Cr	0.8	6.7	2.6	3.8	0.08	0.9	1.2	2.2	1.3	3.2	4.5
Pb	1.8	4.6	2.0	2.2	0.2	0.6	0.9	1.3	1.1	1.9	3.0
Fe	307.5	8925	956.7	4400	30.6	1212.6	452.9	2627.7	483.4	3840.3	4323.8

b.

Sept 1997	Tissue concentrations (ug/g for metals and mg/g for TKN & TP)				Extrapolated Totals (g for metals and kg for TKN & TP)				Wetland system total (g for metals and kg for TKN & TP)		
	Greenhouse		Wetland		Greenhouse		Wetland		Above	Below	Total
	Above	Below	Above	Below	Above	Below	Above	Below			
TKN	11.4	5.2	11.4	8.7	2.9	1.0	41.3	14.9	44.2	15.9	60.1
TP	2.0	2.2	2.5	3.4	0.5	0.4	9.0	5.9	9.5	6.2	15.8
Cu	4.8	13.2	6.6	19.0	1.2	2.4	23.7	32.8	24.9	35.1	60.1
Ni	1.2	3.0	1.7	2.9	0.3	0.5	6.2	5.0	6.6	5.5	12.1
Zn	69.8	78.8	31.5	68.2	17.8	14.3	113.7	117.5	131.5	131.8	263.3
Cd	BD	0.2	0.2	0.4	NA	0.04	0.7	0.6	0.7	0.7	1.4
Cr	1.5	6.2	1.9	5.2	0.4	1.1	6.9	8.9	7.2	10.0	17.3
Pb	0.6	1.0	0.7	1.9	0.2	0.2	2.5	3.2	2.7	3.4	6.1
Fe	625	6025	358.3	3500	159.6	1094	1293.1	6032.9	1452.7	7126.9	8579.6

While concentrations of several constituents decreased, the mass of metals and nutrients increased from June to September. The overall increase in above-ground mass tended to be greater than below-ground mass during this period. Nutrient masses in greenhouse plants were similar in June and September, probably because plant growth in the greenhouse was more advanced than the wetland in June. In the entire wetland, masses of metals and nutrients increased significantly from June to September.

In September, TKN and TP comprised 1.0 and 0.03% of the total dry weight of all wetland system plants, respectively. These proportions are considerably lower than the 14% total nitrogen and 2% phosphorus dry weight proportions reported by Kadlec and Knight (1996) for Typha plants in wastewater wetland treatment systems.

The TKN mass in wetland plant tissues constituted 35 and 82% of the 1996-97 and 1997-98 surface inflow TKN loads, respectively. Similarly, the mass of TP in wetland plant tissues represented 35 and 121% of the 1996-97 and 1997-98 influent loads, respectively. Copper, nickel, zinc, chromium and iron in plants accounted for approximately 10, 16, 9, 12 and 33% of the monitored inlet loads in 1997-98, respectively. Although the relationship between influent pollutant loads and plant tissue pollutant masses could not be established, these proportions provide a general sense of the relative importance of plants as seasonal (plant tissue) or long term (roots) sinks for different pollutants.

6.2 Sediment

Sediment provides the reactive surfaces required for microbial activity and absorption/adsorption processes. Plant derived organic matter accumulating in sediments and as fallen litter is particularly important in this regard, sequestering organic bound pollutants, buffering nutrient release and providing a significant energy and carbon source to microbial communities (Tanner, 2001).

6.2.1 Sediment chemistry and particle size distribution

Sediment chemistry results from samples collected in December 1996, June 1997 and August 1997 are presented in Table 6.3. Forebay, greenhouse and wetland sediment concentrations of chloride, oil and grease, zinc and copper were considerably greater than sediment concentrations for the same constituents at the control site adjacent to the facility (see Figure 3.4). Concentrations of total phosphorus, arsenic, chromium, and several other metals were within the range of control site concentrations for these constituents.

Sediment concentrations were compared to Lowest Effect Level (LEL) and Severe Effect Level (SEL) concentrations for the protection of aquatic habitat, as defined in the Provincial Sediment Quality Guidelines (Persaud *et al.*, 1992). Most benthic species can tolerate LEL sediment concentrations, but are harmed by concentrations at Severe Effects Levels (SEL). Sediment pollutant concentrations less than the Lowest Effect

Level (LEL) included TKN, TP, lead, zinc, cadmium, chromium, nickel, and iron. Copper concentrations slightly exceeded the LEL, but were significantly lower than the SEL.

Table 6.3: Constituent concentrations of sediment collected in December 1996, June 1997 and August 1997. Control samples were not collected in June 1997.

	Forebay			Greenhouse			Wetland*			Control	
Constituent	**Dec. 1996**	**Jun. 1997**	**Aug. 1997**	**Dec. 1996**	**Jun. 1997**	**Aug. 1997**	**Dec. 1996**	**Jun. 1997**	**Aug. 1997**	**Dec 1996**	**Aug. 1997**
TKN (mg/g) (dry)	1.20	1.00	0.20	0.40	0.87	0.60	1.02	0.93	0.45	0.23	1.10
TP (mg/g) (dry)	0.85	0.68	0.3	0.51	0.59	0.2	0.84	0.75	0.5	0.73	0.9
Chloride (dry)	-	69	21	-	270	82	-	867	180	-	28
Sulphate (dry)	-	1077	590	-	957	92	-	300.00	205	-	41
pH	-	7.5	7.6	-	7.6	7.6	-	7.67	7.8	-	7.5
Conductivity (µS/cm)	-	810	700	-	700	1100	-	1498	640	-	240
Oil and grease (mg/kg)	1367	6200	-	5633	3900		26500	2383		35	
TOC (mg/g)	17.00	-	22	14.33	-	27	20.33	-	25	4	23
Arsenic (µg/g)	4.1	-	0.2	1.5	-	0.3	2.8	-	0.3	6.2	0.2
TS, LOI (mg/g) (dry)	34.00	-	-	22.67	-	-	37.17	-	-	6.5	-
Metals											
Al (µg/g)	12000	7100	-	6067	13000	-	9833	10983	-	5667	
Cr (µg/g)	26	20	16	14	29	26	22	25	23	45	23
Mn (µg/g)	537	303	250	320	467	430	365	490	445	323	540
Fe (µg/g)	17000	11000	-	9700	17667	-	14167	16167	-	11333	
Co (µg/g)	6.5	4.5	-	4.8	6.4	-	5.9	6.0	-	3.8	-
Ni (µg/g)	12.7	8.8	7	7	14	15	12	12	12	6	14
Cu (µg/g)	18	18	18	14	22	23	18	18	18	6	12
Zn (µg/g)	71	73	58	50	89	88	73	73	66	20	46
Cd (µg/g)	0.5	0.5	0.6	0.4	0.5	0.7	0.5	0.6	0.6	0.3	0.7
Pb (µg/g)	15	16	11	7	17	16	10	16	8	4	11

Note: underlining indicates concentrations above Lowest Effect Levels (LEL) for the protection of aquatic habitat.
*Wetland samples represent an average of two stations (station 2 and 3 in Figure 3.4)

From December 1996 to August 1997, concentrations of most metals in the wetland, forebay and greenhouse decreased or changed very little. TP and TKN concentrations also decreased at all sediment sampling stations in the facility. Nutrient uptake from sediments during plant growth may partly explain the decrease in nitrogen and phosphorus concentrations from June to August.

Concentrations of nutrients (TKN and TP) and most metals were within the range of sediment concentrations reported for 11 stormwater detention ponds in Ontario (MMM, 1992). Zinc and lead sediment concentrations at the Aurora facility were less than the lower limit reported for ponds.

Sediment particle size analysis results for the June collection are presented in Figure 6.2 (see Appendix H for methods). The forebay sediment (median = 25.0 μm) had a coarser distribution than the sediments collected in the wetland (median = 9.9 μm) and greenhouse (median = 5.1 μm). The greatest difference was found in the sand size class (62 μm - 999 μm), which accounted for 33, 17 and 7% of particles by volume in the forebay, wetland and greenhouse, respectively.

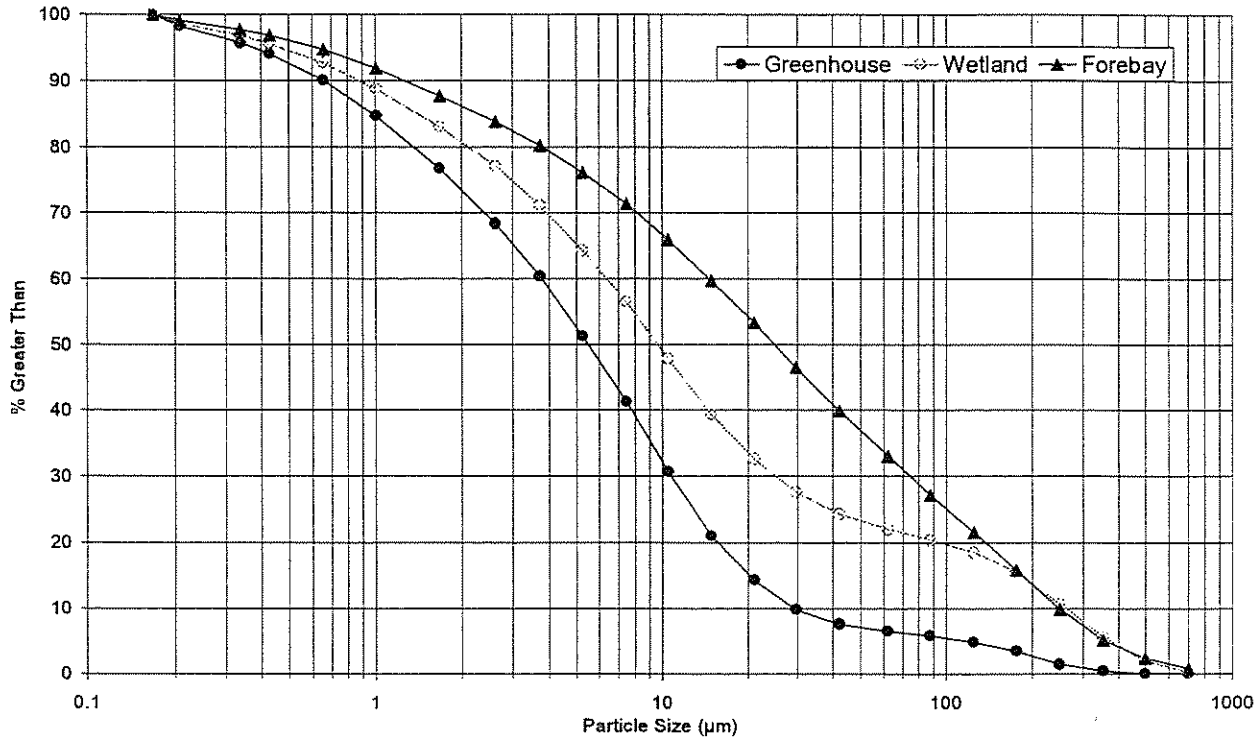


Figure 6.2: Average sediment particle size distributions in the forebay, greenhouse and wetland.

6.2.2 Sediment bioassay

Sediment samples were collected for bioassay in late August, 1997 at the same five stations as those for sediment geochemistry. The full report of the study is presented in Appendix H. The main findings of the sediment bioassay were as follows:

- Overall, the Aurora wetland sediments tested in the laboratory were considered to be of good sediment quality based on mayfly, midge and minnow lethal and sublethal endpoints, chemical uptake and bulk sediment chemistry. The only exception was the sediment collected from the facility forebay which had a total PAH sediment concentration of 11 μg/g.

- Forebay sediment was found to be sublethal to mayfly nymphs and midge larvae resulting in moderate to severe levels of growth reduction. The effect was attributed to the sublethal effects concentration of total PAHs measured in the bulk sediment. The LC50 concentration corresponded with other cited sublethal effect-level concentrations.
- Sediment collected from the greenhouse and wetland stations elicited no negative impact on organism survival or growth. There were no obvious differences in sediment chemistry among these sites despite their spatial distribution within the stormwater facility.
- Chemical bioaccumulation data within the stormwater facility indicate fairly low chemical sediment concentrations that were not related to any lethal biological effect or chemical uptake. The occurrence of benthic growth impairment was limited to the forebay station and may warrant continued monitoring of forebay sediments. The total PAH sediment concentration of 11 μg/g is inferred as the factor that best explained the differences in the sublethal test endpoint. Toxic effects may become apparent at levels approximately one order of magnitude higher than that currently measured, assuming no change in sediment total organic carbon content.

7.0 CONCLUSIONS AND RECOMMENDATIONS

7.1 Conclusions

The Aurora facility is characterized in this report as a wetland because it has a hydroperiod similar to that of many wetlands, the water table is always close to the surface, and it supports a dense cover of wetland plants. However, it also has characteristics of an extended detention dry pond, because it releases storm runoff slowly over a 4 to 5 day period and, except for a small permanent pool in the forebay (40 m^3), the facility drains dry during periods of infrequent rainfall. This unique design did not meet several of the provincial (OMOEE, 1994a) design criteria for stormwater wetland facilities. However, results of this study indicate that it still provided satisfactory levels of treatment, and was comparable in performance to other wetland and wet pond systems cited in the literature.

A greenhouse placed within the wetland, and heated year round was expected to improve performance during the winter by enhancing biological, physical and chemical mechanisms of removal. Results show that the greenhouse was successful in raising air temperatures by an average of 5°C and extending the growing season by 12 to 14 weeks, but water and substrate temperatures in the greenhouse and wetland were not significantly different. Greenhouse removal rates and effluent concentrations also did not differ significantly from those of the wetland. This result is attributed to the key role of root-zone biota in plant removal processes, and the close dependence of root-zone biota on water and soil temperatures, as well as the effect of water temperature on physical and chemical mechanisms of pollutant removal.

Specific findings related to water quantity, water quality, vegetation and sediment were as follows.

7.1.1 Water quantity

Although the Aurora wetland did not support a permanent pool (other than 40 m^3 in the forebay), and extended detention volumes were less than recommended for wetland facilities (OMOEE, 1994a), the system was still effective in attenuating peak flows and providing adequate retention of stormwater for treatment purposes. This effectiveness is indicated by post-event stormwater drawdown times of 3 to 5 days, an estimated mean hydraulic detention time of 36 hours, peak flow reduction of greater than 80% during rain events, and discharge rates in the downstream channel not exceeding 58 L/s. Although the facility was originally designed with the expectation that 41% of rainfall within the catchment would enter the facility as runoff, monitoring for 29 events indicated that this proportion was actually only 21%. A large agricultural area within the catchment, where much of the rainwater infiltrated, partly explains the low runoff coefficient.

During the summer, groundwater recharge accounted for approximately 16% of influent stormwater, resulting in significantly reduced outflow volumes. The volumetric balance calculated over 29 rainfall events, which included a range of different storm sizes during four seasons, closed to within an average of 4.7%.

7.1.2 Water quality

Average effluent event mean concentrations and load-based removal efficiencies are summarized for each season and over the entire study period in Table 7.1. The major findings relating to water quality were as follows.

- Load based removal efficiency and average effluent concentration for TSS over the study period were 86% and 24 mg/L, respectively. Both of these statistics are well within acceptable limits for this type of facility and the level of aquatic habitat protection desired in the downstream channel.

Table 7.1: Summary of wet weather mean effluent concentrations and load-based removal efficiencies for selected parameters.

Parameter	Winter+		Spring		Summer		Fall		Study Period		PWQOs
	Eff. Conc.	R.E. (%)	Eff. Conc.	R.E. (%)	Eff. Conc.	R.E. (%)	Eff. Conc.	R.E. (%)	Eff. Conc.	R.E. (%)	
General Chemistry											
TSS (mg/L)	28.7	46	16.8	90	25.6	91	26.0	87	23.8	86	
BOD (mg/L)	1.6	7	3.1	55	3.1	34	2.1	22	2.5	32	
COD (mg/L)	29.3	6	30.6	51	31.4	36	22.6	39	28.4	33	
Oil/Grease (mg/L)	1.5	25	1.4	82	1.4	44	0.7	78	1.2	61	
Chloride (mg/L)	429	-15	34	69	105	-51	56	24	45	-1	
Phenolics (µg/L)	1.4	21	1.3	48	1.8	-6	1.1	55	1.5	26	1
E. Coli (c./100ml)	252	42	319	68	475	94	1107	55	464	84	100
Nutrients											
TP (mg/L)	0.16	8	0.16	56	0.10	83	0.18	64	0.14	58	0.03
PO_4 (mg/L)	0.07	26	0.06	14	0.02	86	0.09	31	0.05	44	
TKN (mg/L)	0.88	36	1.19	52	0.81	63	0.79	51	0.90	49	
NH_3+NH_4 (mg/L)	0.05	50	0.17	79	0.03	52	0.05	40	0.06	63	
NO_3 (mg/L)	0.63	27	0.36	26	0.08	82	0.50	11	0.28	41	
Metals											
Copper (µg/L)	5.8	37	5.7	66	4.8	69	5.0	57	5.3	58	5
Zinc (µg/L)	63.8	-87	30.4	59	24.0	53	26.3	58	31.8	17	20
Chromium (µg/L)	2.5	20	1.2	50	1.0	58	1.8	32	1.5	36	8.9*
Iron (µg/L)	470	6	353	65	386	52	481	31	415	41	300

Underlining indicates concentrations greater than PWQOs

*1 µg/L in its less common hexavalent form. + Mean concentrations and removal efficiencies over the winter period are based on grab samples collected near the end of the runoff event and, therfore, should be interpreted with caution.

- TSS removal during the winter was only 46%. This poor removal rate during the winter may have been a result of low influent concentrations (average concentration = 43 mg/L). Those concentrations may have been partly a consequence of grab sample collection, which would have missed high solids loading associated with the 'first flush' of runoff. Consequently, poor efficiency in winter should not be attributed exclusively to cold weather effects on removal mechanisms.

- In contrast to TSS, overall removal efficiencies for nutrients ranged from 41% for nitrate to 63% for total ammonia, and metal efficiencies were mostly less than 60% (Table 7.1). In general, removal efficiencies were greatest during the growing season, although seasonal differences may partly reflect differences in the method of inlet sample collection during the summer and winter (*i.e.* composite vs. grab). Effluent concentrations were similar during warm and cold weather.

- Effluent detection frequencies for several metals were less than 50%, including lead, cadmium, cobalt, nickel and molybdenum. Other more frequently detected metals such as zinc, copper and iron often had effluent concentrations greater than PWQOs for these constituents.

- *E.coli*, TP and phenolics also had average effluent concentrations above PWQOs (OMOEE, 1994b).

- The average particle size distribution of suspended particles over the study period indicated a substantial shift to finer particle sizes after the first day of treatment. Based on samples analyzed, the average particle size of the influent was 3.4 μm (average of PSD medians) compared to the average effluent particle size of 1.7 μm on day 1 of the treatment period, and 1.4 μm on day 4.

- Over the treatment period, average daily effluent concentrations of TSS decreased significantly after the first day, while other constituents displayed a more gradual decline. This gradual decline for the majority of pollutants suggests that drawdown up to at least 3 days is a crucial factor in wetland pollutant removal. The results are also consistent with the hypothesis that the pollutants are either in dissolved form or associated with the finer suspended particles.

- The maximum water temperature in the wetland (24°C) during the summer was above the 21°C maximum recommended for cold water fisheries habitat. Unlike the situation in wet ponds, the water temperature in the Aurora wetland increased very little from the inlet to the outlet, probably due to the absence of a significant permanent pool and shading of the water column by plants.

7.1.3 Vegetation and sediment analysis

- The wetland and greenhouse was dominated by *Typha latifolia* (common cattail) and *Scirpus dominus* (bulrush). Vitality testing on both plants indicated that, during the winter, active metabolic functioning was occurring in the below-ground rhizomes, even though the above-ground tissues were dead.

- Biomass analysis of above and below-ground tissues indicated that the dry-to-wet weight ratio was 21% in June and September. In September, shortly before plant senescence in the fall, the dry weight of above ground tissues was almost double that of below ground tissues. The greenhouse and wetland plants were similar, except that the greenhouse above-ground tissues had slightly higher moisture content.

- Tissue chemistry analysis showed that, as biomass increased from June to September, the concentrations of most pollutants in the tissues decreased. In terms of mass, however, all nutrients and metals showed significant increases in both above and below-ground tissues over the summer. TKN and TP masses tripled from June to September. The nutrient masses represented a much higher proportion of influent loads to the wetland than did the metals.

- With the exception of TP, arsenic and chromium, concentrations of pollutants in wetland and greenhouse sediments were greater than concentrations at the control site. However, among pollutants analyzed, only copper had concentrations that exceeded the lowest effect tolerance level for benthic invertebrates.

- Wetland sediments were found to be of good sediment quality based on mayfly, midge and minnow lethal and sublethal endpoints, chemical uptake and bulk sediment chemistry. Forebay sediments showed moderate to severe levels of growth reduction to mayfly nymphs and midge larvae, probably due to high concentrations of PAHs (11 μg/g) in the sediments.

7.2 Recommendations

7.2.1 Maintenance and safety issues

A considerable amount of trash and debris was carried into the facility during the study period. This trash and debris should be cleaned out at least once a year.

Signs were placed in the facility during the study period warning of human hazards related to fluctuating water levels. Further restrictions on human use access to the flooding zone could be achieved by the use of dense perimeter vegetation or fencing with natural materials. Well defined trail grids outside of the flooding zone would help to retain the public amenity function of the facility.

7.2.2 Facility improvement

Runoff events during the winter are less frequent and intense compared to the summer, but can be large and extend over long periods of time. Treatment of stormwater under this type of runoff regime can be improved by lengthening the cold season detention period through temporary alterations to the outlet control structure. Due to the longer interevent periods and slower release of runoff during the winter, longer extended detention periods may be possible without compromising the water quantity control function of the facility.

If development expands beyond the 58 hectare area documented in this study, some modifications may be warranted such that the facility more closely approximates OMOE design criteria for constructed wetlands. These modifications may include the following: (i) deepening the forebay to the 1 m depth recommended in the SWMP manual; (ii) increasing the length-to-width ratio from 2:1 to 3:1 by moving the forebay closer to the inlet; (iii) reconfiguring the basin such that it retains a 35 cm permanent pool; and (iv) increasing the total depth of extended detention to 1 m, or up to the bottom of the existing outlet weir through installation of a Hickenbottom riser device similar to that installed as part of the study.

Wetlands provide food for several species of mammals and water fowl as well as habitat for amphibians. It was often observed during periods of flooding that waterfowl used the wetland area to forage. Since waterfowl can contribute significantly to the fecal matter in the facility, as well as stirring up bottom sediments, waterfowl use of the facility should be restricted by, for example, planting vegetation in open areas.

7.2.3 Future considerations

Once plant uptake and sediment adsorption pools in the Aurora wetland approach their limits (as defined by site- specific environmental conditions and loading regime), removal rates may suffer substantial declines. Deterioration of performance in this regard may indicate the need for dredging, especially in the forebay, and natural or artificial re-establishment of vegetation cover. Short-term monitoring of system performance should be conducted every 3 to 5 years to determine whether maintenance for this purpose is warranted.

The Aurora facility was a dry pond converted to a wetland, but also functioned partly as an infiltration basin, with average water losses to groundwater accounting for 16% of influent runoff during the summer. These losses helped to further reduce pollutant loading to receiving waters. Enhancing the infiltration component of stormwater wetland facilities should be considered when meeting OMOE permanent pool volume guidelines is not practical, and the potential for contamination of groundwater resources is low.

REFERENCES

Bavor, H.J., Davies, C.M., Sakadevan, K. 2001. Stormwater treatment: do constructed wetlands yield improved pollutant management performance over a detention pond system? *Water Science and Technology*. V. 44, No. 11-12 pp 565-570.

