Treatment Performance Assessment and Modeling of a Natural Tundra Wetland Receiving Municipal Wastewater

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ABSTRACT

The application of natural tundra wetlands for municipal wastewater treatment is an option to meet upcoming federal wastewater systems effluent regulations for Canada's Far North. A treatment performance assessment with physical, hydraulic and biogeochemical contextual data was conducted on the wetland treatment area of Coral Harbour, Nunavut. A modified Tanks-In-Series model was used to model treatment kinetics. The study showed seasonal variability in treatment performance and hydraulic characteristics. A decrease in vegetation diversity and deposition of organic detritus was observed in high effluent loading areas. Effective reduction in effluent concentrations was observed. Dilution from watershed contributions accounted for much of the reductions observed. The importance of the determination of the hydraulic residence time, field delineation of the wetted area, and delineation of the watershed was demonstrated. First order rate coefficients determined suggested that the selection of the lowest percentiles from literature of southern treatment wetlands is conservative in this case.

LIST OF ABBREVIATIONS AND SYMBOLS USED

%	Percent
PO_4^{3-}	Phosphate ion
Q_{WS}	Non-Effluent Watershed Discharge
Q_{in}	Wetland Segment Inflow
Q_{out}	Wetland Segment Outflow
e_v	Volumetric Efficiency
τ_{an}	Adjusted Nominal Residence Time
τ_n	Nominal Residence Time
°	Degree
°C	Degrees Celsius
µg/L	Micrograms per Liter
µS/cm	Micro Siemens per Centimeter
AANDC	Aboriginal Affairs & Northern Development Canada
ASTM	American Society for Testing and Materials
BLR	Areal Biochemical Oxygen Demand Loading Rate
BMP	Best Management Practice
BOD	Biochemical Oxygen Demand
BOD_5	Five-day Biochemical Oxygen Demand
BOD_7	Seven-Day Biochemical Oxygen Demand
Br^-	Bromide ion
$C*$	Background Concentration
$CBOD_5$	Five-Day Carbonaceous Biochemical Oxygen Demand
CCME	Canadian Council of Ministers of the Environment
CFU/100 mL	Colony Forming Units per 100 mL
chla	Chlorophyll a

cm	Centimeter
COD	Chemical Oxygen Demand
CSRS	Canadian Spatial Reference System
CWRS	Center for Water Resources Studies
d	Day
DEM	Digital Elevation Model
DO	Dissolved Oxygen
E	Eastern Hemisphere
E. coli	*Escherichia coli*
e.g.	*Exempli gratia*
EC	Environment Canada
et al.	*Et alii*
etc.	*Et cetera*
FAC	Facultative Species
FACU	Facultative Upland Species
FACW	Facultative Wetland Species
FWS	Free Water Surface
g	Gram
GN	Government of Nunavut
GPS	Global Positioning System
h	Head
ha	Hectare
HLR	Hydraulic Loading Rate
hr	Hour
HRT	Hydraulic Residence Time
HSSF	Horizontal Subsurface Flow
i.e.	*Id est*

IC	Ion Chromatography
ICP-ms	Inductively Coupled Mass Spectrometry
ISE	Ion-Selective Electrode
k	First-Order Rate Coefficient
k_{20}	First-Order Rate Coefficient Corrected to 20°C
K_{fs}	Field Saturated Hydraulic Conductivity
kg	Kilogram
km	Kilometer
k_T	First Order Rate Coefficient at Field Temperature
L	Liter
LO	Lower Output
m	Meter
m^2	Meters Squared
m^3	Meters Cubed
MDL	Method Detection Limit
mg/L	Milligrams per Liter
M_{in}	Mass of Injected Tracer
mL	Milliliter
mm	Millimeter
MPN/100 mL	Most Probable Number of Colony Forming Units per 100 mL
N	Northern Hemisphere
n	Number of Samples
N	Number of Tanks-In-Series
n.d.	Non-Detect
NaBr	Sodium Bromide
NAD83	North American Datum 1983
NH_3	Un-ionized Ammonia

NH_3-N	Un-ionized Ammonia Nitrogen
NH_4	Ammonium
NH_4-N	Ammonium Nitrogen
No.	Number
NO_3-N	Nitrate Nitrogen
NPS	National Performance Standards
NRI	Nunavut Research Institute
NU	Nunavut
NWB	Nunavut Water Board
NWT	Northwest Territories
Θ	Temperature Correction Factor
OBL	Obligate Wetland Species
PET	Potential Evapotranspiration
pH	$-log_{10}[H^+]$
ppm	Parts per Million
PPP	Precise Point Positioning
PVC	Polyvinyl Chloride
Q	Discharge
R^2	Coefficient of Determination
RBG	Red Blue Green
RTD	Residence Time Distribution
RTK	Real-Time Kinematic
RWT	Rhodamine WT
Spp.	Species
SSF	Subsurface Flow
t	Advective Time of Travel
TAN	Total Ammonia Nitrogen

TC	Total Coliforms
TIS	Tanks-In-Series
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
U.S. EPA	United States Environmental Protection Agency
UPL	Obligate Upland Species
USA	United States of America
USCS	Unified Soil Classification System
UTM	Universal Transverse Mercator
V	Volt
VF	Vertical Flow
VSS	Volatile Suspended Solids
W	Western Hemisphere
WS	Wetland Segment
WSER	Wastewater Systems Effluent Regulations
WSP	Wastewater Stabilization Pond
WWTP	Wastewater Treatment Plant
y	Year
ν	Average Linear Velocity
σ	Standard Deviation
τ	Mean Residence Time

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Chapter 1: Introduction

1.1. Project Context

In 2009, the Canadian Council of Ministers of the Environment (CCME) released the *Canada-Wide Strategy for the Management of Municipal Wastewater Effluent*; the strategy aims to harmonize wastewater treatment regulations across the provinces and territories by applying National Performance Standards (NPS) to all facilities. Within the strategy, the necessity for careful consideration of Canada's northern treatment systems was identified (CCME, 2009). The strategy has since been adopted by Environment Canada (EC) with the development of the *Wastewater Systems Effluent Regulations* (WSER). As a result, wastewater treatment facilities across Canada are required to comply with a set of federal standards (Government of Canada, 2010). To accomplish this, the Northern provinces and territories were granted a five-year grace period to conduct research on existing systems in order to develop appropriate standards within a northern context. This grace period expires in 2014.

In recognition of the need for a better understanding of the northern operating environment, research has commenced on selected wastewater treatment systems located in the territory of Nunavut (NU), Canada. The research is contracted by the Community and Government Services (CGS) Department of the Government of Nunavut (GN) and conducted by the Center for Water Resources Studies (CWRS) at Dalhousie University. Overall, the objectives of the site-specific wastewater treatment studies are to conduct comprehensive treatment performance and risk assessments on the systems to understand how they are functioning, and to characterize associated human health and environmental risks. Eventually, the data collected from the broader northern wastewater treatment research program will be used to inform policy makers identifying best management practices (BMPs) for northern wastewater treatment.

1.2. Wastewater Treatment in Nunavut

The territory of Nunavut has 25 communities that range in population from approximately 148 to 6 813 (Government of Nunavut, 2012). The relatively small and remote communities are distributed sparsely across the 1 877 788 km^2 territory (Statistics Canada, 2012). Those 25 communities are only accessible by aircraft and ships.

The municipal wastewater in Nunavut communities is primarily generated from domestic and light commercial activities. There are exceptions, such as in Nunavut's capital city of Iqaluit, where wastewater is generated from larger commercial establishments, backwash water from a drinking water treatment plant, and a hospital, and in Pangnirtung, where a fish processing plant contributes to the generated wastewater. Typically, wastewater is collected by pump trucks from individual households and commercial establishments, and transported to a common wastewater disposal area in each respective community. Less commonly, utilidor heated piping infrastructure is used to collect and convey the wastewater to pumping stations or disposal areas.

Wastewater in Nunavut is typically managed with passive treatment methods which include: 16 systems that use a combined configuration of a Wastewater Stabilization Pond (WSP) and wetland; two systems that have either a WSP or a wetland; three systems that discharge raw wastewater to a natural pond or ponds; and one system that discharges directly to a marine outfall. The use of mechanical treatment plants is less common, with only three systems in Nunavut employing this approach. Reasons for the lack of mechanical treatment facilities in Nunavut include being expensive due to high capital and maintenance costs, the optimization of performance is challenging because of increasing demands from rapidly growing populations, and the requirement for extensive technical supervision. The most common wastewater treatment system configuration in Nunavut is the coupled use of single or multiple cell WSPs which discharge into natural tundra wetland areas. Some of the areas in the tundra where wastewater is discharged may not have been wetlands prior to the commencement of wastewater application; however, discharging wastewater to the tundra created wetland areas (Brix, 1994). It has been suggested by Doku and Heinke (1995) that the deliberate attainment of better than secondary wastewater treatment with natural wetlands be termed "engineered natural wetlands".

The WSPs typically store frozen wastewater as it accumulates throughout the winter. The ice and snow break-up occurs normally in June for most of the systems in Nunavut. Treatment of the wastewater in the WSPs and wetlands occurs in the ice-free season spanning approximately three months, from the June break-up to mid-September, when the ice formation returns with colder temperatures. Many of the systems such as in Grise Fiord, Kugluktuk, and Pond Inlet, are manually decanted with a pump and generator at scheduled times over the treatment season. Some of the treatment systems do not have a controlled annual decant, such as in Coral Harbour, where wastewater exfiltrates out of a WSP berm into a wetland area. Other systems may have a semi-controlled decant, such as in Kugaaruk, where a decant cell is used to passively distribute effluent from the WSP onto a wetland area.

Currently, the regulatory requirements of wastewater treatment in Nunavut communities are prescribed by the Nunavut Water Board (NWB). The NWB issues water licenses that establish the minimum treatment requirements for each system on a site-specific basis. The federal government agency Aboriginal Affairs and Northern Development Canada (AANDC) is responsible for inspection of the systems and compliance monitoring. The natural tundra wetland treatment areas are commonly not recognized as part of the wastewater treatment system in Nunavut by the regulatory authorities; a primary concern is uncertainty in their physical extent and treatment potential. As a result, the treatment performance of the systems is not well documented, and the natural wetland systems treat wastewater in an "un-engineered" manner.

1.3. Objectives

Natural treatment wetlands could be a viable passive treatment technology in remote northern areas because they can be engineered to operate in a low-maintenance and controlled manner; however, cold-climate conditions in Canada's Far North present unique operating challenges. In particular, the complex roles that the hydrological setting and biogeochemical interactions play in the overall treatment performance of tundra wetlands will be essential to understand prior to responsible application of this ecological engineering technology.

The site that is the topic of this thesis is located in Coral Harbour, NU. The Coral Harbour municipal wastewater system is of particular interest as it is comprised of a natural tundra wetland that receives effluent from a single-cell WSP. A solid technical understanding of this natural tundra wetland treatment system will help in the development of effluent discharge quality objectives appropriate for northern sites, which incorporate similar wetland treatment systems.

The treatment wetland study in Coral Harbour, NU was undertaken in recognition of the research gaps identified in the following literature review with regards to the treatment performance of natural tundra wetlands receiving municipal wastewater. The specific research objectives of this study are to:

1. identify the main hydraulic, physical, and biogeochemical factors that influence the treatment performance of the wetland;
2. evaluate the temporal variability in treatment performance over the course of the treatment season;
3. compare the overall treatment performance of the wetland to literature findings from other wetlands operating in cold climate regions; and
4. develop a modeling approach to characterize the treatment kinetics in the wetland.

CHAPTER 2: LITERATURE REVIEW

2.1. Natural Wetlands

Wetlands are features that occur naturally in the environment and are characterized by water occurring at or near the surface which support unique ecological communities. They are defined by the U.S. Army Corps of Engineers and U.S. EPA as (U.S. Army Corps of Engineers, 1987; Clean Water Act, 1972):

"*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*."

Conversely, an upland area tends to be characterized by higher ground, greater relief, and with vegetation adapted for drier soil conditions. Wetlands are characterized by hydric soils, which are high in organics and are anaerobic in the upper layer, formed from prolonged saturation (U.S. Army Corps of Engineers, 1987). Characteristically, wetlands have hydrophytic vegetation species adapted to hydric soil conditions which are indicative of wetland habitats. The hydrophytic vegetation can be used to help discern wetland from upland areas.

There are a variety of different types of natural wetlands, which are characterized by diverse geology, water quality, hydrology, vegetation, and oxygen availability. The National Wetlands Working Group (1988) identified five different classes of wetlands that occur naturally in Canada which consist of: bogs, fens, swamps, marshes, and shallow open water (Table 2.1).

Table 2.1 Natural wetland classifications and descriptions.

Wetland class[a]	**Description**
Bog	• Peat-covered wetlands in which the vegetation shows effects of a high water table and is generally poor in nutrients. • Common vegetation types include mosses, especially *Sphagnum* spp. and ericaceous shrubs.
Fen	• Peatlands characterized by a high water table, with a very slow drainage by seepage down low gradient slopes. Relatively low oxygen saturation. • Nutrient content typically moderate. • Vegetation consists mainly of sedges, especially *Carex*, grasses, reeds, brown mosses, shrubs, and sometimes a sparse tree layer.
Swamp	• Standing or gently moving waters occurring seasonally or persisting for long periods leaving the subsurface continuously saturated. • Typically abundant in nutrients and oxygen. • Vegetation is generally a dense cover of deciduous or coniferous trees or shrubs, herbs and some mosses.
Marsh	• Periodically inundated by standing or slowly moving water and are therefore rich in nutrients. • Mainly wet, mineral-soil areas, but shallow, well-decomposed peat may be present. • Gravitational water table, but water remains within the rooting zone of plants for most of the growing season. • Relatively abundant in oxygen saturation. • Dominant vegetation of sedges, grasses, rushes, bulrushes, reeds and white water lily.
Shallow open water	• Relatively small, non-fluvial bodies of standing water representing a transitional stage between lakes and marshes. • Water depth typically less than 2 m at mid-summer levels. • Surface waters impart an open aspect, free of emergent vegetation, but floating, rooted, aquatic macrophytes may be present.

[a]National Wetlands Working Group, 1988; Doku and Heinke, 1993.

2.2. Wastewater Treatment Wetlands

Inherently, wetlands have the unique tendency to convert waste streams into benign natural by-products through ecologically complex processes. Wetlands are frequently termed "nature's kidneys" for their unique ability to improve water quality (Kennedy and Mayer, 2002; Brix, 1994). The natural processes acting to degrade or transform contaminants consist generally of sedimentation of solids, microbial metabolism, ultraviolet irradiation, adsorption onto substrate, chemical precipitation, volatilization, plant assimilation, and soil filtration (Crites and Tchobanoglous, 1998). The harnessing of the natural treatment tendencies of wetlands for use in wastewater applications is not a new practice, and has been used for at least the past 45 years (U.S. EPA, 1999). Initial work on wastewater treatment using wetlands was mostly conducted on natural wetlands receiving effluent (Nichols, 1983). The use of natural wetlands for wastewater treatment subsided in 1987 when the U.S. EPA limited the use of natural wetlands for solely tertiary treatment of wastewater. (U.S. EPA, 1987). Some of the reasons for this limitation may include risk of contaminant exposure to humans, wildlife and/or waterfowl, treatment performance reliability, loss of engineered control of the effluent, and an overall lack of knowledge about the treatment mechanisms and effectiveness of treatment (Breaux *et al.*, 1995).

The treatment abilities observed in natural wetlands have been more recently emulated and applied with the use of artificial engineered constructed wetlands. The preference for the use of constructed wetlands instead of natural wetlands for wastewater treatment resulted from the uncertainty in regards to the long-term consequences of using natural wetlands for disposal (Kadlec and Wallace, 2009). There are three main types of constructed wetlands which are: Free Water Surface (FWS); Horizontal Subsurface Flow (HSSF); and Vertical Flow (VF). The latter two are different types of subsurface flow (SSF) wetlands. Natural wetlands can be characterized with similar terminology as the effluent flow in natural wetlands can vary from surface flow (FWS) to SSF, or a combination of both. One of the key differences between natural and constructed wetlands is hydrology. Natural wetlands tend to experience large fluctuations in water level either diurnally and/or seasonally, whereas, constructed wetlands usually maintain a relatively constant water level throughout the year except when wastewater additions vary (Gopal, 1999). Vymazal (1998) suggested that constructed wetlands are favorable due to the control of hydraulic pathways and Hydraulic Residence Time (*HRT*). Another main difference is the amount of biodiversity between natural and constructed wetlands, with natural systems being much more biodiverse (Gopal, 1999). The use of both natural and constructed wetlands for municipal wastewater treatment can provide both a low-cost and effective treatment technology when they are well understood and designed (Nichols, 1983; Breaux *et al.*, 1995).

2.2.1. Natural Treatment Wetlands

This section of the literature review includes a synthesis of the studies that have been conducted on the treatment performance, and potential impacts, of natural wetlands used for municipal wastewater treatment. Natural wetlands operating specifically in cold climates are discussed later in Section 2.4. These studies are important for the quantitative demonstration of the treatment performance of natural wetlands, to garner an understanding of all the repercussions of using this method of wastewater disposal, and to identify knowledge gaps that need to be addressed to improve the application.

The main application of natural wetlands for municipal wastewater treatment has been for tertiary municipal wastewater treatment (Cooke, 1994; Kadlec and Tilton, 1979; Nichols, 1983; Breaux *et al.*, 1995). It was estimated that in the 1990s approximately half of the 200 FWS wetlands used for wastewater treatment in North America were natural wetland systems (Brix, 1994). In recent years, the move to use constructed wetlands instead of natural systems for wastewater treatment has been encouraged due to uncertainties in the long-term effects on natural wetland ecosystems and the implementation of regulations (Kadlec and Wallace, 2009).

Despite the discouragement of the use of natural wetlands, it has been demonstrated on numerous occasions that they are effective in the treatment of municipal wastewater. For example, Knight *et al.* (1987) conducted a treatment performance assessment on a FWS natural wetland (35 ha) receiving tertiary treated and disinfected wastewater from a resort complex in Reddy Creek, Florida. The system yielded average annual effluent concentrations of 2.0 mg/L for Five-Day Biochemical Oxygen Demand

(BOD_5), 2.1 mg/L for Total Suspended Solids (TSS), 1.6 mg/L for Total Nitrogen (TN), and 3.5 mg/L for Total Phosphorus (TP). Interestingly in that study, it was stated that too much pre-treatment prior to discharge of influent into the wetland can result in higher effluent concentrations of BOD_5 and TSS. This was attributed to microbial populations which account for the removals of organic matter, becoming starved at low loadings. Others, such as Nichols (1983), Tilton and Kadlec (1979), Dubuc *et al.* (1986), Doku and Heinke (1995), Wright (1974), Kadlec (1987), and Nõges and Järvet (2002), have all noted favorable treatment capabilities of natural wetlands for wastewater treatment.

A major advantage of using natural wetlands for treatment of secondary or tertiary municipal wastewaters is the economic benefits (Knight *et al.,* 1987; Breaux *et al.,* 1995; Doku and Heinke, 1995; Nichols, 1983; Kadlec, 1987). The economic benefit of using natural wetlands for wastewater treatment has been highlighted by Breaux *et al.* (1995) in a review paper focused on the economic aspects of the technique. There are potential costs to be considered when discharging wastewater into a natural wetland which include: the cost of piping and distribution structures, engineered improvements and restricted uses from previous un-impacted site conditions. The paper presents a case study of a Louisiana wetland of 231 ha and states capital cost savings ranging from $73 500 to $140 200 USD in 1995, in comparison to constructing a traditional tertiary treatment plant. Monetary values could not be placed on the added benefit of potential water quality improvements in receiving water bodies that received effluent prior to discharge into the natural wetland. The authors describe the use of natural wetlands for wastewater treatment as a low-cost alternative to traditional treatment practices. They demonstrated that in many cases natural wetlands are capable of treating to a higher level than feasible with traditional methods. Likewise, a cost savings estimate by Kadlec and Tilton (1979) showed that the use of a natural wetland for tertiary treatment of municipal wastewater would cost $395 000 USD for construction compared to $1.1 million USD for spray irrigation and $1.7 million USD for chemical treatment without inflation adjustment. The authors also note that costs savings can be substantial for small communities that are located in close proximity to available wetlands.

The ecological enhancement of natural wetlands as a result of nutrient introduction from wastewater is another potential benefit highlighted by Doku and Heinke (1995) and Breaux *et al.* (1995). Potential benefits of nutrient addition to an arctic environment are different from southern environments; typically tundra regions are nutrient poor compared to other less cold and temperate regions (Doku and Heinke, 1995). Another favorable factor is that natural treatment wetlands require minimal maintenance and do not require skilled operators (Nõges and Järvet, 2002).

The use of natural wetlands for wastewater treatment is not without controversial disadvantages (Cooke, 1994; Jenssen *et al.,* 1993). Their use has been limited for several years due to U.S. EPA regulations limiting their use (Breaux *et al.*, 1995). Concerns with their use for wastewater treatment include: the potential accumulation of toxic substances in benthic invertebrates, fish and waterfowl; wetland degradation; eutrophication; long-term performance reliability; and shifts in the biotic communities (Gopal and Ghosh, 2008; Gopal, 1999; Knox *et al.,* 2008). Aesthetic concerns with the use of natural wetlands for wastewater treatment include odour and undesirable visual changes; however, those concerns are not well documented and is likely due to the lack of significant observable negative impacts (Kadlec, 1987). Jenssen *et al.* (1993) suggest that northern environments are especially pristine and un-impacted. That, combined with their low ecological diversity, renders them particularly vulnerable to anthropogenic impacts. Brix (1994) implied that natural wetlands should not be purposefully used for wastewater treatment due to ecosystem changes and the inability to predict the consequences of long-term use. Concerns with the extreme variability and unpredictability of natural wetlands for wastewater treatment were noted by Vymazal (1998). It was noted that this inherent variability would render it challenging to translate performance expectations across geographic regions.

There are advantages and disadvantages associated with the use of natural wetlands for wastewater treatment. The major benefits are the cost-saving advantages, and low maintenance, and technical skill requirements for operation. The assessment of the impact of wastewater discharge into natural wetland receiving environments on local aquatic, terrestrial fauna and vegetation requires further research (Kennedy and Mayer, 2002). Further research is required to understand the potential ecological and human health implications associated with this practice in Canada's Far North.

2.3. Canadian Arctic Wetlands

The National Wetlands Working Group (1988) estimates that wetlands cover approximately 27 800 000 ha of the Northwest Territories (NWT) and Nunavut. That accounts for ≥ 20% of the total wetland area in Canada (Heinke and Doku, 1995). The spatial distribution of the regions of wetlands in the Canadian arctic is illustrated in Figure 2.1. The wetlands found in Northern Canada are geographically defined by three main regions consisting of the low, mid- and high arctic. A description of the most common wetland forms, and predominant vegetation types, for each of the arctic wetland regions is summarized in Table 2.2.

The National Wetlands Working Group (1988) states that the extreme climate of the Canadian arctic influences the development of wetlands in Northern Canada. The arid conditions restrict the spatial extent of wetlands to poorly drained depressions or to areas where a surplus of water is available. The Canadian arctic is in a zone of continuous permafrost, and as a result, the free internal drainage of wetlands is inhibited. Therefore, most of the moisture is either concentrated near or at the surface or stored frozen in the permafrost. Arctic wetlands are also subject to frost churning (cryoturbation) meaning that frost action mixes underlying mineral-rich soil with peat at the surface whereby boulders may be heaved to the surface through the peat. Arctic wetlands are also subject to frost cracking, whereby the development of ice-wedges effectively cracks the peat layer and forms ice-wedge polygons (National Wetlands Working Group, 1988).

Table 2.2 Summary of arctic wetland classes and regions.

Wetland region[a]	**Common wetland forms**	**Predominant wetland vegetation**
High arctic	Low-center lowland polygon fens; peat mound bogs.	Sedges (*Carex* spp.) mainly on fens; lichen and moss are associated with bogs.
Mid-arctic	Low-center lowland polygon fens; salt marshes along low-lying coastal lowlands.	Sedges and cotton grasses (*Eriophorum* spp.) on fens.

Wetland region[a]	Common wetland forms	Predominant wetland vegetation
Low arctic	Low-center lowland polygon fens and bogs; marshes along coasts and in deltas; shallow-water ponds.	Sedges and cotton grasses with mosses associated with low-center polygons; low willows, sedges, and grass vegetation are associated with delta marshes.
Sub-arctic	Peat plateau bogs separated by fens in high sub-arctic; fens and peat plateau-plasa bog complexes in low sub-arctic.	Sparse spruce-lichen woodland associated with bogs; sedges and cotton grasses with mosses found on fens.
High boreal	Fens and bogs; floodplain swamps; delta marshes.	Sedges and mosses on fens; spruce, birch, cotton grasses on bogs, sedges and common reed grass on delta marshes; spruce, birch and ferns on floodplain swamps.

[a] Table sourced from Heinke and Doku, 1995; National Wetlands Working Group, 1988.

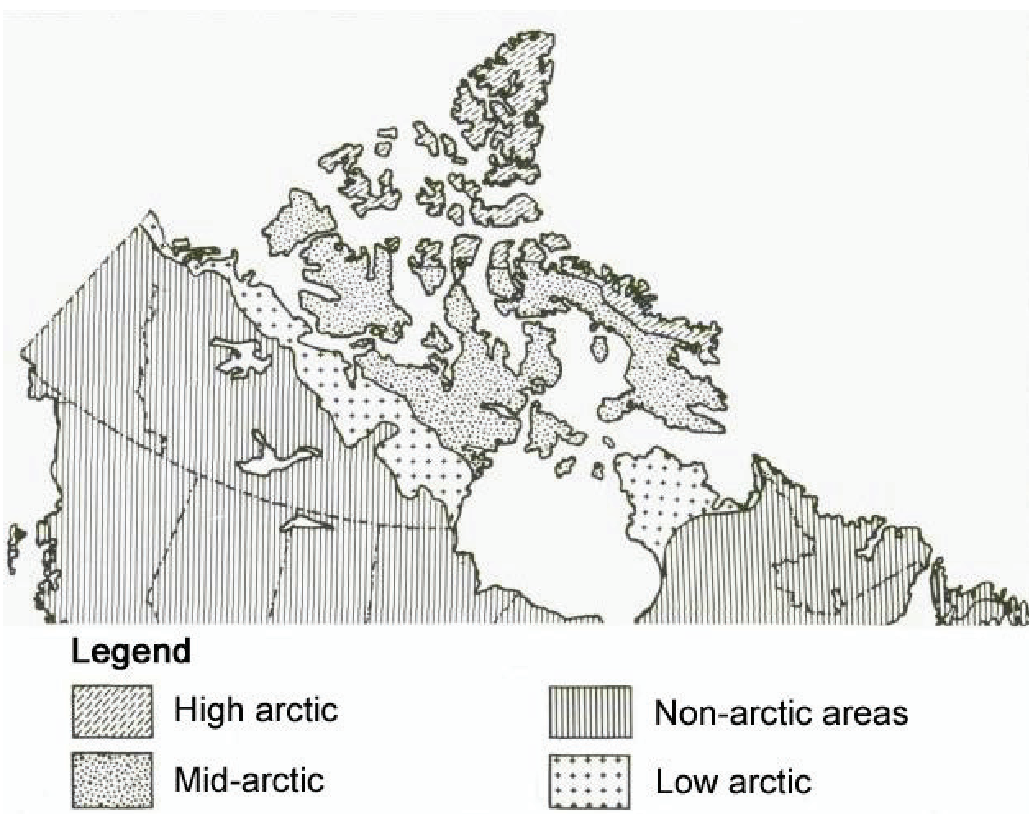

Figure 2.1 Arctic natural wetland regions of Canada (Heinke and Doku, 1993; National Wetlands Working Group, 1988).

2.4. Cold Climate Treatment Wetland Studies

A review of the pertinent literature on the use of both engineered natural wetlands and constructed wetlands operating in cold climate regions is summarized in the following sections. It should be noted that "cold climate" encompasses a subjective broad range of annual air temperature ranges. Wittgren and Mæhlum (1997) define "cold climate" being representative of systems that have a coldest monthly mean temperature below -3°C and a warmest monthly average temperature above 10°C. This definition is mostly analogous to systems that experience snow cover for at least one month of the year. According to their definition, most wetlands in Canada, Alaska, Northern USA, Eastern Europe, and Scandinavia, are classified as cold climate. That said, most of the cold climate wetlands reviewed are operating in much milder climatic conditions than the Canadian arctic.

Published studies on wastewater treatment in arctic environments are limited. The literature on cold climate treatment wetlands in non-arctic regions is the closest comparison, which is reviewed here. It is important to note that many of these studies have much longer treatment seasons with higher average temperatures. For example, some reviewed wetlands operate the entire year, which is much longer than the approximate three month snow- and ice-free treatment season in the arctic (*i.e.*, mid-June to mid-September). Additionally, the use of treatment wetlands in Canada's Far North includes many more challenges than other cold climate wetlands due to permafrost, low precipitation, extreme temperatures, and a 24 hour photoperiod (Kennedy and Mayer, 2002; Pries, 1994).

Constructed wetlands were included in this review because the treatment performance of constructed systems operating in cold climates has been studied more prevalently than in natural wetlands. The reasons for the lack of cold climate natural treatment wetland studies is likely due regulatory bodies, which steer municipalities towards the use of alternative municipal wastewater treatment methods. Previous studies and reviews of numerous treatment wetlands have demonstrated that wastewater treatment is effective in cold regions (Jenssen *et al.,* 1993; Wittgren and Mæhlum, 1997; Kennedy and Mayer, 2002; Mander and Jenssen, 2002). The following cold climate treatment wetland review summarizes research conducted to date on cold climate treatment wetlands in Canada, particularly the arctic and sub-arctic, Northern United States, and Northern Europe.

2.4.1. Northern Canada

One of the first engineered natural wetland treatment studies in northern Canada was conducted by Wright (1974) who studied a natural swampland in Hay River, NWT. That 32 ha swamp received primary treated municipal wastewater from three WSPs. The effluent discharge area was estimated at 110 m^2 per capita (Wright, 1974). Hydrological context for the swampland area was not quantified. Therefore, dilutive effects could not be estimated. Wright (1974) sampled the swampland monthly from August 1972 to September 1973. In Wright's study, favorable percent reductions in concentrations of treatment performance parameters were observed including: 97.7% for Biochemical Oxygen Demand (BOD); 96.8% for TSS; 98.0% for Volatile Suspended Solids (VSS); 96.2% for Un-ionized Ammonia Nitrogen (NH_3-N); 97.6% for TP; and 98.7% for Total Coliform (TC). Wright (1974) did observe an ecological alteration to the swamp in terms of decreased zooplankton, nektonic, and benthic organism diversity, and different community composition. Flows in the swamp were observed to be transient over the treatment season with a high spring freshet flow. During the spring freshet, increased bacterial concentrations were observed at the swamp outlet. Therefore, Wright recommended guidelines for the use of similar natural swamplands as part of the wastewater treatment process. Those guidelines stipulated that the: i) wetland treatment areas be sited away from groundwater recharge zones; ii) alteration of the natural wetlands should be deemed acceptable; and iii) wetland morphology should be such that the *HRT* is relatively long. The Hay River, NWT swampland was studied for over twenty years and during that time it consistently met the regulatory requirements for the system (Doku and Heinke, 1993).

Doku and Heinke (1993) discussed the potential of using natural wetlands for wastewater treatment in the NWT and Yukon prior to the establishment of Nunavut. They noted that Canada's northern communities are small and remote, experience extreme climates, have limited infrastructure funds for design, construction and maintenance of facilities, and have a chronic shortage of skilled labour. As a result, they suggested that the treatment technology implemented must be practical and straightforward to operate, whilst maintaining protection of human health and the environment.

Although widespread, the natural tundra wetland areas in northern territories are generally nutrient deficient. Doku and Heinke (1993) suggested that the addition of municipal wastewater to tundra wetlands could improve the ecological functioning of the area through the addition of nutrients while, minimizing deleterious water quality impacts to other receiving environments. According to the authors, the relatively small size of northern communities, and the primarily domestic origin of the wastewater generated, renders the use of natural tundra treatment wetlands an appropriate option for use in the North. This is particularly true for communities that have sufficient land available for the application. Doku and Heinke (1995) also recommended increased long-term monitoring data collection and interpretation as a precursor to the identification of the most appropriate design criteria for use in the arctic and sub-arctic regions (Kennedy and Mayer, 2002).

Doku and Heinke (1995) also discussed the use of natural wetlands in the Yukon. At that time, the Yukon Water Board had issued five licenses to communities in the territory to permit the use of natural wetlands for secondary municipal wastewater treatment. The Yukon Water Board required that site-specific hydraulic and treatment performance assessments be conducted prior to use of the systems as part of the wastewater treatment train. The demonstration of long-term treatment of effluent discharge in the absence of ecological impacts was a stipulation of acceptance by the regulatory body. Primary treatment of wastewater, at a minimum, prior to discharge into the wetlands was also a regulatory requirement.

As part of their study, Heinke and Doku (1993) suggested recommendations to guide the use of natural treatment wetlands as a viable and effective secondary or tertiary treatment technique. Their recommendations included: 1) requiring that all natural wetland treatment systems be capable of meeting treatment guidelines established by the NWT government during the time of their study; 2) that site-specific ecological studies be conducted to determine local vegetation pollutant removal capabilities; 3) that a minimum of primary treatment occur in advance of wastewater discharge to the a natural wetland;

and 4) that an areal BOD loading rate (*BLR*) not exceed 8 kg BOD_5/ha·d, and that a Hydraulic Loading Rate (*HLR*) of between 100 and 200 m^3/ha·d be maintained. In a subsequent publication (Doku and Heinke, 1995), they provided several additional recommendations: 5) conduct further research on the use of natural treatment wetlands to assist in the establishment of design criteria; 6) that the NWT government foster the informed and responsible use of natural wetlands for wastewater treatment; and 7) that communities be informed and become engaged on the responsible use of natural wetlands for treatment.

The most recent arctic natural wetland treatment study by Yates *et al.* (2012) assessed the treatment performance of six natural tundra wetlands that received municipal wastewater from six hamlet communities in the Nunavut region of Kivalliq. Specifically, the study sites were located in the hamlet communities of Arviat, Baker Lake, Chesterfield Inlet, Coral Harbour, Repulse Bay, and Whale Cove. Three of the six treatment wetlands were combined with pre-treatment from WSPs or lakes. Those systems were in the communities of Arviat, Coral Harbour, and Whale Cove. The other wetlands in Baker Lake, Chesterfield Inlet, and Repulse Bay received either raw wastewater from trucks directly, or wastewater that had received minimal pre-treatment. All of the wetlands varied in terms of size, geographic local, geological composition, flow area dimensions, and vegetation communities. Each wastewater treatment system received from 28 to 163 m^3/d of wastewater over the course of the year; however, the wastewater was stored frozen during the winter months.

Yates *et al.* (2012) collected influent and effluent treatment performance water quality samples weekly from the wetlands for the duration of the treatment season, which was from June 21, 2008 to September 24, 2008. They observed a range of reductions in treatment performance parameters across the wetlands ranging from 47 – 94% for Five-Day Carbonaceous Biochemical Oxygen Demand ($CBOD_5$), 57 – 96% for Chemical Oxygen Demand (COD), 39 – 98% for TSS, >99% for TC, >99% for *Escherichia coli* (*E. coli)*, 84 – 99% for NH_3-N, and an 80 – 99% reduction in TP. The treatment performance assessments they conducted on the natural tundra wetlands demonstrated that the passive wastewater treatment technology has promising potential to be an effective technology for use in communities in the Canadian Arctic.

Dubuc *et al.* (1986) demonstrated that a natural peat wetland in Northern Quebec was highly effective at treating domestic wastewater. The study site was a mid- to high boreal wetland area near the 55^{th} parallel located at the James Bay Energy Company's Fontanges construction camp for the James Bay Energy Company in Northern Québec. The peatland receiving the camp wastewater was approximately 1.5 km long. Average percent reductions of over 90% were reported for most treatment performance parameters, which suggested that the natural peatland system was highly effective at treating the wastewater. The authors noted that the dilution of the effluent from the wetland affected the water quality results. However, they did not quantify the baseline hydrologic system. The treatment performance assessment did not include a bacteriological component, due to the lack of hydrological context and significant background levels, which would have rendered a data interpretation challenge. They noted that including of the quantification of the hydrological context would have significantly strengthened their interpretation of the water quality results.

In terms of winter treatment abilities of arctic wetlands, Edwards and Jeffries (2010) recently debunked the common assumption that arctic wetland treatment mechanisms cease in the winter. Their study focused on nitrogen uptake by *Carex aquatilis* in a low arctic meadow in Churchill, Manitoba. They determined that plant uptake still occurs at temperatures below 0°C and they suggested that decomposition processes occur year-round in tundra ecosystems. During their study, vegetation nitrogen uptake was elevated during spring melt.

These studies that were conducted on Northern Canadian sites demonstrate the effective use of natural arctic and sub-arctic wetlands for use in municipal wastewater treatment. The consideration of the use of natural wetland areas as part of the overall treatment process in Canada's North was recommended by Doku and Heinke (1993 and 1995), Yates *et al.* (2012), Wright (1974), and Dubuc *et al.,* (1986). All those authors described the need for additional research that includes the characterization of the hydrological and hydraulic context of the wetlands coupled with treatment performance assessments. Additional research is needed on the long-term treatment performance and ecological consequences prior to their formal recognition for wastewater treatment.

2.4.2. Southern Canada

A review of the primary historic and current studies that have been conducted on FWS wetlands, in Southern Canada, is included as part of the review as insight for the performance and challenges associated with treatment wetlands in cold regions. The case studies reviewed include the natural wetland in Cootes Paradise, Ontario; the artificial marsh systems in Listowel, Ontario; and the FWS constructed wetland in Albert, Ontario.

An early study on the effects of municipal wastewater effluent discharge on a natural marsh in Southern Canada was conducted by Mudroch and Capobianco (1979). Their study site was the 170 ha Cootes Paradise natural marsh wetland. Cootes Paradise is located to the west of Hamilton, Ontario in the Dundas Valley. The wetland had been receiving municipal WasteWater Treatment Plant (WWTP) primary effluent from 1919 to 1971 and secondary treatment from 1971 onwards (Semkin *et al.*, 1977). The authors were investigating nutrients and heavy metals uptake by vegetation and sediment profiles in the marsh. They noted a long-term decrease in the diversity of vegetation in the marsh in response to the effluent discharge. The sediment profiling showed higher than background concentrations in the marsh substrate of lead, chromium, nickel, copper, and zinc. Increases above background concentrations of nitrogen, phosphorus, and carbon were observed in top 5 cm of the substrate. Large numbers of migratory waterfowl were attracted to the wetland in the early spring to nest and breed.

