NATURALIZED TREATMENT WETLANDS FOR CONTAMINANT REMOVAL: A CASE STUDY OF THE BURNSIDE ENGINEERED WETLAND FOR TREATMENT OF LANDFILL LEACHATE

by

Catherine C. Galbrand

Submitted in partial fulfillment of the requirements for the degree of
Master of Environmental Studies

at

DALHOUSIE UNIVERSITY
Halifax, Nova Scotia
December 2003

Your file Votre référence
ISBN: 0-612-89735-4
Our file Notre référence
ISBN: 0-612-89735-4

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

A naturalized engineered treatment wetland was constructed in Burnside Park, Dartmouth, Nova Scotia in order to remediate the negative effects of landfill leachate influent from the former Burnside Drive landfill on the natural ecosystem of Wright's brook. It was hypothesized that the native vegetation and the vegetation establishment strategy selected for the site would yield a successfully established wetland bearing biological integrity, and that a naturalized system would effectively remediate the contaminated influent, which was high in iron, manganese, ammonia and suspended solids. Using a natural wetland as a vegetative model for the treatment wetland site, chosen native plants were screened for their phytoremediative potential, sedimentation and erosion control abilities, habitat facilitation potential, public deterrence abilities, growth requirements and tolerances. The selected plants were then transplanted into the treatment site. The biological integrity of the treatment wetland was compared to that of a pristine reference wetland using vegetation and macroinvertebrate indices in the second growing season of the site. Plant tissue analysis was performed on the plants in order to assess their phytoremediation ability to iron and manganese. Water quality analysis was performed in order to assess the water purification ability of the site.

Overall, the establishment methodology selected for the Burnside treatment wetland provided satisfactory quantities and varieties of plant species for the site and yielded a high establishment success rate. The results revealed that the reference wetland generally supported greater biological integrity than the treatment wetland which was mostly attributed to the immaturity of the treatment site and the overwhelming presence of iron particulate in the cells. The water quality analysis revealed that the treatment wetland ineffectively purified the influent which was also attributed in part to the site immaturity, its inadequate size, its compromised detention times due to breaks in the berms and the overwhelming presence of iron particulate in the cells. The plant tissue analysis showed that the broad-leaved cattail was the most effective accumulator of iron and the fowl mannagrass was the most effective accumulator of manganese. Several native plants demonstrated greater phytoremediative ability than the exotic canary reed grass, signifying that aggressive growth behaviour is not the only characteristic conducive to high phytoremediation capability.

LIST OF ABBREVIATIONS AND SYMBOLS

AMD	Acid Mine Drainage
ANOVA	Analysis of Variance
ASPB	American Society of Plant Biologists
ASPT	Average Score per Taxon
B	Bog
BAMR	Bureau of Abandoned Mine Reclamation
BCMELP	British Columbia Ministry of Environment, Land, and Protection
BMWP	Biological Monitoring Working Party
BOD	Biological Oxygen Demand
BTEX	Benzene, Toluene, Ethylbenzene and Xylenes
CCME	Canadian Council of Ministers of the Environment?
CLR	Chemical Loading Rate
COD	Chemical Oxygen Demand
COSEWIC	Committee on the Status of Endangered Wildlife in Canada
CPEO	Centre for Public Environmental Oversight
CPOM	Coarse Particulate Organic Matter
DDT	Dichlorodiphenyltrichloroethane
DNR	Department of Natural Resources
DO	Dissolved oxygen
DOC	Dissolved Organic Compounds
DW	Deepwater marsh
EC	Environment Canada
EDTA	Ethylene-diamine-tetraacetic acid
EQL	Estimated Quantitation Limit
ETSD	Ephemeroptera, Trichoptera, Sphaeriidae, and Odonata biotic index
FPOM	Fine Particulate Organic Matter
FRTR	Federal Remediation Technologies Roundtable
FWS	Free Water Surface
HLIS	Halifax Land Information Services
HRT	Hydraulic Residence Time
M	Meadow emergent marsh
n/a	Not applicable

nd	Not detected
NAP	National Academy Press
NGA	No Guideline Available
NRC	Natural Resources Canada
OW	Open water marsh
PAH	Polycyclic Aromatic Hydrocarbons
PCB	Polychlorinated Biphenyl
SM	Shallow marsh
SS	Shrub swamp
SSF	Sub-surface flow
TCE	Trichloroethylene
TDS	Total Dissolved Solids
TKN	Total Kjeldahl Nitrogen
TSS	Total Suspended Solids
USEPA	United States Environmental Protection Agency
VOC	Volatile Organic Compound
WCMC	World Conservation Monitoring Centre
WMPP	Wetlands Mapping Protection Program
A	Wetland area (m^2)
C	Celsius
C_e	Target effluent concentration (mg/L)
C_i	Influent concentration (mg/L)
C^*	Irreducible background concentration (mg/L)
d	Depth of submergence (m)
df	Degrees of freedom
E	Evaporation (m)
E_h	Redox potential
ET	Evapotranspiration
G_i	Groundwater inflows
G_o	Groundwater outflows
H'	Shannon-Weiner diversity index
H'_{var}	Variance in diversity
H_0	Null hypothesis
H_1	Hypothesis

k	First order rate constant (m/y)
mV	Microvolts
N	Total number of individuals of all species
n	Porosity of the medium (i.e. is typically 1.0 for FWS)
n_i	Number of individuals of a given species (i)
O&M	Operation and Management
P	Precipitation (m)
p_i	Proportional abundance of a given species (i).
P_n	Net precipitation
Q	Flow rate (m^3/y)
R	Runoff (m^3/d)
rc	Runoff coefficient
S	Species richness
S_i	Surface inflows
S_o	Surface outflows
T	Tidal inflows (+) or outflows (-)
T1	Transect 1
T2	Transect 2
T3	Transect 3
t	t-statistic
V	Volume of wetland water storage
WA	Watershed area (m^2)
$\Delta V/\Delta t$	Change in volume of wetland water storage per unit time
°	Degrees
α	Alpha
Σ	Summation

ACKNOWLEDGEMENTS

The successful completion of this study was made possible through the help of several persons. I would like to express my sincerest thanks to my supervisors Dr. A.E. Ghaly, Professor of Environmental Engineering, Biological Engineering Department, for his guidance, generosity and faith in my abilities, and Professor Ray Cote, Professor of Industrial Ecology, School for Resources and Environmental Studies for his encouragement, advice, and support throughout this process.

I would also like to take this opportunity to thank our technician Mr. Jack Vissers for the time and effort he put forth creating monitoring tools and various components of infrastructure for the wetland site. I would also like to extend my gratitude to fellow graduate students Mr. Mohab Kamal for his unconditional aid and friendship and Mr. Ricardo Bosshard for his hard work in the wetland.

I would also like to extend personal thanks to my dear friend Mr. Blayne Brown for being my unwavering champion, my fiancé Rex Turgano for his limitless patience and emotional support throughout my time here, and finally, to my parents and family whose encouragement and confidence has always driven me forward.

Without the financial support of the ELJB Foundation of Montréal and the TD Bank Friends of the Environment Foundation of Toronto, this thesis would not have been completed.

1. INTRODUCTION

1.1. Burnside Industrial Park

Industrial parks are urban areas usually located on the outskirts of cities and are zoned for industrial and business activities. They are typically distinguished by the close proximity of firms, sharable infrastructure, and often barren appearance (Grant, 1996). Unfortunately, as a result of current design and operational practices, industrial parks are notorious for high waste production, high energy and material consumption, and air and water pollution, and are consequently often linked with increased ecological health impacts (Grant, 2000). However, with proper planning and management reform, industrial parks can become environmentally compatible components of the urban ecosystem.

There are over 12,600 industrial parks worldwide, with 1000 in Canada. One of these is Burnside Industrial Park located in Dartmouth, Nova Scotia (Figure 1.1). It is the largest industrial park in the Atlantic Provinces with over 3000 acres and more than 1300 businesses supporting over 17,000 employees. Established in 1967, the park currently supports businesses in sales and service (48%), construction (11%), light manufacturing (10%), distribution and warehousing (9%), retail (8%), and professional, financial and other business services (14%). It houses a large range of local and multinational businesses, however, 90% of the businesses located in Burnside Park are small and medium-sized enterprises (SMEs), most of which employ less than 20 people (Eco-Efficiency Centre, 2003). Cumulatively, these operations are a significant generator of solid waste, wastewater discharges and air pollution in the region (Côté et al., 1996; and Noronha, 1999). In addition, like most industrial parks, Burnside Park was not established with ecological health in mind, and had limited regard for the natural landscape of the area. The guiding principle behind its design was the facilitation of growth by the most economical means. As a result, many forested and wetland areas were cleared to make way for its development (Grant, 1996). Landfills to accommodate increasing waste loads were hastily implemented where most convenient, and operated and decommissioned with little consideration for the environment (Sibbald, 2003).

In the early 1990s, Burnside Industrial Park was investigated as having potential for the application of the principles of industrial ecology which takes a systemic approach to the park and its industries (Noronha, 1999). Currently, the park is in the early stages of transformation into a model of environmental management. This reform extends to architecture, landscaping, and the protection of the parks' natural areas, especially its waterbodies. However, progress has been very slow due to continuing challenges such as limited enforcement of current regulations, flawed economic instruments, financial disincentives, and the slow development of partnerships (Côté, 2001; and Reid, 1995).

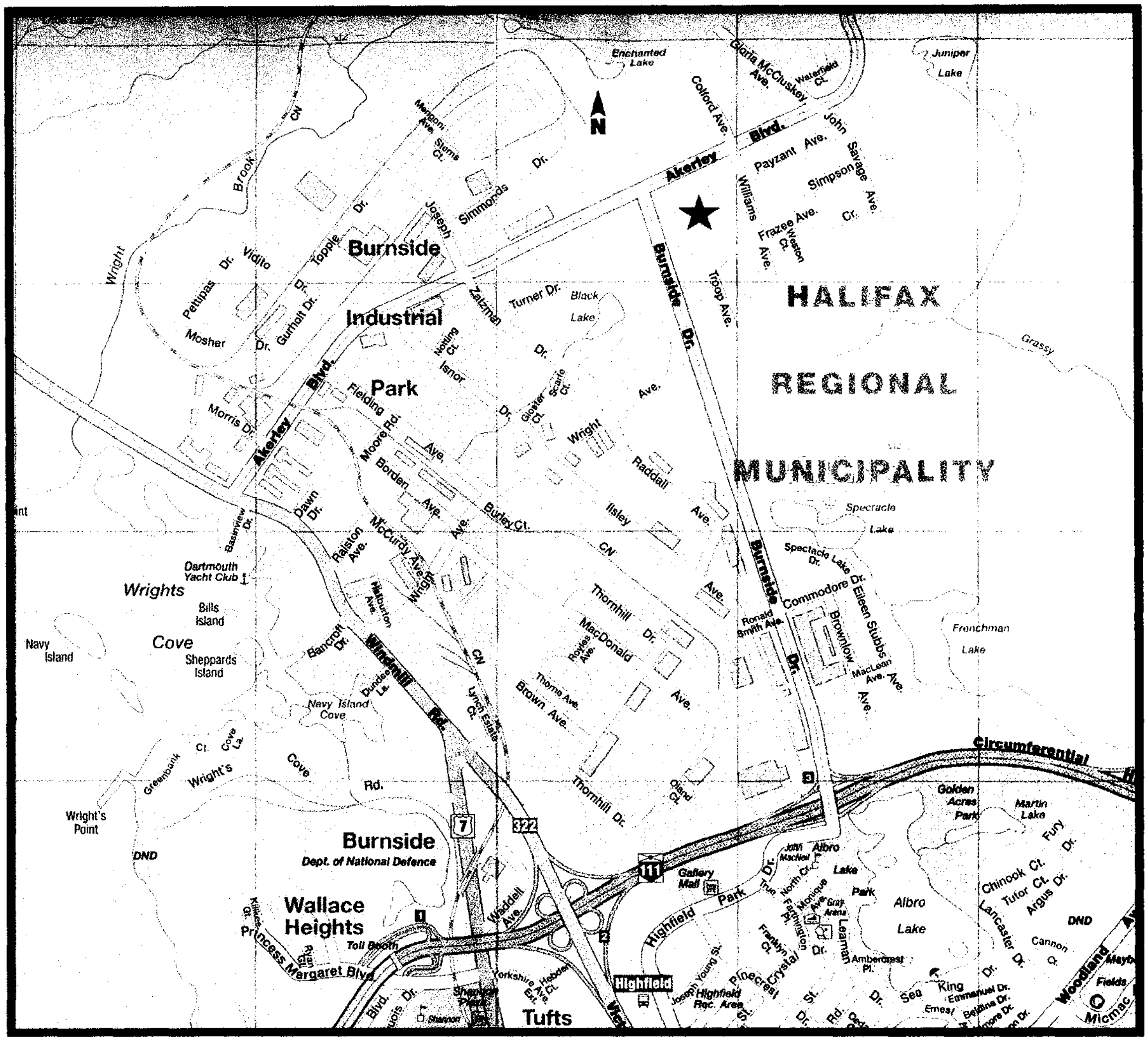


Figure 1.1. Burnside Industrial Park Regional Location (adapted from MapArt, 2001).

1.2. Burnside Drive Landfill

Almost all human activities create waste, which inevitably is discarded. Up until the 1970s, this typically involved open dumping due to the belief that this was the cheapest and most efficient disposal method. Landfill location was often based solely on convenience, and commonly included abandoned quarries, 'useless' swampland, roadsides and so on. Landfills were frequently left uncovered, emitting fowl odours, and providing a breeding ground for rodents and bacteriological and protozoal pathogens. In addition, wastes were often burned to reduce volume, releasing particulate matter and unburned gasses into the air. Waste disposed of in landfill sites was typically not separated, and often included persistent hazardous wastes including solvents and cleaners, petroleum products, commercial chemical products such as PCBs, DDT and formaldehyde, and even radioactive wastes (USEPA, 2002a). Consequently, the range of effects that dump sites and landfills manifested upon the environment were diverse.

Two of the most notorious products of landfills that continue to affect the surrounding environment years after a landfill has been decommissioned are methane gas and leachate. Methane (CH_4), a potent greenhouse gas, is a by-product of the anaerobic decomposition of organic solid waste. Notably, landfills are the largest anthropogenic source of methane on the planet. In addition to contributing to the global warming epidemic, methane build-up at old dumps and landfills has resulted in on-site explosions and fires (USEPA, 2002a).

Landfill leachate is created when soluble components are dissolved (leached) out of landfill waste materials by percolating water such as precipitation and runoff. Leachate may also carry insoluble liquids (such as oils) and small particles in the form of suspended solids. Contaminants which are often associated with leachate typically include volatile organic compounds (from fuel oils, solvents, and cleaners), ammonia, dissolved solids, heavy metals, toxic compounds (from hazardous wastes), and other products of the decomposition of the various waste materials disposed and contained within the landfill (USEPA, 2001). The organic strength alone of landfill leachate can be 20 to 100 times greater than raw sewage (USEPA, 2002a). Of course, the nature and concentrations of any leachate contaminants are as highly variable as the nature and concentrations of the wastes contained within the landfills themselves (Barlaz and Gabr, 2001). In seeing that most landfills of the day were unlined and absent of proper drainage systems, landfill leachate became and continues to be an infamous polluter of soils, surface water and even groundwater used for drinking water supplies (USEPA, 2002a).

Fortunately, landfilling practices have since improved with the adoption of "sanitary" landfills in 1970s, which by definition require the isolation (especially hydrogeological isolation) of wastes from the environment until they are rendered innocuous through biological, chemical and physical degradation processes. Modern sanitary landfills are well-engineered facilities which typically include impermeable liners, leachate and gas collection systems, the daily covering of waste with soil, sand or other material to prevent fires, reduce odours and discourage rodents, and permanent control by trained staff overseeing regular operation, maintenance and monitoring of gas and leachate control systems. As a result, modern landfills are far less likely to cause significant adverse effects on public health and the environment (Rushbrook and Pugh, 1999).

However, thousands of decommissioned and since long forgotten landfills and dumpsites continue to impact their surroundings, specifically through groundwater and surface water pollution (USEPA, 2002a). One of these landfills is the Burnside Drive landfill, (now decommissioned and currently known as the Don Bayer Sports Field) which is located near the northern boundary of the park, at the corner of Akerley Boulevard and Burnside Drive (Figure 1.2). This former landfill has been discharging leachate since its closure in the 1970s. This 13.4 acre open waste disposal site had accepted municipal, agricultural and industrial wastes, old tires, abandoned cars and demolition wastes (all of which were reportedly burned to reduce volume) from the Dartmouth Municipality. The dumpsite was graded, compacted and covered with two feet of soil upon closure, as was common in the day, with no regard to pollution control or aesthetics (Ghaly and Côté, 2001).

Leachate from the decomposing waste beneath the sports field, as well as stormwaters draining from a 55.1 hectare watershed (watersheds 1, 2 and 3) surrounding the landfill ultimately discharge into Wright's Brook (Figure 1.3). Watershed 1 is approximately 38.7 ha in size and encompasses the former Burnside Drive landfill, as well as the north and south sides of Frazee Avenue and areas surrounding Troop and Williams Avenue which connect at Frazee and subsequently drain into the former landfill's stormwater ditches. This watershed area supports several commercial buildings and adjoining parking areas as well as ungraded lots currently undergoing development along the more southern and eastern boundaries of the site. Watershed 2 is approximately 3.4 ha in size and supports two commercial buildings, accompanying parking lots, and a natural open-water wetland. Watershed 3 is approximately 13 ha in size and includes a decommissioned municipal landfill known as the Black Lake Landfill, which currently serves as a disposal site and storage area for the Halifax Regional Municipality. This landfill also contributes leachate to the Wright's brook ecosystem, however, dense, cattail-dominated wetland systems are able to buffer the leachate before it reaches the brook.

The leachate from the former landfill as well as stormwater from the surrounding watersheds 1, 2 and 3 ultimately drain into 1 of 3 stormwater ditches located on the western, northern and eastern

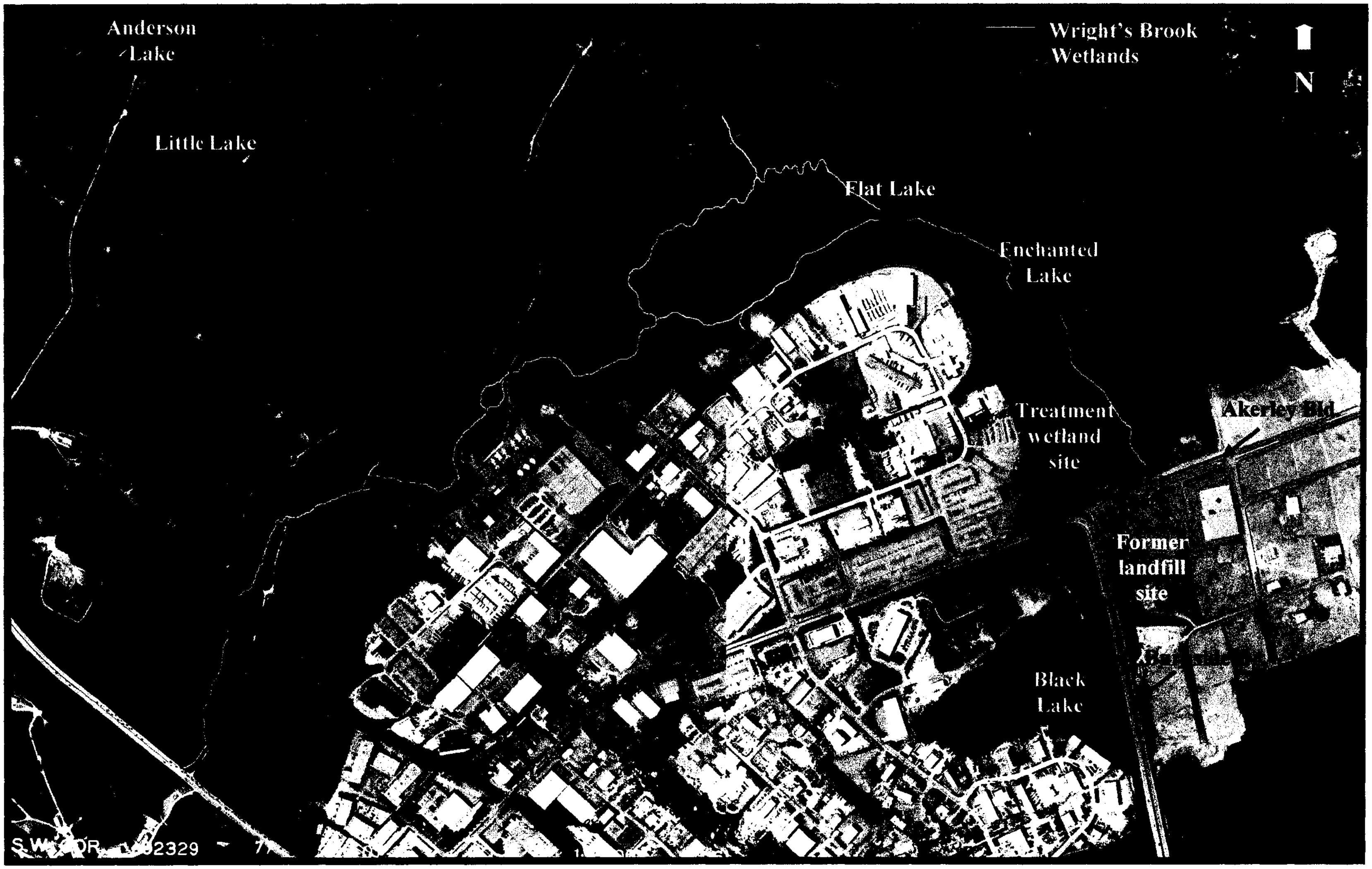


Figure 1.2. Aerial Photograph of Northern Boundary of Burnside Industrial Park and Surrounding Areas (adapted from HLIS, 1992).

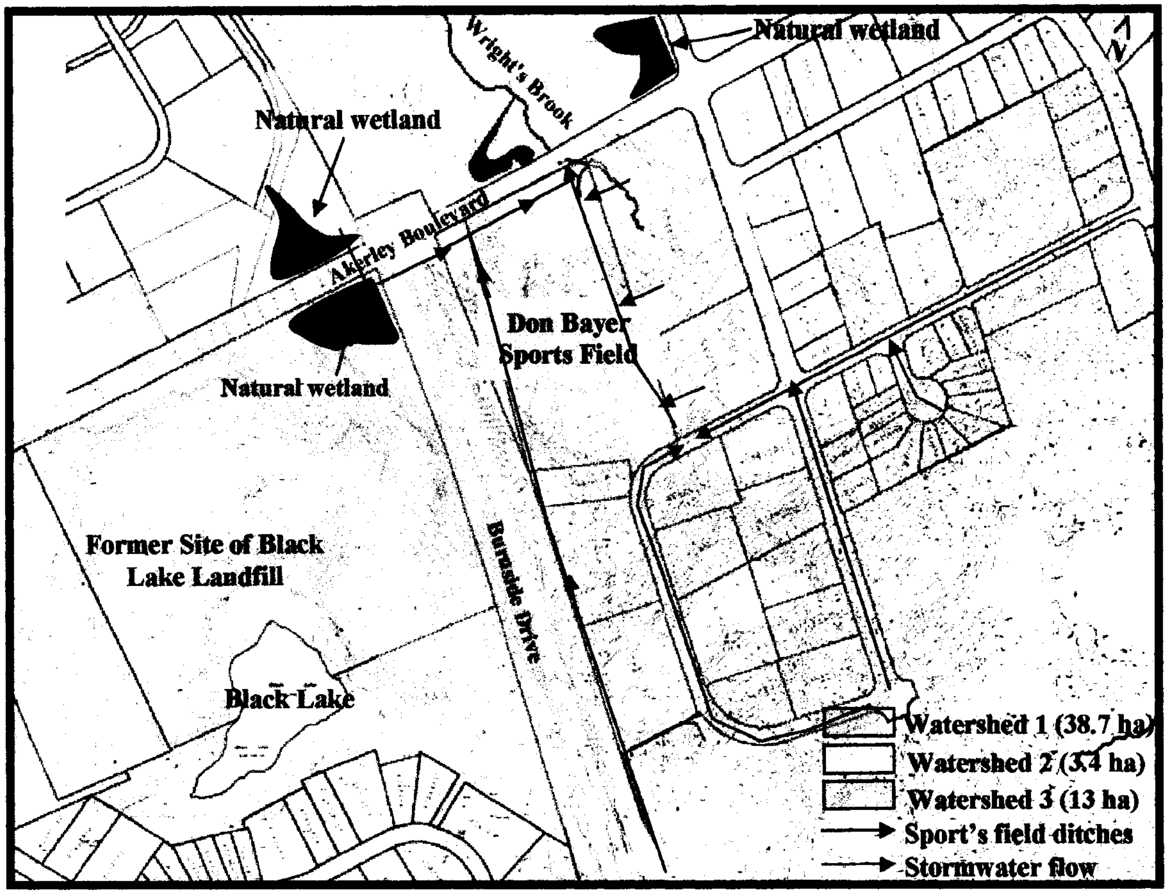


Figure 1.3. Wright's Brook Headwaters Watershed (adapted from HLIS, 2003).

borders of the sports field (Figure 1.4). All three stormwater ditches connect and discharge via gravitational forces through large culverts located beneath Akerley Boulevard into what was once a degraded, natural marsh at the head of the Wright's Brook system (Figure 1.5 and Figure 1.6). Water quality analyses of the stormwater ditches (Table 1.1) indicated that the leachate-contaminated waters contained elevated levels of iron, manganese, ammonia, and suspended solids. This wastewater discharge has had visibly adverse effects on Wright's Brook and its associated riparian wetlands, bogs and lake ecosystems. Signs of disturbance include iron hydroxide coagulations, orange staining, oily sheens, algal growths, scum-like substances, debris and elevated turbidity as shown in Figure 1.7 (Ghaly and Côté, 2001). Wright's Brook is a natural feature of Dartmouth's Burnside Industrial Park. The shallow, slow-moving brook traverses 4.6 km, passing through Enchanted and Flat lakes before discharging into the Bedford Basin of the Halifax Harbour (Figure 1.8). The system also boasts many natural wetland environments, which support valuable wildlife habitat, specifically for waterfowl (WMPP, 1988). Obviously, the continuing contamination and subsequent degradation of the Wright's Brook ecosystem as a result of the former municipal landfill leachate and area stormwater is not fitting with the improved environmental standards the industrial park is trying to adopt. Wright's Brook and its associated ecosystems should be made a conservation priority in this industrial area which is currently undergoing rapid development. Its conservation and protection from further degradation suit municipal covenants to improve the environmental management of the park.

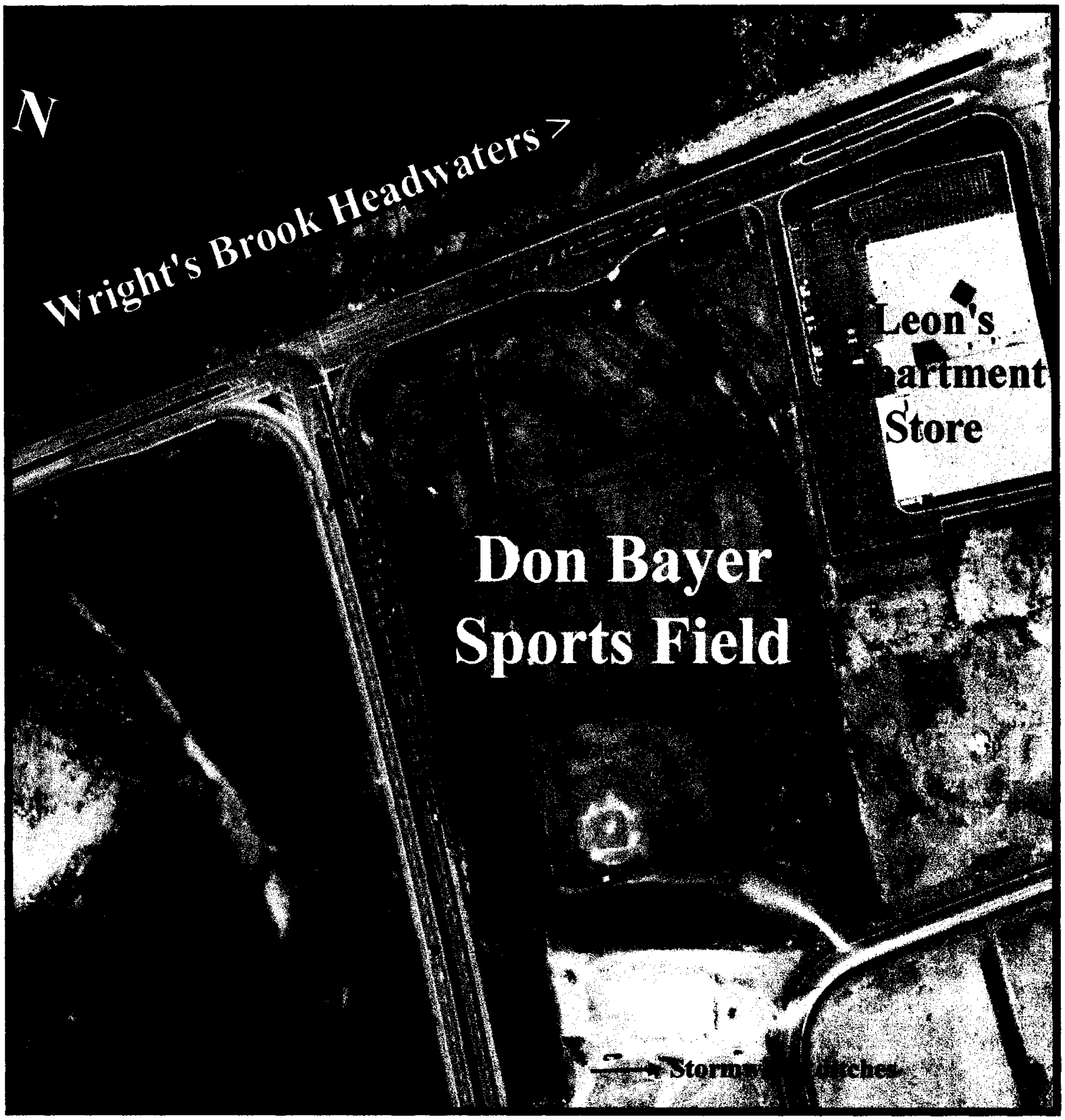


Figure 1.4. Close-up of Don Bayer Sports Field; Site of Former Burnside Drive Municipal Landfill (adapted from HLIS, 1992).

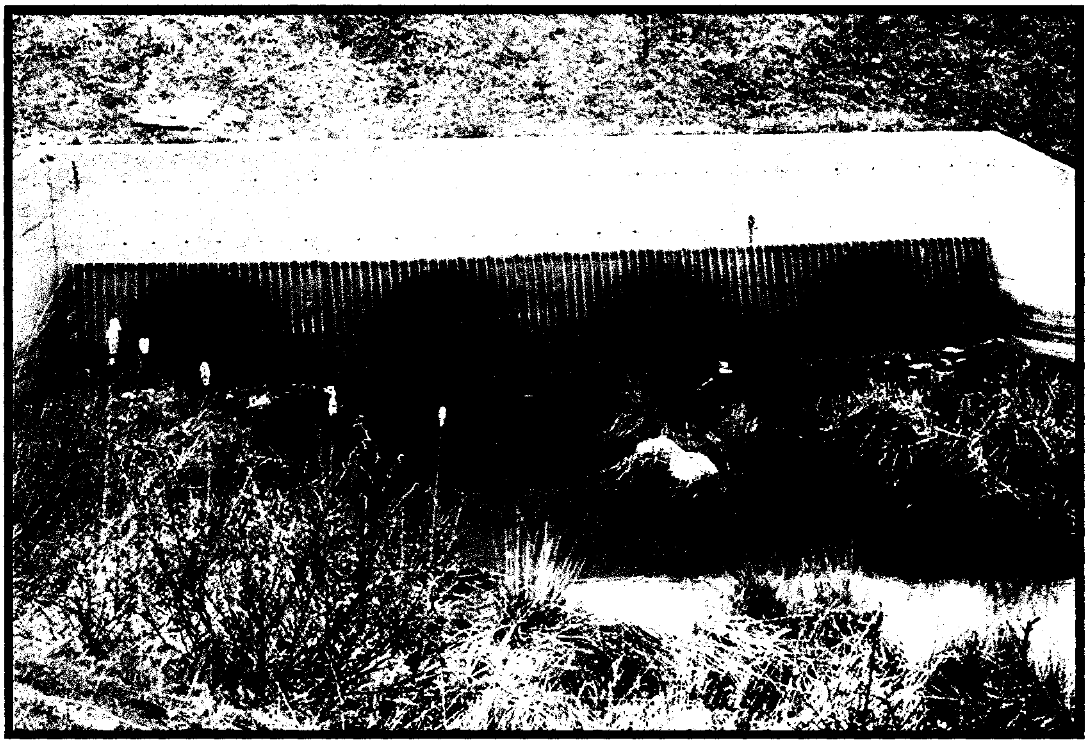


Figure 1.5. Culverts Leading from Don Bayer Sports Field Ditches to Wright's Brook (from Ghaly and Côté, 2001).

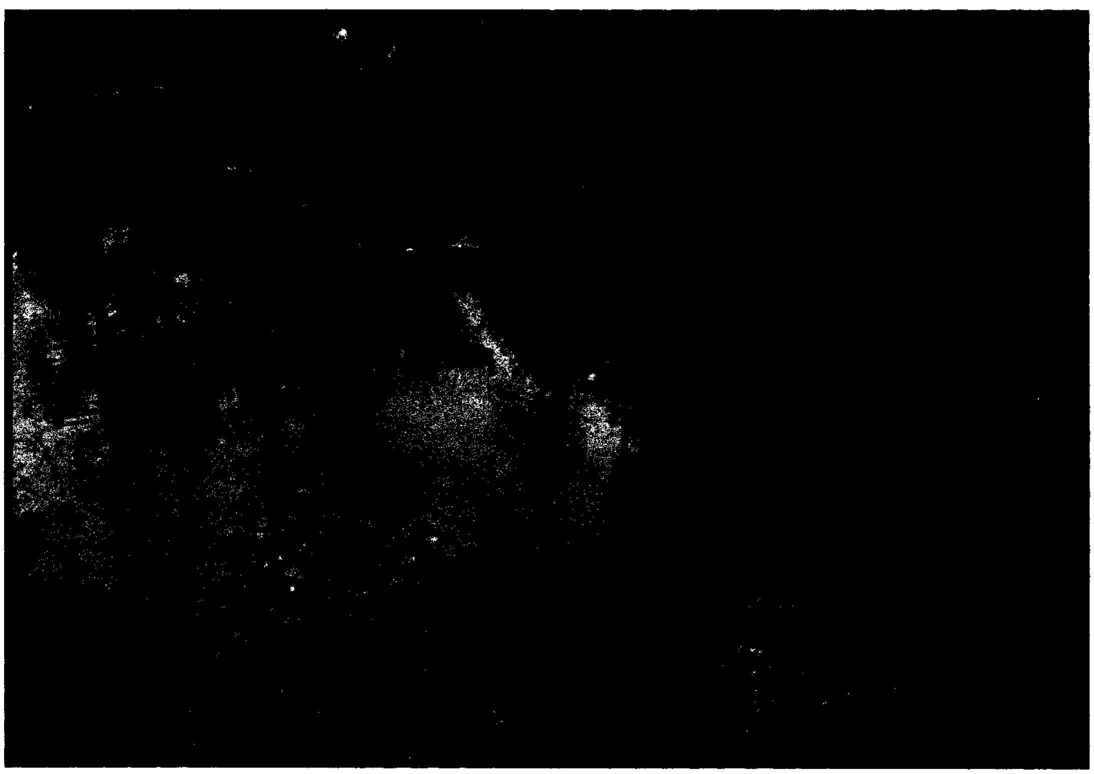


Figure 1.6. Headwaters of Wright's Brook Leading from Culverts (from Ghaly and Côté, 2001).

Table 1.1. Water Quality Results for Samples Taken from the Don Bayer Sports Field Stormwater Ditches in October, 2000 (adapted from Ghaly and Côté, 2001).

PARAMETRE	CONCENTRATION* (μg/L)	GUIDELINE
ELEMENTS		
Aluminum	10.00	5-100[1]
Antimony	0.00	20[3]
Barium	0.00	1000[2]
Beryllium	0.00	11[3]
Bismuth	0.00	NGA
Boron	57.33	200[3]
Cadmium	0.00	0.012[1]
Calcium	43300.00	NGA
Chloride	75370.00	NGA
Chromium	0.67	1-8.9[1]
Cobalt	2.00	NGA
Copper	0.00	2-4[1]
Iron	6166.67	300[1]
Lead	0.00	1-7[1]
Magnesium	4000.00	NGA
Manganese	1800.00	1000-2000[2]
Molybdenum	0.00	73[1]
Nickel	0.00	25-150[1]
Phosphorus	0.00	NGA
Potassium	2100.00	NGA
Selenium	0.00	1[1]
Silver	0.00	0.1[1]
Sodium	41200.00	NGA
Strontium	190.00	NGA
Thallium	0.00	0.8[1]
Titanium	0.00	NGA
Uranium	0.00	NGA
Vanadium	0.00	NGA
Zinc	6.67	30[1]
COMPOUNDS		
Ammonia (as N)	1258.18	NGA
Bicarbonate (as CaCO3)	94281.00	NGA
Carbonate (as CaCO3)	181.00	NGA
Nitrate (as N)	0.00	0.6[1]
Nitrite	0.00	NGA
Nitrate + Nitrite (as N)	0.00	NGA
Ortho Phosphate (as P)	0.00	NGA
Sulfate	6330.00	NGA

*values are the average of four measurements.
[1]Canadian Council of Ministers of the Environment Water Quality Guidelines for the Protection of Freshwater Life.
[2]British Columbia Minister of Environment Lands and Parks Water Quality Guidelines.
[3]Ontario Ministry of Environment Environmental Water Quality Objectives.
NGA = No guideline available.

Table 1.1. Continued.

PARAMETRE	CONCENTRATION* (µg/L)	GUIDELINE
WATER QUALITY PARAMETRES		
Alkalinity (as CaCo3) (mg/L)	95000.00	NGA
Color (TUC)	0.00	NGA
Conductance (uS/cm)	478.67	NGA
Dissolved Organic Carbon (mg/L)	2300.00	NGA
Hardness (as CaCO3) (mg/L)	124670.00	NGA
pH	7.10	<6.5, >9.0[1]
Reactive Silica (as SiO2) (mg/L)	4330.00	NGA
Saturation pH @4C	8.25	NGA
Saturation pH @20C	7.85	NGA
TDS (Calculated) (mg/L)	235330.00	NGA
Turbidity (NTU)	3.30	NGA

*values are the average of four measurements.
[1]Canadian Council of Ministers of the Environment Water Quality Guidelines for the Protection of Freshwater Life.
NGA = No guideline available.

1.3. Burnside Engineered Treatment Wetland

As a direct result of their variable natures, wetlands are difficult ecosystems to accurately define. However, for the most part, wetlands are broadly characterized as saturated land areas supporting aquatic processes as indicated by poorly drained soils, hydrophilic vegetation, and various kinds of biological activity that are adapted to a wet environment (EC, 1987). Wetlands are nature's purifiers, cycling and retaining nutrients, pollutants and sediments through unique, naturally adapted mechanisms which include biogeochemical reactions (i.e. reduction/oxidation transformations), phytoremediation (i.e. plant uptake of contaminants), bioremediation (i.e. microbial degradation), and physical processes (i.e. sedimentation) (Davis, 1995; and USEPA, 1995). Increasingly, these mechanisms have been adapted for use in constructed wetland systems designed and constructed to capitalize on the intrinsic water quality amelioration functions of natural wetlands for human use and benefits. When designed properly, these artificial systems are capable of effectively purifying wastewater using the same processes carried out in natural wetland habitats by vegetation, soils, and their associated microbial assemblages, but do so within a more controlled environment, optimizing water quality functions (Osmund et al., 1995b; and Hammer, 1992).

(a) Iron precipitate

(b) Hydrocarbon sheens

(c) Orange iron staining

(d) Turbid waters with surface scums

Figure 1.7. Evidence of Iron Precipitate and Staining, the Presence of Hydrocarbons and Surface Scums (from Ghaly and Côté, 2001).

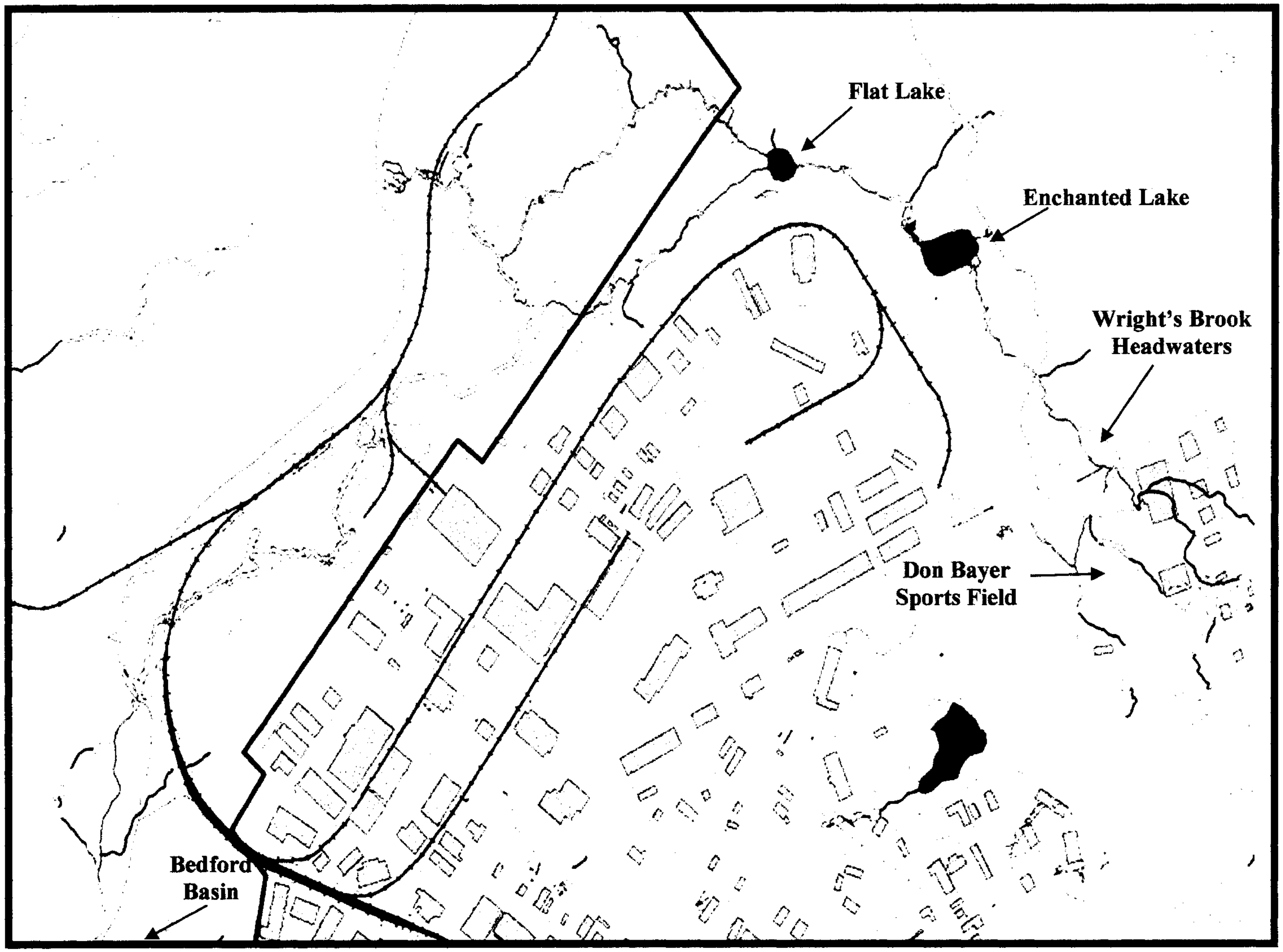


Figure 1.8. Diagram of Wright's Brook (adapted from Ghaly and Côté, 2001).

To remediate the effects of the former Burnside Drive landfill leachate and stormwater input on the natural Wright's Brook ecosystem, a surface flow engineered treatment wetland was designed with the intended purpose of purifying the influent before discharging it back into the impacted brook. The wetland was designed to take on the appearance of a natural marsh by flowing with the contours of the land. The wetland was also designed to curve in a kidney shape (Figure 1.9) in order to increase the length to width ratio to about 5 to 1. In the late fall of 2001 and spring of 2002, a seven celled, deep-water (greater than 1m) system separated by shallow interior earth berms 1 to 2 metres in width was constructed in the marshy area receiving the leachate wastewaters via the culverts beneath Akerley Boulevard, which totalled approximately 5000m^2 in area (Figure 1.10). The first wetland cell (Figure 1.11) was deeper than the others (approximately 1.5m) in order to facilitate the settling and accumulation of suspended solids. The till of the area was found to support 15-25% silt/clay with dense to very dense consistency and a permeability of 10^{-4} to 10^{-6} cm/sec (Lewis et al., 1998). Subsequently, it was concluded that compaction of the area substrates would provide adequate lining for the site. The natural gravitational flow facilitated by the site topography negated the need for any mechanical infrastructure such as pumps that would require regular maintenance. Based on the local maximum monthly (November) rainfall for the period of 1971 to 2000 of 13.09 cm and the watershed area of 55.1 ha, the average flow received by the treatment system was calculated to be 947.1 m^3/day (EC, 2003). The iron loading rate received by the treatment wetland system was calculated to be 1.17 g/m^2 day. Similarly, the manganese loading rate was 0.34 g/m^2 day and the ammonia loading rate was 0.024 g/m^2 day (Table A.1 in Appendix A). In wetlands designed to remove iron from acid mine drainage wastewater, whereas iron loading rates of 2-10 g/m^2 day will result in approximately 90% iron removal, and iron loading rates of 20-40 g/m^2 day can facilitate up to 50% iron removal (Richardson and Nichols, 1985; and Fennessy and Mitsch, 1989). Therefore, the iron loading rate experienced by the treatment wetland was deemed adequate for iron treatment, which is the greatest contaminant of concern. The estimated hydraulic residence time of the treatment wetland system was 5.28 days (Table A.2 in Appendix A). According to Richardson and Nichols (1985) and Fennessy and Mitsch (1989), retention times greater than 24 hrs are best suited to iron removal in constructed wetland systems, hence the retention time experienced by the treatment wetland was also deemed adequate for iron treatment. Similar examples could not be found for ammonia or manganese.

However, preliminary sizing calculations based on the mass balance model determined the optimum wetland area required to treat the Burnside Drive landfill leachate influent would be

5.22 ha for the iron, 0.5 ha for the manganese, and 2.23 ha for the ammonia (Kadlec, 1996). The total treatment area required with peripherals (~25%) was calculated to be 6.53 ha (Table A.3 in Appendix A). Clearly, a 6.53 ha treatment wetland would be very difficult and expensive to construct, especially all in one attempt. Instead, the treatment wetland was constructed with the intention of adding further cells as funding and resources permitted. In addition, Wright's brook is bordered by riparian wetlands all along its length up to Enchanted Lake. Although not as effective as engineered cells, it was concluded that these natural wetlands would cumulatively work to remediate the wastewaters effluent received by the treatment wetland, adding to the total treatment area.

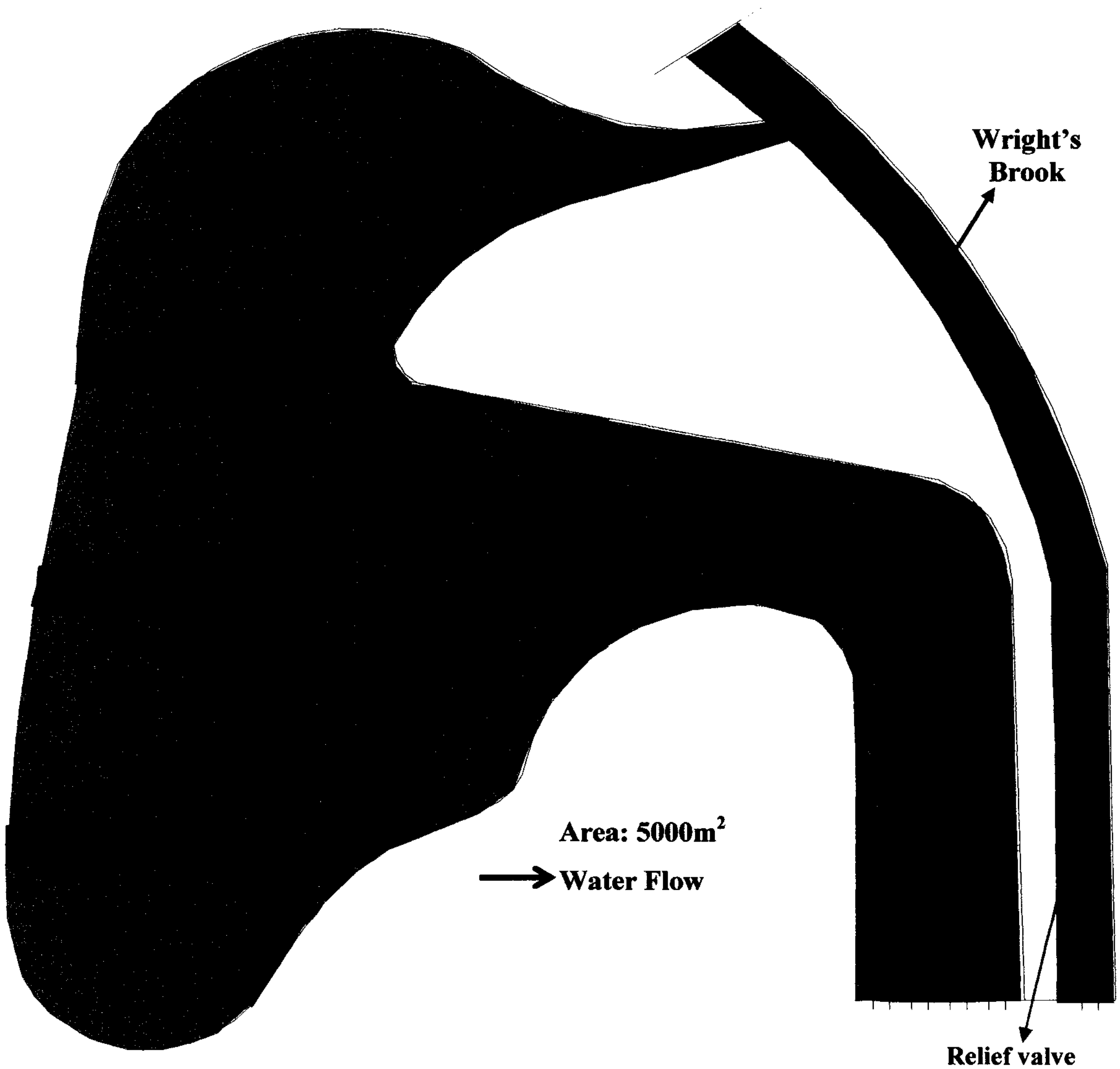


Figure 1.9. Burnside Treatment Wetland Diagram.

Figure 1.10. Photograph of Burnside Wetland Immediately Following Construction, Spring, 2002.

Figure 1.11. Photograph of Cell 1 of Burnside Wetland Immediately Following Construction, Spring, 2002.

Through remediating the effects the leachate and stormwater input, it was hoped that the treatment wetland would help preserve Wright's Brook as a natural ecosystem component of Burnside Industrial Park, as a part of the overall strategy to enhance the environmental management of the park. Although engineered treatment wetlands have become an increasingly popular wastewater management alternative in Canada, it appears that this technique has not been applied in an industrial park setting.

A 24 acre parcel of municipal land, which traversed the length of Wright's Brook from Akerley Boulevard to Enchanted Lake, was made available to the project. The land, which consists of untamed, early successional brush, mature softwood stands and riparian marsh and bog environments, will be shaped into a naturalized ecopark which will be trailed and supplied with benches and picnic tables, and interpretive signage (Figure 1.12). It is intended that the future eco-park will provide peaceful refuge to employees and residents of the area, as well as provide useful educational platforms for environmental learning, with the treatment wetland as well as the natural features of the park serving as its fundamental focus. It is hoped that this site will function as a community model for future projects, and will serve to substantiate that problems need not necessarily be tackled with concrete and artificial infrastructure in order to be remedied effectively. It is also hoped that the site will help cast a new light and instil a new respect for nature in predominately industrial areas. It is anticipated that the project will serve as a sustainable wastewater management demonstration site catering to public and private interests, as well as to educators interested in the subject (Ghaly and Côté, 2001).

Eco-Park 3 Year Development Plan

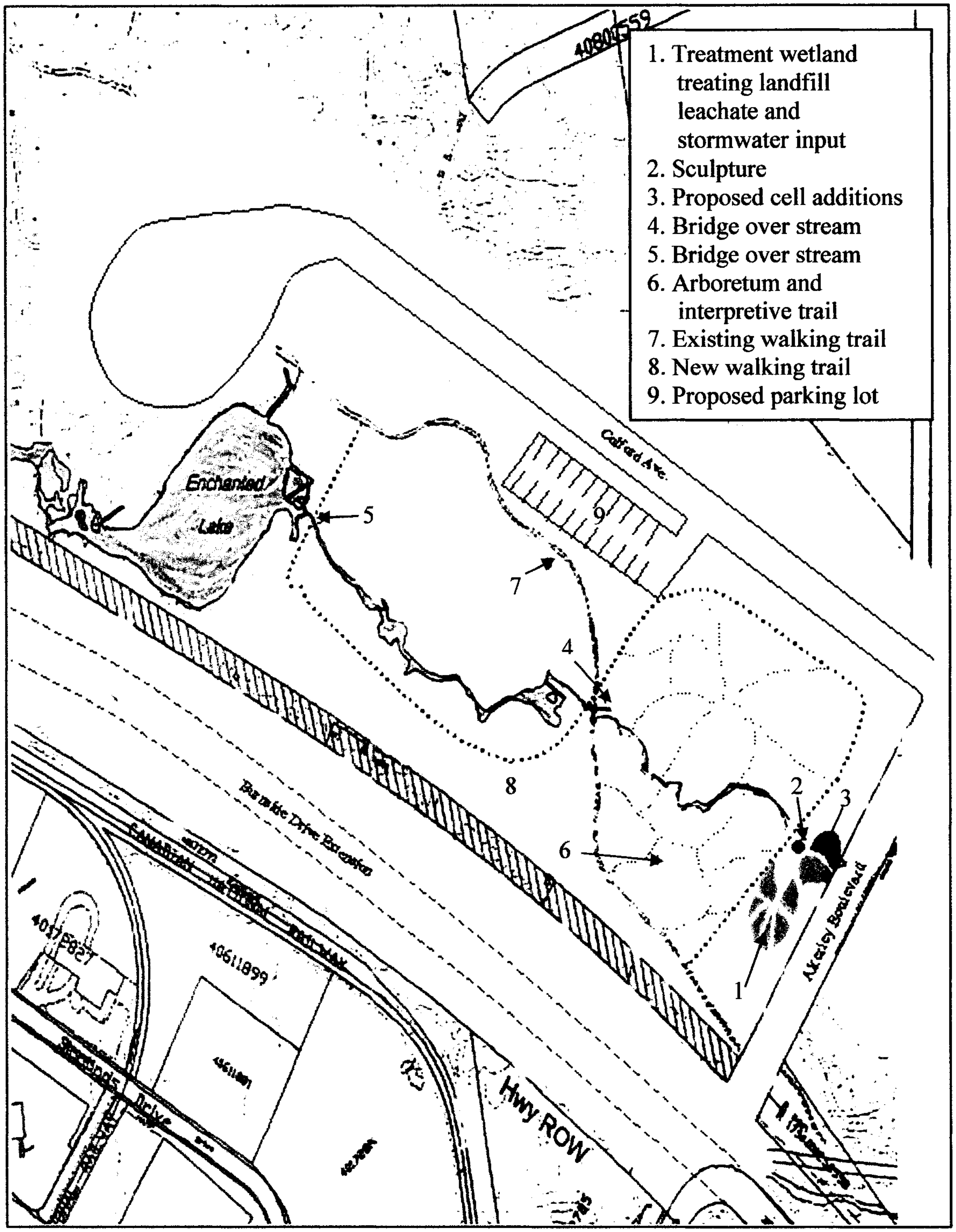


Figure 1.12. Site Plan for the Burnside Treatment Wetland and Adjoining Ecopark.

2. OBJECTIVES

Wetland vegetation is an extremely important component of any treatment wetland system. Not only do wetland plants work to reduce contaminant levels via phytoremediation processes, but also facilitate contaminant remediation and immobilization by supporting processes such as bioremediation (by facilitating microbial growth), sedimentation (by stabilizing sediments and filtering and trapping suspended solids), and biogeochemical reactions (by working to oxygenate the water column and sediments). Therefore, an outdoor treatment wetland should be planted with a diverse arrangement of native, non-aggressive plant species. This is because naturalized systems are more likely to: (a) be self-sustaining, (b) be more adapted to the local environment (i.e. climate, soils, pests etc.) and therefore more resistant and resilient to regionally common disturbances, (c) provide suitable wildlife habitat for local wildlife, and (d) help maintain the ecological health and native biodiversity of natural systems in the region by eliminating the potential threat of biological pollution. According to Daigle and Havinga (1996), biological pollution occurs when exotic and aggressive species spread to natural systems and out-compete native vegetation, causing habitat degradation.

The aim of this study was to establish a diverse, self-sustaining, locally-modelled, native vegetative community bearing biological integrity in the Burnside treatment wetland site that effectively decontaminates the leachate and stormwater input via phytoremediative, physiochemical and biophysical means. The hypothesis of this study is that "the selected native vegetation and vegetation establishment strategy will yield a successfully established site bearing biological integrity and that a naturalized system supporting biological integrity will effectively remediate the characterized contaminated leachate and stormwater input".

The specific objectives of this study were to:

(a) select the appropriate native vegetation for the site,
(b) select the appropriate vegetation establishment strategy for the site,
(c) establish plants in both the wetland and wetland buffer areas,
(d) evaluate the plant establishment success of site following one growing season,
(e) evaluate the biological integrity of the site following one growing season, and
(f) evaluate the water purification ability of the site following one growing season.

3. LITERATURE REVIEW

3.1. Defining Wetlands

Official definitions of the term "wetland" were not attempted until 1970s, as up until then, wetlands were broadly perceived as valueless wastelands. This lack of understanding and appreciation of wetland functions made these valuable ecosystems vulnerable to abuses and their considerable global depletion. It was not until the 1970s when wetland benefits were finally recognised and translated into law which was formed to help prevent further wetland loss. However, formulating regulations proved more difficult than originally surmised, as defining these diverse ecosystems is an extremely complicated task. To this day, wetland definitions remain unclear even amongst scientists and most definitions are often confusing and contradictory. This is due to the fact that these global land features, which occur in every country on the planet, are so variable by nature as they are a halfway world between terrestrial and aquatic ecosystems but exhibit some characteristics of each (Mitsch and Gosselink, 2000). Because they are a diverse phenomenon with many forms, a definition must adequately encompasses the features that typify them.

The International Ramsar Convention on Wetlands, an intergovernmental treaty which was signed in Ramsar, Iran in 1971, provides the framework for national action and international cooperation for the conservation and wise use of wetlands and their resources (Ramsar Convention Bureau, 2001). It has 133 Contracting Parties to the Convention, and has designated 1198 wetland sites, totalling 103.4 million hectares, as 'Wetlands of International Importance' (Ramsar Convention Bureau, 2002a). With such a strong agenda, one would think that the Parties to the Convention would have surmised a rather lengthy and technical definition of the term wetland. However, the Convention has adopted the following, rather ambiguous definition: "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish, or salt including areas of marine water, the depth of which at low tide does not exceed six metres".

The following official definition of a wetland by Environment Canada's National Wetlands Working Group (EC, 1987) is also ambiguous: "land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation, and various kinds of biological activity that are adapted to a wet environment".

The United States government was prompted to create a slightly more specific definition due to regulatory concerns surrounding policy for wetland protection. The following definition is used by the EPA and the U.S. Army Corps of Engineers (Corps) in implementing section 404 of the Clean Water Act: "those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs and similar areas" (Mitsch and Gosselink, 2000).

However, the most commonly used and accepted American definition of wetlands adopted by the U.S. Fish and Wildlife Service since 1979 is: "lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water and must have one or more of the following three attributes: (a) at least periodically, the land supports predominantly hydrophytes, (b) the substrate is predominantly undrained hydric soil, and (c) the substrate is non-soil and is saturated with water or covered by shallow water at some time during the growing season of each year" (Cowardin et al., 1979).

The vague nature of most definitions generally assigned to wetlands proves to be problematic as land features which are obviously not wetlands such as small creeks or lakes, or eutrophified, concrete, and man-made canals, can easily fall within the these broad definitions. However, either an abundance of water-borne plants (hydrophytes) or indicators of hydric soil conditions are generally sufficient enough to most to indicate that a system is indeed a wetland, with boundaries delineated by changes in vegetation structure and soil characteristics (Banner and MacKenzie, 2000).

The lack of fine-tuning of the definition of wetlands has made it rather difficult for scientists to estimate just how many of the unique land features occupy our planet. The exact area covered by wetlands globally remains relatively unknown, however, estimates by the World Conservation Monitoring Centre (WCMC) have wetlands covering approximately 6% of the Earth's land surface, or roughly 5.7 million km^2. The WCMC further divides that estimate into relative areas occupied by the different wetland classes. Of the 5.7 million km^2, 2% are marshes, 3% are bogs, 26% are fens, 20% are swamps, and 15% are floodplains (Ramsar Convention Bureau, 2001).

Canada supports over 127 million hectares of wetland environments, which is approximately 14% of Canada's total land area. Their distribution across the country varies greatly, with most wetlands being situated in Manitoba, Ontario and the Northwest Territories (NRC, 2002). The specific land area occupied by wetland environments in each province is shown in Table 3.1. Although appearing significant in area, it is important to note that pressures from agriculture, urban and industrial development, and other anthropocentric sources, have resulted in drastic declines in total Canadian wetland area, as well as dramatic global declines of these unique and important ecosystems.

Table 3.1. Distribution of Wetlands in Canada (adapted from NRC, 2002).

PROVINCE OR TERRITORY	WETLAND AREA IN HECTARES	PERCENT OF AREA IN PROVINCE OR TERRITORY	PERCENT OF TOTAL CANADIAN WETLANDS
British Columbia	3 120 000	3	2
Alberta	13 704 000	21	11
Saskatchewan	9 687 000	17	8
Manitoba	22 470 000	41	18
Ontario	29 241 000	33	23
Quebec	12 151 000	9	10
New Brunswick	544 000	8	<1
Nova Scotia	177 000	3	<1
Prince Edward Island	4 000	<1	<
Newfoundland and Labrador	6 792 000	18	5
Northwest Territories	27 794 000	9	1
Yukon Territory	1 510 000	3	22
CANADA	127 199 000	14	100

3.2. Classifying Natural Wetlands

Wetland ecosystems are variable in nature and support various different hydrological and biological regimes. One way wetlands can be further classified is based on their location. Fluvial, Riverine or Riparian wetlands are those freshwater wetlands which border rivers and streams. Shore or Lacustrine wetlands are those which border lakes. Freshwater wetlands which are isolated or lacking influence from hydrological features such as oceans, rivers or lakes are referred to as Terminal Basin or Palustrine systems. Marine systems are those wetlands which occur along coastlines. Estuarine wetlands are those which are located in sheltered coastal areas where fresh and salt waters mix, such as salt marshes and mangrove swamps (Ferren et al., 1996).

The problem with this classification system is the large amount of overlap existing between the terms, making identification difficult.

The more commonly used method of wetland classification places emphasis on a system's hydrological status rather than location. The hydrodynamics of a site, which is defined as the degree of vertical water table fluctuation and rate of lateral groundwater flow, is typically the limiting factor influencing all features in a given wetland. Sites with relatively stable water regimes tend to accumulate peat. Peat-accumulating systems are typically classified as either bogs or fens, or more generally, as peatlands. Sites with more active water regimes are more likely to support decomposition of organic matter, support greater aeration, and have greater nutrient availability. These systems generally support either marshes or swamps (Banner and MacKenzie, 2000). Figure 3.1 shows the environmental gradient for these classes of wetlands.

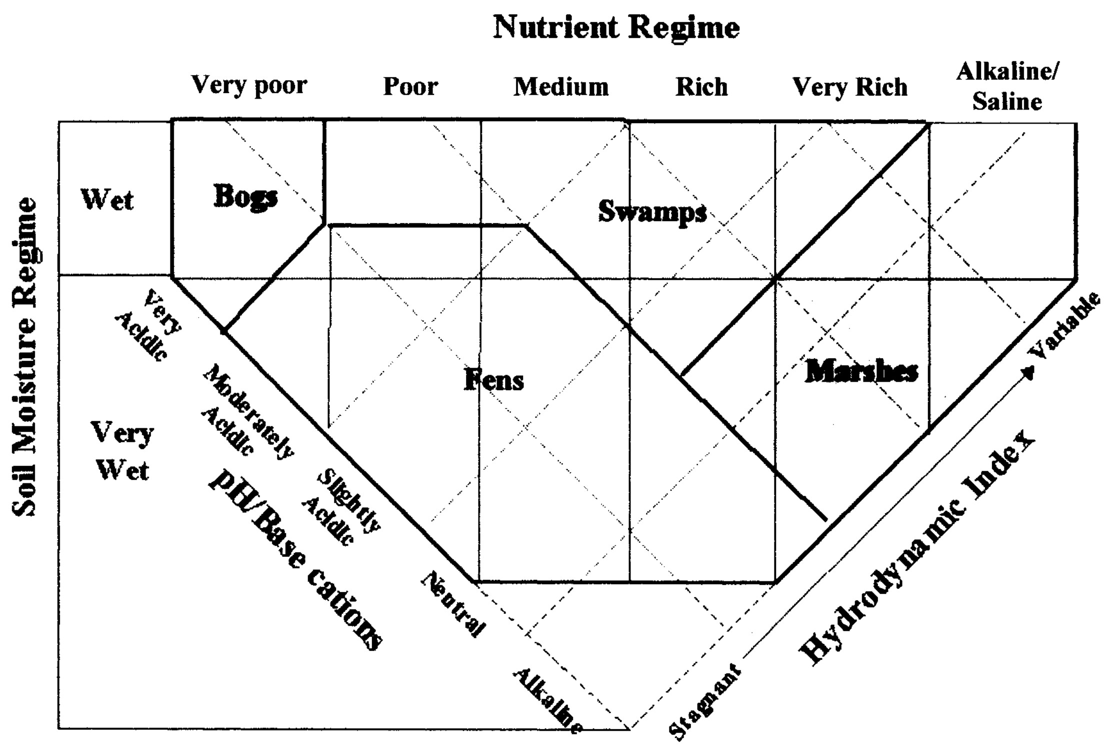


Figure 3.1. Marsh Wetland Environmental Gradients (adapted from Banner and MacKenzie, 2000).

3.2.1. Marshes

Marshes are wetlands which are frequently or continually inundated with (standing or inflowing) water (USEPA, 2002b). They are found in a variety of forms in Canada, including tidal marshes, freshwater marshes, wet meadows, and prairie potholes. All of these marsh subclasses are surface water fed. However, many also receive groundwater input as well. Most marshes support mineral soils and do not accumulate peat. Mineral soils are soils which typically contain less than 20 to 35% organic matter, bear a neutral pH, have high nutrient availability and low porosity. These properties make mineral soils particularly well-suited to supporting abundant and diverse plant and animal life (USEPA, 2002c). Marsh plant life is typically dominated by non-woody, emergent aquatic macrophytes with extensive root and rhizome systems that are morphologically adapted to saturated soil conditions. Oxygen transported through the root systems can then leak from the roots into the soils immediately surrounding them (known as the rhizosphere), creating oxidized conditions which support the decomposition of organic matter and the growth of aerobic microorganisms which would not otherwise occur in the anoxic conditions typical of saturated soils (Brix, 1993). Specific macrophytes which are typical of this class of wetland are cattails, rushes and pondweeds. Marshes are notably biologically productive, not only supporting abundant plant life but also providing critical habitat for a broad array of wildlife, including waterfowl, aquatic mammals, and a wide variety of amphibians, reptiles and aquatic insects (DNR, 1998).

3.2.1.1. Tidal Marshes. Tidal marshes, also known as coastal marshes, are wetland ecosystems which typically occur along coastlines. They are periodically or permanently inundated by shallow saline water, and contain mineral soils with little or no peat deposition. The hydrology of these type of marshes is highly influenced by the motion of oceanic tides. However, these systems can also be influenced by freshwater from runoff, rivers, or groundwater and many which are located upstream of estuaries are actually classified as freshwater systems (Tiner, 1999; and USEPA, 2002c).

Saline tidal marshes, or salt marshes, are typically covered by water at irregular intervals in accordance with the tides. There are two cycles of high and low tide daily, hence, salt marshes are continually in a state of change. This has resulted in unique wildlife adaptations. Despite being unstable, these systems support a diverse range of bird and fish species. Salt marshes have extremely high rates of primary productivity due to the massive inflow of nutrients and organics from surface and/or tidal water inputs. These systems are typically characterized by salt-tolerant plants such as smooth cordgrass, saltgrass, spike grass and glasswort and black grass. Coastal marshes which contain brackish or freshwater are typically dominated by cattail, wild rice, pickerelweed, and arrowhead. Brackish and fresh tidal marshes tend to have a greater variety of plant life than saline marshes due to their stable nature. Both saline and non-saline tidal marshes are known to be of particular importance to several species of migratory waterfowl (USEPA, 2002c; Mitsch and Gosselink, 2000; and Lyon, 1993).

Tidal marshes are extremely valuable ecosystems, not only for the habitat and wildlife diversity they support, but for anthropocentrically based reasons as well. Tidal marshes effectively buffer land from stormy waters, protect shorelines from erosion, and remove/immobilize terrestrial contaminates destined for ocean waters. Despite widespread recognition of their benefits, costal development projects as well as urban and rural pollution continue to threaten these unique ecosystems (USEPA, 2002c; and Tiner, 1999).

3.2.1.2. Freshwater Marshes. A freshwater marsh is a nutrient-rich wetland system which is usually permanently inundated by fresh water, although some may be brackish. Water levels typically vary from a few decimetres to over a metre in depth. These marshes often form on the edges of freshwater systems such as lakes or streams, and are dominated by emergent aquatic plants such as cattails, rushes, reeds and sedges and floating plants and submerged plants such as water lily, pondweed and duckweed. Freshwater marshes are also characterized by highly organic, mineral rich soils which are non-specific in texture class as their composition of sand, silt, and clay can be highly variable from site to site. As a result of their high nutrient levels, these ecosystems are considered to be one of the most productive on earth, they are up to three times as productive as agricultural land and four times as productive as lakes and streams. This high level of productivity provides ideal habitat for a wide variety of wildlife, especially waterfowl and songbirds (USEPA, 2002c; Lyon, 1993; and DNR, 1998).

Freshwater marshes also provide a large variety of anthropocentric services such as flood mitigation and contaminant filtration, however, they too continue to degrade or be lost to human-related pollution and development (USEPA, 2002c, d; and Mitsch and Gosselink, 2000).

3.2.1.3. Wet Meadows. Wet meadows are considered freshwater marshes as they contain many of the same defining characteristics of marsh systems such as highly fertile soils. However, unlike typical marshes, wet meadows contain standing water only during the wet season and stay relatively dry and during the remainder of the year. However, high water tables typically result in wet meadow soils remaining relatively saturated. Wet prairies are often confused with wet meadows as the two are very similar. However, wet meadows are known to remain saturated longer and wet prairies may receive water input from sources other than precipitation. The vegetative community supported by these systems characteristically resembles that of a grassland ecosystem, supporting a variety of grasses and sedges, as well as rushes and facultative wildflowers. The vegetation combined with the high nutrient availability of the soils provides for quality habitat for many insects, herptiles, birds and mammals (USEPA, 2002c; Lyon, 1993; and Tiner, 1999).

Wet meadows usually occur in areas which are poorly drained such as low-lying farmland, and are therefore commonly victim to draining and filling as a means of increasing the agricultural productivity of a given farmer's lands. This is unfortunate as these systems effectively collect runoff during rain events, and as a result, provide for valuable flood control. Waters which collect in these systems are then filtered and cleansed of excess nutrients and pollutants via settling, phytoremediation and bioremediation processes. These important systems are now receiving some attention in terms of conservation (USEPA, 2002d; c and Mitsch and Gosselink, 2000).

3.2.1.4 Prairie Potholes. Prairie potholes, also known as sloughs, are unique type of freshwater marsh which can be found scattered about formerly glaciated landscapes such as those found in the upper Midwest potions of the Canada and the U.S. The glacial retreat which occurred in these regions approximately 12,000 years ago left the landscape scattered with scores of differently shaped and sized depressions, which evolved into wetland ecosystems. Some of these systems are only seasonally inundated, while others may contain permanent standing water. Like wet meadows, prairie potholes detain and purify runoff thereby reducing flood and contamination risks to systems downstream. Unfortunately, these systems are also vulnerable to drainage for agricultural gains. About 40 to 50% of the American upper Midwest regions prairie pothole wetlands have fallen victim to human-induced drainage (USEPA, 2002c; Lyon, 1993; and Tiner, 1999).

These systems contain diverse arrays of vegetation, from submerged and floating species in the deeper central portions of the systems to rushes, bulrushes and cattails forming in the shallower littoral zones to grasses and sedges dominating the systems not containing standing water. Prairie potholes are extremely important in terms of habitat value, as over half of the migratory waterfowl in North America depend on prairie potholes for their survival and reproduction. As a result, these areas are commonly utilized for birding and hunting activities (USEPA, 2002d; a; and c).

3.2.2. Swamps

Swamps, which can occur in either freshwater or saltwater floodplains, characteristically contain highly organic, nutrient-rich, black mineral soils which are saturated throughout the year. Standing water can also be supported in these systems during wet seasons. However, swamps typically tend to be more waterlogged than wet. Swamps are the only type of wetland dominated by water-tolerant woody plants such black spruce and alder. Mosses, grasses and ferns also tend to inhabitant these damp systems. The nutrient-rich state of these wetlands provide excellent habitat for frogs, salamanders, songbirds and mammals such as deer and snowshoe hare (DNR, 1998). Like marshes, they also serve vital roles in flood protection and nutrient removal, and also suffer threat of drainage from agriculture, as their nutrient-rich soils make for ideal farmland (USEPA, 2002c). Two general types of swamp exist in Canada, namely shrub swamps and forested swamps.

3.2.2.1. Shrub Swamps. Shrub swamps are wetlands which are dominated by shrubby vegetation such as alder, sweet gale and rhodora. Ground cover species typically consist of various mosses and ferns, as well as grasses and sedges. Shrub swamps tend to be wetter than forested swamps, with standing water present for much of the year. Shrub swamps, which are mostly found along slow moving streams and in floodplains, tend to evolve into wooded swamps over time (DNR, 1998; and Lyon, 1993).

3.2.2.2. Forested Swamps. Forested swamps, also known as wooded swamps, are characteristically dominated by large woody plants such as red maple, black spruce, balsam fir and tamarack (larch), which are less tolerant of the highly saturated soils of shrub swamps. Like shrub swamps, the hydrological conditions of forested swamps typically arise as a result of floodwater input from nearby rivers and streams. As a result, forested and shrub swamps are frequently found neighbouring one another (DNR, 1998).

3.2.3. Bogs

Bogs are very distinctive wetlands and easily classified with little problem. They are generally associated with the northern hemisphere (especially Canada and Russia) as they develop in areas with low temperatures, short growing seasons, and ample precipitation. In fact, Canada has more bogs and fens than any other country on the planet, covering about 12% of the land area (USEPA, 2002c; and Banner and MacKenzie, 2000). Unlike swamps and marshes, bogs are poorly drained, nutrient deficient, acidic freshwater systems that are characterized by spongy acid-forming peat deposits and thick sphagnum moss floors. Peat is partially decayed organic matter which slowly accumulates via a process referred to as humification. The decomposition rate is restricted due to the lack of oxygen present in the anoxic, saturated conditions provided in the bog ecosystem. Oxygen is needed to support microorganisms responsible for decomposition processes. The growth of peat occurs when rates of decomposition are less than rates of accumulation of dead vegetation. Dissolved organic compounds (DOCs) known as fulvic acids readily leach from bog ecosystems, particularly from the peat formations. This results in tea coloured staining and natural acidity levels which are typically a pH of around 4.5 (NAP, 1995; and Lyon, 1993).

The presence of the sphagnum also contributes to the acidity of bog waters as it actively removes basic minerals from soil water, exchanging them with hydrogen ions, thereby increasing acidity. Bogs also contain specialized coniferous trees (such as black spruce), shrubs (such as leatherleaf) and other plants (such as the carnivorous sundew), which are adapted to low nutrient levels, waterlogged conditions, and the acidic waters. These plants are of considerable interest to biologists as acidic conditions usually inhibit the ability of most plants to take up nutrients. Bogs receive no water input other than that from precipitation, hence their nutrient levels are low. This nutrient-deficient environment results in autogenic succession, where plant litter is decomposed very slowly. This, occurring over thousands of years, yields the characteristic peat deposits which exist within bogs at depths varying from less than 50 cm over 10 m depending on climate, terrain, and hydrology. Bogs can develop overtime by two different ways: terrestrialization and paludification. Terrestrialization occurs when sphagnum mosses grow over a lake or pond, slowly filling it. Paludification occurs when sphagnum mosses grow over dry land, preventing water from leaving the surface. Bogs tend to develop in shallow, low-lying areas above layers of bedrock (USEPA, 2002c; Banner and MacKenzie, 2000; and DNR, 1998).

Like marshes and swamps, bogs also help to prevent upstream flooding and contamination by absorbing runoff. However, they have suffered much abuse in the past. In addition to being drained for use as cropland, peat was once considered an effective fuel source and is still considered a valuable soil conditioner and mined as a result. Peat, which had required hundreds, if not thousands of years to form naturally, can be harvested in a matter of days, leaving a bog ecosystem devastated. Reclamation of these systems proves extremely difficult as they are centuries in the making. These unique systems continue to suffer degradation and loss even as they are being recognised for their valuable contribution to global warming abatement by acting as substantial carbon sinks (USEPA, 2002d,c).

3.2.4. Fens

Fens also develop where hydrological gradients are less dynamic. Like bogs, they are relatively exclusive to the northern hemisphere. Fens contain very similar vegetative communities to that of bogs including the characteristic sphagnum peat deposits. The difference, however, between them is that fens are influenced by groundwater and surface water inputs. This results in decreased nutrient deficiency and acidity, allowing for less sensitive vegetation such as rushes, sedges and grasses to establish themselves. Wildlife that will make use of fens include brown bullheads (catfish), pickerel frogs, and porcupine. One of Nova Scotia's species at risk, the Blandings turtle, is notably a resident of fens. Overall, fens are still relatively nutrient-low habitats as compared to marshes and swamps (DNR, 1998; and Banner and MacKenzie, 2000).

Over time, a fen may build up so much peat that it acts as a physical barrier to the fen's groundwater supply. With its nutrient source depleted, a fen can consequently turn into a bog. Fens, like bogs, provide flood reduction/prevention, improve water quality, and provide habitat for unique plant and animal communities. They have also experienced a decline mostly due to draining for cropland and mining of peat for fuel and fertilizer. These increasing rare ecosystems also take up to 10,000 years to form naturally, and are not easily replaced (USEPA, 2002c,d; and Mitsch and Gosselink, 2000).

3.3. Constructed Wetlands

Constructed wetlands are becoming increasingly common features emerging in landscapes across the globe. Although similar in appearance to natural wetland systems (especially marsh ecosystems), they are usually created in areas that would not naturally support such systems to facilitate contaminant or pollution removal from wastewater or runoff (Hammer, 1992; and Mitsch and Gosselink, 2000). In essence, constructed wetlands are designed and constructed to capitalize on the intrinsic water quality amelioration functions of natural wetlands for human use and benefits. Fields (1993) stated that constructed wetlands are built specifically for water quality improvement purposes, typically involving controlled outflow and a design that maximizes certain treatment functions. Wetland creation can also encompass wetland enhancement, in which a natural wetland is enhanced (i.e. via diversion of flow, the creation of berms, etc.) to provide one or more additional functions (Mitsch and Gosselink, 2000; and Osmund et al., 1995a).

When designed properly, constructed wetlands are capable of effectively purifying wastewater using the same processes carried out in natural wetland habitats by vegetation, soils, and their associated microbial assemblages, but do so within a more controlled environment (Osmund et al., 1995b; and Hammer, 1992). The specific water quality treatment mechanisms emulated include gravitational settling of suspended matter, the facilitation of chemical transformations, and the facilitation of bioremediation and phytoremediation processes (Davis, 1995; and USEPA, 1998b).

The application of this type of biomimicry is not a new idea, as constructed wetlands were frequently utilised for the purpose of pollution abatement by ancient Chinese and Egyptian cultures (Brix, 1993). European experimentation with phytoremediation techniques began in the early 1950s. The use of treatment wetlands subsequently began to grow increasingly popular. Denmark, Germany and the United Kingdom currently have approximately 600 established sewage treatment wetlands between them (Brix, 1993). In fact, treatment wetlands can now be found on every continent on the planet except for Antarctica (Tousignant et al., 1999).

Constructed treatment wetlands are being used to treat a variety of wastewaters (which can be either pumped or gravity-fed), including municipal wastewater, stormwater runoff, agricultural runoff, mining drainage, landfill leachate, industrial wastewaters, ammunition, and radionuclides as shown in Table 3.2. (Davis, 1995). Their increasing popularity can be attributed to several benefits: (a) they are capable of providing removal rates ranging from 60% to over 95% for many pollutants (Tousignant et al., 1999), (b) they are much less costly to build and operate than other conventional treatment facilities as construction of treatment wetland areas costs anywhere from one half to one tenth the cost of building conventional treatment facilities (Hammer, 1992), (c) they provide important functions such as habitat enhancement, and (d) they are a less intrusive and provide more environmentally sensitive approach to pollution abatement (Davis, 1995).

Some of the disadvantages include: (a) they generally require larger land areas than conventional wastewater treatment systems, which can prove disadvantageous in regions where land is scarce or costly, (b) bioremediation and phytoremediation processes require more time than conventional treatments, (c) monitoring can sometimes prove difficult, and (d) the reliability of

Table 3.2. Common Constructed Wetland System Applications.

ACTIVITY	ASSOCIATED POLLUTANT	REMOVAL MECHANISMS	EFFICIENCY	TYPE/USE STATUS
Agriculture (i.e. faecal wastes, fertilizers, erosion, pesticides)	-nitrogen	-bioremediation (esp. denitrification, nitrification,), phytoremediation, volatilization[bde]	-25 to 95%[a,c]	FWS/ Common, Widespread
	-phosphorus	-soil sorption (greatest is soils high in Al)[f], phytoremediation [b,d,e]	-30 to 90%[a,f]	
	-suspended solids	-sedimentation, filtration, phytoremediation [b,d,e]	-35 to 90% [f]	
	-pesticides	-bioremediation, phytoremediation, volatilization, photolysis [d]	- variable depending on pesticide	
	-pathogens (parasites, bacteria and viruses)	-sedimentation, filtration, natural die-off, predation, and UV degradation [d,f]	-80 to 90%[d]	
Municipal Wastewater (sewage effluent)	-BOD	-decomposition of organics, oxidation of inorganics	-up to 100%[f]	SSF (tertiary treatment) & FWS (polishing ponds)/ Currently Limited use, but growing quickly, esp. in Europe, Australia, and New Zealand[g]
	-nitrogen	-bioremediation (esp. denitrificaton, nitrification,), phytoremediation, ammonia volatilization [b,d,e]	-25 to 95%[a,c]	
	-phosphorus	-soil sorption (greatest is soils high in Al)f, phytoremediation [b,d,e]	-30 to 90%[a,f]	
	-suspended solids	-sedimentation, filtration, phytoremediation [b,d,e]	-35 to 90% [f]	
	-pathogens	-sedimentation, filtration, natural die-off, predation, and UV degradation [d,f]	-80 to 90 % [d]	

[a]USEPA, 1993 [b]Liehr et al., 2000 [c]Hammer, 1992 [d]Tousignant et al., 1999
[e]Diekelmann and Schuster, 1982 [f]Osmund et al., 1995c [g]Mitsch and Gosselink, 2000

Table 3.2. Continued.

ACTIVITY	ASSOCIATED POLLUTANT	REMOVAL MECHANISMS	EFFICIENCY	TYPE/USE STATUS
Stormwater (road, roof, lawn, greenspace drainage, etc.)	-suspended solids	-sedimentation, filtration, phytoremediation [b d e]	-80 to 90 %[f]	FWS/ Common, Widespread
	-nitrogen (i.e. fertilizers, waterfowl/ pet faecal matter)	-bioremediation (esp. denitrificaton, nitrification,), phytoremediation, ammonia volatilization[b d e]	-25 to 95%[a c]	
	-phosphorus (i.e. car washing detergents)	-soil sorption (greatest is soils high in Al)[f], phytoremediation [b d]	-30 to 90%[a f]	
	-pesticides	-bioremediation, phytoremediation, volatilization, photolysis [d]	- variable depending on pesticide	
	-roadway pollutants (salts, oils, heavy metals)	-bioremediation, phytoremediation, trapping	-highly variable, depending on contaminant[g]	
Mining Effluent (i.e. acid mine drainage)	-metals	-oxidation/reduction reactions yielding precipitation, complexation with organic matter, filtration, phytoremediation[f]	-up to 99% dissolved Al, Cd, Cu, Cr, Zn[h], up to 99% Fe, 94% Pb, 84% Ni, 9 to 44% Mn[i]	FWS/ Relatively common, over 400 systems in U.S. alone[j]
	-sulphate	-oxidation/reduction reactions, bioremediation[f]	-variable	
	-low pH	-added limestone or other calcium source[f]		

[a]USEPA, 1993 [b]Liehr et al., 2000 [c]Hammer, 1992 [d]Tousignant et al., 1999 [e]Diekelmann and Schuster, 1982 [f]Osmund et al., 1995c [g]Wren et al., 1997 [h]Schnoor, 2002 [i]Lorion, 2001 [j]Mitsch and Gosselink, 2000.

Table 3.2. Continued.

ACTIVITY	ASSOCIATED POLLUTANT	REMOVAL MECHANISMS	EFFICIENCY	TYPE/USE STATUS
Radio-nuclides	90Sr, 137Cs, 239Pu, 238,234U	-phytoextraction, rhizofiltrationg	-n/a	Laboratory, pilot, and field applications[d]
Landfill Leachate	-metals	-oxidation/reduction reactions yielding precipitation, complexation with organic matter, filtration, phytoremediation[f]	-up to 99% dissolved Al, Cd, Cu, Cr, Zn[e], up to 99% Fe, 94% Pb, 84% Ni, 9 to 44% Mn[i] -18-70%[h]	FWS/ Recent application[c]
	-ammonia	-oxidation/reduction reactions, bioremediation[f]	-25 to 95% [a b]	
Explosives and Ammunition	-RDX3, TNT	-degradation by enzymes, phytoremediation	-n/a	SSF/ Experimental stages

[a]USEPA, 1993 [b]Hammer, 1992 [c]Osmund et al., 1995c [d]Schnoor, 2002 [e]Schnoor, 2002
[f]Lorion, 2001 [g]USEPA, 2000c [h]Demchik and Garbutt, 1999

wetland treatment systems can often be less consistent than that of traditional treatment systems, as nature, can be unpredictable and external factors such as weather and pests often cause sites to display inconsistent contaminant removal rates (FRTR 1998; CPEO, 1998; and Davis, 1998).