Bedient, P.B. and Huber, W.C. 1988. *Hydrology and Floodplain Analysis*. Addison-Wesley Publishing Co., Don Mills, Ontario.

Bennett D. 1994. *The Status of Wetland Stormwater Management in Ontario*. M.A. Thesis, University of Waterloo, Waterloo, Ontario.

Canadian Council of Ministers of the Environment, 1999, updated 2001. *Canadian Water Quality Guidelines*.

Carlisle T., G. Mulamoottil and B. Mitchell. 1991. Attitudes toward artificial wetlands in Ontario for stormwater control and waterfowl habitat. *Water Resources Bulletin*. V.27, N.3, p.419-427

Daugherty, R.L., Franzini, J.G. and Finnemore, E.J. 1985. *Fluid Mechanics with Engineering Applications*. McGraw-Hill, New York.

Eaton, A.D., Clesceri, L.S., and Greenberg, A.E. (Eds.). 1995. *Standard Methods for the Examination of Water and Wastewater*, 19th 3rd edition, published jointly by: American Public Health Association, American Water Works Association and Water Environment Federation, Washington D.C.

Environment Canada, 2000. *Road Salts: Assessment Report and Supporting Documentation*. Prepared for the Canadian Environmental Protection Act, 1999 Priority Substances List. Internet Publication.

Ferguson, B. and Debo, T.N. 1990. *On-site Stormwater Management*. Van Nostrand Reinhold, New York, 270 p.

Freeze, R.A, and Cherry, J.A., 1979. *Groundwater*. Prentice-Hall Inc., Englewood, N.J.

Gietz, R. , 1983. *Rideau River Stormwater Management Study - Urban Runoff Treatment in Kennedy - Burnett Settling Pond*. Regional Municipality of Ottawa - Carleton, Ontario.

Gilliam, J.W. 1994. Riparian Wetlands and Water Quality. *Journal of Environmental Quality*. Vol. #23, p. 896-900.

Jansson, M. Anderson, R., Berggren, H. and Leonardson, L. 1994. Wetlands and lakes as nitrogen traps, *Ambio*, V. 23, No. 6, pp 320-325.

Kadlec, R.H. and Knight, R.L. 1996. *Treatment Wetlands*. Lewis Publishers, Boca Raton Fla. 893p.

Kadlec, R.H., Williams, R.B. and Scheffe, R.D., 1987. Wetland evapotranspiration in temperate and arid climates. IN: Hook, D.D. (ed.), *The Ecology and Management of Wetlands*. Croom Helm, Beckenham, p.146 -160.

Kachka, A.M. and Turner, C. 1996. Performance evaluation of a water pollution control pond - Lake Annan. In: *Proceedings of the WaterTech Conference*, Sydney, May 27th and 28th, 1996, p. 204-11.

Lake Simcoe Region Conservation Authority (LSRCA), 1995. *Lake Simcoe Environmental Management Strategy. Implementation Program, Summary of Phase I Progress and Recommendations for Phase II.*, Newmarket, Ontario,

Marshall, Macklin and Monahan (MMM), 1988. *Updated Stormwater Management Report*. Prepared for Coscan Developments Corp. Report No. 10-86024-101 (unpublished).

Maunder, D., Whyte, R. and D'Andrea, M., 1995. *Wet Weather Discharges to the Metropolitan Toronto Waterfront, Phase II.*, Metropolitan Toronto and Region Remedial Action Plan, Queen's Printer, Toronto.

Mitsch W. and Gosselink, J., 1993. *Wetlands*. Van Nostrand Reinhold Company, New York.

National Technical Advisory Committee (NTAC), Federal Water Pollution control Administration, 1968. *Report of the Committee on Water Quality Criteria*, US Department of the Interior, Washington D.C., 234 p.

Neill, C. 1990. Effects of nutrients and water levels on emergent macrophyte biomass in a prairie marsh. V.68, No. 5, p. 1007-1014

Noller, B.N.; Woods, P.H. and Ross, B.J. 1994. Case studies of wetland filtration of mine waste water in constructed and naturally occurring systems in northern Australia. *Water, Science and Technology*. V. 29, No. 4

Nolte and Associates, 1999. *Sacramento Regional Wastewater Treatment Plant Constructed Wetlands Demonstration Project: 5 Year Summary Report*, 1994-1998. Sacramento, California.

Oberts, G.L. and Osgood, R.A. 1991. Water quality effectiveness of a detention/wetland treatment system and its effect on an urban lake. *Environmental Management*. v. 15, no. 1, pp 121-138.

Ontario Ministry of Natural Resources (OMNR), 1994. *Natural Channel Systems: An Approach to Management and Design*, MNR, 103 p.

Ontario Ministry of Environment and Energy (OMOEE), 1994a. *Stormwater Management Practices Planning and Design Manual*, Toronto, Ontario.

Ontario Ministry of Environment and Energy (OMOEE), 1994b. *Water Management: Policies, Guidelines, Provincial Water Quality Objectives of the Ministry of Environment and Energy*. Queen's Printer, Toronto.

Ontario Ministry of Environment and Energy (OMOEE), 1979. *Rationale for the Establishment of Ontario's Water Quality Objectives*. OMOEE, 236 p.

Persaud, D., Jaagumagi, R. and Hayton, A. 1992. *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario*. Ontario Ministry of the Environment, Toronto. 30p.

Peterson, C.A. and Waite, J.L., 1996. The effect of *suberin iamellae* on the vitality and symplasmic permeability of the onion root exodermis. *Canadian Journal of Botany*, V.74, p. 1220-26.

Raven, P.H., Evert, R.F., Eichhorn, S.E. 1992. *Biology of Plants*. 5th edition. Worth Publishers, N.Y. 792p.

Richardson, C.J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands, *Science*, 228:1424-27.

Sharpley, A.N., Smith, S.J., Jones, O.R., Berg, W.A. and Coleman, G.A. 1992. The transport of bioavailable phosphorus in agricultural runoff. *Journal of Environmental Quality*, 21:30-35.

Soil-Eng Ltd. 1988. *A soil investigation for a proposed residential subdivision, Part of Lots 73, 74, and 75, Concession 1, W.Y.S, Town of Aurora*. Unpublished report prepared for Marshall, Macklin, Monaghan Ltd. 25 p.

Stasovski, E. and Peterson, C.A., 1991. The effects of drought and subsequent rehydration on the structure and vitality of *Zea mays* seedling roots. *Canadian Journal of Botany*, V.69, p. 1170-78.

Stormwater Assessment Monitoring and Performance (SWAMP) Program, 2002a. Performance Assessment of a Stormwater Retrofit Pond - Harding Park, Richmond Hill, Ontario., Metropolitan Toronto and Region Conservation Authority, Toronto, Ontario.

Stormwater Assessment Monitoring and Performance (SWAMP) Program, 2002b. Performance Assessment of a Pond/Wetland Stormwater Management Facility - Markham, Ontario., Toronto and Region Conservation Authority, Toronto, Ontario.

Strecker, E.W., Kersnar, J.M., Driscoll, E.D., and Homer, R.R. 1992. *The Use of Wetlands for Controlling Stormwater Pollution*. Terrene Institute, Washington, D.C, EPA/600.

Tanner, C.C. 2001. Plants as ecosystem engineers in subsurface-flow treatment wetlands. *Water Science and Technology*, v. 44, No. 11-12., p. 9-17.

Thunhorst, G.A. 1993. *Wetland Planting Guide for the Northeastern United States*. Environmental Concern, Inc. St.Micheals, Maryland, 179 p.

United States Environmental Protection Agency (USEPA), 1983. *Final Report of the National Urban Runoff Program*. Water Planning Division Report, WH-554, Washington D.C.

Viessman, W. and Lewis, G.L. 1996. *Introduction to Hydrology*, (4th ed.). Harper Collins College Publishers, New York, 760 p.

Vought, L., Dahl, J., Pedersen, C.L., Lacoursiere, J.O. 1994. Nutrient retention in riparian ecotones, *Ambio*, V. 23, No. 6, p. 342-348.

Wetzel, R.G. 2001. Fundamental processes within natural and constructed wetland ecosystems: short-term versus long-term objectives. *Water Science and Technology*, V. 44, No. 11-12., pp 1-8.

Wilson, E.M. 1990. *Engineering Hydrology*. MacMillan Education Ltd., London 348 p.

APPENDIX A

Historical Context of the SWAMP Program

HISTORICAL CONTEXT OF THE SWAMP PROGRAM

In recent years, the Great Lakes Basin has 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 the Environment

The Ontario Ministry of the Environment (OMOE) 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 OMOE 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 the 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 the Environment, 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 Co-ordinator [July 2001 to May 2003]
David Fellowes	
Rene Gagnon	
Dajana Grgic	
Weng Liang	Program Co-ordinator [1995 to 2000]
Serge Ristic	
Derek Smith	
Sheldon Smith	
William Snodgrass	Program Co-ordinator [December 2000 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 the Environment, 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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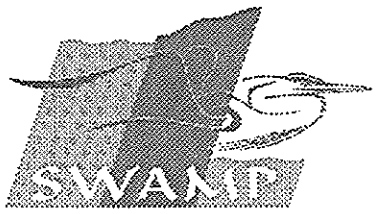


APPENDIX B

Glossary

Glossary

active storage: see 'extended detention storage'.

adsorption: The adherence of a gas, liquid, or dissolved chemical to the surface of a solid (IWA, 2000).

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).

corm: A thickened underground stem, upright in position, in which food is accumulated, usually in the form of starch (Raven *et al.*, 1992)

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

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).

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.

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.

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).

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, 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).

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)

invertebrate: All animals that do not have backbones (IWA, 2000).

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).

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.

rhizome: A more or less horizontal underground stem (Raven *et al.*, 1992).

root: The usually descending axis of a plant, normally below ground, which serves to anchor the plant and to absorb and conduct water and minerals into it (Raven *et al.*, 1992).

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)

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 atmospheric pressure (Parker, 1989).

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).

References

American Society of Civil Engineers (ASCE), 1999. *Development of Performance Measures, Task 3.1 – Technical Memorandum for Determining Urban Stormwater Best Management Practice (BMP) Removal Efficiencies.* URS Greiner Woodward Clyde, Urban Drainage and Flood Control District of Denver, and Urban Water Resources Research Council (UWRRC).

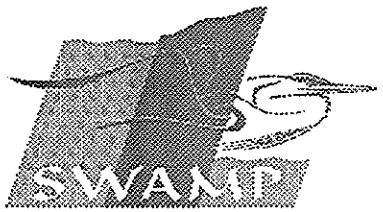
Badelt, B., 1999. *Geomorphic Assessment of the Upper Morningside Tributary.*, Master of Science thesis, University of Guelph, pp. 306.

International Water Association (IWA), 2000. *Constructed Wetlands for Pollution Contro*l, Scientific and Technical Report #8, IWA.

James, W., James, R., 2000 (eds), *Water Systems Models: Hydraulics. User's guide to SWMM4 TRANSPORT, EXTRAN and STORAGE modules*, From the 1988 SWMM manuals by: Huber, W.C., Dickinson, R.E., Roesner, L.A. and Aldrich, J.A. Computational Hydraulics International, Guelph, Ontario.

Parker, S.B. (editor in chief), 1989. *Dictionary of Scientific and Technical Terms.* 4th edition, McGraw-Hill Book Company, New York.

Raven, P.H., Evert, R.F., Eichhorn, S.E. 1992. *Biology of Plants*, 5th edition, Worth Publishers, New York.



APPENDIX C

Literature Review

LITERATURE REVIEW OF STORMWATER TREATMENT WETLANDS

Constructed wetlands have been used extensively for treatment of domestic sewage, industrial effluents, acid mine drainage and agricultural wastes, including manure (Hammer, 1989; Moshiri, 1993; Cooper and Findlater, 1990; Bastian and Reed, 1979; USEPA, 1994; Mitsch and Gosselink, 1993; Reed *et al.*, 1995; Kadlec and Knight, 1996). The use of constructed wetlands in stormwater management is less well documented, but is rated as one of the most effective stormwater best management practices by the Ontario Ministry of the Environment (OMOE, 1994).

Table C-2 presents an overview of stormwater pollutant reduction efficiencies based on several studies of stormwater treatment wetlands. Although performance varied considerably among studies, the review demonstrates the general effectiveness of wetland stormwater treatment systems. Reported removal efficiency ranges were,

- 75 to 90% for total suspended solids,
- 0 to over 90% for nitrogen compounds,
- 43 to over 90% for phosphorus, and
- less than 0 to greater than 90% for heavy metals.

These wide ranges in removal may reflect the biological complexity of wetland systems and the important influence design may have on performance. Unfortunately, specific explanations of variations among studies could not be provided because of insufficient information on influent concentrations, timing of sample collection, total number of samples collected, hydraulic residence time, wetland size and type, substrate and plant type, plant harvesting and pretreatment. Cold weather monitoring data were sparse, but most researchers expected lower temperatures in the late fall, winter and early spring to reduce wetland treatment performance (Livingston, 1989; Witthar, 1993; Ferlow, 1993; Shutes *et al.*, 1993; Findlater *et al.*, 1990).

Limitations of Stormwater Wetland Treatment Systems in Cold Climates

Wetlands in northern regions provide reduced levels of treatment during the winter because cold temperatures slow down the metabolic activity of the biota and the rate of reaction of many chemical processes. Further, the freezing of open water prevents oxygen diffusion from the atmosphere. Thus cold temperatures place severe restrictions on the performance of wetlands used in wastewater treatment (Watson *et al.*, 1989; Reed *et al.*, 1995), as reported at Listowel, and other Ontario sites (Herskowitz, 1986; Miller, 1989; Lemon and Smith, 1993).

Table C-2: Review of observed and expected constituent removal efficiencies in studies of stormwater treatment wetlands.

Study	HRT	Removal Efficiencies	Comments
Johengen and LaRock, 1993	Varied; 5 day retention ?	Nitrate: 67%/day Ammonium: 87%/ day Phosphate: 62%/ day Nearly 2 : 1, N : P reduction ratio	Artificial enrichment of Nitrate, ammonium and phosphate to 0.5 – 1.5 mg/L in 2.5 ha, emergent and floating macrophytic marsh for 5 storms (July – December)
Mungur *et al.*, 1995	Variable	Lead: + 33.3% Zinc: +122.4% Copper: + 41.7% Cadmium: + 16.7%	Mean removal values for highway runoff to a 0.2 ha natural wetland from Mar/93 – Feb/94. Found active metal uptake in sediments and emergent macrophytes, but aqueous metal conc.s remained unchanged
Alder et al, 1995	3 – 4 hours in 2, 4 and 6 day sets	Phosphorus: 90% Nitrate: 40%	Artificial enrichment of NO_3, P, K, Fe, B, Mn, Zn, Cu, Mo. Surface and root zone treatment in troughs in wetland designed to meet stringent P discharge requirements. 50% of N and 80% of P removal from effluent in biweekly grass clippings
Reuter *et al.*, 1992	Variable	TKN: + 3% Nitrate: 85 – 90% Particulate P: 47% Total Reactive Iron: 84% Soluble Reactive Iron: 78% Turbidity and SS: ~ 85%	Aug/87 – May/89 (no June – July results) operation of 660 m^2 constructed wetland treating runoff from a 10,000 m^2 athletic field.
Oberts and Osgood, 1991.	Variable	TSS: 96%; COD: 89%; TP: 77%; DP: 48% TKN: 78%; Nitrate: 64%, TN: 76%; Total Pb: 93%	Warm season results from a 1.0 ha detention pond/ wetland system having a permanent storage of 3415 m^3
Martin, 1988		N: 36%; P: 43%, DP: 57%, Total solids: 55%; Total Pb: 83%; Zn: 70%	Pond / wetland system
Carr and Rushton, 1995		Inorganic N, SS and Zn: > 85% Cu and P: > 71% Cd: 92% Net export of Cl (increase)	Overall annual results from a 1.16 ha natural herbaceous wetland receiving runoff from a 5.8 ha drainage basin. Pretreatment in 2 sedimentation ponds
Harper *et al.*, 1986		Cd: 70.7%, Cr: 72.5%, Cu: 39.9% Fe: +90.1%, Pb: 54.8%; Ni: 70% Zn: 40.9%	Forested swamp receiving stormwater in Florida
Schiffer, 1989		Cr: 40%; Cu: 87%; Ni: 25%; Pb: 83.3%; Zn: 66.7%	Freshwater marsh receiving urban runoff in Florida
Strecker *et al.*, 1992		SS: 81% Pb: 83%* Zn: =42% **	Survey of 14 stormwater surface flow wetlands (*9 and **5 wetlands)
Cutbill, 1993		Nitrate: 92.9%; Nitrite: 81.3%; Phosphate: 46.7%; Sulphate: 11%; Zn: 78.3%; Cd: 18%; Pb: 5.6%	Weekly sampling from July – Sept. of a 13, 125 m^2 artificial wetland receiving urban stormwater

HRT: hydraulic residence or retention time

The Stormwater Management Practices Planning and Design Manual (SWMP manual) (OMOEE, 1994) suggests that water freezing and plant dormancy renders most urban stormwater management practices ineffective in water quality improvement during the winter and early spring. Reduced flow volumes in the winter help to mitigate this problem. Large volumes of stormwater runoff generated during the spring are considered impractical to treat, but dilution is thought to reduce the impact of contaminants on receiving waters (OMOEE, 1994).

Overland runoff peaks during the spring account for up to 1/3 of total annual runoff (Price, 1988). In forested watersheds, the first third of the snowmelt can release 50 to 80% of contaminants accumulated in the snowpack (Jefferies et al, 1979; Davies *et al.*, 1987; Stein *et al.*, 1994). Several studies report elevated stormwater pollutant concentrations during the winter, especially for chloride, lead and suspended solids (Wilde, 1989; Ku and Simmons, 1986; Striegl, 1985; Kronis, 1978; Battaglia, 1976). Pollutant removal efficiencies in wetlands or detention ponds are generally lower during late fall, winter and early spring (Cutbill, 1993; Miller, 1987).

Wetland Treatment Mechanisms During Cold Weather

Several methods have been proposed to reduce impacts of cold temperatures on wetland treatment processes. One such technique is to create an aerated zone beneath the ice by first flooding the wetland, allowing ice to form, then lowering water levels. In this approach, senescent wetland plants support the ice sheet and trap snow providing an insulating layer under consistently cold temperatures. Another technique has been to modify the facility so that cold season flows are stored until late spring or early summer, when warmer conditions facilitate greater chemical and biological treatment (Pries, 1994). This latter technique would not be feasible in the urban areas of southern Ontario due to considerable temperature fluctuations during the cold season and limitations of the availability of land for storage of winter flows.

Several authors suggest that, in order to maintain the metabolic activity of a wetland ecosystem, minimum temperatures of 10°C and an extended photoperiod is required (Hataro *et al.*, 1993; Surrency, 1993; Watson *et al.*, 1989; Stuckey *et al.*, 1978). A mechanically heated greenhouse maybe one method of maintaining such a thermal environment. During periods of shorter daylight, the photoperiod of the greenhouse environment can be extended by the use of artificial lighting. This technology would allow for continuous oxygen diffusion and enhance microbial activity during the winter.

The concept of enclosing a wetland within a greenhouse to reduce the adverse effects of cold and freezing temperatures is not a new idea. In the mid 1970's the Solar Aquacell Treatment Facility was pioneered by Solar Aquasystems Inc. of Encintas, CA for the treatment of municipal sewage (Stewart and Serfling, 1979; Serfling and Alsten, 1979; Stewart *et al.*, 1979). This system attempted to use a greenhouse covered with an

air-filled plastic membrane to provide solar insulation and reduce water losses to evaporation. However, for unknown reasons, the project was abandoned.

John Todd's 'Living Machine' is another example of a technology that employs a greenhouse envelope for sewage treatment using wetland systems (Guterson, 1993; Todd, 1991). Greenhouses have also been employed in several other treatment wetlands as a means of providing a more stable growth environment for the wetland ecosystem. In all cases, the greenhouse was considered to be successful in maintaining the thermal environment needed for wetland treatment of domestic wastewater.

The use of a transparent film in an air bladder configuration has been documented as an effective and economical greenhouse cover alternative. An air bladder film can reduce thermal loss by as much as 50% over conventionally glazed applications (Markov, 1992; Avezov, *et al.*, 1991; Stewart and Serfling, 1979; Keveren, 1973). Even in the cold Canadian winter, significant passive solar heat gain can be achieved. With appropriate circulation, this heat gain has been found to account for up to 58% of fall, winter and spring thermal energy requirements in agricultural greenhouses (Bernier *et al.*, 1991).

References

Alder, P.R., S.T. Summerfelt, D.M. Glenn and F. Takeda, 1995. Evaluation of a wetland system designed to meet stringent phosphorus discharge requirements. *Water Environment Research* v. 68, no, 5, p. 836 - 840

Avezov R., Kasymov, A. Niyazov, S. and Rakhimov, M. 1991. The effects of thermotechnical properties of transparent protective film coating on greenhouse efficiency. *Geliotekника* N.5, Sept-Oct., pp. 3-6.

Bastian, R. and Reed, S. 1979. *Aquaculture Systems for Wastewater Treatment: Seminar Proceedings and Engineering Assessment.* U.S. E.P.A., Office of Water Program Operations, Municipal Construction Division, Washington, D.C., 20460 (EPA 430/9-80-006)

Battaglia G. 1976. *Pollution Characteristics of Urban Snowmelt Runoff.* M.S. Thesis, PB-261, National Technical Information Service, Springfield, VA, 120pp

Bernier H., Raghavan, G. and Paris, J. 1991. Evaluation of soil heat exchanger system for a greenhouse: Part 1: System performance; Part 2: Energy saving aspects. *Canadian Agricultural Engineering* V.33, N. 1, Part 1: p. 93 - 98, Part 2: p. 99 - 105.

Carr, D.W. and B.T. Rushton, 1995. *Integrating a native herbaceous wetland into stormwater management. Stormwater Research Program,* Southwest Florida Water Management District, Brooksville, Florida.

Cooper P. and B. Findlater (eds.) 1990. Constructed wetlands in water pollution control. In: *Proceedings of the International Conference on the Use of Constructed Wetlands in Water Pollution Control,* Cambridge, UK. Pergamon Press, Oxford, 605 pp.

Cutbill L. 1993. Urban stormwater treatment by artificial wetlands: a case study. IN: J. Marsable and H. Torno (eds.) *Proceedings of the Sixth International Conference on Urban Drainage,* Niagara Falls

Davies T., P. Brimblecombe, M. Tranter, S. Tsiouris, C. Vincent, P. Abrahams and I. Blackwood 1987. The removal of soluble ions from melting snowpacks. IN: *Seasonal Snowcovers: physics, chemistry and hydrology,* NATO ASI Ser., Ser. C, 211, p. 337 - 392

Ferlow D. 1993. Stormwater runoff retention and renovation: a back lot function or integral part of the landscape? IN: G. Moshiri (ed.) *Constructed wetlands for water quality improvement.* Lewis Publishers, Chelsea, Michigan, p. 373 - 379.

Findlater B., J. Hobson and P. Cooper 1990. Reed bed treatment systems: Performance evaluation. IN: Cooper P. and B. Findlater (eds.). *Constructed wetlands in water pollution control. Proceedings of the International*

Conference on the Use of Constructed Wetlands in Water Pollution Control, Cambridge, UK. Pergamon Press, Oxford, p. 193 - 204.

Guterson M. 1993. Living Machines: putting human waste back in its place: at the bottom of the food chain. In *Context* No. 35, p. 37 - 38

Hammer, D. (ed.) 1989. *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural.* Lewis Publishers, Chelsea, Michigan

Harper, H.H., M.P. Wanielista, B.M. Fries, and D.M. Baker. 1986. *Stormwater treatment by natural systems.* Florida Dept. of Environmental Regulation Report 84-026.