Semkin *et al.* (1977) observed that Cootes Paradise demonstrated dilutive capabilities of up to 70% for TP, TN, and BOD near the discharge point of the WWTP and inlet of the wetland. They attributed this to a non-effluent stream convergence. The authors also noted that the high production of algae in the marsh during the daytime led to high dissolved oxygen (DO) concentrations. That resulted in favorable nitrifying conditions to remove ammonia and organic nitrogen. Chow-Fraser *et al.* (1998) showed a decrease in phytoplankton diversity and insect diversity in Cootes Paradise. Those authors also demonstrated a 75% loss in the spatial coverage of vegetation and the conversion of benthic species to those that thrive in anaerobic conditions. The Cootes Paradise marsh system is no longer in use for wastewater disposal because years of use resulted in over-sedimentation and subsequent unfavorable BOD concentrations (Kennedy and Mayer, 2002). This site could be considered an example of the long-term negative consequences of the un-engineered use of a natural wetland system for wastewater treatment.

A major study on the cold climate treatment performance of constructed wetlands receiving secondary municipal wastewater was conducted for the Ontario Ministry of the Environment by Herkowitz (1986) in Listowel, Ontario. The Listowel artificial marsh consisted of five constructed wetland cells with a total surface area of 8 670 m^2, which were operated continuously throughout the winter. Average temperatures of the marsh effluents ranged from 0°C to 19.6°C. Between December and March, the average effluent temperature was < 2°C. The marsh systems demonstrated that they were effective at reducing BOD, suspended solids, and bacteria over the entire year. Despite the cold operating temperatures, the systems were capable of producing secondary to tertiary quality effluent. Herskowitz *et al.* (1987) noted that the systems were optimized when the DO was sufficient to facilitate aerobic metabolism. The BOD and TSS removal performance was reduced in the summer due to algal growth (Jenssen *et al.,* 1993) and a reduction in Ammonium Nitrogen (NH_4-N) removal was attributed to low water temperatures when the DO was above 2 mg/L. The nitrogen removal rates were up to twice as high in the summer (Jenssen *et al.,* 1993).

The Alfred, Ontario Municipal FWS constructed wetlands study was reported on by Cameron *et al.* (2003). The 0.47 ha wetlands receive decanted effluent from a two-cell WSP. The wetland subsequently discharges annually during the spring to a brook receiving environment. The authors state that the increased stream flow in the spring freshet from snow melt runoff acts to dilute the pond effluents such that they comply with the discharge criteria. Reductions of 34% for BOD_5, 93% for TSS, 58% for *E. coli*, 52% for Ammonium (NH_4), and 90% for TP were observed (Cameron *et al.*, 2003).

The Southern Canadian treatment wetlands demonstrated effective treatment performance; however, their operating climate is far less harsh than Canada's Far North. Unfortunately, the Cootes Paradise wetland was not sustainable and ecological degradation resulted. The need for research on the optimal contaminant loading rates is required to avoid adverse ecological consequences associated with the use of natural wetlands. Detailed biogeochemical data was useful in the Listowel studies to further understand the treatment mechanisms. Further research on the biogeochemical processes of natural wetlands will allow optimization of favorable treatment mechanisms in design procedures.

2.4.3. Northern United States

There are many examples of wetlands used for municipal wastewater treatment in the United States. For this review, only the natural and/or cold regions FWS wetlands are relevant. The pertinent case studies that can provide insight on cold regions and natural wetland performance are located in Michigan and North Dakota.

The tertiary wastewater treatment performance of a natural wetland was conducted in Houghton Lake, Michigan (Kadlec, 1987; Tilton and Kadlec, 1979; Kadlec *et al.,* 1979). A 700 ha peatland receives secondary treated wastewater through distribution pipes which encourage the dispersed application of effluent rather than a single-point discharge configuration. The authors noted that engineered inlet structures work more effectively to reduce contaminants than single point discharge sites. Vegetation in the peatland was dominated by sedges and willow. Overland flow was the mechanism by which water flowed across the wetland. The Houghton Lake system yielded a 99% and 63% reduction in Nitrate Nitrogen (NO_3-N) and Ammonium Nitrogen (NH_4-N) respectively, within 1 ha of wetland area (Tilton and Kadlec, 1979).

The use of northern natural wetlands for wastewater treatment is not without caution and limitations. For instance, Kadlec (1987) reported that the fall freeze-up usually slows or stops many wetland treatment processes. Regardless, wetlands with winter discharges have still shown water quality improvements. Kadlec (1987) stated that flow distribution can be problematic with northern natural treatment wetlands in addition to, short *HRT*s. Kadlec and Tilton (1979) explain that a hydrologically overloaded wetland would yield a decrease in treatment potential. The authors also suggested that background coliform levels and suspended solids can be present at sufficiently high levels in a natural wetland, even without wastewater addition; thus, rendering it a difficult parameter to regulate. The authors cautioned that the environmental impact of using a natural wetland for wastewater treatment should be considered, because the potential long-term changes to vegetation and fauna communities and structures, are difficult to predict.

The intra- and inter-system variability of northern natural wetlands render highly site-specific treatment performance expectations. For instance Kadlec (1987) reported high intrasite variability in treatment performance for 10 natural wetland wastewater treatment sites in northern environments. At the Houghton Lake wetland, replicate samples showed standard deviations one-third to one-half of the mean treatment performance parameter concentrations. Tilton and Kadlec (1979) cautioned that an extrapolation between the Houghton Lake system and other treatment wetlands can be erroneous due to inherent intersystem differences in effluent quality, climate, hydraulic and hydrologic characteristics. They identified that the perimeter flows into and out of a natural wetland are extremely site-specific and are governed by site topography, soil permeability, watercourses, and anthropogenic flow diversion structures. The authors emphasized that it is extremely important to consider wastewater additions to a natural wetland as it relates to overall wetland water budget. Kadlec (1987) describes natural wetlands as being capable of providing advanced wastewater treatment in northern environments; however, constraints of their use have not been determined and are likely to vary on a site-specific basis (Kadlec and Tilton, 1979).

Hammer and Burckhard (2002) observed the effect of low temperatures on the pollutant removal from secondary treated municipal wastewater in Minot's wetland located at a latitude of 48°15' N in North Dakota. Average monthly air temperature ranged from a low of -19°C in January to a high of 14°C in July. The wetland configuration consisted of a 51 ha, four cell marsh and pond wetland. The limiting effluent discharge quality criteria was 1 mg/L NH_3 and the wetland did not operate annually between January to March. For water temperatures above 10°C, the removals were 42.7% for BOD_5, 52.2% for TSS, and 69.4% for NH_3. For water temperatures between 5° and 10°C, removals were 33.7% for BOD_5, 32.8% for TSS, and 46.8% for Un-ionized Ammonia (NH_3). Removals of 27.2% for BOD_5, 40.6% for TSS, and 46.8% for NH_3 were reported for water temperatures below 5°C. For this wetland system, treatment performance was less effective at lower temperatures. Although apparent TSS removal was higher at lower temperatures, the authors attributed this to the absence of algae at lower temperatures.

The northern United States studies suggest that natural and cold region wetlands do treat wastewater; however, treatment may be limited in colder temperatures. Research is needed to define the intra- and inter-system differences in treatment performance.

2.4.4. Northern Europe

Natural, semi-natural, and constructed wetlands have been used in cold climate wastewater treatment in many northern European countries. The prevalence of FWS and SSF wetlands tends to favour one treatment technology or the other for most European countries. This may be due to regulatory bodies recommending, or providing design criteria, for one configuration over another. For this review, the focus is directed on cold regions European countries where literature is available on FWS constructed wetlands, including Estonia, Norway, and Sweden.

The use of constructed wetlands has become more common in Estonia because many of the small traditional mechanical treatment plants that have been performing poorly and have been closed (Mander and Mauring, 1997; Mander *et al.*, 2000). Average monthly air temperatures in Estonia range from -5°C in January to 17°C in June. Mean annual precipitation is 700 mm.

In a study by Mander and Mauring (1997), the effect of cold temperatures on treatment performance within constructed and semi-natural treatment wetlands was assessed. At the time of the study, adequate design criteria for constructed and semi-natural wetlands for wastewater treatment in Estonia were not established. One of the constructed wetlands studied consisted of a 140 m^2 vegetated drainage channel (a bioditch) that received agricultural wastewater from ponds with a high HLR of 90 cm/d. Another constructed wetland consisted of a semi-natural 0.24 ha wet meadow covered with *Phalaris arundinacea,* which received agricultural effluent from primary treated ponds with a lower HLR of 5 cm/d. The successful treatment of effluent was demonstrated for the *Phalaris* slope. The bioditch showed treatment potential; however, it was hydraulically overloaded. As a result, cold temperatures were demonstrated to have negligible effects on treatment performance. More recently, Mander *et al.* (2000) studied a 1.2 ha FWS constructed wetland treating secondary municipal wastewater in Põltsamaa, Estonia. That system showed a 68% reduction in Seven-Day Biochemical Oxygen Demand (BOD_7), a 73% reduction in nitrogen, and a 24% reduction in phosphorus. Again, the system did not exhibit a decrease in treatment efficiency during the cold season. Nõges and Järvet (2002) studied the largest natural wetland (3 km^2) receiving untreated municipal wastewater in a wetland in the headwaters of Tänassilma River, Estonia. Average mass reductions showed 99% TC, 96% TSS, 94% COD, 87% BOD_5, 65% TN, and only 17% TP from 1985 to 1991. The minimal phosphorus reduction was likely due to saturation from prolonged application of wastewater in the wetland since 1948. The authors noted long-term changes in upstream vegetation from diverse grassland species to cattail, and downstream changes from swamp vegetation to common reed, birches and willows (Nõges and Järvet, 2002).

Jenssen *et al.* (1993) studied the use of constructed wetlands for wastewater treatment in Norway where SSF wetlands are primarily used (Mæhlum and Jenssen, 1998). The authors demonstrated that constructed wetlands are an appropriate technology for cold conditions characteristic of the sub-arctic. They indicated that biological degradation can occur at temperatures of $\leq$ 5°C. Jenssen *et al.* (1993) identified the potential necessity for aerobic pre-treatment prior to discharging to a treatment wetland in order to obtain a high removal of organic matter and nitrogen in cold climates. The possible requirement for larger sizing and longer *HRT*s was also suggested by the authors in order to achieve desirable treatment performance.

In a review paper on treatment wetlands in cold climates, Wittgren and Mæhlum (1997) stated the importance of the wetland-to-catchment ratio on the *HRT* of the wetland, which is particularly important during melt spring in cold climate wetlands. For wetlands with small wetland-to-catchment ratios, they recommend short-term storage of runoff to prevent dramatic *HRT* decreases. For extremely cold regions, such as the arctic, they suggest providing requisite for seasonal water storage.

FWS constructed wetlands are used in Sweden primarily for nitrogen removal in municipal wastewater treatment (Andersson *et al.*, 2005). Typically, FWS wetlands are used to treat secondary or tertiary pre-treated wastewater (Sundblad, 1998). For an 18 ha FWS wetland in Oxelösund, Sweden, Wittgren and Mæhlum (1997) reported areal first order rate coefficients normalized to 20°C (k_{20}) of 12 m/y BOD, 6 m/y TN, and 17 m/y TP. Temperature dependency on reaction rate kinetics was observed and temperature coefficients of 1.01 for BOD, 1.09 for TN, and 1.02 for TP were reported. Nitrogen removal showed significant temperature dependence; however, BOD and TP removal were not as dependent on temperature.

Sundblad (1998) summarized the treatment performance of five Swedish constructed FWS wetlands. Two of the wetlands reviewed were very small in area and received wastewater with no pre-treatment. The 1.5 ha Emmaljunga FWS wetland is a plant-soil system with an overland flow section. Treatment performance results showed over a 75% reduction in BOD, a 60% decrease in TP, and 50% reduction in TN. The author noted that the wetland's overland flow section and stone-lined canal were important for phosphorus removal. Similar to the Emmaljunga wetland, the 0.23 ha Järna wetland is a FWS wetland receives effluent with no pre-treatment. That wetland has an *HRT* of over 5 months and was constructed with passive flow form aeration features. The Järna wetland showed a high nitrogen reduction but was less effective at treating phosphorus and BOD_7. Sundbald (1998) stated that the use of FWS wetlands will continue to expand in Sweden due to their good performance record; however, there remains a need for a database of systems monitoring to refine their design and operation criteria. The author identified a lack of knowledge on the systems and therefore a loss of opportunity for ecosystem enhancement.

In summary, northern Europe treatment wetlands have proven effective in their treatment of municipal wastewater. A need for long-term monitoring on multiple systems has been identified to develop and refine design criteria in the northern and arctic climate regions.

2.5. Treatment Wetland Design Models

The complex decay, transformation, and removal processes that occur in treatment wetlands are typically quantified using the reaction kinetics for contaminant of interest. Wetland designers commonly use chemical reactor theory to represent the wetland treatment kinetics and hydraulic behavior.

Particular contaminants are removed or transformed from the waste stream in a reactor or series of reactors according to the magnitude of the rate coefficient (*k*) or *k*-value. For wastewater treatment, reaction kinetics is typically best characterized by first order reactions. This is because most removal reactions yield an exponential relationship between contaminant removal rates and contaminant concentration (*i.e.,* removal rates decrease exponentially as contaminant concentrations decrease and vice versa). A volumetric based first order rate coefficient has units of inverse time (*e.g.,* d^{-1}, y^{-1}, *etc.*), and an areal based first-order rate coefficient has units of length per unit time (*e.g.,* cm/d, m/y, *etc.*). Determining the *k*-values for the treatment performance parameters is important in order to quantify the rate at which each parameter is reduced in the wetland. Generally, a higher *k*-value is indicative of faster contaminant removal. Wetland design sizing, and predictions for contaminants removal, is completed using rate coefficients, which vary for each treatment performance parameter. Rates vary in part due to the complex removal mechanisms which may be unique for each parameter (*e.g.,* BOD removal from settling, phosphorus assimilation by algae, *etc.*).

The temporal variability in *k*-values for a specific treatment parameter renders it a challenging parameter to fully characterize, especially in the absence of comprehensive long-term hydrological, hydraulic, and water quality datasets (*e.g., k*-values are dependent on variable temperature, *HLR*, *etc.*). Furthermore, *k*-values may vary spatially, especially in natural systems that are often physically heterogeneous. To further complicate the selection of representative *k*-values for design applications, there is high inter-system variability, especially for natural wetlands, such as those prevalent in the arctic.

Many models have been used to quantify reaction kinetics in treatment wetlands. The modeling techniques used for treatment wetland design can be generalized into four main categories consisting of: 1) empirical methods; 2) rational methods based on ideal chemical reactor design; 3) rational methods based on non-ideal chemical reactor design; and 4) process-based computer simulation methods.

The simplest method for designing and sizing wetlands is using a rule of thumb approach with literature values for *HLR*s, areal BOD loading rates, *HRT*s, and per-capita areal wetland requirements. It is common for designers to adopt these ranges from literature. The drawback is that the suggested ranges are arbitrary, and may or may not be appropriate or conservative depending on the wetland. Ideally, the recommended rule of thumb method should only be used as a check on calculations founded on site-specific data; however, realistically, many designers blindly follow rule of thumb procedures due to lack of data and standardized design criteria. Rousseau *et al.* (2004) suggest that using rule of thumb for design is often the most conservative approach; however, the wetland may be over designed and costly.

Commonly, empirical equations, due to their relative simplicity, have been used to predict wetland treatment performance. However, empirical models have been described by Kumar and Zhao (2011) and Rousseau *et al.* (2004) as black box models because they are based on input and output data and do not represent the internal wetland hydraulics. Therefore, regression equations have been used to relate various parameters such as *HLR* and inlet concentration to expected effluent concentrations (Kadlec and Wallace, 2009). Typically, empirically derived regression equations are only applicable for the range of data used to develop the relationship (Kumar and Zhao, 2011). The empirically based models are not the best suited for design applications due to intersystem variability.

Until recently, the most common modeling technique used to represent treatment performance was the ideal chemical reactor model proposed by Kadlec and Knight (1996). They introduced the first order k-C^* model, which only requires two primary variables of k and background concentration (C^*) (Kumar and Zhao, 2011). The ideal chemical reactor model is favored as a design model, because it requires minimal parameterization, and hence is easy to use. The k-C^* model assumes plug flow hydraulics, which was known to be non-representative of true wetland hydraulics. The assumption of plug flow hydraulics in a wetland has been shown to be flawed because short-circuiting and attenuation lead to deviations in the residence times of the wetland. Use of the model revealed that in cases, the *k-C** produced unacceptable non-conservative design effluent concentrations (Kadlec and Wallace, 2009).

Numerous studies have demonstrated that first order *k*-values are dependent on influent concentrations, *HLR*, and *HRT* (Carleton and Montas, 2010; Kadlec, 2000; Jamieson *et al.*, 2007). The selection of constant values for *HLR*s, inlet concentrations, and *k*-values is not a realistic approach for a wetland system that has tremendous inherent variability in a number of aspects. The assumption of static hydraulic and kinetic conditions for a natural wetland is especially inaccurate. Andersson *et al.* (2005) noted that the performance of wetlands receiving effluent at a relatively constant *HLR* are more predictable than wetlands receiving water from non-point sources with variable flows and water quality. As a result of the inadequacies discussed, Kadlec (2000) recommended that the use of plug flow first order models be replaced with models more representative of the internal hydraulics of wetlands. However, the *k-C** model is still widely used and recommended. For instance, Rousseau *et al.* (2004) describes it as the best available design tool. Much of the acceptance of its use is due to the breadth of literature available for first-order *k*-values in treatment wetlands.

The use of non-ideal chemical reactor models has been more recently recommended for use in treatment wetland design (Kadlec and Wallace, 2009; Carleton and Montas, 2010). Among the non-ideal models, the Tanks-In-Series (TIS) model is most frequently selected for contaminant decay modeling in treatment wetlands. Kadlec and Wallace (2009) represent the TIS mass balance equation over an entire sequence of tanks as:

$$\frac{(C - C^*)}{C_i - C^*} = \left(1 + \frac{k\tau}{Nh}\right)^{-N} \qquad [2.1]$$

where:

C = final effluent concentration,

C^* = background concentration,

C_i = initial influent concentration,

k = first order rate coefficient,

τ = hydraulic residence time, and

N = number of tanks-in-series.

The TIS model contains two parameters consisting of a first-order rate constant and α, which represents the number of tanks in the system. Parameterization of the model can involve the determination of α directly in the field with hydrodynamic tracer studies; in this case the model is termed a TIS wetland performance model. Alternatively, α is considered a fitting parameter that is calibrated in conjunction with k using treatment performance data; hence it is termed a relaxed TIS model (Carleton and Montas, 2010). The new recommendation is the use of a relaxed TIS model for representing contaminant decay in FWS wetlands instead of ideal chemical reactor models.

Process based computer simulation models are another type of wetland treatment model that can be developed for steady and unsteady state conditions. They can also be considered as a mechanistic compartmentalization of the wetland and individual treatment processes. However, these types of process based models often require comprehensive parameterization, that render simpler (*i.e.,* k-C^* or TIS) approaches that are more favorable for practical design applications (Rousseau *et al.*, 2004).

After a review of the available treatment wetland models, the most favorable model for a northern context is the TIS non-ideal chemical reactor model. This type of model may best represent the hydraulics and treatment kinetics of natural treatment wetlands characteristic of the Canadian arctic. Reasons for the suitability of the TIS model for northern wetlands include: i) the particularly non-ideal hydraulics in terms of the non-uniform mixing zones that characterize natural wetlands; ii) the parameterization and calibration data is limited yet not overly simplified, straightforward, and feasible to collect; and iii) the model concept is not overly complicated to encourage eventual widespread use in design.

The use of a non-ideal chemical reactor model, such as the TIS model, to represent treatment kinetics in a natural arctic wetland is not without limitations. Arctic wetlands are natural or semi-engineered systems, and as such they generally operate in unsteady hydraulic conditions (*e.g.,* flows and *HLR* may be more variable than a constructed wetland, *etc.*). The unsteady nature of arctic wetlands requires special consideration when modeling for worst-case treatment scenarios. Research is needed to develop, parameterize, calibrate, and verify wetland performance models in an arctic environment. The range of first-order rate coefficients characteristic of arctic wetlands is currently unknown and requires further research to ensure accurate design procedures.

2.5.1. Current Northern Canadian Approach

Treatment wetland design sizing in northern Canada is not formalized and it is mostly left to wetland designers to select the methodology. Alberta offers guidance for wetland design within the province. Other provinces and territories rely on wetland designers to adopt design criteria that achieve treatment objectives on a site-specific basis.

Alberta has a guidance document that provides direction to wetland designers, as well as, regulators (Alberta Environment, 2000). That document recommends *HLR*s ranging from 0.2 cm/d for secondary treated effluent and 0.5 cm/d for nitrified secondary effluent. In cases where BOD_5, TSS, and TP are reduced in the pretreatment, a conservative *HLR* of 2.5 cm/d is recommended. An *HRT* of 14 to 20 days is recommended for a natural treatment wetland (Alberta Environment, 2000; Kadlec and Knight, 1996). The empirical *k*-*C** model ideal chemical reactor is recommended for sizing wetlands and predicting effluent concentrations with the guidance document. The recommended *k*-values and background concentrations are founded on data from hundreds of treatment wetlands across North America. The design criteria are summarized in Table 2.3. The first-order rate coefficients for the proposed model are constant and specified.

Table 2.3 Recommended k-C^* model parameters in Alberta Environment's Guidelines for treatment wetland design.

Performance parameter[a]	***k* (m/y)**	***C** (mg/L)**
BOD_5	34	$3.5 + 0.053C_i$[b]
TSS	1000	$7.8 + 0.063C_i$
TP	12	0.05
TN	22	2
NH_4-N	18	0
Organic nitrogen	17	1.5
Fecal coliform	77	100

[a] Alberta Environment (2000).
[b] C_i = influent concentration.

Empirical design criteria for wetlands in northern Canada were proposed by Doku and Heinke (1993). They recommended an *HLR* between 100 – 200m^3/d and that areal BOD loading rates (*BLR*) be ≤ 8 kg BOD_5/ha·d. In their approach, areal *BLR* is only used as verification that DO will be sufficient to maintain aerobic conditions.

A case study of treatment wetland design in Cambridge Bay, Nunavut was provided by Kadlec (2008). That study describes the model approach taken to predict treatment performance expectations from a 2.9 ha semi-natural tundra wetland. The wastewater system in Cambridge Bay was undergoing renovation. Therefore, the natural wetland was to receive wastewater from primary and secondary WSPs. The first order rate coefficients for a northern context were not developed so the author adopted coefficients from wetlands in southern climates. A conservative approach was taken to estimate rates of removal by applying temperature coefficients, and selecting *k*-values in lower percentiles than most southern systems. Assumptions were made in regards to the expected *HRT* and water depth. Kadlec (2008) carried out the wetland design with a procedure that consisted of three main steps. The first step involved the calculation of a detailed seasonal water budget for the site. The water budget estimation involved determination of inflows and outflows to the wetland, and an estimation of the evapotranspiration and precipitation from the wetland. The second consisted of a review of the treatment performance results from sampling in previous years to use in model parameterization. The third step used the parameterization data to calculate pollutant mass balances, which required the selection of first order rate coefficients to describe the removal kinetics of each contaminant. The only parameter that was assumed to be temperature dependent was nitrogen. The author's selection of *k*-values included 50 m/y TSS and 30 m/y $CBOD_5$ (Kadlec, 2008).

The rate coefficients used in the design process should be determined in an arctic setting to strengthen the use of treatment wetlands in Canada's Far North. Currently, wetland designers must conservatively select arbitrary first order rate coefficients that have been extrapolated from southern and more temperate climates. It is unknown whether the first order rate kinetics in the arctic environment lie in the lower end of the reported literature values, as is assumed by designers. Research is needed to assess the first order rate coefficients characteristic of arctic wetlands. Specific hydraulic, hydrological, and water quality data will be required to parameterize the models. The intrasystem and intersystem variability in rate coefficients requires research. The determination of appropriate rate coefficients would be enabled by conducting long-term monitoring programs at multiple wetlands sites.

2.6. Research Gaps

The literature review suggests that there is a considerable lack of knowledge regarding the use of natural wetlands for wastewater treatment in northern climates. In order to appropriately assess the feasibility of using natural wetlands in the Canadian arctic, additional research is required to ensure their appropriate use and to optimize design criteria and operation procedures. There are specific research needs that need to be fulfilled prior to the realization of the use of natural tundra wetlands as an integral part of an appropriate and effective municipal wastewater strategy for Canada's Far North. In summary, the research needs identified within this literature review include the requirement for:

i) Long-term monitoring of hydrological, hydraulic, and biogeochemical parameters coupled with treatment performance assessments;

ii) Quantification of intra- and inter-system variability of treatment performance parameter first-order rate coefficients; and

iii) Risk assessment of the practice of wastewater addition to natural tundra wetlands in relation to aquatic and terrestrial flora and fauna and human health.

The fulfillment of these research needs will facilitate informed, ecologically responsible and safe incorporation of natural tundra wetlands into the overall northern wastewater BMPs. To meet the research needs outlined above, there is a requirement for hydraulic, treatment performance, and modeling studies on multiple northern wetlands, which entails comprehensive monitoring. As an initial step in fulfillment of the grander research requirements, the Coral Harbour, NU treatment performance assessment and modeling of a natural tundra wetland receiving municipal wastewater study was initiated.

Specifically in reference to the Coral Harbour wetland treatment area, Purdy and Lawson (2006) recommended further monitoring of the wetland contaminant concentrations and flows during the spring freshet. They suggested that the monitoring is a perquisite to garnering acceptance of the wetland in Coral Harbour as part of an approved water license by NWB. They stated that a detailed topographic survey was required in the wetland to determine the flow directions, and identify the need and preferred location of diversion berms, to prevent flow towards the community.

CHAPTER 3: METHODOLOGY

3.1. Site Description

The hamlet of Coral Harbour, NU is the sole community located on Southampton Island; which is situated in the northern Hudson Bay at a latitude of 64° 08’ 13” N, and a longitude of 083° 09’ 51” W (Figure 3.1). The natural tundra receiving wetland study was conducted at the wastewater treatment facility located there; and hereafter termed as “the site.” The hamlet population is estimated at 893 (Government of Nunavut, 2012). Average monthly air temperatures range from a maximum and minimum, respectively of: –26 ºC and –34 ºC in January to 14 ºC and 5 ºC in July. The site is characterized by a dry climate with average annual precipitation comprising of 155 mm as rainfall and 1 335 mm as snowfall, which is equivalent to 286 mm of total precipitation (Environment Canada, 2011).

The wastewater treatment facility is shown in relation to the hamlet and other municipal infrastructure in Figure 3.2. Construction of municipal infrastructure in Coral Harbour is commissioned by the CGS Department of the GN and the hamlet is responsible for the maintenance of the infrastructure. The solid waste facility is located just south of the wastewater treatment facility. The hamlet’s drinking water is sourced from the Post River, which is located to the west. The water is pumped to and stored in a blasted rock reservoir immediately north of the hamlet. Wastewater generated in Coral Harbour is predominately domestic in nature, with no major commercial or industrial inputs. The wastewater is collected from individual dwellings and establishments using pump trucks and is transported to the disposal site. An unlined single-cell WSP and natural tundra receiving area, located approximately 3 km north of the hamlet, is where the wastewater is disposed of.

Historically, the wastewater effluent was discharged straight onto the tundra. To improve treatment, the WSP was constructed in 2003 (Purdy and Lawson, 2006). The berm material of the WSP was designed to be impermeable as it was assumed that the berm would remain frozen over the entire year. However, due to coarse soil properties and seasonal thaw, wastewater is exfiltrated along the south-eastern section of the berm at a variable and uncontrolled rate throughout the treatment season (*i.e.,* June to September). The Coral Harbour water license stipulates that the water quality criteria be met by the "last point of control." Because of the permeable berms, that point is where the truck discharges the waste (Purdy and Lawson, 2006). The natural tundra receiving area is comprised of a mixture of wetland and upland areas and from this point onwards is referred to as the "wetland treatment area". The trucks transport approximately 95 m^3/d (34 779 m^3/year) of wastewater from hamlet residences and establishments to the wastewater treatment facility (Government of Nunavut, 2013)

3.1.1. Wetland Treatment Area

Approximately 14 ha of natural tundra wetland treatment receives the wastewater from the hamlet. This wetland falls within lower end of the 10 – 100 ha range that most natural wastewater treatment wetlands fall within (Brix, 1994). The wetland treatment area is characterized by gently sloping downward west to east topography with elevated regions of north-south striking exposed bedrock. The elevation at the wetland inlet denoted by the circle (A) icon in Figure 3.3 is 29.349 m. The elevation of the wetland outlet denoted by the circle (C) icon in Figure 3.3 is 17.556 m. The straight-line distance between (A) and (C) is 427 m, which yields a site slope of approximately 3%. The receiving environment of the wetland treatment area is a 7 ha shallow freshwater lake that eventually discharges to a marine coastal estuary.

Figure 3.3 indicates the direction of effluent flow within the wetland treatment area, which was variable over the course of the treatment season because of the uncontrolled nature of the wastewater introduction to the wetland treatment area. During the spring freshet, the effluent discharged out of the south-eastern toe of the berm, and flowed approximately 720 m in a south-east to north-east direction, towards the wetland treatment area outlet. During the post-spring freshet period, the effluent discharged from the eastern side of the WSP and flowed approximately 575 m in a west to east direction, prior to discharging into the receiving environment. The locations of the wetland treatment area inlet, mid-point, and outlet changed depending on the period of observation as indicated in Figure 3.3. During post-spring freshet conditions, the effluent flow in the wetland treatment area was at times low and discontinuous, and traveled in the upper wetland treatment area exclusively as subsurface flow, as denoted by the groundwater flow area in Figure 3.3.

The ambiguity in the location of the inlet and outlet points of the wetland treatment area over the treatment season required tracer studies to clarify appropriate sample locations. It is important to note that the exact location of wetland inlets and outlets are not always obvious for natural wetlands. Authors such as Yates *et al.* (2012) have presented treatment performance data without exact outlet sample locations. This ambiguity may have a major impact on the treatment performance results. For example, in a natural wetland, non-effluent watershed contributions in natural wetlands can act to dilute the effluent stream. Long-term monitoring and comparison to previous datasets becomes cumbersome if outlet locations are not consistent.

3.1.2. Reference Wetland

A reference wetland site was selected for baseline comparison of an un-impacted wetland to compare to the study site. The reference wetland site was used to compare biogeochemical observations, to develop an understanding of the background treatment performance concentrations, and to compare land cover types, and distribution of vegetation species. The reference wetland was located approximately 1.3 km north-east of the wastewater treatment facility at a latitude of 64° 10’ 28” N, and a longitude of 083° 10’ 44” W. The reference site wetland was characterized as a mid-arctic intermediate to rich fen according to the National Wetlands Working Group classification scheme (National Wetlands Working Group 1988; 1997); however, the wetland was not low in oxygen saturation. The reference site consisted of a small stream that discharges into a small shallow freshwater lake approximately 60 m wide and 170 m long, which then discharges to another stream. The reference site was characterized by similar gently sloping topography, and surficial and bedrock geology.

3.1.3. Watershed Setting of the Treatment Area Wetland

An important characteristic of the Coral Harbour wetland treatment area is the overall watershed setting. Figure 3.4 shows the watershed setting of the wetland treatment area. The watershed setting is important because dilutive effects are related to the wetland-to-catchment ratio. The total watershed area that contributes runoff to the treatment area outlet point at the receiving water lake is 83 ha. The sub-basins that contribute water to the wetland treatment area are composed of two main sub-basins, hereafter referred to as "wetland watershed sub-basins." These wetland sub-basins preferentially receive effluent from the WSP depending on the timing in the treatment season. The wetland watershed sub-basin where effluent was observed to be flowing during the spring freshet had an area of 7 ha. The wetland watershed sub-basin where the effluent was observed to be flowing in the wetland treatment area during the post-spring freshet period had an approximate area of 15 ha. Which is slightly more than double that of the spring freshet watershed sub-basin.

Figure 3.1 Site locator map of Coral Harbour, NU, Canada.

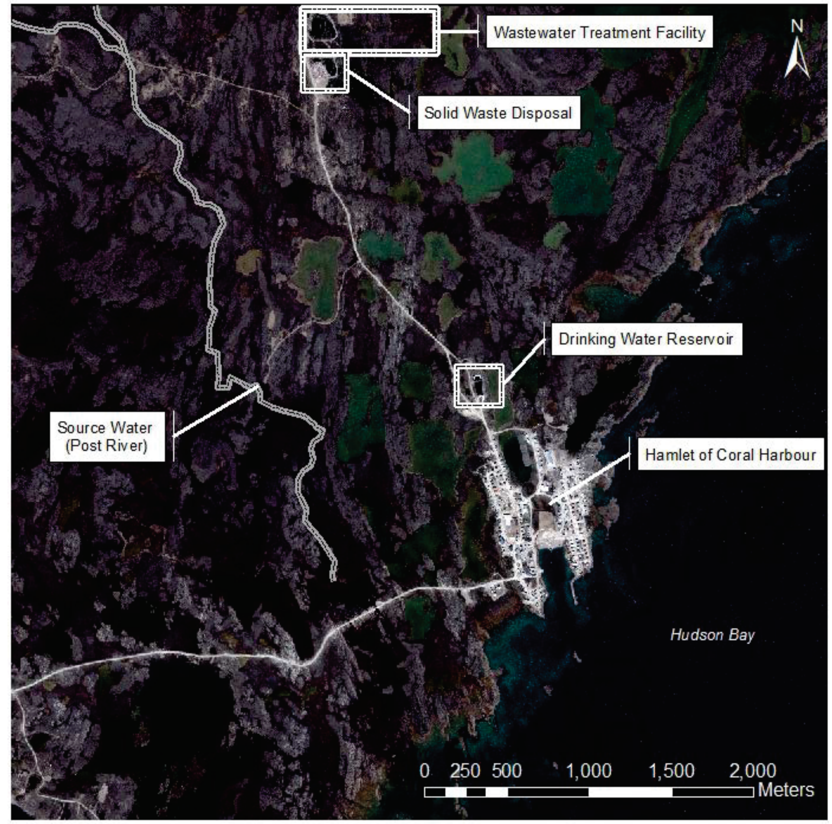

Figure 3.2 Plan view of the hamlet of Coral Harbour, NU showing the locations of municipal infrastructure. QuickBird satellite imagery (July 13, 2006).

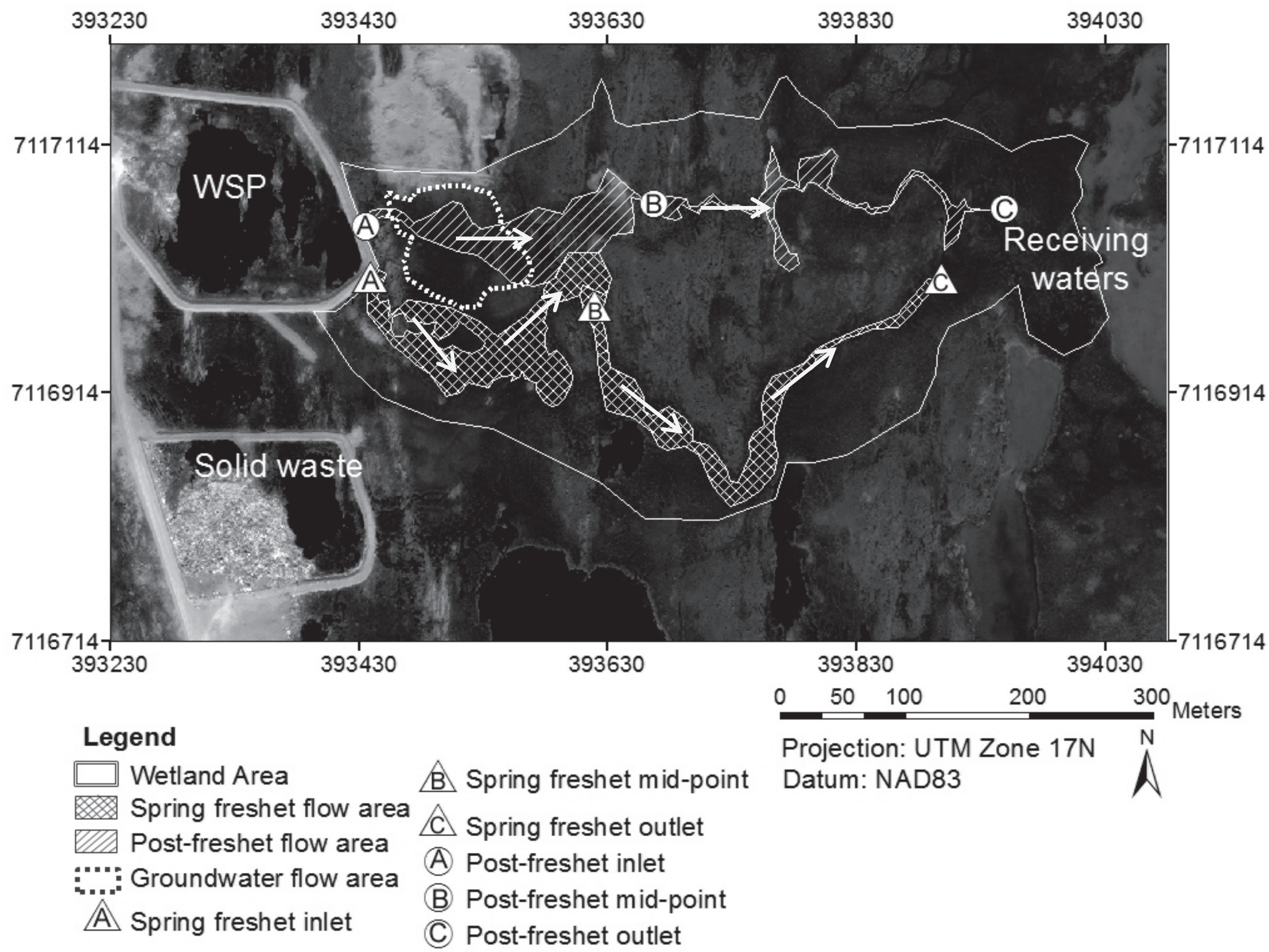

Figure 3.3 The WSP and receiving wetland treatment area in Coral Harbour, NU. The main flow paths of the effluent over the treatment season are indicated. The arrows denote the direction of effluent flow. The locations the inlet (A), mid-point (B), and outlet (C) indicate where treatment performance sampling, water quality monitoring, and stream gauging were conducted over the treatment season. QuickBird satellite imagery (July 13, 2006).