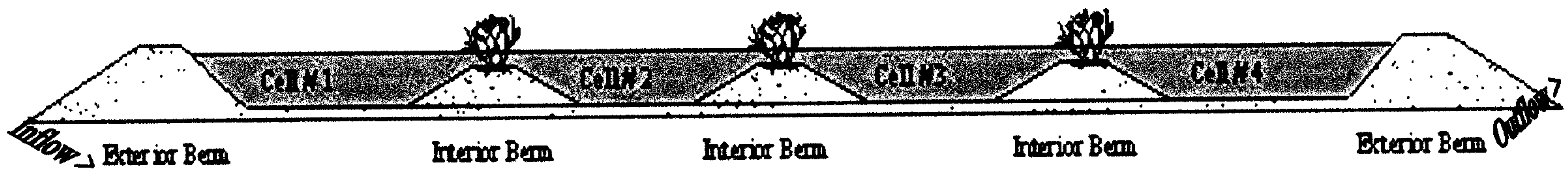
Some constructed wetlands are designed and operated solely to treat wastewater in the most efficient manner possible, while others are implemented with multiple-use objectives in mind, such as the creation of wetland habitat for wildlife use, educational applications, and aesthetic enhancement (USEPA, 2001; and Hammer, 1992). Constructed treatment wetlands are emerging as a cost-effective, environmentally friendly alternative to customary wastewater treatment, especially in the United States and Europe. The use of these systems in Canada, however, has yet to gain widespread acceptance. This is mostly due to lingering uncertainties surrounding treatment wetland performances in cold weather, especially for the removal for BOD, NH_4^+, and NO_3^- (Davis, 1995).

There are two types of constructed wetlands (Figure 3.2): free water surface (FWS) wetlands (also known as surface flow wetlands) and subsurface flow (SSF) wetlands (also known as root zone method wetlands or rock-reed filters). Both these systems typically consist of a series of cells or channels lined with impermeable material (such as clay or plastic liners) to limit potential groundwater infiltration. They both support additional substrates composed of soil or stone, which are established with some sort of aquatic vegetation, such as cattails and reeds (*Typha* spp. and *Phragmites* spp.) or water hyacinth (*Eichhornia crassipes*). Wastewaters enter the wetlands via simple gravitational forces or can be more stringently directed and controlled through the use of pumping mechanisms (Liehr et al., 2000). Specialized devices such as mechanized aerators, trickling filters, artificial medias, facultative lagoons or greenhouse-like enclosures can be added to treatment wetland systems in order to increase their efficiency. However, with each mechanism added, construction and maintenance costs increase. Most treatment wetland designers and managers strive to make these systems as self-sustaining and maintenance-free as possible (Tousignant et al., 1999).

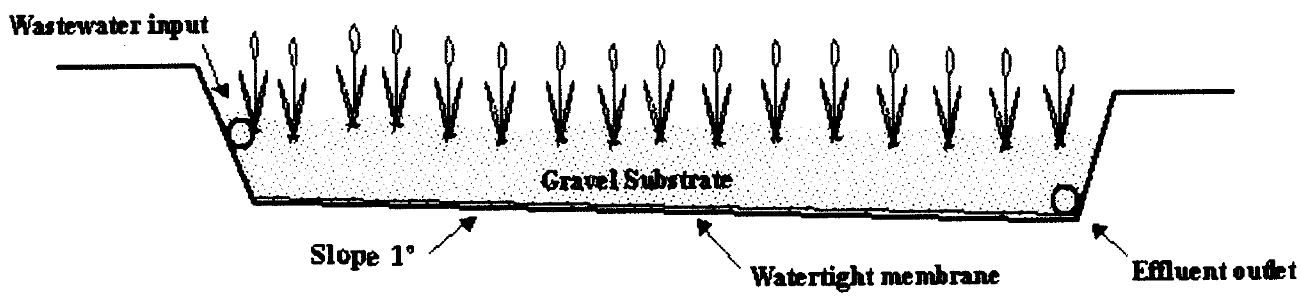
3.3.1. Free Water Surface Wetlands

Free water surface (FWS) wetlands are treatment wetlands in which the surface water flowing through them is exposed to the atmosphere. They typically consist of several basins or cells with the water surface being 0.15 to 2 metres above the substrate (Tousignant et al., 1999). They appear much like natural marshes, containing much emergent aquatic vegetation, and therefore tend to provide wildlife habitat and aesthetic benefits in addition to water treatment. In FWS wetlands, the near-surface layer is aerobic while the deeper waters and substrate are usually anaerobic. These systems are primarily constructed to treat municipal wastewaters, mine drainage, urban storm water, agricultural runoff and livestock wastes, and landfill leachate (USEPA, 2000c).

These treatment wetlands support a considerable sediment layer above their impervious liners in which emergent macrophytes, such as cattails (*Typha* spp.), rushes (*Juncus* spp.) and bulrushes (*Scirpus* spp.), are planted. Suspended solids are removed by gravitational settling. Nutrients and pollutants absorbed to the settled sediments are then exposed to aerobic rhizome areas created by the macrophytes, result and Davis, 1995). Free water surface wetlands can be further sub-classified according to their dominant type of vegetation and include: emergent macrophyte wetlands, free-floating macrophyte wetlands, and submerged macrophyte wetlands. The most common system is the emergent macrophyte based system. Free floating macrophyte based wetlands are similar in nearly every facet except these FWS systems make use of floating plants, such as duckweed (*Lemna* spp.) and water hyacinth (*Eichhornia crassipes)* to remove nutrients and other pollutants in wastewater. A floating barrier grid is used to support the growth of



(a) Cross-section Diagram of Common Free Water Surface Wetland System.



(b) Cross-section Diagram of Common Sub-Surface Flow Wetland System (adapted from USEPA, 2000d).

Figure 3.2. Constructed Wetlands.

floating macrophytes and to reduce wind effects, which would otherwise cause the plants to drift. These systems have the added benefit of being able to mitigate algae growth, a persistent problem in treatment wetlands brought on by the nutrient rich nature of the wastewaters received. In free floating systems, the densely packed floating plants work to block out sunlight, thereby preventing photosynthesis and inhibiting algae growth (Tousignant et al., 1999; and Davis, 1995). Little information is available on submerged macrophyte based FWS wetlands as they are still in the experimental stage, but as their name suggests, these systems would rely on the use of submerged macrophytes such as pondweed (*Potamogeton* spp.) to remove nutrients and other pollutants from received wastewaters (Brix, 1993; and Tousignant et al., 1999).

FWS wetlands have many advantages including: (a) lower and straightforward construction and operating costs, (b) little requirements for mechanical equipment, energy, and skilled operator, (c) superiority in their abilities to remove biochemical oxygen demand (BOD), chemical oxygen demand (COD), and total suspended solids (TSS), (d) significant reduction in the levels of nitrogen and phosphorus, metals, persistent organics and fecal coliforms, and (e) routine harvest and removal of FWS wetland vegetation is typically unnecessary, as phytoremediation represents a relatively secondary removal pathway of contaminants in these systems, hence plant removal does not provide a significant treatment benefit (USEPA, 2000c; Davis, 1995; and Liehr et al., 2000). The main disadvantages of FWS systems are: (a) they require a larger land area than other systems, (b) the wastewaters are exposed and are therefore accessible to humans and animals, hence it may not prove prudent to establish these wetlands in high-use areas such as parks, playgrounds, or similar public facilities, (c) pollutants such as phosphorus, metals, and some persistent organics can become bound in wetland sediments and accumulate over time, and (d) the open water environments of FWS wetlands can attract unwanted pests such as mosquitoes which may ultimately need control (USEPA, 2000c; Davis, 1995; and Liehr et al., 2000).

3.3.2. Sub-Surface Flow Wetlands

A sub-surface flow (SSF) wetland consists of a sealed basin with a porous substrate of rock, gravel or coarse sand planted with emergent macrophytes such as reeds (*Phragmites* spp.), Eurasian watermilfoil (*Myriophyllum spicatum*), and duckweeds (*Lemna* spp.). The depth of the substrate ranges from 0.3 to 0.9 metres. The water level is designed to remain below the top of the substrate allowing the same mechanisms as FWS wetlands to remove contaminants (Liehr et al., 2000). SSF wetlands are most commonly used to treat wastewaters from small-scale sources such as individual homes, schools, apartment complexes, commercial establishments, parks, and other recreational facilities. They can be sub-classified according to their flow patterns: horizontal flow and vertical flow. Horizontal flow SSF wetlands involve the continuous, horizontal flow of wastewaters through the medium. Oxygen is transferred into the system via atmospheric diffusion through the emergent aquatic plants. In vertical flow SSF wetlands, wastewater is added at timed intervals, and the system is allowed to drain between dosing. As a result, these systems tend to be less anoxic than horizontal flow systems as oxygen is able to diffuse easily from the atmosphere into the drained, porous substrates. However, horizontal flow systems remain by far the more commonly used and documented SSF systems (Tousignant et al., 1999).

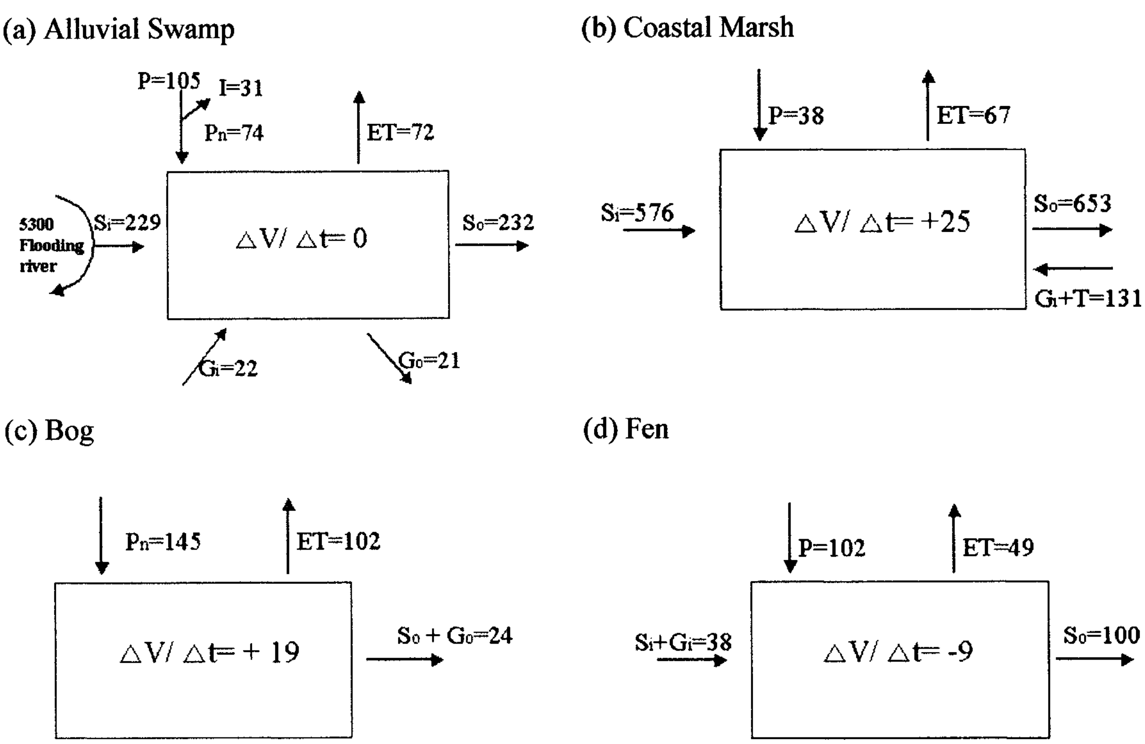
The major advantages of SSF wetlands include: (a) the rocky substrates provide greater surface area for microbial reactions and therefore SSF wetlands can be smaller in size yet treat larger flow volumes than FWS wetlands, (b) they are often better suited to projects in which the available land area is limited, (c) they are typically more suitable to public areas as contaminated wastewaters are not exposed, and (d) the nature of their substrates and flow regimes also allow for better thermal protection and are, therefore, considered to be more effective in colder climates than FWS systems (USEPA, 2000d). The disadvantages of SSF wetlands are (a): they are more expensive to construct, maintain and repair, (b) they have problems with clogging and as a result are best suited to wastewaters with relatively low solids concentrations and under relatively uniform flow conditions which often requires the installation of appropriate inlet and outlet structures designed to ensure the uniform distribution and collection of the applied wastewater, (c) as vegetation is the prominent removal mechanism of pollutants in these systems, plants can reach their points of saturation in terms of pollutant absorption, rendering them no longer effective thus requiring costly and time consuming harvesting, (d) they tend to be anoxic, which limits the biological removal of ammonia nitrogen via nitrification unless costly aerators are be adopted to mitigate this problem, (e) phosphorus removal rates are also inferior to that of FWS systems, (f) they can have problems associated with the accumulation of pollutants in sediments over time, and (g) they provide less habitat value than FWS (Davis, 1995; Tousignant et al., 1999; and USEPA, 2000c,d). The latter is actually advantageous when dealing with substances of high toxicity or treatment systems located in areas where wildlife is not desirable. For example, at the Edmonton Airport in Alberta, Canada, a SSF wetland was created in order to treat runoff contaminated with glycol in de-icing fluid. The creation of habitat in these systems was prohibited as increased birds attracted to the site could have proven problematic to planes utilizing the adjacent runways (Davis, 1995; and USEPA, 2000d).

3.4. Wetland Hydrology

Wetland hydrology is considered to be the most significant determinant of the defining physiochemical and biotic characteristics of a given wetland ecosystem. The physiochemical factors include: salinity, oxygen, nutrient and contaminant concentrations of soils and water. These physiochemical factors determine what vegetation and wildlife will be established in a wetland site. The hydroperiod of a wetland is its seasonal pattern of water level change, as a result of the varying water budget of the site, which is essentially determined by the balance between inflows and outflows of water, wetland landscape geomorphology and subsurface conditions (i.e. permeability of soils). Hydroperiods typically experience fluctuation as a result of seasonal (wet spring, dry fall) and annual weather variations (some springs can be wetter than others while some autumns can be drier than others as reported by Mitsch and Gosselink (2000).

Water inflow sources include precipitation, surface water, over bank flooding, groundwater input and tides. Water outflow can be facilitated by evapotranspiration, surface overflows, groundwater seepage and tidal outflow. The high degree of water input also directly influences the chemistry of soils present in the site as high input flows facilitate high sediment, nutrient and contaminant loading. The physiochemical environment in turn influences the biotic communities which are adapted to site conditions. In addition, water input creates aeration, which provides the oxygen needed to support microorganisms which facilitate the characteristically high rates of decomposition associated with marsh ecosystems. In contrast, wetlands which have little water input such as bogs become nutrient poor, and as a result support exceptionally different biotic communities. In addition, the standing water becomes anoxic hence productivity is lessened and decomposition rates which are dependent on oxygen-consuming microorganisms, slow dramatically. If the hydrological budget of a wetland changes, the wetland ecosystem would likely evolve into a completely different class (Mitsch and Gosselink, 2000; USEPA, 2002c; and Tiner, 1999). The annual hydrological budgets of the different classes of wetlands are often be characterized descriptively using the wetland water budget diagrams illustrated in Figure 3.3.

The physiochemical and biotic environments which arise as a result of a site hydrological regime are also able to impact the hydrology of a site, as well as each other, thereby forming a system of feedback loops. For example, the macrophytes (such as reeds and rushes) which establish as a result of the high nutrient input brought on by high water inputs in turn affect the physiochemical environment by aerating soils, trapping sediments, shading waters, building peat and removing



Where:

V	= volume of wetland water storage
$\Delta V/\Delta t$	= the change in volume of wetland water storage per unit time
P_n	= net precipitation
S_i	= surface inflows
G_i	= groundwater inflows
ET	= evapotranspiration
S_o	= surface outflows
G_o	= groundwater outflows; and
T	= tidal inflows (+) or outflows (-)

Values are expressed in centimetres per year

Figure 3.3. Example Annual Water Budgets for Each of the Four Main Wetland Classes (adapted from Mitsch and Gosselink, 2000).

or stabilising contaminants and pollutants. They also affect hydrological regimes by reducing flow velocities, reducing erosion, trapping sediments and building peat. Particular biotic environments may also attract certain wildlife such as the beaver which can have a dramatic influence on wetland hydrological regimes by building dams and blocking flows altogether (Mitsch and Gosselink, 2000; USEPA, 2002e; and Tiner, 1999). Figure 3.4 illustrates the connection between wetland hydrology and the physiochemical and biotic environment.

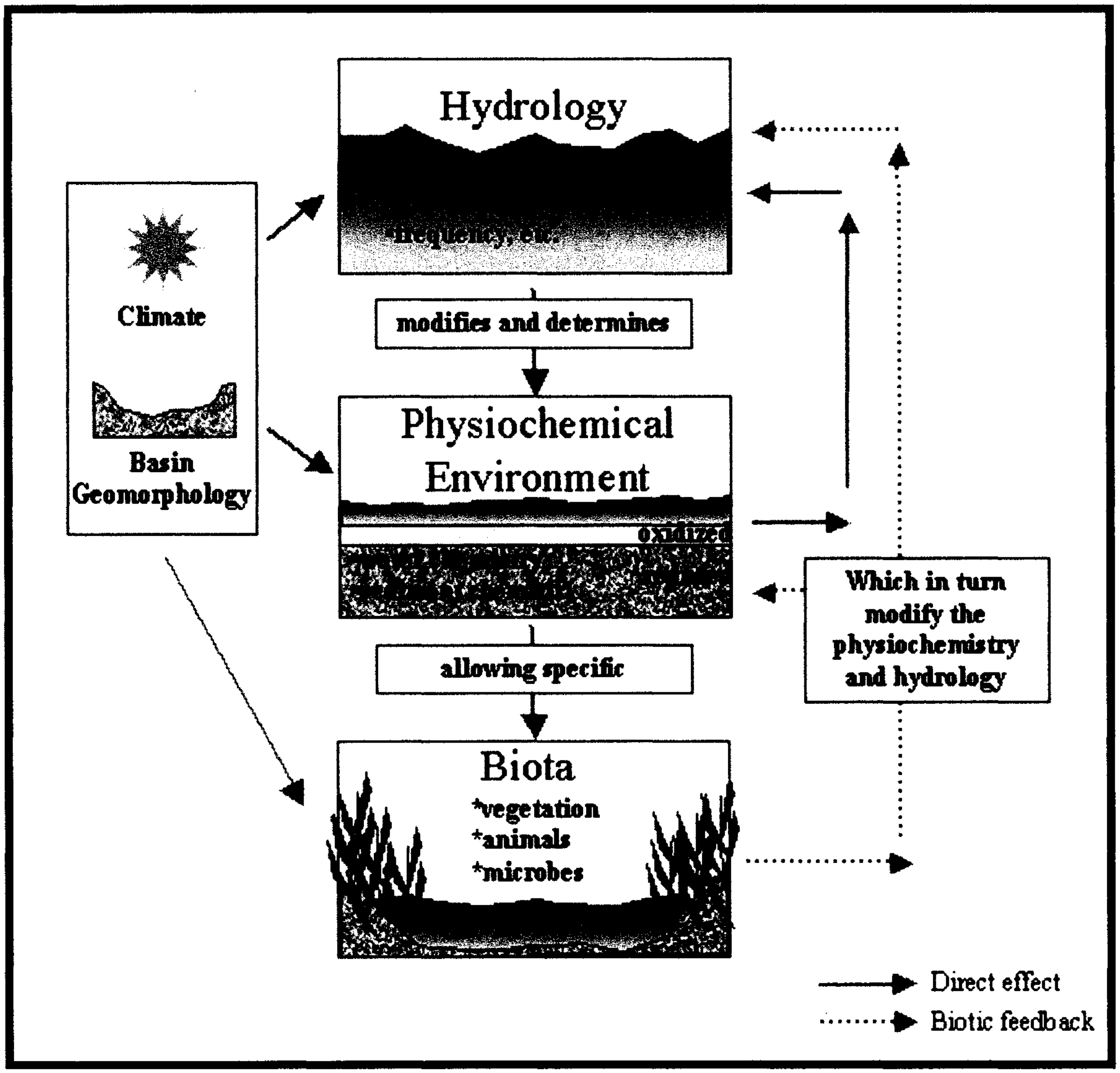


Figure 3.4. Illustration of Effects of Hydrology on Wetland Function (adapted from Mitsch and Gosselink, 2000).

3.5. Wetland Biogeochemistry

Biogeochemical cycling, which is the transport and transformation of chemicals in an ecosystem, involves vast amounts of interconnected physical, biological, and chemical processes. Biogeochemical cycling in wetlands occurs both within the system and wetland surroundings. Although few chemical transformation processes are exclusive to wetland ecosystems, the unique hydrological regimes of wetland environments cause certain processes to be more dominant in them than in other terrestrial or aquatic ecosystems (Mitsch and Gosselink, 2000). The chemicals cycled in wetlands include many ions and elements which are generally termed nutrients.

Nutrients are divided into two categories: macronutrients and micronutrients. Macronutrients are nutrients that comprise greater than 1% of the organic matter in a site and include: phosphorus, nitrogen, sulphur, magnesium and calcium. Micronutrients are required in minute amounts in systems and include: iron, manganese, copper, zinc, silicon, molybdenum, vanadium and cobalt. It is important to note that any of these nutrients in excess can cause toxic effects (Boulton and Brock, 1999). The main sources of these macro and micronutrients are precipitation, surface flow, groundwater and tides. Wetland systems which are able to receive much nutrients from these sources, such as tidal marshes, are considered to be biogeochemically open. Conversely, wetland systems which are somewhat isolated from hydrological transport mechanisms, such as bogs which typically receive very little input or output, are considered to be biogeochemically closed. Once received by the wetland system, macro and micronutrients are subject to a wide variety of biogeochemical reactions. Some of the major reactions which occur in wetland environments including reduction-oxidation (redox), nitrogen transformations, carbon transformations, phosphorus transformations, sulphur transformations, and metal transformations are discussed in the sections to follow (Mitsch and Gosselink, 2000).

3.5.1. Reduction-Oxidation

Reduction-oxidation (or redox reactions) are chemical processes which involve the transfer of electrons from one compound to another. When a substance is chemically reduced, it gains an electron, whereas when a substance is oxidized, it loses an electron. For example, when ferric oxide (Fe_3^+) gains an electron, it is reduced to ferrous oxide (Fe_2^+) and vice versa. The redox potential (E_h) of a solution is a measure of its ability to oxidize or reduce substances. In sediments, at neutral pH, oxidised conditions are indicated by a redox potential exceeding +400 mV, moderately reducing conditions occur between +100 and +400 mV, reducing conditions occur between -100 and 100 mV and highly reducing conditions occur at less than –100 mV (Boulton and Brock, 1999). Reducing conditions typically occur in oxygen deficient soils, while oxidizing conditions occur in oxygen rich soils. This is because oxidation is most rapid in presence of O_2, which is a very efficient electron acceptor. Redox reactions can also take place in the presence of any other terminal electron acceptors available. However, the rate of decomposition will be much slower (i.e. peatlands). In the absence of oxygen, nitrate, manganese, iron, sulfate, and then carbon or organic matter typically act as the terminal electron acceptors in wetlands (Boulton and Brock, 1999; and Mitch and Gosselink, 2000).

Redox reactions are very important to wetland and other systems as they facilitate the cycling and transformations of many chemicals such as nitrogen, carbon, hydrogen, sulphur and metals. In general, inorganic substances such as metals are reduced, and organic matter such as leaves and other vegetative debris is oxidized. Oxidized ions which typically dominate aerobic substrates include nitrate (NO_3^-), ferric oxide (Fe_3^+) and sulphate (SO_4^{2-}). Reduced ions dominating anaerobic soils typically include ferrous oxide (Fe_2^+), manganous (Mn^{2+}), ammonia (NH_4^+) and sulfides (S_2^-) (Mason, 1998).

Waterlogged soils such as those in wetlands are typically anaerobic as water displaces oxygen which would otherwise be present in the porous spaces of the substrate. Wetlands, however, contain specially adapted aquatic macrophytes such as reeds and rushes which thrive in reduced or deoxygenated conditions. These plants contain extensive internal air spaces known as lacunae or aerenchyma which occupy up to 70% of the plant mass. These specialised air spaces facilitate the diffusion and/or convective flow of atmospheric gases such as oxygen from the aerial portions of a plant down into the roots and into the surrounding soils. Conversely, gases in soils formed primarily by decomposition can diffuse into roots and subsequently into the atmosphere. As a result, the areas surrounding plant roots and capillaries become oxygen-rich. These aerobic zones are known as the rhizosphere (Hammer, 1992; and Brix, 1993). Hence, wetlands are unique systems in that their soils support both aerobic (oxidized) and anaerobic (reduced) zones. Redox reactions are particularly significant as they affect the solubility, availability and mobility of substances such as nutrients and pollutants. Many substances in their reduced forms (i.e. iron and manganese) are bioavailable to organisms and can be quite toxic. Hence, oxidation reactions are often responsible for the chemical conversion of many toxic substances (especially inorganic substances) to compounds which are less toxic or not toxic at all (FRTR, 1998). Several nutrient transformations facilitated by redox reactions include N, C, P, and S.

3.5.2. Nitrogen Transformations

Nitrogen is an essential nutrient to all life and a limiting factor to plant growth. It has a very complex biochemistry, and involves several microbiological processes when oxidized or reduced into its various forms. This reduction and oxidation cycling is described in wetland environments as the nitrogen cycle. Organically-bound nitrogen in dead organic matter is released as ammonium (NH_4^+) as the matter is broken down by bacteria and fungi. This process is called nitrogen mineralization or ammonification (Mitsch and Gosselink, 2000). In areas of the wetland where oxygen is not readily available, such as in the deeper sediments, most of this ammonium will be converted back to organic matter via re-assimilation by microbes, diffusion into the waterbody of the system, or taken up by vegetation; although most plants cannot use ammonium effectively, and require nitrate as their main nitrogen nutrient (Lorion, 2001). Where oxygen is available, the ammonium will be oxidized by Nitrosomonas bacteria to nitrite (NO_2^-) which is then oxidized to nitrate (NO_3^-) by Nitrobacter bacteria via a process referred to as nitrification (Boulton and Brock, 1999). Nitrate, which is extremely mobile, is usually quickly assimilated by plants or microorganisms (assimilatory reduction), reduced by anaerobic bacteria to nitrogen gas (N_2) via denitrification or reduced back to ammonium by both aerobic and anaerobic bacteria (i.e. Rhizobium, organisms which live in roots of legumes) via nitrogen fixation process known as dissimilatory reduction (Mitsch and Gosselink, 2000; Boulton and Brock, 1999; and Lorion, 2001). Figure 3.5 illustrates the main components of the nitrogen cycle.

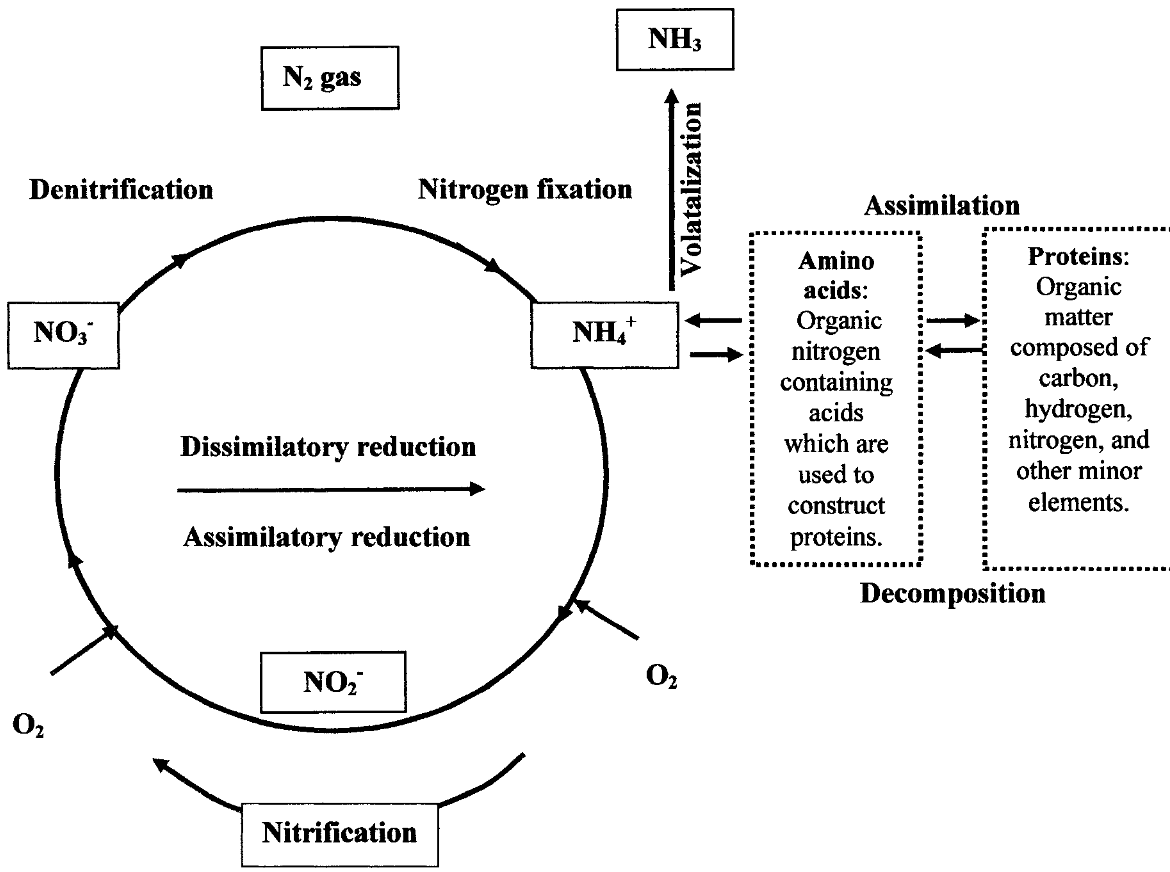


Figure 3.5. Main Components of the Nitrogen Cycle (adapted from Boulton and Brock, 1999).

Nitrification requires oxygen as its electron acceptor. Optimal temperature for nitrification is in the range of 25-35°C, and nitrification rates are drastically lowered at temperatures less than 15°C. Denitrification occurs in anaerobic environments with nitrate serving as the electron acceptor while organic carbon serves as the electron donor (Liehr et al., 2000). Optimal temperature for denitrification is in the range of 25-65°C (Hammer, 1992). Ammonification on the other hand can occur under both aerobic and anaerobic conditions (Liehr et al., 2000). Ammonium can be converted into ammonia gas via a process called volitization. Ammonia and ammonium shift in equilibrium with each other according to pH. Ammonium, which is relatively non-toxic, is typically the more commonly found form of the two in waters with pH less than 9. In highly basic waters (>9 pH), ammonia, which is highly toxic at elevated concentrations, is the more dominant form (Boulton and Brock, 1999).

3.5.3. Carbon Transformations

The carbon cycle describes the process by which carbon cycles from the atmosphere to plants and animals and back to the atmosphere again. Atmospheric carbon exists mainly as carbon dioxide (CO_2), but also in the form of carbon monoxide (CO), methane gas (CH_4) and other gases. In wetland environments, carbon is sequestered from the atmosphere into biomass via photosynthesis and is contained in vegetation, peats, organic soils and sediments. Given characteristics such as wetland size, the depth of soils, groundwater levels, nutrient levels, pH and other factors, wetlands can be significant carbon reservoirs. However, wetlands also simultaneously release carbon as carbon dioxide, dissolved carbon, and methane. Carbon sequestering and release roles of wetlands are highly complex and vary from site to site. Overall, net gradual sequestration occurs in these systems, especially in sites which accumulate biomass, such as bogs and fens (Mitsch and Gosselink, 2000).

Carbon cycles in a wetland system via various physical, chemical and biological processes. Physical processes include: (a) inflow of organic matter from catchment area, (b) aggregation and flocculation of fine particles, (c) settling of particulate matter, and (d) resuspension of particulate matter. Chemical processes include: (a) dissociation and breakdown of compounds and formation of new compounds, (b) adsorption or adhesion of molecules to surface of solids or fluids, (c) desorption, (d) formation of precipitation and insoluble substances as solids, (e) diffusion, and (f) photolysis or degradation of dissolved organic carbon (DOC) by sunlight. Biotic processes and fluxes include: (a) respiration, (b) photosynthesis, (c) consumption (d) death, (e) secretion (organic compounds secreted during photosynthesis such as algal exudates), (f) autolysis or cell breakdown via enzymes, (g) plant transfer (methane in sediments travels through plants and released to atmosphere), (h) excretion by fauna, (i) movement of organisms, and (j) microbial decomposition. Specific decomposition processes include: (a) aerobic decomposition, (b) anaerobic decomposition, (c) methanogenesis (conversion of organic acids to methane and CO_2

by methane-producing bacteria), (d) methane oxidation (methane microbially oxidised and CO_2 produced), and (e) bubbling (escape of methane gas from sediments by bubble formation). The interactions of the various transformations occurring in wetland systems are illustrated in Figure 3.6 (Sommer, 2001; and Mason, 1998).

3.5.4. Phosphorus Transformations

Phosphorus enters wetlands with suspended solids or as dissolved phosphorus. It is an essential macronutrient to all life, and is considered the major limiting factor to the growth of aquatic plants, and algae (Osmond et al., 1995a). The cycling of phosphorus involves transformations between organic and inorganic phosphorus existing in soluble and insoluble forms (Boulton and Brock, 1999; and Mitsch and Gosselink, 2000). At any given time, the majority of phosphorus available in a wetland system is typically bound and immobilised in wetland sediments and litter or bound in living plants and organisms. Soluble organic phosphorus and insoluble forms of both organic and inorganic phosphorus are generally not bioavailable to plants and organisms until they are transformed into their soluble inorganic forms which typically occur as (PO_4^{3-}, HPO_4^{2-} and $H_2PO_4^-$). Microbial removal of phosphorus from wetland soil or water is particularly rapid and highly efficient. However, phosphorus is released back into the water column once the organisms die and decompose. Similarly, phosphorus is also released when plant matter decomposes. Phosphorus can also form complexes with aluminium, iron, and calcium in aerobic conditions (Boulton and Brock, 1999; and Mitsch and Gosselink, 2000). Figure 3.7 illustrates phosphorus transformations in wetlands.

3.5.5. Sulphur Transformations

Sulphur is an essential macronutrient for life. Natural sources of sulphur include rainfall and weathering rock. In aerobic wetland conditions such as in the rhizosphere, sulphur will be present as sulphate (SO_4^{2-}), which is in its oxidized form. In anaerobic areas of wetland systems, sulphur will be present as sulphide (mostly in the form of hydrogen sulphide (H_2S)), which is in its reduced form. Reduced hydrogen, methyl and dimethyl sulphides are often recycled to the atmosphere. However, most reduced sulphur compounds quickly react with metals to form metal sulfides (ie. FeS_2) which precipitate (Lorion, 2001). Once precipitated, the sulphur present in the formulated metal compounds often becomes permanently removed from the sulphur cycle (Mitsch and Gosselink, 2000). After a system has reached its capacity for metal sorption, metal sulfide formation often becomes the main method of metal removal in wetland systems (Kadlec and Knight, 1996). Sulphur compounds are the fourth major electron acceptors after nitrates, iron

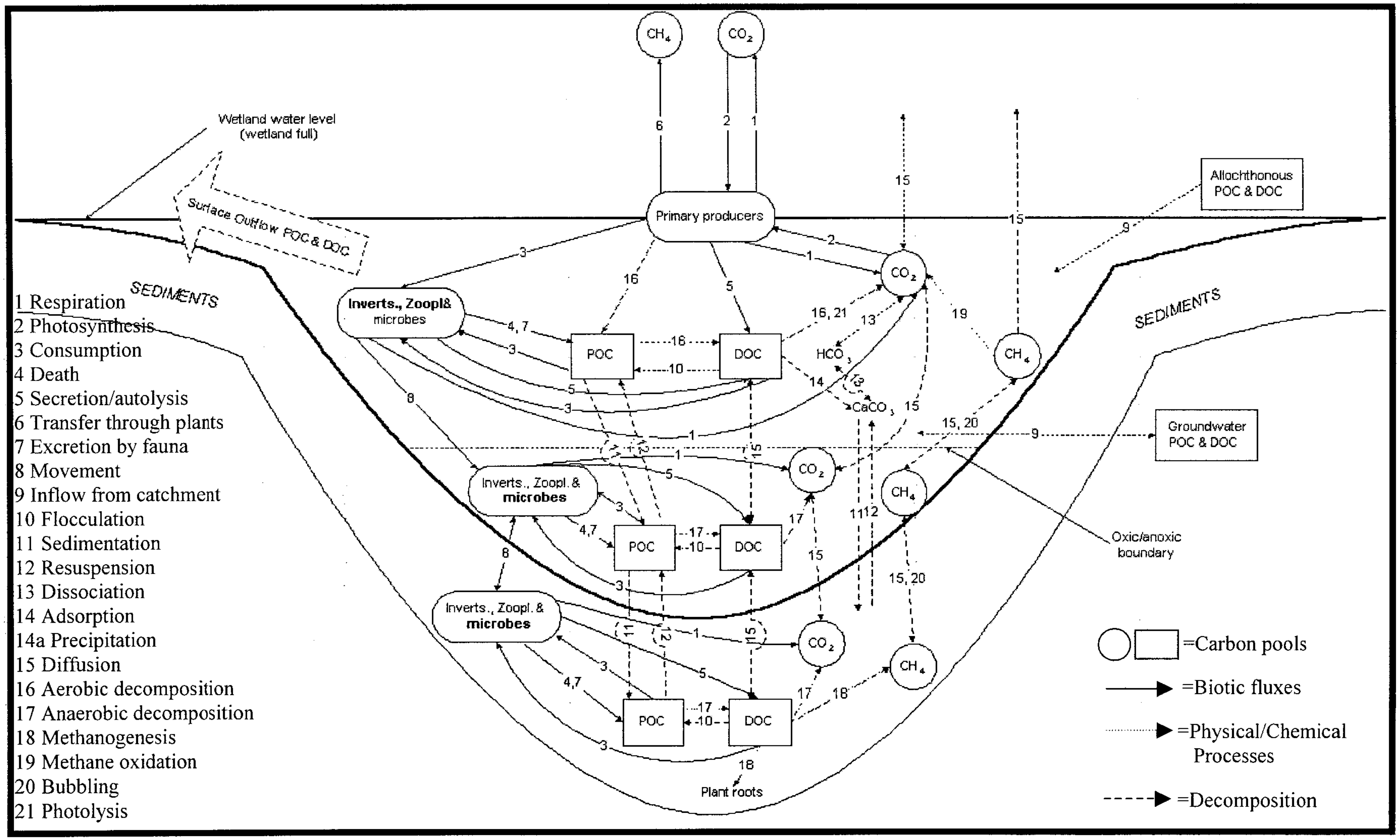


Figure 3.6. Simplified Carbon Cycle in a Wetland Ecosystem (from Sommer, 2001).

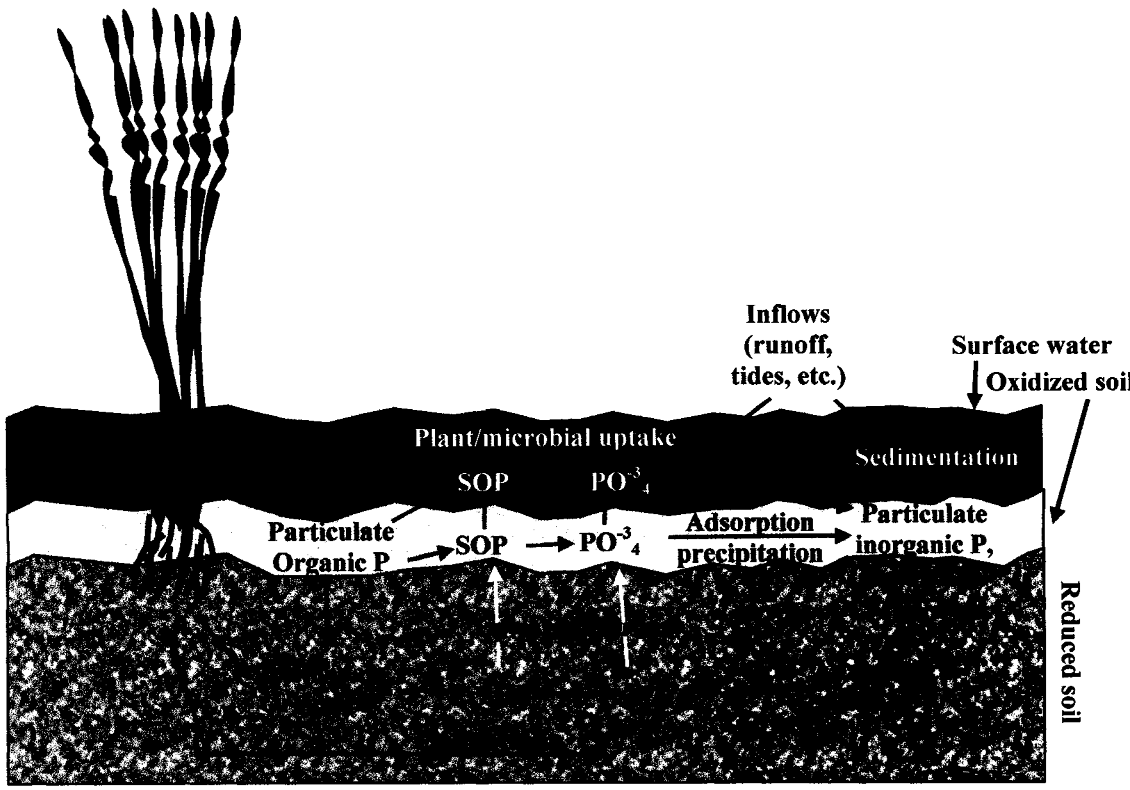


Figure 3.7. Simplified Phosphorus Cycle in Wetlands (adapted from Mitsch and Gosselink, 2000).

and manganese (Mitsch and Gosselink, 2000). Figure 3.8 illustrates a typical sulphur cycle in a wetland ecosystem.

3.5.6. Metal Transformations

Metals common in wetland environments include: aluminium, arsenic, cadmium, cobalt, copper, iron, lead, manganese, molybdenum, mercury, nickel, tin, selenium, vanadium and zinc. Metal cycling in wetlands is complex and highly dependent on existing wetland conditions. In most circumstances, metals entering wetland systems oxidize and either bind to sediments, particulates or soluble organics, or precipitate as insoluble salts such as sulphides and oxyhydroxides. Metals in their reduced, soluble forms can also be taken up or transformed by plants and microorganisms Kadlec and Knight, 1996; and Osmond et al., 1995b). For example, iron is an essential element required by both plants and wildlife. On the redox scale, iron is an important electron acceptor, second only to nitrogen. Oxidized iron (Fe_3^+) is dominant in aerobic conditions, such as in the rhizosphere of the sediments. Reduced iron (Fe_2^+) is found in deeper, anaerobic soils of wetland environments. Soluble ferrous oxide and less soluble ferric oxide can be converted to each other by aerobic microbes (i.e. Thiobacillus-Ferrobacilus bacteria). In surface waters, ferric oxide often

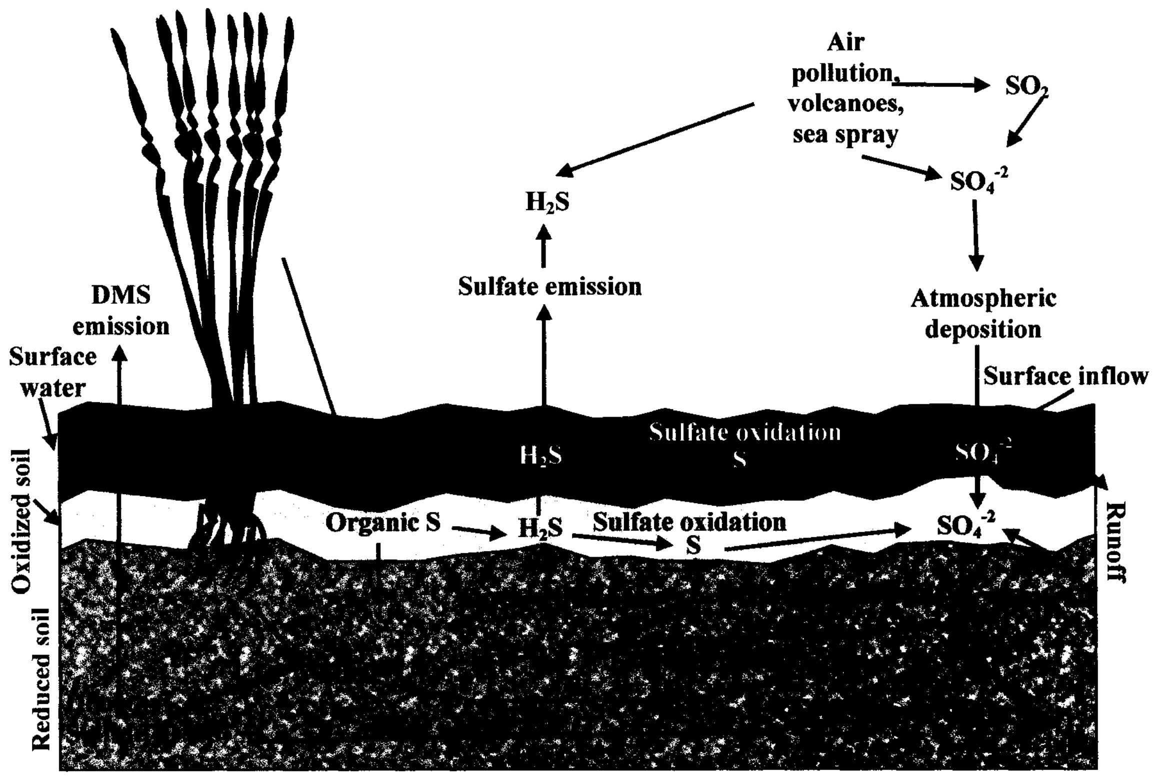


Figure 3.8. Simplified Sulphur Cycle in Wetlands (adapted from Mitsch and Gosselink, 2000).

joins with hydroxide to form ferric hydroxide [$Fe(OH)_3$] precipitate, or with other substances (such as phosphate, creating ferric phosphate ($FePO_4$). The precipitation of metals is highly dependent on pH and oxygen presence. For example, in oxygen rich zones, iron precipitates at pHs greater than 3.5, aluminum at pHs greater than 5 and manganese at pHs greater than 7. In oxygen depleted zones, pHs must be much greater in order for precipitation to occur, if at all (Kadlec and Knight, 1996; Brix, 1993 and Ayers et al., 1994).

Although metals are essential micronutrients for life in wetlands ecosystems, any of these metals can cause severe toxicity at elevated concentrations. Toxicity is dependent on the form of the metal (i.e. reduced, soluble form versus insoluble, oxidized form) and its relative toxicity as some metals such arsenic, cadmium and lead are highly toxic at low concentrations. In addition, some metals such as mercury can biomagnify in species, causing small concentrations to cause devastating effects to wildlife at the top of the food chain, as well as humans (Kadlec and Knight, 1996). Precipitates are not readily bioavailable and are, therefore, generally not toxic to organisms. However, the physical presence of the precipitates can have indirect effects on habitat quality by smothering vegetation and forming barriers over substrates, and by increasing turbidity (Kadlec and Knight, 1996; and Mitsch and Gosselink, 2000).

3.6. Phytoremediative Processes in Wetlands

Phytoremediation is a general term used to describe various mechanisms by which living plants are able to remove, extract, breakdown, stabilize, volatize and transform a variety of macro and micronutrients and organic substances as a result of specific adaptations and plant morphology (Raskin and Ensley, 2000; and Schnoor, 2002). The exploitation of these natural phytoremediation processes for the treatment of contaminants has become an increasingly popular practice in the last few decades and plants with significant phytoremediative capabilities are being increasingly utilized as remediative tools in constructed wetlands for the treatment of wastewaters contaminated with nutrients (i.e. N, P, metals), pesticides, solvents, explosives, hydrocarbons, VOCs, and radionuclides (Schnoor, 2002). Many wetland plants are showing the capacity to withstand relatively high or toxic concentrations of contaminants (USEPA, 2000b). There are six main types of phytoremediation processes occurring in wetland environments. These are phytoextraction, phytotransformation, rhizosphere bioremediation, phytostabilization, phytovolatilization and rhizofiltration (Figure 3.9).

3.6.1. Phytoextraction

Phytoextraction, also called phytoaccumulation, involves the physical uptake (extraction) of chemicals by plant roots and their subsequent accumulation in the aerial portions of plant biomass such as shoots and leaves (FRTR, 1998; and CPEO, 1998). All aquatic plants extract macro and micronutrients from their environments, but most do not accumulate them beyond their metabolic needs (typically less than 10ppm). However, some plants (termed hyperaccumulators) are capable of accumulating pollutants at levels 100-fold greater than those typically found in common non-accumulator plants (i.e. having hyperaccumulating ability of 10 ppm Hg, 100 ppm Cd, 1,000 ppm Co, Cr, Cu, and Pb, 10,000 ppm Ni and Zn) (FRTR, 1998; CPEO, 1998; and Lasat, 2000). It is thought that these plants extract these substances for storage in times of nutrient deficiency, or perhaps to allow them to evade predators (i.e. insects, bacteria) by building up toxic levels of metals in their foliage (Lasat, 2000).

Conventionally, remediation of metal-contaminated soils and sediments involved physical extraction and disposal in landfill sites. However, the use of plants to extract metals from

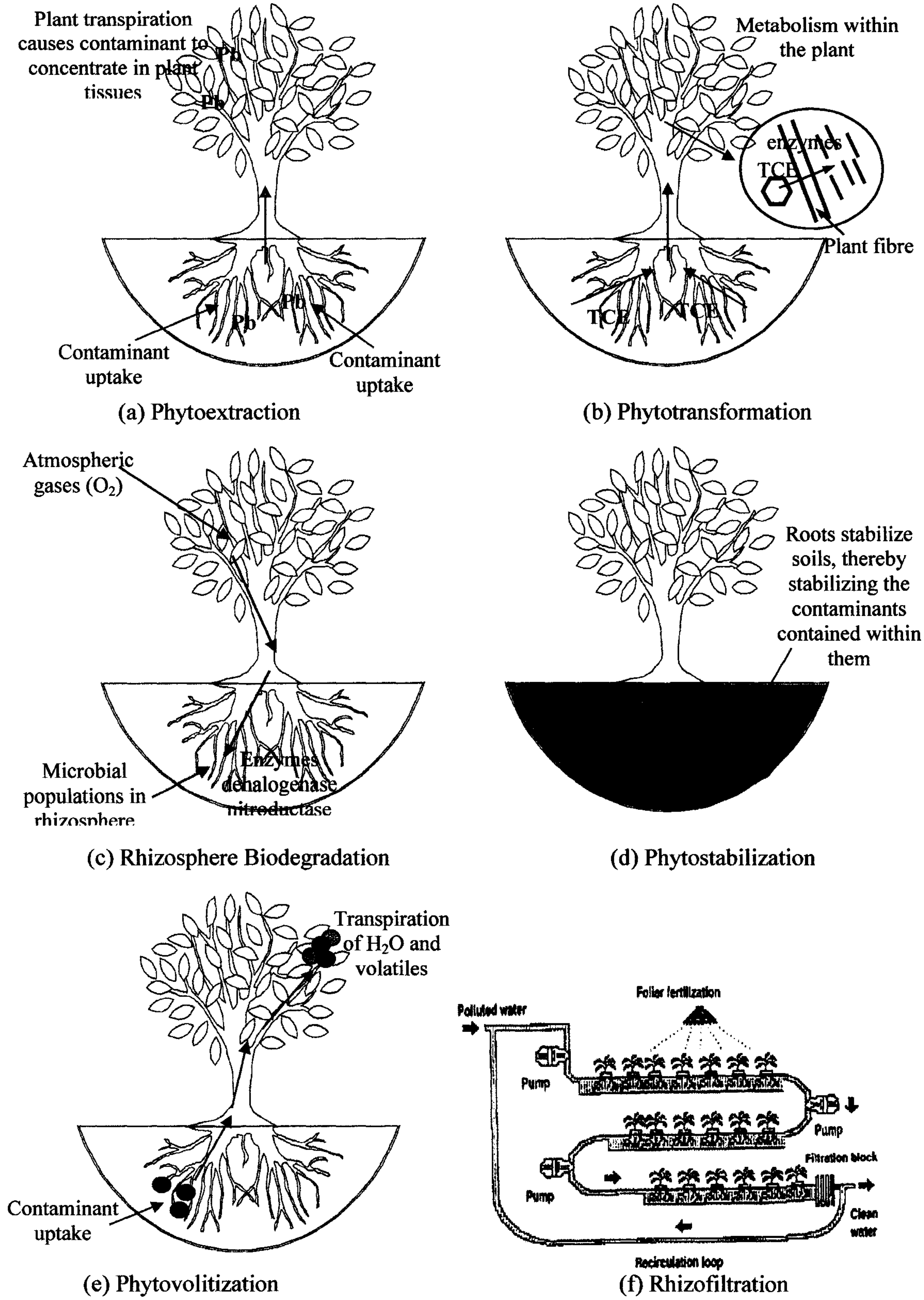


Figure 3.9. Types of Phytoremediation (adapted from USEPA, 2000b; and ASPB, 2003).

contaminated sites has emerged as a cost-effective, environment-friendly cleanup alternative to this invasive remediation methodology (FRTR, 1998; and CPEO, 1998). As of the year 2000, there were approximately 400 known metal hyperaccumulators from at least 45 plant families in the world (Lasat, 2000). Most known hyperaccumulators are those that bioconcentrate nickel. They make up over 300 upland plants belonging to the Brassicaceae (mustard), Euphorbiaceae (spurge), Asteraceae (aster), Lamiaceae (mint), or Scrophulariaceae (figwort, foxglove and speedwell) plant families (USEPA, 2000b). Hyperaccumulator plants have also been successful at some radionuclide remediation including ^{90}Sr, ^{137}Cs, ^{234}U, ^{238}U and ^{239}Pu (Liehr et al., 2000).

Table 3.3. Example Metal Hyperaccumulator Species (adapted from Lasat, 2000).

SCIENTIFIC NAME	METAL ACCUMULATED	LEAF CONTENT (PPM)
Streptanthus polygaloides	Ni	16,400
Thlaspi caerulescens	Zn: Cd	39,600: 1,800
Ipomea alpina	Cu	12,300
Haumaniastrum robertii	Co	10,200
Astragalus racemosus	Se	14,900

As a general rule, phytoextraction of metals and other contaminants will only occur if they are bioavailable (absorbable by roots). Typically, only metals which exist in their soluble forms and complexes or metals that are adsorbed to inorganic soil constituents at ion exchange sites are readily available for plant uptake. Many metals in their oxidized-insoluble forms, metals which are bound to soil organic matter, or metals which are precipitated as oxides, hydroxides or carbonates are not readily bioavailable (Lasat, 2000). Metals which tend to be bioavailable in wetland environments include cadmium, nickel, zinc, arsenic, selenium, and copper. Moderately bioavailable metals include: cobalt, manganese, and iron. Lead, chromium, and uranium are typically not bioavailable in wetland environments. To date, there are no known lead bioaccumulators (Lasat, 2000). Insoluble metals such as lead, however, can be made more available for extraction with the addition of the synthetic chelate ethylene-diamine-tetraacetic acid (EDTA) to soils which prevents metal precipitation.

Hydraulic Control is the term used to describe the process of phytoextraction in larger species such as willow (*Salix* spp.) poplar and cottonwood (*Populus* spp.) trees. Plant roots sorb the contaminants along with other nutrients and water, thus accumulating them in their biomass. Poplar trees, for example, can transpire between 50 and 300 gallons of water per day out of the ground. This high water consumption decreases the tendency of surface contaminants to move towards groundwater. This method is used primarily for treatment of contaminated groundwater (CPEO, 1998; and USEPA, 1998b).

One of the downfalls of using hyperaccumulators in site remediation is that plants may need to be harvested in order to prevent the contaminants from re-entering the system once the plants die and decompose. Harvested plants are usually either incinerated or composted. The ash resulting from incineration processes is disposed of in a hazardous waste landfill (USEPA, 1999a; FRTR, 1998; CPEO, 1998; and Schnoor, 2002). The exception to this is the hydraulic control technique since trees do not need to be harvested due to the long life cycle of the species. Because most hyperaccumulator plants are small in size and slow growing, this remediation technique is sometimes considered slow and inefficient (USEPA, 2000b). In addition, plants used in phytoextraction all have finite points of saturation and once saturation is reached, the contaminated media may become toxic to the plants, causing premature mortality and the subsequent release of the remediated contaminants back into the sediments (Lasat, 2000). The use of phytoextraction can also be hurtful to wildlife consuming the foliage of plants containing high levels of contaminants (USEPA, 2000b).

3.6.2. ***Phytotransformation***

Phytotransformation, also know as phytodegradation, involves the breakdown of chemicals extracted by plants via metabolic processes, or the breakdown of chemicals external to plants via excreted compounds (such as the enzymes dehalogenase and oxygenase) produced by plant roots. This effectively transforms or degrades substances to inert or less toxic forms for use as nutrient (Schnoor, 2002). Application of this natural phytoremediative process in contaminated sites would eliminate the need for harvesting (USEPA, 2000b).

Currently, the applications of this technique revolve around the phytotransformation of organic compounds. However, research on the ability of plant enzymes to degrade and convert chlorinated solvents (ie. TCE), phenols, ammunition wastes, and agricultural chemicals has also being undertaken (USEPA, 2000b). In addition, research is proceeding to determine which species are capable of effectively performing this technique (FRTR, 1998; and CPEO, 1998). Plant species which are currently being studied include hybrid poplars, aspens and cottonwoods (*Populus* spp.), and willows (*Salix* spp.). Certain grasses such as rye (*Lolium* spp.), Bermuda grass (*Cynodon dactylon),* sorghum (*Sorghum* spp.), fescue (*Festuca* spp.) and legumes such as clover (*Trifolium* spp.), alfalfa (*Medicago sativa*) and cowpeas (*Vigna* spp.) are also showing potential, particularly for binding and transforming hydrophobic contaminants such as TPH, BTEX, and PAHs (Schnoor, 2002; and USEPA, 2000e). Species of algae and *Chara* (i.e. stonewort) are also proving to have contaminant transformation abilities (USEPA, 2000b). In seeing that this phytoremediation technique is still in its early research stages, the advantages and disadvantages of its application have yet to be fully recognized. One potential disadvantage of this technique is that some by-products produced during transformation processes may actually be more harmful than the parent substance. Hence, careful scientific consideration of the specific contaminants of concern, as well as the plants present in the site must be taken into consideration (USEPA, 2000b).

3.6.3. ***Rhizosphere Biodegradation***

Rhizosphere biodegradation (also called rhizosphere bioremediation, phytostimulation, plant-assisted bioremediation and rhizodegradation) is the breakdown of chemicals (especially organic chemicals including fuels, solvents and pesticides) in the oxygen-rich rhizosphere soils surrounding plant roots through microbial activity (Schnoor, 2002; FRTR, 1998; and CPEO, 1998). Chemicals are also degraded by the release of root exudates and enzymes, but the results are insignificant as compared to the microbial degradation (Schnoor, 2002). Aquatic macrophytes such as reeds and rushes contain extensive internal air spaces known as lacunae or aerenchyma occupying up to 70% of a plant mass. The lacunae facilitate the diffusion and/or convective flow of oxygen from aerial parts of the plants into the roots for the purpose of respiration. Some of this oxygen leaks out of the plant causing the soils immediately surrounding plant roots and capillaries to become oxygen-rich, thereby supporting the growth of aerobic microorganisms (i.e. yeast, fungi, or bacteria) which would not otherwise occur in the anoxic conditions typical of saturated wetland soils (Hammer, 1992; Brix, 1993; USEPA, 2000e; and Brix, 1993). The rhizosphere also provides food for microorganisms, as plant roots release sugars, alcohols, and acids. In addition, plant roots physically create paths for transport of water and aeration, thereby facilitating microbial activity even further (USEPA, 1999a; FRTR, 1998; and CPEO, 1998).

Although categorized as a phytoremediation process, this technique is actually a form of bioremediation, as it is the microorganisms, not the plants that are in actuality remediating the contaminants by consuming and digesting them for nutrition and energy (FRTR, 1998; and CPEO, 1998). Plants which are most effective for creating suitable environments for microorganisms are those with large, deeply penetrating, fibrous root systems such as rushes and bulrushes (*Juncus* and *Scirpus* spp.) and certain grasses such as rye (*Lolium* spp.), fescue

(*Festuca* spp.) and Bermuda grass (*Cynodon dactylon*) (USEPA, 2000e; and USEPA, 2000b). Some of the disadvantages of this technique include: (a) it requires large root densities to work effectively, which can take time to develop, (b) not all microorganisms form symbiotic relationships with plants, and may actually compete for nutrients with the plant, leading to plant nutrient deficiency, and (c) not all microorganisms living in the aerobic rhizosphere will be contaminant degraders, hence rates of degradation can be variable from site to site despite the presence of abundant root mass (USEPA, 2000b).

3.6.4. Phytostabilization

Phytostabilization involves the physical confinement of chemicals at plants roots, the immobilization of chemicals via adsorption onto roots and/or precipitation within the root zone as a result of chemical compounds produced by wetland plants. Phytostabilization processes simply immobilize substances and do not degrade them. In terms of remediation applicability, phytostabilization helps minimize hazard and exposure of contaminants to organisms by preventing their migration to the groundwater or air (FRTR, 1998; and CPEO, 1998). Contaminants which have been treated by this technique include lead, cadmium, zinc, arsenic, copper, chromium, selenium, uranium, and hydrophobic organics such as PAHs, PCBs, dioxins, furans, pentachlorophenol, DDT, dieldrin (USEPA, 2000b). This process serves as a more physical method of contaminant control. For example, plant roots can act to stabilize contaminated soils and prevent erosion. If erosion does not occur, the contaminants within the soils will remain contained and would not be available to harm organisms (Schnoor, 2002). By the simplicity of its definition, virtually all plants are phytostabilizers. However, there are of course some plants, especially those with strong and extensive root systems such as *Juncus*, *Scirpus* and grasses, which are better immobilizers than others (USEPA, 2000e).

3.6.5. Phytovolatilization

Phytovolatilization involves the uptake and transpiration of a chemical by plants, and the subsequent release of these chemical in its original or modified form into the atmosphere. Phytovolatilization occurs as growing plants take up water, hence the more water a plant takes up, the more chemicals it will transpire. Common remediative applications of this naturally occurring process include treatment of chlorinated solvents, trichloroethylene (TCE), selenium, mercury, and arsenic, which ultimately volatize into the atmosphere at comparatively low concentrations, or as less harmful substances (USEPA, 1999a; and Lasat, 2000). Plants which are particularly effective at phytovolitization are species belonging to the *Populus* family which take up and transpire large amounts of water. Poplar trees are capable of volatilizing 90% of TCE extracted from the ground (USEPA, 1998b). Other species proven effective at phytovolitization include Indian mustard (*Brassica juncea*), and alfalfa (*Medicago sativa*).

Some of the disadvantages of this technique includes: (a) some contaminants may be released into the atmosphere at harmful levels, and (b) bioconcentration of contaminants sometimes occurs in plant leaves and fruit, which could potentially have adverse effects on wildlife that consume them (USEPA, 2000b).

3.6.6. Rhizofiltration

Rhizofiltration involves the adsorption or precipitation of chemicals present in a solution surrounding the root zone onto or into plant roots. Unlike the other natural phytoremediative processes, plants used in remediative applications of this process are raised hydroponically in greenhouses. Once plant roots are sufficiently developed, wastewater collected from a waste site is transported to the greenhouse where it is used as a water source. Once acclimated, the plants can either continue to treat the wastewater received (ex-situ), or be transplanted into the source contaminated area (in situ). When the roots become saturated with contaminants, they commonly need to be harvested and ultimately disposed of (USEPA, 2000b; and USEPA, 1998b). This technique is primarily used to treat only contaminated waters, not sediments and soils. Contaminants successfully treated are typically metals including: lead, cadmium, chromium, zinc, nickel, and copper. The technique is also capable of remediating radionuclides ^{137}Cs, ^{90}Sr, and U (Schnoor, 2002, USEPA, 2000b; and USEPA 1998b). Sunflowers were used successfully to remove radioactive contaminants from pond water in experiments conducted at Chernobyl, Ukraine (USEPA, 1998b). Plants typically utilized in rhizofiltration include water hyacinth (*Eichornia crassipes*), bulrushes (*Scirpus* spp.), cattails (*Typha* spp.), coontail (*Ceratophyllum demersum)*, pondweeds (*Potamogeton spp.)*, arrowroot (*Thalia geniculata*), duckweed (*Lemna* spp.), algae, stonewort (*Chara* spp.), parrot feather (*Myriophyllum aquaticum*), Eurasian water milfoil (*Myriophyllum spicatum*), Hydrilla (*Hydrilla verticillata*), sunflowers *Helenium sp.),* and Indian mustard (*Brassica juncea).*

Rhizofiltration is unique compared to other phytoremediation techniques in that it can be conducted ex-situ year round in enclosed structure, minimizing disturbance to the environment and the effect of winter climates. However, ex-situ treatment is much more expensive to create, operate and maintain. Even if in-situ treatment is used, the cost of this technique is still significantly higher than other phytoremediation techniques since plants are grown in specialized greenhouses. Plants tend to be slow growers with low biomass, making some researchers view this remediative technique as inefficient (USEPA, 2000b).

3.7. Bioremediative Processes in Wetlands

Microbial activities are the primary mechanism responsible for the breakdown of organic matter in wetland environments (FRTR, 1998; and USEPA, 1996). In the presence of sufficient oxygen (aerobic conditions) and nutrients, microorganisms (bacteria, fungi, algae and protozoa) will ultimately convert organic contaminants to carbon dioxide, water and microbial cell mass. In the absence of oxygen (anaerobic conditions), either sulphate, nitrate, or carbon dioxide will serve as an electron acceptor and the contaminants will be ultimately metabolized to methane, limited amounts of carbon dioxide and a trace amount of hydrogen gas (Van Cauwenberghe, and Roote, 1998; and USEPA, 1996). With the exception of a few very complex, large molecular weight compounds (i.e. polycyclic compounds), given enough time a population of microbes can degrade virtually any compound that contains a potential energy source (Hammer, 1992).

Bioremediation techniques have been successfully used to remediate soils and groundwater contaminated with organic compounds such as petroleum hydrocarbons, solvents, pesticides, wood preservatives, and others. As a rule of thumb, the higher the molecular weight (and the more rings with a PAH), the slower the degradation rate of the contaminant, and the more chlorinated or nitrated the compound, the more difficult it is to degrade. Although bioremediation techniques have no direct effect on inorganic contaminants such as metals, they can be used to change the valence state of inorganics and cause adsorption, uptake, accumulation, and concentration of inorganics in macro or microorganisms (FRTR, 1998). For example, some microorganisms excrete organic compounds which increase bioavailability of Fe, Mn, and Cd and facilitating their absorption by plant roots (Lasat, 2000). Some bacteria have also been shown to increase precipitation of iron, manganese and other metals (Witthar, 1993). Bioremediation processes are also responsible for the majority of nitrogen removal in constructed treatment wetland systems, as ammonification, nitrification, and denitrification are all driven by microbial activity (Liehr et al., 2000).

Bioremediation technologies have their ability to ideally transform toxic compounds to harmless substances with little disturbance. In most cases, the contaminants are destroyed and little to no

Table 3.4. Example Bacteria which Effectively Treat Contaminants of Concern (adapted from Lasat, 2000; and Liehr et al., 2000).

BACTERIA	CONTAMINANT TREATED
Nitrosomonas	ammonium
Nitrobacter	nitrite
Rhizobium	nitrate
White rot fungus	wood preservatives and pesticides
Methanotrophic bacteria	methane
Geobacter metallireducens	Can reduce uranium
Marine bacteria	accelerate metal precipitation
Thiobacillus spp.	Reduce and oxidize iron
Pseudomonas maltophilia	Reduce the mobile and toxic Cr_6^+ to nontoxic and immobile Cr_3^+, and also minimizes the mobility of Hg_2^+, Pb_2^+, and Cd_2^+

residual treatment is required. However, some compounds are broken down into more toxic by-products during the bioremediation process (i.e. TCE to vinyl chloride) and may be mobilized in groundwater if no control techniques are used. Therefore, it is important that bioremediation be performed above a low permeability soil layer and with groundwater monitoring wells downgradient of the remediation area (FRTR, 1998). Currently, only a few (mainly petroleum hydrocarbon derived) chemicals have established processes and are accepted widely as suitable candidates for bioremediation. In order to determine whether bioremediation would be an effective technique for treatment of contaminants within a given site, it is essential to know their biodegradability, mobility, frequency of occurrence, and partitioning characteristics (FRTR, 1998). Overall, the rate at which microorganisms degrade contaminants is influenced by: (a) the chemical properties of the contaminants and their concentrations, (b) temperature, (c) oxygen supply, (d) nutrient supply, (e) pH, (f) the availability of the contaminant to the microorganism (i.e. clay soils can adsorb contaminants making them unavailable to the microorganisms), (g) the amount of surface area provided for microbial attachment (i.e. plants with dense, extensive roots provide greater surface areas for attachment), and (h) the presence of substances toxic to the microorganism, (i.e. mercury; or inhibitors to the metabolism of the contaminant) (FRTR, 1998; and USEPA, 1996).

Biodegradation rates slow with decreasing temperature, and bioremediation processes may not, therefore, be effective in areas experiencing northern climates. Microorganisms will remain viable at temperatures below freezing and will resume activity when temperatures rise in spring. In some cases, the speed of the remediation process can be maximized through heating the contaminated site using warm air injection (FRTR, 1998). Aerobic conditions need to be maintained in treatment wetland systems as it is aerobic microbial metabolism that is mostly responsible for the detoxification of contaminants, the modification of nutrients and trace organics and the cycling of metals (Hammer, 1992). If wetland plants insufficiently aerate soils, forced air, liquid oxygen, or hydrogen peroxide can be added to contaminated sediments in order enhance aerobic microbial degradation (FRTR, 1998; and USEPA, 1996). Certain nutrients such as nitrogen, phosphorous, potassium, sulphur, magnesium, calcium, manganese, iron, zinc, and copper are required to sustain microbial growth. If any of these nutrients are not available in sufficient amounts, microbial activity will cease. The pH of the contaminated media will affect the solubility, and consequently the availability, of many contaminants. For example, many metals that are potentially toxic to microorganisms are insoluble at elevated pH (FRTR, 1998 and USEPA; 1996).

It is often argued that one of the principal functions of vegetation in constructed wetland systems is to facilitate ideal environments for microbial populations. They do so by providing oxygenated zones around their roots and capillaries (rhizospheres) and by providing surface areas for microbial attachment (forming biofilms) (Tousignant et al., 1999 and Hammer, 1992). Plants that are particularly effective at supporting microbial niches include those with lengthy, dense roots such as bulrushes (*Scirpus* spp.) rushes (*Juncus* spp), and Carex species (grasses). The more voluminous the roots, the greater surface area provided for rhizosphere development (thereby soils become more oxygenated) and microbial attachment. Hence, a healthy vegetative community is essential to effective and efficient bioremediation (Hammer, 1992). The aerobic microorganisms often in turn, enhance phytoremediation processes through symbiotic relationships which enhance metal uptake by roots (Lasat, 2000).

3.8. Physical Processes in Wetlands

A process common to all wetland systems is the physical settling of suspended particulate matter such as silt or clay, or fine particles of organic and/or inorganic matter. Suspended matter can severely degrade water quality and habitat for a host of reasons. Pollutants such as hydrocarbons, fixed forms of nitrogen and phosphorus, heavy metals, bacteria and viruses often bind to these particulates. When effective particulate settling is facilitated, toxicity is reduced as the pollutants settle to the bottom of the wetland along with the suspended solids they are absorbed to (USEPA, 1993; and Osmund et al., 1995a). The subsequent oxidation or reduction of these settled particulates then releases soluble forms of the pollutants to the wetland environment, which become available for adsorption or removal by soil, microbial populations and wetland plants (USEPA, 2000c). Therefore, it is wetland sediments that provide the largest sink for contaminant removal in wetland environments (USEPA, 1993; USEPA, 2000c; and Osmund et al., 1995a). In addition, increased turbidity arising from abundant suspended matter leads to decreased light availability, reduced photosynthesis of phytoplankton, algae and macrophytes leading to food chain disruptions, increased water temperature that causes decreased dissolved oxygen levels, decreased water depth, decreased visibility for aquatic wildlife (which especially affects visual predators), the clogging of fish gills, and the smothering of plants, eggs larva and food (Boulton and Brock, 1999; and Osmund et al., 1995c).

Wetland environments facilitate the gravitational settling of this matter by the physical nature of their lentic (still water) hydrology, and through the physical presence of the vegetative community, which reduces incoming water flows, traps and filters suspended matter, and stabilizes bottom sediments (USEPA, 1993; USEPA, 2000c; and Osmund et al., 1995a). Constructed wetlands are typically designed to detain turbid waters for long periods of time. This is because the longer wastewaters are detained, the more the suspended matter will settle and be subjected to the phytoremediation and bioremediation processes (Tousignant et al., 1999; USEPA, 2000c; and Liehr et al., 2000). This wastewater retention is known as the hydraulic residence time. Typically, constructed wetlands are designed to retain wastewater input for an average of five days, which generally facilitates suspended solid removal rates of greater than 60% (Witthar, 1993; and Osmund et al., 1995b). In many treatment wetland applications, a sedimentation pond is added upstream of the wetland cells to promote the effective removal of larger suspended particles in order to minimize the chance of clogging wetland cells. Establishing dense, deeply-rooting populations of vegetation in constructed wetland systems also facilitates settling by reducing erosion, obstructing flows and reducing velocities, and by physically trapping sediments (USEPA, 1993; and USEPA, 2000c). According to Osmund et al. (1995a), wetland vegetation typically traps 80 to 90% of sediment from runoff.

3.9. Wetland Benefits

Wetland ecosystems are diverse natural phenomena which provide countless benefits and perform a number of essential functions valuable to both the human and natural world. Marshes and swamps provide foods and habitats for the countless species which depend on them for survival. Bogs and fens provide the unique habitat needs of specialized and often rare species dependent on the distinctive conditions presented by these ancient systems. Benefits provided to both human and natural systems by the interactions of the physical, biological and chemical functions unique to wetland ecosystems can include: (a) wildlife biodiversity, (b) storm protection, (c) water purification, (d) commercial products, (e) recreation, education and culture, and (f) climate change control (Ramsar Convention Bureau, 2002b).

3.9.1. Wildlife Biodiversity

In terms of both plant and animal wildlife, wetlands ecosystems are among the top most productive environments on the planet (Mitsch and Gosselink, 2000). Wetlands provide for the habitat needs of countless species of birds, mammals, invertebrate and microbial species, and the majority of all herptiles (especially amphibians) and fish (Osmond et al., 1995a; and Mitsch and Gosselink, 2000). In the United States, wetlands, which only encompass anywhere from 3.5 to 5% of the country's land surface, supports 31% of plant species (USEPA, 2002d). Globally, freshwater wetlands, which cover approximately 1% of the earth's surface, support over 40% of the world's plant species and 12% of all animal species (Ramsar Convention Bureau, 2002b). In addition, it has been estimated that up to 50% of all endangered species in the United States inhabit and in some way depend on wetland ecosystems at some period in their lifetime (Mitsch and Gosselink, 2000).

Canadian statistics concerning biodiversity and wetland environments are more difficult to come by, however, according to the Canadian Wildlife Service (2002), approximately 60% of the species designated by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) as at risk of extinction are associated with wetland environments. Some examples are listed in Table 3.5.

Wetlands are especially important to millions of birds across the globe which depend on them to provide breeding and nursery sites, resting areas, and refuge from predators. According to Mitsch and Gosselink (2000), as much as 80% of the American breeding bird population, and over 400 species of protected migratory birds in the U.S. rely on wetlands for their survival and reproduction. It is estimated that over half of the tens of millions of migratory waterfowl in North America, and over two-thirds of the waterfowl in the U.S. depend on prairie pothole wetland ecosystems alone. Unfortunately, as a result of continuing wetland degradation and destruction, these bird populations continue to decrease. Over that past 15 years, continental duck breeding populations in the U.S. have declined 31% (USEPA, 2002e; and DU, 2002).

Table 3.5. Canadian Wildlife at Risk Associated with Wetland Habitat (adapted from Canadian Wildlife Service, 2002).

SPECIES	COSEWIC STATUS
Marshes	
Bigmouth Buffalo	Vulnerable
Black Tern	Vulnerable
Brindled Madtom	Vulnerable
Eastern Fox Snake	Threatened
Fowler's Toad	Threatened
King Rail	Endangered
Lake Chubsucker	Vulnerable
Least Bittern	Vulnerable
Prairie White Fringed Orchid	Vulnerable
Queen Snake	Threatened
Small White Lady's-slipper	Endangered
Short Eared Owl	Vulnerable
Spiny Softshell Turtle	Vulnerable
Spotted Gar	Vulnerable
Spotted Turtle	Vulnerable
Swamp Rose Mallow	Vulnerable
Swamps	
American Water-willow Common	Threatened
Greenbrier	Threatened
Cucumber Tree	Endangered
Louisiana Waterthrush	Vulnerable
Prothonotary Warbler	Endangered
Queen Snake	Threatened
Red-shouldered Hawk	Vulnerable
Wood Turtle	Vulnerable
Bogs	
Massasauga Rattlesnake	Threatened
Prairie White Fringed Orchid	Vulnerable
Spotted Turtle	Vulnerable
Fens	
Branched Bartonia	Vulnerable
Prairie White Fringed Orchid	Vulnerable
Spotted Turtle	Vulnerable

Wetland ecosystems also provide essential habitat for countless numbers of amphibians, reptiles and invertebrates which require aquatic environments for breeding, egg development, and larval growth. Many endangered and threatened amphibians and reptiles acquired their status as a direct result of wetland habitat loss (Shaw and Fredine, 1999; and USEPA, 1993).

Coastal marshes and estuaries provide for the fundamental needs of most commercial and game fish. According to Mitsch and Gosselink (2000), over 95% of U.S. commercial fish and shellfish are wetland dependent. Menhaden, flounder, salmon, sea trout, and striped bass are among the more familiar fish that breed and/or develop in coastal marshes and estuaries. In addition, shrimp, oysters, clams, and certain crabs also rely on these wetland environments for food and habitat. Freshwater fish also have very strict dependencies on wetland habitats. Currently, about one-third of native North American freshwater fish species are either endangered, threatened or of special concern. The declines in population of almost all these species are linked inevitably to wetland habitat loss (Shaw and Fredine, 1999).

3.9.2. Storm Protection

Wetlands help mitigate the potentially devastating effects of storm events. Storm events are threatening to natural ecosystems because they can cause flooding of downstream areas and cause erosion. Storm waters also carry nutrients and chemicals from surface runoff leading eutrophication and potentially toxic effects. Flooding has resulted in millions of dollars of property damage, as well as loss of life (USEPA, 2002e; and Azous and Horner, 2000).

Flood damage caused to homes, crops, businesses and other property can have devastating societal costs amounting in the billions. Lose of life can be very real consequence of flood events as well. Wetlands can prove exceptionally effective at flood mitigation as they intercept waters associated with storm events thereby preventing or reducing the effects of flooding to down stream areas. In 1991, an American study conducted by the Minnesota Department of Environment concluded that each acre of wetland intercepting and holding approximately 12 inches of floodwater would cost approximately $300 dollars to replace with technology doing the same function. At that time, the state of Minnesota was losing an estimated 5000 acres of wetland a year. Hence the cost of replacing this loss was calculated to be about 1.5 million dollars annually. Similarly, a study conducted on the wetlands paralleling the Charles River in Boston, Massachusetts, concluded that the flood mitigation provided by the wetlands saved the City about $17 million dollars a year in potential flood damage (USEPA, 2002d,e; and Ramsar Convention Bureau, 2002b).

Wetlands effectively intercept and store harmful storm waters, purify them, and then release them slowly back into the watershed. This slowing of stormwater momentum greatly reduces flooding heights, contributes to base flows of adjacent stream and lakes drying dry periods, allows for groundwater recharge, and dramatically reduces the water's erosive potential (Azous and Horner, 2000; and Witthar, 1993).

The effectiveness of wetland's ability to mitigate storm-related effects is dependent on factors such as its proximity to the incoming stormwaters, as well as the area of the given wetland site. For example, by virtue of their location, coastal wetlands buffer shorelines against oceanic tides and other stress brought on by tropical storms (Mitsch and Gosselink, 2000; and Witthar, 1993). According to the U.S. EPA (2002d), in general, an acre of wetland can store approximately about 1 to 1.5 million gallons of floodwater. However, the effectiveness of smaller wetlands in storm mitigation need not be discounted. Small wetlands presented together as a network can be equally if not more effective at floodwater retention and general storm mitigation than one large one. Also, wetland vegetation can have great bearing on its ability to mitigate the effects of storm events. A densely vegetated wetland can better reduce runoff velocity, as well as stabilize and/or remove more pollutants via the various processes of phytoremediation (Azous and Horner, 2000; Witthar, 1993; and Mitsch and Gosselink, 2000).

3.9.3. Water Purification

Natural wetlands help improve water quality by intercepting surface water runoff and removing or retaining nutrients, and metals, processing organic wastes, and reducing sediments before they reach open water. They accomplish water quality improvement through a variety of biogeochemical, phytoremediative, bioremediative and physical processes operating both independently and interactively with each other. They have been called the "kidneys of the planet" because of the natural filtration processes that occur as water passes through them (Lum, 1998). Consequentially, constructed wetlands designed and tailored to take advantage of these processes in more controlled, efficient manners are increasingly becoming accepted wastewater treatment alternatives.

The water purification functions provided by natural wetlands free of charge save national economies billions of dollars annually. For example, the hardwood swamps of South Carolina in the U.S. are known for their ability to cleanse and filter intercepted water of sediments, pollutants and excess nutrients. It would cost approximately $7 million dollars for construction alone to replace the functions provided by these swamps with water treatment facilities (USEPA, 2002d). Similarly, a 2,500 acre wetland located in Georgia is estimated by scientists to have annual savings of over one million dollars in water pollution control costs (Osmund et al., 1995a). As well, a $1.5 billion investment in wetland areas surrounding reservoirs in upstate New York resulted in savings of $3 to 8 billion dollars on development of new wastewater treatment plants.

3.9.4. Commercial Products

Products of commercial importance which are harvested directly from wetlands in Canada include: fish, shellfish, cranberries, timber (black spruce), wild rice, horticultural peat and sphagnum mosses, and various game animals (such as mink, muskrat, and beaver) (NRC, 2002). However, the economic value of wetlands in terms of their product contribution to the Canadian economy has not been thoroughly investigated in Canada.

The United States, however, has undertaken several studies on the economic benefits of wetlands in terms of their commercial product contribution. For example, in the U.S., 95% of harvested commercial fish and shellfish are dependent on wetlands for food or habitat. Likewise, 90% of all recreational fish species harvested in the U.S. are also either directly or indirectly dependent on wetland habitat. This is economically significant as the commercial and recreational fishing industry adds some $111 billion dollars to the American economy annually. The coastal marshes of Louisiana alone produce over 1 billion pounds of fish and shellfish, amounting to over $250 million in sales (Mitsch and Gosselink, 2000; USEPA, 2002d; and Osmund et al., 1995a). Muskrat pelts in the United States bring over $70 million to the American economy annually (Osmund et al., 1995a). Similarly, the international trade in crocodilian skins is worth over $500 million U.S. dollars a year (Ramsar Convention Bureau, 2002b). The estimated value of flood-tolerant pine and hardwood timber in swamp ecosystems in the United States is approximately $600 dollars per hectare (Mitsch and Gosselink, 2000).

Of course, the harvesting of commercial products from wetlands is not an exclusive practice to North America. In the eastern areas of the globe such as China, wetland species such as water hyacinth and smooth cordgrass are grown and harvested as valuable commodity crops used to produce energy (i.e. fermented to produce methane) fibre, and other materials. Rice, a common wetland plant grown all over the world, is the staple diet of more than half of the humans on the planet. Palms in African wetlands provide essential oils for cooking. The Nipa palm grown in wetlands in Asia produces up to 3 tonnes of sugar per hectare. In Europe reed-growing for roofing material is becoming a budding industry. In Brazil, the 1 million hectare Mamirauá Reserve is the source of wetland products worth $4.4 million U.S. a year. In Bangladesh, a 650,000 hectare mangrove forest produces 45% of the country's timber and all of its newspaper print (Ramsar Convention Bureau, 2002b; and Azous and Horner, 2000).

3.9.5. Recreation, Education and Culture

Wetland ecosystems draw in millions of people interested in recreational activities such as hunting, fishing, bird watching, canoeing; hiking and wildlife photography. It is predicted that North American tourism profits attributed to wetland ecosystems add tens of billions of dollars. For example, the Ramsar Convention Bureau (2002b) states that just birdwatching and waterfowl hunting in North America generate about $20 billion U.S. annually in economic activity. Similarly recreationsl fishing in coastal wetlands is believed to bring in $20 billion in the U.S. (USEPA, 2002d; Osmund et al., 1995a; and NRC, 2002).

Wetlands also serve as invaluable educational and research tools. These systems are often used as outdoor biological laboratories by all areas of the educational sector from primary school students to post graduates. The diversity and complexity of the biological and chemical processes which occur in wetlands provide fascinating opportunities for study, leading to the further understanding of these processes. These systems also provide habitat for many rare and interesting species which can be observed for study. Wetlands have also provided research opportunities in the field of cultural history and anthropological studies. The chemical properties and low decomposition rates of bogs ecosystems in particular have startlingly effectively preserved ancient artefacts including human bodies which have provided invaluable insight into the cultural practices and lifestyles of our predecessors, especially that of Native Americans. Wetlands continue to play significant roles in cultural aspects of various communities. They have formed the basis of many important local traditions, are linked to various religious and cosmological beliefs, and can also be great sources of aesthetic inspiration (Ramsar Convention Bureau, 2002b; NRC, 2002; Azous and Horner, 2000; and Mitsch and Gosselink, 2000).

3.9.6. Climate Change Control

Wetlands are able to help moderate the temperature extremes around them and in adjacent uplands. This is because they return over two-thirds of their annual water inputs to the atmosphere through evapotranspiration, which acts to cool the air (Osmund et al., 1995a). Perhaps more significant however, is the ability of wetland ecosystems to function as carbon sinks. The reducing conditions of bogs in particular act to sequester significant amounts of carbon dioxide that would otherwise enter the atmosphere and contribute to the global warming phenomenon. According to the Ramsar Convention on Wetlands, wetland environments may actually be responsible for as much as 40% of global terrestrial carbon storage (Whiting and Chanton, 2001; and Ramsar Convention Bureau, 2002b). Of course, when these systems are drained for whatever purpose, the increased oxidizing conditions which result increase organic matter decomposition, thus release significant amounts of carbon dioxide back into the atmosphere, amplifying the problem. Hence it is critical that these systems and their functions be protected from disturbance (Whiting and Chanton, 2001; and Kusler and Burkett, 1999). Unfortunately, despite their enormous benefits to both humans and the environment, anthropocentrically-based wetland threats continue to degrade and destroy these unique and valuable natural systems.