Hataro K. C. Trettin, H. House and A. Wollum II 1993. Microbial populations and decomposition activity in three subsurface flow constructed wetlands. IN: G. Moshiri (ed.) *Constructed Wetlands for Water Quality Improvement.* Lewis Publishers, Chelsea, Michigan

Herskowitz, J. 1986. *Town of Listowel Artificial Marsh Project Final Report*, Project No. 128RR. OMOEE, Toronto,Ontario

Jefferies D., Cox, C. Dillon, P. 1979. Depression of pH in lakes and streams in central Ontario during snowmelt. *Journal of the Fisheries Research Board of Canada* V. 36, p. 640 - 646.

Johengen, T.H. and P.A. LaRock, 1993. Quantifying nutrient removal processes within a constructed wetland designed to treat urban stormwater runoff. Ecological Engineering v.2, p. 347 – 366.

Kadlec, R.H. and Knight, R.L. 1996. *Treatment Wetlands.* Lewis Publishers, Boca Raton Fla. 893pp.

Keveren R. 1973. *Plastics in horticultural structures.* Rubber and Plastics Research Association of Great Britain, Shaebury, England, 158 pp.

Kronis H. 1978. *Characterization and Treatment of Snowmelt Runoff from an Urban Catchment.* Research publication No. 73, 10pp.

Ku, H. and Simmons, D. 1986. *Effect of Urban Stormwater runoff on Ground Water Beneath Recharge Basins on Long Island, New York.* Water Resources Investigations Report 85-4088, USGS, Denver, Colorado, 67pp.

Lemon E. and Smith, I. 1993. *Sewage Amendment Marsh Process (SWAMP), Interim Report.* OMOEE Project 597C

Livingston E. 1989. Use of wetlands for urban stormwater management. IN: D. Hammer (ed.) *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural.* Lewis Publishers, Chelsea, Michigan

Markov I. 1992. Dynamic of the optical and mechanical characteristics of new polyvinyl chloride films for greenhouses. *Geliotekhnika* N. 5 , p. 63 - 67

Martin, E.H. 1988. Effectiveness of an urban runoff detention pond-wetland system. *Journal of Environmental Engineering* V.114 (4):810 - 827.

Miller G. 1989. Use of artificial cattail marshes to treat sewage in northern Ontario, Canada. IN: D. Hammer (ed.) *Constructed Wetlands for WastewaterTtreatment:Municipal, Industrial and Agricultural.* Lewis Publishers, Chelsea, Michigan

Miller T. 1987. *Appraisal of Stormwater Quality near Salem, Oregon.* Water Resources Investigations Report 87 - 4064, USGS, Denver, Colorado, 29pp.

Mitsch W. and J. Gosselink, 1993. *Wetlands.* Van Nostrand, Reinhold Company, New York, p.577 - 615

Monahan, S.E. 1994. *Environmental Chemistry.* Sixth Edition, Lewis Publishers, Ann Arbor, Michigan. 811 pp.

Mungur, A.S., Shutes, R., Revitt, D. and House, M. 1995. An assessment of metal removal from highway runoff by a natural wetland, *Water Science and Technology*, V.32(3):169-175.

Moshiri, G.(ed.) 1993. *Constructed Wetlands for Water Quality Improvement.* Lewis Publishers, Chelsea, Michigan

Oberts, G.L. and Osgood, R.A. 1991. Water quality effectiveness of a detention/wetland treatment system and its effect on an urban lake. *Environmental Management.* v. 15, no. 1, pp 121-138.

Ontario Ministry of the Environment and Energy (OMOEE), 1994. *Stormwater management practices planning and design manual.* OMOEE, Toronto, Ontario.

Price A. 1988. Prediction of snowmelt rates in a deciduous forest. *Journal of Hydrology* V. 101, p. 145 - 157.

Pries, J. 1994. *Wastewater and stormwater applications of wetlands in Canada.* Issue paper No. 1994-1, Sustaining Wetlands, Ottawa, Ontario. CH2M Hill Engineering Ltd., Environment Canada, Canadian Wildlife Service, North American Wetlands Conservation Council (Canada).

Reed S.C., Crites, R.W and Middlebrooks, E.J. 1995. *Natural systems for waste management and treatment.* 2nd Edition, McGraw-Hill, New Tork. 433pp.

Reuter, J.E., T. Djohan and C.R. Goldman, 1992. The use of wetlands for nutrient removal from surface runoff in a cold climate region of California – Results from a newly constructed wetland at Lake Tahoe. *Journal of Environmental Management,* v. 36, p. 35 – 53.

Schiffer, D.M. 1989. Water – Quality variability in a central Florida wetland receiving highway runoff. IN: *Water: Laws and Management*, F.E. Davis (Ed.), Bethesda, MD., American Water Resources Association.

Serfling S. and C. Alsten 1979. An integrated, controlled environment aquaculture lagoon process for secondary or advanced wastewater treatment. IN: E. Middlebrooks *et al.* (eds.) *Performance and Upgrading of Wastewater Stabilization Ponds.* EPA 600/9-79-011. U.S. E.P.A., Cincinnati, Ohio, 217 pp.

Shutes R., J. Ellis, D. Revitt and T. Zhang 1993. The use of *Typha latifolia* for heavy metal pollution control in urban wetlands. IN: G. Moshiri (ed.) *Constructed wetlands for water quality improvement.* Lewis Publishers, Chelsea, Michigan, p. 407-414

Stein J., S. Proulx and D. Levesque 1994. Forest floor frost dynamics during spring snowmelt in a boreal forested basin. *Water Resources Research* V. 30, N. 4, P. 995 - 1007.

Stewart W. and S. Serfling 1979. The Solar Aquacell system for primary, secondary or advanced treatment of wastewaters. IN: Bastian and Reed, 1979, p. 377 - 412

Stewart, W. C. Alsten, S. Serfling and D. Mendola 1979. Pilot studies of the Solar Aquacell controlled aquaculture process for wastewater reclamation. IN: *Proceedings - Vol.1, Water Reuse - From research to Application.* AWWA Research Foundation, Denver , Colorado, 781 pp.

Strecker, E.W., Kersnar, J.M., Driscoll, E.D., and Homer, R.R. 1992. *The Use of Wetlands for Controlling Stormwater Pollution.* Terrene Institute, Washington, D.C, EPA/600.

Striegl R. 1985. *Effects of Stormwater Runoff on an Urban Lake, Lake Ellyn at Glen Ellyn,* Illinois Open-file Report 84-603 USGS, Lakewood, Colorado, 19pp.

Stuckey R., J. Wehrmeister,and R. Bartolotta 1978. Submersed aquatic vascular plants in ice-covered ponds of central Ohio. *Rhodora* V.80, N. 821, p. 575 - 580.

Surface, J.M., Peverly, J.H., Steenhuis, T.S. and Sanford, W.E, 1993. Effect of season, substrate composition and plsant growth on lanfill leachate treatment in a constructed wetland. IN: G. Moshiri (ed.) *Constructed Wetlands forWater Quality Improvement.* Lewis Publishers, Chelsea, Michigan, p. 461 - 472

Surrency D. 1993. Evaluation of aquatic plants for constructed wetlands. IN: G. Moshiri(ed.) *Constructed Wetlands forWater Quality Improvement.* Lewis Publishers, Chelsea, Michigan

Thompson, T.E., Conrad, R. and Zeikus, J.G., 1984. The regulation of carbon and electron flow in *Propionispiria arboris* : physiological function of hydrogenase and its role in propionate formation. *FEMS Microbiological Letters* 22: 265.

Todd J. 1991. Living machines for pure water: sewage as a resource. IN: P. Rothkrug and R. Olson (eds.) *Mending the Earth: a World for Our Grandchildren.* North Atlantic Books, Berkley, California and The Environmental Rescue Fund, San Francisco, California, p. 109 - 12

Urbanc-Bercic, O. 1994. Investigation into the use of constructed reedbeds for municipal waste dump leachate treatment. *Water Science and Technology* V.29, No. 4, pp.289 - 294

USEPA (United States Environmental Protection Agency) 1994. *Wetlands Treatment Database (North American wetlands for water quality treatment database, Version 1).* USEPA Risk Reduction Engineering Laboratory, Cincinnati, Ohio and Environmental Research Labor

Watson J., S. Reed, R. Kadlec, R. Knight and A. Whitehouse 1989. Performance expectations and loading rates for constructed wetlands. IN: D. Hammer (ed.) *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural.* Lewis Publishers, Chelsea, Michigan, p. 319 - 351.

Wetzel, R.G. Fundamental processes within natural and constructed wetland ecosystems: short-term versus long-term objectives. *Water Science and Technology,* V. 44, No. 11-12., pp 1-8.

Wigh, R.J., 1984. *Landfill Research at Boone County Field Site.* U.S. National Technical Information Service Rep. No. PB 84 - 161546, Springfield, Virginia.

Wilde F. 1989. Effects of storm-water impoundments on major-ion ratios in ground water. IN: *Ground water Issues and Solutions in the Potomac River Basin, Chesapeake Bay Region,* National Water Well Association, Dublin, Ohio, p. 287 - 311.

Witthar S. 1993. Wetland water treatment systems. IN: G. Moshiri (ed.) *Constructed Wetlands for Water Quality Improvement.* Lewis Publishers, Chelsea, Michigan, p. 147 - 156.

APPENDIX D:

Analytical Procedures

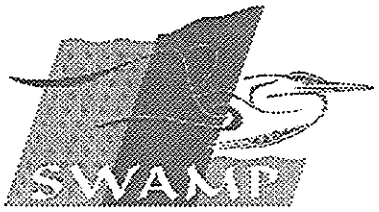
Table D1: OMOE analytical procedures employed with liquid samples in the Aurora stormwater wetland study*

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
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
E3170A	COD3170	Chemical Oxygen Demand	Samples mixed with an acidified Potassium Dichromate Solution to suppress chloride interference. Sulphuric acid containing silver sulphate is added and mixture is digested in a mechanical convection oven for 3 h at 149 ±1°C	
E3172A	SULP3172	Sulphate	Via ion chromatography, sulphate is separated from other anions using columns packed with ion exchange resin and an eluent solution of sodium bicarbonate and sodium carbonate. After separation, sulphate is converted to the acid form by ion exchange using a micromembrane suppressor and its concentration is determined from the conductivity of the sulphuric acid produced.	
E3179A	PHEN3179	Phenolic Compounds	Unfiltered sample aliquot is autoanalyzed and the distillate is mixed with a tartrate-borax buffer, pH 9.4, and 4-aminoantipyrine to produce an antipyrine dye which is oxidized by alkaline ferricyanide. The absorbance of the red antipyrine dye is measure colourimetricallly in a 5 cm flow cell at 505 nm. The result in mmg/L phenol is read from the chart recorder trace by comparison with peaks produced by a similarly treated series of Standards.	The term 'phenolic compounds' is applied to those hydroxy-derivatives of benzene which react under the conditions of the tests, with the reagents used.
E3182A	BOD3182	Biochemical Oxygen Demand	Sample is diluted such that 50% of dissolved oxygen is depleted after 5 days incubation. Dissolved oxygen is determined after preparation and again after incubation. BOD expressed as the amount of dissolved oxygen in mg utilized by 1 litre of sample during a 5-day incubation period at 20°C	
E3188B	TSD3188	Total, suspended and dissolved solids	Suspended solids are determined as the material removed from suspension by a 1.5 to 2.0 µm glass fibre filter after dring at 103 ±2°C. Dissolved solids is the material that remains in solution after suspended solids are filtered out, as determined by evaporating to dryness at 103 ±2°C. Total solids is the sum of suspended and dissolved solids.	
E3201B	SXT3201	Solvent Extractable	Liquid – liquid extraction using dichloromethane as the extraction solvent.	includes non-volatile petroleum hydrocarbons, vegetable oils, animal fats, soaps, greases and waxes.

* See Appendix H for methods used on sediment samples.

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 m 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
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	
E3188B	TSD3188	Total, Suspended and Dissolved 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. dissolved solids is the material that remains in solution after suspended solids are filtered out, as determined by evaporating to dryness at 103° ±2°C. Total solids is the sum of suspended and dissolved solids.	
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	

E3371A	EC3371	Escherichia coli	Membrane filtration procedures are used to recover and enumerate several bacteria or bacterial groups.	
E3386A	MET3386	Metals	Inductively coupled plasma (ICP) following ultrasonic nebulizer	Digestion is not used.



APPENDIX E

Water Quality Statistics

Table E1: Summary of influent water quality statistics for the entire study period from July 1996 to November 1998

Parameter	Unit	RMDL*	N	%>D.L.	MIN	MAX	MEAN	SD	95%CI-LL	95%CI-UL
Dissolved oxygen	mg/L	1	13	100	4.0	11.6	8.7	1.44	7.2	10.6
Biological Oxygen Demand	mg/L	0.2	29	100	0.8	14.6	3.0	1.81	2.4	3.8
Chemical Oxygen Demand	mg/L	1	29	100	12.0	102.0	34.5	1.71	28.4	42.0
pH	none	n/a	29	100	7.3	9.4	8.0	1.04	7.9	8.2
Conductivity	µS/cm	1	29	100	105.0	3230.0	338.7	2.64	237.9	482.0
Turbidity	FTU	0.01	29	100	19.9	691.0	90.9	2.59	64.3	128.4
Suspended solids	mg/L	2.5	29	100	14.0	945.0	110.6	3.41	70.7	172.9
Dissolved solids	mg/L	10	29	100	32.0	1690.0	182.9	2.49	131.3	255.0
Total solids	mg/L	10	29	100	100.0	1880.0	388.5	2.15	294.0	513.5
Chloride	mg/L	0.2	29	100	1.8	949.0	29.8	6.67	14.9	59.4
Phosphate	mg/L	0.0005	29	100	0.02	0.44	0.11	2.04	0.09	0.15
Total phosphours	mg/L	0.002	29	100	0.08	2.15	0.31	2.16	0.24	0.42
Total ammonia	mg/L	0.002	29	100	0.010	5.650	0.135	3.70	0.084	0.218
Nitrite	mg/L	0.001	29	100	0.00	0.23	0.04	2.44	0.03	0.06
Nitrate	mg/L	0.006	29	100	0.2	2.1	0.7	1.66	0.6	0.8
TKN	mg/L	0.02	29	100	0.6	9.5	1.4	1.93	1.1	1.8
Sulphate	mg/L	0.25	17	100	4.5	37.5	12.7	1.68	9.9	16.3
Phenolics	mg/L	0.2	17	100	1.0	4.8	2.0	1.54	1.6	2.4
Solvent Extractable	mg/L	0.5	29	90	0.3	11.0	2.3	2.63	1.6	3.3
E.coli	c./100 ml	4	24	100	40.0	24000.0	1190.0	4.45	655.1	2161.5
Copper	µg/L	1.6	29	100	3.4	37.2	9.5	1.86	7.6	11.9
Nickel	µg/L	1.3	29	69	0.7	3.7	1.6	1.81	1.3	2.0
Zinc	µg/L	0.6	29	100	1.6	162.0	34.3	2.59	24.2	48.5
Cadmium	µg/L	0.6	29	62	0.3	3.30	0.95	2.57	0.67	1.34
Cobalt	µg/L	1.3	29	38	0.7	5.0	1.0	1.92	0.8	1.3
Chromium	µg/L	1.4	29	76	0.7	7.8	2.0	2.00	1.6	2.6
Lead	µg/L	10	29	10	5.0	12.8	5.5	1.31	5.0	6.0
Iron	µg/L	0.8	29	100	11.9	1920.0	593.5	2.47	427.1	824.7
Manganese	µg/L	0.2	29	100	0.3	384.0	75.7	3.60	47.5	120.7
Aluminum	µg/L	11	29	100	38.3	2420.0	489.3	2.37	357.6	669.5
Vanadium	µg/L	1.5	29	76	0.8	5.9	2.0	1.81	1.6	2.5
Molybdenum	µg/L	1.6	29	0	all data less than the detection limit					
Barium	µg/L	0.2	29	100	7.3	55.7	21.2	1.60	17.9	25.2
Beryllium	µg/L	0.02	29	76	0.01	0.27	0.04	2.51	0.03	0.05
Strontium	µg/L	0.1	29	100	57.5	378.0	147.7	1.63	123.7	176.4
Titanium	µg/L	0.9	29	89.7	0.45	25.00	4.07	2.48	2.93	5.67
Calcium	mg/L	0.005	16	100	19.3	101.0	41.5	1.61	32.9	52.3
Magnesium	mg/L	0.008	16	100	0.8	7.9	3.0	1.89	2.2	4.1

Note: The mean, standard deviation and 95% confidence intervals are based on log normally distributed data.

* RMDL = Reporting Method Detection Limit

Table E2: Summary of effluent water quality statistics for the entire study period from July 1996 to November 1998 .

Parameter	Unit	RMDL*	N	%>D.L.	MIN	MAX	MEAN	SD	95%CI-LL	95%CI-UL
Dissolved oxygen	mg/L	1	13	100	3.4	10.7	7.8	1.44	6.4	9.5
Biological Oxygen Demand	mg/L	0.2	29	100	0.8	7.0	2.5	1.57	2.1	2.9
Chemical Oxygen Demand	mg/L	1	29	100	15.3	43.7	28.4	1.35	25.5	31.7
pH	none	n/a	29	100	7.5	8.2	7.9	1.02	7.8	7.9
Conductivity	µS/cm	1	29	100	194.3	4770.7	441.7	2.46	318.2	613.2
Turbidity	FTU	0.01	29	100	3.1	112.2	23.5	2.58	16.6	33.2
Suspended solids	mg/L	2.5	29	100	4.8	155.4	23.8	2.28	17.6	32.1
Dissolved solids	mg/L	10	29	100	126.2	2637.7	272.4	2.27	202.2	367.0
Total solids	mg/L	10	29	100	30.1	2672.6	295.6	2.38	215.6	405.3
Chloride	mg/L	0.2	29	100	4.9	1566.7	45.1	5.13	24.9	81.7
Phosphate	mg/L	0.0005	29	100	0.01	0.25	0.05	2.52	0.04	0.07
Total phosphours	mg/L	0.002	29	100	0.04	0.38	0.14	1.73	0.12	0.18
Total ammonia	mg/L	0.002	29	97	0.002	0.439	0.060	3.70	0.037	0.096
Nitrite	mg/L	0.001	29	100	0.01	0.20	0.03	2.12	0.02	0.04
Nitrate	mg/L	0.006	29	97	0.0	0.9	0.3	3.67	0.2	0.4
TKN	mg/L	0.02	29	100	0.5	2.0	0.9	1.46	0.8	1.0
Sulphate	mg/L	0.25	17	100	9.4	24.3	15.6	1.29	13.8	17.6
Phenolics	mg/L	0.2	17	100	0.7	11.6	1.5	1.82	1.1	2.0
Solvent Extractable	mg/L	0.5	29	90	0.3	3.6	1.2	2.06	0.9	1.6
E.coli	c./100 ml	4	29	100	43.8	3208.2	464.4	2.76	321.0	671.9
Copper	µg/L	1.6	29	100	2.6	28.3	5.3	1.55	4.5	6.2
Nickel	µg/L	1.3	29	38	0.7	3.5	1.0	1.67	0.8	1.2
Zinc	µg/L	0.6	29	100	11.3	192.7	31.8	2.11	24.2	41.7
Cadmium	µg/L	0.6	29	14	0.3	1.67	0.35	1.56	0.30	0.42
Cobalt	µg/L	1.3	29	3	0.7	1.4	0.7	1.15	0.6	0.7
Chromium	µg/L	1.4	29	66	0.7	7.2	1.5	1.84	1.2	1.8
Lead	µg/L	10	29	3	5.0	12.2	5.2	1.18	4.9	5.5
Iron	µg/L	0.8	29	100	129.7	1081.0	414.9	1.80	335.0	513.8
Manganese	µg/L	0.2	29	100	18.6	252.4	53.8	1.87	42.9	67.5
Aluminum	µg/L	11	29	100	64.3	1067.7	334.4	2.12	254.3	439.7
Vanadium	µg/L	1.5	29	59	0.8	22.7	1.5	2.12	1.1	2.0
Molybdenum	µg/L	1.6	29	0	all data less than detection limit					
Barium	µg/L	0.2	29	100	8.9	48.7	18.7	1.50	16.2	21.7
Beryllium	µg/L	0.02	29	41	0.01	0.10	0.02	2.10	0.01	0.02
Strontium	µg/L	0.1	29	100	93.0	381.3	157.9	1.46	137.6	181.2
Titanium	µg/L	0.9	29	100	1.62	18.01	5.25	1.67	4.36	6.32
Calcium	mg/L	0.005	16	100	26.8	134.8	41.1	1.57	33.0	51.3
Magnesium	mg/L	0.008	16	100	1.5	5.6	2.5	1.57	2.0	3.1

Note: The mean, standard deviation and 95% confidence intervals are based on log normally distributed data.

* RMDL = Reporting Method Detection Limit

Table E3: Summary of influent water quality statistics during the spring (March 21 to June 20) in 1997 and 1998

Parameter	Unit	RMDL*	N	%>D.L.	MIN	MAX	MEAN	SD	95%CI-LL	95%CI-UL
Dissolved oxygen	mg/L	1	3	100	9.8	10.5	10.2	1.04	10.6	9.8
Biological Oxygen Demand	mg/L	0.2	7	100	2.8	14.6	4.7	1.78	7.2	3.1
Chemical Oxygen Demand	mg/L	1	7	100	37.0	102.0	50.4	1.45	66.4	38.2
pH	none	n/a	7	100	7.3	8.4	7.9	1.04	8.1	7.6
Conductivity	µS/cm	1	7	100	146.0	1719.5	334.0	2.43	645.0	172.9
Turbidity	FTU	0.01	7	100	19.9	197.0	88.5	2.41	169.8	46.1
Suspended solids	mg/L	2.5	7	100	14.0	335.0	110.4	3.15	258.4	47.2
Dissolved solids	mg/L	10	7	100	94.0	921.0	208.8	2.32	389.0	112.0
Total solids	mg/L	10	7	100	150.0	1117.7	361.2	2.13	632.4	206.3
Chloride	mg/L	0.2	7	100	11.8	507.3	43.3	3.71	114.3	16.4
Phosphate	mg/L	0.0005	7	100	0.02	0.27	0.10	2.33	0.18	0.05
Total phosphours	mg/L	0.002	7	100	0.18	1.03	0.36	1.76	0.55	0.24
Total ammonia	mg/L	0.002	7	100	0.182	5.650	0.495	3.36	1.216	0.202
Nitrite	mg/L	0.001	7	100	0.03	0.23	0.09	1.99	0.16	0.06
Nitrate	mg/L	0.006	7	100	0.4	1.4	0.8	1.66	1.2	0.6
TKN	mg/L	0.02	7	100	1.3	9.5	2.1	2.01	3.5	1.2
Sulphate	mg/L	0.25	3	100	10.0	23.3	15.2	1.53	24.5	9.4
Phenolics	mg/L	0.2	3	100	1.6	2.7	2.1	1.29	2.8	1.6
Solvent Extractable	mg/L	0.5	7	100	1.5	11.0	4.3	2.12	7.6	2.5
E.coli	c./100 ml	4	7	100	410.0	5400.0	1517.7	2.60	3080.0	747.9
Copper	µg/L	1.6	7	100	4.1	20.1	9.6	1.80	14.8	6.2
Nickel	µg/L	1.3	7	71	1.1	3.2	1.9	1.51	2.6	1.4
Zinc	µg/L	0.6	7	100	21.6	162.0	49.3	1.95	80.9	30.1
Cadmium	µg/L	0.6	7	43	0.3	2.80	0.67	2.86	1.45	0.31
Cobalt	µg/L	1.3	7	29	0.7	1.7	0.9	1.62	1.2	0.6
Chromium	µg/L	1.4	7	57	0.7	4.2	1.6	2.26	2.9	0.9
Lead	µg/L	10	7	14	5.0	12.8	5.7	1.43	7.4	4.4
Iron	µg/L	0.8	7	100	357.0	1560.0	757.3	1.70	1122.2	511.0
Manganese	µg/L	0.2	7	100	57.5	180.0	100.1	1.60	141.9	70.7
Aluminum	µg/L	11	7	100	197.0	1000.2	554.2	1.79	852.3	360.3
Vanadium	µg/L	1.5	7	86	1.4	4.0	2.6	1.40	3.3	2.0
Molybdenum	µg/L	1.6	7	0	all data below the detection limit					
Barium	µg/L	0.2	7	100	12.3	33.1	19.0	1.48	25.4	14.2
Beryllium	µg/L	0.02	7	86	0.01	0.10	0.04	2.29	0.08	0.02
Strontium	µg/L	0.1	7	100	74.5	378.0	145.6	1.77	222.0	95.5
Titanium	µg/L	0.9	7	100	1.90	7.41	4.74	1.62	6.79	3.31
Calcium	mg/L	0.005	4	100	19.3	85.1	44.4	2.12	92.7	21.2
Magnesium	mg/L	0.008	4	100	1.4	7.9	3.4	2.53	8.5	1.4

Note: The mean, standard deviation and 95% confidence intervals are based on log normally distributed data.