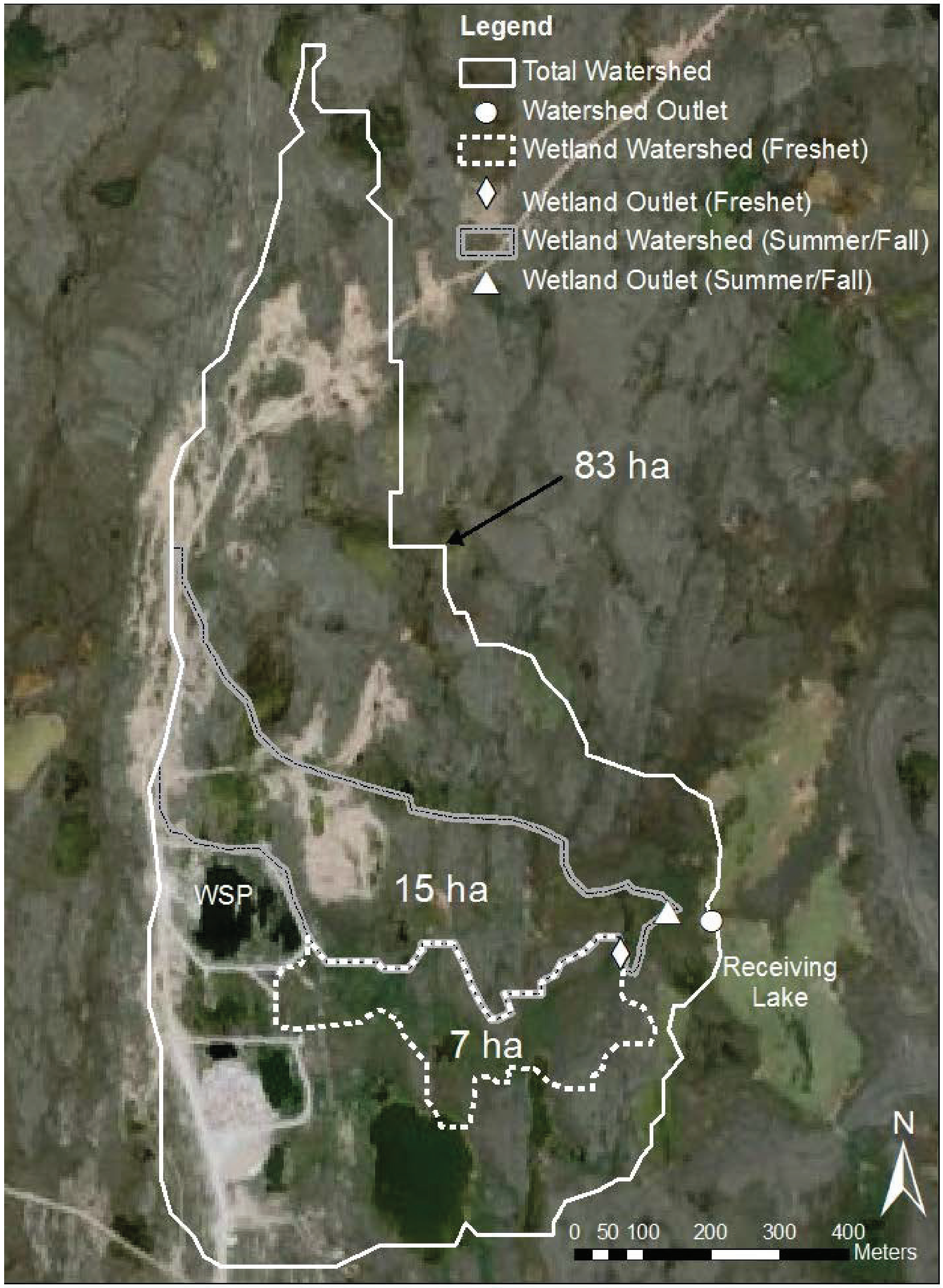

Figure 3.4 Total watershed delineation of hydrologic contribution to the wetland treatment area outlet discharge at the receiving lake. The wetland treatment area is part of two main watershed sub- basins where effluent discharge direction varies seasonally between the spring freshet and post-freshet. The total area of the treatment area sub-basin watershed is 22.4 ha. The overall watershed outlet is indicated, as well as, the wetland treatment area watershed sub-basin outlets.

3.2. Data Collection Strategy

The identification of seasonal variations in the wetland hydraulics, biogeochemistry, and treatment performance, was facilitated by studying the site on multiple occasions over the duration of two treatment seasons. Sampling trips were conducted on four occasions from June 2011 to September 2012. The dates of site visits, representative study periods, number of field days, and rounds of treatment performance samples collected are summarized in Table 3.1.

Table 3.1 Schedule of site visits.

Dates of site				
Arrival	**Departure**	**Study period**	**Number of days**	**Rounds of treatment performance sampling**
14/6/2011	25/6/2011	Spring freshet	12	2
6/9/2011	17/9/2011	Fall	12	2
28/6/2012	14/7/2012	Mid-summer	17	2
31/8/2012	1/9/2012	Fall	2	1

In situ monitoring devices were deployed to collect longer term datasets in between site visits for some parameters. The site was studied during the spring freshet (June 2011) to assess wetland treatment area performance during the snowmelt, when thawed stored wastewater began entering the wetland treatment area. The mid-summer trip was scheduled to capture the tail-end of the spring freshet runoff in 2012 for verification of 2011 results, and to monitor the wetland treatment area in warmer temperature conditions. The fall trips were scheduled to monitor the wetland treatment area at the end of the treatment season when the treatment performance was anticipated to be the most favorable. Three main tasks were conducted in the wetland treatment area monitoring program, which consisted of: (i) physical characterization; (ii) hydraulic characterization; and (iii) treatment performance assessment.

3.3 Physical Characterization

The physical characterization of the site consisted of the collection of vegetation and topographic data in the wetland treatment area. The vegetation survey facilitated characterization of effluent effects on the vegetation and land cover types in the wetland treatment area. The topographic survey data was used for the watershed delineation and determination of hydraulic heads in the subsurface flow area.

3.3.1. Topography

The topography of the wetland treatment area and surrounding area, the position and elevation of the sample points, and effluent flow paths, were determined with a Real-Time Kinematic (RTK) topographic survey. A HiPer Ga (Topcon Positioning System, Inc., Livermore, California, United States) Global Positioning System (GPS) unit was used to conduct the survey. The base station was set-up over an existing Natural Resources Canada monument numbered CP95259m.11O. The monument was positioned at latitude of 64° 08’ 41.0” N and a longitude of 083° 10’ 26.7” W, and had an orthometric elevation of 13.770 m. The monument was coordinated in the NAD83 CSRS datum, using an 8 hour static survey, processed with the Precise Point Positioning (PPP) service offered by Natural Resources Canada. The baseline length between the base station and the site was approximately 2.4 km. The survey was designed to cover an approximate area of 35 ha with a grid spacing of approximately 10 m. A larger spatial extent Digital Elevation Model (DEM) with a 30 m spatial resolution was sourced from the Natural Resources Canada online database Geobase for supplementary use in the watershed delineation (Government of Canada, 2004). A DEM is a raster dataset representation of terrain elevation.

3.3.2. Vegetation and Land Cover

The vegetation survey was performed in July 2012 by positioning transects across the effluent flow path along the length of the wetland treatment area from the inlet to the outlet. Each transect was approximately 120 m in length. A total of 108 sample points were taken at approximately 20 to 25 m spacing along the length of each transect The sample point distribution is shown in Figure 4.1. The data collection procedure at each sample point consisted of laying a 1 m x 1 m quadrat on the ground, a photograph was taken at distance to show all of the vegetation and land cover within the quadrat, and supplementary photographs were taken at a closer range to provide additional detail for desktop identification of vegetation species. A soil moisture content measurement was taken with a HydroSense soil moisture probe (Campbell Scientific Australia, Garbutt, Queensland, Australia). The average height of the vegetation was estimated with a measuring tape. The location of each sample point was recorded with a handheld GPS.

The desktop analysis of the field data was conducted by assessing the photographs taken in the field to identify the species of vegetation and land cover types for each of the sample points. Two main resources were used to identify the vegetation, which were: Mallory and Aiken (2004), and Polunin (1948). The primary dominant vegetation or land cover type was estimated for each sample point based on estimated spatial distribution in the quadrat. The remaining vegetation and land cover types were noted in the order of prevalence in each quadrat. The primary and secondary dominant vegetation or land cover types were listed for each sample point in a Microsoft Excel™ spreadsheet. Main classes of vegetation and land cover types were identified by grouping together vegetation and land cover types that were frequently observed to occur in association with one another. A new class was created for each primary dominant vegetation or land cover type except in cases when the primary vegetation types were also observed as a secondary dominant vegetation type and had a small sample size representation (*i.e.,* <5% of the total sample population). For each of the classes created, the frequency of occurrence with secondary dominant types was noted as a percentage of the total number of samples for each respective class. When the frequency of occurrence of secondary vegetation or land cover types was greater than 10% of the class sample size, it was reported as a dominant secondary vegetation or land cover type.

A map of the spatial distribution of the classes of vegetation and land cover was developed from a three band (RBG) QuickBird satellite image taken in July 13, 2006 with 0.6 m spatial resolution. A supervised image classification was performed on the satellite image using ESRI ArcGIS ArcMap 10© software. The maximum likelihood tool in Image Classification toolbar in ArcMap 10© was used to perform the supervised image classification. The sample points were used to develop a sample training signature file; this was done by selecting sample points representative of a particular class and training surrounding area pixels to be recognized as that class. The output image classification raster file was representative of the highest probability class assignment for each of the pixel values in the original multi-band raster. All cells of the input raster were classified for this analysis. After the classification was performed, the Majority Filter tool in the Spatial Analyst toolbar in ArcMap 10© was applied twice to rasterize the classified image to smooth small scale variations in the classified output raster image.

To determine the accuracy of the supervised image classification, the ground truthed dominant vegetation or land cover type from the survey was verified for agreeance with the class designation from the classification output results. The overall accuracy of the classification was 70%. The spatial resolution is a potential explanation for the accuracy because dominant vegetation types vary on small scales of less than 1 meter. Even 0.6 m spatial resolution data is problematic because the dominant vegetation varies on a spatial scale of less than a meter in the wetland treatment area. Each dominant class of vegetation had other secondary dominant vegetation and land cover types that may have comparable pixel values to other classes, which could also lead to small inaccuracies in the classification. Temporal resolution is also a potential problem as the vegetation and land cover distribution may have changed since the date of the image collection in 2006, as the WSP was constructed in 2003, and the field data was collected in 2012. Despite this date discrepancy, the results of the supervised classification were still useful to deduce some basic trends observable from the spatial distribution of the vegetation and land cover in relation to effluent flow areas.

3.4. Hydraulic Characterization

The hydraulic characterization of the wetland treatment area provides context for the treatment performance assessment. Characterization of the hydrological and hydrogeological setting of the site provides an understanding of the underlying mechanisms that influence the treatment performance of the wetland treatment area.

3.4.1. Hydrological Characterization

The hydrology of the wetland treatment area was characterized over various periods in the treatment season to characterize temporal changes.

3.4.1.1. Watershed Delineation

The watershed delineation was performed with ESRI ArcGIS ArcMap 10© software. Two DEMs were used for the watershed delineation of the wetland treatment area and the area surrounding the WSP. One of the DEMs was sourced from the Natural Resources Canada online database Geobase (Government of Canada, 2004). The Geobase DEM provided a large spatial coverage; however, the Geobase DEM has a poor spatial resolution due to a raster cell size of 30 m. A second DEM developed from the RTK topographic survey data was used to refine the watershed delineation for the wetland treatment area. The approximate spatial resolution of the RTK topographic dataset was 10 m. Therefore, the point file was converted to a 10 x 10 m cell size DEM with the Point to Raster (Conversion) tool. The terrain preprocessing tool within the Arc Hydro toolbar was used to perform the watershed delineations on each DEM. The total watershed area contributing to the wetland treatment area outlet at the receiving lake, and the sub-basins of the wetland treatment area, were determined using the Point Delineation tool in the Arc Hydro toolbar.

3.4.1.2. Discharge Measurement

Surface water flow rates in the wetland treatment area were measured on a near daily frequency at key spatial locations during the study periods. The locations of the stream gauging points in the wetland are indicated in Figure 3.3. Stream gauging points were established at various locations along the main effluent flow paths where changes in flow amounts were suspected (*e.g.*, if two separate stream channels converged or diverged). In particular, stream gauging points were established in areas of the wetland where non-effluent watershed hydrologic contributions were suspected of entering the wetland treatment area. Stream gauging locations were selected in stream channels characterized by relatively straight and uniform flow. The two most important stream gauging points were located at the wetland inlet and outlet, and were critical in forming an understanding of the overall wetland treatment area water budget.

The velocity – area method was used to determine the discharge at stream gauging locations (Dingman, 2002). The apparatus used to measure the current velocity and depth was a 625DF2N digital pygmy meter (Gurley Precision Instruments, Troy, New York, United States). The pygmy meter was equipped with a 2 m wading rod, cable and 1100 model indicator digital read-out. Occasionally, a 10 m measuring tape was used to measure the stream channel width in place of the wading rod.

The stream gauging procedure involved measuring the width of the cross sectional area of the stream section where the discharge was to be determined. The stream gauging cross-sections were oriented perpendicular to the principal direction of flow. Measurements were made along the width of the stream cross-section to capture discharge variations along the width of the channel; the locations of these measurements were termed “sub-sections.” Typically, the spacing of sub-sections where individual velocity and depth measurements were made ranged from 5 to 30 cm depending on the overall cross-sectional width of the channel. Smaller sub-sectional spacing was used for narrow channels (*e.g.,* ~ 10 cm), and larger sub-sectional spacing was used for the relatively wider channels (*e.g.,* ~ 100 cm). The cross-sectional area was cleared of obstructions, such as rocks or vegetation, to facilitate ease of velocity and depth measurements, and to reduce the risk of back eddy formation. At each sub-section the total depth of the stream channel was measured with the wading rod. In all cases, the stream channel depth was less than 0.75 m, therefore it was valid to employ the Six-Tenths-Depth method to measure the current velocity at each point across the stream section. Within this method, current velocity measurements are made at a 60% of the total depth down from the water surface (Dingman, 2002). The pygmy meter was oriented in the direction of flow and, perpendicular to the stream cross-section. The digital read-out for the pygmy meter was set to output average current velocity every 16 seconds. Three sets of average current velocity measurements were made for each observation point across the stream cross-section.

The total discharge, Q (m^3/s), for each stream gauging section was calculated with:

$$Q = \sum_{i=1}^{N} v_{avg} \cdot A_i \qquad [3.1]$$

where:

v_{avg} = average current velocity for each sub-section (m/s), and

A_i = area of each sub-section (m^2).

In instances when the discharge was too small for stream gauging with the pygmy meter, the discharge was measured by recording the amount of time required for the stream flow to fill a bottle or bucket of a known volume. The discharge was calculated by dividing the volume of the receptacle by the time required to fill.

Stage–discharge relationships were developed for the wetland treatment area inlet and outlet, and continuous discharge hydrographs were created using automated stage measurement systems. The water levels at the inlet and outlet locations were measured with HOBO® U20 Water Level Loggers (Onset® Computer Corporation, Bourne, Massachusetts, United States). The loggers were deployed in anchored slotted PVC pipes 152 mm in length to protect the loggers from damage caused by debris. One logger was also deployed open to atmospheric pressure to use for correction due to barometric pressure changes. All the loggers were set at a sampling frequency of 2 hours for long-term deployment. The discharge at inlet and outlet locations was manually determined with stream gauging at various times during the deployment of the pressure transducers, and was typically conducted at least once daily while present on site. The pressure readings from the transducers were corrected for atmospheric pressure fluctuations by subtracting the atmospheric pressure from the submerged pressure. The water levels, h (m), at the deployment locations of the pressure transducers were determined using:

$$h = \frac{P}{\rho g} \qquad [3.2]$$

where:

P = absolute pressure (Pa),

ρ = density of water at field temperature (kg/m^3), and

g = acceleration due to gravity (m/s^2).

The water levels obtained from the loggers were plotted with the corresponding instantaneous discharge measurements for the inlet and outlet locations to develop stage-discharge relationships. A regression analysis was conducted on the discharge data and corresponding water levels by selecting the best R^2 fit of the trend line fits obtained with the Microsoft Excel™ Trendline tool. In all cases, the relationship between water level and discharge was best represented by an exponential equation. The exponential trend line equations were then used to estimate discharge from the water level data when no discrete measurements of discharge were available from stream gauging.

3.4.1.3. Evapotranspiration and Evaporation

The evaporation rate from the WSP and evapotranspiration rate of the wetland treatment area were estimated using a method by Hamon (1963). This method estimates potential evapotranspiration (*PET*) using mean daily air temperatures, and was selected based on the climate data available to parameterize a *PET* estimation method. The mean daily temperatures were obtained from Environment Canada's historical climate data records for Coral Harbour, NU (Environment Canada, 2013). The *PET* (mm/d) was estimated with the following equation (Dingman, 2002; Hamon, 1963):

$$PET = 29.8 \cdot D \cdot \frac{{e_a}^*}{T_a + 273.2} \qquad [3.3]$$

where:

D = day length (hr),

${e_a}^*$ = saturated vapor pressure at mean daily air temperature (kPa), and

T_a = mean daily air temperature (°C).

The saturated vapor pressure was calculated using an equation from Dingman (2002) as follows:

$${e_a}^* = 0.611 \cdot exp(\frac{17.3 \cdot T}{T + 237.3}) \qquad [3.4]$$

where:

T = temperature (°C).

The day length, D, was calculated using the following equation (Dingman, 2002):

$$D = T_{hr} + |T_{hs}|, \tag{3.5}$$

where T_{hr} (hours), and T_{hs} (hours), are the time of sunrise and sunset in relation to solar noon given by:

$$T_{hr} = -\frac{cos^{-1}[-tan(\delta) \cdot tan(\Lambda)]}{\omega}, \tag{3.6}$$

and

$$T_{hs} = +\frac{cos^{-1}[-tan(\delta) \cdot tan(\Lambda)]}{\omega} \tag{3.7}$$

where:

ω is the angular velocity of the earth's rotation (15°/hr = 0.2618 radian/hour), Λ is the latitude (decimal degrees), δ (decimal degrees) is the declination of the sun given as follows:

$$\delta = \left(\frac{180}{\pi}\right) \cdot [0.006918 - 0.399912 \cdot cos(\Gamma) + 0.070257 \cdot sin(\Gamma) - 0.006758 \cdot cos(2 \cdot \Gamma) + 0.000907 \cdot sin(2 \cdot \Gamma) - 0.002697 \cdot cos(3 \cdot \Gamma) + 0.00148 \cdot sin(3 \cdot \Gamma)],, \tag{3.8}$$

and Γ (radians), is the day angle determined by:

$$\Gamma = \frac{2 \cdot \pi \cdot (J - 1)}{365} \tag{3.9}$$

where:

J = day number (J = 1 on January 1 and J = 365 on December 31).

3.4.1.4. Areal Hydraulic Loading Rate

The areal *HLR* is defined as the rate of volumetric wastewater addition to a wetted treatment area (Kadlec and Wallace, 2009). The *HLR* is an important parameter in treatment wetland design because there are ranges in which wetlands treat most optimally (Jing *et al*., 2002). It has been shown that *HLR*s are strongly correlated to the first order rate coefficients that are typically used in wetland design sizing calculations (Jamieson *et* *al*., 2007). Therefore it is important to select an *HLR* range that is representative of the site to avoid erroneous sizing calculations. The *HLR* calculation presented here is based primarily on instantaneous daily influent discharge measurements at the wetland inlet; therefore the *HLR*s reported are only representative of the daily *HLR*s during the periods studied. The daily *HLR* was calculated for the treatment wetland area and the wetland watershed sub-basin. The $HLR_{wetland}$ on the wetland treatment area, (cm/d), was calculated with the following:

$$HLR_{wetland} = (\frac{Q_{in}}{A_w}) \cdot 100 \qquad [3.10]$$

where:

Q_{in} = influent discharge into the wetland treatment area from WSP (m^3/d), and

A_w = wetted area of the wetland treatment area observed in active flow (m^2).

The HLR_{ws} on the wetland watershed sub-basin (cm/d), was calculated as follows:

$$HLR_{ws} = (\frac{Q_{in}}{A_{ws}}) \cdot 100 \qquad [3.11]$$

where:

A_{ws} = area of wetland watershed sub-basin (m^2).

3.4.1.5. Wetland Treatment Area Water Budget

The water budget for the wetland treatment area was represented by:

$$V_{WSP} + P - V_{PET} - V_{out} = \Delta S \qquad [3.12]$$

where:

V_{WSP} = volume of influent discharged from WSP into the wetland treatment area (m^3),

P = volume of precipitation on the wetland watershed sub-basin (m^3),

V_{PET} = volume of evapotranspiration from the wetland watershed sub-basin (m^3),

V_{out} = volume of effluent discharged from the wetland treatment area (m^3), and

ΔS = change in storage volume in the wetland watershed sub-basin (m^3).

The water budget was estimated by calculating the volumetric addition of water to the wetland watershed sub-basin from precipitation, and influent flow from the WSP, as well as, the volume of water lost from the wetland treatment area from evapotranspiration, and effluent outflow. The precipitation data was obtained from historical environment Canada climate data records (Environment Canada, 2013). The volume of precipitation onto the wetland watershed sub-basin was calculated by multiplying the total daily precipitation by the 22 ha wetland watershed sub-basin area. The volume of water lost to *PET* was obtained by multiplying *PET* rates by the wetland watershed sub-basin area. The daily volume of wetland inflow and outflow was taken as the daily flow volumes projected from instantaneous discharge measurements obtained during stream gauging. Differences between the additions and abstractions from the wetland treatment area are representative of water gained or loss in storage. The wetland water budget assumed no seepage to groundwater due to continuous permafrost coverage.

3.4.2. Hydrogeological Characterization

The hydrogeology of the wetland treatment area was characterized to provide additional context for the treatment performance results. During the periods when the effluent is infiltrating into the groundwater and traveling exclusively as subsurface flow, there is potential for treatment of the effluent by way of adsorption of effluent constituents onto subsurface media and filtration.

3.4.2.1. Observation Wells

Observation wells were installed in the groundwater flow area to: determine the depth of the active layer, characterize groundwater flow by monitoring water table levels, and collect treatment performance samples. The boreholes for the wells were extended using a hand auger that had a core barrel end piece designed for stony soils. The well depths were selected based on the maximum depth achievable prior to caving in of materials, which ranged from 40 cm to 110 cm. The wells were constructed on-site with 25 mm diameter PVC casing and slotted screen sections. The depth to water in the boreholes was determined with a 30 m dipper-T water level indicator (Heron Instruments Inc., Dundas, Ontario, Canada). The length of screened sections for each observation well was chosen to assure that the water level measured would fall within the screened interval of the well. The well casing and screens were either screwed together using pre-made threads or by using 25 mm diameter PVC pipe couplers. The well bottom was fitted with a pointed PVC end cap and the top of the well was protected with a locking monitor well cap. A minimum of 10 cm of casing stick-up was used to avoid contamination of the well water by surface flow. The well bore was back-filled with silica sand and sealed with hydrated granular bentonite clay. The water levels and the depth to well bottom were monitored on a near daily frequency during the study periods.

3.4.2.2. Soil Classification

Soils were classified according to the Unified Soil Classification System (USCS) ASTM Standard Practice D-2487-11 (ASTM Committee D18 on Soil and Rock *et al.*, 2011). The soil samples were analyzed by performing a grain size distribution analysis. The method used for the grain size analysis was a modified version of the ASTM Standard Test Method D422 – 63 (ASTM Committee D-18 on Soil and Rock *et al.*, 2007). The procedure was modified to facilitate on-site analysis in the absence of a laboratory oven for drying soils. Each sample collected for grain size analysis had a dry weight of between 1 and 1.5 kg. The representative sample was collected in the field using a shovel and stored in a plastic bag. The soil sample was dried in a large stainless steel tray with a naphtha fuel burning WhisperLite™ stove (MSR®, Seattle, Washington, United States) while mixing with a metal spatula. The dry weight of the sample was measured with a scale and recorded. The sample was then washed over a No. 200 sieve multiple times until the wash water was relatively clear. The sample was dried and weighed again after washing over the No. 200 sieve. The remaining dried sample was shaken through a stack of sieves composed of the following: 25.4 mm., 19.1 mm, 12.7 mm, No. 4, No. 10, No. 30, No. 100, No. 200, and a bottom collection pan. The weight retained on each sieve was weighed and recorded. The grain size distribution curves were plotted with GEOSYSTEM® Software. The percent finer (%) for each sieve size was calculated according to:

$$Percent\ finer = \left(\frac{M_s - M_r}{M_s}\right) \cdot 100 \qquad [3.13]$$

where:

$M_s =$ total mass of dried sample (g), and

$M_r =$ mass of sample retained on the sieve (g).

3.4.2.3. Hydraulic Conductivity

The field saturated hydraulic conductivity (K_{fs}) was determined on-site with a Guelph permeameter system. The operation of the permeameter and calculation of K_{fs} was derived from the Model 2800K1 Guelph Permeameter Operating Instructions (Soilmoisture Equipment Corp., 2008). A total of four permeameter tests were conducted at various locations in the groundwater flow area. The depth of the tests was limited by the instrument and hand augers. Therefore, the K_{fs} values are representative of the upper layer of the overburden (<1 m). A constant head permeability test was performed in the laboratory as a verification on the K_{fs} values determined on site according to ASTM Standard Test Method D2434 – 68 (ASTM Committee D18 on Soil and Rock *et al.,* 2006).

3.4.2.4. Groundwater Characterization

The hydraulic gradient and direction of groundwater flow was determined using a spreadsheet method GRADIENT.xls in Microsoft® Excel™ which was developed by Devlin (2003). Three observation wells in triangular formation were used for the determination of hydraulic gradient and direction. The location of the observation wells are indicated in Figure 4.10. The average linear velocity, (m/s), of the groundwater in the groundwater flow area indicated in Figure 4.10 was estimated with the following equation (Fetter, 2001):

$$v = \frac{K_{fs} i}{n} \quad [3.14]$$

where:

$i =$ hydraulic gradient (m/m), and

$n =$ effective porosity (dimensionless).

The advective time of travel, *t* (days),through the groundwater flow area is estimated with:

$$t = \frac{d}{v} \cdot 60\ seconds \cdot 60\ minutes \cdot 24\ hours \quad [3.15]$$

where:

d = horizontal distance of travel (m).

3.4.3. Hydraulic Residence Time Determination

Tracer tests were conducted to characterize the hydraulic conditions at the site. In particular, the tracer tests allowed for determination of the effluent HRT within the wetland, which related to the treatment potential of the wetland. The HRT is representative of the average amount of time required for water and conservative solutes to move through the wetland. Generally, the longer the HRT of a wetland, the greater the amount of treatment is attainable (Knight *et al.*, 1987; Knox *et al.*, 2008). The HRT determination was facilitated by conducting a tracer test, which involved the injection of a conservative solute upstream and measurement of the concentration over time at a defined downstream location.

3.4.3.1. Location and Timing of Tracer Tests

Individual areas in the wetland where a tracer test was conducted were termed "Wetland Segments" (WS). The locations of each WS are indicated in Figure 3.5. The approximate areas and average depths of the WSs are provided in Table 3.2.

Table 3.2 Approximate dimensions of wetland segments during the treatment season.

Study period	Wetland segment	Average depth (m)	Area (m^2)
Spring freshet	WS-1	0.07	8 374
	WS-2	0.06	6 246
Post-freshet	WS-3	0.04	7 508
	WS-4	0.08	3 138

The surface water tracer tests on both the WS-1 and WS-2 wetland segments were performed in June 2011. The wetland segments WS-1 and WS-2 represent the main effluent surface water flow paths resulting from the hydrological conditions during the June 2011 spring freshet discharges. The WS-1 area is comprised of the wetland inlet and upper half of the wetland. The WS-2 area spans the lower half of the wetland from the mid-point to the approximate outlet discharge point into the receiving lake. The WS-3 area represents the upper half of the wetland and wetland inlet, during the moderate flow conditions, observed in post-freshet period in September 2011 and July 2012. Tracer testing conducted on segment WS-3 in July 2012 comprised of both a surface water and a groundwater tracer test. The wetland segment WS-4 is representative of the lower half of the wetland where surface water flow was observed during all study periods. The wetland segment WS-4 was the only surface water effluent flow path from mid-season in July 2012 through September 2012. The tracer testing for the wetland segment WS-4 was performed in September 2011.

3.4.3.2. Tracer Selection

The fluorescent dye tracer Rhodamine WT (RWT) was selected for use in the surface water tracer tests. The standard concentration of the dye was 200 g/L RWT (*i.e.,* 20% RWT by weight) (Keystone Aniline Corporation, Inman, South Carolina, United States). RWT is commonly used to perform hydraulic tracer studies in surface water wetland environments (Headley and Kadlec, 2007). RWT does possess some limitations; particularly, RWT does not act entirely as a conservative tracer as it undergoes photolysis and sorption onto stream channel substrate and vegetation, which makes it an unfavorable tracer in some environments. Additionally, RWT has been demonstrated to be temperature dependent (Smart and Laidlaw, 1977). Dierberg and Debusk (2005) suggest that the determination of key hydraulic parameters such as *HRT*, number of tanks-in-series (TIS), and dispersion are not affected by the slight non-conservative behavior of RWT in cases where a well-defined tracer response curve is obtained. Headley and Kadlec (2007) state that for wetlands with short nominal residence times (*i.e.,* less than one week), RWT is a suitable tracer choice. The site characteristics were such that the RWT tracer use did not impact the accuracy of the hydraulic parameter estimation, due to relatively short residence times within the wetland, and well defined tracer response curves.

The WS where effluent flow was characterized as subsurface flow for part of the mid-summer is denoted by the WS-3 area in Figure 3.5. The characterization of the hydraulic properties of the subsurface flow region of WS-3 was of interest from a treatment perspective because this portion of the wetland treatment area could provide enhanced treatment due to filtration, prolonged residence times, and sorption of wastewater constituents onto soil media. The use of RWT for the groundwater tracer test was unfavorable due to the suspected longer residence time within the subsurface segment and sorptive tendencies of RWT. For this reason, the conservative tracer sodium bromide (NaBr) was selected for the tracer test of the subsurface flow segment of the wetland treatment area. The ≥99% extra pure, anhydrous NaBr used for the tracer test was sourced from Acros Organics (Thermo Fisher Scientific Inc., New Jersey, United States). NaBr is another common tracer choice for tracer studies in wetlands (Wachniew *et al.*, 2003; Headley and Kadlec, 2007). In this context, it was favorable for application as a groundwater tracer due to conservative hydraulic behavior, non toxic nature, and low adsorption potential, which would facilitate an accurate estimate of the HRT of the subsurface flow segment of the wetland treatment area (Bridson-Pateman *et al.*, 2013). A potential problem encountered with NaBr tracer is the tendency for concentrated brine solutions to behave non-conservatively due to the generation of density gradients, causing the denser bromide solution to sink to the bottom of the water column (Headley and Kadlec, 2007). The generation of density gradients was avoided with use of minimal amounts of tracer, adequately diluted prior to injection.

3.4.3.3. Tracer Injection

The amount of RWT tracer injected for each surface water tracer study was determined such that the RWT concentration at the tracer termination point (*i.e.,* the outlet of the wetland segment) was targeted at 200 μg/L. The mass of NaBr injected into the subsurface flow segment of the wetland treatment area was calculated such that the outflowing bromide (Br^-) concentration was above 26 μg/L; which is the minimum detection limit on the Ion Chromatography (IC) instrument used for analytical determination of Br^- concentration. In addition, Kadlec and Wallace (2009) suggest using a peak tracer response concentration of at least 50 times the minimum detection limit of the tracer. The target Br^- concentration desired at the tracer termination point used in the injection mass estimate was 25 mg/L. An estimation of the mass of injected tracer, M_{in} (g), required to obtain the target concentration at the termination point of the tracer was calculated according to the re-arranged Equation 3.16 from Kadlec and Wallace (2009) as follows:

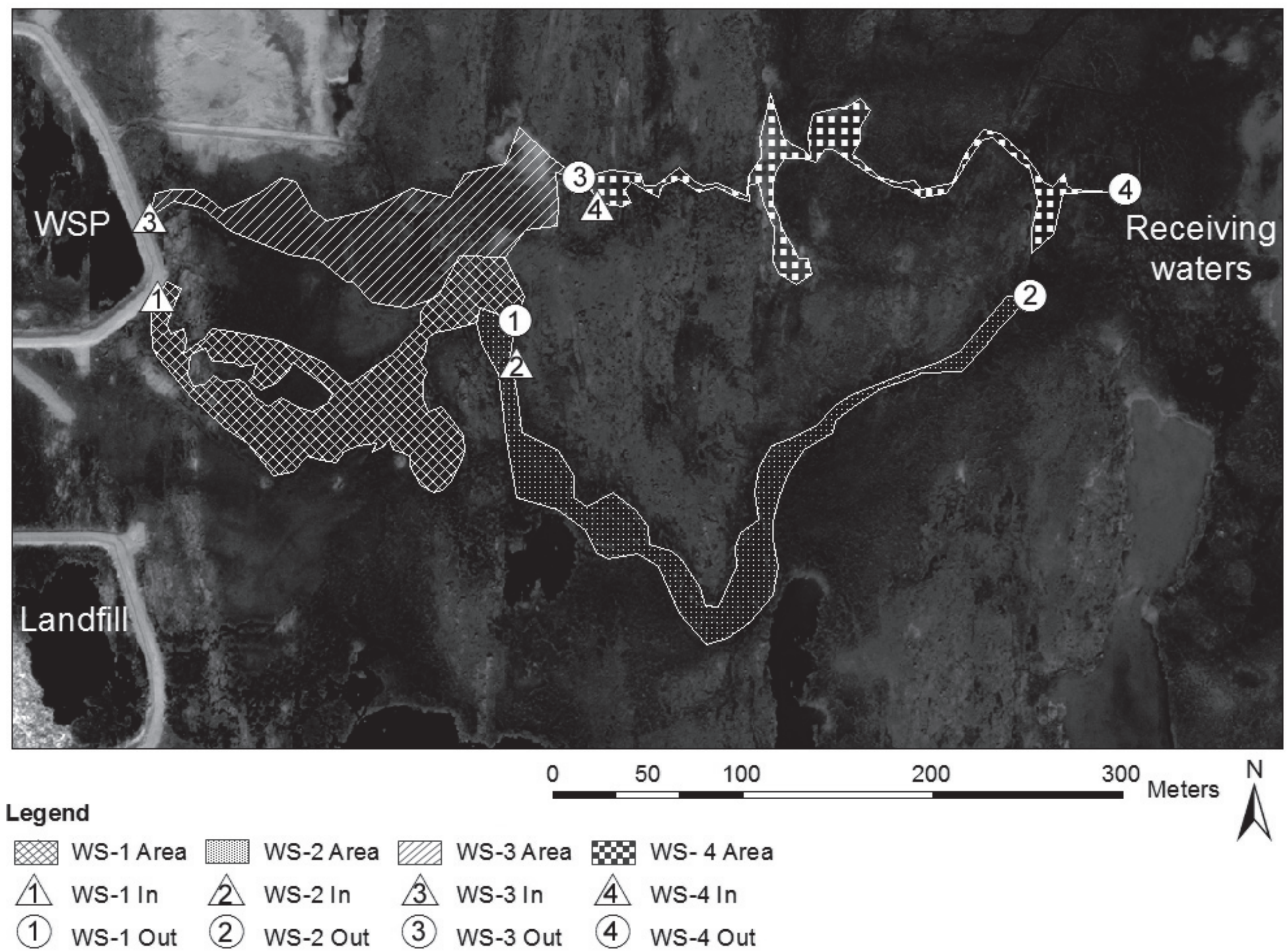

Figure 3.5 Wetland segments WS-1, WS-2, WS-3, and WS-4, which each underwent separate tracer tests. The main flow area during the spring freshet is represented by WS-1 and WS-2. The summer and fall flow area is denoted by WS-3 and WS-4. The injection points of the tracer for each segment are denoted by triangle symbols (In). The downstream tracer monitoring locations are represented by the circle symbols (Out). QuickBird satellite imagery (July 13, 2006).

$$M_{in} = C_t V_n \quad [3.16]$$

where:

$C_t =$ target concentration at the tracer termination point (g/m^3), and

$V_n =$ nominal wetland water volume (m^3).

Dierberg and Debusk (2005) found that the injection rates of tracers can cause the formation of density gradients. Therefore, the tracers were injected over a prolonged time interval such that the ambient discharge at the injection location was not overwhelmed by the injection of the tracer. Additionally, the tracer was diluted with water collected on site prior to the tracer test, and stored in 11 and 38 L buckets. Remaining tracer residue in the mixing buckets was rinsed and dumped into the injection stream at least three times following tracer injection. The specific details of each tracer test performed are summarized in Table 3.3.

Table 3.3 Description of the tracer tests performed, including tracer type, tracer amount, dilution water used, and duration of injection time.

Wetland segment	**Tracer date (m/d/yyyy)**	**Flow type**	**Tracer type**	**Tracer amount**	**Dilution water volume (L)**	**Duration of injection (min)**
WS-1	6/18/2011	Surface	RWT	500 ml	44	10
WS-2	6/17/2011	Surface	RWT	500 ml	44	10
WS-3	7/2/2012	Surface	RWT	300 ml	72	10
WS-3	7/8/2012	Subsurface	NaBr	1 kg	87	7
WS-4	9/7/2011	Surface	RWT	500 ml	109	15

3.4.3.4. In-situ Tracer Monitoring Techniques

The RWT tracer concentration at the termination point of the tracer test in the wetland was measured *in-situ* with an optical fluorometer YSI 6130 RWT sensor installed on a YSI multi-parameter water quality sonde (YSI Inc., Yellow Springs, Ohio, United States). The 6600V-2 sonde model was used during the 2011 study periods, and the 6920 sonde model was used in the 2012 study period. A two-point true dye calibration was performed according to the procedure stated in the YSI 6 series sondes user manual (YSI Inc., 2011). The upper limit for linear response of the optical sensor is 200 µg/l.

The Br^- concentration was measured *in-situ* with an AquiStar® TempHion™ Ion Selective Electrode (ISE) (Instrumentation Northwest, Inc., Kirkland, Washington, United States). A two-point calibration was performed prior to deployment in accordance with the procedure stated in the probe instruction manual (INW, 2012). The measurement range of the ISE probe is 0 - 10 000 ppm Br^- with a 0.1 ppm resolution.

A 6712C Compact Portable Autosampler (Teledyne ISCO, Inc., Lincoln, Nebraska, United States) powered by a 12V battery was positioned at the outlet of the WS for each tracer test to collect grab samples in the event of probe failure.

3.4.3.5. Sampling Frequencies

Due to the short hydraulic residence times (*i.e.,* < 1.4 days), the sampling frequency of the *in-situ* sondes used in the surface water RWT tracer tests ranged from 1 to 10 minutes. This was to ensure the rising limb of the tracer response curve was well defined. Whereas, Headley and Kadlec (2007) suggest that 30 to 40 samples are sufficient to define the tracer response curve, the RWT tracer tests were sampled at a much finer scale as the battery life was the only limitation on the number of sample points possible. The *in-situ* Br^- ISE probes were programmed to sample at 15 to 60 minute intervals for the first three days of the groundwater tracer test. The sampling frequency of the probes was decreased after the Br^- plume was undetected in the observation wells; which indicated slow movement of groundwater. The Br^- ISE probes were programmed to measure samples at two hour frequencies for the long-term deployment of 54 days to conserve battery life. In addition, the autosampler was positioned at the mid-point of the wetland, and was programmed to collect grab samples every two days to provide a back-up method of Br^- tracer concentration measurement.