3.10. Treatment Wetland Design and Implementation

Generally, wetland designs which attempt to mimic natural wetlands in overall structure tend to be the most successful systems (Davis 1995; and Mitsch and Gosselink, 2000). This is mainly because highly engineered and complex technological approaches to wetland design frequently invite failure as with increased complexity comes increased chances of mechanical problems as well as increased maintenance requirements. For example, Subsurface Flow (SSF) wetlands are typically much more "engineered" and do not emulate natural systems as they contain no surface waters. They are also tend to contain a monoculture of effective, yet highly aggressive and often invasive wetland species such as Eurasian watermilfoil (*Myriophyllum spicatum*), duckweeds (*Lemna* spp.), and reeds (*Phragmites* spp.) and are commonly enclosed within greenhouse-like structures. As a result, these systems are frequently quite expensive to construct, maintain and repair (Davis, 1995; and Tousignant et al., 1999). Conversely, Free Water Surface (FWS) wetlands tend to be kept simple. Those which attempt emulate natural systems as much as possible (i.e. flow with topography, use native vegetation, and use natural energies such as gravity flow) are generally much more self-sustaining and resilient to fluctuations such as increased contaminant concentrations and climatic dilemmas such as flooding and drying. Natural treatment systems are also generally much more multifunctional than highly engineered systems; capable of providing benefits other than water purification such as habitat provision and passive recreational opportunities (Tousignant et al., 1999). Design considerations and implementation strategies for these naturalized, FWS systems include: (a) problem identification and quantification, (b) site selection, (c) site planing, (d) wetland size and structure, (e) wetland substrates, (f) site hydrology and hydraulics, (g) wetland vegetation, (h) wildlife considerations, (i) public considerations, and (j) operation and management strategies (Davis, 1995; and Daigle and Havinga, 1996).

3.10.1. Defining the Problem and Articulating Goals

The design considerations for constructed treatment wetlands systems are varied and most often dependent on the treatment goals to be achieved. For example, municipal wastewater treatment wetlands are most concerned with the effective reduction of suspended solids and BOD in controlled, highly monitored wastewater input, while stormwater wetlands must often be designed to facilitate a multitude of contaminants delivered in unpredictable wastewater flows (Lorion, 2001). It is extremely important that wetland designers have a full and thorough understanding of the contaminants they wish for the constructed wetland to treat, for it is the contaminant characteristics and their flow volume and concentration which will formulate the majority of project goals and objectives, as well as be critical in determining much of the design features of the site, including wetland size and structure, necessary retention times and the species of vegetation that should be established (Daigle and Havinga, 1996; and Tousignant et al., 1999). Contaminant characteristics include chemical properties, loading rates, behaviour (i.e. redox behaviour, solubility, potential susceptibly to phytoremediation), and toxicity. The regulatory requirements for effluent concentrations of the contaminants must also be taken into consideration.

According to Daigle and Havinga (1996), project goals should depict an ideal, overall project vision and set out measurable targets defining the intended short-term and long-term outcomes. These measurable targets to be fulfilled will serve as criteria for assessing the extent to which the goals have been fulfilled. The presiding goals for naturalized, multifunctional FWS wastewater treatment wetlands are commonly to create a system that provides a high level of treatment discharging relatively clean water, is inexpensive to build and simple to operate, and is self-maintaining. Depending on the project, other goals may be incorporated into the project plan, such as site restoration, biodiversity and habitat enhancement, ecosystem connectivity restoration, educational and recreational outcomes, and landscape aesthetics (Kadlec and Knight, 1996; Daigle and Havinga, 1996).

3.10.2. Site Selection

Detailed site inventories and analyses of the potential project locations should occur in order to determine the site's suitability to the project, and the projects suitability to the site and surrounding area. Site selection should be based on a number of factors including land use and access, the availability of the land, potential effects on neighboring residents (i.e. foreseeable odour or insect problems), and the potential effects on the ecological resources of the site and adjoining land (ie. vulnerable species, critical habitat, alteration of existing hydrologic regimes, potential introduction of noxious weeds, etc. (Davis, 1995; USEPA, 2000a; and Hammer, 1992).

The site should be located as close to the source of the wastewater as possible, and downgradient if at all possible so that wastewater can move through the system by gravity. The site should be above the water table, and not located in groundwater recharge areas or over fissured rock as to avoid groundwater contamination. In addition, areas that are prone to flooding, such as floodplains, should also be avoided since they may cause a flooding hazard and may be subject to erosion, scouring, sedimentation and high groundwater tables (Tousignant et al., 1999; Hammer, 1992; Kadlec and Knight, 1996; and Davis, 1995).

3.10.3. Site Plan

Once project goals and objectives have been completed, and the site location selected, the next step in the design process is to prepare a site plan. A site plan is an effective and practical way to spatially organize the site, and to graphically communicate intentions. The plan should contain a fairly detailed depiction of the layout and dimensions of the wetland itself, including the arrangement of wetland cells and berms and the general distribution and location of wetland vegetation, the location of inlet and outlet, and other control structures (i.e. muskrat netting), and the location of buffer areas. Additional features to be incorporated may include the delineation of special habitat zones, special research areas, educational areas (i.e. interpretive trails and signage) and other built elements to be integrated across the site (Daigle and Havinga, 1996; and Kadlec and Knight, 1996).

3.10.4. Wetland Size and Structure

The area of the treatment wetland is an important design consideration which is dependent on: (a) flow rates, (b) the nature and concentration of the contaminants to be removed, and (c) pollutant discharge requirements (U.S. BAMR, 2002; and Witthar, 1993). The wetland area required to treat a given contaminant is commonly estimated using the mass balance model, which is calculated using the following equation (Kadlec and Knight, 1996):

$$A = \frac{Q}{k} \ln \left(\frac{C_e - C^*}{C_i - C^*} \right) \tag{3.1}$$

Where:

A = Wetland area (m^2)

Q = Flow rate (m^3/y)

k = first order rate constant (m/y)

C_i = influent concentration (mg/L)

C_e = target effluent concentration (mg/L)

C^* = irreducible background concentration (mg/L)

According to Witthar (1993), the bottom line when it comes to treatment wetlands is the wetland design that produces the best water quality is the one with the largest ratio of treatment area to base flow, hence, the bigger the better. However, when considering size, designers must be cautious not to design the site so large that it is susceptible to drying. The shape of the wetland will be dependent on flow, topography and ownership constraints. Designers attempt to design systems which have large length-to-width ratios in order to: (a) minimize the occurrence of stagnant water (which invites problems such as avian botulism and mosquito production), (b) minimize and short-circuiting (faster, channellized water flow routes resulting in lowered residence times), and (c) increases the distance wastewaters travel, resulting in greater contact with sediments and vegetation, thus increasing purification efficiency (Kadlec and Knight, 1996; Witthar, 1993; and USEPA, 2000a). Ideal length to width ratios are in the range of 3:1 to 5:1. Constructed treatment wetland designs should flow with the natural contours of the landscape, and be integrated with the natural features of the site. Rectangular basins should be avoided whenever possible and wetlands should be indistinguishable, at first glance, from natural wetlands (Daigle and Havinga, 1996; and Davis, 1995). In addition, wavier edges increase surface area for wastewater contact. Sloping and grading should also be gradual, not rigid, in order reduce problems associated with erosion and public safety issues and provide for better wildlife access. Relatively flat littoral zones maximize areas suitable to emergent plant growth, as well as give them room to move 'uphill' should water levels raise (USEPA, 2000a; and Mitsch and Gosselink, 2000).

Most treatment wetlands are designed to support multiple cells. This facilitates the flexibility and use of different cells for different functions. For example, some designers plan for the cells nearest the influent source to facilitate the settling of sediment, for central cells to facilitate certain phytoremediation and bioremediation processes, and for final cells polish any residual contaminants or algae produced within the system. In addition, the use of multiple cells reduces problems associated with short-circuiting (USEPA, 2000a). Wetlands designed to accommodate wastewaters during colder months often need to be made deeper than conventional cells to prevent a solid freeze, and also need to be designed to facilitate longer retention times in order to allow adequate time for slowed biological processes to work (Reagin, 2002; and Davis, 1995).

Treatment wetland cells are commonly separated using vegetated berms typically created with the soils from the excavated wetland cells. The berms must be compact and stable, and high enough to contain wastewater volumes, as well as accommodate occasional high flows. Some planners choose to enforce berms with wire fencing, which also proves effective at preventing muskrats and other species from burrowing into the berms. Some designers may select to enforce the berms using riprap or erosion control fabric on the slopes. Some may also opt to create emergency spillways or install discharge pipes to mitigate seasonally extreme flows (Davis, 1995; and Tousignant et al., 1999). Berm width should ideally be two metres or greater in order to allow for easy maintenance and to help discourage aquatic mammals from burrowing through them. They should be planted as soon as possible in order to prevent erosion. During the initial operation of the wetland, erosion and channeling of the berms may develop, especially if wastewaters are added before berm vegetation has had a chance to mature. This should be immediately mitigated by raking the substrate, filling channels by hand and/or immediately reinforcing the affected areas with deep-rooting vegetation (Davis, 1995; Kadlec and Knight, 1996; and Hammer, 1992).

3.10.5. Wetland Substrates

As a general rule, constructed treatment wetlands, regardless of their location, should be lined with impermeable substrates that have saturated permeabilities greater than 10^{-7} cm/sec, a clay content of at least 15% and a plasticity index of at least 15%. The thickness of this substrate should be at least 30 cm. Often, compaction of on-site soils proves sufficient to line the wetland (Tousignant et al., 1999; Witthar, 1993; and Davis, 1995). Impermeable, synthetic liners made of substances such as plastics (i.e. high density polyethylene) and rubber can also be used to line the wetland. These linings are required to prevent infiltration and exfiltration factors such as groundwater permeation, which may upset carefully computed water budgets; it is extremely important that they be strong enough to deter root penetration and attachment. Of course, the use of synthetic liners increases the construction costs of the treatment wetland considerably. In addition, the liner must also be overlain with substrates suitable to supporting wetland plants. In subsurface flow wetlands, this substrate is gravel based, which can prove quite costly. However, surface flow wetland utilize soil substrates. The requirements for soil substrates are not as particular and typically, soils excavated from the cells prove quite adequate as topsoil for plant growth (USEPA, 2000a; Kadlec and Knight, 1996; Hammer, 1992; and Campbell and Ogden, 1999).

3.10.6. Wetland Hydrology and Hydraulics

Hydrologic factors in wetland design pertain to the water volumes received, its reliability and extremes, and its movement through the site. Water may enter into a constructed wetland from various pathways (precipitation, surface runoff, and wastewater influent) and may exit the wetland via effluent discharge, evaporation, and transpiration (Tousignant et al., 1999; and Davis, 1995). Water budget analysis will determine if the wetland cells are adequately sized in order to best prevent drying and/or flooding. For example, if additional water is required to avoid concentrating pollutants to toxic levels, the designer can try to capture more water input, or in some situations, a well can be added to supply additional water to the system (Davis, 1995; Kadlec and Knight, 1996; and Tousignant et al., 1999). If evaporation is occurring too rapidly, deterrents such as shading vegetation or windbreaks can be added. Commonly, the following formula is used to calculate flows received by the constructed wetland project (Campbell and Ogden, 1999):

$$R = rc[P-(E+ET)]WA \tag{3.2}$$

Where:

R = runoff (m^3/d)
rc = runoff coefficient
P = precipitation (m)
E = evaporation (m)
ET = evapotranspiration (m)
WA= watershed area (m^2)

Wetland hydraulics play an important role in determining factors such as wastewater retention. hydraulic residence time (HRT) is the time that it takes for the wastewater to pass through the wetland system. It is extremely important that wastewater detention times be adequate in order to facilitate the necessary bioremediation, phytoremediation and settling processes required to produce a clean effluent. The time the wastewater should be detained in the wetlands for treatment is dependent upon the treatment area of the wetland system as well as flow velocities and volumes. Treatment wetlands have been designed to have residence times ranging from 0.25 to 75 days, depending on these variables (Witthar, 1993). The average treatment wetland system supports HRTs of about 5 days. In general, the longer wastewater remains in the wetland system, the greater the treatment. The HRT of a given wetland system can be determined using the following equation (Campbell and Ogden, 1999; and Mitsch and Gosselink, 2000):

$$HRT = And/Q \tag{3.3}$$

Where:

HRT= hydraulic residence time (days)

A = area of the system (m^2)

n = porosity of the medium (i.e. is typically 1.0 for FWS)

d = depth of submergence (m)

Q = average flow through the system (m^3/day)

Similarly, the chemical loading rate is used to calculate how much contaminant is being received by the system to be treated. Calculating the loading rate is important in order to gauge if whether the system will be capable of treating the loading received. The chemical loading rate of a given wetland system in g/m^2 day can be determined using the following equation:

$$CLR = QC_i/A \tag{3.4}$$

Where:

CLR= chemical loading rate (g/m^2 day)

Q = design flow (m^3/d)

C_i = influent concentration (mg/L)

A = area of system (m^2)

3.10.7. Vegetation

Wetland vegetation is one of the most important components of treatment wetland systems because wetland plants effectively facilitate: (a) phytoremediation processes, (b) bioremediative processes by facilitating microbial growth, (c) sedimentation by slowing flows, stabilizing sediments and filtering suspended solids, and (d) biogeochemical reactions by oxygenating the water column and sediments (EC, 2000).

FWS treatment wetlands should be designed to be as 'natural' as possible through the establishment of a wide variety of local, non-invasive wetland species in the treatment wetland site. Native plants are defined by Environment Canada (2000) as species that occur naturally in an area since the last glaciation period, and prior to European settlement. Since species can be genetically differentiated across a range of spatial and environmental gradients (i.e. climate, soil, hydrology, topography), it is important to select species that are native to the local region and of local source (EC, 2000).

The importance of establishing locally native species in treatment wetland systems is particularly emphasized. Estimates on what constitutes a "local" source range from 10 km to upwards of 200 kms from a project site. Others argue that climate is the key consideration with plants sourced from distant locations being appropriate so long as they occur in similar latitude. As a rule of thumb, plants for the treatment wetland should be obtained as close to the project site as possible (Daigle and Havinga, 1996). The reason local species are so highly desirable is that they are capable of providing many advantages. Firstly, the more locally adapted species the system contains, the more likely the system will become more self-sustaining, requiring less maintenance. This is because local species are genetically adapted to local climate, soils, diseases and surrounding plant and animal communities, and are therefore most likely to do well. Likewise, local wildlife are most adapted to local species for food and habitat, hence if habitat provision is a project goal, the use of local species is essential. Overall, the establishment of local species can help support local wildlife populations and ensure the viability of the system and reduce the chance of failure as multiple species diversity would be clearly more responsive to variations (i.e. loading, climatic, etc.) than would monocultures (USEPA, 2000a; Reed et al., 1995; and EC, 2000).

Some designers purposely choose to establish exotic, robust species notorious for adapting to virtually any environment in their wetland sites as these species (i.e. reeds (*Phragmites* spp.) and duckweeds (*Lemna* spp.)) are not only practically guaranteed to grow successfully, but, by their very nature, are typically excellent pollutant removers. Some scientists even choose to establish genetically engineered plants in their systems which are specially designed for particularly efficient pollutant removal. However, these species also have grave tendencies to become overly invasive or aggressive. With a lack of natural competitors, and occasionally allelopathic nature, these exotic species can take over areas beyond those for which they were intended. Invasive species can out-compete native vegetation and destroy the natural diversity of the community the created system in situated in (Thunhorst, 1993).

Establishing high vegetative diversity in FWS treatment wetland systems similar to that of natural systems is also important. As stated, some highly engineered, enclosed wetlands (i.e. SSF) are purposely planted with very few, to as little as one species, in order to increase the site's efficiency and the control a manager has over the site (Boulton and Brock, 1999). However, in open, outdoor systems, the establishment of vegetative diversity can prove quite advantageous as it increases the likelihood of recovery from disturbance, increases the ability of the system to resist invasive species and pests and is aesthetically pleasing. Also, in some cases, having high plant diversity has been proven to actually enhance the removal of certain contaminants, such as iron and acidity in mine drainage wetlands (Stark and Williams, 1995). In addition, the numbers of wildlife attracted to a wetland generally increases as vegetation diversity increases, which is valuable especially if habitat provision a specific site goal. Wetlands with healthy and abundant aquatic plants also have greater macroinvertebrate species richness and microbial populations, which can lead to greater bioremediation activity (Davis, 1995). It is a general rule that in any type of plant community establishment, despite its purpose, it is more logical to establish a higher system (more diverse and complex) than a lower system (homogeneous and simple) as a higher system has the ability to also support the functions of a lower system and the higher systems are inherently more stable and able to withstand disturbances. They are therefore more sustainable as they are less likely to require intensive management to ensure effective operation (Boulton and Brock, 1999).

3.10.7.1. Wetland Community Modeling. One of the best ways to ensure one is establishing wetland vegetative communities adapted to the local region is to model the treatment site after a local, natural wetland most similar to the treatment wetland's type (i.e. freshwater marsh). It involves the surveying of plant communities inhabiting a local, natural wetland, including vegetation composition structure, abundance and dynamics. The community surveyed then acts as a template to guide the selection of appropriate species for the treatment wetland plantings (Daigle and Havinga, 1996; and Hoag, 2000). According to Daigle and Havinga (1996), the fundamental premise behind ecological community modeling is that similar physical environments within the same local area or bioregion typically sustain or give rise to similar assemblages of plant species. Hence, a plant community indigenous to a local area or bioregion whose underlying environmental gradients most closely resemble those of the treatment wetland site is probably most likely to succeed there (Daigle and Havinga, 1996; and Hoag, 2000).

However, some of the plants in the natural community may be sensitive or rare which would obviously not be suited to a contaminated site. Others may not be commercially available for planting. The natural wetland will likely also contain undesirable species such as invasive or nuisance species such as poison ivy which should obviously not be modelled. In addition, once the treatment wetland vegetative communities are established with the appropriate species, energy should not be spent on relentlessly maintaining the established community composition and structure, as in the end, natural processes will ultimately determine which species thrive and reproduce and which do not and which later successional species will successfully migrate in. Rather, the community model should simply be used as a tool to guide the wetland designer in his or her selection of plant species to be established in the treatment site (Daigle and Havinga, 1996; and Hoag, 2000).

Once the model community is sufficiently surveyed, and the plant lists and associated research information compiled, the next step is to create the planting plan for the treatment site using the model as a guide, however, the planting plan should be 'enhanced' to best suit project goals. Plant selection, abundance and distribution in the treatment wetland site should be specifically tailored to tolerate and address the contaminants of concern to be received by the treatment system. For example, if research on three specific species identified in the model revealed that they are effective at extracting a contaminant of concern in the site, despite the fact that these three species may not have been highly abundant in the model, the designer may choose to dominate treatment wetland cells with these species in order to best facilitate project remediation goals. The same applies for secondary project goals such as erosion control, aesthetics and habitat provision (Daigle and Havinga, 1996).

In some cases, the species lists obtained from the model site may prove inadequate for project purposes (i.e. no metal extractors were in the model site), and additional species lists are required for the plan. Even in these cases, designers need not resort to establishing exotic species in the site to achieve project goals. There are many additional resources other than the model site that can be explored to enhance the model plant lists. Many NGOs and various public agencies such as local conservation authorities, naturalist or field botanist groups, and regional natural history texts often have or contain extensive knowledge of native species compiled for local areas

(Daigle and Havinga, 1996; and Kusler, and Kentula, 1989).

If the model community plant lists and other native plant lists acquired lack plants which are tolerant of or capable of remediating the specific contaminates of concern at the future constructed wetland site, one way to identify additional suitable plants for the wetland is to look at plant communities growing naturally and successfully in local areas contaminated with similar pollutants to those the constructed site intends to treat. Or, if the undeveloped, treatment project site is already receiving intended contaminated wastewaters (i.e. degraded wetland receiving wastewaters is being enhanced to better treat contaminates), reviewing all native, naturally occurring vegetation identified on (or directly adjacent to) the site during preliminary vegetative inventories of the project site can also prove extremely useful for enhancing site plant lists (Diekelmann and Schuster, 1982).

The investigation of naturally occurring vegetation in contaminated sites can prove extremely useful as it identifies plants that are capable of surviving contaminated conditions. Many of these plants also prove capable of contributing to the degradation of the contaminants. For example, if native *Scirpus lacustris* is found thriving in a metal-contaminated site, it can presumed that this plant can tolerate the conditions, and may very likely hyperaccumulate the metal. This type of investigation is often referred to as "Forensic Phytoremediation", which is defined as the investigation of naturally-revegetated contaminated areas to determine which plants have become established and why, and to determine the impact of these plants on contamination (USEPA, 2000b). The use of forensic phytoremediation not only identifies native plants which are adapted to local climatic, biophysical and biological conditions, but has the added benefit of identifying plants adapted to specific contaminates of concern, which will go a long way towards ensuring the success of the designed system. Of course, forensic phytoremediation often requires extensive characterization of the type and degree of contamination at the site, which does not only consume much time, but can prove rather costly. Ideally, contaminant characterization of the contaminated site would already be known, such as if the site *is* the future site of the treatment wetland, or if contaminant characterizations have already been conducted by previous agencies who would be willing to share their data (USEPA, 2000b; and Kadlec and Knight, 1996).

3.10.7.2. <u>Plant Sources and Planting Techniques</u>. There are four main planting techniques used to establish vegetation in constructed wetland sites: seeding, transplanting whole plants, planting commercial stocks and planting clonal cuttings. The planting techniques chosen will be highly dependent on plant availability. There are also three main potential sources of plants for site establishment are: commercial nurseries, local seed collection, and donor sites.

3.10.7.2.1. Direct Seeding. Seeding is possibly the most cost-effective method of establishing vegetation in a site, as purchasing partially or full-grown plants can prove quite costly (EC, 2000). Seeds can also be collected from local sites, (in some cases, even the model site), saving additional costs. In addition, seed collection causes minimal damage to the donor site as compared with transplanting. The other advantages of seeding are that it requires minimal planting effort, and that collected seeds can be stored for several years (Davis, 1995).

However, there are several disadvantages to this community establishment methodology, most of which relate to its unpredictable nature, which decreases probability of success. Seeds are vulnerable to pests (i.e. eaten by birds) and the elements (i.e. rains and winds wash seeds away). New growth is also very vulnerable to the elements (i.e. pests, cold, wet), reducing survival success. Some literature suggests that planners establishing a site via seeding should only expect a 30% survival rate for the plants (Hoag 2000; and EC, 2000). As well, plants established by seeding take long time to mature as compared to transplanting whole plants, hence if time restrictions exist for site establishment, this methodology may not be ideal.

If seeds are being collected as opposed to purchased, seed collectors must be well aware of the appropriate seed collection times for all the different species being collected as well as how they germinate. Collectors should also be aware of proper handling, processing, storage, and treatment techniques for each species (Daigle and Havinga, 1996).

Soil from a nearby wetland resembling the conditions of the project site can also be used as a source of plants since this soil will contain seeds of a number of native species that are well-adapted to local conditions. In this methodology, cores of wetland soil from a donor marsh about 8 to 10 cm in diameter are transplanted directly into the constructed wetland substrates (Davis, 1995). Of course, core extraction can prove quite damaging to the donor site, and is generally not recommended. If conducted, the appropriate approvals and permits for the soil extraction must be obtained from the appropriate agencies before proceeding. Also, collecting, transporting, and planting the soil masses can be costly and time consuming (Davis, 1995; Kusler and Kentula, 1989; and EC, 2000).

3.10.7.2.2. Commercial Stocks. Many treatment wetland site planners chose to purchase the majority of their plant stock from local nurseries carrying local species. Commonly, plants obtained from nurseries are available in the form of tubers, rhizomes, container plants and bareroot plants. Both tubers and rhizomes or 'rootstocks' are large, dormant, underground stems which providing food storage for the plants. Container plants are young plants that have their roots surrounded by soil in a peat, fibre or plastic pot. Bareroot plants are young plants with very little or no soil surrounding roots (EC, 2000).

Should commercial plants be the primary source of the wetland site, prior to completing plant selection for the site, area plant supplier inventories should be examined to assess the availability of required plant stock. It is highly likely that two or more nurseries will need to be relied on in order to obtain a well-rounded selection. Substitutions for species that are out of stock should strictly prohibited unless confirmed that they comply with the model plant lists (Daigle and Havinga, 1996).

Tubers and rhizomes should be securely planted by digging a hole with a shovel or spade, and pushing the root into the hole approximately five centimetres below the surface of the wetland substrates. Soil must then be firmly packed over top of the root. Commonly, site establishers make the mistake of planting emergent wetland plants in already flooded waters. However, frequently, too much water creates problems for newly planted wetland plants as roots are not fully developed hence the inherent mechanisms which provide for anoxic tolerance are too immature to handle the stressful environment, and the plants often subsequently 'drown'. Therefore, for best survival and growth, it is important that the wetland substrates not be flooded until plants are significantly developed (10 to 12 cm or 2 to 3 months of growth) (Davis, 1995; and Tousignant et al., 1999). In addition, should wastewaters set for treatment be particularly impure, it would be a good idea to give the plants even longer to mature before the waters are added in order to reduce the chance of the plants being overcome by stress. Perhaps a gradual rather than sudden input of wastewater could be facilitated, again, to reduce the shock to the vegetation (Hoag, 2000; Kusler and Kentula, 1989; and Davis, 1995).

As for spacing, propagules are usually spaced at 0.3 to 1 metre intervals, depending on the spreading rates of individual species, as well as how rapidly complete vegetative cover of the site is desired. In addition, propogules are ideally planted in rows running perpendicular to the direction of flow in order to improve coverage and reduce channelling while the vegetation is filling in However, it should also be noted that overly uniform arrangements of plants should be avoided as this may result in the wetland sporting an artificial appearance (Hoag, 2000; and Daigle and Havinga, 1996).

Some of the main advantages of obtaining wetland plants from commercial nurseries is that survival success of the plants significantly higher than that of seed propagation (EC, 2000). However, purchasing species can obviously prove quite costly, especially if the site is rather large, requiring expansive buffer plantings as well. Another downfall to obtaining plants from nurseries is stock availability, as native plants have yet to be a commodity of significant demand. In addition, in many cases, wetland nursery plants are pampered in ideal conditions and are often not obtained from local, wild sources, hence nursery plants are often not as hardy or locally adapted as plants transplanted from local natural areas (Daigle and Havinga, 1996). For these reasons, constructed wetland sites are rarely established solely by nursery-grown plants. Rather, nursery plants are generally purchased to compliment the wetland with highly desirable species unattainable by other means such as slow-growing ornamentals (i.e. blue flag (*Iris versicolor*), cardinal flower (*Lobelia cardinals*)), or species which if removed will cause major disturbance to the donor site (i.e. submerged species) (EC, 2000).

3.10.7.2.3. Transplanting. Transplanting involves collecting seedlings single or clumps of adult plants, or rhizome sections from donor wetland sites and planting them directly into the project site. Transplantation of plant species has a multitude of advantages. Most significant perhaps is that transplanting plants from surrounding areas which are genetically adapted to local environmental conditions (i.e. soil type, microclimate, disease tolerance, etc.), can significantly increases the chance of successful plant establishment as even populations in close proximity to each other can become somewhat distinct in ways that allow them to survive. In addition, this methodology sees significant cost savings as compared to the purchasing of seed or nursery plants. This planting methodology can be very labour intensive and may take longer than conventional planting, however, time is usually less of an issue with this method as successful transplantation can be carried out from early spring on through to late autumn (EC, 2000). In addition, transplanted plants establish themselves readily, and relative survival success is the highest of all planting methodologies. Also, the placing of full-grown plants in the wetland as opposed to seedlings or just seed is also very advantages as the more mature the site, the more quickly it will become effective at treatment (EC, 2000; and Hoag, 2000). In addition, an added benefit is that all transplanted species will contain soil around their roots which contain dormant seeds of plants local to the site. These seeds will germinate when the conditions are right, adding to the diversity and total vegetative cover of a subject area. In addition, the transported soils will also contain microbes, invertebrates, eggs and so on which will help accelerate the establishment of microbial, insect and herptile communities in the site significantly (Hammer, 1992; Kusler and Kentula, 1989, and Hoag, 2000).

Transplants can be removed relatively easily by shovel. Plants with dense, rhizomatous roots systems such as *Juncus* and *Scirpus* species can be split into sections, creating several small plants. It is best to transplant species in clumps with numerous stems and soil surrounding their roots. This helps weigh down the plant and provides a stable base for the root mass. Also, leaving the soil on the plug can increase establishment success by about 30% (Hoag, 2000; and EC, 2000). As with nursery plants, it is best to transplant species into unflooded substrates. Wastewaters should be added gradually after the plants have had a chance to re-establish themselves in their new environment. Taller species should have their stalks trimmed until roots have re-developed in order to minimize damage caused by windthrow (Tousignant, et al, 1999; EC, 2000; and Davis, 1995).

About the only major disadvantage of this methodology is the potential damage which can be caused to the donor site (EC, 2000). Collection should never take place in highly sensitive sites, sites that contain rare species, or sites that contain vulnerable habitat. In addition, collection should never take place without permission of the landowners. It is suggested by Environment Canada that 95% or more of the donor vegetation area be left to regenerate in order to minimize damage to the donor site. In addition, in order to aid recovery, collection should occur from a variety of locations, and not be concentrated in one area. Ideally, donor sites would be those designated for future clearing, development or major alteration, deeming donor site damage a non-issue, as collection could almost be viewed as a rescue (EC, 2000; and Diekelmann and Schuster, 1982).

3.10.7.2.4. Clonal Plants. Clonal plant propagation from stem cuttings or root divisions is commonly used by nurseries to reduce production costs, as well as to perpetuate desirable plant forms, sizes, or leaf and flowers colour. Clones are genetically identical to the plant from which they are sourced. However, when using clonal cuttings, aesthetic uniformity is gained at the expense of genetic diversity. Excessive use of clones can reduce the long-term survival of site plantings and their offspring by reducing genetic variability within the population, rendering plants less resilient and adaptable to environmental change, hence the use of clonal plants should be avoided unless the plants cut reproduce via cloning anyhow (i.e. *Populus* spp.) When this technique is used, stem cuttings should be collected from a variety of different plants to enhance the plantings genetic diversity (Daigle and Havinga, 1996; and Hoag, 2000).

3.10.8. Wildlife

Some constructed wetlands have the additional project goal of enhancing/facilitating wildlife habitat. This can be accomplished by a variety of means. Vegetation can be tailored to suit the habitat needs of specific or broad ranges of species. In addition, landscaping techniques can also be tailored to better accommodate wildlife populations. For example, if attracting waterfowl is desirable, roosting areas should be created, wetland edges should be made to slope gradually to make water access easy, and sufficient open water areas as well as cover should be facilitated. Likewise, some wetland mangers may need to deter certain populations of wildlife from inhabiting the site. For example, muskrats and beavers can damage a treatment system by burrowing through berms and altering water flows. Overabundant waterfowl can cause excessive nutrient loading, overgrazing, trampling, and disease problems (USEPA, 2000a,c). Some sites may need to be tailored to prevent wildlife populations from utilizing the site or portions of the site altogether, especially if the waters being remediated prove to be acutely toxic to wildlife. Potential chronic toxicity should also be considered as adverse health impacts and mutations brought on by contaminants can be equally adverse to populations as direct mortality. Other factors such as water clarity, bacterial presence and pH (should be greater than 4.0) should also be weighed for wildlife suitability (USEPA, 2000a,c; and Osmond et al., 1995a). In addition, in some instances, even small concentrations of certain substances (i.e. persistent organic pollutants (POPs) such as DDT and PCBs and certain heavy metals such as the organic methyl mercury), which may not be harmful of themselves can bioconcentrate or biomagnify in the environment depending on their partitioning tendencies, resulting in adverse affects to wildlife (USEPA, 2000c).

Should the wastewater being treated present a tangible threat to wildlife, hazing or wildlife exclusion devices such as fencing or noise making devices may need to be installed in the site. In some cases, populations may just need to be restricted to areas where the water quality is acceptable, such as in the latter cells. This can be accomplished with fencing and other devices, or by limiting open water zones and by using dense stands of emergent vegetation. Selecting vegetation with little food value may also help deter inhabitants from certain cells (USEPA, 2000c; and Davis, 1995). If the influents are so toxic that wildlife must be completely restricted from the system, then obviously, an open, naturalized treatment system was not an appropriate treatment option for that particular project and a different type of system such as an enclosed solar aquatics facility or a more traditional treatment facility should have been considered. This drives the point that it is it is imperative that nature of the contamination being treated be thoroughly characterized before all other design aspects and project goals such as wildlife facilitation are considered.

3.10.9. Public Consideration

Consideration of how the treatment site will affect public stakeholders and how the site can facilitate visitors is essential when designing the site layout. Potential concerns to take into account include things such as drinking water contamination, unpleasant odours, mosquitoes, access by small children and other safety and health issues. Local stakeholders such as adjacent property owners should be consulted prior to site development in order to ensure all concerns are appropriately accommodated (USEPA, 2000a; and Kadlec and Knight, 1996).

Likewise, the potential effects of public access to the site on the site itself must also be considered and accommodated. One of the most effective ways to facilitate public access to a site is through the use of trails. Trails can be used to concentrate hikers, birdwatchers and other nature enthusiasts to a narrow movement corridor thereby reducing the amount of random trampling throughout the site. They can also help direct people away from sensitive habitats while simultaneously leading to the best vantage points and areas of interest. In this way trails can also help complement educational and/or recreational objectives by providing a range of opportunities to experience and appreciate the different areas of the site (Daigle and Havinga, 1996; and Campbell and Ogden, 1999).

The key challenge is to maximize safe accessibility while minimizing or altogether avoiding conflicts and or interference with site's ability to cleanse incoming influents, or other project goals such as habitat facilitation. Prior to planning the layout of trails across the site, the sensitivity of existing areas should be assessed. Things to look out for include areas which may be particularly susceptibility to trampling, soil compaction or erosion such as the berms or marginal areas of the cells. Areas supporting sensitive species or habitat and areas which may pose dangers to the public, especially children, such as steeply sided slopes, and open water areas should also be identified (Hammer, 1992; USEPA, 2000a; and Daigle and Havinga, 1996). Once investigated, trails can be planned to steer traffic away from these areas or be designed to minimize impacts (i.e. create boardwalks, bridges over wetter areas, stairs on slopes, etc.). Additional deterrent mechanisms such as signage, fences, or vegetative barriers (i.e. thorny species) may also be necessary in areas where access is particularly undesirable (Daigle and Havinga, 1996; Kusler and Kentula, 1989; and USEPA, 2000a).

If enhanced educational outcomes are a project objective, site managers should consider installing interpretive signage along the trails and immediately around the wetland site, explaining the function of the system and other information pertaining to site goals such as the use of phytoremediation, the benefits of naturalisation, the importance habitat creation and so on (USEPA, 2000a). The types of recreational activity (i.e. hiking, cross-country-skiing, biking) anticipated to be facilitated by the trails should also be considered when designing the trail layout. For example, many site managers choose not to facilitate cycling on site as this high impact activity can quickly degrade trails, cause compaction, and pose a hazard to other users (i.e. if trails have bad sight lines) (Daigle and Havinga, 1996; Hammer, 1992; and Campbell and Ogden, 1999).

If trails are lengthy, one may consider placing appropriate resting stops along the paths. Picnic tables and benches could also be considered. If littering is anticipated to prove problematic, site managers should consider placing trash bins on site. In urban, relatively heavily used areas, lighting may need to be incorporated into the site design for safety reasons. However, planners should avoid lighting forest interiors or other sensitive habitats as some plant and animals may be adversely affected by prolonged exposure to artificial nightlight. Even in less populated areas, user safety should always be a key design consideration. Good visibility should always be maintained along trails and other areas of the site. Open zones in which plants not exceeding 60 to100 cm in height should be maintained along both sides of paths. Once again, thorny species such as rose and raspberry could be considered for between paths and bordering adjacent woods in order to reduce access to hiding spots (Campbell and Ogden, 1999; and Daigle and Havinga, 1996).

3.10.10. Develop an Operation and Management Strategy

With any site development project, the project does not end with implementation. In order to properly address potential aftercare considerations for the treatment wetland site, a proper operation and management strategy (O&M) must be developed. The strategy should above all be adaptive, as it may have to be refined, revised, or altogether rethought as the site evolves, the success or failure of techniques becomes evident, and as new management needs become evident. The O&M should clearly delineate management staff roles and responsibilities for both the anticipated short-term and long-term operation of the site. It should also include the roles of anticipated community volunteers. It is also imperative that the O&M contain a budget which spans the entire lifetime of the site. Many project managers make the mistake of pooling nearly all funding into implementation and temporary aftercare, without thinking of their responsibilities to long-term aftercare and monitoring (Daigle and Havinga, 1996; Moshiri, 1993; and Hammer, 1992).

For constructed treatment wetland sites, the shorter, aftercare activities often concern the nurturing of new plantings to ensure optimal growing conditions and maximum survival rates. The first year or two following a planting is the most critical stage in the vegetative community's development. Root systems have yet to become firmly established, and are thus vulnerable to both drought and desiccation as well as drowning due to over saturation. Tender seedlings and saplings are particularly susceptible to herbivory and predation. Also, exposed soils are vulnerable to erosion (Daigle and Havinga, 1996). Hence, short-term management activities commonly executed during this initial vulnerable period include: (a) watering and mulching, (b) cell water level management (to reduce drowning), (c) flood relief (i.e. reduce water retention levels of cells), (d) protection from predation and herbivory (i.e. temporary fencing), (e) aggressive species and noxious weed control (to minimize competition and abide local by-laws. Sometimes use pesticide application, although this is not recommended except in extreme circumstances), (f) placement of erosion control measures (i.e. mulching, woodchip application, hay application, sedimentation netting, temporary filtration devices such as gravel berms to reduce added turbidity etc.), (g) pruning and trimming (i.e. to reduce windthrow), and (h) the removal and replacement of deceased or vandalized plants (Daigle and Havinga, 1996; Moshiri, 1993; Reed et al., 1995; and Davis, 1995).

Longer-term O&M will be required to ensure the system is performing as it was designed to perform, not just for a few years following site development, but for the duration of the project's anticipated life. Operational maintenance issues which should be addressed in the O&M could include: (a) scheduling for the cleaning and maintaining of inlet and outlet structures, valving, and monitoring devices, (b) the inspection of embankments and structures for damage (i.e. vandalism, after heavy storm events, muskrat damage, etc.), (c) the inspection of sediment accumulation and anticipated dredging requirements, (d) the operation of water levels (specify acceptable ranges of fluctuation), (e) the inspection of proper inlet and outlet water flows and retention times (i.e. stagnant water, depressions in berms causing premature releases, debris blockage, etc.); and (f) wastewater application schedule, if this is part of the system design. Short, high-flow discharges to a wetland are more likely to erode or damage established vegetation than lower velocity, more continuous flows (Davis, 1995; Moshiri, 1993; Reed et al., 1995; and Tousignant et al., 1999).

Beyond the immediate aftercare, long-term management strategies for the vegetative communities of the site in particular will also be required to ensure the stability and sustainability of the site (Daigle and Havinga, 1996). Ultimately, the aim of the creation of a naturalised site is to create a reasonably self-sustaining natural community. However, on-going management and intervention, although minimal, will be inevitable. The purpose of long-term management is to help maintain the integrity of the established community, and to eliminate, control or mitigate degradation (i.e. pests, humans, weather, etc.) (Daigle and Havinga, 1996). A long-term management strategy for the vegetative communities of the site may involve some or all of the following considerations: (a) the periodic control and removal of noxious weeds and competing vegetation, (b) pest management strategies (i.e. mosquitoes, abundant herbivores), (c) controlling or restricting human and pet access to particularly vulnerable areas of the site such as wetland edges and berms, (d) the management of formal edges, (e) damage control strategies (i.e. erosion/compaction mitigation, removal/replacement of abundant dead vegetation as a result of sporadic disturbance (i.e. drought, storm, chemical overloading etc.) to avoid decreases in site efficiency or unacceptable increases of BOD levels due to excess decomposition activity), and (f) some sites require vegetation to be harvested as some species become no longer effective once they have reached a point of saturation (Daigle and Havinga, 1996; Moshiri, 1993; and Davis, 1995).

The need for these and other measures must be determined through regular inspections and monitoring. Monitoring in particular is an essential component of any O&M as it is only through monitoring that project managers can gauge the success or failure of the site, and the degree to which project goals are being met. An effective monitoring program will detect both positive and negative changes to the system, allowing management to respond accordingly with the appropriate management interventions, many of which have been discussed above (i.e. remove excess litter, decrease influent flows, etc.) (Daigle and Havinga, 1996; Moshiri, 1993; Reed et al., 1995; Tousignant et al., 1999; and Davis, 1995).

The primary goal of most treatment wetland systems is water quality improvement. Hence it is essential that a practical, long-term water quality monitoring program be specified and budgeted for in the O&M to gauge the success of this goal, and to trigger mitigative measures should problems be identified. Monitoring will also need to occur to gauge the progress of other project goals such as habitat facilitation, public education and so on. Sampling methods may range in sophistication from simple, periodic insect and wildlife censuses, plant inventories and photograph documentation to detailed water quality chemical analyses, hydrologic measurements, GIS applications and so on. The more accurate the data collected, the more likely the site will be managed effectively. However, excessive analysis can be extremely time consuming, costly, and even harmful to the site (i.e. frequent plant surveying can cause trampling and compaction). Hence project managers need to be able to balance the need for accuracy with monitoring efficiency. One should strive to keep the monitoring strategy as simple as possible without comprising the integrity of the site (Daigle and Havinga, 1996). The monitoring phase of the project will ideally never end, but will scale down once managers are satisfied that the wetland is meeting initial objectives. This peak performance should typically not be anticipated until after the system has matured, this usually taking two or three growing seasons. Strategies that try to short-circuit this natural maturity often fail (Bastian and Hammer, 1993; Bennet, 1994; Davis, 1995; Balla, 1994; and Daigle and Havinga, 1996).

4. METHODOLOGY

4.1. Selection of Vegetation

4.1.1. Wetland Community Model

A natural wetland system located approximately 200 m downstream of the Burnside treatment wetland was selected to act as the vegetative community model for the Burnside wetland site (Figure 4.1). The selected model is a riparian, open water marsh dominated by emergent macrophytes (Figure 4.2). Although thriving and appearing healthy and highly productive, the site did show many visual signs of disturbance including orange staining and iron particulate coagulations (Figure 4.3). This site, however, was selected as the vegetation model for the treatment wetland site for several reasons:

(a) it supported many locally-native wetland plant species adapted to local conditions,

(b) it belongs to the same water system and therefore was similar in biophysical characteristics and environmental gradients (i.e. substrates, climate, etc.) to that of the treatment site,

(c) as the Wright's brook marsh received high iron, manganese and ammonia loading, it was likely that plant species selected from the model would be capable of surviving the contaminant loading received by the treatment wetland site,

(d) in addition to increased survival potential, species selected from the model would also likely have potential for hyperaccumulation of the contaminants of concern as per the principles of forensic phytoremediation (USEPA, 2000b), and

(e) modelling the treatment wetland after a naturally occurring wetland area (that is a part of the Wright's brook ecosystem) would ensure habitat continuity, and preserve biological integrity by not introducing species to the system that would not occur there naturally and thus the treatment wetland would appear as a natural extension of the Wright's brook ecosystem.

Although deemed appropriate to act as the vegetative model for the treatment site as per the above arguments, it should be noted that ultimately, the Wright's brook marsh was rejected as the 'reference' or control site for the monitoring and analysis aspects of the study. Instead, a more pristine wetland located within the Waverley Game Sanctuary was selected for this purpose.

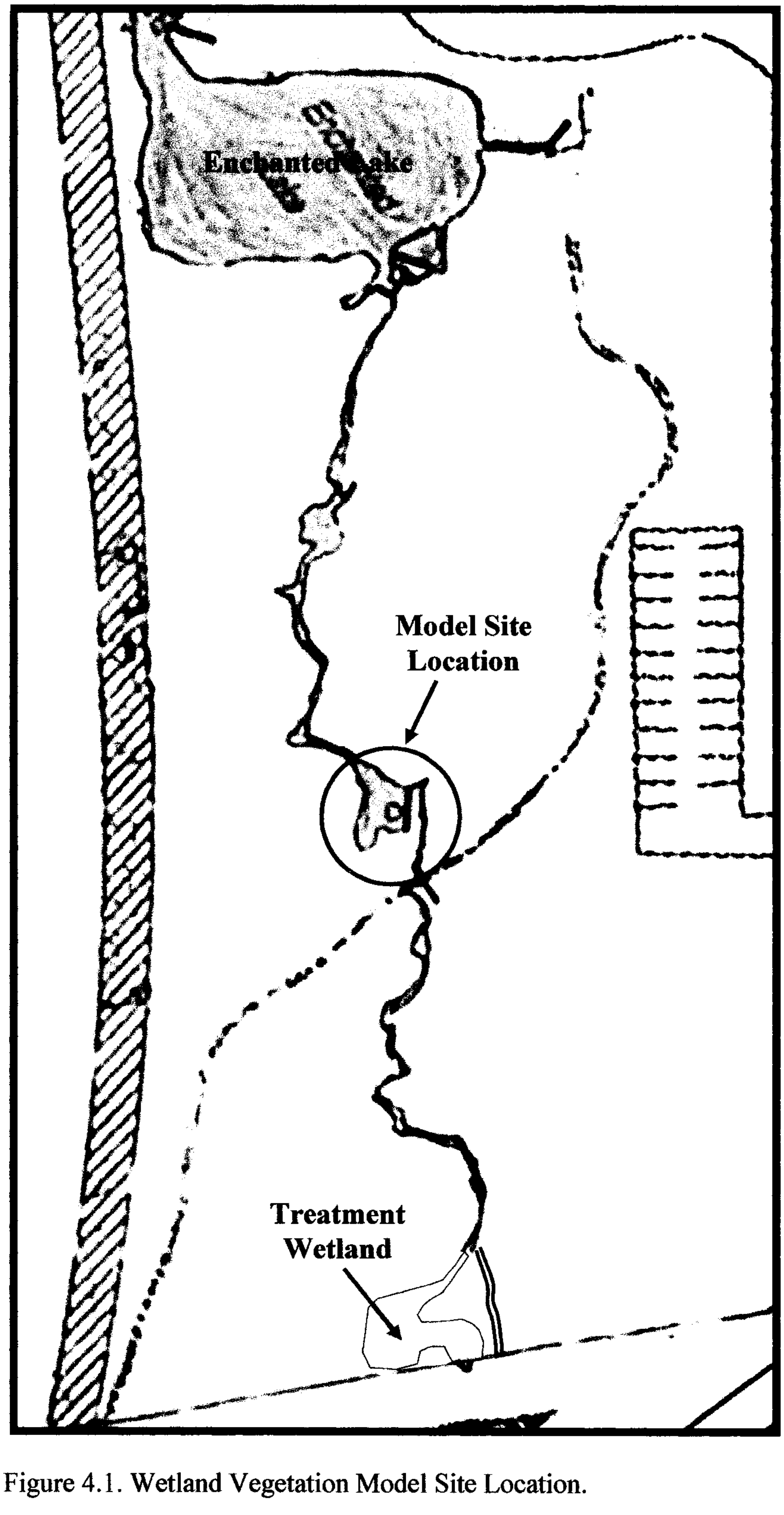


Figure 4.1. Wetland Vegetation Model Site Location.

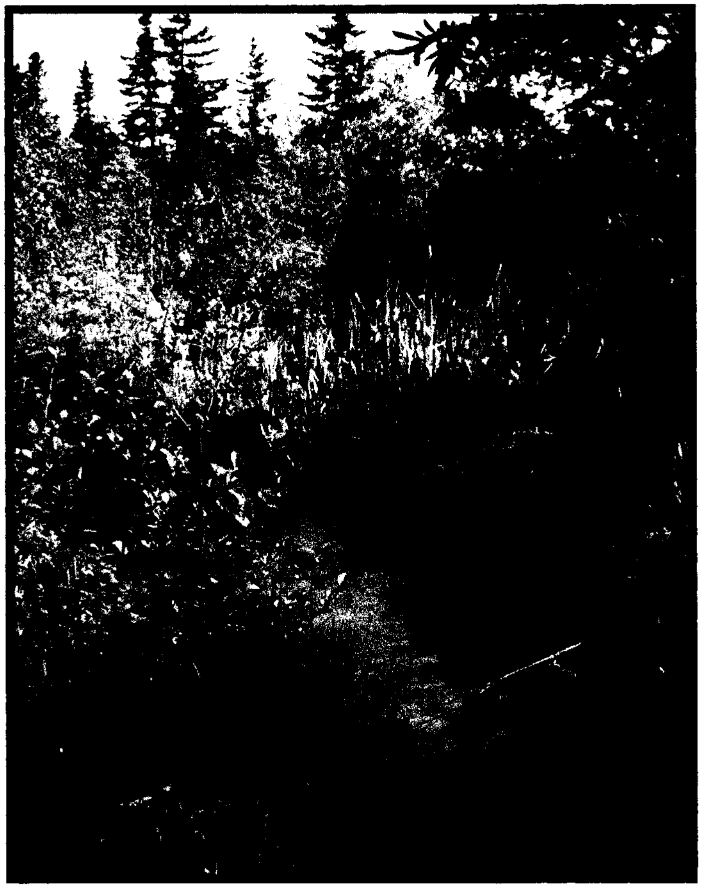


Figure 4.2. The Model Site is a Riparian Open Water Marsh.

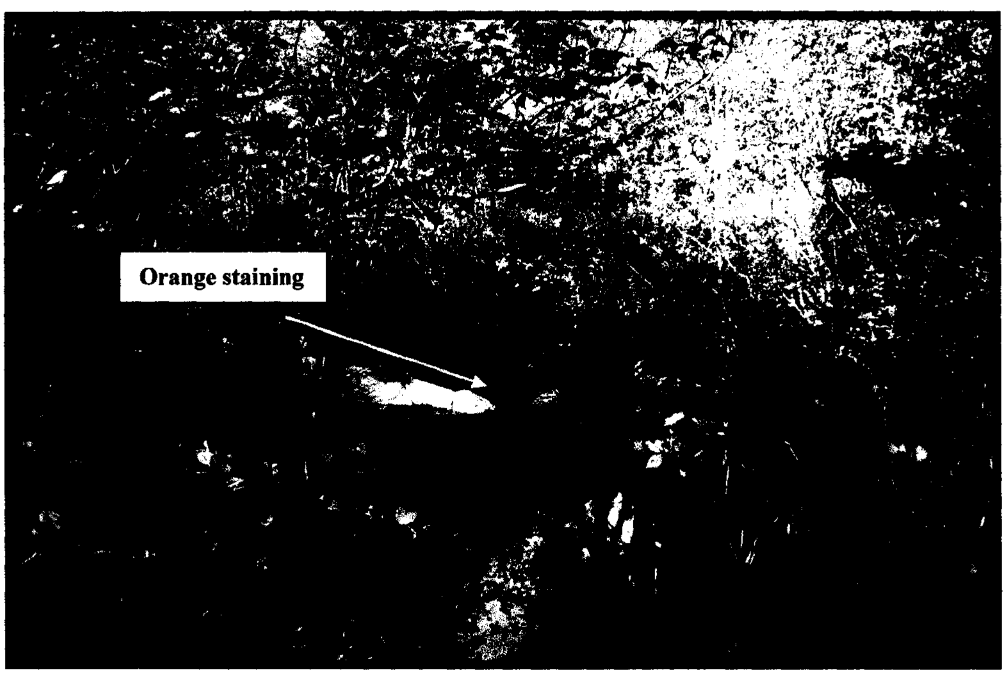


Figure 4.3. Orange Staining and Iron Particulate in the Model Site.

Baseline plant species surveying of the Wright's brook marsh took place in spring of 2002. Identification commenced with the dominant species present, then was refined to include less abundant species. The identification of species present in the model site were verified using several keys including: (a) Roland's Flora of Nova Scotia (Zinck, 1998), (b) Aquatic and Wetland Plants of Northeastern North America (Crow and Hellquist, 2000), and (c) Newcomb's Wildflower Guide (Newcomb, 1977). Species which proved too difficult to identify with absolute certainty were bagged, labelled and submitted to the Nova Scotia Museum of Natural History in Halifax for identification by qualified botanists. All species identified in the model site were recorded and organized into four categories (Emergent Wetland Plants, Upland Vascular Plants, Upland Shrubs and Upland Trees) and assigned coefficients between 1 and 5 indicating dominance, with a ranking of 5 representing very high abundance and a ranking of 1 indicating scarce abundance. Identified species which were non-native were immediately dismissed for potential use in the treatment site.

4.1.2. Burnside Treatment Wetland Plant Species Selection Criterion

Once the Wright's brook marsh plant list (derived from the vegetation survey) was complete, the plant selection for the Burnside treatment wetland commenced using the model plant list as a guide. The species from the model plant list were screened for suitability in the treatment site with regard to the following criterion: (a) phytoremediation potential (especially metal uptake), (b) sedimentation and erosion control, (c) habitat function, (d) public deterrent potential, and (e) rate of plant establishment, tolerances and requirements.

4.2. Vegetation Sources and Establishment Methodology

Transplantation was chosen as the main vegetation establishment methodology to be used for the Burnside treatment wetland site. Fortunately, in early May of 2002, several excellent donor sites including the Wright's brook wetland community model were identified within project property bounds as containing abundant plants for transplant (Figure 4.4 – 4.6). By transplanting the vast majority of the treatment wetland's plants, not only would the established species be naturally locally adapted, but also significant cost savings would be realized as nursery plants can be extremely expensive. In addition, using plants which occurred in communities immediately surrounding the treatment wetland site would ensure habitat continuity, and preserve biological integrity. Creating a site which supported high vegetative diversity similar to that of natural wetland systems was an important design consideration in this constructed wetland project.

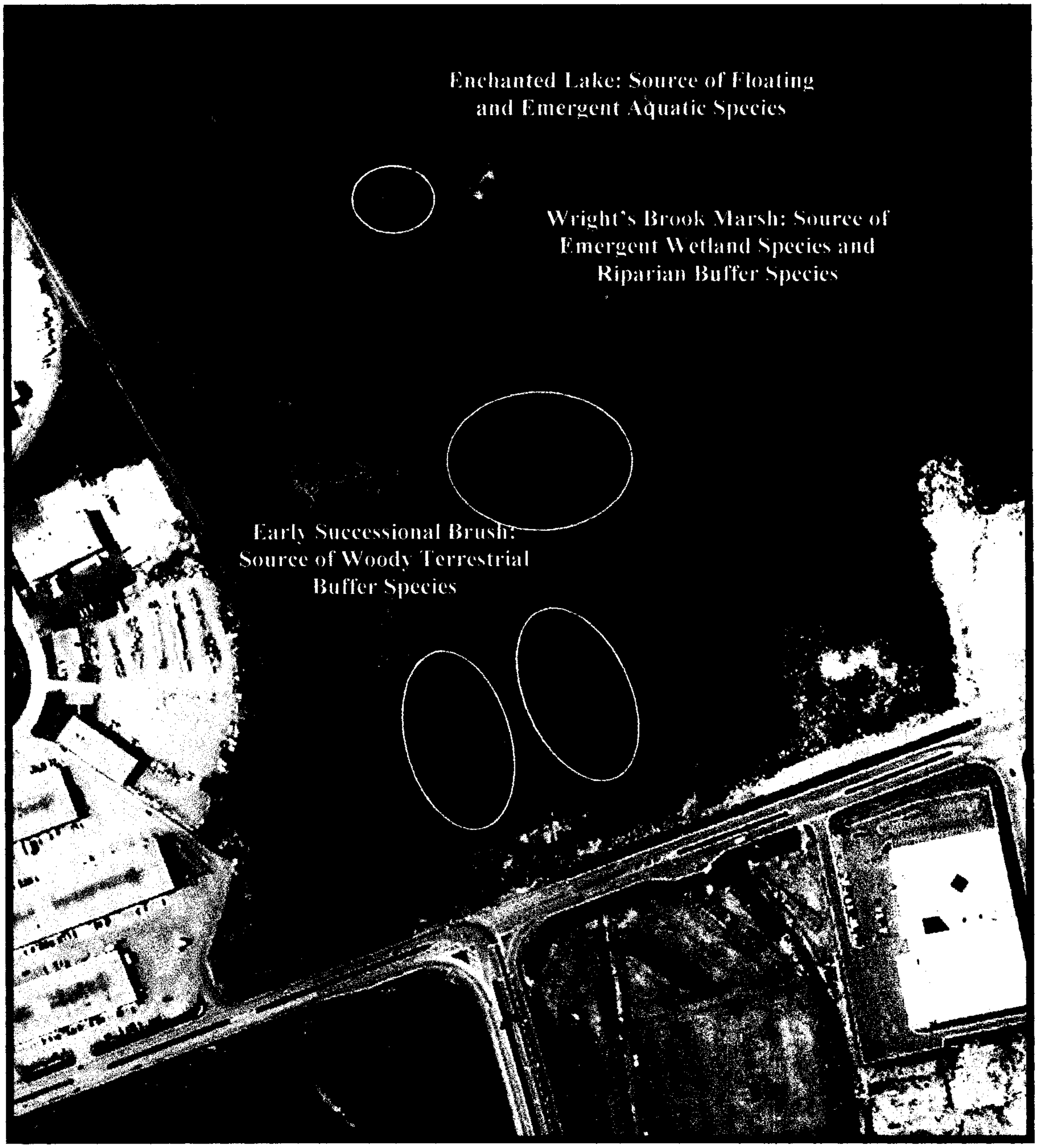


Figure 4.4. Plant Donor Site Locations (adapted from HLIS, 1992).

It has been suggested that 95% or more of a donor vegetation patch should be left to regenerate to minimize damage to the donor site (EC, 2000). Fortunately, given the minor amount of species required (about 1000 plants) and the large cumulative area of donor sites available, minimal damage to the donor sites resulted. Although difficult to quantify, it is likely that less that 0.5% of the donor sites were harvested. In addition, only those species which ranked a dominance coefficient of 2 or higher in the model site were considered for transplant in the treatment wetland

Figure 4.5. Successional Brush Vegetation Donor Site.

Figure 4.6. Enchanted Lake Vegetation Donor Site.

site to minimize the potential over-harvestation of less common species. It was decided that seed collection and application of woolgrass (*Scirpus cyperinus*), soft rush (*Juncus effusus*), and fowl mannagrass (*Glyceria striata*), three dominant native species present in the model site, would also take place in the fall of 2002. Given the wide diversity and availability of wetland species in the donor sites, there was no need to purchase nursery plants for establishment in the site. Once the final treatment wetland site plant candidate list was completed, and the establishment techniques and plant sources identified, the general planting layout for the site was prepared based on the plant functions and characteristics discovered during the screening process.

In May 2002, plant transplantation from the donor sites into the constructed wetland as per the general planting layout plan began. Establishment priority was the system's berms because they are central to the functionality of the site. The establishment of Berm 6 and Cell 4 was deemed not a priority as it was necessary to maintain Berm 6 as a roadway for vehicle traffic for later construction and expansion activities, and it was also expected that Cell 4 (which was already naturally vegetated along its northern boundary), would see alterations. Species identified in the donor sites as desirable for establishment in the treatment site were dug out using shovels and a pick maddock. Care was taken to make ensure the root systems of the extracted plants remained as in tact as possible. Care was also taken to keep the soils extracted with the plant roots intact. Once the wheel barrel was full of plants, the plants were immediately brought to the treatment wetland site and planted. Holes large enough to facilitate each plant root system were dug where the plants were to be established. Spacing of each species coincided with what the research indicated would be required to achieve uniform cover in 1 year (Table B.1 in Appendix B). The roots of the plants were then placed into the holes, buried with the extracted soils, and stomped securely into place. Topsoil was applied in areas which lacked sufficient stratum for planting. Since it was intended that the site should ultimately appear as indistinguishable as possible from that of a natural wetland site, care was taken to ensure plantings were relatively non-linear and non-horticultural. Planting continued through until August, 2002. During this time, water levels in the cells had to be lowered (Figure 4.7) by diverting waters from the system via the culvert located beneath the eastern border of Cell 1 (Figure 4.8). This was done because in surface flow constructed wetlands, water level can be the most critical aspect of plant survival during the first year after planting. Too much water can kill most immature aquatic macrophytes which need to receive abundant oxygen at their roots (Davis, 1995).

Two temporary berms were constructed in the Wright's brook headwaters directly adjacent to the

Figure 4.7. Lowered Water Level in Cell 1.

Figure 4.8. Valved Culvert used for Draining Burnside Cell 1.

treatment wetland to remediate any effects the draining water may have on the natural brook during the excavation period. The established berms were constructed of piled gravel and silt fencing reinforced with large rocks (Figures 4.9 and 4.10). Buffer and riparian vegetation was also established around the perimeters of the cells during the course of the 2002 summer to minimize the effects of wind and water erosion on the site's exposed edges. In addition, straw was placed on any exposed edges of the site to mitigate erosion until the buffer areas became established (Figures 4.11 and 4.12)

Fairly intensive maintenance of the site was necessary during the course of the vegetative establishment. This included watering (Figure 4.13), weed pulling and the removal and replacement of species which were not establishing themselves adequately (Figure 4.14). Berm repair was also a common maintenance practice as high velocity water flows during rain events often caused breaks in the fragile, unvegetated berms (Figure 4.15 and 4.16).

4.3. Evaluation of Vegetation Establishment Success

The survival rates of the plants established in the Burnside treatment wetland (and hence the success of the vegetation establishment strategy), was tested on May 20, 2003 via visual observation. The exercise was not meant to be exhaustive; the location of observed deceased plants established the previous year were simply hand-plotted on site maps of the treatment wetland. Cause of mortality was also noted if obvious (i.e. washout, insect infestation, etc.).

4.4. Evaluation of Biological Integrity of the Site

Biological integrity is defined as the ability of an ecosystem to support and maintain a healthy, balanced, adaptive community of organisms having a species composition, diversity, and functional organization comparable to those of natural habitats within a region (Karr and Dudley, 1981). The biological integrity of the established treatment wetland was tested in the second growing season of the site (spring of 2003) through the use of various metrics designed to evaluate the condition of a wetland's biological assemblages compared to those of a healthy, reference wetland site as recommended by the USEPA (2002h). The biological assemblages selected for this examination were vegetation and aquatic macroinvertebrates.

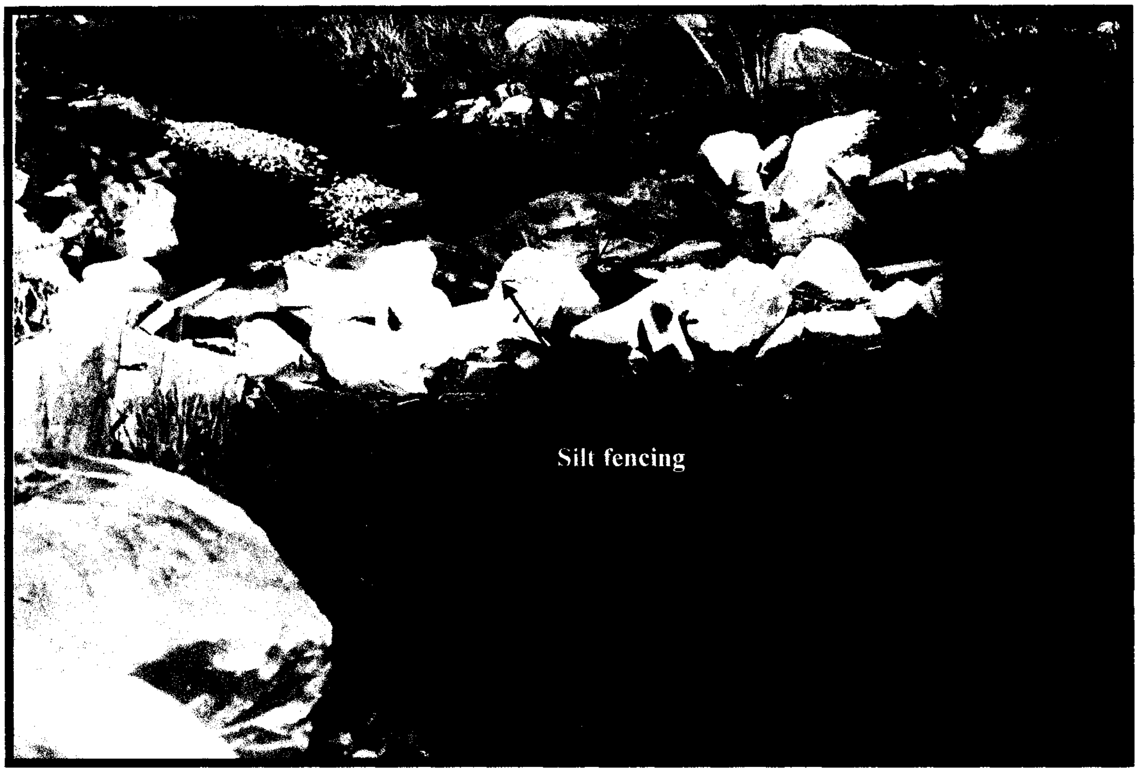


Figure 4.9. The First Temporary Berm in Wright's Brook.

Figure 4.10. The Second Temporary Berm in Wright's Brook.

Figure 4.11. Straw Placed on the Western Edge of Cell 1 to Minimize Erosion.

Figure 4.12. Straw Placed on the Eastern Edge of Cell A, B and C to Minimize Erosion.

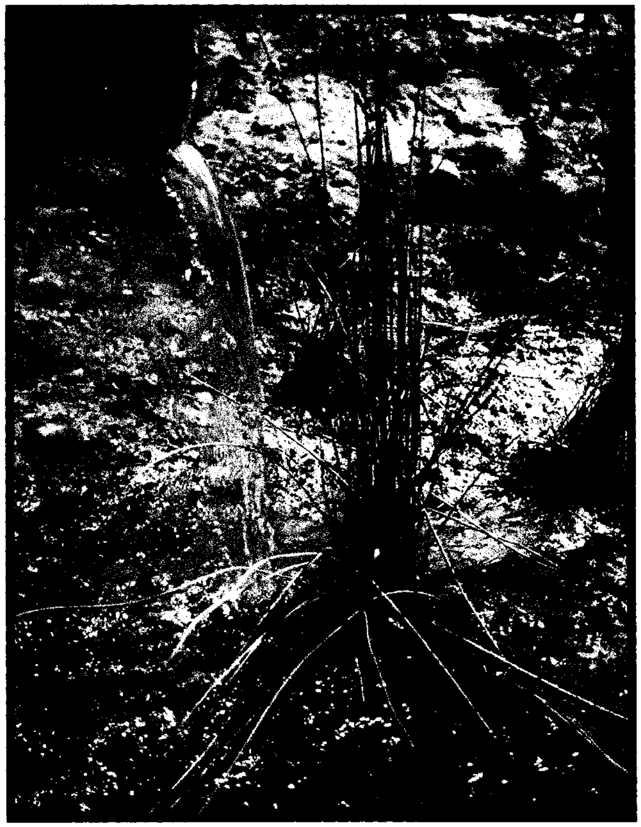


Figure 4.13. Watering the Vegetation in the Wetland, 2002.

Figure 4.14. Weeding, Removal and Replacement of Unestablished Species, 2002.

Figure 4.15. Break in Berm 2 that Needed Repair, Fall 2002.

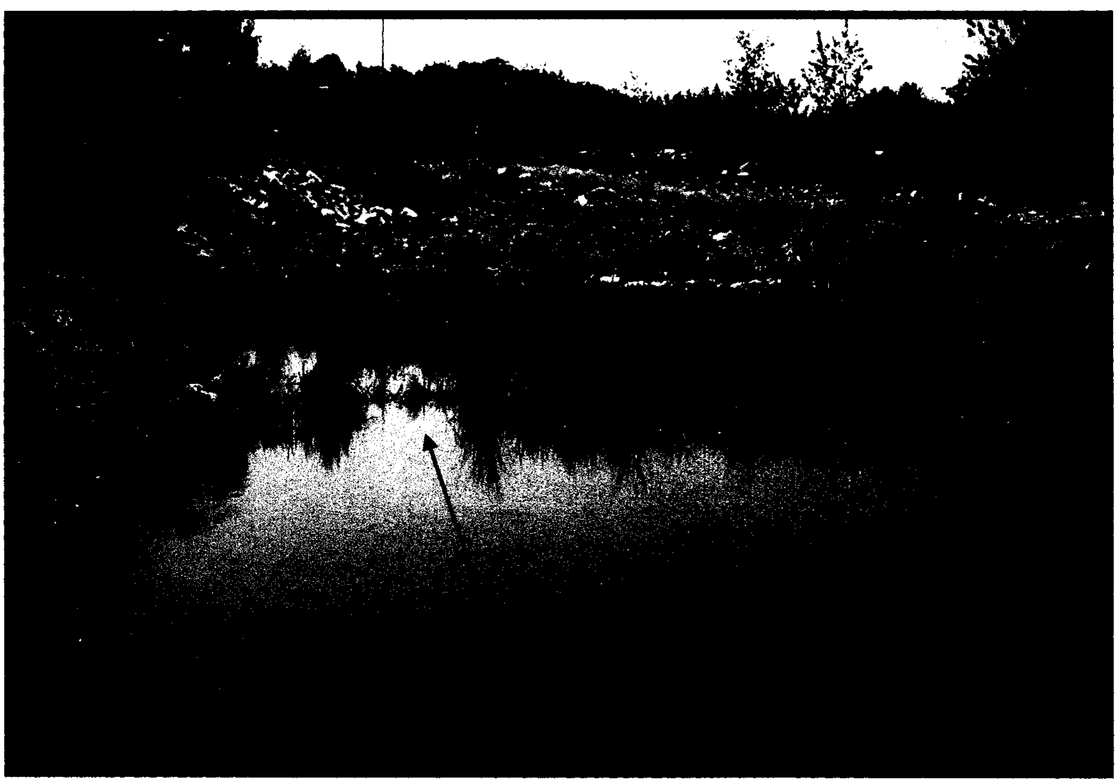


Figure 4.16. Break in Berm 1 that Needed Repair, Fall 2002.

4.4.1. Selecting a Reference Site

To be effectively assessed, the data (i.e. diversity values, dominance values, etc.) from the treatment site had to be compared to the data collected via the same sampling methodology from a healthy, local reference site. It is through evaluation of reference site conditions that the basis for comparison and impairment detection can be made, and hence the improper selection of the reference site may result in the inaccurate assessment of site health (USEPA, 2002h).

The reference site for the Burnside treatment wetland study was selected by examining wetland inventories conducted by the Nova Scotia Wetland Mapping Protection Program (1988). Classifications were based on factors such as dominant vegetation, the presence, depth and permanence of surface water, wetland size, topographic and hydrologic location, surrounding habitat, proportions and interspersion of cover, water and vegetation, and water chemistry. Based on these and other criteria, freshwater systems have been scored values between 36 to 108, with values greater than 60 perceived as having high biodiversity, high habitat value and overall ecological health (WMPP, 1988). The freshwater marsh systems with high-ranking scores located in proximity to the Burnside treatment wetland site are indicated in blue in Figure 4.17. Although difficult to discern, the first set of numbers associated with the surveyed wetland sites indicates their designated number, the second set indicates the wetland's classification, the third set indicates their Gotlet score (with scores less than 65 not indicated), and the fourth set indicates their size in hectares. As can be seen, the wetland classifications are not definitive, as most are given more than one classification, with the more dominant classifications listed in descending order (i.e. B-SS: the site is predominantly bog but supports shrubby areas as well). For purposes of clarification, a small key to the acronyms used in the wetland classification figures is provided in Table 4.1.

As a result of its fitting classification and close location (approximately 25 km from the Burnside site in the Waverly Game Sanctuary), marsh A-8 was selected as the reference site for the biological integrity assessment (Figure 4.18). Water samples from the site were submitted for fingerprint analysis to establish that the site would be an effective control, as disturbance is not always discernible to the eye.

4.4.2. Metrics of Biological Integrity

Metrics use a variety of measurements compared to that of healthy ecosystems to assess the biological health of tested systems. By definition a metric is a parameter or variable which represents some feature, status, or attribute of biotic assemblage, chemical state, or physical condition. In a multi-metric approach, several different metrics are recommended in order to effectively indicate a site's condition (USEPA, 2002h).

The biological integrity of the Burnside treatment wetland was assessed by examining and comparing the vegetation and aquatic macroinvertebrate populations of the treatment wetland and those of reference wetland sites via three metrics commonly used in biological integrity

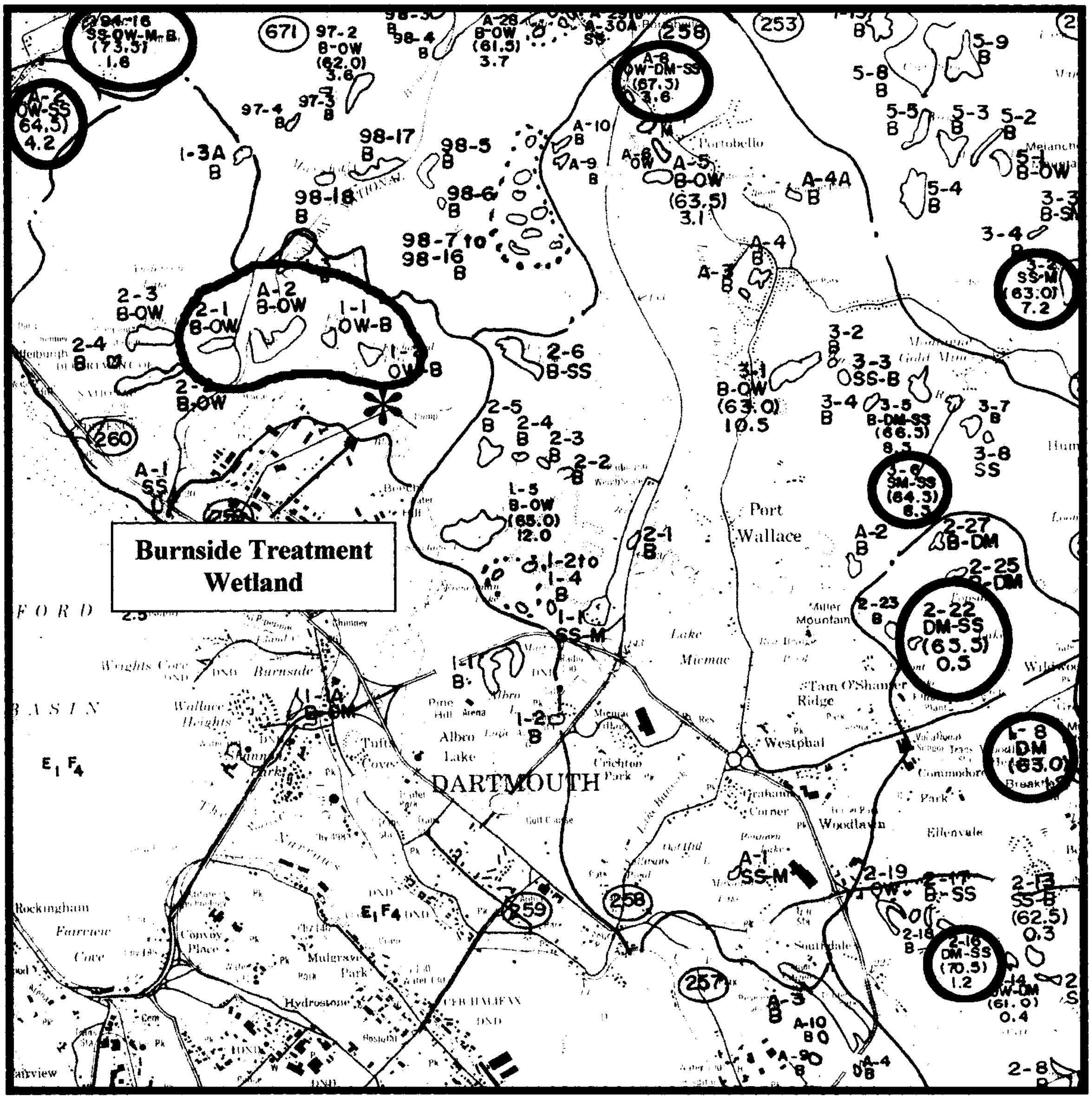


Figure 4.17. High Scoring Freshwater Marshes in Proximity to the Burnside Treatment Wetland Site (adapted from the WMPP, 1988).

Table 4.1. Wetland Map Key to Wetland Types (WMPP, 1988).

ACRONYM	MEANING
OW	Open Water 0.9 – 3.0 m deep
DW	Deepwater Marsh - avg depth >0.15 m. Emergent marsh vegetation usually dominant with surface and subsurface vegetation present in open areas.
SS	Dominated by shrubs where soil surface is seasonally or permanently flooded with as much as 0.3 m water Carex sp. and other meadow emergent occupy open areas.
SM	Shallow marsh – avg. water depth < 0.15 m. Surface water may be seasonally absent
M	Dominated by meadow emergents with up to 0.15 m of surface water in wet seasons.
B	Bog – wetland where the accumulation of peat determines the nature of the plant community.

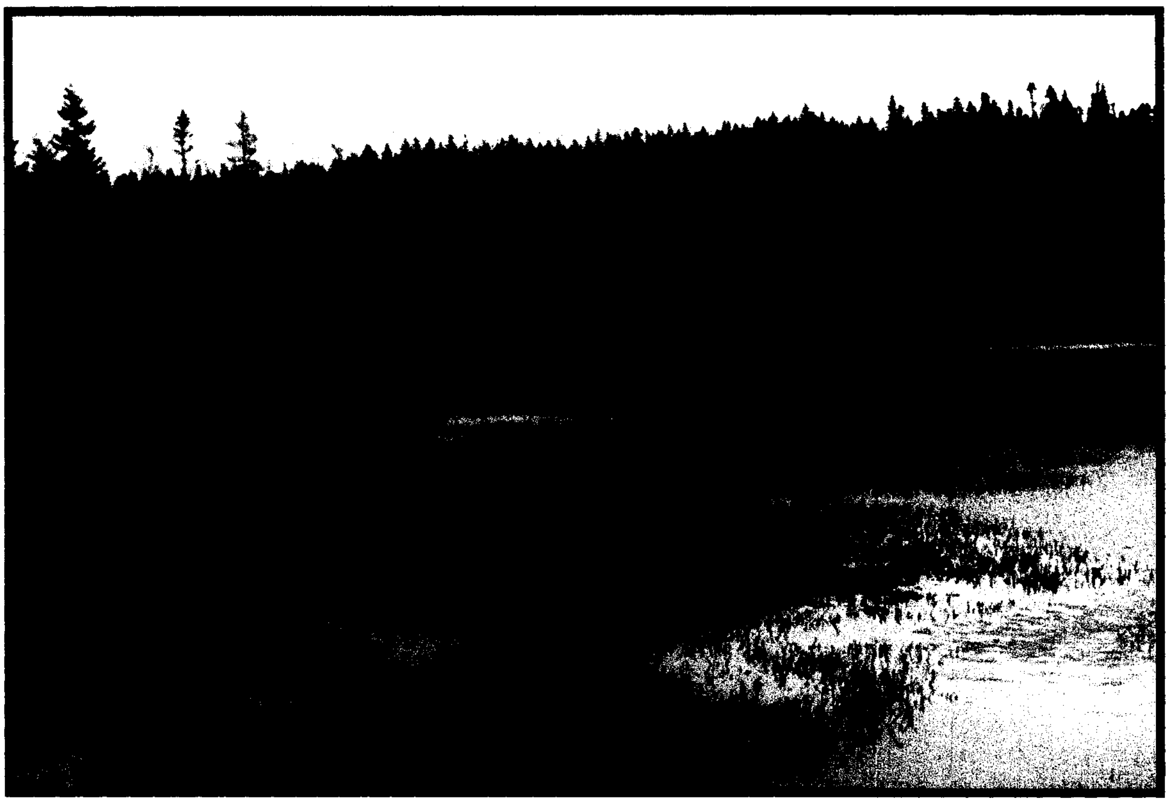


Figure 4.18. Photographs of Reference Site, Marsh A-8.

assessments. For the vegetation, the metrics analyzed were species diversity, species heterogeneity (dominance) and exotic invasive species abundance (Table 4.2). For the macroinvertebrates, the metrics analyzed were diversity, heterogeneity, and trophic structure (Table 4.3). It was very important to keep things as simple as possible without compromising scientific validity, as this monitoring would need to be reproducible in subsequent years by others who may have little expertise in biological monitoring.

Table 4.2. Selected Vegetation Metrics of Biological Integrity for Burnside Treatment Site.

METRIC	PREMISE	COMPARISON
Population Diversity via Shannon-Weiner Diversity Index	Sites supporting high plant diversity support high biological integrity	Wetland site to reference site
Plant population heterogeneity	Sites supporting high plant species heterogeneity support high biological integrity	Wetland site to reference site
Exotic/invasive species abundance	Higher proportions of exotic/invasive species indicative of poor biological integrity	Wetland site to reference site

Table 4.3. Selected Aquatic Macroinvertebrate Metrics of Biological Integrity for Burnside Treatment Site.

METRIC	PREMISE	COMPARISON
Population Diversity via Shannon-Weiner Diversity Index.	Sites supporting high macroinvertebrate diversity support high biological integrity	Cell 1, wetland outlet and reference site
Macroinvertebrate population heterogeneity	Sites supporting high macroinvertebrate heterogeneity support high biological integrity	Cell 1, wetland outlet and reference site
Trophic Structure	Sites supporting skewed ratios functional feeding groups indicative of poor biological integrity	Cell 1, wetland outlet and reference site

4.4.3. Vegetation Sampling

The materials used for the vegetation sampling were: four 1 m long pieces of string, a one-meter long piece of string attached to a pen, a 1 m^2 squared quadrat, a 50 m measuring tape, a clipboard, datasheets, generated random number tables and plastic baggies. The plant identification guides used were Roland's Flora of Nova Scotia, Vol. I and II (Zinck, 1998), Aquatic and Wetland Plants of Northeastern North America, Vol. I and II (Crow and Hellquist, 2000), and Newcomb's Wildflower Guide (Newcomb, 1977).

Vegetation sampling of the Burnside treatment wetland and reference site was conducted in July, 2003. The sampling regime used for the vegetation sampling for the site was a stratified random sample. In essence, a stratified sample allows for a target area to be assessed while still remaining statistically viable by sampling at random (Underwood, 1997). In seeing that the vegetated berms of the Burnside wetland were the priority of the site establishment regime for this thesis, it was these areas which were the focus of the vegetation assessment (green areas in Figure 4.19).

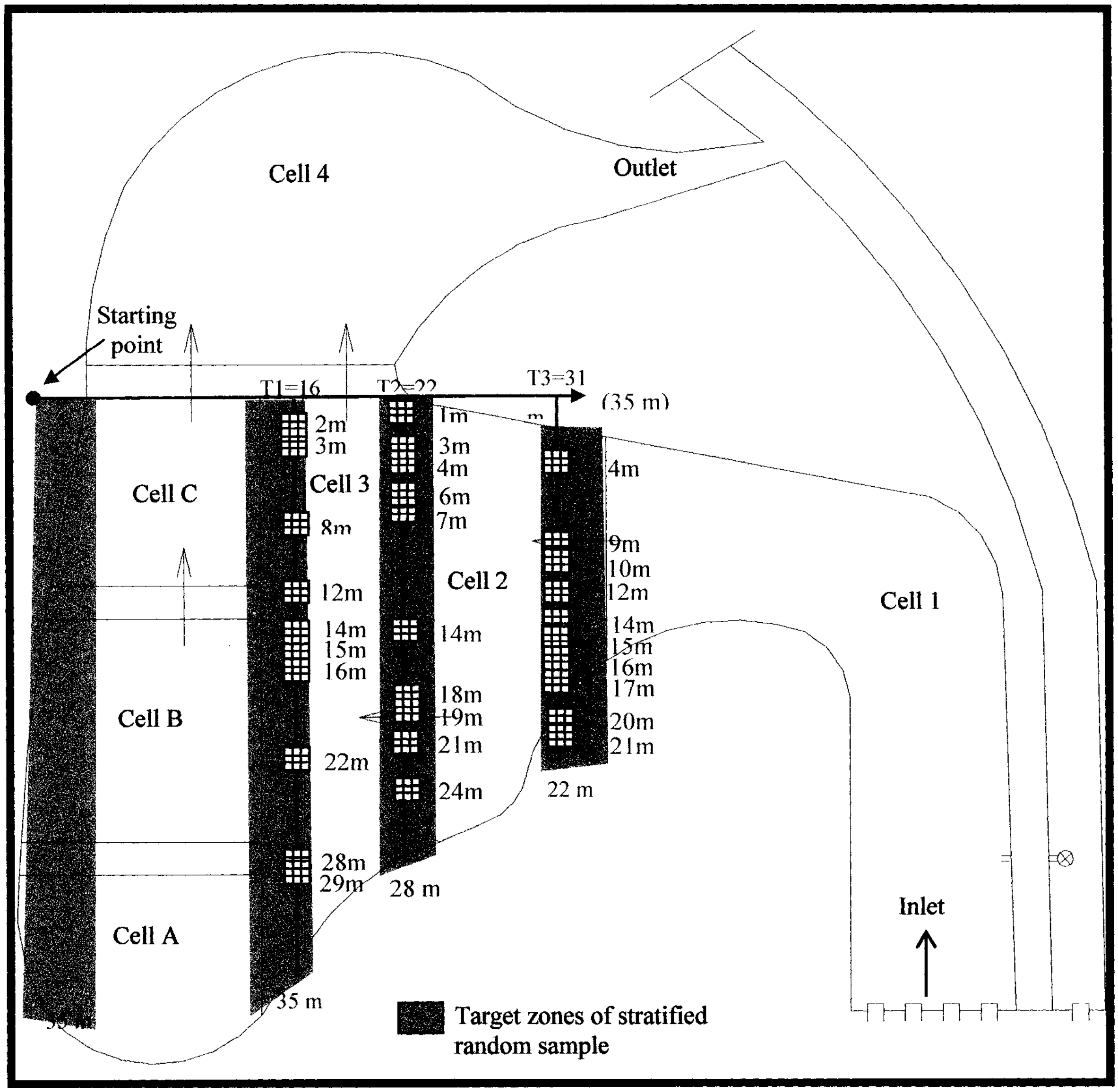


Figure 4.19. Transect and Quadrat Locations at Burnside Treatment Wetland Site.

A starting point was simply arbitrarily chosen and a reference line transect along the northern edge of the target communities was designated. Using excel, a random numbers table (70 numbers ranging between 1 to 35 (length in metres of transect) was generated. The first 3 random numbers (eliminating any repeated numbers or numbers that did not lead to the target areas) were used to locate the 3 line transects in the target areas set perpendicular to the reference line transect. Along each line transect, 10 randomly chosen quadrats, based on the same series of random numbers were sampled, eliminating any repeated numbers or numbers that exceeded the transect range (ie. >22 for T3 in treatment wetland). The quadrat was a 1m^2 metal frame, divided into 9 equal sections with the 1m long sections of string (Figure 4.20). Within each quadrat, the string with the pen attached was dropped 1m above the quadrat, sampling each of the 9 sectors. The plant located at the tip of the pencil was identified and recorded as described by Murphy (1999) and Underwood, (1997). Those plants not identifiable in the field were labeled and bagged for later identification. In total, 270 sectors were sampled for each site.