* RMDL = Reporting Method Detection Limit

Table E4: Summary of effluent water quality statistics during the spring (March 21 to June 20) in 1997 and 1998.

Parameter	Unit	RMDL*	N	%>D.L.	MIN	MAX	MEAN	SD	95%CI-LL	95%CI-UL
Dissolved oxygen	mg/L	1	3	100	10.2	10.7	10.5	1.02	10.8	10.3
Biological Oxygen Demand	mg/L	0.2	7	100	2.7	3.5	3.1	1.11	3.3	2.9
Chemical Oxygen Demand	mg/L	1	7	100	16.8	43.7	30.6	1.37	38.6	24.2
pH	none	n/a	7	100	7.7	8.2	7.9	1.03	8.0	7.7
Conductivity	µS/cm	1	7	100	217.9	500.8	325.9	1.36	409.0	259.6
Turbidity	FTU	0.01	7	100	3.1	62.8	16.1	2.75	34.1	7.6
Suspended solids	mg/L	2.5	7	100	4.8	36.4	16.8	2.01	28.2	10.1
Dissolved solids	mg/L	10	7	100	142.0	326.2	210.4	1.37	265.5	166.7
Total solids	mg/L	10	7	100	153.9	362.0	228.9	1.38	291.3	179.9
Chloride	mg/L	0.2	7	100	13.9	97.4	34.5	2.19	61.7	19.3
Phosphate	mg/L	0.0005	7	100	0.02	0.23	0.06	2.22	0.12	0.04
Total phosphours	mg/L	0.002	7	100	0.09	0.36	0.16	1.73	0.25	0.11
Total ammonia	mg/L	0.002	7	100	0.045	0.420	0.172	2.25	0.314	0.095
Nitrite	mg/L	0.001	7	100	0.02	0.11	0.05	1.73	0.08	0.03
Nitrate	mg/L	0.006	7	100	0.0	0.9	0.4	3.28	0.9	0.1
TKN	mg/L	0.02	7	100	0.7	2.0	1.2	1.57	1.7	0.9
Sulphate	mg/L	0.25	3	100	17.3	24.3	20.2	1.19	24.5	16.6
Phenolics	mg/L	0.2	3	100	1.2	1.5	1.3	1.15	1.6	1.1
Solvent Extractable	mg/L	0.5	7	100	0.6	3.6	1.4	1.85	2.2	0.9
E.coli	c./100 ml	4	7	100	43.8	1115.0	319.4	3.33	778.8	131.0
Copper	µg/L	1.6	7	100	2.6	28.3	5.7	2.21	10.3	3.2
Nickel	µg/L	1.3	7	43	0.7	1.5	0.9	1.52	1.2	0.7
Zinc	µg/L	0.6	7	100	16.6	127.6	30.4	2.07	52.1	17.7
Cadmium	µg/L	0.6	7	29	0.3	1.67	0.43	1.97	0.72	0.26
Cobalt	µg/L	1.3	7	0	all data less than the detection limit					
Chromium	µg/L	1.4	7	57	0.7	2.3	1.2	1.71	1.8	0.8
Lead	µg/L	10	7	0	all data less than the detection limit					
Iron	µg/L	0.8	7	100	188.2	649.4	353.5	1.55	489.0	255.5
Manganese	µg/L	0.2	7	100	21.3	252.4	46.0	2.35	86.6	24.4
Aluminum	µg/L	11	7	100	64.3	522.6	245.1	2.20	440.1	136.5
Vanadium	µg/L	1.5	7	43	0.8	22.7	1.6	3.52	4.2	0.6
Molybdenum	µg/L	1.6	7	0	all data less than the detection limit					
Barium	µg/L	0.2	7	100	8.9	19.9	13.7	1.28	16.5	11.4
Beryllium	µg/L	0.02	7	57	0.01	0.04	0.02	1.84	0.03	0.01
Strontium	µg/L	0.1	7	100	101.4	181.6	128.1	1.23	149.2	109.9
Titanium	µg/L	0.9	7	100	1.62	12.24	4.51	1.90	7.26	2.80
Calcium	mg/L	0.005	4	100	26.9	40.9	32.1	1.20	38.4	26.8
Magnesium	mg/L	0.008	4	100	1.6	3.3	1.9	1.44	2.8	1.4

Note: The mean, standard deviation and 95% confidence intervals are based on log normally distributed data.

* RMDL = Reporting Method Detection Limit

Table E5: Summary of influent water quality statistics during the summer (June 21 to Sept 21) in 1996, 1997 and 1998

Parameter	Unit	RMDL*	N	%>D.L.	MIN	MAX	MEAN	SD	95%CI-LL	95%CI-UL
Dissolved oxygen	mg/L	1	4	100	4.0	9.2	5.6	1.43	7.9	3.9
Biological Oxygen Demand	mg/L	0.2	9	100	1.2	6.2	3.3	1.73	4.7	2.3
Chemical Oxygen Demand	mg/L	1	9	100	18.0	92.0	35.6	1.71	50.4	25.1
pH	none	n/a	9	100	7.7	8.3	8.0	1.02	8.1	7.9
Conductivity	µS/cm	1	9	100	105.0	220.0	152.8	1.28	179.7	129.9
Turbidity	FTU	0.01	9	100	27.7	564.0	120.2	2.56	222.4	64.9
Suspended solids	mg/L	2.5	9	100	15.5	856.0	193.8	3.62	448.7	83.7
Dissolved solids	mg/L	10	9	100	68.0	144.0	100.7	1.29	119.0	85.2
Total solids	mg/L	10	9	100	158.0	966.0	354.0	2.00	557.3	224.9
Chloride	mg/L	0.2	9	100	1.8	11.2	4.2	1.89	6.4	2.8
Phosphate	mg/L	0.0005	9	100	0.04	0.44	0.11	2.25	0.19	0.07
Total phosphours	mg/L	0.002	9	100	0.08	1.08	0.30	2.51	0.54	0.16
Total ammonia	mg/L	0.002	9	100	0.010	0.480	0.082	3.66	0.192	0.035
Nitrite	mg/L	0.001	9	100	0.00	0.10	0.03	3.33	0.08	0.02
Nitrate	mg/L	0.006	9	100	0.3	2.1	0.7	1.84	1.1	0.5
TKN	mg/L	0.02	9	100	0.6	3.3	1.3	1.77	2.0	0.9
Sulphate	mg/L	0.25	8	100	4.5	37.5	11.4	1.94	18.1	7.2
Phenolics	mg/L	0.2	8	100	1.0	4.2	1.8	1.65	2.6	1.3
Solvent Extractable	mg/L	0.5	9	89	0.3	7.5	1.9	2.66	3.6	1.0
E.coli	c./100 ml	4	6	100	1780.0	24000.0	4658.7	2.44	9498.7	2284.9
Copper	µg/L	1.6	9	100	3.4	37.2	11.0	2.20	18.5	6.6
Nickel	µg/L	1.3	9	78	0.7	3.6	1.9	1.99	2.9	1.2
Zinc	µg/L	0.6	9	100	1.6	99.4	22.5	4.24	57.8	8.7
Cadmium	µg/L	0.6	9	78	0.3	2.96	1.46	2.19	2.43	0.87
Cobalt	µg/L	1.3	9	67	0.7	5.0	1.5	2.02	2.3	0.9
Chromium	µg/L	1.4	9	67	0.7	4.4	1.8	2.13	3.0	1.1
Lead	µg/L	10	9	11	5.0	12.2	5.5	1.35	6.7	4.5
Iron	µg/L	0.8	9	100	11.9	1920.0	507.7	4.50	1356.1	190.1
Manganese	µg/L	0.2	9	100	0.3	340.0	67.4	8.21	266.6	17.0
Aluminum	µg/L	11	9	100	38.3	2420.0	570.4	3.29	1240.7	262.2
Vanadium	µg/L	1.5	9	89	0.8	5.9	2.6	1.93	4.0	1.7
Molybdenum	µg/L	1.6	9	0	all data less than the detction limit					
Barium	µg/L	0.2	9	100	7.3	48.4	21.3	1.91	32.5	14.0
Beryllium	µg/L	0.02	9	56	0.01	0.24	0.03	3.35	0.07	0.01
Strontium	µg/L	0.1	9	100	57.5	230.0	119.1	1.56	159.1	89.2
Titanium	µg/L	0.9	9	89	0.45	25.00	4.14	3.11	8.68	1.97
Calcium	mg/L	0.005	5	100	23.7	101.0	40.7	1.75	66.3	25.0
Magnesium	mg/L	0.008	5	100	0.8	7.8	2.4	2.27	4.9	1.2

Note: The mean, standard deviation and 95% confidence intervals are based on log normally distributed data.

* RMDL = Reporting Method Detection Limit

Table E6: Summary of effluent water quality statistics during the summer (June 21 to Sept 21) in 1996, 1997 and 1998.

Parameter	Unit	RMDL*	N	%>D.L.	MIN	MAX	MEAN	SD	95%CI-LL	95%CI-UL
Dissolved oxygen	mg/L	1	4	100	3.4	7.6	5.0	1.40	6.9	3.6
Biological Oxygen Demand	mg/L	0.2	9	100	1.8	7.0	3.1	1.64	4.2	2.2
Chemical Oxygen Demand	mg/L	1	9	100	23.3	41.8	31.5	1.24	36.1	27.4
pH	none	n/a	9	100	7.6	8.2	7.8	1.02	8.0	7.7
Conductivity	µS/cm	1	9	100	194.3	281.3	228.6	1.17	253.3	206.3
Turbidity	FTU	0.01	9	100	4.8	73.8	17.9	2.82	35.3	9.1
Suspended solids	mg/L	2.5	9	100	5.4	155.4	25.7	2.99	52.5	12.5
Dissolved solids	mg/L	10	9	100	126.2	185.0	151.0	1.18	168.5	135.3
Total solids	mg/L	10	9	100	30.1	281.6	160.2	1.94	247.2	103.8
Chloride	mg/L	0.2	9	100	4.9	23.2	10.5	1.54	13.9	7.9
Phosphate	mg/L	0.0005	9	100	0.01	0.08	0.02	2.38	0.04	0.01
Total phosphours	mg/L	0.002	9	100	0.04	0.19	0.10	1.74	0.15	0.07
Total ammonia	mg/L	0.002	9	89	0.002	0.439	0.034	4.23	0.087	0.013
Nitrite	mg/L	0.001	9	100	0.01	0.20	0.02	2.80	0.04	0.01
Nitrate	mg/L	0.006	9	89	0.0	0.6	0.1	4.43	0.2	0.0
TKN	mg/L	0.02	9	100	0.5	1.1	0.8	1.33	1.0	0.7
Sulphate	mg/L	0.25	8	100	9.4	18.6	14.1	1.28	16.7	11.9
Phenolics	mg/L	0.2	8	100	1.1	11.6	1.8	2.21	3.1	1.0
Solvent Extractable	mg/L	0.5	9	100	0.6	2.5	1.4	1.60	1.9	1.0
E.coli	c./100 ml	4	9	100	134.8	1351.2	475.4	2.29	816.8	276.7
Copper	µg/L	1.6	9	100	3.1	6.6	4.8	1.30	5.7	4.0
Nickel	µg/L	1.3	9	33	0.7	3.5	1.0	1.87	1.6	0.7
Zinc	µg/L	0.6	9	100	15.2	63.5	24.0	1.67	33.5	17.1
Cadmium	µg/L	0.6	9	0	all data less than the detection limit					
Cobalt	µg/L	1.3	9	0	all data less than the detection limit					
Chromium	µg/L	1.4	9	33	0.7	2.5	1.0	1.76	1.5	0.7
Lead	µg/L	10	9	0	all data less than the detection limit					
Iron	µg/L	0.8	9	100	129.7	1081.0	385.6	2.24	652.6	227.9
Manganese	µg/L	0.2	9	100	21.5	179.9	57.8	1.87	86.9	38.4
Aluminum	µg/L	11	9	100	69.9	1067.7	321.5	2.68	612.1	168.8
Vanadium	µg/L	1.5	9	78	0.8	3.2	1.7	1.68	2.4	1.2
Molybdenum	µg/L	1.6	9	0	all data less than the detection limit					
Barium	µg/L	0.2	9	100	11.7	29.9	17.0	1.43	21.5	13.5
Beryllium	µg/L	0.02	9	22	0.01	0.08	0.01	2.02	0.02	0.01
Strontium	µg/L	0.1	9	100	93.0	192.4	127.1	1.27	148.7	108.7
Titanium	µg/L	0.9	9	100	2.16	18.01	5.19	1.82	7.68	3.50
Calcium	mg/L	0.005	5	100	26.8	134.8	39.6	1.99	72.4	21.7
Magnesium	mg/L	0.008	5	100	1.5	2.3	1.8	1.16	2.1	1.6

Note: The mean, standard deviation and 95% confidence intervals are based on log normally distributed data.

* RMDL = Reporting Method Detection Limit

Table E7: Summary of influent water quality statistics during the fall (Sept 22 to Dec 20) in 1996, 1997 and 1998

Parameter	Unit	RMDL*	N	%>D.L.	MIN	MAX	MEAN	SD	95%CI-LL	95%CI-UL
Dissolved oxygen	mg/L	1	4	100	9.0	11.6	10.5	1.12	11.7	9.4
Biological Oxygen Demand	mg/L	0.2	7	100	1.6	5.2	2.8	1.50	3.8	2.1
Chemical Oxygen Demand	mg/L	1	7	100	12.0	85.0	27.6	1.96	45.4	16.8
pH	none	n/a	7	100	7.9	8.2	8.0	1.01	8.1	8.0
Conductivity	µS/cm	1	7	100	146.0	1410.0	351.3	2.42	676.3	182.5
Turbidity	FTU	0.01	7	100	23.0	691.0	99.3	3.69	261.2	37.8
Suspended solids	mg/L	2.5	7	100	30.0	945.0	121.0	3.93	333.6	43.9
Dissolved solids	mg/L	10	7	100	32.0	312.0	124.1	2.09	214.0	72.0
Total solids	mg/L	10	7	100	100.0	1260.0	298.4	2.43	575.5	154.7
Chloride	mg/L	0.2	7	100	5.2	402.0	37.6	5.61	135.1	10.5
Phosphate	mg/L	0.0005	7	100	0.03	0.35	0.13	2.16	0.23	0.07
Total phosphours	mg/L	0.002	7	100	0.12	2.15	0.41	2.67	0.84	0.20
Total ammonia	mg/L	0.002	7	100	0.052	0.132	0.091	1.39	0.116	0.072
Nitrite	mg/L	0.001	7	100	0.02	0.05	0.03	1.46	0.04	0.02
Nitrate	mg/L	0.006	7	100	0.2	0.8	0.5	1.45	0.6	0.3
TKN	mg/L	0.02	7	100	0.6	4.1	1.3	1.96	2.2	0.8
Sulphate	mg/L	0.25	4	100	9.0	19.0	13.1	1.43	18.6	9.3
Phenolics	mg/L	0.2	4	100	1.4	4.8	2.2	1.74	3.7	1.3
Solvent Extractable	mg/L	0.5	7	86	0.3	9.0	2.2	3.20	5.3	0.9
E.coli	c./100 ml	4	5	100	100.0	8600.0	940.8	5.07	3901.7	226.8
Copper	µg/L	1.6	7	100	5.2	21.1	9.4	1.88	15.1	5.9
Nickel	µg/L	1.3	7	57	0.7	3.7	1.6	2.02	2.7	0.9
Zinc	µg/L	0.6	7	100	19.0	102.0	41.3	1.84	64.8	26.3
Cadmium	µg/L	0.6	7	86	0.3	3.30	1.41	2.24	2.57	0.78
Cobalt	µg/L	1.3	7	29	0.7	4.2	1.1	2.29	1.9	0.6
Chromium	µg/L	1.4	7	100	1.6	4.5	2.4	1.44	3.2	1.8
Lead	µg/L	10	7	14	5.0	11.2	5.6	1.36	7.0	4.5
Iron	µg/L	0.8	7	100	359.0	1480.0	673.4	1.65	977.9	463.7
Manganese	µg/L	0.2	7	100	36.3	384.0	97.9	2.64	201.1	47.6
Aluminum	µg/L	11	7	100	68.5	2100.0	427.5	2.81	918.6	199.0
Vanadium	µg/L	1.5	7	71	0.8	4.2	1.7	1.87	2.7	1.1
Molybdenum	µg/L	1.6	7	0	all data less than the detection limit					
Barium	µg/L	0.2	7	100	16.2	55.7	24.1	1.63	34.5	16.7
Beryllium	µg/L	0.02	7	100	0.02	0.27	0.05	2.24	0.09	0.03
Strontium	µg/L	0.1	7	100	79.4	357.0	154.6	1.70	228.7	104.4
Titanium	µg/L	0.9	7	71	0.45	7.32	2.50	3.31	6.07	1.03
Calcium	mg/L	0.005	3	100	27.6	37.9	32.7	1.17	39.2	27.3
Magnesium	mg/L	0.008	3	100	2.5	3.3	3.0	1.19	3.7	2.5

Note: The mean, standard deviation and 95% confidence intervals are based on log normally distributed data.

* RMDL = Reporting Method Detection Limit

Table E8: Summary of effluent water quality statistics during the fall (Sept 22 to Dec 20) in 1996, 1997 and 1998

Parameter	Unit	RMDL*	N	%>D.L.	MIN	MAX	MEAN	SD	95%CI-LL	95%CI-UL
Dissolved oxygen	mg/L	1	4	100	8.4	9.8	9.1	1.06	9.6	8.5
Biological Oxygen Demand	mg/L	0.2	7	100	1.6	4.1	2.1	1.44	2.8	1.6
Chemical Oxygen Demand	mg/L	1	7	100	15.6	31.2	22.6	1.30	27.4	18.5
pH	none	n/a	7	100	7.7	8.1	7.9	1.02	8.0	7.8
Conductivity	µS/cm	1	7	100	213.0	1569.2	460.3	2.05	784.3	270.2
Turbidity	FTU	0.01	7	100	8.8	112.2	32.4	2.70	67.7	15.5
Suspended solids	mg/L	2.5	7	100	9.4	83.2	26.0	2.45	50.4	13.4
Dissolved solids	mg/L	10	7	100	138.4	566.2	264.2	1.72	394.2	177.1
Total solids	mg/L	10	7	100	176.4	653.2	305.5	1.59	430.9	216.6
Chloride	mg/L	0.2	7	100	8.0	428.6	55.8	4.42	167.7	18.6
Phosphate	mg/L	0.0005	7	100	0.02	0.25	0.09	2.11	0.16	0.05
Total phosphours	mg/L	0.002	7	100	0.07	0.32	0.18	1.66	0.26	0.12
Total ammonia	mg/L	0.002	7	100	0.026	0.123	0.050	1.89	0.080	0.031
Nitrite	mg/L	0.001	7	100	0.01	0.06	0.03	1.53	0.04	0.02
Nitrate	mg/L	0.006	7	100	0.3	0.7	0.5	1.42	0.6	0.4
TKN	mg/L	0.02	7	100	0.5	1.1	0.8	1.34	1.0	0.6
Sulphate	mg/L	0.25	4	100	13.6	21.7	17.6	1.22	21.4	14.5
Phenolics	mg/L	0.2	4	100	0.7	1.9	1.1	1.58	1.8	0.7
Solvent Extractable	mg/L	0.5	7	57	0.3	2.8	0.7	2.91	1.6	0.3
E.coli	c./100 ml	4	7	100	562.9	3208.2	1106.7	1.93	1801.9	679.8
Copper	µg/L	1.6	7	100	4.1	8.4	5.0	1.27	6.0	4.2
Nickel	µg/L	1.3	7	43	0.7	2.3	1.0	1.76	1.5	0.7
Zinc	µg/L	0.6	7	100	15.7	42.2	26.3	1.49	35.4	19.6
Cadmium	µg/L	0.6	7	29	0.3	1.04	0.41	1.74	0.62	0.27
Cobalt	µg/L	1.3	7	14	0.7	1.4	0.7	1.32	0.9	0.6
Chromium	µg/L	1.4	7	86	0.7	2.9	1.8	1.60	2.6	1.3
Lead	µg/L	10	7	14	5.0	12.2	5.7	1.40	7.3	4.4
Iron	µg/L	0.8	7	100	200.2	1045.6	480.9	1.97	793.5	291.5
Manganese	µg/L	0.2	7	100	18.6	147.0	52.1	2.00	87.0	31.2
Aluminum	µg/L	11	7	100	175.0	1048.7	424.1	2.00	709.2	253.6
Vanadium	µg/L	1.5	7	57	0.8	2.9	1.4	1.85	2.2	0.9
Molybdenum	µg/L	1.6	7	0	all data less than the detection limit					
Barium	µg/L	0.2	7	100	14.7	32.3	20.9	1.31	25.5	17.1
Beryllium	µg/L	0.02	7	57	0.01	0.10	0.03	2.67	0.06	0.01
Strontium	µg/L	0.1	7	100	130.4	266.4	172.9	1.26	205.5	145.5
Titanium	µg/L	0.9	7	100	3.59	8.19	5.23	1.36	6.56	4.17
Calcium	mg/L	0.005	3	100	31.2	43.1	37.6	1.18	45.4	31.1
Magnesium	mg/L	0.008	3	100	2.8	3.3	3.0	1.08	3.3	2.8

Note: The mean, standard deviation and 95% confidence intervals are based on log normally distributed data.