3.4.3.6. Analytical Methods

The *in-situ* monitoring techniques were sufficient to characterize the tracer response curves for the surface water tracer tests conducted with RWT. Therefore, laboratory analytical techniques were not necessary for this parameter.

A Br^- ISE probe (Thermo Fisher Scientific Inc., Beverly, Massachusetts, United States) was used to analyze bromide concentrations in the groundwater grab samples extracted from the observation wells during the initial stages of the groundwater tracer test. A Nalgene hand vacuum pump (Thermo Fisher Scientific Inc., Beverly, Massachusetts, United States) fitted with 6.4 mm diameter vinyl tubing was used to collect the groundwater samples. The ISE probe was used in conjunction with an Orion Star™ meter (Thermo Fisher Scientific Inc., Beverly, Massachusetts, United States). The calibration of the probe with the meter, and measurement of the Br^- concentrations in the groundwater samples, was performed in accordance with the Br^- ISE probe and meter operating manuals (Thermo Fisher Scientific Inc., 2008 and 2010).

The grab samples collected from the autosampler were analyzed for Br^- at the laboratory using Ion Chromatography (IC). A filtration sample processor (Metrohm 788) was used, followed by injection into a Metrohm 761 Compact Ion Chromatograph. Calibration curves were created by measuring five standard concentrations of Br^- in deionized water. The following concentrations were used for instrument calibration: 0.8, 2, 4, 8 and 12 mg/L Br^-. Quality control samples of 0.8 mg/L were run every 10 samples. The method detection limit (MDL) was 0.26 mg/L.

3.4.3.7. Hydraulic Residence Time

The *HRT* of each WS was calculated with a moment analysis of the residence time distribution (RTD) curves generated from the tracer tests. The analysis of the RTD curves, and determination of *HRT*s were performed with Microsoft® Excel™ and checked with a quadrature algorithm in MATLAB®.

The output of each tracer test is a concentration versus time curve, which is termed a tracer response curve, represented by the notation $C(t)$ (µg/L or mg/L). An average background concentration of the tracer was measured at the WS outlets prior to tracer injections. The tracer concentrations of each response curve were corrected prior to numerical integration by subtracting the averaged background concentrations from the measured concentrations during the tracer tests. The RTD curve is a fractional representation of the amount of tracer that still exists in the wetland (or chemical reactor) at a particular time after tracer injection in the flow path. The common notation for the RTD is denoted by $E(t)$ (dimensionless). The RTD curve is defined by (Fogler, 2006):

$$E(t) = \frac{C(t)}{\int_0^{\infty} C(t)dt} \quad [3.17]$$

The mean residence time, τ (minutes), of the WS was determined by taking the first moment of the RTD as defined by (Fogler, 2006):

$$\tau = \int_0^{\infty} tE(t)dt \quad [3.18]$$

The tracer response and RTD curves were numerically integrated using Simpson's rule quadrature formula (Fogler, 2006). Depending on the number of data points, *n* (integer), one of two methods was chosen:

1. when (*n*/3) was an integer for *n* + 1 datum points;

$$\int_{X_0}^{X_n} f(X)dX = \frac{3}{8}h\,([f_0 + 3f_1 + 3f_2 + 2f_3 + 3f_4 + 3f_5 + 2f_6 + \cdots + 3f_{n-1} + f_n] \quad [3.19]$$

alternatively,

2. when *n* was an even number for *n* + 1 datum points;

$$\int_{X_0}^{X_n} f(X)dX = \frac{h}{8}[f_0 + 4f_1 + 2f_2 + 4f_3 + 2f_4 + \cdots + 4f_{n-1} + f_n] \quad [3.20]$$

where:

f(X) = y value at time *x*, and

h (minutes) is given by:

$$h = \frac{X_n - X_0}{n} \quad [3.21]$$

where:

$X_n =$ time at end of tracer test, and

$X_0 =$ time at start of tracer test.

3.4.3.8. Volumetric Efficiency

The term "nominal" describes the best case hydraulic conditions in the wetland where the theoretical maximum volume is available to accommodate actively flowing water, and hence facilitate treatment. In reality, the actual wetland volume that conveys active flow is smaller than the nominal volume due to preferential flow paths, the formation of dead zones, and physical obstructions within the wetland. The nominal residence time, τ_n (days), is the corresponding maximum theoretical residence time assuming the entire wetland volume is active in the conveyance of flow. The actual *HRT* of a wetland is always smaller than the τ_n, because 100% of the wetland volume is not available for active flow. The nominal residence time is given by Kadlec and Wallace (2009) as follows:

$$\tau_n = \frac{V_n}{Q} \qquad [3.22]$$

where:

V_n = nominal wetland volume (m^3), and

Q = discharge (m^3/d).

All segments of the Coral Harbour wetland were characterized by non-uniform flow, where outlet flows from each WS were generally greater than inlet flows due to external hydrologic inputs. To account for this, a flow rate adjusted nominal residence time, τ_{an} (d), was approximated using the following equation (Charazenc, 2003; Kadlec and Wallace, 2009):

$$\tau_{an} = \tau_i \left(\frac{ln(R)}{R-1}\right) \qquad [3.23]$$

where:

τ_i = inlet flow-based nominal residence time (d),

$R = \frac{Q_o}{Q_i}$, water recovery fraction (dimensionless),

Q_i = inlet flow rate (m^3/d), and

Q_o = outlet flow rate (m^3/d).

The volumetric efficiency describes the amount of short-circuiting and presence of dead zones in a wetland. Kadlec and Wallace (2009) define the volumetric efficiency of a wetland (e_v) by the following equation:

$$e_v = \frac{V_{active}}{V_n} \qquad [3.24]$$

where:

V_{active} = volume of the wetland that contains water in active flow (m^3).

The volumetric efficiency of each WS was determined with the following equation (Kadlec and Wallace, 2009):

$$e_v = \frac{\tau}{\tau_{an}}. \qquad [3.25]$$

3.5. Water Quality Monitoring

Basic indicators of water quality were monitored to characterize the overall water quality and biogeochemistry in the wetland treatment area, and the reference wetland. Basic water monitoring provides context for the treatment performance observations, as the biogeochemistry influences the treatment processes and dynamics within the wetland.

3.5.1. Discrete Sampling

Discrete (*i.e.,* instantaneous) water quality measurements of DO, pH, conductivity, and temperature were taken at each of the sample locations indicated in Figure 3.3. Discrete water quality measurements were conducted daily, at minimum, on most occasions. The measurements were made with a handheld YSI 600R or 600QS multiparameter water quality sonde (YSI Inc., Yellow Springs, Ohio, United States). The sondes were calibrated for all basic water quality parameters prior to each site visit according to the manufacturer's specifications (YSI Inc., 2011). The DO probe was calibrated daily on-site prior to use.

3.5.2. *In-situ* Sampling

Continuous water quality measurements were made with a 6600 V2 or 6920 V2 *in-situ* multiparameter water quality sonde (YSI Inc., Yellow Springs, Ohio, United States). The *in-situ* sonde was programmed to record measurements on a 1 or 2 hour sample frequency. The sondes were calibrated for all basic water quality parameters prior to each site visit according to the manufacturer's specifications (YSI Inc., 2011).

3.5.3. Light and Temperature

Light intensity and temperature were monitored continuously *in-situ* in the WSP and the wetland treatment area with HOBO Pendant® Temperature/Light data loggers (Onset® Computer Corporation, Bourne, Massachusetts, United States). The temperature and light pendants were programmed to measure on a 1 to 2 hour frequency.

3.5.4. Chlorophyll a and Pheophytin

Samples were collected for chlorophyll a and its degradation product pheophytin in the WSP, wetland, and reference wetland in July 2012. Samples were filtered through GF/C Whatman glass fiber filters and 1 mL of a 1% magnesium carbonate solution was added immediately after collection, the filters were placed in plastic petrie dishes, covered in aluminum foil, frozen, and stored in the dark until analysis. The method used for analysis was according to Yentsch and Menzel (1963), as modified by Holm-Hansen *et al.* (1965), and recommended by Strickland and Parsons (1968).

3.6. Treatment Performance Assessment

The treatment performance assessment of the wetland was performed by collecting samples from key locations in the wastewater treatment system and analyzing for a suite of parameters. The following describes the methodology followed to facilitate the collection of the treatment performance data.

3.6.1. Field Sampling

Treatment performance samples were taken raw from the wastewater trucks, the WSP, various points in the wetland, and the reference wetland as indicated in Figures 3.2 and 3.3. The water samples were collected as grab samples in 50 mL, 500 mL and/or 1 L clear plastic sample bottles sterilized with 70% ethyl-alcohol. The sample bottles were rinsed three times with water from the sample point prior to sample collection. Approximately three liters of sample water was collected at each sample point. Samples that were collected for metals analysis were acidified to a pH of ≤ 2 with nitric acid immediately following sample collection. A sample collection pole was used to collect samples from the WSP. Two rounds of treatment performance samples spaced approximately one week apart were collected and analyzed for each site visit. A minimum composite sample of 16 hours was collected at the mid-point of the wetland with a 6712C Compact Portable Autosampler (Teledyne Isco Inc., Lincoln, New England, United States) for each round of treatment performance samples. A grab sample was also collected at the composite sample location to use for *E. coli* analysis. Basic water quality measurements of DO, pH, conductivity and temperature were collected along with each treatment performance sample with the handheld 600R or 600QS multiparameter water quality sonde (YSI Inc., Yellow Springs, Ohio, United States). The treatment performance samples were stored in insulated coolers with ice packs for transportation from the site to the laboratory.

3.6.2. Analytical Methods

The treatment performance samples were stored in a refrigerator at 4 ºC upon arrival at the laboratory and analyzed within their respective holding times specified by the manufacturers and/or APHA (1998). Each treatment performance sample was tested for an array of parameters including: $CBOD_5$, *E. coli*, TSS, VSS, TN, Total Ammonia Nitrogen (TAN), NH_3-N and TP. The $CBOD_5$, TSS, and VSS were analyzed according to Standard Methods (APHA, 1998). *E. coli* was analyzed using IDEXX Colilert®-18 and Quanti-Trays® according to the manufacturer's procedure in IDEXX Laboratories, Inc. (2012). TN was analyzed using Hach® TN Test 'N Tubes™ (0.5 to 25.0 mg/L N), according to the manufacturer's procedure in Hach[1] (2012). The low concentration TP samples were analyzed using the Hach® TP Test 'N Tube™ for low range samples ranging from 0.06 to 3.50 mg/L PO_4^{3-} according to the procedure described in Hach[2] (2012). For higher concentration phosphorus samples, Hach® TP Test 'N Tube™ for high range samples ranging from 1.0 to 100.0 mg/L PO_4^{3-} were used following the procedure in Hach (2008). Ammonia/ammonium was analyzed using a Thermo Fisher Scientific High Performance Ammonia Ion Selective Electrode (ISE) (Thermo Fisher Scientific, Beverly, MA, United States) according to the method specified in the User Guide (Thermo Fisher Scientific, 2007). The ISE probe was used in conjunction with a Thermo Scientific Orion Star™ meter (Beverly, Massachusetts, United States). As a result of the portability of the ammonia ISE, the majority of the ammonia/ammonium samples were collected and analyzed immediately on site. The NH_3-N concentration was calculated using pH levels measured in the field according to the method described in CCME (2010). In July and

September 2012, the treatment performance samples were also analyzed for a suite of 23 metals with an X-Series 2 Inductively Coupled Mass Spectrometry (ICP-ms) (Thermo Fisher Scientific, Beverly, Massachusetts, United States). The suite of metals analyzed had MDLs that varied from 0.4 to 10 µg/L. The metal samples were analyzed according to standard methods (APHA, 1998). Metals samples were all nitric acid heat digested and centrifuged prior to analysis.

3.6.3. Interpretive Methods

The percent (%) and log reductions for the treatment performance parameter concentrations observed at the wetland inlet, mid-wetland, and outlet were calculated using Equations 3.26 and 3.27 respectively as follows:

$$\%\,reduction = \frac{\left(C_{raw} - C_{sp}\right)}{C_{raw}} \cdot 100 \qquad [3.26]$$

$$log\ reduction = log(C_{raw}) - log(C_{sp}) \qquad [3.27]$$

where:

C_{raw} = raw treatment performance parameter concentration (mg/L or MPN/100mL), and

C_{sp} = sample point treatment performance parameter concentration (mg/L or MPN/100mL).

3.6.4. Areal Biochemical Oxygen Demand Loading Rate

The areal BOD loading rate, *BLR* (kg/ha·d), describes the rate at which BOD is applied to a wetland area. The *BLR* is an important design parameter as it affects the potential BOD treatment rate for the effluent (Kadlec and Wallace, 2009). The *BLR* was calculated with:

$$BLR = \frac{Q_{in}C_{in}}{A} \qquad [3.28]$$

where:

Q_{in} = influent discharge into wetland segment (m^3/d),

C_{in} = influent BOD_5 concentration into wetland segment (kg/m^3), and

A = area of wetland segment (ha).

3.7. Natural Wetland Treatment Model

Determination of the rate coefficients was facilitated with the development of a chemical reactor model to represent the mixing and decay of wastewater constituents over time in the treatment wetland. The development of the chemical reactor model is described in the following section.

3.7.1. Modified Tanks-In-Series

The TIS non-ideal chemical reactor model was selected to represent the hydraulic behavior and treatment reactions in the wetland. The conventional TIS reactor model was modified to account for the additional non-effluent watershed contributions (*e.g.*, precipitation and snowmelt runoff) that are progressively added throughout the wetland. Within this model, the wetland is represented hydraulically as a series of completely mixed stirred tanks, each possessing an equivalent *HRT*. The general mass balance for each tank in the modified TIS model is shown as follows:

$$Q_{out}C_{out} = \left[Q_{in}C_{in} + \left(\frac{Q_{ws}}{N}\right)C^*\right] - \frac{k\tau Q_{out}}{Nd}(C_{out} - C^*) \qquad [3.29]$$

where:

Q_{in}= flow into tank '*N*' (m^3/d),

C_{in} = concentration into tank '*N*' (kg/m^3),

Q_{out} = flow out of tank '*N*' (m^3/d),

C_{out} = concentration out of tank '*N*' (kg/m^3),

Q_{ws} = non-effluent watershed flow into wetland segment (m^3/d),

C^* = background concentration (kg/m^3),

N = number of tanks (dimensionless),

k = areal rate coefficient (m/d),

τ = actual hydraulic residence time (d), and

d = average wetland depth (m).

The mass balance assumes evapotranspiration, precipitation, and infiltration to be negligible over the studied segments of the wetland. This assumption was suitable for the wetland during the studied periods due to the short residence times of surface flow in the wetland observed on the order of hours to just over a day.

Conceptually, the watershed additions are cumulatively added in equal amounts to each TIS. The conceptual basis for the modified TIS model is illustrated in Figure 3.6. Within the model, the wastewater effluent and watershed contribute a mass addition of a particular contaminant into the first TIS. The addition of the mass is represented by a flow multiplied by a concentration. The mass of the contaminant is then removed by decay or settling reactions that are limited by the *HRT* of each tank. The outgoing concentration of a wastewater contaminant is then further reduced by dilution from watershed additions and the decay reactions in each subsequent TIS.

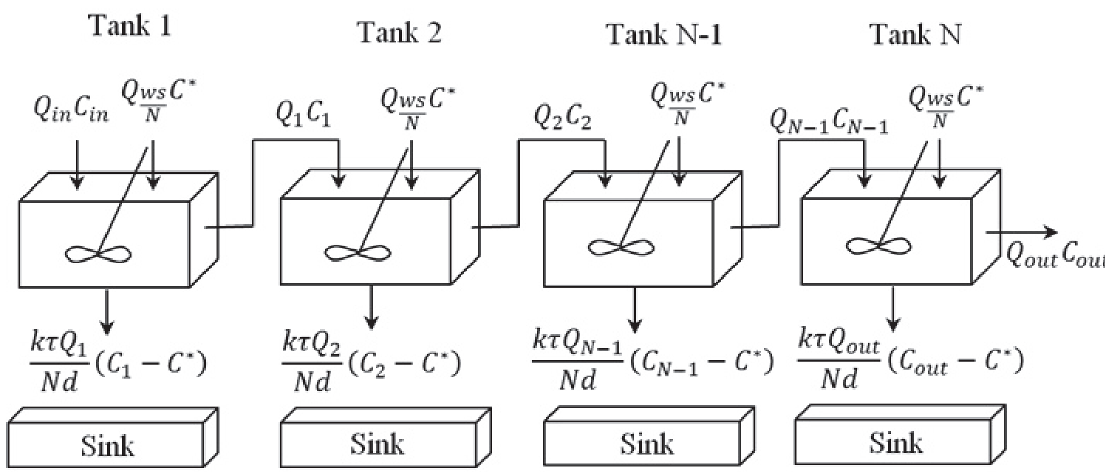

Figure 3.6 Schematic representation of the modified TIS model for "*N*" number of tanks.

The mass balance was re-arranged to solve for the contaminant concentration leaving, C_{out} (kg/m^3), each modeled segment of the wetland as shown with the following equation:

$$C_{out} = \frac{\left(\frac{Q_{in}}{Q_{out}}\right) C_{in} + \left(\frac{\frac{Q_{ws}}{N}}{Q_{out}}\right) C^* + \frac{k\tau C^*}{Nd}}{1 + \frac{k\tau}{Nd}}. \qquad [3.30]$$

Modified TIS models were developed for the main segments of the wetland where surface water flow was observed periodically during the site investigations as shown with WS-1, WS-2, and WS-3 in Figure 3.5.

3.7.2. Compartmentalization

The determination of the appropriate number of tanks is termed the compartmentalization of the WS. The number of tanks (*N*) that best represented the hydraulics of each segment of the wetland was determined using the measured RTDs. This was done by fitting a gamma distribution of detention times to the RTDs developed from the field data following a methodology proposed by Kadlec and Wallace (2009). The gamma distribution of detention times was represented by:

$$g(t) = \frac{N}{\tau\Gamma(N)}\left(\frac{Nt}{\tau}\right)^{N-1} exp\left(-\frac{Nt}{\tau}\right) \qquad [3.31]$$

where:

$\Gamma(N)$ = gamma function of *N,* = *(N-1)!,* factorial, if *N* is an integer (min^{-1}), and

t = time (min).

The gamma function is represented mathematically as (Kadlec and Wallace, 2009):

$$\Gamma(N) = \int_0^\infty exp(-t)t^{N-1}dt. \qquad [3.32]$$

The gamma distribution function given in Eq. 3.31 was developed for each wetland tracer test using the GAMMADIST function in Microsoft® Excel™. The SOLVER tool in Microsoft® Excel™ was used to fit the gamma distribution function (Eq. 3.31) to each tracer dataset by selecting the parameters *N* and to minimize the sum of squared errors (SSQE) between the gamma distribution function and the RTD from the field data. The *N* value obtained from the gamma fit to the RTD was rounded up to the nearest integer.

3.7.3. Parameterization

The modified TIS treatment model was parameterized for each WS where a surface water tracer test was conducted. Model parameterization consisted of the following input parameters determined from field data:

- WS inflow (Q_{in}),
- non-effluent watershed discharge (Q_{ws}),
- WS outflow (Q_{out}),
- influent treatment performance parameter concentration (C_{in}),
- average depth of the wetland segment (d),
- HRT from tracer tests (τ),
- background parameter concentration (C^*), and
- number of tanks from the gamma model fit (N).

The discharge data used to parameterize the TIS models is summarized in Table 3.5. The treatment performance parameter influent and effluent concentrations for each modeled WS are specified in Table 3.6. Only one set of treatment performance sample results were used in the WS-3 model due to unsteady state hydraulic conditions in the wetland during the site visit in July 2012.

3.7.4. Calibration – Rate Coefficient Determination

The areal decay rate coefficient (k-value) was determined for a suite of treatment performance parameters for each WS where a tracer test was conducted. When multiple data sets of flow, depth, and treatment performance data were collected during periods of similar hydraulic conditions, average k-values were calculated for each set of observed field conditions. Areal first order rate coefficients were used consistently throughout the modeling.

Outgoing treatment performance parameter concentrations for each of the respective tanks of the model was calculated using Eq. 3.30 and Microsoft® Excel™. The model was calibrated by setting the outgoing effluent concentration (C_{out}) to equal the field data concentration by optimizing a universal k-value for each treatment performance parameter. The optimization of the set of mass balance equations was performed using the

SOLVER function in Microsoft® Excel™. The requirement of this methodology is that the initial k-value guess must be near the true optimized value, as the SOLVER function ceases iteration at the closest solution to the initial guess (*i.e.*, it can select a local minimum). A poorly selected initial guess would result in a k-value not representative of the global optimized solution. In cases where modeled outgoing effluent concentration was lower than the measured effluent concentration, with a k-value of zero, it was not possible to differentiate between reduction of concentrations by dilution and those explained by treatment processes. For those indistinguishable cases, the reduction in concentrations observed could be explained by dilution alone before factoring in treatment potential.

3.7.5. Temperature Correction

The decay rate coefficients for parameters that are affected by temperature were normalized to 20°C using the Arrhenius equation shown as follows (Kadlec and Wallace, 2009):

$$k_T = k_{20}\theta^{(T-20)} \tag{3.33}$$

where:

k_t = rate coefficient at the field temperature (m/d),

k_{20} = rate coefficient normalized to 20°C (m/d),

T = field temperature in ºC, and

θ = temperature correction factor (dimensionless).

Various temperature correction factors were used for parameters with temperature sensitive decay rate coefficients. The correction factors used during this study are summarized in Table 3.4. As no long-term water quality monitoring data exists for the site, the temperature correction factors were selected from literature. The justification for the selection of particular temperature correction factors is also provided in Table 3.4. The decay rate coefficients for TSS and VSS were not temperature corrected as it was indicated that the decay reactions for these parameters are not directly related to temperature (Kadlec and Wallace, 2009).

Table 3.4 Temperature correction coefficients selected for decay rate coefficient normalization to 20°C.

Parameter	**Temperature correction factor (*Θ*)**	**Justification for selection**	**Reference**
$CBOD_5$	1.012	Average of three overland flow systems.	Kadlec and Reddy, 2001
E. coli	1.070	Assumption of doubling the rate of bacterial loss for a 10°C temperature rise.	Chapra, 1997; Boutilier *et al*., 2009
Total Nitrogen	1.030	Average of four FWS wetlands with average annual operating water temperatures of 8°C and average total nitrogen loading of 234 $g/m^2 \cdot yr$	Kadlec and Wallace, 2009
Total Ammonia Nitrogen	1.053	Average of four FWS wetlands with average annual operating water temperatures of 7.5°C and average ammonia loading of 178 $g/m^2 \cdot yr$.	Kadlec and Wallace, 2009
Total Phosphorus	0.991	Average of ten FWS wetlands with average operating watertemperatures of 1.7 to 21°C and average total phosphorus loading of 4 $g/m^2 \cdot yr$.	Kadlec and Wallace, 2009; Kadlec and Reddy, 2001

Table 3.5 Discharge data used in model parameterization.

Wetland segment	**WS-1**		**WS-2**		**WS-3**
Date sampled (dd/m/yyyy)	21/6/2011	25/6/2011	21/6/2011	25/6/2011	7/7/2012
Q_{in} (m^3/d)	789	403	801	498	21
Q_{ws} (m^3/d)	12	95	442	274	102
Q_{out} (m^3/d)	801	498	1243	772	123

Table 3.6 Treatment performance parameter concentrations used in model parameterization.

Wetland segment	**WS-1**				**WS-2**				**WS-3**	
Date sampled (dd/m/yyyy)	21/6/2011		25/6/2011		21/6/2011		25/6/2011		7/7/2012	
Treatment parameter	C_{in}	C_{out}	C_{in}	C_{out}	C_{in}	C_{out}	C_{in}	C_{out}	C_{in}	C_{out}
$CBOD_5$ (mg/L)	72	23	127	33	23	19	33	21	89	14
E. coli (MPN/100mL)	7.7×10^5	1.7×10^5	2.0×10^6	2.4×10^5	1.7×10^5	1.9×10^4	2.4×10^5	5.4×10^3	3.6×10^3	6.6×10^1
TSS (mg/L)	33	26	29	24	26	18	24	17	106	4
VSS (mg/L)	29	23	26	19	23	16	19	11	103	3
TN (mg/L)	*	*	47	31	*	*	31	23	34.0	4.0
TAN (mg/L)	35	21	55	37	21	20	*	*	22	2
TP (mg/L)	*	*	4	3	*	*	3	3	3	1

*Data not collected on these dates.

Chapter 4: Results and Discussion

The following chapter presents the results from the wetland treatment performance assessment and modeling outcomes in conjunction with a discussion of the findings.

4.1. Physical Characterization

The characterization of the physical aspects of the wetland treatment area consisted of a survey of the vegetation and topography. The results of the vegetation survey are summarized in the following section. The topographic survey results were used to facilitate the watershed delineation, the determination of the sample point positions, the mapping of effluent flow paths, and determination of the observation well elevations; therefore, the topographic survey results are included as elements in subsequent sections.

4.1.1. Vegetation

The spatial distribution of dominant vegetation and land cover from the supervised maximum likelihood classification in the wetland treatment area is illustrated in Figure 4.1. The main classes of vegetation and land cover are separated into eight distinct classes of which six are illustrated in Figure 4.1. Photographic examples of the main classes of vegetation and land cover are shown in Appendix A. Generally, land in the vicinity of the effluent flow areas is characterized by wetland species (Classes 1, 2, 5, and 6), the land around the perimeter of the flow areas is mainly a transition area of diverse wetland and upland species (Class 3), while the remaining land is classified as upland which is associated with higher ground and bedrock outcrops (Class 4). The sample points show the locations where 108 quadrats were performed in the wetland treatment area.

The effluent flow areas for both the spring and post-spring (mid-summer and fall) hydrological conditions are overlayed in Figure 4.1. There are two basic differences in the dominant vegetation between the spring and post-spring flow areas. The first difference is the dominance of *Salix richardsonii* in the spring flow area in contrast to the sparseness of *Salix richardsonii* in the post-spring flow area, which is characterized more predominantly by *Bryophyta* spp. and *Hippuris vulgaris.* The difference in dominant vegetation classes between the seasonal flow areas could suggest that the difference in effluent water quality observed in the wetland between spring and post-spring conditions may have an impact on the vegetation dominance within the wetland flow areas. Additionally, the *Salix richardsonii* was noted to be on average 35 cm in height in the spring flow are of the wetland in comparison to a 21 cm average height in the reference wetland. This may suggest that nutrient loading from the WSP effluent enhances the growth and prominence of *Salix richardsonii.* Tilton and Kadlec (1979) noted an increase in plant biomass near the inlet of a natural treatment wetland receiving municipal wastewater. Interestingly, the use of *Salix* spp. (willow) in constructed wetlands has been successful in Denmark and Sweden for phytoremediation of nutrients and metals in domestic wastewater (Gregersen and Brix, 2001; Wittgren and Mæhlum, 1997).

The second observed difference was the spatial distribution of organic detritus between the spring and post-spring flow areas. The spring flow area shows a spatial distribution of organic detritus along the entire length of the flow path from the inlet to the outlet. In contrast, only the upper portion of the post-spring flow path, near the inlet, is dominated by organic detritus from effluent solids. This difference is explained by the shorter *HRT* during spring conditions, which resulted in the settling of solids along the entire length of the flow path. This contrasts to the post-spring conditions, where most of the settling of solids occured near the inlet due to the lower *HLR* and longer *HRT*s. Yates *et al.* (2012) similarly noted depositions of organic matter in many other Nunavut natural wetland systems.

A detailed summary of the dominant vegetation and land cover classes, along with secondary vegetation and land cover types that were observed from the vegetation survey quadrats, are summarized in Table 4.1. The probable land type designation is stated for each of the eight main classes of vegetation or land cover and, the percent coverage observed in the 14.2 ha wetland treatment area is provided. In total, it is estimated that 54% of the 14.2 ha treatment area was characterized by vegetation and land cover types indicative of wetland environments (Classes 1, 2, 5, and 6). Approximately 17% of the wetland treatment area was likely composed of vegetation characteristic of the transition area between wetland and upland environments (Class 3). An estimated 23% of the wetland treatment area was classified by vegetation and land cover indicative of upland environments (Class 4). The remaining 6% of the wetland treatment area was characterized by surface water in ponds and stream channels, and to a lesser degree, sand from disturbed soil resulting from previous earthworks on-site (Classes 7 and 8). Overall, the diversity of vegetation species was greatest in the Class 3 wetland/upland transition land type where *Carex* spp. was the most dominant species. The least amount of diversity was observed in Class 2 which was mainly characterized by organic detritus. Interestingly, the Class 2 areas were distributed predominantly around the wetland treatment area inlet and along the spring flow area. This is in contrast to Class 3 areas, which were mostly distributed around the lower half the post-spring flow area. The difference in the spatial distribution of classes may suggest that areas where there is relatively higher strength wastewater discharging observe a decrease in vegetation diversity. Other authors similarly noted a decrease in vegetation diversity in natural wetlands in response to wastewater addition (Kadlec, 1987; Mudroch and Capobianco, 1979). Similarly, changes in the community structure of the tundra vegetation in response to nutrient addition have been noted by Yates *et al.* (2012) and Gough *et al.* (2002).

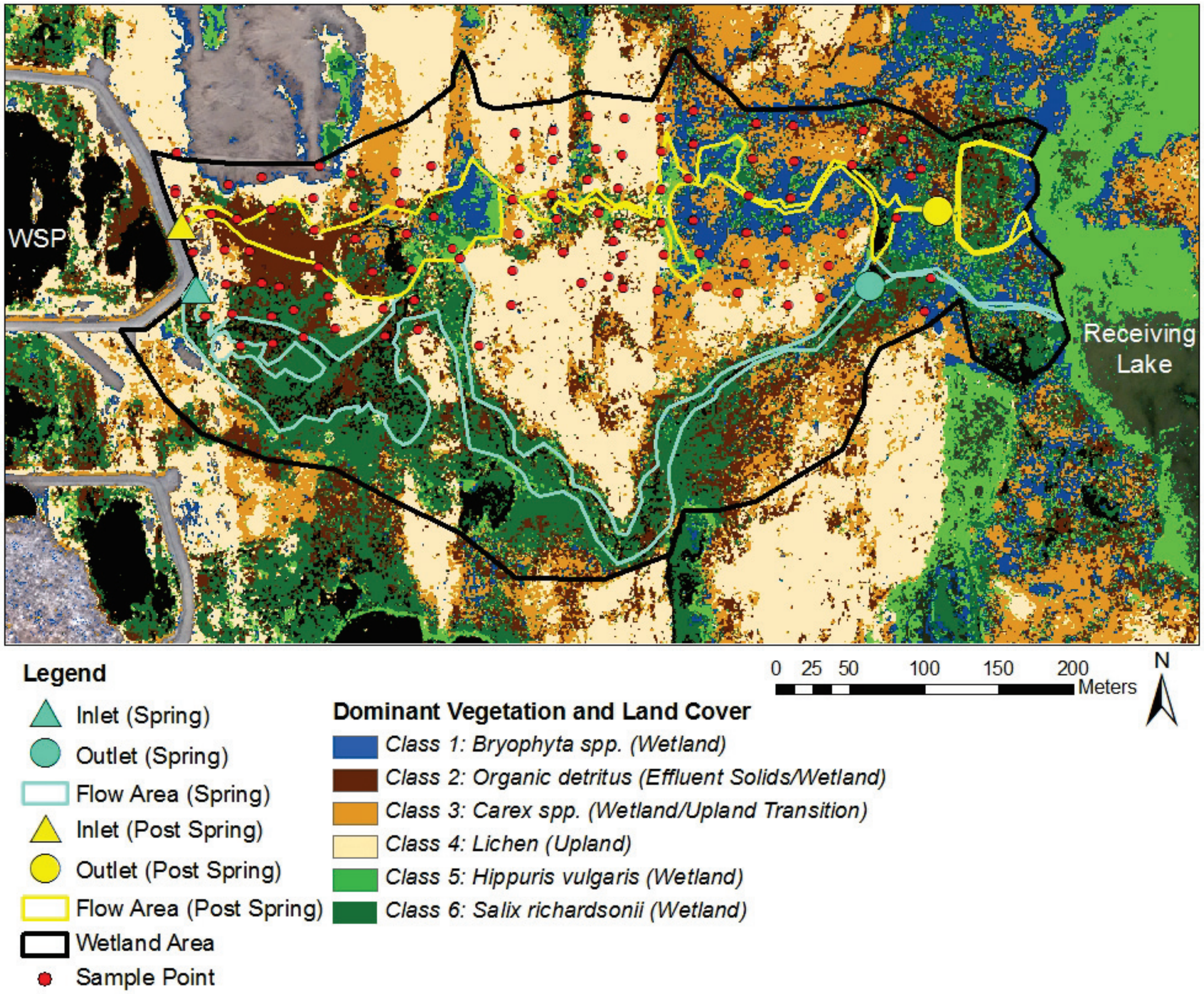

Figure 4.1 Spatial distribution of vegetation and land cover in the wetland treatment area and surrounding area obtained from the classification. The map shows six main groups of dominant vegetation or land cover types that define the wetland treatment area. The sample points indicate where quadrats were used to assess the vegetation and land cover in the field.

Table 4.1 Summary of dominant and secondary vegetation types and land cover classes with probable land type designation and percent coverage of the wetland treatment area.

Class	Dominant Vegetation or Land Cover (Classification)[a]	Secondary vegetation or Land Cover (Classification)	Probable land type designation	Cover
1	*Bryophyta* spp.	*Carex* spp. *Salix richardsonii* (FAC)	Wetland	11 %
2	Organic detritus	*Salix reticulata* (FACU, FACW) *Bryophyta* spp.	Effluent solids deposition/wetland	17 %
3	*Carex* spp.	*Bryophyta* spp. *Salix reticulata* (FACU, FACW) *Dryas octopetala* *Salix richardsonii* (FACU, FACW) *Calluna vulgaris* *Vaccinium uliginosum* *Saxifraga oppositifolia* (FACU, FAC) *Chamerion latifolium*	Wetland/upland transition	17 %
4	Lichen	*Carex* spp. *Calluna vulgaris* *Dryas octopetala* *Vaccinium uliginosum* *Bedrock outcrop*	Upland	23 %
5	*Hippuris vulgaris* (OBL)	*Ranunculus hyperboreus* *Shallow ponded water*	Wetland	6 %
6	*Salix richardsonii* (FAC)	*Organic detritus* *Carex* spp. *Salix arctica* (FAC, OBL)	Wetland	21 %
7	Water		Stream Channel or Pond	5 %
8	Sand		Disturbed Soil	1 %

[a]Vegetation classification scheme is as follows (Reed[1], 1988):
OBL = obligate wetland (>99% of occurrence in wetlands),
FACW = facultative wetland (67 – 99% probable to occur in wetlands),
FAC = facultative (34 – 66% probable to occur in wetlands),
FACU = facultative upland (67 – 99% probably to occur in non-wetlands), and
UPL = obligate upland (>99% of occurrence in uplands).

4.2. Hydraulic Characterization

The hydraulic characterization of the wetland treatment area is described within this section focusing on the hydrology and hydrogeology.

4.2.1. *Hydrology*

The hydrological characteristics of the wetland treatment area including the watershed delineation, influent and effluent discharge rates, areal hydraulic and BOD loading rates, and the water budget are described and discussed in the following sections.

4.2.1.1. Watershed Delineation

The watershed delineation was performed to determine the total watershed area that contributes water to the wetland outlet at the receiving lake. Additionally, the sub-basin watershed areas of the wetland treatment area were delineated to characterize the non-effluent contributions to the wetland treatment area. The total watershed and sub-basin watershed delineations are shown in Figure 3.4. During the spring freshet, the capacity for dilution of the effluent from non-effluent contributions was reduced due to a relatively small wetland watershed sub-basin area (7.2 ha) contributing water to the effluent stream. The wetland-to-catchment ratio during the spring freshet was 1 to 5. However, some of the decreased dilutive capacity was overcome by non-effluent inputs from the WSP and wetland treatment area resulting from thawing snow and ice that had accumulated over the winter season. The wetland watershed sub-basin was larger during the post-spring freshet conditions (22.4 ha). Correspondingly, the wetland-to-catchment ratio in the post-freshet conditions was much greater at 1 to 21. Due to this difference, there is increased potential during the post-spring freshet periods for dilution of the effluent stream from watershed contributions.

4.2.1.2. Influent and Effluent Discharge

Wetland influent and effluent discharge rates during all study periods are summarized as a time series plot in Figure 4.2. The daily discharge amounts in Figure 4.2 were calculated from instantaneous flow measurements obtained from streamgauging. Therefore, the measured values do not account for flow variability on a finer than daily timescale. Of note is the level of temporal variability in inlet and outlet discharge rates over the treatment season. Standard deviations for inlet and outlet discharges are 372 and 480 m^3/d respectively. This variability is important for three main reasons: i) the potential to dilute the wastewater constituents is dependent on the variable ratio of inlet to outlet discharges; ii) discharge variations cause fluctuations in the *HRT* of the wetland treatment area; and iii) as a result, the water quality in the wetland treatment area can fluctuate in accordance with the discharge variations.

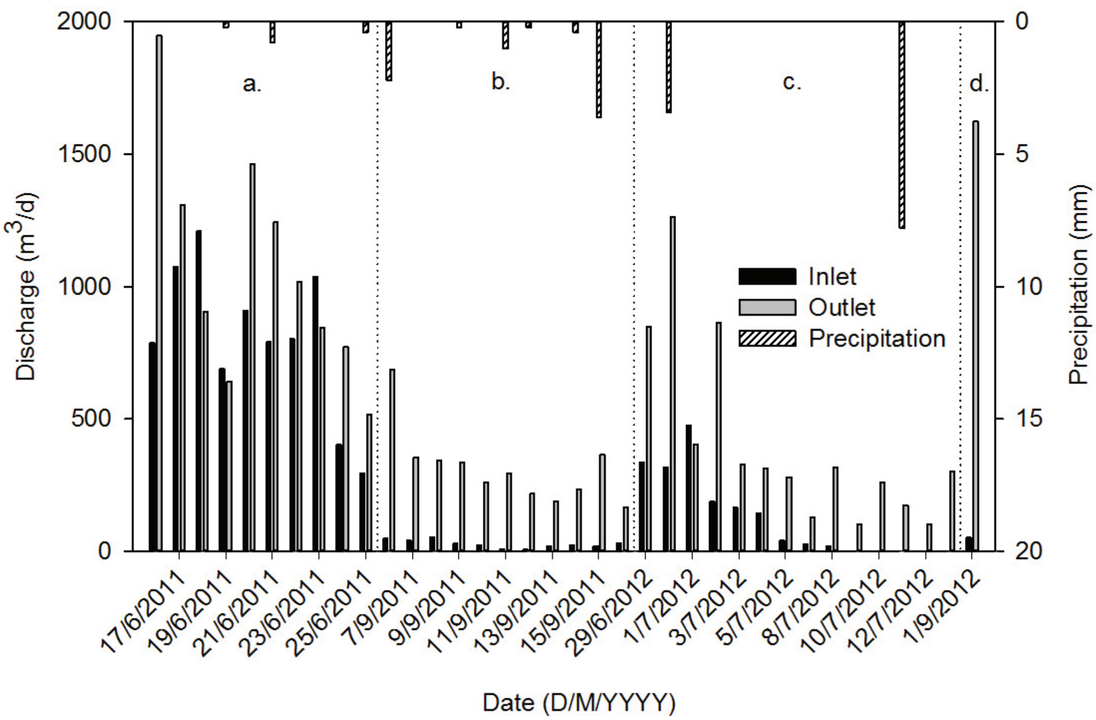

Figure 4.2 Daily influent and effluent discharges from the wetland inlet and outlet calculated from instantaneous flow measurements with precipitation. Section a. represents discharges in June 2011, b. is September 2011, c. is June and July 2012, and d. is September 2012.