Figure 4.20.Vegetation Sampling Quadrat.

4.4.3.1. Species Diversity. The data collected was assessed for plant species diversity (α-diversity) using the following Shannon-Weiner index (Murphy, 1999; and Cousins, 1991):

$$H' = -\Sigma[p_i(\ln p_i)] \tag{4.1}$$

Where:

H'= Shannon-Weiner diversity index

p_i = proportional abundance of a given species (i).

The proportional abundance (pi) is calculated as follows:

$$Pi = n_i/N \qquad (4.2)$$

Where:

n_i = the number of individuals of a given species (i)

N = the total number of individuals of all species.

The Shannon-Weiner index is a measure of the proportional abundance of each species. It assumes that: (a) all individuals are sampled at random, (b) all individuals are sampled from infinitely large populations (too much to count individuals) and (c) all species present will be sampled. For a large number of samples, the values will have a log-normal distribution, which is convenient as t-tests, ANOVA and other parametric statistics only work if data that have a log-normal distribution (Murphy, 1999; and Cousins, 1991). The higher the Shannon-Weiner index, the more diverse the system is implied to be in terms of number of species. In order to accurately measure the difference in diversity between the two communities, the variance in diversity was calculated for both sites using the following formula (Murphy, 1999):

$$H'_{var} = N^{-1}\{\Sigma p_i(\ln p_i)^2-[\Sigma p_i(\ln p_i)]^2\}-\{(2N^2)^{-1}\{S-1]\} \qquad (4.3)$$

Where:

H'_{var} = variance in diversity

N = the total number of individuals of all species

p_i = proportional abundance of a given species (i)

S = species richness

Once the variances were calculated, the diversity of the two sites was statistically compared using parametric t-test statistics. The t-test determines whether the means of the two samples are significantly different from one another by comparing the distribution of values derived from the samples to a statistical distribution (the t-distribution). In the case of the Shannon-Weiner comparison, the formula used for calculating the t-statistic was as follows (Murphy, 1999):

$$t = \frac{[H'_{\text{ref site}}-H'_{\text{treatment site}}]}{H'_{var}\text{ (ref site)} + H'_{var}\text{ (treatment site)}]^{0.5}} \qquad (4.4)$$

Where:

$\underline{t}$ = $\underline{t}$-statistic

H'= Shannon-Weiner diversity index

The degrees of freedom which exist for the test were then calculated in order to gauge the accuracy of the test . The greater the degrees of freedom, the more accurate the test (Murphy, 1999).

$$df = \frac{[H'_{var}(\text{ref site})+H'_{var}(\text{treatment site})]}{\{[H'_{var}(\text{ref site})]^2/N_{\text{ref site}}+[H'_{var}(\text{treatment site})]^2/N_{\text{treatment site}}\}} \tag{4.5}$$

Where:

df = degrees of freedom

H'_{var} = variance in diversity

N =the total number of individuals of all species

The values were then compared to a table of critical values of the $\underline{t}$-distribution, significance level "α2", P value of 0.05. If the values were greater than the relative values in the table, it could be implied that the observed t-value was greater than the expected (random) $\underline{t}$-values, and the two sites compared were significantly different in terms of their species diversity, with the site supporting the higher value of H' supporting the greater species diversity. Likewise, if the values were close to the values in the table of critical values, then the sites could be deemed statistically similar (Murphy, 1999). This latter result was highly desirable as it was the aim of this study to produce a site with biological integrity (gauged in this case through biodiversity) similar to that of a natural wetland site.

4.4.3.2. Heterogeneity (Dominance). The heterogeneity of the wetland site and reference site was assessed by using the following equation (Murphy, 1999):

$$\text{Heterogenity} = \frac{\text{abundance of the top 3 species}}{\text{total number of species}} \times 100 \tag{4.6}$$

4.4.3.3. Exotic Species Abundance. The data collected during the vegetation sampling was also assessed for total proportional abundances of invasive and exotic species. This was assessed using the following equation (Murphy, 1999):

$$\text{Proportional abundance} = \frac{\text{total number of exotic species}}{\text{total number of species}} \times 100 \qquad (4.7)$$

4.4.4. Macroinvertebrate Sampling

The materials used for the macroinvertebrate sampling were a sweep net (0.6 mm mesh size), alcohol, a microscope, 20 one-litre sample bottles, a turkey baister, petri dishes, a clipboard and datasheets. The aquatic macroinvertebrate identification guide used was Freshwater Macroinvertebrates of Northeastern North America (Peckarsky et al., 1990).

Macroinvertebrate sampling for the Burnside treatment wetland and reference wetland site was conducted on July 3, 2003. The macroinvertebrate sampling regime differed from that of the vegetation monitoring in that in addition to the reference site, macroinvertebrate samples were taken from both Cell 1 and the outlet of the treatment wetland (Figures 4.22 and 4.23). This was done in an attempt to gauge any differences in the biological integrity of the macroinvertebrate populations between the most contaminated area of the system, and the hypothetically 'purified' area of the system.

Each dipnet sample contained a large number of organisms. As is the case with most macroinvertebrate sampling schemes, complete census would have been unreasonable. Consequently, the organisms were subsampled to 100 organisms. The contents of the sweep net were divided as equally as possible into 5 1L sample jars filled with approximately 750 ml of alcohol and water (50/50), which killed and preserved the captured organisms. This was repeated for each of the three sample sites. The subsample was obtained by randomly selecting one of the five jars from each sample site. Using the turkey baister, samples were drawn from the jars and placed in Petri dishes for examination until 100 organisms were obtained. If the jar yielded less than 100 organisms, another jar was randomly selected from the same sample site location and sampled until 100 organisms for the site were retrieved as recommended by the USEPA (2002l). In total, sweeps of each sample site yielded 300 macroinvertebrates; 100 for each sample site. Using the indicated identification using keys, the insects were identified to the family level.

Figure 4.21. Cell 1 Sampling Location for Aquatic Macroinvertebrates.

Figure 4.22. Outlet Sampling Location for Aquatic Macroinvertebrates.

4.4.4.1. Population Diversity. The macroinvertebrate samples collected were assessed for family diversity using the Shannon-Weiner index (Equation 4.3). The assumption here is that a site supporting high macroinvertebrate diversity similar to that of a pristine, natural wetland site possess biological integrity (USEPA, 2002l; and Mandaville, 2002). The values obtained from the reference site, Cell 1, and the wetland outlet were compared to each other using parametric t-test statistics (Equation 4.4). The accuracy of the tests were gauged by calculating the degrees of freedom existing for the tests (Equation 4.5) (Murphy, 1999).

4.4.4.2. Heterogeneity (Dominance). The macroinvertebrate populations were also assessed using Equation 4.6 on the family level. The assumption here is that sites having high dominance figures (i.e. top 3 families dominate 95% of sample site), are not as biologically healthy as those sites supporting a more balanced distribution of organisms (USEPA, 2002l; and Mandaville, 2002).

4.4.4.3. Trophic Structure. The term 'trophic structure' relates to the feeding habits of the macroinvertebrates. The proportions of each feeding group are expected to fluctuate to unnaturally occurring proportions in unhealthy wetland systems. Proportional abundance values were obtained from the following equation:

$$\text{Proportional abundance} = \frac{\text{total number of individuals per feeding group}}{\text{total number of individuals}} \text{ X } 100 \qquad (4.7)$$

4.4.5. Wildlife Observations

In addition to detailed vegetation and macroinvertebrate population sampling and analysis, the biological integrity of the Burnside Treatment wetland site was also informally gauged by recording wildlife species which were casually observed in and around the treatment wetland site during site visits. These records were complemented with photographic records when possible.

4.5. Evaluation of the Water Purification Ability of the Wetland

The effectiveness of the naturalized wetland system in purifying the leachate from the landfill was tested via several assessment methods including: chemical water quality analysis, plant tissue analysis, and macroinvertebrate bioassessment.

4.5.1. Chemical Water Quality Analysis

Bi-weekly sampling of the water quality of the site commenced in May, 2003 through September, 2003. Water quality samples were collected from Cell 2, 3 and the outlet and compared to baseline influent concentrations. The water quality between Cells 2 and 3 were compared in order to gauge some semblance of purification progress from Cell 1 to Cell 3. The water quality of the outlet leading from Cell 4 was monitored and compared to influent concentrations to gauge the effectiveness of the entire system. Samples were collected on varying days to ensure samples taken were representative. The parameters analysed included: iron, manganese, phosphorus (orthophosphate), pH, dissolved oxygen (DO), nitrogen (ammonia, nitrate, nitrite and TKN), chemical oxygen demand (COD), total suspended solids (TSS) and total dissolved solids (TDS). Nitrogen, pH, COD, TSS and TDS were analyzed at the Biotechnology Laboratory at Dalhousie University. The parameters iron, manganese, and orthophosphate were analyzed at the Mineral Engineering Laboratory at Dalhousie University.

4.5.2. Plant Tissue Analysis

The phytoremediation capacity of several plants in the Burnside treatment wetland site was tested by conducting chemical analyses of sampled plant tissues compared to that of plant tissues of the same species taken from a control site. The parameters analysed were iron and manganese. The species chosen for analyses were woolgrass (*Scirpus cyperinus*), soft rush (*Juncus effusus*), tweedy's rush (*Juncus brevicaudatus),* pickerelweed *(Pontederia cordata),* yellow-green sedge *(Carex lurida),* fowl mannagrass *(Glyceria striata),* soft stem bulrush *(Scirpus validus)* and fringed sedge (*Carex crinita*). The exotic species reed canary grass (*Phalaris arundinacea*) and aggressive species broad-leaved cattail *(Typha latifolia)* which were also present in the treatment wetland site in abundance were also assessed for their metal hyperaccumulation ability. The control specimens were collected from the Waverley Provincial Game Sanctuary marsh (Figure 4.18)

Plant specimens collected were gently washed to rid them of any soils which may have contained the contaminants of concern as recommended by A&L Canada Laboratories (2002). Using scissors, the roots, stem, leaves, and flowering head of the plant specimens were separated, placed into labelled plastic baggies, and stored in freezers until the analysis was conducted. Specimens were then dried in a convection oven for 24 hours at 45 °C. After drying, the plants were ground and digested with hydrochloric-nitric-hydrofluric-perchloric acids (30+10+10+5 mL/g sample) in a closed vessel at a temperature of 100 °C. Metal concentrations were determined using an

Atomic Absorption Spectometer (Varion SpectrAA, Model Number: 55B, Varion, Mulgrave, Victoria, Australia). The metal analysis was conducted at the Mineral Engineering Laboratory.

4.5.3. Aquatic Macroinvertebrate Monitoring

In addition to biological integrity, aquatic macroinvertebrate can also prove to be excellent indicators of water quality. The basis of most of the metrics involved in water quality bioassessment are focused on the presence or absence of tolerant or intolerant indicator organisms. Intolerant taxa, by definition, are more likely to disappear under impaired conditions, hence their presence indicates good conditions. Tolerant taxa, on the other hand, are capable of surviving in both stressed *and* pristine systems, therefore, the presence of tolerant taxa will not necessarily indicate the presence of poor water quality. According the USEPA (2002l) superior metrics are those which evaluate the presence of intolerant taxa.

The aquatic macroinvertebrate samples collected on July 3, 2003 were examined as indicators of water quality using several specific water quality metrics: (a) Average Score per Taxon (ASPT) via the Biological Monitoring Working Party (BMWP) biotic index, (b) the ETSD (Ephemeroptera, Trichoptera, Sphaeriidae, and Odonata) biotic index, (c) percent abundance of mayflies; and trophic structure (Table 4.4).

4.5.3.1. BMWP Biotic Index and ASPT. The family level-based Biological Monitoring Working Party (BMWP) biotic index described by Kirsch (1999) and Mandaville (2002) was used in this test. The index is devised to limit the taxonomic requirement of many biotic indices to identify organisms to species level which required a rare kind expertise. The BMWP assigns pollution tolerance scores between 1 and 10 to all indicator organisms present at the family level. The greater their tolerance towards pollution, the lower their BMWP score. Individual scores are then tabulated to get a total BMWP score (Table 4.5).

The Average Score Per Taxon (ASPT) represents the average tolerance score of all taxa within the community. The ASPT is calculated by dividing the BMWP score by the number of indicator families present in the sample (Kirsch, 1999; and Mandaville, 2002). The water quality assessment generally associated with the ASPT scores are listed in Table 4.7. ASPT scores were calculated for each of the macroinvertebrate samples taken from each of the study sites.

Table 4.4. Selected Aquatic Macroinvertebrate Metrics of Water Quality.

METRIC	PREMISE	COMPARISON
ASPT via BMWP biotic index	The Average Score per Taxon (ASPT) represents the average tolerance score of all taxa within the community and is calculate by dividing the BMWP (Biological Monitoring Working Party) score by the # of families in the sample.	Cell 1, wetland outlet and reference site
ETSD	Measure abundance of pollution-sensitive groups in wetlands, specifically Ephemeroptera (E) (mayflies), Trichoptera (T) (caddisflies); Sphaeriidae (S) (fingernail clams), and Odonata (dragonflies/damselflies) (D). These "intolerant" taxa are sensitive to pollution, hence greater abundances of these families indicated greater site health.	Cell 1, wetland outlet and reference site
Mayfly abundance	Mayflies sensitive to ammonia, metals, low dissolved oxygen, chlorine, pesticides and acidity	Cell 1, wetland outlet and reference site
Trophic structure *Completed in Biological Integrity Assessment	Altered trophic structures are indicative of specific pollutant loading (i.e. increased ratio of predators to others indicative of heavy metal pollution; decreased ratios indicative of organic pollution and low DO	Cell 1, wetland outlet and reference site

Table 4.5. Pollution Sensitivity Grades for Aquatic Macroinvertebrate Families; adapted for Maritime Studies (Mandaville, 2002; Kirsch, 1999; and Hynes, 1998).

FAMILY	GRADE		FAMILY	GRADE		FAMILY	GRADE	
	N	B		N	B		N	B
Acariformes	6	-	Gammaridae	4	6	Peltoperlidae	9	-
Aeolosomatidae	2	-	Gerridae	5	5	Perlidae	8	10
Aeshnidae	6	8	Glossiphoniidae	3	3	Perlodidae	8	10
Agrionidae	4	8	Glossosomatidae	10	-	Philopotamidae	7	8
Ancylidae	4	6	Gomphidae	6	8	Phryganeidae	7	-
Anthomyiidae	4	-	Gordiidae	8	10	Physidae	2	3
Anthuridae	4	-	Gyrinidae	5	5	Piscicolidae	5	4
Asellidae	2	3	Haliplidae	5	5	Planariidae	4	5
Arctiidae	5	-	Haplotaxidae	1	1	Planorbidae	3	3
Arrenuridae	4	-	Helicopsychidae	7	-	Platyhelminthidae	6	-
Astacidae	4	8	Helodidae	5	5	Pleidae	5	5
Athericidae	6	-	Heptageniidae	7	10	Pleuroceridae	4	-
Atractideidae	4	-	Hirudinea	0	-	Polycentropodidae	4	7
Baetidae	5	4	Hyalellidae	2	-	Polychaeta	4?	-
Baetiscidae	6	-	Hydridae	5	-	Polymetarcyidae	8	-
Belostomatidae	5	-	Hydrobiidae	4	3	Potamanthidae	6	10
Blephariceridae	10	-	Hydrometridae	5	5	Psephenidae	6	-

N = Number of families occurring in North America
B = BWMP Score

Table 4.5. Continued.

FAMILY	GRADE		FAMILY	GRADE		FAMILY	GRADE	
	N	B		N	B		N	B
Branchiobdelidae	4	-	Hydrophilidae	5	5	Psychodidae	8	8
Brachycentridae	9	10	Hydropsychidae	6	5	Psychomyiidae	8	8
Caenidae	5	7	Hydroptilidae	5	6	Pteronarcidae	10	-
Calopterygidae	4	-	Hygrobiidae	5	5	Ptychopteridae	1	-
Capniidae	8	10	Idoteidae	5	-	Pyralidae	5	-
Ceratopogonidae	4	-	Isotomidae	5	-	Rhyacophilidae	9	-
Chaoboridae	2	-	Lebertiidae	4	-	Sabellidae	4	-
Chironomidae	1	2	Lepidostomatidae	-	-	Sialidae	6	4
Chloroperlidae	10	10	Leptoceridae	6	10	Simuliidae	5	-
Chrysomelidae	5	5	Leptophlebiidae	7	10	Siphlonuridae	8	10
Coenagrionidae	2	6	Lestidae	1	-	Sphaeriidae	4	3
Collembola	5?	-	Leuctridae	10	10	Spurchonidae	4	-
Corbiculidae	4	-	Libellulidae	8	8	Sisyridae	5	-
Corduliidae	7	8	Limnephilidae	7	7	Tabanidae	5	-
Cordulegasteridae	7	8	Limnesidae	4	-	Taeniopterygidae	8	10
Corixidae	5	5	Limnocharidae	4	-	Talitridae	2	-
Corydalidae	6	-	Lumbriculidae	2	1	Thiaridae	6	-
Culicidae	1	-	Lymnaeidae	4	3	Tipulidae	7	5
Dixidae	10	-	Mesoveliidae	5	5	Tricorythidae	6	-
Dolichopodidae	6	-	Mideopsidae	4	-	Tubificidae	1	1
Dreissenidae	2	-	Molannidae	4	10	Tyrellidae	4	-
Dryopidae	5	5	Muscidae	4	-	Unionidae	4	6
Dytiscidae	5	5	Naididae	3	1	Unionicolidae	4	-
Elmidae	5	5	Nemouridae	8	7	Valvatidae	2	3
Empididae	4	-	Nepidae	5	5	Veliidae	5	-
Enchytreidae	1	1	Nepticulidae	5	-	Viviparidae	4	6
Ephemerellidae	10	10	Notonectidae	5	5			
Ephemeridae	8	10	Odontoceridae	10	10			
Ephydridae	4	-	Oedicerotidae	4	-			
Erpobdellidae	3	3	Oligochaeta	2	1			

N = Number of families occurring in North America
B = BWMP Score

Table 4.6. Average Score per Taxon (Mandaville, 2002).

ASTP VALUE	WATER QUALITY
>6.0	Excellent Water Quality
5.5 – 6.0	Very Good Water Quality
5.0 - 5.5	Good Water Quality
4.5 – 5.0	Moderate Water Quality
4.0 - 4.5	Moderately-poor Water Quality
<4.0	Poor Water Quality

4.5.3.2. ETSD Biotic Index. The ETSD biotic index is an acronym for Ephemeroptera (mayflies), Trichoptera (caddisflies); Sphaeriidae (fingernail clams), and Odonata (dragonflies and damselflies). It summarizes the taxa richness of the listed taxonomic groups. These intolerant taxa are considered highly pollution-sensitive in wetland waters, hence greater abundances of these groups typically indicate greater site health. Although this index was developed for use with species-level identification, Mandaville (2002) indicated that it is also valid for use with family level identifications. The percent abundance was calculated using the following equation:

$$\text{ETSD abundance} = \frac{\text{total number of ETSD families}}{\text{total number of families}} \times 100 \qquad (4.8)$$

4.5.3.3. Mayfly Abundance. Although mayflies (Ephemeroptera) are a component of the ETSD Biotic Index, it was believed potentially significant to examine the abundance of this indicator group on its own, as mayflies are specifically sensitive to the Burnside treatment wetland site contaminants of concern (ammonia and metals). They are also sensitive to low dissolved oxygen (less than 5 ppm), chlorine, pesticides and acidity. The mayfly abundance was calculated as follows:

$$\text{Mayfly abundance} = \frac{\text{total number of mayfly families}}{\text{total number of families}} \times 100 \qquad (4.9)$$

4.5.3.4. Trophic Structure. Balance shifts in functional feeding groups (i.e. scrapers, shredders, collectors, predators) can be indicative of specific pollution-related stress. According to the USEPA (2002l) and Osmund et al. (1995c) increased ratios of predators to scrapers, shredders, and collectors can be indicative of heavy metal pollution, whereas decreased ratios of predators to scrapers, shredders, and collectors indicative of organic pollution and low dissolved oxygen. The results of the macroinvertebrate trophic structure analysis conducted in Section 4.4.4.3 using Equation 4.8 were re-examined with these specific pollution indicators in mind.

5. RESULTS

5.1. Selection of Vegetation

5.1.1. Model Wetland Plant Community Survey

The results of the Wright's brook marsh vegetation survey are shown in Tables 5.1 to 5.4. The assigned abundance coefficients (with 5 representing high abundance and 1 scarce abundance) for each species identified during the survey are indicated in the Rank column. In total, 21 emergent wetland plant species, 40 upland vascular plant species, 17 upland shrub species and 13 upland tree species were identified in the model site. Of those identified, 2 emergent wetland species, 20 upland vascular plants, and 1 upland shrub were disqualified as candidate species for the treatment wetland for reasoning indicated in the Note column of each table, which included the aggressive or invasive nature of the plant or the plant being considered exotic or a weed. No trees were disqualified as candidate species for the site.

Table 5.1. Wright's Brook Marsh Wetland Emergent Plants (Zinck, 1998).

SCIENTIFIC NAME	COMMON NAME	RANK (1-5)	NOTE*
Alisma plantago-aquatica	Water plantain	2	
Calamagrostis canadensis	Blue joint grass	3	
Carex brunnescens	Grey sedge	2	
Carex crinita	Fringed sedge	3	
Carex lurida	Yellow-green sedge	2	
Carex pseudocyperus	Cypernus sedge	2	
Carex stipata	Awl-fruited sedge	2	
Eleocharis acicularis	Needle spike rush	2	
Glyceria grandis	Reed manna-grass	2	
Glyceria striata	Fowl mannagrass	4	
Iris versicolor	Blue flag	1	
Juncus brevicaudatus	Tweedy's rush	2	
Juncus canadensis	Canada sedge	2	
Juncus effusus	Soft rush	4	
Nuphar variegata	Cow-lily	1	
Polygonum pensylvanicum	Pinkweed	2	
Pontederia cordata	Pickerelweed	1	
Scirpus cyperinus	Woolgrass	4	
Scirpus pungens	Common three square	1	
Scirpus validus	Soft stem bulrush	2	
Typha angustifolia	Narrow-leaved cattail	1	aggressive
Typha latifolia	Broad-leaved cattail	4	aggressive

*These plants were disqualified for the reasons noted

Table 5.2. Wright's Brook Marsh Wetland Upland Vascular Plants (Zinck, 1998).

SCIENTIFIC NAME	COMMON NAME	RANK (1-5)	NOTE*
Aralia nudicaulis	Wild sarsaparilla	4	
Aster spp.	Asters	3	
Bidens discoidea	Common beggar ticks	2	exotic
Brassica rapa	Field mustard	1	invasive
Capsella bursa-pastoris	Shepherd's-purse	1	weed
Chrysanthemum leucanthemum	Oxeye daisy	3	exotic
Cichorium intybus	Chicory	1	exotic
Cornus canadensis	Bunchberry	4	
Cypripedium acaule	Pink lady's slipper	1	
Equisetum arvense	Field horsetail	4	aggressive
Erigeron annuus	Daisy fleabane	2	exotic
Frageria virginiana	Strawberry	3	
Galium palustre	Common bedstraw	2	
Hesperis matronalis	Dame's rocket	2	exotic
Hieracium florentinum	Yellow hawkweed	2	invasive
Maianthemum canadense	Wild lily-of-the-valley	1	
Oenothera biennis	Evening primrose	2	
Onoclea sensibilis	Sensitive fern	3	
Osmunda cinnamomea	Cinnamon fern	2	
Phalaris arundinacea	Reed canary grass	2	exotic and invasive
Phleum pratense	Timothy hay	2	exotic
Potentilla simplex	Common cinquefoil	1	
Pteridium aquilinum	Bracken fern	4	
Ranunculus acris	Creeping buttercup	2	poisonous to grazers
Rorippa sylvestris	Creeping yellow cress	1	invasive
Rumex crispus	Curled dock	1	exotic
Senecio jacobaea	Tansy ragwort	1	exotic
Solanum dulcamara	Bittersweet	1	
Solidago spp.	Goldenrod	4	
Spartina alterniflora	Smooth cord grass	4	exotic
Thalictrum pubescens	Meadow rue	3	
Thlaspi arvense	Stinkweed	1	
Trientalis borealis	Starflower	3	
Trifolium pratense	Red clover	2	exotic
Tussilago farfara	Colt's foot	4	invasive
Verbascum thapsus	Common mullein	1	weed
Vicia cracca	Cow vetch	2	invasive
Viola conspersa	Dog violet	1	
Viola macloskeyi	Small white violet	1	

*These plants were disqualified for the reasons noted

5.1.2. Burnside Treatment Wetland Plant Species Selection Criterion

The selection criteria included phytoremediation potential, sedimentation and erosion control, habitat function, public deterrent potential and rate of plant establishment, tolerances, and requirements (Tables B.1 to B.4 in Appendix B). Species that were found to have high phytoremediation potential during the screening are shaded yellow in the list. Species that were found particularly effective at sediment and soil stabilisation are shaded blue. Species specially suited to habitat facilitation are shaded pink. Species found to be specially suited to the purpose of public deterrence are shaded orange. Finally, species which could simply be used to increase diversity, enhance habitat or aesthetics are shaded green. Those species found to have multiple benefits are shaded in accordance to what was deemed their most valuable asset.

Table 5.3. Wright's Brook Marsh Wetland Upland Shrubs (Zinck, 1998).

SCIENTIFIC NAME	COMMON NAME	RANK (1-5)	NOTE*
Alnus viridis	Speckled alder	2	
Amelanchier arborea	Shadbush/wild pear	3	
Aronia arbutifolia	Red chokeberry	1	
Comptonia peregrina	Sweet fern	2	
Diervilla lonicera	Bush honeysuckle	2	
Kalmia angustifolia	Sheep laurel	4	
Ledum groenlandicum	Labrador tea	2	
Prunus virginiana	Choke cherry	1	
Rhododendron canadense	Rhodora	4	
Rosa multiflora	Multiflora rose	3	exotic
Rosa nitida	Bristly-rose	2	
Rosa palustris	Swamp rose	2	
Rosa virginiana	Common wild rose	2	
Rubus strigosus	Raspberry	2	
Spiraea alba var. latifolia	Meadowsweet	5	
Vaccinium angustifolium	Blueberry	4	
Viburnum cassinoides	Witherod	3	

*These plants were disqualified for the reasons noted

Table 5.4. Wright's Brook Marsh Wetland Upland Trees (Zinck, 1998).

SCIENTIFIC NAME	COMMON NAME	RANK (1-5)	NOTE
Abies balsamea	Balsam fir	2	
Acer rubrum	Red maple	2	
Betula papyrifera	White birch	2	
Betula populifolia	Grey birch	4	
Fraxinus americana	White ash	2	
Larix laricina	Tamarack	1	
Picea glauca	White spruce	2	
Picea mariana	Black spruce	2	
Picea rubens	Red spruce	2	
Pinus strobes	White pine	2	
Populus grandidentata	Large toothed aspen	4	
Populus tremuloides	Trembling aspen	1	
Quercus rubra	Red oak	2	

5.1.2.1. Phytoremediation Potential. Woolgrass (*Scirpus cyperinus),* soft rush (*Juncus effusus),* and broad-leaved cattail (*Typha latifolia)* were all present in abundance in the Wright's brook marsh model site (Table 5.1). All three support large root systems and large biomass, and are easily transplanted. The results of the phytoremediation screening of these plants are presented in Tables 5.5 to 5.7. Given their potential phytoremediation capabilities, these three species would be excellent candidates for the domination of the treatment wetland berms. However, woolgrass and soft rush were selected as the species to dominate the site for several reasons: (a) woolgrass and soft rush have thick, rhizomous roots which are capable of penetrating to a depth of 2.5 to 3 feet or greater and are thus extremely useful in sediment stabilisation and oxygenation, (b) woolgrass and soft rush provide greater surface area for facilitation of microbial growth (Campbell and Ogden, 1999), (c) woolgrass and soft rush both have limited litter and do not contribute much detritus to a system (USEPA, 2001). Cattail was not selected for several reasons: (a) cattail are not likely to extend roots down to depths greater than one foot and therefore are not as efficient as rushes and bulrushes in providing aerobic surface area for microbes and biogeochemical cycling, (b) cattail contributes much litter to the waterbody every autumn, which results in increased oxygen consumption, causing decreased dissolved oxygen levels, (c) upon decay, the abundant cattail litter potentially releases much of the contaminants taken up back into the system and (d) cattails notoriously take over when competing with other plants for space and resources (Campbell and Ogden, 1999; USEPA, 2001). However, given their effectiveness at cleansing contamination, it would be unwise not to utilize these potentially aggressive plants in the future as a resource. The establishment of *Typha* in the future in the subject treatment wetland would have to be executed with extreme caution. According to Terry and Banuelos (1992), the pre-planting of bulrush or other species can prevent the invasion of *Typha* in a wetland system for decades. Therefore, it was decided that *Typha* would not be planted in the site until all other populations were established.

5.1.2.2. Sedimentation and Erosion Control. The species selected from the model plant lists as candidates for the treatment wetland as a result of their excellent sediment stabilization and erosion minimization capabilities are presented in Table 5.8 and 5.9, respectively.

5.1.2.3. Habitat Function. Species identified in the Wright's brook wetland model plants lists as supporting superior habitat facilitation capability are listed in Table 5.10.

5.1.2.4. Public Deterrent Potential. Species identified in the Wright's brook wetland model as supporting public deterrence capabilities are listed in Table 5.11.

Table 5.5. Phytoremediation Screening for Woolgrass (*Scirpus cyperinus*).

CONTAMINANT	CONCLUSIONS	SOURCE
Iron and Manganese	-Abundant in successful AMD treatment site, but not specifically studied	Ye et al., 2001b
Iron	-Abundant in successful treatment wetland; wetland reduced total iron discharge from 10 to 1 mg/l.	Campbell and Ogden, 1999
Metals	-Abundant in successful treatment wetland, but not specifically studied	Tousignant et al., 1999
Acid Mine Drainage (AMD)*	-Abundant in successful AMD treatment site, but not specifically studied	Demchik and Garbutt, 1999
Ammonia, Domestic wastewater	-Woolgrass and cattail treatment wetlands reduced ammonia and TKN 18-67.5% depending on location	Demchik and Garbutt, 1999
Manganese and Zinc, AMD metals	-Mn, Zn, Cu, Ni, B and Cr accumulated in Woolgrass, but only accounted for small % of overall removal	Mays and Edwards, 2000

*AMD wastewater is typically high in iron, manganese and ammonia

Table 5.6. Phytoremediation Screening for Soft Rush (*Juncus effusus*).

CONTAMINANT	CONCLUSIONS	SOURCE
Iron, Manganese, and AMD	-Soft rush and cattail pond 99% effective at Fe removal and 58% effective at Mn removal. -Aboveground and belowground tissues of soft rush had higher concentrations of S, Fe, Mn, Zn, Cd than cattail. -Concentrations of Fe and Mn in soft rush shoots approximately 4x greater cattail shoots. -Notably, overall uptake by both only accounted for less than 2.5% of the annual element loading rates: sediments were primary sink.	Ye et al., 2001b.
Iron	-Fe uptake by common wetland plants potentially key metal removal process in 'polishing treatment' applications (where wetland removing last few mg/L) -future studies to concentrate on *Typha latifolia, Juncus effusus*	Younger and Batty, 2002.
TSS, BOD, TKN, Ammonia, Phosphate	Soft rush 1 of 3 species in wetland which collectively removed 70% TSS and BOD, 60-60% TKN, ammonia and phosphate.	Coleman et al., 2001.
AMD	-Soft rush and cattail dominated successful AMD treatment site, but not specifically studied.	Treacy and Timpson, 1999.
Manganese and Zinc, AMD metals	-Mn, Zn, Cu, Ni, B and Cr accumulated in soft rush, but only accounted for small % of overall removal.	Mays and Edwards, 2000.

Table 5.7. Phytoremediation Screening for Broad-leaved Cattail (*Typha latifolia*).

CONTAMINANT	CONCLUSIONS	SOURCE
Iron and Manganese	-Series of 4 *Typha* wetland cells decreased Fe and Mn from inlet water by 94 and 94% respectively. -Notably, overall uptake by plants only accounted for less than 0.91 and 4.18 % of Fe and Mn uptake: Sediments were primary sink.	Ye et al., 2001a.
Iron and Manganese	-*Typha* wetlands effective removers of iron and manganese.	Snyder and Aharrah, 1985.
Iron	-Typha wetland reduced iron concentrations from 20-25 mg/L to 1 mg/L.	Kleinmann, 1985.
Ammonia	-Ammonia concentration reductions by laboratory-grown cattail shoots in tailings water of 40 mg/L to 3.7 mg/L in 1 day. -Cattail islands removed 0.8 to 0.9 g of ammonia from 300 L of mine water in 24 hours.	Boojum Research Limited, 2002.
TSS, BOD, TKN, Ammonia, Phosphate	Cattail 1 of 3 species in wetland which collectively removed 70% TSS and BOD, 60-60% TKN, ammonia and phosphate.	Coleman et al., 2001.
Iron	-Fe uptake by common wetland plants potentially key metal removal process in 'polishing treatment' applications (where wetland removing last few mg/L) -future studies to concentrate on *Typha latifolia, Juncus effusus.*	Younger and Batty, 2002.
Ammonia, Domestic wastewater	-Woolgrass and cattail treatment wetlands reduced ammonia and TKN 18-67.5% depending on location	Huang et al., 1999.

Table 5.8. Sediment Stabilizer Candidates for Burnside Treatment Wetland Cells (Thunhorst, 1993; Zinck, 1998; and Crow and Hellquist, 2000).

COMMON NAME	SCIENTIFIC NAME	RATIONALE
Woolgrass	*Scirpus cyperinus*	Dense root systems
Soft rush	*Juncus effusus*	Dense root systems
Pickerelweed	*Pontederia cordata*	Dense and prolific around littoral zones
Canada bluejoint grass	*Calamagrostis canadensis*	Dense root systems

5.1.2.5. Rate of Plant Establishment, Tolerances and Requirements. The Final Site Planting Lists shown in Tables B.1 to B.4 in Appendix B contain detailed summaries of all the candidate native species deemed suitable for potential establishment in the Burnside treatment wetland site. This list includes brief descriptions and photographs of the species, establishment rates, specific tolerances and growth requirements.

Table 5.9. Erosion Minimizing Candidates for Burnside Treatment Wetland Buffer Zones (Thunhorst, 1993 and Zinck, 1998).

COMMON NAME	SCIENTIFIC NAME	RATIONALE
Meadowsweet	*Spiraea latifolia*	Prolific growth, deep, dense rooting
Speckled alder	*Alnus viridis*	Prolific growth, deep, dense rooting
Trembling aspen	*Populus tremuloides*	Great in steep areas. Also effective hydrocarbon removers via evapotranspiration in upper 2-3 metres of soil (Raskin and Ensley, 2000).
Pinkweed	*Polygonum pensylvanicum*	Dense rooting
Large toothed aspen	*Populus grandidentata*	Great in steep areas. Also effective hydrocarbon removers via evapotranspiration in upper 2-3 metres of soil (Raskin and Ensley, 2000).

Table 5.10. Habitat Facilitator Candidates for Burnside Treatment Wetland Cells (Thunhorst, 1993).

COMMON NAME	SCIENTIFIC NAME	RATIONALE
Broad-leaved cattail	*Typha latifolia*	Food and cover
Narrow-leaved cattail	*Typha angustifolia*	Food and cover
Raspberry	*Rubus strigosus*	Food
Meadowsweet	*Spiraea latifolia*	Cover and noise barrier
Speckled alder	*Alnus viridis*	Cover and noise barrier
White spruce	*Picea glauca*	Cover and noise barrier
Black spruce	*Picea mariana*	Cover and noise barrier
Red spruce	*Picea rubens*	Cover and noise barrier
Witherod	*Viburnum cassinoides*	Food
Blueberry	*Vaccinium angustifolium*	Food
Choke cherry	*Prunus virginiana*	Food
Red chokeberry	*Aronia arbutifolia*	Food
Shadbush/wild pear	*Amelanchier arborea*	Food

Table 5.11. Deterrent Candidates for Burnside Treatment Wetland Cells (Thunhorst, 1993; Daigle and Havinga, 1996; and Zinck, 1998).

COMMON NAME	SCIENTIFIC NAME	RATIONALE
Speckled alder	*Alnus viridis*	Dense and very difficult to traverse through
Meadowsweet	*Spiraea latifolia*	Dense and difficult to traverse through
Swamp rose	*Rosa palustris*	Thorns
Bristly-rose	*Rosa nitida*	Thorns
Common wild rose	*Rosa virginiana*	Thorns

5.2. Vegetation Sources and Establishment

In total, approximately 1080 plants were transplanted into the wetland cells and buffer areas. Transplant success was high, with few mortalities observed over the course of the spring and summer of 2002. Figure 5.1 shows that general planting plan for the site whereas Figure 5.2 shows the planting layout for the location of each species established in the site in 2002. It should be noted that the locations of the species are approximate, and the representative symbols do not account for the size of the plants. In addition, the layout does not include incidental plantings associated with transplanted species (ie. grasses naturally intertwined with transplanted soft rush), nor does it include the many natural wetland plants which found their way to the wetland site on their own via wind dispersion of seeds, rhizomes and seed stock in existing and transplanted soils, and so on. By the fall of 2002, the treatment wetland site had already begun transformation from a barren landscape (Figure 5.3 to 5.6) to a lush wetland environment (Figures 5.7 to 5.10). Overall transplant success was gauged via mortality census in the spring of 2003.

5.3. Vegetation Establishment Success

The vegetation establishment success of the Burnside treatment wetland site is demonstrated in Figure 5.11, which displays the dead transplants observed in the site as per plant mortality census conducted in May, 2003. Overall, 138 dead transplants were observed, many of which had died as a direct result of washout. This computes to an overall site establish success rate of about 87.3%. The species which suffered the highest mortality rates were the pickerelweed, with approximately 50 dead plants, the meadowsweet with 32 observed dead plants and woolgrass with 27 dead plants. Next were the sheep laurel (7 dead plants), grey birch (6 dead plants), soft rush, large-toothed aspen, the roses and speckled alder (tied with 4 dead plants). According to Environment Canada (2000b), seeding in wetland projects typically sees a 30% survival rate. The seeding success of the woolgrass, soft rush and fowl mannagrass could never be accurately gauged, as the observed new shoots could be the result of natural spreading. However, new shoots of the three seeded species were notably abundant in the wetland, particularly in the latter cells of the site.

5.4. Biological Integrity Assessment

The results of the water quality analysis conducted for the wetland selected to act as the reference site for the biological integrity assessment are presented in Table 5.12. No parameters analysed were reported above guideline concentrations.

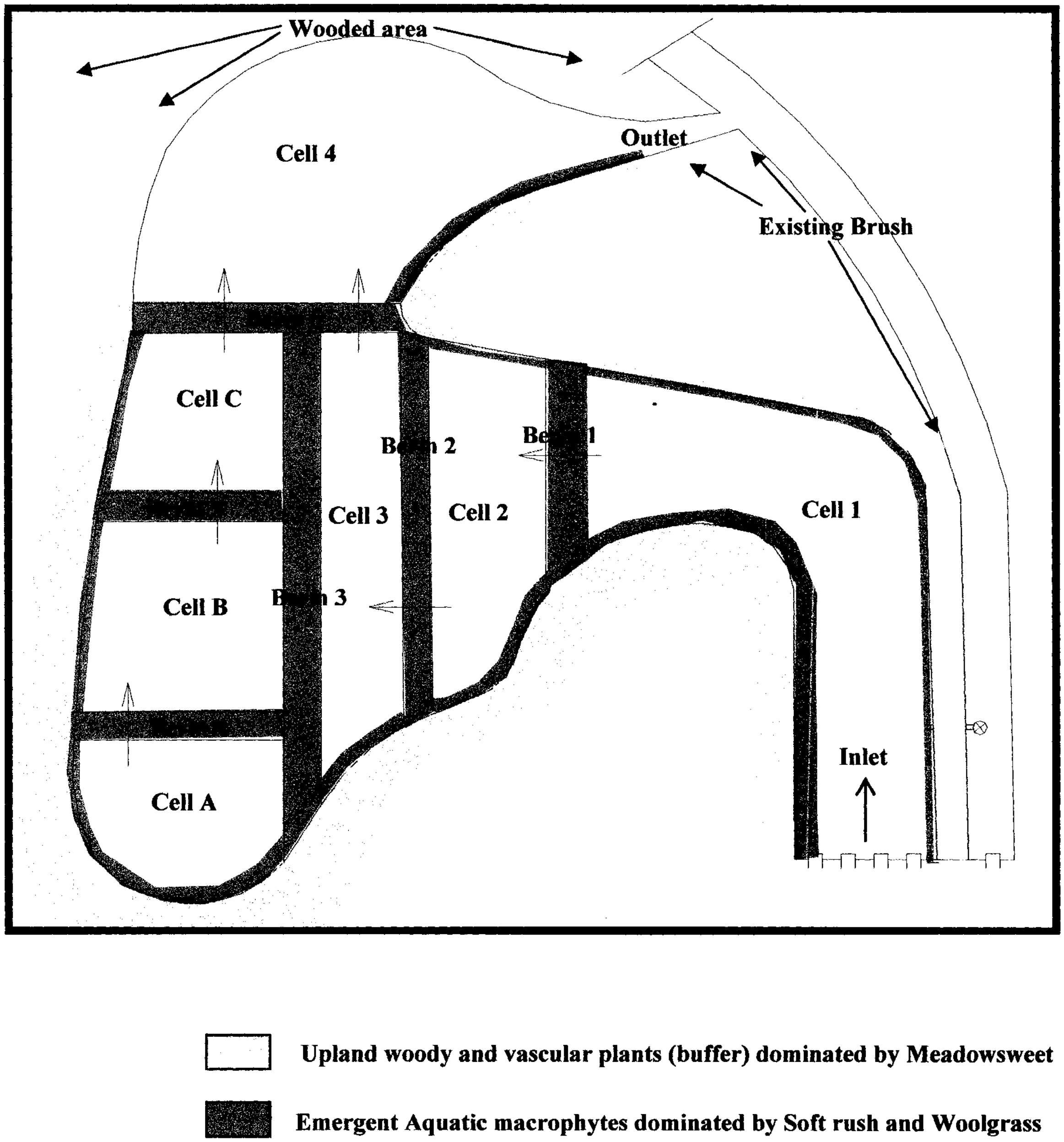
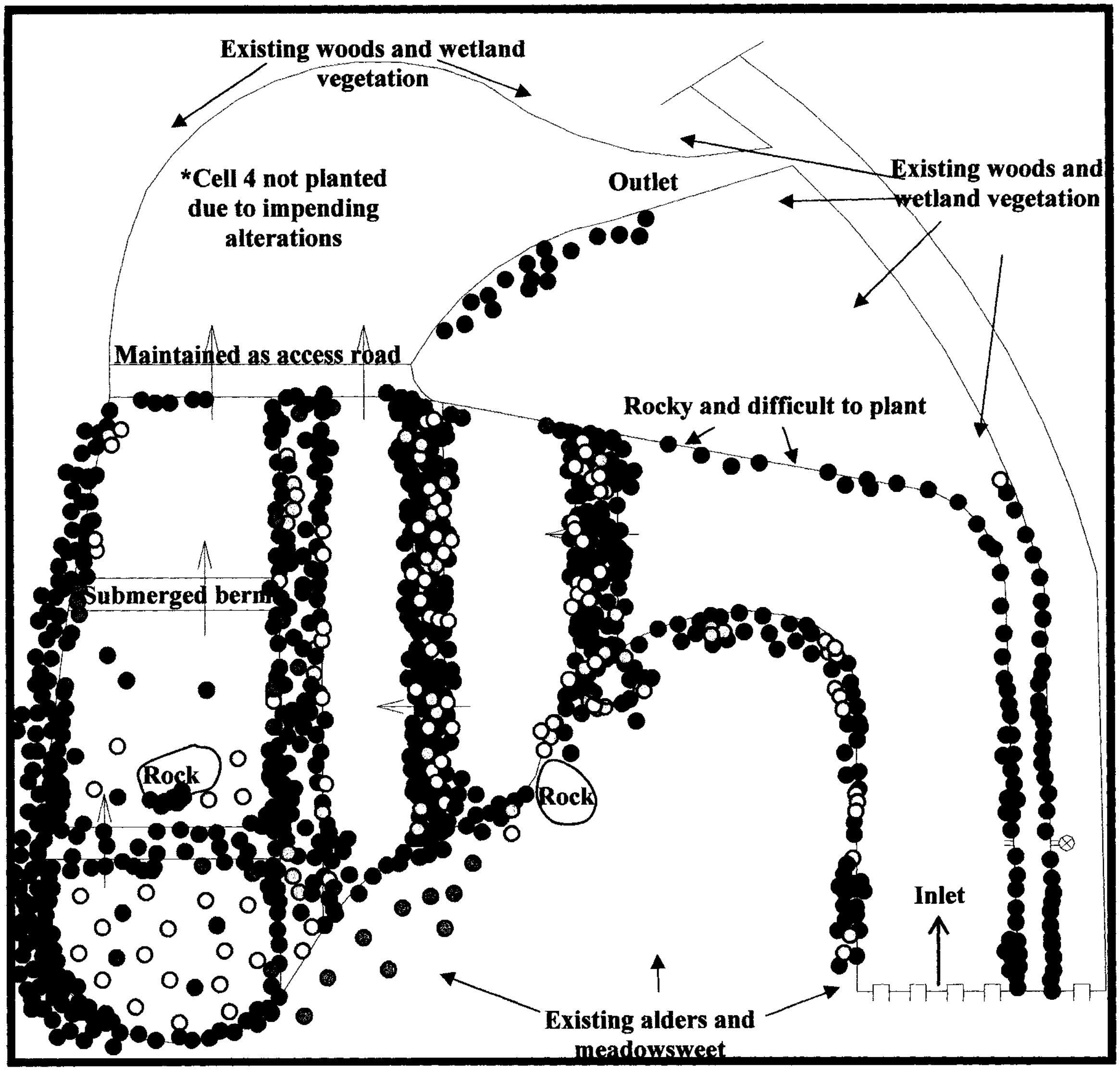


Figure 5.1. General Planting Plan.



- Woolgrass *(Scirpus cyperinus)*. Total =191
- Soft rush *(Juncus effusus)*. Total =216
- Pickerelweed *(Pontederia cordata)*. Total =79
- Yellow-green sedge *(Carex lurida)*. Total =38
- Cypernus sedge *(Carex pseudocyperus)*. Total =42
- Rhodora*(Rhododendron canadense)*. Total =5
- Witherod *(Viburnum cassinoides)*. Total =4
- Meadow-rue *(Thalictrum pubescens* Total=5
- Awl-fruited sedge *(Carex stipata)*. Total =44
- Cow lily (*Nuphar variegata*). Total=21
- Speckled alder (*Alnus rugosa*). Total=9
- Grey birch (*Betula populifolia*). Total=17
- Water plantain (*A. subcortum*). Total =14
- Tweedy's rush *(Juncus brevicaudatus)*. Total =33
- Trembling aspen *(Populus tremuloides)*. Total = 16
- Blueberry *(Vaccinium angustifolium)*. Total = 6
- Sheep laurel *(Kalmia angustifolia)*. Total =22
- Meadowsweet *(Spiraea alba var. latifolia)*. Total =264
- Oxeye daisy *(Chrysanthemum leucanthemum)*. Total =20
- Soft stem bulrush *(Scirpus validus)*. Total =2
- Rose species *(Rosa spp)*. Total =16
- Sweet fern (*Comptonia peregrina*). Total=8
- Blue flag (*Iris versicolor*). Total =4
- Grey sedge (*Carex brunnescens*). Total=6

Figure 5.2. Final Planting Layout, September, 2002.

Figure 5.3. Cell 1 and Portion of Berm 1, May 2002.

Figure 5.4. Berm 1, Cell 2 and Berm 2, May, 2002.

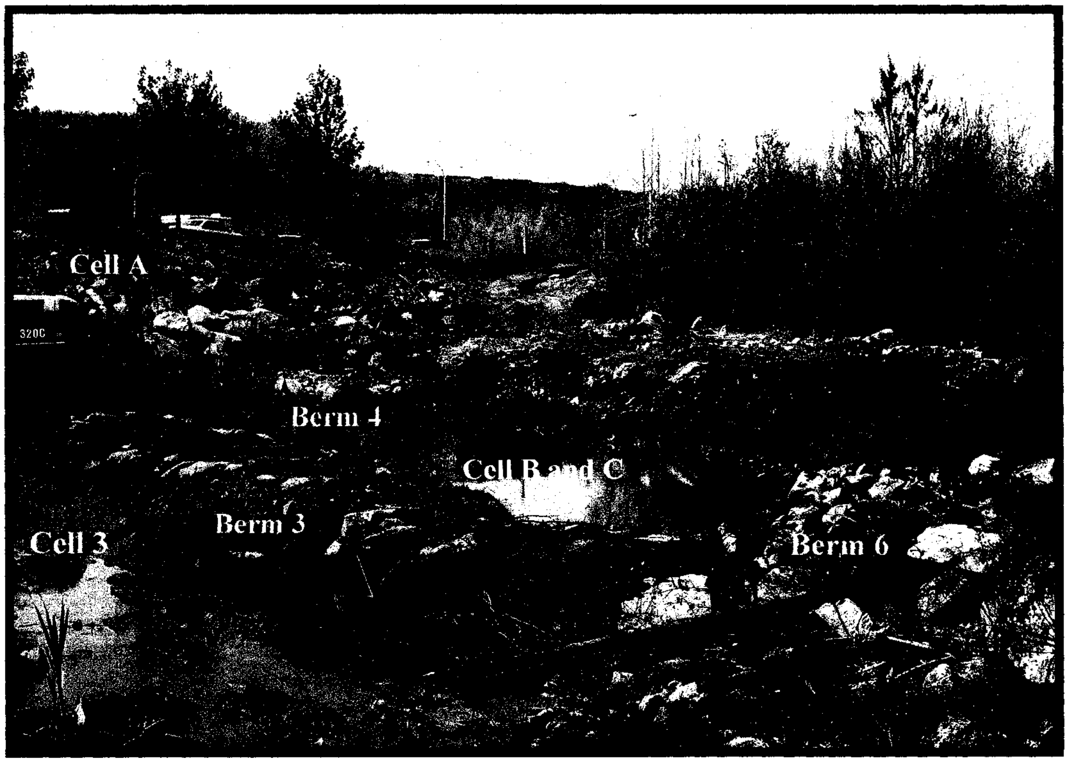


Figure 5.5. Cell 3, Berm 3 and 4, Cell A, B, and C, May, 2002.

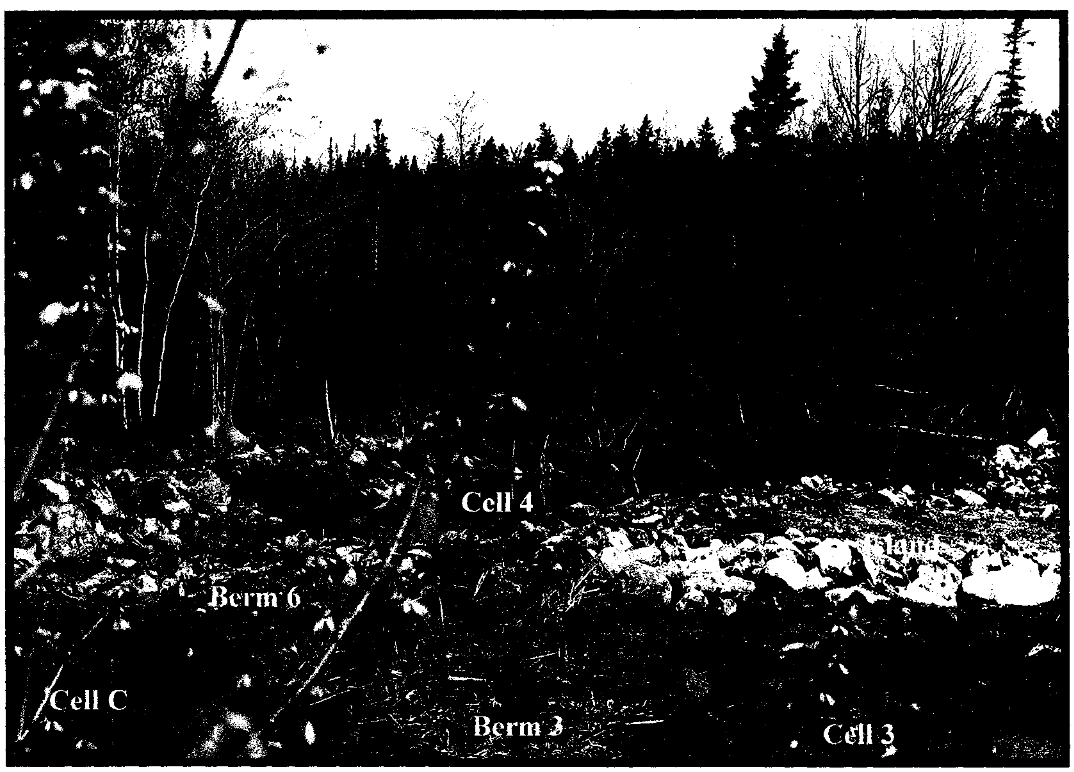


Figure 5.6. Cell C, Berm 6, Cell 4 and Island, May, 2002.

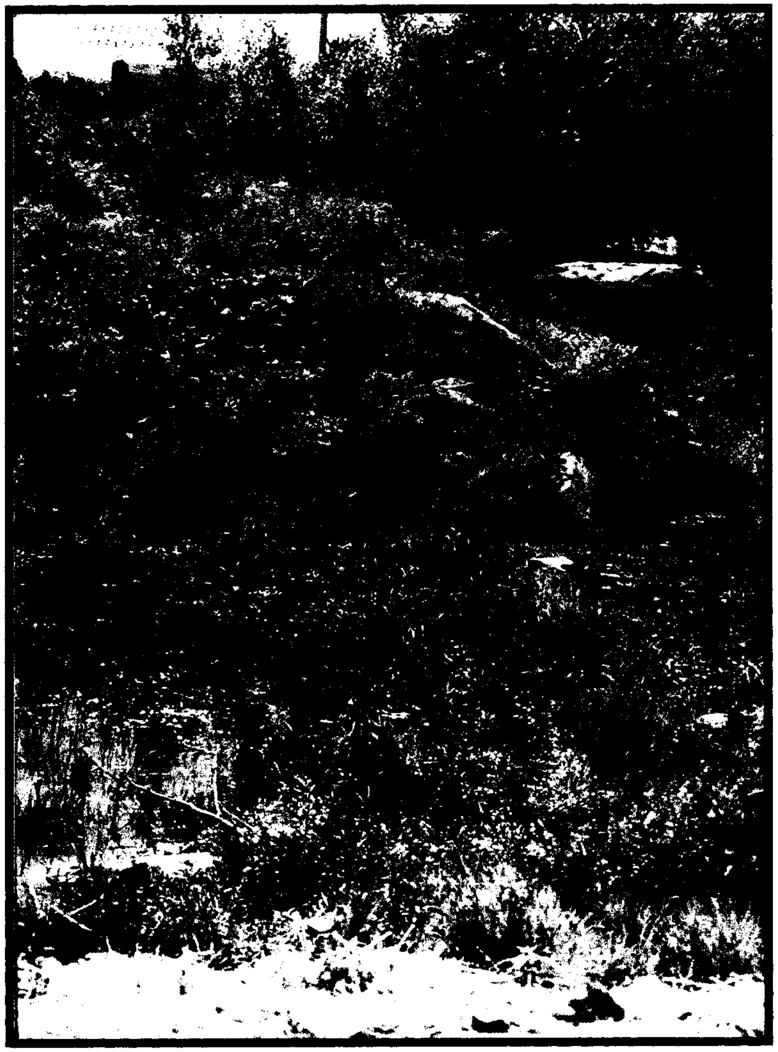


Figure 5.7. Berm 1, Fall, 2002.

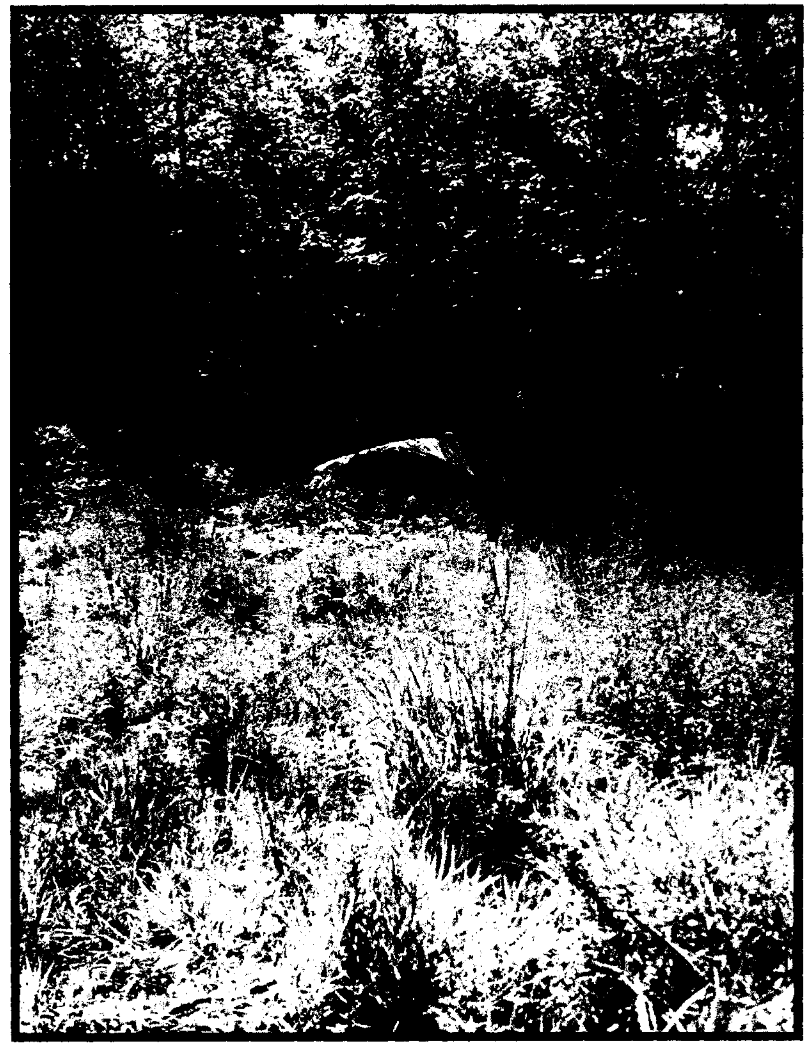
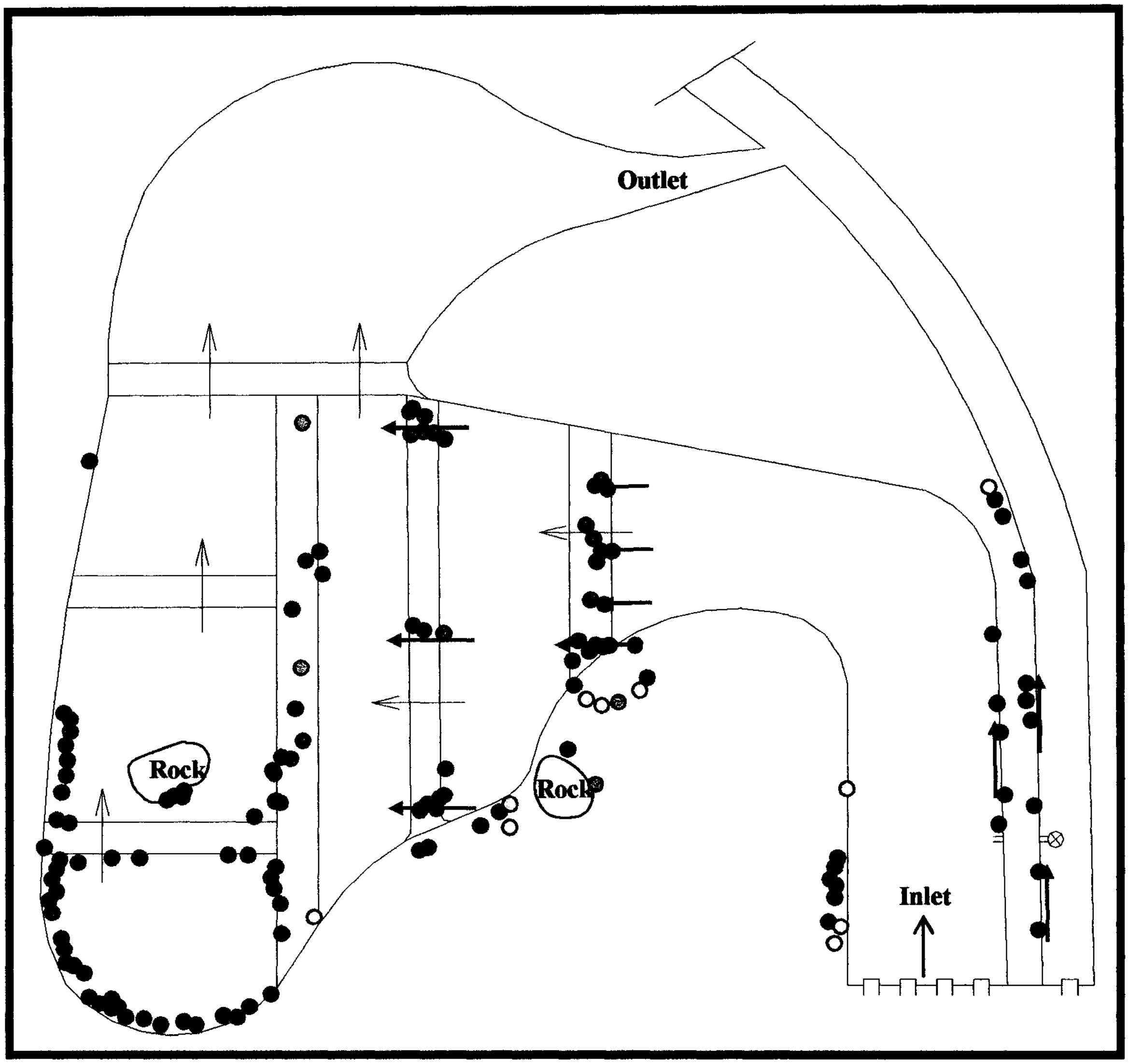


Figure 5.8. Berm 2, Fall, 2002.

Figure 5.9. Western Edge of Cell 1, Fall, 2002

Figure 5.10. Berm 3 and Cell B, Fall, 2002.



● **Pickerelweed (*Pontederia cordata*). 50/74**

● **Woolgrass (*Scirpus* cyperinus). 27/185**

● **Soft rush (*Juncus effusus*). 4/197**

● **Trembling aspen (*Populus tremuloides*). 4/15**

● **Meadowsweet (*Spiraea alba var. latifolia*). 32/264**

● **Sheep laurel (*Kalmia angustifolia*). 7/22**

○ **Rose species (*Rosa spp)*. 4/16**

● **Speckled alder (*Alnus rugosa*). 4/7**

○ **Grey birch (*Betula populifolia*). 6/17**

←——— **= Washout**

Figure 5.11. Transplant Mortalities and Washout, 2003.

Table 5.12. Reference Wetland Water Quality Results.

PARAMETER	EQL[1]	UNITS	CONCENTRATION	GUIDELINE
Benzene	0.001	mg/L	< 0.001	0.370[2]
Toluene	0.001	mg/L	< 0.001	0.002[2]
Ethylbenzene	0.001	mg/L	< 0.001	0.900[2]
Xylenes	0.002	mg/L	< 0.002	0.002[3]
C6 - C10 HC {less BTEX}	0.01	mg/L	< 0.010	NGA
Modified TPH - Tier 1	0.2	mg/L	< 0.200	NGA
>C10-C21 (Fuel Range)	0.05	mg/L	< 0.050	NGA
>C21-C32 (Lube Range)	0.1	mg/L	< 0.100	NGA
Sodium	0.1	mg/L	14.100	NGA
Potassium	0.1	mg/L	0.500	NGA
Calcium	0.1	mg/L	3.600	NGA
Magnesium	0.1	mg/L	0.600	NGA
Alkalinity (as CaCO3)	5	mg/L	6.000	NGA
Sulfate	2	mg/L	10.000	NGA
Chloride	1	mg/L	22.000	NGA
Reactive Silica (as SiO2)	0.5	mg/L	< 1.000	NGA
Ortho Phosphate (as P)	0.01	mg/L	< 0.010	NGA
Nitrite	0.01	mg/L	< 0.010	0.6[2]
Nitrate + Nitrite (as N)	0.05	mg/L	< 0.050	NGA
Nitrate (as N)	0.05	mg/L	< 0.050	NGA
Ammonia (as N)	0.05	mg/L	0.060	NGA
Color	5	TCU	18.000	NGA
Turbidity	0.1	NTU	1.400	NGA
Conductance (RCAp)	1	uS/cm	108.000	NGA
pH	-	Units	6.500	<6.5, >9.0[2]
Hardness (as CaCO3)	0.1	mg/L	11.500	NGA
Bicarbonate (as CaCO3)	1	mg/L	6.000	NGA
Carbonate (as CaCO3)	1	mg/L	< 1.000	NGA
TDS (Calculated)	1	mg/L	55.000	NGA
Cation Sum	0.1	meq/L	0.860	NGA
Anion Sum	0.1	meq/L	0.950	NGA
Ion Balance	-	%	5.090	NGA
Saturation pH @ 4C	-	Units	10.500	NGA
Saturation pH @ 4C	-	Units	10.500	NGA
Saturation pH @ 20C	-	Units	10.100	NGA
Aluminum	10	ug/L	110.000	5-100[2]
Antimony	2	ug/L	< 2.000	204.000
Arsenic	2	ug/L	< 2.000	52.000
Barium	5	ug/L	< 5.000	10003.000
Beryllium	2	ug/L	< 2.000	114.000
Bismuth	2	ug/L	< 2.000	NGA

[1]Estimated Quantitation Limit, which is the minimum concentration that can be reliably reported.
[2]Canadian Council of Ministers of the Environment Water Quality Guidelines for the Protection of Freshwater Life.
[3]British Columbia Minister of Environment, Lands, and Parks Environmental Water Quality Guidelines.
TPH Total Petroleum Hydrocarbons.
NGA No Guideline Available.

Table 5.12. Continued.

PARAMETER	EQL1	UNITS	CONCENTRATION	GUIDELINE
Boron	5	ug/L	< 5.000	200.000[4]
Cadmium	0.3	ug/L	< 0.300	0.012[2]
Chromium	2	ug/L	< 2.000	1-8.900[2]
Cobalt	1	ug/L	< 1.000	0.900[4]
Copper	2	ug/L	2.000	2-4.000[2]
Iron	50	ug/L	270.000	300.000[2]
Lead	0.5	ug/L	< 0.500	1-7.000[2]
Manganese	2	ug/L	150.000	1000.000-2000.000[3]
Molybdenum	2	ug/L	< 2.000	73.000[2]
Nickel	2	ug/L	< 2.000	25.000-150.000[2]
Selenium	2	ug/L	< 2.000	1.000[2]
Silver	0.5	ug/L	< 0.500	0.100[2]
Strontium	5	ug/L	13.000	NGA
Thallium	0.1	ug/L	< 0.100	0.800[2]
Tin	2	ug/L	< 2.000	NGA
Titanium	2	ug/L	2.000	NGA
Uranium	0.1	ug/L	< 0.100	NGA
Vanadium	2	ug/L	< 2.000	NGA
Zinc	5	ug/L	< 5.000	30.000[2]
Phosphorus	0.1	mg/L	< 0.100	NGA
Total Org. Carbon (by UV)	0.5	mg/L	3.600	NGA

[1]Estimated Quantitation Limit, which is the minimum concentration that can be reliably reported.
[2]Canadian Council of Ministers of the Environment Water Quality Guidelines for the Protection of Freshwater Life.
[3]British Columbia Minister of Environment, Lands, and Parks Environmental Water Quality Guidelines.
[4]Ontario Ministry of Environment Environmental Water Quality Objectives.
TPH Total Petroleum Hydrocarbons.
NGA No Guideline Available.

5.4.1. Vegetation

The data and calculations of the biological integrity assessment of the vegetation populations in both the treatment wetland site and the reference wetland site are presented in Tables C.1 and D.1 of Appendix C and D. The results are presented in Table 5.13. Species abundance and richness are illustrated graphically in Figure 5.12. Population heterogeneity as well as exotic species abundance are illustrated in Figures 5.13 and 5.14.

5.4.2. Aquatic Macroinvertebrates

The data and calculations of the biological integrity assessment of the aquatic macroinvertebrate populations in both the treatment wetland site and the reference wetland site are presented in Tables E.1 in Appendix E. The results are presented in Table 5.14. Population richness, heterogeneity, and trophic structure are illustrated graphically in Figure 5.15 to 5.17.

Table 5.13. Results of Biological Integrity Assessment of Wetland Plants.

METRIC	TREATMENT WETLAND	REFERENCE SITE
Species Richness	45	41
Species Diversity (H')	2.069	2.053
Variance (H'_{var})	0.025	0.025
Heterogeneity (Dominance)	47%	32%
Exotic Species Abundance	10.7%	3.8%

t statistic (t) = 0.071
Degrees of Freedom (df) = 530
Critical t values (α0.05) = 1.960 and 1.980

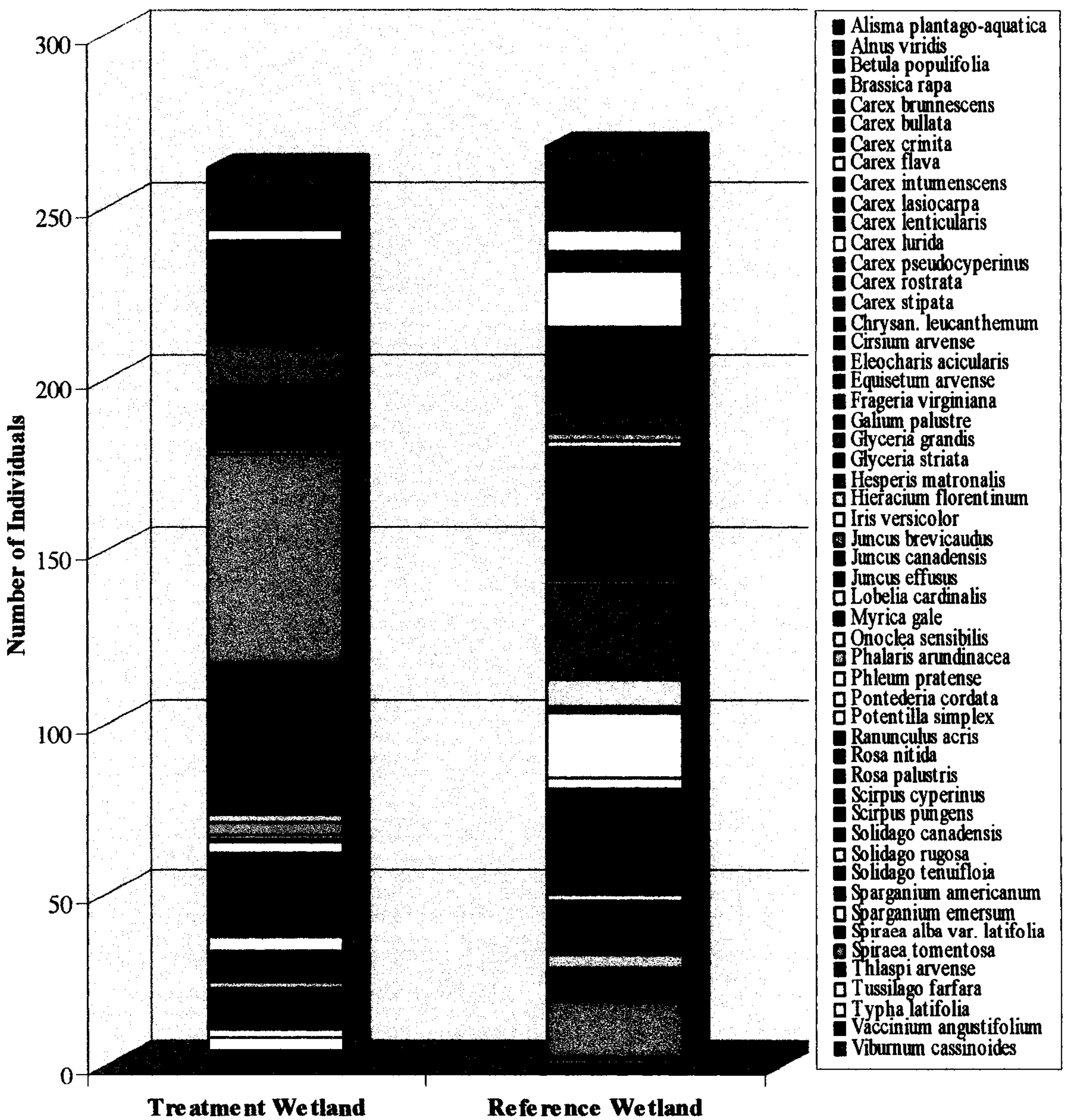
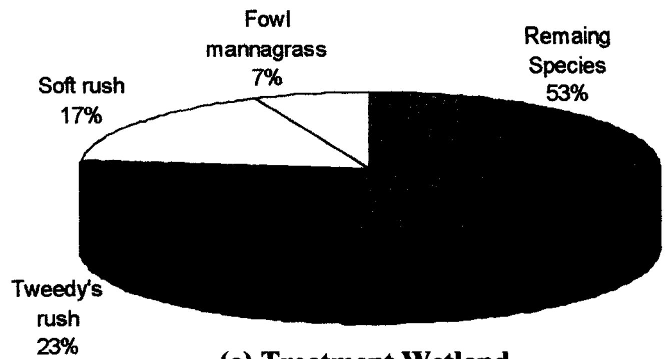
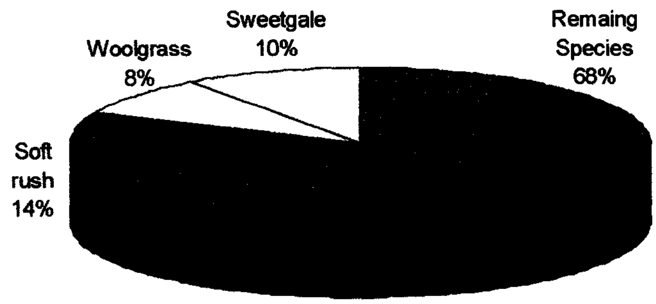


Figure 5.12. Vegetation Species Richness in Treatment Wetland and Reference Wetland Site.

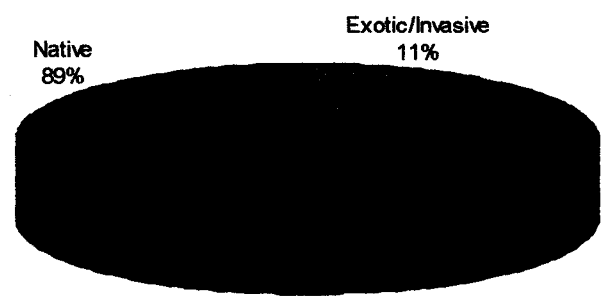


(a) Treatment Wetland

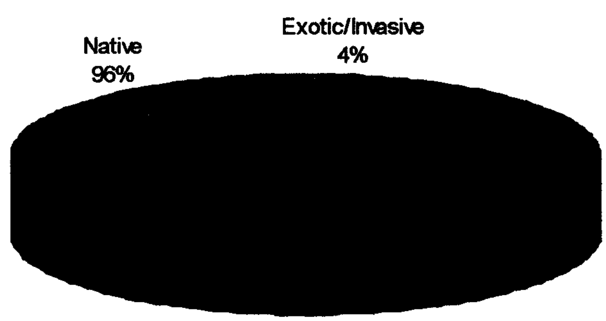


(b) Reference Wetland

Figure 5.13. Abundance of Top Three Plant Species in Treatment and Reference Wetland Sites.



(a) Treatment Wetland



(b) Reference Wetland

Figure 5.14. Abundance of Exotic/Invasive Plant Species in Treatment and Reference Wetland Sites.

Table 5.14. Results of Biological Integrity Assessment of Wetland Aquatic Macroinvertebrates.

METRIC	REFERENCE SITE	CELL 1	OUTLET
Population Richness	22	13	16
Population Diversity (H')	1.115	0.882	0.675
Variance (H'_{var})	0.028	0.022	0.023
Heterogeneity (Dominance)	58%	64%	67%
Trophic Structure			
Scrapers:	2%	0%	0%
Shredders:	2%	2%	0%
Collectors:	75%	96%	97%
Predators:	21%	2%	3%

t statistic (t)
Ref. to Cell 1 = 0.690
Ref. To Outlet = 1.595
Cell 1 to Outlet = 0.968

Degrees of Freedom (df)
Ref. to Cell 1 = 197
Ref. To Outlet = 198
Cell 1 to Outlet = 200

Critical t values ($\propto 2$ 0.05) = 1.96 and 1.98

5.4.3. Wildlife Observations

The wildlife observed in the Burnside treatment wetland site over the course of 2 years are listed in Table 5.15. Photographs of some of the wildlife observed in the site are displayed in Figure 5.18.

5.5. Water Purification Effectiveness

5.5.1. Chemical Water Quality Analysis

The weather data and some general observations of the site are presented in Table 5.16. The results of the water quality analysis conducted for the Burnside treatment wetland are presented in Table 5.17 and 5.18 and are illustrated graphically in Figures 5.19 to 5.29.

5.5.2. Plant Tissue Analysis

The results of the plant tissue analyses conducted on species from both the reference and treatment wetlands are presented in Table 5.19. The total iron and manganese concentrations in the various plants in the treatment wetland are presented in Figures 5.30 and 5.31. The iron and manganese concentrations in various plants of the reference and treatment wetlands are presented in Figure 5.32 and 5.33. The iron and manganese concentrations in roots, stems, leaves and flowers of the various plants in the treatment wetland are presented in Figure 5.34 and 5.35.

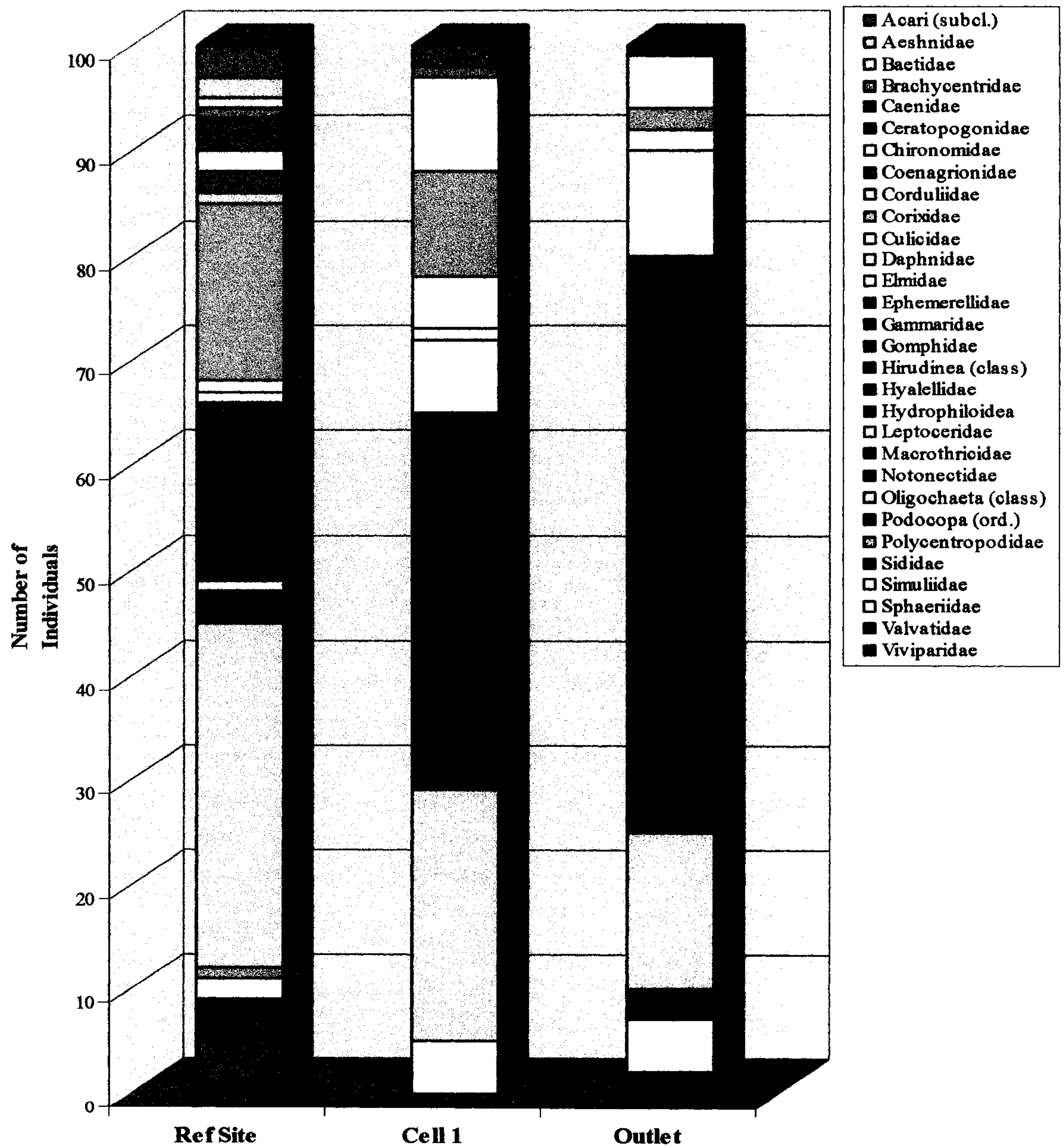
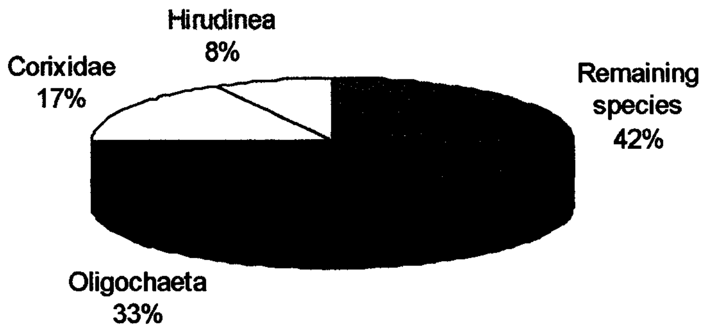
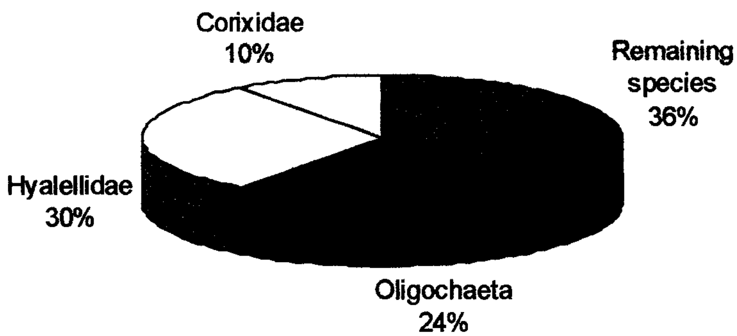


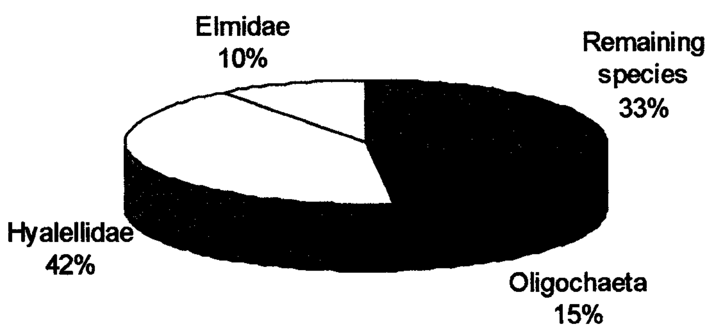
Figure 5.15. Aquatic Macroinvertebrate Richness in Reference Wetland, Cell 1 and Outlet.



(a) Reference site



(b) Cell 1



(c) Outlet

Figure 5.16. Abundance of Top Three Aquatic Macroinvertebrate Taxa in Reference Wetland, Cell 1 and Outlet.

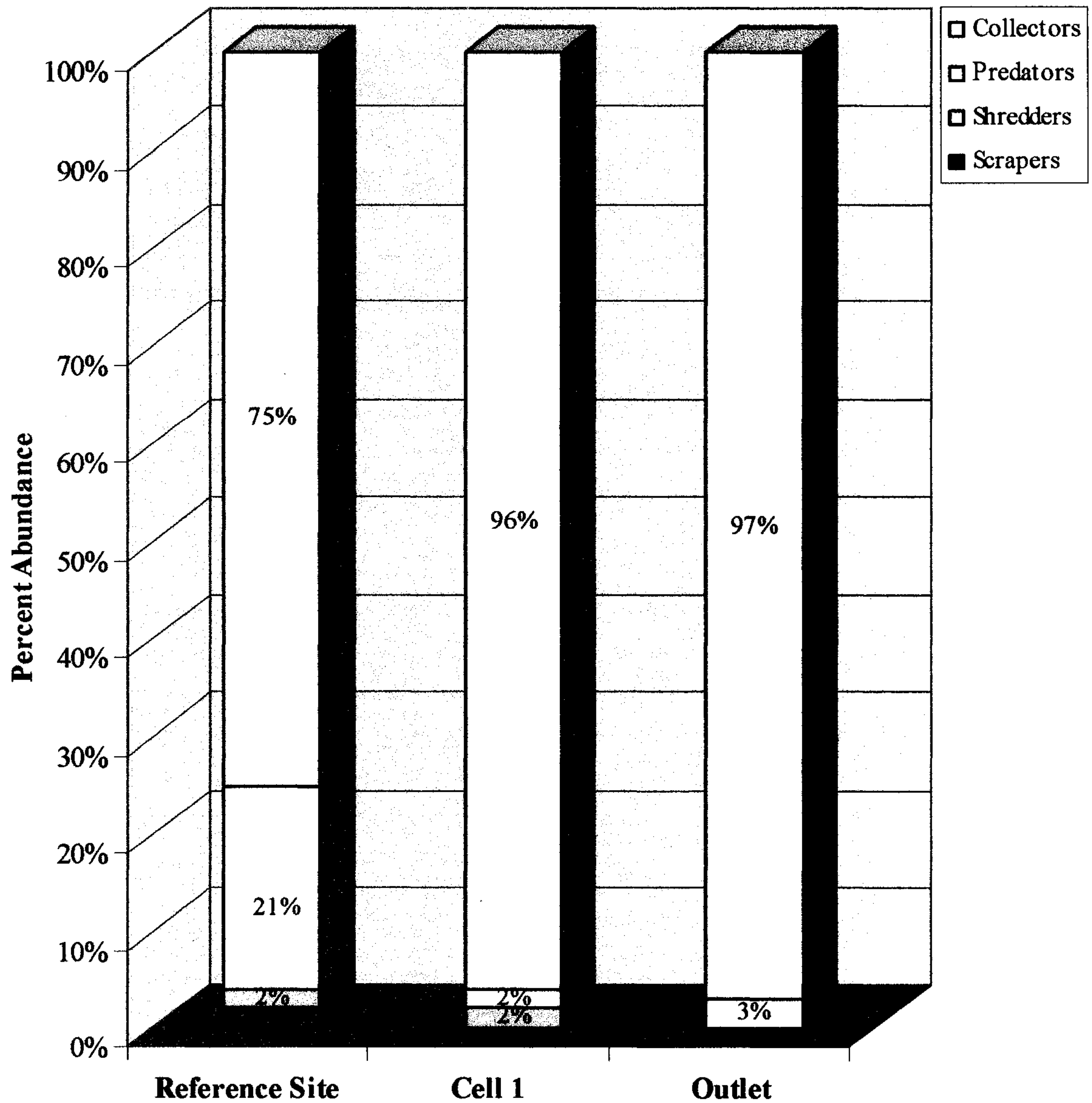


Figure 5.17. Trophic Structure of Aquatic Macroinvertebrates in Reference Wetland, Cell 1 and Outlet.