* RMDL = Reporting Method Detection Limit

Table E9: Summary of influent water quality statistics during the winter (Dec 21 to March 20) in 1997 and 1998

Parameter	Unit	RMDL*	N	%>D.L.	MIN	MAX	MEAN	SD	95%CI-LL	95%CI-UL
Dissolved oxygen	mg/L	1	2	100	11.5	11.5	11.5	1.00	11.5	11.5
Biological Oxygen Demand	mg/L	0.2	6	100	0.8	4.0	1.8	1.67	2.7	1.2
Chemical Oxygen Demand	mg/L	1	6	100	22.0	53.0	27.6	1.40	36.2	21.1
pH	none	n/a	6	100	7.8	9.4	8.2	1.07	8.7	7.8
Conductivity	µS/cm	1	6	100	423.0	3230.0	1088.8	2.03	1921.0	617.1
Turbidity	FTU	0.01	6	100	35.3	113.0	55.5	1.61	81.2	38.0
Suspended solids	mg/L	2.5	6	100	24.0	108.0	43.0	1.75	67.3	27.5
Dissolved solids	mg/L	10	6	100	226.0	1690.0	603.8	1.98	1044.5	349.0
Total solids	mg/L	10	6	100	284.0	1880.0	661.8	1.92	1114.4	393.1
Chloride	mg/L	0.2	6	100	86.0	949.0	277.4	2.32	543.8	141.6
Phosphate	mg/L	0.0005	6	100	0.08	0.24	0.12	1.53	0.17	0.08
Total phosphours	mg/L	0.002	6	100	0.13	0.33	0.21	1.40	0.28	0.16
Total ammonia	mg/L	0.002	6	100	0.010	0.700	0.099	3.90	0.293	0.033
Nitrite	mg/L	0.001	6	100	0.02	0.04	0.03	1.38	0.04	0.02
Nitrate	mg/L	0.006	6	100	0.4	1.1	0.7	1.42	0.9	0.5
TKN	mg/L	0.02	6	100	0.6	3.0	1.0	1.87	1.6	0.6
Sulphate	mg/L	0.25	2	100	10.0	20.0	14.1	1.63	27.9	7.2
Phenolics	mg/L	0.2	2	100	2.0	2.2	2.1	1.07	2.3	1.9
Solvent Extractable	mg/L	0.5	6	83	0.5	5.0	1.6	2.16	2.9	0.8
E.coli	c./100 ml	4	6	100	40.0	780.0	278.4	3.18	702.2	110.3
Copper	µg/L	1.6	6	100	4.8	14.1	7.5	1.48	10.3	5.5
Nickel	µg/L	1.3	6	67	0.7	1.6	1.1	1.54	1.6	0.8
Zinc	µg/L	0.6	6	100	20.5	74.3	33.9	1.56	48.4	23.7
Cadmium	µg/L	0.6	6	33	0.3	2.01	0.48	2.21	0.91	0.25
Cobalt	µg/L	1.3	6	17	0.7	1.3	0.7	1.34	0.9	0.6
Chromium	µg/L	1.4	6	83	0.7	7.8	2.6	2.18	4.8	1.4
Lead	µg/L	10	6	0	5.0	5.0	5.0	1.00	5.0	5.0
Iron	µg/L	0.8	6	100	312.0	707.0	487.1	1.41	641.7	369.8
Manganese	µg/L	0.2	6	100	26.1	104.0	48.3	1.62	71.2	32.8
Aluminum	µg/L	11	6	100	292.0	556.0	393.5	1.28	479.1	323.2
Vanadium	µg/L	1.5	6	50	0.8	2.4	1.3	1.61	1.9	0.9
Molybdenum	µg/L	1.6	6	0	all data less than the detection limit					
Barium	µg/L	0.2	6	100	15.8	28.7	20.8	1.26	25.0	17.2
Beryllium	µg/L	0.02	6	67	0.01	0.07	0.03	1.97	0.05	0.02
Strontium	µg/L	0.1	6	100	116.0	298.0	196.9	1.39	255.9	151.5
Titanium	µg/L	0.9	6	100	4.22	7.95	5.89	1.24	7.01	4.95
Calcium	mg/L	0.005	4	100	36.3	56.7	47.5	1.23	58.0	38.8
Magnesium	mg/L	0.008	4	100	2.7	5.1	3.5	1.36	4.8	2.6

Note: The mean, standard deviation and 95% confidence intervals are based on log normally distributed data.

* RMDL = Reporting Method Detection Limit

Table E10: Summary of effluent water quality statistics during the winter (Dec 21 to March 20) in 1997 and 1998)

Parameter	Unit	RMDL*	N	%>D.L.	MIN	MAX	MEAN	SD	95%CI-LL	95%CI-UL
Dissolved oxygen	mg/L	1	2	100	8.9	9.4	9.1	1.04	9.6	8.6
Biological Oxygen Demand	mg/L	0.2	6	100	0.8	2.7	1.6	1.56	2.3	1.2
Chemical Oxygen Demand	mg/L	1	6	100	15.3	42.2	29.3	1.44	39.2	21.9
pH	none	n/a	6	100	7.5	8.1	7.9	1.02	8.0	7.7
Conductivity	µS/cm	1	6	100	520.3	4770.7	1612.0	2.52	3372.1	770.6
Turbidity	FTU	0.01	6	100	20.9	54.1	37.7	1.50	52.2	27.2
Suspended solids	mg/L	2.5	6	100	16.4	39.9	28.7	1.38	37.1	22.3
Dissolved solids	mg/L	10	6	100	333.8	2637.7	925.1	2.34	1827.8	468.2
Total solids	mg/L	10	6	100	363.0	2672.6	961.1	2.28	1860.4	496.5
Chloride	mg/L	0.2	6	100	104.9	1566.7	429.4	2.98	1027.8	179.4
Phosphate	mg/L	0.0005	6	100	0.03	0.21	0.07	2.06	0.12	0.04
Total phosphours	mg/L	0.002	6	100	0.09	0.38	0.16	1.62	0.24	0.11
Total ammonia	mg/L	0.002	6	100	0.003	0.322	0.049	5.25	0.184	0.013
Nitrite	mg/L	0.001	6	100	0.02	0.06	0.03	1.71	0.05	0.02
Nitrate	mg/L	0.006	6	100	0.4	0.9	0.6	1.28	0.8	0.5
TKN	mg/L	0.02	6	100	0.5	1.6	0.9	1.50	1.2	0.6
Sulphate	mg/L	0.25	2	100	11.1	13.9	12.4	1.17	15.5	9.9
Phenolics	mg/L	0.2	2	100	1.2	1.7	1.5	1.27	2.0	1.1
Solvent Extractable	mg/L	0.5	6	100	0.7	2.8	1.5	1.69	2.2	1.0
E.coli	c./100 ml	4	6	100	57.4	616.1	252.0	2.32	494.9	128.3
Copper	µg/L	1.6	6	100	4.0	9.5	5.8	1.37	7.5	4.5
Nickel	µg/L	1.3	6	33	0.7	2.1	1.0	1.65	1.5	0.7
Zinc	µg/L	0.6	6	100	11.3	192.7	63.9	2.86	148.0	27.5
Cadmium	µg/L	0.6	6	0	all data less than the detection limit					
Cobalt	µg/L	1.3	6	0	all data less than the detection limit					
Chromium	µg/L	1.4	6	100	1.7	7.2	2.5	1.70	3.8	1.6
Lead	µg/L	10	6	0	all data less than the detection limit					
Iron	µg/L	0.8	6	100	388.0	574.7	469.9	1.17	531.0	415.8
Manganese	µg/L	0.2	6	100	40.9	72.8	60.3	1.23	71.3	51.0
Aluminum	µg/L	11	6	100	310.0	480.2	386.4	1.23	455.2	327.9
Vanadium	µg/L	1.5	6	50	0.8	2.0	1.1	1.60	1.7	0.8
Molybdenum	µg/L	1.6	6	0	all data less than the detection limit					
Barium	µg/L	0.2	6	100	16.2	48.7	27.4	1.57	39.4	19.1
Beryllium	µg/L	0.02	6	33	0.01	0.03	0.01	1.61	0.02	0.01
Strontium	µg/L	0.1	6	100	154.6	381.3	251.3	1.52	351.8	179.5
Titanium	µg/L	0.9	6	100	4.47	14.76	6.40	1.57	9.21	4.45
Calcium	mg/L	0.005	4	100	43.3	74.3	59.0	1.28	75.1	46.4
Magnesium	mg/L	0.008	4	100	2.3	5.6	4.2	1.51	6.3	2.8

Note: The mean, standard deviation and 95% confidence intervals are based on log normally distributed data.

* RMDL = Reporting Method Detection Limit

APPENDIX F

Removal Efficiencies

Table F-1: Load-based removal efficiencies (%) for the entire study period

Date/ Parameter	15-Jul-96	19-Jul-96	2-Aug-96	13-Sep-96	18-Oct-96	3-Nov-96	1-Dec-96	17-Dec-96	5-Jan-97	18-Feb-97	25-Mar-97	3-May-97	15-May-97	9-Jul-97	18-Jul-97	15-Aug-97	21-Aug-97	17-Sep-97	27-Oct-97	5-Nov-97	4-Dec-97	5-Jan-98	17-Feb-98	8-Mar-98	18-Mar-98	16-Apr-98	11-May-98	5-Jun-98	11-Jun-98	Overall Removal Efficiencies
Water Chemistry																													
DO	40	43	36	41	39	44	33	10	19	23	17	11	32																27
BOD	12	34	28	69	-50	77	52	30	6	-37	55	25	63	-35	66	20	23	-59	40	56	-7	23	53	17	-5	48	79	42	10	32
COD	27	51	59	-14	-9	79	30	12	31	4	53	19	52	-10	69	46	8	-17	35	78	-179	-68	27	18	-44	61	86	47	30	33
Oil & Grease	-2	41	49			100	79	51	34	9	63	88	85	42	83	70	67	37	89	87	-70	4	48	-42	-10	87	91	-12	56	61
Conductivity	-21	-15	28	10	14	-2	-24	-40	-23	66	80	-97	-3	-6	-17	-60	-50	-40	76	-120	-98	-390	-14	-14	-166	-51	74	30	3	4
Turbidity	74	91	90	86	39	86	90	42	45	30	91	72	84	74	95	75	95	93	81	89	17	16	62	61	-43	87	94	81	87	79
TSS	56	96	93	91	40	92	94	59	52	26	92	88	88	49	96	79	95	94	86	94	53	14	74	53	-60	91	95	84	71	86
DS	-9	-18	18	10	14	-3	-23	-35	-48	57	75	-100	-2	-5	-15	-60	-48	-40	-573	-100	-133	-375	-16	-14	-119	-52	74	29	4	-6
TSS	21	83	61	75	19	80	65	-6	-28	56	78	38	32	-1	80	66	48	84	-125	2	-112	-345	-6	-10	-116	42	80	44	10	26
Chloride	-31	-93	-7	33	7	-23	-57	-50	-22	69	87	-185	-51	-75	-132	-290	-217	-305	87	-370	-122	-526	-22	-40	-129	-83	84	80	20	-1
Sulphate	-25	68	34	12	27	-5	11	-75	-39	45	43	-70	-10	26	4	-52	-143													21
Phenolics	-95	37	66	33	41	58	73	57	44	14	56	26	64	51	25	-7	14													26
E. coli	74	96	87			80	44	-8	21	64	61	45	93	97			89	91	-656		-91	17	77	-544	-10	20	69	98	97	84
Nutrients																													
Phosphate	99	88	97	38	19	72	42	20	15	14	-22	-46	70	70	70	55	74	71	67	-80	66	53	53	72	45	-173	87	88	73	44
TP	69	91	87	81	13	92	85	30	25	-16	50	19	72	32	89	-90	80	63	52	68	39	43	43	52	3	57	91	73	56	58
Amm. -N	100	-8	16	-28	-2	79	13	74	32	54	-10	-10	87	75	72	-52	73	64	79	56	22	-48	59	-14	95	48	99	80	31	63
Nitrite-N	92	-41	87	68	47	63	16	-200	17	-58	51	-53	47	48	83	26	50	-1159	33	48	-21	-28	4	47	-12	65	73	60	86	22
Nitrate-N	92	60	99	80	40	60	29	-40	-1	43	21	-120	62	100	80	65	60	90	-22	-76	21	-1	-37	52	-23	-55	82	94	97	41
TKN	59	74	62	66	18	87	68	26	-3	45	34	-13	62	23	77	12	49	13	24	62	14	0	30	44	-53	28	94	62	42	49
Metals																													
Copper	51	26	83	67	34	79	13	31	29	49	65	78	43	40	89	60	92	-23	79	82	8	16	38	37	-28	66	74	-66	46	58
Nickel	32	18	74	87	19	66	18	47	60	0	83	30	75	63	83	14	70	-63	77	64	-7	6	-18	66	-129	57	82	44	50	48
Zinc	-52	-285	81	84	55	71	41	31	45	-13	64	50	44	-51	86	-28	66	-1033	53	83	-138	-404	-49	-121	-193	49	86	-23	51	17
Cadmium	90	65	88	91	91	78	68	4	0	0	23	73	32	91	91	76	87	30	80	90	66	85	63	31	-6	79	90	-158	15	77
Cobalt	75	91	40	82	19	89	69	4	0	0	71	13	32	65	68	14	69	10	20	26	-7	-2	55	31	-6	69	12	53	15	58
Chromium	37	20	86	68	36	67	-24	11	40	25	71	34	32	15	84	14	79	10	28	45	65	15	15	34	-152	68	-18	53	15	36
Lead	30	30	40	29	19	30	32	4	0	0	23	13	32	15	62	14	23	10	20	67	-161	-2	8	31	-6	68	12	53	15	22
Iron	-5	22	78	67	-11	53	29	28	24	-24	80	51	65	35	76	54	80	-886	66	78	-9	22	36	15	-39	55	73	67	55	41
Manganese	-5	86	60	83	-65	83	72	42	26	-6	84	75	76	-37	82	39	71	-5507	80	66	4	-44	47	-77	-143	65	73	-104	71	56
Aluminum	9	25	78	73	-5	65	-677	20	20	2	73	45	62	63	77	56	77	-65	62	76	-8	24	8	4	-17	53	64	84	72	43
Vanadium	60	6	76	64	21	53	-155	24	18	0	83	33	-970	24	71	-3	64	10	73	73	-7	-2	11	24	49	52	78	86	69	25
Barium	21	53	55	67	11	67	45	-12	-10	23	64	28	39	17	70	11	27	-49	52	24	-35	-74	4	6	-177	19	76	48	13	29
Berylium	30	30	40	77	12	75	5	24	35	-33	81	60	52	71	93	62	80	10	70	85	56	86	72	72	-6	72	81	79	15	59
Strontium	-41	50	22	43	28	57	49	-51	-33	16	56	11	0	25	57	-1	-6	-64	-31	-67	1	-31	1	13	-133	12	76	33	5	14
Titanium	-533	88	21	-48	-37	-549	-627	-5	12	38	44	-61	-43	62	14	-11	55	-291	12	30	3	14	-11	-51	39	-61	-13	66	79	13
inlet vm (m^3)	5720	7479	4955	7462	12275	3058	3818	7565	5619	18246	5578	7443	3767	1190	1422	1849	3084	2370	3006	3144	352	4710	7986	4206	2779	3349	2512	1228	1422	
outlet vm (m^3)	4019	5243	2985	5310	9974	2140	2585	7287	5619	18246	4269	6486	2557	1005	1317	1594	2388	2144	2412	2312	377	4815	7338	2884	2943	2753	2206	571	1202	
flow bal. (%)*	30	30	40	29	19	30	32	4	0	0	23	13	32	15	7	14	23	10	20	26	-7	-2	8	31	-6	18	12	53	15	16

Note: Equations used to calculate single and multiple event removal efficiencies are provided in chapter 3.

* Groundwater recharge accounts for most of the difference between inlet and outlet runoff volumes (see chapter 4 for overall and event based estimates of the facility water balance).

Table F-2: Load-based removal efficiencies (%) for rainfall events during the spring

Date/ Parameter	25-Mar-97	3-May-97	15-May-97	16-Apr-98	11-May-98	5-Jun-98	11-Jun-98	Total Efficiency
General chemistry								
DO	17	11	32					18
BOD	55	25	63	48	79	42	10	55
COD	53	19	52	61	86	47	30	51
Oil and Grease	63	88	85	87	91	-12	56	82
pH	25	15	33	20	8	53	11	22
Conductivity	80	-97	-3	-51	74	30	3	51
Turbidity	91	72	84	87	94	81	87	82
TSS	92	88	88	91	95	84	71	90
Dissolved Solids	75	-100	-2	-52	74	29	4	44
Total Solids	78	38	32	42	80	44	10	62
Chloride	87	-185	-51	-83	84	80	20	69
Sulphate	43	-70	-10					-1
Phenolics	56	26	64					48
E. coli	61	45	93	20	69	98	97	68
Nutrients								
Phosphate	-22	-46	70	-173	87	88	73	14
TP	50	19	72	57	91	73	56	56
Ammonium-N	-10	-10	87	48	99	80	31	79
Nitrite-N	51	-53	47	65	73	60	86	52
Nitrate-N	21	-120	62	-55	82	94	97	26
TKN	34	-13	62	28	94	62	42	52
Metals								
Copper	65	78	43	66	74	-66	46	66
Nickel	83	30	75	57	82	44	50	64
Zinc	64	50	44	49	86	-23	51	59
Cadmium	23	73	32	79	90	-158	15	70
Cobalt	71	13	32	69	12	53	15	50
Chromium	71	34	32	68	-18	53	15	50
Lead	23	13	32	68	12	53	15	34
Iron	80	51	65	55	73	67	55	65
Manganese	84	75	76	65	73	-104	71	71
Aluminum	73	45	62	53	64	84	72	59
Vanadium	83	33	-970	52	78	86	69	-20
Barium	64	28	39	19	76	48	13	47
Berylium	81	60	52	72	81	79	15	68
Strontium	56	11	0	12	76	33	5	38
Titanium	44	-61	-43	-61	-13	66	79	16
inlet volume (m^3)	5578	7443	3767	3349	2512	1228	1422	
outlet volume (m^3)	4269	6486	2557	2753	2206	571	1202	
flow balance (%)*	23	13	32	18	12	53	15	21

Note: Equations used to calculate single and multiple event removal efficiencies are provided in chapter 3.

* Groundwater recharge accounts for most of the difference between inlet and outlet runoff volumes (see chapter 4 for overall and event based estimates of the facility water balance).

Table F-3: Load-based removal efficiencies (%) for rain events during the summer

Date/ Parameter	15-Jul-96	19-Jul-96	2-Aug-96	13-Sep-96	9-Jul-97	18-Jul-97	15-Aug-97	21-Aug-97	17-Sep-97	Total Efficiency
Water Chemistry										
DO	40	43	36	41						40
BOD	12	34	28	69	-35	66	20	23	-59	34
COD	27	51	59	-14	-10	69	46	8	-17	36
Oil and Grease	-2	41	49		42	83	70	67	37	44
pH	33	31	41	30	17	9	17	26	12	29
Conductivity	-21	-15	28	10	-6	-17	-60	-50	-40	-8
Turbidity	74	91	90	86	74	95	75	95	93	88
TSS	56	96	93	91	49	96	79	95	94	91
Dissolved Solids	-9	-18	18	10	-5	-15	-60	-48	-40	-8
Total Solids	21	83	61	75	-1	80	66	48	84	71
Chloride	-31	-93	-7	33	-75	-132	-290	-217	-305	-51
Sulphate	-25	68	34	12	26	4	-52	-143		35
Phenolics	-95	37	66	33	51	25	-7	14		-6
E. coli	74	96	87		97			89	91	94
Nutrients										
Phosphate	99	88	97	38	70	70	55	74	71	86
TP	69	91	87	81	32	89	-90	80	63	83
Ammonium-N	100	-8	16	-28	75	72	-52	73	64	52
Nitrite-N	92	-41	87	68	48	83	26	50	-1159	31
Nitrate-N	92	60	99	80	100	80	65	60	90	82
TKN	59	74	62	66	23	77	12	49	13	63
Metals										
Copper	51	26	83	67	40	89	60	92	-23	69
Nickel	32	18	74	87	63	83	14	70	-63	56
Zinc	-52	-285	81	84	-51	86	-28	66	-1033	53
Cadmium	90	65	88	91	91	91	76	87	30	86
Cobalt	75	91	40	82	65	68	14	69	10	80
Chromium	37	20	86	68	15	84	14	79	10	58
Lead	30	30	40	29	15	62	14	23	10	31
Iron	-5	22	78	67	35	76	54	80	-886	52
Manganese	-5	86	60	83	-37	82	39	71	-5507	70
Aluminum	9	25	78	73	63	77	56	77	-65	58
Vanadium	60	6	76	64	24	71	-3	64	10	53
Barium	21	53	55	67	17	70	11	27	-49	51
Berylium	30	30	40	77	71	93	62	80	10	74
Strontium	-41	50	22	43	25	57	-1	-6	-64	27
Titanium	-533	88	21	-48	62	14	-11	55	-291	34
inlet volume (m^3)	5720	7479	4955	7462	1190	1422	1849	3084	2370	
outlet volume (m^3)	4019	5243	2985	5310	1005	1317	1594	2388	2144	
flow balance (%)*	30	30	40	29	15	7	14	23	10	27

Note: Equations used to calculate single event and multiple event removal efficiencies are provided in chapter 3.

* Groundwater recharge accounts for most of the difference between inlet and outlet runoff volumes (see chapter 4 for overall and event based estimates

Table F-4: Load-based removal efficiencies (%) rain events during the fall

Date/ Parameter	18-Oct-96	3-Nov-96	1-Dec-96	17-Dec-96	27-Oct-97	5-Nov-97	4-Dec-97	Total Efficiency
General Chemistry								
DO	39	44	33	10				31
BOD	-50	77	52	30	40	56	-7	22
COD	-9	79	30	12	35	78	-179	39
Oil and Grease		100	79	51	89	87	-70	78
pH	20	30	33	5	22	28	-5	20
Conductivity	14	-2	-24	-40	76	-120	-98	13
Turbidity	39	86	90	42	81	89	17	79
TSS	40	92	94	59	86	94	53	87
Dissolved Solids	14	-3	-23	-35	-573	-100	-133	-29
Total Solids	19	80	65	-6	-125	2	-112	40
Chloride	7	-23	-57	-50	87	-370	-122	24
Sulphate	27	-5	11	-75				2
Phenolics	41	58	73	57				55
E. coli		80	44	-8	-656		-91	55
Nutrients								
Phosphate	19	72	42	20	67	-80	66	31
TP	13	92	85	30	52	68	39	64
Ammonium-N	-2	79	13	74	79	56	22	40
Nitrite-N	47	63	16	-200	33	48	-21	11
Nitrate-N	40	60	29	-40	-22	-76	21	11
TKN	18	87	68	26	24	62	14	51
Metals								
Copper	34	79	13	31	79	82	8	57
Nickel	19	66	18	47	77	64	-7	50
Zinc	55	71	41	31	53	83	-138	58
Cadmium	91	78	68	4	80	90	66	82
Cobalt	19	89	69	4	20	26	-7	53
Chromium	36	67	-24	11	28	45	65	32
Lead	19	30	32	4	20	67	-161	26
Iron	-11	53	29	28	66	78	-9	31
Manganese	-65	83	72	42	80	66	4	56
Aluminum	-5	65	-677	20	62	76	-8	26
Vanadium	21	53	-155	24	73	73	-7	30
Barium	11	67	45	-12	52	24	-35	30
Berylium	12	75	5	24	70	85	56	46
Strontium	28	57	49	-51	-31	-67	1	17
Titanium	-37	-549	-627	-5	12	30	3	-16
inlet volume (m^3)	12275	3058	3818	7565	3006	3144	352	
outlet volume (m^3)	9974	2140	2585	7287	2412	2312	377	
flow balance (%)*	19	30	32	4	20	26	-7	18

Note: Equations used to calculate single and multiple event removal efficiencies are provided in chapter 3.