As a result of the temporal variation observed in the influent and effleunt discharges, the treatment performance was not consistent. The influent and effluent discharges during the spring freshet in June 2011 averaged 800 and 1066 m^3/d respectively. The influent and effluent discharges were highest during the spring freshet, resulting from snow melt runoff and release of stored thawed wastewater from the WSP. Wetland treatment area discharge rates gradually declined as the summer progressed. In July 2012, the average influent and effluent discharges were respectively, 118 and 487 m^3/d. At the end of treatment season in September 2011, influent and effluent average discharges were comparatively lower at 27 and 313 m^3/d respectively. The influent discharged into the wetland treatment area from the WSP was greatest in June 2011, when the water level of the pond was highest, as a result of the snow and wastewater accumulation over the winter.

The rate of influent discharge to the wetland treatment area was related to the WSP water level. As the WSP water level lowered, the discharge out of the berm and into the wetland treatment area decreased. The flow path of the influent after discharge into the wetland treatment area was governed by the bedrock topography at the south-eastern toe of the berm (wetland inlet). When the water level of the influent at the toe of the berm was high, such as in June 2011, the influent overcame the topographic barriers at the toe of the berm to flow in a south-eastern direction. In contrast, at low pond water level conditions, such as during the post-spring freshet, the topographic barriers at the toe of the berm restricted flow to an exclusively eastward direction. The difference in wetland treatment area inlet locations over the treatment season is illustrated in Figure 3.3. Figure 4.3 illustrates the difference observed in WSP water levels between the spring freshet and post-spring freshet conditions. The datum used as a benchmark for WSP water level measurement was an inoperable decant pipe located in the eastern WSP berm. The WSP water level was transient during the spring freshet and mid-treatment season in June 2011 and July 2012; however, in September 2011, the pond water level remained relatively constant.

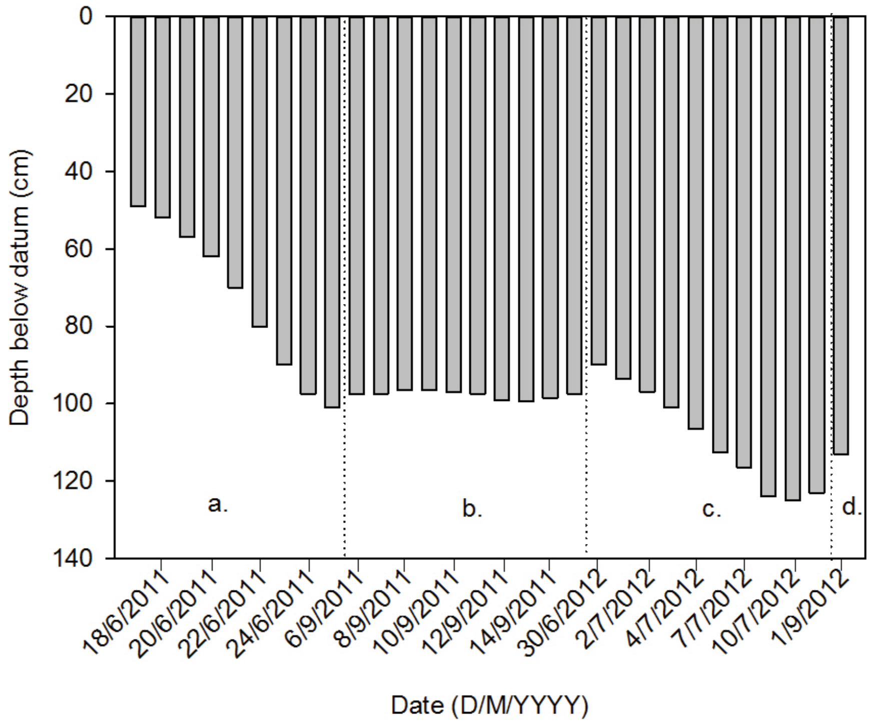

Figure 4.3 The WSP level shown as a depth below a datum. Section a. represents water levels in June 2011, b. is September 2011, c. is June and July 2012, and d. is September 2012.

The decrease in water levels was attributed mainly to pond discharge into the wetland treatment area and not evaporation. This was demonstrated with the pond evaporation rate estimates summarized in Table 4.2. The pond evaporation rate estimate was much smaller in comparison to the water level change in the pond. This suggests that by late-treatment season the pond water level was such that wastewater addition to the pond from the truck discharges and precipitation events acted only to maintain a sporadic small influent discharge into the wetland treatment area.

Table 4.2 Estimated pond evaporation rates for the study periods.

Study Period	**Estimated pond evaporation (mm/d)**
June 2011	2.0
September 2011	1.1
July 2012	2.5
August 2012	1.3

Figures 4.4 and 4.5 represent the wetland treatment area influent and effluent hydrographs developed for mid- to late summer 2012 conditions. Figure 4.4 shows the decline in influent discharge into the wetland treatment area, as the system transitioned

from spring freshet to post-spring freshet conditions. Generally, the influent discharges ranged from 0 m^3/d to 163 m^3/d. The influent discharges were variable over the period monitored with a standard deviation of 19 m^3/d. There were notable instances when the influent discharge from the WSP was either at or close to zero. The wetland treatment area influent discharge increased in response to precipitation events. Figure 4.5 represents the wetland outlet effluent hydrograph for the mid- to late summer 2012 conditions. The wetland effluent discharge ranged from 130 m^3/d to 927 m^3/d, with a standard deviation of 132 m^3/d. The wetland effluent discharge fluctuated in response to precipitation events. The main distinction between the wetland influent and effluent discharges was the difference in the discharge magnitudes as a result of non-effluent watershed additions within the wetland treatment area. For instance, the 45 mm precipitation event on August 26, 2012, produced an inlet discharge response of 26 m^3/d from near zero inlet discharge conditions (Figure 4.4). In comparison, the same precipitation event produced a 927 m^3/d wetland treatment area outlet discharge response (Figure 4.5). This is explained by the difference in the catchment area of the WSP of 2.9 ha (29 273 m^2) compared to the 22.4 ha wetland treatment area watershed sub-basin outlined in Figure 3.4.

4.2.1.3. Areal Hydraulic Loading Rates

The daily *HLR*s were calculated using two different approaches to demonstrate the difference in resulting *HLR* values in relation to the effective area over which treatment is assumed to occur: i) using the 22 ha area of the wetland watershed sub-basins; and ii) using the 1.1 to 1.5 ha wetted areas where effluent was observed to be flowing via ground-truthing. Areas of the wetland treatment area where effluent discharge varied seasonally, which accounted for the range of wetted areas stated. The 1.5 ha (14 620 m^2) wetland segments WS-1 and WS-2 are representative of where effluent was observed to discharge during spring freshet in June 2011. The 1.1 ha (10 646 m^2) area represents wetland segments WS-3 and WS-4 where effluent was observed to discharge during the post-spring freshet.

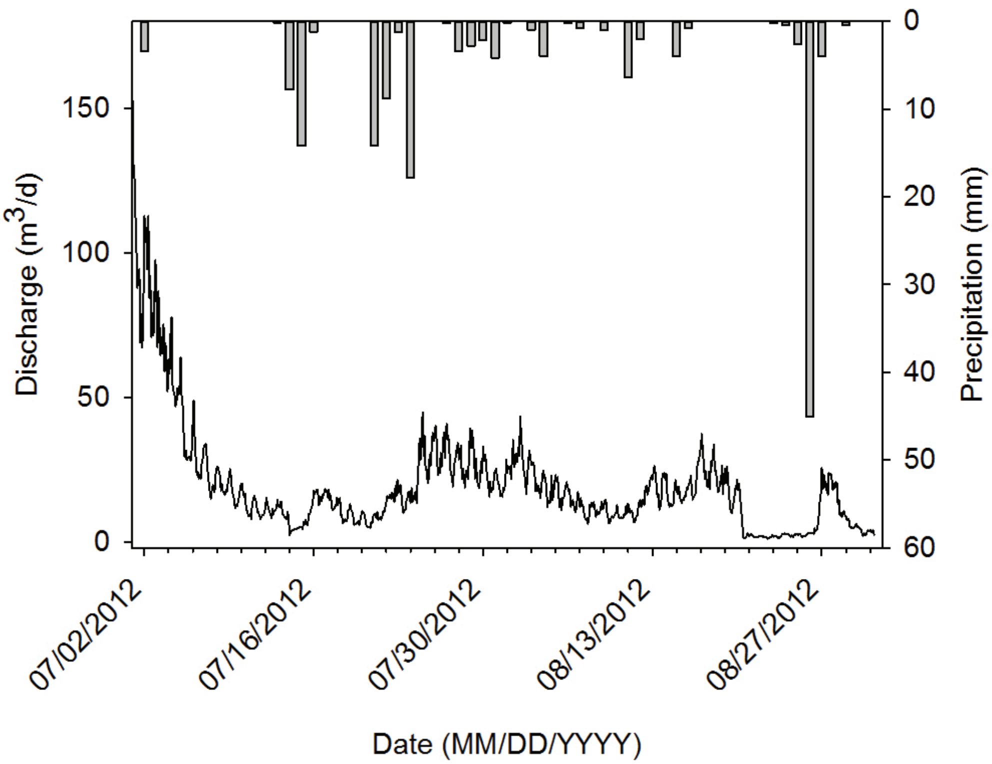

Figure 4.4 Wetland influent hydrograph with precipitation from July to September 2012.

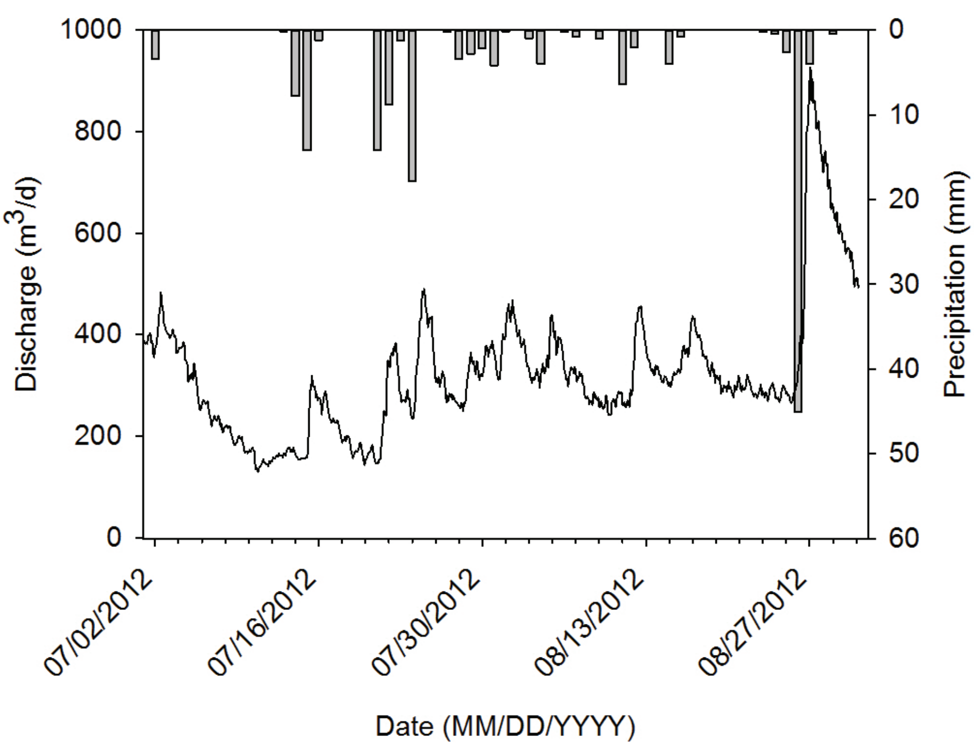

Figure 4.5 Wetland effluent hydrograph with precipitation from July to September 2012.

The effluent *HLR*s from the WSP onto the 22 ha wetland watershed sub-basin during the study periods are shown in Figure 4.6. The effluent *HLR* from the WSP discharging onto the wetland watershed sub-basin ranged from a minimum of 0 cm/d at times during the post-spring freshet to a maximum of 0.54 cm/d (54 m^3/ha·d) during the spring freshet. An average *HLR* of 0.06 cm/d (6 m^3/ha·d) was observed across the study periods. The *HLR* of the effluent from the WSP onto the 1.1 – 1.5 ha wetted area of the wetland treatment area is shown for the periods studied in Figure 4.7; the *HLR*s range from 0 to 8.3 cm/d (0 to 827 m^3/ha·d), with an average *HLR* of 1.0 cm/d (97 m^3/ha·d). For context, typical FWS design criteria values range from 2.5 to 12.5 cm/d (Water Environment Federation, 2010); which is well above the *HLR*s tabulated using the entire wetland watershed sub-basin area. The *HLR*s determined using the wetted areas of the wetland treatment area fall approximately in the middle of the reported literature values, which suggests that the use of the wetted area for *HLR* determination produces more representative *HLR*s for the wetland treatment area. *HLR*s at other northern Canadian natural wetland sites were estimated at 43 m^3/ha·d (7 ha wetland in Teslin, Yukon); 63 m^3/ ha·d (32 ha wetland in Hay River, NWT); and 125 m^3/ ha·d (6 ha wetland in Haines Junction, Yukon) (Doku and Heinke, 1995 and 1993). Although Coral Harbour had a maximum *HLR* much greater than the literature reported values, the *HLR* was not sustained.

The difference in *HLR*s between the wetted treatment area and the entire wetland watershed sub-basin reveals that the definition of the area where effluent is distributed within the wetland treatment area is important for determination of the most representative *HLR*s. For a natural wetland system, such as in Coral Harbour, the determination of the actual wetted area receiving wastewater can be complicated due to seasonal variations in the spatial extent and direction of influent discharge. The only way to ascertain the actual wetted area, which is extremely critical information for wetland design, is to conduct tracer studies or very detailed site-specific topographic and hydrological assessments. Additionally, the variability in the *HLR*s can contribute to uncertainty in the treatment performance expectations of the wetland, particularly for wetlands receiving water from non-point sources with variable flows and water quality (Andersson *et al.*, 2005).

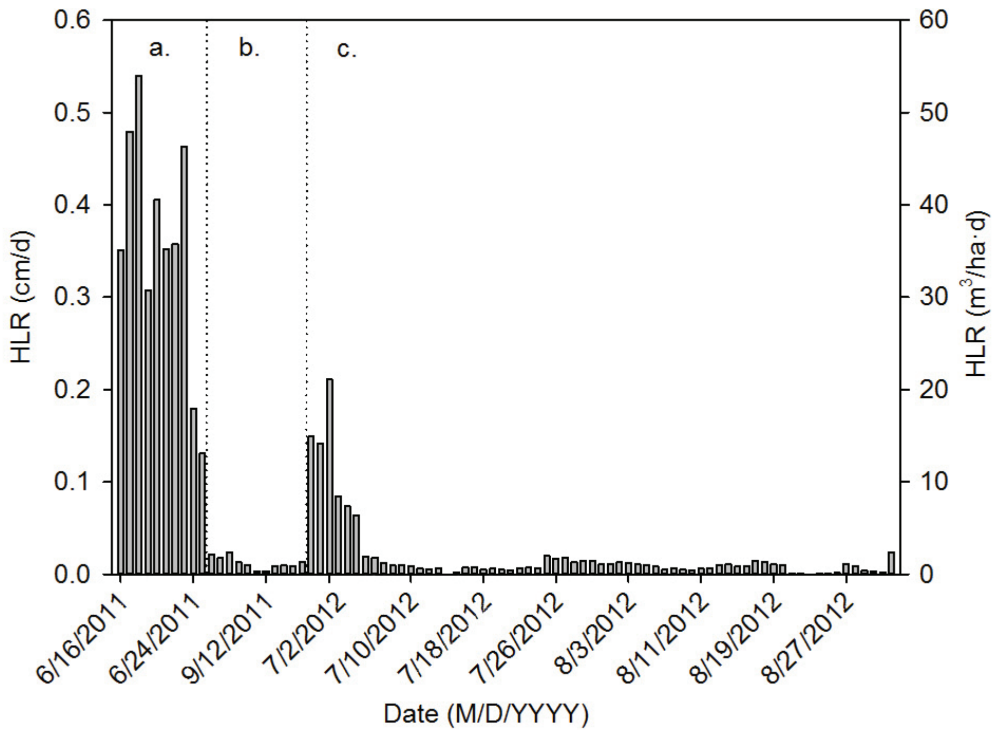

Figure 4.6 *HLR* of effluent from the WSP on the wetland watershed sub-basin area (22 ha). Section a. denotes spring freshet *HLR* in June 2011, b. is September 2011 and c. is the *HLR* from July to September 2012.

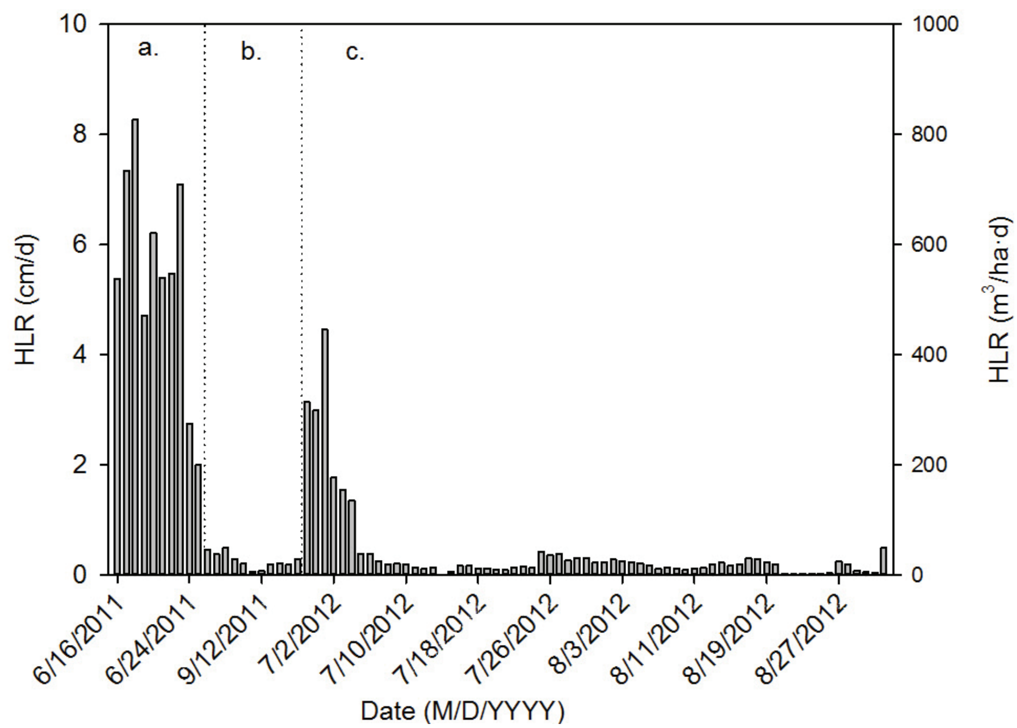

Figure 4.7 *HLR* of effluent from the WSP on the wetted area of the wetland treatment area. Section a. denotes spring freshet *HLR* in June 2011, b. is September 2011 and c. is the *HLR* from July to September 2012.

4.2.1.4. Water Budget

The water budget gives an indication of the relative amount of water added to the wetland treatment area from precipitation and from WSP effluent, as well as, the water loss from the system via *PET* and outflow at the wetland at the outlet. When the additions to the wetland treatment area are less than the system losses, the wetland treatment area is losing stored water which may come from a variety of sources including active layer seepage melt, groundwater, and snow and/or ice melt. Wittgren and Mæhlum (1997) note that in a cold climate, the snow influences the water budget causing flooding in the spring. The severity of the spring freshet flooding is described as being dependent on the ratio of wetland-to-catchment area.

The water budgets estimated for the wetland treatment area watershed sub-basin during the study periods are illustrated in Figures 4.8 and 4.9. The spring freshet water budget estimated in 2011, denoted by section a. of Figure 4.8, suggests that the volume of influent flowing into the wetland treatment area accounts for on average 75% of outflowing effluent from the wetland treatment area. During that time water additions are less than losses, therefore stored water was released in the form of snow and ice melt, as well as, groundwater seepage. In June 2011, the total volume of influent introduced to the wetland treatment area observed was 7 992 m^3 and the total outflow volume measured at the outlet over the same period was 10 662 m^3.

The water budget estimated for the end of the treatment season in September 2011, represented by section b. of Figure 4.8, indicates that the WSP effluent volume discharged onto the wetland treatment area during that period was on average 9% of the volume of water flowing from the wetland treatment area outlet. In the fall, the difference between water added and lost was much greater than the spring, which suggests that stored water was being lost from the system. During that time, the total measured volume of influent discharging onto the wetland treatment area from the WSP was 300 m^3 compared to 3 443 m^3 of outflowing effluent measured flowing from the wetland outlet. The difference in inflow and outflow water volumes over the treatment season emphasizes the variability in the potential for dilution provided by the wetland watershed sub-basins over the course of the treatment season. The proportion of effluent volume to non-effluent volume discharging at the wetland treatment area outlet is an important consideration in outflowing water quality.

The estimated weekly water budget for July and August 2012 is shown in Figure 4.9. Figure 4.9 suggests that the volume of influent discharging onto the wetland treatment area accounted for 11% of the outflow volume at the wetland treatment area outlet. The total volume of influent discharging onto the wetland treatment area during the 2012 study period was 2 692 m^3 compared to an outflow volume at the wetland treatment area outlet of 24 138 m^3. There were instances when water additions to the wetland treatment area, due to precipitation events, were greater than water losses. In those cases, the system was storing water. Similar to conditions during the spring freshet 2011, the early July 2012 conditions were such that WSP influent volume discharging onto the wetland treatment area accounts for a greater proportion of the outflow volume at the outlet.

Overall, during most of the treatment season, the inflow volume from WSP influent was minimal in comparison with non-effluent hydrologic inputs onto the watershed from snow melt runoff, permafrost melt, and precipitation. Generally, the estimated *PET* volume during the spring freshet was 442 m^3/d on average, compared to 233 m^3/d, at the end of the treatment season in 2011. The average estimated *PET* volume in July and August 2012 was slightly higher at 494 m^3/d.

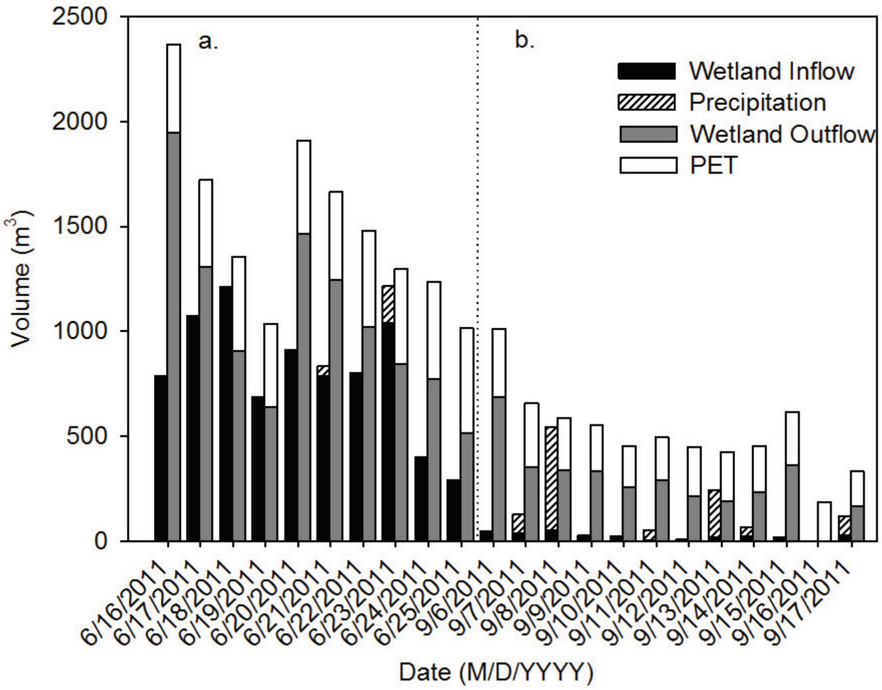

Figure 4.8 Time series of the wetland treatment area water budget for the 2011 study period for the 22 ha wetland treatment area watershed sub-basin. Section a. is representative of the spring freshet conditions in June and b. is the September estimate.

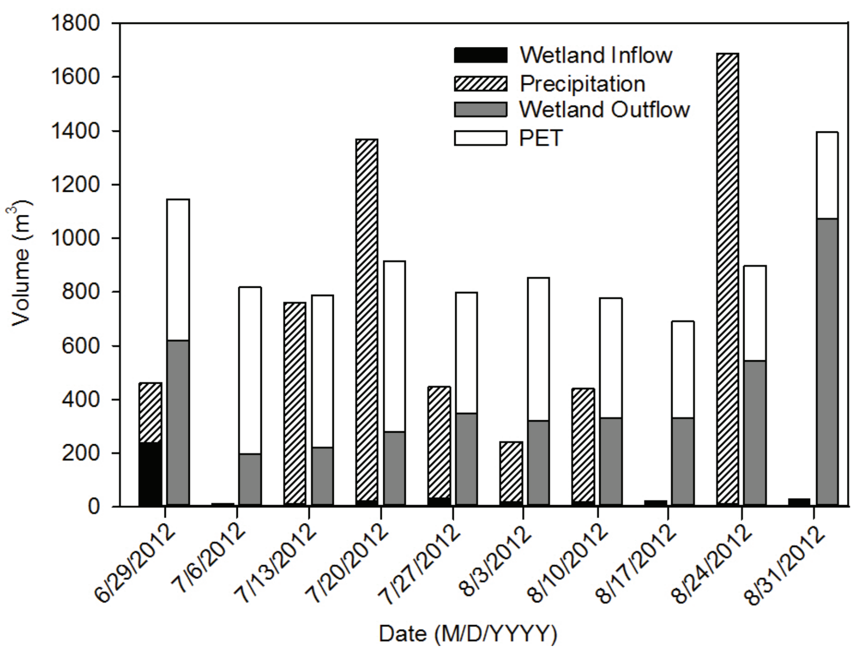

Figure 4.9 Weekly averages of the wetland treatment area water budget for 2012 study period for the 22 ha wetland watershed sub-basin.

4.2.2. Hydrogeology

The hydrogeological setting of the wetland treatment area was characterized because there were periods during the treatment season when the effluent infiltrated into the subsurface near the wetland treatment area inlet (Figure 4.10). The hydrogeological setting of the wetland treatment area is summarized in the following section with a description of the surficial and bedrock geology, the groundwater flow direction, the hydraulic gradient, and velocity of the groundwater. Subsurface flow of effluent was not consistently observed, and therefore it is not a primary treatment mechanism. However, characterization of the hydrogeology in this context is valuable to: i) develop a complete understanding of the wetland setting; and ii) evaluate the basic characteristics of a natural arctic subsurface flow wetland setting receiving wastewater for future research initiatives.

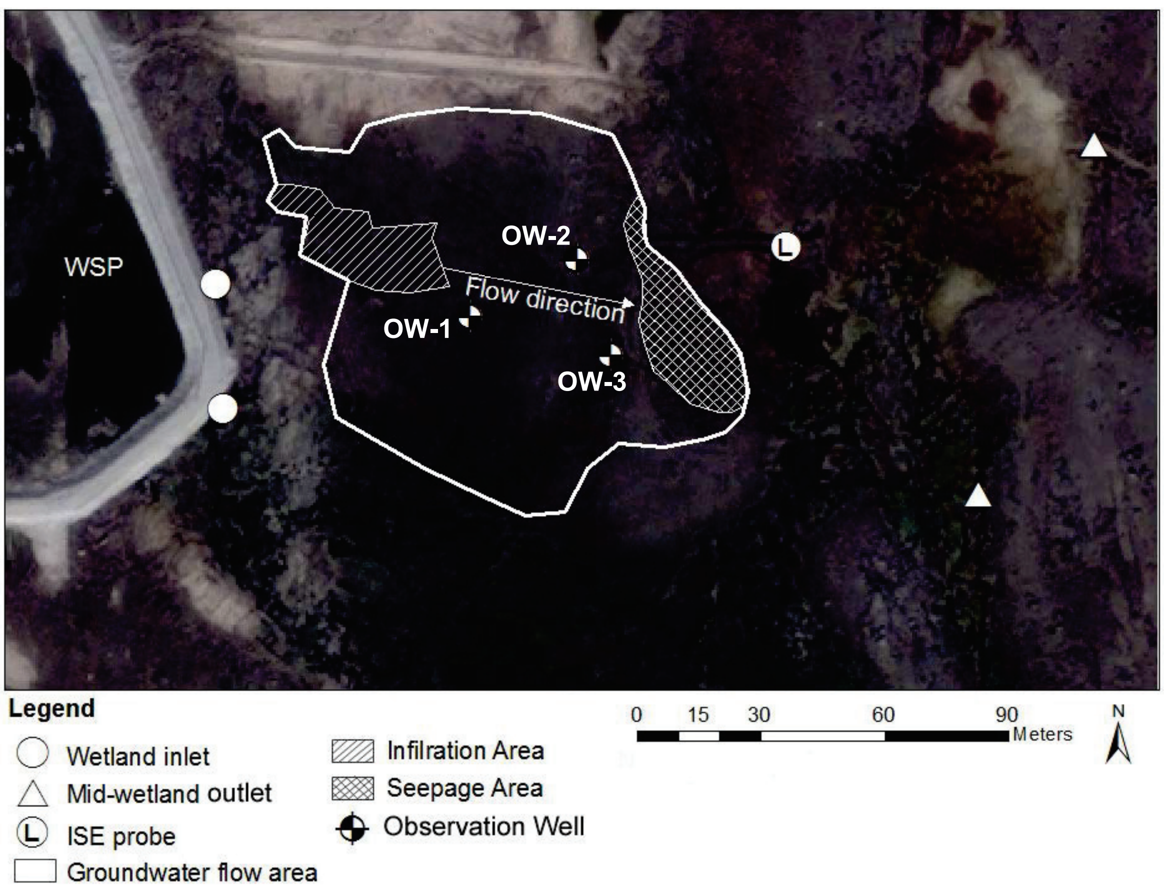

Figure 4.10 Groundwater flow area in the wetland treatment area indicating the infiltration and seepage areas, the location of the observation wells, and the Br^- ISE probe during the groundwater tracer test. QuickBird satellite imagery (July 13, 2006).

4.2.2.1. Surficial Geology

The overburden soil was a poorly sorted light brown sand with gravel, carbonates, and shells fragments. Surficial geology records classify the surficial geology in the region as glacio-marine lag that is composed of sand and gravel to cobble-sized materials mixed with marine sediments of glacial origin (Fontaine and Mallory, 2011). Test pit logs from previous earthworks on-site near the groundwater flow area indicate an upper soil layer of sand with gravel, an underlying layer of well sorted sandy gravel ranging from 0.5 to 1.0 m thickness, underlain by bedrock or permafrost (EBA Engineering Consultants Ltd., 2005). This underlying coarse gravel layer was suspected but not confirmed on-site. Figure C-1 in Appendix B shows the grain size distribution of three separate soil samples collected from 0 to 1 m in the upper layer of soil in the groundwater flow area. The K_{fs} averaged 6.7×10^{-3} cm/s (5.8 m/d) and ranged from 1.9×10^{-3} cm/s (1.7 m/d) to 1.3×10^{-2} cm/s (11 m/d). The laboratory determined hydraulic conductivity measurement was slightly higher, likely due to soil disturbance, at 4.0×10^{-2} cm/s (34 m/d).

4.2.2.2. Active Layer

Coral Harbour, NU is located in a zone of continuous permafrost. The surficial geology in the groundwater flow area of the wetland treatment area is ≥ 1 m in depth; however logs from previously dug test pits near the groundwater flow area suggest the depth to bedrock or permafrost ranges from 0.5 m to 1.5 m (EBA Engineering Consultants Ltd., 2005). Figure 4.11 shows the change in the elevation of the active layer during the July 2012 study period for two of the observation wells. As temperatures increased, the elevation of the active layer decreased in the observation wells as thawing occurred. A total decrease in active layer elevation of 15.7 cm and 13.4 cm was observed over the monitoring period in July 2012 for wells OW-3 and OW-2 respectively. The dynamic nature of the depth of active layer could suggest that the infiltrated effluent may have a transient *HRT* in the subsurface flow area as the volume of the subsurface available for groundwater flow increases with seasonal thawing.

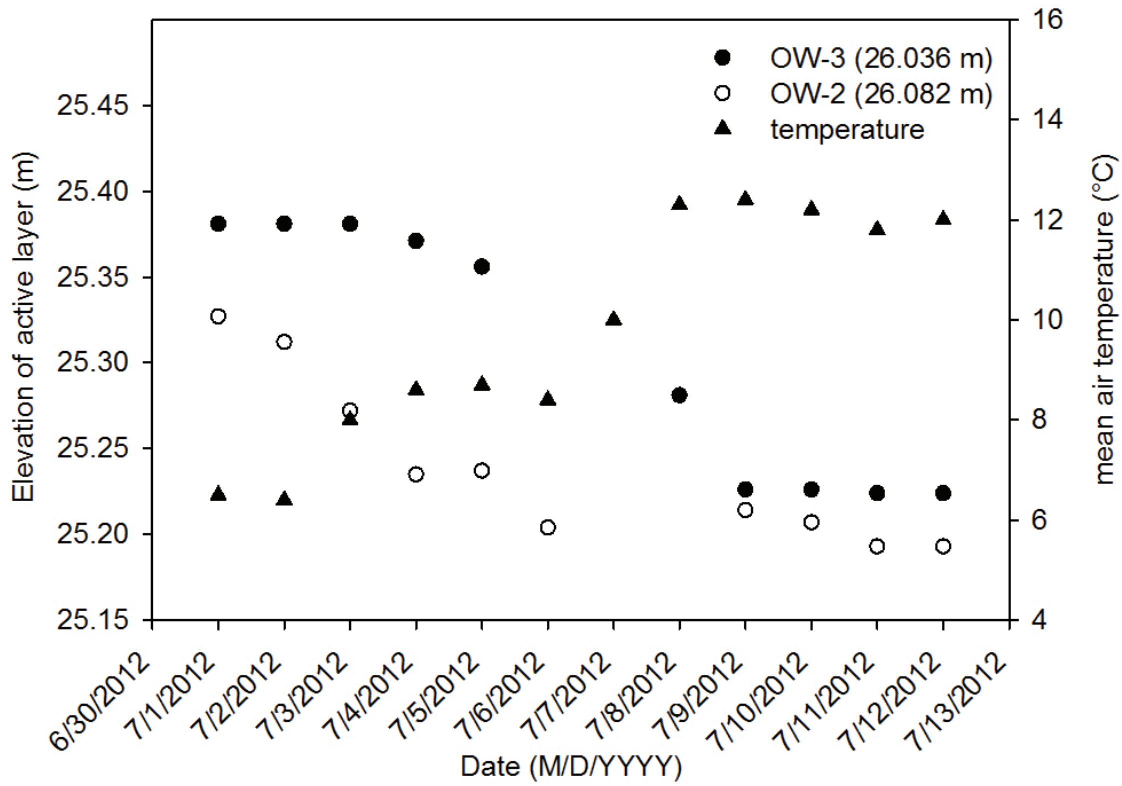

Figure 4.11 Elevation of active layer in the groundwater flow area in July 2012 for two observation wells and associated mean air temperatures.

4.2.2.3. Bedrock Geology

The on-site bedrock geology was identified as granodiorite gneiss which is a metamorphic rock type. The bedrock was exposed in the areas of the WSP and wetland treatment area. The bedrock outcrops were approximately north-south striking in the wetland treatment area and were fractured from weathering. Bedrock geology in the area of the site is Precambrian Neoarchean-Paleoproterozoic undivided granulite-facies gneiss (Fontaine and Mallory, 2011).

4.2.2.4. Groundwater Characteristics

The average hydraulic gradient of the groundwater flow area from June 28, 2012 to July 12, 2012 was 0.04. The average direction of groundwater flow during that period had a bearing of 110°. The estimated advective travel time of effluent from the infiltration area to the seepage area was calculated to be 45 days assuming the average K_{fs} value of 5.8 m/d, a horizontal subsurface travel distance of 50 m, and an effective porosity of 0.20

typical of clean sand and gravel (Fetter, 2001). The travel time of the effluent determined from the tracer test was shorter than that estimate, as discussed in the following section.

4.2.3. Hydraulic Residence Times

The *HRT* of the wetland treatment area varies throughout the treatment season depending on the flow type (*i.e.*, surface or subsurface). Figure 4.12 shows the Br^- tracer response curve used to estimate the *HRT* of the subsurface flow segment of the wetland segment (WS-3) during the low inlet discharge conditions observed in mid- to late summer July 2012. The bromide tracer response curve is not well defined; this is evident with the low peak concentration of 0.7 mg/l Br^- observed approximately 14 days after the tracer injection into the subsurface infiltration area. Grab samples collected 80 meters downstream of the ISE probe at the outflow area of WS-3 provided verification of the tracer response curve bromide concentrations with IC analysis. The grab samples demonstrate an increase in bromide concentration that correlates with the peak observed in the ISE data. As a result of the poor tracer response, the *HRT* of 14 days for the subsurface flow segment of the wetland treatment area is an approximation.

Figure 4.12 also shows two minor pulses of bromide after the main tracer front which could be explained by flushing of the subsurface flow segment in response to precipitation events. This suggests that the groundwater flow rate in the subsurface segment of the wetland treatment area is variable in accordance to variable hydraulic loading rates on the wetland segment. A minor peak is notable in advance of the main tracer front and this is explained by ISE probe stabilization to the field conditions. The probe can take a few hours to stabilize to field conditions after deployment.

The rough estimate of the groundwater *HRT* of 45 days discussed in Section 4.2.2.4 is higher than the tracer determined *HRT* of 14 days. The difference between the two estimates could be attributable to a variety of causes. One explanation could be due to the suspected layer of coarse gravel which likely underlies surface soil discussed in Section 4.2.2.1. The gravel layer would have a higher hydraulic conductivity and act as a preferential flow path leading to faster travel time of effluent through the groundwater flow area, which could explain the discrepancy between the estimated *HRT*s for the groundwater flow area. Other reasons include spatially and temporally varying hydraulic gradients, and lateral subsurface inflow from the WSP and surrounding area. The estimate of 14 days would be the most conservative in this case as it has the least assumptions and was the shortest estimate.