Table 5.15. Wildlife Observations, 2002-2003.

COMMON NAME	SCIENTIFIC NAME	LOCATION	EVIDENCE
Killdeer	*Charadrius vociferus*	Constructed wetland	Heard/observed
American Robin	*Turdus migratorius*	Constructed wetland	Heard/observed
American Crow	*Corvus brachyrhynchos*	Constructed wetland	Heard/observed
Great Blue Heron	*Ardea herodias*	Constructed wetland	Observed
American Black ducks	*Anas rubripes*	Constructed wetland	Observed
Mallard ducks	*Anas platyrhynchos*	Constructed wetland	Observed
Green-winged Teal ducks	*Anas crecca*	Constructed wetland	Observed
Hermit Thrush	*Catharus guttatus*	Constructed wetland	Observed/heard
Leopard Frogs	*Rana pipiens*	Constructed wetland	Observed
Green Frogs	*Rana clamitans*	Constructed wetland	Observed
Muskrat	*Ondatra zibethicus*	Constructed wetland	Observed
American Bullfrogs	*Rana catesbeiana*	Constructed wetland	Observed
Red-bellied Snake	*Storeria occipitomaculata*	Constructed wetland buffer	Observed
Osprey	*Pandion haliaetus*	Wetland –fly over	Observed
Belted Kingfisher	*Ceryle alcyon*	Wetland –fly over	Observed
Ovenbird	*Seiurus aurocapillus*	Mature softwood stand[1]	Heard
Northern Flicker	*Colaptes auratus*	Mature softwood stand[1]	Heard
Swainson's Thrush	*Catharsis ustulatus*	Mature softwood stand[1]	Heard
Hairy Woodpecker	*Picoides villosus*	Mature softwood stand[1]	Heard
Veery	*Catharus fuscescens*	Mature softwood stand[1]	Heard
Ruby-crowned Kinglet	*Regulus calendula*	Mature softwood stand[1]	Heard
Red Squirrel	*Tamiasciurus hudsonicus*	Mature softwood stand[1]	Heard
Black-and-white Warbler	*Mniotilta varia*	Successional woods[2]	Heard
Dark-eyed Junco	*Junco hyemalis*	Successional woods[2]	Heard
Chestnut-sided Warbler	*Dendroica pensylvanica*	Successional woods[2]	Heard
Magnolia Warbler	*Dendroica magnolia*	Successional woods[2]	Heard
Common Grackle	*Quiscalus quiscula*	Successional woods[2]	Heard
Golden-crowned Kinglet	*Regulus satrapa*	Successional woods[2]	Heard
Blackburnian Warbler	*Dendroica fusca*	Successional woods[2]	Heard/observed
Black-capped Chickadee	*Poecile atricapillus*	Successional woods[2]	Heard/observed
American Goldfinch	*Carduelis tristis*	Successional woods[2]	Observed
Ruffed Grouse	*Bonasa umbellus*	Successional woods[2]	Observed
Porcupine	*Erethizon dorsatum*	Successional woods[2]	Observed
Common Garter Snake	*Thamnophis sirtalis*	Successional woods[2]	Observed
Eastern Smooth Green Snake	*Liochlorophis vernalis borealis*	Successional woods[2]	Observed

[1] Mature softwood stand dominated by white and red spruce located along eastern side of Wright's brook.

[2] Successional woods dominated by low shrubs, birch and poplar located along western side of Wrights brook.

Table 5.15. Continued.

COMMON NAME	SCIENTIFIC NAME	LOCATION	EVIDENCE
White-throated Sparrow	*Zonotrichia albicollis*	Successional woods[2]/ mature softwood stand[1]	Heard
Black-throated Green Warbler	*Dendroica virens*	Successional woods[2]/ mature softwood stand[1]	Heard/observed
White-tailed Deer	*Odocoileus virginianus*	Successional woods[2]/ mature softwood stand[1]	Observed/scat
Snowshoe Hare	*Lepus americanus*	Successional woods[2]/ mature softwood stand[1]	Scat

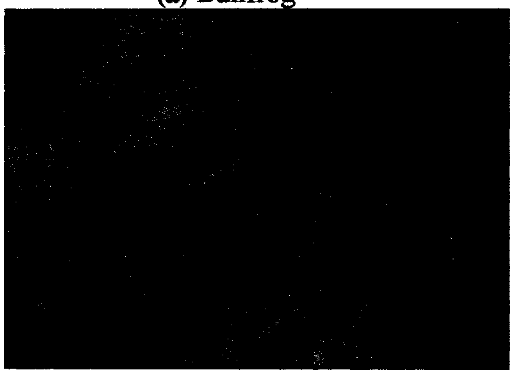
[1] Mature softwood stand dominated by white and red spruce located along eastern side of Wright's brook.
[2] Successional woods dominated by low shrubs, birch and poplar located along western side of Wrights brook.

Table 5.16. Weather, Water Temperature and General Observations.

SAMPLING DATE	WEATHER	WATER TEMPERATURE	OBSERVATIONS
May 24, 2003	Cloudy with fog. Temperature: 9.8°C. Rain 24 hrs prev?: No	10.6°C	Prolific algal bloom in Cell C. There was no rain in May
June 1, 2003	Mostly cloudy with rain. Temperature: 12.3°C. Rain 24 hrs prev?: No	12.2°C	Algal bloom in Cell C.
June 18, 2003	Mostly cloudy. Temperature: 18.1°C. Rain 24 hrs prev?: No	13.4°C	Algal bloom mostly dissipated
June 30, 2003	Mainly clear with drizzle. Temperature: 20.6°C. Rain 24 hrs prev?: No.	15°C	Water levels normal. Little sign of algal bloom.
July 21, 2003	Mostly cloudy. Temperature: 19.5°C. Rain 24 hrs prev?: No.	19.4°C	Much iron floc in cells and stagnation evidence
Aug. 6, 2003	Cloudy and rainy. Temperature: 20.1°C. Rain 24 hrs prev?: Yes (29.1 mm on 5th and 17.9 mm of 4th).	20.3°C	Water levels high- evidence water flows were powerful
Aug. 25, 2003	Mainly clear. Temperature: 13.1°C. Rain 24 hrs prev?: No	15.1°C	Much iron floc in cells, evidence of stagnation
Sept. 11, 2003	Mainly clear. Temperature: 11.8°C. Rain 24 hrs prev?: No	15.7°C	Much iron floc evident in cells

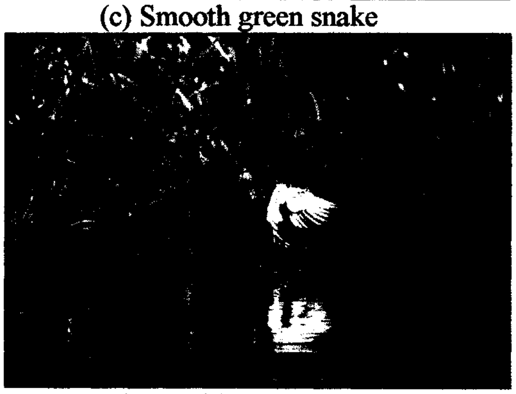
(a) Bullfrog

(a) Porcupine



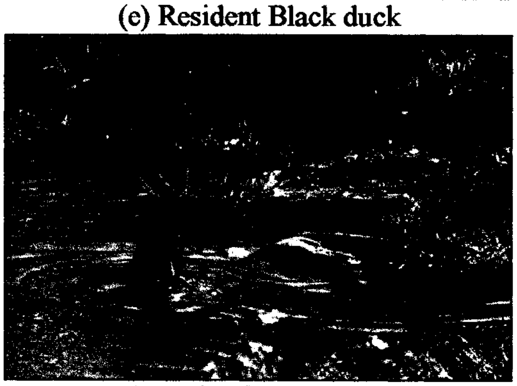
(c) Smooth green snake

(d) Family of Black ducks



(e) Resident Black duck

(f) White-tailed deer track



(g) Pair of mallard ducks

(h) Mammalian skull

Figure 5.18. Photographs of Wildlife at Burnside Wetland Site.

Table 5.17. Iron, Manganese, Orthophosphate and Nitrogen Compounds.

SAMPLING DATE	LOC.	Fe (mg/L)	Mn (mg/L)	NO_2 (mg/L)	NO_3 (mg/L)	NH_4^+ (mg/L)	TKN (mg/L)	PO_4 (mg/L)
May 24, 2003	Cell 1*	10.55	1.61	Nd	5.65	Nd	Nd	Nd
	Cell 2	5.62	1.42	Nd	2.39	0.50	4.00	Nd
	Cell 3	7.16	1.69	Nd	6.88	Nd	9.00	0.01
	Outlet	7.00	1.78	0.01	7.10	Nd	3.00	0.01
June 1, 2003	Cell 1*	11.05	1.48	0.02	4.49	0.50	7.50	Nd
	Cell 2	13.34	1.73	0.02	3.62	1.75	4.00	Nd
	Cell 3	7.37	1.76	0.01	2.46	0.25	9.25	Nd
	Outlet	9.38	1.78	0.01	6.74	Nd	4.25	Nd
June 18, 2003	Cell 1*	5.50	1.10	Nd	Nd	0.47	0.47	Nd
	Cell 2	5.73	1.18	0.02	7.03	Nd	4.25	Nd
	Cell 3	1.40	0.18	0.03	3.12	Nd	11.75	Nd
	Outlet	0.67	0.50	Nd	3.04	Nd	4.75	Nd
June 30, 2003	Cell 2	3.82	1.72	Nd	1.45	Nd	3.50	Nd
	Cell 3	5.96	1.50	Nd	2.10	Nd	19.00	Nd
	Outlet	3.75	1.76	0.02	2.90	Nd	5.25	Nd
July 21, 2003	Cell 2	7.69	1.45	Nd	5.73	3.00	4.25	0.03
	Cell 3	10.89	3.81	0.02	4.42	0.25	5.25	0.01
	Outlet	2.51	0.73	0.59	4.28	0.25	6.50	0.01
Aug. 6, 2003	Cell 2	1.05	0.44	0.24	2.46	0.25	1.25	Nd
	Cell 3	3.13	0.40	0.04	3.91	0.25	4.00	0.03
	Outlet	1.84	0.42	0.07	5.29	0.25	5.00	0.01
Aug. 25, 2003	Cell 2	8.68	2.12	Nd	1.88	2.00	7.00	Nd
	Cell 3	13.88	2.32	Nd	1.59	1.50	5.25	0.03
	Outlet	7.52	2.55	0.01	1.96	0.25	0.50	Nd
Sept. 11, 2003	Cell 2	10.43	1.37	0.03	1.38	0.25	2.00	0.01
	Cell 3	13.98	1.36	0.01	1.16	1.50	5.00	Nd
	Outlet	8.65	1.52	0.01	1.74	1.50	4.00	Nd

*Concentrations indicative of inlet concentrations as water sample taken in proximity to inlet in unvegetated area of Cell 1.

Nd = Not detected

TKN= Total Kjeldahl Nitrogen

Table 5.18. pH, Dissolved Oxygen, Chemical Oxygen Demand, Total Suspended Solids and Total Dissolved Solids.

DATE	LOC.	TSS (mg/L)	TDS (mg/L)	COD (mg/L)	DO (mg/L)	pH
May 24, 2003	Cell 1*	365	305	689	6.2	7.2
	Cell 2	390	376	546	6.1	7.3
	Cell 3	615	611	754	6.2	7.1
	Outlet	490	236	650	5.7	7.6
June 1, 2003	Cell 1*	460	247	351	4.3	7.5
	Cell 2	405	395	728	4.7	7.5
	Cell 3	410	388	676	4.3	7.6
	Outlet	513	462	884	4.0	7.6
June 18, 2003	Cell 1*	NA	255	NA	NA	6.7
	Cell 2	845	762	741	5.0	7.2
	Cell 3	970	883	806	6.4	7.5
	Outlet	893	871	676	4.5	7.6
June 30, 2003	Cell 2	400	345	637	4.2	7.6
	Cell 3	758	500	858	2.5	6.9
	Outlet	518	493	1027	3.9	7.5
July 21, 2003	Cell 2	523	509	1092	4.1	7.2
	Cell 3	605	576	1079	3.5	6.9
	Outlet	513	276	1001	4.4	7.19
Aug. 6, 2003	Cell 2	375	115	897	4.1	7.1
	Cell 3	365	124	1131	4.3	6.8
	Outlet	370	136	1066	4.4	6.8
Aug. 25, 2003	Cell 2	525	440	1209	4.1	6.8
	Cell 3	515	383	1053	4.4	6.7
	Outlet	550	356	858	3.7	7.7
Sept. 11, 2003	Cell 2	535	407	741	4.6	7.4
	Cell 3	410	306	1053	4.4	7.0
	Outlet	410	334	1092	4.6	6.7

*Concentrations indicative of inlet concentrations as water sample taken in proximity to inlet in unvegetated area of Cell 1.

NA = Not analysed
DO = Dissolved Oxygen
COD = Chemical Oxygen Demand
TDS = Total Dissolved Solids
TSS = Total Suspended Solids

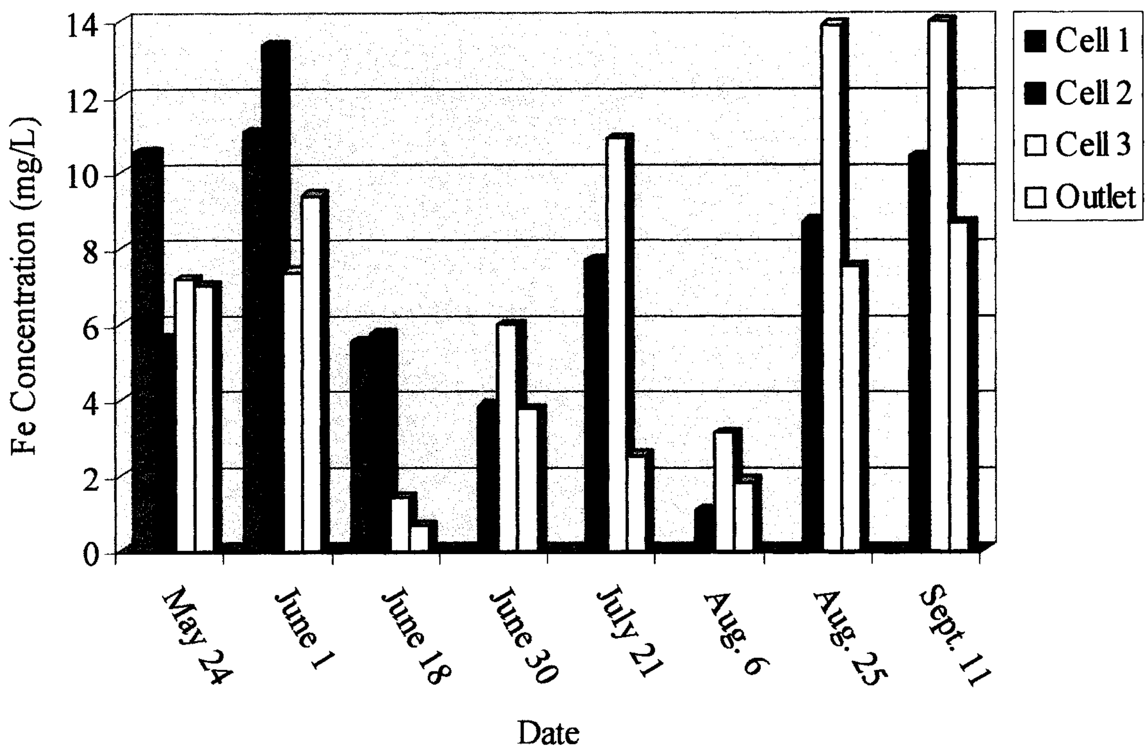


Figure 5.19. Iron Concentration in the Water of the Treatment Wetland, 2003.

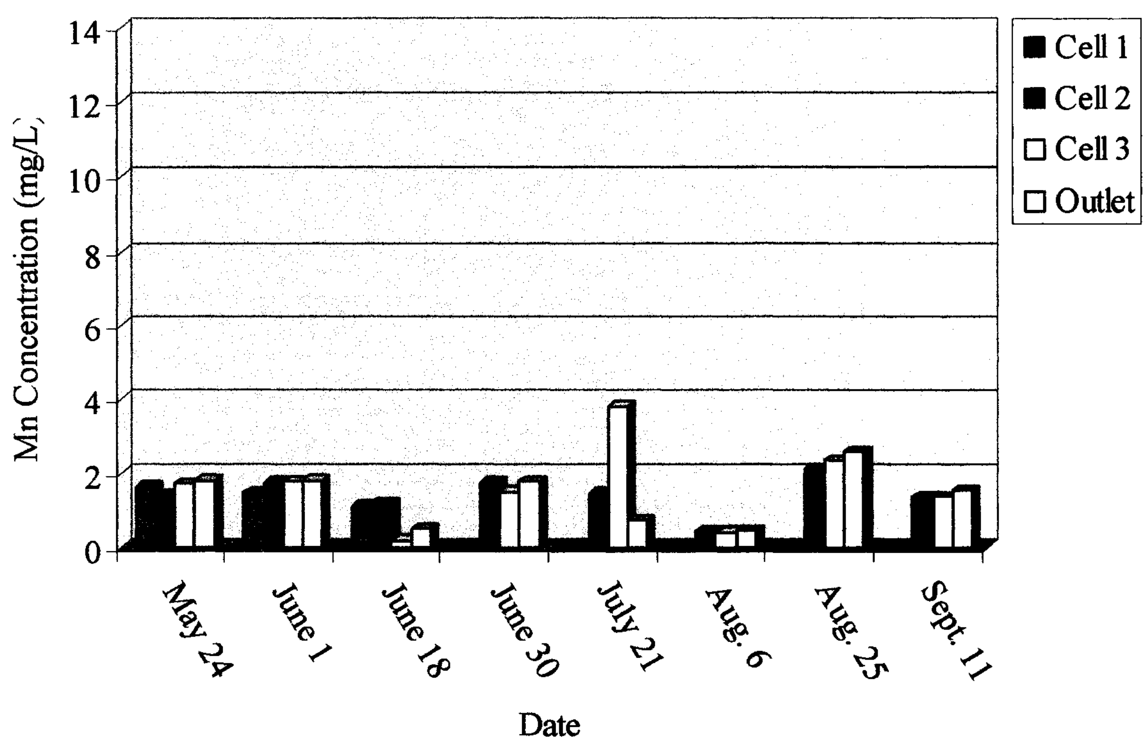


Figure 5.20 Manganese Concentration in the Water of the Treatment Wetland, 2003.

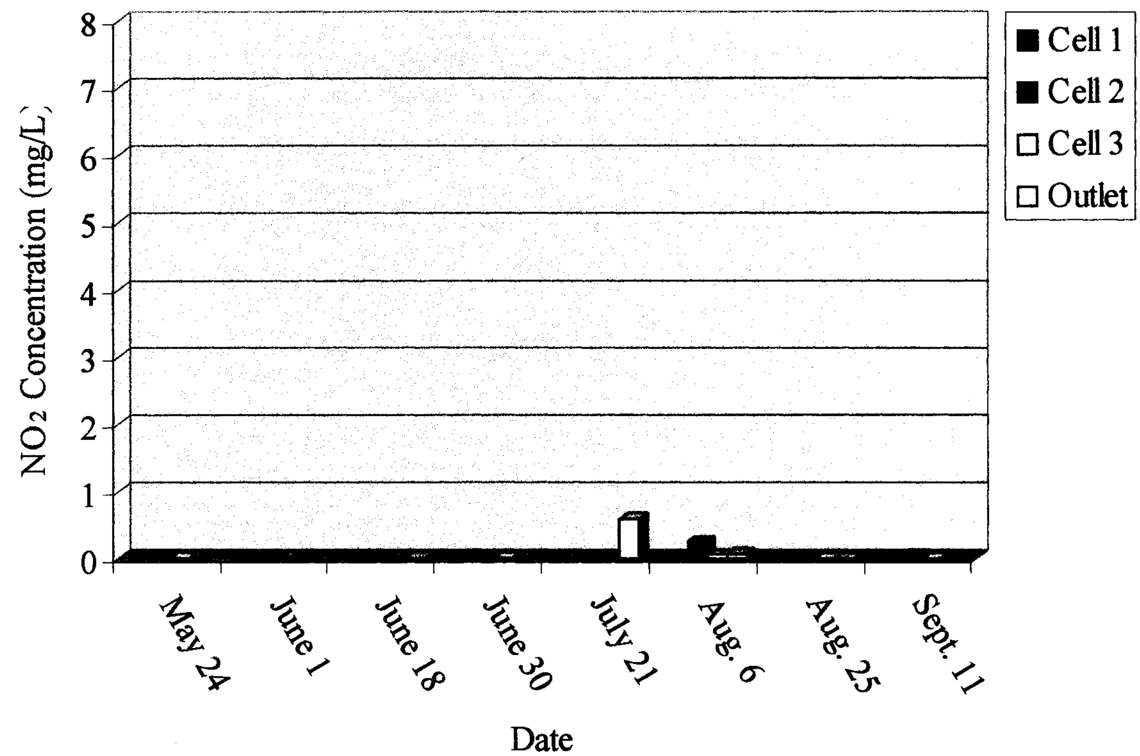


Figure 5.21. Nitrite Concentration in the Water of the Treatment Wetland, 2003.

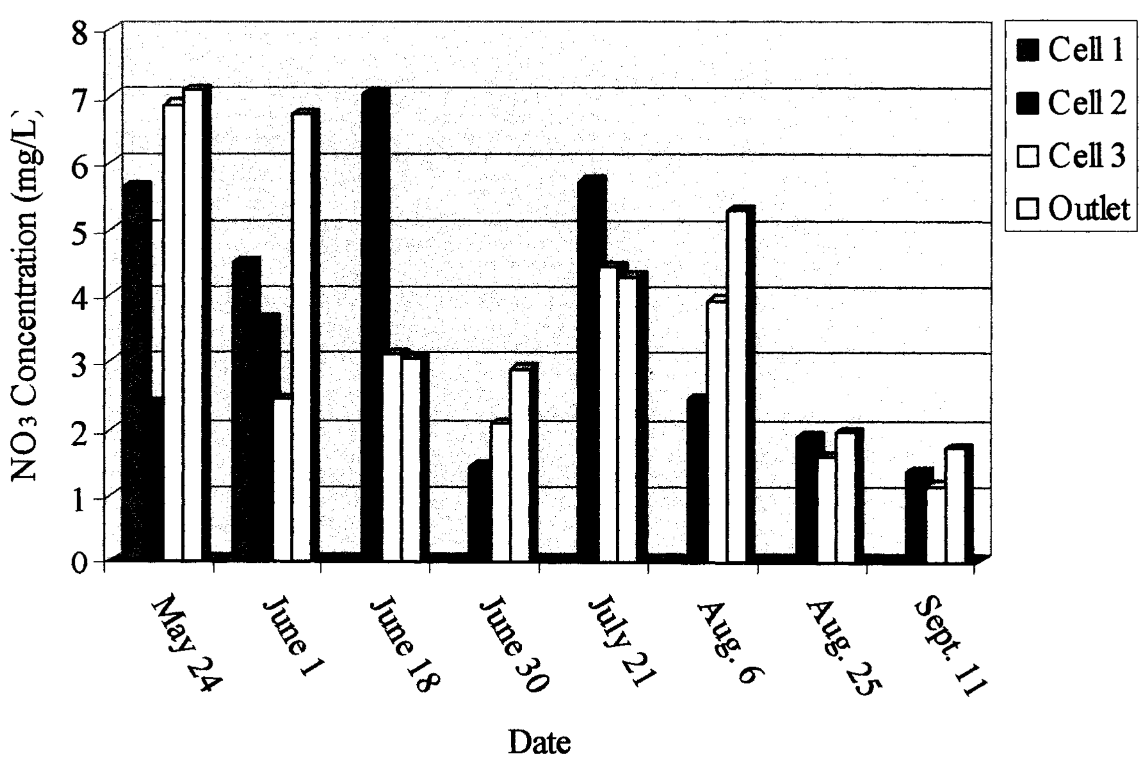


Figure 5.22. Nitrate Concentration in the Water of the Treatment Wetland, 2003.

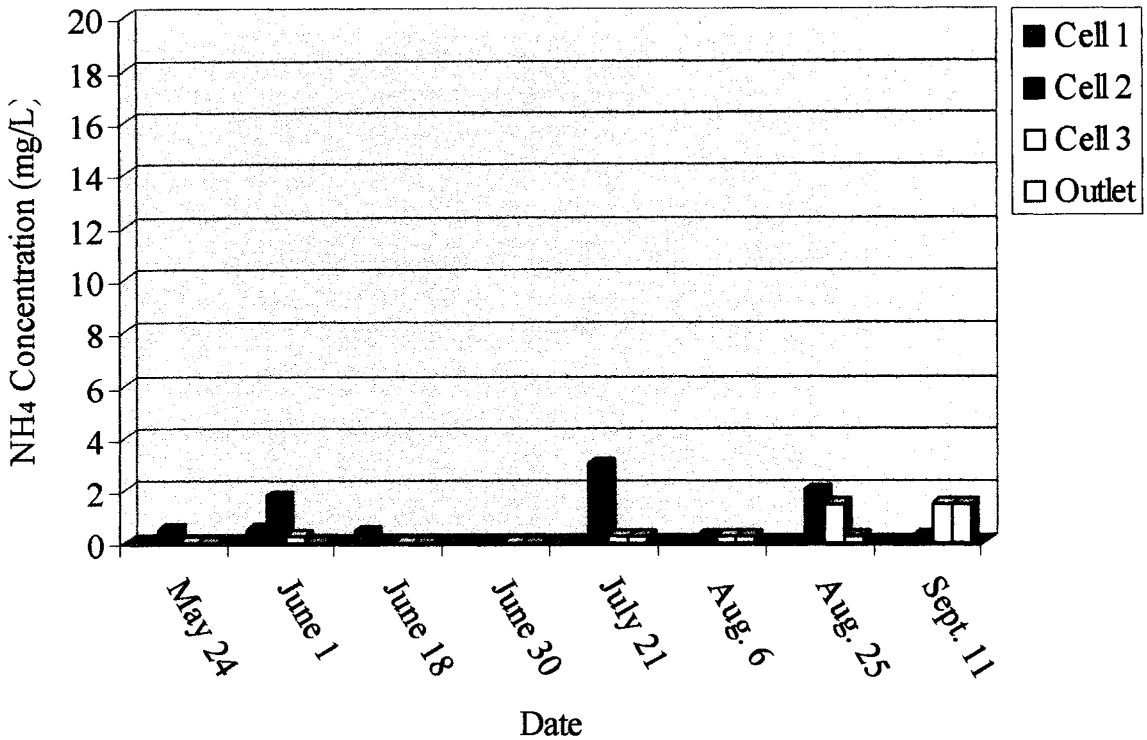


Figure 5.23. Ammonia Concentration in the Water of the Treatment Wetland, 2003.

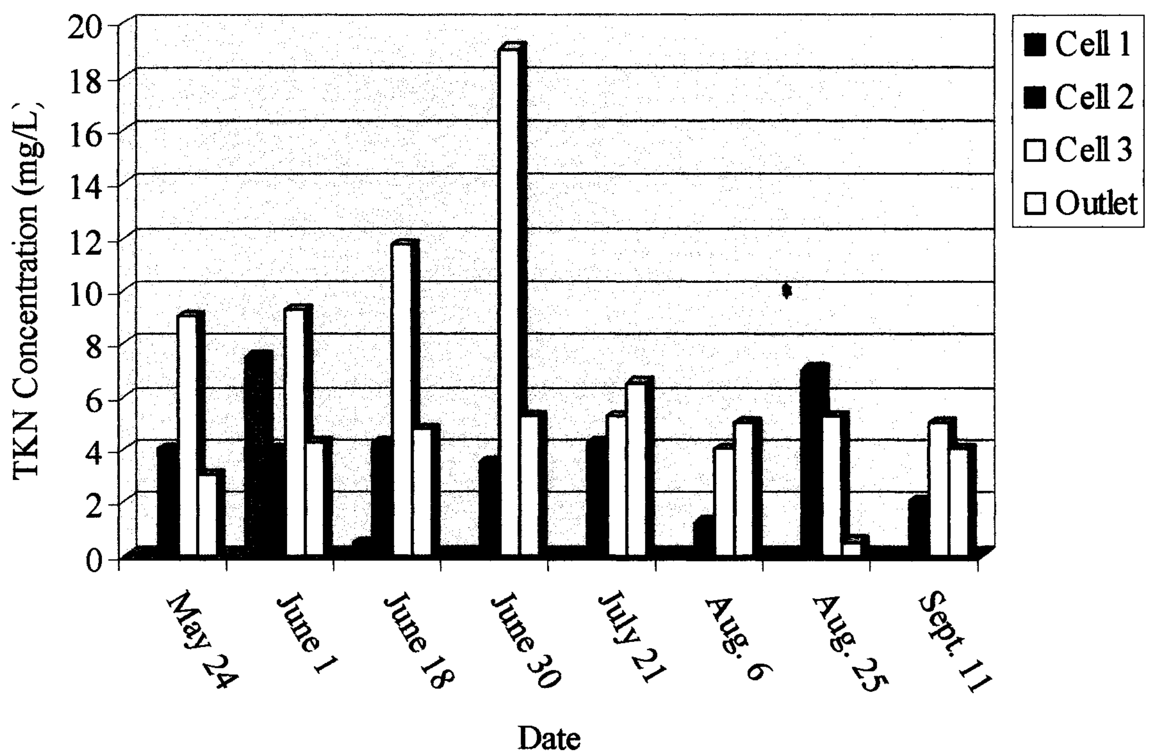


Figure 5.24. Total Kjeldahl Nitrogen Concentration in the Water of the Treatment Wetland, 2003.

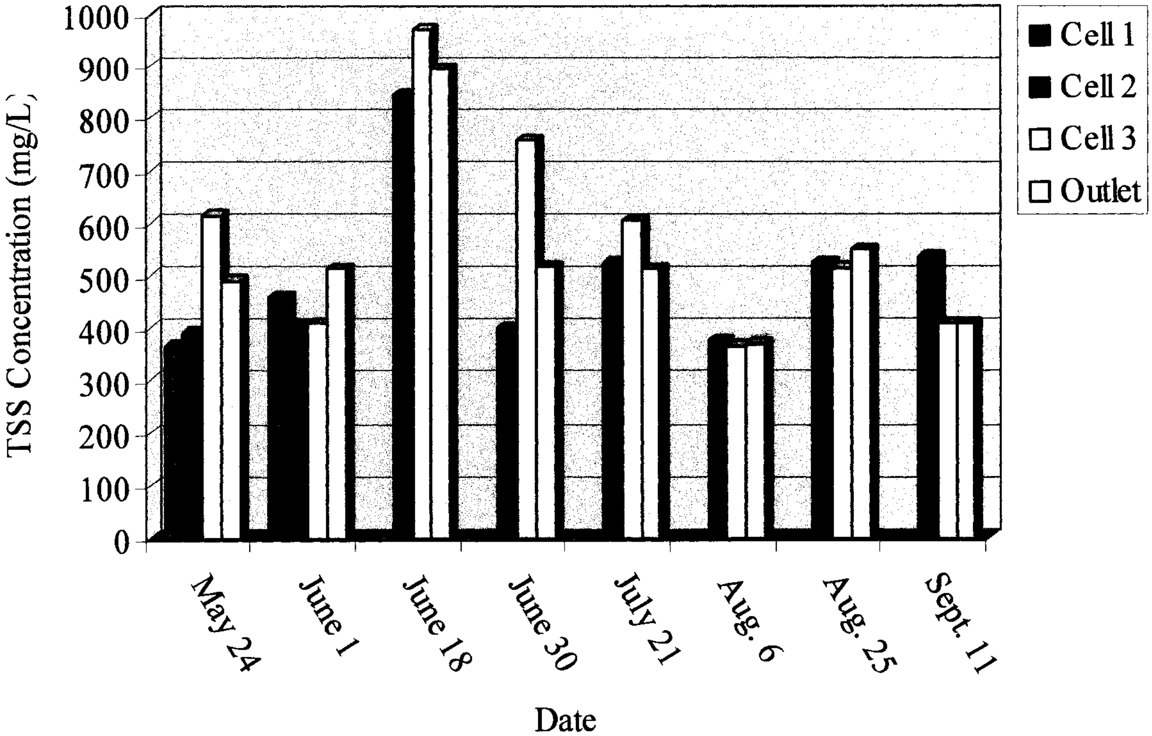


Figure 5.25. Total Suspended Solids Concentration in the Water of the Treatment Wetland, 2003.

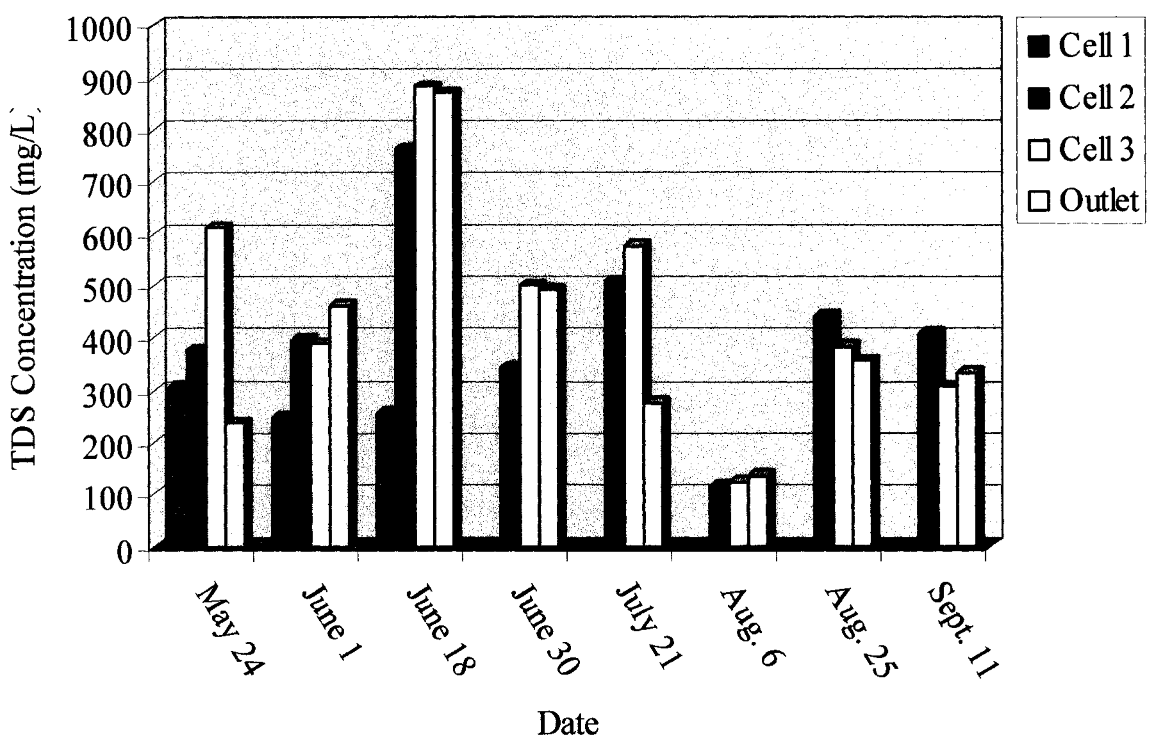


Figure 5.26. Total Dissolved Solids Concentration in the Water of the Treatment Wetland, 2003.

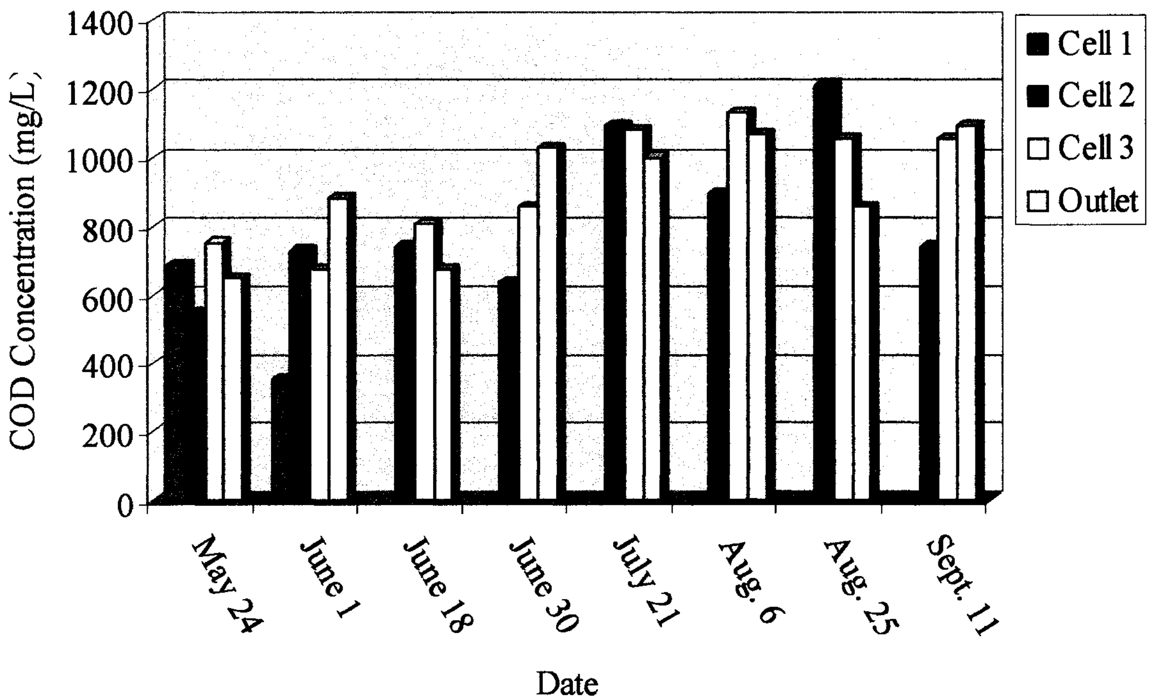


Figure 5.27. Chemical Oxygen Demand Concentration in the Water of the Treatment Wetland, 2003.

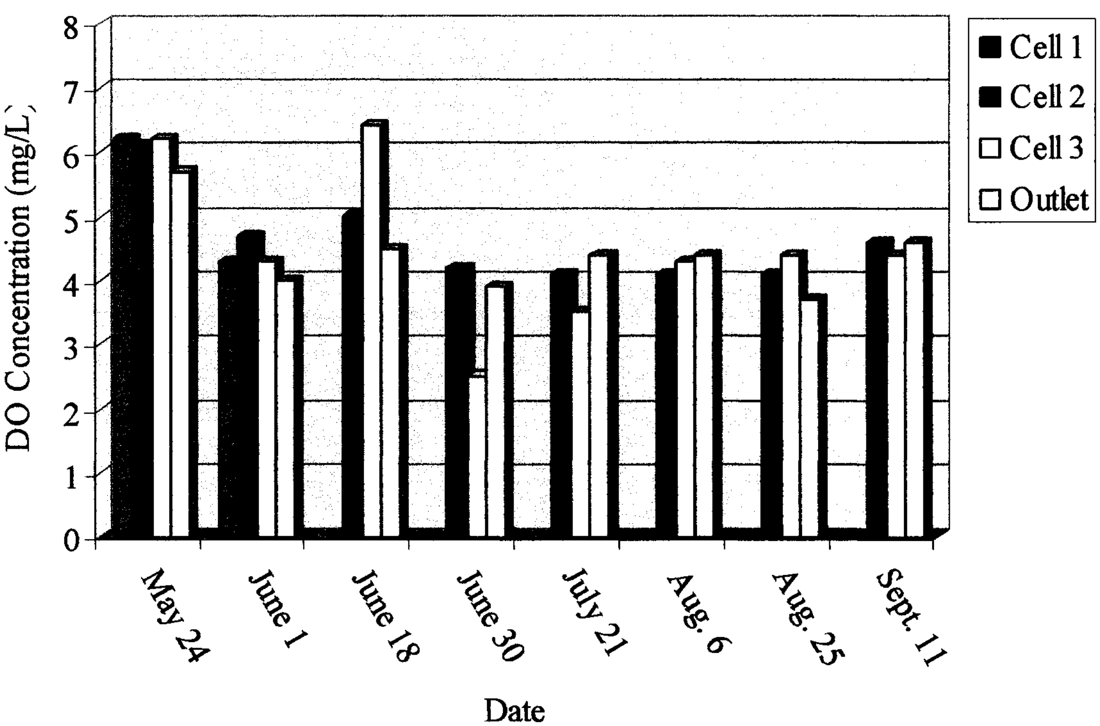


Figure 5.28. Dissolved Oxygen Concentration in the Water of the Treatment Wetland, 2003.

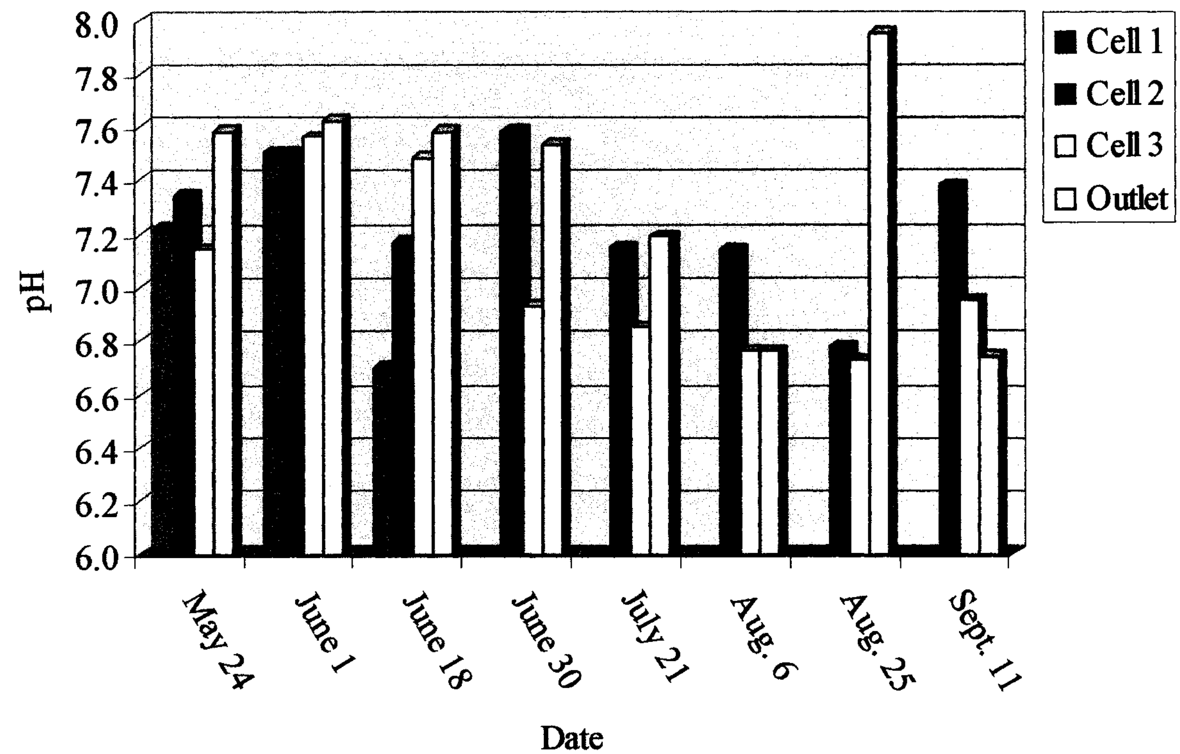


Figure 5.29. pH Concentration in the Water of the Treatment Wetland, 2003.

Table 5.19. Results of the Plant Tissue Analysis.

PLANT SPECIES	PLANT PART	REFERENCE SITE		TREATMENT SITE	
		Fe (mg/kg)	Mn (mg/kg)	Fe (mg/kg)	Mn (mg/kg)
Woolgrass (*Scirpus cyperinus*)	Root	3124	72	9620	90
	Stem	73	103	260	289
	Leaves	252	191	221	148
	Flower	415	351	105	56
	Total	3864	717	10206	583
Soft rush (*Juncus effusus*)	Root	6432	97	4052	217
	Stem	130	313	160	171
	Leaves	No leaves	No leaves	No leaves	No leaves
	Flower	596	471	1094	412
	Total	7875	881	5306	800
Pickerelweed *(Pontederia cordata)*	Root	4532	129	12868	269
	Stem	137	83	85	52
	Leaves	418	432	104	555
	Flower	184	212	122	198
	Total	5271	856	13179	1074
Fowl mannagrass *(Glyceria striata)*	Root	7524	270	6172	86
	Stem	35	129	116	309
	Leaves	246	499	1002	1480
	Flower	471	190	779	386
	Total	8276	1088	8069	2261
Yellow-green sedge *(Carex lurida)*	Root	2338	71	10519	200
	Stem	122	210	302	103
	Leaves	1141	371	809	167
	Flower	1564	265	229	392
	Total	5165	917	11859	862
Fringed sedge (*Carex crinita*).	Root	1827	427	3474	52
	Stem	176	43	129	44
	Leaves	780	150	3936	130
	Flower	466	84	433	230
	Total	3249	704	7972	456
Soft stem bulrush (*Scirpus validus*)	Root	3062	140	13903	156
	Stem	18	88	813	268
	Leaves	No leaves	No leaves	No leaves	No leaves
	Flower	6695	419	4222	778
	Total	9775	647	18938	1202
Tweedy's rush (*Juncus brevicaudatus*)	Root	4572	84	17302	447
	Stem	169	308	461	225
	Leaves	No leaves	No leaves	No leaves	No leaves
	Flower	138	374	1714	443
	Total	4879	766	19477	1115
Broad-leaved cattail *(Typha latifolia)*	Root	119	34	14171	127
	Stem	13	44	5788	277
	Leaves	253	783	173	730
	Flower	24	105	1466	467
	Total	409	966	21598	1601
Reed canary grass (*Phalaris arundinacea*	Root	1967	49	11834	74
	Stem	178	49	245	129
	Leaves	572	283	775	488
	Flower	739	79	619	249
	Total	3456	460	13473	940

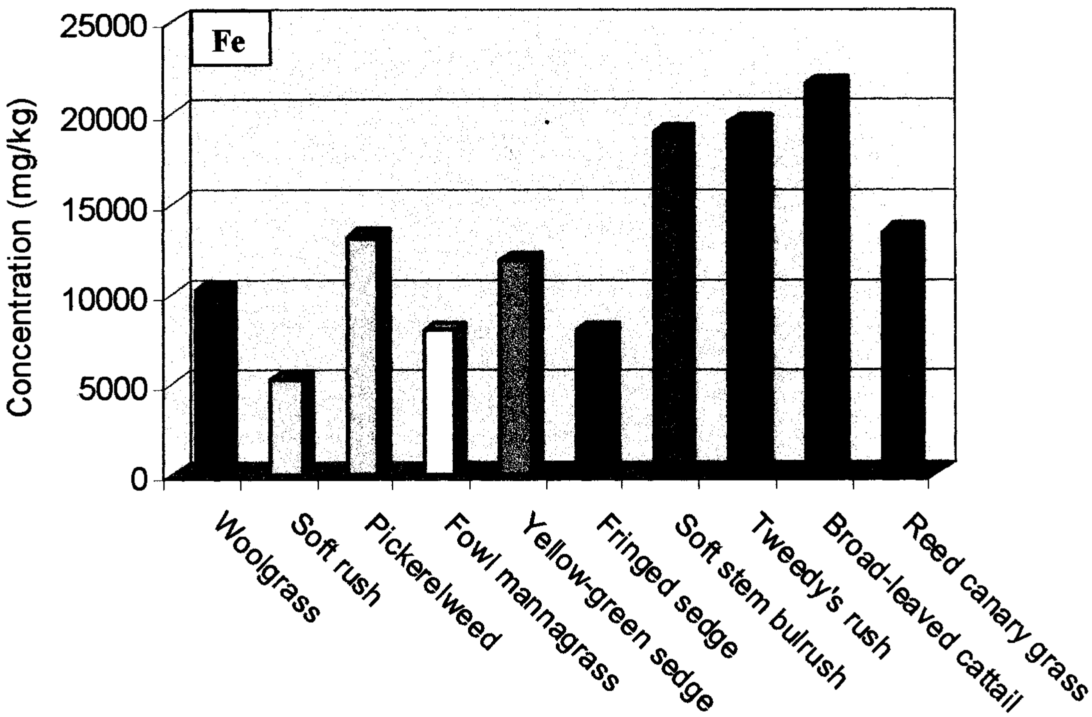


Figure 5.30. Total Iron Concentrations in the Plants Obtained from the Treatment Wetland.

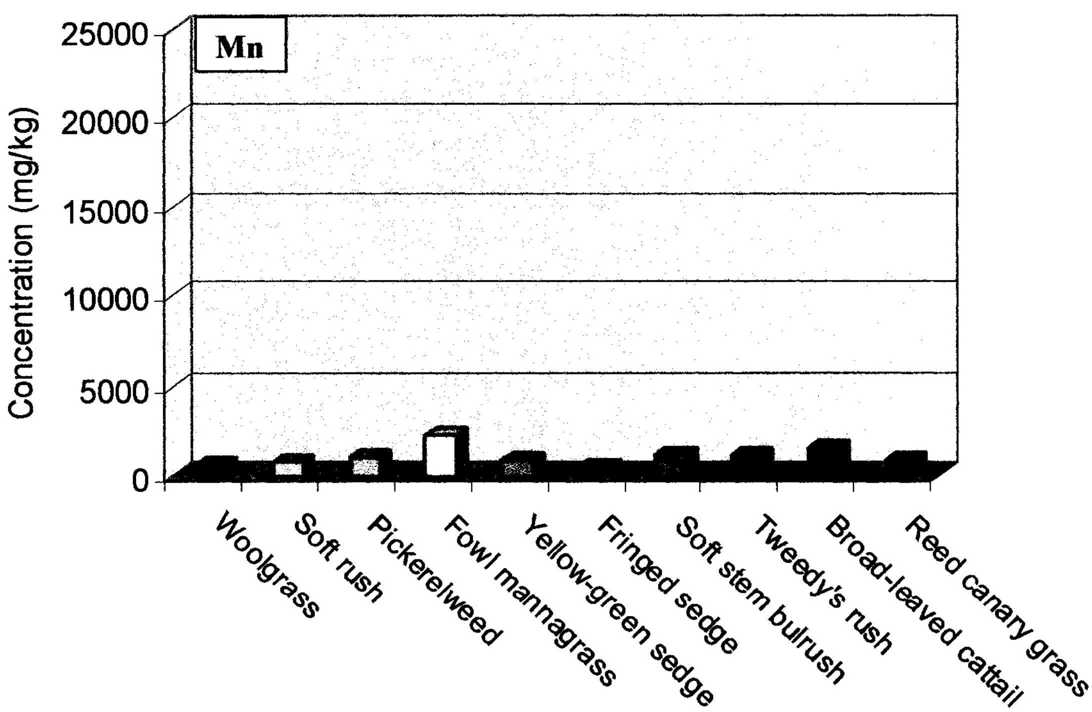
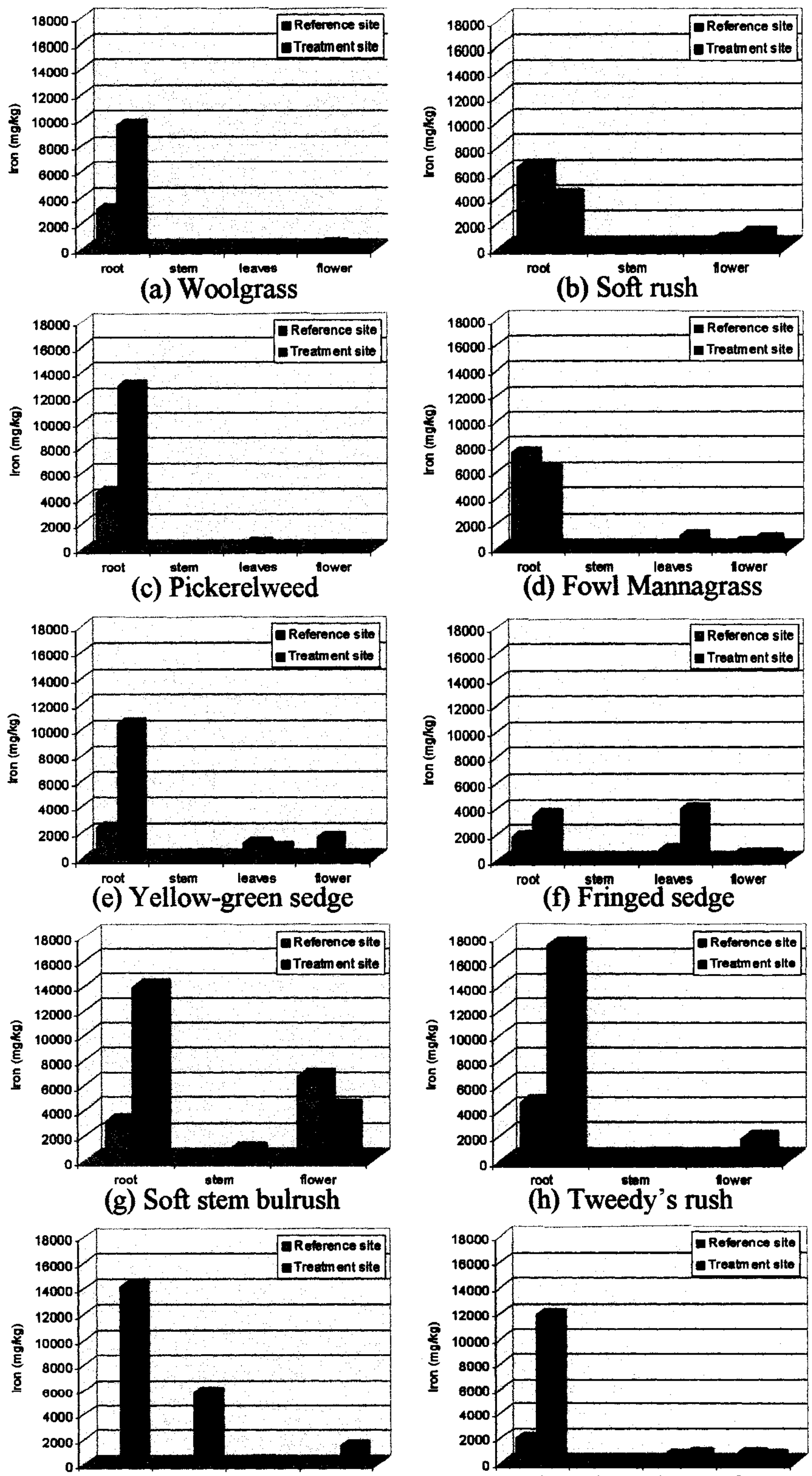


Figure 5.31. Total Manganese Concentrations in the Plants Obtained from the Treatment Wetland.



(a) Woolgrass (b) Soft rush
(c) Pickerelweed (d) Fowl Mannagrass
(e) Yellow-green sedge (f) Fringed sedge
(g) Soft stem bulrush (h) Tweedy's rush
(i) Broad-leaved cattail (j) Reed canary grass

Figure 5.32. Iron Distribution in the Tissues of the Various Plants Obtained from the Treatment and Reference Wetland Sites.

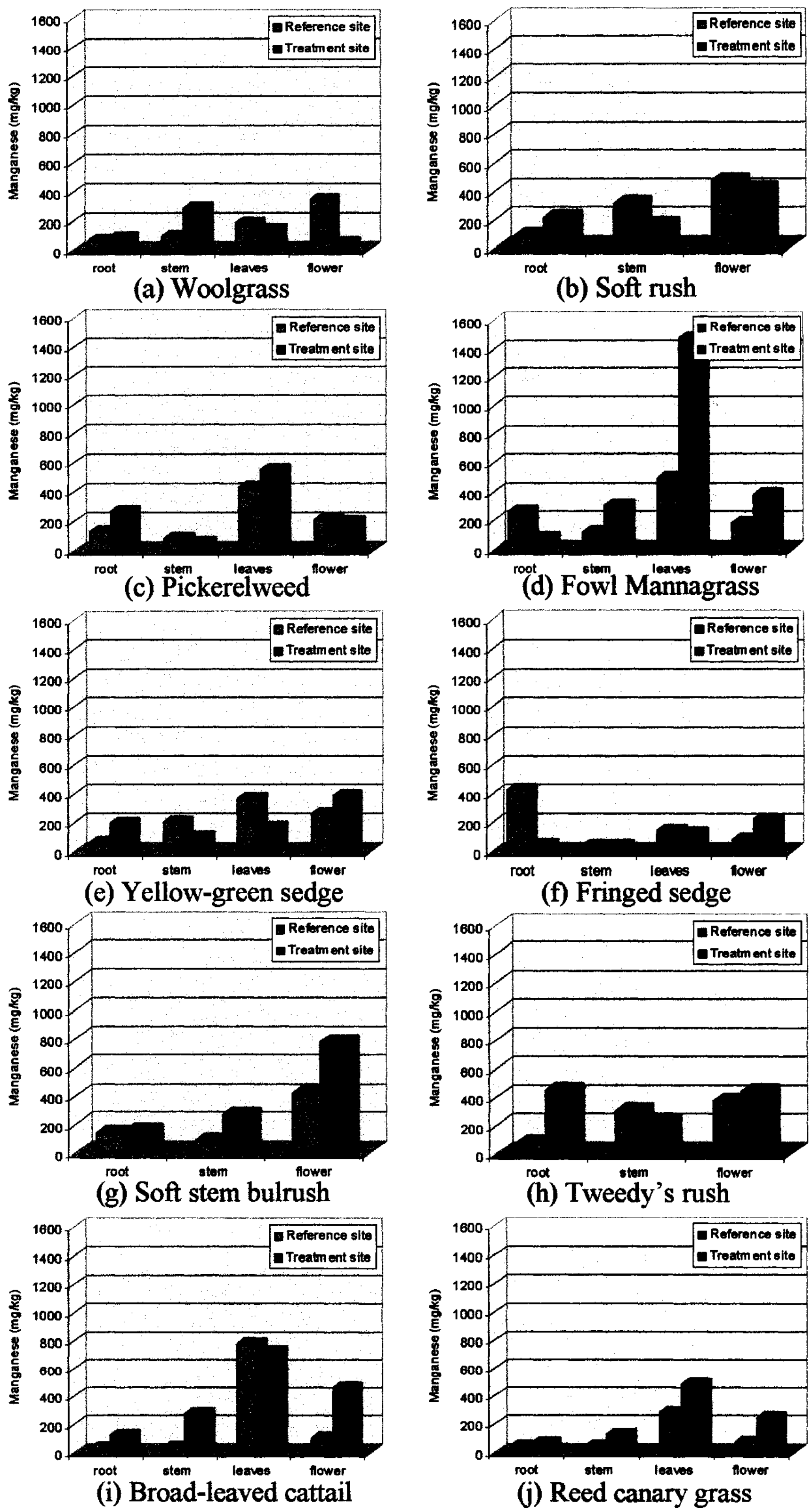


Figure 5.33. Manganese Distribution in the Tissues of the Various Plants Obtained from the Treatment and Reference Wetland Sites.

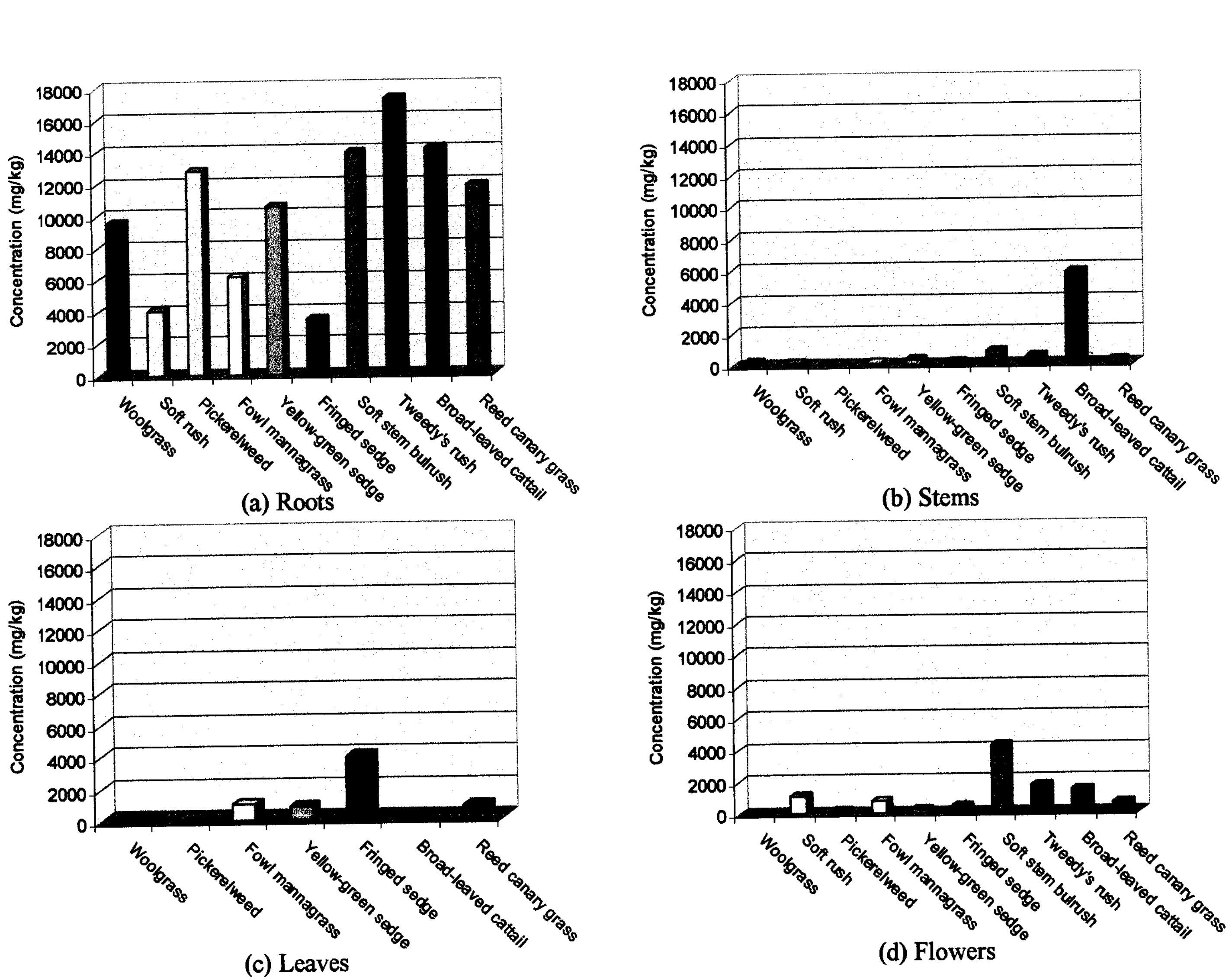


Figure 5.34. Iron Concentration (mg/kg) in the Roots, Stems, Leaves and Flowers of Various Plants Obtained from the Treatment Wetland.

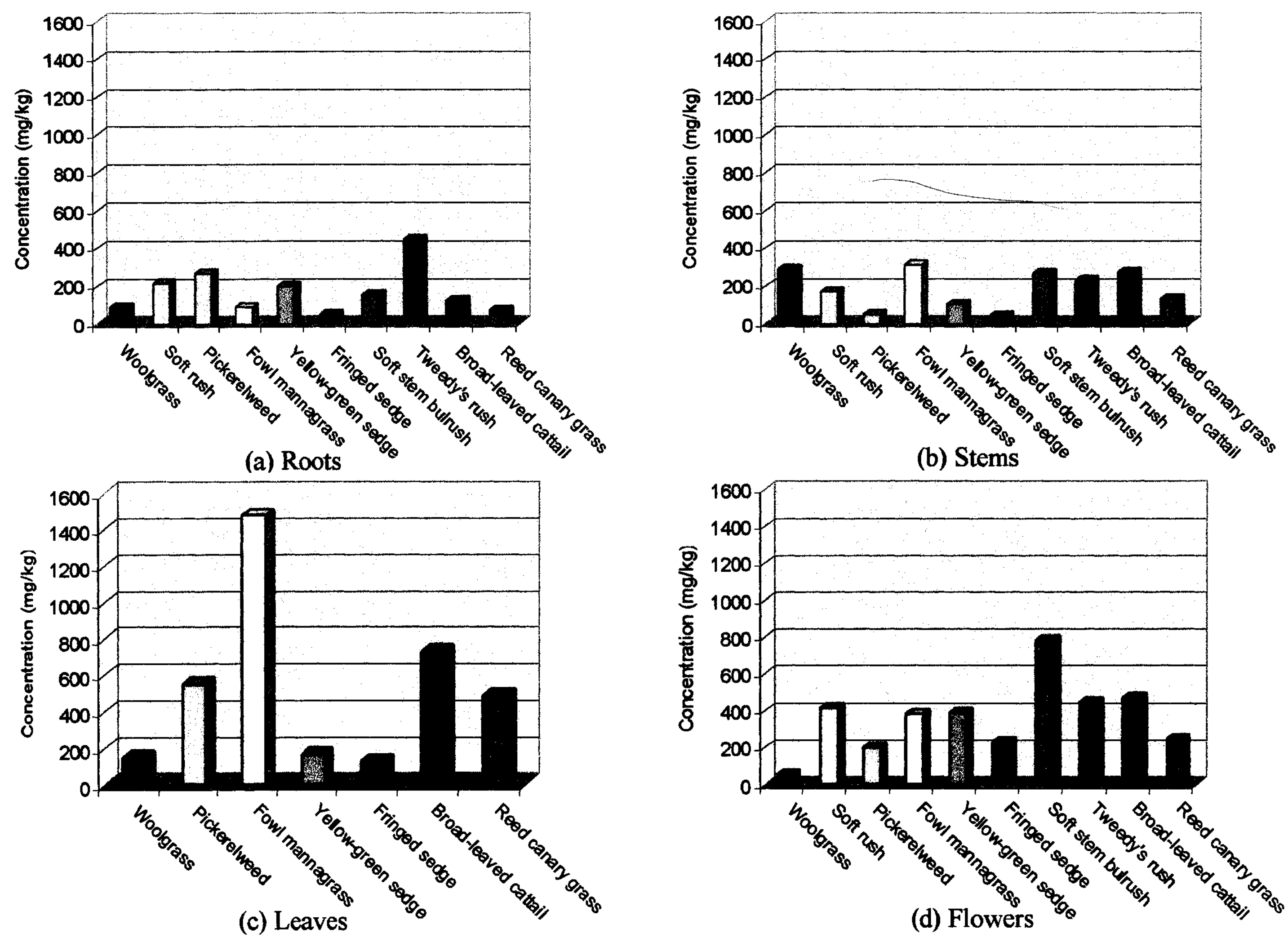


Figure 5.35. Manganese Concentration (mg/kg) in the Roots, Stems, Leaves and Flowers of the Various Plants Obtained from the Treatment Wetland.

5.5.3. Aquatic Macroinvertebrate Assessment

The results of the aquatic macroinvertebrate water quality assessment are presented in Figure 5.20. Raw calculations are presented in Table F.1 of Appendix F.

Table 5.20. Results of Selected Aquatic Macroinvertebrate Metrics of Water Quality.

METRIC	REFERENCE SITE	CELL 1	OUTLET
ASPT via BMWP biotic	5.94	4.86	4.29
ETSD biotic index	45.45%	7.69%	0%
Mayfly abundance	13.64%	0%	0%
Trophic structure			
Scrapers:	2%	0%	0%
Shredders:	2%	2%	0%
Collectors:	77%	96%	97%
Predators:	13%	2%	3%

ASPT = Average Score per Taxon
BMWP = Biological Monitoring Working Party
ETSD = Ephemeroptera (E) (mayflies), Trichoptera (T) (caddisflies); Sphaeriidae (S) (fingernail clams), and Odonata (dragonflies/damselflies) (D).

6. DISCUSSION

6.1. Selection of Vegetation

6.1.1. Wetland Community Model

The initial objective of this study was to select the appropriate native vegetation for establishment in the Burnside treatment wetland. One of the best ways to ensure effective naturalization is to model the vegetation populations to be established in the treatment site after a local, natural wetland of a similar type (Hoag, 2000). According to Daigle and Havinga (1996), community modeling involves the surveying of plant communities inhabiting the natural wetland including vegetation composition, structure and abundance. The community surveyed then acts as a template for the selection of the appropriate species for planting the treatment wetland. Abundance proportions are then altered in final planting strategies to best suit project goals.

In the case of the Burnside treatment wetland the appropriate natural wetland type to model was a freshwater marsh ecosystem. The community selected to act as the vegetation template for the Burnside treatment wetland was the disturbed freshwater water marsh located in Wright's Brook (200m downstream). Only native plants which would not pose a threat of biological pollution were considered for placement in the wetland. In total, 68 native species were identified in the Wright's brook model wetland. The native emergent wetland species dominating the wetter areas of the model site included fowl mannagrass *(Glyceria striata)*, broad-leaved cattail (*Typha latifolia*), soft rush (*Juncus effusus*) and woolgrass (*Scirpus cyperinus*). These four plants were dispersed in clusters (especially the cattail), and were not heterogeneous throughout the site. The native vascular plants dominating the drier, upland portions of the model site were wild sarsaparilla *(Aralia nudicaulis)*, bunchberry *(Cornus canadensis)*, bracken fern *(Pteridium aquilinum)*, goldenrod *(Solidago* spp.*)* and smooth cordgrass *(Spartina alterniflora)*. Similar to the wetland emergent plants, goldenrod and smooth cordgrass were dispersed in clusters. The remaining species were spread more evenly throughout the site. The upland shrub species dominating the model site were sheep laurel *(Kalmia angustifolia)*, rhodora *(Rhododendron canadense)*, meadowsweet *(Spiraea alba var. latifolia)* and blueberry *(Vaccinium angustifolium)*. The meadowsweet was notably the most dominant shrub in the site and was often observed growing in the wet areas of the marsh, sometimes in completely submerged conditions. The remaining species were mostly present homogenously throughout the drier upland portions of the site. The final plant list for the Burnside treatment wetland was developed using the model plant list as a guide. However, the abundance proportions and structures observed in the model site were altered using selection criteria designed to screen the species in the model lists for characteristics which were most conducive to the specific goals of the site.

6.1.2. Burnside Treatment Wetland Plant Species Selection Criteria

Once the model community was sufficiently surveyed and the plant list was complete, the next step was to create the plant list to best suit project goals for the treatment site based on the model list. The idea behind site modelling is not to imitate the model site exactly, but rather to use the community model to guide the appropriate plant selection for the site (Daigle and Havinga, 1996). For example, if research leads to the conclusion that the wetland emergent species dominating the model site have little phytoremediation potential, while wastewater remediation is the primary goal of the treatment site, these species may be excluded from the treatment wetland altogether. Likewise, species which may not be as abundant in the model site may be selected to dominate certain areas of the treatment wetland such as the edges as a result of their inherent soil stabilization capabilities. The same applies for additional project goals such as aesthetics and habitat provision.

For the Burnside treatment wetland, plant selection, abundance and distribution in the treatment wetland site were based on their phytoremediation potential, sedimentation and erosion control abilities, their habitat facilitation abilities, their public deterrent potential, as well as their rates of plant establishment, tolerances and requirements. However, given that the dominant goal of the Burnside treatment wetland is to effectively cleanse the identified contaminants (iron, manganese and ammonia) present in the landfill leachate, the plants chosen to dominate the wetland cells (especially the interior berms) were those species identified from the model lists as being capable of treating these contaminants.

6.1.2.1. Phytoremediation Potential. The wetland plants which are most often effective at facilitating contaminant removal are emergent aquatics with extensive roots systems, which included cattails (*Typha* spp.), reeds (*Phragmites* spp.), bulrushes (*Scirpus* spp.), and various sedges (*Carex* and *Juncus* spp.). These plants are ideal for remediative purposes as they all tend to: (a) shade waters keeping them cool and minimizing algal growth, (b) facilitate the settling of suspended particulate matter via physical obstruction and the slowing of water velocities, (c) stabilise bottom sediments via dense root systems, (d) diffuse oxygen into soil creating oxygen-rich zones around their roots known as the rhizosphere, in which aerobic microbes carry out biochemical processes and effectively cycle many contaminants, and (e) many of these species are reputable metal and nutrient extractors and transformers (Hammer, 1992; Boulton and Brock, 1999; and USEPA, 1999b).

Several cattail, bulrush and sedge species were identified in the model wetland site. However, only those species which were dominant, or present in high abundance (ranked a coefficient of 4 or higher) in the model site were considered for the phytoremediative-capability screening. This was simply because those species capable of dominating or thriving in the model community which was impacted by the leachate input, would naturally be more likely to thrive in the treatment wetland site. Dominance was also an issue because the model site was ultimately going to serve as the primary donor site for the treatment wetland's species, hence plant availability was an important selection consideration. Also, only those candidate species which supported large biomasses were considered for site domination as hyperaccumulators with small biomass are not very effective in large scale sites such as the Burnside treatment wetland (USEPA, 2000b). This resulted in the elimination of many *Carex* and *Cypernus* as candidate species to dominate the treatment wetland site, as most are relatively small plants.

Three emergent wetland plants from the model plant list met the above conditions: woolgrass (*Scirpus cyperinus*), soft rush (*Juncus effusus*) and broad-leaved cattail (*Typha latifolia*). The three identified plants all had inherent metal and nutrient hyperaccumulation abilities. However, due to their limited root mass, abundant litter, aggressive behaviour, and occasionally allelopathic nature, cattails were ultimately eliminated as candidate species to dominate the Burnside treatment wetland site in this phase of the project. The woolgrass and soft rush were consequently selected to dominate the site's berms and littoral areas.

6.1.2.2. Sedimentation and Erosion Control. Species with considerable root development which are particularly effective at sediment stabilization were selected from the model list. The woolgrass and soft rush species, which were selected to dominate the wetland berms and littoral areas as a result of their phytoremediative potentials, were also excellent candidates for sediment stabilization due to their dense root systems. Species selection for this category was applied not only to the interior and littoral zones of the cells but to the drier upland buffer areas of the site which desperately needed enhancement in order to reduce erosion and sedimentation (Figures 6.1 and 6.2). According to Davis (1995), the presence of buffer zones made up of trees, shrubs, and semi-aquatic plants around wetlands are known to be of considerable value in the maintenance of good environmental quality in wetland ecosystems (Castelle et al., 1994). This fringing vegetation forms an important structural component of wetland systems, minimizing invasion by weed species, stabilising and aerating soils and providing a filtering mechanism for pollutants, nutrients and sediment loads in surface runoff (Campbell and Ogden, 1999; and Hammer, 1992). Mostly woody species with deeply penetrating root systems such as meadowsweet (*Spiraea alba var.latifolia)* and speckled alder (*Alnus viridis)* were considered for the buffer areas. Although as a result of their deeply penetrating roots, pinkweed (*Polygonum pensylvanicum)* was initially identified as an ideal candidate for erosion control in the buffer areas. Later characterization lead to the discovery that the plant is an annual, and was therefore disqualified.

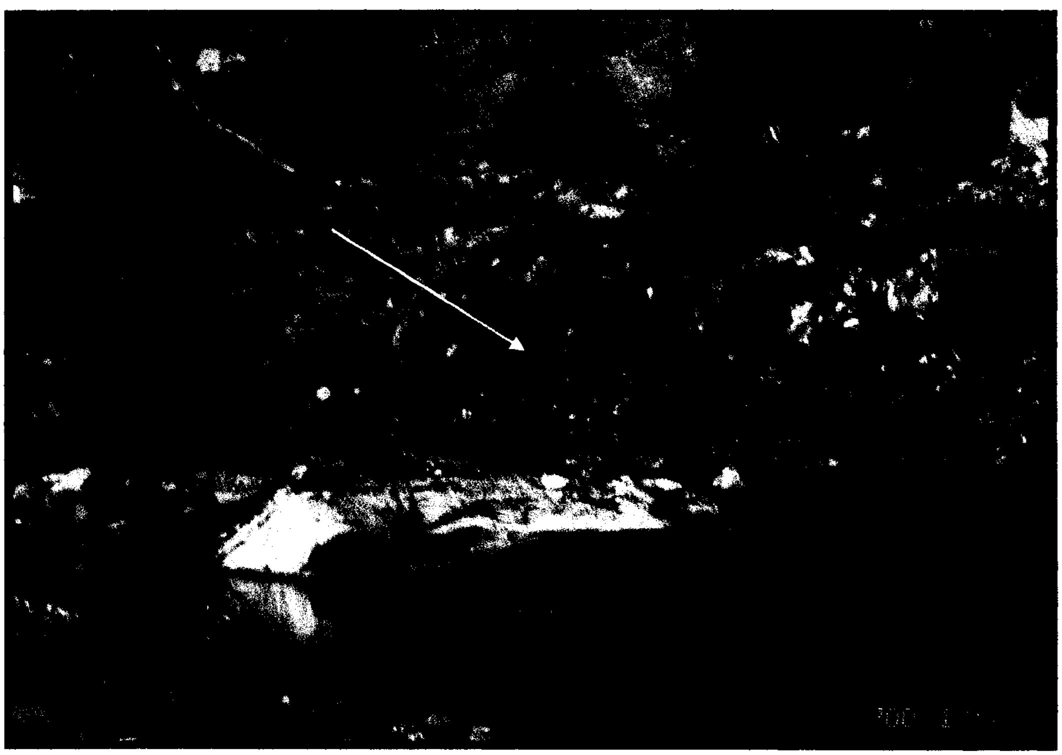
6.1.2.3. <u>Habitat Function</u>. One of the secondary goals of the Burnside treatment wetland project was to facilitate the creation of habitat. The chemical analysis of the water samples indicated that iron is the contaminant available in the largest concentrations in the leachate received by the treatment system. However, the majority of the metal is present in its oxidized form, which is not readily bioavailable (Mitsch and Gosselink, 2000). Because of the non-persistent and non-bioaccumulative nature of this contaminant, habitat facilitation was deemed an important project objective.

It was intended that the aquatic macrophytes, shrubs and trees which make up the vegetative community of the site, would provide both a source of food and a range of habitats for aquatic and terrestrial fauna including: amphibians, birds, and mammals. Rushes, bulrushes and cattails provide denning and nesting sites, as well as shelter in harsh weather. Dense, woody shrubs and trees such as alders and spruces established in buffer zones reduce faunal disturbance from noise, movement, light and other aspects of the surrounding urban environment. Fruiting shrubs and trees such as blueberry and wild cherry are used by birds and mammals as food. Foliage supports insect populations that aid in the cycling of contaminants as well as provide a valuable link to the food chain (Hammer, 1992).

6.1.2.4. <u>Public Deterrent Potential</u>. Also worth noting in the screening exercise were those species which possessed public deterrent attributes such as sharp thorns. Their strategic placement was intended to help restrict public access to sensitive areas such as berms and nesting habitats.

6.1.2.5. <u>Rate of Plant Establishment, Tolerances and Requirements</u>. Understanding of the spreading rates of species being placed in the site was important in order to ensure that rapidly

(a) Buffer Area Needing Vegetation in Cell 1.



(b) Cave-in of Eastern Slope of Cell A after Storm Event.

Figure 6.1. Areas that Needed Erosion Control.

spreading species such as *Typha* do not take advantage and out-compete other slower spreading species placed in the site such as *Carex* or *Scirpus* species. Care was taken to ensure slow growers were given equal opportunity to establish themselves in the treatment site as suggested by Davis (1995). Spreading rates as indicated by the planting distance required for uniform cover to be achieved in 1 year (or UC1), for the species identified as potential candidates for the Burnside treatment wetland site were researched. Unfortunately, this information is limited for non-horticultural plants and was unavailable for several of the candidate species.

Other potentially valuable species characteristics which were researched for the potential candidates included tolerances to permanent inundation, shade, high pH, and salinity, as well as any requirements necessary for healthy growth such as full sun, protection from windthrow, and well drained soils. The knowledge of these tolerances and requirements is crucial to the success of the planting strategy (Daigle and Havinga, 1996). Without proper recognition, understanding and accommodation of these characteristics, planting success would not be maximized.

In general, of the emergent aquatic species, sedges (*Carex* spp.) and rushes (*Juncus* spp.) appeared to be the slowest spreaders, and cattails (*Typha* spp.) appeared to be the fastest (Thunhorst, 1993). Cattails and woolgrass (*Scirpus cyperinus*) support large biomass, while sedge species tend to support small biomass. However, sedges could not be dismissed as they support high habitat value and are often prone to hyperaccumulate metals and nutrients (Thunhorst, 1993; and Campbell and Ogden, 1999). The aquatic emergent species chosen to dominate the interior berms of the treatment wetland site were woolgrass (*Scirpus cyperinus*) and soft rush (*Juncus effusus*), hence, detailed knowledge of the growth habits, tolerances and requirements of these species were particularly emphasized. Soft rush is a very slow spreader (UC1=0.5 ft), while woolgrass is a moderate spreader at UC1=1 ft (Thunhorst, 1993). Consequently, in order to allow soft rush a better chance at competing with the established woolgrass, soft rush individuals were placed at tighter intervals and more individuals were used. Notably, both species are tolerant of permanent inundation, support prolific roots, produce abundant seed, and have high habitat value. However, woolgrass appeared to be the more hardy of the two as it is shade, drought and flood tolerant, as well as tolerant of waters and soils with varying pHs (Zinck, 1998, Thunhorst, 1993, and Larson, 1993).

According to Thunhorst (1993), the handling of the roots of the aquatic emergent species blue flag (*Iris versicolor*), which although is a beautiful ornamental species, tolerant of high nutrient levels and fluctuating water levels, could cause severe dermatitis. Hence, with the exception of a few carefully handled individuals, this species was eliminated as a candidate species for transplant. Also, according to Thunhorst (1993), reed meadow grass *(Glyceria grandis)* does not compete well with other species. As a result, it too was eliminated as a major transplant candidate for the site. In addition, the species pinkweed (*Polygonum pensylvanicum*), a good soil stabilizer with high habitat value was eliminated as a candidate species for transplant as the plant is in fact an annual (Thunhorst, 1993).

Information on spreading rates, requirements and tolerances for the terrestrial vascular plants was rather limited. However, in seeing that nearly all of these plants were selected as candidates to serve the primary purpose of increasing diversity of the terrestrial areas of the site, the characteristics were not as essential to the overall establishment success of the site. With the exception of the ferns, all of the terrestrial vascular plants are flowering, and most are small in biomass. In addition, most reproduce by rhizome as well as seed, and all are perennial (Thunhorst, 1993; Runesson, 2002; Rook, 2002; and Zinck, 1998).

According to Thunhorst (1993), Runesson (2002), and Rook (2002), the fastest spreading candidate shrubs for placement in the treatment wetland are meadowsweet (*Spiraea alba var. latifolia)* and speckled alder *(Alnus viridis)*. Consequently, these species were initially selected to dominate the buffer zones in order to quickly and effectively facilitate bank stabilization. However, the slower spreaders, which include the larger fruiting shrubs such as red chokeberry (*Aronia arbutifolia*) and witherod (*Viburnum cassinoides*), were also included in the buffer as these species provide high habitat value as a result of their abundant fruit. In addition, stouter, less prominent shrubs such as sweet fern (*Comptonia peregrine*), Labrador tea (*Ledum* groenlandicum), rhodora (*Rhododendron canadense*), sheep laurel (*Kalmia angustifolia*) and blueberry (*Vaccinium angustifolium*) were also included in the buffer areas as these species are notoriously tolerant of harsh soil conditions.

6.2. Vegetation Sources and Establishment Methodology

There are four common planting techniques used to establish vegetation in constructed wetland sites. These are: direct seeding, transplanting whole plants, planting commercial stocks in the form of tubers, rhizomes, rootstocks, planting containers and bareroot plants, and planting clonal cuttings. Although potentially labour intensive, compared to other planting techniques, transplantating of mature plants from local donor sites was selected as it is by far the most ideal site establishment methodology for several reasons: (a) it is very cost effective, (b) plants are already genetically adapted to local environmental conditions, increasing their survival success and habitat significance, (c) transplantation can generally be carried out three season of the year, (d) plants establish themselves readily and have the highest survival success of all the plant establishment techniques, (e) by planting adult plants, site establishment occurs much more quickly than with seeding or other means, (f) transplanted species contain soil around their roots holding dormant seeds of plants which will naturally add to the diversity and total vegetative cover of the area, and (g) transplant soils also contain microbes, invertebrates, and eggs, which accelerate the establishment of microbial, insect and herptile communities in the site (Hammer, 1992; EC, 2000b; and Daigle and Havinga, 1996).

For their inherent phytoremediation capabilities, sediment stabilisation abilities, large aboveground and belowground biomass, and abilities to facilitate microbial growth and effectively aerate sediments, the species woolgrass (*Scirpus cyperinus*) and soft rush (*Juncus effusus*) were chosen to dominate the interior berms and littoral edges of the treatment wetland cells. In seeing that the soft rush had a slower rate of spread than the woolgrass, the soft rush was placed at tighter intervals than the woolgrass. The exception to this was the first interior berm, which is naturally the most vulnerable to damage caused by intense water flows during storm events. Consequently, woolgrass was planted at tight intervals within this berm because: (a) the woolgrass would spread faster, working to stabilize the berm more quickly than would the soft rush (Thunhorst, 1993), (b) woolgrass visibly had denser and more prolific roots than did the soft rush, and was therefore believed to have greater potential to stabilize soils, and (c) the woolgrass appeared to take transplanting more easily than the soft rush, which were easily damaged and quick to brown when transplanted. The model wetland site was the exclusive source of these two plants. Although their large, dense, heavy root systems often made them a challenge to extract and transport, harvesting and subsequent planting of the woolgrass and soft rush proved slow but straightforward and unproblematic. The planting of these species was complimented by additional aquatic species from the final planting list to increase diversity and provide other functions characteristic to those species. Woody species were purposely excluded from the interior berms as the roots of shrubs and trees can create channels and subsequent leakages through berms. Although pickerelweed and cowlily were scarce in the model site and consequently assigned coefficients of 1 in the model plant list (which would have them eliminated as transplant candidates), the donor site located near Enchanted Lake supported abundant numbers of them, and in fact was dominated by these species. Given their potential benefits (identified during the screening process), and their high abundance, these species were re-introduced as transplant candidates. Cowlily was placed in the deep water zones of Cell A and B mostly for habitat and aesthetic purposes, as the calm and non-contaminated nature of the waters in this cell was found to be well-suited to this species. The pickerelweed was established in the littoral zones of these cells for the same reasons. Ideally, pickerelweed would have also been established in the remaining cells of the wetland as well. However, access to the donor location of the pickerelweed and cowlily was relatively remote and extremely difficult to access. The donor location was approximately 600 metres north of site through heavy brush and bog. Consequently, extraction only took place once. A variety of woody shrubs and facultative wetland plants from the model and successional brush donor sites were established in the barrier berm located between Cell 3 and Cells A, B and C. However, the successional brush site was nearly dismissed altogether as a primary donor source in the early stages of the site establishment phase as the plants proved difficult to successfully extract from the sites dry, hard soils. The buffer areas were dominated by meadowsweet (*Spiraea alba var. latifolia)* plantings due in part to their availability and ease of removal from the model site. Additional species from the planning list were planted in the buffer and wetland transitional zones in order to increase diversity and provide the functions characteristic to those additional species.