* Groundwater recharge accounts for most of the difference between inlet and outlet runoff volumes (see chapter 4 for overal

Table F-5: Load-based removal efficiencies (%) for precipitation and snowmelt events during the winter

Date/ Parameter	5-Jan-97	18-Feb-97	5-Jan-98	17-Feb-98	8-Mar-98	18-Mar-98	Total Efficiency
General Chemistry							
DO	19	23					22
BOD	6	-37	23	53	17	-5	7
COD	31	4	-68	27	18	-44	6
Oil and Grease	34	9	4	48	-42	-10	25
pH	1	1	-1	14	31	10	6
Conductivity	-23	66	-390	-14	-14	-166	-7
Turbidity	45	30	16	62	61	-43	41
TSS	52	26	14	74	53	-60	46
Dissolved Solids	-48	57	-375	-16	-14	-119	-14
Total Solids	-28	56	-345	-6	-10	-116	-9
Chloride	-22	69	-526	-22	-40	-129	-15
Sulphate	-39	45					34
Phenolics	44	14					21
E. coli	21	64	17	77	-544	-10	42
Nutrients							
Phosphate	15	14	53	53	72	45	26
TP	25	-16	43	43	52	3	8
Ammonium-N	32	54	-48	59	-14	95	50
Nitrite-N	17	-58	-28	4	47	-12	-25
Nitrate-N	-1	43	-1	-37	52	-23	27
TKN	-3	45	0	30	44	-53	36
Metals							
Copper	29	49	16	38	37	-28	37
Nickel	60	0	6	-18	66	-129	10
Zinc	45	-13	-404	-49	-121	-193	-87
Cadmium	0	0	85	63	31	-6	49
Cobalt	0	0	-2	55	31	-6	19
Chromium	40	25	15	15	34	-152	20
Lead	0	0	-2	8	31	-6	4
Iron	24	-24	22	36	15	-39	6
Manganese	26	-6	-44	47	-77	-143	4
Aluminum	20	2	24	8	4	-17	8
Vanadium	18	0	-2	11	24	49	14
Barium	-10	23	-74	4	6	-177	-9
Berylium	35	-33	86	72	72	-6	40
Strontium	-33	16	-31	1	13	-133	-7
Titanium	12	38	14	-11	-51	39	44
inlet volume (m^3)	5619	18246	4710	7986	4206	2779	
outlet volume (m^3)	5619	18246	4815	7338	2884	2943	
flow balance (%)*	0	0	-2	8	31	-6	4

Note: Equations used to calculate single and multiple event removal efficiencies are provided in chapter 3.

* Groundwater recharge accounts for most of the difference between inlet and outlet runoff volumes (see chapter 4 for overall and event based estimates of the facility water balance).

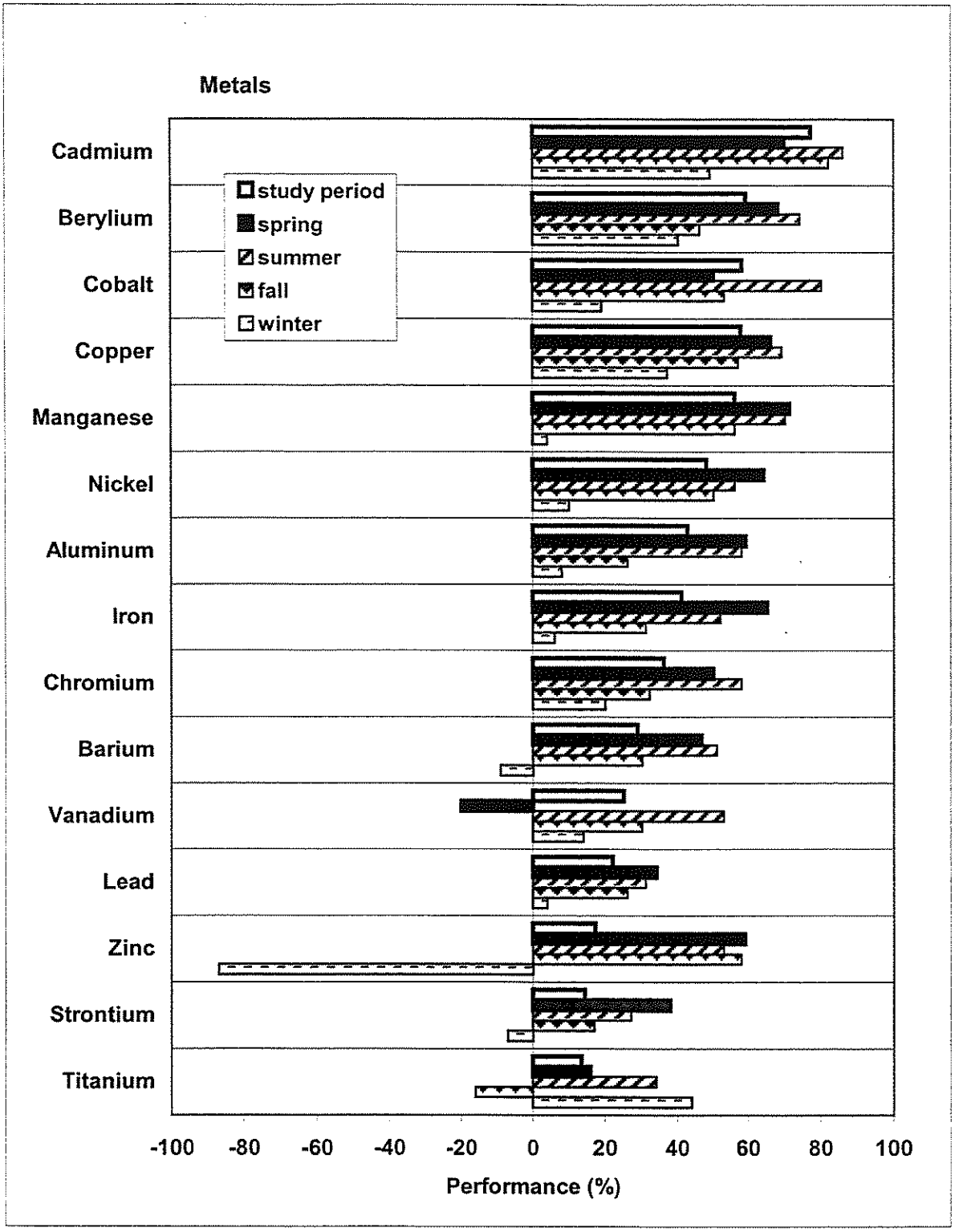


Figure F1: Load-based removal efficiencies for metals

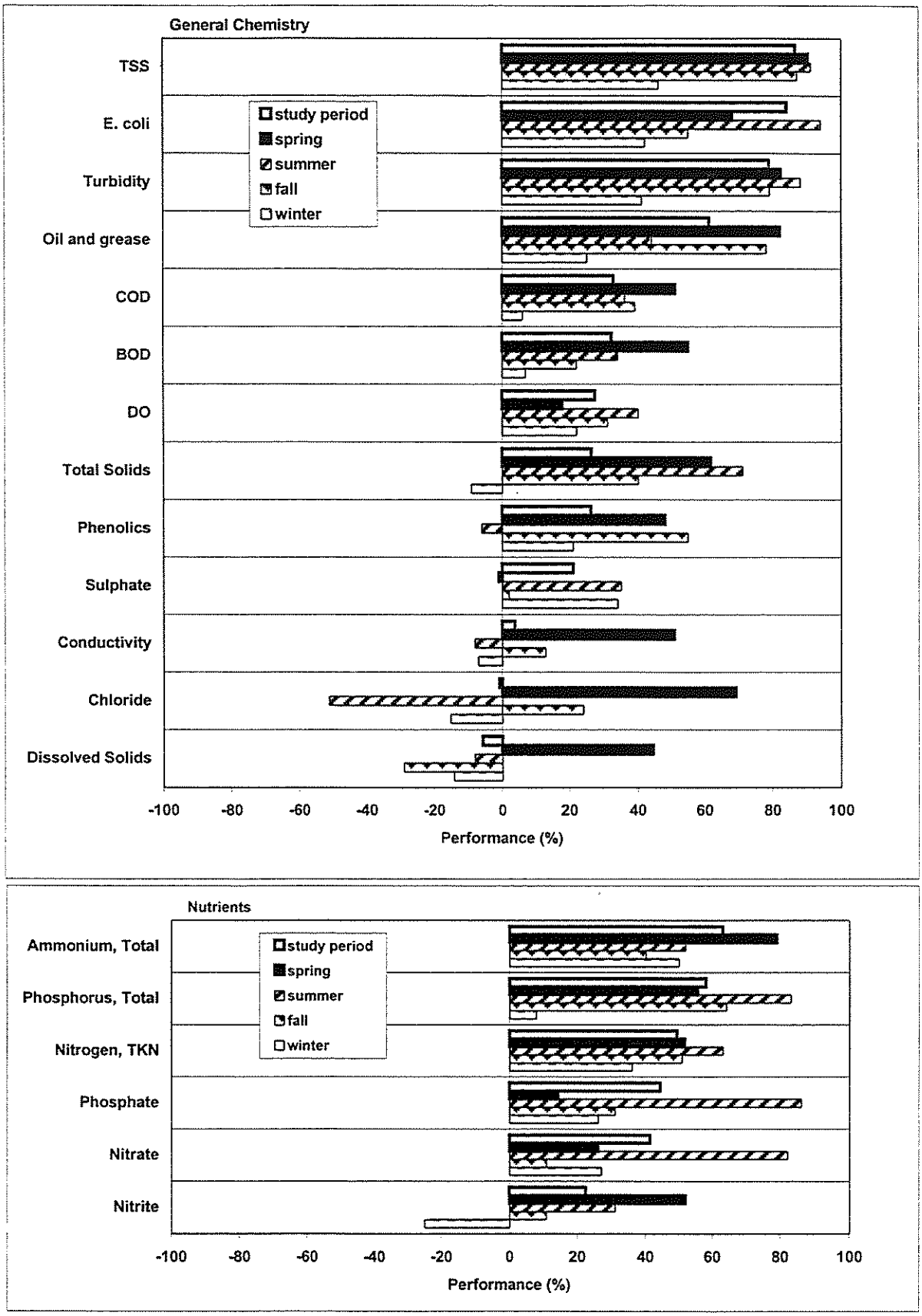


Figure F2: Load-based performance for nutrients and general chemistry constituents.

APPENDIX G

Particle Size Distributions for Individual Events

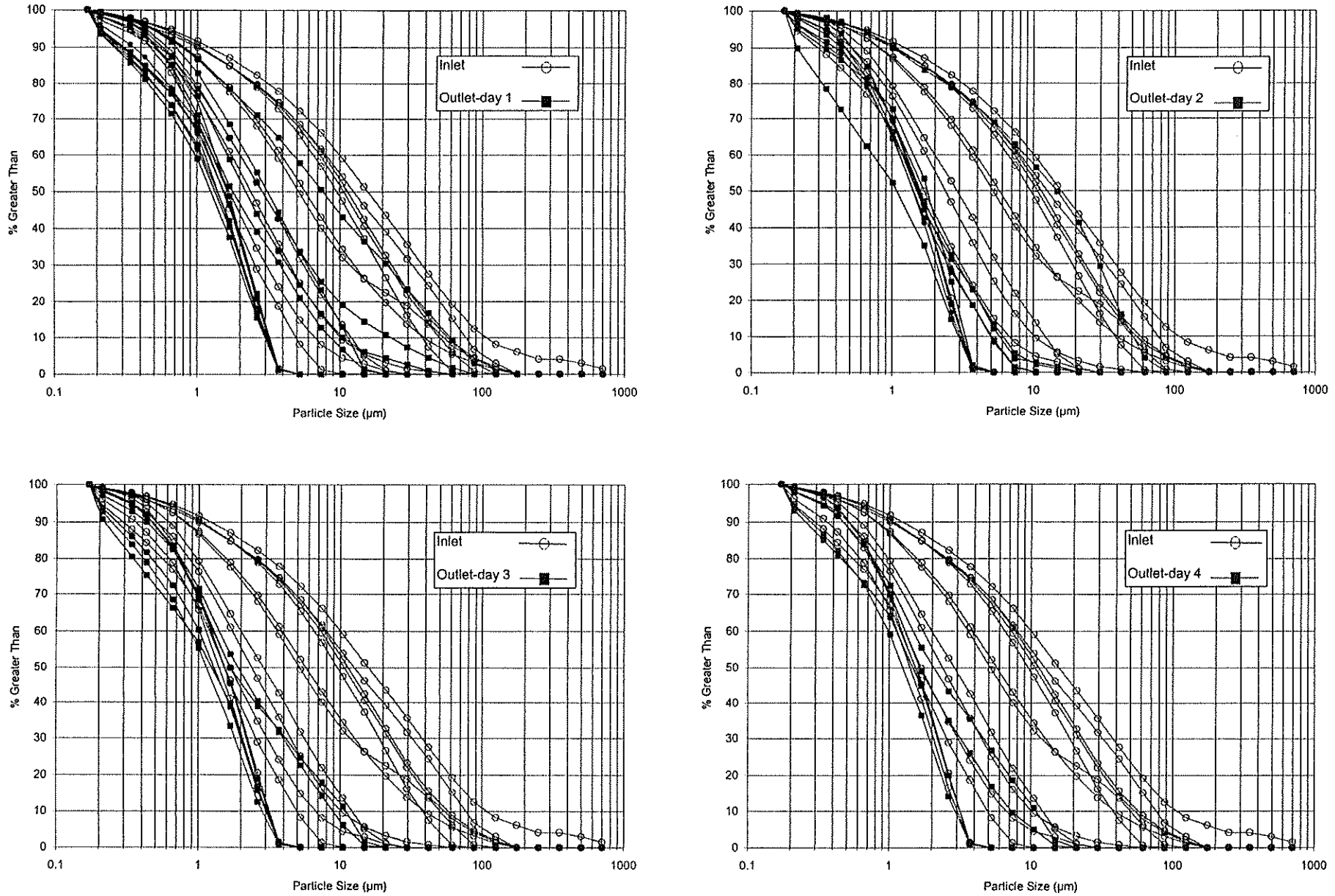


Figure G1: Comparison of inlet and outlet particle size distributions for each day of the 4-day drawdown period.

APPENDIX H

Laboratory Sediment Bioassay Report on Sediments from a Constructed Wetland in Aurora, Ontario, 1997

The following report was produced by Donna Bedard from the Ontario Ministry of the Environment (OMOE). The document has been reformatted but is otherwise reproduced as submitted by the OMOE.

LABORATORY SEDIMENT BIOASSAY REPORT

ON SEDIMENTS FROM A CONSTRUCTED WETLAND IN

AURORA, ONTARIO 1997

D. Bedard
Ontario Ministry of the Environment
Standards Development Branch
Etobicoke, Ontario

NOVEMBER 1998

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1.0 INTRODUCTION

Laboratory sediment toxicity tests are a useful tool for examining biological effects and chemical uptake and availability and are part of an integrated approach in evaluating sediment quality (Jaagumagi and Persaud, 1996). Sediment quality guidelines are chemical criteria developed for the protection of aquatic life in the province of Ontario (Persaud *et al.*, 1992). These guidelines provide a measure of potential sediment toxicity on a generic scale, while the laboratory biological tests are a direct assessment of contaminant toxicity that yield site-specific information.

Whole-sediment toxicity tests were conducted for five field locations in 1997, using the mayfly nymph, *Hexagenia limbata* (21-day exposure, survival and growth), the midge larvae, *Chironomus tentans* (10-day exposure, survival and growth) and the juvenile fathead minnow, *Pimephales promelas* (21-day exposure, survival) (Bedard *et al.*, 1992). The battery of sediment toxicity tests provides a number of endpoints using organisms representing different trophic levels in order to measure differences in sediment quality. The laboratory toxicity tests provide a cost-effective technique for determining if sediment-associated contaminants are harmful to benthic organisms or are being released into the water-column. In conjunction with appropriate negative and reference control sediments, spatial differences in sediment quality, the relative availability of contaminants and their potential impacts can be ascertained. Sediment contaminant concentrations were based on samples prepared for laboratory toxicity testing. The sediment was analysed for particle size, nutrients, metals, major ions, total PCBs, total PAHs, organochlorine pesticides and chlorinated benzenes.

2.0 MATERIALS AND METHODS

2.1 Sample Collection and Site Description

On August 12, 1997, surficial sediment was collected at four test locations within the Aurora stormwater facility. The sites were selected in order to study any changes in environmental conditions at various stages in the treatment of incoming stormwater (Smith, 1998). Station 1 was situated within the greenhouse, stn 2 at the outlet, stn 3 was located in the southern portion of the wetland and stn 4 within the forebay (Figure 1).

Sampling was done using a stainless steel scoop. At each station, approximately 10 L of composited surficial sediment (top 5 cm) was collected from several grabs. Some samples were reconstituted with site water depending on sediment moisture content at the time of collection. The composited sediment was placed into 20 L plastic buckets lined with food-grade polyethylene bags and transported to the Toronto, Ontario laboratory where they were stored at 4°C until required.

A reference control sediment (stn 5) was sampled in a relatively, low-level contaminated area from a nearby creek that was located upstream of the facility. The reference control

sediment should be representative of naturally occurring background contaminant levels for the study area and be physically similar to the test sediment to discriminate effects due to physical or chemical causes. Sediment collected from Honey Harbour, Georgian Bay, Ontario served as a negative control. The negative control sediment is a relatively uncontaminated sediment that provides a measure of test acceptability (ASTM, 1997). Both control sediments are a basis for comparing the biological responses from the test sediments.

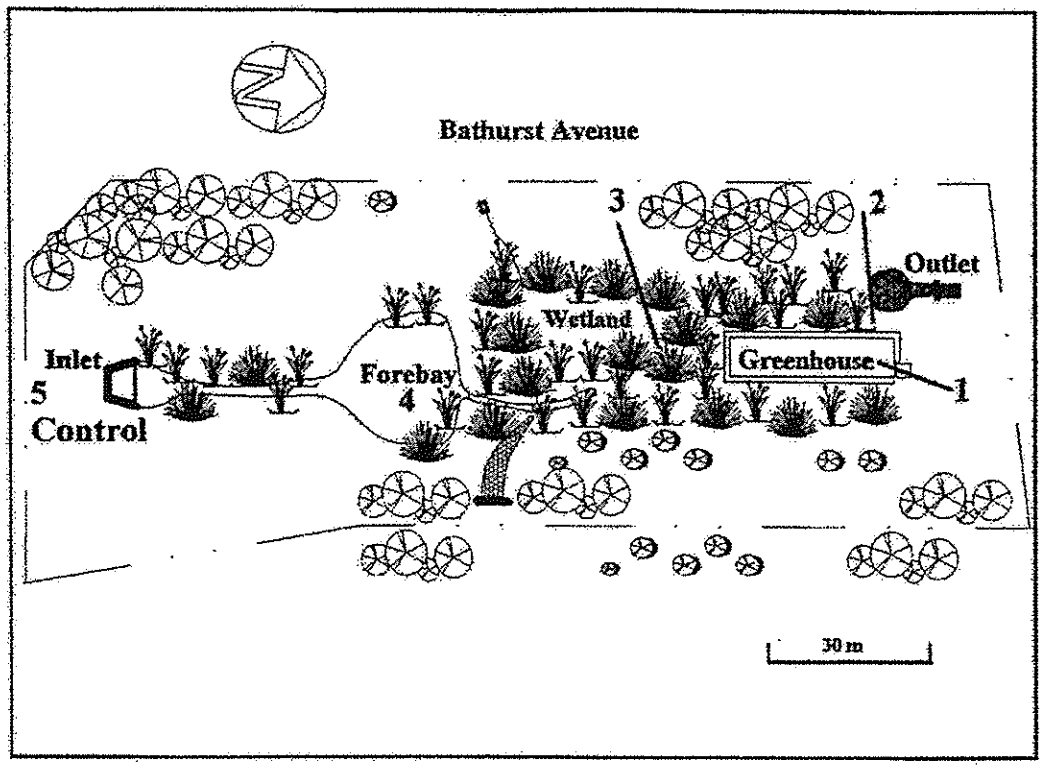

Figure 1: Sediment sampling locations

2.2 Analytical Methods

Chemical analysis of sediment and biota samples was carried out by the OMOE, Laboratory Services Branch, located in Toronto. Test methods are described in the *OMOE Handbook of Analytical Methods for Environmental Samples* (OMOE, 1983). Quality assurance procedures included method blanks, spikes, duplicates and standard reference materials.

Sediment Nutrients and Particle Size Characterization

Homogenized bulk sediment (<2 mm fraction) was measured for total phosphorus (TP), total Kjeldahl nitrogen (TKN) and percent weight loss on ignition (LOI) which measured approximate organic content. Sediment total organic carbon (TOC) was determined with a LECO carbon

analyzer using a dry combustion technique which oxidized carbon to CO_2. Particle size was measured on 50 g, air-dried samples using a Microtrac particle size analyzer for the size range 1.0 mm to 0.1 µm. This was to provide data for sand (2mm -62 µm), silt (62- 3.7 µm) and clay (3.7 - 0.1 µm) size classes. Detailed test methodology is described in OMOE laboratory procedure documents (OMOEE, 1995a; 1995b).

Trace Metals in Sediment

Prepared sediment samples were digested using a concentrated aqua-regia acid mixture (1 part HNO_3 to 3 parts HCl). The dissolved trace metals including As, Cd, Cr, Cu, Fe, Pb, Mn, Ni and Zn in the digestates were detected by inductively coupled argon plasma atomic emission spectroscopy (ICP-AES), and Hg by flow injection vapour generated flameless atomic absorption spectroscopy (AAS). Detailed test methodology is described in OMOEE (1994a).

Organic Chemicals in Sediment

Moist sediment samples were solvent-extracted with acetone and dichloromethane. The extract was transferred to a rotary evaporator, concentrated and fractionated on a Florisil column. Different solvent combinations were used to elute the extracts into three groups: fraction A1 contained total PCBs, five Aroclor groups, hexachlorobenzene, heptachlor, aldrin, octachlorostyrene, pp-DDE and mirex; fraction A2 contained a- & b-BHC, a- & b-chlordane, op-DDT, pp-DDD, pp-DDT; and fraction A3 included heptachlor epoxide, oxychlordane, dieldrin, endosulfan I & II, endosulfan sulphate, endrin and methoxyclor. Analytes were identified and quantified using capillary gas chromatography equipped with a Ni^{63} electron capture detector (GLC-ECD). Detailed test methodology is described in OMOEE (1994b; 1994c).

Organic Chemicals and Percent Lipid in Biota

Pooled whole fish samples (~5 g) were thawed, homogenized and acid digested using concentrated hydrochloric acid (H Cl) on duplicate samples per station. The digestate was reacted with a mixture of 25% dichloromethane in hexane. The extract was treated with sodium bicarbonate to ensure neutralization and dried with anhydrous sodium sulphate. Dichloromethane-cyclohexane was added to the evaporated samples, followed by clean-up and detected by capillary gas chromatography equipped with a mass selective detector. Final results are reported on a wet weight basis for 14 chlorinated organic compounds and 14 pesticides. Detailed methodology is described in OMOE (1990). The samples were also analyzed for 16 individual PAH compounds following standard test methods (OMOE, 1995c). Percent lipid was determined on an aliquot (25 mL) of the final extract obtained before clean-up. The solvent was evaporated by air-drying in a fumehood for 24 hours and lipid residues were measured.

2.3 Laboratory Biological Testing Methods

Basic Experimental Design

Sediment biological tests were conducted according to OMOE standardized procedures (Bedard *et al.*, 1992) and are briefly described below. The bioassays were static, single-species tests using whole-sediment. The experimental unit was a 1.8 L test chamber containing prepared sediment and dechlorinated municipal tap water (1:4, v:v). The chambers were randomly placed into a holding tank at ambient room temperature and maintained under a 16:8 hour, light:dark photoperiod and continuous aeration.

Moist field-collected bottom sediment was pressed through a 2-mm stainless-steel sieve to remove existing large biota and debris prior to use. Sieving was completed on August 15th and 18th, 1997. Subsamples of this homogenized sediment were submitted for chemical and physical characterization according to standard OMOE procedures (OMOE, 1989). The sieved sediment was homogenized with a spatula and stored in 4 L acid-rinsed glass jars until required. Three hundred and twenty-five millilitre aliquots of homogenized sediment were placed into the test chamber and overlaid with the test water. After settling overnight, the chambers were aerated continuously until the end of the test. A clean, negative control sediment was collected from Honey Harbour, Georgian Bay. Negative control mortality must not exceed 15% for mayflies and fathead minnows and 25% for chironomids or the test is declared invalid.