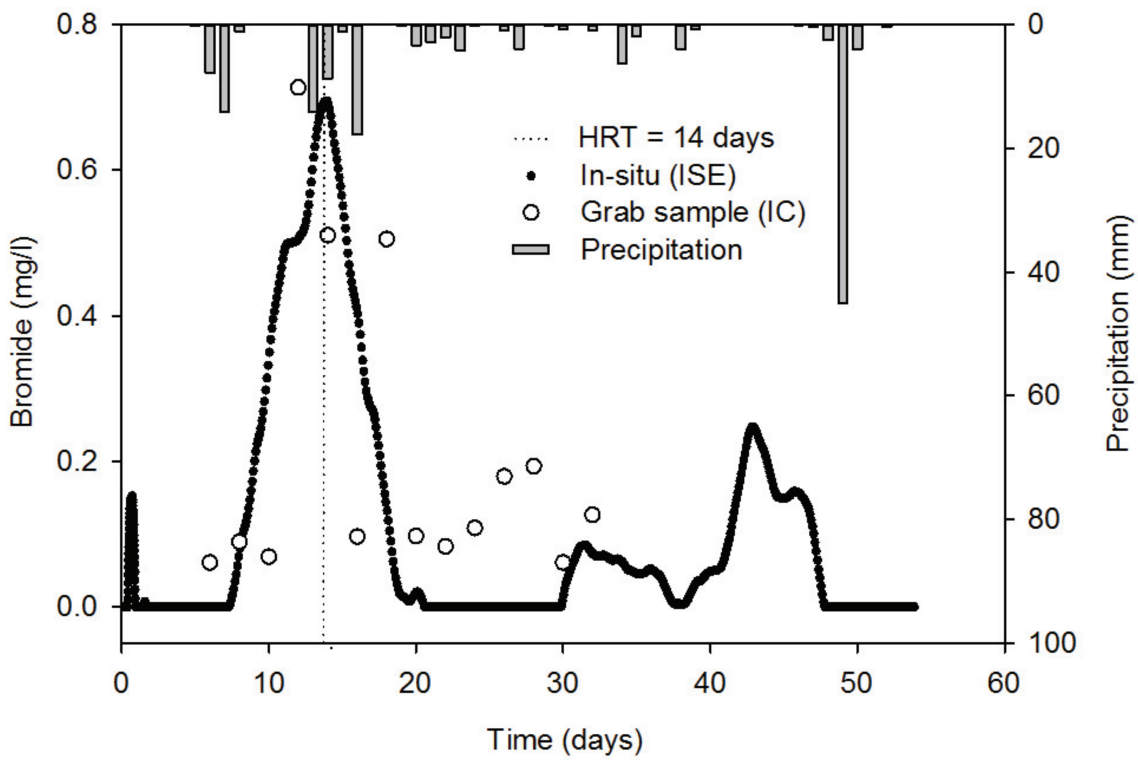

Figure 4.12 Br$^-$ tracer response curve for subsurface flow segment of the wetland treatment area. The tracer test was conducted on July 8, 2012. The *in-situ* ISE was deployed in a surface water pond down gradient of the subsurface flow segment of the wetland treatment area. The ISE data was used for the determination of *HRT*. The surface water grab samples were collected as an analytical check on the probe response at a sample point 80 m downstream from the ISE probe.

Figures 4.13 to 4.16 illustrate the RWT tracer response curves that were used to compute the *HRT*s of the wetland segments characterized by surface flow (*i.e.,* WS-1, WS-2, WS-3, and WS-4). The tracer response curves shown in Figures 4.13 and 4.14 indicate that the tracer concentration measured by the YSI rhodamine WT optical probe are above the calibration limit of 200 μg/l. The over range concentrations in the tracer response curves are not ideal; however, the maximum tracer concentrations are near the upper linear calibration limit, which would limit the amount of error.

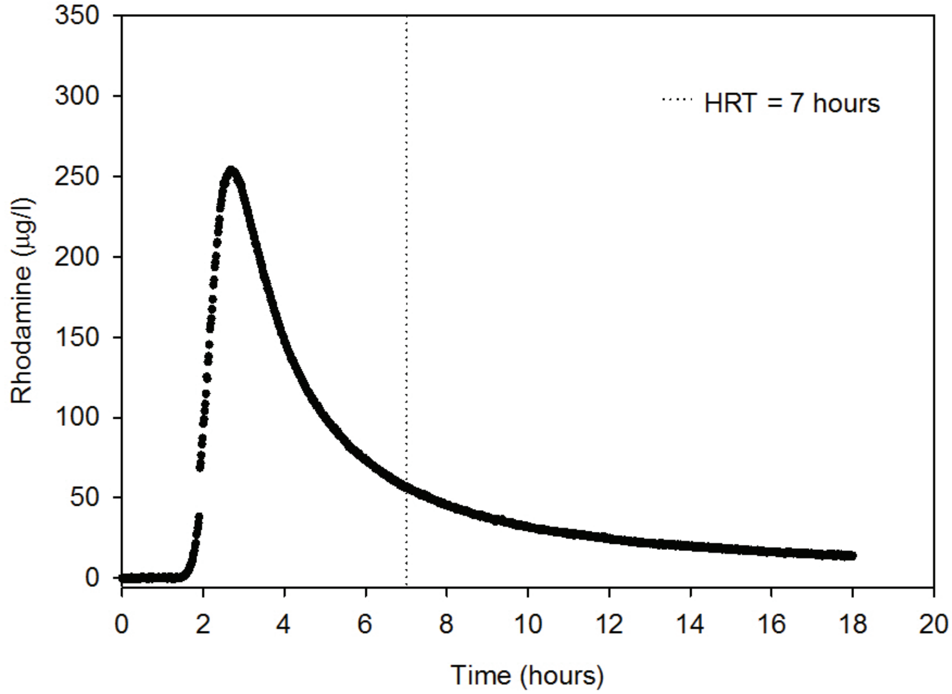

Figure 4.13 Tracer response curve for wetland segment WS-1 (upper half of the wetland during spring freshet). The tracer test was conducted on June 18, 2011.

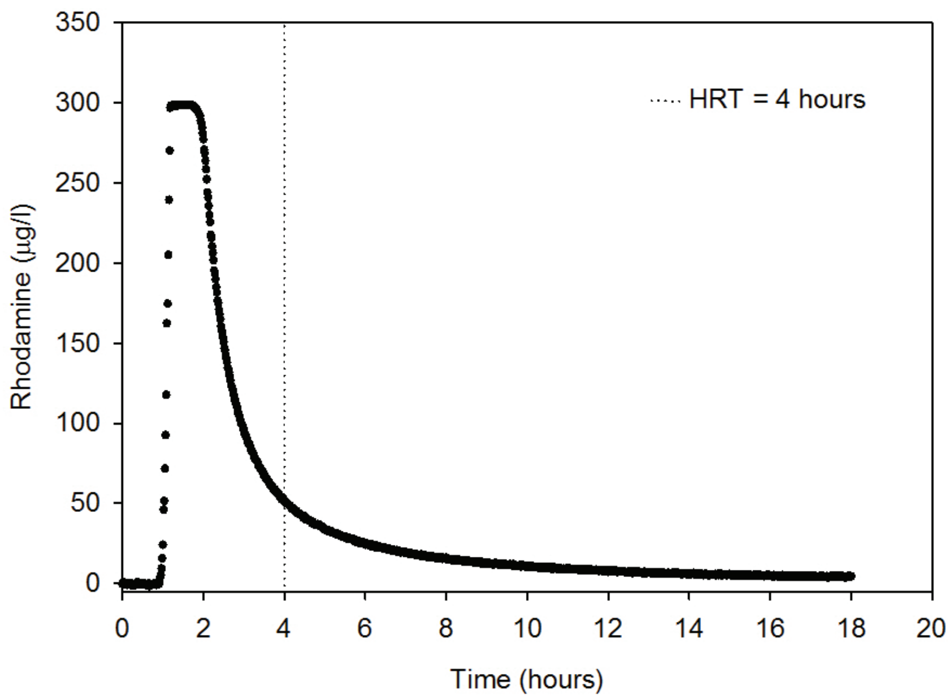

Figure 4.14 Tracer response curve for wetland segment WS-2 (lower half of the wetland during spring freshet). The tracer test was conducted on June 17, 2011.

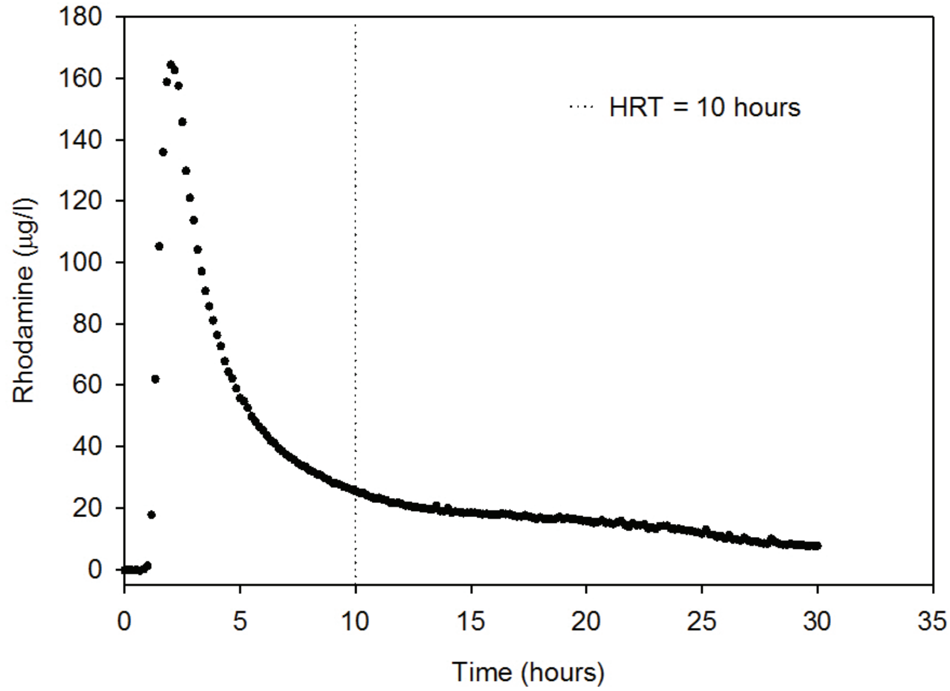

Figure 4.15 Tracer response curve for wetland segment WS-3 (upper half of the wetland after spring freshet). The tracer test was conducted on July 2, 2012.

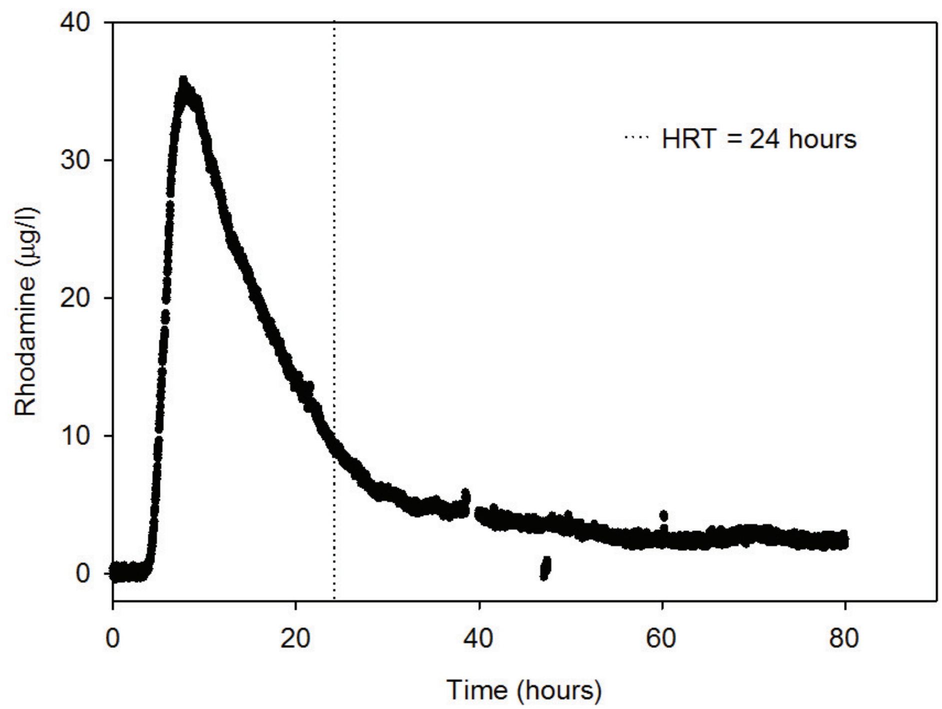

Figure 4.16 Tracer response curve for wetland segment WS-4 (lower half of the wetland after spring freshet). The tracer test was conducted on September 7, 2011.

The total *HRT*s determined from the RWT and Br^- tracer tests are summarized in Table 4.3. The *HRT*s for surface flow wetland segments range from a minimum of 11 hours during the spring freshet in June 2011 to 1.4 days by mid-summer in July 2012. The longest *HRT* during the study period is two weeks in July 2012. This coincides with effluent infiltration into the subsurface, near the wetland inlet. During this time, the effluent travels partially as subsurface flow through the wetland treatment area. Generally, the *HLR* on the wetland treatment area governs whether the flow is characterized by surface or subsurface flow. As a result of the seasonally variable *HLR*, the *HRT* of the wetland varies with the seasonal conditions.

Table 4.3 The total hydraulic residence times of the wetland treatment area during the study periods over the treatment season.

Study period	**Total wetland treatment area *HRT***
Spring freshet	11 hours
Mid-summer	1.4 to 14 days
Fall	1 to 14 days

In comparison to other FWS wetlands, the site has more variation in the range of *HRT*s observed over the treatment season resulting from uncontrolled influent discharge onto the wetland treatment area. The *HRT*s during the spring freshet and mid-summer were shorter than most other FWS wetlands, for instance Kadlec and Wallace (2009) list 34 FWS wetlands as having a mean *HRT* of 6.5 days, which is greater than the range of 11 hours to 1.4 days determined from the tracer studies in this study. Two other natural wetlands in northern Canada at Teslin (7 ha), and Haines Junction (6 ha), Yukon were reported to have *HRT*s of 2 – 4 d and 4.5 days respectively, which are short (Doku and Heinke, 1995); however, still greater than the minimum *HRT* observed in Coral Harbour.

The variability in *HRT*s observed on site is a product of the natural and un-engineered nature of the wetland treatment area. Likewise, Mudroch and Capobianco (1979) noted seasonal variability in *HRT*s of a natural marsh receiving municipal wastewater in Dundas, Ontario, Canada. Within their study, they noted a spring *HRT* of 2.2 days, whereas the summer *HRT* was much longer at 11.8 days. The decrease in *HRT* of wetlands operating in cold climates during snow melt has been described by Wittgren and Mæhlum (1997). They suggested that the wetland-to-catchment ratio will influence the magnitude of variation in *HRT* observed during the spring melt, with a small ratio leading to a more dramatic decrease in *HRT*.

4.2.3.1. Volumetric Efficiency

The dimensions and volumetric efficiencies of each WS where a tracer test was conducted are summarized in Table 4.4. The average volumetric efficiency over all four tracer tests is 0.49; therefore, approximately half of the wetland segments are active and being utilized for treatment.

Table 4.4 Average depths, areas, and volumetric efficiencies of each wetland segment on the date of the tracer tests.

Tracer test date	Wetland segments	Average depth (m)	Area (m^2)	Volumetric efficiency (e_v)
18/06/2011	WS-1	0.07	8374	0.67
17/06/2011	WS-2	0.06	6246	0.39
07/09/2011	WS-3	0.04	7508	0.22
02/07/2012	WS-4	0.08	3138	0.67

4.3. Biogeochemistry

The wetland treatment area biogeochemistry is described with water quality indicators. The water quality indicators measured in the wetland treatment area show seasonal, diurnal, and spatial trends as demonstrated in the following sections. The biological production in the treatment area is likely a strong influencer on the overall water quality trends observed.

4.3.1. Seasonal Trends

Table 4.5 summarizes the average, maximum, and minimum values of pH, DO, conductivity, and temperature observed *in-situ* in the mid-wetland at various intervals over the treatment season. The pH of the mid-wetland was lower on average earlier in the treatment season during the spring freshet. Likewise, the DO values were lower on average in June 2011 (2.2 mg/L), than in the mid-summer July – August 2012 period (9.2

mg/L), and September, which averaged 12.2 mg/L. The lower pH and DO values early in the treatment season were likely attributed to decreased biological production early in the treatment season. The conductivity measurements were lowest in the mid-summer study period at 445 μS/cm, which differs from 654 and 767 μS/cm, in spring and fall respectively. The lower average conductivity in mid-summer could be attributed to increased metabolism of ionic species within the wetland treatment area due to algal growth and potentially increased microbial activity resulting from higher average temperatures. The average temperature was highest in the wetland treatment area during the mid-summer at 11°C, compared to 7.8 and 2.7°C, in spring and fall respectively.

Table 4.5 Summary of basic water quality over the treatment season measured at the mid-wetland shown as average, maximum, and minimum values.

Study period (m/d/yyyy)	**Parameter**	**Average**	**Maximum**	**Minimum**
Spring freshet 6/15/2011 - 6/24/2011	pH	7.7	8.0	6.9
	DO (mg/L)	2.2	12	1.4
	conductivity (μS/cm)	654	789	588
	temperature (°C)	7.8	17	0.0
Mid-summer 6/30/2012 - 8/31/2012	pH	7.9	8.8	7.5
	DO (mg/L)	9.2	17	3.4
	conductivity (μS/cm)	445	608	255
	temperature (°C)	11	25	0.3
Fall 9/11/2011 - 9/17/2011	pH	8.0	8.7	7.2
	DO (mg/L)	12.2	18	0.9
	conductivity (μS/cm)	767	902	637
	temperature (°C)	2.7	11	-0.1

4.3.2. Diurnal Trends

The water quality data shows diurnal trends in pH and DO levels in the mid-wetland which were associated with changes in wetland biogeochemistry related to algal growth. The diurnal trends in pH and DO during the spring freshet and mid-summer are illustrated with 48-hour time series in Figures 4.17 and 4.18 respectively. Refer to Appendix C for an example of a 48-hour time series for the fall study period in September 2011. Figures 4.17 and 4.18 show an increase in pH and DO associated with the solar maxima. The diurnal fluctuations in pH and DO were more pronounced and DO was supersaturated during the post-spring freshet conditions likely due to increased algal growth after the spring freshet. The algal growth raised pH and produced DO in conjunction with daily solar maxima (Crites and Tchobanoglous, 1998). Herskowitz *et al.* (1987) noted similar DO trends in the Listowel, Ontario artificial marsh treatment system, where DO depletion in the winter was followed by DO peaks as a result of algal blooms in the spring.

The diurnal trends in conductivity were less pronounced during the spring freshet when compared to post-spring freshet conditions (Figures C-2 and C-3 in Appendix C). This difference may be due to increased algal growth during the post-freshet period lending to increased metabolism of ionic species in the wetland treatment area in conjunction with daylight.

Overall, the pH, DO, and temperature demonstrate diurnal fluctuation, whereby they increase in the daytime and decreased at night. Conductivity exhibited an inverse behavior, where it decreased in the daytime and increased at night.

4.3.3. Spatial Trends

Figures 4.19 and 4.20 summarize the discrete measurements of pH and DO during the four study periods at the inlet, mid-wetland, outlet of the wetland, and the reference site. Section a. of both Figures 4.19 and 4.20 shows the pH and DO across the wetland treatment area during the spring freshet of June 2011. The pH and DO levels at the inlet during that time suggest that anoxic conditions were present. Herskowitz *et al.* (1987) also reported similar anoxic influent conditions in the Listowel, Ontario artificial marsh treatment systems as a result of anoxic conditions in the WSP during the winter due to ice cover. The pH and DO levels at the mid-wetland and at the outlet increased steadily in advance of the inlet during the spring freshet, which may suggest that biological production occurred earlier further away from the inlet location. The reason for the difference in pH and DO response may be due to shallower water depths and associated higher water temperatures at the mid-wetland and outlet which would be a favorable environment for algal growth.

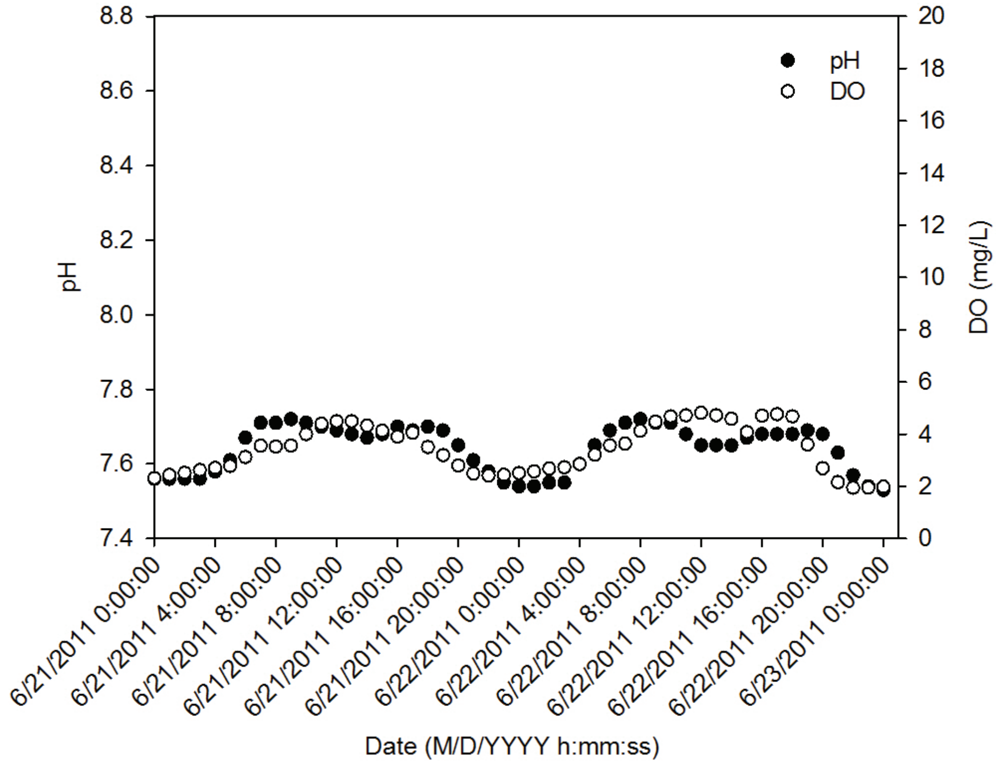

Figure 4.17 Diurnal 48-hour time series of *in-situ* sonde measurements of pH and DO at the mid-wetland during the spring freshet in June 2011.

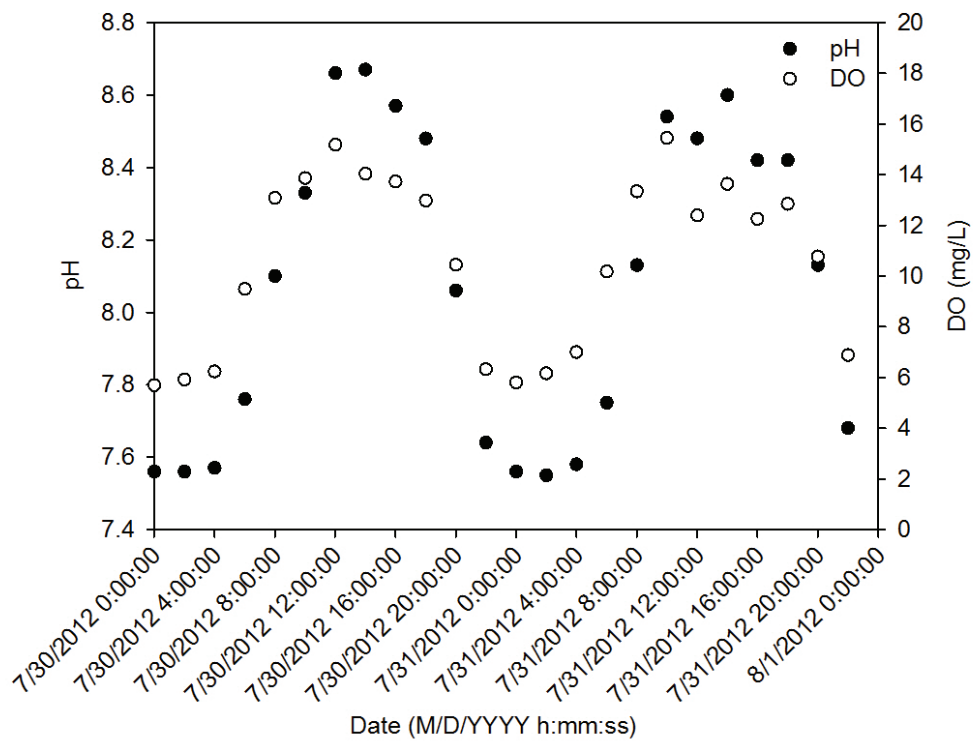

Figure 4.18 Diurnal 48-hour time series of *in-situ* sonde measurements of pH and DO at the mid-wetland during the post-spring freshet conditions in July 2012.

The pH and DO levels across the wetland treatment area during the fall of September 2011 are shown in Section b. of Figure 4.19 and 4.20. The pH and DO levels at the wetland inlet were generally higher as the treatment season progressed. During the fall, the pH levels at all three locations in the wetland treatment area were similar and did not trend up or down. The DO concentrations in the fall were slightly lower at the inlet; however they suggested oxic conditions. The mid-wetland and outlet DO concentrations were stabilized and very similar.

The mid-summer pH and DO conditions in July 2012 are shown in section c. A sudden increase in pH and DO at the inlet is notable on July 5, 2012 and July 3, 2012 respectively. Those increases likely reflect the conversion event of the inlet from anoxic to oxic conditions due to algae proliferation in the WSP. During that time, the mid-wetland pH and DO were slightly lower than the outlet, which was possibly due to the concentration of effluent at the wetland mid-point at that time.

Overall, the pH in the reference wetland was most comparable to the values observed at the mid-point in the wetland treatment area (Figure 4.19). The DO concentrations in the reference site were comparable to the outflow from the wetland (Figure 4.20). There were spatial differences in basic water quality parameters measured in the wetland treatment area. For example, the inlet exhibited different characteristics than the wetland mid-point and the outflow from the wetland. Those differences impact the treatment dynamics as the wastewater moves through the wetland treatment area. There were also temporal differences in the pH and DO levels measured across the wetland treatment area. Of most significance was the inlet varying between anoxic and oxic conditions during the treatment season. Temporal variability of water quality at the inlet impacts the loading rates of wastewater constituents into the wetland and subsequently affects the overall treatment performance.

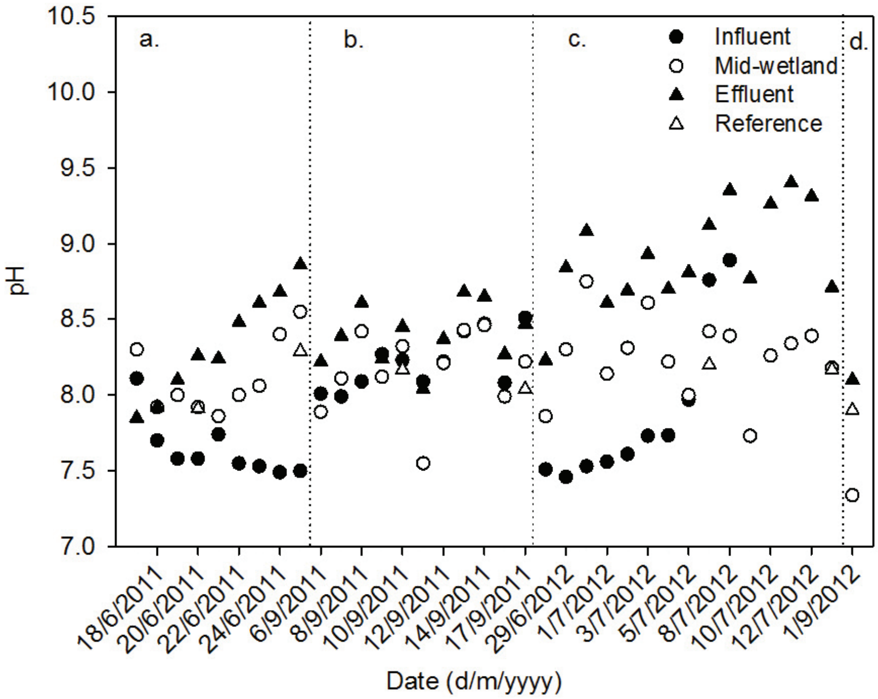

Figure 4.19 Instantaneous pH measurements at the inlet, mid-wetland, outlet, and reference site over the study periods. Section a. represents measurements taken in June 2011, b. is September 2011, c. is June and July 2012, and d. is September 2012.

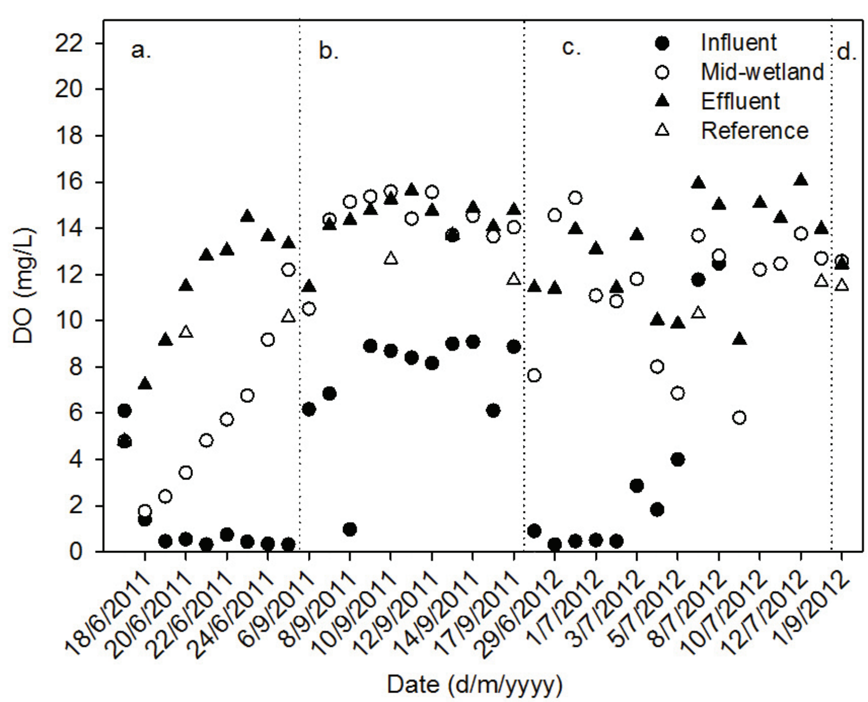

Figure 4.20 Instantaneous DO measurements at the inlet, mid-wetland, outlet, and reference site over the study periods. Section a. represents measurements taken in June 2011, b. is September 2011, c. is June and July 2012, and d. is September 2012.

4.3.4. Biological production

Table 4.6 displays the chlorophyll a (chla) and pheophytin concentrations measured in the WSP, wetland treatment area, receiving waters, and reference wetland in July 2012. The WSP had the highest chla concentration, followed by the wetland inlet. At the mid-wetland location onwards the chla concentrations were relatively low (<12 μg/L) with some indication of biological production within the wetland treatment area as higher concentrations were measured at the outlet than upstream locations. The uptake of nutrients by wetland vegetation may account for the limited growth of algae by the mid-wetland. This was also noted as an algae growth inhibition mechanism by Kennedy and Mayer (2002).

Table 4.6 Chlorophyll a and pheophytin concentrations observed on July 11, 2012 in the WSP, wetland treatment area, receiving lake, and reference wetland.

Sample location	**Chlorophyll a (μg/L)**	**Pheophytin[a] (μg/L)**
WSP	2 887	1 396
Inlet	352	97
Mid-wetland	12	7.0
¾ Wetland	8.1	2.7
Outlet	12	3.3
Receiving lake	4.8	2.8
Reference site	<0.5	<0.5

[a]Pheophytin is a degradation product of algae. The chla reported has been corrected for pheophytin presence.

4.4. Treatment Performance Assessment

The treatment performance of the wetland treatment area is assessed in the following sections. The percent and log reductions of each treatment performance parameter concentrations in comparison to the raw concentrations from the trucks are also summarized below. Finally, the metal concentration results are presented and discussed in the following section.

4.4.1. Raw Wastewater Quality

The raw wastewater quality from the truck sampling is summarized in Table 4.7. Tchobanoglous *et al.* (2003) summarized typical values for domestic wastewater constituents based on wastewater strength. They classify a medium strength wastewater as being a production rate of 460 L/capita/day and a high strength wastewater as being a production rate of 240 L/capita/day. The domestic wastewater production rate in Coral Harbour approximately 107 L/capita/day (Government of Nunavut, 2013). The raw wastewater quality for the samples collected at various times during the study periods for all parameters fall within or above the composition typical; of medium to high strength wastewater values given in Tchobanoglous *et al.* (2003). The mean $CBOD_5$, VSS, TN, and TP concentrations of the Coral Harbour wastewater are above the typical high strength values suggested in Tchobanoglous *et al.* (2003).

Table 4.7 Summary of the raw truck treatment performance parameter concentrations, standard deviations (σ), and sample numbers (n). The samples were collected on multiple occasions during the study periods.

Parameter	**Literature**[a]	**mean**	**max**	**min**	**σ**	**n**
$CBOD_5$ (mg/L)[b]	350	535	900	253	188	8
E. coli (MPN/100mL)	$10^5 – 10^8$	$9.8x10^7$	$4.1x10^8$	$9.8x10^6$	$1.3x10^8$	14
TSS (mg/L)	400	390	477	258	54	15
VSS (mg/L)	315	345	430	233	55	15
TN (mg/L)	70	101	125	78	14	11
TAN (mg/L)	-	59	78	39	13	7
NH_3-N (mg/L)	-	1.3	1.8	0.6	0.4	7
TP (mg/L)	12	13	16	10	1.9	11

[a]Literature values are based on typical high strength values for domestic wastewater in Tchobanoglous *et al.* (2003).
[b]Literature value is BOD_5.

4.4.2. Background Water Quality

The background water quality obtained from the reference wetland is summarized in Table 4.8. The reference wetland values indicate that the majority of the treatment performance parameter concentrations present as background levels were relatively low. The exception was TSS concentrations, which averaged 6 mg/L; however, that value is only slightly above the typical literature suggested value of 3 mg/L (Water Environment Federation, 2010). The background concentrations of *E. coli* averaged 1 MPN/100mL at the reference wetland, which was less than the literature reported typical value of 200 CFU/100mL (Water Environment Federation, 2010).

Table 4.8 Summary of the treatment performance parameter concentrations, standard deviations (σ), and sample numbers (n) for the reference site representative of background wetland constituents. The samples were collected on multiple occasions during the study periods.

Parameter	**Literature**[a]	**mean**	**max**	**min**	**σ**	***n***
$CBOD_5$ (mg/L)[b]	5	2.2	5.0	1.0	1.5	5
E. coli (MPN/100mL)[c]	200	0.5	2.0	n.d.[d]	0.8	6
TSS (mg/L)	3	5.6	21	n.d.	7.8	5
VSS (mg/L)	-	3.8	14	n.d.	5.1	5
TN (mg/L)	2	0.7	1.4	n.d.	0.6	5
TAN (mg/L)	-	0.1	0.3	n.d.	0.1	6
NH_3-N (mg/L)	1	n.d.	0.0	n.d.	0.0	6
TP (mg/L)	0.3	n.d.	0.1	n.d.	0.0	5

[a]Literature values are based on typical background concentrations for treatment performance parameters in Water Environment Federation (2010).
[b]Literature value is BOD_5.
[c]Literature value units in CFU/100mL.
[d]Non-detect concentration.

4.4.3. Five-Day Carbonaceous Biochemical Oxygen Demand

The $CBOD_5$ wetland inlet, mid-point, and outlet concentrations tabulated in Table 4.10 indicate that the concentrations were highest across the wetland treatment area during the spring freshet of June 2011. The $CBOD_5$ concentrations ranged from 54 to 152 mg/L, and non-detect to 25 mg/L, at the inlet and outlet respectively, during the study periods. The higher concentrations were likely attributed to the short *HRT*s of the wetland treatment area during the spring freshet which would limit the removal of BOD by settling or microbial metabolism. The maximum concentration observed at the wetland outlet during the study periods was 25 mg/L. For context, the wetland effluent met the EC WSER regulatory concentration of 25 mg/L that is not currently applicable to Nunavut (Environment Canada, 2010). However, effluent $CBOD_5$ concentrations above the EC WSER concentration were noted for the Coral Harbour wetland treatment area by Yates *et al.* (2012). During their 2008 treatment season; they noted a range from 5 to 54 mg/L.

Table 4.11 shows that the majority of the BOD reduction occured in the upper half of the wetland treatment area. $CBOD_5$ concentrations were the greatest there, which suggests the BOD removal was acting as a first-order process. The rapid reduction in

BOD near the wetland inlet is common among FWS wetlands as a result of the generally laminar like flow conditions within the wetland which encourages sedimentation (Water Environment Federation, 2010; Kennedy and Mayer, 2002). Reed[2] *et al.* (1988) suggest that it is common for the majority of settleable BOD to be removed very close to the wetland inlet. The percent reductions observed in the wastewater exiting the WSP ranged from 52.0 to 88.5%, while by mid-wetland, the reductions range from 84.5 to 99.7%. In comparison, there was minimal difference in the BOD reduction between the mid-wetland and the wetland outlet. For instance, for the periods studied, $CBOD_5$ averaged 96.1% reduction at the mid-wetland, and 97.3% reduction at the outlet.

The Listowel, Ontario FWS wetlands operating at average annual air temperatures of 7 to 8 ºC yielded an 83% BOD removal with an average influent BOD concentration of 56.3 mg/L (Water Environment Federation, 2010; Herskowitz *et al*., 1987). Mander *et al.* (2000) observed a 68% reduction in BOD_7 in a nearly identical sized 1.2 ha FWS constructed wetland in Põltsamaa, Estonia. In the Hay River, NWT 32 ha natural wetland, a 97.7% reduction was observed over the entire treatment season from June to October 1973 (Wright, 1974). Dubuc *et al.* (1986) observed a 99.0% reduction in BOD_5 in a natural peatland in northern Québec. For comparison, the minimum reduction observed in the Coral Harbour treatment area in June 2011 (87.4%) is higher than that observed in the Listowel, and Põltsamaa wetlands, but lower than the Hay River and Fontanges peat wetland.

The $CBOD_5$ percent reduction by the wetland treatment area inlet was lowest during the spring freshet of June 2011 with an 57.3% average reduction; in contrast to later in the treatment season when reductions ranged from 80.5 to 88.5%. The difference may be attributed to reduced BOD removal in the WSP and berm material earlier in the treatment season due to colder water temperatures resulting in slowed microbial metabolism.