6.3. Evaluation of Vegetation Establishment Success

According to Daigle and Havinga (1996), in all site establishment projects, it is inevitable that plants will die and that replanting will be necessary. However, it is the degree to which re-establishment will be necessary that remains unknown. Despite some observed mortalities, overall, the site establishment strategy chosen effectively yielded a successfully established site. This success can be observed by examining the high productivity of the established berms and cells in photographs taken in late summer of 2003 (Figures 6.2 to 6.7). Those plants which did see mortality in the site are pickerelweed (*Pontederia cordata*), meadowsweet (*Spiraea alba var. latifolia*), woolgrass (*Scirpus cyperinus*), sheep laurel (*Kalmia angustifolia*), soft rush (*Juncus effusus*), rose species (*Rosa* spp.), trembling aspen (*Populus tremuloides*), grey birch (*Betula populifolia*) and speckled alder (*Alnus viridis*).

6.3.1. Pickerelweed (Pontederia cordata)

It was believed that pickerelweed would make an excellent candidate for placement in the

Figure 6.2. Vegetation in Berm 1, Summer 2003.

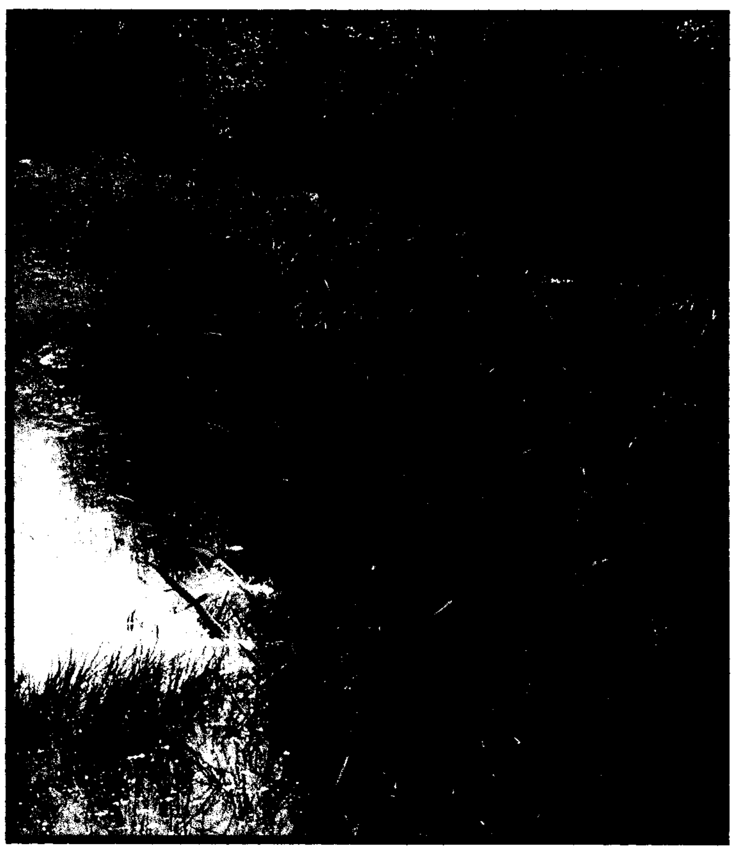


Figure 6.3. Vegetation in Berm 2, Summer 2003.

Figure 6.4. Vegetation in Berm 2, Summer 2003.

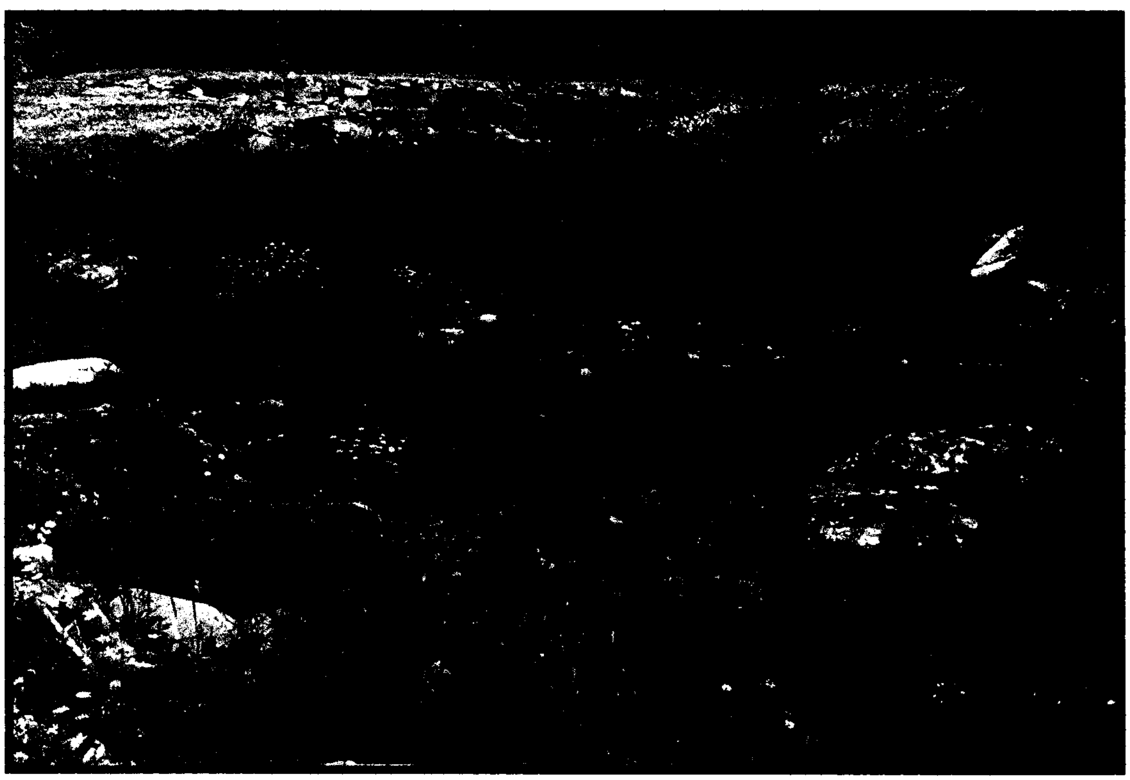


Figure 6.5. Vegetation in Cells 3, A and B, Summer 2003.

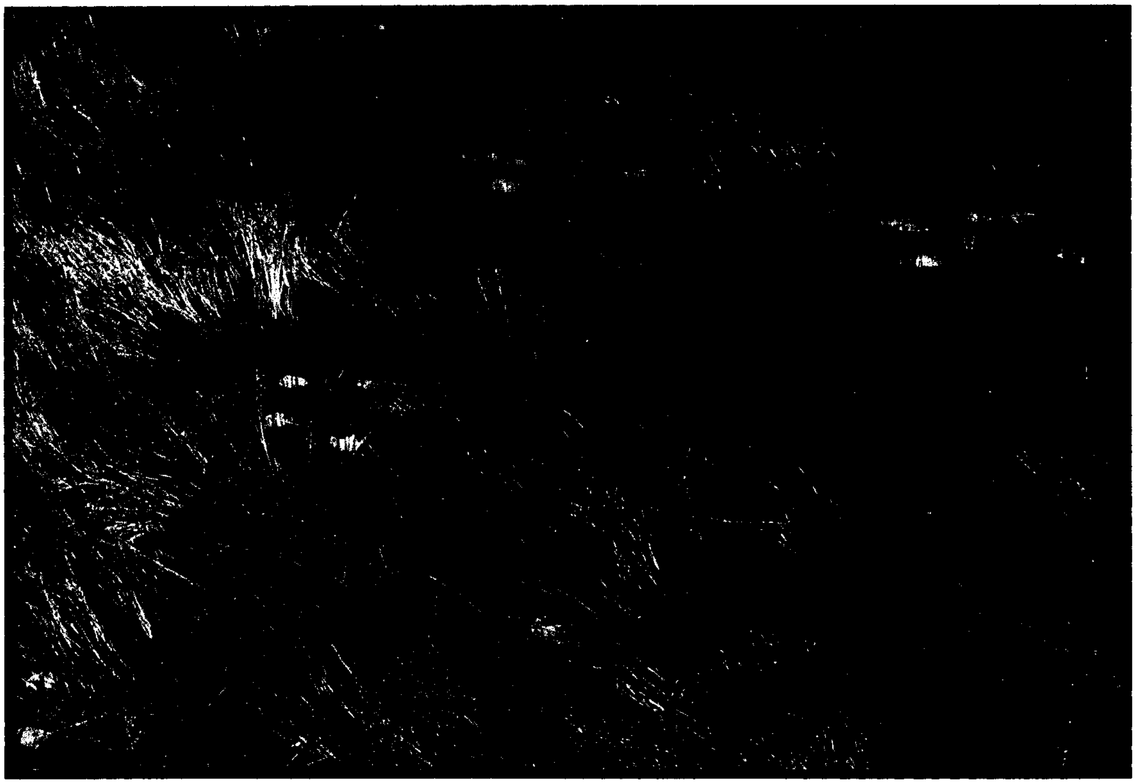


Figure 6.6. Vegetation in Cell A, Summer 2003.

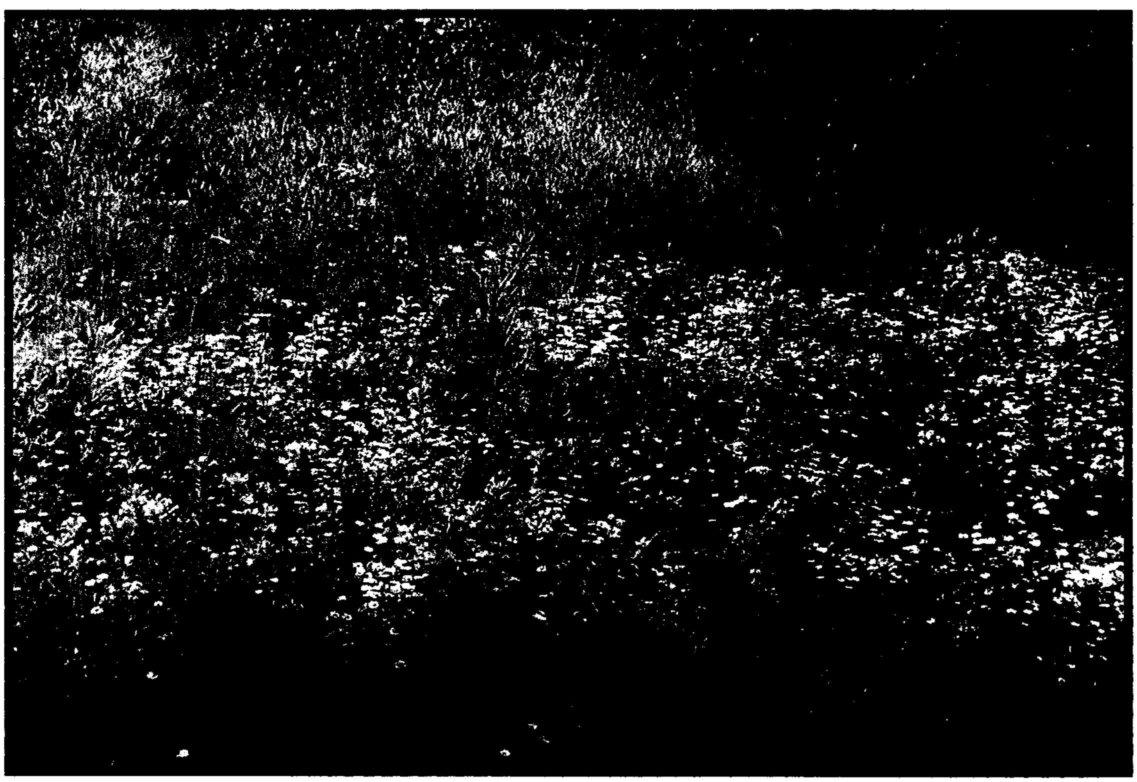


Figure 6.7. Vegetation on the Western Edge of Wetland, Summer 2003.

treatment wetland site (Thunhorst, 1993; and Zinck, 1998). It has a moderate spread of growth, grows in waters up to 1 ft deep, tolerates permanent inundation and shade, is an effective shoreline stabilizer, loses much water through transpiration, is potentially effective at phytoextraction, and is a close relative of water hyacinth (*Eichhornia crassipes*) which is a common wetland plant used in treatment wetlands. In addition, the species was observed growing quite prolifically in the littoral areas of Enchanted Lake, despite its polluted and turbid condition. However, this species suffered the highest mortality rate (about 67%) of all the transplanted species in the site. Clearly, its heightened mortality was not the result of toxicity exposure as this plant was only established in Cells A and B, which were not impacted by the contaminated leachate wastewaters. The high mortality rates may have occurred as a result of the fact that this plant was one of the latter species to be established in the site in late July, or perhaps it was overwhelmed by the prolific Tweedy's rush (Figure 6.8). However, many other species were transplanted late in the growing season and also had to contend with the Tweedy's rush with no observable effects. Literature review revealed no information documenting this or any other apparent reasoning for its failure to establish. Due consideration should be given to any future use of this species in the Burnside treatment wetland site.

6.3.2. Meadowsweet (Spiraea alba var. latifolia)

In comparison to the other species established in the site, the meadowsweet appeared to have suffered a high mortality rate (32 plants). However, this only represents a 12% of the individuals transplanted, as over 264 of these plants were established in the buffer areas of the wetland site. This species proved incredibly hardy, as most of the plants including those which had had little root mass, were planted in extremely harsh substrates, or those planted in mid-August were observed thriving with new shots the following growing season (Figure 6.9). In fact, the main cause of mortality in this species was the actual physical removal from the substrates as a result of washout. Zinck (1998) states that meadowsweet is a fast spreading, densely rooting shrub forming dense thickets. However, other than this, specific information on this particular variety of *Spiraea alba* was limited. According to Thunhorst (1993), *Spiraea alba* prefers full sun and requires moist soils to thrive, however, the meadowsweet in the wetland survived the extremely dry areas of the wetland as well as survived in permanently inundated conditions. Perhaps its preferences will take precedence in later growing seasons, and its mortality will increase. No literature could be found to support this. Hence, there is little doubt that in a few years, this species will form a full and effective buffer around the treatment wetland site.

Figure 6.8. Pickerelweed in Cell B Surrounded by Tweedy's Rush, Summer 2003.

Figure 6.9. New Growth Emerging from Suffering Meadowsweet Transplant, Spring 2003.

6.3.3. Woolgrass (Scirpus cyperinus)

In general, despite its robust appearance, the transplanted woolgrass species appeared to have been the most susceptible to the scouring action of the high waters in the wetland cells, suffering a 32% mortality rate. Despite forced drawdown via the valved culvert leading directly to Wright's brook, the first berm in particular was constantly being flooded and washed out in some areas, leading to the species demise. This was a particularly unanticipated outcome as woolgrass was specifically chosen to dominate the first berm as a result of its apparent ability to stabilize soils. However, it appears that this species is limited in its ability to do so. According to Thunhorst (1993), woolgrass is an excellent soil stabilizer, as it has prolific roots which hold soils in place. These roots were observably larger and more prolific than the soft rush, a species which unexpectedly did not see high mortality in the first berm. In addition, woolgrass has been the subject of treatment wetland studies conducted by Ye et al. (2001b), Campbell and Ogden (1999), Tousignant et al. (1999), Demchik and Garbutt (1999), Huang et al. (1999), and Mays and Edwards (2000), none of which identified any vulnerability of the species to wash out. However, those individuals which survived the flooding thrived (spreading horizontally as well and vertically) and grew to support the largest biomass of all species transplanted into the site (Figure 6.10). The woolgrass plants are currently the most distinguishable plants in the site.

6.3.4. Sheep laurel (Kalmia angustifolia)

The sheep laurel was established in berm 3 where the soil is harsh and dry. Several of the plants, which suffered a 32% mortality rate, were observably brittle and stressed over the course of the summer and fall of 2002 (Figure 6.11). According to Zinck (1998), sheep laurel is an extremely hardy species capable of thriving in harsh, dry soils, and can also be found in wet soils. This was observed in the donor sites from which the species was harvested. In most cases, the ground was so hard where this species was abundant that only the pick maddock could remove the plants from the soils. Conversely, this species was also observed in the swampy areas of the donor sites. The literature reviewed provided no insight into why this species suffered high transplant mortality in the site.

6.3.5. Soft rush (Juncus effusus)

According to Thunhorst (1993), Larson (1993), Ye et al. (2001b), Coleman et al. (2001), Treacy and Timpson (1999), and Mays and Edwards (2000), soft rush is a good candidate for treatment wetland planting as it has a moderate spread of growth, has deep roots, is tolerant of permanent inundation, is virtually an evergreen, and is an effective metal and nutrient remediator. However,

Figure 6.10. Woolgrass in Berm 2, Summer 2003.

Figure 6.11. Sheep Laurel in Berm 3, Fall 2002.

when the soft rush plants were initially extracted and transplanted into the treatment site, they were expected to suffer a relatively high mortality rate as their shoots were delicate and easily broken, and the plants were quick to brown (Figure 6.12). According the Zinck (1998), Thunhorst (1993), and Larson (1993) unlike woolgrass, soft rush is not shade or drought tolerant, nor is it tolerant of waters and soils with varying pHs. Therefore, it was concluded to be the less hardy of the two candidate species selected to dominate the treatment site. However, the woolgrass proved extremely hardy, and despite its slighter root mass, was particularly resistant to the harsh scouring waters of the cells which lead to much of this plant's demise. Overall, the plants only suffered a 2% mortality rate, with most individuals growing tall and robust. It is the soft rush above all other transplanted species which ultimately grew to dominate the treatment wetland site.

Figure 6.12. Transplanted Soft Rush, Summer 2002.

6.3.6. Rose species (Rosa spp.)

According to Thunhorst (1993) and Daigle and Havinga (1996), rose species are excellent candidates for wetland buffer planting as they are aesthetically pleasing, provide erosion control and effectively deter access to sensitive areas due to their thorns (Figure 6.13). It had been originally planned to plant much more of these species in the buffer areas of the wetland cells, specifically to help deter public visitors from sensitive areas of the site by way of their thorns. However, despite the use of gloves and other tools, these same thorns made it extremely difficult to extract these species from the donor site without injury. As a result, those that were extracted and transplanted were extracted and transplanted poorly. Consequently, only 16 rose plants were transplanted into the site, and of those transplanted, 4 died, suffering a 25% mortality rate.

6.3.7. Trembling aspen (Populus tremuloides), Grey birch (Betula populifolia) and Speckled alder (Alnus viridis)

According to Thunhorst (1993) and Zinck (1998), trembling aspen, grey birch and speckled alder are all excellent candidates for wetland buffer planting as all are hardy, fast growing, early successional species which are easily transplanted. However, many of the large toothed aspen, grey birch and speckled alder transplanted into the buffer areas of the wetland site in spring of 2002 were all dead within a few days (Figure 6.14). As a result, it was decided that these species were not ideal candidates for transplant, and their establishment in the site was ultimately terminated. Nearly all of the species transplanted were over 3 ft. in height. According to Daigle and Havinga (1996), the larger the individual transplanted, the more difficult it is for the plant to re-establish itself, as too much energy is required to sustain aboveground biomass. The transplantation of smaller individuals did occur. However, smaller individuals were more difficult to find, especially for the aspen and the alder. Given the phytoremediation-potential of the trembling aspen, it was later decided to attempt placing clonal cuttings of the species along the southern border of the site in the late fall of 2002. According to Daigle and Havinga (1996), clonal plant propagation from stem cuttings of *Populus* spp. can be an effective means of establishing the species in a site. Remarkably, all of the cuttings placed in this steep slope showed buds in the spring of 2003.

6.3.8. Tweedy's rush (Juncus brevicaudatus)

Nothing remarkable had been noted in the literature about this seemingly inconspicuous species. This small, slender plant was not abundant in any of the donor sites. As a result, only a few plants of this species were transplanted into the treatment site during the course of the summer of 2002 simply to help increase the biodiversity of the site. For reasons that remain unknown, this species spread profusely and prolifically throughout the wetland site, forming dense mats of vegetation, especially in Cells A and B (Figures 6.15 and 6.16). In the spring of 2003, a botanist with the Museum of Nature in Halifax identified this species as soft rush. This made sense since the cells had been thoroughly seeded with that species during the previous fall. However, as the growing

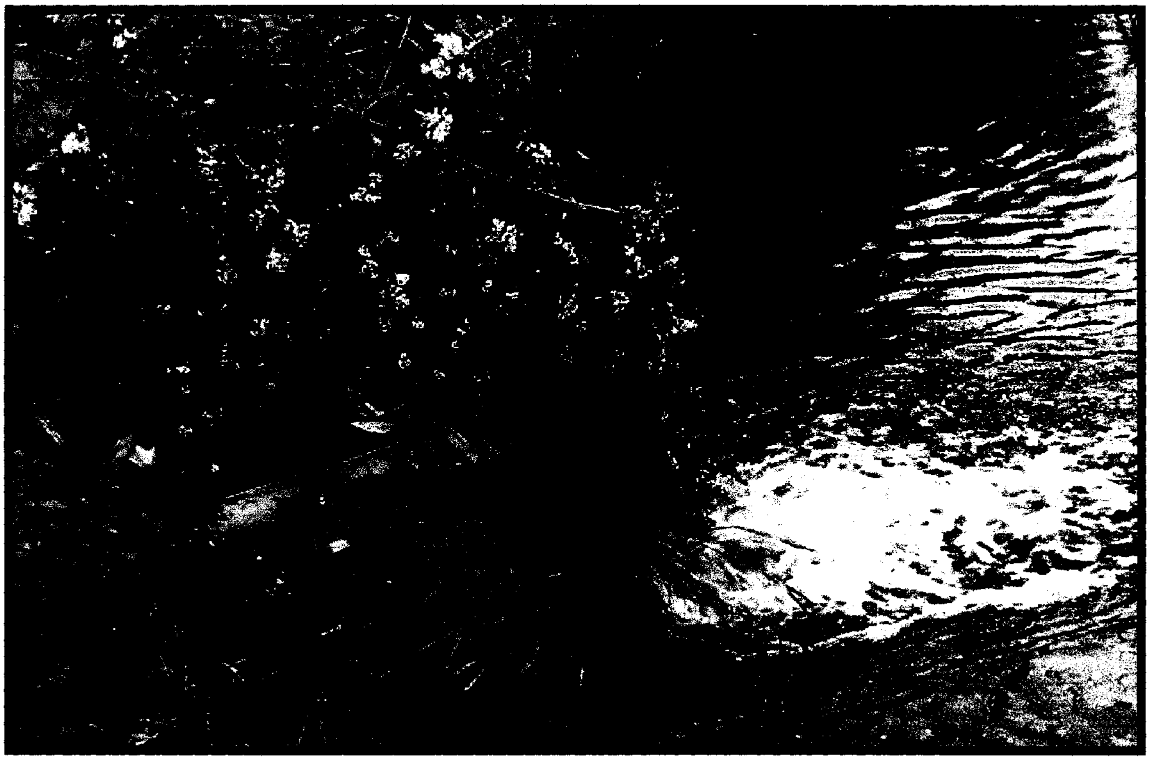


Figure 6.13. Rose Plant Established in Area Vulnerable to Human Disturbance.

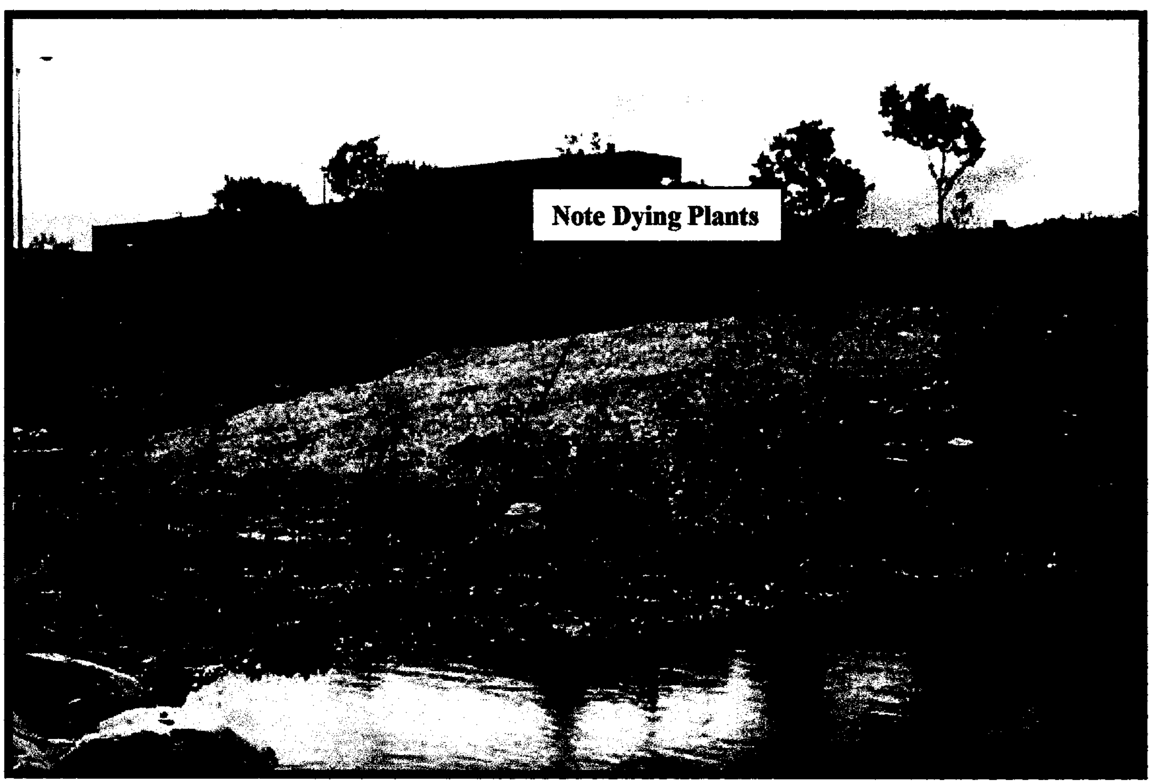


Figure 6.14. Established Trembling Aspen, Grey Birch, and Speckled Alder Species.

had been thoroughly seeded with that species during the previous fall. However, as the growing season continued, the plant began to take on features uncharacteristic of soft rush. Once flowers formed mid-summer, the plants was re-sampled and taken back to the museum where they were identified as Tweedy's rush. It is unknown whether this species may prove problematic in the future and overwhelm the site. No literature could be found explaining its rapid spread. This native species is not noted as being invasive or aggressive.

6.4. Evaluation of Biological Integrity of the Site

Biological assessments (bioassessments) evaluate the health of an ecosystem by directly measuring the condition of one or more of its taxonomic assemblages (i.e. fish, plants) compared to that of natural, healthy sites local to the area (reference sites). The major premise of bioassessments is that the community of plants and animals will reflect the underlying health of the ecosystem in which they live. The goal of such sampling is not to measure every attribute of the system, but rather to find efficient indicators of biological integrity that are expressed adequately with a minimal amount of sampling (USEPA, 2002h, m). Several different biological assemblages can be used to gauge the biological integrity in fresh water systems including: birds, fish, algae, amphibians, aquatic macroinvertebrates and plants. Of all these, the latter two are the most commonly used in wetland bioassessment.

Assessing the vegetative components of wetland systems is an excellent way to gauge the biological integrity of wetland systems as a whole, as they are excellent indicators of the overall ecosystem health. For example, wetland vegetation is an essential component of the food chain, commonly driving all other populations inhabiting a site. Hence, the composition and diversity of vegetative populations influences the populations of all other taxonomic groups (USEPA, 2002k). In addition, habitat for all taxonomic groups (from bacteria to fish) is largely facilitated through vegetation. Therefore, the integrity of biological populations is often directly influenced by the habitat quality of a site (USEPA, 2002k; and Karr, 1997). For these reasons, the vegetative assemblages of the treatment wetland and reference site were selected for examination in the bioassessment study.

The aquatic macroinvertebrate assemblages in the treatment wetland and reference site were also selected for the bioassessment study. Aquatic macroinvertebrates are by far the most commonly used bioindicators of biological integrity in fresh water systems including: steams, lakes, and wetlands. Monitoring macroinvertebrate community composition over time can provide useful data on trends and overall habitat and system health (USEPA, 2002l). The reasoning and benefits of monitoring this group include: (a) since aquatic macroinvertebrates are a highly diverse group, consisting of more than 4000 species in Canada, they are naturally excellent indicators of biological condition, especially biodiversity (Mandaville, 2002), (b) macroinvertebrates are typically the simplest and cheapest bioindicators to measure requiring few people, inexpensive gear, and having minimal impact on the system being sampled (Osmund et al., 1995c; and Hynes, 1998), (c) macroinvertebrates are essential components of wetland food webs (USEPA, 2002l), (d) unlike fish which may not even be present in wetland sites, aquatic macroinvertebrates are abundant in wetland systems, and are small enough to be easily collected, yet unlike plankton, are

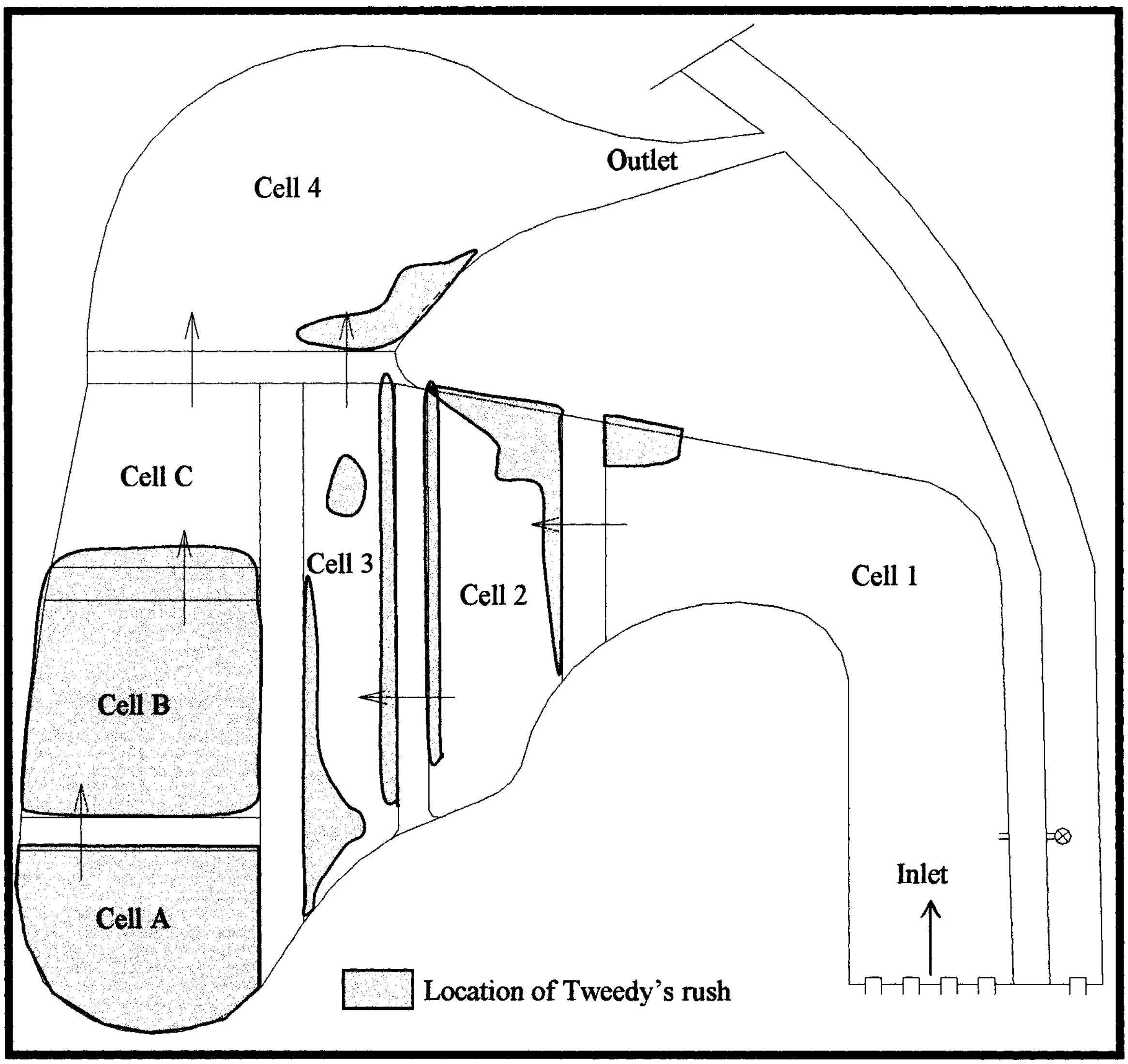
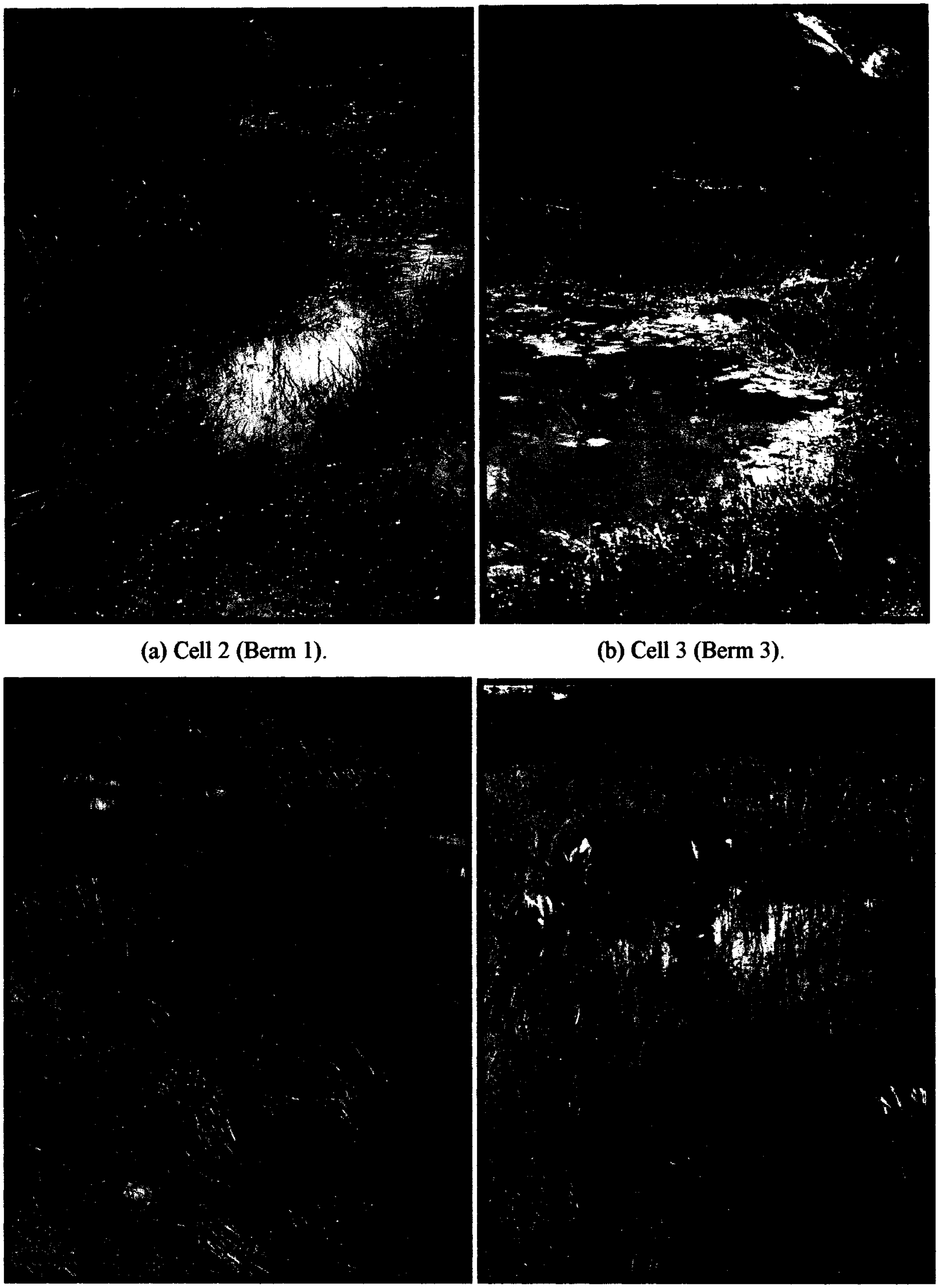


Figure 6.15. Spread of Tweedy's Rush (*Juncus brevicaudatus*) (where density approx. > 20 per m^2).



(a) Cell 2 (Berm 1).

(b) Cell 3 (Berm 3).

(c) Cell A.

(d) Cell B.

Figure 6.16. Tweedy's Rush in Treatment Wetland.

large enough to be easily identified (Osmund et al., 1995c; Hynes, 1998; Barbour et al., 1999; and Kirsch, 1999), (e) due to their limited mobilities and the fact that most species complete their life cycles in a relatively small area, aquatic macroinvertebrates are particularly well-suited for assessing site-specific health (USEPA, 2002l), and (f) aquatic macroinvertebrates are excellent indicators of system health as they are known to respond with a range of sensitivities to many kinds of stressors, and (USEPA, 2002l). However, there are disadvantages of using aquatic macroinvertebrates in bioassessments which include: (a) identification can be difficult, especially if identifying to species, and requires some level of expertise, (b) sampling can be complex, labour intensive, very time consuming, and expensive if identification is contracted, and (c) improper sampling procedure (especially when subsampling) can lead to unrepresentative and inaccurate results (Mandaville, 2002).

6.4.1. The Reference Site

When selecting a reference site, two major criterion must be considered (USEPA, 2002h). First, the reference site condition must be representative of a site with minimal human impact, and be relatively pristine. Second, the reference site must also be similar in classification, hydrological regime and vegetative structure in order to reduce variation, and detect genuine differences. The Burnside constructed treatment wetland, in a natural wetland setting, would be classified as a deep, open, freshwater, marsh ecosystem that is pH neutral as well as permanently inundated and dominated by emergent and submerged populations of vegetation. The reference site selected had to support these characteristics for the comparisons to be valid.

At first, a wetland model community located 200m northeast of the treatment wetland site was considered for use as a reference site. However, although similar in class and other characteristics, it was quickly dismissed as an inappropriate reference site as signs of disturbance such as staining, algal blooms and iron precipitate coagulations were visibly present in the site. It had also been hoped that one of the wetlands associated with Wright's Brook (indicated with purple in Figure 4.18) would be a suitable reference for the treatment site, as they would be regionally appropriate and most likely to contain the characteristics similar to the treatment wetland which is part of the Wright's Brook ecosystem. However, all the inventoried areas along the brook seem to be associated with bog ecosystems, none of which received a Gotlet score greater than 60. It is extremely likely that marsh ecosystems do exist along the course of the brook, however, they were likely too small to be included in the inventory.

Consequently, it was decided that a more pristine site needed to be selected. This pristine site was located through the use of area wetland surveys conducted by the Wetland Mapping Protection Program in the late 1980s. Marsh A-8 from the WMPP surveys was selected as the control site for the Burnside wetland. This pristine 3.6 ha open water marsh was also an excellent control candidate as it was dominated by emergent *Juncus* and *Scirpus* species, and incidentally supported abundant populations of soft rush (*Juncus effusus),* woolgrass (*Scirpus cyperinus*) and meadowsweet (*Spiraea alba*); three of the focal species in the Burnside treatment wetland. No signs of disturbance were observed with the exception of the access road located in close proximity to the site. Site quality was confirmed through water quality analysis.

6.4.2. Vegetation Sampling

According the Wetlands Division and Health and Ecological Criteria Division of the USEPA, for multiple-year monitoring such as that intended for the Burnside site, one-time annual sampling for indicators of biotic integrity including vegetation *and* macroinvertebrates should be adequate (USEPA, 2002j). This one-time sampling was purposely conducted later in the growing season as many species of plants (especially those of the Cyperaceae and Juncaceae family) are difficult to identify unless flowering.

6.4.2.1. Species Diversity. Biodiversity is often sited as the pre-eminent gauge of biological integrity. All biological integrity assessments tend to integrate some gauge of biodiversity as part of their study (Daigle and Havinga, 1996; Murphy 1999; and USEPA, 2002g,h,k). The H' values for both sites were perceptibly close (2.069 and 2.053). With little scrutiny, one could easily deduce that in terms of species diversity, the sites were very similar. This was confirmed statistically through the use of t-tests. The derived hypotheses were:

> H_0: Significant differences exist between the plant species diversities of the sites
> H_1: No significant difference between the plant species diversity of the sites

The calculated test statistic must be higher than the critical $\underline{t}$ –value (P=0.05) in order for the null hypothesis to be rejected (Murphy, 1999). The $\underline{t}$-statistic value of 0.071 which was significantly less than the critical $\underline{t}$ values of 1.960 and 1.980 ($\alpha 2$ P=0.05), indicated that there was no significant difference between the plant species diversities in the two wetland sites. This result is significant as one of the aims of this study was to create a treatment wetland site with high vegetative diversity, similar to that of a pristine, natural wetland site.

6.4.2.2. Heterogeneity (Dominance). Natural, healthy wetlands not only support diverse populations of plants, but balanced proportions of species. Disturbed sites, especially those undergoing early succession such as the treatment wetland site, can often support high diversity, but in skewed proportions (e.g. domination of roadside ditches by duckweed or purple loosestrife). The domination of one or a few species can be early signs that vegetative populations are becoming out-competed by invasive species. Sites that are more heterogeneous are more likely to maintain biodiversity by preventing competitive exclusion (Stevens and Carson, 2002; and Crawley, 1997). Hence, the proportional abundance and distribution of plant species can be an important indicator of system health (Osmund et al., 1995c; USEPA, 2002k; and Murphy, 1999) and can lead to effective and informed management decisions concerning site maintenance (i.e. proportion of cattail increasing annually, hence cattail control may need to be implemented).

The top three species in the treatment wetland site were tweedy's rush (*Juncus brevicaudatus)* with 60 individuals sampled, soft rush (*Juncus effusus*) with 44 individuals sampled, and fowl mannagrass (*Glyceria striata)* with 17 individuals sampled. In total, these three species occupied 46.4% of the sampled population. The top three species in the reference site were soft rush (*Juncus effusus*) with 38 individuals sampled, sweetgale (*Myrica gale*) with 27 individuals sampled and woolgrass (*Scirpus cyperinus*) with 22 individuals sampled. In total, these three native species occupied a more reasonable 32.6% of the sampled population (USEPA, 2002k). Therefore, the reference wetland site supported higher heterogeneity than the treatment wetland site. Although the two wetland sites appeared to be equally biodiverse and more species were found in the constructed wetland site overall, nearly 50% of the constructed wetland site is occupied by 3 species, which indicates that the site is not as healthy as indicated by the richness and diversity results. The results indicated that although currently diverse, the distribution of the species is skewed. This is often viewed as a sign that the biodiversity of a site is under threat as other species are being out-competed by aggressive stands (USEPA, 2002k; Osmund et al., 1995c; and Carlisle, 1998). The tweedy's rush in particular seems to have a ruling presence in the treatment wetland site. However, the tweedy's rush, soft rush and mannagrass are all notably native plants which are not reported to exhibit invasive or aggressive characteristics (Zinck, 1998; and Thunhorst, 1996). Had any of these species been exotic or known to exhibit invasive tendencies, there might have been cause for greater concern. Regardless, the heterogeneity of the treatment site is far from ideal. Perhaps as slower growing species become more established as the site matures over the years, the vegetative heterogeneity of the treatment site will improve

(Thunhorst, 1992; Zinck, 1998; and Daigle and Havinga, 1996). However, if the tweedy's rush or any other species continues to show signs of increasing domination, control measures such as weeding may need to be implemented.

6.4.2.3. Exotic Species Abundance. Wetlands which support biological integrity support healthy assemblages of native plant life. Conversely, sites supporting abundant populations of exotic and invasive species are perceived as systems of poor health, suffering from what is termed "biological pollution". Abundant invasive and exotic species tend to out-compete native vegetation and destroy the natural diversity (and habitat) of the community (Thunhorst, 1993; and Daigle and Havinga, 1996).

The treatment site had a 10.73% abundance of exotic and invasive species while the reference site had only a 3.75% abundance of exotic and invasive species. Although a community ideally would support 100% native species, this is rarely seen even in the most pristine environments, as most exotic plant seeds have highly adapted transport mechanisms which allow them to get just about anywhere (Daigle and Havinga, 1996). The sampling regime may have brought to light the potential problem of exotic, aggressive species out-competing native vegetation in the treatment wetland site in the future. The exotic and/or aggressive species sampled in the treatment wetland included reed canary grass (*Phalaris arundinacea)*, oxeye daisy (*Chrysanthemum leucanthemum*), broad-leaved cattail (*Typha latifolia*), stinkweed (*Thlaspi arvense)*, timothy hay (*Phleum pratense)*, dame's rocket (*Hesperis matronalis),* Canada thistle (*Cirsium arvense),* coltsfoot (*Tussilago farfara),* horsetail (*Equisetum arvense),* field mustard (*Brassica rapa)* and yellow hawkweed (*Hieracium florentinum).* Of these, reed canary grass and broad-leaved cattail are possible invaders, especially in disturbed areas (Thunhorst, 1993). More aggressive control of these species may become necessary in the future in order to prevent them from overtaking the established native vegetation in the site. However, none of these species were present in high numbers, so it is just as likely that their presence is simply a symptom of early succession. Succession occurs when areas suffer a disturbance such as a fire or tornado (or in this case, excavation), which removes the existing vegetation. The first colonizers in areas are typically aggressive species which are adapted to take advantage of full sunlight exposure and inhabit bare areas quickly (Crawley, 1997). However, as a disturbed site matures, and space and light availability become lessened, these successionary colonizers often give way to more hardy, slow-spreading species which are better light competitors. Hence, as the treatment wetland system matures, its plant biodiversity may actually decrease. However, its integrity, as measured by exotic and invasive species abundance as well as heterogeneity, may actually increase, so long as invasive species do not overwhelm the site (Crawley, 1997).

6.4.3. Aquatic Macroinvertebrate Sampling

There are numerous monitoring techniques utilized to sample aquatic macroinvertebrate populations, some of which are more complex, difficult and time consuming than others. Recent trends, however, have been towards more standardized, rapid bioassessment techniques, which utilize semi-quantitative collecting methods such as dip-netting (Mandaville, 1999). The key to selecting an appropriate technique is to balance scientific validity with sampling efficiency in a manner that does not compromise the integrity of the monitoring program (Daigle and Havinga, 1996). In the case of the Burnside wetland, it was important that the techniques and analysis methods selected would be easily reproducible, as future monitoring of the site will be a carried out by volunteers. Dip-netting was selected because it is probably the most common method for sampling invertebrates in wetlands (USEPA, 2002l; and Mandaville, 2002). In the interest of time and simplicity, a subsample of 100 organisms was randomly selected from the samples collected from the dipnetting for the analysis. According to David et al. (1998) and Mandeville (2002), subsamples of 200 or 300 animals are superior to subsamples of 100 animals. However, according to statistical analyses and associated power calculations conducted by the Ontario Ministry of the Environment, only modest gains in sample representativeness were seen in subsamples using 200 or 300 animals. In addition, these gains decreased with the use of multi-metrics, such as those being used in this study (David et al., 1998; and Mandaville, 2002.).

The late sampling (July 2003) of the aquatic macroinvertebrates proved to be conducive to the identification of most macroinvertebrate taxa, as the immature stages of macroinvertebrates present in spring can be quite difficult to identify. In addition, one-time phosphorus loading (a likely consequence of seasonal greenspace fertilizing) occurred in early spring, resulting in hyper-eutrophic conditions temporarily triggering prolific algal growth. Had the sampling taken place in the spring, the results would not have been representative of the conditions occurring in the site during the remainder of the summer and autumn.

The aquatic macroinvertebrates for each sample site were separated into general groups for easier identification (ie. worms and leeches, dragonflies, mayflies and other large predators, beetles, diptera, and so on). It is typically recommended that all samples be identified to the lowest possible taxonomic level, as different sensitivities and functions can be attributed to different individuals within some taxonomic groups (Mandeville, 2002; and USEPA, 2002l). Unfortunately, this can prove difficult, even for experts, especially when identifying to species. For example, fingernail clam and leech species identification is based on chemical shell processing, viewing hinge teeth, or dissecting reproductive structures (USEPA, 2002l). This level of identification would not be easily reproducible for subsequent monitoring teams examining the Burnside treatment wetland site. According to Mandaville (2002) and the USEPA (2002l), many rapid assessment approaches or citizen monitoring programs suggest only identifying to the family level. Therefore, identification using keys to at least family level was attempted for all samples. If identification required complex procedures (i.e. dissection), the individual was identified to its nearest practical taxonomic level.

6.4.3.1. Population Diversity. Biodiversity is often sited as the pre-eminent gauge of biological integrity (Daigle and Havinga, 1996). The derived Shannon-Weiner diversity values (H') for the reference wetland, Cell 1 and outlet area of the treatment wetland indicate that in terms of their aquatic macroinvertebrate populations, the reference wetland is the most diverse (1.115), followed by Cell 1 (0.882), and the outlet (0.675). Since one of the aims of the study was to create a naturalized treatment wetland site that would have a diverse aquatic macroinvertebrate population similar to that of a pristine, natural wetland site and it was expected that this similarity would increase from Cell 1 to the outlet as the water quality and habitat value of the cells improved. The significance of these differences were tested statistically by comparing the values derived from the Shannon-Weiner diversity index for each site via t-tests. The derived hypotheses were:

H_0: Significant differences exist between the aquatic macroinvertebrate diversities of the three locations (reference wetland, Cell 1 and outlet of treatment wetland)

H_1: No significant differences exist between the aquatic macroinvertebrate diversities of the three locations (reference wetland, Cell 1 and outlet of treatment wetland)

The calculated test statistics must be higher than the critical t –values (P=0.05) in order for the null hypotheses to be rejected (Murphy, 1999). The t-statistic values of 0.690, 1.595, and 0.968 obtained for the macroinvertebrate diversity comparisons between the reference wetland and treatment wetland Cell 1, the reference wetland and the treatment wetland outlet, and Cell 1 and the outlet respectively, were less than the critical t-values of 1.960 and 1.980 for all the comparisons. Hence, the null hypothesis was not rejected, which implies that there were no significant differences between the aquatic macroinvertebrates diversity at the test sites. However, the reference wetland supported nearly 40% more families of macroinvertebrates than the outlet, and nearly 30% more families than Cell 1. The macroinvertebrates in the treatment wetland were sampled only during the second growing season of the site. As the vegetation in the treatment wetland matures, superior micro-habitat and water quality improvement is expected to occur. Therefore, it is likely that the macroinvertebrate diversity in the site will improve in subsequent growing seasons.

6.4.3.2. Heterogeneity (Dominance). Natural, healthy wetlands not only support diverse populations of macroinvertebrates, but balanced proportions of species as well (USEPA, 2002l; Voshell, 2002; and Osmund et al., 1995c). The top taxon in the reference wetland site were the Oligochaeta (aquatic worms) with 33 individuals, the Corixidae (water boatmen) with 17 individuals and the Hirudinea (leeches) with 8 individuals, all of which accounted for 58% of the sampled population. The top three taxon in Cell 1 of the treatment wetland were the Hyalellidae (scuds) with 30 individuals, the Oligochaeta with 24 individuals and the Corixidae with 10 individuals, all of which accounted for 64% of the sampled population. Finally, the top three taxon from the treatment wetland outlet were the Hyalellidae with 42 individuals, the Oligochaeta with 15 individuals and the Elmidae (riffle beetles) with 10 individuals, all of which accounted for 67% of the sampled population. Hence, the reference wetland site supported higher taxon heterogeneity than both Cell 1 and the outlet, and Cell 1 supported slightly higher heterogeneity than the outlet. It had been hoped the heterogeneity of the macroinvertebrates in the treatment wetland would be similar to that of the reference wetland, and that heterogeneity would be improved with the hypothesized improving water quality, but the results revealed that the treatment wetland site lacked the biological integrity conducive to supporting appropriately distributed populations of macroinvertebrates (USEPA, 2002l; and Voshell, 2002). However, it is likely that as the system matures, the heterogeneity of the macroinvertebrates population in the site will improve. In addition, the following noteworthy observations must be noted:

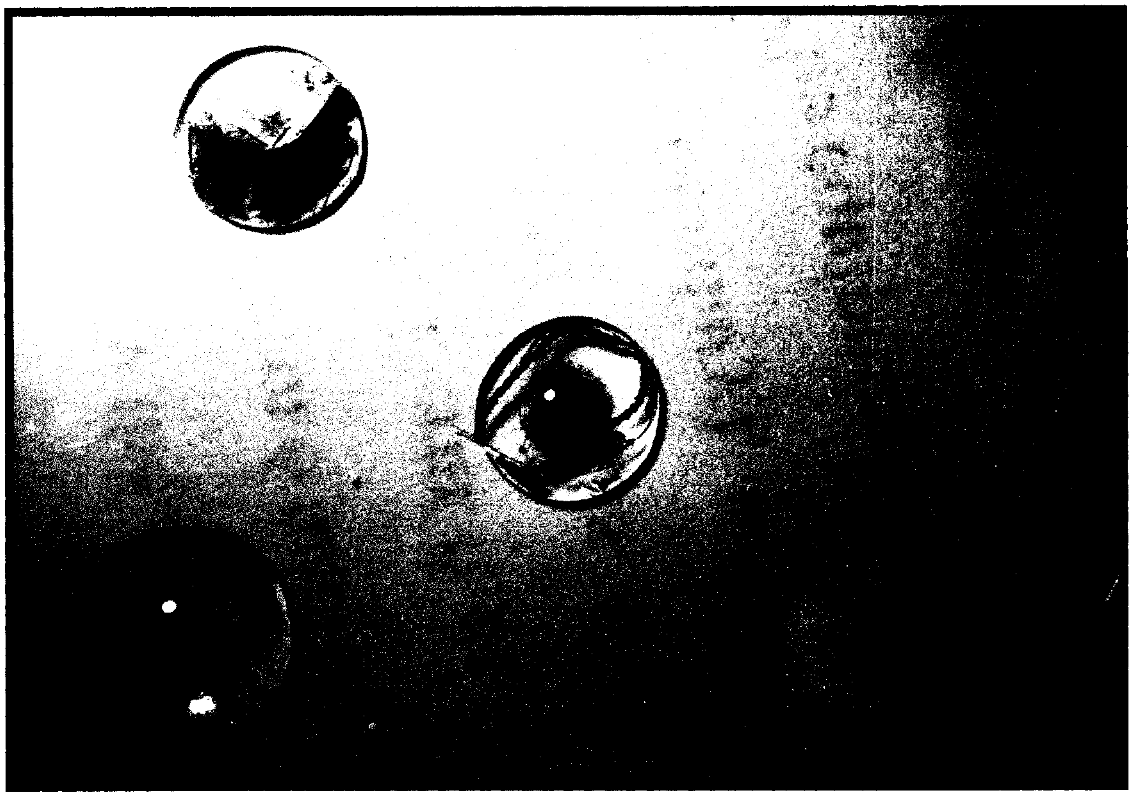
(a) both the reference and treatment wetland sites supported high abundances of Oligochaeta (Figure 6.17). Aquatic worms are typically very abundant in healthy wetlands, especially in deeper areas of the waterbodies. They are actually very important to overall biological health, as their feeding habits cause them to continuously mix the top 5 to 10 cm of the bottom sediment. This vertical mixing exposes otherwise anaerobic sediments to dissolved oxygen from the waterbody, keeping sediments oxygen-rich, which is vital to a diverse assemblage of organisms. As a result, the presence of aquatic worms is often cited as indicators of biological integrity (Voshell, 2002). Hence, their high abundance in the treatment wetland demonstrates that the site does support some semblance of health.

(b) the treatment wetland site supported a high abundance of Hyalellidae, especially in the samples from the outlet (Figure 6.18). Often, these insects are very abundant in small habitats without fish. In fact, some small, spring-fed streams with thick rooted vegetation and abundant detritus for food support up to 10,000 scuds per m^2. They are important to the breakdown of organic matter and are an essential to the diet of many invertebrate predators, including amphibians and waterfowl. Although dominating the treatment wetland site, their high abundance is not indicative of poor biological health (Voshell, 2002).

(c) Corixidae were observably abundant in the reference site as well as Cell 1 of the treatment site (Figure 6.19). This family of insect is notably not dependent on dissolved oxygen levels in the water column, as they breath air from air bubbles held under their wings, which is obtained from surface air. These insects are very tolerant of chemical and biological stress and are typically capable of surviving in nearly any permanent open-water environment. Although their presence is not cited as a gauge of biological integrity, it does not indicate poor biological health (Voshell, 2002; and USEPA, 2002l).

(d) the outlet supported a relatively high abundance of Elmidae larvae. Like the Corixidae, most species of riffle beetle are capable of thriving in both healthy and disturbed systems, and are therefore not cited as indicators of biological integrity (Voshell, 2002).

6.4.3.3. Trophic Structure. In aquatic environments, four major trophic groups of macroinvertebrates exist: scrapers, shedders, collectors, and predators. Scrapers are the organisms most often found clinging to rocks, 'scraping' off and feeding on algae for food. Collectors consume fine particulate organic matter (FPOM), usually detritus, either by filter-feeding or obtaining materials from substrates. Shredders consume larger particulate matter (coarse particulate organic matter or CPOM) such as aquatic plants and detritus through maceration.



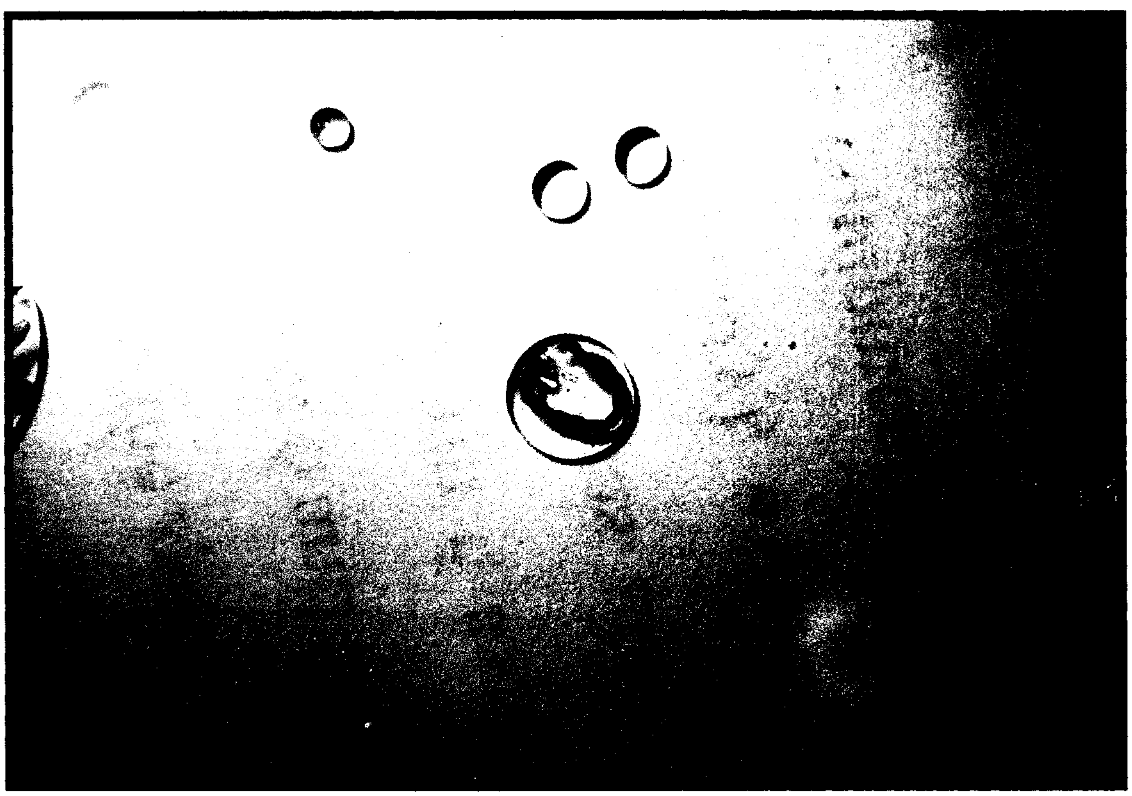


Figure 6.17. Oligochaeta (Aquatic Worms) in Reference Wetland Site Sample.

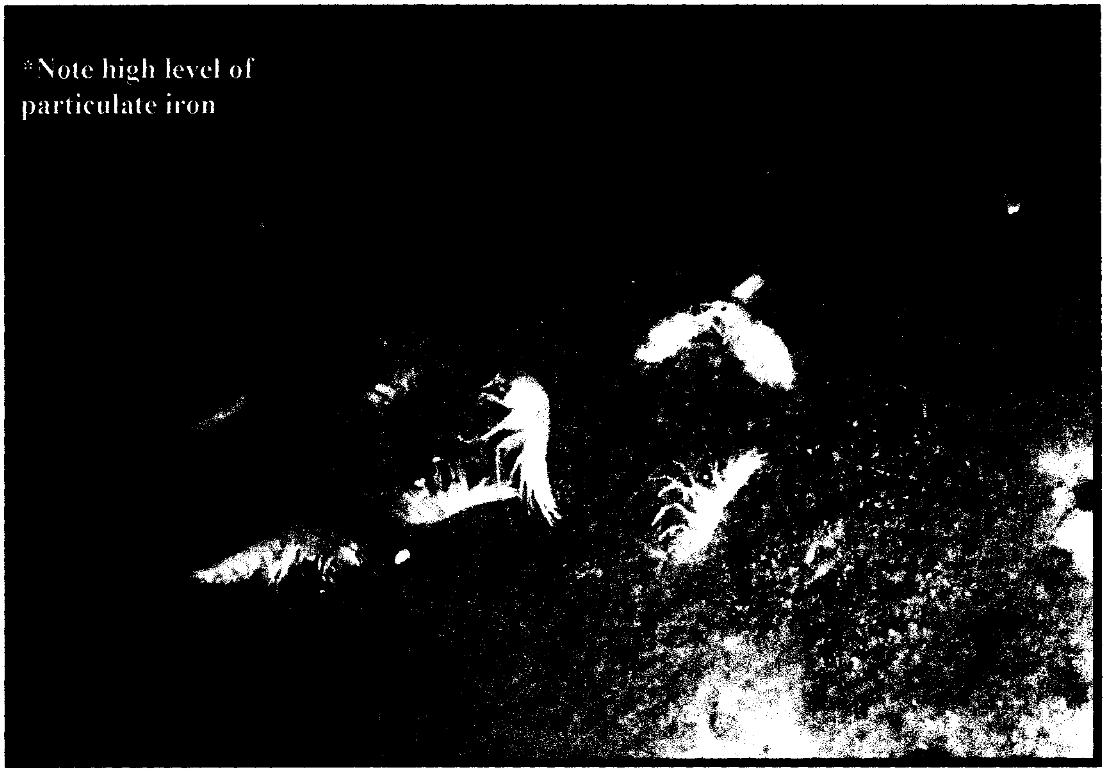


Figure 6.18. Hyalellidae in Outlet Sample.

Figure 6.19. Corixidae (Water Boatman) in Cell 1 Sample.

Finally, predators prey on living animals (typically other macroinvertebrates) for food. Shifts in trophic structure are often indicative of a community responding to an overabundance of a particular food source, or to disturbance (Mason, 1998; and Voshell, 2002).

In the reference wetland, the shredder, scraper, predator and collector ratio was 2:2:21:75, which is normal for natural wetland sites (Mitsch and Gosselink, 2000; and Mason, 1998). In Cell 1, the ratio was 0:2:2:96, and in the outlet the ratio was 0:0:3:97. Clearly, the functional feeding group ratios in the treatment wetland were dissimilar to the reference site. Therefore, according to this metric, the treatment wetland currently does not support the same level of biological integrity as the natural wetland site. Specifically, the treatment wetland was lacking in shredders. The absence of this group is often interpreted as a sign of poor water and habitat quality, as toxic substances and particulate matter often bind to CPOM which is their food source (Osmund et al., 1995c). This may be an indication that the wetland is not purifying the leachate-contaminated waters as effectively as it was designed to do. The treatment wetland site was also lacking in scrapers, which tend to feed on algae that grow attached to solid objects (periphyton) such as rocks. The absence of this feeding group in the treatment wetland site is a likely consequence of the presence of abundant iron precipitate observed in the waters, especially in the outlet portion of the site. Iron precipitate is non-toxic to organisms as once iron is oxidized, it is no longer bioavailable. However, iron precipitate coats sediment, gravel and rocks, limiting the ability of periphyton to grow. Of the algae that are able to sustain themselves, they too become smothered in particulate, making them inaccessible to the scrapers. Other effects associated with increased suspended particulate matter include the smothering of eggs and larvae, and decreased dissolved oxygen concentrations, which is commonly a limiting factor to predators, that tend to have high oxygen demands (Mitsch and Gosselink, 2000; Sample et al., 1997; and Voshell, 2002). Predators are also notably scarce from the treatment wetland site. The fact that the site supports abundant collectors is also explained by the presence of the high particulate content of the waters, as collectors feed on particulate matter. As reported by Mason (1998) and Voshell (2002), trophic structure shifts often occur in response to the overabundance of a particular food source. As the system matures and its remediative ability improves, the trophic structure the macroinvertebrates in the treatment wetland site may improve.

6.4.4 Wildlife Observations

In little more than one growing season, the Burnside treatment wetland site transformed from a barren, muddy series of holes which appeared incapable of ever supporting life to a lush wetland environment supporting abundant number of insects, frogs and waterfowl. The abundance of bullfrogs was particularly noteworthy. Frogs are sensitive creatures, and their presence is often cited as an indicator of sound ecosystem health (USEPA, 2002g).

6.5. Evaluation of the Water Treatment Ability of the Constructed Wetland

6.5.1. Chemical Water Quality Analysis

It should be noted that constructed wetlands do not become completely functional overnight and years may elapse before performance reaches optimal levels (Bennet, 1994). At the time of monitoring, the constructed wetland was only in its first year of operation, and continues to be altered as sufficient funds become available.

Cells A, B and C were neglected in the water quality analyses for two reasons: (a) they were initially separated from Cells 1-3 because they received substantial influent flow from beneath Akerley Boulevard, and (b) water quality analysis of the input from beneath the road revealed no elevated concentrations of contaminants of concern , hence Cells A, B and C simply serve as a source of dilution in the site.

A biweekly sampling regime was selected to accurately characterize the nature of the influent, as well as to gauge the progressing success of the wetland. Given the potentially variable nature of the contaminated input due to the continued decomposition and maturity of the landfill, regular monitoring was particularly justified (i.e. affected by state of waste decomposition, varying natures of wastes being decomposed, effect of precipitation on contaminants, etc.). In cases of such variability, Kadlec and Knight (1996) and Hicks and Stober (1989) suggested that monthly or quarterly site sampling would be inadequate. It was also intended that this sampling regime would result in more accurate analysis and recognition of site trends and variabilities in relation to internal and external influences. However, it is anticipated that the intensity of this monitoring regime will trail off slowly to a few times a year once water quality objectives begin to be consistently met after 2-3 growing seasons (Hammer, 1992).

The four major stages of landfill decomposition are shown in Figure 6.20 (Tchobanoglous et al., 1993). Each stage results in leachate of varying compositions. In the early stages of a landfill life, oxygen trapped within the buried refuse is quickly consumed by aerobic microbial activity in the breakdown of organic matter. This initial stage of landfill decomposition is referred to as the aerobic stage. It is characterized by the production of large quantities of carbon dioxide, as well as temperature increases within the waste. This oxygen is depleted in a few days and the landfill site transforms into an anaerobic system (Barlaz, 1996).

The next stage of landfill waste decomposition (the anaerobic stage) lasts anywhere from one to ten years. It has been observed that leachate from the second stage tends to be high in volatile fatty acids (carboxylic acids such as acetate, propionate and butyrate), derived mostly from anaerobic biological activity. These organic acids typically cause the resulting leachate to support a low pH (typically 4.5 –5.5), and a high chemical oxygen demand (between 20,000 and 35,000 mg/L) (Silva et al., 2003; and Cusso et al., 2001). Consequently, this stage of landfill leachate production is often referred to as the acid phase. The acidic nature of the leachate results in significantly higher metal concentrations because of weathering of soil metals. The metals which are most commonly present in landfill leachate at high concentrations are iron and manganese in their reduced forms (Fe^{2+} and Mn^{2+}), which are the two main contaminants of concern in the Burnside landfill leachate. Zinc is also commonly found in landfill leachate (Liehr et al., 2000). Acid phase leachates also typically contain elevated levels of biochemical oxygen demand (BOD) and ammonia as a result of high decomposition activity of landfill organic matter (Barlaz, 1996; and Barlaz and Gabr, 2001).

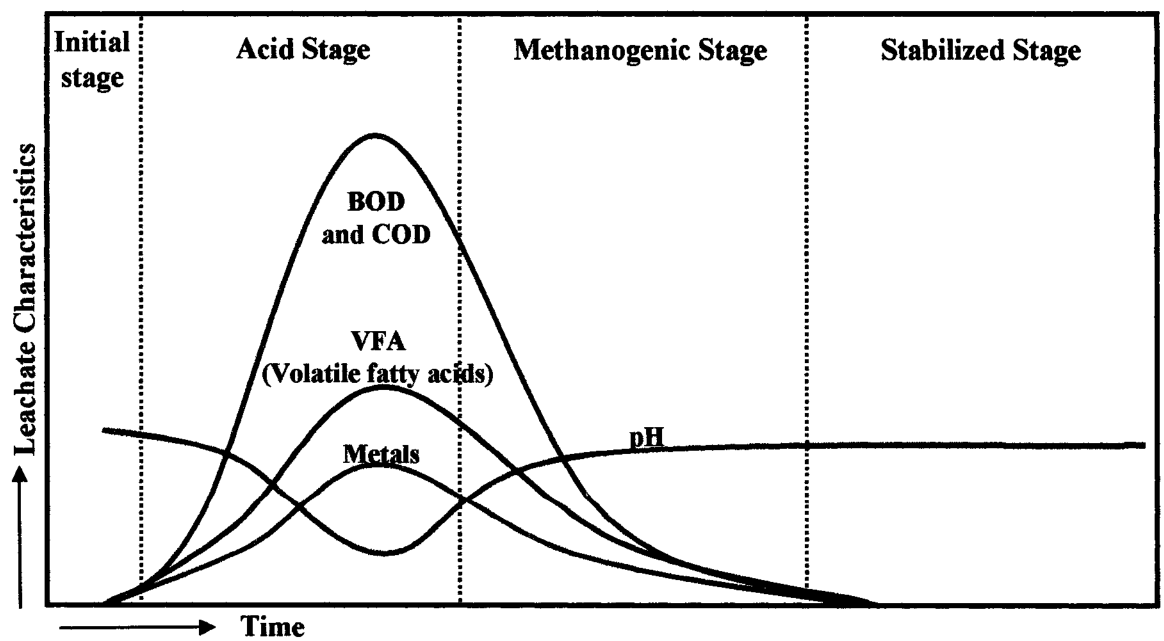


Figure 6.20. Typical Landfill Leachate Characteristics Over Time (adapted from Tchobanoglous et al., 1993).

As a landfill continues to mature, the accumulated carboxylic acids begin to convert to methane and carbon dioxide by methanogenic bacteria. As their activity increases, significant methane and carbon dioxide production occurs, marking the onset of what is termed the methanogenic stage of landfill decomposition. As the acids are consumed, COD and BOD levels decrease and pH begins to rise. This phase of decomposition can be expected to last anywhere from 30 to 200 years (Barlaz, 1996). Several studies suggest that there is no mechanism for ammonia degradation under methanogenic conditions (Robinson, 1995; Burton and Watson-Craik, 1998; Barlaz, 1996; and Kruempelbeck and Ehrig, 1999). In a study of 50 landfills in Germany, Krumpelbeck and Ehrig (1999) found that none of sites exhibited significant decreases in ammonia concentrations up to 30 years after closure.

As time continues to pass and a system matures, waste degradation will progress closer to completion as all the readily biodegradable waste becomes converted to methane and carbon dioxide. The carboxylic acid concentration decreases and the pH continues to increase until ultimately the system becomes stabilized. Hence this final phase of landfill decomposition as referred to as the stabilized or maturation stage. It is in this final stage of degradation that leachates would cease to be hazardous to the environment. The stabilized stage is somewhat theoretical, as the time it would take for complete waste degradation to occur and the landfill to convert back to an aerobic system could very well be on a geologic scale, and has not fully been observed (Barlaz, 1996; and Tchobanoglous et al., 1993).

The indicators of the degree of waste decomposition in landfills include: iron, manganese, COD, BOD, and ammonium concentrations, redox potential, alkalinity, ionic strength, BOD:COD, and the sulfate to chlorine ratio (Barlaz, 1996). In this study, iron, manganese, nitrite, nitrate, ammonium, total Kjeldahl nitrogen, orthophosphate, total solids, and chemical oxygen demand were measured. The pH and dissolved oxygen were also assessed in relation to these parameters. In general, the 2003 water quality results of the Burnside treatment wetland influent saw high iron and manganese concentrations, which is an indication of a landfill in the acid stage of decomposition. However, neutral pHs as well as minimal concentrations of COD and ammonia were also observed, indicating that the landfill decomposition is much more advanced. Given the former landfill age and the constituent concentrations observed, it is likely that the Burnside landfill is between the acid and methanogenic phases of decomposition.

6.5.1.1. <u>Iron.</u> Iron was identified as one of the contaminants of concern in the Burnside landfill leachate effluent. Iron is an essential micronutrient element required by both plants and wildlife at significant concentrations, as it is a vital part of metabolic enzyme formation and the oxygen transport mechanism in the blood (haemoglobin) of all vertebrate and some invertebrate animals (King et al., 1992). Iron also occurs as one of the principle chemical constituents in water. Natural waters contain variable amounts of iron depending on the geological area and other chemical components of the waterway and is present at concentrations typically less than 500 μg/L. Natural concentrations of iron in surface water usually occur as a result of igneous and sandstone rocks (Boulton and Brock, 1999; and Mason, 1998).

Iron exists in two forms: soluble ferrous (Fe^{2+}) and insoluble ferric (Fe^{3+}). At a pH greater than 3.5 with oxygen present, ferrous iron will oxidize to ferric iron. If oxygen is low, this transformation will not occur until the pH reaches 8.5 (Fripp et al., 2000). Certain microbes (*Thiobacillus-Ferrobacilus* bacteria) can increase the rate of iron oxidation The oxidation of Fe^{2+} to Fe^{3+} occurs as the following reaction (Colorado School of Mines, 1999):

$$4Fe^{2+}_{(aq)} + O_{2(g)} + 4H^{+}_{(aq)}\,(\text{Acidity}) \dashrightarrow 4Fe^{3+}_{(aq)} + 2H_2O_{(l)} \tag{6.1}$$

In most aquatic systems supporting waters with adequate oxygen concentrations, all iron present in the waterbody and other aerobic parts of the system such as the rhizosphere of the sediments will be insoluble (oxidized Fe^{3+}). Fe^{2+} can persist in waters void of dissolved oxygen, and commonly resides in anaerobic bottom sediments (EPRI, 2001 and Boulton and Brock, 1999). In surface waters, ferric iron frequently forms iron hydroxide ($Fe(OH)_3$) precipitate as a result of hydroxylation (Fe^{3+} reacting with H_2O molecules) as follows (Colorado School of Mines, 1999):

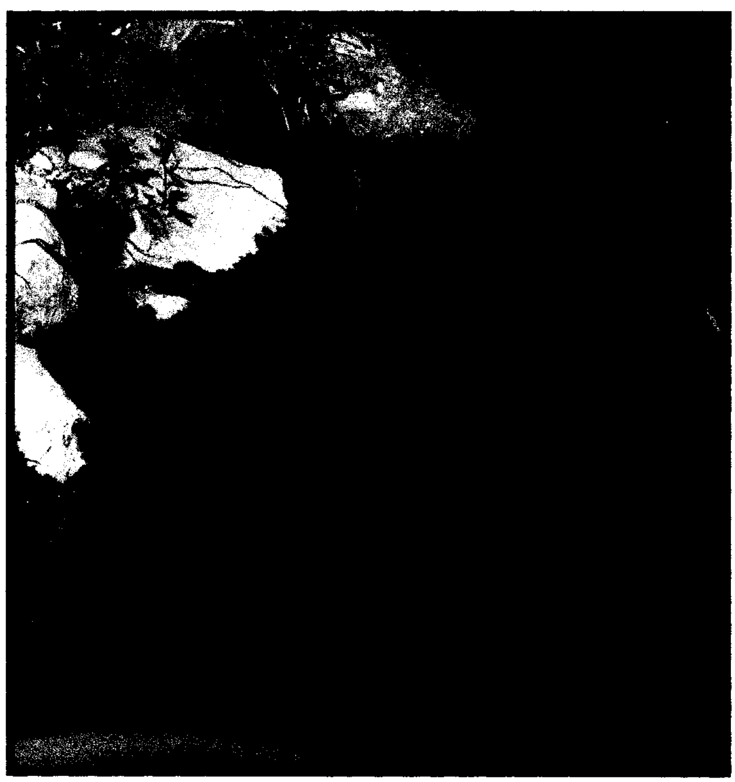
$$Fe^{3+}_{(aq)} + 3\,H_2O_{(l)} \dashrightarrow Fe(OH)_{3(s)} + 3H^{+}_{(aq)}\,(\text{Acidity}) \tag{6.2}$$

Iron precipitates such as iron hydroxide and ferric phosphate ($FePO_4$) cause orange staining and the formation of slick orange coagulations commonly referred to as ocher, yellow boys or yellow dogs (Boulton and Brock, 1999). Despite its odious appearance, Fe^{3+} and its associated compounds are not acutely toxic to wildlife, as they are not readily bioavailable. This is because the primary pathway for most metal uptake by aquatic life is through respiratory organs or via the consumption of other organisms (i.e. aquatic invertebrates). In seeing that only dissolved ionic forms of metals can pass through cell membranes, the direct toxicity of insoluble metals such as Fe^{3+} is limited (USEPA, 1995; and Boulton and Brock, 1999). The reduced form of iron (Fe^{2+}) is soluble and is, therefore, bioavailable and considered to be extremely toxic to wildlife (Mitsch and Gosselink, 2000). However, this iron ion is mostly found in deeper, anaerobic sediments, it is not as physically available to wildlife and is only available for uptake by plants with deeply penetrating roots. Given that the pH of the Burnside landfill leachate is neutral, and that substantial orange staining and abundant ochre have been observed in the treatment wetland and Wright's Brook (Figure 6.21), it is highly likely that the majority of the iron detected in the 2003 water quality analysis is non-toxic Fe^{3+} and its associated precipitates.

The remediation of metal contaminated sites in general can be particularly challenging, as unlike organic contaminants, metals cannot be degraded (FRTR, 1998; and CPEO, 1998). Although phytoremediation technologies have been showing much promise in remediating metal contaminated sites, it must be noted that phytoextraction of metals and other contaminants will only occur if they are bioavailable for plant uptake. Metals like iron which are oxidized, bound to soil organic matter, or precipitated as the oxides, hydroxides, and carbonates are not readily bioavailable, hence their removal via phytoextraction is not typically ineffective (FRTR, 1998; CPEO, 1998; and Lasat, 2000).

Engineered wetlands for the treatment of acid mine drainage (AMD), which is extremely high in Fe^{3+} and its associated precipitates are effective at removing high iron concentrations. Lorion (2001) reports removal rates of up to 99% for iron in AMD treatment wetland sites. Similarly, in treatment wetlands designed to treat iron in contaminated wastewaters from coal combustion processes, Ye et al. (2001a) found FWS systems successfully removed up to 91% of iron. In a review of constructed wetlands in the USA, Skousen et al. (1994) reported iron removal rates from acid mine drainage ranging from 28 and 99%. However, these figures can be unintentionally deceiving to the reader, as the term "removed" is typically inclusive of both plant uptake *and* the forming of particulate. In fact, it is immobilization via oxidative precipitation that generally accounts for the majority of the reported iron removal rates in treatment wetland systems (Wieder, 1988; Perry and Kleinmann, 1991; Brodie et al., 1993; National Rivers Authority, 1992; Henrot and Weider, 1990; and Skousen et al., 1994). In the specific case of AMD FWS wetlands, according to Henrot and Wieder (1990), Calabrese et al. (1991) and Wieder (1992), about 50 to 70% of total iron removal is via iron hydroxylation (Skousen, 1998). Clearly, when iron precipitates, it is not physically removed from the system, but becomes unavailable to biological components, so in a sense, the harmful toxic metal is removed (Ye et al., 2001a). Ye et al. (2001a) observed that in their wetland systems (which achieved 91% removal of iron), only

(a) Cell 1.



(b) Outlet.

Figure 6.21. Iron Particulate in Treatment Wetland, 2003.

0.91% of iron removal was carried out by the wetland plant population. Similarly, in studies conducted by Sencindiver and Bhumbla (1988), Faulkner and Richardson (1989), Mitsch and Wise (1998) and Whiting and Terry (1999), iron accumulation by plants only accounted for less than 1% of the iron removal in several study sites (Ye et al., 2001a). The primary sink for metal particulates consequently becomes bottom sediments as they settle out and accumulate. As accumulation occurs and oxygen becomes more and more depleted in the lower portions of the build up, oxidized iron may be reduced back to its bioavailable dissolved form, where it then can be taken up by vegetation (Mitsch and Gosselink, 2000). However, in areas accommodating high loading rates, accumulation will likely overwhelm any removal progress via phytoremediation processes (Ye et al., 2001a). As the end result, most systems receiving high suspended solid loads will need to be physically dredged of their sediment build-up once every few years (Davis, 1995).

However, the role of plants in treatment wetlands should not be discounted as they still play a crucial part in metal removal by actively facilitating the physiochemical reactions responsible for the majority of their removal. This supporting activity includes physical trapping and filtration of particulates, facilitation of sedimentation, and sediment stabilization. Most importantly, however, wetland plants facilitate metal removal by oxygenating the water column and the soils via radial oxygen loss (ROL), thereby promoting the oxidation processes which render the metals immobile (i.e. ferric iron and its associated precipitates). This phytoremediative process is known as phytostablization (Davis, 1995; and Skousen, 1998). In addition, decomposing plants provide organic matter to which metals readily bind, once again impeding their bioavailability.

Since the Burnside engineered wetland was intended to accommodate passive treatment, it was decided that the most effective treatment method for this constituent would be to facilitate settling, accommodated by a large, settling pond at the beginning of the wetland. Iron removal would also be facilitated by establishing dense, deep rooting vegetation which would act to immobilize the oxidized iron through phytostabilization and extract reduced iron from the deeper, anaerobic sediments.

The iron concentrations observed in the treatment wetland water quality analysis generally decreased through the treatment wetland cells. However, from 6 out the 8 of the sampling events, the iron concentration actually increased in Cell 3 from Cell 2. Observably, much floc was present in both Cell 2 and Cell 3, however, due to the cell layouts, access to the deeper surface waters in Cell 3 was constrained. Subsequently, it was difficult no to stir up bottom sediments or disturb floc deposited on vegetation when collecting the samples from Cell 3. Thus, more iron particulate may have been collected in these samples than was actually represented in the surface water, skewing results. However, the most significant factor which likely contributed to the increased iron concentrations observed in Cell 3 was inadequate settling occurring in Cells 1 and 2 due to significant breaks in the vegetated berms which reduced retention times (Figure 6.22). Although several berm reparation attempts were made, the power of flows would ultimately reopen the channels in the sensitive, newly repaired and planted areas.

Although overall, iron removal did occur, the iron concentrations observed in the outlet effluent never dropped below the CCME Guideline of 0.3 mg/L for the Protection of Freshwater Aquatic Life (CCME, 2001). The lowest iron effluent concentration observed was 0.67 mg/L, the highest was 9.38 mg/L, and the average iron concentration in the effluent was 5.17 mg/L. The amount of total iron observed in the surface water in the site overall was observably affected by rain events which diluted the leachate (ie. June 30, August 6). Also, over the course of sampling, the iron concentrations in Cell 2, 3 and the outlet fluctuated drastically, decreasing throughout the summer, then increasing again in August and September as rain likely brought more iron into the treatment wetland.

6.5.1.2. Manganese. Manganese was also identified as one of the contaminants of concern in the Burnside landfill leachate effluent needing treatment. Manganese (Mn) is a transition metal which is grey, white or silver in colour. It is soft and ductile if pure but usually occurs in compounds and complexes with organic compounds (Sample et al., 1997). Manganese is an essential micronutrient forming a vital part of the enzyme systems that metabolise proteins and energy in all animals. In surface water, natural levels are usually less than 0.2 mg/L and rarely exceed 1.0 mg/L, but it can be naturally present at levels as high as 40 mg/L (BCMELP, 2001). Natural sources of manganese include metamorphic and sedimentary rocks, mica biotite and amphibole horneblende minerals (Sample et al., 1997).

In freshwater systems, manganese is present as either soluble, reduced manganous (Mn_2^+) or less soluble oxidized manganic (Mn_4^+) (Boulton and Brock, 1999). Reduced manganese is considered to be the most toxic form of the metal as it is readily bioavailable (BCMELP, 2001). High manganese concentration in freshwater is reported to cause fish gill damage in similar fashion as that of iron. It is also known to cause decreases in white and red blood cells, haemoglobin, and protein concentrations in fish (Sample et al., 1997). In general, manganese is known to be only

(a) Break in Berm 1.

(b) Breaks in Berm 2.

Figure 6.22. Breaks in the Wetland Berms.

slightly to moderately toxic to aquatic organisms in excessive amounts. Several factors such as salinity, pH, and the presence of other contaminants can affect the toxicity of manganese in waterbodies. However, water hardness appears to be the most influential factor affecting manganese toxicity. As hardness increases, manganese toxicity decreases (BCMELP, 2001; and Boulton and Brock, 1999).

Previous studies have showed that constructed wetlands are less successful in removing manganese (Stillings et al., 1988; Fennessy and Mitsch, 1989; Hedin, 1989; Skousen et al., 1994; and Stark et al., 1994). This is mostly due to the fact that oxidized manganese does not readily form precipitate unless the pH of waters are at least 7.0, with precipitation most commonly occurring in waters with pHs greater than 10.0. In addition, manganese oxidation is sensitive to the presence of Fe^{2+}, which can prevent or even reverse manganese oxidation (Skousen, 1998). Phytoremediation mechanisms prove relatively insignificant, accounting for approximately 1 to 4% of manganese removal (Ye et al. 2001a). Fortunately, the pH of the leachate wastewaters at the Burnside site are not acidic, hence some removal via precipitation may be accommodated in the treatment wetland. In addition, the facilitation of sedimentation via the use of settling ponds and vegetation can be an effective means of immobilization in the more turbid waters of the Burnside leachate influent (BCMELP, 2001). Some treatment site managers try to force manganese precipitation (i.e. MnO_2) by increasing the alkalinity of the treatment site via limestone addition. However, this is a temporary solution which is difficult to maintain without regular adjustment (McMillen et al., 1994). Ultimately, the most effective removal mechanism for manganese in treatment wetland systems is settling, as 90 to 95% of total waterborne manganese binds to particulate matter (BCMELP, 2001).