Water in the exposure chambers was regularly monitored for pH, conductivity, total ammonia, un-ionized ammonia and dissolved oxygen. Dead organisms were removed and the numbers recorded daily. Any signs of abnormal behaviour of the test organisms or changes in appearance of the test chambers were noted. Water loss due to evaporation was replenished as needed.

Hexagenia limbata Lethality and Growth Assay

The toxicity test used four month old laboratory-reared mayfly nymphs with an average wet weight of 4.45 mg ± 0.36 (s.e.) (n=28). The nymphs were raised from eggs collected by Dr. J.J.H. Ciborowski at the University of Windsor, Windsor, Ontario. Mayflies were reared according to OMOE procedures (Bedard *et al.*, 1992) and methods described in the literature (Friesen, 1981).

The rearing procedure involved the transfer of 600 newly-hatched nymphs to a 6.5 L aquarium which contained 2 cm of autoclaved sediment and 5.6 L dechlorinated tap water. Animals were maintained at ambient room temperature, 16:8 hour, light:dark photoperiod, constant aeration and fed a vegetable diet.

Test organisms were retrieved from the rearing aquaria by sieving small portions of sediment in a 500-µm mesh brass sieve. The nymphs were washed into an enamel tray which held a fine mesh sieve and aerated, dechlorinated water. A Pasteur pipette (5-mm opening) was used to transfer the mayflies into 100 mL beakers of water and the contents were gently poured into the test chambers. Ten nymphs were added to each of the three replicate test chambers for

a period of 21 days. Animals were not fed during the length of the test.

At the end of the test, the contents of each test chamber were emptied and rinsed in a sieve bucket. Surviving animals were counted and transferred to 150 mL beakers holding 100 mL dechlorinated water. The nymphs were immobilized with Alka-Seltzer®, blotted dry and individuals weighed to the nearest 0.01 mg.

Chironomus tentans Lethality and Growth Assay

The toxicity test used 10-12 day old, cultured chironomid larvae weighing an average wet weight less than 1 mg. The OMOE maintains continuous cultures *C. tentans* larvae from egg to adult following standard methods (Bedard *et al.*, 1992, Mosher *et al.*, 1982, Townsend *et al.*, 1981). Egg masses were originally supplied by Dr. J. Giesy at Michigan State University, Lansing, Michigan and have been cultured for several generations in our laboratory.

Initially, the midges were reared in enamel trays for 10 to 12 days and then maintained in a 21 L aquarium containing 1.6 L of silica sand. The cultures were held at ambient room temperature with continuous aeration and under a 16:8 hour, light:dark photoperiod. The larvae were provided a vegetable diet *ad libitum*.

Second and third instar larvae were directly transferred from the enamel rearing pans into the test chamber using the 5-mm opening of a Pasteur pipette. A total of 15 animals were added per chamber to each of the three replicates. Animals were fed daily 26 mg of a Cerophyll®:Tetra Conditioning Vegetable® (3:2, w:w) diet.

After 10 days, the contents of the test chambers were emptied and washed in a sieve bucket. Surviving animals were sorted, removed and placed into 150 mL beakers holding 100 mL dechlorinated water and 15 mL silica sand. The larvae were counted, blotted dry and individuals weighed to the nearest 0.01 mg.

Pimephales promelas Lethality and Bioaccumulation Assay

The tests used cultured, juvenile fathead minnows with an average wet weight of 384 mg ± 19 (s.e.) (n=31). The minnows were cultured at the OMOE laboratory following techniques which for the most part are US EPA procedures (USEPA, 1987) with minor revisions (Bedard *et al.*, 1992).

Cultures were maintained at 20°C in a flow-through dechlorinated water system and under a 16:8 hour, light:dark photoperiod. Breeders were kept in 60 L glass aquaria and eggs were laid on spawning tiles. The tiles were incubated in a 25°C water-bath and the developing larvae were transferred to 400 L fibreglass holding tanks. Larval fish were fed 48-hour old live brine shrimp while juveniles and breeders were provided frozen brine shrimp. Each size class was fed *ad libitum*.

Each test chamber received 10 juvenile minnows in triplicate per sample. The minnows

were sorted into 250 mL glass beakers in groups of five. The contents of the beakers were emptied into a small net and the minnows released into the test chamber.

The minnows were exposed for 21 days and fed a NutraFin Staple® diet daily in an amount equivalent to 1% of the average starting wet weight. After 21 days the surviving fathead minnows were pooled from each replicate, counted, immobilized with Alka-Seltzer® and placed into 30 mL glass vials and frozen.

Reference Toxicant Testing

A water-only reference toxicity ($CuSO_4$) test was conducted with *H. limbata* and *C. tentans* for 48-hours and LC50s were calculated. The static tests consisted of four test concentrations and a control. The nominal copper concentrations were 0.05, 0.25, 0.5, 1.0 and 3.0 mg/L. Ten mayfly nymphs or midge larvae were placed into each of four replicate 250 mL beakers. To help reduce stress, five glass tubes were placed into the mayfly test beakers and a fine layer of silica sand was added to the midge test containers. Water quality parameters were recorded at 0 and 48 hours. The mayfly test used four month old laboratory-reared mayfly nymphs with an average wet weight of 4.9 mg ± 0.4 (s.e.). The midge larvae were 10-12 day post-hatch with an average wet weight < 1 mg.

Bioassay Schedule for Aurora Wetland 1997 Sediment Samples

Test Organism	Species	Starting Date ('97)	Completion Date ('97)	Test Duration
Mayfly	*Hexagenia limbata*	August 21	September 11	21 days
Chironomid	*Chironomus tentans*	August 29	September 8	10 days
Minnow	*Pimephales promelas*	August 21	September 11	21 days

2.4 Statistical Methods

Statistical analyses were performed using the SAS® software package (SAS, 1985). Comparisons were made among the test and control sediments using One-Way Analysis of Variance (ANOVA) and Tukey's studentized range test (HSD) and planned comparisons (Steel and Torrie, 1960). Dunnett's one-tailed *t*-test was used solely to compare mortality between the control and test sediments and the associated minimum significant difference (MSD) was described as a measure of test sensitivity. Analysis was made on arc-sine transformed mortality data. Homogeneity of variance across groups was tested using Bartlett's test. Coefficients of variation (C.V. %) were calculated for each endpoint as a measure of test precision. LC50's (including the associated 95% confidence limits) were calculated using software developed by Stephan (1977) and were estimated by probit analysis.

3.0 RESULTS

3.1 Water Quality Test Parameters

Conductivity, pH, total ammonia, un-ionized ammonia and dissolved oxygen parameters were periodically measured on the overlying water for each test species and summarized in Table A. Values are reported as mean ± standard deviation.

Similar pH water quality measurements were recorded among the test sites, regardless of test species, and the average pH ranged from 7.8 to 8.2. In comparison, the overall average pH for the reference sediment was 8.2. Conductivity readings averaged 563 µmho/cm, 537 µmho/cm and 598 µmho/cm in the mayfly, midge and minnow tests, respectively and agreed favourably with the conductivity measured in each of the respective reference control exposure (Ave: 540 µmho/cm). Dissolved oxygen within the test jars remained above the minimum acceptable level (>4 mg/l) throughout the test (OMOEE, 1994d). Test temperature was at or near 20°C for each bioassay.

The amount of total ammonia present in the overlying test water and the converted un-ionized ammonia readings, based on test temperature and pH, are provided. Several of the test sediments resulted in un-ionized ammonia above the PWQO of 0.02 mgNH_3/L in all three toxicity tests. Average values were lowest in the mayfly test (0.04 mgNH_3/L), increased in the midge test (0.06 mgNH_3/L) and were highest in the minnow test (0.12 mgNH_3/L). Elevated un-ionized ammonia readings were also found in the reference sediment (stn 5) for the midge and minnow tests. In general, it appears water quality characteristics at all test sites were representative of local conditions.

3.2 Sediment Characterization

The following sections summarize the sediment physical and chemical parameters to aid in the interpretation of the biological toxicity results. Chemical analysis is based on the sediment prepared for toxicity testing and results may differ from those reported for any field samples collected concurrently. Any dissimilarities are likely due to *in-situ* chemical heterogeneity and/or

TABLE A. Mean (± s.d.) water quality characteristics in Aurora wetland 1997 sediment bioassays.

Test Organism: Mayfly (Hexagenia limbata)[a] Test Temperature: 20.0°C (0.6)

Station	*pH*	*D.O.* mg/L	*Conductivity* µmho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.87 (.04)	8.7 (0.2)	304 (4)	<0.10 (0)	<0.003
Reference	8.34 (.11)	8.5 (0.3)	551 (82)	0.22 (0.14)	0.019
Stn 1	8.15 (.05)	8.5 (0.2)	514 (58)	1.05 (1.27)	0.064
Stn 2	8.39 (.08)	8.7 (0.3)	687 (92)	0.21 (0.18)	0.021
Stn 3	8.32 (.07)	8.6 (0.2)	526 (51)	0.14 (0.08)	0.011
Stn 4	8.11 (.20)	8.3 (0.5)	526 (107)	1.02 (1.12)	0.061

Test Organism: Midge (Chironomus tentans)[b] Test Temperature: 19.5°C (0.5)

Station	*pH*	*D.O.* mg/L	*Conductivity* µmho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.84 (.12)	8.6 (0.3)	314 (9)	0.14 (0.04)	0.004
Reference	8.20 (.06)	8.7 (0.1)	502 (74)	0.69 (0.46)	0.043
Stn 1	8.17 (.01)	8.7 (0.1)	467 (44)	2.45 (1.62)	0.092
Stn 2	8.23 (.08)	8.7 (0.1)	681 (118)	0.11 (0.01)	0.007
Stn 3	8.07 (.16)	8.5 (0.2)	501 (51)	0.23 (0.17)	0.009
Stn 4	8.17 (.04)	8.6 (0.1)	500 (86)	3.86 (1.10)	0.146

Test Organism: Minnow (Pimephales promelas)[a] Test Temperature: 20.0°C (0.6)

Station	*pH*	*D.O.* mg/L	*Conductivity* µho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.35 (.26)	7.7 (1.4)	352 (41)	0.34 (0.24)	0.003
Reference	8.12 (.11)	8.4 (0.3)	567 (96)	1.35 (1.79)	0.052
Stn 1	7.78 (.16)	7.8 (0.8)	596 (116)	7.87 (7.80)	0.162
Stn 2	7.95 (.04)	8.3 (0.3)	685 (103)	0.37 (0.55)	0.014
Stn 3	7.88 (.20)	7.9 (1.0)	545 (59)	2.22 (3.55)	0.083
Stn 4	7.90 (.25)	7.6 (1.5)	596 (141)	9.03 (8.93)	0.251

a Sample size N=4; b Sample size N=3;

Underlining indicates un-ionized ammonia concentrations that exceed the PWQO of 0.02 mg/L

sampling depth and sample handling.

Physical and Nutrient Properties

Sediments were characterized for sand (2 mm-62 µm), silt (62-3.7 µm), clay (3.7-0.1 µm), % loss on ignition (%LOI), total organic carbon (TOC), total phosphorus (TP) and total Kjeldahl nitrogen (TKN) (Table B).

The test sediments encompassed a range in substrate types according to the particle size distribution (Millar *et al.,* 1965). The coarsest textured sediment occurred at stn 4 which had the highest sand fraction of 33%. The sediment was classified as loam. The negative control sediment was also classified as loam and had an identical particle size distribution as stn 4. The reference control sediment was categorized as silt loam and differed from each of the test sediments. Stations 1 and 3 had the highest silt fraction of at least 56% and considered to be silty clay loam. Station 2 sediment was a silty clay and represented the finest textured substrate.

The gradation in particle size among test sediments appeared to coincide with their relative position within the stormwater facility. As expected, the heavier, larger-sized particles were retained within the settling basin, as represented by stn 4. Progressively finer textured sediments occurred with increasing distance away from the inlet. Eventually only fine-grained silty sediment existed within the greenhouse area and proximity to the stormwater outlet.

Despite the differences in substrate type among test sites, there were no concurrent changes in the amount of organic enrichment. In fact, the TOC concentrations among the test sediments of 22 to 28 mg/g were comparable to that associated with the reference sediment situated outside the wetland environment (TOC: 23 mg/g). Very low nutrient content e.g. TP, TKN, was evident in test and control sediments and concentrations were either at or below Provincial Sediment Quality Guideline (PSQG) Lowest Effect Levels (LELs).

Most of the samples contained vegetative matter in the form of twigs and fibrous weeds which were removed during sample preparation. The reference control, stn 2 and stn 3 sediment was light brown in color and stns 1 and 4 were black-grey. Varying quantities of indigenous oligochaetes were found in the samples. Since the presence of oligochaetes was limited to only the test sediments and not found in the reference material, this may be a result of periodic episodes of low-oxygen conditions existing within the stormwater facility and would depend on water flow and possibly even areas of stagnation. Indeed, a lack of moisture was evident at the time of sample collection for some sampling sites. Fluctuations in water temperature on a seasonal basis could also influence local dissolved oxygen concentrations (Smith and Mulamoottil, 1998).

Trace Metal Sediment Concentrations

Bulk sediment samples were analysed for 11 trace metals (Table C). The sediment metal concentrations were compared to Severe Effect Level (SEL) and Lowest Effect Level (LEL) concentrations as outlined in the Provincial Sediment Quality Guidelines (PSQGs) (Persaud *et al.*, 1992). The SEL is defined as that chemical concentration in the sediment that could be detrimental to the majority of the macrobenthos and the LEL is the sediment contaminant

TABLE B. Sediment physical and nutrient characteristics in control(s) and Aurora wetland 1997 sediment used in sediment bioassays.

Station	*% Sand* (2mm-62um)	*% Silt* (62-3.7um)	*% Clay* (3.7-0.1um)	*% LOI*	*TOC* mg/g	*TP* mg/g	*TKN* mg/g
Honey Harbour Control	33.0	44.0	22.3	8.6	40	1.0	3.1
Station 5 Reference Control	20.0	53.4	26.9	4.7	23	0.9	1.1
Greenhouse Station 1	3.0 <T	57.0	39.5	3.7	27	0.2	0.6
Outlet Station 2	7.0	51.9	41.1	1.3	22	0.5	0.4 <T
Wetland Station 3	16.0	56.8	26.7	3.4	28	0.5	0.5
Forebay Station 4	33.0	46.2	21.0	2.8	22	0.3	0.2 <T
PSQG SEL Conc (mg/g dry weight)					100	2.0	4.8

< T - Trace Amount.

TABLE C. Bulk concentrations of trace metals in control(s) and Aurora wetland 1997 sediment (µg/g dry weight) used in sediment bioassays.

Station	*Al %*	*As*	*Cd*	*Cr*	*Cu*	*Fe %*	*Hg*	*Mn*	*Ni*	*Pb*	*Zn*
Honey Harbour Control	2.2	5.0	1.4	44	24	3.7	0.09	980	35	54	140
Station 5 Reference Control	1.3	0.2 <W	0.7 <T	23	12	1.8	0.03 <T	540	14	11	46
Greenhouse Station 1	1.1	0.3 <T	0.7 <T	26	23	1.7	0.02 <T	430	15	16	88
Outlet Station 2	1.2	0.3 <T	0.5 <T	24	17	1.6	0.02 <T	500	14	16	62
Wetland Station 3	0.9	0.2 <W	0.6 <T	22	18	1.4	0.02 <T	390	11	14	70
Forebay Station 4	0.5	0.2 <W	0.6 <T	16	18	0.9	0.03 <T	250	7	11	58
PSQG SEL Conc.	NA	33	10	110	110	4.0	2.0	1100	75	250	820
PSQG LEL Conc.	NA	6.0	0.6	26	16	2.0	0.2	460	16	31	120

<W - Not Detected; <T - Trace Amount; Underlining indicate sediment trace metal concentrations that exceed PSQG-LELs; NA -Not Available.

concentration which can be tolerated by most benthic species.

None of the metals reached SEL concentrations and 7 of 10 metals remained below LEL concentrations. Cadmium and copper were most often measured above LEL concentrations but approximated those levels that are naturally-occurring as exemplified by the reference control sediment. Metal sediment concentrations also matched those levels typically found in Great Lakes nearshore sediments (Ankley *et al.*, 1994), as represented by the negative control sediment collected from Georgian Bay. The inorganic sediment data indicated no metal enrichment in the test sediments above background levels.

Major Ion Sediment Concentrations

Bulk sediment was analysed for four major ions (Na, P, Fl, Cl) and other miscellaneous parameters (Table D). Concentrations of sodium, chloride and sulphate were elevated at some test sites above those encountered outside the stormwater facility (stn 5). An increased salt content was found at stn 2 and may be a result of a build-up of nutrient amendments that were applied to the greenhouse plants. Substantially higher sulphate concentration was associated with stn 4 sediment of 590 µg/g and concentrations declined with increasing distance. The lowest concentration was recorded for stn 1 of 92 µg/g. A similar distribution in dissolved substance concentrations have been measured *in-situ* on water samples (Smith and Mulamoottil, 1998). Sediment quality guidelines are not available for any of the above substances and are not expected to be to toxicologically important.

Organic Chemical Sediment Concentrations

Concentrations of 19 organochlorine pesticides and 14 chlorinated organic compounds, including total PCBs, in the Aurora wetland test sediments were below the respective detection limits (Table E). Trace amounts of pp-DDE (2-3 ng/g) were found in three test sediments.

Individual and total PAH sediment concentrations varied among sites (Table F). The highest total PAH concentration was reported for stn 4 of 11 µg/g, intermediate concentrations for stns 1 and 3 (3 µg/g) and the lowest concentration for stn 2 (0.8 µg/g). None of the 16 individual PAH compounds were detected in the reference control sediment. At least 10 of the 12 individual PAHs with existing PSQGs were above the LEL for stn 4. At this site, flouranthene, phenanthrene and pyrene contributed 47% of the total PAH sediment concentration. There appears to be a progressive decline in sediment PAH depending on proximity to the wetland inlet with highest concentrations occuring within the settling basin. A decrease in total PAH of 92% was apparent between the inlet (stn 4) and outlet (stn 2) test locations.

3.3 Mayfly (*Hexagenia limbata*) 21-day Lethality and Growth Results

The biological data for the two endpoints, mortality and growth, is summarized in Table G. Mayfly mortality was nil for both control sediments and was statistically similar to each test sediment (0% to 3% mortality; $p=0.57$). Mayfly body weights varied significantly among test and control sediments (ANOVA; $p<0.0001$). The lowest mayfly growth occurred for stn 4, which had

TABLE D. Other sediment parameters in reference control and Aurora wetland 1997 sediment used in sediment bioassays.

Station	*Sodium* µg/g	*Potassium* µg/g	*Fluoride* µg/g	*Chloride* µg/g	*Sulphate* µg/g	*pH*	*Conductivity* µS/cm
Station 5 Reference Control	260	2300	100	28	41	7.5	240
Greenhouse Station 1	480	2600	70	82	92	7.6	1100
Outlet Station 2	730	2200	35	280	100	7.8	740
Wetland Station 3	500	1900	49	79	310	7.7	540
Forebay Station 4	280	950	53	21	590	7.6	700

TABLE E. Bulk concentrations of chlorinated organics and pesticides in Aurora wetland 1997 sediment (ng/g dry weight) used in sediment bioassays.

All Stations (exceptions listed below)	Total PCBs	20 <W
	Heptachlor	1 <W
	Aldrin	1 <W
	Mirex	5 <W
	a-BHC	1 <W
	b-BHC	1 <W
	g-BHC	1 <W
	a-Chlordane	2 <W
	g-Chlordane	2 <W
	Oxychlordane	2 <W
	op-DDT	5 <W
	pp-DDD	5 <W
	pp-DDT	5 <W
	Methoxychlor	5 <W
	Heptachlor epoxide	1 <W
	Endosulphan I	2 <W
	Dieldrin	2 <W
	Endrin	4 <W
	Endosulphan II	4 <W
	Endosulphan sulphate	4 <W
	Hexachlorobutadiene	1 <W
	Octachlorostyrene	1 <W
	Hexachlorobenzene	1 <W
	123-Trichlorobenzene	2 <W
	124-Trichlorobenzene	2 <W
	135-Trichlorobenzene	2 <W
	1234-Tetrachlorobenzene	1 <W
	1235-Tetrachlorobenzene	1 <W
	1245-Tetrachlorobenzene	1 <W
	Hexachloroethane	1 <W
	Pentachlorobenzene	1 <W
	236-Trichlorotoluene	1 <W
	245-Trichlorotoluene	1 <W
pp-DDE	Reference (Stn 5)	1 <W
	Station 1	3 <T
	Station 2	1 <W
	Station 3	2 <T
	Station 4	3 <T

<W - Not Detected; <T - Trace Amount.

TABLE F. Bulk concentrations of polycyclic aromatic hydrocarbons (ng/g dry weight) in Aurora wetland 1997 sediment used in sediment bioassays.

Parameter	Reference Station 5	Greenhouse Station 1	Outlet Station 2	South Station 3	Forebay Station 4
Acenaphthene	20 <W	20 <W	20 <W	20 <W	120
Acenaphythylene	20 <W	20 <W	20 <W	20 <W	20 <W
Anthracene	20 <W	20 <W	20 <W	20 <W	230
Benzo[a]anthracene	20 <W	150	20 <W	200	710
Benzo[b]fluoranthene	20 <W	410	120	410	1200
Benzo[k]fluoranthene	20 <W	150	38 <T	170	380
Benzo[ghi]perylene	40 <W	240	89	250	730
Benzo[a]pyrene	20 <W	220	58	240	870
Chrysene	20 <W	210	20 <W	250	840
Dibenzo[ah]anthracene	40 <W	40 <W	40 <W	40 <W	40 <W
Fluoranthene	20 <W	560	100	480	2200
Fluorene	20 <W	20 <W	20 <W	20 <W	120
Indeno[123-cd]pyrene	40 <W	260	89	280	890
Naphthalene	20 <W	20 <W	20 <W	20 <W	20 <W
Phenanthrene	20 <W	190	37 <T	170	1300
Pyrene	20 <W	540	120	620	2100
Total PAHs	380 <W	3070	831	3210	11770

<W - Not Detected; <T - Trace Amount; Underlining indicate sediment PAH concentrations that exceed PSQG-LELs; PSQG's not available for acenapthene, acenaphthylene, benzo[b]flouranthene and naphthalene.

TABLE G. Summary of biological results on mayfly, midge and minnow sediment bioassays for control(s) and Aurora wetland 1997 sediments.

Mean values (± standard deviation) where sample size n=3 replicates for mayfly, midge and minnow tests.

Test Organism	Mayfly : *Hexagenia limbata*		Midge : *Chironomus tentans*		Fathead Minnow : *Pimephales promelas*
Station	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*
Honey Harbour Control	A 0 (0)	C 6.36 (0.1)	A 6.6 (11)	B 8.08 (0.3)	A 0 (0)
Station 5 Reference Control	A 0 (0)	B 14.69 (1.1)	A 6.6 (7)	A 10.78 (0.8)	A 3.3 (6)
Greenhouse Station 1	A 0 (0)	A 20.26 (0.3)	A 4.4 (4)	B 7.37 (1.4)	A 3.3 (6)
Outlet Station 2	A 3.3 (6)	A 20.06 (1.4)	A 2.2 (4)	B 8.46 (1.3)	A 3.3 (6)
Wetland Station 3	A 3.3 (6)	AB 17.81 (4.3)	A 0 (0)	AB 8.96 (0.8)	A 3.3 (6)
Forebay Station 4	A 0 (0)	C 8.64 (0.9)	A 4.4 (4)	C 4.52 (0.8)	A 10.0 (10)
% MSD	6.8	-	12.4	-	12.7
% C.V.	184.6	9.8	108.5	12.7	147.8
D.P.	1.3	3.0	1.7	6.1	0.9

A Means sharing a common letter within a column are not significantly different; Tukey's HSD test for % Mortality ($p<0.05$) and planned comparisons using LSMEANS for comparing Body Weight ($p<0.01$).