4.4.3.1. Areal Biochemical Oxygen Demand Loading Rate

The areal *BLR* is typically not a primary design criterion; however, it is used as a check on aerobic conditions of the wetland (Reed[2] *et al.*, 1988). The *BLRs* for the wetland during the study periods are summarized in Table 4.9. The *BLR*s were highest during the spring freshet when effluent discharge onto the wetland treatment area was greatest. For context, an assembly of free surface wetlands receiving primary treated effluent (100 – 200 mg/l BOD) yielded a *BLR* range of 20 to 400 kg/ha·d resulting in BOD outlet concentrations ranging from 10 to 400 mg/L (Kadlec and Wallace, 2009). Additionally, a typical design criteria for *BLR* is noted as less than 110 kg/ha·d (Water Environment Federation, 2010; Crites and Tchobanoglous, 1998). The Coral Harbour wetland treatment area *BLR*s, summarized in Table 4.9, fall below the suggested design *BLR* value from the literature. Additionally, the highest *BLR* of 46.8 kg/had observed in June 2011 was just slightly above the 40 kg/ha·d design guideline for constructed wetlands suggested by the U.S. EPA (U.S. EPA, 2000). However, it should be noted that this was well above the *BLR* of 8 kg/ha·d recommended for natural treatment wetlands by Doku and Heinke (1995). The *BLR*s during the spring freshet are consistent with the ranges reported in literature (Water Environment Federation, 2010; Kadlec and Wallace, 2009). Contrastingly, during the post-spring freshet, the *BLRs* observed were well below the ranges reported in literature for FWS wetlands receiving primary treated effluent (Crites and Tchobanoglous, 1998). At an average 12 kg/ha·d, the Listowel four FWS wetland system received a higher *BLR* than in Coral Harbour; however, *BLR* near the inlet, was as high as 60 kg/ha·d for the Listowel marshes was observed, which was attributed to settleable BOD removal near the inlet (Reed[2] *et al.*, 1988)

Table 4.9 Areal *BLR* on the wetted area of the wetland treatment area on the dates when BOD measurements were obtained.

Study period	Date (m/d/yyyy)	*BLR* (kg/ha·d)
Spring Freshet	6/21/2011	47
	6/25/2011	31
Mid-summer	7/7/2012	2.7
	7/14/2012	0.1
Fall	9/10/2011	1.9
	9/17/2011	4.9
	9/1/2012	3.2

4.4.4. *Escherichia coli*

The *E. coli* concentrations observed in the wetland treatment area were extremely variable over the course of the treatment season (Table 4.10). The wetland influent *E. coli* concentrations ranged over three orders of magnitude from $1.0\text{x}10^3$ MPN/100mL in July 2012 to $2.0\text{x}10^6$ MPN/100mL in June 2011. Table 4.11 shows that inlet E. coli concentration reductions equates to a 1.1 log in June 2011 and, a 3.5 and 4.1 log in September 2011 and July 2012 respectively. Over the 2008 treatment season, Yates *et al.* (2012) observed an influent *E. coli* range in the wetland treatment area from $6.0\text{x}10^2$ to $1.5\text{x}10^5$ CFU/100mL, which affirms the tremendous variability of influent *E. coli* concentrations. The lowest *E. coli* concentrations were observed in conjunction with the timing of the conversion of the WSP and wetland inlet from anoxic to oxic conditions due to proliferation of algae growth in these areas; the resulting pH and DO increases are shown in section c. of Figures 4.19 and 4.20. The sudden decrease of the WSP's *E. coli* in response to pH increases attributed to algal growth was also observed by Parhad and Rao (1974). It is hypothesized that the rapid increase in pH and DO contributes to relatively lower *E. coli* concentrations observed at the wetland inlet post algae formation.

Over the course of the treatment season, *E. coli* concentrations also varied by over several orders of magnitude at the wetland mid-point and the outlet. Variability in *E. coli* concentrations has been observed in other natural wetlands receiving municipal wastewater (Kadlec and Tilton, 1979). The mid-wetland *E. coli* concentrations range from $4.5\text{x}10^1$ MPN/100mL in September 2011 to $2.4\text{x}10^5$ MPN/100mL in June 2011. Similarly, the wetland effluent concentrations ranged from 1 MPN/100mL in September

2011 to 1.9×10^4 MPN/100mL in June 2011. Yates *et al.* (2012) noted similar variability in effluent *E. coli* concentrations. Their values ranged from 3 to 1.2×10^3 CFU/mL during the 2008 Coral Harbour wetland treatment season. Table 4.11 shows the minimum reduction of *E. coli* at the outlet was a 3.0 log reduction during June 2011.

Later in the treatment season, up to an 8.4 log reduction of *E. coli* was noted. A possible reason for the comparatively lower reduction of *E. coli* during the spring freshet may be the shorter *HRT*s observed. For 14 other wetlands, the median $\log_{10}$ reduction in fecal coliforms was 2.14 (Kadlec *et al.*, 2010; Kadlec and Wallace, 2009). The Coral Harbour wetland demonstrated higher reductions in *E. coli* compared to the other wetlands.

There are two potential reasons for the variation in *E. coli* concentrations and log reductions observed over the treatment season which are: i) the shorter *HRT* during the June 2011 spring freshet, thereby decreasing the potential for removal by treatment; and ii) the waterfowl presence in the wetland treatment area coinciding with the spring freshet runoff, which could act to increase the *E. coli* concentrations in the wetland treatment area. Yates *et al.* (2012) also noted variable *E. coli* concentrations and deduced that it may be attributable to wildlife presence in the northern wetlands. A study by Orosz-Coghlan *et al*. (2006) of the Tres Rios constructed FWS wetland in Arizona demonstrated that for the wetland, *E. coli* strains sourced from both anthropogenic and passerine sources were detectable as being main sources on different occasions. They note that *E. coli* sourced from humans may indicate a greater pathogen risk than that sourced from birds and waterfowl. The contribution of waterfowl to elevated *E. coli* levels, which are non-representative of the wastewater effluent quality in a wetland is also noted by the authors McLain and Williams (2008). Within their work on a constructed wetland receiving municipal effluent, the highest levels of *E. coli* were observed in autumn when migratory waterfowl were present in the wetland.

There was an instance in July 2012, where the mid-wetland *E. coli* concentration (5.8 MPN/100mL) was lower than further downstream at the wetland outlet (1.1×10^2 MPN/100mL). It is possible that the apparent increase in *E. coli* then was non-point source addition of *E. coli* from waterfowl within the wetland treatment area, which is reinforced by Werker *et al.* (2002).

4.4.5. Total and Volatile Suspended Solids

The TSS concentrations at the wetland inlet are lower early in the spring than during post-freshet conditions (Table 4.10). For instance, the TSS concentrations ranged from 29 to 33 mg/L in June 2011. Those values, were much less than the post-freshet concentrations which ranged from 71 to 150 mg/L. Yates *et al.* (2012) reported a similar wide range of influent TSS concentrations during the 2008 treatment season in Coral Harbour. Their TSS values ranged from 6 to 560 mg/L. The difference in TSS concentrations observed at the wetland inlet over the treatment season was attributed to the growth of algae which was noted during all study periods except during the spring freshet in June 2011.

Williamson and Swanson (1979) noted the same effect of algae presence on increasing TSS concentrations in WSP effluent. They observed an increase in TSS concentrations to 105 mg/L during the summer, which was well above the average of 35 mg/L observed during the dormant season (November to May). Likewise, Herskowitz *et al.* (1987) observed higher than average TSS concentrations as a result of algae growth and evapotranspiration in the late spring and summer. Table 4.11 shows higher percent reductions in TSS at the inlet during the spring freshet ranging from 89.9 to 93.2%, while the post-spring freshet TSS reductions ranged from 60.3 to 81.8%. The difference is due to algae growth in the post-spring conditions at the wetland inlet, which resulted in higher TSS concentrations non-representative of raw wastewater. Hammer and Burckhard (2002) noted a similar algae contribution to TSS concentrations and the influence on apparent reduction of TSS. They observed that the apparent removal rate for TSS was higher at lower temperature conditions due to decreased biological production.

For all study periods, the maximum TSS effluent concentration measured at the wetland outlet was 18 mg/L, which is below the 25 mg/L stipulated by the EC WSER which is not currently applicable to Nunavut (Environment Canada, 2010). Yates *et al.* (2012), in their Coral Harbour study, reported a maximum TSS concentration of 27.5 mg/L for the 2008 treatment season, which was above the EC WSER. Table 4.11 shows the TSS percent reductions in the effluent ranged from 94.5% reduction in June 2011 to 99.9% in September 2011 and 2012. The minimum reduction of TSS observed was quite comparable to the 93% removal noted in the Listowel FWS wetlands, which received influent with an average TSS of 111 mg/L operating at average annual water temperatures of 7 to 8 ºC (Water Environment Federation, 2010; Herskowitz *et al*., 1987). Cameron *et al*. (2003) observed a 93% reduction in TSS in a FWS wetland located in Alfred, Ontario. The average percent reduction of TSS at the Hay River, NWT natural wetland (32 ha) was reported as 96.8% (Wright, 1974).

The reductions observed suggest that the wetland treatment area was more effective at reducing TSS during the post-spring freshet conditions. It is possible that a component of the TSS measured at the wetland outlet during the spring freshet conditions resulted from re-suspended particulate matter along the length of the wetland treatment area flow channel. That may have been caused by higher water velocities associated with spring freshet runoff conditions.

The VSS generally followed the same trends as the TSS with respect to concentrations and percent reduction; as such the parameter was not included as part of Tables 4.10 and 4.11. The volatile constituents represented by VSS are considered a reflection of the organic component of the suspended solids. An important observation in the VSS performance data is the similarity in the TSS and VSS concentrations, which suggests much of the TSS is composed of organic constituents. For example, the VSS concentrations on average composed between 87 to 97%, 21 to 100%, and 53 to 100% of the influent, mid-wetland, and effluent TSS concentrations respectively. By comparison, the Hay River, NWT wetland demonstrated a 98.0% reduction in VSS over a treatment season (Wright, 1974). The similarity in TSS and VSS concentrations supports the notion that much of the TSS is concentrations measured in the wetland treatment area are attributable to algae presence.

4.4.6. Nitrogen

The TN concentrations entering the wetland treatment area were highest during the spring freshet in June 2011 at 46.8 mg/L, whereas, the post-spring freshet concentrations ranged from 26.8 to 34.4 mg/L (Table 4.10). Similarly, the TN concentrations were highest in the mid-wetland and at the outlet during the spring freshet. The highest TN concentration at the outlet was 23 mg/L, which was much higher than the post-spring freshet maximum concentration of 4.1 mg/L. The minimum amount of TN reduction was observed at the inlet, mid-wetland, and outlet during the spring freshet as shown in Table 4.11.

Likewise to the trends noted in the TN, the TAN concentrations are highest at the wetland inlet, mid-point and outlet during the spring freshet in June 2011. The influent concentrations range from 28.7 to 45.5 mg/L TAN in June 2011 compared to the post-spring freshet range of 8.6 to 26.9 mg/L TAN. The highest effluent concentration observed during the spring freshet was 16.2 mg/L TAN compared to 6.1 mg/L in the post-spring freshet period. The NH_3-N concentrations were all below the EC WSER of 1.25 mg/L NH_3-N, which is currently non-applicable to Nunavut, in all periods studied at the wetland outlet. The pH conditions observed in the wetland treatment area were such that the ionized form of ammonia (ammonium NH_4^+) was preferentially favored. Due to the pH conditions present on-site (Figure 4.19); the NH_3-N concentrations were all below the EC WSER regulatory concentration except in two cases at the wetland inlet in September 2011 and July 2012. In some cases the NH_3-N concentrations in the wetland treatment area increased in relation to the raw samples; in these cases, the percent reduction in Table 4.11 is noted with a negative sign due to the differing and generally lower pH's measured in the raw water.

A TN reduction of 45.8% was observed at the wetland inlet during the spring freshet, whereas, a range of 64.6 to 70.4% TN reduction was noted during the post-spring freshet conditions. Similar differences were observed in the mid-wetland with 64.5% reduction in TN noted during the spring freshet compared to a range of 94.9 to 98.3% TN reduction during the post-spring freshet conditions. At the outlet, the TN percent reduction was 73.4% during the spring freshet, while the TN reductions observed for all other post-spring freshet conditions ranged from 96.3 to 99.2%. Dubuc *et al.* (1986) noted a 93.4% average reduction for a natural peatland in northern Québec.

The June 2011 spring freshet effluent TAN reduction had a wide range from 72.3 to 96%. The minimum percent reduction in TAN at the inlet of 34.3 % was observed during the spring freshet. Overall, during the post-spring freshet conditions, the TAN reductions observed at the outlet were much greater than during the spring, ranging from 86.4 to 100.0%. The FWS wetland in Alfred, Ontario, Canada demonstrated a 52% reduction in total ammonia which was much less than the reductions observed in the Coral Harbour wetland treatment area (Cameron *et al.*, 2003). Dubuc *et al.* (1986) noted a higher ammonia reduction of 96.6% for a northern Québec peatland. The Coral Harbour wetland is receiving dilutive inputs which accounts for the intersystem differences observed.

The seasonal difference in TN and TAN reduction may be explained by the volatilization of the ammoniacal component of TN due to the pH increase and uptake from algae growth (Tilton and Kadlec, 1979). The resulting pH increases from algal proliferation during the post-spring freshet study periods in the WSP, and at the inlet, are exemplified in Figure 4.19. Additionally, the removal of ammonium and ammonia nitrogen by nitrification requires a sufficient oxygen supply (Herskowitz *et al.*, 1987); which was not available during the spring freshet conditions at the wetland treatment area inlet.

4.4.7. Phosphorus

TP concentrations shown in Table 4.10 were the highest at the wetland inlet, mid-point, and outlet during the spring freshet in June 2011, in comparison to the other periods studied. Generally, the post-spring freshet conditions yielded low or non-detect (n.d.) TP concentrations; however, there were instances when concentrations of up to 2.8 mg/L TP were observed at the wetland outlet. Comparatively, the study by Herskowitz *et al.* (1987) on the Listowel, Ontario artificial marsh wetlands observed effluent phosphorus concentrations all below 1 mg/L. The wetland treatment area was only slightly effective at reducing the TP during the spring freshet in June 2011; however, this was when the most elevated TP concentrations were observed across the wetland treatment area. During the spring freshet period at the wetland outlet, a 73.3% reduction in TP was observed, which is close to the 64.5% reduction in TP that was observed entering the wetland treatment area (Table 4.11). For context, Cameron *et al.* (2003) observed that the Alfred, Ontario wetland demonstrated a 90% TP removal; which is quite a bit greater than the Coral

Harbour wetland treatment area. Dubuc *et al.* (1986) reported a 98.3% reduction in TP in a northern peatland. Mander *et al.* (2000) reported a 24% reduction of phosphorus for a near identical sized FWS wetland located in Põltsamaa, Estonia. They cited phosphorus as the most limiting parameter with reduced removal from March to May; which they suggested may be connected to the thermal regime of the system.

A major mechanism for phosphorus removal in wetlands is adsorption onto oxy-hydroxides of iron and aluminum which requires aerobic wetland conditions (Herskowitz *et al.*, 1987). The DO concentrations at the wetland inlet suggested anaerobic conditions during the spring freshet, which may have been a contributing factor to elevated concentrations of phosphorus observed at the wetland outlet. Short *HRT*s is a main reason for low phosphorus removal (Mudroch and Capobianco, 1979; Knight *et al.*, 1987). It has also been suggested that the contact between bottom substrate and the effluent is important to remove nutrients, and contact time is low in deep waters (Kadlec and Tilton, 1979). Schreijer *et al.* (1997) observed low phosphorus removal from a FWS constructed wetland in the Netherlands with a *HRT* of 2 days and suggested that for their system, an *HRT* of seven days would be required. The reduction in TP observed in the Coral Harbour wetland treatment area is likely mostly attributable to dilution from non-effluents contributions, as the short *HRT*s would not be conducive to facilitate phosphorus contact and adsorption onto wetland substrate and plant/algae assimilation.

4.4.8. Metals

The metals concentrations for the raw truck samples, inlet, the mid-wetland, outlet, and the reference site are summarized in Table 4.12. The results show that the background concentrations of magnesium, potassium, and calcium in all the samples were elevated, likely due to geological conditions of the area, as the reference wetland water quality is consistent with the rest of the sample locations. The wetland treatment area shows varying effectiveness at metals concentration reduction. The concentrations of magnesium and calcium increased in the wetland treatment area likely due to the carbonate containing surficial soil in the wetland treatment area. Lower percent (<90 %) reductions were observed for potassium, iron, manganese, lithium, vanadium, cobalt, nickel and barium; however, some of these have low level concentrations in the raw water. The iron, aluminum, and copper concentrations exceed the CCME *Canadian Guidelines for the Protection of Aquatic Life* (CCME, 2013) at the wetland inlet. Only iron and copper exceed the guidelines by the wetland outlet; however, the copper concentrations at the outlet are less than in the reference wetland. Other metals that are not included in the CCME guidelines but that were observed in greater concentrations in the wetland treatment area than the reference site included potassium, calcium, manganese, lithium, titanium, vanadium, and cobalt.

4.4.9. Subsurface Treatment Potential

The subsurface flow area indicated in Figure 4.10 showed some potential to provide treatment of the effluent during the limited period in the post-freshet when it was not bypassed by surface water flow. Two rounds of treatment performance samples were taken from two of the observation wells labeled OW-1 and OW-2 in Figure 4.10. The observation well OW-1 was approximately 30 m cross and up gradient of OW-2. The wells were sampled on July 14, 2012 and September 1, 2012. The treatment performance results from the wells were variable and did not consistently show reductions in concentrations within the subsurface flow area. Finer temporal scale sampling than a one point grab sample per week would be required to quantify the treatment potential of this area of the wetland. Since, this was only an ephemeral flow area, focus was not placed on further fine scale sampling. The groundwater flow area could be providing some treatment as was suspected with limited sampling results. However, this treatment mechanism in the wetland treatment area is not optimized and is bypassed much of the treatment season by overland surface flow.

Table 4.10 Summary of the treatment performance parameter concentrations for the wetland treatment area at the inlet, mid-wetland, and outlet.

		Influent			**Mid-wetland**[b]			**Effluent**		
Paramete	**Sample period**	mean	max	min	mean	max	min	mean	max	min
$CBOD_5$ (mg/L)	June 2011[a]	119	152	87	34	40	28	24	25	23
	Sept 2011 (2012)	108 (54)	144	72	3 (2)	5	1	1 (2)	2	n.d.
	July 2012	86	89	82	10	14	7	5	8	1
E. coli (MPN/100 mL)	June 2011	$1.4x10^6$	$2.0x10^6$	$7.7x10^5$	$2.0x10^5$	$2.4x10^5$	$1.7x10^5$	$1.2x10^4$	$1.9x10^4$	$5.4x10^3$
	Sept 2011 (2012)	$1.1x10^5$ ($4.0x10^4$)	$2.0x10^5$	$2.2x10^4$	$5.4x10^1$ ($1.3x10^2$)	$6.3x10^1$	$4.5x10^1$	2 (3.6)	3	1
	July 2012	$2.3x10^3$	$3.6x10^3$	$1.0x10^3$	$6.2x10^1$	$6.6x10^1$	$5.8x10^1$	$5.9x10^1$	$1.1x10^2$	7
TSS (mg/L)	June 2011	31	33	29	25	26	24	17	18	17
	Sept 2011 (2012)	148 (71)	150	146	34 (2)	64	4	1 (n.d.)	2	n.d.
	July 2012	100	110	90	4	4	4	13	18	8
TN (mg/L)	June 2011	47	-	-	31	-	-	23	-	-
	Sept 2011 (2012)	34 (27)	-	-	1.8 (4.6)	-	-	1.6 (0.7)	-	-
	July 2012	33	34	32	5.0	5.5	4.5	2.6	4.1	1.2
TAN (mg/L)	June 2011	34	46	29	20	30	13	10	16	2.8
	Sept 2011 (2012)	15 (8.6)	23	8.6	1.6 (0.1)	5.3	n.d.	1.7 (n.d.)	6.1	n.d.
	July 2012	22	27	15	5.8	12	0.3	0.9	3.0	n.d.
NH_3-N (mg/L)	June 2011	0.4	0.4	0.3	0.9	2.7	0.3	0.5	0.7	0.4
	Sept 2011 (2012)	0.7 (0.1)	1.3	0.2	0.1 (n.d.)	0.3	n.d.	0.1 (n.d.)	0.4	n.d.
	July 2012	0.8	2.5	0.2	0.6	1.4	n.d.	0.1	0.3	n.d.
TP (mg/L)	June 2011	3.7	-	-	3.4	-	-	2.8	-	-
	Sept 2011 (2012)	3.7 (2.3)	-	-	0.1 (0.1)	-	-	n.d.	-	-
	July 2012	3.1	3.1	3.0	0.4	0.6	0.1	0.4	0.5	0.2

[a]June 2011 results are presented here as BOD_5.
[b]Mid-wetland samples are minimum 16 hour composite except for grab samples for *E. coli*.
[c]n.d. denotes a non-detectable concentration.

Table 4.11 Summary of the percent and log reductions for treatment performance parameters in relation to the raw truck wastewater quality for the wetland treatment area at the inlet, mid-wetland, and outlet.

		Influent			**Mid-wetland**[b]			**Effluent**		
Parameter	**Sample period**	Mean	Max	Min	Mean	Max	Min	Mean	Max	Min
$CBOD_5$ (% reduction)	June 2011[a]	57.3	62.6	52.0	87.3	90.1	84.5	90.7	93.9	87.4
	Sept 2011 (2012)	80.5 (88.5)	80.6	80.5	99.5 (99.7)	99.6	99.4	99.9 (99.6)	100.0	99.7
	July 2012	81.7	82.4	81.0	97.8	98.5	97.1	99.0	99.7	98.2
E. coli (log reduction)	June 2011	1.2	1.3	1.1	2.0	2.0	2.0	3.3	3.7	3.0
	Sept 2011 (2012)	3.5 (2.7)	4.1	2.9	6.6 (5.2)	6.6	6.5	8.1 (6.7)	8.4	7.7
	July 2012	4.1	4.3	3.8	5.6	5.6	5.6	5.9	6.6	5.3
TSS (% reduction)	June 2011	91.6	93.2	89.9	93.2	94.3	92.1	95.3	96.1	94.5
	Sept 2011 (2012)	62.6 (81.8)	65.0	60.3	91.1 (99.5)	99.0	83.2	99.7 (99.9)	99.9	99.5
	July 2012	74.8	77.7	71.9	99.0	99.0	98.9	96.8	98.1	95.5
TN (% reduction)	June 2011	45.8	-	-	64.5	-	-	73.4	-	-
	Sept 2011 (2012)	67.1	-	-	98.3 (95.5)	-	-	98.5 (99.3)	-	-
	July 2012	67.5	70.4	64.6	95.2	95.4	94.9	97.5	98.8	96.3
TAN (% reduction)	June 2011	42.6	50.9	34.3	63.3	70.6	56.1	84.2	96.0	72.3
	Sept 2011 (2012)	79.8 (78.2)	80.2	79.3	93.1 (99.8)	98.1	88.1	92.0 (100.0)	97.6	86.4
	July 2012	72.3	72.6	72.0	98.3	99.4	97.2	100.0	100.0	100.
NH_3-N (% reduction)	June 2011	70.2	74.7	65.8	-0.8	73.1	-74.7	55.6	70.4	40.9
	Sept 2011 (2012)	27.9 (78.2)	42.0	13.8	76.6 (99.9)	95.7	57.4	63.1 (100.0)	90.7	35.6
	July 2012	-36.9	-	-	96.2	99.2	93.2	99.9	100.0	99.8
TP (% reduction)	June 2011	64.5	-	-	67.1	-	-	73.3	-	-
	Sept 2011 (2012)	72.6 (78.5)	-	-	99.5 (99.5)	-	-	99.7 (99.9)	-	-
	July 2012	77.8	78.8	76.8	97.3	99.2	95.4	97.2	98.3	96.1

[a]June 2011 results are presented here as BOD_5.
[b]Mid-wetland samples are minimum 16 hour composite except for grab samples for *E. coli*.

Table 4.12 Summary of metals concentrations (± standard deviation) including raw, the wetland treatment area inlet, mid-wetland, the outlet, and the reference wetland. The samples were collected in July and September 2012.

Metal	**CCME Guideline**	**Raw**	**Inlet**	**Mid-wetland**	**Outlet**	**Reference wetland**	**% reduction**
(µg/L)	(µg/L)	n = 7	n = 3	n = 3	n = 3	n = 3	
Magnesium	-	4791 ± 656	3450 ± 702	5973 ± 829	5324 ± 624	5612 ± 1713	-11
Potassium	-	25890 ± 2350	15550 ± 2743	9754 ± 867	5323 ± 1467	2841 ± 778	79.4
Calcium	-	36348 ± 4757	34190 ± 11095	96420 ± 24449	77775 ± 13725	57520 ± 15053	-114
Manganese	-	41 ± 3.7	72 ± 38	56 ± 37	20 ± 12	1.3 ± 0.4	51.2
Iron	300	822 ± 147*	653 ± 140*	341 ± 168*	389 ± 221*	52 ± 4.8	52.7
Lithium	-	5.5 ± 2.1	4.1 ± 0.7	4.0 ± 0.8	4.5 ± 1.6	2.5 ± 0.2	18.2
Aluminum	100	3018 ± 714*	128 ± 51*	18 ± 1.7	18 ± 3.7	32 ± 5.9	99.4
Titanium	-	39 ± 3.7	11 ± 1.2	1.5 ± 0.7	1.5 ± 0.7	1.3 ± 0.1	96.2
Vanadium	-	0.9 ± 0.1	0.8 ± 0.1	0.6 ± 0.2	0.5 ± 0.2	n.d.	44.4
Chromium	1	3.7 ± 0.5*	0.5 ± 0.1	n.d.	n.d.	n.d.	100
Cobalt	-	0.7 ± 0.1	0.6 ± 0.1	1.6 ± 0.6	0.6 ± 0.3	n.d.	14.3
Nickel	25	8.0 ± 7.1	3.2 ± 0.4	5.5 ± 0.8	2.8 ± 0.9	7.7 ± 1.7	65.0
Copper	2	102 ± 29*	11 ± 2.7*	4.5 ± 2.3*	3.0 ± 0.8*	5.3 ± 0.3*	97.1
Zinc	30	290 ± 41	14 ± 3.5	6.8 ± 2.8	6.7 ± 3.6	n.d.	97.7
Arsenic	5	1.0 ± 0.3	0.8 ± 0.1	0.7 ± 0.2	n.d.	n.d.	100
Selenium	1	1.2 ± 0.2*	n.d.	n.d.	n.d.	n.d.	100
Silver	0.1	1.5 ± 2.3*	n.d.	n.d.	n.d.	n.d.	100
Cadmium	0.017	n.d.	n.d.	n.d.	n.d.	n.d.	-
Antimony	-	3.1 ± 2.3	n.d.	n.d.	n.d.	n.d.	100
Barium	-	23 ± 7.2	3.7 ± 1.1	15 ± 1.5	13 ± 1.3	14 ± 3.4	43.9
Cerium	-	2.0 ± 0.8	n.d.	n.d.	n.d.	n.d.	100
Lead	1	5.5 ± 2.2*	0.6 ± 0.0	n.d.	n.d.	n.d.	100
Uranium	15	1.0 ± 0.4	n.d.	n.d.	n.d.	n.d.	100

*Concentration exceeds the Canadian Water Quality Guidelines for the Protection of Aquatic Life (CCME, 2013).

4.5. Modified Tanks-In-Series Model

The areal rate coefficients for each treatment performance parameter were determined with a modified TIS wetland performance model. Each model was compartmentalized to determine the number of tanks (*N*) required to best characterize the WS hydraulics. The *k*-values were determined and compared to cold-climate wetland rate coefficients cited in literature.

4.5.1. Model compartmentalization

Model compartmentalization involved determining the appropriate number of theoretical tanks required to represent the hydraulic behavior of each WS. For the spring freshet hydrological conditions, the gamma model fit to the tracer RTD for the wetland segment WS-1 is shown in Figure 4.21. A least squares fit of the gamma model was obtained with four TIS. For the downstream wetland segment WS-2, the gamma model least squares fit to the RTD was obtained with five TIS (Figure 4.22).

During the mid-treatment season hydrological conditions represented by WS-3, the gamma model least squares fit to the tracer test RTD was optimized with two TIS (Figure 4.23). The fit between the tracer test RTD and the gamma model was relatively poor for WS-3 compared to the other wetland segments. For WS-4, the gamma model least squares fit to the tracer data is best approximated by three TIS as shown in Figure 4.24. Generally, comparable numbers of TIS have been found from other FWS wetland studies, with the most common being four TIS (Kadlec and Wallace, 2009). For a specific example, FWS wetland of comparable size and *HRT* of 4.2 days in Ekeby, Sweden was best approximated with three TIS (Kjellin *et al.*, 2007). The number of tanks obtained from the compartmentalization, and used for the determination of rate coefficients, varied according to the least squares gamma fit of the RTD of each wetland segment.

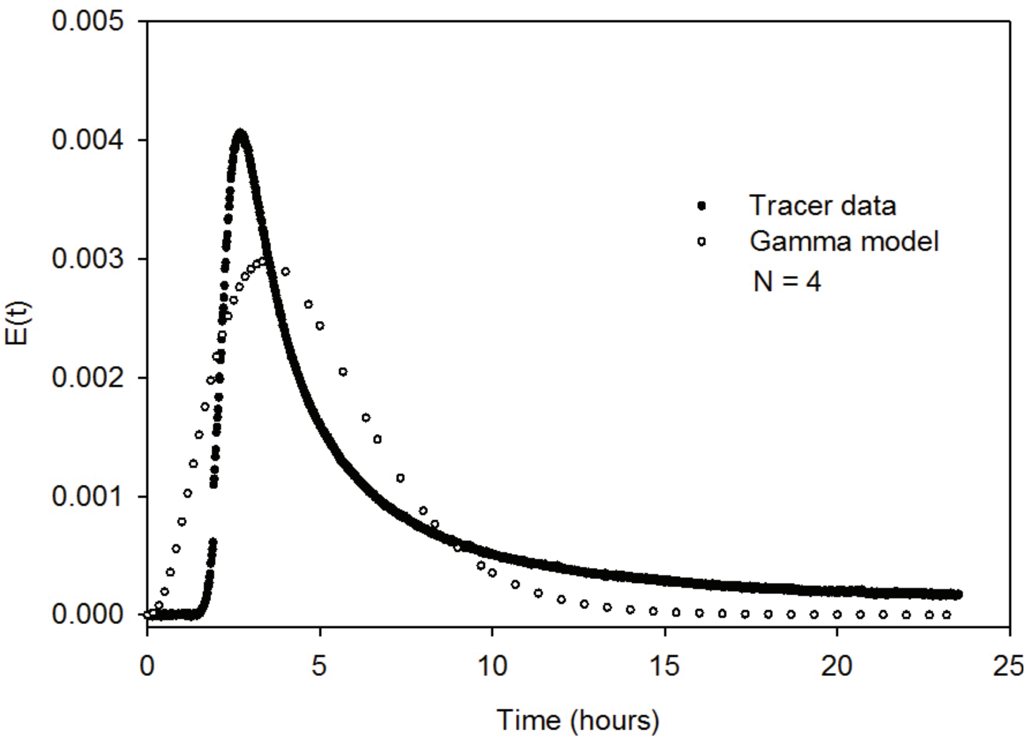

Figure 4.21 Tracer test residence time distribution *E(t)* and the gamma model least squares fit for WS-1 (upper half of the wetland during spring freshet).

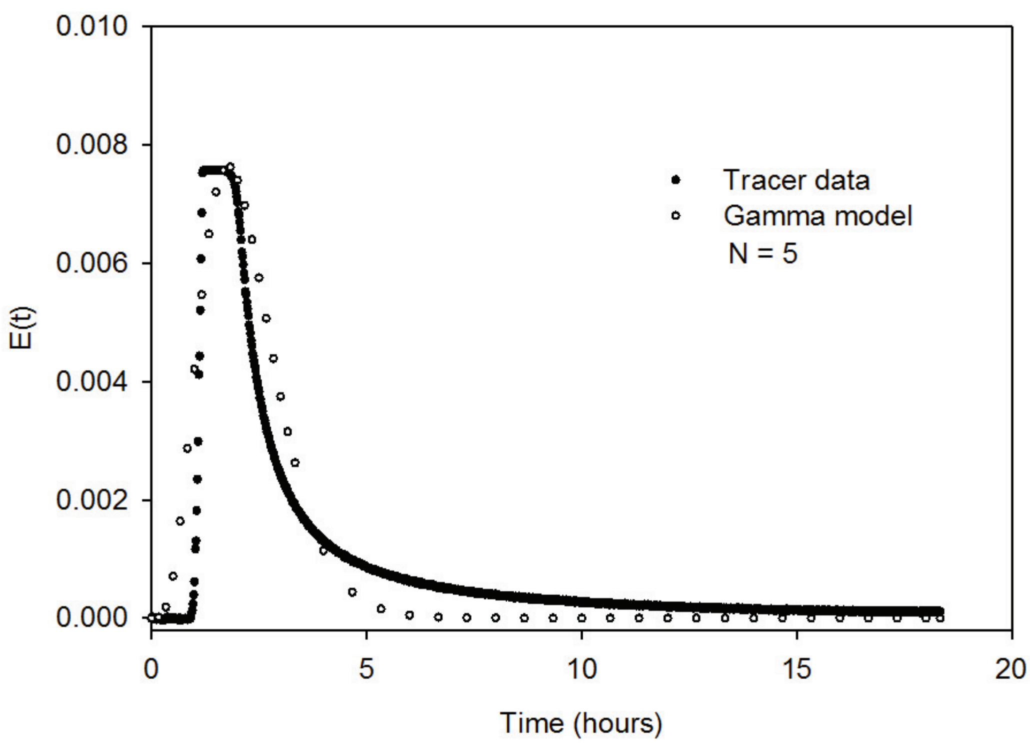

Figure 4.22 Tracer test residence time distribution *E(t)* and the gamma model least squares fit for WS-2 (lower half of the wetland during spring freshet).

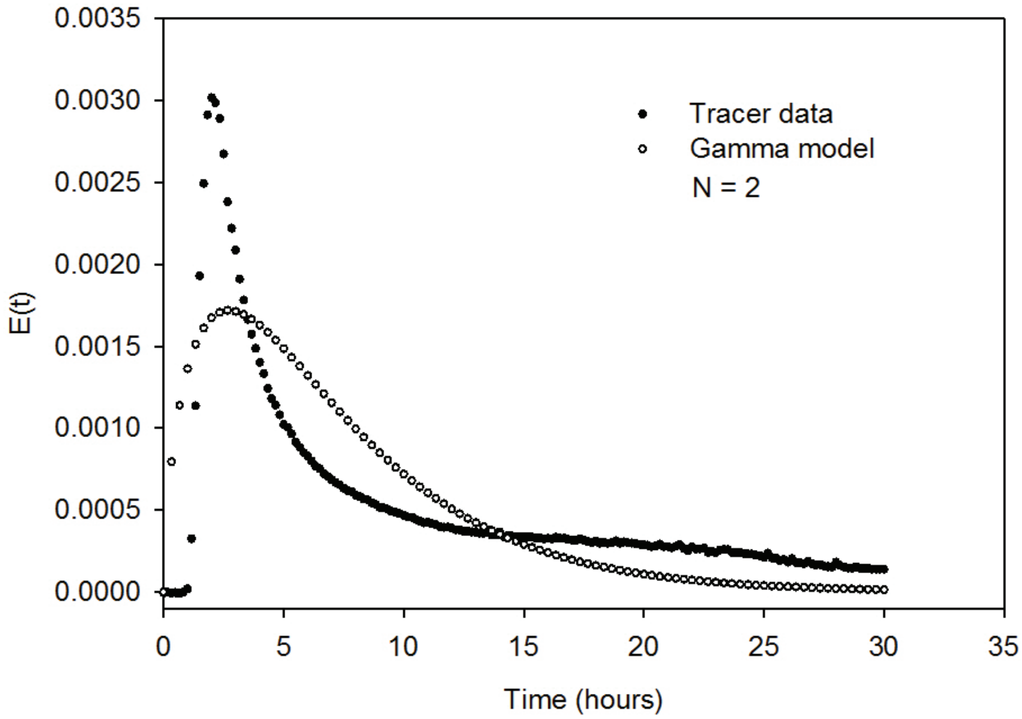

Figure 4.23 Tracer test residence time distribution *E(t)* and the gamma model least squares fit for WS-3 (upper half of the wetland after spring freshet).

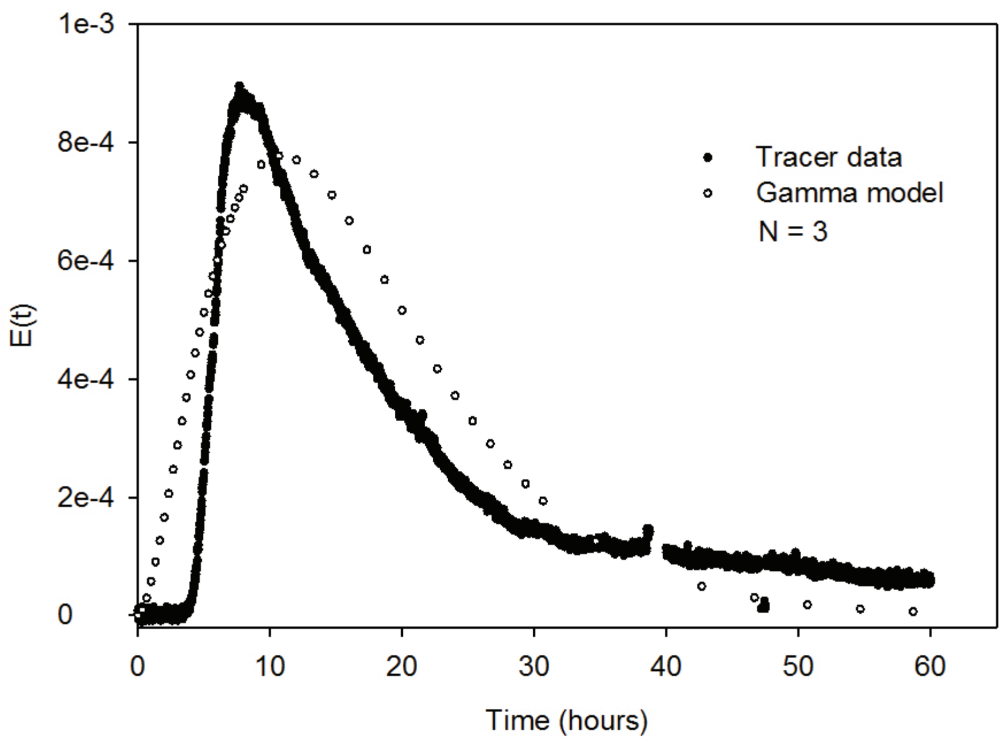

Figure 4.24 Tracer test residence time distribution *E(t)* and the gamma model least squares fit for WS-4 (lower half of the wetland after spring freshet).