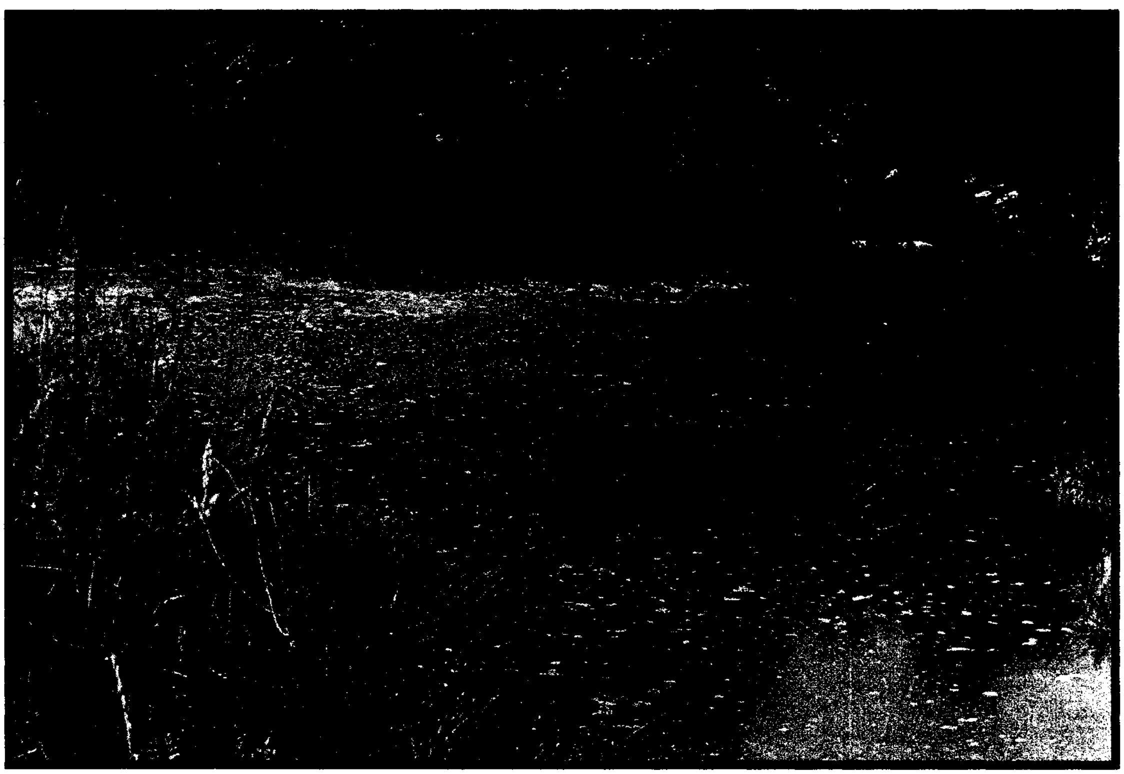
The manganese concentrations in the leachate input generally did not decrease as it flowed through the treatment wetland cells. Although the CCME currently does not publish water quality guidelines for manganese, the British Columbian Ministry of Environment, Land and Parks (BCMELP) have published acute and chronic manganese guidelines for the protection of freshwater aquatic life. The manganese concentrations observed in the outlet effluent exceeded the BCMELP acute guideline of 1.914 mg/L on the August 22nd sampling date and exceeded the BCMELP chronic guideline of 1.15 mg/L on May 23, June 1, June 30, August 25 and September 11. Also, over the course of the sampling season, the manganese concentrations in Cell 2, 3 and the outlet remained relatively stable, showing little to no improvement as site vegetation matured. Mitigative measures must be taken to improve the treatment wetland ability to remove manganese from the leachate influent. Since settling is a major mechanism of its removal, one of the most significant factors which likely contributed to the inadequate settling in Cells 1 and 2 was the significant breaks in the vegetated berms which drastically reduced retention times. In addition, preliminary wetland sizing calculations indicated that the size of wetland required to treat the manganese received by the Burnside treatment system would be about 0.5 ha, which is substantially larger than the current wetland size. Additionally, the amount of total manganese observed in the surface water in the site overall was markedly affected by the heavy rain events occurring in August and September.

6.5.1.3. Nitrite. In aerobic conditions, most ammonium is oxidized to nitrite by *Nitrosomonas* bacteria (Boulton and Brock, 1999). Nitrite is an inorganic compound of nitrogen that is very unstable and typically oxidizes to nitrate rapidly. It has been termed an invisible killer as it has no visible affect on the water column, but is often toxic to fish at low concentrations. The presence of excessive nitrite also depletes dissolved oxygen in the water column as the oxidation of nitrite to nitrate consumes oxygen (Mason, 1998).

The CCME nitrite water quality guideline for the protection of freshwater life is 0.06 mg/L (CCME, 2002). Overall, the nitrite levels observed in the treatment wetland were minimal until the July 21st sampling date, where the nitrite concentration in the outlet was 0.59 mg/L, which exceeds the CCME Guidelines. In addition, the nitrite concentration in Cell 2 and the outlet taken on August 6th were 0.24 mg/L and 0.07 mg/L respectively, which also exceed the CCME Guideline. The exceedances observed on the August 6th date may be correlated with the heavy rain events occurring during and previous to the day of sampling as stormwaters often carry excess nutrients (Bennet, 1994). The high nitrite levels observed in the outlet on the July 21st sampling date may be correlated with the evidence of stagnation observed in the site that day (Figure 6.23). Nitrite can persist in waters which suffer from oxygen depletion (Mason, 1998).

6.5.1.4. Nitrate. In aerobic conditions, nitrite is typically rapidly oxidized to nitrate by *Nitrobacter* bacteria. Nitrate is an inorganic compound of nitrogen which is bioavailable for plant uptake and is essential to plant growth (Boulton and Brock, 1999; and Freedman, 2001). Natural levels of nitrate in waterbodies are typically lower than 1 mg/L. Where nitrite and ammonia are toxic, nitrate is virtually harmless, with direct toxic effects typically not observed until concentrations greater than 1000 mg/L (Mason, 1998). However, if phosphorus concentrations

(a) Cell 2.



(b) Cell 3.

Figure 6.23. Stagnated Surface Waters, July 21, 2003.

are sufficient, high nitrate content in waters can increase the severity of eutrophication, which can have chronic effects of aquatic life.

There are no CCME nitrate water quality guidelines for the protection of aquatic life. However, the BCMELP (1986) guideline for the protection of freshwater aquatic life is 40 mg/L. Overall, the nitrate levels observed in the treatment wetland were minimal. Concentrations generally decreased during the course of the summer. This is a likely result of increased nitrate uptake by the aquatic vegetation as the plants matured and spread over the growing season.

Ammonia. Ammonia was identified as one of the contaminants of concern in the Burnside landfill leachate effluent. Ammonia nitrogen is a pungent, gaseous compound of nitrogen and hydrogen and includes both ammonia (NH_3) and ammonium ion (NH_4^+). More common in landfill leachates is the ionized form NH_4^+ which is formed when NH_3 is combined with water at pH less than 8.5 and low temperatures to produce an ammonium ion and a hydroxide ion (OH^-). Ammonia concentrations are typically less than 0.1 mg/L in natural waters. It is notably very soluble, one volume of water will dissolve 1,300 volumes of ammonia (CCME, 2000). Although concentrations will vary given the specific nature of decomposing wastes and the age of the landfill.

Ammonium is fairly harmless, whereas ammonia (the most toxic by-product of the nitrogen cycle) can be deadly at high levels, especially to fish. Ammonium and ammonia shift in equilibrium with each other according to pH. At the pH of 8.5 or greater, ammonia will be the more dominant form. Increased water temperatures also harbour the unionized form. Consequently, as pH or temperature increase, the proportion of un-ionized ammonia increases, resulting in increased toxicity (USEPA, 1995). Acute effects of fish exposure to ammonia include convulsions, coma, and death. Chronic effects of exposure to ammonia include damage to gill membranes, preventing fish from carrying on normal respiration, alteration of metabolisms or increases in body pH. Ammonia can cause osmoregulatory damage and is an irritant to delicate tissues such as the internal organs. Even trace amounts of ammonia can stress fish, suppressing the immune system. This can cause a gradual breakdown of fish health, developmental problems, and disease outbreaks (USEPA, 1998a).

In both natural and treatment wetland ecosystems, ammonia nitrogen removal occurs through oxidation/reduction reactions, volatilization into the atmosphere and plant uptake. However, the most effective removal of ammonia is not by wetland vegetation, but by aerobic microbes and can therefore be limited in saturated anaerobic areas. Although not the primary removal mechanism, wetland vegetation proves invaluable to ammonia removal in wetland systems as it facilitates microbial growth by oxygenating soils and surface water, and by providing microhabitat and surface area for microbial attachment (Knight et al., 1986, Osmund et al., 1995b; USEPA, 2000c; and USEPA, 1993). Ammonia removal by volatilization is generally insignificant unless the pH of the water is greater than 8.5, as ammonium, which becomes the dominant form of the compound in waters with pH less than 8.5, is not volatile. Additional measures which are occasionally undertaken in treatment wetland to improve the efficiency of ammonia removal involve actively reducing the pH and water temperature of treatment wetlands by liming and shading (USEPA, 1995).

The CCME guidelines for the protection of aquatic life (2001) shown in Table 6.1 reflect when NH_3 becomes the dominant form of ammonia and therefore becomes toxic. The ammonia concentrations in the leachate input generally decreased as it flowed through the treatment wetland cells. None of the ammonia concentrations observed in the waters of the treatment wetland exceeded the CCME guidelines and ammonia was actually not detected in 37% of the samples analysed. The concentrations of ammonia did not appear to be particularly affected by rain events. Also, the ammonia concentrations in the cells appeared to increase slightly over the course of the summer. This could be due to the conversion of organic nitrogen to ammonia nitrogen.

Table 6.1. CCME Ammonia Guidelines (mg/L) for Protection of Aquatic Life (adapted from CCME, 2001).

pH	TEMPERATURE °C						
	0	5	10	15	20	25	30
6.0	231.00	153.00	102.00	69.70	48.00	33.50	23.70
6.5	73.00	48.30	32.40	22.00	15.20	10.60	7.50
7.0	23.10	15.30	10.30	6.98	4.82	3.37	2.39
7.5	7.32	4.84	3.26	2.22	1.54	1.08	0.77
8.0	2.33	1.54	1.04	0.72	0.50	0.35	0.26
8.5	0.75	0.50	0.34	0.24	0.17	0.13	0.09

6.5.1.5. TKN. Total Kjeldahl Nitrogen (TKN) is the sum of both ammonia and organic nitrogen. The combination of the organic nitrogen and all forms of inorganic nitrogen (NH_4, NO_3, NO_2) make up total nitrogen. Natural levels of TKN in waterbodies are typically less than 2.0 mg/L (Cameron et al., 2003). Sources of TKN include the decay of organic material such as plants and animals wastes, as well as anthropocentric sources such as sewage and agricultural runoff. Organic nitrogen is relatively immobile in water and soils because it readily binds to particulate. However, ammonia is susceptible to nitrification under aerobic conditions, hence removal is best facilitated through and oxygenation of the waterbody and uptake by abundant aquatic plant life (Mason, 1998).

There are no specific TKN water quality guidelines for the protection of aquatic life. However, concentrations above 3 mg/L are considered excessive in natural waters (Cameron et al., 2003). The TKN concentrations observed in the treatment site ranged from 0.0 to 19.0 mg/L, with an average of 4.98 mg/L. The TKN did not decrease as the waters moved through the cells. Concentrations were notably high in Cell 3 for the May 24th to June 30th sampling dates. The organic forms of nitrogen measured in TKN analyses include nitrogen that is bound to algae. Commonly, sites which support prolific algae contain elevated levels of TKN (Mason, 1998; and Boulton and Brock, 1999). Algal blooms were developing and decaying in the treatment site during the period when TKN levels in the water samples appeared exceptionally high.

6.5.1.6. Orthophosphate. Phosphorus is an essential macronutrient that is a limiting factor to plant growth. It is essential to all life as a component of nucleic acids and a universal energy molecule (Lee and Jones-Lee, 2001). In excess, phosphorus triggers eutrophic conditions which involves the prolific growth of algal and other aquatic plants. Algal growths can have lethal impacts on aquatic life and, at high concentrations, can be toxic in itself. The absorption of sunlight by algal blooms reduces amount of light reaching aquatic plants in sediment. If an algal bloom is prolonged, aquatic plants will die. Large amounts of decaying algae result in the consummation of large quantities of oxygen by the bacteria and fungi that break it down. This results in the dramatic reduction of oxygen concentrations in the water column, particularly at night. This reduction affects invertebrate predators with high oxygen requirements. The subsequent lack of predators results in critical disruptions in the food chain and increases of nuisance species such as mosquitoes. Algal blooms can also contain toxic strains of blue-green algae which may kill birds, domestic animals, aquatic macroinvertebrates and even humans if consumed (Lee and Jones-Lee, 2001; Sharpley et al, 1994; and Vollenweider, 1968).

In waters, phosphorus is often biologically unavailable as it binds readily to particles. Soluble phosphorus which is available for uptake is called orthophosphate. Orthophosphate concentrations in the treatment wetland waters were analysed in order to identify the cause of the prolific algal blooms occurring in Cells B and C of the site in early May of 2003. Smaller blooms were also occurring along the southern edges of Cells 2 and 3. The orthophosphate concentrations in the waters of Cell 1, 2, 3, and the outlet where minimal, fluctuating between undetectable levels to 0.03 mg/L. Since the large bloom occurred in the latter cells of the site, it was hypothesized that the phosphorus input was from leakage occurring under the road into Cell A. However, sampling of the leaking waters revealed low orthophosphate concentrations of 0.02 mg/L. According to Boulton and Brock (1999), low concentrations of orthophosphate are not necessarily reflective of conditions as it is rapidly taken up by water plants and microorganisms. Hence, despite the findings of the water quality results, it remains likely that the primary source of the phosphorus is the leaking waters beneath Akerley Boulevard for had the influent been the source, prolific algal blooms would have occurred in the earlier cells of the site as well. This loading appears to only occur once every spring, as the algal blooms have been observed to disappear by June.

6.5.1.7. Total Suspended Solids. Suspended solids were identified as one of the contaminants of concern in the Burnside landfill leachate effluent. Total suspended solids (TSS) include all particles suspended in water that will not pass through a filter. Abundant suspended solids such as clay and silt, fine particles of organic and inorganic matter (such as iron particulate), soluble coloured compounds and phytoplankton can result in: (a) decreased light penetration in water reducing photosynthesis of water plants, (b) decreased water depth due to sediment build-up, (c) the smothering of aquatic vegetation, habitat and food, (d) the smothering of macro and micro-organisms, larva, eggs and the clogging of fish gills, (e) the reduced efficiency of predation by visual hunters, and (f) increased heat absorbed by the water, lowering dissolved oxygen, facilitating parasite and disease growth and increasing the toxicity of ammonia (Mason, 1998; and Boulton and Brock, 1999).

Suspended solid removal in constructed wetlands is best facilitated through the encouragement of settling. Effective settling in treatment wetland systems is most commonly accommodated by the creation of a settling pond or a forebay at the head of the wetland which is larger and deeper than the rest of the cells in the system. They are typically designed to support long retention times in order to allow the suspended solids and other debris to settle out. The increased depth accommodates sediment build up, reducing the need for frequent dredging (Tousignant et al., 1999). Densely vegetated wetland sites facilitate the settling of suspended matter by obstructing water flows, stabilizing bottom sediments and physically filtering and trapping the matter (Davis, 1995).

There are no specific CCME Water Quality Guidelines for TSS. The TSS concentrations in Cell 2, Cell 3 and the outlet remained relatively stable over the course of the sampling season, showing little improvement as sites vegetation matured. The most likely explanation for this result is the ineffective facilitation of settling due to the breaks in the wetland berms which caused dramatic reductions in retention times.

6.5.1.8. Total Dissolved Solids. Total dissolved solids (TDS) is a measure of the concentration of dissolved constituents in water, which commonly include carbonate, bicarbonate, chloride, sulfate, phosphate, nitrate, calcium, magnesium, sodium, organic ions, and other ions. A certain level of these ions in water is essential nutrients for aquatic life. Changes in TDS concentrations can be harmful to aquatic organisms by affecting the density of water. Excessive TDS can reduce water clarity, hinder photosynthesis, and lead to increased water temperatures (Mason, 1998; and Boulton and Brock, 1999).

There are no specific CCME Water Quality Guidelines for TDS. The TDS concentrations in Cell 2, Cell 3 and the outlet remained relatively stable over the course of the sampling season, showing little improvement as site vegetation matured. The most likely explanation for this result is reductions in retention times experienced by the site as a result of breaks in the berms.

6.5.1.9. Chemical Oxygen Demand. Chemical oxygen demand (COD) is a measure of the amount of oxygen required to chemically oxidize reduced minerals and organic matter. It does not differentiate between biologically available and inert organic matter. Typical concentrations of COD in domestic sewage range from 200-1000 mg/L (Mason, 1998). In landfill leachate, typical COD concentrations range from 22,000 - 35,000 mg/L in the first 3 years of decomposition, then tend to drop to concentrations ranging from 400 - 800 mg/L. In general, the greater the COD value in water, the more oxygen the influent demands from the waterbody, thus resulting in depleted dissolved oxygen which is essential to the metabolism of all aerobic aquatic organisms (Silva et al., 2003; Cusso et al., 2001; and CCME, 1999b).

There are no CCME COD standards for the protection of aquatic life. However, the COD concentrations observed in the site are notably higher (averaging about 857.5 mg/L) than the

COD concentrations of domestic sewage. COD concentrations did not appear to decrease as the water flowed through the site.

6.5.1.10. Dissolved Oxygen. Dissolved oxygen is one of the most fundamental parameters in water, as it is essential to the metabolism of all aerobic aquatic organisms. It is added to the water column via photosynthesizing plants and stream flow aeration, and is consumed from the waterbody by bacterial, plant and animal respiration, decaying plants and organisms, and chemical oxidation (CCME, 1999b).

The CCME guideline for dissolved oxygen for the protection of freshwater aquatic life is between 5.5 and 6.0 mg/L for warm water ecosystems such as the treatment wetland (CCME, 2001). With the exception of the May 24th sampling date, virtually all samples analysed were below the minimum guideline of 5.5 mg/L. CME, 1999b). The dissolved oxygen concentrations in the treatment wetland waters generally decreased as they flowed through the treatment wetland cells until late June when the concentrations appeared to drop slightly and level off. This was unexpected as it had been assumed that as the water flowed through the cells it would become more oxygenated as a result of the growing vegetation and water purification. Low dissolved oxygen levels are often the result of organic pollution (Mason, 1998). In seeing that colder waters have greater oxygen capacity, the decreasing dissolved oxygen concentrations may be correlated with water temperatures which were greater than 15^{o}C for the June 30 to September sampling dates (CCME, 1999b). The low dissolved oxygen concentrations may also be explained by the abundant iron ochre observed in the site as the physical oxidation of iron consumes oxygen (Lee and Jones-Lee, 2001; and Fripp et al., 2000).

6.5.1.11. pH. Exceedances of pH guidelines have been associated with many adverse effects. However, one of the most significant impacts of pH in waterbodies is the effect that it has on the solubility and thus the bioavailability of other substances such as iron, manganese and ammonia as discussed (CCME, 1999a). The pH of the treatment wetland remained relatively neutral, fluctuating between 6.7 and 7.7 with an average of 7.0. No pH levels were observed above or bellow the CCME pH guidelines for the protection of freshwater aquatic life of less than 6.5 and greater than 9.0 (CCME, 2002). No general increasing or decreasing of the pH was observed as the waters flowed through the site, although a general decrease in pH appears to have occurred over the course of the summer as the system matured. No correlation was observed between the metal levels and pH levels, rain events, or any other constituent.

6.5.2. Plant Tissue Analysis

Water quality analysis is an essential component of any constructed wetland monitoring program. However, this type of assessment only indicates whether or not parameters have increased or decreased, and does not indicate what mechanisms were responsible for any observed changes. As discussed previously, the major mechanisms of contaminant removal in both natural and constructed wetland systems are sedimentation, biogeochemical reactions, bioremediation, and phytoremediation. Since the aim of this study was to establish a native vegetative community that would help decontaminate the iron, manganese and nutrient-rich leachates received by the treatment wetland system, it seemed fitting to test the phytoremediative capability of the plants established in the site.

The species analysed from the treatment wetland which contained the highest concentration of iron in its roots was the tweedy's rush (*Juncus brevicaudatus)* at 17302 mg/kg, followed by broad-leaved cattail (*Typha latifolia*) and soft stem bulrush (*Scirpus validus*) at 14171 and 13903 mg/kg, respectively. The species analysed from the treatment wetland which contained the highest concentration of manganese in its roots was the tweedy's rush at 447 mg/kg, followed by pickerelweed (*Pontederia cordata)* and soft rush (*Juncus effusus*) at 269 and 217 mg/kg, respectively.

The species analysed from the treatment wetland which contained the highest concentration of iron in its stems was the broad-leaved cattail at 5788 mg/kg, followed by soft stem bulrush and tweedy's rush at 813 and 461 mg/kg, respectively. The species analysed from the treatment wetland which contained the highest concentration of manganese in its stems was the fowl mannagrass (*Glyceria striata*) at 309 mg/kg, followed by woolgrass (*Scirpus cyperinus*) and broad-leaved cattail at 289 and 277 mg/kg, respectively.

The species analysed from the treatment wetland which contained the highest concentration of iron in its leaves was the fringed sedge (*Carex crinita*) at 3936 mg/kg, followed by fowl mannagrass and yellow-green sedge (*Carex lurida*) at 1002 and 809 mg/kg, respectively. The species analysed from the treatment wetland which contained the highest concentration of manganese in its leaves was the fowl mannagrass at 1480 mg/kg, followed by broad-leaved cattail and pickerelweed at 730 and 555 mg/kg, respectively.

The species analysed from the treatment wetland which contained the highest concentration of iron in its flowers was the soft stem bulrush at 4222 mg/kg, followed by the tweedy's rush and broad-leaved cattail at 1714 and 1466 mg/kg, respectively. The species analysed from the treatment wetland which contained the highest concentration of manganese in its flowers was the soft stem bulrush at 778 mg/kg, followed by broad-leaved cattail and tweedy's rush at 467 and 443 mg/kg, respectively.

Cumulatively, the highest iron accumulator was broad-leaved cattail at 21598 mg/kg, followed by tweedy's rush and soft stem bulrush at 19477 and 18938 mg/kg, respectively. Cumulatively, the highest manganese accumulator was the fowl mannagrass at 2261 mg/kg, followed by broad-leaved cattail and soft stem bulrush at 1601 and 1202 mg/kg, respectively.

For all species analysed, iron concentrations were highest in plant roots, while manganese concentrations were higher in plant flowers or leaves. In addition, significantly higher concentrations of iron were present in the analysed plant tissues than manganese, which for the most part was accounted for by dramatically high iron concentrations in the plant roots. Plant roots remove metals via two mechanisms: absorption and adsorption. Absorption involves the actual uptake of the metal (phytoextraction), whereas adsorption involves the immobilization of the metal via precipitation of the metal onto the roots (phytostabilization). This precipitation occurs because wetland plants translocate oxygen from the aerial portions of the plants to the root and rhizomes which leaks creating a thin oxidized layer around the surface of the roots known as the rhizosphere (Skousen et al., 1994; Hansel et al, 2002; and Matagi et al., 1998). At a pH greater than 3.5 with oxygen present, soluble and bioavailable of ferrous oxide will oxidize to insoluble ferric oxide which cannot be readily taken up (or absorbed) by plants. Under these circumstances, a coating of ferric oxide precipitate (or plaque) will accumulate on the outside of the roots. This iron coating can account for 32-93% of metal concentration in root tissue analyses. Consequently, some metal concentrations for the roots are quite high when compared to the other plant parts (Mays and Edwards, 2000; Taylor and Crowder, 1983; and Shutes et al., 1993). Since the pH of the waters in the Burnside treatment wetland are neutral, most of the iron which came in contact with the plants oxygen-rich rhizospheres likely oxidized, adhered (or adsorbed) to the outside of roots, and was subsequently not absorbed by the plant. The main mechanism for iron removal in the wetland plants analysed was therefore likely phytostabilization. Conversely, manganese will often not form precipitate unless the pH occurring in waters is greater than 10.0 (Skousen et al., 1994 and Ye et al. 2001a). In addition, the presence of reduced iron can prevent or even reverse manganese oxidation (Skousen, 1998). Manganese, in its soluble, reduced phase is very bioavailable and is readily and rapidly absorbed and translocated into aerial plant tissues such as the leaves and flowers (Kabata-Pendias and Pendias, 1992; and Ye et al., 2001a). Therefore, it is likely that the main mechanism for manganese removal in the wetland plants analysed was phytoextraction.

6.5.2.1. Woolgrass (*Scirpus cyperinus*). Woolgrass was selected as a species to dominate the treatment wetland because it was abundant in the model site which is impacted by the leachate input and because the results of the phytoremediative screening of the model site plant list indicated that this bulrush had high phytoremediative potential. According to Ye et al. (2001b), Campbell and Ogden (1999), Tousignant et al. (1999), Demchik and Garbutt (1999), and Mays and Edwards (2000), this species was utilized in several successful Acid Mine Drainage (AMD) treatment wetlands which treated heavy metals and ammonia. However, none of these studies examined tissue concentrations as part of their analyses.

Of the 10 plant species analysed, woolgrass, was the 7th highest accumulator of iron, and the 9th highest accumulator of manganese. The roots contained the largest concentrations of iron, and the stem contained the largest concentrations of manganese. The same was observed of the iron concentrations in the woolgrass from the control site. However, it was the flower, not the stem that held the highest concentration of manganese in the control site sample. Regardless, these results are fitting with the mobility behaviour of iron and manganese as described by Mays and Edwards (2000), Taylor and Crowder (1983), Shutes et al. (1993), Skousen et al. (1994), Ye et al. (2001a) and Kabata-Pendias and Pendias (1992).

It had been expected that the woolgrass highly rhizomous roots would be more conducive to iron adsorption than the cattail which supports less rhizosphere in terms of surface area. However, this was not the case. Although on a mg/kg basis, the woolgrass roots did not fair as well as the cattail roots in terms of metal accumulation, woolgrass roots penetrate up to 3 times the depth and support several times the biomass as cattail roots (Campbell and Ogden, 1999). Hence, ultimately, a treatment wetland dominated woolgrass plants versus a treatment wetland dominated by the same number of cattail plants may likely remove more iron and manganese overall.

6.5.2.2. Soft rush (*Juncus effusus*). Soft rush was also selected as a species to dominate the treatment wetland site because it was abundant in the model site and the results of the phytoremediative screening of the model site plant list indicated that this rush had high phytoremediative potential. According to Ye et al. (2001b), Younger and Batty (2002), Coleman et al. (2001), Treacy and Timpson (1999) and Mays and Edwards (2000), this species was utilized in several successful Acid Mine Drainage (AMD) treatment wetlands which treated heavy metals and ammonia. However, none of these studies examined tissue concentrations as part of their analyses.

Of the 10 plant species analysed, soft rush was the least accumulator of iron, and the 8th highest accumulator of manganese. The roots of this plant contained the largest concentrations of iron, and the flower contained the largest concentrations of manganese. The same was observed of the iron and manganese concentrations in the soft rush from the control site. These results are fitting with the mobility behaviour of iron and manganese as described by Mays and Edwards (2000), Taylor and Crowder (1983), Shutes et al. (1993), Skousen et al. (1994), Ye et al. (2001a) and Kabata-Pendias and Pendias (1992).

The results of the soft rush tissue analysis were unexpected as the literature review led to the presumption that this species would be one of the most effective iron and manganese phytoremediators which was not the case in the site. However, the species should not be discounted as a candidate for application in the field of metal-contaminated wastewater remediation although it was not as successful as the other species analysed, because it did accumulate significant amounts of iron and manganese. Like the woolgrass, soft rush also has large, deeply penetrating roots (Campbell and Ogden, 1999). Taking its root biomass into account, the soft rush still likely contributes significantly to iron and manganese removal in the treatment wetland site (USEPA, 2000b).

6.5.2.3. Pickerelweed (*Pontederia cordata*). Pickerelweed was selected as a candidate species for the treatment wetland site because it grows profusely around the edges of Enchanted Lake which is affected by the leachate input, and because it has high evapotranspiration rates, which is typically indicative of high phytoremediation potential (Tousignant, 1999). It is also a close relative to water hyacinth; an exotic and highly invasive species which has been proven to be very effective at contaminant removal and has realized several wastewaters treatment applications (Brix, 1993).

Of the 10 plant species analysed, pickerelweed was the 4th highest accumulator of iron, and the 5th highest accumulator of manganese. The roots of this plant contained the largest concentrations of iron, and the leaves contained the largest concentrations of manganese. The same was observed of the iron and manganese concentrations in the pickerelweed from the control site. These results are fitting with the mobility behaviour of iron and manganese as described by Mays and Edwards (2000), Taylor and Crowder (1983), Shutes et al. (1993), Skousen et al. (1994), Ye et al. (2001a) and Kabata-Pendias and Pendias (1992). Thus, in addition to be a very aesthetically pleasing species, pickerelweed is contributing significantly to iron and manganese removal in the treatment wetland site. However, due to its small biomass, significantly more individuals would likely required to remove the same amount of metals as some of the larger species such as the woolgrass or cattail (USEPA, 2000b).

6.5.2.4. Fowl mannagrass (*Glyceria striata*). Fowl mannagrass was chosen as a candidate for the tissue analysis because it was common in the model site and because it established quite readily in the treatment wetland site, mostly from seed, becoming one of the more dominant species in the berms. In addition, many grass *(Poaceae)* species have been documented to exude a class of organic compounds termed siderophores (mugenic and avenic acids) capable of enhancing the availability of iron for uptake, however, this typically occurs in areas of iron deficiency, not iron abundance (Johnson et al., 2002, Jones, 1998; and Roemheld and Awad, 2000). A review of the literature revealed no information concerning fowl mannagrass's or any other *Glyceria* species' ability to uptake metals.

Of the 10 plant species analysed, fowl mannagrass was the 8th highest accumulator of iron, and the highest accumulator of manganese. The roots contained the largest concentrations of iron, and the leaves contained the largest concentrations of manganese. The same was observed of the iron and manganese concentrations in the fowl mannagrass from the control site. These results are fitting with the mobility behaviour of iron and manganese as described by Mays and Edwards (2000), Taylor and Crowder (1983), Shutes et al. (1993), Skousen et al. (1994), Ye et al. (2001a) and Kabata-Pendias and Pendias (1992). Iron uptake into the aerial portions of this species was limited, indicating that fowl mannagrass does not likely employ the use of sideophores as a major mechanism for iron absorption (Lasat, 2002).

The superior manganese accumulation ability of all the species tested was unanticipated. Clearly, the fowl mannagrass contributes significantly to manganese removal in the treatment wetland site. One possible explanation for the high concentrations of manganese found in the fowl mannagrass tissues as compared to the other plants analysed from the treatment site lies in the fact that the vast majority of the fowl mannagrass present in the treatment site was established from seed. Manganese preferentially transports to meristematic tissues, which are the young, expanding tissues in plants. As the fowl mannagrass grew from seed, the plant tissues analysed would have supported virtually all young, new tissues, as opposed the majority of the other plants analysed (with the exception of the cattail and the reed canary grass), which were transplanted into the site as mature plants. It is expected that the manganese accumulation ability of this plant may decline as its tissues mature over consecutive growing seasons (Kabata-Pendias and Pendias, 1992). However, the mature fowl mannagrass tissues analysed from the control site still supported the highest concentrations of manganese, (however less dramatically) of all the control species analysed.

6.5.2.5. Yellow-green sedge (*Carex lurida*). Yellow-green sedge was chosen as a candidate for the tissue analysis because it is common in the model site which is impacted by the leachate input, because it established readily in the treatment wetland site, and because *Carex* species are renowned for their ability to uptake metals (Sanders et al., 2000; Kastning-Culp et al., 1993; Haberl et al., 1999; and Campbell and Ogden, 1999). In addition, yellow-green sedge, along with the fringed sedge, supports a relatively small biomass. According to the USEPA (2000b) and Kamnev and van der Lelie (2000), most plants which hyperaccumulate metals on a significant level are small in size. A review of the literature revealed no specific studies on the metal uptake ability of this particular species.

Of the 10 plant species analysed, yellow-green sedge was the 6th highest accumulator of iron, and the 7th highest accumulator of manganese. The roots contained the largest concentrations of iron, and the flower contained the largest concentrations of manganese. Although, it was the leaves and not the flower that held the highest concentration of manganese in the control site sample, the results are fitting with the mobility behaviour of iron and manganese as described by Mays and Edwards (2000), Taylor and Crowder (1983), Shutes et al. (1993), Skousen et al. (1994), Ye et al. (2001a) and Kabata-Pendias and Pendias (1992). Although not as successful as some of the other species analysed, this sedge did hyperaccumulate significant amounts of iron and manganese and still likely contributes significantly to iron and manganese removal in the treatment wetland site. However, due to its small biomass, significantly more individuals would likely required to remove the same amount of metals as some of the larger species such as the woolgrass or cattail (USEPA, 2000b).

6.5.2.6. <u>Fringed sedge (*Carex crinita*)</u>. Like the yellow-green sedge, fringed sedge was selected as a candidate for the tissue analysis because it is common in the model site, it established readily in the treatment wetland site, because *Carex* species are renowned for their ability to uptake metals, and because it supported a relatively small biomass (Sanders et al., 2000; Kastning-Culp et al., 1993; Haberl et al., 1999; Campbell and Ogden, 1999; USEPA, 2000; and Kamnev and van der Lelie, 2000). However, a review of the literature revealed no specific studies on the metal uptake ability of this particular species.

Of the 10 plant species analysed, fringed sedge was the 9th highest accumulator of iron, and the least highest accumulator of manganese. The leaves contained the largest concentrations of iron, and the flower contained the largest concentrations of manganese. Conversely, the iron and manganese concentrations were both highest in the roots of the fringed sedge from the control site. The iron behaviour in the fringed sedge from the treatment site, and the manganese behaviour in the fringed sedge from the control site is not fitting with the mobility behaviour of iron and manganese as described by Mays and Edwards (2000), Taylor and Crowder (1983), Shutes et al. (1993), Skousen et al. (1994), Ye et al. (2001a) and Kabata-Pendias and Pendias (1992). (Kabata-Pendias and Pendias, 1992; and Ye et al., 2001a). A review of the literature revealed no insight as to why these inconsistencies may have incurred.

Overall, the fringed sedge was one least effective metal accumulator of the 10 species analysed. These results were unexpected as the literature review led to the presumption that the *Carex* species analysed would be very effective iron and manganese removers. In addition, according to the USEPA (2000b) and Kamnev and van der Lelie (2000), most plants which hyperaccumulate metals on a significant level support small biomass. However, this is not fitting with the results of this analysis as the higher tissue concentrations of iron and manganese were found in species supporting large biomass such as the cattail, soft stem bulrush and fowl mannagrass.

6.5.2.7. <u>Soft stem bulrush (*Scirpus validus*).</u> Although not present in abundance in the model site or the treatment site, this species was selected as a candidate species for the tissue analysis because Mitchell and Karathanasis (1995) observed that soft stem bulrush was an effective hyperaccumulator of metals and was actually more effective at metal removal than cattail. However, metals tested in the analysis were cadmium, chromium, copper, nickel, lead, and zinc, and did not include iron or manganese.

Of the 10 plant species analysed, soft stem bulrush was the 3rd highest accumulator of both iron and manganese. The root contained the largest concentrations of iron, and the flower contained the largest concentrations of manganese. These results correspond with the mobility behaviour of iron and manganese as described by Mays and Edwards (2000), Taylor and Crowder (1983), Shutes et al. (1993), Skousen et al. (1994), Ye et al. (2001a) and Kabata-Pendias and Pendias (1992). Notably, the iron concentrations in the flower of the soft stem bulrush were exceptionally high at 4222 mg/kg. The second highest iron concentrations observed in a flower were nearly 60% lower than this figure (1714 mg/kg in the Tweedy's rush flower). In addition, in the control site specimen, it was the flower and not the roots that supported the highest concentrations of iron. Clearly, this plant effectively facilitates iron absorption. As stated, some plants can regulate metal solubility in the rhizosphere by exuding a variety of organic acids from roots (Johnson et al., 2002, Romheld and Marschner, 1986; and Kanazawa et al., 1995). For example, in order to gain access to unavailable phosphorus, some species exude these organic acids in order to mobilize iron-bound phosphates in areas of phosphorus deficiency (Shen et al., 2002; and Jones, 1998). In many of the water quality samples analysed from the treatment wetland cells, orthophosphate was not detected. Although a review of the literature revealed no insight as to which specific mechanisms are utilized by this particular family of plant, it is clear that some mechanism is at play which facilitates iron mobilization. Clearly, soft stem bulrush is a valuable species in terms of its metal extraction ability. The establishment of more individuals of this species in the treatment wetland cells should be considered.

6.5.2.8. Tweedy's rush (*Juncus brevicaudatus*). Tweedy's rush was not particularly common in the model site. However, this low-growing, inconspicuous rush was selected as a candidate species for the tissue analysis because it spread prolifically throughout the wetland, becoming the dominant plant in the site. A review of the literature revealed no specific studies on the metal uptake ability of this particular species. Of the 10 plant species analysed, tweedy's rush was the 2^{nd} highest accumulator of iron and 4^{th} highest accumulator of manganese. The root contained the largest concentrations of both iron and manganese, although the manganese concentration in the tweedy's rush flower was quite akin to that of the root (447 mg/kg versus 443 mg/L). Likewise, the manganese concentrations were highest in the flower of the tweedy's rush from the control site, hence, the result are fitting with the mobility behaviour of iron and manganese as described by Mays and Edwards (2000), Taylor and Crowder (1983), Shutes et al. (1993), Skousen et al. (1994), Ye et al. (2001a) and Kabata-Pendias and Pendias (1992).

This species is one of the most effective iron and manganese phytoremediators in the treatment site, which is an unexpected result, as a screening of its growth habits and phytoremediation potential indicated nothing of its ability. Obviously, the tweedy's rush contributes significantly to iron and manganese removal in the treatment wetland site.

6.5.2.9. Broad-leaved cattail *(Typha latifolia).* Broad-leaved cattail is accepted by the scientific community as a highly effective hyperaccumulator of metals and nutrients and consequently has seen wide-spread application in the field of wastewater treatment (Mitsch and Gosselink, 2000; Kadlec and Knight, 1996; Ye et al., 2001a; Snyder and Aharrah, 1985, Kleinmann, 1985; Coleman et al., 2001; Younger and Batty, 2002; and Huang et al., 1999). Despite its phytoremediation potential and abundance in the model site, broad-leaved cattail was not selected as a candidate species for the treatment wetland due to its characteristically aggressive behaviour, occasionally allelopathic nature, and abundant litter production (Campbell and Ogden, 1999; and Davis, 1996). Despite efforts to prevent its establishment, broad-leaved-cattail did establish sporadically throughout the treatment wetland site. This species was included in the tissue analysis in order to gauge how the phytoremediation ability of the native, non-aggressive species of the site compared in terms of their uptake ability to this notorious hyperaccumulator.

Of the 10 plant species analysed, broad-leaved cattail was the highest accumulator of iron and 2nd highest accumulator of manganese. The root contained the largest concentrations of iron and the leaves contained the largest concentrations of manganese, which is fitting with the mobility behaviour of iron and manganese as described by Mays and Edwards (2000), Taylor and Crowder (1983), Shutes et al. (1993), Skousen et al. (1994), Ye et al. (2001a) and Kabata-Pendias and Pendias (1992). Notably, however, the cattail had comparably high concentrations of iron in its stem, and the iron concentrations in the cattail from the control site were highest in its leaves. Overall, the broad-leaved cattail was the most effective iron absorber of all the species analysed, with the treatment wetland specimen holding 7427 mg/kg of iron in its aerial tissues. Hence, like the soft stem bulrush, some intrinsic mechanism must be at play which work to increases the mobility of iron within this species (Johnson et al., 2002, Romheld and Marschner, 1986; Kanazawa et al., 1995; Shen et al., 2002; and Jones, 1998).

In treatment wetlands designed to treat pH neutral, metal contaminated waters, Ye at al. (2001a) reported iron concentrations of up to 268 mg/kg in analyzed above-ground cattail tissues, and

8000 mg/kg in below ground cattail tissue. Manganese concentrations in analysed above-ground and below-ground cattail tissues were up to 2440 and 1650 mg/kg, respectively. The iron concentrations observed in the above-ground and below-ground tissues of cattail taken from the Burnside treatment wetland were 7427 and 14171 mg/kg, respectively, which is significantly higher than the concentrations observed by Ye et al. (2001a). Conversely, manganese concentrations observed in the above-ground and below-ground tissues of cattail taken from the Burnside treatment wetland were 1474 and 127 mg/kg, respectively, which is significantly lower than the concentrations observed by Ye et al. (2001a). However, the iron concentration of the inlet waters in the treatment wetland tested by Ye et al. (2001a) was 1.3 mg/L, whereas the average iron concentration in the inlet waters of the Burnside treatment wetland was approximately 11 mg/L, which is significantly higher. The manganese concentration of the inlet in the treatment wetland tested by Ye et al. (2001a) was 2.5 mg/L whereas the average manganese concentration in the inlet of the Burnside treatment wetland was approximately 1.5 mg/L. Similarly, the specimens analysed from the control site, which supported a low surface water iron concentration of 0.27 mg/L and a low surface water manganese concentration of 0.15 mg/L, had very little metal uptake. Hence it would appear that iron and manganese accumulation within broad-leaved cattail plants is correlated with water concentration.

Also, one possible explanation for the high manganese concentrations observed the cattail tissues compared to that of the other species analysed may lie in the fact that, similarly to the fowl mannagrass, the vast majority of the cattail present in the treatment site established mostly from seed. As discussed, manganese preferentially transports to young, expanding tissues in plants. In seeing that the cattail grew from seed, the plant tissues analysed have been all young, new tissues. Hence, the manganese accumulation ability of this plant may decline as its tissues mature over consecutive growing seasons (Kabata-Pendias and Pendias, 1992).

6.5.2.10. Reed canary grass (*Phalaris arundinacea*). The exotic, invasive species reed canary grass was selected as a candidate species for the tissue analysis because aggressive species are often the focus of phytoremediation studies as their rapid growth often makes them conducive to contaminant uptake (USEPA, 2001; EC, 2000b; and Davis, 1995). In addition, Hansel et al. (2002) observed elevated concentrations in analysed root tissues of reed canary grass in wetlands treating mine tailings. Reed canary grass became self-established in the Burnside wetland site and without weeding, would likely become one of the most dominant species in the treatment wetland. Like the cattail, this species was included in the tissue analysis in order to gauge how the phytoremediation ability of the native, non-aggressive species of the site compared in terms of their uptake ability to this aggressive weed.

Of the 10 plant species analysed, reed canary grass was the 4th highest accumulator of iron and 6th highest accumulator of manganese. The root contained the largest concentrations of iron and the leaves contained the largest concentrations of manganese. The same was observed of the iron and manganese concentrations in the reed canary grass from the control site (Figure 5.38). These results are fitting with the mobility behaviour of iron and manganese as described by Mays and Edwards (2000), Taylor and Crowder (1983), Shutes et al. (1993), Skousen et al. (1994), Ye et al. (2001a) and Kabata-Pendias and Pendias (1992). Overall, the native plants tweedy's rush and soft stem bulrush accumulated more iron, and the native plants fowl mannagrass, soft stem bulrush, tweedy's rush and pickerelweed accumulated more manganese than did this exotic plant. In addition, since this plant self-established from seed, its tendency to accumulate manganese may decline as its tissues mature (Kabata-Pendias and Pendias, 1992). Clearly, aggressive growth behaviour is not the only characteristic conducive to phytoremediation capability.

6.5.3. Macroinvertebrate Monitoring

Chemical analyses of water quality provides useful information about the presence of contaminants in the waters sampled at the treatment site at the exact moment in time they wre taken. However, this "snapshot" approach to monitoring has several limitations. Chemical analyses can be a poor indicator of long-term environmental variations causing cumulative or chronic effects. The biological effect of many chemicals is often much longer lasting than the pollution event itself. Chemical analyses cannot detect degradation from non-chemical sources as erosion or competition from introduced species, nor can they assess the effect of fluctuating or erratic conditions. They also fail to detect degradation caused by parameters omitted from assessment (USEPA, 2002g, m). These type of analyses are also of limited value in identifying less tangible indicators of declining system health such as habitat degradation. Aquatic organisms on the other hand must live for weeks or months in water, tolerating every fluctuation and variable the system provides, hence complementing traditional water quality analysis with biomonitoring techniques can provide useful integrative measures of water quality (Boulton and Brock, 1999; Barbour et al., 1999; and Osmund et al., 1995c).

Generally, bioindicators used in this water quality monitoring technique are aquatic macroinvertebrates, fish, and/or plankton. However, aquatic macroinvertebrates are most frequently used for the same reasons discussed in Section 6.4 (Osmund et al., 1995c; Hynes, 1998; Mandaville, 2002; and USEPA, 2002l). Decreased species diversity, heterogeneity, and skewed trophic structure are excellent indicators of an unhealthy system undergoing stress. However, more specific metrics exist which are indicative of the cause of that stress such as increased turbidity, nutrient and metal enrichment and low pH. Significant differences between the pollution-indicator populations at the outlet compared to that of the inlet would demonstrate a marked improvement in water quality. Similarly, significant similarities between the outlet and the reference wetland site would demonstrate water quality equivalent to that of a pristine wetland site.

A mid-summer sampling regime was appropriate for this analysis as the one-time P-loading of the site in the early spring would have resulted in datum that were non-representative of the typical condition of the site. For example, dragonflies (Leucorrhinia, Libellula), caddisflies (Triaenodes, Oecetis), chironomids (Tanytarsus, Procladius) and fingernail clams (Sphaeriidae) are all intolerant of phosphorus ranging from 0.015 to 1.38 mg/L. All of these taxa, which are excellent indicators of good water quality, may have been absent from samples in spring, but present in the site once P-levels abided.

6.5.3.1. BMWP Biotic Index and ASPT. The Biological Monitoring Workshop Party (BMWP) score and corresponding Average Score per Taxon (ASPT) for the reference wetland site was 107 and 5.94, respectively. This score is indicative of very good water quality. The BMWP and corresponding ASPT score for Cell 1 of the treatment wetland site was 34 and 4.86, respectively. This score is indicative of moderate water quality. Finally, the BMWP and corresponding ASPT score for the outlet of the treatment wetland site was 30 and 4.29, respectively. This score is indicative of moderately-poor water quality (Mandaville, 2002). It was hoped that the scores of the inlet and outlet would be significantly different, indicating improved water quality, and that the scores of the outlet would be somewhat similar to that of the reference site, indicating the effectiveness of the treatment wetland. The results showed that the treatment site supported less pollution sensitive taxa than did the reference site, and the outlet supported less pollution sensitive taxa than Cell 1, which would indicate that the water quality is actually degrading as it flows through the system. This index was designed to be used as a broad indicator of water quality and is limited in its ability to pinpoint any potential causes of the observed degradation (Mandaville, 2002; and Kirsch, 1999).

6.5.3.2. ETSD Biotic Index. ETSD families are particularly sensitive to pollution, and are typically only found in abundance in waterbodies supporting high water quality. It was hoped that percent abundance of Ephemeroptera (mayflies), Trichoptera (caddisflies), Sphaeriidae (fingernail clams), and Odonata (dragonflies and damselflies) (ESTD) families in Cell 1 would differ significantly from the outlet, indicating improving water quality as the leachate wastewaters flow through the site and that the percent abundance of ESTD families in the outlet and reference site would be similar thereby demonstrating that the treatment wetland was capable of remediating the wastewaters to a quality similar to that of a natural, pristine wetland ecosystem. However, neither of these scenarios were evident from the results. The ETSD families in the reference wetland site accounted for 45.45% of the site's families. Conversely, ETSD families only accounted for 7.69% of the families sampled in Cell 1, and 0% of the families sampled in the outlet of the treatment wetland. The fact that no ETSD families were sampled in the outlet shows not only that the water quality is very poor, but that it is actually not improving as it moves through the site, which concurs with the ASPT results.

The potential causes of the results can be explored by examining the specific sensitivities and requirements of the individual ETSD families. Trichoptera larvae are silk producing creatures that create portable cases used as retreats similar to that of snails (Figure 6.24). Using their silk, they attach themselves to solid, unmoving objects; usually rocks. Cell 1 and the outlet of the treatment wetland had observably high levels of precipitated iron which densely coats the substrates and other objects such as rocks. This slimy coating makes it extremely difficult for larvae to securely attach themselves to anything. In addition, many Trichoptera larvae breathe through sensitive gills which are easily damaged and clogged by particulate matter. As a result, many species of this family are unable to withstand turbid conditions. The presence of the iron particulate in the treatment wetland has caused the systems waters to be exceedingly turbid. Hence, the high abundance of the precipitated iron in the treatment wetland is a likely cause of their absence (Voshell, 2002; and Peckarsky et al., 1990).

Sphaeriidae are small clams with thin, fragile shells (Figure 6.25). They move slowly along bottom sediments by revealing a foot through a small opening, which makes contact with the substrate. They feed by using their specially adapted large gills to filter organic particles from the water. However, an overabundance of non-organic particles such as those associated with siltation or the iron particulate can actually harm their feeding mechanisms. In addition, unstable substrates coated with overabundant fine particulates hinder their ability to attach and move along the bottom sediments. Hence the presence of the particulate is a likely cause of their absence as well (Voshell, 2002; and Peckarsky et al., 1990).

Odonata larvae are a diverse group of predatory insects which exist both within the substrates and water column (Figure 6.26). Like Trichoptera and Sphaeriidae, Odonata breathe through gills. However, most species have special adaptations which protect the gills from damage associated with turbid conditions. Some can even live without their gills if damaged, relying on diffusion through the skin for their source of oxygen. Unfortunately, the presence of abundant particulate matter affects them in a different way. Odonata are visual predators, hence in turbid waters, their ability to capture food to sustain themselves is extremely compromised. Hence, the presence of the iron particulate is also the likely cause of the absence of Odonata in the treatment wetland site (Voshell, 2002; and Peckarsky et al., 1990).

Ephemeroptera larvae are typically collectors or scrapers with three characteristically thin, long tails (Figure 6.27). They are considered the most sensitive of the ETSD group. They are highly vulnerable to turbid conditions, low dissolved oxygen, and heavy metals (Voshell, 2002; and Mandaville, 2002). For these reasons, mayfly abundance was examined as its own water quality index.

6.5.3.3. Mayfly Abundance. As with the ASPT and ETSD indices, it was also hoped that the Ephemeroptera (mayfly) abundance in Cell 1 would be significantly different from the outlet, indicating improving water quality in the system, and the mayfly abundance in the reference site and outlet would be significantly similar, indicating water quality comparable to a pristine wetland. However, this was not demonstrated by the results. Mayfly abundance in the reference site was 13.64%, but no mayflies were identified in either Cell 1 or the outlet of the treatment wetland site.

Like Trichoptera and Sphaeriidae, Ephemeroptera breathe through delicate gills which are easily damaged and clogged by particulate matter. Mayflies also require firm substrates to carry out their daily tasks. The high abundance of iron particulate in the treatment wetland makes it an extremely hostile environment for Ephemeroptera to thrive in as it would likely be the cause of irreparable damage to their vital breathing systems, as well as make the substrates critically unstable (Voshell, 2002; and Mandaville, 2002). Ephemeroptera are also extremely sensitive to low dissolved oxygen levels. Low dissolved oxygen levels are often the result of organic

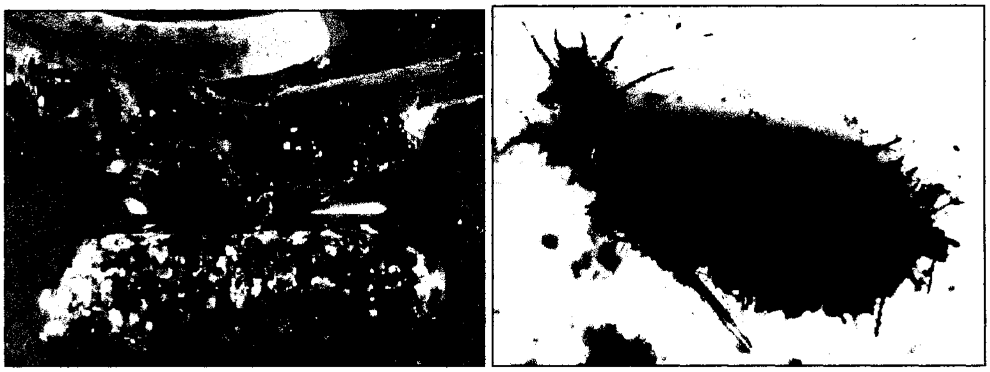


Figure 6.24. Trichoptera larvae (Mandaville, 2002).

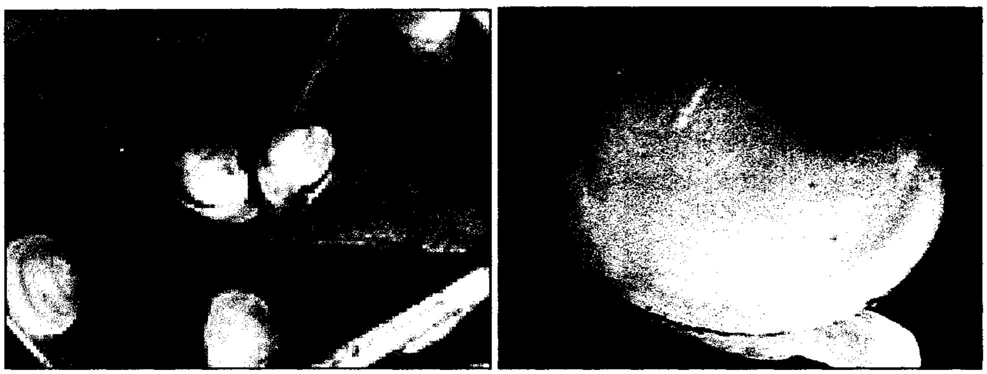


Figure 6.25. Sphaeriidae (Mandaville, 2002).

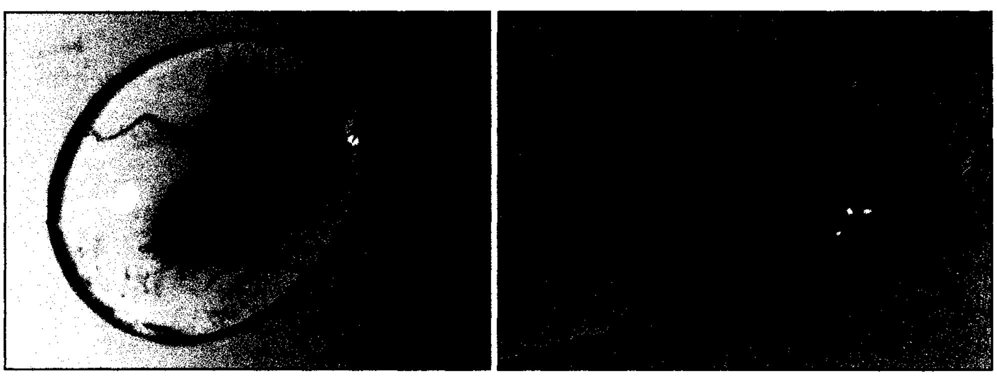


Figure 6.26. Odonata Larvae in Reference Wetland.

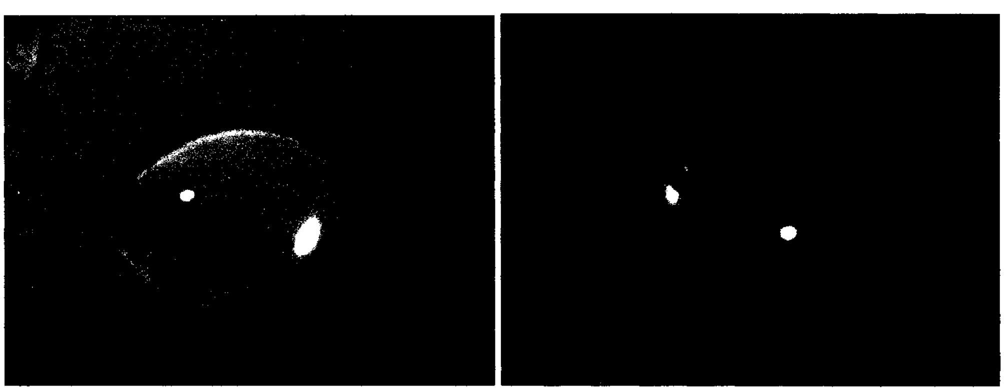


Figure 6.27. Ephemeroptera Larvae in Reference Wetland.

pollution such as raw sewage, although the contaminants treated by the engineered wetland do not include organic wastes. However, high particulate content in waters increases heat absorption which lowers dissolved oxygen levels. In addition, dissolved oxygen is also consumed during iron oxidation (Mason, 1998; Voshell, 2002; Boulton and Brock, 1999; and CCME, 2001). According to Mason (1998), mayflies are one of the most sensitive groups to heavy metalcontamination in waters. Hence, the presence of the abundant iron particulate in the treatment wetland is the likely cause of the absence of Ephemeroptera in the treatment wetland site.

6.5.3.4. Trophic Structure. The shredder, scraper, predator and collector ratio in the reference wetland was 2:2:21:75, which is normal for natural wetland sites (Mitsch and Gosselink, 2000; and Mason, 1998). Conversely, the shredder, scraper, predator and collector ratio in Cell 1 was 0:2:2:96, and 0:0:3:97 in the outlet. Clearly the ratios of functional feeding groups are distributed significantly different from the reference site, which can be interpreted for specific water quality indications. Scraper and collector abundances often decrease in response to metal contamination, as metals tend to bind to their food, leading to both acute and chronic toxic effects. However, abundant collectors were sampled in the treatment wetland site, which can be interpreted as an indication that the iron and manganese in the waters are mostly present in their unbioavailable, non-toxic, oxidized states. If abundant reduced iron and manganese were present in the site, collectors would not be abundant (Mason, 1998; and Voshell, 2002).

The lack of scrapers can be explained by the overbearing presence of the oxidized iron particulate. Scrapers feed on periphyton that grow attached to solid objects. Iron precipitate tends to coat sediment, gravel and rocks, limiting the ability of periphyton to grow. The algae that does survive often becomes smothered in floc, making it inconsumable.

Shedder functional feeding group shifts are also most commonly a response to a lack or abundance of food. In seeing that the wetland site is only in its second growing season, it has not had much opportunity to build up large amounts of coarse particulate organic matter (CPOM) in the form of leaves and detritus, which is their primary food source (Osmund et al, 1995c). However, this does not explain their complete absence from the treatment site. An absence of shredders is also cited as an indication of toxic substances such as metals which often to bind to CPOM, resulting in shredder mortalities (Osmund et al, 1995c). Iron and manganese present in the leachate may have bound to their food source, causing toxic effects. Just as likely, however, is the less dramatic explanation that their food source had simply become so smothered in iron particulate that it too became inaccessible to consume (Voshell, 2002).

In most circumstances, the feeding group which suffers the highest declines in unhealthy ecosystems are predators. This is because predators tend to be highly sensitive, mostly in part of their larger biomass, their high oxygen demands and bioaccumulation factors. The treatment wetland site supported 2 predators in Cells 1 and 3 predators in the outlet while the reference site supported 21 predators. Their absence is an indication of low dissolved oxygen, and potential bioaccumulating chemicals such as persistent organic pollutants (POPs). Interestingly enough, their absence is not indicative of bioavailable metals causing toxicity, as in the case of metal contamination, predator ratios typically appear high against that of the declining scrapers and collectors (USEPA, 2002l; and Osmund et al., 1995c). However, the opposite has occurred here, which again supports the conjecture that the majority of metals in the wetland are not bioavailable, and not causing toxic effects. However, the high abundance of oxidized iron particulate in the treatment wetland is associated with various non-toxic, secondary effects, such as the smothering of eggs, food, substrates, and habitat, the decreasing of dissolved oxygen due to increased water temperatures, and the reduction of visibility which affects visual predators. Hence the iron particulate is also the most likely cause of the disproportionate trophic structures in the treatment wetland site. It is hoped that as the system matures, its iron remediation ability will improve as well. However, given the amount of iron received by the system, and its degree of impact, additional measures and improvements may need to be made in order to increase the site's iron treatment capability.

7. CONCLUSIONS

1. The Wright's brook marsh served as an effective vegetation model for the Burnside treatment wetland. It provided the template necessary to meet the study objective of selecting the appropriate native vegetation for establishment in the Burnside treatment wetland.

2. Through the species screening and research exercises, it was established that the Wright's brook marsh model supported a myriad of species which collectively and effectively met all project goals.

3. The vegetation donor sources and the establishment methodology selected for the Burnside treatment wetland site provided satisfactory quantities and varieties of final plant list species for the site.

4. The vegetation establishment strategy selected for the Burnside treatment wetland was successful. Although the aquatic species pickerelweed, the shrubs sheep laurel and rose, and the trees trembling aspen, grey birch and speckled alder did see high mortality rates in the wetland site, and the woolgrass proved susceptible to the scouring effects of high water levels observed during rain events, the site establishment strategy yielded a high overall 87.3% success rate.

5. There were no significant differences between the plant species diversities of both the reference and treatment wetland site.

6. The treatment wetland supported a higher plant species richness than the reference site. However, the reference site supported greater biological integrity as it had a small abundance of exotic and invasive species (3.75% versus 10.73%) and had greater heterogeneity than the treatment site (32.6% versus 46.4%).

7. Although poor heterogeneity and the presence of weedy, exotic species can be a sign of degraded biological health and future problems, these are also common indicators of a system simply undergoing early succession. The first colonizers in areas are typically aggressive species which are adapted to take advantage of full sunlight exposure and inhabit bare areas quickly. However, as a disturbed site matures, and space and light availability become

lessened, these successionary colonizers often give way to more hardy, slow-spreading species which are better light competitors.

8. As the treatment wetland system matures, its plant biodiversity may actually decrease, but its integrity, as measured by exotic and invasive species abundance as well as heterogeneity, is expected to increase, so long as invasive species present in the treatment wetland site remain controlled through weeding during the first few growing seasons.

9. The aquatic macroinvertebrate component of the biological integrity assessment demonstrates that the treatment wetland is still lacking biological integrity as it does not support macroinvertebrate populations similar to that of a natural wetland. The site is still immature and may simply need more time to develop a healthy macroinvertebrate community. As the site becomes more densely vegetated with time, the health and integrity of the macroinvertebrate populations will improve as vegetation increases dissolved oxygen levels in water and sediments, provides a vital food source, and provides surface area for attachment.

10. The abundant iron floc, existing throughout the treatment site, results in the smothering of macroinvertebrate habitat and eggs and causes increases in water temperatures and decreases in dissolved oxygen levels.

11. The Burnside treatment wetland site attracts and provides habitat for a wide variety of wildlife species.

12. The water quality analysis shows that little water quality improvement is occurring in the Burnside treatment wetland. This is attributed in part to the immaturity of vegetative populations of the site, which are only in their first season of growth.

(a) The deficient water improvement seen by the treatment wetland site is most likely attributed to its inferior size for iron and manganese treatment, as well as the reduced water retention experienced by the system as a result of breaks in the berms.

(b) As the vegetation matures and spreads, removal efficiency is expected to improve as microbial activity and dissolved oxygen levels will be increased, more contaminants will be absorbed through phytoremediative processes, and trapping and stabilization of particulates and sediments will be accommodated.

13. Broad-leaved cattail was the most effective accumulator of iron in the treatment wetland and fowl mannagrass was the most effective manganese accumulator in the treatment wetland.

14. The native plants tweedy's rush and soft stem bulrush accumulated more iron than the exotic canary reed grass, and the native plants fowl mannagrass, tweedy's rush, soft stem bulrush and pickerelweed accumulated more manganese than the exotic canary reed grass, demonstrating that aggressive growth behaviour is not the only characteristic conducive to high phytoremediation capability.

15. The macroinvertebrate water quality monitoring component of this study concurs with the findings of the water quality analysis in that it demonstrates that little water quality improvement is occurring in the treatment wetland at this early stage. The most prevalent problem identified through this analysis is the overabundant and overwhelming presence of the iron floc in the system.

8. RECOMENDATIONS

1. Vegetation selection for the treatment site should be based on natural wetland community populations.

2. Given its apparent success and ease of execution, the seeds of native species should continue to be collected and applied annually in the treatment wetland to increase the vegetative density of the cells and to help maintain a healthy, native plant population in the wetland.

3. The buffer areas should continue to be developed with densely growing, deep rooting species in order to help stabilize banks and improve cover and shading.

4. The progressing establishment, growth and spread of the vegetation in the treatment wetland should continue to be closely monitored over the next few growing seasons, scaling down in intensity as the site matures and populations become more stable.

5. The tweedy's rush as well as other aggressive and exotic species in the site such as the cattail and the reed canary grass should be continually monitored for increasing domination. If it should appear that any aggressive species are overtaking the site, control measures such as weeding or even herbicide application may be necessary.

6. The abundant iron floc present should be captured earlier in the treatment wetland site to improve the macroinvertebrate populations in the system as measured by taxa diversity, heterogeneity and trophic structure would very likely improve. An alternative means of treating the iron other than the natural processes in the treatment wetland should be considered. Typically, a trickling filter consists of a 1 to 3 m deep bed of 3.8 to 5 cm diameter gravel, stone, clinker, plastics or slag. The area of the tricking filter is dependent on the amount of particulate to be treated. The bed is lined with an impermeable liner, usually composed of durable plastic. Leachate is distributed across the surface of the trickling filter through perforated PVC pipes. The flow through the perforated pipes is typically adjusted through ball valves aligned along the pipe manifold system. The trickling filter aerates the leachate, causing iron to precipitate. The precipitated iron then adheres to the media or settles out in the bottom of the filter. The rock media must typically be replaced every 4 to 5 years. For economy and to prevent clogging of the distribution nozzles, trickling filters are

commonly preceded by primary sedimentation tanks equipped with scum collecting devices (Jarvis and Younger, 2001; Gouzinis et al., 1998; and Mason, 1998).

7. Wildlife habitat should continue to be facilitated in the treatment wetland site

8. Berms must be redesigned and strengthened in order to maintain a constant water retention time in the wetland.

9. Plants from the treatment wetland site should continue to be analysed to gauge their phytoremediative ability. In general, native plants are effective accumulators of iron and manganese and should be considered more often in treatment wetland projects dealing with these constituents.

9. REFERENCES

A&L Canada Laboratories Inc. (2002). Technical Bulletin [online].
Available:
http://www.al-labs-can.com/gps_services.html [2002, 15 March]

American Society of Plant Biologists. (2003). Phytoremediation: Using plants to remove pollutants from the environment [online].
Available:
http://www.aspb.org/publicaffairs/briefing/phytoremediation.cfm [2002, 16 March].

Ayers, D.M., A. P. Davis, and P.M. Gietka. (1994). Removing Heavy Metals from Wastewater [online].
Available:
http://www.erc.umd.edu/TES/techtips/PMG%20metal%20precip%20man.pdf [2002, 16 March].

Azous, A.L. and R.R. Horner. (2000). Wetlands and Urbanization: Implications for the Future. Lewis Publishers: Boca Raton, FL.

Balla, S. (1994). Wetlands of the Swan Coastal Plains Vol. 1.: Their nature and management. Water Authority of Western Australia and the Western Australia Department of Environmental Protection: Perth, WA.

Banner, A. and W. MacKenzie. (2000). Ecology of Wetland Ecosystems: Extension Note 45. B.C. Ministry of Forests, Research Branch, Forest Science Program Extension Notes [online].
Available:
http://www.for.gov.bc.ca/hfd/pubs/Docs/En/En45.p [2002, 6 February].

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. (1999). Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water: Washington, D.C.

Barlaz, Morton A. (1996). Microbiology of solid waste landfills. In: Microbiology of Solid Waste. A.C. Palmisano and M.A. Barlaz (eds). CRC Press: Boca Raton, FL.

Barlaz, Morton A. and Gabr, Mohammed A. (2001). Closing the Gaps in the Regulation of Municipal Solid Waste Landfills: Defining the End of the Post-Closure Monitoring Period and the Future Stability of Leachate Recirculation Landfills [online].
Available:
http://www4.ncsu.edu:8030/~barlaz/post_closure/ [2003, 14 January].

Bastian, R.K. and Hammer, D.A. (1993). The use of Constructed Wetlands for Wastewater Treatment and Recycling. In: Constructed Wetlands for Water Quality Improvement. G.A. Moshiri (ed.). Lewis Publishers: Boca Raton, Florida. Pp. 59-68.

Bennet, D.E. (1994). The Status of Wetland Stormwater Management in Ontario. University of Waterloo: Waterloo, ON.

Boojum Research Limited. (2002). Nitrogen Removal from Mine Effluent: An Ecological Approach [online].
Available:
http://www.boojumresearch.com/nitrogen.htm [2003, 14 January].

Boulton, A. J. and M. A. Brock. (1999). Australian Freshwater Ecology: Processes and Management. Cooperative Research Centre for Freshwater Ecology and Gleneagles Publishing: SA.

Brix, H. (1993). Wastewater Treatment in Constructed Wetlands: System Design, Removal Processes, and Treatment Performance. In: Constructed Wetlands for Water Quality Improvement. G.A. Moshiri (ed.). Lewis Publishers: Boca Raton, FL. Pp. 9-22.

British Columbian Ministry of Environment, Land and Parks (BCMELP). (1986). Water Quality Criteria for Nitrogen (Nitrate, Nitrite, and Ammonia). Overview Report. Prepared pursuant to Section 2(e) of the Environment Management Act, 1981 [online].
Available:
http://wlapwww.gov.bc.ca/wat/wq/BCguidelines/nitrogen.html#tab1 [2003, November 19].

British Columbian Ministry of Environment, Land and Parks (BCMELP). (2001). Ambient Water Quality Guidelines for Manganese. Overview Report. Prepared pursuant to Section 2(e) of the Environment Management Act, 1981 [online].
Available:
http://wlapwww.gov.bc.ca/wat/wq/BCguidelines/manganese.html [2002, 14 September].

Brodie, G.A., C.R. Britt, T.M. Tomaszewski, and H.N. Taylor. (1993). Anoxic limestone drains to enhance performance of aerobic acid drainage treatment wetlands: Experiences of the Tennessee Valley Authority. In: Constructed wetlands for water quality improvement. G.A. Moshiri (ed.). Lewis Publishers: Boca Raton, FL. Pp. 922-932.

Burton, S.A.Q., and I.A Watson-Craik. (1998). Ammonia and Nitrogen Fluxes in Landfill Sites: Applicability to Sustainable Landfilling. Waste Management and Research. 16(1): 41-53.

Calabrese, J.P., A.J. Sexstone, D.K. Bhumbla, G.K. Bissonnette, and J.C. Sencindiver. (1991). Application of constructed cattail wetlands for the removal of iron from acid mine drainage. Proceedings, Second International Conference on the Abatement of Acidic Drainage, Vol. 3, MEND. September 16-18, Montreal, Canada.

Campbell, C.S. and M. Ogden. (1999). Constructed Wetlands in the Sustainable Landscape. John Wiley and Sons, Inc: Toronto, ON.

Cameron, K, C. Madramootoo, A. Crolla, and C. Kinsley. (2003). Pollutant removal from municipal sewage lagoon effluents with a free-surface wetland. Water Research. 37: 2803–2812.

Canadian Council of Ministers of the Environment. (1999a). Canadian Water Quality Guidelines for the Protection of Aquatic Life: pH (Marine). In: Canadian Environmental Quality Guidelines. CCME: Winnipeg.

Canadian Council of Ministers of the Environment. (1999b). Canadian Water Quality Guidelines for the Protection of Aquatic Life: Dissolved oxygen (Freshwater). In: Canadian Environmental Quality Guidelines. CCME: Winnipeg.

Canadian Council of Ministers of the Environment. (2000). Canadian Water Quality Guidelines for the Protection of Aquatic Life: Ammonia [online].
Available:
http://www.ccme.ca/ceqg_rcqe/english/E5_01a.pdf [2002, 14 September].

Canadian Council of Ministers of the Environment (CCME). (2001). Summary of Existing Canadian Environmental Quality Guidelines [online].
Available:
http://www.ccme.ca/ceqg_rcqe/english/E1_06.pdf [2002, 14 September].

Canadian Council of Ministers of the Environment (CCME). (2002). Summary of Existing Canadian Environmental Quality Guidelines. CCME: Winnipeg, MB.

Canadian Wildlife Service. (2002). Wetland Conservation Highlights [online].
Available:
http://www.on.ec.gc.ca/wildlife/docs/glwcap1-e.html [2003, 22 January].

Carlisle, B.K. (1998). Wetland Ecological Integrity: An Assessment Approach. The Coastal Wetlands Ecosystem Protection Project [online].
Available:
http://www.state.ma.us/czm/wetlandecologicalintegrity.pdf [2003, 11 March].

Castelle, A.J., A.W. Johnson, and C. Conolly. (1994). Wetland and Stream Buffer Size Requirements: A Review. Journal of Environmental Quality. 23(5): 878-893.

Centre for Public Environmental Oversight (CPEO). (1998). Enhanced Bioremediation: Description [online].
Available:
http://www.cpeo.org/techtree/ttdescript/ensolmx.htm [2002, 12 September].

Coleman, J., K. Hench, Keith Garbutti, S. Sexstone, G. Bissonnette, and J. Skousen. (1999). Treatment of Domestic Wastewater by Three Plant Species in Constructed Wetlands. Department of Biology and Division of Plant and Soil Sciences, West Virginia University: Morgantown, WV.

Colorado School of Mines. (1999). Acid Mine Drainage Experiments [online].
Available:
http://www.mines.edu/fs_home/jhoran/ch126/yellow.htm [2003, 17 April].

Cousins, S.H. (1991). Species diversity measurement: Choosing the right index. Trends in Ecology and Evolution. 6: 190-192.

Côté, R., T. Kelly, H. Reid, and T. Smolenaars. (1996). The Industrial Park as an Ecosystem: Sectoral Case Studies. Dalhousie University, School for Resource and Environmental Studies: Halifax, Nova Scotia.

Côté, R. (2001). The Evolution of an Industrial Park: The Case of Burnside. Presentation for the International Conference and Workshop on Industrial Park Management Manila, Philippines [online].
Available:
http://www.uneptie.org/pc/ind-estates/pdf_documents/ResourceKit/05%20Case%20 Studies /06%20Burnside%20Case%20Study.pdf [2003, 9 July].

Cowardin LM, V. Carter, F.C. Golet, and E.T. LaRoe. (1979). Classification of wetlands and deepwater habitats of the United States. FWS/OBS-79/31. U.S. Fish and Wildlife Service: Washington, DC.

Crawley, M.J. (1997). Structure of plant communities. In: Plant Ecology. M.J. Crawley (ed.). Blackwell Science: Oxford.Pp. 475-531.

Crow, G.E. and C.B. Hellquist. (2000). Aquatic and Wetland Plants of Northeastern North America, Vol. I and II. The University of Wisconsin Press: Madison, Wisconsin.

Cusso, R., K. Haarstad, M. C. Lavagnolo and P. Littarru. (2001). Removal of municipal solid waste COD and NH4–N byphyto-reduction: A laboratory–scale comparison of terrestrial and aquatic species at different organic loads. Chemosphere. 16: 459-470.

Daigle, J.M. and D. Havinga. (1996). Site Level Restoration Planning and Implementation. In: Restoring Nature's Place: A Guide to Naturalizing Ontario's Parks and Greenspace. J.M. Daigle and D. Havinga (eds.). Ontario Parks Association: Toronto. Pp. 61-92.

David, S.M., K.M. Somers, R.A. Reid, R.J. Hall, and R.E. Girard. (1998). Sampling Protocols for the Rapid Bioassessment of Streams and Lakes using Benthic Macroinvertebrates. Second Edition. Ontario Ministry of the Environment: Toronto, ON.

Davis, L. (1995). A Handbook of Constructed Wetlands: A Guide to Creating Wetlands for Agricultural Waste, Domestic Wastewater, Coal Mine Drainage, and Stormwater in the Mid-Atlantic Region. Vol. 1: General Considerations. Natural Resources Conservation Service, US Environmental Protection Agency-Region III and the Pennsylvania Department of Environmental Resources: Washington, DC.

Demchik, M. and K. Garbutt. (1999). Wetlands and Aquatic Processes. Journal of Environmental Quality. 28(1): 243-249.

Department of Natural Resources (DNR). (1998). Nova Scotia Wet Places: Eastern Habitat Joint Venture [online].
Available:
http://www.gov.ns.ca/natr/WILDLIFE/wetlands/page1.htm [2002, 2 December].

Diekelmann, J. and R. Schuster. (1982). Natural Landscaping: Designing with Native Plant Communities. McGraw-Hill, Inc.: New York.

Ducks Unlimited (DU) Canada. (2002). Facts and Figures [online].
Available:
http://www.ducksunlimited.ca/aboutdu/facts.html [2002, 22 September].

Eco-Efficiency Centre. (2003). Primer: Burnside Industrial Park. Dartmouth, Nova Scotia: Eco-Efficiency Centre [online].
Available:
http://www.mgmt.dal.ca/sres/eco-burnside/burnsideindustrialpark.html [2003, 3 September].

Electronic Power Research Institute (EPRI). (2001). Revision of the Freshwater Ambient Water Quality for Iron [online].
Available:

http://www.epri.com/corporate/products_services/ project_opps/environment/101540.pdf [2002, 13 November].

Environment Canada. (1987). Canadian Wetland Classification System. Ecological Land Classification Series No. 21. National Wetlands Working Group, Land Conservation Branch, Canadian Wildlife Service, Environment Canada: Ottawa, ON.

Environment Canada. (2000). Planting the Seed: A Guide to Establishing Aquatic Plants [online]. Available:
http://www.on.ec.gc.ca/wildlife/docs/doc-planting-e.html [2002, 2 January].

Environment Canada. (2003). Climate Normals 1971-2001 from the Shearwater A weather station, Nova Scotia [online].
Available:
http://climate.weatheroffice.ec.gc.ca/climate_normals/ [2003, 10 July].

Faulkner, S.P., and C.J. Richardson. (1989). Physical and chemical characteristics of freshwater wetland soils. In: D.A. Hammer (ed.). Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural. Proc. from the 1st Int. Conf. on Constructed Wetlands for Wastewater Treatment. 13-17 June 1988. Chattanooga, TN. Pp. 41–72.

Federal Remediation Technologies Roundtable (FRTR). (1998). Remediation Technologies Screening Matrix and Reference Guide Version 3.0 [online].
Available:
http://www.frtr.gov/matrix2 [2002, 4 December].

Fennessy, M.S., and W.J. Mitsch. (1989). Treating coalmine drainage with an artificial wetland. Res. J. Water Pollut. Control. Fed. 61:1691-1701.

Ferren W.R., P.L. Fiedler and R.A. Leidy. (1996). Wetlands of the Central and Southern California Coast and Coastal Watersheds: A Methodology for their Classification and Description [online].
Available:
http://www.mip.berkeley.edu/wetlands/tabofcon.html [2002, 27 March].

Fields, S. (1993). Regulations and policies relating to the use of wetlands for nonpoint source pollution control. In: R.K. Olson (ed.). Created and Natural Wetlands for Controlling Nonpoint Source Pollution. CRC Press: Boca Raton, FL. Pp. 151-158.

Freedman, B. (2001). Agriculture and Environment. In: B. Freedman (ed). Environmental Science: A Canadian Perspective. 2nd Ed. Pearson Education Canada Inc.: Toronto, ON.

Fripp, J., P.F. Ziemkiewicz and H. Charkavorki. (2000). Acid Mine Drainage Treatment. EMRRP Technical Notes Collection (ERDC TN-EMRRPSR-14) [online].
Available:
http://www.wes.army.mil/el/emrrp/pdf/sr14.pdf [2002, 5 November].

Ghaly, A.E. and R. Côté. (2001). Engineered Wetland Technology for Treatment of Industrial Park Contaminants: A Progress Report. Dalhousie University: Halifax, NS.

Gouzinis, A., N. Kosmidis, D.V. Vayenas, and G. Liberatos. (1998). Removal of Mn and simultaneous removal of NH3, Fe and Mn from potable water using a trickling filer. Water Research. 32(8): 2442-2450.

Grant, J. (1996). Designing Industrial Parks for the Future. Plan Canada. 36(3): 17-22.

Grant, J. (2000). Industrial Ecology: Planning a New Type of Industrial Park. Journal of Architecture and Planning Research. 17(1): 64-81.

Haberl, R., G. Langergraber and J. Laber. (1999). Constructed wetlands for the treatment of water containing heavy metals [online].
Available:
http://lbewww.epfl.ch/COST837/madrid2000/session4.pdf [2002, 22 September].

Hagen, Andy. (2000). Planting the Seed: A Guide to Establishing Aquatic Plants [online].
Available:
http://www.on.ec.gc.ca/wildlife/docs/doc-planting-e.html [2002, 17 January].

Halifax Land Information Services. (1992). Aerial photograph 181, 92329-77 and 183, 92329-7. 92-07-16, 11D-12, 111-65. Halifax Land Information Services, Government of Nova Scotia: Halifax, N.S. Scale 1: 10, 000.

Halifax Land Information Services. (2003). Topographic Map, Dartmouth, Nova Scotia Land Information Services, Government of Nova Scotia: Halifax N.S. Scale 1: 5, 000.

Hammer, D.A. (1992). Creating Freshwater Wetlands. Lewis Publishers: London.

Hansel, C.M., M.J. La Force, S. Fendorf, and S. Sutton. (2002). Spatial and temporal association of As and Fe species in aquatic plant roots. Environmental Science and Technology. 36(9): 1988-1994.

Hedin, R.S. (1989). Treatment of coal mine drainage with constructed wetlands. In: S.K. Majumdar et al. (ed.). Wetlands ecology and conservation: Emphasis in Pennsylvania. The Pennsylvania. Academy of Science: Easton, PA. Pp. 349-362.

Henrot, J., and R.K. Wieder. (1990). Processes of iron and manganese retention in laboratory peat microcosms subjected to acid mine drainage. J. Environ. Quality 19:312-320.

Hicks, D. B. and Q. J. Stober. (1989). Monitoring of constructed wetlands for wastewater. In: D.A. Hammer (ed). Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural.. Lewis Publishers: Chelsea, MI. Pp. 447-455.

Hoag, C. (2000). Harvesting, Propagating and Planting Wetland Plants. Riparian/Wetland Project Information Series No. 14 [online].
Available:
http://www.plant-materials.nrcs.usda.gov/pubs/idpmcarwproj14.pdf [2002, 23 January].

Hoeg, R. (2001). Weeds to Watch for. East Coast Gardener [online].
Available:
http://www.klis.com/fundy/ecg/weeds3-5.htm [2002, 18 January].

Huang, J., R.B. Reneau, and C. Hagedorn. (1999). Nitrogen removal in constructed wetlands employed to treat domestic wastewater. Water Research. 34(9): 2582-2588.

Hynes, K.E. (1998). Benthic Macroinvertebrate Diversity and Biotic Indices for Monitoring of 5 Urban and Urbanizing Lakes within the Halifax Regional Municipality (HRM), Nova Scotia, Canada. Soils and Water Conservation Society of Metro Halifax: Dartmouth, Nova Scotia.

Jarvis, A.P. and P.L. Younger. (2001). Passive treatment of ferruginous mine waters using high surface area media. Water Research. 35(15): 3643-3648.

Johnson, G.V., A. Lopez and N. La Valle Foster. (2002). Reduction and transport of Fe from siderophores. Plant and Soil. 241(1): 27-33.

Jones, D.L. (1998). Organic acids in the rhizosphere: A critical review. Plant and Soil. 205(1): 25-44.

Jordan, William R., Michael E. Giplin, and John D. Aber. (1991). Restoration Ecology: A synthetic approach to ecological research. Cambridge: Cambridge University Press.

Kabata-Pendias, A. and H. Pendias. (1992). Trace elements in soils and plants. 2nd Edition. CRC Press: Boca Raton, Florida.

Kadlec, Robert H. and Robert L. Knight. (1996). Treatment Wetlands. CRC Press LCC: Boca Raton, Florida.

Kamnez, A.A. and D. van der Lelie. (2000). Chemical and biological parameters as tools to evaluate and improve heavy metal phytoremediation. Bioscience Reports. 20 (4): 239-258.

Kanazawa K, K. Higuchi, N.K. Nishizawa, S. Fushiya, and S. Mori. (1995). Detection of two distinct isozymes of nicotianamine aminotransferase in Fe-deficient barley roots. Journal of Experimental Botany. 46: 1241-1244.

Karr, J.R. and D.R. Dudley. (1981). Ecological Perspectives on Water Quality Goals. Environmental Management. 5: 55-68.

Karr, J.R. (1997). Measuring biological integrity. In: G.K. Meffe and C.R. Carroll (eds.). Principles of Conservation Biology. Sinauer: Sunderland, MA. Pp. 483-485.

Kastning-Culp, N., L. DeBrey, and J. Lockwood. (1993). Development of High Mountain Plant Communities as Wetland Mitigation Systems for Copper Mine Effluent. Department of Plant, Soil and Insect Sciences, University of Wyoming: Wyoming.

King, B.R., G.M. Long, and J.K. Sheldon. (1992). Practical Environmental Bioremediation. Lewis Publishers: Boca Raton, Florida.

Kirsch, P.E. (1999). Benthic Macroinvertebrate Diversity and Biotic Indices Analysis of Lakes Wrights, Springfield, McGrath, Kearney and Morris and an Upgraded Analysis of Lakes Dollar, Russell, Stillwater, Papermill, and Kinsac within the Halifax Regional Municipality (HRM), Nova Scotia, Canada. Soil and Water Conservation Society of Metro Halifax: Dartmouth, Nova Scotia.

Kleinmann, R.L. (1985). Treatment of acid mine water by wetland. Control of Acid Mine Drainage. Proc. of a technology transfer seminar. Bureau of Mines circulator #9027. US Dept. of Interior. Pp. 48-52.

Knight, R.L., B.H. Winchester, and J.C. Higman. (1986). Ecology, hydrology, and advanced wastewater treatment potential of an artificial wetland in North-central Florida. Wetlands. 5:167-180.

Krebs, C.J. (1994). Distribution and abundance at the community level. In: C.J. Krebs (ed). Ecology: the Experimental Analysis of Distribution and Abundance. 4th Edition. Addison-Wesley: Don Mills, ON.

Kruempelbeck, I., and H.J. Ehrig. (1999). Long Term Behavior of Municipal Solid Waste Landfills in Germany. Sardinia, 99, Seventh International Waste Management and Landfill Symposium, October 4-7, Vol. I. Pp. 27-36. add to landfill section??

Kusler, J., and V.R. Burkett. (1999). Wetlands and climate change: Scientific approaches and management options. National Wetlands Newsletter. 21(2):16-17.

Kusler, J.A. and M.E. Kentula. (1989). Wetland Creation and Restoration: The Status of the Science, Vol. I: Regional Reviews. U.S. Environmental Protection Agency, Environmental Research Laboratory: Corvallis, OR.