MSD - Minimum Significant Difference; C.V. - Coefficient of Variation; D.P. - Discriminatory Power.

an average body weight that was 41% lower than the reference control animal weight. Growth for stn 4 was considered to be moderately impaired. Organism weight for stn 4 was also substantially lower than that observed for each of the other test sediments. Nymph growth measured for stns 1, 2 and 3 was either equal to or higher than the reference control weight and is indicative of no negative sublethal effect.

The fairly low rate of growth observed in the negative control sediment was attributed to the longer storage time (~12 months) versus the freshly collected test sediments (~ 1 week). Prolonged storage of the control sediment may have affected the nutritive value of the sediment, thereby altering feeding rates (Boese *et al.*, 1996). Mayfly growth depends on the quality and quantity of detrital material found in the sediment, since supplemental feeding was not provided throughout the test. For the purpose of data interpretation, the reference control sediment would be a better indicator of mayfly growth potential because it was collected on-site, had similar nutrient content and was subjected to storage conditions similar to that of test sediments.

3.4 Chironomid (*Chironomus tentans*) 10-day Lethality and Growth Results

Results for chironomid growth and lethality are presented in Table G. Chironomid mortality was similar among sediments (ANOVA; $p=0.69$) and ranged from 0% to 6%. In terms of sublethal effects, a significant and severe growth reduction was noted for stn 4 (Ave: 4.5 mg). The final average body weight was at least 52% lower than the combined average midge weight attained in both the negative and reference control sediments (ANOVA; $p< 0.0001$). Acceptable levels of midge growth were observed for the remaining test sediments and received a similar ranking to either one or both of the control sediments.

3.5 Fathead Minnow (*Pimephales promelas*) 21-day Lethality Results

Juvenile fathead minnow percent mortality data is reported in Table G. Negative control survival was 100%. The reference control sediment incurred a loss of 3%, along with stns 1, 2, and 3. Station 4 resulted in 10% minnow mortality but was not significantly higher than the control(s) (Dunnett's t-test) and other test sediments (Tukey's Multiple Range test). There was no evidence of fish avoidance behaviour throughout the bioassay according to daily laboratory observations made on the test chambers, suggesting little or no organism stress. Gross measurements made on fish weight on Day-21 also indicated no obvious differences in feeding behaviour. Test weight (Range: 0.46 to 0.53 mg) fell within 10% of the weight attained in the control animals (0.52 mg).

3.6 Quality Assurance Data

An evaluation was made on the biological data in order to determine the repeatability of the test results (Table G). There was a clear distinction in data quality between the lethal and sublethal data sets. This applied to test precision, as well as the ability to differentiate between toxic and non-toxic sites. Organism mortality consistently failed to identify any toxic sites and is reflected in the very poor coefficient of variation or % C.V. The poor test precision (108% to 184% C.V.) was a direct result of low-level sediment contamination. In other words, all test

sediments shared an identical degree of lethality and resulted in very poor discriminatory power (D.P. = 0.9 to 1.7). Nevertheless, each lethality endpoint was equally effective in detecting small differences in mortality among sites as shown by the minimum significant difference (MSD) values of 6% to 12%. In other words, the test design was adequate in determining even small differences in mortality as being significant and met the same quantitative standards found in other sediment toxicity tests that followed MOE standard test methods. Previous series of bioassays had similar reported average MSDs of 12% (*H. limbata*; n=8), 18% (*C. tentans*; n=10) and 15% (*P. promelas*; n=8) (D. Bedard, OMOE, unpublished data).

The most effective endpoint in determining differences in sediment quality applied to both mayfly and midge growth. The test precision was excellent for the two benthic species (9% and 12% C.V.). Both sublethal endpoints identified the same test sediment (stn 4) as eliciting a significant growth effect. *Chironomus* growth had a slightly better discriminatory power (D.P. = 6) relative to the mayfly growth data (D.P. = 3) as reflected in the level of growth impairment observed between the two benthic species.

The 48-hour copper LC50 (95% C.I.) for the water-only reference toxicant exposure for *H. limbata* was 0.33 (0.33 - 0.33) mg/L. This value was within the acceptable 48-h LC50 (± 2 s.d.) range of 1.19 (1.34) mg/L, according to a previous series of reference toxicant tests. Similarly, for *C. tentans*, the LC50 was 0.97 (0.80 - 1.19) mg/L, as compared to an expected 48-h LC50 (± 2 s.d.) of 1.23 (0.94) mg/L. This indicates that the relative sensitivity of the test organisms fell within a normal response range.

3.7 Chemical Bioaccumulation in *Pimephales promelas*

The examination of chemical availability to aquatic organisms is valuable for assessing the potential for chemical transfer through the food web. The primary objective of this test design is to make general observations on whole organism tissue concentrations as they relate to overall bulk chemical concentrations in the sediment and differences in chemical uptake among sites. Surviving fathead minnows were submitted for the analysis of total PCBs, total PAHs and pesticides. All values (wet weight) are based on unpurged, whole body tissue concentrations.

The sources of organic compound accumulation to forage fish include direct contact with the sediment and uptake from the overlying water. Factors that control chemical accumulation by forage fish include those that affect chemical adsorption and desorption such as sediment organic content, particle size distribution and the chemical's solubility properties commonly expressed by the octanol-water partition coefficient, Kow (Lake *et al.*, 1990). Biotic factors affecting uptake include metabolism and lipid content (Boese *et al.*, 1995).

Table H reports the total PCB, total PAH and pp-DDE tissue concentrations (wet weight) as an average value ± standard deviation for each station. Total PCBs and total PAHs were not detected in fathead minnows for most test sediments. Trace amounts of total PAHs were measured for stn 1 and were just slightly higher than the detection limit. The pesticide, pp-DDE was present in trace amounts (4 - 7 ng/g) at all test sites, including the reference control fish.

TABLE H. Total PCBs, total PAHs and pp-DDE concentrations (ng/g, wet wt) in fathead minnows exposed to control(s) and Aurora wetland 1997 sediments in the laboratory.

Station	*Total PCBs* ηg/g	*Total PAHs* ηg/g	*pp-DDE* ηg/g
Honey Harbour Control	30 (14) <T	460 (84) <T	12.5 (6)
Station 5 Reference Control	30 (14) <T	400 (0) <W	7.5 (3) <T
Greenhouse Station 1	20 (0) <W	420 (28) <T	7.0 (0) <T
Outlet Station 2	20 (0) <W	400 (0) <W	4.5 (5) <T
Wetland Station 3	20 (0) <W	400 (0) <W	7.5 (1) <T
Forebay Station 4	20 (0) <W	400 (0) <W	7.5 (2) <T

Values reported as average ± standard deviation; Sample size n=2.

<W - Not Detected; <T- Trace Amount.

Residual amounts of pp-DDE were also measured in the sediment at three of the test sites. The lack of chemical uptake in the test animals above those levels reported in the control(s) animals suggest minimal organic chemical availability and appear to coincide with the low exposure concentrations reported in the sediment.

4.0 DISCUSSION

A ranking system was used to identify differences in sediment quality among sites for the four test sediments. This was determined by the magnitude of an effect using statistical test methods. Each endpoint was considered as being either a significant, toxic (T) or non-significant, non-toxic (N) response. In addition, the lethality endpoint received a greater weighting over the respective sublethal endpoint, where applicable. The final rating is based on the total number of positive hits recorded for each of the five biological endpoints. Each sediment fell into one of the following classifications (listed from least impacted (high quality) to most impacted (very low quality)): non-impacted (high); slightly impacted (slight); intermediately impacted (moderate); strongly impacted (low); and very strongly impacted sites (very low) (Table I).

The final ranking of the Aurora wetland sediments was governed solely by the results obtained using the sublethal endpoint. A slightly lower rating in sediment quality of 'slightly impacted' was assigned to stn 4, based on the poor growth observed in both the mayfly and midge bioassays. Benthic growth was moderately to severely reduced by as much as 50% relative to the other test sites and that measured under 'clean' reference conditions. Each of the other test sediments received a high quality rating due to the absence of any significant lethal or sublethal effects. In addition, there was no evidence of chemical bioaccumulation at any test sites for a number of organic compounds.

Interpretation of the bioassay data typically involves correlation analysis, but this study component was excluded due to the small sample size ($n<5$). Instead, a cursory examination of the data is provided. The outcome of the sediment bioassays detected only one of the test sediments having a negative biological effect. An examination of all the sediment parameters including physical characteristics, nutrient parameters, metal and organic chemical concentrations is required, in order to determine the causative factor(s). For the most part, stn 4 sediment shared many sediment physicochemical parameters to those found in either control or other test sediments. For example, substrate type was comparable to the negative control sediment, nutrient content was analogous to each of the other sediments within the study area and metal sediment concentrations were often the lowest measured. The only difference noted was with respect to total PAH sediment concentration. Station 4 sediment yielded a total PAH concentration of 11 µg/g that was approximately 4-times higher than that measured for any of the remaining three test sediments (Range: 0.8 - 3 µg/g).

The likelihood of PAH-related toxicity at the sublethal level is further supported by the PSQGs and effect-level concentrations cited in a number of sediment bioassay reports. Unfortunately the test results did not permit an effect-concentration to be calculated specifically for the Aurora site due in part to the inadequate sample size and the absence of any gradation in biological response. Instead, information obtained at other PAH-contaminated sites that used similar MOE laboratory sediment bioassay test methods will be examined. Sediment toxicity tests that were conducted in 1995 for 14 samples at the Northern Wood Preservers site, near Thunder

TABLE I. Spatial variability in sediment toxicity and sediment quality for Aurora wetland 1997 samples.

Station	Sediment Quality	Sediment Total PAHs µg/g	Mayfly Mortality	Mayfly Ave wt	Midge Mortality	Midge Ave wt	Minnow Mortality
Reference Station 5	High	0.3	N	N	N	N	N
Greenhouse Station 1	High	3.0	N	N	N	N	N
Outlet Station 2	High	0.8	N	N	N	N	N
Wetland Station 3	High	3.2	N	N	N	N	N
Forebay Station 4	Slight	11.7	N	T	N	T	N

N - Not Toxic, % mortality less than control criteria or $p>0.05$ and $p>0.10$ for growth data;
T - Toxic, % mortality greater than control criteria or $p<0.05$ and $p<0.10$ for growth data.

Bay, Ontario found significant levels of toxicity (Jaagumagi *et al.,* 1998). A 21-day mayfly LC50 of 538 µg/g and a 10-day midge LC50 of 718 µg/g for total PAHs were derived using linear regression analysis (Bedard and Petro, 1997). These values agreed favourably to the PSQG-SEL concentration of 10,000 µg/g corrected for sediment organic content. The normalized SEL concentration was 460 µg/g using an average TOC of 4.6%. The acute effect-concentration can then be converted to a sublethal effect-concentration using a safety factor of 10 (McCarty, 1986; McCarty *et al.*, 1992). The estimated IC50 (IC50 is the chemical sediment concentration associated with a 50% growth reduction) would be 46 µg/g. A similar projection can be made to determine the IC50 that would apply to the Aurora test sediment. Station 4 had a TOC content of 2.2% which would yield a SEL concentration or LC50 of 220 µg/g for total PAH. The estimated IC50 would then be 220 µg/g divided by 10 or 22 µg/g. The actual total PAH sediment concentration was 11 µg/g and closely approximates the sediment concentration likely to elicit a growth reduction of 50%.

In another study, a sublethal effect-concentration was directly measured for a series of field sediments collected within the St. Marys River in 1992 (Bedard and Petro, 1997). Varying degrees of benthic growth impairment were observed using the mayfly, *Hexagenia* and the chironomid, *Chironomus* for sediments with total PAH concentrations ranging from 0.8 µg/g to 14 µg/g. This was a similar range in total PAH sediment concentration found in the Aurora test sediments. The laboratory-derived sublethal effect-concentration associated with 50% growth reduction was 25 µg/g using *Hexagenia* in a 21-day test and 11 µg/g for *Chironomus tentans* in a 10-day test.

According to the above direct and indirect calculations of sublethal effect-concentrations for total PAHs, it appears there is reasonable evidence to suggest stn 4 sediment had an exposure concentration with the potential for eliciting a 50% growth reduction using benthic organisms.

The forebay appears to be subjected to higher contaminant loadings of PAHs relative to other sites within the stormwater facility. Station 4 is situated closest to the inlet and initially receives stormwater entering from the surrounding watershed. PAHs have been associated with urban runoff (Wren *et al.,* 1997) and road runoff (Boxall and Maltby, 1997). The differences in PAH sediment concentration among test sites can be related to the function of various structures within the stormwater facility. The purpose of the forebay is to act as a settling basin for the accumulation of heavier sediment particles and presumably any associated contaminants. The forebay was effective in retaining a higher fraction of sand-sized particulates relative to other sites, along with a higher PAH sediment concentration. In contrast, little or no substantive concentrations of PAHs were evident at the other locations.

The fathead minnow tissue data indicate the sediment is not contributing to elevated contaminant levels in fish under laboratory conditions. Pesticide, PAHs and PCBs were typically found at trace amounts or less, not unlike concentrations attained for the control minnows.

5.0 CONCLUSIONS

1. Overall, the Aurora wetland sediments tested in the laboratory were considered to be of good sediment quality based on mayfly, midge and minnow lethal and sublethal endpoints,

chemical uptake and bulk sediment chemistry. The only exception was the sediment collected from the forebay which had a total PAH sediment concentration of 11 µg/g.

2. Station 4 sediment was found to be sublethal to mayfly nymphs and midge larvae resulting in moderate to severe levels of growth reduction. The effect was attributed to the sublethal-effects concentration of total PAHs measured in the bulk sediment. The IC50 concentration corresponded with other cited sublethal effect-level concentrations.

3. Station 1, 2 and 3 sediments elicited no negative impact on organism surivival or growth. There were no obvious differences in sediment chemistry among these sites despite their spatial distribution within the stormwater facility.

4. Chemical bioaccumulation data indicated minimal availability of total PCBs, PAHs and DDE and reflect those concentrations obtained at low-level conditions. Tissue concentrations were either at trace or non-detected.

5. Current conditions within the stormwater facility indicate fairly low chemical sediment concentrations that were not related to any lethal biological effect or chemical uptake. The occurrence of benthic growth impairment was limited to one location (station 4) and may warrant continuing monitoring of the forebay sediments. The total PAH sediment concentration of 11 µg/g is inferred as the factor that best explained the differences in the sublethal test endpoint. Toxic effects may become apparant at levels approximately one order of magnitude higher than that currently measured, assuming no change in sediment total organic carbon content.

REFERENCES

American Society for Testing and Materials (ASTM). 1997. *Standard Test Methods for Measuring the Toxicity of Sediment-associated Contaminants with Fresh Water Invertebrates.* E1706-95b. In Annual Book of ASTM Standards, Vol. 11.05. Philadelphia, PA. pp.1138-1220.

Ankley, G.T., D.A. Benoit, J.C. Balogh, T.B. Reynoldson, K.E. Day and R.A. Hoke. 1994. Evaluation of Potential Confounding Factors in Sediment Toxicity Tests with Three Freshwater Benthic Invertebrates. *Environ. Toxicol. Chem.* 13:627-635.

Bedard, D. and S. Petro. 1997. *Laboratory Sediment Bioassay Report on St. Marys River Sediments 1992 & 1995.* Ontario Ministry of the Environment, Standards Development Branch, ISBN 0-7778-6674-9, Toronto, Ontario. 59 p.

Bedard, D., A. Hayton and D. Persaud. 1992. *Ontario Ministry of the Environment Laboratory Sediment Biological Testing Protocol*, Ontario Ministry of the Environment, Water Resources Branch, ISBN 0-7729-9924-4, Toronto, Ontario. 23 p.

Boese, B.L., M. Winsor, H. Lee II, S. Echols, J. Pelletier and R.Randall. 1995. PCB Congeners and Hexachlorobenzene Biota Sediment Accumulation Factors for *Macoma nasuta* Exposed to Sediments with Different Total Organic Carbon Contents. *Environ. Toxicol. Chem.* 14: 303-310.

Boese, B.L., H. Lee II, D.T. Specht, J. Pelletier and R. Randall. 1996. Evaluation of PCB and Hexachlorobenzene Biota-Sediment Accumulation Factors Based on Ingested Sediment in a Deposit-feeding Clam. *Environ. Toxicol. Chem.* 15:1584-1589.

Boxall, A.B.A. and L. Maltby. 1997. The Effects of Motorway Runoff on Freshwater Ecosystems: 3. Toxicant Confirmation. *Arch. Environ. Contam. Toxicol.* 33:9-16.

Friesen, M.K. 1981. Hexagenia rigida (McDunnough). In: S.G. Lawrence (Ed.), *Manual for the Culture of Selected Freshwater Invertebrates,* Can. Spec. Publ. Fish. Aquat. Sci., 54:127-142.

Jaagumagi, R. and D. Persaud. 1996. *An Integrated Approach to the Evaluation and Management of Contaminated Sediments.* Ontario Ministry of Environment and Energy, Standards Development Branch, ISBN 0-7778-3845-1, Toronto, Ontario. 50 p.

Jaagumagi, R., D. Bedard and S. Petro. 1998. *Sediment and Biological Assessment of the Northern Wood Preservers Inc. Site Thunder Bay - July 1995 and September 1995.* Final Report. Ontario Ministry of the Environment, ISBN 0-7778-7728-7, Toronto, Ontario. 27 p.

Lake, J.L., N.I. Rubinstein, H. Lee II, C.A. Lake, J. Heltshe and S. Pavignano. 1990. Equilibrium Partitioning and Bioaccumulation of Sediment-associated Contaminants by Infaunal Organisms. *Environ. Toxicol. Chem.* 9: 1095-1106.

McCarty, L.S. 1986. The Relationship between Aquatic Toxicity QSARs and Bioconcentration for some Organic Chemicals. *Environ. Toxicol. Chem.* 5:1071-1080.

McCarty, L.S., G.W. Ozburn, A.D. Smith and D.G. Dixon. 1992. Toxicokinetic Modelling of Mixtures of Organic Chemicals. *Environ. Toxicol. Chem.* 11:1037-1047.

Millar, C.E., L.M. Turk and H.D. Foth. 1965. *Fundamentals of Soil Science*, 9th Ed., John Wiley and Sons, New York, NY.

Mosher, R.G., R.A. Kimerle and W.J. Adams. 1982. *MIC Environmental Assessment Method for Conducting 14-day Partial Life Cycle Flow-through and Static Sediment Exposure Toxicity Tests with the Midge Chironomus tentans*, Report No. E5-82-M-10, Monsanto, St. Louis.

Ontario Ministry of the Environment (OMOE). 1979. *Rationale for the Establishment of Ontario's Water Quality Objectives.* 236 p.

OMOE. 1983. *Handbook of Analytical Methods for Environmental Samples.* Vol. 1 and 2. Laboratory Services and Applied Research Branch, Toronto, Ontario.

OMOE. 1989. *A Guide to the Collection and Submission of Samples for Laboratory Analysis.* Laboratory Services Branch, Toronto, Ontario. 81 p.

OMOE. 1990. *The Determination of Polychlorinated Biphenyls (PCB), Organochlorines (OC), and Chlorobenzenes (CB) in Fish and Clams by GLC-ECD.* PFAOC-E3136A. Laboratory Services Branch, Etobicoke, Ontario. February 19, 1990. 45 p.

OMOEE. 1994a. *The Determination of Trace Metals in Sediments by Spectro Inductively Coupled Plasma-Optical Emission Spectrometer (ICP-OES).* HMARSOIL-E3062A. Laboratory Services Branch, Etobicoke, Ontario. March 28, 1994. 45 p.

OMOEE. 1994b. *The Determination of Polychlorinated Biphenyls (PCBs), Organochlorines (OCs), and Chlorobenzenes (CBs) in Soil and Sediments by Gas Liquid Chromatography-Electron Capture Detection (GLC-ED).* PSAOC-E3270A. Laboratory Services Branch, Etobicoke, Ontario. July 18, 1994. 37 p.

OMOEE. 1994c. *The Determination of Polynuclear Aromatic Hydrocarbons (PAH) in Soil and Sediments by Gas Chromatography-Mass Spectrometry (GC-MS).* PSAPAH-E3350A. Laboratory Services Branch, Etobicoke, Ontario. July 20, 1994. 28 p.

OMOEE. 1994d. *Water Management Policies, Guidelines, Provincial Water Quality Objectives of the Ministry of Environment and Energy,* Toronto, Ontario. 30 p.

OMOEE. 1995a. *The Determination of Total Carbonate-Carbon in Soil and Sediments by Coulometry.* CARBONAT-E3012A. Laboratory Services Branch, Etobicoke, Ontario.

OMOEE. 1995b. *The Determination of Particle Size on Sediments by MICROTRAC Particle Size Analyzer Model 7991-2.* SMPART-E3128A. Laboratory Services Branch, Etobicoke, Ontario.

OMOEE. 1995c. *The Determination of Polynuclear Aromatic Hydrocarbons (PAH) in Biota by Gas Chromatography-Mass Spectrometry (GC-MS).* PFAPAH-E3351A. Laboratory Services Branch, Etobicoke, Ontario. July 20, 1995. 32 p.

Persaud, D., R. Jaagumagi and A. Hayton. 1992. *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario.* Ontario Ministry of the Environment, Toronto. 30 p.

SAS. 1985. Statistical Analysis Systems, *SAS/STAT User's Guide*, Version 6.03 Edition, SAS Institute, Cary, S.C.

Smith, S. 1998. *Aurora Stormwater Wetland Project - Phase II Progress Report.* Ontario Ministry of Environment, Standards Development Branch, Toronto, Ontario. 6.

Smith, S. and G. Mulamoottil. 1998. Constructed Wetlands for Stormwater Treatment. SWAMP Presentation, *Stormwater/CSO Technology Transfer Conference,* February 23-24, 1998, Toronto, Ontario. 29 p.

Steel, R.G.D. and J.H. Torrie. 1960. Principles and procedures of statistics. McGraw-Hill Book Company Inc., New York, 481 p.

Stephan, C.E. 1977. Methods for Calculating an LC50, In: F.L. Mayer and J.L. Hamelink (Eds.) *Aquatic Toxicology and Hazard Evaluation*, ASTM STP 634, Philadelphia PA, pp.65-84.

Townsend, B.E., S.G. Lawrence and J.F. Flannagan. 1981. Chironomus tentans (Fabricius). In:

S.G. Lawrence (Ed.), *Manual for the Culture of Selected Freshwater Invertebrates*, Can. Spec. Publ. Fish. Aquat. Sci., Vol. 54, pp. 127-142.

U.S. Environmental Protection Agency (US EPA). 1987. *Guidelines for the Culturing of Fathead Minnows Pimephales promelas for use in Toxicity Tests.* EPA-600-3-87-001. Environmental Research Laboratory, Duluth, MN, 42 pp.

Wren, C.D., C.A. Bishop, D.L. Stewart and G.C. Barrett. 1997. *Wildlife and Contaminants in Constructed Wetlands and Stormwater Ponds: Current State of Knowledge and Protocols for Monitoring Contaminant Levels and Effects in Wildlife.* Technical Report Series No. 269. Canadian Wildlife Service, Ontario Region, 109 p.