4.5.2. Areal Rate Coefficients

The areal rate coefficients for each treatment performance parameter at field temperatures, and normalized to 20 ºC (k_{20}), were determined using modified TIS models (Table 4.14). During some of the model runs, the effluent concentration (C_{out}) generated from the model was less than the true concentration that was observed in the field. For these cases, the reduction in parameter concentrations can be explained entirely by dilutive effects within the wetland treatment area (*i.e.,* setting k to equal zero can produce modeled effluent concentrations which are lower than those observed in the field). Treatment may still be occurring for these lower output (LO) cases, however, it is not quantifiable given the hydraulic conditions. The LO model results can be explained by the temporal variability in flows and water quality used in the model calibration which add uncertainty in the model output.

4.5.2.1. Five-day Carbonaceous Biochemical Oxygen Demand

The average k_{20} value for $CBOD_5$ for the upstream wetland segment WS-1 during the spring freshet was 113 m/y. The downstream wetland section WS-2 had a k_{20} of 8.8 m/y. The k_{20} for the mid-season wetland segment represented by WS-3 was halved at 4.4 m/y. These k-values suggest that there was removal of organic matter near the wetland inlet, which coincides with visual observations of organic deposition near the inlet noted during the vegetation survey (Figures B-3 and B-4). Kadlec and Wallace (2009) list a distribution of tabulated rate coefficients for a group of 63 free water surface wetlands. The $CBOD_5$ k_{20} for WS-1 was in the 70th percentile of that group of wetlands. The k_{20} for WS-2 lies within the 5th percentile of the same group of wetlands. More specifically within their work, six wetlands are identified with average operating temperatures of 8.3°C which have an average k_{20} of 41 m/y.

The contrast between the $CBOD_5$ rate coefficients of WS-1 and WS-2 suggests that organic matter removal occurs primarily near the wetland inlet. Removal of BOD near the wetland inlet is a common observation in many FWS wetlands as settleable BOD components are deposited near the inlet due to laminar or near-laminar discharge conditions (Reed[2] *et al.*, 1988). Deposited BOD then undergoes aerobic or anaerobic decomposition depending on oxygen availability. The remaining soluble BOD in dissolved or colloidal forms is removed progressively with microbial contact along the wetland (Water Federation, 2010). Kadlec and Wallace (2009) have noted that the effect of weathering is very prevalent for BOD which means that lower *k*-values are produced as the reactions proceed within the wetland. Werker *et al.* (2002) suggest that apparent first-order BOD rate coefficients are commonly developed for an entire wetland based on influent and effluent concentrations; whereas actual BOD removal may only occur along a portion of the total wetland length. Dubuc *et al.* (1986) similarly noted that most BOD_5 reduction occurred in the initial 100 m of the inlet for a natural peat wetland in northern Québec. In that study, the flow was dispersed on the surface near the inlet similar to Coral Harbour, and the authors suggested the organic matter removal by microorganisms was effective in this particular area.

BOD weathering did not exclusively contribute to the difference observed in the rate coefficients between the upper and lower wetland segment halves, as there was a dependency of first order areal rate coefficients on *BLR*s. Kadlec and Wallace (2009) indicated that rate coefficients are dependent on *BLRs*, with a positive relationship between *k*-values and *BLR*s. Table 4.13 shows that the upper half of the wetland treatment area (WS-1) k_{20} values average 113 m/y with an average *BLR* of 67 kg/ha·d; whereas, the lower half of wetland treatment area (WS-2) k_{20} value is 8.8 m/y with a corresponding *BLR* of 32 kg/ha·d. The lowest *BLR* of 3.7 kg/ha·d corresponded with a lower k_{20} value of 4.4 m/d, for the upper section of the wetland treatment area (WS-3) in July 2012.

Table 4.13 $CBOD_5$ areal rate coefficients and areal BOD Loading Rates for the wetland segments during the study periods.

Wetland segment	**Date (m/d/yyyy)**	**$CBOD_5$ k_{20} (m/y)**	***BLR* (kg/ha·d)**
WS-1	6/21/2011	120	82
	6/25/2011	106	53
WS-2	6/21/2011	LO[a]	46
	6/25/2011	8.8	19
WS-3	7/7/2012	4.4	3.7

[a] LO denotes Lower Output signifying that the model outflow treatment performance parameter concentration was lower than the field observed concentration.

4.5.2.2. Escherichia coli

The average k_{20} for *E. coli* for the upper wetland segment WS-1 is 241 m/y. In contrast to the decreased removal along the wetland observed with the $CBOD_5$, the *E. coli* rate coefficient increased to an average of 699 m/y for the lower wetland segment, WS-2. The increase in the rate coefficient of *E. coli* along the wetland treatment area may be explained by progressive loading of *E. coli* along the course of the wetland from waterfowl *Branta canadensis* and *Chen caerulescens*. The waterfowl were present in the wetland treatment area during the spring freshet when the parameterization data for wetland segments WS-1 and WS-2 was collected. The k_{20} for *E. coli*, for the wetland segment WS-3 in July 2012 was 78 m/y. The lower *E. coli* *k*-value during the mid-summer period may be explained by a decreased number of waterfowl present, and a lower concentration of *E. coli* at the wetland inlet, due to growth of algae and unfavorable pH conditions for *E. coli* survival (Parhad and Rao, 1974). In the context of a grouping of *k*-value of fecal coliforms for 23 wetlands assembled by Kadlec and Wallace (2009), WS-1 in 2011 falls in the 80th percentile and WS-2 lies in the 90th percentile and WS-3 in July 2012 lies in the 50th percentile. The *E. coli* *k*-values are much higher than an average *k*-value (fecal coliform) of 34 m/y obtained from 5 FWS wetlands operating with a mean operating temperature of 8°C (Kadlec and Wallace, 2009).

4.5.2.3. Total and Volatile Suspended Solids

The k_{20} for TSS and VSS for the upper wetland segment during the spring freshet WS-1 were 20 and 15 m/y. The model results yielded lower output concentrations than the field observed outflow concentrations for WS-2; therefore, the TSS *k*-values for this WS were not determined. The k_{20} for VSS for WS-2 during the 2011 spring freshet was 22 m/y. The TSS and VSS *k*-values for the mid-season treatment period in 2012 represented by WS-3 were higher at 161 and 204 m/y respectively. Suspended sediments removal have traditionally been reported in terms of influent and effluent water quality and percent removal (Kadlec and Wallace., 2009; Kadlec, 1987). Therefore, comparison of rate coefficients to literature values is not possible.

The difference between the spring freshet and mid-season TSS and VSS model results is likely attributed to the absence of algae growth for WS-1 and WS-2 model trials;

whereas, algae was present at the wetland inlet for the WS-3 sampling period. The algae growth leads to elevated TSS and VSS concentrations near the inlet location. The TSS and VSS concentrations are generally higher at the wetland inlet after the production of algae and conversion from anoxic to oxic conditions; which were observed to have occurred by July 2012.

Similar instances of seasonal dependence of TSS concentrations on algae growth in stabilization ponds and wetlands have been noted in literature (Hammer and Burckhard, 2002). Within their work, lower temperatures contributed to lower TSS concentrations in the wetland due to decreased biological production inhibiting algae proliferation. Generally there is a strong correlation between TSS removal rates and loading rate (Knight *et al.*, 1987). Hence, increased loading from higher concentrations may explain the higher rate coefficients observed during the post-spring freshet study periods.

4.5.2.4. Nitrogen and Phosphorus

The k_{20} TN is 20 m/y for WS-1 and not quantified for WS-2 due to a lower model output than field observed concentrations. For WS-3, a total nitrogen k_{20} was determined to be 2.7 m/y. The upper half of the wetland treatment area in June 2011 (WS-1) TN *k*-value lies in the 70th percentile in comparison to an assembly of 116 FWS wetlands summarized by Kadlec and Wallace (2009). Whereas, the upper wetland segment WS-3 in July 2012 falls in the 10th percentile of the same group of wetlands. The TN *k*-value results are similar to an average of 15 m/y of four wetlands with an average operating temperature of 8 ºC tabulated by Kadlec and Wallace (2009). Andersson *et al.* (2005) reported first order k_{20} TN removal rates of 7.3 and 13.3 m/y for two FWS wetlands (23 and 28 ha) in Sweden; the highest removal rate was observed in a wetland with a 2 ha overland flow area. The authors attributed the increased nitrogen removal by promotion of nitrification in the overland flow section of the wetland. The Coral Harbour wetland treatment area may demonstrate similar dentrification capabilities with its DO concentrations, shallow water depth, and resulting contact with the substrate and vegetation.

The average k_{20} for TAN for WS-1 was 18 m/y, which is in the 60th percentile of a group of 118 FWS wetlands compiled by Kadlec and Wallace (2009). The k_{20} value of the lower wetland segment WS-2 was not determined due to lower output model results than the field observed effluent concentrations. The mid-season July 2012 k_{20} from WS-3 was 12 m/y, which is in the 50^{th} percentile of the group of wetlands. These values are also similar to an average TAN k_{20} of 20 m/y from four free water surface wetlands with an average operating temperature of 7.5°C tabulated by Kadlec and Wallace (2009).

The *k*-values for TP were not determined due to lower outputs from the model than the field observed outflow concentrations.

4.5.2.5. Variability and Design Values

The variations in the *k*-values for individual wetland segments during a particular seasonal period can be attributed to fluctuations in *HLR*s on the wetland treatment area. An important general observation of the discharge data used in the model parameterization is that temporal variations in discharge occur over short time periods of less than a week ranging from 47 to 155 percent differences between measurements (Table 3.5). The temporal variations in treatment performance *k*-values are partly explained by the differences in hydraulic behaviour of the wetland treatment area between performance sampling events.

Overall, the first order rate coefficients for most of the treatment performance parameters estimated from the modified TIS wetland performance model were comparable to literature values from southern systems. The spatial and temporal variability in the *k*-values may render very conservative adoption of design *k*-values preferable for design applications. The determination of *E. coli* *k*-values representative of actual removal rates was complicated by non-point source additions to the wetland treatment area. A different approach to bacteria removal prediction may be required.

Table 4.14 Rate coefficients determined from modified TIS performance models.

Wetland segment	WS-1				WS-2				WS-3	
Date sampled (d/m/y)	21/06/2011		25/06/2011		21/06/2011		25/06/2011		07/07/2012	
Rate coefficient	*k* (m/y)	k_{20} (m/y)	*k* (m/y)	k_{20} (m/y)	*k* (m/y)	k_{20} (m/y)	*k* (m/y)	k_{20} (m/y)	*k* (m/y)	k_{20} (m/y)
Parameter										
$CBOD_5$	111.2	120.2	102.0	106.3	LO[a]	LO	8.4	8.8	4.7	4.4
E. coli	157	244	188	237	329	511	701	887	122	78
TSS	20.4	20.4	LO	LO	LO	LO	LO	LO	160.6	160.6
VSS	18.6	18.6	10.9	10.9	LO	LO	21.9	21.9	204.4	204.4
TN	*	*	17.9	19.8	*	*	LO	LO	3.3	2.7
TAN	*	*	15.3	18.3	LO	LO	*	*	16.5	11.8
TP	*	*	LO	LO	*	*	LO	LO	LO	LO

* Data not collected for these dates.

[a] LO denotes Lower Output signifying that the model outflow treatment performance parameter concentration was lower than the field observed concentration.

Chapter 5: Conclusions

5.1. Conclusions and Design Recommendations

The following section briefly summarizes the main findings of the Coral Harbour, NU wetland treatment performance assessment and modeling study. The overall conclusions are listed and resulting recommendations are bulleted.

The outlet point location of the wetland can be difficult to define. Water quality can change dramatically in short distances from dilution from non-effluent watershed contributions and other factors. Additionally, the wetland outlet point location was shown to change over the treatment season. This ambiguity in outlet location complicates monitoring and performance assessment of the system.

- The wetland outlet sample locations, and any other important sample locations, should be strategically located, representative of the waste stream, and be well defined to assure long-term monitoring is consistent. The correct sample location siting may require site-specific hydrodynamic studies such as tracer testing, as described within this study.

The results of the vegetation survey and supervised classification suggest that the effluent flow areas show a decrease in species diversity and deposition of organic detritus from the effluent waste stream. Specific species dominated the effluent flow areas such as *Salix richardsonii* and *Bryophyta Spp.* An increase in vegetation height was observed in the effluent flow areas and was likely from nutrient uptake. The effluent discharge onto the natural tundra wetland area has likely changed the native vegetation, and in some places, localized near the inlet, solids deposition has caused vegetation die-off.

- There may be changes to the native tundra vegetation in similar other wetland treatment sites. Further research is required to determine optimal loading rates to avoid unfavorable vegetation die-off. It is likely that non-point source dispersal methods would help to mitigate this impact.

Two flow paths in the wetland were identified which varied in between spring freshet and post-spring freshet conditions. The watershed delineation indicated that two watershed sub-basins contribute water to the spring freshet and post-freshet flow paths, which are 7 ha and 15 ha respectively. The difference between wetland-to-catchment areas is important from a water quality perspective (*i.e.,* dilution). Water quality may change along the wetland length with dilutive non-effluent contributions. The wetland-to-catchment ratio varied over the treatment season from 1 to 5 during the spring freshet and 1 to 21 during the post-spring freshet. For this site, a detailed RTK topographic survey was required to refine the sub-basin watershed delineations.

- The watershed delineation and catchment-to-wetland ratio of the wetland treatment area are important. Particularly, to develop an understanding of the wetland water budget, quantify dilutive inputs, strategically choose sample locations, and mitigate unfavorable *HRT* decreases.

The wetland area selected for the calculation of the *HLR*s was demonstrated to be very important. At this site, the wetted area changed over the treatment season. An accurate determination of the wetted area of the wetland required multiple tracer studies during the freshet and mid-summer. GPS delineation of the wetted area in the field was required.

- Field delineation of the wetted area receiving effluent in the wetland is required for accurate *HLR* determination for design calculations and this may require site-specific tracer studies.

At times, the effluent from the WSP at times traveled exclusively as subsurface flow in the upper half of the wetland treatment area, particularly during low *HLR* periods of the treatment season. The *HRT* of the wetland was observed to be 14 days during these low HLR periods. There was some evidence to suggest that the SSF area could provide some treatment of the effluent; however, finer temporal scale sampling would be required to quantify treatment performance of the SSF area accurately. The SSF area does not provide consistent treatment because is not a primary flow path for the effluent during most of the treatment season.

- Optimization of a SSF area of the wetland has great potential to enhance the treatment performance of the wetland. Engineering would be required to take advantage of the SSF area (*e.g.,* inlet re-configuration).

The wetland *HRT* varied dramatically from ≤ 0.5 days during the spring freshet to 14 days in the summer. The variation in wetland *HRT* affects how much effluent treatment can occur, and is an important design parameter for assessing treatment potential. The worst case in treatment availability is in conjunction with the shortest *HRT*, which for this system was during the spring freshet. In comparison to other wetlands, an HRT of 0.5 days is very short.

- Tracer testing to determine site-specific *HRT*s should be conducted at strategic times during the spring freshet or in conjunction with the highest effluent discharge period for decanted systems. Engineering of natural wetland systems would help to increase the hydraulic efficiency and treatment performance.

The presence of algae in the wetland likely led to an increase in pH and DO particularly at the inlet. A decrease in *E. coli* concentrations and an increase in TSS was observed in conjunction with the conversion of the inlet from anaerobic (<1 mg/L DO) to aerobic DO conditions (~8.4 mg/L DO). The pH at the inlet accordingly increased with algal growth from approximately 7.6 to 8.2. It is hypothesized that the volatilization of the ammoniacal component of TN occurs from the pH increase and assimilation of nitrogen by algae.

- Wetland biochemical parameters, particularly DO, temperature, and pH, provide insight into the treatment mechanisms in the wetland. The measurement of basic geochemical parameters is simple, inexpensive, and may be used as a tool for deciding optimal decant or discharge times.

Assessment of the treatment performance demonstrated that the Coral Harbour wetland was effective at reducing contaminant concentrations to relatively low levels at the wetland outlet during all periods studied. Percent reductions over the periods studied were as follows: 87.4 – 100% for $CBOD_5$; 3.0 – 8.4 log for *E. coli*; 94.5 – 99.9% for TSS; 94.3 – 100% for VSS; 73.4 – 99.3% for TN; 72.3 – 100% for TAN; 55.6 – 100% for NH_3-

N; and 73.3 – 99.9% for TP. The lower contaminant reductions were observed during the spring freshet and was likely due to decreased wetland-to-catchment ratio, and the very short *HRT* observed.

- The worst case treatment performance samples should be taken when *HRT*s are the shortest for a tundra treatment wetland.

E. coli concentrations varied up to four orders of magnitude over the treatment season. Additionally, the rate coefficients were relatively high compared to the literature, and in cases higher in the downstream half of the wetland. Results suggest the addition of non-point source addition of *E. coli* from migratory waterfowl during the spring freshet. The shallow water depth, vegetation and settling of *E. coli* sorbed particulates may have all contributed to the high *E. coli* rate coefficients.

- The dynamic and non-point nature of *E. coli* for arctic natural wetland systems renders it a difficult parameter to clearly assess treatment performance for pathogen removal, and deduce overall pathogenic risk attributable directly to wastewater. It may be favourable to differentiate between anthropogenic and passerine sources of *E. coli* for treatment performance assessments.

Total Suspended Solids concentrations were higher at the inlet when biological production increased in the post-spring freshet conditions. The increase in TSS concentrations from algal proliferation led to an apparent decrease in TSS treatment performance. However, TSS was non-representative of the raw wastewater.

- Special consideration should be given to account for the algal component of the TSS during the post-spring treatment period. TSS may not be preferable for use as a performance indicator when algae is present in the water sample. Alternatively, supplementary provision should be made to account for the algae component of TSS in a water sample if it is to be used as a performance indicator (*e.g.,* chlorophyll concentration measurement in combination with TSS, *etc.*)

The first order rate coefficients determined with the modified TIS treatment wetland model were temporally and spatially variable. The k_{20} values and percentiles compared to literature from the modified TIS model ranged from: 4.4 (5^{th}) – 120 (70^{th}) m/y for $CBOD_5$; 78 (50^{th}) – 887 (90^{th}) m/y for *E. coli*; 20 – 161 m/y for TSS; 11 – 204 m/y for VSS; 2.7 (10^{th}) – 20 m/y (70^{th}) for TN; and 12 (50^{th}) – 18 (60^{th}) m/y for TAN. Phosphorus was not quantified due to low model output and concentrations in the wetland.

- Generally, the adoption of low *k*-values from the breadth of literature on treatment wetland rate coefficients from southern systems would be conservative for this wetland treatment area.

The modified TIS treatment wetland modeling revealed that much of the reductions observed in treatment performance parameter concentrations could be attributable to dilution. The quantification of dilution was facilitated with the mass balance approach. The distinction between dilution and treatment is important to accurately characterize the treatment mechanisms. There is danger in extrapolating the treatment performance of the Coral Harbour wetland to other sites, because the dilutive potential will be different.

- There will be a need to characterize the treatment performance capabilities of each wetland on a site-specific basis due to the variable and uncontrolled nature of natural tundra wetlands.

5.2. Recommendations for Future Research

There are limitations to the Coral Harbour natural tundra wetland treatment performance and modeling study. As such, recommendations for future research initiatives are provided to identify areas that still require research to facilitate proper development of natural tundra wetland design criteria.

Further research on the effluent effects on arctic vegetation in the wetland treatment areas for other systems is required. Eventually, determination of the threshold *HLR*s and *BLR*s to avoid detrimental effects to the native vegetation would be useful for design purposes.

The differentiation of *E. coli* sources between anthropogenic and passerine sources in the wetland treatment areas could be researched with *E. coli* source tracking methods such as with quantitative Polymerase Chain Reaction (qPCR) analytical techniques. This information would be particularly useful to determine temporal trends in passerine sources to identify specific periods when *E. coli* analysis would be unfavorable for use as a pathogen indicator in the treatment wetlands (*e.g.,* during migratory periods). The fate of *E. coli* and other pathogens in the wetland treatment areas is an avenue for future research

The subsurface flow area of the Coral Harbour wetland treatment area was potentially providing treatment intermittently during the treatment season. Further research would be required to assess the treatment performance potential of natural tundra wetlands characterized primarily by SSF. A different modeling approach than the modified TIS presented within may be required for SSF applications. The performance model development for an arctic SSF treatment wetland would be an area for future research. Furthermore, the permafrost melt rates and depths of active layers, of the SSF areas receiving effluent would be an important parameter to characterize, especially in relation to *HRT*.

Further work is required to quantify the intrasystem and intersystem variability in first order rate coefficients. For Coral Harbour, the intrasystem variability in *k*-values may be further studied on a desktop basis with modified TIS model run simulations. This would involve selecting a range of input parameters and running the model numerous times to estimate worst case *k*-values. Additionally a sensitivity analysis could be conducted with the model to determine what parameters are most critical in the resulting effluent concentrations. Development of wetland performance models for other sites will be a requirement to facilitate the quantification of intersystem variability in the *k*-values. Therefore, other natural or semi-natural tundra wetlands will need to be monitored with a similar approach as was taken in Coral Harbour, as was described within this study.

The development of appropriate, effective, and conservative design criteria for the use of natural and semi-natural tundra wetlands as part of the overall wastewater treatment BMPs for Canada's Far North will require at least partial fulfillment of the above stated further research initiatives.

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APPENDIX A EXAMPLES OF WETLAND TREATMENT AREA VEGETATION AND LAND COVER CLASSES

Class 1: *Bryophyta* spp. (Wetland)

Figure A-1 Dominant vegetation *Bryophyta* spp. (center) and secondary species *Salix richardsonii* (top center) and less dominant *Carex* spp. (bottom left). Quadrat located at 64° 09' 48.4" N, 083° 11' 18.1" W.

Figure A-2 Dominant vegetation *Bryophyta* spp. and secondary dominant species *Salix richardsonii* (top left). Quadrat located at 64° 09' 49.8" N, 083° 10' 53.9" W.

Class 2: Organic detritus (Effluent Solids Deposition / Wetland)

Figure A-3 Dominant organic detritus land cover and dead vegetation (top left). Quadrat located at 64° 09’ 48.3”N, 083° 11’ 29.1” W.

Figure A-4 Dominant organic detritus land cover and dead vegetation (center and bottom) with secondary dominant species *Salix reticulata* (top left and right). Quadrat located at 64° 09’ 47.5” N, 083° 11’ 27.2” W.

Class 3: *Carex* spp. (Wetland / Upland Transition)

Figure A-5 Dominant vegetation *Carex* spp. and secondary dominant species *Salix richardsonii* (bottom left). Quadrat located at 64° 09’ 50.2” N, 083° 11’ 5.2” W.

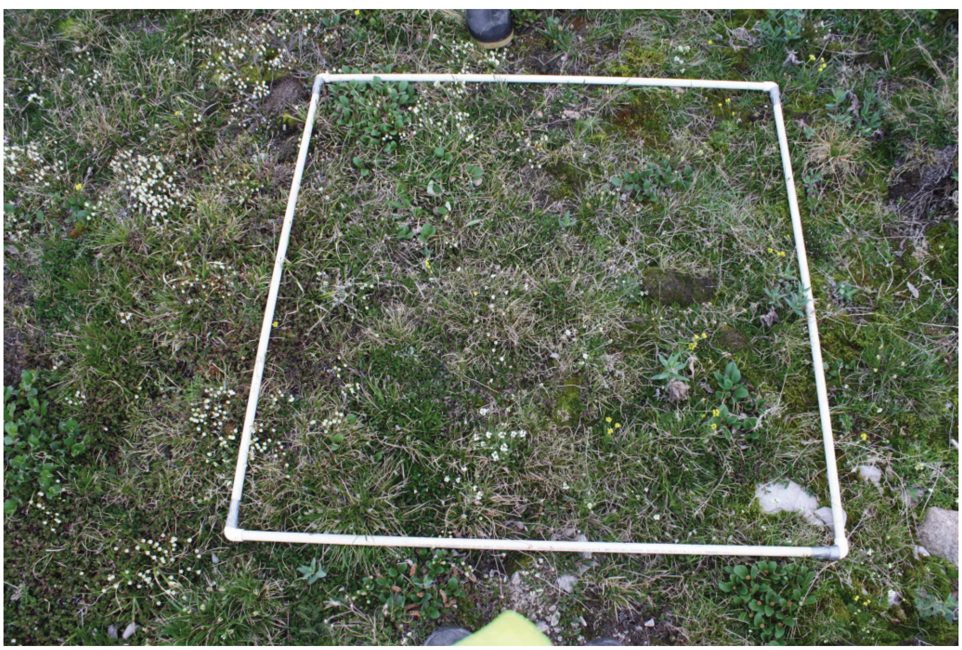


Figure A-6 Dominant vegetation *Carex* spp. and secondary dominant species *Calluna vulgaris*, and less dominant species *Bryophyta* spp., *Dryas octopetala*, *Salix reticulata*, *Salix richardsonii*. Quadrat located at 64° 09’ 48.7” N, 083° 11’ 31” W.

Class 4: *Lichen* (Upland)

Figure A-7 Dominant vegetation *Lichen*, and gneiss cobbles, and secondary dominant species *Vaccinium uliginosum,* and less dominant species *Saxifraga oppositifolia, Rhododendron tomentosum, Calluna vulgaris,* and *Dryas octopetala.* Quadrat located at 64° 09' 48.1" N, 083° 11' 13.8" W.

Figure A-8 Dominant vegetation Lichen and secondary dominant species *Carex* spp.*,* and less dominant species *Saxifraga oppositifolia, Dryas octopetala*, and *Calluna vulgaris*. Quadrat located at 64° 09' 47.3" N, 083° 11' 14.1" W.

Class 5: *Hippuris vulgaris* (Wetland)

Figure A-9 Dominant vegetation *Hippuris vulgaris* and secondary dominant species *Ranunculus hyperboreus* (top right and center right). Quadrat located at 64° 09' 47.2" N, 083° 11' 19.1" W.

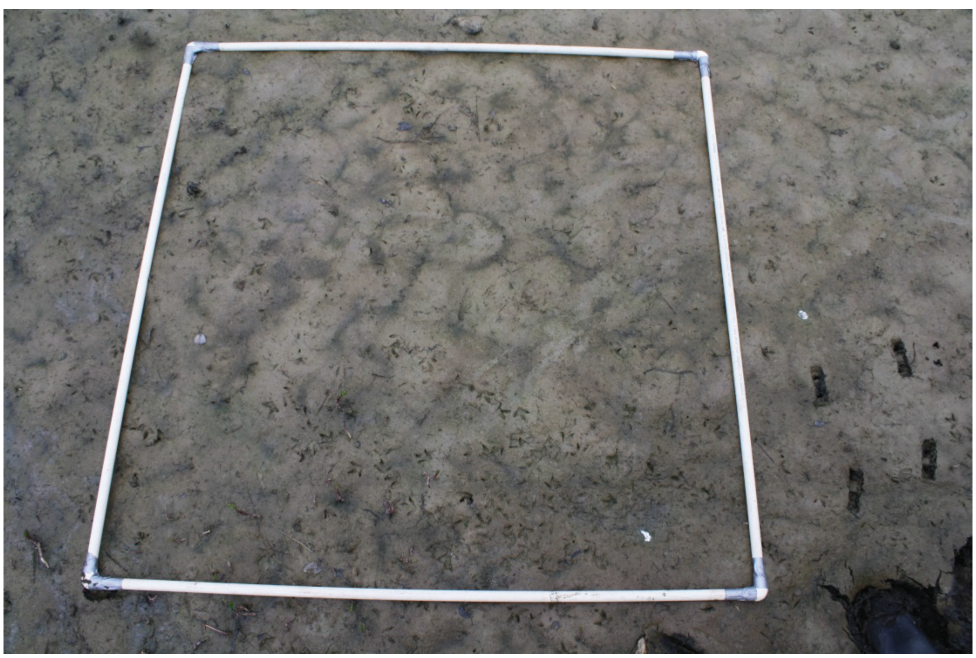


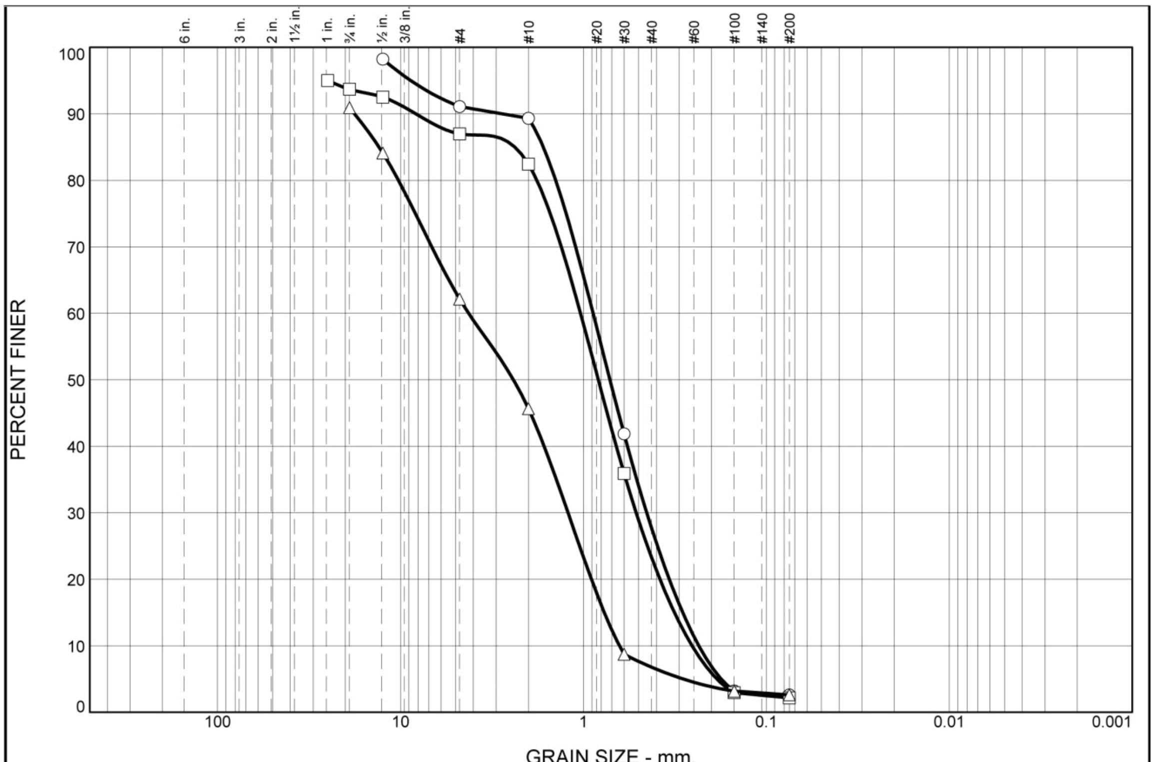
Figure A-10 Shallow ponded water and dominant vegetation *Hippuris vulgaris.* Quadrat located at 64° 09' 47.7" N, 083° 11' 17.1" W.

Class 6: *Salix richardsonii* (Wetland)

Figure A-11 Dominant vegetation *Salix richardsonii* and secondary dominant vegetation *Bryophyta* spp. (top right). Quadrat located at 64° 09' 48.4" N, 083° 11' 8.8" W.

Figure A-12 Dominant vegetation *Salix richardsonii* and secondary dominant vegetation *Salix reticulata* (top center), and less dominant *Carex* spp., *Dryas octopetala* and *Bryophyta* spp. Quadrat located at 64° 09' 49.3" N, 083° 11' 22.3" W.

APPENDIX B GRAIN SIZE DISTRIBUTION



	% +3"	% Gravel		% Sand			% Fines	
		Coarse	Fine	Coarse	Medium	Fine	Silt	Clay
○				1.8	61.5	25.2	2.6	
□			6.7	4.6	59.1	21.1	2.2	
△				16.5	38.9	4.2	2.6	

	LL	PL	D_{85}	D_{60}	D_{50}	D_{30}	D_{15}	D_{10}	C_c	C_u
○			1.6575	0.8855	0.7163	0.4509	0.2866	0.2340	0.98	3.78
□			2.3728	1.0376	0.8292	0.5150	0.3181	0.2548	1.00	4.07
△			13.1106	4.2388	2.4147	1.2141	0.7690	0.6357	0.55	6.67

Material Description	USCS	AASHTO
○ Loose Brown Sand and Gravel	SP	
□ Loose Brown Carbonaceous Sand and Gravel	SP	
△ Loose Brown Carbonaceous Sand and Gravel	SP	

Project No. **Client:**

Project: Wastewater treatment investigation

○ **Location:** Piezometer 3 **Sample Number:** P3

□ **Location:** Piezometer 1 **Sample Number:** P1

△ **Location:** Quadrant 1 **Sample Number:** Q1

Remarks:

Figure

Tested By: J.Hayward

Figure B-1 Grain size distribution plots of three surficial soil samples taken in the groundwater flow area (depth 0 to 1 m).

APPENDIX C SUPPLEMENTARY WATER QUALITY DATA

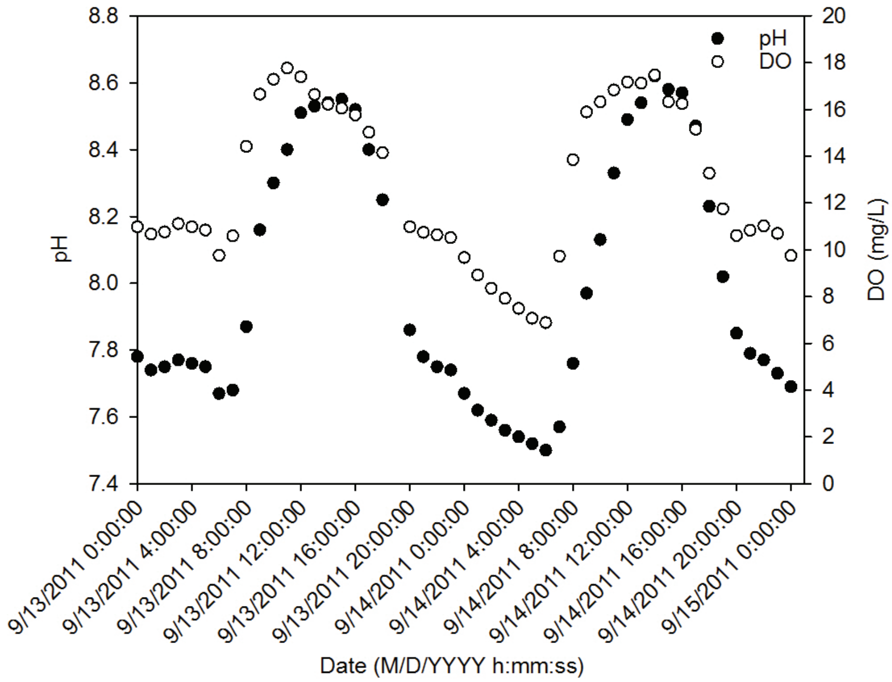


Figure C-1 Diurnal 48-hour time series of pH and DO trends at the mid-wetland during the post-spring freshet conditions in September 2011.

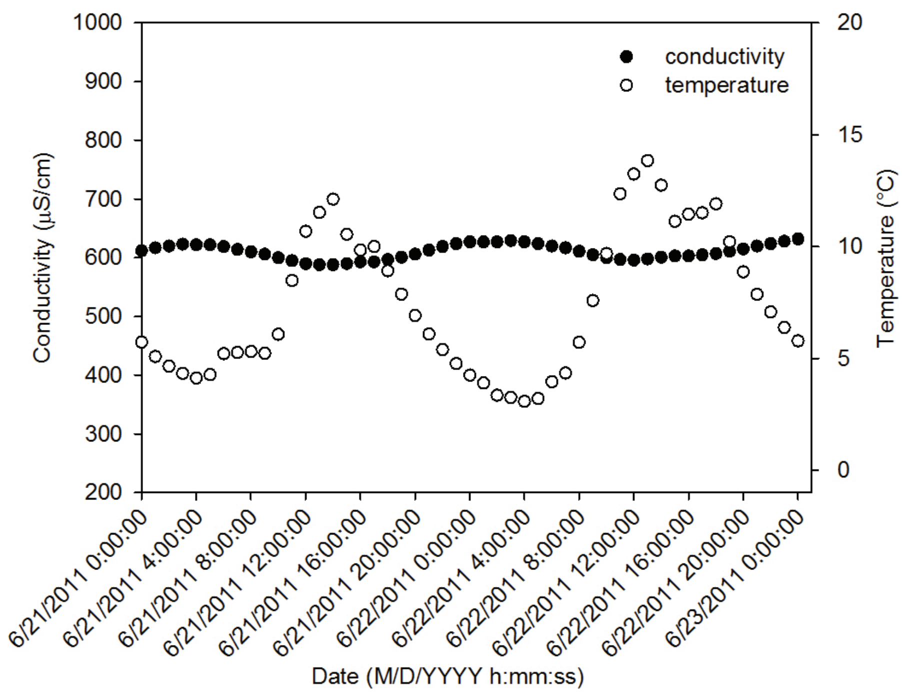


Figure C-2 Diurnal 48-hour time series of conductivity and temperature at the mid-wetland during the spring freshet conditions in June 2011.

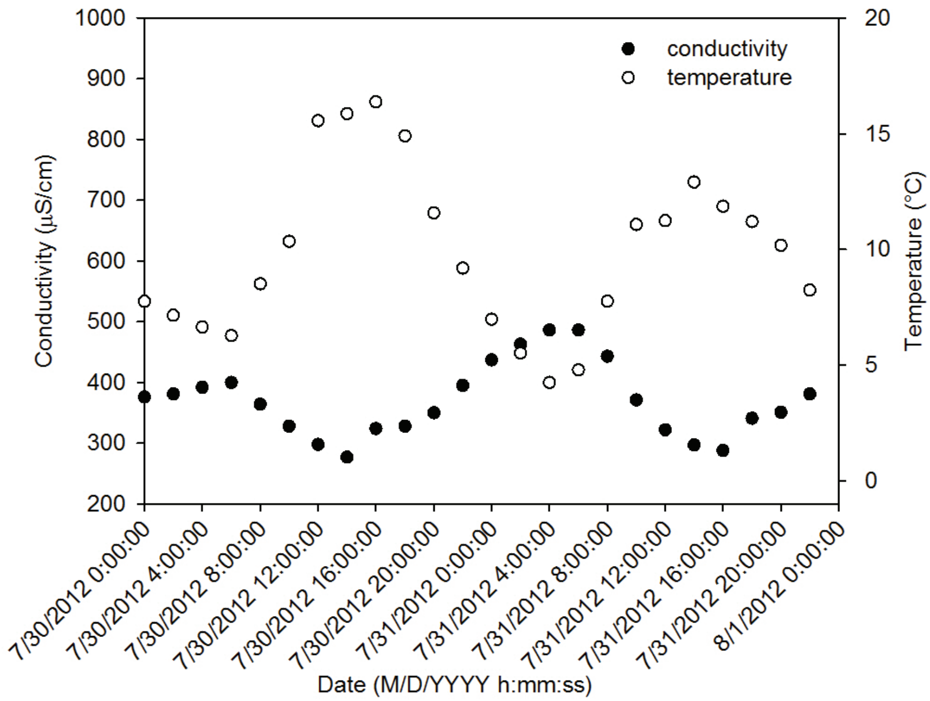


Figure C-3 Diurnal 48-hour time series of conductivity and temperature at the mid-wetland during the post-spring freshet conditions in July 2012.