Larson, G.E. (1993). Aquatic and wetland vascular plants of the northern Great Plains. Gen. Tech. Rep. RM-238 [online].
Available:
http://www.npwrc.usgs.gov/resource/1999/vascplnt/vascplnt.htm [2002, 5 February].

Lasat, M. (2000). The Use of Plants for the Removal of Toxic Metals from Contaminated Soil [online].
Available:
http://www.clu-in.org/download/remed/lasat.pdf [2002, 3 March].

Lasat, M. (2002). Phytoextraction of toxic metals: a review of biological mechanisms. Journal of Environmental Quality. 31: 109-120.

Lee, G. F. and A. Jones-Lee. (2001).Assessing the Water Quality Impacts of Phosphorus in Runoff from Agricultural Land. Proceedings of American Chemical Society Agro Division Symposium: Environmental Impact of Fertilizer Products in Soil, Air and Water. American Chemical Society: Chicago, IL.

Lewis, M.C.F., B.B. Taylor, R.R. Stea, G.G.J. Fader, R.J. Horme, S.G. MacNeil and J.G. Moore. (1998). Earth Science and Engineering: Urban Development in the Metropolitan Halifax Region. Minerals and Energy Branch, Nova Scotia Natural Resources: Halifax, NS.

Liehr, S., D. Kozub, J. Rash, M. Sloop, B. Doll, R. Rubin, H. House, S. Hawes, and D. Burks. (2000). Constructed Wetlands Treatment of High Nitrogen Landfill Leachate. Water Environment Research Foundation: Alexandria, Virginia.

Lorion, R. (2001). Constructed Wetlands: Passive Systems for Wastewater Treatment: Technology Status Report [online].
Available:
http://www.clu-in.org/download/remed/constructed_wetlands.pdf [2002, 3 March].

Lum, K. (1998). The key role of wetlands in addressing the global water crisis. International Conference, Water and Sustainable Development, March 19-20-21,1998. Paris, France.

Lyon, J.G. (1993). Practical Handbook for Wetland Identification and Delineation. Lewis Publishers: Boca Raton, FL.

Mandaville, S.M. (1999). Bioassessment of Freshwaters Using Benthic Macroinvertebrates: A Primer. Soil and Water Conservation Society of Metro Halifax: Dartmouth, Nova Scotia.

Mandaville, S.M. (2002). Benthic macroinvertebrates in Freshwaters: Taxa Tolerance Values, Metrics, and Protocols. Soil and Water Conservation Society of Metro Halifax: Dartmouth, Nova Scotia.

MapArt. (2001). Halifax Regional Municipality Plus. MapArt Publishing Corporation: Oshawa, ON. Scale 1:25 000.

Mason, C.F. (1998). Biology of Freshwater Pollution: Third Edition.Addison Wesley Longman Limited: England.

Matagi, S.V., D. Swai, and R Mugabe. (1998). A review of heavy metal removal mechanisms in wetlands. Afri. J. Trop. Hydrobiol. Fish. 8: 23-35.

Mays, P.A. and G.S. Edwards. (2001). Comparison of heavy metals accumulation in a natural wetland and constructed wetlands receiving acid mine drainage. Ecological Engineering. 18(2): 251-252.

McMillen, T.L., M.M. DeLeo, P.J.S. Colberg, N. Svoboda-Colberg, J.L. Smith. (1994). Constructed Vertical Flow Aerated Wetlands: Manganese Removal from Acid Mine Drainage. AWRA Effects of Human Induced Changes on Hydrology Symposium. June 26-29, 1994. Jackson Hole, WY.

Mitsch, W.J. (1992). Ecological indicators for ecological engineering in wetlands. In: McKenzie, D. H., E.D. Hyatt, and V.J. McDonald (eds). Ecological Indicators. Volume 1. Elsevier: Barking England. Pp. 537-558.

Mitsch, W.J., and K.M. Wise. (1998). Water quality, fate of metals, and predictive model validation of a constructed wetland treating acid mine drainage. Water Research. 32: 1888-1900.

Mitsch, W. J. and J.G. Gosselink. (2000). Wetlands: Third Edition. New York: John Wiley and Sons, Inc.

Mitchell, L.K. and A.D. Karathanasis. (1995). Treatment of metal-chloride-enriched wastewater by simulated constructed wetlands. Environmental Geochemistry and Health. 17: 119-126.

Moshiri, Gerald. (1993). Constructed Wetlands for Water Quality Improvement. CRC Press, Inc.: Boca Raton, Florida.

Murphy, S.D. (1999). ERS 380 Restoration Ecology Course Reader. University of Waterloo: Waterloo.

National Academy Press (NAP). (1995). Wetlands: Characteristics and Boundaries. National Academy Press: Washington D.C.

National Rivers Authority. (1992). Constructed wetlands to ameliorate metal-rich mine waters R&D Note 102. NRA, Bristol, UK.

Natural Resources Canada (NRC). (2002). Learning Resources: Wetlands [online]. Available:
http://atlas.gc.ca/site/english/learning_resources/wetlands/index.html [2002, 17 March].

Newcomb, Lawrence. (1977). Newcomb's Wildflower Guide. Little, Brown and Company Limited: Toronto, ON.

Noronha, J. (1999). Scavengers and Decomposers in an Industrial Park System: A Case Study of Burnside Industrial Park. Dalhousie University, School for Resource and Environmental Studies: Halifax, Nova Scotia:

North Carolina State University. (2002). Specific Wetland Types and Associated Plants: Soft Rush (Juncus effusus) [online].
Available:
http://www.soil.ncsu.edu/lockers/Broome_S/vmmiller/juncus_2.htm [2002, 14 January].

Osmond, D.L., D.E. Line, J.A. Gale, R.W. Gannon, C.B. Knott, K.A. Bartenhagen,, M.H. Turner, S.W. Coffey, J. Spooner, J. Wells, J.C. Walker, L.L. Hargrove, M.A. Foster, P.D. Robillard, and

D.W. Lehning. (1995a). Wetland Management: WATERSHEDSS: Water, Soil and Hydro-Environmental Decision Support System [online].
Available:
http://h2osparc.wq.ncsu.edu/info/wetlands/manage.html [2002, 2 February].

Osmond, D.L., D.E. Line, J.A. Gale, R.W. Gannon, C.B. Knott, K.A. Bartenhagen,, M.H. Turner, S.W. Coffey, J. Spooner, J. Wells, J.C. Walker, L.L. Hargrove, M.A. Foster, P.D. Robillard, and D.W. Lehning. (1995b). Mining and Acid Mine Drainage: WATERSHEDSS: Water, Soil and Hydro-Environmental Decision Support System [online].
Available:
http://h2osparc.wq.ncsu.edu/wetland/aqlife/mining.html#amdm[2002, 2 February].

Osmond, D.L., D.E. Line, J.A. Gale, R.W. Gannon, C.B. Knott, K.A. Bartenhagen,, M.H. Turner, S.W. Coffey, J. Spooner, J. Wells, J.C. Walker, L.L. Hargrove, M.A. Foster, P.D. Robillard, and D.W. Lehning. (1995c).Biomonitoring WATERSHEDSS: Water, Soil and Hydro-Environmental Decision Support System [online].
Available:
http://h2osparc.wq.ncsu.edu/info/biomon.html [2002, 2 February].

Peckarsky, B.L., P.R .Fraissinet, M.A. Penton, and D.J. Conklin Jr. (1990). Freshwater Macroinvertebrates of Northeastern North America. Cornell University Press: London.

Perry, A., and R.L.P. Kleinmann. (1991). The use of constructed wetlands in the treatment of acid mine drainage. Natural Resource Forum. 15(3): 178-184.

Plafkin, J.L. M.T. Barbour, K.D. Porter, S.K. Gross, R.M. Hughes. (1989). Rapid Assessment Protocols for Use in Streams and Rivers: Benthic Macroinvertebrates and Fish. D.M. Rosenberg. and V. H. Resh (eds.). Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman & Hall: New York, NY.

Price, J.S.; and J.M. Waddington. (2000). Advances in Canadian wetland hydrology and biogeochemistry. Hydrological Processes. 14(9): 1579-1589.

Qian, Jin-Hong, Adel Zayed, Yong-Liang Zhu, Mei Yu, and Norman Terry. (1999). Phytoaccumulation of Trace Elements by Wetland Plants: III. Uptake and accumulation of ten trace elements by twelve plant species. Journal of Environmental Quality. 28: 1448-1445.

Ramsar Convention Bureau for the Protection of Wetlands. (2001). Ramsar Information Paper no. 1: What are Wetlands? [online].
Available:
http://www.ramsar.org/about_infopack_1e.htm [2003, 28 January].

Ramsar Convention Bureau for the Protection of Wetlands. (2002a). Contracting Parties to the Ramsar Convention on Wetlands [online].
Available:
http://www.ramsar.org/key_cp_e.htm [2002, 19 December].

Ramsar Convention Bureau for the Protection of Wetlands. (2002b). Background Papers on Wetland Values and Functions [online].
Available:
http://www.ramsar.org/values_intro_e.htm [2002, 19 December].

Raskin, Ilya and Burt Ensley. (2000). Phytoremediation of Toxic Metals: Using Plants to Clean up the Environment. John Wiley and Sons Inc.: New York, NY.

Reagin, Misty. (2002). Nature's filter: Constructed wetlands. The American City and County. 117(6): 20-26.

Reed, S.C., R.W. Crites, and E.J. Middlebrooks. (1995). Natural Systems for Waste Management and Treatment. McGraw-Hill: New York, NY.

Reid, H.E. (1995). Wasted Opportunities: A Case Study of the Waste Diversion Barriers facing Small Businesses in the Burnside Industrial Park, Dartmouth, Nova Scotia. Dalhousie University, School for Resource and Environmental Studies: Halifax, Nova Scotia.

Richardson, C.J. and D.S. Nichols. (1985). Ecological analysis of wastewater management criteria in wetland ecosystems. In: P.J. Godfrey, E.R. Kaynor, S. Pelczarski and J. Benforado (eds.). Ecological considerations in wetlands treatment of municipal wastewaters. VanNostrand Reinhold Company: New York, N.Y. Pp. 351-391.

Robinson, H.D. (1995). A Review of the Composition of Leachates from Domestic Wastes in Landfill Sites. UK Department of the Environment, Reference: DE0918A/FR1.

Roemheld, V. and F. Awad. (2000). Significance of root exudates in acquisition of heavy metals from a contaminated calcareous soil by graminaceous species. Journal of Plant Nutrition. 23(11-12): 1857-1866.

Romheld, V., and H. Marschner. (1986). Mobilization of iron in the rhizosphere of different plant species. Advances in Plant Nutrition. 2:155-204.

Rook, E.J.S. (2002). Aquatic Wetland Plants [online].
Available:
http://www.rook.org/earl/bwca/nature/aquatics [2002, 3 March].

Runesson, U.T. (2002). Common_herb species of the northwest forest [online].
Available:
http://www.borealforest.org/herbs [2001, 19 September].

Rushbrook, P., and M.Pugh. (1999). Solid Waste Landfills in Middle- and Lower-Income Countries: A Technical Guide to Planning, Design, and Operation. World Bank Technical Paper No. 426. The World Bank: Washington D.C.

Sample, B.E., G. W. Sutter II, M.B. Sheaffer, D.S. Jones, and R.A Efroymson. (1997). Ecotoxicological Profiles for Selected Metals and other Inorganic Chemicals. U.S. Department of Energy: Tennessee.

Sanders, F., S. Gustin, and P. Pucel. (2000). Natural treatment of CMB produced water: Field observations [online].
Available:
http://www.ipams.org/sanders.pdf [2003, 28 March].

Saunders, G.L. (1989). Trees of Nova Scotia: A Guide to the Native and Exotic Species. Forestry Canada, Department of Lands and Forests: Halifax, N.S.

Schnoor, J.L. (2002). Phytoremediation of Soil and Groundwater. Evaluation Report TE-02-01. Center for Global and Regional Environmental Research and the Dept. of Civil and Environmental Engineering, University of Iowa for the Ground-Water Remediation Technologies Analysis Center Technology: Iowa City.

Sencindiver, J.C., and D.K. Bhumbla. (1988). Effect of cattail (Typha) on metal removal from mine drainage. Mine drainage and surface mine reclamation. Info. Circ. 9183. U.S. Bureau of Mines, Pittsburgh, PA. Pp.359-366.

Shen, H., X Yan, M. Zhao, S. Zheng, and X. Wang. (2002). Exudation of organic acids in common bean as related to mobilization of aluminum- and iron-bound phosphates. Environmental and Experimental Botany. 48(1): 1-9.

Skousen, J., A., K. Sexstone, J. Garbutt, and J. Sencindiver. (1994). Acid mine drainage treatment with wetlands and anoxic limestone drains. In: D.M. Kent (ed.). Applied Wetland Science and Technology. Lewis Publishing: Boca Raton, FL. Pp. 263-281.

Skousen, J.A. (1998). Overview of Passive Systems for Treating Acid Mine Drainage [online].
Available:
http://www.wvu.edu/~agexten/landrec/passtrt/passtrt.htm [2002, 8 December].

Sharpley, A.N., S.C. Chapra, and R. Wedepohl. (1994). Managing Agricultural Phosphorus for Protection of Surface Waters: Issues and Options. Journal of Environmental Quality. 23: 437-451.

Shaw, S.P. and C.G. Fredine. (1999). Wetlands of the United States: their extent and their value to waterfowl and other wildlife, Circular 39 [online].
Available:
http://www.npwrc.usgs.gov/resource/1998/uswetlan/uswetlan.htm [2002, 9 January].

Shutes, R.B., J.B. Ellis, D.M. Revitt, and T.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 Publishing: Boca Raton, FL. Pp. 407-414.

Sibbald, J. (April, 2003). Personal Communication. Pollution Prevention Coordinator, Environment and Right of Way Services, Public Works and Transportation, Halifax Regional Municipality.

Silva, A.C., M. Dezotti, G.L. Sant'Anna Jr. (2003). Treatment and detoxication of a sanitary landfill leachate. Chemosphere. In Press, Corrected Proof, Available online 18 November 2003.

Snyder, C,D. and E.C. Aharrah. (1985). The Typha community: a positive influence on mine drainage and mine restoration. Wetlands and Water Management on Mined Lands: Proc. of a Conf. 23-24 Oct. 1985. The Pennsylvania State University: University Park, PA. Pp. 187-188.

Sommer, B. (2001). Predicting wetland response to environmental change: Using carbon cycling as a surrogate for wetland function [online].
Available:
http://www.ecu.edu.au/chs/cem/wetlands/ [2003, 19 July].

Stark, L.R., F.M. Williams, S.E. Stevens, and D.P. Eddy. (1994). Metal removal in wetland treatment systems. Proc. Int. Land Reclamation and Mine Drainage Conf. 2429 Apr. 1994. Vol. 2. Bureau of Mines SP 063-94. U.S. Dep. of Interior, Pittsburgh, PA. Pp. 89-98.

Stark, L.R and F.M. Williams. (1995). Assessing the performance indices and design parameters of treatment wetlands for H, FE, and Mn retention. Ecological Engineering. 5: 433-444.

Stevens, M.H.H and W.P. Carson. (2002). Resource quantity, not resource heterogeneity, maintains plant diversity. Ecology Letters. 5: 420-426.

Stillings, L.L., J.J. Gryta, and E.A. Ronning. (1988). Iron and manganese removal in a Typha-dominated wetland during ten months following wetland construction. Mine drainage and surface mine reclamation. Info. Circ. 9183. U.S. Bureau of Mines. Pittsburgh, PA. Pp. 317-324.

Sytsma, Kenneth J. (2002). Wisconsin Vascular Plants [online].
Available:
http://wiscinfo.doit.wisc.edu/herbarium [2002, 6 April].

Taylor, G.J. and A.A. Crowder. (1983). Uptake and accumulation of heavy metals by *Typha latifolia* in wetlands of Sudbury, Ontario region. Canadian Journal of Botany. 61: 63-73.

Tchobanoglous, G., H. Theisen, and S. Vigil. (1993). Integrated Solid Waste Management, Engineering Principles and Management Issues. McGraw-Hill, Inc.: New York, NY.

Terry, Norman and Gary Banuelos. (2000). Phytoremediation of Contaminated Soil and Water. Lewis Publishers: Boca Raton, Florida.

Thunhorst, Gwendolyn. (1993). Wetland Planting Guide for the Northeastern United States: Plants for Wetland Creation, Restoration and Enhancement. Environmental Concern Inc.: Maryland.

Tiner, R.W. (1999). Wetland Indicators: A Guide to Wetland Identification, Delineation, Classification, and Mapping. CRC Press: Boca Raton, FL.

Tousignant, E., O. Fankhauser and S. Hurd. (1999). Guidance Manual for the Construction and Operations of Constructed Wetlands for Rural Applications in Ontario [online].
Available:
http://res2.agr.ca/initiatives/manurenet/download/wetlands_manual.pdf [2002, 8 November].

Treacy, P. and P. Timpson. (1999). The Use of Wetlands to Prevent Environmental Pollution from Acid Mine Drainage [online].
Available:
http://journals.eecs.qub.ac.uk/RIA/ProcBI/1999/PB99I1/PDF/99107BI.pdf [2002, 9 December].

Underwood, A.J. (1997). Experiments in Ecology: Their logical design and interpretation using analysis of variance. Cambridge University Press: New York, N.Y.

University of Florida. (2002). Centre for Aquatic and Invasive Plants [online].
Available:
http://aquat1.ifas.ufl.edu/ [2001, 19 December].

U.S. BAMR. (2002). The Science of Acid Mine Drainage and Passive Treatment [online]. Available:
http://www.dep.state.pa.us/dep/deputate/minres/bamr/amd/science_of_AMD.htm [2003, 18 January].

USEPA. (1993). Natural Wetlands and Urban Stormwater: Potential Impacts and Management [online].
Available:
http://www.abe.msstate.edu/csd/references/stormwat.html [2002, 13 September].

USEPA. (1995). Linking Restoration Practices to Water Quality Parameters: Ecological Restoration. Office of Water, US. Environmental Protection Agency: Washington. EPA 841-F-95-007 [online].
Available:
http://www.epa.gov/OWOW/NPS/Ecology/chap3.html [2002, 19 September].

USEPA. (1996). A Citizen's Guide to Bioremediation. EPA 542-F-01-001 [online].
Available:
http://www.epa.gov/swertio1/download/citizens/bioremediation.pdf [2002, 28 October].

USEPA. (1998a). 1998 Update of Ambient Water Quality Criteria for Ammonia. EPA 822-R-98-008 [online].
Available:
http://www.epa.gov/ost/standards/nh3_rpt.pdf [2001, 29 October].

USEPA. (1998b). A Citizens Guide to Phytoremediation: Technology Fact Sheet. EPA 542-F-01-002 [online].
Available:
http://www.clu-in.org/products/citguide/phyto2.htm [2002, 28 October].

USEPA. (1999a). Phytoremediation Resource Guide. EPA 542-B-99-003 [online].
Available:
http://www.clu-in.org/download/remed/phytoresgude.pdf [2002, 28 October].

USEPA. (1999b). Manual: Constructed Wetlands Treatment of Municipal Wastewaters. EPA/625/R-99/010 [online].
Available:
http://www.epa.gov/ordntrnt/ORD/NRMRL/Pubs/2001/wetlands/625r99010.pdf [2002, 19 October].

USEPA. (2000a). Guiding Principles for Constructed Treatment Wetlands: Providing for Water Quality and Wildlife Habitat. EPA 843-B-00-003 [online].
Available:
http://www.epa.gov/owow/wetlands/construc/backgrnd.html [2002, 19 October].

USEPA. (2000b). Introduction to Phytoremediation. EPA/600/R-99/107 [online].
Available:
http://www.clu-in.org/download/remed/introphyto.pdf [2002, 28 October].

USEPA. (2000c). Wastewater Technology Fact Sheet: Free Water Surface Wetlands. EPA 832-F-00-024 [online].
Available:
http://www.epa.gov/owm/mtb/free_water_surface_wetlands.pdf [2002, 19 October].

USEPA. (2000d). Wastewater Technology Fact Sheet: Wetlands: Subsurface Flow. EPA 832-F-00-023 [online].
Available:
http://www.epa.gov/npdes/pubs/wetlands-subsurface_flow.pdf [2002, 19 October].

USEPA. (2000e). Hot Technologies: Brownfeilds technology [online].
Available:
http://www.brownfieldstech.org/technology/index.html [2002, 19 October].

USEPA. (2001). Constructed Wetlands for Wastewater Treatment and Wildlife Habitat [online].
Available:
http://www.epa.gov/owow/wetlands/construc/backgrnd.html [2002, 19 October].

USEPA. (2002a). Technical Approaches to Characterizing and Redeveloping Brownfields Sites: Municipal Landfills and Illegal Dumps.EPA/625/R-02/002 [online].
Available:
http://www.epa.gov/ORD/NRMRL/Pubs/625R02002/625r02002chap2.pdf [2002, 20 October].

USEPA. (2002b). Wetlands Overview. EPA 843-F-01-002a [online].
Available:
http://www.epa.gov/owow/wetlands/ [2002, 14 September].

USEPA. (2002c). Types of Wetlands [online].
Available:
http://www.epa.gov/owow/wetlands/types/ [2002, 14 September].

USEPA. (2002d). Functions and Values of Wetlands. EPA 843-F-01-002c [online].
Available:
http://www.epa.gov/watertrain/wetlands/ [2002, 14 September].

USEPA. (2002e). Threats to Wetlands. EPA 843-F-01-002d [online].
Available:
http://www.epa.gov/owow/wetlands/facts/threats.pdf [2002, 14 September].

USEPA. (2002f). 17 Case Studies: Free Water Surface Constructed Wetlands Systems [online].
Available:
http://www.epa.gov/owow/wetlands/construc/freewatr.html [2003, 17 March].

USEPA. (2002g). Biological Assessment of Wetlands [online].
Available:
http://www.epa.gov/owow/wetlands/bawwg/ [2003, 22 January].

USEPA. (2002h). Methods for Evaluating Wetland Condition: Developing Metrics and Indexes of Biological Integrity. EPA-822-R-02-016 [online].
Available:
http://www.epa.gov/waterscience/criteria/wetlands/6Metrics.pdf [2003, 22 January].

USEPA. (2002i). Methods for Evaluating Wetland Condition: Wetlands Classification. EPA-822-R-02-017 [online].
Available:
http://www.epa.gov/waterscience/criteria/wetlands/7Classification.pdf [2003, 22 January].

USEPA. (2002j). Methods for Evaluating Wetland Condition: Study Design for Monitoring Wetlands. EPA-822-R-02-015 [online].
Available:
http://www.epa.gov/waterscience/criteria/wetlands/4StudyDesign.pdf [2003, 22 January].

USEPA. (2002k). Methods for Evaluating Wetland Condition: Using Vegetation To Assess Environmental Conditions in Wetlands. EPA-822-R-02-020 [online].
Available:
http://www.epa.gov/waterscience/criteria/wetlands/10Vegetation.pdf [2003, 22 January].

USEPA. (2002l). Methods for Evaluating Wetland Condition: Developing an Invertebrate Index of Biological Integrity for Wetlands. EPA-822-R-02-019 [online].
Available:
http://www.epa.gov/waterscience/criteria/wetlands/9Invertebrate.pdf [2003, 22 January].

USEPA. (2002m). Methods for Evaluating Wetland Condition: Introduction to Wetland Biological Assessment.EPA-822-R-02-014 [online].
Available:
http://www.epa.gov/waterscience/criteria/wetlands/1Introduction.pdf [2003, 22 January].

Van Cauwenberghe, L. and Roote, D.S. (1998). In Situ Bioremediation.. Doc, No. TO-98-01 [online].
Available:
http://www.gwrtac.org/pdf/insbio_o.pdf [2002, 5 November].

Vollenweider, R.A. (1968). Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters, with Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication. DAS/CSI/68.27. Organization for Economic Cooperation and Development: Paris.

Voshell, R. J. (2002). A Guide to Freshwater Invertebrates of North America. McDonald and Woodward Publishing: Blacksburg, Virginia.

Wetlands Mapping Protection Program (WMPP). (1988). Nova Scotia Wetlands Atlas. Canadian Wildlife Service, Environment Canada: Sackville, N.S.

Wieder, R.K. (1988). Determining the capacity for metal retention in man-made wetlands constructed for treatment of coalmine drainage. Mine drainage and surface mine reclamation. Info. Circ. 9183. U.S. Bureau of Mines: Pittsburgh, PA. Pp. 375-381.

Wieder, R.K. (1992). The Kentucky wetlands project: A field study to evaluate man-made wetlands for acid coal mine drainage treatment. Final Report to the U.S. Office of Surface Mining, Villanova University: Villanova, PA.

Whiting,G.J. and J.P. Chanton. (2001). Greenhouse carbon balance of wetlands: Methane emission versus carbon sequestration. Tellus B. 53: 521-528.

Whiting, S.N., and N. Terry. (1999). The Tennessee Valley Authority constructed wetland at Widows Creek: Role of vegetation in the removal of trace elements. TR-114220. Electric Power Res. Inst.: Palo Alto, CA.

Whitinger, D. (2002). The Plants Database [online].
Available:
http://plantsdatabase.com/ [2002, 13 March].

Witthar, S.R. (1993). Wetland Water Treatment Systems. In: G.A. Moshiri (ed.). Constructed Wetlands for Water Quality Improvement. Lewis Publishers: Boca Raton, Florida. Pp. 147-155.

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 number 269. Canadian Wildlife Service, Environmental Conservation Branch, Ontario Region: Hamilton, Ontario.

Ye, Z.H., S.N. Whiting, Z.-Q. Lin, C.M. Lytle, J.H. Qian, and N. Terry. (2001a). Removal and distribution of iron, manganese, cobalt, and nickel within a Pennsylvania Constructed Wetland Treating Coal Combustion By-Product Leachate. Journal of Environmental Quality. 30:1464-1473.

Ye, Z.H., S.N. Whiting, Z.-Q. Lin, C.M. Lytle, J.H. Qian, and N. Terry. (2001b). Wetlands and aquatic processes: Trace element removal from coal ash leachate by a 10-year-old constructed wetland. Journal of Environmental Quality. 30: 1710-1719.

Younger P.L, and L.C. Batty. (2001). The Role of Vegetation in Wetland Treatment of Mine Waters. Proceedings of the 1st IMAGE-TRAIN Cluster Meeting [online]. Available:
http://www.image-train.net/products/proceedings_first/chapter_19.pdf [2002, 15 January].

Zinck, M. (1998). Roland Flora of Nova Scotia, Vol. I and II. Nimbus Publishing and the Nova Scotia Museum: Halifax, Nova Scotia.

APPENDIX A. BURNSIDE TREATMENT WETLAND CHEMICAL LOADING RATE, HYDRAULIC RESIDENCE TIME, AND PRELIMINARY SIZING CALCULATIONS

Table A.1. Burnside Wetland Chemical Loading Rates.

PARAMETRES		AMMONIA	IRON	MANGANESE
Design flow (m^3/d)[a]	Q=	947.1	947.1	947.1
Influent concentration (mg/L)	C_i=	1.26	6.17	1.80
Area of system (m^2)	A=	5000	5000	5000
Chemical loading rate (g/m^2 day)=		**CLR=Q*C_i/A**		
		0.24	**1.17**	**0.34**

[a]Design flow based on the maximum monthly (November) rainfall of 13.09 cm and the watershed area of 55.1 ha (EC, 2003).

Table A.2. Burnside Wetland Hydraulic Residence Time (HRT).

Design flow (m^3/d)[a]	Q=	947.1
Area of system (m^2)	A =	5000
Porosity of the medium	n =	1.0[b]
Depth of submergence (m)	d =	1.0
Hydraulic Retention Time	**HRT (days) = And /Q**	
	5.28 days	

[a]Design flow based on the maximum monthly (November) rainfall of 13.09 cm and the watershed area of 55.1 ha (EC, 2003).

[b]Porosity as suggested by Kadlec and Knight, 1996.

Table A.3. Burnside Wetland Preliminary Sizing Calculations.

PARAMETRES		AMMONIA	IRON	MANGANESE
Design flow (m^3/d)[a]	Q=	947.1	947.1	947.1
Influent concentration (mg/L)	C_i=	1.26	6.17	1.80
Target effluent concentration (mg/L)	C_e=	0.35	0.30	1.15
Wetland background limit (mg/L)[b]	C*=	0.06	0.27	0.15
Reduction fraction target	F_e=	0.73	0.95	0.36
Reduction fraction to background	F_b=	0.95	0.96	0.92
Areal Rate constant[c]	k=	22	35	35
Required wetland area (ha)	**A= (0.0365*Q/k)*ln(C_i-C*/C_e-C*)**			
C_i -C* =		1.20	5.90	1.65
C_e – C*=		0.29	0.03	1.00
(C_i-C*)/(C_e-C*) =		4.14	196.67	1.65
ln ((C_i-C*)/(C_e-C*)) =		1.42	5.28	0.50
((0.0365 * Q)/k)=		1.57	0.99	0.99
	A=	**2.23**	**5.22**	**0.50**
Required area for active treatment (ha)=				**5.22** (largest area)
Total required area with peripherals (~25%) (ha)=				**6.53**

[a]Design flow based on the maximum monthly (November) rainfall of 13.09 cm and the watershed area of 55.1 ha (EC, 2003).

[b]Background figures derived from the control wetland in Waverley game park.

[c]Areal rate constant as suggested by Kadlec and Knight, 1996.

APPENDIX B. FINAL SITE PLANTING LISTS

Table B.1. Final Site Planting List for Emergent Aquatic Species in Cell Interiors, Littoral Zones and Berms (Thunhorst, 1993; Zinck, 1998; Rook, 2002; Crow and Hellquist, 2000; Schnoor, 2002; Davis, 1996; and Larson, 1993).

☐ High phytoremediation potential

☐ Effective at sediment and soil stabilization

☐ Suited for habitat facilitation

☐ Suited for public deterrence

☐ Increase diversity, enhance habitat or aesthetics

SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Alisma plantago-aquatica	Water plantain	-Obligate aquatic species with white flowers -Up to 3.5ft -UC1=1ft -Flowers June- Sept. -Permanently inundated -Can tolerate salinity[a] -Perennial: reproduces vegetatively by rhizome and sexually by seed-grows well from seed -Food source for waterfowl	 (Galbrand, 2002)
Calamagrostis canadensis	Canada bluejoint grass	-Tall herbaceous grass forming clumps -Up to 5ft -UC1=0.5ft -Flowers June-Aug. -Soil stabilizer. Tolerates permanent inundation up to 0.5 ft. -Perennial: reproduces vegetatively by rhizome and sexually by seed-grows well from seed -Food for mammals, muskrats, deer	 (Sytsma, 2002)
Carex brunnescens	Grey sedge	-Slender, facultative sedge forming dense clumps -6 inches to 2 ft -Flowers June –Aug. -Perennial	 (Whitinger, 2002)
Carex crinita	Fringed sedge	-Facultative sedge growing in clumps with dropping terminal flower heads -Up to 3 ft tall -Flowers May- Aug. -Perennial	 (Galbrand, 2003)

Table B.1. Continued.

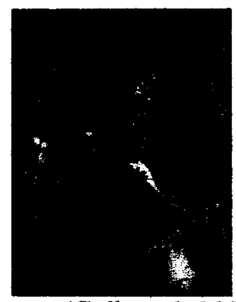
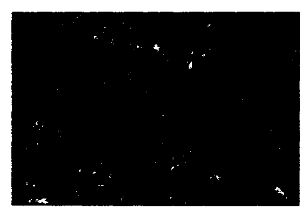
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Carex lurida	Yellow-green sedge	-Facultative sedge with unique, spiked flower heads -1.5-3ft tall -Flowers June – October -Tolerates shade -Perennial -Food for various songbirds	 (Galbrand, 2003)
Carex pseudocyperus	Cypernus sedge	-Facultative sedge with robust terminal flower heads -Up to 2.5 ft tall -Flowers June – Aug. -Tolerates shade	 (Whitinger, 2002)
Carex stipata	Awl-fruited sedge	-Obligate wetland sedge with prominent terminal flower -1 to 3ft. high. -UC1=0.5ft -Flowers May –Aug. -Tolerate inundation 25% of growing season. Tolerates drought. -Perennial: reproduces vegetatively by rhizome and sexually by seed Food for various songbirds and black duck	 (Galbrand, 2003)
Eleocharis acicularis	Needle spike rush	-Indiscreet spike rush with tiny terminal flower heads growing in dense mats -1 to 2ft. -UC1=1ft. -Flowers June-Oct. -Dense rhizomous roots excellent for erosion control -Perennial: reproduces vegetatively by rhizome and sexually by seed -Food for waterfowl	 (Galbrand, 2002)
Glyceria striata	Fowl mannagrass	-Large, erect obligate wetland species growing in tussocks -Up to 4 ft -Food for waterfowl, muskrat, deer -Perennial: reproduces vegetatively by rhizome and sexually by seed. -Seeds easily collected from heads in late fall -Unlike *G. grandis*, this species intermixes well with other species	 (Galbrand, 2003)

Table B.1. Continued.

SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Glyceria grandis	Reed meadow grass	-Large, erect obligate wetland species with showy head -3 to 5 feet high -UC1=2ft -Tolerates inundation in spring but requires draw-down -Perennial: reproduces vegetatively by rhizome and also by stolon and sexually by seed. -Browsed by waterfowl, muskrat and deer -Does not compete well with other species.	 (Sytsma, 2002)
Iris versicolor	Blue Flag	-Obligate wetland iris with beautiful blue/violet flowers -Up to 4 ft -UC1=0.5ft -Flowers June- July -Tolerant of high nutrient levels and fluctuating water levels -ideal mix of tolerance and beauty -Caution: severe dermatitis may result from handling rhizome -food for wildfowl and waterfowl	 (Galbrand, 2001)
Juncus brevicaudatus	Tweedy's rush	-Small, slender facultative sedge with reddish flowers -Up to 2 ft tall -Flowers June –Sept. -Considered rare Species of Concern in some northern states including WY, ND, and TN.	 (Galbrand, 2002)
Juncus canadensis	Canada sedge	-Facultative rush -Up to 1 to 3ft -Flowers July-Oct. -Perennial: reproduces vegetatively by rhizome and sexually by seed	 (Galbrand, 2002)

Table B.1. Continued.

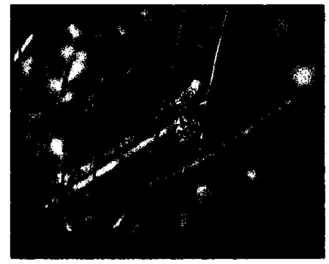
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Juncus effusus	Soft rush	-Facultative wetland rush, evergreen with pithy, leafless stems round in cross section supporting a small lateral, inflorescence flower (approx. 1 in. long). Flower sustains compact spikelets of 30 to 100 small flowers per branch that are brown in colour, each on own stalk. The fruit is a capsule bearing numerous seeds. An erect, rolled, pointed involucral bract, 1 to 2 in. in length, appears as a continuation of the stem above the flower. Stems emerge from thick, lengthy, finely divided rhizomes forming tussocks. -Up to 6 feet tall -UC1=0.5ft -Flowers July-Sept. -Max water depth 3 in. can tolerate permanent inundation but prefers dry-down. Requires full sun. Prefers neutral soils and waters ranging from pH of 4 to 6. -Nutrient and metal hyperaccumulator, deep roots (2-3 feet) = support microbial growth and stabilize soils. Phytovolatilization of VOCs -Perennial: reproduces vegetatively by rhizome and sexually by seed (up to 1,500,000 seeds per ounce). Seed dispersal primary means of reproduction. Seeds can remain viable for greater than 60 years in sediments. -A transplanted unit must contain at least 3 to 5 culms (stems) to grow successfully -Food for waterfowl, song and terrestrial birds, muskrat, good cover, supports spawning ground for fish	 (Galbrand, 2002)

Table B.1. Continued.

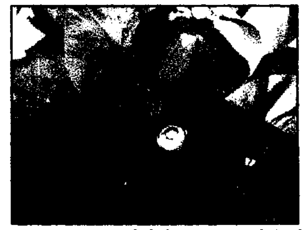
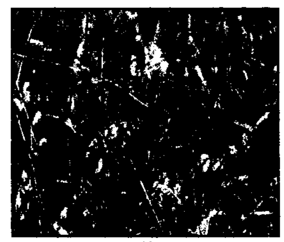
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Nuphar variegata	Cowlily	-Aquatic obligate wetland species with bright yellow flower -Up to 16 inches high -UC1=0.5ft -Flowers May-Oct. -Grows in waters up to 6 ft deep. -Shades water body= reduce water temp. Tolerates acidic water and shade. -Perennial: reproduces vegetatively by rhizome and sexually by seed -Food for waterfowl, porcupine, deer and muskrat	 (Whitinger, 2002)
Polygonum pensylvanicum	Pinkweed	-Large herbaceous facultative wetland plant with reddish stems and pink flowers -Up to 6.5 ft -Flowers May-Oct. -Tolerates poor soil fertility, good for erosion control. Tolerates permanent inundation. -Annual: reproduces sexually by seed -Food and cover for songbirds, waterfowl, game birds and various mammals	 (Galbrand, 2002)
Pontederia cordata	Pickerelweed	-Obligate fringing aquatic species with bright violet flower -Up to 3.5ft -UC1=1ft -Flowers June-Nov. -Grows in water up to 12 in. Permanent inundated, tolerates shade -Effective shoreline stabilisers -Related to water hyacinth. Losses much water through transpiration therefore potential phytoextraction . Potential ammonia remover -Perennial: reproduces vegetatively by rhizome and sexually by seed -seed food for waterfowl, foliage and root foot for muskrat and Canada goose, nectar for butterflies	 (University of Florida, 2002)

Table B.1. Continued.

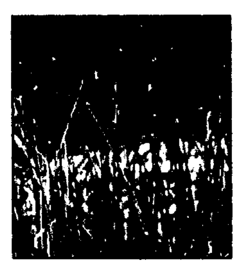
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Scirpus cyperinus	Woolgrass	-Facultative wetland bulrush with large 'woolly' head in fall, abundant basal leaves, often growing in "tussocks". Upper part of its stems obtusely 3-angled; with enclosed leaf sheaths and flat, smooth leaves which droop near the tips. Leaves are <1 cm wide and are mostly crowded near base. The dense inflorescence is terminal with spikelets subtended by 3 to 6 involucral bracts which are leaf like and not appearing as a continuation of the stem. The numerous spikelets, which are ovate and approximately 0.6 cm long and 0.4 cm wide, are green in their flowering stage, becoming brown in late Aug. to Oct. The achene which is whitish, are elongated and 0.6 to 1 mm long has 6 long, curly bristles when mature, which cumulatively add to give the spikelets their characteristically "woolly" look. -4 to 6.5m high. -Flowers Aug.-Sept. -UC1=1ft -Nutrient and metal hyperaccumulator. Tolerates waters with low pH. Deep, prolific roots (up to 3 feet) = support microbial growth and stabilize soils. Drought and flood tolerant (inundation up to 25% of growing season). Prefers full sun, but tolerates shade. Tolerates soils with pH ranging from 4.8 to 7.2. -Perennial: reproduces vegetatively by rhizome and sexually by seed (up to 1,800,000 seeds per ounce) -Food source for much waterfowl	 (Galbrand, 2002)
Scirpus pungens	Common three square	-Facultative wetland bulrush with characteristic triangular-profiled stem -Up to 4 ft. high -UC1=2ft -Flowers June-Sept. -Tolerates permanent inundation -Soil stabilizer, drought tolerant, tolerates brackish water -Perennial: reproduces vegetatively by rhizome and sexually by seed -Cover and food for various waterfowl	 (University of Florida, 2002)

Table B.1. Continued.

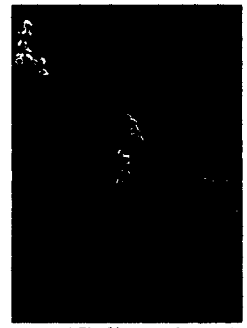
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Scirpus validus	Soft stem bulrush	-Obligate wetland bulrush with stem round in cross-section -Up to 10ft. high -UC1=2ft -Flowers June-Sept. -Tolerates permanent inundation and brackish water -Prefers pH between 6.5 and 8.5, requires full sun -Potential metal accumulator -Perennial: reproduces vegetatively by rhizome and sexually by seed -Cover and food for various waterfowl and fish	 (Galbrand, 2003)
Typha angustifolia	Narrow-leaved cattail (not included in phyto-screening due to limited abundance in model site)	-Large obligate wetland species with brown terminal spike -Up to 10 ft. high. -UC1=2ft -Flowers June -July -Max water depth 12 in- tolerates permanent inundation and drought. -Nutrient and metal hyperaccumulator, moderate root depth (1-2 feet) = support microbial growth and stabilize soils. Tolerates low pH. Phytovolatilization of VOCs -Perennial: reproduces vegetatively by rhizome and sexually by seed -Food for waterfowl and terrestrial birds, good cover, supports spawning ground for fish -tubers fav. food of muskrat and beaver WARNING: Possible Invasive. Do not establish until all other populations have been established	 (University of Florida, 2002)

Table B.1. Continued.

SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Typha latifolia	Broad-leaved cattail	-Large obligate wetland species with round, erect, tapering stems up to 2.5 cm in diameter at the base, with thick, white, creeping rhizomes up to 2 ft. in length. It has 12 to 16 long, narrow, alternate, greyish green basal leaves that are 6 to 23 mm wide, erect and twisting, D-shaped and pithy in cross section, which sheath at the base of the stem. The flower structure is a terminal, cylindrical spike-like inflorescence, with the pistillate and staminate portions usually contiguous. The pistillate which is 2.5 to 20 cm long and at least 2.5 cm in diameter when mature is located below narrower staminate, which is 7 to 13 cm long. The pistillates are green when flowering, and various browns when fruiting. In fruiting, the pistillate bears innumerable, tiny, downy hair-tufted and persistent stigma-bearing achene, which are fusiform, light brown, ellipsoidal, about 1 mm long and designed to float in wind or water. These seeds are released as the pistillate gradually breaks apart in the fall, taking on the appearance of a wooly mass. -Up to 10 ft. high. -UC1=2ft -Flowers May-June -Max. water depth- 12-18 in but prefers <6 in. -Tolerates permanent inundation and drought. -Nutrient and metal hyperaccumulator, moderate root depth (1-2 feet) = support microbial growth and stabilize soils[a]. Tolerates brackish conditions. Phytovolatilization of VOCs -Perennial: reproduces vegetatively by rhizome and sexually by seed (up to 268,000 seeds per spike) -Food for waterfowl, song and terrestrial birds, supports spawning ground for fish -tubers fav. food of muskrat and beaver WARNING: Possibly Invasive. Do not establish until all other populations are established. Also possible allelopathic characteristics	 (Galbrand, 2002)

Table B.2. Final Site Planting List for Terrestrial Vascular Plants in Wetland Edges and within Buffer Areas (Thunhorst, 1993; Runesson, 2002; Rook, 2002; and Zinck, 1998).

☐ High phytoremediation potential

☐ Effective at sediment and soil stabilization

☐ Suited for habitat facilitation

☐ Suited for public deterrence

☐ Increase diversity, enhance habitat or aesthetics

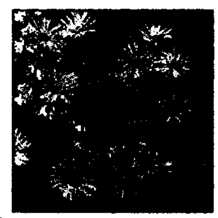
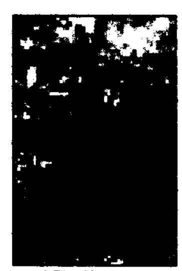
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Aralia nudicaulis	Wild sarsapailla	-Herbaceous plant with small, white; in globular flowers and blue-black fruits -Up to 50 cm high -Flowers May-July -Food for deer, songbirds, and grouse -Perennial: reproduces vegetatively by rhizomes and sexually by seed	 (Galbrand, 2002)
Aster spp.	Asters	-Herbaceous plant with variable flower colours -Up to 6 feet -Generally tolerate shade and drought -Perennial: reproduces vegetatively by rhizomes and sexually by seed -food for many small game animals	 (University of Florida, 2002)
Cornus canadensis	Bunchberry	- Small 4 petalled white flower, bright red unpalatable fruit -8 in. tall -Flowers May-June -Shade loving -Perennial: reproduces vegetatively by runners and sexually by seed -fruit forage material for wildlife	 (Galbrand, 2002)
Cypripedium acaule	Pink lady's slipper	-Bright pink orchid -8 in. tall -Flowers June to Aug.	 (Galbrand, 2002)

Table B.2. Continued.

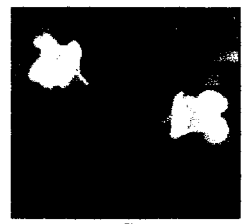
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Frageria virginiana	Strawberry	-Small plant with white flowers -4 inches tall -Flowers late May-June -Perennial -Edible berries enjoyed by humans and wildlife	 (Galbrand, 2002)
Maianthemum canadense	Wild Lily-of-the-valley	-Small plant with white flowers -3 to 8 in. tall -Flowers May – June -Shade tolerant -Perennial: reproduces vegetatively by rhizomes and sexually (but poorly) by seed	 (Galbrand, 2002)
Oenothera biennis	Evening primrose	-Robust herbaceous plant with yellow flowers -2 -5 feet -Flowers June-Aug. -Perennial	 (Sytsma, 2002)
Onoclea sensibilis	Sensitive fern	-Stout, light-green fern -Up to 3.5 feet high -UC1=1ft -Tolerates acidic soils (>4.5) and permanently saturated soils -Perennial: reproduces vegetatively by rhizomes and sexually by spores -Food source for terrestrial birds and mammals	 (Galbrand, 2002)
Osmunda cinnamomea	Cinnamon fern	-Tall robust fern growing in circles -Up to 6 feet high -UC1=0.5 -Tolerates drought and permanently saturated soils. Transplants easily. -Perennial: reproduces vegetatively by rhizomes and sexually by spores -Food source for terrestrial birds and mammals	 (Galbrand, 2002)

Table B.2. Continued

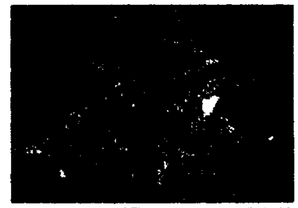
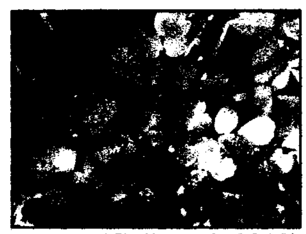
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Pteridium aquilinum	Bracken fern	-Common fern with characteristic 3-part leaf -Up to 3 feet high -Perennial: reproduces vegetatively by rhizomes and sexually by spores -releases of allelopathic chemicals to dominate other vegetation, but is not very threatening	 (Sytsma, 2002)
Solidago spp.	Goldenrod	-Tall herbaceous plants with yellow flowers -Up to 5 ft -Flowers from July through September -Perennial: reproduces vegetatively by rhizomes and sexually by seed -Grazed by deer, game birds, songbirds	 (Galbrand, 2002)
Thalictrum pubescens	Meadow rue	-Tall herbaceous plant with 'starburst' flowers. -3-8 feet tall -Flowers July –Sept. -Perennial	 (Galbrand, 2002)
Trientalis borealis	Starflower	-Small six-petalled white flower -3 in tall -Flowers May-Aug. -Tolerant of poor soil conditions -Perennial: reproduces vegetatively by rhizomes and sexually by seed	 (Galbrand, 2002)
Viola conspersa	Dog violet	-Tiny violet with purple flower -2-6 in tall -Flowers May-June -Perennial	 (Galbrand, 2001)
Viola macloskeyi	Small White Violet	-Tiny violet with white flower -2-6 in tall -Flowers May-June -Perennial	 (Galbrand, 2002)

Table B.3. Final Site Planting List for Shrubs in Buffer Areas (Thunhorst, 1993; Runesson, 2002; Rook, 2002; Zinck, 1998; and MSB, 2001).

- High phytoremediation potential
- Effective at sediment and soil stabilization
- Suited for habitat facilitation
- Suited for public deterrence
- Increase diversity, enhance habitat or aesthetics

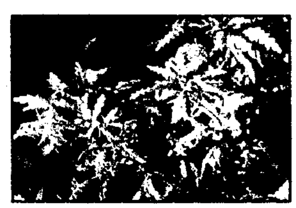
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Alnus viridis	Speckled alder	-Robust shrub with woody cones. Often grows in clumps or thickets. -Up to 6 feet high -Effective bank stabiliser -Fast spreading, transplants well	 (Sytsma, 2002)
Amelanchier arborea	Shadbush/ Wild pear	-Large shrub with abundant white flowers and edible fruit. -Up to 50 feet high -Flowers in May, fruits in June -Food source for much wildlife, nesting cover for several birds	 (Sytsma, 2002)
Aronia arbutifolia	Red chokeberry	-Large shrub with abundant unpalatable fruit. Leaves turn red in fall. -Flowers May-June, fruits Sept-Dec -Up to 3m high -Slow spread rate by suckers -Tolerates drought and shade -Food for numerous birds and mammals	 (Whitinger, 2002)
Comptonia peregrina	Sweet fern	-Stout, hardy, fragrant woody plant with bur-like seeds -Up to 4 ft high -Flowers May -Drought, shade, acidic soil and salt tolerant -Reproduces by rhizomes and seed Food for rabbits, deer in winter, and flickers. Food and cover for ruffed grouse	 (Galbrand, 2002)

Table B.3. Continued.

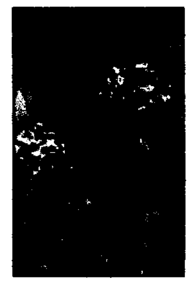
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Diervilla lonicera	Bush honeysuckle	-Low, discrete shrub with ornate yellow-orange flowers -Up to 4 ft tall -Flowers June-July	 (Galbrand, 2002)
Kalmia angustifolia	Sheep laurel or Lambkill	-Stout woody evergreen with bright pink flowers -Up to 2ft high -Flowers May- Aug. -Annual growth rate <1 ft. -Extremely hardy, grows in poor soils, can adapt to wet soils -Poisonous if consumed	 (Galbrand, 2002)
Ledum groenlandicum	Labrador tea	-Low spreading evergreen shrub with spicy fragrance and white flowers -1-2.5 ft tall -Flowers May-June -Tolerates nutrient poor, acidic soils	 (Galbrand, 2002)
Prunus virginiana	Choke cherry	-Shrub or small tree with white flowers and abundant fruit -Up to 20 ft. -Flowers May- early June -Fruit desirable to wildlife, especially bees, butterflies and birds	 (Sytsma, 2002)
Rhododendron canadense	Rhodora	-Woody species with beautiful purple flowers in spring -Up to 6 ft tall -Tolerates acidic soils -Food for deer, songbirds and small mammals	 (Galbrand, 2002)

Table B.3. Continued.

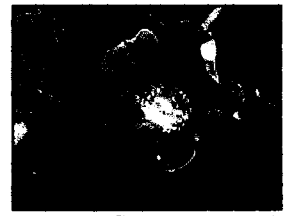
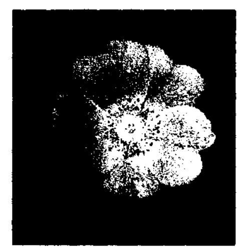
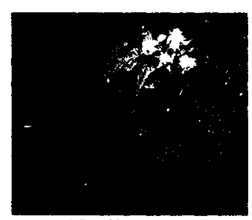
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Rosa nitida	Bristly rose	-Dense, bristly species with bright pink flowers. -Up to 7 feet high -Flowers June-Oct -Tolerates poor soil and shade -Food source for songbirds, terrestrial birds and mammals	 (Sytsma, 2002)
Rosa palustris	Swamp rose	-Thorny, obligate wetland species with large pink flowers -Up to 7 feet high -Flowers June-Oct. -Prefers full sun -Tolerates saturated soils for 75% or growing season -Food source for songbirds, terrestrial birds and fox	 (Galbrand, 2001)
Rosa virginiana	Common wild rose	-Thorny species with large pink flowers -4 to 6 ft tall -Fast growth rate -Flowers in June-July -Prefers acidic soils and full sun. Salt tolerant -Food source for songbirds, terrestrial birds and mammals	 (Sytsma, 2002)
Rubus strigosus	Raspberry	-Prickly plant with white flowers and edible fruits -Up to 4 ft high -Flowers June-July -Food for abundant forms of wildlife	 (Galbrand, 2002)
Spiraea alba var. latifolia	Meadowsweet	-Dense, white-flowered hardy shrub forming clumps or thickets -Up to 5 ft high -Flowers June-Sept. -Dense roots excellent for bank stabilization -Fast spreading -Prefers full sun, needs moist soils	 (Galbrand, 2002)

Table B.3. Continued.

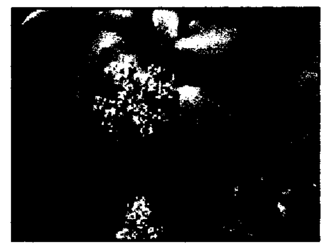
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Vaccinium angustifolium	Blueberry	-Stout woody plant with pink/white flowers and abundant, edible fruit. Foliage turns bright red in fall. -Up to 2 ft high -Flowers late May-mid-June -Tolerates acidic soils -Food for wide variety of wildlife	 (Sytsma, 2002)
Viburnum cassinoides	Witherod	-Glossy woody shrub with white flowers, edible black fruit, and brightly coloured fall foliage -5 to 15 ft tall -Flowers mid-June -Tolerates shade -Food for wide variety of wildlife, esp. songbirds -slow spreading	 (Sytsma, 2002)

Table B.4. Final Site Planting List for Trees in Buffer Areas (Saunders, 1989; Thunhorst, 1993; Schnoor, 2002; and Runesson, 2002).

- ☐ High phytoremediation potential
- ☐ Effective at sediment and soil stabilization
- ☐ Suited for habitat facilitation
- ☐ Suited for public deterrence
- ☐ Increase diversity, enhance habitat or aesthetics

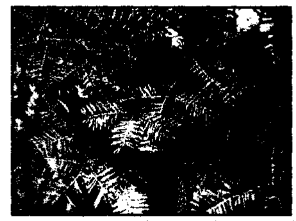
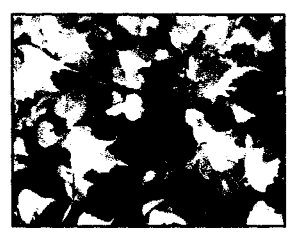
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Abies balsamea	Balsam fir	-Soft coniferous species with flat, singular plain needles -Up to 20m high -Drought tolerant, stream bank stabilizers. -Food source for birds and mammals, good noise buffer	 (Galbrand, 2002)
Acer rubrum	Red maple	-Medium-sized deciduous with red flowers in spring and red leaves in fall -Up to 40m high -Sensitive to wind and ice damage. Tolerates drought. -Fast growing early successional species -Food for birds, squirrels and deer -Noted in literature to treat leachate	 (Sytsma, 2002)
Betula papyrifera	White birch	-Small-medium sized, white-barked deciduous -Up to 20m high -Flowers late May-June -Fast growing -Young regenerating stands provide prime browse and cover for deer and moose -Good bank stabilisers	 (Sytsma, 2002)
Betula populifolia	Grey birch	-Silver-barked, deciduous tree -35-50 ft high. -Flowers April, fruits Sept-Dec -Fast growing, 2-2.5 ft per year. Spreads by suckers. Easily transplanted. -Tolerates poor soil conditions, drought and flooding -Good bank stabilisers -Food source for waterfowl, songbirds, squirrel and deer	 (Galbrand, 2002)

Table B.4. Continued.

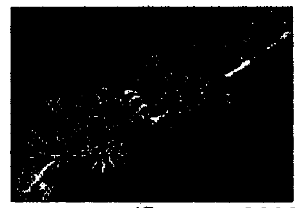
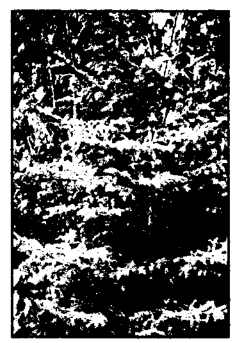
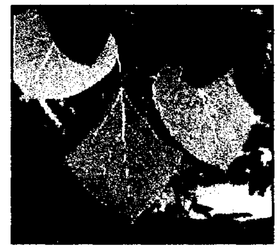
SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Fraxinus americana	White ash	-Hardy deciduous with characteristic compound leaves -Up to 20m high -Establishes quickly -Seeds eaten by birds, shoots by deer and beaver	 (Galbrand, 2002)
Larix laricina	Tamarack	-Uniquely deciduous conifer species with soft needles -Up to 20m -Shade intolerant, prefers moist organic soils -Fast growing, shallow roots spreading wide -Needles food for rabbits, seeds eaten by birds and grouse, branchlets browsed by deer	 (Sytsma, 2002)
Picea glauca	White spruce	-Medium to large-sized coniferous with sharply pointed, stout needles -Up to 40m -Habitat for birds and mammals, food for squirrels and porcupines, good noise buffer -slow growing	 (Galbrand, 2002)
Picea mariana	Black spruce	-Small to medium-sized conifer with sharply pointed, stout needles -up to 25m -slow growing -Habitat for birds and mammals, good noise buffer	 (Galbrand, 2001)

Table B.4. Continued.

SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Picea rubens	Red spruce	-Small to medium-sized coniferous with sharply pointed, stout needles -Up to 28 m -Habitat for birds and mammals, good noise buffer -Porcupines enjoy the bark -Slow growing	 (Galbrand, 2002)
Pinus strobus	White pine	-Medium-sized conifer with long, pointed needs in groups of 5 -Up to 30 m tall -Twigs eaten by deer, squirrels and birds eat seeds -Good noise and wind buffer	 (Galbrand, 2002)
Populus grandidentata	Large toothed aspen	-Medium -sized deciduous tree with deeply toothed leaves -Up to 25m high -Fast growing -Populus trees transpire between 50 and 300 gallons of water per day out of the ground. This rapid extraction can decrease surface contaminant flow towards groundwater, especially organic contaminants	 (Galbrand, 2002)

Table B.4. Continued

SCIENTIFIC NAME	COMMON NAME	DESCRIPTION	PHOTOGRAPH
Populus Tremuloides	Trembling aspen	-Medium sized deciduous tree with finely-toothed leaves -21m tall -UC5=5 feet Habitat for birds and mammals; noise buffer, decrease wind blown dust, decrease vertical migration of contaminants -Fast growing, easily transplanted -Used for remediation of petroleum hydrocarbons, landfill leachate, nutrients (nitrate, ammonium, phosphate), chlorinated solvents, MTBE, municipal wastewater, and pesticides -Populus trees transpire between 50 and 300 gallons of water per day out of the ground. This rapid extraction can decrease surface contaminant flow towards groundwater, especially organic contaminants	 (Sytsma, 2002)
Quercus rubra	Red oak	-hardy, medium sized to large deciduous with large, deeply toothed leaves that turn red in fall. -Up to 20m high -Fast growing	 (Galbrand, 2002)

APPENDIX C. VEGETATION SAMPLING DATA

Table C.1 Treatment Wetland Site Transect 1 Data.

Transect 1 (16m)	Quad 1 (2m)	Quad 2 (3m)	Quad 3 (8m)	Quad 4 (12m)	Quad 5 (14m)
1	*J. brevicaudatus*	*S. pungens*	*J. effusus*	*R. nitida*	*G. palustre*
2	*O. sensibilis*	*J. brevicaudatus*	*E. acicularis*	*B. rapa*	*B. populifolia*
3	rock	*S. rugosa*	rock	*R. acris*	*V. angustifolium*
4	*S. tenuifolia*	*J. brevicaudatus*	*G. striata*	*J. brevicaudatus*	*V. angustifolium*
5	*J. brevicaudatus*	*C. lurida*	*R. acris*	*P. pratense*	*J. effusus*
6	*A. viridis*	*A. viridis*	*S. alba*	*C. leucanthemium*	*S. tenuifolia*
7	*F. virginiana*	*R. nitida*	*J. brevicaudatus*	*J. brevicaudatus*	*S. tenuifolia*
8	*S. alba*	*C. lurida*	*E. acicularis*	*J. effusus*	*P. simplex*
9	*J. effusus*	*T. arvense*	*J. effusus*	*C. bullata*	*P. simplex*

Transect 1 (16m)	Quad 6 (15m)	Quad 7 (16m)	Quad 8 (22m)	Quad 9 (28m)	Quad 10 (29m)
1	*J. brevicaudatus*	*S. rugosa*	*J. effusus*	*J. effusus*	*S. tenuifolia*
2	*J. brevicaudatus*	*A. plantago-aquatica*	*C. rostrata*	*J. effusus*	*S. tenuifolia*
3	*B. populifolia*	*S. alba*	*C. crinita*	*S. tenuifolia*	*J. brevicaudatus*
4	*T. farfara*	*S. alba*	*C. crinita*	*S. tenuifolia*	*E. arvense*
5	*C. brunnescens*	*G. striata*	*C. crinita*	*J. brevicaudatus*	*G. striata*
6	*C. leucanthemium*	*A.plantago-aquatica*	*J. effusus*	*T. latifolia*	*P. cordata*
7	*C. stipata*	*C. leucanthemium*	*J. brevicaudatus*	*J. effusus*	*S. rugosa*
8	*C. bullata*	*V. angustifolium*	*C. stipata*	*A. viridis*	*J. brevicaudatus*
9	*J. brevicaudatus*	*J. effusus*	*C. crinita*	*T. latifolia*	*J. effusus*

Table C.2. Reference Wetland Site Transect 1 Data.

Transect 1 (3m)	**Quad 1** (2m)	**Quad 2** (4m)	**Quad 3** (5m)	**Quad 4** (13m)	**Quad 5** (16m)
1	*J. effusus*	*L. cardinalis*	*C. crinita*	*M. gale*	*S. alba*
2	*S. alba*	*P.pratense*	*C. bullata*	*M. gale*	*P. cordata*
3	*T. arvense*	*S. alba*	*C. bullata*	*J. effusus*	*C. lurida*
4	*J. effusus*	*C. stipata*	*M. gale*	*C. bullata*	*C. lurida*
5	*J. effusus*	*S. tenuifolia*	*J. effusus*	*S. cyperinus*	*C. lurida*
6	*S. canadensis*	*C. stipata*	*M. gale*	*S. cyperinus*	*S. rugosa*
7	*P. arundinacaea*	*M. gale*	*M. gale*	*S. cyperinus*	*P. cordata*
8	*C. leucanthemium*	*S. canadensis*	*G. grandis*	*J. effusus*	*C. stipata*
9	*S. alba*	*S. alba*	*M. gale*	*C. bullata*	*S. alba*

Transect 1 (3m)	**Quad 6** (20m)	**Quad 7** (21m)	**Quad 8** (22m)	**Quad 9** (23m)	**Quad 10** (30m)
1	*P. cordata*	*P. cordata*	*M. gale*	*P. cordata*	*E. arvense*
2	*C. stipata*	*P. cordata*	*C. brunnescens*	*P. cordata*	*S. tenuifolia*
3	*J. effusus*	*P. cordata*	*G. palustre*	*S. cyperinus*	*M. gale*
4	*J. effusus*	*R. acris*	*G. palustre*	*G. palustre*	*S. tenuifolia*
5	*S. cyperinus*	*J. effusus*	*S. cyperinus*	*G. palustre*	*R. acris*
6	Open water	*R. acris*	*G. palustre*	*S. cyperinus*	*P. cordata*
7	*P. cordata*	*J. effusus*	*C. brunnescens*	*P. cordata*	*J. effusus*
8	*P. cordata*	*C. pseudocypernus*	*M. gale*	*C. lurida*	*T. latifolia*
9	*P. cordata*	*C. pseudocypernus*	*M. gale*	*S. cyperinus*	*J. effusus*

Table C.3. Treatment Wetland Site Transect 2 Data.

Transect 2 (22m)	**Quad 1** (1m)	**Quad 2** (3m)	**Quad 3** (4m)	**Quad 4** (6m)	**Quad 5** (7m)
1	*S. canadensis*	*J. brevicaudatus*	*J. canadensis*	*P. arundinacaea*	*J. brevicaudatus*
2	*J. brevicaudatus*	*S. cyperinus*	*P. arundinacaea*	*P. arundinacaea*	*C. stipata*
3	*J. brevicaudatus*	*J. brevicaudatus*	*J. brevicaudatus*	*P. arundinacaea*	*C. crinita*
4	*S. alba*	*S. cyperinus*	*J. effusus*	*J. effusus*	*J. brevicaudatus*
5	*S. cyperinus*	*C. stipata*	*C. pseudocypernus*	*G. striata*	*J. effusus*
6	*S. rugosa*	*G. striata*	*J. brevicaudatus*	*G. palustre*	*G. palustre*
7	*J. brevicaudatus*	*J. brevicaudatus*	*J. brevicaudatus*	Bare ground	*J. brevicaudatus*
8	*G. striata*	*S. cyperinus*	*J. brevicaudatus*	*T. latifolia*	Bare ground
9	*S. cyperinus*	*S. alba*	*J. brevicaudatus*	*J. effusus*	*C. rostrata*

Transect 2 (22m)	**Quad 6** (14m)	**Quad 7** (18m)	**Quad 8** (19m)	**Quad 9** (21m)	**Quad 10** (24m)
1	*C. stipata*	*G. striata*	*J. brevicaudatus*	*G. palustre*	*G. palustre*
2	*J. brevicaudatus*	*G. striata*	*J. brevicaudatus*	*S. cyperinus*	*O. sensibilis*
3	*I. versicolor*	*G. striata*	*C. intumenscens*	*J. brevicaudatus*	*J. brevicaudatus*
4	*J. effusus*	*J. brevicaudatus*	*S. cyperinus*	*R. acris*	*J. brevicaudatus*
5	*T. latifolia*	*J. brevicaudatus*	*J. effusus*	*G. striata*	*R. acris*
6	*J. brevicaudatus*	*G. striata*	*S. cyperinus*	*J. effusus*	*J. effusus*
7	*S. emersum*	*G. striata*	*P. simplex*	*R. acris*	*C. pseudocypernus*
8	*G. striata*	*E. arvense*	*G. striata*	*J. brevicaudatus*	*T. arvense*
9	*J. brevicaudatus*	*E. arvense*	*J. brevicaudatus*	*G. striata*	*E. arvense*

Table C.4. Reference Wetland Site Transect 2 Data.

Transect 2 (6m)	**Quad 1** (6m)	**Quad 2** (9m)	**Quad 3** (11m)	**Quad 4** (13m)	**Quad 5** (16m)
1	*S. alba*	*S. cyperinus*	*P. simplex*	*M. gale*	*P. cordata*
2	*G. grandis*	*S. rugosa*	Bare ground	*M. gale*	*J. effusus*
3	*J. effusus*	*C. lurida*	*C. bullata*	*M. gale*	*J. effusus*
4	*J. effusus*	*S. cyperinus*	*S. americanum*	*S. americanum*	*J. effusus*
5	*G. grandis*	*S. cyperinus*	*S. americanum*	*S. americanum*	*P. cordata*
6	*G. grandis*	*J. canadensis*	*S. americanum*	*C. crinita*	*P. cordata*
7	*G. grandis*	*J. effusus*	*J. effusus*	*J. effusus*	*G. grandis*
8	*G. grandis*	*J. effusus*	*C. lenticularis*	*C. crinita*	*P. cordata*
9	*G. grandis*	*C. lurida*	*C. crinita*	*S. americanum*	*S. alba*

Transect 2 (6m)	**Quad 6** (22m)	**Quad 7** (24m)	**Quad 8** (25m)	**Quad 9** (26m)	**Quad 10** (28m)
1	*P. cordata*	*C. brunnescens*	*S. cyperinus*	*J. effusus*	*C. lurida*
2	*P. cordata*	*G. grandis*	*P. simplex*	*S. alba*	*C. lurida*
3	*C. bullata*	*C. rostrata*	*C. lasiocarpa*	*C. lurida*	*S. cyperinus*
4	*C. lurida*	*C. brunnescens*	*C. lasiocarpa*	*C. intumenscens*	*S. alba*
5	*J. effusus*	*A. viridis*	*S. americanum*	*P. simplex*	*S. cyperinus*
6	*J. effusus*	*A. viridis*	*C. bullata*	*S. americanum*	*G. palustre*
7	*S. tomentosa*	*A. viridis*	*C. bullata*	*S. americanum*	*G. grandis*
8	*M. gale*	*S. tomentosa*	*S. cyperinus*	Bare ground	*J. effusus*
9	*M. gale*	*J. effusus*	*S. tomentosa*	*S. cyperinus*	*J. effusus*

Table C.5. Treatment Wetland Site Transect 3 Data.

Transect 3 (31m)	**Quad 1** (4m)	**Quad 2** (9m)	**Quad 3** (10m)	**Quad 4** (12m)	**Quad 5** (14m)
1	*J. brevicaudatus*	*J. effusus*	*J. effusus*	*J. brevicaudatus*	*J. effusus*
2	*J. brevicaudatus*	*S. cyperinus*	*J. brevicaudatus*	*C. lenticularis*	*J. effusus*
3	*J. brevicaudatus*	*S. cyperinus*	Open water	*J. effusus*	*J. effusus*
4	*J. effusus*	*J. effusus*	*C. leucanthemium*	*C. lurida*	*J. effusus*
5	*G. striata*	*S. cyperinus*	*J. brevicaudatus*	*J. brevicaudatus*	*J. effusus*
6	*J. effusus*	*J. brevicaudatus*	Open water	*J. effusus*	*J. brevicaudatus*
7	*E. arvense*	Open water	*G. striata*	*J. effusus*	*J. brevicaudatus*
8	*S. alba*	Open water	*J. effusus*	*J. effusus*	*J. effusus*
9	*S. alba*	*J. brevicaudatus*	*J. effusus*	*J. brevicaudatus*	*J. brevicaudatus*

Transect 3 (31m)	**Quad 6** (15m)	**Quad 7** (16m)	**Quad 8** (17m)	**Quad 9** (20m)	**Quad 10** (21m)
1	*J. effusus*	*S. emersum*	*G. palustre*	*G. grandis*	*G. grandis*
2	*J. effusus*	*R. palustris*	*J. brevicaudatus*	*G. grandis*	*G. grandis*
3	*J. effusus*	*J. brevicaudatus*	*E. arvense*	*G. palustre*	*C. arvense*
4	*J. brevicaudatus*	*J. brevicaudatus*	*G. grandis*	*R. nitida*	*R. nitida*
5	*S. cyperinus*	*J. brevicaudatus*	*G. grandis*	*T. arvense*	*G. grandis*
6	*J. brevicaudatus*	*T. farfara*	*S. tenuifolia*	*G. grandis*	*G. grandis*
7	*R. palustris*	*J. effusus*	*S. tenuifolia*	*G. grandis*	*S. alba*
8	*J. effusus*	*J. effusus*	*H. matronalis*	*V. cassinoides*	*G. grandis*
9	*J. effusus*	*H. matronalis*	Open water	*J. brevicaudatus*	*G. grandis*

Table C.6. Reference Wetland Site Transect 3 Data.

Transect 3 (19m)	**Quad 1** (1m)	**Quad 2** (3m)	**Quad 3** (5m)	**Quad 4** (6m)	**Quad 5** (12m)
1	*S. americanum*	*J. effusus*	*E. arvense*	*M. gale*	*C. lurida*
2	*S. cyperinus*	*J. effusus*	*S. tomentosa*	*M. gale*	*S. tomentosa*
3	*S. cyperinus*	*S. cyperinus*	*C. lurida*	*S. cyperinus*	*M. gale*
4	*M. gale*	*S. cyperinus*	*H. florentinum*	*A. viridis*	*I. versicolor*
5	*C. stipata*	*C. brunnescens*	*H. florentinum*	*A. viridis*	*C. flava*
6	*M. gale*	*C. brunnescens*	*S. tomentosa*	*C. flava*	*I. versicolor*
7	*M. gale*	*S. tomentosa*	*J. effusus*	*C. flava*	*J. effusus*
8	*S. americanum*	*S. tomentosa*	*S. tomentosa*	*E. acicularis*	*J. effusus*
9	*S. tomentosa*	*C. stipata*	*S. tomentosa*	*E. acicularis*	*J. effusus*

Transect 3 (19m)	**Quad 6** (13m)	**Quad 7** (14m)	**Quad 8** (16m)	**Quad 9** (17m)	**Quad 10** (19m)
1	*M. gale*	*M. gale*	*J. effusus*	*O. sensibilis*	*J. effusus*
2	*G. grandis*	*C. stipata*	*C. lasiocarpa*	*S. americanum*	*C. lurida*
3	*S. cyperinus*	*M. gale*	*S. tomentosa*	*S. americanum*	*C. lurida*
4	*G. grandis*	*C. lurida*	*C. bullata*	*S. tomentosa*	*S. pungens*
5	*C. flava*	*C. lurida*	*O. sensibilis*	*S. tomentosa*	*C. stipata*
6	*C. flava*	*O. sensibilis*	*S. emersum*	*S. tomentosa*	*M. gale*
7	*S. emersum*	*C. stipata*	*C. lasiocarpa*	*M. gale*	*J. effusus*
8	*C. flava*	*O. sensibilis*	*J. effusus*	*S. pungens*	*E. acicularis*
9	*S. emersum*	*S. emersum*	*S. tomentosa*	*S. pungens*	*J. effusus*

APPENDIX D. VEGETATION BIOLOGICAL INTEGRITY ASSESSMENT CALCULATIONS

Table D.1. Number of Plants, Proportional Abundance and Native Status of Plant Species in Reference and Treatment Wetland Sites.

SPECIES	TREATMENT SITE		REFERENCE SITE		NATIVE STATUS
	# plants	Proportional abundance (pi)	# plants	Proportional abundance(pi)	
Juncus brevicaudus	60	0.230	0	0	native
Scirpus cyperinus	12	0.046	22	0.082	native
Juncus effusus	44	0.169	38	0.142	native
Frageria virginiana	1	0.004	0	0	native
Spiraea alba var latifolia	9	0.034	10	0.037	native
Glyceria grandis	12	0.046	6	0.022	native
Solidago rugosa	4	0.015	2	0.007	native
Carex lurida	3	0.011	16	0.060	native
Thlaspi arvense	3	0.011	1	0.0034	exotic
Carex stipata	5	0.019	9	0.034	native
Chrysanthemum leucanthemum	4	0.015	1	0.004	exotic
Carex bullata	2	0.008	9	0.034	native
Galium palustre	7	0.029	6	0.022	native
Equisetum arvense	6	0.023	2	0.007	invasive
Iris versicolor	1	0.004	2	0.007	native
Juncus canadensis	1	0.004	1	0.004	native
Carex pseudocyperinus	2	0.008	2	0.007	native
Solidago canadensis	1	0.004	2	0.007	native
Alisma plantago-aquatica	2	0.008	0	0	native
Glyceria striata	17	0.065	0	0	native
Carex lenticularis	1	0.0048	1	0.004	native
Phalaris arundinacea	4	0.015	1	0.004	exotic
Carex crinita	5	0.019	4	0.015	native
Typha latifolia	4	0.015	1	0.004	invasive
Scirpus pungens	1	0.004	3	0.011	native
Eleocharis acicularis	2	0.008	3	0.011	native
Sparganium emersum	2	0.008	4	0.015	native
Phleum pratense	1	0.004	1	0.004	exotic
Carex rostrata	2	0.008	1	0.004	native
Solidago tenuifolia	9	0.034	3	0.011	native
Hesperis matronalis	2	0.008	1	0.004	exotic
Carex brunnescens	1	0.004	6	0.022	native
Onoclea sensibilis	2	0.008	8	0.030	native
Carex intumenscens	1	0.004	1	0.004	native
Cirsium arvense	1	0.004	0	0	exotic
Tussilago farfara	2	0.008	0	0	exotic and invasive
Potentilla simplex	3	0.011	3	0.011	native

pi=number of individuals per species/total number of individuals

Table D.1. Continued.

SPECIES	TREATMENT SITE		REFERENCE SITE		NATIVE STATUS
	# plants	Proportional abundance (pi)	# plants	Proportional Abundance (pi)	
Vaccinium angustifolium	3	0.011	0	0	native
Betula populifolia	2	0.008	0	0	native
Ranunculus acris	5	0.019	4	0.015	native
Rosa nitida	4	0.015	0	0	native
Rosa palustris	2	0.008	0	0	native
Brassica rapa	1	0.004	0	0	exotic
Viburnum cassinoides	1	0.004	0	0	native
Alnus viridis	3	0.011	5	0.019	native
Myrica gale	0	0	27	0.101	native
Hieracium florentinum	0	0	2	0.007	exotic
Spiraea tomentosa	0	0	16	0.060	native
Sparganium americanum	0	0	13	0.049	native
Carex flava	0	0	6	0.022	native
Pontederia cordata	1	0.004	19	0.071	native
Carex lasiocarpa	0	0	4	0.015	native
Lobelia cardinalis	0	0	1	0.004	native

pi=number of individuals per species/total number of individuals

Table D.2. Results of Species Density, Richness, Diversity and Heterogeneity and Exotic Species Abundance Analyses.

MEASUREMENT	TREATMENT SITE	REFERENCE SITE
Total Density (N)*	261	267
Species Richness (S)	46	41
Species Diversity (H')	2.069	2.053
Variance in diversity (H'_{var})	0.0247	0.025
Heterogeneity	47%	33%
Exotic species abundance	28	10
Exotic species proportional abundance	10.73	3.75

*does not add to 270 as bare ground, open water or rock areas were observed

N = sum of individuals present in site
S = sum of species present in site
$H' = -\Sigma$ (pi ln pi)
$H'_{var} = N^{-1}\{\Sigma p_i(\ln p_i)^2 - [\Sigma p_i(\ln p_i)]^2\} - \{(2N^2)^{-1}\{S-1]\}$
Heterogeneity = (abundance of the top 3 species/total number of species)*100
Exotic species abundance = sum of exotic species present in site
Exotic species proportional abundance = (total number of exotic species/total number of species)*100
$\underline{t} = [H'_{\text{ref site}} - H'_{\text{treatment site}}]/H'_{var}\text{ (ref site)} + H'_{var}\text{ (treatment site)}]^{0.5}$
$df = [H'_{var}\text{(ref site)} + H'_{var}\text{(treatment site)}]/\{[H'_{var}\text{(ref site)}]^2/N_{\text{ref site}} + [H'_{var}\text{(treatment site)}]^2/N_{\text{treatment site}}\}$

t-statistic (t) = 0.071
Degrees of freedom (df) = 530
Critical t values ($\propto$0.05) = 1.960 and 1.980

APPENDIX E. AQUATIC MACROINVERTEBRATE BIOLOGICAL INTEGRITY ASSESSMENT CALCULATIONS

Table E.1. Abundance, Proportional Abundance and Type of Feeding Group Status of Aquatic Macroinvertebrate Taxa in Reference Wetland and Cell 1 and Outlet of the Treatment Wetland.

AQUATIC MACROINVERTEBRATES			REFERENCE SITE		CELL 1		OUTLET		FEEDING GROUP
Class	Order	Family	# individuals	Proportional abundance (pi)	# individuals	Proportional abundance (pi)	# individuals	Proportional abundance (pi)	
Hirudinea			8	0.08			1	0.01	predator
Oligochaeta			33	0.33	24	0.24	15	0.15	collector
Gastropoda	Basommatophora								
		Valvatidae	4	0.04			1	0.01	collector
		Viviparidae	5	0.05			1	0.01	collector
Bivalvia	Veneroida	Sphaeriidae	2	0.02					collector
Crustacea	Amphipoda								
		Hyalellidae			30	0.3	42	0.42	collector
		Gammaridae	7	0.07	2	0.02	9	0.09	collector
	Isopoda								collector
	Decapoda								collector
	Podocopa						2	0.02	collector
Branchipoda	Spinicaudata								collector
	Cladocera								
		Daphnidae			1	0.01			collector
		Macrothricidae			2	0.02	1	0.01	collector
		Chydoridae							collector
		Sididae					1	0.01	collector
		Holopediidae							collector
Copepoda	Calanoida								collector
	Cyclopoida								collector
	Harpacticoida								collector
Insecta	Hemiptera	Notonectidae	3	0.03			2	0.02	predator
		Corixidae	17	0.17	10	0.1	2	0.02	collector
	Coleoptera								
		Scirtidae							scraper
		Elmidae	1	0.01	7	0.07	10	0.1	collector
		Hydrophiloidea			2	0.02			shredder
		Dytiscidae							predator
		Hydroptilidae							predator

pi=number of individuals per taxa/total number of individuals

Table E.1. Continued

AQUATIC MACROINVERTEBRATES			REFERENCE SITE		CELL 1		OUTLET		FEEDING GROUP
Class	**Order**	**Family**	**# individuals**	**Proportional abundance (pi)**	**# individuals**	**Proportional abundance (pi)**	**# individuals**	**Proportional abundance (pi)**	
Insecta cont'd	Odonata								
		Corduliidae	1	0.01					predator
		Aeshnidae	2	0.02					predator
		Gomphidae	1	0.01					predator
		Coenagrionidae	2	0.02					predator
	Plecoptera								
		Leuctridae							shredder
		Perlodidae							predator
		Perlidae							predator
	Ephemeroptera								
		Baetidae	1	0.01					scraper
		Emphemeridae							collector
		Caenidae	3	0.03					collector
		Ephemerellidae	1	0.01					scraper
		Leptophlebiidae							collector
		Heptageniidae							scraper
	Megaloptera								predator
	Trichoptera								
		Polycentropodidae	1	0.01					collector
		Limnephilidae							shredder
		Lepidostomatidae							shredder
		Leptoceridae	1	0.01					predator
		Hydroptilidae							scraper
		Brachycentridae	1	0.01	1	0.01			collector
	Diptera								
		Chironomidae			9	0.09	5	0.05	collector
		Ceratopogonidae					1	0.01	collector
		Empididae							predator
		Tipulidae	2	0.02	5	0.05	2	0.02	shredder
		Culicidae	1	0.01	5	0.05	5	0.05	collector
		Simuliidae			2	0.02			collector
Arach-nida	Acari (subclass)		3	0.03					predator

pi=number of individuals per taxa/total number of individuals

Table E.2. Results of Taxa Density, Richness, Diversity, Heterogeneity and Trophic Structure Analyses.

MEASUREMENT	REFERENCE SITE	CELL 1	OUTLET
Total Density Sampled (N)	100	100	100
Population Richness (S)	22	13	16
Population Diversity (H')	1.037	0.882	0.675
Variance in diversity (H'_{var})	0.029	0.022	0.023
Heterogeneity	58%	64%	67%
Trophic Structure			
Scrapers:	2%	0%	0%
Shredders:	2%	2%	0%
Collectors:	75%	96%	97%
Predators:	21%	2%	3%

N = sum of individuals present in site
S = sum of taxa present in site
$H' = -\Sigma$ (pi ln pi)
$H'_{var} = N^{-1}\{\Sigma p_i(\ln p_i)^2 - [\Sigma p_i(\ln p_i)]^2\} - \{(2N^2)^{-1}\{S-1]\}$
Heterogeneity = (abundance of the top 3 taxa/total number of taxa)*100
Trophic structure = (total number of individuals per feeding group/ total number of individuals) * 100
$t=[H'_{ref\ site}-H'_{cell\ 1}] / H'_{var}(\text{ref site}) + H'_{var}(\text{cell 1})]^{0.5}$
$df=[H'_{var}(\text{ref site})+H'_{var}(\text{cell 1})]/\{[H'_{var}(\text{ref site})]^2/N_{ref\ site}+[H'_{var}(\text{cell 1})]^2/N_{cell\ 1}\}$

t statistic (t)
Ref. to Cell 1 = 0.690
Ref. To Outlet = 1.595
Cell 1 to Outlet = 0.968

Degrees of Freedom (df)
Ref. to Cell 1 = 197
Ref. To Outlet = 198
Cell 1 to Outlet = 200

Critical t values ($\propto$2 0.05) = 1.96 and 1.98

APPENDIX F. AQUATIC MACROINVERTEBRATE WATER QUALITY ASSESSMENT CALCULATIONS

Table F.1. Abundance and BMWP Scores for the Aquatic Macroinvertebrates in the Reference Wetland and Cell 1 and Outlet of the Treatment Wetland.

AQUATIC MACROINVERTEBRATES			REF. SITE	CELL 1	OUTLET	BMWP index*
Class	**Order**	**Family**				
Hirudinea			8		1	
Oligochaeta			33	24	15	1
Gastropoda	Basommatophora					
		Valvatidae	4		1	
		Viviparidae	5		1	6
Bivalvia	Veneroida	Sphaeriidae	2			3
Crustacea	Amphipoda					
		Hyalellidae		30	42	
		Gammaridae	7	2	9	6
	Isopoda					
	Decapoda					
	Podocopa				2	
Branchipoda	Spinicaudata					
		Daphnidae		1		
		Macrothricidae		2	1	
		Chydoridae				
		Sididae			1	
Copepoda	Calanoida					
	Cyclopoida					
	Harpacticoida					
Insecta	Hemiptera	Notonectidae	3		2	5
		Corixidae	17	10	2	5
	Coleoptera					
		Scirtidae				5
		Elmidae	1	7	10	5
		Hydrophiloidea		2		5
		Dytiscidae				5
		Hydroptilidae				6
	Odonata					
		Corduliidae	1			8
		Aeshnidae	2			8
		Gomphidae	1			8
		Coenagrionidae	2			6
	Plecoptera					
		Leuctridae				10
		Perlodidae				10
		Perlidae				10

*BMWP = Biological Monitoring Working Party (Table 4.5)

Table F.1. Continued.

AQUATIC MACROINVERTEBRATES			REF. SITE	CELL 1	OUTLET	BMWP index*
Class	**Order**	**Family**				
Insecta cont'd	Ephemeroptera					
		Baetidae	1			4
		Emphemeridae				10
		Caenidae	3			7
		Ephemerellidae	1			3
		Leptophlebiidae				10
		Heptageniidae				10
	Megaloptera					
	Trichoptera					
		Polycentropodidae	1			7
		Limnephilidae				7
		Lepidostomatidae				10
		Leptoceridae	1			10
		Hydroptilidae				6
		Brachycentridae	1	1		10
	Diptera					
		Chironomidae		9	5	2
		Ceratopogonidae			1	
		Empididae				
		Tipulidae	2			5
		Culicidae	1	5	2	
		Simuliidae		5	5	
Arachnida	Acari (subclass)		3	2		

*BMWP = Biological Monitoring Working Party (Table 4.5)

Table F.2. Results of BMWP and, ASPT Scoring, ETSD Biotic Index, Mayfly Abundance and Trophic Structure.

CALCULATION	REFERENCE SITE	CELL 1	OUTLET
BMWP score	107	34	30
ASPT score	5.94	4.86	4.29
ETSD biotic index	45.45%	7.69%	0%
Mayfly abundance	13.64%	0%	0%
Trophic Structure			
Scrapers:	2%	0%	0%
Shredders:	2%	2%	0%
Collectors:	75%	96%	97%
Predators:	21%	2%	3%

BMWP = Biological Monitoring Working Party
ASPT = Average Score per Taxon
ETSD = Ephemeroptera (E) (mayflies), Trichoptera (T) (caddisflies); Sphaeriidae (S) (fingernail clams), and dragonflies (D).

BMWP score = sum of BMWP scores
ASPT score = sum of BMWP scores / number of BMWP indicator families present
ETSD biotic index = (total number of ETSD families / total number of families) * 100
Mayfly abundance = (total number of mayfly families / total number of families) * 100
Trophic structure = (total number of individuals per feeding group/ total number of individuals) * 100